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An Optimization Approach to Evaluate the Role of Ecosystem Services in Chesapeake Bay Restoration Strategies

United States Environmental Protection Agency
Office of Research and Development

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Acronyms and Abbreviations

AFO	animal feeding operation
BMP	best management practice
BNR	biological nutrient removal
CAFO	confined animal feeding operation
CAIR	Clean Air Interstate Rule
CBPO	Chesapeake Bay Program Office
CBWM	Chesapeake Bay Program's Phase 5.3 Community Watershed Model
CH_4	methane
CO	carbon monoxide
CO_2	carbon dioxide
CO_2e	carbon dioxide equivalent
COLE	Carbon On-Line Estimator
CWA	Clean Water Act
DED	duck energy day
EO	Executive Order
EPA	U.S. Environmental Protection Agency
ESRP	Ecosystem Services Research Program
FASOM	Forest and Agricultural Sector Optimization Model
GAMS	General Algebraic Modeling System
GHG	greenhouse gases
HI	high-density impervious
HP	high-density pervious
LI	low-density impervious
LP	low-density pervious
IPCC	Intergovernmental Panel on Climate Change
LWD	large woody debris
MILP	mixed integer linear programming
MS4	municipal separate stormwater system
N_2O	nitrous oxide
NO_2	nitrogen dioxide
NCASI	National Council for Air and Stream Improvement
NHD	National Hydrography Dataset
O&M	operation and maintenance
POTW	publicly owned treatment works
RESAC	2000 Regional Earth Science Application Center
SSURGO	Soil Survey Geographic database
STATGO2	State Soil Geographic database
TMDL	total maximum daily load
UFORE	Urban Forest Effects Model
USDA	U.S. Department of Agriculture
USFS	U.S. Forest Service

USGS	U.S. Geological Survey
WIP	Watershed Implementation Plans
WWTP	wastewater treatment plant

Chesapeake Bay Facts

Area:	64,000 square miles
No. of Streams:	More than 100,000 streams
Land to Water Ratio:	14:1
States:	Delaware, Maryland, New York, Pennsylvania, Virginia, West Virginia, and Washington, D.C.
Major River Basins:	James River Eastern Shore Patuxent River Potomac River Rappahannock River Susquehanna River (50% of the freshwater draining to the Bay) Western Shore York River

Land Use (acres):		
	Forest	28,693,725
	Cropland	3,333,949
	Pasture/Hay	5,668,917
	Urban	2,958,157

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EXECUTIVE SUMMARY

Many of the nation's watersheds and estuaries are suffering from the effects of high pollutant loads. Under the Clean Water Act (CWA), one of the main mechanisms for addressing these water quality impairment problems is through the establishment of total maximum daily loads (TMDLs), which limit the allowable amount of pollutant loads to a water body. Despite significant progress over the past two decades, meeting TMDL limits often presents challenging tradeoffs regarding where and how to control upstream pollutant sources. For example, relatively large and easily verifiable nutrient load reductions often can be achieved through controls of point-source discharges; however, these controls may entail relatively large costs and minimal reductions in sediment loads. In contrast, nonpoint-source controls are generally more diffuse and difficult to monitor; however, they may offer lower-cost options, more sediment control, and other ecological benefits.

The purpose of this project is to develop an analytic framework to assist policymakers in evaluating these TMDL-related tradeoffs. The framework is designed to incorporate measures of both the cost-effectiveness and ecosystem service impacts associated with individual pollution-control projects. The inclusion of ecosystem services is a unique feature of this framework. It accounts for not only the targeted pollutant reductions but also the ancillary societal benefits—i.e., "bonus" ecosystem services—provided by certain pollution-control projects. For example, riparian forest buffers not only reduce nutrient runoff to streams, they also sequester carbon through increased biomass. When these ancillary benefits are expressed in monetary terms, they can be thought of as offsetting some of the costs of the pollution-control projects.

This report describes how the analytic framework can be used to explore the following key questions: (1) what mix of pollution-control projects provides the least costly way to achieve water quality goals in an impaired watershed and (2) how does the consideration of bonus ecosystem services affect the desired mix of projects?

The application described in this report is intended to illustrate how the framework can be used to investigate the tradeoffs between project costs, load reductions, and bonus ecosystem services. **The reported results should NOT be interpreted as policy recommendations, because the framework does not yet include all of the information needed for a complete assessment of the socially optimal mix of pollution controls.** In particular, the report **does**

NOT advocate large-scale changes in agricultural land use. Rather, it examines the tradeoffs between a range of point- and nonpoint-source controls, under alternative and simplified modeling assumptions.

ES.1 APPLICATION OF THE ANALYTIC FRAMEWORK TO THE CHESAPEAKE BAY WATERSHED

To demonstrate the analytic framework, we use the Chesapeake Bay watershed and its recently established TMDL as a case study application (**Figure ES-1**). It must be noted that, due to time and resource constraints for development, the analytic framework provides a somewhat simplified representation of the sources and control measures available in the Chesapeake Bay watershed. These limitations also apply to the estimates of costs, pollutant load reductions, and ecosystem services associated with these options. As a result, the analytic framework is not currently suitable for examining very specific and detailed policy options. Nevertheless, given these constraints, this report shows that the framework can provide a number of useful insights into the more general tradeoffs (including costs and ecosystem service benefits) associated with meeting TMDL targets.

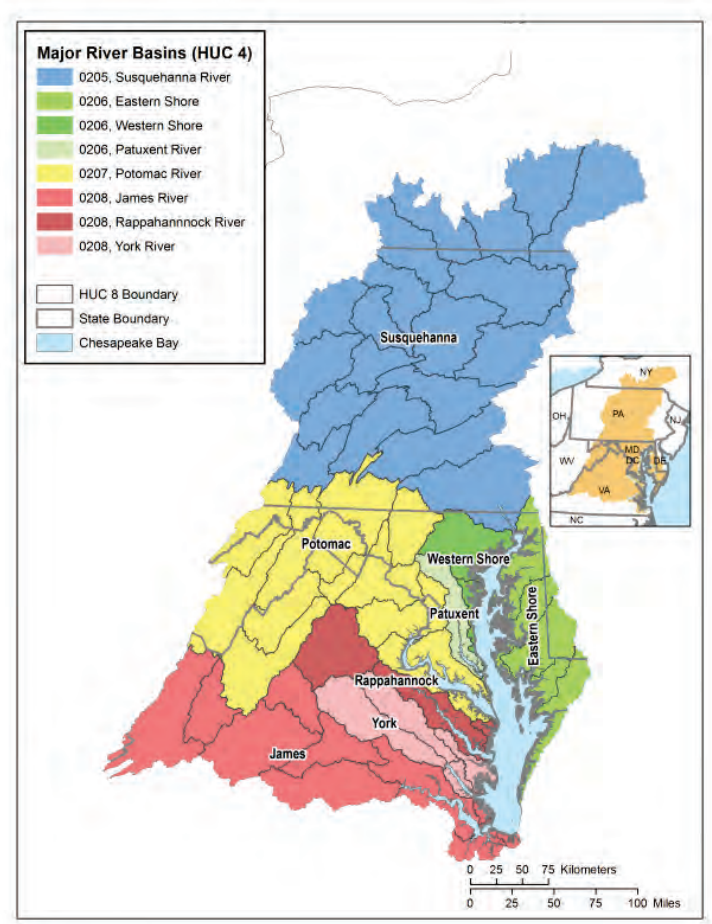


Figure ES-1. Chesapeake Bay – major river basins.

Evaluating Nitrogen, Phosphorus, and Sediment Reductions Needed to Restore the Chesapeake Bay

As part of its commitment to meet the objectives articulated in Executive Order (EO) 13508, the U.S. Environmental Protection Agency (EPA) established a Chesapeake Bay TMDL in December 2010. The TMDL sets nutrient- (i.e., nitrogen and phosphorus) and sediment-load allocations for the major tributaries to the Bay to be achieved by 2025. These allocations can be translated into load-reduction targets, which are summarized in **Table ES-1**. To help achieve these targets, the EPA has statutory authority to regulate point sources of nutrients and/or sediments, such as municipal and industrial wastewater treatment plants (WWTPs), municipal separate storm sewer systems (MS4s), and confined animal feeding operations (CAFOs). The EPA does not have statutory authority to regulate nonpoint sources, such as agriculture and silviculture, "on-site" wastewater treatment, and some types of stormwater runoff; however, states and local governments often do have such regulations.

Table ES-1. Load Reduction Targets by Basin (millions of lbs)

Basin	Nitrogen[a]	Phosphorus	Sediment
Eastern Shore of Chesapeake Bay	4.74	0.27	38.88
James River Basin	8.18	0.89	326.23
Patuxent River Basin	0.20	0.05	7.67
Potomac River Basin	6.77	1.03	509.72
Rappahannock River Basin	1.01	0.18	51.90
Susquehanna River Basin	33.14	1.16	529.02
Western Shore of Chesapeake Bay	4.91	0.26	38.24
York River Basin	0.95	0.08	23.80
Total	**59.91**	**3.92**	**1525.47**

[a] Excludes expected reductions in delivered loads attributable to non-tidal atmospheric deposition in the watershed.

Considering Green Alternatives to Gray Treatment: The Costs and Ecological Co-benefits

The pollution controls associated with the selected sources can be categorized as either "gray" or "green" infrastructure. Gray infrastructure refers to common urban and suburban wastewater and stormwater controls, such as municipal and industrial WWTPs and certain MS4 pollutant-control technologies. Green infrastructure represents pollution-control practices that have the potential to naturally filter out nutrients and sediment, while also providing added ecosystem services such as water storage and carbon sequestration. Green infrastructure includes several best management practices (BMPs) for the treatment of nonpoint-source runoff, such as the installation of buffer strips at stream edges, the use of pervious surfaces to reduce stormwater runoff, and the restoration of wetlands.

This report presents the results of a systems analysis designed to explore the implications—the costs and ecosystem service benefits—of alternative mixes of green and gray infrastructure to achieve nutrient load and sediment load reduction targets under the Chesapeake Bay TMDL. The conceptual framework for the systems analysis is shown in **Figure ES-2**.

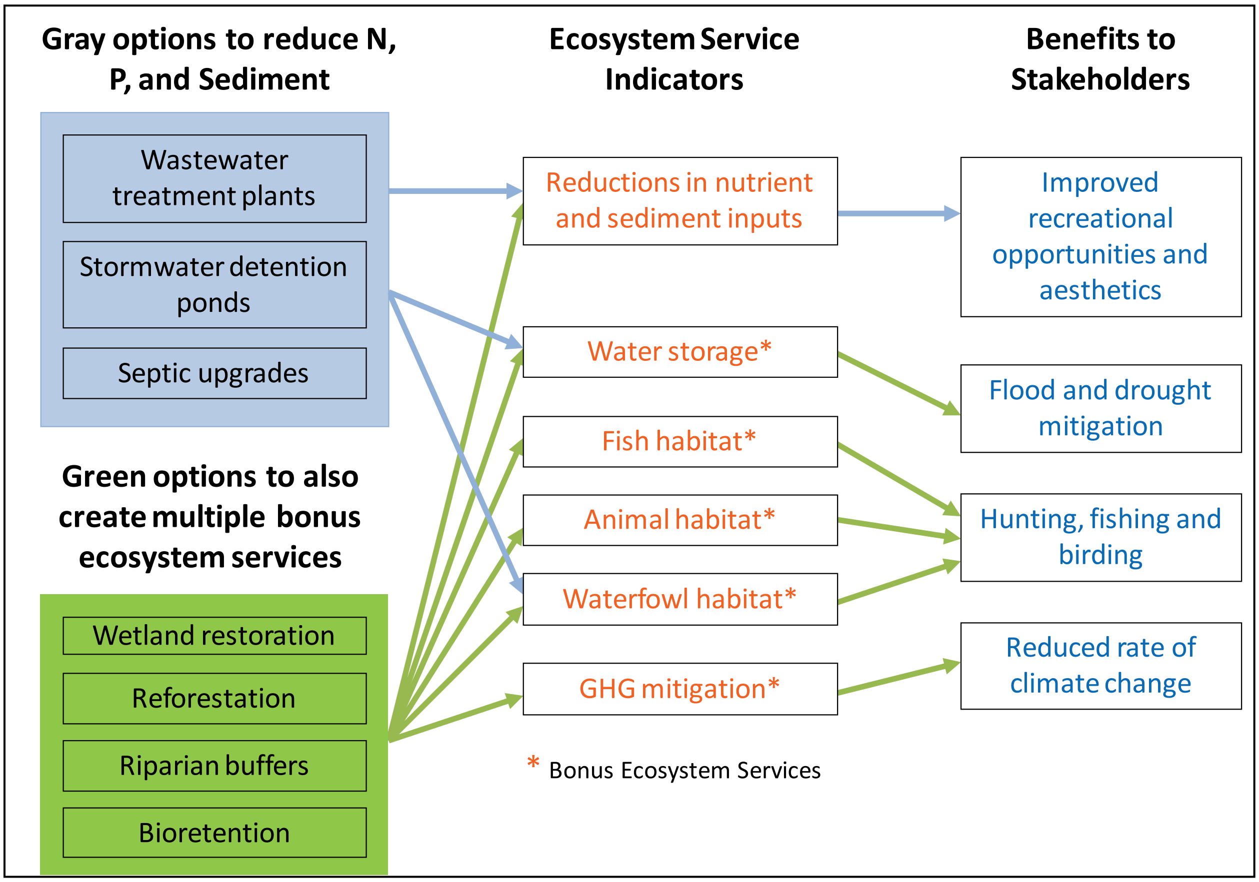


Figure ES-2. Gray vs. green infrastructure pollution controls, associated ecosystem services, and stakeholder benefits.

Evaluating Alternatives for Achieving Pollution-reduction Targets

The analytic framework provides a tool that can be used to evaluate alternative approaches for meeting TMDL load-reduction requirements cost-effectively. The optimization analysis identifies a mix of pollution-control projects that minimizes the total costs of control and maximizes the bonus ecosystem benefits provided by these controls. In many cases, achieving an optimal mix of projects requires striking a balance between private and public interests, because the costs of controls are primarily borne by the owners or users of the sources (e.g., utility customers, land owners), whereas the ecosystem benefits (e.g., greenhouse gas [GHG] mitigation, flood protection) are often externalized and distributed more broadly across society.

This analysis is a subset of analyses that would be needed to identify a “socially optimal” mix of control options that considers the tradeoffs between producing bonus ecosystem services and other goods and services, such as agricultural products. Therefore, the results from the analytic framework cannot be interpreted as the socially optimal solution, because the framework does not account for the full range of benefits and tradeoffs associated with using particular pollution-control approaches. Nevertheless, the framework incorporates a considerable amount of information regarding implementation costs, pollution-reduction effectiveness, and production of ecosystem services that informs tradeoff analysis.

The approach used to develop and apply the analytic framework is represented in **Figure ES-3.** The core steps are described as follows:

Step 1. Define the aggregate nutrient and sediment load reduction targets of interest. For this application, the targets are based on the basin-specific TMDL allocations shown in Table ES-1.

Step 2. Create a spatial inventory of the main point and nonpoint sources in the watershed, and identify control projects (gray or green infrastructure) for reducing nutrient and sediment loads from these sources. The pollutant sources and related controls are grouped into three main categories: (1) point-source control technologies, (2) nonpoint-source agricultural BMPs, and (3) nonpoint-source urban stormwater BMPs. Additional details are provided below.

Step 3. Develop estimates of the annual costs and effectiveness of representative pollution-control projects. In this context, effectiveness is measured as the annual decrease in pollutant load delivered to the tidal waters of the Chesapeake Bay. The costs include, as appropriate, annualized capital and installation costs, land costs, and operation and maintenance costs.

Step 4. Develop estimates of the bonus ecosystem services associated with selected pollution-control projects. As shown in Figure ES-2, pollutant-load reductions from the gray and green projects provide the core ecosystem services. However, other *bonus* ecosystem services can also be provided, depending on the mix of control projects selected. Therefore, as feasible, methods were developed to quantify the effects of pollution-control projects on a number of bonus ecosystem service indicators. Where possible, these ecosystem service changes are also measured in monetary terms. This monetization is designed to represent the societal value of the service or a potential payment to the control project's provider for supplying the ecosystem service.

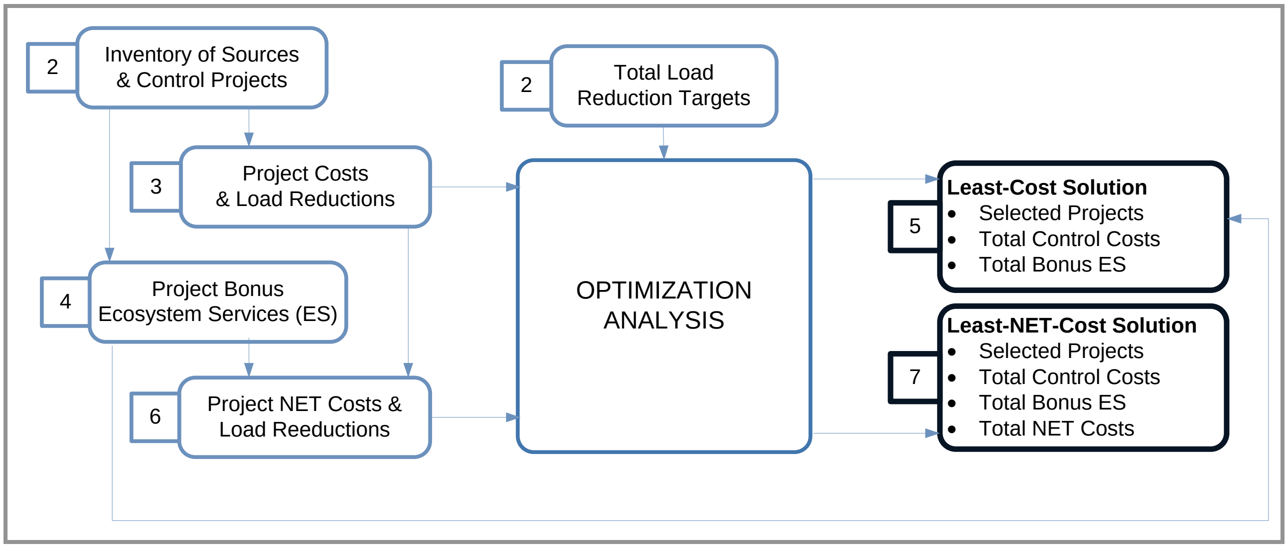


Figure ES-3. Flow diagram representation of the analytic approach.

Step 5. Generate the least-cost solution. Using a linear programming optimization analysis, this process identifies the combination of nutrient- and sediment-control projects included in the analytic framework that achieves targeted load reductions to the Chesapeake Bay at the lowest total cost. In addition to identifying the least-cost combination of projects and their associated costs, the optimization solution can be used to characterize and quantify the bonus ecosystem services delivered by these selected projects.

Step 6. Estimate the NET costs associated with each project. In short, NET costs are equal to costs of the project minus its monetized bonus ecosystem service benefits provided by the project.

Step 7. Generate the least-NET-cost solution. In this case, both the costs of control and the bonus ecosystem services are directly incorporated into the optimization analysis. It identifies the combination of control projects that achieves the load-reduction targets at the lowest total NET cost. This result is the framework's closest approximation of the socially optimal solution, given the data limitations described above.

Selecting and Characterizing Sources and Pollution Controls.

Given the time and resources available, this study included as many sources, pollution controls, and BMPs as feasible. The sources and BMPs selected for the analysis were carefully chosen to be as representative as possible; however, it is important to emphasize that it does not include all of the possible BMPs and control measures. The results of the analysis must be interpreted with this limitation in mind.

The point sources in the analytic framework include the more significant municipal and industrial WWTPs in the Bay watershed. To reduce loading from these sources, the framework includes WWTP upgrades for enhanced nitrogen and/or phosphorus removal at 332 significant municipal (≥ 0.5 million gallons per day capacity) and 58 industrial WWTPs. The most advanced nitrogen and phosphorus removal technologies available for full-scale implementation are referred to in the report as “Tier 4” technologies. The costs and effluent concentrations associated with Tier 4 technologies are summarized in **Table ES-2**.

Table ES-2. Annual Average Concentrations and Costs for Tier 4 WWTP Technology

Facility Type	Parameter	Effluent Concentration (mg/L)	Average Cost ($/additional lb removed/yr)
Significant Municipal (generally, ≥ 0.5 MGD)	TN	3.0	$21.58
	TP	0.1	$14.92
Significant Industrials (generally, >75 lb/day TN and 25 lb/day TP)	TN	3.0 or permit limit, if less	$19.10
	TP	0.1 or permit limit, if less	$407.21

TN = total nitrogen
TP = total phosphorus

Five urban stormwater BMPs are also included in the framework. The costs and removal efficiencies used to represent these technologies are summarized in **Table ES-3**.

Table ES-3. Average Costs and Removal Efficiencies for Selected Urban Stormwater BMPs

BMP	Total Annual Cost per BMP Acre ($/acre/yr)	Removal Efficiencies (%)		
		Total Nitrogen	*Total Phosphorous*	*Total Suspended Solids*
Extended Detention	$4,460	20%	20%	60%
Bioretention	$66,647	48%	60%	68%
Grass Buffer	$6,676	32%	40%	53%
Forest Buffer	$364	50%	60%	60%
Wetlands	$601	20%	45%	60%

Nine representative nonpoint-source agricultural BMPs were also included in this study. **Table ES-4** presents ranges of costs and removal efficiencies used in the analytic framework for seven of the BMPs. The framework also includes natural revegetation (i.e., allowing land to return to a more natural vegetative cover, by suspending agricultural production activities on the land, but possibly allowing other activities such as hunting) and conversion to forest. Load reductions for these BMPs are estimated as the difference between crop/pastureland loads and forest/open land loads, and their cost ranges are $14–141 to $129–257 per acre per year, respectively.

Table ES-4. Average Costs and Removal Efficiencies for Selected Agricultural BMPs

BMP	Total Annual Cost per BMP Acre ($/acre/yr)	Removal Efficiencies (%)		
		Total Nitrogen	*Total Phosphorous*	*Total Suspended Solids*
Forest Buffers	$163–291	19–65%	30–45%	40–60%
Grass Buffers	$99–226	13–46%	30–45%	40–60%
Wetland Restoration	$236–364	7–25%	12–50%	4–15%
Livestock Exclusion	$81–117	9–11%	24%	30%
Cover Crops	$31	34–45%	15%	20%
No-till	$14	10–15%	20–40%	70%
Reduced Fertilizer Application	$37	15%	0%	0%

In addition to quantifying the costs and effectiveness of urban stormwater and agricultural BMPs, the analytic framework includes estimates of selected bonus ecosystem services for 12 of the 14 BMPs. Due to data and resource constraints, this quantitative analysis is limited to the ecosystem service categories shown in **Table ES-5**. Monetary estimates (expressed in dollars per acre) were estimated for four categories of service. In particular, values for carbon sequestration and/or reduced GHG emissions were associated with all 12 of the BMPs listed in Table ES-5. Non-monetary indicators were also included for two categories of bonus ecosystem services.

Table ES-5. Summary of Bonus Ecosystem Services Included for Selected BMPs

	Monetized Ecosystem Services				Non-monetized Ecosystem Services	
BMPs	**Carbon Sequestration & Reduced GHG Emissions**	**Non-waterfowl Hunting**	**Duck Hunting**	**Air Quality**	**Brook Trout Habitat**	**Wetland Water Storage**
Urban Stormwater						
Extended Detention	●					
Bioretention	●			●		
Grass Buffer	●			●		
Forest Buffer	●			●		
Wetlands	●			●		●
Agriculture						
Forest Buffers	●	●			●	
Grass Buffers	●					
Conversion to Forest	●	●			●	
Natural Revegetation	●	●			●	
Wetland Restoration	●	●	●		●	●
No-till	●					
Reduced Fertilizer Application	●					

Selecting Scenarios for Conducting Sensitivity Analyses

Ten scenarios were created (**Table ES-6**) to test the analytic framework, investigate the effects of different potential approaches for achieving the TMDLs, and examine the effect of uncertainties regarding inputs to the analysis.

Table ES-6. Scenarios for Conducting Sensitivity Analyses of the Analytic Framework

Scenario	Description	Options
1	Uses Bay Strategy's load reduction for nitrogen, phosphorus, and sediment	
2	The same as Scenario 1, but with (1) added transaction costs for agricultural and stormwater BMPs and (2) increased agricultural land costs	a) Agricultural and stormwater BMP transaction costs 10% greater than Scenario 1 (Base Case) b) Agricultural and stormwater BMP costs 25% greater than Scenario 1 c) Increase land rental costs for agricultural BMPs by 120%
3	The same as Scenario 2(a), but with BMP pollution removal effectiveness adjustments	a) Assume agricultural and stormwater BMPs are 50% as effective as point sources (2:1 credit ratio) b) Assume agricultural and stormwater BMPs are 67% less effective as point sources (3:1 credit ratio)
4	The same as Scenario 2(a), but with sediment options	a) Lower target sediment load allocation (higher reduction target) b) Higher target sediment load allocation(lower reduction target) c) No sediment reduction
5	The same as Scenario 2(a), but with single-pollutant controls	a) Nitrogen reduction target only b) Phosphorus reduction target only c) Sediment reduction target only
6	The same as Scenario 2(a), but with carbon price options	a) \$26 per ton of carbon (\$7 per ton of CO_2e) b) \$92 per ton of carbon (\$25 per ton of CO_2e)
7	The same as Scenario 2(a), but with WWTP technology requirements	a) WWTP operating at Tier 4 levels b) WWTP operating at Tier 4 levels and a 2:1 credit ratio for point sources compared to nonpoint sources (i.e., nonpoint-source BMPs are 50% as effective as point-source control project)

Scenario	Description	Options
8	The same as Scenario 2(a), but with agricultural land conversion restrictions	a) No agricultural land conversion beyond 100 feet from stream b) No natural revegetation c) No agricultural land conversion beyond 1,044 feet from stream
9	The same as Scenario 2(a), but with minimum agricultural wetland restoration restrictions	a) Minimum of 30,000 acres of agricultural wetland restoration b) Minimum of 60,000 acres of agricultural wetland restoration
10	The same as Scenario 5(a), but with Bay-wide nitrogen reduction targets and inter-basin BMP credit ratios (rather than river basin-level targets)	

Scenario 1(TMDL Basin-level Targets) uses the load-reduction targets specified in Table ES-1. It includes all of the point-source, agricultural BMP, and urban stormwater control projects described in ***Section 3*** of this report.

Scenarios 2 through 9 are included to assess how the results from applying the analytic framework are affected by changes to BMP costs; BMP effectiveness; varied sediment reduction targets; controlling one pollutant at a time (as opposed to simultaneous control of all 3 pollutants); changing carbon prices; WWTP technology; and agricultural land conversion constraints.

Scenario 10 (Bay-Wide Targets with Basin-level Load Adjustment Factors) disregards river basin-level targets and opens the optimization analysis across the entire Bay watershed while accounting for the differential effects of loadings from different basins on Chesapeake Bay water quality.

All of the scenarios are run for the two optimization conditions described in ***Sections 1 and 3*** of this report: (1) a least-cost solution and (2) a least-NET-cost solution.

ES-2 KEY RESULTS AND CONCLUSIONS

Figure ES-4 highlights and **Table ES-7** summarizes the cost and bonus ecosystem service results of the scenarios. Some of the main findings from these analyses are the following:

- Given the inventory of point- and nonpoint-source controls included in the analytic framework, green infrastructure projects (agricultural BMPs, in particular) account for approximately two-thirds of the project costs in most of the least-cost solutions to the TMDLs.

- For example, as shown in the first bar in Figure ES-4, the estimated aggregate annual control costs in Base Case Scenario 2(a) are $218 million, with 64% of these costs attributable to agricultural BMPs and 36% attributable to point-source controls.

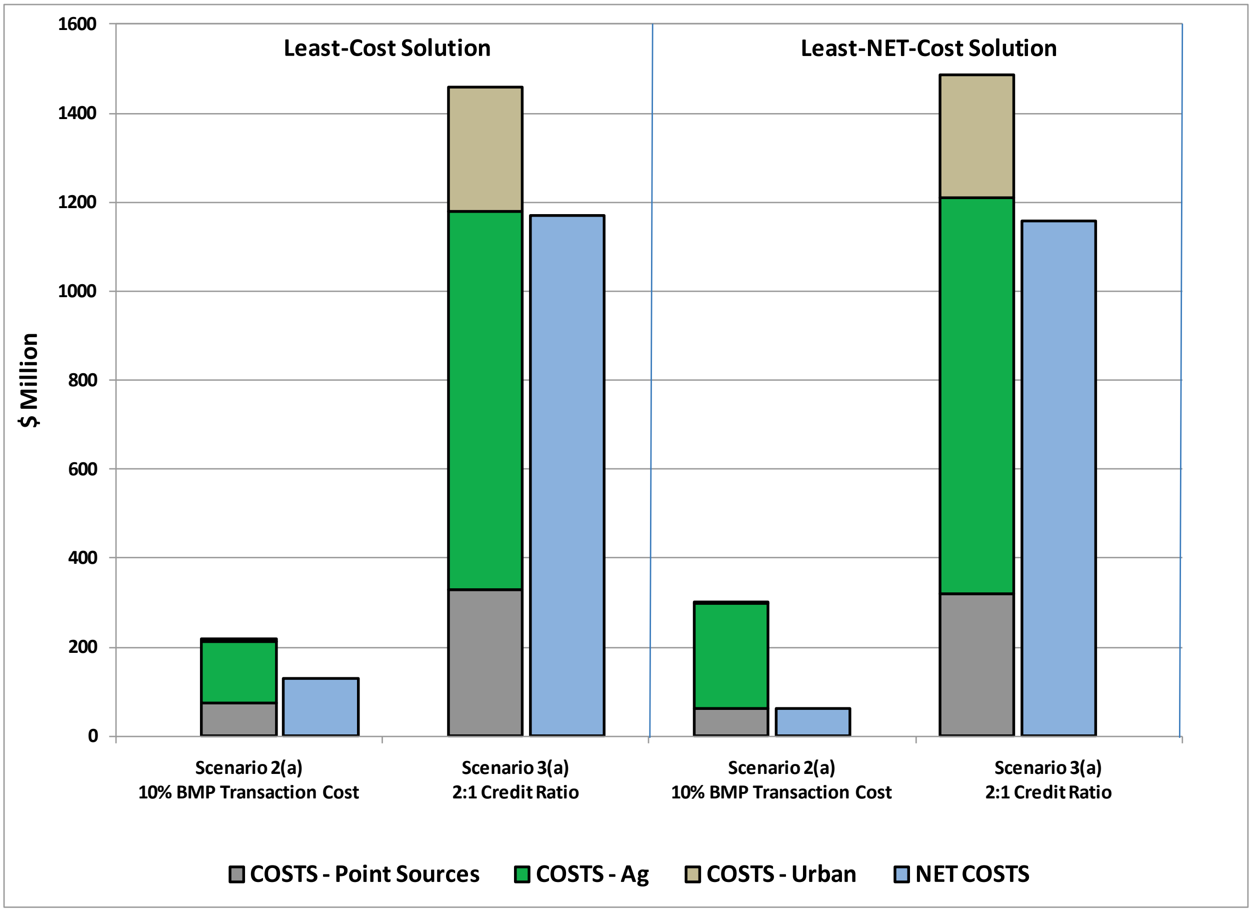


Figure ES-4. Costs, bonus ecosystem services, and NET costs for selected scenarios and optimization solutions.

- Green infrastructure contributes substantial offsetting ecosystem service values to the cost of achieving the TMDL targets; gray infrastructure contributes ecosystem service *disbenefits*.
  - For example, as shown in the second bar in Figure ES-4, the offsetting value of the bonus ecosystem services in Base Case Scenario 2(a) is $90 million and is mainly from the value of carbon sequestered through the natural revegetation BMP.

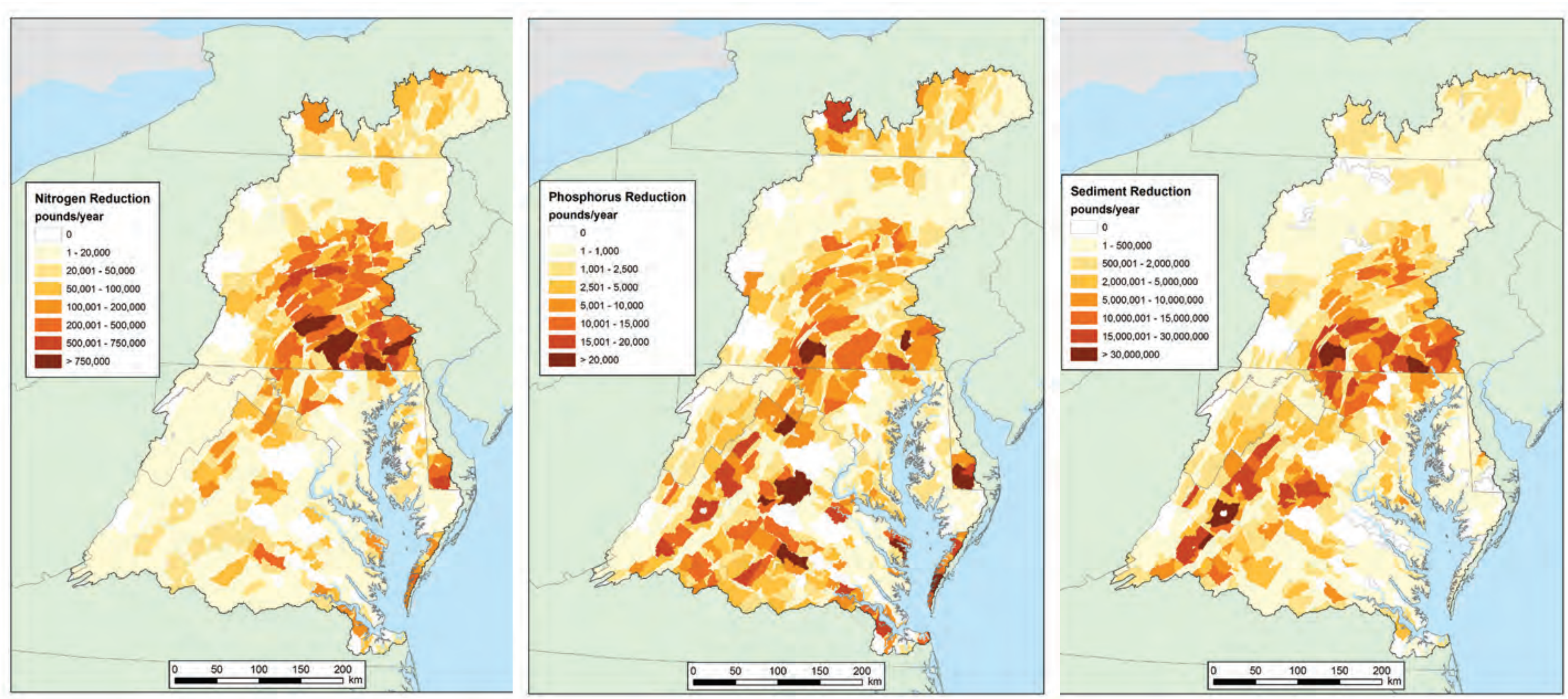
  - In the scenario requiring maximum technology upgrades for WWTPs (Scenario 7), the total social *costs* of GHG and other air pollutant emissions from these sources are $17.7 million.
- Including monetized ecosystem services in the optimization analysis (i.e., estimating a least-NET-cost solution) shifts the solution towards the inclusion of more nonpoint-source controls, in particular, natural revegetation on agricultural land. The size of this shift is particularly sensitive to the assumed per-ton value of carbon sequestration.
  - In Base Case Scenario 2(a), monetized value of bonus ecosystem services more than double to $238 million per year (see the sixth bar in Figure ES-4) when the value of these services is included in the optimization analysis. The acreage in natural revegetation increases more than three-fold. When the assumed price of carbon is doubled in Scenario 6(b), the value of bonus ecosystem services increase to $666 million per year.
- As expected, the total cost of control increases and the value of bonus ecosystem services decrease significantly when (1) transaction and land rental costs are increased for nonpoint-source BMPs, (2) the pollution removal effectiveness of BMPs is reduced, (3) the availability of agricultural BMP projects is restricted, and (4) the technological requirements on WWTPs are made more stringent. The highest aggregate control costs (over $2 billion per year) were estimated for the scenario that combined lower BMP pollution removal effectiveness and more stringent WWTP technology requirements.
- Uncertainty about the pollution removal effectiveness of agricultural or stormwater BMPs may substantially increase the costs of achieving the load-reduction targets, in particular, if states require offset ratios for point- to nonpoint-source trades. While Scenario 3(a) does not specifically model such trades, the large increase in control costs resulting from a 2:1 credit ratio (to over $1.4 billion, as shown in Figure ES-4) is indicative of the expected result of such policies. Although BMP credit ratios are currently being used in a precautionary fashion to promote beneficial environmental outcomes, they may also add significant costs. Consequently, an improved understanding of performance risks of BMPs could help to substantially reduce the costs of achieving TMDLs through nonpoint-source controls.

- Of the three pollutants' reduction targets, the nitrogen targets tend to have the largest effect on the optimization solutions. Removing the phosphorus and sediment reduction targets from the framework has a relatively small effect on total costs and land-use changes. The sediment reduction targets are the least influential.
  - For example, when only nitrogen reduction targets are included in the optimization analysis, the estimated total costs decrease by less than 10% compared to the Base Case, where all three pollutants are controlled. In contrast, the costs decline by 65% and 91%, respectively, when only phosphorus and sediment reduction targets are included.
- As shown in **Figures ES-5**, **ES-6**, **and ES-7**, the load reductions are more heavily concentrated in the near-tidal areas and the basins with the highest load-reduction targets.
  - For example, over 55% of the total nitrogen load reductions are required from the Susquehanna basin, which makes up 43% of the land area of the watershed. As a result, the largest concentration of nitrogen load reductions in the watershed is located in the lower reaches of the Susquehanna basin.
- Finding the least-cost solution for meeting the TMDL targets does not necessarily mean selecting the control projects with the lowest *ratio* of costs per delivered load reduction, particularly if the *total* load reductions from these projects are relatively small. For example, grass buffers often have a relatively low-cost-per-pound ratio for nitrogen reduction; however, they do not often figure significantly into the least-cost solutions because they do not contribute enough total load reductions to meet the basin-level targets.
- The costs of achieving desired improvements in Chesapeake Bay water quality could be further reduced by replacing basin-specific load-reduction targets with Bay-wide targets and inter-basin BMP credit ratios (based on their relative impact on Bay water quality). The added flexibility is expected to result in a less costly mix of control projects; however, it would also change the spatial distribution of load reductions in the Chesapeake Bay watershed.

Table ES-7. Summary of Optimization Results by Scenario

Scenario	Least-Cost Solution			Least-NET-Cost Solution		
	Annual Control Costs ($ millions/yr)	Bonus Ecosystem Services ($ millions/yr)	Annual NET Costs ($ millions/yr)	Annual Control Costs ($ millions/yr)	Bonus Ecosystem Services ($ millions/yr)	Annual NET Costs ($ millions/yr)
Scenario 1 — TMDL Basin-level Targets	205.4	91.0	114.5	292.8	251.8	40.9
Scenario 2a — Basin-level Targets with 10% BMP Transaction Costs (Base Case)	218.4	89.8	128.6	301.4	238.0	63.4
Scenario 2b — Basin-level Targets with 25% BMP Transaction Costs	237.8	86.9	150.9	307.5	213.5	93.9
Scenario 2c — Basin-level Targets with 2.2x Land Rental Costs	287.8	59.4	228.4	335.1	133.8	201.2
Scenario 3a — Basin-level Targets with 10% BMP Transaction Costs, 2:1 Credit Ratio	1,457.1	287.2	1,169.9	1,487.3	329.3	1,158.0
Scenario 3b — Basin-level Targets with 10% BMP Transaction Costs, 3:1 Credit Ratio	2,020.9	374.4	1,646.5	2,031.0	381.4	1,649.6
Scenario 4a — Basin-level Targets with 10% BMP Transaction Costs, Low Sediment Load Allocation	227.8	91.2	136.6	308.6	232.8	75.8
Scenario 4b — Basin-level Targets with 10% BMP Transaction Costs, High Sediment Load Allocation	218.6	89.8	128.8	300.7	237.4	63.3
Scenario 4c — Basin-level Targets with 10% BMP Transaction Costs, No Sediment Reduction Target	217.1	86.8	130.3	298.7	235.8	62.9
Scenario 5a — Basin-level Nitrogen Target Only with 10% BMP Transaction Costs	199.9	79.4	120.5	282.1	224.0	58.0
Scenario 5b — Basin-level Phosphorus Target Only with 10% BMP Transaction Costs	75.7	42.3	33.4	151.3	176.1	(24.8)
Scenario 5c — Basin-level Sediment Target Only with 10% BMP Transaction Costs	20.1	6.9	13.2	118.0	150.9	(33.0)
Scenario 6a — Basin-level Targets with 10% BMP Transaction Costs, Low Carbon Price	218.4	52.7	165.7	249.2	101.6	147.7
Scenario 6b — Basin-level Targets with 10% BMP Transaction Costs, High Carbon Price	218.4	179.4	39.1	439.2	666.0	(226.8)
Scenario 7a — Basin-level Targets with 10% BMP Transaction Costs, Tier 4 Upgrades	1,024.5	51.4	973.1	1,106.6	190.9	915.7

Scenario	Least-Cost Solution			Least-NET-Cost Solution		
	Annual Control Costs ($ millions/yr)	Bonus Ecosystem Services ($ millions/yr)	Annual NET Costs ($ millions/yr)	Annual Control Costs ($ millions/yr)	Bonus Ecosystem Services ($ millions/yr)	Annual NET Costs ($ millions/yr)
Scenario 7b — Basin-level Targets with 10% BMP Transaction Costs, Tier 4 Upgrades, 2:1 Trading Ratio	2,044.0	269.9	1,774.1	2,068.3	307.8	1,760.5
Scenario 8a — Basin-level Targets with 10% BMP Transaction Costs, No Conversion Beyond 100ft	1,730.1	15.2	1,714.9	1,732.2	16.6	1,715.6
Scenario 8b — Basin-level Targets with 10% BMP Transaction Costs, No Application Of Natural Revegetation	361.7	51.9	309.8	366.6	60.1	306.4
Scenario 8c — Basin-level Targets with 10% BMP Transaction Costs, No Conversion Beyond 1,044 Ft	337.4	72.7	264.7	377.1	145.6	231.4
Scenario 9a — Basin-level Targets with 10% BMP Transaction Costs, Minimum Of 30,000 Converted Wetland Acres	224.7	89.1	135.6	307.8	237.7	70.1
Scenario 9b — Basin-level Targets with 10% BMP Transaction Costs, Minimum Of 60,000 Converted Wetland Acres	231.6	89.1	142.5	312.1	235.1	77
Scenario 10 — Bay-Wide N Reduction Target (No River Basin level Targets)	163	87.8	75.2	249.9	243.5	6.4



Figures ES-5, ES-6, and ES-7. Optimization results: nitrogen, phosphorus, and sediment load reductions for Scenario 2(a).

Another notable result is that, *given the data and assumptions used in the analytic framework*, many of the optimization solutions involve a substantial portion of agricultural acreage being taken out of production. The optimization analytic framework often selects these kinds of projects (in particular, natural revegetation and conversion to forest) because they provide relatively large estimated load reductions at a relatively low estimated cost. For example, the Base Case Scenario 2(a), which costs roughly one-fifth as much as Scenario 7 that requires maximum upgrades of WWTPs, takes 1.6 million acres of cropland and 0.4 million acres of pastureland out of production (about 22% of all agricultural land in the Chesapeake Bay watershed).

These results regarding agricultural land conversion must be interpreted with caution. In particular, the analysis does not include the full social costs of taking working agricultural lands out of production. What the analysis does include are the direct costs to agricultural producers, which is a large portion of costs. However, it does not consider the indirect economic effects of economic productivity created by the agricultural sector, nor the social values associated with being able to buy affordable local produce or view farmland. Nor does it consider the environmental impacts of new land that might be brought into production outside the watershed and the externalities that would generate elsewhere. These and other considerations would need to be included before the analytical results could be interpreted as a truly socially optimal solution.

Nevertheless, as represented in **Figure ES-8**, the analysis illuminates potentially important tradeoffs between agricultural production and a competing set of ecosystem services. Point D represents conditions with no regulations or government incentives for taking land out of production. In this case, some agricultural producers opt to forego production activities and use the land to support other ecosystem services (e.g., by leasing land for hunting); however, the overall level of production is very high and bonus ecosystem services are relatively low.

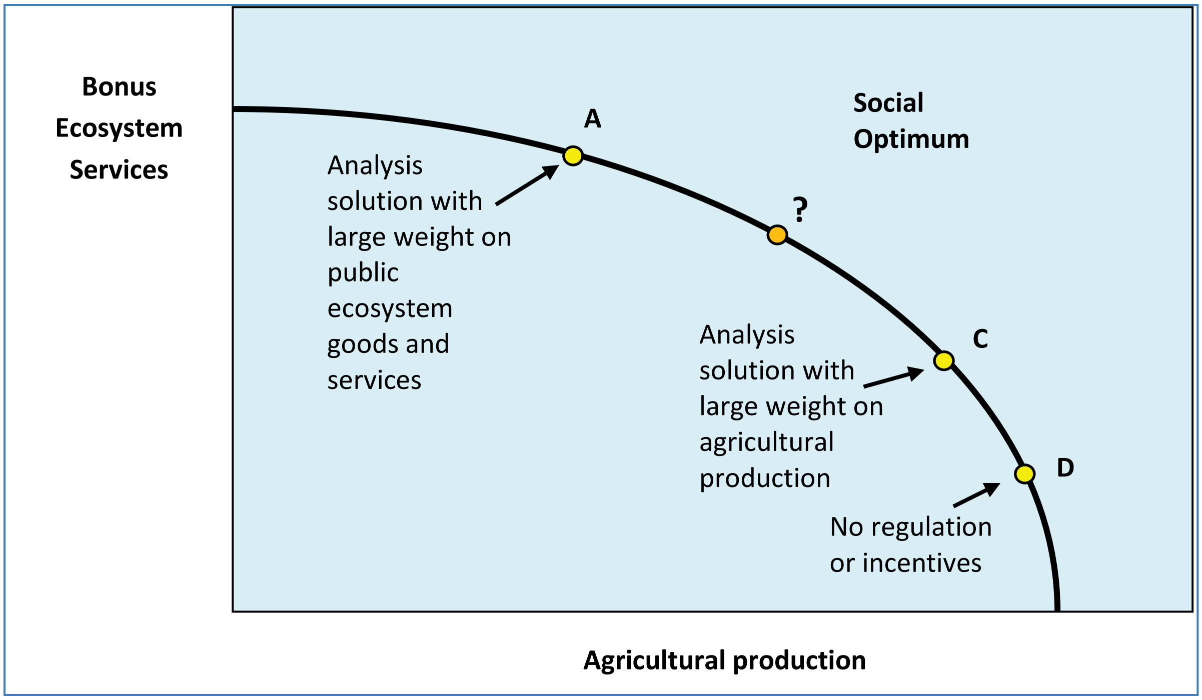


Figure ES-8. Representation of potential tradeoffs between agricultural production and bonus ecosystem services in the Chesapeake Bay watershed.

Points C and A represent conditions when agricultural producers are encouraged to forego even more production, as a way to achieve the TMDL required reductions in nutrient and sediment loads. For Point C, a relatively high emphasis (i.e., weight) is placed on maintaining agricultural production levels, and a relatively low weight is placed on bonus ecosystem services. For example, Point C could be interpreted as a representation of the least-cost solution to Scenario 2(c), which includes a 120% upward adjustment to the opportunity costs of agricultural land. Alternatively, it could be interpreted as the least-cost solution to Scenario 8(a), which disallows agricultural land conversion beyond the 100 ft buffer area. In contrast, Point A assumes a relatively high weight on bonus ecosystem services and a relatively low weight on agricultural production. For example, Point A could be interpreted as a representation of the least-NET-cost solution to the Base Case Scenario 2(a). This scenario directly accounts for bonus ecosystem services in the optimization analysis, but it places no additional restrictions or disincentives on agricultural land conversion.

As shown in Figure ES-8, the true socially optimal point might lie between points A and D; however, without full information on the relevant costs and benefits, the current framework cannot identify this point. The scenarios examined are not meant to suggest a socially preferred solution, but rather to provide points on the tradeoff curve in order to better understand where the optimal mix may lie. In other words, they demonstrate how the framework can be used to illustrate and examine the tradeoffs between agricultural production and bonus ecosystem services, when evaluating options for achieving TMDL requirements.

ES-3 LIMITATIONS AND FUTURE DIRECTIONS

As demonstrated by the results from the varied scenarios, the analytic framework described in this report provides a rich framework for analyzing economic implications and tradeoffs associated with alternative nutrient- and sediment-control strategies in the Chesapeake Bay watershed. However, by necessity and design, it also provides a simplified representation of the options and trade-offs involved in meeting selected pollution-reduction targets for a large watershed. Due to data and resource limitations and to make the analysis tractable, a number of simplifying assumptions have been made in developing the framework. Some of the key limitations of the framework include the following:

- The analytic framework does not include some potentially important sources of nutrients and sediment in the watershed. For point sources, it does not include CAFOs or a number of smaller treatment facilities and animal operations, nor does it include septic systems. For nonpoint sources, it does not include active construction sources.
- The framework does not include all of the possible pollution-control measures available in the watershed. For example, due to lack of data on costs, effectiveness, or potential locations, BMPs such as conservation planning, decision agriculture, erosion and sediment control, and stream restoration are not included. The lack of coverage for certain source categories and pollution-control measures implies that some technically feasible and potentially low-cost alternatives are not included. For this reason, the results described in this report may overstate the minimum costs of achieving the TMDL goals.
- The framework also does not quantify all the possible bonus ecosystem services associated with nutrient-control measures. For this reason, the framework most likely underestimates the total potential effects of including bonus ecosystem services in the optimization framework.
- Some of the bonus ecosystem services in the analytic framework are measured in monetary terms. These estimates are mainly designed to represent their societal value, but

they may also be interpreted as potential payments to project providers for supplying the ecosystem services. In this report, we do not evaluate specific policy mechanisms for funding these types of payments; however, the framework does provide a framework for exploring these policy issues.

- The estimates of control costs, pollutant-removal effectiveness, and bonus ecosystem services are often based on limited data and are subject to uncertainty. For example, the opportunity costs of agricultural land conversion are based on county-level average land rental rates, which are rough approximations and perhaps underestimates of the true opportunity costs.
- The framework provides a fundamentally static representation of alternative outcomes within the watershed. It does not include dynamic adjustments, staged implementations of BMPs, or assumptions about other future changes in demographics or urbanization.
- The estimated impacts are confined to the Chesapeake Bay watershed and the directly affected entities within the watershed; therefore, they do not account for possible leakage or ripple effects into other areas or sectors. For example, the carbon sequestration resulting from natural revegetation or conversion of agricultural land to forest within the watershed may be offset by increased land clearing and carbon releases in other areas (e.g., to meet food demand).
- The framework is designed to make as much use as possible of the data and modeling framework provided by the Chesapeake Bay Program's Phase 5.3 Community Watershed Model (CBWM). Although the CBWM system offers several important advantages, it also has its own limitations that, in turn, affect our framework.

The analytic framework is also a work in progress. As shown below, a number of potential extensions are being considered to further strengthen and broaden this framework and to address existing limitations:

- Investigating methods for including additional control projects in the framework, such as strategies for controlling sediment runoff from construction sites in the watershed and reducing runoff from CAFOs. Further investigation into the role of atmospheric deposition could also be conducted.
- Modifying the framework to include regional- or state-level load-reduction targets within the basin-level framework currently being considered, or the framework could account for

the hydrologic role of each tributary in pollutant retention in the Bay (Scenario 3) and, in turn, eutrophication or other impacts in applying restrictions on pollutant reductions.

- Adding more ecosystem service effects to the analytic framework, including the GHG emission changes associated with point-source controls, cover crops, and other nutrient management strategies.

SECTION 1. INTRODUCTION

1.1 BACKGROUND AND PURPOSE

Many of the nation's watersheds and estuaries are suffering from the effects of high pollutant loads. Nationally, over 40,000 individual water bodies are listed as impaired under Section 303(d) of the Clean Water Act (CWA), and according to a recent national assessment of 141 major estuaries, 13% exhibited high levels of eutrophication (NOAA, 2007) due to heavy nutrient loads.

One of the main mechanisms for addressing water quality impairment problems under the CWA is through the establishment of total maximum daily loads (TMDLs), which limit the allowable amount of pollutant loads that a water body can accept. Since 1995, the U.S. Environmental Protection Agency (EPA) has approved almost 44,000 TMDLs. Despite this progress, meeting TMDL limits often presents an important set of "tradeoffs" regarding where and how to control upstream pollutant sources. For example, relatively large and easily verifiable nutrient load reductions can often be achieved through controls of point-source discharges; however, these controls also may entail relatively large costs and minimal reductions in sediment loads. In contrast, nonpoint-source controls are generally more diffuse and difficult to monitor; however, they may offer lower-cost options, more sediment control, and other ecological benefits.

In 2009, the EPA's Ecosystem Services Research Program (ESRP) embarked on a project to develop a modeling framework that would assist policymakers in evaluating these types of tradeoffs. This framework was specifically designed to incorporate measures of both the cost-effectiveness and ecosystem service impacts associated with individual pollution-control projects. The inclusion of ecosystem services is a unique feature of this framework, and it accounts for not only the targeted pollutant reductions, but also the ancillary societal benefits (i.e., "bonus" ecosystem services) that are provided by certain pollution-control measures. For example, riparian forest buffers not only reduce nutrient runoff to streams, they also sequester carbon, thereby increasing biomass. When the ancillary ecosystem benefits of a control measure are expressed in monetary terms, they also can be thought of as offsetting some of the costs of pollution control. This deduction of ecosystem service benefits provides an estimate of the NET costs (to society) of the control option.

The framework includes a detailed inventory of pollutant sources and control measures—referred to as “projects” in this report—and an optimization model that is designed to address the following research questions:

- What combination of point-source and nonpoint-source pollution controls achieves total pollutant-load limits at the lowest total cost?
- What are the cost and bonus ecosystem service implications of relying more heavily on certain types of agricultural and urban stormwater controls (rather than point-source controls) to meet the total load limits?
- How does the inclusion of bonus ecosystem services in the model affect the optimal choice of point- and nonpoint-source controls?

The modeling framework also provides a structure for examining some of the implications of market-based strategies or payment systems for nutrient reductions or other bonus ecosystem services.

The purpose of this report is, therefore, to present a method to determine (1) how to most cost effectively achieve water quality goals in an impaired watershed, and (2) how the consideration of ecosystem services might change the optimal solution and/or change the net cost of achieving water quality goals. Further, the report is intended to describe the modeling framework and to demonstrate how it can be applied in a large, complex watershed.

To demonstrate the framework, we use the Chesapeake Bay watershed and its recently established TMDL as a case study application (details regarding the Chesapeake Bay context are provided ***Section 2***). It must be noted that, due to time and resource constraints for model development, the model provides a somewhat simplified representation of the sources and control measures available in the watershed. These limitations also apply to the estimates of costs, pollutant-load reductions, and ecosystem services associated with these options. As a result, this framework is not currently suitable for examining very specific and detailed policy options. Nevertheless, *given these constraints*, we show that the framework can provide a number of useful insights into the more general tradeoffs (including costs and ecosystem service impacts) associated with meeting TMDL targets.

1.2 OVERVIEW OF THE ANALYTICAL APPROACH

The analytical approach described in this report is designed to explore the implications—the costs and ecosystem service benefits—of alternative mixes of "green" and "gray" infrastructure to achieve water pollution load-reduction targets. Green infrastructure represents pollution-control practices that have the potential to naturally filter out nutrients and sediment, while also providing added ecosystem services such as water storage and carbon sequestration. Examples of green infrastructure include riparian buffers and restored wetlands. Gray infrastructure refers to common wastewater and stormwater controls, such as publicly owned treatment works (POTW) upgrades and many municipal separate storm sewer system (MS4) pollutant-control technologies that typically do not provide these types of ancillary benefits. While it is possible for some gray infrastructure projects to provide bonus ecosystem services and for some green infrastructure projects to have negative environmental impacts, this analysis focuses on green infrastructure's potential to provide bonus ecosystem services.

The development and application of this analytical approach involved seven steps, which are represented in **Figure 1-1** (see ***Section 3*** and the appendixes for details).

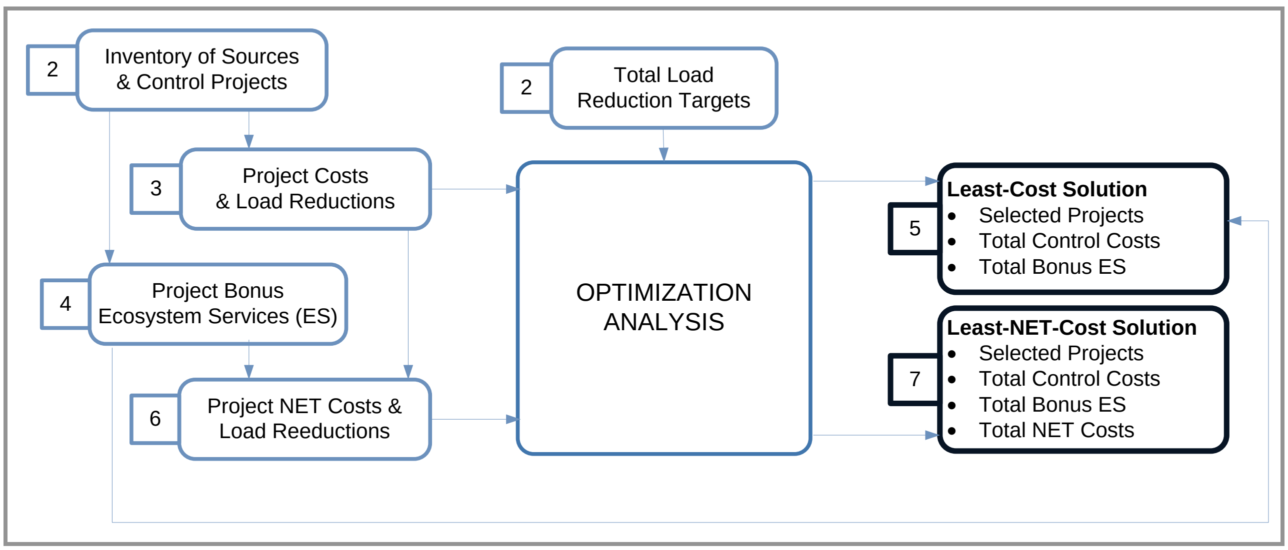


Figure 1-1. Flow diagram representation of the analytical approach.

Step 1. Define the aggregate nutrient- and sediment-load reduction targets of interest. In this case study, the targets were based on the basin-specific load allocations defined by the Chesapeake Bay TMDL.

Step 2. Create a spatial inventory of the main point and nonpoint sources in the watershed and identify control projects (gray or green infrastructure) for reducing nutrient and sediment loads from these sources. By combining this information, we created a spatial inventory of available pollution-control "projects" in the watershed.

Step 3. Develop estimates of the annual costs and effectiveness of each pollution-control project. In this context, effectiveness is measured as the annual decrease in pollutant load delivered to the tidal waters of the Chesapeake Bay.

Step 4. Develop estimates of the bonus ecosystem services associated with each of the pollution-control projects. As shown in Figure 1-1, pollutant-load reductions from the gray and green projects provide the core ecosystem services of interest. However, other *bonus* ecosystem services can also be provided, depending on the mix of control projects selected. Therefore, as feasible, we developed methods to quantify the effects of pollution-control projects on a number of *bonus* ecosystem service indicators. Due to data and resource constraints, this quantitative analysis is limited to the ecosystem service indicators shown in **Figure 1-2**.[1] Where possible, these ecosystem service indicators are also measured in monetary terms. This monetization is designed to represent the societal value of the service or a potential payment to the water pollution control project's provider for supplying the ecosystem service. These values are estimates of what might be appropriate to fund with public money or payments or that might be available from individuals or firms under some policy scenarios; however, in this report, we do not evaluate the mechanisms necessary for providing such funds.

[1] As noted in ***Section 2-5,*** several other potential bonus ecosystem services associated with these projects are not quantified in this analysis.

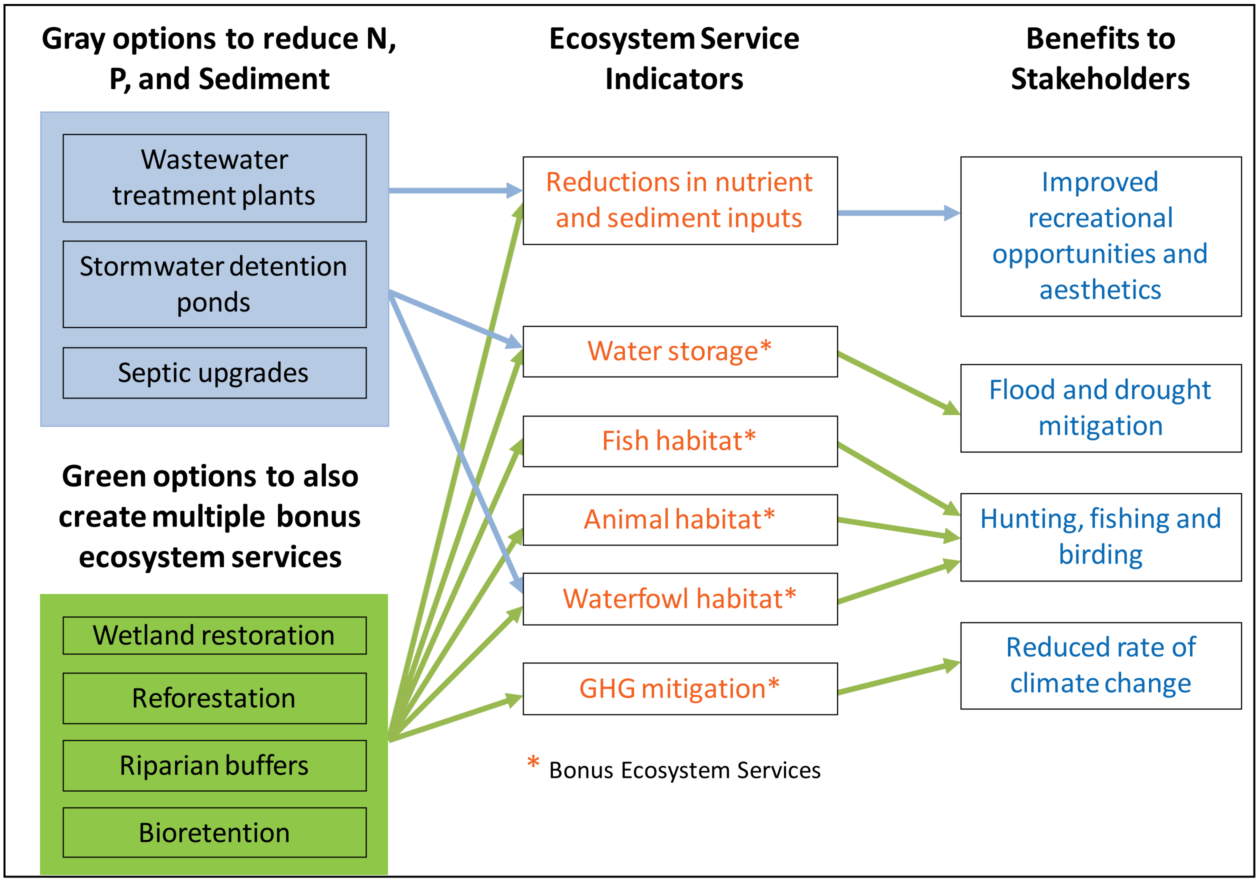


Figure 1-2. Gray vs. green infrastructure pollution controls, associated ecosystem services, and stakeholder benefits.

Step 5. Apply an optimization model to identify the combination of gray and green treatment projects that achieves the specified nutrient and sediment load-reduction targets in each basin at the lowest total cost. Using the General Algebraic Modeling System (GAMS), the objective is to minimize the combined costs (summed across pollution-control projects) of reducing loads to the Chesapeake Bay, subject to the constraints that the resulting combined load reductions for nitrogen, phosphorus, and sediment must be greater than or equal to those specified by the TMDL.

The model output from this optimization process includes (1) a list of selected cost-minimizing projects, (2) an estimate of the aggregate costs of meeting the load-reduction targets, and (3) an estimate of aggregate bonus ecosystem services derived from the selected projects. The cost and ecosystem service estimates also can be segregated according to project type and geographic location.

Step 6. Estimate the NET costs associated with each project for all projects in the inventory. The concept of NET costs for this analysis is represented in **Figure 1-3**. In short, NET costs are equal to costs of the project minus the monetized bonus ecosystem service benefits provided by the project. It is important to emphasize that, unlike the costs, the ecosystem service benefits derived from control projects will often accrue to individuals other than the owner of the project. In other words, they are often *external* benefits that accrue to society, but not necessarily to the project owner. The presence of bonus ecosystem services does not reduce pollution-control costs for project owners, but it does offset some of the losses to society resulting from certain pollution-control measures[2]. As a result, the NET cost estimates represent net costs *to society* of pollution-control projects.

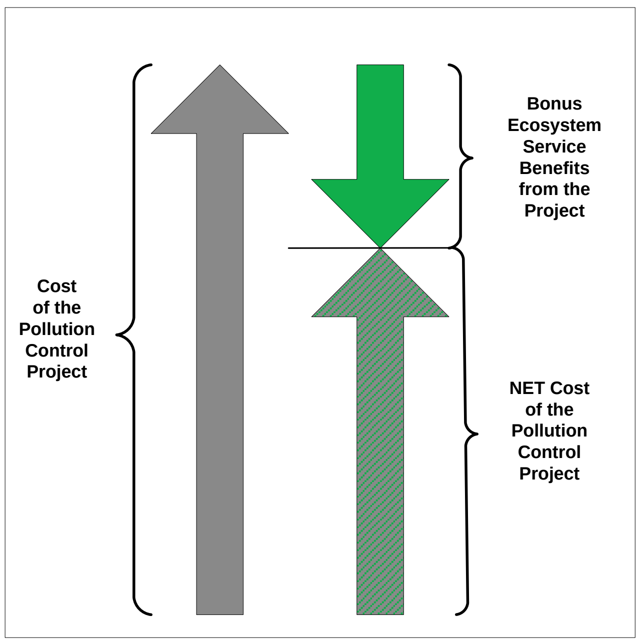


Figure 1-3. Relationship between costs, bonus ecosystem services, and NET costs of pollution control.

Step 7. Re-run the optimization model to identify the combination of gray and green treatment projects that achieves the specified nutrient and sediment load-reduction targets in each basin at the lowest total NET cost. In this case, the objective is to minimize the aggregate NET costs, subject to meeting the load-reduction targets. The GAMS model output from this optimization process includes (1) a list of selected NET-cost minimizing projects,

[2] Unless a market or payment system exists to compensate owners for providing these services, their benefits are not likely to be fully factored into a project owner's incentives or decision making.

(2) estimates of the aggregate costs and NET costs of meeting the load-reduction targets, and (3) an estimate of aggregate bonus ecosystem services derived from the selected projects.

1.3 KEY ASSUMPTIONS AND LIMITATIONS

By necessity and design, the framework described in this report provides a simplified representation of the options and trade-offs involved in meeting selected pollution-reduction targets for a large watershed. Due to data and resource limitations and to make the analysis tractable, a number of simplifying assumptions have been made in developing the framework. These assumptions and their implications are described throughout this report; however, as part of this introduction, it is important to stress a few main caveats.

First, the model includes the largest sources of nutrients and sediment in the watershed, but not all of the sources. For point sources, it includes the largest and most significant wastewater treatment plants (WWTPs); however, it does not include concentrated animal feeding operations (CAFOs) or a number of smaller treatment facilities and animal operations, nor does it include septic systems. For nonpoint sources, it includes the main agricultural and urban sources (cropland, pastureland, and urban post-construction stormwater sources); however, it does not include active construction sources[3].

Second, the model includes many but not all of the possible pollution-control measures available in the watershed. For point sources, it includes the main control technologies available at large WWTPs. For nonpoint sources, it includes five urban stormwater and nine agricultural best management practices (BMPs). However, the lack of coverage for certain source categories and pollution-control measures implies that some technically feasible and potentially low-cost alternatives are not included in the model's cost-minimization results. For this reason, the results described in this report may overstate the minimum costs of achieving the TMDL goals, and the exact mix of selected practices may be sub-optimal.

Third, due to data and resource limitations, the model also does not quantify all of the possible bonus ecosystem services associated with nutrient control measures. It does include monetary estimates for carbon sequestration, selected recreational hunting benefits, and atmospheric pollutant removal, and it includes quantitative indicators of water storage and brook

[3] Although stormwater dischargers are considered point sources by EPA for permitting purposes, urban runoff is more generally considered to be nonpoint source of pollution (http://water.epa.gov/polwaste/nps/categories.cfm). In this report, we classify urban stormwater as a nonpoint source of nutrients and sediment.

trout habitat changes. However, as discussed in ***Section 2***, a wide variety of other potential services are not accounted for in the framework. For this reason, the model most likely underestimates the total potential effects of including bonus ecosystem services in the optimization framework.

Fourth, the estimates of control costs, pollutant-removal effectiveness, and bonus ecosystem services included in the model are often based on limited data and are subject to uncertainty. The cost and removal effectiveness estimates are based on a limited number of existing studies of control techniques and generally represent average or best mid-range estimates[4]. As a result, they incorporate relatively little spatial or other sources of variation in costs or removals. For agricultural BMPs that require removing land from production, the associated land cost estimates are based on county-level average rental rates for agricultural land. Similarly, to quantify bonus ecosystem services such as carbon sequestration, recreational hunting, and water storage, we transfer models and study results from other areas and adapt them to the Chesapeake Bay context. All of the methods used to value changes in ecosystem services rely on relatively simple "benefit transfer" techniques (Boyle and Bergstrom, 1992). That is, they apply unit value estimates (e.g., value per unit of carbon sequestered, value per recreation day, value per ton of pollutant removed), which are drawn from summary studies of the empirical nonmarket valuation literature. None of these value estimates were specifically developed for the Chesapeake Bay watershed; however, in our judgment, they provide the best *available* estimates for valuing bonus ecosystem services from land conversion in the watershed.

Fifth, by design, the model does not include monetary estimates for ecosystem services that accrue directly from reductions in nutrient and sediment loads to the Bay (i.e., it does not quantify "non-bonus" ecosystem services). If the purpose of the analysis were to estimate and compare the net gains to society (i.e., "total value") of different pollution-reduction targets, then it would be important to also monetize as many non-bonus ecosystem services as possible. However, for this analysis, the objective is to evaluate the implications of different strategies for

[4] One of the ways we address the uncertainty in the pollutant removal efficiencies for nonpoint source controls is to include "efficiency ratios" for trade-offs between point and non-point source controls. This approach is described in Section 4, and the results are reported in Section 5.

attaining one set of load-reduction targets.[5] The modeling framework could be expanded in the future to address broader total objectives, but these are beyond the current scope of this project.

Sixth, the model provides a fundamentally static representation of alternative outcomes within the watershed. It does not include dynamic adjustments, staged implementations of BMPs, or assumptions about other future changes in demographics or urbanization. Rather, it compares (1) a representation of current conditions with (2) a menu of alternative conditions involving instantaneously lower levels of loadings, higher annual costs, and different levels of annual bonus ecosystem services. This simplified representation tends to overstate the costs (i.e., present annualized value) of controlling current sources because, in practice, these controls will need to be phased in over several years. On the other hand, it understates control costs by not including the increase in loadings expected to result from future growth and urbanization in the watershed.

Seventh, the modeled impacts are confined to the directly affected entities within the Chesapeake Bay watershed; therefore, they do not account for possible leakage or ripple effects into other sectors or areas. For example, the carbon sequestration resulting from revegetation or conversion of agricultural land within the watershed may be offset by increased land clearing and carbon releases in other areas (e.g., to meet food demand). These secondary effects are not captured in the model. Similarly, changes in agricultural production due to land conversion may have ripple effects on incomes, jobs, and economic activity, both inside and outside the watershed. As land is taken out of crop production, the remaining cropland may increase in value, requiring larger payments to landowners for them to change practices as more nonpoint-source options are adopted. These indirect economic impacts are also not included in the modeled impacts.

In particular, the analysis does not include the full social costs of nutrient controls that result in large amounts of working agricultural lands being taken out of production. What the analysis does include are the *direct* costs to agricultural producers, which is a large portion of costs. However, it does not consider the indirect economic effects of economic productivity created by the agricultural sector, nor the social values associated with being able to buy affordable local produce or view farmland. Nor does it consider the environmental impacts of

[5] In other words, rather than developing a framework for cost-benefit analysis, we develop a framework that is more closely associated methods for cost-effectiveness analysis.

new land that might be brought into production outside the watershed and the externalities that would generate elsewhere. These and other considerations would need to be included before the analytical results could be interpreted as a truly socially optimal solution.

Eighth, the model described in this report is designed to make as much use as possible of the data and modeling framework provided by the Chesapeake Bay Program's Phase 5.3 Community Watershed Model (CBWM). Although the CBWM system offers several important advantages, it also has limitations that, in turn, affect our model. A recent review of the CBWM (Band et al., 2008) points to three particularly relevant issues: (1) the sediment and nutrient loading and transport models in the CBWM do not account for the effects of extreme weather events, (2) the model does not capture the potentially significant time lags between management actions and environmental responses, and (3) the model does not fully couple groundwater and surface water systems and, therefore, does not adequately capture nitrate loads via groundwater. According to a recent study of conservation practices in the Chesapeake Bay watershed (Conservation Effects Assessment Project [CEAP], 2011), "[t]he most critical conservation concern related to cropland in the region is the need to reduce nutrient losses from farm fields, especially nitrogen in subsurface flows"(p.6); therefore, the limited coverage of groundwater pathways in the CBWM is particularly noteworthy.

Finally, although the analysis presented in this report does not specifically address trading or offset programs for nutrient reductions or other ecosystem services, it does have potentially important implications for these types of programs. The model does not specify the policy mechanism(s) by which the cost-minimizing or NET cost-minimizing solutions are obtained. They could be obtained under direct regulations of point and nonpoint sources; through grants, subsidies, or other incentives to nutrient sources; or through offset and trading programs. However, trading and offset programs are of particular interest because, in principle, they can be designed to offer land owners, facility owners, and other parties the necessary flexibility and incentives to find and implement relatively low-cost approaches to meeting pollutant-reduction targets.

1.4. ORGANIZATION OF THE REPORT

The remaining sections of this report are organized as follows:

- ***Section 2*** provides a more detailed description of the background and issues associated with Chesapeake Bay's water quality impairments and TMDL. It also describes the ecosystem services affected by poor water quality in the Bay
- ***Section 3*** provides a summary description of the main methods and data used in the model. It describes (1) the source categories and the gray and green infrastructure projects included in the model for reducing nutrient and sediment loads, (2) the methods used to quantify the costs, pollutant-removal effectiveness, and bonus ecosystem services associated with alternative control projects, and (3) the optimization model used to solve of the least cost (and least NET cost) combination of control projects. More detailed descriptions of these methods are provided in the technical appendices to this report.
- ***Section 4*** defines the specific set of model scenarios, which are used in the following section to demonstrate the framework and to analyze the effects of alternative policies and model specifications.
- ***Section 5*** summarizes and compares optimization results for each of the model scenarios. These results include estimates of the total costs, load reductions, land-use changes, and bonus ecosystem services associated with each scenario.
- ***Section 6*** summarizes the main conclusions drawn from the model application and results.
- ***Section 7*** presents the references.

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SECTION 2. MODELING FRAMEWORK CASE STUDY: THE CHESAPEAKE BAY WATERSHED AND TMDL

In this proof-of-concept analysis, we applied the modeling framework to the Chesapeake Bay watershed and used the recently established TMDL policy as the context for a case study analysis. The Chesapeake Bay offers several advantages for this purpose, as shown below:

- The Chesapeake Bay is one of the most intensively studied watersheds in the country. As a result, a relative abundance of data and models are available as input to our framework. In particular, we make extensive use the data and structure provided by the Chesapeake Bay Program's Phase 5.3 CBWM, as discussed in ***Section 1***.
- The Chesapeake Bay is the largest and arguably the most ecologically diverse and economically important estuary in the country. These features alone make it an inherently interesting context for analysis.
- The Chesapeake Bay has recently been the subject of the EPA's largest and most complex TMDL setting process. For decades, the Bay has experienced significant water quality impairments, due primarily to excessive nutrient and sediment loads. A lack of progress in addressing these issues led to the issuance of Executive Order (EO) 13508 in May 2009, which gave the federal government and the EPA increased authority to coordinate and implement a strategy for restoring the Bay. As a key step in this strategy, the EPA established a multipollutant TMDL in December 2010. This TMDL defined the maximum allowable annual loadings of nitrogen, phosphorus, and sediment to the Chesapeake Bay and its tidal tributaries. In this report, the model seeks to attain constraints that are based directly on the TMDL limits for all three pollutants concurrently in each of the eight major basins within the Chesapeake Bay watershed.

> The 2010 Bay TMDL sets nitrogen, phosphorus, and sediment-loading targets for the major tributaries to the Bay to be achieved by 2025. These targets are pollution limits necessary to meet applicable water quality standards in the Bay and its tidal rivers and embayments. Specifically, the TMDL sets Bay watershed limits that will result in a 25% reduction in nitrogen, a 24% reduction in phosphorus, and a 20% reduction in sediment.

- The broader goals of EO 13508 include the protection and restoration of aquatic and terrestrial ecosystems throughout the watershed. One of the main objectives of this modeling framework is to examine how different pollution-control strategies contribute to these protection goals through the enhancement of bonus ecosystem services.

The following sections provide an overview of the Chesapeake Bay, highlighting features analyzed in this case study.

2.1 THE CHESAPEAKE BAY WATERSHED AND WATER QUALITY IN THE ESTUARY

The Chesapeake Bay is the largest and arguably the most economically important estuary in the United States. The water surface area of the mainstem and tidal tributaries of the Bay is roughly 4500 square miles. The drainage area for the estuary—the Chesapeake Bay watershed —is shown in **Figure 2-1.** This area extends over six states (Delaware, Maryland, New York, Pennsylvania, Virginia, and West Virginia) plus the District of Columbia, and it covers 64,000 square miles. The overall watershed can be separated into eight distinct basins, which are also shown in Figure 2-1. The largest of these basins is the Susquehanna, which accounts for 43% of the watershed area and roughly half of the freshwater inflow to the Bay.

The health and water quality of the Chesapeake Bay estuary has been a major concern for decades. In particular, heavy nutrient and sediment loads from across the watershed have contributed to low oxygen levels; algal blooms; decreased water clarity; loss of submerged aquatic vegetation (SAV); declines in fish and shellfish abundance; and fish kills. As a result, most of the Chesapeake Bay and tidal tributaries have been listed as impaired waters for several years under Section 303(d) of the CWA. These impairments have, in turn, required the establishment of a TMDL for the Chesapeake Bay.

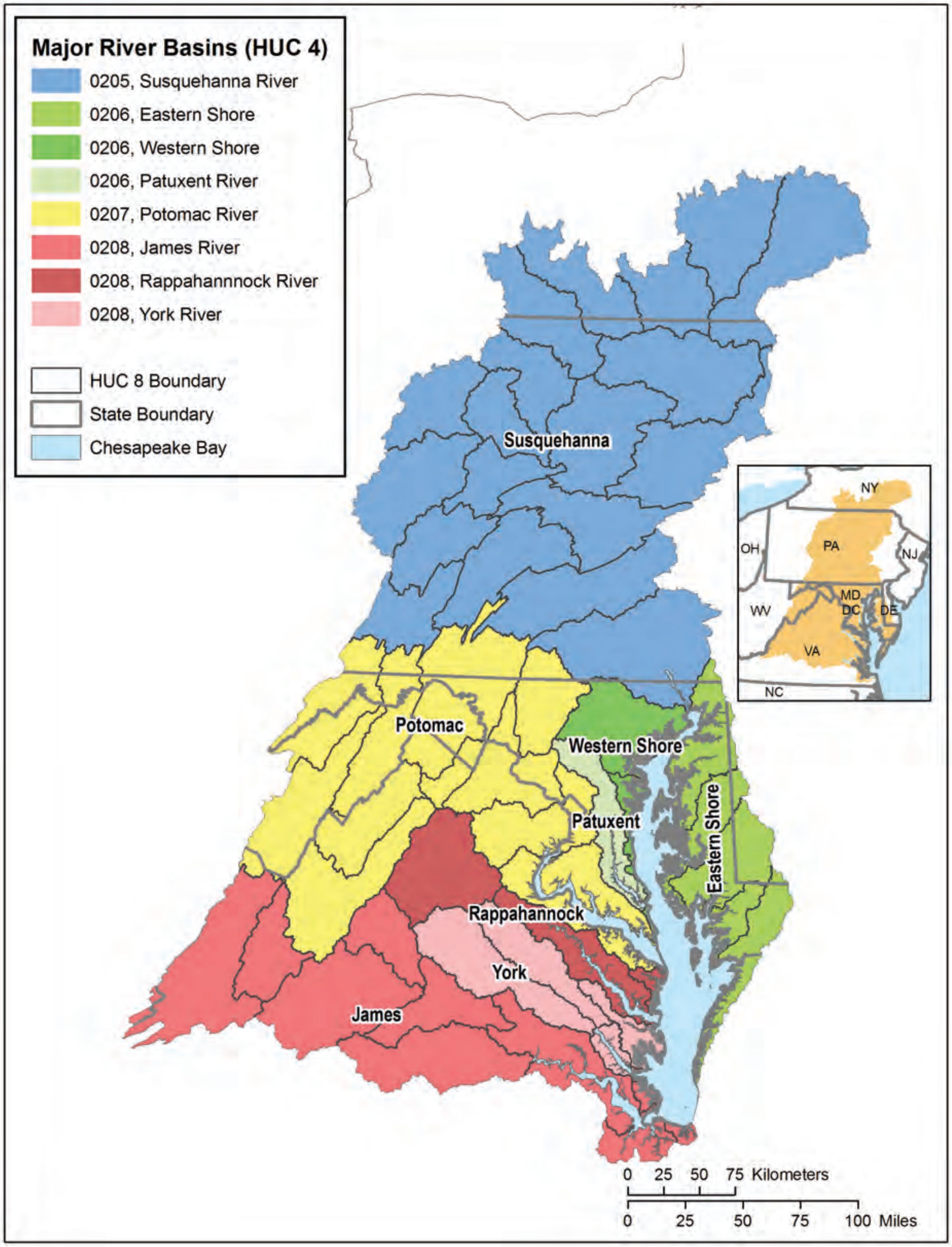

Figure 2-1. The Chesapeake Bay watershed and its major river basins.

2.2 MAIN SOURCES OF NUTRIENTS AND SEDIMENT IN THE CHESAPEAKE BAY WATERSHED

The main sources of nutrients and sediment in the Chesapeake Bay watershed are shown in **Figure 2-2** and profiled in the EPA's *The Next Generation of Tools and Actions to Restore Water Quality in the Chesapeake Bay* (U.S. EPA, 2009). These sources include the following:

- **Agriculture—**The watershed hosts an estimated 87,000 farms that occupy about 8.5 million acres, or 25% of the watershed. Nutrient and sediment sources from agriculture include fertilizer application, erosion resulting from tillage and irrigation, and manure generated at animal feeding operations (AFOs).
- **Urban and Suburban Lands—**The population of the Chesapeake Bay watershed is about 17 million. Development ranges from small subdivisions to large municipalities. Population growth and land development generate runoff from construction, landscaping, and impervious surfaces, carrying sediment and nutrients from fertilizer and animals. Due to continued population growth and associated development, developed lands are the only pollution source that is increasing in the Bay despite control strategies.
- **Wastewater—**Although current pollution-reduction technologies used by the major municipal and industrial WWTPs remove significant amounts of nitrogen and phosphorus, they are not sufficient to protect the Bay (U.S. EPA, 2007). As a result. WWTPs are being upgraded to perform higher levels of treatment, but the costs of these measures are also significantly higher. Additional wastewater treatment capacity will be needed as the population of the watershed continues to grow. Septic systems also continue to increase the load to the waters upstream of the Bay.

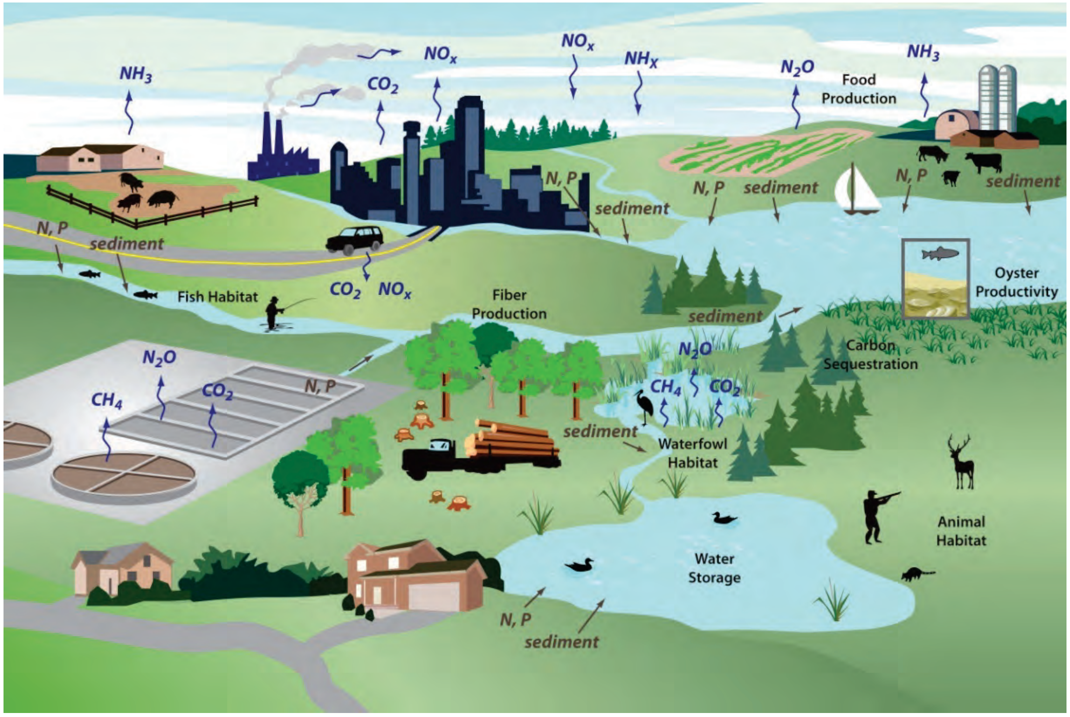

Figure 2-2. Overview of Chesapeake Bay watershed resources, pollution sources, and ecosystem services.

- **Atmospheric Emissions and Deposition**—In addition to releasing nutrients via runoff and groundwater seepage, AFOs, WWTPs, mobile sources, and industrial sources also emit oxidized and reduced nitrogen to the atmosphere, both of which contribute to the atmospheric chemistry that leads to the deposition of ambient nitrogen species in the environment. Reduced atmospheric nitrogen species include ammonia (NH_3) and ammonium ion (NH_4^+), the sum of which is referred to as reduced nitrogen (NH_x). Nitrous oxide (N_2O) is generated from both anthropogenic and biogenic sources. The spatial coverage of the Chesapeake Bay's nitrogen airshed is significantly greater than the watershed. As a result, approximately half of the nitrogen deposited in the watershed originates from emission sources outside of the watershed.

2.3 EXECUTIVE ORDER (EO) 13508 — CHESAPEAKE BAY PROTECTION AND RESTORATION

Despite the efforts of the EPA, the states, and other parties involved in the Chesapeake Bay Program, by 2009, progress continued to be slow for developing a comprehensive plan to reduce nutrient and sediment loads to the Bay and improve water quality conditions. In response, the President issued EO 13508 in May 2009, which called on a Federal Leadership Committee chaired by the Administrator of the EPA to restore and protect the Chesapeake Bay and its tributaries in the watershed. In partnership with the Chesapeake Bay watershed's jurisdictions and other federal agencies, the EPA's primary role has been to establish TMDLs for nitrogen, phosphorus, and sediment to the Bay to meet water quality standards for dissolved oxygen, clarity/SAV, and chlorophyll-*a*.[6]

The broader goals of EO 13508, for which other federal agencies have the lead role, include effective management of agricultural and forest lands; restoration of fish, game, and bird habitat and populations; preservation of treasured landscapes; and better management of federal lands in the watershed.[7] Many of the performance targets associated with EO 13508 reflect ecosystem services (Daily et al., 1997; NRC, 2005) that are not only important and valuable to residents of the Bay states, but also may generate revenue streams for landowners and developers who are willing to implement BMPs or participate in conservation programs that promote such services, while at the same time reducing nutrient and sediment loading to the Bay.

2.4 THE CHESAPEAKE BAY TMDL

As part of its commitment to meet the strategy articulated in EO 13508, the EPA established a Chesapeake Bay TMDL in December 2010 (U.S. EPA, 2010a). The TMDL sets nutrient- (i.e., nitrogen and phosphorus) and sediment load targets for the major tributaries to the Bay to be achieved by 2025. These targets are pollution limits necessary to meet applicable water quality standards in the Bay and its tidal rivers and embayments. Specifically, the TMDL sets Bay watershed limits of 185.9 million pounds of nitrogen, 12.5 million pounds of phosphorus, and 6.45 billion pounds of sediment per year—a 25% reduction in nitrogen, 24% reduction in phosphorus, and 20% reduction in sediment. These pollution limits are further divided by jurisdiction and major river basin based on state-of-the-art modeling tools, extensive monitoring data, peer-reviewed science, and close interaction with jurisdiction partners.

Table 2-1 reports the final TMDL load allocations for nutrients and sediments, by state and by basin.

6 EPA's role also includes expanded rulemaking for stormwater and concentrated animal feeding operations (CAFOs) and stronger enforcement and compliance activities (Federal Leadership Committee, 2010).

7 These goals and the associated performance targets are documented in the Draft Strategy to Restore the Chesapeake Bay (Federal Leadership Committee, 2010).

Table 2-1. Chesapeake Bay TMDL Watershed Nutrient and Sediment Allocations by Major River Basin and Jurisdiction (U.S. EPA, 2010c)

Jurisdiction	Basin	Nitrogen: final allocations (million lbs/year)	Phosphorous: final allocations (million lbs/year)	Sediment: final allocations (million lbs/year)
Pennsylvania	Susquehanna	68.9	2.49	1,741.17
	Potomac	4.72	0.42	221.11
	Eastern Shore	0.28	0.01	21.14
	Western Shore	0.02	0.00	0.37
	PA Total	**73.93**	**2.93**	**1,983.78**
Maryland	Susquehanna	1.09	0.05	62.84
	Eastern Shore	9.71	1.02	168.85
	Western Shore	9.04	0.51	199.82
	Patuxent	2.86	0.24	106.3
	Potomac	16.38	0.9	680.29
	MD Total	**39.09**	**2.72**	**1,218.1**
Virginia	Eastern Shore	1.31	0.14	11.31
	Potomac	17.77	1.41	829.53
	Rappahannock	5.84	0.9	700.04
	York	5.41	0.54	117.8
	James	23.09	2.37	920.23
	VA Total	**53.42**	**5.36**	**2,578.9**
District of Columbia	Potomac	2.32	0.12	11.16
	DC Total	**2.32**	**0.12**	**11.16**
New York	Susquehanna	8.77	0.57	292.96
	NY Total	8.77	0.57	292.96
Delaware	Eastern Shore	2.95	0.26	57.82
	DE Total	2.95	0.26	57.82
West Virginia	Potomac	5.43	0.58	294.24
	James	0.02	0.01	16.65
	WV Total	**5.45**	**0.59**	**310.88**
Total Basin/Jurisdiction Final Allocation		**185.93**	**12.54**	**6,453.61**
Atmospheric Deposition Final Allocation		**15.7**	**N/A**	**N/A**
Total Basin-wide Final Allocation		**201.63**	**12.54**	**6,453.61**

[a] Cap on atmospheric deposition loads direct to Chesapeake Bay and tidal tributary surface waters to be achieved by federal air regulations through 2020.

Figures 2-3, 2-4, and 2-5 compare the final TMDL allocations for the entire watershed with historical annual loadings estimates from the CBWM (U.S. EPA, 2010a).

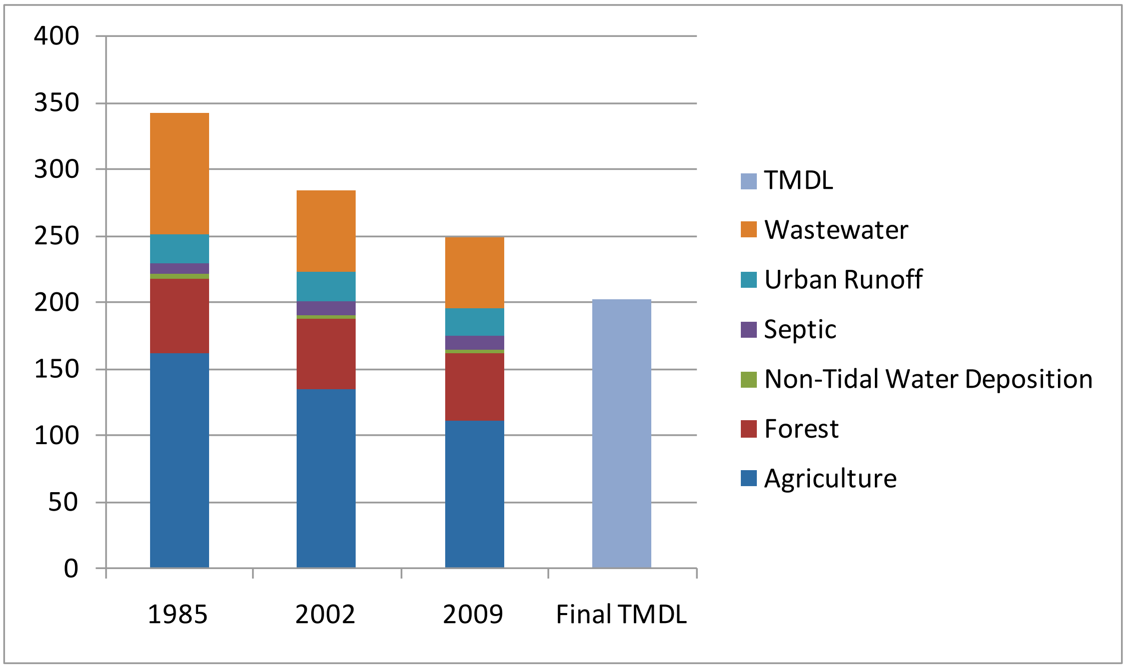

Figure 2-3. Annual nitrogen loads to the Chesapeake Bay, by sector (millions of lbs).

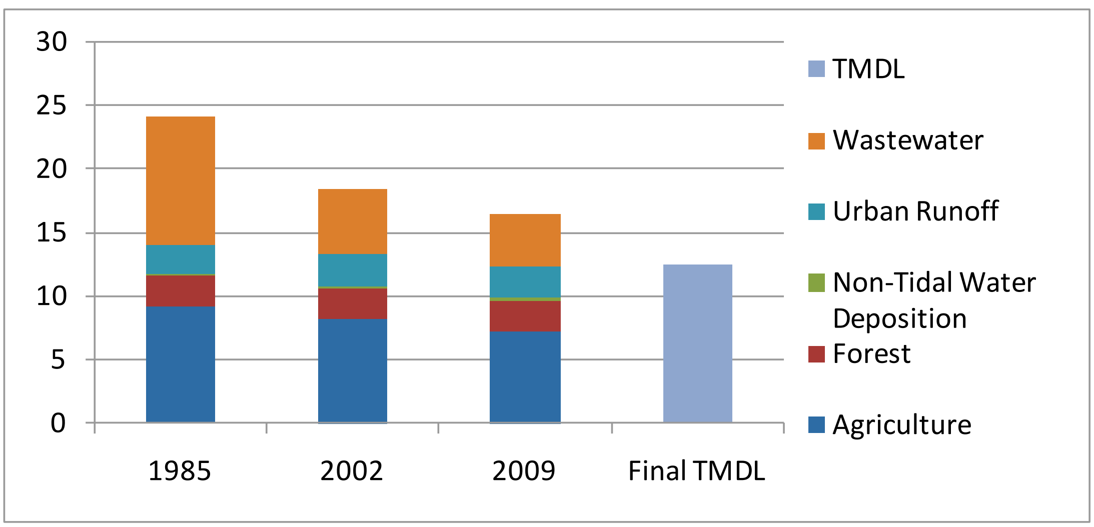

Figure 2-4. Annual phosphorus loads to the Chesapeake Bay, by sector (millions of lbs).

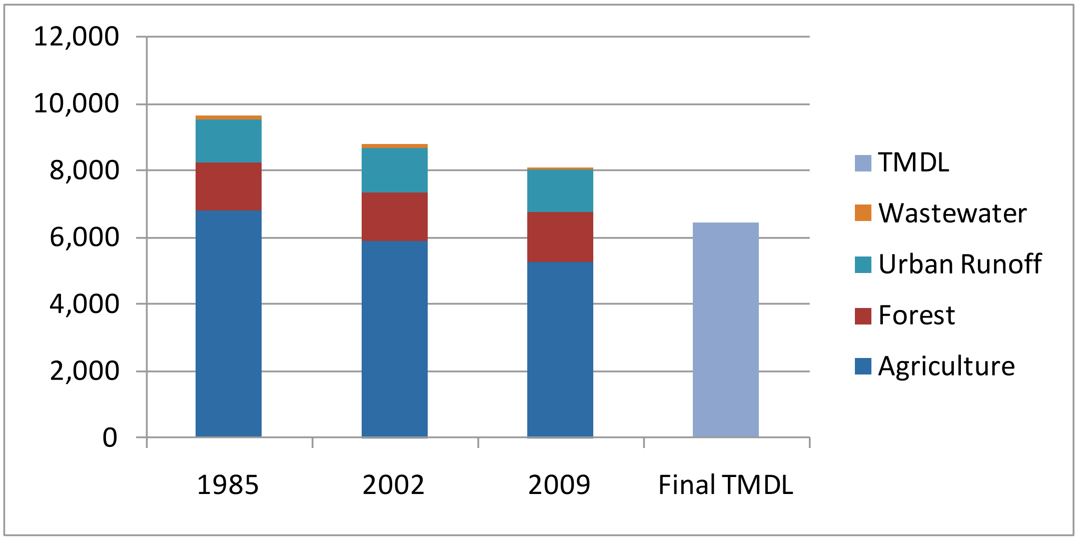

Figure 2-5. Annual sediment loads to the Chesapeake Bay, by sector (millions of lbs).

Watershed Implementation Plans (WIPs) that detail how and when the six Bay states and the District of Columbia will meet pollution allocations have played a central role in shaping the TMDL. The WIPs consider such things as ecological restoration and sustainability while allowing for greater transparency and accountability for improved performance. Each of the seven Bay watershed jurisdictions has created a WIP, which is intended to provide reasonable assurance that the jurisdiction will achieve and maintain water quality standards. The WIP documents how the jurisdiction will partner with federal and local governments to achieve and maintain water quality standards. The final WIPs enabled the EPA to reduce and remove most federal backstops set based on the draft WIPs, leaving a few targeted backstops and a plan for enhanced oversight and contingency actions to ensure progress[8]. The final TMDL is shaped in large part by the jurisdictions' plans to reduce pollution— a long-standing priority for the EPA and why the Agency always provided the jurisdictions with flexibility to determine how to reduce pollution in the most efficient, cost-effective, and acceptable manner.

The WIPs address a combination of regulatory and non-regulatory actions designed to control sources of nitrogen, phosphorus, and sediment pollution. The EPA has statutory authority to regulate point sources of nutrients and/or sediments, such as municipal and industrial WWTPs, MS4s, and CAFOs. The EPA also has statutory authority to set emission standards for

[8] The draft WIPs did not sufficiently identify the programs needed to reduce pollution or provide assurance that the programs could be implemented. As a result, the draft TMDL issued September 24, 2010, contained moderate- to high-level backstop measures to tighten controls on federally permitted point sources of pollution.

mobile and stationary sources of air pollutants. One beneficial effect of these standards is a decrease in atmospheric deposition of nitrogen to watersheds and open Bay waters.

The EPA has three levels of federal backstops: minor (modifying WIP wasteload allocations); moderate (more stringent modification of wasteload allocations); and high (completely rewriting of the WIP). The EPA's authority may limit effluent concentrations or loads of pollutants, or it may require that certain technologies or practices be applied to such sources. Although this is not always the case, controls of point sources of pollutants are most often associated with "gray" infrastructure, i.e., built infrastructure that is not likely to produce on-site ecosystem services, but may even reduce such services (e.g., by generating greenhouse gases [GHGs] during construction and operation, or by replacing natural habitat).

The EPA does not have statutory authority to regulate nonpoint sources such as agriculture and silviculture; "on site" wastewater treatment; and some types of stormwater runoff[9]; however, states and local governments often have such regulations, and other policy approaches do exist for encouraging nonpoint-source participation in nutrient-control strategies. Treatment of nonpoint-source runoff primarily involves BMPs and conservation strategies such as nutrient and manure management plans for farms; the installation of buffer strips at stream edges, rain gardens, and pervious surfaces to reduce stormwater runoff; the restoration of wetlands; and natural revegetation. Financial incentives for implementing these kinds of BMPs are provided through federal programs such as the Conservation Reserve Program or Wetland Restoration Program. In addition, Virginia, Pennsylvania, Maryland, and West Virginia have created (or are in the process of developing) nutrient trading programs. These programs allow certain point sources to reduce the costs of meeting their pollution-reduction requirements by purchasing equivalent nutrient reductions from other sources (including, in many cases, nonpoint sources). Of particular interest in this study is the potential for combining these incentive systems with markets or payment systems for other ecosystem services. In principle, combined systems would allow BMP projects to generate revenue from multiple separate ecosystem services. This type of "stacking" or "bundling" of ecosystem service payments would, for example, allow the owners of a restored wetland to sell credits not only for pollutant-load reductions but also for bonus ecosystem services, such as GHG reductions, wetland habitat, and water storage for flood mitigation. The desirability of allowing ecosystem service payments to be

[9] EPA does regulate stormwater runoff from construction and development activities.

stacked in this way is a source of debate and controversy (Fox et al., 2011); however, as long as separate payments are not earned for the *same* ecosystem service (i.e., no "double dipping" occurs), then stacking may be an appropriate way to incentivize nutrient- and sediment-control projects that provide multiple ecosystem services.

2.5 POTENTIAL ECOSYSTEM SERVICE BENEFITS FROM GREEN INFRASTRUCTURE PRACTICES IN THE BAY WATERSHED

In addition to reducing nutrient and sediment loads, agricultural and urban stormwater BMPs can play an important role in enhancing the Chesapeake Bay watershed's ability to support and provide ecosystem services. EPA's ESRP defines ecosystem services as "outputs of ecological functions or processes that directly or indirectly contribute to social welfare or have the potential to do so in the future."

To highlight the potential benefits of green infrastructure practices, **Table 2-2** identifies several types of ecosystem services that may be supported by land conversion involving riparian buffers, afforestation, or wetlands as pollution control measures. Using the framework developed by the Millennium Ecosystem Assessment (MEA, 2005), these services are divided into three main categories: provisioning, cultural, and regulating services[10]. In addition to improving water quality and enhancing ecosystem services in the Bay estuary, these land-use conversions are expected to augment other bonus ecosystem services in the upland watershed.

[10] The MEA framework is not the only system for categorizing ecosystem service and it has been criticized for not adequately distinguishing between "intermediate" and "final" services (Boyd and Banzhaf, 2007; Wallace, 2007); however, for this project it provides a useful and simple structure for organizing and identifying the main ecosystem services affected by green infrastructure practices.

Table 2-2. Summary of Key Ecosystem Services Derived from Riparian Buffers, Wetlands, and Forests in the Chesapeake Bay Watershed

Ecosystem Service Category	Riparian Buffers	Wetlands	Forests
Provisioning	Habitat for subsistence, commercially caught, and other consumed fish	Habitat for subsistence, commercially caught, and other consumed fish	Provision of wood and other forest products
Cultural	Recreation ▪ Fishing: habitat for sport fish ▪ Hunting: habitat for waterfowl ▪ Birding: habitat for birds Aesthetic ▪ Open-space benefits Nonuse ▪ Habitat for preserving wildlife and plant biodiversity	Recreation ▪ Fishing: habitat for sport fish ▪ Hunting: habitat for waterfowl ▪ Birding: habitat for birds Aesthetic ▪ Open-space benefits Education ▪ Wetlands education centers Nonuse ▪ Habitat for preserving wildlife and plant biodiversity	Recreation ▪ Hunting: habitat for deer and other game ▪ Hiking and other nature enjoyment Aesthetic ▪ Open-space benefits Education ▪ Educational forests Nonuse ▪ Habitat for preserving wildlife and plant biodiversity
Regulating	Carbon storage and other GHG regulation Soil regulation ▪ Erosion control ▪ Sediment retention Hydrological regulation ▪ Flood control ▪ Groundwater recharge Water quality regulation ▪ Nutrient removal ▪ Filtration	Carbon storage and other GHG regulation Soil regulation ▪ Erosion control ▪ Sediment retention Hydrological regulation ▪ Flood control ▪ Groundwater recharge/discharge Water quality regulation ▪ Nutrient removal ▪ Filtration	Carbon storage and other GHG regulation Soil regulation ▪ Erosion control ▪ Sediment retention Hydrological regulation ▪ Flood control ▪ Groundwater recharge Water quality regulation ▪ Nutrient removal ▪ Filtration Air quality regulation

2.5.1 Provisioning Services

Chesapeake Bay provides an abundance of aquatic life that can be harvested for consumption. In 2007, commercial fishing in the Chesapeake Bay region generated $52 million in value in Maryland and $131 million in Virginia (NOAA, 2007). Riparian buffers and wetlands can augment these provisioning services by providing and improving spawning habitat for anadromous and other fish. Chesapeake Bay forests provide wood and other forest products for consumption. Approximately $11 billion of total industry output within the Chesapeake Bay watershed comes from timber management, harvesting activities, and primary manufacturing of wood products, such as lumber and pulp (Sprague et al., 2006). Increasing forested land within the Bay watershed can further contribute to these provisioning services.

2.5.2 Cultural Services

Land conversion–related pollution control measures can also contribute to the recreational, aesthetic, educational, and nonuse values of the area. Hunting opportunities increase with increased forest and wetland habitat, but a number of other cultural ecosystem services are also supported. For example, riparian buffers and wetlands contribute to recreational fishing services by providing improved aquatic habitat. In a study evaluating the ecosystem response of northern Virginia streams to riparian buffers on agricultural land, Teels and colleagues (2006) found that buffers significantly improved both the physical condition and aquatic community when compared to sites without buffers. Riparian forests also provide support for fish habitat by introducing leaf litter and woody debris. Large woody debris (LWD) inputs increase habitat diversity and provide habitat for important prey for juvenile fish (Greene et al., 2009). The presence of woody debris also contributes to the suitability of aquatic habitat for the American shad and gamefish popular among Bay anglers (Bilkovic et al., 2002). By supporting and protecting these types of aquatic habitat, green practices also provide services for individuals who hold non-use values for these resources because, for example, they place value on the knowledge that healthy ecosystems are being better preserved for future generations.

Buffers, wetlands, and forests also provide habitat for birds and mammals that are important for recreational activities, such as hunting and bird watching. In a study relating the width of riparian forests to avian community structure on the Delmarva Peninsula, Keller and colleagues (1993) estimate that the number of neotropical migrant bird species in buffers that are 300 meters wide is twice as large as in buffers 25 meters wide. In addition, riparian buffers, wetlands, and forests are noted for their “open space” aesthetic services, and a large literature exists demonstrating and measuring the economic value of open-space services (McConnell and Walls, 2005).

2.5.3 Regulating Services

Many of the land conversions that would reduce nutrient and sediment loading to the Bay are expected to increase carbon capture and reduce GHG emissions relative to existing land uses, which attenuate the level of GHGs in the atmosphere. In addition, the land-use changes help to regulate water flow and thus provide benefits through flood control. Wetlands and bioretention planters/ponds, in particular, have been shown to support a wide variety of these services, including groundwater recharge and discharge and sediment retention (Turner, Georgiou, and Fisher, 2008).

Riparian buffers and forests help stabilize stream banks and prevent erosion. One study found that stream bank stabilization that included a forested riparian buffer prevented the loss of 0.2 hectares of land to erosion each year per 400 meters of bank length (Williams et al., 2004). Riparian forests also provide shade to streams, which reduces the maximum stream temperature and temperature volatility in streams. A study comparing stream temperatures following clear-cutting relative to a reference stream found that the clear-cut stream had higher maximum stream temperatures, as well as an increased diurnal variation in temperatures (Johnson and Jones, 2000). In urban areas in particular, vegetation-based buffers and bioretention planters/ponds also provide air quality regulation services by removing atmospheric pollutants and improving air quality.

SECTION 3. METHODS

3.1 SOURCES AND CONTROL PROJECTS FOR NUTRIENT AND SEDIMENT LOADS

3.1.1 Source and Control Projects Included in the Model

This subsection identifies and describes the management practices and technologies for control of nutrient and sediment loadings that are included in the modeling framework. The sources and related controls are grouped into three main categories: (1) point-source control technologies, (2) nonpoint-source agricultural BMPs, and (3) nonpoint-source urban stormwater BMPs.

3.1.1.1 Point-source Controls

The point sources included in the model are significant municipal and industrial WWTPs. There are a total of 402 significant municipal (≥ 0.5 million gallons per day capacity), 2,798 insignificant municipal, and 81 industrial WWTPs in the Bay watershed (U.S. EPA, 2009). Of these facilities, cost and effectiveness data were available for 332 significant municipal and 58 industrial WWTPs (CBP, 2002; 2004). Although significant WWTPs represent a relatively small percentage of all WWTPs in the Bay watershed, they account for a large majority of the total nitrogen and phosphorus loadings from these facilities. Based on 2000 data, WWTP categories discharged the following share of total WWTP loadings: significant municipal WWTPs accounted for 86% and 79% of TN and TP loadings, respectively; industrial WWTPs accounted for 13% and 20% of TN and TP loadings, respectively; and non-significant WWTPs accounted for 1% of both TN and TP loadings (CBP, 2002).

The model includes WWTP upgrades for enhanced nitrogen and/or phosphorus removal at these 390 significant WWTP facilities. Based on the technology classification system outlined in CBP (2002), four discrete tiers of nutrient reductions were defined, with Tier 4 representing the limits of technology. None of these tiers are defined to include green infrastructure elements, such as wastewater wetlands.

3.1.1.2 Agricultural BMPs

For this modeling framework, loadings from agricultural lands are divided into two main source categories: cropland and pastureland. These lands account for roughly 13% and 9% of the 64,000 square miles in the watershed, respectively. According to estimates from the CBWM P5.3, in 2009, cropland accounts for 33% of nitrogen and 25% of phosphorus loadings to the Bay, and pastureland accounts for 8% and 16%, respectively.

To account for nutrient- and sediment-control alternatives for these sources, the following nine agricultural BMPs are included in the model:

- **Riparian forest buffers—**Installing these buffers involves planting strips of trees on land located between a potential pollutant source (e.g., an agricultural field) and a body of surface water.
- **Riparian grass buffers—**Installing these buffers involves planting strips of grasses on land located between a potential pollutant source (e.g., an agricultural field) and a body of surface water.
- **Wetland conversion—**This practice involves returning agricultural land that was drained to allow crop and livestock production to their natural/historic function as wetlands.
- **Natural Revegetation—**This practice involves suspending agricultural activities and allowing the land to lie fallow and revert to a more natural vegetative cover of shrubs, grasses, and/or trees[11]. Although the land is being taken out of crop production, from an environmental standpoint, the land is still "working" to provide ecosystem services.
- **Conversion to forest—**This practice involves planting and nurturing trees to convert existing agricultural land to forest.
- **Livestock stream exclusion—**This practice involves establishing fences and other structures to exclude livestock from streams and other waterways (only applied to pastureland).
- **Cover crops—**This practice involves planting secondary crops (not for harvest) for soil enhancement and erosion prevention.
- **No-till agriculture—**This practice for growing crops excludes the step of tilling the soil, with the objective of increasing water and nutrient retention and reducing soil erosion.

[11] In accordance with the CBWM P5.3, retired land in the model is assumed to convert to the “hay-unfertilized” land-use category, which is described as “hay and other herbaceous agricultural areas that do not receive fertilizer and are not harvested.” (U.S. EPA, 2010c, p.4-26)

- **Reduced fertilizer application—**This practice involves reducing nitrogen applied to cropland as chemical and natural fertilizer, such that plant uptake is matched with nutrient availability. Referred to as "enhanced nutrient management" in the CBWM P5.3, this BMP is assumed to result in a 15% reduction in nitrogen application.

3.1.1.3 Urban Stormwater BMPs

Storm water from developed lands can be categorized as runoff that is discharged during active construction or post-construction. In this framework, we only consider post-construction stormwater runoff from urban, developed land. The relative contribution of active and post-construction storm water to loadings of total nitrogen, total phosphorus, and sediment to the Bay are shown in **Table 3-1**. Although active construction storm water contributes loadings at a higher rate per acre, post-construction storm water contributes a greater percentage of total loadings to the Bay.

Table 3-1. Loadings Delivered to the Bay in 2009 from Stormwater Runoff

Source of Stormwater Runoff	Acres of Land Use	Total Nitrogen			Total Phosphorous			Sediment		
		(lbs/yr)	(lbs/ acre/yr)	% of total load to CB	(lbs/yr)	(lbs/ acre/yr)	% of total load to CB	(tons/yr)	(tons/ acre/yr)	% of total load to CB
Active Construction	**31,241**	**615,462**	**19.7**	**0.3%**	**180,494**	**5.8**	**1.1%**	**82,733**	**2.6**	**2.9%**
Post-Construction	**2,605,062**	**18,750,567**	**7.2**	**7.7%**	**1,878,022**	**0.7**	**11.6%**	**258,336**	**0.1**	**9.2%**

Notes: CB = Chesapeake Bay

Table is based data from the Chesapeake Bay Program Watershed Model, Phase 5.3, for the 2009 Annual Assessment scenario

To account for nutrient- and sediment-control alternatives for post-construction stormwater runoff, the following five urban stormwater BMPs are also included in the model:

- **Extended detention basins**—These basins are engineered structures designed to capture and store runoff and release it slowly for control of peak runoff and velocities with pollutant removal by settling, but the basins are not designed to promote infiltration[12]. They were selected to represent generic, site-level, and traditional stormwater BMPs that

[12] Some basin systems can be designed to promote infiltration or to include green infrastructure features; however, these "infiltration basins" or "enhanced" extended detention basins are not included as part of this BMP category.

provide minimal ancillary ecosystem services.[13] They are the only gray stormwater infrastructure BMP considered in this analysis

- **Bioretention planters—**These planters are vegetated, landscaped depressions that allow for retention and infiltration of runoff; they are also referred to as a rain gardens. Bioretention planters were selected to represent generic, site-level, and vegetation-based stormwater management practices with ancillary ecosystem services. These practices are often referred to as low-impact development techniques or green infrastructure.
- **Urban, riparian grass buffers—**These buffers are areas of vegetation located adjacent to a water body and designed to accept runoff as overland sheet flow from upstream development.
- **Urban, riparian forest buffers—**These buffers are areas of trees, shrubs, and other vegetation located adjacent to a water body and designed to accept runoff as overland sheet flow from upstream development.
- **Urban wetlands—**These are lands inundated or saturated by ground or surface water at a frequency and duration sufficient to support a prevalence of vegetation typically adapted for life in saturated soil conditions (USACE, 1987). In urban settings, they are often referred to as pocket wetlands and/or constructed wetlands.

3.1.2 Caveats

Although the model covers many of the most important nutrient and sediment sources and control options available in the Chesapeake Bay watershed, due primarily to data and resource limitations, some potentially significant BMP options have not yet been incorporated into the framework.

As noted in ***Section 1***, AFOs are one important source of nutrient loads in the watershed, contributing 17% of the total nitrogen and 26% of the total phosphorus to the Chesapeake Bay; however, sufficient information on manure application was not available at the time of this analysis to assess the effects of decreasing nitrogen and phosphorus runoff loads from AFOs' manure land application on crops and pastures. As a result, AFOs/ CAFOs could not be

[13] For example, blue roofs (holding tanks on roofs that capture and gradually release precipitation) and rain barrels are detention-based technologies that do not provide ecosystem services, but can be implemented in high-density urban settings.

considered in the least-cost analysis described in later sections. A more detailed discussion of the data limitations for runoff from the actual feedlots is provided in **Appendix B**.

In addition, a few potentially significant categories of agricultural and stormwater BMPs have not been included in the model. For example, the following BMPs, which are listed as part of the CBWM, have not been incorporated into our modeling framework:

- Conservation planning (agricultural BMP)
- Decision agriculture (agricultural BMP)
- Erosion and sediment controls for development activities (urban stormwater BMP)
- Stream restoration (urban and non-urban).

3.2 SPATIAL MAPPING OF SOURCES AND BASELINE LOADINGS IN THE CHESAPEAKE BAY WATERSHED

An important step in modeling the least-cost combination of control practices that meet the basin-level load-reduction targets is to identify and map the locations of existing sources and loadings in relation to the river and stream network of the Chesapeake Bay watershed.

3.2.1 Point Sources: Significant Municipal and Industrial WWTPs

Location and discharge data for municipal and industrial WWTPs for 2008 were obtained from EPA as part of analyses completed for the *Report on Next Generation of Tools and Actions to Restore the Bay* (U.S. EPA, 2009), in response to EO 13508, *Chesapeake Bay Protection and Restoration* (Federal Register, 2009). These data included delivery factors from each facility to the Bay, which account for the attenuation of pollutants between the point of discharge and the Bay.

3.2.2 Nonpoint Sources: Agricultural and Urban Stormwater Runoff

The main data source and framework used to characterize agricultural and urban stormwater sources is the CBWM. This model is the most recent and comprehensive version of the Chesapeake Bay Program Office's (CBPO) watershed-level simulation model, which was released in its original version in 1982. The following data elements were drawn from the CBWM:

- **Watershed network and segmentation**—The model subdivides the Chesapeake Bay watershed into a linked network of 1,955 "land-river segments." This segmentation combines county-level boundaries with subwatershed boundaries for the watershed's river reach network.

- **Land use/land cover segmentation—**The model subdivides each land-river segment into 26 land-use categories, including 14 types of agricultural land and 5 types of developed land, resulting in 54,740 model elements.
- **Delivered loadings—**For each land-use category in each land-river segment, the model provides total annual delivered loadings estimates (in 2009) for nitrogen, phosphorus, and sediment.

Other key sources of spatial data used in this analysis include the following:

- **The National Hydrography Dataset (NHD)—**The NHD provides digital spatial data representing the surface water network in the Chesapeake Bay watershed. For this analysis, we used the "medium resolution" NHD, which is based on 1:100,000-scale topographic mapping because it provides a more detailed (i.e., higher resolution) representation of the reach network than the CBWM. Therefore, it was used to identify the land area within each land-river segment that is potentially available for installation of grass or forest buffer BMPs.
- **The Soil Survey Geographic (SSURGO) and State Soil Geographic (STATSGO2) Databases—**These databases (NRCS, 2006; NRCS, 2010) were used to identify land areas in each land-river segment with soils that are classified as hydric (or partially hydric). This designation was used to identify lands that were potentially suitable for installation of wetland BMPs. In addition, these databases provide descriptors of certain geomorphic features of the soil component (e.g., flood plains, backswamps, depressions) that were used to characterize water storage potential for potential wetland areas.
- **The USGS Hydrogeomorphic Regions in the Chesapeake Bay Watershed Dataset**. These data were used to account for the fact that several agricultural BMPs vary in effectiveness according to the hydrogeomorphic region in which they are applied.

Step-by-step descriptions of how these and other data were incorporated into the economic model are provided in **Appendices A and B.**

3.2.3 Mapping Areas Available and Suitable for Agricultural BMPs

To identify and map areas available and suitable for agricultural BMPs, we used the CBWM data on land-use classification systems to specify two main agricultural land uses—cropland and pastureland. For both land uses in each land-river segment, we then extracted CBWM data on (1) total acres, and (2) delivered pollutant loadings in 2009 (i.e., net of expected reductions in delivered loads attributable to non-tidal atmospheric deposition in the watershed).

We then placed a series of restrictions on where agricultural BMPs could be implemented in the watershed. For example, as shown in **Table 3-2**, certain BMPs, such as no-till agriculture, are not applicable on pastureland, whereas livestock exclusion is not applicable for cropland. Using CBWM data on current (2009) levels of BMP implementation in each land-river segment, we defined and excluded ineligible areas for new BMP implementation (e.g., an area being treated by an existing forest or grass buffer is not available for a new forest or grass buffer). In addition, several BMPs are assumed to be placed only within a fixed distance from the stream; for example, forest and grass buffers are assumed to be 100 feet deep. To estimate the specific number of acres within a land-river segment that are available for installation of a forest and grass buffer, we multiplied the eligible area of unbuffered agricultural acres in the land-river segment by the ratio of 100 feet to segment's average depth from streams.[14]. In addition, we assume that wetland restoration can only be applied to hydric soils (based on the SSURGO dataset that are within 1,044 feet from the edge-of-stream)[15].

[14] We used the polylines in the 1:100,000 NHD to estimate average distance depth from the stream.
[15] This distance is equal to one side of a square 5-acre wetland (i.e. 5 acres = 1,044 ft x 1,044 ft).

Table 3-2. Assumed Relationship between Land Characteristics and Suitability for Selected Agricultural BMPs

Land Characteristics	BMP Type								
	Grass Buffer	Forest Buffer	Conversion to Forest	Natural Revegetation	Livestock Exclusion	Restored Wetlands	Cover Crops	No Till	Reduced Fertilizer Application
Cropland	●	●	●	●		●	●	●	●
Pastureland	●	●	●	●	●	●			
Hydric Soil	●	●	●	●	●	●	●	●	●
Non-hydric Soil	●	●	●	●	●		●	●	●
Within 100-foot Buffer	●	●	●	●	●	●	●	●	●
Outside 100-foot Buffer			●	●		●[a]	●	●	●

[a] Assumed to only be applicable within 1,044 feet of the stream. (This distance is equal to one side of a square of 5 acres.)

3.2.4 Mapping Areas Available and Suitable for Urban Stormwater BMPs

To identify and map areas available and suitable for urban stormwater BMPs, we selected four urban land-use categories from the CBWM P5.3: high-density impervious (HI), high-density pervious (HP), low-density impervious (LI), and low-density pervious (LP)[16]. For each land-river segment, we then extracted and compiled data from the CBWM for these four categories, including (1) total acres, (2) number of acres with BMPs implemented in 2009, and (3) delivered pollutant loadings in 2009. Based on assumptions detailed in **Section 2.2.4.2 of Appendix B**, about existing BMPs in 2009, we calculated the acres of land still available for BMP placement for each of the four urban land uses in 2009.

In addition to the availability of land, we determined the suitability of land for implementing each of the five urban stormwater BMPs based on land-use data from the CBWM ; soil-type data (hydric vs. non-hydric) from STATSGO2; and spatial location relative to (i.e., distance in feet from) surface waters based on the NHD. Application rules for all five BMPs are summarized in **Table 3-3**. Grass and forest riparian buffers have significantly higher BMP-area-to-treatment-area ratios than the other three BMPs. Considering space limitations in urban settings, these practices were assumed to potentially apply to all urban land uses within 50 feet of streams and rivers. The 50-foot width was chosen based on the range of urban riparian buffer widths suggested in the literature on BMP pollution removal effectiveness and costs and physical limitations in urban settings. In the case of wetlands, while it is possible to construct pocket wetlands that require only a relatively small footprint, it is unlikely that high-density urban areas would be able to readily accommodate these BMPs. Therefore, wetlands were assumed to be applicable only to low-density urban land uses. In addition, only areas with hydric soils were considered suitable for the construction of wetlands.

[16] These developed land used categories are also referred to as high-intensity impervious, high-intensity pervious, low-intensity impervious, and low-intensity pervious developed lands, and are defined in U.S. EPA, 2010c, p. 4-20.

Table 3-3. Assumed Relationship between Land Characteristics and Suitability for Selected Urban Stormwater BMPs

	BMP Type				
Land Characteristics	**Extended Detention Basin**	**Bioretention**	**Forest Buffer**	**Grass Buffer**	**Wetlands**
High-density Urban (HI+HP)	●	●	●	●	
Low-density Urban (LI+LP)	●	●	●	●	●
Hydric Soil	●	●	●	●	●
Non-hydric Soil	●	●	●	●	
Within 50-foot Buffer	●	●	●	●	●
Outside 50-foot Buffer	●	●			●

We overlaid land availability and land suitability criteria to determine the available *and* suitable land for each urban stormwater BMP. For example, for wetlands, we added land areas that are low-density urban land use, have hydric soil, and are located either inside or outside a 50-foot buffer from a stream without existing BMPs in 2009. The full details of these calculations are described in **Appendix A**.

3.2.5 Caveats

The NHD-resolution scale of 1:100,000 used to define the waters for this analysis provides a detailed characterization of the surface water network within the Bay watershed; however, it does not provide the resolution required to identify features such as man-made ditches constructed to drain hydric soils for agriculture and silviculture in the watershed. These features are particularly prominent in the Eastern Shore basin. The omission of these features leads to an underestimation of the agricultural acreage suitable and available for riparian buffer installation. It also most likely understates the costs required for nutrient control with buffers in areas with high densities of drainage ditches.

3.3 ESTIMATION OF CONTROL COSTS AND LOAD REDUCTIONS FOR THE SELECTED CONTROL PROJECTS

This section summarizes the methods used to estimate the average control costs and load reductions for the selected nutrient- and sediment-control projects included in the optimization model. For a more detailed description of how control costs and load reductions are calculated, refer to **Section 2 of Appendix A** and **Section 2 of Appendix B**. In addition, **Section 4 of Appendix A** describes how the optimization model is expressed and operates mathematically.

3.3.1 Significant Municipal and Industrial Wastewater Point-source Controls

The analysis considers upgrades to significant municipal and industrial WWTPs as one of the possible means of achieving additional nitrogen and phosphorus removal. The cost and effectiveness of upgrades was based on data from two Chesapeake Bay Program studies (CBP, 2002; 2004) that provided estimated capital and operation and maintenance (O&M) costs for each WWTP to incrementally reduce total nitrogen and total phosphorus effluent concentrations.

As shown in **Table 3.4**, the study defined four tiers of nutrient reductions, with Tier 4 representing the limits of technology. As part of analyses completed for the *Report on Next Generation of Tools and Actions to Restore the Bay* (U.S. EPA, 2009), costs were restated in 2008 dollars and adjusted to reflect effluent concentrations reported in 2008 by each facility (i.e., accounting for upgrades implemented between 2004 and 2008); for a full methodology, see Abt Associates (2009). Total annual costs (capital and O&M) and reductions in total nitrogen and total phosphorus delivered to the Bay to meet each tier's effluent discharge limit were considered as distinct potential projects to be implemented for meeting the total pollutant-reduction targets.

Table 3-4. Annual Average WWTP Nutrient Effluent Concentrations for Tier Classifications

Facility Type	Parameter	Effluent Concentration (mg/L)			
		Tier 1	Tier 2	Tier 3	Tier 4
Significant Municipal (generally, ≥ 0.5 MGD)	TN	8.0 if BNR or year 2000 concentrations	8.0 or permit limit, if less	5.0 or permit limit, if less	3.0
	TP	Year 2000 concentrations	1.0 or permit limit, if less	0.5 mg/L or permit limit, if less	0.1
Significant Industrials (generally, > 75 lb/day TN and 25 lb/day TP)	TN	Year 2000 concentrations or permit limit, if less	50% reduction from Tier 1 (or year 2000 concentrations) or permit limit, if less	80% reduction from Tier 1 (or year 2000 concentrations) or permit limit, if less	3.0 or permit limit, if less
	TP				0.1 or permit limit, if less

TN = total nitrogen
TP = total phosphorus
Source: CBP, 2002

Table 3-5. Average Incremental Costs of WWTP Nutrient Removal by Tier Classifications

Facility Type	Parameter	Average Cost ($/lb removed/yr)		
		Tier 2[a]	Tier 3	Tier 4
Significant Municipal	Nitrogen	**11.50**	**30.74**	**21.58**
	Phosphorus	**1.61**	**997.38**	**14.92**
Significant Industrial	Nitrogen	**13.05**	**13.80**	**19.10**
	Phosphorus	**8.17**	**3.45**	**407.21**

[a]Tier 2 costs and pollutant reductions include all facilities meeting Tier 2 standards from 2008 conditions, and, therefore, include meeting Tier 1 standards. Tier 1 costs and pollutant reductions were not calculated separately

Note that the average cost per pound removed does not monotonically increase. Costs are plant-specific and depend on the baseline technology already present. In some cases, the infrastructure for a specific technology is built for Tier 3 (e.g., tanks, pumps, and alum for a specific dosing rate for total phosphorus [TP] treatment by chemical precipitation) and only additional materials are needed to meet Tier 4 (e.g., purchase of additional chemicals to achieve higher dosing rates and lower effluent concentrations).

3.3.2 Agricultural Best Management Practices

To incorporate agricultural BMPs into the cost-minimization analysis, we developed estimates of the annual reduction in delivered loads per acre and the annual cost per acre for each BMP.

The amount of nutrients and sediment removed by each BMP per acre is a function of (1) the baseline loadings per acre, (2) the loadings per acre of the new land use, and (3) the removal effectiveness of the BMP. As described in ***Section 3.2.2***, the baseline loadings estimates are based on data from the CBWM P5.3, which vary across land-river segments and land-use categories. The removal effectiveness estimates are based on estimates reported in Simpson and Weammert (2009), Brinson (1993), and the SSURGO database (NRCS, 2006). [17]

The effectiveness of agricultural BMPs varies by nutrient type and, in some cases, by hydrogeomorphic region. **Table 3-6** presents ranges of efficiencies for seven of the BMPs. As in the CBWM, conversion to forest and natural revegetation is not assigned removal efficiencies. Instead, the per-acre load reductions for these BMPs are estimated as the difference between (1) average per-acre loads for cropland or pastureland in the land-river segment and (2) average per

[17] Same values used by the Chesapeake Bay Watershed Model for nonpoint-source BMPs (CBP, 2009).

acre forest or “hay-unfertilized” loads in the land-river segment. As seen in this table, no-till agriculture is assumed to be the most effective at reducing sediment, whereas planting early cover crops is assumed to be, on average, the most effective in reducing nitrogen.

The annual cost per acre is the sum of three components: (1) the annualized installation (i.e., capital) cost of the BMP, (2) the annual O&M costs, and (3) the value of the land being converted or replaced by the BMP (i.e., the cash rental rate for crop and pastureland). Estimates of installation and O&M costs were based on data reported in Wieland et al. (2009) and Wainger and King (2007). Land rental rates were based on county-level estimates for crop and pasture land, as reported in the 2008 Cash Rents Survey (NASS, 2008). As a result, all of the spatial variation in per-acre cost estimates is attributable to the county-level variation in average land rental rates. Ranges of average costs per acre are reported for the nine BMPs in **Table 3-7**. Natural revegetation and no-till have the lowest estimated costs per acre, whereas wetland restoration and forest buffers have the highest.

Table 3-6. Summary of Treatment Efficiencies for Selected Agricultural BMPs

BMP	Removal Efficiencies		
	Total Nitrogen	***Total Phosphorous***	***Total Suspended Solid***
Forest Buffers	19–65%	30–45%	40–60%
Grass Buffers	13–46%	30–45%	40–60%
Wetland Conversion	7–25%	12–50%	4–15%
Livestock Exclusion	9–11%	24%	30%
Cover Crops	34–45%	15%	20%
No-till	10–15%	20–40%	70%
Reduced Fertilizer Application	15%	0%	0%

Table 3-7. Summary Cost for Selected Agricultural BMPs

BMP	Total Annual Cost per BMP Acre[a] ($/acre/yr)	
	Low	**High**
Forest Buffers	$163	$291
Grass Buffers	$99	$226
Wetland Conversion	$236	$364
Natural Revegetation	$14	$141
Conversion To Forest	$129	$257

BMP	Total Annual Cost per BMP Acre[a] ($/acre/yr)	
	Low	**High**
Livestock Exclusion	$81	$117
Cover Crops	$31	$31
No-till	$14	$14
Reduced Fertilizer Application	$37	$37

[a] Values shown are non-weighted averages across all land-river segments and land types.

3.3.3 Urban Stormwater BMPs

A literature review was conducted to determine the range of pollutant-removal efficiencies and costs for the removal of total nitrogen (TN), TP, and sediment for urban stormwater BMPs. An overview of treatment efficiencies and costs (annualized capital and O&M) used in the study is summarized in **Tables 3-8 and 3-9**.

Table 3-8. Summary of Treatment Efficiencies for Selected Urban Stormwater BMPs

BMP	Removal Efficiencies		
	Total Nitrogen	***Total Phosphorous***	***Total Suspended Solid***
Extended Detention	20%	20%	60%
Bioretention	48%	60%	68%
Grass Buffer	32%	40%	53%
Forest Buffer	50%	60%	60%
Wetlands	20%	45%	60%

Table 3-9. Summary Costs for Selected Urban Stormwater BMPs

BMP	Total Annual Cost per BMP Acre ($/acre/yr)	Ratio of BMP to Treated Area	Total Annual Cost per Treated Acre ($/acre/yr)
Extended Detention	$4,460	0.03	$134
Bioretention	$66,647	0.04	$2,666
Grass Buffer	$6,676	0.58	$3,872
Forest Buffer	$364	0.58	$211
Wetlands	$601	0.03	$18

Details on the range of values for pollutant-removal efficiencies and costs compiled during the literature review are provided in **Sections 2.2.4.4 and 2.2.4.5 of Appendix B**. These sections in **Appendix B** also include a discussion of BMP performance and the challenges with and assumptions implicit in selecting one treatment efficiency for each BMP across the Bay without a predefined regulatory requirement. In general, parameters that determine the final BMP treatment efficiency include the interaction of site and BMP design characteristics. Site characteristics determine the volume of runoff and associated pollutant loadings that are generated, whereas the BMP design determines the volume of runoff and pollutant(s) that will be prevented and/or mitigated. The BMP design can vary within certain limits to meet different levels of treatment or regulatory requirements. **Appendix B** also discusses the rationale for selecting the treatment efficiency and cost values used to model each BMP in the analysis.

3.3.4 Caveats

The cost and pollution removal efficiencies included in the model represent best-available approximations for the selected control projects, given the scope and scale of the model and project. Nevertheless, they are approximations and should be interpreted accordingly. In particular, the annual cost and removal estimates incorporate the effects of spatial heterogeneity in only a limited way. For example, the land values included in the cost estimates for agricultural BMPs are based on county-level averages for cropland and pasture. The removal effectiveness estimates for BMPs also represent average conditions by hydrogeomorphic region and do not incorporate spatial heterogeneity in factors such as topography.

3.4 QUANTIFICATION AND VALUATION OF BONUS ECOSYSTEM SERVICES

This section summarizes the methods used to identify, quantify, and (where possible) value the bonus ecosystem services provided by alternative nutrient- and sediment-control methods. The discussion focuses primarily on the bonus services provided by the conversion and management of agricultural lands and some urban stormwater BMPs, because point-source control projects provide few, if any, bonus ecosystem services. A more detailed and technical discussion of these methods is provided in **Section 3 of Appendix A** and **Section 3 of Appendix B**.

This section also describes the different ways in which ecosystem services estimates are incorporated into the larger economic modeling framework. In some cases, it is possible to estimate a value of increased services per unit (e.g., per acre) of land conversion. In these cases, the ecosystem service values can be thought of and treated as reductions in the NET cost of nutrient controls, and they can be directly incorporated into the optimization (i.e., cost minimization) framework described in the previous section. In other cases, the gains in bonus ecosystem services can only be estimated at an aggregate level or in nonmonetary terms; however, they still provide useful information for understanding the overall costs and benefits of different nutrient- and sediment-control strategies in the Chesapeake Bay watershed.

Although ecological benefits from vegetation-based control strategies, such as riparian buffers, wetlands, and forest, have been extensively studied and documented in the research literature, methods for quantifying the specific cause-and-effect relationships remain limited. Relatively few transferable and scalable models are available to reliably estimate the change in ecological services resulting from an acre of land-use conversion. Nevertheless, for this analysis, methods were adopted to quantify and/or value the following ecosystem services: (1) carbon sequestration and reduced GHG emissions from conversion to buffers, wetlands, bioretention ponds, and forests; (2) duck hunting services from wetland restoration; (3) non-waterfowl hunting services from increased forest cover; (4) air pollutant removal by urban stormwater buffers, wetlands, and bioretention ponds; (5) brook trout habitat and recreational fishing services from increased forest cover; and (6) water storage (flood control) services from wetland restoration.

It should be noted that prices from wetland mitigation banks were also considered as a potential source of information on the value of wetland restoration. Unfortunately, published data on these prices are very limited and suggest high variation in prices, ranging from $3,000 to $653,000 per acre in the United States (Madsen et al., 2010). Moreover, it is somewhat difficult to interpret what these prices represent and to what extent they reflect values for controlling nutrient and sediment loads, rather than for bonus ecosystem services.

3.4.1 Changes in Greenhouse Gas Emissions and Carbon Sequestration

Although carbon sequestration associated with certain control options (e.g., afforestation) is a potentially important source of bonus ecosystem services, it is also important to account for changes in GHG emissions. By accounting for both carbon sequestration and GHG emissions, this analysis predicts the net carbon balance of selected control projects in the Chesapeake Bay watershed.

3.4.1.1 Greenhouse Gases and Air Pollutant Emissions from Significant Wastewater Municipal and Industrial WWTP Upgrades

WWTP upgrades result in additional emissions of GHGs directly as part of the treatment process, and/or indirectly by requiring additional electricity, the production of which, in turn, results in additional indirect emissions of GHGs and other air pollutants. To quantify the additional GHG and air pollutant emissions associated with WWTP upgrades, we used the Chesapeake Bay Program (2002, 2004) data updated to 2008 (described in ***Section 3.3.1***) and followed assumptions made in the Chesapeake Bay Program 2002 study, such as the use of biological nutrient removal (BNR) technology for nitrogen removal and chemical precipitation for phosphorus removal. The magnitude of direct nitrous oxide (N_2O) emissions is negligible relative to the additional electricity requirements (WERF, 2010; Hwang et al., 2006); therefore, we did not quantify N_2O emissions. For chemical precipitation, we used data from Jiung and colleagues (2005) on the incremental energy use associated with decreasing concentrations of effluent phosphorus using chemical precipitation. The phosphorus treatment process does not have significant direct emissions of GHG or other air pollutants. To meet the Tier 4 phosphorous limit, we assumed chemical precipitation and the use of data from Jiung and colleagues (2005) (this differs slightly from the Chesapeake Bay Program study, which assumed low-pressure membrane treatment with the addition of metal salts). Based on these assumptions and data sources, which are described in further detail in **Appendix C**, we calculated the methanol and additional energy use associated with upgrading WWTPs to Tier 4 effluent discharge limits.

To calculate the social cost, or disamenities, of these emissions, we made the following assumptions:

- The social cost of GHG emissions was calculated assuming \$45 per ton of carbon, or approximately \$12 per ton of carbon dioxide (CO_2) equivalent (CO_2e), as suggested by the Intergovernmental Panel on Climate Change (IPCC).
- The social costs of air pollutant emissions were calculated using \$9,171 per ton of ozone (O_3) and \$2,245 per ton of sulfur dioxide (SO_2), based on median values from Murray and colleagues (1994) as reported by Nowak and colleagues (2006).

The resulting facility-level social costs reflect the additional GHG and air pollutant emissions associated with upgrading WWTPs to meet Tier 4 discharge limits for total nitrogen and total phosphorus. These social costs are disamenities that result from implementing WWTP upgrades to meet nitrogen- and phosphorus-reduction targets.

Based on this methodology, the total social costs of GHG and air pollutant emissions associated with upgrading significant municipal and industrial WWTPs to meet Tier 4 effluent discharge concentrations are $6.3 million and $11.4 million, respectively. The additional GHG emissions are almost 140,000 metric tons of carbon. To put these GHG emission numbers into perspective, they are equivalent to emissions resulting from annual electricity use by roughly 56,000 households (less than 1% of the Chesapeake Bay population), or the emissions of about 89,000 cars per year (U.S. EPA, 2010c).

3.4.1.2 Greenhouse Gas Emissions and Carbon Sequestration Associated with Cropland, Pastureland, and Wetlands

Greenhouse Gas Emissions. To estimate changes in GHG emissions associated with specific land-use change scenarios, the analysis focused on three emission types — CO_2, N_2O, and methane (CH_4) —all expressed in the common unit of CO_2e. Carbon dioxide emissions occur as a result of decomposition and aerobic degradation and can be temporarily accelerated following conversion of lands to wetlands. Nitrous oxide emissions are most common with croplands with higher emissions, such as corn, which require nitrogen fertilization, unlike nitrogen-fixing crops, such as soybeans. Methane, a product of anaerobic degradation, also commonly occurs in wetlands due to the low oxygen availability with a high water table.

Average per-acre emission rates were estimated for three main land cover types — cropland, pastureland, wetland— using two primary reference sources. The Forest and Agricultural Sector Optimization Model (FASOM) (Adams et al., 1996) was used for crop and pasture N_2O emission rates, and the IPCC was referenced to identify CO_2 and CH_4 emission rates, where available, for wetlands (IPCC, 2006). Zero GHG emissions were assumed for forest land and riparian buffers. (See **Section 3 of Appendix A** for details.)

For BMPs involving the conversion of cropland and pastureland to other uses, we assumed that the crop- and pasture-specific GHG emissions would be reduced to zero on the affected acres. For acres converted to wetlands, we added the wetland-specific GHG emission rate. To estimate the reductions in GHG emissions resulting from reduced fertilizer application, we assumed that N_2O emissions (and the CO_2e) from the affected cropland would decline in proportion to the decline in fertilizer application (i.e., 15%).

Carbon Sequestration. Carbon sequestration and potential offset of climate change is one of the ecosystem services offered with conversion of agricultural lands to forests and wetlands in the Chesapeake Bay watershed. Conversion to forests will result in accumulation or sequestration of carbon in aboveground and belowground vegetation, as well as soil pools during stand development. Conversion to wetlands will sequester carbon in vegetation and soils, with a large amount of carbon accumulating in the soils due to higher water tables and anoxic conditions, which slows decomposition. Conversion to grasslands also will result in carbon sequestration, mostly below ground.

For this analysis, carbon sequestration rates were calculated for the land conversions to forests, wetlands, grasslands, and natural revegetation using a five-step process, as outlined below.

- **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 Chesapeake Bay watershed.
- **Step 2: Select tree species.** For crop or pastureland converted to forest, it was 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 modeled to involve planting of the wetland area with bald cypress/water tupelo forest type (Neely, 2008). For natural revegetation, it was assumed that 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 (NCASI)/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 non-soil carbon storage values report for 5- to 10-year increments during years 0–90 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 the Steps 1 through 3, described above, and assigning counties to their respective main ecoregions, carbon sequestration rates were calculated 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, other agricultural BMPs are also likely to have an effect on carbon sequestration. For this model, we only added carbon sequestration estimates for no-till agriculture. (See **Appendix A** for details on the methodology.)

3.4.1.3 Carbon Sequestration by Urban Stormwater BMPs

The design and landscape characteristics of urban stormwater BMPs, such as the type and age of the vegetative cover, are primary determinants of the magnitude of carbon sequestration services provided by urban stormwater BMPs. These services were quantified in two parts—services from changes in vegetative cover and from changes in soils. Vegetative cover within each practice was distributed among four general categories: grasses, shrubs, trees, and emergent aquatic vegetation. Classifications of soil and associated values of soil organic carbon were urban forest, park use grass, wetlands, and clean fill. Detailed descriptions of the type of vegetative covers and soils assumed for each urban stormwater BMP are shown in **Section 3.1.2 of Appendix B**. Based on these assumptions, the estimated carbon sequestration service by each urban stormwater BMP is shown in **Table 3-10**. In calculating the vegetation associated with urban stormwater BMP implementation, we assumed that there was no pre-existing vegetation. This may lead to an overestimation of ancillary carbon sequestration benefits associated with these BMPs.

Table 3-10. Summary of Estimated Carbon Sequestration Services

BMP	Annualized Carbon Sequestration (lbs/acre of BMP /yr)		
	Low	**Mid**	**High**
Extended Detention	1,360	1,548	1,735
Bioretention	1,539	1,790	1,977
Grass Buffer	1,543	1,795	1,982
Forest Buffer	2,329	2,661	2,755
Wetlands	6,098	6,151	6,193

The mid-range estimates of carbon sequestration shown in Table 3-10 were used, along with the unit-value estimates of carbon (described below), to value these ancillary ecosystem services provided by urban stormwater BMPs.

3.4.1.4 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 one ton of carbon to the atmosphere (also referred to as the social cost of carbon [SCC]). We assumed a value of $45 per metric ton of carbon sequestered which is a mid-range estimate from studies reviewing the empirical literature on the marginal social costs of carbon (Tol, 2005 and 2008; IPCC, 2007). Equivalently, we assume a value of $12.27 per ton of CO_2e emissions reduced.[18] Applying this estimate of SCC to the estimated time paths of carbon flux reported**,** the present value of carbon storage associated with each land-use conversion category was calculated using a 3% discount rate, and an annualized value of carbon storage for each acre of land conversion was determined. These estimates are reported in **Table 3-11**.

[18] The 3.666 conversion rate is the molecular weight of CO_2 –44- divided by the molecular weight of carbon - 12. This conversion rate between CO_2 emissions reduced and carbon sequestered also assumes that the carbon is permanently stored.

Table 3-11. Per Acre Value of Reduced Greenhouse Gas Emissions and Carbon Sequestration Services from BMP Application ($/ac)

BMP Application	Annualized Value[a], $	
	From Cropland	From Pastureland
To Forest	$31.98–$60.39	$29.71–$44.50
To Wetland	$36.55–$49.67	$36.55 – $36.57
To Grass Buffer	$3.52–$16.64	$0–$0.02
To Natural Revegetation	$27.23–$49.21	$28.88–$39.88
To No-Till	$1.59	NA
To Reduced Fertilizer Application	$0.53–$2.50	NA

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

One alternative to the SCC valuation approach described above is to use the actual or expected market price of carbon credits; however, depending on the information source, this price can vary from close to zero (Chicago Climate Exchange, 2010) to over $30/t CO_2e. To address this value uncertainty, we also include a sensitivity analysis in our model runs, which varies the price between $7/t CO_2e and $25/t CO_2e.

3.4.2 Duck Hunting Services from Wetland Restoration

Although several studies have estimated values for hunting services in the United States, few studies have estimated the effect that changes in landscape composition (e.g., habitat loss or gains) differences in land-cover characteristics have on these values. One exception is Murray and colleagues (2009), who estimated the effects of wetland restoration in the Mississippi Alluvial Valley on duck hunting services. This report adapts the methodology from that study to estimate the effects of wetland conversion in the Chesapeake Bay watershed on duck hunting services. This methodology only applies to wetlands converted from cropland and pasture. It is not used to assess ecosystem services from urban wetlands, where hunting is unlikely to occur.

The first step is to develop a model for estimating energetic carrying capacity of the Chesapeake Bay watershed for ducks. To accomplish this, a “duck energy day” (DED) model was applied. DEDs are the number of ducks that can meet their daily energy requirements from an area of foraging habitat for a single day (Reinecke and Kaminski, 2005). Based on a review of the literature, parameter values were selected that allowed for the estimation of average DEDs per acre for corn and soybean cropland and for freshwater and tidal wetlands.

The second step is to estimate baseline DEDs in the Chesapeake Bay watershed by multiplying the number of acres in each land-cover category by the corresponding DED/acre estimates from the first step. For cropland, the baseline area was defined as the cropland that is currently in corn and soybeans. For baseline wetland area, all Chesapeake Bay watershed freshwater and tidal wetland acres were included.

The third step is to estimate the baseline value of duck hunting services in the Chesapeake Bay watershed by state in 2008; see **Table 3-12**. Estimates of the total number of duck hunting days by state in the watershed were based on state-level duck hunting data from Richkus and colleagues (2008) and other hunting data based on the National Survey of Fishing, Hunting, and Wildlife Associated Recreation (FHWAR). To estimate the annual value of these duck hunting days, the regional average value of a duck hunting day was estimated (Rosenberger and Loomis, 2001). The estimated aggregate annual value of duck hunting in the Chesapeake Bay watershed varies from less than $40,000 in West Virginia to $4.3 million in Maryland.

Table 3-12. Baseline Value of Chesapeake Bay Watershed Duck Hunting in 2008

State	Duck Hunting Days 2008 (000s)[a]	Value per Duck Hunting Day (2008 $)[b]	Aggregate Annual Value of Duck Hunting (2008 $)
Delaware	28.2	$52.62	$608,036
Maryland	95.6	$52.62	$4,287,369
New York	29.9	$52.62	$2,157,428
Pennsylvania	66.3	$52.62	$3,092,266
Virginia	55.8	$45.86	$2,356,575
West Virginia	0.6	$45.86	$37,465

[a] FHWAR

[b] 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. Following the method and assumptions used in Murray and colleagues (2009), it was assumed that the aggregate value of duck hunting in each state increases in direct proportion to the increase in total DEDs. Therefore, we first estimated the percentage increase in total DEDs per state, for each individual acre of land conversion, and applied this percent adjustment to the baseline hunting values. The results of this step are reported in **Table 3-13**. The largest increments are associated with conversion from pastureland (lowest DEDs per acre) to tidal wetlands (highest DEDs per acre).

Table 3-13. Incremental Annual Value of Duck Hunting Services Per Acre of Wetland Restoration

State	Value by Type of Land-Use Conversion (2008 $)			
	Cropland to Tidal Wetland	**Cropland to Freshwater Wetland**	**Pastureland to Tidal Wetland**	**Pasture to Freshwater Wetland**
DE	$7.10	$3.04	$8.17	$4.11
MD	$7.56	$3.33	$8.49	$4.27
NY	NA	$3.12	NA	$6.77
PA	NA	$2.24	NA	$4.01
VA	$3.78	$1.69	$4.21	$2.12
WV	NA	$0.94	NA	$1.59

3.4.3 Non-Waterfowl Hunting Services from Increases in Forest Cover

To estimate the effects of land-use/land-cover change on other hunting services, results from a hedonic price study of hunting leases by Shrestha and Alavalapati (2004) were applied to the Bay. Although this study was conducted in central Florida rather than in the Chesapeake Bay watershed, it is geographically the closest study that has estimated the effect of different types of land cover on hunting values. From this study, the elasticity parameter estimate of hunting values with respect to forest cover was used to estimate the effects of increased forest cover in the watershed on non-waterfowl hunting services. In particular, our model assumes that each 1% increase in forest cover (per state) increases the average annual value of non-waterfowl hunting in the state by 0.132%.

First, baseline county-level hunting day estimates were derived from Ribaudo and colleagues (2008) and the FHWAR survey. Estimates of non-waterfowl hunting days by state were derived by deducting the duck hunting day estimates reported in Table 3-13. The total value of these non-waterfowl hunting days were then valued using estimates of average per-day consumer surplus for small- and big-game hunting (Rosenberger and Loomis, 2001).

Second, using (1) these baseline hunting value estimates, (2) baseline forest cover estimates per state (from RESAC, i.e, the 2000 Regional Earth Science Application Center), and

(3) the elasticity parameter from Shrestha and Alavalapati (2004), we estimated the incremental annual value of non-waterfowl hunting per acre of additional forest cover. These estimates are also reported in **Table 3-14**.

This methodology was applied to assess bonus ecosystem service benefits from land conversion associated with agricultural BMPs, but not from installation of urban forest buffers.

Table 3-14. Baseline and Incremental Annual Value of Non-waterfowl Hunting Services in the Chesapeake Bay Watershed

State	Non-Waterfowl Hunting Days in 2008 (000s)	Per-Day Value of Non-Waterfowl Hunting (2008 $)	Aggregate Annual Value of Non-Waterfowl Hunting Days (2008 $)	Incremental Value of Non-Waterfowl Hunting per Additional Forest Acre (2008 $)
DE	231	$49.66	$11,453,207	$1.19
MD	1,846	$49.66	$91,691,165	$1.90
NY	3,105	$49.66	$154,183,935	$4.58
PA	6,847	$49.66	$340,019,762	$3.24
VA	3,706	$42.60	$157,876,788	$1.21
WV	649	$42.60	$27,658,629	$1.83

3.4.4 Atmospheric Pollutant Removal by Urban Stormwater BMPs

In addition to carbon sequestration, vegetation-based urban stormwater BMPs remove atmospheric pollutants and improve air quality. The Urban Forest Effects Model (UFORE) estimates changes in the atmospheric pollutants associated urban trees and shrubs (Nowak et al., 2006). Module D of UFORE ("UFORE D: Dry Deposition of Air Pollution") calculates dry deposition rates for O_3, SO_2, nitrogen dioxide (NO_2), carbon monoxide (CO), and particulates (PM_{10}). These estimates of pollutant removal incorporate various factors, including leaf-area index, species composition, atmospheric data (e.g., air temperature, wind speed), and ambient concentrations of atmospheric pollutants. Low, mid, and high estimates for pollutant removal per tree, per acre of urban tree cover, and per acre of shrub cover from Nowak and colleagues (2006) are provided in **Section 3.2.2 of Appendix B**. Value per ton of pollutant removed was also acquired from the UFORE model, which uses values from Murray and colleagues (1994), a study that reports externality values used in energy decision making across various studies. These values are also provided in **Section 3.2.2 of Appendix B**. Using the mid estimate for pollutant-removal rates and annualized values per ton of pollutant removed, we calculated the annual value of atmospheric pollutant removal services per acre of urban stormwater BMP, as shown in **Table 3-15**.

Table 3-15. Annual Value of Pollutant Removal by Urban Stormwater BMPs ($/acre of BMP/year).

BMP	O_3	PM_{10}	NO_2	SO_2	CO	Total
Extended Detention	$0	$0	$0	$0	$0	$0
Bioretention	$23	$12	$15	$2	$0	$51
Grass Buffer	$21	$10	$13	$2	$0	$46
Forest Buffer	$72	$36	$46	$5	$1	$160
Wetlands	$3	$2	$2	$0	$0	$7

3.4.5 Brook Trout Habitat and Recreational Fishing Services from Increases in Forest Cover

Brook trout were once present throughout most coldwater streams and rivers in the eastern United States and are the only trout species native to this area. However, habitat loss, water quality degradation, and introduction of non-native species have all contributed to large declines in their populations. Wild populations of brook trout have been extirpated or greatly reduced from nearly half of the subwatersheds where they previously lived (Eastern Brook Trout Joint Venture, 2006).

Results from a recent study by Hudy and colleagues (2008) were used to estimate the effects of changes in land cover on brook trout habitat. In this study, the researchers examine the relationship between the status of brook trout populations in eastern U.S. subwatersheds and selected land-use characteristics, including the percentage of forest cover within the sub-watershed. For our analysis, we applied this model to predict (1) the current status of brook trout in 1,414 subwatersheds (12-digit HUCs) within the Chesapeake Bay watershed, and (2) the change in status associated with changes in forest cover. To predict changes in brook trout habitat status for each subwatershed, conversion to forest, forest buffer installation, natural revegetation, and wetland restoration were all assumed to increase forest cover on an acre-per-acre basis.

Unfortunately, it is difficult to use the Hudy and colleagues model to value the benefits of forest cover changes on a per-acre basis. First, it treats forest cover as a discrete rather than continuous variable for predicting changes in brook trout habitat. It also defines a specific forest cover threshold (68.1%), above (below) which brook trout habitat is improved (degraded). Second, it is difficult to identify an economic valuation study or model that is compatible with the Hudy and colleagues framework. One possible valuation approach is described in **Section 3 of Appendix A**; however, it tends to generate value estimates that are implausibly large.

3.4.6 Water Storage and Flood Control from Freshwater Wetlands

Wetland restoration BMPs are intended to return natural/historic functions to former wetland areas. Depending on its location, a wetland can serve several functions (**Figure 3-1**) and provide a variety of services. In addition to controlling nutrient runoff to nearby surface waters and providing habitat for waterfowl and other wildlife, wetlands can provide downstream flood protection through water storage and buffering of storm waters.

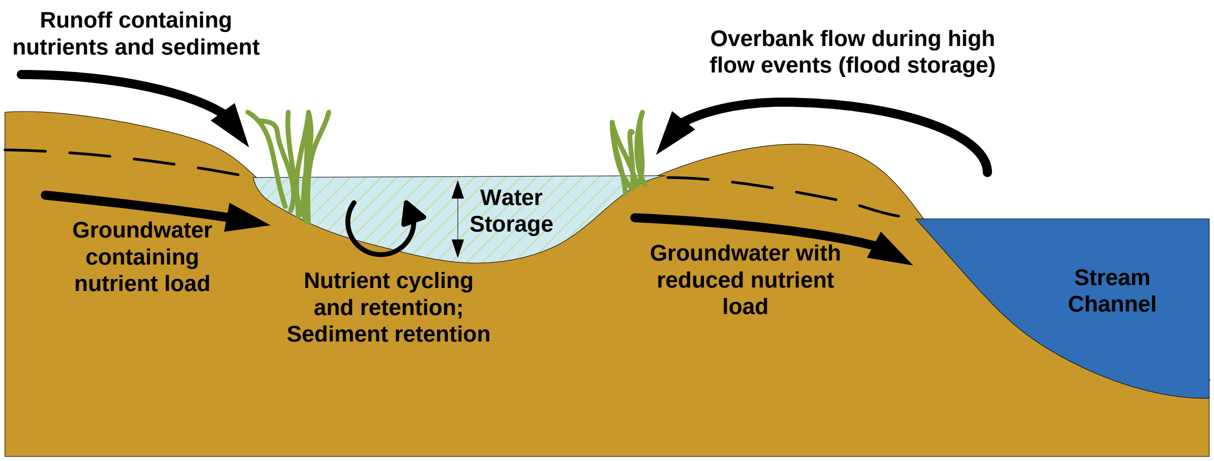

Figure 3-1. Overview of key wetland functions.

Although flood-control services through water storage are difficult to quantify reliably, estimating the total amount of additional water stored in a basin as a result of wetland restoration provides at least a rough indicator of potential flood-control services.

To estimate total water storage, a simple framework was used that multiplies the total acres of restored freshwater wetlands in each basin by an assumed average storage capacity of 3 acre-ft of water per acre of wetlands (U.S. EPA, 2006). This simple model does not account for differences in wetland characteristics, which can affect the rates of water input and the amount of water stored; nor does it account for the downstream characteristics (e.g., dams, populations, properties) that influence the potential damages associated with flood events.

To account for differences in potential and expected water storage services, four key wetland characteristics were defined—landform suitability, wetland suitability, stormwater protection, and water storage potential. **Table 3-16** displays these characteristics and a simple scoring matrix, which are used to develop an index of water storage services provided per acre of converted freshwater wetland. This composite index takes on a value of 1 for acres providing the highest potential water storage services and a value of 9 for the lowest. Using this composite index, we define three overall ratings for lands' water storage potential—low potential (> 7), medium potential (3–7), and high potential (< 3).

In this analysis, the use of wetland conversion as an agricultural BMP is assumed to only be applicable for crop and pasture lands that lie (1) within the 1,044-foot area and (2) in areas with hydric soils (i.e., most likely to be former wetland areas). Therefore, the ranking described in **Table 3-17** only applies to these lands. Because estuarine and freshwater wetlands differ significantly in their functions, the two types were also differentiated using locations and landforms underlying the areas as identified by SSURGO data.

Table 3-16. Scoring Matrix and Characteristics of Potential Wetlands

Wetland Characteristic	Data Source/Method	Purpose of Characteristic	Rank of Values (Best to Worst)		
			1	2	3
C1: Landform Suitability: SSURGO	Underlying soil hydrogeomorphic features	General suitability of area to provide a functional wetland	Categories involving floodplains, swamps, and former riverine features	Categories involving depressions	All other categories
C2: Wetland suitability: NWI	Location in regards to NWI polygons	General suitability of area to provide a functional wetland	Contained completely within an NWI polygon	Partially contained or adjacent (within 30 m) to NWI polygon	All other locations
C3: Stormwater protection	Distance from upstream of urban areas and downstream of flood-control dams	Measure of water storage, considering property value, that can be protected	Not within 5 miles downstream of dam, but within 2 miles upstream of urban area	Not within 5 miles downstream of dam, but within 5 miles upstream of urban area	All other locations
C4: Water storage potential: DEM	Difference in elevation between centroid of potential wetland and midpoint of nearest stream segment	Measure of water storage by depth of water that may be stored within the wetland	Elevation Difference < 0 m	0 m < Elevation difference < 2 m	All other values

3.4.7 Caveats

All of the methods described above for quantifying and monetizing ecosystem services involve a number of simplifying assumptions and, therefore, must be interpreted as providing rough approximations of bonus services. For example, due primarily to data limitations, the methods for quantifying changes in carbon sequestration and duck habitat assume relatively little spatial variation in carbon storage and habitat provided by alternative land cover types. The carbon sequestration estimates require several assumptions regarding the type and timing of change in the mix of vegetation resulting from BMP implementation. The methods used to quantify changes in hunting services are based mainly on studies that were not conducted in the Chesapeake Bay watershed. In addition, all of the methods used to value changes in ecosystem services rely on relatively simple "benefit transfer" techniques (Boyle and Bergstrom, 1992). That is, they apply somewhat generic unit value estimates (e.g., value per unit of carbon sequestered, value per recreation day, value per ton of pollutant removed), which are drawn from summary studies of the empirical nonmarket valuation literature. None of these value estimates were specifically developed for the Chesapeake Bay watershed; however, in our judgment, they provide the best *available* estimates for valuing bonus ecosystem services from land conversion in the watershed. More detailed descriptions of these methods and discussions of the main limitations and areas of uncertainty associated with them are provided in **Appendix A** and **Section 3 of Appendix B.**

Perhaps the most important uncertainties associated with assessing bonus ecosystem services from nutrient- and sediment-control projects are from those that cannot be reliably quantified or monetized. Table 2-2 provides a long list of potential services; however, due to a lack of available data and models, only a small subset of them are quantified and included in the model. Consequently, the results described in ***Section 5*** are best interpreted as providing lower-bound estimates for the *total* value and potential significance of *all* bonus ecosystem services.

3.5 THE OPTIMIZATION MODEL

As described in ***Section 1***, the first objective of the optimization is to identify the combination of nutrient- and sediment-control projects included in the model that achieves targeted load reductions to the Chesapeake Bay at the lowest total cost. In this regard, the framework is similar to the other cost-minimization studies, such as Schwartz (2010), which solves for the least-cost nutrient strategy in the Potomac River Basin. However, in addition to identifying the least-cost combination of projects and their associated costs, the model solution can be used to characterize and quantify the bonus ecosystem services delivered by these selected projects.

The second objective is to incorporate the bonus ecosystem services directly into the optimization framework. We do this by deducting, from the costs of each control project, the value of the bonus ecosystem services delivered by the project (i.e., by calculating the NET cost of each project). We then apply the optimization model to identify the combination of control projects that achieves the load reduction targets at the lowest total NET cost.

To identify and compare the least-cost solution and least-NET-cost solution under alternative scenarios, the optimization is formulated as a mixed integer linear programming (MILP) problem[19]. We solve for the optimal solutions using the GAMS modeling system. The optimization model uses a branch-and-cut algorithm to search across the possible combinations of costs and load reductions and identify the combined set of projects that achieves the targeted aggregate load reduction for the lowest aggregate cost or the lowest aggregate NET cost. A detailed technical description of the optimization model is provided in **Section 4 of Appendix A**.

3.5.1 Optimization Model Inputs

The main model inputs are the following:

(1) An inventory of available and mutually exclusive point- and nonpoint-source control projects available in each basin in the watershed (e.g., acres of urban land available for riparian grass buffer placement);

(2) Costs (in \$/year) and reductions in delivered nitrogen, phosphorus, and sediment loads (in pounds/year) for each of the mutually exclusive removal projects;

(3) Monetized bonus ecosystem services (\$/year) associated with each project (e.g., carbon sequestration, hunting, air quality values);

(4) Non-monetized ecosystem services associated with each project (e.g., water storage or impact on brook trout habitat); and

[19] More complex genetic and evolutionary algorithms have also been used to evaluate cost-effective strategies for watershed-level nutrient control (Bekele and Nicklow, 2005; Arabi et al., 2006). These methods are particularly helpful for addressing problems with multiple and competing objectives (e.g., minimizing costs and maximizing load reductions); however, they are not required in this case, where the load reduction targets are held constant and the single objective is to minimize costs (or NET costs).

(5) The optimization constraints (i.e., the nutrient- and sediment-load reduction targets).

For our analysis, we define load-reduction targets based on the goals of the Chesapeake Bay TMDL. These targets are calculated as the difference between baseline annual loadings (in 2009) and the final TMDL load allocations for non-atmospheric sources. To describe baseline loadings, **Table 3-17** reports CBWM estimates of 2009 basin-level nutrient and sediment loads delivered to the Bay. To focus the analysis on load reductions from non-atmospheric sources, these reported baseline estimates have already been adjusted downwards to reflect expected future (2020) reductions in atmospheric nitrogen deposition attributable to the Clean Air Interstate Rule (CAIR). **Table 3-17** also reports the final TMDL load allocations by pollutant and basin (U.S. EPA, 2010a). In addition, it reports the low and high load-allocation values for sediments. Based on the draft TMDL allocations (U.S. EPA, 2010b), these ranges are included to support sensitivity analyses, which are described in ***Sections 4 and 5*** of this report. **Table 3-18** reports the corresponding load-reduction targets by basin.

Table 3-17. Baseline Loads and TMDL Load Allocations by Basin (millions lbs)

Basin	2009 Baseline Loadings			TMDL Load Allocations				
	N[a]	P	Sediment	Final N Allocation	Final P Allocation	Sediment		
						Low Allocation[b]	Final Allocation	High Allocation[b]
Eastern Shore of Chesapeake Bay	19.0	1.7	298	14.3	1.4	256	259	281
James River Basin	31.3	3.3	1,263	23.1	2.4	852	937	937
Patuxent River Basin	3.1	0.3	114	2.9	0.2	82	106	90
Potomac River Basin	53.4	4.5	2,546	46.6	3.4	1,920	2,036	2,113
Rappahannock River Basin	6.9	1.1	752	5.8	0.9	681	700	750
Susquehanna River Basin	111.9	4.3	2,626	78.8	3.1	2,013	2,097	2,214
Western Shore of Chesapeake Bay	14.0	0.8	238	9.1	0.5	155	200	171
York River Basin	6.4	0.6	142	5.4	0.5	107	118	118
Total	245.8	16.5	7,979	185.9	12.5	6,066	6,454	6,674

[a] These baseline estimates are net of expected reductions in delivered loads attributable to non-tidal atmospheric deposition in the watershed.

[b] Specified in the Draft TMDL (U.S. EPA, 2010b)

Table 3-18. Load Reduction Targets by Basin (millions lbs)

Basin	N[a]	P	Sediment		
			Low Allocation	Midpoint	High Allocation
Eastern Shore of Chesapeake Bay	4.74	0.27	42.00	38.88	17.00
James River Basin	8.18	0.89	411.11	326.23	326.11
Patuxent River Basin	0.20	0.05	31.97	7.67	23.97
Potomac River Basin	6.77	1.03	626.05	509.72	433.05
Rappahannock River Basin	1.01	0.18	70.94	51.90	1.94
Susquehanna River Basin	33.14	1.16	612.99	529.02	411.99
Western Shore of Chesapeake Bay	4.91	0.26	83.43	38.24	67.43
York River Basin	0.95	0.08	34.60	23.80	23.60
Total	59.91	3.92	1,913.10	1,525.47	1,305.10

[a] Excluding expected reductions in delivered loads attributable to non-tidal atmospheric deposition in the watershed

3.5.2 Optimization Model Outputs

For both the least-cost and least-NET-cost optimization problems, the following model outputs are generated for each land-river segment:

- An inventory of *selected* point- and nonpoint-source projects
- The number of additional acres converted and treated by each agricultural and urban stormwater BMP
- The total costs of pollutant removal ($/yr) by BMP and point-source category
- The total reductions in delivered loads of nitrogen, phosphorus, and sediment (lbs/yr) by BMP and point-source category
- The total monetized bonus ecosystem services ($/yr) by BMP
- The total NET costs ($/yr) by BMP and point-source category
- The total non-monetized bonus ecosystem services by BMP and point-source category.

3.5.3 Simplified Representation of the Optimization Process

Figure 3-2 provides a simplified representation of the optimization process for one pollutant (nitrogen) and one basin (Susquehanna). For the optimal solution to achieve the load reduction target of 33 million pounds, the figure orders control projects from the least to the most costly per pound (net of the value of phosphorus and sediment control), and thus, traces out a marginal cost curve for nitrogen reduction in the Susquehanna River basin.[20] In this simplified representation, the model finds and adds the most cost-effective projects until the load-reduction target is met. In Figure 3-2, the cost of the "last" project added is roughly $5 per pound of nitrogen. The lower-cost ($/lb) projects tend to be agricultural BMPs, followed by urban stormwater BMPs, but the optimal solution also includes point-source controls that cost less than $5/ lb.

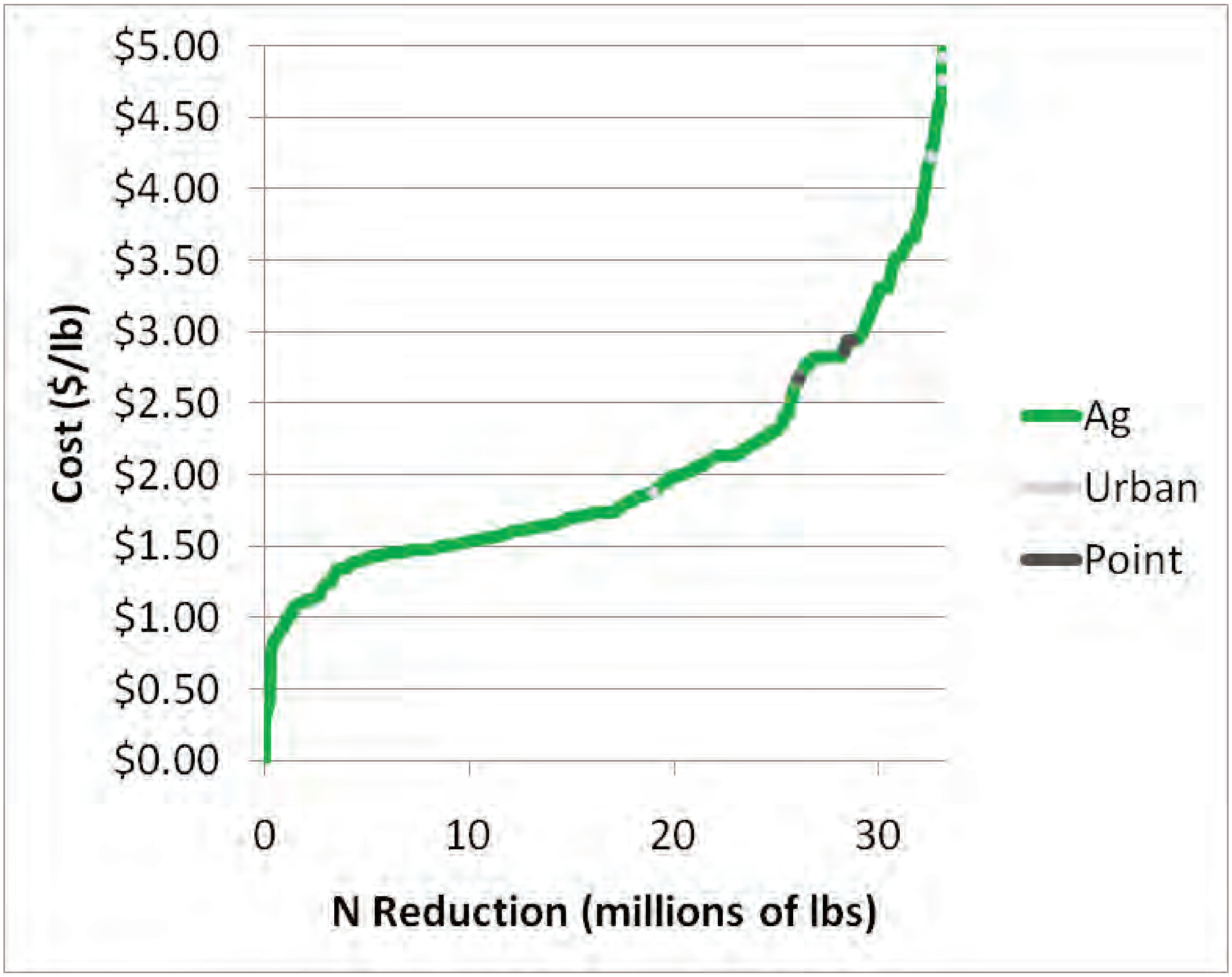

Figure 3-2. Marginal cost curve for nitrogen (Susquehanna Basin).

In practice, the optimization process can be a bit more complicated, because finding the least-cost solution for meeting the TMDL targets does not *necessarily* mean selecting the control projects with the lowest *ratio* of costs per delivered load reduction. Projects with low cost-to-load reduction ratios may not be selected if the *total* load reductions from these projects are

[20] The estimates are based on the assumptions for Scenario 1, which are described in ***Section 3***.

relatively small. For example, grass buffers often have a relatively low-cost-per-pound ratio for nitrogen reduction; however, as discussed in the following sections, they do not often figure significantly into the least-cost solutions because they do not contribute enough total load reductions to meet the basin-level targets.

3.5.4 Optimization When Target Load Reductions Cannot be Achieved with the Available Projects in the Model

Not all of the model solutions were as straightforward as the one shown in Figure 3-2. In some scenarios, such as requiring a 2:1 nonpoint to point credit ratio, the source controls included in the model are not sufficient to meet the target reductions in all of the basins. To successfully operationalize the model when target reductions cannot be achieved, we included a penalty (i.e., “safety valve”) price in the model for each pollutant. This price is imposed on each pound by which pollutant reductions fall below a target. Most importantly, it allows the model to find a “second-best” solution when the available control projects included in the model are insufficient to meet a target (i.e., when there is no true solution to the optimization problem).

The penalty prices are set at the high end of actual costs for the projects in the model—1,557 $/lb and 5,110 $/lb, respectively, for nitrogen and phosphorus, and $15/lb for sediment. The nitrogen and phosphorus penalty prices are equivalent to the most expensive point-source controls available in the model. The sediment penalty price, which has no point-source control, was estimated by equating the value of sediment reduction to the value of phosphorus reduction at its penalty price[21].

The penalty also acts as a cap on additional cost per pound for the pollutant (i.e., it is less costly to incur the penalty than to reduce the pollutant by another unit). Using a higher penalty price would result in a solution that is closer to the target, but possibly much more costly and still below the target level. As a result, it should be noted that when a basin-level load reduction cannot be achieved, the model solution is somewhat dependent on the choice of penalty price. By selecting very high penalty prices, we have placed relatively little constraints on the size of the total cost estimates.

[21] For nitrogen and phosphorus, the penalty price was set equal to the cost per pound of the most expensive Tier 4 POTW upgrade in the model. For sediment, the phosphorus penalty price was used, prorated by the ratio of Bay-wide phosphorus-to-sediment load reduction targets.

3.5.5 Caveats

Given the set of input data and constraints to be applied for a given scenario, the model solves for the least-cost solution that satisfies all constraints. Clearly, the ability of the model to accurately reflect the full costs of alternative policies is dependent on the quality of the model inputs. While we have taken care to assess the available data and incorporate estimates in the model that reflect the best-available information, there is uncertainty regarding the parameter values. As with any model of a complex system, it is not possible to capture all of the underlying detail of the system.

For instance, there is a great deal of heterogeneity across regions and landowners such that the actual costs per acre of a mitigation option may vary considerably. However, based on available information, the model assumes that a given BMP has the same annualized installation and maintenance cost per acre regardless of where it is applied. Similarly, in scenarios where transactions costs are applied, they are implemented as a constant percentage of control cost for all nonpoint-source projects, whereas transactions costs are likely to vary by BMP type, size of land holding where they are applied, and other factors.

Another caveat to keep in mind is that there are no market adjustments captured within this framework. As land is taken out of crop production, the remaining cropland may increase in value, requiring larger payments to landowners for them to change practices as more nonpoint-source options are adopted.

In addition, the current version of the model is static; therefore, it does not reflect possible future changes in market conditions, land use, nutrient and sediment loading, GHG emissions and sequestration, or levels of other ecosystem services that may take place over time. While a dynamic model could potentially be developed, the data and resource requirements associated with developing a dynamic characterization are considerably higher than for a static model and must be weighed against the potential benefits.

Finally, the optimization model does not include any dynamic adjustments, staged implementations of BMPs, or assumptions about other future changes in demographics or urbanization within the Chesapeake Bay watershed. Rather, it compares (1) a representation of current conditions with (2) a menu of steady-state alternatives involving instantaneously lower levels of loadings (designed to meet 2025 goals), higher annual costs, and different levels of annual bonus ecosystem services. The model results described in the next section of the report must be interpreted from the perspective of this simplified framework.

SECTION 4. MODEL SCENARIOS

In this section, we define a number of model run scenarios, which we use to (1) demonstrate the model, (2) investigate the effects of different potential approaches for achieving the TMDLs, and (3) examine the effect of uncertainties regarding model inputs. These scenarios are described below, and the results of these model runs are reported in ***Section 5***.

- **Scenario 1. TMDL Basin-level Targets.** This scenario uses the load-reduction targets specified in Table 3-18. For sediment reductions, it uses the mid-range target load reduction. It includes all of the point-source, agricultural BMP, and urban stormwater control projects described in ***Section 2*** of this report.
- **Scenario 2. Sensitivity to BMP Costs.** This scenario is the same as Scenario 1, except that selected adjustments are applied to the costs of the agricultural and/or urban stormwater BMPs (no additional costs are applied to point sources). The following three "sub-scenarios" are investigated:
  - **Scenario 2(a)** includes a 10% increase on the transaction costs per pound removal for all agricultural and urban stormwater BMPs. Scenario 2(a) is treated as the Base Case for this analysis overall.
  - **Scenario 2(b)** includes a 25% increase on the transaction costs per pound removal for all agricultural and urban stormwater BMPs
  - **Scenario 2(c)** includes an increase in the land rental costs for all agricultural BMPs by a factor of 2.2.

  The motivation for including these adjustments is to account for the likelihood that including nonpoint sources in a nutrient trading framework is likely to involve significant transactions costs. These costs include the time and resources needed to identify and exchange information with potential trading partners, inspect and verify performance, and establish contractual arrangements (Stavins, 1995). For farmers, participating in a water quality market may involve a significant learning process and require large changes in their business practices (Ribaudo et al., 2008). Empirical evidence regarding these costs is very limited; therefore, we apply 10% and 25% cost adjustments to reflect the large uncertainties and potential range of magnitude of these additional costs. The motivation for including the land rental adjustment is to account for the likelihood that the land rental estimates included in the model do not fully capture the compensation that farmers would need to shift working agricultural lands into alternative uses. A 120% increase in land rental rates represents a high-end adjustment to these costs (Hellerstein, 2010).

- **Scenario 3. Sensitivity to BMP Effectiveness.** This scenario and Scenarios 5 through 10, described below, incorporate and build on **Scenario 2(a)**. In addition to the 10% transaction cost adjustment, the pollutant-removal effectiveness of agricultural and urban stormwater BMPs are adjusted for uncertainty in the following ways:
  - **Scenario 3(a)** assumes a 2:1 credit ratio for point sources compared to nonpoint sources by reducing the nutrient removals from each nonpoint-source BMP by 50%.
  - **Scenario 3(b)** assumes a 3:1 credit ratio for point sources compared to nonpoint sources by reducing the nutrient removals from each nonpoint-source BMP by 66.7%.

  The purpose of including these two downward adjustments on BMP effectiveness is to account for their higher uncertainty compared to point-source controls. Credit ratios of this magnitude are often incorporated into water quality trading programs to address this type of uncertainty (Conservation Technology Information Center, 2006).
- **Scenario 4. Sensitivity to Sediment Reduction Targets.** For this scenario, different load-reduction targets for sediment are included in the optimization runs. Using the ranges of sediment load allocations defined in Table 3-17, and the corresponding load-reduction targets in Table 3-18, the following alternatives are investigated:
  - **Scenario 4(a)** includes comparatively large sediment-reduction targets, based on sediment load allocations that are assumed to be below the final TMDL targets.
  - **Scenario 4(b)** includes comparatively small sediment-reduction targets, based on sediment load allocations that are above the final TMDL targets.
  - **Scenario 4(c)** includes no sediment-reduction targets. It solves exclusively for the optimal combination of controls that meets the basin-specific nitrogen and phosphorus targets.

  These sensitivity analyses are included to investigate how the stringency of the sediment targets affects the model solution. In particular, because the point-source controls included in the model are not expected to provide large reductions in sediment loads and the reductions that do occur are not quantified as part of the analysis, we investigate how the stringency of the sediment targets determines the relative contribution of agricultural and urban stormwater BMPs in the model solution.

- **Scenario 5. Sensitivity to Single-Pollutant Reduction Targets.** For this scenario, load-reduction targets are specified for only one of the three pollutants (i.e., no restrictions are placed on loadings of the other two pollutants):
    - **Scenario 5(a)** only includes load-reduction targets for nitrogen.
    - **Scenario 5(b)** only includes load-reduction targets for phosphorus.
    - **Scenario 5(c)** only includes load-reduction targets for sediment.

  These sensitivity analyses are included to investigate which of the three individual pollutant-reduction targets has the largest effect on total costs and total pollutant removals.
- **Scenario 6. Sensitivity to Carbon Prices.** For this scenario, alternative values are assumed for the GHG emission reductions and carbon sequestration. Based on the discussion in ***Section 3.4***, rather than assuming $45 per ton of carbon ($12 per ton of CO_2e), the following alternative low- and high-end values were assumed
    - **Scenario 6(a)** assumes $26 per ton of carbon ($7 per ton of CO_2e)
    - **Scenario 6(b)** assumes $92 per ton of carbon ($25 per ton of CO_2e)

  In addition to addressing some of the uncertainty associated with the value of reducing GHG emissions and sequestering carbon, this scenario allows us to investigate how the NET cost minimization solution (including bonus ecosystem services) is affected by changes in value of these services.
- **Scenario 7. WWTP Technology Limits.** In this scenario, WWTPs are required to implement all of their available control projects (i.e., Tier 4 technology is required). The model then solves for optimal combination for the remaining nonpoint-source controls. The following two main sub-scenarios are considered:
    - **Scenario 7(a)** assumes Tier 4 requirements for WWTPs
    - **Scenario 7(b)** assumes Tier 4 requirements for WWTPs <u>and</u> a 2:1 credit ratio for nonpoint-source BMPs
- **Scenario 8. Land Conversion Limits.** In this scenario, conversion of agricultural land through conversion to forest, wetland restoration, or natural revegetation is limited in different ways. The purpose is to investigate how limits on land conversion (in favor of

"working land" BMPs) affects costs and bonus ecosystem services. The following sub-scenarios are considered:

- **Scenario 8(a)** assumes no land conversion beyond 100 feet from the stream network. In other words, forest and grass buffers and livestock exclusion are included as options, but conversion to forest, wetland restoration, and natural revegetation are not
- **Scenario 8(b)** assumes no application of natural revegetation is included as an option
- **Scenario 8(c)** assumes no land conversion beyond 1,044 feet from the stream network.

- **Scenario 9. Minimum Wetland Conversion.** This scenario restricts land conversion by requiring a *minimum* amount of wetland conversion. The purpose of this scenario is to examine the cost implications of requiring wetland conversion as a component of achieving the TMDL. The following sub-scenarios are considered:
  - **Scenario 9(a)** assumes at least 30,000 acres of wetland conversion from agricultural land
  - **Scenario 9(b)** assumes at least 60,000 acres of wetland conversion from agricultural land.
- **Scenario 10. Bay-Wide Targets with Basin-level Load Adjustment Factors.** In this scenario, we applied Bay-wide load-reduction targets rather than basin-level targets. As in Scenario 5(a), we also only included nitrogen targets. The Bay-wide targets for nitrogen were set equal to the sum of the basin-level targets in Scenario 1. However, in the optimization model, loads from each basin were adjusted (weighted) by their respective Estuarine Delivery Factors[22] such that the weighted sum of basin-level nitrogen load reductions had to meet the Bay-wide targets. This scenario explores the implications of increased location flexibility in meeting overall load reduction goals, while accounting for the differential effects of loadings from different basins on Chesapeake Bay water quality. Because basin-specific delivery factors were only available for nitrogen, this scenario excluded the phosphorus and sediment reduction targets.

[22] The following sub-basin factors were averaged at the basin-level estuarine delivery factor: Susquehanna, 10.319; WestShore, 7.914; Patux AFL, 3,093; Patux BFL, 6.377; Potomac AFL, 6.188; Potomac BFL, 6.174; Rapp AFL, 2.81; Rapp BFL, 4.482; York AFL, 0.798; York BFL, 1.854; James AFL, 0.533; James BFL, 0.793; Upper EastShore, 7.502; Mid EastShore, 6.93; Lower EastShore, 7.971; VA EastShore, 5.716.

Table 4-1 provides a summary of the scenarios. Scenarios 2, 3, 4, 5, and 6 primarily involve sensitivity analyses, allowing for and investigating the effect of varying selected model parameters, assumptions, and constraints. The main objectives in conducting these sensitivity analyses are the following:

- Test and demonstrate the validity of the model (i.e., are the direction and magnitude of the changes in the model solution reasonable and consistent with expectations?)
- Investigate how strongly the cost minimizing mix of controls depends on selected features of the model.

In Scenarios 7, 8, and 9, we explore the effects of different possible approaches and restrictions for achieving the TMDL goals.

In addition, all of the scenarios are run for the two optimization conditions described in this section and in ***Sections 1 and 3***—a least-cost solution and a least-NET-cost solution. In the first case, the model solves for the combination of control projects that meets the load-reduction targets at the lowest total cost. In the second case, NET costs are estimated by deducting monetized bonus ecosystem services from control costs, and the model solves for the combination of control projects that meets the load-reduction targets at the lowest total NET cost.

Table 4-1. Summary of Model Scenarios

Scenario	Description	Options
1	Uses Bay Strategy's load reduction for nitrogen, phosphorus, and sediment	
2	The same as Scenario 1, but with (1) added transaction costs for agricultural and stormwater BMPs and (2) increased agricultural land costs	a) Agricultural and stormwater BMP transaction costs 10% greater than Scenario 1 (Base Case) b) Agricultural and stormwater BMP costs 25% greater than Scenario 1 c) Increase land rental costs for agricultural BMPs by 120%
3	The same as Scenario 2(a), but with BMP pollution removal effectiveness adjustments	a) Assume agricultural and stormwater BMPs are 50% as effective as point sources (2:1 credit ratio) b) Assume agricultural and stormwater BMPs are 67% less effective as point sources (3:1 credit ratio)

Scenario	Description	Options
4	The same as Scenario 2(a), but with sediment options	a) Lower target sediment load allocation (higher reduction target) b) Higher target sediment load allocation(lower reduction target) c) No sediment reduction
5	The same as Scenario 2(a), but with single-pollutant controls	a) Nitrogen reduction target only b) Phosphorus reduction target only c) Sediment reduction target only
6	The same as Scenario 2(a), but with carbon price options	a) $26 per ton of carbon ($7 per ton of CO_2e) b) $92 per ton of carbon ($25 per ton of CO_2e)
7	The same as Scenario 2(a), but with WWTP technology requirements	a) WWTP operating at Tier 4 levels b) WWTP operating at Tier 4 levels and a 2:1 credit ratio for point sources compared to nonpoint sources (i.e., nonpoint-source BMPs are 50% as effective as point-source control project)
8	The same as Scenario 2(a), but with agricultural land conversion restrictions	a) No agricultural land conversion beyond 100 feet from stream b) No natural revegetation c) No agricultural land conversion beyond 1,044 feet from stream
9	The same as Scenario 2(a), but with minimum agricultural wetland restoration restrictions	a) Minimum of 30,000 acres of agricultural wetland restoration b) Minimum of 60,000 acres of agricultural wetland restoration
10	The same as Scenario 5(a), but with Bay-wide nitrogen reduction targets and inter-basin BMP credit ratios (rather than river basin-level targets)	

SECTION 5. RESULTS

In this section, we summarize some of the key findings and compare results from the multiple model runs and scenarios. Detailed results for each scenario are provided in **Appendix D**.

5.1 BASE CASE RESULTS: SCENARIO 2(a)

To provide context for the model results for the rest of the scenarios, we use Scenario 2(a) as the Base Case (i.e., point of reference). As described in ***Section 4***, this scenario is the same as Scenario 1 (TMDL Basin-level Targets), except that it also includes transaction costs for agricultural and urban stormwater nonpoint-source BMPs. More specifically, it uses the basin-level TMDL load-reduction targets specified in Table 3-18, and it includes all of the point-source, agricultural BMP, and urban stormwater control projects described in ***Section 2*** of this report. To make the scenario somewhat more realistic and relevant for policy purposes, it also includes a supplemental 10% transaction cost for all nonpoint-source controls.

Tables 5-1 through 5-6 report detailed results for Base Case Scenario 2(a). [23] As shown in Table 5-1, the least-cost solution for Scenario 2(a) produces a total annual control cost estimate[24] (including the transaction costs) of $218 million, with 64% of these costs attributable to agricultural BMPs, 34% attributable to point-source controls, and the remainder attributable to urban stormwater BMPs. Agricultural BMPs also account for 82%, 67%, and 96% of the total nitrogen, phosphorus, and sediment reductions, respectively. Even though bonus ecosystem services are not included in the optimization routine for this model run, the BMPs still contribute $90 million per year in measurable bonus ecosystem services (99% from agricultural BMPs), resulting in total NET costs of $129 million per year. Overall, natural revegetation of agricultural land accounts for most of the land-use change (2 million acres), pollutant removals (52–66%), costs (49%), and bonus ecosystem services (98%). Interestingly, several agricultural BMP categories (i.e., conversion to forest, wetland restoration, and reduced fertilizer) application and urban stormwater BMP categories (i.e., urban buffers, bioretention planters, and extended detention ponds) are never selected as part of the least-cost solution for this scenario. In other

[23] Throughout the remainder of this section, the detailed cost charts will not be displayed for each scenario; these can be found in **Appendix D**.

[24] All costs in this report are reported as average annual costs over the period 2010–2025.

words, the model is able to meet the basin-level load-reduction targets at a lower total cost by selecting other available point- and nonpoint- source controls.

Tables 5-2 and 5-3 report results by basin for the least-cost solution. Of the eight basins, the Susquehanna contributes the largest portion of nutrient removals (55% of nitrogen and 28% of phosphorus), sediment removals (34%), BMP acres (44%), costs (36%), and bonus ecosystem services (49%).

Overall, the least-cost solution reduces total phosphorus and sediment loads by more than their respective targets (by 4% and 33%, respectively). For phosphorus, these exceedances are achieved in the Eastern Shore and Rappahannock basins. For sediments, the exceedances primarily occur in five basins—the Potomac, Patuxent, Rappahannock, Susquehanna, and Western Shore. Therefore, the optimization results are primarily driven by the nitrogen-reduction targets and, to a lesser extent, the phosphorus targets. The sediment-reduction targets play less of a role and mainly affect the least-cost solution in three basins—James, York, and Eastern Shore.

Table 5-3 shows that carbon sequestration and reduced GHG emissions account for over 96% of the total monetized bonus ecosystem services generated by the least-cost solution. In addition, the least-cost solution is estimated to restore brook trout habitat to intact condition in 13 subwatersheds and to provide 20 thousand acre feet of water storage in urban wetlands.

Tables 5-4 and 5-6 report results for the least NET cost solution to Base Case Scenario 2(a). In these model runs, the monetized bonus ecosystem services are deducted from the costs of each pollution-control project, and the model solves for the combination of projects that meets the basin-level targets at the lowest total NET cost. In this case, the total nutrient- and sediment-control costs actually increase by $83 million, compared to the least-cost solution of $218 million in Table 5-1; however, these increases are more than offset by a $148 million increase in bonus ecosystem services (compared to $89.83 million in Table 5-1). As a result, the total NET costs are $65 million (51%) lower than in the least-cost solution reported in Tables 5-1 and 5-2. The increase in costs and ecosystem services occur mainly as a result of an even larger selection of natural revegetation (6.1 million acres, compared to 1.996 million acres in Table 5-1). The model solution is dominated by large-scale natural revegetation because, in many instances, the estimated bonus ecosystem services provided by an acre of natural revegetation (in particular, carbon sequestration) are larger than the costs (i.e. the NET costs are *negative*). Consequently, the total NET costs of natural revegetation are negative (-$8.4 million) in the model solution.

Tables 5-5 and 5-6 report results by basin for the least NET cost solution. Once again, the Susquehanna accounts for the largest portion of BMP acres and pollutant reductions; however, the Western Shore accounts for the largest NET costs ($42 million). Due to the large ecosystem services derived from natural revegetation, NET costs are negative in four of the basins (Patuxent, Rappahannock, Potomac, and York). Table 5-6 shows that the large increase in bonus ecosystem services (compared to the least-cost solution) is primarily attributable to the increased carbon sequestration, which mainly comes from the large increase in natural revegetation.

Figures 5-1, 5-2, and 5-3 provide maps of the Chesapeake Bay watershed showing the spatial distribution of loading reductions (i.e., delivered loads to the Bay) for Base Case Scenario 2(a) (least-cost solution). As expected, the load reductions are more heavily concentrated in the basins with the highest load-reduction targets (per acre) and their near tidal areas.

Table 5-1. Least-Cost Solution for Scenario 2(a): Load Reductions, Costs, and Bonus Ecosystem Services by Control Category

Control Category	Area (million acres)	Annual N Reduction (million lbs/yr)	Annual P Reduction (million lbs/yr)	Annual Sediment Reduction (billion lbs/yr)	Annual Control Costs ($millions/yr)	Bonus Ecosystem Services ($millions/yr)[b]	Annual NET Costs ($millions/yr)
POTW Advanced Nutrient Removal	-	10.03	1.00	-	68.65	-	68.65
Industrial Advanced Nutrient Removal	-	0.67	0.21	-	6.65	-	6.65
Point Source Subtotal	**-**	**10.70**	**1.21**	**-**	**75.30**	**-**	**75.30**
Forest Buffers	0.006	0.51	0.01	0.014	1.76	0.34	1.42
Grass Buffers	0.005	0.20	0.01	0.004	0.99	0.09	0.90
Conversion to Forest	-	-	-	-	-	-	-
Natural Revegetation	1.996	39.40	2.13	1.133	108.08	88.05	20.02
Livestock Exclusion	0.056	2.13	0.35	0.430	5.96	0.00	5.96
Restored Wetlands	-	-	-	-	-	-	-
Cover Crops	0.509	5.75	0.09	0.090	17.59	-	17.59
No-till	0.287	0.85	0.16	0.283	4.45	0.46	3.99
Reduced Fertilizer Application	-	-	-	-	-	-	-
Agriculture BMP Subtotal[a]	**2.860**	**48.85**	**2.75**	**1.954**	**138.83**	**88.94**	**49.89**
Extended detention pond	-	-	-	-	-	-	-
Bioretention Planters	-	-	-	-	-	-	-
Urban Grass Buffers	-	-	-	-	-	-	-
Urban Forest Buffers	-	-	-	-	-	-	-
Urban Wetlands	0.007	0.35	0.12	0.073	4.31	0.90	3.42
Urban SW BMP Subtotal	**0.007**	**0.35**	**0.12**	**0.073**	**4.31**	**0.90**	**3.42**
Total	**2.866**	**59.91**	**4.08**	**2.027**	**218.44**	**89.83**	**128.61**

[a] Acre subtotal may be less than the sum of individual BMP acres because more than one BMP may be applied to individual acres.

[b] Includes only monetized ecosystem services (i.e., carbon sequestration, GHG emission reduction, hunting, and air quality benefits).

Table 5-2. Least-Cost Solution for Scenario 2(a): Load Reductions, Costs, and Bonus Ecosystem Services by Basin

Control Category	Area (million acres)	Annual N Reduction (million lbs/yr)	Annual P Reduction (million lbs/yr)	Annual Sediment Reduction (billion lbs/yr)	Annual Control Costs ($millions/yr)	Bonus Ecosystem Services ($millions/yr)[a]	Annual NET Costs ($millions/yr)
Eastern Shore of Chesapeake Bay	0.438	4.74	0.35	0.039	22.227	9.366	12.861
James River Basin	0.452	8.18	0.89	0.326	40.693	17.699	22.994
Patuxent River Basin	0.017	0.20	0.05	0.016	0.884	0.814	0.069
Potomac River Basin	0.533	6.77	1.03	0.693	23.215	11.785	11.430
Rappahannock River Basin	0.062	1.01	0.26	0.171	3.143	2.673	0.470
Susquehanna River Basin	1.251	33.14	1.16	0.697	78.463	43.654	34.809
Western Shore of Chesapeake Bay	0.053	4.91	0.26	0.062	45.588	2.078	43.510
York River Basin	0.061	0.95	0.08	0.024	4.229	1.763	2.466
Total	**2.866**	**59.91**	**4.08**	**2.027**	**218.441**	**89.833**	**128.608**

[a] Includes only monetized ecosystem services -- i.e., carbon sequestration, GHG emission reduction, hunting, and air quality benefits

Table 5-3. Least-Cost Solution for Scenario 2(a): Bonus Ecosystem Services

Basin	Monetized Ecosystem Services ($millions/yr)					Brook Trout Habitat		Wetland Water Storage (Acre-feet)
	Carbon Sequestration & Reduced GHG Emissions	Non-Waterfowl Hunting	Duck Hunting	Air Quality	Total	Number of Additional Subwatersheds with "Reduced" Brook Trout Habitat	Number of Additional Subwatersheds with "Intact" Brook Trout Habitat	
Eastern Shore of Chesapeake Bay	9.20	0.16	-	0.00	9.36	-	-	16
James River Basin	17.30	0.39	-	0.02	17.71	9	(6)	7,357
Patuxent River Basin	0.79	0.02	-	0.00	.081	-	-	52
Potomac River Basin	11.44	0.33	-	0.01	11.77	3	-	3,359
Rappahannock River Basin	2.63	0.05	-	-	2.68	-	1	0
Susquehanna River Basin	41.24	2.42	-	0.00	43.66	70	18	189
Western Shore of Chesapeake Bay	2.01	0.05	-	0.02	2.08	-	-	8,567
York River Basin	1.73	0.03	-	0.00	1.76	-	-	63
Total	**86.34**	**3.45**	0.00	**0.05**		**82**	**13**	**19,603**

Table 5-4. Least-NET-Cost Solution for Scenario 2(a): Load Reductions, Costs, and Bonus Ecosystem Services by Control Category

Control Category	Area (million acres)	Annual N Reduction (million lbs/yr)	Annual P Reduction (million lbs/yr)	Annual Sediment Reduction (billion lbs/yr)	Annual Control Costs ($millions/yr)	Bonus Ecosystem Services ($millions/yr)[b]	Annual NET Costs ($millions/yr)
POTW Advanced Nutrient Removal	-	9.05	0.44	-	58.54	-	58.54
Industrial Advanced Nutrient Removal	-	0.46	0.11	-	4.20	-	4.20
Subtotal	**-**	**9.51**	**0.55**	**-**	**62.74**	**-**	**62.74**
Forest Buffers	0.000	0.03	0.00	0.000	0.07	0.01	0.06
Grass Buffers	-	-	-	-	-	-	-
Conversion to Forest	-	-	-	-	-	-	-
Natural Revegetation	6.116	46.74	3.72	1.813	229.01	237.37	(8.36)
Livestock Exclusion	0.007	0.40	0.05	0.085	0.88	0.00	0.88
Restored Wetlands	-	-	-	-	-	-	-
Cover Crops	0.143	3.03	0.02	0.039	4.93	-	4.93
No Till	0.096	0.50	0.06	0.105	1.50	0.15	1.34
Reduced Fertilizer Application	-	-	-	-	-	-	-
Subtotal	**6.363**	**50.70**	**3.85**	**2.043**	**236.39**	**237.54**	**(1.14)**
Extended detention pond	-	-	-	-	-	-	-
Bioretention Planters	-	-	-	-	-	-	-
Urban Grass Buffers	-	-	-	-	-	-	-
Urban Forest Buffers	-	-	-	-	-	-	-
Urban Wetlands	0.003	0.19	0.07	0.038	2.31	0.48	1.83
Subtotal	**0.003**	**0.19**	**0.07**	**0.038**	**2.31**	**0.48**	**1.83**
Total	**6.366**	**60.40**	**4.46**	**2.081**	**301.44**	**238.02**	**63.42**

[a] Acre subtotal may be less than the sum of individual BMP acres because more than one BMP may be applied to individual acres

[b] Includes only monetized ecosystem services -- i.e., carbon sequestration, GHG emission reduction, huting, and air quality benefits

Table 5-5. Least-NET-Cost Solution for Scenario 2(a): Load Reductions, Costs, and Bonus Ecosystem Services by Basin

Control Category	Area (million acres)	Annual N Reduction (million lbs/yr)	Annual P Reduction (million lbs/yr)	Annual Sediment Reduction (billion lbs/yr)	Annual Control Costs ($millions/yr)	Bonus Ecosystem Services ($millions/yr)[a]	Annual NET Costs ($millions/yr)
Eastern Shore of Chesapeake Bay	0.378	4.74	0.48	0.043	25.87	17.99	7.88
James River Basin	0.835	8.18	0.89	0.349	46.758	32.07	14.69

Control Category	Area (million acres)	Annual N Reduction (million lbs/yr)	Annual P Reduction (million lbs/yr)	Annual Sediment Reduction (billion lbs/yr)	Annual Control Costs ($millions/yr)	Bonus Ecosystem Services ($millions/yr)[a]	Annual NET Costs ($millions/yr)
Patuxent River Basin	0.042	0.30	0.05	0.023	1.865	1.97	(0.10)
Potomac River Basin	1.473	7.01	1.03	0.510	46.844	51.19	(4.34)
Rappahannock River Basin	0.333	1.16	0.44	0.386	8.279	13.04	(4.76)
Susquehanna River Basin	2.975	33.14	1.16	0.688	115.725	107.22	8.51
Western Shore of Chesapeake Bay	0.104	4.91	0.26	0.058	47.104	4.70	42.41
York River Basin	0.226	0.95	0.15	0.024	8.999	9.85	(0.85)
Total	**6.366**	**60.40**	**4.46**	**2.081**	**301.44**	**238.02**	**63.42**

[a] Includes only monetized ecosystem services -- i.e., carbon sequestration, GHG emission reduction, huting, and air quality benefits

Table 5-6. Least-NET-Cost Solution for Scenario 2(a): Bonus Ecosystem Services

Basin	Monetized Ecosystem Services ($millions/yr)				Brook Trout Habitat		Wetland Water Storage (Acre-feet)
	Carbon Sequestration & Reduced GHG Emissions	Non-Waterfowl Hunting	Duck Hunting	Air Quality	Number of Additional Subwatersheds with "Reduced" Brook Trout Habitat	Number of Additional Subwatersheds with "Intact" Brook Trout Habitat	
Eastern Shore of Chesapeake Bay	17.59	0.39	-	-	-	-	0
James River Basin	31.33	0.73	-	0.01	17	(8)	4,780
Patuxent River Basin	1.91	0.06	-	-	-	-	0
Potomac River Basin	49.31	1.88	-	0.00	(27)	57	258
Rappahannock River Basin	12.75	0.29	-	-	18	(12)	0
Susquehanna River Basin	100.08	7.14	-	0.00	43	129	47
Western Shore of Chesapeake Bay	4.54	0.14	-	0.01	3	-	5,238
York River Basin	9.65	0.20	-	0.00	-	-	172
Total	**227.16**	**10.84**	0.00	**0.02**	**54**	**166**	**10,494**

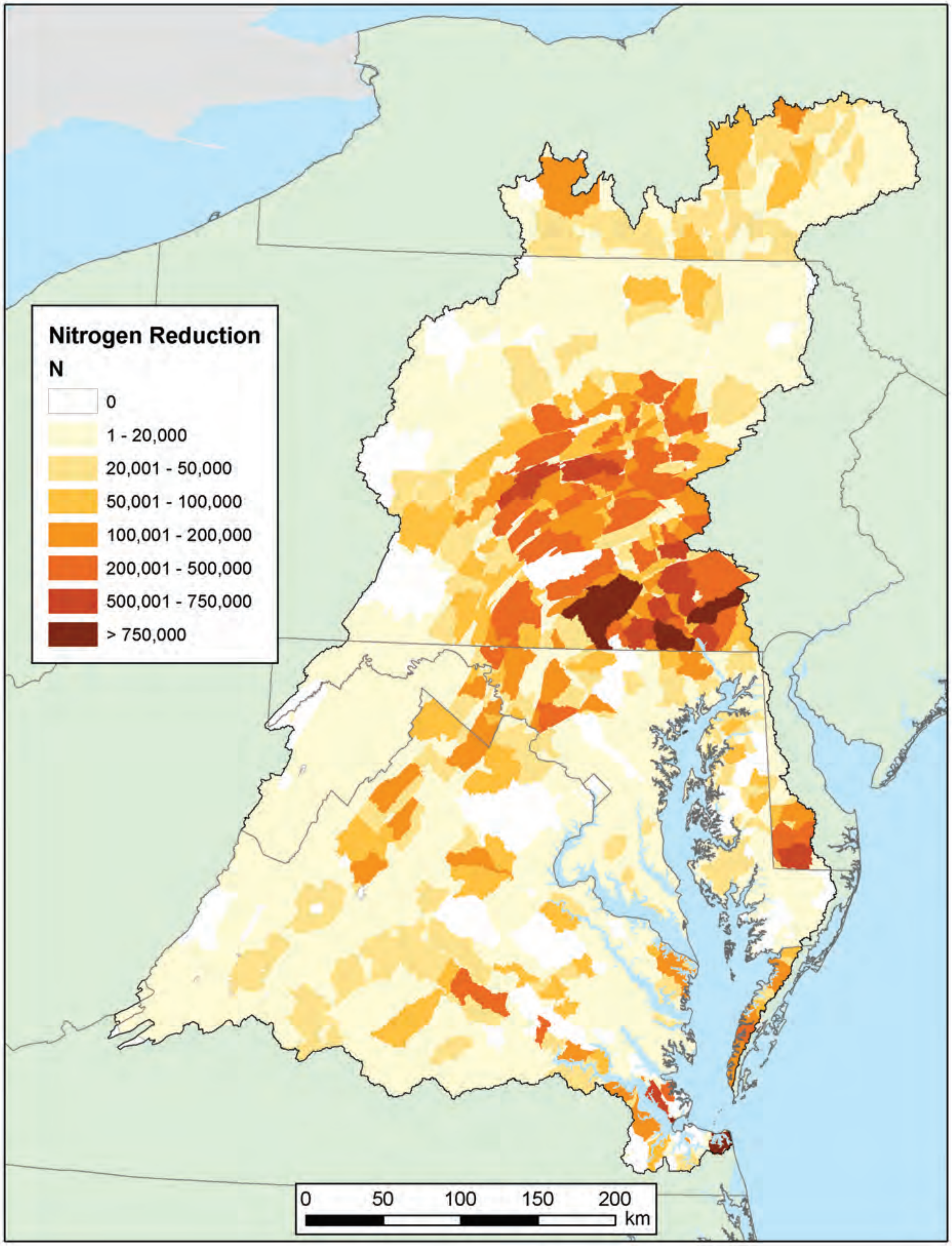

Figure 5-1. Reductions in Delivered Nitrogen Loads by Land-River Segment: Scenario 2(a), Least-Cost Solution.

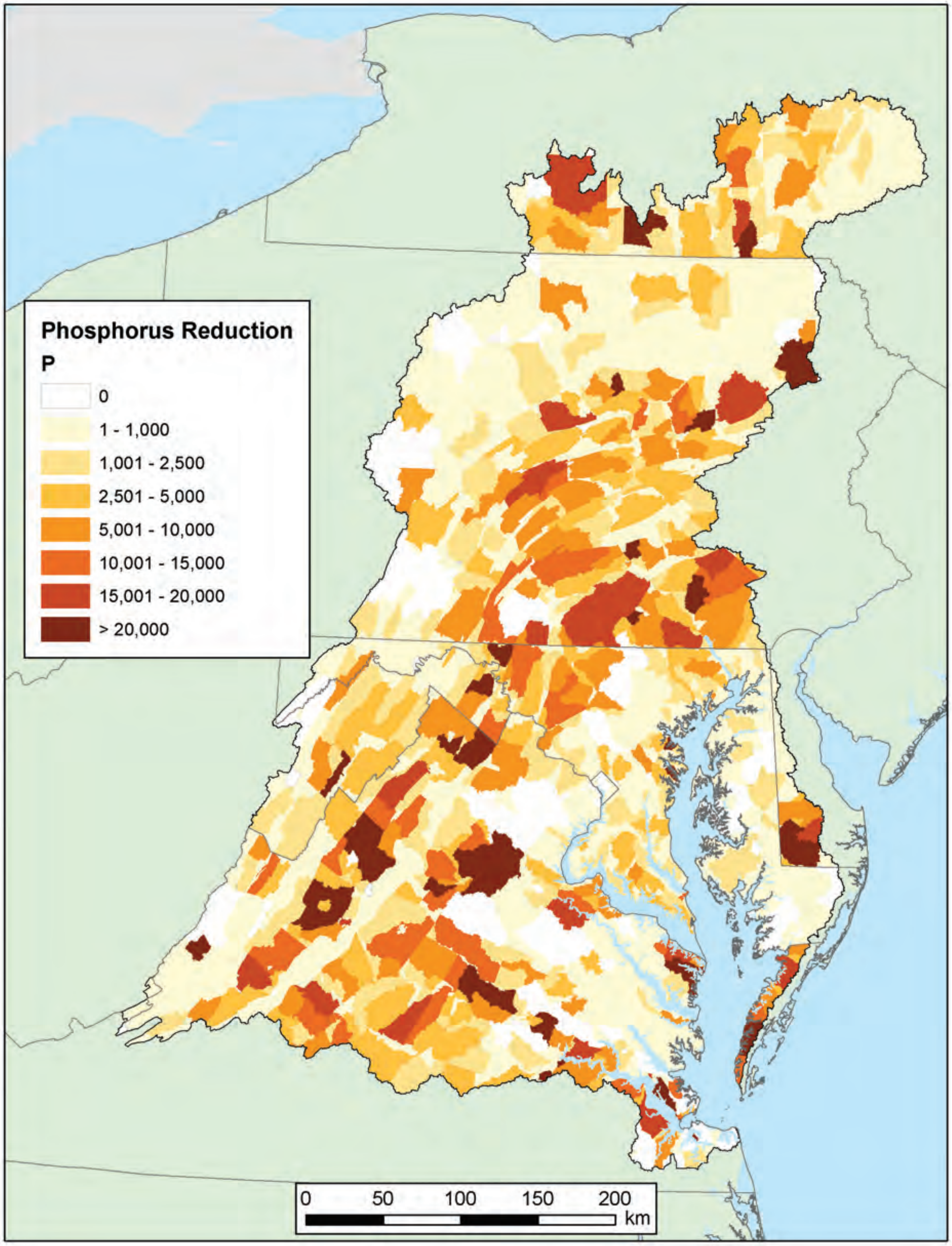

Figure 5-2. Reductions in Delivered Phosphorus Loads by Land-River Segment: Scenario 2(a), Least-Cost Solution.

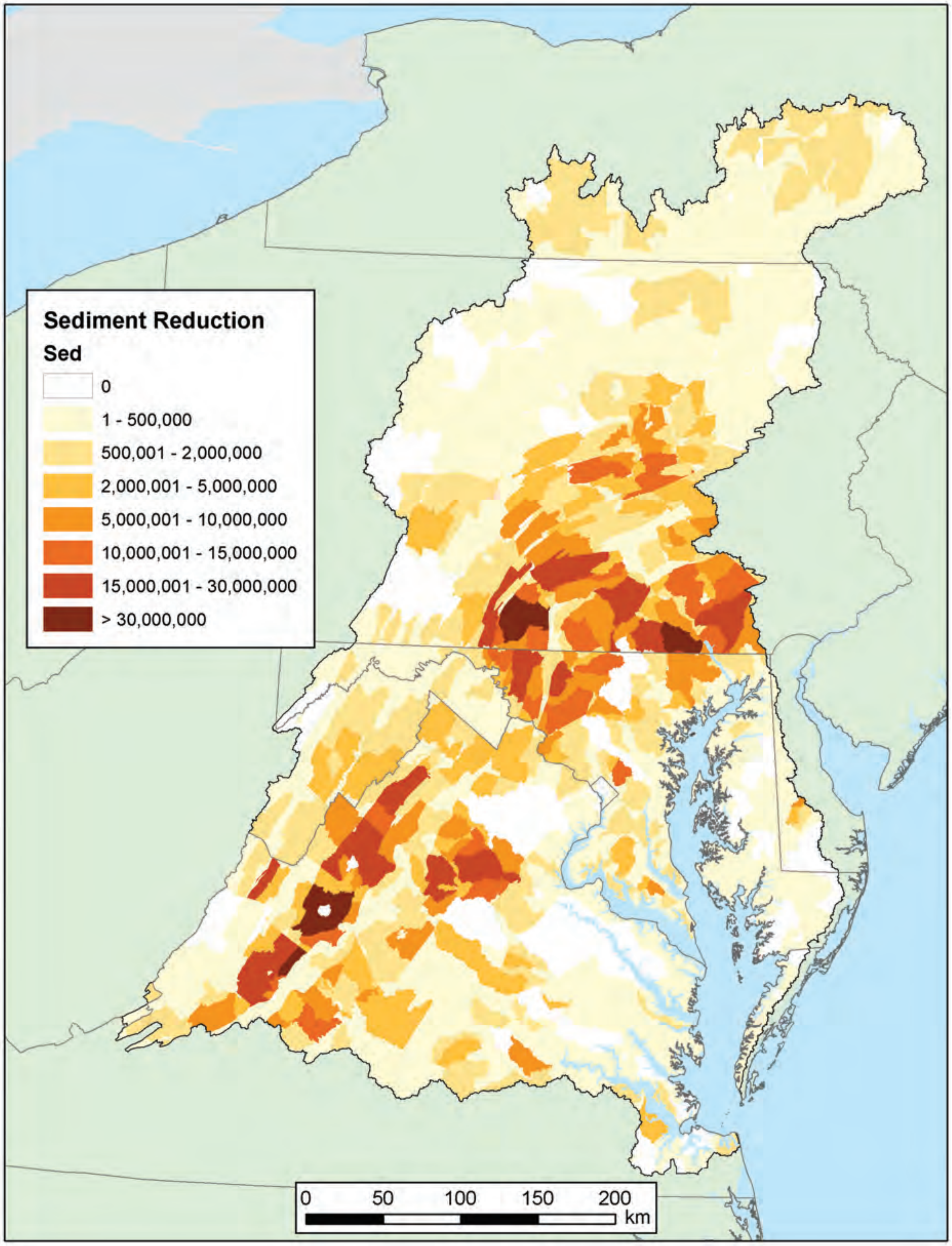

Figure 5-3. Reductions in Delivered Sediment Loads by Land-River Segment: Scenario 2(a), Least-Cost Solution.

5.2 SENSITIVITY ANALYSES

Figure 5-4 compares the total costs and NET cost of nutrient and sediment controls for the Scenario 1 and Scenario 2 model runs. Although not explicitly shown in the graph, the difference between the cost and NET cost values are equal to the bonus ecosystem services, which include different adjustments for the costs of agricultural and urban stormwater BMPs. The figure also shows (1) how these costs and benefits are distributed among point sources, agricultural nonpoint sources, and urban stormwater nonpoint sources, and (2) how these costs and benefits differ between the least-cost solution and the least-NET-cost solution.

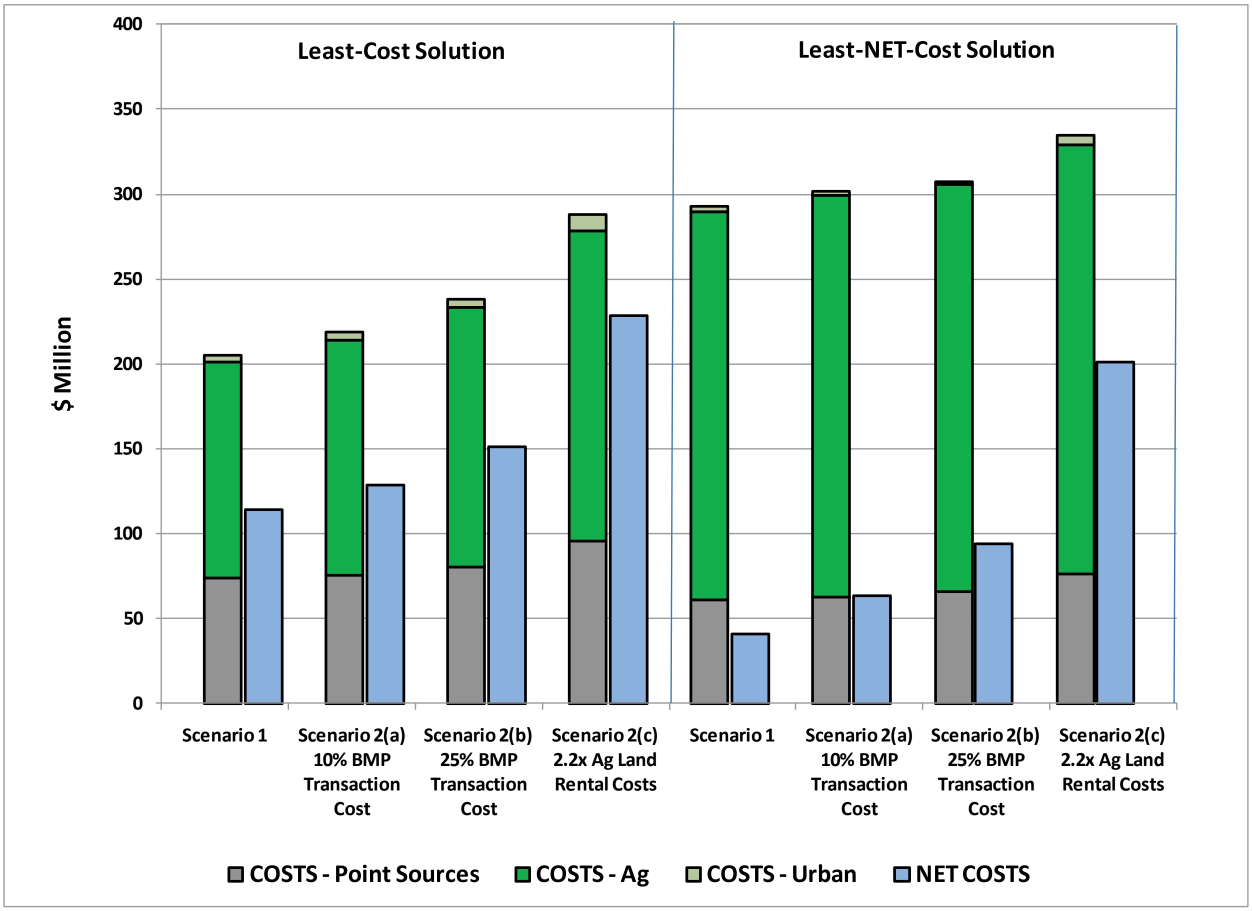

Figure 5-4. Annual costs and NET costs under alternative adjustments to nonpoint-source BMP costs.

Under the least-cost solution, the total control costs for Scenario 1 are $205 million per year, with 62% attributable to agricultural BMP controls and 2% to urban stormwater BMP controls. In Scenarios 2(a), 2(b), and 2(c), the supplementary costs attached to nonpoint-source controls increase the overall costs of nutrient and sediment control by 6%, 16%, and 40%, respectively, compared to Scenario 1. As nonpoint-source controls become more expensive, point-source controls play a larger role in the model solution, and their costs increase by 29% from Scenario 1 to Scenario 2(c). The increasing role of point sources also leads to a 35% decrease in the value of monetized bonus ecosystem services, from $91 million to $59 million. In all four scenarios, urban stormwater BMPs account for less than 4% of the monetized ecosystem services.

Under the least-NET-cost solution, both the control costs and the ecosystem service benefits are significantly higher than under the least-cost solution, but moving from Scenario 1 to Scenario 2(c), they show a similar pattern of increasing costs and declining ecosystem services. For Scenario 1, the costs are $87 million (42%) higher than under the least-cost solution, and the ecosystem services are $161 million (177%) higher. As expected, the resulting total NET costs are lower. For Scenario 2(c), the costs are $47 million (16%) higher compared to the least-cost solution, and the ecosystem services are $74 million (125%) higher; consequently, the NET costs decrease by $27 million.

Figure 5-5 compares the acres of BMP-related land-use change for Scenarios 1, 2(a), 2(b), and 2(c) under both the least-cost and least-NET-cost solutions. Under all scenarios and solutions, the large majority of BMP acres are associated with natural revegetation (i.e., cropland and pastureland) and “working land” options (i.e., cover crops, no-till agriculture, and reduced fertilizer application). Under the least-cost solution, the most notable difference across scenarios occurs as a result of the large increase in agricultural land rents under Scenario 2(c). Although the change in total BMP acres is relatively small across scenarios, the percent attributable to natural revegetation decreases from 2 million acres (70%) in Scenario 1 to 1.2 million acres (42%) in Scenario 2(c), while the percent attributable to working land options increases.

Under the least-NET-cost solution, the amount of land-use change increases significantly compared to the least-cost solution. Due to the relatively large net benefits associated with carbon sequestration, the largest increases are associated with natural revegetation of pastureland. In Scenario 1, natural revegetation of pastureland increases from less than 400,000 acres to over 4.5 million acres. As in the least-cost solution, from Scenario 1 to Scenario 2(c), there is a large shift away from natural revegetation, going from 6.5 million acres (97%) to 3.3 million acres (73%).

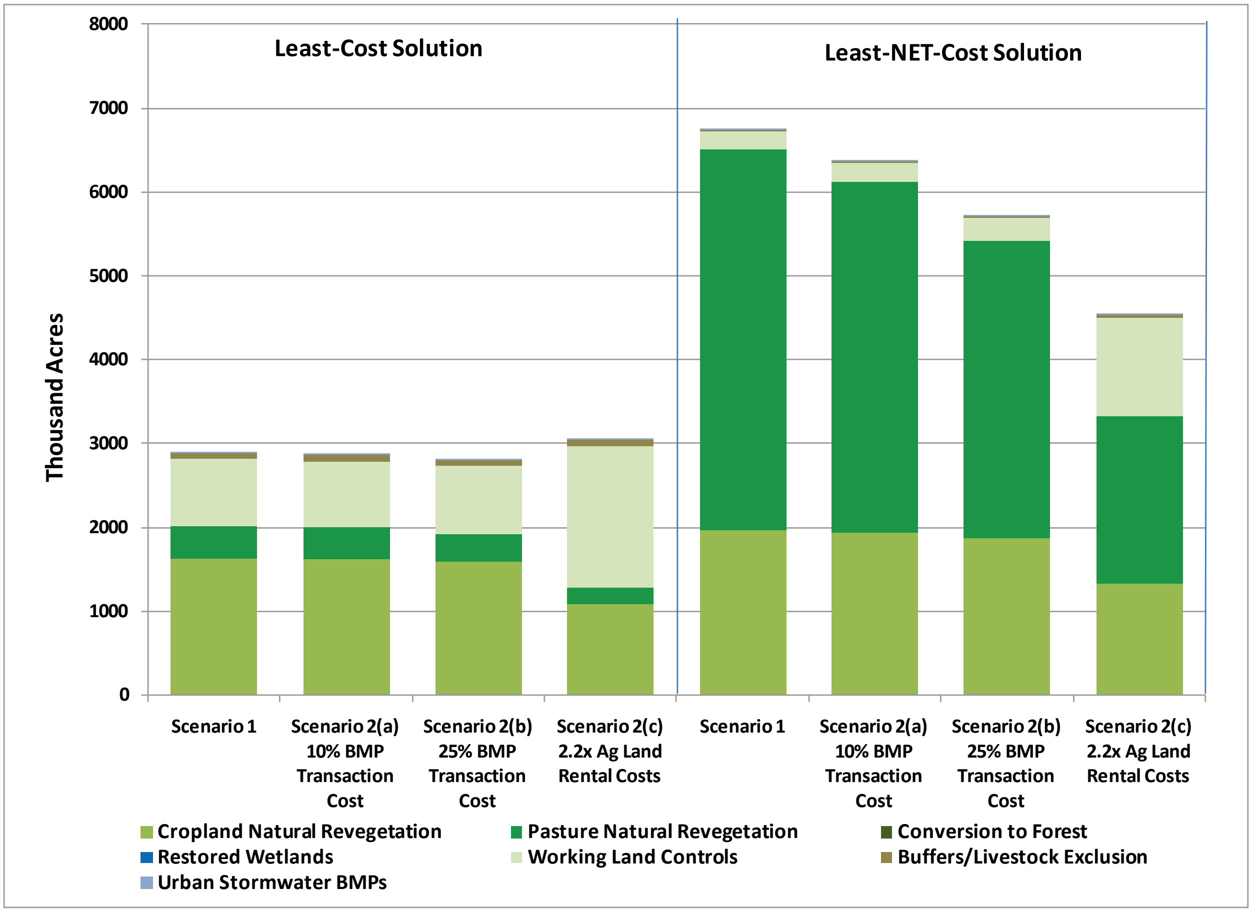

Figure 5-5. Additional BMP acres under alternative adjustments to nonpoint-source BMP costs.

Figure 5-6 compares control cost and ecosystem service estimates based on alternative assumptions regarding BMP effectiveness. The reference (Base Case) scenario is Scenario 2(a), which includes the 10% transaction cost increment for BMPs. The alternatives are the Scenario 3 model runs, which, in addition to the 10% transaction cost increment, also include adjustments to BMP effectiveness. In the least-cost solution, lowering the removal effectiveness of nonpoint-source BMPs in Scenarios 3(a) and 3(b) by 2:1 (50%) and 3:1 (66.7%), respectively, leads to 5.7-fold and 8.3-fold ($1.24 billion and $1.80 billion) increases in total control costs, respectively.[25] To meet the load-reduction targets with BMPs that are overall less effective, the model solution

[25] These total cost estimates are constrained by the model, and may therefore be underestimates. As shown in **Appendix D**, under Scenarios 3(a) and 3(b), some of the basin-level loading reduction targets are not attainable with the control projects included in the model. For example, under Scenario 3(a), the James basin cannot attain the sediment target and the Susquehanna basin cannot attain the nitrogen target. Nonattainment is even more prevalent under Scenario 3(b). As described in Section 2, in these cases, a penalty price is used, which in essence puts a high cap on the costs of additional loading reductions.

relies much more on land conversion to forest. This change entails higher costs, but also increases bonus ecosystem services by $197 million and $285 million, respectively.

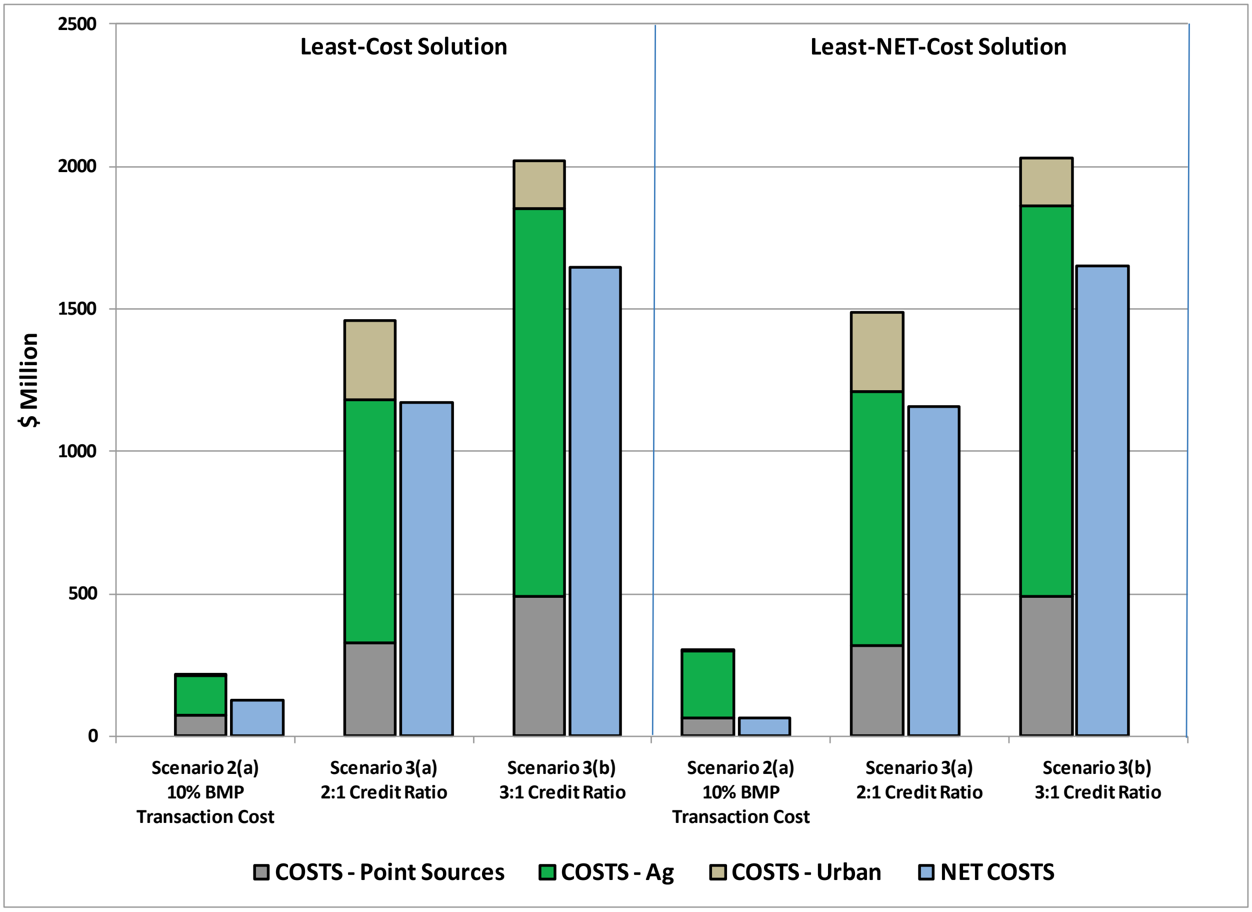

Figure 5-6. Annual costs and NET costs under alternative adjustments to nonpoint-source BMP removal effectiveness.

The shift towards conversion to forest, with the inclusion of 2:1 and 3:1 credit ratios, is shown in **Figure 5-7**. Whereas conversion to forest is not included in the least-cost or least-NET-cost solution for Scenario 2(a), it accounts for roughly 4 million acres in Scenario 3(a) and 7.4 million acres in Scenario 3(b). Interestingly, there is overall very little difference between the least-cost and least-NET-cost solution for Scenario 3(b). In other words, when the effectiveness of the BMPs is significantly reduced, and the target load reductions become more difficult (or impossible) to attain with the control projects included in the model, including the ecosystem services in the optimization routine has much less of an effect on the model solution.

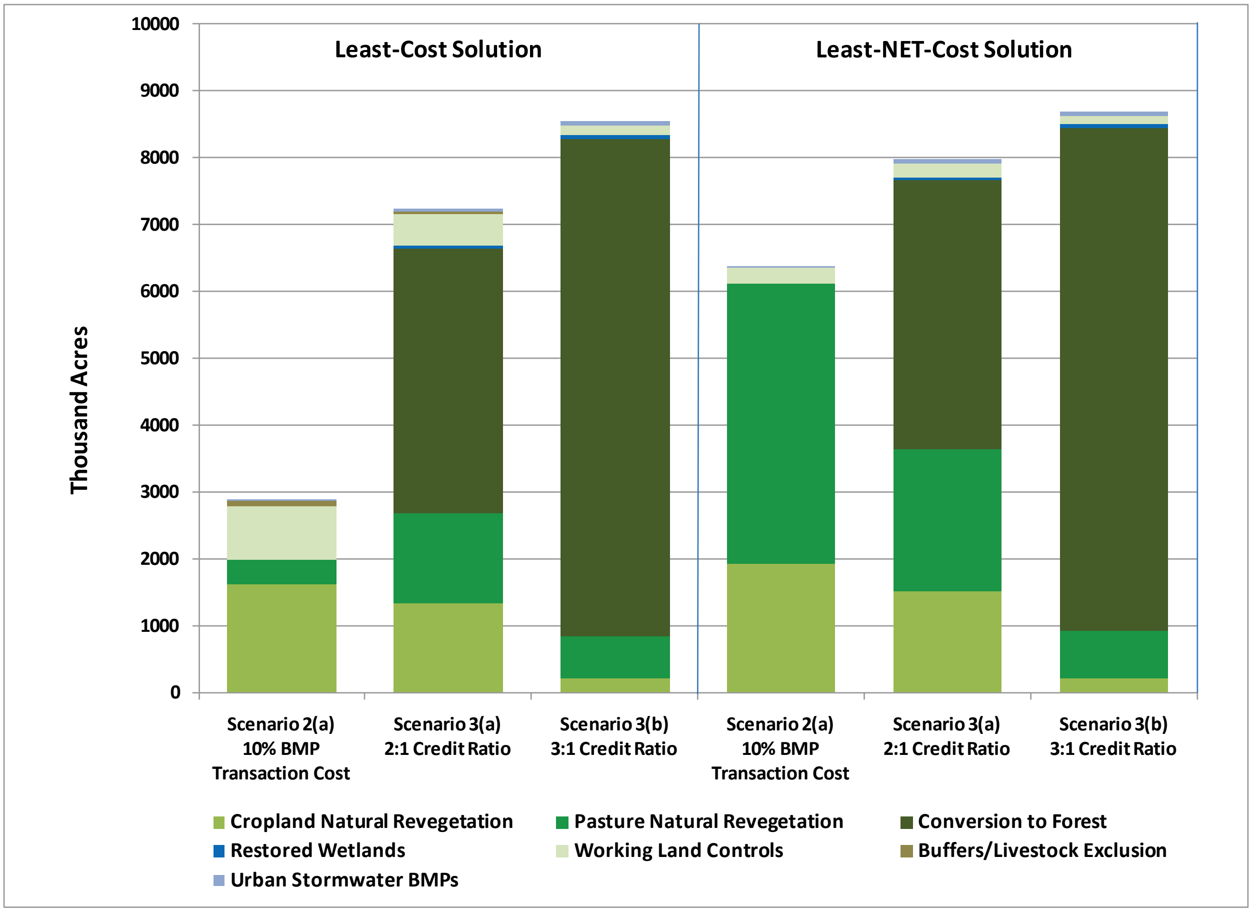

Figure 5-7. Additional BMP acres under alternative adjustments to nonpoint-source BMP removal effectiveness.

Figure 5-8 shows the sensitivity of the model results to changes in the sediment-reduction targets. For comparison, the Base Case is again represented by Scenario 2(a), which includes basin-level sediment-reduction targets based on the TMDL loading allocations. As expected, the total control cost estimate for achieving the lower sediment allocation—Scenario 4(a)— is higher, but by a very small amount (4%) under the least-cost solution. The costs of achieving the higher sediment allocation (i.e., Scenario 4[b]) are essentially identical to Scenario 2(a). As discussed in ***Section 5.1***, under Scenario 2(a), the total sediment reduction target is exceeded by 33%; therefore, the results are primarily driven by the nitrogen and phosphorus targets. The sediment reduction targets are only a binding constraint in three of the eight basins: James, York, and Eastern Shore. Consequently, changing the sediment-reduction targets has relatively little effect on the overall model solution. This conclusion is reinforced by the results of Scenario 4(c), which excludes the sediment-reduction target entirely. In this case, the total control costs decline by less than 1% relative to Scenario 2(a).

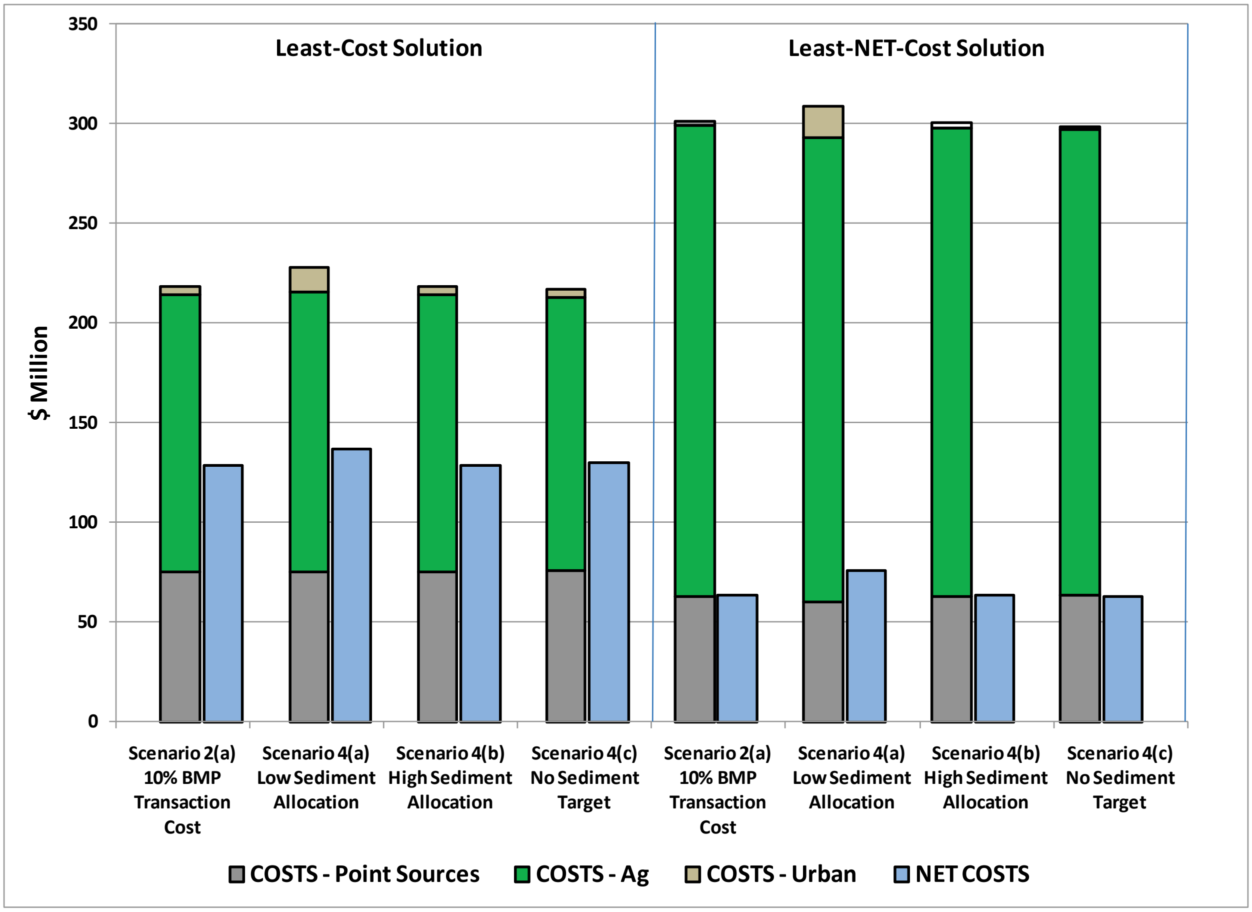

Figure 5-8. Annual costs and NET costs services under alternative adjustments to sediment load allocations and reduction targets.

Figure 5-9 shows how the estimated BMP acres are affected by changes in the sediment-reduction targets. As expected, the changes are again very small, particularly for Scenarios 4(a) and 4(b). When the sediment-reduction target is increased in Scenario 4(a), the main differences are a 29% increase no-till acreage and a 2% increase in natural revegetation under the least-cost solution. When the sediment-reduction target is entirely excluded from the optimization (i.e., Scenario 4[c]), the main effect is to reduce natural revegetation by 5%.

Although not shown in these graphs, another interesting result of Scenario 4(c) is that, even when sediment reduction targets are omitted from the model, the estimated sediment reductions still *exceed* the overall target for the Bay watershed by over 300 million pounds. (See **Appendix D** for details.) For example, in the least-cost solution, the sediment reductions fall below the targets for the Eastern Shore, James, and York River basins; however, these shortfalls are more than offset by large exceedances in the other five basins.

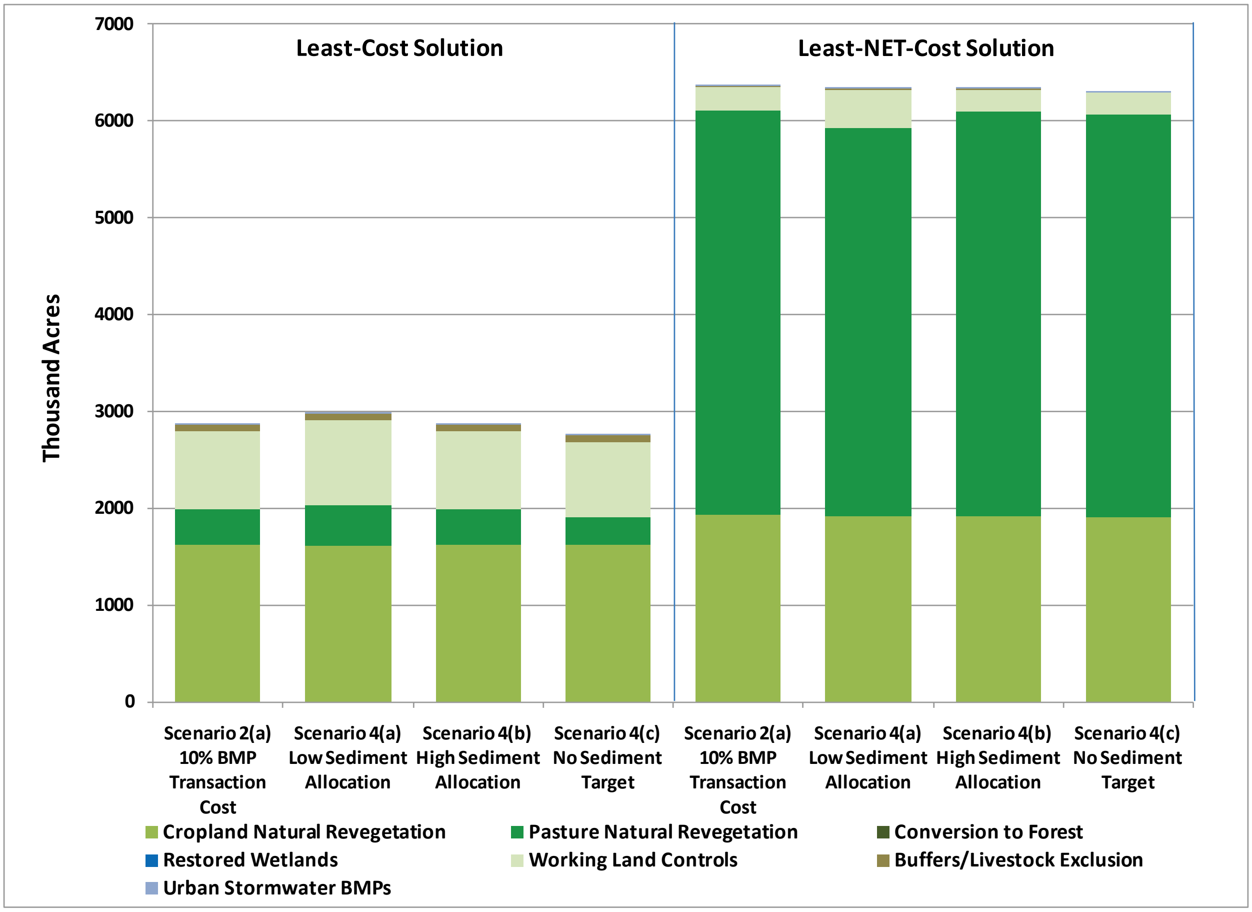

Figure 5-9. Additional BMP acres under alternative adjustments to sediment load allocations and reduction targets.

Figures 5-10 and 5-11 summarize model results for scenarios that only include load reduction targets for one of the three pollutants. Compared to Scenario 2(a), the total costs decline the least for Scenario 5(a), which only includes a nitrogen-reduction target, and decline the most of for Scenario 5(c), which only includes a sediment-reduction target. Under the least-cost solution, total costs decline by 8%, 65%, and 91%, respectively, for the nitrogen-only, phosphorus-only, and sediment-only reduction targets. These results again show that the nitrogen-reduction targets have the strongest effect on the model solution. In contrast, the sediment-reduction targets can be achieved at relatively low cost, in part because they do not require any point-source controls.[26]

[26] Unlike Scenario 4(c), which includes just the nitrogen and phosphorus reduction targets, the least-cost solution to the nitrogen-only scenario (5[a]) does not generate sediment reductions that exceed the Bay-wide sediment reduction target. However, the least-NET-cost solution to 5(a) does generate excess sediment reductions for the watershed as a whole (see **Appendix D** for details).

Figure 5-10 also shows differences in how bonus ecosystem services are affected under the least-cost and least-NET-cost solutions. In both cases, the value of these services declines relative to Scenario 2(a) because fewer BMPs are required; however, they decline by a smaller percentage under the least-NET-cost solution. Interestingly, in the latter case, the value of the bonus ecosystem services *exceeds* the total control costs when the model only includes a phosphorus- or sediment-reduction target (i.e., NET costs are negative). The reason for these relatively high ecosystem service values in Scenario 5(b) and Scenario 5(c) is further shown in **Figure 5-11**. Under the least-NET-cost solution, the single pollutant reduction targets are met almost entirely through natural revegetation of pastureland, which in this modeling framework is assumed to involve relatively low costs per acre and relatively high carbon sequestration potential.

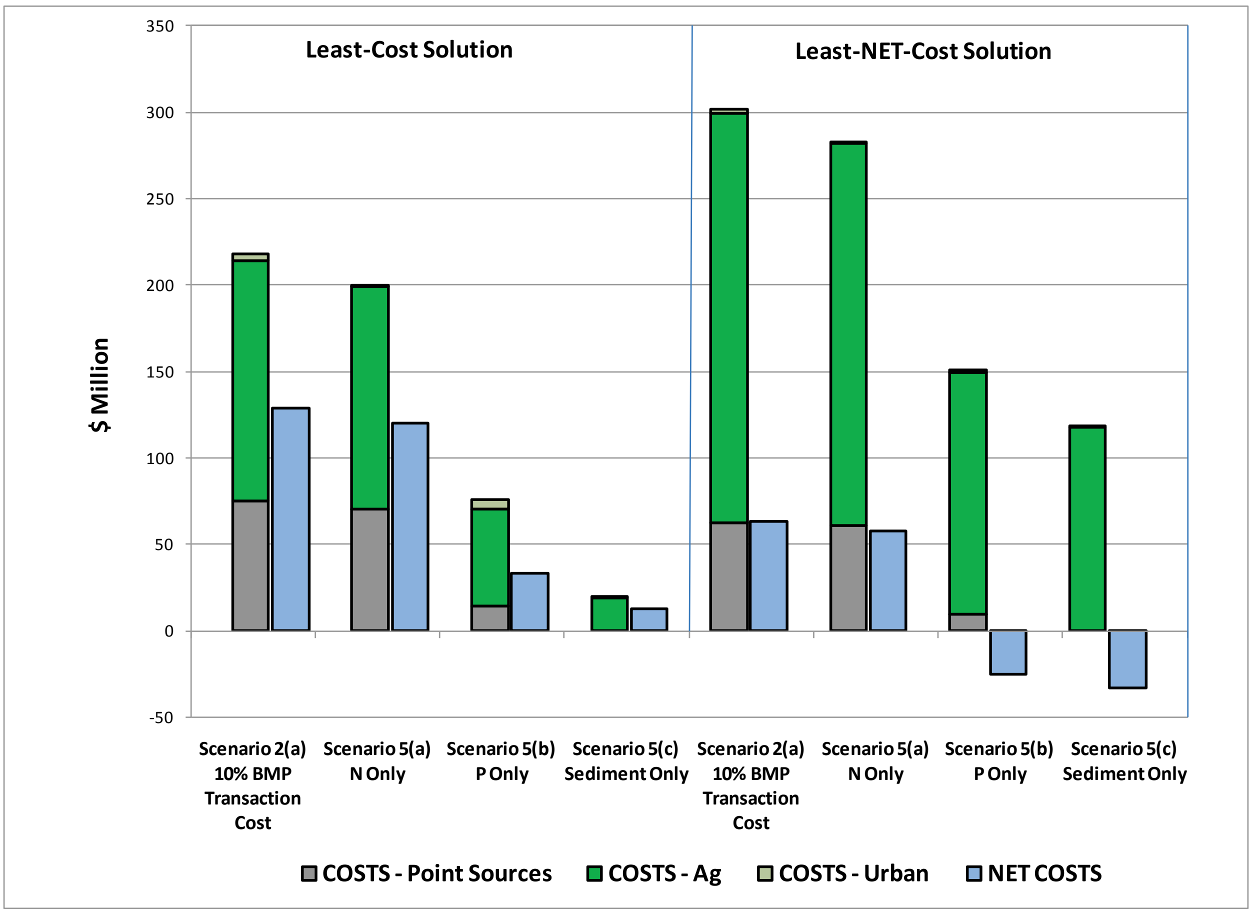

Figure 5-10. Annual costs and NET costs under alternative adjustments to sediment load allocations and reduction targets.

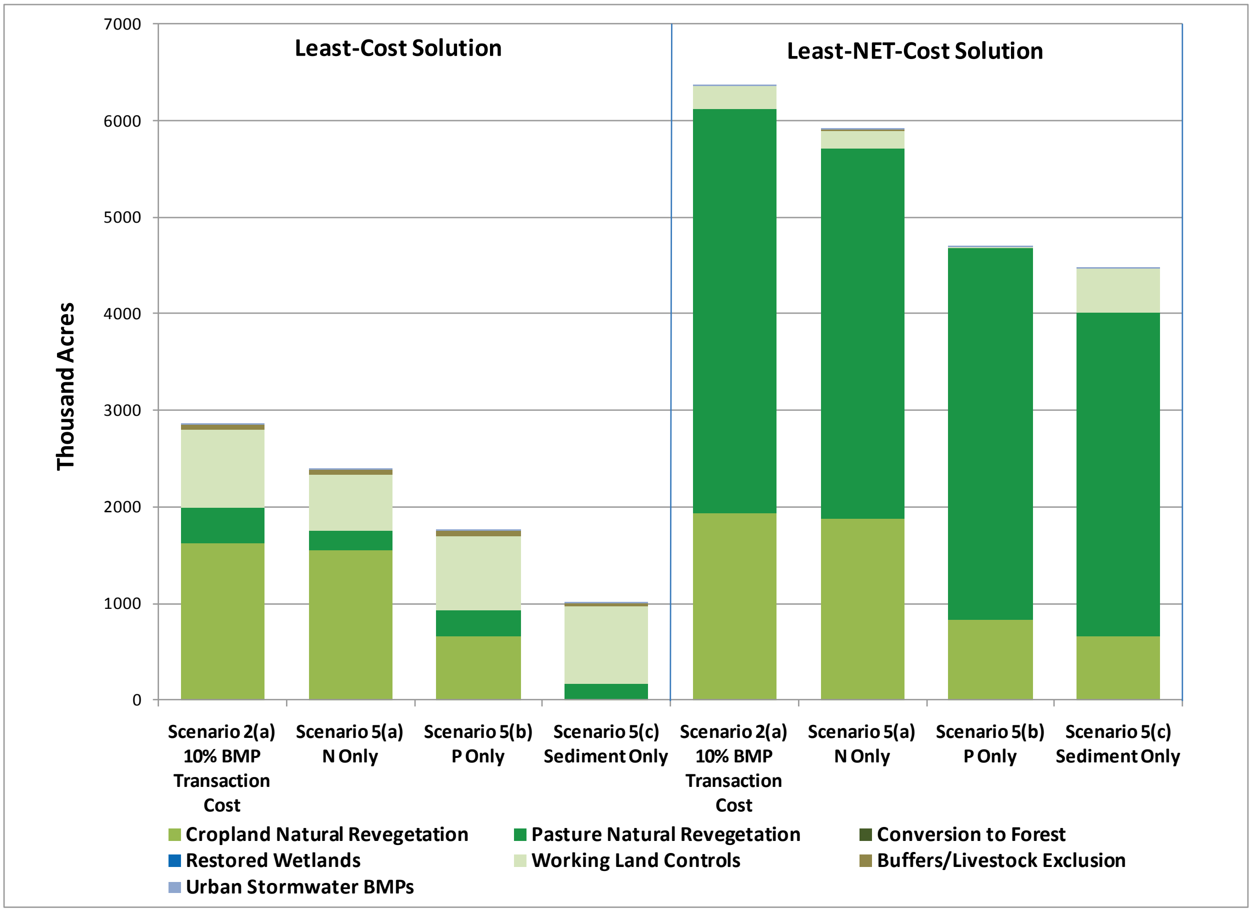

Figure 5-11. Additional BMP acres under alternative adjustments to sediment load allocations and reduction targets.

Figures 5-12 and 5-13 investigate the sensitivity of the model results to alternative carbon prices. As expected, changing the carbon price does not change the costs or the selected control projects in the least-cost solution because the monetized bonus ecosystem services do not affect the optimization. Only the value of the monetized ecosystem services changes. In contrast, for the least-NET-cost solution, a lower carbon price in Scenario 6(a) results in a $52 million decline in costs relative to Scenario 2(a) and a $136 million decline in bonus ecosystems services, such that NET costs increase by 133%. With a higher carbon price in Scenario 6(b), costs increase by $138 million relative to Scenario 2(a), but bonus ecosystem services increase by even more ($428 million). The high carbon price leads to a model solution with bonus ecosystem services that exceed costs by $227 million. The effect of the high carbon price on BMP acres is shown in Figure 5-13. It leads to a solution that would rely almost entirely on large amounts of natural revegetation (over 8.5 million acres).

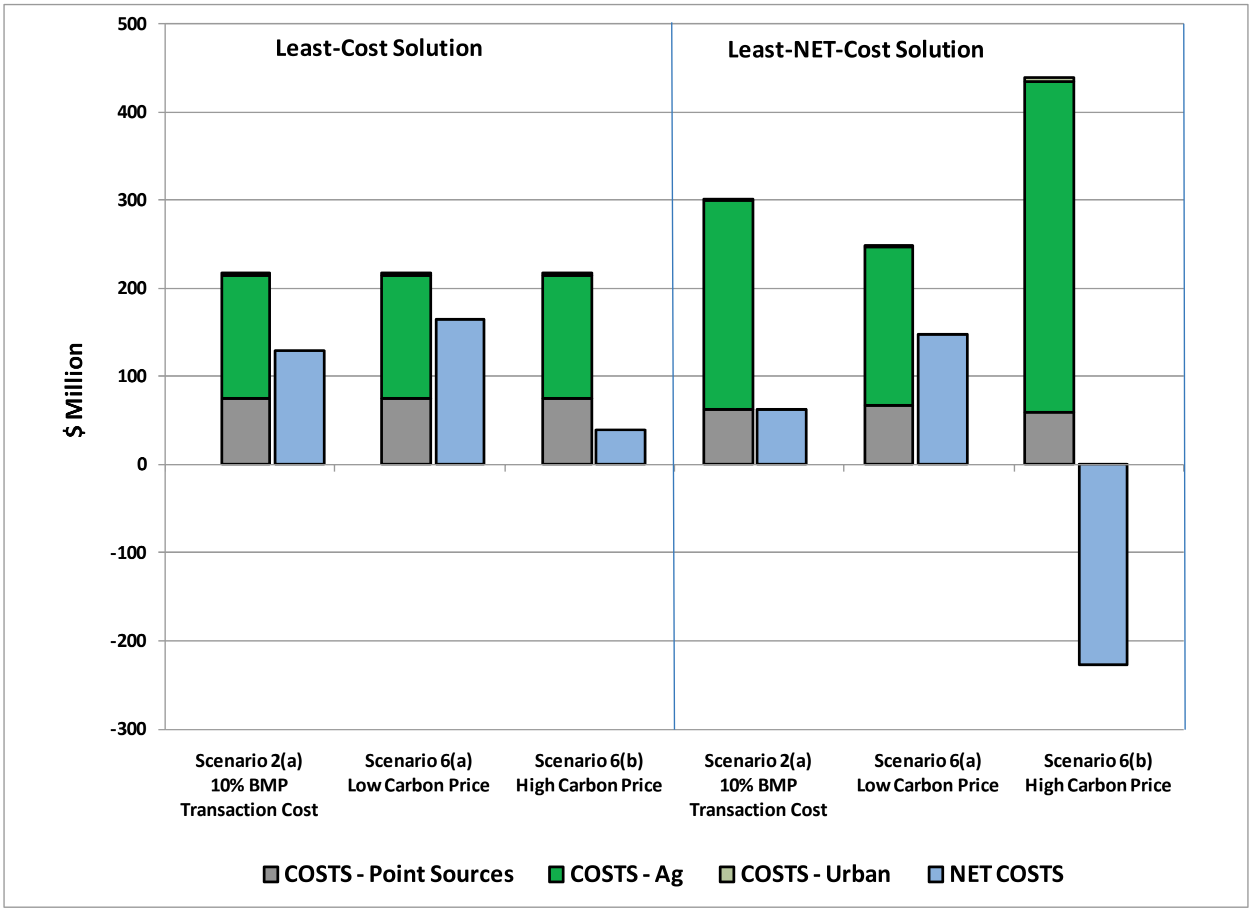

Figure 5-12. Annual costs and NET costs under alternative carbon prices.

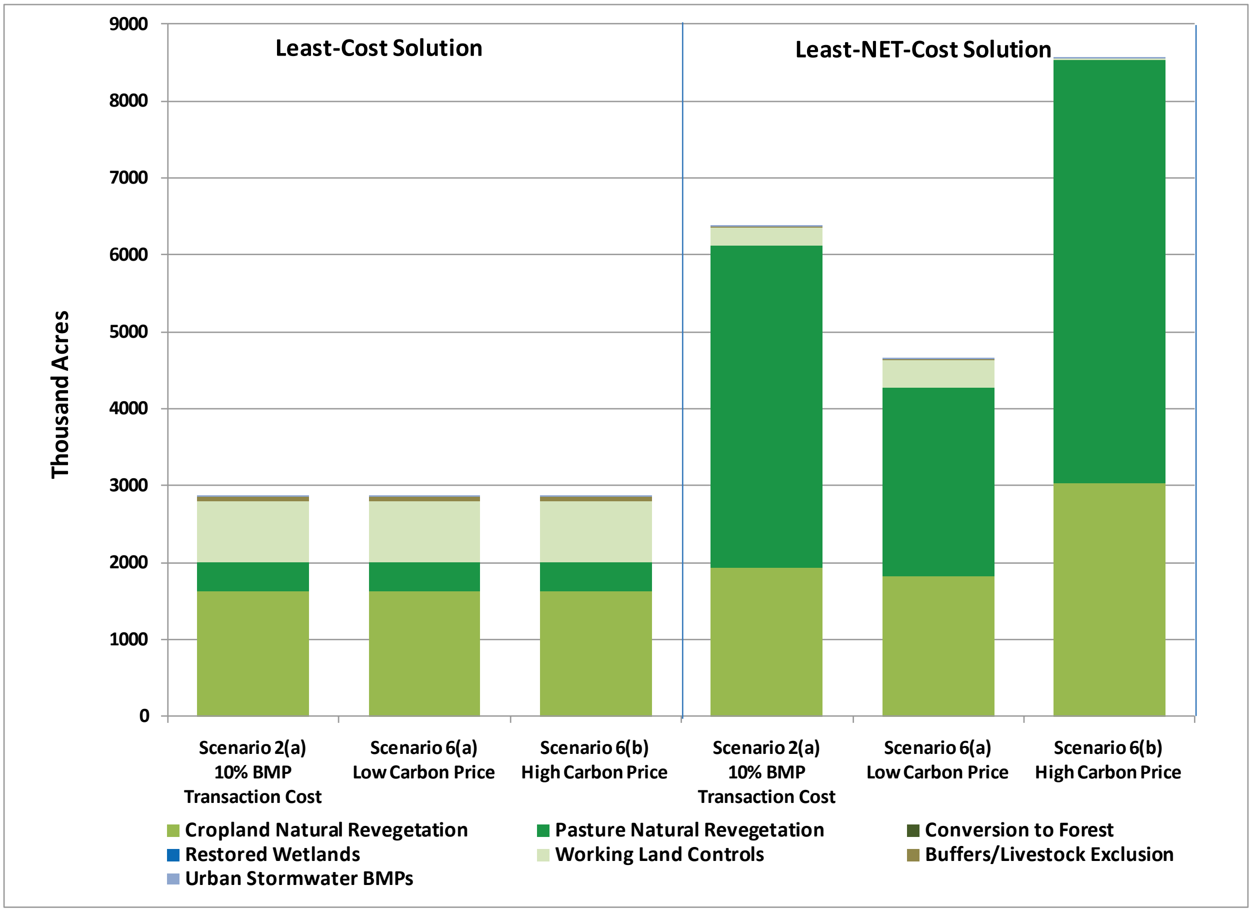

Figure 5-13. Additional BMP acres under alternative carbon prices.

5.3 MODEL RESULTS FOR ALTERNATIVE LOAD-REDUCTION APPROACHES AND RESTRICTIONS

Figure 5-14 compares total control costs and monetized ecosystem services estimates for alternative policy configurations for achieving the load-reduction targets. Base Case Scenario 2(a) is again included for reference. Scenario 7(a) restricts the tradeoffs between point- and nonpoint-source controls by imposing Tier 4 technology requirements on WWTPs. Scenario 7(b) includes these same WWTP technology requirements, plus a 2:1 credit ratio for point to nonpoint sources.

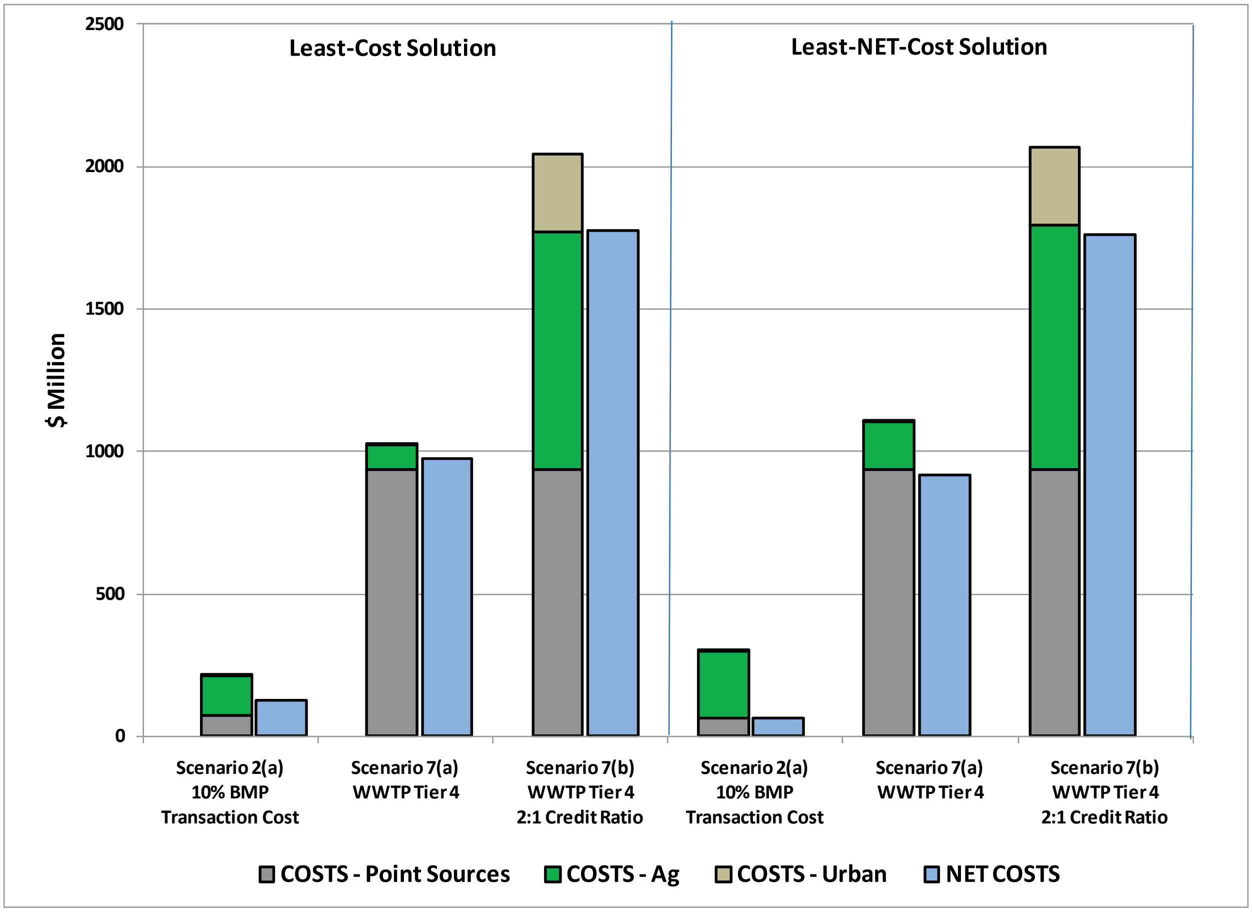

Figure 5-14. Annual costs and NET costs under alternative requirements.

As expected, Scenarios 7(a) results in very large total control costs (over $1 billion in the least-cost solution) compared to the Base Case, and a relatively large contribution to these costs is from point sources (91%). Despite the high cost, these point-source controls only achieve 49% of the nitrogen reduction and 55% of the phosphorus reductions required. None of the point-source controls reduce sediment. The agricultural and urban stormwater BMPs required to meet these targets account for 8% and less than 1%, respectively, of the total control costs in this scenario. Under the least-NET-cost solution, the costs increase by another $82 million, as additional nonpoint-source controls (in particular, natural revegetation) are included. The annual ecosystem services increase to $191 million; however, they are still dwarfed by the total costs.

When the 2:1 credit ratio is included in Scenario 7(b), the costs double to over $2 billion in both the least-cost and least-NET-cost solutions. As in Scenario 3(a), which does not include the WWTP technology restriction, the James River basin cannot meet its sediment reduction target with the menu of control projects included in the model, and the Susquehanna cannot meet its nitrogen-reduction target. As shown in **Figure 5-15**, another similarity with Scenario 3(a) is that both the least-cost and least NET-cost solutions include large amounts conversion to forest (over 3.9 million acres).

Although they are not shown in Figure 5-14, requiring Tier 4 is expected to also result in increased air pollutant emissions. When monetized, the impacts of these emissions can be interpreted as reductions in ecosystem services or as increase in social costs. The total social costs of GHG and air pollutant emissions associated with upgrading significant municipal and industrial WWTPs to meet Tier 4 effluent discharge concentrations are $6.3 million and $11.4 million, respectively. The additional GHG emissions are almost 140,000 metric tons of carbon, which is equivalent to emissions from over 56,000 households' annual electricity use or the emissions of almost 89,000 cars per year (U.S., EPA 2010c).

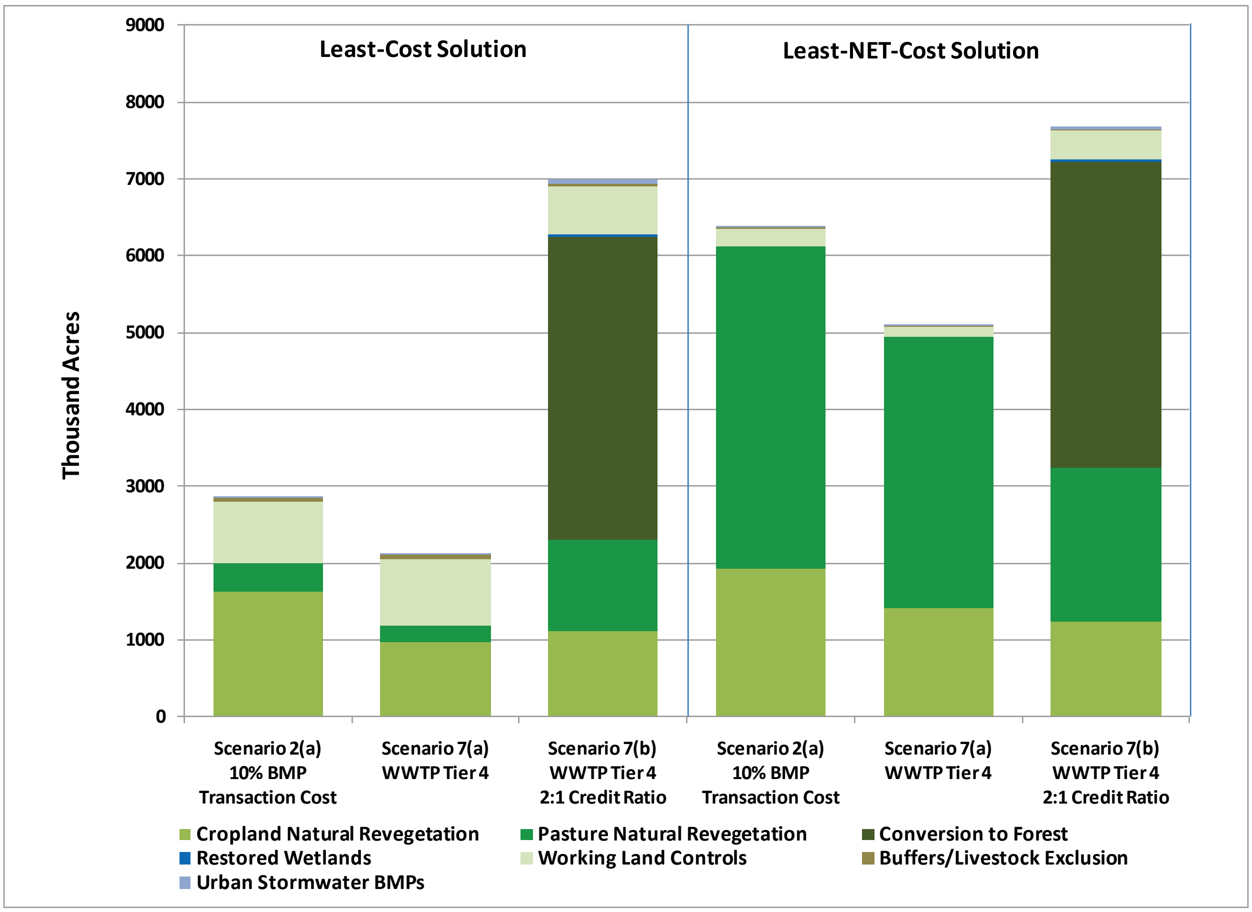

Figure 5-15. Additional BMP acres under alternative requirements.

Figures 5-16 and 5-17 investigate the sensitivity of the model results to different restrictions on agricultural land conversion. The purpose of these model runs is to see how the mix of selected control projects is affected by limiting the amount of natural revegetation, conversion to forestry, and wetland restoration that is available (and to see whether the load-reduction targets are attainable under these restricted conditions).

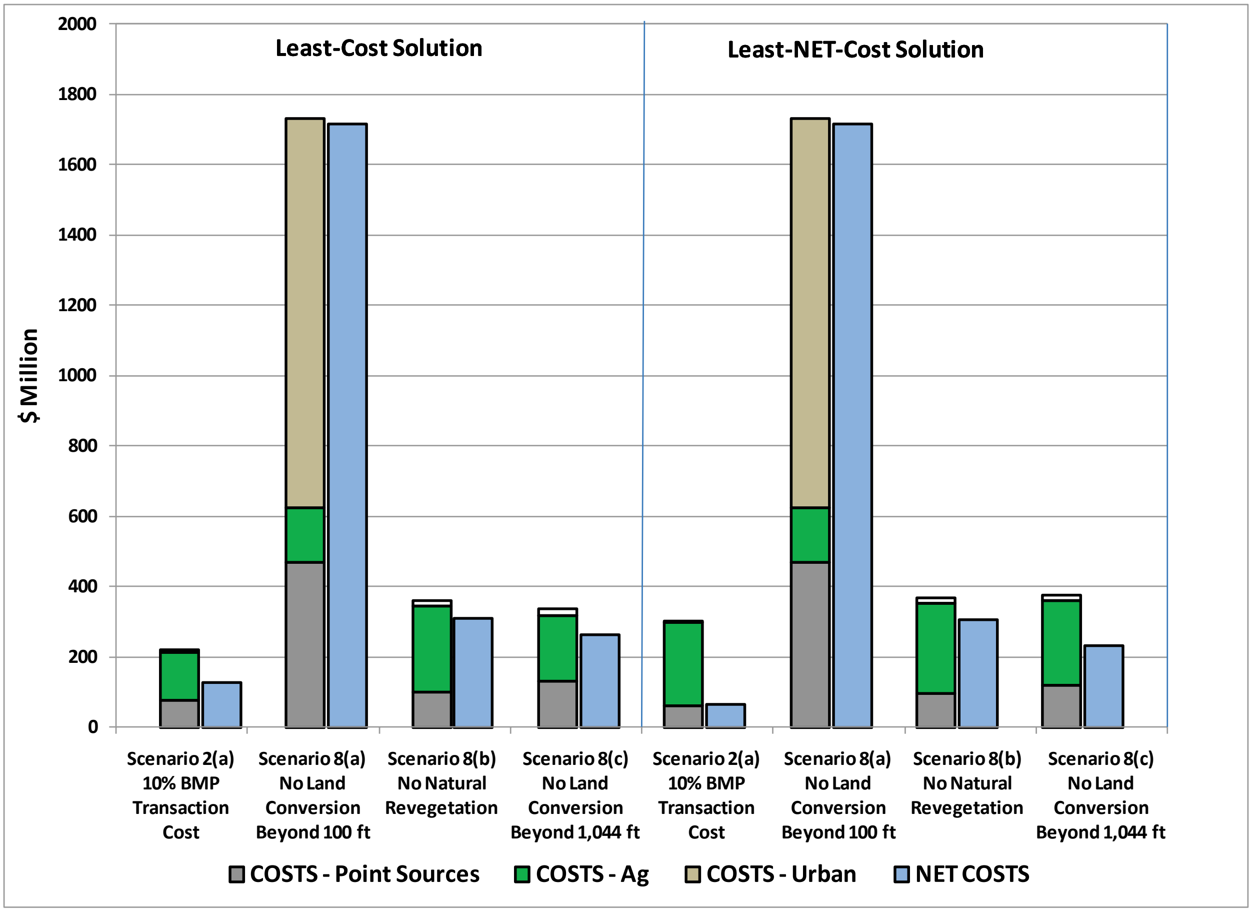

Figure 5-16. Annual costs and NET costs under alternative land conversion restrictions.

In Scenario 8(a), no agricultural land conversion is allowed beyond 100 feet from the stream. Under both the least-cost and least-NET-cost solutions, the estimated annual costs increase to over $1.7 billion, and the urban stormwater controls now account for 64% of these costs (primarily associated with bioretention planters). The annual costs of agricultural BMPs—primarily from cover crops and reduced fertilizer application—are $154 million; however, this amounts to only 9% of the total costs.[27] In both cases, the bonus ecosystem services are less than $17 million. As shown in Figure 5-1, even though agricultural BMPs account for a small fraction of the total costs compared to urban stormwater BMPs, they still account for a large majority of the BMP acreage.

In Scenario 8(b), only natural revegetation is excluded as a BMP option. Due to the large role played by natural revegetation in the Base Case Scenario 2(a), this restriction increases costs for the least-cost solution by 66%, to $362 million per year. It also decreases bonus ecosystem services by 42% to $52 million per year. As shown in **Figure 5-17**, rather than 2 million acres of natural revegetation, the model selects almost 900,000 acres for conversion to forest.

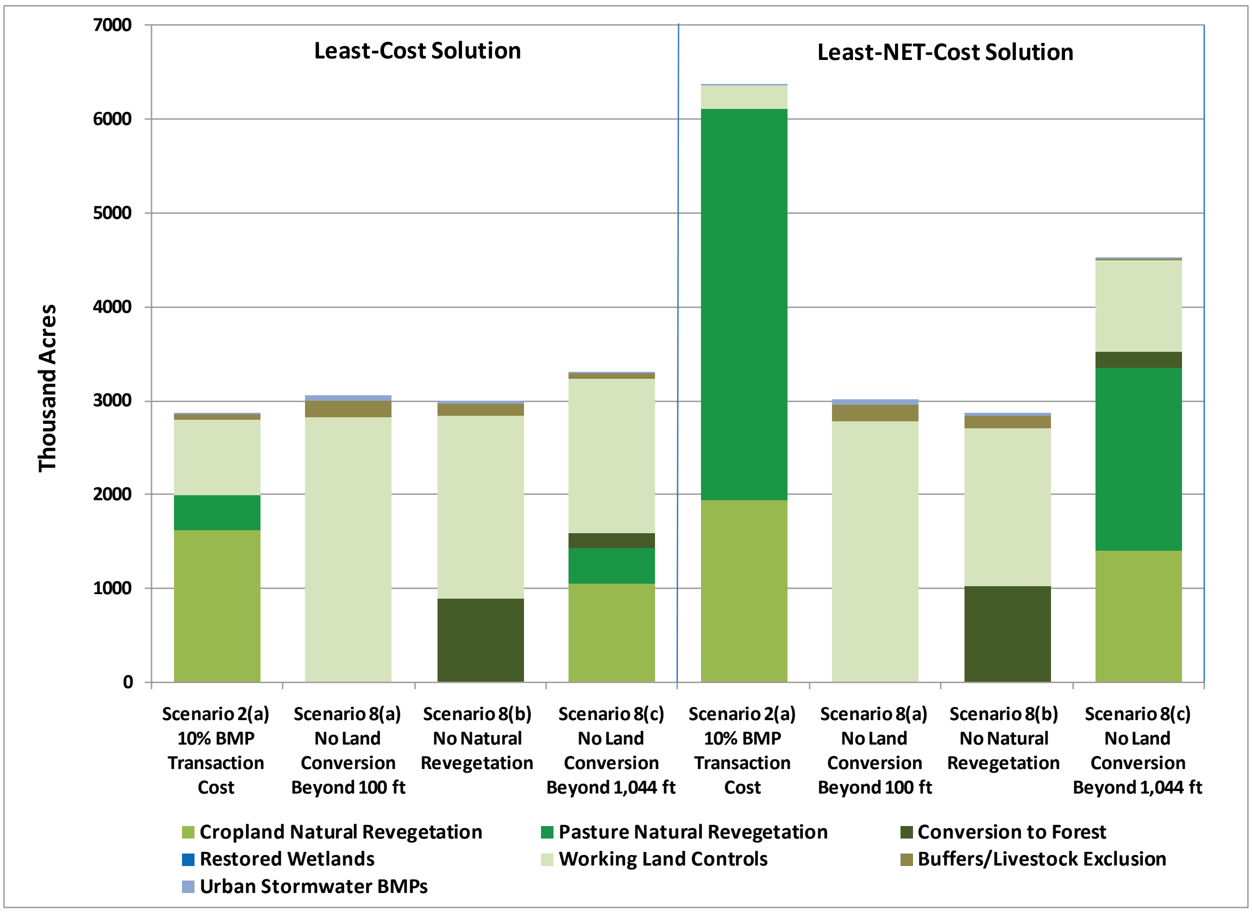

Figure 5-17. Additional BMP acres under alternative land conversion restrictions.

[27] Under the Scenario 8(a) restrictions, three basin-level load reduction targets are not attainable with the inventory of control projects included in the model—the sediment target in the James basin, the phosphorus target in the Potomac, and the nitrogen target in the Susquehanna (see **Appendix D** for details). Therefore, even these high total costs estimates are capped by the penalty prices included in the model.

In Scenario 8(c), no agricultural land is allowed beyond 1,044 feet from the stream. Compared to the Base Case Scenario 2(a) least-cost solution, this restriction increases total costs by 54% and reduces bonus ecosystem services by 19%. As shown in Figure 5-17, it reduces natural revegetation by 28% to 1.4 million acres; however, it also brings in 150,000 acres of conversion to forest. In addition, it increases the amount of land using "working land" controls (in particular cover crops and reduced fertilizer application) by roughly 1 million acres.

Figures 5-18 and 5-19 illustrate the sensitivity of the model results to two additional restrictions on agricultural land conversion. In Scenarios 9(a) and 9(b), a *minimum* of 30,000 and 60,000 acres, respectively, of wetland conversion are required. As expected, the total costs of the least-cost solution increase; however, the increment is relatively small. Relative to the Base Case Scenario 2(a), costs increase by 3% with a 30,000 acre wetland conversion requirement and by 6% with a 60,000 acre requirement. In contrast, the total additional acres of BMPs actually *decrease*, since wetland acres are relatively efficient at pollutant removal. Under Scenario 9(a) they decrease by roughly 24,000 acres (less than 1%), and under Scenario 9(b), they decrease by almost 33,000 (1.1%). Total bonus ecosystem services also decline slightly (by less than 1%) in both scenarios. A very similar pattern occurs under the least-NET-cost solution, with relatively small increases in costs and NET costs and relatively small declines in bonus ecosystem services.

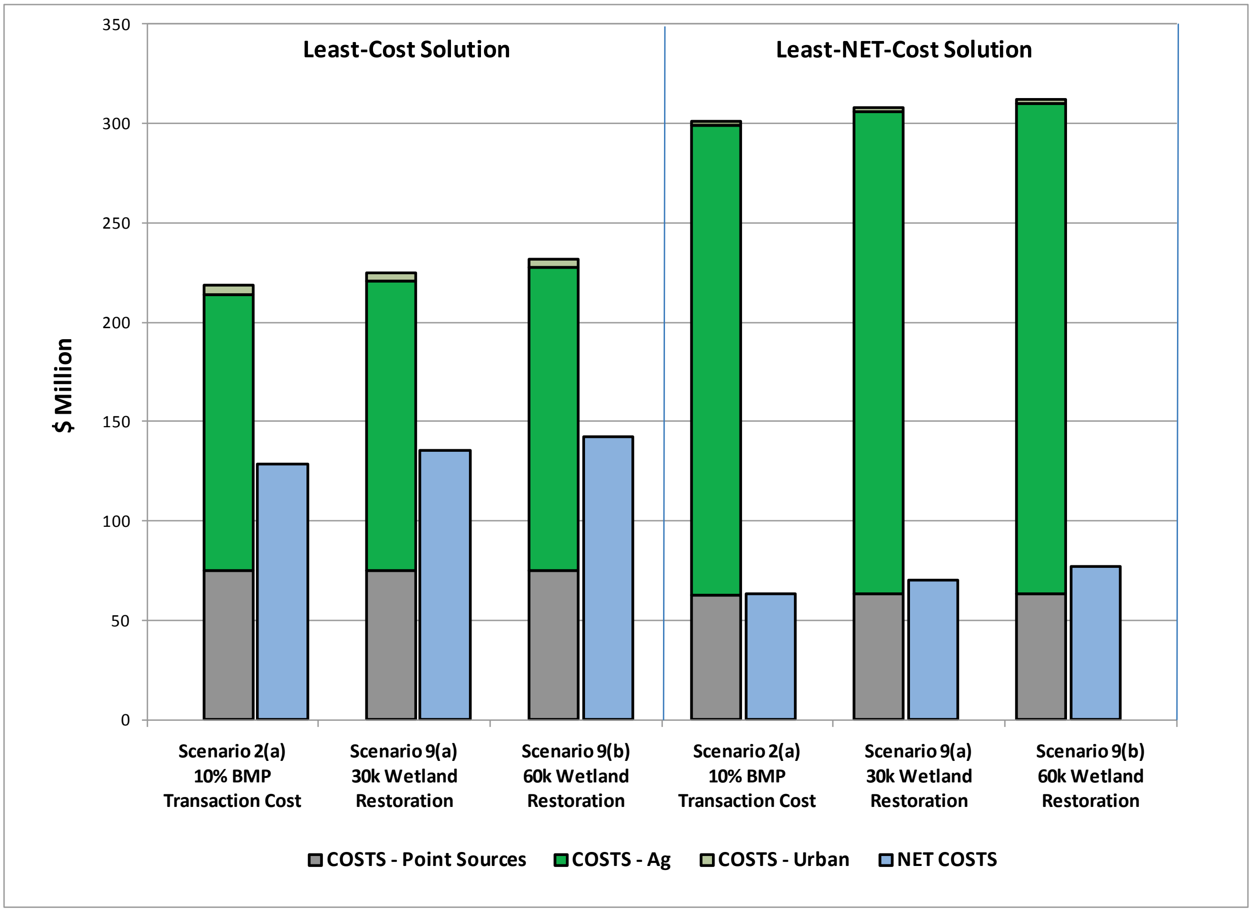

Figure 5-18. Annual costs and NET costs with minimum wetland conversion requirements.

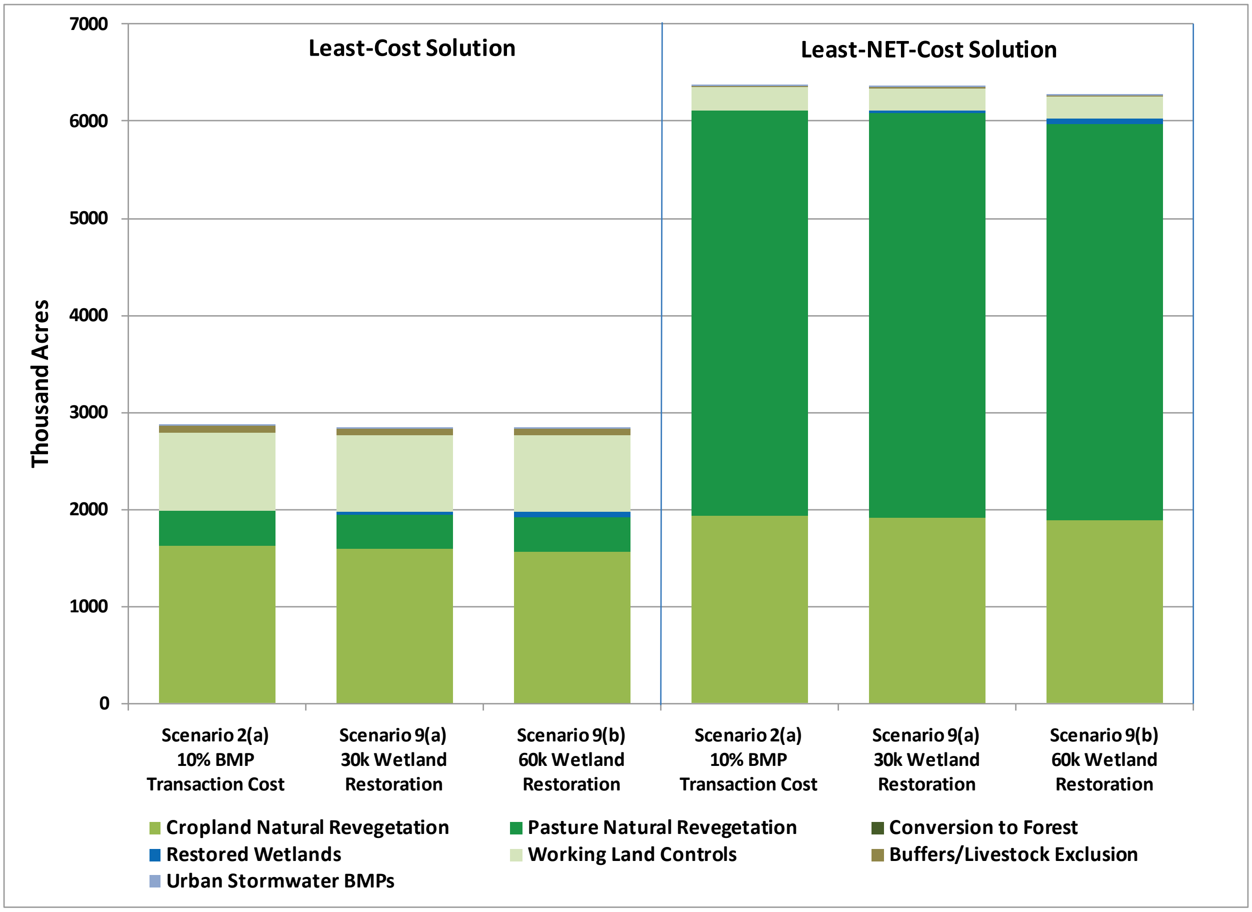

Figure 5-19. Additional BMP acres with minimum wetland conversion requirements.

Finally, **Figures 5-20 and 5-21** compare Scenarios 5(a) and Scenario 10 and examine how costs, bonus ecosystem services, and land conversion are affected by relaxing the basin-level nitrogen reduction targets, while keeping the overall Bay-wide nitrogen target. In Scenario 10, the individual basin-level nitrogen load reduction targets in Scenario 5(a) are replaced with basin-level loading impact factors and Bay-wide load reduction targets (equal to the sum for the basin-level targets in Scenario 1). As expected, the additional flexibility results in significantly lower total control costs, which are estimated to be 18% lower than in Scenario 5(a) under the least-cost solution. The fraction of these costs attributable to agricultural BMPs increases to 89%, while the point-source contribution is roughly 10%. Monetized bonus ecosystem services increase by 11% relative to Scenario 5(a), but the NET costs are still positive ($75.2 million).

Under the least-NET-cost solution, the additional flexibility offered by Scenario 10 results in a much lower role for point-source controls. Point sources account for only 6% of the nitrogen reductions and only 2% of the total costs. Over 95% of the costs and bonus ecosystem service benefits are associated with natural revegetation, and the total NET costs are only $6 million per year.

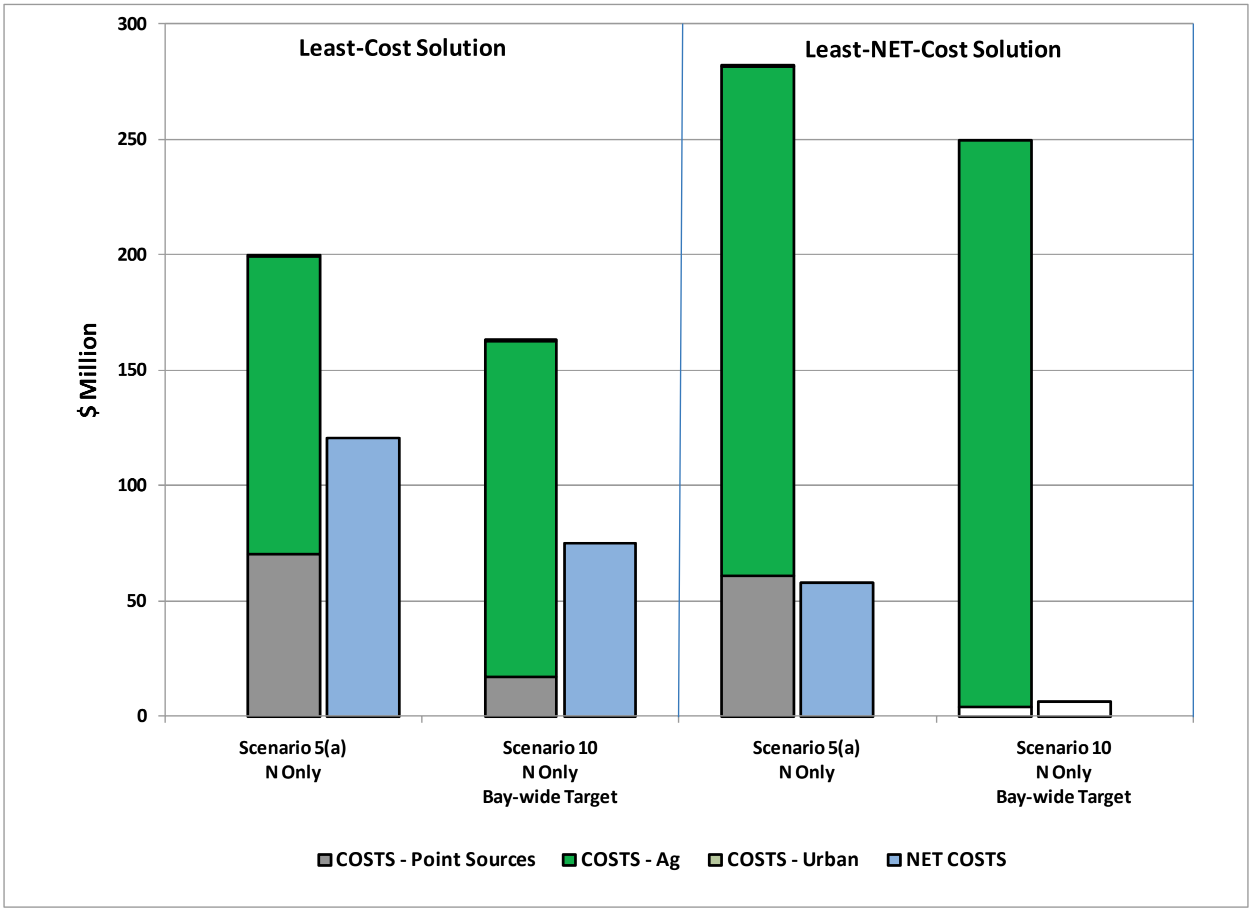

Figure 5-20. Annual costs and NET costs under basin-specific and basin-wide nitrogen reduction targets.

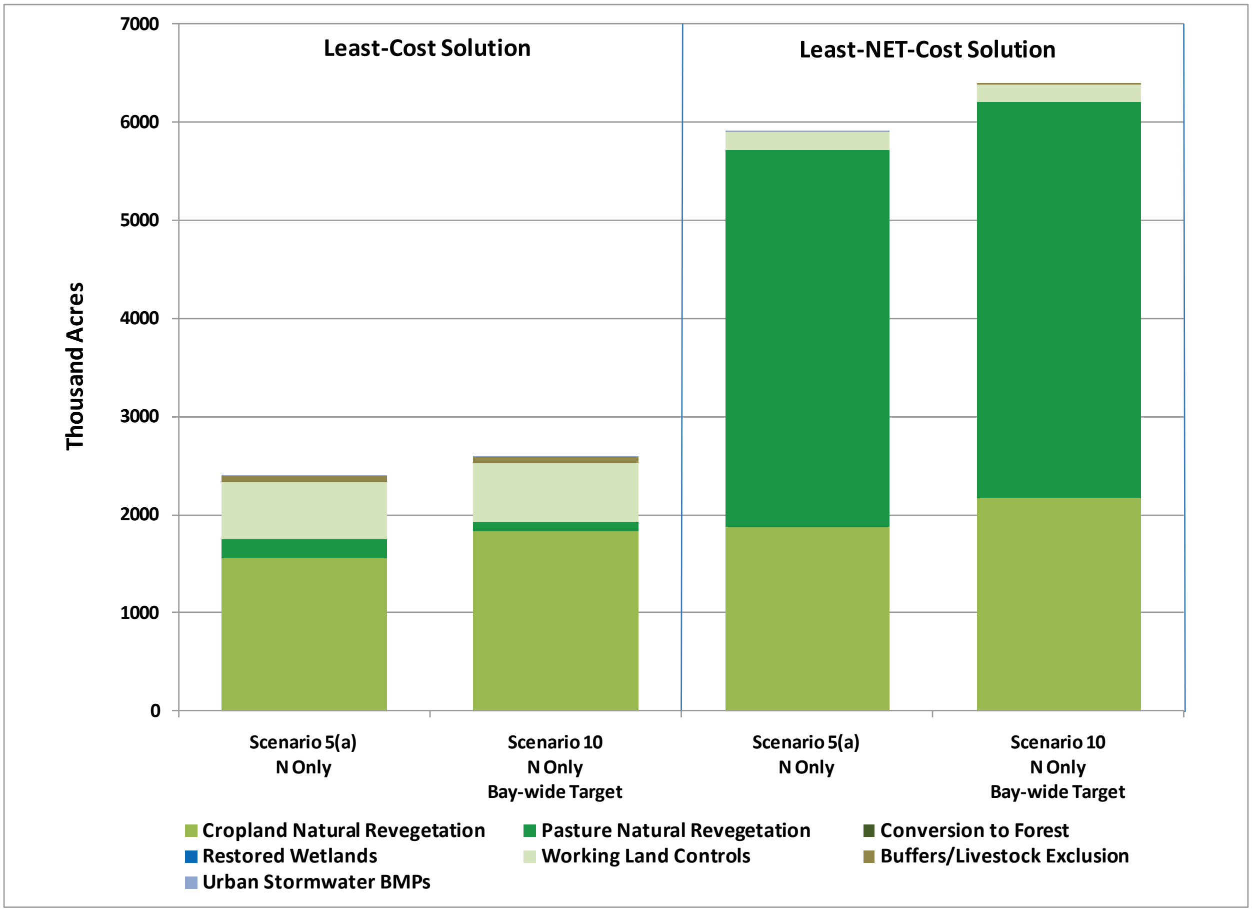

Figure 5-21. Additional BMP acres with under basin-specific and basin-wide nitrogen reduction targets.

SECTION 6. CONCLUSIONS

This report demonstrates an optimization modeling approach for estimating the least-cost combination of nutrient- and sediment-control methods in the Chesapeake Bay watershed and the resulting bonus ecosystem services delivered. The model includes a wide variety of point- and nonpoint-source control projects across the watershed, and it investigates the magnitude and potential role of ecosystem services in determining an optimal mix of control projects.

Applying the model to a wide range of scenarios, some of the main findings from this ongoing work are the following:

- Given the inventory of point- and nonpoint-source controls included in the model, and their cost and removal effectiveness estimates, green infrastructure projects (agricultural BMPs, in particular) account for approximately 2/3 of the project costs in most of the least-cost solutions for achieving the load-reduction targets.
  - For example, the estimated aggregate annual control costs in the Base Case scenario (Scenario 2[a]) are $218 million: 64% of these costs is attributable to agricultural BMPs, and 36% is attributable to point-source controls (a very small percent is attributable to urban stormwater BMPs in this scenario).
- Green infrastructure contributes substantial offsetting ecosystem service value to the cost of achieving the TMDL targets; gray infrastructure contributes ecosystem service *disbenefits*.
  - For example, the offsetting value of the bonus ecosystem services in the Base Case is $90 million, mainly from carbon sequestered through natural revegetation. It also results in 20,000 additional acres of urban wetlands and restores brook trout habitat in 95 subwatersheds.
  - In the scenario that would require maximum (Tier 4) technology upgrades for WWTPs, the total social costs of GHG and air pollutant emissions are $17.7 million. The additional GHGs are equivalent to emissions from over 56,000 households, or the emissions of almost 89,000 cars per year.
- Including monetized ecosystem services as cost offsets in the optimization model—i.e. estimating a least-NET-cost solution—shifts the solution towards the inclusion of more nonpoint-source controls; in particular, natural revegetation of cropland and pastureland.

The size of this shift is particularly sensitive to the assumed per-ton value of carbon sequestration.

- In the Base Case scenario (Scenario 2[a]), monetized bonus ecosystem services more than double to $238 million per year (and acreage in natural revegetation increases more than three-fold) when the value of these services is included in the optimization routine. When the assumed price of carbon is doubled to $92 per ton of carbon in Scenario 6(b), the bonus ecosystem services increase to $666 million per year.

- Most model solutions involve a substantial portion of agricultural acreage being taken out of production. This strategy results in substantially lower costs and greater bonus ecosystem services than a strategy that emphasizes traditional gray infrastructure; however, taking agricultural land out of production may not be a feasible option for other reasons.
  - For example, the Base Case Scenario 2(a) costs roughly 20% as much as the scenario that requires maximum (Tier 4) upgrades of WWTPs, but it takes 2 million acres of agricultural land out of production (that is about 22% of all agricultural land in the Chesapeake Bay watershed). In contrast, the scenario that assumes higher opportunity costs for agricultural land (Scenario 2[c]) takes only 1.3 million acres out of production, but costs increase to $288 million.
- As expected, the total costs of control increase and bonus ecosystem services decrease significantly when (1) transaction and land rental costs are increased for nonpoint-source BMPs, (2) the pollutant removal effectiveness of BMPs is reduced, (3) the availability of agricultural BMP projects is restricted, and (4) the technological requirements on WWTPs are made more stringent. The highest aggregate control costs (over $2 billion per year) were estimated for the scenario that combined lower BMP pollution removal effectiveness and WWTP technology requirements (i.e., Scenario 7[b]).
- Of the three pollutants, the nitrogen load reduction targets tend to have the largest effect on the model solution, whereas the sediment load reduction targets are the least influential.
  - For example, when only nitrogen load reduction targets are included in the optimization (Scenario 5[a]), the estimated total costs decrease by less than 10% compared to the Base Case Scenario 2(a). In contrast, they decline by 65% and 91%,

respectively, when only phosphorus load and sediment load reduction targets are included.

  - Even so, failure to account for sediment load reduction targets increases the reliance on point-source controls and reduces bonus ecosystem services. In Scenario 4(c), which excludes the sediment load reduction target, overall costs decrease by $1.3 million compared to the Base Case Scenario 2(a), but point-source costs increase by $0.8 million and bonus ecosystem services decrease by over $3 million.

- Uncertainty about the effectiveness of agricultural or stormwater BMPs may substantially increase the costs of achieving the load reduction targets if states require 2:1 credit ratios for point- to nonpoint-source trades. While Scenario 3(a) does not specifically model such trades, the large increase in control costs resulting from a 2:1 credit ratio (to over $1.4 billion, as shown in Figure ES-4) is indicative of the expected result of such policies. Although credit ratios are currently being used in a precautionary fashion to promote beneficial environmental outcomes, they may also add significant costs. Consequently, an improved understanding of performance risk could help to substantially reduce the costs of achieving TMDLs through nonpoint-source controls.
- Assigning TMDL targets to the Bay as a whole, as opposed to major tributary basins, had a major effect on control costs. When the least-cost projects were drawn from throughout the Bay watershed, costs were substantially lower than when TMDL targets were required to be met at the level of individual basins. This hypothetical example demonstrated the general concept that if larger areas are used to generate nutrient or sediment offsets costs will be lower when the least-cost options are not distributed evenly throughout the basin. However, this approach could have implications for local water quality.

6.1 POLICY-RELEVANT OBSERVATIONS

Developing a modeling framework that would assist policymakers in evaluating TMDL-related tradeoffs poses a number of questions. This framework was specifically designed to incorporate measures of both the cost effectiveness and ecosystem service impacts associated with individual pollution-control projects. Implementation of a program that considers both cost-effectiveness and ecosystem service impacts compounds the challenges. The policy-relevant questions and observations made include the following:

- What institutional arrangements are needed to coordinate and promote restoration goals such as TMDLs and ensure performance of nutrient- and sediment-management practices and ecosystem-service restoration to achieve pollutant allocations limits at the lowest total cost? Trading and offset programs are potentially important methods for achieving cost-effective TMDLs. These programs offer the potential to restore multiple environmental endpoints simultaneously. Although several barriers to trading nutrient or ecosystem service credits may exist, the potential cost savings from these approaches suggest that they are worth serious consideration.
- How does the absence of a CWA regulatory structure for certain nonpoint sources impact implementation, and what are the cost and bonus ecosystem service implications of relying more heavily on certain types of agricultural and urban stormwater controls (rather than point-source controls) to meet the load allocations? This is one of the most challenging obstacles, i.e., no regulation of nonpoint-source dischargers, such as farmers and foresters, who would be the suppliers of low-cost nutrient reductions. However, states do not have this limitation and can impose loading caps on nonpoint-source sectors to facilitate trading programs. Further, the CWA has sufficient flexibility to allow multiple innovations and program types. Where state laws have created restrictions on dischargers in ways that support development of trading (e.g., North Carolina and Virginia), various nutrient trading programs have been implemented and have reduced costs of compliance with caps. Our analysis suggests that significant cost savings and bonus ecosystem services can be gained from strategies that take advantage of cost-effective green BMPs. Nutrient trading programs offer one potentially promising approach for realizing these gains.
- **What are the broader market implications of nutrient control methods that rely on conversion of agricultural land?** Changes in agricultural production due to large-scale land conversion are likely to have ripple effects on incomes, jobs, and economic activity, both inside and outside the watershed. Moreover, as land is taken out of production, the remaining land may increase in value, requiring larger payments to landowners for them to change practices as more nonpoint-source options are adopted. These macroeconomic and indirect economic impacts are not included in the current model; however, they must also be considered in a broader evaluation of the strategies investigated in this report.

- **What if no political or basin boundaries were considered?** That is, if trading were allowed throughout the Chesapeake Bay watershed, what operational and statutory hurdles would exist? Limitations on the geographic scope of trading or offsets limit the ability to achieve cost savings by allowing credit buyers to purchase the lowest-cost nutrient credits. Therefore, the fewer the restrictions that are placed on the trading, the more cost efficient it is to meet loading reduction targets. However, the environmental impact of large geographical trading boundaries must be evaluated to prevent "hot spots" or areas of high or increased pollution.
- What would be the outcome if the watershed were modeled based on state boundaries rather than river basin boundaries? What are the operational and statutory barriers to achieving basin-wide allocations where river basins cross state lines? This modeling analysis focused on achieving river basin–specific pollutant-reduction targets. Shifting to state boundaries limits the availability of nonpoint sources for intrastate trading and also limits the amount of ecosystem services that could be provided. Otherwise, states would need to establish interstate trading programs to increase opportunities for nonpoint-source pollution controls that provide expanded ecosystem services.
- **Do existing regulatory programs hinder implementation?** Some types of regulation may be seen to interfere with development of offset and trading programs. For example, requiring permit holders to adopt particular technologies or direct payments for implementing particular practices would reduce the ability of nutrient credit markets to form and potentially reduce the potential for innovation that can lead to cost efficiencies.
- **How should a trading program account for groundwater?** The role of groundwater as a source of nutrients to the Bay was not addressed in this analysis due to time and resources; however, it merits analysis. Groundwater policy for nutrients would require considerable study from the perspective of source control and temporal implications given the lag time in delivery.
- **Is phased implementation of trading system to achieve TMDL targets feasible programmatically?** Phased implementation is a valuable approach to dealing with uncertainty of stressors and responses. Time is needed to engage communities, to capture ideas and innovative approaches, and to establish the required performance metrics and monitoring. Similarly, adaptive implementation allows TMDLs to be modified in response

to new knowledge and provides flexibility in order to incorporate innovation in approaches.

The barriers to implementing new approaches to restoration are complex and include finding sources of funding to implement ideas, managing uncertainty, finding the time to engage communities, disseminating information about what works, and adapting policies to reflect what works. Solutions will require the recognition of temporal and spatial scales, community resilience and capital, and the importance of sharing new ideas and information.

6.3 FUTURE DIRECTIONS

As demonstrated by the results from the varied scenarios, the model described in this report provides a rich framework for analyzing economic implications and tradeoffs associated with alternative nutrient- and sediment-control strategies in the Chesapeake Bay watershed. However, it is also a work in progress. A number of potential model extensions deserve consideration to further strengthen and broaden this framework and to address existing limitations. First, including additional pollutant-control methods and technologies in the model is a priority. They include, for example, strategies for (1) controlling sediment runoff from construction sites in the watershed, and (2) reducing runoff from CAFOs. Further investigation into the role of atmospheric deposition could also be conducted.

Second, the model can be modified to include regional- or state-level load-reduction targets within the basin-level framework currently being considered, or the model could account for the hydrologic role of each tributary in pollutant retention in the Bay and, in turn, eutrophication or other impacts in applying restrictions on pollutant reductions.

Third, options for including additional ecosystem service effects (including the GHG emission changes associated with point-source controls) in the model need to be further investigated and implemented as appropriate.

Fourth, the framework could be expanded to also examine the broader economic and ecological impacts of the modeled changes. In particular, how might large-scale changes in the amount of land devoted to agricultural production affect employment, incomes, and other ecosystem services, both inside and outside the watershed?

Fifth, options for transferring and applying the framework to other watersheds will be considered. The policy importance of the Chesapeake Bay and the availability of data and models for the watershed have made it an ideal context for investigating and testing the framework. Many of the techniques and lessons learned in this model development process should be transferable for analyzing similar issues in other watersheds.

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