Tradeoffs among Ecosystem Services, Performance Certainty, and Cost-efficiency in Implementation of the Chesapeake Bay Total Maximum Daily Load

Lisa A. Wainger, George Van Houtven, Ross Loomis, Jay Messer, Robert Beach, and Marion Deerhake

The cost-effectiveness of total maximum daily load (TMDL) programs depends heavily on program design. We develop an optimization framework to evaluate design choices for the TMDL for the Potomac River, a Chesapeake Bay sub-basin. Scenario results suggest that policies inhibiting nutrient trading or offsets between point and nonpoint sources increase compliance costs markedly and reduce ecosystem service co-benefits relative to a least-cost solution. Key decision tradeoffs highlighted by the analysis include whether agricultural production should be exchanged for low-cost pollution abatement and other environmental benefits and whether lower compliance costs and higher co-benefits provide adequate compensation for lower certainty of water-quality outcomes.

Key Words: Chesapeake Bay, cost-effectiveness, ecosystem services, environmental policy, optimization model, pollution control, TMDL, water quality trading

Total maximum daily loads (TMDLs) are authorized by the Clean Water Act (33 U.S.C. § 1344) to limit pollutant inputs to local water bodies for which water quality is a concern. Historically, TMDLs most commonly have been applied in relatively small nontidal watersheds. Increasingly, in response to a variety of forces (some are explained in Copeland (2005)), many states are developing

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more comprehensive listings of impaired water bodies that include tidal waterbodies and large watersheds. Once such waterbodies are listed as impaired, environmental agencies develop TMDLs to bring them into compliance with their designated uses (e.g., swimmable, fishable, aquatic habitat). TMDLs for tidal waterbodies (e.g., Long Island Sound and Chesapeake Bay) often require multi-state initiatives and involve high compliance costs, suggesting a need for regional-scale design strategies to promote cost-effectiveness.

A TMDL is designed by (i) estimating the maximum amount of one or more pollutants that can enter a waterway without compromising its designated use, and (ii) allocating the allowable amount of pollution among sectors of the sources of that pollution (e.g., municipal, agricultural, and industrial). The designated use in Chesapeake Bay and its tributaries is "aquatic habitat," a designation that sets high standards for in-water conditions. Load allocations generally can be enforced only for permitted point source (PS) emitters, which include wastewater treatment plants (WWTPs), some stormwater (SW) systems, and, most recently, large confined animal feeding operations. However, states are increasingly looking to allocate loads (and, consequently, load *reductions*) to nonpoint-source (NPS) emitters, which include agricultural producers, property developers, and SW management entities, to more equitably distribute reductions across all emitters.

Because NPS emitters are not regulated at the federal level, state regulators must either create legislation to limit their emissions or seek creative approaches to incentivize them to generate low-cost reductions voluntarily. In Chesapeake Bay, regulators are promoting participation by NPS emitters in multiple ways, including substantial support for water-quality trading markets. That support includes providing regulatory flexibility that allows emitters to buy and sell nutrient and sediment credits, developing web-based tools to reduce market transaction costs, and creating institutions to manage legal risks.

However, the tools created to facilitate markets so far have not examined how specific program rules affect market development, such as their influence on the supply of available credits or the availability of trading partners. Such details can have dramatic effects on whether the markets will develop (Ghosh, Ribaudo, and Shortle 2011, Ribaudo and Gottlieb 2011, Shabman and Stephenson 2007, King and Kuch 2003) and whether the offsets will be cost-effective (Nickerson, Ribaudo, and Higgins 2010). In addition, market rules can affect the environmental neutrality of the trading and offset programs relative to alternative policies. To be environmentally neutral, the programs must not increase pollution emissions or cause adverse environmental outcomes in the process of reducing costs (e.g., additionality issues) (see Duke et al. *forthcoming*). Because TMDL design and implementation choices can alter the viability of water-quality markets and produce unintended environmental outcomes (from nonlinear responses and complex interactions in social and environmental systems), optimization analyses are valuable for examining potential implications of TMDL design choices on their overall cost-efficiency.

Two previous modeling efforts have used data from Chesapeake Bay watersheds to evaluate the effects of programmatic and environmental factors on the cost-saving potential of PS-NPS trading or equivalent programs. One study used a detailed optimization model to show substantial potential for cost savings from targeting effort to the most cost-efficient PS and NPS options compared to applying limit-of-technology requirements to emitters (Schwartz

2010). That study did not explicitly consider trading so it largely omitted the effects of transaction costs and trading ratios. Hanson and McConnell (2008) used a simpler but data-rich model to estimate potential cost savings from an "administered" trading system. The model optimized reductions in nitrogen using a dedicated fund available through the State of Maryland to pay for either WWTP upgrades or cover crops. The authors compared their results to the program's existing allocation formula that dedicated most of the fund to WWTP upgrades. They found that gains from PS-NPS trading depended largely on the level of watershed urbanization and estimated potential savings of 14 to 16 percent in urban basins and 31 to 61 percent in rural basins.

We build on those previous efforts to evaluate potential economic efficiencies of PS-NPS trading by developing a detailed optimization analysis that includes (i) updated and enhanced model functions and structure to reflect current TMDL program rules and newly available data, (ii) additional sources of cost and performance uncertainties associated with trading programs, and (iii) consideration of the effect of PS-NPS trading policies on the production of ecosystem service co-benefits.

Our approach is similar to one used in a recent economic analysis of nutrient credit trading for Chesapeake Bay (Van Houtven et al. 2012); our analysis differs primarily in its focus on elucidating tradeoffs associated with a broader range of TMDL policy options and environmental benefits, such as alternate restrictions on conversion of agricultural land to natural vegetation. Maintaining agricultural viability and competitiveness in the Chesapeake Bay watershed while cost-effectively meeting the TMDL is a significant consideration when designing a bay restoration strategy. We present scenarios that are not intended to be policy recommendations but rather are intended to enrich the discussion.

The primary goal of this study was to quantify the potential for alternative TMDL policies to achieve a suite of environmental goals in terms of achieving water-quality goals and ecosystem service co-benefits. Ecosystem services are outputs of natural systems from which humans derive benefits (Boyd and Banzhaf 2007). In our application, co-benefits are additional ecosystem services produced from pollution-control practices that are ancillary to the benefits derived from water quality improvements in the listed receiving waterbody. Understanding the production of co-benefits is relevant because the ability of alternative policies to deliver co-benefits can suggest how the TMDL can be leveraged to meet multiple bay restoration goals efficiently and may be important in promoting voluntary compliance by some emitters. Prior work has revealed widespread spatial mismatches between the emitters that can reduce emissions at the lowest cost and those that have the most to gain from an improved estuary (Environmental Protection Agency (EPA) 2011, Figure ES-5) if one assumes that proximity to the bay implies greater potential for accruing benefit. Thus, an analysis of ecosystem service co-benefits from pollution-control practices provides insights into how local priorities might be served through alternative TMDL policies.

We first describe the case study, the optimization framework, and the scenarios tested. We then discuss the results and implications of three main types of policy decisions in the TMDL design: (i) which best management practices (BMPs) are allowed or subsidized; (ii) the level of risk-aversion applied when managing BMP performance uncertainty; and (iii) the effects of using ecosystem service co-benefits to offset TMDL costs. Incentives for these co-benefits could be provided by, for example, "stacking" policies that allow BMP developers to receive separate payments or credits for each type of ecosystem service generated by a specific practice.

Case Study

The analytical framework we used was originally developed for and applied to the entire 64,000-square-mile Chesapeake Bay watershed shown in Figure 1. The methods and results of that work are reported in a detailed technical report (EPA 2011). For this application, we focused on the Potomac River basin and introduced policy scenarios that were not examined in the prior research. The Potomac River basin is the second largest sub-basin within the Chesapeake Bay watershed and includes portions of four states (Maryland, Pennsylvania, Virginia, and West Virginia) plus the District of Columbia. Chesapeake Bay is a particularly useful test case because of the existence of a rich data set with which we can inform extensive analyses of spatially explicit nutrient and sediment sources and transport dynamics plus facility-by-facility estimates of the costs associated with upgrading WWTPs to reduce nutrient loading. In addition, Executive Order 13508 (74 CFR 23099, 2009) provides the impetus

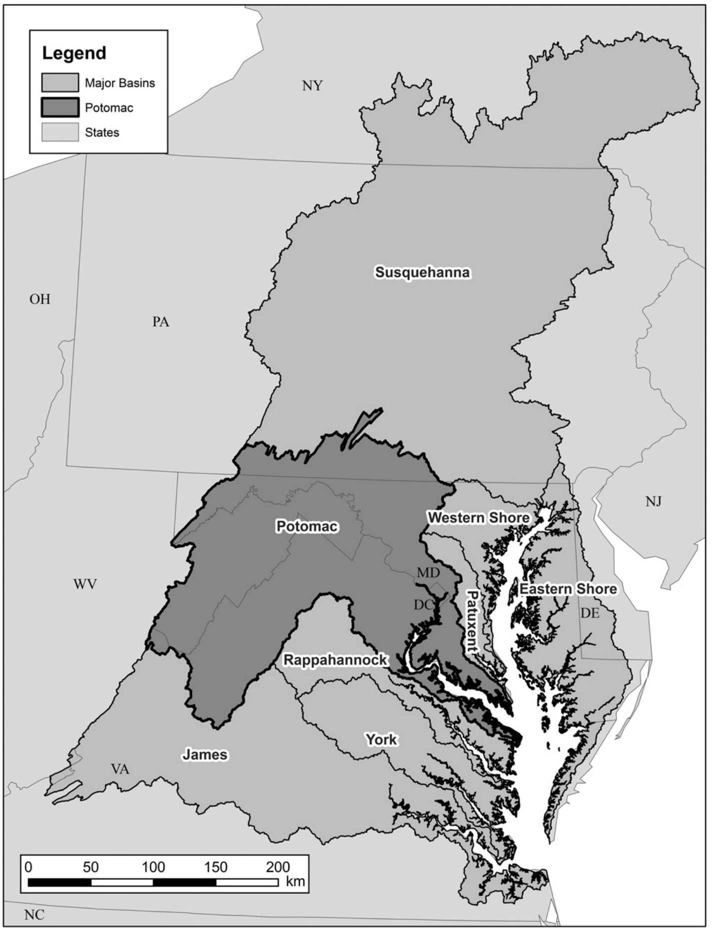

Figure 1. Major Basins of the Chesapeake Bay Watershed

for examining co-benefits of activities aimed at improving water quality because it established goals for a broad range of ecosystem services as part of efforts to restore Chesapeake Bay. Land use in the Potomac River basin is sufficient to allow it to represent the relevant issues; approximately 25 percent of the basin's area is agricultural and 10 percent is urban, and there are 97 significant PS dischargers (wastewater and industrial operations).

Our analysis applies two main restoration objectives for the Potomac River basin. The first, and primary, objective is meeting the TMDL cap by reducing loads from the basin to the bay by 6.77 million pounds of nitrogen (N), 1.03 million pounds of phosphorus (P), and 509.72 million pounds of sediment. The second objective is to maximize the benefits of a bundle of ecosystem goods and services. Note that our case study is a simplified version of the states' TMDL implementation strategies. As such, we intend to provide general insights rather than specific policy recommendations.

Modeling Framework

We developed and solved the optimization framework in the General Algebraic Modeling System (GAMS, 2012) and used new and existing models and data sources to characterize the costs and effectiveness of practices by location in the watershed. The model relies in many ways on the structure, data, and model output from the Chesapeake Bay Program's (CBP's) Phase 5.3 Watershed Model (CBWM) (EPA 2010a), which simulates sources, controls, and transports of nutrient and sediment throughout freshwater portions of the Chesapeake Bay watershed. Our model incorporates selected CBP model inputs and outputs but also adds several components to capture ecosystem service outputs, the cost of nutrient and sediment reduction practices, and spatial representation of landscape capacity to implement various NPS best management practices (Figure 2).

The effectiveness of pollution-control practices was expressed in terms of reductions in nutrient and sediment loads delivered to the bay's tidal segments

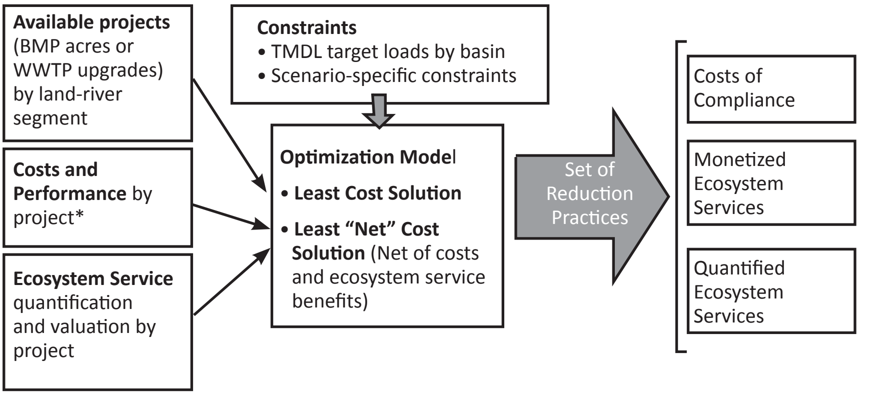

Figure 2. Major Analysis Components

* Some components of the cost data are averaged by county while others reflect watershed averages. The performance of a NPS project depends on conditions within the land-river segment and within the hydrogeomorphic region (agricultural BMPs) or tributary basin (urban BMPs). PS project costs were estimated for specific projects.

and was based on modeled responses from the CBP model. However, we added models to estimate how the practices also led to changes in a set of ecosystem service co-benefits. We developed ecological production functions for six types of ecosystem service co-benefits—climate regulation, duck hunting, non-waterfowl hunting, health and aesthetic benefits from air quality improvements, habitat-related services derived from brook trout, and flood-risk reductions—using wetland water storage as a proxy. The first four services were valued using benefit transfer and were used in the optimization. The other two services were reported as quantitative proxies of benefits and not used directly in the optimization.

The optimization was solved for multiple scenarios using two objective functions: least cost and least net cost. In both, we treated the unit cost and load reduction from each potential control option as spatially independent; therefore, the optimization could be solved as a mixed-integer linear programming problem rather than requiring more complex and computationally intensive optimization routines (see Kling 2011). The least-cost objective was to minimize the total cost (*TC*) of reducing loads to Chesapeake Bay subject to the constraints of achieving TMDLs for N, P, and sediment in each major tributary basin.

(1) $$\min_{A,E} TC = \sum_{ij} C_{ij} A_{ij} + \sum_{kl} D_{kl} E_{kl}$$

subject to:

Reductions for all pollutants (N, P, sediment) ≥ target reductions;

A_{ij} ≤ available acres for NPS practice *i*; and

No more than one option *k* is used per plant *l*

where C_{ij} is cost per acre of NPS-pollution-control practice *i* in location *j*, A_{ij} is acres of implementation of BMP *i* within land-river segment *j*, D_{kl} is the cost of project *k* at plant *l*, and E_{kl} is a binary variable indicating whether project *k* at plant *l* is used. In the function, *i* represents the NPS nutrient or sediment reduction practice (including all agricultural and urban SW practices), *j* is the land-river segment (cost and effectiveness of BMPs vary by location), *k* is the PS upgrade project (one of three mutually exclusive options developed per plant), and *l* is the treatment plant.

Although the costs and effectiveness of BMPs are represented as varying by land-river segment, the data set includes some variables that differ by county, hydrogeomorphic region, and land use type. Within each geographic unit, however, the per-acre cost of applying each BMP (C_{ij}) was assumed to be constant. For the performance data by land-river segment, the model uses the mix of land cover, geology, and existing level of implementation of BMPs.

Our second objective was to minimize the *least net cost*, which is defined as the difference between the pollution control costs and the value of the ecosystem service co-benefits, subject to the same constraints as equation 1.

(2) $$\min_{A,E} TC = \sum_{ij} \left(C_{ij} - \sum_{n} S_{nij} \right) \mathrm{A_{ij}} + \sum_{kl} D_{kl} E_{kl}$$

where S_{nij} is the per-acre value of ecosystem service type *n* produced by NPS pollution control practice *i* in location *j*.

Data and Methods for Quantifying Costs and Load Reductions from Projects

We created an inventory of discrete potential nutrient and sediment reduction projects for use in the optimization. The projects represented controls on either PS or NPS emitters. PS control projects include alternative "tiers" of nutrient removal at WWTPs that represented various levels of technology adoption. NPS controls included a representative set of agricultural and urban SW BMPs.

Point Sources and Wastewater Treatment Technologies

The model included 97 significant municipal and industrial wastewater facilities, which accounted for a large majority of the N and P loads from PSs in the Potomac basin. Data available from EPA (2009) included the facilities' locations, quantity of discharges, and factors determining the proportion of facility load delivered to tidal waters of the bay. Those factors account for attenuation of pollutants between the point of discharge and the bay.

Based on the technology classification system from the CBP, we defined four discrete tiers of nutrient reduction options for these facilities up to the limit of technology. The cost and effectiveness of upgrades were based on two CBP studies (2002, 2004) that provided the estimated capital and operation and maintenance (O&M) costs for each WWTP to incrementally reduce N and P effluent concentrations. For each facility, the total annualized cost (capital and O&M) and reductions in N and P delivered to the bay were evaluated by tier to define distinct potential projects.

Agricultural and Urban Stormwater Sources and Best Management Practices

The main data source and framework used to characterize agricultural and urban SW sources was the CBWM, from which the following main data elements in our analysis were drawn.

> *Watershed network and segmentation*. Our model subdivided the Chesapeake Bay watershed into a linked network of 1,955 "land-river segments."
>
> *Land use / land cover segmentation*. Our model subdivided each land-river segment into twenty-six land use categories that we aggregated into two agricultural categories (crop and pasture land), four urban land categories, and one "other" category.
>
> *Delivered loadings*. For each land use category in each land-river segment, our model provided estimates of total annual delivered loadings in 2009 for N, P, and sediment.

Using CBWM data (CBP 2009) on 2009 levels of BMP implementation in each land-river segment, we defined areas that would be eligible for implementation of new agricultural and urban SW BMPs (e.g., an area being treated by an existing forest or grass buffer would not be available for a new forest or grass buffer). We limited the areas available for wetland restoration to a zone within 1,044 feet of surface waters (the width of a 5-acre square) because that zone was assumed to be most likely to support wetlands and produce water-quality benefits. The model also allowed compatible practices to be applied to the same acre. In those cases, we adjusted the nutrient and sediment removals to account for the interdependencies of load reductions for combined practices.

To incorporate agricultural BMPs into the cost-minimization analysis, we developed estimates of annual reductions in delivered loads per acre and annual cost per acre for each BMP. The amount of nutrients and sediment removed by each BMP per acre was a function of (i) baseline loadings per acre, (ii) loadings per acre of new land use, and (iii) removal effectiveness of the BMP given existing BMPs. We based the effectiveness of removal on estimates created for the CBWM (CBP 2009) and varied the estimates according to the hydrogeomorphic region and other location variables.

Our model includes nine agricultural BMPs and five urban SW practices. Table 1 provides watershed-wide ranges of removal efficiencies for seven of the nine practices used. The two practices not shown are natural revegetation (allowing land to lie fallow and revert to natural vegetative cover) and conversion to forest (planting and nurturing trees to convert existing agricultural land to

Table 1. Summary of Treatment Efficiencies for Selected Best Management Practices

	Removal Efficiencies (percent)		
Best Management Practice	**Total Nitrogen**	**Total Phosphorus**	**Total Suspended Solids**
Agricultural Best Management Practices[a]			
Forest buffers[b]	19–65	30–45	40–60
Grass buffers[b]	13–46	30–45	40–60
Wetland conversion[c]	7–25	12–50	4–15
Livestock exclusion[d]	9–11	24	30
Cover crops[e]	34–45	15	20
No-till[f]	10–15	20–40	70
Reduced fertilizer application[g]	15	0	0
Urban Stormwater Best Management Practices			
Extended detention[h]	20	20	60
Bioretention[i]	48	60	68
Grass buffer[b]	32	40	53
Forest buffer[b]	50	60	60
Wetlands[c]	20	45	60

[a] Load reductions from conversion to forest and natural revegetation BMPs were based on differences in loading rates between land cover categories rather than removal rates.

[b] Planting strips of trees or grasses on land located between a potential pollutant source (e.g., agricultural or urban land) and a body of surface water.

[c] Returning drained agricultural or urban land to its natural/historic function as wetlands.

[d] Establishing fences and other structures to exclude livestock from streams and other waterways.

[e] Planting secondary crops (not for harvest) for soil enhancement and erosion prevention.

[f] Excluding the soil tilling step in crop production to increase water and nutrient retention and reduce soil erosion.

[g] Reducing nitrogen applied to crop land as chemical and natural fertilizer by 15 percent.

[h] Engineered structures designed to capture and store runoff and release it slowly for control of peak runoff and velocities with pollutant removal by settling (not designed to promote infiltration).

[i] Vegetated, landscaped depressions that allow for retention and infiltration of runoff.

forest), which do not have associated removal efficiencies. For these practices, we estimated load reductions by comparing the loads associated with the land cover before and after conversion by land-river segment, as was done in the CBWM. We assumed that fallow land produced loads comparable to "hay-unfertilized" lands.

The annual cost per acre per practice is the sum of three components: the annualized installation cost of the BMP (i.e., capital), the annual O&M costs, and the value of the land being converted from an agricultural use (i.e., the cash rental rate for crop and pasture land), where needed. 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 reported in the 2008 Cash Rents Survey (National Agricultural Statistics Service (NASS) 2011). We report ranges of average costs per acre for the nine BMPs in Table 2; the values used in the analysis depend on the location being analyzed. Values for the costs and efficiency of nutrient removal for urban SW BMP were drawn from a literature review (Morin et al. 2010) and are summarized in Tables 1 and 2.

Data and Methods for Quantifying Ecosystem Service Co-benefits from Agricultural and Urban Stormwater Best Management Practices

Table 3 lists the ecosystem service co-benefits included in the model and the BMPs to which they were applied. A variety of data sources and benefit-transfer methods were used to generate the monetary and nonmonetary estimates of the

Table 2. Summary Unit Costs for Selected Best Management Practices

Practice	Total Annual Cost per BMP per Acre (dollars per acre per year)	
	Low	**High**
Agricultural Best Management Practices		
Forest buffers	163	291
Grass buffers	99	226
Wetland conversion	236	364
Natural revegetation	14	141
Conversion to forest	129	257
Livestock exclusion	81	117
Cover crops	31	31
No-till	14	14
Reduced fertilizer application	37	37
Urban Stormwater Best Management Practices		
Extended detention	4,460	4,460
Bioretention	66,647	66,647
Grass buffer	6,676	6,676
Forest buffer	364	364
Wetlands	601	601

benefits of these services. Despite the detailed methods we used to generate the estimates (see EPA (2011) for further details), it is important to note that, due primarily to data limitations, the estimates provide fairly rough approximations of co-benefit value and they only account for a subset of the potential co-benefits from the selected BMPs. Nevertheless, they serve our purpose of investigating the potential role of ecosystem service co-benefits in TMDL policy design. To examine how the uncertainty surrounding these estimates affects our results, we include a sensitivity analysis in the final section.

Table 3. Summary of Ecosystem Service Co-benefits Included for Selected Best Management Practices

	Monetized Ecosystem Service				Nonmonetized Ecosystem Service	
Practice	Carbon Sequest. and Reduced GHG Emissions	Non-waterfowl Hunting	Duck Hunting	Air Quality	Brook Trout Habitat	Wetland Water Storage
Agricultural Best Management Practices						
Forest buffers	•	•			•	
Grass buffers	•					
Conversion to forest	•	•			•	
Natural revegetation	•	•			•	
Wetland restoration	•	•	•		•	•
No-till	•					
Reduced fertilizer	•					
Urban Stormwater Best Management Practices						
Extended detention	•					
Bioretention	•			•		
Grass buffer	•			•		
Forest buffer	•			•		
Wetlands	•			•		•

Table 4. Per-acre Value of Reduced Greenhouse Gas Emissions and Carbon Sequestration Services from Best Management Practice Application

	Annualized Value[a] in Dollars per Acre	
Best Management Practice Application	From Crop Land	From Pasture
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] $45 per ton of carbon; 90-year period; 3 percent discount rate.

Changes in Greenhouse Gas Emissions and Carbon Sequestration

Greenhouse Gas Emissions. Our analysis focused on three greenhouse gas (GHG) emission types—carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) —all expressed in a common unit of emissions, CO_2e. We estimated average per-acre emission rates for three land cover types: crop land, pasture land, and wetlands. The N_2O emission rates for crop and pasture land were based on data from Adams et al. (1996), and CO_2 and CH_4 emission rates for wetlands were derived from data provided by the Intergovernmental Panel on Climate Change (IPCC) (2006). For BMPs involving conversion of crop and pasture land to other uses, we assumed that emissions would be reduced to zero. For acres converted to wetlands, we added the wetland-specific GHG emission rate. To estimate the reductions from reduced fertilizer applications, we assumed that N_2O emissions from the affected crop land would decline in proportion to the decline in fertilizer application (i.e., 15 percent).

Carbon Sequestration. We calculated carbon sequestration rates for conversion of agricultural lands to forests, wetlands, or grasslands for no-till agriculture and for most SW management practices (Table 4). For crop or pasture land afforestation, we assumed that the land would be planted with the most common tree species found in the dominant forest type of each ecoregion. For conversion to wetlands, we assumed planting of a bald cypress and water tupelo type of forest (Neely 2008). For natural revegetation, we assumed that the land would naturally regenerate to an even mixture of all of the forest types found within the ecoregion. The National Council for Air and Stream Improvement and the U.S. Forest Service's Carbon On-Line Estimator (COLE) (Van Deusen and Heath 2011) was used to calculate the amount of carbon sequestered by agricultural BMPs. Carbon sequestrations from urban SW BMPs and from no-till agriculture were estimated using the Urban Forest Effects (UFORE) Model (Nowak and Crane 2000) and the IPCC (2006) methodology, respectively.

To express GHG and carbon-related ecosystem services in monetary terms, we considered a range of values based on estimates of the marginal social cost of carbon sequestered and the large degree of uncertainty associated with estimating and projecting such values. As a central value estimate, we used $45 per metric ton of carbon ($12 per ton of CO_2) based on results in Tol (2005, 2008) and IPCC (2006). To consider sensitivity of the results to this value, we applied a range of values—from a low estimate of $26 per metric ton of carbon ($7 per ton of CO_2) to a high estimate of $92 per metric ton ($25 per ton of CO_2)—that correspondsd broadly with the range of recommended values by the Interagency Working Group on Social Cost of Carbon (2010). Applying the estimates to the estimated time paths of carbon flux reported, we calculated the present value of carbon storage and net change in GHG emissions associated with each land-use-conversion type using a 3 percent discount rate over a 90-year time horizon to create an annualized value of carbon storage. Table 4 shows values based on the central value of $45 per ton of carbon or carbon equivalent for other GHGs.

Duck Hunting Services from Wetland Restoration

We estimated the effects of wetland conversion on duck hunting services using methodology from Murray et al. (2009). Their methodology applied only to wetlands created by conversion from crop and pasture land. The first step was to evaluate changes in the number of ducks potentially supported by wetlands by estimating the energetic carrying capacity of land cover types using a "duck energy day" (DED) model (Reinecke and Kaminski 2005). Based on our review of the literature, we estimated average DEDs per acre for corn and soybean crop lands and for freshwater and tidal wetlands. The second step was to estimate baseline DEDs by multiplying the number of acres in each land cover category by the corresponding estimates of DEDs per acre from the first step.

The third step was to estimate the baseline value of duck hunting services, which involved multiplying the total number of duck hunting days per state (Richkus et al. 2008) by the regional average consumer surplus value of a duck hunting day (Rosenberger and Loomis 2001). The final step was to estimate the increase in the value of duck hunting services associated with each acre of land converted from crop or pasture to freshwater or tidal wetland. We assumed that the aggregate value of duck hunting in each state increased in direct proportion to the increase in total DEDs. An assumption that duck hunting activity would increase in proportion to the duck supply is likely to have inflated the value of this change. However, in analyzing the data, we found a positive correlation between hunting participation by state and acres of wetland (unpublished analysis) that supported this assumption. We did not have a model to support a more specific assumption of the rate of increase in participation with supply. The results of our analysis are reported in Table 5 as annual per-acre values associated with wetland restoration.

Non-waterfowl Hunting Services from Increases in Forest Cover

To estimate the effects of land use and land cover changes on other hunting services, we applied the results from a hedonic price study of hunting leases in central Florida by Shrestha and Alavalapati (2004). We chose that study because, among available studies on the effects of land cover on hunting values, it examined the area closest geographically to Chesapeake Bay. Using their estimates of the elasticity of hunting values with respect to forest cover, we assumed that each 1 percent increase in forest cover (per state) would increase the average annual value of non-waterfowl hunting in that state by 0.132 percent. We derived the estimates of baseline hunting values from hunting participation data in Ribaudo et al. (2008) and from average per-day consumer surplus estimated for small- and big-game hunting by Rosenberger and Loomis (2001). We estimated the value of changes in consumer surplus from non-waterfowl hunting by first multiplying user days by average consumer surplus under baseline conditions. We then multiplied those baseline values by the product of the change in forest cover (in acres) and the elasticity parameter that represented the incremental annual value of additional forest cover. The resulting estimates of average values of hunting per acre of additional forest cover are reported in Table 5.

Removal of Atmospheric Pollutants by Urban Stormwater Best Management Practices

Vegetation-based urban SW BMPs are thought to remove atmospheric pollutants and thereby improve air quality and promote human health outcomes and other benefits. The UFORE Model incorporates research on the removal efficiencies of plants to estimate changes in atmospheric pollutants associated with urban trees and shrubs and the value per ton of pollutant removed (Nowak, Crane, and Stevens 2006). Module D of UFORE ("UFORE D: Dry Deposition of Air Pollution") calculates dry deposition rates for ozone (O_3), sulfur dioxide (SO_2), nitrogen dioxide (NO_2), carbon monoxide (CO), and particulates (PM_{10}). We generated values per ton of pollutant removed using the UFORE Model (Nowak, Crane, and Stevens 2006). The UFORE calculations were based on Murray, Marsh, and Bradford (1994), which summarized externality values used in multiple studies of energy decision-making. Table 6 provides annualized estimates of the values of atmospheric pollutant removal by pollutant species based on mid-range estimates for pollutant-removal rates.

Brook Trout Habitat and Recreational Fishing Services

To estimate the effects of changes in land cover on brook trout habitat status, we adapted a regression tree model (Hudy et al. 2008) to characterize the status of stream populations (eastern United States) as a function of selected land use characteristics, including the percentage of forest cover within the subwatershed and the condition of the riparian buffer. Increases in forest cover and forested riparian cover can create the cool water conditions that promote fish survival and can thereby provide a restoration opportunity. We applied this model to predict the current status of brook trout within 1,414 subwatersheds (12-digit HUCs) and change in population status (intact, reduced, and extirpated). Intact

Table 5. Incremental Annual Value of Hunting Services per Acre of Land Conversion (dollars per acre per year)

	Duck Hunting Values by Land Conversion Category				Non-waterfowl Hunting Values
State	Crop Land to Tidal Wetland	Crop Land to Freshwater Wetland	Pasture to Tidal Wetland	Pasture to Freshwater Wetland	Pasture or Crop Land to Forest
Maryland	7.56	3.33	8.49	4.27	1.90
Pennsylvania	NA	2.24	NA	4.01	3.24
Virginia	3.78	1.69	4.21	2.12	1.21
West Virginia	NA	0.94	NA	1.59	1.83

Table 6. Annual Value of Pollutant Removal by Urban Stormwater Best Management Practices (dollars per acre per 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

watersheds have self-sustaining brook trout populations, reduced watersheds support stocked fish, and extirpated watersheds have no brook trout. We report only conversions to intact status because we assume that other factors besides land use are likely to limit the transition from extirpated to reduced status.

Water Storage and Flood Control from Freshwater Wetlands

Although we expected freshwater wetlands to provide a variety of ecosystem services, we used a metric of potential water-holding capacity weighted by location characteristics as a rough but readily quantifiable proxy for potential flood-control services. The metric is an average value for water storage capacity that we modified to reflect geophysical variability, proximity of property to be protected, and the presence of existing flood-control structures. An average storage capacity of three acre-feet of water per acre of wetlands was derived from literature sources (EPA 2006), and four characteristics were used to account for geographic differences in potential water storage services per acre-foot: landform suitability (using soil data from NRCS (2006, 2010)), wetland suitability (proximity to wetlands identified by the National Wetlands Inventory (USFWS 2011)), flood protection (distance upstream of urban areas and downstream of flood control dams), and water storage potential (GIS analysis of elevation data). These factors were combined into an index that rated wetlands as having a low, medium, or high degree of potential to produce flood risk reduction benefits.

Scenario Analysis

We used multiple scenarios to explore the implications of decisions to (i) limit the types of BMPs allowed, (ii) require ratios for trading between NPS and PS emitters, and (iii) require a minimum percentage of load reduction from the urban SW sector (Table 7). A major assumption underlying all of the scenarios is that programs for trading or offsets will work perfectly to shift load reduction activities from emitters with relatively high costs of compliance to those with low costs of compliance. The assumption that all potential gains from trade will be fully realized is highly optimistic (Shortle, this issue). However, our intent was to show the potential benefits of trading and offsets rather than estimate levels of participation. Except where noted, all of our scenarios assumed that load allocations (i.e., allocations of wasteloads) were given exclusively to regulated PS emitters. In addition, to account for the relatively high transaction costs associated with arranging trades or offsets from the NPS sectors, we added a 10 percent increment to the unit costs (C_{ij}) of all of the BMPs.

Results and Policy Implications

The optimization analysis identified the mix of practices that would meet the watershed's TMDL for the lowest cost with and without considering offsetting effects of ecosystem service co-benefits. The results of the two optimization approaches are identified as (i) the *least-cost* solution when optimization minimizes compliance costs (equation 1) and (ii) the *least-net-cost* solution when the analysis minimizes the difference between compliance costs and ecosystem service co-benefits (equation 2). The value of ecosystem service benefits used in the least-net-cost solution captures only a narrow set of benefits that are derived from terrestrial systems and that are ancillary to the benefits generated by water-quality improvements. Therefore, the net cost calculations did not include all sources of benefits and were intended only to compare relative benefits among scenarios rather than true net benefits. All of the scenarios that succeeded in meeting the N, P, and sediment load limits (Table 7) were assumed to produce the same level of benefits that could be derived from improvements to tidal aquatic systems.

Total compliance costs and ecosystem service outcomes for all of the scenarios for the Potomac River basin are shown in Table 8. Results for scenarios other than the baseline are grouped by the policies that they reflect. The results for the Potomac River basin differ in some important ways from results for other basins in the Chesapeake Bay watershed and for the bay TMDL overall, and we discuss those differences where they are most relevant.

Base-case Scenario

The least-cost solution for the base case (scenario 1 in Table 7), which involved a 1:1 NPS credit ratio, no restrictions on conversion of agricultural land, and no sector-specific load reduction requirement, resulted in $23.2 million in annual compliance costs and produced $11.8 million in annual ecosystem service co-

Table 7. Scenarios Used in Optimization

		Scenario-specific Features[a]		
Scenario		Nonpoint-source Credit Ratio	Agricultural Land Conversion Restrictions	Sector-specific Load Reduction Requirements
1	Base case	1:1	Unrestricted	None
2a	High farm land conversion restrictions	1:1	No conversion outside of 100-foot buffer	None
2b	High farm land opportunity costs	1:1	Unrestricted but farmers are reimbursed at 2.2 times the rental rate	None
2c	Intermediate farm land conversion restrictions	1:1	No more than 10 percent farm land per land-river segment	None
3	Untradeable stormwater allocation	1:1	Unrestricted	20 percent of reduction of pollutants must be met by storm-water sector
4a	Moderately precautionary use of BMPs	2:1	Unrestricted	None
4b	Highly precautionary use of BMPs	3:1	Unrestricted	None
5	Intermediate farm land conversion restrictions and moderately precautionary use of BMPs	2:1	No more than 10 percent farm land per land-river segment	None

[a] All scenarios include the same PS and NPS control options, unit costs (including a 10-percent transaction cost increment for NPS controls), and ecosystem service co-benefits.

Table 8. Scenario Results for the Potomac Basin: Least-cost Solutions

	Scenario							
	1	2a	2b	2c	3	4a	4b	5
	Base Case	No Agricultural Conversion beyond Buffer	2.2 Times Rental Rate	10 Percent Ag Conversion	20 Percent Stormwater Allocation	2:1 Credit Ratio	3:1 Credit Ratio	10 Percent Ag Conversion; 2:1 Trading
Area of applied practices (acres)	532,644	587,228	574,787	764,776	376,036	1,559,762	2,261,941	757,056
Annual N reduction (pounds/year)	6,766,832	6,766,832	6,766,832	6,766,832	6,766,832	6,766,832	6,788,666	6,766,832
Annual P reduction (pounds/year)	1,031,620	1,016,222	1,031,620	1,031,620	1,031,620	1,031,620	**857,319**	**690,965**
Annual sediment reduction (pounds/year)	692,952,976	817,977,117	697,527,584	808,204,723	677,935,222	509,721,953	**490,947,933**	**455,547,137**
All ecosystem service co-benefits	11,784,837	4,112,109	10,225,393	10,245,970	11,089,931	55,903,835	91,403,197	11,445,641
Carbon sequestration and reduced GHG emissions (dollars/year)	11,444,637	3,070,206	9,943,021	9,501,201	9,549,568	53,920,648	87,008,216	10,466,879
Non-waterfowl hunting (dollars/year)	332,362	0	265,289	253,202	236,268	1,916,024	3,882,739	317,093
Duck hunting (dollars/year)	0	913	0	20,351	0	0	11,051	58,309
Air quality (dollars/year)	7,838	1,040,989	17,083	471,217	1,304,095	67,163	501,191	603,360
Additional intact subwatersheds for brook trout (HUC-12)	0	0	0	4	0	35	63	4
Wetland storage potential (acre-feet)	3,359	30,330	7,321	40,752	24,137	28,202	35,862	101,408
Annual control cost	23,214,669	374,445,496	33,484,984	89,967,073	555,696,016	87,443,514	464,164,739	262,065,935
Annual NET cost (dollars per year)	11,429,832	370,333,388	23,259,591	79,721,103	544,606,085	31,539,679	372,761,541	250,620,294

Note: Bolded numbers represent values that are below TMDL target reductions.

benefits for an estimated net cost (excluding water-quality benefits) of $11.4 million annually (Table 8). In addition to the monetized ecosystem services, the base case produces about 3,400 acre-feet of water storage capacity in wetlands in locations with moderate to high potential to protect properties from floods. About 99 percent of the N load reduction, 71 percent of the P reduction, and 96 percent of the sediment reduction were met using agricultural BMPs. Of the agricultural practices used in the least-cost solution, 24 percent of the N reduction, 43 percent of the P reduction, and 68 percent of the sediment reduction came from working-land options (BMPs that do not require taking land out of production) and the remainder from practices that convert agricultural land to permanent vegetation. Urban SW controls accounted for 1 percent of the N reduction, 3 percent of the P reduction, and 4 percent of the sediment reduction. PS controls accounted for less than 1 percent of the N reduction, 26 percent of the P reduction, and 0 percent of the sediment reduction. The Potomac results differ from results for the entire watershed, where PS controls accounted for 36 percent of the effort in the base-case solution. This greater use of NPS controls in the Potomac suggests that PS controls in the Potomac are more costly than in some other basins.

Protection of Working Lands

Two main findings are revealed by comparing the cumulative cost curves and the practices used in the least-cost solution for the base case and scenario 2a, which prohibited conversion of working land located outside of a 100-foot stream buffer. First, without policies that restrict land conversion, revegetation of farm land would represent a substantial portion of the nutrient and sediment reduction effort in a least-cost solution that complies with the TMDL. Second, when urban SW and PS pollution controls are substituted for agricultural options, the total cost of compliance increases substantially and marginal costs increase dramatically as effort approaches the compliance target (Figure 3).

The increase in the total compliance cost from $23 million for the base case to $374 million for scenario 2a reveals the potentially substantial cost savings from using a full suite of agricultural BMPs compared to a policy that precludes conversion of farm land (Table 8). The cost increase under the conversion restriction results from substitution of more costly SW and PS options to meet the target limits. The cumulative cost curve for scenario 2a has two inflection points, one at about 65 percent of compliance and one at 92 percent of compliance with all pollution targets (Figure 3). These inflection points represent transitions—first to more expensive PS controls and later to more expensive urban SW practices. The dramatic increase in cumulative cost near the TMDL caps under scenario 2a demonstrates that without substantial activity in trading or offset programs, a large proportion of spending could be required to meet a relatively small proportion of nutrient and sediment reductions.

The increase in cost between the base case and scenario 2a occurs because BMPs for working lands do not appear to have sufficient capacity to substitute completely for farm land conversion options (Figure 4). In other basins in the Chesapeake Bay, some pollution reduction targets cannot be met when agricultural conversion is restricted without using some of the most costly reduction practices from SW and PS emitters (EPA 2011). A caveat to our finding that working-land options have insufficient capacity to substitute for

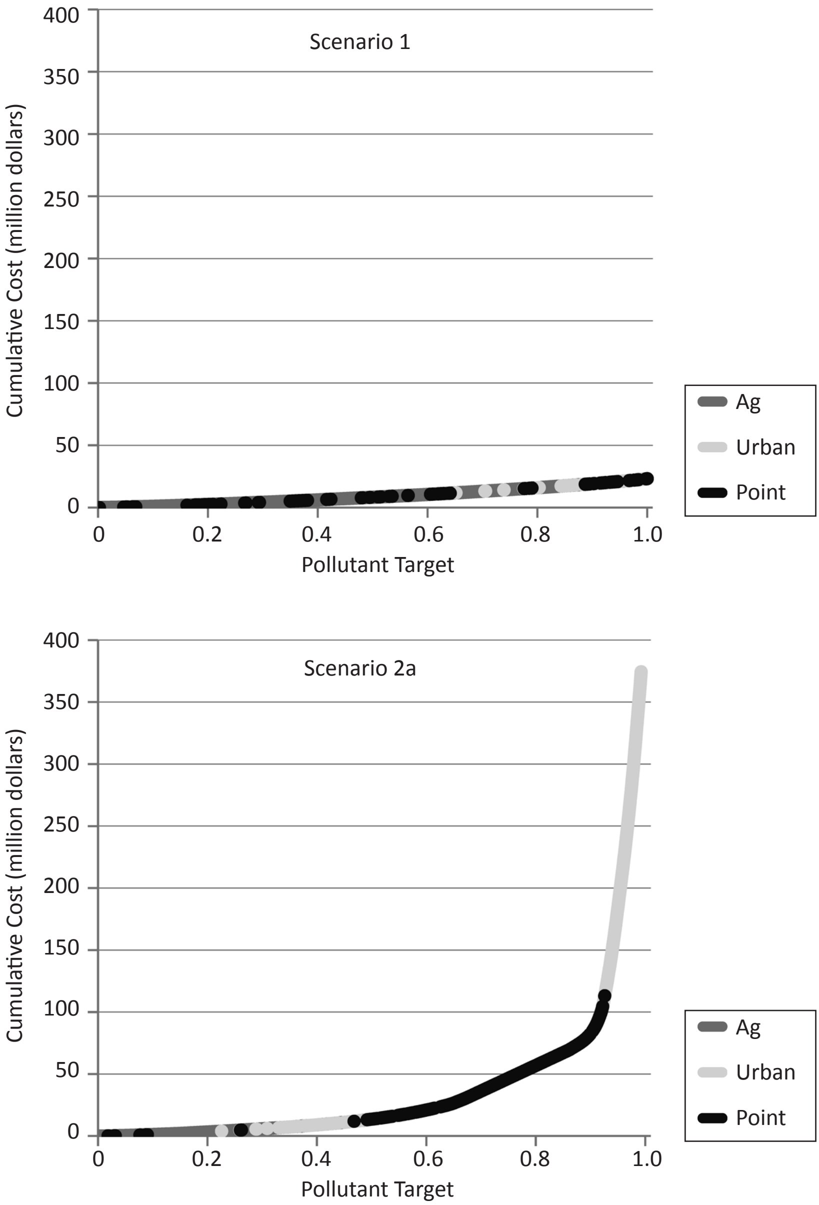

Figure 3. Comparison of Cumulative Cost Curves without (scenario 1) and with (scenario 2a) Restrictions on Agricultural Land Conversion for the Potomac Basin

farm land conversion options is that our framework does not include the full suite of working-land options available for credit by the CBP, but it does include most practices with broad applicability and our model allows multiple practices to be applied per farm. Our framework also omits options for confined animal feeding operations and new septic connections or upgrades, which account for about 8 percent of existing N loads in the Potomac basin (EPA 2010b). If working

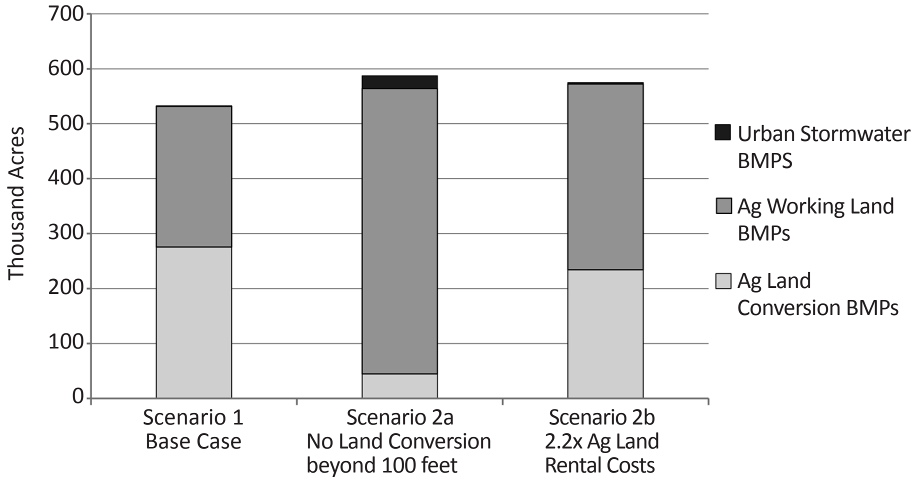

Figure 4. NPS Acres by Practice Type under Alternative Treatments of Agricultural Land

lands have substantially greater reduction capacity than our analysis suggests, they may represent an effective substitute for land conversion options.

The base case results likely represent an unrealistic amount of conversion of crop land to permanent vegetation: 243,000 acres converted (41 percent of the crop land in the watershed). For context, consider that the U.S. Department of Agriculture (USDA) Conservation Reserve Program (CRP) limits enrollment to 25 percent of crop land by county (HR 6124, Section 2708.f.1.A). Despite its lack of realism, the scenario clarifies a key tradeoff—namely that efforts to maintain crop production at or near existing levels compete with goals to reduce costs of meeting the TMDL. However, the savings in the least-cost scenario require \$10.5 million in compensation annually for lost farm rent, which suggests that substantial impacts may be associated with converting that amount of agricultural land. Loss of farm land could be associated with several negative consequences, including decreased availability of local animal feed and/or produce, higher commodity prices, and loss of farm land amenities, none of which were considered in our model.

Agricultural land-conversion options appear to be highly cost-effective relative to many working-land options. However, this result is sensitive to the value used to represent the opportunity cost of taking land out of agricultural production. When rental rates are multiplied by 2.2 to better represent both current opportunity costs and expectations of future commodity price increases (Hellerstein 2010), the proportion of working-land options in the least-cost solution increases substantially (scenario 2b in Figure 4). With the higher rental rate, the proportion of nutrients and sediment removed by working-land options increases by 7 percent relative to the base case and accounts for 31 percent of N, 50 percent of P, and 75 percent of sediment.

To test the potential effects of alternative policies aimed at preventing conversions of agricultural land, we compared total TMDL compliance costs for all possible levels of restriction on conversion of agricultural land. We found that the effects are nonlinear and that the cumulative cost of TMDL compliance in the Potomac basin increases substantially when the maximum allowable

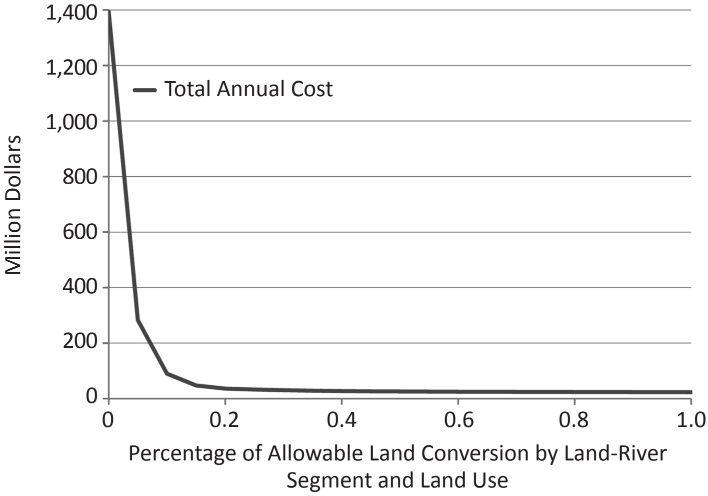

Figure 5. Cumulative TMDL Annual Compliance Cost (million dollars) under Different Levels of Restriction on Agricultural Land Conversion within the Potomac Basin

amount of converted agricultural land drops below 10 percent (Figure 5). The point at which restrictions on conversion of agricultural land become binding is around 15 percent, but costs begin rising rapidly at 10 percent and continue to increase as the maximum percent of allowable agricultural land conversion declines. These results differ somewhat from those for the bay watershed as a whole. In the whole-basin scenario, costs begin to increase dramatically when the maximum allowable conversion drops below 20 percent. The slope of the cumulative cost curve increases as cost-effectiveness of a given practice declines, as the capacity of a NPS practice is exhausted, and as a more advanced technology (higher tier of effort) is used at a given PS facility.

Minimum Load Reductions from the Stormwater Sector

In scenario 3, 20 percent of pollution reductions must be met within the SW sector without using trading or offsets. With a 20-percent allocation to SW, the cost of compliance increases by 2,500 percent of the baseline cost to $556 million, the highest cost of all of the scenarios (Table 8). This result is consistent with the results from the scenarios with restrictions on conversions of agricultural land. In both cases, when reduction capacity comes from the urban or SW sector instead of the agricultural sector, costs increase substantially.

We tested the sensitivity of the SW allocation cap by considering multiple percentage allocations to SW (Figure 6). As with the agricultural restrictions, the slope of the cumulative cost curve rapidly exceeds the point at which the allocation forces the model from the least-cost solution. The allocation becomes binding between 10 and 15 percent and increases steadily for allocations greater than 20 percent. These results suggest that policies that restrict the use of trading or offsets between urban SW and agricultural sectors will increase the aggregate cost of compliance.

Ecosystem service co-benefits under scenario 3 were estimated to be $11 million, an amount similar to co-benefits under the base case, but in scenario 3 that $11 million offsets only 2 percent of the cost (Table 7). The scenario generates lower carbon sequestration benefits than the base case but a substantial increase in benefits related to air quality improvements. The increase in urban ecosystem service co-benefits stems from the heavy use of "green" urban BMPs in the analysis framework rather than more traditional "gray" infrastructure approaches.

Managing Uncertainty of NPS Performance

The results from scenarios 4a and 4b, which apply different ratios for PS-NPS trades or offsets (described in Table 7), reveal the effects of policies aimed at managing BMP performance uncertainty. The base case (scenario 1) used the CBP's best estimates of the amount of pollution reduction expected by practice and location. However, due to variability of on-farm conditions, maintenance practices, weather, and other factors not already included in the efficiency estimate, a BMP's performance in reducing NPS pollution in any given year and at any particular location is highly uncertain. Trading and offset programs can apply ratios to represent the greater uncertainty of performance for NPS practices relative to PS practices. For example, the 2:1 trading or offset ratio used in scenario 4a effectively reduced the assumed effectiveness of all NPS BMPs by 50 percent. The higher the ratio used, the more precautionary the approach to dealing with such uncertainty is. As previously noted, all of the scenarios assumed a 10-percent transaction cost increment for trades between PS and NPS sectors.

Even though the cost per pound of nutrient or sediment increases substantially for the NPS BMPs in scenarios 4a and 4b, the NPS BMPs are still less expensive than many PS or urban NPS options in the Potomac basin, as evidenced by the extensive use of NPS BMPs in the least-cost solution (Figure 7). When the offset ratio is increased from 1:1 (used in scenario 1) to 2:1 (used in scenario 4a), the

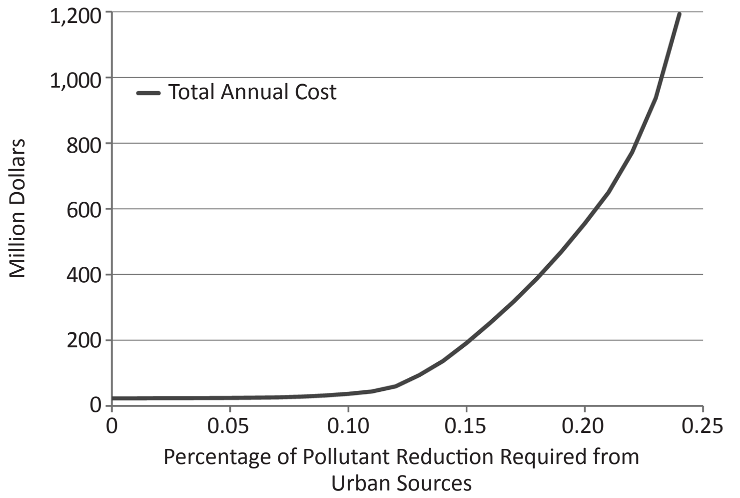

Figure 6. Cumulative TMDL Compliance Cost under Alternative Levels of Required Reductions from the Urban Stormwater Sector within the Potomac Basin

overall agricultural BMP effort (measured as pounds removed) drops slightly—by 4 percent of N effort, 3 percent of P effort, and 10 percent of sediment effort. At a 3:1 trading or offset ratio (scenario 4b), agricultural BMP effort declines by 11 percent of N effort, 14 percent of P effort, and 7 percent of sediment effort relative to the base case. We know from our results (not presented here) that the proportion of PS and SW effort increases more substantially in some of the other bay basins as the credit ratio increases, so this result is specific to the Potomac basin.

An unanticipated result of scenarios with higher trading or offset ratios (4a and 4b) is that the amount of working-land options in the least-cost solution *decreases* substantially under higher credit ratios (Figure 8). This results from the model's need to use the most effective practice on a given acre to reach the reduction targets without using high-cost options. With higher trading ratios, many more acres are needed to meet the targets and the number of available acres of agricultural or urban land can become binding. Therefore, to control costs, the model selects practices with the greatest effectiveness per acre. In the 2:1 scenario, natural revegetation replaces the working-land options, but in the 3:1 scenario, active reforestation becomes the most cost-effective way to meet the targets. In the 2:1 scenario (4a), revegetation is applied to 1.5 million acres; in the 3:1 scenario (4b), 2.2 million acres are revegetated. As a result, ecosystem co-benefits increase substantially (Figure 7). In other basins, working lands are conserved as the trading ratio increases because the number

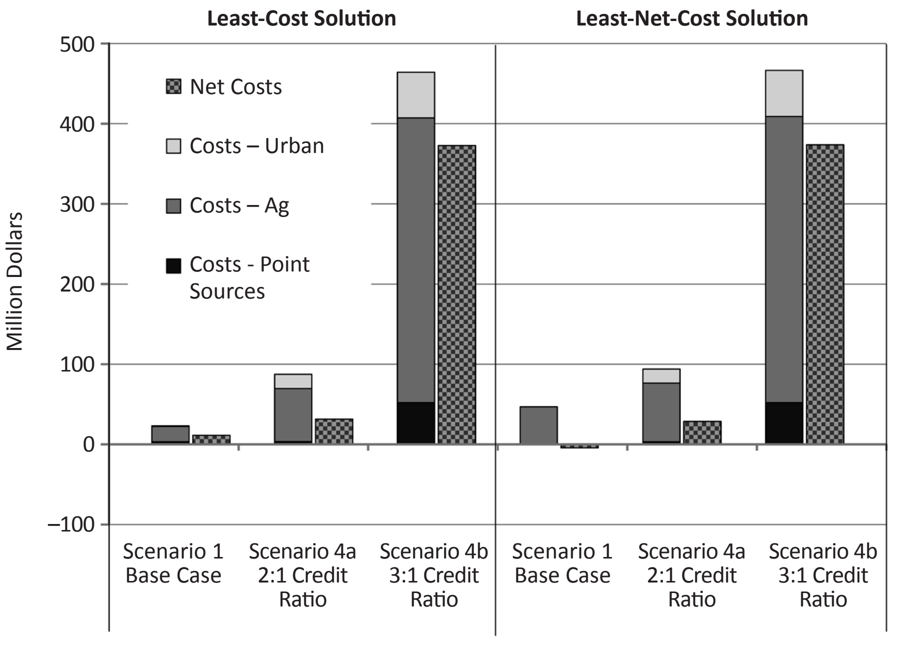

Figure 7. Annual Costs and "Net" Costs under Alternative PS-NPS Ratios for Trading and Offsets

Notes: The left side of the figure shows total costs and the distribution of costs across source sectors when the project costs are minimized by a scenario. The right side shows results when the differences between cost and co-benefit are minimized. On both sides, the checkered bars show the total cost per scenario after subtracting the value of ecosystem service co-benefits.

of acres is not binding and/or substitutes available from the PS and SW sectors are more cost-effective in those basins than in the Potomac.

For both of the scenarios that include a precautionary approach to managing BMP performance uncertainty (4a and 4b), the aggregate compliance cost rises substantially relative to the base case. In scenario 4a, the compliance cost is nearly three times the base case cost; in scenario 4b, the compliance cost is 19 times the base case cost (Figure 7). These much higher costs are partially offset by substantially higher ecosystem service co-benefits.

Figure 7 also provides a comparison of least-cost solutions (left panel) and least-net-cost solutions (right panel) for scenarios 1, 4a, and 4b. The most significant difference seen from minimizing net costs (right panel) is for scenario 1, in which the aggregate control cost is higher for the least-net-cost solution than for the least-cost solution but the net cost becomes *negative* because ecosystem service co-benefits more than offset the additional control cost. Figure 8 shows that the substantial co-benefits in the least-net-cost solution are generated by a shift away from working-land options under the least-cost solution (left panel) to natural revegetation options. In contrast, little is gained in scenario 4b by optimizing for the least net cost in the Potomac basin. The two alternative optimization approaches generate similar results because the BMPs that are most effective at creating ecosystem service benefits are already being used to capacity to reach the sediment targets with the 3:1 credit ratio (Table 8).

These results reveal the potential value of improved information about the performance of BMPs. If the values used by the CBP were reasonably

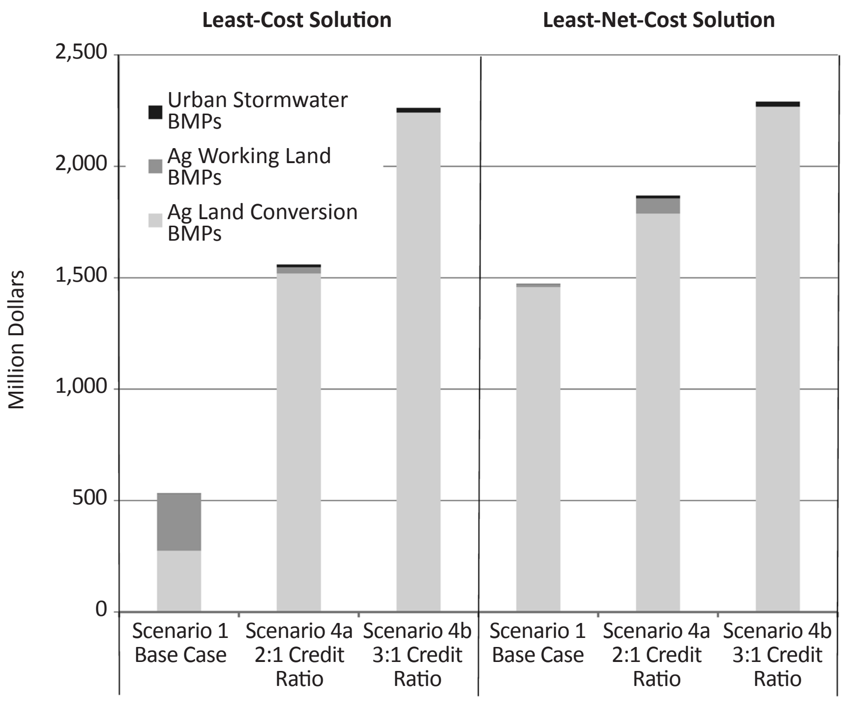

Figure 8. NPS Acres by Practice Type under Alternative PS-NPS Ratios of Trading and Offsets

accurate averages, reducing uncertainty could save between $64 million and $440 million in compliance costs by reducing the PS-NPS trading or offset ratio needed to ensure outcomes in a TMDL program. Conversely, if the estimated efficiencies were not accurate, then improved certainty could enhance the environmental performance of trading and offset programs.

Tradeoffs between Costs, Water Quality Certainty, and Watershed Co-benefits

Given current policies and market conditions, incentives for converting large amounts of farm land to permanent vegetation (scenarios 1, 4a, and 4b) are not likely to materialize, nor would they necessarily be desirable, in the bay watershed or elsewhere. Yet scenario 2a, which puts the greatest restriction on conversions of farm land (no conversion outside of 100-foot stream buffers), reveals some of the opportunity costs of such policies. That scenario generated the third highest total cost among the scenarios ($374 million) and the lowest ecosystem service co-benefits ($4 million) (Table 8).

Two scenarios with an intermediate level of farm land conversion demonstrate the effect of combining conversion restrictions with policies aimed at managing the higher level of uncertainty associated with BMPs for controlling NPS emissions relative to PS controls. Scenarios 2c and 5 represent cases in which conversion of agricultural land is restricted to 10 percent of the agricultural land by land-river segment and either a 1:1 or 2:1 PS-NPS trading or offset ratio is applied to manage uncertainty. The results from scenario 2c resemble the base case. Monetized ecosystem services are $12 million in scenario 1 and $10 million in scenario 2c while water storage potential increases from 3,000 to 41,000 acre-feet and the cost rises modestly (compared to other scenarios) from $23 million to $90 million. In contrast, the results from scenario 5 show

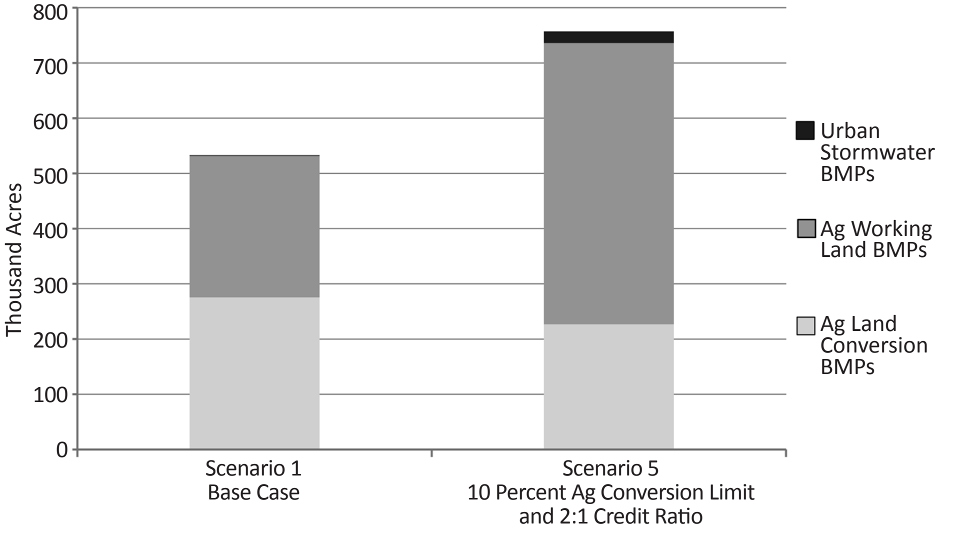

Figure 9. NPS Acres by Practice Type under the Unrestricted Base Case and Multiple Restrictions

that a 2:1 ratio raises the total cost to $262 million (a 1,000 percent increase over the base case) while ecosystem service co-benefits decrease only slightly to $11 million (Table 7). The model used a wider variety of practices to meet targets in scenario 5 (Figure 9) but could not identify a solution that would meet the P and sediment targets when both a 10% land conversion restriction and a 2:1 trading ratio were applied simultaneously.

The P and sediment targets are not met in scenario 5 for several reasons, including the issue previously described that the model does not include the full suite of available urban and agricultural practices and, under an imposed price cap, does not allow some of the most expensive SW practices to be used. In addition, our assumptions limited apparent capacity in ways that are distinct from the process currently used for Chesapeake Bay. In the existing bay TMDL, nutrient and sediment reductions are directly allocated to NPS sectors and no offset or trading ratios are applied. In our framework, we assumed that all allocations go to regulated PS sectors. When we apply trading or offset ratios, we apply them to all reductions generated by the NPS sector. Consequently, ratios that exceeded 1:1 reduced the apparent capacity of the NPS sectors. The model thus reveals that a decision to ignore some sources of uncertainty in NPS performance has the effect of making it easier to meet targets.

Our results further suggest that high PS-NPS trading or offset ratios (used either in assigning allocations to source sectors or in trading or offset programs), by themselves, promote ecosystem service co-benefits but only when sufficient offsets are available and realistically could be used. However, the high ratios also create pressure to convert more agricultural land to permanent vegetation since that produces more reduction per acre when available acres are limiting. In contrast, a 1:1 ratio reduces the cost of compliance and pressure to convert agricultural land but is not as precautionary about ensuring water-quality outcomes. Our work reveals that the uncertainty in water-quality outcomes that is generated by offsets, trading, and NPS sector allocations is likely to be offset, at least partially, by generation of ecosystem service co-benefits. Further, if trading ratios that exceed 1:1 do not prevent trading, they will likely promote production of a higher level of co-benefits in addition to greater assurance of the water-quality outcomes.

Sensitivity of the Results to Carbon Values

Because of the high degree of uncertainty associated with estimating the social value of carbon sequestration, we tested alternative values to evaluate their effect on the results. Changing the social value of carbon would not change the mix of practices used in any of the least-cost solutions. However, as shown in Figure 10, it would affect the least-net-cost solutions by changing both the direct cost and the net cost. For the base case (scenario 1), going from the lowest carbon value to the highest (from $7 to $26 per ton of CO_2e) causes the total cost of the least-net-cost solution to increase by $59 million per year while the ecosystem co-benefits increase by $145 million. As a result, net cost becomes $74 million in net returns, as shown by the negative value for net cost. When a 2:1 credit ratio is applied (scenario 2a), a similar pattern emerges but the magnitude of the change is somewhat smaller. In that case, going from the lowest carbon value to the highest value increases the total cost by $28 million and the ecosystem co-benefits by $126 million. Overall, a lower carbon value reduces the value of the co-benefits in both least-cost and least-net-cost

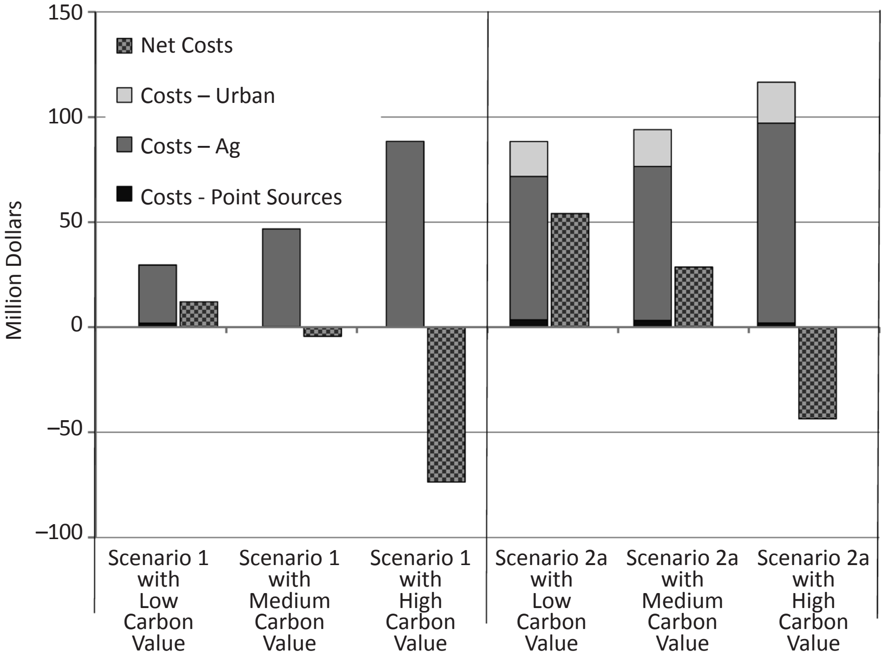

Figure 10. The Effect of Carbon Value on the Least-Net-Cost Solution for the Base Case and Scenarios Involving a 2:1 Ratio for Trading and Offsets

scenarios and, in least-net-cost scenarios, reduces direct costs and increases net costs.

Conclusions

A variety of legal, political, and social factors associated with TMDL design and implementation choices restricts the possibility and/or desirability of achieving the least-cost solution. However, some constraints can be created unintentionally if the cost and benefit implications of decisions are not recognized. Therefore, this analysis explores how various policy choices and system conditions can affect outcomes in terms of the aggregate cost of TMDL compliance and generation of ecosystem service co-benefits.

Our optimization analysis suggests that the aggregate compliance cost for the Chesapeake Bay TMDL could be substantially reduced by generating the majority of nutrient and sediment reductions from NPS management practices and by creating opportunities for ecosystem service co-benefits to offset costs. However, using the least-cost combination of practices, which was predominantly a 50:50 mix of working land and agricultural conversion options, may conflict with a desire to promote social and economic goals associated with maintaining agricultural production at the current level. Further, a heavy reliance on NPS options, either in initial sector allocations or through use of trading and offset programs, may reduce certainty of water-quality outcomes.

The analysis thus reveals two key tradeoffs created when TMDL policies influence whether reductions are generated by PS or NPS sectors. A smaller compliance cost and greater ecosystem service co-benefit often must be weighed against (i) the implications of reduced agricultural production and (ii) less certainty regarding water-quality outcomes. Although neither tradeoff is particularly desirable, each represents an opportunity for managing the cost of the TMDL. The analysis also shows that direct allocations to the SW sector or restrictions on agricultural conversion increase the marginal costs near the nutrient and sediment caps, which suggests that costs under these policies may be sensitive to small changes in the TMDL cap. For example, in scenario 2a, 85 percent of the compliance cost is attributable to the last 20 percent of effort invested.

Under certain policies, TMDLs have the potential to produce both multiple ecosystem service co-benefits and water-quality-derived benefits at the same time for no additional cost. However, targeting effort to minimize the net cost of jointly producing water quality and ecosystem service co-benefits almost always results in higher direct costs. These higher costs might be justified if they were less than the cost of pursuing bay restoration goals for both terrestrial and aquatic ecosystem services through separate programs but we have not demonstrated that case. Instead, our work demonstrates that the reduced certainty of water-quality outcomes that may result from using trading, offsets, and NPS allocations is likely to be offset, at least to some degree, by the production of a range of other ecosystem service benefits.

We do not mean to suggest that the least-cost solutions represented here are socially optimal. Many questions remain. For example, the sensitivity of the type of agricultural BMP selected (working land or conversion) to the opportunity cost of land suggests that a comprehensive social value of agricultural land is needed to more effectively weigh the tradeoffs of reducing nutrients by forgoing agricultural production. Conversion of agricultural land is likely to cause the loss of a wide variety of cultural, aesthetic and hunting-derived benefits that were not included in our model. Further, if BMP efficiency is being overestimated or the transaction costs of NPS engagement (including the costs of PS-NPS trading) are being underestimated, the benefits of using a high proportion of NPS pollution controls to meet the TMDL diminish substantially because insufficient acreage is available to achieve the TMDL. As a result, we conclude that key elements for achieving cost-efficiency in TMDL programs are reducing uncertainty associated with the performance of NPS management practices and decreasing the cost of monitoring and verifying performance.

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