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Establishing Trading Ratios for Point – Non-Point Source Water Quality Trades:

Can we capture environmental variability without breaking the bank?

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Establishing Trading Ratios for Point – Non-Point Source Water Quality Trades: Can We Capture Environmental Variability without Breaking the Bank?

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Abstract

This report evaluates and demonstrates technical methods of comparing point source (PS) and non-point source (NPS) nutrient discharges that can be used for developing credit scoring methods for a water quality trading system. We evaluate the state of the science for evaluating hydrologic functions within watersheds and for linking those physical functions to water quality and environmental outcomes. Many factors influence the links between cause and effect, and therefore models are used to consider how heterogeneity in biophysical characteristics and built infrastructure may influence the effectiveness of nutrient management practices at different locations within the watershed. We demonstrate these techniques using a case study of the Patuxent River in Maryland, USA and a suite of models developed for that river by an interdisciplinary team of ecologists, hydrologists and physical scientists who have been working together on Patuxent River water quality problems for many years.

We find that reasonably appropriate data and models are available to inform a scoring system for nutrient credit trading in the Patuxent watershed. The system we develop for scoring PS to NPS trades is able to capture some important aspects of landscape heterogeneity and importance of the location of the best management practice (BMP) to measure nutrient reduction effectiveness. However, the models used to inform the scoring do not provide a means to assess potential error of scores and do not capture variability of performance with weather conditions or other temporal factors. The most poorly constrained data appear to be those for nutrient trapping performance by specific BMP. The success of trading programs in achieving environmental goals could be limited by any systematic biases in estimations of BMP effectiveness generated by the process-based models that provided the most comprehensive information and a logical method for scoring nutrient trades (e.g., Chesapeake Bay Program watershed model).

1. Introduction

1.1 Pollutant Credit Trading Fundamentals

Pollution credit trading programs are underway or are being developed in over 80 regions in the US (Breetz et al. 2004), and they are being widely promoted as an innovative market-based approach to meeting water quality standards earlier and at lower cost (USEPA 2003a). Tighter clean water standards are driving the interest in trading programs as waterways that are designated as "impaired" under the Clean Water Act are being required to develop Total Maximum Daily Loads (TMDLs) and implement plans to reduce pollution. TMDLs are a mandated cap on the amount of nutrients that may be discharged to a given waterbody, where the total nutrient cap is allocated among different sources, such as point and non-point sources. Discharge permits are used to enforce compliance with point source discharge allowances.

Proposed nutrient trading programs use the well-established "cap and trade" system developed by resource economists to improve efficiency of meeting environmental regulations (Baumol and Oates 1988). The system has been successfully implemented in several cases, most famously for controlling the air pollutant SO_2 (Stavins 1998). In a cap and trade system, regulators set regional caps on pollution emissions and assign pollution allowances to emitters (e.g., power plants in the case of air pollution or waste-water treatment plants (WWTPs) in the case of TMDLs). The formulas for determining allowances are usually contentious and include evaluating plant capacity, previous investments in pollution reduction, and other factors.

In theory, a system of tradeable credits creates incentives for emitters that have low costs of compliance to reduce emissions below their allowances and sell the credits to emitters who have high emission reduction cost. Emitters with high costs of compliance have an incentive to buy credits from the emitters that can reduce their emissions more cost-effectively. One key to the successful development of a credit market, therefore, is heterogeneity of discharge reduction costs among emitters. Where there is relatively heterogeneous discharge, the trading program allows the overall discharge cap to be met at lower cost to emitters than would be possible if a fixed reduction requirement was applied to all emitters.

To take advantage of perceived heterogeneity in nutrient discharge reduction costs, many jurisdictions are developing nutrient trading systems in which trading is intended to occur between point source (PS) dischargers as buyers, which are primarily WWTPs who have high costs of reducing nutrient dischargers, and non-point source (NPS) dischargers as sellers, which are typically owners of agricultural or urban lands with relatively low discharge reduction costs. In implementing a system that allows credits to be traded between point and non-point sources, many questions arise regarding how well we can characterize the effectiveness of nutrient reduction activities and how to trade off the certainty of nutrient releases from WWTPs with the uncertainty of nutrient discharge reductions from non-point sources. NPS emitters may be far from the stream, and emissions may vary seasonally and annually depending on crop choices, weather patterns, installation quality of the management practice and the degree to which the practice is maintained. Conversely, point sources often discharge directly into streams and generate more consistent emissions with high certainty. While new regulations are providing the impetus to implement trading programs that allow or even encourage point-non-point trading, the details of how to implement them to achieve water quality benefits are not well developed.

1.1.1 Evaluating trades

In evaluating any trades or offsets, the activities designed to sequester or remove nutrients must be assigned a number of credits proportional to the amount of nutrients that are prevented from reaching the waterway, sometimes at a specific point in the waterway such as at the mouth of a river or near some sensitive resource. A large number of factors go into such calculations include the expected efficiency of the activity, the form of the nutrient being kept out of the waterway, location within the watershed, timing of maximum discharge without the activity, and uncertainty of performance. Taking account of these various “scoring” considerations usually results in trading ratios that determine how many pounds of reduction from a non-point source at a particular time at a particular place is equivalent to a pound of reduction at a point source.

Credit trading systems differ in their approach to scoring, and it is not uncommon to find a 1:1 trading ratio being used for all trades suggesting that nutrient discharge gains and losses or resulting gains and losses in water quality are equivalent regardless of their PS or NPS origins (Environomics 1999, Breetz et al. 2004). Since trading goals vary widely, each program is tailored to the case at hand. Different pollutants create different responses in the waterways, and therefore, rules must be adapted to reflect details of the situation.

In this paper we focus on the issues related to trading nitrogen and phosphorus between PS and NPS dischargers. The differences in characteristics of emissions between PS and NPS emitters means that achieving water quality goals depends on developing equivalency measures for nutrient credits that goes beyond comparing pounds of nutrient reduction at the source of the discharge. Non-point sources create nutrient credits by implementing changes in land use practices or employing technology. Best management practices (BMPs) comprise many structural and non-structural methods including: limiting tillage of cropland, planting riparian buffers, building storm water holding ponds, among many other options (USDA ARS 1994, Center for Watershed Protection 2000, US EPA 2004). In the case of NPS discharges, the amount of nutrient discharged to the stream is difficult to measure accurately and varies with site characteristics and location in the watershed. In the case of PS dischargers, the nutrients are discharged directly to the stream or river and can be accurately and easily measured. In both cases, the affect that nutrients have on the ecology of the stream may vary with the season and with weather patterns, but the seasonal patterns of emissions usually differs between the two types of nutrient sources.

This report evaluates and demonstrates technical aspects of comparing PS and NPS nutrient discharges that can be used for developing credit scoring methods for a water quality trading system. We evaluate the state of the science for evaluating hydrologic functions within watersheds and for linking those physical functions to water quality and environmental outcomes. Many factors influence the links between cause and effect, and therefore models are used to consider how heterogeneity in biophysical characteristics and built infrastructure may influence the effectiveness of nutrient reductions at different locations within the watershed. We demonstrate these techniques using a case study of the Patuxent River in Maryland, USA and a suite of models developed for that river by a team of researchers.

1.1.2 The baseline issue

The particular activities that can be used to earn credits are those that go beyond activities required by law or other baseline requirement that may be established. If activities that are used to generate credits do not represent some new action, then they do not result in any reduction in nutrients to the waterway beyond "business as usual" reductions. To deal with the fact that many agricultural lands or developed lands may not be in compliance with existing laws, some jurisdictions are proposing to limit the amount of credit that can be claimed by BMPs. For example, in Virginia, it has been proposed that credit-seekers who are implementing riparian buffers must meet a baseline requirement for a minimal buffer width (35') and only get credit for the width of the buffer that goes beyond this Natural Resources Conservation Service (NRCS) standard. Pennsylvania, on the other hand, appears to be moving forward with a program that would not require that farmers exceed any baseline activity. Under that program, farmers receiving money to implement BMPs through various USDA programs may be able to sell the credits associated with those same activities. Many consider this "double-dipping" and point out that in such cases, no new activity has been generated to offset the additional releases of nitrogen that will be produced as a result of the credit sale (Blankenship 2007, Feb).

1.1.3 Regulatory cost vs. environmental effectiveness

Accounting for site-specific heterogeneity in delivery of nutrients and effectiveness of BMPs adds to the transaction costs of trading and could result in an unwieldy trading system that would realize no trades. The goal in developing scoring criteria, therefore, is to strive for balance between keeping transaction costs of trading low enough to facilitate trades while still ensuring trades are evaluated in terms of their relative environmental benefits. Achieving consensus on how much detail to include in scoring systems is the subject of much debate at present.

1.1.3.1 Previous examples of intermediate complexity systems: Conservation Reserve Enhancement Program (CREP)

The issues related to finding a balance between complexity and effectiveness have been grappled with previously in other programs aimed at nutrient reductions. An example is the Conservation Reserve Program (CRP) and the related Conservation Reserve Enhancement Program (CREP), managed jointly by USDA and participating states. In these programs, farms are enrolled to take land out of production to create environmental benefits. Farmland being considered for enrollment is rated according to cost-effectiveness for achieving environmental goals. The system ranks parcels according to the intended cover of the land once retired and uses an environmental benefit index (EBI) based on site and location characteristics to quantify benefits. The index is based primarily on the expected environmental enhancements from reduced erosion, water quality improvements, and provision of wildlife habitat. The specific indicators that make up the EBI include proximity of the site to water, adjacency to protected land, erodability and other factors. States may target particular locations (e.g., high quality ecosystems) by designating Conservation Priority Areas that get factored into scoring.

1.1.4 The basis for credit scoring: Production functions of ecosystem services

It is easy to lose site of the fact that nutrient credit trading is ostensibly a method to achieve an environmental benefit at lower cost. The emissions or discharge cap that is used to determine the basis of trading creates the environmental benefits but ensuring equivalent trades is what allows the cap to represent a meaningful reduction in pollution. The accounting needed to compare nutrient reductions can be thought of as similar to the process of developing an ecological "production function." A production function is an economic tool used to quantify how much and what type of inputs are needed to create a certain quantity of outputs. The function usually incorporates substitutability of inputs to show how producers might trade off some inputs based on availability or cost.

A nutrient credit scoring system is a type of environmental production function in which the environmental benefits produced at a given site are predicted as a function of site and landscape characteristics. The inputs to production are typically site features (e.g., vegetation, hydrologic characteristics) and location characteristics (position down-gradient from nutrient sources, position near a stream). These features combine to produce an environmental function (e.g., fish habitat). Site factors are critical to understanding the ability of a site to produce an ecological function such as nutrient removal, however, as typically used, they do not fully describe: 1) the opportunity of the site to provide that function (e.g., will a particular species use the site and thus benefit from water quality improvements) or whether that function rises to the level of an environmental service that people will value (e.g., fishing or swimming opportunity).

To evaluate a production function that considers both the ability to produce a function and a service people will value requires an evaluation of the level of ecological function (nutrient removal) and complementary factors that control its value for the intended service. Characteristics such as how a site fits into a network of working and natural lands, connections to infrastructure, and accessibility are necessary to understand whether an area will produce a valued service. For example, a function such as erosion control may be considered more valuable where it provides the service of protecting sensitive recreational fish spawning areas (i.e., coarse sediment areas) within a receiving stream. In addition to landscape aspects, temporal aspects may need to be considered such as whether the conservation practice protects resources during vulnerable periods such as spawning seasons.

Although previous indicator systems such as CRP have considered some of these location elements in their environmental benefit index, only limited use has been made of formulas that quantify the relative importance of various inputs to production. In other words, we do not typically have information on how much weight to apply to different location characteristics. Instead, we see simple indicator systems that treat all indicators with equal importance. The quantification of how characteristics contribute to the value of services is extremely difficult to produce, but examples do exist (Wainger et al. 2007, Popp et al. 2001, Wainger et al. 2001). The nutrient credit trading market is built on the assumption that any reduction in pollution is equivalent, however the social benefits of a program could be improved by considering the additional payoffs of environmental services that are produced through BMPs, particularly those involving planting natural vegetation.

1.2 Organization of report

In this report, we characterize the types of data and information that are required to evaluate credit trades for equivalency. We discuss what might be required to accurately evaluate nutrient reductions and set trading ratios for different activities and discuss trade-offs required to develop a scoring system that is not overly burdensome to implement. Using a case study, we demonstrate how to apply specific models to quantify the relationship between site and location characteristics and nitrogen removal efficiencies of BMPs. A similar approach was used to evaluate phosphorus, but that information is not presented here. We choose among available data and models to develop a system of credit accounting and discuss the state of the science and the relative uncertainty of such calculations.

2. Case Study Setting

The Patuxent River watershed lies on the western shore of the Chesapeake Bay and is one of the nine largest rivers that drain to the Bay. The Patuxent River has non-tidal and tidal portions and grades from fresh to brackish water along its length. The river crosses two geophysical provinces, the Piedmont in the headwaters and the Coastal Plain over the majority of the watershed. The configuration of the river and marshes affects nutrient retention along the downstream gradient. The upper portion of estuary (river kilometer (rkm) 40-95) is narrow (50-300 m), very turbid and vertically well-mixed with an average depth of 1.1 meter. Extensive tidal marshes flank this portion of estuary. The Lower estuary (rkm 40 to mouth) is much wider (1-5 km), deeper (5.4 m), clearer, and seasonally stratified.

The Patuxent River watershed drains about 900 square miles of land that includes parts of seven counties, all of which have rapidly growing populations (Table 1). Land use is a mix of developed and undeveloped uses in all counties, but the estuarine portion of the basin retains a greater proportion of forested lands than the upper basin. Farmland is distributed throughout the basin and comprises 21% of the acreage (USGS Mid Atlantic Integrated Assessment (MAIA)), and farm size is generally smaller on average than for the US as a whole (National Agricultural Statistical Survey (NASS)). Both agricultural land and forested lands are on the decline in all parts of the watershed (Taylor and Acevedo 2006).

Table 1. Patuxent Counties Population Trends (US Census)

County	1990 population	2004 population	% change
Anne Arundel	427,239	508,356	19.0%
Calvert	51,372	86,293	68.0%
Charles	101,154	135,702	34.2%
Howard	187,328	266,532	42.3%
Montgomery	757,027	921,631	21.7%
Prince George's	729,268	841,642	15.4%
St Mary's	75,974	94,950	25.0%
Total	2,329,362	2,693,674	15.6%

2.1 Regulatory Setting and Management Goals

The Patuxent River is marked by the same high nutrient concentrations that have degraded water quality, aesthetics and habitat throughout the Chesapeake Bay estuary. Water quality is a concern since it affects drinking water quality, commercial and recreational fisheries, water clarity, and health and safety of those that come in direct contact with water. From a regulatory perspective, one of the primary concerns is that poor water quality enhances the size and duration of an anoxic-hypoxic zone that forms seasonally in the Chesapeake Bay mainstem and in other tributaries such as the lower Patuxent estuary. These low oxygen zones are thought to contribute to stress of benthic and pelagic organisms, and potentially reduce recruitment of certain fish (Breitburg 1994, Roman et al. 1993).

Efforts have been underway for decades to reduce nutrients in the Patuxent River and all tributaries of the Chesapeake Bay. Nutrient reductions in the Patuxent have been significant, but

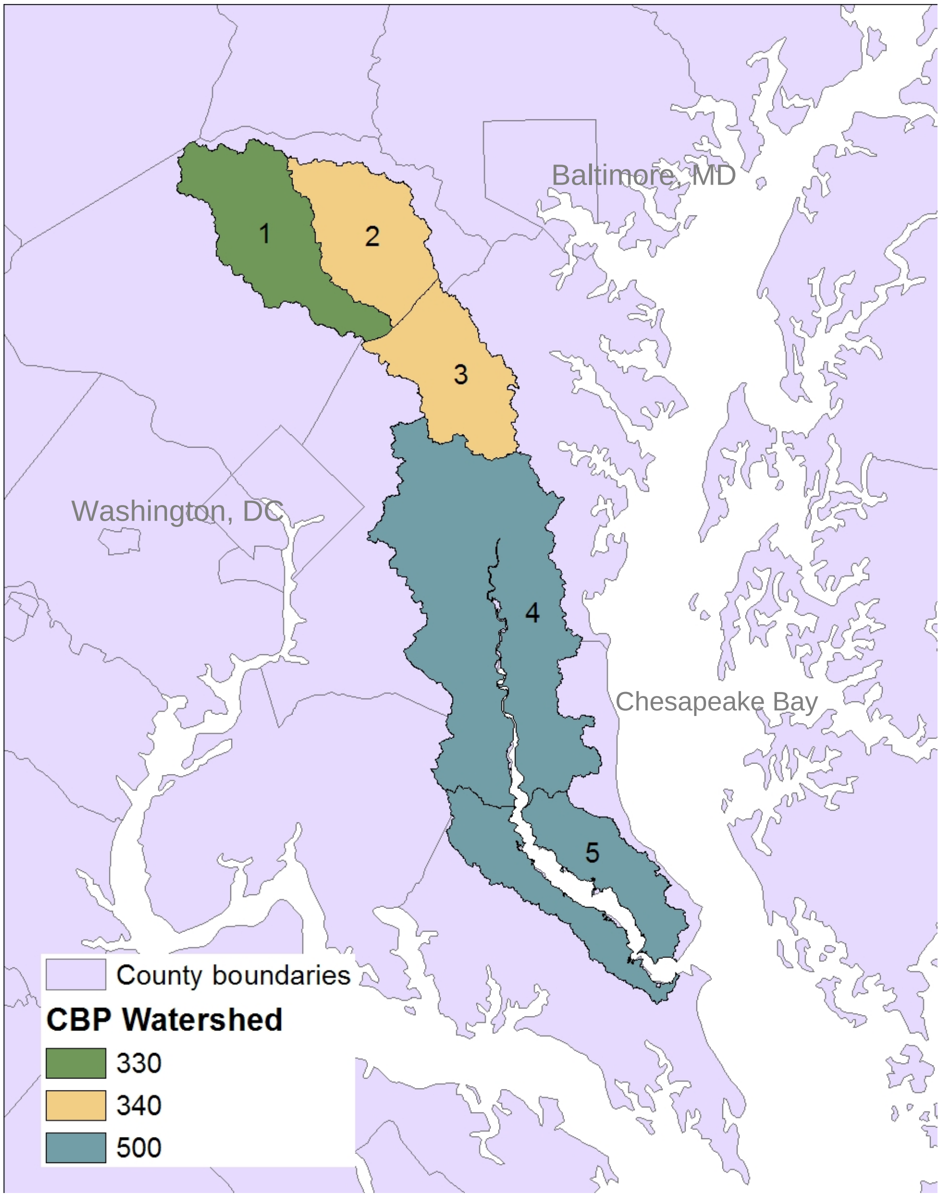

Figure 1. Patuxent watershed sub-basins
Numbered zones within the watershed show basins with different nutrient delivery characteristics based on research by Jordan and Weller. Zones 1 and 2 are Piedmont watersheds and Zones 3, 4 and 5 are Coastal Plain Watersheds. Zone 4 drains to the middle estuary and Zone 5 to the lower estuary. The colored sub-basins represent the 3 subwatersheds distinguished by the Chesapeake Bay Program (CBP) watershed model, and each has a distinct basin delivery rate.

they did not reach the original goals (set as part of the Chesapeake Bay agreement of 1987) of slashing nutrients by 40% by the year 2000. Failure to meet goals was due, in large part, to population gains within the watershed that countered some of the reductions in discharge. Most of the nutrient reductions to date have been achieved by improving treatment of outflow at major WWTPs. Over the period of 1990-1993, the major treatment plants on the Patuxent River were upgraded to include biological nutrient removal, and new upgrades at WWTPs are ongoing in Maryland and are being funded through a "flush tax," implemented in 2005, on all users of municipal sewer systems. A similar tax was levied on septic owners.

A new Chesapeake Bay Agreement was signed in 2000, and subsequently new nutrient emission targets were developed and allocated among the major tributaries of the Bay. The nutrient goals were designed to protect the Chesapeake Bay's living resources and enable the Bay to be removed from the U.S. Environmental Protection Agency's list of impaired waters by 2010. A 2003 agreement with EPA's Chesapeake Bay Program (CBP) set nutrient allocations for all major tributaries and states in the watershed prescribing the maximum amount of nitrogen, phosphorus, and sediment releases allowed into the Chesapeake to meet Bay water quality standards (US EPA 2003b).

Based on the new agreement, Maryland established new nitrogen and phosphorus targets in October 2002. The new targets will reduce nitrogen discharges to no more than 38 million pounds and cap the discharge of phosphorus at 3.1 million pounds, requiring reductions of 19 million pounds of nitrogen (about 30% reduction) and 700,000 pounds of phosphorus (about an 18% reduction). Maryland's plan to achieve the reductions involves allocating pollution caps to each tributary and undertaking activities that will reduce nitrogen emissions and prevent future increases in emissions.

These tributary allocations effectively create the cap that would be used in a cap and trade system. However, since Maryland is funding new WWTP upgrades through taxes, little incentive currently exists to create a trading system. However, trading remains a possibility for generating the required offsets for all new major WWTPs.

Other regulation with a major potential to affect water quality trading in Maryland is the Water Quality Improvement Act (WQIA) enacted in 1998 by the Maryland General Assembly. The statute required that all agricultural operations with gross income greater than $2,500 or sales of more than 8 animal units (one animal unit is roughly equal to 1,000 pounds live weight) develop and implement a nitrogen-based and phosphorus-based nutrient management plan. By mid-2005, all regulated farms were required to have implemented both types of plans. A variety of other regulations affect NPS polluters (agriculture, urban, marinas, etc.) as described on the Maryland DNR website (http://www.dnr.state.md.us/bay/czm/nps/whatisnps.html). This regulation potentially limits some of the on-farm activities for which farmers can claim and sell nutrient credits.

The Chesapeake Bay partners hope to voluntarily achieve the new nutrient and sediment reductions by 2010 in order to remove Bay and tributary waters from EPA's listing of impaired waterways and thus avoid mandatory TMDLs in these waters. However, EPA has predicted that states will fail to meet established goals. EPA predicts the states will achieve just 59 percent of their nitrogen reduction goals, 74 percent of their phosphorus goals and 74 percent of their sediment goals by 2010 (Blankenship 2006a). Therefore, EPA will be required to impose a TMDL plan by 2011 (Blankenship 2006b). A TMDL plan will require Maryland and other states to meet their current voluntary emission reduction goals.

3. Methods

We use the Patuxent River Watershed in Maryland, USA, as a case study for evaluating the knowledge and data available to evaluate equivalency of (or to "score") nutrient trades. This watershed, which has both freshwater and estuarine segments, is part of the Chesapeake Bay watershed and has undergone extensive study by many researchers. The watershed is completely contained within one state and therefore offers a simplified governmental structure for conceptualizing scoring policy. All in all, the watershed provides the potential to examine a wide range of options for scoring and evaluating trades.

To generate some of the information needed for scoring trades, we worked with a group of modelers who had previously developed a set of models to characterize nutrient sources and sinks and potential environmental benefits of nutrient reductions in the Patuxent watershed both on the land and in the estuary (Breitburg et al. 2003). We supplemented those models with other research including compiled databases, output from models, and expertise of field scientists working within the system in order to characterize the potential for trading.

3.1 General Issues in Scoring Nutrient Trades

3.1.1 Establishing the point of equivalency

The first major consideration in developing a nutrient trading scoring system is determining the environmental endpoint of interest and selecting a location or locations at which the endpoint is sensitive to nutrient concentrations. Given the need to meet TMDL regulations, the endpoint of interest for most states will simply be an outflow point of a surface water body, and the goal will be to stay below an established nutrient concentration threshold (instantaneous and/or time-averaged). In these cases, the equivalency of nutrient reductions is judged in terms of how any activity reduces the amount of nutrients reaching the outflow point such as the mouth of a river. In the Chesapeake Bay, the point of equivalency is a location above the deep channel in the mid-Bay where hypoxia and anoxia are a major concern because this is the region defined as "impaired" under the Clean Water Act.

The point of equivalency essentially determines where the environmental benefits will be judged. Sites will be scored differently depending on the definition of this point. All else equal, sites that are farther away from the point of equivalency will receive fewer credits for implementing the same BMP. Sites that are downstream of the point of equivalency may receive no credits at all. However, in estuaries nutrients can flow both up and downstream, so it could be inefficient to use criteria based on a simplified unidirectional flow.

Once the point of equivalency has been established, the relative amount of nutrients delivered from any location in the watershed to that point can be evaluated. The characteristics that determine delivery to a given point depend on many conditions along the flowpath but perhaps most significantly are 1) whether water travels above ground or below ground to reach the stream; and 2) the distance from the point where water reaches the stream (or other surface water body) and the point of equivalency (e.g., the river mouth) and the size of stream between those two points. Smaller streams are generally considered to do more attenuation than large streams although such relationships have not been clearly established (Seitzinger et al. 2002). Therefore, deciding where to judge the "equivalency" or comparability of pounds of nutrients is critical to establishing scoring criteria.

3.1.2 Understanding net movement of nutrients from watersheds

As part of establishing the basis for nutrient credit trading and the context of evaluating particular trades, it is useful to have an accounting of the sources and sinks of nutrients and sediments within the watershed. This baseline assessment can help in determining appropriate trading areas and determining the level of the nutrient cap that will generate environmental benefits. Thus, such models are useful for setting up a trading system and providing background necessary for accurate scoring of BMP effectiveness.

A variety of models and data are available to characterize entire watersheds and to assist governments in evaluating the potential for trading and for setting nutrient emission caps to realize environmental goals and contribute to scoring BMPs at particular sites. The type of model used will vary by the type of pollution of interest: phosphorus, nitrogen, sediment or other (e.g., toxin or pathogen). For purposes of this discussion, we will focus on phosphorus and nitrogen modeling, since these are the pollutants of interest in the Patuxent River, the case study location. The particular form of nutrient will also influence the techniques used since soluble forms will move with water while bound forms will move with sediment.

Nutrient trading is primarily focused on controlling nutrients arriving via freshwater flow paths (i.e., overland flow), and has had an emphasis on reducing concentrations in surface water. However, nitrogen reaches estuaries through three main pathways: 1) atmospheric deposition, 2) freshwater inputs (surface and groundwater) and 3) inter or intra-estuarine transport. Phosphorus is not deposited from the atmosphere but is moved in freshwater, similar to nitrogen, and is moved with sediment that may be suspended in surface water. To understand the potential of nutrient trading to change nitrogen concentration in estuaries, therefore, requires an accounting of all three sources. Recently the idea of cross-media trading between air emitters and water emitters has been discussed, but this is a new and untested idea (USDA 2006).

The quantity of nitrogen deposited from the air varies spatially depending on the presence of local and distant upwind sources. Atmospheric effects may not be important determinants of nutrients in most estuaries (Mayer et al. 2002) but can be locally important. While nitrogen naturally makes up the bulk of our atmosphere and some atmospheric deposition of nitrogen occurs naturally, enhanced deposition of nitrogen atmospheric deposition occurs due to increased inputs of nitrogen from the combustion of fossil fuels, fertilizer application and high-intensity animal operations (USGS 2000). Nitrogen deposited from the air, by plants, or as a result of human activities is available to run off land through surface and groundwater flow. Nitrogen is present on vegetated and developed land alike, but it is largely the nitrogen on developed land that is considered "controllable", meaning that actions might be taken to reduce nitrogen availability and runoff. Nitrogen also fluxes between sediments and surface water and between different surface water bodies. Since estuaries receive both fresh and salt-water inputs, nutrients can come from the land-side or the ocean-side. The availability of nutrients is determined by the rate of atmospheric deposition and land use/management practices, and the three pathways, atmospheric, freshwater, and saltwater movement, determine how available nitrogen moves between locations.

Nutrients have the potential to be trapped or transformed as they move along flowpaths. The potential for nutrients to be taken up by organisms, trapped in sediment or soil sinks, or transformed to different chemical forms and states is a function of the types of systems through which the nutrients move. For example, nitrogen in water moving through a plant root zone has the potential to be taken up by those plants, while nitrogen in anoxic muds has the potential to be transformed into gas. Understanding the relative importance of each component controlling availability and movement is important for understanding how well nitrogen can be controlled through any particular activity.

The Patuxent River is typical of many riverine-type estuaries that exchange nutrients with Bays. Nutrients are exchanged between the Patuxent estuary and the Chesapeake Bay estuary through normal estuarine circulation. The Patuxent is considered a seasonally-stratified estuary and can generally be characterized in terms of the typical 2-layer water circulation pattern that develops when fresh water flows into salt water. In the typical case, fresher water coming from the river flows downstream along a top (shallow) layer while saltier heavier water from the Bay flows upstream along a bottom (deep) layer. The circulation in the Patuxent is sometimes more complex than this, but 2-layer circulation is dominant.

Models and data are available to create general estimates of nutrient availability and understand the relative magnitude of the three pathways of movement. However, the amount of data and understanding able to support the quantification of such fluxes (i.e., the specific parameters used in the models) will vary by watershed. For watersheds nested within the Chesapeake Bay watershed, a great deal of measurement and modeling has been conducted and it can inform such analyses, although not all site-specific heterogeneity is likely being captured, even in this well-studied system. In other locations, data may be sparse and those wishing to quantify trades may need to rely on general tools. Here we review the national and regional data sources and tools available to those wishing to score trades of nitrogen and phosphorus and discuss how we made estimates for the Patuxent watershed.

3.1.3 Understanding differences between nutrient sources: types of nutrients and timing of outputs

To ensure that trades among different types of nutrient discharges have a neutral effect on the environment, it has been proposed that trades and trading ratios take into account the form of the nutrient (e.g., bio-available vs. non bio-available forms) and the timing of discharges. For example, the type of phosphorus emitted from cropland is significantly different from the phosphorus emitted by WWTPs. Much of the total phosphorus runoff from cropland is bound to sediment and is not immediately available for uptake by plants while the phosphorus from WWTPs is soluble and readily available for uptake. If the target of the nutrient cap is to improve estuarine water quality, the type of nutrient and the season in which it is released may control the environmental impact. Evidence suggests that in the Chesapeake Bay, nitrogen and phosphorus are limiting nutrients at different times of year, and that nutrient releases in the spring may have a bigger impact on summer low oxygen levels in the deep portions than nutrients released at other times.

The sensitivity of the estuary to the timing of nutrient runoff means that trading systems may want to account for the seasonal nature of agricultural sources relative to the more consistent discharges from WWTPs. The annual hydrological cycle also has an effect on the potential impact of nutrient runoff at different times of year. Seasonal and event-driven changes in streamflow can influence the dilution rate of nutrients and the environmental effects.

EPA has proposed that a common metric be used to track nutrient trades overall, but that different forms of nutrients or other chemicals be considered when trading ratios are determined (EPA 2004). Such details of accounting, which are important to controlling the environmental impacts, are also the types of details that begin to make the transaction costs of trading more expensive and difficult. The movement of nutrients and their effect on the environment depends on the form of the chemical being evaluated.

3.2 Existing Models and Data

More specific to nutrient trading are calculations designed to assess how much of a nutrient is likely to be prevented from reaching the point of equivalency due to the implementation of a particular management practice at a particular location. Only some of the available modeling tools can be directly used to calculate the credits generated by a certain activity. However, it is important to keep in mind that watershed assessment tools may be needed to assess characteristics of particular basins (i.e., "basin delivery factors") that are then used to assign credits to particular activities within those watersheds. We demonstrate how both types of tools are used in Section 3.3.

A variety of web tools have been developed to assist potential traders in making calculations about how basins emit and store nutrients with and without implementation of BMPs. Some of the available and well-used tools are shown below.

3.2.1 Widely used tools for making basin-wide assessments:

- **The Long-Term Hydrologic Impact Assessment (L-THIA)** model is an online tool to assess the water quality impacts of land use change for any basin. http://www.ecn.purdue.edu/runoff/
- **SPAtially Referenced Regressions on Watershed attributes (SPARROW)** model. A method for regional interpretation of water-quality monitoring data. New versions are being developed for the Upper Midwest and entire Mississippi River Basins. USGS. (http://water.usgs.gov/nawqa/sparrow/)
- **Non-point Source Pollution and Erosion Comparison Tool (N-SPECT)** is designed to predict potential water-quality impacts from non-point source pollution and erosion and gives nutrient concentrations in receiving waters. NOAA. http://www.csc.noaa.gov/crs/cwq/nspect.html
- **Soil and Water Assessment Tool (SWAT)** was developed to quantify the impact of land management practices in large watersheds in terms of water quantity and water quality by specific nutrient categories (e.g., organic N). http://www.brc.tamus.edu/swat/ USDA ARS Temple, TX.
- **Better Assessment Science Integrating point and Non-point Sources (BASINS)** incorporates the SWAT model and provides additional tools to evaluate water quality using an integrated analysis of point and non-point sources.

- **Agricultural Non-Point Source Pollution model (AGNPS)** The Agricultural Research Service developed this program designed to evaluate non-point source pollution from agricultural watersheds. Outputs related to soluble nitrogen and phosphorus for surface water and infiltration are provided. Sediment yield and runoff are calculated, and sediment transported nitrogen and phosphorus are determined. USDA http://www.il.nrcs.usda.gov/technical/waterq/index.html

3.2.2 Tools for making field-scale assessments:

- **Nutrient Net** (http://www.nutrientnet.org/) is an online calculator available for selected watersheds that is designed to calculate nutrient credits produced by particular activities at particular locations. This model is further described below.
- **Internet based Urban Best Management Practices and Cost Analysis** shows the effectiveness and costs of various structural BMPs. Purdue University http://pasture.ecn.purdue.edu/~jychoi/ubmp0/.
- **Nitrate Leaching and Economic Analysis Package** (NLEAP) is field scale assessment of potential nitrate N leaching into groundwater. Basic information concerning farm management practices, soils, and climate are translated into projected N budgets and nitrate N leaching indices. Output can be provided as an annual screening, a monthly screening and an event-by-event analysis. USDA Agricultural Research Service (ARS). http://www.il.nrcs.usda.gov/technical/waterq/index.html
- **Erosion/Productivity Impact Calculator** (EPIC) model developed for field calculations of the effect of management strategies on water quality. EPIC provides: (1) Volume of surface runoff, days of runoff, and percolation below the root zone. (2) N loss in percolate and subsurface flow. (3) Nutrient loss in surface runoff and erosion - soluble and attached N, and soluble P and P loss with sediment. (4) Erosion assessment - sediment loss and associated chemicals. And other outputs. USDA ARS http://www.il.nrcs.usda.gov/technical/waterq/index.html

3.2.3 Evaluating nutrient sources and sinks using a nutrient budget approach

A combination of measurement and modeling is typically needed to calculate nutrient movements within a watershed. One of the primary approaches to quantifying the relative contributions of different sources of nutrients along alternative flow pathways is through a “nutrient budget” approach, also called a “mass balance” approach. Although researchers take many field measurements of nutrient concentrations, it is virtually impossible to completely characterize nutrient movement by field measurements alone. The problem is that it can be hard to isolate some flow paths from others and that spatial and temporal variability make it difficult to determine annual flows solely through field measurements that may capture conditions over a limited area (e.g., at fixed monitoring sites) or at limited points in time (e.g., occasional grab samples and not continuous monitoring). Therefore, it is typical to use field measurements as estimates for some of the flow pathways and to estimate the pathways for unmeasured flows with models that balance the inputs with the outputs. This “mass balance” approach can be loosely or tightly constrained depending on available information.

3.3 Modeling nutrient movement in the Patuxent

Models that are used to estimate nutrient flows under *current* conditions are also applied to *project* nutrient flows after management practices have been implemented. Within the Chesapeake Bay watershed, both simple and complex models have been developed. At the complex end is the Watershed Model produced by the EPA Chesapeake Bay Program Office that calculates nutrient flows for many subwatersheds as well as the Bay watershed as a whole. Recent work is making the model available in an on-line interface called Vortex that allows users to access set scenarios or develop their own management scenarios. Access is currently restricted while development is on-going. On the simple side are "box" models that apply field measurement to understand some estuarine segment and typically employ simplifying assumptions, such as assuming the resident water is well-mixed (e.g., Hagy et al. 2000). Other models are described at the Chesapeake Community Model Program (CCMP 2006).

To estimate nutrient credits that might be generated by activities at specific locations within the Patuxent watershed, we started with a simple nutrient budget framework but used a variety of models and research to inform the equations used in the framework. In other words, although we started with a simple equation aimed at balancing the nutrients within the basin, we estimated the values for stocks and flows using more complex models. As described previously, field measurements are generally insufficient to understand basin response under a variety of conditions.

3.3.1 The nutrient budget framework

We developed our nutrient budget framework by adapting a framework already in use for nutrient trading. NutrientNet is a model developed by World Resources Institute to assign credits to proposed nutrient reduction activities on farms. The methods used by NutrientNet are adapted from the Nutrient Management Yardstick, first developed by the Center of Environmental Science (CML) in the Netherlands and later adapted by the Institute for Agriculture and Trade Policy (IATP) for the US. So, the system has a long pedigree.

The NutrientNet version was first developed for the Kalamazoo Watershed in Michigan and later for the Potomac River, a tributary of the Chesapeake Bay. Complementary modeling that was ongoing in those watersheds was used to parameterize those models. Using GIS data, model-derived parameters, and farm-specific data which are input by the user, the model applies a mass-balance approach to calculate nutrient reductions within a river/estuarine mainstem as a result of BMP adoption on a farm.

We start with this simple mass-balance framework and examine what information is available to parameterize this relationship for our case study watershed. We then discuss what other factors might be considered in evaluating trades.

A five-part system is used to establish a nitrogen emission baseline from a site. Using the example of total nitrogen in a cropped system, the framework consists of these five steps:

Step 1 Locate the containing watershed

Assign the farm location to a watershed and locate the outflow point of that watershed; Spatial (GIS) data can be queried to characterize soil or other factors if necessary.

Step 2 Calculate Nitrogen available at edge of field

Nitrogen into a cropped system is a function of:

Atmospheric deposition
Fertilizer applied
Irrigation
Fixation

Nitrogen moving out occurs through:

Crop Uptake

Step 3 Determine if management practices are excluded

If a BMP is required to some level, then the available nutrients on the field are reduced by a basin-specific efficiency for that practice at the baseline level.

Step 4 Apply edge of stream runoff coefficient

This coefficient reflects the proportion of nutrients available from the field that are likely to reach the stream

Step 5 Apply in-stream attenuation coefficient

This coefficient reflects the proportion of nutrients entering stream that reach the point of equivalency.

In summary, the total nitrogen (N) baseline loading is calculated as follows:

> Total Nitrogen Baseline Loading = (N on field)*(N removal efficiency of required BMPs) *(Edge of Stream Coefficient)*(Attenuation Coefficient)

This equation and underlying parameters are used to estimate the total pounds of nitrogen delivered from the farm to the mouth of the river under baseline conditions of current farming practices and farm location within the watershed. Two aspects of nutrient transport are incorporated. First, not all nutrients available on the field will move to the edge of stream, and second, not all nutrients that reach the stream will be delivered to the mouth, which is the point of equivalency in our case study. What this equation does not *directly* consider is how the containing basin affects the coefficients that determine delivery from a particular farm. That information comes from external models that are used to assign different coefficients to the sub-watersheds within the trading area based on land use, topography and other factors specific to the watershed. Some type of data pre-processing (i.e., modeling) is required to use this simple nutrient budget approach to estimating credits.

The farm area for which the baseline nitrogen loading is calculated will change with the BMP being evaluated. For example, if a riparian buffer is proposed along 100 feet of the stream, only the farm acreage adjacent to the buffer may be counted. The Chesapeake Bay Program uses 4 upland acres as the source area for the baseline loading for every acre of buffer area.

Once the baseline loading is established, the amount of credits generated by a particular activity can be evaluated using two types of relationships. In the simple case, the baseline loading is multiplied by a removal efficiency:

Credit calculations: Apply BMP efficiency

Pounds removed = Total nitrogen loading * BMP efficiency

Then the number of credits generated is evaluated as:

Credits = (pounds TN removed) / x

where x typically ranges from 1 to 2.

The relationship between pounds removed and credits generated is determined by regulators. Issues regarding the uncertainty of delivery or seasonal or temporal variability are not *explicitly* addressed in this formulation, but by increasing x above 1, allows some uncertainty to be implicitly addressed. If the point source must buy two credits for every pound of nutrient discharged, then there is a greater chance of a neutral effect on in-stream nutrients.

3.3.2 Adapting the model to the Patuxent River

We used this framework as the basis of our accounting system for our case study area, and evaluated how well it would work for our watershed. To adapt the model for policy analysis within the Patuxent River watershed, we used available models, data and model output, to develop spatial heterogeneous model parameters as described in the next section.

3.4 Sources for calculating model variables and estimating parameters for the Patuxent Watershed

The Chesapeake Bay Program serves as a rich resource of model parameters and other data for estimating nutrient flows in areas within the Bay watershed (or airshed for atmospheric deposition). The Chesapeake Bay Watershed, Airshed or Estuary models are desirable sources of information because they have been developed by teams of researchers over many years using the best available data. Further, that modeling has been validated by comparing model outputs to field measurements, and the techniques used and quality of model performance have been reviewed by external scientists. Because the Bay Program was a consistent source of all required data, we started with these values to develop our nutrient budget model. However, we also compared model parameters and estimated loadings to studies done specifically for the Patuxent and nearby watersheds to see how calculations compared. Data derived from model output were used to create basin-specific loadings by land use category, edge of field delivery coefficients, and BMP efficiencies.

3.4.1 Locating the farm field (Step 1)

Since our major source of information for the model was the CBP models, we first characterized all areas in the basin as to their location within three watershed segments used in the Bay Program Models (Figure 1). A GIS database of watershed segments had been created by SERC researchers (Weller, pers. comm.), and their watershed boundaries were sufficiently close to the Bay Program boundaries, that we only had to reclassify those watersheds by CBP segment number.

3.4.2 Nutrients on the Field (Step 2)

3.4.2.1 Atmospheric Deposition Models

Atmospheric deposition contributes to the quantity of nutrients that are available to runoff from a farm field or other location. Therefore, atmospheric flows are commonly modeled as part of the available nitrogen on a field. The quantity deposited varies widely by location but may be of sufficient magnitude that such flows cannot be ignored.

Parameters for atmospheric deposition of nitrogen were developed from a Chesapeake Bay Program statistical model of both wet and dry deposition (US EPA CBP 1997, Appendix D). The atmospheric inputs include wet nitrate (NO_3), dry NO_3, organic nitrogen (OrN), organic phosphorus (OrP), and dissolved inorganic phosphorus (DIP). Dry ammonia (NH_4) deposition is assumed to be negligible. These values were generated from regression modeling by the CBP that characterized a logarithmic relationship between precipitation and the NH_3 and NO_3 concentrations in the precipitation as measured through monitoring stations of the National Air Deposition Program (NADP). Calculations have been made for the entire Chesapeake Bay watershed and average values per subwatershed are available. Deposition parameters for wet and dry NO_3 that were specific to the Patuxent River watershed segments were incorporated in our model (Table 2).

Table 2. Atmospheric Deposition by Basin

Physiographic Province	Reservoir	River Segment	Zone	Chesapeake Bay Model Zone Number	Atmospheric Wet Deposition of NO3 (in lbs./acre-year)	Atmospheric Dry Deposition of NO3 (in lbs./acre-year)
Piedmont	Yes	Head Water	1	330	3.46	3.49
Piedmont	No	Head Water	2	340	3.46	3.49
Coastal Plain	No	Head Water	3	340	3.46	3.49
Coastal Plain	No	Middle Estuary	4	500	3.20	3.30
Coastal Plain	No	Lower Estuary	5	500	3.20	3.30

The CBP has also developed the Regional Acid Deposition Model (RADM) that provides information on deposition of nitrogen species for the area of the airshed. Other models have been used to make site-specific estimates for small areas based on digital elevation models (e.g., Ollinger et al. 1993).

3.4.2.2 Data on farming practices: Fertilizer application, Irrigation, Fixation and Crop Uptake

Details of the farm and farming practices will determine what nutrients are available to move from a field, and simple formulas are typically used to make estimates of total available nutrients (e.g., Simpson et al. 1993, Penn State University 1997). Details of the crop, the types and quantities of fertilizer applied, whether irrigation is used, and whether nitrogen-fixing crops or cover crops are grown are some of the most important factors in determining available nitrogen. Crop type and crop yield will reflect how much of the available nitrogen was taken up by plants and removed from the farm site. Soil type is a factor in the quantity of fertilizer the farmer chooses to apply and probably yields and also determines how easily nutrients move out of the field, which is a topic for the next section.

How the nutrient budget model is used determines the source of this information. In an on-line calculator, such as NutrientNet, the credit-seeker (e.g., farmer) would enter this information directly. For a policy analyst seeking to understand the potential for nutrient trading, information on average farm characteristics is available from agricultural surveys such as the

National Agricultural Statistical Survey. Within the Chesapeake Bay watershed, the CBP analyzes available agricultural survey information and can produce estimates of agricultural activity by watershed, county or by the intersection of county and watershed.

3.4.3 Determine obligatory or baseline activities by type of BMP (Step 3)

Because trading programs are not underway in Maryland, it is difficult to determine what practices might be excluded from receiving credit. As described earlier, the WQIA requires all Maryland farmers to have nutrient management plans. As a result, we excluded that practice from consideration as one of the BMPs. Trading programs are being developed in the neighboring states of Pennsylvania and Virginia, and these states provide case studies to suggest which practices might be excluded from credit trades. Pennsylvania requires little to nothing from farmers who wish to sell credits. At present, it appears that all BMP implementation is eligible for credit trading. On the other hand, Virginia has proposed some rather significant requirements for “baseline” activities that are not eligible for credit trading (Virginia Department of Environmental Quality (VDEQ) 2006). Many of these baselines are based on NRCS recommendations and existing storm water regulations for which many areas are in non-compliance. The assumptions we made about baseline activities are shown in Table 3.

3.4.4 Nutrient Runoff: Edge of stream coefficients (Step 4)

The annual yield of nutrients from land within a watershed, or the load per unit area, varies by land use type as well as basin characteristics such as geology, topography and landscape position. Therefore, it is important to at least characterize differences in yield by basin and by land cover type to capture spatial differences in nutrient yield per acre.

Nutrients are carried in both surface and subsurface water flow. Since water flow varies with weather and season, nutrients in surface and shallow subsurface flow also vary with these factors. The variability of water flow makes it difficult to quantify NPS discharges to waterways, and much uncertainty surrounds estimates of current and future NPS discharges (Jordan et al. 2003).

A complicating factor is that releases of nutrients by different land types in a watershed are not independent and additive. Some ecosystems (e.g., riparian buffers) can take up nutrients in response to release of nutrients from adjacent ecosystems. Therefore, adding an acre of cropland to a watershed that is only 10% cropland would probably not result in as much of an increase in watershed nutrient discharge as adding an acre of cropland to a watershed that is 90% cropland. A watershed with 90% cropland would not be as well buffered as one with 10% cropland (Jordan pers. comm.).

Therefore, models are needed to characterize the likely level of basin response to a BMP. Researchers have investigated the site and landscape factors that control release of nutrients from

Table 3a. BMP Credit Accounting Table for Patuxent Basin 500: Riparian Restoration

Management Options	Initial Condition		Nitrogen Reduction		Costs					Cost-Effect
	Initial land use type	Existing nitrogen load at edge of stream (EOS) (lbs/acre)	Total TN offset / acre or unit (lbs / acre/ yr)	TN offset / acre or unit beyond baseline (credits)	Install Cost	Annual O&M (2006 $)	Annual land rental payment $/ac/yr (land taken out of production)	One-time incentive payment	Annual cost (10-year horizon; no annual-ization, 2006 $)	Annual cost per credit (2006 $)
Riparian Restoration										
Riparian forest buffers[1]	Ag, High-Till	18.808	80.22	31.34	$1,729		$190	$100	$373	$10.05
Riparian forest buffers	Ag, Low-Till	13.054	54.90	21.56	$1,729		$190	$100	$373	$13.19
Riparian forest buffers	Pasture	5.871	23.29	9.35	$1,729		$190	$100	$373	$25.04
Riparian forest buffers	Hay	7.125	28.81	11.48	$1,729		$190	$100	$373	$35.94
Riparian forest buffers	Mixed Open	4.270	16.25	8.13	$1,729		$190	$100	$373	$47.05
Riparian forest buffers	Urban (20% Imperv)	7.181	29.06	14.53	$1,729		$190	$300	$393	$25.56
Riparian forest buffers*	Urban - Perv	7.089	28.65	14.33	$1,729		$190	$300	$393	$26.46
Wetland restoration[2]	Ag, High-Till	18.808	80.22	62.68	$1,392	$42	$125	$100	$316	$9.12
Wetland restoration	Ag, Low-Till	13.054	54.90	43.11	$1,392	$42	$125	$100	$316	$11.96
Wetland restoration	Pasture	5.871	23.29	18.69	$1,392	$42	$125	$100	$316	$22.71
Wetland restoration	Hay	7.125	28.81	22.96	$1,392	$42	$125	$100	$316	$32.60
Wetland restoration	Mixed Open	4.270	16.25	13.25	$1,392	$42	$125	$100	$316	$42.68
Wetland restoration	Urban (20% Imperv)	7.181	29.06	23.14	$5,000	$84	$125		$709	$65.62
Wetland restoration	Urban - Perv	7.089	28.65	22.83	$5,000	$84	$125		$709	$46.56
Riparian grass buffers	Ag, High-Till	18.808	59.92	22.69	$206	$60	$170	$100	$261	$9.37
Riparian grass buffers	Ag, Low-Till	13.054	38.98	15.10	$206	$60	$170	$100	$261	$12.52
Riparian grass buffers	Pasture	5.871	12.83	5.61	$206	$60	$170	$100	$261	$25.48
Riparian grass buffers	Hay	7.125	17.39	7.27	$206	$60	$170	$100	$261	$27.54
Livestock Stream Exclusion (offstream watering with fencing, 35' riparian zone)	Pasture	5.871	14.09	0.00	$403	$20			$60	NA[3]
Livestock Stream Exclusion (offstream watering w/ fencing & 100' of riparian grassland)	Pasture	5.871	17.29	7.84	$403	$20			$60	$3.93
Livestock Stream Exclusion (offstream watering w/ fencing & 100' riparian forest)	Pasture	5.871	23.29	9.35	$403	$20			$60	$3.60

1. Buffer BMPs assume a 50' baseline buffer is required as a baseline so only 50' of the 100' buffer is credited in nitrogen reductions.
2. Wetland restoration was assumed to be at the inexpensive end, e.g., tile drains broken to allow land to revert to wetland.
3. This practice is not considered to go beyond baseline, therefore no credits are assigned

Blue shading = lowest cost options; red text = urban land use; credit ratio pounds to credits is 1:1

Table 3b. BMP Credit Accounting Table for Patuxent Basin 500: Annual Crop Practices and Land Conversions

Management Options	Initial Condition		Nitrogen Reduction		Costs					Cost-Effect
	Initial land use type	Existing nitrogen load at edge of stream (EOS) (lbs/acre)	Total TN offset / acre or unit (lbs / acre/ yr)	TN offset / acre or unit beyond baseline (credits)	Install Cost	Annual O&M (2006 $)	Annual land rental payment $/ac/yr (land taken out of production)	One-time incentive payment	Annual cost (10-year horizon; no annual-ization, 2006 $)	Annual cost per credit (2006 $)
Annual Crop Practices										
Cereal Cover Crops - Early Planting (Late planting is baseline)	Ag, High-Till	18.808	8.46	2.82		$31			$31	$10.49
Cereal Cover Crops - Early Planting (Late planting is baseline)	Ag, Low-Till	13.054	5.87	1.96		$31			$31	$13.63
Commodity Cereal Crop/Small Grain Enhancement Early Planting	Ag, High-Till	18.808	4.70	4.70		$31			$31	$6.29
Commodity Cereal Crop/Small Grain Enhancement Early Planting	Ag, Low-Till	13.054	3.26	3.26		$31			$31	$8.18
Commodity Cereal Crop/Small Grain Enhancement Late Planting	Ag, High-Till	18.808	3.20	0.00		$31			$31	NA[3]
Commodity Cereal Crop/Small Grain Enhancement Late Planting	Ag, Low-Till	13.054	2.22	0.00		$31			$31	NA[3]
Land Conversions										
Agricultural land retirement to mixed open	Ag, High-Till	18.808	29.08	29.08	$93		$150	$100	$169	$7.77
Agricultural land retirement to mixed open	Ag, Low-Till	13.054	17.57	17.57	$93		$150	$100	$169	$11.05
Tree planting on former agricultural land	Ag, High-Till	18.808	35.08	35.08	$1,392	$42	$100	$100	$291	$8.30
Tree planting on former agricultural land	Ag, Low-Till	13.054	23.57	23.57	$1,392	$42	$100	$100	$291	$11.19
Impervious surface reduction (from urban imperv to urban perv)	Urban Imperv	7.547	0.92	0.92	$1				$0	$0.04
Forest conservation (during development, onsite forest land set aside)	Urban	7.181	11.82	11.82	$1				$0	$0.01
Urban stream restoration (436 linear feet)		NA	174.20	174.20	$79,453				$7,945	$45.61
Forest harvesting practices	Commercial Forest	25.000	12.50	12.50	$58				$6	$0.47

Blue shading = lowest cost options; red text = urban land use; credit ratio pounds to credits is 1:1

3. This practice is not considered to go beyond baseline, therefore no credits are assigned

Table 3c. BMP Credit Accounting Table for Patuxent Basin 500: Urban BMPs

Management Options	Initial Condition		Nitrogen Reduction		Costs					Cost-Effect
	Initial land use type	Existing nitrogen load at edge of stream (EOS) (lbs/acre)	Total TN offset / acre or unit (lbs / acre/ yr)	TN offset / acre or unit beyond baseline (credits)	Install Cost	Annual O&M (2006 $)	Annual land rental payment $/ac/yr (land taken out of production)	One-time incentive payment	Annual cost (10-year horizon; no annual-ization, 2006 $)	Annual cost per credit (2006 $)
Urban BMPs										
Urban wet ponds & wetlands - new construction	Urban	7.181	2.15	2.15	$3,500				$350	$153
Urban wet ponds & wetlands - retrofits	Urban	7.181	2.15	2.15	$6,300				$630	$275
Dry detention ponds & hydrodynamic structures - new construction	Urban	7.181	0.36	0.36	$3,500				$350	$919
Dry detention ponds & hydrodynamic structures - retrofits	Urban	7.181	0.36	0.36	$6,300				$630	$1,655
Dry extended detention ponds - new construction	Urban	7.181	2.15	2.15	$3,500				$350	$153
Dry extended detention ponds - retrofits	Urban	7.181	2.15	2.15	$6,300				$630	$275
Urban infiltration practices	Urban	7.181	3.59	3.59						NA
Urban filtering practices	Urban	7.181	2.87	2.87						NA
Urban erosion & sediment control			0.00	0.00						
Urban nutrient management	Urban	7.181	1.22	1.22	$7				0.70	$0.54
Urban nutrient management	Mixed Open	4.270	0.73	0.73	$7				0.70	$0.95
Septic pumping	per person on septic	0.018	0.00	0.00		$76			$76	$17,390
Septic upgrade (add denitrification)	per person on septic	0.018	0.04	0.04	$3,648	$692			$1,057	$24,061

Blue shading = lowest cost options; red text = urban land use; credit ratio pounds to credits is 1:1

land into water using various types of models. Multiple investigators have found that the proportion of a watershed in agriculture or developed land can be used to predict water quality in streams (Weller et al. 2003, Jordan et al. 2003, Mayer et al. 2002, Johnson et al. 1997), and slope, soil characteristics, and land cover are generally regarded to be important variables in runoff generation. However, for any given site, the relationship between land use and nutrient runoff can deviate widely from the expected. Therefore, it is important to assess factors of both the site and the watershed when estimating BMP effectiveness.

Baseline nitrogen runoff estimates for subwatersheds of the Patuxent Basin are derived from estimates from the CBP model and compared for some land covers to empirical modeling conducted by Weller et al. (2003) and Jordan et al. (2003) (Table 4 CBP factors by basin). The CBP estimates yield based on land cover proportions and other factors and calibrates the delivery factor for each watershed based on monitoring data of stream concentrations. The Weller and Jordan research determined that the primary factors affecting nitrogen yields by basin were land cover proportions, physiographic province (Piedmont vs. Coastal Plain), and presence of a reservoir. The primary effect of the reservoir was to shunt nutrients from one watershed to another since the population served by the reservoir discharged wastewater into a different river.

We could not directly compare the land use loading factors developed by Jordan and Weller (Jordan pers. comm.) with the factors we derived from CBP watershed model output because the two models differed as to how they evaluated in-stream attenuation and how they drew basin boundaries. Land use loading factors developed by Jordan and Weller are shown in Table 5.

Table 4. Land use loading factors from the Chesapeake Bay Program watershed model

	Basin 330 (Piedmont, Upper Watershed)		Basin 550 (Coastal Plain, Lower Watershed)	
	N at Edge of Stream (kg / ha)	N Delivered to Mouth (kg / ha)	N at Edge of Stream (kg / ha)	N Delivered to Mouth (kg / ha)
Cropland High-Till	23.42	2.84	16.78	16.78
Cropland Low-Till	20.16	2.44	11.65	11.65
Pasture	5.88	0.71	5.24	5.24
Hay	9.10	1.10	6.36	6.36
Mixed Open	5.05	0.61	3.81	3.81
Urban – Impervious	7.20	0.87	6.73	6.73
Urban – Pervious	8.47	1.03	6.32	6.32

Table 5. Land use loading factors for the Patuxent River Watershed by land use type (based on modeling of Jordan and Weller, pers. comm.)

Land use	Physiographic Province	TN discharge (kg N / ha / yr)
Cropland	Coastal Plain	18
Cropland	Piedmont	18
Forest	Coastal Plain	2.9
Forest	Piedmont	1.2
Developed land	Coastal Plain	10
Developed land	Piedmont	Insufficient information

3.4.5 In-Stream Nutrient attenuation (Step 5)

Nitrogen attenuation in a stream, river or estuary is a function of two primary processes: burial and denitrification. These two processes cause nitrogen to become unavailable to organisms in the estuary. Quantifying the relative effect of these processes has been the subject of much research.

The issue of nutrient attenuation in the stream turned out to be troublesome because existing models differed in terms of how they separated edge of field effects with in-stream attenuation effects. The CBP model provided a straightforward estimate of attenuation; one coefficient is used per watershed segment on both fresh and estuarine segments. The Jordan and Weller studies lumped the two effects of field-to-stream and in-stream attenuation, and only evaluated the freshwater portions of the river. However, their work provided an explicit accounting of the effect of water diversions via a reservoir to eventual nitrogen delivery to the mouth. Those results showed that about 56% of water in the subwatershed is diverted in an average rainfall year (Weller, pers. comm.). Finally, a box model created for the Patuxent by other researchers (Boynton et al, in review) was used to estimate attenuation between the mid and lower estuary.

The CBP model estimated average attenuation within a basin, thereby limiting the amount of spatial variability that could be captured. For the upper Patuxent (Segment 330), only 12% of nitrogen that reached streams was estimated delivered to the mouth. For the middle Patuxent (Segment 340) 82% of nitrogen that reached the stream also reached the mouth and for Segment 500 close to the mouth of the watershed, all the nitrogen was estimated to reach the mouth (= no attenuation). These results fit with emerging data showing that headwater streams are able to remove significantly more nitrogen than streams lower in a watershed (Peterson et al. 2001).

The Boynton et al. study separated out burial and denitrification effects of attenuation and linked these processes to area of tidal and sub-tidal marsh and showed different effects between the middle and lower estuary. Their work provided an opportunity to increase the spatial heterogeneity of attenuation effects if we assumed linear removal with stream mile. Using the nutrient budget they developed, we calculated that the TN loss per river kilometer (km) per year is 15,770 kg for the middle basin and 23,900 kg for the lower basin. By cross-referencing data on area of marsh, we estimated that each million meters square of *tidal* marsh buries and denitrifies 26,200 kg N in the middle basin and 22,400 kgN in the lower basin. And the *sub-tidal* areas bury and denitrify 14,200 kg N per year per Mm2 in the upper basin and 6,800 kg N in the lower basin.

3.5 Other factors: Spatial heterogeneity of nutrient runoff and threshold effects

It is well-established that nutrient runoff will vary by many location factors in the landscape. The work by Jordan and Weller (Jordan et al. 2003 and Weller et al. 2003) demonstrated the importance of land cover proportions, physiographic province and reservoirs for determining nutrient runoff in a particular sub-basin. Their work complements other work that showed that percentage of land cover was the major factor that explained differences in nutrient in streams (e.g., Johnson et al. 1997, Jones et al. 2000, Omernik 1977, Wickham et al. 2002). Work by Jordan and Weller suggests a linear relationship between percent cropland and nutrient concentrations in streams indicating NPS BMPs might reasonably be scored using constant efficiencies. However, other work (Lee et al. 2000 and Lee et al. 2001) found that above 60% cropland in a watershed, the stream concentrations increased more rapidly with increasing percent land cover in cropland. Such non-linearities can be evaluated by using basin-specific efficiencies.

On the other hand, very specific details of whether a BMP, such as a riparian buffer, is positioned to intercept nutrients are important for understanding potential BMP effectiveness (Lowrance et al. 1984, Mayer et al. 2006). Riparian buffer effectiveness will vary by location characteristics that may not be captured in a scoring system based on regional (e.g., GIS) data. In a recent review of the literature Mayer et al. (2006) suggest that "soil type, watershed hydrology (e.g., soil saturation, groundwater flow paths, etc.), and subsurface biogeochemistry (organic carbon supply, high nitrate inputs) may be more important factors dictating nitrogen concentrations" delivered to streams than buffer vegetation type.

Some researchers have explicitly demonstrated that certain best management practices differ in their ability to prevent nutrient runoff depending on site-specific details that are not captured in basin descriptors. In particular, the hydrologic regime of the site (e.g., depth to water table, position in groundwater gradient, etc.) has been shown to be important (Mayer et al. 2005, Band et al. 2001). The Mayer et al. work suggested that the type of vegetation in the buffer and even buffer width was not the most important determinant of nitrogen removal, but that soil type, watershed hydrology (e.g., soil saturation, groundwater flow paths, etc.), and subsurface biogeochemistry (organic carbon supply, high nitrate inputs) were potentially more important possibly due to their influence on denitrification rates.

A key hydrologic variable in determining effectiveness of riparian buffers and other BMPs is the length and type of the flow path that water running off land follows to get to the stream. Various methods have been used to quantify the flow path length and test whether it might be used to predict effectiveness of riparian buffers, but results of the effectiveness of such a measure have been mixed (Baker et al. 2006; Hunsaker and Levine 1995; Jones et al. 2001, Schueler and Galli 1992). Similarly, site-specific studies to link buffer effectiveness to measurable characteristics have had mixed success. Baker et al. (2006) found that when flowpath effects were examined basin-wide, the metric was highly correlated with proportion of the basin in particular land uses and therefore did not always add predictive value to the relationship between presence of riparian buffers and in-stream nutrients.

Some work suggests that the level of development in a watershed may influence the importance of some location factors and therefore different levels of effort might be needed to score trades in watersheds with different levels of development. Gergel (2005) showed that the spatial arrangement of different land uses was most important in watersheds in which more than 65% of land was a source of nutrient runoff. Many others have tested the importance of spatial arrangement of land such as the level of connectivity or fragmentation of land uses and the location of source and sink lands relative to streams without generating measures that were more effective than proportion of the basin in particular land uses (Jones et al. 2001, Hunsaker et al. 1994). The result of studies to quantify BMP effectiveness based on location factors have generally not generated metrics that improved predictive capability beyond that due to the proportion of the basin in various land covers. Future work may identify factors for which some consensus can be reached. However, at present such factors are not available to incorporate into scoring models.

3.5.1 Ground water movement and lagged effects due to long travel times

Something that is sometimes not well captured in BMP efficiencies is the potential to shift dissolved forms of nitrogen from surface flow to subsurface flow. Efforts to prevent runoff and phosphorus movement can have the unintended consequence of increasing nitrates in groundwater (McFarland 1995). This shifting of nitrogen to groundwater is important because of the different paths the two types of flow take. Within the Bay watershed, groundwater takes an average of 10 years to reach the mainstem (Phillips et al. 1999), although this varies by aquifer rock type and physiographic province. This lagged delivery of nutrients to the estuary means that nitrogen in groundwater can create a delayed response of the estuary to BMP implementation and could cause future increases in nitrogen from a shift in farming practices.

3.6 Cost-effectiveness of BMPs

To understand which BMPs are likely to be adopted by farmers, it is useful to understand the relative cost-effectiveness for removing nutrients. We compiled the best available information on BMP effectiveness and BMP costs in order to evaluate cost-effectiveness.

3.6.1 Effectiveness of BMPs

Location characteristics will determine the potential of BMP implementation sites to sequester nutrients and prevent delivery to a point of interest. In other words, the ability of a BMP to prevent nutrients from reaching a point in a river or estuary will depend on where that BMP is implemented because 1) location and basin characteristics will alter effectiveness and 2) landscape position affects the ability of nutrients to reach the site.

A variety of methods are used to capture the effectiveness of BMPs, although the data are poorly constrained at this point. For some types of BMPs, the reduction in nutrients may be due to a land conversion or a combination of a land conversion and a removal efficiency. An example of the latter case is riparian buffers. For a typical buffer, crop or pasture land next to the stream is converted to permanently vegetated cover. The area of the buffer is therefore a land use conversion, and the new land use will have a different nutrient loading factor than the former land use. In addition, the buffer also acts to intercept nutrients coming off the land. Therefore, a removal efficiency can be applied to adjacent nutrient sources. The total credits from developing a buffer are therefore the sum of the change in loadings from the land conversion and the interception efficiency.

Most land use conversion factors and NPS BMP efficiencies used here were developed from US EPA CBP (2006), although other sources were used in a few cases. The source US EPA CBP (2003) was used to evaluate efficiency of urban stormwater management, streambank protection measures, and low impact development. The values and their sources are shown in Table 3. The efficiency factors are basin specific where data supported such differentiation and the table shows efficiencies for Basin 330, a Piedmont Basin in the Upper watershed.

3.6.2 Costs to farmers of implementing BMPs

The supply of nutrient credits from non-point sources will be a function of farmer costs to implement nutrient control measures. Those costs include implementation costs of building any structures, implementing new farm management, and maintaining such practices through time. In addition, transaction costs will arise from the time spent learning about trading opportunities, interacting with officials who are overseeing trades, and identifying the quantity of credits that farmers are eligible to sell.

We used a few sources to evaluate the potential costs of implementing BMPs but did not attempt to estimate transaction costs. The main source of cost data are existing government cost-sharing programs. These programs typically receive bids from farmers who are willing to implement BMPs for a given cost and are looking for cost-sharing assistance. We primarily drew cost figures from a document produced by the US EPA (2003) that estimated average costs of both agricultural and urban BMPs. Other sources used were a report prepared for the Chesapeake Bay Commission (2003) and data by Patuxent County for cost-sharing on practices that we collected from an agricultural extension office in each county (Table 6). Other cost estimates have been prepared by University of Maryland researchers (Nelson et al. 2005) but were not used in this study. We used the implementation costs to evaluate the relative cost-effectiveness of various BMPs for the Patuxent River watershed (Table 3).

For the most part, the sources we used to estimate costs did not evaluate how costs were likely to vary in space. We know that effectiveness will vary spatially and temporally which provides some understanding of where BMPs might be implemented most cost-effectively. It may be important to capture spatial differences in cost-effectiveness since BMPs may become clustered in a particular part of the watershed if credits can be produced more cheaply due to location-specific factors. The net effect on water quality outcomes should be positive from such a bias, but understanding whether costs differ by location will be necessary to understand the likelihood that BMPs will be implemented where they can do the most ecological good.

Table 6. Maryland Agricultural Water Quality Cost-Share Figures

		Contour Farming	Contour Orchard	Strip-cropping	Grassed Waterway	Diversion	Terrace	Critical Area Seeding
County	Year	$/acre	$/acre	$/acre	$/linear ft	$/linear ft	$/linear ft	$/acre
Anne Arundel	2003	$20.00	$20.00	$20.00	$3.00	$2.90	$2.90	$1,100.00
Calvert	2005	$20.00	$20.00	$20.00	$9.00	$9.00	$3.83	$1,500.00
Charles	2005	$20.00	$20.00	$20.00	$3.30	$3.30	$2.90	$900.00
Howard	2002	$20.00	$20.00	$20.00	$3.00	$3.63	$3.75	$719.07
Howard	2006	$20.00	$20.00	$20.00	$3.00	$3.63	$3.75	$844.41
Montgomery	2006	$20.00	$20.00	$20.00	$4.65	$5.35	$5.35	$1,500.00
Prince George's	2004	$20.00	$20.00	$20.00	$3.00	$2.90	$2.90	$800.00
St Mary's	2005	$20.00	$20.00	$20.00	$3.75	$3.75		$2,200.00

We found only one model that suggested how costs of BMP implementation would vary spatially. Lichtenberg (2004) developed a model using the county-level differences in reimbursement rates for BMPs in Maryland. His work interpreted such differences in rates to the differences in availability of machinery, topography, soil types and wages for off-farm labor. The driving factor in costs is proximity to urban areas where costs tend to be higher

4. Results and Discussion

4.1 Lessons learned from COASTES modeling project and next steps

4.1.1 General considerations when selecting models

All watershed-based model projections have the potential for high error when used to make calculations for specific locations. However, the models might still be used effectively for making estimates of change in loadings if enough sites are evaluated and if model error is random. With a large number of sites and random errors, the errors should cancel out and total nutrient loadings should be within an acceptable range of error. However, any *systematic* problems in projections (e.g., if nutrient loadings are always overestimated) will create problems for trading systems and can result in goals not being achieved.

As previously described, other watershed models are increasingly available from academic or government sources and can readily provide inputs to trading scoring systems in watersheds outside of the Chesapeake Bay. However, applying general models to specific watersheds can lead to distortions of local conditions. A model developed specifically for a watershed is preferable to a general model that has not been tested and fit for the watershed of interest.

The scoring criteria for credit trading can only be as complex as the data will support. In a well-studied watershed such as the Patuxent, data are available to create highly disaggregated scoring systems to evaluate potential for BMPs to sequester nutrients. However, even with extensive study and modeling, the scientific community has not reached a consensus on the most important site characteristics to determine nutrient movement, transformation and sequestration. Many scientists feel that individual sites need to be monitored to develop appropriate data to score and vet nutrient trades (Jordan, pers. comm.).

Clearly the costs of implementing highly site-specific scoring systems must be balanced against the desire to encourage participation in a trading system and reach desired ecological outcomes. Some jurisdictions might conclude that the spatial and temporal variability of nutrient loadings is too complex and would prefer to assume that all BMPs generate the same level of nutrient reductions. However, for areas that have not been well studied, many simple models are available and in widespread use. At a minimum accounting for basic differences in landscape position of sites, has the ability to improve the quality of trades and provide enhanced ecosystem benefits relative to scoring all trades the same.

4.1.2 Landscape Effects

Although the variety of computer tools to calculate nutrient runoff is growing, many tools rely on models to predict nutrient runoff that have not been thoroughly tested and measurements that may not be generalizable to different geophysical settings. For example, a method originally developed to predict soil losses from farm fields has been modified for use at coarser scales and is in widespread use to predict phosphorus runoff in surface water. The Universal Soil Loss

Equation (USLE) (USDA 1978 and USDA 1997) or the Revised Universal Soil Loss Equation (RUSLE) is the basis of many tools available to estimate phosphorus runoff. This equation relies on local characteristics of precipitation, soil properties, slope length and steepness, and cropping and management practices.

Work by colleagues at the Smithsonian Environmental Research Center has demonstrated some of the inaccuracies of models that are in widespread use for predicting sediment delivery (and related phosphorus loads) to streams (Boomer et al. in review). They found that for basins within the Chesapeake Bay watershed, sediment yield predictions, derived using the USLE, and the RUSLE over-predicted annual average sediment delivery by 75 to 200 percent. Predictions from USLE and RUSLE differed but were highly correlated and accuracy, measured in terms of the ability of the algorithm to rank basins in terms of sediment delivery, was similar.

In addition to testing USLE-based methods, they compared five Sediment Delivery Ratio (SDR) algorithms, which are empirical models used to estimate annual average sediment delivery. They found these algorithms were more accurate than USLE or RUSLE but still exceeded observed sediments loads by more than 100%. They tested the models in over 100 watersheds, ranging in size from 100 to over 90,000 ha, within the Chesapeake Bay watershed for which they had water quality monitoring data. The USLE and RUSLE models were designed for use at the plot scale and provide reasonably accurate estimates at that scale. However, the models have not been validated for use at the watershed scale and this work in combination with other research demonstrates that estimates from these models have high uncertainty.

4.1.2.1 Estuarine Effects / Boundary Conditions

One of the lessons learned about the Patuxent River and estuary from the joint modeling work is that the system is closely connected with the rest of the Chesapeake Bay. Modeling suggests that nutrient levels within the lower estuary are controlled largely by the influx of nutrients from the Bay mainstem (Lung and Nice, in press). However, this research does not agree with earlier field-based research showing that the River is typically a net exporter of nutrients to the Bay (Boynton et al. 1995). Nonetheless, all research suggests the possibility that the River can be a net importer of nutrients under certain conditions, particularly if nutrient loads in the Patuxent were dramatically reduced. The modeling work suggests that nutrient reductions must occur Baywide before Patuxent water quality goals can be met (Lung and Nice, in press).

Local vs. regional effects on water quality

If what the estuary model suggests is true (Lung and Nice 2007), then it does not effect the lower estuary if we alter the point of equivalency used in a Patuxent watershed trading scenario. Since nutrient levels in the local estuary appear to be strongly controlled by nutrients coming in from the Bay mainstem, nutrient reductions in the watershed will not have a significant effect on the hypoxia/anoxia in that point of the river. However, nutrient reductions in the upper estuary and in the non-tidal portions of the river may have locally beneficial effects and will reduce outputs to the Bay mainstem. A system-wide reduction in nutrients appears necessary to affect the hypoxia/anoxia in the Patuxent.

4.1.2.2 Availability of models and research to inform trade scoring

For our watershed, we found a good variety of models had been developed to inform credit scoring. However, we also discovered that the models were not generally designed to provide simplified scoring rules. Most models are designed to capture a great deal of complexity and therefore do not lend themselves easily to generalizing across many sites (e.g., Weller et al. 2003). Alternatively, models may be generalizable, but too complex to run for such scoring applications (e.g., the CBP watershed model).

The basic approach we were able to use was to use simplified output from the complex models and put that information into a simple formula for a nutrient budget. This approach works well as long as the modelers are willing to provide appropriate output. In our case, the primary model used was the Chesapeake Bay Program watershed model because this model had been designed for water quality management purposes and was able to provide a comprehensive view of the watershed. Other models did not necessarily completely agree with that model, so finding absolute consensus on basin responses to BMPs was elusive.

We found the state of the science was marked by mature models but immature field data to support certain aspects of the models. The long history of model development in the watershed and associated monitoring data provided a solid foundation for projecting changes in nutrients with land use change. However, these complex models come at a high cost in terms of time and money invested. Empirical models are simpler to build and provide readily understandable results with an understanding of the error rates associated with the model. It is therefore easy to test whether empirical (statistical) models have random errors as opposed to biased errors that might affect the quality of scoring systems. Biased errors could have a tendency to over-or underestimate nutrient changes with BMPs that could make the system inefficient or cause nutrient targets to be missed. The types and magnitude of the error of process-based models such as the CBP watershed model are virtually impossible to evaluate.

While a great deal of field data has been collected and used to inform models, there remains a poor understanding of current BMP implementation rates (Thompson, 2004). In addition, because of the complexity of the models used to inform the scoring, the dynamics of changing land use cannot be easily incorporated. This is important because basin response can change non-linearly as more land is converted to developed uses (Jordan, pers. comm.).

4.1.2.3 Factors important in effectively scoring trades in Patuxent River

As discussed, not all aspects of spatial heterogeneity of BMP cost-effectiveness could be evaluated. What we could capture based on available models and understanding was:

- Effects of atmospheric deposition
- Regional effects of underlying geology/hydrology
- Effects of land use cover at a fixed point in time (static relationships)
- Effect of estuarine circulation
- Effects of built water diversions (e.g., reservoirs)
- In-stream attenuation

The CBP watershed model created distinct watershed segments for three areas of the Patuxent basin despite the fact that one of the CBP basins contained two different physiographic provinces (Coastal Plain and Piedmont) and the lower basin drained both into the middle and lower estuary. The CBP creates model outputs that suggest these basins are homogeneous in contrast to research suggesting that it is important to distinguish at fine-scale detail. The model does incorporate some heterogeneity within basins by using model coefficients that represent weights representing the proportion of basin area with different characteristics (e.g., physiographic province). This technique works fine for evaluating effects across the scale of one or more watersheds but limits the model's ability to score site-specific activities. The CBP model may be appropriate for scoring trades consistently across the watershed but does not necessarily capture some of the important processes going on within the Patuxent watershed.

The CBP model was developed for the scale of the entire Chesapeake Bay watershed and therefore has limited ability to make fine-scale distinctions for any individual river. Further, the tools needed to make fine-scale distinctions would be cost-prohibitive to apply in this situation since information on the site would likely be needed to provide model information.

4.1.2.4 Factors not readily captured

The factors that we were not able to capture in our scoring system were:

- Explicit land cover / geology / hydrology along flow path to stream
- Effects of changing land cover, particularly whether basin response may change non-linearly as proportion of development increases
- BMP effect on lagged deliveries of nutrients to estuary (due to shift to groundwater flow from surface flow)
- Details of different movement of nutrient forms and remobilization
- Site specific factors controlling BMP effectiveness
- Temporal variability of BMP effectiveness with weather
- BMP effectiveness under extreme events

It has been suggested that stochastic modeling might be used to evaluate the net behavior of BMPs under weather and climate variability. However, a great deal more information needs to be collected on BMP effectiveness to inform such modeling.

5. Conclusions

In summary, reasonably appropriate data and models were available to inform our scoring system for nutrient credit trading in the Patuxent watershed. The system we developed for scoring was able to capture some important aspects of landscape heterogeneity and importance of BMP location in measures of effectiveness. However, the models used did not provide a means to assess potential error of scores and did not capture variability of performance with weather conditions or other temporal factors. The most poorly constrained data appear to be those for BMP performance by specific practice. Also, the inability to determine the error rates associated with the process-based models used (e.g., CBP watershed model) and whether there might be systematic biases in estimations of BMP effectiveness have the potential to limit the success of trading programs in achieving environmental goals.

Using model output from our research team, we concluded that the point of equivalency used for establishing trading ratios between point and non-point sources is not important for improving low oxygen conditions in the Patuxent lower estuary. Rather, modeling suggests that if nutrients are lowered only in the Patuxent, the estuary will become a net importer of nutrients and that local oxygen conditions will not improve. Further research is needed to confirm this result.

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