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Assessing and improving the outcomes of nonpoint source water quality trading policies in urban areas: A case study in Virginia

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ABSTRACT

Nonpoint source (NPS) water quality trading (WQT) is a market-based approach to improving water quality. Past work has shown that these programs could increase localized pollutant loadings, in part by exporting water quality controls from urban to rural areas. Virginia's NPS WQT program has enabled thousands of transactions and may provide a model for other programs, but its impacts on urban water quality have not been thoroughly assessed. We quantify the impact of NPS WQT purchases in Virginia on water quality and hydrology in an urban catchment. We go on to assess outcomes of a policy alternative where buyers and sellers are collocated in the urban catchment. Simulation results show that NPS WQT increased total phosphorus (TP) loading by an average of 0.8 lbs TP/year for each 1.0 offsite credits purchased in the analyzed catchment. The TP loading increased in years with greater rainfall, such that TP loads were increased by up to 1.2 lbs TP/year for each offsite credit purchased. These loading increases may or may not be acceptable, depending on the cumulative number of purchases within an urban catchment and existing local water quality issues. In our policy alternative with buyers and sellers collocated in the catchment, we found that the TP increase from development was completely offset at the catchment scale, with a decrease of 4.3 lbs TP/year for each 1.0 credits purchased. This suggests that credits awarded for urban mitigation practices are undervalued compared with water quality requirements for credit purchasers. This undervaluation is a result of the Virginia trading program using one approach to compute the credit value for buyers and a different approach to compute the credit value for sellers. We demonstrate how using a single model to determine both buyer and seller credit values in urban areas could provide greater transparency and mitigate the risk of urban pollution hot spots. This work demonstrates the importance of consistency in the scale of pollutant load calculations between buyers and sellers for NPS WQT, and contributes novel insight into the implications of WQT for urban NPS pollution.

Keywords: water quality trading; stormwater management; nonpoint source pollution; watershed management

1. INTRODUCTION

Nonpoint source (NPS) water pollution represents one of the greatest engineering and policy challenges of this century (Patterson et al., 2013). Many human activities, including agriculture and urbanization, lead to NPS pollutants entering waterways through runoff. These pollutants are challenging to measure or attribute to specific polluters, but can cause considerable damage to human and ecosystem health. Controlling NPS pollution through regulation is exceedingly costly, with NPS water quality programs in the United States costing billions of dollars per year (Ribaudo et. al, 2011).

Water Quality Trading (WQT) is a form of environmental credit trading that can minimize the cost of controlling water pollution. Under WQT, a regulated entity is permitted to purchase pollutant offsets ("credits") from an entity with a lower cost of abatement, generally within a watershed boundary. More than a hundred WQT programs involving NPS polluters (NPS WQT programs) were developed across the U.S. between 2003 and 2013 (BenDor et al., 2021). About half of these NPS WQT programs failed to generate an active market (BenDor et al., 2021; Hoag et al., 2017; Stephenson and Shabman, 2017), but where markets have become active, considerable cost savings of have been reported over traditional approaches (Corrales et al., 2014, 2017; Nobles et al., 2017). Given the theoretical environmental and economic benefits for NPS WQT, the U.S. Environmental Protection Agency (USEPA) has recently encouraged new development of highly flexible and innovative markets that encourage greater participation than past WQT efforts (USEPA, 2018; 2019). Inclusion of new categories of NPS buyers and sellers is likely to be one way that future markets can generate participation (Heberling et al., 2018). Efforts to encourage WQT in urban areas, in particular, have increased in recent years (Malinowski et al., 2020).

NPS WQT programs can be designed to achieve water quality goals at a lower cost, but such market-based controls have been shown to result in unintended outcomes that might put waterbodies at risk. The Virginia NPS WQT program provides an important example of this trend. In Virginia, credits are most commonly generated through the permanent conversion of agricultural land to forest. Credits are most commonly purchased by regulated land developers who forgo construction of permanent water quality controls, such as bioretention basins, at development sites. Therefore, credit purchases mainly occur in urban areas, with credit generation sites ("banks") located an average distance of 164 kilometers away along the stream network in agriculturally-dominated watersheds (Saby et al., 2021b). Over the past five years, this program has grown considerably, with over 300 NPS credit banks approved and thousands of credits purchased annually (USACE, 2022).

If widespread use of NPS WQT credits can lead to lower urban water quality under this market structure, this could introduce issues not only for urban streams but for regional water quality health. Large areas of connected impervious surface in urban areas increases the mobilization and transport of pollutants, allowing contaminants to travel long distances downstream to receiving estuaries and coastal areas (Jackson and Pringle, 2010; McGrane, 2016). Urban environments are also subject to a unique set of pollutant sources and annual load timings compared with surrounding environments (Carey et al., 2013). Therefore, pollutants from urban areas can be delivered downstream more efficiently and at different times of the year compared with pollutants from rural areas.

Despite the potential risk to urban water quality, the impact of Virginia's NPS WQT program on urban catchments has never been assessed using detailed simulations. Land disturbance entities that purchase NPS WQT credits calculate site-based requirements using the Virginia Runoff Reduction Method (VRRM), a simple spreadsheet tool that estimates annual pollutant loads for land disturbance activities with limited adjustments for site-specific considerations. VRRM depends on major simplifying assumptions, including consistent BMP performance regardless of storm intensity. Much literature on WQT has focused on program development and design (e.g., Lee and Douglas-Mankin, 2011; Zhang et al., 2019, 2022.), but this body of work has not focused on evaluating program outcomes in existing programs.

This study is therefore motivated by the finding that NPS WQT is a frequently proposed policy solution, but the outcomes of these policies for water quality have rarely been assessed. In particular, efforts to implement NPS WQT programs in urban areas have accelerated, yet there is no study examining the outcomes of these programs in the context of urban catchments. This study has three objectives to fill this knowledge gap:

1. *Quantify outcomes:* Quantify the impact of NPS WQT purchases on water quality and hydrology in a simulated urban catchment in Virginia.
2. *Assess alternatives:* Quantify the outcome of a policy alternative in which NPS WQT buyers and sellers are collocated in the same catchment.
3. *Describe potential improvements:* Describe how NPS WQT policies could be updated to improve water quality outcomes, including by improving equivalency for WQT buyers and sellers in urban areas.

We accomplish these objectives using the Stormwater Management Model (SWMM) to simulate water quality and hydrology under different policy scenarios in a small urbanized catchment. SWMM enables an intermediate-scale stormwater analysis between the field-scale estimates used to calculate TP reduction requirements for credit buyers (calculated in VRRM), and the lumped watershed-scale estimates that are used to calculate NPS WQT credits for sellers (calculated in the Chesapeake Assessment Scenario Tool, CAST). Our SWMM model contains more detailed hydrology, with 9 years of hourly precipitation data, Green-Ampt infiltration parameters, and dynamic wave flow routing. This method enables evaluation of NPS WQT policy outcomes at a finer timestep, with more detailed hydrology, and with more complex land use representation within the catchment scale compared with models that have been used in NPS WQT program development and implementation. Our results demonstrate trends in water quality and hydrology across years with variability in precipitation, and compare impacts for daily precipitation rates. SWMM is one of the most widely-used models to assess urban stormwater hydrology, and is commonly used for stormwater system design and to support policy analysis and decision-making in urban areas around the world. No study has previously simulated the outcome of NPS WQT policies in this detail or at the urban catchment scale.

Results of this study demonstrate the successes and drawbacks of the pioneering NPS WQT program in Virginia for the first time. While many NPS WQT programs have failed to become operational, Virginia's program provides an important example of a functioning market that could offer some reprieve to the costly problem of addressing NPS pollution. The outcomes of this market structure in practice, including the research presented here, provide important insights to both widespread adoption and continued expansion of the program in Virginia.

1.1 Program structure and credit calculation in Virginia's NPS WQT program

Virginia's NPS WQT program was created to increase flexibility and reduce costs for achieving the total nitrogen (TN), total phosphorus (TP), and sediment Total Maximum Daily Loads (TMDLs) for the Chesapeake Bay. One NPS credit is equivalent to one pound of pollution per year prevented from entering the Chesapeake Bay. In Virginia, NPS credits may be generated for each of the three pollutants with Chesapeake Bay TMDLs (TN, TP, and sediment). Sales are negotiated between the credit banks (sellers) and regulated entities (buyers).

In this program, credit sellers most often generate credits through the permanent conversion of agricultural land to conserved forest that may be timbered. These credit banks are often managed by professional mitigation bankers, who are contracted by the landowners to manage land conversion, maintenance, and credit sales procedures. The Virginia Department of Environmental Quality (VADEQ) calculates the number of TN, TP, and sediment credits awarded per acre at the 8-digit Hydrologic Unit Codes (HUC-8) scale using CAST. CAST is a web tool based on the Chesapeake Bay Model simulation of TN, TP, and sediment loads (Chesapeake Bay Program, 2020). VADEQ has made these conversion values public in a lookup table (VADEQ, 2020). Landowners must meet baseline requirements before credits are awarded, including establishing a 35-foot riparian buffer. Once the conversion is complete and credits are approved by VADEQ, credit availability is noticed on the Regulatory In-Lieu fee Bank Information Tracking System (RIBITS; USACE, 2022). Banks must report the number of credits sold to VADEQ periodically, and RIBITS is subsequently updated.

These NPS WQT credits are most often purchased by developers in urban areas, which are regulated under the Virginia Stormwater Management Program (VSMP). By Virginia law, certain land-disturbing activities are required to mitigate impacts to water quality. In particular, land developers must limit TP runoff from developed sites to 0.41 lbs/acre/year. TP is treated as a keystone pollutant, meaning that developers are only required to calculate reductions to TP, and other pollutants are assumed to be reduced by ratios calculated using the Chesapeake Bay Model. For each project, developers calculate their annual TP reduction requirements in the VRRM spreadsheet, and then decide whether to purchase credits or place onsite controls to meet the reduction requirements. Increasingly, developers are buying TP credits to cover all of their water quality requirements. Water quantity requirements, such as stormwater basins for flood protection, must still be met onsite. If a developer chooses to purchase credits, they contact the WQT bank manager directly to negotiate a purchase. Credits are sold at a 1:1 trading ratio, meaning one credit awarded for agricultural land conversion is assumed to offset one credit purchased. Once a developer purchases a TP credit, the associated ratio of TN and sediment credits are retired from the bank (removed from the market).

For the purposes of this study, it is important to highlight that this program targets the Chesapeake Bay, and does not consider more localized impacts to water quality. Some local restrictions do exist, including that credits must be purchased from within the 12-digit hydrologic unit code if there is a local nutrient-based TMDL, or from the closest available bank if there is a nutrient-related impairment. Despite these restrictions, credit buyers are clustered in urban areas, and sellers are found in agricultural areas (Saby et al., 2021b). Credit banks may generally be located upstream or downstream from the purchase site, and the average distance between buyers and sellers is 164 km measured along the stream network (Saby et al., 2021b). Because the agricultural credit banks are generally far, and sometimes downstream, from the urban purchase sites, our model assumes that water quality in the urban catchment is not impacted by the credit banks.

It is also important to emphasize that there is inconsistency in the scale of models applied in Virginia's NPS WQT program. In this subsection, we have discussed that conversion credits are awarded to banks based on CAST calculations of pollutant load changes at the Chesapeake Bay. Meanwhile, credits are purchased by developers based on VRRM calculations of pollutant load changes at the edge of the development site. The equivalency of these edge-of-tide (CAST) and edge-of-site (VRRM) models has never been verified in the literature. This mismatch provides greater context for our use of an intermediate-scale model (SWMM) to calculate tradeoffs for urban settings.

2. METHODS

Virginia WQT purchase sites were simulated according to VSMP regulatory requirements for development. We use a SWMM model of the Meadow Creek Watershed in Charlottesville, Virginia for this analysis, which was calibrated for streamflow (Herbst et al., 2022). We add subbasins to this model in which we simulate hypothetical development. Three regulatory compliance scenarios are applied. Results are analyzed in R code. Based on these results, alternative policy scenarios are developed. An overview of this study workflow is presented in Figure 1.

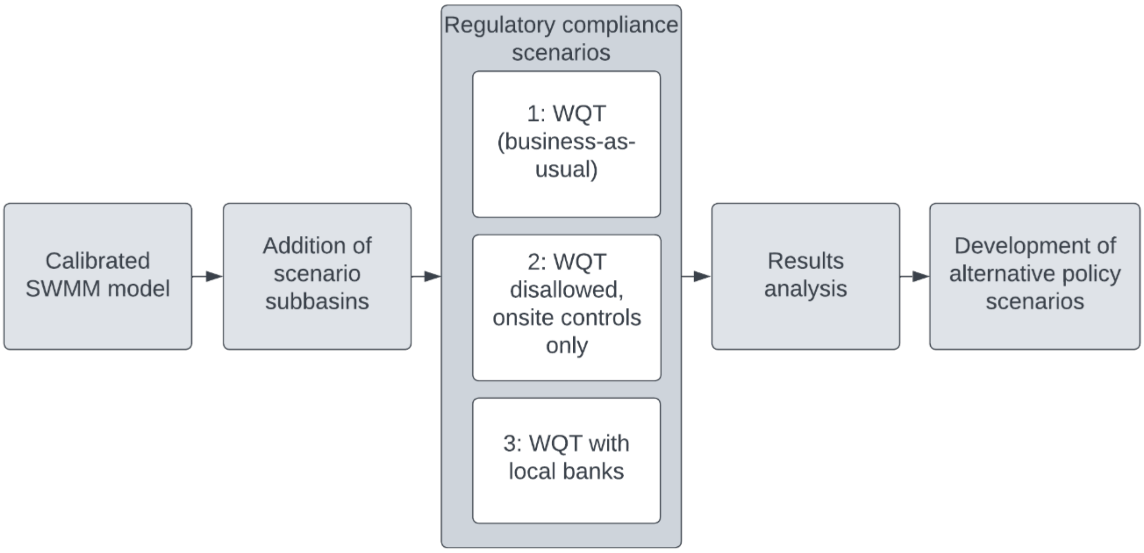

Figure 1. Methodological framework

2.1 Study area

The study area is the 5.5-square mile upper portion of Meadow Creek catchment located in Charlottesville, Virginia. Figure 2D shows the Chesapeake Conservancy Land Use Project (CCLUP) data for the area. These data were produced from the National Agricultural Imagery Program (NAIP), Light Detection and Ranging (LIDAR) data, and other ancillary data to classify one-meter land cover in 2013 and 2014 (CCLUP, 2018). The study area is 45% impervious land cover. The majority of the pervious areas have hydrologic group B soils, which has a relatively low potential for runoff when saturated (Soil Survey Staff, NRCS, 2022). Considerable areas of group D soils, with high runoff potential, are present underlying impervious areas and in riparian zones.

This study area was selected because it is a small, highly urbanized watershed in Virginia with available data for model development. The goal of this analysis is to present unit differences in pollutant export between regulatory compliance strategies in a theoretical policy scenario, rather than specific solutions to water quality issues found in the Meadow Creek watershed. As a result, our model includes accurate land use and topographic characteristics, but does not detail regulated point and nonpoint sources nor BMPs already implemented in the watershed.

2.2 Stormwater Management Model (SWMM) description and calibration

SWMM is used to simulate the three scenarios. SWMM is an urban hydraulic and hydrologic model developed by the USEPA capable of simulating water quality and quantity conditions within a local area continuously or during storm events. SWMM simulates rainfall-runoff processes in urban areas and stormwater drainage networks. The application of SWMM in this study includes the generation of NPS pollutant loadings using the pollutant and land use editor modules, and the use of the Low Impact Development (LID) control module. SWMM models LID controls as a combination of vertical layers, where pollutants are removed through infiltration or a user defined pollutant removal efficiency (James et al., 2010).

The upper Meadow Creek watershed SWMM model was originally constructed for the City of Charlottesville and included 87 subbasins and the stormwater pipe and stream network (Charlottesville Stormwater Services, 2010). PCSWMM is modeling software that runs SWMM with additional functionality (CHI, 2022). Using PCSWMM, we updated the model to include four additional subbasins (a total of 91 subbasins), with subbasin areas ranging in size from 5.9 to 260 acres, with a median area of 22 acres. In the new model, the Green and Ampt infiltration parameters were updated using tabulated reference values and Soil Survey Geographic Database (SSUURGO) soil type data (NRCS Soil Survey Staff, 2022). Impervious surface percentages were also updated using the CCLUP data (CCLUP, 2018).

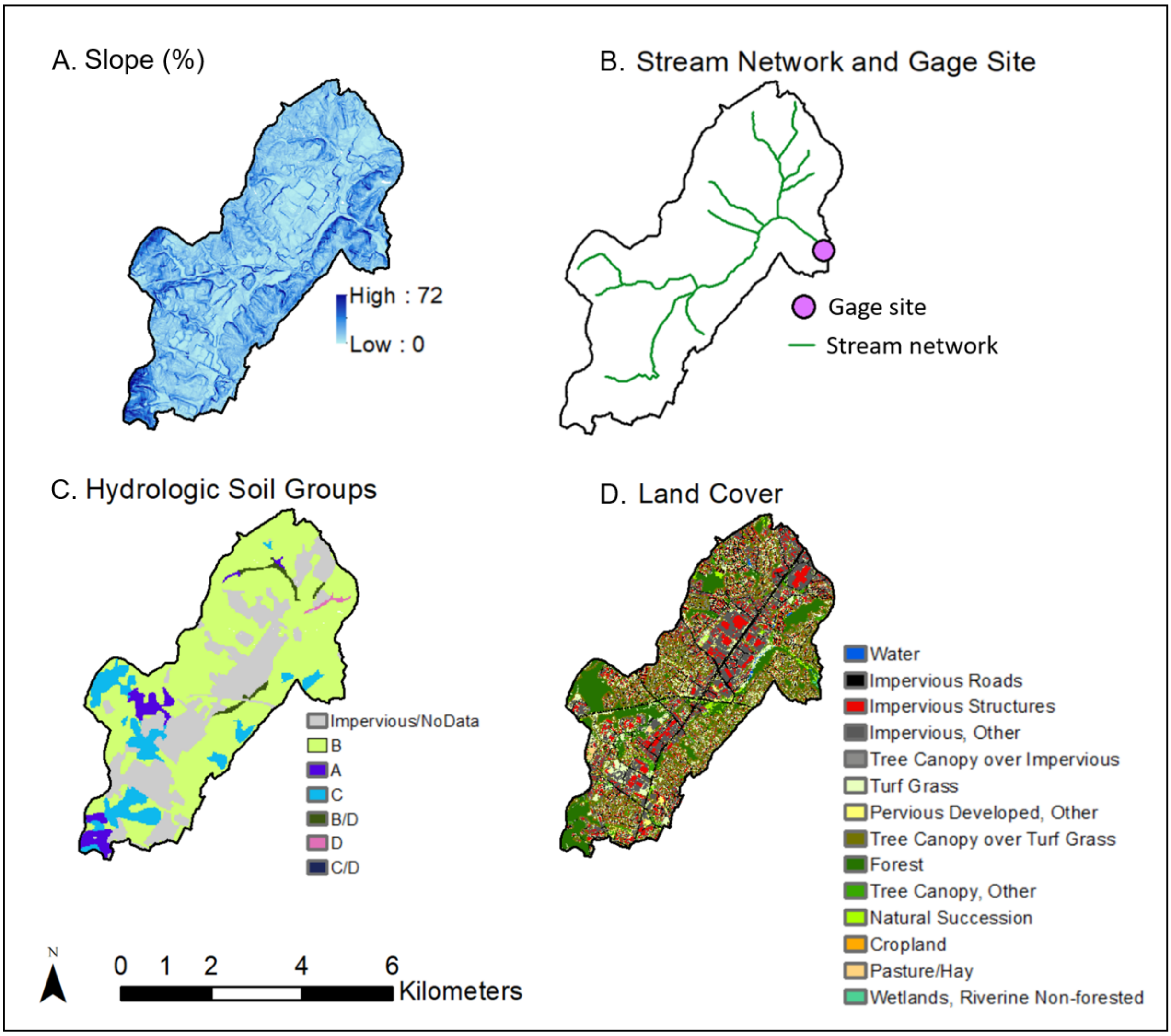

Figure 2. Slope (A), stream network (B), hydrologic soil groups (C), and land cover (D) for the study area.

Meadow Creek stream flow is derived from a depth-discharge relationship curve using data from August 2020 to February 2022. Water level was measured continuously at a 15-minute time interval and recorded with an ONSET HOBO U20L-04 water level logger installed inside a PVC stilling well that is anchored at a cross-section at the Meadow Creek outlet. Discharge measurements were recorded for 22 periods at the outlet cross-section with a SONTEK Flow Tracker 2 Velocimeter. A rating curve was constructed using the depth versus discharge data and fitted with a power function. Rain data were collected using a HOBO Rain Gauge Data Logger that recorded cumulative rainfall at 15-minute intervals coinciding with the water level time steps. See Herbst (2022) for more details on this data collection project.

Subcatchment and conduit parameters were calibrated using the PCSWMM sensitivity-based radio tuning calibration (SRTC), using a storm event from April 8th, 2021. Characteristics of the storms that were used during validation and calibration are summarized in Table 1. The SWMM parameters that were adjusted in the calibration process include subbasin width, depression storage, Manning's n for impervious surfaces, and conduit roughness. Nash-Sutcliffe Efficiency (NSE) is a common hydrologic "goodness-of-fit" index that is used to assess a model's ability to predict flow conditions relative to observed flow conditions. NSE values closer to one indicate better predictive ability of the model. The calibration was validated three additional storms, with results shown in Table 2. These storms were selected based on the quality and availability of rainfall data and storm flow data. Larger storm flow events would be difficult to measure accurately as these could result in significant overland flooding, and the rating curve does not capture out-of-channel flow.

Table 1: Characteristics of storms used in model calibration and validation. The 04/08/2021 storm was used in calibration.

Storm Date	03/31/2021	04/08/2021 (calibration)	06/11/2021	08/08/2021	08/18/2021
Total rainfall (inches)	0.75	1.07	1.51	0.77	1.68
Peak Runoff (CFS)	417	699	1,148	816	1,591
Storm Duration (hours)	4.5	9.5	4	2	7.5
Total Runoff (cubic feet)	260,900	4,021,000	5,534,000	2,677,000	7,244,000

This SWMM model is applied to compare to regulatory models that do not include a baseflow component, so baseflow was not incorporated into SWMM to provide a clearer comparison. This model was designed to represent discharge following precipitation, and not to characterize flow between precipitation events. For this reason, the calibration was performed on a storm event, rather than over continuous time. This means that in both VRRM and the SWMM model, response is based only on stormwater runoff, with no baseflow representation. Therefore, SWMM provides a finer time-step and more detailed land use and hydrology than VRRM, and shares the assumption that pollutant loadings are dominated by stormflow response.

Table 2. Performance metrics for the storm events used for model calibration and validation. NSE = Nash-Sutcliffe Efficiency. RMSE= Root Mean Square Error.

Event date	NSE	R^2	RMSE (CFS)
03/31/2021	0.625	0.913	141
04/08/2021 (calibration)	0.798	0.942	162
06/11/2021	0.873	0.908	325
08/08/2021	0.775	0.838	207
08/18/2021	0.923	0.924	215

2.3 Simulating water quality using SWMM

Water quality modeling in PCSWMM required assigning land use percentages to each subbasin in the model and assigning EMC values to each land use category. CCLUP data was used to identify land use in the watershed because this classification includes detailed urban land covers that separate impervious from other land covers, as shown in Figure 2D. In the absence of locally observed EMC data for land uses in the study area, information was applied from the literature. The Washington D.C. Department of the Environment produced a technical memorandum that summarizes EMC values for TP by land use from literature (District Department of the Environment, DDE, 2014). The Long-Term Hydrologic Analysis (L-THIA) also includes EMC values by land use, as referenced by USACE (2017). We combined these sources to match TP EMCs to CCLUP land uses as summarized in Table 3.

The output of the EMC water quality model was validated by comparing with a U.S. Geological Survey (USGS) summary of loads at the Chesapeake Nontidal Network Stations for water years 2009-2018. A 9-year simulation of our model (2012-2021) showed an average loading of 1.32 lbs TP/acre/year. The USGS report showed TP loads of 0.13 to 2.01 lbs/acre/year based on the monitoring network, with a combined average of 0.52 lbs/acre/year (Moyer & Blomquist., 2020).

This indicates that the loadings found in this study are on the high end of the loading distribution in the Chesapeake Bay Watershed, which was deemed reasonable given the high level of urbanization in the study area.

Table 3: Land use categories and Event Mean Concentrations (EMCs) used in the SWMM model.

Land Use in SWMM	Percentage of watershed	EMC (mg TP/L)	Source(s)	Equivalent land use in respective source(s) (land use weight)	CCLUP categories
Impervious, Road	12.4	0.4	DDE, 2014	Highways/roads (100%)	Impervious, Road
Impervious, Non-Road	26	0.32	USACE, 2017	Commercial (100%)	Impervious, Non-road
Tree canopy over impervious	3.3	0.25	USACE, 2017; DDE, 2014	Commercial (70%), Forest/open (30%)	Tree canopy over impervious
Water	0.01	N/A	N/A	N/A	Water
Forest	20.7	0.01	USACE, 2017	Forest (100%)	Forest, wetlands
Tree canopy over Turf	20.8	0.23	USACE, 2017; DDE, 2014	Agricultural (10%), Forest/open (90%)	Forest, turf grass
Mixed open	3.9	0.2	USACE, 2017; DDE, 2014	Residential (20%), Forest/open (80%)	Mixed open, small and medium fractional turf
Turf Grass	11.8	1.3	USACE, 2017	Agricultural (100%)	Turf grass, pasture
		Weighted average: 0.35			

A second validation of our water quality model was performed using CAST. CAST calculates a per-acre loading for developed areas in the HUC-12 Meadow Creek watershed of 1.23 lbs/acre/year in the 2020 Watershed Progress scenario. Natural load sources contributed 0.26 lbs TP/acre/year. The land uses in SWMM were mapped to developed or undeveloped land covers, and a weighted average of per-acre loadings for the study area was calculated. An average per-acre loading rate of 1.10 lbs TP/acre/year was found, which is 0.22 lbs/acre/year less than the SWMM model predicted. This error was within the reported uncertainty range of reported TP EMC values.

Studies have reported a range of TP EMC values by land use type, with values generally ranging +/- 0.3 mg/L (DDE, 2014). This study ignores this uncertainty by assuming a single, representative TP EMC value for each land use type included in the modeling domain. Future research could extend this study by including the TP EMC uncertainty in the modeling scenarios to better understand its impact on the study results.

2.4 Addition of scenario subbasins to simulate development

NPS WQT purchase sites are represented in the model as identical subbasins. These subbasins were added to the calibrated model to represent areas of development where scenarios could be implemented. Using additional land area, instead of updating existing subbasins, allows use of identical regulatory requirements and calculations for each of the purchase subbasins, and allows the assumptions about pre-development conditions to be consistent across sites. These subbasins are referred to as "scenario subbasins." Land use change was simulated according to Figure 3, with each 2.4-acre site converted to 50% impervious (non-road) and 50% turf. Parameters for the scenario subbasins and rationale and are summarized in Table 4.

Figure 3. Land use conversion implemented in SWMM in each scenario subbasin.

Table 4. Key parameters in the scenario subbasins applied in SWMM.

Parameter	Pre-development	Post-development	Note
Forest (%)	100	0	Pre-development site assumed to be good condition forest.
Impervious (%)	0	50	
Managed turf (%)	0	50	
Area (acres)	2.4	2.4	Median area for a NPS WQT purchase site in Virginia (Saby et al., 2021b).
Slope (%)	5.7	5.7	Average slope across subbasins used in the upper Meadow Creek SWMM model. Shown in Figure 2A.
Hydrologic soil type	B	B	Predominant soil type in the region according to National Resource Conservation Service data (2022)

Twenty-five scenario subbasins were placed randomly throughout the calibrated model. The total combined area of the scenario subbasins added to the model is 60 acres, or representing a 1.6% increase in the total catchment area. The subbasins were created using an R script and uploaded to PCSWMM. A screenshot of the PCSWMM model with details for one of the scenario subbasins is shown in Figure 4 for illustrative purposes. Peak runoff flows and depths for 1-year and 10-year 24-hour storms from the scenario subbasins were generated using the pre-development model with forested scenario subbasins, and then used in compliance calculations for water quantity control. All sites were adjusted using the post-development parameters shown in Table 4 before applying scenarios 1, 2, and 3.

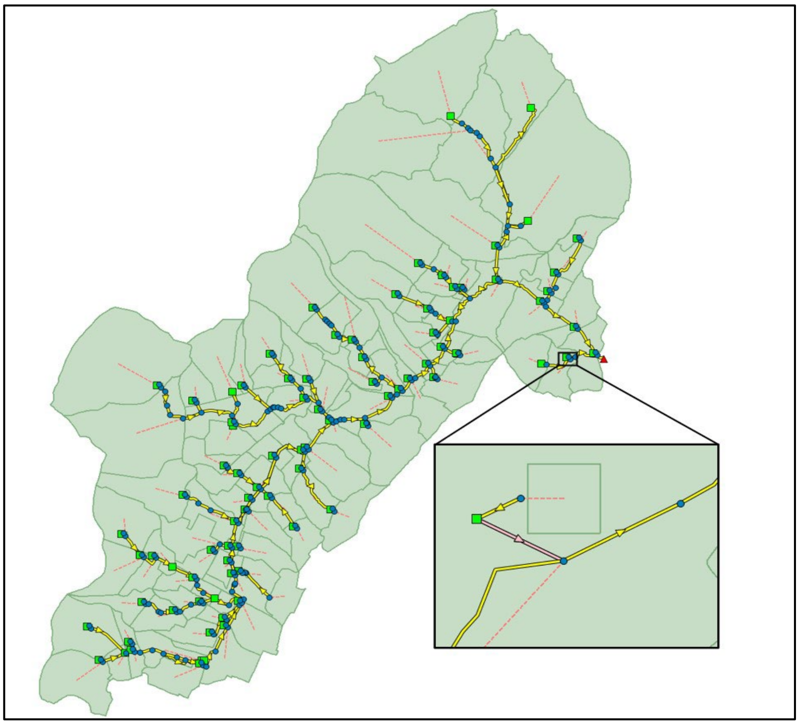

Figure 4. The SWMM model with a detail of a scenario subbasin. Twenty-five identical 2.4-acre scenario subbasins were distributed randomly throughout the model.

The selection of 25 scenario subbasins represents an estimate of the typical density of urban development projects, but does not represent the actual number of purchases in the catchment. The current quantity of actual credit purchases in Virginia is not publicly available due to the timing and nature of credit purchasing procedures, and Charlottesville does not maintain city-level records of credit purchases. A land developers' intent to purchase WQT credits is approved by VADEQ during the Construction General Permit (CGP) permitting process, but the actual number of credits purchased and bank name is not reported until the impact has occurred and the permit is terminated. CGP permit terminations occur at the end of a 5-year permit cycle. Therefore, credit purchasing information is not centrally available for all active and proposed construction permits. Data curated by Saby et al. (2021b) showed 6 credit purchases in the study area in permits that had been terminated in 2014 and 2019. Given that the NPS WQT program was established in 2012, and an increase in annual transactions is clear from available data (Saby et al., 2021a, 2021b; USACE, 2022), the available records of number of credits purchased is expected to underestimate the number of the actual credits sold. Therefore, greater than 6 purchases are assumed to have occurred in the study area. CGP records from VADEQ show that there are 151 active permits issued or reissued in the City of Charlottesville in the 2019 permit cycle, all of which may or may not include credit purchases. Given the general bounds of 6 and 151 purchase sites, 25 scenario subbasins were placed within the study area.

2.5 Scenario design

In general, the design of development sites (including WQT purchase sites) must be responsive to regulatory requirements. To represent a realistic WQT purchase site in our model, we followed calculation procedures that are required by Virginia law for regulated land developers in the commonwealth. In particular, VRRM was used to calculate the post-development TP load reduction requirements (VADEQ, 2016), and the Energy Balance method was used to calculate post-development water quantity control requirements.

Using parameters from these calculations, the SWMM model was updated to represent three policy scenarios. In this way, the scenarios in this study represent real-world actions that a Virginia land developer could take. In this subsection, the Virginia requirements used to determine SWMM model parameters are described.

2.5.1 Calculation of post-development total phosphorus (TP) reduction requirements

Calculating the post-development TP load requirements for the simulated WQT sites is the first step in scenario design. Virginia regulations require that developed areas achieve a post-development loading rate of less than or equal to 0.41 lbs TP/acre/year (Virginia Code 9VAC25-870-63(A)). This value is based on a very rough estimation of the impervious cover expected to support good stream quality (Haile, 2012). Either offsite or onsite measures can generally be used to achieve compliance.

The required TP reduction must be calculated with VRRM (VADEQ, 2016). The VRRM approach is an instance of the generalized Runoff Reduction Method (RRM), which credits the performance of stormwater BMPs based on expected pollutant efficiency and runoff volume reduction (Battiata et al., 2010). VRRM applies research-based BMP efficiencies and parameters to the RRM approach that are appropriate for Virginia. BMP efficiencies are assumed to be constant across ecoregions and position in the landscape. This system is based on treating the 1-inch storm, which is the 90th percentile rainfall depth in Virginia (VADEQ, 2016). A complete guide to the VRRM approach is included in VADEQ (2016). For concision, this section describes only the key concepts related to the calculation of pollutant values. The post-development TP load for the site is calculated according to Equation 1.

$$L = P * P_j * \frac{Tv_{site}}{R_d} * \mathrm{C} * 2.72 \qquad (1)$$

Where L is post-development TP load for site (pounds/year), P is the statewide average total annual rainfall (43 inches), P_j is the literature-based fraction of rainfall that produces runoff (assumed to be 0.9), Tv_{site} is the post-development treatment volume, R_d is the 90th percentile rainfall depth (1 inch), C is the mean concentration of TP in urban runoff, assumed to be 0.26 mg/L across all urban land covers, and 2.72 is a unit adjustment factor to convert milligrams to pounds and liters to acre-feet.

The required TP reduction is calculated as L–(0.41*A), where A is the site area in acres, and 0.41 lbs TP/acre/year is the allowable loading rate, as set forth in Virginia code. Note that since the allowable loading rate is greater than the pre-development forested loading rate (0.07 lbs TP/ acre/year), an increase in TP compared with pre-developed conditions is allowed. The scenario subbasins were assumed to fall on type B soils in VRRM. Soil type has a minimal impact on the overall load and required reduction in VRRM, but it does impact BMP design and curve number adjustments. The rationalization for each of these terms in VRRM is explained in the model documentation (VADEQ, 2016).

Using the VRRM spreadsheet, we calculated a required reduction of 2.16 lbs TP/year, or 0.90 lbs/acre developed/year for each 2.4-acre site. Each of the three scenarios in this study represent different methods of achieving compliance with this required reduction of 0.90 lbs TP/ acre developed/year.

2.5.2 Calculation of post-development water quantity reduction

Calculating the post-development requirements for water quantity control for the simulated WQT sites is the next step in scenario design. Water quantity requirements must generally be met onsite in Virginia. Specifically, the 1-year 24-hour storm post-development peak release rate must be evaluated using the Energy Balance method, which is intended to address both the increased peak and volume of runoff caused by development (Virginia Code 62.1-415:24). The Energy Balance method was developed in Virginia to enable onsite control of water quantity, before it is released into receiving channels. This method is designed to reduce the burden of high flows on the receiving channel, and help ensure adequate channel stability downstream (Rolband and Graziano, 2012). The Energy Balance method requires that runoff from developed sites adhere to Equation 2.

$$qp(\text{1 year allowable}) \leq IF * \frac{qp_{predev} * Q_{predev}}{Q_{postdev}} \quad (2)$$

Where qp is peak discharge, IF is the improvement factor, Q is runoff depth, and *postdev* and *predev* subscripts indicate post-development or pre-development conditions. The IF is 0.9 for projects less than one acre, and 0.8 for projects greater than or equal to one acre. The IF values are intended to ensure that redevelopment projects improve the site and retrofit with stormwater management technology as needed. Currently, Virginia is the only state to have adopted the Energy Balance method for stormwater control requirements, though at least 12 other states have incorporated volume-based restrictions rather than only reducing peak flow (USEPA, 2016). Some of these programs include minimum infiltration rates as a part of volume reduction, while others employ simple volume reduction formulas similar to the Energy Balance.

The Energy Balance formula can sometimes require highly restricted flows, particularly if the pre-development conditions are forested. To avoid requiring unrealistically low runoff rates, the rules include a caveat that the allowable developed condition peak discharge can be equal to or less than forested conditions, according to Equation 3. This avoids application of the improvement factor for forested areas.

$$qp(\text{1 year allowable}) \leq \frac{qp_{forest} * Q_{forest}}{Q_{postdev}} \quad (3)$$

Where qp_{forest} is peak discharge if the site was entirely forested, and Q_{forest} is the associated runoff depth under forested conditions. In most cases, the 10-year 24-hour storm must be controlled by matching the peak flow for the pre-developed conditions.

$$qp(\text{10 year allowable}) \leq qp_{forest} \quad (4)$$

Where qp is peak discharge.

The 100-year 24-hour storm does not need to be controlled, but facilities must be structurally sound for a storm of this magnitude. If there is a public safety concern at the site (e.g., risk of flooding a roadway), additional flow restrictions may be applied. There are assumed to be no public safety restrictions in the scenario subbasins.

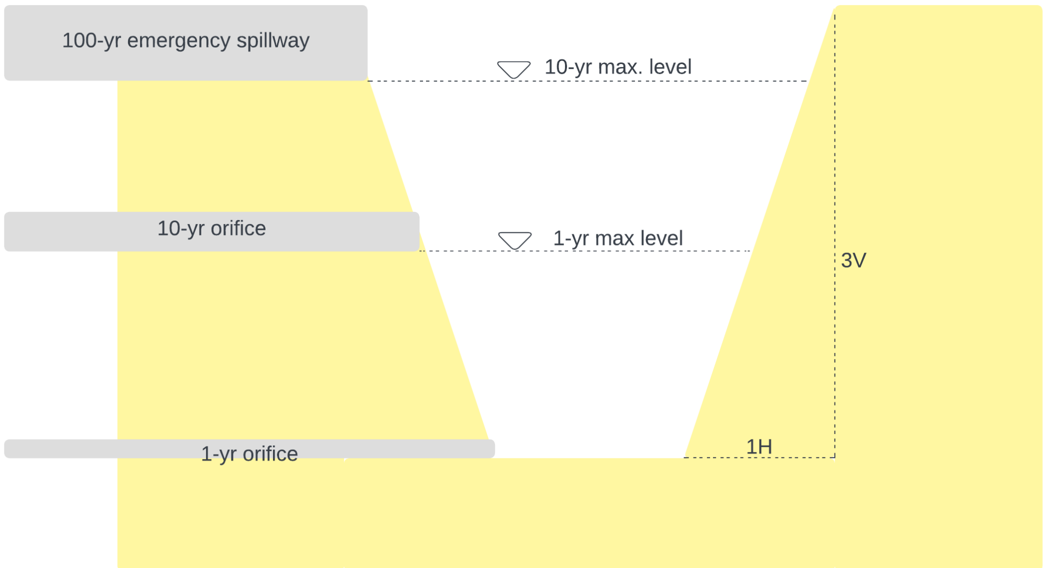

Figure 5. Diagram of the stormwater detention basin implemented in each scenario subbasin to satisfy Virginia stormwater quantity control requirements. This design was completed according to specifications in the Virginia Stormwater BMP Clearinghouse (VADCR, 2013b). All three scenarios include an onsite stormwater detention basin in the scenario subbasins.

Each of the three policy scenarios include onsite water quantity controls that comply with these requirements. Detention basins are frequently used to meet water quantity requirements because they are often the least-cost option to achieve compliance (see Nobles et al., 2017; Hirschman et al., 2009). An onsite detention basin was incorporated at each site to meet the Virginia quantity control requirements. Since the pre-developed site is forested, Equation 3 was used to calculate the required post-development flows for the 1-year 24-hour SCS Type II storm for the area. The basins also match the pre-development peak flow rate for the 10-year 24-hour design storm, and include an emergency spillway to convey the 100-year 24-hour storm. A diagram of the detention basin is shown in Figure 5. The value of the required quantity reductions, and corresponding orifices sizes, varied by scenario due to the added storage offered by onsite water quality control. Since there is no water quality volume detained in this basin, Virginia regulations assume that this detention basin would have a negligible effect on water quality.

2.5.3 Scenario 1: NPS WQT under current rules (business-as-usual, BAU)

In scenario 1, business-as-usual NPS WQT is assumed. This means that water quality credits are purchased to cover 100% of the water quality requirements, and only Virginia water quantity rules are met onsite. Saby et al. (2021b) show that buyers and sellers are collocated within the same HUC-12 catchment less than 1% of the time. Therefore, this scenario includes no water quality controls, because all mitigation is assumed to occur outside of the modeling domain.

Quantity controls were achieved using a detention basin following the diagram in Figure 5. Once the allowable flows were calculated as described in section 2.5.2, the orifice sizing was completed in SWMM. Pre- and post-development flow rate for 1-,10-, and 100-year storms was simulated using the SWMM model. Each orifice was sized by iteratively adjusting the diameter until the required outflow was achieved. Detention basin parameters are shown in Table 5.

Table 5: Detention basin parameters applied in scenario 1 for quantity control.

SCS Type II storm intensity	Maximum allowable flow (CFS)	Orifice area (square feet)	Inlet offset (feet)	Orifice type	Notes
1-year 24hour	0.33	0.04	0	Pipe	Matches peak predevelopment volume.
10-year 24-hour	7.21	2.63	2.6	Pipe	Matches peak predevelopment flow.
100-year 24-hour	n/a	7.83	5.8	Weir	Allows 1 foot of freeboard and conveys peak flow.

2.5.4 Scenario 2: WQT disallowed, all water quality controls implemented onsite

In scenario 2, both quantity and quality rules are met onsite. This scenario eliminates the WQT option for achieving water quality compliance.

Bioretention basins represent one of the most commonly used design options for meeting quality requirements in the U.S. (Liu et al., 2014). Bioretention basins have the highest potential TP removal rate and second highest total removal rates (considering both runoff volume reduction and TP load reduction) of the 15 BMPs included in VRRM (Hirschman et al., 2008). The high treatment efficiency rate of the bioretention basin facilitates compliance with treatment requirements of the design site without using additional treatment BMPs (Kavehei et al., 2021). Both a bioretention and detention basin were designed according to Virginia stormwater requirements and placed in each scenario subbasin.

Sizing of the bioretention basin was completed following procedures for Virginia stormwater design specifications (VACDR, 2013a). Following procedures required for Virginia stormwater engineers, VRRM was used to determine the percentage of the scenario subbasin area that would need to be treated to achieve the required TP reduction. In Virginia, the drainage area for a bioretention basin should be no more than 50% impervious or 2.5 acres (VADCR, 2013a). Treating 0.92 acres (40,075 square feet) of impervious area and 0.92 acres of pervious area with Level II Bioretention (77% of each cover type area, or 1.84 acres total) achieved the required reduction of 2.16 lbs/year for each site in VRRM, or 0.90 lbs TP/acre developed/year.

Sizing the Level II Bioretention facility then required calculating a total bioretention storage depth in feet according to Equation 5.

$$\text{Total storage depth} = (\text{depth})_{\text{surface}} + (\eta * \text{depth})_{\text{soil media}} + (\eta * \text{depth})_{\text{gravel}} \quad (5)$$

Where η is porosity, and dry conditions are assumed at the start of the storm. Depth and porosity for each layer are shown in Figure 6. Subsequently, treatment volume (Tv) in cubic feet was calculated according to Equation 6.

$$T_v = 1.25 * \frac{R_v * A * 1}{12} \quad (6)$$

Where R_v is the unitless runoff coefficient and is provided by VRRM, and A is the area that drains to the BMP in square feet, 1 is the rainfall depth in inches, and 12 is a unit conversion factor (inches/foot). Finally, the surface area (SA) for each bioretention basin was calculated according to Equation 7.

$$SA = \frac{Tv}{storage\ depth} \quad (7)$$

The surface area for each of the bioretention facilities was 3,254 square feet. This is calculated with a BMP drainage area of 1.84 acres (80,150 square feet), porosity and depths shown in Figure 6, and an R_v of 0.58.

Bioretention effectiveness in SWMM is sensitive to the underlying soil conditions that control infiltration rates. To test a range of realistic infiltration rates, we adjusted the conductivity of surrounding soils in four subscenarios. Subscenario 2A assumes type A soils with conductivity of 0.7 inches/hour surrounding the bioretention. For soils with this conductivity, no underdrain is required in the bioretention basin (VACDR, 2013a). In scenario 2B we reduced the conductivity to 0.5 inches/hour to represent type B soils, which would not require an underdrain. In scenario 2C, we added an underdrain and storage area and reduced the conductivity of the surrounding soils to 0.3 inches/hour to represent type C soils. In scenario 2D, we again included an underdrain and storage area, and reduced the conductivity of the surrounding soils to 0.1 inches/hour to represent type D soils. We assigned a bioretention treatment efficiency of 50%, as specified in VRRM, which is applied to any outflow from the underdrain in scenarios 2C and 2D. Scenarios 2A, 2B, 2C and 2D are summarized in Table 6. Each of these subscenarios can be considered a reasonable implementation of bioretention basins in the study area.

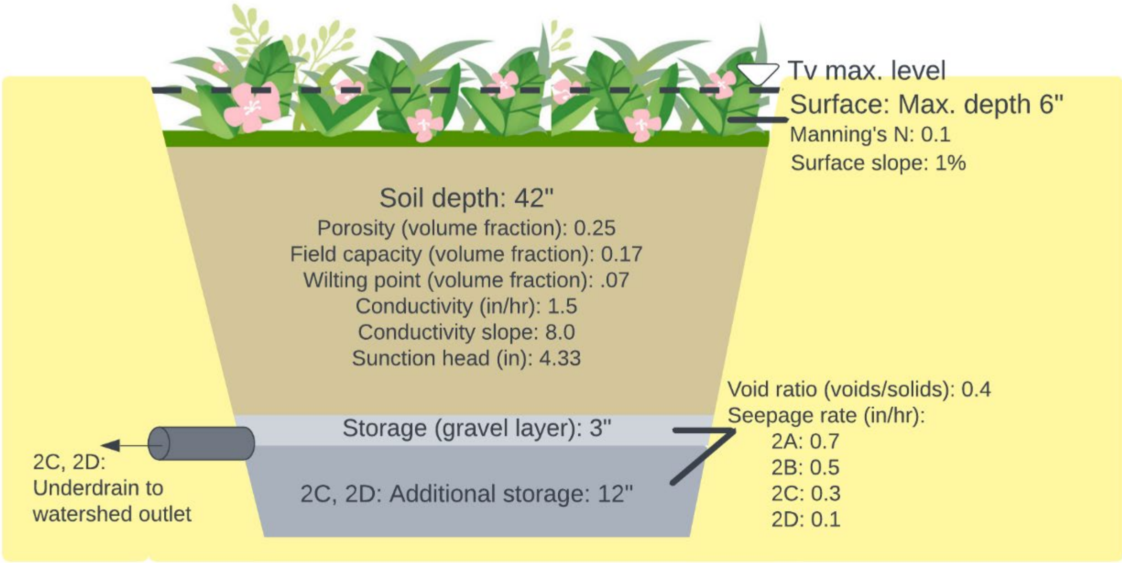

Figure 6. Diagram of features implemented in scenario 2 for onsite quantity control. Parameters are based on Virginia requirements for Level 2 bioretention. Infiltration rates were adjusted across subscenarios 2A, 2B, 2C, and 2D, and an underdrain included in scenarios with seepage rates lower than 0.5 inches/hour. Facilities were designed to remove 0.90 lbs TP/acre/year.

Table 6. Bioretention basin parameters changed between scenarios 2A, 2B, 2C and 2D.

Scenario name	Underdrain present	Conductivity of surrounding soils (inches/hr)	Efficiency in underdrain (%)	Notes
2A	No	0.7	N/A	No underdrain is required for soils with this conductivity.
2B	No	0.5	N/A	No underdrain is required for soils with this conductivity.
2C	Yes	0.3	50	Surrounding soils conductivity is reduced, and some flow exits through underdrain.
2D	Yes	0.1	50	Surrounding soils conductivity is reduced further, increasing flow through underdrain.

Bioretention basins increase stormwater storage, so we recalculated the energy balance requirements with an adjusted curve number. This resulted in a slightly higher allowable flow compared with scenario 1, as shown in Table 7.

Table 7. Detention basin parameters applied for quantity control in scenario 2 (all subscenarios). Due to the increased storage from the bioretention basin, allowable flows are slightly greater and the detention facility size is reduced compared with scenario 1.

SCS Type II storm intensity	Maximum allowable flow (CFS)	Orifice area (square feet)	Inlet offset (feet)	Orifice type	Notes
1-year 24hour	0.41	0.054	0	Pipe	Matches peak predevelopment volume.
10-year 24hour	7.21	2.83	2.1	Pipe	Matches peak predevelopment flow.
100-year 24-hour	n/a	10.22	3.6	Weir	Allows 1 foot of freeboard and conveys peak flow.

2.5.5 Scenario 3: WQT with local banks, offsets generated within the catchment

In this hypothetical scenario, scenario subbasins are identical to scenario 1, with onsite control of water quantity and offsite WQT purchases. However, unlike scenario 1, all credits are purchased from banks established within the local urban catchment instead of from agricultural areas outside of the urban area.

In this scenario, NPS WQT credits are generated through the conversion of impervious area to forest. Studies have demonstrated how urban population shrinkage in many American cities offers an opportunity to add urban green space and address stormwater management problems within cities (Desimini, 2013; Haase et al., 2014; Schilling and Logan, 2008). Vacant land within cities has been shown to infiltrate 51-54% additional annual runoff compared with the pre-demolition conditions (Kelleher et al., 2020), and some cities across the northeastern and mid-Atlantic U.S. have vacancy rates of greater than 10% (Schilling and Logan, 2008).

To create the urban WQT banks, impervious area in the model was converted to forest according to official WQT credit generation rates. The per-acre credit value is calculated by VADEQ based on CAST. VADEQ generated WQT conversion values in CAST using the average per-acre TP loadings for each land use in each HUC-8 in Virginia. The number of credits awarded for each acre of conversion is calculated as the difference between edge-of-tide average loads for forested and impervious areas. For example, in the Rivanna watershed containing the study area, CAST estimated that an acre of impervious land contributes 0.49 pounds more TP per year to the Chesapeake Bay more than an acre of forest. Therefore, VADEQ awards 0.49 NPS WQT credits for each acre of impervious acre converted in the Rivanna (VADEQ, 2020). In SWMM, the required amount of impervious land cover (non-road) was converted to forested land cover, and the percent of impervious cover in each subbasin was reduced accordingly. The converted area was distributed by uniform percent throughout the existing 91 subbasins in the watershed using an R script. The adjusted subbasins with reduced impervious area were then used in the SWMM model.

2.6 Model simulation and post-processing

The model was run for each scenario using an hourly precipitation time series for the period October 1, 2012, to September 30, 2021, totaling 9 water years, to drive the model. These data were obtained from the National Oceanic and Atmospheric Administration (NOAA) climate data repository for the Charlottesville-Albemarle Regional Airport site (NOAA station ID WBAN:93736), and are shown in Figure 7. Over this nine-year period, the average annual precipitation ranged from 39 inches (water years 2011-2012 and 2015-2016) to 58 inches (water year 2017-2018), with a mean of 49 inches.

The SWMM output files were processed in R using the SWMMR package (Leutnant et al., 2019; R Core Team, 2017). Removal efficiency was used to describe the effectiveness of bioretention, is which is a common metric used calculated as a percentage decrease in pollutant load entering and exiting the facility (Wang et al., 2019).

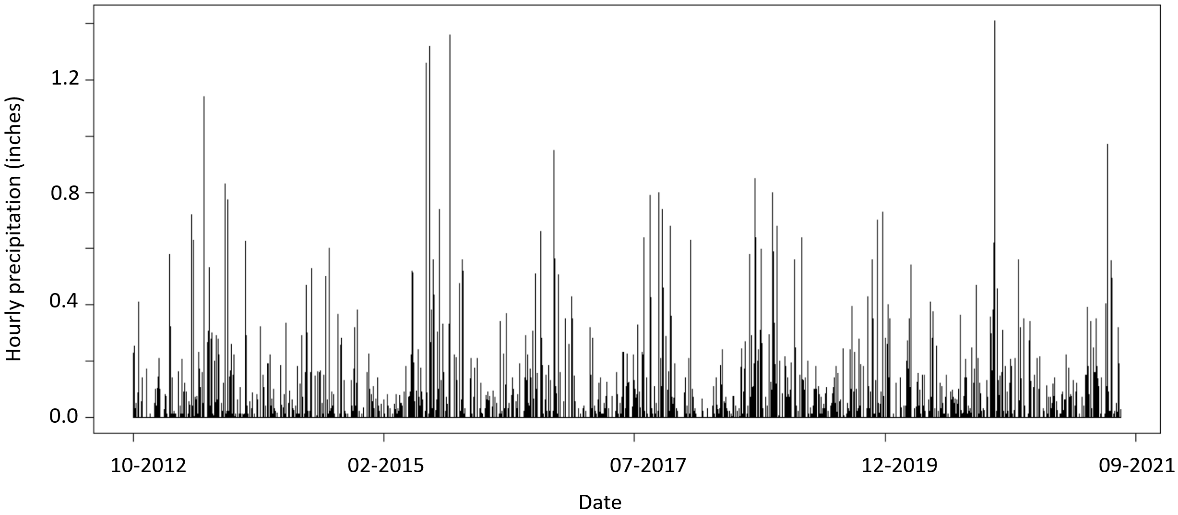

Figure 7. Precipitation data used in the model from National Oceanic and Atmospheric Administration station ID WBAN:93736, approximately 5 miles north of the study area centroid. Average annual precipitation was 49 inches. The model calibration storm was in April, 2021, and validation storms were in March through August, 2021.

3. RESULTS

Results are presented as a comparison in TP loading rates for scenario 1 (NPS WQT BAU) and scenario 2 (onsite bioretention, no WQT), followed by scenario 1 compared with scenario 3 (local trading policy alternative). Daily and watershed-level comparisons are presented subsequently.

3.1 TP reduction for onsite bioretention (Scenario 2) compared with BAU WQT (Scenario 1)

Scenario 2 (onsite bioretention) reduced TP loads by 0.42 to 0.92 lbs/acre/year, with an average removal rate of 0.72 lbs/acre/year across scenarios 2A, 2B, 2C and 2D, as shown in Figure 8 panel A. This corresponds with an annual TP removal efficiency of 45% to 83%, with a mean of 65% for the bioretention basin drainage area, shown in figure 8 panel B. The effectiveness of the onsite bioretention depended on rainfall, with years of higher precipitation showing a higher per-acre reduction (Figure 9). In all subscenarios, the expected reduction from bioretention increased by 0.02 lbs/acre treated/year for each inch of additional rainfall. VRRM assumes a constant reduction in TP load of 0.90 lbs/acre/year for this scenario, with an annual rainfall of 43 inches.

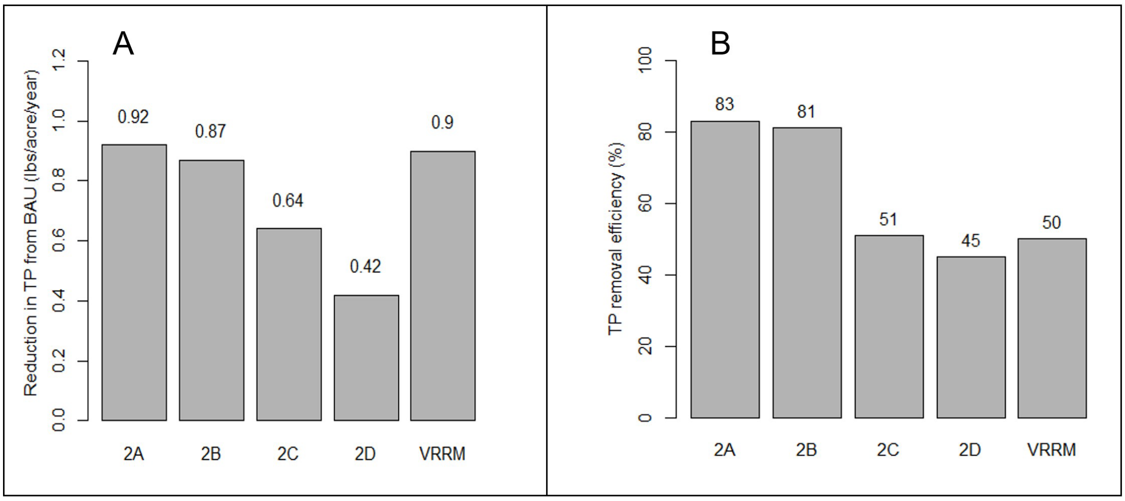

Figure 8. Comparison of TP load reduction (panel A) and removal efficiency (panel B, percentage reduction in TP loads entering and exiting the facility). Subscenarios A-D represent type A-D soils with increasing runoff potential.

It is also useful to present results in terms of the catchment TP increase for each credit purchased. Since 0.9 credits were needed for each acre developed, TP loading increased by an average of 0.8 lbs TP/year for each 1.0 NPS WQT credit purchased. Depending on rainfall and soil type, this value ranged from 0.23 to 1.22 lbs TP/year for each NPS WQT credit purchased.

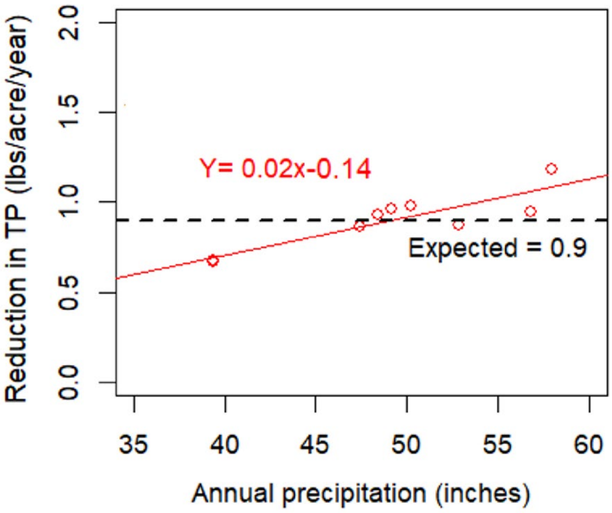

Figure 9. Scenario 2B annual reduction in TP loads generated by the onsite bioretention basin for each acre of development treated. The mean value is 0.87 lbs/acre/year. The "Expected" value is the amount calculated using the Virginia Runoff Reduction Method (VRRM) and used in the design of bioretention basins in the model.

In the SWMM model, there are three routes that runoff can take through and bypassing the bioretention basin, each contributing to the overall TP removal. First, runoff can infiltrate into the bioretention basin and exfiltrate into surrounding soils, with an assumed 100% treatment efficiency. Second, runoff can infiltrate into the bioretention basin and exit through the underdrain, if present, with a user-defined TP removal efficiency of 50%. Third, runoff can be routed around the bioretention basin if the treatment volume is exceeded during an intense storm. This bypassed runoff is not treated, so has a 0% TP removal rate.

The proportion of runoff contained in each of these flow paths, with a 0%, 50%, or 100% TP removal efficiency, respectively, explains the differences in loading rates and removal efficiencies shown in Figure 8. In VRRM, 100% of runoff from the BMP drainage area is assumed to enter the bioretention facility, and 50% of the TP contained is assumed to be removed. Therefore, while the VRRM efficiency of 50% is directly assigned in the formula for the annual load calculation (Equation 1), the SWMM efficiency depends on dynamic simulation of runoff that determines the proportion of flow between the three flow paths.

According to Virginia BMP specifications, bioretention basins in areas with soil conductivity greater than or equal to 0.5 inches/hour (Type A/B soils) do not require an underdrain. In scenario 2A, soils surrounding the bioretention basins in the SWMM model were assigned a conductivity of 0.7 inches/hour, and all runoff that entered the bioretention exited via exfiltration. In scenarios 2B, 2C and 2D, conductivity of surrounding soils was decreased to 0.5, 0.3, and 0.1 inches/hour, respectively. An underdrain was added in scenarios 2C and 2D. As conductivity was reduced, more flow exited through the underdrain and overall efficiency was reduced, as less water exited the system via infiltration. Therefore, the effectiveness of bioretention in SWMM compared with VRRM depended on the amount of runoff that is allowed to exfiltrate compared with the amount of runoff that flows through the underdrain with the user-defined efficiency. Scenarios 2A and 2B showed very similar TP removal rates. This is because infiltration was not generally limited by the conductivity in the surrounding soils with rates of greater than 0.5 inches/hour, so reduced conductivity did not limit infiltration in scenario 2B.

3.2 TP reduction with local trading (Scenario 3) compared with BAU WQT (Scenario 1)

Scenario 3 explores the potential for conversion of urban impervious areas to forest (WQT credit sellers) to offset the impact of development (WQT credit buyers) within the local watershed. This is our policy alternative scenario. Subsection 2.2.5 provides a detailed description of this scenario development.

The SWMM model included 25 WQT purchase sites that each required 2.16 lbs TP WQT credits for compliance. Therefore, a total of 54 TP credits were needed to meet the required reduction for the development in the watershed. VADEQ assigns a per-acre credit rate of 0.49 lbs TP/acre converted/year, calculated with CAST for the Rivanna HUC-8 watershed. Therefore, 110.2 acres of impervious area needed to be converted across the watershed, or 1.84 acres of impervious conversion for each acre of mixed impervious and turf development. 110.2 acres comprises 6.5% of total impervious area and 10.6% of the non-road impervious land cover in the study area. This large theoretical mitigation area relative to the development area reveals disparities in TP loading in VRRM and CAST demonstrated in Figure 10.

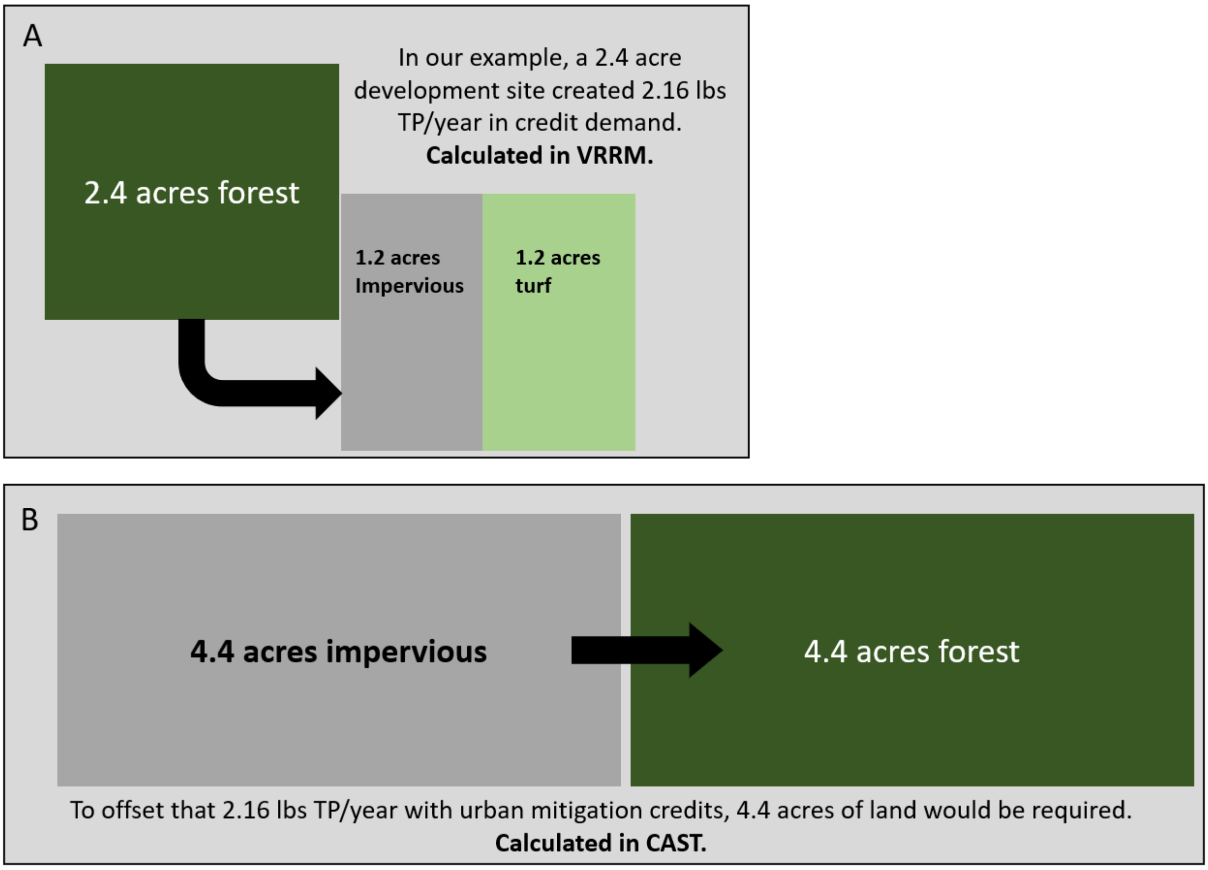

Figure 10: Demonstration of requirements for NPS WQT credit buyers (A) and sellers (B). This shows disparities in VRRM and the Chesapeake Assessment and Scenario Tool (CAST) that lead to large estimated reductions for scenario 3.

When implemented in SWMM, scenario 3 resulted in an average annual TP reduction of 3.9 lbs/year for each acre developed, with greater reductions in years with more precipitation, as shown in Figure 11. This is more than 4.3 lbs of TP reduced for each 1.0 impervious conversion WQT credit purchased.

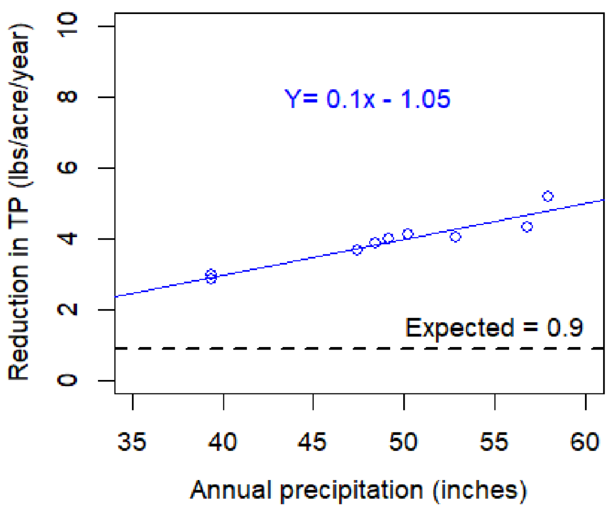

Figure 11: Annual reduction in TP for each acre of development treated in scenario 3. The mean value is 3.9 lbs/acre/year. The "Expected" value is the amount calculated using VRRM, and is the amount of NPS WQT credits required per acre.

The first reason for this disparity is the edge-of-tide (EOT) crediting procedure. NPS WQT urban conversion credits are currently calculated to offset loads at the Chesapeake Bay, rather than locally. Therefore, the difference between EOT and edge-of-stream (EOS) loadings can account for some of the overcompliance. To assess the impact of the delivery ratio, we calculated the EOT:EOS ratio for TP in the Rivanna watershed using CAST. A ratio of 0.58 for the 2020 progress scenario was found (Chesapeake Bay Program, 2020). When this ratio is applied to the 0.49 lbs/acre/year used to award NPS WQT credits, the value is increased to 0.84 lbs/acre/year at EOS.

VRRM was also used to calculate the per-acre increase in TP for an acre of forest converted to 100% impervious area. In VRRM, the per-acre loading for a forested acre is 0.07 lbs TP/year, and the loading for TP for an acre of impervious area is 2.17 lbs/ year. Therefore, the change in TP for a 1-acre conversion is therefore 2.10 lbs/acre/year. For each acre of impervious area converted in SWMM, 2.11 lbs TP/year were reduced. The change in TP found for an acre of forest to impervious area for CAST (EOS), VRRM, and SWMM are compared in Figure 12. Figure 12 shows CAST EOS, rather than EOT which is used in crediting. This is to demonstrate that the asymmetry between VRRM and CAST cannot be explained by the delivery ratio alone. The delivery ratio accounts for 0.35 lbs/acre/year, or 22% of the difference between VRRM and CAST, but the VRRM estimate remains 2.5 times greater than the CAST EOS estimate. A per-acre difference of 1.26 lbs TP/acre converted/year between CAST and VRRM remains after the delivery ratio is applied.

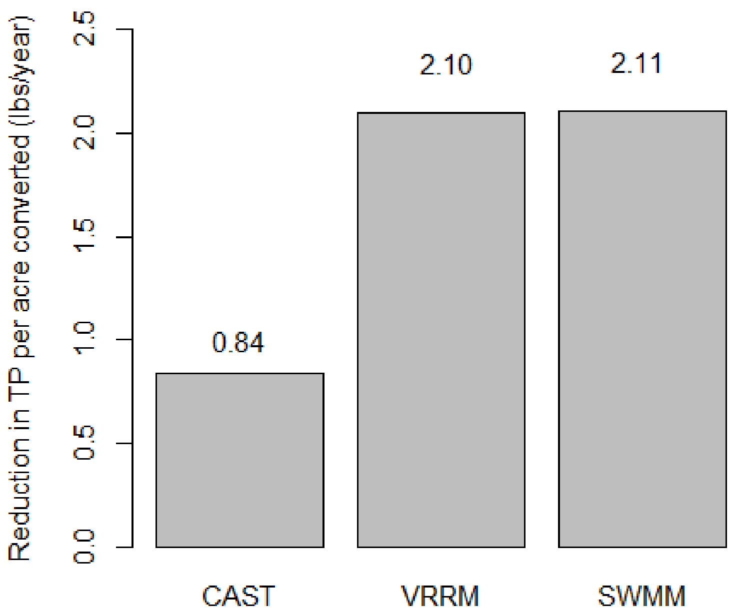

Figure 12: Comparison of per-acre change in TP loading estimates when one acre of impervious area is converted to an acre of forest in the study watershed. The CAST loading is edge-of-stream (EOS) for the Rivanna watershed. Edge of tide (EOT) estimates are currently used for WQT crediting, with an annual reduction of 0.49 lbs/acre converted/year.

SWMM and VRRM estimates were very similar, at 2.11 and 2.10 lbs TP reduced/acre converted/year, respectively. This level of agreement suggests that VRRM estimates do a good job of summarizing the complexity included in the SWMM model at the annual scale. In particular, this includes land use EMC heterogeneity and changes in runoff related to development. The SWMM model includes adjustments to TP loading rates due to flow changes, as well as due to land-use specific EMCs (0.32 mg/L for impervious, 0.01 mg/L for forested land cover). VRRM relies only on flow changes to adjust the TP loading, with a constant EMC of 0.26 mg/L across forested and impervious land uses. Therefore, changes in the selected EMC values in the SWMM model could have altered the SWMM estimated load reduction, and created a disparity with the VRRM model.

The SWMM model also did not include all environmental conditions that could have impacted this result. Figure 11 demonstrates how SWMM estimates vary depending on annual rainfall, but a range of other environmental conditions not included in either model could impact loading rates. For example, the position of land use conversion in the watershed, rainfall climatology, and groundwater characteristics could all impact the quantification of TP loads for impervious conversion to forest.

3.3 Comparison of daily responses and hydrologic impact

We assessed TP export and total runoff at the daily timescale. Mean values for scenarios 2A, 2B, 2C and 2D are used to represent scenario 2. For scenario 2, Figure 13A and Figure 13B both show an increasing reduction up to 1 inch of rainfall, then a constant reduction for days with greater rainfall. Bioretention basins offered minimal additional pollution control for larger storms, with precipitation after one inch bypassing the bioretention basin. This pattern aligns with expectations, because bioretention facilities in the model were designed to control runoff from the 1-inch storm according to Virginia rules. Other Virginia BMPs designed to control the 90th percentile storm could be expected to show a similar pattern.

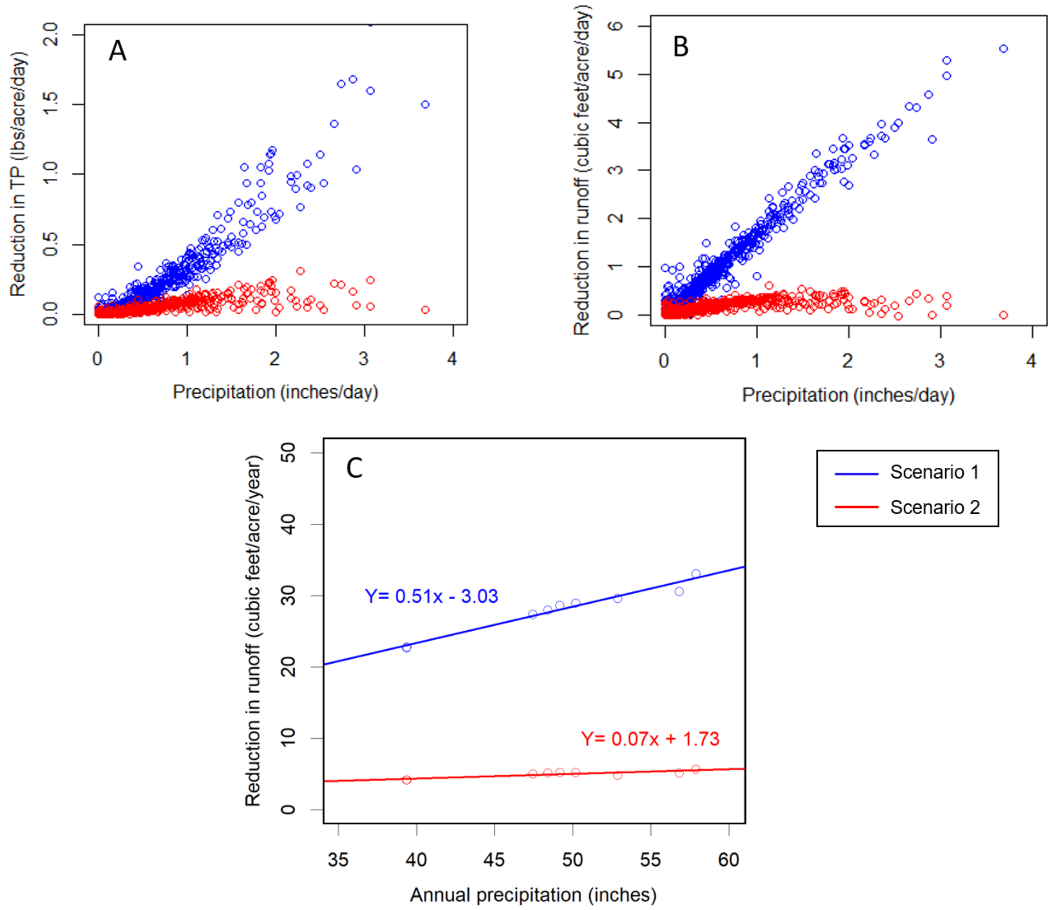

Figure 13. Daily reduction in TP load (A) and runoff (B) for each acre of development, and annual runoff reduction (C). Scenario 2 (onsite bioretention) offered TP and runoff control for precipitation events less than 1 inch in magnitude, while scenario 3 (impervious conversion) continued to control TP and runoff for larger storm events.

The reduction in TP load and runoff associated with scenario 3 continued to increase across storm events, with larger storm events showing a larger TP reduction per acre compared with smaller storms. At the annual scale (Figure 13C) greater reductions were also observed with greater annual precipitation, with the infiltration benefits of impervious conversion increasing linearly with annual precipitation but remaining fairly constant for bioretention. In practice, ground and surface water interactions would limit the effectiveness of infiltration in reducing runoff, an effect that is not captured in this model.

The difference in runoff between scenario 1 and scenario 2 is minimized because the added storage from the bioretention basin in scenario 2 is factored into onsite quantity requirements. As described in the Methods section, design of scenario 2 included re-analyzing energy balance runoff control requirements with the added onsite storage offered by bioretention. This resulted in a higher allowable flow from the development site for scenario 2 compared with scenario 1 (0.4 cfs instead of 0.33 cfs). This saves project costs by reducing the required size of the detention basin, but limits the added hydrologic benefit of the bioretention basins.

3.4 Overall catchment response

Figure 14 shows the percentage reduction in average annual TP load and runoff at the watershed outlet. This is the cumulative catchment-scale impact of the 25 2.4-acre scenario subbasins throughout the study area. Scenario 2 results are averaged across scenarios 2A, 2B, 2C, and 2D. Onsite bioretention basins offered a limited but statistically significant impact on hydrology at the watershed scale in the scenario assessed ($p<.05$), with an average annual reduction in total runoff of 1.0% (Figure 14A). A 0% reduction in runoff was expected since 100% of quantity control requirements are met onsite in all scenarios, including scenario 1, BAU. Overall, scenario 3 showed good potential to control stormwater quantity, with an average annual reduction in flow of 5.7%. This is the result of widespread impervious conversion implemented in scenario 3, which reduced imperviousness by 6.5% across the study area watershed. The quantity control benefits described here are not currently valued as a part of a NPS WQT credit.

Onsite bioretention basins resulted in a 0.9% reduction of TP loads at the watershed outlet compared with WQT BAU, or an average reduction of 43 lbs/year out of a total of 4,778 lbs/year total load. Impervious conversion reduced TP by 4.7% or 233 lbs/year. The Expected TP reduction is the amount calculated using VRRM, which represented 1.1% of the overall TP loads in the watershed or 53 lbs/year.

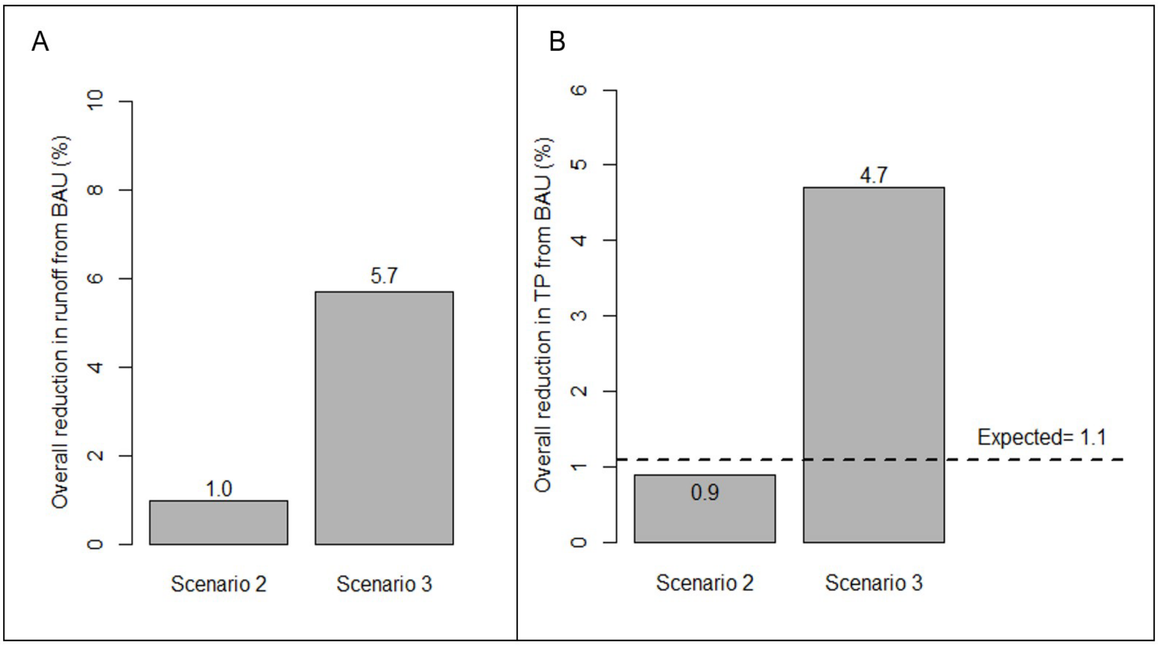

Figure 14. Reduction in annual runoff (A) and TP load (B) and at the watershed outlet compared with scenario 1, BAU NPS WQT. In panel A, the expected reduction in runoff is zero because quantity control requirements are met onsite. In panel B, the Expected TP reduction is the required reduction, which was 1.1% of the overall watershed loads calculated in SWMM. Scenario 2 results are averaged across 2A, 2B, 2C, and 2D.

4. DISCUSSION

4.1 Inconsistent TP load calculations in CAST and VRRM

The comparison of scenario 1 (BAU) and scenario 3 (impervious conversion) shown in Figure 11 indicates that impervious conversion WQT credits may currently represent considerable overcompliance with Virginia water quality requirements. This is due to inconsistency between calculations with CAST and VRRM. Model assumptions in both VRRM and SWMM may lead to overestimation of the TP decrease associated with converting impervious area to forest. In particular, these models both rely on increased infiltration to reduce TP loads, but do not include representation of groundwater and surface water interactions that could limit the effectiveness of infiltration in reducing TP and peak flows. CAST includes groundwater representation and is based on extensive data, but is modeled at a much larger scale and does not include gauge data from the study area. This limits the applicability of CAST for site-specific inquiries. This lack of local applicability could be a topic of discussion for future versions of the Chesapeake Bay Model.

While the relative accuracy of the three models is challenging to gauge, a key takeaway of this analysis is the inconsistency itself. Since CAST estimates lower TP than VRRM, urban conversion WQT credits are undervalued and disincentivized in the current program. Results from SWMM show agreement with VRRM (Figure 12), and result in the considerable watershed-level impact shown in the overall response (Figure 14). These quantitative loadings cannot be validated without real-world data, but qualitatively, the SWMM result presented here does suggest that a reasonable offset for land disturbance may be found within the catchment through converting impervious areas to forest. Incorporating the local benefit into per-acre credit value could incentivize this action, and allow better equivalency between buyers and sellers.

In many WQT programs, trading ratios are applied to account for uncertainty in loading rates between buyers and sellers (Breetz et al., 2004). The lower TP estimate in CAST compared with VRRM could be considered to account for uncertainty in a similar manner, by requiring a greater reduction by bankers than by purchasers. However, using disparate model estimates for buyers and sellers is not a methodical way to apply a trading ratio. This method can lead to random and variable requirements for buyers and sellers across the state that cannot usually be verified with data. Results presented here suggest that inconsistency between VRRM and CAST results in a more conservative trade than would likely be required to offset loads within this urban zone.

We emphasize that this analysis and comparison with VRRM is limited to impervious conversion crediting procedures, and does not apply to the agricultural conversion credits. VRRM cannot be used to estimate agricultural loading rates, and we did not compare CAST loadings for agricultural conversion with any other method. Future work should review the crediting procedures for agricultural areas.

4.2 Policy implications and proposed "nested" NPS WQT

Results demonstrate how local pollutant load increases from NPS WQT can occur in urban areas as a result of spatially-clustered purchases at land development sites. This cumulative impact is not likely to be offset by local urban credit generation, at least in part because of incongruity between VRRM and CAST disincentivizes creation of urban banks. In some areas, there could be a need for greater protections to uphold Virginia's water quality goals and avoid degradation from pollutant loads, especially as participation in the NPS WQT program increases across the Commonwealth.

While increased TP loads alone do not necessarily constitute degradation, these loads may contribute to aquatic habitat stressors present in some urban areas. Clean Water Act (CWA) anti-degradation rules limit the ways in which waters of the United States can be degraded. Under CWA Section 101(a), designated waterbody uses must be protected, including recreation, wildlife habitat, and natural resources. If water quality is better than the minimum to protect the designated uses, some degradation may be allowed (Virginia Code 9VAC25-260-30). In Virginia, all waters are protected for recreation, aquatic life, wildlife, and marketable natural resources. In urban areas where these designated uses are compromised, it may be reasonable to examine whether allowing NPS WQT purchases is congruent with achieving water quality goals.

Several potential solutions could help mitigate increased pollutant loadings from NPS WQT and avoid risk of degradation. First, credits could be required to be generated upstream of the land development impact. This is already a requirement for streams with a nutrient-related TMDL in place, but not for most Virginia streams. This safety measure is a part of other trading programs, and might provide a good solution for some urban areas. However, an offset located at a distant upstream location cannot be assumed to provide a reasonable offset for the unique sources and pollutant transport patterns that are found in urban areas. For example, an urban bioretention system has been shown to reduce loadings of urban oil, road salts, and heavy metals (Li and Davis, 2009), while WQT mitigation banks, whether upstream or downstream, are unlikely to mitigate these pollutants in urban catchments.

Scenario 3 results suggest that more localized trading may be able to help advance water quality goals at the catchment-scale. Conversion of impervious surfaces to forest offered combined water quality and hydrology benefits. In addition to nutrient reduction and runoff control, other benefits of urban reforestation could be expected, such as reduction of the urban heat island effect, reduction of urban influence on regional precipitation patterns, and social benefits of urban green space (McGrane, 2016). To harness this potential, we introduce the concept of a "nested" WQT approach. In a nested WQT program, areas at risk for a local pollutant hot-spot would be subject to different rules than the surrounding low-risk area. Specifically, offset credits would be valued based on the local impact, rather than the impact at the river basin outlet. This could incentivize local trades and allow NPS WQT to help meet more localized water quality goals.

In Virginia, a nested NPS WQT program would include that land developers in urban or urbanizing areas would not be permitted to purchase credits from outside of urban zones (Figure 15). Credits generated in urban areas would be valuated based on local, rather than EOT, loads so that they are accurately incentivized based on the local benefit. This could offset the higher costs of implementing urban mitigation practices, and lead to creation of a highly-local market where the cost of water quality improvement is minimized within each urban zone, instead of being minimized across the entire HUC-6 river basin. Trading as usual could continue outside of urban areas, where risk of hot-spots is limited. These urban credits could be generated by the burgeoning community of WQT bankers in Virginia, who could work with stormwater managers and community members to determine high-priority and low-cost actions that could generate credits. An annual pollutant load reduction for a proposed project could be modeled and agreed upon, and credits subsequently approved for sale within the urban area. By allowing communities and stormwater managers to decide where WQT banks are placed across urban landscapes, social equity and municipal water quality goals can be integrated with NPS WQT. Assessment of urban land prices costs for a wide range of mitigation practices is needed to establish the feasibility of this market approach in Virginia.

Key to any NPS WQT program is to mitigate opportunities for unintended consequences. For nested WQT, this involves three key areas. First, the program must incentivize water quality improvement actions that would not otherwise have occurred. In a nested program, this could mean avoiding awarding credits for actions that would have occurred as a part of other government-led programs to improve urban water quality. This could also mean developing a strategy for stacking credits with other municipal programs to offset costs and encourage greater participation. Clear guidelines should be put in place to guide urban credit development and prevent redundancy.

Second, a nested program would need to account for placement of offset and disturbance sites within the catchment. Within urban zones, areas with lower land values are more likely to become mitigation sites. To account for social implications of this, credit and purchase approval processes should include meaningful public participation to ensure that the program incentivizes actions that residents approve of and benefit from. There are also environmental implications for market-based site selection. Many WQT programs apply trading ratios to account for uncertainty between buyers and sellers. In a nested system, a trading ratio could be applied to reduce risk associated with heterogeneity and uncertainty across the catchment. For example, if 0.90 credits are needed to meet Virginia rules, a 2:1 trading ratio would require the purchase of 1.8 credits to ensure good environmental outcomes. The outcomes of subscenarios 2A, 2B, 2C and 2D in Figure 8 are one demonstration of how variable the impact of onsite controls can be based on local soil conditions for a single development scenario. Trading ratios should be based on a risk analysis that takes into consideration heterogeneity of pollutant loading across the landscape and over time.

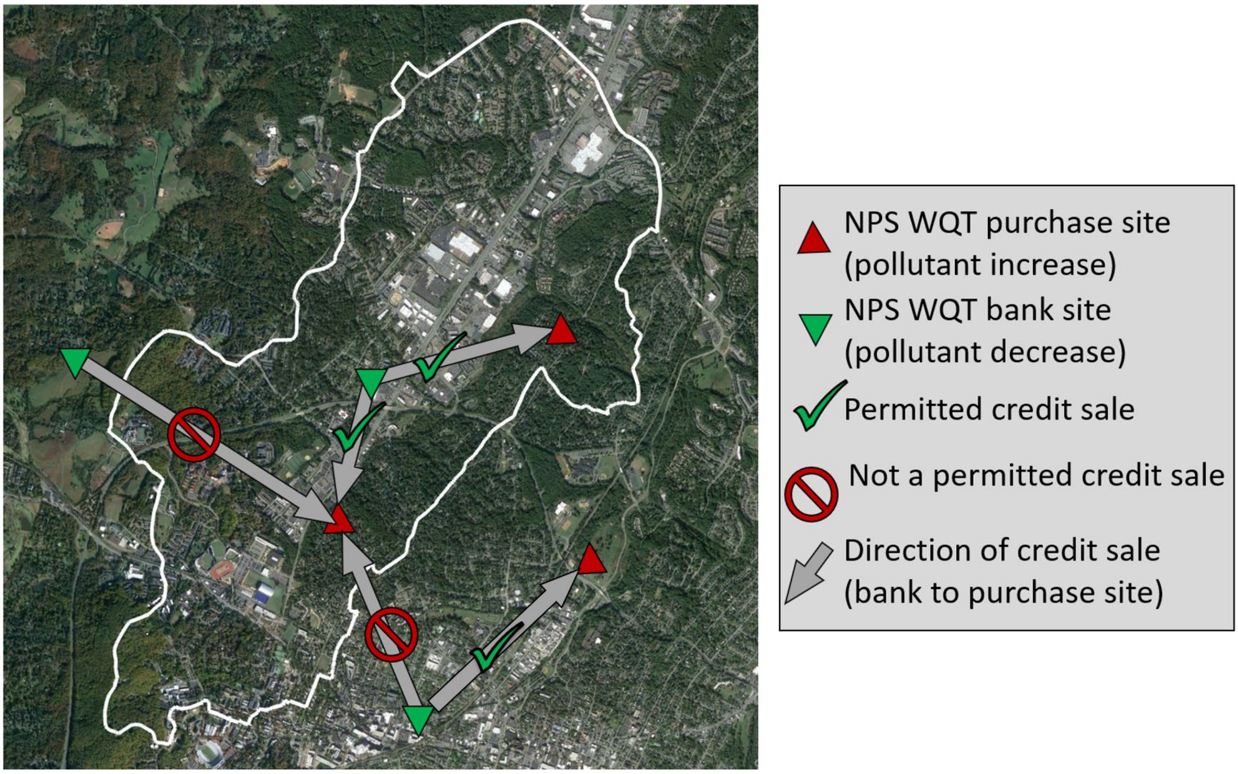

Figure 15. Illustration of a potential "nested" NPS WQT program in the study area watershed. In this proposed system, land developers seeking to purchase NPS WQT credits would do so from banks within the urban or urbanizing catchments deemed at-risk of degradation. This increases the incentives for mitigation actions within a target area to alleviate localized pollutant increases. Banks outside of the target area could continue to sell to other areas outside of the area, as shown.

Third, a nested program would need to mitigate the possibility of incentivizing urban sprawl or creating inequity across the state. If the cost of water quality control for developers in urban areas becomes considerably higher than the cost of developing outside of urban limits, sprawl is incentivized. If the cost of urban development becomes considerably higher in one locality than another, inequity is introduced. Localized economic analysis as well as state-led guidelines would need to be a part of program development to ensure that these issues are not introduced.

Implementation of more complex rules associated with a nested WQT program or a similar urban incentive could lead to confusion among market participants and lowered compliance and enforcement. Automation of purchasing and approval processes could facilitate implementation of new rules, while reducing transaction costs and increasing program transparency (Saby et al., 2021a). For example, development of an online WQT marketplace would streamline enforcement and compliance of trading boundary rules. Offering automated applications and procedures for market participants is an important first step to ensuring good program outcomes.

This study centers on impervious conversion to forests as a method to produce water quality credits in urban areas, which may not be the most cost-effective option to drive an urban WQT market in many areas. In many cases, less land-intensive options, such as generating credits through an offsite bioretention basin, could be more economically favorable. In Virginia, credits can be created for any BMP approved by the Chesapeake Bay Program (9VAC25-900-10; see Chesapeake Bay Program, 2018, for details). Currently, NPS WQT credits for all practices are valued based on the estimated pollutant load offset at the Chesapeake Bay (CAST EOT), rather than local loads.

Therefore, the value of credits for other credit-generating BMPs could be revaluated to be more equivalent with onsite mitigation practices, as described in this study. Future work can re-estimate an appropriate mitigation value for a range of urban BMPs to accurately capture local benefits and incentivize urban WQT banks.

The relative cost of mitigation actions across an urban area would determine the feasibility of a nested WQT program for a given area. This study does not provide this cost analysis. The cost of urban land and construction of BMPs can vary greatly between and within cities. The intent of this study is to provide insight into the environmental outcomes of NPS WQT and potential paths forward for credit generation and program design. Cost assessment falls outside of this scope but is an important topic for future work.

4.3 Calculation method for equivalent loading rates for NPS WQT bankers and purchasers

The key benefit to credit bankers in a nested NPS WQT model in Virginia would be that offset credits within the urban zone would be calculated based on the local impact, consistent with onsite requirements for land disturbance, rather than the impact at the river basin outlet. Recalculating pollutant loading estimates could help achieve greater consistency between buyers and sellers as a first step to incentivizing urban mitigation credits in Virginia.

We demonstrate the effect of using one model for both buyer and seller requirements in Table 8. Here, we calculated loading rates for NPS WQT bankers (Hypothetical TP credit) and purchasers (Hypothetical TP requirement) using CAST (EOT and EOS), VRRM, and SWMM. The Hypothetical TP credit column uses data presented in Figure 12, comparing the outcomes of CAST, VRRM, and SWMM. The Hypothetical TP requirement column allows a post-development loading rate of 0.41 lbs TP/acre developed/year, in accordance with Virginia code 9VAC25-87063. This allowable load is assumed to be constant across scales measured in VRRM, SWMM, and CAST EOS. The CAST EOT estimate is adjusted for delivery to the Bay (EOT:EOS is 0.58).

The trading ratio in Table 8 is provided to account for areas, such that if the trading ratio is applied to the Hypothetical TP requirement, the area required to offset new development would equal the area converted for credits for equivalent land uses. This ratio could be increased to account for variable site conditions that could influence equivalency between buyers and sellers, including the higher relative ecological value of established forest compared with new plantings. For reference, under current rules, one acre of 100% impervious development has a required reduction of 1.76 lbs TP/acre/year. With credit generation rates of 0.49 lbs TP/acre/year, this means that 3.5 acres of mitigation is required to offset one acre of development. The method presented here, therefore, represents a more controlled method for applying a trading ratio, rather than relying on model differences.

This analysis demonstrates how each model would lead to different TP mitigation requirements, which would ultimately influence Virginia's BMP specifications and a range of other regulatory factors. In lieu of revising other stormwater management requirements, VRRM represents a logical choice to recalculate the value of urban impervious conversion in Virginia. More analysis is needed to determine an appropriate trading ratio to account for expected impacts to the Chesapeake Bay as well as a range of other environmental, economic, and land disturbance conditions. There is also potential to offset urban bank costs further through credit stacking for flood control, temperature control, carbon sequestration, air quality, or other ecological services to humans that urban mitigation practices can provide, which future work can quantify.

Table 8. Results from four different models used to calculate loading rates with consistent values between NPS WQT buyers and urban conversion credit sellers. The allowable post-development runoff rate in Virginia is 0.41 lbs/acre/year (adjusted to 0.24 lbs/acre/year EOT in the Rivanna watershed).

	Hypothetical TP credit: Credited reduction for one acre of impervious conversion, lbs TP /acre/year	**Hypothetical TP requirement:** Required reduction for one acre of forest to 100% impervious development, lbs TP/acre/year	Acres required if no trading ratio was applied	Net allowed TP increase lbs TP/ acre/year	Trading ratio to achieve a 1:1 acreage offset and 0 estimated net TP	Comment
CAST EOT	0.49	0.29	0.59	0.20	1.69	Would require recalculation of BMP requirements based on CAST
CAST EOS	0.84	0.50	0.59	0.34	1.69	Would require recalculation of BMP requirements based on CAST
VRRM	2.10	1.76	0.84	0.34	1.19	Allows good accordance with existing onsite requirements and BMP specifications
SWMM	2.11	1.77	0.84	0.34	1.19	Computational complexity may be prohibitive for regulatory application
Calculation	= Pre-conversion load – Post-conversion load.	= Post-development load – allowable load (0.41 lbs TP/acre developed/year, or 0.24 EOT)	= Hypothetical requirement / Hypothetical credit		= Hypothetical credit/ Hypothetical requirement	

4.4 Limitations

The quantitative SWMM results presented in this study cannot be assumed to apply to watersheds across Virginia or the U.S. Pollutant loads and hydrologic response to stormflow can be highly heterogeneous across the landscape, and our methods do not explore response across a range of environmental conditions and pollution transport dynamics. Differences in soil type and geology, land use changes, climate, and ecology would all impact quantitative results, and are not represented in the specific parameterization of our model.

Lack of groundwater representation in the model may exaggerate the ability of both impervious conversion and bioretention basins to reduce TP loads. Without representation of groundwater and subsurface processes, pollutants that infiltrate are simply assumed to be removed from the system in the SWMM model. The implication of this for our results is that there may be conditions under which the infiltrated pollution in scenarios 2 and 3 would flow back to surface water without sufficient treatment, where groundwater rise would limit infiltration, or where subsurface pollutants, such as from leaking sanitary infrastructure, could be further mobilized. Each of these would result in less overall water quality improvement, or even increased loads as a result of infiltration. Assuming that infiltration leads to pollution removal is common in models used to design and implement regulations. Contaminated groundwater can interact and degrade ecosystems and drinking water, but detailed site-specific geotechnical analysis is needed to understand these dynamics. Assuming pollutants are removed upon infiltration may be a reasonable assumption for TP, since this pollutant tends to bind to soil, but this is an important limitation for modeling of more mobile pollutants (including nitrogen), which are not included in this study. Much future research is needed to better understand how groundwater moves within heterogeneous urban zones, and to inform when, where, and how much infiltration can help reduce nutrient pollution.

The EMC-based representation of water quality in this study simplifies several important pollution transport dynamics. In EMC modeling, the pollution runoff remains constant in the model. Models that include nutrient cycling parameters can better capture transport dynamics. Many models also incorporate build-up and wash-off dynamics that can more accurately capture larger pollutant loadings during different-sized storms and after dry periods. However, these require real-world data and parameterization to perform well (Tuomela et al., 2019), and such data are not available for our study area. In regulatory settings, it is often not feasible to require extensive locally available information, so the EMC approach provides the best option.

Bioretention basins were selected as the onsite compliance option in scenario 2, but in practice developers can use other BMPs following Virginia Code 62.1-44.15:33. This study does not test alternative onsite mitigation practices or compare the effectiveness of different onsite BMPs. The intent of this study is to provide a first insight into the impact of replacing onsite water quality treatment with offsite credits, and we felt that a thorough comparison of possible BMPs would have shifted the focus to comparing BMP effectiveness, rather than describing policy outcomes.

This study also does not provide a cost or feasibility analysis of the nested WQT structure centering on urban trading. It is not within the scope of the study to determine how much of the watershed is likely to be available for conversion or the relative costs of different urban mitigation actions, but to identify policy scenarios that could improve environmental outcomes.

5. CONCLUSION

This study has provided a novel assessment of an active NPS WQT program in Virginia, and shed light on considerations for water quality managers seeking to develop or improve water quality trading programs. In particular, this work shows how NPS WQT programs can face issues related to the scale of analysis for load calculations. The Virginia program calculates credits based on expected impacts at the river basin scale, yet purchasers are regulated based on their impact at localized sites. In this study we applied an intermediate scale model to evaluate the program requirements for buyers and sellers. As we have shown, the scale mismatch in Virginia is reflected in disparate load estimations for buyers and sellers, which ultimately disincentivizes collocation of buyers and sellers in urban areas.

Watershed managers may want to more explicitly consider water quality goals across scales in the design of trading programs. This can ensure that safeguards are in place to reasonably balance water quality goals from the subcatchment to regional scales. We propose "nested" NPS WQT as one possible option to meet water quality goals across scales, in which collocation of buyers and sellers within an urban catchment could be required when deemed necessary by local water quality managers. Using a single model to calculate NPS WQT credit value for both buyers and sellers could create greater consistency and transparency across requirements in urban catchments.

This study also found that the number of credits sold provided a good proxy for the annual TP increase in the study catchment (i.e., one credit purchased was reasonably equivalent to an increase of one pound TP/year at the catchment scale). This increase may or may not be acceptable, depending on the local water quality goals. We encourage public accessibility of NPS WQT credit transaction data to enable local decision makers to evaluate whether and how to involve NPS WQT in local watershed management.

This research used a theoretically-based SWMM model, and quantitative results presented here should not be extrapolated to apply to other areas. This study also focused on the impacts of WQT on urban areas, and did not include any analysis of the crediting procedures for banks such as agricultural conversion banks. Detailed simulation of agricultural conversion credit banks, and best management practices to optimize water quality returns, are important areas for future work. Economic analysis of a range of options for trading within urban areas could further inform best practices for trading within urban areas. This study sets a trailhead for future work assessing the outcomes of NPS WQT programs.

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