ABSTRACT

Title of Document:	THE EFFECTS OF CHANGES IN LAND COVER AND LAND USE ON NUTRIENT LOADINGS TO THE CHESAPEAKE BAY USING FORECASTS OF URBANIZATION
	Allen Derrick Roberts, Doctor of Philosophy, 2010
Directed By:	Professor, Stephen D. Prince, Department of Geography

This dissertation examined the effects of land cover and land use (LC/LU) change on nutrient loadings (mass for a specified time) to the Chesapeake Bay, after future projections of urbanization were applied. This was accomplished by quantifying the comprehensive impacts of landscape on nutrients throughout the watershed. In order to quantify forecasted impacts of future development and LC/LU change, the current (2000) effects of landscape composition and configuration on total nitrogen (TN) and total phosphorus (TP) were examined. The effects of cover types were examined not only at catchment scales, but within riparian stream buffer to quantify the effects of spatial arrangement. Using the **SPA**tially **R**eferenced **R**egressions **O**n **W**atershed **A**ttributes (SPARROW) model, several compositional and configurational metrics at both scales were significantly (p value ≤ 0.05) correlated to nutrient genesis and transport and helped estimate loadings to the Chesapeake Bay with slightly better accuracy and precision.

Remotely sensed forecasts of future (2030) urbanization were integrated into SPARROW using these metrics to project TN and TP loadings into the future. After estimation of these metrics and other LC/LU-based sources, it was found that overall nutrient transport to the Chesapeake Bay will decrease due to agricultural land losses and fertilizer reductions. Although point and non-point source urban loadings increased in the watershed, these gains were not enough to negate decreased agricultural impacts.

In catchments forecasted to undergo urban sprawl conditions by 2030, the response of TN locally generated within catchments varied. The forecasted placement of smaller patches of development within agricultural lands of higher nutrient production was correlated to projected losses. However, shifting forecasted growth onto or adjacent to existing development, not agricultural lands, resulted in projected gains. This indicated the importance of forecasted spatial arrangement to projected TN runoff from the watershed.

In conclusion, comprehensive landscape analysis resulted in differences in simulations of current and future nutrient loadings to the Chesapeake Bay, as a result of urbanization and LC/LU change. With eutrophication from excess nutrients being the primary challenge to the estuary, information gained from the estimation of these effects could improve the future management and regulation of the Chesapeake Bay.

THE EFFECTS OF CHANGES IN LAND COVER AND LAND USE ON NUTRIENT LOADINGS TO THE CHESAPEAKE BAY USING FORECASTS OF URBANIZATION

By

Allen Derrick Roberts

Dissertation submitted to the Faculty of the Graduate School of the
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Advisory Committee:
Professor Stephen D. Prince, PhD, Chair
Professor Ralph Dubayah, PhD
Professor In-Young Yeo, PhD
Professor Brian Needelman, PhD
Gregory Schwarz, PhD

Dedication

This dissertation is dedicated to my mother who passed away back in 1999. Without her wisdom, knowledge, and drive for me to succeed, I would not be in the place where I am in today. Additionally, I would like to dedicate this work to my wife who influenced me to go to the University of Maryland and receive all of the training needed to complete not only my dissertation, but a life-long dream. To the end, thank you both for everything that you have given me. I will always love you both and it is more than you will ever know.

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To Dr. Gregory Schwarz, a special appreciation and gratitude on my behalf towards you is warranted. Without your countless responses to my emails, meetings with me at the United States Geological Survey (USGS) headquarters in Reston,

Virginia, phone conversations, and invaluable insight and knowledge into the inner workings of the **SPA**tially **R**eferenced **R**egressions **O**n **W**atershed Attributes (SPARROW) model, this dissertation could have never been completed. Through your mentoring and encouragement, learning and understanding the SPARROW model was not only quite easier, but rather enjoyable. Thank you once again for taking the time out of your busy schedule to accommodate all of my requests in regards to the SPARROW model and my research. Additionally to John Brakebill and Dr. Stephen Preston, also of the USGS, without your guidance, original datasets, and SAS codes representing the Chesapeake Bay version of the SPARROW models, this research would have not been possible. To the both of you, thank you very much and it was truly the starting point for the modeling analysis conducted in this research. Furthermore, I would also like to thank Dr. Richard Alexander and Dr. Richard Smith of the USGS in Reston for helpful input regarding how to utilize the SPARROW model more effectively in nutrient loading assessments for the Bay.

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To my all of my Department of Geography cohorts, especially those who are or were part of the Dr. Prince group, I would love to thank you for the memories, laughter, and lifelong friendships that were developed here. So to Jyothy, Khaldoun, Nancy, and Katie, I would like to thank you all personally for the academic discussions and scholarly exchanges that we were able to bounce off one another for the past few years. It was a learning experience that I will cherish. To Tao, Karl, Karen, Sage, Evan, Kelley, Jennifer, Angira, Ritvik, Derrick and others, I wanted to say thank you all for the hallway conversations, updates, and overall understanding of this process that we all chose to endure for the sake of academic enlightenment. I wish you all the best in life and hope through some venue, we can all stay connected.

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You will always have a special place in my heart for allowing me to undertake the greatest challenge of life. This is the culmination of our hard work and we did it!

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Chapter 1: Introduction

1.1 Background and nutrient history

The Chesapeake Bay is the largest estuary system occurring within the United States (U.S.). Its watershed has an area of 166,534 km^2 (64,299 mi^2) covering the mid-Atlantic region of the eastern seaboard. The watershed drains land masses in six states (New York (NY), Pennsylvania (PA), Delaware (DE), Maryland (MD), West Virginia (WV) and Virginia (VA)) and the District of Columbia (DC). The watershed also has the largest ratio of land-to-water volume of any estuary in the U.S., making runoff from the terrestrial landscape extremely important to the overall, ecological functioning of the estuary (Shuyler *et al.*, 1995).

Once considered a natural treasure due to its rich ecological diversity, recreational uses, and numerous fishing industries that it supported, the Chesapeake Bay has been in steady decline for over 400 years. The primary cause of this decline was the first significant, anthropogenic-driven transformations in land cover and land use (LC/LU) by European settlement that removed forests in favor of agricultural and urban cover types in the watershed in the 17th and 18th centuries (Boesch, 2006). Since this initial influx of population into the watershed, LC/LU changes have contributed greatly to problems of: increased sedimentation, turbidity (water opaqueness), nutrients, eutrophication (nutrient enrichment), hypoxia (low levels of dissolved oxygen (DO)), and anoxia (devoid of DO) (Bratton *et al.*, 2003). Furthermore, these changes have lead to a lowering in the population of submerged aquatic vegetation (SAV), numerous aquatic species, and the overall sustainability of the watershed and Chesapeake Bay itself (Bratton *et al.*, 2003).

Environmental and ecological concerns resulting from nutrient runoff associated with agriculture and increased urbanization have been cited as driving factors threatening the long-term sustainability of the Chesapeake Bay (Burke *et al*., 2000; Paolisso and Maloney, 2000). The two nutrients of greatest concern to degraded water quality and sustainability are nitrogen (N) and phosphorus (P). These nutrients enter into the Chesapeake Bay primarily through six major basins (the James, Patuxent, Potomac, Rappahannock, Susquehanna, and York) and dozens of other smaller rivers and streams at an estimated freshwater inflow rate of 8.7 x 10^6 m^3/h (Sims and Coale, 2002). The nutrients are transported to the estuary from point sources that include commercial, industrial, and municipal wastewater discharges directly into streams and from non-point (diffuse) sources that include atmospheric deposition, fertilizer and manure applications, and impervious surfaces (Alexander *et al.,* 2000b; and Castro and Driscoll, 2002) (Figure 1.1). The introduction of nutrients may have significant impacts on their biogeochemical cycling not only at the local and regional scale, but quite possibly on a global level.

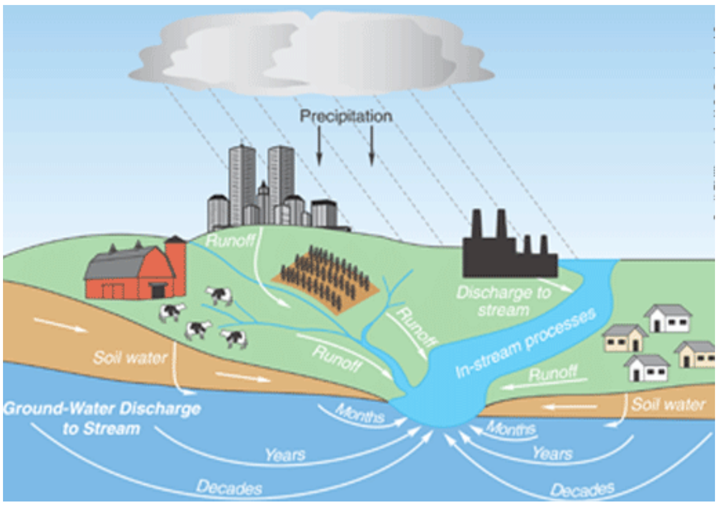

Figure 1.1: Example of how point and non-point N and P sources may be transported to the Chesapeake Bay (Goldman, 2005).

1.2 Current urbanization in the Chesapeake Bay

Since the end of World War II, urbanization and rapid population growth in the watershed have significantly increased the rate of genesis and delivery of nutrients to the Chesapeake Bay from stormwater runoff and point-source discharges (Sims and Coale, 2002). In fact, studies have indicated that the current rates of N and P reaching the Bay are 5-8 and 13-24 times their pre-colonial rates, respectively, (Boynton *et al*., 1995). These increases are primarily due to the conversion of pervious non-urban cover types that include wetlands and forests to impervious cover types indicative of urban development, such as residential and commercial uses. Additionally, over the last couple of decades, an even greater rate of the transference of non-urban LC/LUs by urbanization has been indicated throughout the watershed (Goetz *et al*., 2004a-b; Jantz *et al*., 2005). The LC/LU transformations can be directly correlated with the estimated doubling of population from 8 million to approximately 16 million over this time period (McConnell, 1995). One estimate of current LC/LU indicates that the watershed is comprised of: 58% forest, 33% agriculture, 8% urban, and 1% barren and wetlands (Sims and Coale, 2002).

Whereas estimates associated with population are not always easily quantified, urban and non-urban LC/LUs can be measured with more certainty using remotely sensed data derived from satellite imagery. In the watershed, LC/LU has been previously (1992) and currently (2001) mapped using the National Land Cover Dataset (NLCD) created from the Multi-Resolution Land Characteristic (MRLC) program (Homer *et al.*, 2004). However, current (2000) LC/LU has been mapped even more comprehensively using LANDSAT data collected by the **R**egional **E**arth **S**cience **A**pplications **C**enter (RESAC). RESAC products that are available for analysis include watershed-wide LC/LU, percent impervious surface area (% ISA), and percent tree cover (% TC) maps (Goetz *et al.*, 2003, 2004a, b; Jantz *et al.*, 2005) (Figure 1.2a-c).

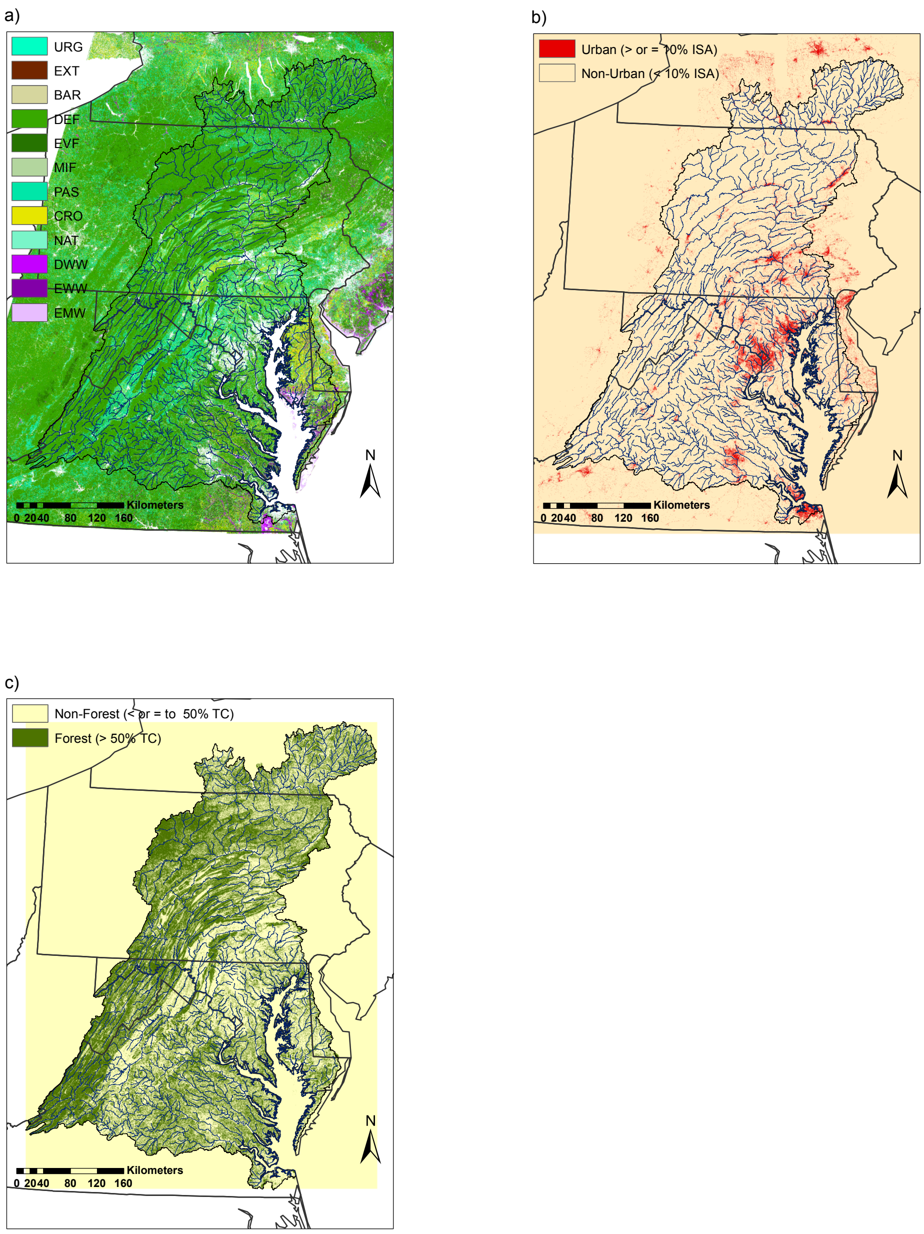

Figure 1.2: Current (2000) Chesapeake Bay watershed-wide maps of **a)** LC/LU, **b)** % ISA, and **c)** % TC. In **Figure 1.2a**, URG = Urban/residential/recreational grasses, EXT = Extractive, BAR = Barren, DEF = Deciduous forest, EVF = Evergreen forest, MIX = Mixed (deciduous-evergreen) forest, PAS = Pasture, CRO = Cropland, NAT = "Natural" grass, DWW = Deciduous wooded wetland, EVW = Evergreen wooded wetland, and EMV = Emergent (sedge-herb) wetland classification.

1.3 Forecasted urbanization in the Chesapeake Bay

By 2020, the population in the watershed is estimated to increase by approximately 20% over levels estimated in the mid 1990s (McConnell, 1995). As population is forecasted to increase over this time period and beyond, urbanized areas are likely to continue to expand throughout the developed and undeveloped regions of the watershed. With this expansion of urban growth, impervious surface areas capable of the concentrated buildup of non-point N and P that can runoff to streams during storm events will also continue to rise (Figure 1.3).

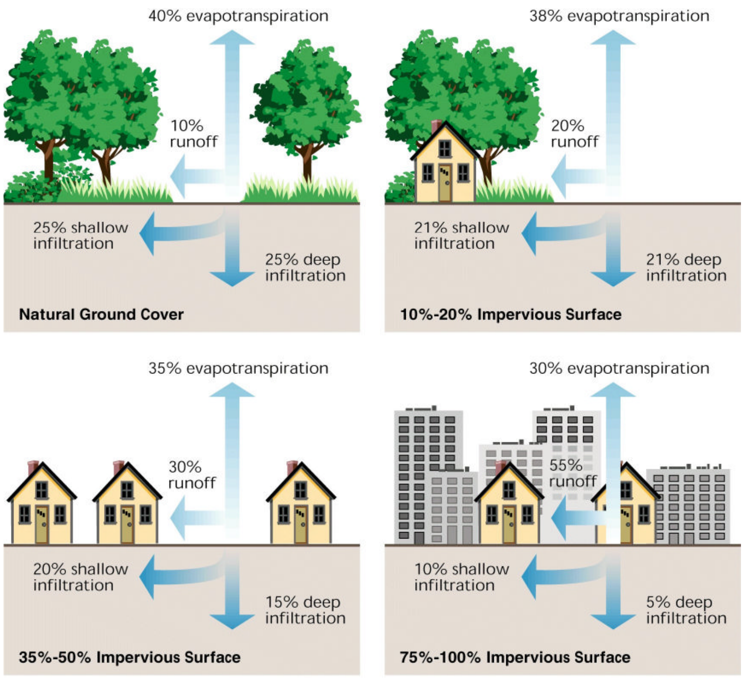

Figure 1.3: Schematic showing effects on the water balance of impervious cover and surface runoff in the Chesapeake Bay when urbanization is forecasted to increase over time (Federal Interagency Stream Restoration Working Group, 1998).

Projections of future urbanization have also been completed and incorporated within studies of the Chesapeake Bay watershed in an effort to quantify the effects of this increase in estimated impervious cover. The studies were conducted only for smaller areas and were used to correlate this forecasted LC/LU change to a variety of environmental processes. In a near comprehensive examination of the watershed (the New York portion was excluded), Wickham *et al.* (2002) used non-linear regression techniques to forecast urbanization to quantify changes in N and P export. In the Patuxent Basin, Costanza *et al.* (2002) used an economic land-use model to project development to determine its impacts on N and P export and net primary production (NPP). For the headwaters of the York Basin, Im *et al.* (2003) used the continuation of historical LC/LU change patterns provided by Caroline County's (VA) comprehensive development plan to evaluate forecasted urbanization on N and P export, peak flow, sediment mass, and total runoff volume. Finally, in the Susquehanna Basin tributaries of southeastern PA, Chang (2004) also used Lancaster County Planning Commission projections to forecast growth to 2030 to quantify N and P export responses.

Although it has been shown that other methods have been used for forecasting future urbanization to quantify environmental impacts in Chesapeake Bay subwatersheds, another method has been developed with the distinct advantage of modeling these estimated changes at the scale of the entire watershed with spatially-explicit forecasts of the locations of development and measures of uncertainty (Jantz *et al.*, in press). The predictions were provided by a model, which is referred to as **S**lope, **L**and use, **E**xclusion, **U**rban extent, **T**ransportation, **H**illshade or (SLEUTH), a cellular automata simulation that represents urban growth/expansion processes in a spatially-explicit, two-dimensional grid (Clarke *et al.*, 1997). Similarly to the other methods described earlier, SLEUTH has also been applied at much smaller regional scales in the watershed to predict urbanization (Clarke and Gaydos, 1998; Carlson, 2004).

A large regional application of this model in the watershed, that consists of a 23,700 km^2 section of the Baltimore-Washington metropolitan region (approximately 15% of the total watershed area), was used to forecast urbanization to 2030 (Jantz *et al.*, 2004; Claggett *et al.*, 2005; Jantz and Goetz, 2005). This application described projected urbanization under three distinct scenarios of development that included: 1) current trend, 2) managed/smart growth, and 3) ecological sustainable (Jantz *et al.,* 2004). In another watershed-wide application of SLEUTH, low-density development indicative of continued urban sprawl has been modeled for the entire Chesapeake Bay watershed (Jantz *et al.*, in press) (Figure 1.4).

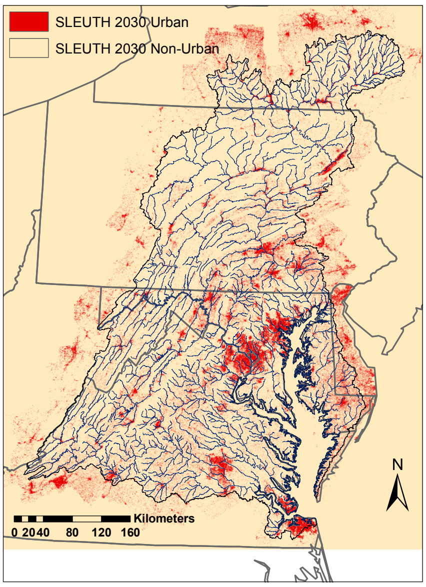

Figure 1.4: Projected (2030) Chesapeake Bay watershed-wide map of SLEUTH estimated urban and non-urban LC/LU.

1.4 Regional implications of effects of current and forecasted urbanization on nutrients in the Chesapeake Bay

According to Boesch *et al*. (2001), the multistate effort to restore the Chesapeake Bay ecosystem by reducing the inputs of nutrients that stimulate organic over-enrichment is one of the world's most ambitious attempts at large-scale ecosystem restoration. By the late 1990s, the Susquehanna Basin (the largest tributary to the estuary) was estimated to deliver over 1 x 10^9 metric tons of sediment, 5 x 10^4 tons of N, and 3 x 10^3 tons of P to the Chesapeake Bay annually (Preston and Brakebill, 1999; Langland and Cronin, 2003; Goetz *et al.*, 2004a; United States Environmental Protection Agency, 2009a). As an N-limited system in which its excess quantities control the trophic condition of the estuary, the reduction in N and surplus P delivered to its tributaries is paramount to reversing the trends of eutrophication and related ecological and environmental declines seen in the Chesapeake Bay (Weller, 2003). Thus, in an effort to mitigate these aforementioned regional impacts, the Chesapeake Bay Agreement was enacted in 1987 to reduce controllable nutrient levels 40% by 2000 from an initial 1985 baseline (Chesapeake Executive Council, 1987). Additionally, new amendments to the agreement in 2000 have called for the removal of all nutrient and sediment-related water quality issues impairing the Chesapeake Bay and its removal from the list of impaired waters under the Clean Water Acts of 1972 , 1977, and 1990 (Wang *et al.*, 2006).

Of the watershed traits such as: soil properties; vegetative cover; moisture conditions; size; shape; topography; orientation; geology; and channel characteristics that impact nutrient genesis and transport, cultural practices, such as LC/LU, may be the most pivotal (Wolfe, 2001). Throughout the Chesapeake Bay watershed, increasing human-populations have caused LC/LU changes from forest to agriculture to urban uses that have resulted in greater freshwater flows and increases in N and P loadings (mass for a specified time) from subwatersheds to the estuary (Fisher *et al.,* 2006). Although these LC/LU changes have increased inputs into the Chesapeake Bay in a widespread manner, the pattern of response has been not been universal (Fisher *et al.,* 2006). This fluctuation in the pattern of response to nutrient genesis and transport is based upon the variability in the properties of composition and configuration associated with these LC/LU changes.

In regards to LC/LU composition, current estimates indicate that the greatest (about 40%) contributor of excess nutrients delivered to the Chesapeake Bay are through agricultural practices, such as the non-points source delivery of applied fertilizer and manure (Goetz *et al.*, 2004a). The next largest contributor is from urban-induced point discharges that account for 23% of the N and 34% of the P measured in the estuary (Boesch *et al.*, 2001). Additionally, atmospheric deposition accounts for 11% of the N and 6% of the P measured in the estuary (Boesch *et al.*, 2001). In conjunction with the losses from non-point urban uses (commercial, industrial, residential, and transportation) that account for 9% and 8% of the N and P, respectively, in the Chesapeake Bay (Boesch *et al.*, 2001), composition of LC/LU is a primary factor controlling regional nutrient conditions reported in the estuary. Thus, with the anticipated increase of urban uses in the watershed, distinct changes in the amount of N and P reaching the estuary are expected to occur. However, the composition of these and other LC/LUs capable of impacting N and P are not evenly distributed watershed-wide and are likely to continue to cause variations in their long-term response.

Whereas the widespread pattern of uneven N and P responses to variability in composition is more discernible in the watershed, the responses to LC/LU configuration have only been extensively studied at much smaller, localized scales. Configuration can be envisioned as the arrangement of LC/LU at the catchment, basin, or regional watershed scale, but has been only quantified at the riparian stream buffer scale in smaller regional studies. In a small tributary located on the Chesapeake Bay's western shore, deciduous forest in the buffer adjacent to croplands removed over 80% of the nitrate (NO_3) and phosphate (PO_4) in overland runoff and nearly 85% of the NO_3 in shallow subsurface runoff (Correll *et al*., 1992; Jordan *et al*., 1993; Boesch *et al*., 2001). In localized watersheds in central MD, it was determined that urban riparian stream buffer zones had a lower potential to remove N via denitrification than forested riparian stream buffer zones (Groffman *et al*., 2002; Goetz *et al.,* 2003). However, in other smaller watersheds on the eastern shore of MD, some forested riparian stream buffers were shown to increase runoff to streams (Norton and Fisher, 2000). Although an attempt has been made to quantify the impacts of riparian LC/LU at more of a regional scale (Baker *et al*., 2006), the comprehensive configurational effects of watershed-wide riparian and catchment LC/LU on nutrient genesis and transport has yet to be completed. Yet, similarly to composition, the uneven distribution of configuration throughout the watershed should also cause discrepancies in the response of N and P.

By using the watershed-wide maps of current development and non-urban LC/LU with forecasted development, the comprehensive effects of landscape properties on present and projected nutrient loadings to the Chesapeake Bay could be quantified. This is an important management and research priority that would give insight into the potential correlations between the terrestrial and aquatic environments that were previously unknown at regional scales. Additionally, the potential findings here may also generate greater insight into the mechanisms and processes controlling biogeochemical cycling at local and global scales. Finally, if these potential findings were to lead to similar findings substantiated in other watersheds, a new approach to the management and restoration of nutrient-enriched ecosystems could be introduced.

1.5 Priority questions regarding effects of current and forecasted urbanization on nutrients in the Chesapeake Bay

Due to enactment of the Chesapeake Bay Agreements and enforcement of the Clean Water Acts, new research depicting how forecasts of future urbanization and the resulting LC/LU will affect total nitrogen (TN) and total phosphorus (TP) loadings to the Chesapeake Bay is needed. This is in response to local, state, and federal government entities endeavoring to meet these mutually agreed upon regional and federally-binding nutrient reduction mandates. With that in mind, several priority management questions regarding quantifying the effects of current and forecasted urbanization on nutrient loadings include:

- What LC/LUs at catchment and riparian stream buffer scales are significant in the current (2000) non-point (diffuse) sources and delivery of TN and TP loadings estimated in the Chesapeake Bay watershed?
- What are the causes of the observed effects of LC/LUs at catchment and riparian stream buffer scales on the current (2000) non-point (diffuse) sources and delivery of TN and TP loadings estimated in the Chesapeake Bay watershed?

- What specific types of LC/LUs lost to future urbanization will have the greatest impacts on projected TN and TP loadings estimated to the Chesapeake Bay?
- What changes in TN and TP loadings will occur in the Chesapeake Bay between 2000 and the future (2030) under the urbanization scenario applied here?

These priority management questions are addressed in this dissertation by combining satellite-based, remotely sensed LC/LU data with a watershed-wide, nutrient loading simulation approach. Studies in this dissertation concentrate on how current and forecasted changes in landscape composition and configuration can be used to quantify their comprehensive effects on nutrient loadings to the Chesapeake Bay. This region is chosen for this dissertation due to its varied LC/LU types, the intensity of anticipated changes, the availability of key data, and its potential for application to reducing eutrophication that is now and has been the top priority for the management, policy-making and restorative efforts of the estuary for nearly three decades (Malone *et al.,* 1993; Boesch *et al*., 2001).

1.6 Objectives

The specific objectives of this dissertation were to:

1. Quantify the generation and transport of nutrient loadings to the Chesapeake Bay using the current (2000), spatially-explicit (30 m resolution) distribution of urban and non-urban cover types,
2. Characterize changes in the generation and transport of nutrient loadings to the Chesapeake Bay using the future (2030), spatially-explicit (30 m resolution) distribution of urban and non-urban cover types, and

3. Compare and evaluate the characteristics of sprawling catchments projected to have losses with those projected to have gains in TN by 2030 to assess potential changes in LC/LU that may lead to new management actions in the Chesapeake Bay further reducing eutrophication.

1.7 The dissertation and its organization

In this chapter (**Chapter 1**), a brief overview of the nutrient background of the Chesapeake Bay is presented, in regards to how current and forecasted urbanization and resulting LC/LU changes could affect TN and TP loadings. The implications of how forecasted changes in urbanization could potentially impact these biogeochemical cycles regionally and how evaluating the comprehensive effects of landscape composition and configuration could help the overall quantification of TN and TP loadings to the estuary are also briefly described. The information here provides the framework for the research presented in this dissertation.

Chapter 2 describes a new methodology to quantify the comprehensive effects of current (2000) urban and non-urban LC/LU on nutrient loadings to the entire Chesapeake Bay. This is accomplished using the RESAC watershed-wide maps of LC/LU, % ISA, and % TC (Goetz *et al.*, 2003, 2004a, b; Jantz *et al.*, 2005) integrated within the Chesapeake Bay Version 3.0 TN and TP **SPA**tially **R**eferenced **R**egressions **O**n **W**atershed Attributes (SPARROW) models (Brakebill and Preston, 2004). The discovery and evaluation of newly, significant (p value ≤ 0.05), compositional and configurational landscape metrics representing non-point source and land-to-water delivery variables to the Chesapeake Bay are presented in new RESAC SPARROW models. Additionally, the spatially-explicit distribution of these landscape metrics and the estimates of current TN and TP loadings to the Chesapeake Bay are also presented. Finally, comparisons of the new RESAC SPARROW models with the original Chesapeake Bay Version 3.0 SPARROW models and the Phase 4.3 **H**ydrologic **S**imulation **P**rogram-**F**ORTRAN (HSPF) models are presented in regards to loadings to the Chesapeake Bay, loadings within the six largest basins draining the estuary, and the accuracy and precision of estimates.

Chapter 3 discusses a new approach to determining the forecasted effects of future (2030) urbanization on nutrient loadings to the Chesapeake Bay using the new RESAC SPARROW models developed in Chapter 2. The spatially-explicit, future changes in watershed-wide urbanization are derived from the SLEUTH forecasts and the RESAC LC/LU map and then integrated within the SPARROW models using the significant landscape metrics. Other LC/LU-correlated, significant variables in the SPARROW models are also projected into the future in an effort to more accurately portray estimated TN and TP loadings. The changes in TN and TP loadings from current to projected estimates for the entire Chesapeake Bay and the six largest basins are discussed. In addition, the changes to the significant landscape and non-landscape metric variables leading to these results are also evaluated and presented.

Chapter 4 documents the application of the TN results concluded in **Chapter 3** to catchments only forecasting an increase in low-intensity development (urban sprawl), but with differing responses of either a gain or loss in these projected loadings. The sprawling catchments are divided into two groups according to their projected TN loading response and compared for significant statistical differences. The statistical comparisons are based upon the changes or quantities of all significant variables found in the RESAC SPARROW model from the current to projected simulation. The statistical comparisons are evaluated and all landscape and non-landscape metric variables indicated with significant statistical differences are presented. The spatial arrangement of urban sprawl, in regards to the replacement of non-urban LC/LUs that would lead to gains or losses in projected TN loadings is also discussed.

Finally, **Chapter** 5 presents the conclusions and the management and policy recommendations developed from the results presented in **Chapters 2-4**. This dissertation commences with a discussion of directions for future research.

Chapter 2: Effects of urban and non-urban land cover on nitrogen and phosphorus runoff to Chesapeake Bay[1]

2.1 Abstract

The aim of this study was to determine the effects of catchment and riparian stream buffer-wide urban and non-urban land cover and land use (LC/LU) on total nitrogen (TN) and total phosphorus (TP) runoff to the Chesapeake Bay. The effects of the composition and configuration of LC/LU patches were explored in particular. A hybrid statistical-process model, the **SPA**tially **R**eferenced **R**egression **O**n **W**atershed Attributes (SPARROW), was calibrated with year 1997 watershed-wide, average annual TN and TP discharges to Chesapeake Bay. Two variables were predicted: 1) yield per unit watershed area and 2) mass delivered to the estuary. The 166,534 km^2 watershed was divided into 2,339 catchments averaging 71 km^2. LC/LU was described using sixteen classes applied to both the catchments and also to riparian stream buffers alone. Seven distinct landscape metrics were evaluated. In all, 167 (TN) and 168 (TP) LC/LU class metric combinations were tested in each model calibration run. Runs were made with LC/LU in six fixed riparian buffer widths (31, 62, 125, 250, 500, and 1,000 meters (m)) and entire catchments. The significance of the non-point source type (land cover, manure and fertilizer application, and atmospheric deposition) and factors affecting land-

[1] The material in Chapter 2 was previously published. Roberts, A. D., and Prince, S. D., 2010. Effects of urban and non-urban land cover on nitrogen and phosphorus runoff to Chesapeake Bay. Ecological Indicators 10(2): 459-474.

to-water delivery (physiographic province and natural or artificial land surfaces) was assessed. The model with a 31 m riparian stream buffer width accounted for the highest variance of mean annual TN (r^2 =0.9366) and TP (r^2 =0.7503) yield (mass for a specified time normalized by drainage area). TN and TP loadings (mass for a specified time) entering the Chesapeake Bay were estimated to be 1.449 x 10^8 and 5.367 x 10^6 kg/yr, respectively. Five of the 167 TN and three of the 168 TP landscape metrics were shown to be significant (p value ≤ 0.05) either for non-point sources or land-to-water delivery variables. This is the first demonstration of the significance of riparian LC/LU and landscape metrics on water quality simulation in a watershed as large as the Chesapeake Bay. Land cover metrics can therefore be expected to improve the accuracy and precision of estimated TN and TP annual loadings to the Chesapeake Bay and may also suggest changes in land management that may be beneficial in control of nutrient runoff to the Chesapeake Bay and similar watersheds elsewhere.

2.2 Introduction

Land cover and land use (LC/LU) and its changes have large effects on water quality of streams, rivers, lakes, and estuaries. Urbanization is a pervasive form of LC/LU alteration that is rapidly growing (Paul and Meyer, 2001). This involves conversion of croplands, forests, grasslands, pastures, wetlands, and other cover types to residential and transportation and also commercial and industrial uses, thereby increasing the areas of impervious surfaces (Tsegaye *et al*., 2006). Impervious surfaces are quantifiable indicators that correlate very closely with increases in non-point (diffuse) sources of polluted runoff which degrades the quality of aquatic resources (Arnold and

Gibbons, 1996). When combined with other anthropogenic and natural processes, landscape variables affect non-point nitrogen (N) and phosphorus (P) transport from land to the receiving water bodies and can contribute to eutrophication (nutrient enrichment) leading to poor water quality. Thus, LC/LU are critical properties that affect waterway pollution.

One such region where LC/LU changes are said to have affected regional water quality is the Chesapeake Bay watershed (Figure 2.1a). The Chesapeake Bay is the largest estuary in the United States (US); its watershed (166,534 km^2) encompasses portions of six states: New York (NY), Pennsylvania (PA), Delaware (DE), Maryland (MD), West Virginia (WV), and Virginia (VA)) and the District of Columbia (DC). Once considered a natural treasure due to its rich wildlife habitat and seafood industries, the estuary has been in steady decline starting with the colonial landscape transformations in the mid 1600s. By the 1990s, the human population of the watershed was approximately 16 million (McConnell 1995), concentrated in fast-growing urban corridors (Figure 2.1b). With the increase in population and the built environment, LC/LU modifications within the watershed have contributed to sedimentation, turbidity, eutrophication, and hypoxia, consequently reducing submerged aquatic vegetation (SAV) and affecting many other aspects of the aquatic ecosystem (Breitburg 1992; Hassett *et al.,* 2005). As stricter regulations involving point source discharges, fertilizer and manure applications, and fossil fuel emissions that lead to atmospheric deposition are enacted, spatial information on how landscape properties affect regional nutrient runoff is needed to meet reduction goals.

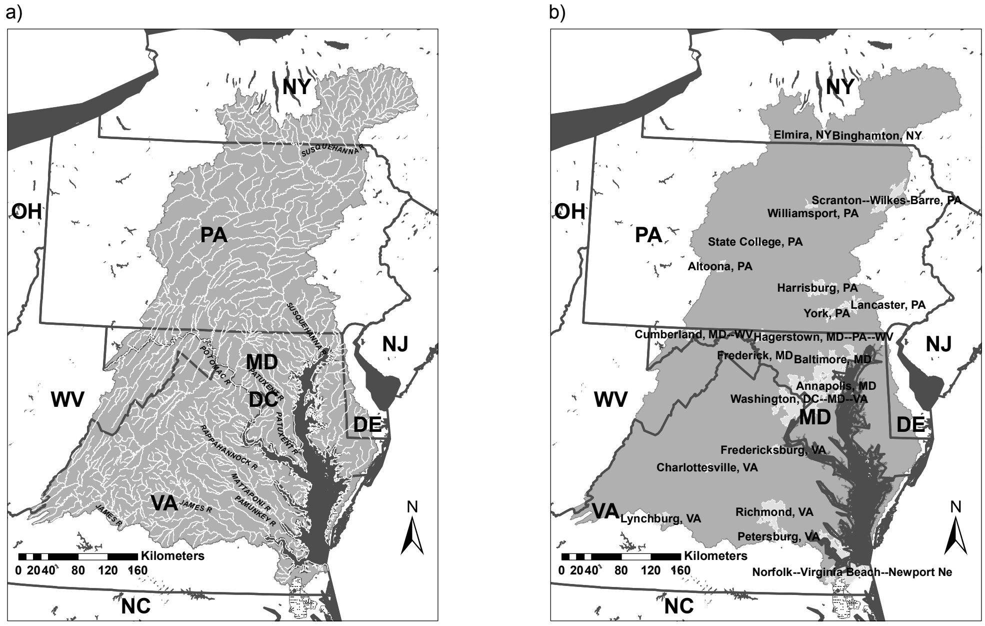

Figure 2.1: The Chesapeake Bay watershed showing the locations of: **a)** Streams and rivers draining the estuary and **b)** Urban centers located within its boundaries.

Remotely sensed LC/LU data have recently become available for landscape analysis of the Chesapeake Bay watershed, including the **R**egional **E**arth **S**cience **A**pplication **C**enter's (RESAC) LC/LU, percent impervious surface area (% ISA), and percent tree cover (% TC) maps for 2000 (Goetz *et al.,* 2003, 2004a, b; Jantz *et al.,* 2005). Other remotely sensed datasets include the National Land Cover Dataset (NLCD) developed by the Multi-Resolution Land Characteristics (MRLC) (Homer *et al.,* 2004) consortium for 1992 and 2001 which cover the entire conterminous United States. In particular, watershed-wide % ISA may yield important new information in linking the effects of urbanized areas to the estuary's water quality to compliment previous findings, within the watershed's smaller tributaries, correlating ISA to changes in stream hydrology (Carlson and Arthur, 2000; Jennings and Jarnagin, 2002; Dougherty *et al.,* 2004).

Water quality models, such as the **H**ydrologic **S**imulation **P**rogram-**F**ORTRAN (HSPF) (Bicknell *et al.,* 1996, 2001) and the **SPA**tially **R**eferenced **R**egressions **O**n **W**atershed **A**ttributes (SPARROW) (Smith *et al.,* 1997), have been applied to the Chesapeake Bay watershed (Preston and Brakebill, 1999; Linker *et al.,* 2000; Brakebill *et al.,* 2001; Brakebill and Preston, 2004; Goetz *et al.,* 2004a). The HSPF model has often been used by management and regulatory entities, such as the Chesapeake Bay Program (CBP) and the United States Environmental Protection Agency (USEPA). HSPF is a process-based deterministic model that simulates nutrient loadings (mass) to the tidal tributaries (Linker *et al.,* 2000). Complex process models of this type, however, require extensive temporally variable data (such as hourly rainfall, temperature, wind, and evapotranspiration) and detailed calibration. These requirements, generally limit application to a few watersheds (Alexander *et al.,* 2002a).

The SPARROW model developed by the United States Geological Survey (USGS) has been used to estimate stream export and improve interpretability of model parameters (Alexander *et al.,* 2002a). SPARROW utilizes a hybrid-statistical process structure that implements deterministic functions with spatially-distributed components, thus accounting for the dendritic features of watersheds (Alexander *et al.,* 2002b). The model addresses many shortcomings of purely statistical or regression-based models by incorporating deterministic components of nutrient transport that includes flow paths, first-order loss functions, and mass-balance constraints (Alexander *et al.,* 2002b). Furthermore, unlike HSPF, SPARROW provides robust measures of uncertainty. SPARROW was designed to reduce problems associated with data interpretation caused by sparse stream sampling measurement networks, network sampling biases, and basin heterogeneity. SPARROW has been applied at national (Smith *et al.,* 1997; Smith *et al.,* 2003; Alexander *et al.,* 2004), regional (Alexander *et al.,* 2000; Moore *et al.*, 2004), and even localized watershed scales (Alexander *et al.,* 2002a, b; McMahon *et al.,* 2003).

Previous versions of SPARROW for the Chesapeake Bay watershed (Preston and Brakebill, 1999; Brakebill *et al.,* 2001; Brakebill and Preston, 2004) have found no correlation between LC/LU and land-to-water delivery of non-point N and P. In these versions, however, LC/LU types were represented only as total areas of each cover type and did not take into account their configuration, or the possibility of differences in the role of LC/LU types in the riparian zones alone. Developed urban land cover in riparian zones have been shown to increase non-point N losses to streams (Groffman *et al.,* 2002), unlike non-developed (forested) zones (Goetz *et al.,* 2003).

Previous research has found that landscape metrics applied in catchment nutrient export models can improve nutrient predictions over those that use total LC/LU areal extent (Carle *et al.,* 2005). Landscape metrics describe the spatial structure of patches, the cover classes of patches, and patch mosaics, and provide other measures of composition and configuration (Leitao *et al.,* 2006). Landscape composition is the variety and abundance of patch types without regard to their spatial character or arrangement, whereas configuration quantifies spatial character and arrangement, position, and orientation of landscape elements. Although landscape metrics have been shown to improve correlations between the land surface and nutrient loading dynamics at catchment and riparian scales in subwatersheds of the Chesapeake Bay (Jones *et al.,* 2001; King *et al.,* 2005; Baker *et al.,* 2006), there has not been a comprehensive analysis of the entire watershed.

Thus, the overall purpose of this study was to use the SPARROW model with improved LC/LU maps from remotely-sensed data to determine the effects of LC/LU on TN and TP runoff from entire catchments and from the riparian stream buffer alone. Furthermore, the effects of landscape composition and configuration on runoff were explored.

2.3 Materials and methods

2.3.1 Chesapeake Bay watershed land cover and land use data

Three RESAC maps of LC/LU of the entire watershed were used, all at 30m resolution. The first map had 18 distinct LC/LU types (Goetz *et al.,* 2004a, b; Jantz *et al.,* 2005); the second depicted % ISA (Goetz *et al.,* 2004a, b; Jantz *et al.,* 2005); the third was of % TC (Goetz *et al.,* 2003, 2004a).

The watershed-wide % ISA map was partitioned into urban (≥ 10% ISA, class 1) and non-urban (< 10% ISA, class 2). Non-urban areas were coded as the appropriate LC/LU class. Twelve LC/LU classes were used: urban/residential/recreational grasses; extractive land; barren land; deciduous forest; evergreen forest; mixed (deciduous-evergreen) forest; pasture/hay; cropland; natural grass; deciduous wooded wetland; evergreen wooded wetland; and emergent (sedge-herb) wetland. The four built classes (low, medium, and high intensity developed, and transportation) were included in the urban grouping. The two remaining non-urban RESAC LC/LU classes: open water (not applicable) and mixed wetland (negligible areal coverage) were omitted. The % TC map was partitioned into non-forested (≤ 50% TC) and forested (> 50% TC) classes. Overall, a total of sixteen classes were created (Table 2.1).

2000 land cover class	Class number
Urban (≥ 10% ISA)	1
Non-urban (< 10% ISA)	2
Urban/residential/recreational grasses	3
Extractive	4
Barren	5
Deciduous forest	6
Evergreen forest	7
Mixed (deciduous-evergreen) forest	8
Pasture/hay	9
Croplands	10
Natural grass	11
Deciduous wooded wetland	12
Evergreen wooded wetland	13
Emergent (sedge-herb) wetland	14
Non-forested (≤ 50% TC)	15
Forested (> 50% TC)	16

Table 2.1: The sixteen Chesapeake Bay watershed cover classes created from the 2000 RESAC LC/LU, % ISA, and % TC maps.

2.3.2 Landscape metrics

Seven landscape metrics, previously shown to be significant indicators of downstream water impairment in catchments within the northeastern United States (Leitao *et al.,* 2006), were used. These seven landscape metrics were calculated and applied to the sixteen LC/LU classes. Five metrics (1-3, 6, 7) measure landscape configuration and two (4, 5) composition. The complete definitions for all seven metrics are given in Leitao *et al.* (2006).

1) **Contagion** quantifies the degree to which LC/LU types were clumped as opposed to dispersed in many smaller fragments

2) **Area-weighted mean radius of gyration** (distance) measures connectivity by correlation length. Correlation length is the average distance one might traverse across a map from a random starting point and moving in a random direction while remaining in the same LC/LU (Keitt *et al.,* 1997). Larger values of area-weighted mean radius of gyration indicate more connected landscapes.

3) **Patch number** measures total LC/LU fragmentation by the total number of patches of a particular LC/LU.

4) **Percentage of the landscape** area composed of a specified LC/LU.

5) **Area-weighted mean patch size** quantifies the sum, across all patches of a particular LC/LU, of patch area multiplied by proportional abundance of the patch. Since this metric weights each patch by its size relative to the total area of that particular LC/LU, larger patches exert greater influence than smaller patches, reducing the effects of extremely small patches.

6) **Area-weighted mean edge contrast** (percentage) quantifies the amount of contrast between adjacent LC/LU patches. In this application, contrast is defined as physical characteristics of differing cover types influencing nutrient transport. The metric quantifies the functional edge based on predetermined contrast weights assigned to pairwise comparisons of LC/LU types giving greater influence to larger patches.

7) **Area-weighted mean Euclidean nearest neighbor distance** quantifies the shortest distance from one patch to the next patch of the same LC/LU type, weighted in favor of larger patch sizes.

2.3.3 The SPARROW model

SPARROW estimates TN and TP loadings (mass for a specified time) and yields (mass for a specified time normalized for drainage area) from spatially-referenced watershed networks using source, land-to-water delivery, and stream and reservoir decay variables. The model is most frequently parameterized using average watershed-wide conditions for one specific year that represents a "snapshot" in time and is therefore not event-based (unlike HSPF). However, this allows for SPARROW to predict loadings and yields at a substantially higher number of points throughout these networks than HSPF is capable of by using linked nested stream reaches and their contributing catchment areas that are much smaller in magnitude. Catchments surrounding these nested reaches are denoted as *J(i)* and are the set of all reaches upstream that include reach *i*, except for those containing, or upstream of, monitoring stations of reach *i* (Figure 2.2) (Alexander *et al.,* 2002b). Unlike many other source-transport watershed models, SPARROW can simulate large watersheds using land-to-water delivery variables and simplified, yet process-based, descriptions of the sources (Schwarz *et al.,* 2006).

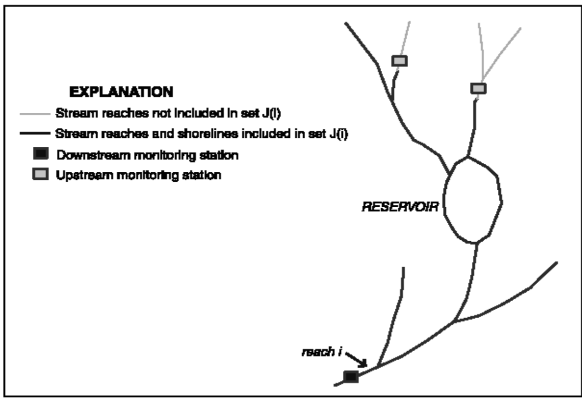

Figure 2.2: Illustration of a set of nested stream reaches and reservoir shorelines in relation to monitoring stations (Alexander *et al.,* 2002b). In model calibration, reach *i* refers to any reach containing a monitoring station. In model application, reach *i* refers to any reach for which a prediction can be made.

Both point and non-point source variables resulting in river discharges into the receiving water body (in this case the Chesapeake Bay) are included. Land-to-water delivery variables describe properties of the landscape relating climatic and other natural and human-induced surface processes affecting non-point N and P transport to streams. Stream decay is described by first-order losses of TN and TP loadings along stream channels, whereas reservoir decay is described by attenuation factors that influence TN and TP losses through large lakes and reservoirs. SPARROW uses nonlinear least squares regression to determine which variables are significant (p value ≤ 0.05) at the (Chesapeake Bay) watershed stations.

To assess model robustness, SPARROW utilizes a sensitivity analysis with Monte Carlo data resampling bootstrapping methods. The analysis calculates whether any coefficient has the wrong sign (Smith *et al.,* 1997) using the mean coefficient value (bootstrap estimate), confidence interval, and probability. Coefficients for significant variables are generated in 200 separate model runs by resampling (with replacement) from the original data.

The structure of Version 3.0 Chesapeake Bay model (Brakebill and Preston, 2004), referred to as B & P, was used so that the overall topology of the stream network and B & P non-LC/LU variables, such as point sources, could be used. Variables not related to landscape properties were the most current (1997) watershed-wide estimated datasets available. In addition, this version most closely approximated the year (2000) of the RESAC landscape maps. The difference in this present application from B & P was that landscape metrics were added by replacing catchment-wide non-point nutrient land sources and creating new catchment and riparian stream buffer-wide land-to-water delivery variables in order to determine the total significance of LC/LU composition and configuration. Riparian stream buffers are defined here as fixed, transitional areas between terrestrial landscapes and stream reaches created from the linked, spatially-referenced watershed network. In Version 3.0, 2,339 separate catchments averaging 71 km^2 were modeled (Figure 2.3a) using the enhanced river reach file (E3RF1) (Brakebill and Preston, 2004).

The TN and TP models were calibrated with observations at 87 and 104 stream loading sites, respectively, collected in 1997 by federal and state agencies. Stream nutrient loadings were calculated from the data using a log-linear regression model know as ESTIMATOR (Cohn *et al.,* 1989), which uses the 1950-2000 averages of daily stream discharge, specifying 1997 as the trend component (Brakebill and Preston, 2004).

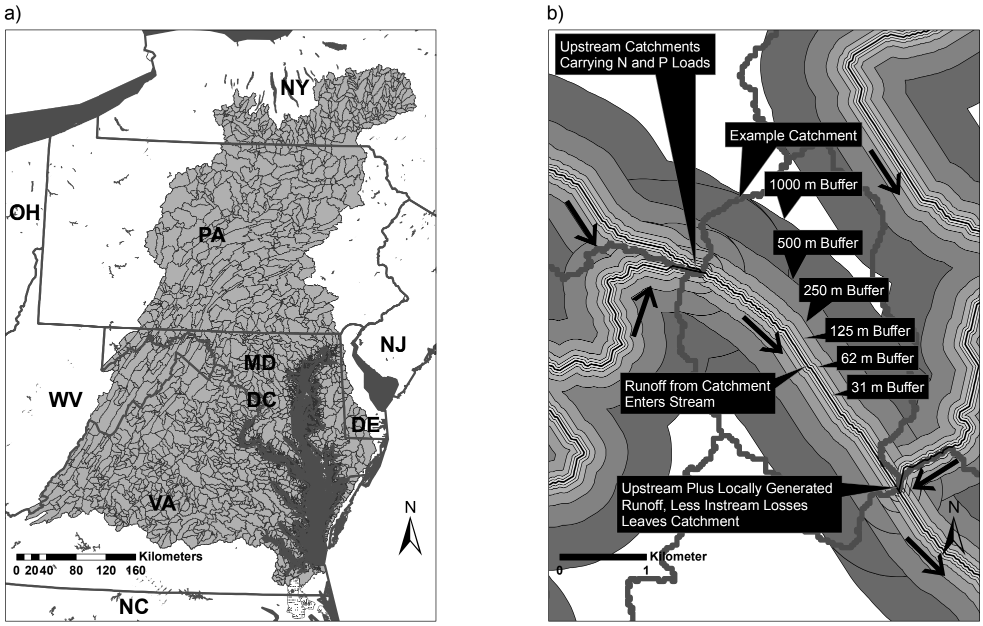

Figure 2.3: Map of: **a)** 2,339 catchments used and **b)** Schematic diagram of an example catchment area showing fixed 31, 62, 125, 250, 500, and 1,000 meters (m) riparian stream buffer width areas surrounding stream reach and corresponding upstream and downstream runoff paths.

2.3.4 Model calibration

Using the 2,339 modeled catchments and their associated stream reaches, fixed riparian stream buffers of 31, 62, 125, 250, 500, and 1,000 m (Figure 2.3b) were specified for a total of 4,678 land areas per model. A model run consisted of analyzing 2000 land cover classes within each of the 2,339 catchments with the only differences being which land cover area within one of the six fixed riparian stream buffer areas was also analyzed. Using FRAGSTATS (McGarigal *et al.,* 2002), five of the seven metrics (area-weighted mean radius of gyration, patch number, percentage of landscape, area-weighted mean edge contrast, and area-weighted mean Euclidean nearest neighbor distance) were created from the sixteen cover classes at both catchment and riparian stream buffer width scales, adding 160 new variables per run evaluated for land-to-water delivery significance.

The sixth metric (contagion) was calculated for land cover classes in 2000 at the catchment and the six riparian stream buffer widths. The values were calculated per map, not using LC/LU data cross-referenced between the other two maps. Contagion was calculated at the landscape map level between the cover classes of: 1) urban (class 1) and non-urban (class 2) in the % ISA map, 2) the twelve remaining non-urban classes (classes 3-14) in the LC/LU map, and 3) non-forested (class 15) and forested (class 16) from the % TC map at both catchment and riparian stream buffer width scales to add six more variables. By adding in the six contagion variables, a total of 166 new variables were evaluated for each TN and TP model run.

Finally, the 1997 B & P model urban (TN and TP models) and non-agricultural/non-urban (TP model only) areas were replaced by the area-weighted mean patch size for 2000 urban and non-agricultural/non-urban land. Non-agricultural/non-urban land area in the B & P TP model was modified here for new TP models by formation of area-weighted mean non-agricultural/non-urban patch sizes obtained by subtracting area-weighted mean cropland patch sizes (class 10) from area-weighted mean non-urban patch sizes (class 2). By adding catchment-wide area-weighted mean urban (≥ 10% ISA) patch sizes (TN and TP models), along with area-weighted mean non-agricultural/non-urban patch sizes (TP model only) as non-point source regions, the final number of variables increased to 167 and 168 for each TN and TP model run, respectively.

The area-weighted mean patch size metric for the sixteen landscape classes was not evaluated for land-to-water delivery significance since it could not be normalized for catchment and riparian stream buffer width area. Any type of land-to-water delivery variable directly correlated with catchment and riparian stream buffer width size would tend to confuse the relationship between scale-dependent, non-point source variables and non-point N and P stream loadings.

2.4 Results

2.4.1 TN mode1

Of the six models that used riparian stream buffers, the 31 m buffer out-performed all other simulations (yield coefficient of determination (r^2) 0.9366; root mean squared error (RMSE) 0.2407%) (Table 2.2). This was also indicated by the smallest Akaike Information Criteria (AIC) value (-2.7020) (Table 2.2) that indicated a better model and evidence for its selection over the other simulations (Luke, 2004). The 31 m model explained nearly 94% of variations in the mean annual TN yield resulting in a RMSE of 24%. The r^2 between the natural log of the observed and predicted loadings at the 87 Chesapeake Bay TN monitoring stations was 0.9896 (Figure 2.4a). Buffers of 62-1,000 m had lower yield r^2 and higher RMSE values, but the same significant coefficients.

Model run	Model yield r^2	Model RMSE (%)	Model AIC
1997 B & P TN	0.9073	0.2834	-2.4140
2000 RESAC 31 m TN	0.9366	0.2407	-2.7020
2000 RESAC 62 m TN	0.9332	0.2454	-2.6727
2000 RESAC 125 m TN	0.9332	0.2454	-2.6727
2000 RESAC 250 m TN	0.9332	0.2454	-2.6727
2000 RESAC 500 m TN	0.9332	0.2454	-2.6727
2000 RESAC 1000 m TN	0.9332	0.2454	-2.6727
1997 B & P TP	0.7413	0.3257	-2.1610
2000 RESAC 31 m TP	0.7503	0.3126	-2.2345
2000 RESAC 62 m TP	0.7457	0.3246	-2.1591
2000 RESAC 125 m TP	0.7353	0.3329	-2.1086
2000 RESAC 250 m TP	0.7262	0.3368	-2.0853
2000 RESAC 500 m TP	0.7220	0.3393	-2.0705
2000 RESAC 1000 m TP	0.7209	0.3400	-2.0664

Table 2.2: Comparison of r^2, RMSE, and AIC values between the Brakebill and Preston (2004) (1997 B & P) and the 2000 RESAC 31-1,000 m TN and TP models.

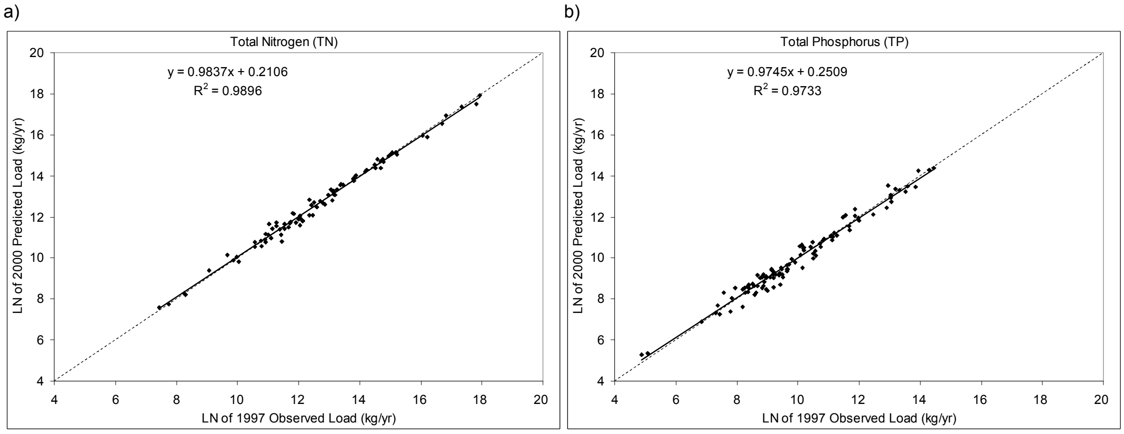

Figure 2.4: Comparison between the natural log of observed versus predicted loadings (kilograms per year (kg/yr)) in the: **a)** 2000 RESAC 31 m TN model utilizing 87 Chesapeake Bay water quality N monitoring stations and **b)** 2000 RESAC 31 m TP model utilizing 104 Chesapeake Bay water quality P monitoring stations.

Of the 167 landscape variables tested for either source or land-to-water delivery potential in the 31 m TN model, only five were significantly related to non-point N sources or delivery to streams. The five metrics were: area-weighted mean urban (≥ 10% ISA) patch size (source), percentage of extractive land (land-to-water delivery), area-weighted mean edge contrast of deciduous forest (land-to-water delivery), percentage of cropland (land-to-water delivery), and percentage of evergreen forest within the riparian stream buffer (land-to-water delivery) (Table 2.3). The only difference between the 31 m model and the other models was the highly significant (p value = 4.87 x 10^{-2}) effect of percentage of evergreen forest at this buffer width (Table 2.4). The percentage of evergreen forest in models of buffer widths of 62-1000 m was omitted as a land-to-water delivery variable (p-value > 0.05). Sensitivity analysis results comparing initial parametric with a final set of averaged bootstrapped coefficient estimates showed good agreement between the two for all significant variables (Table 2.5).

2000 RESAC 31 m model	Significant (p value ≤ 0.05) landscape metric variables
TN	Area-weighted mean urban (≥ 10% ISA) patch size (source) Percentage of cropland (land-to-water delivery) Percentage of extractive land (land-to-water delivery) Area-weighted mean edge contrast of deciduous forest (land-to-water delivery) Percentage of evergreen forest within the riparian stream buffer (land-to-water delivery)
TP	Area-weighted mean non-agricultural/non-urban patch size (source) Percentage of barren land within the riparian stream buffer (land-to-water delivery) Area-weighted mean urban (≥ 10% ISA) patch size (source)

Table 2.3: Comparison of the significant (p value ≤ 0.05) landscape metrics in the 2000 RESAC 31 m TN and TP models.

Variable (units)	Model run (Year)	B & P (1997)	RESAC (2000)					
			31 m	62 m	125 m	250 m	500 m	1000 m
	Yield r^2	0.9073	0.9366	0.9332	0.9332	0.9332	0.9332	0.9332
	RMSE (%)	0.2834	0.2406	0.2454	0.2454	0.2454	0.2454	0.2454
	AIC	-2.4140	-2.7020	-2.6727	-2.6727	-2.6727	-2.6727	-2.6727
	Model category							
Point sources (kg/yr)	Nitrogen source	1.530 (1.1 x 10^{-4})	1.173 (1.2 x 10^{-4})	1.169 (1.6 x 10^{-4})	1.169 (1.6 x 10^{-4})	1.169 (1.6 x 10^{-4})	1.169 (1.6 x 10^{-4})	1.169 (1.6 x 10^{-4})
Applied fertilizer (kg/yr)	Nitrogen source	0.294 (6.8 x 10^{-13})	0.175 (3.9 x 10^{-6})	0.194 (1.0 x 10^{-6})	0.194 (1.0 x 10^{-6})	0.194 (1.0 x 10^{-6})	0.194 (1.0 x 10^{-6})	0.194 (1.0 x 10^{-6})
Atmospheric deposition (kg/yr)	Nitrogen source	0.215 (3.5 x 10^{-7})	0.492 (2.0 x 10^{-7})	0.476 (4.0 x 10^{-7})	0.476 (4.0 x 10^{-7})	0.476 (4.0 x 10^{-7})	0.476 (4.0 x 10^{-7})	0.476 (4.0 x 10^{-7})
Applied manure (kg/yr)	Nitrogen source	0.065 (7.3 x 10^{-3})	0.078 (7.7 x 10^{-4})	0.087 (5.1 x 10^{-4})	0.087 (5.1 x 10^{-4})	0.087 (5.1 x 10^{-4})	0.087 (5.1 x 10^{-4})	0.087 (5.1 x 10^{-4})
Urban land (kg/ha/yr)	Nitrogen source	9.157 (8.7 x 10^{-6})	ONU	ONU	ONU	ONU	ONU	ONU
* Area-weighted mean urban (≥ 10% ISA) patch size (kg/ha/yr)	Nitrogen source	MNU	24.885 (7.2 x 10^{-7})	23.200 (1.2 x 10^{-6})	23.200 (1.2 x 10^{-6})	23.200 (1.2 x 10^{-6})	23.200 (1.2 x 10^{-6})	23.200 (1.2 x 10^{-6})
Percentage of Coastal Plain (%)	Landscape delivery	-0.735 (1.6 x 10^{-7})	-0.729 (4.1 x 10^{-8})	-0.679 (1.9 x 10^{-7})	-0.679 (1.9 x 10^{-7})	-0.679 (1.9 x 10^{-7})	-0.679 (1.9 x 10^{-7})	-0.679 (1.9 x 10^{-7})
* Percentage of extractive land (%)	Landscape delivery	MNU	0.270 (7.4 x 10^{-4})	0.283 (6.5 x 10^{-4})	0.283 (6.5 x 10^{-4})	0.283 (6.5 x 10^{-4})	0.283 (6.5 x 10^{-4})	0.283 (6.5 x 10^{-4})
* Area-weighted mean edge contrast of deciduous forest (%)	Landscape delivery	MNU	0.014 (7.2 x 10^{-3})	0.011 (3.1 x 10^{-2})	0.011 (3.1 x 10^{-2})	0.011 (3.1 x 10^{-2})	0.011 (3.1 x 10^{-2})	0.011 (3.1 x 10^{-2})
* Percentage of cropland (%)	Landscape delivery	MNU	0.021 (1.1 x 10^{-4})	0.020 (2.0 x 10^{-4})	0.020 (2.0 x 10^{-4})	0.020 (2.0 x 10^{-4})	0.020 (2.0 x 10^{-4})	0.020 (2.0 x 10^{-4})
* Percentage of evergreen forest within the riparian stream buffer (%)	Landscape delivery	MNU	0.013 (4.7 x 10^{-2})	MIS	MIS	MIS	MIS	MIS
Small streams (m/day)	Stream decay	0.375 (1.8 x 10^{-2})	0.249 (5.5 x 10^{-2})	0.299 (2.4 x 10^{-2})	0.299 (2.4 x 10^{-2})	0.299 (2.4 x 10^{-2})	0.299 (2.4 x 10^{-2})	0.299 (2.4 x 10^{-2})
Intermediate streams (m/day)	Stream decay	0.135 (2.2 x 10^{-1})	0.090 (3.2 x 10^{-1})	0.088 (3.4 x 10^{-1})	0.088 (3.4 x 10^{-1})	0.088 (3.4 x 10^{-1})	0.088 (3.4 x 10^{-1})	0.088 (3.4 x 10^{-1})
Large streams (m/day)	Stream decay	0.031 (5.4 x 10^{-1})	0.030 (4.8 x 10^{-1})	0.028 (5.1 x 10^{-1})	0.028 (5.1 x 10^{-1})	0.028 (5.1 x 10^{-1})	0.028 (5.1 x 10^{-1})	0.028 (5.1 x 10^{-1})
Reservoir (m/yr)	Reservoir decay	19.036 (2.8 x 10^{-2})	14.224 (2.3 x 10^{-2})	14.466 (2.3 x 10^{-2})	14.466 (2.3 x 10^{-2})	14.466 (2.3 x 10^{-2})	14.466 (2.3 x 10^{-2})	14.466 (2.3 x 10^{-2})

Table 2.4: All significant (p value ≤ 0.05) variables in the 1997 B & P and 2000 RESAC TN models using 31-1,000 m riparian stream buffer widths with model coefficients and p values (in parentheses). * Denotes all significant landscape metrics. ONU = original land source area "not utilized" as a result of being replaced with new surrogate landscape source area metric in 2000 model runs. MNU = new landscape source area and land-to-water delivery metrics "not utilized" in 1997 model run. MIS = new landscape land-to-water metric found to be "insignificant" (p value > 0.05) for that 2000 model run.

Variables (units)	Model run (Year)	RESAC (2000)	
		31 M initial parametric	31 M averaged bootstrapped
	Model category		
Point sources (kg/yr)	Nitrogen source	1.173	1.189
Applied fertilizer (kg/yr)	Nitrogen source	0.175	0.170
Atmospheric deposition (kg/yr)	Nitrogen source	0.492	0.508
Applied manure (kg/yr)	Nitrogen source	0.078	0.080
* Area-weighted mean urban (≥ 10% ISA) patch size (kg/ha/yr)	Nitrogen source	24.885	28.555
Percentage of coastal plain (%)	Landscape delivery	-0.729	-0.757
* Percentage of extractive land (%)	Landscape delivery	0.270	0.280
* Area-weighted mean edge contrast of deciduous forest (%)	Landscape delivery	0.014	0.015
* Percentage of cropland (%)	Landscape delivery	0.021	0.021
* Percentage of evergreen forest within the riparian stream buffer (%)	Landscape delivery	0.013	0.019
Small streams (m/day)	Stream decay	0.249	0.261
Intermediate streams (m/day)	Stream decay	0.090	0.103
Large streams (m/day)	Stream decay	0.030	0.037
Reservoir (m/yr)	Reservoir decay	14.224	13.669

Table 2.5: Sensitivity analysis comparison of all significant (p value ≤ 0.05) estimated variables between initial parametric and averaged bootstrapped coefficient estimates in 2000 RESAC 31 m TN model. * Denotes all landscape metrics found to be significant.

Of the 2,339 catchments in the 31 m model, the largest TN yields (> 18 kilograms per hectare per year (kg/ha/yr)) were identified in: the lower Susquehanna Basin containing cities such as Harrisburg, Lancaster, and York (PA); along tributaries of the middle Potomac Basin in north central MD; the eastern shore of MD; and southwestern DE (Figure 2.5a). Smaller areas of high locally generated TN yields were also found near the urban areas of DC and Richmond (VA). Although much attenuation occurred after accounting for stream and reservoir loss processes, there were only local differences in the amount of largest TN yields (> 18 kg/ha/yr) delivered to the estuary (Figure 2.5b). The highest locally generated urban patch size N yields (> 0.99 kg/ha/yr) per catchment were associated with the major urban centers of: DC; Baltimore (MD); Scranton/Wilkes-Barre, Harrisburg, Lancaster, and York (PA); Richmond, Petersburg, and Norfolk-Virginia Beach-Newport News (VA); and Elmira and Binghamton (NY) (Figure 2.6a). Stream and reservoir decay reduced area-weighted mean urban patch size N yield per catchment. However, largest yields (> 0.99 kg/ha/yr) to the estuary were still shown to originate from these areas (Figure 2.6b).

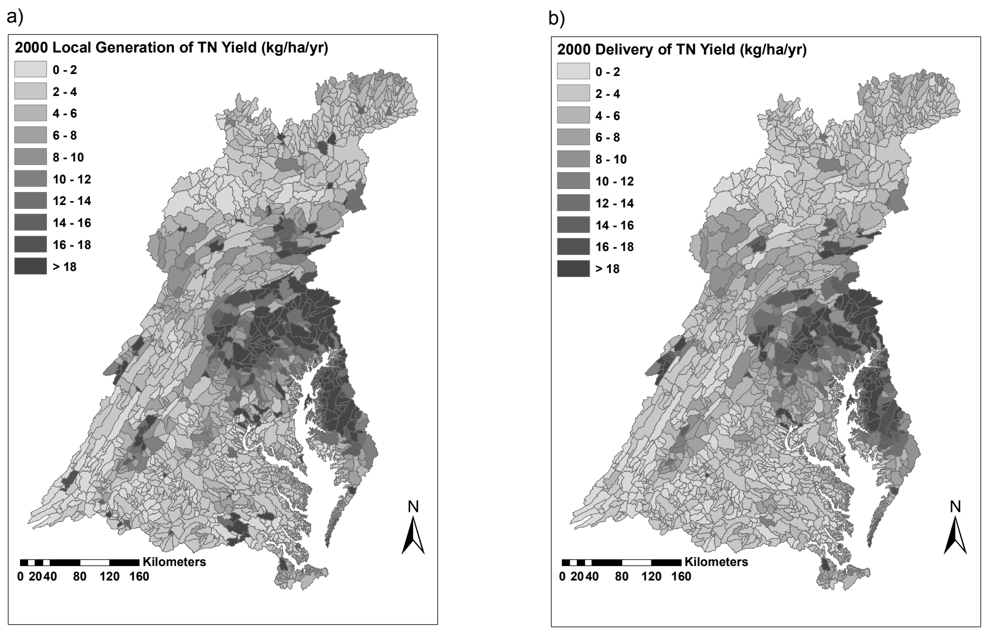

Figure 2.5: Per catchment estimated TN yield (kg/ha/yr) map from the 2000 RESAC 31 m model of: **a)** Local generation and **b)** Delivery to the Chesapeake Bay.

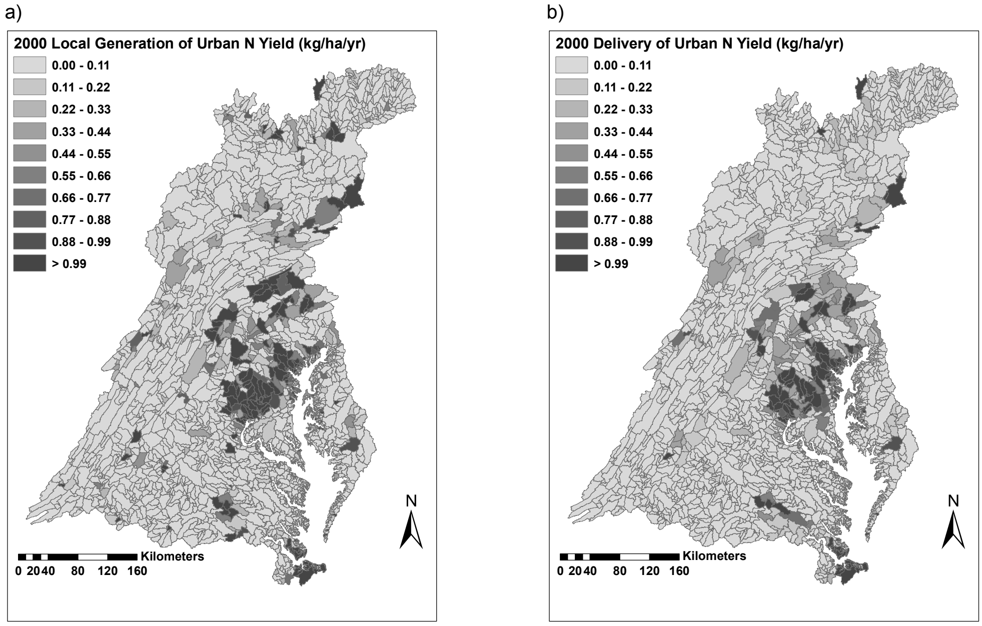

Figure 2.6: Per catchment estimated area-weighted mean urban (≥ 10% ISA) patch size N yield (kg/ha/yr) map from the 2000 RESAC 31 m model of: **a)** Local generation and **b)** Delivery to the Chesapeake Bay.

2.4.2 TP mode1

As in the case of the TN models, restriction of metrics to the 31 m riparian stream buffer gave the best results. This model also had the highest yield r^2 (0.7503) and lowest RMSE (0.3216) and AIC (-2.234) values (Table 2.2). The 31 m model explained about 75% of variation in the mean annual TP yield, a small improvement over the 62 m model. The r^2 between the natural log of the observed and predicted loadings at the 104 TP monitoring stations was 0.9733 (Figure 2.4b). Model significance decreased with each of the wider buffers.

Variables (units)	Model run (Year)	B & P (1997)	RESAC (2000)					
			31 m	62 m	125 m	250 m	500 m	1000 m
	Yield r^2	0.7413	0.7503	0.7457	0.7353	0.7262	0.7220	0.7209
	RMSE (%)	0.3257	0.3216	0.3246	0.3330	0.3367	0.3394	0.3400
	AIC	-2.1610	-2.2345	-2.1591	-2.1086	-2.0853	-2.0705	-2.0664
	Model category							
Point sources (kg/yr)	Phosphorus source	0.673 (4.7 x 10^{-6})	0.738 (1.3 x 10^{-6})	0.743 (1.4 x 10^{-6})	0.710 (5.4 x 10^{-6})	0.734 (3.4 x 10^{-6})	0.724 (4.5 x 10^{-6})	0.725 (4.5 x 10^{-6})
Applied fertilizer (kg/yr)	Phosphorus source	0.016 (3.3 x 10^{-4})	0.016 (1.1 x 10^{-3})	0.016 (1.1 x 10^{-3})	0.019 (3.4 x 10^{-4})	0.016 (1.9 x 10^{-3})	0.016 (1.8 x 10^{-3})	0.016 (1.9 x 10^{-3})
Applied manure (kg/yr)	Phosphorus source	0.007 (4.5 x 10^{-2})	0.008 (3.0 x 10^{-2})	0.008 (3.5 x 10^{-2})	0.009 (2.9 x 10^{-2})	0.008 (3.6 x 10^{-2})	0.008 (3.9 x 10^{-2})	0.008 (3.8 x 10^{-2})
Non-agricultural/non-urban land (kg/ha/yr)	Phosphorus source	0.093 (9.0 x 10^{-13})	ONU	ONU	ONU	ONU	ONU	ONU
* Area-weighted mean non-agricultural/non-urban patch size (kg/ha/yr)	Phosphorus source	MNU	0.110 (5.8 x 10^{-13})	0.110 (1.1 x 10^{-12})	0.094 (2.5 x 10^{-11})	0.105 (6.6 x 10^{-12})	0.104 (7.6 x 10^{-12})	0.102 (6.6 x 10^{-12})
Urban land (kg/ha/yr)	Phosphorus source	0.442 (7.8 x 10^{-5})	ONU	ONU	ONU	ONU	ONU	ONU
* Area-weighted mean urban (≥ 10% ISA) patch size (kg/ha/yr)	Phosphorus source	MNU	0.921 (1.0 x 10^{-4})	0.893 (1.3 x 10^{-4})	0.975 (1.1 x 10^{-4})	0.873 (2.3 x 10^{-4})	0.861 (2.8 x 10^{-4})	0.867 (2.8 x 10^{-4})
* Percentage of barren land within the riparian stream buffer (%)	Landscape delivery	MNU	0.281 (1.2 x 10^{-6})	0.279 (6.1 x 10^{-5})	0.213 (1.8 x 10^{-3})	0.167 (7.0 x 10^{-3})	0.132 (1.6 x 10^{-2})	0.114 (2.0 x 10^{-2})
Small streams (m/day)	Stream decay	-0.260 (4.9 x 10^{-2})	-0.198 (1.3 x 10^{-1})	-0.199 (1.4 x 10^{-1})	-0.210 (1.3 x 10^{-1})	-0.194 (1.7 x 10^{-1})	-0.191 (1.9 x 10^{-1})	-0.184 (2.1 x 10^{-1})
Intermediate streams (m/day)	Stream decay	0.028 (7.9 x 10^{-1})	0.150 (1.9 x 10^{-1})	0.144 (2.1 x 10^{-1})	0.137 (2.4 x 10^{-1})	0.095 (4.1 x 10^{-1})	0.082 (4.8 x 10^{-1})	0.071 (5.3 x 10^{-1})
Large streams (m/day)	Stream decay	0.039 (4.6 x 10^{-1})	0.034 (5.2 x 10^{-1})	0.035 (5.1 x 10^{-1})	0.053 (3.5 x 10^{-1})	0.037 (5.0 x 10^{-1})	0.037 (5.1 x 10^{-1})	0.038 (5.0 x 10^{-1})
Reservoir (m/yr)	Reservoir decay	17.004 (5.4 x 10^{-2})	19.019 (5.7 x 10^{-2})	18.473 (6.3 x 10^{-2})	17.669 (7.5 x 10^{-2})	19.611 (6.1 x 10^{-2})	19.965 (5.9 x 10^{-2})	19.700 (5.8 x 10^{-2})

Table 2.6: All significant (p value ≤ 0.05) variables in the 1997 B & P and 2000 RESAC TP models using 31-1,000 m riparian stream buffer widths with model coefficients and p values (in parentheses). * Denotes all significant landscape metrics. ONU = original land source areas "not utilized" as a result of being replaced with new surrogate landscape source area metrics in 2000 model runs. MNU = new landscape source area and land-to-water delivery metrics "not utilized" in 1997 model run.

Of the 168 landscape metric variables evaluated in the 31 m model, only three were significantly related to non-point P sources or delivery processes (Table 2.3). Within the riparian stream buffer, the percentage of barren land was the only significant land-to-water delivery metric (Table 2.6). However, significance of this variable fell as the width of the riparian stream buffer increased. Comparisons between the initial parametric and the final set of averaged bootstrapped coefficient estimates indicated little to no deviation for all significant variables (Table 2.7).

Variables (units)	Model run (Year)	RESAC (2000)	
		31 M Initial Parametric	31 M Averaged Bootstrapped
	Model category		
Point sources (kg/yr)	Phosphorus source	0.738	0.738
Applied fertilizer (kg/yr)	Phosphorus source	0.016	0.016
Applied manure (kg/yr)	Phosphorus source	0.008	0.008
* Area-weighted mean non-agricultural/non-urban patch size (kg/ha/yr)	Phosphorus source	0.110	0.110
* Area-weighted mean urban (≥ 10% ISA) patch size (kg/ha/yr)	Phosphorus source	0.921	1.139
* Percentage of barren land within the riparian stream buffer (%)	Landscape delivery	0.281	0.277
Small streams (m/day)	Stream decay	-0.198	-0.231
Intermediate streams (m/day)	Stream decay	0.150	0.138
Large streams (m/day)	Stream decay	0.034	0.044
Reservoir (m/yr)	Reservoir decay	19.019	20.529

Table 2.7: Sensitivity analysis comparison of all significant (p value ≤ 0.05) estimated variables between initial parametric and averaged bootstrapped coefficient estimates in 2000 RESAC 31 m TP model. * Denotes all landscape metrics found to be significant.

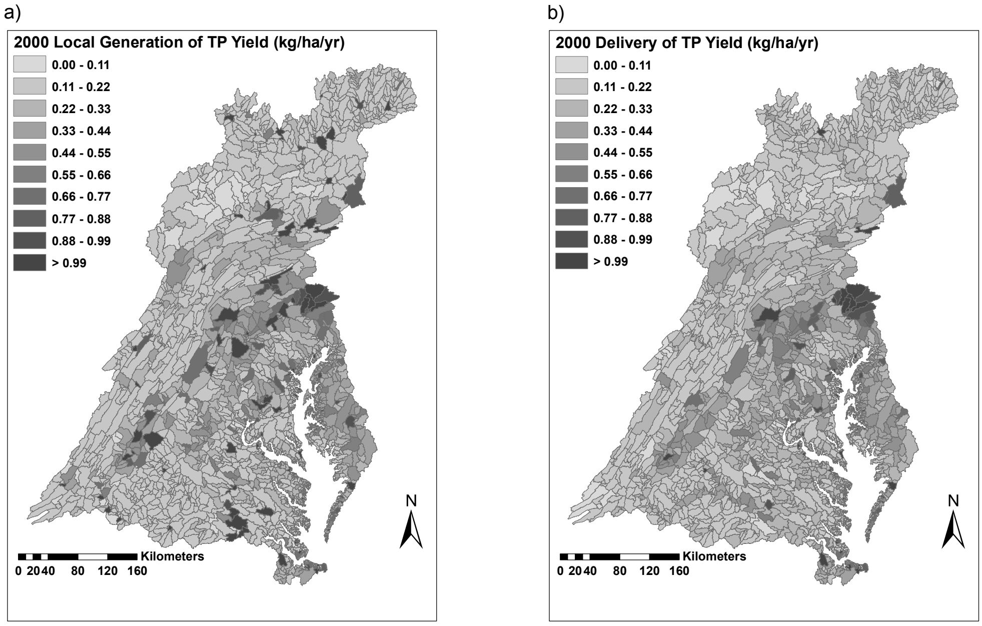

Figure 2.7: Per catchment estimated TP yield (kg/ha/yr) map from the 2000 RESAC 31 m model of: **a)** Local generation and **b**) Delivery to the Chesapeake Bay.

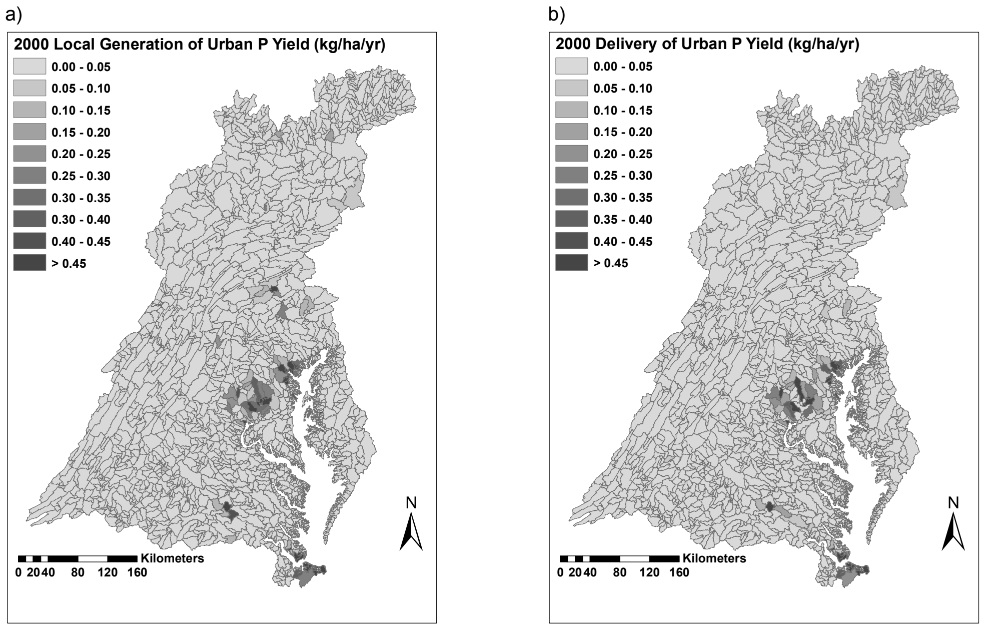

Figure 2.8: Per catchment estimated area-weighted mean urban (≥ 10% ISA) patch size P yield (kg/ha/yr) map from the 2000 RESAC 31 m model of: **a)** Local generation and **b)** Delivery to the Chesapeake Bay.

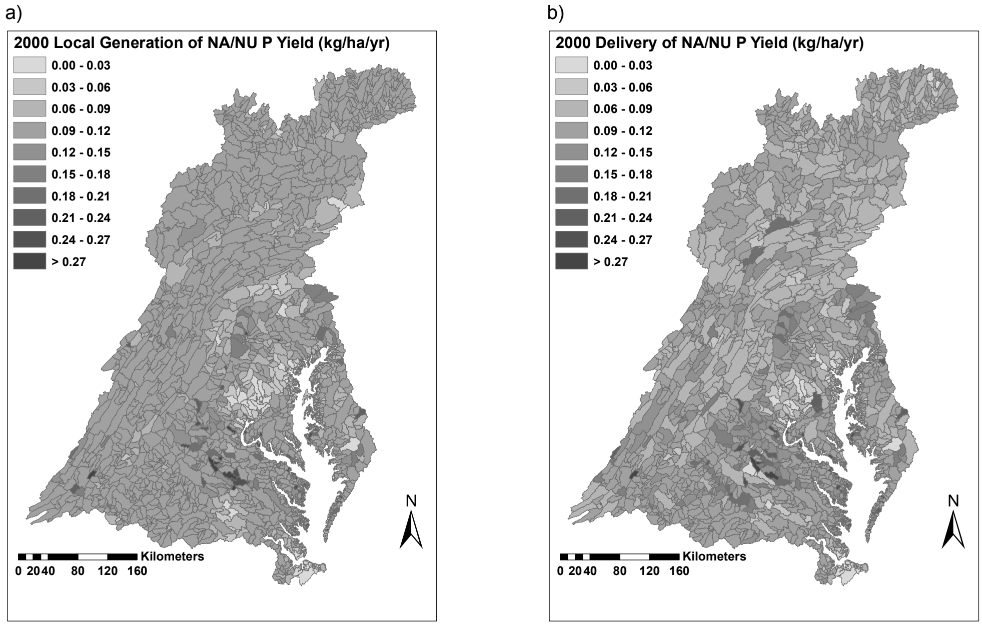

Figure 2.9: Per catchment estimated area-weighted mean non-agricultural/non-urban (NA/NU) patch size P yield (kg/ha/yr) map from the 2000 RESAC 31 m model of: **a)** Local generation and **b)** Delivery to the Chesapeake Bay estuary.

A map of locally generated TP yields showed that highest yields (> 0.99 kg/ha/yr) were found in: the central and lower Susquehanna Basin near Scranton/Wilkes Barre, Harrisburg, Lancaster, and York (PA); central MD; and Richmond and Petersburg (VA) in the middle-to-lower James Basin (Figure 2.7a). After accounting for stream and reservoir losses, areas with the highest TP yields (> 0.99 kg/ha/yr) delivered to the estuary were significantly reduced, with the exception of near Lancaster (PA) (Figure 2.7b). A map of locally generated area-weighted mean urban (≥ 10% ISA) patch size P yields per catchment showed that higher values (> 0.45 kg/ha/yr) were estimated in: Harrisburg (PA); DC; Baltimore (MD); and Richmond and Norfolk-Virginia Beach-Newport News, (VA) (Figure 2. 8a). Highest delivered yields (> 0.45 kg/ha/yr) were also found in these urban municipalities (Figure 2.8b). Finally, although lower in magnitude, the greatest locally generated area-weighted mean non-agricultural/non-urban patch size P yields per catchment (> 0.27 kg/ha/yr) were in northern and central VA (Figure 2. 9a). The largest delivered area-weighted mean non-agricultural/non-urban patch size P yields (> 0.27 kg/ha/yr) were also from these areas (Figure 2.9b).

2.5 Discussion

2.5.1 Effects of non-urban and urban LC/LU on TN and TP runoff to the Chesapeake Bay

The natural log of the 2000 area-weighted mean urban (≥ 10% ISA) patch sizes derived from the RESAC map was found to be well correlated with the 1997 NLCD urban, as used by B & P, which suggests that the new parameterization did not change the fundamental structure of B & P, and so the use here of non-LC/LU elements of B & P was justified (Figure 2.10a). Numerous studies (Bannerman *et al.,* 1993; Rushton, 2001; Sonada *et al.,* 2001; Stow *et al.,* 2001; Line *et al.,* 2002; Shinya *et al.,* 2003; Coulter *et al.,* 2004; Groffman *et al.,* 2004; Law *et al.,* 2004; Osmond and Hardy, 2004; Caccia and Boyer, 2005; Erickson *et al.,* 2005; Wakida and Lerner, 2005, 2006; Williams *et al.,* 2005; Gilbert and Clausen, 2006) have found that non-point N and P generated from low, medium, and high intensity developed, as well as transportation LC/LU classes are linked to stream loadings. The natural log of the 1997 non-agricultural/non-urban land used by B & P was also well correlated with the 2000 area-weighted mean non-agricultural/non-urban patch size derived from RESAC in the 31 m model (Figure 2.10b).

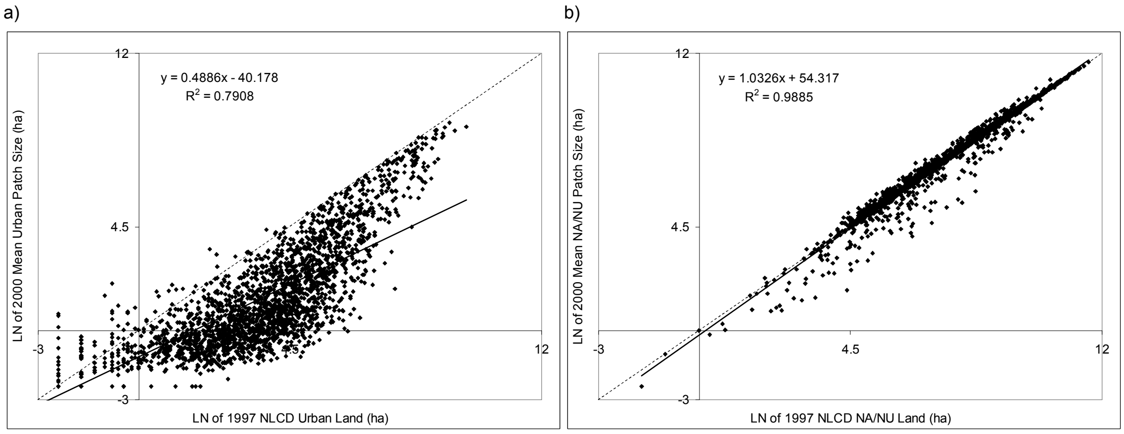

Figure 2.10: 2,339 SPARROW catchment comparison of the natural log of the: **a)** Significant (p value ≤ 0.05) 1997 NLCD source factor of total urban land area (ha) used in the 1997 B & P TN and TP models versus the significant source factor of 2000 area-weighted mean urban (≥ 10% ISA) patch sizes (ha) used in the 2000 RESAC 31 m TN and TP models and **b)** Significant 1997 NLCD source factor of non-agricultural/non-urban land area (ha) used in the 1997 B & P TP model versus the significant source factor of 2000 area-weighted mean non-agricultural/non-urban patch sizes (ha) used in the 2000 RESAC 31 m TP model.

Nearly 25 kg of non-point N in the 31 m TN model, as compared to close to 1 kg of non-point P in the 31 m TP model, were estimated to be generated and measured from streams draining the estuary from each ha of area-weighted mean urban patch (≥ 10% ISA) size annually (Table 2.4; Table 2.6). The 31 m TN model coefficient value was well within the expected range of export values for all urban land use yields (3-40 kg/ha/yr) (Schwarz *et al.,* 2006). The largest area-weighted mean urban (≥ 10% ISA) patch sizes (>270 ha) were identified in: DC; Baltimore (MD); Scranton/Wilkes-Barre, Harrisburg, Lancaster, and York (PA); Richmond, Petersburg and Norfolk-Virginia Beach-Newport News (VA); and Elmira and Binghamton (NY) (Figure 2.11a).

Just over 0.1 kg of non-point P in the 31 m model was estimated to be generated and measured from streams draining the Chesapeake Bay from each ha of area-weighted mean non-agricultural/non-urban patch size annually (Table 2.6). The highest area-weighted mean non-agricultural/non-urban patch sizes per catchment (> 27,000 ha) were located in PA and western VA (Figure 2. 11b).

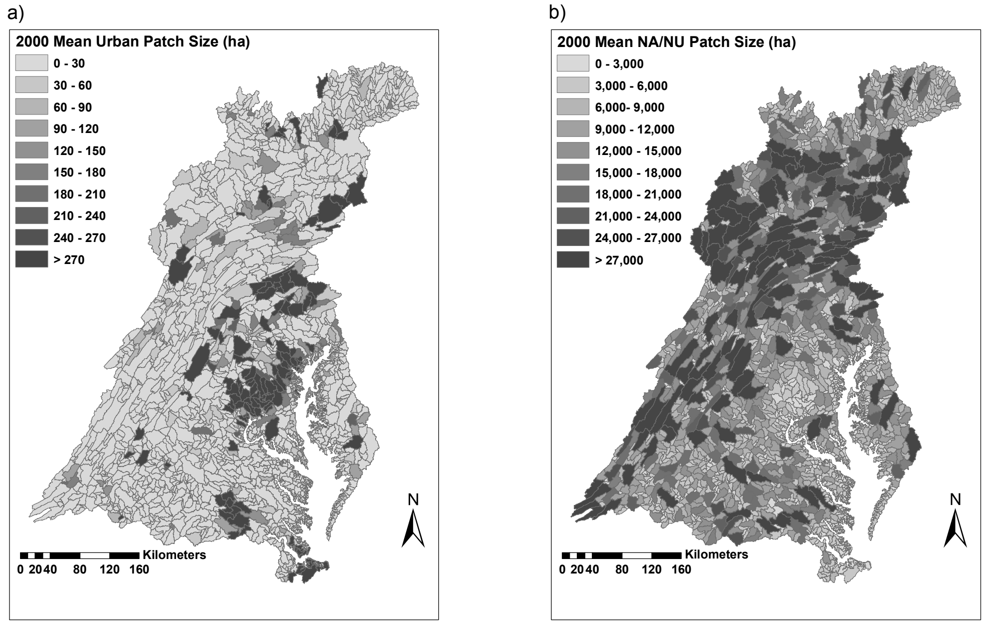

Figure 2.11: Per catchment source factor metrics of: **a)** Area-weighted mean urban (≥ 10% ISA) patch size (ha) in the 2000 RESAC 31 m TN and TP models and **b)** Area-weighted mean non-agricultural/non-urban (NA/NU) patch size (ha) in the 2000 RESAC 31 m TP model.

For every one percent of extractive land composition in a catchment, a 0.27% increase in delivery from all non-point N sources to streams draining the estuary was estimated. Characteristics associated with extractive land may help explain this. Low infiltration capacities tied to decreased hydraulic conductivity (the ability of a porous medium to transmit a specific fluid under a unit hydraulic gradient) (Ward and Trimble, 2004) of mine soils that increase overland runoff have been found throughout the watershed's PA tributaries (Ritter and Gardner, 1993; Guebert and Gardner, 2001). Furthermore N-fixing trees, such as black locust (*Robinia pseudoacacia*), used in mine spoil reclamation in the Susquehanna Basin (Bruns, 2005) may also increase non-point N stream delivery. The highest percentages of extractive land per catchment (> 4.5%) were found in central PA, western MD, and northeastern WV (Figure 2.12a).

Area-weighted mean edge contrast of deciduous forest measures the contrast between the eleven non-urban LC/LU classes and deciduous forest in the LC/LU map. Greatest differences in this metric were between deciduous forest and urban/residential/recreational grasses, extractive, barren, pasture/hay, croplands, and natural grass. The greater the incidence of dissimilar LC/LU classes configured around deciduous forest, the higher the area-weighted mean edge contrast. This metric is also related to forest fragmentation. For each one percent of the area-weighted mean edge contrast of deciduous forest in a catchment, delivery from all non-point N sources to streams draining the estuary was estimated to increase by 0.014%. Fragmentation-related increases in runoff based upon observed lower hydraulic conductivities in nearby non-forested LC/LU classes, as compared to forested LC/LU classes, have been found elsewhere (Chandler and Walter, 1998; Giambelluca, 2002; Ziegler *et al.,* 2004a, 2006, 2007; Zimmermann *et al.,* 2006). The highest area-weighted mean edge contrasts of deciduous forest per catchment (> 63%) were found in southeastern PA and northern MD (Figure 2.12b).

For every one percent of cropland composition in a catchment, delivery from all non-point N sources to streams draining the estuary was estimated to increase by 0.021%. This may be a result of decreased hydraulic conductivity and increased runoff due to cultivation practices that may have contributed to applied fertilizer, manure and other non-point N delivery from these areas. Field compaction from tillage and machinery alone may promote surface sealing and overland runoff (Logsdon and Jaynes, 1996). The highest percentages of cropland per catchment (> 45%) were in central and southeastern PA, the eastern shore of MD, and southwestern DE (Figure 2.12c).

For every one percent of evergreen forest composition within riparian stream buffers, delivery from all non-point N sources to streams draining the estuary was estimated to increase by 0.013%. Increased overland and shallow subsurface N runoff correlated with greater evergreen forest has been found elsewhere, albeit in small catchments, resulting from decreased soil hydraulic conductivities (Allan *et al.,* 1993; Allan and Roulet, 1994; and Wetzel, 2003). A similar conclusion was also reached in North Carolina's (US) piedmont region where a greater proportion of forest in some riparian buffers increased N loadings to streams due to increased overland and shallow subsurface runoff linked to decreased soil hydraulic conductivity in these buffers (Verchot *et al.,* 1997). Highest percentages of evergreen forest within riparian stream buffers per catchment (> 22.5%) were in central PA, central and western VA, and south central NY (Figure 2.12d).

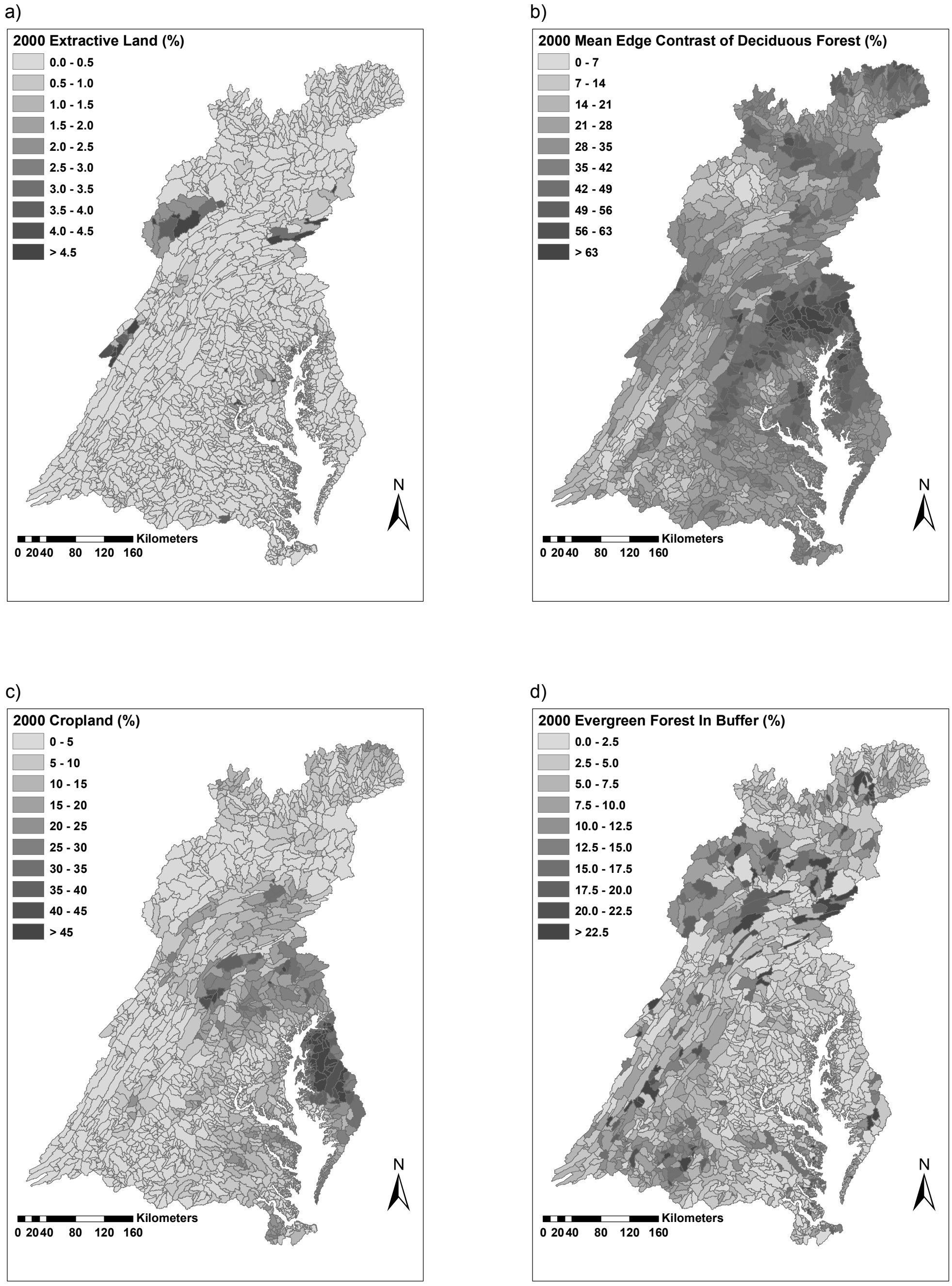

Figure 2.12: Per catchment land-to-water delivery metrics of: **a)** Extractive land (%), **b)** Area-weighted mean edge contrast of deciduous forest (%), **c)** Cropland (%), and **d)** Evergreen forest within the riparian stream buffer (%) in the 2000 RESAC 31 m TN model.

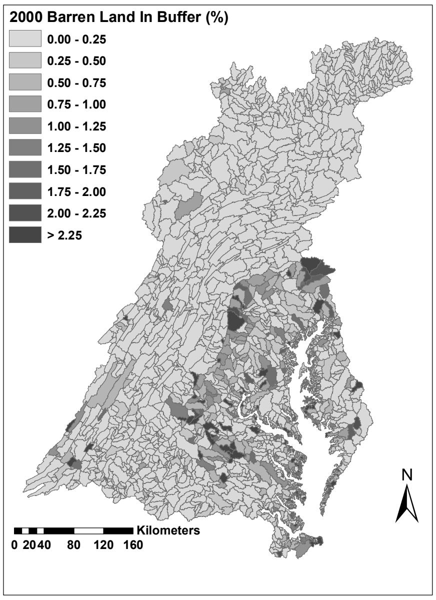

Figure 2.13: Per catchment land-to-water delivery metric of barren land within the riparian stream buffer (%) in the 2000 RESAC 31 m TP model.

Finally, for every one percent of barren land composition within the riparian stream buffer, delivery from all non-point P sources to streams draining the estuary was estimated to increase by 0.281%. In other studies elsewhere, higher overland runoff related to greater compositions of barren land with lower hydraulic conductivity has been found at catchment scales (Ziegler and Giambelluca, 1997, 1998; Ziegler *et al.,* 2001, 2004b; Assouline and Mualem, 2002; Perkins *et al.,* 2007). In addition, a recent study conducted in Mississippi Basin (US) tributaries within the states of Arkansas and Missouri determined that compositions of this same landscape metric found significant in this study (percent barren land in the riparian stream buffer) were correlated to increase P in overland flow delivered to streams (Lopez *et al,* 2008). However in that study, percent barren land in the riparian stream buffer was found to be significant at a fixed 120 m width. The highest percentages of barren land (> 2.25%) in riparian stream buffers were in southeastern PA, central and the eastern shore of MD, southwestern DE, and central to southeastern VA (Figure 2.13).

2.5.2 Comparisons with B & P models of the Chesapeake Bay watershed

Since both the 31 m TN and TP SPARROW models and the B & P models are driven by the same non-LC/LU data, predictions of loadings and yields were similar. However, loading and yield allocations within similar catchments varied between the 31 m and B & P models. Utilizing an F-test, the inclusion of the new additional explanatory landscape metrics in the 31 m TN model indicated a significant (p value ≤ 0.05) difference from the B & P TN model. Additionally, a t-test indicated that barren land in the 31 m TP model was highly significance (p value ≤ 0.05) and provided evidence that the two TP models were statistically different. Thus, with the inclusion of these new metrics, initial biases found in the B & P models of their original explanatory variables not accounting for the entire variability in the observed, watershed-wide nutrient loading station estimates were slightly reduced (Figure 2.14a-d). This slight increase in the estimated accuracy between the observed and predicted measurements at stream loading sites was also indicated by the increased yield r^2 and decreased RMSE and AIC values from the 31 m to the B & P TN and TP models (Table 2.2).

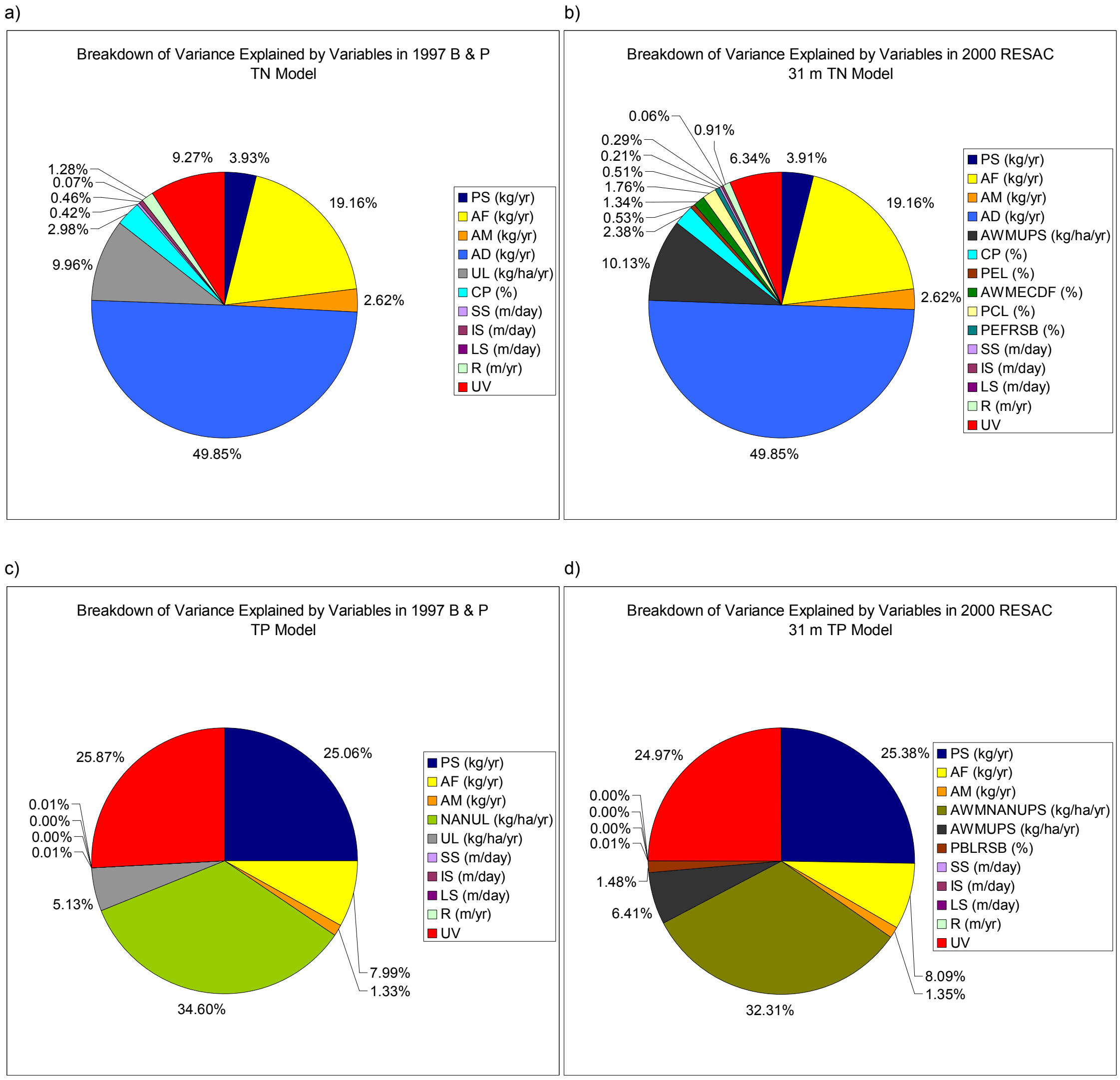

Figure 2.14: Comparison of the breakdown of variance explained by variables in the: **a)** 1997 B & P TN, **b)** 2000 RESAC 31 m TN, **c)** 1997 B & P TP, and **d)** 2000 RESAC 31 m TP models. PS = point sources, AF = applied fertilizer, AM = applied manure, AD = atmospheric deposition, UL = urban land, CP = coastal plain, SS = small streams, IS = intermediate streams, LS = large streams, R = reservoir, UV = unexplained variance, AWMUPS = area-weighted mean urban (≥ 10% ISA) patch size, PEL = percentage of extractive land, AWMECDF = area-weighted mean edge contrast of deciduous forest, PCL = percentage of cropland, PEFRSP = percentage of evergreen forest in the riparian stream buffer, NANUL = non-agricultural/non-urban land, and AWMNANUPS = area-weighted mean non-agricultural/non-urban patch size.

The TN annual loadings (kg/yr) estimated to enter the Chesapeake Bay from the 31 m model were 1.449 x 10^8, as compared to 1.480 x 10^8 generated from the B & P model, approximately 98% of the B & P model value. The mean annual TN yield (kg/ha/yr) per catchment draining directly to the Chesapeake Bay from the 31 m model was about 55.03, as compared to 62.58 from the B & P model. The largest increases in delivered yield (> 3.00 kg/ha/yr) to the estuary per catchment from the new model, as compared to the B & P model, were found near: central NY, central and southeastern PA, and northeastern MD in the upper, middle, and lower Susquehanna Basin; Cumberland (MD), northeastern WV, and northern VA in the upper-to-middle Potomac Basin; the eastern shore of MD; and DE (Figure 2.15a). The largest decreases in delivered yield (> 3.00 kg/ha/yr) to the estuary per catchment from the 31 m model, as compared to the B & P model, were found near: DC in the lower Potomac Basin and Richmond and Petersburg (VA) in the middle-to-lower James Basin (Figure 2.15a).

The B & P and 31 m models showed close agreement in coefficients for LC/LU sources (urban land and area-weighted mean urban (≥ 10% ISA) patch size), other sources (point, fertilizer and manure applications, and atmospheric N deposition), land-to-water delivery (coastal plain), stream decay (small, intermediate, and large), and reservoir decay (Table 2. 4). A comparison of the natural log of the 1997 B & P and 2000 31 m TN model loadings (kg/yr) in the six largest basins (James (27,019 km^2), Patuxent (2,479 km^2), Potomac (38,000 km^2), Rappahannock (7,405 km^2), Susquehanna (71,225 km^2), and York (6,915 km^2)) had an r^2 value of 0.9984 (Figure 2.16a). The York Basin is formed by the confluence of the Mattaponi and Pamunkey Basins in southeastern VA.

The TP annual loadings (kg/yr) estimated to reach the Chesapeake Bay from the new model were 5.367 x 10^6 versus 5.210 x 10^6 kg/yr estimated to reach the estuary from the B & P model, approximately 3% higher than the predicted loadings in the B & P model. The mean annual TP yield (kg/ha/yr) per catchment draining directly to the Chesapeake Bay from the 31 m model was about 2.38, as compared to 2.14 from the B & P model. The largest increases in delivered yield (> 0.08 kg/ha/yr) to the Chesapeake Bay per catchment from the 31 m model to the B & P model were near: Lancaster and Harrisburg (PA) in the lower Susquehanna Basin; Baltimore (MD); Frederick and southern MD, DC, and northern VA in the middle-to-lower Potomac Basin; Fredericksburg (VA) in the middle Rappahannock Basin; central VA in the upper York Basin; Norfolk-Virginia Beach-Newport News (VA) in the lower James Basin; the eastern shores of MD and VA; and DE (Figure 2.15b). The largest decreases in delivered yield (> 0.08 kg/ha/yr) to the estuary per catchment from the 31 m model to the B & P model were near: Baltimore (MD) and Lynchburg and Norfolk-Virginia Beach-Newport News (VA) in the upper-to-lower James Basin (Figure 2.15b).

Similarly to the TN model, the comparisons of the significant coefficients of LC/LU sources (urban land, area-weighted mean urban (≥ 10% ISA) patch size, non-agricultural/non-urban land, and area-weighted mean non-agricultural/non-urban patch size), other sources (point, fertilizer and manure applications), stream decay (small, intermediate, and large), and reservoir decay between the B & P and 31 m model were in close agreement (Table 2. 6). The comparison of the natural log of the 1997 B & P and 2000 31 m TP model loadings for the six largest basins in the watershed indicated an r^2 value of 0.9991 (Figure 2.16b).

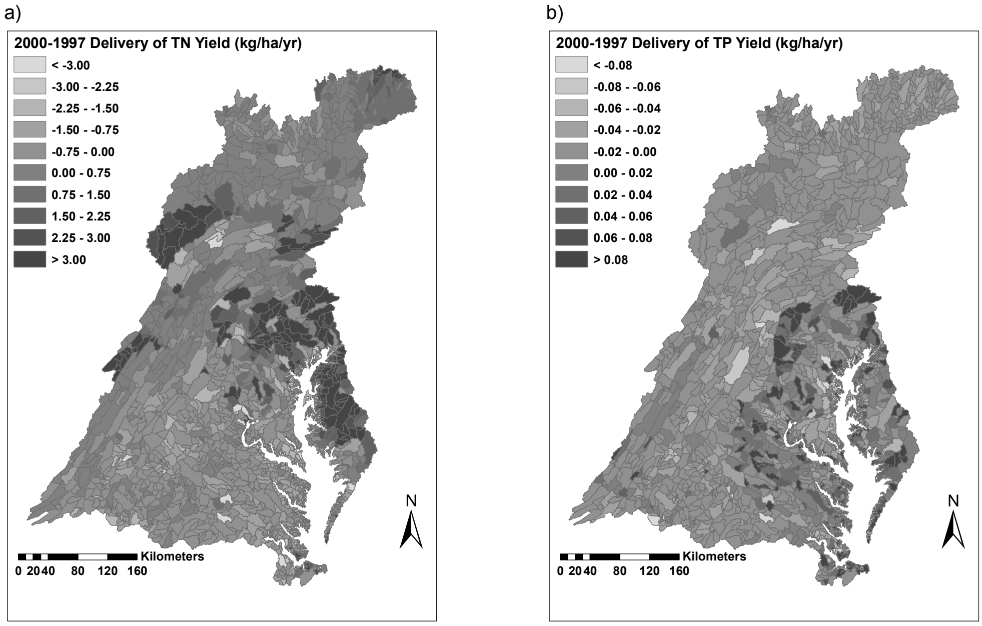

Figure 2.15: Per catchment estimated difference between the 2000 RESAC 31 m and 1997 B & P model in predicted yield (kg/ha/yr) delivered to the Chesapeake Bay for: **a)** TN and **b)** TP.

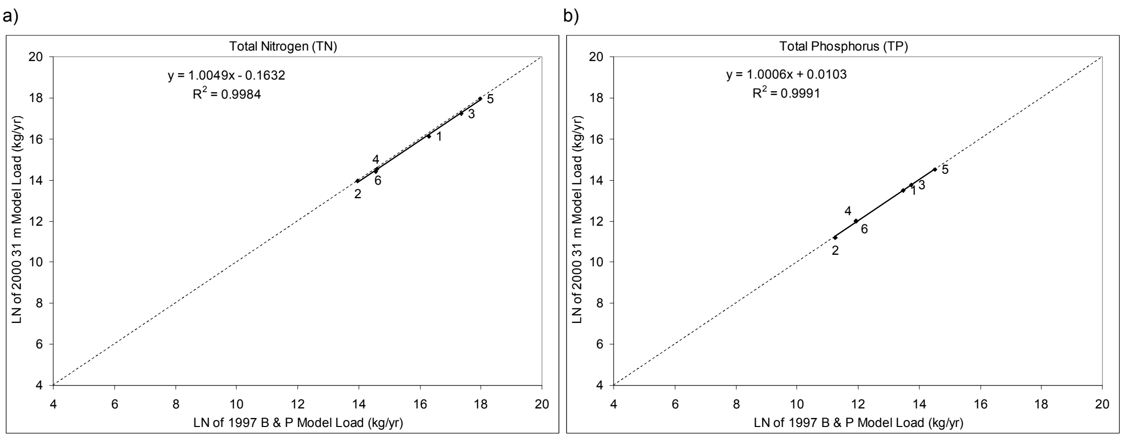

Figure 2.16: Comparison of the natural log of the 1997 B & P and 2000 RESAC 31 m model estimated loadings (kg/yr) for the James, Patuxent, Potomac, Rappahannock, Susquehanna, and the York Basins within the Chesapeake Bay watershed for: **a)** TN) and **b)** TP. 1 = the James River Basin, 2 = the Patuxent River Basin, 3 = the Potomac River Basin, 4 = the Rappahannock River Basin, 5 = the Susquehanna River Basin, and 6 = the York River Basin.

2.5.3 Comparisons between the Chesapeake Bay HSPF and SPARROW TN and TP models

The new TN and TP runoff model results were compared with the results of the Chesapeake Bay HSPF Phase 4.3 simulation parameterized upon 64 to 94 basins used from 1985-2000 (Wang *et al.,* 2001; Chesapeake Bay Program, 2007). Chesapeake Bay HSPF modeling has been in progress since 1982 as a management tool for the estuary's restoration. In comparison with the mean 1985-1994 TN loading of 1.420 x 10^8 kg/yr estimated to enter the Chesapeake Bay from the Phase 4.3 HSPF simulation (Wang *et al.,* 2001), the 2000 31 m TN model loadings were approximately 102% of the HSPF value. The 2000 31 m TN model loadings were nearly 112% of the 2000 Phase 4.3 HSPF TN model value of 1.292 x 10^8 kg/yr (Chesapeake Bay Program, 2007). The comparison between the natural log of the estimated 2000 Phase 4.3 HSPF and the 31 m model TN loadings of the six largest basins in the watershed indicated a r^2 of 0.9883 (Figure 2.17a). The close agreement between the SPARROW and the HSPF estimates suggest that the inclusion of new compositional and configuration representations of catchment and riparian stream buffer-wide urban and non-urban LC/LU in the SPARROW models could improve the precision of annual Chesapeake Bay TN runoff simulation..

The mean 1985-1994 estimated HSPF Phase 4.3 Bay TP loading was 9.991 x 10^6 kg/yr (Wang *et al.,* 2001). The 2000 31 m TP model results were only 54% of this HSPF value. Furthermore, the 2000 31 m TP model loadings were approximately 62% of the estimated 2000 Phase 4.3 HSPF Bay TP loadings of 8.673 x 10^6 kg/yr (Chesapeake Bay Program, 2007). A comparison between the natural log of the TP loadings for the 2000 Phase 4.3 HSPF and the SPARROW model in the six largest basins in the watershed also indicated a high r^2 of 0.8692 (Figure 2.17b). Although 2000 was 16% below the normal long-term mean Chesapeake Bay watershed-wide stream runoff (United States Geological Survey, 2001), these findings still suggest an underprediction of TP in SPARROW. However, the inclusion of new compositional metrics of watershed and riparian stream buffer-wide urban and non-urban LC/LU within SPARROW slightly improved the precision of predicted annual Chesapeake Bay-wide TP runoff against HSPF estimates.

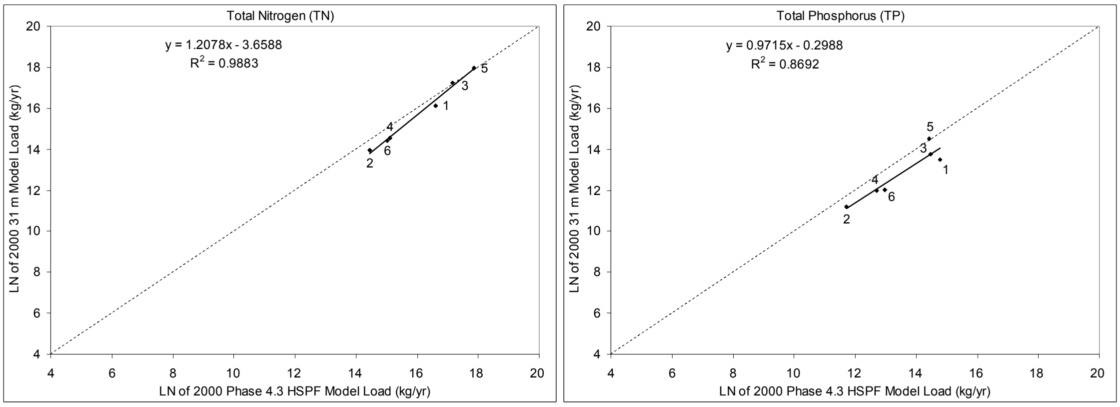

Figure 2.17: Comparison of the natural log of the 2000 Phase 4.3 HSPF and 2000 RESAC 31 m model estimated loadings (kg/yr) for the James, Patuxent, Potomac, Rappahannock, Susquehanna, and the York Basins within the Chesapeake Bay watershed for: **a)** TN) and **b)** TP. 1 = the James River Basin, 2 = the Patuxent River Basin, 3 = the Potomac River Basin, 4 = the Rappahannock River Basin, 5 = the Susquehanna River Basin, and 6 = the York River Basin.

2.6 Conclusions

This paper investigated the effect of LC/LU on simulations of regional nutrient loading to the Chesapeake Bay estuary using a modification of the USGS Version 3.0 Chesapeake Bay 1997 SPARROW (Brakebill and Preston, 2004 (B & P)) TN and TP models with the same catchments, but new compositional and configurational landscape metrics and a consideration of riparian stream buffers for non-point source and land-to-water delivery significance.. New watershed wide maps of LC/LU, % ISA, and % TC for 2000 were used. It was concluded that the new models improved the predictive ability of SPARROW and the precision of simulated annual TN and TP loadings reaching the estuary, as compared with HSPF estimates, indicating that these two approaches may now be more complimentary.

Catchment-wide compositional landscape metrics of area-weighted mean urban (≥ 10% ISA) and non-agricultural/non-urban patch sizes significantly depicted the generation of non-point N and P land sources eventually reaching the Chesapeake Bay. Whereas, compositional and configurational landscape metrics in catchments (percentage of extractive land, area-weighted mean edge contrast of deciduous forest, and percentage of cropland) that are thought to be associated with water quality at localized scales, were shown here to be significantly correlated with the delivery of non-point N loadings to larger, nested river networks draining the estuary.

Additionally, at the localized scale, riparian stream buffers are thought to attenuate nutrients eventually reaching the stream channel. However, at the scale of the entire watershed, the demonstration here of the significant effects of LC/LU on TN and TP runoff extends to riparian stream buffer compositional percentages of evergreen forest and barren land increasing delivery of non-point N and P, respectively, to the entire Chesapeake Bay. Hence, these representations of LC/LU at catchment and riparian stream buffer width scales should be used in future data-driven water quality models representative of the entire Chesapeake Bay TN and TP watershed runoff. The increased transport of non-point N and P from all compositional and configurational land-to-water delivery landscape metrics to Chesapeake Bay streams at both catchment and riparian stream buffer-wide scales suggest the vital role that overland and shallow subsurface flow processes may play in enhanced nutrient transport, as a result of decreased soil hydraulic conductivity linked to these and adjacent cover types.

In regards to the greatest reduction of non-point N transported to the Chesapeake Bay, future land management should be focused on reducing the highest percentages of extractive land in catchments located in central PA, western MD, and northeastern WV. Likewise, in terms of decreasing non-point P transported to streams draining the estuary, projected land management strategies should involve reducing percentages of barren land in riparian stream buffers within southeastern PA, central and the eastern shore of MD, southwestern DE, and central to southeastern VA. Thus, these and the rest of our findings are relevant to land managers, planners, lawmakers, and other stakeholders in making better-informed landscape decisions involving not only the overall nutrient health of Chesapeake Bay, but in similar watersheds elsewhere.

Chapter 3: Effects of projected future urban land cover on nitrogen and phosphorus runoff to Chesapeake Bay[2]

3.1 Abstract

This paper examined the effects of simulated land cover and land use (LC/LU) change from 2000 to 2030 on nutrient loadings to the Chesapeake Bay. The **SPA**tially **R**eferenced **R**egression **O**n **W**atershed Attributes (SPARROW) model was used with anticipated watershed-wide LC/LU change from a growth forecast model that provides spatially-explicit probabilities of conversion to impervious surface. The total nitrogen (TN) and total phosphorus (TP) loadings estimated to enter the Chesapeake Bay were reduced by 20% and 19%, respectively. In general, as development replaced other LC/LUs from 2000 to 2030, TN and TP runoff was significantly reduced by losses of non-point, non-urban source loadings, yields, and land-to-water delivery. The simulation results suggest future changes in landscape composition and configuration at catchment and riparian stream buffer width scales could lower TN and TP runoff to the estuary.

[2] The material in Chapter 3 was previously published. Roberts, A. D., Prince, S. D., Jantz, C. A., and Goetz, S. J., 2009. Effects of projected future urban land cover on nitrogen and phosphorus runoff to the Chesapeake Bay. Ecological Engineering 35(12): 1758-1772.

3.2 Introduction

Urbanization is a primary form of land cover and land use (LC/LU) change that is accelerating and has significant influence on watershed-wide environmental conditions. Urbanization converts croplands, forests, grasslands, pastures, wetlands, and other cover types to, in particular, residential and transportation, but also commercial and industrial uses, increasing significantly areas of impervious surfaces (Tsegaye *et al*., 2006). Globally, as population increases and shifts from rural areas to cities, urban expansion is inevitable. Furthermore, within the United States, population and its associated development is growing twice as fast in coastal areas as compared to inland areas (Bartlett *et al*., 2000; Conway and Lathrop, 2005). As urban land cover continues to increase, the incidence of non-point (diffuse) source nutrients, such as nitrogen (N) and phosphorus (P), in streams from impervious cover can be expected to rise significantly. These nutrients travel from land surfaces to streams as eroded organic and dissolved inorganic species via overland, shallow interflow, and even baseflow runoff processes. Excess nutrients are the main causes of eutrophication, hypoxia, and anoxia in rivers, estuaries, and coastal oceans (Paerl, 2006). Thus, the impacts on nutrient loading (mass for a specified time) estimates within rivers, estuaries, and coastal oceans of projected future urban growth are of interest. Some studies have examined scenarios of future urbanization on non-point source N and P in smaller watershed regions (Tsihrintzis *et al.,* 1996; Bhaduri *et al*., 2000; Costanza *et al*., 2002; Chang, 2004; Filoso *et al*., 2004; Tang *et al*., 2005; Wang *et al*., 2005), however larger regions have not been studied thoroughly, yet significant impacts on regional, national, and even global nutrient loadings can be expected.

To quantify the potential future nutrient loadings of a significant larger region, the Chesapeake Bay watershed was examined. The Chesapeake Bay is the largest estuary in the United States, with a watershed (166,534 km^2) encompassing portions of six states (New York (NY), Pennsylvania (PA), Delaware (DE), Maryland (MD), West Virginia (WV), and Virginia (VA)) and the District of Columbia (DC) (Figure 3.1a-b). The Chesapeake Bay also has the highest watershed land area per volume of water of any estuary in the United States, making runoff from the land surface critically important in determining the nutrient status of the estuary (Shuyler *et al*., 1995). While human-induced LC/LU transformations that lead to increases in N and P first appeared in the watershed in the mid-1600s (Boesch, 2006), this rate increased after the end of World War II in the late 1940s when the Chesapeake Bay population was still under 8 million (McConnell, 1995). By 2000, the watershed's population was approximately 15.7 million (Chesapeake Bay Program, 2008a), with expectations of close to 20 million by 2030 (Chesapeake Bay Program, 2008b). Clearly this increase will further drive human-induced LC/LU changes in the form of urbanization. Increases in nutrient delivery to the estuary resulting from these population and consequent LC/LU changes have been the primary focus of research and policy efforts relating to restoring the Chesapeake Bay through the Federal Water Pollution Control Act (FWPCA) of 1972 and the Clean Water Act (CWA) of 1977 (Morgan and Owens, 2001). Thus, watershed-wide examinations of projected future urbanization on delivered N and P loadings to the estuary are warranted.

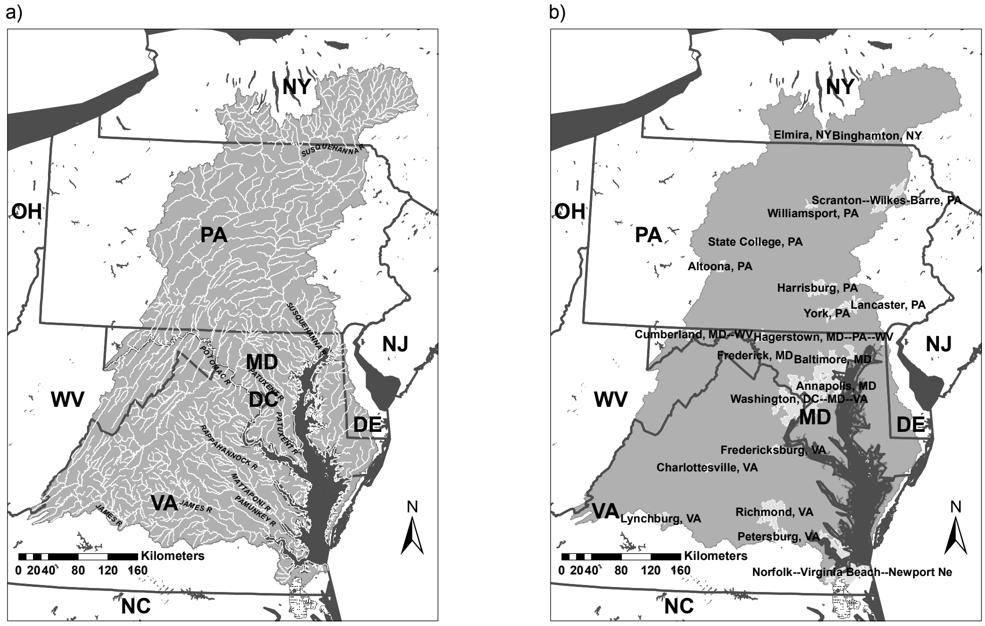

Figure 3.1: The Chesapeake Bay watershed showing the locations of: **a)** Streams and rivers draining the estuary and **b)** Urban centers located within its boundaries.

The spatial pattern of urban development in the Chesapeake Bay watershed is increasingly taking the form of low-density, decentralized residential and commercial development (Jantz *et al.,* 2004). Previous research has indicated that between 1970 and 2000, lot sizes throughout the watershed increased by 60% and that the average home size also increased from 1500 to 2265 ft^2 (Chesapeake Bay Program, 2008c). These trends are expected to continue over time (Chesapeake Bay Program, 2008c). It is reasonable to assume that water quality and aquatic habitats in the watershed will decline due to this urbanization, but low density development may have less effect than earlier, more concentrated expansion. Projection of the current trend of growth to 2030 may provide a better insight into the probable effects on the Chesapeake Bay nutrient status.

Watershed-wide, spatially-explicit, predictions of urbanization with 30 m resolution have been modeled by Jantz *et al.* (in press) using the **S**lope, **L**and use,

Exclusion, **U**rban extent, **T**ransportation, and **H**illshade (SLEUTH) urban growth model. SLEUTH is a cellular automaton, pattern-extrapolation model calibrated using urban development patterns in the past and forecasts of these patterns into the future (Jantz and Goetz, 2005). This version of SLEUTH was developed from the Clarke urban growth (Clarke *et al*., 1997) and land cover change models (United States Geological Survey, 2008a). SLEUTH has been applied to model urban growth in numerous areas (Clarke and Gaydos, 1998; Silva and Clarke, 2002; Arthur-Hartranft *et al*., 2003; Herold *et al*., 2003; Yang and Lo, 2003; Dietzel and Clarke, 2004; Solecki and Oliveri, 2004; Xian and Crane, 2005; Xian *et al*., 2005). In addition to the Jantz *et al.* (in press) recent application of SLEUTH to the entire Chesapeake Bay watershed, the model has been previously applied (although to smaller regions) in the watershed near Baltimore (MD), DC, and State College (PA) (Clarke and Gaydos, 1998; Carlson, 2004; Jantz *et al*., 2004; Claggett *et al*., 2005).

Increases in impervious surface areas may contribute more non-point N and P to the Chesapeake Bay as a result of increases in: leaf litter, vehicle emissions, residential and roadside landscaping (fertilizers), urban wildlife and pets, construction, and infrastructure (Minton, 2002).

The projected effects of watershed-wide urban growth and consequent effects on LC/LU and its changes on nutrient loadings were estimated using the United States Geological Survey's (USGS) **SPA**tially **R**eferenced **R**egressions **O**n **W**atershed **A**ttributes (SPARROW) model (Smith *et al*., 1997; Schwarz *et al.*, 2006). SPARROW estimates total nitrogen (TN) and total phosphorus (TP) runoff from watersheds of various sizes by statistical functions that relate upstream point and non-point sources, land-to-water delivery variables, and stream and reservoir nutrient attenuation (loss) processes that change TN and TP loadings as they travel downstream through nested river channel and reservoir networks. Quantities of contaminants in streams may be expressed as either loadings or yields (mass loading normalized by drainage area). Land-to-water delivery variables describe properties of the landscape relating climatic, natural, and human-induced surface processes affecting non-point N and P transport to streams.

The hybrid statistical-process structure of the model allows for the implementation of deterministic functions (such as first-order stream loss functions) with spatially-distributed components (such as sources and land-to-water delivery variables within stream networks) to account for the dendritic nature of watersheds (Alexander *et al.*, 2002b). SPARROW is based upon an average year and does not account for individual storms (that is, it is not event-based). The model simulates average annual discharge in terms of loadings and yields locally generated and delivered to the estuary, using just those input variables that have significant effects (p value ≤ 0.05). Local generation is the amount of TN or TP generated from within each catchment independent of any upstream loadings or yields, whereas delivered refers to the amount of TN or TP reaching the estuary after accounting for any upstream loadings and yields and also in-stream and reservoir losses. The model is applied to each of 2,339 catchments of the Chesapeake Bay watershed (Figure 3.2a) using a watershed map by Brakebill and Preston (2004). The model has been applied to the Chesapeake Bay watershed using the National Land Cover Dataset (NLCD) (Preston and Brakebill, 1999; Brakebill *et al*., 2001; Brakebill and Preston, 2004) and by Roberts and Prince (2010) incorporating the **R**egional **E**arth **S**cience **A**pplication **C**enter's (RESAC) remotely sensed LC/LU and percent impervious surface area (% ISA) maps for 2000 (Goetz *et al.*, 2003, 2004a, b; Jantz *et al.*, 2005).

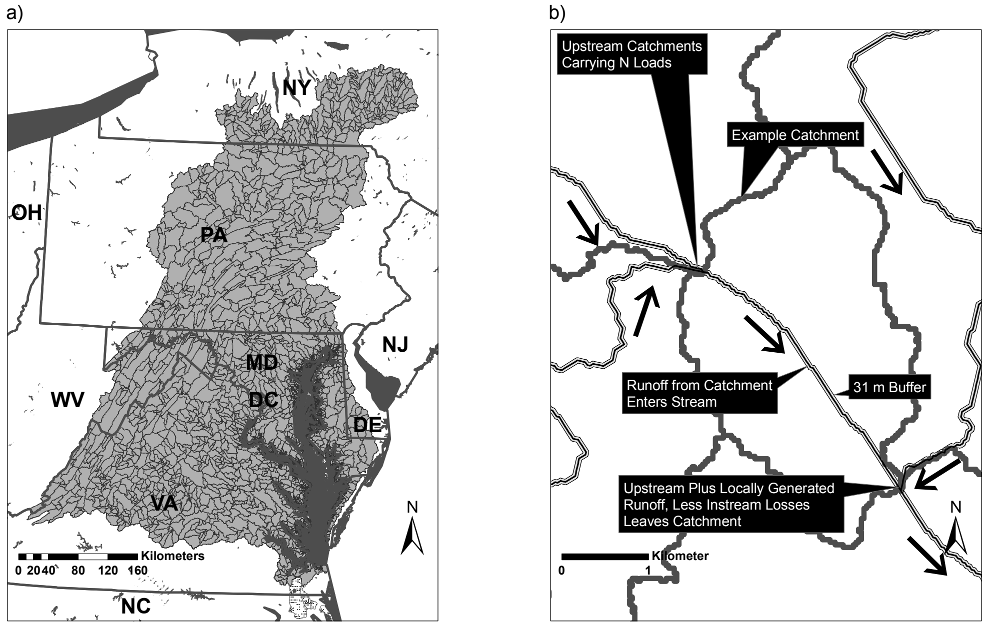

Figure 3.2: 2,339 catchments used in the: **a)** 2000 RESAC 31 m SPARROW Chesapeake Bay TN and TP models with **b)** Schematic diagram of an example catchment area showing fixed 31 m riparian stream buffer width area surrounding stream reach and corresponding upstream and downstream runoff paths.

Roberts and Prince (2010) used entire catchment and 31 m riparian stream buffer landscape metrics to specify non-point sources and land-to-water delivery variables that affect TN and TP loadings to the Chesapeake Bay. Landscape metrics describe the spatial structure of patches, the cover classes of patches, and patch mosaics, thus providing measures of composition (the variety and abundance of patch types) and configuration (spatial character and arrangement, position, and orientation of landscape elements (Leitao *et al.,* 2006)). The use of landscape metrics, in conjunction with projected future urban growth, has previously indicated that spatial alterations in LC/LU affect predicted N and P loadings in streams throughout a significant portion of Chesapeake Bay watershed (Wickham *et al.*, 2002), although a holistic watershed approach was not implemented. Even in remote reaches of the watershed, locally-dependent land development decisions that lead to more urban growth can adversely affect downstream loadings to the estuary. Thus, holistic watershed management is needed to bridge this gap between land use planning and comprehensive natural resource management (Conway and Lathrop, 2005). Modeling is needed to integrate the local catchment level to impacts on the entire Chesapeake Bay watershed.

Thus, the overall purpose of this study was to estimate future TN and TP runoff to the Chesapeake Bay, using SPARROW models, with maps of projected future urbanization.

3.3 Materials and methods

3.3.1 Future Chesapeake Bay watershed land cover and land use

Maps of the projected development in 2030 were derived from SLEUTH model runs for the Chesapeake Bay watershed (Jantz *et al.*, in press). The model was calibrated based upon data delivered to the Chesapeake Bay Program that was stratified by sub-region and two dates of impervious cover. For more information about the parameters and validation exercises used to calibrate these SLEUTH model runs, please consult Jantz *et al.* (in press).

Results from these SLEUTH model runs were then used to make projections of the probabilities of development in 2030 by acting as surrogates for impervious surface area (ISA) at the 30 x 30 m (900 m^2) pixel scale used to define urban areas in 2000 for the entire watershed. The probabilities were converted to a binary pixel classification of 0 that equaled projected non-urban areas and 1 that equaled projected urban areas by 2030. These new classes were then used to represent urban (≥ 10% ISA) (class 1) and non-urban (< 10% ISA) (class 2) areas characterized in 2000 by Roberts and Prince (2010). Class 2 pixels were assumed to remain in their existing twelve LC/LU classes, as given by the 2000 RESAC LC/LU map (Goetz *et al.*, 2004a, b; Jantz *et al.*, 2005). All fourteen LC/LU used in this study are given in Table 3.1. Complete accounts of the LC/LU mapping methodologies are given in Goetz *et al.* (2004b) and Jantz *et al.* (2005). Ten 2030 SLEUTH map runs were averaged to create mean values of 2030 LC/LU.

2000 and 2030 land cover classes	Class number
Urban (≥ 10% ISA)	1
Non-urban (< 10% ISA)	2
Urban/residential/recreational grasses	3
Extractive	4
Barren	5
Deciduous forest	6
Evergreen forest	7
Mixed (deciduous-evergreen) forest	8
Pasture/hay	9
Croplands	10
Natural grass	11
Deciduous wooded wetland	12
Evergreen wooded wetland	13
Emergent (sedge-herb) wetland	14

Table 3.1: The fourteen Chesapeake Bay watershed land cover classes used.

3.3.2 Landscape metrics and Chesapeake Bay SPARROW models

The models used here addressed some shortcomings of previous Chesapeake Bay TN and TP SPARROW models (Preston and Brakebill, 1999; Brakebill *et al.,* 2001; Brakebill and Preston, 2004), including: adding a relationship between LC/LU composition and the land-to-water delivery of non-point N and P; consideration of landscape configuration; and other spatial configuration factors such as LC/LU in riparian zones. Roberts and Prince (2010) found that LC/LU composition and configuration in catchments and composition in riparian stream buffers improved the accuracy and precision of TN and TP loadings estimates for 2000. A complete account of the definitions, techniques, and other methods used regarding the incorporation of landscape metrics and the 2000 SPARROW model calibrations is given in Roberts and Prince (2010).

In the 2000 RESAC 31 m models, five metrics (1-3, 6,7) measuring landscape configuration and two metrics (4-5) measuring composition (Leitao *et al.*, 2006) were initially evaluated in SPARROW for each catchment to quantify the effects of non-urban and urban land cover on current TN and TP runoff to the Chesapeake Bay. The following summaries indicate the properties that each metric measures. Complete definitions for all seven metrics are given in Leitao *et al.* (2006).

1) **Contagion** quantifies the degree to which LC/LU types were clumped in larger patches as opposed to dispersed in many smaller fragments.

2) **Area-weighted mean radius of gyration** measures connectivity using correlation length. This is the average distance one might traverse across a map from a random starting point and moving in a random direction while remaining in the patch.

3) **Patch number** indicates total number of patches of a particular LC/LU.

4) **Percentage of the landscape** area composed of a specified LC/LU.

5) **Area-weighted mean patch size** quantifies the sum, across all patches of a particular LC/LU, of patch area multiplied by proportional abundance of the patch.

6) **Area-weighted mean edge contrast** quantifies the amount of contrast between adjacent LC/LU patches. In this application, contrast is defined as physical characteristics of differing cover types that may influence nutrient transport and delivery.

7) **Area-weighted mean Euclidean nearest neighbor distance** quantifies the shortest distance from one patch to the next patch of the identical LC/LU type.

All area-weighted mean landscape metrics weight each patch by its size relative to the total area of that specific LC/LU, meaning that larger patches will exert greater influence than smaller patches and they are insensitive to extremely small patches.

Of the seven landscape metrics analyzed, only three (4-6) were found to be significant in the 2000 models (Table 3.2) and so the others were excluded. Thus, the SPARROW models were based upon landscape metrics for LC/LU in whole catchments and in just a 31 m riparian stream buffer. Riparian stream buffers are defined here as fixed, transitional areas between terrestrial landscapes and stream reaches created from the linked, spatially-referenced watershed network (Figure 3.2b). Non-point sources and land-to-water delivery variables from the models calibrated with 2000 data (Roberts and Prince, 2010) were used to model future Chesapeake Bay TN and TP loadings using maps of LC/LU classes in 2030.

The predictor variables (Table 3.2) used in the 2000 models are described below; for further details see Roberts and Prince (2010). In the TN model, for each kilogram (kg) of: 1) point sources discharged, 2) fertilizer and 3) manure applied to agricultural land (crop and pasture) and 4) atmospheric N deposited on the watershed, approximately 1.2, 0.2, 0.1, and 0.5 kg of N, respectively, were estimated in Chesapeake Bay streams annually. For each hectare (ha) of: 5) area-weighted mean urban ($\geq$ 10% ISA) patch size, nearly 25 kg of N was estimated in Chesapeake Bay streams annually. Urban patch size was used to estimate the buildup and washoff of N to sewers from urban non-point sources. For each percent (%) of: 6) land on the coastal plain, non-point N delivered to streams was estimated to decrease by 0.73% annually. Unlike the coastal plain, for each % of: 7) extractive, 8) cropland, and 9) area-weighted mean edge contrast of deciduous forest, non-point N delivered to streams was estimated to increase by 0.27, 0.021, and 0.014% annually, respectively. Area-weighted mean edge contrast of deciduous forest was used to measure the N transport differences between eleven non-urban classes and deciduous forest in the 2000 RESAC LC/LU map. Greatest differences between deciduous forest were with: urban/residential/recreational grasses, extractive, barren, pasture/hay, croplands, and natural grass. For each % of: 10) evergreen forest in the riparian stream buffer, non-point N delivered to streams was estimated to increase by 0.013% annually. In the case of all four of these land-to-water delivery metrics, these variables estimate land properties (reduced hydraulic conductivities), associated with these or surrounding cover types at the soil and shallow subsurface scales, that increased N-enriched overland and shallow subsurface runoff. For every meter (m) traveled in: 11) small (mean flow ≤ 100 cubic feet per second (ft^3/s)), 12) intermediate (mean flow >100 and ≤ 500 ft^3/s), and 13) large (mean flow > 500 ft^3/s) streams per day, about 25, 9, and 3% of the instream N was estimated to be lost, respectively. Finally, for any: 14) reservoir in the watershed, N was removed from the water column at an average settling velocity of over 14 m/yr.

Model component (units)	TN model Yield r^2 = 0.9366 RMSE = 0.2406			TP model Yield r^2 = 0.7503 RMSE = 0.3216		
	Variable	Coefficient Value	P value	Variable	Coefficient value	P value
Source (^{1} = kg/yr, ^{2} = kg/ha/yr)	Point^1	1.173	1.2×10^{-4}	Point^1	0.738	1.3×10^{-6}
	Applied fertilizer^1	0.175	3.9×10^{-6}	Applied fertilizer^1	0.016	1.1×10^{-3}
	Atmospheric deposition^1	0.492	2.0×10^{-7}	Applied manure^1	0.008	3.0×10^{-2}
	Applied manure^1	0.078	7.7×10^{-4}	* Area-weighted mean non-agricultural/non-urban patch size^2	0.110	5.8×10^{-13}
	* Area-weighted mean urban (≥ 10% ISA) patch size^2	24.885	7.2×10^{-7}	* Area-weighted mean urban (≥ 10% ISA) patch size^2	0.921	1.0×10^{-4}
Landscape delivery (%)	Percentage of coastal plain	-0.729	4.1×10^{-8}	* Percentage of barren land within the riparian stream buffer	0.281	1.2×10^{-6}
	* Percentage of extractive land	0.270	7.4×10^{-4}			
	* Area-weighted mean edge contrast of deciduous forest	0.014	7.2×10^{-3}			
	* Percentage of cropland	0.021	1.1×10^{-4}			
	* Percentage of evergreen forest within the riparian stream buffer	0.013	4.7×10^{-2}			
Stream decay (m/day)	Small streams	0.249	5.5×10^{-2}	Small streams	-0.198	1.3×10^{-1}
	Intermediate streams	0.090	3.2×10^{-1}	Intermediate streams	0.150	1.9×10^{-1}
	Large streams	0.030	4.8×10^{-1}	Large streams	0.034	5.2×10^{-1}
Reservoir decay (m/yr)	Reservoir	14.224	2.3×10^{-2}	Reservoir	19.019	5.7×10^{-2}

Table 3.2: All significant (p value ≤ 0.05) variables in the 2000 RESAC 31 m TN and TP SPARROW models (Roberts and Prince, 2010). RMSE = root mean squared error. * Denotes all significant landscape metrics.

In the TP model, for each kg of: 1) point sources discharged, 2) fertilizer and 3) manure applied to agricultural land, approximately 0.7, 0.02, and 0.01 kg of P, respectively, were estimated in Chesapeake Bay streams annually. For each ha of: 4) area-weighted mean non-agricultural/non-urban patch size, over 0.1 kg of P was estimated in Chesapeake Bay streams annually. This metric represented patches of mainly forest that exported P to streams as smaller predominant quantities of dissolved inorganic P via groundwater (baseflow) and shallow subsurface discharges. It was created by subtracting area-weighted mean cropland patch sizes from area-weighted mean non-urban patch sizes. For each ha of: 5) area-weighted mean urban (≥ 10% ISA) patch size, approximately 1 kg of P was estimated in Chesapeake Bay streams annually. For each % of: 6) barren land in the riparian stream buffer, non-point P delivered to streams was estimated to increase by 0.281% annually. This metric also represented reductions in hydraulic conductivity at the soil surface that increased P-enriched overland runoff. For every m traveled in: 7) small streams per day, an increase of about 20% P was estimated to occur, thus indicating small streams were a source by acting as a mechanism to erode P-enriched sediments. For each m traveled in: 8) intermediate and 9) large streams per day, about 15 and 3% of the instream P was estimated to be lost, respectively. Finally, for any: 10) reservoir in the watershed, P was removed from the water column at an average settling velocity of over 19 m/yr.

Although some stream and reservoir decay coefficients have p values > 0.05 that indicate these variables are statistically insignificant, these variables are still included in model calibrations on the grounds of being mechanistically significant within the SPARROW model structure.

In all, five TN and three TP landscape metrics were significant non-point sources or land-to-water delivery variables in the model (Table 3.2) (Roberts and Prince, 2010). The predicted values of these metrics in 2030 were calculated using forecasted 2030 LC/LU data (Table 3.1). ISA was modeled using SLEUTH while the other LC/LU classes in 2030 were estimated using the 2000 RESAC LC/LU map for all areas that were not predicted to become ISA.

3.3.3 Projected fertilizer and manure applications and point source loadings

The values of those source loading variables that were significant in the TN and TP models that are subject to LC/LU change, such as fertilizer and manure applications and point source loadings, were projected forward to 2030. The 2000 atmospheric deposition of N was unchanged since the aim of this work was to determine the effects of urbanization on source loadings and because of the difficulty in forecasting a variable that is largely determined by policy, regulation, and legal changes.

The 2030 annual commercial fertilizer and manure loadings were only considered for pasture and row-crop (croplands), as in the simulations by Brakebill and Preston (B & P) (2004). This was done to ensure consistency with B & P, since SPARROW is subject to changes in model structure; for example the number of source or land-to-water delivery variables found to be significant may change if the models are calibrated with new datasets. Fertilizer and manure application rates used in the Phase 5.0 **H**ydrologic **S**imulation **P**rogram **F**ORTRAN (HSPF) Chesapeake Bay model (Chesapeake Community Modeling Program, 2008) were used. These data provided fertilizer and manure loading rates in terms of several chemical forms of N and P, applied to cropland and pasture within 1,000 watershed segments. Fertilizer was defined as applications of ammonia-N (NH_3N) and/or nitrate-N (NO_3N) for N and phosphate-P (PO_4P) for P. Manure was defined as applications of ammonia-N (NH_3N), nitrate-N (NO_3N), and/or organic N for N and phosphate-P (PO_4P) and/or organic P for P. The data also included several management strategies (high till, low till, no till, and nutrient management) used on cropland and pasture for each month in the year. From these data, annual mean applied rates of N fertilizer were calculated; 28.02 and 15.83 kg/ha/yr, for cropland and pasture, respectively, whereas P fertilizer had rates of 17.19 and 5.16 kg/ha/yr for these same cover types. Annual mean applied manure rates for N manure were 9.79 and 52.74 kg/ha/yr, for cropland and pasture. Finally, P manure rates used for cropland and pasture were 5.56 and 23.77 kg/ha/yr.

Using the 2030 areas of cropland and pasture in all of the 2,339 model catchments, 2030 annual fertilizer and manure application loadings were tabulated. To determine the total 2030 fertilizer and manure application loadings of N and P, cropland and pasture quantities per catchment were combined. The 2000 and projected 2030 watershed-wide estimates of fertilizer and manure applications are given in Table 3.3.

Population throughout the watershed in 2030 was predicted for each catchment using an empirical correlation of population with SLEUTH ISA output transformed to housing density (United States Geological Survey, 2008b). For 2000, a non-urban land density of 0.0615 housing units/acre and an urban land density of 2.1 housing units/acre were used.

$$\boldsymbol{\log(\text{population density}) = 3.18 + \log(\text{housing density})} \quad (1)$$

Utilizing Equation 1, these housing densities lead to population densities of 93 people per square mile for non-urban land and 3,178 people per square mile for urban land. The 2000 population of the Chesapeake Bay watershed was 15,710,840 (Chesapeake Bay Program, 2008a) while the estimate using Equation 1 was 15,761,476 only 0.003% higher than the official tally. To project population densities in 2030 based upon housing densities, a non-urban land density of 0.0615 housing units/acre was once again used. However, a lower housing density of a 1.5 housing units/acre replaced the 2000 urban land value to represent the effect of continued increases in area occupied by each dwelling. These housing densities lead to population densities of 93 people per square mile for non-urban land and 2,270 people per square mile for urban land in 2030. Thus, the population estimated for 2030 was 19,761,581, quite similar to the near 20 million estimate made for the catchment as a whole by the Chesapeake Bay Program (2008b). Equation 1 was preferred, rather than whole-watershed estimates of future population, since the modeling was based on populations in each of the 2,339 catchments, not a single, Chesapeake Bay watershed-wide projection.

Recent wastewater treatment plant (WWTP) estimates show that, on average, 2.72 and 0.16 kg/yr of N and P are discharged per person into the watershed (Cummins, 2004). Using the projected urban population gains, in conjunction with these discharge values, provided estimates for 2030 N and P point source loadings (kg/yr) from each catchment with municipal WWTPs discharging into Chesapeake Bay waterways, as of 2000. For the 2,339 catchments, all estimated increases in point discharges were then assigned to their nearest WWTP for 2030 projections. A comparison of 2000 to the projected 2030 estimated increase in the point source N and P loadings discharged to streams draining the Chesapeake Bay is shown in Table 3. 3.

TN model					TP model				
Variable	2000	2030	2030-2000 change	2030-2000 % change	Variable	2000	2030	2030-2000 change	2030-2000 % change
Point	3.6605×10^7	4.8122×10^7	$+1.1517 \times 10^7$	+31.46	Point	2.6955×10^6	3.3730×10^6	$+6.7750 \times 10^5$	+25.13
Applied fertilizer	1.9346×10^8	7.6587×10^7	-1.1687×10^8	-60.41	Applied fertilizer	6.6409×10^7	3.6437×10^7	-2.9972×10^7	-45.13
Applied manure	8.7020×10^7	1.4673×10^8	$+5.9710 \times 10^7$	+68.62	Applied manure	7.3470×10^7	6.7662×10^7	-5.8080×10^6	-7.91
* Area-weighted mean urban (≥ 10% ISA) patch size	109	182	73	+66.97	* Area-weighted mean non-agricultural/ non-urban patch size	6,629	2,398	-4,231	-63.83
					* Area-weighted mean urban (≥ 10% ISA) patch size	109	182	73	+66.97
Percentage of extractive land	0.21	0.16	-0.05	-23.81	Percentage of barren land within the riparian stream buffer	0.43	0.35	-0.08	-18.60
Area-weighted mean edge contrast of deciduous forest	31.84	28.34	-3.50	-10.99					
Percentage of cropland	10.05	8.68	-1.37	-13.63					
Percentage of evergreen forest within the riparian stream buffer	5.66	5.31	-0.34	-6.18					

Table 3.3: Comparison of 2000 and projected 2030 Chesapeake Bay watershed-wide total discharged point source and applied fertilizer and manure loadings (kg/yr), land-based source variables (ha), and land-to-water delivery variables (%). * Denotes the averaged value of these land-based source variables from all 2,339 catchments.

3.4 Results

3.4.1 TN

TN annual loadings projected to be delivered to the Chesapeake Bay by 2030 were 1.171 x 10^8 kg/yr, as compared to 1.449 x 10^8 kg/yr estimated in 2000 (Roberts and Prince, 2010), about 19% less than the 2000 quantity (Table 3.4). Using the root mean squared error (RMSE) of the TN model (0.2406) (Table 3.2), uncertainties of the model predictions were obtained and indicated that this projected 19% reduction in total loadings was well within the range of change that predicted total loadings could of decreased by as much as over 50% or increased by upwards of 32% between 2000 and 2030 (Table 3.5). The highest increases in projected TN yield (> 4 kg/ha/yr) were predicted near: Harrisburg, Lancaster and York (PA); the northern and the eastern shore of MD; DE; and central VA (Figure 3.3a). Catchments with the largest decreases in projected TN yield (> 4 kg/ha/yr) were predicted near Baltimore (MD) and DC (Figure 3.3a). A comparison of the six largest basins - the James (27,019 km^2), Patuxent (2,479 km^2), Potomac (38,000 km^2), Rappahannock (7,405 km^2), Susquehanna (71,225 km^2), and York (6,915 km^2 basin formed by the confluence of the Mattaponi and Pamunkey in southeastern VA) from 2000 to 2030 indicated that annual TN loadings in three of these basins (the James, Patuxent, and Rappahannock) were predicted to increased, while the others decreased. In all six basins, the overall annual TN loadings were predicted to decrease by nearly 17% from 1.1001 x 10^8 in 2000 to 9.1776 x 10^7 by 2030 and were close to the Chesapeake-wide projected decline of 19%.

TN model					TP model				
Variable	2000	2030	2030-2000 change	2030-2000 % change	Variable	2000	2030	2030-2000 change	2030-2000 % change
Point	4.4105×10^7	5.2200×10^7	$+8.0950 \times 10^6$	+18.35	Point	1.9665×10^6	2.3677×10^6	$+4.0120 \times 10^5$	+20.40
Applied fertilizer	4.8395×10^7	1.5615×10^7	-3.2780×10^7	-67.73	Applied fertilizer	1.0226×10^6	5.4169×10^5	-4.8091×10^5	-47.03
Applied manure	9.7110×10^6	1.2783×10^7	$+3.0720 \times 10^6$	+31.63	Applied manure	5.2862×10^5	4.8027×10^5	-4.8350×10^4	-9.15
Atmospheric deposition	3.5803×10^7	2.7355×10^7	-8.4480×10^6	-23.60	Area-weighted mean non-agricultural/ non-urban patch size	1.5772×10^6	5.0174×10^5	-1.0755×10^6	-68.19
Area-weighted mean urban (≥ 10% ISA) patch size	6.9311×10^6	9.1806×10^6	$+2.2495 \times 10^6$	+32.46	Area-weighted mean urban (≥ 10% ISA) patch size	2.7222×10^5	4.0387×10^5	$+1.3165 \times 10^5$	+48.36
Total	1.4495×10^8	1.1713×10^8	-2.7820×10^7	-19.19	Total	5.3671×10^6	4.2953×10^6	-1.0718×10^6	-19.97
Point	16.43	16.98	+0.55	+3.35	Point	0.79	0.82	+0.03	+3.80
Applied fertilizer	3.84	1.01	-2.83	-73.70	Applied fertilizer	0.08	0.04	-0.04	-50.00
Applied manure	0.42	0.75	+0.33	+78.57	Applied manure	0.03	0.03	+0.00	+0.00
Atmospheric deposition	2.39	1.91	-0.48	-20.08	Area-weighted mean non-agricultural/non-urban patch size	0.11	0.04	-0.07	-63.64
Area-weighted mean urban (≥ 10% ISA) patch size	0.85	0.84	-0.01	-1.18	Area-weighted mean urban (≥ 10% ISA) patch size	0.03	0.04	+0.01	+33.33
Total	23.93	21.49	-2.44	-10.20	Total	1.04	0.97	-0.07	-6.73

Table 3.4: Comparison of 2000 and projected 2030 Chesapeake Bay watershed-wide total loadings (kg/yr) delivered to the estuary from all significant sources and mean yield (kg/ha/yr) from all 2,339 catchments.

TN model					TP model				
Variable	2000	2030	2030-2000 change	2030-2000 %	Variable	2000	2030	2030-2000 change	2030-2000 %
Point	4.4105×10^{7} $\pm 1.0612 \times 10^{7}$	5.2200×10^{7} $\pm 1.2559 \times 10^{7}$	-1.5076×10^{7} - $+3.1266 \times 10^{7}$	-27.55 - +93.35	Point	1.9665×10^{6} $\pm 6.3243 \times 10^{5}$	2.3677×10^{6} $\pm 5.6991 \times 10^{5}$	-9.9268×10^{5} - $+1.7951 \times 10^{6}$	-38.20 - +134.56
Applied fertilizer	4.8395×10^{7} $\pm 1.1644 \times 10^{7}$	1.5615×10^{7} $\pm 3.7570 \times 10^{6}$	-4.8181×10^{7} - -1.7379×10^{7}	-80.25 - -47.29	Applied fertilizer	1.0226×10^{6} $\pm 3.2887 \times 10^{5}$	5.4169×10^{5} $\pm 1.3038 \times 10^{5}$	-9.8399×10^{5} - $+2.2166 \times 10^{4}$	-72.81 - +3.20
Applied manure	9.7110×10^{6} $\pm 2.3365 \times 10^{6}$	1.2783×10^{7} $\pm 3.0756 \times 10^{6}$	-2.3401×10^{6} - $+8.4841 \times 10^{6}$	-19.42 - +115.05	Applied manure	5.2862×10^{5} $\pm 1.7000 \times 10^{5}$	4.8027×10^{5} $\pm 1.1560 \times 10^{5}$	-3.7281×10^{5} - $+2.7611 \times 10^{5}$	-53.64 - +76.99
Atmospheric deposition	3.5803×10^{7} $\pm 8.6142 \times 10^{6}$	2.7355×10^{7} $\pm 6.5816 \times 10^{6}$	-2.3644×10^{7} - $+6.7478 \times 10^{6}$	-53.23 - +24.82	Area-weighted mean non-agricultural/ non-urban patch size	1.5772×10^{6} $\pm 5.0723 \times 10^{5}$	5.0174×10^{5} $\pm 1.2077 \times 10^{5}$	-1.7440×10^{6} - -4.0687×10^{5}	-83.67 - -38.03
Area-weighted mean urban (≥ 10% ISA) patch size	6.9311×10^{6} $\pm 1.6676 \times 10^{6}$	9.1806×10^{6} $\pm 2.2089 \times 10^{6}$	-1.6270×10^{6} - $+6.1260 \times 10^{6}$	-18.92 - +116.39	Area-weighted mean urban (≥ 10% ISA) patch size	2.7222×10^{5} $\pm 8.7546 \times 10^{4}$	4.0387×10^{5} $\pm 9.7212 \times 10^{4}$	-8.5781×10^{4} - $+3.4908 \times 10^{5}$	-23.84 - +189.03
Total	1.4495×10^{8} $\pm 3.4875 \times 10^{7}$	1.1713×10^{8} $\pm 2.8181 \times 10^{7}$	-9.0876×10^{7} - $+3.5236 \times 10^{7}$	-50.54 - +32.01	Total	5.3671×10^{6} $\pm 1.7261 \times 10^{6}$	4.2953×10^{6} $\pm 1.0339 \times 10^{6}$	-4.1792×10^{6} - $+2.0356 \times 10^{6}$	-58.92 - +55.91
Point	16.43 ±3.95	16.98 ±4.09	-7.49 - +8.59	-36.74 - +68.83	Point	0.79 ±0.25	0.82 ±0.20	-0.48 - +0.54	-46.72 - +102.21
Applied fertilizer	3.84 ±0.92	1.01 ±0.24	-4.00 - -1.66	-83.90 - -57.03	Applied fertilizer	0.08 ±0.03	0.04 ±0.01	-0.08 - +0.00	-74.33 - -2.59
Applied manure	0.42 ±0.10	0.75 ±0.18	+0.05 - +0.61	+9.31 - +191.72	Applied manure	0.03 ±0.01	0.03 ±0.01	-0.20 - +0.20	-48.67 - +94.81
Atmospheric deposition	2.39 ±0.58	1.91 ±0.46	-1.51 - +0.56	-51.08 - +30.56	Area-weighted mean non-agricultural/non-urban patch size	0.11 ±0.04	0.04 ±0.01	-0.12 - -0.02	-81.33 - -29.16
Area-weighted mean urban (≥ 10% ISA) patch size	0.85 ±0.20	0.84 ±0.20	-0.42 - +0.40	-39.51 - +61.44	Area-weighted mean urban (≥ 10% ISA)patch size	0.03 ±0.01	0.04 ±0.01	-0.01 - +0.03	-31.56 - +159.75
Total	23.93 ±5.76	21.49 ±5.17	-13.37 - +8.49	-45.03 - +46.71	Total	1.04 ±0.33	0.97 ±0.23	-0.72 - +0.58	-52.12 - +81.70

Table 3.5: Comparison of 2000 and projected 2030 Chesapeake Bay watershed-wide total loadings (kg/yr) delivered to the estuary from all significant sources and mean yield (kg/ha/yr) from all 2,339 catchments with prediction errors for each year, range of 2030-2000 change, and range of 2030-2000% change.

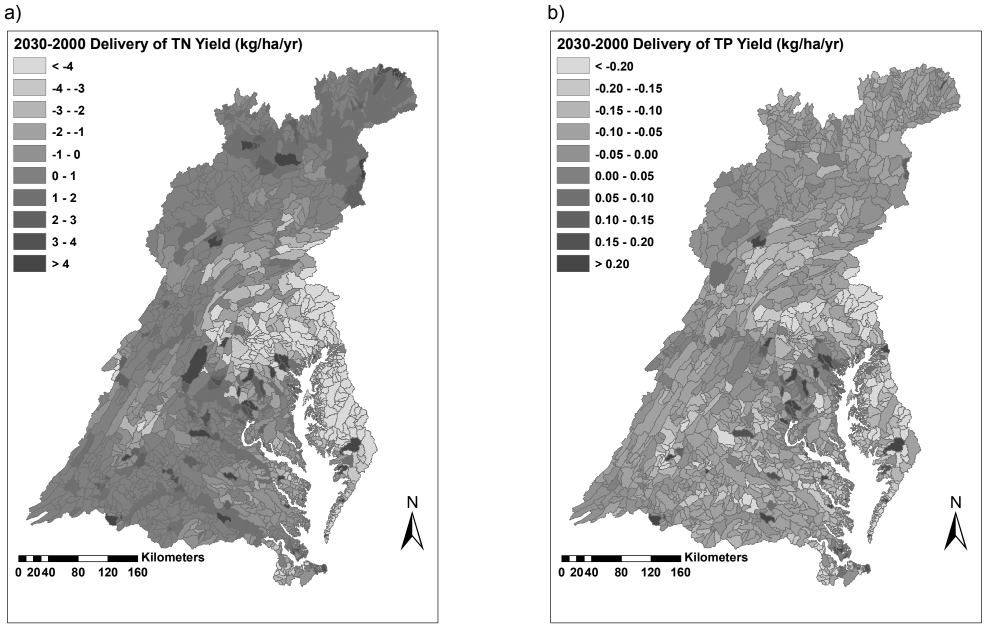

Figure 3.3: Per catchment estimated 2030-2000 difference maps of the total yield in kg/ha/yr per year for: **a)** N and **b)** P delivered to the Chesapeake Bay.

3.4.2 TP

2030 TP annual loadings projected to reach the Chesapeake Bay estuary were 20% lower (4.295 x 10^6 kg/yr) than the 5.367 x 10^6 kg/yr predicted to enter the estuary in 2000 (Roberts and Prince, 2010) (Table 3.4). Based upon the RMSE for the TP model (0.3216) (Table 3.2), predictions of model uncertainty for the change in TP loadings indicated that this 20% reduction value was also well within the range of change that projected total loadings could of decreased by as much as 59% or increased by 56% between 2000 and 2030 (Table 3.5). Catchments with the highest increases in projected TP yield (> 0.20 kg/ha/yr) were also near: Harrisburg, Lancaster, and York (PA); northern and the eastern shore of MD; DE; and central VA (Figure 3.3b). The largest decreases in projected TP yield (> 0.20 kg/ha/yr) were predicted to occur in the same regions as the largest decreases in TN yield (Figure 3.3b). The annual TP loadings comparison of the watershed's six largest basins from 2000 to 2030 was also projected to show an overall decline of approximately 21%, with TP loadings decreasing in all individual basins.

3.5 Discussion

3.5.1 Agricultural land losses and reductions in total and agricultural non-point loadings

Overall, the 2030 SPARROW modeled results showed that the predicted conversion of agricultural land to urban uses throughout the Chesapeake Bay watershed can be expected to result in significant reductions in delivered TN and TP. Agriculture is currently the single largest contributor of Chesapeake Bay nutrient pollution, representing 39% and 49% of its N and P loadings (Sims and Coale, 2002). Furthermore, since World War II, the geographic intensification of the use of commercial chemical fertilizers and animal agriculture within regions, such as the lower Susquehanna Basin in southeastern PA the eastern shore of MD, and DE, have increased agricultural nutrient runoff to the Chesapeake Bay by substantially increasing N and P available for non-point source runoff to streams. Thus, as a result of the predicted conversion of agricultural (crop and pasture) land and the lower estimated rates of N and P fertilizer loadings to be applied to these remaining lands by 2030, the projected overall reduction in TN and TP seen here result from smaller quantities of fertilizer loadings delivered to Chesapeake Bay streams. Lower applications of fertilizers represented smaller loadings of eroded organic and dissolved inorganic N and P species available on the land surface and in the shallow subsurface for non-point source delivery to the Chesapeake Bay when transported to streams via overland, shallow interflow, and even baseflow runoff processes.

With the recent adoption of agricultural best management practices (BMPs), such as conservation-tillage and off-season (winter) cover crop conservation programs, total applications of commercial fertilizer are expected to decline (Sims and Coale, 2002). Applications of manure are expected to increase or stay similar to 2000 quantities to provide for crop nutrient needs. Both of these trends are incorporated in the projections of total fertilizer and manure loadings applied for 2030 (Table 3.3) and help explain the overall, estimated 2030 nutrient reduction trends.

Watershed-wide agricultural land was predicted to decrease from approximately 25% in 2000 to 22% in 2030, with the greatest decreases (> 9%) predicted to occur in the most intensely farmed catchments of the lower Susquehanna Basin of southeastern PA, the eastern shore of MD, and DE. This finding has great significance since these catchments were also found in the results for 2000 to produce disproportionately the highest TN (> 18 kg/ha/yr) and TP (> 0.99 kg/ha/yr) delivered yields to the Chesapeake Bay (Roberts and Prince, 2010). This was mainly a product of the substantial, agricultural, non-point source losses associated with the highest applications rates of fertilizer and manure that occurred throughout the watershed. However, by 2030, the mean delivered yield from these highest producing catchments in 2000 was projected to decrease nearly 11% and 1% for TN and TP, respectively. The declines in mean TN and TP yield seen in these 2000 highest producing catchments were correlated with the anticipated, substantial decreases in applied fertilizer N and P.

The results reported here are similar to those of other studies of the effects of development on future nutrient loadings in smaller watersheds. The increase in TN of between 0.13-0.21% for the Saint Louis, Missouri region estimated by Wang *et al*. (2005)

from 2005-2030 was an effect of their projected extreme urbanization event. Similarly, Tang *et al.* (2005) evaluated non-point source nutrient loading differences in north-central Michigan from 1978-2040 with predicted urbanization and determined that after development, TN and TP losses would also only slightly increase (< 3%). As in the previous study, Tang *et al.* (2005) projected an extreme increase in urban land from 4.2 to 11.5%, nearly 300%, as compared to <200% in the present study. Thus, in both these studies, the small increases on TN and TP runoff were obtained with overwhelming gains in impervious-based runoff to streams. Without these very high changes in imperviousness, the reduction in non-point losses of fertilizer and manure would of lead to declines in TN and TP loadings. Notwithstanding, even though there were gains in non-point N and/or P due to urbanization, much greater nutrient increases were limited by the conversion of agricultural lands that have higher nutrient contribution than urban uses (Tang *et al*., 2005; Wang *et al*., 2005).

Bhaduri *et al.* (2000) utilized a land use simulator to estimate impervious surface growth near Indianapolis, Indiana from 1973-1991 and found that, over this period, an 18% increase in impervious areas resulted in a 15% decrease in non-point source N and P loadings. The decline in non-point N and P loadings was directly attributed to losses of agricultural lands. Furthermore, in Miami, Florida, Tsihrintzis *et al.* (1996) found that a specific agricultural land area would have significant non-point source N and P reductions of 54% and 35%, respectively, if it was entirely converted to urban use. In both of these studies, these reductions were primarily due to the decreases in fertilizer use.

By substantially lowering the amount of estimated, delivered fertilizers and manure through land conversion in those catchments that contributed the most TN and TP, such as those in the lower Susquehanna Basin of southeastern PA, a greater proportional effect on Chesapeake Bay water quality than elsewhere can be expected. In the Susquehanna Basin alone, TN and TP fell by about 33 and 30%, respectively. This finding is quite significant since the Susquehanna Basin contributes about 50% of water that enters the Chesapeake Bay annually (Susquehanna River Basin Commission, 2008), thus even moderate percentage reductions in loadings would significantly reduce TN and TP entering the estuary. Chang (2004) used a land use change model of proposed development from the late 1990s to 2030 in several lower Susquehanna Basin catchments and also found that agricultural land conversion to urban uses decreased overall, non-point source P loadings. These trends in smaller parts of the watershed support the findings reported here, that, for the entire watershed, the mean delivered yield will decrease by about 10% and 7% for TN and TP from 2000 to 2030 (Table 3.4).

3.5.2 Forest (and other non-agricultural and non-urban) land losses and reductions in non-point loadings

Conversion of forest and other non-agricultural and non-urban land areas to development was also found to correlate with the predicted reduction in TP by 2030. P export from these primarily forested regions of the watershed was shown by the yield of area-weighted mean non-agricultural/non-urban patches that indicated larger patches have greater influences on non-point source generation (Roberts and Prince, 2010). By 2030, these large, contiguous patches that contributed dissolved inorganic P to streams were predicted to be substantially smaller as a result of development that stopped these infiltration-based P runoff processes. This was seen in the averaged area-weighted mean non-agricultural/non-urban patch size that decreased nearly 64% from 2000 to 2030 (Table 3.3), in step with a 67% reduction in its mean delivered yield (Table 3.4), and the over 19% decline in overall TP delivered to the Chesapeake Bay (Table 3.4). The catchments with the largest losses in area weighted mean non-agricultural/non-urban patch size (>36,000 ha) were predicted to occur throughout the watershed (Figure 3.4a). Catchments with the highest decreases in delivered P yield (> 0.27 kg/ha/yr) from this source were predicted to occur only near central VA (Figure 3.5).

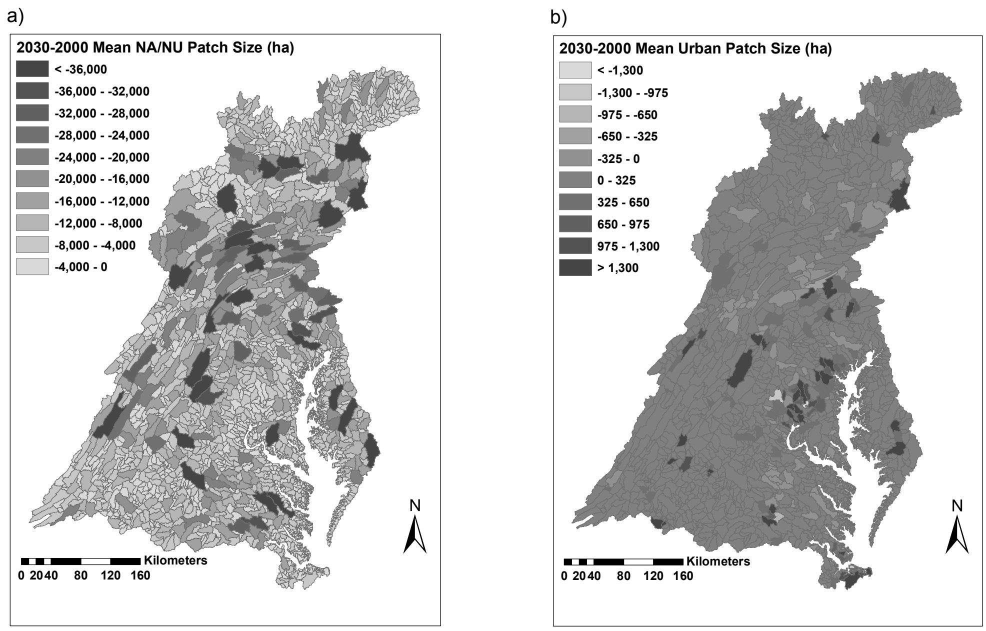

Figure 3.4: Per catchment 2030-2000 difference maps of the area-weighted mean: **a)** Non-agricultural/non-urban (NA/NU) and **b)** Urban (≥ 10% ISA) patch size source metrics in ha.

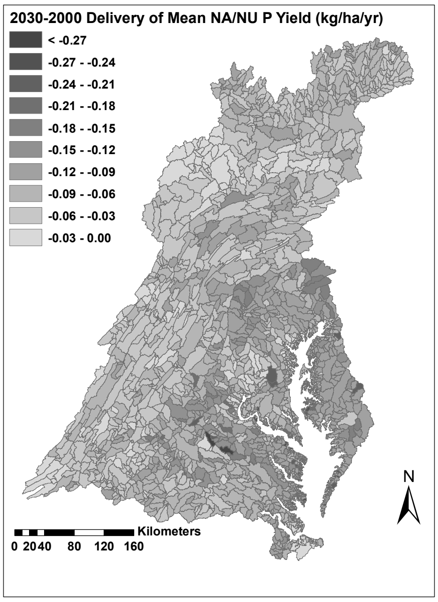

Figure 3.5: Per catchment estimated 2030-2000 difference map of the NA/NU patch size yield in kg/ha/yr for P delivered to the Chesapeake Bay.

3.5.3 Impervious surface area gains and changes in urban non-point loadings

As a direct result of increases in urban non-point source loadings, TN was projected to increase in the range of 1-33% from 2000 to 2030 in the James, Patuxent, and Rappahannock Basins. This result is similar to that of Costanza *et al.* (2002) for the Patuxent, who compared mean delivered TN concentration to the estuary in 1997 with a future "buildout" scenario and found an overall 14% gain. The catchments with the greatest increases in non-point urban N yields (> 2.00 kg/ha/yr) were near: Baltimore, Cumberland, and Frederick (MD); northeastern WV; DC; and Richmond, Norfolk-Virginia Beach-Newport News, and Lynchburg (VA) (Figure 3.6a). Catchments with the largest increases in non-point urban P yields (> 0.08 kg/ha/yr) were projected to occur in the same regions as urban N with the exceptions of: Cumberland and Frederick (MD) and northwestern WV (Figure 3.6b). Catchments with the greatest decreases in non-point N and P yields (> 2.00 and > 0.08 kg/ha/yr, respectively) were projected to occur only near DC (Figure 3.6b).

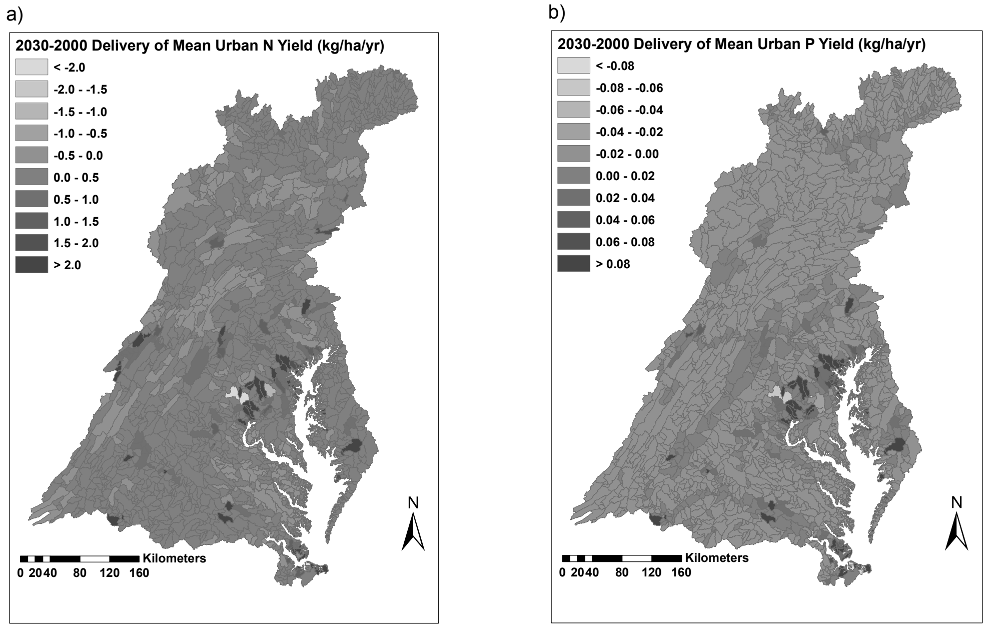

Figure 3.6: Per catchment estimated 2030-2000 difference maps of the area-weighted mean urban (≥ 10% ISA) patch size yield in kg/ha/yr for: **a)** N and **b)** P delivered to the Chesapeake Bay.

Surprisingly however, from 2000 to 2030, the mean non-point urban N and P yields delivered to the Chesapeake Bay for all 2,339 catchments were projected to remain virtually the same, decreasing and increasing by only 0.01 kg/ha/yr, respectively (Table 3.4). Non-point urban yields were modeled using area-weighted mean urban (≥ 10% ISA) patches so that larger patches of development, as opposed to the same area in smaller patches, had greater influences on non-point N and P generation (Roberts and Prince, 2010).

The results of the simulation of non-point N and P yields to streams by 2030 are counter-intuitive since the increase in urbanization might have been expected to increase the runoff of nutrients. The expectation of an increase in pollution by urbanization is also suggested by the SLEUTH projections that non-urban LC/LUs will be converted to development from 7% in 2000 to 13% in 2030. The expectation is yet further reinforced since the averaged area-weighted mean urban (≥ 10% ISA) patch size for all 2,339 catchments was predicted to increase by 67% from 2000 to 2030 (Table 3.3). Catchments with the largest increase in area weighted mean urban (≥ 10% ISA) patch size (>1,300 ha) were predicted to occur near: Harrisburg (PA); Baltimore and Hagerstown (MD); DC; and Richmond, Norfolk-Virginia Beach-Newport News, and Lynchburg (VA) (Figure 3.4b). Catchments with the highest decreases in this patch size (>1,300 ha) were predicted only near DC (Figure 3.4b).

However, TN and TP loadings in streams and reservoirs are attenuated by processes that include denitrification under anaerobic conditions, biological uptake by stream organisms, and sedimentation onto stream and reservoir floors (Alexander *et al*., 2000). Thus, any increase in non-point N and P discharge from impervious surfaces may be diminished by cumulative downstream water attenuation processes. The greatest increases in imperviousness were predicted to occur in catchments within the small and intermediate stream categories that were estimated to attenuate the highest percentages of instream N loadings throughout the watershed. In the case of P, however, no attenuation occurs in small streams and may explain the overall slight increase in mean delivered P yield projected to the Chesapeake Bay by 2030.

Filoso *et al.* (2004) found that after the projected conversion of 44% forested land to urban uses from 1991 to 2101 in the Ipswich Basin of Massachusetts, gains in ammonium-N concentrations were trivial (0.2-0.5 μM). The lack of effect on N could have been a result of ammonia volatilization and/or ammonium sorption to sediments within the stream channel (Filoso *et al.*, 2004). Thus, the expected increases in non-point N, as a result of urbanization, were reduced in part due to significant stream attenuation.

Only in catchments with unusually high rates of estimated urbanization were the projected non-point urban N gains large enough to overwhelm downstream attenuation. Thus, expected substantial gains in mean non-point urban yield delivered to the Chesapeake Bay from 2000 to 2030 did not occur.

3.5.4 Changes in point sources with urbanization

Gains in population and urban land throughout the watershed must lead to an increase in point source loadings, as well as changes in non-point sources discussed above. Using the models, it was shown that projected increases in delivered point source loadings from 2000 to 2030 would offset the TN and TP loading losses to the Chesapeake Bay caused by non-urban, non-point source reductions. The delivered N and P point loadings to the estuary were projected to increase over 18 and 20%, respectively, between 2000 and 2030 (Table 3.4). Similarly to the non-point loadings, the only attenuation processes reducing point N and P loadings delivered to the estuary occurred in streams and reservoirs presumably via denitrification, biological uptake, sedimentation, etc.

The negation of the reductions caused by reduction in agriculture, as a result of increases in point sources, must be qualified by recognition of the uncertainties in estimation of future population size and geographical distribution. Furthermore, the projected 2030 discharge additions were based on WWTP estimates of domestic effluent discharge that do not take into account gains or losses from industrial or commercial point sources. Neither do these loadings take into account the future locations of WWTPs, nor any future advances in effluent removal technology that can be expected to decrease these loadings substantially. All of these are limitations of the point loadings with the result that they cannot be quantified with great accuracy.

3.5.5 Land-to-water delivery losses and reductions in non-urban, non-point loadings

The results indicated that reductions in land-to-water delivery variables, resulting from the conversion of non-urban LC/LUs to development, also contributed to the decreases in delivered TN and TP to the Chesapeake Bay. Roberts and Prince (2010) showed that several landscape metrics were significantly related to increased non-point delivery to Chesapeake Bay streams. For N these were: percentage of extractive land, percentage of cropland, area-weighted mean edge contrast of deciduous forest, and percentage of evergreen forest within the riparian stream buffer and, for P, percentage of barren land within the riparian stream buffer.

The significance of these findings is that changes in the spatial composition and configuration of LC/LU can be expected to provide a means of reduction in land-to-water delivery. Increases in non-point N and P transport by landscape compositional and configuration changes might be caused by processes such as reduced soil hydraulic conductivity properties and increases in overland and shallow subsurface flow paths (Roberts and Prince, 2010). Overland and shallow subsurface flows are favored by compacted or saturated non-urban surfaces often found associated with extractive lands. Croplands, evergreen forests, and barren lands have been shown to have this effect.

Stormflow in some Chesapeake Bay tributaries has been suggested to provide pathways for non-point N to reach streams, allowing surface flow to traverse forested riparian buffers (Norton and Fisher, 2000). Changes of LC/LU associated with development may eliminate some surfaces with limited hydraulic conductivity, such as barren land, and contribute to reduction of non-point N and P transport.

The catchments in which significant changes in factors that affect surface and shallow subsurface flow were not evenly distributed across the Chesapeake Bay watershed. The largest decreases in percentage of extractive land (> 0.27%) were predicted in: central and east-central PA; western and central MD; northeastern WV; DC; and northern VA (Figure 3.7a). Catchments with the greatest decreases in percentage of cropland (> 9%) were predicted in: southern PA; the eastern shore of MD; and DE (Figure 3.7b). The catchments with the greatest decreases in area-weighted mean edge contrast of deciduous forest (> 2.0%) were predicted throughout the watershed, whereas the greatest increases in area-weighted mean edge contrasts of deciduous forest (> 2.0%) were predicted near: east-central MD and Norfolk-Virginia Beach-Newport News (VA) (Figure 3.7c). Catchments with the largest decreases in percentage of evergreen forest within the riparian stream buffer (> 1.35%) were predicted watershed-wide (Figure 3.7d). Finally, the catchments with the greatest decreases in the percentage of barren land within the riparian stream buffer (> 0.45%) were predicted near: Lancaster (PA); Baltimore and Frederick (MD); DC; and Norfolk-Virginia Beach-Newport News (VA) (Figure 3.8).

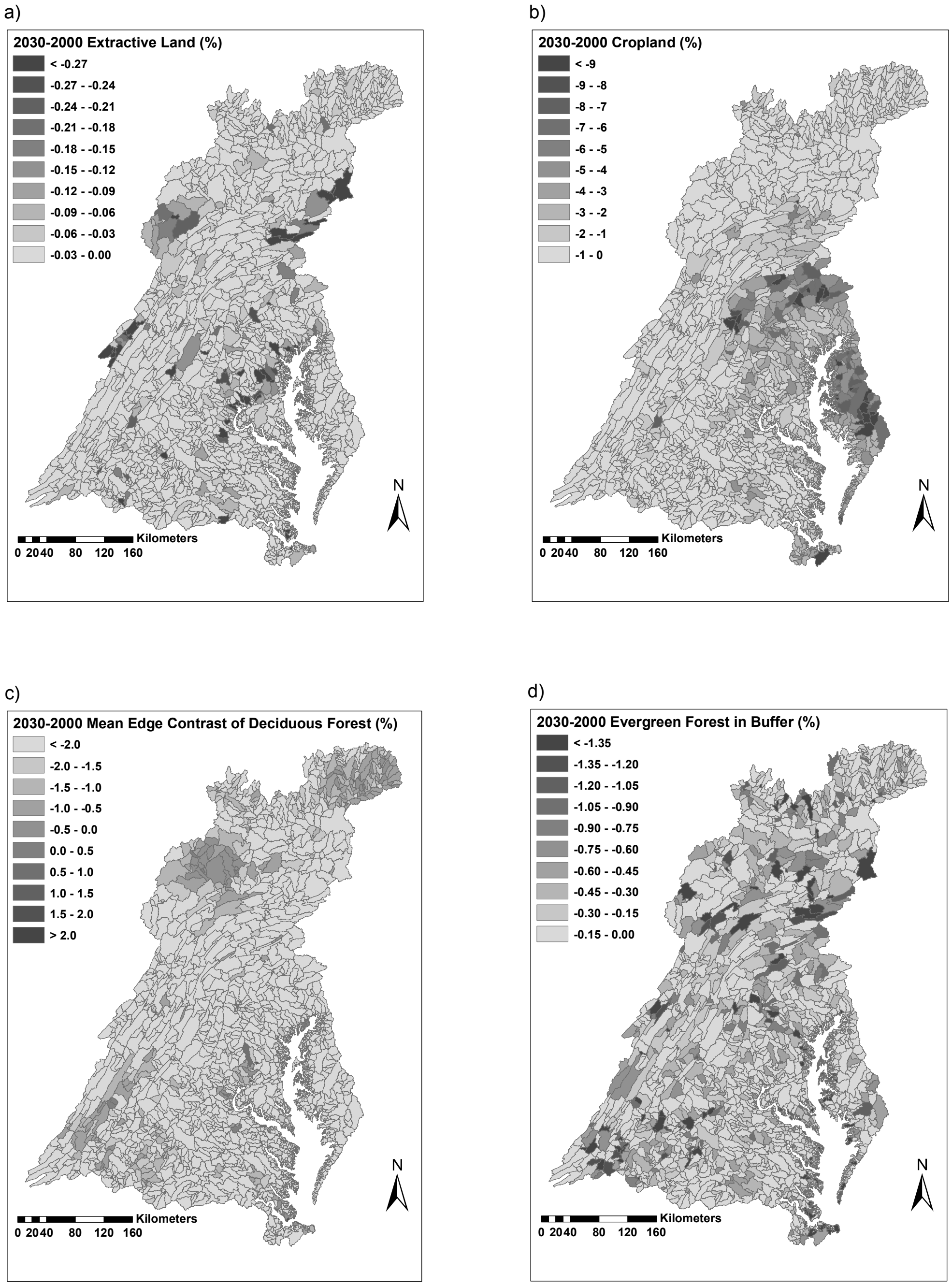

Figure 3.7: Per catchment 2030-2000 difference maps of the: **a)** Percentage of extractive land, **b)** Area-weighted mean edge contrast of deciduous forest land (%), **c)** Percentage of cropland, and **d)** Percentage of evergreen forest land in the 31 m riparian stream buffer land-to-water delivery metrics for the RESAC TN model.

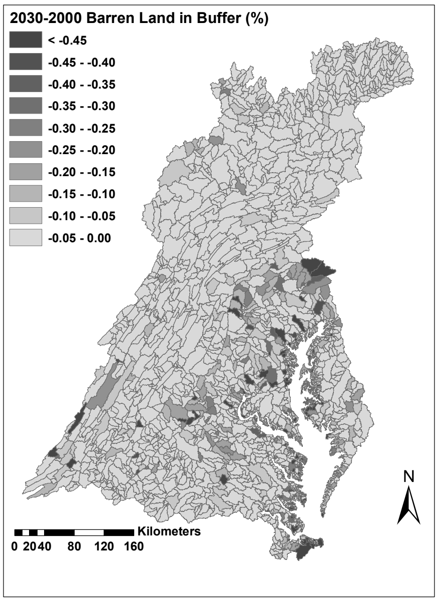

Figure 3.8: Per catchment 2030-2000 difference map of the percentage of barren land in the 31 m riparian stream buffer land-to-water delivery metric for the RESAC TP model.

3.6 Conclusions

Previously, the quantification of the impacts of projected future urbanization on nutrient loading estimates has been limited to smaller watersheds and impacts on regional, national, and even global nutrient loadings have been deduced from the results of local studies. To substantiate the expectations based on the small catchments, substantially larger watersheds regions that drain into large estuaries and coastal oceans need to be examined from the detailed small catchment scale aggregated to the larger catchments in which they occur. The present study of the potential future sources and transport of TN and TP using projections of urbanization in the Chesapeake Bay watershed is an attempt to undertake such an assessment. The effects of LC/LU change from 2000 to 2030 in the Chesapeake Bay watershed, in particular as a result of forecast population increases and consequent increases in urbanization, was modeled and the effects on nutrient loadings to the estuary assessed.

There was an estimated 19% and 20% reduction in overall delivered TN and TP to the Chesapeake Bay. Although substantial increases in development-induced, point source N and P loadings were apparent watershed-wide, the estimated conversion of agricultural lands leading to declines in delivered fertilizer loadings to streams was the primary reason for the overall reductions in TN and TP delivery to the estuary that were simulated to occur from 2000 to 2030. In contrast to the non-point source changes, the projected increases in point source N and P loadings are necessarily imprecise because future improvements in effluent removal technologies, future WWTP locations that could alter their watershed-wide distribution, and possible gains or losses in industrial and commercial sources cannot be predicted. Increases in impervious surfaces associated with urbanization that would otherwise have increased the mean, delivered non-point urban N and P yields for all catchments from 2000 to 2030 were negated due to downstream water attenuation processes decreasing delivered TN and TP to the Chesapeake Bay from all sources.

The relative magnitude of TN and TP contributions by point sources, fertilizers, and other non-point sources to future nutrient loadings depends on the land cover mosiacs of a watershed (Anbumozhi *et al.,* 2005), as well as total area. The results suggest that lowering area-weighted mean patch sizes within catchments of significant non-point LC/LU sources could reduce future TN and TP loadings to the Chesapeake Bay. This is especially true of mean urban patch sizes, where decreases in its total area would be likely to limit its yields by reducing larger impervious surface areas capable of capturing greater quantities of non-point N and P that could be delivered to streams. In addition, limiting urban growth to the replacement of agricultural source lands and other non-urban cover types associated with large land-to-water delivery of non-point N and P to Chesapeake Bay streams would be particularly effective.

Thus, to minimize projected TN and TP loadings to the Chesapeake Bay, the estimated, spatial distribution of catchment and riparian stream buffer-wide LC/LU should be examined and evaluated in the future prior to development. This would allow for the impacts of the compositional and configurational landscape properties demonstrated here to be quantified and taken into account before any potential management actions are considered.

Chapter 4: Effects of future urban sprawl on nitrogen runoff in subwatersheds of Chesapeake Bay[3]

4.1 Abstract

The effects of sprawl on non-point N runoff from the Chesapeake Bay watershed was studied using the **SPA**tially **R**eferenced **R**egressions **O**n **W**atershed Attributes (SPARROW) model, a nutrient runoff model, applied to maps of impervious surface in 2000 and modeled impervious surface in 2030. Projected catchment-wide losses and gains in TN runoff were calculated for 2,339 catchments. Results suggested leapfrog growth that was predicted to occur in smaller, detached patches on crop and pasture will lower catchment total nitrogen (TN) runoff from 2000 to 2030. This was in contrast to gains in TN runoff that were predicted to occur with infill and peripheral growth, predominantly in cover types other than crop and pasture. Furthermore, development in smaller, detached patches would also reduce urban, non-point N runoff in catchments over this time period. These results suggest that the strategic placement of leapfrog development may be capable of reducing future TN runoff to the Chesapeake Bay and in similar watersheds elsewhere.

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[3] The material in Chapter 4 is currently in review. Roberts, A. D., Prince, S. D., Jantz, C. A., and Goetz, S. J., Submitted for publication. Effects of future urban sprawl on nitrogen runoff in subwatersheds of Chesapeake Bay. Ecological Engineering.

4.2 Introduction

Total nitrogen (TN) runoff from the land is the primary causative factor in the formation of eutrophic, hypoxic, and anoxic conditions in estuaries and coastal oceans (Paerl, 2006). Thus, any land cover and land use (LC/LU) change through urbanization, including the form referred to as sprawl, that converts non-urban cover types to development will most certainly alter nitrogen (N) runoff to streams (Tang *et al.*, 2003). According to Gilliam (2002), urban sprawl can be defined as a form of urbanization distinguished by leapfrog patterns of development, commercial strip development, low-density land uses, large expanses of separated single land-uses, automobile dominance, and a minimum of public open space. LC/LU change from non-urban to urban has been shown to increase N runoff significantly (Carpenter *et al.*, 1998). Long-term water quantity is reduced by sprawl through the channeling of rainwater to drains and the reduction in the volume of water that infiltrates, percolates, and recharges soil profiles and aquifers, although water quality only generally declines (Tu *et al.*, 2007).

Urban sprawl can lead to an increase in TN caused by increases in impervious cover that allow for the buildup and consequential washoff of N to sewers from urban non-point sources that include vehicle emissions, lawn fertilizers, and pet wastes (Babbitt, 2007). Sprawl can also lead to TN gains through increases in point source discharges associated with increased N release from wastewater treatment plants (WWTPs) because of population gains. As the population expands and moves from centralized, higher-density, developed areas to form lower-density, suburban areas, the effects of sprawl may decrease TN if growth is arranged to replace higher yielding (mass for a specified time normalized by drainage area) cover types, such as cropland or pasture. In contrast, sprawl effects may increase TN if growth is configured to replace lower yielding cover types, such as forest or wetlands. Thus, the impacts of the spatial patterns of future sprawl on projected TN loadings (mass for a specified time) in rivers, estuaries, and coastal oceans are of interest.

The correlation between urban sprawl and reduced water quality is not universal. While nitrate (NO_3^-) and TN have been found to be positively correlated to urban and agricultural area in eastern Massachusetts watersheds (Williams *et al.*, 2005; Tu *et al.*, 2007), the Waquoit Bay watersheds surrounding Cape Cod, Massachusetts (Bowen and Valiela, 2001), and several South Carolina coastal streams (Tufford *et al.*, 2003), other dissolved organic nitrogen (DON) species and ammonia (NH_3) were negatively correlated to sprawl in these same eastern Massachusetts and South Carolina coastal stream studies.

The effects of anticipated sprawl on TN in smaller watersheds in Michigan (Tang *et al.,* 2005), Missouri (Wang *et al.,* 2005), Sweden (Jansson and Colding, 2007), and Georgia (Dale *et al.,* 2008) have all indicated an increase in loadings over time. However, it is not clear in large, nested basins if future sprawl could produce losses or gains in projected TN loadings, based upon where growth may occur in the watershed. Although briefly examined by Roberts *et al.*, (2009), this has not been studied exclusively, yet it may have significant management implications.

Here, the Chesapeake Bay watershed, a large, nested basin, was examined to determine which characteristics associated with future sprawl could affect TN loading losses or gains. The Chesapeake Bay is the largest estuarine system in the United States (166,534 km^2 watershed) draining portions of six states (New York (NY), Pennsylvania

(PA), Delaware (DE), Maryland (MD), West Virginia (WV), and Virginia (VA)) and the District of Columbia (DC) that include numerous urban centers (Figure 4.1a-b). The substantial population increase and intensification of agriculture leading to increases in application of chemical fertilizer after the end of World War II has increased TN loadings to the detriment of the overall nutrient health of the estuary (Sims and Coale, 2002). As agricultural non-point runoff, N-based fertilizers can be transported to streams in enriched sediments and DON via surface overland flow, shallow subsurface interflow, and subsurface percolation (Pellerin *et al.,* 2006). Additionally, a significant portion of these TN loadings are generated from impervious cover-based non-point runoff and point discharges from wastewater treatment plants (WWTP), industrial, and commercial sources directly into Chesapeake Bay streams.

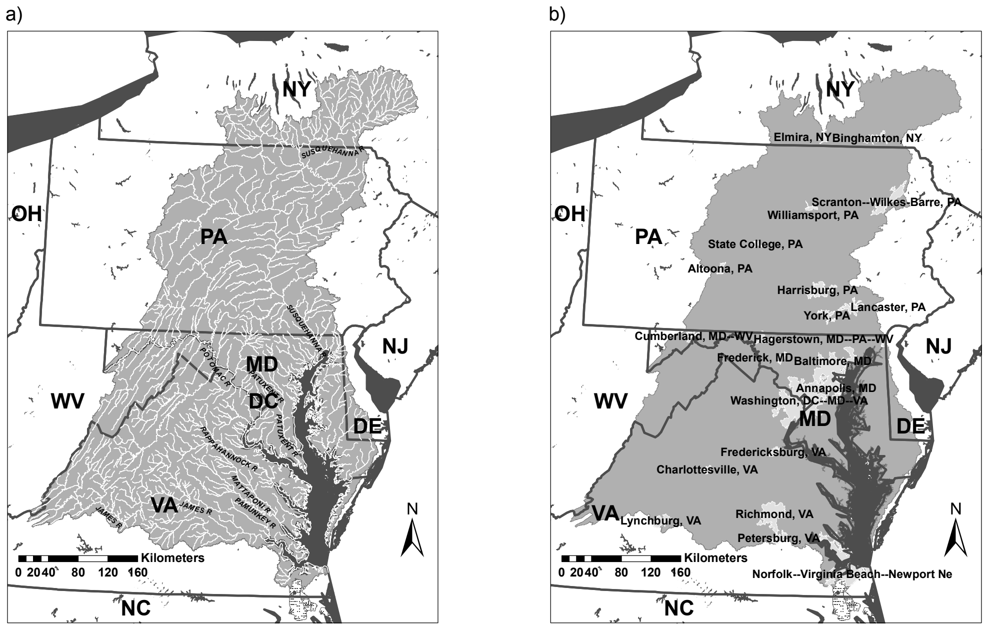


Figure 4.1: The Chesapeake Bay watershed showing the locations of: **a)** Streams and rivers draining the estuary and **b)** Urban centers located within its boundaries.

Since the early 1980s, federal, regional, and state entities have made significant efforts in research and policy initiatives aimed at restoring the estuary to pre-World War II conditions. These include: the United States Environmental Protection Agency (USEPA), Chesapeake Bay Program (CBP), National Oceanic and Atmospheric Administration (NOAA), United States Geological Survey (USGS), states boarding the Chesapeake Bay (MD and VA) and DC, and various other local jurisdictions in the watershed. The estuary is N-limited, with a nutrient input ratio of TN to total phosphorus (TP) well above the Redfield ratio of 16:1 for the normal production of phytoplankton (Weller, 2003). This is caused by an excess of available N rather than deficiencies of TP (Weller, 2003). Thus, a comparison of subcatchments with high and low TN runoff is relevant to understanding and management of TN loadings to the Chesapeake Bay.

Anticipated urban growth to 2030 has shown that expected sprawl will not be spatially-uniform throughout the watershed (Jantz *et al.*, 2004, in press). Simulations of future, urban land cover derived from the **S**lope, **L**and use, **E**xclusion, **U**rban extent, **T**ransportation, and **H**illshade (SLEUTH) urban growth model (Jantz *et al.,* 2004, in press) at 30 m resolution show a continued trend from the last several decades of significant increases in developed land that is detached and distributed further away from existing and new primary urban centers, that is, sprawl. With sprawl predicted to occur across non-urban lands of varying N generation and transport abilities, its spatial arrangement is quite likely to be an important indicator of projected TN.

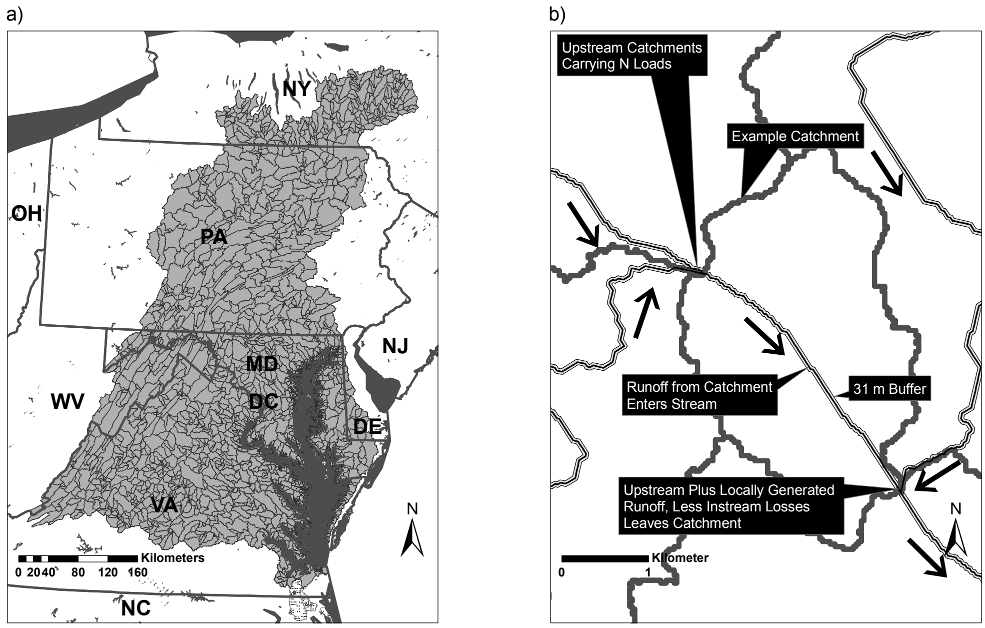


Figure 4.2: 2,339 catchments used in the: **a)** 2000 RESAC 31 m SPARROW Chesapeake Bay TN model with **b)** Schematic diagram of an example catchment area showing fixed 31 m riparian stream buffer width area surrounding stream reach and corresponding upstream and downstream runoff paths.

Using watershed-wide 2000 non-urban and urban cover types derived from the **R**egional **E**arth **S**cience **A**pplication **C**enter's (RESAC) LC/LU and percent impervious surface area (% ISA) maps (Goetz *et al.*, 2004a, b; Jantz *et al.*, 2005), Roberts and Prince (2010) modeled current TN loadings to the Chesapeake Bay. The model was a recalibrated version of the 1997 USGS Version 3.0 **SPA**tially **R**eferenced **R**egressions **O**n **W**atershed **A**ttributes (SPARROW) model (Brakebill and Preston, 2004). 2,339 catchments draining to the estuary (Figure 4.2a) were modeled. Unlike the Brakebill and Preston (B & P) model, the RESAC version of SPARROW added compositional and configurational landscape metrics. In addition, metrics were applied within a 31 m riparian stream buffer (Figure 4.2b). These metrics were shown to have significant effects on non-point sources and land-to-water delivery variables affecting TN loadings to the estuary (Table 4.1). The recalibrated model was used to estimate the effects of the anticipated expansion and its spatial distribution of development from 2000 to 2030.

Landscape metric variables	Effect	
	Source	Land-to-water delivery
Area-weighted mean urban (≥ 10% ISA) patch size	X	
Percentage of cropland		X
Percentage of extractive land		X
Area-weighted mean edge contrast of deciduous forest		X
Percentage of evergreen forest In the riparian stream buffer		X

Table 4.1: Landscape metrics that were found to be significant (p value ≤ 0.05) in the 2000 RESAC 31 meter SPARROW model.

Of the 5 landscape metrics tested by Roberts and Prince (2010), area-weighted mean urban (≥ 10% ISA) patch size was found to have the strongest relationship to TN runoff from 2000 to 2030. The term ≥ 10% ISA described watershed-wide urbanization as any LANDSAT pixel with the proportion of its 30 m x 30 m (900 m^2) area greater than or equal to 10% impervious surface area in 2000 (Goetz *et al.*, 2004a, b; Jantz *et al.*, 2005). Projected urban areas by SLEUTH model runs (Jantz *et al.,* in press) were then used as surrogates to represent 2000 urban areas derived from this ≥ 10% ISA threshold for 2030 estimates. The area-weighted mean urban (≥ 10% ISA) patch size metric measures one aspect of landscape composition and quantifies the sum, across all patches of urban, of patch area multiplied by proportional abundance of the patch. Thus, larger patches exert greater influence than smaller patches. The calibrated 2000 model found that per ha of area-weighted mean urban (≥ 10% ISA) patch size increased non-point N by about 25 kg to the Chesapeake Bay annually (Roberts and Prince, 2010).

By 2030, if growth occurs by creation of smaller patches of development unattached to existing infrastructure or added to smaller patches of urban land, area-weighted mean urban (≥ 10% ISA) patch size will decrease from 2000 values. With potential losses in urban non-point N yields, TN losses would also be possible. However, reductions in TN loadings would depend upon the current LC/LU of non-urban lands to be converted, since they have different impacts on catchment TN runoff.

In contrast, 2030 growth will increase discharged point sources that could cause increases in TN runoff in catchments. If gains in point sources are greater than non-point N losses, overall increases in TN would occur. Therefore, discharges of point sources for 2030 were estimated from projected gains in urban population (Roberts *et al.*, 2009).

Thus, the overall purpose of this study was to compare the characteristics of future sprawling catchments that are projected to have losses with those expected to gain in TN loadings.

4.3 Materials and methods

4.3.1 Categories of sprawling catchments

Between 2000 and 2030, the RESAC 31 m SPARROW model (Roberts and Prince, 2010) indicated that 157 catchments out of 2,339 would decrease in area-weighted mean urban (≥ 10% ISA) patch size (Figure 4.3a). Here, the 157 catchments were divided into those in which the comparison of 2000 with 2030 ISA had an increase with those that had a decrease in locally generated TN yield. There were 106 catchments that were modeled to decreased (Figure 4.3b) and 51 catchments (Figure 4.3c) to increase in locally generated TN yield. In the SPARROW model, local generation of TN indicated the yields and loadings generated from within a catchment independent of any upstream loadings.

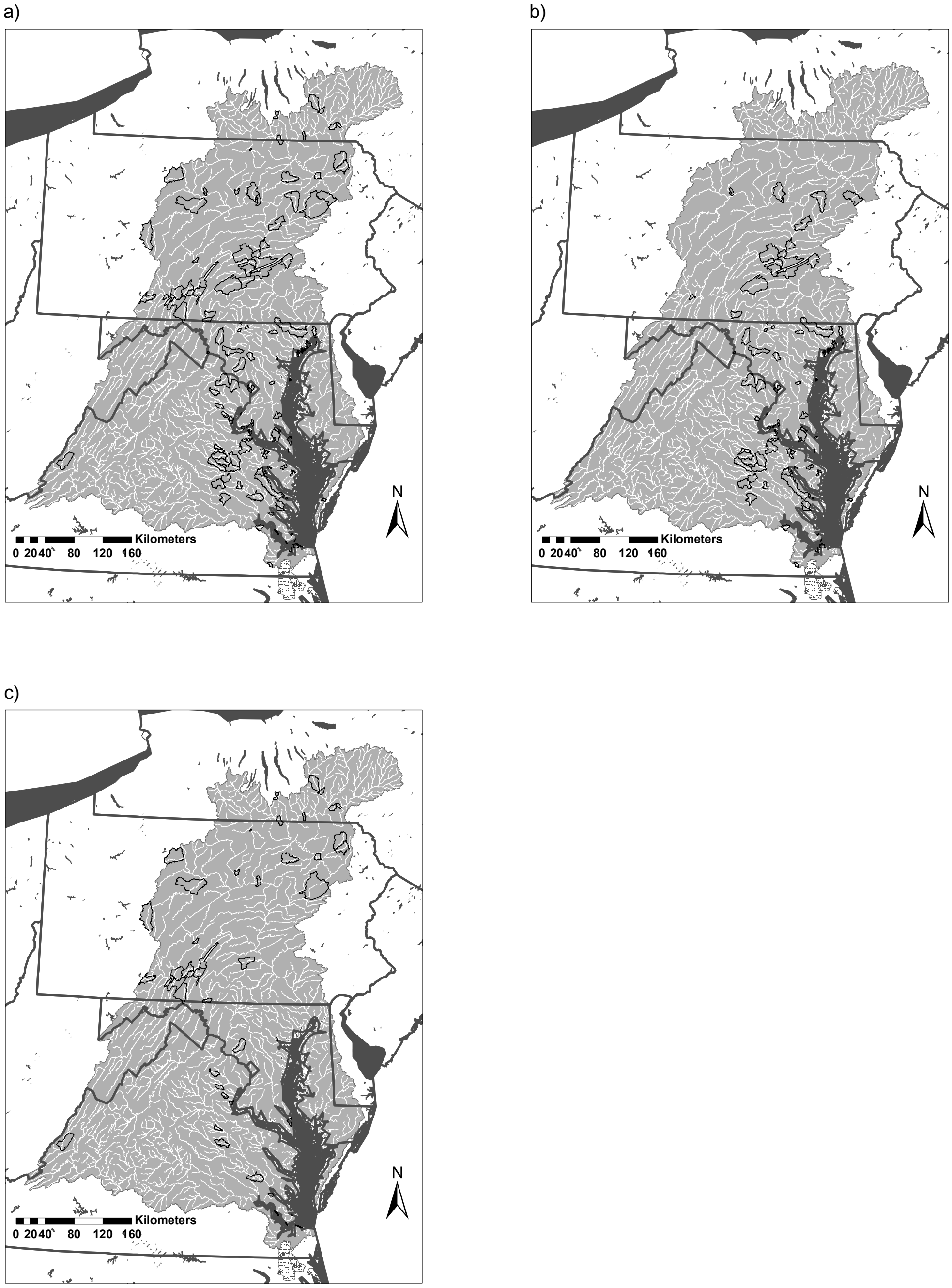


Figure 4.3: Map of the Chesapeake Bay watershed locations of the: **a)** 157 catchments showing decreases in area-weighted mean urban (≥ 10% ISA) patch size with **b)** 106 catchments indicating losses and **c)** 51 catchments indicating gains in locally generated TN yield from 2000 to 2030.

In addition to the area-weighted mean urban (≥ 10% ISA) patch size, ten other variables (Table 4.2) that were significant in the 2000 model (Roberts and Prince, 2010) were analyzed in the two categories of catchments. The only other significant variable that was not evaluated was an increase in atmospheric deposition of N from 2000 to 2030, owing to the difficulty in forecasting a variable that is largely determined by policy, regulation, and legal changes. A complete description of these model variables are given by Roberts and Prince (2010) and Roberts *et al*. (2009).

4.3.2 Results for the entire Chesapeake Bay watershed (Roberts and Prince, 2010)

For each kilogram (kg) of: point sources 1) discharged, 2) fertilizer and 3) manure applied to agricultural land (crop and pasture) and atmospheric N deposited on the watershed, approximately 1.2, 0.2, 0.1, and 0.5 kg of N, respectively, were estimated in Chesapeake Bay streams. For every percent (%) of: 4) land on the coastal plain, non-point N delivered to streams was estimated to decrease by 0.73%. In contrast, for each % of: 5) extractive, 6) cropland, and 7) area-weighted mean edge contrast of deciduous forest, non-point N delivered to streams was estimated to increase by 0.27, 0.021, and 0.014%, respectively. Area-weighted mean edge contrast of deciduous forest was used to measure the contrast between eleven non-urban classes and deciduous forest in the 2000 RESAC LC/LU map. Contrast here was defined as the physical characteristics of differing non-urban cover types influencing N transport. Greatest contrast differences for deciduous forest were with: urban/residential/recreational grasses, extractive, barren, pasture/hay, croplands, and natural grass. For each % of: 8) evergreen forest in the riparian stream buffer, non-point N delivered to streams was estimated to increase by 0.013%. For every meter (m) traveled in small (mean flow ≤ 100 cubic feet per second

(ft^3/s)), intermediate (mean flow >100 and ≤ 500 ft^3/s), and large (mean flow > 500 ft^3/s) streams per day, about 25, 9, and 3% of the instream N was estimated to be lost to processes, such as denitrification, biological uptake, and sedimentation, respectively. Thus, each catchment has only one: 9) mean flow value used to determine the estimated % of N lost based upon the categories just outlined. Finally, for any: 10) reservoir in the watershed, N was removed from the water column at an average settling velocity of over 14 m/yr.

Model component (units)	Variable	Coefficient value
Nitrogen source (^{1} = kg/yr, ^{2} = kg/ha/yr)	Point sources^1	1.173
	Applied fertilizer^1	0.175
	Atmospheric deposition^1	0.492
	Applied manure^1	0.078
	Area-weighted mean urban (≥ 10% ISA) patch size^2	24.885
Landscape delivery (%)	Percentage of coastal plain	-0.729
	Percentage of extractive land	0.270
	Area-weighted mean edge contrast of deciduous forest	0.014
	Percentage of cropland	0.021
	Percentage of evergreen forest in the riparian stream buffer	0.013
Stream decay (m/day)	Small streams	0.249
	Intermediate streams	0.090
	Large Streams	0.030
Reservoir decay (m/yr)	Reservoir	14.224

Table 4.2: Variables in the 2000 RESAC 31 m TN SPARROW model found significant (p value ≤ 0.05).

4.3.3 Yields used for 2030 TN runoff projections

For 2030 estimates, fertilizer rates for cropland and pasture were taken to be 28.02 and 15.83 kg/yr, respectively, and manure at 9.79 and 52.74 kg/yr (Roberts *et al.,* 2009). Furthermore, for each person added to the urban population, it was estimated that an additional 2.72 kg/yr of N would be added to streams from the nearest WWTP servicing these areas in 2000 via increased point discharges (Roberts *et al.,* 2009).

4.3.4 Sprawling catchments and parametric testing

Normality evaluations of: point source discharges; applied fertilizer; applied manure; area-weighted mean urban (≥ 10% ISA) patch size; percentage of land on the coastal plain; percentage of extractive land; area-weighted mean edge contrast of deciduous forest; percentage of cropland; percentage of evergreen forest in the riparian stream buffer; stream size; and reservoirs were conducted using the Shapiro-Wilk, Anderson-Darling, Lillefors, and Jargue-Bera tests (Thode, 2002). After it was determined that all datasets were non-normal and that the mean differences in these variables could not be examined using parametric tests, median differences were evaluated with the non-parametric Mann-Whitney U test (Kvam and Vidakovic, 2007). Only significant (p value ≤ 0.05) median differences between each of the variables in the two categories of catchments are reported.

4.4 Results

4.4.1 Categories of sprawling catchments

In catchments with projected TN yield losses from 2000 to 2030, the median reduction was estimated at -1.10 kg/ha/yr. In the model, the highest decreases in projected TN yield (> 1.8 kg/ha/yr) were predicted to occur near: DC; Baltimore and Frederick (MD); Harrisburg, (PA); and Richmond (VA) (Figure 4. 4a, c). In these catchments, the median decrease in the extent of area-weighted mean urban (≥ 10% ISA) patch size was -2.89 ha. The largest decreases in area-weighted mean urban (≥ 10% ISA) patch size (> 27 ha) were predicted to occur near: DC; Scranton/Wilkes-Barre, Harrisburg, and Williamsport (PA); and Richmond (VA) (Figure 4. 4b, d).

In contrast, in catchments with projected TN yield gains, equivalent values were +0.53 kg/ha/yr and -1.52 ha. The highest increases in projected TN yield (> 1.8 kg/ha/yr) were predicted to occur near: DC; Hagerstown (MD); and Scranton/Wilkes-Barre, Harrisburg, and Williamsport (PA) (Figure 4.5a, c). The largest decreases in area-weighted mean urban (≥ 10% ISA) patch size (> 27 ha) will be located in the same regions (Figure 4.5b, d).

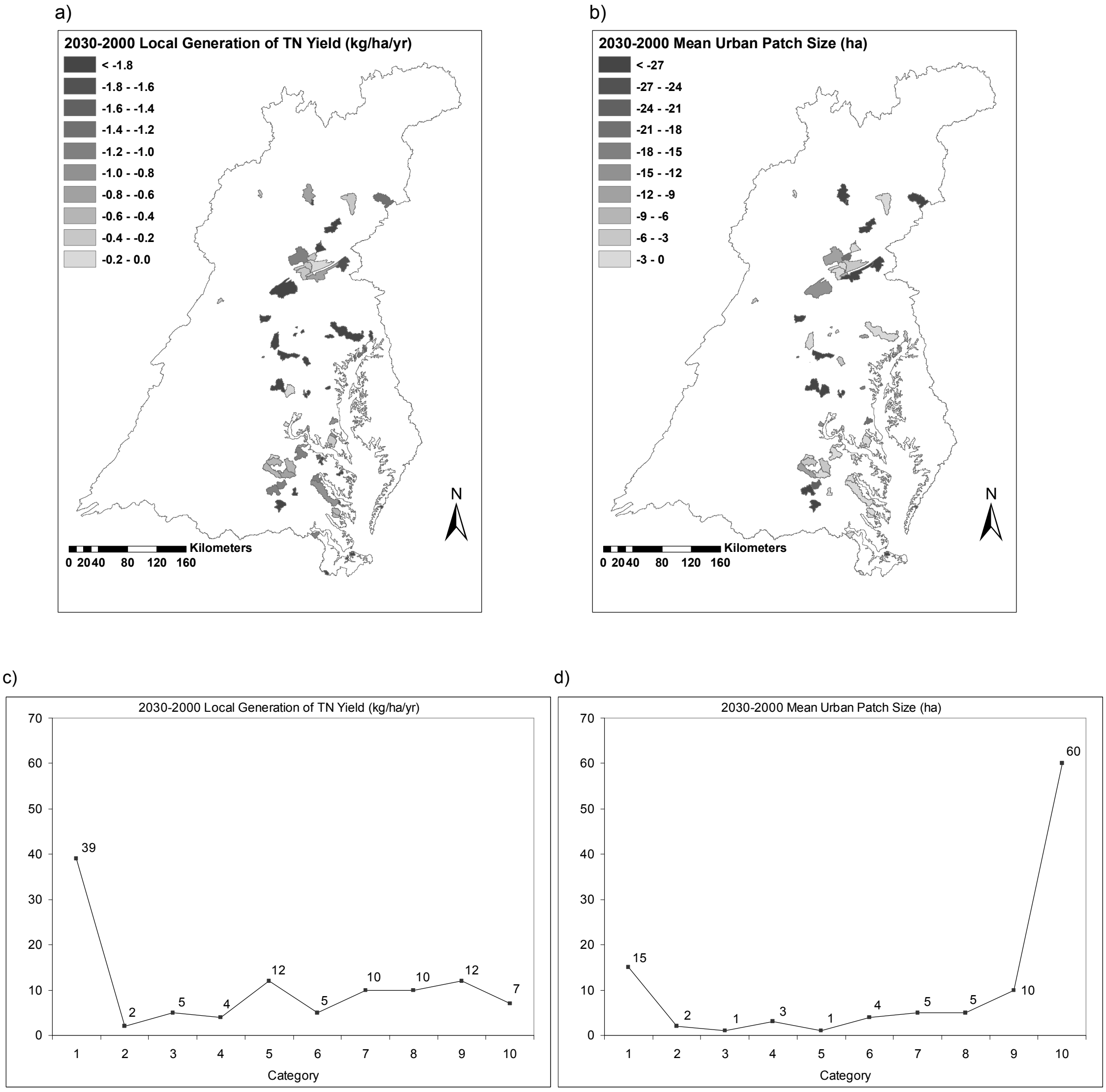


Figure 4.4: Per catchment 2030-2000 change maps of: **a)** Losses in locally generated TN yield (kg/ha/yr) with **b)** Decreases in the area-weighted mean urban (≥ 10% ISA) patch size (ha), and **c-d)** Corresponding frequency distributions of the number of catchments falling in each category. For **Figure 4.4c**, category classes are: 1 = < -1.8, 2 = -1.8 - -1.6, 3 = -1.6 - -1.4, 4 = -1.4 - -1.2, 5 = -1.2 - -1.0, 6 = -1.0 - -0.8, 7 = -0.8 - -0.6, 8 = -0.6 - -0.4, 9 = -0.4 - -0.2, and 10 = -0.2 - 0.0 kg/ha/yr. For **Figure 4d**, category classes are 1 = < -27, 2 = -27 - -24, 3 = -24 - -21, 4 = -21 - -18, 5 = -18 - -15, 6 = -15 - -12, 7 = -12 - -9, 8 = -9 - -6, 9 = -6 - -3, and 10 = -3 - 0 ha.

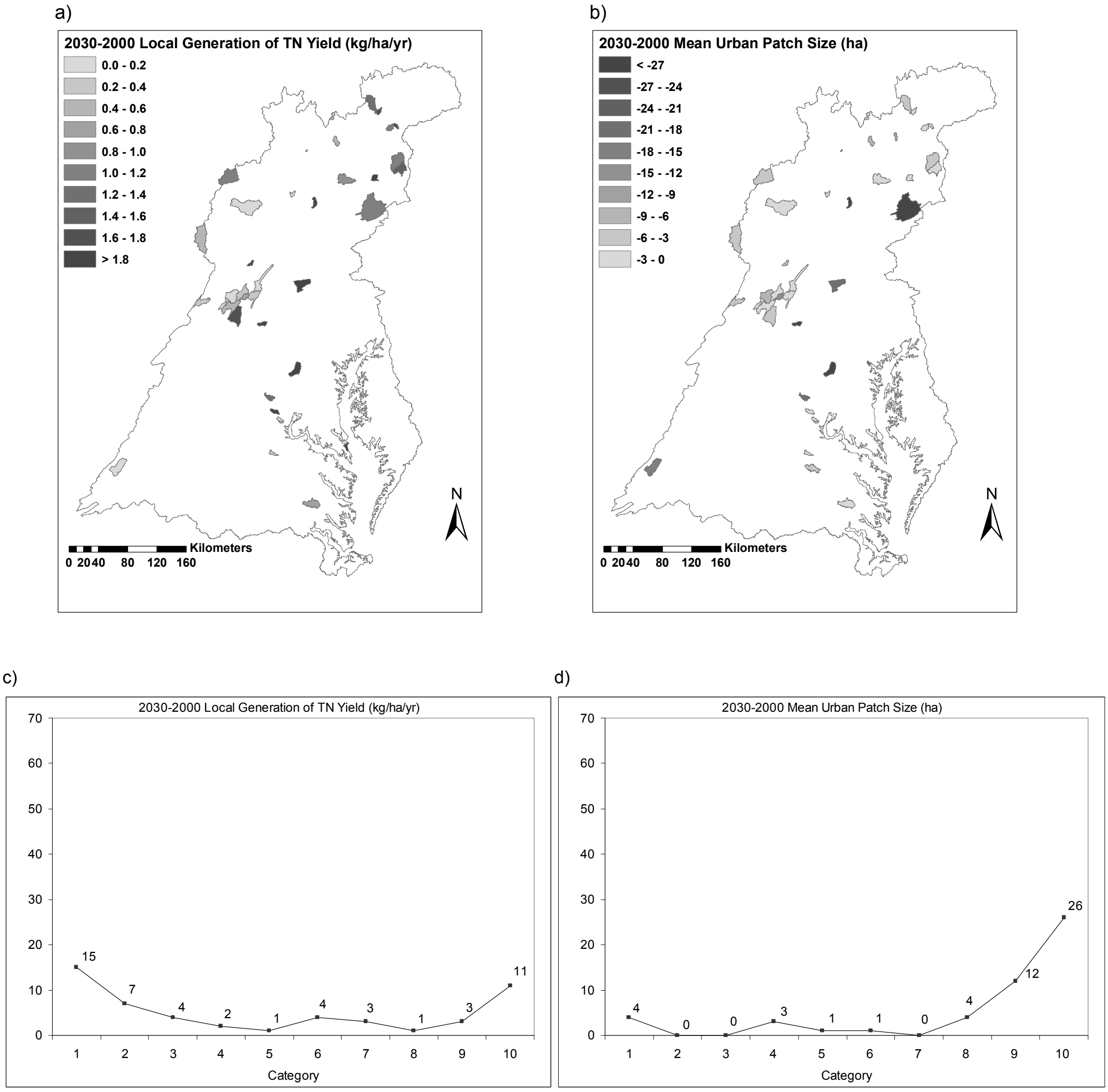


Figure 4.5: Per catchment 2030-2000 change maps of: **a)** Gains in locally generated TN yield (kg/ha/yr) with **b)** Decreases in the area-weighted mean urban (≥ 10% ISA) patch size (ha) and **c-d)** Corresponding frequency distributions of the number of catchments falling in each category. For **Figure 4.5c,** category classes are: 1 = 0.0 -0.2, 2 = 0.2 -0.4, 3 = 0.4 - 0.6, 4 = 0.6 -0.8, 5 = 0.8 - 1.0, 6 = 1.0 - 1.2, 7 = 1.2 - 1.4, 8 = 1.4 - 1.6, 9 = 1.6 - 1.8, and 10 = > 1.8 kg/ha/yr. For **Figure 4.5d**, category classes are 1 = < -27, 2 = -27 - -24, 3 = -24 - -21, 4 = -21 - -18, 5 = -18 - -15, 6 = -15 - -12, 7 = -12 - -9, 8 = -9 - -6, 9 = -6 - -3, and 10 = -3 - 0 ha.

4.4.2 Sprawling catchments and SPARROW

Of the eleven variables tested, five were found to be significant (Table 4.3). These were: area-weighted mean urban (≥ 10% ISA) patch size, applied fertilizer, applied manure, percentage of coastal plain, and percentage of cropland.

Significant SPARROW variables	Catchments with losses in local generation of TN yield	Catchments with gains in local generation of TN yield
Area-weighted mean urban (≥ 10% ISA) patch size	-0.03 kg/ha/yr	-0.01 kg/ha/yr
Applied fertilizer	-37.48 kg/ha/yr	-2.28 kg/ha/yr
* Percentage of land on coastal plain	87%	0%
Percentage of cropland	-0.72%	-0.25%
Applied manure	+28.23 kg/ha/yr	+31.70 kg/ha/yr

Table 4.3: Comparison of the significant (p value ≤ 0.05) median changes from 2000 to 2030 for the RESAC 31 m SPARROW variables in the catchments with losses and gains in locally generated TN yield. * Denotes static hydrogeomorphic province variable that does not vary from 2000 to 2030.

The median area-weighted mean urban (≥ 10% ISA) patch size N yield loss was estimated to be -0.03 kg/ha/yr in catchments with projected TN yield losses (Table 4.3). In contrast, the median area-weighted mean urban (≥ 10% ISA) patch size yield loss was estimated to be only -0.01 kg/ha/yr in catchments with projected gains (Table 4.3). In TN yield loss catchments, applied fertilizer yield decreased substantially (-37.48 kg/ha/yr), whereas applied manure yield increased by + 28.23 kg/ha/yr (Table 4.3). For catchments with TN yield gains, applied fertilizer decreased only -2.28 kg/ha/yr, in contrast to the sharp increase in applied manure yield of +31.70 kg/yr (Table 4.3). The results of the changes in locally generated applied fertilizer and manure yield in the two categories of catchments are given in Table 4.4.

Local generation	Catchments with losses in local generation of TN yield	Catchments with gains in local generation of TN yield
Applied fertilizer Applied manure	-5.96 kg/ha/yr +1.73 kg/ha/yr	-0.30 kg/ha/yr +2.90 kg/ha/yr

Table 4.4: Comparison of the median changes in locally generated applied fertilizer and manure yield (kg/ha/yr) from 2000 to 2030 in catchments with losses and gains in locally generated TN yield.

In catchments with projected TN yield losses, the median percentage on the coastal plain was 87%, as compared to 0% in catchments with gains (Table 4. 3). The median percentage of cropland decreased by -0.72% in catchments with projected losses, but decreased by only -0.25% in catchments with gains (Table 4.3). Thus, the conversion of cropland to urban was nearly 3:1 in catchments with projected losses.

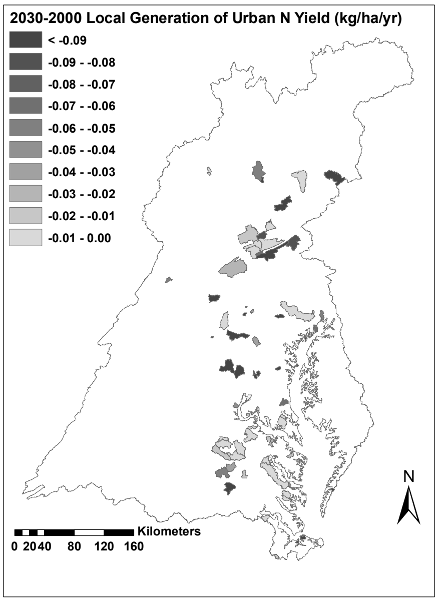
4.5 Discussion

4.5.1 Area-weighted mean urban (≥ 10% ISA) patch size

The finding of a greater (-0.03 kg/ha/yr) yield loss attributed to urban patch size in catchments with projected TN losses is interesting since this could be indicative of variations in urban non-point N losses dependent upon where growth will occur. Catchments with projected TN yield losses are predicted to have smaller patches of growth on currently undeveloped land that is unattached to existing urban areas. This type of sprawl is referred to as "leapfrog" growth. With the introduction of isolated patches, urban non-point N loadings may fall catchment-wide in spite of an increase in development, since these patches represent smaller areas and runoff volumes. Additionally, smaller patches may help to disperse concentrated, urban non-point sources, such as lawn fertilizer, pet waste, vehicles, etc; among more stream networks leading to attenuation of loadings. Lawn fertilizer rates can be even higher and more concentrated than rates applied to some crops (Bowen and Valiela, 2001). The highest decreases in area-weighted mean urban (≥ 10% ISA) patch size yield (> 0.09 kg/ha/yr) in catchments with projected TN yield losses were predicted to occur near: DC; Annapolis, Baltimore, Frederick, and Hagerstown (MD); Scranton/Wilkes-Barre and Harrisburg (PA); and Richmond (VA) (Figure 4.6a-b).

In contrast, the finding of patch size related yield loss (-0.01 kg/ha/yr) in catchments projecting TN gains is interesting since it suggests that the spatial pattern of future sprawl will occur around smaller areas of existing urban land. This type of sprawl is referred to as "infill and peripheral" growth. Although this type of growth in catchments with TN gains still led to a median decrease in patch size, there would be smaller declines in urban non-point N runoff. In contrast to leapfrog growth, higher concentrations of non-point N would reach these streams. The largest decreases in area-weighted mean urban (≥ 10% ISA) patch size yield (> 0.09 kg/ha/yr) in catchments with projected TN yield gains were predicted to occur near: DC; Hagerstown (MD); and Harrisburg and Williamsport (PA) (Figure 4.7a-b).

a)



b)

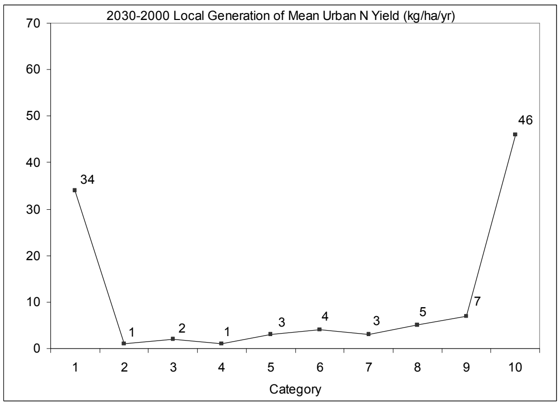
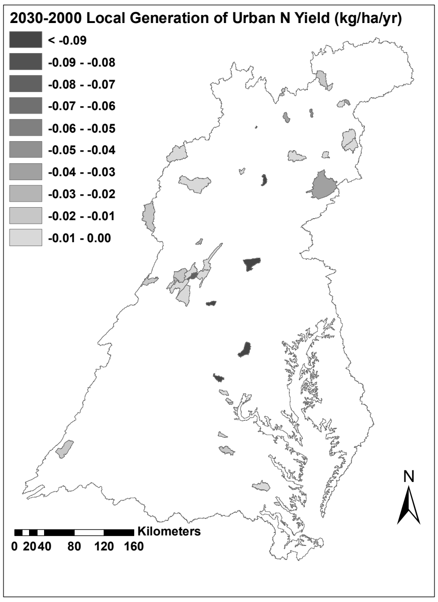


Figure 4.6: Per catchment 2030-2000 change map of: **a)** Losses in locally generated area-weighted mean urban (≥ 10% ISA) patch size yield (kg/ha/yr) with losses in locally generated TN yield and **b)** Corresponding frequency distribution of the number of catchments falling in each category. Category classes are: 1 = < -0.09, 2 = -0.09 - -0.08, 3 = -0.08 - -0.07, 4 = -0.07 - -0.06, 5 = -0.06 - -0.05, 6 = -0.05 - -0.04, 7 = -0.04 - -0.03, 8 = -0.03 - -0.02, 9 = -0.02 - -0.01, and 10 = -0.01 - 0.00 kg/ha/yr.

a)



b)

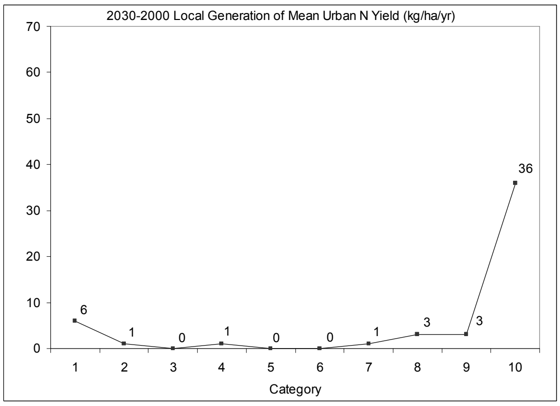


Figure 4.7: Per catchment 2030-2000 change map of: **a)** Losses in locally generated area-weighted mean urban (≥ 10% ISA) patch size yield (kg/ha/yr) with gains in locally generated TN yield and **b)** Corresponding frequency distribution of the number of catchments falling in each category. Category classes are: 1 = < -0.09, 2 = -0.09 - -0.08, 3 = -0.08 - -0.07, 4 = -0.07 - -0.06, 5 = -0.06 - -0.05, 6 = -0.05 - -0.04, 7 = -0.04 - -0.03, 8 = -0.03 - -0.02, 9 = -0.02 - -0.01, and 10 = -0.01 - 0.00 kg/ha/yr.

4.5.2 Applied fertilizer and manure applications

In catchments with projected TN yield losses, the net application in yield of fertilizer and manure was -9.25 kg/ha/yr less (the projected -37.48 kg/ha/yr decrease in applied fertilizer yield added to the projected +28.23 kg/ha/yr increase in applied manure yield (Table 4. 3)), thus non-point source N fell. Overall, lower quantities of fertilizer and manure were predicted to be transported to streams, as seen by the net loss of -4.23 kg/ha/yr (the projected -5.96 kg/ha/yr decrease in locally generated fertilizer yield added to the projected +1.73 kg/ha/yr increase in locally generated manure yield (Table 4.4)) from these sources. The findings also suggest that because future sprawl will occur in these catchments, possibly as leapfrog growth on cropland and pasture patches, overall TN decreases. The highest decreases in applied fertilizer yield (> 30 kg/ha/yr) were predicted to occur near: Baltimore and Frederick (MD) and Richmond and Norfolk-Virginia Beach-Newport News (VA) (Figure 4.8a, c). The largest decreases in locally generated fertilizer yield (> 30 kg/ha/yr) were predicted to occur only in small catchments near Baltimore (MD) (Figure 4.8b, d). The highest increases in applied manure yield (> 26 kg/ha/yr) were predicted to occur throughout the watershed (Figure 4.9a, c). However, the largest increases in locally generated manure yield (> 26 kg/ha/yr) were also predicted to occur only in small catchments near Baltimore (MD) (Figure 4.9b, d).

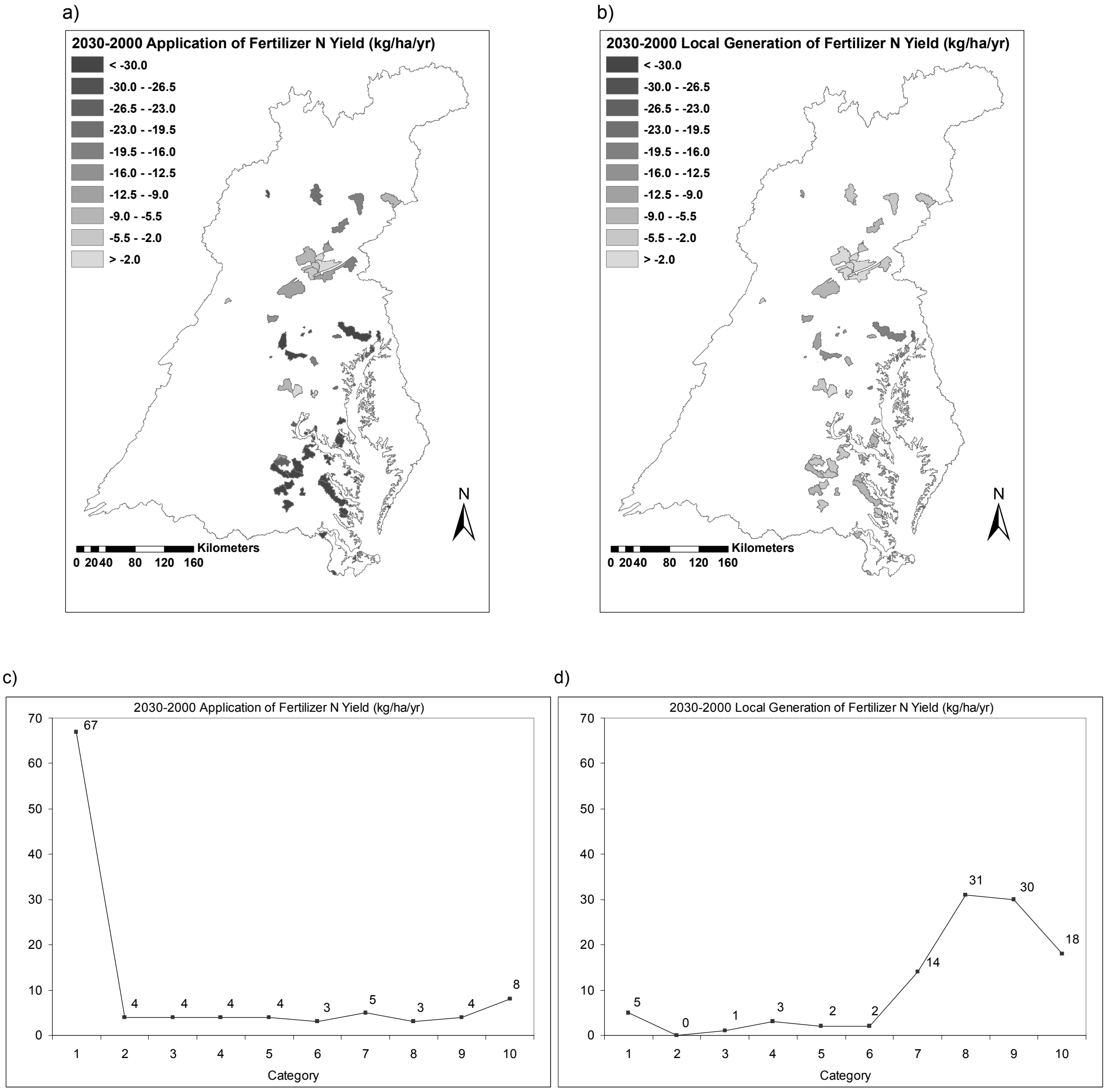


Figure 4.8: Per catchment 2030-2000 change maps of: **a)** Application and **b)** Local generation of fertilizer N yield (kg/ha/yr) with losses in locally generated TN yield and **c-d)** Corresponding frequency distributions of the number of catchments falling in each category. Category classes are: 1 = < -30.0, 2 = -30.0 - -26.5, 3 = -26.5 - -23.0, 4 = -23.0 - -19.5, 5 = -19.5 - -16.0, 6 = -16.0 - -12.5, 7 = -12.5 - 9.0, 8 = -9.0 - -5.5, 9 = -5.5 - -2.0, and 10 = > -2.0 kg/ha/yr.

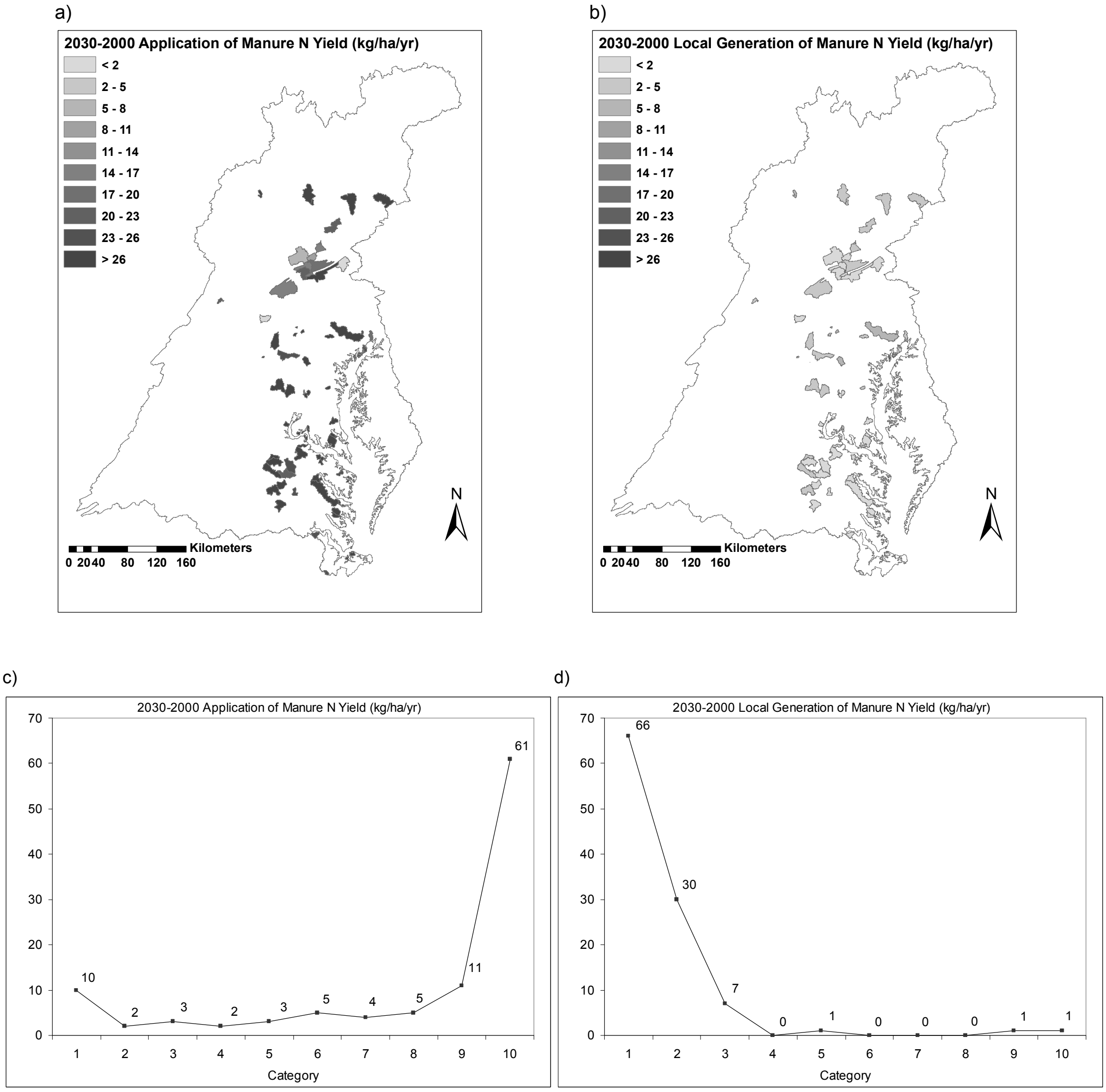


Figure 4.9: Per catchment 2030-2000 change maps of: **a)** Application and **b)** Local generation of manure N yield (kg/ha/yr) with losses in locally generated TN yield and **c-d)** Corresponding frequency distributions of the number of catchments falling in each category. Category classes are: 1 = < 2, 2 = 2 - 5, 3 = 5 - 8, 4 = 8 - 11, 5 = 11 - 14, 6 = 14 - 17, 7 = 17 - 20, 8 = 20 - 23, 9 = 23 - 26, and 10 = > 26 kg/ha/yr.

In contrast, in catchments projecting TN yield gains, the net application of fertilizer and manure yield was higher by +29.42 kg/ha/yr (the projected -2.28 kg/ha/yr decrease in applied fertilizer yield added to the projected +31.70 kg/ha/yr increase in applied manure yield (Table 4.3)). This was seen by the net gain in median yield within streams of +2.60 kg/ha/yr (the projected -0.30 kg/ha/yr decrease in locally generated fertilizer yield added to the projected +2.90 kg/ha/yr increase in locally generated manure yield (Table 4.4)) attributed to these sources. Future sprawl here suggests that infill and peripheral growth will occur where agricultural land conversions will be less likely. This will be especially important in the case of pasture and its higher rate of manure applications by 2030. The highest decreases in applied fertilizer yield (> 4.8 kg/ha/yr) were predicted to occur near: DC; Hagerstown (MD); Harrisburg, Williamsport, State College, and Altoona (PA); and Richmond (VA) (Figure 4.10a,c). The largest decreases in locally generated fertilizer yield (> 4.8 kg/ha/yr) were predicted to occur near these same regions (Figure 4.10b,d). The highest increases in applied manure yield (> 4.0 kg/ha/yr) will be scattered throughout the watershed (Figure 4.11a,c). The largest increases in locally generated manure yield (> 4.0 kg/ha/yr) will be near: DC and Scranton/Wilkes-Barre, Williamsport, State College, and Altoona (PA) (Figure 4. 11b,d).

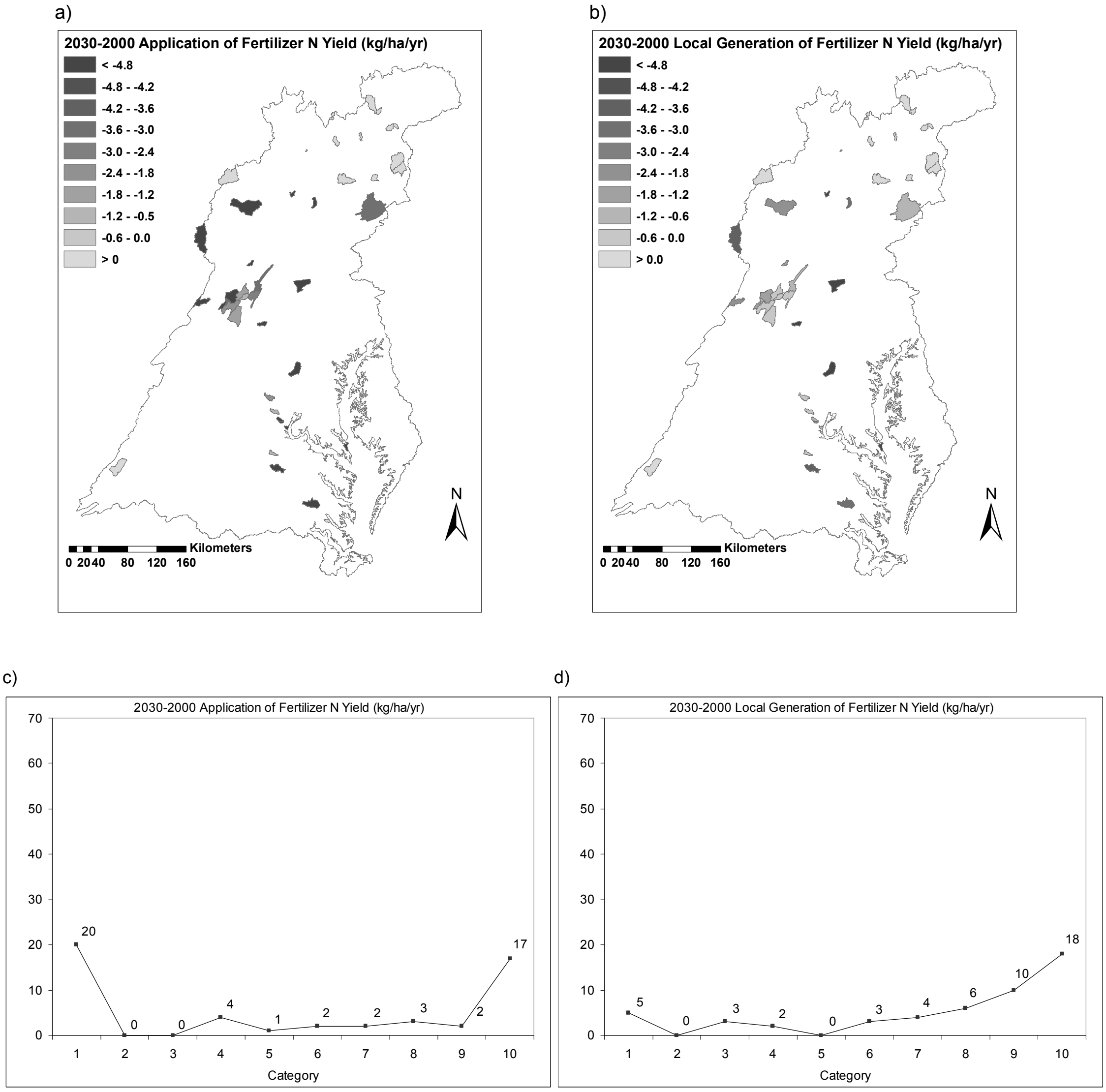


Figure 4.10: Per catchment 2030-2000 change maps of: **a)** Application and **b)** Local generation of fertilizer N yield (kg/ha/yr) with gains in locally generated TN yield and **c-d)** Corresponding frequency distributions of the number of catchments falling in each category. Category classes are: 1 = < -4.8, 2 = -4.8 - -4.2, 3 = -4.2 - -3.6, 4 = -3.6 - -3.0, 5 = -3.0 - -2.4, 6 = -2.4 - -1.8, 7 = -1.8 - 1.2, 8 = -1.2 - -0.6, 9 = -0.6 - -0.0, and 10 = > 0.0 kg/ha/yr.

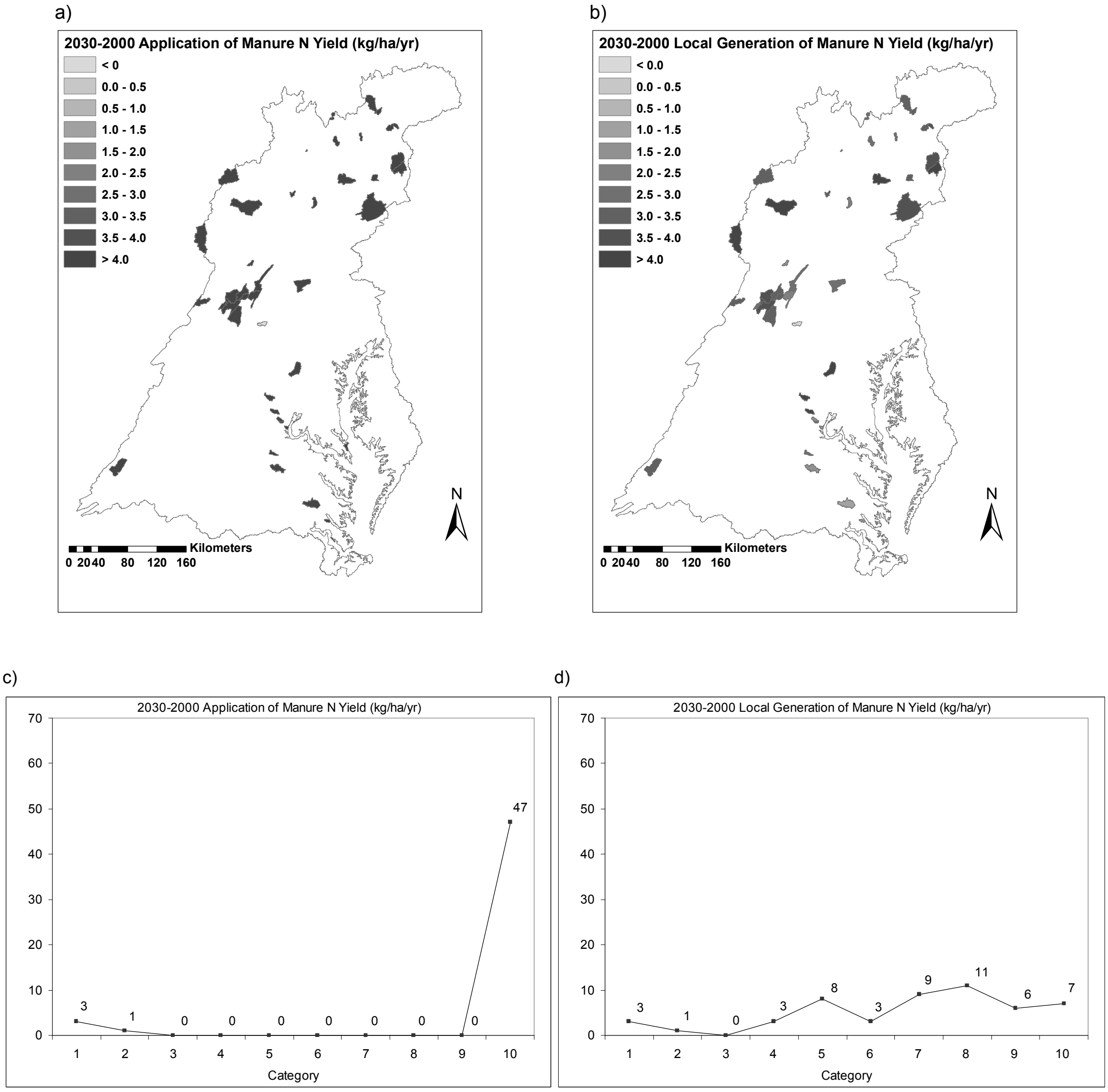


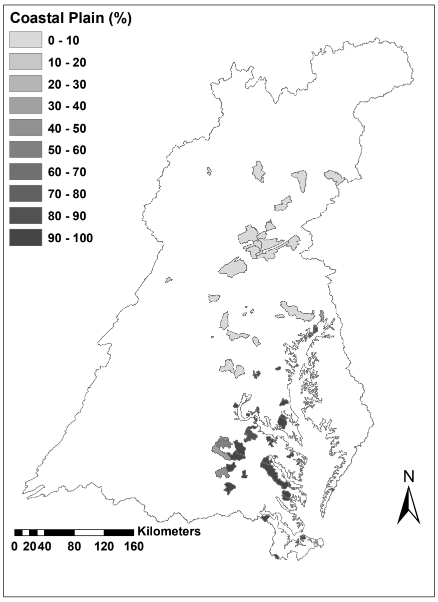
Figure 4.11: Per catchment 2030-2000 change maps of: **a)** Application and **b)** Local generation of manure N yield (kg/ha/yr) with gains in locally generated TN yield and **c-d)** Corresponding frequency distributions of the number of catchments falling in each category. Category classes are: 1 = < 0.0, 2 = 0.0 - 0.5, 3 = 0.5 - 1.0, 4 = 1.0 - 1.5, 5 = 1.5 - 2.0, 6 = 2.0 - 2.5, 7 = 2.5 - 3.0, 8 = 3.0 - 3.5, 9 = 3.5 - 4.0, and 10 = > 4.0 kg/ha/yr.

4.5.3 Percentage of land on coastal plain

Non-point N transport to Chesapeake Bay streams was less the more land on the coastal plain. This could be a result of DON (NO_3^-) transported in surface overland flow and shallow subsurface interflow being able to infiltrate deeper into soil profiles (Preston and Brakebill, 1999). At these deeper subsurface depths, bacterial-induced denitrification processes may occur under reduced (anoxic) conditions when in the presence of riparian vegetation (Jordan *et al.*, 1997; Willems *et al.*, 1997; Cornwell *et al.*, 1999; Hayashi and Rosenberry, 2002). Increased infiltration and percolation has been linked with greater hydraulic conductivity correlated to higher macropores found in coastal plain soils of the Chesapeake Bay watershed (Willems *et al.*, 1997; Wynn *et al.*, 2000).

The finding that catchments with a loss in TN consisted of more coastal plain suggested that leapfrog growth will occur primarily in areas not directly benefiting from these larger regions of N attenuation. Thus, infill and peripheral growth will occur predominantly in areas where larger regions of potential non-point attenuation would have been more advantageous. The highest percentages of land on the coastal plain (90-100%) in catchments with projected losses in TN are near: DC; Baltimore and Annapolis (MD); and Richmond and Norfolk-Virginia Beach-Newport News (VA) (Figure 4.12a-b), as compared to only small catchments near DC and Baltimore (MD) in those with projected gains in TN (Figure 4.13a-b).

a)



b)

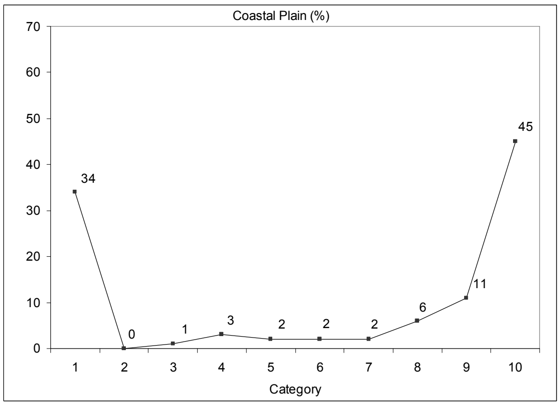
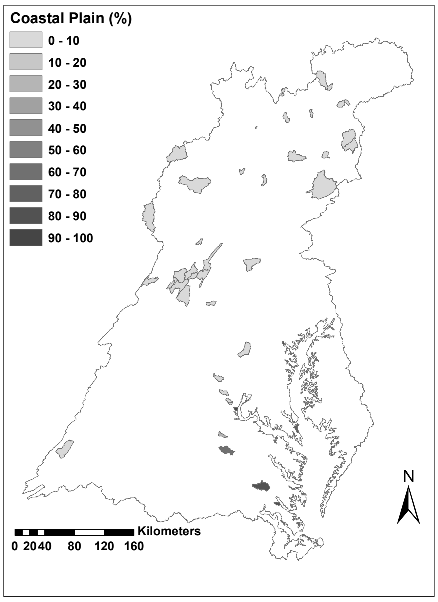


Figure 4.12: Per catchment map of: **a)** Percentage of land on the coastal plain (%) with losses in locally generated TN yield and **b)** Corresponding frequency distribution of the number of catchments falling in each category. Category classes are: 1 = 0 - 10, 2 = 10 - 20, 3 = 20 - 30, 4 = 30 - 40, 5 = 40 - 50, 6 = 50 - 60, 7 = 60 - 70, 8 = 70 - 80, 9 = 80 - 90, and 10 = 90 - 100%.

a)



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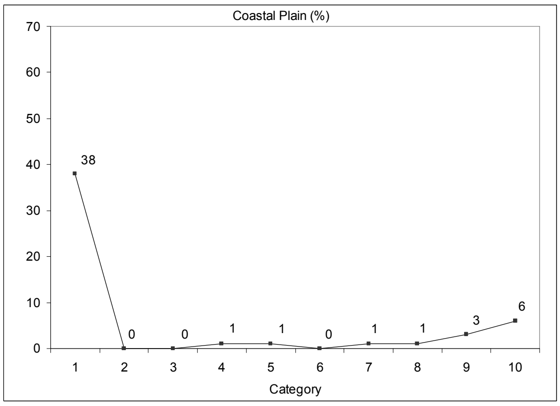


Figure 4.13: Per catchment map of: **a)** Percentage of land on the coastal plain (%) with gains in locally generated TN yield and **b)** Corresponding frequency distribution of the number of catchments falling in each category. Category classes are: 1 = 0 - 10, 2 = 10 - 20, 3 = 20 - 30, 4 = 30 - 40, 5 = 40 - 50, 6 = 50 - 60, 7 = 60 - 70, 8 = 70 - 80, 9 = 80 - 90, and 10 = 90 - 100%.

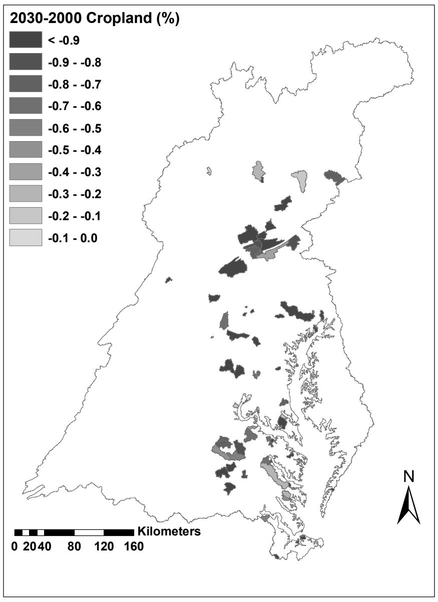
4.5.4 Percentage of Cropland

In catchments with projected TN losses, the higher amounts of cropland that would be converted to urban would also lower the quantities of fertilizer and manure being applied to the surface and thus, available for potential non-point leaching and transport. It was estimated that cropland increased non-point N transport to streams (Roberts and Prince, 2010). This has been attributed to decreases in hydraulic conductivity from cultivation practices, such as tillage and machinery field compaction, thus promoting surface sealing and overland runoff (Logsdon and Jaynes, 1996).

Catchments with TN losses also represented a transformation from cropland and its higher N impacts to urban uses with lower N impacts. In another study of future sprawl in the watershed's Patuxent Basin (Costanza *et al.*, 2002), it was concluded that there is a considerable gain in water quality by the conversion of agricultural land to residential uses. In catchments with projected TN losses, the highest decreases in percentage of cropland (> 0.9%) were predicted to occur scattered throughout the watershed (Figure 4.14a-b).

In contrast, in catchments with projected gains in TN, it was suggested that even with reduced, urban non-point impacts, the remaining agricultural non-point effects would be sufficient to increase overall loadings. This was based in part on the lower conversion of cropland that would occur under infill and peripheral growth conditions by 2030. In catchments with projected gains, the highest decreases in percentage of cropland (> 0.9 %) were also predicted to occur scattered throughout the watershed-wide (Figure 4.15a-b).

a)



b)

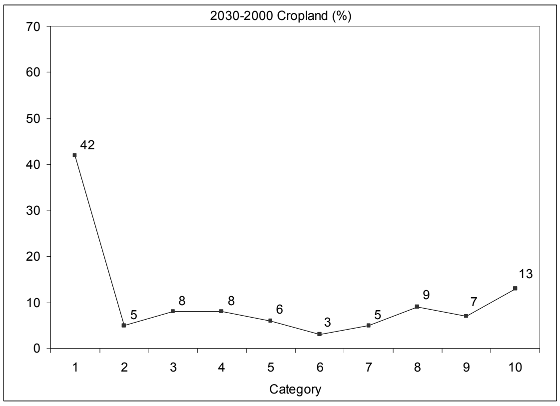
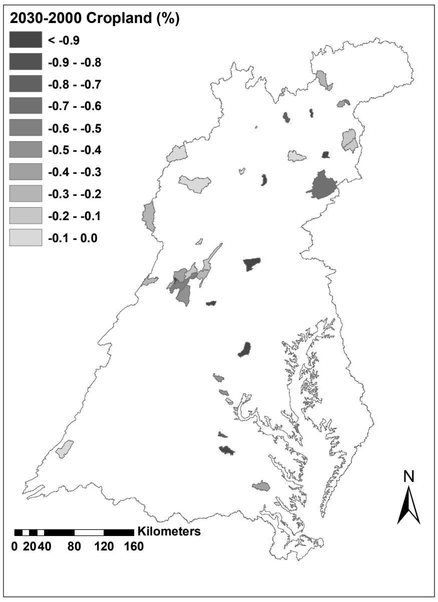


Figure 4.14: Per catchment 2030-2000 change map of: **a)** Decreases in percentage of cropland (%) with losses in locally generated TN yield and **b)** Corresponding frequency distribution of the number of catchments falling in each category. Category classes are: 1 = < -0.9, 2 = -0.9 - -0.8, 3 = -0.8 - -0.7, 4 = -0.7 - -0.6, 5 = -0.6 - -0.5, 6 = -0.5 - -0.4, 7 = -0.4 - -0.3, 8 = -0.3 - -0.2, 9 = -0.2 - -0.1, and 10 = -0.1 - 0.0 %.

a)



b)

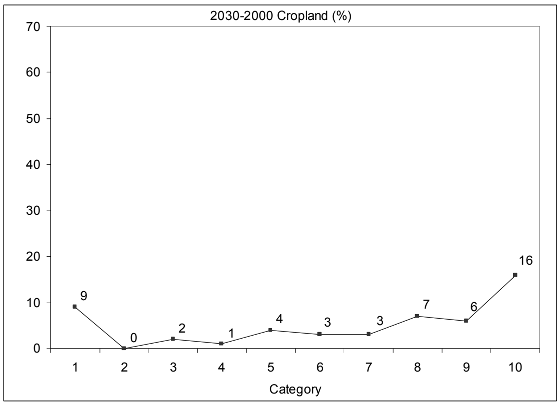


Figure 4.15: Per catchment 2030-2000 change map of: **a)** Decreases in percentage of cropland (%) with gains in locally generated TN yield and **b)** Corresponding frequency distribution of the number of catchments falling in each category. Category classes are: 1 = < -0.9, 2 = -0.9 - -0.8, 3 = -0.8 - -0.7, 4 = -0.7 - -0.6, 5 = -0.6 - -0.5, 6 = -0.5 - -0.4, 7 = -0.4 - -0.3, 8 = -0.3 - -0.2, 9 = -0.2 - -0.1, and 10 = -0.1 - 0.0 %.

4.6 Conclusions

To decrease projected TN yields in sprawling catchments, it is suggested that whenever possible, management should confine urbanization to smaller patches in cropland and pasture. This leapfrog growth could reduce the non-point agricultural N effects from fertilizer and manure applied to these cover types. This type of sprawl aids in the transformation of land to urban uses with an overall lower N impact. Additionally, leapfrog growth may lead to higher attenuation of urban non-point loadings by dispersing these concentrated sources over a greater number of stream networks.

Although only five of the eleven SPARROW variables had significant changes or differences between the catchment categories, all variables should be examined. On a catchment-by-catchment basis, other variables may help explain projected TN losses or gains. An example of this is point sources that may cause increases in catchments with projected TN gains. This scenario was realized in a study of anticipated sprawl in Sweden that concluded point sources were the cause of the overall gain in future TN to the Baltic Sea (Jansson and Colding, 2007). Also, Bowen and Valiela (2001) concluded that a factor of 17 increase in delivered point sources was a primary cause for the substantial, overall TN gain to estuaries near Cape Cod, Massachusetts from 1938 to 1990.

In conclusion, the results of this study further confirmed our earlier demonstration that landscape pattern has a significant effect on TN runoff. Also, it suggested management practices and policies that could be enacted for sustainable future growth. It was concluded that significant rates of urbanization can occur, while still improving the overall nutrient health of the Chesapeake Bay, findings that could be beneficial to watersheds projected to undergoing similar, developmental, transformations elsewhere.

Chapter 5: Lessons learned from current and projected future urban land cover in Chesapeake Bay watershed[4]

5.1 Abstract

This overview summarizes findings from recent studies of the Chesapeake Bay that combined remotely sensed, land cover and land use (LC/LU) data within a watershed-wide, nutrient loading (mass for a specified time) model to quantify effects of current and projected future urban land cover on total nitrogen (TN) and total phosphorus (TP) runoff. Important findings included: 1) slightly improved accuracy and precision of current estimated nutrient loadings to the Chesapeake Bay when using landscape configuration and riparian stream buffers, 2) overall nutrient loadings to the Chesapeake Bay decreased due to agricultural land losses and reductions in fertilizers after projected urbanization, and 3) the importance of spatial arrangement of forecasted urban sprawl to projected TN response within catchments of the watershed. Using these and other key findings, it is concluded that several management and policy recommendations could be implemented in the watershed to help mitigate nutrient-related issues in the Chesapeake Bay. Furthermore, if climate change can be evaluated in addition to this landscape analysis, a more comprehensive understanding of the anthropogenic impacts on projected watershed-wide nutrient loadings can potentially be quantified.

[4] The material in Chapter 5 is currently in review. Roberts, A. D., Submitted for publication. Lessons learned from current and projected urban land cover in Chesapeake Bay watershed. Environmental Science and Policy.

5.2 Introduction

Since the introduction of European settlement into the Chesapeake Bay watershed in the 17th and 18th centuries, land cover and land use (LC/LU) changes in the form of deforestation that created agricultural and urban cover resulted in steadily declining water quality (Boesch, 2006). Although initially detrimental to the overall functioning of this ecosystem, widespread degradation of the Chesapeake Bay did not become apparent until after World War II when rapid urbanization and population growth and the intense use of commercial fertilizers watershed-wide significantly increased the genesis and delivery of nutrients to the Chesapeake Bay (Sims and Coale, 2002). These excess nutrients in the form of total nitrogen (TN) and total phosphorus (TP) have contributed greatly to problems of: increased sedimentation, turbidity (water opaqueness), hypoxia (low levels of dissolved oxygen (DO)), anoxia (devoid of DO), a lowering in submerged aquatic flora and fauna populations, and a lowering in the overall sustainability of the watershed and Chesapeake Bay itself (Breitburg 1992; Bratton *et al.*, 2003; Hassett *et al.,* 2005).

In addition to being the largest estuary in the United States (U.S.), the Chesapeake Bay also has the largest ratio of land-to-water volume of any estuary in the world, making runoff from the surface extremely critical to the overall health of this ecosystem (Shuyler *et al.*, 1995). The 166,534 km^2 watershed drains portions of six states: (New York (NY), Pennsylvania (PA), Delaware (DE), Maryland (MD), West Virginia (WV) and Virginia (VA)) and the District of Columbia (DC) from six major basins (the James, Patuxent, Potomac, Rappahannock, Susquehanna, and York) and other smaller rivers and streams at an estimated freshwater inflow rate of 8.7 x 10^6 m^3/h (Figure 5.1a) (Sims and Coale, 2002). The watershed also contains numerous large, intermediate, and small urban centers that have been previously correlated to the estuary's overall trophic status (Figure 5.1b) (Preston and Brakebill, 1999; Brakebill *et al.,* 2001; Brakebill and Preston, 2004).

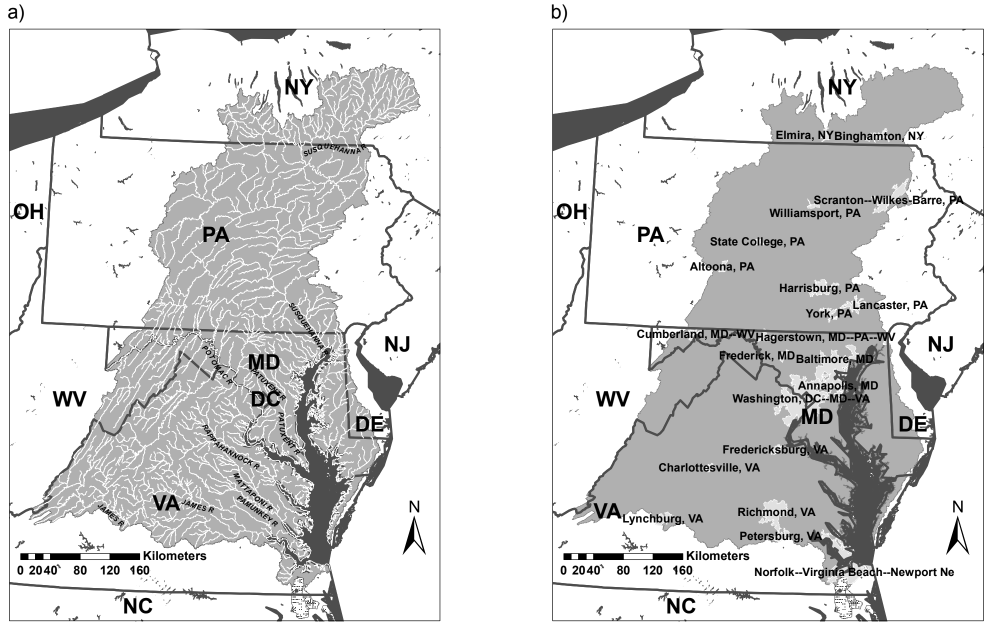


Figure 5.1: The Chesapeake Bay watershed showing the locations of: **a)** Streams and rivers draining the estuary and **b)** Urban centers located within its boundaries.

To comprehensively examine current (2000) urban and non-urban land cover in the watershed, the **R**egional **E**arth **S**cience **A**pplications **C**enter (RESAC) used LANDSAT data to map LC/LU, percent impervious surface area (% ISA), and percent tree cover (% TC) (Goetz *et al.*, 2003, 2004a, b; Jantz *et al.*, 2005). Additionally, spatially-explicit forecasts of the 2030 locations of watershed-wide development with measures of uncertainty were provided by the **S**lope, **L**and use, **E**xclusion, **U**rban extent, **T**ransportation, **H**illshade (SLEUTH) model, a cellular automata model that represented urban growth and expansion processes in a two-dimensional grid (Jantz *et al.*, in press).

Using these land cover datasets, current (2000) and projected (2030) TN and TP loadings (mass for a specified time) to the Chesapeake Bay were simulated using the **SPA**tially **R**eferenced **R**egressions **O**n **W**atershed **A**ttributes (SPARROW) model (Roberts *et al.*, 2009; Roberts and Prince, 2010). SPARROW estimates these nutrient loadings and yields (mass for a specified time normalized for drainage area) from spatially-referenced watershed reaches, catchments, and networks using point and non-point source, land-to-water delivery, and stream and reservoir decay variables (Schwarz *et al.,* 2006). For these models, 2,339 catchments draining to the estuary were simulated with a 31 m riparian stream buffer (Figure 5.2a-b) using significant (p value ≤ 0.05) compositional (variety and abundance) and configurational (orientation and position) metrics of land cover (Roberts *et al.*, 2009; Roberts and Prince, 2010). Furthermore, other land cover-based variables (such as point sources and applied fertilizer and manure) were estimated to 2030 in an effort to more accurately portray overall projected loadings.

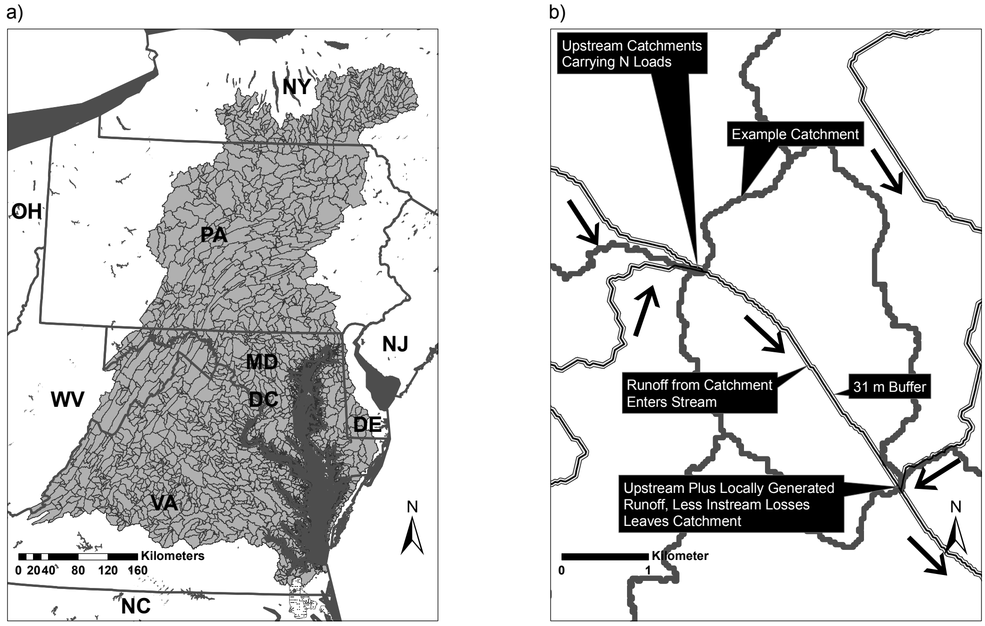


Figure 5.2: 2,339 catchments used in the 2000 and 2030 runs of the: **a)** RESAC 31 m Chesapeake Bay SPARROW models with **b)** Schematic diagram of an example catchment area showing fixed 31 m riparian stream buffer width area surrounding stream reach and corresponding upstream and downstream runoff paths.

The multistate effort to restore the Chesapeake Bay ecosystem to pre-colonial times by reducing the inputs of nutrients leading to enrichment is one of the world's most ambitious attempts at large-scale ecosystem restoration (Boesch *et al*., 2001). A prime example of this was demonstrated with the enactment of the Chesapeake Bay Agreement in 1987 that endeavored to reduce controllable 1985 nutrient levels by 40% by 2000 (Chesapeake Executive Council, 1987). In addition, new 2000 amendments called for the removal of all nutrient-related issues impairing the Chesapeake Bay and its removal from the list of impaired waters under the Clean Water Acts (Wang *et al*., 2006).

As a result of these efforts, it is concluded that using current and projected watershed-wide urban land cover to quantify comprehensive landscape effects on nutrient loadings to the Chesapeake Bay is an important management and research priority that may help local, state, and federal stakeholders meet these reduction mandates. This is due to the findings summarized here that hint towards new correlations between the terrestrial and aquatic environments that may lead to new approaches to the management and restoration of nutrient-enriched ecosystems. Findings that not only address these correlations at the regional scale of the Chesapeake Bay watershed, but may also generate greater insight into mechanisms and processes controlling nutrient-related, biogeochemical cycling at local and global scales.

Thus, using these findings, several priority research and management questions are addressed in this overview and include:

- **What LC/LUs at catchment and riparian stream buffer scales are significant in the current (2000) non-point (diffuse) sources and delivery of TN and TP loadings estimated in the Chesapeake Bay watershed?**
- **What are the causes of the observed effects of LC/LUs at catchment and riparian stream buffer scales on the current (2000) non-point (diffuse) sources and delivery of TN and TP loadings estimated in the Chesapeake Bay watershed?**
- **What specific types of LC/LUs lost to future urbanization will have the greatest impacts on projected TN and TP loadings estimated to the Chesapeake Bay?**
- **What changes in TN and TP loadings will occur in the Chesapeake Bay between 2000 and the future (2030) under the urbanization scenario applied here?**

Additional findings and management and policy recommendations developed from these summarized results are also included within this overview. Finally, this overview commences with a discussion of directions for future research.

5.3 Significance of findings for priority management questions

- **What LC/LUs at catchment and riparian stream buffer scales are significant in the current (2000) non-point (diffuse) sources and delivery of TN and TP loadings estimated in the Chesapeake Bay watershed?**

At the entire Chesapeake catchment scale, the LC/LU sources of non-point TN to the estuary were mostly related to agriculture (cropland and pasture) with the exception of urban (Roberts and Prince, 2010). In regards to TP loadings, all LC/LUs (anthropogenic and non-anthropogenic) were non-point sources to the Chesapeake Bay at the catchment scale (Roberts and Prince, 2010). The significant cover types for TP were urban, non-agricultural/non-urban, and agricultural.

For each hectare (ha) of urban (≥ 10% ISA) that weighted each patch of this LC/LU by its size relative to the total area of this LC/LU per catchment (area-weighted mean urban (≥ 10% ISA) patch size), nearly 25 and 1 kilograms per year (kg/yr) of N and P were estimated in Chesapeake Bay streams (Roberts and Prince, 2010). In terms of TN and TP loadings from agriculture, the application of fertilizers and manure was the main cause of the high loadings. For each kg of fertilizer and manure N applied to agricultural lands, approximately 0.2 and 0.1 kg/yr of N were estimated in streams (Roberts and Prince, 2010). Each kg of fertilizer and manure P applied to these lands yielded an estimated 0.02 and 0.01 kg/yr of P in streams (Roberts and Prince, 2010).

In the case of TP, non-agricultural/non-urban lands that consisted primarily of forests were a major source of P. This was due to large area-weighted non-agricultural/non-urban patch sizes being distributed watershed-wide, even though delivery in yield from this land cover was quite low (Figure 5.3a-b). For every ha of this patch size, about 0.1 kg/yr of P was estimated to streams (Roberts and Prince, 2010).

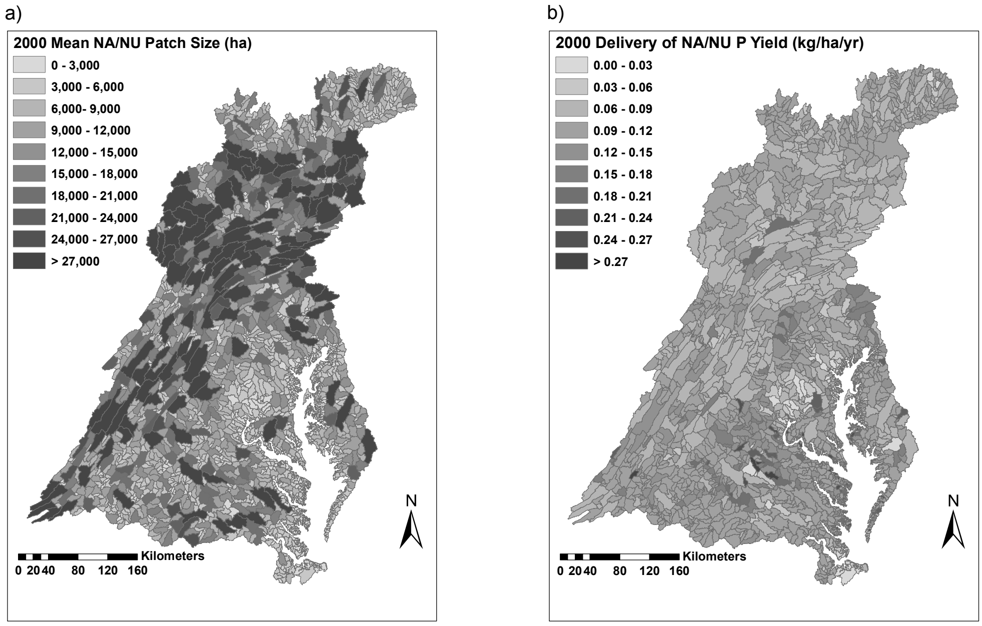


Figure 5.3: Per catchment source factor metric of: **a)** Area-weighted mean non-agricultural/non-urban (NA/NU) patch size in ha with delivery of **b)** P yield to Chesapeake Bay in kg/ha/yr in the 2000 RESAC 31 m TP model.

Significant LC/LUs estimated to contribute to the delivery of non-point N to the Chesapeake Bay included: extractive land and cropland at the catchment scale and evergreen forest isolated to the 31 m riparian stream buffer (Roberts and Prince, 2010). For each percent of extractive, cropland, and evergreen forest at these scales, N delivered to streams from anon-point sources was estimated to increase by 0.27%, 0.021%, and

0.013%, respectively. Barren land isolated to the 31 m riparian stream buffer was the only significant LC/LU correlated to P transport by increasing delivery of non-point P to streams by 0.281% for each percent of its composition (Roberts and Prince, 2010). For both nutrients, this result was the first demonstration of the significance of riparian stream buffer LC/LU on transport at the scale of the entire watershed. With the exception of cropland, an interesting finding was that all of these LC/LUs are ones that are not commonly associated with the non-point source delivery of N and P to the estuary. Additionally, these significant land-to-water delivery cover types occurred throughout the watershed in smaller percentages (Table 5.1) (Goetz *et al.,* 2004a, b; Jantz *et al.,* 2005).

Significant land-to-water delivery LC/LUs	2000 watershed-wide composition (%)
Extractive land	0.21
Cropland	8.18
Evergreen forest within 31 m riparian stream buffer	0.06
Barren land within 31 m riparian stream buffer	0.003

Table 5.1: 2000 watershed-wide compositions of significant land-to-water delivery LC/LUs.

Extractive land was located predominantly near central PA, western MD, and northeastern WV (Figure 5.4a). Cropland was clustered near southern Pennsylvania, the eastern shore of MD, and DE (Figure 5.4b). Although scattered throughout the watershed, the highest concentrations of evergreen forest within the 31 m riparian stream buffer were found in central PA, south central NY, and central and western VA (Figure 5.4c). Finally, barren land within the 31 m riparian stream buffer was predominantly clustered near central VA and southeastern and south central PA (Figure 5.4d). Thus, at the regional scale of the entire Chesapeake Bay, the high impact LC/LUs varied in location.

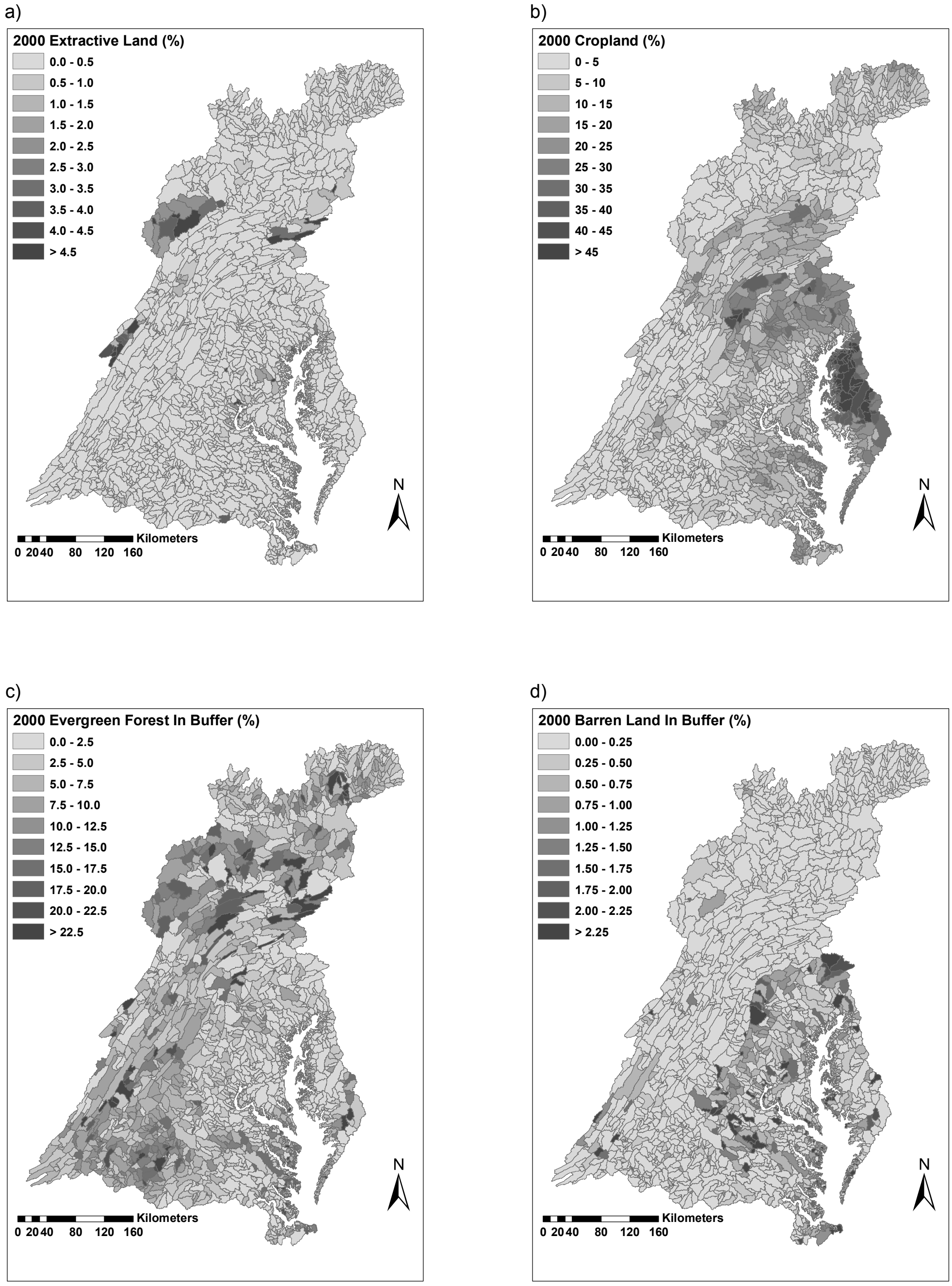


Figure 5.4: Per catchment land-to-water delivery metrics of: **a)** Extractive land (%), **b)** Cropland (%), and **c)** Evergreen forest within the riparian stream buffer (%) in the 2000 RESAC 31 m TN model and **d)** Barren land within the riparian stream buffer (%) in the 2000 RESAC 31 m TP model.

Finally, another LC/LU correlated to the non-point transport of N was catchment-wide area-weighted mean edge contrast of deciduous forest (Roberts and Prince, 2010). However, unlike the other significant delivery cover types, this variable was based upon landscape configuration. Deciduous forest fragmentation influencing nutrient runoff was quantified by either adjacencies to other non-urban, natural LC/LUs (such as mixed forest and wetland) or non-urban, anthropogenic-altered LC/LUs (such as extractive, cropland, and pasture). For each percent gain in this variable (meaning increased overall adjacencies to non urban, anthropogenic-altered LC/LUs at the catchment scale), delivery of non-point N to Chesapeake Bay streams was estimated to increase by 0.014% (Roberts and Prince, 2010).

- **What are the causes of the observed effects of LC/LUs at catchment and riparian stream buffer scales on the current (2000) non-point (diffuse) sources and delivery of TN and TP loadings estimated in the Chesapeake Bay watershed?**

It was surmised that developed (urban) and agricultural LC/LUs yielded significant N and P loadings, either through urban sources, such as vehicle emissions, lawn fertilizers, and animal wastes or through agricultural sources, such as fertilizer and manure applications (Roberts and Prince, 2010). The nutrient sources were actively delivered to streams after precipitation and irrigation events. During these sources of runoff water, urban non-point N and P were delivered to streams via developed patches connected to sewers, culverts and stormwater drains (Figure 5.5a-c) (Roberts and Prince, 2010). Agricultural non-point N and P were delivered to streams via non-urban LC/LUs that were previously mentioned and capable of transport. For both LC/LU sources, the concentrated buildup of non-point N and P occurred primarily as a result of the impervious components within these cover types. The penetration of N and P into deeper soil profiles was prevented by paving (urban), certain tillage practices, (agriculture) and use of heavy-machinery that compact the surface (agriculture) (Logsdon and Jaynes, 1996), and allowed for a buildup of these nutrients at the surface and shallow-subsurface level. Thus, both LC/LUs were substantial sources of non-point N and P to the estuary.

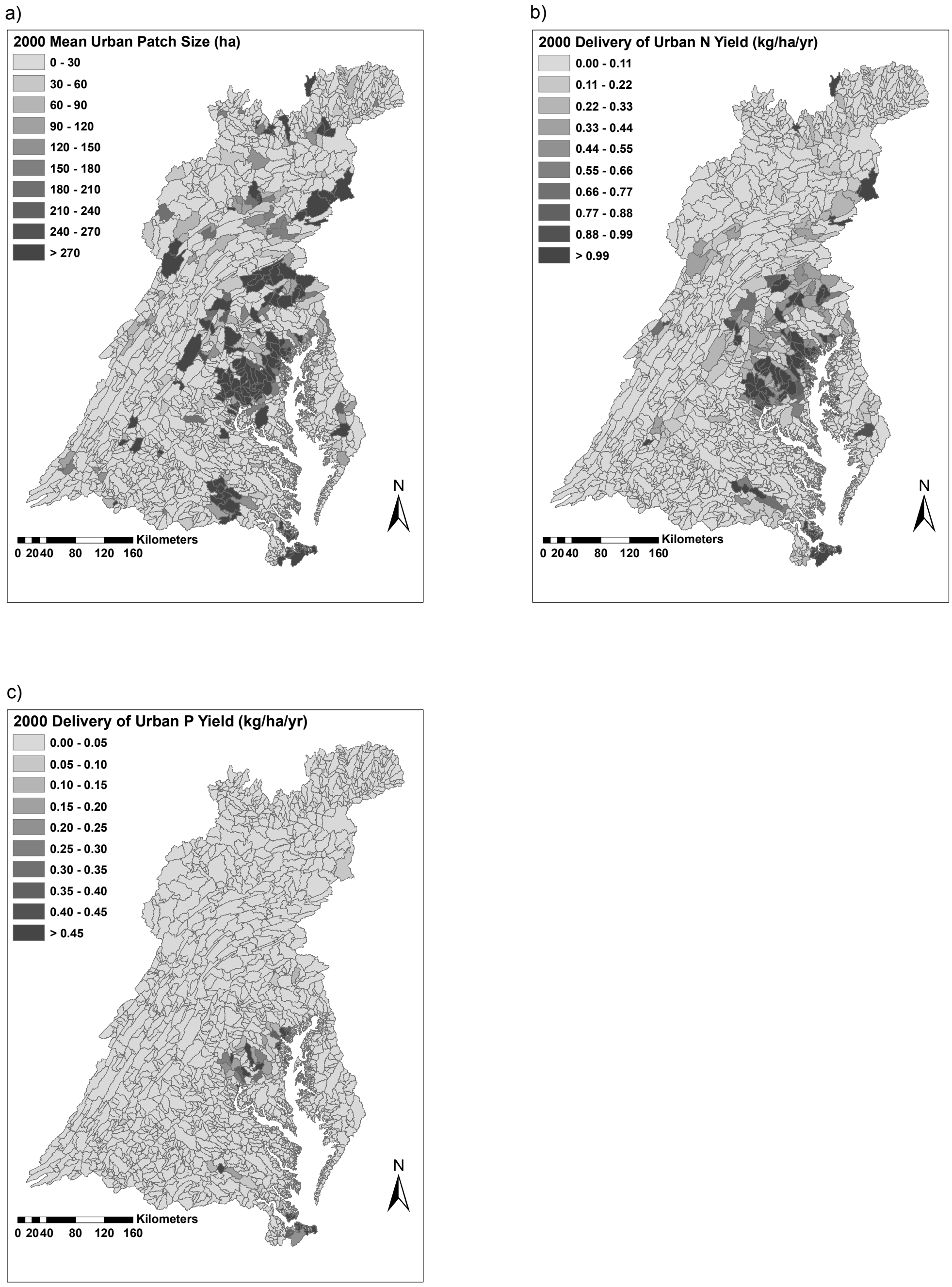


Figure 5.5: Per catchment source factor metrics of: **a)** Area-weighted mean urban (≥ 10% ISA) patch size (ha) with delivery of **b)** N and **c)** P yield to Chesapeake Bay in kg/ha/yr in the 2000 RESAC 31 m models.

In regards to TP, non-agricultural/non-urban LC/LU was the background (natural) source of P in the Chesapeake Bay watershed before modern anthropogenic influences started (Roberts *et al.,* 2009). It was concluded that P from this LC/LU grouping was not the cause of current eutrophication in the Chesapeake Bay and its watershed. This P was generated from the leaching of decomposed organic matter on the forest floor (such as litter fall) and subsurface (such as rooting systems) to groundwater sources. The groundwater sources of dissolved P can subsequently runoff to Chesapeake Bay streams, over temporal scales that can range from months to decades. This naturally-occurring process is responsible for the generation of background P in streams and has been found in other studies of areas similar to those of the Chesapeake Bay watershed (Yanai, 1992).

Finally, extractive land, cropland, barren land, evergreen forests and non-urban, anthropogenic-impacted LC/LUs surrounding deciduous forests were all cover types that can increase non-point N and P delivery to the Chesapeake Bay probably because of their diminished permeability (Roberts and Prince, 2010). In effect, they were non-urban lands that mimicked the impervious properties of urban uses. By the reduction of hydraulic conductivity that would allow for the initial infiltration and subsequent percolation of runoff through soil pores into subsurface groundwater reserves, these cover types encouraged overland and shallow subsurface runoff. Thus, the cover types were capable of transporting dissolved and sediment-enriched nutrients to the estuary. Other studies have noted this effect, especially in regards to barren lands (Ziegler and Giambelluca, 1997, 1998; Ziegler *et al.,* 2001, 2004; Assouline and Mualem, 2002; Perkins *et al.,* 2007; Lopez *et al.,* 2008). The occurrence of this effect at the catchment and riparian stream buffer scale throughout the entire Chesapeake Bay watershed was a significant finding. Furthermore, forested riparian buffers previously thought to attenuate non-point N delivery at local scales (Correll *et al*., 1992; Jordan *et al*., 1993; Boesch *et al*., 2001) were found here to increase non-point N delivery at the entire watershed scale.

- **What specific types of LC/LUs lost to future urbanization will have the greatest impacts on projected TN and TP loadings estimated to the Chesapeake Bay?**

The LC/LU types that had the greatest impacts on projected nutrient loadings estimated to the estuary were cropland and pasture (Roberts *et al.,* 2009). This resulted from their higher nutrient production, as opposed to those of developed uses. The conversion of agricultural land to urban cover estimated to occur in the lower Susquehanna Basin had the greatest proportional impact on projected TN and TP loadings, since the river provides nearly half of the freshwater volume entering the Chesapeake Bay (Figure 5.6a). Thus, any conversion here would be the key to reducing nutrient-related issues in the estuary and its watershed. In the southeastern PA portion of the Susquehanna alone, several catchments were forecasted to have > 9% of their existing cropland converted to urban by 2030 (Figure 5.6b). Catchments that had the highest delivered TN (> 18 kg/ha/yr) and TP yields (> 0.99 kg/ha/yr) to the Chesapeake Bay in 2000 (Figure 5.6c-d). Reduction in cropland in these catchments occurred with westward expansion of suburbs from Philadelphia (PA), northern expansion of suburbs from Baltimore (MD), and the growth of urbanized areas surrounding Harrisburg, York, and Lancaster (PA) (Figure 5.6a). This same growth was predicted on the eastern shore of MD and DE (Figure 5.6a). The forecasted growth surrounding the Salisbury (MD) area illustrated how increased urbanization rates in areas of the highest yields in 2000 had the greatest impacts on reducing TN and TP loadings to the Chesapeake Bay.

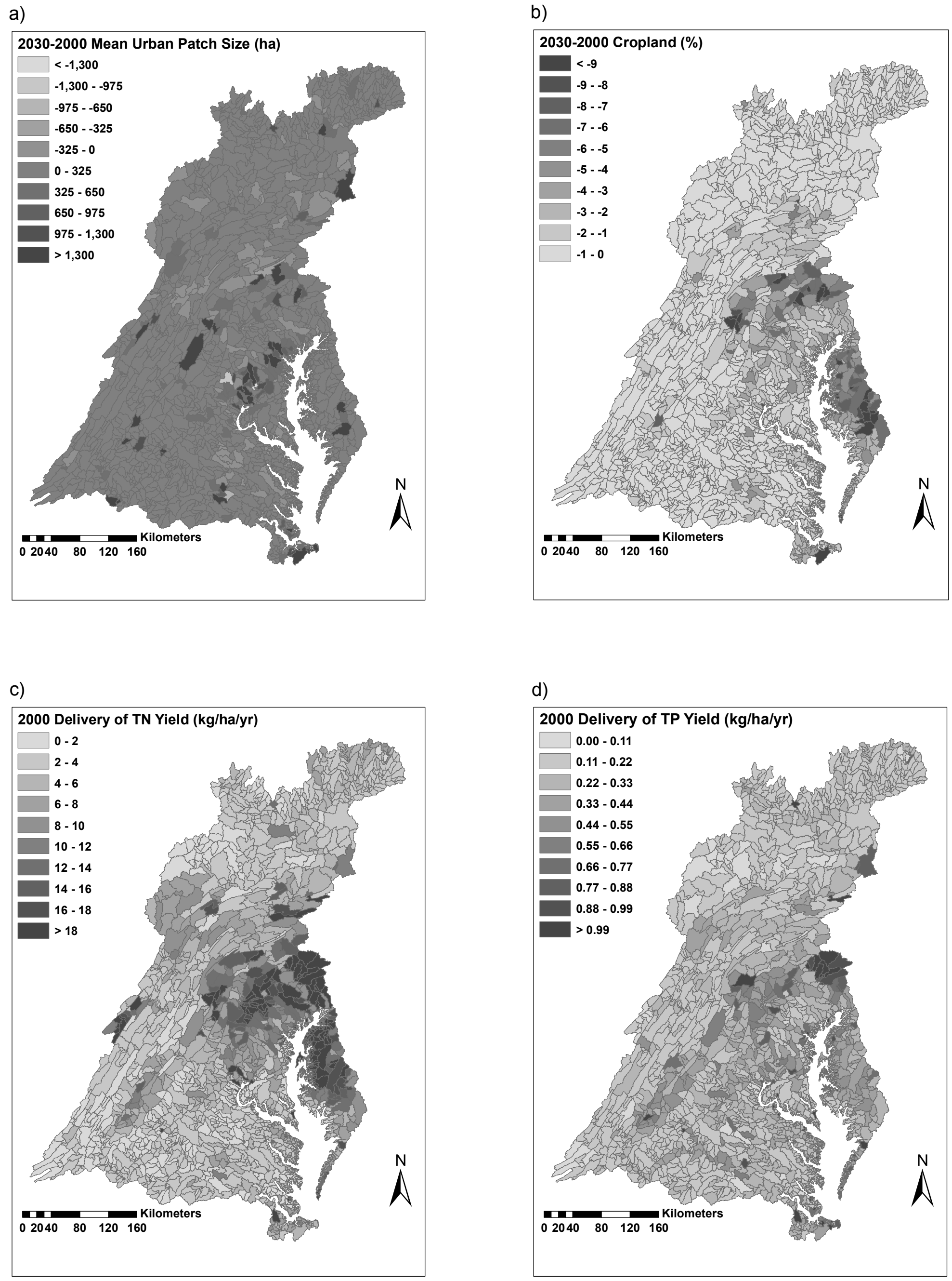


Figure 5.6: Per catchment 2030-2000 differences of the: **a)** Area-weighted mean urban (≥ 10% ISA) patch size source metric in ha and **b)** Cropland (%) projected to have the greatest impacts on catchments with the highest 2000 delivery of **c)** TN and **d)** TP yield to Chesapeake Bay in kg/ha/yr by 2030.

The conversion of extractive land (Figure 5.7a) to development (Figure 5.6a) was predicted to have the greatest impact in catchments currently producing the highest TN yields near Scranton-Wilkes Barre (PA) in the middle Susquehanna Basin and Cumberland (MD) in the upper Potomac Basin (Figure 5.6c) (Roberts *et al.,* 2009; Roberts and Prince, 2010). In addition, the forecasted conversion of barren land within the 31 m riparian stream buffer (Figure 5.7b) to urban (Figure 5.6a) was also predicted to reduce the highest yielding TP catchments currently in the lower Susquehanna Basin near Lancaster (PA) (Figure 5.6d) (Roberts *et al.,* 2009; Roberts and Prince, 2010). In these catchments, the conversion of these cover types reduced lands that were correlated to increased transport of non-point N and P loadings to streams from overland runoff.

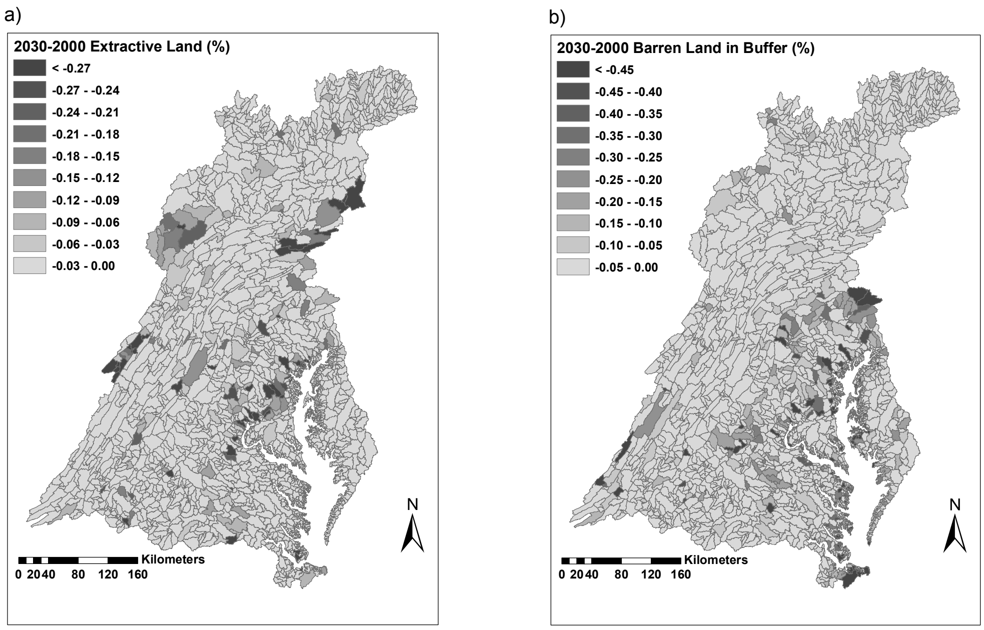


Figure 5.7: Per catchment 2030-2000 differences of the land-to-water delivery metrics of: **a)** Extractive land (%) and **b)** Barren land within the riparian stream buffer (%) in the RESAC 31 m models.

- **What changes in TN and TP loadings will occur in the Chesapeake Bay between 2000 and the future (2030) under the urbanization scenario applied here?**

After urbanization, TN and TP loadings delivered to the Chesapeake Bay by 2030 were projected to be reduced by approximately 19% and 20% (Table 5.2; Figure 5.8a-d) (Roberts *et al.,* 2009). Reductions in TN and TP loadings indicated the pivotal role of decreases in the application of fertilizers and their non-point source runoff (Roberts *et al.,* 2009). Reductions in non-point source losses from fertilizers occurred not only through the conversion of cropland and pasture to development, but the use of a smaller mean application rate for agricultural lands that decreased from 39.78 and 13.65 kg/ha/yr for N and P in 2000 to an estimated 15.75 and 7.49 kg/ha/yr for N and P in 2030.

TN model run					TP model run				
Variable	2000	2030	2030-2000 change	2030-2000 % change	Variable	2000	2030	2030-2000 change	2030-2000 % change
Point	4.4105×10^7	5.2200×10^7	$+8.0950 \times 10^6$	+18.35	Point	1.9665×10^6	2.3677×10^6	$+4.0120 \times 10^5$	+20.40
Applied fertilizer	4.8395×10^7	1.5615×10^7	-3.2780×10^7	-67.73	Applied fertilizer	1.0226×10^6	5.4169×10^5	-4.8091×10^5	-47.03
Applied manure	9.7110×10^6	1.2783×10^7	$+3.0720 \times 10^6$	+31.63	Applied manure	5.2862×10^5	4.8027×10^5	-4.8350×10^4	-9.15
* Atmospheric deposition	3.5803×10^7	2.7355×10^7	-8.4480×10^6	-23.60	Area-weighted mean non-agricultural/ non-urban patch size	1.5772×10^6	5.0174×10^5	-1.0755×10^6	-68.19
Area-weighted mean urban (≥ 10% ISA) patch size	6.9311×10^6	9.1806×10^6	$+2.2495 \times 10^6$	+32.46	Area-weighted mean urban (≥ 10% ISA) patch size	2.7222×10^5	4.0387×10^5	$+1.3165 \times 10^5$	+48.36
Total	1.4495×10^8	1.1713×10^8	-2.7820×10^7	-19.19	Total	5.3671×10^6	4.2953×10^6	-1.0718×10^6	-19.97

Table 5.2: Comparison of 2000 and projected 2030 total loadings in kg/yr delivered to the Chesapeake Bay from all significant sources in RESAC 31 m models. * Denotes significant source variable in the RESAC 31 m TN models that used the same inputs for 2000 and 2030 and was projected to decrease by 2030 due to conversions of significant land-to-water delivery LC/LUs.

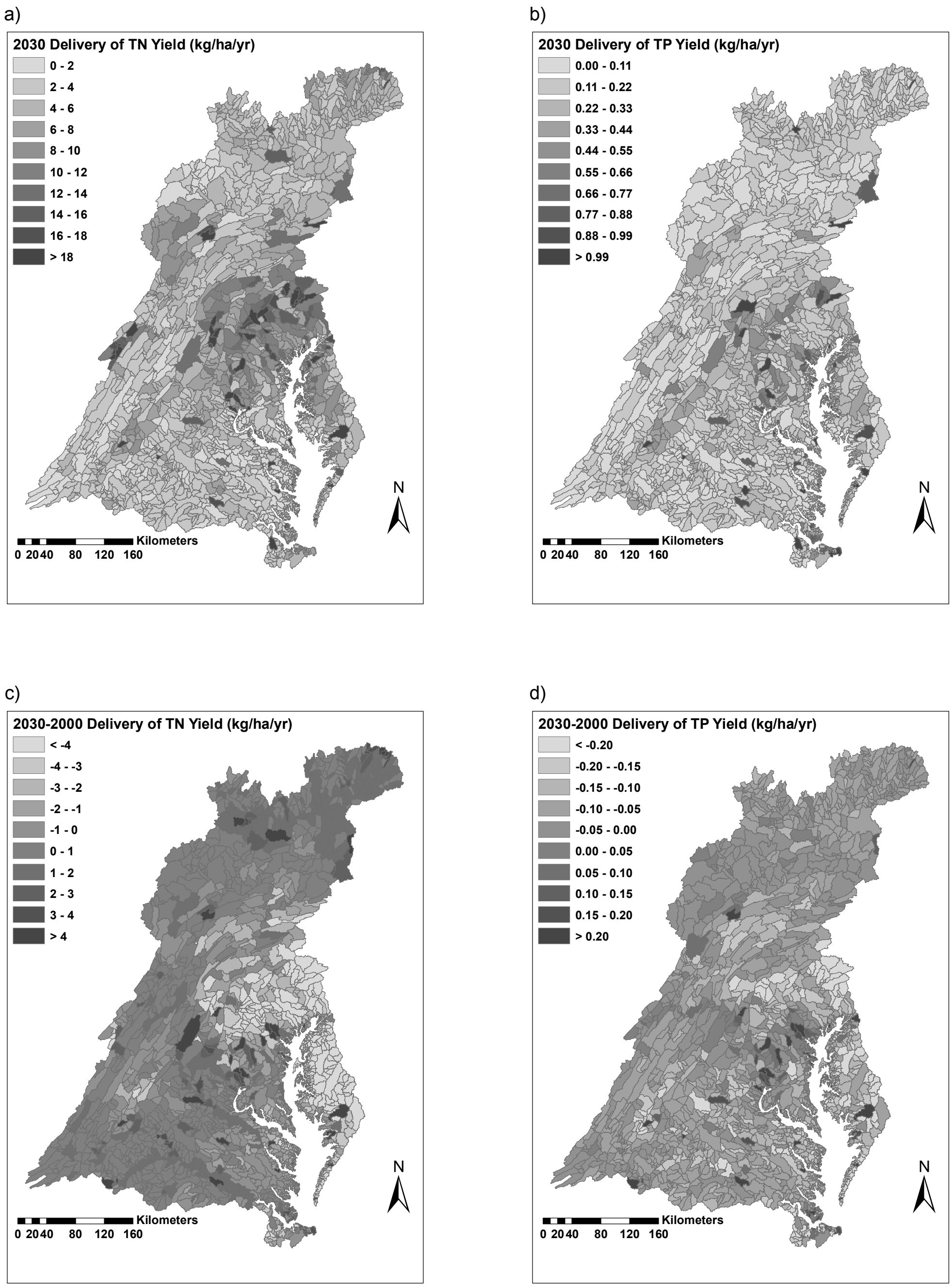


Figure 5.8: Per catchment delivery of: **a)** TN and **b)** TP in yield to Chesapeake Bay in kg/ha/yr in the 2030 RESAC 31 m models with 2030-2000 differences of delivery of: **c)** TN and **d)** TP yield to Chesapeake Bay in kg/ha/yr in the RESAC 31 m models.

Increases in point and impervious-based, non-point source loadings from urban lands can be expected to have lower overall N and P loadings than the cropland and pasture they replace. Even with a projected population increase of four million that lead to an estimated 18% and 20% gain in point N and P loadings and a 32% and 48% gain in non-point N and P loadings (Table 5.2), total urban loading increases were not sufficient to negate the reduction in losses from agricultural lands (Roberts *et al.,* 2009). Recent studies in smaller watersheds in the midwestern U.S. have also concluded that the replacement of agricultural lands and their higher nutrient production with urban LC/LUs of lower nutrient production would lower overall TN and TP (Tang *et al.,* 2005; Wang *et al.,* 2005). The result reported here was the first demonstration that this phenomenon would occur at the large regional scale, represented by the Chesapeake Bay watershed.

5.4 Additional Findings

The integration of compositional and configuration LC/LU metrics at catchment and riparian stream buffer scale within the 2000 SPARROW models gave slightly more accurate stream loading estimates when compared with 87 TN and 104 TP measurement sites, as opposed to the last SPARROW models applied to the Chesapeake Bay in 1997 (Table 5.3) (Brakebill and Preston, 2004; Roberts and Prince, 2010). In addition, the use of metrics at these scales in SPARROW yielded slightly more precise results when compared with the 2000 Phase 4.3 **H**ydrologic **S**imulation **P**rogram-**F**ORTRAN (HSPF) model estimates for loadings to the estuary, also as opposed to the 1997 models (Table 5.4) (Roberts and Prince, 2010). Since the enactment of the Chesapeake Bay Agreement in the 1980s, HSPF has been developed by the Chesapeake Bay Program (CBP) and

United States Environmental Protection Agency (USEPA) as the primary tool aiding in federal and state management and regulatory decisions regarding the estuary (United States Environmental Protection Agency, 2009). Without quantifying comprehensive landscape and riparian stream buffer effects, the 1997 TN model overestimated 2000 HSPF loadings by 15%, as compared to 12% using the RESAC 31 m model (Table 5.4) (Roberts and Prince, 2010). The 1997 TP model underestimated 2000 HSPF by 40%, as compared to 38% using the RESAC 31 m model (Table 5.4) (Roberts and Prince, 2010).

Model run	Model yield r^2	Model RMSE (%)
1997 B & P TN	0.9073	0.2834
2000 RESAC 31 m TN	0.9366	0.2407
1997 B & P TP	0.7413	0.3257
2000 RESAC 31 m TP	0.7503	0.3126

Table 5.3: Comparison of yield coefficient of determination (r^2), and root mean squared error (RMSE) between the 1997 Brakebill and Preston (B & P) and 2000 RESAC 31 m SPARROW models.

Nutrient (kg/yr)	1997 B & P	2000 RESAC 31 m	Mean 1985-1994 Phase 4.3 HSPF	2000 Phase 4.3 HSPF	1997 B & P – mean 1985-1994 Phase 4.3 HSPF % change	2000 RESAC 31 m – mean 1985-1994 Phase 4.3 HSPF % change	1997 B & P – 2000 HSPF Phase 4.3 % change	2000 RESAC 31 m – 2000 HSPF Phase 4.3 % change
TN	1.480×10^8	1.449×10^8	1.420×10^8	1.292×10^8	+4.23	+2.04	+14.55	+12.15
TP	5.210×10^6	5.367×10^6	9.991×10^6	8.673×10^6	-47.85	-46.28	-39.93	-38.12

Table 5.4: Comparison of 1997 B & P and 2000 RESAC 31 m SPARROW model runs with the mean 1985-1994 and 2000 HSPF Phase 4.3 model runs for total loadings in kg/yr delivered to the Chesapeake Bay.

A surprising finding was that catchments long distances away from the Chesapeake Bay delivered significant amounts of TN and TP (Roberts and Prince, 2010). Catchments in the headwaters of the Potomac Basin in western MD near Cumberland and northeastern WV and in the middle Susquehanna Basin near Scranton/Wilkes Barre (PA) were estimated to deliver > 18 kg/ha/yr of TN to the Chesapeake Bay (Figure 5.6c). Similarly, catchments in the headwaters of the Susquehanna Basin near Elmira (NY), in the middle Susquehanna Basin near Scranton/Wilkes-Barre (PA), and in the upper James Basin near Charlottesville (VA) were estimated to deliver > 0.99 kg/ha/yr of TP to the Chesapeake Bay (Figure 5.6d). Seemingly, once nutrients were able to leave these catchments, most were delivered to the Chesapeake Bay. This indicated that larger streams can be pipelines to the estuary with minimal attenuation by stream and reservoirs.

The estimated 2030 applications of manure to cropland and pasture may be greater in regions of the Chesapeake Bay watershed with larger combined animal feeding operations (CAFOs). According to the Gollehon *et al.* (2001), manure nutrient production exceeded the assimilative capacity of the land in the watershed in south central NY and southern and southeastern PA in the upper and middle Susquehanna Basin, the eastern shore of MD, DE, and in west central VA in the upper James and Potomac Basins. In these areas, a surplus of 2.0 to 5.1 x 10^5 tons of dry manure was produced. Thus, the annual estimated mean rate of 9.79 and 52.74 kg/ha/yr manure N for cropland and pasture and 5.56 and 23.77 kg/ha/yr manure P for cropland and pasture used for 2030 projections (Roberts *et al.,* 2009) may underestimate the true future values in these regions after LC/LU changes. Additionally, throughout the rest of the watershed with smaller or no CAFOs, estimated rates may be too high for 2030 after urbanization. In this scenario, findings gathered and presented in Roberts *et al.* (2009) may be too low.

5.5 Management and Policy Recommendations

5.5.1 The case for using more comprehensive LC/LU effects to estimate nutrient loadings

Effects of LC/LU composition on TN and TP loadings to the Chesapeake Bay estuary are being used by local, state, and federal entities (Prince Georges County, MD and the City of Bowie (MD), 2003; Commonwealth of Pennsylvania, 2003; Commonwealth of Virginia, 2005; West Virginia Tributary Strategy Stakeholders Working Group, 2005; and United States Geological Survey (Senus *et al.,* 2005)). However, findings presented by Roberts and Prince (2010) and Roberts *et al.* (2009) indicated that analysis of LC/LU should also consider landscape configuration and spatial arrangement to capture more comprehensively the effects of LC/LU. Roberts and Prince (2010) indicated that the use of compositional and configurational metrics of LC/LU slightly improved the accuracy between predicted and observed loadings throughout the watershed and precision of nutrient loading estimates to the estuary (Tables 5.3-4).

At the local and state level, several mechanistically-based, nutrient loading models that include the effects of LC/LU composition on nutrient loadings have been implemented. These include HSPF within tributaries of the York Basin (Im *et al.,* 2003), the **G**eneralized **W**atershed **L**oading **F**unction (GWLF) on the eastern shore of MD (Lee *et al.,* 2000, 2001), the **S**oil **W**ater **A**ssessment **T**ool (SWAT) model in the Rappahannock Basin (Meng *et al.,* 2009), and SWAT and the **ANN**ualized **AG**ricultural **N**on **P**oint **S**ource Pollution model (AnnAgNPS) on the eastern shore of MD (Ali *et al.,* 2007). At the federal level, the CBP and USEPA are using HSPF to manage the effects of LC/LU change on the watershed-wide nutrient loadings (Chesapeake Community Modeling Program, 2009). As continuous (hourly and/or daily) models for the long-term simulation of nutrients, only LC/LU composition is implemented to calculate TN and TP loadings in watersheds (Borah and Bera, 2003). By incorporating landscape configuration and spatial arrangement at the catchment and riparian stream buffer scale, improved agreement between all of these models may occur.

5.5.2 The case for targeting and prioritizing non-urban LC/LU for urbanization

As a management tool at localized scales, specific LC/LU types should be targeted in the future for urbanization to reduce projected nutrient loadings to the Chesapeake Bay. Even in remote reaches of the watershed, locally-dependent land development can impact TN and TP loadings that reach the estuary. As previously indicated, numerous catchments that were long distances away from the Chesapeake Bay still delivered high TN and TP yields (Roberts and Prince, 2010). Thus, a second policy recommendation is for municipalities at the local (cities and counties) level to inventory and prioritize all non-urban LC/LUs for urban conversion prior to the start of future development. Based on findings from Roberts and Prince (2010) and Roberts *et al.* (2009, submitted) non-urban LC/LUs that should be targeted and prioritized for future urbanization are cropland then pasture, followed by extractive and barren land.

Development of cropland may have dual impacts in regards to reducing nutrient runoff. Results outlined in Roberts and Prince (2010) and Roberts *et al.* (2009, submitted) demonstrated cropland's unique function as: 1) a non-point N and P source through the application of fertilizer and manure and as 2) a land-to-water delivery variable for non-point N resulting from reduced hydraulic conductivity properties that allowed for N in overland and shallow subsurface runoff to reach streams. Thus, conversion of cropland may have multiple benefits in regards to reducing eutrophication in the estuary.

Following cropland, pasture should be prioritized as the next LC/LU targeted for urban conversion due to its vast potential for fertilizer and manure-based, non-point source nutrient genesis (Roberts *et al.,* 2009; Roberts and Prince 2010). Furthermore, using the results from Roberts *et al.* (submitted), forecasted sprawl in catchments was shown to be most beneficial (as judged by TN reductions) if allowed to replace cropland and pasture in smaller developed patches that are unattached to existing urban. In catchments with forecasted sprawl onto or near existing urban, gains in TN loading were projected due to smaller reductions in cropland and pasture (Roberts *et al.,* submitted)

Extractive land is the third LC/LU type that should be prioritized for development due to its non-point N contribution to Chesapeake Bay from overland runoff processes (Roberts and Prince, 2010). Barren land is the fourth LC/LU to be targeted for future urbanization. Although barren land within the riparian stream buffer was a significant land-to-water delivery variable for non-point P to Chesapeake Bay streams, it was not a significant transport mechanism at the catchment scale (Roberts and Prince, 2010). Additionally, studies have indicated that revegetating barren land can counteract its runoff transporting potential (Ziegler and Giambelluca, 1998). To attenuate overland runoff from riparian barren lands, management activities should focus on revegetation.

Thus, by prioritizing development initiatives to the conversion of targeted cover types and the strategic placement of growth, a reduction in adverse impacts on projected nutrient loadings to the estuary can be expected.

5.6 Future directions

One important variable that was not evaluated in these studies that could affect urban-induced LC/LU change on nutrient loadings to the Chesapeake Bay was climate change. Climate change, (such as increased temperature, precipitation, and evapotranspiration) that is predicted for the region by the Intergovernmental Panel on Climate Change (IPCC) (Goldman, 2006) has not been used in forecasts of nutrient runoff, mainly because changes in these were insignificant annually (Preston and Brakebill, 1999; Brakebill *et al.,* 2001; Brakebill and Preston, 2004).. In these existing studies using previous Chesapeake Bay SPARROW models, the authors found no correlation between climatic variables and nutrient loadings using the current ranges of conditions. This can be expected because of the low range of interannual variability in precipitation and temperature throughout the extent of the watershed.

Nevertheless over the time span considered here (30 years), significant climate changes are expected. Since precipitation is the primary mechanism transporting nutrients from their source to streams in watersheds (Wolfe, 2001), the forecasted increase in precipitation in the Chesapeake Bay watershed may lead to detectable changes in nutrient runoff in this time frame. Alexander *et al.* (1996) suggested that with current TN in the watershed, loadings to the estuary could increase with higher precipitation.

Thus, the addition of climate change to obtain a comprehensive forecast of nutrient loadings to the estuary 30 years into the future is warranted. This integration of climate change and LC/LU variables could be accomplished if climatic variables and derivatives, such as precipitation, evaporation, temperature, or even soil moisture of the magnitude anticipated are incorporated within the RESAC 31 m SPARROW models presented in Roberts and Prince (2010) and Roberts *et al.,* (2009, submitted)**.** In addition to annual effects, these variables should be evaluated with seasonal SPARROW models to determine significance of seasonality changes.

Seasonal SPARROW models could separate nutrient loading responses in the watershed based upon the four seasons (three month intervals) and could quantify the impacts of climatic variables that could change watershed-wide, such as snow pack (winter), snow melt and thaw (spring), convective thunderstorms (summer), and soil moisture recharge (autumn). In the watershed, it was already determined that on average the highest volume of runoff occurs during the spring months of March to May (Preston and Brakebill, 1999). Changes in management practices applied to agricultural lands, such as spring tillage to coincide with times of increased runoff, could also be evaluated in these seasonal models. Furthermore, all other significant variables in these models that are subject to changes could be collected and evaluated seasonally as well. Thus, through the use of seasonal models, landscape and climate change could be potentially analyzed in unison to quantify the comprehensive effects of anthropogenic transformation in the Chesapeake Bay and its watershed for better management practices.

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