Watershed-Scale Analysis of Riparian Buffer Function

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Watershed-scale Analysis of Riparian Buffer Function

by

Molly Van Appledorn

A thesis submitted in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

in

Ecology

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2009
ABSTRACT

Watershed-scale Analysis of Riparian Buffer Function

by

Molly Van Appledorn, Master of Science

Utah State University, 2009

Major Professor: Dr. Matthew E. Baker
Program: Ecology

The ability of riparian buffers to filter undesirable nutrients from upland sources has long been recognized as an important ecosystem service for maintaining or improving water quality, and as a result, many land management strategies have been built around the preservation or restoration of buffer zones. Newly derived flow-path metrics have shown great promise as a way to assess riparian buffer function at the watershed scale but a thorough investigation of metric performance was necessary. The goals of this study were to: 1) test the independence of flow-path metrics from traditional metrics using a spatially extensive, independent sample of watersheds, 2) evaluate the effects of stream map resolution on riparian characterization and the ability to predict nitrate discharges, and 3) explore whether nutrient retention estimates may improve the performance of flow-path metrics. The results of this study validated initial findings that flow-path metrics provided more flexible, detailed, and independent measures of land cover patterns compared to traditional methods. Buffer characterization by flow-path metrics was affected by stream map resolution, as were models using metrics to relate nitrate
discharge to watershed land cover patterns. Retention-informed metrics showed promise in improving the ability to relate nitrate-nitrogen discharges to measures of riparian function, especially in certain physiographic contexts. A thorough understanding of flow-path metrics and how they are affected by sampling regime, stream map resolution, and estimates of retention is necessary toward the development of a tool useful to land use managers.

(149 pages)
DEDICATION

“I feel a difference between large, deep-rooted stones and the debris lying at the foot of a cliff, pebbles on a beach, stones rolled to the side of a field…These are loose and unsettled as if on a journey, and I can work with them in ways I couldn’t with a long resting stone. To take such a stone would be like extracting a tooth and I would have missed the greater opportunity of knowing the stone in the place it has become a part of.”

Andy Goldsworth, Stone

To my parents, David and Linda Van Appledorn, who taught by word and by example which stones I should pick up and which stones I should let rest.
ACKNOWLEDGMENTS

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I would especially like to thank my committee members, Bethany Neilson and Tamao Kasahara, for their time and thoughtful insights. And to Matt Baker whose advice, encouragement, and support have been invaluable to me.

Molly Van Appledorn
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Anthropogenic pollution has contributed to the loss of biodiversity, eutrophication, and overall poor ecosystem health in many estuarine and freshwater habitats around the world (Malone et al., 1993; Boesch et al., 2001; Rabalais et al., 2001). In the United States alone forty-three “dead zones,” coastal areas that support very little life due to low oxygen levels, had been reported by Dybas (2005). The Chesapeake Bay is one such area that has experienced a dramatic decline in water quality and ecosystem health, with dead zones reaching up to 40% of the bay’s area at times (Dybas, 2005). Millions of dollars have been spent on mitigating the effects of nonpoint source pollution by engaging in land preservation initiatives, watershed development planning, and installation of stream-side vegetation within the bay’s 160,000 km² watershed (Bernhardt et al., 2005). Near-stream vegetated areas known as riparian buffer zones have been a conservation priority because of their potential to filter nonpoint source pollutants from upslope sources thus reducing negative impacts on downstream ecosystems (Dosskey, 2001).

The attenuation of nutrients by riparian buffers has been well documented along transects from upland areas to streams. Peterjohn and Correll (1984) measured nutrient (carbon, nitrogen, and phosphorous) concentrations along a water flow path from an upslope agricultural area (nutrient source), through a forested buffer, and to a stream. They found decreases in nutrient concentrations, particularly for nitrate-nitrogen, with distance traveled through the buffer that they could not attribute to dilution. Field studies
conducted in a variety of physiographic settings and with an array of vegetative and hydrologic characteristics have also revealed patterns of nitrate-nitrogen reduction with distance through buffers of similar magnitude to Peterjohn and Correll’s (1984) findings (e.g., Lowrance et al., 1984; Jacobs and Gilliam, 1985; Pinay and Decamps, 1988; Dilliha et al., 1989; Lowrance 1992; Jordan et al., 1993; Daniels and Gilliam, 1996; Dukes et al., 2002), or with more varied results (e.g., Schnabel, 1986; Brusch and Nilsson, 1993).

Based on these observed patterns of nutrient reduction with distance through a buffer, a great deal of research has attempted to quantify likely buffer function for whole watersheds. In doing so, researchers hoped to not only test for potential buffer effects on water quality at stream outlets, but also to build statistical models to predict patterns of nutrient discharge. Weller et al. (1996) used measures of riparian wetland area within watersheds to predict phosphorus loads with multiple regression models that explained between 57% and 88% of model variance. In addition to whole-watershed proportions of land cover, Johnson et al. (1997) quantified land cover patterns within a fixed-distance (100m) of streams. Whole-watershed and fixed-distance measures were then related to one of several chemical variables, including nitrate-nitrogen, in a series of multiple regression models. Models using fixed-distance measures of land cover and models using whole-watershed land cover proportions were both strong predictors of nitrate-nitrogen. Jones et al. (2001) also found strong relationships between land cover proportions and nitrogen yields to streams (50 – 86% explained variance) within the Chesapeake Bay watershed using a stepwise regression analysis. They included independent variables such as watershed proportions of land cover, fixed-distance
proportions of land cover, and other factors such as road density, slope, and potential soil loss.

Not all analyses linking watershed land cover patterns to nutrient discharges demonstrated strong relationships, however. Omernik et al. (1981) found only weak relationships between whole-watershed land cover proportions and total nitrogen and inorganic nitrogen levels even when proximity to streams was considered (less than 60% explained variance), and Osborne and Wiley (1988) reported large temporal variation in statistical model outcomes predicting nitrate-nitrogen which demonstrated that buffers may exhibit very weak effects on nutrient discharges. Evidence of both strong and weak buffering effects were reported by Hunsaker and Levine (1995). Although no more than 50% of linear regression model variance was accounted for in this study, the authors concluded that whole-watershed measures of land cover were slightly better predictors of total nitrogen than measures of land cover within a fixed distance of streams and thus accounting for proximity to streams was not important in their modeling approach. However, in a second watershed where proximity to streams was considered more explicitly, they found stronger relationships between land cover proportions and nutrient discharges.

Such ambiguous and conflicting results among studies may suggest that either whole-watershed proportions of land cover or land cover within fixed distances of streams were inadequate representations of landscape processes, or that riparian buffers have no effect on water quality. Assuming that buffers have the potential to reduce nutrient discharges from upslope sources as evidenced by transect studies (e.g., Peterjohn
and Correll, 1984; Lowrance et al., 1984), methods of quantifying potential buffer function for whole watersheds need to be revisited. By representing potential buffer function as gross proportions of forest or wetland within a watershed or within fixed-distance of streams, no distinction is made between forests with or without nutrient contributions from upslope sources, nor is the direction of flow paths (i.e., preferential routes that water follows over the landscape according to topographic constraints) throughout a watershed considered. Interpretations based on these fixed-distance proportions of land cover are not only variable and unreliable, they have potential to misguide watershed land management strategies (Baker et al., 2006).

In response to unsuccessful attempts at linking land cover patterns to nutrient discharge, Weller et al. (1998) used heuristic models to determine which characteristics of riparian buffers may be important considerations when quantifying potential buffer function. By representing nutrient source areas and buffers in simulated landscapes, they were able to explore the relationships among buffer widths between nutrient sources and streams, buffer continuity along stream margins, and the ability of buffers to retain nutrients. They found the best predictor of nutrient discharge to be the frequency of gaps when buffers were assumed to be highly retentive, average buffer width when buffers were relatively leaky, and variability in buffer width when buffers were moderately retentive. The results from this study suggested that statistical models linking land cover patterns to nutrient discharge may be improved by including measures of buffer continuity, average width, and variation in buffer width.
These conclusions led to the development of a new method to quantify riparian buffer potential that linked patterns of land cover to the potential for nutrient retention in an ecologically meaningful way. Baker et al. (2006) used surface flow paths from upland source areas to streams to define riparian buffers as contiguous areas of forest or wetland adjacent to a stream and along a source-to-stream flow pathway. In so doing, Baker et al. (2006) were able to calculate the width of buffer along a flow pathway for any particular unit source area and identify any unbuffered sources. They then aggregated these buffer measures across entire watersheds to quantify land cover patterns in a manner similar to those suggested by Weller et al. (1998). For example, the unique buffer widths assigned to every unit source area were averaged across each watershed to calculate Mean Buffer Width. The distribution of buffer widths was also summarized by its Coefficient of Variation. A third measure called the Frequency of Gaps was defined as the percentage of unbuffered source-to-stream flow paths in a watershed. The definition of riparian buffers used by Baker et al. (2006) also allowed the calculation of an additional measure relating the potential for buffers to reduce nutrient delivery to streams: the Mean Inverse Buffer Width (MIBW). The inverse width of a buffer along a flow pathway for a unit source area reflects expected decrease in delivery with increasing width that has been previously observed in transect studies (e.g., Peterjohn and Correll, 1984). Implicit in the MIBW calculation is the assumption that all riparian buffers filter nutrients from upland sources uniformly and maximally. Therefore the MIBW is a measure of buffer potential under ideal conditions for nutrient attenuation. The MIBW may be used as a weight for calculating the proportion of cropland within a watershed adjusted to account for the
expected effects of retentive buffers. This Adjusted Cropland value is an expression of the proportion of watershed cropland that should reach streams if buffers retain nutrients as well as in the published literature (Baker et al., 2006). The Adjusted Cropland calculation is useful for making comparisons of potential buffer impacts among watersheds. Baker et al. (2006) referred to the measures outlined above as “flow-path metrics” because of their reliance on source-to-stream flow pathways to isolate areas within a watershed directly involved with nutrient export and delivery.

Underlying flow-path metrics are three concepts that distinguish these calculations from previous attempts to quantify buffer potential for watersheds: aggregation, connectivity, and retention (Baker et al., 2006). The concept of aggregation reflects the idea that potential buffer function for a watershed may be related to patterns of nutrient reduction observed along many individual transects perpendicular to streams. Each flow-path metric is an aggregate measure of land cover pattern that incorporates potential impacts of riparian buffers along individual flow paths. The concept of connectivity is used to define the buffers themselves: buffers must be located in between upslope source areas and stream networks along a topographically defined flow pathway. By this definition, it is possible that some stream-side forests or wetlands would not be considered to be riparian buffers if they could not intercept nutrients from a nutrient source. Stream location is important for identifying forest or wetland as riparian buffers, and altering the location of the stream network may affect characterizations of buffers using flow-path metrics (Baker et al., 2007). The concept of retention refers to the ability of riparian buffers to attenuate nutrients under certain conditions. Although complex
hydrologic and biogeochemical interactions such as varying soil saturation levels and the concentration of nutrients intercepted by buffers affect their ability to retain nutrients (e.g., Groffman et al., 1992; Hill, 1996; Vellidis et al., 2003), Baker et al. (2006) designed the MIBW to reflect a “best case” scenario. By assuming all riparian buffers filtered nutrients uniformly and maximally, the MIBW illustrates potential for extant buffers to impact nutrient discharges at stream outlets. Baker et al. (2006) were able to create interpretable measures of land cover pattern based on the biophysical process of nutrient attenuation by incorporating the concepts of aggregation, connectivity and retention into flow-path metrics. Such process-based measures are foundational for building understanding and making predictions about spatial relationships (Li and Wu, 2004).

Flow-path metrics were developed using a cluster sample of watersheds from the Chesapeake Bay basin (the Smithsonian Environmental Research Center, or SERC, dataset). Watersheds in the SERC dataset were selected to capture a broad range of land cover distributions and to exclude factors that may confound water quality analyses (e.g., nutrient point sources). The results from the initial application of flow-path metrics to the SERC dataset suggested that flow-path metrics may be a valuable tool to aid land managers because of their interpretability, efficiency, and sensitivity to regional land cover patterns (Baker et al., 2006). However, it remained unclear if the clustered sampling used in the SERC study design would have biased initial results and thus reduced potential for broad inference or more detailed application and interpretation. Therefore, the overall purpose of this study was to gain a deeper understanding of the
potential for flow-path metrics to augment and inform watershed management by testing the metrics using an independent and comprehensive sample of watersheds.

This study was completed in three sections which are presented as Chapters II – IV. Each chapter addresses one of the three foundational concepts of flow-path metrics—aggregation, connectivity, and retention—in order to better understand: 1) whether flow-path metrics, due to their ability to capture transect-level processes at the watershed scale, relate information that is different than whole-watershed or fixed-distance measures of land cover, 2) how different representations of source-to-stream connectivity affect the ability of flow-path metrics to predict nitrate-nitrogen concentrations at watershed outlets, and 3) how accounting for site-specific factors that influence nutrient retention may affect the ability of flow-path metrics to predict nitrate-nitrogen concentrations.

To be more specific, in Chapter II of this study I test the independence of flow-path metrics from whole-watershed measures of land cover using watersheds from the Maryland Biological Stream Survey (MBSS) dataset, an extensive probability-based sampling regime established by the Maryland Department of Natural Resources in 1993 to assess the condition of streams statewide. The intent of this chapter was to justify the use of flow-path metrics in subsequent statistical modeling without violating model assumptions of variable independence. Statistical relationships among whole-watershed land-cover proportions, fixed-distance proportions, and flow-path buffer metrics within the MBSS dataset are investigated using two publically available land cover maps. The
results from this chapter form the basis for conducting Chapters II and IV, and it is within the context of this chapter that the rest of the study should be viewed.

Characterization of buffers by flow-path metrics has been shown to be sensitive to the resolution of stream map used as input into metric calculations using the SERC dataset (Baker et al., 2007). In Chapter III, I compare flow-path metrics in the MBSS dataset across two stream map resolutions, a “coarser” stream map of the 1:24,000 scale, and topographically-derived stream map that has a scale finer than 1:24,000. I then relate the resulting two sets of flow-path metrics to baseflow grab-samples of nitrate-nitrogen concentrations in a series of multiple regressions to better understand how representations of connectivity may influence predictions of nutrient discharge.

The results of the analyses conducted in Chapter III suggest that there is potential for flow-path buffer metrics to improve understanding of relationship between land cover and patterns of nitrate-nitrogen stream concentrations. In Chapter IV, I contrast measures of mean inverse buffer width (which assume all riparian buffers are able to attenuate nutrients uniformly and maximally) with 5 novel variations that incorporate relative estimates of site-specific nitrate-nitrogen retention. The methodology of computing these variations is described in this chapter more explicitly. Linear regression models relating nitrate-nitrogen concentrations to either flow-path metrics or the novel variations are compared within each physiographic province to explore potential improvements to the existing flow-path buffer metrics.

This study seeks to explore three concepts foundational to flow-path metrics—aggregation, connectivity, and retention—using an independent sample of watersheds in
order to better understand the potential for flow-path metrics to be incorporated into future watershed management tools. Because of the nature of the MBSS dataset, described in further detail in Chapters II through IV, this study is uniquely well-suited to provide a clearer and more thorough understanding of 1) how flow-path metrics are able to describe land cover patterns relative to whole-watershed and fixed-distance metrics, 2) how different representations of source-to-stream connectivity may alter the ability of flow-path metrics to relate stream nitrate-nitrogen concentrations to land cover patterns, and 3) how accounting for site-specific retention may affect the ability of flow-path metrics to relate nitrate-nitrogen concentrations to patterns of land cover.

**Literature Cited**


CHAPTER II

VALIDATION OF FLOW-PATH METRICS USING AN INDEPENDENT SAMPLE OF MARYLAND WATERSHEDS

Abstract

Riparian buffers have the potential to filter undesirable nutrients from source land before they enter the stream, and as a result, are a priority for land management. Previous studies have used whole-watershed land cover proportions or summaries of land cover within fixed-distances of a stream network in statistical models to predict nutrient discharges to streams or to detect buffers effects with variable success. Newly developed measures of potential riparian buffers along source-to-stream flow paths offer an ecologically meaningful alternative to whole-watershed and fixed-distance measures of land cover. In this study I test the relative independence of “flow-path” buffer metrics from whole-watershed and fixed-distance land-cover proportions using a broad sample of watersheds in order to assess their potential for use in statistical models linking land cover patterns to nutrient discharges. I computed flow-path buffer metrics, whole-watershed proportions of land cover, and proportions of land cover within 100m of streams for nearly 1,500 watersheds comprising four physiographic provinces throughout the state of Maryland. These estimates were repeated for two different land cover maps. Flow-path metrics provided information about buffering potential that was distinct from both whole-watershed and fixed-distance measures of land cover. Compared to fixed-distance measures of land cover, flow-path metrics were more independent of whole-watershed land-cover proportions and more appropriate additional predictors of
watershed-scale nutrient discharges. Despite independence from watershed scale patterns of land use, flow-path metrics remained sensitive to regional differences in land cover distributions. Differences between land cover maps had little effect on the relative independence of flow-path metrics and watershed land-cover proportions. This study validates the initial findings that flow-path metrics provided more flexible, detailed and independent measures of land cover patterns compared to whole-watershed or fixed-distance metrics.

Introduction

Anthropogenic pollution has contributed to the loss of biodiversity, eutrophication, and overall poor ecosystem health in many estuarine and freshwater habitats around the world (Malone et al., 1993; Boesch et al., 2001; Rabalais et al., 2003). The Chesapeake Bay, the largest estuary in the United States, has experienced a dramatic decline in water quality and ecosystem health with dead zones (coastal areas that support very little life due to low oxygen levels) reaching up to 40% of the bay’s area at times (Dybas, 2005). Millions of dollars have been spent to mitigate the effects of nonpoint source pollution through land preservation initiatives, watershed development planning, and installation of stream-side vegetation within the bay’s 160,000 km² watershed (Bernhardt et al., 2005). Near-stream vegetated areas known as riparian buffers have been a conservation priority in the region because of their potential to filter nonpoint source pollutants from upslope sources thus reducing negative impacts on downstream ecosystems (Dosskey, 2001).
The ability of riparian buffers to attenuate nutrients has been well documented along transects from upland sources to streams under a variety of physiographic and hydrologic conditions (e.g., Peterjohn and Correll, 1984; Lowrance et al., 1984; Jacobs and Gilliam, 1985; Pinay and Decamps, 1988; Dilliha et al., 1989; Lowrance, 1992; Jordan et al., 1993; Daniels and Gilliam, 1996; Dukes et al., 2002). Based on these observed patterns of nutrient reduction with distance through a buffer, researchers have attempted to quantify likely buffer function for whole watersheds in order to test for potential buffer effects and to predict patterns of nutrient discharge. For example, Weller et al. (1996) used proportions of wetland area in multiple regression models to predict phosphorus loads. In addition to whole-watershed proportions of land cover, Johnson et al. (1997) used proportions of land cover within a fixed distance of stream networks to predict a variety of chemical variables.

Despite their widespread use, whole-watershed and fixed-distance measures of land cover have led to mixed interpretations of the ability for buffers to attenuate nutrients. Linking nutrient discharge to the amount of forest within a fixed-distance of the stream network has revealed both strong (Weller et al., 1996; Johnson et al., 1997; Jones et al., 2001) and weak relationships (Omernik et al., 1981; Osborne and Wiley, 1988), and has failed to explain more variance in nutrient discharge than whole-watershed measures of forest or cropland (Hunsaker and Levine, 1995). Such ambiguities may be attributed the fact that the methods of quantifying potential buffer function used in these studies (whole-watershed proportions of land cover and/or proportions of land cover within a fixed distance of streams) did not discern between
forests with or without contributing nutrient source areas, nor did they consider directed flow paths (i.e., preferential routes that water follows over the landscape according to topographic constraints). Interpretations made from both whole-watershed and fixed distance measures of land cover were not only variable and imprecise, but also had the potential to misguide current watershed management strategies (Baker et al. 2006b).

In response to unsuccessful attempts at linking land cover patterns to nutrient discharge, Weller et al. (1998) used heuristic models to determine which characteristics of riparian buffers may be important considerations when quantifying potential buffer filtering effects. By representing nutrient source areas and buffers in simulated landscapes, they were able to explore the relationships among buffer widths, the ability of buffers to retain nutrients, and buffer continuity along a stream margin. They found that when buffers were highly retentive, the frequency of gaps was a strong predictor of nutrient discharge. However, when buffers were relatively leaky, average buffer width was a strong predictor. The variability in buffer width was the best predictor when buffers were moderately retentive. The results from this study suggested that statistical models linking land cover patterns to nutrient discharge may be improved by including measures of buffer continuity, average width and variation in buffer width (Weller et al., 1998).

These conclusions led to the development of a new method to quantify riparian buffer potential that linked patterns of land cover to the process of nutrient retention in an ecologically meaningful way. Baker et al. (2006b) used surface flow pathways from upland source areas to streams to define riparian buffers as contiguous areas of forest or
wetland adjacent to a stream and along a source-to-stream flow path. In doing so, Baker 
et al. (2006b) were able to calculate the width of buffer along a flow pathway for any particular unit source area and identify any unbuffered sources. Buffer measures were then aggregated across entire watersheds to quantify land cover patterns in a similar manner to that of Weller et al. (1998). Because the new buffer measures relied on source-to-stream flow paths to isolate areas within a watershed directly involved with nutrient export and delivery, Baker et al. (2006b) referred to the measures outlined above as “flow-path metrics”.

Flow-path metrics were applied to a cluster sample of study watersheds selected to represent a range of cropland proportions and population densities while controlling for factors that may confound nutrient analyses (such as sewage outfalls) in order to maximize the potential to detect changes in land cover patterns among physiographic provinces (Liu et al., 2000). Using this sample, Baker et al. (2006b) explored the relative independence of flow-path and fixed-distance riparian characterizations from whole-watershed land-cover proportions as well as the nature of the differences between fixed-distance and flow-path measures. The results from this study suggested that flow-path metrics provided estimates of potential buffer function that were more precise than fixed-distance proportions and more independent of whole-watershed land cover (Baker et al., 2006b).

There is a need to substantiate these initial observations with an independent and comprehensive watershed sample to explore the utility of this method for practical application. The goal of this study is understand whether flow-path metrics are generally
appropriate for use as additional predictors in multiple regression models linking patterns of watershed land cover to nutrient discharges. In order to justify their use, flow-path metrics must be relatively independent of whole-watershed measures of land cover. Also, in order to improve model fit, flow-path metrics should be able to relate novel information not already contained in other land cover measures. Therefore, this study is designed to test the hypotheses that a) flow-path metrics are more independent from whole-watershed land cover than fixed-distance proportions, and b) flow-path metrics relate information about land cover patterns that is different from the information captured by fixed-distance characterizations. To test these hypotheses, I will compare whole-watershed, fixed-distance and flow-path measures for a broad sample of watersheds spanning four physiographic provinces and two different land cover inputs. Thus, an implicit secondary goal of this study will involve understanding metric sensitivity to land cover inputs and sampling design. By completing the analyses using two different land cover maps for watersheds selected using a distinct sampling regime from the original study of Baker et al. (2006b), I will be able to determine if observed patterns of watershed and riparian land cover are heavily influenced by sampling regime or dataset inputs.

Methods

Study Area

A dataset consisting of 1,489 watersheds throughout the state of Maryland ranging in size from 1.17 hectares to 43,116 hectares was used for this study (Figure 1). The watersheds were chosen originally as part the Maryland Biological Stream Survey
(MBSS), an extensive biological and physical monitoring network established to assess and inventory stream ecosystems, based on a stratified random sampling design according to major drainage basin and stream order (1st to 3rd order, non-tidal streams on a 1:250,000 stream map; Mercurio et al., 1999).

The dataset comprised four physiographic provinces (Coastal Plain, Piedmont, Appalachian Mountain, and Appalachian Plateau; Langland et al., 1995). The easternmost province, the Coastal Plain, is characterized by watersheds of relatively low relief, a great deal of agriculture, and wedge-shaped surficial aquifers created by shallow clay confining layers overlain by other unconsolidated sediments (Vroblesky and Fleck, 1991). The Piedmont has gently rolling hills of moderate topography (typically 30 – 100 m in local relief) that are dissected by dendritic networks of streams (White, 2001). There is a wide variety of bedrock that underlies the region, some of which is fractured or highly karstic, allowing for unpredictable groundwater flow and a great deal of variation in stream flow patterns (White, 2001).

Streams are an important geomorphic feature of the Appalachian Mountain province as they determine the landforms with which they are intimately related (Fenneman, 1938; Hack, 1965; Keaton et al., 2005). Steep mountain sides constrain the location of streams to a latticework of channels that cut through shallow soils and are often in contact with bedrock (Keaton et al., 2005). Because of relatively high relief in this province, agricultural land use is typically restricted to fertile valley bottoms along higher order streams (Keaton et al., 2005; Baker et al., 2006b). Broad ridge tops with steep side slopes caused by folding of sedimentary bedrock define the Appalachian
Plateau province, and agricultural land use is typically restricted to the ridge tops (U.S. Geological Survey, 1984; Langland et al., 1985). The long-term average annual precipitation in Maryland is about 43 inches per year (1901-2001), with greater precipitation in the eastern and extreme western parts of the state than in the central region (Wheeler, 2003).

**Geographic Data**

I analyzed publicly available elevation, stream channel, and land cover data sets using ARC/INFO (ESRI, Inc.). Elevation data were obtained from a 30-meter digital elevation model (DEM; National Elevation Dataset, http://ned.usgs.gov). Stream channels were identified using the 1:24,000-scale National Hydrography Dataset (NHD; United States Geological Survey, www.nhd.usgs.gov). The DEM was preprocessed using a normalized excavation version of the AGREE algorithm in order to correct for stream alignment differences between the DEM and the NHD while minimizing the occurrence of undesirable parallel stream flow pathways and watershed boundary distortions (Baker et al., 2006a). The normalized excavation version of the AGREE algorithm initially lowers the elevation of streams to that of the minimum elevation within a 150m locality, and reconditions the DEM surface to allow for uninterrupted downstream flow (Hellwegger, 1997; Baker et al., 2006a). Watershed boundaries were then delineated using the reconditioned DEM and classified by physiographic province if 80% or more of their area fell within a province’s boundaries.

Two different land cover inputs were used in this study. All analyses were conducted using the 1992 National Land Cover Dataset (NLCD; United States
Whole-watershed and Fixed-distance Land Cover Proportions

The area of different land cover designations, such as forest or wetland (hereafter termed “for+wet”), developed, and row-crop agriculture (hereafter, “cropland”) were calculated for each watershed and summarized as proportions of watershed area. To calculate fixed-distance metrics, a 100-meter corridor was constructed around the stream location within which patterns of land cover were identified according to each land cover map. Fixed-distance proportions were calculated as the areas of each land cover within the 100-meter corridor expressed as percentages of the entire area of the corridor. The distance of 100-meters has been used in previous land-cover analyses because calculations using narrower widths (30 – 100 meters) do not significantly impact land cover estimates when using data of 30-meter resolution (Roth et al., 1996). Additionally, the amount of a particular land cover within a fixed-distance corridor was also expressed as a percentage of whole-watershed land cover. In other words, these “near stream” proportions of land cover related the proportion of whole-watershed land cover that was located within the 100-meter stream corridor. The near-stream land cover proportions were calculated to characterize the tendency of a particular land cover type to be located within 100-meters of streams.
Flow-path Metric Calculations

Flow-path metrics were calculated for each watershed following the methods described in Baker et al. (2006b). Briefly, land cover was summarized from the 1992 and 2001 NLCDs such that cropland pixels were identified as sources. Surface flow paths from all source cells to the stream were conducted based on steepest descent (D8 algorithm; O’Callaghan and Mark, 1984). For wet pixels contiguous along a source-to-stream flow path and adjacent to the stream were identified as buffers. For a watershed, the width of buffer located along each source-to-stream flow path could be averaged across all flow paths to calculate mean buffer width. The coefficient of variation in buffer width, a second flow-path metric, was calculated by dividing the standard deviation of buffer widths across all flow paths in a watershed by the mean. Unbuffered source-to-stream flow paths were identified as “gaps.” A third flow-path metric termed “frequency of gaps” was calculated as the percentage of unbuffered source-to-stream flow pathways within a watershed.

Through characterization of buffers according to the above definition, it was possible to calculate an additional measure related to the potential for buffers to retain nutrients. For any cropland cell the proportion of nutrients potentially reaching the stream, $t$, was calculated as:

$$t = \frac{1}{w + 1}$$

(1)

where $w$ was the width of buffer (in meters) along a flow path from the source cell to the stream. Decreases in the proportion of nutrients delivered to streams with increased widths of transport through riparian buffers are consistent with previous observations.
Values of $t$ were averaged across all cropland cells in a watershed to calculate the Mean Inverse Buffer Width (MIBW). Implicit in the MIBW calculation is the assumption that all riparian buffers filter nutrients from upland sources uniformly and maximally. Therefore the MIBW is a measure of buffer potential under ideal conditions for nutrient attenuation that has been previously used to characterize riparian buffer potential (Baker et al., 2006b; Baker et al., 2007).

The MIBW allows for one further calculation: the proportion of cropland within a watershed that is adjusted to account for the expected effects of buffers. This “Adjusted Cropland” percentage is an expression of the proportion of watershed cropland that is expected to reach streams (Baker et al., 2006b). It would be possible for a watershed with a high proportion of agriculture to have the same adjusted cropland value as a watershed with much less cropland if enough for+ wet cells were located along source-to-stream flow paths. Thus, the adjusted cropland calculation is useful for comparing buffer filtering potential among watersheds.

Quantitative Analysis

In order to provide a context for understanding the patterns of metrics in the MBSS watershed sample, I first described general land cover patterns using descriptive statistics. Distributions of whole-watershed, fixed-distance and near-stream proportions of land cover for each physiographic province were compared using boxplots.

To determine if flow-path buffer metrics and fixed-distance proportions of land cover were statistically independent of whole-watershed patterns, I compared these measures to watershed land cover proportions using Pearson product-moment
correlations within each physiographic province. The goal of making these comparisons was to determine whether flow-path metrics were more independent of whole-watershed land cover than fixed-distance buffer characterizations. Relative independence would be interpreted as strong evidence in favor of using flow-path metrics in addition to watershed land cover in statistical models of nutrient discharges.

Additionally, for each physiographic province I regressed flow-path metrics against fixed-distance proportions of forest cover to determine whether the flow-path metrics provided information that was new and different from that of fixed-distance characterizations. I plotted MIBW and adjusted cropland proportions against whole-watershed cropland to understand differences in potential watershed buffering effects with increasing cropland proportions across physiographic provinces. In order to understand whether buffer characterization by flow-path metrics may enhance predictions of nitrate-nitrogen discharges for certain physiographic provinces, I regressed adjusted cropland against whole-watershed proportions of cropland.

All of the analyses described above were completed for each land cover map, the 1992 NLCD and the 2001 NLCD. By examining above relationships with two separate land cover maps, I evaluated how different land cover data sets and potential land use changes might influence the generality of my findings.

**Results**
Among-province land cover differences

Patterns of watershed land cover varied by physiographic province. In general, the Coastal Plain and Piedmont provinces tended to have greater proportions of watershed cropland (>22% and >13%, respectively) while the Appalachian provinces had relatively little cropland according to either land cover map (<10%; Figure 2). Across provinces proportions of cropland within 100m of streams followed a similar pattern with the largest proportions occurring in the Coastal Plain and Piedmont and low proportions in the Appalachian Mountain and Plateau (Figure 2b,c). Fixed-distance cropland proportions were notably smaller in the Piedmont according to the 1992 NLCD (7.9%) than in the 2001 NLCD (16.9%). Although lower proportions of whole-watershed cropland were found in the Appalachian provinces, cropland tended to occur in near-stream areas, especially in the Appalachian Mountain province (>20%, Figure 2c,d). In contrast to the 2001 NLCD results, the Appalachian Mountain province, not the Piedmont, averaged the greatest proportion of near-stream cropland (20.5%, Figure 2c).

A different pattern was observed for for+wet cover across all provinces. Appalachian provinces tended to have greater proportions of watershed and fixed-distance for+wet than the Piedmont and Coastal Plain (Figure 3). The Piedmont consistently had the least for+wet at the watershed scale (<31%) and within 100m of streams (<44%). However, the Piedmont and Coastal Plain provinces had the greatest proportions of watershed for+wet located within 100m of streams (Figure 3e,f), suggesting for+wet cover was more likely to be located near the stream in these provinces, particularly in the Piedmont, compared to the Appalachian provinces.
Whole-watershed vs. Fixed-distance Proportions and Flow-path Metrics

For the entire MBSS study area whole-watershed proportions of cropland were strongly correlated to fixed-distance percent cropland for both land cover maps \( (r = 0.87, \text{ Table 1}) \). Similarly, fixed-distance proportions of for+wet were also strongly correlated with whole-watershed for+wet \( (r \geq 0.86) \). Flow-path metrics were not strongly correlated with whole-watershed metrics according to either land cover map \( (|r| < 0.65) \) except for adjusted percent cropland (Table 1).

Within provinces similar patterns of correlations existed (Table 1), with the Appalachian Mountain province showing stronger positive correlations between watershed cropland and fixed-distance cropland \( (r = 0.94) \) compared to other provinces. Fixed-distance for+wet was also strongly negatively correlated with whole-watershed cropland in this province \( (r \leq -0.85) \) while other provinces showed more moderate correlations. Within provinces flow-path metrics were not strongly correlated to whole-watershed metrics for either land-cover map with the exception of adjusted percent cropland, and frequency of gaps in the Appalachian Mountain province (Table 1).

Flow-path Buffer Characterizations vs. Fixed-distance Land Cover Proportions

Flow-path buffer characterizations showed weak but positive correlations with fixed-distance percent for+wet, attributable to distinct relationships among physiographic provinces and non-linear relationships within provinces (Table 2; Figure 4). The relationship between fixed-distance percent for+wet and mean buffer width was highly
heteroscedastic: very narrow buffers occurred at low proportions of fixed-distance for+wet while there was great variation in mean buffer width at larger fixed-distance proportions (Figure 4a,d). Gap frequency was most strongly correlated with fixed-distance percent for+wet in the Appalachian Mountain province according to the 1992 NLCD, but in the 2001 NLCD the strongest correlations were found in the Coastal Plain (Table 2). The Appalachian Plateau showed the weakest correlation of this relationship due to the extreme amount of variability in mean buffer width at high proportions of fixed-distance for+wet (Figure 4a,d). Gap frequency was strongly and negatively correlated with fixed-distance percent for+wet in all provinces according to both land cover maps (Table 2; Figure 4b,e). Weak but negative relationships were observed between the coefficient of variation in buffer width and fixed-distance percent for+wet. Although these relationships were province-specific, the high degree of variation of buffer width variability at low percentages of fixed-distance cover influenced these results (Figure 4c,f).

*MIBW and Adjusted Cropland*

Relationships between cropland proportions and the Mean Inverse Buffer Width were province-specific, though there was a high degree of variability in MIBW values at low proportions of cropland (Figure 5a,c). In the Coastal Plain, MIBW remained variable even at greater proportions of cropland, while Piedmont watersheds tended to have high MIBW values at greater proportions of cropland. In the Appalachian Mountain province, MIBW values approached 1.0 for watersheds with more than 20% cropland according to
the 2001 NLCD (Figure 5c), but this relationship was not clear for the 1992 NLCD due to limited watersheds identified with more than 20% cropland area.

The relationship between adjusted proportions of cropland and whole-watershed cropland also varied by physiographic province (Figure 5b,d). Land cover map affected the relationships in the Piedmont and Appalachian Plateau with both provinces showing weaker relationships in the 2001 dataset as evidenced by 15% (Piedmont) and 36% (Appalachian Plateau) decreases in cropland coefficients. This means that Adjusted Cropland was more similar to whole-watershed cropland according to the 1992 NLCD than the 2001 NLCD. Additionally, the amount of variance explained by linear regression models decreased for the 2001 NLCD ($r^2_{adj} = 0.68$ vs. 0.86, Piedmont; $r^2_{adj} = 0.39$ vs. 0.45, Appalachian Plateau). Cropland coefficient values for the Coastal Plain province were intermediate of the Piedmont and Appalachian Plateau (0.645, 1992 NLCD; 0.577, 2001 NLCD). The Appalachian Mountain province had the most consistent relationship across land cover map years with less than a 4% difference in the cropland coefficient (0.913, 1992 NLCD; 0.948, 2001 NLCD) and very high amounts of explained variance ($r^2_{adj} ≥ 0.98$).

**Discussion**

*Land cover patterns*

With few exceptions, similar patterns of land cover distributions were found in this study compared to previous land cover descriptions of Baker *et al.* (2006b) and Jones *et al.* (2001). Coastal Plain and Piedmont watersheds were more likely to have greater proportions of total for+wet located within 100m of streams than in the Appalachian
provinces which is in agreement with previous studies (Baker et al., 2006b; Baker et al., 2007). This spatial relationship implies that there may be a greater potential for buffering in the Coastal Plain and Piedmont than the other provinces, but fixed-distance measures are highly correlated with whole-watershed patterns of land cover. Cropland was also more likely to be located near streams in the Coastal Plain and Piedmont than the Appalachian provinces in the 2001 NLCD, most likely due to large values of whole-watershed cropland in these provinces.

*Are flow-path metrics appropriate for use in statistical models?*

Flow-path metrics provided information about buffering potential that was distinct from both whole-watershed and fixed-distance measures as evidenced by weaker correlations and non-linear relationships with these traditional metrics. Similar to the results of Baker et al. (2006b), flow-path metrics were only weakly correlated with whole-watershed land cover across provinces while fixed-distance measures were more strongly related to whole-watershed land cover proportions. Because of their relative independence from whole-watershed land cover, including flow-path metrics in multiple regression models as additional predictor variables of nutrient discharge would not violate the model’s statistical assumptions of variable independence.

Additionally, heteroscedastic relationships between flow-path metrics and fixed-distance metrics demonstrated that flow-path metrics provided implicit and novel information that was not captured using gross proportions of watershed land cover. These results suggest that it may be possible to improve statistical model fits by
incorporating flow-path metrics as additional predictor variables in multiple regressions that link land cover proportions to nutrient discharge. Although the magnitude of change in mean buffer width varied by dataset, the relationship between flow-path and fixed-distance metrics observed in the SERC dataset remained apparent in MBSS watersheds which suggested that flow-path metrics may be useful for models in a wide range of watersheds.

Relationships between flow-path and fixed-distance metrics varied by physiographic province which suggested that although flow-path metrics were independent from watershed scale patterns of land use, they remained sensitive to regional differences in land cover distributions. Recognizing regional disparities in potential buffer function may be important for effective implementation of broad-scale watershed management and restoration efforts. For example, the comparison of adjusted proportions of cropland and whole-watershed cropland suggested that accounting for the spatial arrangement of source areas and buffers in flow-path metrics could potentially improve the ability to predict nutrient discharges for certain watersheds in the Coastal Plain, particularly those with low adjusted cropland proportions despite a large amount of source area. However, on average nutrient predictions based on land cover patterns would not benefit as much from the implicit spatial information offered by flow-path metrics for watersheds in the Appalachian Mountain region. With millions of dollars spent on riparian restoration in Maryland alone (Bernhardt et al., 2005; National River Restoration Science Synthesis, http://nrrss.nbii.gov), the possibility of using measures
such as flow-path metrics to identify regions that may be inherently well buffered may be a useful tool in a prioritization process.

In contrast to flow-path metrics, fixed-distance measures of land cover were highly correlated to watershed-scale measures in the MBSS dataset, underscoring that the use of such metrics to describe riparian buffer function would not relate any information not already summarized by whole-watershed proportions. The fact that relationships between fixed-distance and whole-watershed proportions of land cover were still strongly correlated despite the high degree of variability in land cover pattern in the MBSS dataset emphasized that such measures are ambiguous and are inappropriate descriptors of riparian filtering potential.

When would statistical models benefit the most from flow-path metrics?

Although flow-path metrics may be appropriate for use in statistical models, there may be instances when they may not improve model predictions of potential buffer nutrient filtering. For example, the potential benefit riparian buffers may have on reducing nutrient discharges may be overwhelmed by large proportions of watershed cropland (Figure 5a,c) such that adding flow-path metrics into statistical models may not improve nutrient predictions. Few benefits from including flow-path metrics are gained in watersheds where Adjusted Cropland and whole-watershed cropland values are very similar. In provinces and watersheds where this pattern is observed, accounting for potential buffering effects may do little to improve nutrient predictions based on cropland proportions. However, simply because flow-path metrics may not enhance predictions of
nutrient discharge in certain areas should not discount the benefits that may be gained by restoring buffers in these areas.

*Land cover map comparison*

Differences in observed land cover patterns between the 1992 and 2001 NLCD may be a result of one or a combination of factors: 1) the use of alternative classification algorithms used in different years, 2) differing atmospheric and terrain correction methods, 3) classification error, or 4) actual land use change. Because direct comparisons between these datasets are discouraged (Homer *et al.*, 2004), the purpose of this study was not to attribute differences in metric performance to any specific confounding factor listed above. Rather, comparing metric relationships using two land cover maps was intended to reveal insights into how robust flow-path metrics are in the context of different land cover descriptions.

Patterns of land cover distribution were generally similar between the 1992 and 2001 NLCD, though near-stream and fixed-distance cropland proportions seemed to be somewhat sensitive to map inputs. Individual watersheds that exhibited extreme sensitivities tended to be smaller in size (see Strayer *et al.*, 2003; King *et al.*, 2005). The responses of these watersheds may not have been apparent in mean and median values reported for each physiographic province.

The most noticeable difference in land cover pattern, though still relatively small, was observed in the Piedmont. Here watersheds tended to have greater proportions of cropland located within 100m of the stream network using the 2001 NLCD despite the observation that cropland was the land cover class least likely to change between 1992
and 2001 datasets (Appendix A). In other provinces average near-stream cropland proportions were comparable across years and may be partially due to physical limitations on the location of arable land (e.g., tillable soil is found in the relatively flat valley bottoms in the Appalachian Mountain province; Keaton et al., 2005).

Differences between the two land cover maps had little effect on the independence of flow-path metrics from either whole-watershed or fixed-distance measures of land cover. Correlation strength and quantitative values of the metrics themselves may have differed by land cover map but the overall qualitative pattern of the relationships remained similar. For example, the maximum mean buffer width observed using the 1992 NLCD was nearly 6 times greater than the same measure according to the 2001 NLCD. Yet the variation in mean buffer width was always high at larger proportions of fixed-distance for both NLCDs.

Even though the independence of flow-path metrics was not dramatically affected by differences in land cover maps, the importance of ensuring the accuracy of datasets used to calculate flow-path metrics should not be underestimated. Errors in map classification may be propagated through landscape metrics leading to erroneous conclusions, sometimes with potentially serious ecological and financial consequences if management decisions are based on landscape assessment (Weller et al., 2003; Gergel et al., 2007). In this study, the slope relationship between adjusted cropland and whole-watershed proportions for Piedmont and Appalachian Plateau watersheds changed a great deal between land cover maps. Although the pattern observed for the Appalachian Plateau may be a consequence of either a small sample size, a limited distribution of
cropland proportions represented in the dataset, or a combination thereof, it is nonetheless an example of how changes in data input can affect perceived buffer potential for a physiographic province.

**Conclusions**

The purpose of this study was to test the statistical independence of flow-path metrics from whole-watershed and fixed-distance metrics in a broad, independent sample of watersheds without controlling for potential confounding factors, and to assess the sensitivity of the metrics to land cover inputs. Consistent with the findings of Baker *et al.* (2006b), flow-path metrics provided implicit, novel information that was different and independent from both whole-watershed and fixed-distance metrics. Flow-path metrics were not insensitive to regional land cover characteristics, but rather reflected distinct patterns of land cover that varied according to physiographic province. The results from this study suggest that flow-path metrics may be incorporated into multiple regression models without violating assumptions of variable independence. Flow-path metrics may also improve the ability to test for potential riparian effects for a broad range of watershed types and land uses more precisely than traditional metrics because of their relative independence from whole-watershed and fixed-distance measures.

While changes in land cover maps had little effect on the ability of these metrics to relate different information about potential buffer function, the magnitude of calculated metric values was sensitive to land cover inputs. It is therefore important to consider the quality of data being used in this type of landscape analysis.
The value in flow-path metrics lies in their ability to efficiently characterize riparian buffer function within a clear conceptual framework. By comparing their performance to traditional metrics in a more comprehensive dataset, my findings support the potential utility of flow-path metrics as part of a management toolbox for land-use planners.

**Literature Cited**


TABLE 1. Pearson correlation of watershed percent cropland and percent forest+wetland (“for+wet”) with other whole-watershed land cover proportions, fixed-distance land cover proportions, and flow-path metrics. In the table, the far left column indicates what whole-watershed land cover percentage (either cropland or for+wet) was used in the correlation tests. Results are shown for the entire study region as well as for each physiographic province: Appalachian Plateau (AP), Appalachian Mountain (AM), Piedmont (PD) and Coastal Plain (CP).

<table>
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<th>Dataset</th>
<th>1992 NLCD</th>
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<tr>
<td>Whole-watershed % Cropland</td>
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All correlations shown are significant ($p < 0.05$) unless noted. Strong correlations (|r| > 0.7) are shown in bold.
TABLE 2. Pearson correlation of fixed-distance percent forest+wetland with flow-path descriptions of mean buffer width, gap frequency, CV buffer width, mean inverse buffer width, and adjusted cropland for the 1992 and 2001 National Land Cover Dataset. Results are shown for the entire study region as well as for each physiographic province: the Appalachian Plateau (AP), Appalachian Mountain (AM), Piedmont (PD) and Coastal Plain (CP).

<table>
<thead>
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<th>2001 NLCD</th>
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All correlations shown are significant (p < 0.05). Strong correlations (|r| > 0.7) are shown in bold.
FIGURE 1. The 1,489 watersheds from the Maryland Biological Stream Survey (MBSS) used in this analysis.
FIGURE 2. Proportions of cropland according to whole-watershed (a,b), fixed-distance (c,d), and fraction of land cover near streams (e,f) for 1992 (left side; a,c,e) and 2001 NLCD (right side; b,d,f).
FIGURE 3. Proportions of for+wat according to whole-watershed (a,b), fixed-distance (c,d), and fraction of land cover near streams (e,f) for 1992 (left side; a,c,e) and 2001 NLCD (right side; b,d,f).
FIGURE 4. Mean buffer width in meters (a, d), frequency of gaps (b, e), and CV of buffer width (c, f) plotted against fixed-distance percentages of forest-wetland cover using the 1992 NLCD (top row) and 2001 NLCD (bottom row). Note scale differences between 1992 and 2001 NLCD.
FIGURE 5. Scatterplots of the relationships of percent cropland with mean inverse buffer width (a,c) and adjusted percent cropland (b,d). The 1992 NLCD was used in calculations for graphs a and b; bottom row graphs (c, d) use the 2001 NLCD.
CHAPTER III
THE EFFECT OF STREAM MAP RESOLUTION ON MEASURES OF RIPARIAN BUFFER FILTERING CAPACITY AND MODELS OF NITRATE DISCHARGE

Abstract

Ability to relate patterns of land cover to nutrient discharge for entire watersheds is of interest to land managers wishing to restore riparian buffers and improve water quality, yet measures of riparian buffers may be sensitive to changes in stream map resolution. The goal of this study was to evaluate the effects of stream map resolution on measures of buffer distributions and their ability to predict nitrate-nitrogen concentrations at watershed outlets throughout physiographic provinces within Maryland. I characterized buffer distributions for an extensive sample of watersheds relative to two different stream maps, and then incorporated the characterizations as additional independent variables in a series of linear regression models predicting concentrations of nitrate-nitrogen at watershed outlets. I found that stream map resolution affected buffer characterization with finer stream maps having narrower and more variable mean buffer widths, a greater frequency of gaps, and greater proportions of nutrients expected to reach streams per cropland cell. Linear regression models based on higher resolution stream maps improved predictions of nitrate-nitrogen concentrations in the Coastal Plain and Piedmont, and models based on both maps were comparable in the Appalachian Mountains. No regression models using fine stream maps were supported in the Appalachian Plateau. The results from this study suggest that perception of how well a
watershed is buffered is dependent on stream map resolution. High resolution stream maps may not be appropriate for all regions when linking patterns of watershed land cover to nitrate discharges, underscoring the importance of carefully considering how source-to-sink connectivity is best modeled for any particular watershed or set of watersheds. Analyses incorporating the most appropriate stream map for the region of interest have the potential to provide robust estimates of nutrient discharges and to aid land managers in restoration efforts.

**Introduction**

Anthropogenic pollution has contributed to the loss of biodiversity, eutrophication, and overall poor ecosystem health in many estuarine and freshwater habitats (Boesch *et al.*, 2001; Rabalais *et al.*, 2001). In the United States alone forty-three dead zones had been reported by 2005 (Dybas, 2005). Mitigating the effects of nonpoint source pollution from agricultural areas through the use of conservation tillage, installation of riparian buffers, and wetland restoration has been a major focus of local and national water quality improvement initiatives (Carpenter *et al.*, 1998; Bernhardt *et al.*, 2005).

Riparian buffer zones have been a conservation priority in watershed management because of their ability to attenuate nutrients, potentially reducing the impact on downstream ecosystems (Dosskey, 2001). Transect-scale studies have demonstrated significant reductions in nitrate-nitrogen loads when nutrients are transported through vegetated areas (e.g., Peterjohn and Correll, 1984; Dilliha *et al.*, 1989; Lowrance, 1992;
Daniels and Gilliam, 1996; Dukes et al., 2002). However, the ability of buffers to do so is affected by their hydrologic connectivity to nutrient sources (Baker et al., 2001).

Until recently spatial analyses designed to quantify buffer potential have neglected the importance of the hydrologic connectivity between source areas and streams that is necessary for effective nutrient attenuation, using either whole-watershed proportions of land cover or proportions of land cover within fixed distances of streams. When such measures of land cover were related to nutrient discharge, both strong (Weller et al., 1996; Johnson et al., 1997; Jones et al., 2001) and weak relationships (Omernik et al., 1981; Osborne and Wiley, 1988) were reported. In response Baker et al. (2006b) incorporated the concept of hydrologic connectivity into an improved measure of buffer potential by explicitly linking riparian buffers to upslope source areas according to surface topography. In this method, buffers are defined as forest or wetland areas contiguous along a source-to-stream flow path and adjacent to a stream (Baker et al., 2006b). Referred to as “flow-path” buffer metrics because of their reliance on flow paths to isolate areas of a watershed directly involved with the export and delivery of nutrients to streams, these land cover measures were a marked improvement over previous methods and have shown great promise as a tool for land-use managers (Baker et al., 2006b; Baker et al., in review).

Because flow-path metrics relied on the location of streams to define riparian buffers, they were sensitive to the resolution of stream map used in the analysis. In a study comparing characterizations of potential buffer function using flow-path metrics across three stream maps of different resolutions, finer stream maps were found to dissect the landscape by reaching farther up into watersheds than coarser stream maps (Baker et
Decreases in scale resulted in narrower buffers, increases in gap frequency, and diminished estimates of the potential for riparian buffers to effectively reduce nutrient loads to the stream (Baker et al., 2007).

Despite expressing sensitivity toward changes in connectivity, flow-path metrics have been used to relate in-stream nitrate concentrations to patterns of watershed land cover (Baker et al., in review). However, it is likely that such relationships are affected by the scale of stream map used in the metric calculation. For example, two different representations of stream channels may dramatically change the perceived flow routing of a watershed such that cropland may appear well-buffered according to one representation but is completely unbuffered according to another, depending on the location of the stream channels relative to land cover patterns (Figure 6).

The goal of this study was to evaluate the effects of stream map resolution on measures of riparian buffer function quantified along flow pathways, and their ability to predict concentrations of nitrate–nitrogen at watershed outlets using a broad, representative sample of watersheds throughout the state of Maryland. Flow-path metrics were computed using two stream maps of different resolutions and were then related to stream nitrate-nitrogen concentrations. Models were compared across stream map resolutions to understand 1) how stream map resolution affected measures of riparian buffer function, 2) which combination of metrics and stream map resolutions could better predict nitrate-nitrogen concentrations at the watershed scale, and 3) how these patterns varied by physiographic province.
Methods

Study Area

A dataset consisting of 1,592 watershed sampling sites throughout the state of Maryland was used for this study (Figure 6). Sites were chosen originally as part the Maryland Biological Stream Survey (MBSS), an extensive biological and physical monitoring network to assess and inventory stream ecosystems, based on a stratified random sampling design according to major drainage basin and stream order (1st to 3rd order, non-tidal streams on a 1:250,000 stream map; Mercurio et al., 1999).

The dataset comprised four physiographic provinces (Coastal Plain, Piedmont, Appalachian Mountain, and Appalachian Plateau; Langland et al., 1995). The eastern-most province, the Coastal Plain, is characterized by watersheds of relatively low relief, a great deal of agriculture, and wedge-shaped aquifers created by a shallow clay confining layer overlain by other unconsolidated sediments (Vroblesky and Fleck, 1991). The Piedmont has gently rolling hills of moderate topography (typically 30 – 100 m in local relief) that are dissected by dendritic networks of streams (White, 2001). There is a wide variety of bedrock that underlies the region, some of which is fractured or highly karstic, allowing for unpredictable groundwater flow and a great deal of variation in stream flow patterns (White, 2001). Streams are an important geomorphic feature of the Appalachian Mountain province as they determine the landforms with which they are intimately related (Fenneman, 1938; Hack, 1965; Keaton et al., 2005). The steep mountain sides constrain the location of streams to a latticework of channels that cut through shallow soils and are often in contact with bedrock (Keaton et al., 2005). Because of the relatively high relief in this province, agricultural land use is typically restricted to fertile
valley bottoms along higher order streams (Keaton et al., 2005; Baker et al., 2006b). Broad ridge tops with steep side slopes caused by folding of sedimentary bedrock define the Appalachian Plateau province, and agricultural land use is typically restricted to the ridge tops (U.S. Geological Survey, 1984; Langland et al., 1995). The long-term average annual precipitation in Maryland is about 43 inches per year (1901-2001), with greater precipitation in the eastern and extreme western parts of the state than in the central region (Wheeler, 2003).

**Nutrient Data**

Grab samples for water chemistry analysis were collected according to the methods outlined in Mercurio et al. (1999). For each watershed, a single grab sample was collected from the stream during the spring index period (March 1 to May 1) on one occasion within a nine-year period from 1999 to 2003. Samples were stored on ice, brought to the laboratory within 48 hours of collection and analyzed for nitrate-nitrogen concentrations (mg/l) using a Dionex 2001i ion chromatograph (Sunnyvale, CA) following EPA standards for water quality analysis (EPA, 1987).

**Geographic Data**

I analyzed publicly available elevation, land cover, and stream channel data sets using ARC/INFO (ESRI, Inc.). Elevation data were obtained from a 30-meter digital elevation model (DEM; National Elevation Dataset, http://ned.usgs.gov) and land cover was derived from the 2001 National Land Cover Dataset (NLCD; United States Geological Survey, http://seamless.usgs.gov). Two alternative stream maps were used in this analysis: “NHD” and “FINE.” NHD stream channels were identified using the
1:24,000-scale National Hydrography Dataset (United States Geological Survey, www.nhd.usgs.gov). In this method, the DEM was first preprocessed using NHD streams according to a normalized excavation version of the AGREE algorithm in order to correct for stream alignment differences between the DEM and the NHD while minimizing the occurrence of undesirable parallel stream flow pathways and watershed boundary distortions (Baker et al., 2006a). The normalized excavation version of the AGREE algorithm initially lowers the elevation of streams to that of the minimum elevation within a 150m locality, and reconditions the DEM surface to allow for uninterrupted downstream flow (Baker et al., 2006a; Hellweger, 1997).

The FINE stream map was generated using a nonparametric deviance reduction method which parses a distribution of slope-contributing area products according to break points that minimize the amount of variance in each group (Hill and Baker, unpublished manuscript). Two slope-contributing area relationships were calculated as the product of local slope and upslope contributing area or contributing area divided by local slope for each physiographic province. Both of these distributions were split to maximize deviance reduction in resulting groups of cell values. The group with higher slope-area values was further subdivided two more times to identify raster cells with large and statistically distinct values. Cells with large values for both slope-contributing area relationships were interpreted as likely channel initiation points due to erosive power or wetness, respectively (Montgomery and Dietrich, 1988). Streams were delineated along flow lines downslope of potential channel initiation points. The resulting stream grid was a version of the NHD that was finer than 1:24,000-scale (hereafter, “FINE” stream map).
Before calculating metrics for each watershed, the raw DEM was first preprocessed to allow for uninterrupted downstream flow by using either the NHD or FINE stream maps as input into a normalized excavation and then by filling spurious pits (Jenson and Domingue, 1988). Watersheds were delimited from the collection location of water chemistry samples and were classified by physiographic province if 80% or more of their area fell within a province’s boundaries; watersheds that did not meet the areal requirement were excluded from this analysis as were watersheds that contained no cropland. The 1,592 watersheds used in this analysis ranged in size from 1.17 hectares to 43,116 hectares.

Land Cover Summaries

Land cover patterns of row crop agriculture, forest or wetland (hereafter, “for+wet”), and development were summarized for whole watersheds, within a fixed-distance of streams, and near-stream cover as a proportion of total cover. To calculate fixed-distance measures, a 100-meter corridor was constructed around a stream network within which patterns of land cover were identified and summarized as percentages of the entire corridor area. The distance of 100-meters has been used in previous land-cover analyses because calculations using narrower widths (30 – 100 meters) do not significantly impact land cover estimates when using data of 30-meter resolution (Roth et al., 1996). The expression of near stream cover as a proportion of total cover was calculated to characterize the tendency of a particular land cover type to be located within 100 meters of streams.
Flow-path Metric Calculations

Flow-path metrics were calculated for each watershed following the methods described in Baker et al. (2006b). Briefly, land cover was summarized from the 2001 NLCD such that cropland pixels were identified as sources. Surface flow paths from all source cells to the stream were conducted based on steepest descent (D8 algorithm; O’Callaghan and Mark, 1984). For wet pixels contiguous along a source-to-stream flow path and adjacent to the stream were identified as buffers. For a watershed, the width of buffer located along each source-to-stream flow path could be averaged across all flow paths to calculate mean buffer width. The coefficient of variation (CV) in buffer width, a second flow-path metric, was calculated by dividing the standard deviation of buffer widths across all flow paths in a watershed by the mean. Unbuffered source-to-stream flow paths were identified as “gaps”. A third flow-path metric termed “frequency of gaps” was calculated as the percentage of unbuffered source-to-stream flow paths within a watershed.

Through characterization of buffers according to the above definition, it was possible to calculate an additional measure related to the potential for buffers to retain nutrients. For any cropland cell the expected proportion of nutrients reaching the stream, \( t \), was calculated as:

\[
 t = \frac{1}{w + 1}
\]

(1)

where \( w \) was the width of buffer (in meters) along a flow path from the source cell to the stream. Decreases in the proportion of nutrients delivered to streams with increased widths of transport through riparian buffers are consistent with previous observations.
Values of $t$ were averaged across all cropland cells in a watershed to calculate the Mean Inverse Buffer Width (MIBW). Implicit in the MIBW calculation is the assumption that all riparian buffers filter nutrients from upland sources uniformly and maximally. Therefore the MIBW is a measure of buffer potential under ideal conditions for nutrient attenuation that has been previously used to characterize riparian buffer potential (Baker et al., 2006b; Baker et al., 2007).

The MIBW allows for one further calculation: the proportion of cropland within a watershed that is adjusted to account for potential effects of buffers. This “Adjusted Cropland I” percentage is an expression of the proportion of watershed cropland that is expected to reach streams (Baker et al., 2006b). It would be possible for a watershed with a high proportion of agriculture to have the same adjusted cropland value as a watershed with much less cropland if enough for+ wet cells were located along source-to-stream flow paths. Thus, Adjusted Cropland I is useful for comparing buffer filtering potential among watersheds. A second adjusted cropland measure, “Adjusted Cropland II,” was calculated as the proportion of cropland entering streams through gaps. This measure assumes that any transport through a buffer results in complete nutrient retention (sensu Baker et al., in review).

Quantitative Analysis

I compared differences in land cover patterns summarized by whole-watershed, fixed-distance, and near-stream proportions between stream map resolutions using paired t-tests for the study region and each province (Zar, 1999). To compare the effect of stream channel representation on flow-path metrics, paired t-tests were used to assess
statistical differences in gap frequency, mean buffer width, CV, and MIBW between stream map resolutions across the study region and within each physiographic province (Zar, 1999).

Patterns of land cover were related to nitrate-nitrogen concentrations using linear regression models. All regression models described below were calculated both with and without including the proportion of development as an additional predictor variable, a factor that has been shown to be a significant secondary source of nitrate-nitrogen in the study area (Weller et al., 2003; King et al., 2005).

A set of baseline response models was constructed by regressing nitrate-nitrogen concentrations against whole-watershed cropland or cropland and development. Mean buffer width, frequency of gaps, and MIBW were added into each baseline regression sequentially as additional predictor variables. A second set of regression models was constructed using adjusted proportions of cropland as predictors of nitrate discharge. Both the Adjusted Cropland I and II were regressed against nitrate-nitrogen concentrations and the results compared with those of previously described models. All 12 models was constructed using both NHD and FINE stream maps resulting in a total of 24 models.

All linear regression models were compared using two methods: a) Akaike Information Criteria, adjusted using a second-order correction to account for small sample size relative to the amount of model parameters (AICc), and b) adjusted coefficients of determination ($R^2_{adj}$). In the AICc approach, the quality of candidate models is compared based on a balance of model parsimony and unexplained variance using a calculated AICc score (Burnham and Anderson, 2002). For any set of models,
the model with the smallest AICc score is considered to be of highest quality. Direct comparisons between the highest quality model and any other candidate may be estimated by the difference in AICc scores ($\Delta_i$) where $\Delta_i < 2$ suggests comparable models, and models with AICc differences greater than 10 have virtually no empirical support. In addition, AICc weights ($w_i$), a rescaling of AICc scores, were calculated and may be interpreted as the relative likelihood that a particular model is the most appropriate given the data.

Regression model parameters and adjusted coefficients of determination were calculated using a 10-fold cross-validation method (Fielding and Bell, 1997). Data were randomly partitioned into 10 equal subsets from which a training dataset, consisting of 9 of the 10 subsets, and a test dataset, the remaining 10% of the data, were constructed. Regression models were fitted to the training dataset and error rates of the model fit were obtained by subsequently applying the model to the test dataset. The process of fitting and validating was repeated 10 times, each time withholding a different data subset as the test dataset. Model parameters and error rates were averaged to obtain the reported mean values. Regression models that accounted for higher explained variance as measured by the adjusted coefficient of determination were interpreted as being of higher quality than other models.

**Results**

**Land Cover Characteristics**

Increasing stream map resolution resulted in small changes in fixed-distance proportions of land cover, few of which were significant (Table 3). Very slight,
insignificant increases in fixed-distance proportions of cropland were found in all provinces but the Coastal Plain, and proportions of for+wet within 100m of streams decreased in all provinces but were only significant in the Appalachian provinces (Table 3). In contrast, the magnitude of change in near-stream proportions of land cover was much greater with differences of up to 55% from one stream map resolution to another (near-stream cropland, Appalachian Plateau; Table 3). The proportion of watershed cropland located within 100m of streams increased significantly by 34% over the entire study area with increasing stream map resolution (t = -20.4, p < 0.001). Differences in near-stream cropland between resolutions were greater in the Appalachian provinces compared to either the Coastal Plain or Piedmont (Table 3). Proportions of near-stream for+wet also increased significantly over the entire study area when the FINE map was used (34.8% to 42.9%; t = -27.5, p = < 0.001). The Coastal Plain and Appalachian Plateau provinces had the greatest proportional increases in near-stream for+wet and the Appalachian Mountain region had the least, though still significant (27.8% vs. 30.2%; t = -3.8, p < 0.001).

*Buffer Characterization by Flow-path Metrics*

The characterization of buffers varied by stream map resolution with finer stream maps having narrower and more variable mean buffer widths, a greater frequency of gaps, and greater proportions of nutrients expected to reach streams for every cropland cell across the entire study region (p < 0.001, N = 1,592; Table 4). Mean buffer widths using FINE maps were 73% narrower than those using the NHD maps for the whole MBSS dataset, and with the exception of the Appalachian Mountain province, mean
buffer widths for each province were about 30% to 90% greater using NHD maps (Table 4). Instead of declines in buffer widths with finer resolution stream maps, mean buffer widths of the Appalachian Mountain province increased significantly from 13.7m to 19.4m (p = <0.01, t = -2.6), though median values were relatively similar between stream maps. Buffers in the Appalachian Plateau were wider than any province for both stream map resolutions, averaging nearly 40 meters in width according to the NHD and over 20 meters using the FINE map (Table 4; Figure 7). Piedmont watersheds consistently had the narrowest buffers of all provinces according to either stream map (4.2m, NHD; 2.5m, FINE). In all provinces mean buffer width values were heavily influenced by the presence of large statistical outliers that caused means to be much greater than median values (Figure 8).

As mean buffer widths tended to decrease with finer resolution, the coefficient of variation in buffer width increased in every province, and with the exception of the Appalachian Mountains, all observed differences were significant (Table 4; Figure 8). Coefficient of variation values were the largest in the Appalachian Mountain province as well, over 1.5 times greater than the second-highest ranking province (Piedmont) according to the NHD. The greatest increases in the coefficient of variation across stream maps were observed in the Appalachian Plateau, nearly 15% change.

Finer map resolutions resulted in significantly more gaps for all provinces, with the Piedmont having the highest occurrence of gaps using either stream map (48.1%, NHD; 55.0% FINE; Table 4; Figure 8). The largest increases in gap frequency between stream maps were observed in the Appalachian Plateau (approximately 25%) though this province had the lowest frequency of gaps using either the NHD or FINE maps. The
rank order of gap frequency among provinces was consistent for both stream map resolutions with the Appalachian Plateau, Coastal Plain, Appalachian Mountain, and Piedmont ranking from lowest to highest (Figure 8).

The estimated proportion of cropland contributions reaching the streams (MIBW) increased with stream map resolution \( (p < 0.001, t = -12.7; \text{Table 4}) \) consistent with the observed decreases in mean buffer width, more frequent occurrences of gaps, and more variation in buffer width. Despite similar MIBW values using the FINE stream map for the Appalachian Mountain and Coastal Plain provinces, a proportionally greater increase in mean inverse buffer width was found in the Coastal Plain (18% increase vs. 4% increase; Table 4). The consequences of changes in MIBW between stream map resolutions affected the adjusted proportions of cropland \((I)\) such that for most provinces values of adjusted percent cropland \((I)\) tended to show less differences from whole-watershed proportions of cropland when FINE stream maps were used compared to NHD maps (Figure 9). The Coastal Plain saw the greatest increase in slope (23%) from the NHD to FINE stream maps. In other provinces, changes in slope were negligible (<6%).

Linear Regression Model Comparison

Linear regression models using the NHD stream map were better predictors of nitrate discharge than models using FINE stream maps in the Appalachian provinces while the opposite was true for the Coastal Plain and Piedmont (Table 5). No model using NHD maps were among the highest ranked models in the Coastal Plain and Piedmont, and no model using FINE stream maps were among the most supported in the Appalachian Plateau. In fact, only one of the 14 candidate models was identified as
clearly being the best fit in the Coastal Plain province with no other models—either NHD- or FINE-based—achieving a 1% chance or better of being selected as a quality model. In the Appalachian Mountain province models based on coarser stream maps were more well-supported and much more likely to be the most appropriate models ($\Delta_i < 2.0; 0.17 < w_i < 37$) compared to models based on the FINE stream map ($\Delta_i > 7.0; w_i = 0.01$). Additionally, NHD-based models explained 89% more variance than FINE stream map models and were nearly 17 times more likely than the highest ranked FINE stream map models to be the best selection according to evidence ratios ($w_1 / w_2$).

For either stream map, models using whole-watershed proportions of cropland to predict nitrate discharges were ranked higher than models using adjusted cropland proportions for all provinces except the Appalachian Plateau, and the majority of models with $w_i$ values greater than 0.01 included proportions of watershed development (Table 5). For all other provinces models based on adjusted cropland proportions never had $w_i$ values greater than 0.01. It should be noted that no models for the Appalachian Plateau province had a positive $r^2_{adj}$ value indicating a lack of sufficient fit to the data.

Discussion

Stream Map Resolution and Flow-path Metrics

Finer-scale stream maps tended to dissect watersheds more than coarser maps and in doing so revealed implicit, province-specific patterns of land cover distributions. This was particularly evident when comparing near-stream land cover proportions. Fixed-distance measures were relatively insensitive to stream map resolution, emphasizing that such metrics may lack the ability to characterize potential buffer function effectively.
Increases in near-stream proportions of cropland and for+wet were not surprising because with more streams mapped at the finer resolution, greater proportions of total watershed land cover were sampled within the fixed-distance of 100m from streams. However, proportionally greater increases in near-stream cropland were observed for the Coastal Plain and Appalachian Plateau provinces compared to other regions which suggested that in these provinces cropland occurred where FINE streams began. Remnant forests were located along larger order streams in Coastal Plain watersheds while cropland was distributed more broadly across the upland areas. The fining of the stream map resulted in streams extending through many of these remnant forest areas and into cropland which led to increases in near-stream cropland measures. Near-stream proportions of cropland increased more than near-stream for+wet proportions in the Appalachian Mountain province because streams tended to be added in the valley bottoms where farming is more likely to occur (Baker et al., 2006b) rather than along the steeper, forested hillslopes. Cropland in Appalachian Plateau watersheds tended to occur on the relatively flat hill tops with for+wet located along the steeper hill slopes and larger order streams. New streams in the FINE map tended to be added along the hill slopes rather than the flat hill tops, thus near-stream measures of for+wet proportions were greater with more detailed stream maps.

Buffers appeared to be narrower and more variable in width, and had more gaps according to the FINE stream map which is consistent previous findings (Baker et al., 2007), though a notable exception occurred in the Appalachian Mountain province. Here the relatively high-relief topography restricted the location of streams such that the FINE stream map often was not different than the existing NHD map, and in many cases, NHD
streams reached up farther into the watershed than FINE streams. Any additional stream channel mapped at the FINE resolution tended to occur as short tributaries to larger order streams rather than in the headwater areas. Although most farming occurs in the valley bottoms, any cropland in the upper portions of an Appalachian Mountain watershed tended to appear especially well-buffered according to the NHD. This effect translated into a pattern different than what was observed for all other provinces in this study and those of Baker et al. (2007): Appalachian Mountain buffer width remained relatively unchanged with finer stream maps.

Stream Map Resolution and Predictions of Nitrate-nitrogen Concentrations

Stream map resolution affected the ability to relate landscape metrics to nitrate-nitrogen concentrations, even dramatically so in the Coastal Plain where a FINE-based model clearly outperformed all other candidate models. In this province as well as in the Piedmont, models based on more detailed stream maps were better predictors of nitrate concentrations than models based on coarser maps. In the Coastal Plain, the improvement in nitrate-nitrogen concentration predictions with finer stream maps may be because a) in reality, streams do indeed extend through the remnant stream-side forests and into agricultural upland, or b) because of ditching or tile drainage systems, discharge from the cropland remained relatively unbuffered such that riparian characterizations using FINE stream maps better represented patterns of nitrate discharge even though the actual stream locations were more accurately depicted by the NHD map. FINE stream maps also greatly improved predictions of nitrate-nitrogen concentrations at watershed outlets in the Piedmont despite a high degree of physiographic variability that may have
potentially confounded the relationship between land cover estimates and nutrient discharges (Jordan et al., 1997; Lowrance et al., 1997). It is possible that channel incision, anthropogenically-altered hydrologic pathways not reflected in stream maps, impervious surface, or the propensity of groundwater to travel through fractured regolith in some areas (Pavich et al., 1989; Jordan et al., 1997; Lowrance et al., 1997) that resulted in cropland run-off bypassing denitrification areas in riparian zones. In this case, buffer characterizations using FINE stream maps would be more representative of source-sink connectivity rather than the actual nutrient transport pathways.

In contrast to the Coastal Plain and Piedmont provinces, finer resolution stream maps did not necessarily improve in-stream nitrate concentration predictions in the Appalachian provinces. Both NHD- and FINE-based models had quality fits in the Appalachian Mountain watersheds, possibly owing to the similarities between the two map resolutions discussed above, though models using the coarser resolution were a slightly better choice than FINE-based models. The relative success of NHD-based models in the Appalachian Mountain province suggests that headwater contributions to stream nitrate discharges may not be as important as previously suspected in other regions (e.g., Alexander et al., 2000; Boyer et al., 2002; Alexander et al., 2007).

Due to the high degree of variability within the MBSS dataset, specific reasons why one particular model was better suited over another model for any given province are unclear, however the statistical underpinnings of regression modeling seemed to drive general patterns of model performance. In every province except for the Appalachian Plateau which produced un-interpretable models, baseline regression models, many with two or more fitted parameters, were better predictors of nitrate discharges than adjusted
cropland models. When flow-path metrics were included with whole-watershed cropland as additional predictor variables in a multiple regression model, the separate statistical effects of cropland and buffering were allowed to vary independently of each other and therefore such models were more likely to outperform adjusted cropland models. In contrast, by integrating the MIBW into measures of watershed cropland to create the Adjusted Cropland I measure, the magnitude of the cropland effect on nitrate discharge was forced to be equivalent to the magnitude of the buffering effect captured by the MIBW, thus variation among the two effects that might have led to a better model fit was not allowed.

The relatively poor ability of adjusted cropland models to predict nitrate discharges for most physiographic provinces may also be due to inappropriate assumptions applied to the flow path metrics. For example, the assumption that nutrient retention was uniform and optimal across all buffers imposed restrictions on site-specific characteristics such as soil moisture, the concentration of nutrients delivered to the buffer, and other complex biological and hydrological interactions (e.g., Hill, 1996; Hill et al., 2000; Vellidis et al., 2003) that may greatly affect buffering potential. By not accommodating fine scale variation in nutrient retention capabilities the adjusted cropland models may have inaccurately portrayed buffer function by overestimating retention, obscuring the relationships between adjusted cropland proportions and nitrate discharges.

In all provinces, the inclusion of development as an additional predictor variable allowed for better model fits which suggests that accounting for urbanization in landscape models is important for understanding buffer function in this dataset. Incised streams,
altered hydrologic flow paths, and impervious surface found in urbanized areas may interact in a way that leads to more nitrogen enriched discharges and a lower near-stream water table (Groffman et al., 2002) that ultimately lead to large nitrogen discharges (Groffman et al., 2004). Previous studies have also emphasized the importance of watershed development contributions to nitrogen discharges in the region (Weller et al., 2003; King et al., 2005). Because watersheds with known point sources were not excluded in the MBSS dataset, incorporating a measure of watershed urbanization may be an important consideration in broad-scale modeling of independent samples.

This analysis should be interpreted as an exploratory tool to investigate general effects of changing stream map resolution on landscape models due to several limitations. First, the grab sampling technique used to obtain nitrate concentrations provided only a single snap shot of nutrient concentrations at a particular site rather than a temporally integrative measure, and nutrient data were collected during different years for different watersheds. Although the NLCD map used in this analysis was selected to best represent the time frame for all nutrient data, it is possible that results from this study could change with more temporally extensive sampling techniques as in-stream nutrient concentrations can be highly variable over space and time (e.g., Spieles and Mitsch, 2000; McClain et al., 2003). In a comparison of landscape models similar to those of this study, Baker et al. (in review) were able to infer the relative retentiveness of riparian buffers in the SERC dataset, but the variability inherent in this dataset may have obscured the ability to draw clear conclusions about whether buffers were relatively retentive or leaky. Also, watersheds in the MBSS dataset were independently selected for the purpose of establishing a state-wide watershed monitoring protocol (see Mercurio et al., 1999). No
effort to exclude watersheds with sewage outfalls or other known pollutant point sources was made, and attaining a representative range of land cover patterns was not a project goal. Without control for such factors, there was an inherent level of variability within the dataset itself. Despite such variability, this study was able to broadly discern the effects of stream map resolution on the ability of different models to predict nitrate-nitrogen concentrations in an independent set of watersheds.

Another limitation with this analysis is in the construction of the FINE stream map itself. Because it was derived from a DEM, any errors from that layer would be propagated to the higher resolution stream map which could result in imprecise channel initiation points and stream locations. Additionally, specific watershed characteristics that may affect where channels begin such as groundwater springs, underground seeps, or karstic terrain may not be well represented by surface topography and would have been overlooked in the creation of topographically-derived stream maps. The ability of FINE-based models to achieve quality data fits in most provinces despite these issues suggests that the amount of source-to-sink connectivity that was captured by the FINE stream map was nevertheless an improvement over connectivity modeled by the NHD stream map. Despite these limitations, this study was well-suited to explore and evaluate the effects of different stream map representations on buffer characterization and prediction of nitrate-nitrogen for a wide variety of watersheds.

**Conclusions**

Because the definition of riparian buffers is dependent on the hydrologic connectivity between nutrient sources and streams, dramatic differences in perceived
ability of buffers to filter nutrients may be observed when different representations of stream channels are used to calculate flow-path metrics. Buffers tended to be narrower and with more gaps, and had more variable mean widths according to finer stream maps. Additionally, a greater proportion of nutrients were expected to reach the streams for every cropland cell when finer stream maps were used. A notable exception was in the Appalachian Mountain province where buffers averaged wider widths according to higher resolution stream maps most likely due to similarities among the two stream map resolutions in this province. Nonetheless, these differences highlight the potential for dramatic variations in perceived filtering ability of buffers when alternative stream maps are used in water quality analyses.

Linear regression models based on higher resolution stream maps improved predictions of nitrate-nitrogen concentrations in the Coastal Plain and Piedmont and were comparable to models based on coarser maps in the Appalachian Mountain province. No regression models using fine stream maps were supported in the Appalachian Plateau. This indicates that the highest resolution stream map may not be appropriate for all regions when linking patterns of watershed land cover to nitrate-nitrogen discharges. Such stream maps may be imprecise representations of hydrologic connectivity, underscoring the importance of carefully considering how source-to-sink connectivity is best modeled for any particular watershed or set of watersheds. Analyses incorporating the most appropriate stream map for the region of interest have the potential to provide robust estimates of nutrient discharges which could potentially aid land managers in restoration efforts.
Literature cited


TABLE 3. Mean and standard deviation for land cover characteristics calculated using two stream map resolutions (NHD, 1:24,000; FINE, finer than 1:24,000) for the MBSS study region, Coastal Plain (CP), Piedmont (PD), Appalachian Mountain (AM) and Appalachian Plateau (AP) physiographic provinces. Bold values indicate a significant difference in mean values between stream maps ($p < 0.5$) using paired t-tests. Note that there was no difference in watershed land cover proportions between the two stream map resolutions.

<table>
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<tr>
<th></th>
<th>MBSS (n=1592)</th>
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<th>AM (n=155)</th>
<th>PD (n=659)</th>
<th>CP (n=635)</th>
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<td>NHD</td>
<td>FINE</td>
<td>NHD</td>
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TABLE 4. Mean and standard deviation for flow-path metrics using NHD and FINE stream maps for the MBSS study area as well as the Coastal Plain (CP), Piedmont (PD), Appalachian Mountain (AM), and Appalachian Plateau (AP) physiographic provinces. Differences between mean values of map resolutions are significant except where indicated by *.

<table>
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<tr>
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<td>NHD</td>
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<td>105</td>
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<td></td>
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TABLE 5. Comparison of models predicting stream nitrate concentrations based on land cover proportions and buffer metrics in watersheds from the Coastal Plain, Piedmont, Appalachian Mountain, and Appalachian Plateau physiographic provinces. Models are compared across stream map resolutions (NHD, 1:24,000; FINE finer than 1:24,000). Linear regression parameters are derived from a 10-fold crossvalidation of each model. Models with $w_i < 0.01$ are not reported. * denotes an insignificant (p > 0.05) coefficient term.
<table>
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<th>Model Parameters</th>
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<th>Intercept</th>
<th>Crp. Coeff.</th>
<th>$R^2$</th>
<th>Adj. $R^2$</th>
<th>AIC</th>
<th>AICc</th>
<th>Δi</th>
<th>$w_i$</th>
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<tr>
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<td>0.68</td>
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<td>n.s.</td>
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<td>0.68</td>
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<td>442.95</td>
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<tr>
<td>Adj. % Cropland (2) + % Development</td>
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<td>0.29</td>
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<td>203.30</td>
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FIGURE 6. A hypothetical example of how changes in stream map representation may affect perceived buffer function within a watershed. Nutrient source areas are located in yellow; potential buffer areas are in green. Stream channels are delineated in blue. In example (a), all nutrient source area appears to be well-buffered, with potential buffer areas being located between the source areas and the streams. However, streams dissect the buffers and reach up into source areas according to an alternative stream channel map (b), thus decreasing the ability for buffers to reduce nutrient loads.
FIGURE 7. The 1,596 watersheds included in the Maryland Biological Stream Survey (MBSS) used in this analysis.
FIGURE 8. Comparison of mean buffer width (a), frequency of gaps (b), and coefficient of variation in buffer width (c) using the NHD (white) and FINE (grey) stream maps across the Appalachian Plateau (AP), Appalachian Mountain (AM), Piedmont (PD) and Coastal Plain (CP) physiographic provinces. Boxed delimit the 25th and 75th percentiles, whiskers the 10th and 90th, and open circles represent outlying observations. Mean values are plotted as solid circles. For graphs (a) and (c), statistical outliers not plotted for clarity.
FIGURE 9. Proportion of cropland adjusted downward to account for the effect of buffers plotted against total watershed cropland. Scatterplots are shown for two stream map resolutions: 1:24,000 (a) and finer than 1:24,000 (b). The dashed line represents a 1:1 relationship, or the expected outcome if accounting for the presence of buffers is no different than whole-watershed proportions of cropland. Solid lines represent linear regression models for each physiographic province.
CHAPTER IV

USING ESTIMATES OF NUTRIENT RETENTION TO INFORM MEASURES OF RIPARIAN BUFFER POTENTIAL

Abstract

Restoration of riparian buffer zones has been a major focus of water-quality initiatives in an effort to reduce the negative effects of nonpoint source pollution. New measures of potential buffer function were developed with the simplifying assumption that buffers filtered nutrients solely as a function of their respective widths along flow pathways from source areas to streams. The purpose of this study was to develop weighted variations of the original measures that incorporated estimates of nutrient retention, and evaluate their ability to predict nitrate-nitrogen concentrations at watershed outlets. Specifically I tested the hypotheses that 1) the ability of buffers to filter nutrients appeared to be reduced according to weighted measures compared to unweighted measures, and 2) models based on weighted measures were better predictors of nitrate-nitrogen concentrations than models using unweighted measures.

I developed weighted measures and used them to describe patterns of land cover in 1,613 watersheds across the state of Maryland. Watersheds appeared to be more well-buffered according to unweighted measures than weighted measures. In the case of some watersheds, differences between weighted and unweighted measures were extreme and dramatically changed the outcome of how well watersheds were perceived to be buffered. Despite the variety of ways to characterize riparian buffers in this study, potential buffering effects were limited at similar proportions of watershed cropland within a given
physiographic province, emphasizing the importance hydrologic connectivity between nutrient sources and streams. Weighted measures showed promise in improving the ability to relate nitrate-nitrogen discharges to measures of riparian function in certain physiographies. The measures developed and tested in this study were an important step towards developing tools to aid restoration and conservation strategies.

**Introduction**

Countering the effects of anthropogenic pollution in an effort to improve water quality has been an important goal for land managers nationwide. Riparian buffers have been a conservation priority in watershed management because of their ability to attenuate nutrients (Dosskey, 2001). In fact, the restoration of stream-side ecosystems has been a major focus of water quality initiatives at the national and local levels, and millions of dollars have been spent annually to reduce the effects of nonpoint source pollution, particularly from agricultural sources (Bernhardt *et al.*, 2005).

Restoration projects focused on reducing the impact of nonpoint source pollution generally have followed one of two approaches. The first simply seeks to increase riparian buffer width which allows for ease of planning, implementation and monitoring (Lee *et al.*, 2004). However, this approach fails to include site-specific attributes that may affect buffer retentiveness and does not consider the spatial configuration of nutrient sources and sinks nor the directional flow pathways that are used for nutrient delivery to streams. The second approach attempts to optimize the spatial configuration of buffers according to a given a suite of topographic, hydrological, land-use or physiographic characteristics (Tomer *et al.*, 2003; Polyakov *et al.*, 2005). For example, Tomer *et al.*
(2005) mapped areas suitable for buffer installation throughout a watershed based on variables such as fine-scale soil characteristics, local slope values and the amount of surface area draining to a particular site. Such geographic analyses are more sophisticated than the previous approach, but still fail to consider spatial configurations of land cover types and their influence on potential nutrient discharges. In fact, successfully linking local characteristics to broader, watershed-scale processes has proven to be a significant challenge (Mayer et al., 2007).

Recently Baker et al. (2006b) proposed a new method of quantifying riparian buffer potential that linked patterns of land cover to the process of nutrient retention in an ecologically meaningful way. In this method, Baker et al. (2006b) used surface flow pathways from upland source areas to streams to define riparian buffers as contiguous areas of forest or wetland adjacent to a stream and along a source-to-stream flow path. In so doing, Baker et al. (2006b) were able to calculate the width of buffer along a flow pathway for any particular unit source area and identify any unbuffered sources. They then aggregated these buffer measures across source areas to quantify land cover patterns. For example, the Mean Inverse Buffer Width (MIBW) has been useful for understanding differences in buffer potential for whole watersheds because it reflects the expected proportion of nutrients reaching streams consistent with transport distance through a buffer (Baker et al., 2006b).

Although the MIBW and other measures were a marked improvement over previous studies and have shown great promise as a tool for land-use managers (Baker et al., 2006b), they were developed with the simplifying assumption that all buffers along a surface flow path had the potential to filter nutrients uniformly and optimally. This
assumption was useful for understanding the impact of buffers on water quality based on a “best case scenario” for buffering filtering potential. For example, Baker et al. (2007) observed that even under the assumed optimal capacity of buffer filtering, very little buffering effect was detected for Coastal Plain watersheds with 25% or more of their area as cropland.

However, it may be unrealistic to assume that riparian buffers filter nutrients uniformly and optimally. Differences in buffer retentiveness are strongly related to complex hydrological and biogeochemical interactions such as varying soil saturation levels and the concentration of nutrients intercepted by the buffer (Hill, 1991; Groffman et al., 1992; Brinson, 1993; Osborne and Kovacic, 1993; Hill, 1996; Hill et al., 2000; Vellidis et al. 2003). It would be reasonable to expect that incorporating some estimate of site-specific nutrient retention into flow-path metrics would improve their precision as a management tool (Gold et al. 2001; Polyakov et al. 2005).

In this study, I developed 5 novel variations of the MIBW that incorporate estimates of nitrate-nitrogen retention (hereafter, “weighted measures”) and contrast them with the original MIBW (hereafter, “unweighted measures”) developed by Baker et al. (2006b). The overall goal of this study was to assess whether accounting for site-specific properties that may influence retention is important for making better predictions of nitrate-nitrogen concentrations from land cover patterns. To accomplish this goal the following hypotheses were tested: 1) the ability of buffers to filter nutrients effectively at the watershed scale appears to be reduced according to weighted measures compared to unweighted measures, and 2) multiple regression models incorporating weighted
measures are better predictors of nitrate-nitrogen discharge than models using
unweighted measures or whole-watershed proportions of cropland.

**Methods**

**Study Area**

A data set of 1,613 watershed sampling sites throughout the state of Maryland
was used for this study (Figure 10). These sites were chosen originally as part of an
extensive biological and physical monitoring network to assess and inventory stream
ecosystems, the Maryland Biological Stream Survey (MBSS; Mercurio et al., 1999). Site
selection for the MBSS was based on a stratified random sampling design according to
major drainage basin and stream order (1st to 3rd order, non-tidal streams on a 1:250,000
stream map).

The dataset comprised four physiographic provinces (Coastal Plain, Piedmont,
Appalachian Mountain, and Appalachian Plateau; Langland et al., 1995). The eastern-
most province, the Coastal Plain, is characterized by watersheds of relatively low relief, a
great deal of agriculture, and wedge-shaped aquifers created by a shallow clay confining
layer overlain by other unconsolidated sediments (Vroblesky and Fleck, 1991). The
Piedmont has gently rolling hills of moderate topography (typically 30 – 100 m in local
relief) that are dissected by dendritic networks of streams (White, 2001). There is a wide
variety of bedrock that underlies the region, some of which is fractured or highly karstic,
allowing for unpredictable groundwater flow and a great deal of variation in stream flow
patterns (White, 2001). Streams are an important geomorphic feature in the Appalachian
Mountain province as they determine the landforms with which they are intimately
related (Fenneman, 1938; Hack, 1965; Keaton et al., 2005). The steep mountain sides constrain the location of streams to a latticework of channels that cut through shallow soils and are often in contact with bedrock (Keaton et al., 2005). Because of the relatively high relief in this province, agricultural land use is typically restricted to fertile valley bottoms along higher order streams (Keaton et al., 2005; Baker et al., 2006b).

Broad ridge tops with steep side slopes caused by folding of sedimentary bedrock define the Appalachian Plateau province, and agricultural land use is typically restricted to the ridge tops (U.S. Geological Survey, 1984; Langland et al., 1995). The long-term average annual precipitation in Maryland is about 43 inches per year (1901-2001), with greater precipitation in the eastern and extreme western parts of the state than in the central region (Wheeler, 2003).

**Nutrient Data**

Grab samples for water chemistry analysis were collected according to the methods outlined in Mercurio et al. (1999). For each watershed, a single grab sample was collected from the stream during the spring index period (March 1 to May 1) on one occasion within a nine-year period from 1999 to 2003. Samples were stored on ice, brought to the laboratory within 48 hours of collection and analyzed for nitrate-nitrogen concentrations (mg/l) using a Dionex 2001i ion chromatograph (Sunnyvale, CA) following EPA standards for water quality analysis (EPA, 1987).

**Geographic Data**

I analyzed publicly available elevation, land cover, and stream channel data sets using ARC/INFO (ESRI, Inc.). Elevation data were obtained from a 30-meter digital

Stream channels were derived from the DEM using a nonparametric deviance reduction method applied to regional slope-contributing area distributions (Hill and Baker, unpublished manuscript). Briefly, the DEM was first preprocessed using a normalized excavation version of the AGREE algorithm in order to correct for stream alignment differences between the DEM and the NHD while minimizing the occurrence of undesirable parallel stream flow pathways and watershed boundary distortions (Baker et al., 2006a). The normalized excavation version of the AGREE algorithm initially lowers the elevation of streams to that of the minimum elevation within a 150m locality, and reconditions the DEM surface to allow for uninterrupted downstream flow (Baker et al., 2006a; Hellweger, 1997). Next, for each physiographic province two slope-contributing area relationships were calculated as the product of local slope and upslope contributing area or contributing area divided by local slope. Each of these distributions was split to maximize deviance reduction in resulting groups of cell values. The group with higher slope-area values was further subdivided two more times to identify raster cells with large and statistically distinct values. Cells with large values for both slope-contributing area relationships were interpreted as likely channel initiation points due to excess erosive power or wetness, respectively (Montgomery and Dietrich, 1988). Streams were identified along flow lines downslope of potential channel initiation points.

Watersheds were delineated from the collection location of water chemistry samples and classified by physiographic province when 80% or more of their area fell
within a province’s boundaries. The 1,613 watersheds ranged in size from 1.17 hectares to 43,116 hectares.

Overview of Weighted and Unweighted Measure Calculation

For each watershed row crop agriculture was identified as nutrient sources. Surface flow paths from source areas to streams were constructed according to topographical constraints. I identified buffers as any contiguous forest or wetland cell located along a flow path and adjacent to the stream. The expected proportion of nutrients reaching the stream for a particular source cell was calculated using both unweighted and weighted distance measures (Figure 11). For unweighted measures, an inverse distance measure assuming optimal and uniform retention for each buffer cell was calculated for each source cell and averaged across all sources. For weighted measures, buffer cells were individually assigned values (decay coefficients) based on ancillary information intended to reflect potential for nutrient attenuation. The proportion of nutrients potentially reaching the stream was then calculated as a function of buffer width and capacity for attenuation using an exponential decay function. For each watershed, the unique buffer-width measures (weighted or unweighted) were aggregated along all flow paths. The end result was 5 weighted and 1 unweighted measure calculations for each watershed (Figure 11).

Calculation of Unweighted Measures

The MIBW was calculated for each watershed following the methods described in Baker et al. (2006b). Briefly, land cover was derived from the 2001 NLCD such that
cropland pixels were identified as sources. Surface flow paths from all source cells to the stream were constructed based on steepest descent (D8 algorithm; O’Callaghan and Mark, 1984). Forest and wetland pixels contiguous along a source-to-stream flow path and adjacent to streams were identified as buffers. For any cropland cell the expected proportion of nutrients reaching the stream, \( t \), was calculated as:

\[
t = \frac{1}{w + 1}
\]  

(1)

where \( w \) was the width of buffer (in meters) along a flow path from the source cell to the stream.

**Calculation of Weighted Measures**

Weighted measures followed the assumption that the nutrient retention of riparian buffers is not uniform, but varied spatially according to controls on constituent transport and biogeochemical removal. For any particular source cell \( t \), the expected proportion of nutrients transmitted to the stream was then calculated as a function of average buffer retention potential along a flow pathway such that:

\[
t = e^{-\left(\sum_{i} r_{i}w_{i}w_{f}w_{f}^{*}\right)}
\]

(2)

where \( r \) was an exponential decay coefficient for buffer cell \( i \) (informed by the retention value from ancillary datasets described in detail below), and \( w \) was the distance in meters traveled, either through a particular buffer cell \( w_{i} \), or the entire source-to-stream flow path \( w_{f} \). This equation reduces to:
such that the proportion of nutrients transmitted to the stream from a particular cropland cell is a function of the decay coefficient and distance traveled through a particular buffer cell, summed over the length of the flow pathway. Equation 3 is an alternative to the inverse distance function, Equation 1, that has been used in previous studies to model decreases in source influence with distance (e.g., Johnson et al., 2007; Soranno et al., 1996; Van Sickle and Johnson, 2008).

Because the range of \( r \) values varied widely among retention estimate methods, \( r \) values for each method were scaled to a range of literature values before use in the weighted distance equation (Equation 3). A range of nitrate-nitrogen retention decay coefficients was calculated based on a review of existing transect-scale field studies. To be included in this analysis, research must have been conducted at physiographically similar sites as those used in this study (Appendix B). For each study the proportion of nitrate-nitrogen remaining was plotted against distance traveled through a buffer, and a curve following Equation 3 was fitted to obtain a range of \( r \) values (Figure 12). This range of values was then used to scale the raw values of \( r \) from retention estimates #1, 2, 3, and 5 described below such that they would vary from 0 to the maximum literature value. In the case of retention estimate #4 (an average of estimates 1 – 3), \( r \) values were scaled prior to the calculation of this estimate thus no subsequent scaling was necessary (see Figure 11).
Retention Estimates for Weighted Measures

Estimates of nutrient retention were based on two factors believed to influence buffer effectiveness: soil wetness and degree of cropland loading. Riparian nitrogen attenuation is most efficient in the shallow subsurface when groundwater interacts with the biologically active zone (Mayer et al., 2007). Here the high water table is in contact with organically rich soils, microbial communities and plant roots allowing for uptake, assimilation, chemical transformations and denitrification that reduce nutrient loads to streams (Groffman et al., 1992; Groffman et al., 1996; Gilliam et al., 1997). Wetlands in particular are likely to exhibit greater nutrient retention relative to other features in the landscape due to anoxic conditions created by a high water table and carbon-rich soils (Johnson et al., 2001; Verhoeven et al., 2006; Zedler, 2003). The second factor, degree of cropland loading, acknowledges that buffers may respond differently to varying levels of cropland contributions and that buffers may experience decreases in their efficiency under greater loadings (Dillaha et al., 1989; Blackwell et al., 1999). The retention estimates described below attempt to relate either soil wetness (estimates 1 – 4) or nutrient loading (estimate 5) to riparian buffer function in a spatially explicit way.

Retention Estimate 1: Topographic Index

I approximated relative soil saturation connectivity by calculating a topographic index (Beven and Kirkby, 1979; Quinn et al., 1991), which is an estimate of relative saturation level during a rainfall event. The topographic index was calculated according to the equation:

\[ \omega = \ln \left( \frac{\alpha}{\tan(\beta)} \right) \]
where \( a \) is the specific contributing area and \( \tan(\beta) \) is the slope. Higher index values indicate a greater likelihood of relative saturation and are typically found in flat areas with large upslope contributing areas. According to this index, buffers located within relatively wet areas (i.e., a higher topographic index value) were expected to retain a greater fraction of their nutrient loads. Due to the aggregative nature of flow-path measures, effective buffering may occur when buffer cells have high topographic index values, but also when buffer width is great enough to accommodate cropland contributions despite inherently low topographic values. For each physiographic province, index values were scaled relative to the maximum and minimum found in that province to account for terrain differences among provinces.

Retention Estimate 2: Wetlands Designation

Buffer retention was modified according to a rule-based function that incorporated information from the 1:24,000 National Wetlands Inventory polygons (NWI; U.S. Fish and Wildlife Service, http://www.fws.gov.nwi) and the 2001 NLCD. All NWI polygons except marine and estuarine wetlands and deepwater habitats were converted to raster format and were used in this analysis. Each buffer cell was then evaluated for land cover type according to the schema in Appendix C with the NLCD weighting more heavily than the NWI because of time-scale relevance. Cells not meeting previous requirements for functional buffers according to the flow-path metrics, despite being classified as wetlands according to the NWI, were excluded from the analysis to maintain consistency among measures. This analysis resulted in the creation of a raster where for wet buffer cells received decay coefficients that reflected the likelihood of
wetland designation; large coefficients reflected a greater probability of wetland occurrence and therefore more retentive buffers.

Retention Estimate 3: Normalized Difference Wetness Index

Remotely sensed satellite imagery was used to calculate the normalized difference wetness index (NDWI), a measure of vegetative liquid water that is also influenced by soil moisture (Gao 1996). The NDWI is calculated using the following formula:

\[
NDWI = \frac{\text{near-infrared} - \text{mid-infrared}}{\text{near-infrared} + \text{mid-infrared}}
\]  

where near-infrared corresponds to Landsat-7/EMT+ band 4 and mid-infrared corresponds to band 5. Values of NDWI range from -1 to 1 with more positive values indicating higher moisture content. Soils lacking vegetative cover and/or have no moisture tend to result in negative NDWI values.

Landsat-7/ETM+ images were obtained from the USGS Global Visualization Viewer download site (online reference – http://glovis.usgs.gov). Eight scenes covering the study area and spanning the late-summer/early-fall months from 1999 to 2002 (Appendix D) were combined into a single NDWI 30-m raster layer for use in this analysis.
Retention Estimate 4: Average of Estimates

An integrative index was also calculated based on the three previous nutrient retention estimates by averaging literature-scaled topographic index, wetland designation, and NDWI values for each buffer cell. Because nutrient removal efficiency depends on multiple biogeochemical, hydrological, and ecological controls (Lowrance et al., 1997; Hill et al., 2000; Gold et al., 2001; Vidon and Hill, 2004a), such an estimate of nutrient retention may be a useful indicator of riparian buffer function.

Retention Estimate 5: Loading Effects

To estimate decreases in buffer retention due to the degree of cropland loading, buffer cells were expected to retain nutrients as a function of the amount of contributing sources so that buffer retention decreased as relative cropland contributions increased. In this modification, the retention for each buffer cell was calculated as $1 / \#$ cropland cells contributing to that buffer cell. Thus, buffer retentiveness would be highest when only a single cropland cell contributed to any length of buffer and all buffer cells in that flow path would be assigned the maximum value decay coefficient. Decay coefficient values would decrease with increasing contributing cropland, reflecting the potential for saturation with large source loads. This equation was used because it is the simplest mathematical model to represent loading and to explore this relationship.

Under certain topographic conditions such as steep terrain or when contributing areas are large in relatively flat regions, channelized flow paths, or gullies, are likely to form (Montgomery and Dietrich, 1988; 1989; 1992). Riparian buffers are ineffective at reducing nutrient loads, particularly nitrate nitrogen, from channelized flow as nutrients
bypass the underground biogeochemically active zone (Vellidis et al., 2003; Vidon and Hill, 2004b). I used the slope-area deviance reduction method (Hill and Baker, unpublished manuscript) to identify locations where the geomorphic relationship of \( \ln(\text{slope} \times \text{area}) \) predicted channelized flow was likely to occur. Buffer cells located downslope of gully initiation sites according to this method were assumed to be completely unretentive and were removed from the analysis.

*Adjusted Cropland Calculations*

For each watershed, the unique buffer-width measures (weighted or unweighted) were aggregated to calculate measures for whole watersheds. The Mean Inverse Buffer Width (MIBW) was calculated by averaging the unweighted buffer width measure (Equation 1) over all cropland pixels for every watershed. The MIBW is a measure of the presumed decrease in cropland effect with distance traveled through a buffer following an inverse decay function. An adjusted proportion of cropland (MIBW Cropland) was then calculated by incorporating the MIBW as an inverse distance weight, essentially adjusting proportions of watershed cropland downward to represent the potential reduction in nutrient delivery to the stream due to the effect of estimated riparian buffer filtering. A second adjusted cropland measure, Gap Cropland, was calculated as the proportion of cropland that enters the stream through gaps. Gap Cropland was based on the assumption that *any* transport through a buffer resulted in complete nutrient retention (sensu Baker et al., in review).

Similar measures were calculated using the weighted buffer width measures. Each weighted measure was averaged over all cropland cells to calculate mean
topographic index, wetland, NDWI, average and loading measure for each watershed. These aggregated measures are computationally similar to the MIBW and should be viewed as a variation of such. Throughout the rest of this paper, these measures will be referred to as TOPO, WET, NDWI, AVEWET, and LOAD. These measures were used to adjust the proportion of cropland to represent the amount of effective cropland in the watersheds after accounting for buffer function as modeled by the five measure variations. Similar to MIBW Cropland from above, these cropland measures will be referred to as TOPO Cropland, WET Cropland, NDWI Cropland, AVEWET Cropland and LOAD Cropland throughout the rest of this paper.

Quantitative Analysis

To understand broad patterns of cropland distribution, summary statistics were calculated for the adjusted and unadjusted cropland proportions across provinces. Mean values of measures were compared in each province using a one-way ANOVA with the null hypothesis that there was no significant difference in mean values. If a significant difference was detected (p-value < 0.05), pair-wise comparisons were made using Tukey’s Honest Significance Difference statistic (Zar, 1999).

The use of weighted and unweighted measures may influence the detection of threshold responses in ecosystem function such that buffers may appear to have little or no effect on nitrate-nitrogen concentrations at certain proportions of watershed cropland. For each adjusted cropland measure, a non-parametric change-point analysis (nCPA; Qian et al., 2003) was used to detect changes in apparent buffer function with patterns of watershed cropland proportions in every physiographic province. This analysis partitions
the dataset into two parts according to variable x (in this case, whole-watershed cropland proportions) in such a way as to minimize within-group variation, and uses a bootstrap technique to estimate uncertainty associated with the partitioning (Baker et al., 2007).

Eight linear regression models were built to compare the relative utility of weighted and unweighted measures in predicting nitrate-nitrogen concentrations. For each model, either whole-watershed cropland or one of the seven adjusted cropland proportions was used as an independent variable to predict nitrate-nitrogen concentrations at watershed outlets. The eight regression models were compared using two methods: a) Akaike Information Criteria, adjusted using a second-order correction to account for small sample size relative to the amount of model parameters (AICc), and b) adjusted coefficients of determination ($R^2_{adj}$).

In the AICc approach, the quality of candidate regression models is compared based on a balance of model parsimony and unexplained variance using a calculated AICc score (Burnham and Anderson, 2002). For any set of models, the model with the smallest AICc score is considered to be of highest quality. Direct comparisons between the highest quality model and any other candidate may be estimated by the difference in AICc scores (“AICc differences”, $\Delta_i$) where $\Delta_i < 2$ suggests comparable models, and models with AICc differences greater than 10 have virtually no empirical support. In addition, AICc weights ($w_i$), a rescaling of AICc scores, were calculated and may be interpreted as the relative likelihood that a particular model is the most appropriate given the data.
Regression model parameters and adjusted coefficients of determination were calculated using a 10-fold cross-validation method (Fielding and Bell, 1997). Data were randomly partitioned into 10 equal subsets from which a training dataset, consisting of 9 of the 10 subsets, and a test dataset, the remaining 10% of the data, were constructed. Regression models were fitted to the training dataset and error rates of the model fit were obtained by subsequently applying the model to the test dataset. The process of fitting and validating was repeated 10 times, each time withholding a different data subset as the test dataset. Model parameters and error rates were averaged to obtain the reported mean values. Regression models that accounted for higher explained variance as measured by the adjusted coefficient of determination were interpreted as being of higher quality than other models.

Results

Comparison of Cropland Adjusted by Weighted and Unweighted Measures

Proportions of cropland adjusted by weighted and unweighted measures ranked similarly across all provinces with TOPO Cropland predicting the greatest amount of effective cropland and Gap Cropland the least amount of cropland (Figure 13). On average TOPO Cropland means were nearly 50% greater than average Gap Cropland values, but this varied greatly by province. The greatest difference between TOPO Cropland and Gap Cropland means was in the Appalachian Plateau province where cropland adjusted using the TOPO measure was 128% larger than Gap Cropland. In all provinces LOAD, AVEWET, WET, NDWI, and MIBW Cropland reliably ranked from highest to lowest but with many cropland proportions having statistically similar means
in each province (Table 7). As expected, the greatest proportions of cropland were measured by the unadjusted percentages of cropland (whole-watershed cropland) in any province (Figure 13).

Although mean values for any cropland proportion were highest in Coastal Plain due to positive skew, median values were highest in the Piedmont province (Figure 13). The Appalachian provinces had lower mean and median values than the other provinces with Appalachian Plateau consistently having the smallest values (Figure 13). The Appalachian Plateau province had the greatest range of mean cropland values with Gap Cropland having a mean value 29% smaller than whole-watershed cropland value, though cropland covered a very small areal extent in this province overall.

In a multiple pair-wise comparison, LOAD Cropland and TOPO Cropland were always significantly different from the MIBW Cropland in each province except for the Appalachian Mountain province where no cropland proportions were statistically different from each other (Table 6). Though not significantly different from each other, LOAD and TOPO Cropland were both statistically different from Gap Cropland and NDWI Cropland in the Coastal Plain and Piedmont. All significant relationships observed in the Coastal Plain and Piedmont provinces were sensitive to sample size (Appendix E).

As expected, cropland proportions adjusted by weighted measures were greater than MIBW Cropland regardless of physiographic province for nearly all watersheds (Figure 14). When this relationship was not true, differences between weighted measures and the MIBW were negligible (<0.04) and most likely attributable to rounding error. Absolute differences in cropland proportions adjusted by weighted and unweighted
measures tended to be relatively small, though some watersheds saw extreme changes in cropland proportions. On average greater differences tended to occur in TOPO Cropland vs. MIBW Cropland and LOAD Cropland vs. MIBW Cropland comparisons with the greatest mean change in cropland proportion occurring in the Coastal Plain.

Watersheds exhibiting dramatic value shifts using weighted measures, particularly for the Coastal Plain and Piedmont provinces, were typically small with relatively little cropland area (<3%), though there were exceptions to this observation.

*Non-parametric Change-point Analysis*

Patterns of potential threshold response to increases in watershed cropland proportions were similar using the six measures (Table 7). Within any given province all measures had similar observed change point values with the exception of the Appalachian Mountain province where MIBW and NDWI had higher thresholds than other values that were outside of the confidence intervals of the other measures. Despite overall similarities among measures, LOAD had smaller mean bootstrap estimates and more limited confidence bounds than other measures in the Piedmont and Appalachian Mountain provinces, but a larger estimate and confidence interval in the Coastal Plain. LOAD also predicted thresholds at smaller amounts of cropland than other measures in the Piedmont but this pattern was not observed consistently in other provinces (Table 7; Figure 15). However, these differences were small and were within the bounds of the confidence intervals of other measures (Table 7). In the Appalachian Mountain and Piedmont provinces, greater observed change point values and bootstrap estimates coincided with larger confidence intervals.
Observed change point values varied by physiographic province (Table 7). Threshold responses were observed at greater proportions of watershed cropland in the Coastal Plain and Piedmont than in either Appalachian province. In fact, observed change point values in the Appalachian Mountain province were approximately an order of magnitude smaller than those of the Coastal Plain. Such sensitivity to increases in cropland proportions reflects the overall low proportions of cropland in the Appalachian provinces in addition to the spatial patterns of land cover observed in these provinces (Baker et al., 2006b).

Regression Model Comparison

Out of the set of candidate models for the Coastal Plain, the model using TOPO-adjusted cropland was clearly the best supported (Table 8). The evidence ratio \( \frac{w_1}{w_2} \) comparing the first- and second-ranked models was 5.64, suggesting the TOPO Cropland model is more than 5 times as likely then the second-ranked MIBW Cropland model as the best model and therefore all other lower ranked models as well. The TOPO model also had the highest explained variance out of all other candidates, though it was a modest improvement (1 - 5% increase of \( R^2_{adj} \)). Despite the similarities in regression parameters among all models, incorporating estimates of retention based on the topographic index substantially improved model performance compared to previous methods using unweighted measures.

In contrast, no model based on informed retention estimates appeared to be the best selection from the set of candidate models in the Piedmont province. The models receiving the most support according to AIC differences scores were MIBW Cropland
and Gap Cropland ($\Delta_i < 0.4$), each having a 45% or 37% probability of being the most appropriate model for the data, respectively. NDWI Cropland, WET Cropland, and AVEWET Cropland were ranked lower and with substantially less support, but LOAD Cropland, TOPO Cropland and whole-watershed cropland had essentially no empirical support ($\Delta_i > 10$).

Model selection was also ambiguous for the Appalachian provinces, though models using whole-watershed cropland had considerably less support ($\Delta_i > 4$) and were ranked among the lowest. In the Appalachian Mountain province, models incorporating estimates of nutrient retention were ranked among the most supported, all of which had AIC difference values of less than 2 but had a high degree of uncertainty associated with their selection ($w_i < 0.25$). For example, LOAD Cropland had the lowest AICc scores out of all candidate models and explained the greatest amount of variance, but the evidence ratio was only 1.1. In the Appalachian Plateau, models using MIBW Cropland and Gap Cropland were clearly the best selections with all other models receiving essentially no empirical support ($\Delta_i > 25$). Models using whole-watershed cropland as a predictor were consistently the least supported by AICc analysis and had the least amount of explained variance for any physiographic province.

**Discussion**

As expected, weighted measures predicted decreases in riparian buffer potential than unweighted measures regardless of physiographic province. Because of the underlying assumption that any transport through a buffer would result in a complete nutrient reduction, it was not surprising that Gap Cropland consistently predicted the
lowest proportions of effective cropland. LOAD Cropland was expected to be the most similar to whole-watershed proportions of cropland in every province because its functionally different construction excluded wet cells which would otherwise be included in other measures. Across provinces mean values of MIBW Cropland were always less than LOAD and TOPO Cropland, and in some provinces, significantly different from other adjusted cropland proportions as well. Greater proportions of nutrients reaching the streams were also predicted by the weighted measures TOPO, WET, NDWI, and AVEWET compared to the MIBW, though these measures were not as statistically distinct from each other or from either MIBW or LOAD.

Mean values of measures ranked similarly across provinces and likely stems from slight but systematic differences in methods for calculating retention estimates. For example, the distribution of decay coefficient values in the WET measure resulted from overall prevalence of land cover types in the larger datasets, even after limiting the analysis to buffers located along flow-paths: NLCD forest was relatively more common than NLCD wetland (43% vs. 2% of total area), and NWI wetlands were limited in areal extent (4% of total area). Therefore, the probability of a particular buffer cell to be identified as forest by the NLCD and non-wetland by the NWI (i.e., the combination modeled to have the lowest retention rates) was greater than other possible forest-wetland combinations for any buffer cell, and far more likely than the combination of NLCD wetland and NWI wetland, the combination assumed to be the most retentive. As a result, WET tended to characterize buffers as being least effective than most other weighted measures, especially MIBW.
In some watersheds, incorporation of retention estimates dramatically changed the outcome of how well the watershed was buffered. Small watersheds tended to reflect greater differences between weighted and unweighted measure values than other watersheds. For example, a 43.3 ha Coastal Plain watershed with 80% of its land cover as cropland saw the greatest variation in adjusted cropland values of any watershed, ranging from 29% (WET Cropland) to 76% (LOAD Cropland). Statistical outliers were consistently the same watersheds across adjusted cropland comparisons and appeared to be more sensitive to both WET, which uses categorical coefficient assignments, and LOAD, which excludes buffers where concentrated flow is likely to occur.

Differences in threshold responses to increases in cropland according to weighted and unweighted measures were minimal suggesting that despite the variety of ways to characterize riparian buffers, the filtering capacity of buffers is limited at similar proportions of cropland and that characterizing retention may not be as important as ensuring source-sink connectivity. These findings are consistent with Baker et al. (2007) who simulated relationships between relative nutrient retention and hydrologic connectivity and found that watershed-scale effects of buffer filtering capacity were more influenced by hydrologic connectivity of cropland to potential buffers. Additionally, thresholds were similar not only within physiographic provinces, but across the Coastal Plain and Piedmont provinces where similar thresholds of approximately 20% cropland were observed. A buffer’s ability to effectively filter nutrients with noticeable impact at the watershed scale was less at lower proportions of cropland according to LOAD than that of other measures in the Piedmont and Appalachian Mountain regions. Although all threshold relationships were observed at less than 10% cropland, the relatively large
differences in threshold values between measures in the Appalachian Mountain province emphasize that the perception of how well-buffered a watershed is may change according to how retention is characterized.

The incorporation of retention estimates improved the ability to relate nitrate-nitrogen concentrations to measures of riparian buffer function in the Coastal Plain province. The regression model based on the topographic index was decidedly the best fit in this region despite statistical similarities of TOPO Cropland with other cropland measures. Based on previous observations that Coastal Plain buffers are highly retentive (Peterjohn and Correll, 1984; Jordan et al., 1993; Lowrance et al., 1997; Baker et al., in prep), I would expect regression models using MIBW Cropland or Gap Cropland to outperform other models because such cropland measures assume that buffers were highly retentive. However, these measures were clearly not as well-supported in the analysis of Coastal Plain watersheds. The Coastal Plain is characterized by unconsolidated sediments and relatively low relief (Ator and Denis, 1997) which would promote saturation excess flow, the hydrologic transport mechanism captured by the Topographic Index calculation. The ability of regression model using TOPO Cropland as an independent variable to predict nutrient discharge in this province may reflect the importance of saturation excess flow in the region.

Regression models using cropland adjusted by weighted measures out-performed MIBW Cropland and Gap Cropland in the Appalachian Mountain province though there was a large amount of uncertainty surrounding the selection of the best model. Average buffer widths in this region (17.7m) were much greater than in the Coastal Plain (6.9m) or Piedmont provinces (2.5m) where model selection was clearer, which would make
weighted measures less sensitive to changes in retention values simply because of the mathematical relationship between buffer width and the retention coefficients. That is, because the weighted decay function used in this analysis tended toward 0 at large proportions of cropland, wide buffers would appear to be retentive simply because of the greater distance nutrients have to travel even if informed estimates indicated the buffers were leaky (Weller et al., 1998). Additionally, the limited extent of cropland in Appalachian Mountain watersheds reduced the potential number of flow pathways and thus the potential for a variety of filtering capacity scenarios which may have favored one adjusted cropland regression model over another.

In the Piedmont and Appalachian Plateau the regression model that accentuated the importance of gap frequency (Gap Cropland) and the model which assumed uniform and maximum retention were of greater quality than other candidates, with regression models incorporating retention estimates receiving considerably less support. It is possible that weighted measures were unable to account for the high degree of variable buffer filtering capacity that has been documented in the Piedmont (Jordan et al., 1997; Baker et al., in prep). The ability of MIBW Cropland and Gap Cropland to successfully predict nitrate discharge relative to other models may attest more towards the value of models with simple assumptions over more complex models. The potential for error propagation in weighted measures was greater than that of unweighted measures which may have influenced the ability of more complex models to effectively characterize buffer retention. For example, weighted measures would include or even amplify the errors of ancillary datasets used to derive retention estimates or even the bias associated with the field studies used for scaling while unweighted flow-path measures did not
comprise such factors. Mean buffer widths in the Appalachian Plateau were the widest of any province (20.1 m) and therefore relatively insensitive to weighted measures. At such buffer widths, nutrient filtering capacity would appear to be maximal simply due to the transport distance alone, thus because MIBW and Gap Cropland models assume uniform and maximum retention by buffers would appear to be more appropriate than other models.

In this analysis it was important to consider the quality of the data used as inputs. For example, classification errors in land-cover maps may be propagated through landscape measures leading to mistaken conclusions, with potentially serious ecological and financial consequences if management decisions are based such conclusions (Gergel et al., 2007; Weller et al., 2003). To achieve a reasonable level of confidence in the measures’ ability to describe buffer function, all land-cover maps used in the analysis, including the NLCD and Landsat imagery, were matched temporally to the MBSS nitrate discharge data as much as possible. Ideally a composite of multiple images per scene area from spring and early summer months would be used to calculate the NDWI to improve index accuracy (Metzler and Sader, 2005), but I was restricted by data availability during this analysis to only one image per scene area.

The spatial resolution of the data also must be considered as changes in grain size may greatly impact land-cover distributions (Hollenhorst et al., 2006) and perceived connectivity of sources to streams (Baker et al., 2007). In this study, the aggregative step of averaging buffer widths over all flow paths for an entire watershed would undoubtedly include unbuffered flow pathways and thus skew average buffer widths. The minimum width for the detection of riparian buffers was one cell width, or 30 meters.
calculating weighted and unweighted flow-path measures, the aggregative step of averaging buffer widths over all flow pathways would undoubtedly include unbuffered flow pathways. The estimates of retention may have limited the ability of weighted measures to represent riparian function. For example, in conjunction with other indices derived from remotely-sensed data the NDWI has been used to identify open water features (e.g., Zou et al., 2006), detect change in forest composition (e.g., Wilson and Sader, 2002), and delineate wetlands (e.g., Sader et al., 1995; Li and Chen, 2005), but is not a measure of soil moisture alone. The ability of the index to represent soil moisture is obscured when canopy cover is present, in which case the index may be more representative of vegetation types rather than soil moisture (Gao, 1996). It is therefore difficult to attribute any observed relationship between NDWI and nitrate discharges to soil moisture alone.

Compared to the NDWI, the topographic index is a more robust and interpretable estimate of relative soil saturation that has been used in previous watershed-scale studies to identify sites likely to intercept nutrients (Moore and Grayson, 1991; Tomer et al., 2003; Tomer et al., 2005), or to identify streamside areas that may experience flooding disturbances (O’Neill et al., 1997; Russell et al., 1997). The application of the topographic index in any study assumes that a) for any area, uniform subsurface runoff represents the fluctuating water table, and b) the slope of the surface topography reflects the hydraulic gradient of subsurface flow (Beven and Kirkby, 1979; Beven, 1997). The broad application of the topographic index in this study has the potential to violate at least one of the assumptions in certain watersheds, particularly in parts of the Piedmont where patterns of subsurface flow may not relate to surface topography because of karstic
terrain. The fact that TOPO Cropland models were ill-suited for modeling nitrate discharges in this province may be partially due to the topographical index’s inability to account for alternative forms of subsurface flow that are not governed by surface topography.

The results from this study underscore the need to ensure buffers’ hydrologic connectivity in order to observe water quality improvement. The similar ranking of measure mean values across provinces despite physiographic differences in land-cover patterns suggests that hydrologic connectivity may be more important than estimates of nutrient attenuation in many watersheds. Additionally, the significant differences in perceived buffer effectiveness according to LOAD when compared to other measures is evidence of the profound effects that can occur when buffers become hydrologically disconnected. In an analysis examining the interaction of connectivity and relative retentiveness, Baker et al. (2007) found that the watershed-scale effects of buffer filtering capacity were more influenced by hydrologic connectivity rather than relative retention. Weller et al. (1998) made similar conclusions from simulations using hypothetical watersheds.

Conclusions

The purpose of this study was to explore the utility of flow-path measures that incorporated estimates of nutrient retention as a means of quantifying watershed-scale effect of riparian buffer function. These results suggest that nutrient retention estimates may be an important consideration when modeling riparian buffer function, especially areas such as the Coastal Plain. Weighted measures tended to provide a more
conservative estimate of riparian buffer function, and in the case of some watersheds
differences between weighted and unweighted measures were extreme. Such sensitivity
suggests that weighted measures may be more appropriate in watersheds with adequate
proportions of source area to eliminate bias resulting from measure reliance on too few
buffer cells.

The ability of a buffer to have a watershed-scale effect on water quality occurred
at lower proportions of watershed cropland using weighted measures. Watershed
development plans guided by measures assuming maximum and uniform retention may
be overestimating the filtering effects of stream-side forests and wetlands. To err in favor
of safety, land-use planners may wish to employ a measure that does not rely on a “best
case scenario” but rather attempts to account for heterogeneous levels of buffer
efficiency, or obtain a range of watershed protection uncertainty by using several
different measures of retention potential.

Retention-informed measures show promise in improving the ability to relate
nitrate-nitrogen discharges to measures of riparian function, especially in certain regions
such as the Coastal Plain. This suggests that there are particular locations within a
watershed that are inherently well-suited as effective filtering sites than others, and that
these places are likely to yield the most benefit to water quality if restored. Analyses
incorporating retention estimates may be used to identify such areas and could potentially
save time and money in restoration efforts. These measures are an important step
towards the development of an efficient and effective management tool to aid land-use
planners.
Literature Cited


TABLE 6. Results from a one-way ANOVA and p-values from subsequent multiple pairwise comparisons between the proportions of cropland adjusted by MIBW, TOPO, WET, NDWI, AVEWET and LOAD as well as Gap Cropland. Bolder values are significant at the 0.05 level.
<table>
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<th>Region</th>
<th>MIBW Cropland</th>
<th>Gap Cropland</th>
<th>TOPO Cropland</th>
<th>WET Cropland</th>
<th>MDWI Cropland</th>
<th>AVEWET Cropland</th>
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<td>&lt;0.001</td>
<td>&lt;0.001</td>
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</tr>
<tr>
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<td>&lt;0.001</td>
<td>WET Cropland</td>
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<td>&lt;0.001</td>
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<td>1.000</td>
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TABLE 7. Results from non-parametric change-point analysis of unweighted (MIBW) and weighted (TOPO, WET, NDWI, AVEWET, LOAD) measures as a function of percent watershed cropland within four physiographic provinces. All relationships are significant to \( p < 0.001 \).

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<td>18.9</td>
<td>18.9</td>
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<th>App. Plateau (145) *</th>
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<td>TOPO</td>
<td>WET</td>
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<tr>
<td>Observed Change Point</td>
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<td>&lt;0.1</td>
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<td>Upper Bound (95% CI)</td>
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*Due to limited range of cropland proportions in this province, change-point analysis results should be considered unreliable.
TABLE 8. Comparison of models predicting stream nitrate concentrations based on land cover proportions and buffer measures in watersheds from four physiographic provinces. Linear regression parameters result from a 10-fold cross validation of each model. Models more strongly supported using AICc comparisons ($\Delta_i < 2$) are in bold. For all AICc models, $K = 3$. 
<table>
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<th>$R^2$</th>
<th>Adj. $R^2$</th>
<th>AIC</th>
<th>AICc</th>
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<th>$w_i$</th>
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<td>2535.09</td>
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<td>2540.62</td>
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<td>0.05</td>
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<td>0.396</td>
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<td>2543.44</td>
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<td>0.398</td>
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<td>0.368</td>
<td>0.367</td>
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<td>2577.81</td>
<td>42.72</td>
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<td>0.364</td>
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<td>0.260</td>
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### Appalachian Mountain (n=162)

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<th>TOPO</th>
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<th>MIBW</th>
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<td>491.70</td>
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<td>495.73</td>
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<td>0.04</td>
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### Appalachian Plateau (n=145)

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<th>Gap</th>
<th>MIBW</th>
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FIGURE 10. The 1,613 watersheds included in the Maryland Biological Stream Survey (MBSS) used in this analysis.
FIGURE 11. Schematic of methodology used in this study for each watershed. See text for detailed explanation.
FIGURE 12. Plot of exponential decay functions fitted to observed patterns of nitrate-nitrogen transport through riparian buffers. Field studies were conducted within the Coastal Plain (Peterjohn and Correll, 1984; Dillaha et al., 1989; Lowrance, 1992, Jordan et al., 1993; Hubbard and Lowrance, 1997; Dukes et al., 2002), Piedmont (Daniels and Gilliam, 1996), and Appalachian Mountain (Dillaha et al., 1989) physiographic provinces.
FIGURE 13. The distribution of unadjusted (% Crop) and adjusted (unweighted: MIBW Crop, Gap Crop; weighted: TOPO Crop, WET Crop, NDWI Crop, AVEWET Crop, LOAD Crop) proportions of cropland for each physiographic province. Boxes delimit the 25th and 75th percentiles, whiskers the 10th and 90th, and solid circles represent mean values. Statistical outliers are not plotted for clarity.
FIGURE 14. Comparison of TOPO Crop (a), WET Crop (b), NDWI Crop (c), AVEWET Crop (d), and LOAD Crop (e) to MIBW Crop. Points plotted above the dashed 1:1 line are watersheds that have a greater proportion of nutrients potentially reaching the stream according to weighted measures than MIBW.
FIGURE 15. Change-point analysis for each physiographic province showing the cumulative probability of a threshold according to MIBW (black), TOPO (blue), WET (green), NDWI (red), AVEWET (yellow), and LOAD (grey). Lines are representative of the uncertainty associated with change-point estimation. Note logged x-axes for Appalachian provinces.
CHAPTER V
CONCLUSION

The purpose of this study was to gain a deeper understanding of the potential for riparian buffer quantifications derived from flow-path analysis to augment and inform watershed management decisions. Chapters II through IV explored three concepts foundational to flow-path analysis—aggregation, connectivity, and retention—in order to better understand the potential for ecologically meaningful metrics to be incorporated into future watershed management tools. Specifically, this study addressed 1) how land cover patterns that were described by flow-path analysis compared to whole-watershed and fixed-distance measures, 2) how different representations of source-to-stream connectivity altered the ability of buffer quantifications from flow-path analysis to predict stream nitrate-nitrogen concentrations, and 3) how accounting for site-specific retention within flow-path analysis affected the relationship between nitrate-nitrogen concentrations and patterns of land cover.

The results of Chapter II demonstrated the flexibility and efficiency in characterizing riparian buffer function through flow-path analysis for a wide range of land cover distributions and physiographic regions. While remaining sensitive to regional patterns of land cover, buffer quantifications using flow-path analysis were relatively independent from patterns of whole-watershed land cover or land cover within fixed distances of streams. This suggested that statistical models relating in-stream nutrient concentrations to land cover patterns could include flow-path measures as additional predictor variables without violating assumptions of statistical independence.
Perhaps even more importantly, flow-path measures of riparian buffers demonstrated the potential to improve such statistical models because they provided implicit and novel information that was not captured using gross land cover proportions.

Because riparian buffers were hydrologically defined using flow-path analysis, Chapter III explored the effects of alternative stream channel maps on buffer characterizations and measures of filtering potential. According to finer stream maps, buffers tended to be narrower, less longitudinally continuous and more variable mean widths. Such differences in buffer characterization between stream maps emphasized that our perception of how well a watershed may be buffered depended on the specific stream map used in flow-path analysis. A comparison of regression models predicting in-stream nitrate concentrations from land cover patterns suggested that finer stream map resolutions may not be appropriate for all physiographic regions. However, analyses incorporating the most appropriate stream map for the region of interest had the potential to provide robust estimates of nutrient discharges. This result underscored that before flow-path analyses are implemented, users should carefully consider how source-sink connectivity is best represented for any particular watershed or set of watersheds.

Although buffer quantifications using flow-path analysis were shown to better predict in-stream nitrate concentrations than gross land cover proportions, the results of Chapter III suggested that there was room to improve measures of potential buffer function. Therefore the goal of Chapter IV was to explore the utility of buffer measures that incorporated various estimates of nutrient retention from ancillary datasets as a means of quantifying watershed-scale filtering effects. Incorporating site-specific characteristics that could influence retention improved statistical models relating land
cover patterns to nutrient discharges, but only in certain areas. Furthermore, the type of retention estimate that improved nitrate predictions the most varied by physiographic province. In general, retention-informed metrics showed promise in improving the ability to relate nitrate-nitrogen discharges to measures of riparian function, especially in certain regions such as the Coastal Plain.

The more detailed understanding of flow-path measures gained from this study may help the development of spatial tools to aid land-use planning. Flow-path measures of buffer potential may be useful in statistical modeling for a broad range of watershed types and land cover distributions, particularly when used in concert with appropriate stream channel datasets. Additionally, incorporating estimates of nutrient retention into flow-path analyses identified particular locations within a watershed likely to be inherently well-suited as effective filtering sites. By using processes similar to those described in this analysis, watershed planners may identify sites that may yield the most benefit to water quality if restored. Analyses incorporating retention estimates may be used to identify such areas and could potentially save time and money in restoration efforts. This study provided an important step towards the development of an efficient and effective management tool to aid land-use planners.
APPENDIX A. Confusion matrix that describes the average proportion of classification change using the 1992 NLCD compared to 2001 NLCD for MBSS study region. For example, 69.0% of 1992 NLCD cropland cells remained as cropland cells according to the 2001 NLCD, while 10.0% were later classified as forest+wetland.

<table>
<thead>
<tr>
<th>1992 NLCD</th>
<th>2001 NLCD</th>
<th></th>
<th></th>
<th></th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cropland</td>
<td>0.690</td>
<td>0.100</td>
<td>0.067</td>
<td>0.143</td>
<td>1.000</td>
</tr>
<tr>
<td>Forest+Wetland</td>
<td>0.102</td>
<td>0.362</td>
<td>0.132</td>
<td>0.404</td>
<td>1.000</td>
</tr>
<tr>
<td>Developed</td>
<td>0.156</td>
<td>0.073</td>
<td>0.663</td>
<td>0.108</td>
<td>1.000</td>
</tr>
<tr>
<td>Other</td>
<td>0.179</td>
<td>0.223</td>
<td>0.130</td>
<td>0.469</td>
<td>1.000</td>
</tr>
</tbody>
</table>
APPENDIX B. Field-based studies included in the retention estimation analysis. To qualify, nitrate-N concentrations, reductions, or loads must have been reported for at least 3 clearly identified sampling locations along a source-to-stream buffered transect and in physiographies similar to those of the MBSS dataset. Studies from all provinces were lumped to create a single range of decay coefficient values due to the limited number of studies appropriate for this analysis. Unless otherwise noted, all values were reported in the original manuscripts in tables.

<table>
<thead>
<tr>
<th>Source</th>
<th>Physiographic Province</th>
<th>Decay Coefficient</th>
<th>R²</th>
</tr>
</thead>
<tbody>
<tr>
<td>Peterjohn &amp; Correll 1984</td>
<td>Coastal Plain, Maryland</td>
<td>0.0453</td>
<td>0.4106</td>
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<tr>
<td>Lowrance 1992</td>
<td>Coastal Plain, Georgia</td>
<td>0.0603</td>
<td>0.5268</td>
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<tr>
<td>Jordan et al. 1993 *</td>
<td>Coastal Plain, Maryland</td>
<td>0.0754</td>
<td>0.7021</td>
</tr>
<tr>
<td>Hubbard &amp; Lowrance 1997 †</td>
<td>Coastal Plain, Georgia</td>
<td>0.0482</td>
<td>0.4423</td>
</tr>
<tr>
<td>Dukes et al 2002</td>
<td>Coastal Plain, North Carolina</td>
<td>0.1069</td>
<td>0.3165</td>
</tr>
<tr>
<td>Dilliha et al 1989</td>
<td>Appalachian Mountain, Virginia</td>
<td>0.119</td>
<td>0.203</td>
</tr>
<tr>
<td>Daniels &amp; Gilliam 1996 ‡</td>
<td>Piedmont, North Carolina</td>
<td>0.087</td>
<td>0.365</td>
</tr>
</tbody>
</table>

* Only two values of nitrate-N were reported; 6 other values were estimated from detailed graph

†Data from the intact (mature) riparian forest were used in this analysis; data from clear-cut and selective thinning sites were excluded

‡Data from riparian forest transects were used in this analysis; nitrate-N values were estimated from detailed graph and text
APPENDIX C. Combinations of the 2001 NLCD and 1980’s NWI designations and their decay coefficients before being scaled to literature values. For example, a cell classified as a wetland by both the NLCD and NWI was given the highest pre-scaled decay coefficient, 1.

<table>
<thead>
<tr>
<th>NLCD</th>
<th>NWI</th>
<th>Decay coefficient prior to scaling</th>
</tr>
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<tbody>
<tr>
<td>Wetland</td>
<td>Wetland</td>
<td>1</td>
</tr>
<tr>
<td>Wetland</td>
<td>Other</td>
<td>0.75</td>
</tr>
<tr>
<td>Forest</td>
<td>Wetland</td>
<td>0.5</td>
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<tr>
<td>Forest</td>
<td>Other</td>
<td>0.25</td>
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</table>
APPENDIX D. Landsat-7/ETM+ data used to calculate the Normalized Difference Wetness Index (NDWI).

<table>
<thead>
<tr>
<th>Path</th>
<th>Row</th>
<th>Sensor</th>
<th>Date of image acquisition</th>
<th>Season</th>
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<tbody>
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<td>14</td>
<td>33</td>
<td>Landsat-7/ETM+</td>
<td>9/23/1999</td>
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<tr>
<td>14</td>
<td>34</td>
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<td>Summer</td>
</tr>
<tr>
<td>15</td>
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<td>10/5/2001</td>
<td>Autumn</td>
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<tr>
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<tr>
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<td>Landsat-7/ETM+</td>
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<td>16</td>
<td>33</td>
<td>Landsat-7/ETM+</td>
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<td>Autumn</td>
</tr>
<tr>
<td>17</td>
<td>33</td>
<td>Landsat-7/ETM+</td>
<td>9/12/1999</td>
<td>Autumn</td>
</tr>
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Appendix E. Results from a one-way ANOVA and p-values from subsequent multiple pairwise comparisons between cropland proportions adjusted by the metrics MIBW, TOPO, WET, NDWI, AVEWET, and LOAD as well as Gap Cropland. Data were re-sampled randomly and without replacement to reflect the smaller sample sizes of the Appalachian provinces.

<table>
<thead>
<tr>
<th>Coastal Plain</th>
<th>MIBW Cropland</th>
<th>Gap Cropland</th>
<th>TOPO Cropland</th>
<th>WET Cropland</th>
<th>NDWI Cropland</th>
<th>AVEWET Cropland</th>
</tr>
</thead>
<tbody>
<tr>
<td>N  F  p  df</td>
<td>14  1.1  0.350  5</td>
<td>0.128 0.739</td>
<td>0.109 0.678</td>
<td>0.941 1.000</td>
<td>0.839 0.836</td>
<td>1.000 1.000</td>
</tr>
<tr>
<td></td>
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</tr>
<tr>
<td>Piedmont</td>
<td>MIBW Cropland</td>
<td>Gap Cropland</td>
<td>TOPO Cropland</td>
<td>WET Cropland</td>
<td>NDWI Cropland</td>
<td>AVEWET Cropland</td>
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<tr>
<td>N  F  p  df</td>
<td>14  0.5  0.727  5</td>
<td>0.152 0.763</td>
<td>0.122 0.706</td>
<td>0.941 1.000</td>
<td>0.839 0.858</td>
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<tr>
<th>Coastal Plain</th>
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<tr>
<td>Piedmont</td>
<td>MIBW Cropland</td>
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<td>WET Cropland</td>
<td>NDWI Cropland</td>
<td>AVEWET Cropland</td>
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<tr>
<td>N  F  p  df</td>
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<td>0.941 1.000</td>
<td>0.839 0.858</td>
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