DEVELOPMENT OF A SPATIALLY DISTRIBUTED MODEL
FOR ESTIMATING NITROGEN REMOVAL IN FORESTED
RIPARIAN BUFFERS

by

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ABSTRACT

Riparian buffers are a widely promoted best management practice for improving stream health and habitat benefits, including reducing instream nitrogen (N) loading. There is a large spatial variability in N removal within riparian buffers, which can largely be explained by buffer width and variations in subsurface water flux. Variable-width buffers are recommended, but there are no simple models to guide restoration efforts, and fixed-width buffers are the norm, also because they are easier to implement and regulate. The goal of our research is to improve riparian buffer restoration for reducing N loading. We developed a simple spatially explicit GIS model that estimates the N removal within riparian buffers and that can prioritize locations for variable-width buffers. The innovative part of our model is the spatially explicit prioritization scheme accounting for removal in both upslope and downslope areas. We tested the model on a watershed in the Mid-Atlantic Piedmont and we applied the model to compare fixed-width and variable-width buffers. Our results show that variable-width buffers require less area to reach N load reduction goals, compared to the fixed-width buffers. Our model will aid watershed managers to make well-informed decisions for prioritizing buffer placement and to maximize N removal and improve water quality.
Chapter 1

INTRODUCTION

Riparian buffers are a widely promoted best management practice (BMP) for improving stream health and habitat benefits, including reducing in-stream nitrogen (N) loading. For example, in the Chesapeake Bay region, in 2014 The Chesapeake Bay Program set a goal to restore 900 miles of forest buffers per year until 70% of streams in the watershed are buffered (Chesapeake Bay Program).

There is large variation in the effectiveness of riparian buffers for N removal. Several transect studies have observed greater than 90% N removal in riparian buffers (Balestrini et al., 2011; Haycock and Pinay, 1993), while other studies have observed less than 30% N removal (Heinen et al., 2012; Sabater et al., 2003). Much of this variability in N removal effectiveness is due to variability in buffer width and subsurface hydrology. To account for this, researchers have proposed variable-width riparian buffer design recommendations based on the slope, hydric soils, or flow routing (Baker, 2010; Kellogg et al., 2010; Lemoine et al., 2006). However, models that would incorporate these controlling factors are very complex. Thus, fixed-width buffers are the norm, because they are easier to implement and regulate.

To promote and implement variable-width buffers requires simple, accurate, and cost-effective models that can be easily transferred between watersheds. Models also need to account for spatial variability of hydrology (Aslan and Trauth, 2014; Sweeney and Newbold, 2014). If we can improve spatial mapping of factors that
contribute to N removal in a watershed, then we can better target areas for riparian buffer restoration and conservation.

The main goal of our research is to improve riparian buffer conservation to reduce N loading. Our objective is to develop a spatially distributed model that can be used as a decision support tool for prioritization of variable-width riparian buffer restoration. Our model was developed using Geographic Information Systems (GIS) and largely builds on previous research, including an empirical model that relates riparian N removal to buffer width and water flux (Sweeney and Newbold, 2014). We tested our model in a small watershed located in the Mid-Atlantic Piedmont, the Red Clay Creek watershed, which exhibits high N loading.

We include a literature review in Chapter 2, discussing the controlling factors involved in N removal and various models that are used to guide riparian buffer placement. A detailed description of our model, model scenarios, and the study area are included in Chapter 3. The intermediate model outputs and results from model scenarios are presented and discussed in Chapter 4. The conclusions are presented in Chapter 5.

Note: The development of the buffer N removal model was a joint effort with my advisor Luc Claessens. The results presented in this thesis are from an earlier version of the model, which does not include the iterative prioritization of variable-width buffers. This feature of the model was later on developed by Luc Claessens.
Chapter 2
LITERATURE REVIEW

2.1 Introduction

The riparian zone plays an important role in the water quality of a watershed, as it connects upslope land areas with the stream. Vegetated riparian buffers enhance stream shading, bank stability, and reduce sediment and N loading. The role of riparian buffers as a BMP for N removal has become more critical as high N loading from agricultural activities has degraded stream health and has led to eutrophication in coastal estuaries. To reduce instream N loading and improve water quality, watershed managers are focusing their efforts on restoring forested buffers along vulnerable streams. The best location for buffer restoration is difficult to determine, as buffers are not effective in all areas. In this literature review, we describe the N removal processes and factors that control buffer effectiveness, including biological factors and hydrological factors. Then, we discuss riparian buffer management and regulation. Finally, we describe current modeling approaches to guide the placement of riparian buffers for N removal.

2.2 N Removal in Riparian Zones

Riparian buffer can be a net N sink depending on its capacity to facilitate two dominant processes: plant uptake and denitrification. Through plant uptake, plants absorb N through the roots and temporarily store reactive N in biomass. Denitrification is a bacterial process by denitrifying bacteria, which convert reactive
nitrate to N gases (both reactive and nonreactive). These bacteria rely on carbon stores in the soil and will only remove nitrate when oxygen levels are low, such as in areas with a shallow water tables. It is often assumed that soils with a developed vegetated buffer have a sufficient carbon supply, as vegetation contributes leaf litter and other organic carbon to the soil. Much of the organic carbon in soils exists closer to the soil surface (0-25cm), in the root zone, which is also where the highest rates of denitrification occur (Clément et al., 2002; Hunter et al., 2006). In fact, denitrification rates decrease exponentially from the soil surface down (Rassam et al., 2008), though deep carbon deposits may exist. In the next two sections we discuss biological and hydrologic factors that control N removal in the riparian zone.

2.3 Biological Factors Controlling N Removal

There are several general characteristics of riparian buffers that control N removal. The capacity of a riparian buffer to facilitate plant uptake and denitrification is dependent on the age of the vegetation, season, and vegetation type. These factors can affect the timing and which process of N removal is more dominant.

The age of the vegetated buffer or length of time since installation can affect the amount of N removal (Heinen et al., 2012). It may take up to 20 years before the buffer reaches maturity and accumulates enough organic matter for denitrification (Fennessy and Cronk, 1997). As the buffer ages, more organic matter will be added to the soil, enabling more denitrification improving buffer effectiveness. Also plant N uptake is affected by the age of buffer. Younger trees and shrubs grow more rapidly, removing more N from the water and store in it the biomass. Rapid growth and subsequent N removal may occur within the first 10 years after buffer installation (Newbold et al., 2010). When the vegetation reaches maturity, growth rates decrease
and less N is absorbed through the roots. One study predicts a annual net zero N removal by plant uptake (Omernik et al., 1981), though periodic harvesting of riparian biomass could maintain N removal efficiencies in the long term (Tomer et al., 2015).

Season is also a factor in N removal. Several studies have found that plant uptake is the dominant N removal process in the growing season and denitrification in the dormant season (Groffman et al., 1992; Simmons et al., 1992). Plant uptake of N occurs almost exclusively during the growing season, from May to October. After the growing season, plants become a source of N as large amounts of N return to the soil as litter (Fennessy and Cronk, 1997; Pinay et al., 1993). Seasonal effects on denitrification vary spatially and may be influenced by seasonal variation in hydrological factors. Studies have shown higher denitrification in the winter and spring when groundwater levels are high (Pinay et al., 1993; Nelson et al., 1995). In contrast, Clement et al. (2002) showed that regardless of changing groundwater levels, denitrification did not vary across season, likely because of high soil moisture levels. Apart from hydrological factors, seasonal variations in denitrification rates could also be influenced by temperature and N availability associated with fertilizer application (Anderson et al., 2014).

The vegetative type and species within a riparian buffer influences the amount of N removal. The riparian buffer design as proposed by the USDA (Welch, 1991), consists of three zones in order from crop field to stream: grasses, managed forest, permanent forest. This design has proven to remove most nutrient and sediment loads from upslope crop fields (Lowrance and Sheridan, 2005). Grasses are ideal for removing sediment from surface flows, while trees have the capacity for removing contaminants from shallow groundwater (Tomer et al., 2008). Despite this combined
design, other buffers consist of only forest or only grasses, which have the potential to
remove 100% and 84% of N, respectively (Haycock and Pinay, 1993). Other studies
have found that there is no difference in N removal effectiveness between vegetation
types: forest, forested wetland, herbaceous, herbaceous/forest, and wetland (Mayer
et al., 2007; Sweeney and Newbold, 2014).

2.4 Hydrologic Factors Controlling N Removal

Regardless of age of buffer, season, or vegetation type, little N removal will
occur if there is a short interaction time between N and carbon. Only when N flows
under anoxic conditions through microbiologically active, carbon rich soil, can large
amounts of N be removed (Heinen et al., 2012). Longer interaction times allow for
more plant uptake and more time for denitrifying bacteria to reduce nitrate to
molecular N. Empirical studies have found a decline in nitrate concentration with
transport through buffers (Lowrance et al., 1997). Thus the width of the buffer is often
used to as an indicator of the interaction time, though buffer width alone cannot
predict interaction time. Interaction time depends on various hydrologic factors,
including the flow distance through the buffer and the rate of flow (Meals et al., 2010;
Passeport et al., 2013).

Flow path is an important factor in N removal as it controls the interaction time
between nitrate and carbon-rich soils in riparian zones. Riparian buffer placement
should consider flow paths to ensure optimal interaction between shallow groundwater
flow and riparian vegetation. Flow paths vary in direction and with depth: surface,
subsurface, and deep groundwater flowpaths. Flow paths are dictated by the depth of
permeable sediment, depth of confining layers, presence of macropores, and the shape
of the watershed, whether concave or convex (Hill et al., 2004; Willems et al., 1997).
Perpendicular flow through a 150m buffer may result in the same retention time as oblique flow through a 10m buffer (Haycock et al., 2009). Nitrate may bypass the organic rich soils in the root zone through deep or overland pathways (Heinen et al., 2012; Vidon and Hill, 2006). Deeper pathways would have negligible denitrification because of a limited carbon supply (Rassam et al., 2008). In places where groundwater flows beneath the buffers, N removal may increase by installing wider buffers (> 25m) (Tomer et al., 2015). A shallow confining layer may allow the water table to interact with plant roots and increase potential for denitrification. Even within these groundwater discharge hotspots, the interaction between the root zone and the upper aquifer may be limited by stream incision (Hill et al., 2004). Flow paths are therefore a critical consideration for estimation interaction time and N removal and should be incorporated into riparian buffer design.

Flow rate is an important factor that affects interaction time and nitrate removal rate. The rate of water flow through the buffer may change the ability of a buffer to remove nitrate per unit buffer width. Several studies found that more N removal occurred in riparian buffers with lower flow rates (Fennessy and Cronk, 1997; Gibert et al., 2008). Though an increase in flow rate may increase denitrification rate initially, high flow rates result in lower denitrification rates because nitrate diffusion to denitrification sites is limited (Rassam et al., 2008; Willems et al., 1997). At very low flow rates however, less nitrate removal will occur because the total volume of water and amount of nitrate passing through would be low (Rassam et al., 2008). Flow rate is controlled mainly by water table gradient. Steeper gradients increase water flow rate and decrease the residence time. Shallow slopes stay saturated longer and have a higher potential for nitrate removal (Hazlett et al., 2008).
The soil characteristics in the riparian zone affect the rate of flow through the buffer and can influence the interaction time between nitrate and carbon. The soil texture, hydraulic conductivity, and porosity of the soil play a role in water flow rate. Hydric soils hold water for longer times (increased residence time), are more frequently saturated and increase N removal capacity. Mineral soils with higher porosities and higher hydrologic conductivity are much less effective because of the faster flow rates.

The surface roughness of the buffer, related to the density of grasses and shrubs, influences the water flow rate and infiltration in a riparian zone. Increasing the roughness of a buffer can decrease the width needed to remove nitrate and sediment (Fennessy and Cronk, 1997). Vegetation increases roughness and slows water flow, allowing more infiltration and residence time in the buffer (Dosskey et al., 2010).

Water flux (flow rate per unit buffer length) has been found to be inversely related to nitrate removal rate (Sweeney and Newbold, 2014). Areas with large water fluxes may experience low nitrate removal rate per unit buffer length because of short interaction time. Higher and more constant water fluxes are associated with larger aquifer size (Vidon and Hill, 2006). Smaller water flux increases the retention time and potential for denitrification. Water flux has been considered a key control of riparian N removal (Sweeney and Newbold, 2014).

2.5 Riparian Buffer Management

2.5.1 Regulation of Riparian Buffers

The regulation of riparian buffers and their management began when impacts from deforestation were detected in streams. Wooded areas were converted to farm
fields and built up land or clear cut for logging, with the riparian zone often being the first area to be logged (Montgomery and Piégay, 2003). This caused increased stream velocity, increased sediment accumulation in the streambed, damaging solar insolation, and increased nutrient concentrations (Thornton et al., 2000). Recommendations for the width of filter strips between logging areas and streams varied from 8 to 50 meters, depending on slope, with doubled distance in municipal areas (Trimble and Sartz, 1957). A study in 1980 suggested that wider buffers, especially those 30m or wider, prevented most of the impact from logging on macroinvertebrate populations (Newbold et al., 1980). Though conservation efforts limited further logging and other riparian degradation, restoring degraded riparian zones is not often required by law and has been met with much criticism.

Riparian buffers provide some of the same ecosystem services as wetlands, such as flood control, wildlife habitat, and nutrient removal. But unlike wetlands, riparian buffers are not protected under the Clean Water Act (NRC, 2002). States are required to implement BMPs to reach any established total maximum daily load (TMDL) reduction requirements. The Environmental Protection Agency (EPA) provides recommendations and funding for riparian buffers through Comprehensive Nutrient Management Plans. The Farm Bill has a Conservation Reserve Program that provides financial incentives for farmers to install forested buffers. De Steven and Lowrance (2011) assessed the frequency of various programs developed from the Farm Bill and found that riparian forest buffers were most likely to be installed in the Piedmont, whereas grass filter strips were more common in the coastal areas.

Many states began to develop their own regulations and guidelines in the past few decades. Recommendations varied from 8-101m with the average of 35m
(Broadmeadow and Nisbet, 2004). Regulations in Georgia require a fixed-width buffer independent of topography (Wenger, 1999), though several other states provided a range of widths, depending on the slope (e.g., wider for steeper slopes). A comprehensive list of state guidelines for riparian buffer width is presented in Phillips et al. (2000).

Pennsylvania established recommendations in 1993 for buffers widths of 15-50m, with an increase by 6m for each 10% increase in slope (Broadmeadow and Nisbet, 2004). In 2010, Pennsylvania increased the minimum buffer width to 100 feet (30m) or 150 feet (45m) for streams designated as Exceptional Value or High Quality waters) for regulatory, voluntary, and grant activities (DEP, 2010). Pennsylvania Act 162 (2014) affected applications for NPDES Stormwater Construction permits proposing an earth disturbance within 150 feet (45m) of a stream designated as Exceptional Value or High Quality waters. Holders of the NPDES permit were required not only to preserve existing buffers but install riparian forest buffers if those waters were not attaining their designed uses (Raphael and Landy, 2015). Although many states have set regulations for timber harvesting and construction in the riparian zone, agriculture activities in riparian zones are still permitted (NRC, 2002). Thus, riparian buffer placement in agriculture areas will continue to be driven by incentive-based volunteer programs until more stringent regulations are established.

### 2.5.2 Fixed-width Buffer Prevalence

Buffer width recommendations have varied over the past several decades. The earliest riparian buffer strategies recommended a uniform or fixed-width for impaired streams and had little scientific bases (Wenger, 1999). Conservationists expected that
increasing the buffer width would increase interaction time between carbon and nitrate and result in more removal.

The scientific literature shows that there is no single fixed-width that is ideal for all stream features. Fixed-width buffers can be insufficient in some locations and overprotective in others (Bren, 2000; Holmes and Goebel, 2011; Qiu, 2009). The main problem of fixed-width buffers is that they do not consider local hydrology or amount of N traveling in the subsurface. Several studies found that buffer width alone was a weak indicator of N removal in the riparian zone (Mayer et al., 2007; Sweeney and Newbold, 2014). A standardized buffer width, therefore, should not be used for all stream locations. Some areas may require wider buffers, or perhaps an alternative BMP. In some cases, widening or narrowing the buffer according to the local hydrology could increase N removal per unit buffer area.

Recently, there is a call for riparian buffer placement strategies to consider hydrologic factors such as flow path, watershed shape, and water flux (Speiran, 2010; Sweeney and Newbold, 2014). Despite recommendations for variable-width buffers, most practitioners implement fixed-width buffers. This is partially due to the uncertainty of potential outcomes of variable-width buffers and the general simplicity of fixed-width buffers (Richardson et al., 2012). Monitoring for compliance is also easier for fixed-width buffers (Lee et al., 2004).

2.6 Modeling Approaches to Guide the Placement of Riparian Buffers

2.6.1 Hydrologic Index Models

Geospatial models can be used to guide buffer placement. Geospatial data include digital elevation models (DEM), land use, and soil data, which are commonly
available on large scales. Several studies incorporate these data in varying degrees of complexity to map hydrologically sensitive areas (HSAs) to improve riparian buffer placement.

One study developed a simple approach to guide land management decisions that emphasizes the location of hydric soils (Kellogg et al., 2010). Dosskey et al. (2006) also used hydric soil data, with depth to water table from the SSURGO database to identify locations where some groundwater pollutant removal may occur in a buffer. Soil capability classifications were used in another study, with land use data to prioritize riparian buffer restoration (Narumalani et al., 1997).

Several hydrologic index methods were developed to identify hydrologically sensitive areas. DEM and land use data along with Hydrologic Engineering Center-Hydrologic Modeling System (HEC-HMS) equations were used to predict travel time to stream for each grid cell and ‘zones of influence’, which would guide buffering (Burcher, 2008). Tomer et al. (2008) tested two landscape analysis techniques: soil surveying and topographic wetness index. They suggest that a combination of these two techniques could help identify locations for riparian buffer restoration. Dosskey et al. (2013) used the Water Inflow Index (WII) and the Topographic Index (TI) combined with soil data to predict hydrology and prioritize buffer placement. They used TauDEM, a multiple direction flow routing tool to calculate both indexes. More research needs to be done to determine threshold values that define the upper limit appropriate for buffer placement (Dosskey et al., 2013). Thus, these indexes can guide riparian buffer placement but are not ideal for precise decision making.
2.6.2 Flow Routing Algorithms

Flow routing can be used to guide riparian buffer placement. The flow distance through a buffer more accurately represents the interaction time, compared to standard buffer calculations which use Euclidean distance from the top of the stream bank to the edge of the buffer (Baker et al., 2006). Geographic Information Systems (GIS) have played a key role in the development of flow routing algorithms. Flow routing algorithms can be categorized into either single-direction or multiple-direction. Single-direction algorithms route flow from one cell to a single adjacent cell along the steepest downslope gradient, while multiple-direction algorithms routes flow into more than one downslope neighbor. In the single-direction algorithm, flow can only be directed in one of eight directions; it is referred to as d8 routing. This single-direction algorithm is an oversimplification of flow (Holmgren, 1994). Single-direction flow routing is used in ArcGIS and Arc Hydro tools (Maidment, 2002), the most widely used GIS software.

Multiple-direction algorithms have been developed to increase dispersion and more realistic of groundwater and subsurface flow (Piechnik et al., 2012; White and Arnold, 2009). The multiple-direction flow algorithms vary by level of dispersion. The method developed by Quinn (1991) distributes flow from a single cell to all neighboring downslope cells, and consequently flow becomes very disperse. As a result, flow in valley bottoms is braided, and model accuracy of this flow routing is questioned in valley bottoms and riparian zones where the subsurface flow meets the stream. They recommend using single-direction flow routing for these areas and overlaying the actual drainage system to improve stream flow routing. Tarboton (1997) developed the d-infinity multiple-direction flow algorithm to reduce unrealistic dispersion. This method apportions flow into only two downslope neighboring grid
cells, reducing dispersion compared to the multiple-direction algorithm. The d-infinity tools can accurately map hydrological characteristics (Wolock and McCabe, 1995), and has been used to estimate surface and shallow subsurface flow to prioritize streams for riparian buffering (Tomer et al., 2008).

2.6.3 Estimating Subsurface Flow

Digital Elevation Models (DEMs) are used as a proxy for water table elevation and soil-water storage dynamics (Baker et al., 2003; Christensen et al., 2013). In addition to various routing methods, surface topography can be smoothed in a GIS to estimate groundwater flow. Several studies used a focal averaging technique to smooth the DEM and estimate the shape of the water table (Baker et al., 2003; Wang et al., 2013). They assume that the smoothed DEM has the same shape as the water table, though this is not always the case (Lanni et al., 2011). Focal mean can be calculated for each cell in a GIS to smooth hills and valleys in a DEM. Smoothing a DEM can improve estimation of water table shape, but not water table elevation (Lanni et al., 2011). There are several DEM smoothing operations that can estimate groundwater flow, shown in Table 2.1.
Table 2.1 Examples of DEM smoothing applications.

<table>
<thead>
<tr>
<th>Smoothing method</th>
<th>Reference</th>
<th>Raster Resolution (m)</th>
</tr>
</thead>
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<td>3 x 3 low-pass filter</td>
<td>(Lanni et al., 2011)</td>
<td>2</td>
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<tr>
<td>7 x 7 moving-window average (60 times)</td>
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</tbody>
</table>

Often streams that are derived from a DEM do not match mapped flow lines. There are various techniques to hydrologically recondition the DEM to ensure that the flow defined streams align with the mapped flow lines, and force the flow into known streams locations. The AgreeDEM Reconditioning tool in the Arc Hydro Tools “burns” the stream into a DEM and forces flow into the mapped flow lines (Maidment, 2002). This tool consists of several sub-processes that divide a watershed into wedges based on the distance to the closest stream cell. The elevation of each cell in the watershed is adjusted based on the elevation of the closest stream cell. An alternative, simple stream burning method is a Euclidean Reconditioning technique (Tarboton, pers. comm., November 19, 2015). Streams are burned into the DEM using a gradual incision controlled by the Euclidean distance to streams, without dividing the watershed into wedges. The main difference between these two methods is that AgreeDEM Reconditioning has this allocation subroutine that relates the elevation of each grid cell in the watershed to the elevation of the closest stream cell.

2.6.4 GIS Models for Prioritization Buffer Restoration

GIS provides a platform in which spatial datasets can be compiled, visualized, and analyzed. Several studies have utilized GIS to prioritize riparian buffer restoration.
Some studies developed GIS approaches to delineate and evaluate riparian buffers based on stream size (Zeilhofer et al., 2011; Apan et al., 2002). Rassam et al. (2005) developed the Riparian Nitrogen Model, which uses a bucket model approach to calculate the potential N removal. This can be used to prioritize buffer placement on the catchment scale and has been applied in South East Queensland, Australia (Rassam et al., 2008). Baker et al. (2006) developed a series of flow-path metrics, such as buffer gap frequency, that can be used to evaluate current buffer conditions and target buffer restoration. These metrics were incorporated in a user-friendly GIS model, the Riparian Analysis Toolbox (Baker, 2010), and used to compare physiographic provinces in the Chesapeake Bay Basin (Weller and Baker, 2014). These models can evaluate N removal and help prioritize watersheds for restoration, but are not specifically designed to prioritize the placement of variable-width buffers.

2.6.5 Empirical Model of N Removal

Methods are needed to determine the buffer width necessary to achieve TMDL load reductions. Knowing the appropriate buffer width for a stream section could reduce under-protection and over-protection of streams. Sweeney and Newbold, (2014) performed an extensive review of scientific literature to determine the buffer width necessary for N removal. Using the data collected in the literature review, they converted a conceptual model developed by Vidon, (2006) into an empirical model. This empirical model uses water flux to calculate nitrate removal rate per unit distance into the buffer, $k_N$ ($m^{-1}$). They chose an exponential model because most studies found that nitrate levels decline faster towards the upslope boundary of the buffer. They found a poor relationship between buffer width and nitrate removal efficiency, which confirmed previous perceptions (Figure 2.1). Subsurface water flux, $q_L$ (l/m/day),
(lateral flow rate per unit buffer length) explained 37% of the variability of N removal efficiency (Figure 2.2). The remaining 63% might be explained by flow path, landscape, soil, or geologic factors. Therefore, buffer width and the water flux have a large influence on N removal.

The authors suggest that this empirical model could be used to identify the necessary buffer width on a watershed scale for a desired fractional N removal, if the average water flux in the basin is known. Here in this study we use the Sweeney and Newbold (2014) empirical model and apply it within a spatially distributed framework to estimate N removal in forested riparian buffers, and to find optimal locations for restoring riparian buffers.

Figure 2.1 Nitrate removal efficiency vs. buffer width (Sweeney and Newbold, 2014).
Figure 2.2 Nitrate removal rate vs. subsurface water flux (Sweeney and Newbold, 2014).
Chapter 3

BUFFER N REMOVAL MODEL

3.1 Introduction

We developed a simple spatially explicit decision-support GIS model that estimates the N removal within riparian buffers and that can prioritize locations for variable-width buffers. The model was built within ArcGIS and uses Arc Hydro tools, TauDEM tools, and customized scripts. Our model is spatially distributed, such that buffer width varies spatially across the watershed depending on spatial variation in existing N sinks, subsurface water flow and N yield. The innovative part of our model is the spatially explicit prioritization scheme accounting for removal in both upslope and downslope areas.

This chapter is presented in three sections: Study Area, Model Description and Buffer Scenarios. We tested the model on a watershed in the Mid-Atlantic Piedmont which is subject to high N loading. We collected the data required for the model and applied the model to compare fixed-width and variable-width restored buffers. The model requires various general data sets, including elevation, hydrography, and land-use. In the Model Description section we explain the GIS model, including the methods for processing elevation data, and for calculating flow, N loading, and N removal for different riparian buffer scenarios. In the Buffer Scenarios section, we explain how we mapped locations for fixed-width buffers and prioritized locations for variable-width buffers.
3.2 Study Area

We tested the buffer N removal model on a watershed in the Mid-Atlantic Piedmont and we applied the model to compare fixed-width and variable-width buffers. The watershed (R01) (25.2 km2) is located in Chester County, Pennsylvania, in the headwaters of the west branch of the Red Clay Creek (Figure 3.1). The Red Clay Creek watershed is one of the four main watersheds of the Brandywine-Christina River Basin, along with the Brandywine Creek, White Clay Creek, and Christina River watersheds. We chose to focus on the Red Clay Creek because of the high concentration of mushroom farming, which is the focus of a related research study. We chose the subwatershed R01 because TMDL studies assigned it a relatively high total N load reduction (U.S. Environmental Protection Agency, 2006) and because headwater catchments play an important role for controlling the water quality (Alexander et al., 2007). Slightly less than half of the watershed area is agriculture with approximately 20% forest or wetlands, which are classified as existing N sinks (Table 3.1, Figure 3.2). The contiguous N sinks vary in width from the stream from 0 (no buffer) to over 100m in some sections.

The Brandywine-Christina River Basin (BCRB) is a heavily researched basin due to its economic value and water quality issues, with a total of 117 miles of streams impaired for nutrients (Brandywine Valley Association, 2013). Nearly 600,000 people living in Delaware, Pennsylvania, and Maryland depend on the waters of the BCRB for drinking water (Jones Jr. and Kauffman, 2015). Land-use in the BCRB is approximately evenly distributed among urban, forest/wetland, and agriculture (Jones Jr. and Kauffman, 2015) – see Figure 3.3 for a map of land-uses. The climate in the region is humid continental with average temperatures varying from 22.6°C in the summer to -0.4°C in the winter and an average annual rainfall of 1,215 mm (NCEI).
The BCRB is in the southern portion of the Delaware River Basin which covers portions of five states and five physiographic provinces (Murdoch et al., 2008) (Figure 3.4). The BCRB falls mostly within the Piedmont physiographic province. The Piedmont has a unique hydrology, where flow patterns are constrained primarily by valley shapes (Lowrance et al., 1997). Depth to groundwater is a less dominant factor for flow patterns in the Piedmont compared to the Coastal Plain. The Piedmont region is subject to high N yields, primarily from corn, soy and alfalfa crops (Fischer and Moore, n.d.) – see Figure 3.5 for a map of all primary sources of N in the Delaware River Basin.

In 2003, the USGS performed simulation studies for each watershed in the Brandywine-Christina River basin based on four years of data collection. These simulations were used to create TMDLs for the subwatersheds with high N loads in the basin. In 2006, the Environmental Protection Agency established total maximum daily loads (TMDLs) to 29 out of the 70 subwatersheds in the Brandywine-Christina River Basin (Figure 3.6).

Table 3.1 Land use summary of R01 subwatershed.

<table>
<thead>
<tr>
<th>Classification</th>
<th>Area (km²)</th>
<th>Area (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Residential</td>
<td>5.25</td>
<td>20.5</td>
</tr>
<tr>
<td>Urban</td>
<td>3.16</td>
<td>12.4</td>
</tr>
<tr>
<td>Agriculture</td>
<td>10.57</td>
<td>41.4</td>
</tr>
<tr>
<td>Mixed agriculture</td>
<td>1.36</td>
<td>5.3</td>
</tr>
<tr>
<td>Forest</td>
<td>4.26</td>
<td>16.7</td>
</tr>
<tr>
<td>Wetland/water</td>
<td>0.88</td>
<td>3.4</td>
</tr>
<tr>
<td>Other/unclassified</td>
<td>0.07</td>
<td>0.3</td>
</tr>
<tr>
<td>Total</td>
<td>25.55</td>
<td>100</td>
</tr>
</tbody>
</table>
Figure 3.1 The R01 study subwatershed is located in the headwaters of the Red Clay Creek watershed.
Figure 3.2. Land use of the R01 subwatershed.
Figure 3.3 Land uses in the Brandywine-Christina River Basin (Jones and Kauffman, 2015).
Figure 3.4. Land use and physiographic provinces of the Delaware River Basin (Murdoch et al., 2008).
Figure 3.5 Nitrogen yield in the Delaware River Basin (Fischer and Moore, n.d.).
Figure 3.6 TMDL N load reduction of the Christiana River Basin (Jones and Kauffman, 2015).
3.2.1 Data Collection

To make our model widely useable, all of the data layers used in our analysis are publicly available. The data sources for this study are summarized in Table 3.2. We obtained elevation data (DEM) from the United States Geologic Survey (http://viewer.nationalmap.gov/) and stream hydrography from Chester County and the National Hydrography Dataset (NHD). The DEM has a 10-m horizontal resolution and a 2.44-m vertical resolution (DEM User’s Manual 2nd Edition). A 10-meter DEM was used because elevation data are widely available at this resolution and 10-meters is suitable for flow path estimation and agricultural field scale BMP placement (Piechnik et al., 2012). The DEM and stream hydrography are presented in Figure 3.7. The shaded relief of the elevation surface can be seen in Figure 3.8, which shows the hillshade of the DEM. The elevation of this subwatershed ranges from 77-168m. We obtained land use/land cover (LULC) data from Chester County.

Table 3.2 Sources of spatial data for the Red Clay Creek study subwatershed.

<table>
<thead>
<tr>
<th>Data</th>
<th>Source</th>
<th>Date</th>
</tr>
</thead>
<tbody>
<tr>
<td>DEM(^1)</td>
<td>National Elevation Dataset - USGS</td>
<td>2015</td>
</tr>
<tr>
<td>Hydrography</td>
<td>National Hydrography Dataset – USGS; Chester County</td>
<td>2014</td>
</tr>
<tr>
<td>LULC</td>
<td>Chester County</td>
<td>2005</td>
</tr>
</tbody>
</table>

\(^1\) horizontal resolution: 10x10 m
Figure 3.7 Elevation of R01 subwatershed.
Figure 3.8 Elevation of R01 subwatershed shown with hillshade effect.
3.3 Model Description

3.3.1 Elevation Pre-processing

The digital elevation model (DEM) data required pre-processing. This was necessary to remove artifacts due to bridges and other road crossings that artificially elevated streambed elevations. These artifacts interfered with flow routing, and affected the results from N removal calculations and prioritized buffer placement. The DEM pre-processing methodology was developed by Luc Claessens. First, a hydrologically corrected DEM and stream raster was created using Arc Hydro tools and the hydrography data layer. Next, streambed elevations at road crossings were adjusted using an iterative local minimum search operation. This produced an adjusted, pre-processed DEM. After that, Arc Hydro tools were run a second time to produce an updated hydrologically corrected DEM and hydrography stream raster.

3.3.2 Elevation Processing

To estimate groundwater flow patterns, we created a smoothed and hydrologically conditioned surface from the pre-processed DEM and hydrography stream raster. We tested several variations of smoothing and stream burning similar to other studies (see section 2.6.3). In initial calculations we found that artificial ‘island cells’ were created in the riparian areas that were disconnected from nearby flow paths. These ‘island cells’ had no flow accumulation and were unrealistic of actual flow patterns. To improve the flow patterns in the riparian areas and remove these ‘island cells’, we performed further elevation processing through DEM hydrological reconditioning and smoothing.
First, we performed hydrological reconditioning to force the flow into the stream, by using a simple Euclidean Reconditioning method (Tarboton, pers. comm., November 19, 2015), as follows:

\[ Elev_{\text{recon}} = Elev_{\text{pre}} - 10 \cdot Elev_{\text{str}} - 0.02 \cdot (50 - Distance) \cdot \begin{cases} 1 & (Distance < 50) \end{cases} \]  

Where: \( Elev_{\text{recon}} \) is the reconditioned elevation, \( Elev_{\text{pre}} \) is the pre-processed elevation, \( Elev_{\text{str}} \) is the pre-processed elevation of stream grid cells and \( Distance \) is the Euclidean Distance to the nearest stream grid cell. This reconditioning lowers the elevation of stream cells, and lowers the elevation of hillslopes within 50m of the stream, such that the elevation slopes down towards the stream.

Next, we smoothed the reconditioned DEM using a filtering operation. We used a sequence of 5 filtering operation, using a 3x3 filter, and omitting the stream cells. Lastly, we used Pit Remove to raise topographical sinks in the hydrologically reconditioned and smoothened DEM that may interfere with the flow routing operations.

### 3.3.3 Flow Routing

To estimate the spatially distributed water flux and N yield we use flow routing. The flow routing steps determine the flow directions, flow accumulation, stream location, and watershed boundary. These are all intermediate outputs in the model and are used throughout the model.

There are several flow routing algorithms that produce various levels of dispersion (see section 2.6.2). The multiple-direction algorithms allow for more dispersion of flow compared to single-direction algorithms (d8), and are more representative of subsurface flow. We used TauDEM toolbox because it has a
multiple-direction algorithm, D-infinity. The D-infinity flow algorithm proportions flow into one or two downslope grid cells with a realistic dispersion (Tarboton et al., 2009). It is based on the steepest slope of a triangular facet (Tarboton, 1997) – see Figure 10. The flow may be routed into one of the eight neighboring cells or divided proportionally into two adjacent cells depending on the direction of the steepest slope. The D-infinity Contributing Area tool uses the d-infinity flow direction grid to calculate the specific catchment area, which includes the contribution of the grid cell itself. We divided the specific catchment area by the grid cell length (10m in our case) to convert the specific catchment area to flow accumulation. We called this the d-infinity flow accumulation grid.

To define the stream layer, we used the Stream Definition tool from the Arc Hydro toolbox, using a catchment area threshold of 0.04 km$^2$ (400 cells). We related this stream against the hydrography data to obtain a final stream grid that is aligned with the DEM and matches the mapped streams. Lastly, we defined the watershed boundary using the D-infinity Upslope Dependence tool.

### 3.3.4 Stream Width Calculation

To accurately map the lateral extent of buffer locations, we calculated stream width. Stream width accuracy was limited by the horizontal resolution of the DEM (10m). A finer resolution DEM may improve stream definition, but may interfere with flow routing. We calculated the drainage area from the previously calculated flow accumulation, adjusting by cell area. Stream width was estimated using discharge, according to hydraulic geometry relationships:

\[
Q = x \cdot DA^y
\]

\[
W = a \cdot Q^b
\]
where $Q$ is discharge ($m^3/s$), $DA$ is drainage area ($km^2$), $W$ is stream width (m), and $a$, $b$, $x$, and $y$ are constants. We used parameters values for the Piedmont region of the Mid-Atlantic, reported by Mohamoud and Parmar (2006) ($a$ is 11.95, $b$ is 0.47, $x$ is 0.015, and $y$ is 0.989).

### 3.3.5 Nitrogen Loading

$N$ yield (load per unit area) is typically estimated from land-use. $N$ yield values were based on nitrate yield values for the Red Clay Creek TMDL simulation study (Senior and Koerkle 1994) (Table 3.3). We assume that nitrate is the dominant nitrogen form and from here forward we refer to nitrogen (N) instead. For residential areas, we averaged the $N$ yield from residential-sewered and residential-unsewered. For agricultural areas, we averaged the $N$ yield from animal/crops, row crops, and mushroom yields. All other land use classifications were similarly classified into a land use group as shown in Table 3.4 and assigned the corresponding $N$ yield values. For restoration scenarios, the $N$ yield values for buffer cells were adjusted according to the change in land use. To calculate $N$ loads, we accumulated the surface of $N$ yields using the flow routing routine. We used the D-infinity Contributing Area tool in the TauDEM Toolbox weighted by the $N$ load grid to accumulate the loading through the watershed. We converted the accumulated $N$ yields into loads (kg/yr) for the subwatershed. This value constitutes the land-use $N$ loading of the watershed, which is the loading estimate without accounting for $N$ removal in existing riparian buffers or other $N$ sinks.
Table 3.3 Simulated mean annual nitrate yield by land use for Segments 7 of the Red Clay Creek Watershed, 1995-97. (Senior and Koerkle, 2003).

<table>
<thead>
<tr>
<th>Land-use Group</th>
<th>N yield (kg/km²/yr)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Residential</td>
<td></td>
</tr>
<tr>
<td>Unsewered</td>
<td>1398</td>
</tr>
<tr>
<td>Sewered</td>
<td>740</td>
</tr>
<tr>
<td>Average</td>
<td>1069</td>
</tr>
<tr>
<td>Urban</td>
<td>731</td>
</tr>
<tr>
<td>Agricultural</td>
<td></td>
</tr>
<tr>
<td>Animal/crops</td>
<td>2712</td>
</tr>
<tr>
<td>Row crop</td>
<td>2354</td>
</tr>
<tr>
<td>Mushroom</td>
<td>3183</td>
</tr>
<tr>
<td>Average</td>
<td>2750</td>
</tr>
<tr>
<td>Forested</td>
<td>158</td>
</tr>
<tr>
<td>Open</td>
<td>517</td>
</tr>
<tr>
<td>Wetlands/water</td>
<td>167</td>
</tr>
<tr>
<td>Impervious</td>
<td></td>
</tr>
<tr>
<td>Urban</td>
<td>229</td>
</tr>
<tr>
<td>Residential</td>
<td>229</td>
</tr>
<tr>
<td>Undesignated</td>
<td>514</td>
</tr>
</tbody>
</table>

3.3.6 Nitrogen Removal

Nitrogen removal is calculated by multiplying N loading times fractional removal. These calculations are done in combination with the flow routing between the grid cells. The calculation of fractional removal is based on the empirical model presented by Sweeney and Newbold (2014), which estimate a first-order decay multiplier. The decay multiplier is a function of water flux and buffer width.

We calculated the water flux \( q_L \) (l/m/day) from the previously derived d-infinity flow accumulation grid (\( fac\ cells \)) using equation 4:

\[
q_L = \frac{100 m^2}{cell} \cdot \frac{Base\ flow}{m^3/yr} \cdot \frac{1000 l}{m^3} \cdot \frac{1 yr}{365\ days} \cdot FD \cdot \frac{1}{m} \tag{4}
\]

We assumed the average annual base flow depth to be 0.2 m/yr, using the same assumption as Sweeney and Newbold (2014). This is representative of humid plains in
the eastern United States where the total annual flow is 0.416m with 47.5% from shallow groundwater contribution (Santhi et al., 2007). We assumed a flow distance \((FD)\) of 12.071m, the average between diagonal and straight flow through a 10x10m cell.

Next, we calculated N removal rate per unit buffer width into the buffer, \(k_N\) \((m^{-1})\), for each cell in the study area using equation 5:

\[
k_N = \frac{\alpha}{q_L}
\] (5)

We assumed the removal factor, \(\alpha\) to be 2.72, as determined by Sweeney and Newbold (2014). Then, we calculated the decay multiplier, \((r(x))\) using equation 6:

\[
r(x) = \exp(-k_N \cdot x)
\] (6)

where \(x\) is the flow distance through a single cell (12.071m).

After calculating nitrogen removal, the model calculates the instream N loading, the N that enters the stream without being removed. We termed this value the final N loading. To calculate total removal at the watershed-scale, we subtracted the final N loading from the land-use N loading. For restoration scenarios, we also calculated the fraction of N removed relative to the baseline loading (after removal in existing sinks), to determine whether the TMDL load reduction was achieved.

### 3.3.7 Nitrogen Load Reduction from LULC Change

Nitrogen load reduction is achieved through two processes: N load reduction from change in land-use N yield, and N load reduction from removal within the buffer (calculated with the decaying accumulation tool). To calculate the portion of N load reduction from a change in land-use N yield (for restoration scenarios), we compared the change in N loading before and after adjusting the N yield. The portion of N load
reduction from removal within the buffer is then calculated as the total N load reduction minus the load reduction from change in land-use N loading.

3.4 Buffer Scenarios

We applied the model to examine N removal of existing buffers and for various buffer restoration scenarios. For restoration scenarios, we restricted buffer placement to only cropland and rangeland areas. Residential, urban, farmsteads, and commercial land uses were not considered suitable because it would be costly and impractical to remove existing infrastructure. The suitability of each land use category for buffer restoration is presented in Table 3.5. Areas with existing forests and wetlands were assumed to have already functioning buffers. These existing buffers were added to the simulated restored buffer locations during model simulations.
Table 3.4. Land use classification equivalents – matching nitrate yield land use categories with land use data classification.

<table>
<thead>
<tr>
<th>Level II Classification(^\d) (Anderson et al., 1976)</th>
<th>Land-use Class 2 Code(^\d)</th>
<th>Land-use Group(^2)</th>
<th>N Yield Values(^2) kg/km(^2)/yr</th>
<th>Suitability for buffering</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Urban or Built-up Land</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Residential</td>
<td>11</td>
<td>Residential</td>
<td>1069</td>
<td>-</td>
</tr>
<tr>
<td>Commercial and service</td>
<td>12</td>
<td>Urban</td>
<td>731</td>
<td>-</td>
</tr>
<tr>
<td>Industrial</td>
<td>13</td>
<td>Urban</td>
<td>731</td>
<td>-</td>
</tr>
<tr>
<td>Transportation</td>
<td>14</td>
<td>Urban</td>
<td>731</td>
<td>-</td>
</tr>
<tr>
<td>Industrial</td>
<td>15</td>
<td>Urban</td>
<td>731</td>
<td>-</td>
</tr>
<tr>
<td>Mixed urban</td>
<td>16</td>
<td>Urban</td>
<td>731</td>
<td>-</td>
</tr>
<tr>
<td>Other urban</td>
<td>17</td>
<td>Urban</td>
<td>731</td>
<td>-</td>
</tr>
<tr>
<td>Institutional</td>
<td>18</td>
<td>Urban</td>
<td>731</td>
<td>-</td>
</tr>
<tr>
<td>Public recreation/conservation forest</td>
<td>19</td>
<td>Open</td>
<td>514</td>
<td>-</td>
</tr>
<tr>
<td><strong>Agriculture</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Crop and pasture</strong></td>
<td>21</td>
<td>Agriculture</td>
<td>2750</td>
<td>Yes</td>
</tr>
<tr>
<td>Orchards, vineyards, and nurseries(^3)</td>
<td>22</td>
<td>Mixed agriculture</td>
<td>1653</td>
<td>Yes</td>
</tr>
<tr>
<td>Confined feeding operations</td>
<td>23</td>
<td>Agriculture</td>
<td>2750</td>
<td>-</td>
</tr>
<tr>
<td>Farmstead/farm related structures</td>
<td>24</td>
<td>Residential</td>
<td>1069</td>
<td>-</td>
</tr>
<tr>
<td>Other agriculture</td>
<td>29</td>
<td>Agriculture</td>
<td>2750</td>
<td>Yes</td>
</tr>
<tr>
<td><strong>Rangeland(^3)</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rangeland - herbaceous</td>
<td>31</td>
<td>Mixed agriculture</td>
<td>1653</td>
<td>Yes</td>
</tr>
<tr>
<td>Rangeland - shrub</td>
<td>32</td>
<td>Mixed agriculture</td>
<td>1653</td>
<td>Yes</td>
</tr>
<tr>
<td>Rangeland – mixed</td>
<td>33</td>
<td>Mixed agriculture</td>
<td>1653</td>
<td>Yes</td>
</tr>
<tr>
<td><strong>Forested</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Deciduous forest land</td>
<td>41</td>
<td>Forested</td>
<td>158</td>
<td>-</td>
</tr>
<tr>
<td>Coniferous forest land</td>
<td>42</td>
<td>Forested</td>
<td>158</td>
<td>-</td>
</tr>
<tr>
<td>Mixed coniferous-broadleaf</td>
<td>43</td>
<td>Forested</td>
<td>158</td>
<td>-</td>
</tr>
<tr>
<td>Clear cut</td>
<td>44</td>
<td>Open</td>
<td>514</td>
<td>-</td>
</tr>
<tr>
<td><strong>Water</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Open water</td>
<td>50</td>
<td>Wetlands/water</td>
<td>158</td>
<td>-</td>
</tr>
</tbody>
</table>
Waterways, streams, canals 51 Wetlands/water 158 -
Natural lakes and ponds 52 Wetlands/water 158 -
Man-made reservoirs 53 Wetlands/water 158 -
Bays and estuaries 54 Wetlands/water 158 -
Other 59 Undesignated 514 -
Wetlands
Wetlands 60 Wetlands/water 158 -
Forested wetlands 61 Wetlands/water 158 -
Non-forested wetlands 62 Wetlands/water 158 -
Barren Land
Sandy Areas other than beaches 73 Open 167 -
Bare Exposed rock 74 Open 167 -
Strip mines, quarries, and gravel pits 75 Open 167 -
Transitional 76 Urban 517 -
Mixed barren land 77 Open 167 -
Unclassified
Unclassified 0 Undesignated 514 -

1 Classifications of the land use data (Chester County, 2005).
2 Land-use Groups and matching N yields obtained from Senior and Koerkle (2003).
3 Rangeland and orchards were not listed as land-use categories in Senior and Koerkle (2003) so mixed agriculture values from Reckhow (1980) were substituted.

### 3.4.1 Fixed-width Buffer Restoration

We created a series of fixed-width buffers with widths from 10 to 50 meters at 10m intervals, and from 50 to 150 at 25m intervals. The positioning of the outer boundaries of these buffers was adjusted to account for the stream width, by adding half of the stream width to the buffer width on each side of the stream. For streams with widths less than 10m, we used 5m as the minimum width for half of the stream, because the narrowest streams are still represented by a 10x10m cell. Thus, the minimum buffering width, which measures from the stream centerline, would be 15m for a 10m wide buffer. We converted this combined stream and buffer width to an
integer and used Raster to Polyline to create a polyline from the stream raster. Then, we used the Buffer tool and buffered this polyline by the value field. This creates a polygon with the correct buffer width, including the stream width. We clipped the stream out from this buffer using a polygon of the stream. This resulted in a raster layer representing fixed-width buffers. The land-use N yield rates were adjusted for the locations of the fixed-width buffers. Next, the N removal calculations were performed as described in section 3.3.6.

3.4.2 Variable-width Buffer Restoration

To prioritize areas for variable-width buffer design we calculated the net N removal for each grid cell. Net N removal \( R_{\text{net}} \) was calculated for each grid cell suitable for buffering using equation 7:

\[
R_{\text{net}} = \text{accumulated } N \text{ loading} \cdot (1 - r(x))
\]  

(7)

The calculation of \( R_{\text{net}} \) is part of an iterative algorithm (see next) that continuously reassigns buffer grid cells. Buffer cells are assigned using a slicing tool that divides the range of net N loading into a specified number of zones of equal area.

The net removal of a given cell depends on whether there are also buffer cells downslope, in which case the actual removal will be less, and adding buffer width may result in only negligible net N removal. To account for this downslope dependency, Luc Claessens developed a model that reevaluates the accumulated N loading of all cells based on the removal that occurs in grid cells downslope. This iterative algorithm is incorporated into the prioritization scheme described above.

At each prioritization iteration we only consider added buffer areas that are contiguous with stream cells and riparian buffer areas. Noncontiguous ‘buffer’ areas were not considered because they are less practical for restoration. To determine
whether buffer areas are contiguous, we temporarily designated the stream as a buffer and then used Region Group to number the contiguous groups with eight-cell neighborhood to define contiguity. We determined the region number of the stream grid cells and then removed all other regions that did not match the region number of the stream. After removing the stream itself, only contiguous buffers remain. The N yield rates were similarly adjusted for the locations of the new variable-width buffers. Next, the N removal calculations were performed as described in section 3.3.6.
4.1 Introduction

The results presented in this thesis are from an earlier version of the model, which does not include the iterative prioritization of variable-width buffers. This feature of the model was later developed by Luc Claessens. The results presented here include the intermediate output results from the model as well as the results from the buffer restoration scenarios. In Appendix A we also include results from preliminary analysis that were produced during earlier phases of model development.

4.2 Intermediate Results

4.2.1 Nitrogen Loading

The land-based N loading is concentrated in the northwest portion of the R01 subwatershed where there is predominantly agricultural land use (Figure 4.1). Average N loading before any N removal is 1,542 kg/km²/yr.
Figure 4.1 Nitrate yield (kg/km²/yr).
4.2.2 DEM Processing

DEM processing creates a hydrologically reconditioned DEM and stream grid. The original DEM and processed DEM are presented in Figure 4.2. Road features visible in the original DEM are removed after processing. The DEM smoothing slightly reduced the elevation range, decreasing the average slope of the subwatershed. It was our intention that with a shallower slope, more dispersion would take place. While we did observe an increase in dispersion from d8 flow routing methods to d-infinity flow routing, only minimal additional dispersion took place after DEM smoothing. The reason for this is because the dispersion of d-infinity flow depends on the relative difference in elevation between a center cell and an adjacent cell versus a center cell and a diagonal cell (Tarboton et al., 2009). But because focal averaging of the elevation does not change the relative difference in elevation between neighboring cells, there was therefore little or no additional dispersion from DEM smoothing. Further, while the d-infinity method produces more dispersion than the d8 method, it is still limited to two cell dispersion, and may result in perhaps overly concentrated flow. Other methods may provide more dispersion and perhaps a more accurate representation of subsurface water flow, including multiple direction flow algorithms such as those developed by Quinn et al. (1991) or McGlynn and Seibert (2003).
Figure 4.2 Elevation images before and after DEM processing.
4.2.3 Flow Routing

The flow routing was performed using a d-infinity algorithm in TauDEM. We created a d-infinity flow directions grid, presented in Figure 4.3. The d-infinity flow directions output is in radians but, for display purposes, the flow directions were split into four quadrants: northeast, northwest, southwest, and southeast. The flow accumulation grid is shown in Figure 4.4. The flow accumulation includes the cell itself, thus the minimum flow accumulation is one. The watershed area for R01 was 25.2 km$^2$. 
Figure 4.3 Flow direction grid divided into four diagonal directions, using D-infinity.
Figure 4.4 Flow accumulation grid, using D-infinity. Values are expressed as number of grid cells of 100m$^2$ each (10x10m). Watershed area is 25.55 km$^2$. 
4.2.4 Stream Width

The calculated stream width for each stream cell was less than 10 meters in this subwatershed. Since the resolution of the DEM is 10x10m, the minimum assigned stream width was 10 meters. It is important to consider stream width when mapping riparian buffers. We considered making narrow streams suitable for buffering since a portion of a 10m cell could be occupied by trees. We decided that accounting for the dual function of a cell that acts as both a stream and a buffer would create unnecessary complexity, and thus decided to limit one function per grid cell.

4.2.5 Nitrogen Removal

The water flux was calculated from flow accumulation (Figure 4.5). Most areas on the upper parts of the hillslopes had a water flux less than 30 l/m/day. Most streamside regions had a water flux between 30 and 200 l/m/day. Stream features started to form where water flux reached 400 l/m/day. Fractional N removal was calculated from the water flux and displayed the same spatial pattern (Figure 4.6). There are flow paths branching off the stream that have high water fluxes and low fractional removal values. The pattern changes when N loading is included in the calculation of fractional N removal. Areas of high water flux tend to be areas of high N loading, so the actual N removal is high, even though the fractional N removal rate is low. Figure 4.7 illustrated this effect across the entire watershed. Note that this map is for illustration purpose only; during model simulations we only account for removal in existing N sinks and restored buffers.
Figure 4.5 Water flux (l/m/day).
Figure 4.6 Fractional N removal.
Figure 4.7 N removal calculated from fractional N removal and N loading.
4.3 Buffer Scenarios

We calculated N removal within existing, fixed-width, and variable-width buffer scenarios. For the restoration scenarios we only considered placement of buffers in agricultural areas (cropland and pastures) (Figure 4.8).

Figure 4.8 Areas suitable for potential buffer restoration.
4.3.1 Existing Buffer Assessment

We calculated the existing buffer area as 5.14 km$^2$, or 20% of the total subwatershed area. Existing buffers are presented in Figure 4.9. Table 4.1 summarizes the N loading and removal in existing buffers in our study area. Our model calculated a 26% load reduction within existing buffers. The final loading was 1,141 kg/km$^2$/yr, which we took as the baseline to determine the net load reduction in buffer restoration scenarios.

Table 4.1 Summary results of N removal in existing buffers.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Watershed area (km$^2$)</td>
<td>25.55</td>
</tr>
<tr>
<td>Buffer area (% of watershed area)</td>
<td>18</td>
</tr>
<tr>
<td>Land-use N loading (kg/km$^2$/yr)</td>
<td>1,542</td>
</tr>
<tr>
<td>N removal in existing buffers (% of total N load)</td>
<td>26</td>
</tr>
<tr>
<td>Baseline N loading (kg/km$^2$/yr)</td>
<td>1,141</td>
</tr>
<tr>
<td>TMDL N load reduction goal$^1$ (%)</td>
<td>48.6</td>
</tr>
</tbody>
</table>

$^1$ TMDL load reduction is from the baseline loading.
Figure 4.9 Existing N sinks.
4.3.2 Fixed-width Buffer Assessment

We examined nine fixed-width buffer scenarios, ranging between 10m and 150m (Table 4.2). An example of a fixed-width buffer is presented in Figure 4.10. Only areas suitable for buffers are shown, including cropland and pasture. For some stream reaches, no buffer is added because the existing buffer is sufficiently wide. As buffer width increased, the N load reduction also increased. There is a steep N load reduction with the addition of 10m buffers, but the benefit from widening the buffer decreases as width increases.

The uniformity of N loading and water flux influences the performance of fixed-width buffers. In cases where the N loading and subsurface water flux is uniform along streams in the watershed, fixed-width buffers would be effective and there would be overlap with variable-width buffers. However, in our study area the N loading and water flux is spatially variable, rendering fixed-width buffers less effective than variable-width buffers.

Table 4.2 Summary results of N load reductions in fixed-width buffers.

<table>
<thead>
<tr>
<th>Buffer width</th>
<th>Buffer area (%)</th>
<th>Load reduction (% of baseline load)</th>
<th>Load reduction from removal in buffer (%)</th>
<th>Load reduction from LULC change (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>10</td>
<td>2.8</td>
<td>13.1</td>
<td>12.4</td>
<td>0.7</td>
</tr>
<tr>
<td>20</td>
<td>4.2</td>
<td>18.3</td>
<td>16.8</td>
<td>1.5</td>
</tr>
<tr>
<td>30</td>
<td>5.8</td>
<td>23.3</td>
<td>20.8</td>
<td>2.5</td>
</tr>
<tr>
<td>40</td>
<td>8.0</td>
<td>29.4</td>
<td>25.2</td>
<td>4.2</td>
</tr>
<tr>
<td>50</td>
<td>9.7</td>
<td>32.9</td>
<td>27.4</td>
<td>5.5</td>
</tr>
<tr>
<td>75</td>
<td>14.4</td>
<td>40.0</td>
<td>30.7</td>
<td>9.3</td>
</tr>
<tr>
<td>100</td>
<td>19.4</td>
<td>45.4</td>
<td>31.9</td>
<td>13.5</td>
</tr>
<tr>
<td>125</td>
<td>24.1</td>
<td>49.2</td>
<td>31.7</td>
<td>17.5</td>
</tr>
<tr>
<td>150</td>
<td>28.8</td>
<td>52.5</td>
<td>31.0</td>
<td>21.5</td>
</tr>
</tbody>
</table>
Figure 4.10 Restored fixed-width buffer (30m).
4.3.3 Variable-width Buffer Assessment

The model presented here prioritizes buffer locations based on the actual removal in a given grid cell, by considering removal in upslope buffered areas (the final model also takes into account removal in downslope buffered areas). The actual removal values for all suitable buffer locations are presented in Figure 4.11. The actual removal grid was ordered from lowest to highest and then divided into 100 slices (Figure 4.12). We prioritized cells based on the actual removal values and removed cells that were noncontiguous with the stream. A map of the contiguous and noncontiguous buffers is presented in Figure 4.13. After developing a relationship between buffer area and N removal, we were able to map buffers that remove a desired percentage of N from the watershed.

We performed seven variable-width buffers scenarios that removed 10, 20, 30, 40, 48.6, 50, and 90% of the baseline loading. The N load reductions from land-use change and removal within the buffer are summarized in Table 4.3. The N load reduction starts to level off around 50% removal. A map of predicted restored buffer locations, presented in Figure 4.14, shows the buffer area needed to meet the TMDL load reduction for the study area (48.6%).

In some cases, variable-width buffers were predicted in areas extending out from existing buffers. These specific locations of extended buffers may be wrong, because the existing buffers could be sufficient. This is a limitation of the intermediate version of the model that was used for this thesis. This effect is accounted for in the final version of the model, which accounts for N removal in both upslope and downslope direction as part of the prioritization iteration scheme. Downslope dependency is described in section 3.4.2.
The variable-width buffer tended to be in areas with a high water flux and high 
N loading. The low N fractional removal rates in these areas was outweighed by the 
high N loading and resulted in high expected N removal. These areas contribute 
significantly to streamflow and thus have a larger influence on water quality (Sweeney 
and Newbold, 2014).

Table 4.3 Summary results of N load reductions in variable-width buffers.

<table>
<thead>
<tr>
<th>Trial</th>
<th>Buffer area (%)</th>
<th>Net load reduction (% of baseline load)</th>
<th>Load reduction from removal in buffer (%)</th>
<th>Load reduction from LULC change (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>1.7</td>
<td>10.1</td>
<td>9.8</td>
<td>0.3</td>
</tr>
<tr>
<td>2</td>
<td>3.5</td>
<td>20.0</td>
<td>18.6</td>
<td>1.5</td>
</tr>
<tr>
<td>3</td>
<td>5.6</td>
<td>30.0</td>
<td>26.5</td>
<td>3.4</td>
</tr>
<tr>
<td>4</td>
<td>8.6</td>
<td>40.0</td>
<td>33.4</td>
<td>6.6</td>
</tr>
<tr>
<td>5</td>
<td>12.8</td>
<td>48.6</td>
<td>37.9</td>
<td>10.7</td>
</tr>
<tr>
<td>6</td>
<td>14.1</td>
<td>50.1</td>
<td>38.6</td>
<td>11.4</td>
</tr>
<tr>
<td>7</td>
<td>51.4</td>
<td>90.3</td>
<td>23.9</td>
<td>66.4</td>
</tr>
</tbody>
</table>
Figure 4.11 N removal calculated from fractional N removal and N loading, for areas suitable for potential buffer restoration.
Figure 4.12 Ranking of restoration prioritization, through statistical slicing of N removal rates. Slice zones with a high value (red) indicate high priority areas.
Figure 4.13 Map showing contiguous and noncontiguous restored variable-width buffers.
Figure 4.14 Location of restored variable-width buffers, needed to meet TMDL load reduction.
4.3.4 Comparing Fixed- and Variable-width Restoration

Benefits from both fixed- and variable-width buffers decreases as buffer area increased, evident in the logarithmic shape of the relationships (Figure 4.15). Variable-width buffers, however, exhibited a higher N removal capacity, compared to fixed-width buffers. For example, to reach the TMDL load reduction (48.6%), variable-width buffers would require 40% less area compared to fixed-width buffers. The results presented in this thesis are for the intermediate version of the model. For the final model, there is even more difference between fixed-width and variable-width buffers. The difference between fixed- and variable-width buffers increases with larger N load reductions.

The N load reduction can be attributed to either LULC change or N removal in the buffer. N removal in the buffer is relatively more important for narrower buffers, and any additional N load reduction from widening the buffer beyond 50m will largely be from LULC N yield reductions rather than from N removal in the buffer (Figure 4.16). Interestingly, the N load reduction due to N removal actually decreases with width for buffers wider than 75m. One possible explanation is that with lower N yield from LULC change, there would be less N removed in the buffer.
Figure 4.15 N load reduction vs. added buffer area, comparing fixed- and variable-width buffers.
Figure 4.16 N load reduction for fixed-width buffers, comparing contribution from LULC change and removal within the buffer.

4.4 Model Evaluation

4.4.1 Model Limitations

The riparian buffer N removal model is limited by several assumptions. We assume a single base flow depth of 0.2 m/yr for the entire watershed. This may not be the case for every cell in the watershed. Incorporating the slope of the terrain and geologic data could improve the accuracy of the water flux. Also, as previously mentioned, alternative multiple direction flow routing algorithms that disperse the flow more could also improve the accuracy of the water flux. Incorporating other factors such as depth to the water table and soil characteristics may also improve water flux estimation. It is also important to mention that in our model, water flux is
considered the main factor controlling N removal and may not be strongly correlated with N removal in some cases.

4.4.2 Additional Applications and Future Research

The riparian buffer N removal model can evaluate different buffer restoration strategies. A combined, fixed/variable-width buffer restoration strategy was evaluated by Luc Claessens using the fully functional model. He evaluated a minimum 10m fixed-width buffer, which in addition to N removal would provide stream temperature moderation, bank stability, and provision for aquatic food habitat and energy supply. The results showed that this combined buffer design was more effective at removing N than the fixed-width buffer design, but not as effective as the variable-width buffer design per unit buffer area.

Our model could also be used to prioritize conservation of existing buffers. The prioritization method used for restoring variable-width buffers can also be applied to existing buffers to determine which existing buffers are most critical to conserve.

The model could also be applied on the watershed scale (e.g., the entire Red Clay Creek watershed), to determine where riparian restoration should be prioritized. Applying the model on a larger scale would provide a broader view and help decision makers prioritize restoration for the entire watershed. Similarly, for case specific targeted restoration, the model can be applied at a smaller scale.

Our model can be used to aid in BMP cost-benefit analyses. A large consideration for riparian buffer restoration is the cost to the landowner. In-depth cost-benefit studies consider sign-up incentives, installation costs, maintenance costs, and opportunity costs (Qiu and Dosskey, 2012; Trenholm et al., 2013). These costs are dependent on the area needed for buffering. For our study watershed R01 (25 km²),
our model estimated the area saved by using variable-width buffers instead of fixed-width buffers. This area can be converted into a cost savings. With buffer installation costs ranging from $218-729 per acre (Lynch and Tjaden, 2000), landowners would collectively save approximately $150,000-$500,000 to meet the TMDL N load reduction. This cost estimate is only for planting and maintenance and does not include opportunity costs and transaction costs. In other words, the total costs savings from using a variable-width buffer approach are substantial. Using our model, common practice and regulations of restoration projects may shift towards promoting variable-width buffers.

Future research efforts are needed to verify the model. Modeled estimates of N loading could be compared against empirical N loading from water quality sampling, for watersheds with varying levels of buffering. This work is currently being conducted by Luc Claessens using data collected across the White Clay Creek watershed. This verification, along with sensitivity analysis of model parameters would provide a measure of model robustness.
Chapter 5

CONCLUSIONS

The goal of our research was to improve riparian buffer restoration for reducing N loading. We developed a simple spatially explicit GIS model that estimates the N removal within riparian buffers and that can prioritize locations for variable-width buffers. Our study presents a simple GIS-based spatially distributed model that estimates N removal in forested riparian buffers. Our model builds on previous research, including an empirical model that relates water flux to N removal rate (Sweeney and Newbold, 2014). The N removal rate, buffer width, and N loading are incorporated into the calculation of N removal. Our model calculates spatially distributed N removal, where calculations are performed in combination with the flow routing between the grid cells. We evaluated N removal for different model scenarios, including existing, fixed-width, and variable-width buffer scenarios. The variable-width buffers were mapped in areas where we expect buffers to become large N sinks, using an innovative spatially explicit prioritization scheme accounting for removal in both upslope and downslope areas. This is an improvement on current models, which are not spatially distributed and rarely link buffer area to N removal.

Our results show a higher N removal per unit buffer area in variable-width buffers compared to fixed-width buffers. Variable-width buffers would require less land to produce the same removal as a fixed-width buffer, thus reducing buffer installation and maintenance costs and cropland conversion. Variable-width buffers are often placed in areas where there is a high water flux and high N loading.
Watershed managers can use our decision support tool to prioritize buffer areas needed to achieve a desired N removal. Our model provides an interpretive aid and should be combined with site evaluations for precise delineation of buffers.

The combined fixed/variable-width approach may be the most appealing scenario for watershed managers. A 10m buffer along the stream should provide sufficient bank stability, stream shading, and habitat, while wider buffers are only applied where necessary to optimize N removal. Our model may be used to identify areas where riparian buffers are not practical, such as areas where wide buffers significantly interfere with farming activities. In these cases, alternate BMPs should be considered (e.g., permeable reactive barriers or treatment wetlands). Riparian buffers may take a long time to reach maturity and should be one component of a comprehensive watershed management plan to meet N load reduction goals in a timely manner.

Our model will help watershed managers make well-informed decisions to effectively manage riparian buffer restoration and conservation. With buffer placement prioritized in areas of high N removal, riparian buffers become more cost-effective and more desirable for landowners. As riparian buffers become more widely used, particularly in agricultural areas, significantly reductions in instream N loading could be achieved which would improve water quality.
REFERENCES


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Appendix A

PRELIMINARY ANALYSIS

A.1 Preliminary Methods

As a first-attempt at developing a spatially distributed model, we conducted a preliminary analysis. We tested a simple semi-distributed approach where N removal was calculated for each discrete riparian grid cell. This approach was based on previous studies that compared buffer function along the stream in a one-dimensional scheme (Baker et al., 2006; Rassam and Pagendam, 2009; Tomer et al., 2008). We first calculated N removal in existing buffers to prioritize stream segments for conservation. Then, we calculated the buffer width needed along each stream segment to attain a desired N removal.

A.1.1 Existing Buffer Assessment

We evaluated N removal in existing buffers. We calculated the buffer width for each riparian grid cell using the Flow Length tool weighted by existing buffer cells, similar to a method used by Baker et al. (2006). This tool uses the flow direction to calculate the accumulated distance through buffered cells upslope from each cell. For this approach, we were only concerned with the weighted flow length for each riparian cell. The previously calculated buffer width and the water flux were used to calculate N removal efficiency using the equation:

\[ E_N = 1 - \exp\left(-\frac{\alpha w}{q_L}\right) \]  

(A1)
We used the *D-infinity Contributing Area* tool in the TauDEM Toolbox weighted by the N loading grid to accumulate the N loading through the watershed. We multiplied this loading by the N removal efficiency to get the actual removal for each riparian cell. This removal was aggregated to determine the total removal within the subwatershed.

### A.1.2 Variable-width Buffer Assessment

For the one-dimensional restoration analysis, we rearranged equation A1 to solve for buffer width.

\[
w = \frac{- \ln(1 - E_N) \cdot q_L}{\alpha}
\]  

(A2)

Sweeney and Newbold (2014) proposed that an appropriate buffer width could be calculated at the watershed scale if the average water flux and the desired N removal was known. Thus, knowing the water flux for each riparian grid cell, we calculated the appropriate buffer width for that grid cell. We set the target N removal efficiency (\(E_N\)) for each riparian cell to match the TMDL load reduction value (48.6%). The removal factor \(\alpha\) was set to 2.72 (Sweeney and Newbold, 2014). This resulted in a map of “desired” buffer width values for each riparian cell.

### A.2 Preliminary Results

The preliminary analysis was a stepping stone towards the final model. This preliminary analysis was semi-distributed, as buffer width was calculated using the flow through the buffer, similar to the approach used in the Riparian Analysis Toolbox developed by Baker (2010). Water flux and N loading values however, were only one-dimensional, where removal calculations were performed on the riparian cells.
We reversed the N removal calculations to calculate the buffer width needed for each riparian grid cell to reach the TMDL load reduction. The required buffer width for each riparian cell is shown in Figure A1. The majority of riparian cells require a buffer width less than 10 meters. In some cases, forests or wetlands are already in places where a large buffer width is required. It should be noted that extremely high buffer width requirements at the stream heads are based on ephemeral stream water flux rather than lateral water flux. To remove these extremely high points at the start of the streams, the threshold for stream definition should be lowered. This analysis is similar to approaches used in other studies (Rassam and Pagendam, 2009; Tomer et al., 2008). This method is useful when prioritizing stream sections for buffering and is also practical since buffer width is a commonly understood buffer characteristic. However, prioritizing buffer restoration based on stream sections does not capture variability at finer spatial scales. Therefore, we decided to develop a spatially explicit model that use a two-dimensional analysis, which would more accurately assess the N removal of riparian buffers.
Figure A1 Required buffer width to meet TMDL N load reduction, using a simple 1-D model developed during preliminary analysis.
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