Identifying Riparian Buffer Effects on Stream Nitrogen in Southeastern Coastal Plain Watersheds

Abstract

Within the Southeastern (SE) Coastal Plain of the U.S., numerous freshwaters and estuaries experience eutrophication with significant nutrient contributions by agricultural non-point sources (NPS). Riparian buffers are often used to reduce agricultural NPS yet the effect of buffers in the watershed is difficult to quantify. Using Corrected Akaike Information Criterion (AIC_c) and model averaging, we compared flow-path riparian buffer models with land use/land cover (LULC) models in 24 watersheds from the SE Coastal Plain to determine the ability of riparian buffers to reduce or mitigate stream total nitrogen concentrations (TNC). Additional models considered the relative importance of headwaters and artificial agricultural drainage in the Coastal Plain. A buffer model which included cropland and non-buffered cropland best explained stream TNC ($r^2 = 0.75$) and was 5 times more likely to be the correct model than the LULC model. The model average predicted that current buffers removed 52% of nitrogen from the edge-of-field and 45% of potential nitrogen from the average SE Coastal Plain watershed. On average, 26% of stream nitrogen leaked through buffered cropland. Our study suggests that stream TNC could potentially be reduced by 34% if buffers were adequately restored on all cropland. Such estimates provide realistic expectations of nitrogen removal via buffers to watershed managers as they attempt to meet water quality goals. In addition, model comparisons of AIC_c values indicated that non-headwater buffers may contribute little to stream TNC. Model comparisons also indicated that artificial drainage should be considered when accessing buffers and stream nitrogen.

Keywords: Agricultural drainage, agricultural headwaters, flow-path analyses, riparian buffer metrics, total nitrogen.

Introduction

Degraded water quality due to nutrient enrichment is a major environmental concern across the United States. The eutrophication of lakes, rivers, and estuaries can threaten ecological systems and human health (Turner and Rabalis 1991; Vitousek et al. 1997; National Research Council 2000; State-EPA Nutrient Innovations Task Group 2009). Nonpoint sources (NPS) of nutrients from agriculture and urban areas often contribute significantly to eutrophication (Carpenter et al. 1998) but are difficult to identify, quantify, and manage. With anticipated increases in population and crop production for both food and biofuels, nutrient loads are expected to continue to increase (Galloway et al. 2008). Within the Southeastern (SE) Coastal Plain of the U.S., agricultural NPS contribute to eutrophication in numerous freshwaters and estuaries (Bricker et al. 2007; Hoos and McMahon 2009). Agricultural practices on the SE Coastal Plain include industrialized farming of soybean, corn, wheat, cotton, and tobacco (USDA-NASS 2009). In recent decades, confined animal feeding operations (CAFOs) for swine and poultry have expanded rapidly (Mallin and Cahoon 2003) and NPS from the applied manure has been implicated in the rise of various nutrients within coastal waters (Burkholder et al. 2006). Nitrogen NPS via atmospheric deposition from industry, transportation, and agricultural practices have also been shown to exert a strong influence on nitrogen concentrations in streams in some models (Hoos and McMahon 2009).

Within watersheds, the management of NPS nutrient pollution occurs at a local scale and has relied primarily on the implementation of best management practices, including the use of riparian buffers. For the purposes of this paper, we define riparian buffers broadly as any non-cultivated land cover (forest, shrubland, grassland, or wetland) adjacent to water bodies. In transect studies, riparian buffers have generally been shown to efficiently reduce NPS nitrogen loads though there is wide variation in removal efficiencies (Mayer et al. 2007; Vidon et al. 2010). This removal is touted as one of many ecosystem services provided by riparian buffers (Brauman et al. 2007).

Based on these transect studies, state and federal riparian programs have been implemented across large watersheds to help states and watershed managers meet nitrogen target reductions. For example, selected watersheds in North Carolina have enrolled 12,900 of a desired 40,500 hectares in stream buffer programs as part of a larger effort to reduce nitrogen by 30 percent (NCDENR 2007). Similar riparian efforts are emphasized in the Chesapeake Bay Program with its multi-state nutrient reduction strategy (www.chesapeakebay.net). Despite the general effectiveness at the field scale and the broad implementation of stream buffers, it is difficult to explicitly quantify the water quality benefits of buffers at a watershed scale (Sutton et al. 2009; Vidon et al. 2010; Tomer and Locke 2011). Yet realistic estimates of buffer effectiveness for watersheds are needed to ensure programs can meet desired nutrient target goals, especially as nutrient trading schemes become more prevalent and available conservation resources grow more limited.

When considering nitrogen at the watershed scale, many studies have correlated NPS nitrogen to watershed-scale metrics of land use / land cover (LULC), such as percent agriculture in the watershed (Liu et al. 2000; Jones et al. 2001; McMahon et al. 2003; Weller et al. 2003; Alexander et al. 2008; Hoos and McMahon 2009). Research specifically interested in riparian areas has correlated nitrogen to LULC within specific distances from the stream (Osborne and Wiley 1988; Johnson et al. 1997; Dodds and Oakes 2008). However, it is difficult to separate buffer effects from watershed effects in correlation analyses since the effect of buffers is inherently included within the watershed LULC metric coefficients (Weller et al. 2011). In an attempt to more accurately depict the source and flow of NPS nitrogen and its interception by buffers (or lack thereof), flow-path buffer models have been developed and incorporated into Geographic Information System (GIS) tools (Baker et al. 2006). Resulting flow-path buffer metrics emphasize the degree of connection between nitrogen sources and the stream (e.g. % of non-buffered cropland in the watershed). Weller et al. (2011) combined these flow-path buffer metrics with LULC metrics in *a priori* hypotheses in the Chesapeake Bay watershed. They tested resulting regressions using corrected Akaike Information Criterion (AIC_c) to identify the most likely models. Resulting models separated the effects of buffers on mean stream nitrate concentrations from the nitrate contributed by all cropland for watersheds within the Chesapeake Bay.

In this paper, we apply the analytical approach of Weller et al. (2011) to watersheds within the SE Coastal Plain to determine the effects of buffers on stream nitrogen concentrations in the region. We first consider the LULC nitrogen sources of cropland, pasture, urban, CAFOs, and atmospheric deposition that past SE Coastal Plain research has linked to stream nitrogen concentrations (McMahon et al. 2003; Hoos and McMahon 2009; Rothenberger et al. 2009). We then incorporate flow path buffer metrics into the model to differentiate between crop and buffer effects on stream nitrogen concentrations. Resulting coefficients can provide realistic watershed estimates of buffer effectiveness that can better inform nutrient management within the region.

Apart from separating buffer effects from cropland effects, the analytical approach allows us to make additional AIC_c comparisons to determine the relative importance of spatial drivers of NPS nitrogen and buffers. For example, headwaters (defined here for simplicity as areas draining into first order streams) have been shown to have an important influence on stream chemistry due to their abundant, close connection to the terrestrial landscape and their sensitivity to disturbances and pollutants (Lowe and Likens 2005; Alexander et al. 2007). Due to smaller stream size, accessibility and ease of agricultural practices, headwaters may have fewer buffers than higher order streams and rivers (Tomer et al. 2003; Baker et al. 2007). By comparing AIC_c values from flow-path buffer metrics in headwaters versus the entire watershed, we may be able to identify the relative importance of buffers in headwaters and higher order streams and rivers in determining overall stream nitrogen concentrations.

Similarly, this comparative AIC_c approach may also identify the importance of artificial agricultural drainage on nitrogen concentrations. Nitrate is soluble in water and can pass from the soil and into agricultural tile and ditch lines during rain events (Randall and Goss 2008). Numerous water quality studies have shown the impact of artificial drainage on nitrogen transport at local scales (Dean and Foran 1992; Randall et al. 2000; Cook and Baker 2001; Randall and Goss 2008; Cuadra and Vidon 2011), including in the SE Coastal Plain (Jacobs and Gilliam 1985; Tesoriero et al. 2005; Harden and Spruill 2008). Studies and models at broader scales have implicated artificial drainage as an important driver of nitrogen concentrations (Petrolina and Gowda 2006; Royer et al. 2006; Alexander et al. 2008; David et al. 2010). Artificial drainage occurs in the SE Coastal Plain and is prevalent on the outer Coastal Plains of North Carolina. Riparian buffers can be rendered ineffective when ditches and tiles transport nitrate-laden water past buffers directly into streams (Jacobs and Gilliam 1985; Gold et al. 2001; Harden and Spruill 2008). In this analysis, we identify cropland that has potential artificial drainage and could bypass buffers, and then combine those drained croplands with non-buffered cropland to provide an estimate of total cropland that is not benefiting from buffers. AIC_c comparisons of models with and without drained cropland may indicate the relative importance of artificial drainage through buffers on SE Coastal Plain stream nitrogen concentrations.

We aim to test the relevance of buffer models and identify the effect of riparian buffers on watershed nitrogen concentrations within the SE Coastal Plain. With resulting models we hope to estimate the watershed-scale effectiveness of buffers in the region. Additionally, we investigate the relative importance of buffers in headwaters and artificial drainage in determining stream nitrogen concentrations in the SE Coastal Plain.

Methods

Study Areas and Underlying Data

The study watersheds are found within watersheds of six main river systems (USGS HUC 0301-0306), namely the Savannah River, Santee-Edisto Rivers, the Pee Dee River, the Cape Fear River, and the rivers of the Albemarle-Pamlico Sound (Fig. 1). Within those main river systems, row crop agriculture takes place predominantly on the Coastal Plain which consists of the Level III ecoregions of the Mid-Atlantic Coastal Plain, the Southern Coastal Plain and the Southeastern Plains (Omernick 2004). The region of interest hereafter will be referred to as the SE Coastal Plain. The region consists of rolling to flat stair-stepped plains that once formed various paleoshorelines along the Atlantic coast. Dominated by sandy to silty soils with variable drainage and a shallow water table, subsurface flows in the Coastal Plain are estimated to contribute up to 70% of stream flows (Tesoriero et al. 2005). Land use in the region is predominantly a mix of row crop agriculture, livestock facilities, pasture, pine plantations, woodlands and wetlands. There are numerous coastal urban areas within the SE Coastal Plain but no large urban centers were included in the final study watersheds.

Total nitrogen concentrations (TNC)(mg N/L) for water quality sampling locations in the SE Coastal Plain were compiled from USGS and state water quality databases by the U.S. Geological Survey's (USGS) Southeast SPAtially Referenced Regression On Watershed attributes (SPARROW) project. From available samples, USGS SPARROW calculated the average annual TNC and detrended those values to the year 2002 to remove bias from different sampling years and to align the water quality data with the approximate years of LULC (see Hoos et al. 2008 for details). In the SE Coastal Plain, USGS SPARROW provided 154 locations with detrended TNC data. From those locations, we limited our study to: 1) watersheds where a large proportion of the watershed was located in the SE Coastal Plain, 2) watershed area was greater than 100 km^2, and 3) the data record for TNC included at least 3 years between 1997 and 2003 and at least 7 observations per year. If multiple selected watersheds were nested within each other, the most data rich watershed was chosen so that only independent watersheds were considered in the statistical analyses. The above criteria resulted in 24 watersheds (Fig. 1) and associated general attributes are provided in Table 1.

Figure 1 approximately here

Table 1 approximately here

Digital Elevation Models (DEMs) with 10 m resolution were downloaded from USGS (http://seamless.usgs.gov) for the region. The stream layer consisted of the National Hydrography Dataset (NHD) - high resolution (1:24,000 or better; http://nhd.usgs.gov) stream lines that we modified to include the edges of connected NHD water bodies (i.e. reservoirs, large rivers or ponds) in order to accurately identify riparian areas found at water body edges. The 10 m DEM was modified to "burn" in the stream layer using the ArcHydro reconditioning function within ArcMap 9.3 (ESRI Inc., Redmond CA.) to minimize differences in flow-path from the DEM and the NHD. The DEM was then "filled" to ensure the flow of all upland cells into the stream network. Watersheds for each data point were delineated and area was calculated using ArcHydro tools.

Metric Development

Basic Inputs

For each watershed, all LULC metrics were created using the National Land Cover Dataset (NLCD) 2001 (Homer et al. 2007) and were converted to a 10 m grid to match the 10 m DEMs. The Analytical Tools Interface for Landscape Assessments (ATtILA) (Ebert and Wade 2004), an ArcView extension (ESRI Inc, Redlands, CA), was used to generate all LULC metrics. LULC metrics included were percent cropland (Cl), percent pasture (Pa), and percent urban (Ur).

The density of CAFOs was generated from data provided by state agency permitting records. Only medium and large CAFOs were included in the permitting records and are defined as bovine CAFOs with more than 300 animals, swine CAFOs with more than 750 animals, and poultry CAFOs with more than 9,000 animals (USEPA, 2008). The selected CAFOs were summed for each watershed and divided by the total area of the watershed to provide the metric for CAFO densities (Cafo).

Annual wet and dry deposition of total oxidized and reduced nitrogen (kg-N/ha/yr) was provided from the Community Multiscale Air Quality (CMAQ) model (Byun and Schere 2006; Schwede et al.

2009). CMAQ is a process based air quality model that inputs meteorological model data and source emissions rates to estimate transport, transformation and deposition of multiple air pollutants, including oxidized and reduced nitrogen species. Sources for wet and dry oxidized nitrogen include power plants, cars, trucks, diesel equipment, industry and space heating. Sources for wet and dry reduced nitrogen include agriculture, CAFOs and cars (Byun and Schere 2006). Total nitrogen deposition consisted of combining the wet and dry oxidized and reduced nitrogen species from CMAQ. The nitrogen wet deposited by CMAQ does not include organic nitrogen, only inorganic nitrogen. Organic nitrogen is estimated to constitute roughly 30% of the true, full total nitrogen wet deposition (Cape et al. 2011). CMAQ output is reported on a 12 km grid resolution which was weighted proportionally to the selected watersheds and an average deposition metric (Atm) was calculated.

Buffer Metric

Flow-path derived riparian data layers were generated using the riparian analysis tool (Baker et al. 2006). The tool connects NPS areas (i.e. croplands) to the existing stream network via DEM-generated flow-paths and enumerates the flow-path length of natural areas (buffers) that intersect the source flow-path prior to reaching the stream. The length of the buffer flow-path or buffer width is then assigned to the source area cells. The use of DEM-derived flow paths necessarily relies on the assumption that both surface and shallow subsurface flows follow the topological gradient. In the SE Coastal Plain, subsurface flows predominate but due to shallow confining layers we assume that flow-paths describe general connectivity between uplands and the stream network (Tesoriero et al. 2005). We ran the tool using cropland for source areas and natural areas (combined NLCD forest, shrubland, grassland, and wetland classes) adjacent to streams and other water bodies were treated as buffers. Urban classes were ignored in this flow-path metric as we were interested in determining the effect of buffers on agricultural nitrogen sources. The modified NHD and 10 m DEMs described earlier were used as inputs into the tool. From the tool, we identified those cropland cells that had a buffer width equal to 0 and calculated the desired metric: percent of the watershed with non-buffered cropland (NB).

Buffer Metric in Headwater Catchments

Headwater catchments were defined as those catchment areas within the first order of NHD-high (resolution of 1:24,000) streams (Strahler 1957). We identified all NHD-high first order stream reaches using the stream order tool in ArcGIS (ESRI Inc. Redlands, CA) using our modified DEM. We then used the watersheds tool in ArcHydro (ESRI Inc. Redlands, CA) and derived the contributing areas to the headwater streams for each of the 24 watersheds. Using that layer as a mask, we recalculated NB for the headwaters in each watershed (HwNB).

Artificial Agricultural Drainage

Few spatial maps of artificial agricultural drainage exist. A proxy for artificial drainage combines existing SSURGO drainage classes with agriculture NLCD classes (Sugg 2007). This proxy assumes that if the pixel is in active agriculture and has poor drainage then the farmer has modified the hydrology to allow for cultivation. Sugg (2007) used poor drainage classes from the State Soil Geographic (STATSGO) database and NLCD 1992 and found the layer generally agreed with the agricultural surveys performed by the U.S. Department of Agriculture. We converted SSURGO drainage classes from polygons into a 10 m raster grid and isolated those poorly and very poorly drained classes that coincided with the 2001 NLCD cropland to give an estimate of artificially drained cropland. We combined the drained cropland layer with the riparian tool output to identify drained cropland situated behind existing buffers. These drained and buffered croplands were combined with non-buffered cropland and summarized for each watershed to create the metric: percent drained and non buffered cropland (DNB).

Statistical Methods

Basic Inputs Model

The first model, termed the Basic Inputs (BI) model, included variables that have been shown to significantly influence nitrogen concentrations in the SE Coastal Plain, including: percent croplands (Cl), percent pasture (Pa) which often receives livestock manure, percent urban lands (Ur), medium and large CAFOs (Cafo), and atmospheric deposition (Atm). The BI model was the following:

$$TNC = \beta_0 + \beta_{cl}Cl + \beta_{pa}Pa + \_\beta_{ur}Ur + \beta_{cafo}Cafo + \beta_{Atm}Atm + \varepsilon \quad \text{(BI)}$$

where TNC is the total nitrogen concentration, β_0 is the intercept or the background nitrogen concentrations, β with the variable subscripts are the fitted model coefficients, and ε is error. Coefficient estimates of the BI model indicated that only the cropland coefficient (β_{cl}) was significant ($t=4.90$, $p<0.001$) and so a Parsimonious Basic Inputs (PBI) model:

$$TNC = \beta_0 + \beta_{cl}Cl + \varepsilon \qquad \text{(PBI)}$$

was considered as well.

Buffer Model

The riparian analysis tool identifies which cropland is not buffered at a 30 m resolution. Weller et al. (2011) determined that cropland sources to stream nitrogen could be separated into two components, nitrogen leached from all cropland ($\beta_{cl}Cl$) and additional nitrogen lost from croplands with no buffer ($\beta_{nb}NB$). The two variables, Cl and NB, are correlated so we must test for collinearity effects on model coefficient stability (see validation section below). The NB variable was incorporated into the basic inputs model making the Buffer model (BF):

$$TNC = \beta_0 + \beta_{cl}Cl + \beta_{NB}NB + \beta_{pa}Pa +\_\beta_{ur}Ur + \beta_{cafo}Cafo + \beta_{Atm}Atm + \varepsilon \qquad \text{(BF)}$$

where β_{NB} is the coefficient for non-buffered cropland and all other coefficients are as described earlier. The NB variable was also incorporated into the PBI model to make the Parsimonious Buffer model (PBF):

$$TNC = \beta_0 + \beta_{cl}Cl + \beta_{NB}NB + \varepsilon. \qquad \text{(PBF)}$$

We used a multiple regression (GLMSELECT; SAS® 9.2; Cary NC) to fit all the regression models.

Model Average

The corrected Akaike Information Criterion (AIC_c) was used to compare, rank, and select the above model(s) that best explained the variation in the data while penalizing for greater numbers of variables. Models (BI, PBI, BF, PBF) were compared to the model with the lowest AIC_c value by quantifying the difference, delta (Δ), between models. A $\Delta<2$ indicates models with similar ability to explain variation in the data, $2<\Delta<4$ has some support, while $\Delta>10$ indicates a strong preference for the lower AIC_c model (Burnham and Anderson 2002; Weller et al. 2011). Akaike weights were calculated for those models with $\Delta<10$, and those models with a weight > 10% were then retained in creating a model average (MA) for coefficients (Royal 1997). Though detailed in the results section below, Akaike weights determined that the MA include:

$$TNC = \beta_0 + \beta_{cl}Cl + \beta_{NB}NB + \varepsilon. \qquad \text{(MA)}$$

Coefficients within each likely model were multiplied by their respective model weights and then summed to create the coefficient estimates for the MA. Variance around each MA coefficient was the combined weighted variance of individual models plus deviation from the MA (Burnham and Anderson 2002; Weller et al. 2011).

Validation

High correlation between predictors in the models can lead to overfitting of the model and poor characterization of model coefficients. Tests were performed to ensure that collinearity was not strongly influencing model results. First, a correlation analysis was performed using PROC CORR (SAS® 9.2; Cary NC) and variables with relationships where $r > 0.70$ were flagged for possible effects of collinearity. Variance Inflation Factors (VIF) were examined for each model coefficient. VIF measures the amount of inflation in the standard error of a coefficient if it is correlated with other coefficients. Model coefficients with a VIF<10 are considered to be stable (Allison 1999). Finally, we performed jack-knife analyses for cross validation on the likely models as well as the MA to determine the reliability of the model coefficients.

Average Watershed Nitrogen Estimates

Weller et al. (2011) reasoned that the coefficients from the MA could be combined with average watershed proportions to estimate concentration amounts of nitrate sources and nitrate removal via riparian buffers for the Chesapeake Bay watershed (see Weller et al. 2011, fig. 2 & 6 and eq. 13). Therefore if we use average percentages for our 24 SE Coastal Plain watershed variables (Table 1), $\beta_{cl}Cl$ is the amount of nitrogen released from all cropland. On buffered cropland (BCl; calculated as Cl – NB) in the average watershed, $\beta_{cl}BCl$ can be considered the amount of nitrogen that would leak through buffers to the stream. On non-buffered cropland, $\beta_{cl}NB + \beta_{NB}NB$ estimates the total nitrogen entering the stream and $\beta_{NB}NB$ can be considered the amount of nitrogen leached due to lack of buffers or the additional nitrogen retained if buffers were restored on all non-buffered cropland. Finally, the current removal of nitrogen by buffers within the watershed can be estimated by multiplying β_{NB} (the coefficient of additional nitrogen leached from non-buffered cropland) by BCl (the amount of buffered cropland currently present). The percent removal by buffers can be calculated by taking β_{NB}BCl and dividing by the amount of nitrogen predicted at the edge of field (β_{Cl}Cl + β_{NB}NB + β_{NB}Cl).

Headwater Buffers Model

Headwater streams can constitute the majority of stream miles within a watershed yet riparian buffers may be less prevalent in headwaters (Baker et al. 2007). In the headwater buffers (HBF) model, we wish to test the relative importance of buffers in the headwaters. NB and HwNB metrics are too highly correlated to include in the same model or to be considered jointly in the model average but comparisons can be made between AIC_c values in separate regression models. The HBF model was set up as follows:

$$TNC = \beta_0 + \beta_{cl}Cl + \beta_{HwNB}HwNB + \varepsilon. \qquad \text{(HBF)}$$

where β_{HwNB} is the coefficient for non-buffered cropland in the headwaters and all other coefficients are as described earlier. If HBF has an equivalent or lower AICc than PBF, then this indicates that the headwaters model explains much of the variation in stream TNC and that buffers in higher- order streams and rivers may have little influence on stream nitrogen concentrations.

Drained and Non-Buffered Cropland Model

To address the hypothesis that artificially drained buffered cropland influence stream nitrogen concentrations, we developed the Drained and Non-Buffered Cropland model (DNBC):

$$TNC = \beta_0 + \beta_{cl}Cl + \beta_{DNB}DNB + \varepsilon. \qquad \text{(DNBC)}$$

where β_{DNB} is the coefficient for drained and non-buffered cropland and all other coefficients are as described earlier. As in the HBF model, if DNBC has a lower AICc than PBF, then this suggests that the drained component is important in describing stream nitrogen concentrations.

Results

Nitrogen Concentrations and Land Use / Land Cover Metrics

Average annual total nitrogen concentrations (TNC) for the 24 watersheds ranged from 0.28 mg/l to 1.98 mg/l, with a watershed mean TNC of 0.86 mg/l (Table 1). Cropland percentages averaged 21.5% and ranged from 1.9% up to 37.7%. Cropland was highly correlated to multiple variables including: CAFO densities, atmospheric deposition (Atm), non-buffered cropland (NB), non-buffered cropland in the headwaters (HwNB), and drained and non-buffered cropland (DNB)(Table 2). Pasture land and urban land averaged 5.9% and 5.2% of the watersheds respectively (Table 1). CAFO densities ranged from 0.0 to 0.22 CAFOs/km^2, with a mean density of 0.05 CAFOs/km^2 and a median density of 0.02 CAFOs/km^2 (Table 1). Atmospheric deposition was highly correlated to CAFO densities which could be expected given that CAFOs are an input into CMAQ models (Byun and Schere 2006). NB covered an average of 6.3% of the watersheds with a wide range of values across the watersheds from 0.2% in watershed 242 up to 19.3% of watershed 133 (Table 1). Interestingly, the average percentage of HwNB was 5.0%, which would account for 79% of the average non-buffered cropland in the entire watershed yet the average area of watershed headwaters was 58% (Table 1). NwNB was strongly correlated to NB in the watershed (r = 0.996). DNB had an average watershed coverage of 9.2% and ranged from 0.4% up to 25.2% of the entire watershed (Table 1). DNB was strongly correlated to NB (r = 0.977) and HwNB (r=0.972).

Table 2 here

Model Results

The results of the multiple regression models with their various input variables are found in Table 3. The BI model had an adjusted R^2 = 0.68 and an AIC_c value of -28.2 (Table 3). The cropland coefficient was significant (t = 4.90, p < 0.001) and the intercept value was marginally significant (t = 1.89, p = 0.07). Coefficients for pasture, urban, CAFO density, and atmospheric deposition were not significant contributors to the stream nitrogen concentrations (p > 0.23) and were not retained in the PBI model. The PBI model increased the adjusted R^2 to 0.69, decreased the AIC_c to -38.2 due to fewer variables (Table 3), and the cropland coefficient was significant (t = 7.21, p < 0.001).

The addition of the non-buffered cropland to the BI model to make the BF model explained more variation (adjusted-R^2 = 0.74) but had a higher AIC_c of -30.3 than the PBI due to the penalty of including 7 variables (Table 3). Of the inputs for the BF model, only the coefficient for NB was significant (t=2.34, p=0.03). The PBF model included cropland and non-buffered cropland and increased the adjusted R^2 to 0.75 while having the lowest AIC_c value of -41.65 (Table 3). Both the intercept and NB coefficients were significant (t=2.33, p=0.03; t=2.53, p=0.02, respectively) while the cropland coefficient was not significant (t=1.71, p=0.10).

Table 3 here

Model Averages and Validation

AIC_c deltas and their weights for each of the models indicated that only the PBI and PBF models should be considered for inclusion in the formation of the model average (MA) since they had $\Delta < 10$ and >10% weight (Royal 1997). Of those models included in the MA, the Akaike weights indicated that the PBF model was approximately 5 times more likely to be the correct model than the PBI model (Table 3). Applying the Akaike weights to the various coefficients of the three models resulted in the following MA equation for stream nitrogen concentrations (Table 4):

$$TNC = 0.229 + 0.017*Cl + 0.050*NB + \varepsilon.$$

The MA accounted for 73.2% of variation and adequately predicted observations for most watersheds though the confidence intervals for watershed 136 did not overlap with the 1:1 line (figure 2).

Table 4 and Figure 2 here

Since both coefficients, β_{Cl} and β_{NB}, incorporated cropland, VIF factors were analyzed to ensure that collinearity was not adversely affecting results. VIF for the three coefficients included in the MA showed that all VIF values were < 4.2 which is below the suggested allowable limit of 10 (Allison, 1999)(Table 5). To ensure reliable prediction of TNC, cross-validation with jack-knifing was used to see the cumulative effect of multiple "minus-one" runs on overall RMSE and adjusted R^2. The percent change in RMSE for the MA was increased by 14.9% and there was a loss of explained variation as the adjusted R^2 dropped 11.5% (Table 6). Overall, correlated variables included in the MA did not overly influence the stability of the MA coefficients. While TNC predictions were influenced by removing one site from a relatively small sample size of 24, the predictions remained relatively robust for the SE Coastal Plain.

Tables 5&6 here

Estimated Nitrogen Contributions

For the SE Coastal Plain, estimated contributions of the various MA components to average stream nitrogen are included in Table 7 and are estimates based on the coefficients of the MA multiplied by the average watershed variable. The MA predicts that average watershed non-crop sources (β_0) account for 0.23 mg N/L, which is 25% of estimated stream TNC. On average 29% of stream nitrogen leaks through buffered cropland, adding an estimated 0.26 mg N/L to streams. Non-buffered cropland contributes 0.41 mg N/L (46%) to SE Coastal Plain streams with 0.31 mg N/L (34%) of that average stream nitrogen attributed to a lack of buffers. Therefore, that same amount might be attenuated if adequate buffers were restored on all cropland. Conversely, 66% of average stream nitrogen is not amenable to reductions through agricultural buffer restorations as it leaks through buffers or comes from other sources. Current buffers are expected to remove 0.75 mg N/L or 52% of nitrogen estimated to come from the edge of cropland fields. Of the potential nitrogen an average SE Coastal Plain stream might receive, buffers from the watershed are estimated to attenuate 45% of that potential nitrogen. If buffers were placed on all croplands, those buffers would be predicted to attenuate 64% of potential nitrogen before it reached the stream, assuming that restored buffers were equally efficient as current buffers.

Table 7 here

HBF and DNBC Model Comparisons

Regression analysis for the HBF model resulted in a significant model ($F=36.32$, $p<.001$) with the HwNB coefficient as significant ($t=2.62$, $p=0.02$). The HBF model had the same adjusted R^2 value as the PBF model, 0.75, and a slightly lower AIC_c, -42.06 versus -41.65 (Table 8).. The DNBC model was significant ($F=32.39$, $p<0.001$) and included a significant DNB coefficient ($t=2.13$, $p=0.05$). The DNBC model had an adjusted $R^2=0.73$ and an AIC_c value of -39.96 (Table 8), which has a delta of 2.1 with HBF and a delta of 1.7 with PBF.

Discussion

Buffer Models and Efficiency

Riparian buffers on croplands in the SE Coastal Plain can have a strong effect on stream nitrogen concentrations. Buffer models explained 5-7% more variation in stream TNC than watershed LULC models and the parsimonious buffer model was 5 times more likely to be correct than the parsimonious basic input model. Weller et al. (2011) found even stronger support for buffer models in the Chesapeake Bay where no LULC models received any weight in their model average though increases in R^2 for the buffer models were small (1.5-2.0%). LULC models have linked agriculture to total nitrogen variables in past findings from the SE Coastal Plain (McMahon and Woodside 1997; McMahon et al. 2003; Hoos and McMahon 2009; Rothenberger et al. 2009). By incorporating spatially-explicit source-to-buffer metrics into models alongside LULC metrics, while checking for collinearity, we are able to quantify buffer coefficients that describe how agricultural nitrogen interacts with buffer spatial patterns to contribute to stream nitrogen concentrations (Weller et al. 2011). Models that incorporate buffers should be considered when attempting to model, understand, and manage these agriculturally dominated landscapes.

Buffers have been used as a best management practice by managers to reduce nitrogen losses at field edges. Despite the generally high efficiency of removal generally shown in field studies (see Mayer et al. 2007 for review), there is wide variation in removal efficiencies due to landscape position, soils, and subsurface flows (Young and Briggs 2005; Vidon and Hill, 2006; Harden and Spruill, 2008; Vidon et al., 2010). Some studies have cautioned that transect studies which focus on overland sheet flow may overestimate watershed buffer efficiencies as concentrated flows or subsurface flows that produce lower efficiencies may be prevalent in many circumstances (Dosskey et al. 2002, Newbold et al. 2010, Pankau et al. 2012). Empirical estimates of effective watershed buffers have been difficult to document; possibly due to lag times, inadequate sampling designs, or inadequate contrasts between paired watersheds (Sutton et al. 2009, Tomer and Locke 2011). The current analysis provided estimates of removal efficiencies by agricultural buffers for the SE Coastal Plain: 53% removal of edge-of-field nitrogen and 45% removal of all watershed nitrogen. These values are lower than many transect studies but may better reflect the efficiencies of buffers across the watershed. Very similar estimates of overall edge-of field buffer removal (50%) and watershed (40%) efficiencies for nitrate were found in Coastal Plain watersheds in the Chesapeake Bay (Weller et al. 2011). Such estimates provide more realistic expectations of nitrogen removal to watershed managers as they attempt to meet water quality goals and become involved in potential nutrient trading markets.

When breaking down the removal efficiency in this study and Weller et al. (2011), it is interesting to note that β_{NB} in our study was only half of β_{NB} in the Mid-Atlantic Coastal Plain, 0.050 versus .101. However, our watersheds contained twice as much buffered cropland, 15.2% versus 7.2% so overall watershed estimates of buffer removal efficiencies were similar. The buffer efficiencies reported here highlight the limits of buffers in controlling stream nitrogen. This study predicted that 39% of average stream nitrogen comes from all cropland regardless of whether it was buffered or not. This leakage value is higher than the estimated value of 7% calculated for nitrate by Weller et al. (2011) for the Coastal Plain in the Chesapeake Bay. The differences in the two studies for removal efficiency and leakage estimates could possibly be due to the increased heterogeneity of the watersheds and their underlying geology in this study. Many of the Chesapeake Bay Coastal Plain watersheds used by Weller et al. (2011) were small and clustered on the Mid-Atlantic Coastal Plain just to the west of the bay in Maryland. Conversely, watersheds in the current study were large, spread across the Mid-Atlantic Coastal Plain, the Southern Coastal Plain and the Southeastern Plains with a few of the upper portions of larger watersheds extending into the Piedmont ecoregion (Omernick 2004). Indeed, buffer leakage estimates for Piedmont watersheds in Weller et al. (2011) constituted 66% of stream nitrate concentrations. Thus our estimates of leakage across these larger watersheds might reflect an intermediate buffer leakage value that incorporates a broader influence of subsurface pathways passing through the coarser substrates of the inner Coastal Plain and fractured substrates of the Piedmont (Harden and Spruill 2008; Tesoriero et al. 2009). It should be noted as well that Weller et al. (2011) considered nitrate concentrations while our study considered total nitrogen concentrations. Organic nitrogen can often be a substantial component of total nitrogen in SE Coastal Plain streams (Stow et al. 2001, Rothenberger et al. 2009) and thus high leakage values may also reflect organic nitrogen inputs from agriculture and the riparian areas themselves (Vidon et al. 2010).

Despite buffer limitations, Coastal Plain buffers are still estimated to remove over half of edge-of-field nitrogen contributions. However, the current buffer estimates should not be applied outside the Coastal Plain as Weller et al. (2011) showed that buffer estimates vary greatly between ecoregions. The buffer estimates could be also strengthened by increasing the number of watersheds included in future analyses. Cross-validation showed reductions in explained variance and greater sample size could reduce uncertainty in the estimates. The range of TNC values was also limited in these large watersheds reaching only a maximum average value of 1.98 mg N/L while Weller et al. (2011) in their much smaller watersheds saw numerous Coastal Plain nitrate concentrations above 5 mg N/L though their average value was similar to this study. Caution should be applied when applying these buffer efficiencies to watersheds that have much higher stream TNC than were observed in this study.

Further research on watershed buffer efficiency should also involve better classification of buffers. The definition for buffers within this study and Weller et al. (2011) was any natural vegetation adjacent to water bodies. Future buffer classifications could include incorporating soils data that focuses on biogeochemical hotspots for nitrogen removal (Rosenblatt et al. 2001; Young and Briggs 2007; Kellogg et al. 2010; Vidon et al. 2010). Buffer classification could also be improved via finer resolution LULC data. The 30 m NLCD data used here does not identify narrow buffers (<15m) yet these narrow buffers may be prevalent in agricultural settings (Floyd et al. 2009) and can be effective nitrogen sinks (Meyer et al. 2007). Overall, the study's findings support the use of buffers as a best management practice to lower stream nitrogen concentrations in the Coastal Plain. However, these findings also highlight the importance of using buffers as one tool of many BMPs implemented in a watershed management strategy to reduce nitrogen concentrations. In-field management of nutrient application, timing, and location should be considered in conjunction with riparian buffers.

CAFOs and Atmospheric Deposition

CAFOs and atmospheric deposition were not considered significant contributors to among-watershed variation in stream TNC in this dataset. In part, this may be a reflection of the small sample size of our dataset. These models are intended to capture the major patterns of TNC across large watersheds and do not have the statistical power to evaluate the importance of each variable. Literature has supported a relationship between CAFOs and nitrogen at the local scale (Stone et al. 1998; Sloan et al. 1999; Karr et al. 2001; Stone et al. 2004, Israel et al. 2005; Whalen and DeBerardinis 2007) and the watershed scale (Glasgow and Burkholder 2000; Weldon and Hornbuckle 2006; Rothenberger et al. 2009; Ciparis et al. 2012). Since the 1990s, CAFOs have trended toward larger facilities that are concentrated in specific regions (Kellogg et al.2000). In the SE Coastal Plain, CAFOs demonstrate this spatially concentrated pattern. For example, 59% of 8.9 million swine raised in North Carolina in 2011 were produced in 4 counties (8,300 km^2) on the Coastal Plain (www.ncagr.gov/stats/). Regression relationships in past watershed CAFO studies were typically dominated by relatively few sub-watersheds with much higher CAFO densities (Weldon and Hornbuckle; Rothenberger et al. 2009; Ciparis et al. 2012). Several watersheds (e.g. 86 and 87) in high density CAFO counties did have high CAFO-to-TNC relationships yet other watersheds (e.g. 133) were outside the major CAFO counties but still had high non-buffered cropland and high TNC. Manure is generally applied onto nearby agricultural fields in wet or dry form (Read et al. 2008) so although CAFO density was used in this study, the influence of CAFOs is likely to be reflected in the cropland coefficients. Agricultural fields receiving manure applications are regulated by state or federal permits that mandate nutrient waste management plans and buffered setbacks from streams and tile inlets. While CAFOs were not significant in our regression analysis across the SE Coastal Plain, in those areas where CAFOs are highly concentrated, further research and nutrient budgets are needed under current CAFO nutrient management plans to ensure that surface waters are adequately protected from nitrogen pollution (Sloan et al. 1999; Israel et al. 2005; Whalen and DeBerardinis 2007).

Atmospheric deposition has been a highly significant input of nitrogen in previous models in the region as well (McMahon and Woodside 1997; Jones et al. 2001; Alexander et al. 2008; Hoos and McMahon 2009). For example, McMahon and Woodside (1997) estimated that 27% of TN inputs in the Albemarle-Pamlico watersheds came from atmospheric deposition. Hoos and McMahon (2009) estimated that atmospheric deposition contributed 30-50% of nitrogen mass delivered to streams for the six major river catchments (HUCs 0301-0306) considered in this study. Their study included areas beyond the Coastal Plain and only looked at extrapolated wet deposition from monitoring which makes direct comparison difficult. Despite its importance within past statistical models, atmospheric N deposition at the coarse resolution of 12 km^2 was not significant in the BI model of our study. A comparison of atmospheric N deposition with TNC does show a positive relationship (r=0.763) but deposition shows an even stronger relationship with CAFOs (r=0.883). Within CMAQ, ammonia emissions are modeled as primarily local deposition of agriculture and CAFOs (Byun and Schere 2006) and watersheds 86 and 87 have very high CAFO densities and high percent row crop. Recent refinements to the ammonia emissions model of CMAQ suggest that ammonia deposition may be more regional than previously thought and that deposition immediately adjacent to ammonia sources (i.e. CAFOs) may be overestimated (Dennis et al. 2010).

Correcting this overestimation in future analyses could strengthen the relationship between TNC and atmospheric deposition.

Headwater Analysis

Non-buffered croplands in the headwaters (HwNB) had the lowest AIC_c of all the models and was statistically equivalent to the PBF model indicating that buffers in higher- order streams and rivers may have little influence on stream nitrogen concentrations and that interaction with buffers occurs primarily with smaller streams. This supports studies that suggest that much of the riparian buffers on larger streams/rivers may interact very little with upland nitrogen sources (McGlynn and Seibert 2003; Tomer et al. 2003). Our analysis highlights patterns of land use in headwaters within agricultural watersheds that have implications for management. The headwaters in our study covered an average of 58% of the total watershed yet we found that 79% of non-buffered cropland occurred in headwaters. In the relatively flat Coastal Plain watersheds, areas surrounding headwater streams are more suitable for agricultural production while areas around higher order rivers are often broad bottomland swamps with wide buffers were it is more difficult to farm. In this setting, entire watershed metrics that include the bottomland buffers may overestimate the amount of actual buffer interaction in the watershed. Additionally, buffer restoration efforts often occur along more perennial rivers and streams (e.g."USGS blue lines") and buffers in some of these locations may have limited impacts. Further study on the importance of buffers in headwaters is needed, including studies using imagery capable of identifying narrow headwater buffers widths that cannot be detected using the current 30m resolution LULC datasets.

If we consider issues related to stream resolution, estimates of HwNB in the watershed could increase if more detailed stream information was used. Here, headwater streams were defined as 1^{st} order USGS high-resolution stream lines which can underestimate true headwater streams (Heine et al. 2004, Roy et al. 2009, Brooks and Colburn 2011). Baker et al. (2007) found that if more detailed headwater streams were included in the Chesapeake Bay Coastal Plain, the percentage of non-buffered cropland could increase from 29% of total croplands to 49%. The definition and identification of headwater streams in agriculture is difficult as the systems are often highly modified by ditches and drains but some research is this arena is underway (Bailly et al. 2008, Lang et al. 2012). Headwaters should continue to be the focus of research and emphasized when management plans consider targeted approaches for riparian preservation and restoration (Dosskey et al. 2006; Baker et al. 2007; Dodds and Oakes 2008; Tomer et al. 2009), especially as finer LULC imagery and stream maps become available.

Drained Buffered Croplands

The Drained and Non-Buffered Cropland (DNBC) model was significant but was less supported by the AICc comparisons than then HBF and PBF models ($\Delta \approx 2$). The difference indicates the model may not be the best model but should not be discounted. Artificial drainage has been shown to influence nutrient transport at the local scale in the SE Coastal Plain (Jacobs and Gilliam 1985; Tesoriero et al. 2005; Harden and Spruill 2008) but across large, variable watersheds in the SE Coastal Plain with limited sample size, a strong influence was not evident. Studies and models at broader scales have implicated artificial drainage as an important driver of nitrogen concentrations in the Midwestern Cornbelt (Petrolina and Gowda 2006; Royer et al. 2006; Alexander et al. 2008; David et al. 2010). Artificial drainage is very prevalent in the Midwest where the most recent USDA surveys from 1987 estimated that Indiana, Illinois, Iowa, and Ohio all had greater than 20% drained cropland. Sixty-five counties in those four states had drained cropland estimates greater than 50% (Sugg 2007 and accompanying data). Conversely, North and South Carolina reported drained cropland at 8% and 5% respectively, with only three counties in North Carolina reporting greater than 5% poorly drained cropland. Only watersheds 119 and 133 had small portions of their watershed within two of those counties. A similar analysis that includes more drained lands and buffers with a greater sample size should be conducted in the region as well as in other regions with higher artificial drainage such as the Upper Midwest.

Lack of a strong influence from drainage may also be related to an inadequate estimation of drainage via the soil drainage proxy. Broad assessments of the influence of agricultural drainage on nutrients remain difficult as little data on the spatial location and extent of artificial drainage exist. The proxy of soil drainage classes used in this study is one of few alternatives and has been successfully used in models in the Midwest (Crumpton et al. 2006). Sugg (2007) suggests that the proxy may be inadequate in areas outside the Midwest and there is no data to assess its accuracy beyond the county scale. New remote sensing efforts may lead to better drainage estimates but thus far are limited in extent (Naz and Bowling

2008; Naz et al. 2009). More research and additional agricultural census data are needed to quantify the extent and impact of artificial agricultural drainage on nutrient exports at the watershed level in areas where it is prevalent. Apart from studying artificial drainage bypassing buffers, research of BMPs that modify or intercept artificial drainage need to continue (Evans et al. 1995; Singh et al. 2007; Iovanna et al. 2008)

Conclusions

In summary, models explicitly accounting for buffers in cropland better explained stream nitrogen concentrations and estimated that buffers attenuate an average of 45% of potential stream nitrogen in rural SE Coastal Plain watersheds. These estimates provide a more realistic picture of what buffers currently provide as an ecosystem service and what restored buffers may achieve when attempting to set and meet watershed water quality goals or when participating in nutrient trading programs. Headwaters experienced disproportionate amounts of non-buffered cropland compared to the watershed as a whole and we highlight the need for focused research to better identify headwaters in agricultural settings and how buffers in those settings can influence stream nitrogen concentrations. Further work with spatially-explicit buffer models should also try to better characterize riparian buffers, incorporating biogeochemical activity and other landscape factors. Artificial agricultural drainage and CAFO studies should also be considered as priorities for future watershed nutrient research in areas where they are heavily concentrated.

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References

Alexander RB, Boyer EW, Smith RA, Schwarz GE, Moore RB (2007) The role of headwater streams in downstream water quality. Journal the American Water Resources Association 43:41-59

Alexander RB, Smith RA, Schwarz GE, Boyer EW, Nolan JV, Brakebill JW (2008) Differences in phosphorus and nitrogen delivery to the Gulf of Mexico from the Mississippi River Basin. Environmental Science and Technology 42:822-830

Allison PD (1999). Logistic Regression Using the SAS System: Theory and application. SAS Inst. Inc., Cary, NC.

Bailly J, Lagacherie P, Millier C, Puech C, Kosuth P (2008) Agrarian landscapes linear features detection from LiDAR: application to artificial drainage networks. International Journal of Remote Sensing 29:3489–350

Baker ME, Weller DE, Jordan TE (2006) Improved methods for quantifying potential nutrient interception by riparian buffers. Landscape Ecology 21:1327-1345

Baker ME, Weller DE, Jordan TE (2007) Effects of stream map resolution on measures of riparian buffer distribution and nutrient retention potential. Landscape Ecology 22:973-992

Brauman KA, Daily GC, Duarte TK, Mooney HA (2007) The nature and value of ecosystem services: an overview highlighting hydrologic services. Annual Review of Environment and Resources 32:67-98

Bricker S, Longstaff B, Dennison W, Jones A, Boicourt K, Wicks C, Woerner J (2007) Effects of nutrient enrichment in the nation's estuaries: a decade of change. NOAA Coastal Ocean Program Decision Analysis Series No 26 National Centers for Coastal Ocean Science, Silver Spring, MD pp 328

Brooks RT, Colburn EA (2011) Extent and channel morphology of unmapped headwater stream segments of the Quabbin Watershed, Massachusetts. Journal of the American Water Resources Association 47:158-168

Burkholder JM, Dickey DA, Kinder CA, Reed RE, Mallin MA, McIver MR, Cahoon LB, Melia G, Brownie C, Smith J, Deamer N, Springer J, Glasgow HB, Toms D (2006) Comprehensive trend analysis of nutrients and related variables in a large eutrophic estuary: a decadal study of anthropogenic and climatic influences. Limnology and Oceanography 51:463-487

Burnham KP, Anderson DR (2002) Models selection and multimodel inference: a practical information-theoretic approach. Springer, New York

Byun D, Schere KL (2006) Review of the governing equations, computational algorithms, and other components of the Models-3 Community Multiscale Air Quality (CMAQ) modeling system. Applied Mechanics Reviews 59:51-77

Cape JN, Cornell SE, Jickells TD, Nemitz E (2011) Organic nitrogen in the atmosphere - Where does it from? A review of sources and methods. Atmospheric Research 102:30-48

Carpenter SR, Caraco NF, Correll DL, Howarth RW, Sharpley AN, Smith VH (1998) Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications 8:559-568

Ciparis S, Iwanowicz LR, Voshell JR (2012) Effects of watershed densities of animal feeding operations on nutrient concentrations and estrogenic activity in agricultural streams. Science of the Total Environment 414:268-276

Cook MJ, Baker JL (2001) Bacteria and nutrient transport to tile lines shortly after application of large volumes of liquid swine manure. Transactions of the ASAE 44:495-503

Crumpton WG, Stenback GA, Miller BA, Helmers MJ (2006) Potential Benefits of Wetland Filters for Tile Drainage Systems: Impact on Nitrate Loads to Mississippi River Subbasins. Final Project Report # IOW06682 Washington DC, US Department of Agriculture

Cuadra PE, Vidon P (2011) Storm nitrogen dynamics in tile-drain flow in the US Midwest. Biogeochemistry 104:293-308

David MB, Drinkwater LE, McIsaac GF (2010) Sources of nitrate yields in the Mississippi River Basin. Journal of Environmental Quality 39:1657-1667

Dean, DM, Foran ME (1992) The effect of farm liquid waste application on tile drainage. Journal of Soil and Water Conservation 47:368-369

Dennis RL, Mathur R, Pleim JE, Walker JT (2010) Fate of ammonia emissions at the local to regional scale as simulated by the community multiscale air quality model. Atmospheric Pollution Research 1:207-214

Dodds WK, Oakes RM (2008) Headwater influences on downstream water quality. Environmental Management 41:367-377

Dosskey MG, Helmers MJ, Eisenhauer DE (2006) An approach for using soil surveys to guide the placement of water quality buffers. Journal of Soil and Water Conservation 61:344-354

Dosskey MG, Helmers MJ, Eisenhauer DE, Franti TG, Hoagland KD (2002) Assessment of concentrated flow through riparian buffers. Journal of Soil and Water Conservation 57:336-343

Ebert DW, Wade TG (2004) Analytical tools interface for landscape assessments (ATtiLA): User manual EPA/600/R-04/083. Washington, DC, US Environmental Protection Agency

Evans RO, Skaggs RW, Gilliam JW (1995) Controlled versus conventional drainage effects on water quality. Journal of Irrigation and Drainage Engineering 121:271–276

Floyd, W.C., S.H. Schoenholtz, S.M. Griffith, P.J. Wigington Jr., and J.J. Steiner. 2009. Nitrate-nitrogen, land use/land cover, and soil drainage associations at multiple spatial scales. Journal of Environmental Quality 38: 1473-1482.

Galloway JN, Townsend AR, Erisman JW, Bekunda M, Cai Z, Freney JR, Martinelli LA, Seitzinger SP, Sutton MA (2008) Transformation of the nitrogen cycle: recent trends, questions and potential solutions. Science 320:889-892

Glasgow HB Jr, Burkholder JM (2000) Water quality trends and management implications from a five-year study of a eutrophic estuary. Ecological Applications 10:1024-1046

Gold AJ, Groffman PM, Addy K, Kellogg DQ, Stolt M, Rosenblatt AE (2001) Landscape attributes as controls on ground water nitrate removal capacity of riparian zones. Journal of the American Water Resources Association 37:1457-1464

Groffman PM, Butterbach-Bahl K, Fulweiler RW, Gold AJ, Morse JL, Stander EK, Tague C, Tonitto C, Vidon P (2009) Challenges to incorporating spatially and temporally explicit phenomena (hotspots and hot moments) in denitrification models. Biogeochemistry 93:49-77

Harden SL, Spruill TB (2008) Factors affecting nitrate delivery to streams from shallow ground water in the North Carolina Coastal Plain. US Geological Survey Scientific Investigations Report 2008-5021, 39 p

Heine, R.A., C.L. Lant, and R.R. Sengupta. 2004. Development and comparison of approaches for automated mapping of stream channel networks. Annals of the Association of American Geographers 94:477-490.

Homer C, Dewitz J, Fry J, Coan M, Hossain N, Larson C, Herold N, McKerrow A, Van Driel JN, Wickham J (2007) Completion of the 2001 National Land Cover Database for the Conterminous United States. Photogrammetric Engineering and Remote Sensing 73:337-341

Hoos AB, McMahon G (2009) Spatial analysis of instream nitrogen loads and factors controlling nitrogen delivery to streams in the southeastern United States using spatially referenced regression on watershed attributes (SPARROW) and regional classification frameworks. Hydrological Processes doi: 101002/hyp7323

Hoos AB, Terziotti S, McMahon G, Savvas K, Tighe KC, Alkons-Wolinsky R (2008) Data to support statistical modeling of instream nutrient load based on watershed attributes, Southeastern United States. US Geological Survey Open-File Report 2008-1163, pp 50

Iovanna R, Hyberg S, Crumpton W (2008) Treatment wetlands; cost-effective practice for intercepting nitrate before it reaches and adversely impacts surface waters. Journal of Soil and Water Conservation 63:14A-15A

Israel DW, Showers WJ, Fountain M, Fountain J (2005) Nitrate movement in shallow ground water from swine-lagoon-effluent spray fields managed under current application regulations. Journal of Environmental Quality 34:1828-1842

Jacobs TC, Gilliam JW (1985) Riparian losses of nitrate from agricultural drainage waters. Journal of Environmental Quality 14:472-478

Johnson LB, Richards C, Host GE, Arthur JW (1997) Landscape influences on water chemistry in Midwestern stream ecosystems. Freshwater Biology 37:193-208

Jones KB, Neale AC, Nash MS, Van Remortel RD, Wickham JD, Ritters KH, O'Neill RV (2001) Predicting nutrient and sediment loadings to streams from landscape metrics: a multiple watershed study from the United States Mid-Atlantic Region. Landscape Ecology 16:301-312

Karr JD, Showers WJ, Gilliam W, Andres AS (2001) Tracing nitrate transport and environmental impact from intensive swine farming using delta nitrogen-15. Journal of Environmental Quality 30:1163-1175

Kellogg DQ, Gold AJ, Cox S, Addy K, August PV (2010) A geospatial approach for assessing denitrification sinks within lower-order catchments. Ecological Engineering 36:1596-1606

Kellogg RL, Lander CH, Moffitt DC, Gollehon N (2000) Manure nutrients relative to the capacity of cropland and pastureland to assimilate nutrients: spatial and temporal trends for the United States. U.S. Department of Agriculture, Natural Resources Conservation Service and Economic Research Service Washington DC.

Lang M, McDonough O, McCarty G, Oesterling R Wilen B (2012) Enhanced detection of wetland-stream connectivity using LiDAR. Wetlands 32:461-473

Liu Z, Weller DE, Correll DL, Jordan TE (2000) Effects of land cover and geology on stream chemistry in watersheds of Chesapeake Bay. Journal of the American Water Resources Association 36:1349-1365

Lowe WH, Likens GE (2005) Moving headwaters to the head of the class. BioScience 55:196-197

Lowrance R, Altier RS, Newbold JD, Schnabel RR, Groffman PM, Denver JM, Correll DL, Gilliam JW, Robinson JL, Brinsfield RB, Staver KW, Lucas W, Todd AH (1997) Water quality functions of riparian forest buffer systems in Chesapeake Bay Watersheds. Environmental Management 21:687-712

Mallin MA, Cahoon LB (2003) Industrialized animal production – a major source of nutrient and microbial pollution to aquatic ecosystems. Population and Environment 24:369-385

Mayer PM, Reynolds SK Jr., McCutchen MD, Canfield TJ (2007) Meta-analysis of nitrogen removal in riparian buffers. Journal of Environmental Quality 36:1172-1180

McGlynn BL, Seibert J (2003) Distributed assessment of contributing area and riparian buffering along stream networks. Water Resources Research 39:1082, doi:101029/2002WR001521

McMahon G, Alexander RB, Qian S (2003) Support of total maximum daily load programs using spatially referenced regression models. Journal of Water Resources Planning and Management-ASCE 129:315-329

McMahon G, Woodside MD (1997) Nutrient mass balance for the Albemarle-Pamlico drainage basin, North Carolina and Virginia, 1990. Journal of the American Water Resources Association 33:573-589

National Research Council (2000) Clean coastal waters: Understanding and reducing the effects of nutrient pollution. National Academy Press, Washington DC

Naz BS, Ale S, Bowling LC (2009) Detecting subsurface drainage systems and estimating drain spacing in intensively managed agricultural landscapes. Agricultural Water Management 96:627-637

Naz B, Bowling LC (2008) A decision analysis system for mapping of subsurface drainage systems. Transactions of the ASABE 51:1937-1950

Newbold JD, Herbert S, Sweeney BW, Kiry P, Alberts SJ (2010) Water quality functions of a 15-year-old riparian forest buffer system. Journal of the American Water Resources Association 46:299-310

North Carolina Department of Environment and Natural Resources (NCDENR)(2007) North Carolina Conservation Reserve Enhancement Program (CREP) annual report 2007. Accessed online April 22, 2011: http://www.enr.state.nc.us/dswc/pages/CREP2007AnnualReport.pdf

Omernik JM (2004) Perspectives on the nature and definition of ecological regions. Environmental Management 34:S27-S38

Osborne LL, Wiley MJ (1988) Empirical relationships between land use/cover and stream water quality in an agricultural watershed. Journal of Environmental Management 26:9-27

Pankau RC, Schoonover JE, Williard KWJ, Edwards PJ (2012) Concentrated flow paths in riparian buffer zones of southern Illinois. Agroforestry Systems 84:191-205

Petrolina DR, Gowda PH (2006) Missing the boat: Midwest farm drainage and the Gulf of Mexico hypoxia. Review of Agricultural Economics 28:240-253

Randall GW, Goss MJ (2008) Nitrate losses to surface water through subsurface, tile drainage. In: Hatfield JL, Follett RF (eds), Nitrogen in the Environment: Sources, Problems, and Management, 2nd Ed. Academic Press, San Diego, pp 145-176

Randall GW, Iragavarapu TK, Schmitt MA (2000) Nutrient losses in subsurface drainage water from dairy manure and urea applied for corn. Journal of Environmental Quality 29:1244-1252

Read JJ, Brink GE, Adeli A, McGowen SL (2008) Swine effluent application timing and rate affect nitrogen use efficiency in common bermudagrass. Journal of Environmental Quality 37:S180-S189

Rosenblatt AE, Gold AJ, Stolt MH, Groffman PM, Kellogg DQ (2001) Identifying riparian sinks for watershed nitrate using soil surveys. Journal of Environmental Quality 30:1596-1604

Rothenberger MB, Burkholder JM, Brownie C (2009) Long-term effects of changing land use practices on surface water quality in a coastal river and lagoonal estuary. Environmental Management 44:505-523

Roy AH, Byvas AL, Fritz KM, Lubbers HR (2009) Urbanization affects the extent and hydrologic permanence of headwater streams in a Midwestern US metropolitan area. Journal of the North American Benthological Society 28: 911-928

Royall RM (1997) Statistical evidence: a likelihood paradigm. Chapman and Hall, New York

Royer TV, David MB, Gentry LE (2006) Timing of riverine export of nitrate and phosphorus from agricultural watersheds in Illinois: implications for reducing nutrient loading to the Mississippi River. Environmental Science and Technology 40:4126-2131

Schwede DB, Dennis RL, Bitz MA (2009) The watershed deposition tool: a tool for incorporating atmospheric deposition in water-quality analyses. Journal of the American Water Resources Association 45:973-985

Singha R, Helmers MJ, Crumpton WG, Lemke DW (2007) Predicting effects of drainage water management in Iowa's subsurface drained landscapes. Agricultural Water Management 92:162-170

Sloan AJ, Gilliam JW, Parsons JE, Mikkelsen RL, Riley RC (1999) Ground water nitrate depletion in a swine-lagoon effluent-irrigated pasture and adjacent riparian area. Journal of Soil and Water Conservation 54:651-656

Spruill TB (2000) Statistical evaluation of effects of riparian buffers on nitrate and ground water quality. Journal of Environmental Quality 29:1523-1538

State-EPA Nutrient Innovations Task Group (2009) An Urgent Call to Action - Report of the State-EPA Nutrient Innovations Task Group. Accessed online Aug 30, 2011: http://water.epa.gov/scitech/swguidance/standards/criteria/nutrients/upload/2009_08_27_criteria_nutrient_nitgreport.pdf

Stone KC, Hunt PG, Humenik FJ, Johnson MH (1998) Impact of swine waste application on groundwater and stream water quality in an eastern Coastal Plain watershed. Transactions of the ASAE 41:1665-1670

Stone KC, Hunt PG, Novak JM, Johnson MH, Watts DW, Humenik FJ (2004) Stream nitrogen changes in an eastern Coastal Plain watershed. Journal of Soil and Water Conservation 59:66-72

Stow CA, Borsuk ME, Stanley DW (2001) Long-term changes in watershed nutrient inputs and riverine exports in the Neuse River, North Carolina. Water Research 35:1489-1499

Strahler AN (1957) Quantitative analysis of watershed geomorphology. Transactions of the American Geophysical Union 8:913–920

Sugg Z (2007) Assessing US Farm Drainage: Can GIS Lead to Better Estimates of Subsurface Drainage Extent? Accessed online Aug 02, 2010: http://pdf.wri.org/assessing_farm_drainage.pdf

Sutton AJ, Fisher TR, Gustafson AB (2009) Effects of restored stream buffers on water quality in no-tidal streams in the Choptank River Basin. Water, Air, & Soil Pollution doi: 101007/s11270-009-0152-3

Tesoriero AJ, Duff JH, Wolock DM, Spahr NE (2009) Identifying pathways and processes affecting nitrate and orthophosphate inputs to streams in agricultural watersheds. Journal of Environmental Quality 38:1892-1900

Tesoriero AJ, Spruill TB, Mew HE Jr., Farrell KM, Harden SL (2005) Nitrogen transport and transformations in a coastal plain watershed: influence of geomorphology on flow paths and residence times. Water Resources Research 41:W02008 doi:101029/2003WR002953

Tomer MD, Dosskey MG, Burkhart MR, James DE, Helmers MJ, Eisenhauer DE (2009) Methods to prioritize placement of riparian buffers for improved water quality. Agroforestry Systems 75:17-25

Tomer MD, James DE, Isenhart TM (2003) Optimizing the placement of riparian practices in watershed using terrain analysis. Journal of Soil and Water Conservation 58:198-206

Tomer MD, Locke MA (2011) The challenge of documenting water quality benefits of conservation practices: a review of USDA-ARS's conservation effects assessment project watershed studies. Water Science and Technology 64: 300-310

Turner RE, Rabalais NN (1991) Changes in Mississippi River water quality this century. Bioscience 41:140-147

Vidon P, Allan C, Burns D, Duval TP, Gurwick N, Inamdar S, Lowrance R, Okay J, Scott D, Sebestyen S (2010) Hot spots and hot moments in riparian zones: potential for improved water quality management. Journal of the American Water Resources Association 46:278-298

Vidon PGF, Hill AR (2006) A landscape-based approach to estimate riparian hydrological and nitrate removal functions. Journal of the American Water Resources Association 42:1099-1112

Vitousek PM, Aber J, Howarth RW, Likens GE, Matson PA, Schindler DW, Schlesinger WH, Tilman GD (1997) Human alteration of the global nitrogen cycle: causes and consequences. Ecological Applications 7:737-750

Weldon MB, Hornbuckle KC (2006) Concentrated animal feeding operations, row crops, and their relationship to nitrate in Eastern Iowa rivers. Environmental Science and Technology 40:3168-3173

Weller DE, Baker MA, Jordan TE (2011) Effects of riparian buffers on nitrate concentrations in watershed discharges: new models and management implications. Ecological Applications 21:1679-1695

Weller DE, Jordan TE, Correll DL, Liu Z (2003) Effects of land-use change on nutrient discharges from the Patuxent River watershed. Estuaries 26:244-266

Whalen SC, DeBerardinis JT (2007) Nitrogen mass balance in fields irrigated with liquid swine waste. Nutrient Cycling in Agroecosystems 78:37-50

US Department of Agriculture (USDA), National Agricultural Statistics Service (NASS) (2009) 2007 Census of Agriculture. Accessed online Aug 17 2010: http://www.agcensus.usda.gov/Publications/2007/

US Environmental Protection Agency (USEPA) (2008) Compiled CAFO NPDES Regulations and Effluent Limitations Guidelines and Standards. USEPA Washington DC, pp44

Young EO, Briggs RD (2005) Shallow ground water nitrate-N and ammonium-N in cropland and riparian buffers. Agriculture, Ecosystems, and Environment 109:297-309

Young EO, Briggs RD (2007) Nitrogen dynamics among cropland and riparian buffers. Journal of Environmental Quality 36:801-814

Table 1. The river, area (km^2), total nitrogen concentration (TNC)(mg N/L), and the percentage of various land use\land cover categories for 24 watersheds from the SE Coastal Plain. WshdID is the ID for each watershed and corresponds to the numbers found in Fig. 1, Cl = % cropland, Pa = % pasture, Ur = % urban, CAFO = density of Confined Animal Feeding Operations (CAFOs/km^2), Atm = average annual wet and dry atmospheric deposition of total oxidized and reduced nitrogen (kg-N/ha/yr), NB = % cropland with no buffer, HwNB = % of cropland with no buffer in headwaters, DNB = % combined drained and non-buffered croplands.

wtrshdID	River/Stream	Area	TNC	Cl	Pa	Ur	CAFO	Atm	NB	HwNB	DNB
23	Sandy Run	133.4	1.27	34.5	6.3	6.4	0.15	19.5	10.0	8.0	16.2
52	Ogeechee	6904.2	0.54	16.0	7.0	5.0	0.00	5.4	3.3	2.6	4.0
61	Ebenezer	497.9	0.46	11.8	3.8	4.8	0.00	5.4	1.5	1.1	4.7
86	Black	1759.4	1.09	37.7	1.0	6.9	0.21	29.1	11.2	9.5	15.0
87	NE Cape Fear	1229.9	1.41	41.2	1.0	4.3	0.22	34.7	11.7	9.9	15.8
99	Blackwater	1922.0	0.96	21.9	11.3	2.1	0.00	8.5	4.5	3.9	7.1
100	Potecasi	578.8	0.80	26.8	11.6	2.2	0.02	10.1	4.2	3.8	8.0
103	Lumber	3177.0	0.75	25.3	2.9	7.9	0.02	12.4	8.2	5.9	11.9
104	Waccamaw	1783.8	0.86	18.3	0.4	4.0	0.03	11.0	4.3	3.3	9.0
114	Contentnea	1892.0	1.19	35.8	8.0	9.2	0.04	15.8	8.5	6.4	12.3
119	Trent	432.5	1.05	24.7	0.5	3.5	0.10	18.9	8.0	6.5	13.8
130	Fishing Cr	1371.8	0.58	6.3	11.3	5.1	0.00	9.0	1.5	1.2	1.7
133	Conetoe Cr	187.3	1.98	35.7	1.9	5.2	0.03	11.6	19.3	14.9	25.0
134	Chicod Cr	111.6	1.17	35.1	4.3	4.4	0.14	10.6	15.6	11.6	25.2
136	New	202.1	1.78	32.2	0.9	4.2	0.13	17.7	10.5	8.1	14.7
166	Salkehatchie	881.2	0.53	14.3	9.3	5.0	0.01	5.4	2.6	2.2	3.4
183	N. Fork Edisto	1878.5	0.73	13.3	9.3	7.6	0.07	7.6	3.4	2.6	3.6
185	S. Fork Edisto	2199.3	0.37	15.7	9.6	5.2	0.03	6.9	3.1	2.7	3.5
195	Black Cr	565.6	0.52	12.8	5.9	8.9	0.00	7.7	2.6	1.7	2.7
197	Lynches	2703.6	0.57	18.1	8.5	5.5	0.04	8.1	6.8	5.3	8.3
203	Black Run	4279.6	0.64	20.5	9.0	6.8	0.02	6.7	7.7	5.7	10.0
240	Upper 3 Runs	521. 8	0.33	4.6	5.3	5.8	0.00	6.5	1.1	0.8	1.2
242	Lower 3 Runs	157.7	0.28	1.9	1.6	1.3	0.00	5.6	0.2	0.0	0.4
287	Chowan	4448.8	0.75	12.1	11.0	3.0	0.00	8.1	2.1	1.6	3.2
	mean	1659.1	0.86	21.5	5.9	5.2	0.05	11.8	6.3	5.0	9.2
	median	1300.8	0.75	19.4	6.1	5.0	0.02	8.7	4.4	3.9	8.1

Table 2. Correlation analysis for dependent and independent variables of the 24 study watersheds in the SE Coastal Plain. Correlations greater than 0.70 are shown in bold. TNC = total nitrogen concentration, Cl = % cropland, Pa = % pasture, Ur = % Urban, CAFO = density of Confined Animal Feeding Operations (CAFOs/km2), Atm = average annual wet and dry atmospheric deposition of total oxidized and reduced nitrogen (kg-N/ha/yr), NB = % cropland with no buffer, HwNB = % of cropland with no buffer in headwaters, DNB = % drained and non-buffered croplands.

	Cl	Pa	Ur	CAFO	Atm	NB	HwNB	DNB
TNC	**0.838**	-0.412	0.013	0.59	0.624	**0.86**	**0.871**	**0.848**
Cl		-0.347	0.175	**0.744**	**0.763**	**0.873**	**0.895**	**0.882**
Pa			0.02	-0.484	-0.515	-0.446	-0.442	-0.474
Ur				0.09	0.098	0.16	0.129	0.082
Cafo					**0.883**	0.629	0.665	0.64
Atm						0.571	0.62	0.567
NB							**0.996**	**0.977**
HwNB								**0.972**

Table 3. Various models considered in regression analyses to explain variation in total nitrogen concentrations for 24 watersheds in the SE Coastal Plain. Independent variables that were included in the various models are denoted by the X, where Cl = % cropland, Pa = % pasture, Ur = % Urban, CAFO = density of Confined Animal Feeding Operations (CAFOs/km^2), Atm = average annual wet and dry atmospheric deposition of total oxidized and reduced nitrogen (kg-N/ha/yr), NB = % cropland with no buffer, k = number of parameters in the model, delta(Δ) = difference between model AIC_c value and the best AIC_c model (PBF) value, weight = Akaike weights for each model where only models with $\Delta<10$ are included (Royal 1997).

model	symbol	independent variables						k	RMSE	Adj R^2	AICc	Delta	wt
		Cl	Pa	Ur	Atm	Cafo	NB						
Basic Inputs	BI	X	X	X	X	X		6	0.251	0.68	-28.22	13.43	0
Parsimonious Basic Inputs	PBI	X						2	0.246	0.69	-38.18	3.47	0.15
Basic Inputs + No Buffer	BF	X	X	X	X	X	X	7	0.225	0.74	-30.3	11.35	0
Parsimonious No Buffer	PBF	X					X	3	0.221	0.75	-41.65	0	0.85

Table 4. Coefficients and standard errors (SE) for 3 models (see Table 3 for abbreviations) included in calculation of the MA (Δ<10) and for model average (MA). β_0 = intercept, β_{Cl} = coefficient for cropland, β_{NB} = coefficient for cropland with no buffer. Significant coefficients within the various regression models are indicated in bold.

Models	β_0	β_{Cl}	β_{NB}
PBI	0.156	**0.033**	
SE	0.110	0.005	
PBF	**0.242**	0.014	**0.050**
SE	0.104	0.008	0.019
Model Average (MA)	**0.229**	0.017	**0.050**
SE	0.110	0.010	0.019
95% confidence	0.443	0.037	0.088
	0.016	-0.003	0.012

Table 5. Variance Inflation Factors (VIF) for coefficients of the PBF model which were all included in the model average. Coefficients in a model are generally considered stable and minimally impacted by collinearity when VIF is below 10.

Coefficient	Value	SE	t value	Pr > t	VIF
β_0	0.262	0.145	1.800	0.086	0.000
β_{CI}	0.013	0.010	1.310	0.206	4.211
β_{NB}	0.049	0.020	2.470	0.023	4.211

Table 6. Jack-knifed cross-validation performed on various models (see Table 3 for abbreviations) that were used to comprise the model average (MA) to determine the effect of individual watersheds on the overall prediction of stream total nitrogen concentrations.

model	all data		cross validation		% change in RMSE	% change in Adj R^2
	RMSE	Adj R^2	RMSE	Adj R^2		
PBI	0.246	0.689	0.264	0.643	0.074	-0.067
PBF	0.221	0.750	0.262	0.649	0.187	-0.135
MA	0.228	0.732	0.262	0.648	0.149	-0.115

Table 7 Estimates of total nitrogen concentrations (TNC) (mg N/L) for model average (MA) components in the SE Coastal Plain. Estimates are based on the mean value of the land cover variables for 24 watersheds multiplied by the coefficients of the model average (MA), where Cl = % cropland, BCl = % cropland with buffer, and NB = % cropland with no buffer. Stream TNC is a combination of non-crop sources, cropland leakage, and current additions from non-buffered cropland. Potential TNC predicts concentrations if there was no buffer removal thus it includes stream TNC components plus current removal. Edge of field includes all potential TNC components except non-crop sources.

MA Components		mg N/L	% of Stream TNC	% of Potential TNC
Non-crop sources	β_0	0.229	25.4	13.9
All cropland leakage	$\beta_{Cl}Cl$	0.366	40.5	22.2
Buffered	$\beta_{Cl}BCl$	0.259	28.7	15.7
Non-buffered	$\beta_{Cl}NB$	0.107	11.8	6.5
Non-buffered addition (potential restoration)	$\beta_{NB}NB$	0.308	34.1	18.7
Current removal	$\beta_{NB}BCl$	0.743	--	45.1
Stream TNC		0.903		
Potential TNC		1.646		
Edge of Field		1.417		
% buffer removal from edge		52.4		

Table 8. Comparison of regression models to test the importance of headwater buffers and artificial drainage in explaining variation in total nitrogen concentrations for 24 watersheds in the SE Coastal Plain. Independent variables that were included in the various models are denoted by the X, where Cl = % cropland, NB = % cropland with no buffer, HwNB = % of cropland with no buffer in headwaters, DNB = % drained and non-buffered croplands, k = number of parameters in the model, delta(Δ) = difference between model AIC_c value and the best AIC_c model (HBF) value.

model	symbol	independent variables				k	RMSE	Adj R^2	AICc	Delta
		Cl	NB	HwNB	DNB					
Parsimonious No Buffer	PBF	X	X			3	0.221	0.75	-41.65	0.41
Headwaters No Buffer	HBF	X		X		3	0.219	0.75	-42.06	0
PNB + Drained Buffer	NBDB	X			X	3	0.229	0.73	-39.96	2.1

Figures Captions

Fig. 1. Watershed boundaries and ID numbers for 24 non-nested watersheds located within six large river basins in the Southeastern Coastal Plain (see Table 1 for watershed descriptions).

Fig. 2. Observed versus predicted Total Nitrogen Concentrations (TNC) for 24 watersheds in the Southeast Coastal Plain based on the MA equation. Error bars represent 95% confidence intervals of predicted TNC for individual watersheds.

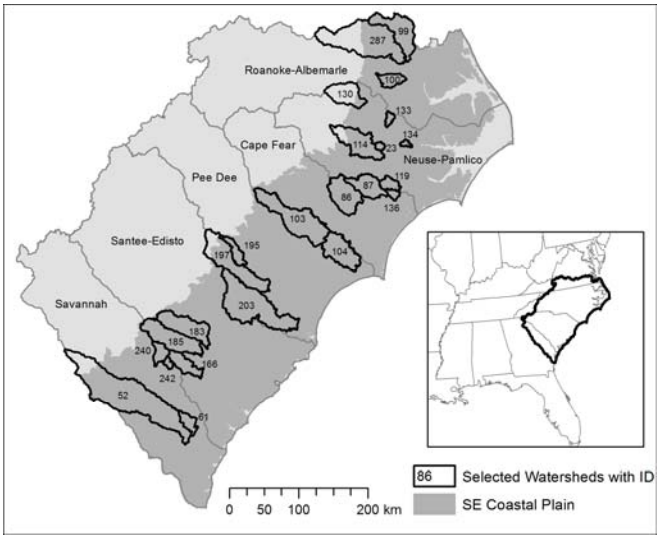
Roanoke-Albemarle
Cape Fear
Neuse-Pamlico
Pee Dee
Santee-Edisto
Savannah
287
99
100
130
133
134
114
23
119
87
86
136
103
104
195
197
203
183
185
240
166
242
52
61
0 50 100 200 km
86 Selected Watersheds with ID
SE Coastal Plain

Fig.1

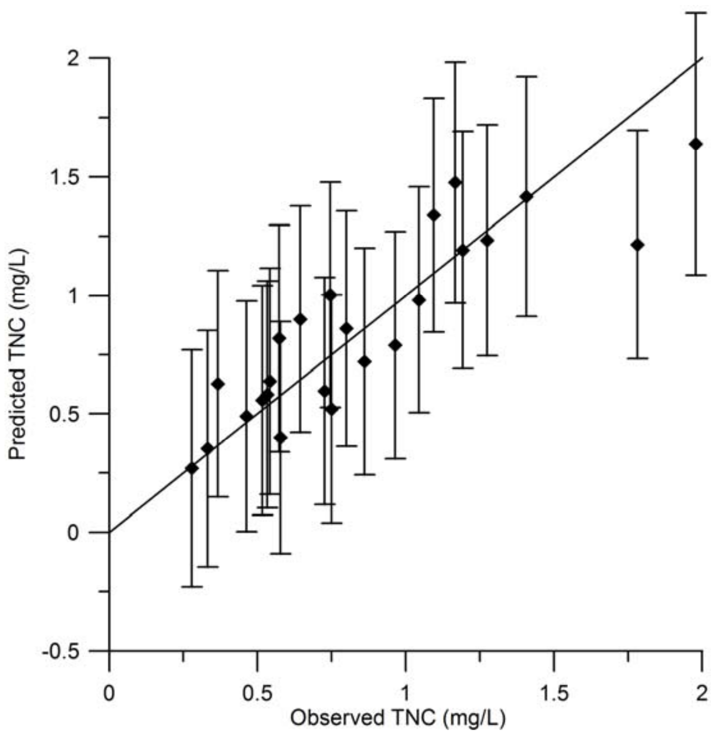

Fig.2