NUTRIENT BUDGETS FOR WATERSHEDS ON

THE SOUTHEASTERN ATLANTIC COAST OF THE UNITED STATES:

TEMPORAL AND SPATIAL VARIATION

by

SYLVIA CATHERINE SCHAEFER

(Under the Direction of Merryl Alber)

ABSTRACT

Nitrogen budgets were constructed for the watersheds of twelve rivers in the southeastern U.S. following the methodology of the SCOPE project. Total inputs ranged from 2,762-6,232 kg/km^2/yr, with the largest contributions from fertilizer and net food/feed import. Export to the estuaries of these rivers was positively related to inputs and averaged 9%. In contrast, reported export to estuaries further north averaged 25%. These differences are consistent with temperature-driven changes in denitrifier activity and suggest a possible control for watershed nitrogen export. In order to evaluate changes in nutrient input over time, nitrogen and phosphorus budgets were constructed for the watershed of the Altamaha River estuary in Georgia for 1954, 1974, and 1992. Nutrient inputs increased between 1954 and 1974 and decreased in 1992 (from 2,007 to 3,553 to 2,977 kg N/km^2/yr and 408 to 532 to 340 kg P/km^2/yr). These changes were largely driven by trends in fertilizer use.

INDEX WORDS: reactive nitrogen, phosphorus, N budgets, P budgets, watersheds, nutrient inputs, stream export, Georgia, Altamaha, southeastern

United States, temperature, denitrification, latitude, climate
change, temporal trend, agriculture, estuary, coastal

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INTRODUCTION

The most important nutrients limiting production in coastal ecosystems are nitrogen (N) and phosphorus (P) (Smith 1984, Howarth 1988, Galloway et al. 1996). Excesses of either N or P lead to eutrophication, which affects numerous estuaries in the United States and worldwide (Bricker et al. 1999, Howarth et al. 2000, NRC 2000). Problems associated with eutrophication include hypoxia, nuisance algae, sea grass dieoff, and changes in the food web assemblage (Howarth 1988, Rabalais 2002). Both nitrogen and phosphorus have been shown to cause such problems in coastal systems on the east coast of the United States (e.g. Lapointe and Clark 1992, Bowen & Valiela 2001, Anderson et al. 2002, Paerl et al. 2004).

The nitrogen export to an estuary has been shown to be related to inputs on a watershed scale (Howarth et al. 1996, Boyer et al. 2002). These inputs are largely derived from non-point agricultural sources such as fertilizer application and manure production by livestock (Jordan and Weller 1996, Carpenter et al. 1998). Urban areas are another potential source of nutrient pollution (Carpenter et al. 1998). Numerous studies have shown that N fluxes in rivers can be predicted from watershed characteristics such as population density, fertilizer application, and atmospheric N deposition (e.g. Howarth et al. 1996, Howarth 1998, Goolsby and Battaglin 2000, Caraco et al. 2003). Phosphorus fluxes to rivers can be predicted based on sewage point sources and fertilizer runoff in watersheds (Caraco 1995). Human influences such as population growth and increased agricultural activity have been shown to result in increased nutrient loading to watersheds, with subsequent downstream effects (e.g. Boynton et al. 1995, Stow et al. 2000, Bowen and Valiela 2001, McIsaac et al. 2002).

Nutrient inputs to watersheds in the southeastern United States have received comparatively less attention than those in other areas. Although the southeast has been included in several large-scale studies (Howarth et al. 1996, Castro et al. 2003), watersheds there have not been examined in great detail. Those studies that did examine nutrient inputs to southeastern watersheds (e.g. Asbury and Oaksford 1997) did not use a methodology that would permit a direct comparison between southeastern watersheds and those in other regions. Furthermore, studies of historical changes in nutrient inputs to southeastern watersheds have not been undertaken.

The purpose of this thesis was to examine the sources of nutrients to estuarine watersheds in the southeastern United States, following a methodology previously used to construct nutrient budgets for watersheds in the northeastern and mid-Atlantic region (Boyer et al. 2002). In Chapter 1, total N inputs were calculated for twelve estuarine watersheds in North Carolina, South Carolina, and Georgia, and the inputs compared to riverine N export. Total nutrient inputs to watersheds were comparable with those obtained for the northeastern and mid-Atlantic region (Boyer et al. 2002), but the predominant inputs differed among the two regions. N inputs due to agriculture predominated in the southeast. Fertilizer (followed by net food and feed import) was the most important source of new N input to southeastern watersheds, whereas in the northeast, atmospheric deposition predominated. The percentage of N inputs that were exported in the southeast was far lower than in the northeast, which is likely a result of differing rates of microbial denitrification in response to temperature.

In Chapter 2, both N and P inputs to Georgia's Altamaha River watershed were calculated for three points in time—1954, 1974, and 1992—in order to examine historical trends in nutrient inputs. Inputs of both N and P increased between 1954 and 1974 and then declined again by 1992. Changes in both nutrients were driven primarily by changes in fertilizer use and, in some sub-watersheds, food and feed import. Sub-basins with a high level of agricultural production tended to have the highest inputs.

Nutrient budgets such as those constructed here are important tools for addressing coastal eutrophication. As human population continues to grow, it becomes increasingly important to prevent excess nutrients from reaching estuaries. Understanding the patterns of nutrient input and the processes controlling export to estuaries will help focus management efforts on those systems most at risk.

CHAPTER 1

A LATITUDINAL GRADIENT IN N EXPORT TO COASTAL ECOSYSTEMS

Background

Human inputs of reactive nitrogen (N) to the global cycle have increased dramatically since the advent of the industrial age through agricultural production (including biological nitrogen fixation and fertilizer production) and emissions from fossil fuel combustion (Vitousek et al. 1997; Galloway and Cowling 2002). In estuaries and coastal areas, this increase of reactive nitrogen in the environment can lead to eutrophication, which is a major problem throughout the United States, particularly along the mid-Atlantic and Gulf coasts (NRC 2000).

As part of the International SCOPE Nitrogen Project, which aimed to improve understanding of human influence on the nitrogen cycle, Boyer et al. (2002) developed a methodology for examining total N input to watersheds. They used this approach to develop N budgets for sixteen watersheds in the northeastern United States, ranging from the Penobscot River in Maine to the James River in Virginia. They found a strong positive relationship between total N input and riverine export to the coastal zone. This export averaged 25% of new inputs. A similar N budget has also been constructed for the Mississippi River; riverine nitrate flux there amounted to 18% of inputs (McIsaac et al. 2001).

Although the SCOPE methodology has not been applied to riverine estuaries of the southeastern U.S., there have been several studies of N inputs. Castro et al. (2003) constructed watershed budgets for 34 estuaries on the Atlantic and Gulf coasts of the United States, including 7 in the southeast. They found that agricultural sources predominated in 20 of the 34, and in 6 out of 7 southeastern estuaries. Asbury and Oaksford (1997) provided estimates of N inputs to the watersheds of a different set of seven southeastern rivers in Georgia and Florida. Fertilizer or animal waste predominated in all systems.

In a large-scale overview of nitrogen inputs to basins draining the North Atlantic, Howarth et al. (1996) reported that riverine N exports in the southeast were lower relative to N inputs than in the northeast. This is consistent with the observations of Castro et al. (2003). Although their reported values for the proportion of N exported to estuaries were higher than those calculated for the SCOPE Project, their results indicate that a smaller proportion of N inputs was exported in southern systems than in those further north. Asbury and Oaksford (1997) also found that riverine N output from these systems ranged between 4.2 and 7.4 percent of inputs, which is a lower proportion than that reported for the northeast. However, all of these studies used a methodology different from that of Boyer et al. (2002), and therefore are not directly comparable to the results obtained by the SCOPE N Project.

In this study, we constructed nitrogen budgets for southeastern watersheds based on the methodology of the SCOPE Project (Boyer et al. 2002). Our objectives were to determine whether the relationship between net N input and riverine export observed in the northeast held in southeastern watersheds, and to evaluate differences as a function of latitude.

Methods

N budgets were constructed for the watersheds of 12 rivers, ranging from Virginia to Georgia (Figure 1.1). The watersheds included in this study were those of the Roanoke (in VA and NC); Pamlico, Neuse, and Cape Fear (NC); Pee Dee and Santee (NC and SC); Black and Edisto (SC); Savannah (NC, SC and GA); and Ogeechee, Altamaha, and Satilla (GA). The St. Mary's watershed, just south of the Satilla, was excluded because data were insufficient for calculating riverine export (but see Appendix A).

Total N input was calculated by summing all external sources of N to a watershed. To ensure maximum comparability between the SCOPE N Project and this study, we followed the basic procedure set forth in Boyer et al. (2002) (see also Appendix B). Sources of N considered were atmospheric deposition, fertilizer, biological nitrogen fixation by crops and forest lands, and net food and feed import. The base time period of data used for this study, as for SCOPE (Boyer et al. 2002), was the early 1990s. This period represents typical conditions for the southeast in terms of both temperature and precipitation.

Watershed Characteristics

Watersheds were mapped to the most downstream USGS water quality gage, which was the point used to estimate riverine export to the coast. The area of each watershed was delineated using ESRI GIS software (ESRI 2003; ESRI 2004). Shapefiles of 8-digit hydrologic unit codes (HUCs) (Steeves and Nebert 1994) were downloaded and, for each catchment, Digital Elevation Models (DEMs) (Musser 1996; USGS 1999a) were used to delineate the area within each HUC that contributed to the gaged area. In the case of both the Ogeechee and Santee watersheds, we combined data from gages on different branches of the river in order to include as much of the watershed as possible.

Many of the data sets used in this study were available on a county-by-county basis, so we also determined the fraction of each county located inside each watershed. Shapefiles of the counties in Georgia, North and South Carolina, and Virginia were obtained from the TIGER geographic database of the U.S. Bureau of the Census (USBoC 2005) and overlaid with gaged watershed shapefiles in a GIS to determine the relative proportion of each county located inside each watershed.

Climatic characteristics, including average temperature, precipitation, and number of frost days per year, were calculated for each watershed using gridded data from the DAYMET database (Thornton et al. 1997). Land cover classifications from the early 1990s were obtained from the National Land Cover Characterization dataset (USGS 1999b-d, 2000). As in the SCOPE Project (Boyer et al. 2002), values were averaged over the years 1988-1993. The average slope of watersheds was calculated from the National Elevation Dataset (USGS 1999a) and averaged over each watershed using ArcGIS 9 (ESRI 2004).

Sources of N

Atmospheric Deposition

Wet Deposition. Wet deposition of inorganic N (in kilograms per hectare) was obtained from the National Atmospheric Deposition Network/National Trends Network (NADP 2005) for all stations located in and around the study watersheds (Figure 1.2). At each station, annual values between 1987 and 1996 were averaged in order to bracket the target of the early 1990s. Nine or ten years' worth of data were available for all but five of the nineteen stations used. (Where data were not available for all years, only those values available were averaged.) Average values for each station were plotted in a GIS and the values kriged (ordinary linear kriging, 200 meter pixel size) to create a surface. Surfaces were then clipped to the extent of each watershed and means calculated.

Dry Deposition. The SCOPE Project (Boyer et al. 2002) used a published model for deriving dry deposition estimates from wet deposition measurements. As this model is valid only for the northeastern United States, we obtained annual dry deposition data (in kilograms inorganic N per square kilometer) from the Clean Air Status and Trends Network (CASTNET) (USEPA 2005) for all stations located in and around the watersheds (Figure 1.2). As was the case with wet deposition data, averages of all available data between 1987 and 1996 were taken for each station. Data were kriged using the ordinary linear method with a 200 m pixel size, surfaces were clipped to the watershed extent, and the mean value for each catchment was calculated.

Organic Deposition. Organic nitrogen deposition has been found to be consistently around 30% of total atmospheric N deposition (Neff et al. 2002), which was the estimate used in the SCOPE study (Boyer et al. 2002). The contribution of atmospheric organic nitrogen to total N deposition was therefore calculated from the estimates of inorganic N in wet and dry deposition in each watershed. Following SCOPE (Boyer et al. 2002), half of this organic nitrogen was assumed to be a new input.

Volatilization Export. A proportion of the inorganic N in both fertilizer and animal manure is released to the atmosphere as a result of volatilization. We used information on fertilizer N inputs in each county (described below) to calculate volatilization losses as a percentage of the different types of N found in fertilizer (Battye et al. 1994). For animal manure, we used ammonia emission rates reported in kg N $animal^{-1}$ yr^{-1} (Battye et al. 1994) and multiplied these by the total number of animals in the watershed (see Net Food and Feed Import). Following SCOPE (Boyer et al. 2002), we assumed that 25% of both of these emissions were transported long-range, becoming an export from the watershed, and subtracted them from total atmospheric deposition.

Fertilizer Input

County-by-county fertilizer sales and N content estimates are available for 1985-1991 from a USGS Water Resources Investigations Report (Battaglin and Goolsby 1994). 1991 estimates were used in this study following the SCOPE Project (Boyer et al. 2002). Fertilizer N content data (Battaglin and Goolsby 1994) were weighted by the proportion of agricultural land in each county located inside the watershed. Golf courses and other urban grasses were also considered as fertilized land. To do this, the National Land Cover Characterization dataset (USGS 1999b-d, 2000) was used to extract the area in land cover classes 81-83 (Pasture/Hay, Row Crops, and Small Grains, respectively) and 85 (Urban/Recreational Grasses) for each county and for that portion of each county located inside the watershed. These areas were then used to calculate the proportion of fertilized land inside each watershed. This method assumes that all fertilizer sold in a county is actually used in that same county, which may not be true in all cases.

Net Food & Feed Import

Net food and feed import refers to the total amount of N in food and feed that must be imported into a watershed to sustain the human and animal populations within that watershed; i.e. the quantity of food and feed needed above and beyond that which is already produced in the watershed. This approach was chosen by the SCOPE Project instead of tallying sewage and manure inputs since it represents a new N input to a region rather than a redistribution of N (Howarth 1996). Net food and feed import is calculated by subtracting total N production (by both crops and livestock) from total N consumption (by both humans and livestock). A positive result indicates that food and feed must be brought into a catchment to sustain the N demands of its population; a negative result indicates that the watershed produces more N in food and feed than required by its population, and that there is a net export. Estimating net food and feed import requires estimates of both crop and animal production and animal and human consumption.

Crop Production. Information on crops grown in each county was obtained from the 1992 Census of Agriculture (USDA-NASS 1992) and adjusted for proportion in the watershed. For this study, only crops comprising 1% or more of harvested cropland in a watershed were considered, a total of 18 different crops (Table 1.1). Except for hay, forage, and silage crops, 10% of production was subtracted to account for spoilage and pests (Boyer et al. 2002). Nitrogen content for each crop was calculated in almost all cases through the use of conversion factors available from the USDA's Crop Nutrient Tool (part of the PLANTS database) (USDA/NRCS 2005). The one exception to this was for hay and pastureland, where we used estimates from Lander & Moffitt (1996).

Animal Production. Livestock population data in each county were obtained from the 1992 United States Department of Agriculture's (USDA) Census of Agriculture (USDA-NASS 2004) and adjusted for the proportion of each county in each watershed. Animal populations were multiplied by published annual excretion rates (Van Horn 1998) (Table 1.2). The difference between livestock N consumption and N excretion in manure was assumed to constitute animal production. Following SCOPE (Boyer et al. 2002), 10% was subtracted from the total production to account for spoilage and inedible parts.

Animal Consumption. Animal populations in each watershed, obtained as described above, were multiplied by published annual nitrogen consumption rates for each species (Van Horn 1998) (Table 1.2).

Human Consumption. 1990 county-by-county population data was obtained from the U.S. Bureau of the Census (USBoC 2005). The population in each county was multiplied by the proportion of the county in each watershed, and the total population in the watershed was multiplied by a per-capita annual nitrogen consumption rate of 5 kg yr^{-1} (Garrow et al. 2000).

Biological Nitrogen Fixation

Legumes (family Fabaceae) are able to symbiotically fix nitrogen from the air and thus represent an additional nitrogen input to catchments. Several categories of N-fixing plants were considered.

Crop N fixation. Hay, pastureland, peanuts, and soybeans are all important legume crops in the study area. Acreages of these crops in each county were obtained for 1992 from the Census of Agriculture (USDA-NASS 1992). The area of legumes grown in each watershed (calculated by weighting by proportion of county in each watershed, as above) was multiplied by N fixation rates obtained from the literature (Table 1.3).

Forest N fixation. The SCOPE Project (Boyer et al. 2002) considered both symbiotic and non-symbiotic N fixation in forests. We did not have specific information on the distribution of N-fixing organisms in the southeast, so we followed their methodology for comparability. Non-symbiotic N fixation by free-living soil microbes was assumed to be 40 kg km^{-2} yr^{-1} (Boyer et al. 2002). The symbiotically nitrogen-fixing tree species considered by Boyer et al. (2002) are black locust and alder. Black locust was assumed to make up 10% of oak-hickory forest (Boyer et al. 2002) and to have a fixation rate of 5,000 kg km^{-2} yr^{-1} (Boring and Swank 1984). Alder was assumed to cover 10% of wetland areas (Boyer et al. 2002) and have a fixation rate of 4,000 kg km^{-2} yr^{-1} (Hurd et al. 2001). Applying these same assumptions, we obtained data on forest coverage from the USDA's Forest Inventory and Analysis Program (FIA) (USDA-FS 2005) to determine the acreage of oak-hickory forest (again weighted by proportion of county in each watershed). Wetland areas were derived from early 1990s land cover estimates (USGS 1995), using classes 91 and 92 (Woody Wetlands and Emergent Herbaceous Wetlands, respectively).

One additional input from biological nitrogen fixation that was not included in these budgets is kudzu (genus *Pueraria*). Kudzu is an invasive leguminous vine that covers a substantial portion of the southeastern United States. Estimates of area are difficult to come by, but could range as high as 118,000 ha for the four southeastern states represented in this study (Darryl Jewett, unpublished data). Nitrogen fixation in the species of kudzu prevalent throughout the southeastern U.S. (*Pueraria montana var. lobata*) has not been well studied, and we have therefore not included kudzu in our estimates of biological nitrogen fixation. Estimates of nitrogen fixation in tropical species of kudzu are as high as 3,000 kg N ha^{-1} (Vieravargas et al. 1995). If we assume that temperate species of kudzu have only half the N-fixation rate reported for tropical species, inputs of reactive nitrogen due to fixation by kudzu could represent a substantial source of N to these budgets (possibly as much as an additional 15%). The total N inputs presented here are therefore likely underestimates, and should be adjusted if data for kudzu become available.

Non-food crop export. Cotton and tobacco are two important southeastern crops that are not consumed by humans or livestock. We assumed that virtually all of these crops are harvested for sale elsewhere and therefore considered them as a non-food crop export. 10% of both cotton and tobacco was assumed to be lost to spoilage or pests. Neither cotton nor tobacco is a crop frequently grown in the northeast, so this represents a deviation from the methodology of SCOPE (Boyer et al. 2002). However, the amount of N associated with this adjustment represented less than one percent of the total N.

Export

N export to the coast was estimated based on water quality and discharge information from the USGS National Water Information System (USGS 2005). Measurements of total

N concentration were obtained for the water quality gage located furthest downstream on each river (Table 1.4). Measurements were made irregularly; we used all available measurements between 1987 and 1996. Whole water concentrations of total Kjeldahl nitrogen (TKN) (parameter #625) and nitrite + nitrate (parameter #630) (in mg L^{-1}, as N) were multiplied by discharge on the day of observation at the same or a nearby gaging station to obtain an estimate of load for each parameter. Average load for each constituent was then summed to get total N load, and divided by watershed area.

There was substantial variation in the number of observations available for a given parameter and gage, ranging from 5 to 110 samples (Table 1.4). N export estimates with a high sampling frequency, such as the Savannah River, are therefore more reliable than those with a low number of available samples, such as the Wateree (a branch of the Santee). On average, there were 40 observations of TKN and 34 observations of nitrite + nitrate.

Three of the watersheds posed special difficulties. The water quality station near Eden, GA, on the Ogeechee River (#02202500) was unusually far upstream. In order to be able to include as much of the watershed as possible in this study, nitrogen export from a water quality station near Claxton, GA on the Canoochee River (#02203000), whose confluence with the Ogeechee River is below Eden, was added to the export from the Ogeechee River. The same approach was used for the Santee River watershed in South Carolina, where gages on the Congaree (#02169500) and Wateree (#02148000) Rivers were combined. In the case of the Altamaha watershed, water quality data from the Gardi, GA gage station (#02226010) was correlated with flow data from the Doctortown, GA gage (#02226000), located approximately 7 km away.

In most of the cases, data were available for both organic and inorganic N in whole water (unfiltered) samples. However, in the Canoochee River (a branch of the Ogeechee), only inorganic N was measured. Although there was no clear relationship between TKN and nitrite + nitrate, organic N averaged 64% of total N in the other rivers examined, which corresponds to an additional 31 kg N km^{-2} yr^{-1}. In the two arms of the Santee River (the Wateree and Congaree Rivers), only dissolved nitrite + nitrate samples were available. This does not, however, represent a large source of error since filtered nitrogen values were often equivalent to unfiltered values in other systems.

Results

General Watershed Characteristics

Watershed area ranged from 3,274 km^2 for the Black River to 35,112 km^2 for the Altamaha River. Most watersheds included in the study were dominated by forest, with the area-weighted average for all systems at 60% (Table 1.5). Agriculture was the next-most important land use, averaging 25%. Other land uses were wetland (averaging 7%), urban (4%), water (2%), and other (4%). The area-weighted population density was 58 persons km^{-2}. Temperature increased from north to south, with an area-weighted annual average of 16.3°C. Precipitation was relatively constant, with an area-weighted annual average of 1,244 mm yr^{-1}.

Inputs

The total N inputs into the watersheds studied ranged from 2,762 kg N km^{-2} yr^{-1} in the watershed of the Savannah River to 6,232 kg N km^{-2} yr^{-1} in that of the Santee, with an overall area-weighted average of 3,499 kg N km^{-2} yr^{-1} (Table 1.6).

Nitrogenous fertilizer was the most important N input to most watersheds, accounting for, on average, 34% of total inputs. Fertilizer ranged from 603 kg N km^{-2} yr^{-1} in the Savannah watershed to 2,262 kg N km^{-2} yr^{-1} in the Neuse River watershed. The area-weighted average for all watersheds studied was 1,189 kg N km^{-2} yr^{-1}.

Net import of food and feed varied greatly, but on average was the second-most important input N to these watersheds, accounting for 27% of the total. Import was highest in the Santee River watershed (2,621 kg N km^{-2} yr^{-1}). This was driven mainly by the large livestock population in this watershed, which led to a high animal consumption value. The Black River watershed was the only watershed that exhibited a net export of food and feed (-210 kg N km^{-2} yr^{-1}). The Black River is the smallest watershed included in this study, and it is therefore possible that the export reflects on uneven distribution of animal and human populations. For example, it would be possible for a lot of cropland to be inside a watershed while the human population that those crops sustain lives in a town right outside the watershed boundary, thus causing an N export.

An overall average of 25% of total N was due to biological N fixation in crop and forest lands. The majority of the fixed nitrogen was a result of agricultural N fixation, which accounted for 88% of the total biological N fixation. Forest nitrogen fixation was a relatively small proportion of overall N fixation, with the exception of the Satilla watershed in southern Georgia. The Satilla watershed contains a substantial amount of wetland area (18%) and few N-fixing crops; therefore, symbiotic N fixation from alder represents a larger proportion of N input due to biological fixation here than in most other watersheds in the southeast. Nitrogen fixation ranged from a high of 1,592 kg N km^{-2} yr^{-1} in the Pee Dee watershed to a low of 248 kg N km^{-2} yr^{-1} in the Satilla watershed.

Net atmospheric deposition accounted for, on average, 15% of N inputs. The highest atmospheric deposition was observed in the Roanoke watershed (758 kg N km^{-2} yr^{-1}) and the lowest in the Santee watershed (421 kg N km^{-2} yr^{-1}), with an area-weighted average of 538 kg N km^{-2} yr^{-1}.

Output

Flow ranged from 275 mm yr^{-1} in the Satilla to 1,238 mm yr^{-1} for the combined branches of the Santee River (Wateree and Congaree) (Table 1.6b). The average for the watersheds included in this study was 774 mm yr^{-1}, though all watersheds other than the Santee fell below this average. On an areal basis, total N export was lowest in the Black River (158 kg N km^{-2} yr^{-1}) and highest in the Santee River (899 kg N km^{-2} yr^{-1}), averaging 331 kg N km^{-2} yr^{-1}. Riverine N export was linearly related to the net anthropogenic N input (R^2=0.84, Figure 1.3). On average, 9% of total N inputs was exported in stream flow in these systems.

Comparison with the Northeast

N inputs

The nitrogen inputs reported here for southeastern watersheds can be directly compared to those compiled for northeastern watersheds as part of the SCOPE Project

(Boyer et al. 2002). Total N inputs to the 16 watersheds considered by Boyer et al. (2002) ranged from 835 kg N km^{-2} yr^{-1} to 5,717 kg N km^{-2} yr^{-1} with an area-weighted average of 3,088 kg N km^{-2} yr^{-1}, which overlaps with the N inputs estimated here (2,762-6,232 kg N km^{-2} yr^{-1}). The area-weighted average input for the northeastern watersheds was comparable, although slightly lower (3,088 vs. 3,499 kg N km^{-1} yr^{-1}). However, it should be noted that only the gaged areas of watersheds were considered in these budgets. Since many major northeastern population centers are located on the coast, N inputs from a number of large cities such as Boston, New York, and Washington D.C. are either excluded or only partially included in the budgets compiled in the SCOPE estimates. With the exception of the city of Savannah, there are no major cities located downstream of southeastern gages.

We observed differences in the proportion contributed by atmospheric deposition and nitrogenous fertilizer in the two regions (Table 1.6a). Atmospheric deposition dominated in the northeast, contributing 37% of total nitrogen inputs (Boyer et al. 2002) as opposed to only 15% in the southeast. In contrast, fertilizer was the dominant input in the southeast, contributing 34% of the total N inputs as compared to only 14% of total inputs in the northeast (Boyer et al. 2002). Net N import in food and feed, crop N fixation, and forest N fixation contributed approximately the same proportion to total inputs in both the northeast and the southeast (27% vs. 25%, 22% vs. 21%, and 3% vs. 3%, respectively).

N export

In the northeast, riverine export constituted an average of 25% of total nitrogen input (Boyer et al. 2002). In contrast, the export from the southeast averaged only 9% of inputs

(Table 1.6b). When the data from both studies are plotted together, they fall into two distinct groups (Figure 1.4). Interestingly, the two southernmost systems included in the SCOPE analysis (Boyer et al. 2002), the James and the Rappahannock River watersheds in Virginia, cluster with the southern systems. When these two watersheds were shifted from the northeastern to the southeastern group, relationship between N input and riverine export for northeastern watersheds was greatly improved (R^2 increased from 0.62 to 0.80, whereas that for the southeast fell slightly from 0.84 to 0.82) (Figure 1.4). Therefore, both of these systems were included with the southeastern group of watersheds for the remainder of this analysis.

Discussion

Nitrogen inputs to the watersheds considered in this study ranged from 2,762 kg N km^{-2} yr^{-1} to 6,232 kg N km^{-2} yr^{-1}. Although there are no direct comparisons for these observations given differences in methodology, these numbers are within range of previous reports. Castro et al. (2003) estimated N inputs to the Pamlico River to be 4,320 kg N km^{-2} yr^{-1} and inputs to the Altamaha River to be 2,830 kg N km^{-2} yr^{-1}, which were similar to our estimates of 4,118 kg N km^{-2} yr^{-1} and 3,099 kg N km^{-2} yr^{-1}, respectively. In contrast, estimates of N inputs by Asbury and Oaksford (1997) were substantially different from ours. They calculated inputs of 5,470 kg N km^{-2} yr^{-1} to the Altamaha River (3,099 kg N km^{-2} yr^{-1}, this study), 5,430 kg N km^{-2} yr^{-1} to the Satilla River (3,203 kg N km^{-2} yr^{-1}, this study), and 2,690 kg N km^{-2} yr^{-1} to the Ogeeechee River (3,098 kg N km^{-2} yr^{-1}, this study). It is unclear what caused these discrepancies, as there are numerous differences between the two studies. Much of the difference is probably due to the fact that they used conversion factors for animal excretion that were often two to three-fold greater than those used here. However, both Asbury and Oaksford (1997) and Castro et al. (2003) found that agricultural inputs tended to predominate, which was consistent with this study.

The southeastern watersheds exhibited a lower percent of nitrogen export than watersheds in the northeastern U.S. This is consistent with observations in other large-scale overview studies (Howarth et al. 1996, Castro et al. 2003). The observed differences between southeastern and northeastern watersheds suggested a potential latitudinal effect on the proportion of nitrogen exported in rivers. When the percentage of total inputs that are lost to riverine export is plotted against latitude (Figure 1.5), southeastern watersheds showed very little variation, ranging from 5% to a maximum of 14%, with most (including the James and Rappahannock watersheds) falling between 7 and 11% of total N inputs exported. In the northeastern watersheds, the percent of inputs exported varied substantially, ranging from 19 to 38%. The proportion exported shows an increasing trend with latitude, although no such trend was observed in the southeast. These results suggest that there is a threshold in the efficiency of nitrogen processing between 38 and 39 degrees north latitude (near the Virginia-Maryland border).

We did an exploratory analysis to examine various factors that could have caused the observed differences in proportionate N export between watersheds in the northeastern and southeastern U.S. Although the N budgets varied among watersheds, neither the relative nor the absolute amount of N contributed by the various sources (fertilizer input, net food and feed import, atmospheric deposition, and biological nitrogen fixation) had any relationship to the amount of nitrogen exported in stream flow, nor were any of the subcomponents of these budgets (e.g. NH_X deposition or total animal consumption) related to N exported. The proportion of any given type of land cover in a watershed (e.g. wetland, urban) also did not correlate with the percentage of exported N.

It has been suggested that precipitation may have an effect on nitrogen export, with higher precipitation leading to greater export due to a shorter flushing time through the watershed (Howarth et al. 2006). Although this relationship holds true for northeastern watersheds, some of the more northerly watersheds also receive more precipitation so it is difficult to separate the effects of latitude from precipitation. This is not the case in the southeast region where precipitation increases with decreasing latitude (though with less variation in precipitation than in the northeast). When the southeastern group of watersheds is included in the analysis, there is no relationship with precipitation (Figure 1.6).

Another potential factor that could control N export is watershed slope. Water will run off a steeper slope more quickly, allowing less time for N-processing. Average slope for all watersheds was calculated from the national Elevation Dataset (USGS 1999a) using a GIS. Although the northeastern group of watersheds had a higher average slope than the southeastern group (7 and 4 degrees, respectively), it did not explain N export (Figure 1.7). In fact, the northeastern watersheds show a negative relationship between slope and riverine N export, which is counterintuitive as runoff will increase with increasing slope, and thus one would expect N export to increase as well. The southeastern group of watersheds actually showed a far greater range in slope than the northeastern watersheds, yet there was still no relationship with export.

There are numerous other possibilities for the differences in N-processing, many of which are climate-driven and thus co-vary with latitude. When percentage N exported is plotted versus number of frost days and average watershed temperature, the relationships look very similar to that obtained when exports are plotted against latitude (Figure 1.8a, b). The break point at 39°N latitude corresponds to an average watershed temperature of 12°C and approximately 120 frost days per year. This could be related to numerous processes: a greater rate of tree growth in southern watersheds could sequester more nitrogen, or increased snow cover in the north could control the rates of microbial activity. In order to explain the break at 39°N latitude, however, it would be useful to identify a process that has an identifiable threshold as opposed to factors that vary along a continuum.

Numerous studies have shown that denitrification rates in both terrestrial and aquatic systems decrease most substantially below 10-12°C. Stanford et al. (1975) found the denitrification rate in soils to begin a sharp decline between 10 and 15°C and estimated the break point to be at 11°C. This decline was most pronounced between 5 and 10°C. Malhi et al. (1990) also found denitrification rates in soils to decline most rapidly between 5 and 10°C. This pattern also has been shown in aquatic systems. Seitzinger (1988) reported denitrification rates in sediments from Narragansett Bay to decline by half between 3 and 10°C, but remain steady between 10 and 20°C. Pfennig and McMahon (1997) also found denitrification to be lower in riverbed sediments incubated below 5°C than in those incubated above 10°C. More recently, Addy et al. (2005) found denitrification to be significantly lower in marsh sediment below 12°C.

The data presented here show an increase in the percentage of nitrogen exported from watersheds at temperatures below 12°C, which is consistent with the sharp decline in denitrification described above. Thus, we propose that temperature, by regulating denitrification rates, controls watershed nitrogen export. As one moves further north, the temperature is below 10-12°C for a greater and greater portion of the year, decreasing the time that denitrification can occur at the optimal rate. Thus, this could explain the higher export percentages for more northerly watersheds.

Although the results reported here strongly suggest that watershed N export is controlled by differential rates of denitrification, an examination of the data suggests that other factors could also play a role. It is possible that very high N inputs overwhelm the ability of denitrifiers to process the material, thereby resulting in less removal. The two watersheds with the highest overall total N inputs also showed low removal rates: the Santee had the highest percentage export in the southeastern group and the percent export in the Schuylkill was also higher than might otherwise be expected based on its N input. Denitrification removes a greater amount of N per river mile in higher-order streams than in lower-order streams (Seitzinger et al. 2002), so smaller watersheds without substantial lengths of high-order streams can be expected to show a greater percentage of N export. The two smallest watersheds examined (those of the Charles and Blackstone Rivers) had less proportionate removal than other systems with comparable average temperatures. When these four systems are excluded from the analysis there is a significant piece-wise negative exponential relationship between temperature and proportionate export, with a greater rate of change at lower temperatures (Figure 1.8b). This is in accordance with what would be expected given the reported relationship between denitrification and temperature. We infer from these observations that temperature is the primary factor controlling microbial processing of N (and therefore watershed N export), but the capacity of the microbes can be exceeded by high total loads or in small watersheds. The location of the load in the watershed may also be a factor in this regard, as N input closer to the coast would have less time to be processed.

The results obtained here have implications for eutrophication in light of future climate change. With a doubling of atmospheric CO_2, global average temperature is expected to increase anywhere between 1.5-4.5°C (Houghton et al. 2001) and up to 5°C in winter in the northern section of the United States (Moore et al. 1997). If the relationships presented here hold, then predicted increases in watershed temperature would be expected to shift the break point further north, thereby resulting in increased N-processing within more northerly watersheds and a lower proportionate export to the coastal zone in some areas. It must be noted that concurrent increases in N inputs, which are also predicted (Howarth et al. 2002, Galloway et al. 2004), would act to counterbalance these trends. Nevertheless, the observations presented here suggest that nitrogen removal will decrease the proportion of N exported to estuaries, potentially resulting in reduced rates of eutrophication.

Table 1.1. Conversion factors used to calculate N estimates for various crops

Crop	**lb N**	**Reference(s)**
Apples	0.0006/lb of fruit	USDA/NRCS 2005
Barley (grain)	0.9655/bushel	USDA/NRCS 2005
Cotton	0.0304/lb of seed and lint	USDA/NRCS 2005
Corn (grain)	0.7929/bushel	USDA/NRCS 2005
Corn (silage)	7.7501/ton	USDA/NRCS 2005
Cropland used only for pasture	2000/acre	Lander & Moffitt 1996
Non-crop pastureland	1000/acre	Lander & Moffitt 1996
Hay, alfalfa	55.7759/ton	USDA/NRCS 2005
Hay, non-alfalfa	21.7/ton	Lander & Moffitt 1996
Oats (grain)	0.5984/bushel	USDA/NRCS 2005
Peanuts	0.0448/lb of seed	USDA/NRCS 2005
Soybeans	0.0144/bushel	USDA/NRCS 2005
Rye	1.0557/bushel	USDA/NRCS 2005
Peaches	0.0012/lb of fruit	USDA/NRCS 2005
Pecans	0.238/lb of nut	USDA/NRCS 2005
Sweetpotatoes	0.0028/lb of root	USDA/NRCS 2005
Tobacco	1.7064/hundredweight	USDA/NRCS 2005
Wheat (grain)	1.4366/bushel	USDA/NRCS 2005

Table 1.2. Animal N consumption, excretion, and manure volatilization rates. N consumption and excretion rates from van Horn (1998); manure volatilization rates from Battye et al. (1994). All values in kg N animal^{-1} year^{-1}

Animal	N consumption	N excretion	N volatilization
Cattle, beef	66.75	58.51	18.83
Cattle, dairy	156.00	121.00	18.83
Cattle, young	Same as adult	Same as adult	10.72
Chickens (broilers)	0.13	0.07	0.14
Chickens (layers)	0.84	0.55	0.20
Goats	5.97	5.00	5.26
Horses	44.80	40.00	10.03
Pigs & hogs	8.51	5.84	4.20
Sheep	5.97	5.00	2.77
Turkeys	0.62	0.39	0.71

Table 1.3. Rates of biological nitrogen fixation in crops.

Crop	kg N km^{-2} yr^{-1}	Reference(s)
Hay, alfalfa	22,400	Heichel et al. 1984
Hay, non-alfalfa	11,700	Lander & Moffitt 1996, cited in Boyer et al. 2002
Pastureland, all types	1,500	Jordan & Weller 1996
Peanuts	8,000	Smil 1999
Soybeans	9,600	Lander & Moffitt 1996, cited in Boyer et al. 2002

Table 1.4. USGS water quality gaging stations and sampling information used to calculate riverine N export. Parameter codes refer to USGS methods to sample various N constituents in water samples. 610: Ammonia (unfiltered); 625: Total Kjeldahl Nitrogen (unfiltered); 630: Nitrite plus nitrate (unfiltered); 631: Nitrite plus nitrate (filtered).

Watershed	USGS Station No.	Location	Parameter Code	Sampling Period	No. of Observations
Roanoke (ROA)	02080500	Roanoke River at Roanoke Rapids, NC	625	3/31/87-9/16/92	18
Roanoke (ROA)	02080500	Roanoke River at Roanoke Rapids, NC	630	3/31/87-9/16/92	6
Pamlico (PAM)	02083500	Tar River at Tarboro, NC	625	2/28/87-10/4/95	59
Pamlico (PAM)	02083500	Tar River at Tarboro, NC	630	2/28/87-10/4/95	6
Neuse (NEU)	02089500	Neuse River at Kinston, NC	625	1/8/87-11/14/96	67
Neuse (NEU)	02089500	Neuse River at Kinston, NC	630	12/12/90-9/16/96	11
Cape Fear (CFR)	02105769	Cape Fear River at Lock #1 near Kelly, NC	625	5/14/87-9/29/95	20
Cape Fear (CFR)	02105769	Cape Fear River at Lock #1 near Kelly, NC	630	15/14/87-9/12/92	7
Pee Dee (PEE)	02131000	Pee Dee River at Peedee, SC	625	1/27/87-9/28/95	49
Pee Dee (PEE)	02131000	Pee Dee River at Peedee, SC	630	11/19/90-11/16/92	13
Black (BLK)	02136000	Black River at Kingstree, SC	625	1/14/87-9/15/89	17
Black (BLK)	02136000	Black River at Kingstree, SC	630	1/14/87-11/16/88	12
Santee (SNT)	02148000	Wateree River near Camden, SC	625	12/4/95-11/7/96	5
Santee (SNT)	02148000	Wateree River near Camden, SC	631	12/4/95-11/7/96	5
Santee (SNT)	02169500	Congaree River at Columbia, SC	625	10/23/95-12/5/96	16
Santee (SNT)	02169500	Congaree River at Columbia, SC	631	10/23/95-12/5/96	16
Edisto (EDI)	02175000	Edisto River near Givhans, SC	625	1/28/87-12/2/96	78
Edisto (EDI)	02175000	Edisto River near Givhans, SC	630	11/29/90-11/19/92	13
Savannah (SAV)	02198500	Savannah River near Clyo, GA	625	1/20/87-12/12/96	81
Savannah (SAV)	02198500	Savannah River near Clyo, GA	630	1/20/87-12/12/96	110
Ogeechee (OGE)	02202500	Ogeechee River near Eden, GA	625	1/6/87-4/19/95	49
Ogeechee (OGE)	02202500	Ogeechee River near Eden, GA	630	10/22/90-11/3/92	13
Ogeechee (OGE)	02203000	Canoochee River near Claxton, GA (Ogeechee)	610	1/20/87-6/11/92	66
Ogeechee (OGE)	02203000	Canoochee River near Claxton, GA (Ogeechee)	630	1/20/87-6/11/92	66
Altamaha (ALT)	02226010	Altamaha River near Gardi, GA[1]	625	1/7/87-6/8/93	38
Altamaha (ALT)	02226010	Altamaha River near Gardi, GA[1]	630	1/7/87-12/10/96	107
Satilla (SAT)	02228000	Satilla River at Atkinson, GA	625	1/7/87-8/24/93	27
Satilla (SAT)	02228000	Satilla River at Atkinson, GA	630	1/7/87-2/17/94	39

[1]Streamflow data from USGS Station 02226000 (Altamaha River at Doctortown, GA) used to calculate load (see text).

Table 1.5. General characteristics of watersheds considered in this study.

Watershed	**Abbreviation**	**Area km^2**	**Persons km^{-2}**	**Mean Temp °C**	**Precip mm yr^{-1}**	**Flow mm yr^{-1}**	**Forest %**	**Agric. %**	**Urban %**	**Wetl. %**	**Water %**	**Other %**
Roanoke	ROA	21,984	40	13.8	1,181	352	69.6	22.2	2.8	1.7	2.5	1.4
Pamlico	PAM	5,748	35	15.2	1,155	334	58.8	26.5	2.7	10.3	0.6	1.0
Neuse	NEU	7,033	103	15.7	1,200	341	51.0	29.3	7.6	9.8	1.5	0.7
Cape Fear	CFR	13,599	82	15.7	1,186	355	62.8	20.8	7.0	6.0	1.5	1.8
Pee Dee	PED	21,448	62	15.4	1,220	467	61.2	27.0	5.5	3.8	1.1	1.4
Santee	SNT	11,105	205	15.6	1,276	1,238	69.7	18.1	7.0	0.8	2.2	2.1
Black	BLK	3,274	32	17.4	1,213	286	33.3	43.5	3.0	18.1	0.2	1.9
Edisto	EDI	6,944	39	17.9	1,259	337	45.0	32.3	1.6	15.2	0.7	5.3
Savannah	SAV	25,488	34	16.5	1,339	418	65.9	18.0	2.8	4.7	3.6	4.9
Ogeechee	OGE	8,415	12	18.1	1,260	330	44.9	33.6	0.7	14.2	0.6	5.9
Altamaha	ALT	35,112	51	17.8	1,252	339	57.9	24.5	3.5	7.3	1.2	5.7
Satilla	SAT	7,248	14	19.3	1,299	275	45.9	30.4	1.0	14.4	0.6	7.7
Area-weighted average			*58.1*	*16.3*	*1,244*	*425*	*59.7*	*24.5*	*3.9*	*6.6*	*1.7*	*3.5*

Table 1.6a. Watershed nitrogen inputs. All values in kg N km^{-2} yr^{-1}.

Watershed	**Net Atmospheric N Dep.**	**Nitrogenous Fertilizer**	**N fixation in forest lands**	**N fixation in agricult. lands**	**Net N import in food & feed**	**Non-food crop export**	**Total nitrogen inputs**
Roanoke	758	821	46	697	601	34	2,889
Pamlico	604	1,892	97	848	803	127	4,118
Neuse	560	2,262	89	824	1,178	28	4,884
Cape Fear	516	1,061	81	530	1,458	40	3,604
Pee Dee	428	1,181	62	1,530	888	50	4,039
Santee	421	1,603	160	1,430	2,621	3	6,232
Black	609	2,010	118	839	-210	84	3,282
Edisto	516	1,306	104	551	465	29	2,913
Savannah	481	603	110	521	1,053	5	2,762
Ogeechee	638	1,594	159	730	5	27	3,098
Altamaha	529	1,138	132	572	750	22	3,099
Satilla	468	1,678	111	137	817	8	3,203
Area-weighted averages							
This study	*538*	*1,189*	*102*	*770*	*931*	*30*	*3,499*
SCOPE (Boyer et al. 2002)	*959*	*474*	*167*	*740*	*748*	-	*3,088*

Table 1.6b. Watershed nitrogen stream export.

Watershed	**Flow (mm yr^{-1})**	**TKN (kg N km^{-2} yr^{-1})**	**Nitrite + nitrate (kg N km^{-2} yr^{-1})**	**Total Streamflow N export (kg N km^{-2} yr^{-1})**	**% of N inputs exported in stream flow**	**% of N inputs stored or lost in basin**
Roanoke	352	130	67	197	7	93
Pamlico	334	265	181	446	11	90
Neuse	341	224	222	446	9	91
Cape Fear	355	165	83	248	7	94
Pee Dee	467	241	149	390	10	91
Santee	1,238	461	438*	899	14	86
Black	286	139	19	158	5	95
Edisto	337	167	61	228	8	93
Savannah	418	179	93	272	10	91
Ogeechee	330	222†	62	283	9	91
Altamaha	339	185	88	273	9	92
Satilla	275	339	26	365	11	89
Area-weighted average	*774*	*197*	*99*	*328*	*9*	*91*

*Filtered
†Ammonia only for Congaree branch

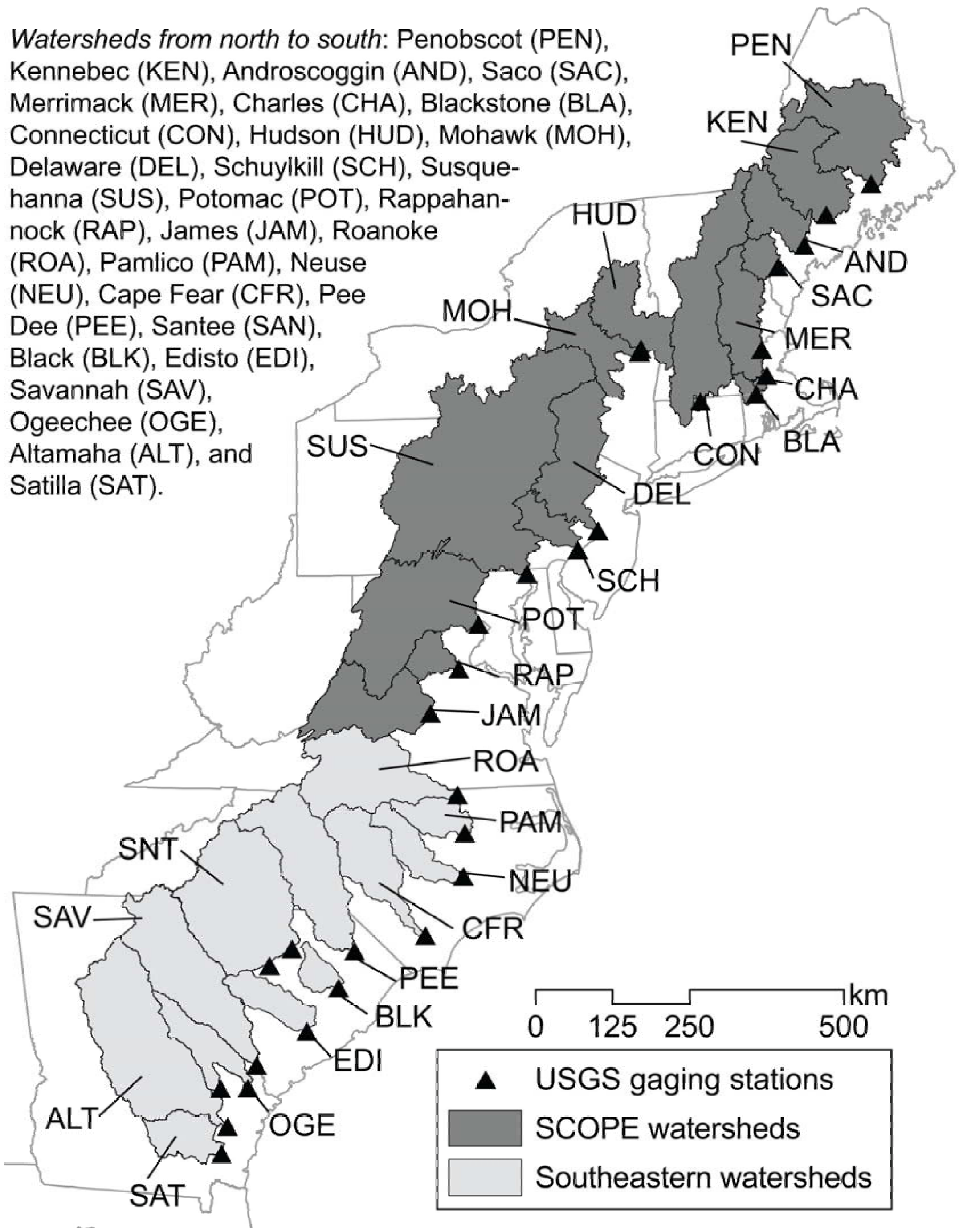

Figure 1.1. Location of eastern U.S. riverine watersheds considered in this study. Watershed areas were delineated to the most downstream USGS water quality gauging stations (represented by triangles) based on data from the National Elevation Dataset (USGS 1999a). For watersheds with two gaging stations, area contributing to either gauge was included. Analyses were done using ESRI ArcGis9 software.

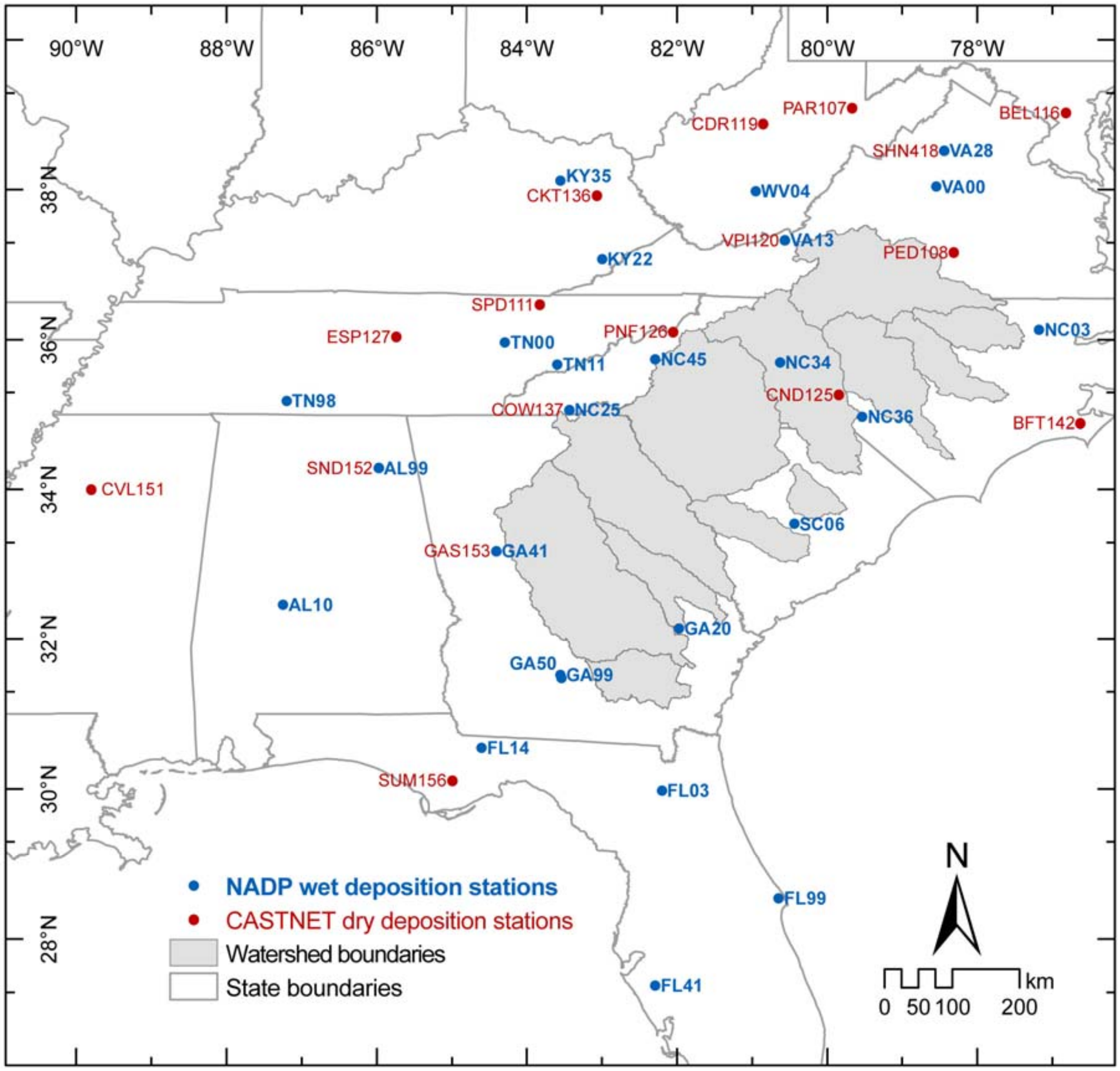

Figure 1.2. Locations and site codes of wet (NADP) and dry (CASTNET) deposition stations used to estimate atmospheric N deposition to the study area.

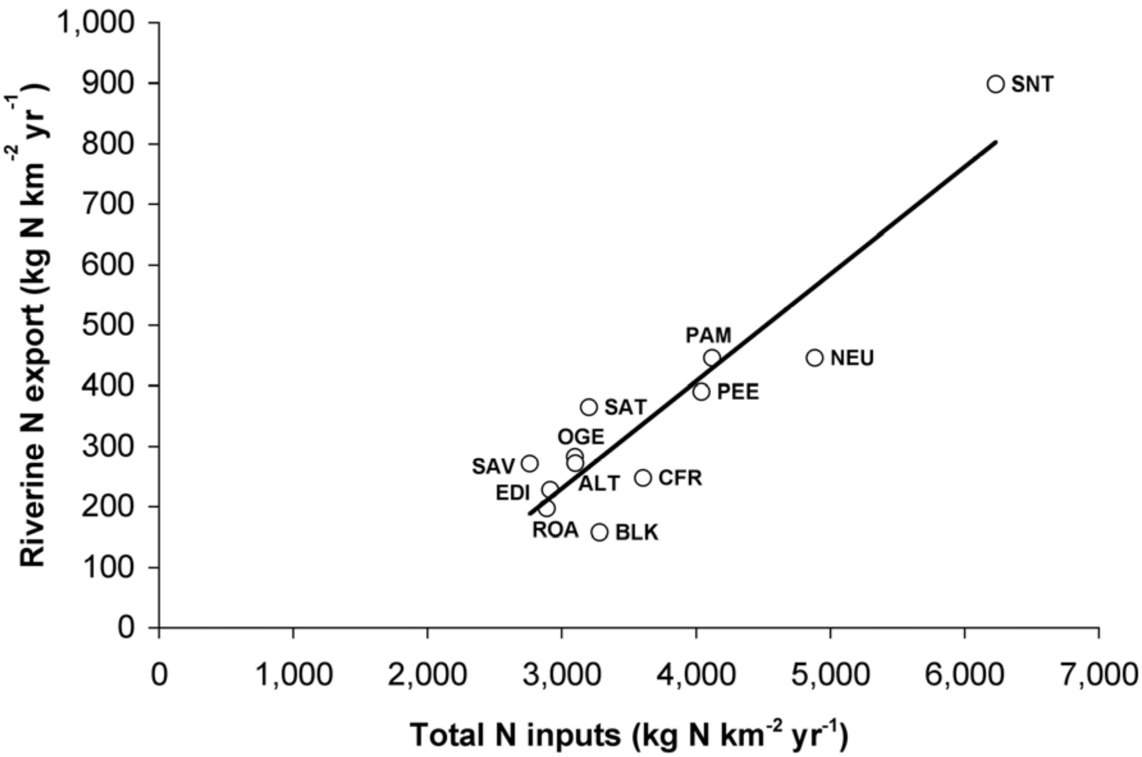

Figure 1.3. Total N inputs vs. stream export for southeastern watersheds. Line equation is riverine N export = 0.18 * total N inputs – 300.95 (R^2=0.84; p<0.0001). Watershed abbreviations as in Figure 1.1.

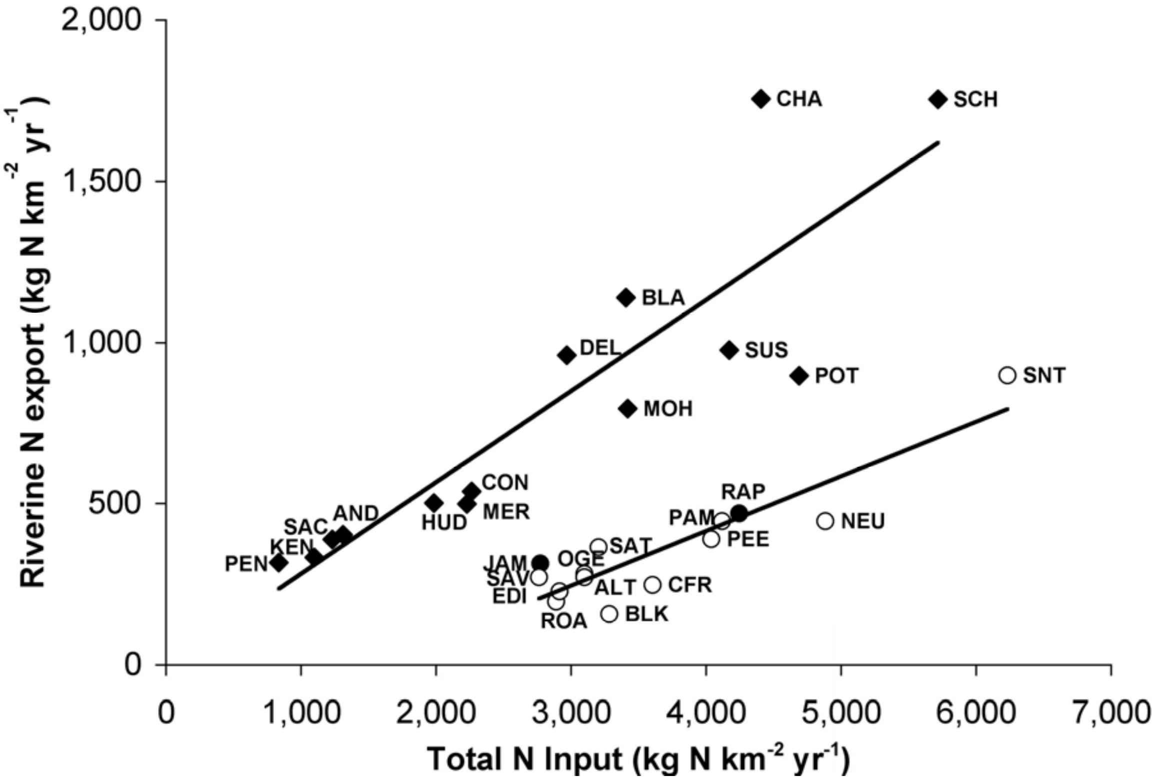

Figure 1.4. Riverine export of N plotted against total nitrogen inputs for watersheds of the eastern U.S. (filled symbols, SCOPE watersheds; open symbols, this study). Riverine N export was calculated from observations of total N concentration and flow at the most downstream USGS water quality monitoring gage in each river. N input is the sum of net atmospheric deposition, fertilizer inputs, net food and feed import, and biological N fixation. For the northern and mid-Atlantic watersheds (diamonds), the line equation is: riverine export = 0.28* N input - 0.04, (R^2=0.80, p<0.0001). For the southeastern watersheds (circles), the line equation is: riverine export = 0.17 * N input – 262.30, (R^2=0.82, p<0.0001). Watershed abbreviations can be found in Figure 1.1. All data have been normalized to watershed area.

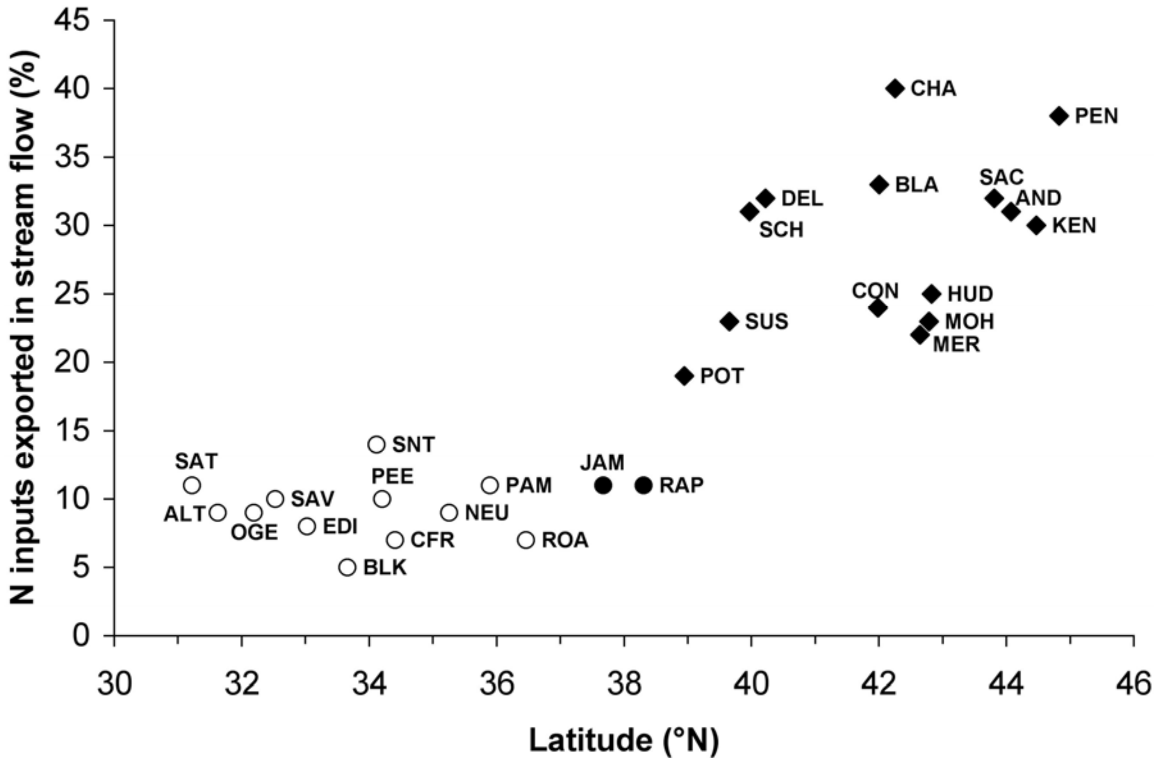

Figure 1.5. Relationship between the percentage of N input exported in stream flow and latitude for watersheds of the eastern U.S. Latitudes are those of USGS water quality stations used to calculate riverine export. Symbols and watershed abbreviations as in Figure 1.4.

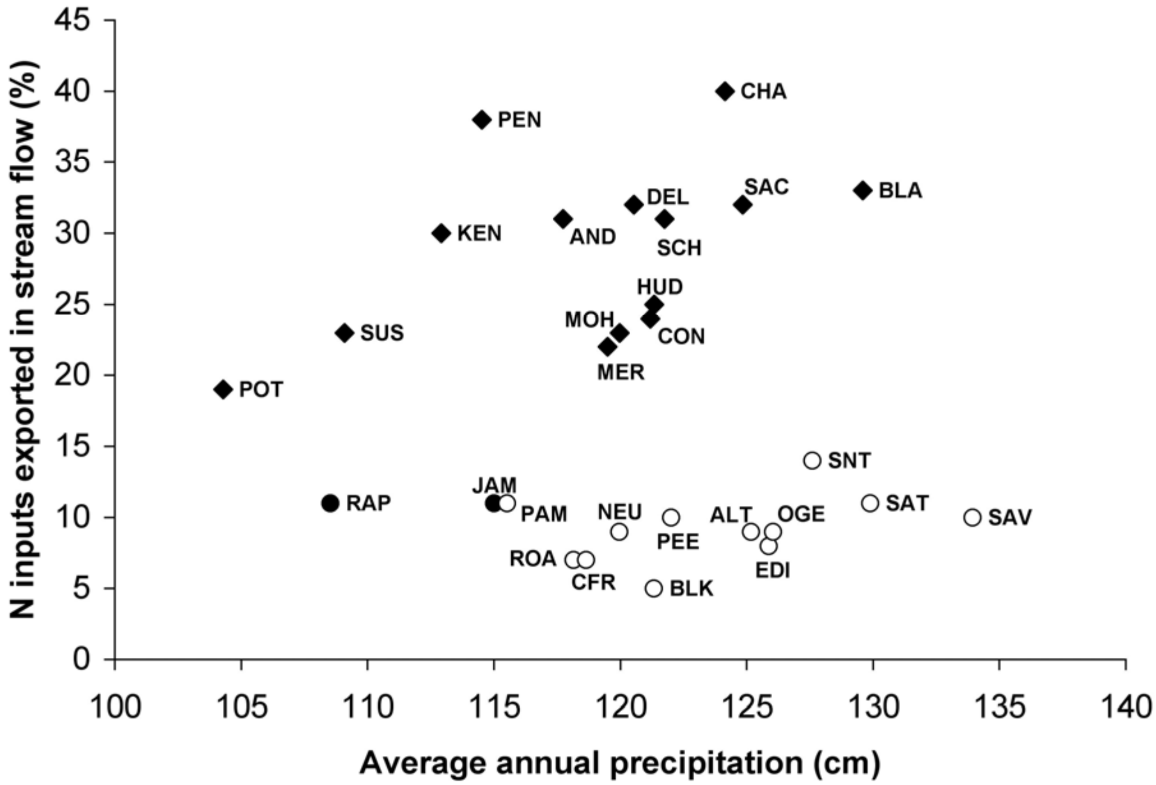

Figure 1.6. Relationship between the percentage of N input exported in stream flow and average precipitation for watersheds of the eastern U.S. Precipitation and temperature were calculated from DAYMET gridded climate data (Thornton et al. 1997) using ESRI ArcGIS 9 software. Neither the southern nor the northern and mid-Atlantic watersheds had a significant relationship between precipitation and % N input exported. Symbols and watershed abbreviations as in Figure 1.4.

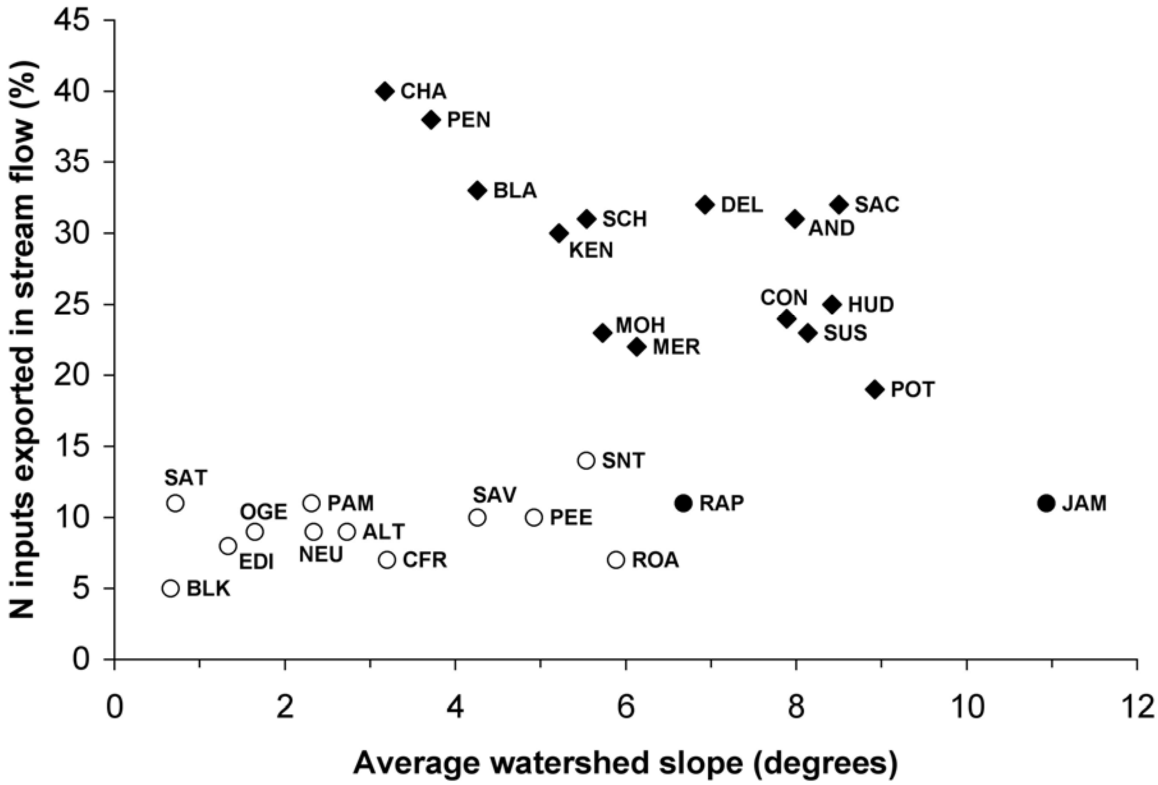

Figure 1.7. Relationship between the percentage of N input exported in stream flow and average watershed slope for watersheds of the eastern U.S. Watershed slopes were calculated using the National Elevation Dataset (USGS 1999a) using ESRI ArcGIS 9 software. Neither the southern nor the northern and mid-Atlantic watersheds had a significant relationship between average watershed slope and % N input exported. Symbols and watershed abbreviations as in Figure 1.4.

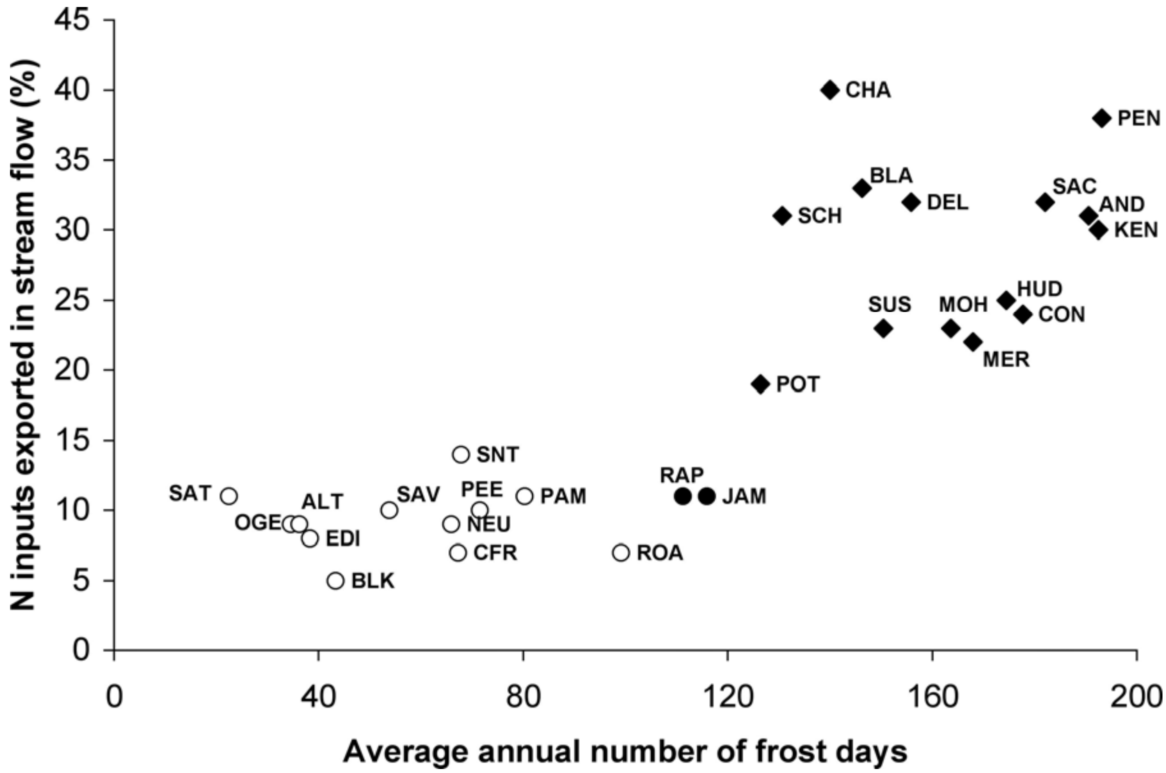

Figure 1.8a. Relationship between the percentage of N input exported in stream flow and average annual number of frost days for watersheds of the eastern U.S. Precipitation and temperature were calculated from DAYMET gridded climate data (Thornton et al. 1997) using ESRI ArcGIS 9 software. Symbols and watershed abbreviations as in Figure 1.4.

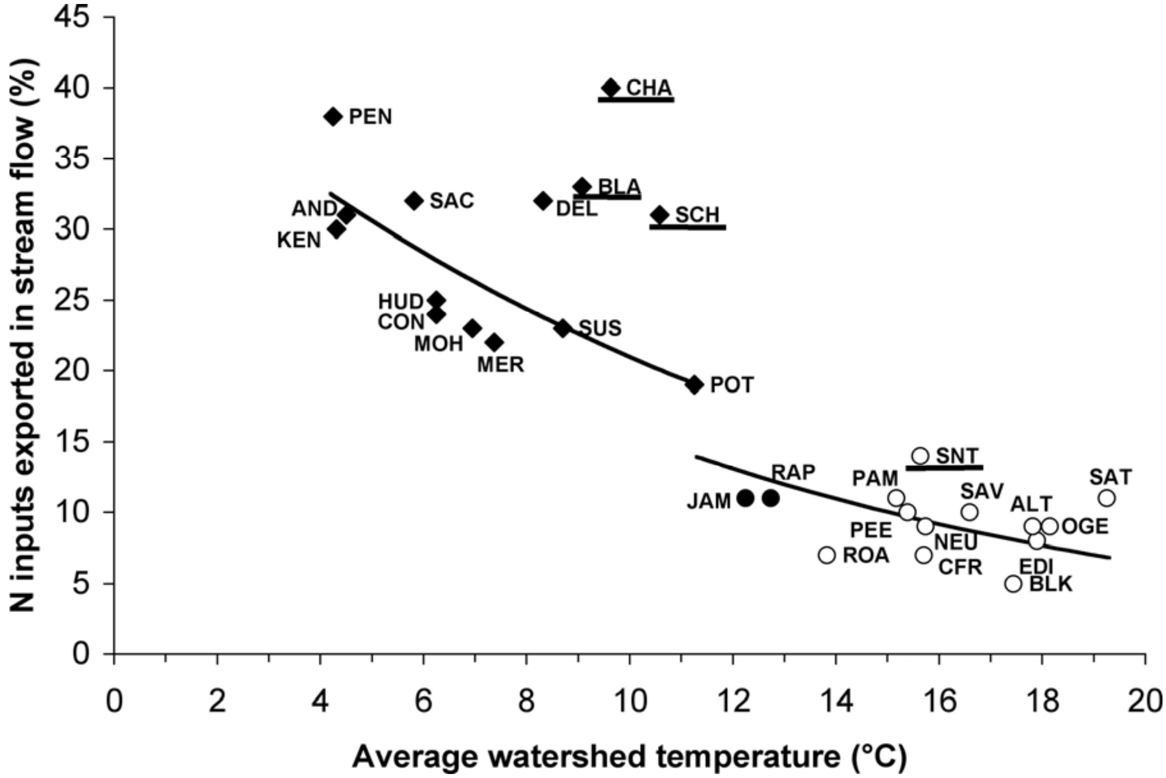

Figure 1.8b. Relationship between the percentage of N input exported in stream flow and average temperature for watersheds of the eastern US. Temperature was calculated from DAYMET gridded climate data (Thornton et al. 1997) using ESRI ArcGIS 9 software. Equations (which exclude underlined data points) are: % N input exported from northern and mid-Atlantic watersheds = 44.64 $e^{-0.08* \text{temperature}}$, R^2=0.51, p=0.01, and % N input exported from southeastern watersheds = 38.10 $e^{-0.09 * \text{temperature}}$, R^2=0.36, p=0.02. Note that the Potomac River watershed, which has an average temperature of 11.3°C, was included in both regressions as it falls on the accepted (Stanford et al. 1975) breakpoint. Symbols and watershed abbreviations as in Figure 1.4.

CHAPTER 2

TEMPORAL TRENDS IN NUTRIENT INPUTS TO THE WATERSHED OF THE ALTAMAHA RIVER, GEORGIA

Background

The input of excess nutrients can cause increased primary productivity in aquatic ecosystems—a process known as eutrophication (Nixon 1995)—potentially leading to other adverse effects, such as hypoxia, fish kills, and loss of submerged aquatic vegetation (Bricker et al. 1999, NRC 2000). The two nutrients most often of concern are nitrogen (N) and phosphorus (P). Both N and P can be limiting nutrients in estuaries, and thus excesses of either puts estuaries at risk for eutrophication (Howarth 1988, NRC 2000).

The source of excess nutrients to estuaries is usually human activity. Humans have substantially altered the global nitrogen cycle, especially since the advent of the industrial age. Inputs of reactive nitrogen into the global system have been increased through agricultural production (both biological nitrogen fixation and fertilizer production) and emissions from fossil fuel use. Between 1890 and 1990, total reactive nitrogen inputs into the global system due to anthropogenic activities increased from 15.6 Tg yr^{-1} to 139 Tg yr^{-1} (Galloway and Cowling 2002). By 2050, these inputs are expected to reach 270 Tg yr^{-1} (Galloway et al. 2004).

Human alterations to the global phosphorus cycle are a result of the intensification of erosion, point source inputs from major cities, and the application of fertilizers (both organic and inorganic). As a consequence of these activities, P mobilization has increased between 2.5 and 5-fold from natural background values on a global scale (Howarth et al. 1995, Smil 2000, Bennett et al. 2001). Since the global cycle of P is controlled primarily by geological processes, this mobilization cannot be reversed on human time scales (Schlesinger 1997, Smil 2000).

These increases in nutrient inputs over time have been documented in a number of watersheds and are reflected in increased loading to coastal systems. In Chesapeake Bay, Boynton et al. (1995) calculated that increases in watershed nutrient inputs have led to bay N loading 4.8-7.9 times and P loading 13-24 times greater than in the pre-colonial era. N loading to the Bay from the Susquehanna River increased linearly by a factor of 2.5 between 1945 and 1989 (Hagy et al. 2004, Kemp et al. 2005). This increase in N loading was positively correlated with the incidence of hypoxia in the mainstem of the Chesapeake Bay (Hagy et al. 2004). Bowen and Valiela (2001) found that a 67% increase in nitrogen input to the Waquoit Bay (Cape Cod, Massachusetts) watershed between 1938 and 1990 was reflected as a 250% increase in the N load to the estuary. In the watershed of the Neuse River in North Carolina, Stow et al. (2000) found that sources of both N and P, but especially N, have increased since the 1950s. However, this increase was not consistently reflected in in-stream nutrient concentrations. N inputs to the watershed of the Mississippi River have been estimated to have increased from 10.4 kg N ha^{-1} yr^{-1} to 17.3 kg N ha^{-1} yr^{-1} between the 1950s and 1990s, with much of the increase taking place in the 1960s (McIsaac et al. 2002). These increases were closely correlated with increases in riverine flux to the lower Mississippi River.

The watershed of the Altamaha River, which is formed by the confluence of the Oconee and Ocmulgee Rivers, is one of the largest on the east coast of the United States, encompassing an area of 36,718 km^2 and draining 24% of the state of Georgia (Figure 2.1). Quantitative information on changes in nutrient loading is limited, but there is evidence that downstream nutrient concentrations were significantly lower in the past than they are today. Dr. Lawrence Pomeroy (University of Georgia) measured ambient NO_2 + NO_3 (NO_X) and PO_4 concentrations in the Altamaha River estuary at a series of 7 stations sampled on 21 cruises between 1956 and 1959. At that time, NO_X concentrations averaged 2.86 ± 2.86 µM and PO_4 concentrations averaged 0.27 ± 0.22 µM. There were then no further systematic observations of nutrient concentrations in the estuary until the 1990s, when nutrient concentrations were sampled on 10 dates between 1990 and 1995 at stations located every 2-4 km along the length of the estuary as part of the Georgia Rivers Land Margins Ecosystems Research (LMER) project. Concentrations were found to average 11.55 ± 8.80 µM NO_X and 0.49 ± 0.17 µM PO_4. Nitrogen measurements made since 2001 as part of the Georgia Coastal Ecosystems Long Term Ecological Research (LTER) project were similar to those observed during the 1990s. There is some evidence that PO_4 concentrations increased. Phosphate concentrations measured on 11 cruises between May 2001 and December 2003 have averaged 0.70 ± 0.44 µM (S. B. Joye, unpublished data).

Information on nutrient loading to the Altamaha River comes from the USGS water quality monitoring program. Weston et al. (*in prep*) found evidence that NO_X

concentrations had significantly increased by approximately 0.2 μM yr^{-1} at the most downstream USGS station (Gardi, station #02226010) between the 1970s and the 1990s. Consistent data for the Gardi station is not available prior to that date, but a compilation of data collected at the nearby Doctortown station (#02226000) had average NO_X concentrations of 4.1 ± 3.9 mg N L^{-1} between 1956 and 1960 as compared to 20.1 ± 8.9 mg N L^{-1} between 1976 and 1980, when observations at Doctortown were discontinued. Taken together, these observations suggest that nitrogen input to the Altamaha increased dramatically between the 1950s and 1970s, and then more slowly thereafter. There were not enough observations of phosphorus concentration to determine whether there was a similar trend over the 1950-1990 time period.

There have been numerous changes in Georgia since the 1950s. The population of the state increased from 3.4 million to 6.5 million individuals between 1950 and 1990 (Forstall 1995). At the same time, agricultural practices were also changing: national shifts in farming practices since the 1950s caused agriculture to become more industrialized and farmland to become concentrated in fewer hands (Centner 2001, 2003). There is no information for the watershed of the Altamaha for the whole time frame, but between 1970 and 1990 the average population density increased from 35 persons km^{-2} to 70 persons km^{-2} and agricultural land declined by about one third (Weston et al. *in prep*). These trends most likely resulted in changes in nutrient loading to the Altamaha watershed, which were then reflected in downstream nutrient concentrations.

This study estimated N and P inputs to the watershed of the Altamaha River. Complete nutrient budgets were constructed for three target years: 1954, 1974, and 1992, to correspond with the Pomeroy observations (1956-1959), the LMER data (1994-1999), and an intermediate time point (in 1974) in order to determine the temporal changes in inputs of both nutrients. Budgets were also broken down by sub-watershed to evaluate the relative importance of various inputs on a spatial basis.

Methods

Watershed delineation

A GIS shapefile of the greater Altamaha River watershed and its major subwatersheds (the Upper and Lower Oconee River; the Upper, Lower, and Little Ocmulgee River; the Ohoopee River; and the Lower Altamaha River) was obtained from the Georgia GIS Clearinghouse (McFadden 2000). The proportion of each county located in the watershed was determined by overlaying a shapefile of Georgia counties (USBoC 2005) onto that of the watershed. Note that for this analysis, we used the area of the entire watershed, rather than just the gaged portion as in Chapter 1.

Nutrient Budgets

The methodology for determining total nitrogen inputs over time is similar to that set forth by Boyer et al. (2002) as part of the International SCOPE Nitrogen Project and has been described in greater detail elsewhere (see Chapter 1; see also Appendix B). N inputs include atmospheric deposition, fertilizer, net food and feed import, and nitrogen fixation. Phosphorus is not transported long-range through atmospheric deposition and there is no mechanism equivalent to biological nitrogen fixation for this element, so the only inputs of P that need be considered in a budget are fertilizer and net food and feed import.

Calculations for sub-watersheds were done in the same manner as those for the overall Altamaha budgets.

Fertilizer. Published estimates of county-by-county commercial nitrogen (as N) and phosphorus (as P_2O_5) fertilizer sales cover the period from 1945 to 1991 (Alexander and Smith 1990, Battaglin and Goolsby 1994). We weighted the annual amount of N and P fertilizer sold in each county by the proportion of the county inside the Altamaha watershed. (In the case of P, the weight of P_2O_5 fertilizer was adjusted to reflect only the element P.) We used 1954 and 1974 estimates for the nutrient budgets for these two years. Fertilizer sales were not available for the 1992 budget, so 1991 data were used instead. It should be noted that these estimates do not include residential fertilizer use, which may be substantial, especially in urban areas.

Net food and feed import. Net food and feed import is calculated by subtracting total population (animal + crop) from total consumption (human + animal) (see Chapter 1). Data on crops comprising 1% or more of total harvested cropland and animal populations were obtained from the USDA Census of Agriculture (USBoC 1956-1984; USDA-NASS 1987-1992). The Census is generally conducted at 5-year intervals, and data were analyzed for censuses covering the period 1954 to 1992. Data for nutrient budgets were taken from the 1954, 1974, and 1992 censuses. Crop data for all but pastureland and non-alfalfa hay were multiplied by N and P conversion factors obtained from the USDA PLANTS database (USDA/NRCS 2005). N content for pastureland and non-alfalfa hay is from Lander and Moffitt (1996, cited in Boyer et al. 2002) (Table 2.1). P conversion factors were not available for pastureland and non-alfalfa hay, but P content averaged 15% of N content for other crops, so we used this conversion to estimate P content for these two crops. Animal populations were multiplied by published per capita N and P consumption and excretion rates (Van Horn 1998) (Table 2.2). The difference between livestock consumption and excretion in manure was assumed to constitute animal production. Following Boyer et al. (2002), 10% was subtracted from crop (excluding hay, forage, and silage crops) and animal production to account for spoilage and inedible parts.

Data on human population were obtained from the U.S. Bureau of the Census for decennial censuses between 1950 and 1990 (Forstall 1995), and 1950, 1970, and 1990 data were used in nutrient budgets. Human consumption was estimated by multiplying the population by annual per capita consumption rates of 5 kg N yr^{-1} (Garrow et al. 2000) and 0.55 kg P yr^{-1} (Smil 2000).

Net atmospheric N deposition. Information on N deposition in the 1990s was obtained from the National Atmospheric Deposition Program (NADP) and Clean Air Status and Trends Network (CASTNET) (NADP 2005, USEPA 2005) for, respectively, wet and dry deposition, for stations in and around the Altamaha. N deposition values for each station were averaged for all years between 1987 and 1996 (the decade bracketing the target year of 1992) for which data were available. These stations were used to interpolate grids of regional atmospheric N deposition by kriging in a GIS (ordinary linear kriging, 200 m pixel size) (ESRI 2004). The grids were then clipped to the watershed extent and averages calculated. Organic N deposition was assumed to be 30% of total atmospheric deposition, and was calculated based on wet and dry deposition values (Neff et al. 2002). Half of this was assumed to constitute a new input to the watershed (Boyer et al. 2002).

Historic data on N deposition could not be obtained from the NADP and CASTNET programs, neither of which were established until the late 1970s. In order to hindcast, we used information on nationwide changes in NO_X emissions compiled by the U.S. Environmental Protection Agency (USEPA 2000). We assumed that the Georgia temporal trend was the same as the nationwide trend and that the proportion of nationwide atmospheric N emissions deposited in the Altamaha watershed remained constant. We calculated the ratio of 1987-1996 inorganic atmospheric deposition in the Altamaha watershed to 1987-1996 nationwide emission, and multiplied 1954 and 1974 emissions estimates by this ratio.

A proportion of the inorganic N in both fertilizer and animal manure is released to the atmosphere as a result of volatilization. 1991 fertilizer data (the last year for which data were available) (Battaglin and Goolsby 1994, see above) were broken down by type of N fertilizer (urea, ammonium nitrate, nitrogen solutions, anhydrous ammonia, and other combined fertilizers). Volatilization for 1992 was calculated based on losses from each type of fertilizer (Battye et al. 1994). Earlier data (Alexander and Smith 1990) did not include information on types of N fertilizer, so we assumed that the proportion of total fertilizer sales represented by each type of fertilizer was the same as in 1991. For animal manure, ammonia emission rates reported in kg N animal^{-1} yr^{-1} (Battye et al. 1994) were multiplied by the total number of animals in the watershed (see Net Food and Feed Import). Following Boyer et al. (2002), we assumed that 25% of both of these emissions were transported long-range, becoming an export from the watershed, and subtracted them from total atmospheric deposition to obtain net atmospheric N deposition.

Nitrogen fixation. Biological N fixation was calculated for the three target time points: 1954, 1974, and 1992. In each case, crop distributions were available from Censuses of Agriculture (USBoC 1956-1984, USDA-NASS 1992). The area of each nitrogen-fixing crop was weighted by the proportion of the county inside the Altamaha watershed and total crop acreages were multiplied by published N fixation rates (Table 2.3).

Biological nitrogen fixation in forestlands was also calculated for each time point. Non-symbiotic N fixation was assumed to be 40 kg N km^{-2} yr^{-1} following Boyer et al. (2002). Forestland area for the 1992 budget was calculated using the Forest Inventory and Analysis (FIA) database (USDA-FS 2005) (using the Georgia inventory done in 1989, which was the closest year available). Since the FIA program was not yet in place in 1974, we used 1974 land cover data (NARSAL 2003) to calculate forested area and non-symbiotic N fixation. Land cover data was not available for 1954, and thus we were unable to estimate forested land at this point in time. Therefore, we used 1974 data for this time point.

Following the methodology of Boyer et al. (2002), we also considered symbiotic N fixation in forestlands. Black locust was assumed to make up 10% of oak-hickory stands (Boyer et al. 2002). We used a symbiotic nitrogen fixation rate of 5,000 kg N km^{-2} yr^{-1} (Boring and Swank 1984) for this tree species. Oak-hickory stand area reported in the FIA database was used for the 1992 time point. We assumed that the proportion of forested area consisting of oak-hickory stands did not change between 1974 and 1992 and thereby calculated symbiotic N fixation by black locust in the 1970s. The 1974 figure was also substituted in the 1954 budget.

Alder, another symbiotically fixing tree species, was assumed to cover 10% of wetland area (Boyer et al. 2002). Wetland area was calculated in a GIS using land cover data for 1992 (USGS 1995) and 1974 (NARSAL 2003). We used a fixation rate of 4,000 kg N km^{-2} yr^{-1} (Hurd et al. 2001). Again, since no data was available for 1954, we used 1974 data as a placeholder.

Our use of 1974 forest N fixation data in the 1954 budget is clearly a potential source of error. However, forest N fixation was a small component of the overall inputs, and thus the budgets are not substantially affected by this substitution. In addition, biological N fixation inputs from the invasive leguminous vine kudzu (genus *Pueraria*) were not included in these budgets, and could represent a substantial additional N input. These budgets should be adjusted if N fixation rates for the species prevalent in the southeastern United States (*Pueraria montana var. lobata*) become available.

Non-food crop export. Cotton and tobacco are two crops grown in the Altamaha watershed that are consumed by neither humans nor animals. We assumed that virtually all of these crops are harvested for sale elsewhere and therefore subtracted their N and P production from the total inputs.

Results

Fertilizer. The amount of fertilizer N sold in the watershed increased steadily between 1954 (503 kg N km^{-2} yr^{-1}) and 1977 (1,618 kg N km^{-2} yr^{-1}), and then declined to 1,055 kg N km^{-2} yr^{-1} in 1991, the last year for which data were available (Figure 2.2). This was in contrast to the U.S. as a whole, where N fertilizer use in 1992 was higher than in 1974 (Howarth et al. 2002). Sales of fertilizer phosphorus followed a pattern similar to that of N, increasing from 274 kg P km^{-2} yr^{-1} in 1954 to 382 kg P km^{-2} yr^{-1} in 1974 before falling back to 188 kg P km^{-2} yr^{-1} in 1991 (Figure 2.2). It would be informative to have estimates subsequent to 1991 to determine whether the decreasing trend has continued, and it would also be helpful to have information on residential use since only commercial use is included in these estimates.

Net food and feed import. Net N and P import in food and feed has increased over time, primarily due to the increasing human population in the watershed (from approximately 808,000 in 1950 to over 1.8 million in 1990; Figure 2.3) and declining crop production. Import of N rose from 486 kg N km^{-2} yr^{-1} in 1954 to 563 kg N km^{-2} yr^{-1} in 1974 and 721 kg N km^{-2} yr^{-1} in 1992. Import of P rose from 138 kg P km^{-2} yr^{-1} in 1954 to 149 kg P km^{-2} yr^{-1} in 1974 and 154 kg P km^{-2} yr^{-1} in 1992. The decline in crop production was in large part due to declines in pastureland and other agricultural lands. Figure 2.4 summarizes the trends in the amount of agricultural land over the study period. The amount of harvested cropland declined from 6,528 km^2 in 1954 to 2,493 km^2 in 1992. Total pastureland (which includes cropland used for pasture) declined from 11,421 km^2 in 1954 to 2,570 km^2 in 2002, with most of the reduction occurring by 1964. Total agricultural land decreased from 19,414 km^2 to 5,721 km^2 over the study period, a decline of more than two-thirds.

Shifts in animal population were also a factor, albeit a smaller one, in the rise in food and feed imports. The most common types of livestock in the Altamaha River watershed are chickens, cattle, and hogs. The watershed also has smaller numbers of other livestock, such as horses, sheep, and turkeys. The number of chickens in the watershed has shown a dramatic increase, from approximately 20 million in 1954 to over 150 million in 1992

(Figure 2.5a). This increase was almost exclusively due to the increase in broiler chickens raised for meat; the standing stock of older layer chickens has increased by only a few million since 1954 and has actually declined from its high in 1969. The total number of cattle and calves remained relatively constant throughout the study period (averaging 395,437), but there was an increase in the number of beef cattle and a concurrent decline in the number of dairy cows, which require more nutrients than beef cows (Figure 2.5b). The number of hogs varied from 300,000 to 450,000 between 1954 and 1979, after which it declined sharply to less than 230,000 animals in 1992 (Figure 2.5c).

Net atmospheric N deposition. Net atmospheric nitrogen deposition rose most dramatically between 1954 and 1974, from 251 to 532 kg N km^{-2} yr^{-1}. The increase from 1974 to 1992, to 534 kg N km^{-2} yr^{-1}, was much smaller. Since these numbers are based primarily on nationwide emissions estimates, they reflect the overall trend in nitrogen emissions in the United States over that period of time.

Biological N fixation. Biological nitrogen fixation by crops increased from 644 to 707 kg N km^{-2} yr^{-1} between 1954 and 1974 and then dropped to 552 kg N km^{-2} yr^{-1} by 1992. This pattern was the result of dramatic rises and subsequent declines in both soybean and non-alfalfa hay production. Most other nitrogen-fixing crops declined steadily over the study period, particularly pastureland (see Figure 2.4).

Forestland N fixation declined overall due to a decline in forested wetland area in the Altamaha watershed, from 4,572 km^2 in 1974 to 3,100 km^2 in 1992. This resulted in a substantial reduction in the amount of N fixed by alders. Non-symbiotic forest N fixation and fixation by black locust increased slightly due to a very small increase in total forest area between 1974 and 1992, but not enough to offset the decline in alder N fixation. N fixation in forestland was 154 kg N km^{-2} in 1974 and 136 kg N km^{-2} in 1992. Total biological nitrogen fixation accounted for 797 kg N km^{-2} yr^{-1} in 1954, 861 kg N km^{-2} yr^{-1} in 1974, and 688 kg N km^{-2} yr^{-1} in 1992.

Non-food crop export. N and P production in cotton and tobacco was quite small. N export fell from 31 kg N km^{-2} yr^{-1} in 1954 to 22 kg N km^{-2} yr^{-1} in 1974 and 21 kg N km^{-2} yr^{-1} in 1992. P export was 4 kg P km^{-2} yr^{-1} in 1954, 1 kg P km^{-2} yr^{-1} in 1974, and 3 kg P km^{-2} yr^{-1} in 1992.

Overall trends. Nitrogen inputs to the Altamaha watershed showed an overall increase between 1954 and 1992, from 2,007 kg N km^{-2} yr^{-1} to 2,977 kg N km^{-2} yr^{-1}. Inputs of N were actually highest, at 3,553 kg N km^{-2} yr^{-1}, in 1974 and then declined by 1992 (Table 2.4a). Total phosphorus inputs also showed an increase between 1954 and 1974, rising from 408 kg P km^{-1} yr^{-1} to 531 kg P km^{-2} yr^{-1}. P inputs then declined to pre-1954 levels by 1992, falling to 340 kg P km^{-2} yr^{-1} (Table 2.4b).

Overall, fertilizer tended to be the most important input of N to the watershed. It was also the primary driver of change in the overall N budget, since changes in other inputs were relatively modest compared to changes in fertilizer inputs. Fertilizer was followed in importance by biological N fixation in agricultural lands and net food and feed import. Fertilizer contributed 25% of total new N inputs in 1954 and 46% of inputs in 1974 before falling to 35% of total inputs in 1992. The importance of biological N fixation in agricultural lands declined over the course of the study period, falling from 40 to 23% of the total N inputs. Net food and feed import accounted for 24% of new N inputs in 1954 and 16% in 1974, then rose again to 24% of total new N inputs in 1992. The proportion accounted for by net atmospheric deposition rose slightly over the study period from 13%

to 18% of total N inputs. N fixation in forestlands was a minor component of the overall budget, averaging only about 6% of the total N inputs.

Fertilizer input was also the dominant source of new phosphorus to the watershed throughout the study period, contributing an average of 65% of new inputs. However, its relative importance declined from 67% in 1954 and 72% in 1974 to only 55% in 1992. Again, changes in the budget were driven primarily by changes in P fertilizer. Net food and feed import, while not having increased much in absolute numbers, is now more important as a percentage of total P inputs than in the past, contributing 45% in 1992.

Spatial distribution. When nutrient budgets were broken down by sub-watershed, most basins followed the overall trend of increased inputs in 1974 as compared to 1954 and then either held constant or declined by 1992 (Figure 2.6; Table 2.5). In the case of N, the decline was not back to 1954 levels, whereas 1992 P inputs were below 1954 levels in several cases (Lower Oconee, Little Ocmulgee, and Ohoopee). Input changes for both N and P were driven primarily by fertilizer, the dominant input to most sub-basins throughout the study period, although changes in net food and feed import also played a role in many of the basins. In the N budgets, net atmospheric deposition and crop and forestland N fixation were lesser contributors. Crop N fixation showed a maximum in 1974 in several sub-watersheds. Forestland N fixation generally declined over the course of the study period; net atmospheric deposition generally increased.

We examined the spatial patterns of inputs of both fertilizer and net food and feed import, as these two inputs accounted for most of the variation both among sub-watersheds and over time. Fertilizer inputs followed the same patterns for both N and P, and tended to be highest in the middle portions of the watershed (Figure 2.7). The Lower

Ocmulgee consistently had the greatest inputs of fertilizer, followed by the Little Ocmulgee and Ohoopee sub-basins. Import of net food and feed followed a pattern opposite from that of fertilizer and was lowest in the middle region of the watershed, in the Little and Lower Ocmulgee and the Ohoopee sub-basins. These sub-basins led in crop production, with the Little and Lower Ocmulgee sub-basins showing a net export of food and feed in 1992. Net food and feed imports were highest in all years in the upper reaches of watershed, in the Upper Oconee and Upper Ocmulgee sub-watersheds (Figure 2.8).

The Upper Oconee and Upper Ocmulgee sub-watersheds were unique in showing continuous increases in both N and P inputs over the entire study period. Although this was due to large increases in net foot and feed import over time for both systems, the main drivers of food and feed imports were different in each case. The Upper Ocmulgee, site of a substantial portion of the major metropolitan area of Atlanta and its extensive suburbs, saw a dramatic increase in human consumption over time. This sub-watershed had the greatest increase in population of any sub-watershed in both percentage and absolute terms (Figure 2.3) and accounts for over 50% of the total population of the greater Altamaha watershed. This growth resulted in increased human nutrient requirements leading to greater import of food. In contrast, the net food and feed import in the Upper Oconee was dominated by increased animal consumption. Nearly all the increase in chicken population in the greater Altamaha watershed was localized within the Upper Oconee sub-basin, and it was also the only basin with an overall increase in cattle and hog populations. Coupled with steadily declining crop production in both sub-watersheds, the increased demand for food and feed in these two sub-watersheds drove a rise in net import.

There was also variation in the other sources of N inputs to the sub-watersheds, although not as great as that observed with fertilizer and net food and feed import. Net atmospheric N deposition was similar in all sub-watersheds except the Upper Oconee, where high animal populations resulted in large losses due to manure volatilization. As a result, net atmospheric N deposition in this sub-basin was consistently lower than in others and, though it increased in 1974, returned to near-1954 levels in 1992 due to the continuous increases in animal populations. N inputs from cropland fixation did not vary much among sub-watersheds, but the Lower Ocmulgee sub-basin led in new N inputs from this source, especially in 1974. N fixation by forestlands was similar and declined in 1992 relative to 1974 in all sub-watersheds except the Lower Altamaha. In this sub-basin, an increase in forested wetland area between 1974 and 1992 caused an increase in the amount of N fixed by alders, resulting in a substantial overall increase in forestland N fixation.

When taken together, total nutrient inputs were fairly evenly distributed among sub-basins in 1954 with slightly lower inputs to the Lower Oconee and Lower Altamaha sub-watersheds and slightly higher inputs to the Upper Oconee and Lower Ocmulgee (Table 2.5; Figure 2.6). Over time, these differences among sub-watersheds increased. By 1992, the Upper Oconee sub-watershed had N inputs twice as high and P inputs more than three times as high as those of the Lower Oconee. These differences were primarily due to the changes in human population and agricultural practices described above.

Discussion

In 1992, estimated nutrient inputs into the watershed of the Altamaha River were 2,977 kg N km^{-2} yr^{-1} and 340 kg P km^{-2} yr^{-1}. These numbers represent an increase of 970 kg N km^{-2} yr^{-1} as compared to 1954, but a decline of 68 kg P km^{-2} yr^{-1}. In both cases, nutrient inputs peaked at the mid-point estimate (1974). There are no older data with which to compare these estimates, but the results obtained for 1992 were highly comparable to nitrogen inputs presented by Castro et al. (2003) in their analysis of a series of watersheds on the Atlantic and Gulf coasts of the United States. Their study (which did not include forestland N fixation) calculated inputs of 2,830 kg N km^{-2} yr^{-1} to the Altamaha watershed, as compared to 2,977 kg N km^{-2} yr^{-1} in this study. Nutrient inputs calculated by Asbury and Oaksford (1997) were far higher than those presented here, at 5,470 kg N km^{-2} yr^{-1} and 1,380 kg P km^{-2} yr^{-1}. This is most likely a result of differences in methodology, including the use of animal excretion rates far higher than those used in the current study.

The majority of the nutrients entering the Altamaha River watershed over the study period have been agriculturally derived, consisting primarily of inputs due to fertilizer and biological N fixation in agricultural lands. This is consistent with previous reports (Asbury and Oaksford 1997; Castro et al. 2003). However, the relative importance of the different sources of nitrogen has shifted over time. Biological N fixation by crops was the most important source of new N to the Altamaha watershed in the 1950s, but was replaced by increasing inputs of fertilizer N (which peaked in 1974). The overall increase in fertilizer use coincided with a decline in agricultural land—both crop and pastureland—in the watershed (Figure 2.5). It is possible that some of the commercially sold fertilizer was applied to golf courses and the like, which would not be included in the estimate of cultivated land, but it is also likely that there has been an increase in the amount of fertilizer used per unit area. This may be due to changes in growing techniques. Crop yields in the watershed have risen substantially since 1954 (in several cases, with most of that increase taking place between 1954 and 1974), reflecting a global pattern of dramatic increases in crop yields per unit area due to increased fertilizer application, especially during the 1960s (Tilman et al. 2002). There have also been shifts in the crops that are cultivated: the percentage of harvested cropland producing soybeans increased from 3 to 21% and the percentage producing peanuts increased from 6 to 12%over the course of the study period.

Net food and feed import, which reflects changing populations of livestock and humans as well as crop production, was another important and growing source of nutrients to the watershed. There is less and less cropland producing food and feed for the growing populations of both animals and people in the Altamaha, resulting in the need to import those nutrients from outside the watershed. The large increase in chicken population (from 20 million to over 150 million between 1954 and 1992), taken together with decreased pastureland, also reflects a national trend towards concentrating livestock in animal feeding operations (Centner 2001, 2003).

The finding that nutrient inputs to the Altamaha watershed were lower in 1992 than 1974 indicates that increases in watershed nutrient loading to the Altamaha took place in the early portion of the study period, between 1954 and 1974. These results are driven largely by the change in fertilizer input in the watershed, which decreased by about one third between 1974 and 1992. This finding fits the long-term pattern of nitrogen input to the Mississippi River watershed, wherein total N inputs leveled off after approximately 1974 (McIsaac et al. 2001). In contrast, N inputs to the Waquoit Bay (Bowen and Valiela 2001) and Neuse River (Stow et al. 2001) watersheds both continued to increase due to rises in wastewater disposal and animal sources, respectively. N loading to the Chesapeake Bay has also continued to increase since the 1950s due to increases in both fertilizer use and import of animal feed (Kemp et al. 2005). There is less information on P, but inputs to the Neuse River watershed remained approximately constant between 1974 and 1992 (Stow et al. 2001). However, Baker and Richards (2002) found a decrease in P inputs to two watersheds in Ohio between 1975 and 1995 due to declines in phosphorus fertilizer inputs.

The amount of nutrients that reach an estuary depend on total watershed input (e.g. Howarth et al. 1996, Boyer et al. 2002, see also Chapter 1). In this study, the nutrient budgets suggest that N loading to the watershed increased by 77% between 1954 and 1974 and then declined again to approximately a 50% overall increase by 1992. Watershed P loading also showed a mid-point peak but did not show an overall increase. Over this time period, nutrient concentrations of both NO_X and PO_4 in the Altamaha River estuary increased. As described in the introduction, there was a four-fold increase in NO_X concentrations between the 1950s (Pomeroy data) and the 1990s (LMER data). There is no information on estuarine nutrient concentrations for the 1970s, but there was evidence for a large (5-fold) increase in USGS water quality measurements between the late 1950s and the late 1970s and then a leveling off, which is consistent with our observation that increases in N loading to the watershed took place primarily during the early part of the study period. There is no evidence for an increase in watershed P loading between 1954 and 1992, but there was an approximate doubling of the PO_4 concentration in the estuary. Phosphate is well buffered in marsh sediments (Pomeroy et al. 1965), and thus concentrations in the estuary may not necessarily reflect loading to the watershed.

Whether or not further increases in nutrients take place in the estuary of the Altamaha River will depend on future trends in agriculture and human populations, since these have been shown to be the primary contributors to nutrient delivery to estuaries (Caraco 1995, Jordan and Weller 1996, Carpenter et al. 1998). Fertilizer inputs to the watershed declined between 1974 and 1992, and if fertilizer application becomes more efficient in the future, inputs could continue to decrease. However, populations of both humans and animals are likely to continue to increase, and the production of food and feed to sustain those populations may also have to intensify. Thus, further increases in nutrient delivery to the estuary can be expected.

Table 2.1. N and P crop conversion factors used in this study. Sources: USDA PLANTS database (USDA/NRCS 2005) and Lander and Moffitt (1996) (pastureland and non-alfalfa hay).

Crop	lb N/unit	lb P/unit
Cotton	0.0304/lb of seed and lint	0.0038/lb of seed and lint
Corn (grain)	0.7929/bushel	0.1514/bushel
Corn (silage)	7.7501/ton	2.2644/ton
Cropland used only for pasture	2000/acre	300/acre
Non-crop pastureland	1000/acre	150/acre
Hay, alfalfa	55.7759/ton	5.226/ton
Hay, non-alfalfa	21.7/ton	3.255/ton
Oats (grain)	0.5984/bushel	0.1092/bushel
Peanuts	0.0448/lb of seed	0.0037/lb of seed
Sorghums (grain)	0.98/bushel	0.0006/bushel
Soybeans (beans)	0.0144/bushel	0.0006/bushel
Soybeans (hay)	45.9418/ton	4.3312/ton
Rye	1.0557/bushel	0.1873/bushel
Peaches	0.0012/lb of fruit	0.0001/lb of fruit
Pecans	0.238/lb of nut	0/lb of nut
Peanuts (hay)	0.0448/lb of seed	0.0037/lb of seed
Peanuts (seed)	31.743/ton	3.3931/ton
Tobacco	1.7064/hundredweight	0.218/hundredweight
Wheat (grain)	1.4366/bushel	0.2309/bushel

Table 2.2a. Animal N consumption and excretion rates used in this study. Source: Van Horn (1998).

Animal	**N consumption rates (kg animal^{-1} yr^{-1})**	**N excretion rates (kg animal^{-1} yr^{-1})**
Cattle, beef	66.75	58.51
Cattle, dairy	156.00	121.00
Cattle, young	Same as adult	Same as adult
Chickens (broilers)	0.13	0.07
Chickens (layers)	0.84	0.55
Pigs & hogs	8.51	5.84
Turkeys	0.62	0.62
Horses	44.80	40.00
Sheep	5.97	5.00
Goats	5.97	5.00

Table 2.2b. Animal P consumption and excretion rates used in this study. Source: Van Horn (1998). All values in kg P animal^{-1} year^{-1}.

Animal	**P consumption rates (kg animal^{-1} yr^{-1})**	**P excretion rates (kg animal^{-1} yr^{-1})**
Cattle, beef	13.91	10.43
Cattle, dairy	39.73	29.48
Cattle, young	Same as adult	Same as adult
Chickens (broilers)	0.02	0.01
Chickens (layers)	0.21	0.17
Pigs & hogs	1.77	0.94
Turkeys	0.15	0.09

Table 2.3. Rates of biological nitrogen fixation used in this study.

Crop	**kg N km^{-2} yr^{-1}**	**Reference(s)**
Hay, alfalfa	22,400	Heichel et al. 1984
Hay, non-alfalfa	11,700	Lander & Moffitt 1996, cited in Boyer et al. 2002
Pastureland, all types	1,500	Jordan & Weller 1996
Peanuts	8,000	Smil 1999
Soybeans	9,600	Average of published values, after Boyer et al. 2002

Table 2.4a. Nitrogen inputs to the Altamaha watershed in 1954, 1974, and 1992. All numbers in kg N km^{-2} yr^{-1}.

Source	**1954**	**1974**	**1992**
Net atmospheric deposition	251	532	534
Fertilizer	503	1,618	1,055
Net food and feed import	486	563	721
N fixation in agricultural lands	644	707	552
N fixation in forest lands	154	154	136
Non-food crop export	31	22	21
Total N input	*2,007*	*3,553*	*2,977*

Table 2.4b. Phosphorus inputs to the Altamaha watershed in 1954, 1974, and 1992. All numbers in kg P km^{-2} yr^{-1}.

Source	**1954**	**1974**	**1992**
Fertilizer	274	382	188
Net food and feed import	138	150	154
Non-food crop export	4	1	3
Total P input	*408*	*532*	*340*

Table 2.5a. Total nitrogen inputs to Altamaha sub-watersheds in 1954, 1974, and 1992. All numbers in kg N km^{-2} yr^{-1}.

Sub-basin	**1954**	**1974**	**1992**
Upper Oconee	2,115	3,896	5,240
Upper Ocmulgee	2,095	3,036	4,597
Lower Oconee	1,699	3,050	2,221
Lower Ocmulgee	2,263	4,495	3,553
Little Ocmulgee	2,132	3,851	2,512
Ohoopee	2,033	3,832	3,105
Lower Altamaha	1,520	2,633	2,287

Table 2.5b. Nitrogen inputs to Altamaha sub-watersheds in 1954, 1974, and 1992. Data are broken down by source. All numbers in kg N km^{-2} yr^{-1}.

	Net atm. N dep.	**Fertilizer**	**Net food and feed import**	**Biol. N fix. in agric. lands**	**Biol. N fix. in forest lands**
Upper Oconee					
1954	220	674	488	453	133
1974	395	915	1,669	798	133
1992	278	686	2,312	626	121
Upper Ocmulgee					
1954	257	259	778	678	146
1974	587	833	916	568	146
1992	599	836	1,197	368	135
Lower Oconee					
1954	270	510	201	553	193
1974	586	1,640	35	611	193
1992	648	800	32	436	159
Lower Ocmulgee					
1954	244	881	249	757	165
1974	539	2,834	1	1,017	165
1992	583	2,006	-319	774	145
Little Ocmulgee					
1954	243	757	327	695	142
1974	545	2,434	22	748	142
1992	620	1,238	-156	532	105
Ohoopee					
1954	263	744	325	626	124
1974	560	2,391	60	755	124
1992	580	1,257	223	756	101
Lower Altamaha					
1954	278	463	354	366	81
1974	561	1,489	126	393	81
1992	561	852	291	434	166

Table 2.5c. Total phosphorus inputs to Altamaha sub-watersheds in 1954, 1974, and 1992. All numbers in kg P km^{-2} yr^{-1}.

Sub-basin	1954	1974	1992
Upper Oconee	608	984	1,198
Lower Oconee	466	507	363
Upper Ocmulgee	536	538	627
Lower Ocmulgee	692	857	800
Little Ocmulgee	649	751	508
Ohoopee	624	735	619
Lower Altamaha	596	502	468

Table 2.5d. Phosphorus inputs to Altamaha sub-watersheds in 1954, 1974, and 1992. Data are broken down by source. All numbers in kg P km^{-2} yr^{-1}.

	Fertilizer	Net food and feed import
Upper Oconee		
1954	155	457
1974	216	770
1992	245	954
Upper Ocmulgee		
1954	141	397
1974	197	343
1992	298	328
Lower Oconee		
1954	278	192
1974	388	121
1992	285	80
Lower Ocmulgee		
1954	480	216
1974	670	195
1992	715	96
Little Ocmulgee		
1954	413	241
1974	575	180
1992	441	71
Ohoopee		
1954	405	224
1974	565	177
1992	448	173
Lower Altamaha		
1954	252	171
1974	352	152
1992	304	166

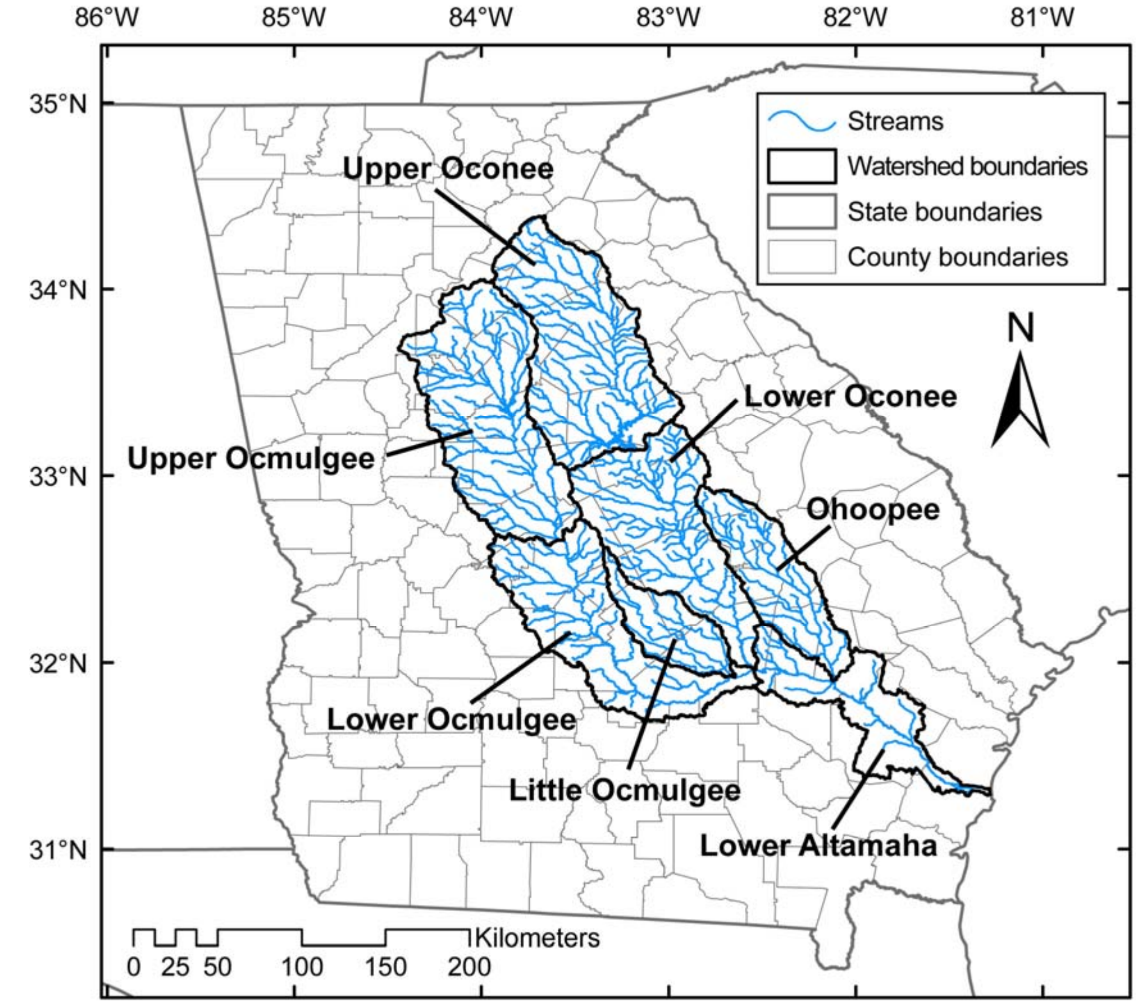

Figure 2.1. Location of the greater Altamaha River watershed and its major sub-basins.

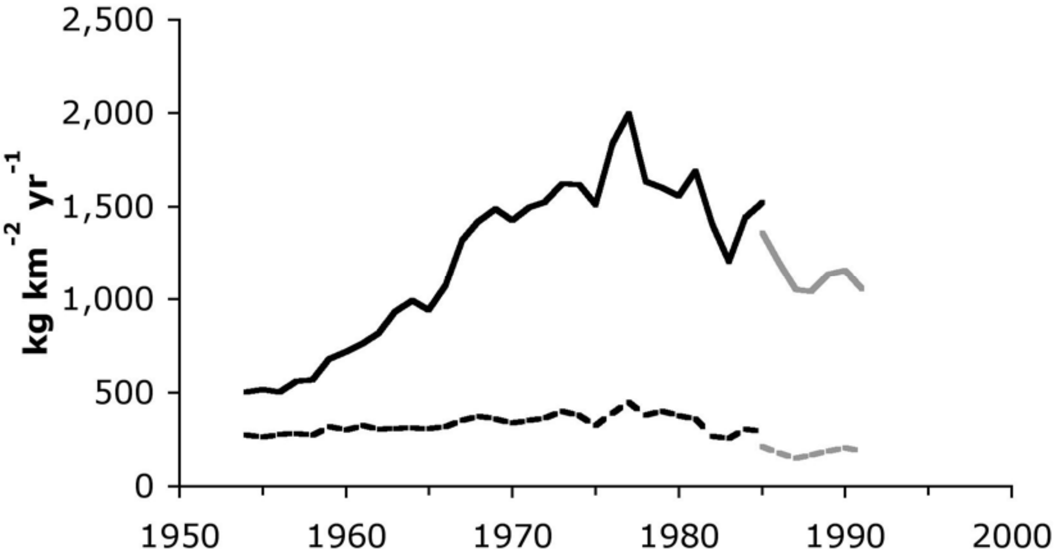

Figure 2.2. Commercial nitrogen and phosphorus fertilizer use in the Altamaha watershed from 1945 to 1991. Solid lines represent N; dashed lines represent P. 1945-1985 data (black lines) are calculated using information from Alexander & Smith (1990); 1985-1991 data (gray lines) use information from Battaglin & Goolsby (1994).

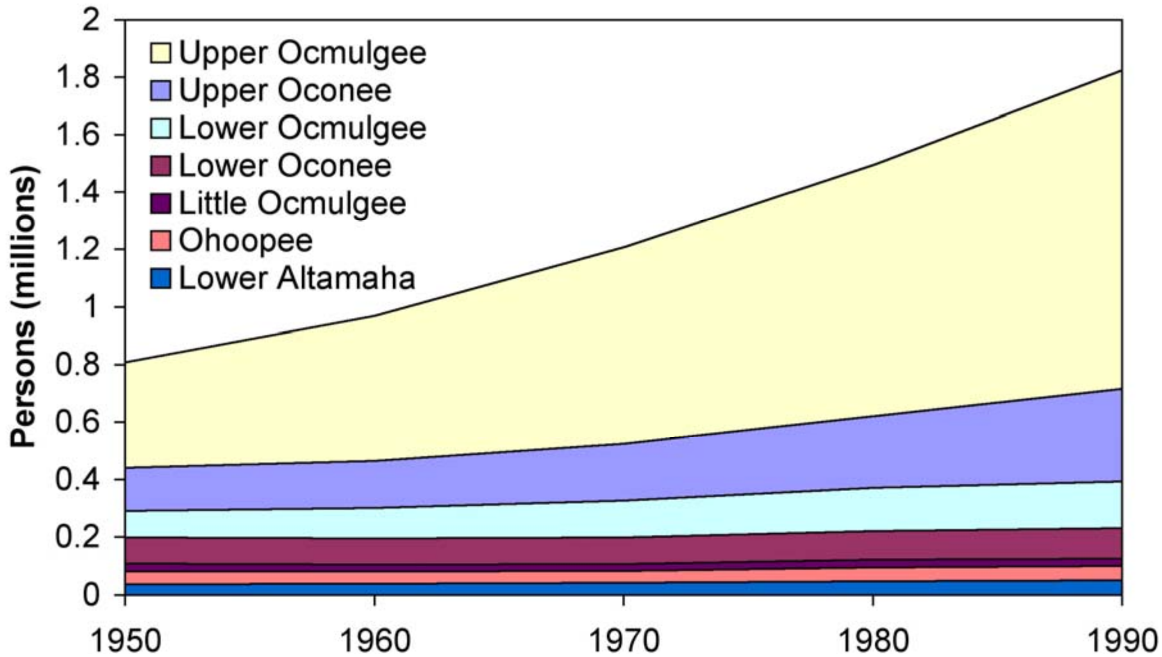

Figure 2.3. Human population in the Altamaha watershed, 1950-1990. Data are broken down by sub-basin. Source: United States decennial censuses (Forstall 1995).

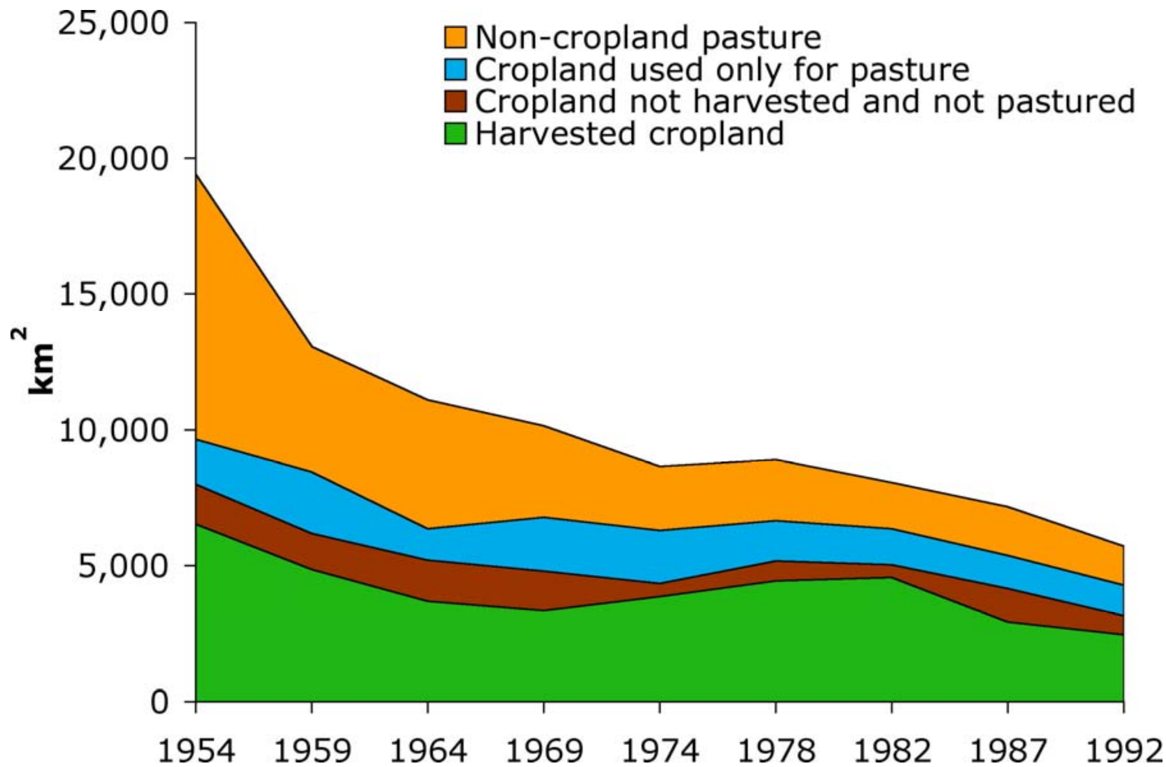

Figure 2.4. Area of land in agricultural production in the Altamaha watershed, 1954-1992. Data are broken down by type. Source: U.S. Bureau of the Census and USDA Agricultural Censuses, 1954-1992.

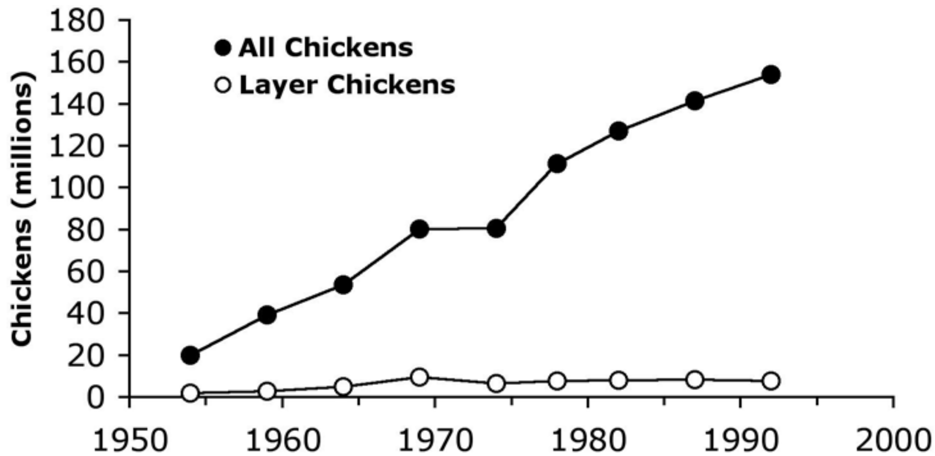

Figure 2.5a. Number of chickens in the Altamaha watershed, 1954-1992. Source: U.S. Bureau of the Census and USDA Agricultural Censuses, 1954-1992.

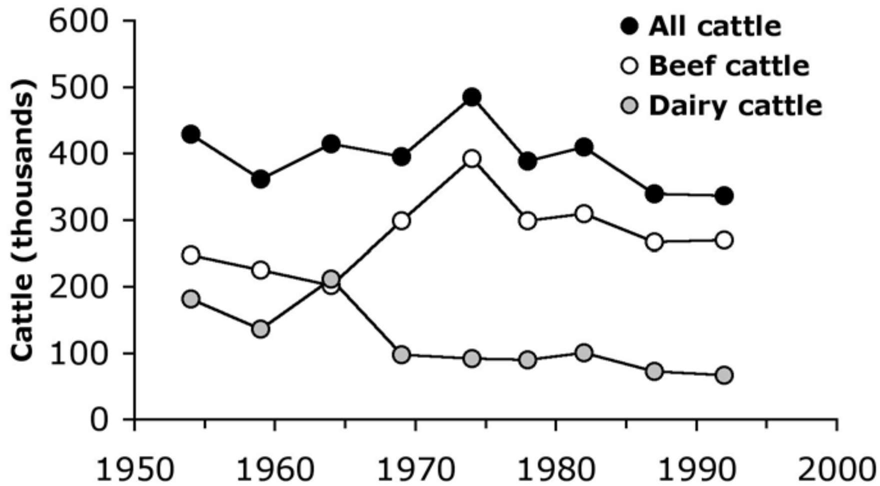

Figure 2.5b. Number of cattle in the Altamaha watershed, 1954-1992. Source: U.S. Bureau of the Census and USDA Agricultural Censuses, 1954-1992.

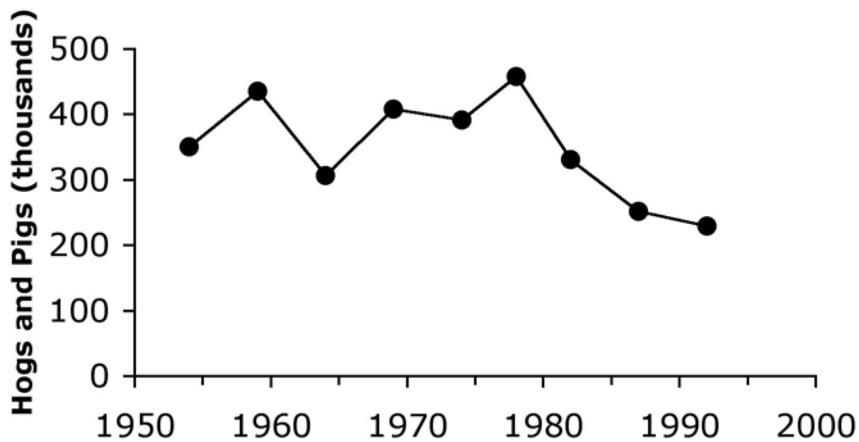

Figure 2.5c. Number of hogs and pigs in the Altamaha watershed, 1954-1992. Source: U.S. Bureau of the Census and USDA Agricultural Censuses, 1954-1992.

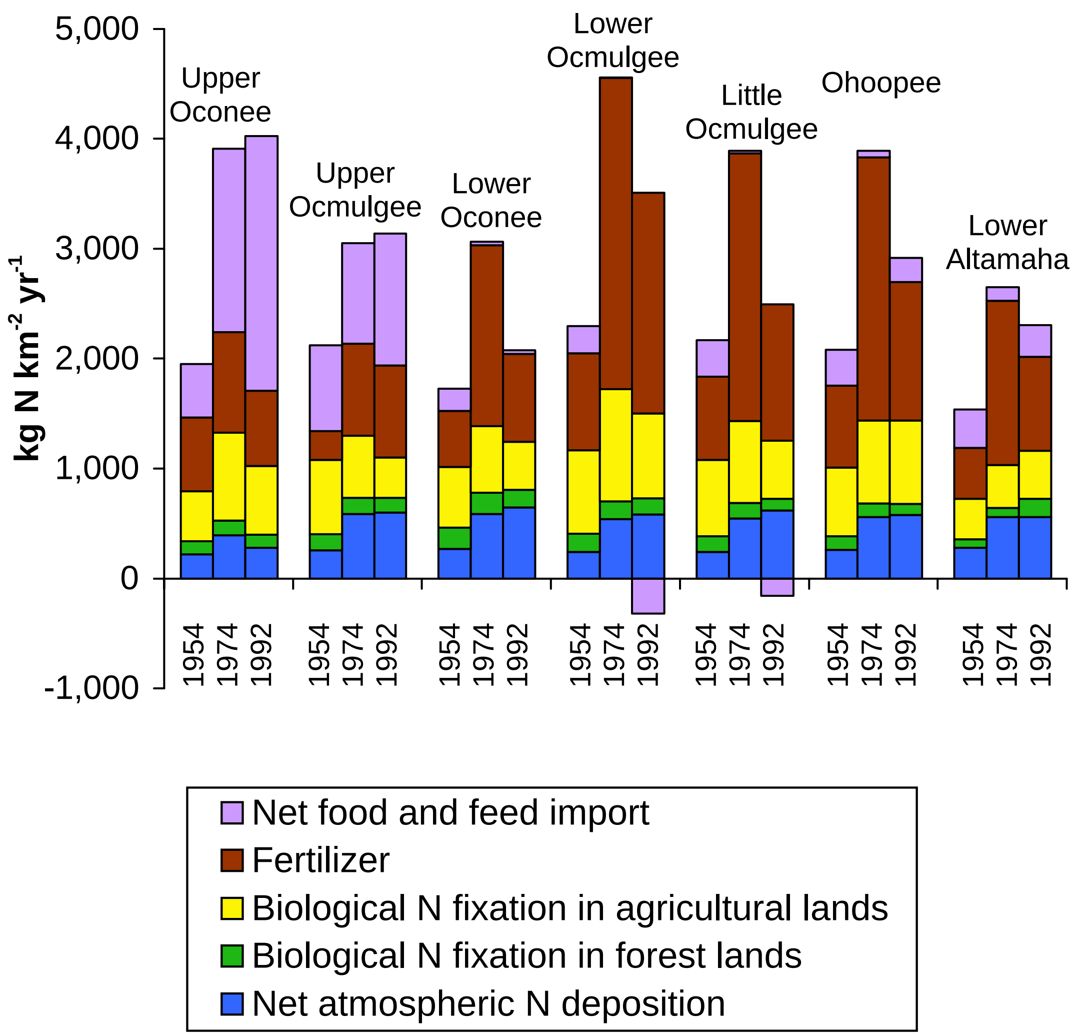

Figure 2.6a. Nitrogen inputs into sub-basins of the Altamaha watershed: 1954, 1974, and 1992.

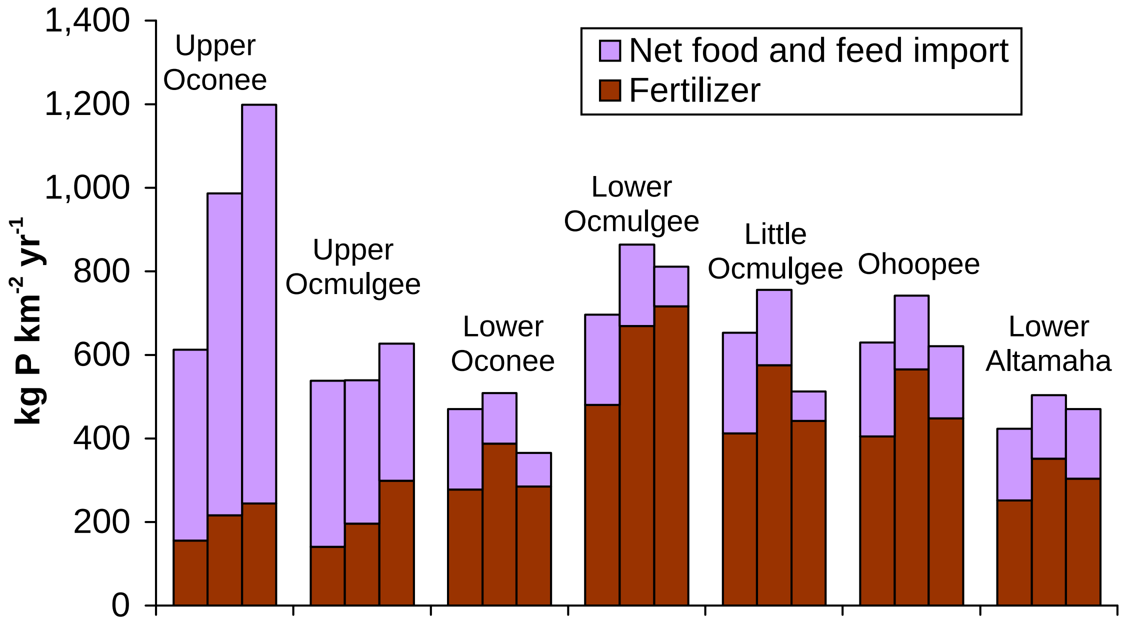

Figure 2.6b. Phosphorus inputs into sub-basins of the Altamaha watershed: 1954, 1974, and 1992.

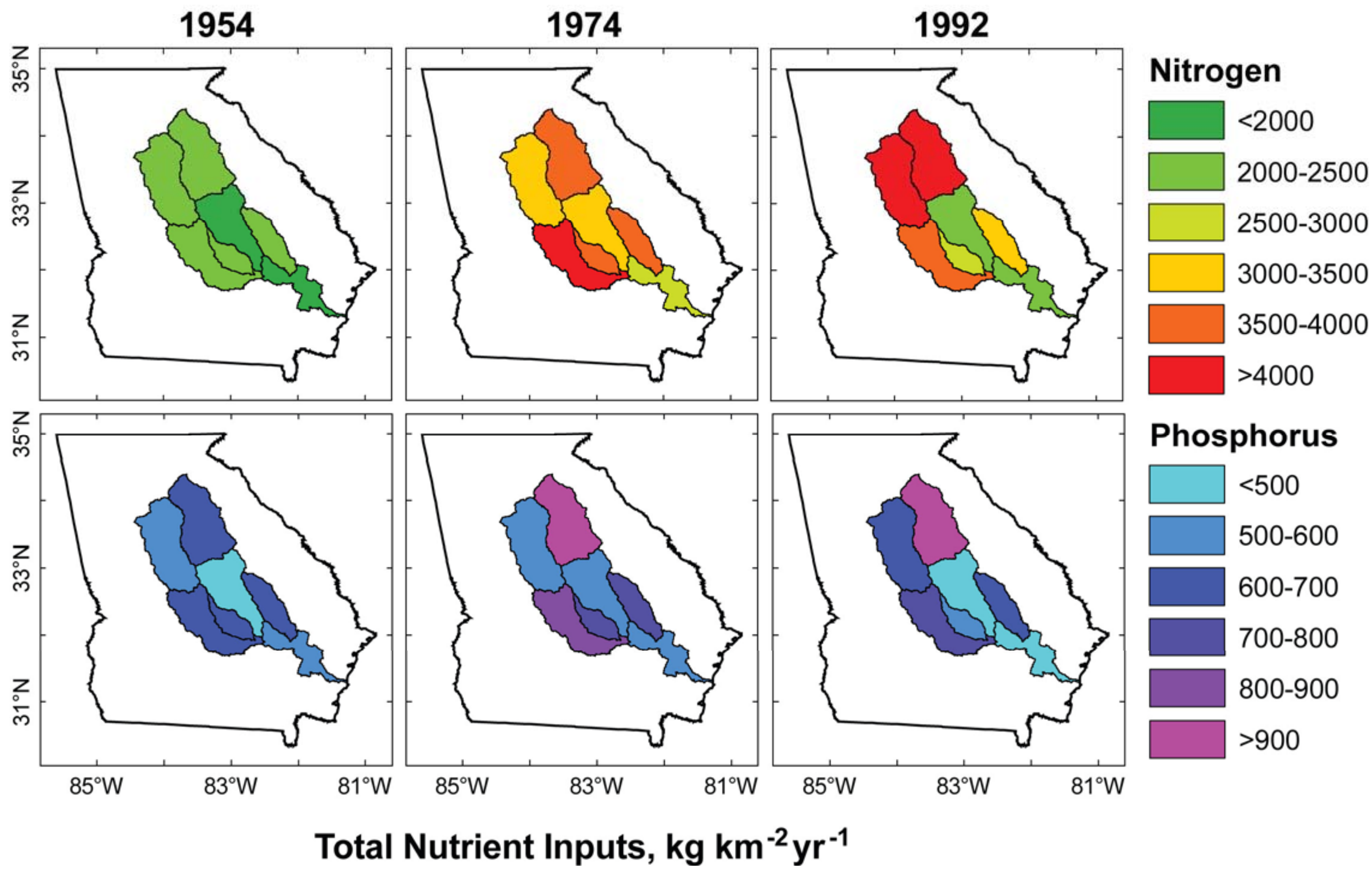

Figure 2.6c. Spatial distribution of total N and P inputs to sub-basins of the Altamaha watershed in 1954, 1974, and 1992.

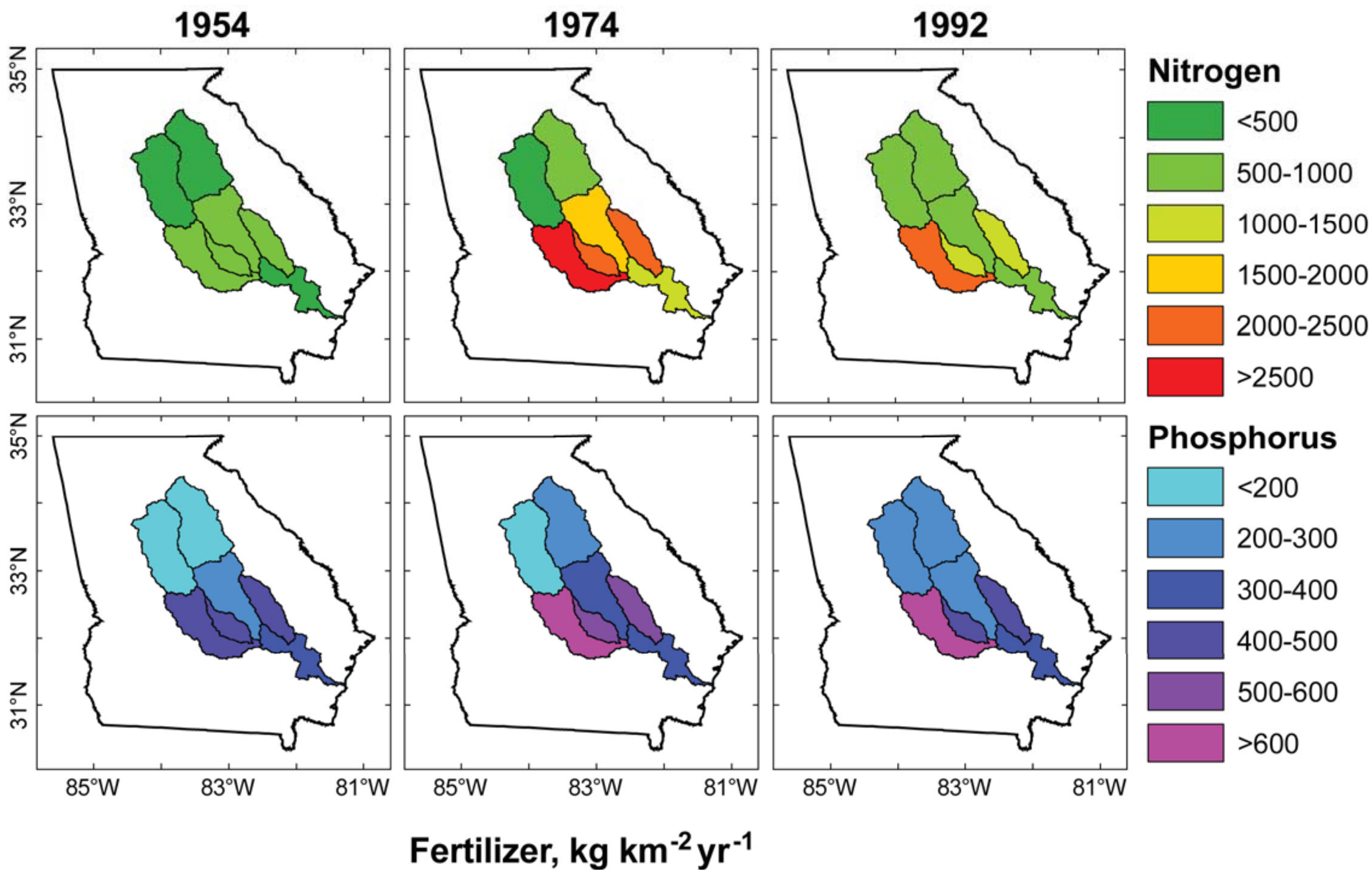

Figure 2.7. Spatial distribution of N and P fertilizer inputs to sub-basins of the Altamaha watershed in 1954, 1974, and 1992.

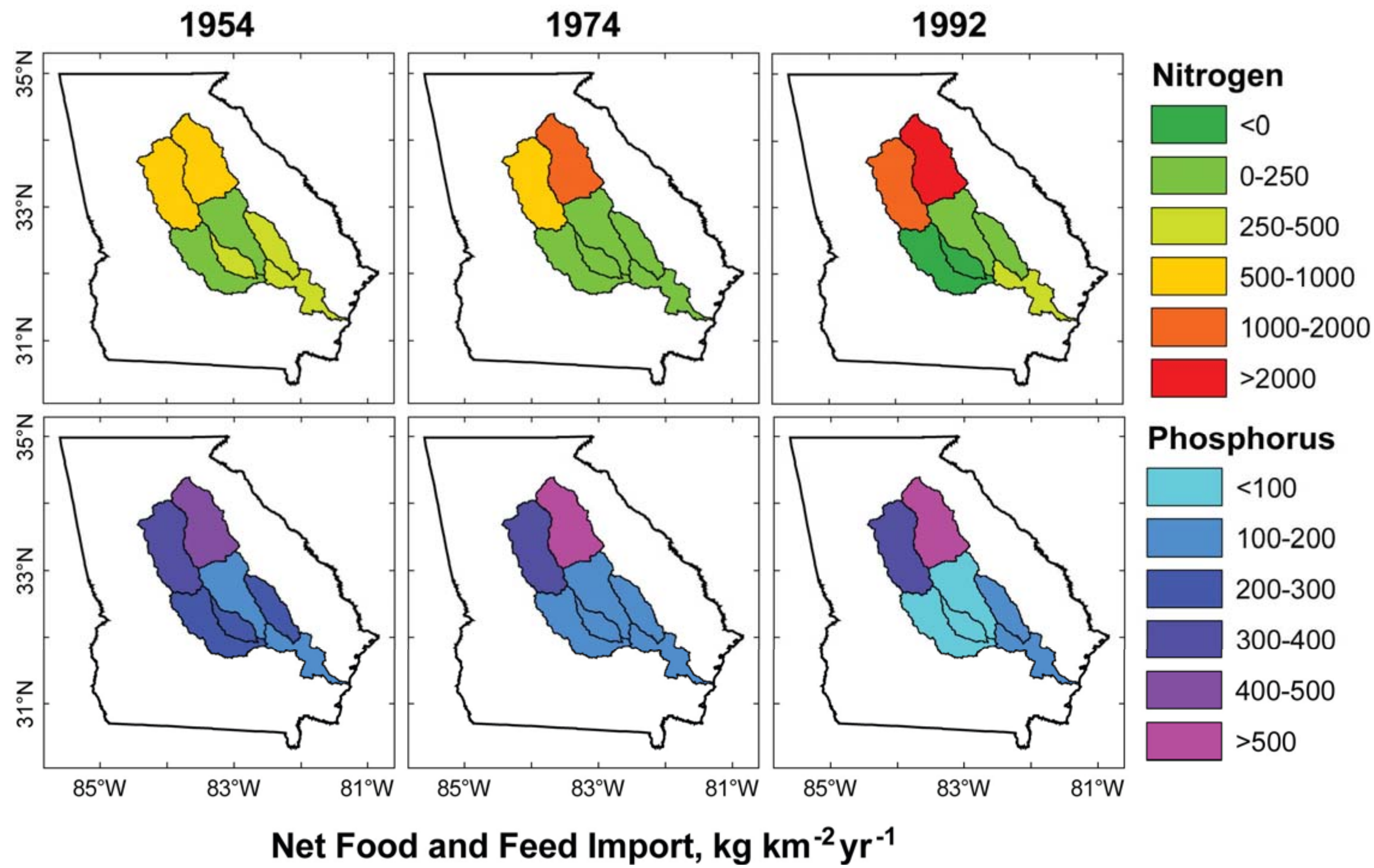

Figure 2.8. Spatial distribution of inputs of N and P in net food and feed import to sub-basins of the Altamaha watershed in 1954, 1974, and 1992.

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APPENDIX A

NITROGEN INPUTS TO THE ST. MARY'S RIVER WATERSHED, AND A PRELIMINARY ESTIMATE OF ESTUARINE EXPORT

Calculations for the St. Mary's watershed followed the same approach as for other watersheds, described in Chapter 1. General watershed characteristics and N inputs are summarized here. The St. Mary's was not included in Chapter 1, however, because information on riverine export was incomplete.

General watershed characteristics	
USGS water quality gage #	02231220
Latitude	30.78
Watershed area (km^2)	2,062.56
Mean watershed slope (degrees)	0.46
Average annual temperature (°C)	20.1
Average annual number of frost days	14.1
Average annual precipitation (cm)	133.7
Population density (persons km^{-2})	32
Land Use	
% Forest	44.1
% Agriculture	1.0
% Urban	2.1
% Wetland	39.1
% Water	2.8
% Other	11.0

N inputs. All values in kg N km^{-2} yr^{-1}.	
Net atmospheric N deposition	580.63
Total atmospheric deposition	648.26
Inorganic N deposition	533.86
Organic N deposition	228.80
Fertilizer volatilization export	0.33

Manure volatilization export	67.29
Fertilizer use	59.24
Net food and feed import	510.28
Crop production	87.51
Animal production	180.74
Animal consumption	617.07
Human consumption	161.47
Biological N fixation in croplands	71.79
Biological N fixation in forestlands	252.34
Non-symbiotic N fixation	35.12
Symbiotic N fixation by black locust	8.40
Symbiotic N fixation by alder	208.89
Non-food crop export	0.87
Total N inputs	1,473.41

Flow data were not available for the USGS water quality gage at Boulogne, FL. A preliminary estimate of N export was calculated using water quality data from USGS gage #02231220 (St. Mary's River at Boulogne, FL). Streamflow was interpolated from data from gages #02231000 (St. Mary's River near MacClenny, FL) and #02231253 (St. Mary's River near Gross, FL) using a proportion based on drainage area of each of these stations:

$$\text{Boulogne stream flow} =$$

$$\text{MacClenny stream flow} + \frac{\text{Gross stream flow} - \text{MacClenny stream flow}}{\text{Gross drainage area} - \text{MacClenny drainage area}}$$

$$\times (\text{Boulogne drainage area} - \text{MacClenny drainage area})$$

where
MacClenny drainage area = 700 miles^2
Boulogne drainage area = 1,180 miles^2
Gross drainage area = 1,360 miles^2

Measurements of organic N were not available.

N export in stream flow. All numbers in kg N km^{-2} yr^{-1}.

NH_4	67.19
NO_2 + NO_3 (unfiltered)	64.05

When these data are plotted on the graph of watershed input vs. N export (see Figure 1.3), the point for the St. Mary's falls close to our regression, with a 9% export. However, organic N is not included in this estimate, and would most likely increase the percentage of N exported substantially.

APPENDIX B

DETAILED GUIDE FOR N BUDGET CALCULATIONS

This guide details N budgets as they were calculated in this study. Budget calculations are based on Boyer et al. (2002). Places where this methodology deviates from Boyer et al. (2002) are indicated in the text.

Net food and feed import

Livestock consumption and production

Cattle:
Obtain from agricultural census:
- Livestock and poultry: cattle and calves inventory (number)
- Beef cows (number)
- Milk cows (number)

Young cattle = inventory – beef cows – milk cows
Calculate proportion of beef and milk cattle:
    Proportion beef cattle = beef cows / (beef cows + milk cows)
    Proportion milk cattle = milk cows / (beef cows + milk cows)
Calculate number of young beef and young milk cattle:
    Young beef cattle = young cattle * proportion beef cattle
    Young milk cattle = young cattle * proportion milk cattle

Hogs and Pigs:
Obtain from agricultural census:
    Hogs and pigs inventory (number)

Sheep:
Obtain from agricultural census:
    Sheep and lambs inventory (number)

Chickens:
Broiler chickens: Broilers and other meat-type chickens sold (number)
Layer chickens: Layers and pullets 13 weeks old and older inventory (see text) (number)

Horses:
Obtain from agricultural census:
Horses and ponies, inventory (number)
Mules, burros, and donkeys – inventory (number)
Sum.

Turkeys:
Obtain from agricultural census:
Any poultry, turkeys (number)
Any poultry, turkey hens kept for breeding (number)
Sum.

Multiply populations in each county by proportion of county in the watershed, sum, and multiply by consumption/excretion factors.

Animal production = consumption – excretion. Subtract 10% for spoilage and inedible parts.

Crop production

Obtain from agricultural census:
- Total cropland, harvested cropland (acres)
- Acreages and quantities harvested for all crops found in counties entirely or partially in the watershed.

Multiply acreage in each county by proportion of county inside the watershed.

For crops whose acreage constitutes 1% or more of total harvested cropland in the watershed, multiply quantities harvested by proportion of county inside the watershed. *(The inclusion of only crops constituting 1% or more of harvested cropland may be a deviation from the Boyer et al. methodology.)*

To calculate hay production, obtain:
a. Hay-alfal, oth tame, small grain, wild, grass silage, green chop, etc (see txt) (tons, dry)
b. Alfalfa hay (tons, dry), harvested (quantity)
Non-alfalfa hay production (tons) = a – b.

To calculate pastureland production, obtain:
c. Pastureland, all types (acres)
d. Total cropland, cropland used only for pasture or grazing (acres)
Non-crop pastureland (acres) = c – d.

If necessary, convert quantities harvested to appropriate units and multiply by conversion factors. Subtract 10% from all crops other than hay and silage.

Human consumption

Obtain county census figures from U.S. Bureau of the Census. Multiply by proportion of county inside the watershed, then by N consumption rate.

Divide all by watershed area to obtain per-square-km figures.

Net food and feed import = (animal consumption + human consumption) – (animal production + crop production)

Fertilizer

From Battaglin & Goolsby (1994) report, obtain the following fields for all counties entirely or partially within the watershed:

- NTOT91 (or year of choice) (total N in fertilizers)
- AMNI91 (ammonium nitrate)
- ANHY91 (anhydrous ammonia)
- NMIS91 (other combined fertilizers)
- NSOL91 (nitrogen solutions)
- UREA91 (urea)

Using the early 1990s National Land Characterization Data Set, calculate for each county, and for the portion of the county located inside the watershed, the area in classes 81-83 (Pasture/Hay, Row Crops, and Small Grains) and 85 (Urban/Recreational Grasses). Total the land area in these classes. Use these areas to calculate the proportion of agricultural land inside the watershed for each county and weight fertilizer sales estimates accordingly.

For fertilizer inputs, use value from NTOT91 field. (The other fields are used to calculate fertilizer volatilization—see Net Atmospheric N Deposition.)

Net atmospheric N deposition

Wet deposition

From National Atmospheric Deposition Program (NADP), obtain annual Total Wet Deposition (kg/ha) data and latitude/longitude for each site. Use “Inorganic N” column.

Plot values for desired years or average (using only data with acceptable completeness values) in GIS and krige surface (ordinary linear, 200m pixel size is acceptable). Clip to watershed and calculate average.

(This study used 1987-1996 averages. Boyer et al. used 1991 values.)

Dry deposition

From Clean Air Status and Trends Network (CASTNET) download “drychem” data file. Use fields:

- hno3_flux
- no3_flux
- nh4_flux

and calculate averages for years with acceptable completeness values.

kg/ha weights are for whole compound, convert to N only using molecular weights.

Obtain latitude/longitude for each site and plot values. Krige surface (ordinary linear, 200m pixel size). Clip to watershed and calculate average.

(Boyer et al. used a published model to calculate dry deposition based on wet deposition data. However, that model is valid only for the northeastern United States and is thus inappropriate to use in other geographical regions.)

Organic N deposition

Using previously calculated inorganic wet and dry deposition weights, calculate organic N deposition (30% of *total* N deposition):

Organic N deposition = (inorganic N deposition/7) * 3

Half of this is considered a new input, so

New organic N deposition = ((inorganic N deposition/7) * 3)/2

Fertilizer volatilization export

Multiply each type of N fertilizer species in the watershed (see Fertilizer Inputs) by fraction volatilized:

- AMNI: 0.02
- ANHY: 0.001
- NMIS: 0.02
- NSOL: 0.025
- UREA: 0.15

Sum; 25% of this is considered an export. Subtract from atmospheric deposition.

Manure volatilization export

Multiply number of animals in the watershed (see Net Food and Feed Import) by the ammonia emission rate per animal for each type of animal. Convert to N by multiplying by 0.776 (the percentage of N by weight in NH_4). 25% of this is considered an export; subtract from atmospheric deposition.

<u>Biological N fixation</u>

Croplands

Obtain from agricultural census:
- Pastureland, all types (acres)
- Acreages of any other legume crops found in counties entirely or partially in the watershed.

Hay and pastureland fixation are calculated as for crop production:
a. Hay-alfalfa, other tame, small grain, wild, grass silage, green chop, etc(see txt)(acres)
b. Alfalfa hay (tons, dry), harvested (acres)
Non-alfalfa hay area = a – b
c. Pastureland, all types (acres)
d. Total cropland, cropland used only for pasture or grazing (acres)
Non-crop pastureland (acres) = c – d.

Multiply acreages in each county by proportion of county in watershed, then multiply by published fixation rates.

Divide by watershed area.

Forestlands

Multiply forest acreages in each county by proportion of county in watershed.

Use total forest land area for calculation of non-symbiotic forest N fixation.

Use 10% of oak-hickory stand area for calculation of symbiotic N fixation by black locust.

For alder fixation, use 10% of the area of classes 91 and 92 in early 1990s National Land Characterization Dataset.

Divide by watershed area.

<u>Non-food crop export</u>

Crops such as cotton and tobacco which are not consumed by humans or animals should be considered a non-food crop export. Calculate N contents as in crop production calculations and subtract from the total N inputs.

(Non-food crop export is not considered in Boyer et al. since cotton and tobacco are not crops commonly grown in the northeastern United States.)

APPENDIX C

DATA SETS USED IN THIS STUDY

Animal populations
Data Source: U.S. Department of Agriculture Census of Agriculture
Data Time Period: 1840-present; conducted approximately every 5 years since 1925
Data Scale: Nationwide; county-level
URL: http://www.nass.usda.gov/Census_of_Agriculture/index.asp

Atmospheric deposition (dry)
Data Source: Clean Air Status and Trends Network (CASTNET)
Data Time Period: 1987-present (varies by station)
Data Scale: Nationwide monitoring stations
URL: http://www.epa.gov/castnet/

Atmospheric deposition (wet)
Data Source: National Atmospheric Deposition Program (NADP)
Data Time Period: 1978-present (varies by station)
Data Scale: Nationwide monitoring stations
URL: http://nadp.sws.uiuc.edu/

Crop production/N fixation
Data Source: U.S. Department of Agriculture Census of Agriculture
Data Time Period: 1840-present; conducted approximately every 5 years since 1925
Data Scale: Nationwide; county-level
URL: http://www.nass.usda.gov/Census_of_Agriculture/index.asp

Elevation
Data Source: U.S. Geological Survey National Elevation Datset (USGS 1999a)
Data Time Period: Unknown
Data Scale: Nationwide; 1 arc-second (approximately 30-meter) grid
URL: http://ned.usgs.gov/

Fertilizer use
Data Source: Alexander and Smith 1990
Data Time Period: 1945-1985; annual
Data Scale: Nationwide; county-level estimates
URL: http://water.usgs.gov/pubs/ofr/ofr90130/report.html

Fertilizer use
Data Source: Battaglin and Goolsby 1994
Data Time Period: 1985-1991; annual
Data Scale: Nationwide, county-level estimates
URL: http://water.usgs.gov/pubs/wri/wri944176/

Forest inventory
Data Source: U.S. Forest Service Forest Inventory and Analysis National Program
Data Time Period: Late 1980s/early 1990s (varies by state)
Data Scale: Nationwide, county-level
URL: http://www.fia.fs.fed.us/

Frost days
Data Source: DAYMET U.S. Data Center (Thornton et al. 1997)
Data Time Period: 1980-1997; annual
Data Scale: Nationwide; 1-km grid
URL: http://www.daymet.org/

Human populations
Data Source: U.S. Bureau of the Census
Data Time Period: 1790-present; decennial
Data Scale: Nationwide, county-level
URL: http://www.census.gov/

Land cover (1992)
Data Source: U.S. Geological Survey National Land Cover Dataset
Data Time Period: 1992
Data Scale: Nationwide; 30-meter grid
URL: http://landcover.usgs.gov/natllandcover.php

Land cover (1974)
Data Source: University of Georgia Natural Resources Spatial Analysis Laboratory (NARSAL 2003)
Data Time Period: 1974
Data Scale: Georgia; 60-meter grid
URL: http://narsal.ecology.uga.edu/glut.html

Precipitation
Data Source: DAYMET U.S. Data Center (Thornton et al. 1997)
Data Time Period: 1980-1997; annual
Data Scale: Nationwide; 1-km grid
URL: http://www.daymet.org/

Stream flow
Data Source: U.S. Geological Survey National Water Information System
Data Time Period: 1899-present (varies by station)
Data Scale: Nationwide monitoring stations
URL: http://waterdata.usgs.gov/nwis

Temperature
Data Source: DAYMET U.S. Data Center (Thornton et al. 1997)
Data Time Period: 1980-1997; annual
Data Scale: Nationwide; 1-km grid
URL: http://www.daymet.org/

Water quality
Data Source: U.S. Geological Survey National Water Information System
Data Time Period: 1899-present (varies by station)
Data Scale: Nationwide monitoring stations
URL: http://waterdata.usgs.gov/nwis

Watershed boundaries
Data Source: U.S. Geological Survey (Steeves and Nebert 1994)
Data Time Period: Unknown
Data Scale: Nationwide; 1:250,000-scale
URL: http://water.usgs.gov/GIS/huc.html

Watershed boundaries (Georgia)
Data Source: McFadden 2000
Data Time Period: Unknown
Data Scale: Georgia; 1:24,000-scale
URL: https://gis1.state.ga.us/