WATER QUALITY TRENDS IN THE SOUTHEASTERN PLAINS AND PIEDMONT ECOREGION AND THE APPLICATION OF POWER ANALYSIS

by

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(Under the Direction of Vernon Meentemeyer)

ABSTRACT

This study contributes to the growing body of literature on water quality monitoring and offers important observations on issues of scale and hypothesis testing. In order to understand the interaction between land use/land cover (LU/LC) and water quality trends and to implement landscape level management options, analysis at large spatial scales is required. The study recognizes that this type of broad scale analyses is confounded by biases and factors ranging from small scale to large scale. Such biases include potential lack of representativeness of specific sites within monitoring networks to biases and inherent assumptions of the ecoregional framework that was employed. The two components of this study are: 1) determination of trends in water quality at multiple spatial scales and assessment of the association between LU/LC changes and trends; and 2) demonstration of power analysis to determine adequacy of monitoring period and/or effect size (change in constituent values over time) for trend detection. It examines trends of six water quality constituents in two adjacent ecoregions at multiple over two time periods.

Trends were not observed for most constituents in many areas except pH. This constituent showed increased values throughout the region, and this trend appeared to be affected by LU/LC changes. The power of a statistical test is rarely reported when water quality trends are cited. Using the software TRENDS, the requisite minimum monitoring duration, effect size for trend detection and stations that met a priori power were determined.

Water quality assessment is confounded by the characteristic of the information, and current and future water quality monitoring efforts should focus more on addressing this issue by designing programs and expanding analysis of the database to meet research needs while satisfying compliance requirements. In order to meet criteria for rigorous statistical tests of spatial and temporal patterns, monitoring design should include longer, uninterrupted recording periods and more strategically sampled sites that best represent the landscape. Reporting power results should be included to strengthen observations and analyses. Analyses that fail to detect trends should be viewed with caution since the data may not be robust enough to support the null hypothesis.

INDEX WORDS: trend analysis; water quality; water quality monitoring; Southeastern Plains ecoregion; Piedmont ecoregion; seasonal Kendall test; effect size; minimum monitoring period; TRENDS

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DEDICATION

To my family for their endurance, support and belief in me during this project and to the many unrecognized efforts to improve knowledge of the interdependence between our physical world and human well-being.

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CHAPTER 1:

INTRODUCTION

Water quality issues are a focus of research because they are important to many segments of our society. Passage of the Water Quality Act of 1965, the 1972 Clean Water Act (CWA)`, and the Safe Drinking Water Act of 1986 and subsequent amendments provide the main legislative framework to address water quality issues in the United States. Recognizing the need to improve water quality, public and private sectors have invested large amounts of capital and made fundamental changes in managing water resources. Public and private investments for sewage and water treatment facilities exceeded $541 billion dollars during the 1972-1996 period alone (Wagsness 1997) to ensure that water quality met health standards and industry needs. Despite investments in pollution abatement technology and programs, monitoring and research efforts, and changes in conservation policy, many streams and rivers currently fail to meet target-use criteria. Of the 23% of total river and stream miles assessed in 1998, 35% were identified as impaired and an additional 10% were considered threatened. The remaining 55% fully supported all designated uses (USEPA 2000a,b). These values have not changed significantly since the biennial Reports to Congress were initiated in 1975. The consistent failure of many surface freshwater systems to meet target objectives of the CWA suggests that much more needs to be done to improve water quality to meet human, ecosystem and societal needs. For example, upgrading water and wastewater systems to comply with U.S. Environmental Protection Agency (USEPA) environmental standards are estimated to cost between $275 billion and $1 trillion between 2002 and 2017 (Allison 2002). Additional efforts and resources need to be directed toward research, policy and management, and education (Bjorkland and Pringle 2001).

Fueled by legislative mandate and research needs, large scale, coordinated and national-level water quality monitoring programs began in the 1960's and 1970's. The National Water Quality Surveillance Systems (NWQSS) of the USEPA, and the Hydrologic Benchmark Network (HBN) and the National Stream Quality Accounting Network (NASQAN) of the U.S. Geological Survey (USGS) provide more than 35 years of data and information on water quality and streamflow. This information is important for research and provides a database for policy decisions and program management. During the period of maximum station operation of the HBN and NASQAN programs (1979-1992) physical, chemical and biological constituents at more than 600 sites throughout the 50 states and Puerto Rico were recorded on a regular basis. The data were used to characterize water quality and to identify spatial and temporal trends. A later monitoring effort, the National Water Quality Assessment Program (NAWQA) was initiated in 1991 to assess the effectiveness of the water pollution abatement programs. Additionally, many local, state and regional governments, public interest groups and businesses operate monitoring programs that provide useful information for specific stream or river characterization needs.

Analysis of data from the early years of the programs did not provide incontrovertible evidence of spatial and temporal trends of water quality in the U.S. Using data from more than 400 sampling sites over the period 1974-1981, Smith et al. (1987) noted that policies and management practices were responsible for the changes they noted. For example, the significant improvements (decreased concentrations) in fecal coliform bacteria and lead in most regions of the U.S. were considered to be a direct result of the improvements in waste treatment facilities and the phasing out of lead from gasoline. The increased concentrations (worsening conditions) of sodium, nitrogen and suspended sediments in water samples reflected increased regional use of road salts and fertilizer applications and expanded regional agricultural activity. A later analysis of NASQAN data (Lettenmaier et al. 1991) suggests that the variable trends in water quality (mostly upward for common ions, total nitrogen, suspended sediments and mostly downward for total phosphorous and trace metals) could not be tightly linked to land use and population changes. Additionally, these early studies did not account for natural variability across the landscape and over short periods of time. Windom et al. (1991) first questioned the accuracy of the downward trends reported for many trace metal concentrations. Subsequent USGS documents (see Alexander et al. 1996) caution that data on trace metal concentrations collected during the early years of monitoring may be biased because of inconsistencies in sampling, analysis and management procedure,s and the “true” values were estimated to be half the reported values. Thus, the early analysis of the temporal trends of trace metals was likely to be seriously flawed, erring on the side of higher concentrations. Recognizing this bias forced a re-evaluation of the accuracy of NASQAN data.

Basic questions about the effectiveness of water pollution abatement investments and policies and the spatial and temporal characteristics of water quality in the U.S. could not be answered satisfactorily when Congress sought the answers in the mid-1980’s (Leahy et al. 1993). This inability to provide a definitive analysis of water quality and trends, despite the large amount of monitoring data, was attributable, in part to flaws in monitoring design, laboratory procedures and data analysis. Most of the data for surface water quality were collected as part of compliance monitoring programs which focused on chemical concentrations and flows discharged from point sources, located upstream and downstream from those sources and water-supply intakes (Alexander et al.1996; Powell 1995). While these data are adequate to satisfy regulatory needs, particularly within a given watershed, they do not provide information required for basin-wide assessments of water quality at larger scales, such as regional, ecosystem or national levels. As used throughout this document, large scale refers to "large extent of geographic coverage, or more commonly, a coarse level of geographic detail" (Goodchild and Proctor 1997, p. 10). Cartographically, this perspective is comparable to a broad scale or small scale map where detail is lost as the geographic area increases.

Establishing the criteria used to delineate areas of interest and selecting the appropriate spatial and temporal scales of assessment are critical challenges in monitoring programs. While much ecological research continues to focus on processes and responses to stressors at small scale points of interest such as stream reach, issues of water quality and stream health are often better addressed at larger scales, such as regions, water basins and landscape. The apparent usefulness of watersheds as study units arises from the understanding that water quality and quantity characteristics at a point on a stream reflect the aggregate characteristics of the watershed above that point (Omernik and Bailey 1997). There is growing momentum to use even broader spatial scales, such as ecosystem, ecoregion, continental and global. This holistic approach is particularly important for water resource management, and requires spatial and temporal frameworks that delineate units of study appropriate for issues of concern. This broad scale approach to assessment is necessary to understand the interaction of multiple stressors on water quality and overall river health. At these large scales, spatial patterns of natural and anthropogenic interrelationships involving ecosystems and their components can be considered. Schramm and Hubert (1996) note that ecosystem management requires consideration of events beyond the watershed level. However, this type of broad scale analysis is confounded by biases and factors ranging from small scale to large scale. Such biases include potential lack of representativeness of specific site within the monitoring networks to biases and inherent assumptions of the ecological framework that was employed.

Traditional units or areas for analysis of surface water resources are watersheds and other hydrologic units. These units delineate the landscape and remain the basis in mapping and area characterization for many research and management applications despite recent suggestions that other spatial frameworks may be more appropriate. Many researchers and water resource managers (e.g., Montgomery et al. 1995; Cannon 1994; Lotspeich 1980) endorse the watershed approach noting that it provides an appropriate spatial framework for total environmental and economic planning. The USGS continues to use a hydrologic unit model in much of its work including the current updating of hydrography maps of the United States. While watersheds, basins and hydrologic units remain the standard for much of the large scale research, monitoring and management, this framework has three serious flaws that jeopardize accuracy of the data and may introduce bias: 1) heterogeneity of ecological characteristics and environmental conditions within a delineated study unit; 2) inability to distinguish differences in surface water and groundwater integrating processes among watershed types (hydric, mesic, and xeric); and 3) dependency on topographic features to clearly delineate study units (Omernik and Bailey 1997).

Some researchers argue that a spatial framework for characterizing water resources should be grounded in criteria that best reflect the unique homogeneous environmental conditions of the areas of interest (Griffith et al. 1999; Wright et al. 1998; Omernik and Bailey 1997; Ravichandran et al. 1996; Hughes and Larson 1988; Whittier and Hughes 1988; Bailey 1983). The ecoregion framework, developed in the 1980's, offers an alternative for landscape delineation. It delineates the landscape based on area commonalities of ecological systems, organisms and environmental conditions (Wright et al. 1998; Omernik and Bailey 1997), provides a spatial framework for ecosystem analysis and monitoring, and may be applied to watershed planning and management activities (Cannon 1994).

While watershed and ecoregion approaches utilize different criteria, each has important purposes and they can be complementary when used correctly. Omernik and Bailey (1997) caution that care must be exercised in choosing the appropriate geographic framework for spatial analysis or to address a given set of questions pertaining to ecosystem management and ecological processes.

Water quality is dependent in part on the integrity of the surrounding landscape, and land uses within the watershed (or region of influence) may account for much of the variability of stream quality (Gove et al. 2001; Omernik 1977). Numerous studies have identified the relationship between water quality and stream health and landscape characteristics (e.g., Jones et al. 2000), nutrients (e.g., Alexander et al. 2000; Smith et al.

1997; Jordan et al. 1997; Puckett 1995) and sediment loads (e.g., Levine et al. 1993). Since assessment of water quality has a spatial component, recognition of the differences in criteria used in the different spatial frameworks (*viz*. watershed, hydrologic unit and ecoregion) and characterization of the landscape and land use at multiple scales are critical for accurate measurement.

Water quality monitoring programs have a variety of information goals including characterization of current conditions, trend analysis and detection of values exceeding regulatory limits (Dixon and Chiswell 1996) and serve to guide future assessment efforts (Ficke and Hawkinson 1975). Identification and assessment of trends is important for policy and management initiatives particularly in the areas of ecosystem health, human health and economic well-being. Testing water quality monitoring data for trends through time has received some attention (see for example, Smith et al. 1997; Smith et al. 1987; Lettenmaier 1978; Steel et al. 1974). Likewise there is a growing body of literature on the development and application of statistical methodologies for trend detection (see, for example, Nickerson and Brunell 1997; Helsel and Hirsch 1992; Hirsch et al. 1991; Gerrodette 1987; Hirsch et al. 1982).

A trend in water quality appears when there is a consistent pattern of change in values in constituent(s) of interest in one direction (increase or decrease) over time. By definition, a trend is detected when the regression has a slope significantly different from zero (Gerrodette 1987). In some applications it may be sufficient to determine if there is a propensity for water quality parameters to deviate from initial values rather than precisely model patterns of change (Nickerson and Brunell 1997). However, Landwehr (1979 p. 466) suggests, "…if an increasing or decreasing trend is suggested by the index values, further examination is necessary to determine if indeed a trend is taking place…or whether the situation has merely become more variable….” Many factors may confound interpretations of trends, including adequacy of data points and the behavior of the data itself.

A statistical operation, power analysis, may be used to address these issues of adequacy of data record length and magnitude of trend (necessary for detection) with a reasonable degree of certainty. The ability of a statistical procedure to distinguish a situation (for example, a trend) different from the null hypothesis is called the power of that procedure (Gerrodette 1987). Power analysis can be used to detect trends in the presence of concomitant variables that may obscure patterns of change (Nickerson and Brunell 1997). It is used to test for a Type II error (which concludes no trend exists when in fact there is a trend). Schlesinger (1989, p.1) notes that “strong inference in science demands testing the null hypothesis” and Peterman (1990, p.2025) also argues that inferences made in science demand “reporting of the probability of making Type II error when the null hypotheses is not rejected as well as the traditional of *P* (probability) values”. When the null hypothesis has not been rejected, inferences can be greatly strengthened by knowing the detectable effect size (Cohen 1988; Rothenbery and Weins 1985) for a desired Type II error rate, the given sample size, and the sampling variability (Peterman 1990). Considering the statistical support power analysis can provide trend detection, combining trend detection and power analysis can provide convincing evidence of the status and changes in water quality.

Goal and Objectives

National monitoring programs provide a rich data source from which to characterize water quality status and trends in the U.S. To date most of the analysis has been made using a spatial framework based on hydrological units. While this model has been useful for compliance monitoring, it may not adequately characterize the "domain of influence" of the landscape on water quality at a large scale. In contrast to the hydrologic unit and watershed models for landscape delineation, there has been little work on water quality analysis at large geographic scales such as ecoregions.

There is a growing awareness of the global nature of many environmental concerns and the interconnectedness of ecological properties. Consideration of the environment as a cohesive whole indicates a change in the scale of problems experienced, an understanding of their causative factors and cumulative impact, and a recognition that requires investigation and management actions on multiple spatial and temporal scales (Harmancioglu and Fistikoglu 1998). Despite its central role in human and social well-being, most research on and management of water resources is still conducted on spatial scales ranging from stream reach to (local) watersheds. There are an increasing number of initiatives that address a wide range of environmental issues at the scale of political boundaries (e.g., state level), region and landscape. This scaling up of research (using data from numerous sites to characterize conditions and trends over a larger area) requires examination of multiple variables simultaneously, synthesis of data, and assessment of trends over space and time. However, most of these efforts of data aggregations across watersheds are based on the traditional hydrologic unit model. Additionally, water quality assessment, characterization and trend detection at larger spatial scales present unique challenges to the analysis of the data and interpretation of results.

The goal of this research project is to identify long-term trends of selected water quality characteristics in a large geographic region at multiple scales and to evaluate effectiveness of current monitoring efforts to detect changes. The objectives are to:

1. Describe long-term water quality trends among stations situated over a large geographic area in the U.S.;

2. Identify differences in trends among constituents at multiple spatial and temporal scales;

3. Identify minimum critical monitoring parameters for the detection of trends in selected water quality constituents, including levels of detection and length of monitoring;

4. Provide observations for improvements to monitoring protocol.

Hypotheses

Despite the enormous investment of resources, policy changes and management initiatives in the public and private sectors, more than a third of all assessed stream and river miles in the United States do not meet water quality standards (USEPA 2000a,b). Water pollution and alteration of river systems have caused a wide range of environmental problems that directly impact on the ecological health and integrity of streams and rivers. While there have been significant improvements to water quality in many rivers and streams over the past quarter of a century, these improvements have not been uniform through time or space or for all constituents of interest. Water quality problems threaten human health and place economic and social burdens on society. Much of the research on characterization of surface water and human impacts on water quality has focused on relatively small spatial and temporal scales. Evaluation of water quality at larger spatial scales (for example, ecoregions or geopolitical regions) offers the advantages of improved synoptic assessment, operational efficiency in monitoring, and inclusion of factors operating at multiple scales. Synthesis of the data presented here highlight commonalities and inter-station differences of water quality trends. It also identifies constraints to improved explanatory and predictive models based on current monitoring programs. This research adds to the growing number of initiatives that "scale up" the results from study units to provide information useful to water resource management.

The hypotheses of this research effort are:

1) Most of the selected water quality constituents will show detectable trends over the period of study in some regions of the study area,and these trends will show a discernable geographic pattern related to land use/land cover;

2) Trends in water quality are more apparent over longer monitoring periods and large regions than at the scale of individual stations;

3) Amount of change in water quality constituent is too small to demonstrate a trend with a high probability of certainty at many stations. Conversely, amount of monitoring data are inadequate or monitoring time is too short time too detect a trend;

4) Power analysis can be successfully applied to water quality data;

5) Power analysis is useful in retrospective interpretation of data and operational design of future monitoring efforts.

Limitations of research

A major challenge to working with a large database is narrowing the scope of investigation to address the basic issues of interest: characterization of water quality trends and validation of the utility of a statistical procedure (power analysis) in water quality monitoring. In consideration of time and resource limitations, this research will not pursue issues that address:

1) Role of environmental factors including chemical, hydrologic and biological on water quality;
2) Evaluation of water quality as a measure of overall stream and river health and integrity;
3) Identification and assessment of stations and larger scale areas that fail to meet compliance criteria;
4) Effects of historical (pre-1973) land use/land cover practices on water quality trends.

CHAPTER 2:

LITERATURE REVIEW

Water quality is an important feature of water resources, and it may be perceived differently by consumer/user groups depending upon their interests and needs. For example, water quality standards for public health are different than those for industrial or recreational needs while the standards for ecosystem integrity differ spatially, depending on in-stream biota, flow regimes and other environmental characteristics. The "fitness" of water to meet the multiple uses may be established by defining parameters for selected constituents (for example, dissolved oxygen, toxins, pathogens, sediment load, etc.) (Novotny and Olem 1994). Scientists, resource managers, and policy managers recognize the wide range of deleterious impacts on water quality that result from human activities. This recognition has led to attempts to stem this degradation through monitoring and remediation programs. The spatial heterogeneity and the dynamic and complex nature of streams and rivers present challenges to the water quality monitoring and management. While these efforts have been on going in the United States for almost a century and have been more focused and coordinated in the past 30 years, much remains to be done in order to ensure that both human needs are met and ecological integrity of these systems maintained and protected.

Many years of research and management have produced an enormous amount of literature on water quality and an in-depth synthesis of any one topic of water quality would be an exhaustive undertaking. This literature review focuses on six issues of water quality that pertain directly to the overall issue of water quality monitoring discussed in the dissertation: 1) water quality; 2) factors that effect water quality; 3) past and current monitoring efforts; 4) spatial scales; 5) spatial frameworks; and 6) trend and power analysis. This brief review will provide the background for the two subsequent chapters on water quality trends in selected ecoregions in the United States and power analysis of trends.

Water quality in the United States

The quality of water affects many aspects of a community's well being, from the individual's health to economic and social institutions. Human activities and land use practices contribute to natural processes that affect water quality, and therefore management and policy options governing resource use must be constructed to safeguard this basic resource. This section briefly examines historical development of water quality policies, the present condition of water quality and factors that continue to contribute to its deterioration in the United States.

Adequate supplies of unpolluted freshwater are critical to long-term sustainability of a society's institutions and maintenance of its ecosystems. Freshwater that does not meet minimum quality standards is a value-depressed resource and may be a significant social and economic burden. Degraded water resources are a risk to human health and quality of life and leads to biological impoverishment (Naiman et al. 1995). Poor water quality also has wide-ranging effects on the ecological integrity and ecosystem services of freshwater systems. For example, degraded water quality is at least partially responsible for declining populations and loss of species diversity for a wide range of aquatic fauna, particularly fish and mussels (Frissell 1993; Williams et al 1989; Karr et al. 1985). This loss is reflected further in reduced revenue from commercial and sports fisheries and recreational activities and added expenses in cleanup efforts.

There are numerous examples where deteriorating quality of streams and other water supplies in the United States became a public issue and subsequently lead to efforts to safeguard them. Initially concerns focused primarily on impacts of the threats to the economy, safety, health and lifestyle. For example, Governor Gage of colonial Virginia issued a proclamation in 1610 that attempted to protect water supplies. *"There shall be no man or woman dare to wash any unclean linen, wash clothes, nor rinse or make clean any kettle, pot, or pan or any suchlike vessel within twenty feet of the old well or new pump. Nor shall anyone aforesaid, within less than a quarter mile of the fort, dare to do the necessities of nature, cinse by these unmanly, slothful, and loathsome immodesties, the whole fort may be choked and poisoned"*. The Refuse Act of 1899 prohibited discharge of any refuse into the nation's navigable waters (Foster and Matlock 2001) but did little to control other types of pollution, such human and animal wastes, toxic material, metals and sediment. In response to high rates of siltation in streams caused by forestry and agriculture, Congress passed the Weeks Act of 1911. The major concern at that period was loss of river navigation, hydropower, and agricultural productivity (Walker 1991). The founding of the U.S. Public Health Service in 1912 signaled federal government involvement in water quality issues as they pertained to human health. Quality of drinking water served in interstate commerce was the first concern of the Public Health Service. Later in the 1940's, the Public Service's attention included ambient water quality in rivers and lakes (Rogers 1993). The Federal Water Pollution

Control Act of 1948 (PL 80-845) acknowledged the rights and responsibilities of states in matters of water quality and provided financial support for technical assistance and research; it also set the groundwork for subsequent legislative and management guidelines for the protection and improvement of the nation's freshwater resources (Foster and Matlock 2001). The 1972 Federal Water Pollution Control Act Amendment (PL 92-500), now called the Clean Water Act and the 1978 amendment stated the goal to "restore and maintain the chemical, physical and biological integrity of the Nation's waters"(Karr and Chu 1997). As instruments for river and stream protection developed, aesthetics, recreational value and ecosystem services of freshwater systems assumed an increasingly important role in policy and management. Technological advances, policies and management during the last quarter of the twentieth century were instrumental in the dramatic improvement of some water quality constituents. In particular, the new generation of sewage treatment facilities and operation guidelines resulted in dramatic declines in fecal coliform bacteria counts in many water bodies in the United States.

Despite many programs aimed at improving water quality, large investments of money for research, monitoring and capital improvement, and the involvement of government and nongovernmental agencies for more than thirty-five years, water quality issues still are not adequately addressed. Water quality in many areas of the United States is below acceptable standards and almost 80% of the American population lives within ten miles of a polluted body of water (USEPA 2000a,b).

Water quality refers to the composition of water, affected by natural processes and cultural activities. The various constituents, usually measured in concentrations per unit volume, are used to characterize water quality. Water pollution refers to undesirable changes in the condition of water or the causes of degradation in quality. Water quality and pollution are determined by measuring its chemical, physical, biological and radiological properties against a set of standards. These standards are limits and are usually based on human health or other criteria, such as visual preference, and they provide the framework for monitoring and compliance (Chesters and Schlerow 1985).

The USEPA classifies water quality based upon its ability to support one or more of the six designated uses, including: aquatic organisms, fish consumption, primary contact (i.e. full body contact, such as swimming or bathing without risk to human health), secondary contact (i.e., recreational activities with minimal water contact, such as boating), drinking supply; and agricultural uses. For each of these uses, USEPA rates water as good (fully supporting), good but threatened, fair (partly supporting), poor (not supporting), and not attainable (USEPA 2000a,b). The 1999 Index of Watershed Indicators (USEPA 2000a) noted that only 15% of the 2262 watersheds in the 50 contiguous United States and Puerto Rico had "good" water quality while almost 59% had some or more serious water quality problems; there was no data on the remaining watersheds. The report further noted that 6% of all watersheds were highly vulnerable to (further) degradation from pollution. The USEPA (2000a) lists siltation, pathogens, nutrients, oxygen-depleting substances, metals, and pesticides as the leading pollutants of streams and rivers nationwide. Primary sources of pollution include agriculture, urban runoff/storm sewers, municipal point sources, resource extraction, forestry, land disposal and habitat modification.

While natural processes and events (e.g., fires, succession, changes in streamflow), and catchment properties (e.g., soil, vegetation type, geology) impact water quality (Bis et al. 2000), human activities and land use practices are the principal factors influencing water quality within developing basins (Gove et al. 2001; Basnyat et al. 2000b; Basnyat et al. 1999). Numerous studies (e.g., Jones et al. 2001; Herlihy et al. 1998; Tufford et al. 1998; Allan et al. 1997; Allan and Johnson 1997; Smith 1992) demonstrate that many characteristic water properties (e.g., anions and cations, nutrients and acid neutralizing capacity) and biota are strongly related to watershed land cover and land use activities. Human activities may increase runoff (e.g., through removal or changes in type of vegetation), decrease streamflows (e.g., through water extraction and diversion), contribute to inputs of nutrients (e.g., fertilizers, human and animal waste products) or pathogens (e.g., sewage and animal wastes), or contaminate water with toxic materials (e.g., pesticides, metals, pharmaceuticals, etc.). Changes in climatic patterns are also likely to influence both stream flow and water quality (DeWalle et al. 2000; Cruise et al. 1999).

Most of the gains in improved water quality were made through the control of point source pollution, such as industrial and municipal discharges. Non-point source pollution, however is the most common type of pollution today and is the leading cause of degraded water quality in the United States (Bhaduri et al. 2001, 2000; USEPA 2000a, 1998; Karr and Chu 1999). The severity of and extent of non-point pollution was articulated in the 1987 Water Quality Act which noted that "*It is the national policy that programs for the control of non-point sources of pollution be developed and implemented in an expeditious manner so as to enable the goals of this act to be met through the control of both point and non-point sources of pollution*" (USEPA 1989). As a result of their diffuse sources, non-point pollution is more difficult to monitor and control.

Non-point source pollution arises from diffuse sources including agriculture, forestry, urban runoff, erosion and discharge of polluted groundwater into surface waters (Novotny and Olem 1994; Heng and Nikolaidis 1998) and the dispersal of airborne material (Chesters and Schlerow 1985). The high contribution of sediment, nutrients, and pesticide residues from agricultural activities to surface water pollution has long been recognized. Urban development is responsible for an increasing portion of the non-point pollutants. A 5-year study by the USEPA concluded that urban runoff contains high concentrations of toxic metals and “priority” pollutants, coliform and pathogens and sediments (USEPA 1983). Numerous other studies have also noted the large contribution of pollutants to water resources originating in urban areas. Aerial discharges from point sources, such as power plants and industrial processes, and the more diffuse sources such as fertilizer and pesticide applications are another major non-point source of pollution affecting water quality. Some airborne pollutants quickly disperse and are deposited over large geographic areas, yielding a wide range of volatile compounds such as hydrocarbons, ammonia, hydrogen sulfide, methane, mercapans and particulates. Such pollutants are frequently delivered to receiving waters in the same concentration and amounts that were released (Chesters and Schlerow 1985). While there is some debate about the extent of the contribution of these airborne pollutants to stream water chemistry (Stottlemyer 1997), researchers have identified strong correlations between atmospheric input and water chemistry (Likens et al. 1996).

Efforts continue to prevent and/or control water pollution. Best Management Plans (BMP’s) have proven useful in controlling some types of non-point pollution. These plans include but are not limited to structural and nonstructural controls and operations and maintenance procedures such as protection of riparian areas, minimum tillage farming, restriction of livestock from access to streams and rivers, more efficient use of fertilizers and pesticides, improved road building techniques, and limitations on types of activities known to be major contributors to pollution (Novotny and Olem 1994).

Factors affecting water quality

Streams are the integrated product of all factors in the watershed and respond to the interaction of ecosystems within the watershed (Lotspeich 1980). A number of studies have demonstrated a strong relationship between water quality, water quantity and run-off to landscape characteristics (Jones et al 2000) and land use within watersheds (Basnyat et al. 2000a; Basnyat et al. 1999; Herlihy et al. 1998; Roth et al., 1996; Hunsaker et al. 1992). This section briefly reviews some of the environmental characteristics that affect water quality and vulnerability of streams to contaminant loadings.

Dillon and Molot (1997) note that chemical composition of surface water depends on *in situ* processes, external supply of substances, the loss rate of substances and modifying factors such as climate. Additionally, streams and rivers are very heterogeneous systems and vary both spatially and temporally in physical properties (e.g., flow rates, temperature, light and substrate availability), chemical composition, nutrient availability and biotic communities. This variability is a function of environmental properties and ecological processes, both in-stream and in the landscape through which the stream flows (Dent and Grimm 1999; Hornberger 1994). For example, the high degree of heterogeneity of available nutrients is related to multiple factors working simultaneously including catchment geology, vegetation, land use, and atmospheric inputs (Dent and Grimm 1999). The range of influence over stream water extends horizontally to the riparian area, longitudinally downstream and vertically to the groundwater. In turn, this chemical signature is a strong control of biotic processes in-stream, in the riparian area, and in the hyporheic zone.

Bedrock type appears to be a major factor influencing stream water chemistry. Bedrock type infers susceptibility of component materials to chemical weathering. Bedrock is non-random and is controlled by geologic factors that cluster rocks of like material and composition. In their study of streams in the Blue Ridge and Valley and Ridge physiographic provinces (Virginia), Puckett and Bricker (1992) noted that even at low amounts, carbonate minerals exerted a disproportionately strong influence in water chemistry; bedrocks containing carbonates readily dissolved in stream water. In contrast, water passing over more resistant, low reactivity quartz-based bedrocks exhibited chemistry similar to that of rainwater.

Climate and hydrology also are strong influences on water chemistry. For example, denitrification rates increase with higher temperatures, thereby decreasing nitrogen delivery into streams or reducing the amount already present (Seitzinger 1988). Atmospherically derived deposits, such as nitrate compounds or sulfates (SO_4^{-2}) are concentrated in intermittent streams in the watersheds in dry periods and released in surges or spikes downstream during precipitation events. Nutrient spiraling also can affect water chemistry through retention and release of nutrients. This ecological process is governed in part by abiotic processes (physical-chemical, hydrologic), and biotic controls (uptake, assimilation, excretion, transport, and transformation). The hydrologic regime influences nutrient uptake stored in biological tissue and thereby affects nutrient concentration in the water column. Nutrient uptake and retention appears to be favored by low flow, a high streambed area to channel volume ratio, retention devices (e.g., debris dams, beaver ponds), and permeable substrates (that allow substantial interstitial flow).

There is wide spatial and temporal variability in the susceptibility of lotic systems to contaminant loading. This vulnerability is a function of natural characteristics of the landscape and anthropogenic influences. Despite similar land use activities and disturbance regimes within the same water basin, concentrations of contaminants and how they are expressed may vary significantly (Hirsch and Miller 2001). Landscape characteristics include biological, physical and chemical factors of the in-stream environment as well as other factors of the water basin. Recognition of this variability is important for effective and accurate monitoring programs (Droppo and Juskot 1995). Accordingly, monitoring efforts must modify procedures and establish a baseline range of values of measured constituents before water quality data can be accurately assessed.

Hydrology and basin characteristics influence the magnitude and timing of contaminant transport, thus directly affecting water quality and its vulnerability to contaminants. Alexander et al. (2000) point out that nitrogen is more readily removed from small streams than from large ones noting that channel depth was the probable explanation. The shallower channels had a higher proportion of the water in contact with the sediment where transformation or retention of the nitrogen can occur. In the deeper channels the rate of this biogeochemical interaction is much smaller, and the nitrogen in the water column is transported out of the area. Therefore, nitrogen loads transported by larger streams or rivers are more likely to travel long distances and exert their effects farther from the point of origin than if the nitrogen enters smaller systems or those more distant (Smith et al. 1997; Hirsch and Miller 2001). This mechanism is important in understanding the spatial distribution of fertilizer-derived nitrogen sources transported intact to estuaries and coastal areas of the Gulf of Mexico where the ecological effects are considered to be the most severe. Researchers believe the anoxic conditions of the "dead zone" are linked to the downstream transport of nitrogen applied to agricultural areas, primarily in regions close to deep-water tributaries of the Mississippi River and Ohio River basins.

Despite the known influence of the landscape on water chemistry, multiple samples in time are required to detect subtle changes in flow-related alterations in chemical composition. In order to accurately characterize the geochemical factors that influence stream chemistry, a hierarchical sampling scheme is indicated (Puckett and Bricker 1992). Droppo and Juskot (1995) argue that transport characteristics of the contaminant are frequently overlooked in monitoring programs and are a major source of prediction error of contaminant concentration. The degrees to which they affect loading estimates are site and contaminant specific and are dependent upon monitoring program objectives and level of accuracy desired. These transport factors include: 1) sediment and particle size, 2) relative proportions of the three operational phases of the contaminants (dissolved, particulate and bedload), 3) flow regime, 4) physical, chemical and biological interactions between sediment, water and the contaminant, and 5) external controls such as contaminant source, nature of contaminant, climate and landscape characteristics. These interacting characteristics are expressed at multiple spatial and temporal scales.

For example, flow rates are determined by seasonality, short-term precipitation events, infiltration rate of water into the adjacent riparian zones, stream channel morphology, and the groundwater-stream interface and flow patterns. In turn, flow rates are a major influence on the proportions of the phases of the contaminants and particle size of the sediment.

Monitoring

Since the 1970's there has been increased emphasis on aquatic monitoring to satisfy regulatory, management and research needs. This increased level of effort in personnel, programs and expenditures has yielded a large amount of data that is both technically and statistically valid (Dixon and Chiswell 1996) and critical to meet a variety of social and environmental needs. Addressing the practical questions of why monitoring should be undertaken is an essential, and sometimes overlooked, first step. Once the goal is established monitoring objectives, protocol, and methodologies for data processing, analysis and synthesis follow. This section reviews the role of monitoring in research and management and the structure of three national-level water monitoring programs.

Historically, an array of federal, state and private monitoring networks and programs has provided data for water quality assessment in the United States. Since the early part of the 20^{th} century monitoring programs and networks have grown in number, size, complexity and sophistication and now support water quality assessment efforts at many geographic scales. Despite the widespread and longstanding interest in using the data from these programs and networks, there remain significant difficulties in interpreting point-level water quality data and generalizing it to larger spatial scales for assessments and trend analysis. Problems arise primarily because of the: 1) sparseness of sampling locations as a function of cost constraints. 2) spatial bias in sampling locations in order to target sampling toward specific pollution sources, and 3) drainage basin heterogeneity. These limitations reduce the utility of the data for larger scale interpretation and identification of in-stream and watershed variables affecting water quality (Smith et al. 1997).

Since the passage of the CWA there has been an increased effort to develop and maintain monitoring programs. These programs collect and analyze stream and water quality data to satisfy a range of regulatory and management objectives and prescriptions (Montgomery et al.1995). Objectives of freshwater monitoring programs include: 1) assessments of river conditions, 2) detection of existing and emerging problems, 3) establishment of priorities for current or emerging problems, 4) design and implementation of water protection and/or restoration programs, 5) evaluation of program or project success, and 6) monitoring emergency responses. Assessments characterize river constituents, detect spatial and temporal patterns, and identify factors and processes that influence river conditions (Smith et al. 1997), and they are important for the guidance of water policies and management (USEPA 1998).

In order to address a wide range of needs, both public agencies and private organizations implement monitoring programs. Federal, state, interstate, regional, local, tribal, and territory agencies are involved with public programs. In 1998, eighteen federal agencies were involved implementing 141 monitoring programs nationally (USEPA 1998). In addition, there is a multitude of environmental nongovernmental organizations (NGO's) and citizen groups that implement monitoring programs. The

Izaak Walton League of America's *Adopt A Stream Program* and the *Riverwatch* initiatives are notable examples of NGO monitoring efforts. Additionally, many industries are required to maintain a monitoring program and submit reports on a periodic basis to government agencies.

Historically monitoring relied on a chemical sampling regime in conjunction with flow measurements to estimate stream contaminant load (Droppo and Jaskot 1995). While the large number of monitoring efforts has provided important information at scales from the national to local level, there are some serious shortcomings to the applicability of the information they generate. Unfortunately, some of the information cannot be aggregated readily to obtain regional and national trends. Problem areas include: 1) inconsistencies among participating parties in data collection, evaluation, assessment and archiving procedures, 2) variation in quality control standards, 3) missing or incomplete spatial and temporal records, 4) incompatibility of recorded information and lack of accompanying descriptors that would allow other users to determine the meaning and utility of the information and, 5) changes in monitoring programs, resulting in modifications to protocol and program emphases (Smith et al. 1987; Wolman 1971).

Much of the research on water quality at the national and regional levels is based on data from the Water Quality Network (WQN). This network includes two monitoring programs, NASQAN and HBN. These programs have a national network of stations that consistently monitor a suite of water properties from watersheds that exhibit diverse physiographic, environmental and cultural characteristics. Usefulness of the programs is enhanced by their structure, including 1) relatively long period of continuous monitoring, 2) high frequency of sampling, 3) large number of monitoring sites, 4) wide geographic distribution, 5) large number of water quality parameters measured, and 6) sites representative of a diverse range of environmental and human influences. While the two monitoring programs have the same overall goal of providing water quality data for research, policy and management, they differ in objectives, design operation and history. Total operational expenditure for both monitoring programs through 1995 is $95 million; NASQAN's share is about $80 million.

NASQAN- Operations began in early 1973 and continue through the present. Primary objectives of the NASQAN program are to: 1) measure quantity and quality of stream water, 2) describe spatial variability in stream water quality, 3) detect long-term trends in stream water quality, and 4) provide information and guidance for future water quality assessments. Approximately 85 physical, chemical and biological properties are routinely monitored including dissolved, suspended and whole water (i.e., total) components of these properties. The number of stations grew from the initial 51 at startup to 513 in 1980. Selection of monitoring station location was based on the hierarchical model of classification of nested hydrologic units (consisting of major regional drainage basins, subregional drainage basins, accounting units and cataloging units). NASQAN stations are located at the outlets of streams from watersheds to account for the chemical mass and water transported from the landscape. Sites were chosen to represent "typical" natural and cultural features of the landscape (e.g., land cover type, land use and population density) of the United States.

Most physical and field measurements, ions and nutrients have been monitored regularly from 1973-1995 while other properties (e.g., fecal coliform bacteria, organic carbon, alkalinity) were added later. In response to administrative considerations (primarily budget), project needs and changes in analytical procedures, monitoring of some properties was discontinued as early as 1982 (e.g., whole water analysis of trace elements). Discovery in 1991 of a bias in of measurements of trace element lead to new sampling and processing protocols for low-level inorganic analyses; these new procedures were implemented in 1994.

The initial monitoring protocol schedule advised samples to be collected on fixed time intervals and frequency varied with constituent of interest. Mean streamflow, initially measured daily, was the most frequently recorded constituent. Other physical/field measurements (e.g., pH, dissolved oxygen and temperature), major ions, most nutrients and some biological properties (e.g., fecal coliform) are collected 4-12 times per year, and radiochemical properties are sampled least frequently, 1-2 times per year. Logistical, technical and management considerations, however, have modified schedules and generally reduced measurement frequency.

Major reviews of NASQAN in 1981, 1986 and 1995 resulted in changes in level of support, network objectives and operations that affected the number and type of stations, sampling frequency and constituents analyzed. Beginning in 1980 the number of analyses decreased from a yearly high of almost 300,000 to less than 50,000 in 1995. The number of stations began to decrease dramatically from a high of almost 500 in 1987 to 142 in 1995. While the objective of monitoring for mass transport was preserved after each round of review, greater emphasis was placed on monitoring water transport to major estuaries and freshwater bodies including the Great Lakes. The 1995 review, implemented the following year, shifted the focus of operations to more intensive sampling, analysis and characterization for a wide range of water properties, including herbicides. Additionally, the geographic area was more narrowly defined to include primarily only the Mississippi, Columbia, Colorado and Rio Grande River basins and the major coastal drainages in the United States (Alexander et al.1996).

HBN- The HBN program was established in 1958 and the first stations began monitoring water samples in 1962. By 1968 virtually all stations were operational. The goal of the program is to establish a benchmark or baseline of water quality in areas where human impact is minimal. The objectives were to investigate the influence of terrestrial and climatic factors on streamflow and water chemistry, understand the statistical characteristics of stationary hydrologic time series, and identify the effects of atmospherically derived pollutants on stream water quality. Located in 37 states, HBN monitoring stations are situated in moderate to small watersheds that represent a cross section of geologic and climatic provinces and a diversity of soil and vegetation types found in the United States. Most HBN monitoring stations are in "protected areas", such as national and state parks and wilderness areas because of the selection criteria of "minimally" impacted site; a few stations are located in moderately disturbed watersheds (e.g., agriculture, logging) where the land use was not expected to change radically.

The constituents measured and field and laboratory protocols for the HBN program are identical to those employed in NASQAN. Similarly, there have been some operational changes in the program during its history, primarily in response to levels of financial support. Most of the changes resulted in decreased analyses. For example, the number of stations in operation at any one time decreased from a high of 59 in 1988 to 53 in 1995 and the number of annual analyses fell from the 1981 high of about 17,000 to about 9,000 in 1995. Beginning in 1983, frequency of sampling decreased from monthly to quarterly for most of the stations.

NAWQA- The National Water Quality Assessment program (NAWQA) currently is the primary focus of the USGS water quality monitoring network. Built on the infrastructure of the NASQAN and HBN monitoring programs, NAWQA is designed to address limitations of earlier water quality programs that were driven primarily by compliance monitoring. Two of NAWQA's objectives are similar to those of the NASQAN program (current water quality conditions and water quality trends over time); a third objective is improvement of understanding of the primary natural and human factors that affect water quality condition. The NAWQA program provides an intensive and in-depth characterization of stream conditions and trend analysis over time, and examines the effect of land use activities on streamflow and water quality in the nation's most important and large river basins and aquifer systems. These study units are distributed throughout the U.S. and represent watersheds that cover 65% of the water used for drinking and irrigation. The 59 study units are divided into three sub-groups of 20 units each. Intensive monitoring for 3-4 years for the first 20 study units began 1991; the second and third sub-groups began monitoring in 1994 and 1997, respectively. This rotation of cycles will extend into the foreseeable future. Trends are assessed every 10 years and low-level monitoring is continuously on-going. Unlike earlier programs, NAWQA also reports on the presence and concentrations of a wide range of pesticides. One of the initial findings was the long-term persistence of certain contaminants, such as DDT, PCB and lead in sediment and fish tissue. Analysis of data from the first monitoring cycle further identified variability of concentration levels, seasonally and among watersheds having different land use patterns and natural features. The second monitoring cycle will focus increasingly on: 1) selected new pesticides with high usage in agriculture and populated areas and pesticide degradation products, 2) indicators of water-borne diseases, and 3) concentrations and fluxes of total and methylmercury. The number of study units in the second monitoring cycle was contracted to 42; the first sub-group of 14 began monitoring in 2001, and other subgroups will commence in 2004 and 2007 respectively (Gilliom, et al. 2001).

While the importance of monitoring programs is well recognized, many current programs are inadequate and fail to provide needed information and data. Reasons for the shortcomings are failure to meet one or more of the following criteria for an effective program: 1) establishment of an hypothesis that is driven with clearly defined objectives, 2) program that is based on sensitive indicators of change, 3) program based on mechanistic or causal relations between observed change and suspected disturbances, 4) sampling strategy appropriate for detecting changes, 5) format and framework for organizing, analyzing, storing and retrieving monitoring data, and 6) procedure for incorporating monitoring results into decision making processes (Montgomery et al. 1995).

Spatial scale of research efforts

Environmental and ecological phenomena often occur at multiple spatial and temporal scales simultaneously. Choice of the appropriate scale for research and management is important in order to meet objectives and to ensure accuracy of data interpretation. Broadly speaking, scale is the term most often used to describe the level of geographic detail. To the general scientific community, large scale implies either a large extent of geographic coverage, or more commonly, a coarse level of geographic detail. The critical point is that the level of geographic detail in data affects the outcome of analysis (Goodchild and Proctor 1997). This section on scale examines the expanding role of large scale analysis and synthesis of data in research and management of natural resources, particularly water.

Within the last two decades, water resources management has become more complicated as the problems experienced have grown both in scope and scale. Consideration of the environment as a cohesive whole indicates recognition of the scale of problems experienced, an understanding of causative factors and cumulative impact, and awareness that solutions require investigation and management actions on large and multiple spatial and temporal scales specific to the issues of interest (Hay et al. 2001; Harmancioglu and Fistikoglu 1998). Historically, scientific knowledge was not applied effectively in resource management in part because landscapes were not viewed as systems and linkages among processes and landscape elements were not addressed (Montgomery et al. 1995). However, some of the most important environmental changes occur at large spatial scales such as landscapes, and ecological interactions may produce spatial patterns at landscape level (O'Neill et al. 1997). Characterization of the status and trends at large spatial and temporal scales is useful for understanding the overall condition of ecological resources and processes (Michener et al. 1997; Graham et al. 1991; Urban et al. 1987) and human impacts on these features. Additionally, political decisions to manage natural resources are made at the broad scale, such as river basins, forest districts and states (O'Neill et al. 1997). Recent advances in freshwater ecology that highlight the importance of multi- and large scale analysis include: nutrient spiraling (Newbald et al. 1981, 1982), the multi-dimensional nature of freshwater systems (Ward 1989), and hydrologic connectivity (Pringle 2001, 2000a, b; 1997; Pringle and Triska 2000). Additionally, the relationship between climate change and freshwater systems is receiving increased attention, and numerous research efforts show that air chemistry affects water quality and the hydrologic cycle (Ramanathan et al. 2001; Meyer et al. 1999). Research has also demonstrated the importance of the two-way coupling of land and water ecosystems in providing nutrients and habitat for wildlife and vegetation along the entire course of a river system, and this relationship may be best understood at a large scale (Willson et al. 1998).

While many research efforts still focus on environmental conditions and ecological processes at the local scale, for example stream reach or small hydrologic unit, there is growing interest in research and management strategies that can be applied at larger spatial scales such as entire river systems and large geographic areas. For example, Perry et al. (1999) examine impacts of riparian forest management on water quality for the dendritic stream network of the entire Litter River (Georgia) system. Lorenz et al. (1997) developed river ecosystem indicators for the Rhine River based on river abiotic and biotic characteristics and Wickham et al. (1997) examined the relationship of stressors to geographic patterns of species richness at continental and global scales. Some researchers are developing tools that assess specific ecological functions across large geographical areas to compare traits of biota among communities across biogeographic regions (Statzner et al. 2001).

Research and analysis of data over large spatial scales is required to assess tradeoffs when balancing economic/social needs with ecological priorities and conservation. Coupled to landscape-level planning, large scale analysis can provide a framework for generating the information required to accountably access performance toward achieving environmental objectives (Montgomery et al. 1995).

Numerous research efforts attempt to model hydrologic and water quality conditions at the macro-scale (Arnell 1999; Srinivasan et al. 1998), and the USEPA is assessing current and future water quality conditions of the southeastern (Cruise et al. 1999) and western (Jones et al 2000) states in the U.S. at the landscape level. The biennial reports to Congress of the National Water Quality Inventory summarize water quality conditions on a state-by-state basis (USEPA 2000a, 1998). There also have been numerous assessments of regional trends in water quality based on USGS's NAWQA data (Smith et al., 1997; Hirsch et al., 1991; Lettenmaier et al., 1991; Smith et al., 1987). The 1994 report of the Committee on USGS Water Resources Research stressed the importance of *scaling up* information collected from individual study units to provide a national synthesis of the status and trends of water resources in the U.S. The report also notes that highlighting important commonalties and major regional differences is critical in obtaining a broader picture of water quality phenomena for use in program decision and policy making at the national level (Hornberger 1994).

While there is interest in monitoring and analyzing status and trends of ecological processes and environmental conditions at very large spatial scales (ecosystem, continental, global) there were few well-coordinated efforts prior to the 1990's. Bailey (1991) noted that with few exceptions, publications about ecosystems rarely use existing information about geographical variability to design monitoring programs, and there were few or no global environmental monitoring programs of ecosystems at this time. A notable exception to this generalization is the Global Environment Monitoring System (GEMS)/Water Program. Launched in 1976, its mission is to archive and analyze data from a global network of reporting stations to determine status and trends of regional and global water quality (Fraser et al. 2001). The report on freshwater systems of the Pilot Analysis of Global Ecosystems (PAGE) program provides an overview of freshwater resources at the continental and global levels. PAGE is also a forerunner of the more intensive and integrated analysis of global ecosystems under the Millennium Ecosystem Assessment Program that began in the late 1990's (Revenga et al. 2000). The international Hydrology for Environment, Life and Policy (HELP) Initiative, supported by the United Nation Educational, Scientific and Organization and the World Meteorological Organization has recently begun work on hydro-socio-ecological sustainability for a number of candidate water-basins (Endreny 2001).

Scientists are making significant progress toward understanding how landscape variables influence the physical, chemical and biological properties of freshwater systems. Stream characteristics, such as water chemistry, biota and geochemical dynamics, are affected by both local and regional measures. The type and scale of data that demonstrate the strongest influence depends on the variable measured and on study design. As scientists build on existing experience with spatially scaled studies, increasing attention should be paid to temporal and spatial characteristics, the intersection of these two dimensions and the hierarchical structure of spatial data. An improved understanding of the spatial and hierarchical relationships among linkages across the land-water ecotone would do much to guide future study designs (Allan and Johnson 1997).

Spatial frameworks: ecoregions and watersheds

Management of rivers and streams requires an understanding of their geographic and temporal patterns. These lotic systems are very heterogeneous at scales ranging from millimeters to tens of kilometers (Cooper et al. 1997) and the appropriate geographic framework must be incorporated into data collection, analysis and synthesis in order to effectively address a wide range of research and management questions. While there are numerous spatial models for delineation of the landscape, the watershed framework is most frequently incorporated in water resource studies. A more recently developed model, the ecoregion, is gaining wider acceptance; it is , based on landscape characteristics. This section explores the definition, development, application and differences in these two spatial frameworks.

The ecoregion model is an important concept in landscape ecology (Hargrove and Hoffman 1999), and it is used increasingly in research and management as an alternative framework for delineating geographic boundaries. It is an *a priori* deductive method of identifying broad-scale patterns of natural features that are functionally related to ecosystems or areas of interest (Whittier and Hughes 1988). As a management and research tool it establishes a logical basis for characterizing ranges of ecosystem conditions or quality that are realistically attainable (Whittier et al. 1988; Omernik 1987) and it may be used in the development, management and protection of natural resources (Ravichandran et al. 1996; Wright et al. 1993; Hughes and Larson 1988).

The ecoregion construct provides a geographic classification and stratification of land in a hierarchical structure that is based on commonalities of recognizable characteristics, such as climate, soil, topography, vegetation type and land use. Ecoregions stratify the landscape into zones that have similar ecological potential and within which the mosaic of ecosystem components is different than adjacent areas in a holistic sense (Griffith et al. 1999; Omernik and Bailey 1997; Bailey 1983). They delimit large areas within which local ecosystems occur more or less throughout the region in a predictable pattern (Omernik and Bailey 1997). Ecoregions are areas of homogeneity with respect to ecological systems involving interrelationships among organisms and their environment (Omernik 1995). This geographic framework focuses on spatial patterns of the aggregate of organisms rather than the characteristics of each group.

Herbertson (1905) made one of the earliest attempts to classify the world into "ecological regions" and he recognized the importance of human impacts (i.e., development) on landscape conditions. Later efforts at landscape classification were based primarily on distribution of climate-vegetation zones (Bailey 1983). While researching models to inventory forestland, Rowe (1962) suggested that geographic classification of land is necessary to identify relational patterns and that uniformity of relief, geology, local climate and native vegetation mimic topographic patterns. He also recognized the role of scale and developed a construct that incorporated multiple levels of classification. Crowley (1967) proposed a hierarchical structure of spatial units that can be applied to a classification system by defining smaller ecosystems within larger ecosystems thereby providing multiple levels of resolution of areas and subjects of interest. Bailey (1983; 1980; 1976) incorporated this hierarchical framework into his four-tier ecoregion classification model where each level has unique basic criteria. The basic criteria at the broadest level, *domain*, describes the prevailing climatic condition characterized by climatic zone or group (e.g., humid temperate). The 4 domains are subdivided into 16 *divisions* that characterize definite vegetation affinities (e.g., prairie) and fall within the same regional climate; basic criteria for this level are climate type, using the Köppen or Thornthwaite classification system. Divisions are subdivided into 64 *provinces* on the basis of climax plant formations (e.g., tall-grass prairie). Provinces are further subdivided into *sections* on the basis of differences in the composition of climax vegetation type (e.g., mesquite-acacia). All levels are needed to fully characterize areas of interest.

While attempting to classify streams for more effective water quality management Omernik (1987) modified the Bailey ecoregion construct to display regional patterns that are reflected in spatially variable combinations of causal factors, including climate, soils and geology, vegetation, and physiography and integrative factors, primarily land use. The Omernik model differs from the Bailey model in the defining criteria, lines of delineation and number of subunits within each level of the hierarchical structure. This model consists of 15 Level I classes (broadest scale), 52 Level II classes, and 99 Level III classes. Some regions of the United States are further subdivided into Level IV classes. Initial efforts to use this model were "at the state level of resource management. They focused on aquatic ecosystems, mainly attainable ranges in chemical quality, biotic assemblages, and lake trophic state"(Omernik 1987 p.123). Like the Bailey model, most of the work in identifying boundaries for the higher-level resolution maps has been done for the conterminous United States.

Despite strong arguments supporting the use of the ecoregions as a spatial framework, science initially was slow to apply it in research and management (Whittier and Hughes 1988), and the use of ecoregions across geopolitical boundaries still is not widespread. Some criticism of the ecoregion framework stems from border delineation. Historically, this regionalization was a subjective process that involved integrating and weighting of environmental characteristics by experts, and the subsequent placement of borders lead to disagreement and frequent revisions of maps. For example, Wright et al. (1998) observe that there was no fidelity between vegetation type and ecoregion boundaries in Idaho, Oregon and Washington for both the Bailey and Omernik models. However, there have been recent attempts to resolve the problem of border placement by using a more rigorous process such as the Multivariate Geographic Clustering process (Hargrove and Hoffman 1999).

Ecoregions provide a common spatial framework for ecosystem assessment, research, monitoring and management for some resource management agencies. Successful application of the ecoregion framework has been applied at various scales including international, national and state. The National Research Council suggests that restoration goals and assessment strategies should be established on an ecoregional basis (NRC 1992). The most common use in the U.S. is at the state level; resource management agencies in some states, (e.g., AR, IW, NE, OH, OR, TX and WA) have used ecoregions primarily to set water quality standards and to develop biological criteria and non-point source pollution management goals (Omernik and Bailey 1997). Canada now uses ecoregions for multiple purposes including reporting on the state of the environment, developing protected area strategies and developing regional indicators of forest disturbance and biodiversity (Government of Canada 1991).

Another widely used spatial framework for delineating the landscape is the watershed model. The suitability of this well-developed model for many applications has long been acknowledged (Omernik and Griffith 1991) and it is used currently in monitoring, research, and management (Omernik and Bailey 1997; Montgomery et al. 1995). Many ecologists consider watersheds as ecosystems with a range of complexities dependent in part upon its size (Lotspeich 1980). Odum (1969) emphasized that if water quality problems are to be effectively addressed the entire watershed (or basin) must be considered in management plans. Most States continue to use this model to organize their semi-annual water quality status reports to Congress and to address other resource assessment and management needs (Omernik and Bailey 1997).

The watershed model is based on drainage patterns of a region. Watersheds are topographic areas in which apparent surface water runoff drains to a specific point in a stream or to a waterbody such as a lake. The USGS uses the watershed as the basis to map out its hierarchically-structured hydrologic unit system. The United States is divided into successively smaller hydrologic units classified into four levels: regions, sub-regions, accounting units, and cataloging units. The hydrologic units are arranged within each other, from the smallest (cataloging units) to the largest (regions). Each hydrologic unit is identified by a unique hydrologic unit code (HUC) consisting of two to eight digits based on the four levels of classification in the hydrologic unit system. There are 21 regions in the United States, and each region contains a major river or the combined drainage of a series of rivers. The second level of classification divides the regions into 222 sub-regions, each of which includes the area drained by a river system, a reach of a river and its tributaries in that reach, a closed basin(s), or a group of streams forming a coastal drainage area. The third level of classification subdivides many of the sub-regions into 352 accounting units; these units nest within or are equivalent to sub-regions. The fourth level of classification and smallest geographical area is the cataloguing unit. It is a geographic unit representing part or all of a surface drainage basin, a combination of basins, or a distinct hydrologic feature. There are approximately 2150 cataloguing units in the United States (USGS 1992; Seaber 1987).

Omernik and Griffith (1991) noted that spatial differences in the quality and quantity of environmental resources correspond to topographic divides. While the USGS watershed-based model is useful, it has some serious weaknesses that may lead to inappropriate management decisions. Omernik and Bailey (1997) point out these problematic areas: 1) these hydrologic units may consist of segments of watersheds or watersheds with adjacent interstices, and consequently, at each level of classification the majority of the hydrologic units may not be not true topographically-defined watershed, 2) areas in which there is a similarity in the aggregate of geographic characteristics frequently do not correspond to the patterns of topographic watersheds, 3) large portions of the U.S. (20% - 33%), including deserts, wetlands and karst and glacier-effected lands (e.g. pothole regions) do not have well-defined drainage networks, making the determination of water flow direction in these landscapes problematic, 4) the watershed model does not consider differences in the ground water and surface water dynamics among the xeric, mesic and hydric environments, and 5) watersheds cut across regions of diverse climate and landform.

CHAPTER 3

TREND ANALYSIS OF WATER QUALITY IN THE SOUTHEASTERN PLAINS AND PIEDMONT ECOREGIONS OF THE UNITED STATES: 1973-1995[1]

[1] Bjorkland, R. A. and V. Meentemeyer. To be submitted to *Journal of American Water Resources Association*.

Abstract

Information about trends is crucial for guidance of water policy, management practices, and regulatory efforts. However, most studies of water quality trends fail to adequately address the importance of spatial and temporal scales and differences in trends detection attributable to inherent landscape characteristics. This study provides new information about spatial patterns of time series trends of constituents indicative of water quality, the effect of monitoring length and spatial scale on trend detection, and the impact of changes in land use/land cover (LU/LC) on trends. It recognizes that this type of broad scale analyses is confounded by biases and factors ranging from small scale to large scale. Such biases include potential lack of representativeness of specific sites within monitoring networks to the biases and the inherent assumptions of the ecoregional framework that was employed.

This study examines trends of six water quality constituents: percent saturation of dissolved oxygen, pH, total nitrogen, total phosphorous, total sediment, and instantaneous stream flow. Water quality data from 41 stations of the U.S. Geological Survey's (USGS) National Stream Quality Accounting Network (NASQAN) program in southeastern United States were analyzed using the seasonal Kendall test. Trend analysis was conducted at multiple spatial scales and for two time periods. Spatial scales included individual stations and the USGS's watershed-based 6-digit and 4-digit Hydrologic Unit Code (HUC) areas. Changes in the LU/LC were calculated from the 2001 Natural Resources Inventory database. Summary of the results show the following: 1) all water quality constituents demonstrated some trend during both monitoring periods, and the longer periods yielded more trends; 2) there was little difference in trend detection among

the three spatial scales examined (stations, 6-digit and 4-digit HUC); 3) pH reported the largest number of trends at all spatial scales; 4) pH levels increased throughout the study area; 5) patterns of trends were less well defined for other constituents; 6) more than 7% of the study area experienced LU/LC change over the 1982-1992 period; the urban category experienced the largest percentage increase in area and cropland the greatest decrease; and 7) increasing pH levels are associated with moderate amounts of regeneration of forest regrowth on barren lands and abandoned farms and conversion of cropland to pastureland.

Retrospective analysis of trends in water constituents is confounded by the quality of the information, including incomplete data, inconsistencies in monitoring schedule within and between stations, and inadequate record length.

Keywords: trend analysis; water quality; water quality monitoring; Southeastern Plains ecoregion; Piedmont ecoregion; seasonal Kendall test

Introduction

There is considerable interest among scientists, policy makers, resource managers and health officials in monitoring and management of freshwater resources. This interest is a result of growing concern over the impact of human activities (direct and indirect) on water quality (Uri 1991) and recognition that water quality, human welfare and ecosystem health and integrity are inter-related (e.g., Naiman and Turner 2000; SCOWAR 1998). One outcome of this interest has been the establishment over the past few decades of monitoring programs that accumulate and analyze reliable long-term water quality data (Hirsch et al. 1991). A common goal of many monitoring programs is to provide information crucial for guidance of water policy, management and regulatory efforts (USEPA 1998). Information derived from these programs is used to characterize water quality status, identify changes over time for a range of constituents (e.g., pathogen loadings, toxin concentrations, dissolved oxygen), and to help identify factors and processes that influence them (Smith et al. 1997).

The importance of monitoring and assessment at larger spatial scales (small map scale) is well recognized. The 1994 report of the Committee on USGS Water Resources Research stressed the importance of scaling up information collected from individual study units to provide a national synthesis of water quality phenomena for use in program decision and policy making (Hornberger 1994). Characterization of the status and trends at large spatial and temporal scales (long time period) is useful for understanding the overall condition of ecological resources and processes (Michener et al. 1997; Graham et al. 1991; Urban et al. 1987) and human impacts on these features. Additionally, political decisions to manage natural resources are made at the broad scale, such as river basins, forest districts and states (O'Neill et al. 1997).

There is increasing interest in evaluating the relative condition of water resources at regional or national levels (Jones et al. 2001); however, many studies of water quality still focus on individual streams or stream segments affected by local pollution sources or disturbances at the watershed or basin level (see for example, USGS Water-Resources Investigations Reports [Bell et al. 1996; Butler 1995; Cary 1989; Allen and Cowan 1985]). Other studies that investigate water quality at larger spatial scales such as entire states (see for example USGS Water-Resources Investigations Reports [Trench 1996; Petersen 1992; Buell and Grams 1985]) fail to consider naturally occurring differences in water characteristics as a result of heterogeneity of landscape features. A hierarchically structured spatial framework that identifies regions with similar physical and biological properties is needed to address these challenges. The ecoregion construct is useful for this purpose and can be used in large scale and multi-station comparative analysis.

The goal of this paper is to examine changes in water conditions in streams and rivers throughout a large geographic area and to identify factors driving these changes. The objectives include: 1) identification of the presence and direction of significant changes in values of percentage dissolved oxygen, pH, total nitrogen, total phosphorous, total sediment, and instantaneous stream flow in monitoring stations in the Southeastern Plains and Piedmont ecoregion for the period 1973 – 1995, 2) comparison of trend data at multiple spatial scales: station, 6 and 4-digit Hydrologic Unit Code (HUC) level, 3) identification of land use/land cover changes during 1982-1992 for the study area, and 4) assessment of the effect of land use/land cover changes on water quality.

Background

Water quality is constantly changing in response to physical, chemical and biological conditions. These changes may occur gradually over multiple time periods (e.g., hourly, daily, seasonally) thus producing a monotonic trend. Other changes in quality may be episodical or "random" and result from specific events, such as periodic discharge of effluents from sewage treatment facilities or storms. These changes produce "step" trends.

Water quality trend detection and assessment serve primarily as a warning signal of change, identifying if the quality is improving or deteriorating. This information is useful for decision-making and remedial or preventative actions. Trend detection provides a basis for predicting future conditions and calculating historical conditions. Additionally, it aids in the assessment of the impact of human and natural disturbances on the landscape and the effectiveness of management schemes.

Assessment of trends in water quality require consistent, long-term, and geographically widespread data from representative sites and appropriate statistical techniques that can treat trend detection problems associated with water quality data (Alexander et al. 1996). In order to collect these data, numerous government agencies and environmental groups maintain water quality monitoring programs. For example, in 1998, eighteen federal agencies were involved in implementing 141 monitoring programs nationally (USEPA 1998). These monitoring efforts help identify emerging trends in overall water quality, as well as changes in individual water constituents over time.

Trend detection involves finding a "signal" in the midst of "background" noise and depends strongly on the number and precision of the samples. In effect the louder the noise or the smaller the trend the more data are required for trend detection. By definition, a trend is detected when the regression has a slope significantly different from zero (Gerrodette 1987).

Accurate, precise and timely detection of water quality trends is confounded by the availability and nature of the data. Primary sources of water quality data for the United States are the numerous intensive studies of selected rivers with historically serious water quality problems, and water quality monitoring programs such as NASQAN (Smith et al. 1987). However, differences among localities in methodology, objectives and completeness of data seriously limit utility of the data for time series analysis and inter-station comparisons. For example, while the NASQAN program has been operating for about thirty years, most stations have been operational for shorter periods. Disparities in operational procedures and start-up and ending dates may not provide data useful for comparative purposes or as a basis for trend analysis. While it is well recognized that water chemistry and quality of rivers naturally vary spatially and temporally (Aulenbach and Hooper 1996; Turk 1983), differences in monitoring procedure tend to confound data analysis and add to the difficulty of detecting water quality trends. Other operational differences within the NASQAN program include: 1) number of constituents measured, 2) frequency of samples collected, 3) date (month) of collection, 4) length of collection period, and 5) location and density of monitoring stations.

The nature of water quality data itself presents a challenge to trend analysis. These characteristics include: 1) a lower bound of zero (no negative values possible), 2) presence of outliers, 3) positive skewness (a function of items 1 and 2), 4) non-normal distribution of the data, 5) censored data (e.g., values below detectable limit), 6) gaps and/or missing data, 7) seasonal patterns for most constituents, 8) autocorrelation of values, and 9) strong covariance with other uncontrolled variables (Helsel and Hirsch 1992; Lettenmaier 1988).

Trend testing is an exploratory statistical technique and requires an adequate sample size. Various statistical tests and estimators, along with the use of exploratory data analysis procedures, are useful in determining water quality trends. Prior to analysis, a number of statistical considerations must be addressed, including: 1) type of trend hypothesis to examine (step trends versus monotonic trend), 2) general category of statistical methods employed (parametric versus nonparametric), 3) nature of data analyzed (concentration data versus flux data), 4) type of data manipulation to achieve the best results from the analysis (e.g., mathematical transformations of the data and removal of sources of variability due to seasonal and stream discharge variations, and 5) choice of technique for trend analysis of data records with censored or 'less than' values (Hirsch et al. 1991).

The Seasonal Kendall test was developed to address many of the challenges presented by constraints inherent in water quality data, and it is widely used to detect trends. The null hypothesis for this test is that there is no trend. This non-parametric test is a generalization of the Mann-Kendall test and adjusts for the effects of variability in data caused by seasonality by only comparing water quality data collected during the same season each year over the monitoring period. In this procedure the season may be user-defined, such as daily, monthly, or 4 times per year, and magnitudes of the data are ignored in favor of the relative ranks (Smith et al. 1982).

Methodology

Data Source – Water Quality: The source of the data used in this study is the United States Geological Survey's NASQAN program; the data were available on CD-ROM (Alexander et al. 1996). While the objectives and operating procedures for the program have been modified since its inception in 1973, it is one of the most comprehensive databases for a range of stream water characteristics including physical, chemical and biological properties. Monitoring stations in this program were set up in watersheds representing diverse climatic, physiographic and cultural characteristics in order to provide the best synoptic perspective of surface freshwater conditions throughout the United States (Alexander et al. 1996). Data used in this study are a subset of the NASQAN file for the southeastern region of the United States and were collected from 41 monitoring stations. Criteria used for selection of the subset include constituents of interest, geographic area and data collection period.

Water Quality Constituents: Trend analysis was conducted on water quality trends for six constituents: dissolved oxygen as a percentage of saturation (DOSAT), pH, total nitrogen (TN), total phosphorous (TP), total suspended sediment (SED) and instantaneous stream flow (FLOW). These constituents were selected because they: 1) are important indicators of water quality and stream health, 2) reflect land use and other human influences on the landscape, 3) are among the most consistently reported data for stream water characterization, 4) are easily detected with available monitoring equipment, and 5) are intensively studied and monitored in research and management programs. They are surrogate measures of the water's capacity to sustain aquatic life

(DOSAT and pH), in situ and contributed nutrients (TN and TP), soil erosion (SED), and stream energy and volume (FLOW).

Geographic region: Data for the analysis came from 41 of the 43 NAWQA stations (Figure 3.1; Table 3.1) in the southeastern United States. Data from two stations, one each on the Congaree River at Columbia, SC and the Roanoke River near Scotland Neck, NC were not used because of short monitoring periods and inconsistent reporting. Selection of the study area was based on the following criteria: 1) large, contiguous geographic area, 2) numerous monitoring stations that had long and continuous water quality records (generally exceeding 15 years), and 3) similarity in environmental characteristics and land use patterns. The relative homogeneity of ecological characteristics within ecoregions provides a convenient spatial framework for delineation of the study area. While the USGS (1999a, 1999b) base map considers the study site as one ecoregion, other maps recognize this area as consisting of two: the Southeastern Plains Ecoregion (65) and the Piedmont Ecoregion (45) (USEPA 1997; Omernik et al. 1987). The Southeastern Plains ecoregion includes portions of nine states (AL, FL, GA, LA, MS, NC, SC, TN and VA) and occupies the area immediately to the west of the Middle Atlantic Coastal Plain and north of the Southern Coastal Plain. There is relatively little relief to the landscape and the sands, silts and clays are cretaceous or tertiary-age. The Piedmont ecoregion, defined by gently rolling hills rising to 183 meters (600 feet) and encompassing parts of five states (AL, GA, NC, SC, and VA) represents the transitional area between the Southeastern Plains and the mostly mountainous ecoregions of the Appalachians to the northwest. The Fall Line forms the boundary between the Piedmont and Southeastern Plains, and streams flowing across the Fall Line experience abrupt change in gradient marked by shoals and rapids. Much of the cultivated land in the Piedmont has reverted to forestland.

This study area is characterized as having irregular plains and a mosaic of cropland, pasture, woodland, and forest (Figure 3.2). The natural vegetation is mostly oak-hickory-pine and Southern mixed forest (USEPA 2000c). Soil types are primarily Ultisols with pockets of Alfisols, Inseptisols and Vertisols (USGS 1985). Climate of the area is characterized as humid temperate with precipitation in all seasons. Watershed size ranges from 23 to 85,000 km^2 (9 to 32,820 mi^2) and the mean area of the 41 watersheds is 12,458 km^2 (4810 mi^2). Streams in this area are generally relatively low gradient and sandy bottomed, especially toward the coastal areas.

This region has experienced major change in land use patterns in the last quarter of the 20th century, primarily reflecting conversion of rural agricultural land to urban and suburban use and changes in types of agricultural and silviculture production. Locally, poultry and hog operations have significantly affected stream water quality. Nutrient levels and fecal coliform bacteria concentrations can be very high in waterbodies downstream of sewage treatment facilities, and the hog and poultry operations, and effluent from livestock facilities poses threats of eutrophication to surface waters. Additionally, silviculture, agriculture and urban development have contributed to sediment loads especially in areas of highly erodible soils. In contrast, streams draining relatively undisturbed and forested watersheds in the study area have low-medium concentrations of fecal coliform bacteria, dissolved solids and phosphorous (USEPA 2000c).

Data Monitoring Period: Maximum monitoring period for this data subset extends from 1963 through 1995; however not all stations were active or had consistent reporting throughout the period. Most stations provide data for five of the constituents used in the study for about 20 years, beginning in the early to mid-1970's. Data for dissolved oxygen began about 10 years later. Station monitoring dates are shown in Table 3.2. For this study, monitoring period was defined as the intervening time period during which monitoring for a given constituent was recorded at least once every six months. In those instances where there were non-recording periods exceeding six months, data prior to this recording gap were discarded. Monitoring frequency varied among stations and constituents. While most stations recorded data on a regular frequency (monthly or bi-monthly) throughout the monitoring period, some stations were monitored less frequently and/or varied the frequency throughout the monitoring period. Instantaneous stream flow was the most frequently and regularly monitored constituent, while dissolved oxygen was the least monitored. In those instances where multiple values for a constituent were recorded in any month, an average monthly value was substituted.

Data structure timeframe: Monitoring period for each station was subdivided and grouped into four classes in order to identify factors affecting water quality (such as land use) and trend detection (such as the length of monitoring period). The four time classes were: 1) beginning date of consistent station monitoring to 01 January 1982, 2) 01 January 1982 – 01 January 1993, 3) 1993 to the end of record, and 4) the entire period, from beginning date of consistent monitoring period to end of record. The period 01

January 1982 - 01 January 1992 was used because it coincided with the first two complete cycles of the Natural Resource Inventory (NRI).

Data Source - Land Use/Land Cover: Land use/land cover data came from the NRI. This inventory is conducted every five years (1982, 1987, 1992, 1997, and 2002) on nonfederal lands for the entire United States by the Natural Resources Conservation Service (NRCS), an agency of the U.S. Department of Agriculture. It categorizes the landscape into twelve land use/land cover types and estimates the amount of area gained from and lost to each category and the associated standard error of estimation. Estimates are made on the basis of surveys using Primary Sampling Units (PSU) of 16 to 259 hectare square parcels (average of 65 hectares), depending on region of the country. In the Western States, the smaller PSU's were used mainly in irrigated areas and larger ones (259 ha) in relatively homogeneous areas containing large tracts of range, forest or barren land. The land use/land cover categories include: cultivated cropland, noncultivated cropland, pastureland, rangeland, forestland, other rural land (e.g., farmsteads and structures, field windbreaks, barren land, and marshland), urban and built-up land, rural transportation land (e.g., roads, railroads and associated rights-of-way), small water areas (inland bodies of water having a surface area of < 16 ha), census water (> 16 ha in size and perennial streams at least 200 m wide), Conservation Reserve Program (CRP) land, and federal land. (Federal land was included to account for complete coverage of U.S. surface area and lands transferred between federal and private entities). NRI data can be used at multiple spatial scales including national, regional, State, and sub-State levels such as USGS' 4 and 6-digit HUC's (NRCS 2001). NRCS supplied land use/land cover inventory data for each of the twenty-two 6-digit HUC's for the periods 1982 and 1992 (pers. com. D. Lund 2002) and percent change in each category was calculated.

Calculation of trends: Time series trends of water quality were calculated by the Seasonal Kendall test that was available on the software WQStat Plus (IDT 1998). The Seasonal Kendall test is a statistical procedure commonly used in the water industry to detect trends in water constituents. It "accounts for seasonality by computing the Mann-Kendall test on each of *m* seasons separately, and then combining the results" (Helsel and Hirsch 1992, p. 338). Trend analysis was performed at each station for each time period and all constituents. The software requires a minimum of 16 cumulative observations over the period of interest in order to calculate trends. Therefore to meet this criterion and to recognize distinct seasonal differences throughout the ecoregion, the calendar year was sub-divided into the four seasons and a "representative" date was established. These dates, defined as the midpoint of each season were: February 3 for winter, May 5 for spring, August 6 for summer, and November 4 for fall. Trend analysis results included number of observations, slope estimator (in units per year), summary of significance test ("yes" or "no") at the 95%, 90% and 80% confidence level, "z" value, and a time series scatter plot with the regression line. In this study, significant trends were recorded for the 95% and 90% confidence levels only.

Aggregation of water quality data: Water quality data from individual stations were aggregated in order to identify the effect of spatial scale on observed data and to examine the relationship between land use/land cover and water quality indices. Since NRI data are statistically accurate at only the 6-digit HUC and larger scale, only water quality data at the 6-digit and 4-digit level can be used. Aggregation of water data was based on group membership of individual stations to 6-digit and 4-digit HUC's. The 41 stations were aggregated into 22 6-digit HUC's and 16 4-digit HUC's. The 6-digit HUC's contained 1-3 stations, while the 4-digit HUC's contained 1-6 stations. Aggregation of the data was based on the assumption that: 1) resulting water quality trend values were representative of the measured constituents for the monitored stations at selected HUC levels, 2) data from contributing stations were equally weighted, and 3) monitored constituents reflect characteristics and changes within watersheds as well as exogenous controls such as transport and deposition of material across watersheds. Monitoring data from all the stations in a given HUC for a selected time period and constituent were combined, and mean values were substituted in those cases where multiple values for a given date resulted from this procedure. The resulting aggregations provided a basis for mapping and comparing trends at multiple spatial scales and for statistical analysis incorporating land use/land cover characteristics.

Stepwise discriminant analysis incorporating NRI data: A statistical analysis based on group membership was used to examine the relationship between changes in water quality and land use/land cover at the 6-digit HUC level. While multivariate regression can be a more robust statistical technique in analyzing relationship between variables, (stepwise) discriminant analysis is justified when describing categorical variables (up trend, down trend, no trend) and as a method for selecting variables that contribute to group differences. Two assumptions were made: changes in the landscape were reflected in specific water constituents, and these impacts were expressed within a relatively short time after disturbance, notwithstanding delayed effects. Data only for comparable time periods were included; therefore, this analysis was made only for the 10 - year inclusive period, 1982 to 1992 during which NRI data were available.

The percentage change for each of the twelve-land use/land cover categories of the NRI data was included for this procedure. A preliminary stepwise discriminate analysis was used to assess potential contribution of the land use/land cover changes to the categorization of 6-digit HUC's into three water quality trends (upward, downward, no trend) for each constituent. In the "forward stepwise" procedure used in this study, a model of discrimination is built step-by-step and all variables are reviewed and evaluated to determine which ones contribute most to the discrimination between groups.

Results and Discussion

Comparison of water quality trends at stations: Trends in water quality differed across length of monitoring period, constituent and geography. The number of stations indicating an upward trend (U), downward trend (D), "no trend" (S) and inadequate data (?) was noted for the 1982 – 1992 and for the entire period of station operation at the 95% and 90% confidence levels (Table 3.3). Results show there were significant trends in all constituents during both time periods and at both significant levels. While the most frequent result was no trend (75% for cumulative total of all constituents, 1982 – 1992 and 95% confidence), 20% of the total combined responses for all constituents reported upward trends for the entire monitoring period. As expected, number of "no trends" decreased as length of monitoring period increased (to entire period) and significance level decreased (less conservative criteria). More than 25% of the stations reported increased values for total nitrogen, total phosphorous and pH for the entire monitoring period at the 90% confidence level. The fewest number of stations (7 downward, 2 upward; = 22%) reported trends for dissolved oxygen while the largest number (22 upward, 2 downward; = 59%) reported a trend for pH. One possible factor for the relatively low incidence of reported trends for dissolved oxygen is the short monitoring period; only 4 stations (10%) reported data prior to late 1982.

There were considerable differences in detected trends across constituents. For example, while 33 stations (80%) reported no trend for total nitrogen and phosphorous, only 24 stations (59%) reported no trend for pH and 13 stations (32%) reported an upward trend for this constituent during the 1982 - 1992 period at the 95% confidence level. pH showed the most consistent pattern of trends of the six constituents; 50% of the increases measured during 1982 - 1992 and 40% during the entire monitoring period were attributable to this constituent. As shown in Table 3.4 sediment was the constituent most frequently showing a downward trend (44%) during the 1982 - 1992 period while flow was responsible for 37% of the downward trends reported during the entire monitoring period.

Water quality trends were mapped to show the geography of changes over two periods: 1982 - 1992 and for the entire period of station operation. Figures 3.3 to 3.8 show the trends for the individual stations. While no one station or group of stations demonstrates significant trends in all measured constituents, the Scape Ore Swamp near Bishopville, South Carolina reported the most activity; significant trends were reported for all constituents except nitrogen for the entire monitoring period. Examining trends by constituent, geographic patterns emerge. Two stations each in VA and SC show a decrease in dissolved oxygen, while the only station showing an increase is located in the Gulf coastal area of LA. One station each in southern Virginia and central South

Carolina (Scape Ore Swamp) show a decrease in pH in contrast to the broad scale trend of increasing values throughout the study site. The majority of stations indicating a trend (upward) in sediment are located in the southern-most reaches of the study site and the trends are variable; in contrast all the VA stations reported a downward trend. Phosphorous and nitrogen show similar upward trends over the entire monitoring period and the majority of stations reporting these trends are located in the southern reaches. All 11 stations reporting a downward trend in streamflow likewise are located in this same region.

Comparison of water quality trends at 6-digit HUC's: Trend analysis of the station data aggregated to the 6-digit HUC level revealed a pattern similar to station data. Data at this aggregated level represent mean values for the stations included in the HUC. While aggregation of stream data introduces some variability as a function of real differences in stream characteristics and measurement factors (for example, sampling protocol), the effect is offset by larger sampling size.

Similar to the spatial pattern at the station level, water quality trends at the 6-digit HUC (Figures 3.9 to 3.14) are most pronounced for the entire monitoring period and for constituents pH and FLOW. No trend was observed for dissolved oxygen in most of the HUC's except for a downward trend in three widely scattered HUC's (020802, 030402, and 030601) located in the northern half of the study area and upward trends in HUC's 031200 and 031800 (located in the southern portion) for the entire monitoring period. One station located within HUC 031200 had the longest data record for dissolved oxygen in the entire area – more than 22 years. In contrast, record length for most of the stations in the other HUC's was slightly more than 11 years. There was no downward trend and nearly equal numbers of upward and "no" trends reported for pH at the 6-digit HUC level for the entire monitoring period. Additionally, trends for both monitoring periods indicated these patterns were distributed throughout the study area. Twice as many HUC's (8) showed an increase in sediment in contrast to a downward trend (4), and 10 showed no trend for the entire monitoring period. There was no apparent geographic pattern of trends in sediment throughout the study area; in some regions contiguous HUC's had opposite trends, such as HUC 031700 (downward) and HUC 031602 (upward). The spatial pattern of trends for total phosphorous and nitrogen were very similar to each other, especially for the entire monitoring period. While the majority of upward and downward trends for these two constituents appeared in HUC's in the northern reaches of the ecoregion, no clear geographic pattern emerges. No HUC reported an upward trend in flow during the entire monitoring period while 55/% (12) reported a downward trend and the remaining 45% (10) reported no trend. The downward trends appear to cluster in the middle portion of the study area, describing a swath from NC through eastern Mississippi. These trends are summarized in Table 3.5.

Comparison of the range and mean values of the coefficients of variation between station and aggregated data for each variable showed no significant difference suggesting that precision is not jeopardized when the data are aggregated. While some 6-digit HUC data reflect a difference in trend from individual station data, most changes show a shift from "no trend" to a trend. This shift can be expected because of the combined effects of the mean values and larger pool of observations from which the trend is calculated. Table 3.6 compares the percentage of stations reporting no trend (S), downward trend (D), upward trend (U), or insufficient data (?) to the percentage of 6-digit HUC's reporting these trends, by variable, time period and confidence level. Generally, percentages for each trend category (up, down, none, insufficient data) at the station and HUC levels are most similar for DOSAT and pH and the least for TN, TP, SED and FLOW. Similarity in percentage values suggests that trends at these two spatial scales are comparable. In contrast changes in trend results (e.g., from no trend to upward or downward trend) when station data are aggregated probably reflect the larger pool of observations included in the analysis. These shifts in trend detection are the greatest for TN, TP, and SED (increased percentage of upward trends) and FLOW (increased percentage of downward trend).

Comparison of water quality trends at 4-digit HUC's: Trends of station data aggregated to the 4-digit HUC were mapped in order to identify patterns at this larger spatial scale. Since this aggregation collapsed the 22 6-digit HUC's into 16 4-digit HUC's, trend results are expected to be similar. The frequency of trends (upward, downward), summarized in Table 3.7, shows a pattern similar to that expressed at the station and 6-digit HUC levels. For example, more trends are detected during the entire monitoring period than the 1982-1992 period. Additionally, trends are well represented for pH (8 HUC's with upward trends), FLOW (8 HUC's with downward trends), and TN (2 HUC's with downward trends and 7 HUC's with upward trends). Spatial patterns of trends for the six constituents for the 1982 – 1992 and entire monitoring periods (Figures 3.15 to 3.20) generally reflect the patterns observed at the smaller area scales.

While the spatial patterns of trends between the 6-digit and 4-digit HUC levels are similar there are some notable exceptions. Some of the upward or downward trends observed at the 6-digit HUC level are expressed as "no trend" at the 4-digit HUC. For example, the 6-digit HUC 030101 shows an increasing pH trend for the entire monitoring period while the adjacent HUC 030102 shows no trend; aggregated data of the 4-digit level (0301) shows no overall trend for the period. This phenomenon can be expected because of the effect of increased sample size and the “averaging” effect resulting from aggregation. In a few instances aggregation produces a trend where none existed at the smaller scale. For example, there is a downward trend in nitrogen for the entire monitoring period in HUC 0316 (which includes portions of the Tombigbee River and Black Warrior River Basins in AL and MS) while neither of the two 6-digit member HUC’s (031601 and 031602) showed a trend. However, one HUC member (031601) showed a downward trend during the 1982-1992 period, and the change in trend observed at the 4-digit HUC level may reflect the strong influences of this downward trend during the short period and the larger database from which the trends were calculated.

Comparison of trend direction at three spatial scales: Comparison of trends at each spatial scale (stations, 6-digit HUC, and 4-digit-HUC) shows that results are generally consistent throughout the range of scales for all constituents at both monitoring periods except DOSAT (Table 3.8). The overall pattern of DOSAT became less clearly defined as the spatial scale increased. Additionally, a clearly defined geographic pattern of trends for SED could not be established for any of three spatial scales. This may suggest that data for sediment is insufficient for pattern recognition or this variable responds in a more random manner.

Descriptive statistics of land use/land cover status and changes: Like many regions in the United States the 510,748 km^2 (197,200 mi^2) of the study area has experienced noticeable changes in land use/land cover over the past twenty-five years to accommodate dynamic economic and social needs. More than 55% of the study area is forestland and the amount remained nearly constant between the 1982 and 1992 NRI censuses. In contrast, while almost 16% of the total study area was in agricultural use (crops and animal production) in 1982, this amount declined to 12.3% by 1992, representing a 26% decrease. A summary of the estimated area for each of the twenty-two 6-digit HUC's for 1982 and 1992 is shown in Table 3.9 and Figure 3.21.

The original 12 categories of land use/land cover of the NRI data were collapsed into 5 categories in order to simplify analysis: crops, pasture, forest, urban and other. The analysis showed that almost 3, 645,000 hectares (9,000,000 acres) or 7.1% of the total study area changed land use/land cover and six 6-digit HUC's recorded changes of more than 10% of the land area between the 1982 and 1992 census (Table 3.10). Overall there was a trend toward retiring cropland and expanding urban areas. All HUC's in the study area showed an increase in urban growth and a decrease in cropland. The urban category showed the largest amount of change, increasing an average of 35%. There was more than a 50% increase in urban area in four of the 6-digit HUC's (030102, 030201, 030202, and 030701). The first three HUC's encompasses the northeastern region of North Carolina including the Tar-Pamlico and Neuse River basins; HUC 030701 encompasses the Oconee-Ocmulgee River basin of central Georgia. The area in cropland decreased the most, showing an average loss of 26%. Eight HUC's (030601, 031401, 031502, 031601, 031602, 031700, 031800, and 080702) showed a decrease by 30% or more, with two HUC's (031602 in southwestern AL and 0318000 and south central MS) showing a decrease of more than 50% (Table 3.11). Some of the area lost to cropland reverted to forestland; this phenomenon is particularly prominent in the southern coastal regions of the study area, including portions of AL, LA, and MS. Overall there was a small decrease in pasture acreage (2%) despite increases of 20% in two HUC's (030401 and 031401). There was almost no change (-0.31%) in forest cover; however one HUC (080902) located in the coastal area of Mississippi lost 12% forest cover. The "other" category showed an overall mean increase of 17%. This category included: rural transportation land, small and census water areas, federal land, other rural land, and areas put under the Conservation Reserve Program (CRP); CRP land represents about 85% of total acreage in this category. CRP is a conservation and price-support program of the U.S. Department of Agriculture that promotes agricultural land retirement and/or intensive conservation management practices. This study made no distinction between land retired from production and other uses in this category.

Trends in pH and land use/land cover changes: Some spatial patterns emerge when rates of urbanization, cropland loss and forest conversion are mapped against 6-digit HUC's that show significant trends in pH. When the aggregated NRI data were used, five of the eight HUC's with significant pH increases occur in areas experiencing the largest increase in forest cover (Figure 3.22). In contrast, there was no discernable pattern between significant trends in pH and amount of cropland (loss) and urbanization (gain) (Figures 3.23; 3.24).

Stepwise discriminant analysis: This statistical procedure was used to identify the the association between land use/land cover and trends in water quality. pH was the only constituent that had sufficient representation of more than one trend "type" to satisfy criteria for discriminant analysis. Additionally, pH trends exhibited only two member groups (upward and "no trend"); no downward trends were reported for the 6-digit

HUC's in the study area. The resulting Model_1 ($\alpha = 0.1$) consists of 11 variables and Model_2 ($\alpha = 0.05$) has 10 variables (Table 3.12). Six variables are common to both models: Crop1 → Rural; Crop2 → Small water; Forest → Forest; Transport → Pasture; Transport → Urban; and Federal → Federal. The analysis identified change from rural to forest as the variable having the greatest influence in discriminating between groups for Model_1. In contrast, the more conservative Model_2 identified change from both cultivated and uncultivated cropland to pasture land as the most significant variables.

Rates of change of the most important variables in Models 1 (rural to forest) and Model 2 (cultivated cropland to pastureland and noncultivated cropland) were mapped by 6-digit HUC . There was no apparent spatial relationship between HUC's with significant pH trends and percentage conversion from cultivated and noncultivated cropland to pastureland (Figures 3.25; 3.26). In contrast, there appears to be an association between the percentage conversions of minor land to forestland. All eight HUC's with significant pH trends are located in regions where the conversion is less than 15% (Figure 3.27). Additionally, 77% (17 out of 22) of the HUC's are represented by this lower conversion rate.

Conclusion

Water is an essential resource for all ecosystems, and its quality is intricately linked to the health and integrity of these systems and human welfare (Naiman et al. 1995). While characteristics of water naturally vary spatially and temporally, human activities (Gove et al. 2001; Basnyat et al. 2000a,b; Basnyat et al. 1999) and natural events (e.g., storms, droughts) (Bis et al. 2000) impact water quality over a wide range of spatial and temporal scales. The extent of these impacts varies with the characteristics of the landscape, type of stressors placed on the water system, and ability of the streams to absorb and assimilate these changes. This variability in characteristics and responses to stressors presents unique challenges to monitoring and management efforts. Identification of trends is an integral part of monitoring and can provide the basis for policy initiatives for water and landscape management.

This study examined water quality trends over a large contiguous area of the southeastern United States at three spatial scales and for two time periods, and it explored the relationship between land use/land cover changes and these trends. While numerous other studies have contributed to understanding temporal changes in stream water quality, this study examines trends from a geographic perspective at multiple scales. The results of this approach are important to an understanding of: 1) spatial variability of trends, 2) impacts of landscape level changes on water quality, 3) adequacy of current monitoring system design, and 4) utility of archived data to explain trends and to predict future conditions based on external factors, such as land use and management activities. It is also well recognized that scale greatly affects interpretation of results, and that use of multiple scale can be helpful in minimizing artifacts and biases (Grove et al. 2001; Loftis et al. 1991; Magnuson 1990).

Data used in the analysis came from only those monitoring stations located within the Southeastern Plains and Piedmont ecoregions. This criterion provided a means to compare water quality trends within a relatively homogeneous region, thereby discounting the effects of inter-station differences on trends resulting from disparate environmental characteristics of the landscape. Geographic patterns of trends therefore reflect, in part, impacts of external factors at the landscape level, such as changes in land use/land cover. The following summarizes the observations and conclusions of this study

1. Overall assessment: Analysis of water quality at the individual station level revealed that with one exception, no one station showed significant changes in all or most of the constituents measured. The large number of "no trends" for most constituents suggests that no one part of the region experienced significant declines or improvements in water quality as measured by the constituents.

2. Geographic patterns of trends by constituents: Trends of some constituents show distinct geographic patterns while for others the pattern is less well defined. The forty percent of the stations reporting increased pH levels (for the entire monitoring period) were located throughout the study area indicating the geographically widespread pattern of this phenomenon. Identification in this study of a region-wide pattern of increasing pH levels seems to differ from other assessments that note increased acidity in streams as a direct result of coal mining activities (Rohn et al. 2002). The apparent contradiction may be a function of spatial scale of observation and/or length or monitoring time on which the assessments were based. Stations reporting a downward trend of stream flow are less widespread and limited to the southern half of the study area. In contrast, geographic patterns of consistent trends are less clear for sediment, nitrogen and phosphorous. It was not possible to determine if the "no trend" pattern of dissolved oxygen throughout the region was a function of short monitoring period or a true reflection of this water quality metric. In comparison, Harned and Davenport (1991) noted upward trends of DOSAT for major rivers flowing into the Albemarle-Pamlico estuarine system in VA and NC for the period 1945-1988. While this riverine system is

not included in the study area, the increasing DOSAT values are a departure from the downward trends and no trends identified for stations in adjacent areas and located within the study area.

3. Lack of observable trends: The majority of stations report “no trend” for many of the constituents. Failure to observe more trends may be a function of: 1) absence of stressors; 2) capacity of streams to absorb changes thus maintaining homeostasis; and 3) inability of monitoring design to detect trends. Comparison of analysis between the 1982-1992 monitoring and entire monitoring period and two significance levels suggest that monitoring design is critical in the ability to detect changes and trends. Trend detection generally increased with longer monitoring period and less rigid significance criteria.

4. Effects of spatial scaling on trend detection: Scaling up is a synthesizing of data from study units or smaller areas to provide information on status and trends over larger spatial scales, such as water basins and other regional units within areas of interest. The value of monitoring at large scales is well recognized and an increasing number of studies are incorporating this element in project design (Arnell 1999; Cruise et al. 1999; Srinivasan et al. 1998).

Water quality trends in this study were scaled up from the station level to the 6-digit and 4-digit HUC’s. Results suggested that the trends were similar to those at the smaller scale (e.g., station). Scaling up has the advantage of providing a more “accurate” picture of water quality in the measured streams because it reduces variability arising from inherent spatial and temporal differences in streams and measurement error. Results calculated by scaling up to larger spatial areas indicate only the average of trends in the measured streams. It is recognized that it is not valid to conclude that the results statistically represent water quality trends in the entire HUC because this study used archival data and therefore did not control sampling design. While the type of data aggregation used in this study cannot provide robust statistical evidence of overall trends in any HUC, it provides the best available indication of trends in the measured streams and may suggest processes ongoing at a larger scale. Additionally, since the relationship between water quality and land use/land cover could only be examined at the 6-digit HUC level, trend data only at this larger spatial scale can be used in the analysis.

5. Land use/land cover changes: Almost eight percent of the total land area changed its land use/land cover classification during the 1982-1992 period. The 3.5% yearly increase of urban areas represented the largest change of the five categories used in the aggregated values: cropland, pastureland, forestland, urban and other. Cultivated and noncultivated cropland decreased by 2.6% yearly, pastureland showed a minor loss of acreage, and forestland remained constant. During this period, CRP retired from production or initiated conservation-oriented intensive management on almost 688,500 hectares (1.7 million acres) or 1.3% of the total land area of the ecoregion. This represents about 18% more land than was converted to urban uses.

6. Land use/land cover change on measurable changes in water quality metrics: There was insufficient group membership for all constituents except pH for discriminant analysis to be applied as a statistical test of the relationship between water quality trends and land use/land cover changes. When applied to increased pH levels, variables reflecting less intensive land use activities were identified, including conversion of cropland to pastureland and rural, non-agricultural use to forest cover.

Lack of group membership for the other constituents may be a function of the short (10 year) monitoring period. Trend analysis of the constituents at most stations showed "no" trend or insufficient data for this period. It is suggested that use of longer monitoring period would substantially increase the likelihood of satisfying criteria for applying discriminant analysis or other statistical test of inter-group differences.

7. Monitoring program – values and needs: The source of the data used in this study provides a history of water quality in representative streams and rivers throughout the United States. The NAWQA program was originally designed for compliance initiatives. In contrast this project used the data for a retrospective study of water quality. The differences in goals highlight areas that require consideration of water quality monitoring design.

Water quality monitoring efforts need to accommodate research and management needs as well as the mandated compliance scope of the efforts. To satisfy these other needs, the design must be revised to reflect criteria for robust statistical operations. Incomplete and inadequate water quality data can jeopardize the effectiveness of management options, including Best Management Practices (Lapp et al. 1998). Data must be recorded at greater frequency, on a continuous basis (without episodes of discontinuity), in a spatial pattern that statistically represents effects of landscape changes, and for longer periods of time.

Adequacy of length of monitoring period can be easily calculated by power analysis. While numerous power analysis programs are available, results of this statistical procedure are seldom reported in literature. Therefore, failure to identify relationships between land use/land cover changes and water quality may result from inadequate monitoring period and number of samples.

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Table 3.1. List of monitoring stations used in this study, grouped by state. Information includes name of river, station and 6-digit HUC identification numbers, latitude and longitude and drainage area.

	Station Attributes						
SL[2] #	State	Name	STAID[3]	HUC[4]	Latitude	Longitude	Drainage area (km^2)
1	AL	Alabama River at Claiborne	02429500	031502	31°32'48"	77°28'32"	56,895
2	AL	Alabama River near Montgomery	02420000	031502	32°24'41"	76°53'55"	39,075
3	AL	Black Warrior River below Selden Dam near Eutaw	02466031	031601	32°46'38"	81°03'00"	15,048
4	AL	Blackwater River near Bradley	02369800	031401	31°01'39"	85°06'37"	228
5	AL	Chattahoochee River at Andrews L&D near Columbia	02343801	031300	31°15'33"	87°14'03"	21,264
6	AL	Tombigbee River Bl Coffeeville L&D near Coffeeville	02469762	031602	31°45'25"	88°15'08"	47,700
7	AL	Tombigbee River at Gainesville	02449000	031601	32°49'30"	90°10'10"	22,357

[2] SL # =Serial number
[3] STAID =Station identification number
[4] HUC = 6-digit hydrologic unit code

	Station Attributes						
SL[2] #	**State**	**Name**	**STAID[3]**	**HUC[4]**	**Latitude**	**Longitude**	**Drainage area (km^2)**
8	**FL**	**Apalachicola River at Chattahoochee**	**02358000**	**031300**	**30°42’03”**	**79°05’23”**	**44,548**
9	**FL**	**Escambia River near Century**	**02375500**	**031403**	**30°57’54”**	**89°53’50”**	**9,886**
10	**FL**	**Ochlockonee River near Havana**	**02329000**	**031200**	**30°33’14”**	**84°20’06”**	**2,953**
11	**FL**	**Perdido River at Barrineau Park**	**02376500**	**031401**	**30°41’25”**	**77°38’04”**	**1,020**
12	**GA**	**Falling Creek near Juliette**	**02212600**	**030701**	**33°05’59”**	**77°09’48”**	**186**
13	**GA**	**Flint River at Newton**	**02353000**	**031300**	**31°18’34”**	**78°05’10”**	**14,867**
14	**LA**	**Bogue Chitto near Bush**	**02492000**	**031800**	**30°37’45”**	**86°42’36”**	**3,142**
15	**LA**	**Pearl River near Bogalusa**	**02489500**	**031800**	**30°47’35”**	**89°49’15”**	**17,024**
16	**LA**	**Tangipahoa River at Robert**	**07375500**	**080702**	**30°30’23”**	**81°37’43”**	**1,673**
17	**LA**	**Tchefuncta River near Covington**	**07375050**	**080902**	**30°29’40”**	**88°09’24”**	**376**
18	**MS**	**Cypress Creek near Janice**	**02479155**	**031700**	**31°01’30”**	**87°50’29”**	**137**

	Station Attributes						
SL[2] #	State	Name	STAID[3]	HUC[4]	Latitude	Longitude	Drainage area (km^2)
19	MS	Escatawpa River near Agricola	02479560	031403	30°48’32”	77°19’57”	1,362
20	MS	Pascagoula River near Benndale	02479020	031700	30°52’42”	88°27’41”	17,301
21	NC	Contentnea Creek at Hookerton	02091500	030202	35°25’40”	87°30’45”	1,898
22	NC	Neuse River at Kinston	02089500	030202	35°15’29”	83°43’25”	6,972
23	NC	Pee Dee River near Rockingham	02129000	030402	34°56’46”	77°31’05”	17,775
24	NC	Roanoke River at Roanoke Rapids	02080500	030101	36°27’37”	77°32’00”	21,720
25	NC	Rocky River near Norwood	02126000	030401	35°08’54”	79°52’11”	3,553
26	NC	Tar River at Tarboro	02083500	030201	35°53’38”	87°26’25”	5,654
27	NC	Yadkin River at Yadkin College	02116500	030401	35°51’24”	80°23’10”	5905
28	SC	Lynches River at Effingham	02132000	030402	34°03’05”	77°32’27”	2,668
29	SC	Scape Ore Swamp near Bishopville	02135300	030402	34°09’02”	80°18’18”	249

	Station Attributes						
SL[2] #	State	Name	STAID[3]	HUC[4]	Latitude	Longitude	Drainage area (km^2)
30	SC	Upper Three Runs near New Ellenton	02197300	030601	33°22’14”	90°21’42”	256
31	TN	Tennessee River at Pickwick Landing Dam (Ll)	03593005	060400	35°03’54”	88°07’30”	85,004
32	VA	Appomattox River at Matoaca	02041650	020802	37°13’30”	77°34’54”	3,481
33	VA	Blackwater River near Franklin	02049500	030102	36°45’45”	84°51’33”	1,598
34	VA	Dan River at Paces	02075500	030101	36°38’32”	89°01’00”	6,605
35	VA	Holiday Creek near Andersonville	02038850	020802	37°24’55”	78°38’10”	23
36	VA	James River at Cartersville	02035000	020802	37°40’15”	77°09’59”	16,206
37	VA	Mattaponi River near Beulahville	01674500	020801	37°53’16”	77°35’09”	1,557
38	VA	Meherrin River at Emporia	02052000	030102	36°41’24”	79°45’15”	1,935
39	VA	Nottoway River near Sebrell	02047000	030102	36°46’13”	84°23’03”	3,680
40	VA	Pamunkey River near Hanover	01673000	020801	37°46’03”	88°46’20”	2,800

	Station Attributes						
SL[2] #	State	Name	STAID[3]	HUC[4]	Latitude	Longitude	Drainage area (km^2)
41	VA	Rappahannock River near Fredericksburg	01668000	020801	38°19’20”	77°23’03”	4,134
						Total	510,715
						Mean	12,456
						Median	3,680
						Min	23
						Max	85,004
						Stdev	18,228

Table 3.2. Monitoring period and duration (in years) for each station and constituent analyzed in the study:DOSAT[5], pH, and TN[6]

		DOSAT		pH		TN	
SL#	STAID	Period	No. Yr	Period	No. Yr	Period	No. Yr
1	01668000	11/15/82 - 04/15/94	11.42	01/01/73 - 04/01/94	21.26	01/01/78 - 04/01/94	16.26
2	01673000	11/01/82 - 05/01/94	11.50	01/01/73 - 05/01/94	21.34	10/01/74 - 04/01/94	19.51
3	01674500	11/01/82 - 05/01/94	11.50	04/01/79 - 05/01/94	15.09	04/01/79 - 04/01/94	15.01
4	02035000	11/01/82 - 05/01/94	11.50	01/01/73 - 09/01/95	22.68	01/01/73 - 09/01/95	22.68
5	02038850	02/01/82 - 07/01/95	13.42	10/01/67 - 07/01/95	27.77	10/01/80 - 01/05/93	12.27
6	02041650	11/01/82 - 05/01/94	11.50	12/01/77 - 05/01/94	16.42	01/01/78 - 04/01/94	16.26
7	02047000	11/01/82 - 07/01/95	12.67	12/01/77 - 07/01/95	17.59	01/01/78 - 07/01/95	17.51
8	02049500	11/01/82 - 01/01/95	12.67	10/01/74 - 01/01/95	20.27	10/01/74 - 01/01/95	20.27
9	02052000	11/01/82 - 09/01/93	12.67	01/01/73 - 09/01/93	20.68	04/01/79 - 08/01/93	14.35
10	02075500	11/01/82 - 08/01/93	12.67	04/01/79 - 08/01/93	14.35	04/01/79 - 08/01/93	14.35
11	02080500	11/01/82 - 05/01/95	12.67	10/01/76 - 08/01/95	18.84	10/01/76 - 08/01/95	18.84
12	02083500	11/01/82 - 07/01/95	12.67	01/01/73 - 09/01/95	22.68	08/01/73 - 07/01/95	21.93
13	02089500	11/01/82 - 06/01/95	12.67	08/01/73 - 09/01/95	22.10	08/01/73 - 06/01/95	21.85
14	02091500	11/01/82 - 06/01/95	12.67	03/01/79 - 09/01/95	16.52	03/01/79 - 06/01/95	16.26
15	02116500	11/01/82 - 09/01/91	12.67	02/01/73 - 09/01/92	19.59	09/01/73 - 09/01/92	19.01
16	02126000	11/01/86 - 06/01/95	12.67	02/01/73 - 08/01/95	22.51	01/01/76 - 08/01/95	19.59
17	02129000	11/01/82 - 09/01/86	12.67	01/01/73 - 09/01/86	13.67	09/01/73 - 09/01/86	13.01
18	02132000	11/01/82 - 09/01/94	12.67	01/01/73 - 09/01/94	21.68	10/01/74 - 09/01/94	19.93
19	02135300	11/01/82 - 09/01/95	12.67	08/01/70 - 09/01/95	25.10	12/01/80 - 09/01/95	14.76
20	02197300	11/01/82 - 09/01/93	12.67	11/01/67 - 09/01/93	25.85	01/01/81 - 04/01/93	12.25
21	02212600	10/01/82 - 06/01/94	12.67	10/01/67 - 06/01/94	26.68	06/01/74 - 11/01/91	17.43
22	02329000	10/01/73 - 06/01/94	12.67	01/01/73 - 06/01/94	21.43	01/01/73 - 06/01/94	21.43
23	02343801	10/01/82 - 09/01/95	12.67	10/01/82 - 09/01/95	12.93	10/01/82 - 07/01/94	11.76
24	02353000	10/01/82 - 08/01/95	12.67	01/01/73 - 08/01/95	22.59	10/01/82 - 08/01/94	11.84
25	02358000	06/01/73 - 09/01/95	12.67	01/01/73 - 09/01/95	22.68	01/01/73 - 09/01/95	22.68
26	02369800	10/01/84 - 09/01/95	12.67	10/01/84 - 09/01/95	31.85	10/01/84 - 02/01/95	10.34
27	02375500	01/01/74 - 09/01/94	12.67	02/01/73 - 09/01/94	21.59	02/01/73 - 09/01/94	21.59
28	02376500	05/01/75 - 09/01/94	12.67	01/01/73 - 09/01/94	21.68	09/01/73 - 06/01/94	20.76
29	02420000	04/01/83 - 08/01/92	12.67	01/10/73 - 08/01/92	19.57	01/01/75 - 08/01/92	17.59
30	02429500	10/01/82 - 06/01/95	12.67	01/01/73 - 06/01/95	22.43	04/01/73 - 06/01/95	22.18
31	02449000	10/01/82 - 06/01/94	12.67	01/01/73 - 06/01/94	21.43	07/01/74 - 06/01/94	19.93
32	02466031	10/01/82 - 08/01/93	12.67	02/01/78 - 08/01/93	15.51	02/01/78 - 08/01/93	15.51
33	02469762	10/01/82 - 04/01/95	12.67	01/01/73 - 04/01/95	22.26	10/01/74 - 02/01/95	20.35
34	02479020	10/01/82 - 06/01/95	12.67	02/01/73 - 06/01/95	22.34	11/01/73 - 01/15/93	19.22
35	02479155	10/01/82 - 05/01/95	12.67	12/01/66 - 05/01/95	28.43	10/01/80 - 05/01/95	14.59
36	02479560	10/01/82 - 08/01/93	12.67	07/01/74 - 08/01/93	19.10	04/01/79 - 08/01/93	14.35
37	02489500	10/01/84 - 08/01/94	12.67	01/01/73 - 08/01/94	21.59	10/01/73 - 08/01/94	20.85
38	02492000	10/01/84 - 09/01/92	12.67	01/01/73 - 08/01/95	22.59	10/01/74 - 08/01/95	20.85
39	03593005	10/01/82 - 09/01/94	12.67	10/01/74 - 09/01/94	19.93	03/01/75 - 09/01/94	19.52
40	07375050	10/01/84 - 07/01/93	12.67	11/01/77 - 07/01/93	15.67	11/01/77 - 08/01/95	17.76
41	07375500	10/01/84 - 08/01/94	12.67	03/01/79 - 08/01/94	15.43	03/01/79 - 06/01/94	15.26

[5] DOSAT = Dissolved oxygen (% saturation)
[6] TN = Total nitrogen

Table 3.2. continued. Constituents: TP[7], SED[8], and FLOW[9]

	TP		SED		FLOW	
SL#	Period	No. Yr	Period	No. Yr	Period	No. Yr
1	12/01/77 - 09/01/93	15.76	12/01/77 - 09/01/93	15.76	12/01/77 - 09/01/93	15.76
2	10/01/74 - 09/01/93	18.93	10/01/74 - 09/01/93	18.93	10/01/74 - 09/01/93	18.93
3	04/01/79 - 01/01/94	14.76	04/01/79 - 01/01/94	14.76	04/01/79 - 01/01/94	14.76
4	02/01/74 - 09/01/95	21.59	03/01/74 - 08/01/95	21.43	02/01/74 - 09/01/95	21.59
5	10/01/73 - 07/01/95	21.76	08/01/74 - 04/01/95	20.68	10/01/73 - 07/01/95	21.76
6	12/01/77 - 09/01/93	15.76	12/01/77 - 09/01/93	15.76	12/01/77- 09/01/93	15.76
7	12/01/77 - 06/01/94	16.51	12/01/77 - 06/01/94	16.51	12/01/77 - 06/01/94	16.51
8	10/01/74 - 08/01/94	19.85	10/01/74 - 08/01/94	19.85	01/15/82 - 08/01/94	12.55
9	04/01/79 - 08/01/93	14.35	04/01/79 - 08/01/93	14.35	10/01/76 - 08/01/93	16.84
10	04/01/79 - 08/01/93	14.35	04/01/79 - 08/01/93	14.35	04/01/79 - 08/01/93	14.35
11	10/01/76 - 12/01/93	17.18	09/01/75 - 12/01/93	18.26	09/01/75 - 12/01/93	18.26
12	08/01/73 - 02/01/94	20.52	08/01/73 - 08/01/94	21.01	08/01/73 - 08/01/94	21.01
13	08/01/73 - 06/01/94	20.85	08/01/73 - 06/01/94	20.85	08/01/73 - 06/01/94	20.85
14	03/01/79 - 08/01/94	15.43	09/01/75 - 08/01/94	18.93	09/01/75 - 08/01/94	18.93
15	09/01/73 - 09/01/92	19.01	11/01/80 - 09/01/92	11.84	09/01/73 - 09/01/92	19.01
16	11/01/86 - 02/01/94	7.26	08/01/76 - 02/01/94	17.52	01/01/76 - 02/01/94	18.10
17	09/01/73 - 09/01/86	13.01	09/01/73 - 09/01/86	13.01	09/01/73 - 09/01/86	13.01
18	10/01/74 - 09/01/94	19.93	10/01/74 - 09/01/94	19.93	01/01/73 - 09/01/94	21.68
19	10/01/72 - 09/01/95	22.93	01/01/73 - 09/01/95	22.68	10/01/72 - 09/01/95	22.93
20	10/01/73 - 09/01/93	19.93	02/01/73 - 09/01/93	20.59	10/01/72 - 09/01/93	20.93
21	09/01/77- 02/01/94	16.43	02/01/75 - 09/01/92	23.35	07/01/64 - 02/01/94	29.61
22	01/01/73 - 06/01/94	21.43	11/01/74 - 06/01/94	19.59	01/01/73 - 06/01/94	21.43
23	10/01/82 - 05/01/94	11.59	10/01/82 - 08/01/93	10.84	10/01/82 - 05/01/94	11.59
24	04/01/73 - 08/01/94	21.35	05/01/81 - 08/01/94	13.26	03/01/73 - 08/01/94	21.43
25	01/01/73 - 09/01/95	22.68	02/01/74 - 05/01/94	20.26	01/01/73 - 09/01/95	22.68
26	10/01/84 - 09/01/95	10.92	10/01/84 - 04/01/95	10.50	10/01/72 - 09/01/95	22.93
27	02/01/73 - 09/01/94	21.59	10/01/74 - 09/01/94	19.93	02/01/73 - 09/01/94	21.59
28	06/01/73 - 09/01/94	21.27	01/01/78 - 09/01/94	16.68	01/01/73 - 09/01/94	21.68
29	09/01/74 - 08/01/92	17.93	01/01/75 - 01/01/93	18.01	01/01/73 - 01/15/93	20.05
30	01/01/73 - 06/01/95	22.43	11/01/73 - 06/01/95	21.59	02/01/73 - 06/01/95	22.34
31	01/01/75 - 10/01/93	18.76	01/01/75 - 08/01/93	18.59	01/01/73 - 10/01/93	20.76
32	03/01/78 - 08/01/93	15.43	01/01/73 - 08/01/93	20.59	03/01/78 - 08/01/93	15.43
33	10/01/74 - 02/01/95	20.35	10/01/74 - 04/01/95	20.51	02/01/73 - 04/01/95	22.18
34	11/01/73 - 06/01/95	21.59	11/01/73 - 06/01/95	21.59	10/01/73 - 06/01/95	21.68
35	08/01/73 - 05/01/95	21.76	10/01/73 - 05/01/95	21.59	10/01/73 - 05/01/95	21.59
36	04/01/79 - 08/01/93	14.35	04/01/79 - 08/01/93	14.35	07/01/74 - 09/01/95	21.18
37	10/01/76 - 06/01/87	10.67	01/01/78 - 06/01/87	9.42	10/01/76 - 06/01/87	10.67
38	10/01/76 - 06/01/87	10.67	10/01/76 - 08/01/90	13.84	10/01/76 - 08/01/90	13.84
39	10/01/74 - 09/01/94	19.93	04/01/75 - 06/01/94	19.18	10/01/74 - 09/01/94	19.93
40	04/01/78 - 10/01/83	5.50	04/01/78 - 07/01/90	12.26	04/01/78 - 07/01/90	12.26
41	03/01/79 - 04/01/83	4.09	03/01/79 - 07/01/90	11.34	04/01/78 - 07/01/90	12.26

[7] TP = Total phosphorus
[8] SED = Sediment
[9] FLOW = Instantaneous stream flow

Table 3.3. Summary table of trends for all stations. Number and percentage of total stations reporting for each constiuent by trend category. The statistics for 1982-1992 and entire monitoring period are at the 95% and 90% confidence levels. Trend categories are None, Down, Up and ?[10]

Period: 1982-1992; 95% confidence level														
	DOSAT		**pH**		**TN**		**TP**		**SED**		**FLOW**		**TOTAL**	
Trend	No.	%	No.	%	No.	%	No.	%	No.	%	No.	%	No.	%
None	35	85%	24	59%	33	80%	33	80%	26	63%	34	83%	185	75%
Down	3	7%	1	2%	4	10%	1	2%	7	17%	0	0%	16	7%
Up	0	0%	13	32%	1	2%	0	0%	3	7%	2	5%	19	8%
?	3	7%	3	7%	3	7%	7	17%	5	12%	5	12%	26	10%
TOTAL	**41**												**246**	

Entire monitoring period; 95% confidence level														
	DOSAT		**pH**		**TN**		**TP**		**SED**		**FLOW**		**TOTAL**	
Trend	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**
None	33	80%	19	46%	26	63%	30	73%	26	63%	29	71%	163	66%
Down	5	12%	2	5%	3	7%	2	5%	7	17%	11	27%	30	12%
Up	1	2%	20	49%	12	29%	9	22%	7	17%	1	2%	50	20%
?	2	5%	0	0%	0	0%	0	0%	1	2%	0	0%	3	1%
TOTAL	**41**												**246**	

[10] ? = Insufficient data to estimate trend

Table 3.3. continued.

Period: 1982-1992; 90% confidence level

	DOSAT		pH		TN		TP		SED		FLOW		TOTAL	
Trend	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**
None	32	78%	22	54%	31	76%	30	73%	24	59%	33	80%	172	70%
Down	5	12%	2	5%	5	12%	2	5%	9	22%	1	2%	24	10%
Up	1	2%	14	34%	2	5%	2	5%	3	7%	2	5%	24	10%
?	3	7%	3	7%	3	7%	7	17%	5	12%	5	12%	26	11%
TOTAL	**41**												**246**	

Entire monitoring period; 90% confidence level

	DOSAT		pH		TN		TP		SED		FLOW		TOTAL	
Trend	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**
None	30	73%	17	41%	23	56%	27	66%	22	54%	26	63%	145	59%
Down	7	17%	2	5%	4	10%	3	7%	10	24%	14	34%	40	16%
Up	2	5%	22	54%	14	34%	11	27%	8	20%	1	2%	58	24%
?	2	5%	0	0%	0	0%	0	0%	1	2%	0	0%	3	1%
TOTAL	**41**												**246**	

Table 3.4. Summary table of trends for all stations. Number and percentage of total stations reporting for each trend category by constituent. Statistics for 1982-1992 and entire monitoring period are at 95% and 90% confidence intervals.

Period: 1982-1992; 95% confidence level														
	DOSAT		**pH**		**TN**		**TP**		**SED**		**FLOW**		**TOTAL**	
Trend	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**
None	**35**	19%	24	13%	33	18%	33	18%	26	14%	34	18%	185	75%
Down	**3**	19%	1	6%	4	25%	1	6%	7	44%	0	0%	16	7%
Up	**0**	0%	13	50%	1	4%	7	27%	3	12%	2	8%	19	8%
?	**3**	16%	3	16%	3	16%	0	0%	5	25%	5	26%	26	10%
TOTAL	**41**												**246**	
Entire monitoring period; 95% confidence level														
	DOSAT		**pH**		**TN**		**TP**		**SED**		**FLOW**		**TOTAL**	
Trend	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**
None	33	20%	19	12%	26	16%	30	18%	26	16%	29	18%	163	66%
Down	5	17%	2	7%	3	10%	2	7%	7	23%	11	37%	30	12%
Up	1	2%	20	40%	12	24%	9	18%	7	14%	1	2%	50	20%
?	2	67%	0	0%	0	0%	0	0%	1	33%	0	0%	3	1%
TOTAL	41												**246**	

Table 3.4 continued.

Period: 1982-1992; 90% confidence level														
	DOSAT		**pH**		**TN**		**TP**		**SED**		**FLOW**		**TOTAL**	
Trend	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**
None	32	19%	22	13%	31	18%	30	17%	24	14%	33	19%	172	70%
Down	5	21%	2	8%	5	21%	2	8%	9	38%	1	4%	24	10%
Up	1	4%	14	58%	2	8%	2	8%	3	13%	2	8%	24	10%
?	3	12%	3	12%	3	12%	7	27%	5	19%	5	19%	26	11%
TOTAL	41												**246**	
Entire monitoring period; 90% confidence level														
	DOSAT		**pH**		**TN**		**TP**		**SED**		**FLOW**		**TOTAL**	
Trend	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**	**No.**	**%**
None	30	21%	17	12%	23	16%	27	19%	22	15%	26	18%	145	59%
Down	7	18%	2	5%	4	10%	3	8%	10	25%	14	35%	40	16%
Up	2	3%	22	38%	14	24%	11	19%	8	14%	1	2%	58	24%
?	2	67%	0	0%	0	0%	0	0%	1	33%	0	0%	3	1%
TOTAL	41												**246**	

Table 3.5. Summary of 6-digit HUC trends. This table shows the number of 6-digit HUC's that reported downward (D) or upward trend (U) for the 1982-1992 and entire monitoring periods at the 95%confidnece level. The number of HUC's showing no trend or insufficient data are not reported.

Constituent	1982-1992		Entire period	
	D	U	D	U
DOSAT	3	0	3	2
pH	0	7	0	12
TN	1	2	1	90
TP	0	1	0	9
SED	1	2	4	8
FLOW	1	0	12	0

Table 3.6. Comparison of the percentage of stations in each trend category with the percentage of 6-digit HUC's for each constituent for 1982-1992 and entire monitoring period. Results shown for 95% and 90% confidence levels.

Period: 1982-1992; 95% confidence level												
	DOSAT		pH		TN		TP		SED		FLOW	
Trend	Station	HUC	Station	HUC	Station	HUC	Station	HUC	Station	HUC	Station	HUC
None	85%	82%	59%	64%	80%	82%	80%	77%	63%	77%	83%	86%
Down	7%	14%	2%	0%	10%	14%	2%	0%	17%	5%	0%	5%
Up	0%	0%	32%	32%	2%	0%	0%	5%	7%	9%	5%	0%
?	7%	5%	7%	5%	7%	5%	17%	18%	12%	9%	12%	9%
Entire monitoring period; 95% confidence level												
	DOSAT		pH		TN		TP		SED		FLOW	
Trend	Station	HUC	Station	HUC	Station	HUC	Station	HUC	Station	HUC	Station	HUC
None	80%	77%	46%	45%	63%	50%	73%	64%	63%	45%	71%	45%
Down	14%	14%	5%	0%	7%	9%	5%	0%	17%	18%	27%	55%
Up	2%	9%	49%	55	29%	41%	22%	36%	17%	36%	2%	0%
?	5%	0%	0%	7%	0%	0%	0%	0%	2%	12%	0%	0%

Table 3.6. continued.

Period: 1982-1992; 90% confidence level												
	DOSAT		pH		TN		TP		SED		FLOW	
Trend	**Station**	**HUC**	**Station**	**HUC**	**Station**	**HUC**	**Station**	**HUC**	**Station**	**HUC**	**Station**	**HUC**
None	78%	73%	54%	59%	76%	82%	73%	68%	59%	73%	80%	82%
Down	12%	18%	5%	0%	12%	14%	5%	5%	22%	5%	2%	5%
Up	2%	5%	34%	36%	5%	0%	5%	9%	7%	14%	5%	5%
?	7%	5%	7%	5%	7%	5%	17%	18%	12%	9%	12%	9%
Entire monitoring period; 90% confidence level												
	DOSAT		pH		TN		TP		SED		FLOW	
Trend	**Station**	**HUC**	**Station**	**HUC**	**Station**	**HUC**	**Station**	**HUC**	**Station**	**HUC**	**Station**	**HUC**
None	73%	68%	41%	36%	56%	36%	66%	55%	54%	36%	63%	32%
Down	17%	23%	5%	0%	10%	14%	7%	5%	24%	23%	34%	68%
Up	5%	9%	54%	64%	34%	50%	27%	41%	20%	41%	2%	0%
?	5%	0%	0%	0%	0%	0%	0%	0%	2%	0%	0%	0%

Table 3.7. Summary of 4-digit HUC trends. This table shows the number of 4-digit HUC's that reported a downward (D) or upward (U) trend for the 1982-1992 and entire monitoring periods at the 95% confidence level. The number of HUC's showing no trend or insufficient data are not reported.

Constituent	1982-1992		Entire period	
	D	U	D	U
DOSAT	1	0	3	2
pH	0	7	0	8
TN	3	0	2	7
TP	0	0	0	5
SED	1	1	2	4
FLOW	0	0	8	0

Table 3.8. Comparison of general spatial/temporal patterns in trends at three spatial scales. This table summarizes the general trend for each constituent over the whole study area at the station, 6-digit and 4-digit HUC levels. The first entry shows the general trend for the 1982-1992 period, and the second entry the trend for the entire monitoring period

Constituent	Station	6-HUC	4-HUC
DOSAT	Up[11]/Up	Mixed[12]/Mixed	?/Mixed
pH	Up/Up	Up/Up	Up/Up
TN	Down[13]/Up	Down/up	Down/Up
TP	Up/Up	Up/Up	? [14]/Up
SED	? /?	?/?	?/Mixed
FLOW	?/Down	?/Down	?/Down

[11]UP = Upward (increasing trends)
[12] Mixed = Both upward and downward (decreasing) trends seen in the period under review
[13] Down = Downward (decreasing) trend
[14] ? = Insufficient data or too few trends detected to establish a pattern at a given spatial scale

Table 3.9. Summary of land use/ land cover of study area listing the amount of land (in 1,000ha) in each of the 12 NRI categories for 1982 and 1992 by 6-digit HUC's. Summary statistics are included. A complete description of each category is found in the text[15]. Data from the NRI program (NRCS 2001).

		CROP1		CROP2		PASTURE		RANGE		FOREST		MINOR	
SL #	**HUC**	**1982**	**1992**	**1982**	**1992**	**1982**	**1992**	**1982**	**1992**	**1982**	**1992**	**1982**	**1992**
1	**020801**	**270.2**	**236.8**	**35.3**	**41.7**	**147.2**	**155.9**	**0.0**	**0.0**	**919.0**	**909.9**	**71.3**	**72.1**
2	**020802**	174.3	99.3	75.9	110.2	232.1	236.0	0.0	0.0	1390.6	1373.2	43.9	44.8
3	**030101**	384.1	273.0	73.0	110.3	256.4	249.2	0.0	0.0	1513.1	1537.3	39.6	49.3
4	**030102**	512.6	461.5	7.3	14.1	35.0	32.7	0.0	0.0	1174.6	1092.3	70.7	74.0
5	**030201**	395.9	359.8	2.3	7.7	39.1	33.8	0.0	0.0	698.5	680.1	81.1	80.5
6	**030202**	427.0	399.1	5.3	14.2	47.3	43.9	0.0	0.0	716.6	676.1	40.6	36.5
7	**030401**	342.6	259.8	39.0	33.4	217.0	259.7	0.0	0.0	905.9	870.5	31.7	34.6
8	**030402**	873.0	780.0	20.1	24.6	75.2	77.5	0.0	0.0	1768.6	1758.9	58.1	63.6
9	**030601**	226.7	140.8	24.1	28.2	300.1	301.2	0.0	0.0	1558.1	1563.4	34.3	32.9
10	**030701**	553.4	379.7	40.6	41.1	320.0	357.8	0.0	0.0	2352.6	2313.6	61.6	56.9
11	**031200**	120.7	100.9	8.3	5.6	19.0	13.8	0.0	0.0	147.8	161.1	4.9	5.5
12	**031300**	827.5	659.3	68.9	75.3	398.6	380.0	0.0	0.0	2478.9	2517.9	66.2	52.5
13	**031401**	144.9	92.5	7.5	4.0	50.3	61.3	0.0	0.0	854.7	842.1	22.5	22.0
14	**031403**	103.2	77.9	10.3	10.2	85.0	72.2	0.0	0.0	832.7	850.1	10.0	14.6
15	**031502**	272.0	126.7	13.9	20.8	255.2	251.5	5.6	6.0	1290.6	1320.9	27.4	33.5
16	**031601**	573.2	321.4	51.6	48.4	498.4	506.5	20.8 *	16.6 *	2382.0	2467.1	78.6	67.2
17	**031602**	92.1	34.8	9.8	14.3	122.5	127.3	7.3 *	32.8 *	1177.3	1193.7	15.1	22.3
18	**031700**	183.2	88.7	21.3	16.5	324.5	340.8	0.0	0.0	1869.1	1898.5	45.0	51.0
19	**031800**	191.7	88.3	26.2	17.5	438.8	430.3	0.0	0.0	1358.2	1421.8	37.4	39.9
20	**060400**	322.4	213.4	50.0	79.3	386.5	377.9	0.0	0.0	1126.9	1141.7	36.9	35.0
21	**080702**	87.0	51.3	19.0	13.0	239.3	211.5	0.6 *	0.0	739.7	767.2	24.2	27.9
22	**080902**	4.6	5.6	1.9	0.8	32.0	28.7	0.0	0.0	132.4	117.0	180.8	181.0
	Total	7082.4	5250.8	611.4	731.2	4519.4	4549.5	5.6	6.0	27387.8	27474.6	1081.8	1097.7
	% Total	14.5%	10.8%	1.3%	1.5%	9.3%	9.3%	0.0%	0.0%	56.1%	56.3%	2.2%	2.2%
	Max	873.0	780.0	75.9	110.3	498.4	506.5	5.6	6.0	2478.9	2517.9	180.8	181.0
	Min	4.6	5.6	1.9	0.8	19.0	13.8	0.0	0.0	132.4	117.0	4.9	5.5
	Mean	321.9	238.7	27.8	33.2	205.4	206.8	0.3	0.3	1244.9	1248.8	49.9	49.9
	STDev	232.36	203.87	23.40	32.56	148.97	150.69	1.28	1.35	645.62	658.74	36.65	35.38

[15] CROP1 = cultivated cropland; CROP2 = uncultivated cropland; MINOR = other rural lands; WATER1 = small bodies of water; WATER2 = larger bodies of water; FEDERAL = Federal ownership; CRP =Conservation Reserve Program; * = questionable estimate-std. error > estimated area.

Table 3.9. continued

		URBAN		RURAL		WATER1		WATER2		FEDERAL		CRP		TOTAL
SL #	**HUC**	**1982**	**1992**	**1982**	**1992**	**1982**	**1992**	**1982**	**1992**	**1982**	**1992**	**1982**	**1992**	
1	**020801**	78.9	112.4	23.5	24.3	15.9	16.9	513.4	513.9	78.5	77.8	0.0	6.0	2153.2
2	**020802**	182.6	226.8	41.1	42.0	26.9	27.7	93.2	94.0	416.3	419.9	0.0	3.0	2676.8
3	**030101**	104.1	136.5	53.5	54.8	27.0	26.7	44.6	44.9	29.2	31.5	0.0	11.1	2524.6
4	**030102**	48.4	78.3	29.2	30.3	18.3	19.5	285.6	285.6	80.2	163.6	0.0	10.2	2262.0
5	**030201**	49.2	80.0	23.6	24.4	13.0	13.2	526.3	526.6	98.8	116.3	0.0	5.5	1927.9
6	**030202**	102.9	158.4	24.3	26.5	11.5	12.8	78.0	78.9	30.4	33.9	0.0	3.4	1483.8
7	**030401**	135.5	193.2	35.6	36.2	16.3	17.2	14.0	14.1	27.3	27.7	0.0	18.5	1764.9
8	**030402**	122.4	166.7	62.3	66.2	25.5	26.4	23.4	23.5	24.5	24.4	0.0	41.2	3053.0
9	**030601**	125.5	164.6	39.6	43.3	25.9	26.7	72.9	74.3	316.7	315.6	0.0	32.9	2723.8
10	**030701**	174.3	273.4	60.1	63.9	43.8	48.4	30.8	31.0	72.7	74.0	0.0	69.9	3709.7
11	**031200**	8.6	11.0	4.1	4.5	5.0	6.4	0.2	0.2	0.0	0.0	0.0	9.5	318.7
12	**031300**	301.4	380.8	74.3	77.6	45.2	48.6	63.1	63.5	153.1	153.6	0.0	68.0	4477.1
13	**031401**	79.1	112.0	17.4	20.1	11.0	12.3	129.8	130.2	232.8	236.5	0.0	17.0	1550.0
14	**031403**	20.5	25.9	17.5	17.7	10.0	11.1	1.2	1.2	5.1	5.2	0.0	9.8	1095.8
15	**031502**	55.4	74.6	28.8	28.9	20.8	23.6	19.8	19.9	45.2	45.6	0.0	82.0	2034.2
16	**031601**	122.1	158.6	58.0	61.1	45.0	49.2	28.5	32.1	168.6	166.7	0.0	132.0	4026.9
17	**031602**	65.7	77.0	20.0	20.5	22.1	23.8	118.5	119.3	6.9	6.9	0.0	10.2	1657.4
18	**031700**	106.9	117.6	48.6	50.5	28.2	30.3	53.3	53.7	270.3	275.8	0.0	26.7	2950.3
19	**031800**	57.5	68.2	38.3	38.2	27.2	30.7	15.6	14.9	48.0	53.6	0.0	35.4	2238.9
20	**060400**	46.7	64.8	28.1	28.7	17.7	18.3	66.7	66.7	18.1	19.2	0.0	55.0	2100.0
21	**080702**	71.7	89.0	16.8	16.9	16.5	18.5	25.1	25.1	1.3	1.3	0.0	19.3	1241.0
22	**080902**	52.3	61.8	2.5	2.2	27.9	30.1	394.5	394.5	19.9	27.2	0.0	0.0	848.9
	Total	2111.8	2831.8	747.3	778.9	500.7	538.2	2598.3	2608.2	2143.9	2276.3	0.0	666.6	48819.1
	% Total	4.3%	5.8%	1.5%	1.6%	1.0%	1.1%	5.3%	5.3%	4.4%	4.7%	0.0%	1.4%	100.0%
	Max	301.4	380.8	74.3	77.6	45.2	49.2	526.3	526.6	416.3	419.9	0.0	132.0	4477.1
	Min	8.6	11.0	2.5	2.2	5.0	6.4	0.2	0.2	0.0	0.0	0.0	0.0	318.7
	Mean	96.0	128.7	34.0	35.4	22.8	24.5	118.1	118.6	97.4	103.5	0.0	30.3	2219.0
	STDev	64.44	85.15	19.16	20.14	11.10	12.00	160.18	160.14	115.40	116.55	0.00	32.98	1017.31

Table 3.10. Summary of land use/ land cover changes. This table summarizes the changes in land use/land cover between 1982 and 1992 NRI censuses, by 6-digit HUC. Changes, expressed as percentage and absolute amounts, are based on the 1982 amounts of land in each category. The five land use /land cover categories are aggregates of the original NRI categories.

	Percentage area changed, 1982-1992							Absolute amount of area changed, 1982-1992						
SL #	HUC	Crop	Pasture	Forest	Urban	Other	Total	Crop	Pasture	Forest	Urban	Other	Total	Area (1,000 ha)
1	020801	-13.57	5.94	-0.99	42.38	-0.53	4.48	-41.43	8.75	-9.11	33.45	-3.73	93.40	2156.54
2	020802	-16.24	1.68	-1.26	24.25	1.60	4.34	-40.62	3.89	-17.50	44.27	9.96	116.24	2643.84
3	030101	-16.14	-2.80	1.60	31.12	12.53	6.41	-73.75	-7.17	24.22	32.40	24.30	161.84	2509.06
4	030102	-8.53	-6.60	-7.00	61.76	13.26	10.21	-44.35	-2.31	-82.26	29.89	99.02	257.82	2242.08
5	030201	-7.71	-13.56	-2.64	62.55	3.18	4.31	-30.70	-5.31	-18.43	30.78	23.65	108.86	1936.22
6	030202	-4.40	-7.03	-5.65	53.92	3.99	0.06	-30.70	-5.31	-18.43	30.78	7.37	92.58	1448.93
7	031401	-36.63	21.72	-1.48	41.70	5.94	8.83	-55.81	10.94	-12.64	32.97	24.54	136.89	1762.56
8	030402	-9.92	3.07	-0.55	36.28	26.58	7.02	-88.29	2.31	-9.64	44.39	69.74	214.37	3032.64
9	030601	-32.64	0.36	0.34	31.21	7.42	6.01	-81.89	1.09	5.31	39.16	36.33	163.78	2695.68
10	030701	-29.16	11.84	-1.66	56.85	27.96	11.44	-173.18	37.87	-38.96	99.10	75.17	424.28	3680.64
11	031200	-19.60	-19.67	2.00	48.71	8.21	8.79	-31.43	-9.23	9.60	13.81	17.25	81.32	946.08
12	031300	-18.76	-5.13	1.77	27.66	11.26	6.88	-186.58	-22.32	51.52	86.06	17.25	363.73	5313.60
13	031401	-36.63	21.72	-1.48	41.70	5.94	8.83	-55.81	10.94	-12.64	32.97	24.54	136.89	1570.75
14	031403	-22.65	-15.05	2.09	26.48	36.08	7.04	-25.80	-12.80	17.37	5.43	15.80	77.19	1111.97
15	031502	-48.38	-1.24	2.35	34.65	64.96	13.91	-138.27	-3.24	30.33	19.20	91.98	283.01	2060.64
16	031601	-40.81	0.74	3.57	29.87	34.18	7.03	-254.95	3.85	85.13	36.49	129.48	283.01	3991.68
17	031602	-51.83	3.68	1.39	17.13	11.15	17.08	-52.81	4.78	16.40	11.26	20.37	283.01	1684.80
18	031700	-48.53	5.03	1.57	10.00	9.62	6.73	-99.27	16.32	29.40	10.69	42.85	198.53	3136.32
19	031800	-51.43	-1.94	4.69	18.60	27.75	8.87	-112.06	-49.01	63.67	10.69	46.21	198.53	2262.82
20	060400	-21.41	-2.22	1.32	38.65	33.11	8.41	-79.74	-8.63	14.82	18.06	55.44	176.70	2076.19
21	080702	-39.31	-11.82	3.72	24.11	30.01	13.56	-41.63	-28.35	55.89	17.29	25.15	168.32	912.38
22	080902	-3.09	-10.37	-11.65	18.20	1.51	4.47	-0.20	-3.32	-15.43	9.52	9.44	37.91	1246.75

Table 3.11. Summary of land use/land cover changes, 1982-1992, expressed as percentage change from 1982 values. Calculation based on aggregation of data from 12 categories to 5. The classification "gain" refers to increased acreage and "loss" refers to decreased acreage for each of the five categories. Area expressed as 1,000 hectares

		Crop	Pasture	Forest	Urban	Other	Total		AREA
	mean	-26.24	-2.06	-0.31	35.05	17.61	7.90		2292
	max	**	**	**	**	**	17.08		5314
	min	**	**	**	**	**	0.06		912
gain	**max**	0.00	21.72	4.69	62.55	64.96	**		**
	min	0.00	0.36	0.34	10.00	1.51	**		**
loss	**max**	-51.43	-19.67	-11.65	0.00	-0.53	**		**
	min	-3.09	-1.24	-0.55	0.00	-0.53	**		**

Table 3.12. Variables identified important in discriminant analysis membership grouping for trends of pH for 6-digit HUC's, at α = 0.1 and 0.05; Pr > F cutoff set at 0.10.

Model_1 pH at α =0.10		**Model_2 pH at α = 0.05**	
VARIABLES (% CHANGE)	Pr > F	VARIABLES (% CHANGE)	Pr > F
Forest → Forest	0.0159	Transport → Urban	0.0180
Crop1 → Rural	0.0239	Crop 1 → Rural	0.0342
Transport → Urban	0.0392	Transport → Pasture	0.0427
Forest → Crop1	0.0433	Crop 2 → Pasture	0.0441
Federal → Federal	0.0441	Federal → Federal	0.0476
Transport → Transport	0.0455	Federal → Forest	0.0594
Crop2 → Small water	0.0570	Forest → Forest	0.0760
Forest → Federal	0.0716	Crop 1 → Pasture	0.0785
Pasture → CRP	0.0941	Crop 2 → Small water	0.0940
Transport → Pasture	0.0968	Forest → Pasture	0.0999
Rural → Forest	0.0971		

Crop1 = Cultivated cropland

Rural = Other rural land

Transport = Rural transportation land

Urban = Urban and built-up land

Federal = Federal land

Crop2 = non cultivated cropland

Small water = Small water areas (rivers < 200 meters wide or bodies of water < 16 ha)

CRP = Conservation Reserve Program

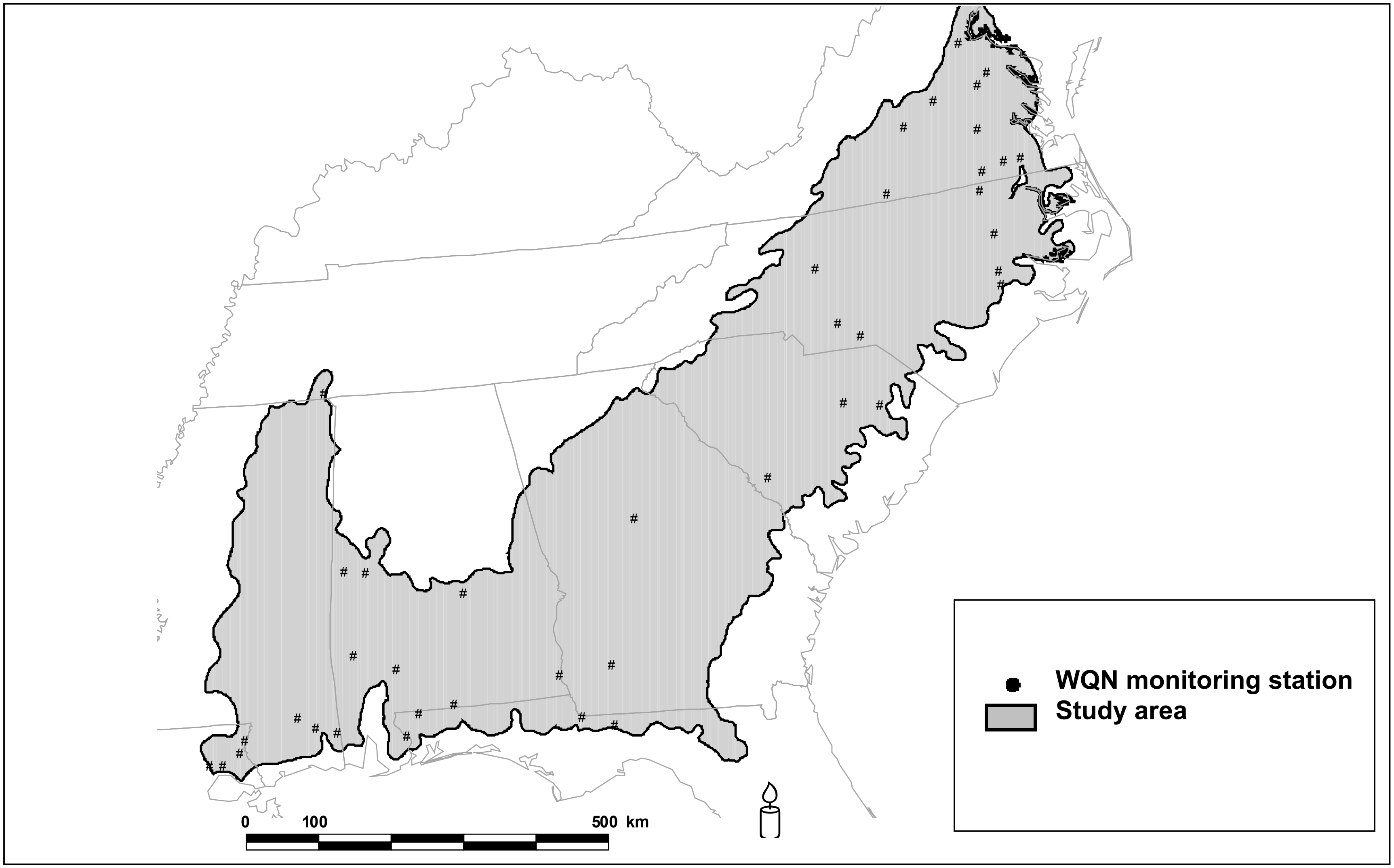


Figure 3.1. Water Quality Monitoring Network (WQN) stations in study area.

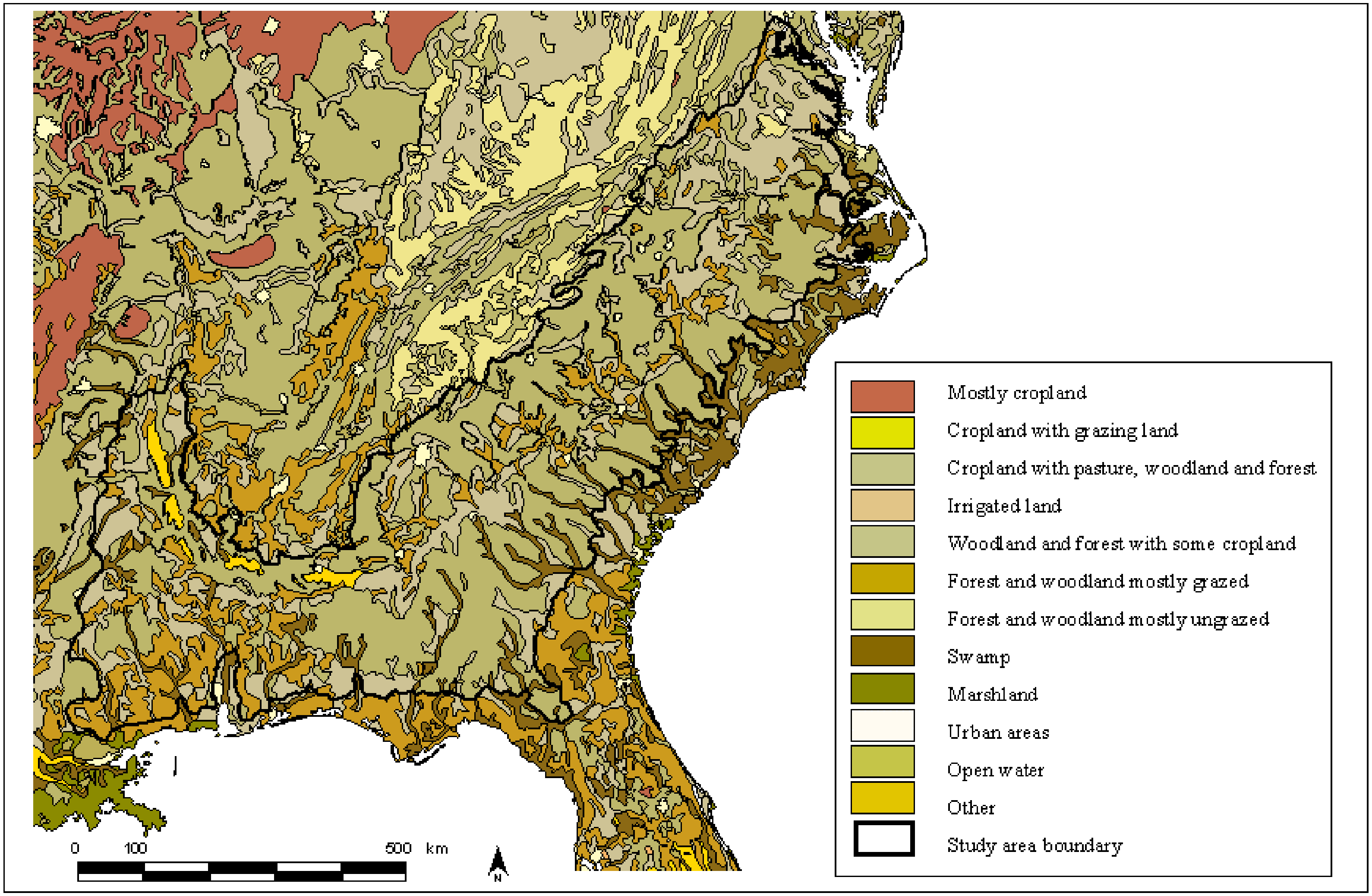


Figure 3.2. A 12-category USGS 1992 land use/land cover map of the Southeastern Plains and Piedmont ecoregions.

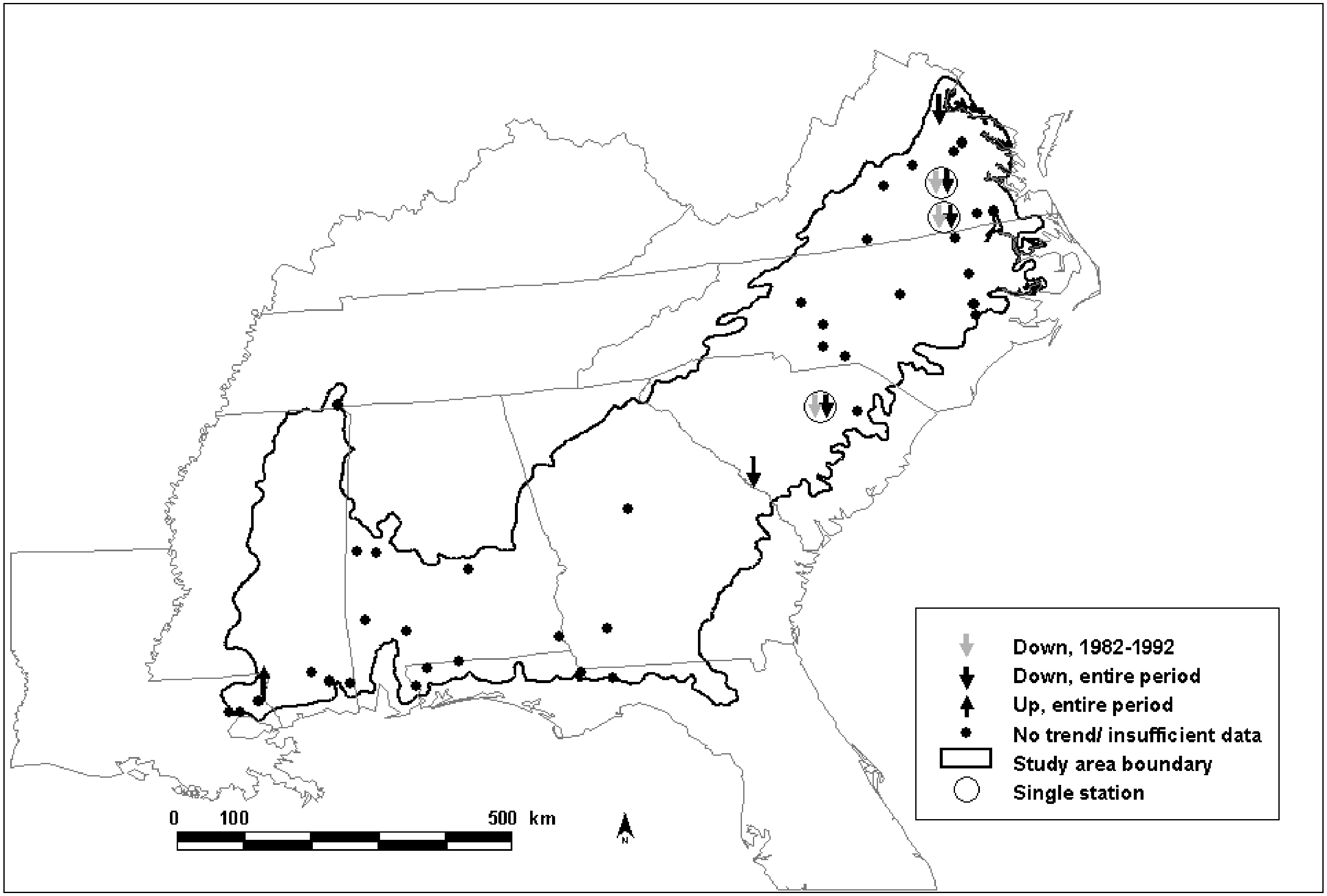


Figure 3.3. Station trends in dissolved oxygen (DOSAT) for two time periods.

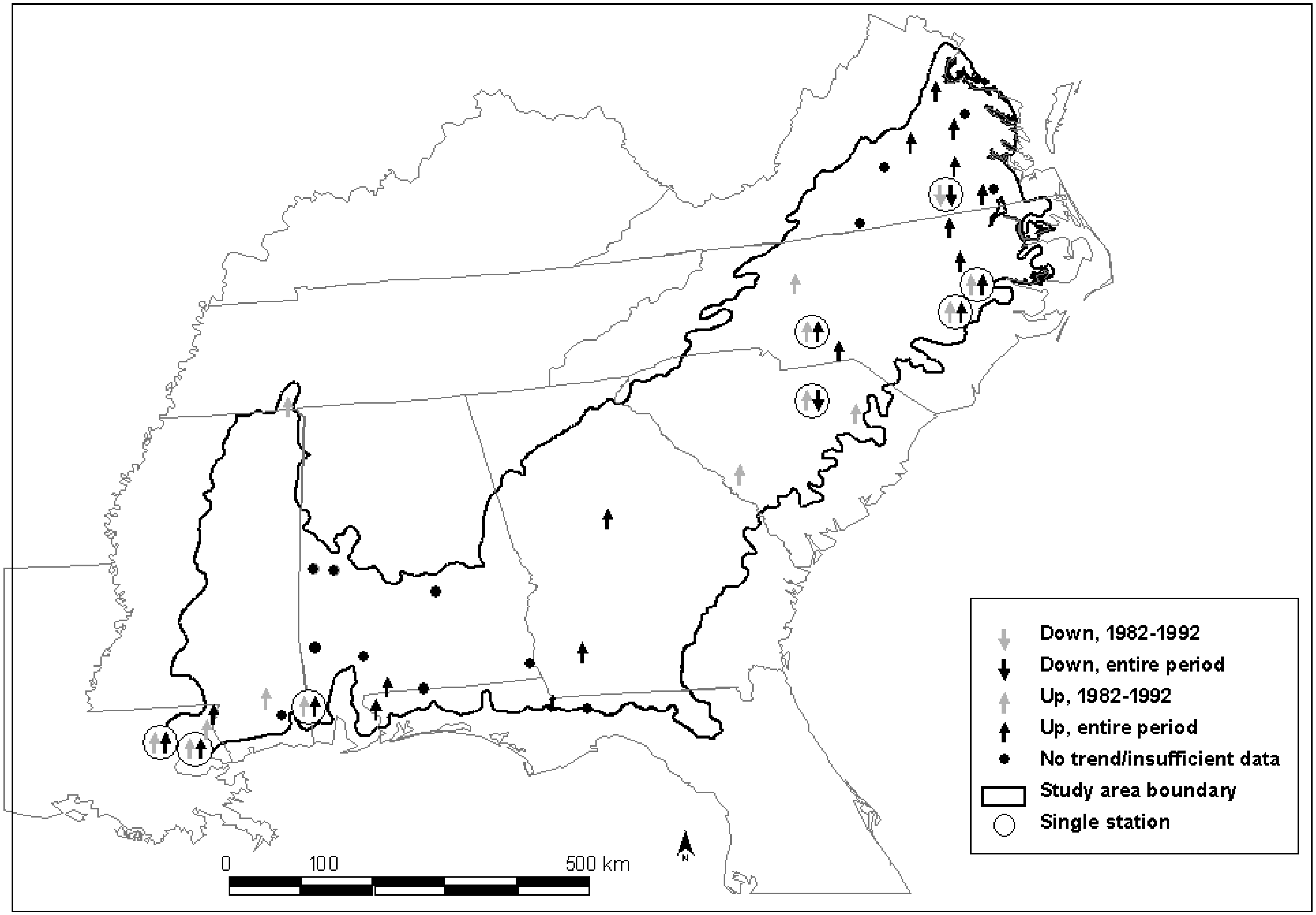


Figure 3.4 Station trends in pH for two time periods.

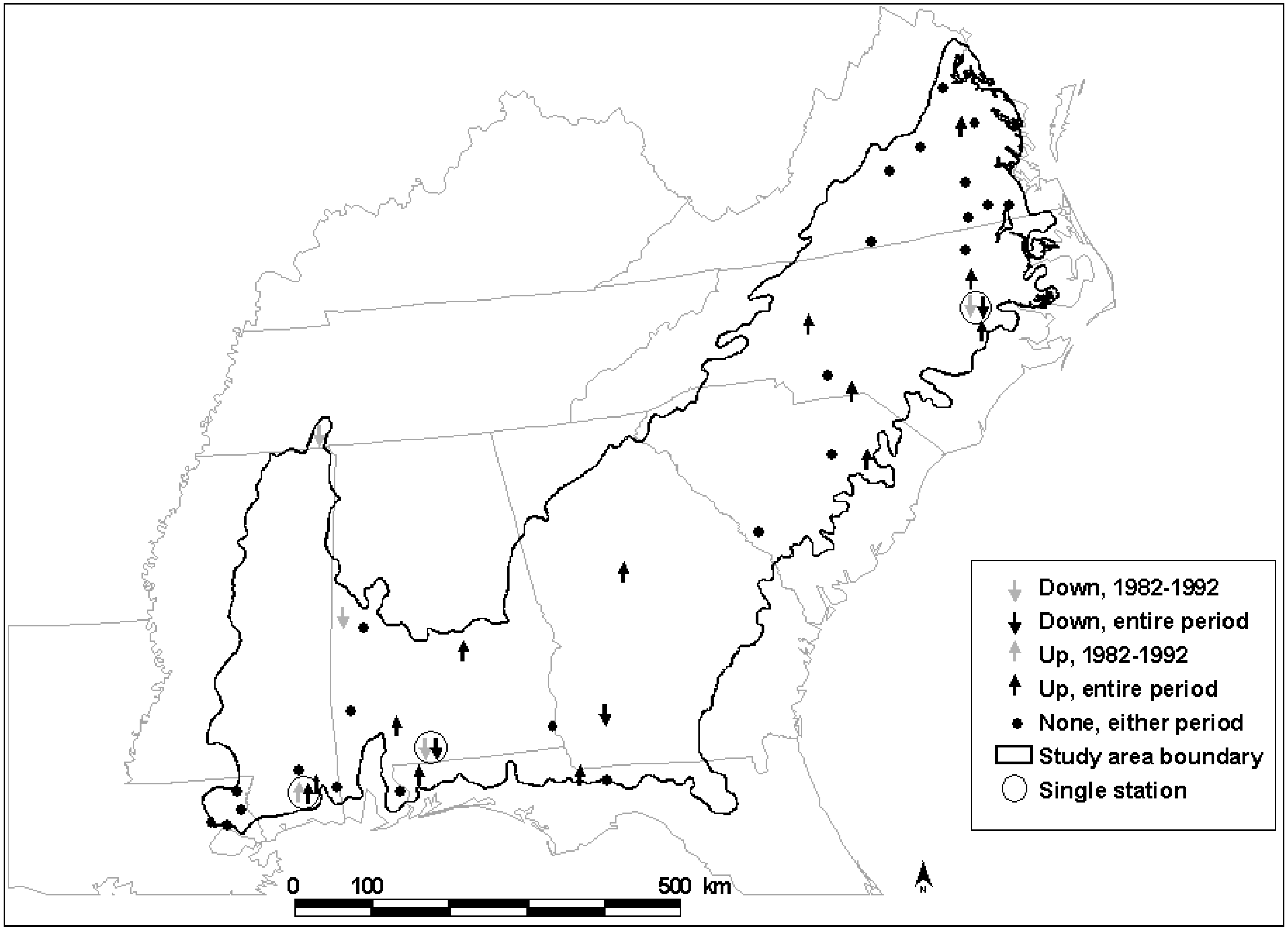


Figure 3.5. Station trends in total nitrogen (TN) for two time periods.

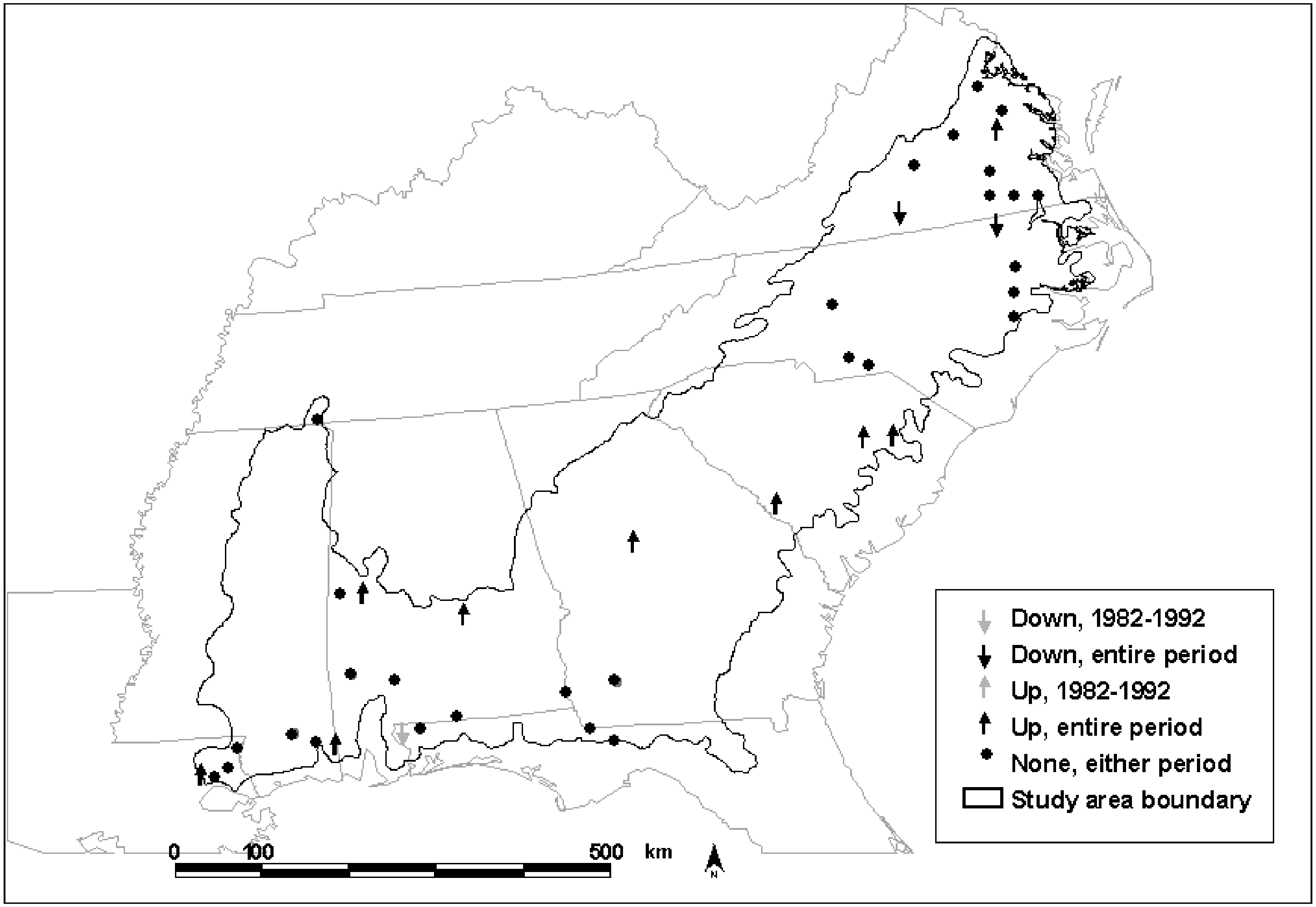


Figure 3.6. Station trends in total phosphorus (TP) for two time periods.

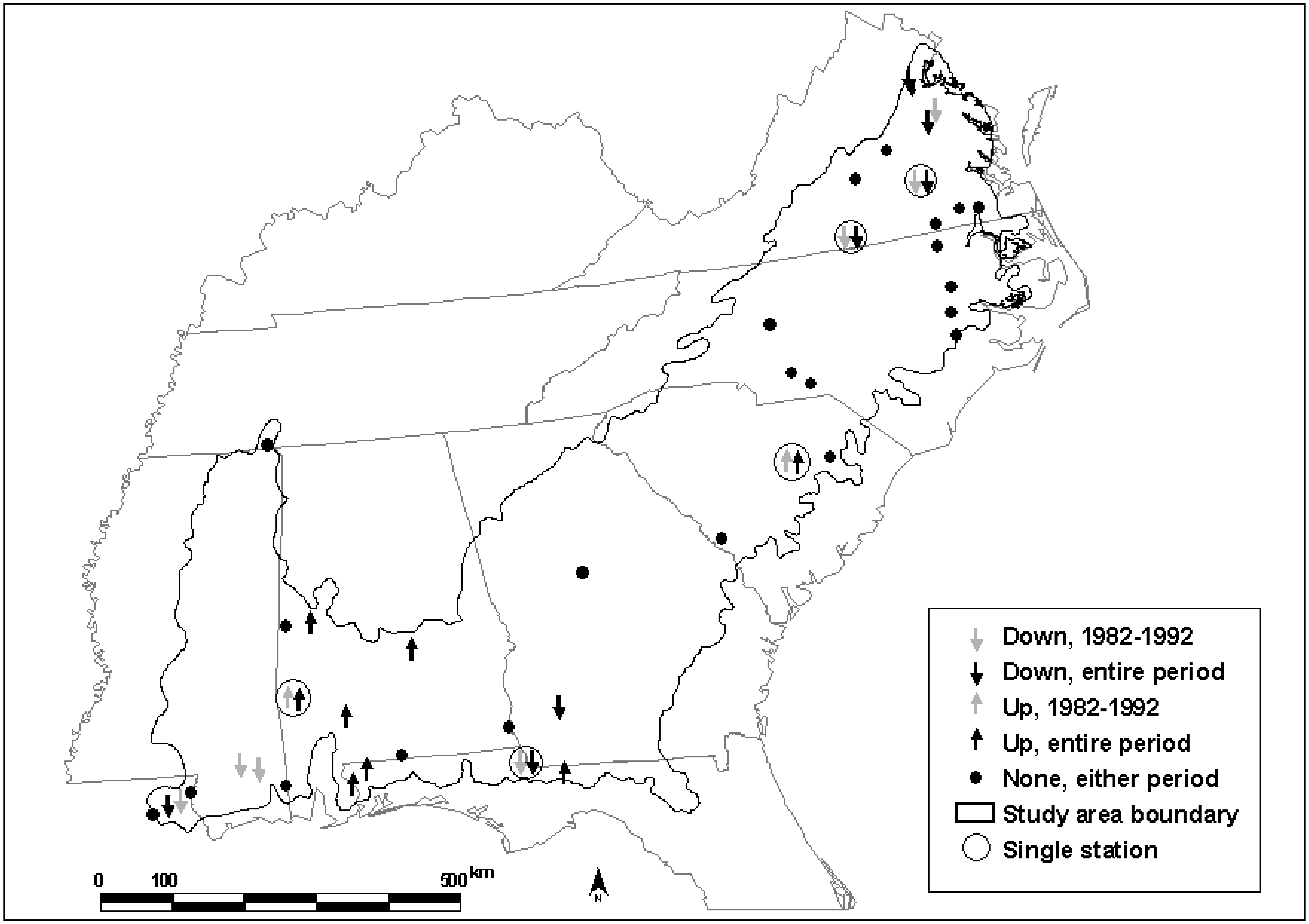


Figure 3.7. Station trends in Sediment (SED) for two time periods.

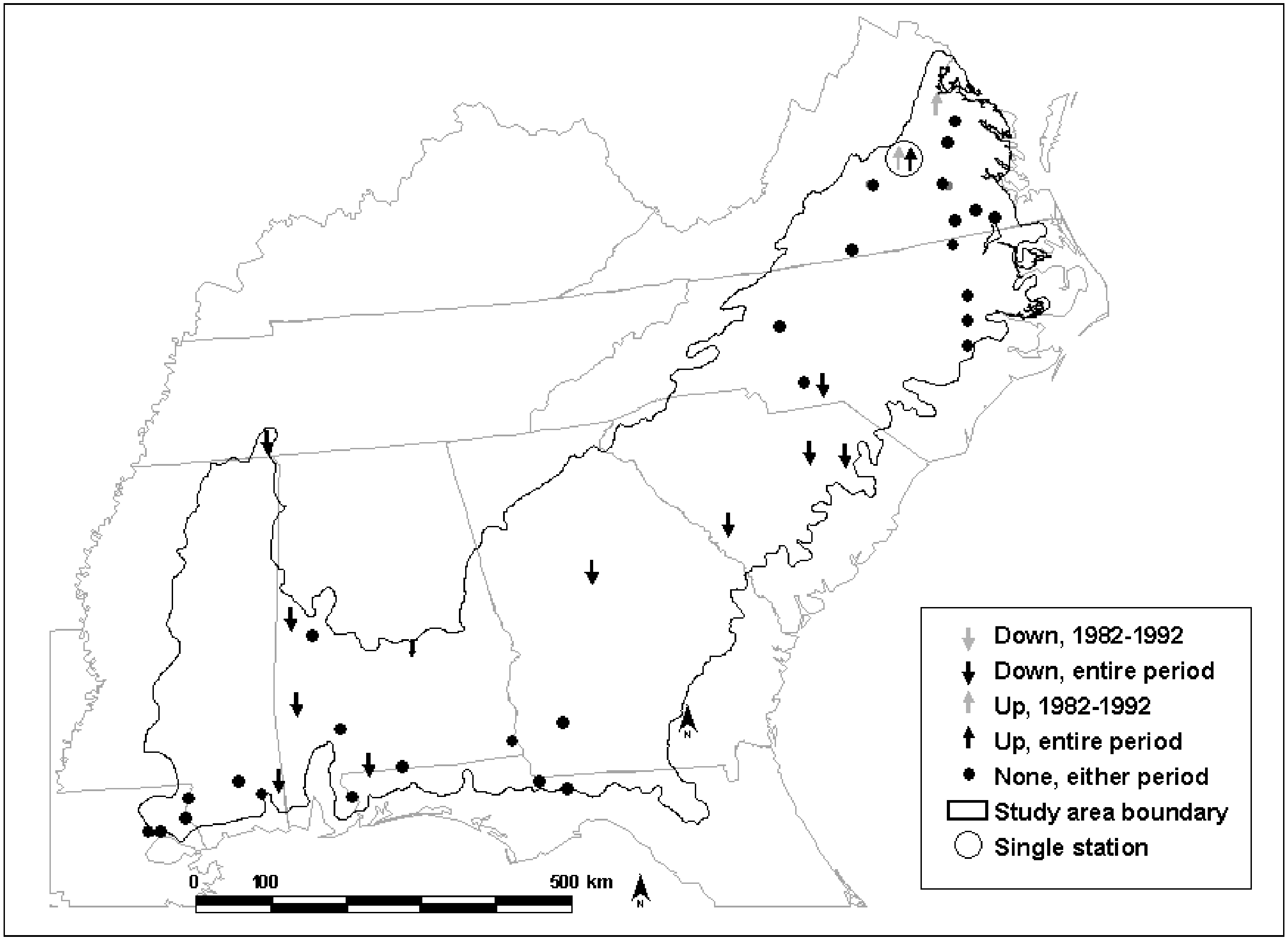


Figure 3.8. Station trends in instantaneous streamflow (FLOW) for two time periods.

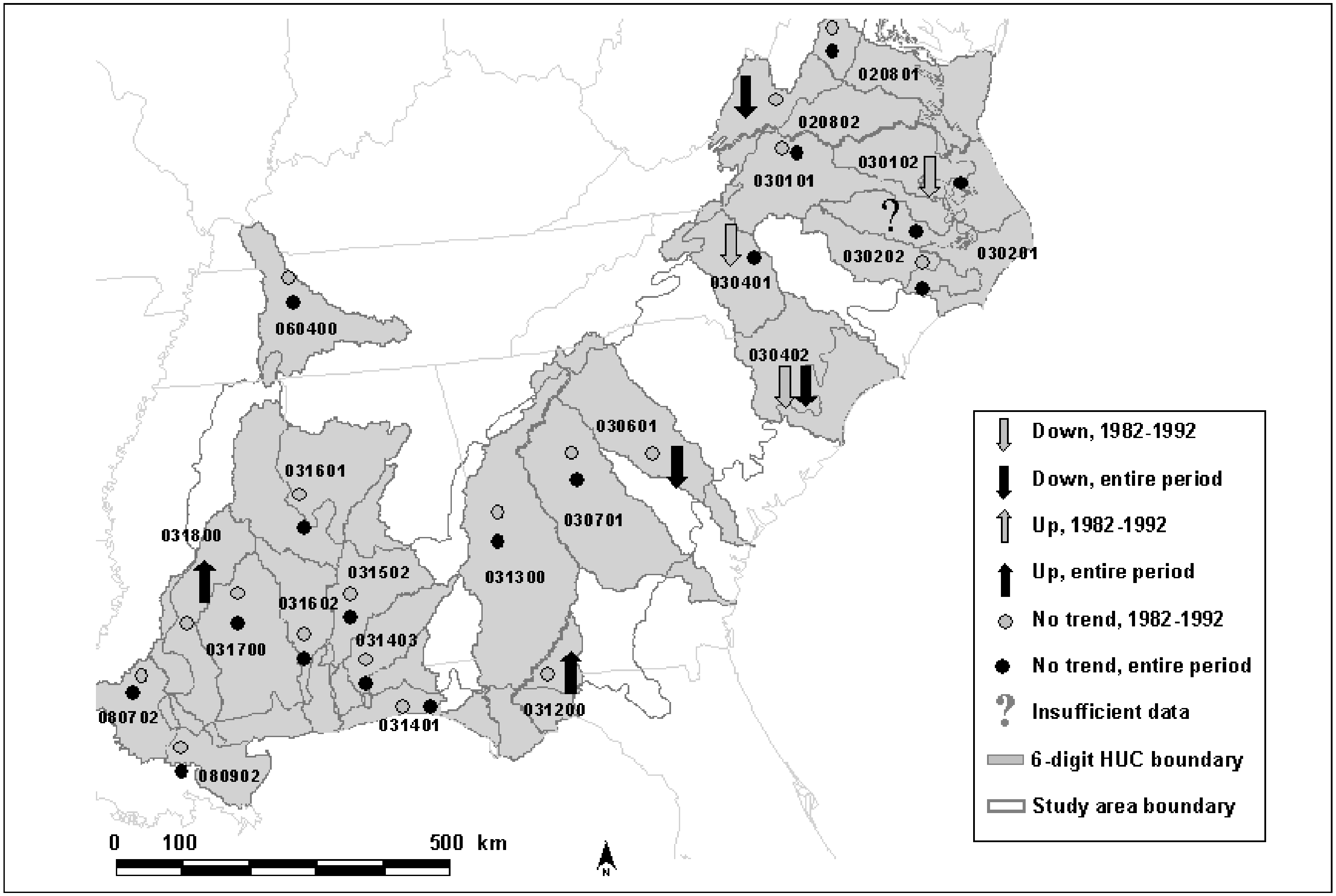


Figure 3.9. 6-digit HUC trends in dissolved oxygen (DOSAT) for two time periods.

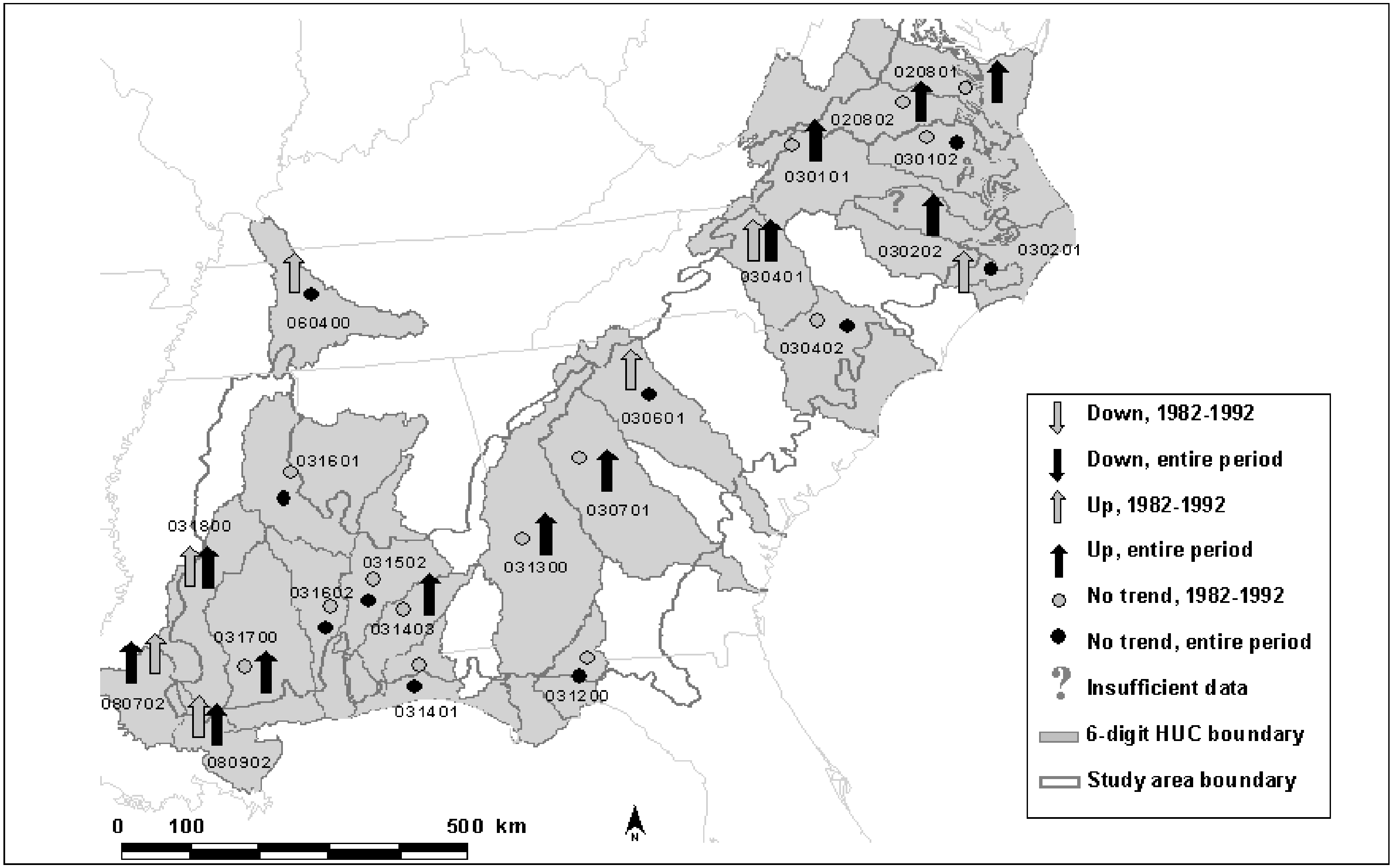


Figure 3.10. 6-digit HUC trends in pH for two time periods

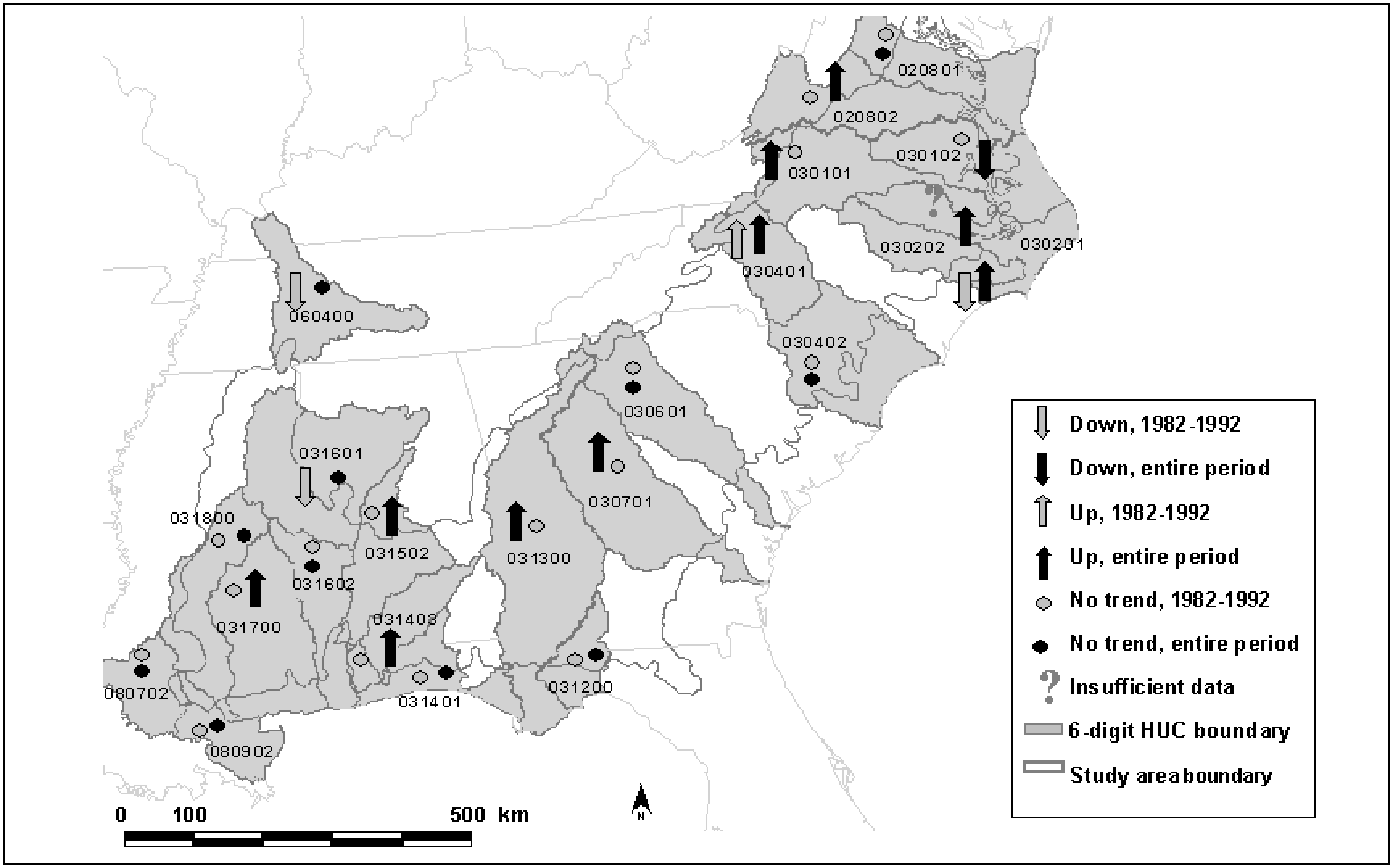


Figure 3.11. 6-digit HUC trends in total nitrogen (TN) for two time periods.

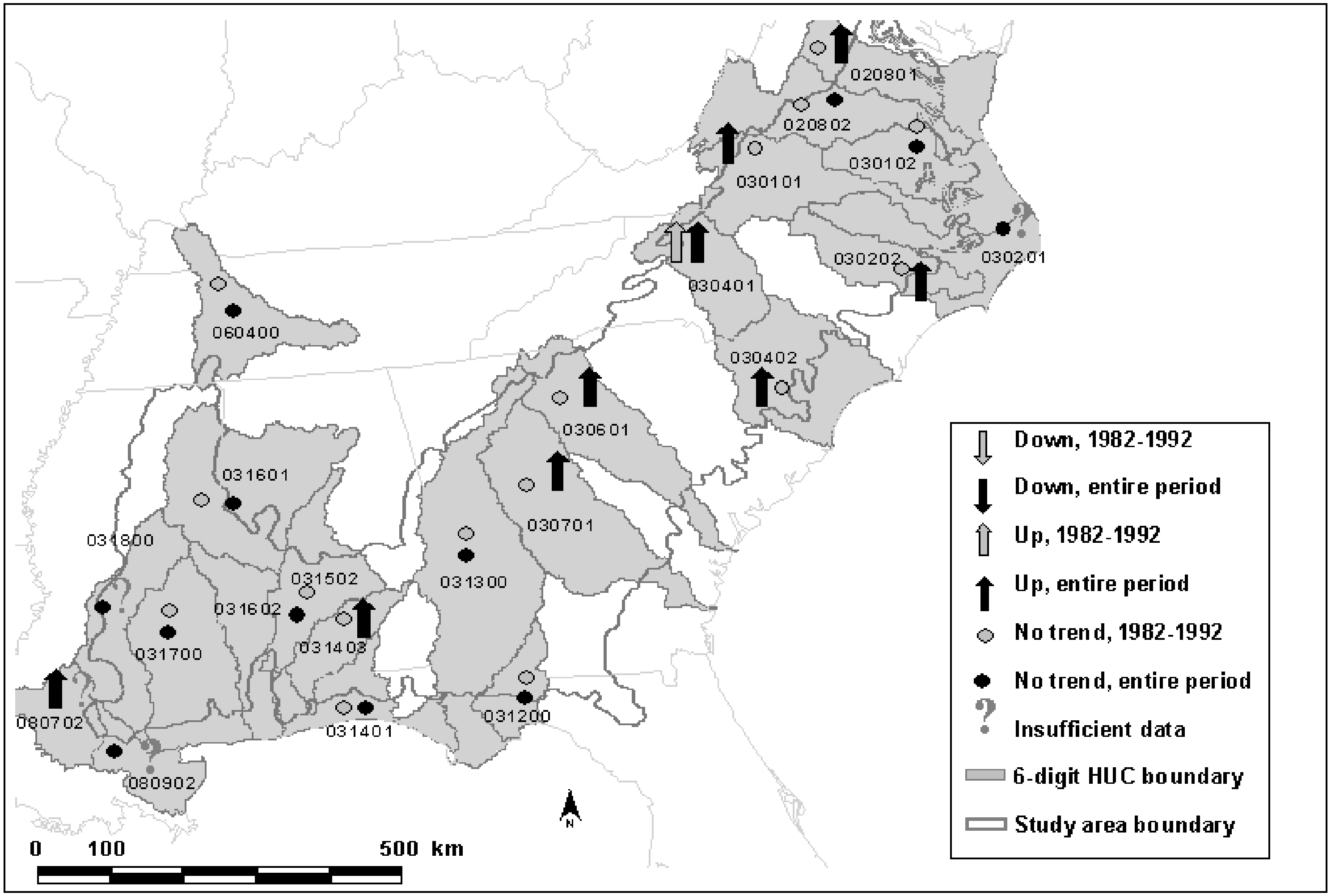


Figure 3.12. 6-digit HUC trends in total phosphorus (TP) for two time periods.

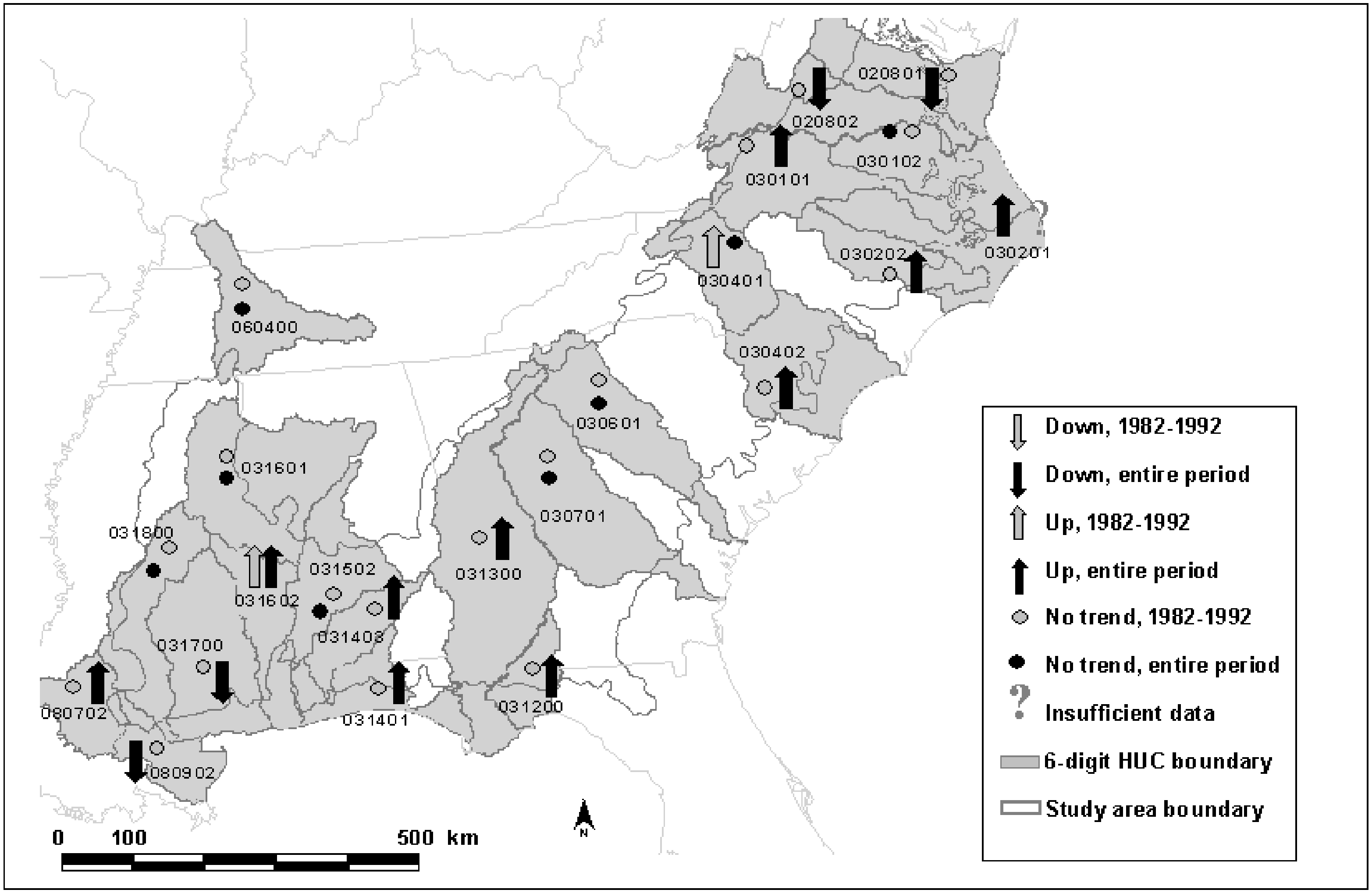


Figure 3.13. 6-digit HUC trends in Sediment (SED) for two time periods.

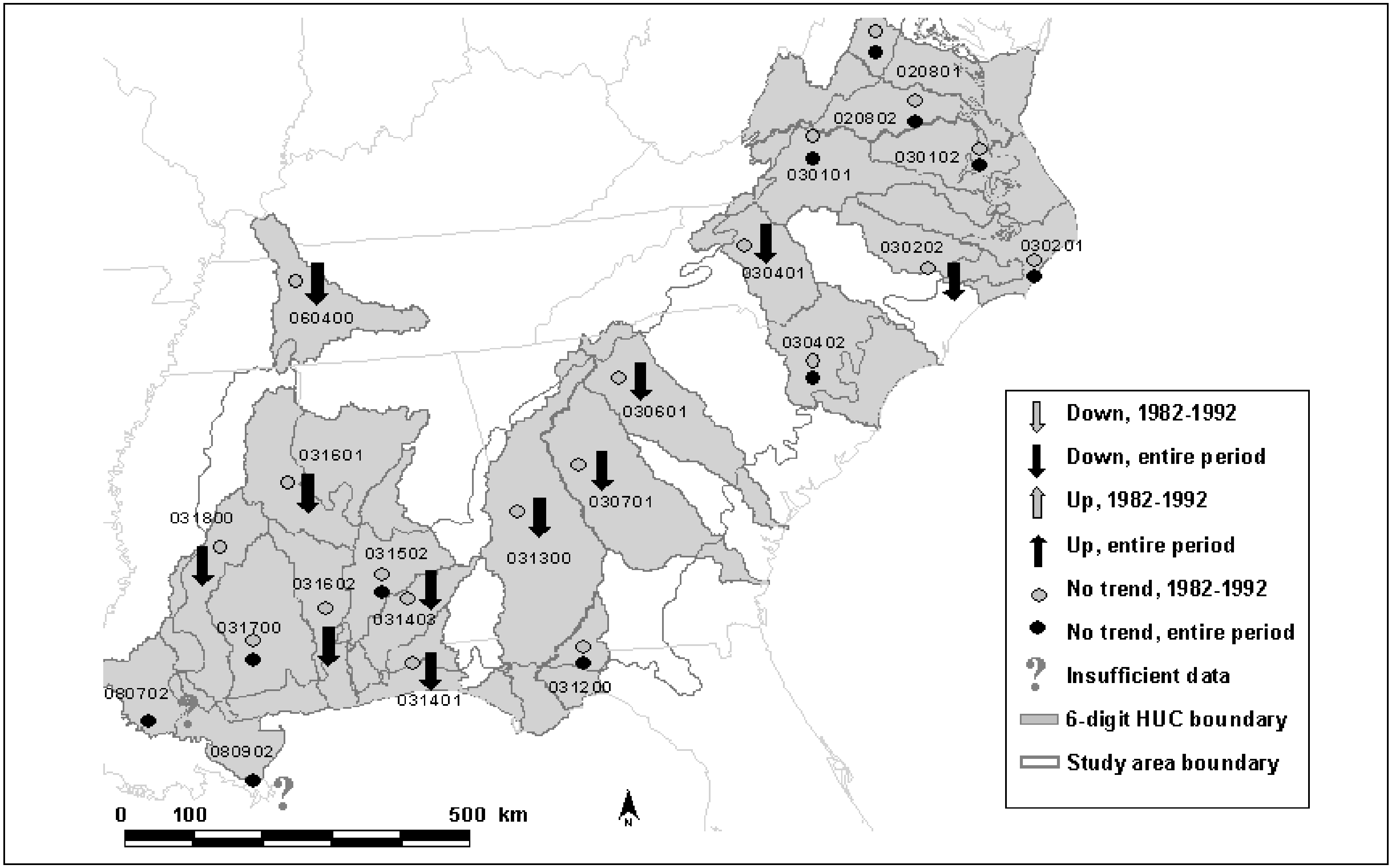


Figure 3.14. 6-digit HUC trends in instantaneous streamflow (FLOW) for two time periods.

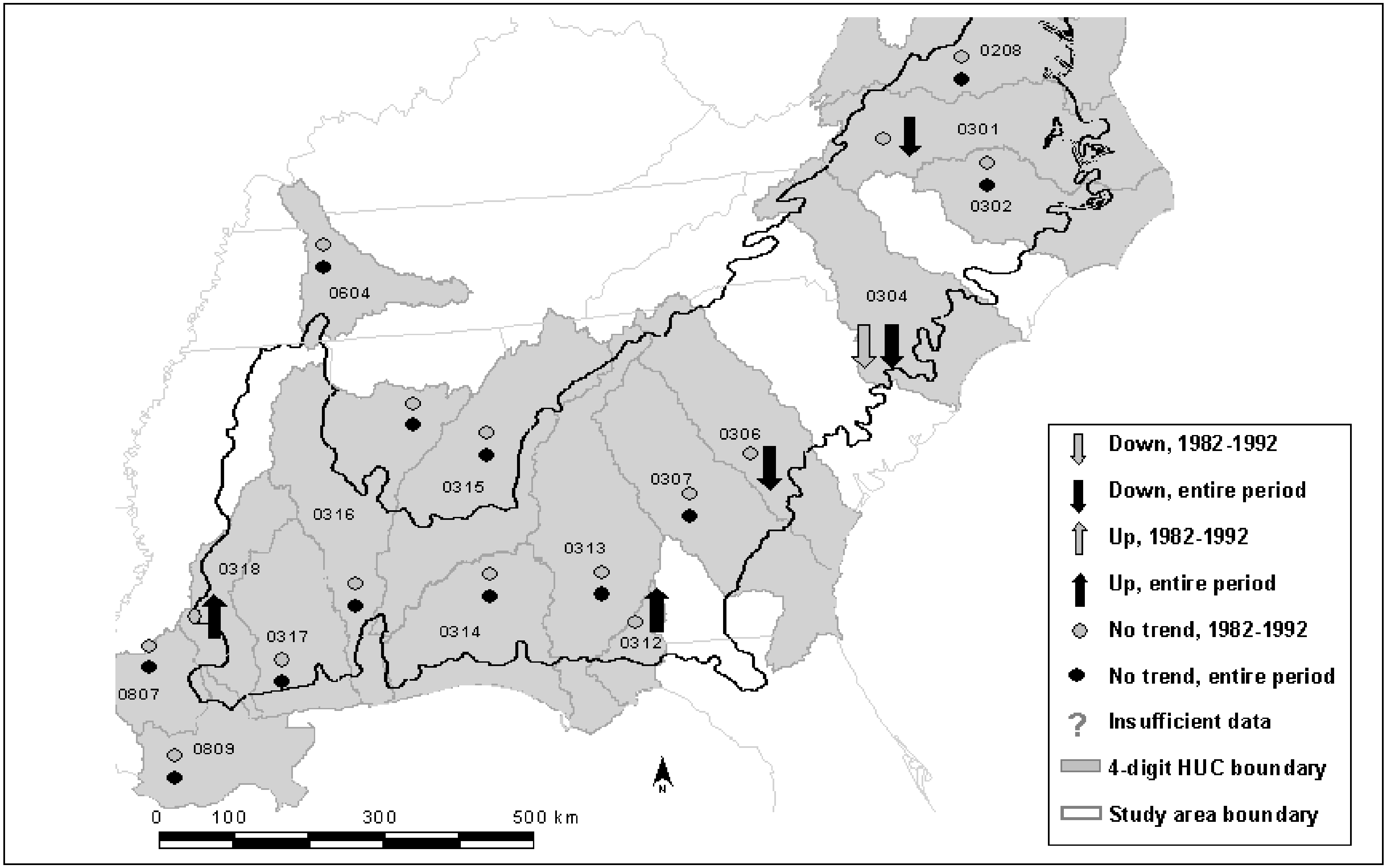


Figure 3.15. 4-digit HUC trends in dissolved oxygen (DOSAT) for two time periods.

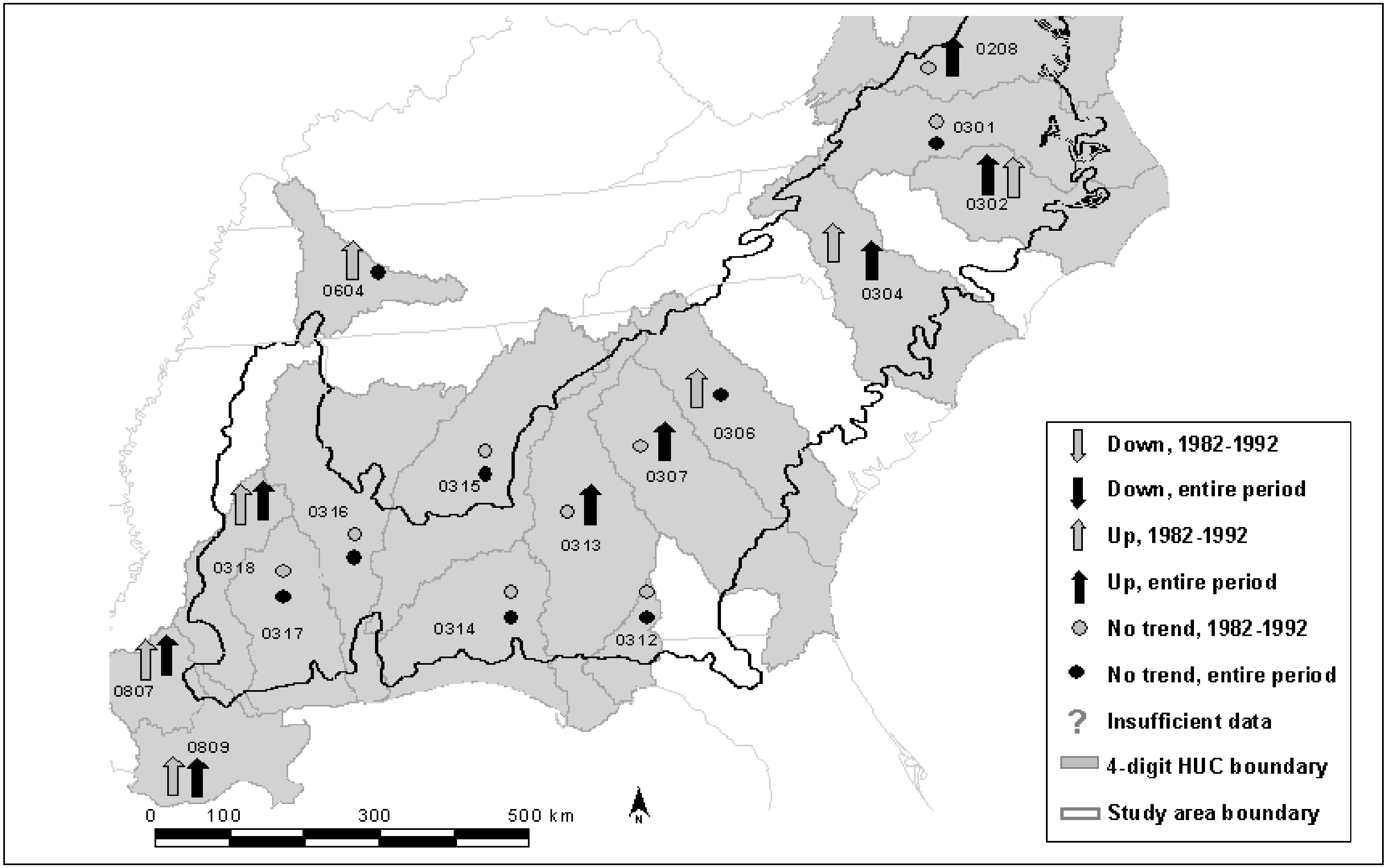


Figure 3. 16. 4-digit HUC trends in pH

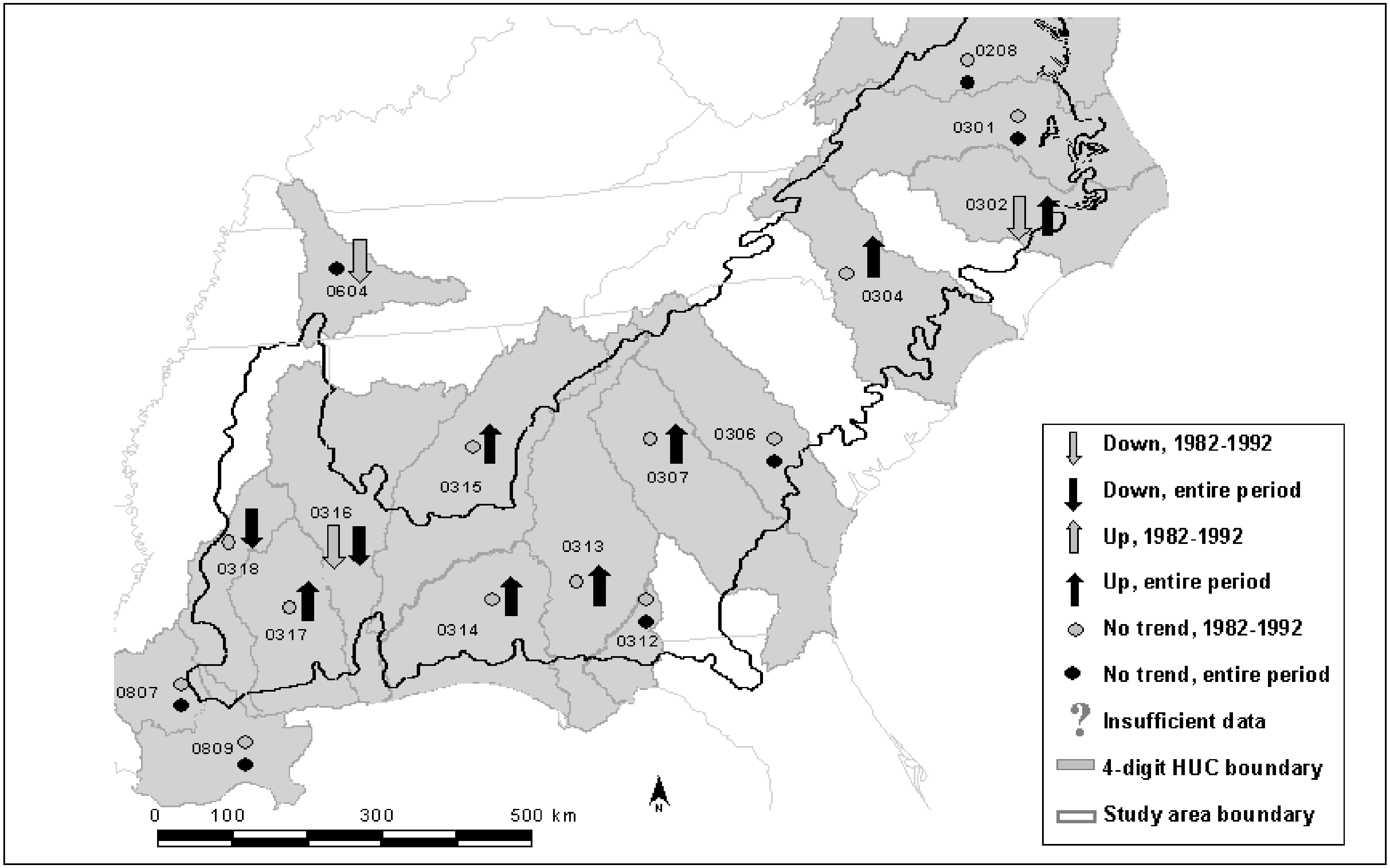


Figure 3.17. 4-digit HUC trends in total nitrogen (TN) for two time periods.

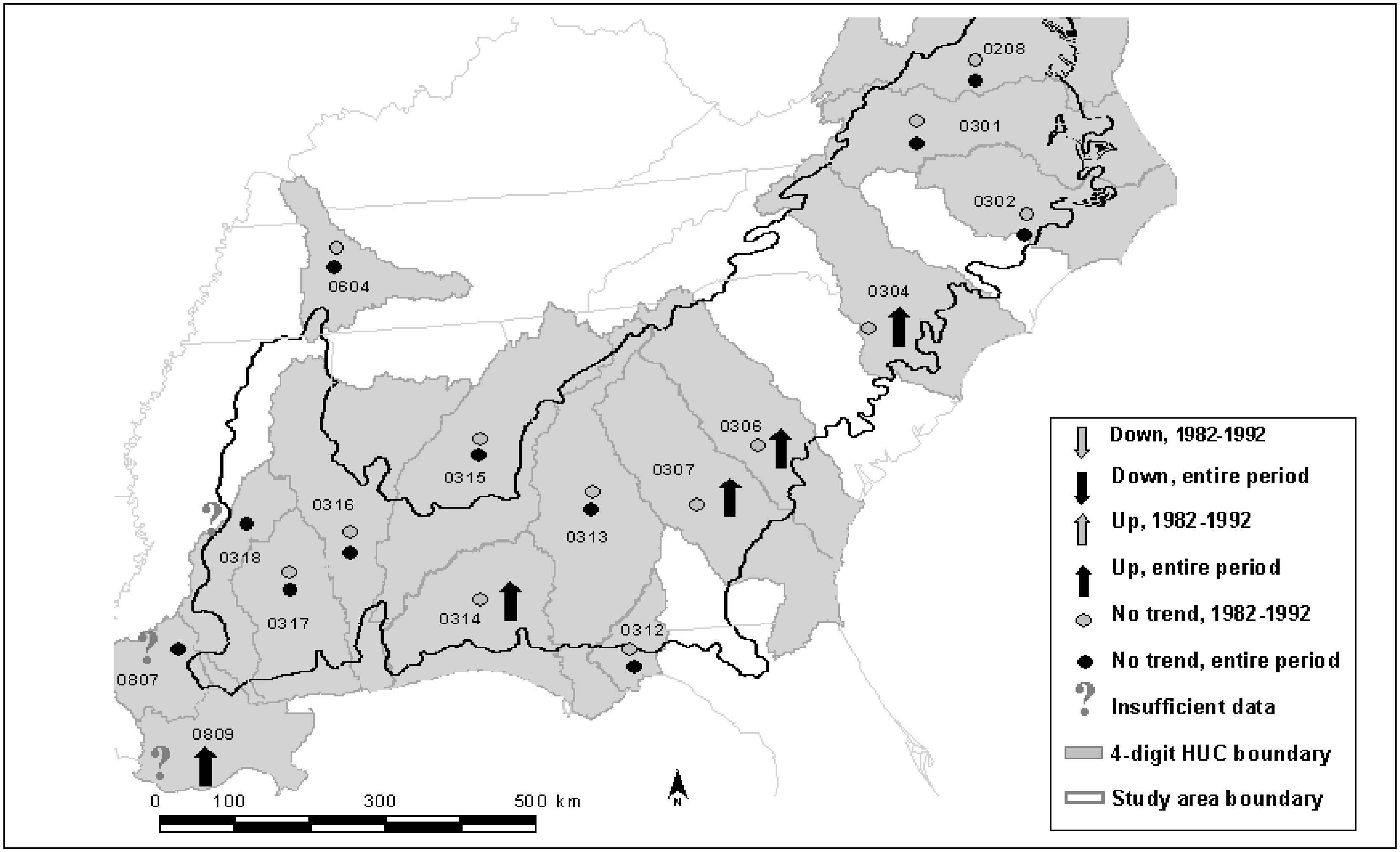


Figure 3.18. 4-digit HUC trends in total phosphorus (TP)

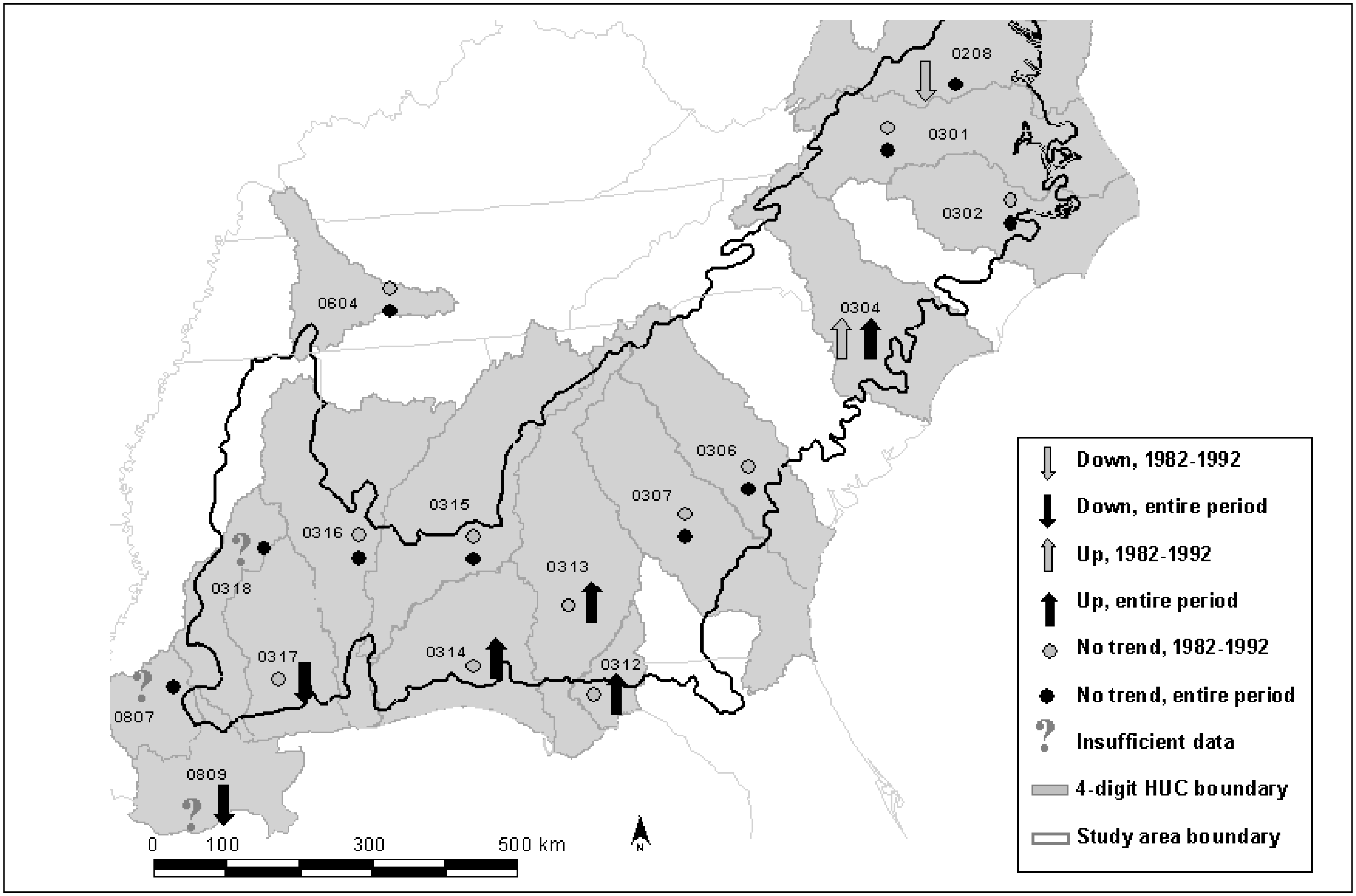


Figure 3.19. 4-digit HUC trends in Sediment (SED) for two time periods.

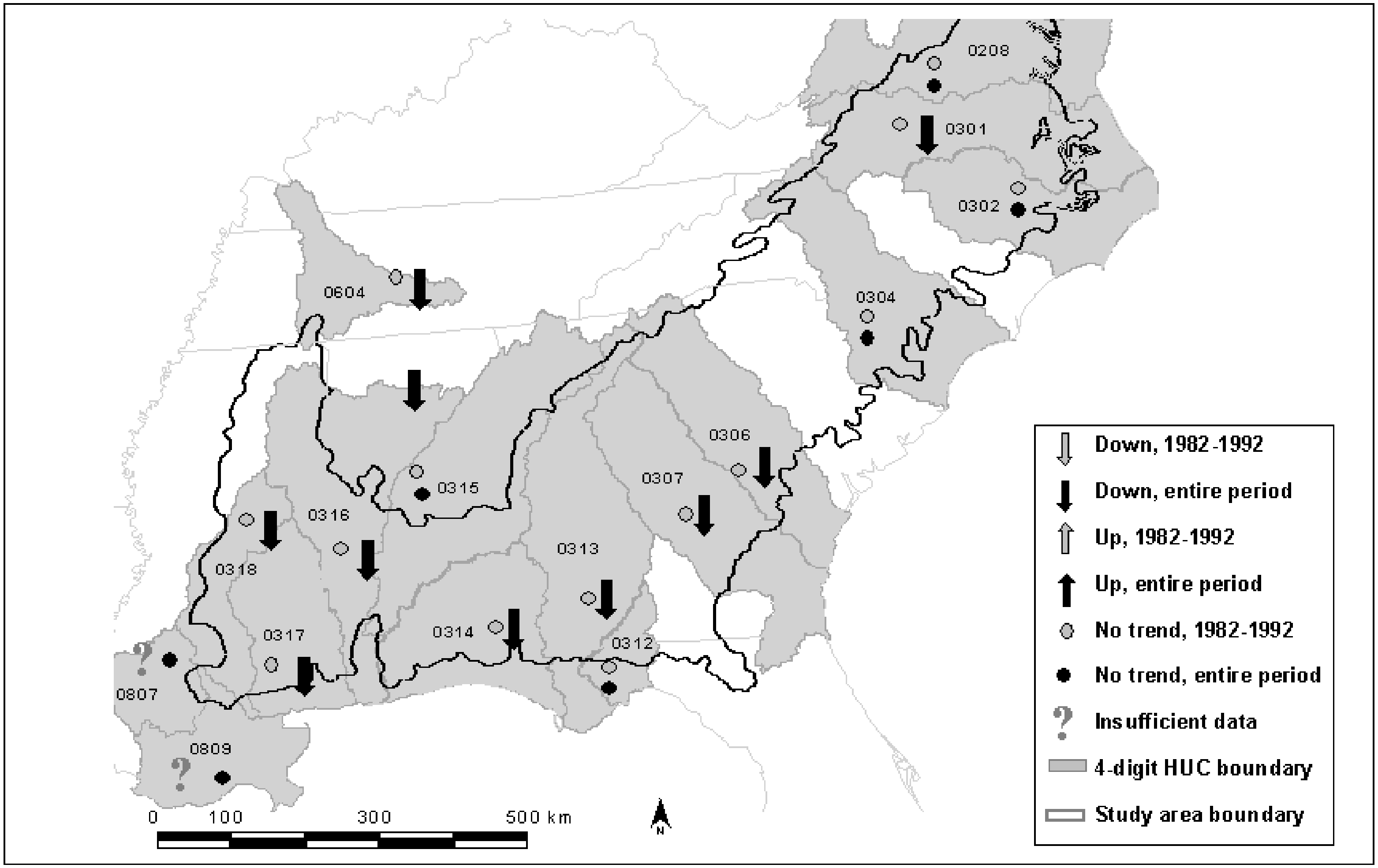


Figure 3.20. 4-digit HUC trends in instantaneous streamflow (FLOW) for two time periods.

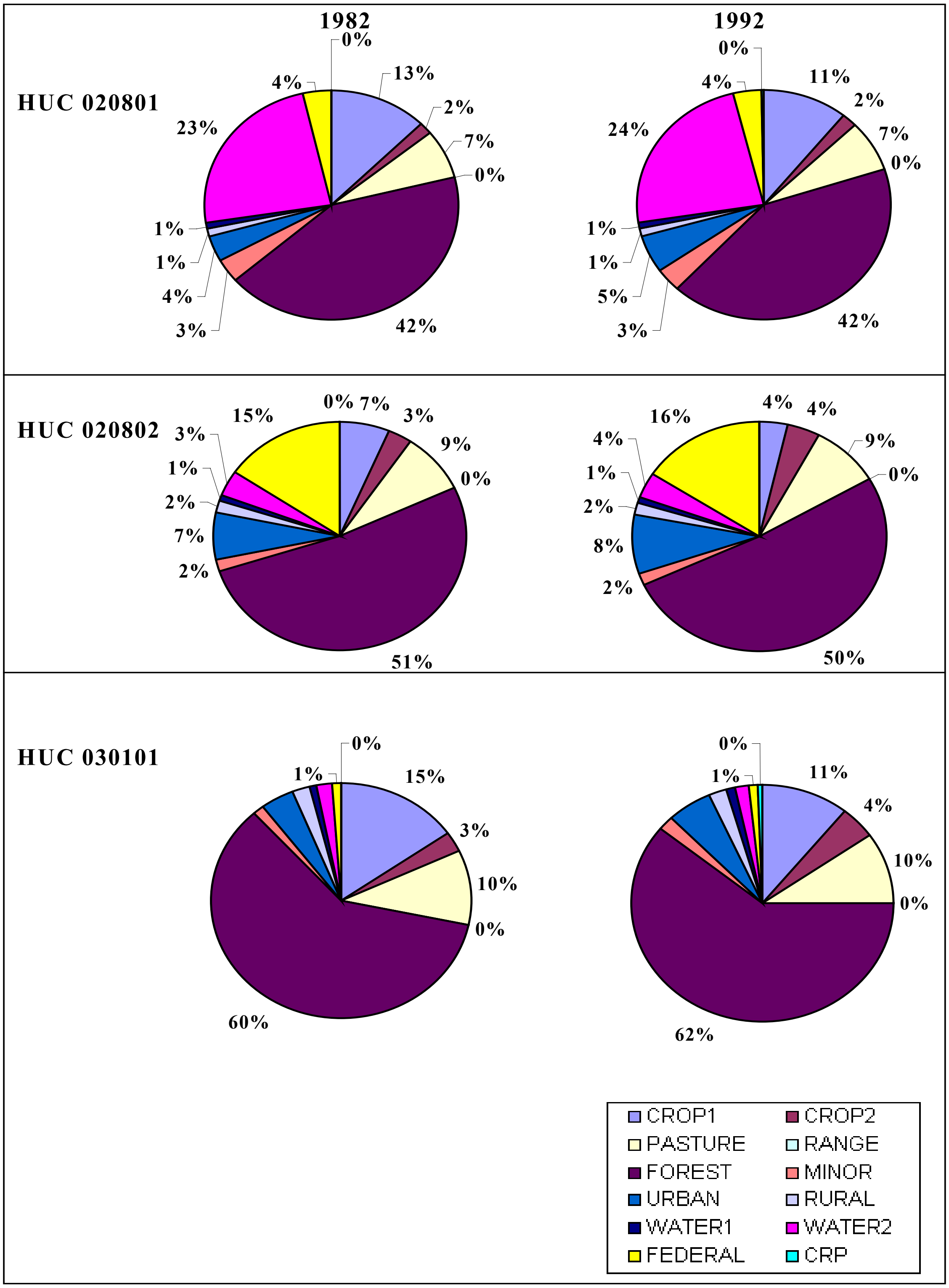


Figure 3.21. Comparison of land use /land cover categories by census year (1982, 1992) and by 6-digit HUC. Values ≤ 1% are shown as "0%".

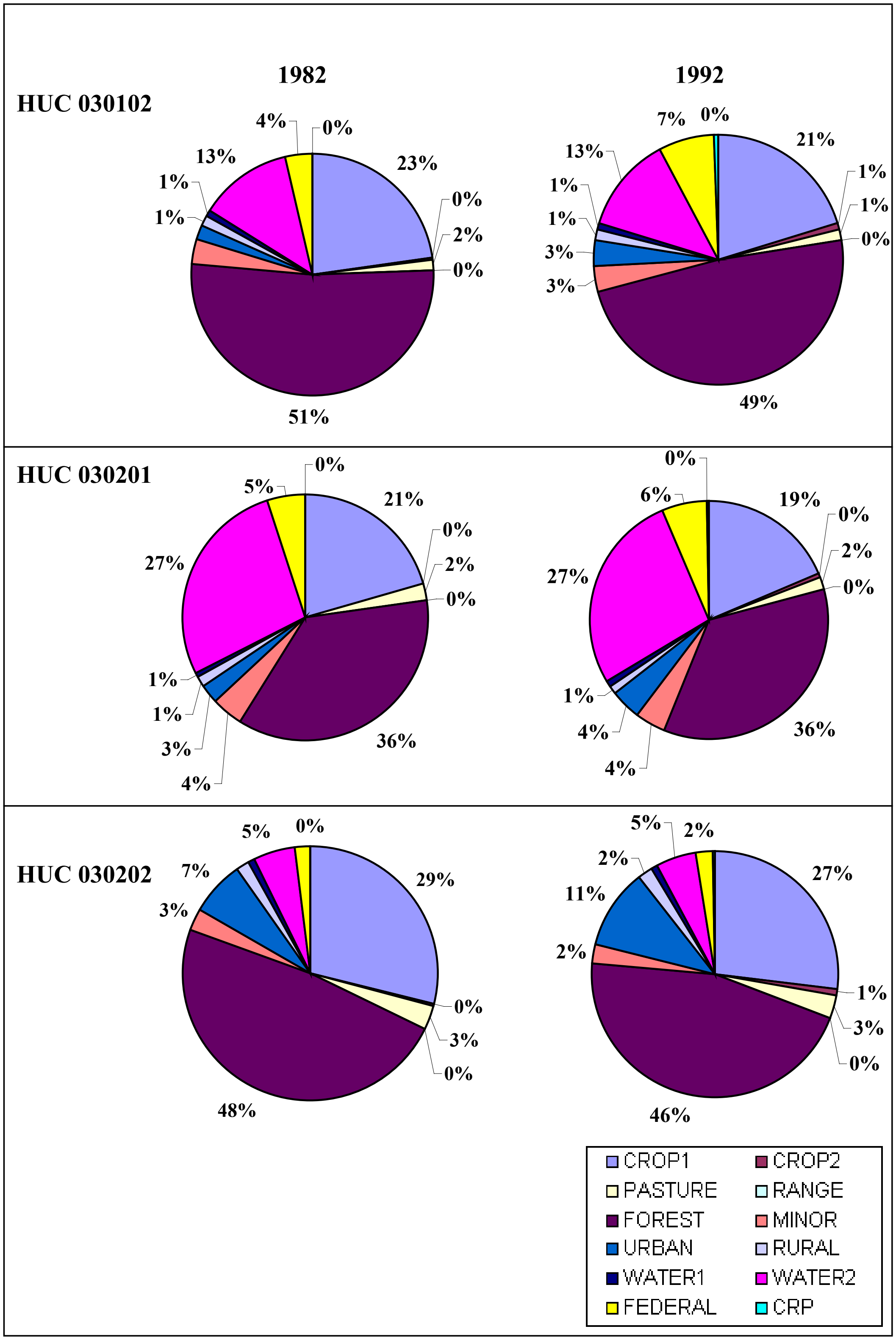


Figure 3.21 continued.

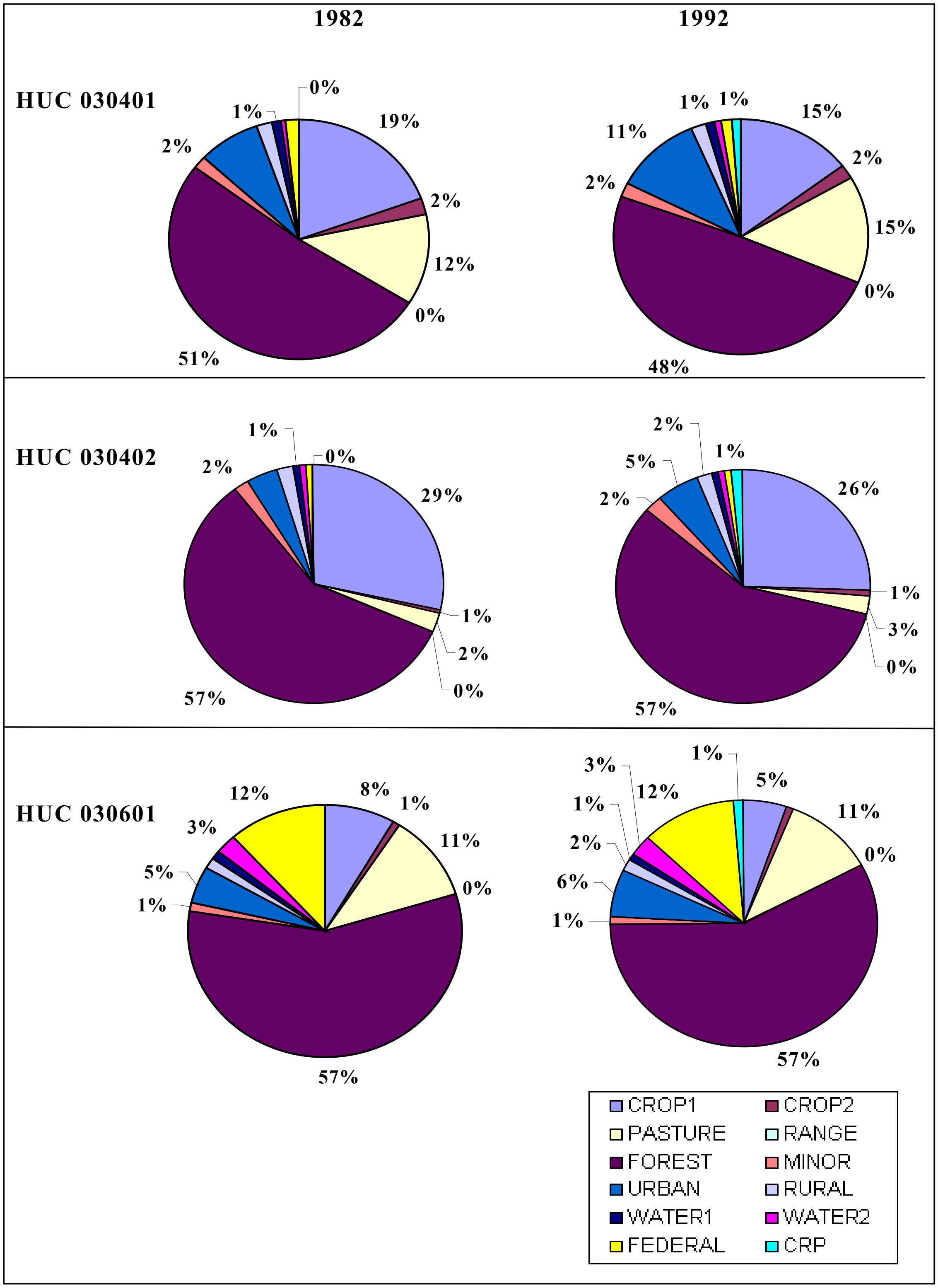
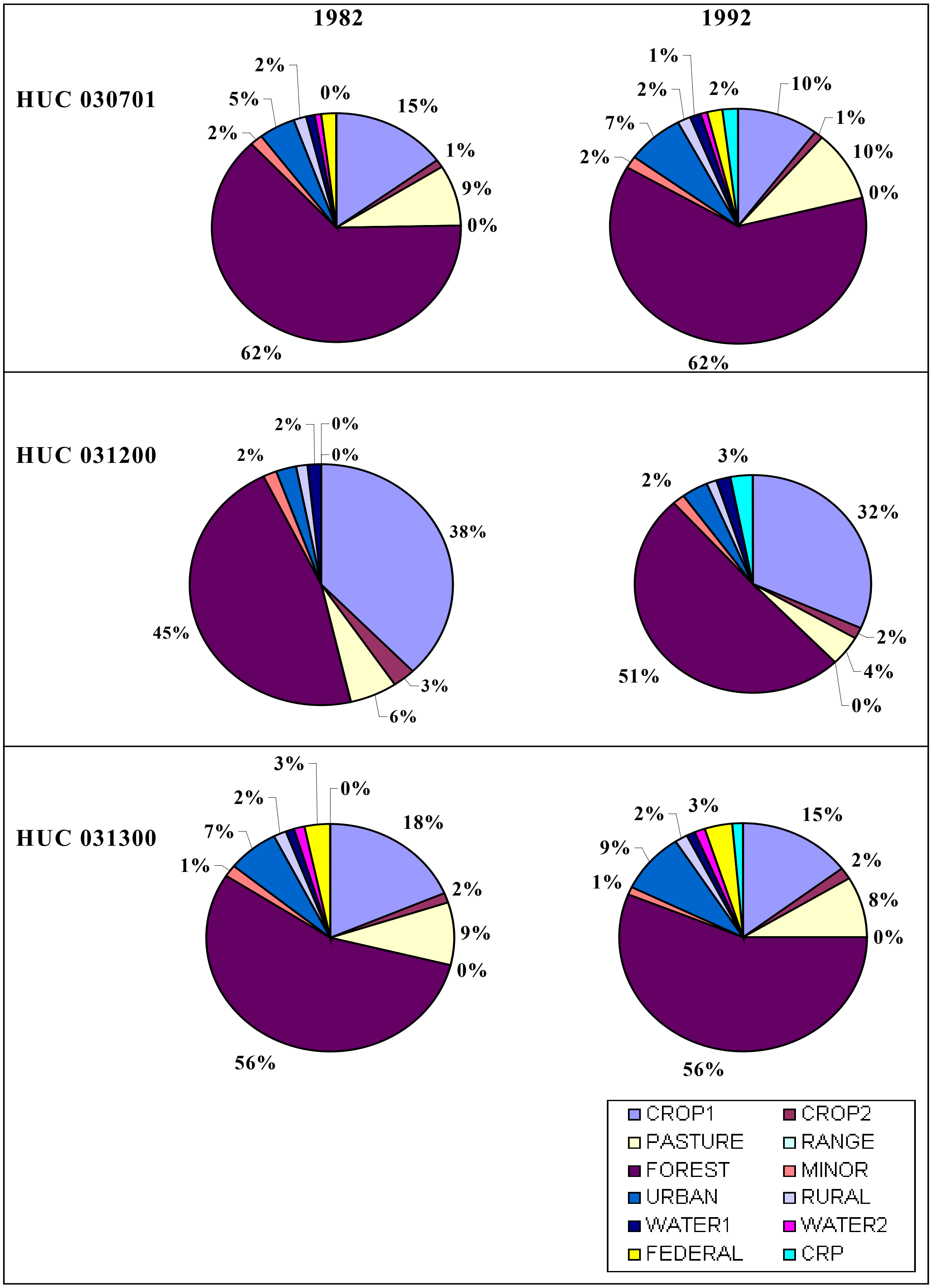


Figure 3.21 continued.



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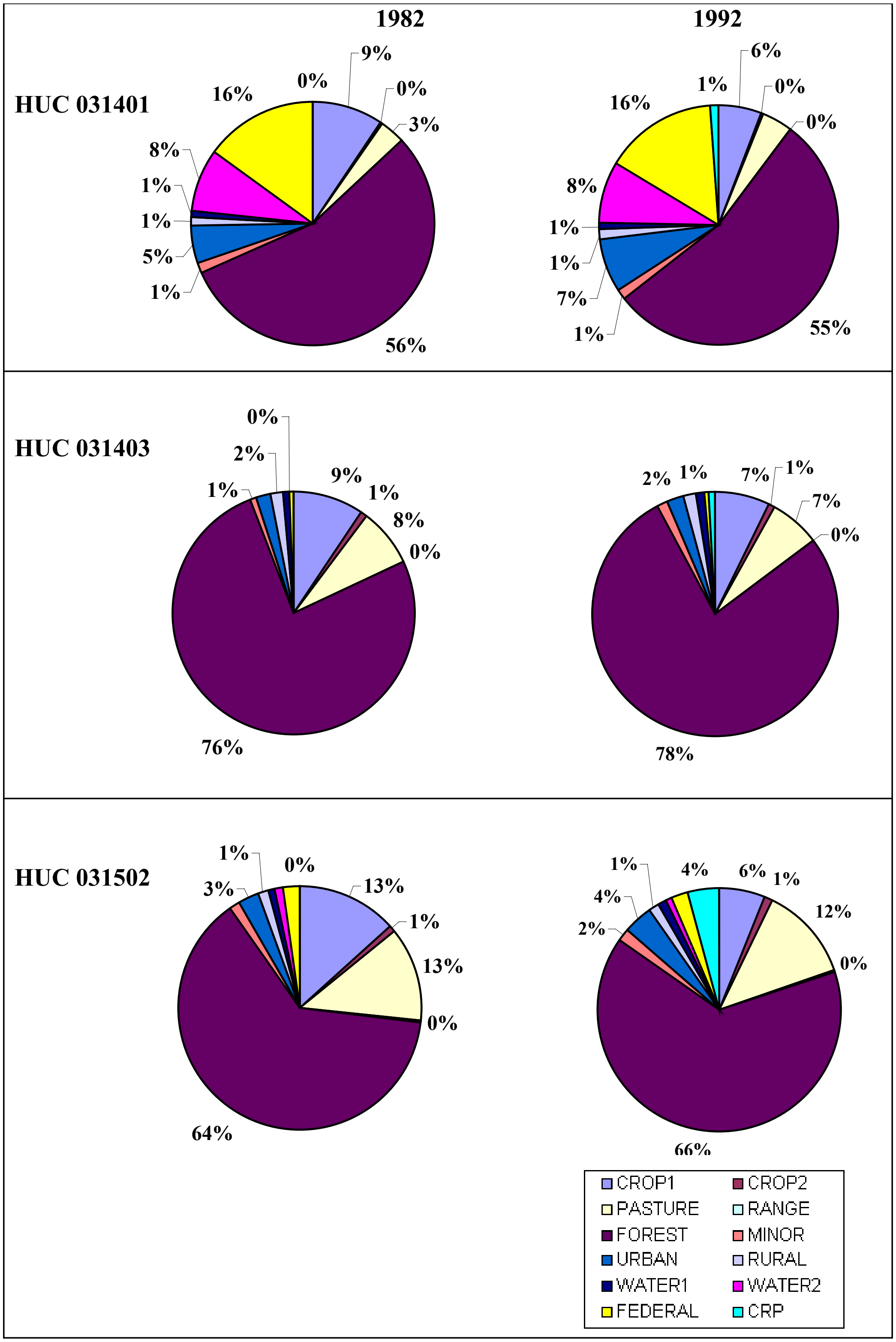
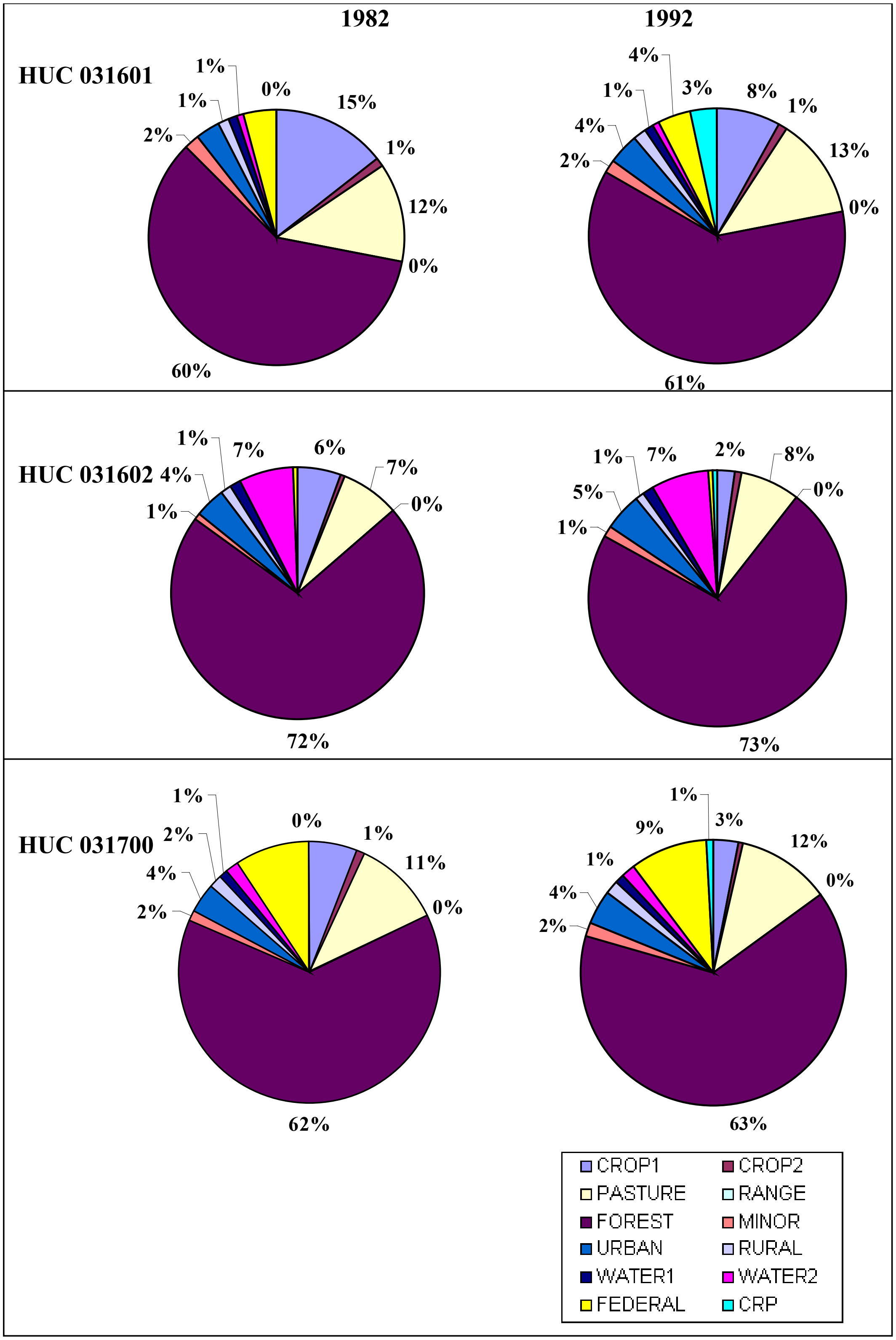


Figure 3.21 continued.



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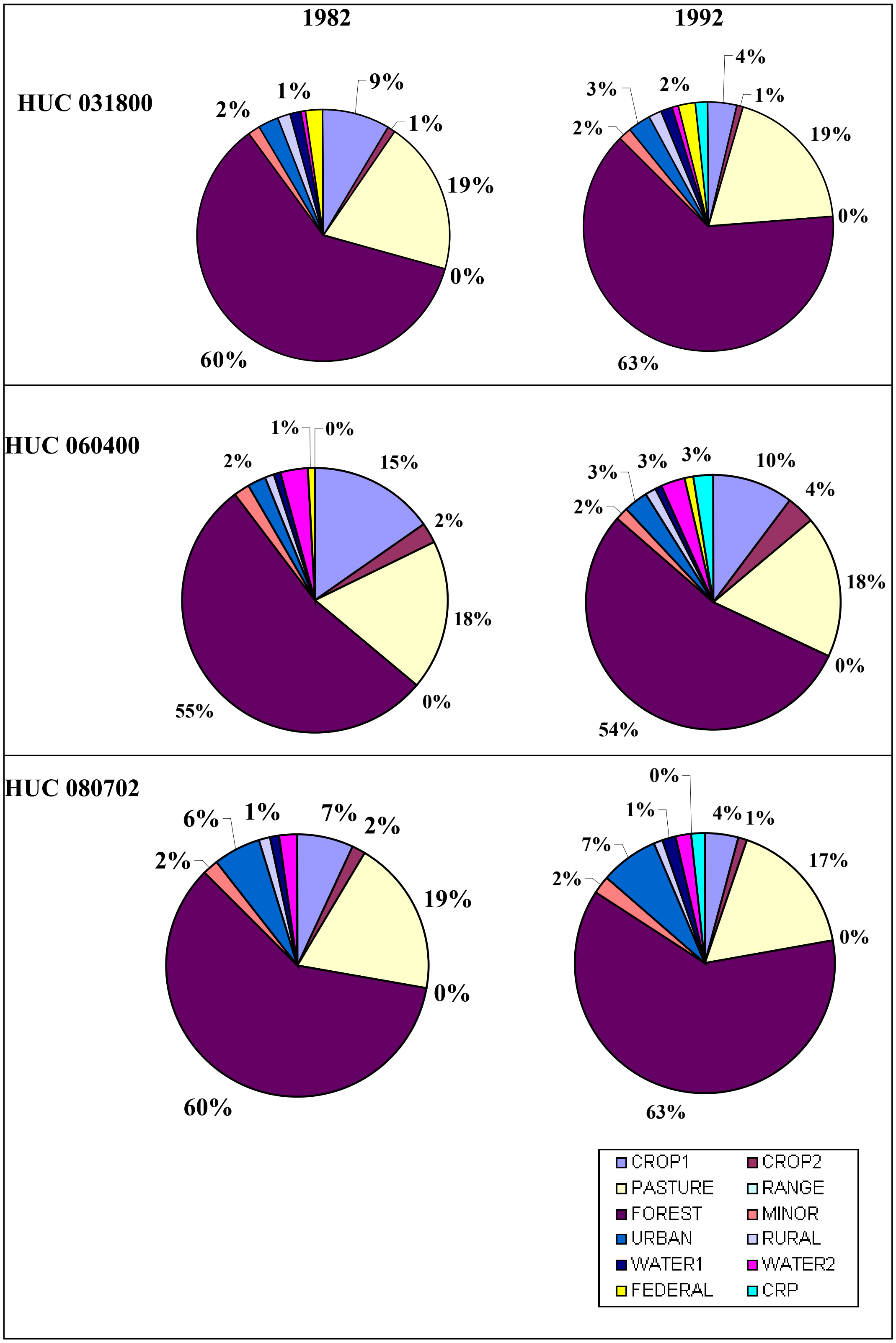

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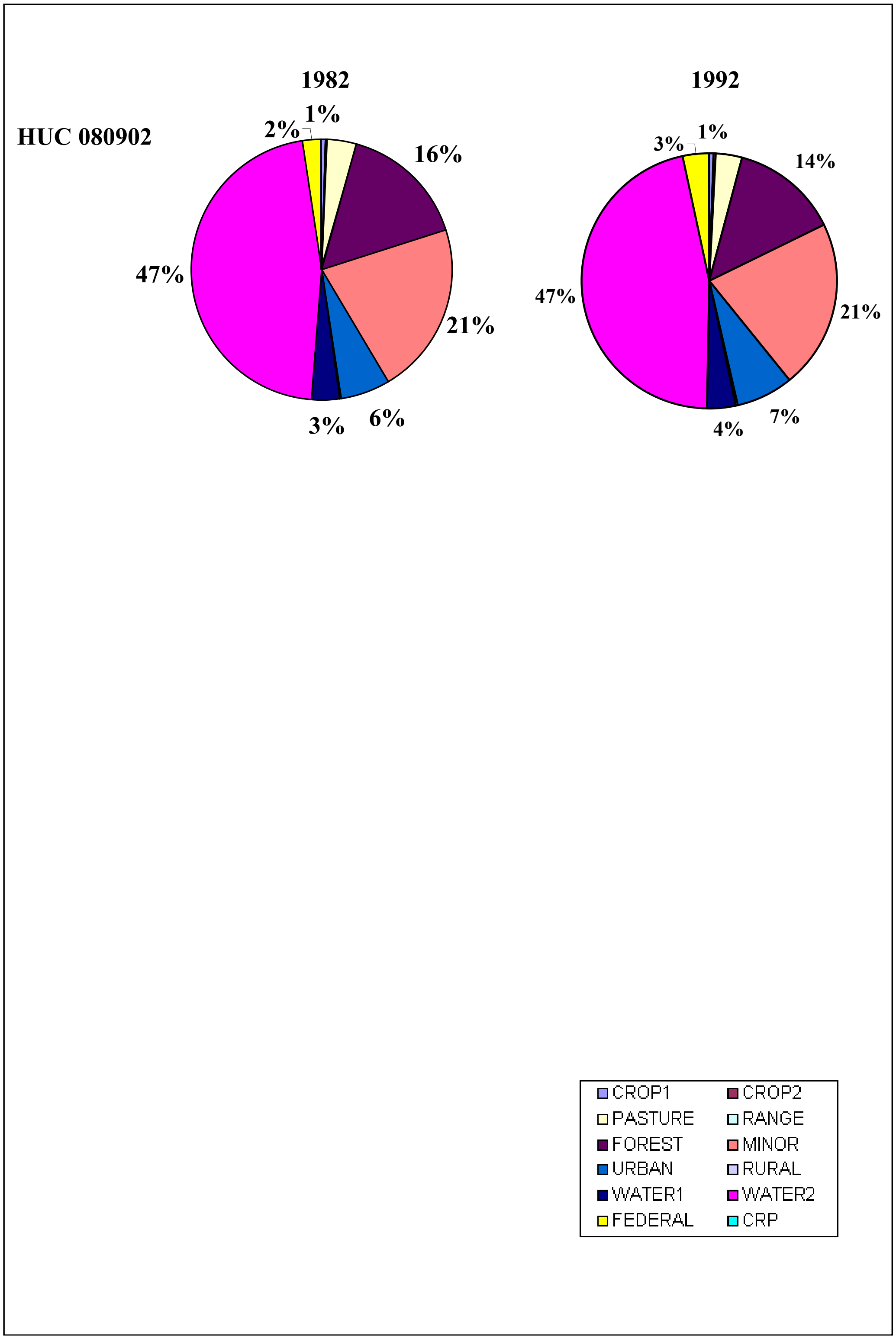

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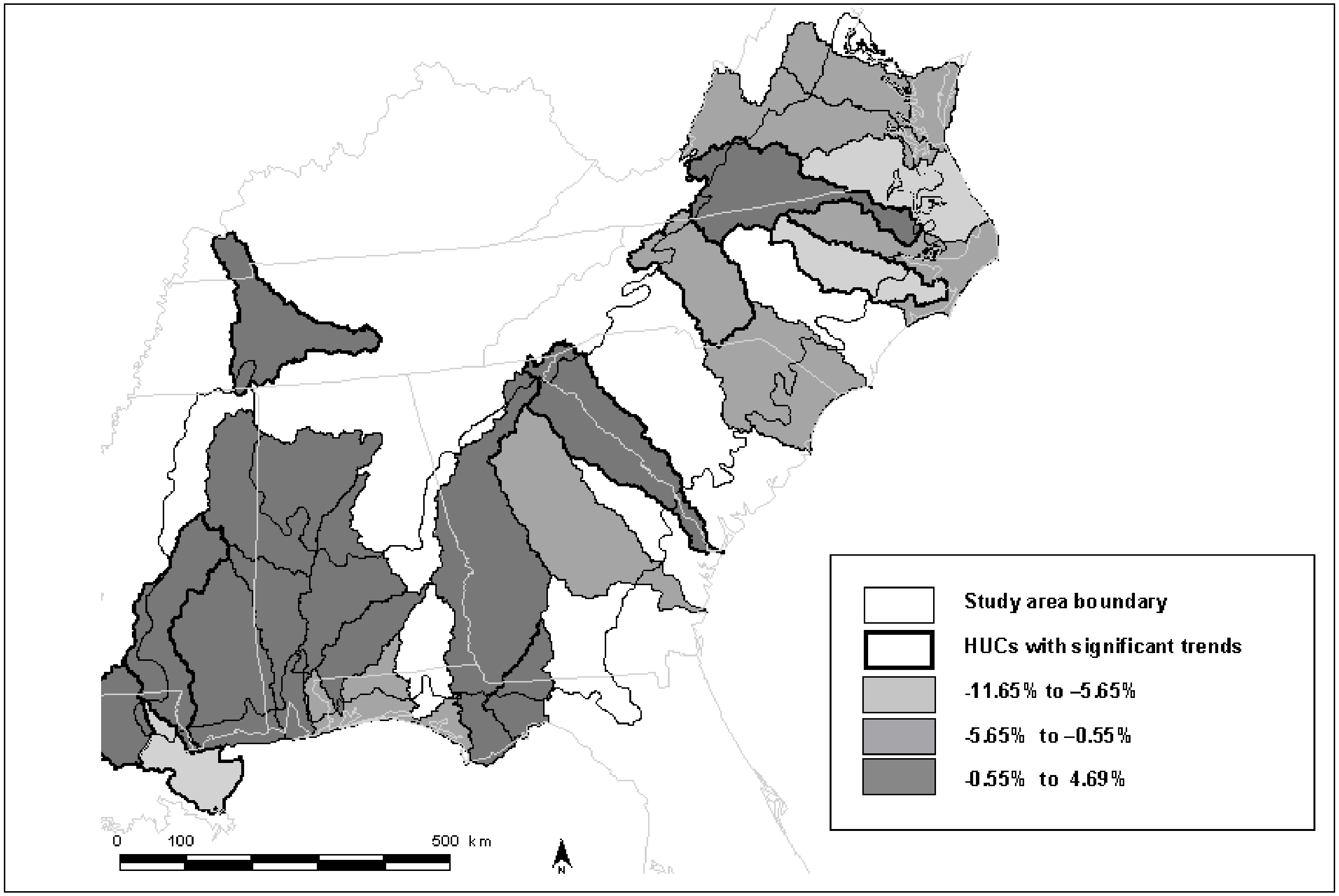

Figure 3.22. Maps showing percentage change in forestland area 1982-1992. Negative values indicate loss. The 6-digit HUC's with significant upward trends in pH are highlighted in bold borders. Classification is based on aggregated NRI data.

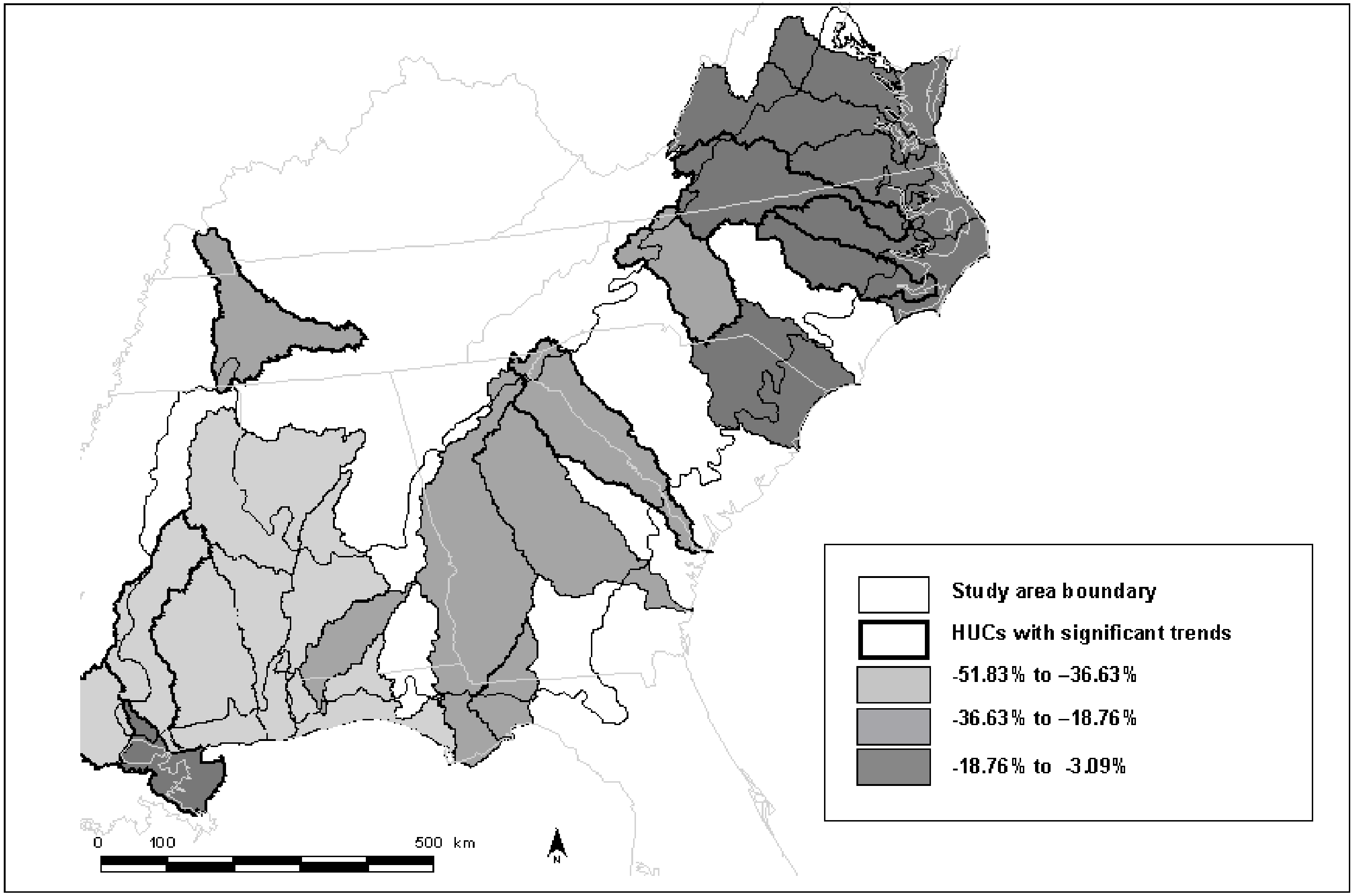

Figure 3.23.Map showing percentage change in cropland area, 1982-1992. All 6-digit HUC's reported losses (negative values) I cropland. Huc's with significant upward trends for pH are highlighted in bold borders. Classification is based on aggregated NRI data.

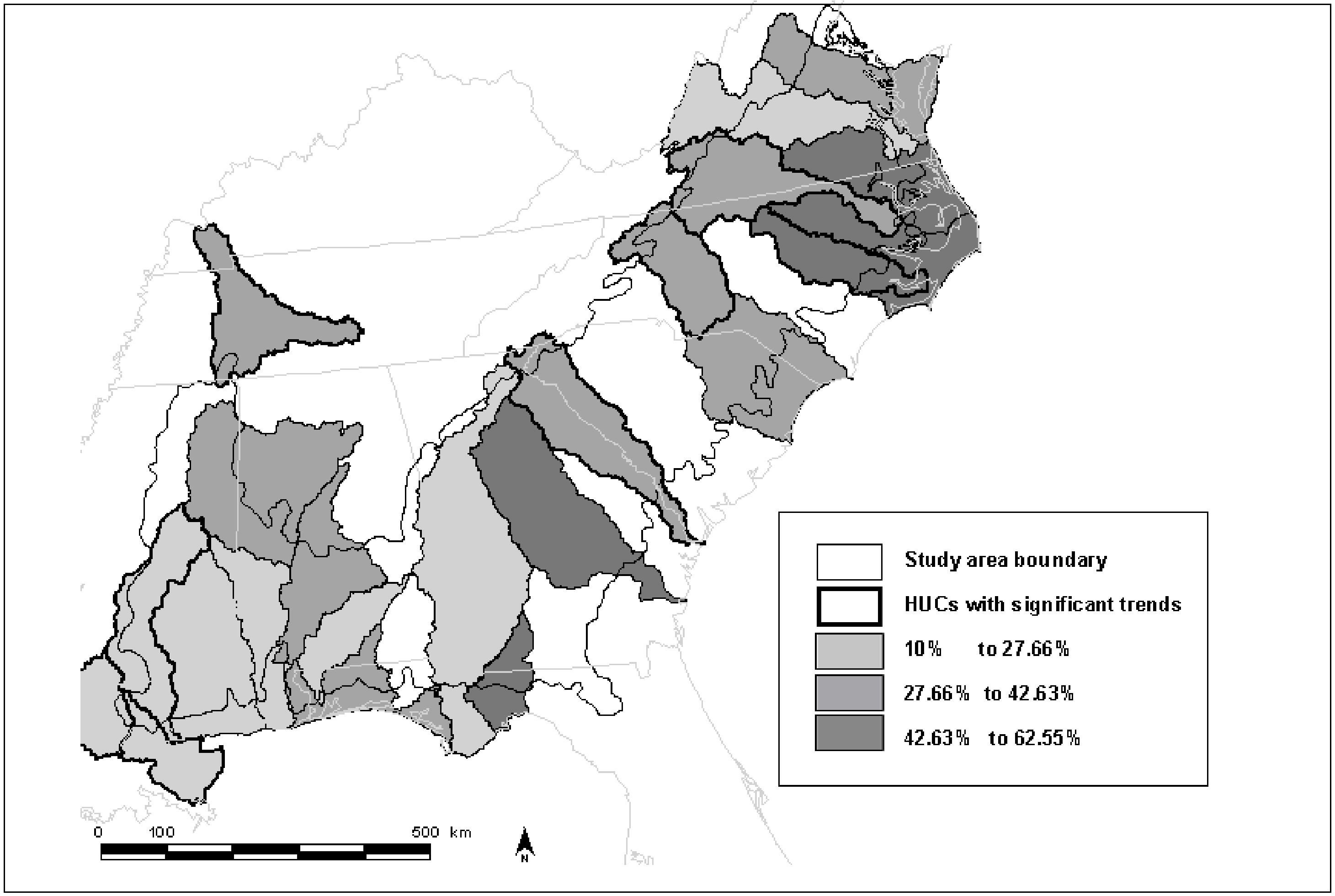

Figure 3.24. Map showing percentage change in urban area, 1982-1992. All 6-digit HUC's reported increases (positive values) in urban area. HUC's with significant upward trends in pH are shown in bold borders. Classification is based on the aggregated NRI data.

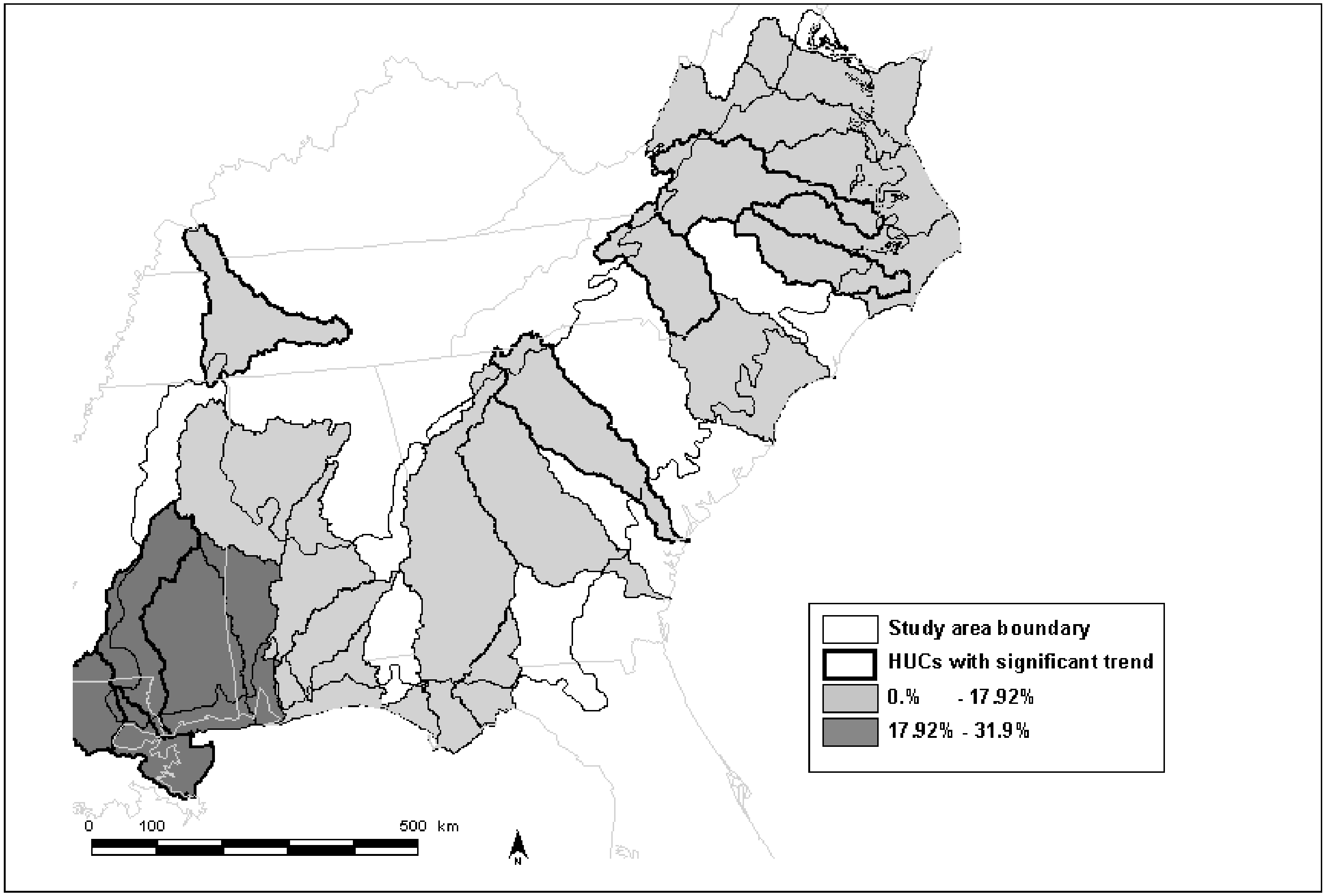

Figure 3.25. Map showing percentage area conversion of cultivated cropland to pasture land for each 6-digit HUC. HUC's with significant upward trends in pH are highlighted in bold borders. Classification is based on twelve categories of NRI data.

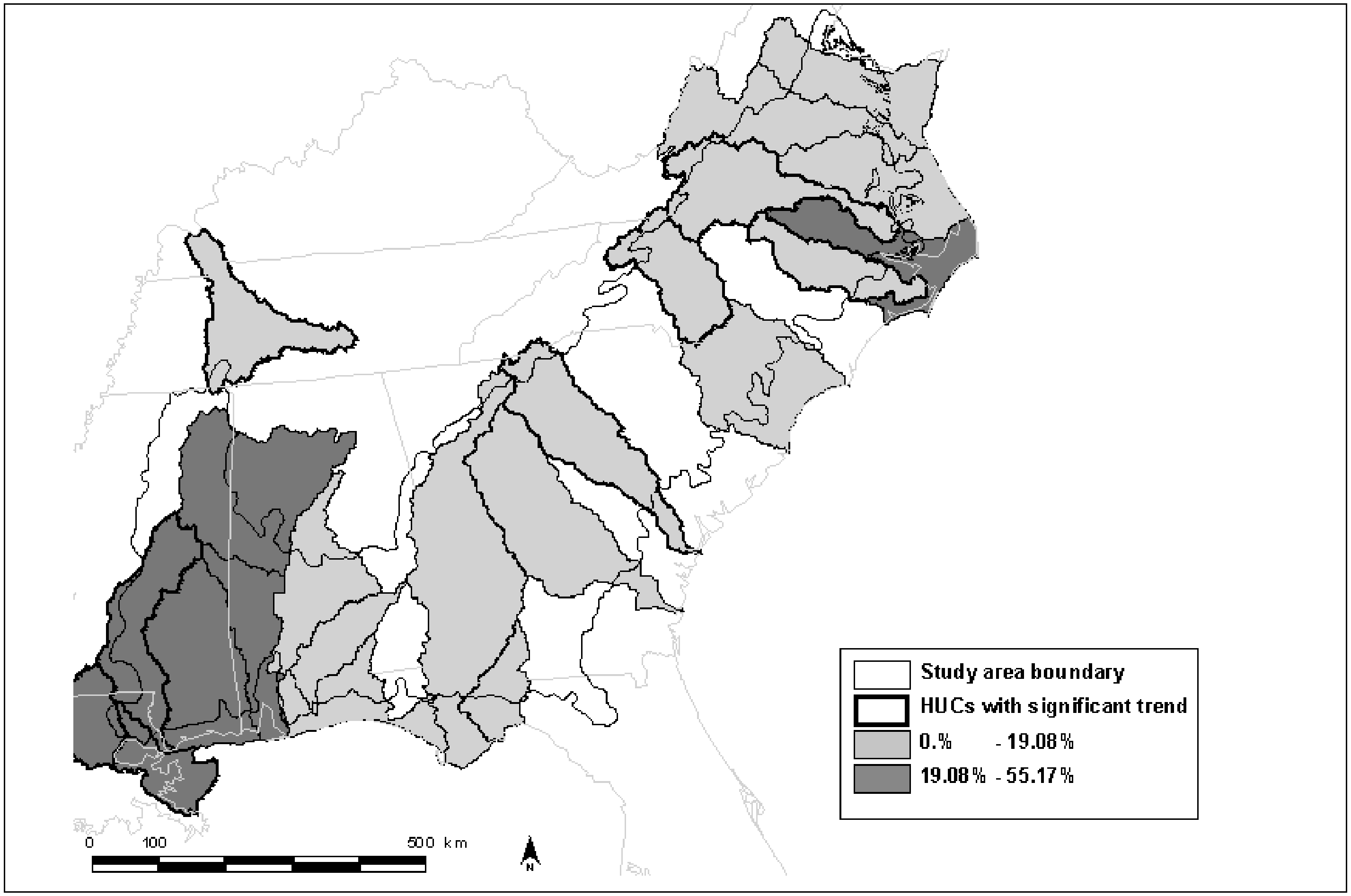

Figure 3.26. Map showing percentage area conversion of non-cultivated cropland to pastureland, for each 6-digit HUC. HUC's with significant upward trends in pH are highlighted in bold borders. Classification is based on 12 categories of NRI data.

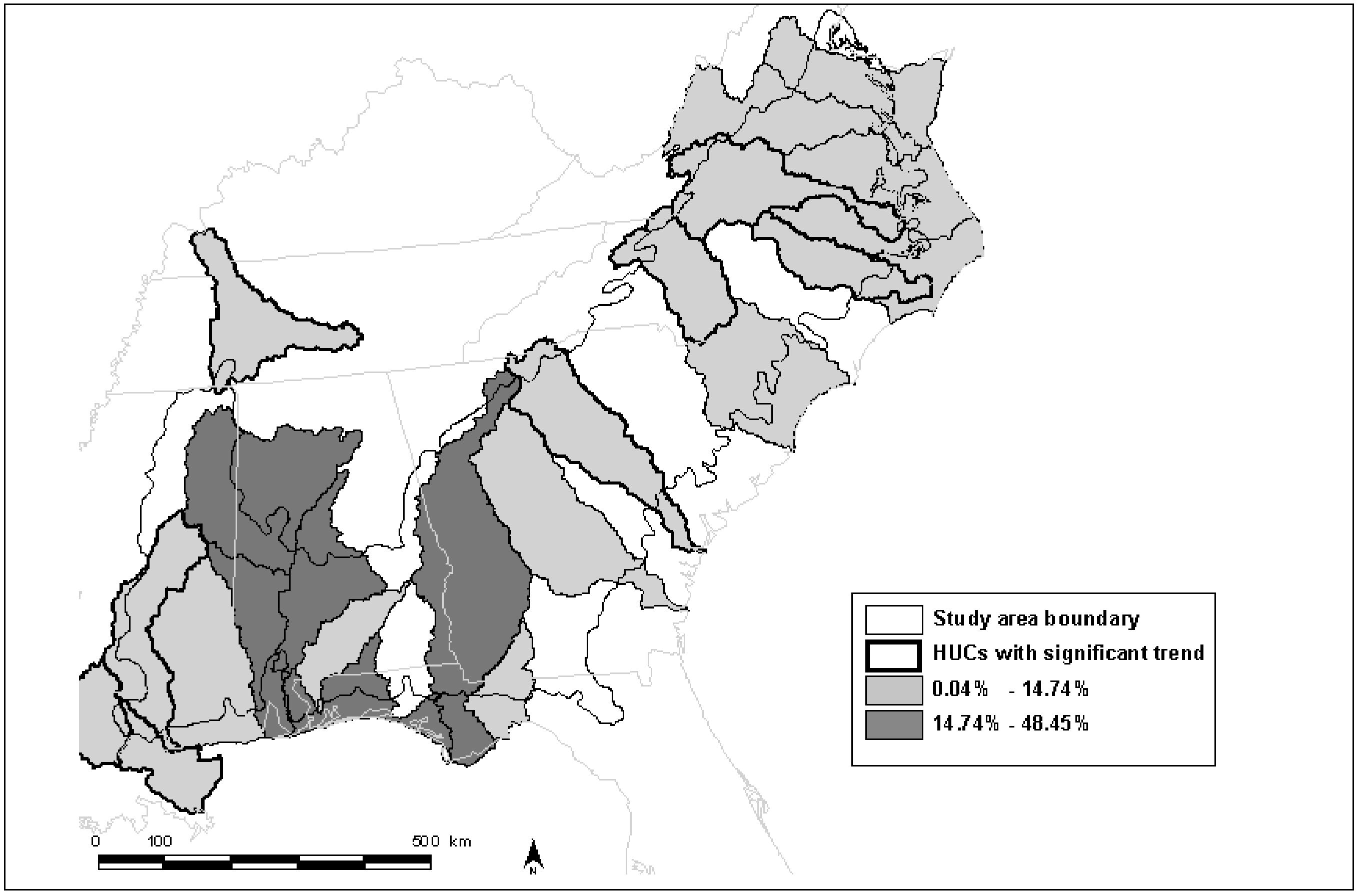

Figure 3.27. Map showing percentage area conversion of minor land to forestlands for each 6-digit HUC. HUC's with significant upward trends in pH are highlighted in bold borders. Classification is based on 12 categories of NRI data. All HUC's with significant trends in pH are located in HUC's where conversion is less than 14.74%

CHAPTER 4

RETROSPECTIVE POWER ANALYSIS OF WATER QUALITY TRENDS IN THE SOUTHEASTERN PLAINS AND PIEDMONT ECOREGIONS[1]

[1] Bjorkland, R. A. and V. Meentemeyer. To be submitted to *Water Resources Research*

Abstract

Numerous water quality studies report on time series trends of measured constituents to satisfy compliance criteria. While these summaries provide information useful for water policy and management efforts, they fail to report on the probability of committing a Type II error (reporting that no trend exists when in reality a trend is present). Power analysis is a statistical procedure that can provide a measure of confidence in accepting or rejecting the null hypothesis. However, power analysis is rarely used in water quality monitoring despite its usefulness in prospective and retrospective analysis and assessment of ongoing projects. This study is one of the first to use power analysis for water quality data collected over a large geographic area. It recognizes that this type of broad scale analyses is confounded by biases and factors ranging from small scale to large scale. Such biases include potential lack of representativeness of specific sites within monitoring networks to the biases and the inherent assumptions of the ecoregional framework that was employed. Nevertheless, it clearly demonstrates the importance of this statistical procedure in analysis of monitoring data and suggests that some reports showing no trend may be biased by insufficient length of monitoring period and/or inadequate effect size (detection level).

Power analysis is used in this study to calculate required minimum length of monitoring period and effect size for trend detection of six water quality constituents: percent saturation of dissolved oxygen (DOSAT), concentration of hydrogen or hydroxide ions (pH), concentration of two common nutrients in water, total nitrogen (TN) and total phosphorous (TP), measure of erosion, total sediment (SED), and instantaneous stream flow (FLOW). Power analysis is also used to identify stations that meet target power

values where effect size and significance values are established *a priori*. In order to demonstrate this application, water quality trends from 41 stations of the U.S. Geological Survey's (USGS) National Stream Quality Accounting Network (NASQAN) program in southeastern United States were analyzed using the seasonal Kendall test, a commonly used statistical procedure in the water industry to detect time series trends. Trend analysis was conducted at two time periods: entire length of monitoring operation (long period) and 1982 – 1992 (short period). A power analysis was conducted on those stations not reporting a trend to determine if the null hypothesis could be supported at the 80% confidence level. All stations were located only in the Southeastern Plains and Piedmont ecoregions in order to minimize differences in trend detection resulting from inherent diversity in landscape characteristics. The software TRENDS, used to calculate power, requires a measure of precision of estimates. These estimates, the coefficients of variation (CV) were calculated from water quality descriptive statistics for each station by dividing mean values by the standard deviation.

The results of this study show: 1) there is an inverse relationship between effect size and monitoring length to satisfy *a priori* criteria and between CV and power; 2) there is great variability in effect size and minimum monitoring time required to reach *a priori* established power value of 0.8; minimum monitoring time ranges from 11 years for pH to 33 years for TP and SED with effect size of 2%; 3) 11 years is adequate to detect a 2% change in pH; however SED requires a change of more than 160% and TP cannot be detected within this time period; 4) all stations in the study area achieved a power of 0.8 for pH for the long period and 27 stations achieved this power for the short period, at 2%

effect size; and 5) none of the stations satisfied the criteria of power of 0.8 and effect size of 2% for constituents TP and SED.

Reporting power results is important to strengthen observations and analyses. Results of this study suggest that many reports that currently show no trend may be biased by a Type II error as a result of insufficient data, and therefore they should not reject the null hypothesis until additional analysis is performed. Consequently, trend analyses that fail to detect trends should be viewed with caution since they may lack adequate data to satisfy a test of power analysis when conservative criteria are applied. Additionally, results point to the need for a more rigorous monitoring design that includes longer monitoring period and lower detection levels.

Keywords: power analysis; water quality trends; water quality monitoring; effect size; minimum monitoring period; TRENDS

Introduction

Information about water quality in streams and rivers is central to a community's well being. And measures of their spatial and temporal variability therefore have important social, economic and environmental implications. In order to evaluate these conditions, numerous monitoring programs have been established in the United States to collect, analyze and synthesize data. A common goal of monitoring programs is identification of the presence and direction of trends of water constituents (e.g., pathogen loadings, toxin concentrations, dissolved oxygen) in water samples. This information is also important to understand the impacts of human activities on freshwater systems. Data from trend analyses, however, may be misleading especially if the statistics initially do not indicate the presence of a trend. Failure to detect a trend does not necessarily indicate that a trend in reality does not exist. This failure may be a function of *a priori* criteria, type of statistical tests used or the nature of the data itself. Scientists and practioners now recognize the importance of identifying and controlling for this Type II error in resource management (e.g., Carr and Biedenbach 1999; Thomas 1997; Gerodette 1991; Peterman 1990) and are beginning to report its probability of occurrence when the null hypothesis is not rejected. Power analysis can help distinguish if the null hypothesis is a result of no effect or if the study design makes it unlikely that a real environmental effect can be detected (Thomas and Krebs 1997).

The goal of this paper is to describe the use of power analysis in evaluating trends in water quality and its role in monitoring programs. The two objectives are: 1) to perform a retrospective power analysis on six water constituents in the Southeastern Plains and Piedmont ecoregions to examine the relationship between estimates of power, length of monitoring period, and detectable levels of change' and 2) to conduct a prospective power analysis on these variables to identify preferred minimum monitoring times to satisfy pre-established acceptable levels of power.

Background

Trend detection involves finding a "signal" in the midst of "background" noise and it depends strongly on the number of samples and effect size (change over time). Consequently, the louder the noise or the smaller the change the more data is required for trend detection. In the absence of sensitivity tests, failure to detect a trend does not always imply there is no statistically significant change in response variable(s). Failure to reject the null hypothesis of a trend may be as much a function of study design as the absence of any "real" trend. A statistical procedure, power analysis, can be employed to distinguish between these alternatives and therefore is a critical component in designing experiments (Thomas and Krebs 1997) and retrospective analysis of the data (Thomas and Krebs 1997; Thomas 1997).

The importance of power analysis has long been recognized, but this statistical tool is not used regularly in basic or applied environmental and ecological studies (Thomas 1997; Thomas and Kreibs 1997; Peterman 1990; Green 1989; Toft and Shea 1983) despite the availability of numerous software packages to calculate power or sample size (Thomas and Krebs 1997). Gerrodette (pers. comm. 2001; 1991) argues that while power analysis on ecological data has been used primarily to detect temporal changes in the size of animal populations, it can be used to analyze water quality data along spatial and temporal gradients. Operationally, a prospective power analysis used during the experimental design of monitoring programs can help ensure that data collected satisfies statistical needs. For example, it may be used in an exploratory manner to investigate the relationship between range of sample sizes deemed feasible, effect sizes considered ecologically or environmentally important, levels of variance that historically occur, and desired alpha levels (Thomas and Krebs 1997). A retrospective power analysis is important for the interpretation of results, especially when the null hypothesis is not rejected. For example, it can be used to help assess whether or not there is a real trend in the variables of interest (Reed and Blaustein 1997). Since the actual sample size, variance and alpha level are known in this scenario, these values may be used to calculate power at the minimum effect size deemed to be important or the effect size detectable with the minimum level of power (Thomas and Juanes 1996; Toft and Shea 1983). While a retrospective power analysis may be conducted using only observed variances (Hayes and Steidl 1997), observed effect size and variances (Reed and Blaustein 1995), and neither the observed effect size nor variance (Rottenberry and Weins 1985), different approaches yield different results. The goal of the analyses should guide the approach used (Thomas 1997).

Power analysis can be used to determine necessary sample size, detectable limit (detection threshold), time required to assess trends in the data with a degree of confidence and probability of detecting a trend. While its application in designing a study (prospective power analysis) is gaining wider acceptance (Hoenig and Heisey 2001; DiStefano; Foster 2001; Nickerson and Brunell 1997; Thomas 1997) its use after the data are collected (retrospective power analysis) is still controversial (Hoenig and Heisey 2001; Hayes and Steidl 1997; Reed and Blaustein 1995). Some recent studies have used power analysis to optimize experimental design by identifying minimum required sampling size (e.g., Lovell et al. 2001; Cheruvelil et al 2000; Sheppard 1999; Somers et al. 1998) and to evaluate monitoring program effectiveness (e.g., Stirrat et al. 2001) and sensitivity of the study design to variables of interest (Marshall 2001; Evans and Viengkham 2001; Carr and Biedenbach 1999; Fore et al. 1994). While still quite uncommon, statistical power has been used in the interpretation of ongoing monitoring studies (Lougheed et al. 1999; Wilson et al. 1999). Power analysis is most frequently used in population studies of animals and plants and occasionally for other applications, such as animal behavior studies (Thomas and Juanes 1996), toxicity testing (Carr and Biedenbach 1999; Muller and Benignus 1992) water sample testing (McBride and Smith 1997; Zangrandi 1997), and decision making in resource management (Hilborn and Peterman 1996). Despite its importance for management and regulatory efforts, it is seldom cited in ecological studies and environmental reports (for example see Aulenbach and Hooper 1996; Lettenmaier et al. 1991; Smith et al. 1987).

By definition, a trend is detected when the regression has a slope significantly different from zero (Gerrodette 1987). The observation that a trend in water quality change is present when in reality it is not, is termed a Type I error (α). In contrast, a Type II error (β) occurs when the observer incorrectly concludes that a trend is not occurring when in reality it is (Table 4.1). Power, defined as 1- β, is the ability to determine if the null hypothesis (H_0) is false and the probability of rejecting the null hypothesis given that the alternative hypothesis is true (Thomas 1997; Muller and Benignus 1992; Peterman 1990; Cohen 1988; Gerrodette 1987; Toft and Shea 1983). Additionally, power analysis can be used to detect trends in the presence of concomitant variables that may obscure patterns of change (Nickerson and Brunell 1997). While

Schlesinger (1989, p.1) notes that "strong inference in science demands testing the null hypothesis…" Peterman (1990, p.2025) argues that inferences also demand "reporting of the probability of making Type II error when ecologists do not reject null hypotheses, as well as the traditional *P* (probability) values." In addition, when H_0 has not been rejected, inferences can be greatly strengthened by knowing the detectable effect size (Rothenbery and Weins 1985; Cohen 1988) for a desired β, the given sample size, and the sampling variability (Peterman 1990).

The power of a statistical test depends on five parameters: type of test, significance criterion, reliability of sample results' sample size' and effect size (Cohen 1988). Power increases with increasing sample size, effect size and higher alpha (α) levels, and declines with increasing sample variance. Variance has two components: measurement error and inherent variability in the parameter being measured. The significance level (variously called Type I error and the alpha error) implies the "critical region of rejection" of the null hypothesis and the value, set in advance of analysis, is frequently very small, such as 0.01 to 0.05. Precision of a sample value is the closeness with which it can be expected to approximate the true population values. It always depends on sample size and may be affected by observational error (e.g., inaccuracies in recording data). Power increases as precision improves. Sample size is the number of observations and involves length of monitoring period and intensity and frequency of monitoring effort. Intuitively, the larger the sample size the smaller the error and the greater the reliability of the results (Cohen 1988). However, Thas et al. (1998, p. 356) noted that increasing sampling frequency (thereby increasing sample size) inflates Type I error and increases power "which is only virtual and of no practical value". For example, increasing sampling frequency of nitrates (NO^{-3}) in water samples to more than once per month (i.e., 12-26 samples per year) affects power to a lower extent than if the data were collected less than monthly (i.e., 2 to 12 samples per year).

Effect size is the difference between the null and alternative hypotheses and it may be measured using either raw or standardized measures. Raw measures, such as differences between means or slope in a regression analysis are easier to visualize and interpret; standardized measures, such as correlation coefficients and R^2, are dimensionless and implicitly incorporate variance (Thomas 1997; Thomas and Krebs 1997). When the null hypothesis is false it is some specific nonzero value and serves as an index of degree of departure from the null hypothesis. Effect size is a determinant of power and size: the larger the effect size the greater the degree to which the phenomenon under study is manifested. For example, a large effect size provides a better power of the test if other factors (e.g. sample size) are held constant; likewise, effect size and required sample size are inversely related. The effect size is the magnitude of change in trend analysis.

Methodology

Data Source: The source of the data used in this study is the United States Geological Survey's NASQAN program. While the objectives and operating procedures for the program have been modified since its inception in 1973, it is one of the most comprehensivedatabases for a range of stream water characteristics including physical, chemical and biological properties. Monitoring stations in this program were set up in watersheds representing diverse climatic, physiographic and cultural characteristics in order to provide the best synoptic perspective of surface freshwater conditions throughout the United States (Alexander et al. 1996). Data used in this study are a subset of the NASQAN file for the southeastern region of the United States and were collected from 41 monitoring stations. Criteria used for selection of the subset include water quality constituents of interest, geographic area and data collection period.

Water Quality Constituents: Power analysis was conducted on water quality trends for six constituents: dissolved oxygen (as a percentage of saturation (DOSAT), pH, total nitrogen (TN), total phosphorous (TP), total suspended sediment (SED), and instantaneous stream flow (FLOW). These constituents were selected because they: 1) are important indicators of water quality and stream health, 2) reflect land use and other human influences on the landscape, 3) are among the most consistently reported data for stream water characterization, 4) are easily detected with available monitoring equipment and, 5) are intensively studied and monitored in research and management programs.

Geographic region: Data for the analysis came from 41 of the 43 NAWQA stations (Figure 4.1) in the southeastern United States. Data from two stations, one each on the Congaree River at Columbia, SC and the Roanoke River near Scotland Neck, NC were not used because of short monitoring periods and inconsistent reporting. Selection of the study area was based on the following criteria: 1) large, contiguous geographic area, 2) numerous monitoring stations that had long and continuous water quality records (generally exceeding 15 years), and 3) similarity in environmental characteristics and land use patterns. The relative homogeneity of ecological characteristics within ecoregions provides a convenient spatial framework for delineation of the study area. While the USGS (1999a, 1999b) base map considers the study site as one ecoregion, other maps recognize this area as consisting of two: the Southeastern Plains Ecoregion

(65) and the Piedmont Ecoregion (45) (USEPA 1997; Omernik et al. 1987). The Southeastern Plains ecoregion includes portions of nine states (AL, FL, GA, LA, MS, NC, SC, TN and VA) and occupies the area immediately to the west of the Middle Atlantic Coastal Plain and north of the Southern Coastal Plain. There is relatively little relief to the landscape and the sands, silts and clays are cretaceous or tertiary-age. The Piedmont ecoregion, defined by gently rolling hills rising to 183 meters (600 feet) and encompassing parts of five states (AL, GA, NC, SC, and VA) represents the transitional area between the Southeastern Plains and the mostly mountainous ecoregions of the Appalachians to the northwest. The Fall Line forms the boundary between the Piedmont and Southeastern Plains, and streams flowing across the Fall Line experience abrupt change in gradient marked by shoals and rapids. Much of the cultivated land in the Piedmont has reverted to forestland.

This study area is characterized as having irregular plains and a mosaic of cropland, pasture, woodland, and forest. The natural vegetation is mostly oak-hickory-pine and Southern mixed forest (USEPA 2000c). Soil types are primarily Ultisols with pockets of Alfisols, Inseptisols and Vertisols (USGS 1985). Climate of the area is characterized as humid temperate with precipitation in all seasons. Watershed size area ranges from 23 to 85,000 km^2 (9 to 32,820 mi^2), and the mean area of the 41 watersheds is 12,458 km^2 (4810 mi^2). Streams in this area are generally relatively low gradient and sandy bottomed, especially toward the coastal areas.

This region has experienced major change in land use patterns in the last quarter of the 20th century, primarily reflecting conversion of rural agricultural land to urban and suburban use and changes in types of agricultural and silviculture production. Locally, poultry and hog operations have significantly affected stream water quality. Nutrient levels and fecal coliform bacteria concentrations can be very high in waterbodies downstream of sewage treatment facilities, and the hog and poultry operations, and effluent from livestock facilities poses threats of eutrophication to surface waters. Additionally, silviculture, agriculture and urban development have contributed to sediment loads especially in areas of highly erodible soils. In contrast, streams draining relatively undisturbed and forested watersheds in the study area have low-medium concentrations of fecal coliform bacteria, dissolved solids and phosphorous (USEPA 2000a).

Data Monitoring Period: The monitoring period for this data subset extends from 1973 through 1995; however not all stations were active or had consistent reporting throughout the period. For this study, monitoring period was defined as the intervening time period during which monitoring for a given constituent was recorded at least once every six months. In those instances where there were non-recording periods exceeding six months, data prior to this recording gap were discarded. Monitoring frequency varied among stations and constituents. While most stations recorded data on a regular frequency, (monthly or bi-monthly) throughout the monitoring period, some stations monitored less frequently and/or varied the frequency throughout the monitoring period. Instantaneous stream flow was the most frequently and regularly monitored constituent while dissolved oxygen was the least monitored. In those instances where multiple values for a constituent were recorded in any month an average monthly value was substituted.

Data structure timeframe: Monitoring period for each station was subdivided and grouped into four classes in order to examine the relationship between power analysis and factors effecting monitoring efforts, such as length of time and number of observations. The four time classes were: 1) beginning date of consistent station monitoring to 01 January 1982, 2) 01 January 1982 - 31 December 1992, 3) 1993 to the end of record, and 4) the entire period, from beginning date of consistent monitoring period to end of record.

Power analysis calculation: Power analysis was calculated using the computer software TRENDS. TRENDS is a specialized power analysis software that was originally developed to calculate theoretical sampling size for animal populations during project design (Thomas and Krebs 1997; Gerrodette 1993; 1987). While this program has not yet been used with water quality data, it can be applied to both prospective and retrospective power analysis of water data. (pers. comm. Gerrodette 2001). It was selected because of ease of use, ready availability, and ability to perform analysis at the spatial scale and level of precision indicated by the type and quality of data employed in the study. Additionally, it uses a measure of the trend for effect size rather than a standardized measure such as R^2 and it can estimate power for different variance structures (Thomas and Krebs 1997).

TRENDS calculates power analysis of linear regressions by incorporating five parameters: monitoring duration (time), rate of change (effect size), precision of estimates as measured by the coefficient of variation (CV), significance level (α) and, power ($1-\beta$). Application of TRENDS is determined by the parameter of interest; values of any one parameter may be calculated if the other four are specified. Figure 4.2 shows a computer “view screen” of TRENDS.

In this study, the effect size was arbitrarily set at 0.01 and 0.02 and 0.05 (representing a linear 1%, 2% and 5% per annum change in concentrations of the constituent). The CV’s were estimated from standard deviation and mean values. These data were calculated for each water constituent at every station by the software WQStat Plus (IDT 1998). The alpha levels, set at 0.05 and 0.1, are standard in many statistical operations on environmental data.

Relationships among the five parameters are affected by a number of factors including: 1) type of change (linear or exponential), 2) direction of change (positive or negative), 3) use of 1- or 2-sided statistical test and 4) variance structure (Gerrodette 1993).

Type of change: In this study the linear option was chosen. A linear change means an equal absolute change occurs at each time step. Under this option concentrations change by a constant amount defined as a fraction of the initial amount. For example, a 2% linear change beginning with an initial value of 100 would be 100, 102, 104, 106, etc. with each increment equal to (0.02) (100) = 2 (pers. comm. T. Gerrodette 2002). These incremental changes in concentration were considered reasonable for detection and having potential environmental and health effects. Time series change in concentrations of the water constituents was assumed to be linear, and in the absence of more detailed information to the contrary, this model is usually assumed in practice (Gerrodette 1987). Additionally, values for the variables nitrogen, phosphorous, sediment and instantaneous flow rate were made linear by suitable transformations (square root or logarithm). The exponential option is best suited for use with biological data that demonstrates nonlinear patterns. For water quality monitoring, this option would be suitable for a constituent such as fecal coliform bacteria where population counts are likely to follow an exponential growth pattern.

Direction of change: There is asymmetry in the ability to detect increasing and decreasing trends. The negative (decreasing) trend is the most conservative, yielding minimum monitoring time and maximum estimates of power. The positive sign option is used when data suggest measured concentration values are increasing while the negative sign is applied when the trend is decreasing. Therefore since power was calculated prior to establishing the trend, the decision was made to calculate power using a conservative model. Additionally, since this study examined power analysis for the entire monitoring period, the per time step option was selected.

Statistical test: A two-tailed test is appropriate when there is interest in detecting a trend in either direction (increasing or decreasing). If a one-tailed test is chosen, only the change in the selected direction (positive or negative) can be detected. For example, it would not be possible to conclude that an increase occurs if a one-tailed test for a decline is chosen, *a priori* (pers. comm. T. Gerrodette 2002).

Variance structure: TRENDS offers three options for variance structure. If the CV's increased as constituent values increased, the CV proportional to SQRT (A) option was used, where "A" represents the constituent values. If the CV's decreased as data values increased, the option of CV proportional to 1/SQRT (A) was used. The third option, CV constant with (A), was used if the CV showed no relationship to constituent values. In order to establish this relationship, mean values for each station were plotted against the

CV's, the trend lines were drawn, and the regression equations calculated. Results of this procedure showed a downward trend for pH, dissolved oxygen, and instantaneous flow, an upward trend for phosphorous, and no trend for sediment and nitrogen (Figures 4.3.1-4.3.6). Therefore, the constant option was used for sediment and nitrogen, the 1/SQRT option for pH, dissolved oxygen and flow, and the SQRT option for phosphorous.

Assumptions: Utility and applicability of power analysis is predicated on meeting basic assumptions. Water quality data often violate these assumptions and to the extent they do, results provided by TRENDS are approximations. Use of transformed data and the non-parametric Seasonal Kendall test to assess for trends minimizes effects of such violations (pers. comm. T. Gerrodette 2001). TRENDS software incorporates general assumptions of linear regression models including: 1) measurement values are taken at regular temporal intervals (e.g., monthly, bi-monthly); this is the most restrictive assumption, 2) all samples are given equal weights in the regression, 3) the regression model best represents the trend of the variables, 4) estimated coefficients of variation (CV) reflect all sources of variation since power is very sensitive to CV's, 5) data have normal error distributions, equal variances, and independence of estimates, 6) change over the time period is monotonic, and 7) calculations are based on numerical approximations even if all assumptions are satisfied. In this study, most of the assumptions were met at least partially.

Trend analysis calculation: Time series trends of the six constituents were calculated by the seasonal Kendall test, a widely accepted statistical procedure in water research. The seasonal Kendall was calculated by the software WQStat Plus for the time periods 1982-1992 and the entire monitoring period.

Results and Discussion

Retrospective power analysis: This portion of the study examined the relationship between monitoring effort and coefficients of variation (CV's). Monitoring effort includes length of monitoring period and number of observations. The independent variable, CV, was used as a measure of variance (see Hayes and Steidl 1997) for the power analysis. The CV's were estimated from residual variance around the regression line, thereby including both real variability as well as measurement (Gerrodette 1987). Scatter plots and regression equations show the absence of any discernable pattern between CV's and length of monitoring period (Figures 4.4.1-4.4.6) and number of observations (Figures 4.5.1-4.5.6). R^2 values ranged from a high of 11 (monitoring time for dissolved oxygen) to a low of 0.002 (number of observations for dissolved oxygen; monitoring time for instantaneous streamflow). Failure to find a relationship between CV's and monitoring effort (length of monitoring time, number of observations) may be due to: 1) unmodelled sources of variation, 2) small changes in number of observations relative to measurement error, or 3) methods of measurement that are insensitive to small changes in concentrations (pers. comm. T. Gerrodette 2001).

TRENDS was used to calculate requisite monitoring times for preset powers of 0.8 and 0.9. from a range of generic CV values. Only results for 0.9 are shown (Figure 4.6). In this study requisite monitoring time is defined as the minimum amount of time needed to achieve a desired power for a range of predetermined effect sizes and significance levels. *A priori* effect sizes of 0.01, 0.02 and 0.05, and α level of 0.05 were established. Results show that as CV's increase requisite monitoring time increases as a curvilinear function. There is also an inverse relationship between effect size and monitoring time. For example, requisite monitoring time for a power of 0.9 is 48 years when CV = 0.4 and effect size = 0.01. When the effect size increases to 0.05, requisite monitoring time falls to 18 years. Results also show there is a little difference in requisite monitoring time between the power levels (0.8 and 0.9) with higher powers requiring more monitoring time. However, these differences appear to decrease as CV's increase and they disappear completely at the upper range of CV values.

Changes in significance level ($\alpha = 0.05$ and 0.10) seem to have little effect on requisite monitoring times at the lower range of CV values (Figure 4.7), but this difference increases as CV's increase. For example, when the CV is 0.025, there is no difference in requisite monitoring time (7 years) for an effect size = 0.02 and a power = 0.9. However, if the CV is 0.45, monitoring time is 29 and 31 years for $\alpha = 0.05$ and 0.10, respectively.

Descriptive statistics of the CV's from the data (Table 4.2) show the wide range of values among the water quality constituents. The constituent pH shows the best precision (mean value = 0.063) and sediment and phosphorous show the worst precision (mean values = 0.595 and 0.643, respectively). Frequency distribution of all station CV's for each variable are shown in Figures 4.8.1-4.8.6.

In order to identify effects of monitoring time (duration) on estimates of power, observed minimum, median, and maximum CV's for all stations were aggregated. Calculated monitoring times were plotted against power values established *a priori* for effect sizes 0.01, 0.02 and 0.05. The results (Figures 4.9.1 - 4.9.9) show a steep sigmoid function for all constituents in the three groups of CV values (minimum, maximum, median). Sediment and phosphorous show a noticeable shift to a much longer monitoring time to reach the desired power. Tables 4.3 – 4.5 summarize the estimated power for monitoring efforts of varying duration using the observed minimum CV's for effect size = 0.02 and $\alpha = 0.05$. When minimum CV values are used, time to reach a power of 0.8 ranges from 5 years for dissolved oxygen to 18 years for sediment (Table 4.3). The range for median CV values is 11 years for pH to 33 years for sediment and phosphorous (Table 4.3). When maximum CV values are used, monitoring period ranges from 20 years for pH to more than 44 years for total nitrogen (Table 4.4).

Minimum detectable effect size for selected time period: Minimum detectable effect size for the 1982 – 1992 period identified in this study (see *Methodology* section this paper) was calculated using the median CV values for the stations; *a priori* power value of 0.8 and $\alpha = 0.05$ were established. Results (Table 4.6) show that the smallest detectable limit is 2% (for pH). In contrast, sediment requires almost a doubling (164%) of the measured effect size for detection during this period. The analysis could not calculate a minimum detectable limit for phosphorous because the monitoring period was too short and limitations of the software.

Spatial distribution of power by stations: Stations were identified on the basis of whether or not a power of 0.80 could be calculated from the data for an effect size = 0.02 and α = 0.05 for the 1982 – 1992, and the entire monitoring periods. Results for pH, dissolved oxygen, total nitrogen and streamflow were mapped out (Figures 4.10-4.13.). A power of 0.8 could not be calculated for phosphorous and sediment because data for these constituents could not meet *a priori* criteria. For dissolved oxygen, most stations meeting the criteria are located in the extreme southern regions of the study site (AL and MS); however, the entire monitoring period was required to reach the desired power level.

There was widespread geographic distribution of stations meeting the power criteria for the 1982-1992 and entire monitoring periods for pH. Only 9 stations could satisfy the criteria for nitrogen; these were located in two distinct groups, one each in the northern and southern regions of the study area. Seventeen stations satisfied the criteria for streamflow and they were generally distributed throughout the study area. However, the entire monitoring period was needed to achieve the power of 0.08 for both of these constituents. These emerging geographic patterns identify where station data was adequate to calculate power as measured by CV's. Table 4.7 summarizes the number of stations meeting these criteria for effect size 0.01 and 0.02 and both time periods.

Conclusion

This study examines the application of power analysis in water quality monitoring and its utility in management of monitoring efforts. Data for six water constituents in 41 stations in the Southeastern Plains and Piedmont ecoregions were used to calculate power. Power provides a statistical measure of confidence in rejecting the null hypothesis when, in reality, an alternative hypothesis exists. While this statistical procedure is well established and there are many software packages available to compute power, it is still infrequently reported in literature. The importance of calculating power was presented in this paper.

Power was used in this study to calculate requisite station monitoring times and to identify those stations that meet target power values where effect size and significance values are established *a priori* (Peterman 1990; Cohen 1988; Rothenbery and Weins 1985). Dissolved oxygen, pH, and streamflow had the largest number of stations that met target power values of 0.8 for the entire length of the monitoring period. Median monitoring time for the stations varied significantly among the constituents (Table 4.8.). Based on the calculated CV's from station data for each constituent (Tables 4.9 - 4.11), requisite monitoring time for many stations for all constituents far exceeds the actual monitoring period; this scenario is particularly true for variables nitrogen, phosphorous and sediment. For example, the two variables with the best measures of precision, pH and dissolved oxygen, require a minimum of 11 and 14 years respectively to satisfy the criteria for a power with an effect size of 2%. However, mean length of monitoring time of all stations in the study area was 21 years for pH 12.5 years for dissolved oxygen. While many stations met the criteria for pH, most stations records were not long enough to support the null hypothesis of no trend for other constituents. Additionally, interruption of recording periods as a result of programmatic changes and equipment malfunction or other contingency shortened the actual period of data that can be used to accurately and precisely characterize status and trends of water quality.

Another finding of interest is the relatively large change in constituent concentration and flow necessary for detection at a statistically significant level. Analysis of the minimum detectable effect over the 10-year period (1982-1992) suggests that a 2% and 3% change in values is needed for pH and oxygen, respectively. The flow-adjusted variables sediment and phosphorous required changes in effect size >100% during this period.

There are some management lessons that can be drawn from this study. While the NASQAN data is a valuable source of information for water quality, the operational period is not sufficient to provide incontrovertible evidence of the absence of trends over time for some constituents, especially those in low concentrations. This study also suggests that there is great variability in effect size or requisite monitoring time within the same ecoregion. This variability may be a function of inherent differences in stream processes and composition at the measured site as well as disparities in monitoring procedures for the entire study area.

The target power of 0.8 used in many parts of the study is a reasonable level. However, this value varies depending on management and research applications and considerations of opportunity costs and ethical tradeoffs (Muller and Benignus 1992). Incorporating power analysis in the analysis of water quality data would serve to strengthen the validity of the observations. Power analysis can be a helpful tool in guiding the design of future and on-going monitoring programs as well as evaluating historical water quality data.

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Table 4.1. Outcomes of trend testing. (Taken from Helsel and Hirsch 1992).

	True Situation	
Decision	**H_0 TRUE**	**H_0 FALSE**
Fail to Reject H_0 (No Trend)	Probability $(1-\alpha)$	Type II error (β)
Reject H_0 (TREND)	(Type I error) Significance level α	(Power) $1-\beta$

Table 4.2. Descriptive statistics on estimated CV's

CONSTITUENT	MIN	MAX	MEAN	MEDIAN	95^{th} Percentile	MODE
DOSAT	0.015	0.610	0.108	0.095	0.206	0.131
pH	0.023	0.183	0.063	0.060	0.115	0.047
SED	0.168	4.581	0.595	0.517	1.238	0.5
TP	0.156	4.496	0.643	0.589	1.137	0.333
TN	0.017	0.901	0.210	0.193	0.407	0.25
FLOW	0.036	0.584	0.180	0.150	0.442	0.118

Table 4.3. Estimated power for range of monitoring durations using the observed minimum CV's (in parentheses) and an effect size of 0.02. ($\alpha = 0.05$).

	Estimated Power					
MONITORING TIME (yrs.)	**DOSAT (0.015)**	**pH (0.023)**	**SED (0.168)**	**TP (0.156)**	**TN (0.017)**	**FLOW (0.036)**
1	0.05	0.05	0.05	0.05	0.05	0.05
2	0.07	0.06	0.05	0.05	0.07	0.05
3	0.12	0.08	0.05	0.05	0.11	0.07
4	0.39	0.21	0.05	0.05	0.34	0.12
5	0.81	0.48	0.06	0.06	0.73	0.24
6	0.99	0.79	0.07	0.07	0.97	0.44
7	1	0.96	0.09	0.09	1	0.68
8		1	0.11	0.12		0.87
9			0.14	0.16		0.97
10			0.18	0.22		1
11			0.24	0.29		
12			0.3	0.37		
15			0.57	0.69		
18			0.84	0.93		
20			0.94	0.99		
25			1	1		

Table 4.4. Estimated power for range of monitoring durations used the observed maximum CV's (in parentheses) and effect size of 0.02 (α = 0.05). Power could not be calculated (N/C) for SED and TP due to the large CV values.

MONITORING TIME (yrs.)	DOSAT (0.610)	pH (0.183)	SED (4.581)	TP (4.496)	TN (0.901)	FLOW (0.584)
1						
2	0.05	0.05	N/C	N/C	0.05	0.05
3	0.05	0.05	"	"	0.05	0.05
4	0.05	0.05	"	"	0.05	0.05
5	0.05	0.06	"	"	0.05	0.05
6	0.05	0.07	"	"	0.05	0.05
7	0.05	0.08	"	"	0.05	0.05
8	0.05	0.1	"	"	0.05	0.05
9	0.06	0.12	"	"	0.05	0.05
10	0.06	0.15	"	"	0.05	0.06
11	0.06	0.19	"	"	0.06	0.06
12	0.07	0.24	"	"	0.06	0.07
15	0.09	0.45	"	"	0.07	0.09
18	0.12	0.7	"	"	0.09	0.12
20	0.14	0.84	"	"	0.1	0.15
25	0.26	1	"	"	0.17	0.27
30	0.43		"	"	0.29	0.46
35	0.65		"	"	0.46	0.69
36	0.69		"	"	0.5	0.73
40	0.85		"	"	0.63	0.88
45	0.96		"	"	0.86	0.97
50	1		"	"	0.97	1

Table 4.5. Estimated power for range of monitoring durations used the observed median CV's (in parentheses) and effect size of 0.02 (α = 0.05).

MONITORING TIME (yrs.)	DOSAT (0.095)	pH (0.06)	SED (0.517)	TP (0.589)	TN (0.193)	FLOW (0.150)
1	0.05	0.05	0.05	0.05	0.05	0.05
2	0.05	0.05	0.05	0.05	0.05	0.05
3	0.05	0.06	0.05	0.05	0.05	0.05
4	0.06	0.08	0.05	0.05	0.05	0.05
5	0.08	0.12	0.05	0.05	0.06	0.06
6	0.11	0.2	0.05	0.05	0.06	0.07
7	0.16	0.32	0.05	0.05	0.08	0.09
8	0.22	0.47	0.06	0.05	0.09	0.12
9	0.31	0.64	0.06	0.06	0.12	0.15
10	0.42	0.79	0.06	0.06	0.15	0.2
11	0.55	0.91	0.07	0.07	0.19	0.26
12	0.67	0.97	0.08	0.07	0.24	0.33
15	0.94	1	0.11	0.1	0.46	0.6
18	1		0.16	0.15	0.73	0.86
20			0.21	0.2	0.87	0.95
25			0.41	0.39	1	1
30			0.68	0.67		
35			0.9	0.9		
36			0.93	0.93		
40			0.99	0.99		
45			1	1		
50						

Table 4.6. Minimum detectable effect size for each constituent with *a priori* power o 0.80 using observed median CV values. N/C indicates effect size estimate could not be calculated.

Constituent	Median CV	Power	Minimum Detectable Effect (11 yrs)
DOSAT	0.095	0.8	.03
pH	0.060	0.8	.02
SED	0.517	0.8	1.64
TP	0.589	0.8	N/C
TN	0.193	0.8	.08
FLOW	0.150	0.8	.05

Table 4. 7. Number of stations in study area for which power estimates of trend results were ≥ 0.80 for effect sizes (0.01, 0.02) and 1982-1992 and entire monitoring period.

Constituent	0.01 Effect Size, 1982-1992	0.01 Effect Size, Entire Period	0.02 Effect Size, 1982-1992	0.02 Effect Size, Entire Period
DOSAT	0	3[2]	4	15
pH	3[3]	38	27	41
TN	0	0	0	10
TP	0	0	0	0
SED	0	0	0	0
FLOW	0	0	2[4]	17

[2] Station identification numbers: 02212600; 02375500; 02376500
[3] Station identification numbers: 02212600; 02343801; 02353000
[4] Station identification numbers: 02197300; 03593005

Table 4.8 . Length of monitoring period, in years for each constituent and all stations in study area. Values reflect only those periods with uninterrupted monitoring schedules.

Constituent	Mean	Maximum	Minimum	STDev
DOSAT	12.45	24.93	3.84	3.93
pH	20.68	31.85	12.93	4.03
SED	17.42	23.35	9.42	3.81
TP	17.08	22.93	4.09	4.84
TN	17.60	22.68	10.34	3.47
FLOW	18.80	29.61	10.67	4.07

Table 4.9. Minimum monitoring duration, in years, required for pre-determined power levels for three effect sizes (0/01, 0.02, and 0.05), calculated using observed minimum CV values (in parentheses).

4.9.1. Effect size = 0.01

Desired Power	DOSAT (0.015)	pH (0.023)	TN (0.017)	TP (0.156)	SED (0.168)	FLOW (0.036)
0.50	6	8	7	21	23	10
0.75	7	9	8	25	27	11
0.80	7	9	8	26	28	12
0.90	8	10	8	29	31	13
0.95	8	11	9	30	33	14

4.9.2 Effect size = 0.02

Desired Power	DOSAT (0.015)	pH (0.023)	TN (0.017)	TP (0.156)	SED (0.168)	FLOW (0.036)
0.50	5	6	5	14	15	7
0.75	5	6	6	16	17	8
0.80	5	7	6	17	18	8
0.90	6	7	6	18	20	9
0.95	6	7	6	19	21	9

4.9.3. Effect size = 0.05

Desired Power	DOSAT (0.015)	pH (0.023)	TN (0.017)	TP (0.156)	SED (0.168)	FLOW (0.036)
0.50	4	4	4	8	9	5
0.75	4	5	4	9	10	5
0.80	4	5	4	9	10	5
0.90	4	5	4	10	11	6
0.95	5	5	5	11	11	6

Table 4.10. **Minimum monitoring duration, in years, required for pre-determined power levels for three effect sizes (0/01, 0.02, and 0.05), calculated using observed maximum CV values (in parentheses).**

4.10.1. Effect size = 0.01

Desired Power	DOSAT (0.610)	pH (0.183)	TN (0.901)	TP (4.496)	SED (4.581)	FLOW (0.584)
0.50	52	25	33	N/C[5]	N/C	50
0.75	N/C	30	47	N/C	N/C	N/C
0.80	N/C	31	50	N/C	N/C	N/C
0.90	N/C	34	N/C	N/C	N/C	N/C
0.95	N/C	36	N/C	N/C	N/C	N/C

4.10.2. Effect size =0.02

Desired Power	DOSAT (0.610)	pH (0.183)	TN (0.901)	TP (4.496)	SED (4.581)	FLOW (0.584)
0.50	32	16	36	N/C	N/C	31
0.75	38	19	42	N/C	N/C	37
0.80	39	20	44	N/C	N/C	38
0.90	42	22	47	N/C	N/C	41
0.95	45	23	N/C	N/C	N/C	44

4.10.3. Effect Size = 0.05

Desired Power	DOSAT (0.610)	pH (0.183)	TN (0.901)	TP (4.496)	SED (4.581)	FLOW (0.584)
0.50	17	5	19	N/C	N/C	17
0.75	20	6	22	N/C	N/C	20
0.80	21	6	23	N/C	N/C	20
0.90	22	7	25	N/C	N/C	22
0.95	24	7	28	N/C	N/C	23

[5] N/C = Could not be calculated for this combination..

Table 4.11. Minimum monitoring duration, in years, required for pre-determined power levels for three effect sizes (0/01, 0.02, and 0.05), calculated using observed median CV values (in parentheses)..

4.11.1. Effect size = 0.01

Desired Power	DOSAT (0.095)	pH (0.06)	TN (0.193)	TP (0.589)	SED (0.517)	FLOW (0.150)
0.50	17	13	26	45	44	22
0.75	20	15	31	N/C[6]	N/C	27
0.80	21	16	32	N/C	N/C	28
0.90	23	17	35	N/C	N/C	30
0.95	24	19	37	N/C	N/C	32

4.11.2. Effect size =0.02

Desired Power	DOSAT (0.095)	pH (0.06)	TN (0.193)	TP (0.589)	SED (0.517)	FLOW (0.150)
0.50	11	9	16	28	27	14
0.75	13	10	19	32	32	17
0.80	14	11	19	33	33	18
0.90	15	11	21	35	35	19
0.95	16	12	22	37	37	20

4.11.3. Effect size = 0.05

Desired Power	DOSAT (0.095)	pH (0.06)	TN (0.193)	TP (0.589)	SED (0.517)	FLOW (0.150)
0.50	7	5	9	20	20	8
0.75	8	6	11	25	25	10
0.80	8	6	11	27	27	10
0.90	9	7	12	30	30	11
0.95	9	7	12	33	33	12

[6] N/C = Could not be calculated for this combination.

Figure 4.1. Water Quality Network Monitoring Stations in study area.

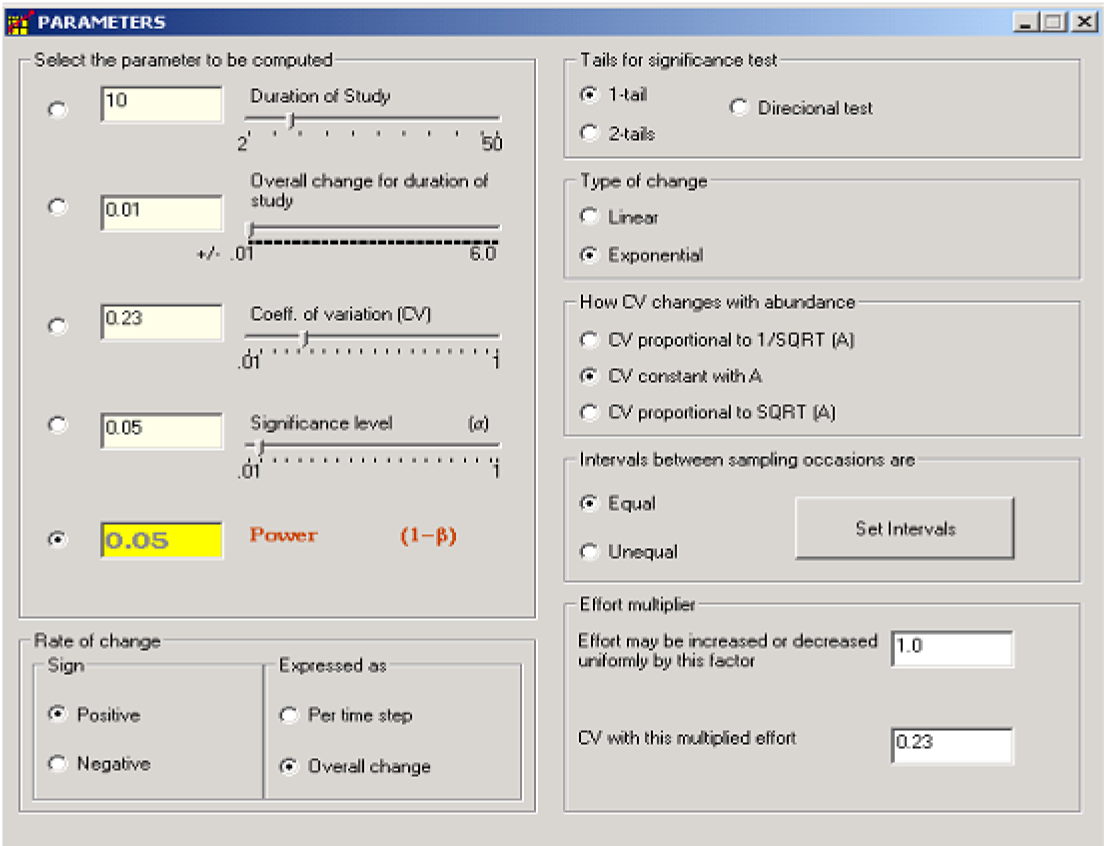
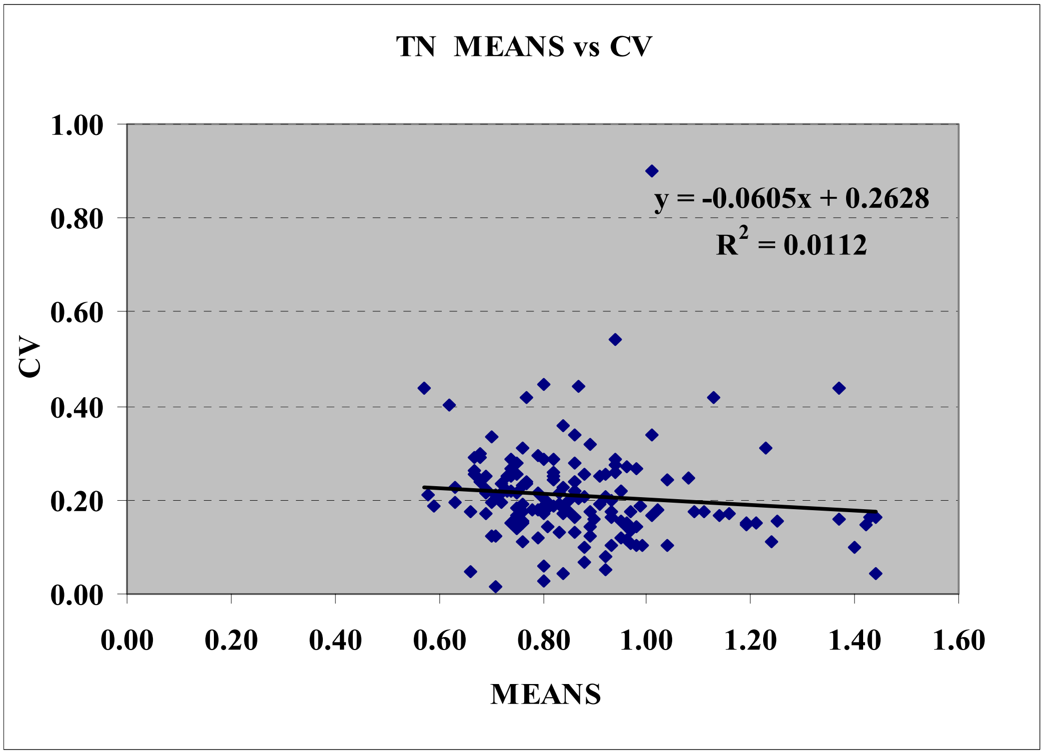
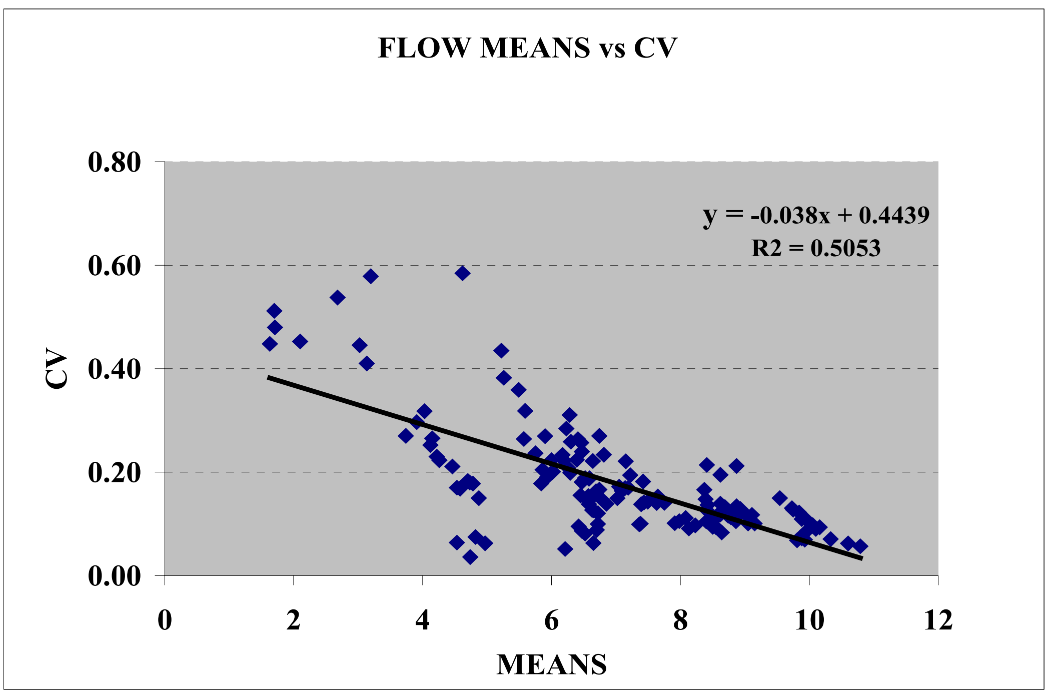
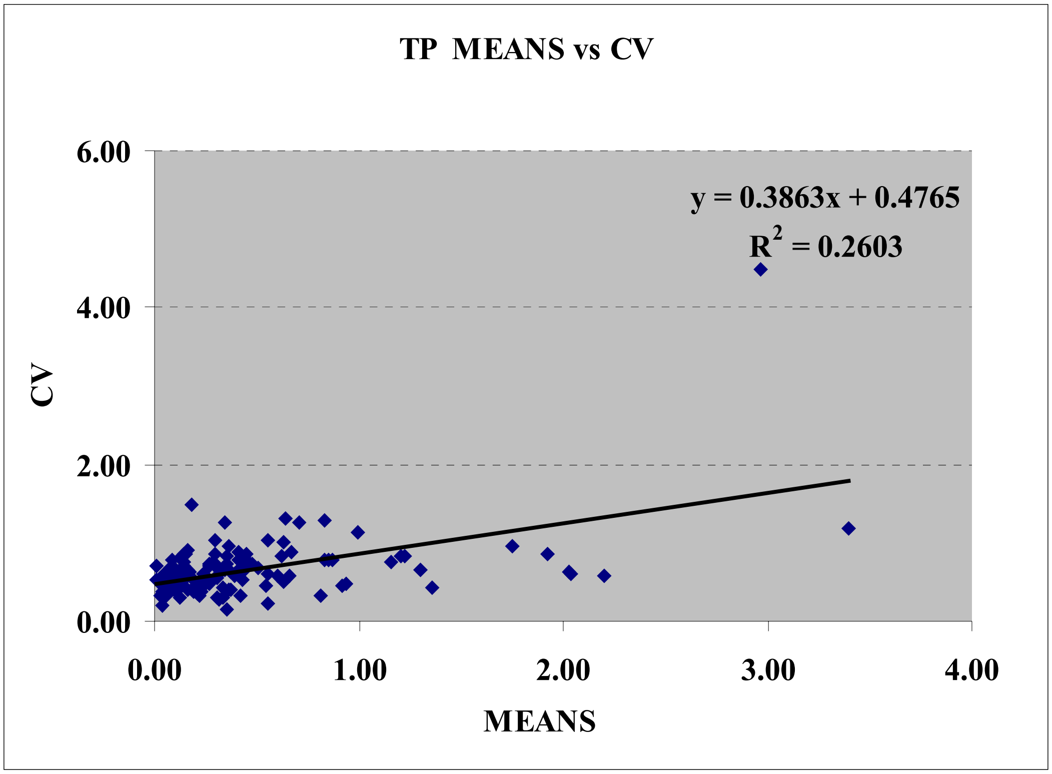


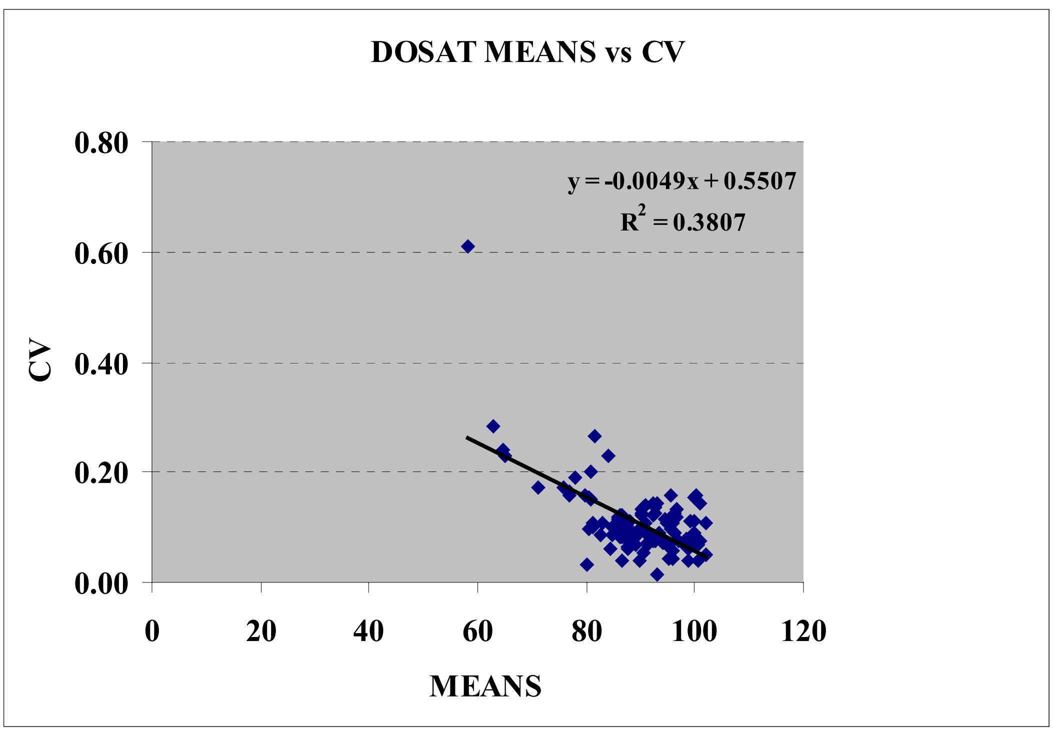
Figure 4.2. Screen view of program TRENDS, the stand-alone power analysis software used in this study to estimate, power, effect size and minimum monitoring duration.



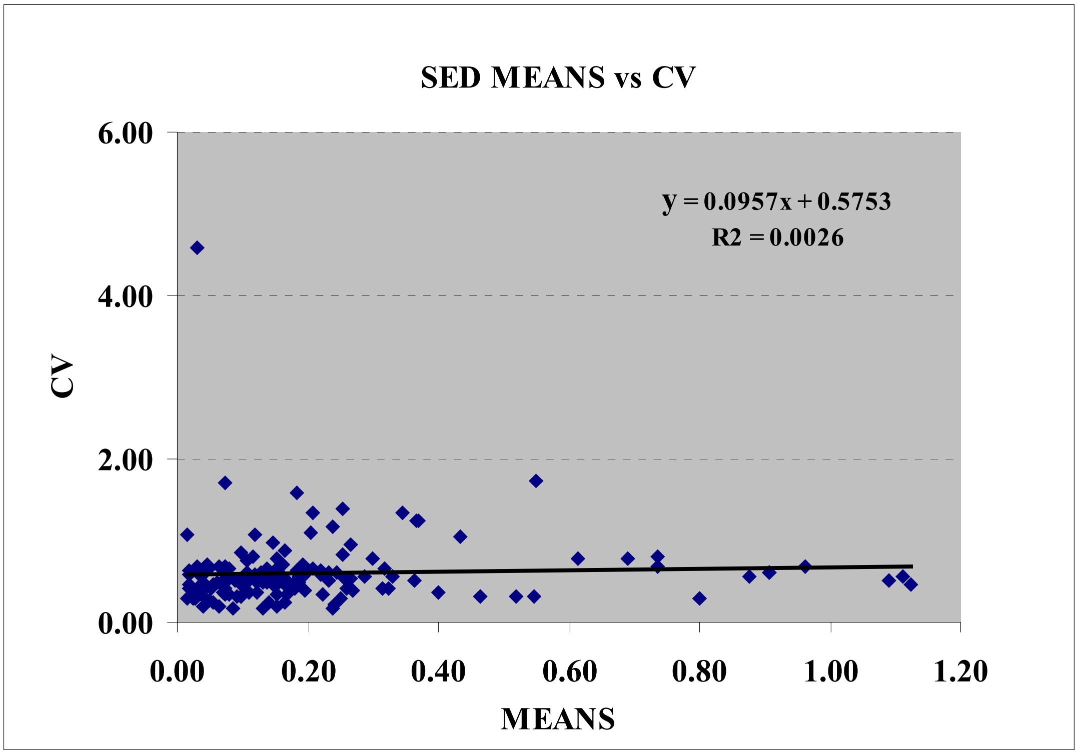


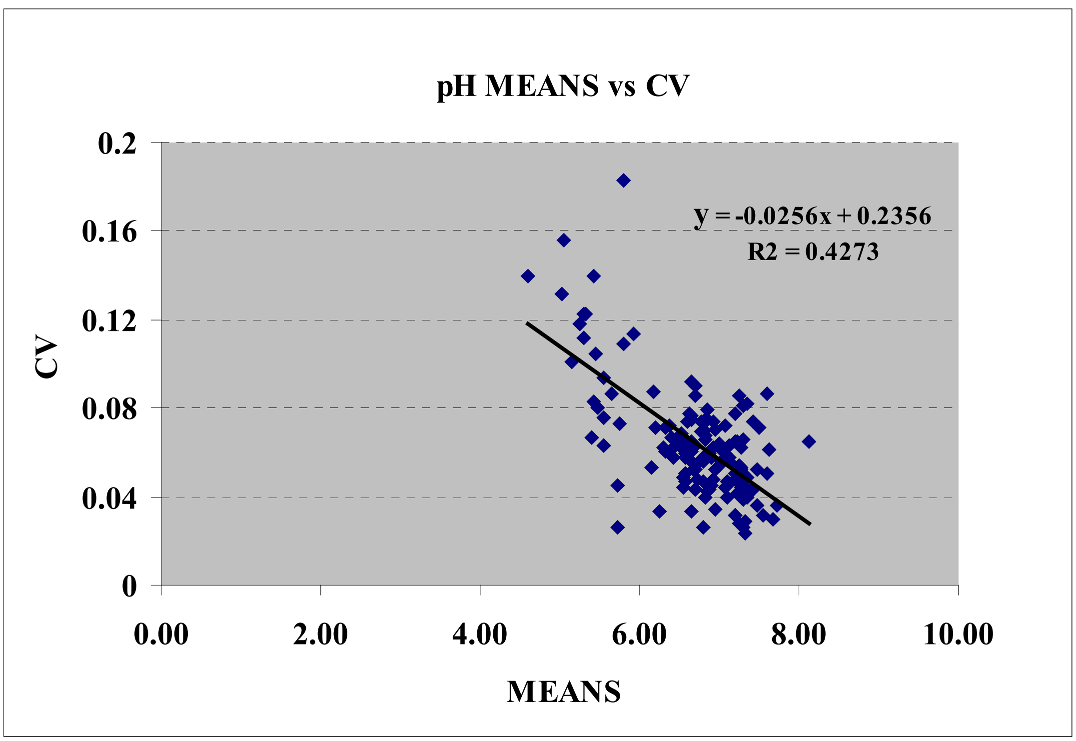
Figures 4.3.1. (upper) and 4.3.2. (lower). Relationship between coefficient of variation and mean concentration of nitrogen (TN) and instantaneous streamflow (FLOW), respectively.



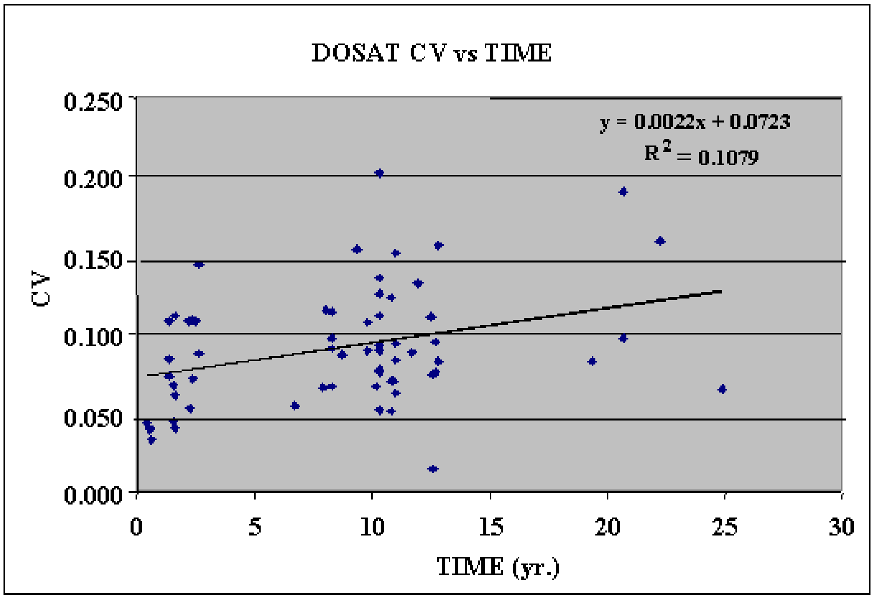


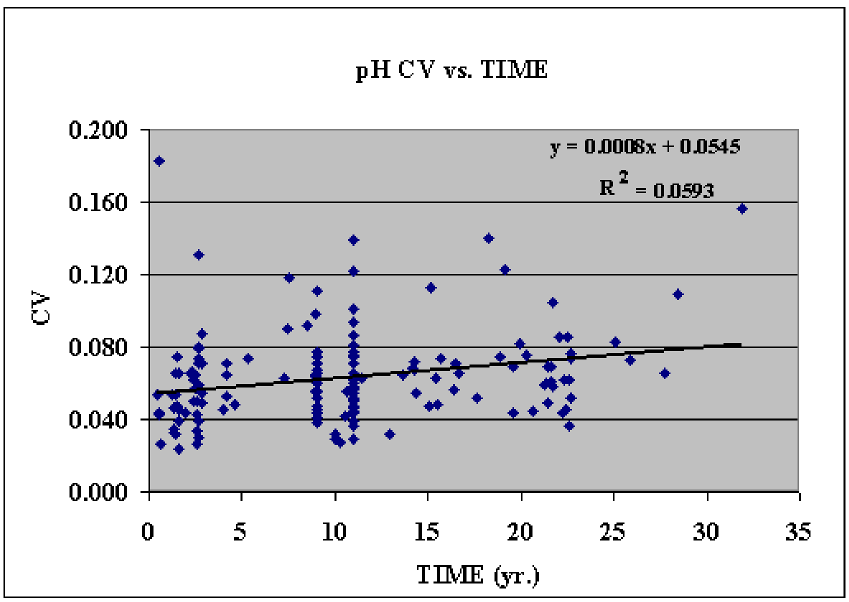
Figures 4.3.3. (upper) and 4.3.4. (lower). Relationship between coefficient of variation and mean concentration of phosphorus (TP) and dissolved oxygen (DOSAT), respectively.



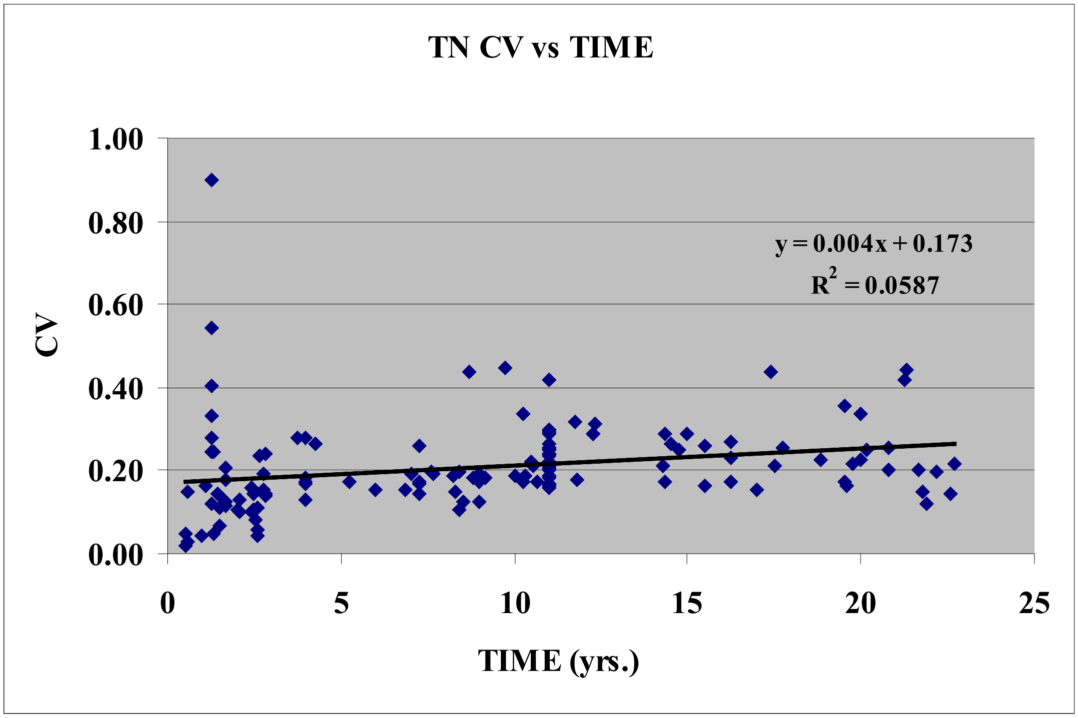


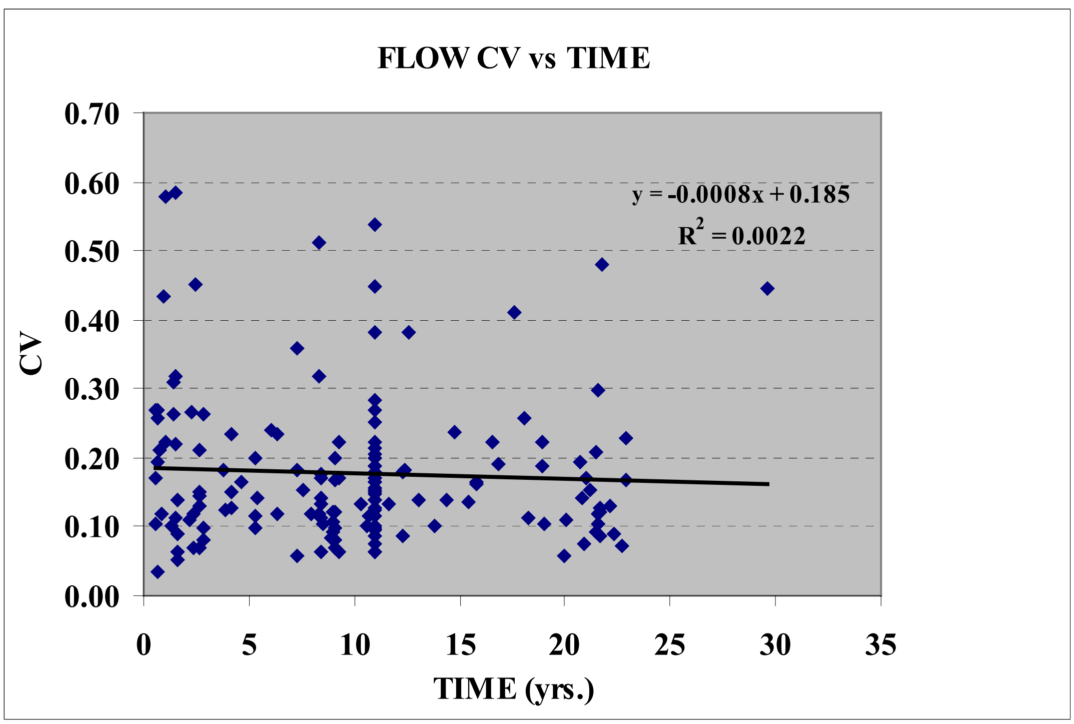
Figures 4.3.5. (upper) and 4.3.6. (lower). Relationship between coefficient of variation and mean concentration of sediment (SED) and pH, respectively.



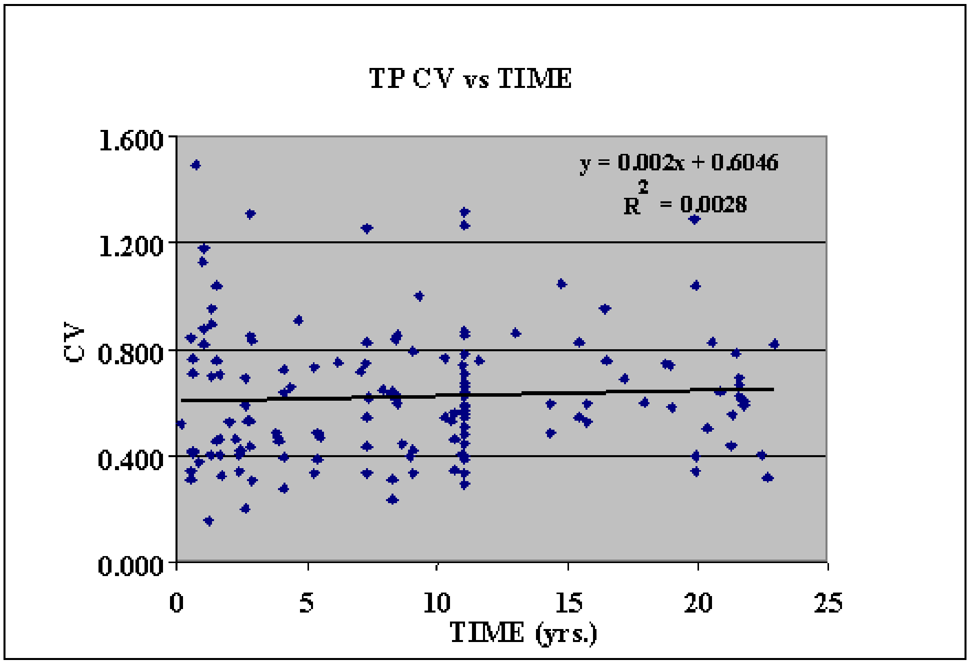


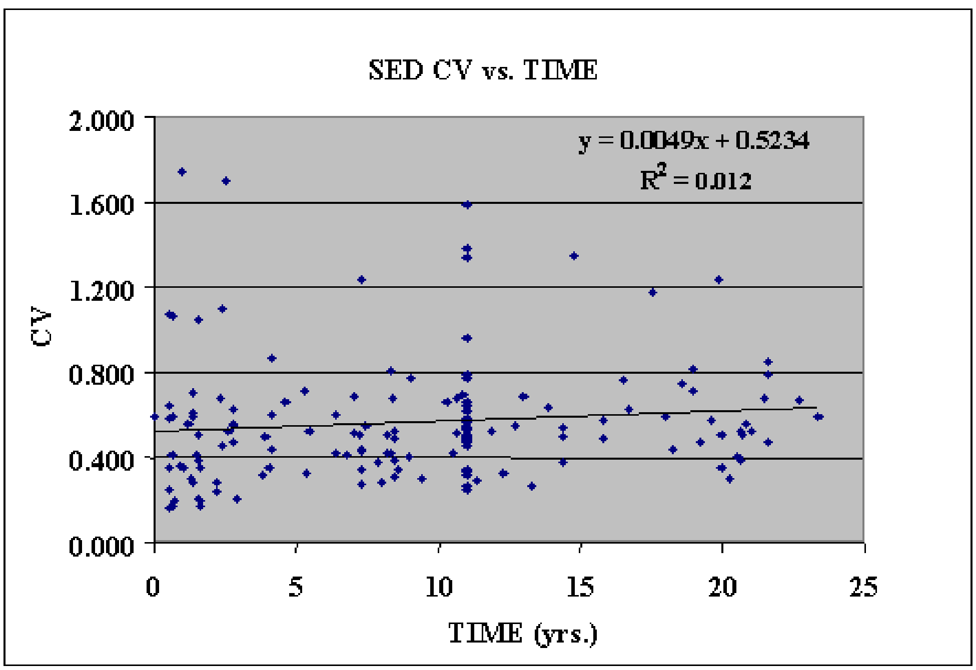
Figures 4.4.1. (upper) and 4.4.2. (lower). The relationship between coefficient of variation and length of monitoring of constituents dissolved oxygen (DOSAT) and pH, respectively.



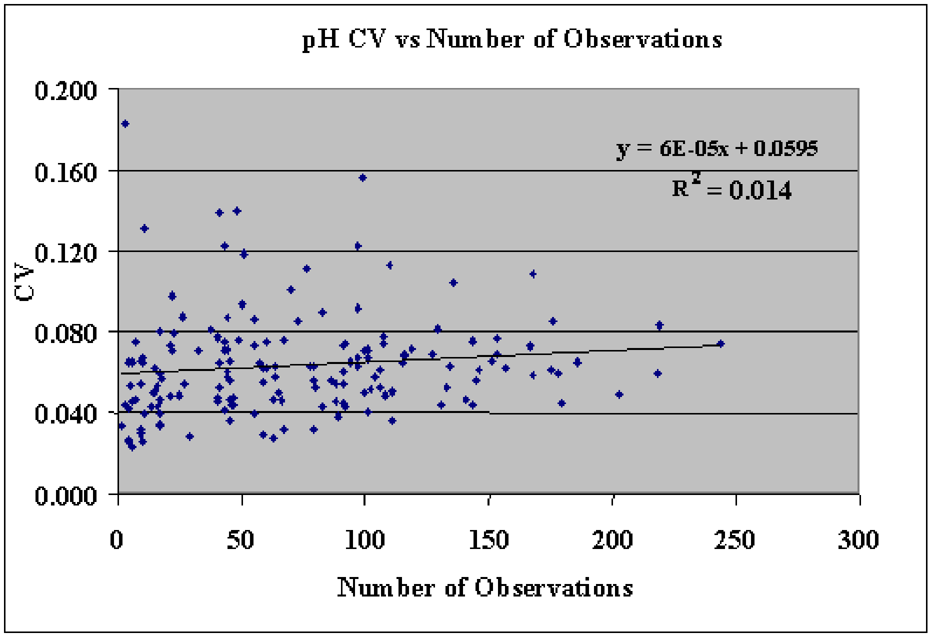


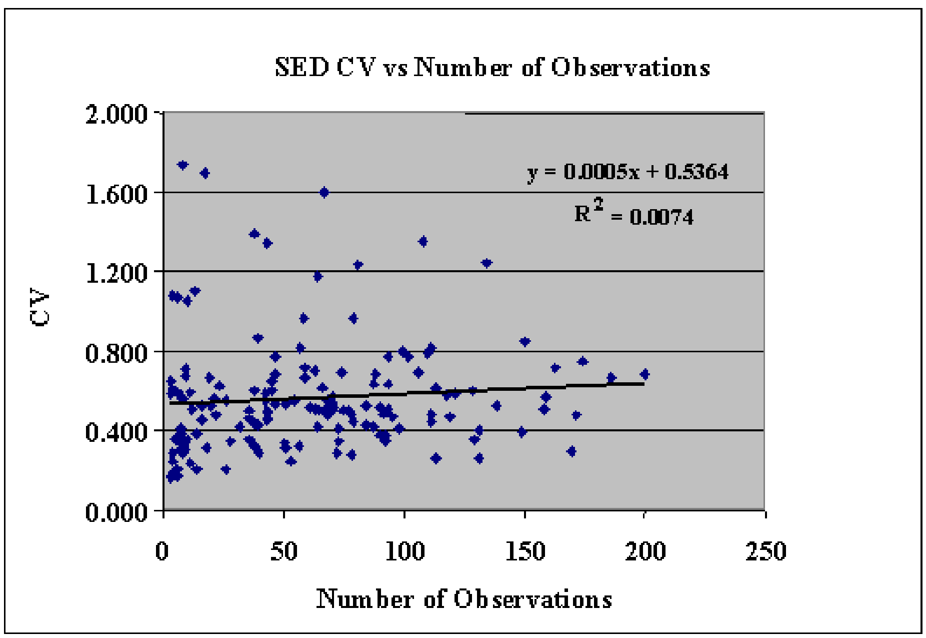
Figures 4.4.3. (upper) and 4.4.4. (lower). The relationship between coefficient of variation and length of monitoring of constituent nitrogen (TN) and streamflow (FLOW), respectively.



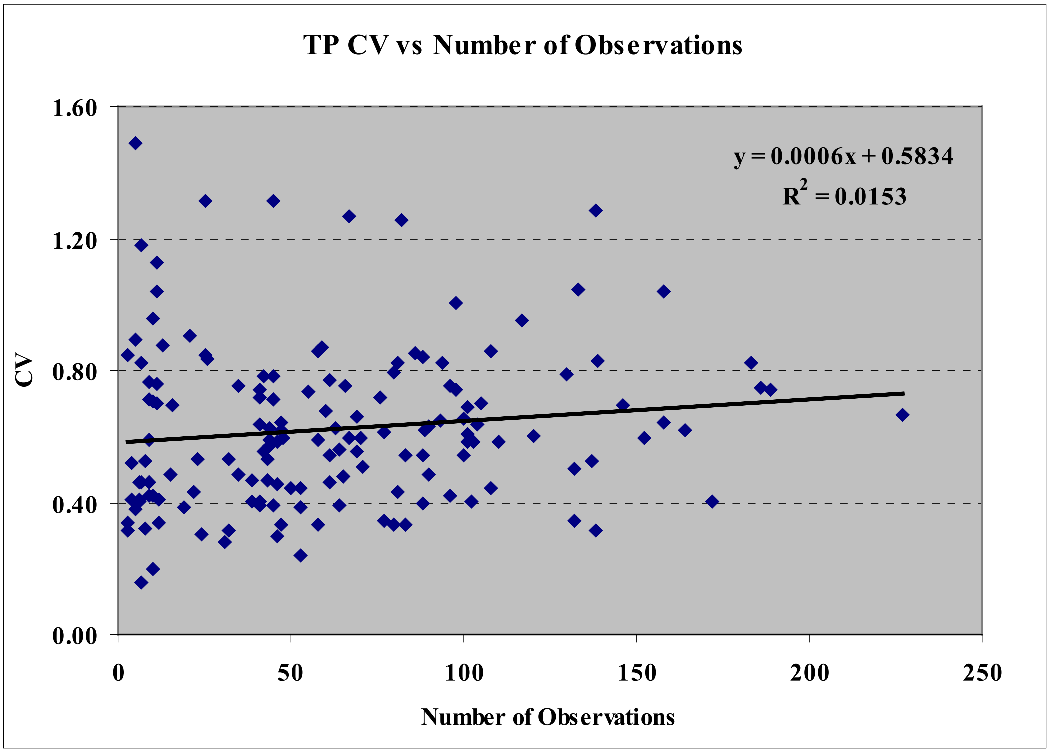


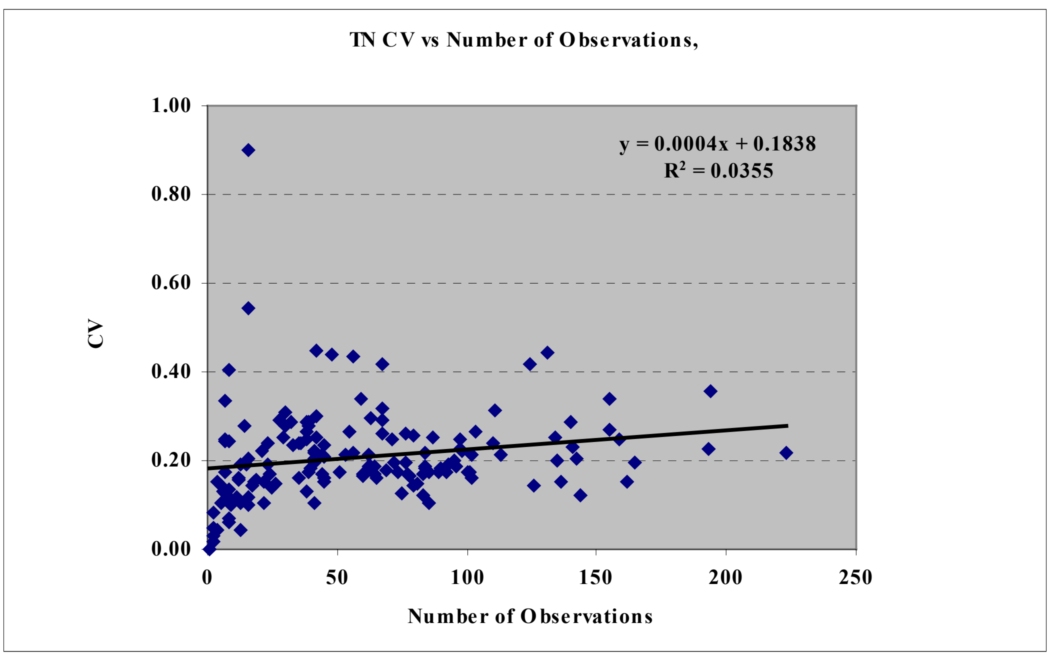
Figures 4.4.5. (upper) and 4.4.6. (lower). The relationship between coefficient of variation and length of monitoring of constituents phosphorus (TP) and sediment, (SED) respectively.



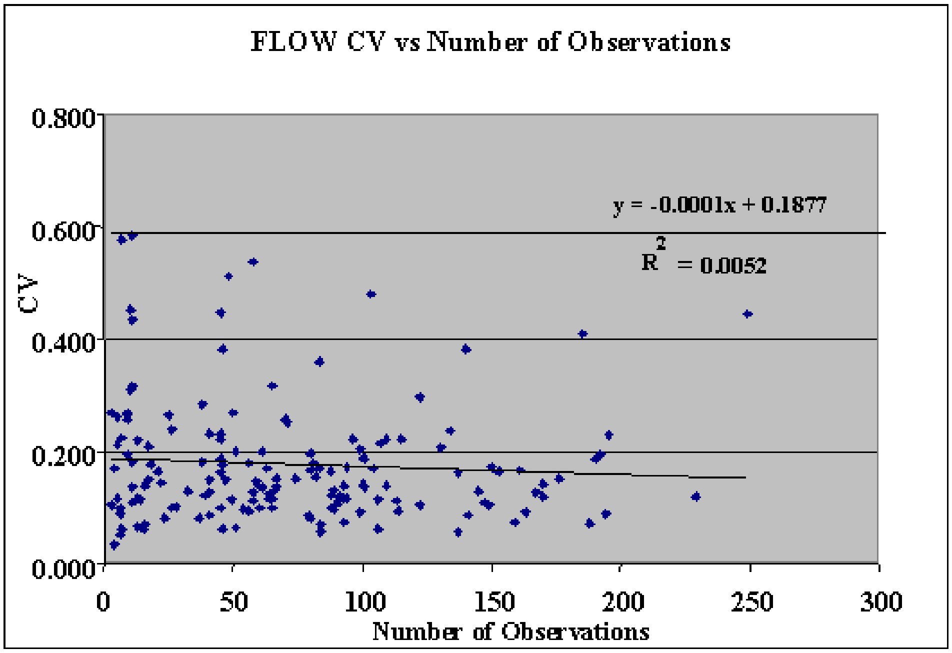


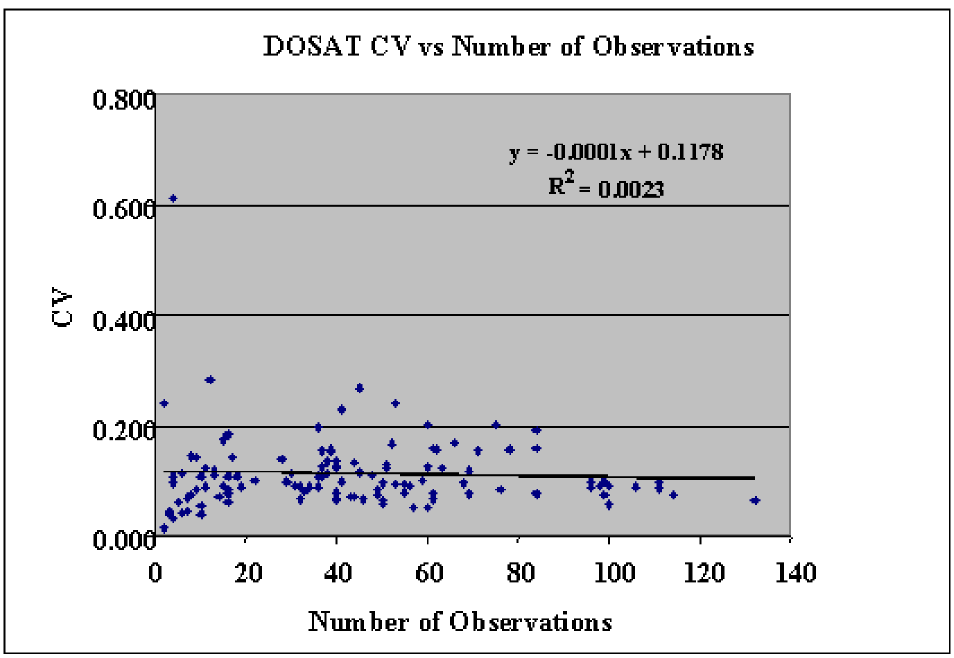
Figures 4.5.1. (upper) and 4.5.2. (lower). The relationship between coefficient of variation and number of station observations. For constituents pH and sediment, (SED) respectively.





Figures4.5.3. (upper) and 4.5.4. (lower). The relationship between coefficient of variation and number of station observations. for constituents phosphorus (TP) and nitrogen (TN) respectively





Figures4.5.5. (upper) and 4.5.6. (lower). The relationship between coefficient of variation and number of station observations. for constituents instantaneous streamflow (FLOW) and dissolved oxygen (DOSAT).

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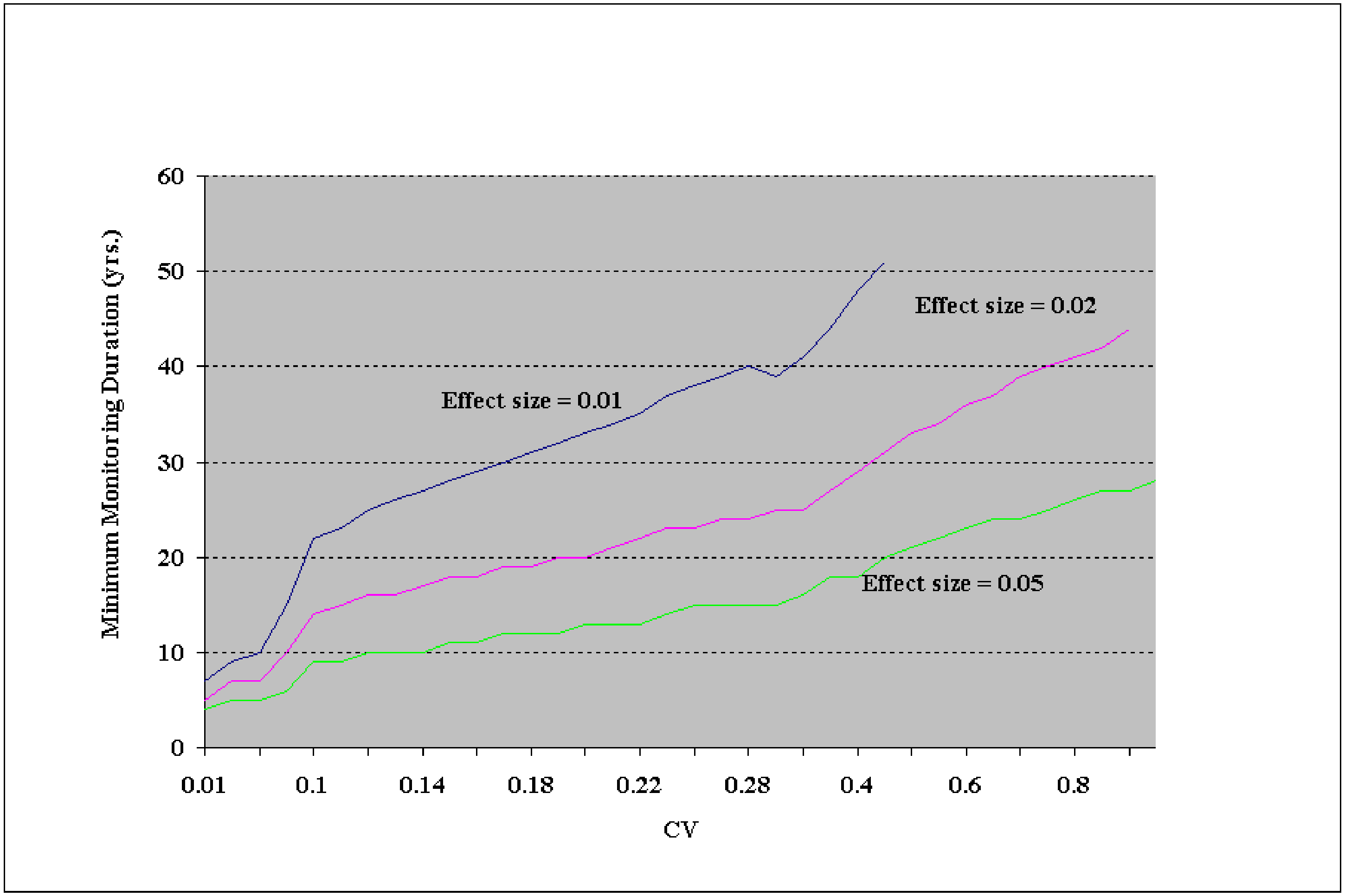


Figurre 4.6. The relationship between coefficient of variation and monitoring durations required to achieve 0.9 power for three effect sizes.

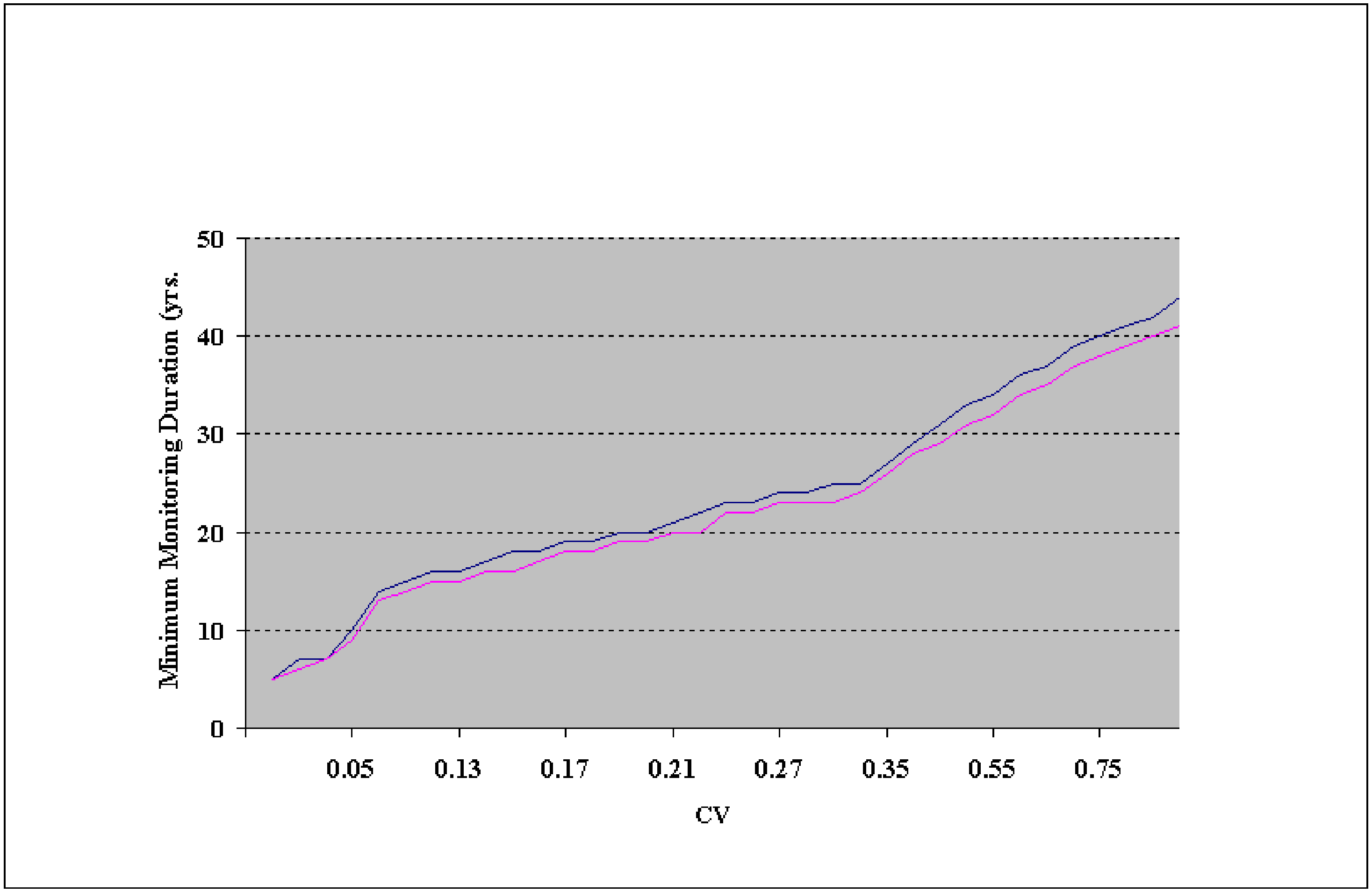
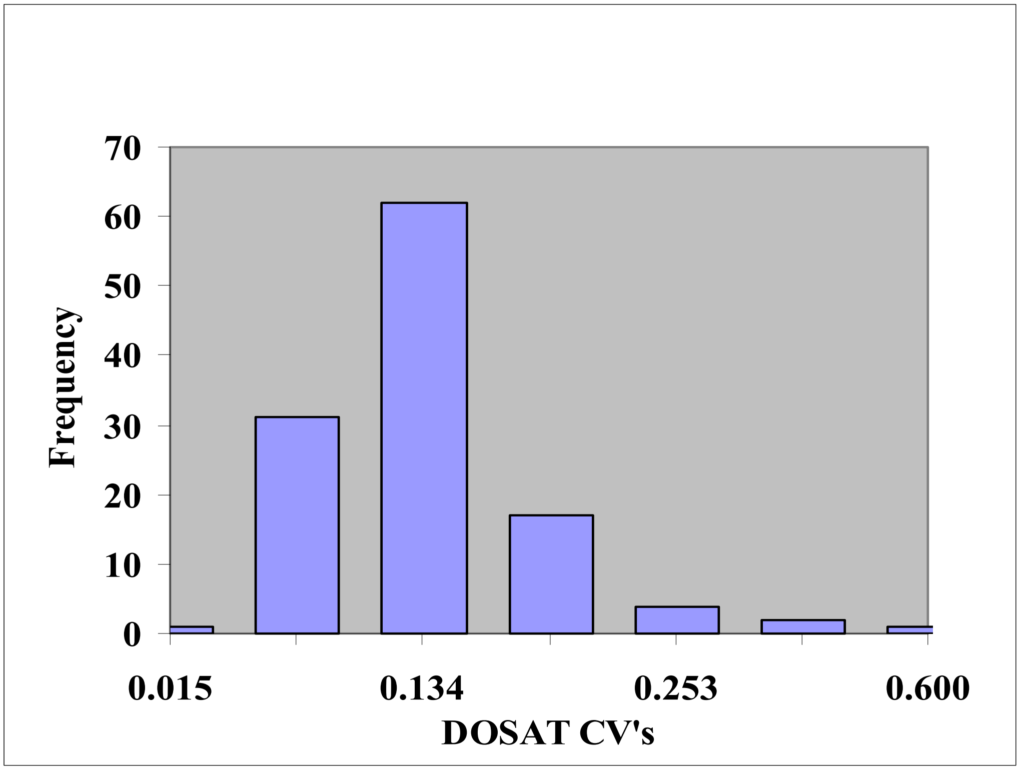
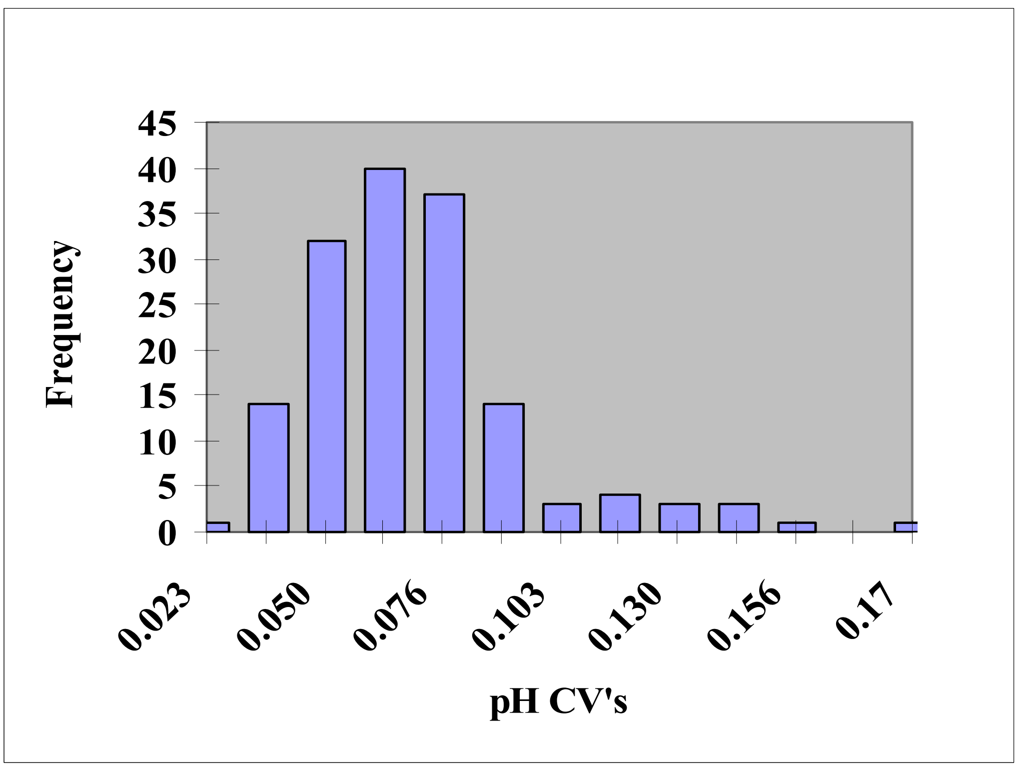
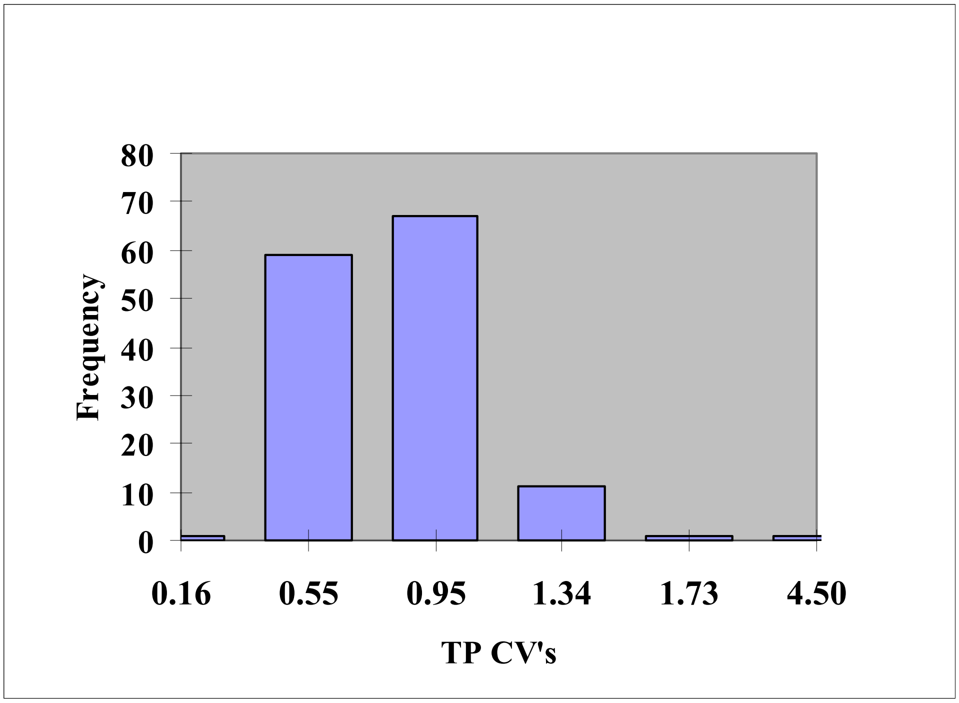


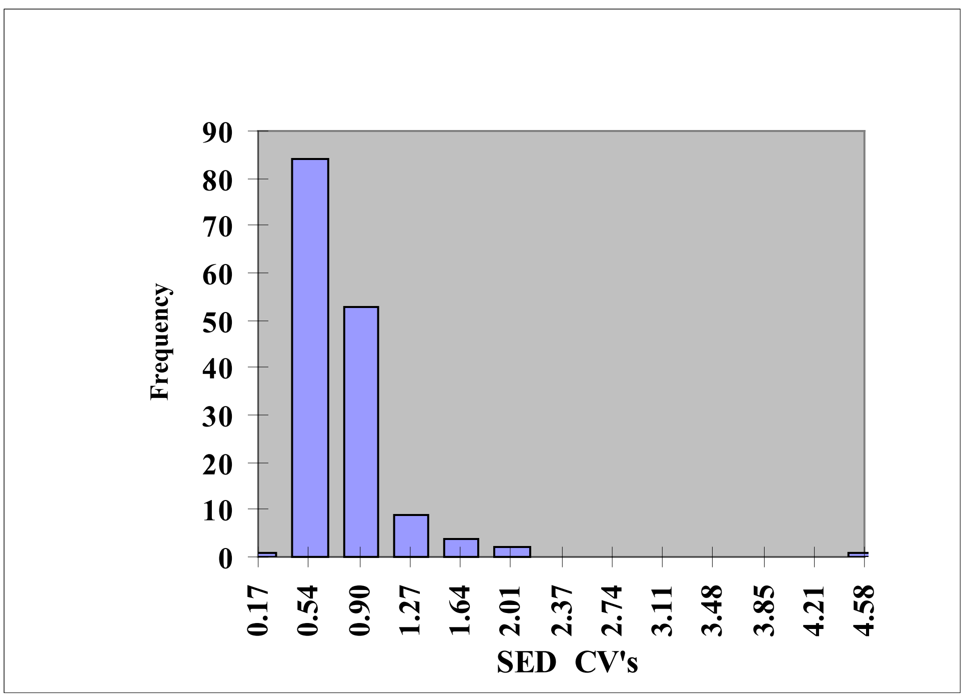
Figure 4.7. The relationship between coefficient of variation and the minimum survey duration required to detect a 0.02 effect (trend) with 0.9 power, for $\alpha = 0.05$ (blue line)and 0.1 (purple line).



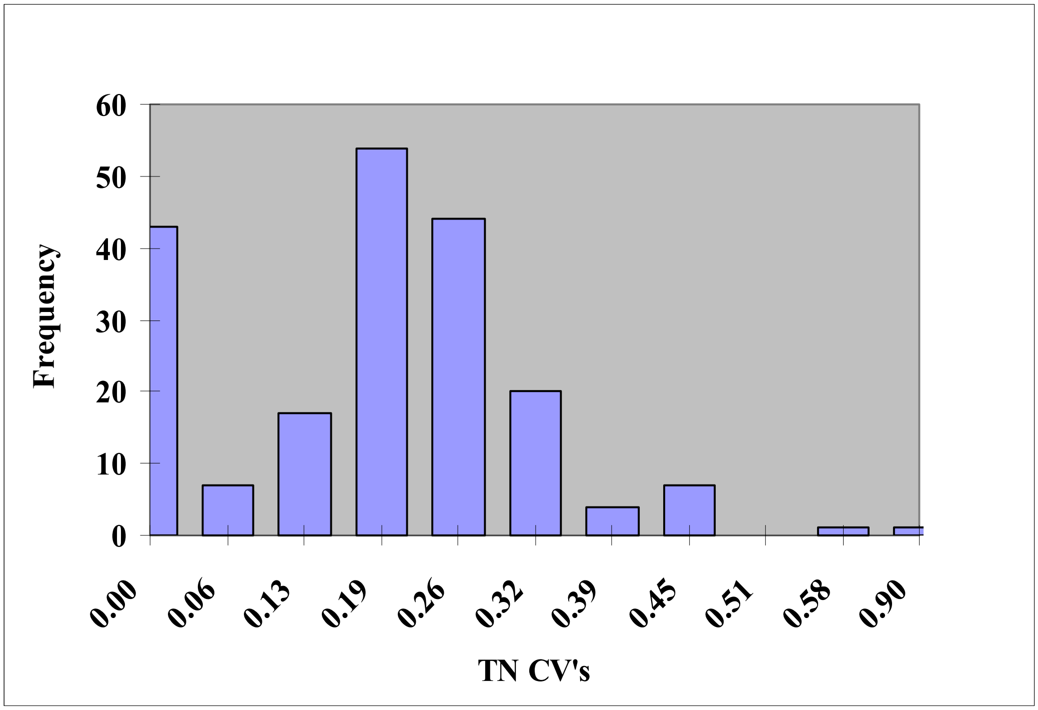


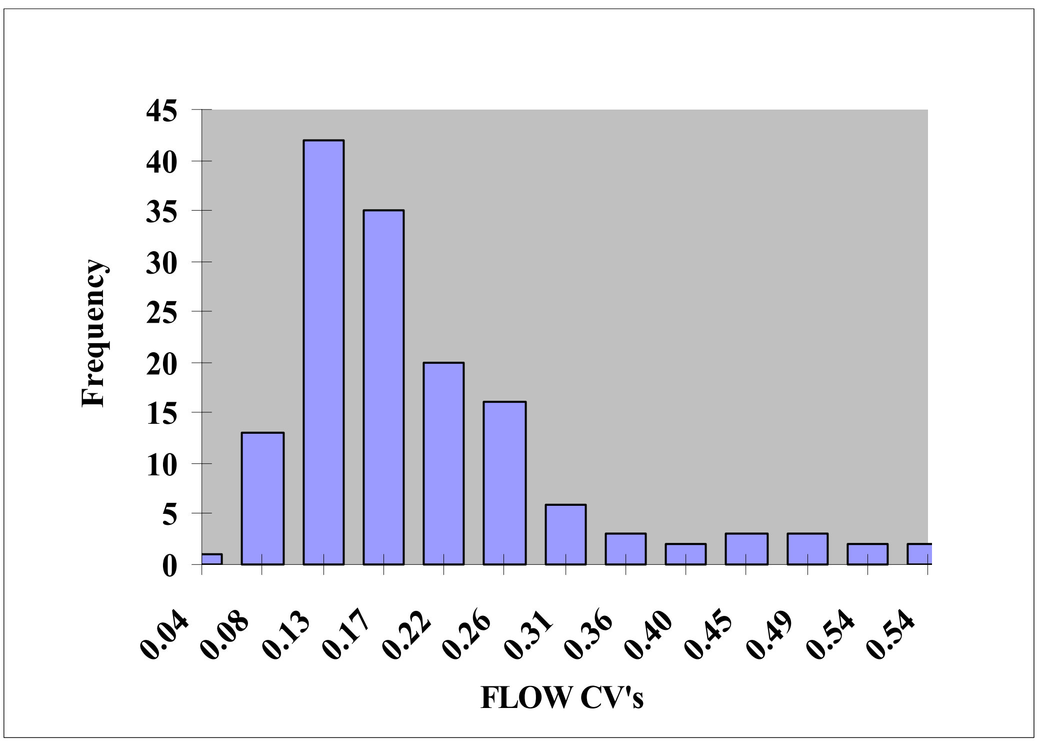
Figures 4.8.1. (upper) and 4.8.2 (lower) The range and frequency of observed coefficients of variation for dissolved oxygen (DOSAT) and pH, respectively.



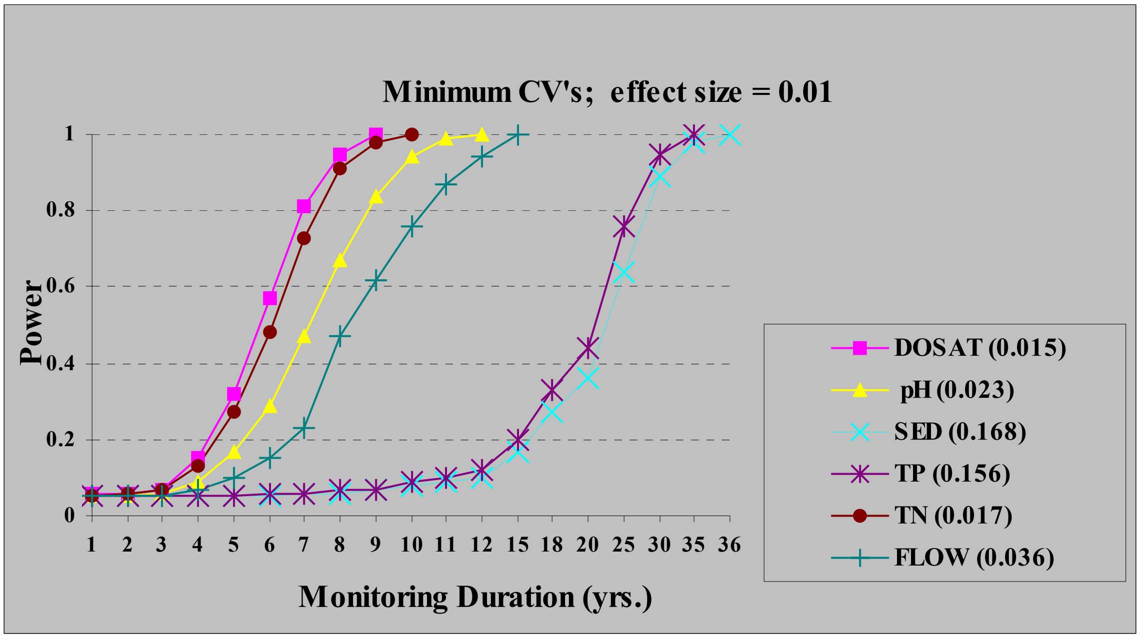


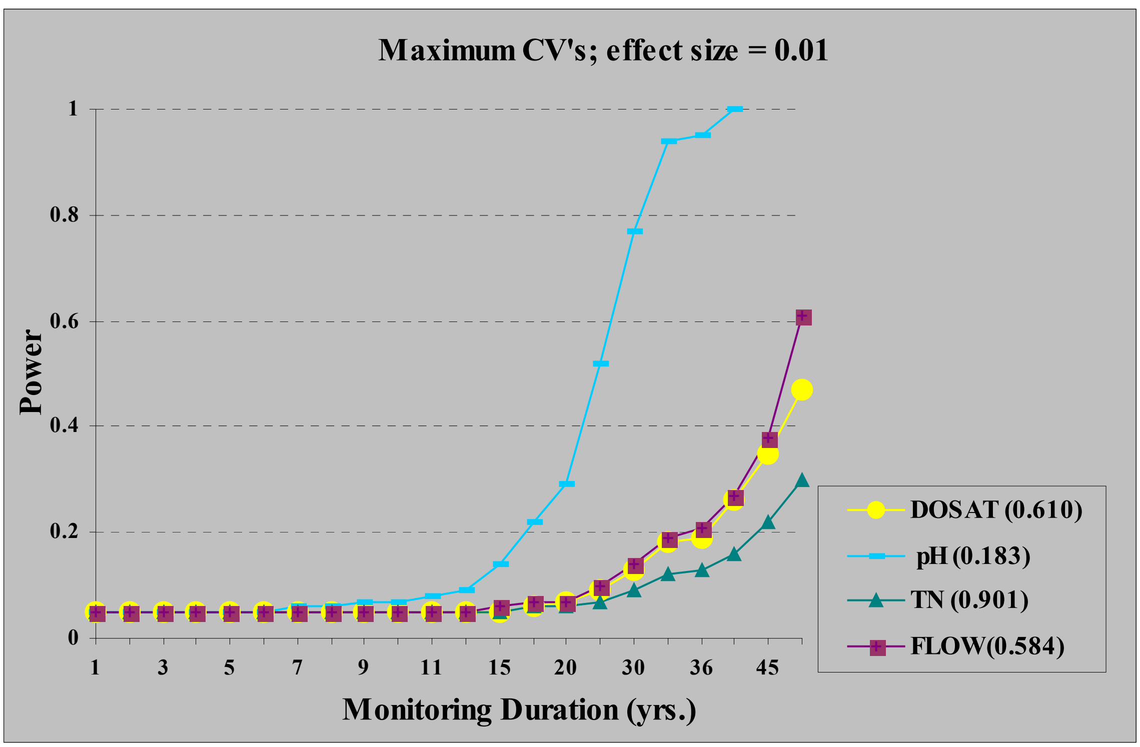
Figures 4.8.3 (upper) and 4.8.4 (lower) The range and frequency of observed coefficients of variation for. phosphorus (TP) and sediment (SED), respectively. Phosphorus and sediment adjusted for flow rates.



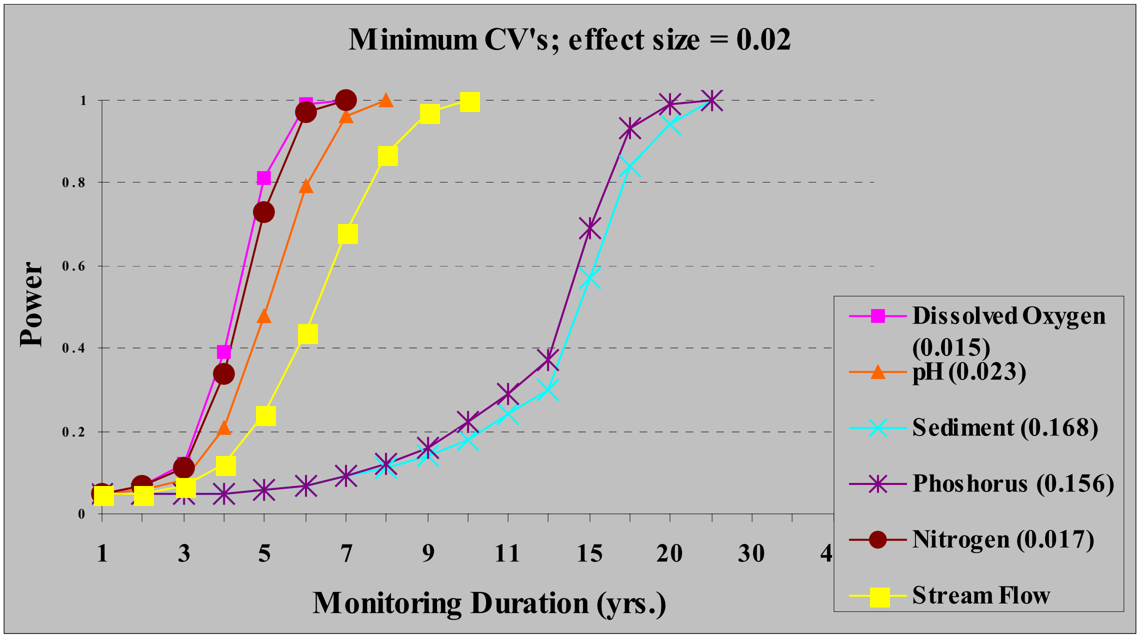


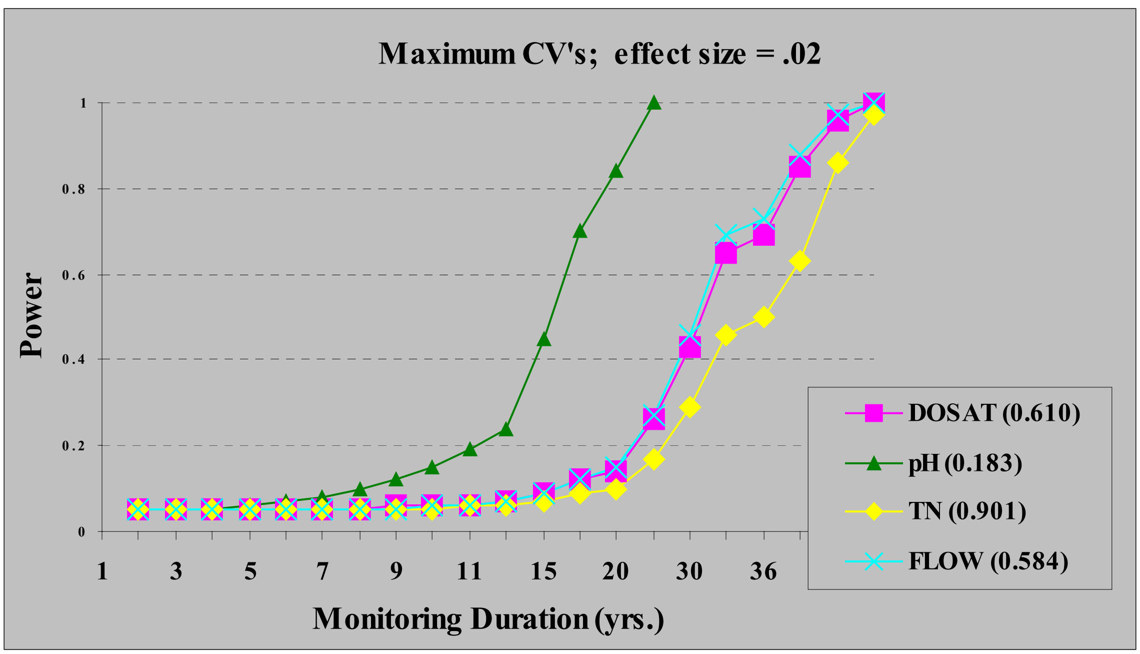
Figures 4.8.5 (upper) and 4.8.6 (lower) range and frequency. The observed coefficients of variation for. nitrogen (TN) and instantaneous streamflow (FLOW), respectively.



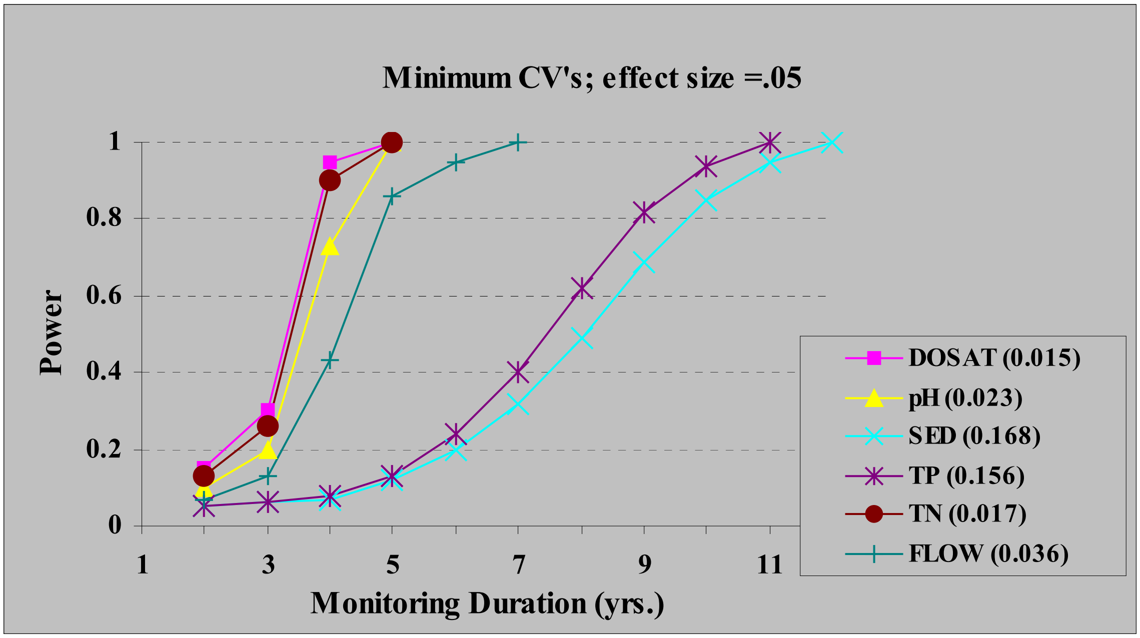


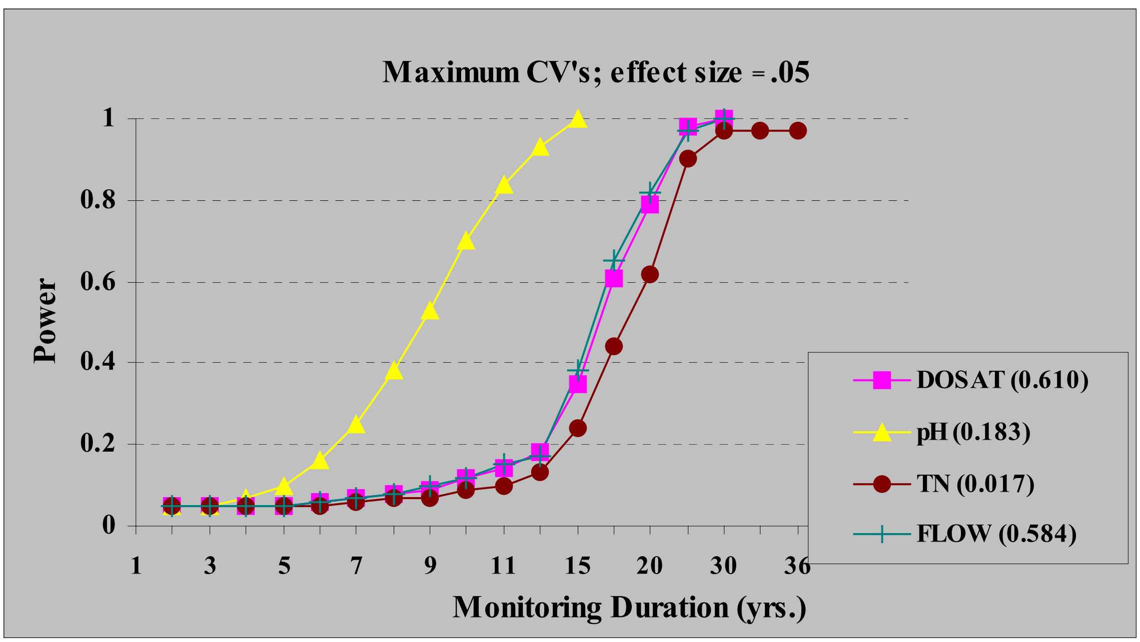
Figures 4.9.1. (upper) and 4.9.2. (lower). The relationship between power estimates and monitoring duration for effect size = 0.01 using observed minimum and maximum CV's (in parentheses).



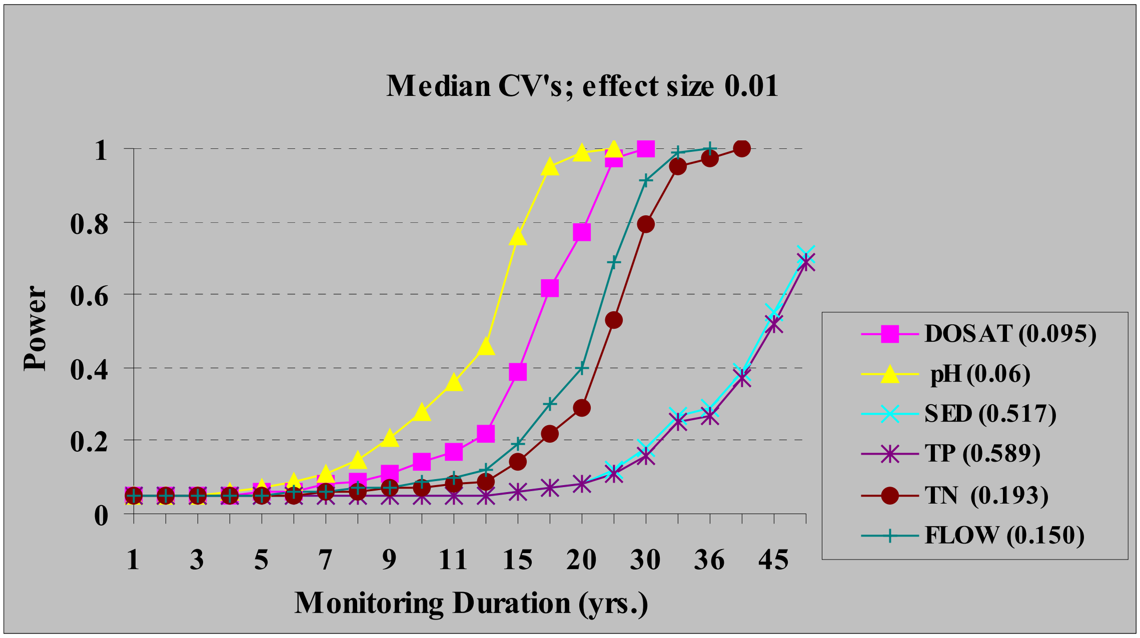


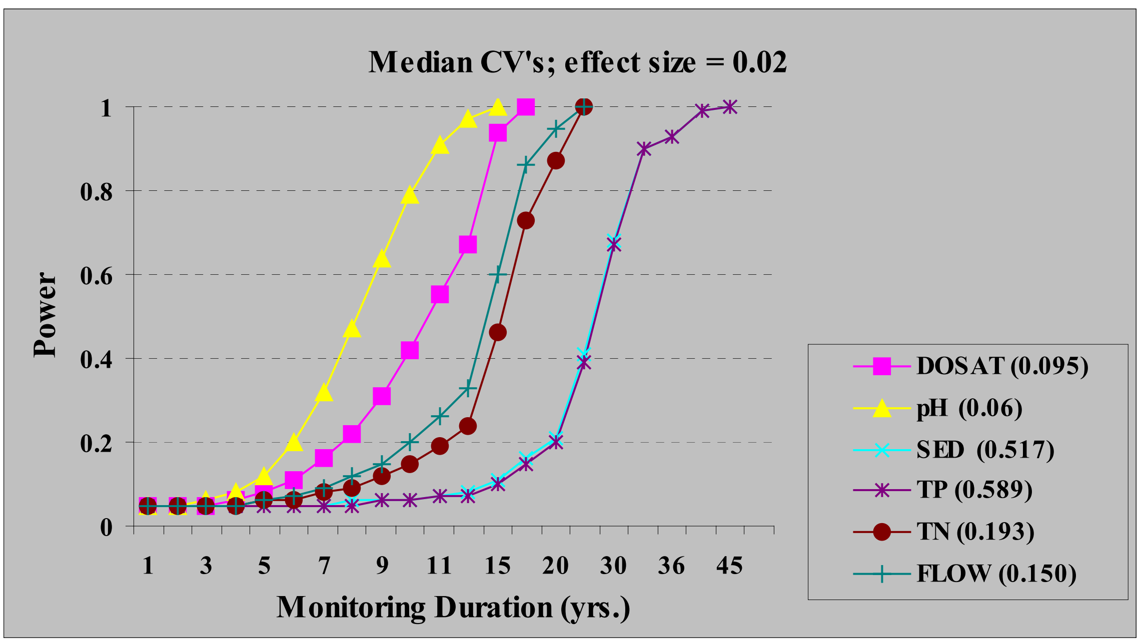
Figures 4.9.3. and 4.9.4. The relationship between power estimates and monitoring duration for effect size = 0.02 using observed minimum and maximum CVs (in parentheses).





Figures 4.9.5. (upper) and 4.9.6. (lower). The relationship between power estimates and monitoring duration for effect size = 0.05 using observed minimum and maximum CV's (in parentheses).





Figures 4.9.7. (upper) and 4.9.8. (lower). The relationship between power estimates and monitoring duration for effect sizes = 0.01 and 0.02 using observed median CV's (in parentheses).

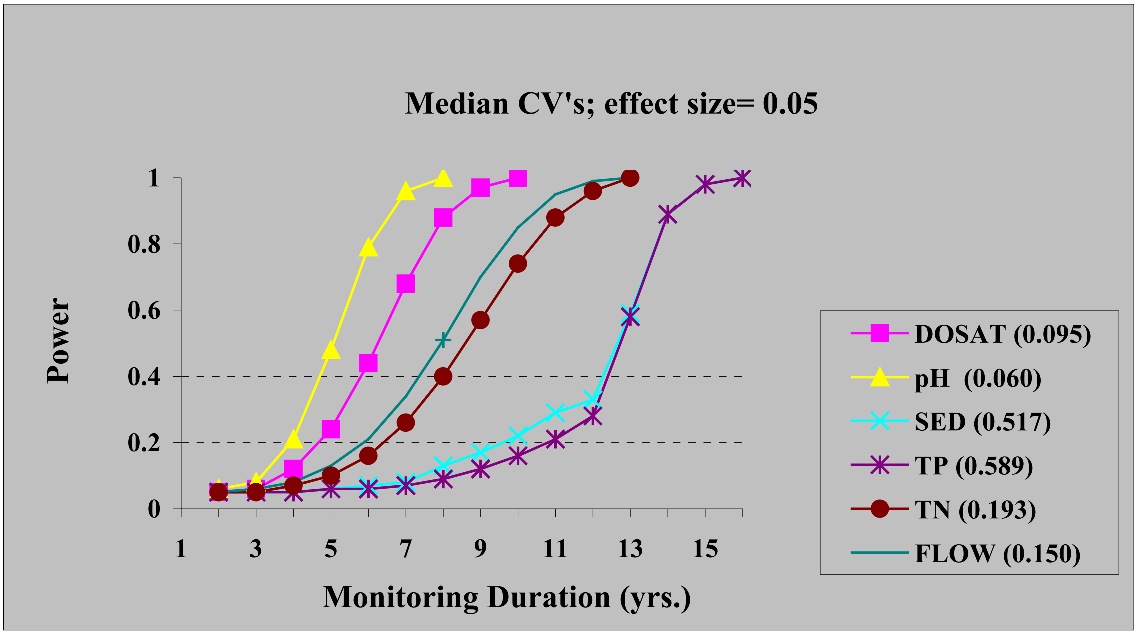


Figure 4.9.9. The relationship between power estimates and monitoring duration for effect size = 0.05 using observed median observed CV's (in parentheses).

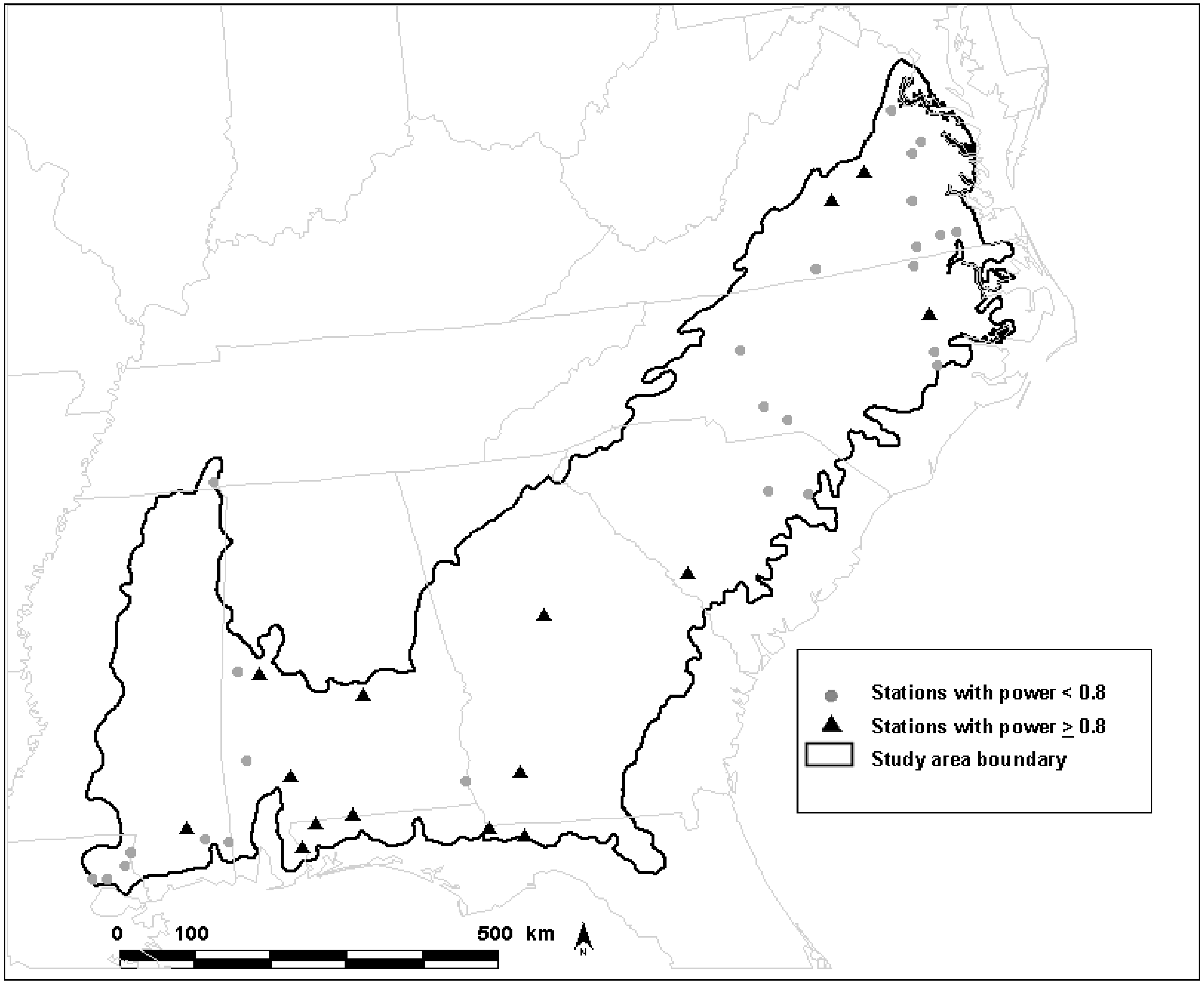


Figure 4.10. Power analysis results by station for dissolved oxygen (DOSAT) for entire monitoring period only for effect size of 0.02.

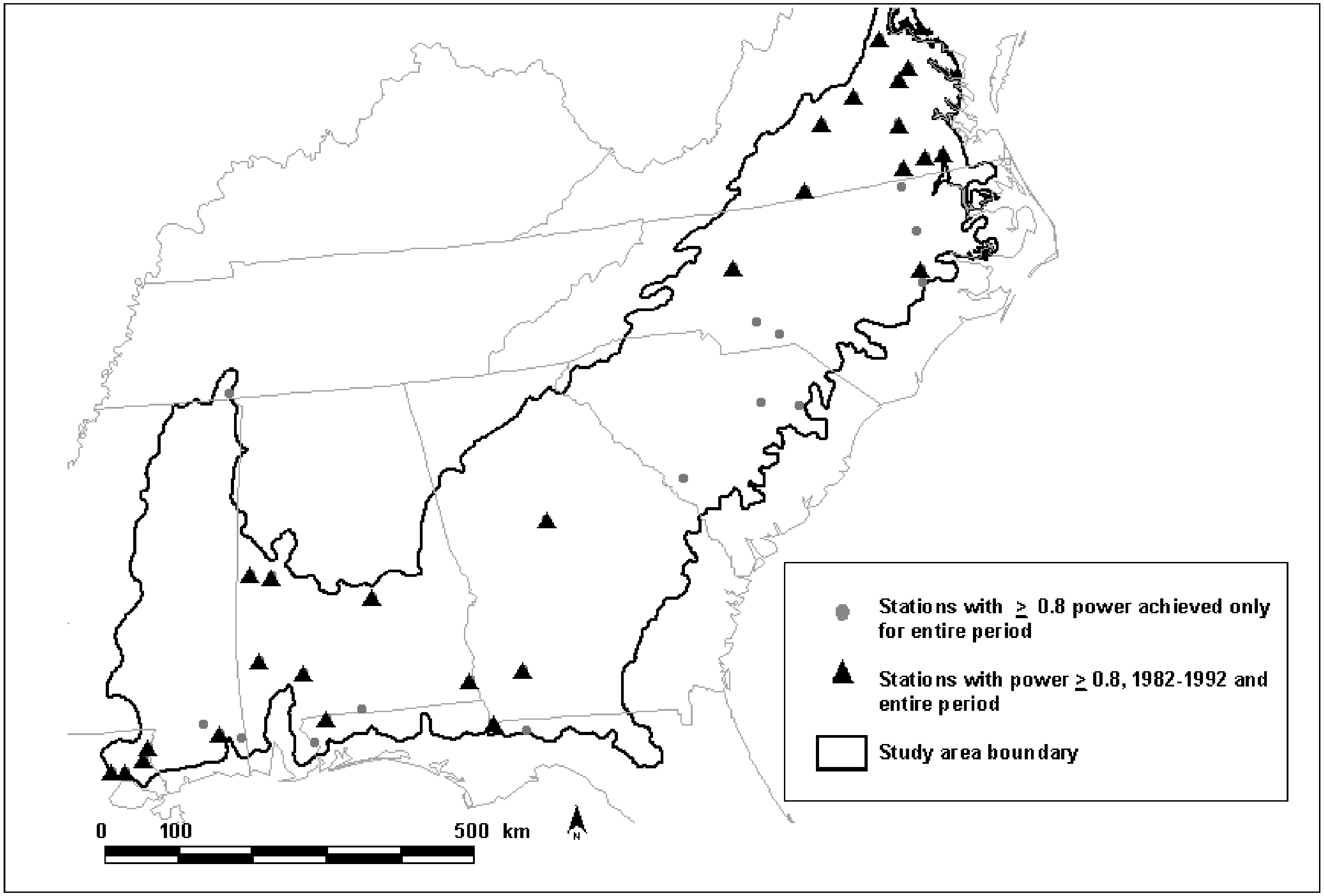


Figure 4.11. Power analysis results by station for pH for1982-1992 and entire monitoring period for effect size of 0.02.

Figure 4.12. Power analysis results by station for nitrogen (TN) for entire monitoring period only for effect size of 0.02.

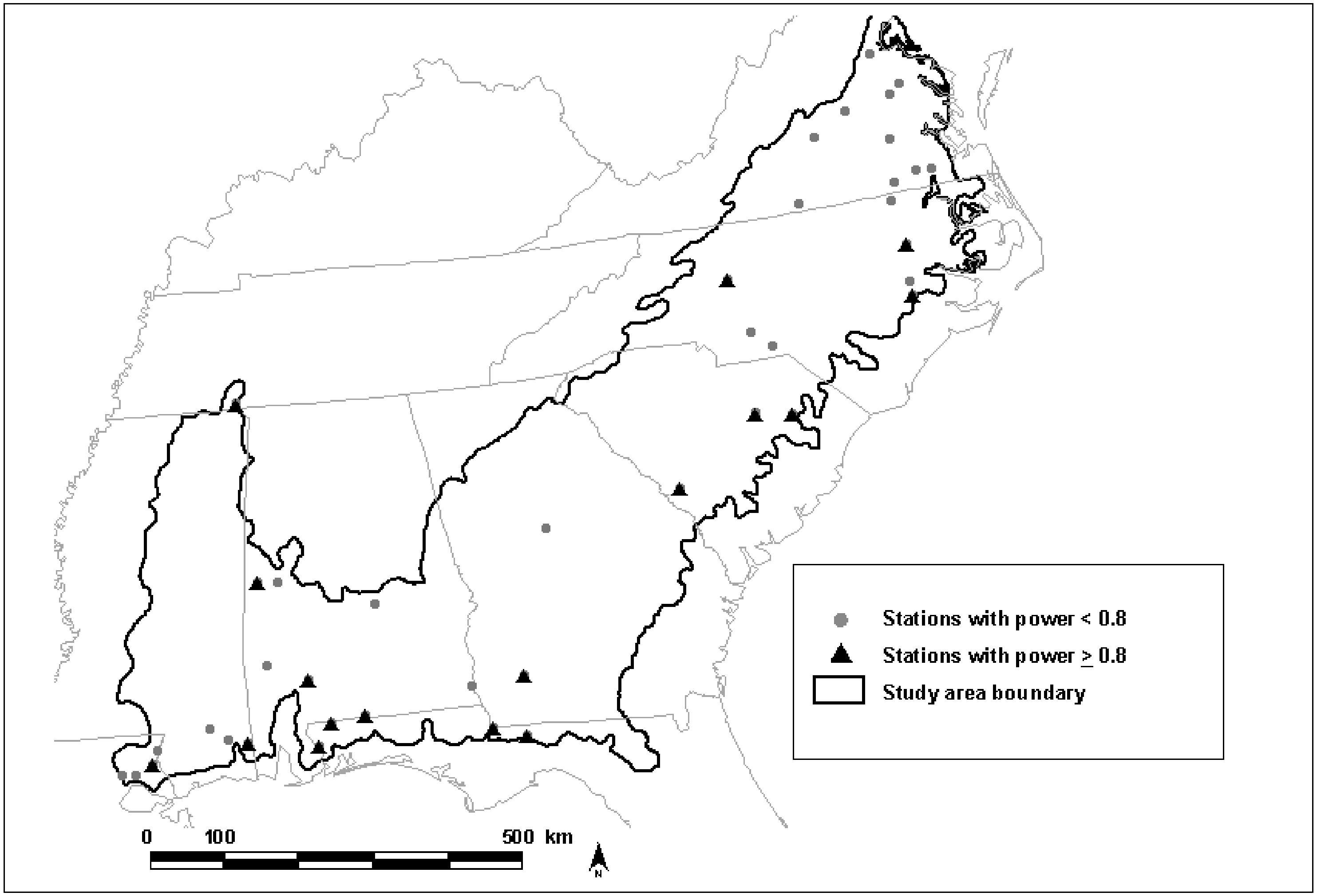


Figure 4.13. Power analysis results by station for instantaneous streamflow (FLOW) for entire monitoring period only for effect size = 0.02..

CHAPTER 5

CONCLUSIONS

There has been some improvement in water quality in the United States over the past thirty years. However, this success is not uniform across the nation and all metrics of water quality constituents. Overall, water quality in the United States still fails to meet goals set by the CWA of 1972; approximately 40 percent of the Nation's assessed waters still do not meet criteria and about half of the Nation's 2000 major watersheds have water quality problems. Studies reveal that significant threats to water quality still remain, including nutrient imbalances, polluted runoff, excessive wet weather flows, habitat degradation, and temperature and flow alterations. These threats put human health at risk and compromise ecosystem services. These facts demonstrate a critical need for improved assessment instruments, water quality standards and a set of tools to implement them (USEPA 1998).

The growing scientific and public interest in water quality has fostered an interest in more efficient monitoring practices to understand and manage for changes in water quality through space and time (Harmancioglu and Alpaslan 1994). Driven in part by recognition of the impacts of human activities (both direct and indirect) on water quality and the effects of water quality on human health, this interest has resulted in the gradual accumulation of long-term water quality data records (Hirsch et al. 1991).

Water quality monitoring is invaluable to ensure compliance with regulations, detect trends or patterns, and advance ecological understanding (Soballe 1998). Statistical analysis of the data is now accepted as a routine part of water quality management programs in both government and industry and it serves as a basis for laws and regulations and as criteria in the determination of remedial action or evaluation of management schemes (Loftis et al 1991).

Statistical detection of time series trends of water quality metrics now receives more attention because of its importance for policy and management. Trend detection depends upon adequate and consistent data observations provided by monitoring programs. "However, monitoring typically measures only a few characteristics in a small fraction of a large and complex system, and thus the information contained in monitoring data depends upon which features of the ecosystem are actually captured by the measurements. Difficulties arise when these data contain something other than intend, but this can be minimized if the purpose of sampling is clear, and the sampling design, measurements, and data interpretations are all compatible with this purpose. The monitoring program and data interpretation must also be properly matched to the structure and functioning of the system. Obtaining this match is sometimes an iterative process that demands a close link between research and monitoring" (Soballe 1998, p. 10). Additionally, two objectives of monitoring programs, estimation of average conditions and evaluation of trends, are scale dependent. An explicit consideration of scale in monitoring programs is important for producing meaningful information since spatial and temporal scales of interest define the best statistical model and affects both sampling design and data analysis (Loftis et al. 1991).

This research examines changes in water quality indices over a large contiguous area in southeastern United States and identifies some of the obstacles inherent in retrospective analysis. The six constituents examined are commonly cited indices, reflecting nutrient loadings (TN and TP), biological activity (dissolved oxygen), and watershed characteristics and land use activities (pH, sediment and streamflow). Numerous research efforts have shown that natural and human-caused events affect water quality by altering chemical composition, physical properties and biological communities of the streams and rivers (for example, Weinberg et al. 2002, Stow et al. 2001, Jones et al. 2001, Basnyat et al. 2000a,b). These events may occur instream or in the watershed. Carpenter et al. (1998) identified agriculture and urban activities responsible for increased loadings of nitrogen and phosphorous nationwide, and Uri (1991) noted that each 1% increase in amount of area of corn and soybeans planted results in about a 0.42 % increase in sediment loading in the Iowa River. Nonurban residential development contributed to increasing total nitrogen concentrations in streams of the Upper Tennessee River Basin (Hampson et al. 2000), and urbanization and nonurban residential development were cited as major factors contributing to general deterioration of multiple water quality indices (for example Santee River Basins [Hughes et al. 2000], Albemarle-Pamlico Drainage Basin [Spruill 1998], and small watersheds in Ontario, Canada [Sliva and Williams 2001]). Petersen (1992, 1990) cited population growth, changes in wastewater treatment methods, agricultural impacts, and other factors responsible for the trends in numerous water quality indices (e.g., total phosphorus, orthophosphates, sulfates, dissolved chloride, total ammonia, biochemical oxygen demand, fecal coliform bacteria and dissolved oxygen) in Arkansas rivers. Franklin (1992) observed that large scale alteration of the vegetative cover (e.g., clear cut logging) in the Pacific Northwest resulted in 90% increase in watershed runoff while Smith (1992) demonstrated that afforestation of pastureland produced the opposite effect, reducing overland water delivery to streams by more than 50%. Detection of trends, however, is highly dependent upon constituent measured, location, sampling and analysis protocol and experimental design. Cary (1989) noted that trends in some constituents (e.g., specific conductance, sodium, chloride) and no trends in other constituents (e.g., hardness, alkalinity, dissolved solids, sulfates) after 9 years monitoring in the Powder River (MT and WY) may be affected by sample size, number of monitoring seasons, heterogeneity, significance level, and autocorrelation.

The USGS and state agencies now regularly provide summary reports on water quality. This information is useful for planning and management objectives, yet results may be difficult or confusing to interpret. The observed complexity of trends and status conditions in many of these reports reflect the difficult challenges inherent in monitoring and assessing water quality data. These challenges are a result of differences in sampling design, monitoring goals and procedures, and variability among water quality metrics. Some of the reports base their conclusions on relative small sample size and/or short monitoring times of less than 16 years (see, for example Hampson 2000; Peterson 1992; Cary 1989). While trends for some constituents may appear within this timeframe, observations of “no” trend may be as much a factor of the monitoring design and subsequent analysis as the absence of any real trend.

Reporting on trends in water quality metrics is usually biased in favor of not committing a Type I error (error of commission), and power analysis tests can provide statistical evidence for accepting or rejecting the null hypothesis where no trend was detected. Despite the recognition of its usefulness in prospective and retrospective analysis and ready availability in numerous computer software packages, it is rarely used on water research. This study demonstrated not only its utility in rejecting the null hypothesis but it calculated minimum length of monitoring time and “preferred” level of error measurement to achieve a range of acceptable powers with significance levels and rates of change established *a priori.*

The Seasonal Kendall estimates of trends over time for constituents at all stations identified differences in trends among stations and regional areas, and across constituents, monitoring length and level of confidence. The study suggests that viewed as a single entity, the Southeastern Plains and Piedmont ecoregions has not shown significant deterioration or improvement in water quality over the entire length of monitoring period (1973 to 1995) with the exception of pH. The ability to detect trends is dependent upon quality of the data and how the data “behave”. For example, two constituents, dissolved oxygen and pH, have low CV values, reflecting low variability in the data. No trend in dissolved oxygen was detected at most stations; however, length of monitoring period at most sites was less than the minimum of 15 years required to achieve a power of 0.80 to support the null hypothesis. In contrast the majority of stations showed an increase in pH when the entire monitoring period was used; mean monitoring period (almost 21 years) exceeded the minimum required to achieve a power of 0.90. Failure to demonstrate trends in other constituents and/or at more stations may reflect insufficient monitoring time and high degree of variability of the data.

There is increasing interest and level of effort to monitor and assess the health of water resources at regional and national scales (Jones et al. 2001; Harmancioglu and Fistikoglu 1998). Large scale analysis provides the spatial framework to address and evaluate performance toward achieving environmental objectives such as protection and improvement of water quality. Aggregated data from multiple stations representing 4 and 6-digit HUC's showed that there was a close approximation of direction and magnitude when compared to individual station data for a given region. This approximation suggests that large scale data represents "average" conditions for all the streams of interest and may be useful when applied to land use/land cover information.

Analysis of land use/land cover data for the entire study area showed that almost 8% of the total land area underwent some change between the 1982-1992 NRI censuses. Largest percentage change occurred in the expansion of urban/built-up areas while cropland experienced the largest loss. A statistical test suggested that the nearly spatially-uniform increase in pH levels was linked to changes in type of agricultural practices. Failure to find other explanatory models was more of a reflection of inadequate and insufficient data than absence of linkage between water quality changes and land use activities.

Retrospective analysis of data is useful for explanatory and predictive models of changes in water quality. There have been numerous efforts to identify and quantify the impact of human activity on water quality specifically and stream health in general. These models provide useful information and guidance for broad-based policy initiatives and action including compliance monitoring and enforcement, remedial activities, and improvements in land management (land treatment).

Development of effective management strategy requires that two goals must guide design and operation of monitoring networks and data analysis: 1) characterization of status and detection of significant (and long-lasting) trends in water quality, and 2) linking trends with changes in land use activities. Short term monitoring may not provide reliable or adequate data because the hydrologic variability and latency in response to some stressors may mask or confound changes in water quality. Required monitoring period varies according to locale, constituent of interest and characteristics of the landscape. For example, smaller watersheds unaffected by large pollutant sources may require shorter monitoring periods. Longer monitoring periods are needed when changes are gradual and response is muted or delayed by the buffering capacity of larger systems or long hydrologic resident times of contaminants (Spooner and Line 1993). Quality assessment should also be coupled to a power analysis that can provide a level of confidence when the null hypothesis is indicated.

Monitoring the condition and "health" of water is a critical to safeguarding and improving this essential resource. There are numerous examples of the strong linkages between quality of life and water resources. In order to ensure that monitoring provides the data necessary to address the multiple demands on water management, future design protocols should more fully encompass research needs.

Future areas of research

This research effort focuses on a few interlinked topics on monitoring. Its development engendered related questions not addressed here but which offer potential for additional work. These include:

1) Comparison of water quality trends of other ecoregions in the United States representative of distinct climatic regions and land use/land cover characteristics;

2) Analysis of trends using a broader range of variables including biological metrics (e.g., fecal coliform bacteria);

3) Predictive and explanatory model building incorporating site-specific land impact variables, such fertilizer application rates and crop yields;

4) Computation of power analysis at multiple sites simultaneously (e.g., program MONITOR);

5) Sensitivity testing of Seasonal Kendall test;

6) Comparison of results of trend analysis using the ecoregion model with other spatial frameworks (e.g., watersheds); and

7) Spatial pattern of excursions (deviations) from compliance levels and relation to monitoring efforts and rates of land use/land cover change.

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