SPATIAL AND TEMPORAL DYNAMICS OF CHLOROPHYLL AND NUTRIENTS IN THE SUWANNEE RIVER AND ESTUARY, FLORIDA, USA

By

ANDREA MARIE KRZYSTAN

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To my family

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LIST OF ABBREVIATIONS

AIC	Akaike Information Criterion
°C	degrees Celsius
CUSUM	Cumulative sum
CWA	Clean Water Act
h	Hours
km	Kilometers
L	Liters
m	Meters
m^3	Cubic meters
mg	Milligrams
µg	Micrograms
NCDC	National Climatic Data Center
OLS	Ordinary least squares
PCU	Platinum-cobalt units
ppt	Parts per thousand
RMSE	Root mean squared error
s	Seconds
SED	Squared Euclidean distance
SRWMD	Suwannee River Water Management District
TN	Total nitrogen
TMDL	Total Maximum Daily Load
UFA	Upper Floridan aquifer
USGS	United States Geological Survey
y	Years

Abstract of Thesis Presented to the Graduate School of the University of Florida in Partial Fulfillment of the Requirements for the Degree of Master of Science

SPATIAL AND TEMPORAL DYNAMICS OF CHLOROPHYLL AND NUTRIENTS IN THE SUWANNEE RIVER AND ESTUARY, FLORIDA, USA

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Andrea Marie Krzystan

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The ability to differentiate between human induced changes and natural variations in water quality is a key element of effective management. An important step in distinguishing these sources of variation involves characterizing the temporal dynamics of specific water quality parameters, which can be accomplished through analysis of long-term datasets. Two long-term datasets were used in this study to examine water quality dynamics within a system of concern: the Suwannee River and its estuary.

The first dataset was used to assess the presence of increasing trends in total nitrogen concentrations along the length of the Suwannee River between 1989 and 2010 using cumulative sum (CUSUM) analyses. CUSUM analyses were applied to data from different sections of the river delineated based on a moving split-window boundary detection analysis. Results demonstrated increasing total nitrogen concentrations in each of the delineated sections of the river across the study period. However, fine-scale differences in CUSUM statistics highlighted spatial differences in the influence of hydrogeology, land use, groundwater quality, and climatic events on nitrogen dynamics, which suggested the need for an adaptive management approach.

A second long-term dataset was used to develop time series and regression models that described relationships between chlorophyll-a concentrations and salinity, temperature, light availability, and concentrations of total nitrogen and total phosphorus within the lower Suwannee River and its estuary from 1998 to 2010. To increase the power to detect relationships, stations were grouped according to results of multivariate ordination, and models were developed using pooled data from 1999-2008. The predictive power of each model was evaluated using data from 1998, 2009, and 2010. As expected, final model parameters varied among groups of stations distributed in the river, oyster reef, and nearshore areas of the system; however, color and total phosphorus concentrations always explained significant amounts of variation in chlorophyll-a values. Chlorophyll-a concentrations exhibited different relationships to model covariates at the station level, although observations at the group level ordinated together. Models for groups of stations fit overall trends at all sites with adjusted R-squared values ranging from 0.34 to 0.72, but the accuracy of predictions differed among sites within each group. Differences in spatial relationships were attributed to variation in the river plume, which can influence top-down grazing pressure or bottom-up influences, such as reduced light availability, due to color.

Results of both analyses highlight the overall complexity of the Suwannee system and emphasize the need to consider multiple sources and scales of variation when attempting to distinguish natural variation from human activities leading to impairment or improvement.

CHAPTER 1
INTRODUCTION

The ecological health and integrity and human use of aquatic ecosystems depends on water quality. This importance is highlighted by examples of negative impacts from degraded water quality including drinking water contamination (Alam et al. 2007), loss of revenue from fisheries closures (Mallin et al. 2002; Macfarlane 2006) and altered ecosystem structure and function (Duarte 1995; Cloern 2001; Wazniak et al. 2007). To address such impacts, managers in the United States initially have focused on controlling threats to surface water quality at their source. At a national level, the Clean Water Act (CWA) was implemented in the 1970s to regulate the point source discharge of industrial pollutants (e.g., outfall pipelines). The scope of the act was broadened in subsequent years to consider potential physical and biological impacts, as well as threats from non-point sources, e.g., urban and agricultural runoff, which are not confined to a single point of introduction (Norgart 2004).

In addition to outlining federal management guidelines, the CWA also provides minimum requirements for water quality management by state governments. More specifically, each state must define designated uses for all water bodies; identify waters not being adequately protected; and apply Total Maximum Daily Loads (TMDLs) to reverse impairment. A water body that cannot serve its designated use because of poor water quality (e.g., criteria for a given parameter are not satisfied) is listed as impaired. Once impairment is verified through sampling, TMDLs are developed to specify the maximum level of a pollutant that can be assimilated without causing harm, and policies are implemented to reduce loads to this level, with the aim of restoring the ability of the impaired waterway to meet its designated use (Norgart 2004; USEPA 2006).

An example of a system currently being managed according to CWA guidelines for threats to water quality is the Suwannee River in Florida. Several segments of the river and its associated estuary have been declared as “impaired” because of a failure to meet state water quality standards for nutrients or other parameters (FDEP 2002). Nutrient impairment is of particular concern because of its potential to lead to eutrophication. Eutrophication, an increase in the supply of organic matter to a system, can result in ecosystem-wide impacts on biogeochemical and trophic processes by causing habitat loss, decreasing biodiversity, and degrading water quality, especially through formation of anoxic or hypoxic zones (Rosenberg 1985; Nixon 1995; Paerl 1999; Cloern 2001).

Thus far, managers have addressed impairment in the Suwannee system with TMDLs and non-regulatory measures, including Best Management Practices for agriculture (Norgart 2004; USEPA 2006). A more recent regulatory initiative is the development of numeric nutrient criteria, which will establish upper limits for nutrient concentrations expected to prevent adverse impacts to water quality and wildlife (FDEP 2010).

To establish effective numeric criteria, managers will need to understand nutrient sources and dynamics within each system, as well as how changing nutrient levels manifest as biological responses. More specifically, managers will need to be able to distinguish natural variation in nutrient levels from variation due to anthropogenic inputs in order to designate attainable magnitude, duration, and frequency components for criteria. Although distinguishing these sources of variation is complex, a first step involves characterizing trends in nutrients and biological responses from long-term water quality datasets.

CHAPTER 2
NITROGEN TRENDS IN THE SUWANNEE RIVER, FLORIDA

Rationale

Nitrogen is a nutrient vital to all life, and its biological availability is determined by the nitrogen cycle. The nitrogen cycle has been altered by human activities, such as fertilizer production, fuel combustion, and crop and animal agriculture (Galloway et al. 1995; Vitousek et al. 1997; Carmargo and Alonso 2006). These activities have accelerated since the industrial revolution, resulting in increased levels of nitrogen in many terrestrial and aquatic ecosystems across the globe (Howarth 2008).

Increasing nitrogen loads to aquatic systems have the potential to negatively impact human health and ecosystems. One health concern is methemoglobinemia, a condition induced by high nitrate levels in drinking water that affects the ability of blood cells to transport oxygen (Carmargo and Alonso 2006). A primary concern for ecosystems is eutrophication, an increase in the supply of organic matter to a system. Eutrophication can have ecosystem-wide impacts and lead to decreased biodiversity by altering biogeochemical cycles, creating hypoxia or anoxia, and decreasing light penetration, which leads to loss of vegetated habitats (Rosenberg 1985; Nixon 1995; Paerl 1999; Cloern 2001).

Although eutrophication can occur naturally, many incidences have been linked to human activities, including detrimental eutrophication in the Wadden Sea, the Black Sea, and the Gulf of Mexico (De Jonge et al. 1996; Humborg et al. 1997; Turner et al. 2007). These systems share a common characteristic, the influence of large rivers, which is not unexpected because large rivers deliver substantial nutrient loads to many coastal ecosystems (Billen and Garnier 1997). For example, the northern Gulf of Mexico received 91% of its nitrogen load and 88% of its phosphorus load from the Mississippi and Atchafalaya Rivers during 1972-1993 (Turner et al. 2007). Nutrients transported to the coast originate from all parts of a river's watershed. Therefore, efforts to manage nutrients in coastal systems with strong riverine influences need to account for sources and dynamics of nutrients not only at the coast, but also along the river.

An example of a system currently being managed as a watershed is the Suwannee River in Florida. Several segments of the Suwannee River have been declared as “impaired” because they did not meet the state's water quality standards for nutrients (FDEP 2002). To address this impairment, managers have implemented a multi-faceted approach involving regulatory and non-regulatory measures. Regulatory measures include Total Maximum Daily Loads, which estimate the maximum level of a pollutant that can be assimilated by the water body without causing harm, and non-regulatory measures include Best Management Practices for agriculture (Norgart 2004; USEPA 2006). A recent regulatory initiative involves the development of numeric nutrient criteria, which will provide upper limits to nutrient levels within a system to prevent adverse impacts to water quality and wildlife (FDEP 2010).

To establish effective numeric criteria, managers will need to understand nutrient sources and dynamics within each system. More specifically, managers need to be able to distinguish natural variation in nutrient levels from variation due to anthropogenic inputs in order to designate attainable magnitude, duration, and frequency components for each criterion.

With this need in mind, the present study characterizes spatial and temporal variation in total nitrogen concentrations in the Suwannee River using a combination of trend and spatial analysis techniques. Cumulative sum (CUSUM) analyses will indicate if and when total nitrogen concentrations at eleven stations have increased from 1989 to 2010. Trend analyses will focus on different sections of the river as delineated with a moving split-window boundary detection technique. These results will provide information about differences in trends along the length of the river that will help managers identify areas in need of increased attention or customized management approaches.

Methods

Study Area

The focus area for this study was the main reach of the Suwannee River within Florida. This portion of the river is 394 km long, has an average annual discharge of 300 m^3 s^{-1}, and drains an 11,000 km^2 watershed (Katz et al. 1997; Wolfe and Wolfe 1985). The watershed as a whole is relatively pristine with numerous forested, wetland, and protected areas; however, anthropogenic land-uses such as crop and animal agriculture and phosphate mining are present (Katz et al. 1997; Bledsoe and Phlips 2000).

As the river meanders towards the Gulf of Mexico, variations in geology, topography, and hydrology result in changing water chemistry and ecology (Wolfe and Wolfe 1985). Managers use these differences to divide the river into three reaches: the Upper, Middle, and Lower Suwannee. The Upper Suwannee provides important spawning habitat for the Gulf Sturgeon and has some karst features like sinkholes, limestone outcrops, and springs, such as White Springs, Suwannee Springs, and Ellaville Springs. Water in the Upper Suwannee River is highly colored, acidic, and typically has low nutrient concentrations. As the Upper Suwannee transitions into the Middle Suwannee near Ellaville at Highway 90, increased groundwater input from the

Upper Floridan Aquifer via 62 mapped springs results in clearer, more alkaline water with increased nutrient concentrations. The Middle Suwannee channel is 79-150 m wide, bordered by swamp and bottomland hardwood plant communities, and inhabited by West Indian manatees during winters. The river widens to 240-300 m in its Lower reach, which runs from Fanning Springs to the Gulf of Mexico, where it splits into two passes. Relative to the Middle reach, there are fewer springs in the Lower Suwannee but major springs like Manatee and Fanning Springs do contribute to high nutrient levels. The Lower reach of the river is tidally influenced up to the Gopher River confluence, and its bottom is characterized by exposed limestone, coarse sand, and sandy mud.

Data Description

Raw data consisted of monthly concentrations (mg L^{-1}) of total Kjeldahl nitrogen, ammonia, and nitrate-nitrite for 1989-2010. Data were acquired from the State of Florida STORET database (http://www.dep.state.fl.us/water/storet) for 11 stations (Table 2-1; Figure 2-1).

For each station and sampling event, total nitrogen (TN) concentrations were calculated as the sum of concentrations of total Kjeldahl nitrogen, ammonia, and nitrate-nitrite. Monthly TN concentrations were aggregated into annual arithmetic means to reduce serial autocorrelation and to minimize the influence of short-term variability on results of tests for trends. Annual means were calculated using all values for a given year, although some years had missing values or multiple sampling events within a given month.

Trend Analyses

Cumulative sum (CUSUM) control charts (Page 1954) and a local boundary detection technique were used to identify spatial and temporal patterns in annual mean TN concentrations in the Suwannee River from 1989 to 2010.

Spatial differences in temporal trends were evaluated by delineating sections of the river using a moving split-window method and then comparing CUSUM results among sections. Sections were delineated based on dissimilarities in annual mean TN concentrations at adjacent sites. Due to the relatively small number of sites, differences were determined using a moving window of 2 sites, the smallest size possible, and the differences were quantified using squared Euclidean distances (SEDs). Calculations of SEDs using Equation 2-1 were based on mean TN concentrations, x_1 and x_2, for each year, t, at adjacent sample sites, S_1 and S_2, across T = 22 y (Fortin and Dale 2005).

$$\mathrm{SED}\ (S_1, S_2) = \sum_{i=1}^{T} (\mathrm{x}_{1\mathrm{t}} - \mathrm{x}_{2\mathrm{t}})^2 \qquad \text{2-1}$$

Ten SED measures were calculated and plotted against pairs of adjacent sites to determine where boundaries occurred along the river. Boundaries were visualized as peaks relative to other SED values. High, narrow peaks were interpreted as sharp boundaries, whereas low, wide peaks were interpreted as gradual boundaries (Fortin and Dale 2005). Stations located between boundaries of either type were grouped into the same section for CUSUM analyses.

CUSUM analyses were used to determine if annual mean TN concentrations at each station in the previously identified sections increased relative to background levels during the 22-y study period. Background levels were defined as the arithmetic mean of annual mean values within a section for 1989-2000. These years were selected because time series plots of annual mean TN concentrations appeared relatively stable during that period.

For each station, annual mean values were converted into standardized deviations by subtracting the appropriate reference mean from the annual mean value and then dividing this quantity by the standard deviation of annual mean values within the section. Standardized values were used to derive CUSUM statistics (CS_t) for each year by taking the maximum of two values: 0 or ($CS_{t\text{-}1} + z_t - k$), where $CS_{t\text{-}1}$ was the CUSUM statistic for the previous time period ($CS_1 = 0$), z_t was the standardized annual mean nutrient concentration for the current year, and k was a term allowing for some variation around the background value, which was equal to one-half the section standard deviation (i.e. equal to 0.5 for the standardized data). To determine if observed changes were significant, a threshold parameter, h, was estimated using Equation 2-2 based on k = 0.5 and the ARL (i.e., the average run length between false alarms) defined as 200 (Rogerson 2006).

$$h \approx \frac{2k^2(ARL+2)}{2k^2(ARL+1)} \times \frac{\ln(1+2k^2 ARL)}{2k} - 1.166 \qquad \text{2-2}$$

The ARL value was chosen to obtain a probability of 0.005 for the occurrence of false alarms (i.e., where the threshold was exceeded due to random chance, not a true change in the process).

Finally, CUSUM statistics were visualized as a function of each year for each station along with the threshold value, h, to determine if and when annual mean TN concentrations had changed relative to background concentrations for each section.

Results

Boundary Detection

The moving split-window boundary detection method indicated three sharp boundaries along the study area based on differences in observed TN concentrations at adjacent sites (Figure 2-2). Based on these boundaries, four sections were delineated for the CUSUM analyses: station 1; stations 2, 3, and 4; stations 5, 6, and 7; and stations 8, 9, 10, and 11.

Trend Analyses

Time series plots of annual mean TN concentrations had generally similar patterns at all stations with maximum values occurring during 2003, 2004, or 2005 (Figures 2-3 to 2-6); however, CUSUM results did exhibit some spatial variability across sections.

For the most upstream section, station 1, the CUSUM statistic was non-zero, but below the threshold during 1994-1999 and 2003, and it exceeded the threshold the first time in 2004 and then steadily increased through 2010 (Figure 2-3).

In the next section (stations 2, 3, and 4), the first non-zero CUSUM statistics at all sites occurred during 1991-1992, and the threshold was first exceeded one year earlier than at station 1, in 2003 (Figures 2-4A to 2-4C). Furthermore, CUSUM statistics for all three stations increased steadily from 2003 to 2010, although the rate of increase slowed between 2005 and 2006 (Figures 2-4A to 2-4C).

Like stations 2, 3, and 4, CUSUM statistics for stations 5 and 7 in the next section were first non-zero during 1991-1992 (Figures 2-5A and 2-5C). However, non-zero CUSUM statistics were not observed at station 6 until 2001-2002 (Figure 2-5B). The first significant increase in annual mean TN concentrations was observed at stations 5, 6, and 7 in 2003, and CUSUM statistics increased steadily through 2010, with only a small change in the rate of increase during 2008 (Figures 2-5A to 2-5C).

Similar to stations 2, 4, 5, and 7, the first non-zero CUSUM statistics were observed in 1991 for stations 8, 10, and 11 in the most downstream section (Figures 2-6A, 2-6C, and 2-6D). However, the first non-zero statistic was not observed until 1999 at station 9 (Figure 2-6C). The threshold was exceeded from 2003 to 2010 for stations 9, 10, and 11 but it was first exceeded in 1999 at station 8, the earliest significant increase at any of the stations. Patterns of significant CUSUM statistics at stations 9 and 11 were similar, with a steady increase through 2007, a slight leveling in 2008, and then an increase through 2009 and 2010 (Figures 2-6B and 2-6D). Significant CUSUM statistics at station 8 exhibited a more exaggerated S-shape due to a relatively slow increase in values from 1999 to 2002 (Figure 2-6A). Finally, significant CUSUM statistics at station 10 increased at a relatively constant rate from 2003 to 2010.

Discussion

Trends in Total Nitrogen

Results of the CUSUM trend analyses showed periods of increased TN concentrations in each section of the Suwannee River by the end of the 22-y study period. These results agree with previously documented trends in nitrogen in the river (e.g., increases of 0.02 μg L^{-1} y^{-1} between 1977 and 1997; Pitman et al. 1997), and they also parallel observed increases in concentrations of nitrate-nitrogen discharged from springs (Katz et al. 1999).

Nitrate-nitrogen concentrations have increased in recent decades from 0.1 μg L^{-1} to 5.0 μg L^{-1} in some springs, with the trend being attributed to human activities, such as agriculture and wastewater discharge (Katz et al. 1999). Nitrogen from springs and other groundwater sources can impact surface water quality because springsheds overlap the river's watershed in many areas and the basin's karst topography leads to high connectivity between surface and ground waters (Katz et al. 1997; Crandall et al. 1999). Other sources of nitrogen loads to the Suwannee River include atmospheric deposition, runoff from swamps or wetlands, and point sources, such as wastewater outfalls (Upchurch et al. 2007; Brown et al. 2008).

The relative impact of different nitrogen sources varies along the length of the river due to changes in hydrology, geology, and land use. The contribution of groundwater to the base flow of the Suwannee increases in the Middle and Lower Suwannee as the number of springs and other karst features increases (Crane 1986; Grubbs 1997). The Middle Suwannee also is dominated by agricultural land use, which contributes to increased nitrogen loading through leaching and runoff of animal waste and fertilizers (Pitman et al. 1997; Cabrera 2004). Because of increasing inputs from springs and increased nitrogen concentrations in groundwater toward the mouth of the Suwannee, higher concentrations of nitrogen and stronger trends in nitrogen were expected to occur at stations in the downstream sections of the river. However, stations 2, 3, and 4 in the second most upstream section of the river had the highest mean TN concentration, 1.33 mg L^{-1}, among all the sections across the whole study period.

Section 2 may not have followed expected spatial TN trends due to the limited influence of groundwater in this region. As part of the Upper Suwannee, this section is coincident with a confined portion of Upper Floridan aquifer (UFA). Where the UFA is confined, interactions between groundwater and surface water are limited and water quality primarily is affected by local factors like land use (Crandall et al. 1999; Upchurch et al. 2007). Furthermore, groundwater along most of section 2 does not flow towards the Suwannee River (Katz et al. 1997). These hydrogeologic characteristics combined with historic land uses (e.g., wastewater discharges, livestock farms, and paper mills) result in relatively high nutrient loads that are less dependent on groundwater than loads in other parts of the system (Hand et al. 1990 in Katz et al. 1997).

Despite these unexpected results in spatial patterns of TN concentrations along the river, trends in annual mean TN concentrations were indeed strongest in the two downstream sections as indicated by the relative magnitude of their CUSUM statistics. CUSUM statistics for section 4 (stations 8, 9, 10, and 11) ranged from 26 to 41 relative to the 1989-2000 reference value and exhibited the largest, positive, cumulative deviations of all sections with non-zero values (Figure 6). The trend was particularly strong at station 8, where a significant increase above the reference value was detected in 1999, the earliest positive deviation for any station (Figure 2-6A). The trend at Station 8 may result from elevated nitrogen concentrations delivered by nearby second-magnitude springs (Poe and Rock Bluff) and the Santa Fe River, which joins the Suwannee River just upstream of this station.

In addition to the spatial variation in the influences of hydrology, geology, and land use, temporal variation in climatic conditions and agricultural practices can influence groundwater contributions and nitrogen loads to surface water bodies. For example, Katz and Bohlke (2000) noted that seasonal trends in groundwater nitrate concentrations in the Middle Suwannee sub-basin followed patterns in fertilizer application. On the other hand, climatic events drive more complex interactions between ground and surface waters and the processes that influence nitrogen dynamics.

During high rainfall events, increased water levels in the Suwannee can cause reverse flows for some springs (Giese and Franklin 1996a). Reduction in spring inputs, along with the potential for dilution and denitrification of nitrate in groundwater, results in less nitrate-nitrogen loading to the Suwannee during periods of high flow (Katz et al. 1997). Where groundwater influences are less significant, increases in TN concentrations are likely because increased rainfall generates runoff containing organic nitrogen from swamps, wetlands areas or agricultural operations (Copeland 2009).

In contrast, periods of drought are characterized by higher relative contributions from springs (Giese and Franklin 1996b). As the proportion of groundwater within the river increases, increased nitrate concentrations are possible depending, in part, on the age of the groundwater (Upchurch et al. 2007; Copeland et al. 2009). Discharged groundwater tends to be older during low flow periods, which can introduce groundwater from deep sources like the Avon Park formation that has high concentrations of organic nitrogen (Copeland et al. 2009). With the potential for both increased groundwater contributions and higher nitrogen concentrations during periods of low rainfall, associated increases in riverine nitrogen levels often are observed (Katz et al. 1997).

Such complex interactions between groundwater, surface water, and climatic conditions make it difficult to interpret patterns in TN concentrations at timescales from hours to months or seasons within years. In fact, Copeland et al. (2009) both documented variable responses for different components of TN during a single time period and observed that variation in flow can mask trends in nitrogen concentrations. With these difficulties in mind, the present study demonstrated the utility of evaluating trends on a longer timescale.

The use of annual mean concentrations highlighted patterns that can be interpreted more easily. For example, annual mean TN concentrations at all stations (except 1 and 8) showed significant cumulative deviations from their respective 1989-2000 means that began in 2003 and continued through 2010. The timing of this increase coincided with above normal rainfall in 2003 (Verdi et al. 2006 *in* Copeland et al. 2009). Above normal rainfall also was experienced during the 1997-98 El Niño event; however, significant increases in annual mean TN concentrations were not apparent in the CUSUM analyses during these years.

The discrepancy between trends following these time periods may be attributable to climatic conditions in the years prior to each period of high rainfall. More specifically, an extended drought occurred from 1999 to 2003 while the 1997-98 El Niño event was preceded by a weak La Nina event in 1994-1995 (characterized by low rainfall) and then above normal rainfall conditions in 1996 (Crandall et al. 1999; Copeland et al. 2009). During the drought from 1999 to 2003, more nitrogen could have been stored within the soils, so when heavy rainfall occurred during 2003, a greater quantity of nitrogen was released causing a larger increase in nitrogen throughout the system. Similar patterns of higher than expected nitrogen concentrations were observed in British rivers after a drought in 1975-76 (Burt et al. 2009). Although the increase after 2003 is apparent in plots of annual mean TN concentrations for stations in the Suwannee River, short-term events like tropical storms may have masked trends calculated from monthly or more frequent data (Burt et al. 2009; Copeland et al. 2009).

CUSUM Analyses

Trends in annual mean TN concentrations were identified readily using CUSUM methods. CUSUM analyses have traditionally been used to monitor industrial processes, but more recently, they have been applied to environmental data to detect changes in indicators of fishery stocks and to examine trends in water quality data (Gibbons 1999; Mesnil and Petigas 2009).

CUSUM analyses are especially useful for environmental analyses because they are robust to missing values, relatively simple to interpret, sufficiently flexible to meet the desired objectives (e.g., changing the sensitivity to false alarms), and designed to detect changes from background noise in datasets where the type of change (e.g., linear or exponential) is unknown (Manly and MacKenzie 2003; Petigas 2009; Tam 2009).

Another advantage of CUSUM analyses is the potential to examine relationships between variables by comparing them over the same time period. Standardized CUSUM statistics, in particular, can be used to compare trends in variables that vary widely in magnitude, thereby providing insight into complex system dynamics (see Petigas 2009 and Briceño and Boyer 2010). This type of analysis would be a logical follow-up to the current study and could be used to explore how nitrogen dynamics relate to dynamics of potential nitrogen sources (e.g., fertilizer application) or potential biological responses (e.g., algal blooms).

Although CUSUM analyses can be effective in many applications, the ability to obtain reliable results is dependent on meeting assumptions of the test, and the results are sensitive to the user-defined reference value. For example, reference values usually are derived from a set of historical data for the process of interest. When the reference dataset encompasses a short period of time, variability may be underestimated and results may contain more violations than actually may have occurred. When too long a period of record is used, real trends may be contained within the reference dataset, and important changes may go undetected (Tam 2009). One way to account for this limitation is to conduct a CUSUM analysis in two stages: i) select a reference period based on expert knowledge and carry out the CUSUM analyses to determine if there are any extreme values during the selected time period then ii) remove any extreme values and re-calculate the reference value for subsequent analyses (Mertens et al. 2008). Another potential complication with CUSUM analyses of environmental time series data is the presence of serial or spatial dependence since independence of sample points is a key assumption. Where dependence is an issue, the spatial or temporal resolution of sample points can be adjusted or analyses can be modified to account for dependence (see Manly and MacKenzie 2000 and 2003).

Summary

This study demonstrated increasing TN concentrations in the Suwannee River between 1989 and 2010. During the same time period, elevated nitrogen levels also have been documented in many aquatic systems around the world, and these increases have been linked to human activities (Galloway et al. 1995; Vitousek et al. 1997; Carmargo and Alonso 2006; Howarth 2008). Where nitrogen concentrations are high, adverse impacts to ecosystems and human health can occur. To mitigate impacts from nitrogen pollution, managers need to have an understanding of nitrogen sources, factors influencing nitrogen dynamics, biological responses to increased nitrogen levels, and factors that moderate these biological responses.

The influences of hydrogeology, land use, groundwater quality, and climatic events on nitrogen dynamics in the Suwannee River were highlighted through the application of CUSUM trend analyses within a spatial context. This information will help managers develop comprehensive strategies to address concerns about nitrogen loads. First, the importance of hydrogeology was emphasized as observed trends differed from trends predicted solely on the basis of differences in geography along the river. This result suggests that management approaches will need to be adapted for different parts of the Suwannee Basin depending, in part, on whether or not the UFA is confined or unconfined. Where the river overlies confined parts of the UFA, managers should target run-off and promote land uses that do not deliver nitrogen. In areas where groundwater is more influential, nitrogen dynamics are complex and still not well understood; therefore, managers will benefit from additional research on groundwater quality and dynamics. Finally, the potential influence of long-term climatic events remains an important consideration because managers will not be able to control these cycles and their impact may mask trends in nitrogen concentrations in the river.

The relationships among nitrogen concentrations, biological responses, and mitigating factors were not addressed by this study. However, future efforts can use experimental approaches, multiple CUSUM analyses, or empirical models to investigate how nitrogen trends manifest as trends in biological indicators, such as chlorophyll-a, and how the negative impacts of these responses can be mitigated by other factors, such as water residence time. In fact, data from the lower Suwannee River and the associated estuarine waters support an empirical approach.

Table 2-1. Sampling site details. River miles are measure from an origin point in Alligator Pass (Tillis 2000) and latitude and longitude are in decimal degrees (WGS 84).

Site	Location	River mile	Latitude	Longitude
1	Near Benton, FL	191.0	30.508115	-82.716569
2	Below White Springs	171.2	30.325833	-82.738611
3	At Suwannee Springs	150.0	30.395000	-82.936111
4	At Ellaville below US 90	127.3	30.376944	-83.180278
5	At Dowling Park	113.0	30.244722	-83.249722
6	At Luraville	89.3	30.098889	-83.171944
7	At Branford	76.2	29.955556	-82.927778
8	At Rock Bluff	57.0	29.791111	-82.924444
9	Near Wilcox	34.5	29.591389	-82.937222
10	At Fowler's Bluff	17.0	29.399167	-83.022778
11	At Gopher River	5.7	29.328056	-83.103056

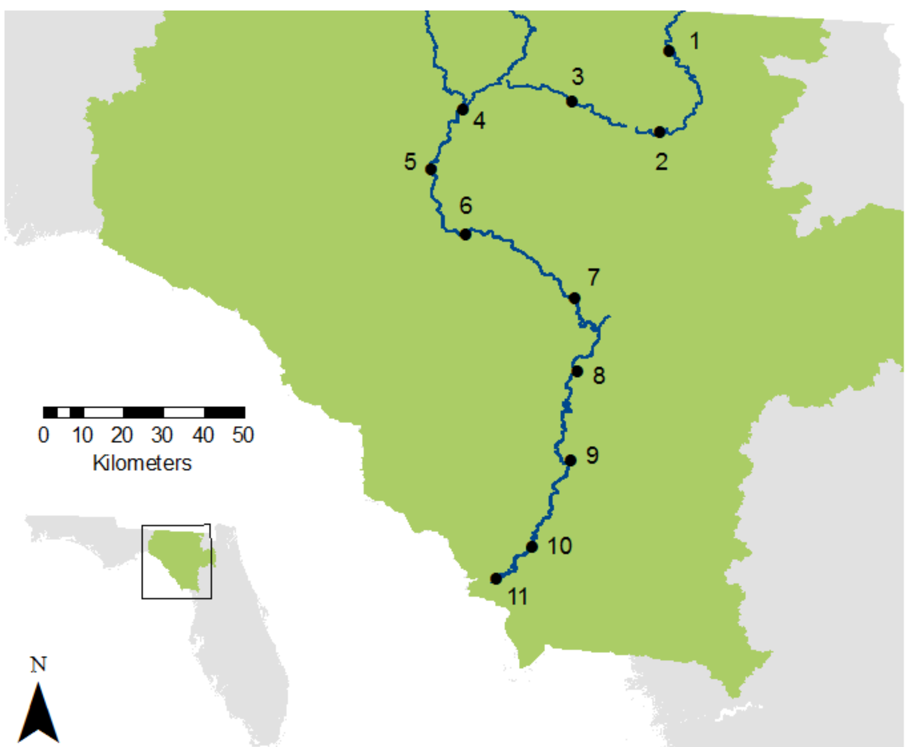


Figure 2-1. Sampling sites and the extent of the Suwannee River Basin (green).

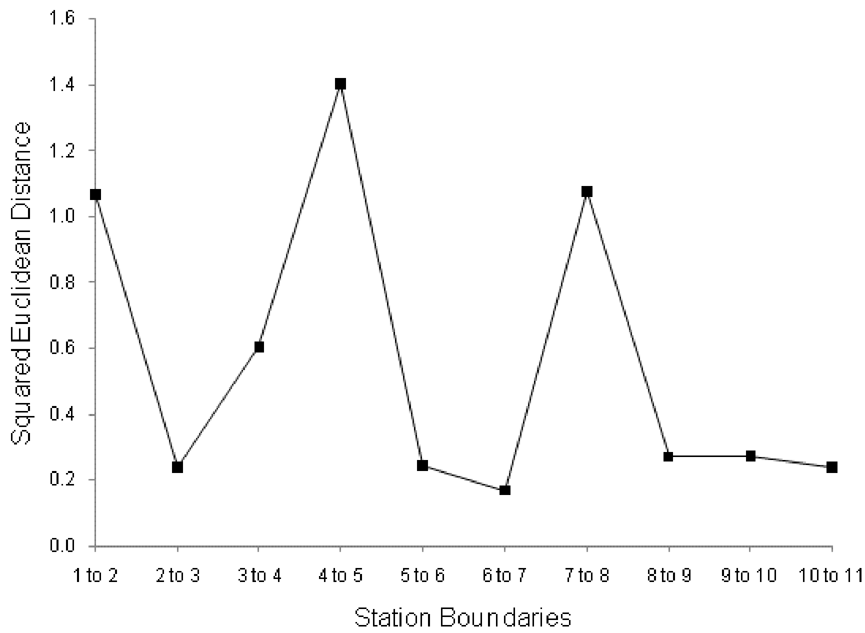


Figure 2-2. Moving split-window boundary detection results displaying differences between adjacent stations based on observed mean annual nitrogen concentrations.

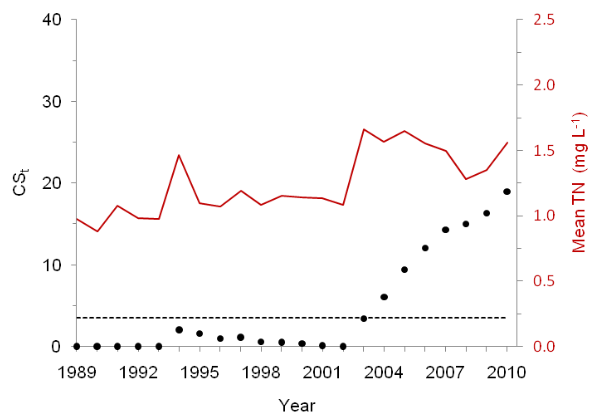


Figure 2-3. CUSUM statistics and mean annual total nitrogen concentrations (red line) for station 1 from 1989 to 2010. Horizontal dashed line represents CUSUM threshold value of 3.49.

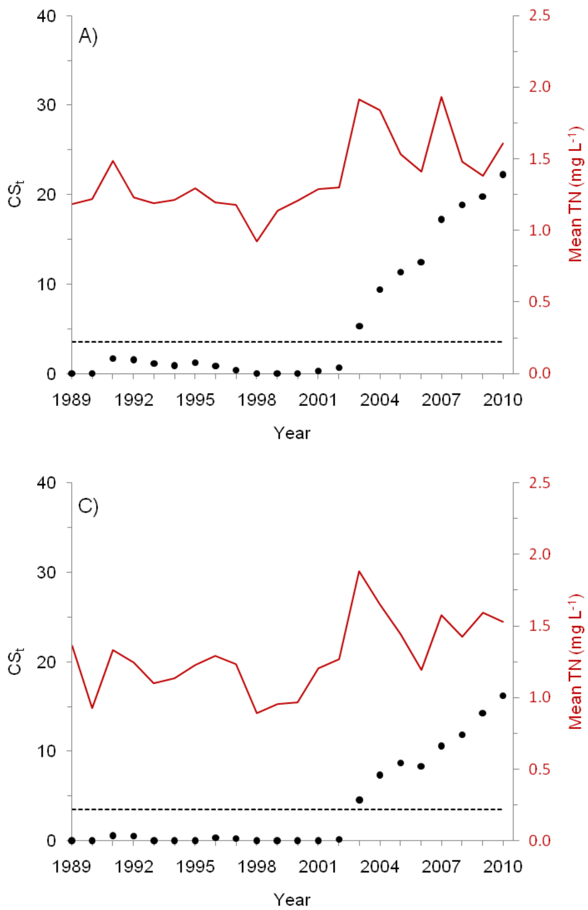


Figure 2-4. CUSUM statistics and mean annual total nitrogen concentrations (red line) for A) station 2, B) station 3, and C) station 4 from 1989 to 2010. Horizontal dashed line represents CUSUM threshold value of 3.49.

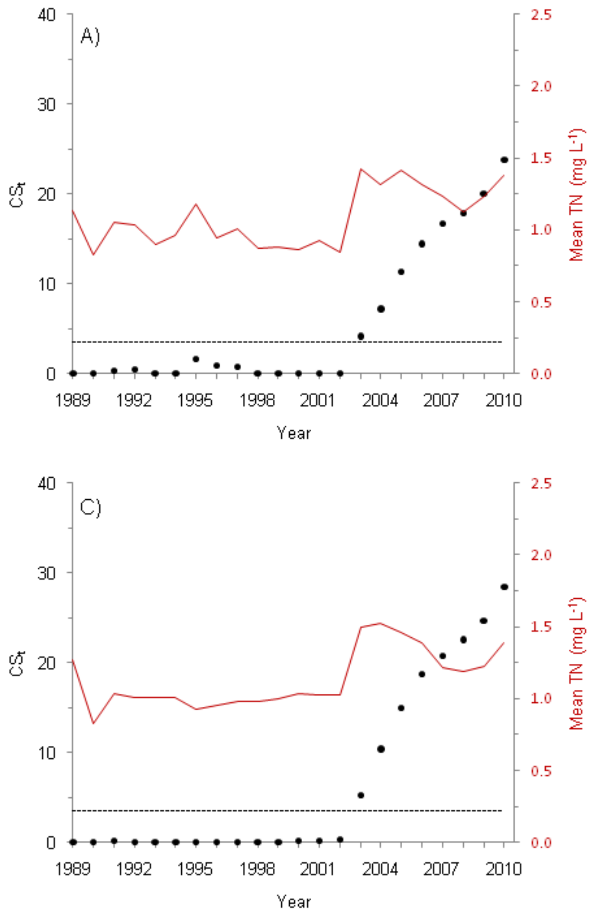


Figure 2-5. CUSUM statistics and mean annual total nitrogen concentrations (red line) for A) station 5, B) station 6, and C) station 7 from 1989 to 2010. Horizontal dashed line represents CUSUM threshold value of 3.49.

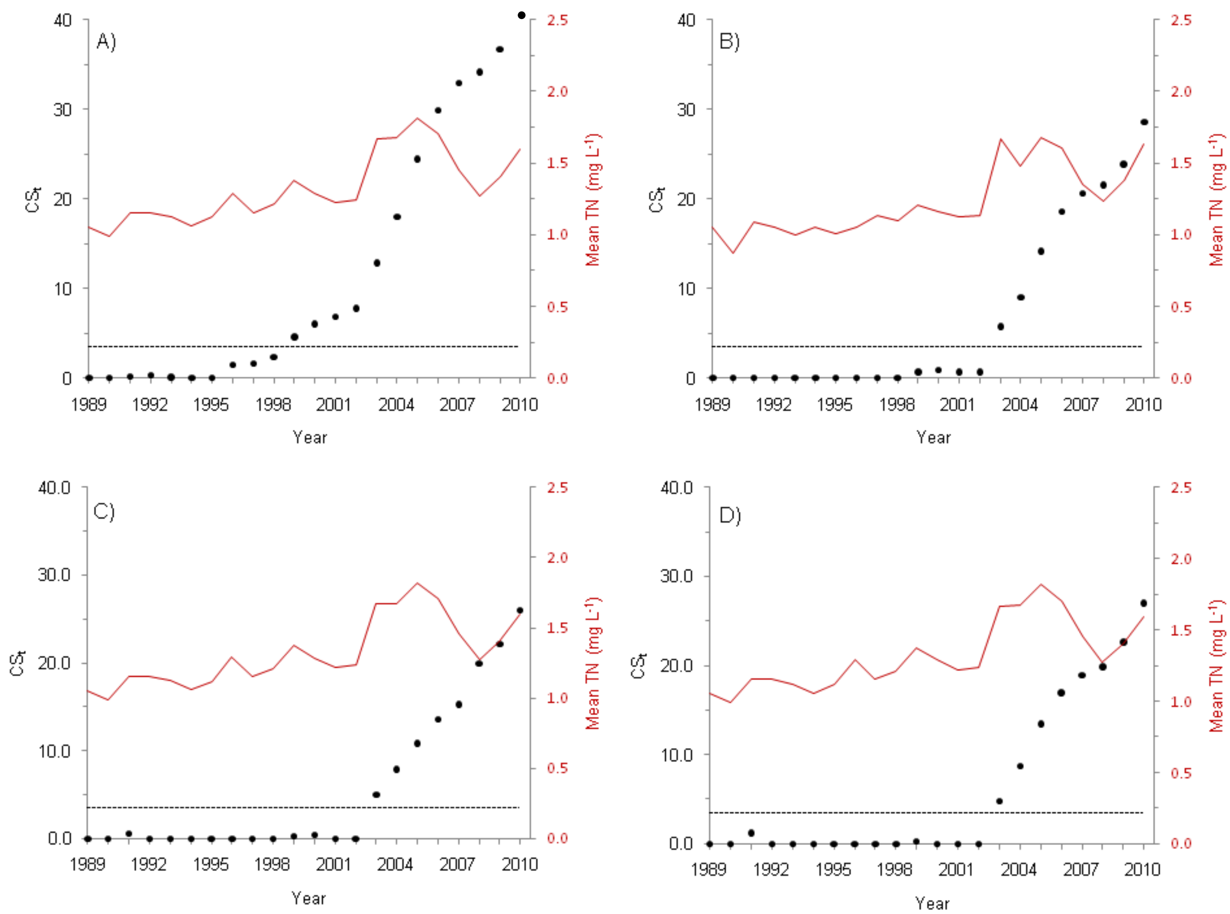


Figure 2-6. CUSUM statistics and mean annual total nitrogen concentrations (red line) for A) station 8, B) station 9, C) station 10, and D) station 11 from 1989 to 2010. Horizontal dashed line represents CUSUM threshold value of 3.49.

CHAPTER 3
CHLOROPHYLL DYNAMICS IN THE LOWER SUWANNEE RIVER AND ESTUARY

Rationale

Coastal and estuarine systems are highly productive, contain diverse habitats, and support commercial and recreational fisheries (Raabe et al. 2007). However, many estuaries have been negatively impacted by human activities in adjacent watersheds (NRC 2000). Nutrient enrichment from land-based sources is of particular concern due to the potential for increased delivery or production of organic matter, i.e., eutrophication (Nixon et al. 1996; Howarth 2008). Eutrophication can lead to increased occurrences of high chlorophyll-a concentrations, toxic algal blooms, hypoxia, or anoxia, which can decrease biodiversity and alter food web dynamics (Rosenberg 1985; Nixon 1995; Paerl 1999; Cloern 2001). The extent of impacts from eutrophication varies among systems based on differences in hydrology, basin morphology, and biological factors (Bricker et al. 2003; Painting et al. 2007). Thus, effective nutrient management requires knowledge of how these regulating factors affect biological responses to increased nutrient delivery.

River-influenced estuarine systems are particularly difficult to manage and susceptible to nutrient enrichment because rivers deliver substantial nutrient loads and contribute to complex hydrodynamics (Billen and Garnier 1997). For example, the northern Gulf of Mexico received 91% of its nitrogen load and 88% of its phosphorus load from the Mississippi and Atchafalaya Rivers during 1973-1993 (Turner et al. 2007). Because these river-introduced nutrients originate from all parts of the watershed, efforts to manage nutrients in coastal systems with strong riverine influences need to account for both upstream influences and coastal impacts.

An example of a system currently being managed as a watershed is the Suwannee River in Florida. Several segments of the Suwannee River and its estuary have been declared as “impaired” because they did not meet the state’s water quality standards for nutrients (FDEP 2002). To address this impairment, managers have implemented a multi-faceted approach involving regulatory and non-regulatory measures. Regulatory measures include Total Maximum Daily Loads, which estimate the maximum level of a pollutant that can be assimilated by the water body without causing harm, and non-regulatory measures include, for example, Best Management Practices for agriculture (Norgart 2004; USEPA 2006). A recent regulatory initiative involves the development of numeric nutrient criteria, which will provide upper limits to nutrient levels within a system designed to prevent adverse impacts to water quality and wildlife (FDEP 2010).

To establish effective numeric criteria, managers will need to understand nutrient sources and dynamics within each system and how these dynamics are manifested as biological responses. More specifically, managers need to be able to distinguish natural variation in nutrients and their associated biological responses from variation due to anthropogenic inputs. This information will help managers designate attainable magnitude, duration, and frequency components for each criterion.

A first step towards separating these sources of variation is the analysis of long-term datasets, which can provide the opportunity to place short-term events in the context of long-term patterns, allow for the assessment of processes that occur over a long timescale, and suggest potential sources of variation by highlighting patterns (Burt et al. 2010). To this end, the present study utilizes a long-term dataset for the lower

Suwannee River and its estuary to support more effective management of nutrients and their associated biological impacts. More specifically, this study provides insight into the spatial and temporal dynamics of water quality in the system by characterizing relationships between chlorophyll-a, nutrients, and other water quality parameters. Chlorophyll-a dynamics were the focus of this study because chlorophyll-a is a common proxy for phytoplankton biomass, and it can also be used as an indicator of eutrophication (Bricker et al. 2003). Other parameters that are potentially related to chlorophyll dynamics across the system also were considered, including causal parameters such as nutrient concentrations and regulating environmental variables like discharge, water color and temperature, salinity, and wind. First, long-term patterns in each variable were evaluated using trend analyses. Relationships between chlorophyll-a and other parameters were then characterized using time series and regression modeling. Results provide insight into the dynamics of and relationships between chlorophyll-a and other water quality parameters in the Suwannee River and its estuary, and results also highlight the importance of considering spatial and temporal variation when managing water quality in estuarine systems.

Methods

Study Area

The Suwannee River, a major feature of southern Georgia and north-central Florida, originates in the Okefenokee Swamp and terminates in a coastal-plain estuary comprising a network of tidal channels and salt marsh habitat along the Gulf Coast of Florida (Bales et al. 2006; Orlando et al. 1993). This river exhibits blackwater characteristics, is 394 km long, has an average annual discharge of approximately 300 $m^3 s^{-1}$, and meanders through a watershed draining roughly 28,600 km^2 (Wolfe and

Wolfe 1985). The watershed as a whole is relatively pristine with numerous forested, wetland, and protected areas; however, anthropogenic land-uses such as phosphate mining and crop and animal agriculture are present (Bledsoe and Phlips 2000; Katz et al. 1997).

The Suwannee River basin has a karstic topography characterized by features like sinkholes and springs. The high porosity of this landscape results in a dynamic exchange between groundwater features like the Upper Floridan Aquifer and surface waters. In addition, several tributaries such as the Alapaha River, the northern Withlacoochee River, and the Santa Fe River are connected to the Suwannee River (Katz et al. 1999, 2001). The drainage patterns of these hydrologically significant tributaries are used to divide the Suwannee River basin into management units known as sub-basins. Within Florida, there are five such divisions: the Alapaha sub-basin, the Withlacoochee sub-basin, the Upper Suwannee sub-basin, the Santa Fe sub-basin, and the Lower Suwannee sub-basin (Katz et al. 1997).

The focus area for this study, the lower Suwannee River and estuary, extends north from Cedar Key, Florida to Horseshoe Beach, Florida, and it is part of the Lower Suwannee sub-basin (Figure 3-1). The river and estuary are ecologically diverse, sustain both commercial and sport fisheries, and provide critical habitat for endangered species such as the Gulf of Mexico sturgeon and the Florida manatee (Raabe et al. 2007). These benefits can be affected by variations in water quality within the system, which are driven by regional and local climate patterns, freshwater withdrawal for drinking water or irrigation, groundwater input from springs, and wind events (Bales et al. 2006; Quinlan 2003; Raabe et al. 2007).

Data Sources

The primary dataset was a thirteen-year record of water quality data for the lower Suwannee River and estuary covering 1998 to 2010 (Frazer, unpublished data). Parameters extracted from this dataset were water temperature, salinity, color, and concentrations of total nitrogen, total phosphorus, and chlorophyll-a. Supplemental data acquired from various sources provided information about mean daily discharge, wind direction, color, climate indices, and extreme weather events (Table 3-1).

Sampling for core water quality parameters was conducted monthly at ten fixed stations distributed among the river, oyster reef, and nearshore zones of the system (Figure 3-2). Measurements for water temperature (°C) and salinity (ppt) were taken in situ at 1 m using a Yellow Springs, Inc. datasonde (Model: 600R). Two surface water samples were collected during each sampling event for analysis at the laboratories of the Fisheries and Aquatic Sciences Program at the University of Florida. First, whole water samples were collected to determine concentrations of total nitrogen (μg L^{-1}) and total phosphorus (μg L^{-1}). Then, a second water sample was collected and a known volume from this sample was filtered through a 47 mm glass fiber filter. The filter was placed over dessicant and frozen for later processing at the laboratory. Processing consisted of pigment extraction in ethanol and measurement of chlorophyll-a concentration (μg L^{-1}) using spectrophotometry. An acidification step was included in the spectrophotomic analyses to determine concentrations of chlorophyll-a corrected for phaeophytin (Method 10200 H; American Public Health Association 1989; Sartory and Grobbelaar 1984). Beginning in 1999, a separate filtered water sample was collected for subsequent determination of color (PCU) in the laboratory using spectrophotometry.

Complete descriptions of field and laboratory protocols are available elsewhere (Frazer et al. 1998, 2001).

Mean daily discharge (m^3 s^{-1}) data were acquired online (http://waterdata.usgs.gov /nwis/rt) for a United States Geological Survey gauging station located near the Gopher River, and they consisted of verified values averaged over a 24-hour period. Data from a gauging station near Wilcox, FL also were acquired for use in estimating values missing from the Gopher River dataset.

Wind directions were extracted from the continuous winds records of the Keaton Beach meteorological station (http://www.ndbc.noaa.gov). Records consisted of ten-minute averages of clockwise deviations from true north in degrees.

Additional color data (PCU) were acquired for two fixed stations near the mouth of the Suwannee River (http://waterdata.usgs.gov/nwis/dv). These surface water quality monitoring stations were sampled irregularly over the period of interest using methods comparable to those used for the primary dataset, and the resulting data were used to estimate values missing from the core water quality dataset.

Finally, data were acquired for extreme weather events and two climatic indices, i.e., the Southern Oscillation Index and the Multivariate El Niño-Southern Oscillation Index. Monthly time series records for the climate indices were used to evaluate potential effects of global or regional climatic trends in the regression analyses. Data extracted from the Storm Event Database of the National Climatic Data Center (NCDC) for use in regression analyses included the dates of weather events (i.e., hurricanes, tropical storms, storm surges or flooding) that were expected to influence water quality by altering discharge, runoff, or rainfall within the study area.

Trend Analyses

The presence of monotonic trends was evaluated at each station for six water quality variables: total nitrogen, total phosphorus, salinity, discharge, color, and chlorophyll-a corrected for phaeophytin (hereafter referred to as chlorophyll-a). Objectives of the analyses were to i) determine if trends in causal parameters, such as nutrient concentrations, were accompanied by trends in a response variable, chlorophyll-a, and ii) compare any significant trends between sample sites and across the study area. Monotonic trends, gradual increases or decreases during the study period, were examined rather than step trends, changes in the mean value of a parameter after a pre-defined date, because there was no a priori knowledge of an event that would cause a step change. Results of tests for normality, the presence or absence of periodicity, and serial independence were used to select specific methods to evaluate trends for each station and parameter. Except where otherwise noted, all analyses were conducted with the R statistical software package (version 2.12.2; 2011-03-25; R Development Core Team 2011).

Normality was evaluated with the Shapiro-Wilk test (shapiro.test; R Development Core Team 2011) to determine if parametric or non-parametric methods were appropriate for each combination of station and parameter. Where the null hypothesis of a normal distribution was rejected at a 90% significance level, non-parametric tests (i.e., Mann-Kendall or Seasonal Kendall) were employed due to their increased power to detect trends (Bouchard and Haemmerli 1992; Carey 2009). Parametric methods consisted of simple linear regressions with time as an explanatory variable.

Next, the presence of periodicity, cyclic variation among months within years due to variation in factors like precipitation, water temperature, and light availability, was assessed with the Kruskal-Wallis Rank Sum test (kruskal.test; R Development Core Team 2011). Because periodicity can obscure the presence of a trend or suggest a trend is significant when it actually is not, methods accounting for periodicity were applied when the null hypothesis of no difference in the distributions of values between months was rejected at a 90% confidence level (Helsel and Hirsch 1991; Qian et al. 2007). For normal data analyzed with parametric regression, periodicity was removed prior to testing for trends by subtracting the monthly median value for that station and parameter from each data point (Qian et al. 2007). Where data were non-normal, periodicity was addressed using the non-parametric Seasonal Kendall test implemented in the program *Kendall.exe* (Helsel et al. 2006). If periodicity was not present and data were non-normal, the Mann-Kendall test was used (MannKendall; McLeod 2011).

Finally, serial independence was tested at a one month lag and a 90% confidence level using the Ljung-Box Q test (Box.test; R Development Core Team 2011) because serial autocorrelation increases Type I errors in trend analyses (Darken et al. 2002; Hirsch and Slack 1984; Ljung and Box 1978). Independence tests on normal data were implemented after significant periodicity was removed by subtracting the monthly median value for that station and parameter from each data point. Non-independence was alleviated for normally distributed time series by modeling the serial autocorrelation with an autoregressive time series model, extracting the residuals from this model, and carrying out the trend test on the independent residuals. For Seasonal Kendall tests on non-normal data, significance levels were adjusted for non-independence by incorporating a covariance term into the estimation of the variance of the test statistic, S (Darken et al. 2002; Hirsch and Slack 1984). For Mann-Kendall tests, a block bootstrap method (tsboot; Canty and Ripley 2010) was used to obtain improved significance tests in the presence of significant autocorrelation (see McLeod 2011).

Regression Analyses

Overview

A combination of multivariate regression and time series modeling techniques were used to assess relationships between chlorophyll-a concentrations and salinity, water temperature, light availability, concentrations of total nitrogen and total phosphorus, as well as several larger-scale factors (i.e., river discharge, wind, weather events, and climate indices). To increase the power to detect relationships and to account for potential spatial variability in relationships, stations were grouped according to results of a non-metric multi-dimensional scaling based method on conditions observed at each site over the ten-year period.

For each group of stations, a regression model was developed using pooled, $\log_{10}$ -transformed data from 1999 to 2008, with coefficients and intercepts estimated using ordinary least squares (OLS). After satisfying diagnostic requirements for normality and homoscedasticity, residuals for individual stations were extracted and evaluated for serial autocorrelation. Where autocorrelation was significant, it was modeled using a combination of first-order and seasonal autoregressive or moving average terms to create residuals that satisfied assumptions associated with the OLS technique. The predicted values from each time series model were added to the corresponding predicted values for the same station from the group model to obtain an adjusted prediction that was evaluated to assess how well the revised predictions fit the real data. Finally, the predictive power of each model was evaluated for each station by generating adjusted predictions for chlorophyll-a concentrations based on data in holdout samples from 1998, 2009, and 2010 and comparing these results to recorded chlorophyll-a concentrations.

Data management

Missing values. Missing values represented 17% of the whole dataset and were replaced with one of three values: an average value derived from relevant data, a value for a similar location and time from another data source, or a predicted value from regression models based on relevant data. Eleven missing water temperatures were replaced with means calculated from data for the same station in the same month of all available years. Data from nearby Suwannee River Water Management District sampling stations (SUW275C1 and SUW305C1) provided replacements for twenty-three missing values in the color dataset for stations 1 and 2. A series of regression analyses provided substitutes for 241 missing values in the corrected chlorophyll-a, total nitrogen, total phosphorus, salinity, color, and river discharge sub-datasets. These regressions were based on data that were pooled across all stations and $\log_{10}$-transformed after adding 0.5 to eliminate zero values. Missing values for the corrected chlorophyll-a series were not replaced if they coincided with missing values for the uncorrected chlorophyll-a series. Missing chlorophyll-a values were not estimated for fourteen sampling events to avoid biasing regression results by inserting new chlorophyll-a values based on other available data.

Station groups. Stations were grouped according to results of an ordination implemented in PRIMER v.6 (Clarke and Gorley 2006). Non-metric multi-dimensional scaling (MDS; Clarke 1993) was used to ordinate a resemblance matrix of Bray-Curtis similarities between stations. Similarities were computed using range-transformed data for the following parameters: temperature, salinity, color, and concentrations of total nitrogen and total phosphorus. Within a given combination of parameter and month, range standardizations involved subtracting the minimum value from across all stations and then dividing by the range of values across all stations. Thus, the transformed values were scaled from 0 to 1 while retaining the relative relationships among stations and reducing the influence of differences in the absolute magnitudes of the parameters.

Descriptive analyses. Data were summarized across the sampling period (1998 to 2010) and for each year using descriptive statistics that were visualized with time series plots. Descriptive statistics calculated in Microsoft Excel consisted of the means, medians, standard deviations, and minimum and maximum values for each year and across the entire sampling period for each station and each parameter. Time series plots of raw data for groups of stations were generated for each parameter across the entire study period.

Additional variables. A number of lagged and interaction terms were generated in Microsoft Excel for consideration in the regression analyses. Terms with a one-month lag were defined for each quantitative variable as the observed value for the month before the given sampling period. Additional lag terms were defined for discharge by extracting the mean daily discharge for one day, two days, and two months prior from the full discharge dataset available online. Second-order interaction terms were generated between all quantitative variables by multiplying the raw values of the respective component variables. A single third-order interaction term was generated by multiplying values for total nitrogen, total phosphorus, and color within each time period.

Several qualitative variables were defined as 1 for present and 0 for absent to account for unique conditions that could affect a sampling event. Dummy variables consisted of terms for wind events, storm events, years with high discharge, and sampling dates with high discharge values. Wind events were defined as continuous episodes of predominantly southwest winds for periods of 12, 24, 36, or 48 h before the time of sampling as derived from average, hourly wind directions. Dummy variables for storm events were defined as a flood, severe thunderstorm, tropical storm, or hurricane preceding sampling by one month or less, with data extracted from the Storm Event Database for Dixie County, Florida. Years with high discharge were defined as those with an average annual discharge greater than the average discharge for 13 years (1998-2010) at the Gopher River station. Periods of increased discharge also were incorporated into the model by way of dummy variables for individual days of sampling with discharge values within the top 50% or top 5% of discharges recorded during the period of study.

Dummy variables with more than two levels were designated for the appropriate month and water season for each observation to incorporate these potential sources of variation in the regressions. Dummy variables for sampling month were defined with January as the base level (i.e., samples collected during January were not coded in the analyses and dummy variables for the remaining months represented effects beyond those observed in the month of January). Dummy variables representing the water season for a given observation (i.e., early dry, late dry, early wet, or late wet) were defined using the early dry season as the base level. Water seasons were defined after the United States Geological Survey (USGS) convention where June, July, and August are the early wet season; September, October, and November are the late wet season;

December, January, and February are the early dry season; and March, April, and May are the late dry season.

Finally, all quantitative variables were $\log_{10}$-transformed after adding a constant value of 0.5 to eliminate undefined values due to zeros.

Ordinary least squares regressions

Model selection. Linear regression models were developed for each group of stations using $\log_{10}$-transformed data from 1999-2008 pooled across stations within each group and an interactive stepwise procedure based on pre-defined variable selection criteria. Subsets of variables considered for each model were selected from the complete set of variables defined above based on individual relationships with the response variable, $\log_{10}$-transformed chlorophyll-a ($\log_{10}$ chl-a) concentrations. These relationships were evaluated with Spearman's rho correlation coefficients and tested for significance using an asymptotic *t* approximation (cor.test; R Development Core Team 2011). If the linear correlation with $\log_{10}$ chl-a concentrations was significant at an 85% confidence level, the variable was retained for consideration.

Variables were prioritized for inclusion according to the absolute value of the appropriate Spearman's rho correlation coefficients, with the most correlated variables being considered first. Variables were evaluated one at a time, with the exception of interaction terms, which were evaluated alongside their component terms, by adding them to the previous candidate model and estimating the new model parameters (lm; R Development Core Team 2011). The initial reference point for future candidate models was an intercept-only model with no additional covariates.

Each variable was evaluated using a combination of four criteria: i) the significance of the coefficient estimates for each variable based on a *t*-test and a 95% confidence level; ii) the significance of an *F*-test comparing nested models at a 90% confidence level; iii) the relative increase or decrease in the Akaike information criterion (AIC) value after the new variable(s) were added to the model; and 4) whether or not the variance inflation factor of the new variable was greater than 10. An additional criterion for retaining interaction terms in a given model was that the component terms, as well as the interaction term, had significant coefficient estimates to ensure that the interaction term provided information above that provided by any of the component terms alone.

Combining these criteria created six potential scenarios that determined if variables were retained in the model or removed from consideration at each step. Variables were kept in the following cases: i) when both the coefficient *t*-tests and model *F*-tests were significant; ii) when the coefficient *t*-tests were significant, the model *F*-test was not significant, and there was a decrease in the AIC value; or iii) when the coefficient *t*-test was not significant, the variance inflation factor for the new variable was greater than 10, and there was a decrease in the AIC value. Variables were not retained in the candidate model when: i) the coefficient *t*-test was significant but the model *F*-test was not significant and there was no decrease in AIC value; ii) the coefficient *t*-test was not significant and the variance inflation factor for the new variable was less than 10; or iii) the coefficient *t*-test was not significant, the variance inflation factor was greater than 10, and there was no decrease in AIC value. Variables were re-evaluated and retained or eliminated at each step in model development.

Terms were added or removed from candidate models until all the prioritized variables in the screened subset were evaluated. Due to the stepwise nature of the model development and the potential for the order of variable evaluation to affect variable significance, linear correlations were tested between model residuals and the rejected variables with Spearman's rho correlation coefficients at the end of the variable selection procedure (cor.test; R Development Core Team 2011). This final check was used to identify any variables that were initially removed from the model, but that could still have a significant relationship with chlorophyll-a. Variables showing significant linear correlation at this step were added to the model and their contributions were re-evaluated by applying the four variable selection criteria. This process was repeated until no additional, significant variation in the model residuals could be explained. Finally, the resulting model for each group of stations was subjected to a last set of diagnostics assessing normality, homogeneous variances and independence of the residuals before it was chosen as the final model.

Model diagnostics. Residuals for each model were assessed for normality and homogeneity of variance at a 95% confidence level using Shapiro-Wilk (shapiro.test; R Development Core Team 2011) and Breusch-Pagan tests (bptest; Zeileis and Hothorn 2002), respectively. If residuals were non-normal or had heterogeneous variance, datasets were examined for influential outliers. Influential observations were identified using plots of the studentized residuals as a function of hat values, which measure distance from the center of the distribution of the explanatory variables, for each observation. Plots were annotated with lines for the threshold values of each statistic. The threshold for potential outliers was a studentized residual for a given observation greater than 3.0. Observations were considered potentially influential, i.e., exerted significant leverage, when the hat value exceeded a threshold, h. This threshold was calculated separately for each model using Equation 3-1 based on k, the number of parameters estimated in the model, and *N*, the number of observations used to estimate model parameters.

$$h = \frac{2(k+1)}{N} \qquad 3\text{-}1$$

Influential outliers were examined further to determine if they were erroneous values or if they were due to unique characteristics of the given sampling effort (e.g., an extended period of heavy rainfall). Values were deleted when there was a known reason for an error (e.g., equipment failure). Otherwise, a new dummy variable was created for the particular situation (e.g., for the time periods associated with a rainfall event). New dummy variables were added to the model, evaluated according to the four criteria outlined previously, and retained in the model where they added significant information. After adding a new dummy variable, the residuals were re-evaluated for normality and homoscedasticity. Where diagnostic tests failed again, identification of outliers and influential values continued until residual diagnostic tests were satisfactory.

Residuals for each station were tested individually using the Ljung-Box test for overall independence (Box.test; R Development Core Team 2011). If residuals for a given station were independent, no further action was taken and the model selected during OLS model selection was considered the final model. For stations with residuals that were not independent at a 90% confidence level, a time series model was developed.

Time series models

Time series models were developed for autocorrelated residuals from individual stations to create a model that yielded residuals satisfying the OLS regression assumption of independence and to improve overall model fit. Serial autocorrelation was modeled using a combination of autoregressive and moving average terms that were either non-seasonal (i.e., with a 1 month period) or seasonal (i.e., with a 12 month period). Potential parameters were identified for each station based on a visual inspection of plots of the partial autocorrelation function (pacf; R Development Core Team 2011) for each set of residuals and the results of the extended sample autocorrelation function (eacf; Chan 2010). In general, where the partial autocorrelation function plot indicated no autocorrelation beyond a lag of p, an autoregressive term of lag p was evaluated, and where the partial autocorrelation function plot exhibited dampening oscillations with peaks at multiples of lag q, a moving average term for lag q was evaluated. The extended autocorrelation function provided a set of potential autoregressive and moving average terms based on iteratively fitting least squares estimates to the dataset and determining which terms were significant (Tsay and Tiao 1984). From these potential parameters, a set of candidate models were defined that ranged from including only one parameter to including as many as six parameters. To minimize model complexity, none of the candidate models included an intercept term and both non-seasonal and seasonal terms were limited to a maximum lag of 2 (i.e., only up to 2 or 24 months). Each candidate model was fit to the appropriate residuals using maximum likelihood criteria (arima; Chan 2010).

The fit of each candidate model was evaluated by examining the significance of coefficients and whether the residuals were independent. Significance of estimated coefficients was tested with a student’s *t*-test at a 90% confidence level, and independence of the new residuals was evaluated using the Ljung-Box test at a 95% confidence level. Models meeting these two criteria were ranked in increasing order of their respective AIC values and the model with the lowest AIC value was selected as the final model. Where there were multiple models with AIC values within a range of 2, the least complex model was selected as the final time series model. Fitted values from the final models were used as a time series adjustment for fitted values from the OLS regression models.

Final model evaluation

Final models were evaluated for each station by assessing fit for data from 1999-2008 and by examining the predictive ability of each model for holdout samples from 1998, 2009, and 2010.

Model fit was quantified with adjusted R-squared and root mean squared error values calculated when the final model was applied to data from 1999-2008 for each station. Fit statistics were calculated for both the final model without time series adjustments and the final model with time series adjustments, wherever applicable. Time series adjustments were incorporated into the predictions by adding the predicted value from the time series model to the predicted value from the OLS regression model.

Furthermore, root mean squared error values were calculated for yearly subsets of the study period to examine where each model performed well and where it did not. These smaller scale statistics were supplemented by visualizations consisting of time series plots of the observed and adjusted predicted values for each station.

Finally, the predictive ability of each station model was evaluated by calculating root mean squared error values when the models were fit to data from 1998, 2009, and 2010. The accuracy of hindcasts and forecasts also were evaluated qualitatively by examining plots of the observed and adjusted predicted values from each model for each year and station.

Results

Trend Analyses

Test selection

Tests were selected to produce consistency and to address violation of assumptions. Non-parametric methods were used for all trend analyses to facilitate comparisons among results even though Shapiro-Wilk tests indicated a violation of the normality assumption in only 1 of the 50 cases tested. Periodicity was detected in 31 cases. Of these, 5 were independent and evaluated with the standard Seasonal Mann-Kendall test while 26 were not independent and evaluated with an adjusted form of the Seasonal Mann-Kendall test. In the remaining cases with no periodicity, 2 were independent and evaluated with the standard Mann-Kendall test while 17 were not independent and the significance of the Mann-Kendall test was adjusted using block bootstrap methods.

Trends

Trends were detected in some parameters at some stations. Increasing trends were significant at a 90% confidence level for total nitrogen at stations 1 and 5 and for total phosphorus at station 9 (Tables 3-2 and 3-3). No trends were significant in the corrected chlorophyll-a series at any stations at a 90% confidence level (Table 3-4). However, chlorophyll-a exhibited a decreasing trend at station 3 and an increasing trend at station 6 that were significant at an 85% confidence level (Table 3-4). Significant increasing trends were detected for total nitrogen at station 7 and total phosphorus at station 10 at an 85% confidence level (Tables 3-2 and 3-3). No significant trends were observed in the color or salinity time series at any station at an 85% significance level or higher (Tables 3-5 and 3-6). Mean daily discharge values at the Gopher River gauging station did not show a significant trend over the course of the study period (*S*=299; Kendall's tau=0.02; *P*=0.68).

Regression Analyses

Data management

Missing values. Regression models were used to estimate missing values for color, salinity, total nitrogen, total phosphorus, and discharge and were based on a variety of explanatory variables (Tables 3-7 to 3-11).

Station grouping. Non-metric multi-dimensional scaling resulted in four groups of stations with a two-dimensional stress value of 0.01, which indicated a reliable result (Figure 3-3). The first group consisted of stations 1 and 2. The second group contained only station 4. The third group consisted of stations 3, 5, and 6 and the fourth group contained stations 7, 8, 9, and 10.

Descriptive statistics. Time series plots and descriptive statistics for chlorophyll-a, total nitrogen, total phosphorus, color, water temperature, and salinity showed similarities and differences in patterns among stations and parameters. Complete sets of descriptive statistics for each parameter and station are available in the Appendix.

Across the study period, chlorophyll-a concentrations were least variable, and lowest, at stations 1 and 2 and most variable, and highest, at stations 3, 5, and 6 (Figures 3-4 and 3-5). In fact, variability in chlorophyll-a at stations 3, 5, and 6 (Tables A-3, A-5, and A-6) was at least twice as high as that observed at stations 1 and 2 (Tables A-1 and A-3). Within years, the period between 2000 and 2002 was the most variable for all stations except station 1 and the lowest variability in chlorophyll concentrations was during 1998, 2006, or 2007 for all but stations 4 and 6 (Tables A-1 to A-10). Despite the low variability during 2006, a large, positive deviation in chlorophyll-a was observed at all stations during this year (Figures 3-4 to 3-7). These deviations during 2006 coincided with the end of a period of relatively high variability in concentrations of total nitrogen, total phosphorus, color, salinity, and discharge, which appeared to begin around 2003.

Relatively higher variability in total nitrogen concentrations between 2003 and 2006 was most apparent at stations 3 through 10 (Figures 3-8 to 3-11); although at stations 1 and 2, the apparent decrease in concentrations between 2003 and 2006 seemed to track the pattern that occurred between 1998 and 2002 (Figure 3-7). A similar, but weaker signal during these two periods also is apparent at stations 3 through 6, but not at stations 7 through 10 (Figures 3-7 to 3-11). Most stations did not experience any large deviations, with the exception of station 7, which had a high value during 2009 that was more than three times the mean value across the study period (Table A-17).

The period of high variability in the middle of the study period also was noticeable in time series plots of total phosphorus concentrations. This relative increase in variability was most conspicuous at stations 7 through 10 (Figure 3-15), but it also was evident at stations 4, 3, 5, and 6 (Figures 3-16 and 3-17). In addition to this period of variability, large deviations in total phosphorus were observed during 2008 at stations 1, 2, and 4 (Figures 3-12 and 3-14) and during 2004 at stations 1, 2, 7, 8, 9, and 10 (Figures 3-12 and 3-15), which was similar to patterns in total nitrogen. Finally, mean concentrations of total phosphorus across the study period were lowest at stations 7, 8, 9, and 10 and almost four times as high at stations 1 and 2 (Tables A-27 to A-28, A-21, and A-22).

In addition to a period of high variability during 2003 to 2006 that also was observed in nutrient concentrations, color exhibited an additional period of relatively high variability at most stations between 2008 and 2010 (Figures 3-16 to 3-19). Furthermore, color was elevated at all stations during 1998 (Figures 3-16 to 3-19). Maximum color values at all stations were observed during one of these three time periods, with color values being consistently highest at stations 1 and 2 (Tables A-31 and A-32). Station 1 also exhibited the largest range in color values, while values at station 8 exhibited the smallest range of all stations (Tables A-31 and A-38).

In contrast to the other parameters, patterns in water temperature did not display noticeable periods of increased variability across the study period. At all stations, water temperatures fluctuated in a cyclical pattern that followed expected seasonal trends (i.e., colder temperatures in the winter months; Figures 3-20 to 3-23). However, stations 3, 4, and 7 were subject to a wider range of temperatures than other stations across the study period and station 4 exhibited the lowest temperature (Tables A-43, A-44, and A-47).

Finally, timing of patterns in salinity and discharge were similar to patterns observed in the color series with the presence of three periods of increased variability (Figures 3-24 to 3-28). However, this pattern was not present at stations 1 and 4 (Figures 3-24 and 3-26). Station 1 had minimal variability in salinity across the study period (Table A-51) while salinity at station 4 was the most variable across the study period with no apparent patterns (Table A-54; Figure 3-26). Where patterns were visible, decreases in salinity tended to occur close to increases in discharge (Figures 3-24 to 3-28). In fact, the minimum salinity value of 0 ppt was observed at half of the stations (1 through 5) during 1998, the same year that the maximum mean daily discharge was observed (Figures 3-24 to 3-26, 3-28).

Model descriptions

Stations 1 and 2. The group model developed using pooled data from stations 1 and 2 had nineteen terms and explained 72% of the variability in $\log_{10}$ chl-a from 1999 to 2008 ($F_{19, 217} = 33.29$, $P = 0.00$; Table 3-12). Two of the terms were customized dummy variables for 99th percentile chlorophyll and November 2007. The 99th percentile chlorophyll dummy variable represented sampling events where the observed chlorophyll-a concentration was in the top 1% of observed values across all stations and the entire study period (i.e., values > 30.23 µg L^{-1}) while the November 2007 term was used to model influential observations during that month. Other variables included an interaction term between salinity and total phosphorus and two lagged terms for discharge and total phosphorus. Each of these terms displayed a negative relationship with $\log_{10}$ chl-a (Table 3-13).

Residuals extracted from the group model were significantly autocorrelated for both stations 1 and 2 (Table 3-14). Autocorrelation in station 1 residuals was modeled with one non-seasonal, first-order autoregressive term (Coefficient = 0.33, S.E = 0.09, P = 0.00) while the model for station 2 residuals consisted of a non-seasonal, first-order autoregressive term (Coefficient = 0.19, S.E. = 0.09, P = 0.01) and a first-order, seasonal moving average term (Coefficient = -0.24, S.E. = 0.11, P = 0.01).

Station 4. The final model for station 4 explained 34% of variation in $\log_{10}$ chl-a, and it had the fewest terms of all the group models, i.e., four ($F_{4,\,114}$ = 16.12, P = 0.00; Table 3-15). Of the four terms, $\log_{10}$ temperature displayed a large, positive relationship with $\log_{10}$ chl-a, while $\log_{10}$ color was the only variable with a negative relationship to the response variable (Table 3-16).

Residuals from this model were significantly autocorrelated (Table 3-14) and a time series model was developed that had one first-order, non-seasonal moving average term (Coefficient = 0.34, S.E. = 0.09, P = 0.01).

Stations 3, 5, and 6. The final model explained 34% percent of the variation in $\log_{10}$ chl-a for pooled data from 1999-2008 ($F_{6,\,351}$ = 31.57, P = 0.00; Table 3-17). The model contained two dummy variables for observations coinciding with the following conditions: greater than 50% discharge (i.e., values greater than 5973 $m^3\ s^{-1}$) or southwest 36-h wind events.

Residuals extracted from the group model were independent for station 3, while residuals for stations 5 and 6 were significantly autocorrelated (Table 3-14). Autocorrelation was modeled for station 5 and 6 residuals with first-order, non-seasonal autoregressive terms (Coefficient = 0.20, S.E. = 0.09, P = 0.00 and Coefficient=0.25, S.E. = 0.09, P = 0.00, respectively).

Stations 7, 8, 9, and 10. The final model for pooled data from stations 7, 8, 9, and 10 explained 55% of variation in the observed $\log_{10}$ chl-a series for 1999-2008 ($F_{19,\,456}$ = 30.99, P = 0.00; Table 3-19). Among the nineteen variables in this model were five dummy variables, including dummy variables for month of the year and one term generated to account for influential outliers during October of 2004. Log_{10} salinity was the most significant parameter overall, and statistics indicated a negative relationship with $\log_{10}$ chl-a (Table 3-20).

Residuals from the group model were significantly autocorrelated for stations 7, 8, and 9, but not for station 10 (Table 3-14). Therefore, time series models were developed for only stations 7, 8, and 9. Time series models for stations 7 and 9 had one first order, non-seasonal autoregressive term (Coefficient = 0.17, S.E. = 0.09, P = 0.05 and Coefficient = 0.24, S.E. = 0.09, P = 0.01, respectively). The time series model for station 8 had one first order, non-seasonal moving average term (Coefficient = 0.27, S.E. = 0.09, P = 0.03) and one first order, seasonal moving average term (Coefficient = 0.22, S.E. = 0.10, P = 0.01).

Model fit

Fitted values generally followed overall trends in observed chlorophyll at individual stations from 1999 to 2008 (Figures 3-29 to 3-32). However, most unadjusted fitted values underestimated large deviations in $\log_{10}$ chl-a. Overall, the accuracy of unadjusted fitted values as measured by root mean squared errors (RMSE) was highest for stations 1 and 2 and lowest for stations 3, 5 and 6, with results for station 3 being least accurate (Table 3-21).

When accuracy was assessed within years, it was highest across all stations during 2000, 2001, 2002, and 2007 and lowest during 1999 (Table 3-21). Throughout the most accurate years, stations 7 and 8 were among the top three most accurate stations (Table 3-21). Station 7 fitted values also were the most accurate relative to other stations during 1999, whereas accuracy for station 8 during this year was among the lowest of all stations (Table 3-21). Relative accuracy varied among stations across other years, although some general patterns existed. First, fitted values from stations 1 and 2 were consistently among the most accurate of all stations for all years (Table 3-21). Second, fitted values from stations 9 and 10 showed the most variability in relative accuracy across the years. In fact, the highest observed RMSE came from station 10 during 1999, and the second-highest accuracy came from this station during 2004 (Table 3-21). Finally, fitted values for stations 3, 4, 5, and 6 tended to be among the least accurate across all years (Table 3-21).

Accuracy of fitted values was differentially affected after time series adjustments were applied. Across the time period used to develop the models, time series adjustments had no effect on individual model accuracy at most stations, with improvements being observed only for stations 4 and 5 (Table 3-21). The improved fit for stations 4 and 5 was apparent in fitted values that more closely followed short-term variation in observed $\log_{10}$ chl-a (Figures 3-30 and 3-31B).

Time series adjustments also increased accuracy in many instances when RMSE values were calculated within years; however, patterns of improvement at this shorter timescale varied among years and stations. Time series adjustments resulted in improvement for all of the eight relevant stations during 1999 (Table 3-21). Other years where accuracy was improved at most stations included 2003, 2005, and 2006 (Table 3-21). On the other hand, time series adjustments were least effective in 2008 where accuracy of fitted values was improved at only stations 2 and 4 (Table 3-21). When considered across years and within stations, time series adjustments were least effective in improving accuracy at stations 5, 7, and 8 and most effective at stations 1, 2, and 4 (Table 3-21). In fact, the largest absolute improvement in accuracy was observed at station 4 during 2005 where the RMSE decreased from 0.30 to 0.26 after applying time series adjustments.

Model evaluation

Hindcasts and forecasts were generally less accurate than models fitted to data from 1999-2008. Averaged across the relevant years and stations, accuracy was poorer for time series adjusted predictions derived for the holdout samples (i.e., RMSE for 1998, 2009, and 2010 = 0.28 and RMSE for 1999-2008 = 0.21). Similarly, unadjusted RMSE values for hindcasts and forecasts were larger than 73% of similar values for data from 1999-2008 (Table 3-21). Accuracy of estimates was improved for 2010 and essentially maintained for 1998 and 2009 when time series adjustments were applied. Time series adjustments also improved accuracy in at least one instance for stations 1, 2, 4, 7, and 8 across the three holdout years (Table 3-21).

Improvements were visible in plots of observed, hindcasted, and forecasted values for stations 1 and 2. For example, unadjusted values missed some of the variation in observed values at station 1 during the fall and spring of 1998, the summer and fall of 2009, and the spring and summer of 2010, but time series adjusted values performed better (Figures 3-33A-C). However, there were periods where time series adjusted forecasts overestimated fluctuations in observed chlorophyll (e.g., for station 2 during April and May of all years; Figures 3-34A-C).

Hindcasts and forecasts for stations 3, 5, and 6 also missed some of the fine-scale variation in observed values. Overall, the predictions were most accurate at station 3 during 2010 (Table 3-21); nevertheless, predicted values did not match the minimal fluctuation in chlorophyll observed at this station during early 1998 (Figure 3-35A) or February through April of 2009 (Figure 3-35B). Variation at station 5 was not captured for May to September 2009 or for 2010 (Figures 3-36B and 3-36C), and unadjusted predicted values for station 5 had the worst fit across all stations for the holdout samples (Table 3-21). Lastly, time series adjustments did not improve accuracy of the model for station 6, where large changes in chlorophyll concentrations occurred (Figures 3-37A-C). In fact, RMSE values were unchanged for 1998 and 2010 and increased for 2009 (Table 3-21).

Like stations 5 and 6, time series adjustments for station 4 had small impacts on the accuracy of predictions, and the only increase in accuracy was observed during 2010 (Table 3-21). This pattern is evident in plots of observed, hindcasted, and forecasted values for each year, with unadjusted predictions tending to miss key variation in observed values (e.g., in September of 1998; March, June, and August of 2009; and November and December of 2010) and time series adjusted values overestimating fluctuations during the same time periods (Figures 3-38A-C).

Finally, the pattern of following overall trends in observed values, but missing larger deviations also was seen in hindcasts and forecasts for stations 7, 8, 9, and 10. For 1998, unadjusted hindcasts at stations 7, 8, and 9 predicted less variation than what was observed (Figures 3-39A, 3-40A, and 3-41A), and hindcasts at station 10 underestimated observed values, particularly in the beginning of the year (Figure 3-42A). However, time series adjusted hindcasts more closely followed observed values as indicated by the reduction in RMSE values for this year (Table 3-21). In addition to missing some variation in observed chlorophyll values, initial values for both unadjusted and adjusted 1998 hindcasts were very different from the observed values for all stations (Figures 3-39A, 3-40A, and 3-41A). In contrast, forecasts for 2009 and 2010 followed overall trends well at stations 7, 8, and 9 (Table 3-21), and time series adjusted forecasts did not overestimate fluctuations as much as in 1998 (Figures 3-39B, 3-40B, and 3-41B). At station 10, forecasts performed well in 2009, with the exception of November where an increase in chlorophyll was not predicted (Figure 3-42B), and the accuracy of forecasts decreased after April 2010 (Figure 3-42C). Overall, forecasts for station 10 were the most accurate of the four stations, with an average RMSE across the three years of 0.27 compared to values of 0.28 for stations 7 and 8 and 0.29 for station 9.

Discussion

Trend Analyses

Few significant trends were detected in water quality time series from the lower Suwannee River and estuary between 1998 and 2010. Additionally, there were no apparent spatial patterns in significant trends and trends in nutrients did not coincide with trends in chlorophyll-a. The small number of significant trends may be a consequence of the absence of monotonic trends over the period of study or may be due to the insensitivity of the trend analysis methods to short-term trends.

The lack of significant trends in discharge, salinity, and color may reflect true characteristics of the system because dynamics of these parameters are influenced by physical processes that fluctuate within or between years, with greater stability on longer timescales. For instance, flow in the Suwannee River consists primarily of input from tributaries driven by precipitation and groundwater flux (Grubbs and Crandall 2007). Patterns in precipitation and groundwater flux are influenced strongly by regional and global climatic cycles that occur across multiple years or longer periods; although, events like tropical storms can cause significant, short-term increases in flow (Copeland et al. 2009; Carlson et al. 2010). In fact, the study period did include El Niño events during 1998 and 2003-2003 and La Niña events during 1999-2001 and 2007-2008 (Grubbs and Crandall 2007); however, periods of high and low discharge associated with these events and other short-term incidents seemed to be sufficiently balanced across the 13-y study period so that no overall trend was detected.

Riverine discharge significantly alters several other parameters. Thus, it is not surprising that no trends were detected in salinity or color because these parameters are related strongly to discharge in the Suwannee estuary (Bales et al. 2006; Quinlan et al. 2009). The lack of trend in discharge also may explain the small number of significant trends in total nitrogen and total phosphorus given that concentrations of these nutrients also can be linked to river discharge (Bledsoe and Phlips 2000). Finally, unquantified spatial and temporal variation in freshwater inputs or surface runoff from salt marshes and tidal creeks to the north and south of the river mouth could have masked overall trends.

Although most parameters did not have significant monotonic trends across the study period, significant trends were detected at some sites for concentrations of total nitrogen, total phosphorus, and chlorophyll-a (Tables 3-2 through 3-4). Significant trends in nutrients did not exhibit obvious spatial patterns as trends were detected in each region of the study area; however, all trends indicated significant increases across the study period. Furthermore, none of the significant nutrient trends coincided with the significant trends in chlorophyll-a concentrations detected at stations 3 and 6. The fact that nutrient trends were not manifested as clearly linked trends in chlorophyll-a concentrations may be due to differences in other factors regulating phytoplankton biomass (e.g., grazing or light availability).

In addition to the possible natural explanations, the small number of significant trends may be an artifact of the trend analysis. One potentially influential aspect was the timescale at which trends were assessed. The potential influence of temporal scale is apparent when trends in total nitrogen are considered. In the present study, total nitrogen trends were assessed over 13 y using methods that were insensitive to trends at shorter timescales (Hirsh and Slack 1984). This insensitivity can explain why significant trends were found at only three stations, although time series plots of total nitrogen concentrations suggested trends at shorter timescales. For example, time series plots for stations 1 and 2 suggested decreasing concentrations of total nitrogen from 1998 through 2002, a large increase the following year, and then a subsequent decrease through 2008 (Figure 3-8). Furthermore, when examined in the context of longer datasets, total nitrogen concentrations are not as stable as the present results suggest. For instance, a 23-y dataset showed increases in annual mean total nitrogen concentrations of almost 50% within downstream portions of the river (see Chapter 2).

Regression Analyses

Model interpretation

Relationships between chlorophyll-a concentrations and a range of explanatory terms varied across time and space, as indicated by differences in model parameters among groups of stations. Nevertheless, there were consistently significant relationships between rates of change in chlorophyll-a concentrations and rates of change in salinity, color, and concentrations of total phosphorus at all stations. The prevalence of these terms, which are related to water residence time and light and nutrient availability, agree with other work showing these factors are important environmental controls of phytoplankton dynamics (Bledsoe et al. 2004; Lawrenz et al. 2010).

Short water residence times negatively affect accumulation of phytoplankton biomass in the Suwannee River and estuary, where flushing rates range from 1 to 6 days, depending on discharge (Bledsoe et al. 2004). Discharge is closely linked to salinity (Bales et al. 2006); thus, the prevalence of a salinity term may represent the influence of residence time on phytoplankton biomass (i.e., residence times increase when discharge is relatively low and salinities rise).

The extent of the riverine plume varies across the Suwannee estuary based on discharges, tides, currents, and coastal winds (Bales et al. 2006). More specifically, these factors influence how far offshore and alongshore the plume reaches, thereby affecting light and nutrient availability across the system.

The water in the plume tends to have high concentrations of colored dissolved organic matter, which lowers light availability (Bledsoe et al. 2004). In fact, color is generally the largest contributor to light attenuation in the estuary (Bledsoe et al. 2004). This relationship between light availability and color helps explain why changes in color concentrations explained considerable variation in chlorophyll-a concentrations across all sites considered in this study. Interestingly, a positive relationship was indicated between $\log_{10}$ color and $\log_{10}$ chl for the group model for stations 7, 8, 9, and 10, contrary to the expected negative relationships that were observed at the other stations (Table 3-20). This result may be due to unique characteristics of the nearshore region, which are discussed below.

Similar to light availability, nutrient availability across the system is impacted by changes in the extent of the Suwannee River plume, with subsequent effects on phytoplankton dynamics. Rivers can contribute substantial nutrient loads to coastal systems (Billen and Garnier 1997; Turner et al. 2007), and large, riverine nutrient loads have been observed in the Suwannee system (Ashbury and Oaksford 1997). Riverine nutrient loads typically support phytoplankton growth, as observed by Bledsoe et al. (2004), with bioassays indicating no nutrient limitation in the lower Suwannee River and parts of the Suwannee estuary during most of that study. However, the same study documented nitrogen and phosphorus limitation in the oyster reef and nearshore areas of the estuary during some periods, and nitrogen was commonly the limiting nutrient (Bledsoe et al. 2004). In fact, nitrogen is generally considered to be the primary limiting nutrient in marine systems (Boynton et al. 1982). Therefore, significant relationships between chlorophyll-a and total phosphorus concentrations, instead of total nitrogen concentrations, represented unexpected results (see Frazer et al. 1998).

These results may be due to differences in sources and dynamics of nitrogen and phosphorus in the system. Nitrogen in the Suwannee River originates primarily from groundwater sources (Katz et al. 1997), and these nitrogen-rich groundwaters sustain the base flow of the river during low flow periods (Katz et al. 1999). On the other hand, phosphorus enters the system through runoff and point sources, which primarily are influenced by weather events and higher discharges (Asbury and Oaksford 1997). Therefore, during periods of low or below-average flow, nitrogen will still be supplied to the estuary through groundwater inputs, but phosphorus inputs will be decreased. Furthermore, the extent of the plume will decrease during low flow periods, resulting in decreasing phosphorus concentrations further from shore. Over the long-term, these differing dynamics could result in nitrogen concentrations being more stable than phosphorus concentrations, which would explain why changes in total phosphorus explained more of the variation in chlorophyll-a than changes in total nitrogen concentrations. Nevertheless, further research will be required to fully elucidate the dynamics of nitrogen and phosphorus limitation in the Suwannee estuary, and the present findings highlight the need for continued investigation into impacts from both nitrogen and phosphorus.

Other terms explaining significant variation in chlorophyll-a also were related to residence time and the availability of light and nutrients, although these relationships varied among stations. Such differences across the study area are not surprising given results of previous studies showing differences in nutrient limitation of phytoplankton growth, primary drivers of light availability, and influence of the river plume among the river, oyster reef, and nearshore areas of the Suwannee Estuary.

Within the river, short residence times and low light availability (relative to other parts of the system) tend to limit accumulation of phytoplankton biomass despite the presence of sufficient nutrients to support growth (Bledsoe and Phlips 2000; Bledsoe et al. 2004). These characteristics were observed in the present study for stations 1 and 2, which had the highest average concentrations of total nitrogen, total phosphorus, and color and the lowest average concentrations of chlorophyll-a across the study period (Appendix A). The importance of residence time also was indicated by the significance of terms for discharge in the previous month and an interaction term between salinity and total phosphorus. The lagged term shows the potential for delayed responses to changes in discharge, and the negative coefficient for the interaction term can be interpreted as an antagonistic interaction between salinity and total phosphorus. More specifically, this interaction indicates the potential for the importance of one explanatory variable to be decreased by the presence of another variable (Cohen et al. 2003). In this case, an expected increase in phytoplankton biomass due to increased total phosphorus concentrations can be reduced if residence time decreases, as indicated by a decrease in salinity.

Residence time and availability of light also impact phytoplankton dynamics in the oyster reef area of the Suwannee estuary. In the oyster reef region, increased discharge shortens residence times and decreases light availability (Bledsoe et al. 2004). However, the reef structure can slow flushing to some extent, which can allow phytoplankton biomass to accumulate (Bledsoe et al. 2004). Residence times in the reef region also can be increased by onshore winds, as indicated by the significance of a term for a 36-h southwest wind event prior to sampling in the model for stations 3, 5, and 6 (Table 3-18). In terms of light availability, models for station 4 and stations 3, 5, and 6 indicated similar, negative relationships between $\log_{10}$ color and $\log_{10}$ chl-a (Tables 3-16 and 3-18). Estimated coefficients for each model were nearly equal but smaller than the coefficient in the model for stations 1 and 2 (Table 3-13), which suggests an increased contribution to light attenuation from tripton in the shallower, reef area as previously demonstrated by Bledsoe et al., (2004). Generally shallower depths at the reef stations (Frazer et al. 1998) also help to explain the significant effect of temperature in each of the relevant models, because shallow water would experience a wider range of temperatures, including periods of cold that could reduce phytoplankton growth rates (Irwin and Finkel 2008).

In contrast to the river and reef, the nearshore region of the estuary is frequently nutrient limited, with sufficient light to support phytoplankton growth (Bledsoe et al. 2004). These characteristics can be linked to the decreased influence of the riverine plume further from shore (Bledsoe et al. 2004; Quinlan and Phlips 2007). However, the plume periodically extends across the nearshore region during periods of high flow, and the significance of these events for phytoplankton dynamics in the region was highlighted in the significant parameters contained in models for stations 7, 8, 9, and 10. For instance, five of the nineteen terms were related to discharge, and three of these terms specifically represented periods of high discharge when the plume is expected to impact the nearshore sites (i.e., dummy variables for Greater than 50% discharge, Storm event, and October 2004). When the plume extends offshore, light attenuation and nutrient availability increase. The influences of these changes are represented in the regression model by significant parameters for total phosphorus concentrations and color (Table 3-20). Coefficient estimates for both terms indicated a positive effect on chlorophyll-a concentrations, which was unexpected for the color term. The positive relationship with color may reflect the dominant effect of increased nutrient concentrations.

Evaluation of modeling methods

The combination of multiple regression and time series techniques provided insight into relationships between chlorophyll-a and water quality parameters in the lower Suwannee River and estuary despite some known limitations.

A primary advantage of the regression approach was the ability to highlight parameters explaining variation in chlorophyll-a dynamics, which can provide more information than basic correlative analyses. For example, Bledsoe et al. (2004) hypothesized that sustained southwesterly winds could increase residence time within the estuary resulting in an accumulation of phytoplankton biomass. This hypothesis was based on observations of elevated chlorophyll concentrations after periods of onshore winds, and it is supported by results for two groups of stations in the present study. Models for reef stations (3, 5, and 6) and the nearshore stations (7, 8, 9, and 10) indicated a positive relationship between $\log_{10}$ chl-a concentrations and a 36-h period of southwesterly winds prior to the sampling event (Tables 3-18 and 3-20). Despite these results, a direct link cannot be confirmed between these or any other explanatory variables until cause and effect relationships have been explored through experimental studies.

In addition to suggesting future experimental studies, results highlighted the need to explore other factors that influence chlorophyll-a dynamics. One area highlighted for further consideration was the influence of salt marshes and tidal creeks around the estuary. Sampling has shown higher nutrient concentrations in these areas relative to the oyster reef and nearshore parts of the estuary, with no straightforward relationships between these sources and discharge from the Suwannee River (FDEP 2010). The unknown influence of these inputs may explain why models for stations 3, 4, 5, and 6 in the reef region had the poorest fit. Future modeling should attempt to quantify freshwater input and nutrient loads from these areas in addition to inputs originating from the main channel of the Suwannee River, which was the focus of this study.

Another advantage of the modeling arose from the time series components, which were used to improve fit and satisfy assumptions regarding residual independence. Although time series adjustments did not improve fit to the extent expected, adjustments did bring predicted values closer to observed values where large deviations occurred. This improvement is likely due to the presence of inherent serial autocorrelation in biomass accumulation, i.e., the amount of biomass at a given time is dependent on the amount of biomass present at one or more earlier time steps. In fact, in a chlorophyll time series model developed by Lehman (1992), exogenous variables explained very little of the observed variance in comparison to temporal correlation among chlorophyll concentrations. On the other hand, decreased fit after time series adjustments to models of the Suwannee data most often was due to exaggeration of large deviations. For example, the time series component in the station 4 model improved predictions of the timing of larger deviations in chlorophyll-a values, but the magnitudes of predicted increases were too high (e.g., in September of 2000; Figure 3-38).

The prevalence of overestimates points to the value of incorporating other controls on phytoplankton biomass. For example, the models did not incorporate competition for resources, grazing pressure, losses due to sinking or senescence, or other factors that can regulate large increases phytoplankton biomass; therefore, predictions exceeded values that would occur naturally.

A failure to incorporate biological controls and other factors also may help explain differences in the performance of models among stations and time periods. For example, grazing by zooplankton, bacteria, and benthic filter feeders has been documented as an important top-down control of phytoplankton biomass in many coastal systems (Strom and Strom 1996; Li and Smayda 1998; Calbet and Landry 2004), with the potential for grazing by microzooplankton, in particular, reported to yield large decreases in phytoplankton standing crop in the Suwannee estuary (Robinson 2007; Quinlan et al. 2009). In the present study, grazing pressure may have varied across sites, as previously demonstrated across zones indentified by differing salinities (Robinson 2007). Temporal variation in grazing pressure also may have been generated by changes in temperature. For instance, Li and Smayda (1998) suggested that warmer temperatures during the winter and spring enhanced zooplankton growth and grazing rates enough to impact phytoplankton blooms in Narragansett Bay, Rhode Island.

In addition to unquantified factors, the use of ordinary least squares (OLS) required transformations to satisfy assumptions for homoscedasticity and normality of residuals, and the method only assessed linear, unidirectional relationships. Water quality data tend to be non-normal and have non-constant variances (Carey 2009); therefore, a log transformation was needed to attain homoscedasticity and normality of residuals. However, this transformation alters the interpretation of relationships identified in the model, which may complicate interpretations for managers attempting to use raw data (Osborne 2002). Furthermore, the inability to model complex, non-linear relationships obviates evaluation of feedback loops, such as increasing competition for nutrients as phytoplanktonic biomass accumulates. Other approaches (e.g., multivariate or hierarchical models) can accommodate such interactions, and they should be considered in future modeling studies.

Summary

The trend and regression analyses presented here provided insight into the dynamic relationships between chlorophyll-a and other water quality parameters in the Suwannee River and its estuary. This information highlights the importance of considering spatial and temporal variability, and the results will be valuable for managers working to address water quality concerns in the lower Suwannee River, its estuary, and other affected areas.

Spatial and temporal variation in relationships between chlorophyll-a and other water quality parameters were highlighted using time series and regression modeling. Spatial variability was linked to river plume dynamics, which generates differences in top-down and bottom-up controls on phytoplankton biomass across the study area. The bottom-up influence of nutrients and light were consistently important as indicated by presence of significant terms for color and concentrations of total phosphorus in all models. The significance of these influences, and other terms related to water residence time, corroborated results from previous studies that noted the influence of the river’s plume. However, the importance of total phosphorus noted here contradicted findings by Bledsoe et al. (2004), which identified nitrogen as the primary limiting nutrient in the estuary. This inconsistency may be due to differences in relative variability in nitrogen and phosphorus loads, but further research will be required to fully understand the relevant dynamics. Other studies also have reported the potential for significant top-down control of phytoplankton biomass that can vary with plume dynamics (Robinson 2007; Quinlan et al. 2009) and previously reported variation in grazing may explain the variable fit of regression models among groups of stations. Overall, the importance of the riverine plume as a regulator of phytoplankton biomass across the study area suggests that managers should contemplate impacts from altered flow, along with nutrient loads and other sources of pollution.

Variations in fit also highlighted the importance of considering climatic cycles. For instance, conclusions based on data collected under only one set of hydrologic conditions may not be applicable in different circumstances, which could lead to misguided management decisions. Furthermore, the importance of timescales was highlighted in the trend analyses, where trends at shorter timescales averaged out over the full study. Such variation becomes important when choosing a baseline for reference or target values for criteria, as well as when assessing the success of management actions (e.g., lower nutrient concentrations).

With these considerations in mind, future work should include a balance between fine-scale studies and large-scale modeling. Fine-scale experimental or descriptive studies could help clarify the causes and magnitude of temporal and spatial variability in processes across the study area. For example, if patterns in phosphorus limitation coincide with agricultural planting seasons, efforts to reduce fertilizer runoff can be stressed. On a larger scale, the modeling approach demonstrated in this study can be applied across the Big Bend region to identify which driving factors remain important at a regional scale. Such information could identify similarities or differences among systems and support customized management initiatives.

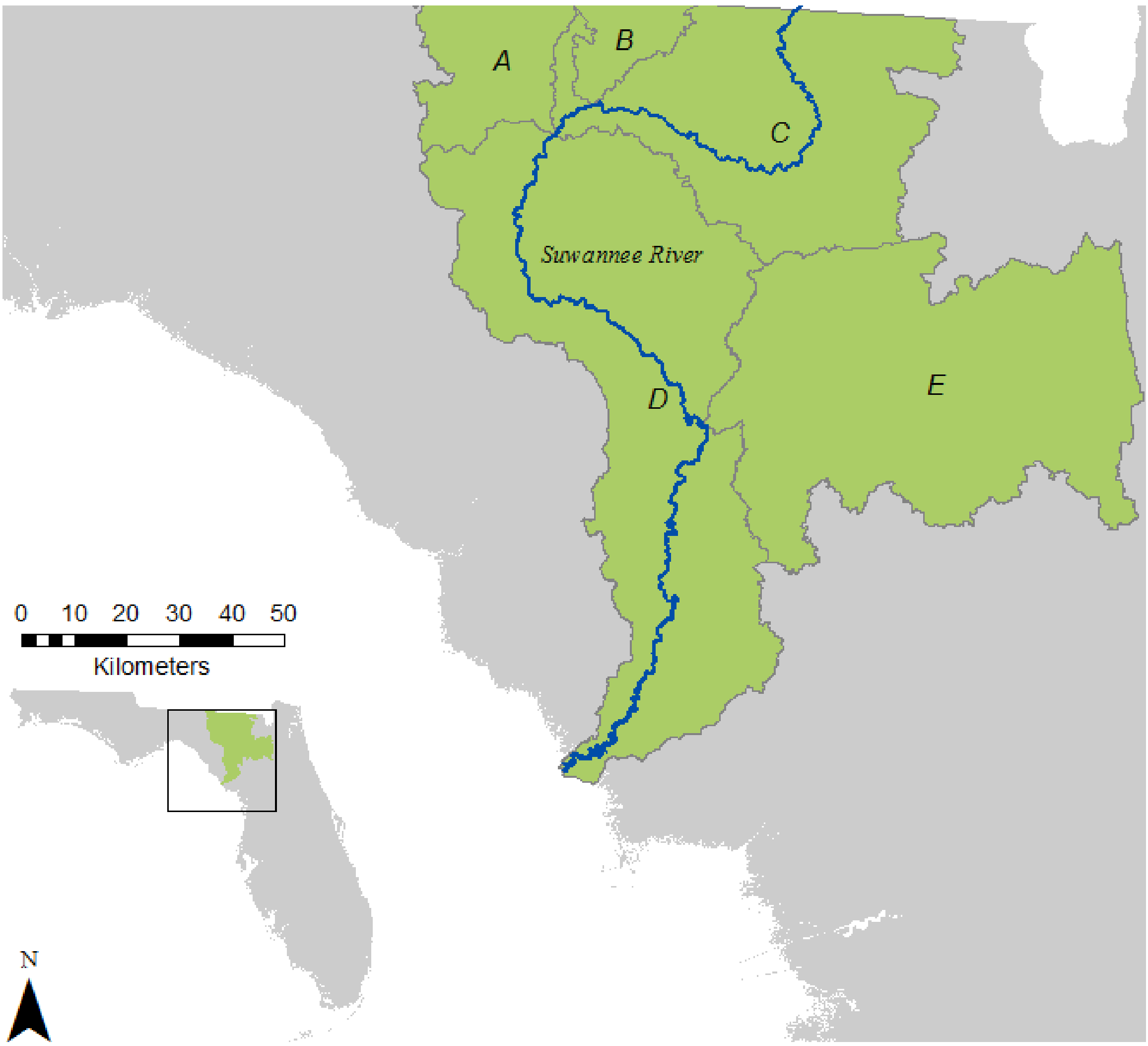


Figure 3-1. Suwannee River basin with the A) Withlacoochee sub-basin, B) Alapaha River sub-basin, C) Upper Suwannee River sub-basin, D) Suwannee River sub-basin, and E) Santa Fe River sub-basin illustrated. Data source: Suwannee River Water Management District (http://www.srwmd.state.fl.us/index.aspx?NID=319).

Table 3-1. Supplementary data sources. (NOAA: National Oceanographic and Atmospheric Association)

Parameter	Data source	Site ID	Site name
Mean daily discharge	United States Geological Survey	2323592 2323500	Above Gopher River Wilcox
Wind Direction	NOAA National Data Buoy Center	KTNF1	Keaton Beach
Water Color	Suwannee River Water Management District	SUW275C1 SUW305C1	At Gopher River West Pass

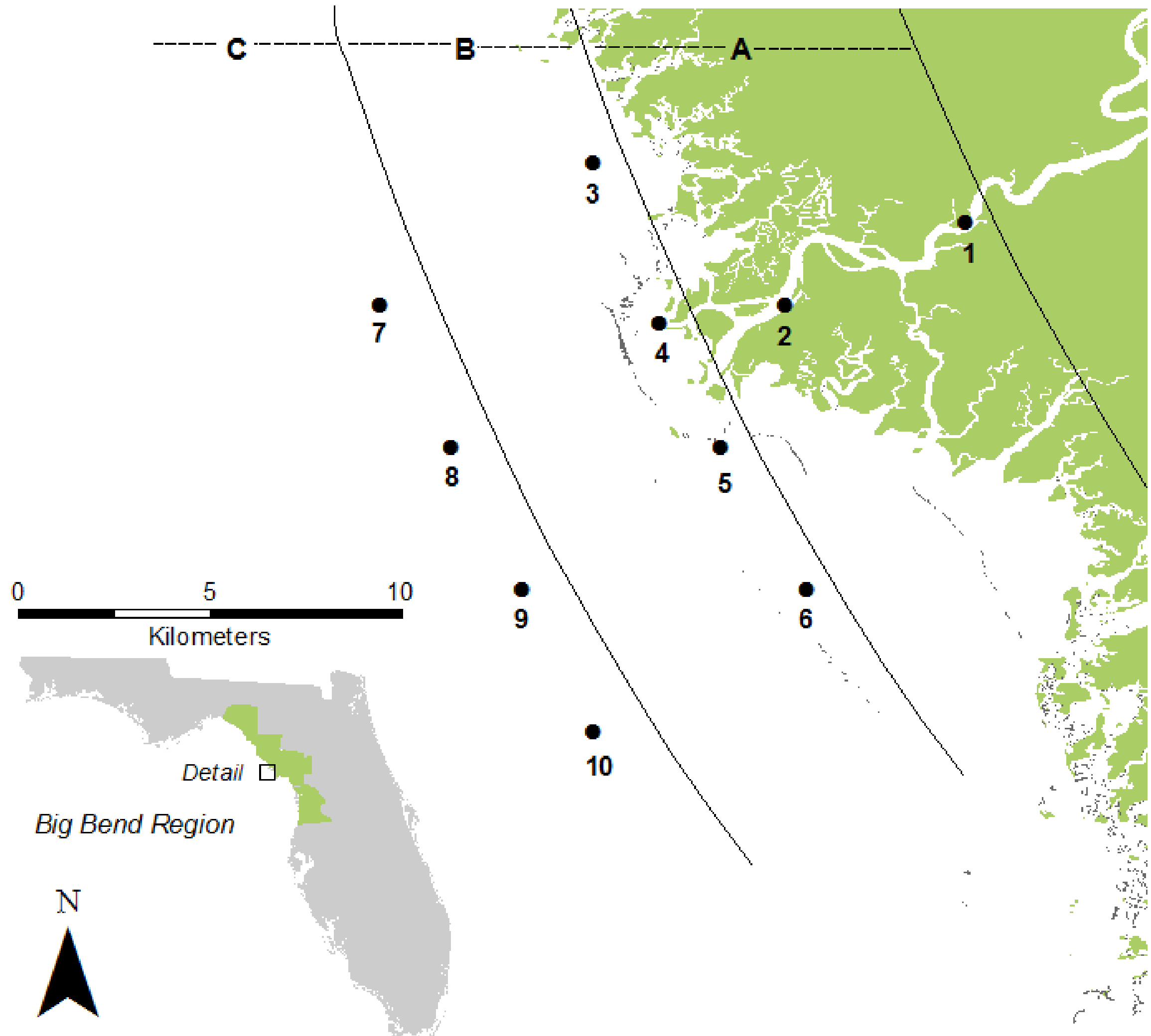


Figure 3-2. Water quality sampling sites with the A) river, B) reef, and C) nearshore regions illustrated (Quinlan 2003; Raabe et al. 2007). Oyster reefs are outlined in gray.

Table 3-2. Details and results of trend analyses for total nitrogen by station.

Station	Trend Test	*S*	Kendall's tau	*P*
1	Adjusted Seasonal Kendall	254	0.271	0.038
2	Mann Kendall with bootstrap	36	0.003	0.956
3	Adjusted Seasonal Kendall	44	0.047	0.698
4	Mann Kendall with bootstrap	39	0.003	0.953
5	Seasonal Kendall	151	0.161	0.008
6	Mann Kendall with bootstrap	6	0.000	0.993
7	Adjusted Seasonal Kendall	171	0.183	0.118
8	Mann Kendall with bootstrap	82	0.007	0.900
9	Mann Kendall with bootstrap	96	0.008	0.882
10	Mann Kendall with bootstrap	-20	-0.002	0.975

Table 3-3. Details and results of trend analyses for total phosphorus by station.

Station	Trend Test	*S*	Kendall's tau	*P*
1	Mann Kendall with bootstrap	-41	-0.003	0.950
2	Mann Kendall with bootstrap	19	0.002	0.976
3	Adjusted Seasonal Kendall	2	0.002	0.993
4	Adjusted Seasonal Kendall	88	0.094	0.354
5	Adjusted Seasonal Kendall	74	0.079	0.451
6	Adjusted Seasonal Kendall	91	0.097	0.370
7	Adjusted Seasonal Kendall	57	0.061	0.626
8	Adjusted Seasonal Kendall	92	0.098	0.447
9	Adjusted Seasonal Kendall	196	0.209	0.075
10	Adjusted Seasonal Kendall	134	0.143	0.135

Table 3-4. Details and results of trend analyses for chlorophyll-a by station.

Station	Trend Test	*S*	Kendall's tau	*P*
1	Seasonal Kendall	38	0.042	0.504
2	Seasonal Kendall	81	0.089	0.151
3	Seasonal Kendall	-92	-0.100	0.106
4	Seasonal Kendall	14	0.015	0.817
5	Adjusted Seasonal Kendall	69	0.076	0.204
6	Adjusted Seasonal Kendall	107	0.117	0.148
7	Adjusted Seasonal Kendall	59	0.064	0.581
8	Mann Kendall with bootstrap	-66	-0.006	0.918
9	Mann Kendall	660	0.055	0.308
10	Mann Kendall	637	0.054	0.320

Table 3-5. Details and results of trend analyses for color by station.

Station	Trend Test	*S*	Kendall's tau	*P*
1	Adjusted Seasonal Kendall	100	0.107	0.441
2	Adjusted Seasonal Kendall	31	0.033	0.821
3	Adjusted Seasonal Kendall	12	0.013	0.934
4	Adjusted Seasonal Kendall	38	0.041	0.786
5	Adjusted Seasonal Kendall	13	0.014	0.929
6	Mann Kendall with bootstrap	13	0.001	0.985
7	Mann Kendall with bootstrap	85	0.007	0.897
8	Mann Kendall with bootstrap	-78	-0.007	0.905
9	Mann Kendall with bootstrap	-58	-0.005	0.929
10	Mann Kendall with bootstrap	-115	-0.010	0.860

Table 3-6. Details and results of trend analyses for salinity by station.

Station	Trend Test	*S*	Kendall's tau	*P*
1	Adjusted Seasonal Kendall	-38	-0.041	0.735
2	Adjusted Seasonal Kendall	37	0.040	0.758
3	Adjusted Seasonal Kendall	-59	-0.063	0.628
4	Adjusted Seasonal Kendall	20	0.021	0.863
5	Adjusted Seasonal Kendall	-31	-0.033	0.737
6	Adjusted Seasonal Kendall	-67	-0.072	0.537
7	Adjusted Seasonal Kendall	-111	-0.119	0.343
8	Mann Kendall with bootstrap	-3	0.000	0.996
9	Mann Kendall with bootstrap	-10	-0.001	0.988
10	Mann Kendall with bootstrap	-9	-0.001	0.989

Table 3-7. Analysis of variance for model used to estimate missing color values with log_{10} color as the response variable. Asterisks indicate a dummy variable.

	SSE	df	*F*	*P*
Intercept	15.46	1	284.86	0.000
Station*	78.19	9	160.03	0.000
Hi flow year*	0.53	1	9.75	0.002
50^{th} percentile discharge*	0.89	1	16.37	0.000
One day lag: log_{10} discharge	4.73	1	87.17	0.000
Two day lag: log_{10} discharge	4.06	1	74.77	0.000
Residuals	68.35	1259		

Table 3-8. Analysis of variance for model used to estimate missing salinity values with log_{10} salinity as the response variable. Asterisks indicate a dummy variable.

	SSE	df	*F*	*P*
Intercept	4.35	1	88.29	0.000
Station*	20.64	9	46.56	0.000
Log discharge	0.39	1	7.85	0.005
One month lag: log_{10} salinity	3.53	1	71.68	0.000
Two month lag: log_{10} salinity	1.36	1	27.54	0.000
Two day lag: log_{10} discharge	0.74	1	14.99	0.000
One month lag: log_{10} discharge	0.37	1	7.41	0.007
95^{th} percentile discharge*	0.90	1	18.29	0.000
Residuals	68.25	1386		

Table 3-9. Analysis of variance for model used to estimate missing total nitrogen values with $\log_{10}$ total nitrogen as the response variable. Asterisks indicate a dummy variable.

	SSE	df	*F*	*P*
Intercept	4.48	1	272.84	0.000
Station*	9.89	9	66.89	0.000
Month*	2.10	11	11.62	0.000
High discharge year*	0.21	1	12.57	0.000
One month lag: $\log_{10}$ total nitrogen	1.19	1	72.26	0.000
Two month lag: $\log_{10}$ discharge	0.11	1	6.95	0.008
One day lag: $\log_{10}$ discharge	0.70	1	42.85	0.000
95^{th} percentile discharge*	0.11	1	6.74	0.010
90^{th} percentile discharge*	0.18	1	10.81	0.001
Residuals	22.14	1348		

Table 3-10. Analysis of variance for model used to estimate missing total phosphorus values with $\log_{10}$ total phosphorus as the response variable. Asterisks indicate a dummy variable.

	SSE	df	*F*	*P*
Intercept	0.70	1	22.01	0.000
High discharge year*	0.17	1	5.27	0.022
One month lag: $\log_{10}$ total phosphorus	58.52	1	1838.71	0.000
One month lag: $\log_{10}$ discharge	3.29	1	103.42	0.000
One day lag: $\log_{10}$ discharge	1.82	1	57.31	0.000
90^{th} percentile discharge*	0.44	1	13.92	0.000
Station*	0.14	2	2.18	0.114
Month*	3.02	11	8.61	0.000
Residuals	43.48	1366		

Table 3-11. Analysis of variance for model used to estimate missing mean daily discharge values for the Gopher River dataset. Asterisks indicate a dummy variable.

	SSE	df	*F*	*P*
Intercept	2.28E+08	1	142.31	0.000
Wilcox discharge	1.52E+10	1	9447.39	0.000
Squared Wilcox discharge	2.32E+07	1	14.48	0.000
Residuals	6.17E+09	3841		

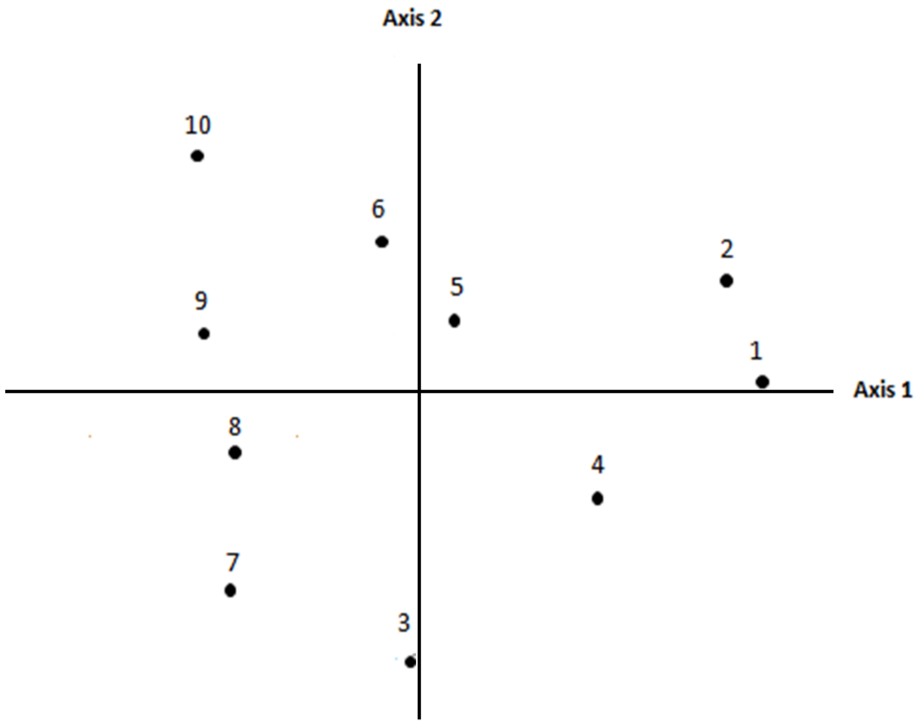


Figure 3-3. Two-dimensional ordination of sampling stations (stress value = 0.01) based on range-transformed values of temperature, salinity, color, and concentrations of total nitrogen and total phosphorus. Short distances between points indicate similarity, with points farther apart being less similar.

Table 3-12. Analysis of variance for group model for stations 1 and 2 with $\log_{10}$ chlorophyll-a as the response variable. Asterisks indicate dummy variables.

	SSE	df	*F*	*P*
Intercept	1.31	1	36.70	0.000
Month*	4.94	11	12.59	0.000
Log_{10} color	1.62	1	45.40	0.000
One month lag: $\log_{10}$ total phosphorus	0.29	1	8.07	0.005
Log_{10} (salinity X total phosphorus)	0.29	1	8.00	0.005
Log_{10} salinity	0.60	1	16.75	0.000
Log_{10} total phosphorus	0.30	1	8.52	0.004
One month lag: $\log_{10}$ discharge	0.86	1	23.96	0.000
99^{th} percentile chlorophyll*	1.38	1	38.63	0.000
November 2007*	0.22	1	6.05	0.015
Residuals	7.75	217		

Table 3-13. Coefficient estimates for group model for stations 1 and 2. Asterisks indicate dummy variables.

	Coefficient	S.E.	*t*	*P*
Intercept	2.06	0.38	5.40	0.000
February*	0.06	0.06	1.06	0.290
March*	0.12	0.06	1.92	0.057
April*	0.31	0.06	5.11	0.000
May*	0.51	0.06	7.99	0.000
June*	0.30	0.06	4.82	0.000
July*	0.26	0.06	4.28	0.000
August*	0.25	0.06	4.04	0.000
September*	0.24	0.06	3.79	0.000
October*	0.26	0.06	4.20	0.000
November*	0.04	0.06	0.67	0.502
December*	-0.04	0.06	-0.58	0.561
Log_{10} color	-0.37	0.05	-6.74	0.000
One month lag: log_{10} total phosphorus	-0.36	0.13	-2.84	0.005
Log_{10} (salinity X total phosphorus)	-0.26	0.09	-2.83	0.005
Log_{10} salinity	0.60	0.15	4.09	0.000
Log_{10} total phosphorus	0.47	0.16	2.92	0.004
One month lag: log_{10} discharge	-0.30	0.06	-4.90	0.000
99th percentile chlorophyll*	0.72	0.12	6.22	0.000
November 2007*	0.37	0.15	2.46	0.015

Table 3-14. Results of Box-tests for autocorrelation for station residuals extracted from group models.

Station	X^2	df	P
1	12.59	1	0.000
2	3.97	1	0.046
3	2.63	1	0.103
4	8.70	1	0.003
5	4.32	1	0.038
6	7.56	1	0.006
7	5.49	1	0.019
8	6.76	1	0.009
9	6.52	1	0.011
10	0.23	1	0.632

Table 3-15. Analysis of variance for station 4 model.

	SSE	df	*F*	*P*
Intercept	0.56	1	8.53	0.004
Log_{10} total phosphorus	0.38	1	5.83	0.017
Log_{10} color	0.53	1	8.08	0.005
Log_{10} salinity	0.83	1	12.63	0.001
Log_{10} temperature	0.68	1	10.31	0.002
Residuals	7.49	111		

Table 3-16. Coefficient estimates for station 4 model.

	Coefficient	S.E.	*t*	*P*
Intercept	-1.09	0.37	-2.92	0.004
Log_{10} total phosphorus	0.51	0.21	2.42	0.017
Log_{10} color	-0.26	0.09	-2.84	0.005
Log_{10} salinity	0.24	0.07	3.55	0.001
Log_{10} temperature	0.72	0.22	3.21	0.002

Table 3-17. Analysis of variance for group model for stations 3, 5, and 6. Asterisks indicate dummy variables.

	SSE	df	*F*	*P*
Intercept	4.10	1	48.83	0.000
Log_{10} salinity	1.44	1	17.16	0.000
Log_{10} total phosphorus	3.14	1	37.33	0.000
Log_{10} color	0.83	1	9.88	0.002
Log_{10} temperature	4.24	1	50.43	0.000
Southwest 36-h wind event*	0.38	1	4.55	0.034
Highest 50% discharge*	1.38	1	16.39	0.000
Residuals	29.49	351		

Table 3-18. Coefficient estimates for group model for stations 3, 5, and 6. Asterisks indicate dummy variables.

	Coefficient	S.E.	*t*	*P*
Intercept	-1.70	0.24	-6.99	0.000
Log_{10} salinity	0.35	0.08	4.14	0.000
Log_{10} total phosphorus	0.64	0.10	6.11	0.000
Log_{10} color	-0.24	0.08	-3.14	0.002
Log_{10} temperature	0.99	0.14	7.10	0.000
Southwest 36-h wind event*	0.17	0.08	2.13	0.034
Highest 50% discharge*	0.17	0.04	4.05	0.000

Table 3-19. Analysis of variance for group model for stations 7, 8, 9, and 10. Asterisks indicate dummy variables.

	SSE	df	*F*	*P*
Intercept	0.85	1	18.86	0.000
Log_{10} total phosphorus	3.89	1	86.34	0.000
Highest 50% discharge*	1.58	1	35.05	0.000
Southwest 36-h wind event*	0.40	1	8.85	0.003
Log_{10} discharge	0.88	1	0.88	0.000
Log_{10} color	0.23	1	5.21	0.023
October 2004*	2.42	1	53.65	0.000
Log_{10} salinity	0.59	1	13.21	0.000
Month*	2.18	11	4.39	0.000
Storm event*	0.60	1	13.44	0.000
Residuals	20.53	456		

Table 3-20. Coefficient estimates for group model for stations 7, 8, 9, and 10. Asterisks indicate dummy variables.

	Coefficient	S.E.	*t*	*P*
Intercept	1.86	0.43	4.34	0.000
Log_{10} total phosphorus	0.71	0.08	9.29	0.000
Highest 50% discharge*	0.19	0.03	5.92	0.000
Southwest 36-h wind event*	0.15	0.05	2.98	0.003
Log_{10} discharge	-0.34	0.08	-4.43	0.000
Log_{10} color	0.14	0.06	2.28	0.023
October 2004*	-1.02	0.14	-7.33	0.000
Log_{10} salinity	-0.79	0.22	-3.63	0.000
February*	-0.18	0.05	-3.73	0.000
March*	-0.06	0.05	-1.28	0.203
April*	-0.08	0.05	-1.57	0.117
May*	-0.13	0.05	-2.44	0.015
June*	0.04	0.05	0.80	0.427
July*	0.00	0.05	-0.09	0.930
August*	0.05	0.05	1.03	0.306
September*	-0.02	0.05	-0.39	0.698
October*	0.04	0.05	0.69	0.489
November*	0.05	0.05	0.94	0.350
December*	0.03	0.05	0.57	0.569
Storm event*	-0.22	0.06	-3.67	0.000

Table 3-21. Root mean squared error (RMSE) values for individual station models calculated for each year. Values in bold are RMSE calculated after the appropriate time series adjustment was applied. Dashes indicate no time series adjustments were applied.

Station	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	1998-2010
1	0.26	0.23	0.17	0.15	0.16	0.16	0.15	0.12	0.18	0.18	0.20	0.20	0.25	0.17
	0.26	**0.21**	**0.19**	**0.15**	**0.14**	**0.13**	**0.15**	**0.11**	**0.18**	**0.20**	**0.20**	**0.17**	**0.24**	**0.17**
2	0.32	0.23	0.20	0.14	0.18	0.16	0.12	0.12	0.17	0.19	0.20	0.20	0.24	0.17
	0.31	**0.22**	**0.19**	**0.18**	**0.16**	**0.15**	**0.11**	**0.12**	**0.17**	**0.16**	**0.20**	**0.20**	**0.24**	**0.17**
3	0.34	0.21	0.26	0.22	0.23	0.26	0.32	0.29	0.32	0.42	0.36	0.39	0.17	0.29
	-	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**
4	0.28	0.23	0.29	0.29	0.22	0.24	0.25	0.30	0.26	0.24	0.21	0.17	0.26	0.25
	0.30	**0.21**	**0.27**	**0.28**	**0.20**	**0.23**	**0.28**	**0.26**	**0.27**	**0.23**	**0.20**	**0.19**	**0.24**	**0.24**
5	0.33	0.32	0.23	0.23	0.22	0.30	0.32	0.33	0.30	0.16	0.35	0.33	0.34	0.28
	0.35	**0.31**	**0.24**	**0.22**	**0.23**	**0.31**	**0.31**	**0.31**	**0.29**	**0.17**	**0.35**	**0.35**	**0.37**	**0.27**
6	0.32	0.32	0.24	0.24	0.27	0.29	0.35	0.35	0.30	0.18	0.19	0.31	0.26	0.27
	0.32	**0.30**	**0.26**	**0.24**	**0.28**	**0.29**	**0.33**	**0.32**	**0.30**	**0.21**	**0.19**	**0.36**	**0.26**	**0.27**
7	0.39	0.18	0.13	0.10	0.14	0.20	0.15	0.32	0.27	0.14	0.19	0.18	0.23	0.18
	0.39	**0.16**	**0.13**	**0.10**	**0.15**	**0.21**	**0.16**	**0.33**	**0.25**	**0.14**	**0.19**	**0.19**	**0.21**	**0.18**
8	0.35	0.29	0.13	0.13	0.14	0.33	0.18	0.23	0.26	0.16	0.14	0.28	0.28	0.20
	0.34	**0.24**	**0.13**	**0.16**	**0.14**	**0.33**	**0.21**	**0.22**	**0.22**	**0.17**	**0.16**	**0.30**	**0.25**	**0.20**
9	0.38	0.23	0.14	0.15	0.13	0.19	0.24	0.18	0.28	0.18	0.13	0.22	0.23	0.18
	0.38	**0.20**	**0.14**	**0.15**	**0.14**	**0.18**	**0.25**	**0.18**	**0.27**	**0.19**	**0.13**	**0.23**	**0.23**	**0.18**
10	0.27	0.49	0.22	0.33	0.32	0.22	0.13	0.20	0.23	0.16	0.21	0.22	0.26	0.25
	-	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**	**-**

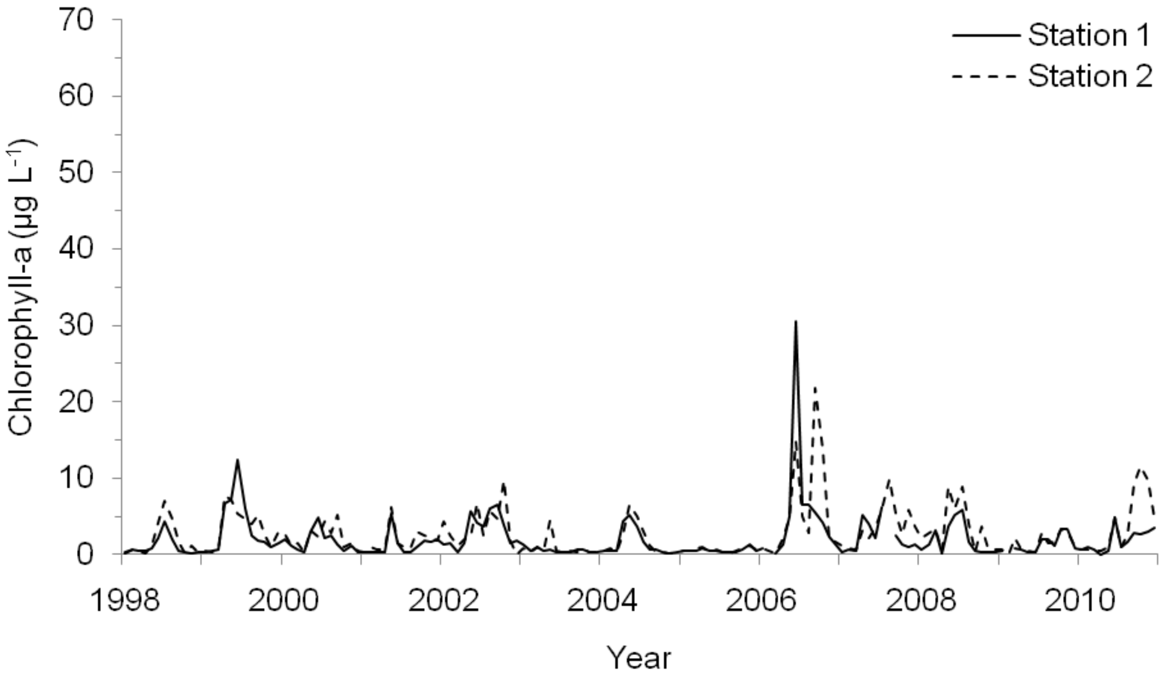


Figure 3-4. Time series plot of chlorophyll-a concentrations (µg L^{-1}) for stations 1 and 2.

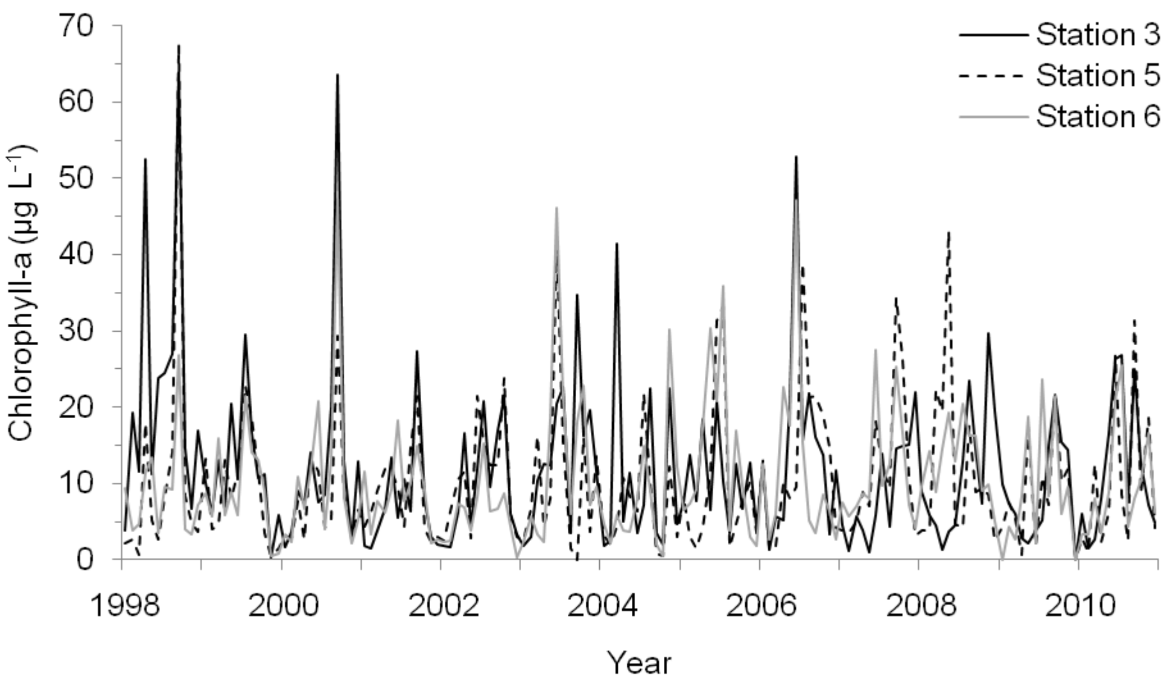


Figure 3-5. Time series plot of chlorophyll-a concentrations (µg L^{-1}) for stations 3, 5, and 6.

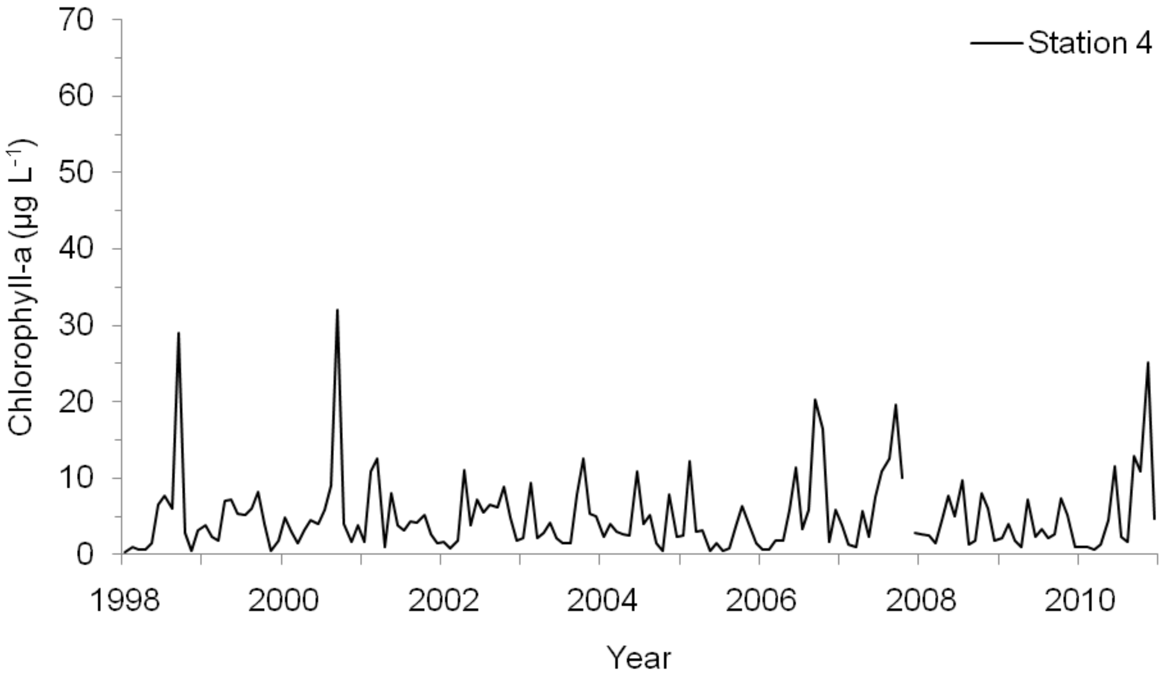


Figure 3-6. Time series plot of chlorophyll-a concentrations (µg L^{-1}) for station 4.

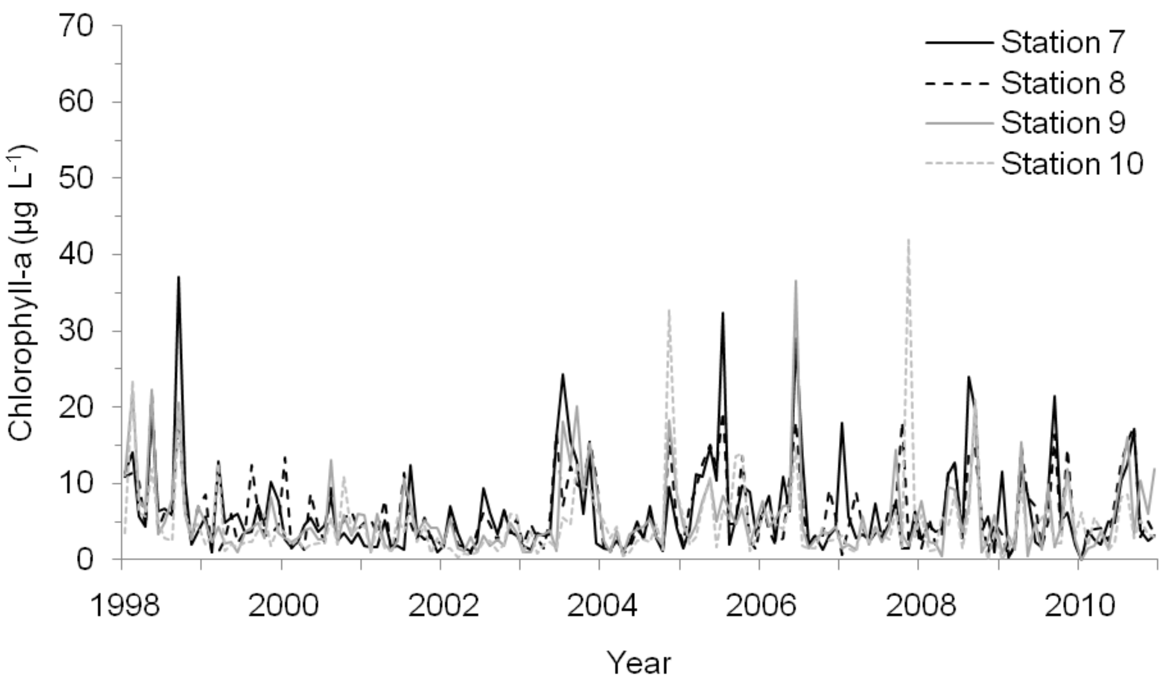


Figure 3-7. Time series plot of chlorophyll-a concentrations (µg L^{-1}) for stations 7, 8, 9, and 10.

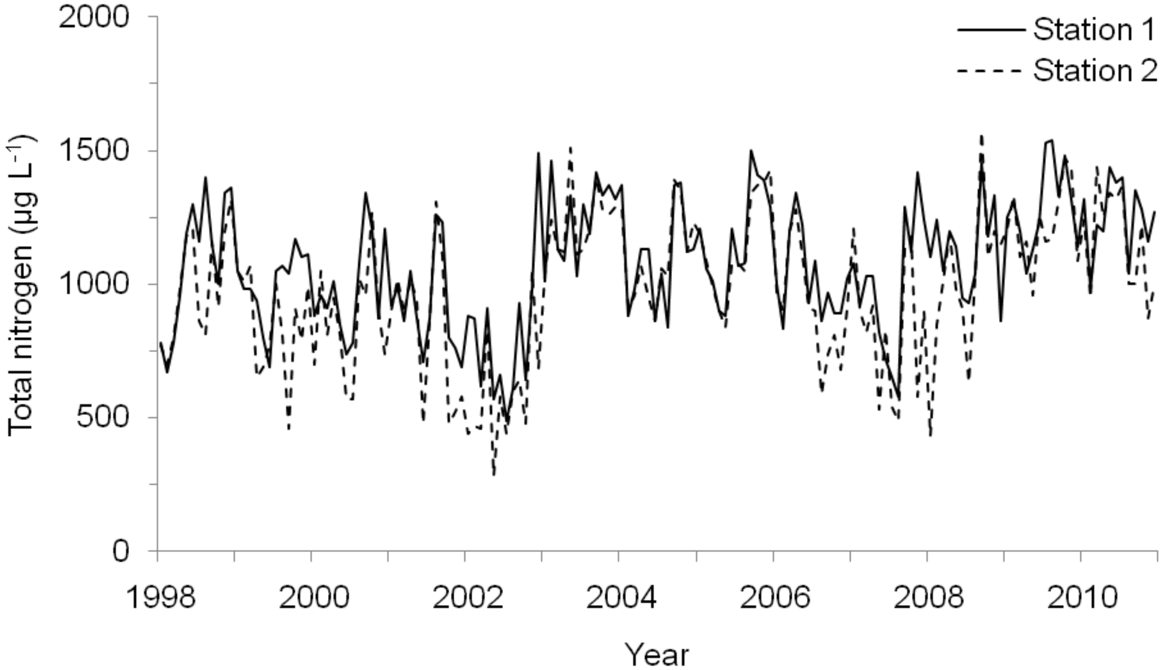


Figure 3-8. Time series plot of total nitrogen concentrations (µg L^{-1}) for stations 1 and 2.

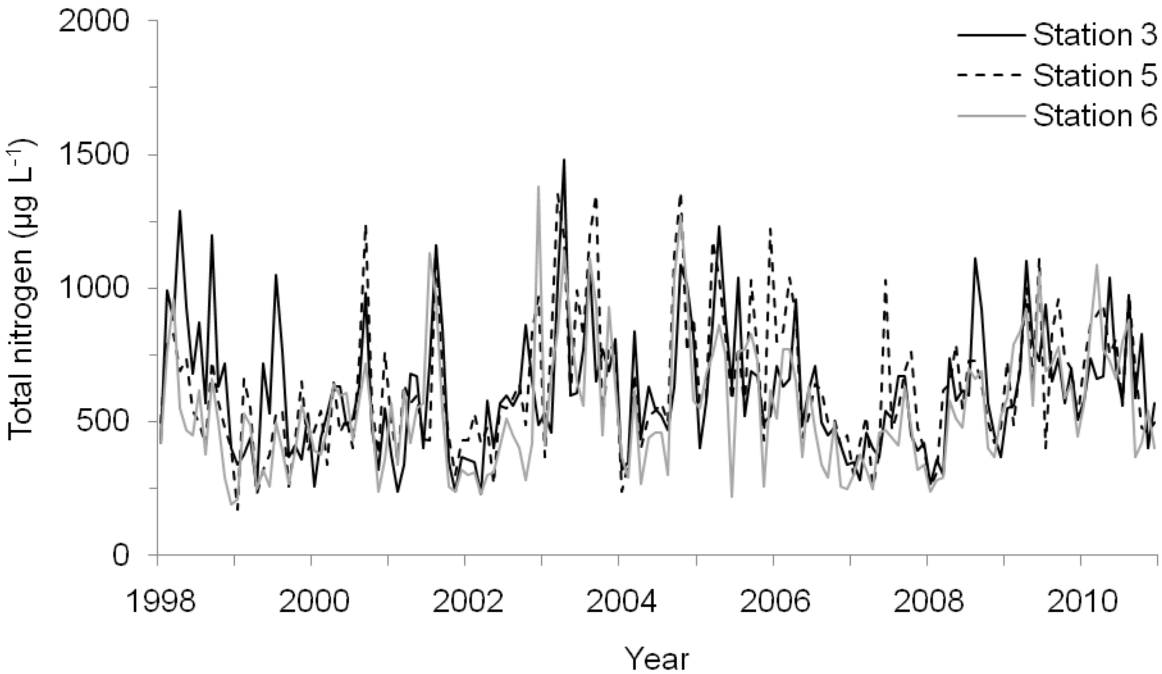


Figure 3-9. Time series plot of total nitrogen concentrations (µg L^{-1}) for stations 3, 5, and 6.

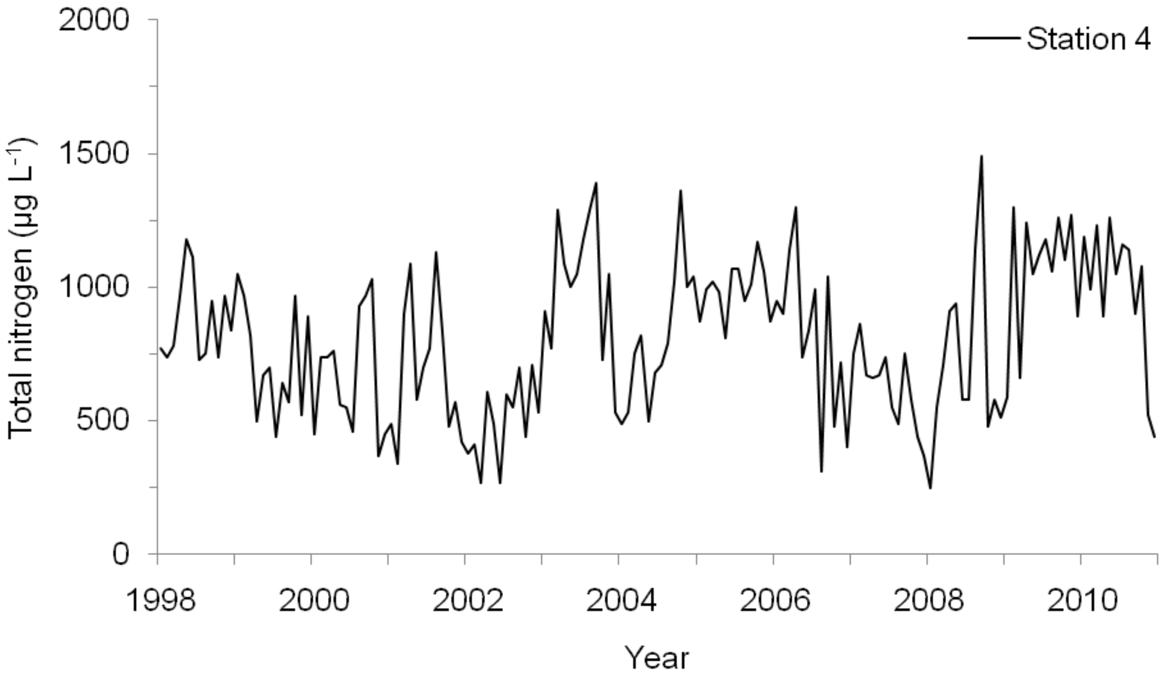


Figure 3-10. Time series plot of total nitrogen concentrations (µg L^{-1}) for station 4.

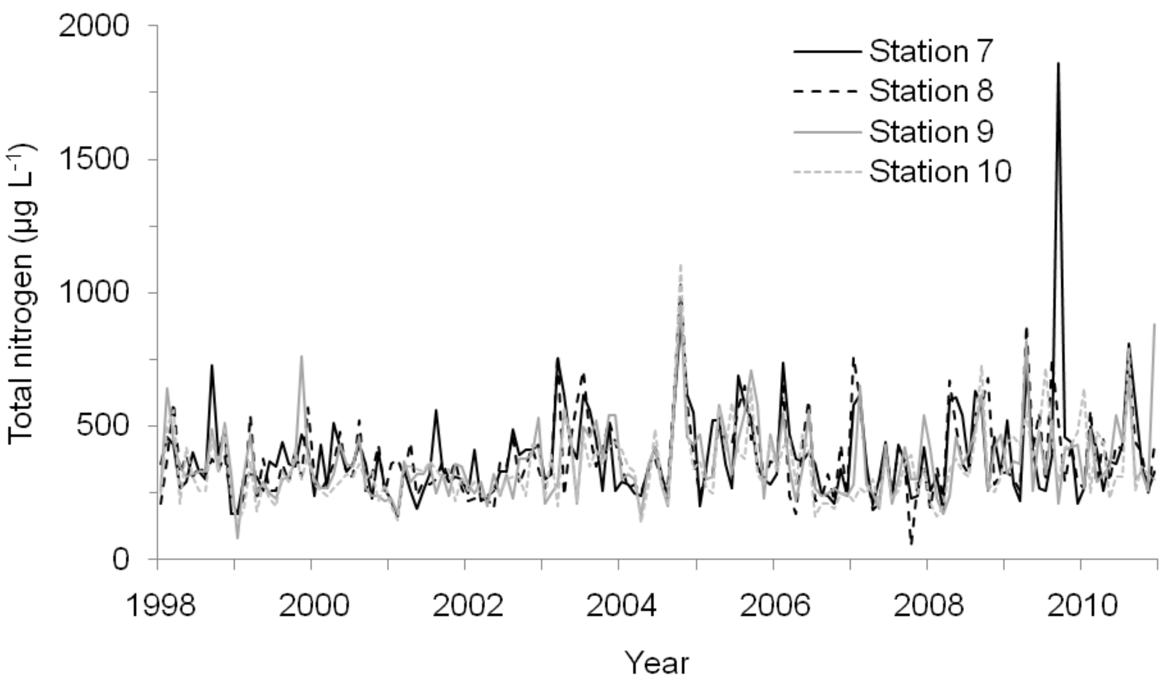


Figure 3-11. Time series plot of total nitrogen concentrations (µg L^{-1}) for stations 7, 8, 9, and 10.

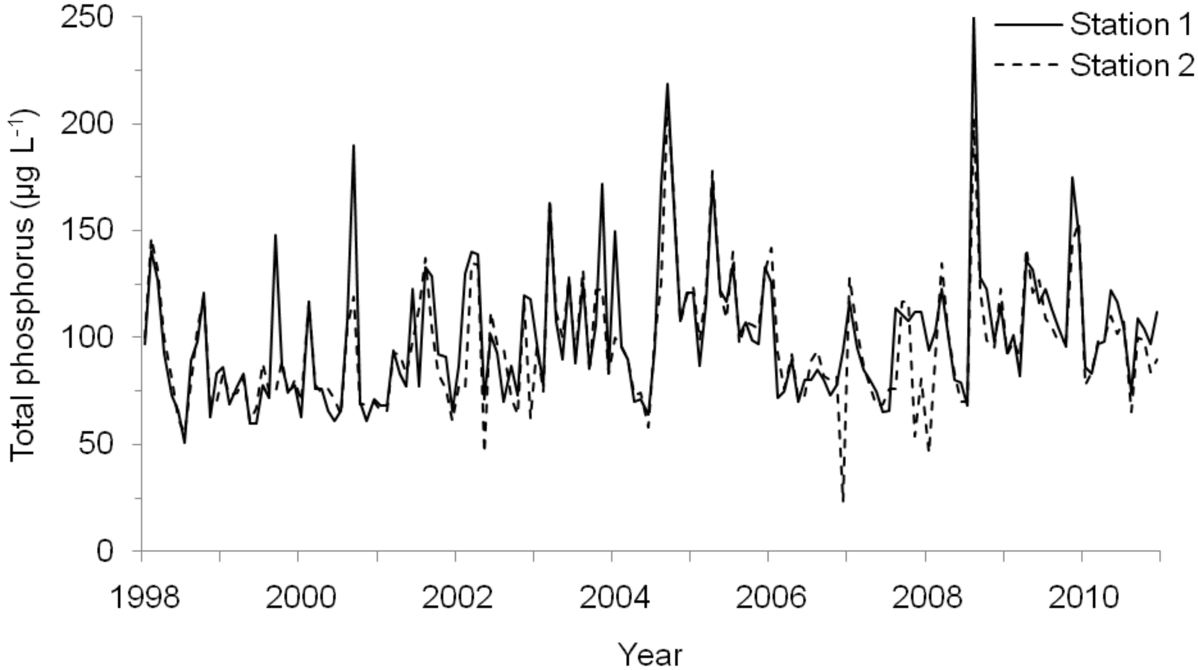


Figure 3-12. Time series plot of total phosphorus concentrations (µg L^{-1}) for stations 1 and 2.

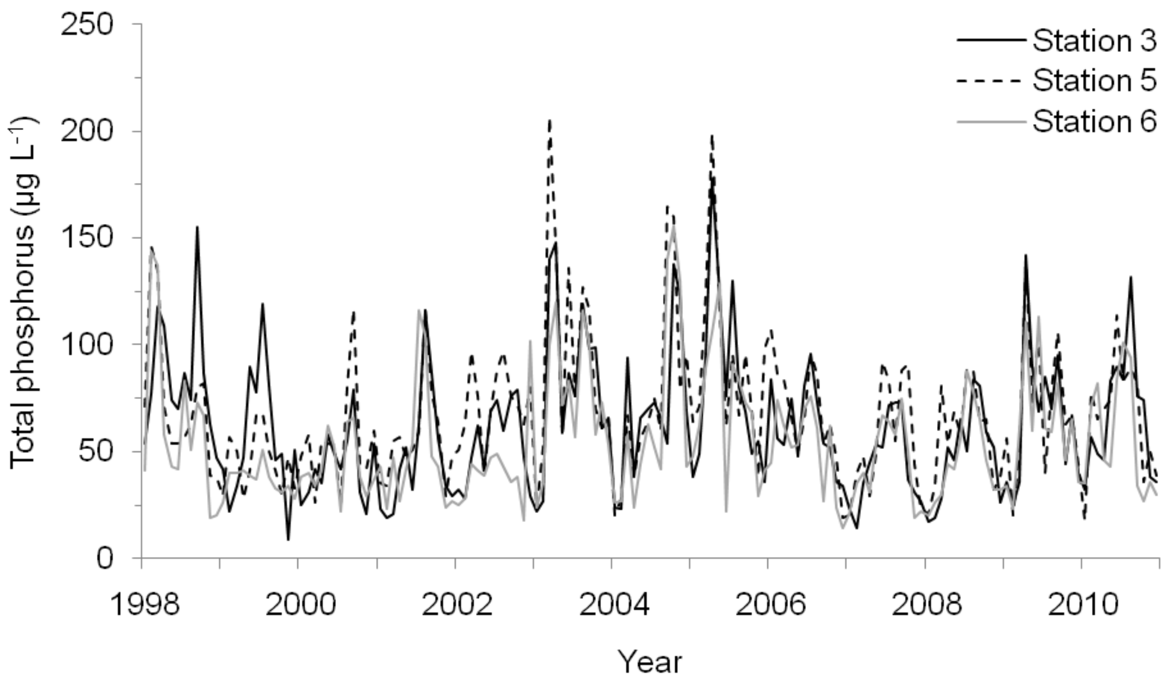


Figure 3-13. Time series plot of total phosphorus concentrations (µg L^{-1}) for stations 3, 5, and 6.

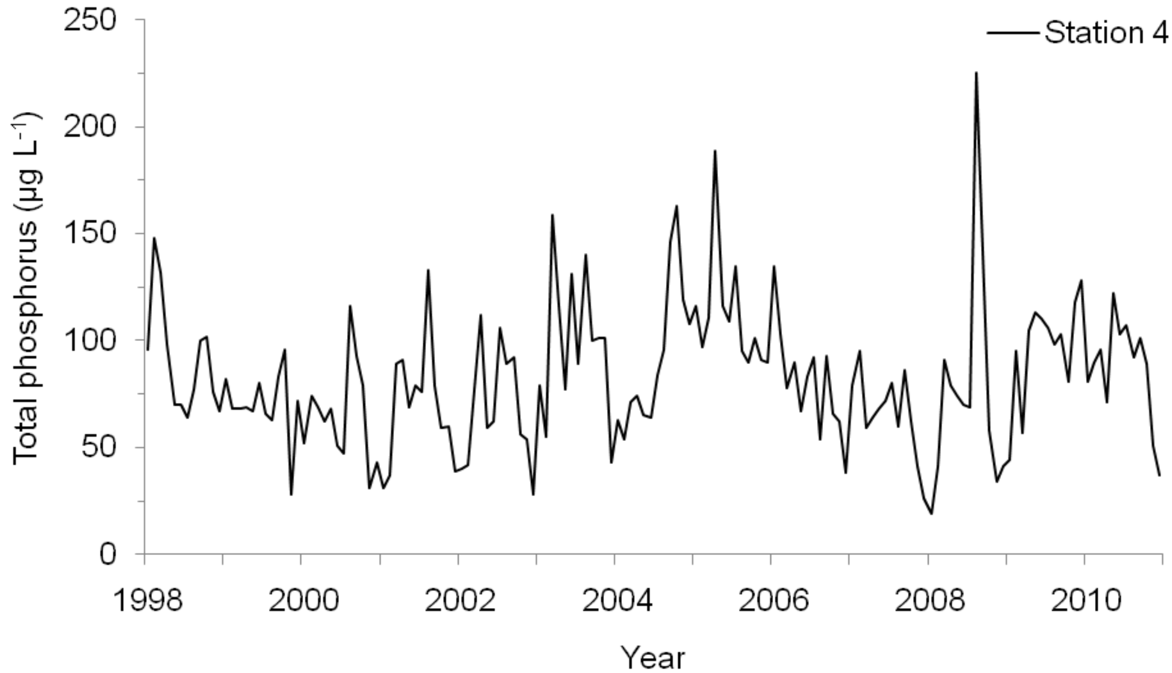


Figure 3-14. Time series plot of total phosphorus concentrations (µg L^{-1}) for station 4.

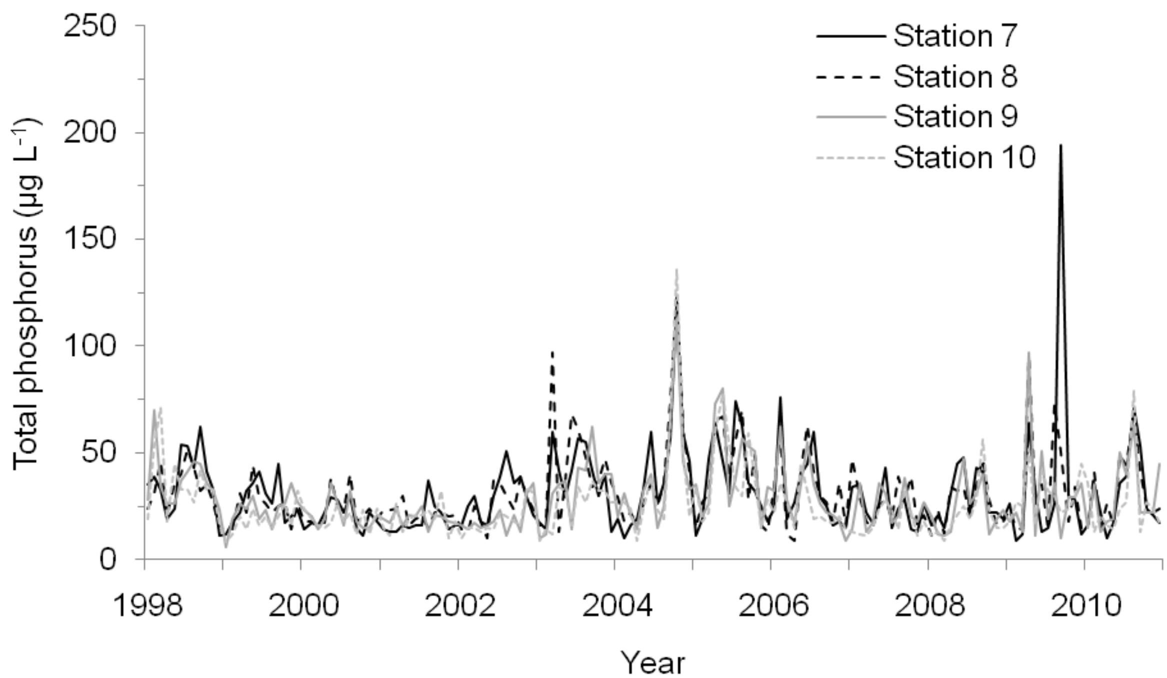


Figure 3-15. Time series plot of total phosphorus concentrations (µg L^{-1}) for stations 7, 8, 9, and 10.

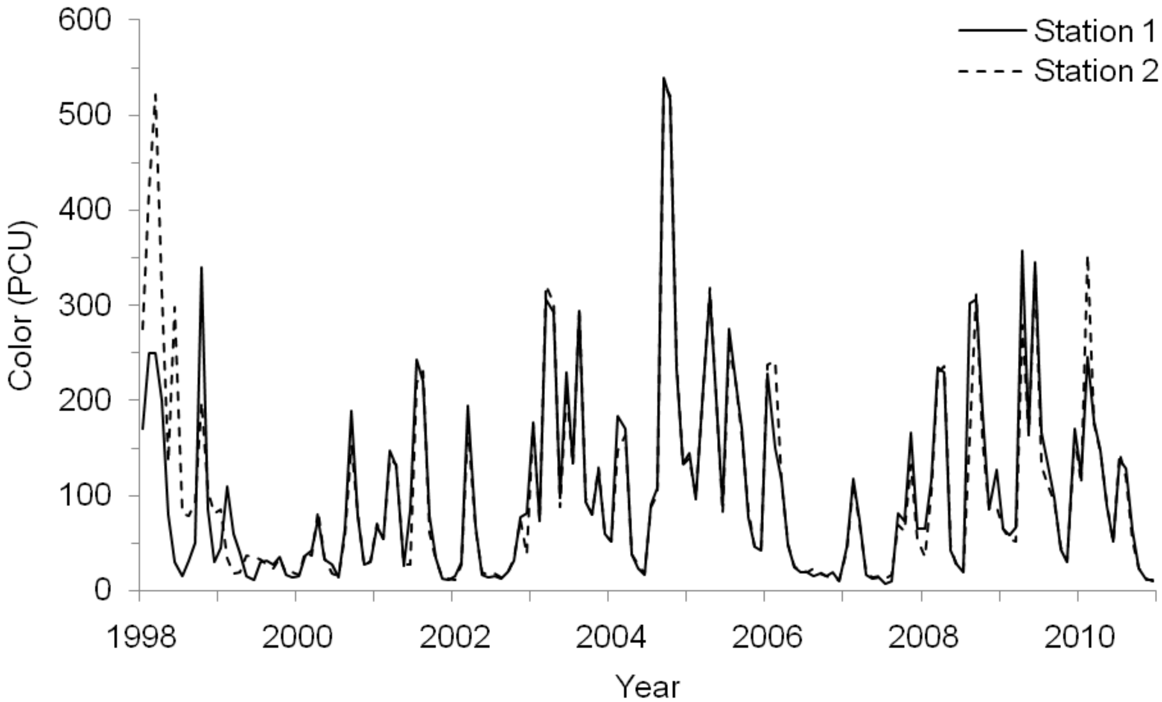


Figure 3-16. Time series plot of color (PCU) for stations 1 and 2.

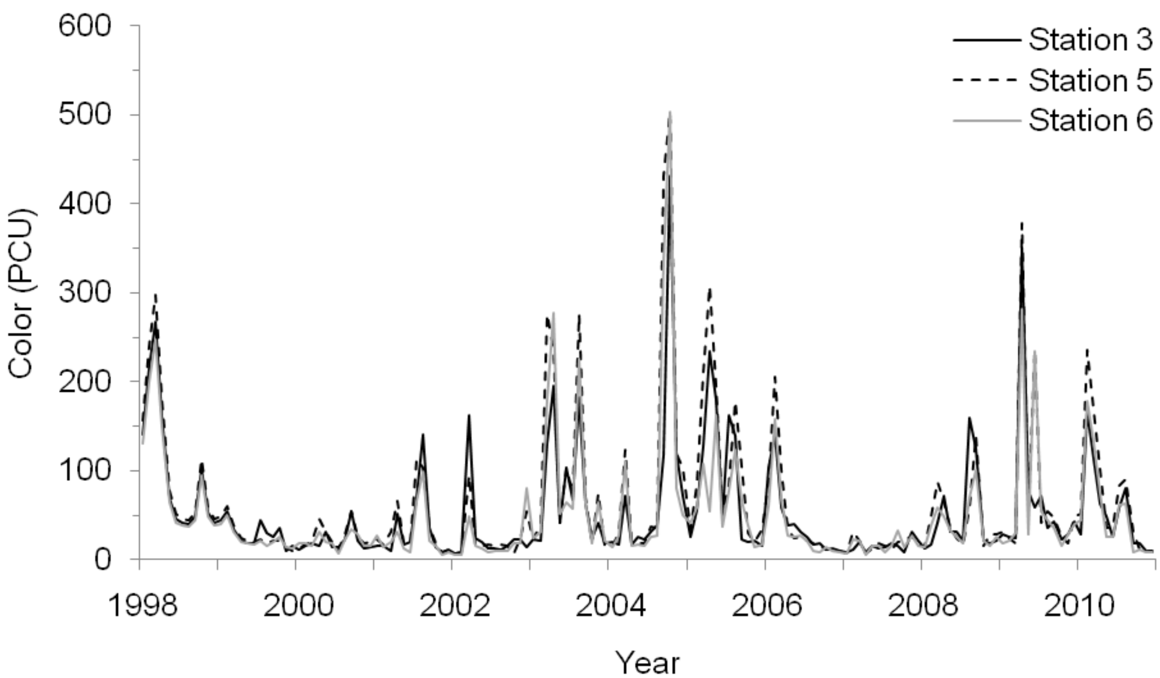


Figure 3-17. Time series plot of color (PCU) for stations 3, 5, and 6.

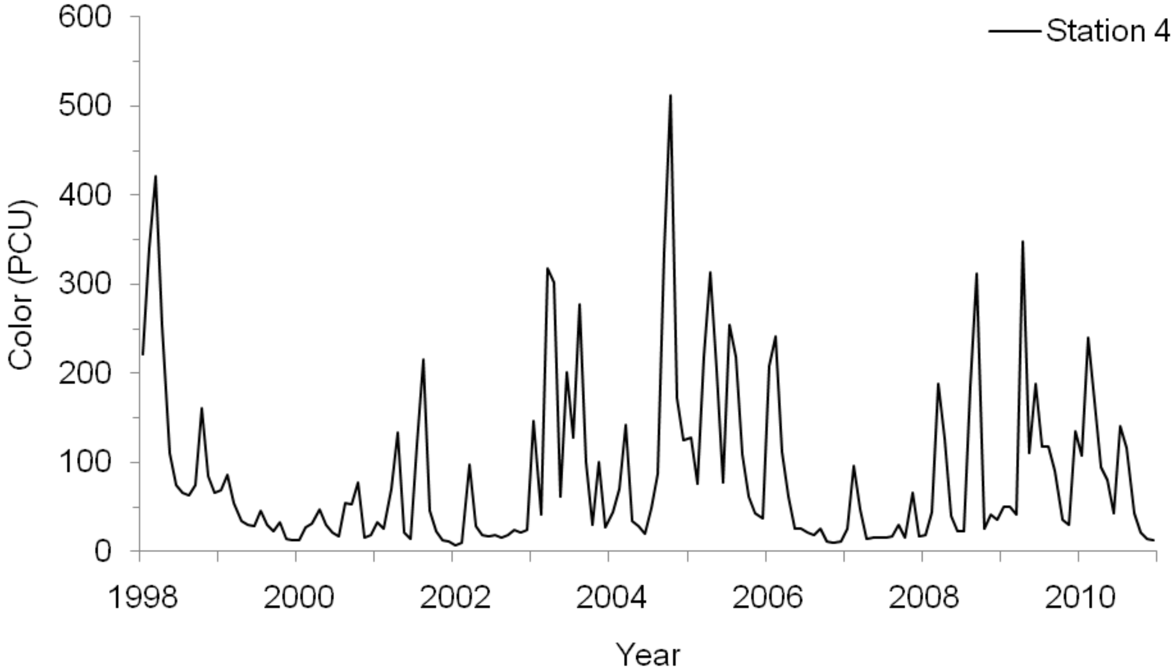


Figure 3-18. Time series plot of color (PCU) for station 4.

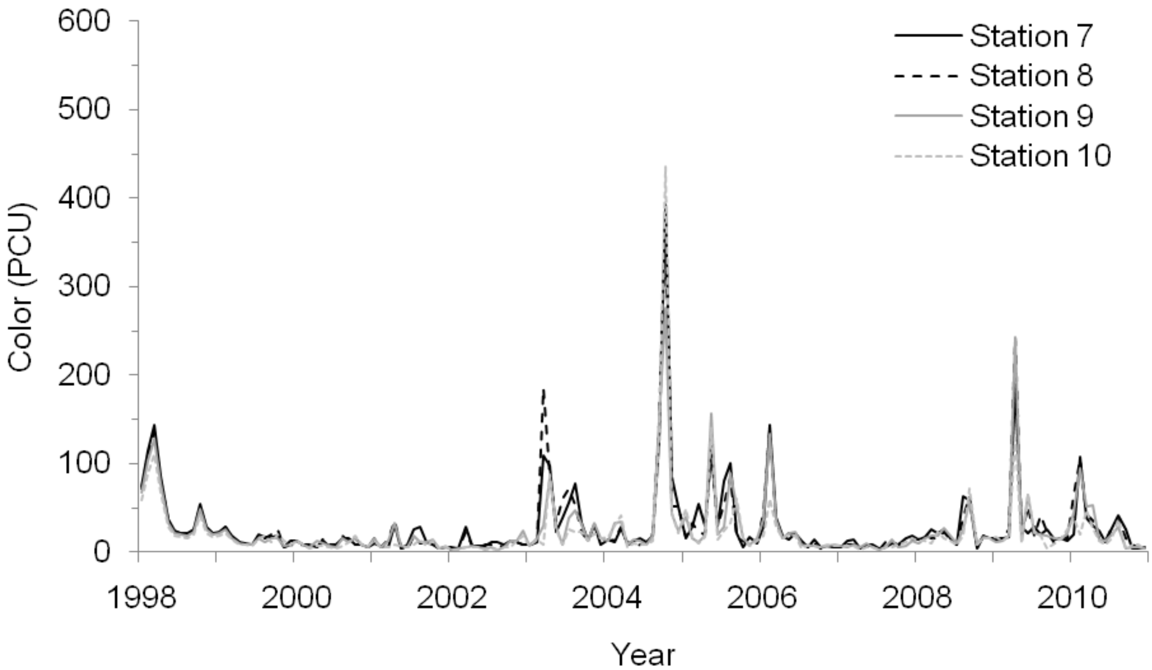


Figure 3-19. Time series plot of color (PCU) for stations 7, 8, 9, and 10.

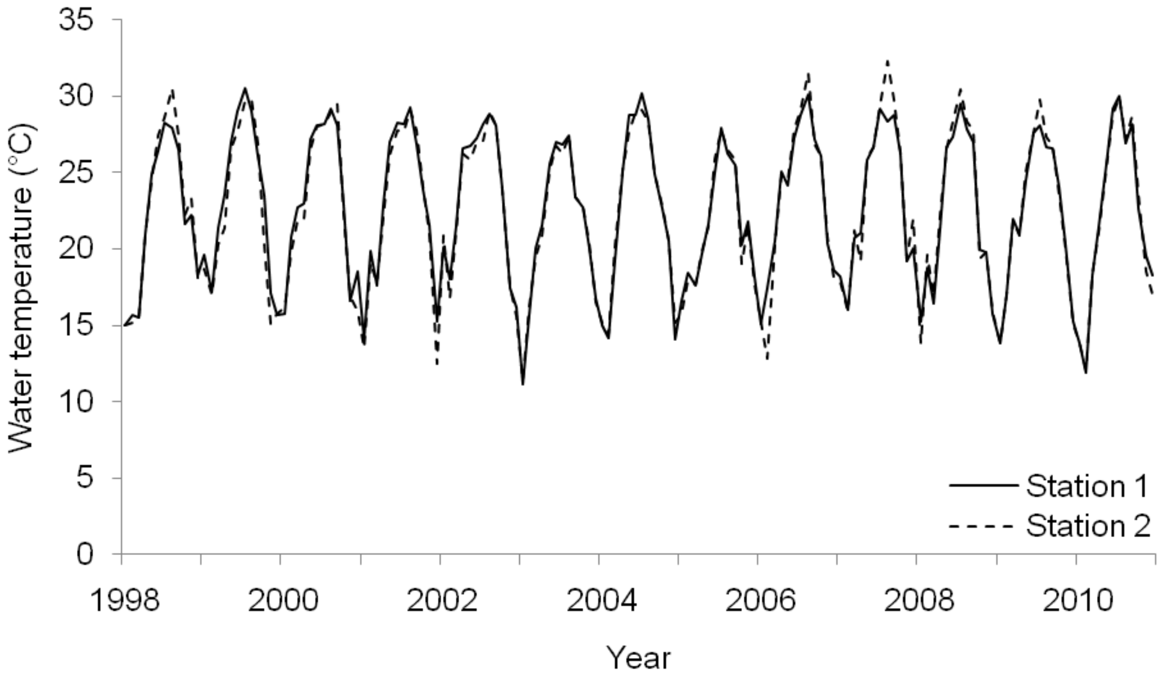


Figure 3-20. Time series plot of water temperature (°C) for stations 1 and 2.

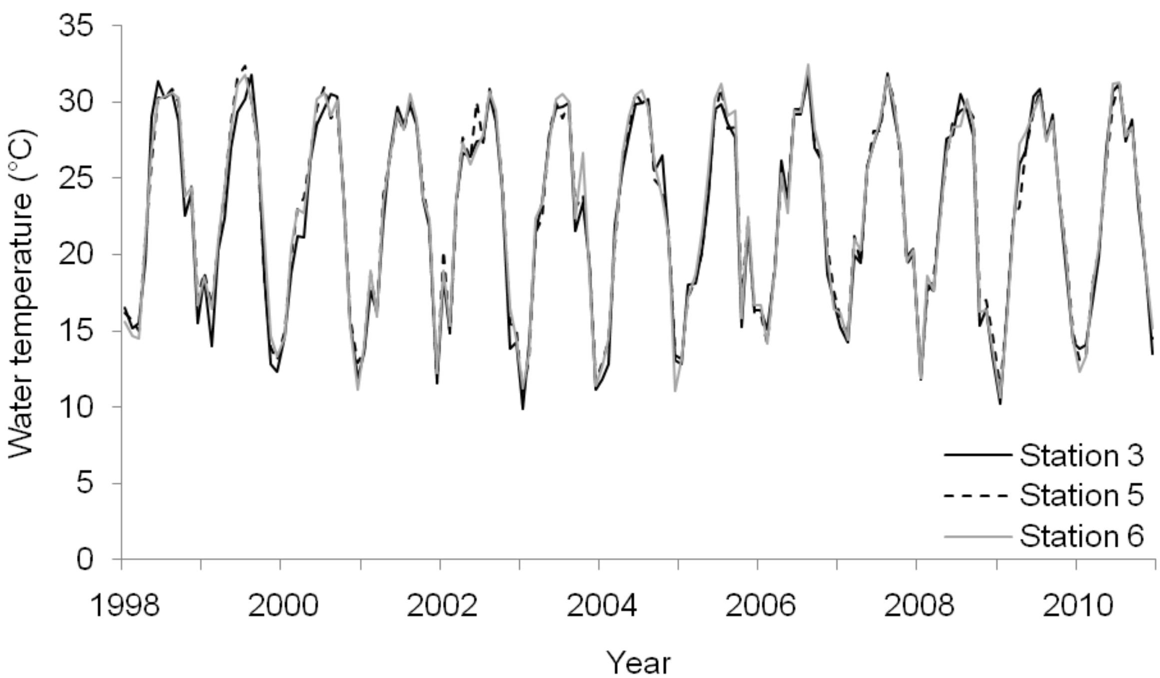


Figure 3-21. Time series plot of water temperature (°C) for stations 3, 5, and 6.

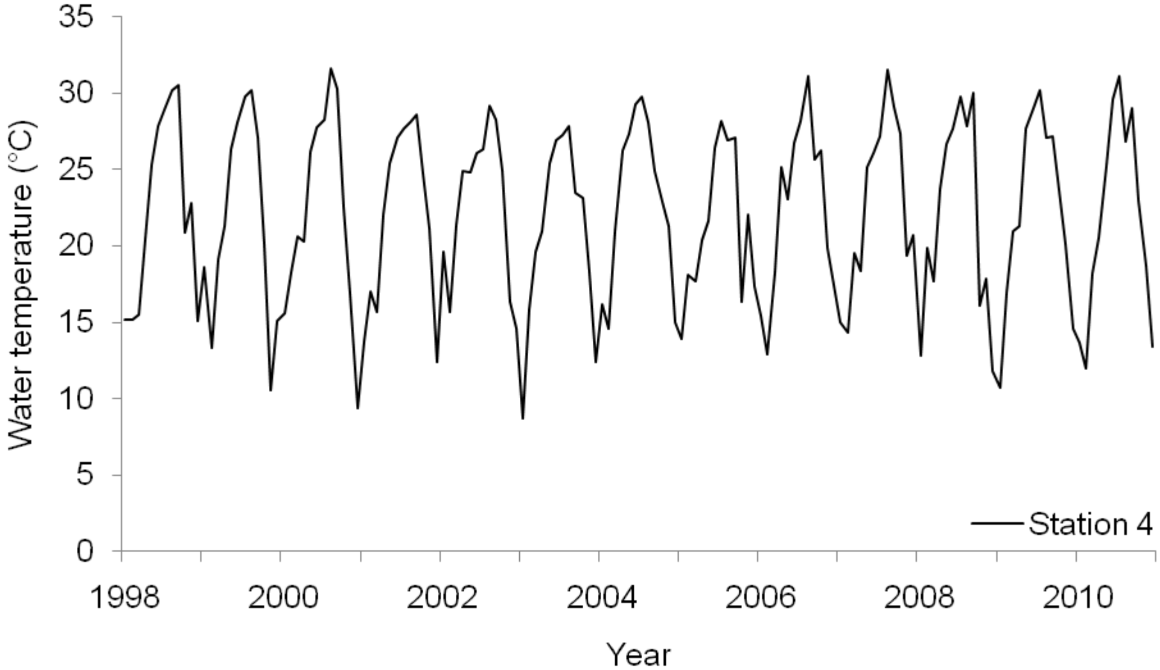


Figure 3-22. Time series plot of water temperature (°C) for station 4.

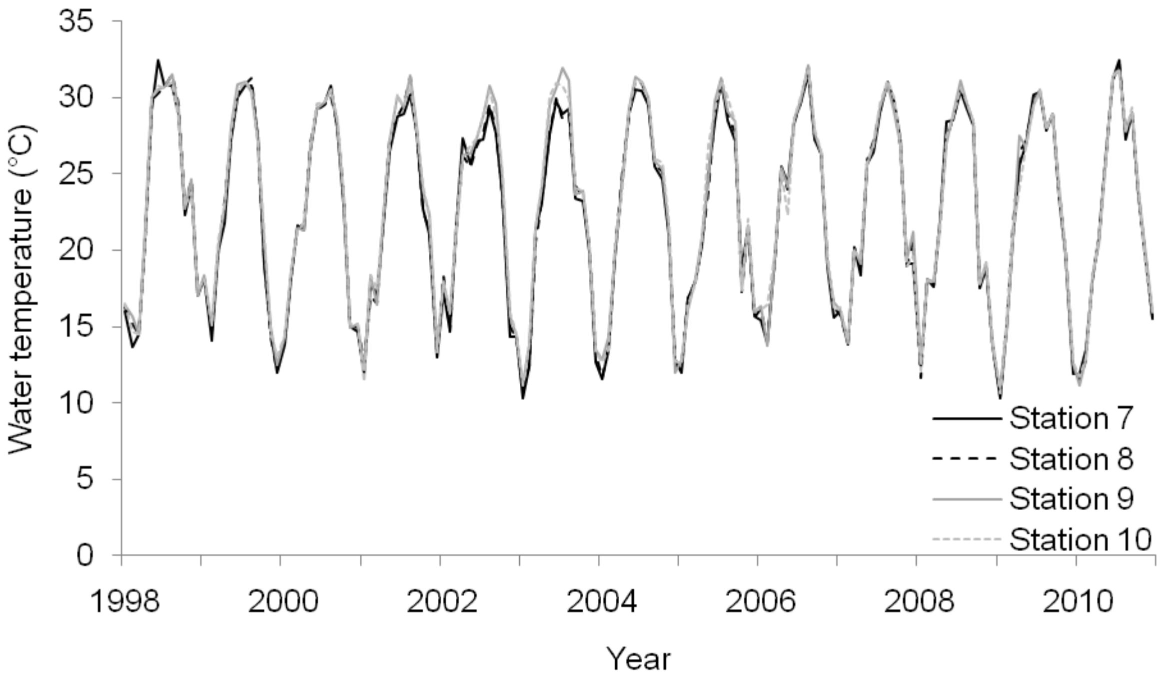


Figure 3-23. Time series plot of water temperature (°C) for stations 7, 8, 9, and 10.

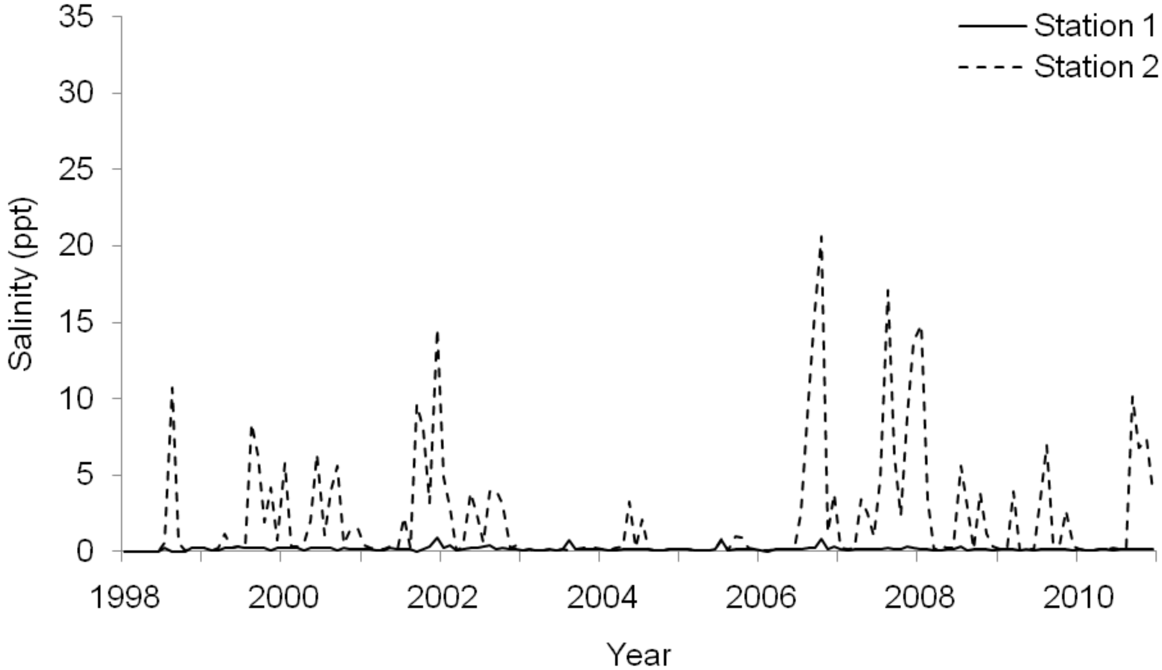


Figure 3-24. Time series plot of salinity (ppt) for stations 1 and 2.

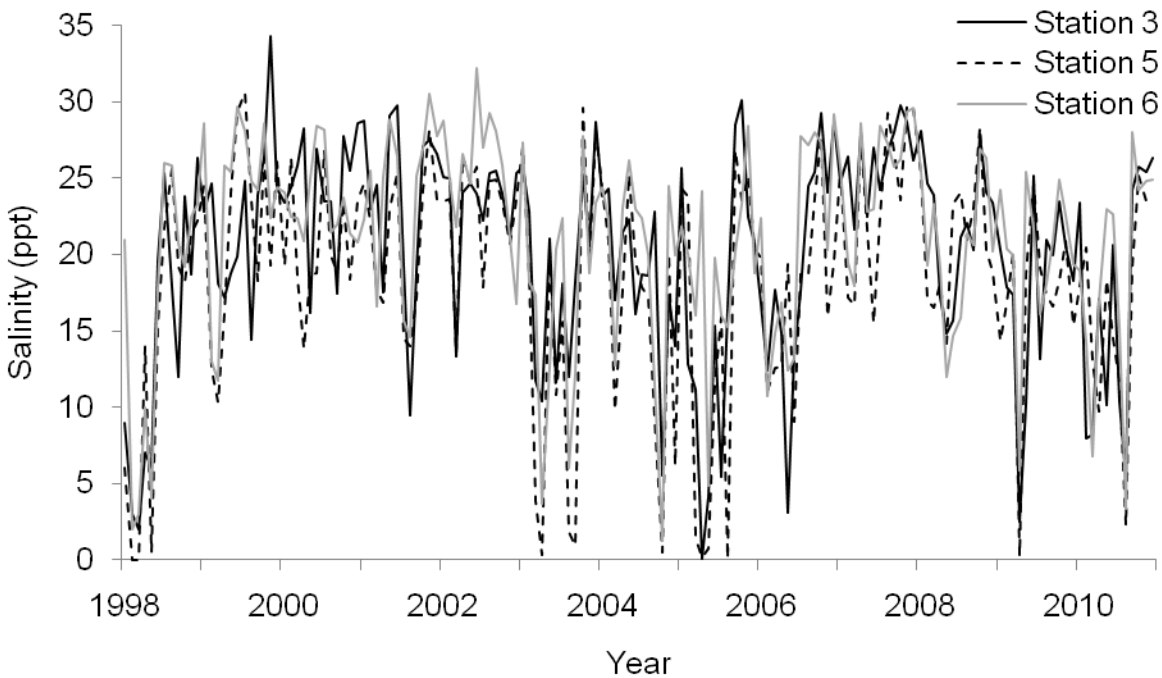


Figure 3-25. Time series plot of salinity (ppt) for stations 3, 5, and 6.

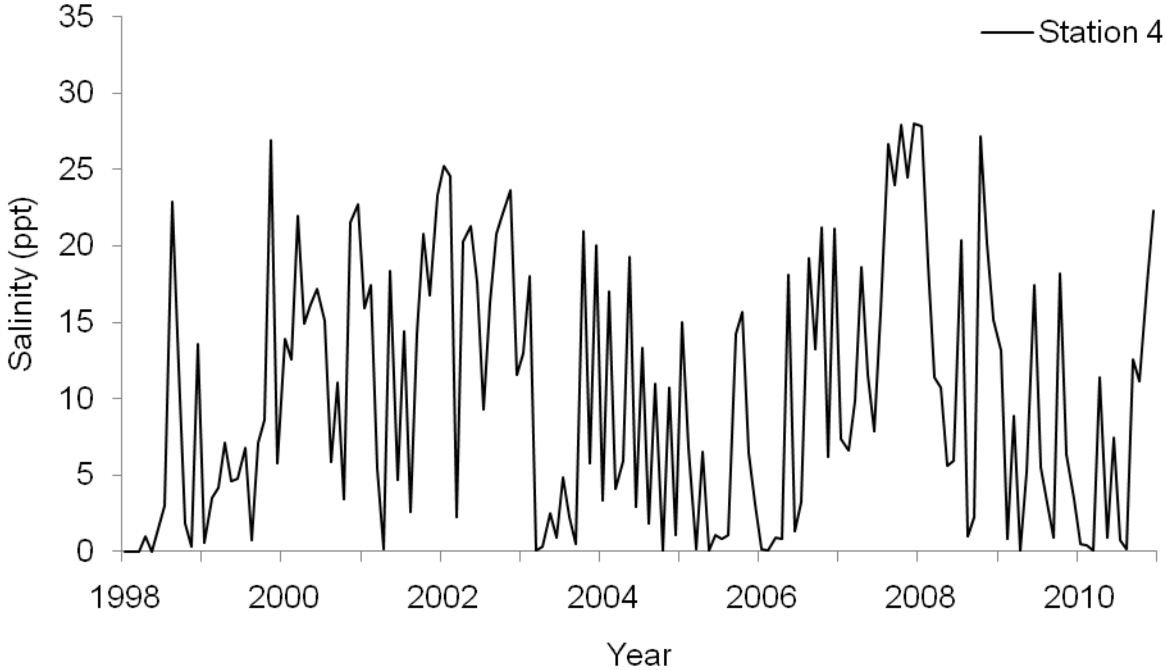


Figure 3-26. Time series plot of salinity (ppt) for station 4.

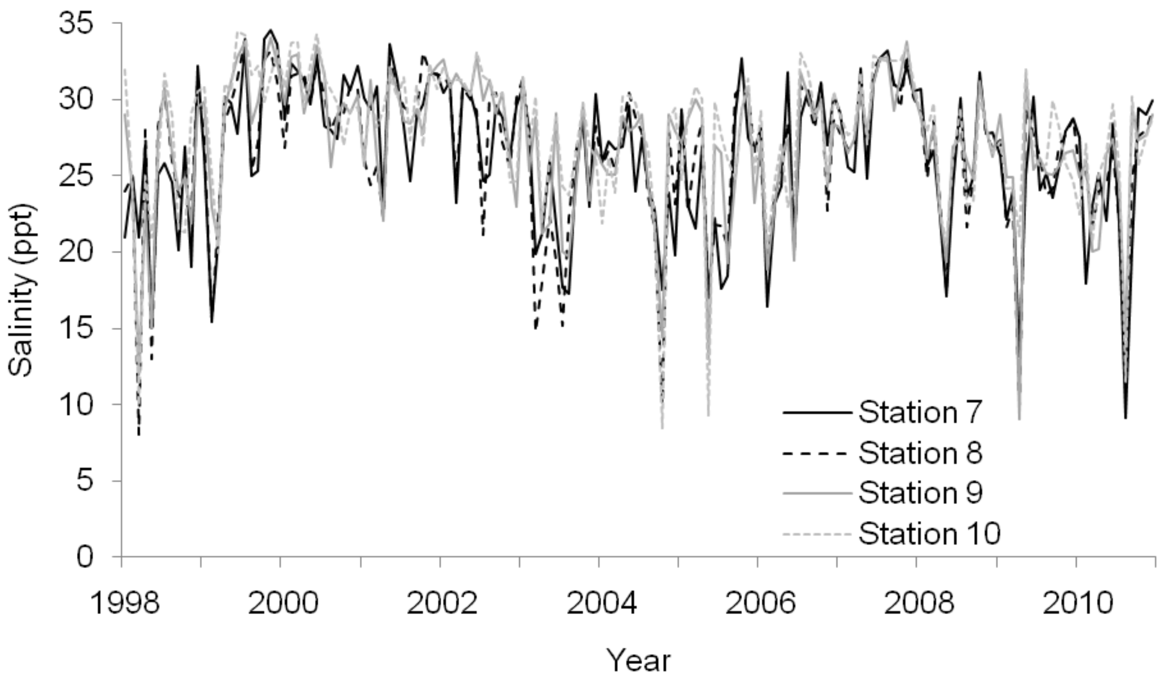


Figure 3-27. Time series plot of salinity (ppt) for stations 7, 8, 9, and 10.

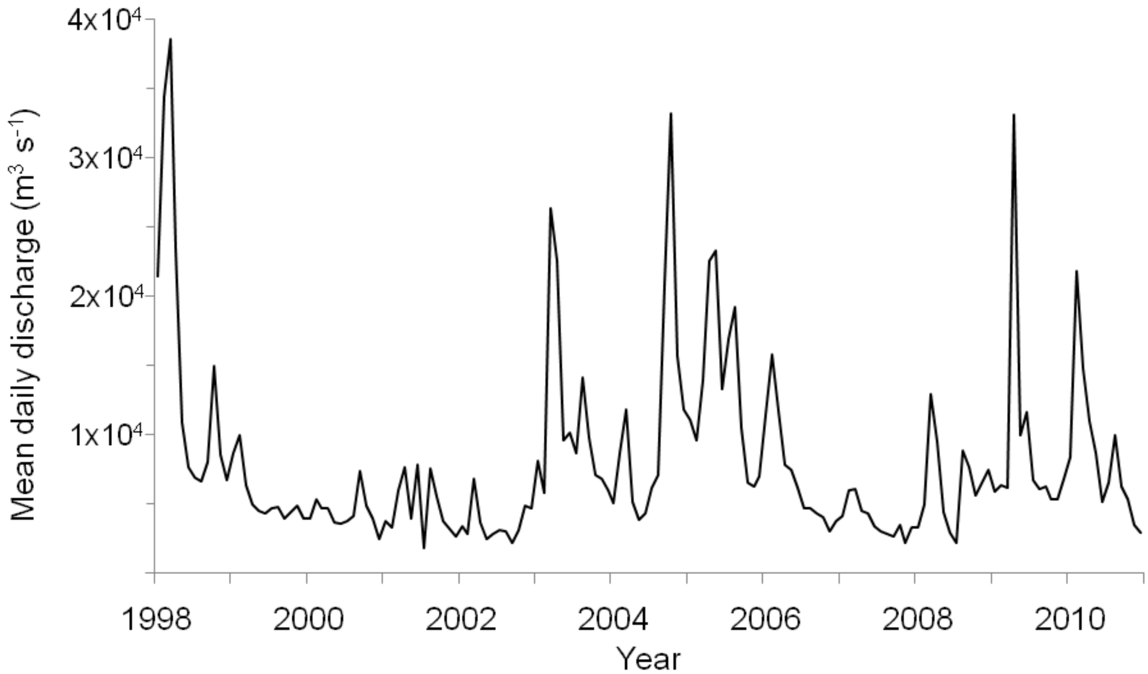


Figure 3-28. Time series plot of mean daily discharge ($m^3 s^{-1}$) at the Gopher River gauge station.

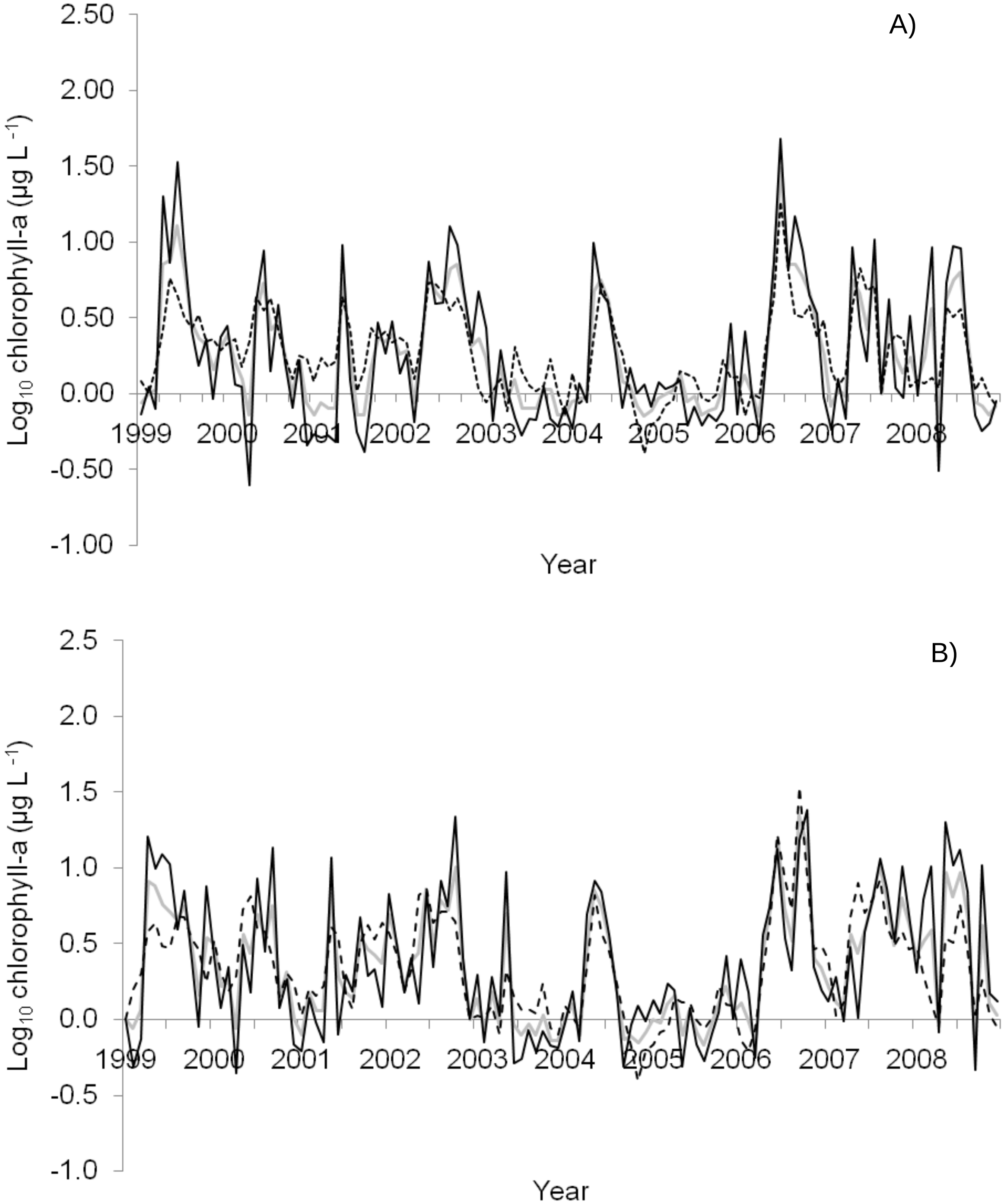


Figure 3-29. Time series plots of monthly observed and fitted values of $\log_{10}$-transformed chlorophyll-a concentrations from models for A) station 1 and B) station 2. Observed values are represented with a solid grey line. Time series adjusted fitted values are represented with a solid black line and unadjusted fitted values are represented with a dashed black line.

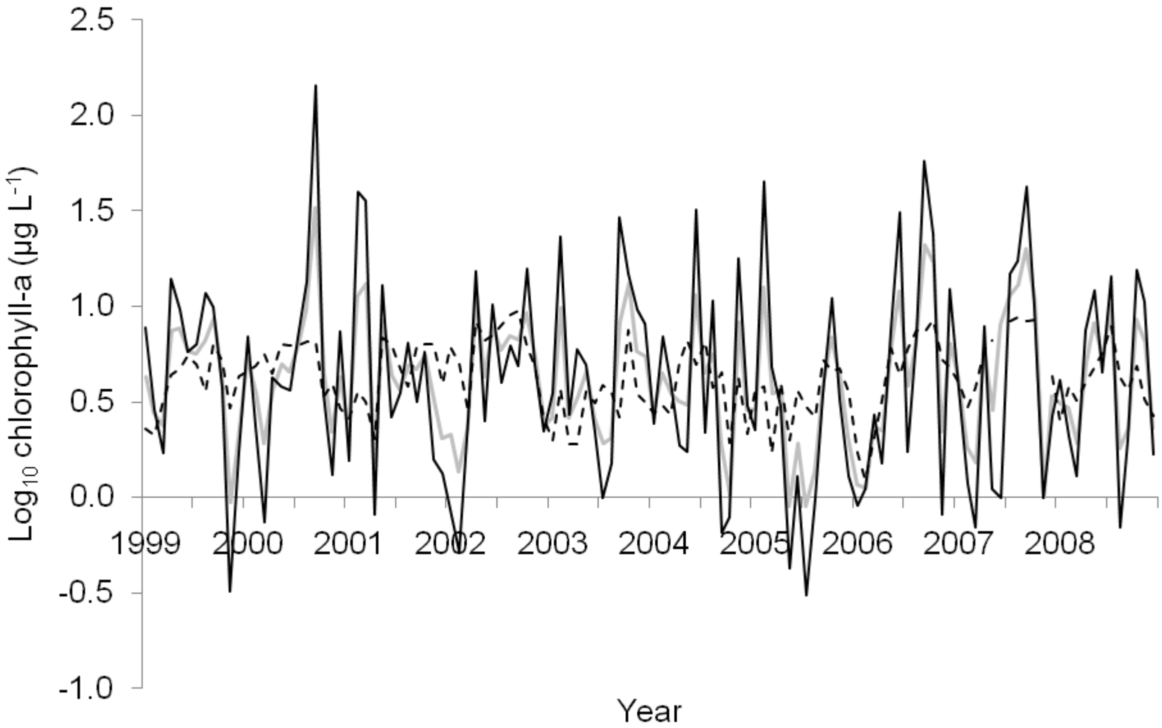
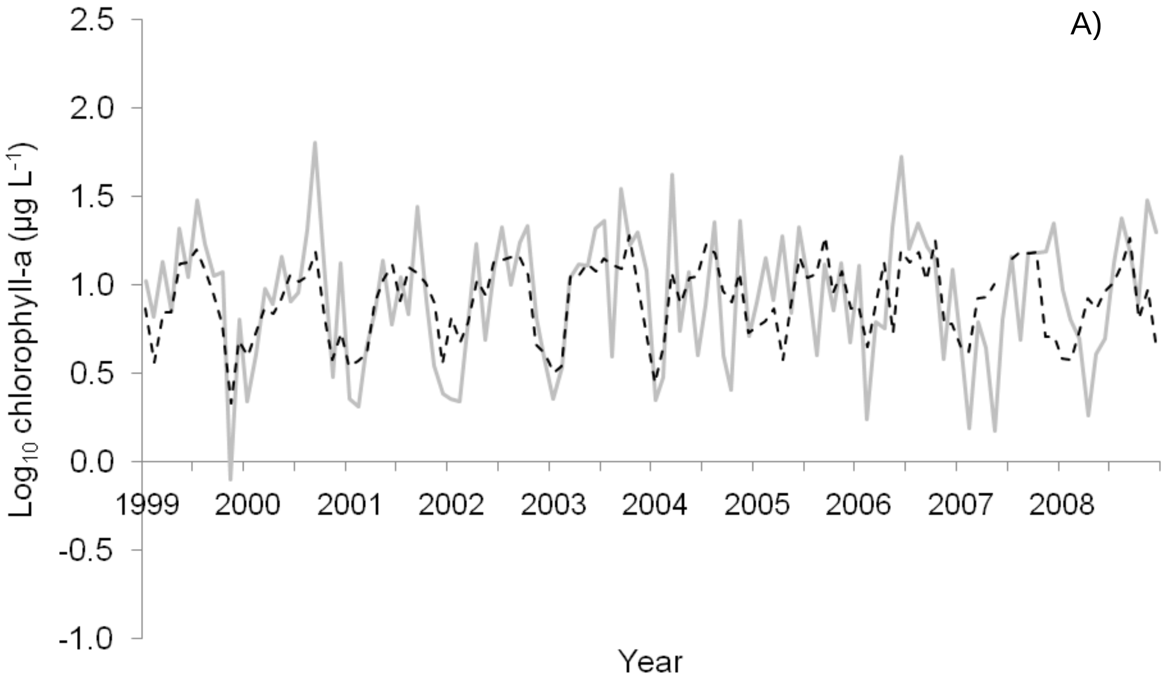


Figure 3-30. Time series plots of monthly observed and fitted values of $\log_{10}$-transformed chlorophyll-a concentrations from models for station 4. Observed values are represented with a solid grey line. Time series adjusted fitted values are represented with a solid black line and unadjusted fitted values are represented with a dashed black line.



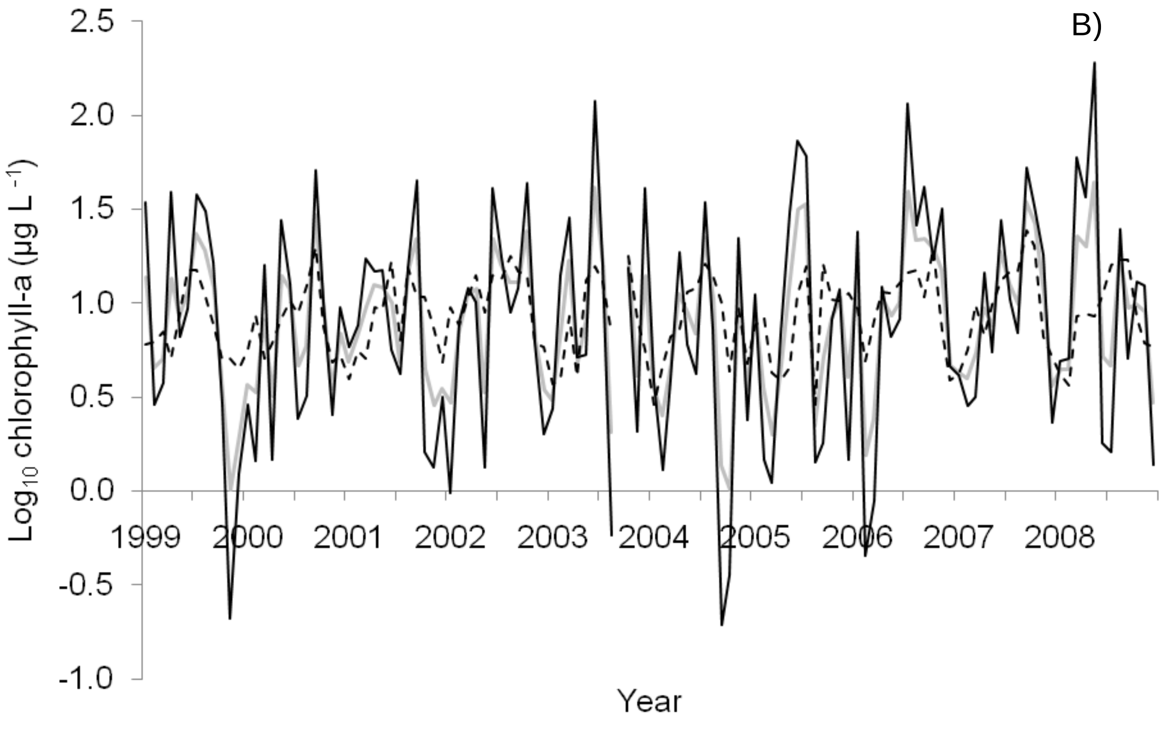


Figure 3-31. Time series plots of monthly observed and fitted values of $\log_{10}$-transformed chlorophyll-a concentrations from models for A) station 3, B) station 5, and C) station 6. Observed values are represented with a solid grey line. Time series adjusted fitted values are represented with a solid black line and unadjusted fitted values are represented with a dashed black line.

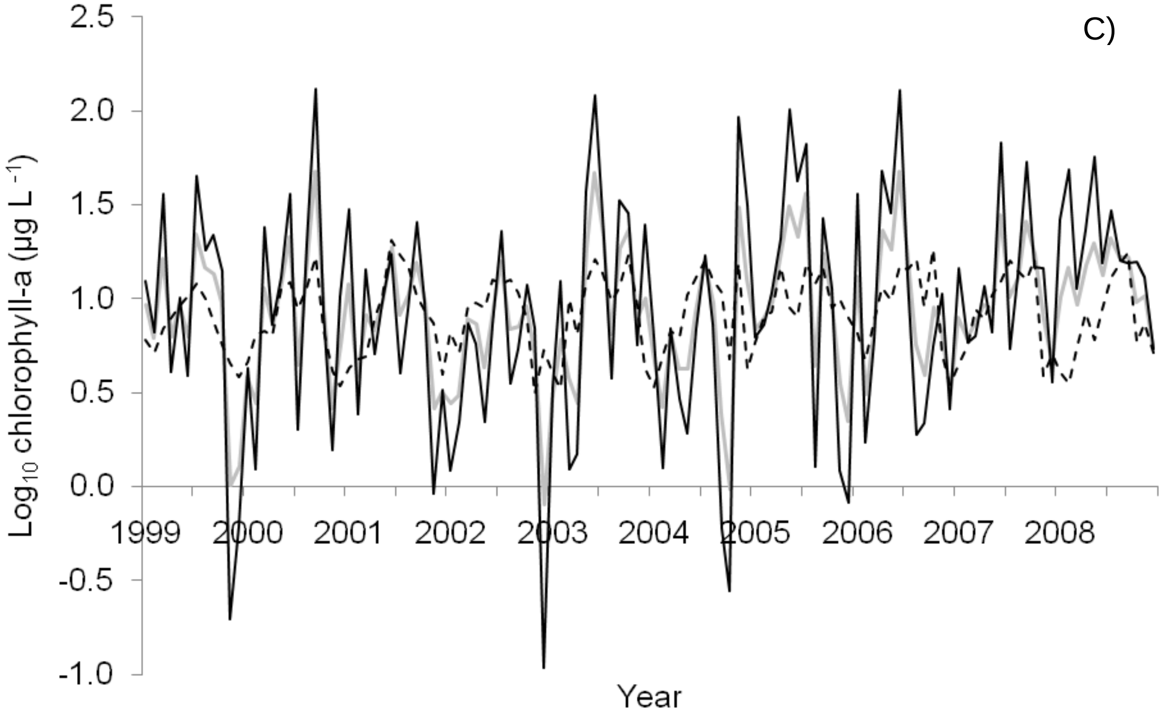


Figure 3-31. Continued

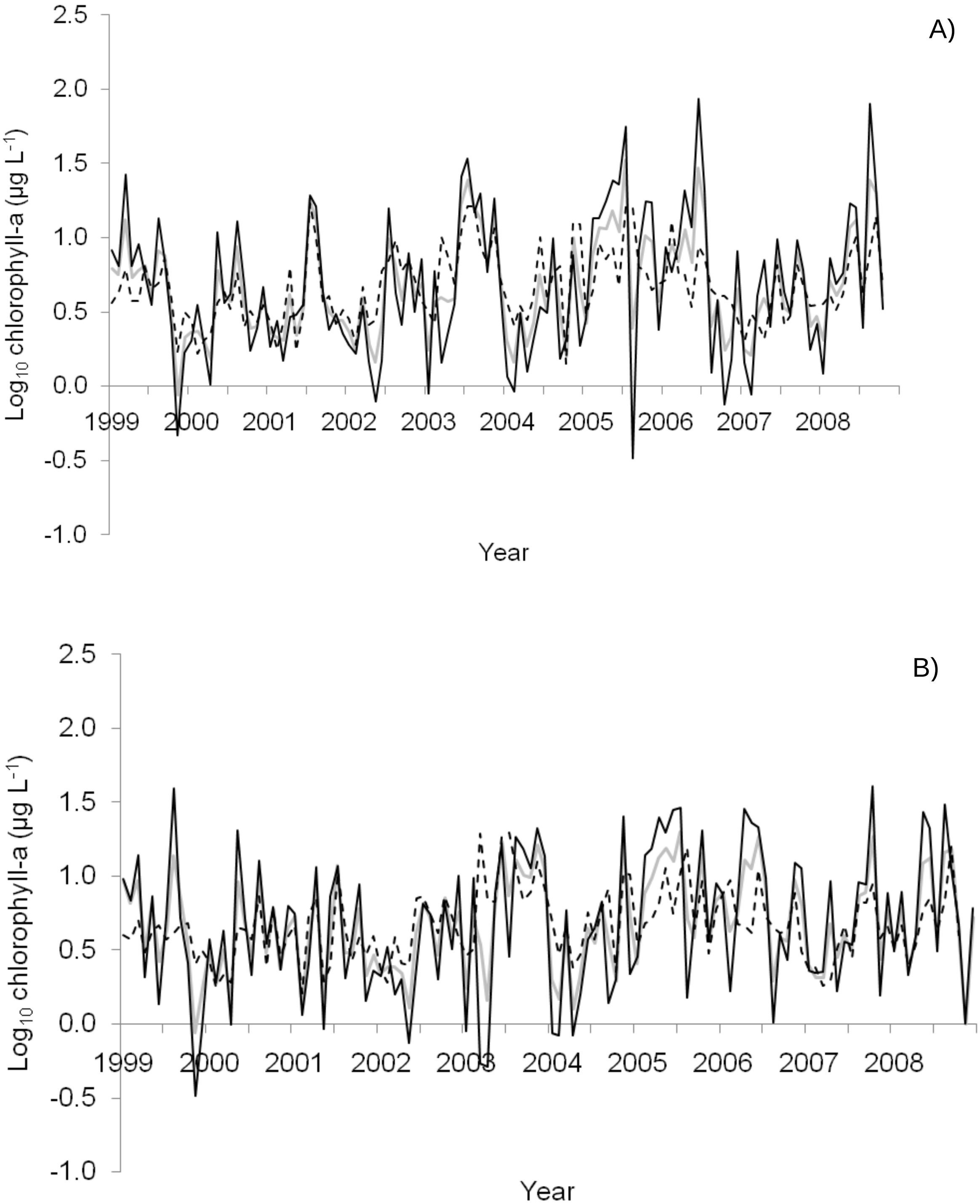
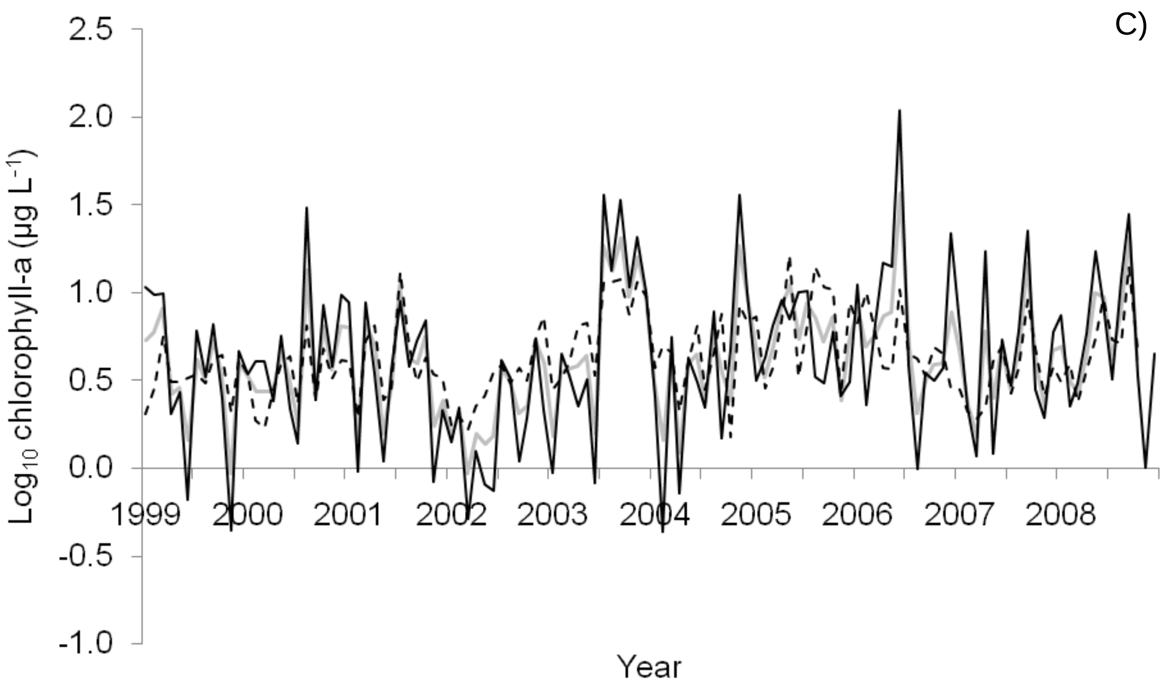


Figure 3-32. Time series plots of monthly observed and fitted values of $\log_{10}$-transformed chlorophyll-a concentrations from models for A) station 7, B) station 8, C) station 9, and D) station 10. Observed values are represented with a solid grey line. Adjusted fitted values are represented with a solid black line. Unadjusted fitted values are represented with a dashed black line.



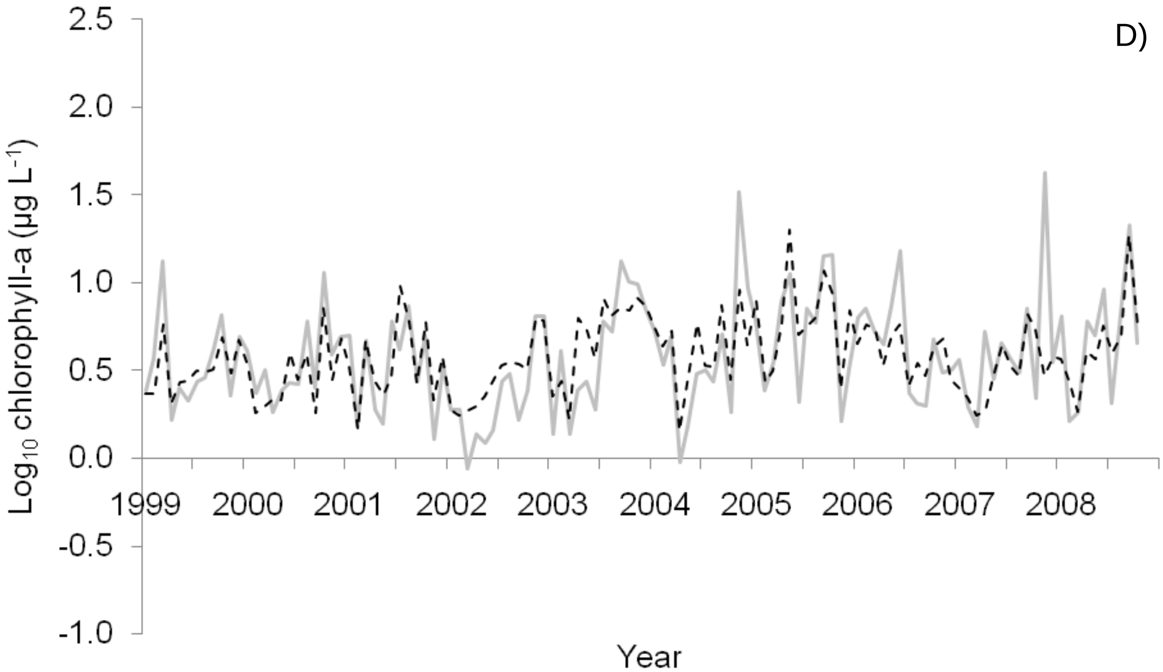


Figure 3-32. Continued

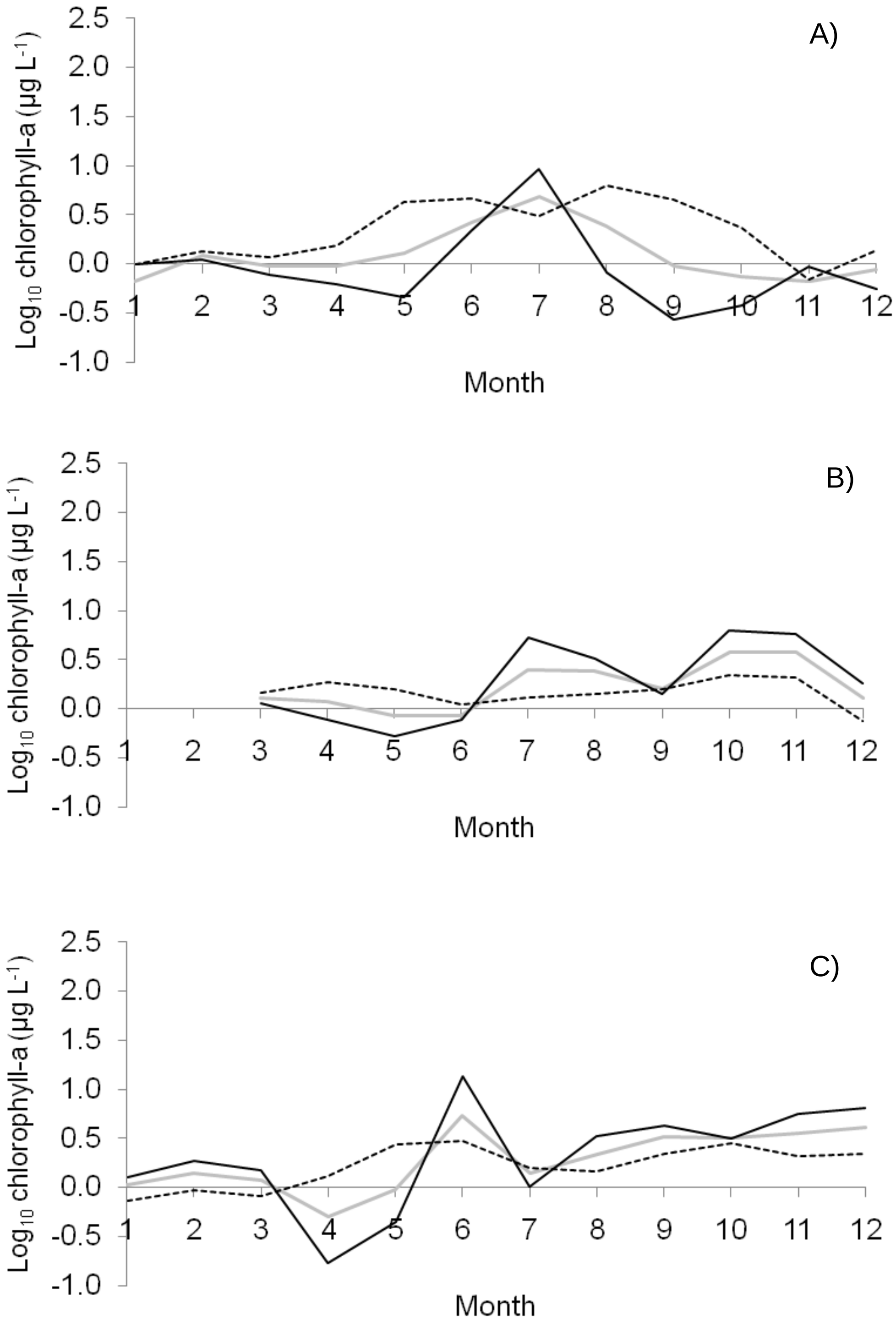


Figure 3-33. Time series plots of hindcasts and forecasts of log_{10}-transformed chlorophyll-a concentrations for station 1 and for holdout samples from A) 1998, B) 2009, and C) 2010. Observed values are represented with a solid grey line. Time series adjusted values are represented with a solid black line and unadjusted values are represented with a dashed black line.

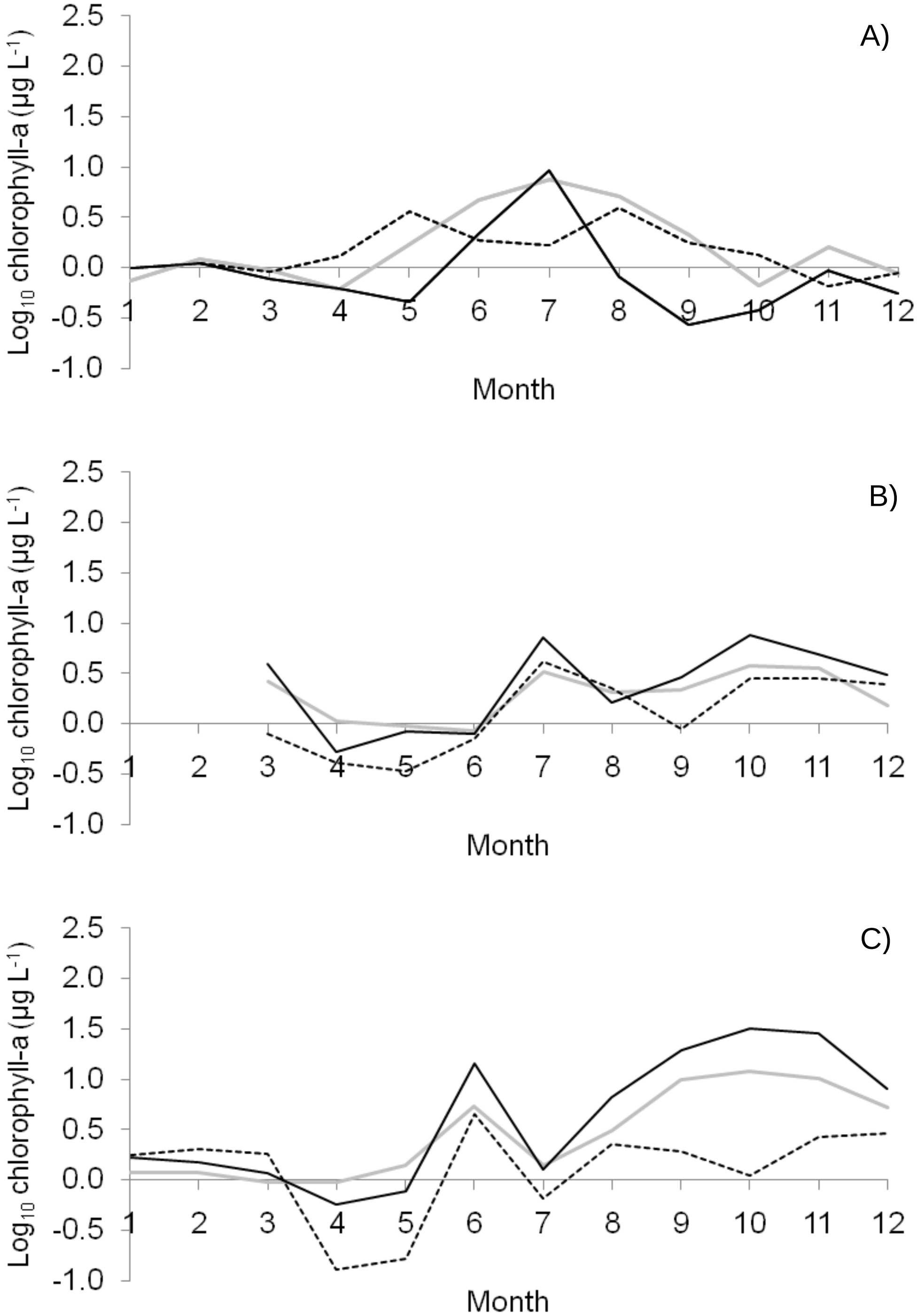


Figure 3-34. Time series plots of hindcasts and forecasts of $\log_{10}$-transformed chlorophyll-a concentrations for station 2 and for holdout samples from A) 1998, B) 2009, and C) 2010. Observed values are represented with a solid grey line. Time series adjusted values are represented with a solid black line and unadjusted values are represented with a dashed black line.

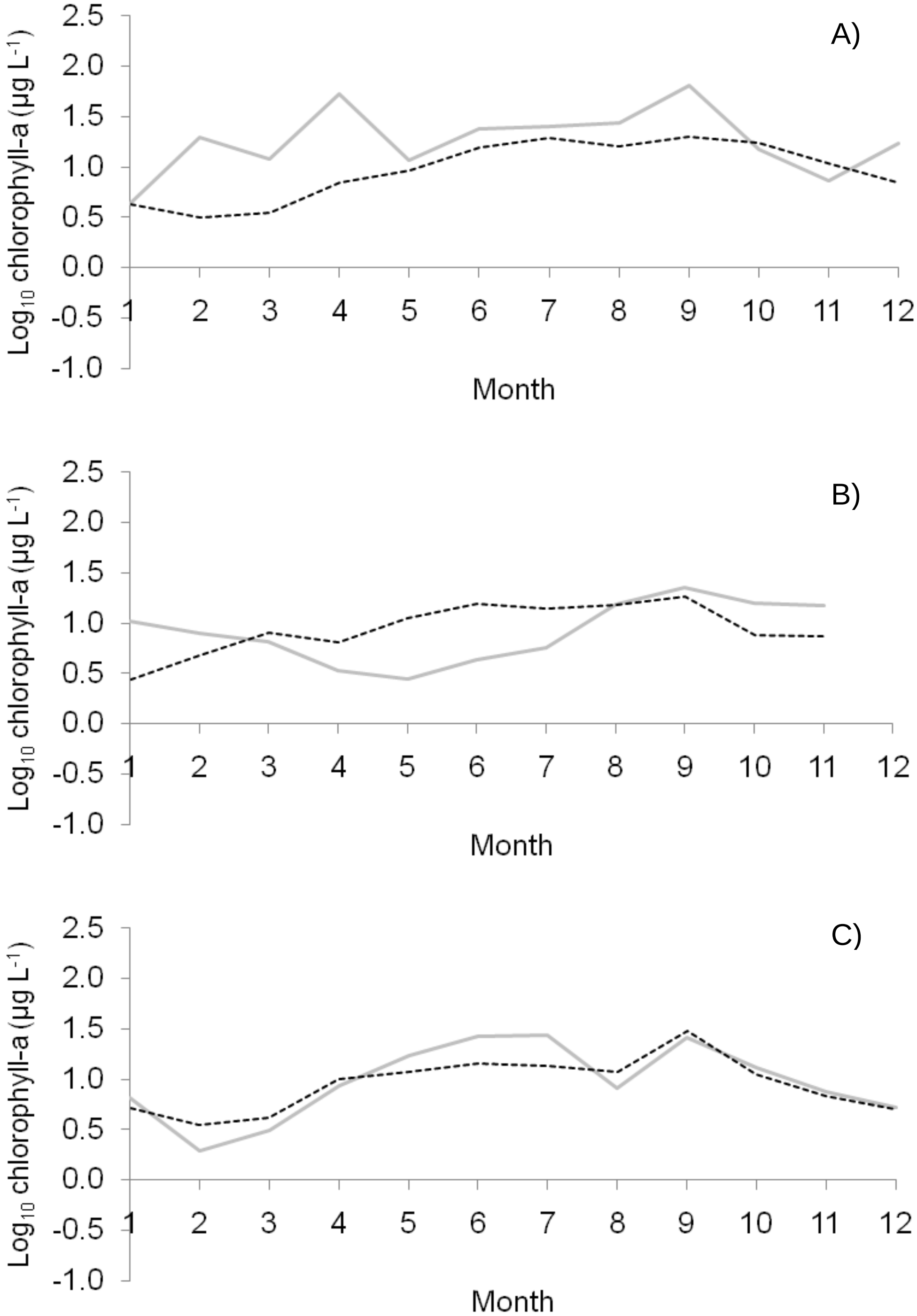


Figure 3-35. Time series plots of hindcasts and forecasts of $\log_{10}$-transformed chlorophyll-a concentrations for station 3 and for holdout samples from A) 1998, B) 2009, and C) 2010. Observed values are represented with a solid grey line. Unadjusted values are represented with a dashed black line.

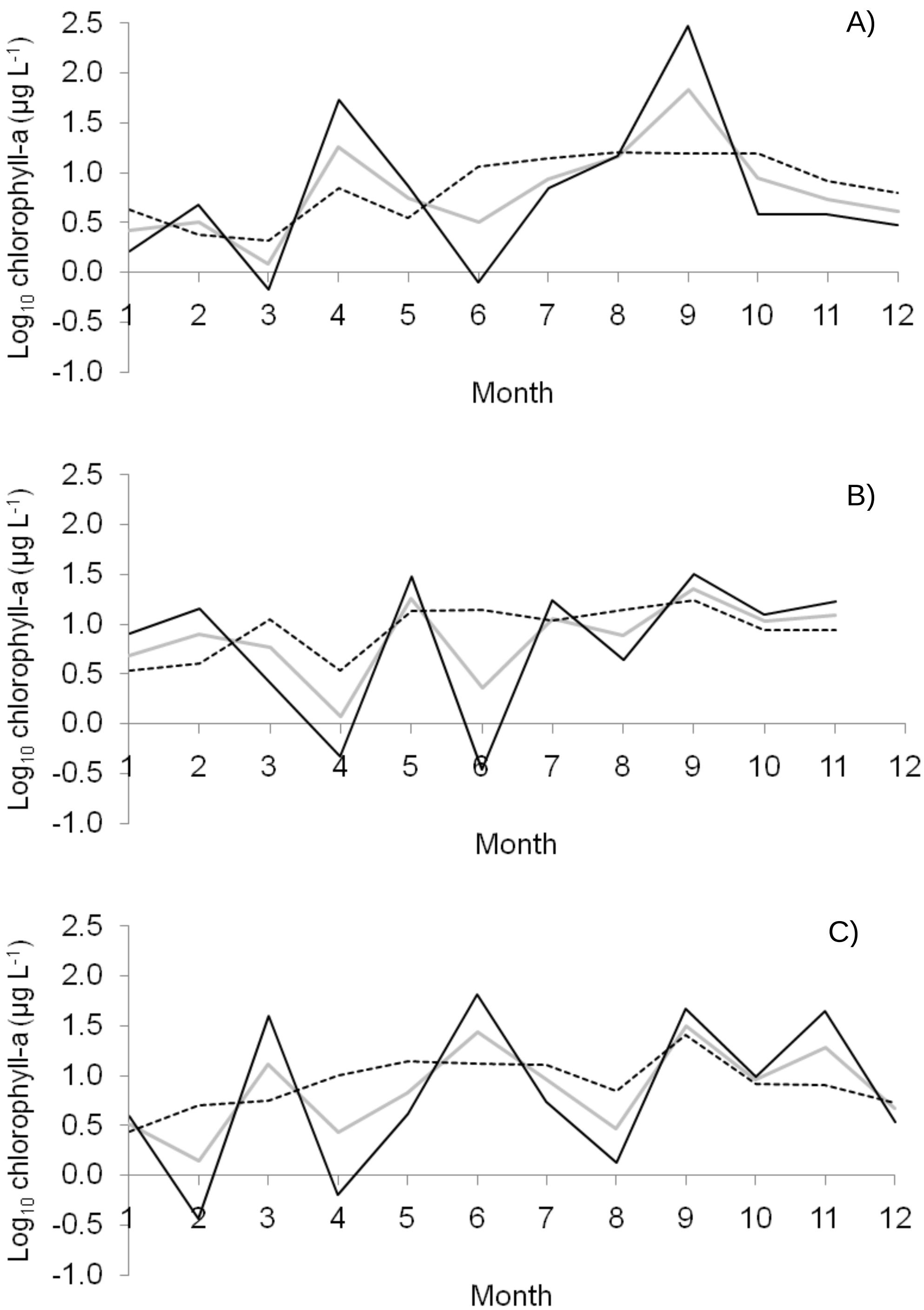


Figure 3-36. Time series plots of hindcasts and forecasts of log$_{10}$-transformed chlorophyll-a concentrations for station 5 and for holdout samples from A) 1998, B) 2009, and C) 2010. Observed values are represented with a solid grey line. Time series adjusted values are represented with a solid black line and unadjusted values are represented with a dashed black line.

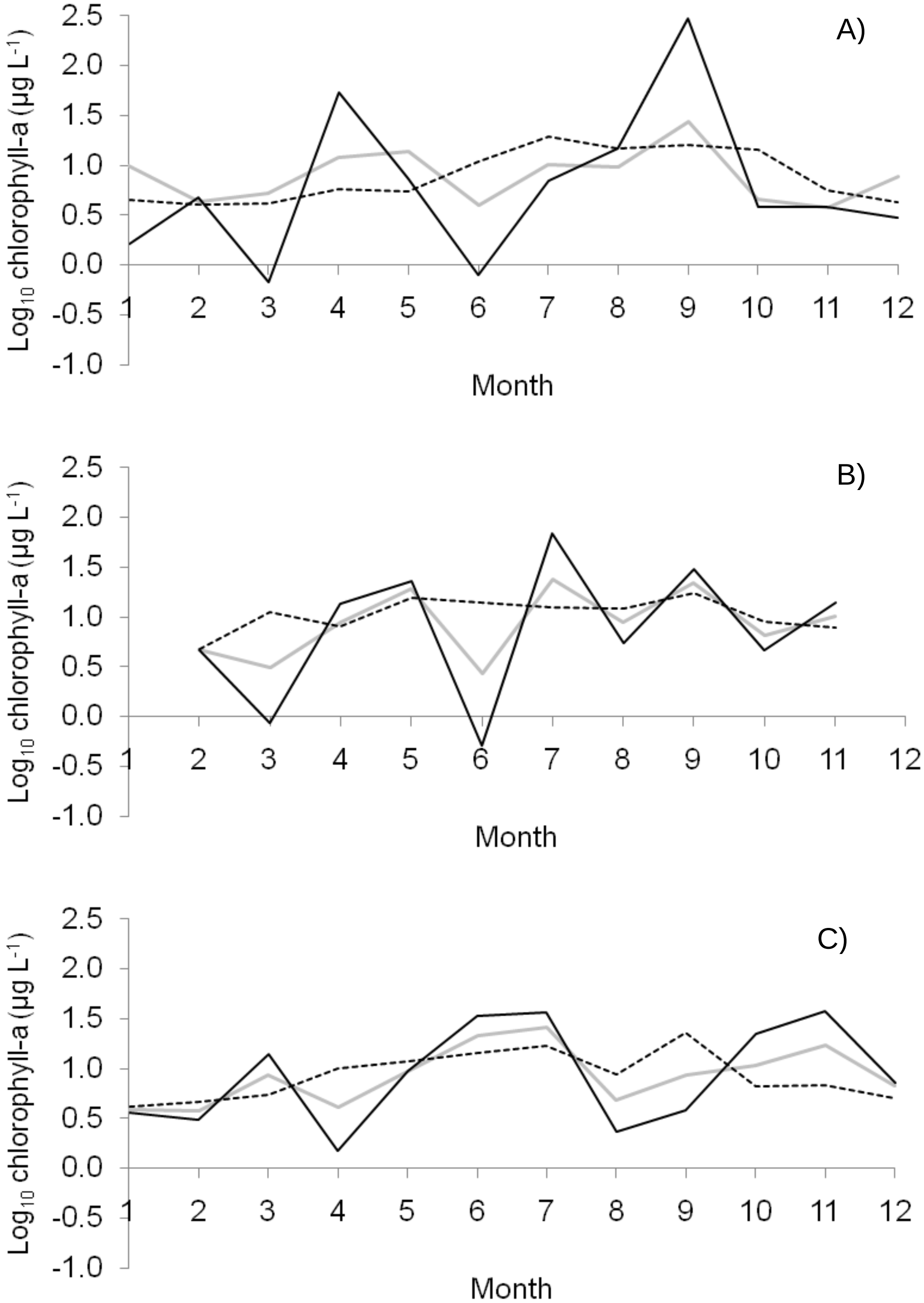


Figure 3-37. Time series plots of hindcasts and forecasts of $\log_{10}$-transformed chlorophyll-a concentrations for station 6 and for holdout samples from A) 1998, B) 2009, and C) 2010. Observed values are represented with a solid grey line. Time series adjusted values are represented with a solid black line and unadjusted values are represented with a dashed black line.

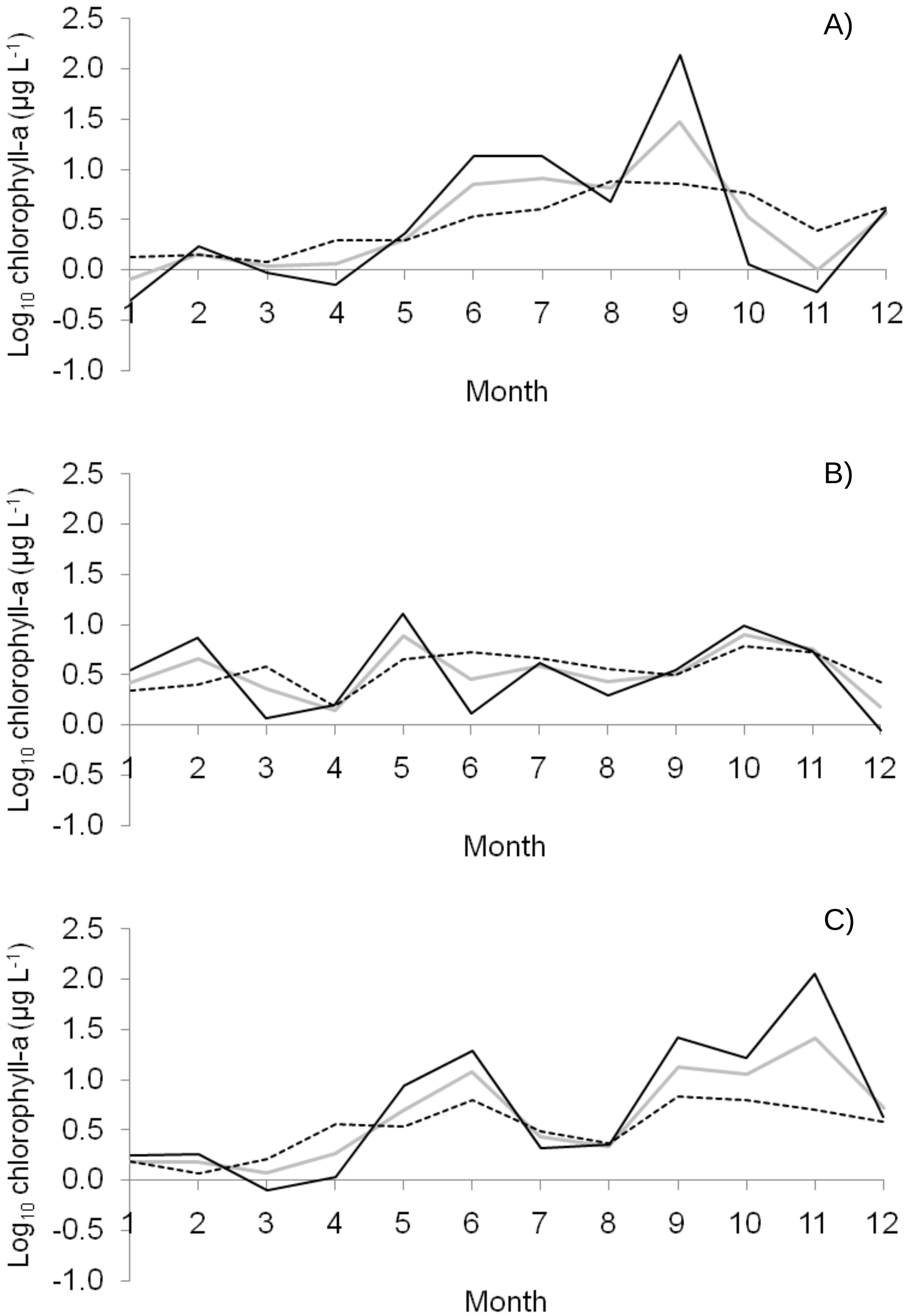


Figure 3-38. Time series plots of hindcasts and forecasts of $\log_{10}$-transformed chlorophyll-a concentrations for station 4 and for holdout samples from A) 1998, B) 2009, and C) 2010. Observed values are represented with a solid grey line. Time series adjusted values are represented with a solid black line and unadjusted values are represented with a dashed black line.

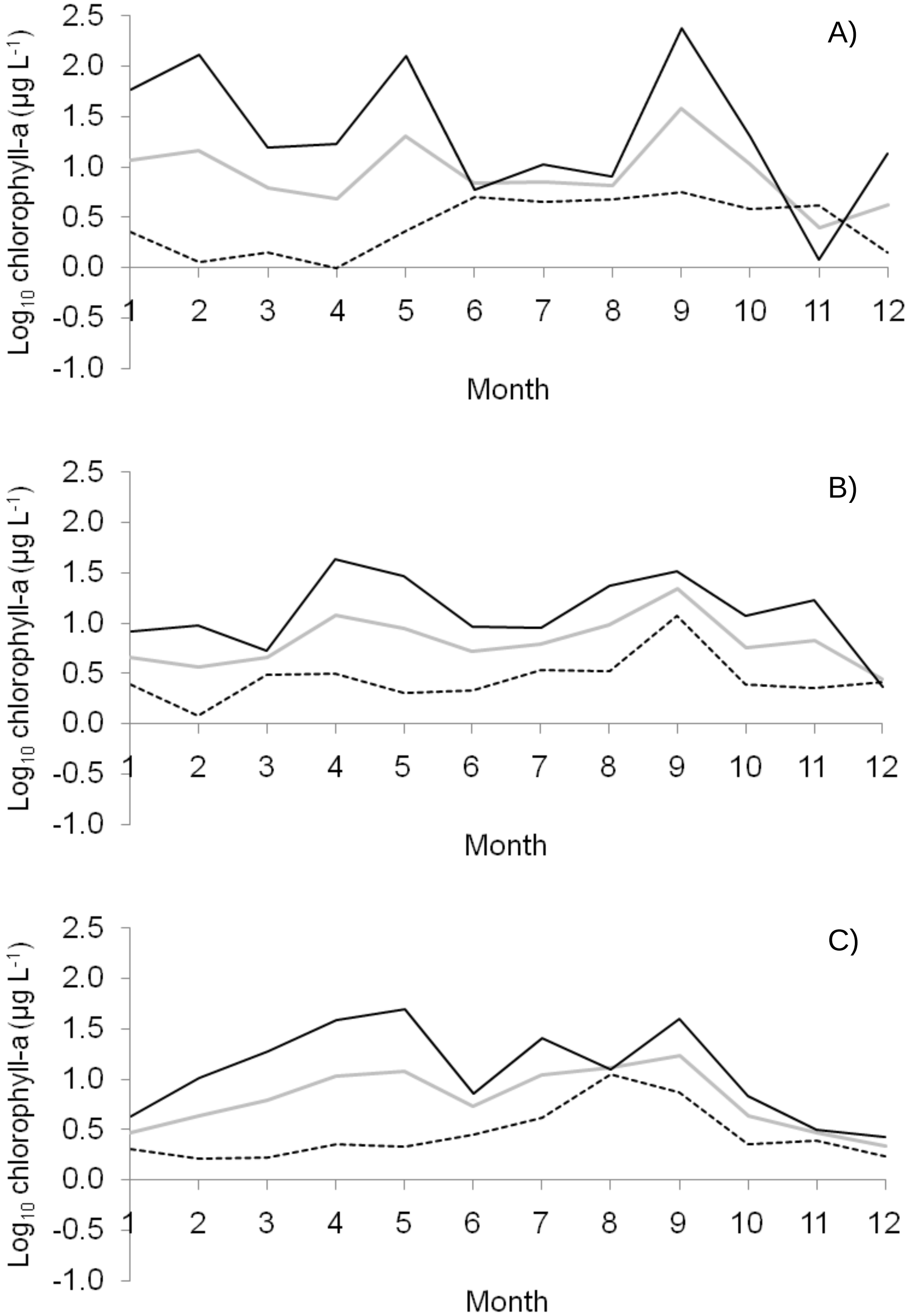


Figure 3-39. Time series plots of hindcasts and forecasts of $\log_{10}$-transformed chlorophyll-a concentrations for station 7 and for holdout samples from A) 1998, B) 2009, and C) 2010. Observed values are represented with a solid grey line. Time series adjusted values are represented with a solid black line and unadjusted values are represented with a dashed black line.

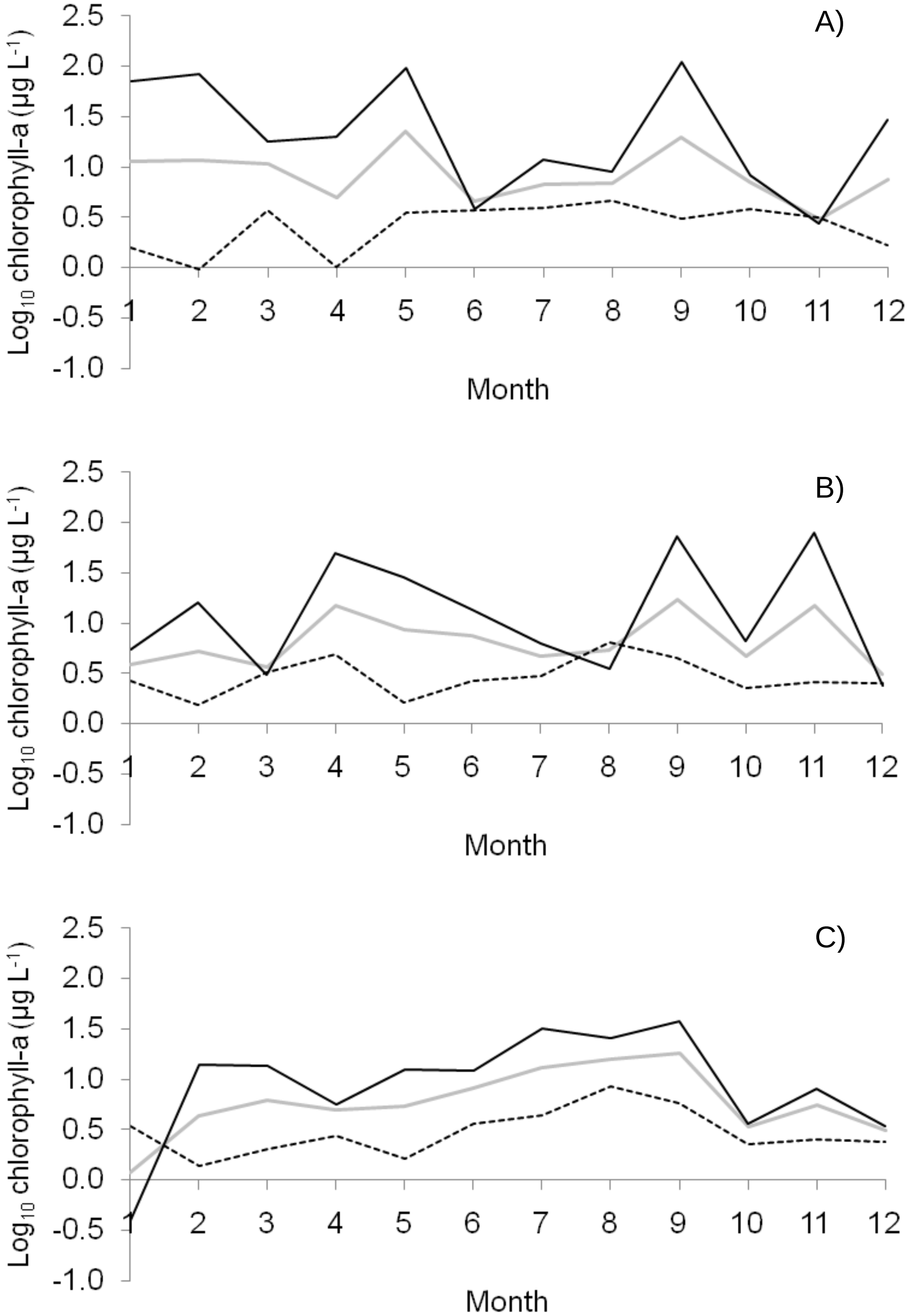


Figure 3-40. Time series plots of hindcasts and forecasts of $\log_{10}$-transformed chlorophyll-a concentrations for station 8 and for holdout samples from A) 1998, B) 2009, and C) 2010. Observed values are represented with a solid grey line. Time series adjusted values are represented with a solid black line and unadjusted values are represented with a dashed black line.

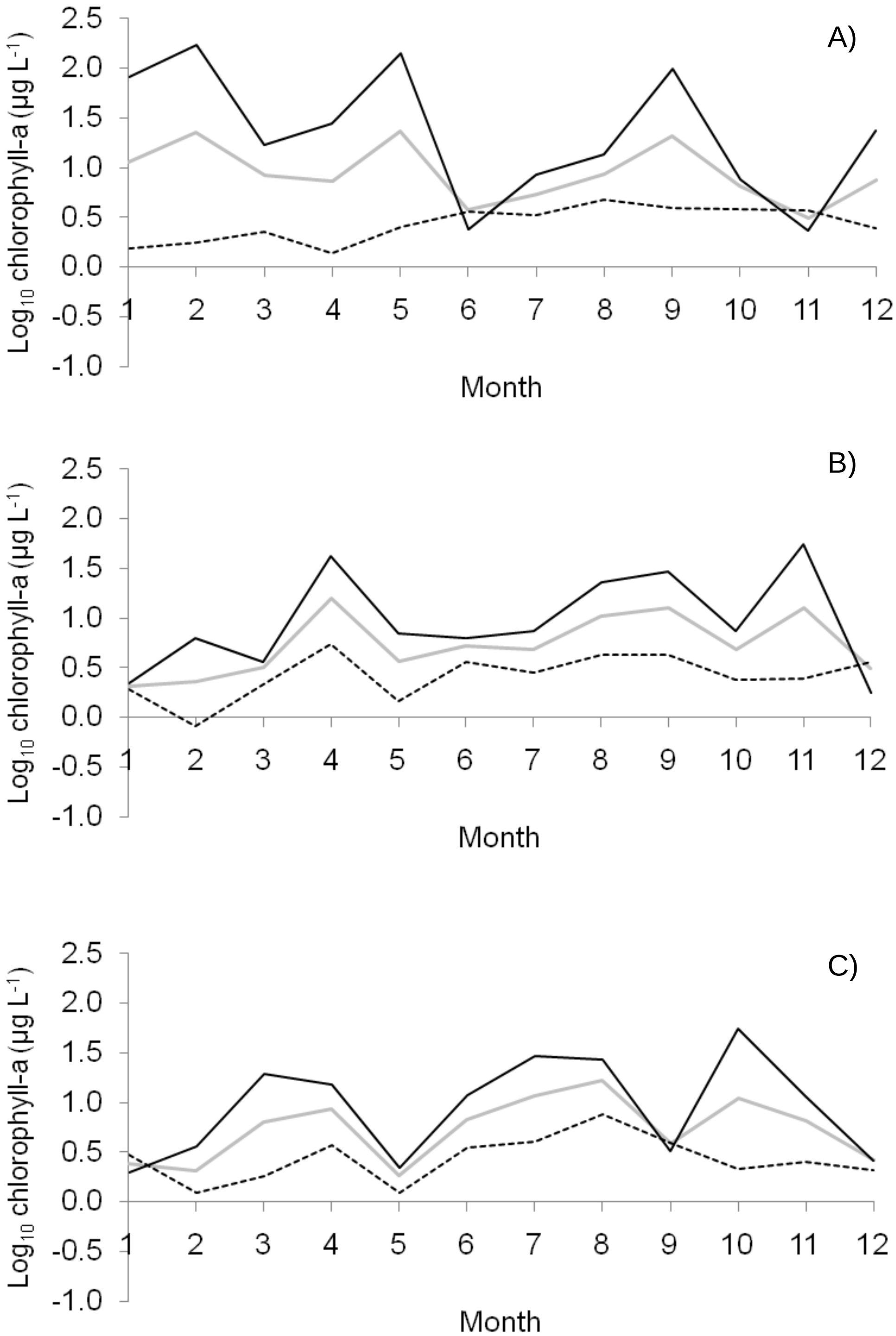


Figure 3-41. Time series plots of hindcasts and forecasts of $\log_{10}$-transformed chlorophyll-a concentrations for station 9 and for holdout samples from A) 1998, B) 2009, and C) 2010. Observed values are represented with a solid grey line. Time series adjusted values are represented with a solid black line and unadjusted values are represented with a dashed black line.

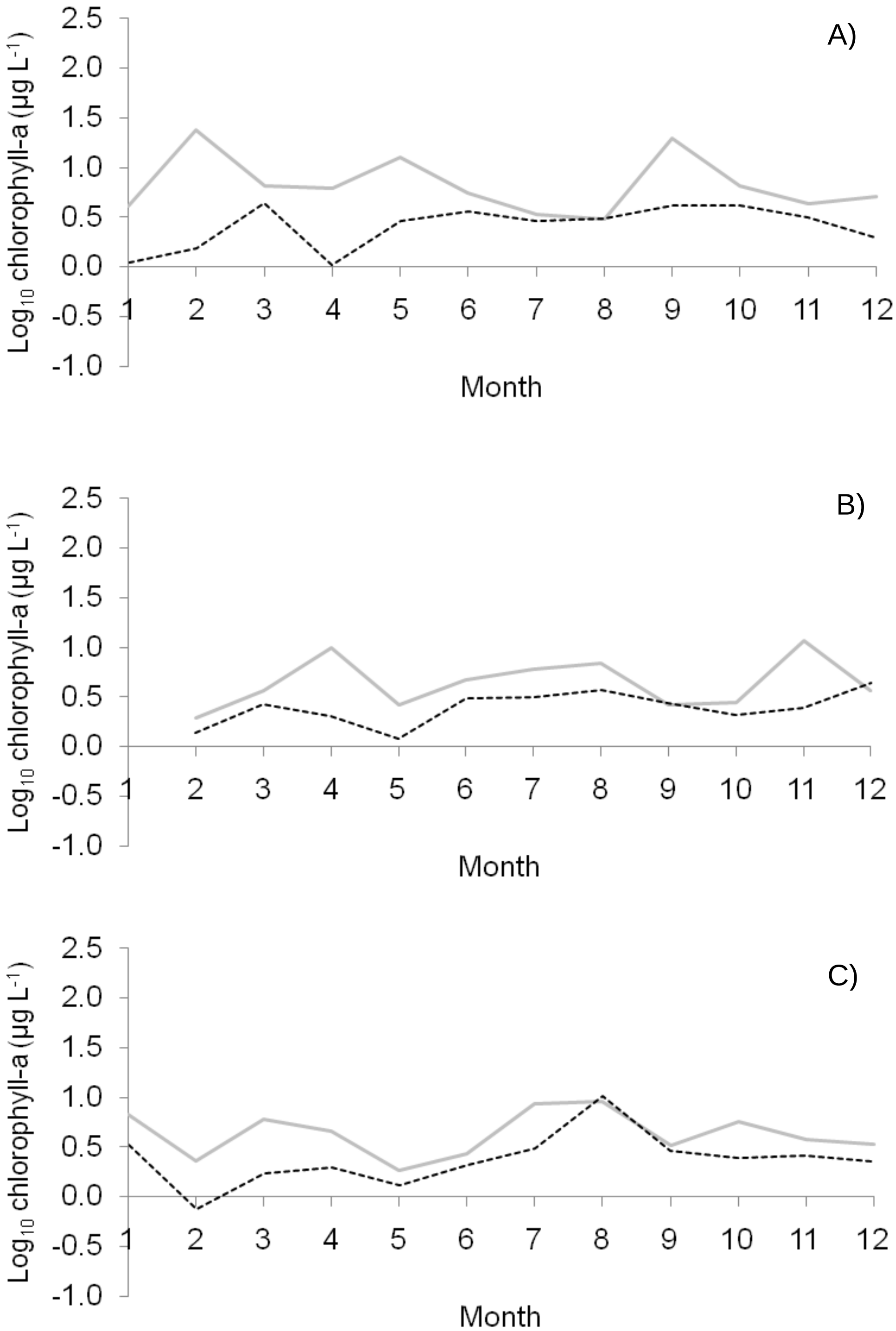


Figure 3-42. Time series plots of hindcasts and forecasts of $\log_{10}$-transformed chlorophyll-a concentrations for station 10 and for holdout samples from A) 1998, B) 2009, and C) 2010. Observed values are represented with a solid grey line. Unadjusted values are represented with a dashed black line.

CHAPTER 4
CONCLUSIONS

This study characterized trends and relationships for water quality parameters measured over decades at multiple stations within the Suwannee River and its estuary. Results of these analyses provided an improved understanding of water quality dynamics in the Suwannee system and highlighted several important considerations for managers.

In spite of an improved understanding of some aspects of nitrogen and chlorophyll dynamics, the overall complexity of the Suwannee system remains a challenge, which has implications for the management of this system and others. This complexity was most apparent in links between results from this study and previous work, where diverse influential factors included short-term weather events, long-term climatic cycles, coastal hydrodynamics, groundwater transport, biogeochemistry, social dynamics (e.g., fertilizer use and land use decisions), and biological processes. Beyond identifying these factors as important components of chlorophyll and nutrient dynamics worthy of further research, the links also highlighted challenges facing managers as they address water quality concerns, such as nutrient impairment.

More specifically, it is unrealistic to expect a complete and detailed understanding of the intricacies of nutrient dynamics or biological responses, such as phytoplankton abundance and distribution. Attaining an adequate understanding of these interactions and influences is especially challenging given the limited amount of time and resources typically available. With these challenges in mind, this study highlighted some key considerations for managers seeking to define attainable magnitude, duration, and frequency components of water quality criteria.

One consideration is the influence of events and climatic cycles, which cannot be managed, but potentially affect nutrient and phytoplankton dynamics. The influence of events and cycles was evident in nutrient trends in the river and estuary where increases in total nitrogen concentrations around 2003 most likely were related to increased rainfall associated with an El Niño event (e.g., Figures 3-8 and 3-1). However, a closer examination of the time series indicated that climatic conditions prior to this event probably affected the system's response. For instance, larger increases in total nitrogen were observed after the 2003 El Niño event than after the relatively stronger 1997-1998 El Niño because the former event was preceded by a sustained drought and the latter was preceded by relatively normal rainfall (Crandall et al. 1999; Copeland et al. 2009). Changes in precipitation due to climatic cycles also can affect phytoplankton abundance and distribution indirectly by altering nutrient loads, residence times, or light availability (Justic et al. 1997). Because impacts from unmanageable factors like climatic cycles can be significant, managers should strive to collect data across a range of conditions and time periods in an effort to document the full range of variability. This information will assist managers as they define the frequency and duration components of water quality criteria.

Another consideration for managers is the spatial scale of sampling and analysis. The scale should be chosen carefully, and it should match the scale at which the process or organism of interest operates (Levin 1992). When scales are mismatched or not considered, results may lead to false conclusions (Warner et al. 1995). Although determining appropriate scales is challenging, modeling, like that implemented in this study, can help identify drivers at a given scale, and multiple models targeting different scales can be used to evaluate how patterns change across scales (Levin 1992). Such information will help researchers and managers interpret and apply results more appropriately, even if the ideal scale of sampling or analysis cannot be attained due to logistical constraints.

Finally, it will be challenging to address the issues above without broad-scale, long-term datasets. Such datasets place short-term and localized events in the context of long-term, broad-scale patterns, which can help identify potential drivers (e.g., climatic cycles) and distinguish unusual events that require further evaluation (Magnuson 1995; Burt et al. 2010). Long-term datasets are also the only way to assess processes that occur over a long timescale. For example, groundwater residence times in the Upper Floridan aquifer are expected to delay responses to nutrient management (Katz et al. 1999; Upchurch et al. 2007). Without the information provided by long-term datasets, the effectiveness of a particular management strategy could be underestimated or overvalued, which will lead to poor outcomes (Burt et al. 2010).

APPENDIX
DESCRIPTIVE STATISTICS

Chlorophyll-a

Table A-1. Descriptive statistics for chlorophyll-a (μg L^{-1}) at station 1.

Time Period	Mean	Median	Standard deviation	Minimum	Maximum
1998	1.01	0.44	1.22	0.17	4.30
1999	3.49	1.71	3.75	0.37	12.43
2000	1.65	1.21	1.34	0.23	4.91
2001	1.23	0.66	1.45	0.23	5.27
2002	3.14	2.75	2.12	0.37	6.52
2003	0.53	0.44	0.32	0.23	1.24
2004	1.51	0.55	1.78	0.21	5.16
2005	0.53	0.49	0.29	0.23	1.27
2006	5.85	4.08	8.51	0.21	30.51
2007	2.23	1.23	2.01	0.34	6.14
2008	1.90	0.95	2.03	0.19	5.81
2009	1.35	0.78	1.09	0.34	3.24
2010	1.84	1.28	1.53	0.00	4.92
1998-2010	2.00	0.94	3.07	0.00	30.51

Table A-2. Descriptive statistics for chlorophyll-a (μg L^{-1}) at station 2.

Time Period	Mean	Median	Standard deviation	Minimum	Maximum
1998	1.82	0.91	2.24	0.11	7.07
1999	3.49	3.49	2.56	0.37	7.62
2000	2.20	1.92	1.49	0.37	5.09
2001	1.80	1.21	1.62	0.30	6.25
2002	3.96	2.52	2.62	1.09	9.76
2003	0.81	0.44	1.18	0.23	4.48
2004	1.75	0.64	2.12	0.21	6.56
2005	0.60	0.56	0.29	0.18	1.17
2006	5.86	2.51	6.99	0.21	21.72
2007	3.72	3.32	2.66	0.67	9.87
2008	3.49	3.07	3.08	0.22	8.94
2009	1.58	1.56	1.09	0.34	3.24
2010	3.90	1.75	4.13	0.45	11.59
1998-2010	2.69	1.48	3.19	0.11	21.72

Table A-3. Descriptive statistics for chlorophyll-a ($\mu g\ L^{-1}$) at station 3.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	22.99	18.10	18.07	3.78	63.98
1999	11.75	10.62	7.68	0.30	29.57
2000	13.65	8.78	16.58	1.71	63.53
2001	7.48	5.89	7.25	1.55	27.37
2002	10.02	7.82	7.26	1.71	21.28
2003	14.07	12.48	9.49	1.79	34.78
2004	10.66	4.84	12.17	1.74	41.47
2005	10.39	8.93	5.46	3.49	20.81
2006	15.02	13.08	13.61	1.24	52.82
2007	8.91	5.81	6.81	1.01	22.01
2008	11.33	8.16	8.88	1.34	29.72
2009	9.43	7.37	6.33	2.23	21.67
2010	12.08	7.82	9.35	1.45	26.81
1998-2010	12.15	9.94	10.91	0.30	63.98

Table A-4. Descriptive statistics for chlorophyll-a ($\mu g\ L^{-1}$) at station 4.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	1.82	0.91	2.24	0.11	7.07
1999	3.49	3.49	2.56	0.37	7.62
2000	2.20	1.92	1.49	0.37	5.09
2001	1.80	1.21	1.62	0.30	6.25
2002	3.96	2.52	2.62	1.09	9.76
2003	0.81	0.44	1.18	0.23	4.48
2004	1.75	0.64	2.12	0.21	6.56
2005	0.60	0.56	0.29	0.18	1.17
2006	5.86	2.51	6.99	0.21	21.72
2007	3.72	3.32	2.66	0.67	9.87
2008	3.49	3.07	3.08	0.22	8.94
2009	1.58	1.56	1.09	0.34	3.24
2010	3.90	1.75	4.13	0.45	11.59
1998-2010	2.69	1.48	3.19	0.11	21.72

Table A-5. Descriptive statistics for chlorophyll-a (μg L^{-1}) at station 5.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	11.47	4.96	18.34	0.72	67.40
1999	9.39	9.69	6.95	0.51	22.81
2000	8.19	5.90	7.52	2.69	29.26
2001	8.54	7.59	5.70	2.36	21.48
2002	10.67	11.04	7.06	2.44	23.84
2003	11.72	7.99	11.23	1.55	41.03
2004	6.93	5.36	6.14	0.52	21.82
2005	10.08	6.23	10.83	1.49	33.15
2006	13.58	11.51	10.62	1.06	38.97
2007	11.89	9.18	9.80	3.13	34.45
2008	12.36	8.72	11.77	2.46	43.13
2009	9.04	7.49	6.43	0.67	21.90
2010	10.44	7.37	10.09	0.89	31.28
1998-2010	10.33	8.02	9.76	0.51	67.40

Table A-6. Descriptive statistics for chlorophyll-a (μg L^{-1}) at station 6.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	8.85	8.23	6.61	3.27	26.85
1999	9.13	8.78	6.15	0.51	21.48
2000	11.31	7.44	12.57	2.11	47.28
2001	8.56	8.36	4.83	2.11	18.25
2002	6.04	6.48	3.90	0.30	15.26
2003	13.58	8.32	12.74	2.28	46.07
2004	8.06	5.21	8.20	0.46	30.20
2005	13.54	9.41	10.98	1.74	35.81
2006	12.49	7.41	12.67	2.60	47.11
2007	11.66	8.40	7.62	4.13	27.48
2008	13.03	13.52	4.64	5.03	20.45
2009	10.53	8.35	7.91	2.23	23.69
2010	9.92	8.04	7.32	3.24	25.47
1998-2010	10.52	8.29	8.67	0.30	47.28

Table A-7. Descriptive statistics for chlorophyll-a ($\mu g\ L^{-1}$) at station 7.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	10.59	6.48	9.73	2.03	37.14
1999	5.25	5.36	3.23	0.37	12.82
2000	3.11	2.57	2.01	1.02	8.17
2001	4.53	2.49	5.14	1.55	17.85
2002	3.71	3.36	2.55	0.94	9.38
2003	8.98	5.32	7.29	1.24	24.26
2004	3.41	2.64	2.64	0.93	9.61
2005	9.82	9.34	8.12	1.93	32.32
2006	7.72	6.15	7.69	1.24	29.06
2007	3.40	2.63	2.13	1.12	7.49
2008	8.33	4.47	7.56	1.56	23.91
2009	7.18	5.48	5.24	2.23	21.45
2010	7.22	5.31	4.93	1.68	16.87
1998-2010	6.39	4.12	6.13	0.37	37.14

Table A-8. Descriptive statistics for chlorophyll-a ($\mu g\ L^{-1}$) at station 8.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	9.18	6.71	5.92	2.52	21.69
1999	5.22	4.93	3.76	0.37	13.40
2000	3.89	2.61	2.86	0.87	10.04
2001	3.96	2.90	3.08	0.87	11.38
2002	3.36	2.94	1.91	0.79	6.52
2003	8.06	8.09	5.27	0.94	16.65
2004	3.91	3.04	4.26	0.76	16.40
2005	9.05	8.17	5.50	2.07	19.36
2006	7.28	6.51	4.72	1.45	18.14
2007	4.81	3.07	4.64	1.56	18.16
2008	7.14	4.80	4.95	1.90	14.52
2009	7.33	4.81	5.04	2.57	16.42
2010	6.93	4.97	5.36	0.67	17.54
1998-2010	6.16	4.68	4.80	0.37	21.69

Table A-9. Descriptive statistics for chlorophyll-a ($\mu g\ L^{-1}$) at station 9.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	10.19	7.39	7.27	2.60	22.30
1999	3.45	3.15	2.02	0.44	7.71
2000	4.01	2.77	3.22	1.24	13.11
2001	3.90	3.61	2.75	0.94	10.52
2002	1.95	1.55	1.30	0.44	4.83
2003	8.36	6.44	6.79	1.02	20.06
2004	4.54	3.16	4.82	0.72	18.23
2005	5.70	4.92	2.40	1.96	10.54
2006	7.55	4.85	9.33	1.52	36.49
2007	4.10	3.74	3.53	1.01	14.41
2008	6.43	4.13	5.20	2.12	20.22
2009	6.19	4.25	4.77	1.56	15.42
2010	6.18	5.92	4.59	1.34	16.01
1998-2010	5.58	4.13	5.25	0.44	36.49

Table A-10. Descriptive statistics for chlorophyll-a ($\mu g\ L^{-1}$) at station 10.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	7.94	5.36	6.85	2.52	23.53
1999	3.61	2.32	3.16	1.17	12.63
2000	3.49	2.44	2.63	1.32	10.90
2001	3.23	3.28	1.99	0.79	6.98
2002	2.12	1.40	1.89	0.37	5.98
2003	4.96	4.18	3.91	0.87	12.63
2004	5.71	2.76	8.79	0.46	32.71
2005	6.11	5.47	4.50	1.13	13.91
2006	4.76	4.12	3.67	1.49	14.76
2007	6.31	3.08	11.34	1.01	42.01
2008	5.79	4.47	5.55	1.12	21.00
2009	4.62	3.13	3.20	1.45	11.17
2010	4.33	3.63	2.40	1.34	8.49
1998-2010	4.84	3.24	5.37	0.37	42.01

Total Nitrogen

Table A-11. Descriptive statistics for total nitrogen (µg L-1) at station 1.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	1090.83	1155	250.03	670	1400
1999	998.33	1045	135.84	690	1170
2000	986.67	935	187.73	740	1340
2001	917.50	870	185.63	690	1260
2002	804.17	765	271.58	490	1490
2003	1248.33	1310	152.90	1010	1460
2004	1102.50	1125	195.13	840	1380
2005	1167.50	1145	201.18	880	1500
2006	1020.42	985	160.77	835	1340
2007	991.67	1030	262.15	580	1420
2008	1122.50	1120	171.10	860	1450
2009	1291.67	1280	160.39	1040	1540
2010	1253.33	1275	141.00	980	1440
1998-2010	1076.57	1075	232.29	490	1540

Table A-12. Descriptive statistics for total nitrogen (µg L^{-1}) at station 2.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	991.92	947	211.12	700	1320
1999	850.00	860	188.05	460	1070
2000	862.50	855	205.08	570	1270
2001	841.67	895	269.51	480	1310
2002	582.00	530	208.84	280	1050
2003	1226.75	1225	133.95	1040	1510
2004	1101.67	1065	176.63	860	1390
2005	1145.00	1085	196.49	840	1430
2006	916.67	905	204.15	590	1280
2007	835.83	865	257.03	490	1210
2008	1004.17	1020	290.31	430	1570
2009	1221.67	1190	148.13	960	1470
2010	1166.67	1230	191.42	870	1440
1998-2010	980.50	1000	272.40	280	1570

Table A-13. Descriptive statistics for total nitrogen (μg L^{-1}) at station 3.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	798.50	795	281.34	400	400
1999	511.67	425	227.51	250	250
2000	537.17	520	181.48	260	260
2001	533.33	390	282.69	240	240
2002	521.67	565	167.49	230	230
2003	790.00	730	285.21	460	460
2004	639.17	590	250.43	320	320
2005	711.67	650	247.71	400	400
2006	580.00	560	169.65	340	340
2007	459.17	435	120.56	280	280
2008	574.17	575	253.64	270	270
2009	710.58	695	172.07	507	507
2010	693.33	665	180.72	400	400
1998-2010	620.03	580	239.63	230	1480

Table A-14. Descriptive statistics for total nitrogen (μg L^{-1}) at station 4.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	877.25	810	155.81	730	1180
1999	728.33	685	206.83	440	1050
2000	667.50	650	226.12	370	1030
2001	691.67	640	257.57	340	1130
2002	496.67	510	148.16	270	710
2003	1023.33	1050	254.78	530	1390
2004	807.50	770	259.97	490	1360
2005	989.17	1000	101.93	810	1170
2006	817.50	870	301.85	310	1300
2007	627.50	665	144.61	370	860
2008	726.67	580	338.94	250	1490
2009	1060.00	1110	234.25	590	1300
2010	987.50	1065	265.44	440	1260
1998-2010	807.74	775	278.06	250	1490

Table A-15. Descriptive statistics for total nitrogen (μg L^{-1}) at station 5.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	598.75	558	150.24	400	830
1999	425.83	410	162.45	170	660
2000	586.67	525	245.30	340	1240
2001	542.08	470	222.73	300	1160
2002	564.17	540	187.44	270	970
2003	894.17	815	325.28	370	1350
2004	660.00	555	327.39	240	1360
2005	795.83	770	266.00	430	1220
2006	638.33	570	206.35	410	1040
2007	521.67	475	215.65	260	1030
2008	555.83	585	167.63	260	790
2009	737.00	725	214.07	400	1110
2010	721.67	735	185.02	460	980
1998-2010	659.38	570	391.60	170	4400

Table A-16. Descriptive statistics for total nitrogen (μg L^{-1}) at station 6.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	520.50	483	209.13	190	960
1999	388.33	390	123.86	210	560
2000	494.17	490	140.03	240	720
2001	544.17	545	271.74	240	1130
2002	441.67	355	306.62	230	1380
2003	765.83	760	240.51	410	1140
2004	590.83	460	331.26	270	1270
2005	648.33	740	208.49	220	860
2006	484.17	485	188.27	250	770
2007	400.00	395	103.92	250	630
2008	476.67	495	162.22	240	700
2009	718.67	705	170.92	444	1060
2010	673.33	695	220.67	370	1090
1998-2010	549.74	510	238.23	190	1380

Table A-17. Descriptive statistics for total nitrogen (μg L^{-1}) at station 7.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	385.25	377	140.56	170	730
1999	327.50	335	76.29	170	430
2000	308.33	305	66.99	220	430
2001	326.67	280	136.80	160	580
2002	351.67	330	93.21	220	510
2003	464.17	440	149.88	260	750
2004	420.83	335	203.71	230	900
2005	422.50	385	141.81	260	690
2006	349.17	340	142.60	210	740
2007	293.75	270	74.93	205	440
2008	375.83	330	167.57	180	630
2009	592.50	435	414.23	350	1860
2010	464.17	465	141.39	260	810
1998-2010	390.95	360	183.75	160	1860

Table A-18. Descriptive statistics for total nitrogen (μg L^{-1}) at station 8.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	349.58	343	96.64	210	590
1999	294.17	295	70.64	170	410
2000	315.83	285	90.50	240	520
2001	296.92	290	60.18	193	440
2002	316.67	310	80.26	220	470
2003	496.67	460	137.20	340	770
2004	423.33	345	226.97	250	1030
2005	431.67	430	120.29	240	650
2006	366.67	350	142.53	220	650
2007	269.00	240	110.13	50	420
2008	350.83	315	126.02	170	580
2009	533.33	480	164.83	330	880
2010	492.50	435	163.66	250	760
1998-2010	379.78	350	150.45	50	1030

Table A-19. Descriptive statistics for total nitrogen (μg L^{-1}) at station 9.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	387.08	338	110.05	260	640
1999	279.17	265	86.18	80	410
2000	301.67	275	78.14	220	460
2001	310.83	320	75.97	170	470
2002	310.83	305	90.90	200	530
2003	419.17	465	125.44	210	560
2004	401.67	340	212.81	160	960
2005	453.33	470	141.38	230	710
2006	325.00	270	116.27	220	560
2007	286.67	270	80.04	190	430
2008	363.33	355	120.70	170	620
2009	495.83	425	179.52	280	880
2010	463.33	455	167.08	260	760
1998-2010	369.07	325	142.29	80	960

Table A-20. Descriptive statistics for total nitrogen (μg L^{-1}) at station 10.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	357.25	319	121.58	210	560
1999	275.83	285	88.67	140	410
2000	288.33	285	39.04	220	360
2001	300.00	335	70.45	150	370
2002	296.67	280	65.55	240	430
2003	374.17	400	110.24	200	570
2004	420.83	345	251.95	140	1110
2005	410.83	390	129.79	250	640
2006	302.50	265	109.55	160	470
2007	287.50	265	65.10	200	390
2008	331.17	310	150.11	160	730
2009	461.67	445	102.76	290	720
2010	390.00	315	171.25	230	790
1998-2010	345.90	320	133.99	140	1110

Total Phosphorus

Table A-21. Descriptive statistics for total phosphorus (µg L^{-1}) at station 1.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	91.83	91.0	27.27	51	141
1999	81.08	77.5	22.83	60	148
2000	85.42	70.5	37.56	61	190
2001	91.75	87.0	24.07	66	133
2002	102.42	97.0	26.17	70	140
2003	110.42	101.5	31.09	78	172
2004	118.00	102.0	47.88	64	219
2005	117.33	116.5	22.99	87	174
2006	83.58	80.0	15.15	70	126
2007	95.00	101.5	19.80	65	117
2008	113.58	102.0	47.26	68	251
2009	118.33	114.5	26.26	82	175
2010	100.33	101.0	14.00	74	122
1998-2010	100.70	95.0	31.44	51	251

Table A-22. Descriptive statistics for total phosphorus (µg L^{-1}) at station 2.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	93.08	90.5	28.97	55	147
1999	76.50	74.5	8.16	61	89
2000	81.83	72.5	19.79	64	119
2001	89.42	87.5	22.03	61	137
2002	90.58	86.0	29.12	46	135
2003	109.17	106.0	25.93	75	165
2004	109.92	100.0	42.26	58	211
2005	119.75	115.0	22.59	97	178
2006	83.08	82.0	26.50	23	142
2007	87.75	79.0	23.12	54	128
2008	103.17	97.0	40.04	46	202
2009	115.08	107.0	22.17	91	154
2010	93.17	98.5	13.26	65	110
1998-2010	96.35	93.0	28.64	23	211

Table A-23. Descriptive statistics for total phosphorus (µg L^{-1}) at station 3.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	84.58	75.5	30.14	47	155
1999	55.67	48.0	30.94	9	119
2000	43.50	42.0	16.46	21	79
2001	47.83	38.0	29.38	19	116
2002	53.75	53.5	18.73	28	79
2003	83.50	81.5	39.91	22	148
2004	69.83	67.5	35.23	23	138
2005	81.50	72.5	43.21	36	178
2006	61.83	55.0	19.60	34	96
2007	44.67	41.0	20.21	14	73
2008	48.25	51.0	22.40	17	84
2009	66.58	68.0	33.33	25	142
2010	66.42	65.5	29.10	32	132
1998-2010	62.15	55.5	31.67	9	178

Table A-24. Descriptive statistics for total phosphorus (µg L^{-1}) at station 4.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	91.67	86.5	26.64	64	148
1999	70.17	68.5	16.35	28	96
2000	65.42	65.0	23.40	31	116
2001	70.17	72.5	28.22	31	133
2002	68.08	60.5	26.96	28	112
2003	99.25	100.5	33.92	43	159
2004	92.25	79.0	35.14	54	163
2005	111.67	105.0	27.87	90	189
2006	80.08	80.5	25.42	38	135
2007	66.00	66.0	19.01	26	95
2008	78.58	69.5	56.07	19	225
2009	96.50	104.0	24.63	44	128
2010	86.67	91.0	23.90	37	122
1998-2010	82.81	79.0	31.97	19	225

Table A-25. Descriptive statistics for total phosphorus ($\mu g\ L^{-1}$) at station 5.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	74.17	67.0	34.05	38	146
1999	45.08	44.0	14.00	28	68
2000	55.17	53.5	24.47	25	117
2001	56.00	53.0	23.24	29	116
2002	72.58	74.0	17.32	39	97
2003	94.83	73.0	52.04	23	207
2004	75.58	65.5	45.84	20	165
2005	91.58	83.5	41.85	38	199
2006	68.50	65.5	24.56	19	107
2007	55.17	48.5	26.79	20	92
2008	57.75	61.0	22.63	21	89
2009	70.17	65.0	33.17	20	139
2010	66.50	72.0	26.61	18	114
1998-2010	67.93	63.0	33.41	18	207

Table A-26. Descriptive statistics for total phosphorus ($\mu g\ L^{-1}$) at station 6.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	64.83	54.5	40.22	19	144
1999	36.50	37.5	6.71	26	51
2000	43.58	39.0	14.22	22	66
2001	49.83	43.0	30.04	23	116
2002	42.50	40.0	20.89	18	102
2003	74.25	72.5	30.64	24	120
2004	67.08	48.0	47.10	24	156
2005	70.08	70.5	31.82	22	129
2006	51.67	57.0	20.38	14	76
2007	44.67	45.0	19.35	19	75
2008	46.58	43.0	21.30	20	88
2009	62.58	60.5	28.06	23	113
2010	57.00	44.5	27.44	27	101
1998-2010	54.71	46.0	29.26	14	156

Table A-27. Descriptive statistics for total phosphorus (µg L^{-1}) at station 7.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	37.25	37.0	14.86	11	62
1999	25.50	22.5	11.49	9	45
2000	18.83	17.0	6.60	10	30
2001	20.83	19.0	8.36	13	37
2002	28.75	29.5	11.05	16	51
2003	37.25	41.0	15.76	14	60
2004	42.25	34.5	29.73	16	120
2005	39.50	34.0	19.46	13	74
2006	33.17	26.5	18.96	15	76
2007	19.08	16.0	9.14	10	43
2008	26.00	21.5	14.04	11	48
2009	43.83	29.0	48.72	19	194
2010	31.67	25.0	16.62	13	71
1998-2010	31.07	26.0	21.26	9	194

Table A-28. Descriptive statistics for total phosphorus (µg L^{-1}) at station 8.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	33.42	33.5	10.94	14	52
1999	22.42	20.0	8.62	13	44
2000	21.75	18.5	8.71	14	40
2001	19.33	19.5	5.21	10	30
2002	25.58	24.5	8.71	14	39
2003	43.92	37.0	22.90	13	97
2004	42.25	32.5	30.91	16	124
2005	40.33	34.5	20.15	15	70
2006	34.08	32.0	15.20	16	63
2007	22.25	21.0	9.85	11	39
2008	22.58	17.5	11.75	9	47
2009	39.58	31.5	22.52	22	95
2010	35.75	34.5	15.04	18	68
1998-2010	31.02	26.0	17.89	9	124

Table A-29. Descriptive statistics for total phosphorus (µg L^{-1}) at station 9.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	37.00	35.0	12.81	23	70
1999	19.92	21.5	7.29	6	30
2000	20.92	19.5	7.49	12	36
2001	20.58	20.0	3.06	16	26
2002	19.08	17.5	7.59	10	36
2003	34.00	36.5	14.24	12	62
2004	38.25	31.5	27.54	14	113
2005	42.25	36.5	20.71	15	80
2006	30.33	27.0	14.98	13	62
2007	20.42	17.5	9.49	10	38
2008	23.33	19.5	11.87	9	47
2009	34.83	29.0	22.74	13	97
2010	31.50	29.0	14.72	13	63
1998-2010	28.65	24.5	16.44	6	113

Table A-30. Descriptive statistics for total phosphorus (µg L^{-1}) at station 10.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	35.33	33.0	15.93	16	71
1999	18.58	18.0	6.56	8	32
2000	19.00	18.5	4.79	12	26
2001	19.00	19.0	6.32	10	32
2002	18.58	15.5	7.24	10	34
2003	25.50	29.5	9.93	9	37
2004	38.67	25.5	34.22	9	136
2005	39.17	34.0	19.41	15	78
2006	23.75	20.0	9.46	14	45
2007	18.67	16.5	7.27	11	33
2008	20.92	17.5	12.44	9	56
2009	30.17	28.0	11.00	16	52
2010	25.92	22.5	18.11	13	79
1998-2010	25.63	22.0	15.99	8	136

Water Color

Table A-31. Descriptive statistics for color (PCU) at station 1.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	127.67	82.5	110.00	15	340
1999	36.58	31.0	27.13	12	110
2000	53.17	35.0	48.00	14	189
2001	92.67	73.5	77.68	11	243
2002	48.17	24.5	52.98	13	195
2003	163.75	130.5	93.65	60	307
2004	175.84	121.0	177.76	17	540
2005	158.07	157.5	89.62	42	312
2006	57.25	20.0	69.98	10	228
2007	57.33	56.5	49.77	7	167
2008	145.50	122.0	104.14	19	306
2009	141.67	115.5	109.53	30	357
2010	100.92	105.5	72.00	11	246
1998-2010	104.51	73.5	99.92	7	540

Table A-32. Descriptive statistics for color (PCU) at station 2.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	217.42	167.5	149.13	79	522
1999	32.50	30.5	18.42	18	86
2000	49.92	36.0	39.94	16	153
2001	85.17	58.0	77.30	13	231
2002	42.83	23.5	45.77	12	173
2003	164.00	131.0	96.58	60	321
2004	171.85	117.5	177.41	18	532
2005	155.66	150.5	88.27	42	318
2006	66.08	23.5	85.92	11	242
2007	51.92	48.5	40.31	13	133
2008	125.50	96.0	94.76	22	312
2009	127.33	102.0	96.54	31	339
2010	107.75	102.0	95.63	10	354
1998-2010	107.53	72.0	107.18	10	532

Table A-33. Descriptive statistics for color (PCU) at station 3.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	102.17	61.0	77.30	40	267
1999	29.25	27.5	13.48	10	54
2000	20.25	15.0	11.98	11	54
2001	34.17	16.5	38.33	9	140
2002	27.83	15.0	42.65	7	162
2003	75.92	52.0	62.98	18	195
2004	82.01	35.0	116.41	17	432
2005	87.85	52.5	76.38	16	234
2006	42.33	29.0	40.37	10	140
2007	15.42	14.5	6.73	8	31
2008	47.83	26.5	47.53	13	160
2009	67.33	39.0	91.27	23	351
2010	53.42	39.0	47.40	9	166
1998-2010	52.75	26.0	63.98	7	432

Table A-34. Descriptive statistics for color (PCU) at station 4.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	161.58	97.5	122.06	63	422
1999	38.33	31.5	21.97	12	86
2000	33.75	28.5	19.85	13	77
2001	60.67	29.5	64.03	11	216
2002	25.08	18.5	23.72	7	98
2003	144.58	114.5	106.30	27	318
2004	134.95	78.6	148.02	20	512
2005	144.89	118.5	92.63	37	314
2006	64.42	26.0	80.65	10	242
2007	31.08	17.0	25.97	14	96
2008	88.67	41.0	94.12	19	312
2009	109.50	100.5	89.28	30	348
2010	90.00	87.0	69.52	12	240
1998-2010	86.73	46.5	93.07	7	512

Table A-35. Descriptive statistics for color (PCU) at station 5.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	113.50	68.0	86.05	45	297
1999	26.67	21.0	14.76	10	60
2000	25.33	20.0	13.12	11	54
2001	38.67	21.0	36.06	8	113
2002	24.25	16.0	25.52	7	95
2003	104.08	72.0	99.78	16	276
2004	121.41	36.0	167.52	15	504
2005	111.33	84.0	91.38	20	308
2006	47.17	23.0	58.23	10	206
2007	17.33	17.0	7.44	7	31
2008	46.92	32.0	36.89	15	139
2009	81.25	44.5	110.44	18	378
2010	72.25	44.0	69.72	10	236
1998-2010	63.86	31.0	83.15	7	504

Table A-36. Descriptive statistics for color (PCU) at station 6.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	94.33	56.5	71.74	37	247
1999	23.75	20.0	11.45	12	50
2000	20.50	19.0	7.69	7	34
2001	27.50	18.0	27.72	6	99
2002	20.25	11.5	22.21	5	81
2003	89.75	60.0	87.08	18	277
2004	101.85	26.9	155.65	14	504
2005	66.37	56.5	44.52	14	162
2006	36.50	24.0	42.88	8	158
2007	16.67	15.0	8.22	6	33
2008	39.08	29.5	25.95	14	100
2009	69.08	32.5	89.52	15	282
2010	53.33	35.0	53.83	8	178
1998-2010	50.69	26.0	68.51	5	504

Table A-37. Descriptive statistics for color (PCU) at station 7.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	54.33	32.5	41.50	21	143
1999	15.75	17.0	6.68	5	29
2000	9.58	9.0	3.42	5	15
2001	12.50	8.5	10.08	4	32
2002	10.17	8.5	6.51	4	29
2003	42.33	32.5	35.09	9	109
2004	64.45	17.0	109.42	10	392
2005	42.38	29.5	36.69	6	115
2006	26.17	14.0	38.46	5	144
2007	9.67	9.0	4.81	4	19
2008	23.92	18.0	18.43	4	63
2009	33.58	18.5	49.26	13	189
2010	28.25	25.0	27.99	4	107
1998-2010	28.70	15.5	42.74	4	392

Table A-38. Descriptive statistics for color (PCU) at station 8.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	52.08	31.0	39.88	20	137
1999	15.50	15.0	7.28	6	28
2000	9.83	9.0	4.13	5	18
2001	11.58	10.5	8.82	3	35
2002	8.83	8.0	5.17	3	23
2003	49.08	32.0	50.26	7	186
2004	61.77	17.7	107.50	7	380
2005	35.83	21.5	34.12	9	119
2006	26.67	13.0	37.53	4	139
2007	8.33	8.0	3.50	2	14
2008	20.92	15.0	15.47	7	63
2009	41.00	18.5	64.75	10	243
2010	31.08	22.0	29.38	5	105
1998-2010	28.66	15.0	44.61	2	380

Table A-39. Descriptive statistics for color (PCU) at station 9.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	49.00	29.5	37.11	19	128
1999	13.67	13.5	5.87	7	26
2000	9.83	9.5	3.64	5	16
2001	11.83	11.0	7.99	4	33
2002	7.67	6.0	6.11	2	24
2003	29.25	29.5	21.68	7	85
2004	52.94	21.0	80.19	9	276
2005	39.15	19.5	43.33	8	156
2006	25.83	12.0	35.89	5	134
2007	7.33	6.5	3.20	3	14
2008	20.17	18.0	12.88	9	57
2009	39.25	18.0	65.70	11	243
2010	28.25	17.0	27.86	4	95
1998-2010	25.71	15.0	37.75	2	276

Table A-40. Descriptive statistics for color (PCU) at station 10.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	42.25	25.5	32.13	16	111
1999	13.75	13.5	5.05	7	22
2000	8.50	7.0	4.10	4	19
2001	8.17	7.0	3.21	4	14
2002	6.17	5.0	4.88	3	19
2003	22.67	18.5	21.77	7	88
2004	64.15	19.1	122.19	6	436
2005	33.08	17.0	37.33	7	140
2006	16.75	12.0	15.57	4	60
2007	7.00	6.5	3.30	3	15
2008	18.58	14.0	17.04	8	71
2009	29.00	17.5	35.17	4	129
2010	19.08	14.0	15.05	2	47
1998-2010	22.24	13.0	40.87	2	436

Water Temperature

Table A-41. Descriptive statistics for water temperature (°C) at station 1.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	21.95	21.90	4.95	15.00	28.30
1999	23.31	23.55	5.13	15.70	30.50
2000	23.43	22.95	4.78	15.80	29.20
2001	22.89	23.40	5.26	14.12	29.26
2002	23.57	25.22	4.68	16.21	28.85
2003	21.51	22.13	5.08	11.17	27.45
2004	22.67	23.81	6.01	14.09	30.16
2005	21.59	20.97	3.76	16.58	27.92
2006	23.40	24.65	4.85	15.06	30.15
2007	23.36	23.44	4.60	16.05	29.16
2008	22.14	20.63	5.23	15.25	29.51
2009	22.24	23.04	4.87	13.87	28.07
2010	22.14	22.05	5.98	11.92	30.00
1998-2010	22.63	22.70	4.90	11.17	30.50

Table A-42. Descriptive statistics for water temperature (°C) at station 2.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	22.40	22.75	5.52	15.00	30.53
1999	22.44	20.85	5.42	15.10	29.80
2000	23.01	22.15	5.11	15.90	29.50
2001	22.37	23.07	5.71	12.52	29.10
2002	23.26	24.80	4.72	15.57	29.05
2003	21.39	21.86	4.89	11.39	27.40
2004	22.71	23.88	5.57	14.39	29.14
2005	21.43	20.66	4.12	15.75	27.78
2006	23.17	24.65	5.74	12.86	31.53
2007	23.85	23.99	5.28	15.96	32.31
2008	22.43	21.01	5.65	13.79	30.49
2009	22.40	22.72	5.18	13.86	29.74
2010	22.04	22.37	6.10	12.08	29.94
1998-2010	22.53	22.20	5.17	11.39	32.31

Table A-43. Descriptive statistics for water temperature (°C) at station 3.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	23.25	23.35	6.61	15.20	31.40
1999	22.08	21.35	7.02	12.30	31.80
2000	22.68	22.45	6.51	11.90	30.50
2001	22.50	22.97	6.45	11.63	29.82
2002	23.01	25.22	6.03	13.85	30.43
2003	21.81	22.48	7.08	9.85	29.95
2004	23.01	25.58	6.92	11.82	30.20
2005	21.85	21.08	5.98	12.86	29.87
2006	23.27	24.89	5.79	15.07	31.71
2007	23.24	23.08	5.74	14.27	31.85
2008	21.56	20.46	6.84	11.80	30.56
2009	23.02	24.94	6.76	10.25	30.85
2010	22.09	21.48	6.79	13.51	31.20
1998-2010	22.57	23.44	6.29	9.85	31.85

Table A-44. Descriptive statistics for water temperature (°C) at station 4.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	22.32	21.85	6.15	15.10	30.50
1999	21.63	20.70	6.64	10.60	30.20
2000	22.32	21.60	6.72	9.35	31.60
2001	21.97	23.40	5.90	12.38	28.57
2002	22.68	24.87	5.03	14.60	29.23
2003	20.83	22.08	6.11	8.72	27.89
2004	23.06	23.94	5.50	14.62	29.82
2005	21.34	21.00	4.85	13.90	28.20
2006	22.50	24.09	5.63	12.92	31.09
2007	22.81	22.96	5.66	14.32	31.54
2008	21.83	21.79	6.61	11.79	30.07
2009	22.45	22.49	6.14	10.75	30.20
2010	21.77	21.77	6.71	12.02	31.10
1998-2010	22.12	22.35	5.80	8.72	31.60

Table A-45. Descriptive statistics for water temperature (°C) at station 5.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	23.42	24.20	6.26	15.00	30.84
1999	22.96	22.40	6.90	13.10	32.40
2000	23.34	23.90	6.21	12.87	31.00
2001	22.73	24.16	6.33	11.55	29.99
2002	23.73	25.51	5.93	14.81	30.87
2003	21.91	22.74	6.97	10.67	30.00
2004	23.09	24.61	6.56	12.91	30.61
2005	22.11	21.14	5.97	13.20	30.97
2006	23.46	24.39	5.83	14.20	32.07
2007	23.36	23.40	5.64	14.66	31.50
2008	21.89	20.52	6.48	12.12	29.72
2009	22.96	23.62	6.28	11.29	30.79
2010	22.14	22.20	6.64	13.09	31.17
1998-2010	22.85	23.89	6.10	10.67	32.40

Table A-46. Descriptive statistics for water temperature (°C) at station 6.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	23.27	24.10	6.58	14.50	30.65
1999	23.15	22.70	6.57	13.30	31.80
2000	23.13	23.40	6.64	11.11	30.60
2001	22.81	23.62	6.29	12.21	30.50
2002	23.45	25.14	5.72	14.40	30.72
2003	22.35	22.85	7.19	11.26	30.56
2004	23.06	24.83	7.06	11.10	30.80
2005	22.57	21.96	6.36	13.18	31.20
2006	23.37	24.01	6.04	14.17	32.49
2007	23.48	23.31	5.58	14.46	31.66
2008	21.59	20.46	6.57	11.94	30.23
2009	23.12	25.64	6.65	10.61	30.38
2010	22.23	22.14	6.81	12.32	31.29
1998-2010	22.89	23.77	6.25	10.61	32.49

Table A-47. Descriptive statistics for water temperature (°C) at station 7.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	23.41	23.30	6.96	13.70	32.50
1999	22.18	20.90	6.87	12.00	30.80
2000	22.66	22.30	6.21	13.80	30.80
2001	22.29	22.19	6.32	11.94	30.19
2002	22.52	24.66	5.84	14.30	29.26
2003	21.85	23.24	6.87	10.33	29.79
2004	22.76	24.76	7.03	11.57	30.53
2005	21.95	20.97	6.03	12.00	31.20
2006	23.05	24.95	6.09	13.91	31.51
2007	22.99	22.95	5.84	13.86	31.03
2008	22.07	20.41	6.54	12.34	30.41
2009	22.58	24.57	7.08	10.29	30.39
2010	22.42	22.21	6.99	11.87	32.48
1998-2010	22.52	23.11	6.29	10.29	32.50

Table A-48. Descriptive statistics for water temperature (°C) at station 8.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	23.64	23.75	6.67	14.50	31.57
1999	22.38	21.50	6.92	12.10	31.30
2000	22.79	22.65	6.07	14.10	30.10
2001	22.50	22.25	6.50	12.75	31.41
2002	22.55	24.48	5.67	14.27	29.62
2003	21.96	23.43	6.73	10.72	29.95
2004	23.00	25.27	7.19	11.68	31.18
2005	21.91	21.01	6.13	12.50	30.98
2006	23.23	24.70	5.94	14.10	32.07
2007	23.17	23.29	5.77	13.91	30.96
2008	22.02	20.50	6.58	11.62	30.50
2009	22.84	25.15	6.96	10.51	30.48
2010	22.36	22.21	7.08	11.47	32.25
1998-2010	22.64	23.43	6.26	10.51	32.25

Table A-49. Descriptive statistics for water temperature (°C) at station 9.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	23.71	23.70	6.62	14.50	31.54
1999	22.68	21.70	6.71	12.50	31.00
2000	22.84	22.53	6.11	14.30	30.50
2001	22.96	23.13	6.61	12.15	31.47
2002	23.23	25.49	5.94	14.38	30.76
2003	22.92	23.76	7.22	11.19	31.92
2004	23.20	25.07	7.09	11.99	31.41
2005	22.17	21.25	6.31	12.79	31.28
2006	23.31	24.77	6.00	13.71	32.11
2007	23.22	23.40	5.62	13.94	30.87
2008	22.18	20.56	6.53	12.58	31.09
2009	22.88	25.45	6.95	10.96	30.53
2010	22.36	22.12	7.05	11.17	31.78
1998-2010	22.90	23.75	6.30	10.96	32.11

Table A-50. Descriptive statistics for water temperature (°C) at station 10.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	23.48	23.35	6.60	14.50	31.06
1999	22.69	22.00	6.60	13.10	31.10
2000	22.94	22.73	6.09	14.40	30.50
2001	22.88	23.25	6.58	11.40	30.75
2002	24.13	25.66	6.97	14.49	30.47
2003	22.86	23.99	7.10	11.17	30.92
2004	23.19	24.97	7.05	12.62	31.13
2005	22.41	21.44	6.41	12.58	31.09
2006	23.38	23.75	5.70	16.11	31.79
2007	23.24	22.95	5.82	14.06	31.11
2008	22.14	20.30	6.52	12.14	30.65
2009	22.58	23.83	6.75	10.66	30.44
2010	22.43	22.29	7.02	11.73	31.84
1998-2010	22.95	23.61	6.33	10.66	32.50

Salinity

Table A-51. Descriptive statistics for salinity (ppt) at station 1.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	0.05	0.00	0.09	0.00	0.20
1999	0.18	0.20	0.06	0.10	0.30
2000	0.18	0.20	0.04	0.10	0.20
2001	0.20	0.12	0.23	0.01	0.89
2002	0.22	0.20	0.11	0.09	0.42
2003	0.17	0.14	0.17	0.03	0.70
2004	0.12	0.12	0.05	0.04	0.17
2005	0.17	0.13	0.17	0.03	0.70
2006	0.23	0.18	0.21	0.00	0.86
2007	0.19	0.18	0.06	0.10	0.36
2008	0.14	0.13	0.08	0.03	0.35
2009	0.12	0.13	0.04	0.03	0.18
2010	0.13	0.13	0.05	0.03	0.19
1998-2010	0.16	0.14	0.13	0.00	0.89

Table A-52. Descriptive statistics for salinity (ppt) at station 2.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	1.03	0.05	3.05	0.00	10.70
1999	2.00	0.50	2.79	0.10	8.40
2000	2.45	1.43	2.33	0.30	6.40
2001	3.28	0.37	4.83	0.11	14.57
2002	2.22	2.65	1.78	0.10	4.87
2003	0.13	0.14	0.06	0.03	0.20
2004	0.57	0.16	1.00	0.04	3.22
2005	0.33	0.15	0.36	0.03	1.01
2006	4.58	0.75	7.07	0.05	20.64
2007	5.04	2.98	5.57	0.13	17.14
2008	2.77	0.79	4.23	0.04	14.83
2009	1.57	0.37	2.21	0.03	6.98
2010	2.43	0.16	3.65	0.04	10.14
1998-2010	2.18	0.30	3.74	0.00	20.64

Table A-53. Descriptive statistics for salinity (ppt) at station 3.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	14.22	14.95	8.79	2.00	26.30
1999	22.47	23.35	5.25	14.40	34.30
2000	24.28	24.95	3.96	16.20	28.60
2001	23.46	25.61	6.12	9.46	29.79
2002	23.30	24.36	3.41	13.33	25.50
2003	18.86	19.44	6.26	10.41	28.70
2004	18.51	18.70	5.24	5.50	24.31
2005	16.08	15.88	9.78	0.09	30.14
2006	18.98	17.63	7.70	3.06	29.26
2007	26.16	26.59	2.46	21.66	29.77
2008	21.93	22.73	4.33	14.87	28.10
2009	17.47	19.09	6.23	2.48	25.19
2010	17.08	18.85	8.11	5.42	26.31
1998-2010	20.21	22.25	7.01	0.09	34.30

Table A-54. Descriptive statistics for salinity (ppt) at station 4.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	4.68	1.25	7.44	0.00	22.90
1999	6.73	5.30	6.81	0.60	26.90
2000	14.72	15.00	6.01	3.40	22.76
2001	12.85	15.21	7.64	0.15	23.29
2002	17.92	20.56	6.99	2.26	25.24
2003	7.43	3.69	8.22	0.04	20.93
2004	7.55	5.05	6.50	0.05	19.25
2005	5.92	4.75	6.02	0.09	15.70
2006	8.79	4.71	9.00	0.06	21.18
2007	17.38	17.12	8.56	6.58	28.03
2008	13.88	13.28	9.20	0.95	27.87
2009	6.94	5.39	6.27	0.04	18.18
2010	7.09	4.17	7.80	0.10	22.28
1998-2010	10.14	7.68	8.44	0.00	28.03

Table A-55. Descriptive statistics for salinity (ppt) at station 5.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	13.70	16.85	9.55	0.00	25.70
1999	21.77	23.85	6.46	10.30	30.60
2000	20.54	19.10	3.76	13.80	26.80
2001	20.96	22.59	4.67	14.02	28.17
2002	22.75	23.67	3.71	13.52	26.46
2003	14.69	17.07	10.95	0.30	29.61
2004	15.72	17.82	7.60	0.52	25.27
2005	14.48	18.15	11.09	0.04	26.85
2006	17.42	18.76	5.43	9.02	27.80
2007	23.81	24.56	5.20	15.49	29.77
2008	20.84	20.63	4.11	14.20	28.36
2009	16.89	17.83	5.92	0.21	24.67
2010	16.61	18.14	6.67	2.24	25.19
1998-2010	18.48	19.30	7.49	0.00	30.60

Table A-56. Descriptive statistics for salinity (ppt) at station 6.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	16.28	19.95	9.03	2.00	26.00
1999	23.89	25.10	5.82	11.70	29.70
2000	23.41	22.45	2.62	20.77	28.40
2001	23.79	25.34	5.36	14.56	30.55
2002	25.57	26.22	4.17	16.77	32.24
2003	17.52	18.51	7.67	3.64	27.73
2004	19.26	22.32	7.25	1.21	26.16
2005	19.06	19.93	5.87	4.83	28.45
2006	20.85	21.34	7.00	10.76	29.15
2007	25.35	26.03	3.83	17.91	29.63
2008	20.43	20.42	4.69	11.95	26.96
2009	19.96	20.63	5.22	5.87	25.40
2010	18.59	20.81	7.62	3.35	28.01
1998-2010	21.07	22.40	6.55	1.21	32.24

Table A-57. Descriptive statistics for salinity (ppt) at station 7.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	23.56	24.8	4.55	15.00	32.20
1999	28.05	28.8	5.87	15.50	34.60
2000	30.43	31.0	1.94	27.40	33.00
2001	29.43	29.9	3.00	23.01	33.60
2002	28.26	29.3	2.71	23.26	31.23
2003	24.12	23.7	4.73	17.30	30.82
2004	24.70	25.2	3.44	17.53	29.64
2005	24.35	24.8	5.09	17.11	32.70
2006	26.74	28.5	4.75	16.43	31.79
2007	29.87	30.8	3.00	24.86	33.24
2008	26.07	26.2	4.07	17.12	31.81
2009	24.67	25.1	4.29	13.16	30.22
2010	23.63	24.1	6.06	9.12	29.94
1998-2010	26.45	27.16	4.77	9.12	34.60

Table A-58. Descriptive statistics for salinity (ppt) at station 8.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	23.67	24.50	6.85	8.00	31.40
1999	28.18	29.15	5.31	15.40	34.00
2000	30.14	30.44	1.68	26.80	32.40
2001	28.85	29.38	3.49	22.26	33.10
2002	28.51	30.07	3.18	21.10	31.65
2003	23.20	23.05	5.54	14.72	31.41
2004	24.96	26.18	5.37	10.00	30.48
2005	25.50	26.67	4.68	16.74	30.90
2006	26.20	27.01	4.02	19.32	31.05
2007	29.98	30.57	2.56	25.89	32.93
2008	25.85	26.51	3.83	17.33	31.41
2009	24.72	26.61	5.23	10.00	29.41
2010	24.33	25.01	4.27	12.96	29.28
1998-2010	26.47	27.64	4.91	8.00	34.00

Table A-59. Descriptive statistics for salinity (ppt) at station 9.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	24.03	24.50	5.66	12.00	30.80
1999	29.84	30.55	4.18	20.70	34.10
2000	30.39	30.04	2.21	25.60	33.58
2001	28.81	28.99	3.06	22.06	32.18
2002	29.95	30.62	2.80	22.96	32.84
2003	25.10	24.55	4.05	19.68	31.45
2004	25.62	26.84	4.22	14.40	29.03
2005	25.76	27.09	5.15	13.03	30.77
2006	26.34	27.34	4.13	18.87	31.81
2007	30.19	30.61	2.37	26.50	33.82
2008	26.33	26.57	3.07	19.37	31.18
2009	25.01	25.57	5.42	9.01	31.61
2010	24.78	26.20	4.43	14.50	29.48
1998-2010	27.09	27.96	4.51	9.01	34.10

Table A-60. Descriptive statistics for salinity (ppt) at station 10.

Time period	Mean	Median	Standard deviation	Minimum	Maximum
1998	25.07	25.40	6.21	10.00	32.00
1999	30.38	31.25	4.08	20.00	34.50
2000	31.00	30.70	2.19	27.03	34.30
2001	29.61	30.24	2.05	26.02	32.36
2002	30.18	31.22	2.72	24.54	33.13
2003	26.91	26.62	2.82	21.99	31.51
2004	25.49	27.45	6.14	8.49	30.26
2005	26.50	28.32	5.93	9.29	31.41
2006	27.31	28.21	3.89	20.27	33.03
2007	30.94	32.13	2.33	27.12	33.46
2008	26.42	27.23	3.26	20.62	30.27
2009	26.26	25.81	2.89	21.04	32.06
2010	25.07	26.16	5.10	11.47	30.16
1998-2010	27.78	28.94	4.51	8.49	34.50

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BIOGRAPHICAL SKETCH

Andrea Krzystan was born in Panama City, Florida but grew up in Severna Park, Maryland. The proximity to the Chesapeake Bay and the Smithsonian museums in Washington D.C. inspired a passion for science and nature from a young age. She pursued these interests by returning to Florida and studying Marine Science at Eckerd College in St. Petersburg. The unique education provided by Eckerd College included extensive field and laboratory experiences, a variety of research internships, and other opportunities including playing varsity volleyball and participating in many service learning experiences. Each of these experiences prepared her for life after graduation.

After graduation in 2006, Andrea worked as a Water Quality Specialist for the Florida Department of Environmental Protection at Tampa Bay Aquatic Preserves. During her time there, Andrea applied her experience to develop a water quality monitoring program and also developed new interests in resource management and geographic information systems. After three years, she decided to further her education in marine ecology by returning to graduate school and joined the lab of Dr. Tom Frazer at the University of Florida in 2009.

Through her graduate experience, Andrea's interests in marine ecology and resource management have been strengthened. After earning her master's degree, Andrea plans to pursue a career that combines these interests.