ABSTRACT

Title of Thesis: REMOVAL OF WASTEWATER NITROGEN AND PHOSPHORUS BY AN OLIGOHALINE MARSH

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Wetlands potentially contribute to water quality improvement. The objective of this thesis was to determine the role of wastewater processing by oligohaline Council Creek marsh (0.16 km^2), which receives wastewater from Easton, MD. Tidal flushing, nutrient distributions, plant uptake, and sediment burial were examined as removal mechanisms. This marsh seems to be little affected by, and has a small impact on wastewater. Wastewater flow can raise mean water levels by 3 cm in addition to seasonal changes of 30 cm induced by thermal expansion and contraction of seawater. During the ~3 d flushing time of water, nutrients passing through the marsh experienced small transformations compared with 25–75% dilution of wastewater in the marsh by Choptank River water, during summer high sea level stands. During the growing season, marsh plants intercepted 30–44% N and 11–17% P, while sediment burial appeared to remove 27% N and 6% P, annually.

REMOVAL OF WASTEWATER NITROGEN AND PHOSPHORUS BY AN

OLIGOHALINE MARSH

by

Jason John Traband

Thesis submitted to the faculty of the Graduate School of the University of Maryland at College Park in partial fulfillment of the degree requirements for the degree of
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Chapter 1

GENERAL INTRODUCTION

Anthropogenic activity has significantly impacted coastal and estuarine waters. Nitrogen (N) and phosphorus (P) inputs from both diffuse and point sources are the major causes of degraded water quality. Eutrophication resulting from these inputs has had severe consequences, including accumulation of phytoplankton and epiphyte biomass, bottom water hypoxia, decreased water clarity, and loss of submerged aquatic plants (Fisher et al 1988, Smith et al. 1999).

Increased diffuse and point source nutrient inputs to estuarine waters are the result of increased anthropogenic disturbance in the upstream watershed (Harding 1994, Lee et al 2000). Rising human populations and subsequent land use changes are major factors causing eutrophication (Beaulac and Reckhow 1982, D'Elia et al. 1986). Inputs of diffuse source nutrient loading primarily result from agricultural use of fertilizers (Bohlke and Denver 1995) and intensive animal operations (Sallade and Sims 1997), but also include atmospheric deposition (Magnien et al. 1992, Paerl et al. 2002) and residential septic systems (McClelland et al. 1997). Nutrient inputs to coastal waters are also linked to point sources such as municipal and industrial wastewaters (Peierls et al. 1991).

One important point source of N and P is the discharge from municipal wastewater treatment plants. Wastewater disposal has been a local public health and aesthetics issue since Greek and Roman times (Stoddard et al. 2002). Increased human population densities and development of urban areas necessitated the development of the urban water cycle, which consists of clean water supply demands and return of wastewater flows. Until about 100 years ago, the primary goal of wastewater disposal was removal from the municipality to reduce the spread of water-borne disease (Stoddard et al. 2002), and this typically meant that wastewater was discharged to rivers and estuaries. With continually rising human populations, degradation of the water supply of downstream municipalities soon became an issue (Stoddard et al. 2002). To address these problems, municipal wastewater treatment methods were developed and implemented for the purpose of reducing the discharge of contaminants and pathogens. However, the increased costs of more effective yet increasingly complex wastewater treatment are important limitations on the widespread use of advanced treatment (Kadlec and Knight 1996).

Wastewater treatment began to be widely implemented around 1900. With the goal of removing solids to reduce contamination of waters downstream of urban centers, primary treatment methods employ settling and screening of wastewater to remove solids, but suspended organic matter, often with a high biological oxygen demand (BOD), is still discharged in effluent water. By 1915, secondary treatment of wastewater, utilizing biological processes such as bacterial metabolism to metabolize this residual organic matter (remove the BOD) had been developed. However, it was not until the Clean Water Act of 1972 that secondary treatment was mandated (Stoddard et al. 2002). Advanced or tertiary wastewater treatment methods greatly enhance wastewater treatment by removing the high concentrations of nutrients remaining in the human waste (e.g. N and P) that remain in solution after secondary treatment. However, improved methods are costly and require high maintenance. In contrast, natural treatment methods such as treatment wetlands that employ natural physical, chemical, and biological processes may provide a more cost-effective means to treat wastewater (Dolan et al. 1981, Kadlec and Knight 1996, Smith et al. 2000, Sartoris et al. 2000).

Wetlands are increasingly used as an alternative treatment method for wastewater (Kadlec and Knight 1996, Kadlec and Reddy 2001). Because they are high productivity ecosystems (Kadlec and Knight 1996, Mitsch and Gosselink 2000), wetlands have the potential to process and remove wastewater contaminants via macrophytic nutrient uptake, burial in sediments, and denitrification. While constructed wetlands are often used for wastewater treatment (Peterson and Teal 1996, Sartoris et al. 2000, Smith et al. 2000, and Spieles and Mitsch 2000), natural wetlands may also have an important role in wastewater processing. One such example is the discharge of wastewater to oligohaline wetlands.

Oligohaline marshes are located at the upper end of the estuarine gradient with salinities ranging from 0.5-5psu. In the Chesapeake Bay region, semidiurnal lunar tidal ranges are approximately 0.5 m but highly variable due to meteorological forcing. Although tidal flushing can regularly reverse flow in oligohaline wetlands, there is a natural hydraulic gradient with net flow toward the estuary. Because of their close proximity to land, these tidal marshes can also be heavily influenced by anthropogenic activity in the upstream watershed. These marshes are characterized by high primary productivity with values ranging from 1000 – 3000 g m^{-2} yr^{-1} (Mitsch and Gosselink 2000). Due to elevation and salinity gradients and seasonal succession, oligohaline marshes exhibit high spatial and temporal species diversity among macrophyte communities. While wetland primary production may result in a largely seasonal nutrient immobilization, nutrient burial in sediments has the potential for long-term removal.

Sediment nutrient burial has been shown to play an important role in wetland nutrient immobilization (Johnston et al. 1984, Merrill 1999).

In recent decades, bioremediation to reduce nutrient loading to natural waters via treatment wetlands has received much attention (Kadlec and Knight 1996). The ultimate goal in using wetlands to reduce nutrient transport to coastal waters is to apply natural and more cost effective methods to reduce anthropogenic impacts on coastal resources. However, this use of marshes has been obscured by the continuing debate over the whether tidal marshes act as nutrient sources or sinks (Nixon 1980, Correll et al. 1992, Childers 2000).

Research Objectives and Organization

Despite the wealth of literature on wetland nutrient processing (e.g., Valiela and Teal 1974, Olde Venterink et al. 2002), there has been less work done on nutrient processing and long-term impacts of wastewater discharge on oligohaline marshes. Other studies have addressed constructed wetlands for wastewater treatment (Peterson and Teal 1996, Sartoris et al. 2000, Smith et al. 2000, and Spieles and Mitsch 2000), wastewater treatment using natural non-tidal wetlands (Tilton and Kadlec 1979, Dolan et al. 1981, Bayley et al. 1985, Cooke et al. 2000), wastewater impacts on salt marshes (Teal 1997), and impacts of short-term (2 years) wastewater application to tidal freshwater marshes (Whigham et al. 1980). The major research objective of this thesis was to determine if a tidal, oligohaline marsh in which municipal wastewater has been discharged for approximately 40 years can process and retain a significant fraction of the wastewater nitrogen and phosphorus. Additionally, I wanted to determine whether there were any major ecological impacts to the marsh as a result of the wastewater discharge.

Four hypotheses have been developed to address this objective. The first hypothesis is addressed in Chapter 2 to test whether the wastewater inflows induced changes in the hydrology of the marsh and are detectable in Council Creek using two small data loggers that record both temperature and pressure (i.e., water level). In addition, the data loggers were used to discern patterns in temperature and water level from daily to seasonal scales. In Chapter 3, it is hypothesized that evidence for nitrogen and phosphorus uptake and release can be detected from water quality sampling. The integrated effect of all marsh processes was tested using two water quality data sets (samples taken at low tide discharging from the marsh and longitudinal transects of water quality through the marsh). Chapter 4 addresses the hypothesis that marsh macrophytes remove a significant fraction of wastewater nitrogen and phosphorus through production of aboveground biomass. In Chapter 5 it is hypothesized that nutrient burial during sediment accumulation provides a more permanent mechanism for removal of significant amounts of wastewater nitrogen and phosphorus.

Study site description

Council Creek is located in the Choptank River basin on the Delmarva Peninsula (eastern shore of the Chesapeake Bay, Figure 1-1). In Figure 1-2, Council Creek basin (within the yellow polygon) has an area of 2.20 km^2 and flows through a 0.16 km^2 natural oligohaline marsh before flowing into the Choptank River. Land use in Council Creek basin is largely composed of agriculture (40.3%), WWTP property (35.3%), part of the

Talbot County landfill (14.1%), and forest (3.3%). The town of Easton, Maryland is located approximately 6.3 km north/northwest of Council Creek (see Figure 1-1). Although population density within Council Creek basin is low (5.5 km^{-2}), Easton's municipal wastewater treatment plant, serving approximately 8600 people (Lee et al. 2001), is located within the Council Creek watershed (Figure 1-2).

The treatment plant consists of three components. The first component is the lagoon system, which is used to settle out solids and oxidize organic matter (Figure 1-2). Water retention time in the lagoons is approximately 90 days. The second component is the overland flow system (Figure 1-2), which consists of 0.28 km^2 of grass terraces (Cronshaw et al. 1990). Water from the lagoons is spray-irrigated onto the terraces to allow plant and soil processes to remove nutrients from the wastewater, and the grass is periodically harvested to remove plant-accumulated nutrients. During this process the treatment plant has the ability to cease all discharge, or to by-pass the overland flow fields to allow them to dry. The third component is the treatment facility (Figure 1-2), where water is first chlorinated to kill any remaining pathogens, de-chlorinated, and finally oxygenated before being discharged into Council Creek. The entire treatment system is designed to discharge a maximum of about 8900 m^{-3} day^{-1} (2.35 MGD) of treated wastewater (Cronshaw et al. 1990). After discharge into Council Creek, the wastewater flows 1.2 km through the meandering creek before entering the Choptank River. Once the wastewater is in the creek, the tides cause creek water to regularly flood over the surrounding marsh, where further removal of N and P may occur in an area of marsh that is equivalent to 57% of the grassed overland flow system (see Chapters 3-5).

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FIGURE LEGENDS

Figure 1-1: Upper panel: Location of the Choptank Basin relative to Chesapeake Bay. Main panel: Location of Easton, Council Creek (see Figure 1-2)

Figure 1-2: Council Creek basin: Composite of aerial photographs of Council Creek basin. Yellow polygon indicates basin boundary, and green polygon indicates marsh boundary. Black horizontal line and red vertical line are the seams between aerial photographs in the composite.

Choptank Basin

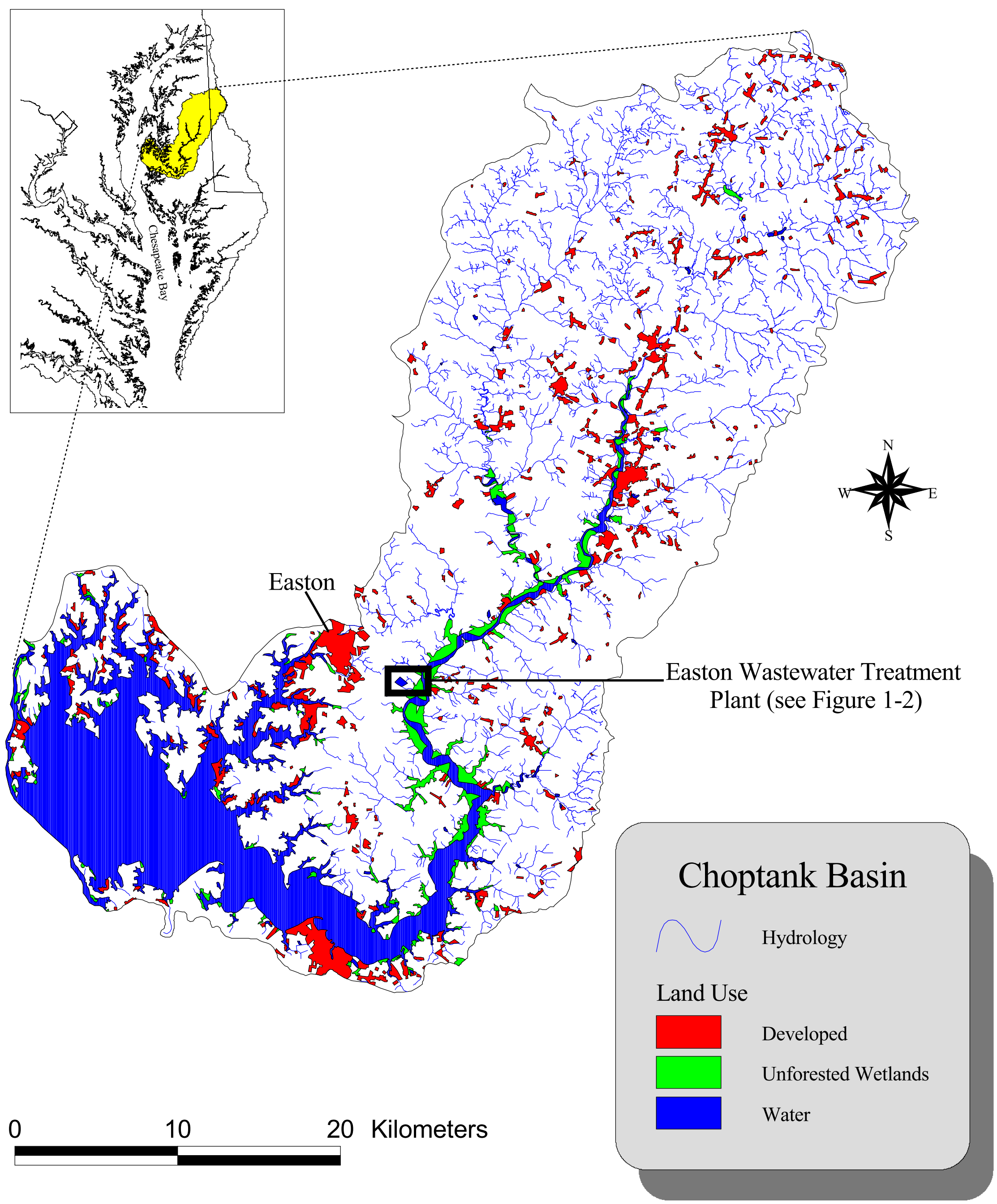

Figure 1-1

Council Creek Basin

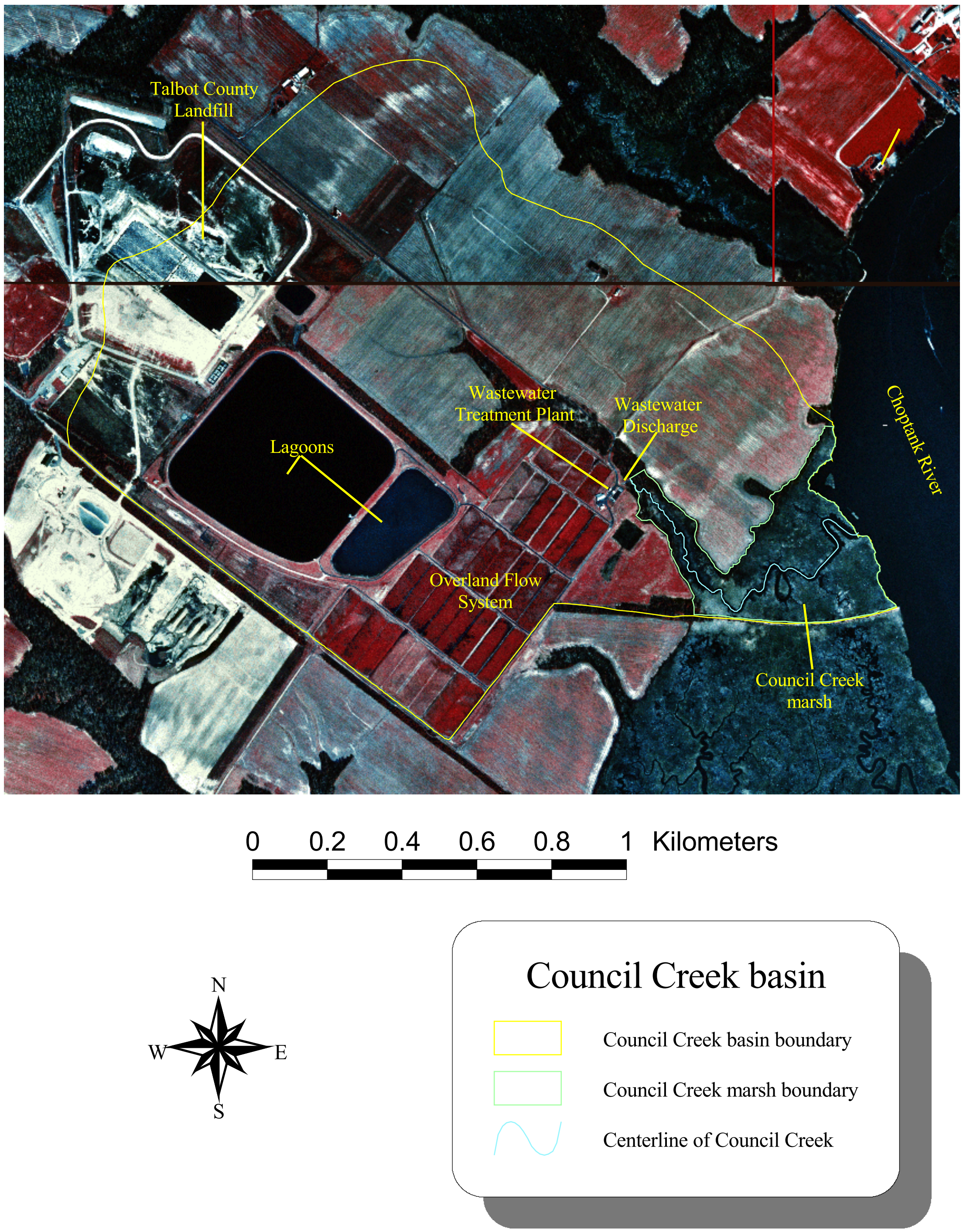

Figure 1-2

Chapter 2

PHYSICAL FORCING AND ANTHROPOGENIC IMPACTS ON AN OLIGOHALINE MARSH

Abstract

Wetlands are increasingly used as alternatives to tertiary treatment of wastewater. In the case of natural wetlands receiving wastewater, augmented hydraulic loading is expected to alter marsh water properties. The hypotheses tested here are (1) that increased hydraulic loading via wastewater inputs increases the hydraulic gradient across a natural oligohaline marsh and (2) that water flowing through the wastewater plant's overland flow system has an effect on marsh water temperature. These hypotheses were tested in Council Creek marsh (0.16 km^2) on the Delmarva Peninsula, which receives an average wastewater flow of 5716 m^3 d^{-1} (1.53 MGD) from the town of Easton, MD. The wastewater plant, located within Council Creek basin (2.20 km^2), employs a 0.27 km^2 overland flow system to provide partial tertiary treatment.

Weather, tides, and WWTP discharge were the main drivers of water temperature and water level in Council Creek marsh. Warming and cooling of water temperature corresponded to air temperature changes at seasonal and weekly time scales. Both water level and water temperature were strongly influenced by frontal passages, with sharp water level and temperature declines occurring immediately following the passage of a cold front and slowly rising water levels and warming temperatures corresponding to the passage of a warm front. Diel temperature cycling resulted from insolation on the overland flow system, with evidence for tidally driven thermal exchanges with the adjacent Choptank River. Diel water level patterns were primarily influenced by tidal

fluctuations, but with a strong influence of wastewater discharge on weekly values of the hydraulic gradient. These data support the hypothesis that increased hydraulic loading from wastewater augments the hydraulic gradient across the marsh, increasing the water flux and decreasing water residence time. Furthermore, the overland flow system acts as a large heat exchanger, which influences marsh temperatures.

Introduction

Wastewater disposal has been a public concern for millennia. The rise of civilization and development of urban areas necessitated the development of the urban water cycle, and the increased human population densities in cities and towns created water supply demands and generated wastewater flows. Up to the early 20th century the goal of wastewater disposal was merely removal from the municipality to reduce the spread of water-borne disease (Stoddard et al. 2002), and this typically meant that wastewater was discharged to rivers and estuaries. With continually rising human populations, degradation of the water supply of downstream municipalities soon became an issue (Stoddard et al. 2002). To address these problems, municipal wastewater treatment methods were developed and implemented for the purpose of reducing contaminants and pathogens. However, more effective and increasingly complex treatment methods are also more costly (Kadlec and Knight 1996).

Wastewater treatment methods are classified into three general categories (Kadlec and Knight 1996, Stoddard et al. 2002). In increasing complexity, effectiveness, and cost, the methods are primary, secondary and tertiary treatment. There are, however, different approaches and varied levels of treatment within each group. Primary treatment methods employ settling and screening of wastewater to remove solids, but suspended organic matter, often with a high biological oxygen demand (BOD), is still discharged in effluent water. Secondary treatment of wastewater utilizes biological processes such as bacterial metabolism to metabolize this residual organic matter (remove the BOD), yet high concentrations of the original nutrients from the human waste (e.g. nitrogen and phosphorus) remain in solution. Tertiary or advanced wastewater treatment is aimed at removal of these wastewater nutrients prior to discharge into receiving water bodies. Tertiary or advanced wastewater treatment can be achieved through costly, high maintenance, chemical or biological processes, but utilization of natural physical, chemical, and biological processes can sometimes provide a cost-effective alternative.

Wetlands are increasingly used as a natural treatment for wastewater (Kadlec and Knight 1996, Kadlec and Reddy 2001). Because they are highly productive ecosystems (Kadlec and Knight 1996, Mitsch and Gosselink 2000), wetlands have the potential to process and remove wastewater contaminants via macrophytic nutrient uptake, burial in sediments, and denitrification. Compared to more expensive alum additions for phosphorus (P) removal or the oxic/anoxic steps required for biological nitrogen (N) removal, wetlands used to remove the contaminants from human wastewater can provide a low cost alternative to these or other tertiary wastewater treatment methods (Dolan et al. 1981, Kadlec and Knight 1996, Smith et al. 2000, Sartoris et al. 2000).

Millions of gallons of wastewater are discharged into wetlands from municipal treatment plants daily as auxiliary treatment (e.g., Lee et al. 2000, Stoddard et al. 2002). The quantity and quality of the effluent that is released can significantly impact biological, chemical and physical processes that occur in the wetland ecosystems receiving effluent. One example is the discharge of wastewater into oligohaline marshes, located at the upper extent of tidal influence. With their direct terrestrial linkage, these marshes may also be highly impacted by other anthropogenic activities. Increased hydraulic loading from wastewater flows, in addition to natural stream flows, may also affect water residence time within the marsh and tidal exchanges between the marsh and the downstream estuary.

Oligohaline marshes have a natural hydraulic gradient, with net flow of water toward the estuary (Figure 2-1). This gradient, defined here as)h/)x (change in water level per unit distance, usually cm km^{-1}), is one of the primary determinants of water residence time within the marsh. Hydrologic inputs and tides are the two main drivers of water residence time and marsh water levels (Whigham et al. 1980, Mitsch and Gosselink 2000). From the upstream watershed there are water inputs due to stream flow, runoff (i.e. precipitation events) and groundwater discharge. From the downstream estuary, water level change is the effect of tides and meteorological forcing (Chuang and Boicourt 1989). Semidiurnal lunar tides are quite predictable because of the regular orbital patterns of the sun and moon. However, meteorological tides are more variable, caused by the more irregular patterns of weather events such as frontal passages and persistent winds (Wang 1979a, Wang 1979c, and Chuang and Boicourt 1989). Variability in the hydraulic gradient and consequently in the water residence time should therefore be a function of changes in watershed inputs and tidal variability. Tidal fluctuations will have varied effects on the residence time and the hydraulic gradient. In general, low tides are expected to increase the hydraulic gradient, increase the flow out of the marsh, and decrease water residence time, while high tides are expected to decrease or reverse the hydraulic gradient and flow and increase water residence time.

This chapter focuses on the physical and hydraulic effects of wastewater on an oligohaline marsh. Council Creek marsh on the Choptank estuary receives an average of 5716 m^3 d^{-1} (1.53 MGD) of wastewater from the town of Easton, Maryland on the Delmarva Peninsula (Figure 2-2). The plant uses primary and secondary treatment in a lagoon system, plus a partial tertiary treatment in a grassed overland flow system. I hypothesize that discharge from the wastewater treatment plant increases the natural hydraulic gradient ($\Delta h_n/\Delta x$) across Council Creek marsh by increasing the hydraulic head ($\Delta h_n+\Delta h_w$) at the landward edge of the marsh (Figure 2-1). Additionally, I hypothesize that wastewater discharge will affect water temperature in Council Creek marsh due to heating and cooling of wastewater in the lagoons and overland flow system. Evidence to support or reject this hypothesis should emerge from an examination of discharge data, weather data, and high-resolution marsh water level and water temperature data. Furthermore, such detailed water level and temperature data are not well documented in marsh literature.

Study Site Description

Council Creek is located in the Choptank River basin on the Delmarva Peninsula (eastern shore of the Chesapeake Bay, Figure 2-2). Council Creek basin (within the yellow polygon) has an area of 2.20 km^2, and the creek flows through a 0.16 km^2 natural oligohaline marsh before flowing into the Choptank River (Figure 2-3). The town of Easton, Maryland is located approximately 6.3 km north/northwest of Council Creek (Figure 2-2), and the municipal wastewater treatment plant is located within the Council Creek watershed (Figure 2-3). Additionally, part of the Talbot County, Maryland landfill is located within the Council Creek watershed.

The treatment plant consists of three components (Figure 2-3). The first component is the lagoon system, which is used to settle out solids and oxidize organic matter. Water retention time in the lagoons is approximately 90 days. The second component is the overland flow system, which consists of 0.28 km^2 of grass terraces (Cronshaw et al. 1990). Water from the lagoons is spray-irrigated onto the terraces to allow plant and soil processes to remove nutrients from the wastewater, and the grass is periodically harvested to remove plant-accumulated nutrients. During this process the treatment plant has the ability to cease all discharge, or to by-pass the overland flow fields to allow them to dry. The third component is the treatment facility, where water is first chlorinated to kill any remaining pathogens, de-chlorinated, and finally oxygenated before being discharged into Council Creek. The entire treatment system is designed to discharge a maximum of about 8900 m^{-3} day^{-1} (2.35 MGD) of treated wastewater (Cronshaw et al. 1990). After discharge into Council Creek, the wastewater flows 1.2 km through the meandering creek before entering the Choptank River. Once the wastewater is in the creek, the tides cause creek water to regularly flood over the surrounding marsh, where further removal of N and P may occur in an area of marsh that is equivalent to 57% of the grassed overland flow system (Chapters 3-5).

Methods

Weather data were obtained from four local sources (Table 2-1). Because of the spatial variability of precipitation, data were obtained from sources that spatially encompass Council Creek (Figure 2-2). These locations were the Cambridge Wastewater Treatment Plant, Horn Point Laboratory, the NWS station in Dover, DE, and the NWS station in Royal Oak, MD. Insolation data from Horn Point Laboratory (Fisher et al. in press), and air temperature and wind data from Royal Oak were also utilized.

Tide data were acquired from two sources. Tides and Currents Pro for Windows (v 3.0h) by Nobeltec Corporation was used as the source for predicted tide data. Tidal predictions for Dover Bridge were used, which is the tidal prediction station closest to Council Creek (1.1 km upstream of the mouth of Council Creek, Figure 2-3). Observed water level and water temperature data were also obtained from two model 3001 LT F15/M5 data loggers manufactured by Solinst Canada Ltd. The data loggers resolve water level changes to 0.1cm with 0.1% accuracy, and resolution of water temperature data was 0.01ºC with an accuracy of 0.1ºC (Solinst website). Levelogger software (v 1.5.0.15) was used for data acquisition and data logger communication; data were downloaded to a portable computer at approximately three-month intervals.

The data loggers were installed in the marsh from 15 November 2001 through 19 November 2002. The inner station data logger was placed at the head of the marsh, while the outer station data logger was located near the confluence of Council Creek and the Choptank River (Figure 2-3). Two 20-cm ID schedule 80 PVC pipes of 1m length were manually inserted vertically into the creek bed about 30 cm below mean low water (Figure 2-4). To prevent damage from boats and debris, the data loggers were suspended inside the PVC pipes from stainless steel rods secured horizontally about 1cm from the top of the pipe (Figure 2-4). Water level and temperature were recorded at 15-minute intervals for the entire 367 days except for three brief periods (1130h to 1300h on 17 December 2001, 1125h to 1405h on 11 April 2002, and 1345h to 1405h on 8 August 2002) when the data loggers were collected to download data. Due to instrument failure, data were lost at the outer station from 8 August through 19 November 2002. Upon investigation of the outer station water level data, there was a declining trend in tidal height from 21 June through 8 August 2002 that was attributed to drift associated with the instrument failure. For this reason, outer station water level data from this period were also not used.

The loggers recorded water level data relative to the default reference level of 100 cm. In order to normalize both loggers to a common horizontal datum, long-term mean (± se) water levels were calculated for the period 15 November 2001 to 20 June 2002 for each data logger (inner station: 142.3 ± 0.2 cm, outer station: 146.9 ± 0.2 cm). I had attempted to install the data loggers at approximately the same water depths at these tidal stations, and the difference between the two long-term means, 4.7 ± 0.4 cm, represents the slightly deeper installation depth of the outer station data logger. To correct for this bias, long-term mean sea level at the outer station was used as a horizontal datum. Inner station data were referenced to the outer station by adding the difference between the two long-term mean values (4.7 cm) to all inner station data points (Figure 2-4) and then subtracting the outer station mean from each 15-minute observation to yield the deviation from the common horizontal datum. Under the assumption that the mean difference over 7 months between the stations is due only to installation differences, all inner station water level values have therefore been referenced to local mean sea level at the outer station.

Results

Weather Data

Air temperature during the study period was compared to long-term means to determine if the study period exhibited anomalous weather conditions. In this region minimum monthly mean temperatures occur in January (Norton and Fisher 2000) and steadily increase to a maximum in July and August. Following the summer maximum, air temperatures then decline throughout fall and into winter. Mean monthly air temperatures at Royal Oak for 2001 and 2002 were compared to long-term (1993-2002) mean monthly air temperatures for this same station (Figure 2-5). Although mean monthly air temperatures in winter, spring, and summer were higher than average and fall was lower than average in 2001, none of the differences fell outside of the 95% confidence intervals of the long-term data ($p > 0.05$). Temperature in 2002 was characterized by a slightly cooler than average winter, average spring and summer, and warmer than average fall. However, again there were no statistically significant deviations ($p > 0.05$) from normal conditions in any month. Therefore, monthly mean temperature data during the period can be characterized as within the normal ranges for each month.

Precipitation data for the study period were also compared with long-term mean values. Mean monthly precipitation for the region is 8-9 cm month^{-1} and relatively uniform throughout the year (Figure 2-6); however, there is a slight increase in precipitation to about 10 cm month^{-1} at high temperatures in July and August due to water vapor recycling (monsoonal effect, Lee et al. 2001). Monthly precipitation totals from Royal Oak for 2001 and 2002 were compared to long-term (1948-2002) mean monthly precipitation totals (Figure 2-6). Summer 2001 was wetter than average, but the fall was drier than average. Conversely, 2002 was characterized by a drier than average spring and summer and a wetter than average fall. Although the dry conditions in fall

2001 through summer 2002 led to a severe drought (Maryland Department of the Environment website), on a monthly time scale none of the differences fell outside of the 95% confidence intervals of the long-term data ($p > 0.05$). Therefore, monthly mean precipitation during the study period can be characterized as within the normal ranges for each month.

Daily wind data during the study period were also obtained from Royal Oak to further characterize the climate during the study period. Data on speed and direction of long-term mean winds (using records from Baltimore, Maryland and Dover, Delaware) show that the general pattern is one of maximum winds (3.5-5 m s^{-1}) occuring during the winter out of the NW (NOAA-National Climatic Data Center website, Figure 2-7). During spring and summer, wind direction shifts to S/SW and velocity declines to a minimum (2-3.8 m s^{-1}) in August. Throughout fall, wind direction continues from the S/SW with increasing speed. Mean monthly wind speeds from Royal Oak were consistently higher than the long-term values reported for Dover and Baltimore, and the differences are probably due to measurement biases. Wind at Dover and Baltimore is measured using totalizing anemometers on a 24-hour basis, whereas winds at Royal Oak are estimated twice daily by the observer at the weather station. Although there is a positive bias in the Royal Oak wind speed data in Figure 2-7, the seasonal pattern of winds during the study period is comparable to the long-term pattern of winds, with a maximum speed in March, primarily from the NW, and an annual minimum in summer from the SW. On shorter time scales (days), shifts in speed and direction typically signify passages of major weather systems such as fronts or low-pressure cells. Passages of cold fronts are accompanied by a shift in wind from the S or SW to NW, while wind directions shifting to S and SW are the result of an overriding warm front.

Marsh Temperature Data

The annual water temperature cycle in Council Creek marsh follows closely the annual air temperature cycle (Figure 2-8). This figure shows mean monthly water temperatures (°C) at the inner and outer stations and the mean monthly air temperatures (°C) during the period of data logger deployment (15 November 2001-19 November 2002). The data from November of 2001 and 2002 were pooled to get one water temperature value for the inner station. Water temperature and air temperature were at a minimum (~5°C) in January, steadily increased to maximum in summer (July and August), and then declined throughout the fall to winter temperatures. During the study period, water temperature at the inner station closely followed air temperature except in the summer, when the inner station was about 2-3°C cooler. Water temperature at the outer station closely followed air temperature even into the summer when the time series ended due to instrument failure.

Examination of the temperature data in more detail helps reveal the mechanisms causing the general seasonal cycle shown in Figure 2-8. Seasonal cooling of marsh water temperature in the fall followed a trend similar to that of air temperature cooling (Figure 2-9). During December 2001, water temperature (15-minute observations) at the inner station and outer station (15-minute observations) cooled more slowly than did mean daily air temperature (twice daily observations). During this period, mean daily air temperature declined at a rate of 0.45°C day^{-1} (Figure 2-9 C), while temperatures at the inner and outer stations decreased at 0.26°C day^{-1} and 0.33°C day^{-1}, respectively (Figure

2-9 A, B). The general cooling trend during this month-long time period was punctuated by sharp decreases due to passage of cold fronts at approximately weekly intervals, as marked by NW winds. Note that temperature at the outer station exhibited more high frequency variation due to tidal exchanges with the Choptank River. Thus the mechanisms of heat exchange controlling outer station water temperature were probably continuous conductive cooling with air, weekly cold front passages, and exchange with the larger thermal mass of the Choptank estuary.

Similar to the cooling trend, seasonal warming tended to follow closely the trend of mean daily air temperature (Figure 2-10). Comparison of water temperatures at the inner and outer stations and mean daily air temperatures from 24 to 31 May 2002 showed that the water temperature at the inner station was rising at a rate of 0.59°C day^{-1}, about equal to that of mean daily air temperature (0.60°C day^{-1}); however, water temperature at the outer station was rising at a higher rate (0.69°C day^{-1}). Note the strong diel pattern in temperature at both stations with more high frequency variability at the outer station due to tidal exchange. Therefore, as in the fall, heat exchange with the atmosphere and Choptank estuary as well as solar heating (described in more detail below) are the dominant processes influencing water temperature at the two stations.

Temperatures at both marsh stations also responded to frontal weather systems. Northwest winds and sharp temperature declines are caused by passage of a cold front (as shown in Figure 2-9). Conversely, passage of a warm front causes S/SW winds and slowly warming temperatures. Figure 2-11 shows plots of inner and outer station water temperature (15-minute observations), mean daily air temperature, and mean daily wind vectors during March 2002. Cold fronts passed through the area at regular intervals of five to six days. Air temperature and marsh water temperature at both stations dropped sharply (5-10°C) following the switch to NW winds (see vertical dashed lines and red arrows in Figure 2-11), despite the overall warming trends during the entire month.

Diel patterns of water temperature at inner and outer stations are quite variable. Diel temperature changes at the inner station are as high as 3°C, while they were up to 10°C at the outer station during the study period (Figures 2-10 and 2-11). To further explore the diel cycles, Figure 2-12 shows examples of inner station water temperature (15-minute observations), hourly insolation, outer station water temperature (15-minute observations), and outer station water level (15-minute observations) data for 24 to 31 May 2002, which were relatively clear, sunny days (Figure 2-12 B). Water temperature at the inner station experienced a daily temperature minimum and maximum at approximately 1000h and 2200h, respectively (Figure 2-12A). These times are ~ 8 to 9 hours out of phase with the daily insolation cycle (compare panels A and B in Figure 2-12). A similar pattern occured throughout the year; however, during November and December the phase shift was only 5 to 7 hours.   This very regular diel heating and cooling cycle at the inner station was unrelated to tides (compare panels A and D in Figure 2-12), but strongly related to insolation (panels A and B in Figure 2-12).

The outer station temperature record is more complex than the inner station due to the close proximity of the Choptank River. Temperature at the outer station also exhibited a diel pattern related to insolation, but here it lags only 5 to 6 hours behind insolation (Figure 2-12 C). However, as high tides shift into phase with the daily temperature peak, the influx of cooler Choptank River water in spring drives the temperature back down until water level falls again. During this part of the record, 5ºC temperature changes occurred 1-2 times per day. In contrast to the spring conditions shown in Figure 2-12, when air temperatures drop below the water temperature of the Choptank River in fall, there is a tidally induced warming at the outer station corresponding to high tides (not shown, but the inverse of the pattern in Figure 2-12). This seasonal reversal is due to the larger thermal inertia of the Choptank River compared to the surface water in Council Creek.

Effluent discharge from the wastewater treatment plant (WWTP) also influenced temperature at the inner marsh station. Inner and outer station water temperature (15-minute observations) and mean daily air temperature are plotted for 13 April to 31 May 2002 in Figure 2-13. During this period, three different management operations took place at the WWTP (Figure 2-13 C). Through 19 April the overland flow system was in use. Starting on 20 April through 8 May and on 12 May there was no wastewater discharged to Council Creek. From 9 to 11 May and 13 to 23 May wastewater was discharged, but the overland system was by-passed, until normal operations resumed on 24 May. The cessation of discharge on 20 April was coincidental with the passage of a cold front, and resumption of normal flow operations was coincidental with a warm front (Figure 2-13 C).

A temperature signal from the WWTP's overland flow system is apparent at the inner station (Figure 2-13 A), but not at the outer station (Figure 2-13 B). At the inner station, diel temperature cycles from diurnal solar heating and nocturnal radiative cooling are apparent until discharge stopped on 20 April (left side of Figure 2-13 A) at which time a sharp temperature decline occurred due to the cold front passage. During the period of no overland flow, inner station water temperature generally followed mean daily air temperatures but showed no regular diel pattern. When direct discharge from the plant initially resumed on 9 May, bypassing the overland flow system, there was little effect on water temperature at the inner station. It was only when overland flow resumed on 24 May that the regular diel oscillations resumed. Water temperatures also continued this seasonal rise with warming air temperatures following the coincidental warm front passage on 23 May. See Figure 2-12 for a more detailed examination of the latter period. Water temperature at the outer station followed the general weather patterns but exhibited no discernable effects from changes in WWTP operations with large tidal effects, as also shown in Figure 2-12. From Figure 2-12 and 2-13 it is clear that the overland flow system of the Easton WWTP is acting as a large solar collector throughout the year and that heat is transferred to the inner marsh by the inflowing wastewater, with about a 5 to 9 hour time lag from the diurnal peak in solar radiation.

Marsh Stage Data

Predicted water level data at Dover Bridge (Figure 2-3) from Tides and Currents Pro was used to estimate lunar tides. These data were exported as 15-minute observations from 14 November 2001 to 20 June 2002 and normalized to mean sea level by subtracting the long-term mean water level value from all data points. Tidal amplitudes averaged about 40-50 cm with a 14-day oscillation due to solar/lunar phasing.

The observed annual cycle of water level in Council Creek marsh follows predictions for Dover Bridge (Figure 2-14). This figure shows mean monthly water levels for both inner and outer stations of Council Creek marsh and predicted data at Dover Bridge during the period of data logger deployment (15 November 2001 – 19

November 2002). The data from November 2001 and 2002 were pooled to get one value for the inner station. While the predicted data are offset because the data were normalized to MSL at Dover Bridge, the general patterns are similar for predicted and observed data. Mean water levels increase from a minimum in January to a peak in late summer due to net sea level increases arising from S winds and thermal expansion of the North Atlantic resulting from solar heating (NOAA-NOS website). Through the fall as water temperature declines and winds change to the N (Figure 2-7), mean water levels fall ~30 cm to winter values.

Marsh water levels in Council Creek are primarily influenced by semidiurnal lunar tides and meteorological forcing. Inner and outer station marsh water level (15 minute observations) in Council Creek, predicted water level at Dover Bridge (15 minute observations), and mean daily wind vectors are plotted in Figure 2-15 for March 2002. Daily tidal oscillations are evident at both inner and outer stations; however, as with marsh water temperature (Figure 2-11), marsh water level responded to passage of frontal weather systems (cold fronts in this case, red arrows in Figure 2-15 D). Both the inner and outer stations showed the same sharp decreases in water level following cold fronts with occasional damped tidal ranges for a cycle or two. Water level declines of 20-40 cm follow the passage of cold fronts at both the inner station (Figure 2-15 A) and outer station (Figure 2-15 B). Passages of cold fronts result in decreases in water level 70% of the time and can occasionally overwhelm the semidiurnal lunar tides, as was the case on 22 March (see black arrows on Figure 2-15 A, B, and C). However, there was no significant relationship between the speed of northerly winds and the magnitude of water level decreases in this data set.

Observed versus predicted tidal ranges were compared for March 2002. In general, the predicted time of high and low tides at Dover Bridge agreed well with observation at the inner and outer stations of Council Creek marsh (Figure 2-15); however, the amplitudes of the observed tides differed considerably from the predictions due the influence of wind tides and local water inputs. Mean tidal ranges were calculated for the inner and outer stations and Dover Bridge for each spring and neap tide (Table 2-2). Each mean was calculated over a 3-day time period that included all tidal ranges on the day of the spring or neap tide and one day before and after each spring or neap tide (Figure 2-15). Due to a significant wind event that would have biased the 20-22 March 2002 observed neap tidal ranges, the last two tidal cycles were not included in the calculation. The data from Table 2-2 are plotted as observed tidal range versus predicted tidal range in Figure 2-16. The outer station ranges correlated well with the predicted ranges at Dover Bridge ($r^2 = 0.65$), although the slope was less than expected (slope = 0.33). At the inner station, the observed tidal range did not exhibit a significant correlation with predicted range ($r^2 = 0.22$). However, all values scatter about the 1:1 line in the vicinity of 40-50 cm, indicating little overall bias in the observed tides relative to the predictions.

Marsh Hydraulic Gradient

The hydraulic gradient ($\Delta h/\Delta x$) in Council Creek marsh was calculated from the two sets of marsh stage data. Because the two data loggers were not exactly synchronized, the difference in water level height between the data loggers (Δh) at any point in time was linearly extrapolated to within one minute by calculating the rate of change of water level between each data point at the inner station. With data points at both stations at the same time (within one-minute), the difference in water level height (Δh) between the data loggers (inner station-outer station) was calculated for all time points and divided by the distance in stream length between the inner and outer stations (Δx = 1.00 km), yielding a time series of values of the hydraulic gradient ($\Delta h/\Delta x$, cm km^{-1}, see Figure 2-17).

The mean value of the hydraulic gradient was 4.7 cm km^{-1}, but the gradient was highly variable, occasionally reversing with the tides (Figure 2-17). The hydraulic gradient usually remained positive over the study period, indicating nearly continuous discharge toward the Choptank at the inner station, even at high tide. Fluctuations in the hydraulic gradient are typically in the 0-5 cm km^{-1} range that result from average tidal fluctuations (as shown in the bottom panel of Figure 2-17). Large positive deviations from normal variability can be attributed to three main processes: 1) basin runoff during precipitation events (as shown for 7 February in the bottom panel of Figure 2-17), 2) large WWTP discharge events, and 3) marsh draining resulting from abnormally low tides resulting from passages of cold fronts (as shown in top panel of Figure 2-17).

The occasional negative excursions of the hydraulic gradient were events when there was net flow of Choptank water into Council Creek. These events were relatively infrequent, occurring on only 25 of the 410 tides (6.10%) from 15 November 2001 – 20 June 2002 and usually lasted for about 0.5–2 hours. Of these 25 negative events, 20 (80%) of them are the result of a tidal phase shift between the inner and outer stations (Figure 2-18). This figure shows a typical example when Choptank River water is flowing into Council Creek on flood tide. Of the remaining five (out of the 25) negative excursions in the hydraulic gradient, only one could be explained as noise in the data due to strong NW winds with gusts of 17-20 m s^{-1}.

The effect of effluent discharge to Council Creek is also clear from the estimated hydraulic gradient values. Due to the variability shown in Figure 2-17, 7-day averages of the data were calculated to remove high frequency signals such as tides. In Figure 2-19 the 7-day hydraulic gradient averages are plotted versus 7-day averages of WWTP discharge data. The averages correspond to the same weekly intervals spanning 18 November 2001 to 16 June 2002. Linear regression of these data indicates that the WWTP discharge is positively correlated with the hydraulic gradient across Council Creek marsh ($r^2 = 0.58$, $p < 0.0001$). In the absence of WWTP flow, the natural hydraulic gradient $\Delta h_n/\Delta x$ across Council Creek marsh approximates 2.3 cm km^{-1} (the intercept) from the stream and groundwater inflows from the drainage basin. However, the 7-day average gradient increases linearly with WWTP discharge as water levels increase at the inner station due to augmented water inputs (Figure 2-1).

Discussion

Weather, tides, and WWTP discharge were the main drivers of water temperature and water level in Council Creek marsh. Seasonal warming and cooling of water temperature corresponded to seasonal air temperature changes (Figures 2-8, 2-9 and 2-10), and at weekly time scales both water level and water temperature were strongly influenced by frontal passages, with sharp water level and temperature declines occurring immediately following the passage of a cold front (Figures 2-11 and 2-15). Conversely, there were also slowly rising water levels and warming temperatures corresponding to the passage of a warm front. Diel temperature cycling resulting from insolation was evident at both stations throughout the year (Figure 2-12 panels A and C). However, the outer station had the added influence of tidally mediated water exchanges with the Choptank River, which added complexity to the observed diel temperature pattern (Figure 2-12 panel C). Diel water level patterns were primarily influenced by the semidiurnal lunar tidal fluctuations, with attenuated tidal ranges at the inner station (Figure 2-15). Seasonal water level patterns in Council Creek marsh followed the expected cycle of a minimum in winter and maximum in summer (Figure 2-14). This pattern means that in winter, not only are the colder temperatures reducing potential biological and chemical processing of wastewater, but also there was less marsh contact with wastewater because water is confined to the creek channel more often with the opposite being true for summer conditions (see Chapter 3).

The hypothesis to be tested in this chapter was that the discharge of wastewater to Council Creek marsh increases the hydraulic gradient across the marsh. Further, I expected to see a wastewater temperature signal in the marsh, because of the large area of the overland flow system (0.28 km^2) relative to the marsh area (0.16 km^2). To test this two-part hypothesis, marsh water temperature and stage data were examined over a one-year period and compared to weather, WWTP management and discharge records, and predicted tidal data.

These data of wastewater discharge supported the hypothesis that there is a physical influence of wastewater discharge on Council Creek marsh. While the overland flow system is designed to remove nutrients through plant uptake and soil adsorption, it also functions as a solar collector during the day and a radiator at night. On clear days, the sun heats the overland flow system, wastewater flowing through the system is warmed, and then this solar energy is advected to the marsh via wastewater discharge. At night, the overland flow system undergoes radiative cooling, which cools the wastewater entering the marsh with a diel phase shift of approximately 5 to 9 hours (Figures 2-12 and 2-13).

The discharge from the plant also influences the hydraulic gradient across the marsh. Under natural conditions (i.e. no wastewater discharge), Council Creek marsh has an average hydraulic gradient toward the Choptank River of ~2.3 cm km^{-1}, resulting from the bathymetric changes and inflowing water from the drainage basin. However, the hydraulic gradient increases linearly with increases in wastewater discharge (Figure 2-19) due to the additional water inputs derived primarily from the deep groundwater pumping (>100 m) from wells around Easton and outside the natural drainage of Council Creek basin. This interbasin transfer of water via wastewater augments the natural inputs and increases the hydraulic gradient from ~2 cm km^{-1} to as much as ~8 cm km^{-1}. In contrast to natural conditions, during average discharge (5716 m^3 day^{-1}), the hydraulic gradient increases to 5.3 cm km^{-1} (Figure 2-19). Furthermore, despite the variability caused by tides, the hydraulic gradient in Council Creek marsh is nearly always positive (i.e. the flow is toward the Choptank estuary at the inner station, Figure 2-17). It is clear, both from the marsh temperature data (Figure 2-12 and 2-13) and marsh stage data (Figure 2-19), that the hypothesis is supported by the observations and that discharge from Easton's wastewater treatment plant has significant physical impacts on Council Creek marsh.

Implications of the physical impacts of wastewater discharge into Council Creek primarily result from the increase in hydraulic gradient across the marsh. A two-fold increase in the hydraulic gradient from natural to average discharge conditions should result in a significant reduction in water residence time in this marsh. This reduction in hydroperiod has a major consequence that increased hydraulic gradient and shorter residence times mean that there is more unprocessed wastewater flowing out of the marsh and into the estuary. In turn, a shorter water residence time means that there is less time for water quality improvement (i.e., nutrient removal) since water residence time represents the contact time between wastewater and the marsh when plant assimilation and/or sediment processes act on wastewater N and P. While Council Creek marsh removes some wastewater N and P (see chapters 3-5), the fraction of the total wastewater N and P that is sequestered by marsh processing is probably reduced by the increased hydraulic gradient and consequently shorter residence times.

In addition to the wastewater discharge influence, other implications of marsh response to physical forcing are the dynamic and rapid responses to weather changes. Rates of seasonal warming and cooling at both the inner and outer stations of Council Creek marsh are not significantly different than the rates of seasonal warming and cooling of air temperatures (Figures 2-9 and 2-10). Marsh water temperatures also closely follow weekly air temperature patterns that result from regional weather patterns like the passage of frontal systems (Figure 2-11), and marsh water levels also show rapid response to the passage of frontal systems (Figure 2-15). Marsh draining events and lower high tides caused by cold front passages reduce marsh contact with wastewater by confining flow to the creek channel with little over bank flow at high tides, making the marsh less effective at nutrient processing. Large diurnal water temperature changes on the order of 5 to 10 ºC regularly occur as result of solar heating and tidal exchange with the Choptank River. These values are slightly higher than those found for other treatment wetlands by Kadlec and Reddy (2001) but within the range of diel variability (~4-13ºC) in a Choptank estuary salt marsh (Shiah and Ducklow 1995).

Biological processes are largely temperature dependent and therefore marsh nutrient processing will covary with the temperature changes described above (Kadlec and Reddy 2001, Mitsch and Gosselink 2001). Kadlec and Reddy's study and review of temperature effects in treatment wetlands surveyed various wetland processes. They concluded that BOD removal slightly decreases in summer, while nitrogen transformations showed a more complex pattern. Ammonification, ammonia volatization, nitrification, and nitrate reduction have been found to increase as seasonal temperatures increase. However, the reduced oxygen availability that occurs with increased temperatures could potentially limit nitrification rates which consequently may affect denitrification rates during warmer months (Kadlec and Knight 1996, Kadlec and Reddy 2001). Macrophytic nitrogen uptake and storage is seasonal, and maximal uptake rates have an optimal temperature (Reddy and Portier 1987). Phosphorus removal is relatively unaffected by temperature changes, despite its importance as a major plant nutrient (Kadlec and Knight 1996, Kadlec and Reddy 2001); however, microbial activity is temperature dependent with production in the Chesapeake Bay area, increasing with temperature up to about 20ºC. Above 20ºC, the effect of temperature is weaker due to other limiting factors such as carbon substrate (Shiah and Ducklow 1994 and 1995). In addition to seasonal changes in microbial activity, the fast temperature response time of bacteria makes them susceptible to the diel temperature changes (Shiah and Ducklow 1994).

In summary, it is clear that the Easton wastewater treatment plant has significant physical effects on Council Creek marsh. Water temperature in Council Creek marsh exhibits rapid responses to physical influences at several time scales. Rates of marsh seasonal cooling and warming are similar to air seasonal cooling and warming, and frontal passages are the major drivers of water temperature at weekly time scales. Water temperature at the inner station exhibits a cyclic diel pattern that is caused by solar heating and radiative cooling of the WWTP's overland flow system. Solar heating and radiative cooling of marsh water is clear at the outer station as well, but there is also an additional temperature influence from tidal exchanges with the larger thermal body of the Choptank River that can result in temperature changes of up 10ºC in one tidal cycle. The augmentation of hydrologic inputs to Council Creek due to wastewater discharge significantly increases the hydraulic gradient across the marsh and reduces water residence time. As a result, the effectiveness of Council Creek marsh as a natural wastewater treatment system is probably significantly reduced.

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Table 2-1: List of weather stations in or near the Choptank Basin from which weather data were obtained for this study

Source	Location	Data	Dates	Reference
Cambridge Wastewater Treatment Plant	Cambridge, Maryland	Precipitation	Aug. 1948 – Jun. 1990	Lee et al 2000, Norton and Fisher 2000
Horn Point Laboratory	Cambridge, Maryland	Precipitation, Insolation	Precip: Jun. 1997 - Jun. 1999 Insolation: Nov. 2001 – Nov. 2002	Lee et al 2000, Norton and Fisher 2000, Fisher et al in press
National Weather Service Station	Dover, Delaware	Precipitation	Aug. 1948 – Dec. 1997	Lee et al 2000, Norton and Fisher 2000
National Weather Service Station	Royal Oak, Maryland	Precipitation, temperature, and wind	Precip: Aug. 1948 – Dec. 2002 Temp: 1993-2002 Wind: 2001-2002	Lee et al 2000, Norton and Fisher 2000, John Swaine, Jr. – NWS observer

Table 2-2: Tidal ranges for March 2002 for three-day periods encompassing each spring/neap tide at Dover Bridge, inner station, and outer station (see also Figure 2-14)

Date	Tide/moon	Dover Bridge (predicted)	Inner station (observed)	Outer station (observed)
4-6 Mar. 2002	Neap/half	47.1 ± 8.4 cm	33.9 ± 2.9 cm	47.0 ± 3.3 cm
12-14 Mar. 2002	Spring/new	46.1 ± 1.6 cm	41.4 ± 3.3 cm	52.8 ± 2.2 cm
20-22 Mar 2002	Neap/half	36.5 ± 9.3 cm	47.4 ± 6.5 cm	49.5 ± 5.7 cm
27-29 Mar. 2002	Spring/full	65.6 ± 2.5 cm	53.7 ± 2.1 cm	58.6 ± 1.4cm

FIGURE LEGENDS

Figure 2-1: Conceptual diagram of wastewater effects on the natural hydraulic gradient ($\Delta h/\Delta x$) of a tidal marsh.

Figure 2-2: Upper panel: Location of the Choptank Basin relative to Chesapeake Bay. Main panel: Location of Easton, MD, Easton Wastewater Treatment Plant (see Figure 2-3), and weather stations.

Figure 2-3: Composite of four 1994 aerial photograph of Easton Wastewater Treatment Plant and Council Creek Basin. Black horizontal line and red vertical line are the seams between photographs in the composite.

Figure 2-4: Data logger installation correction. Correction for the absolute difference in data logger height was found to be 4.65 cm. Since the outer station was used as the horizontal datum, 4.65 cm. was added to all inner station data to correct for installation bias.

Figure 2-5: Monthly mean air temperatures (°C) at Royal Oak, Maryland: long-term monthly mean (1993-2002) and standard errors (heavy black line), monthly means for 2001 (blue line), and monthly means for 2002 (red line). Error bars indicate standard error of the mean.

Figure 2-6: Monthly precipitation (cm month^{-1}): long-term spatial and temporal average of weather stations in the vicinity of Council Creek (see Table 2-1) (heavy black line), ± 1 standard deviation of long-term mean (shaded area between dotted lines), monthly precipitation sum at Royal Oak, Maryland for 2001 (blue line), and monthly precipitation sum at Royal Oak for 2002 (red line).

Figure 2-7: Mean monthly winds (m s^{-1}): The shaded area is the long-term (1930-1996) monthly mean regional variability in wind speed between Dover Air Force Base, Dover Delaware and Baltimore Washington Airport, Baltimore, Maryland. The wind direction in the shaded area represents the mean wind direction between Dover and Baltimore (NOAA-National Climatic Data Center website). The blue line is the mean monthly wind speed at Royal Oak, Maryland for 2001. The red line is the mean monthly wind speed at Royal Oak, Maryland for 2002. Errors represent standard error of the mean.

Figure 2-8: Mean monthly water temperature (°C): Mean monthly water temperature at the inner station of Council Creek during data logger deployment (red line), mean monthly water temperature at the outer station during data logger deployment (blue line), and mean monthly air temperature at Royal Oak, Maryland during data logger deployment (green line). Error bars indicate standard error of the mean.

Figure 2-9: Fall/winter cooling in December 2001. Panel A: Water temperature (15-minute observations) at the inner station of Council Creek marsh, Panel B: Water temperature (15-minute observations) at the outer station, Panel C: Mean daily air temperature at Royal Oak, Maryland.

Figure 2-10: Spring warming in May 2002. Panel A: Water temperature (15-minute observations) at the inner station of Council Creek marsh, Panel B: Water temperature (15-minute observations) at the outer station, Panel C: Mean daily air temperature at Royal Oak, Maryland.

Figure 2-11: Effect of frontal passages on temperature in March 2002. Panel A: Water temperature (15-minute observations) at the inner station of Council Creek marsh, Panel B: Water temperature (15-minute observations) at the outer station, Panel C: Mean daily air temperature at Royal Oak, Maryland, Panel D: Mean daily wind at Royal Oak, Maryland. Wind vectors indicate magnitude and direction. Wind direction originates at zero. Cold fronts are indicated by vertical dashed lines.

Figure 2-12: Diel temperature changes in May 2002. Panel A: Water temperature (15-minute observations) at the inner station of Council Creek marsh, Panel B: Hourly insolation data at Horn Point Laboratory, Panel C: Outer station water temperature (15-minute observations), Panel D: Water level (15-minute observations, cm.) at the outer station.

Figure 2-13: Effect of wastewater plant management. Panel A: Water temperature (15-minute observations) at the inner station of Council Creek marsh in May 2002, Panel B: Water temperature (15-minute observations) at the outer station in May 2002, Panel C: Mean daily air temperature at Royal Oak, Maryland in May 2002. Arrows in panel C indicate the status of WWTP operations.

Figure 2-14: Mean monthly water level: Mean monthly observed water level at the inner station of Council Creek during data logger deployment (red line), mean monthly observed water level at the outer station during data logger deployment (blue line), and mean monthly predicted water level at Dover Bridge during data logger deployment (green line). Error bars indicate standard error of the mean.

Figure 2-15: Marsh water level in March 2002. Panel A: Water level (15-minute observations) at the inner station of Council Creek marsh, Panel B: Water level (15-minute observations) at the outer station. Panel C: Predicted water level (15-minute observations) at Dover Bridge, Panel D: Panel D: Mean daily wind at Royal Oak, Maryland. Wind vectors indicate speed and direction. Wind direction originates at zero. Cold fronts are indicated by vertical dashed lines. Yellow shaded areas represent neap tidal cycles for which the data in Table 2 were calculated. Green shaded areas represent spring tidal cycles for which the data in Table 2 were calculated.

Figure 2-16: Correlation of predicted and observed tidal range (see Table 2 for data).

Mean 3-day tidal range during each spring/neap tide for March 2002. Observed inner station versus predicted Dover Bridge data are plotted in red, observed outer station versus predicted Dover Bridge data are plotted in blue.

Figure 2-17: Variability in the hydraulic gradient of Council Creek marsh: Top panel shows the full record of hydraulic gradient data. The bottom panel is an example of a brief period of the data to illustrate some of the causes of variability. Long-term mean of the hydraulic gradient data is represented by a red dashed line.

Figure 2-18: Example of flooding by Choptank estuary: Inner station water level, cm (red line), outer station water level, cm (blue line). The blue shaded areas are when the hydraulic gradient is positive (i.e. net flow out of Council Creek), while the yellow shaded are is when the hydraulic gradient is negative (i.e. net flow of water into Council Creek).

Figure 2-19: Correlation between wastewater discharge and hydraulic gradient. Data are plotted as 7-day averages for the weeks 18 November 2001 to 16 June 2002 showing a significant positive correlation.

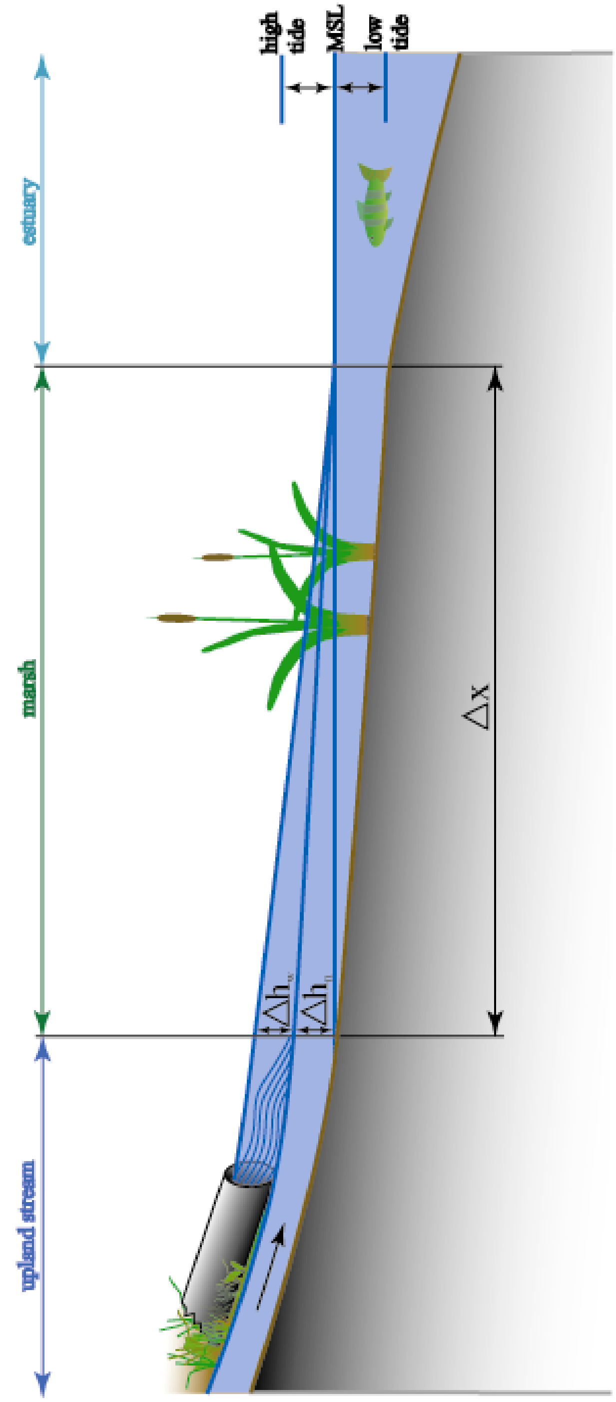


Figure 2-1

Choptank Basin

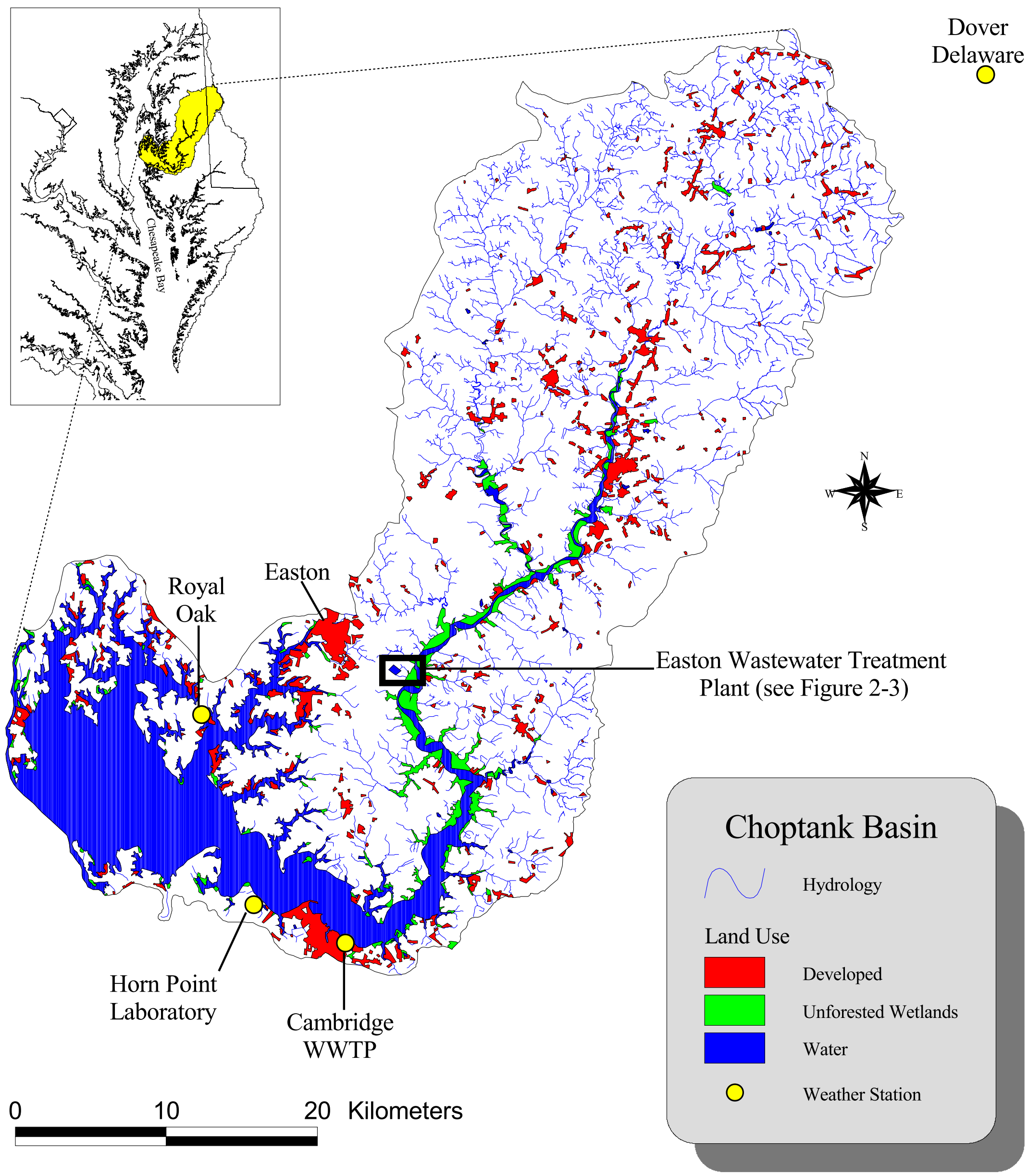


Figure 2-2

Council Creek Basin

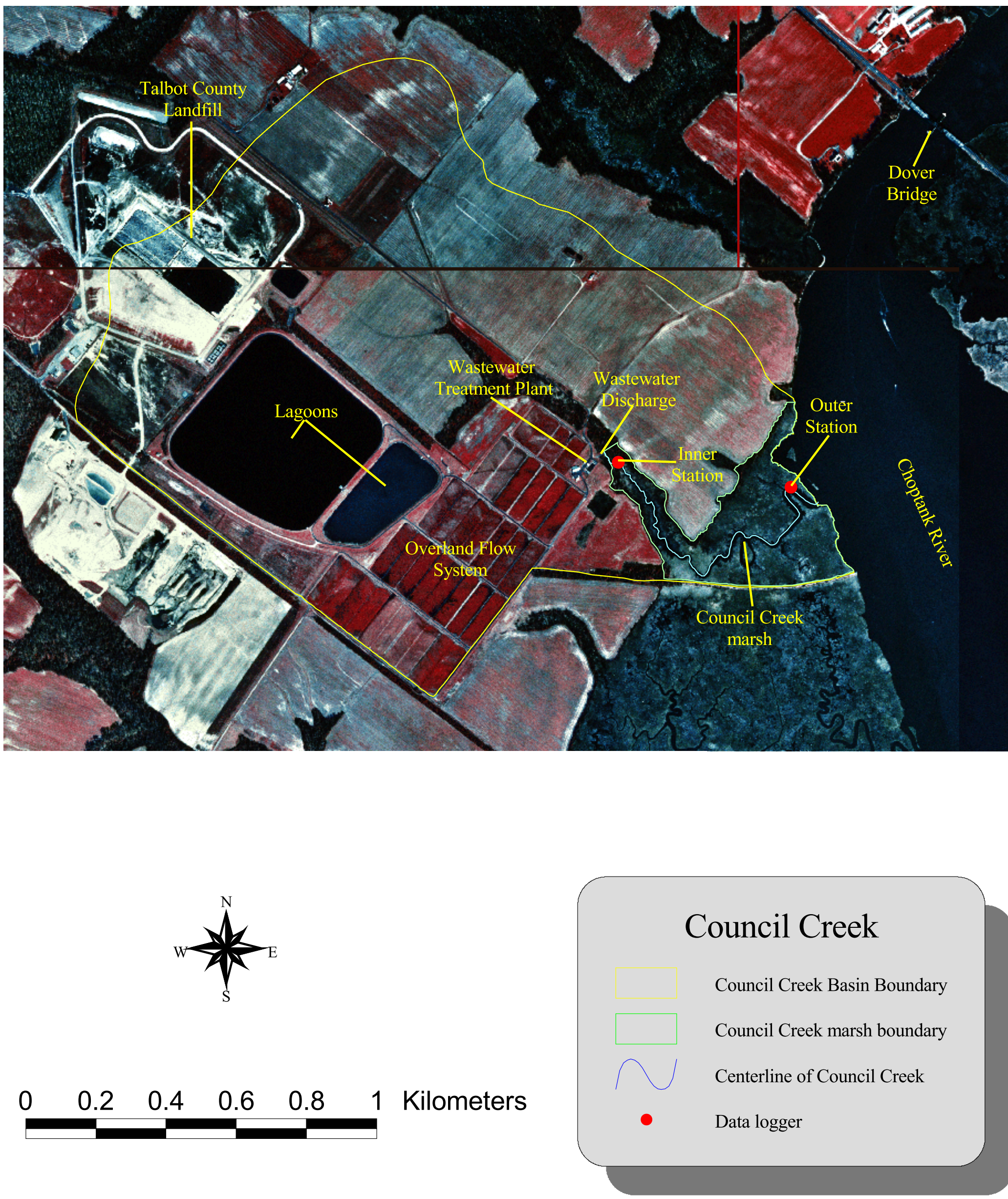
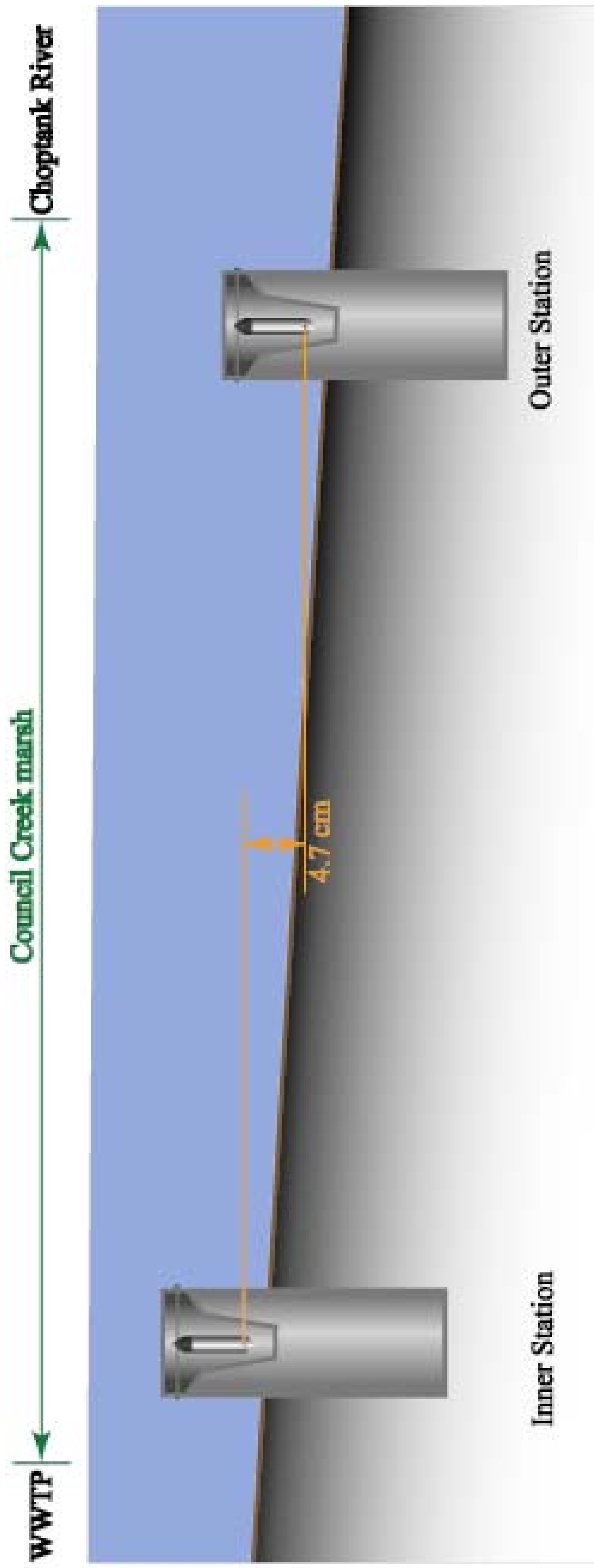
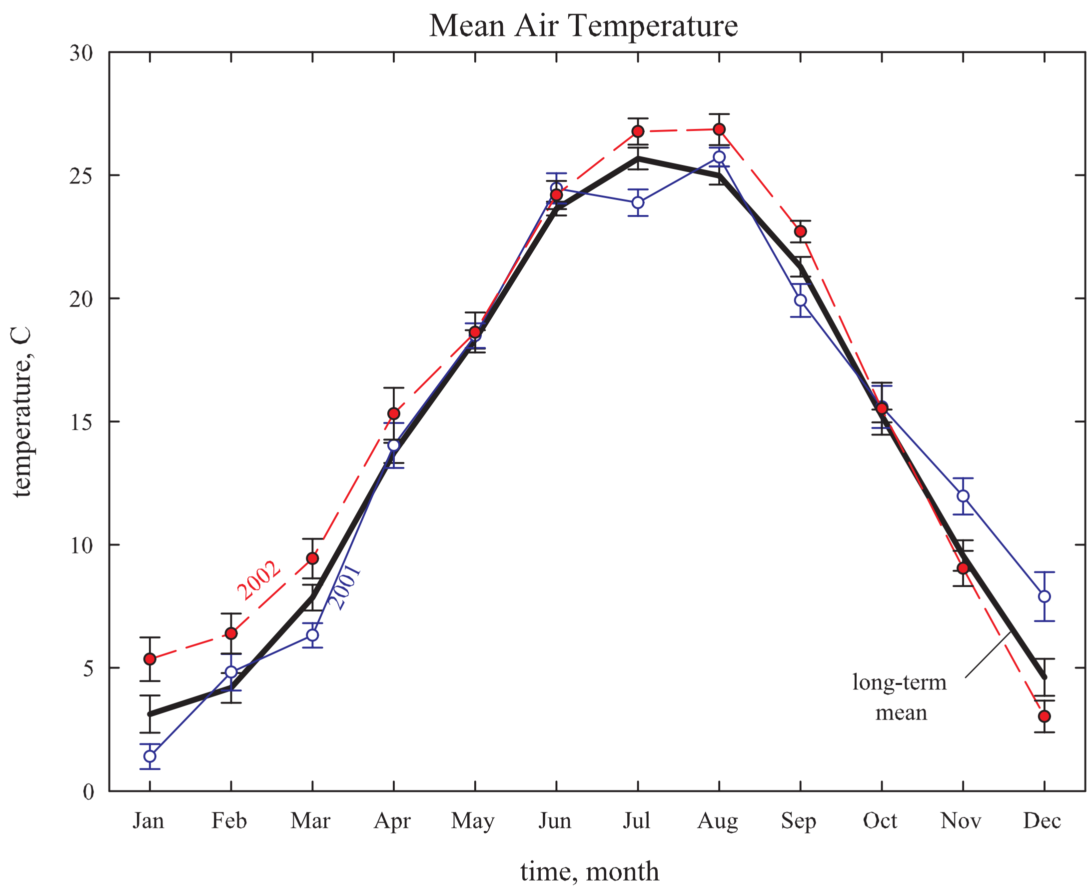


Figure 2-3



Choptank River
Council Creek marsh
WWTP
4.7 cm
Outer Station
Inner Station

Figure 2-4



Mean Air Temperature
30
25
20
15
10
5
0
temperature, C
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
time, month
2002
2001
long-term
mean

Figure 2-5

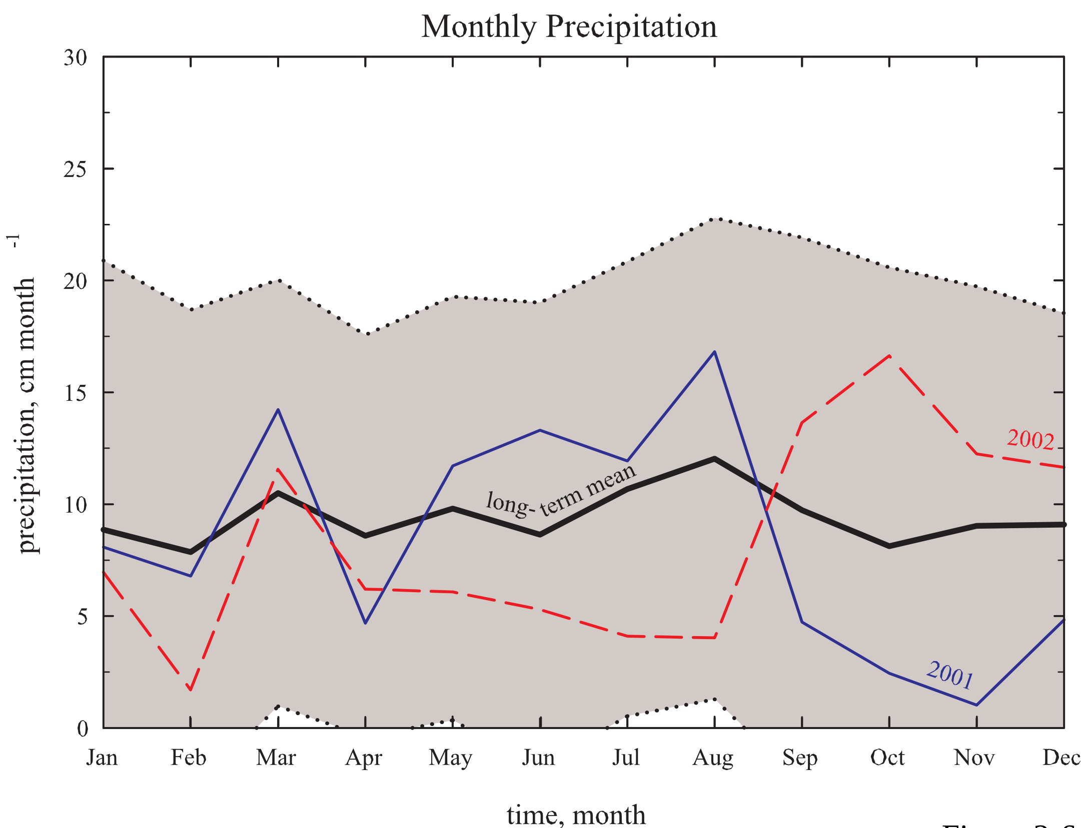

Monthly Precipitation
30
25
20
15
10
5
0
precipitation, cm month -1
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
time, month
long- term mean
2002
2001

Figure 2-6

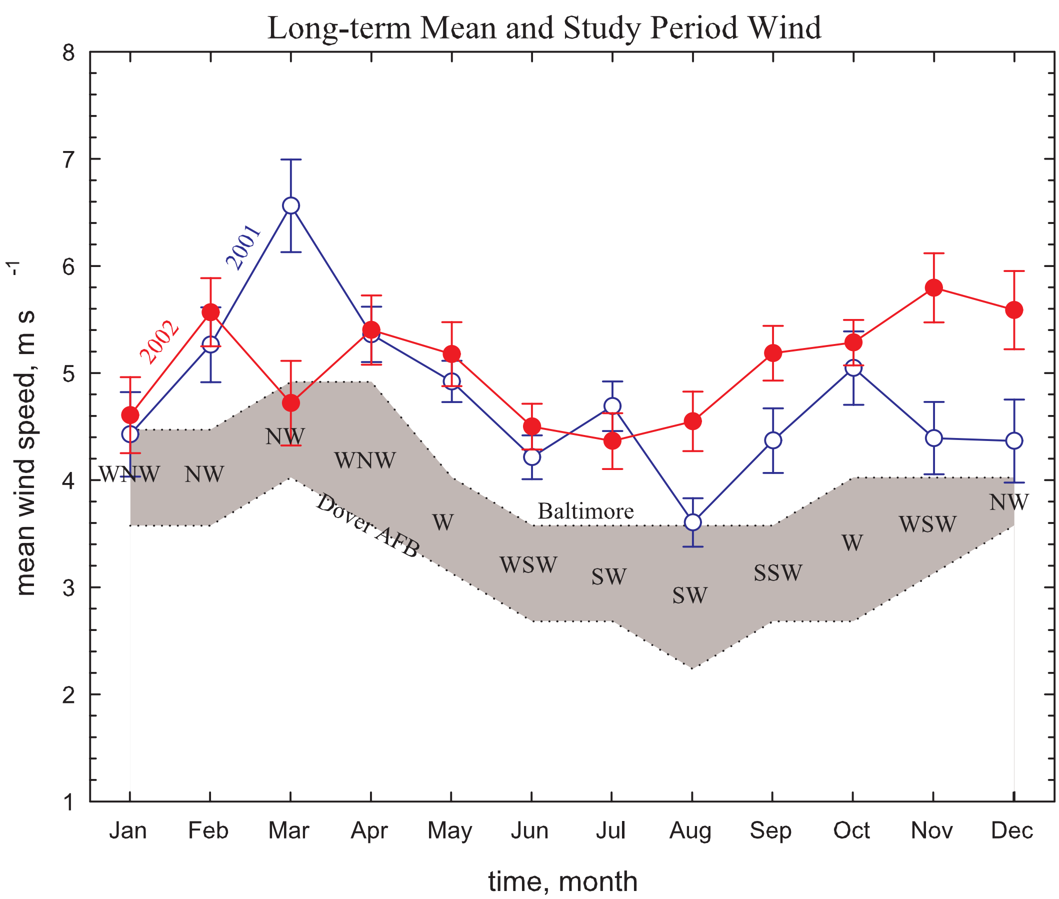

Long-term Mean and Study Period Wind
mean wind speed, m s -1
time, month
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
1
2
3
4
5
6
7
8
2001
2002
WNW
NW
NW
WNW
W
WSW
SW
SW
SSW
W
WSW
NW
Baltimore
Dover AFB

Figure 2-7

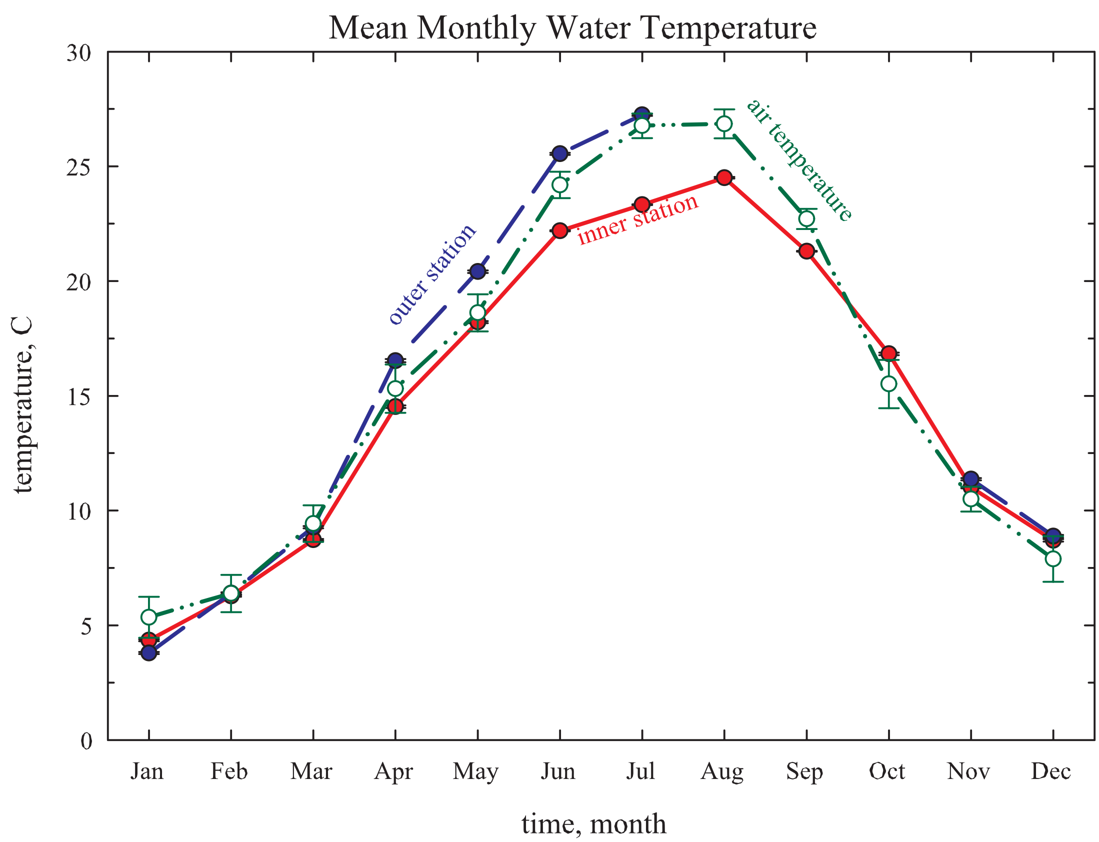
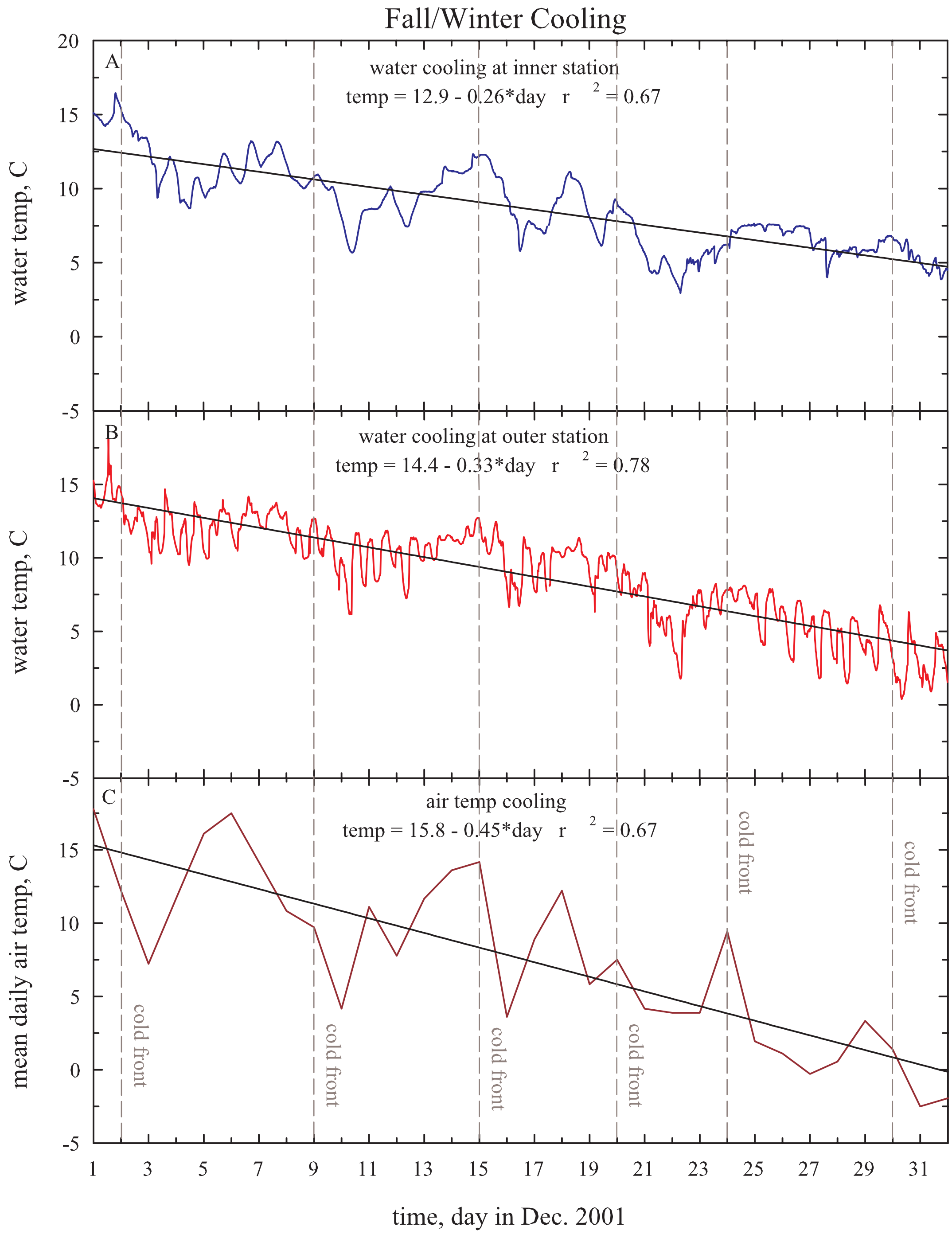

Mean Monthly Water Temperature
30
25
20
15
10
5
0
temperature, C
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
time, month
outer station
inner station
air temperature

Figure 2-8



Fall/Winter Cooling
A
water cooling at inner station
temp = 12.9 - 0.26*day r 2 = 0.67
water temp, C
B
water cooling at outer station
temp = 14.4 - 0.33*day r 2 = 0.78
water temp, C
C
air temp cooling
temp = 15.8 - 0.45*day r 2 = 0.67
mean daily air temp, C
cold front
cold front
cold front
cold front
cold front
cold front
time, day in Dec. 2001

Figure 2-9

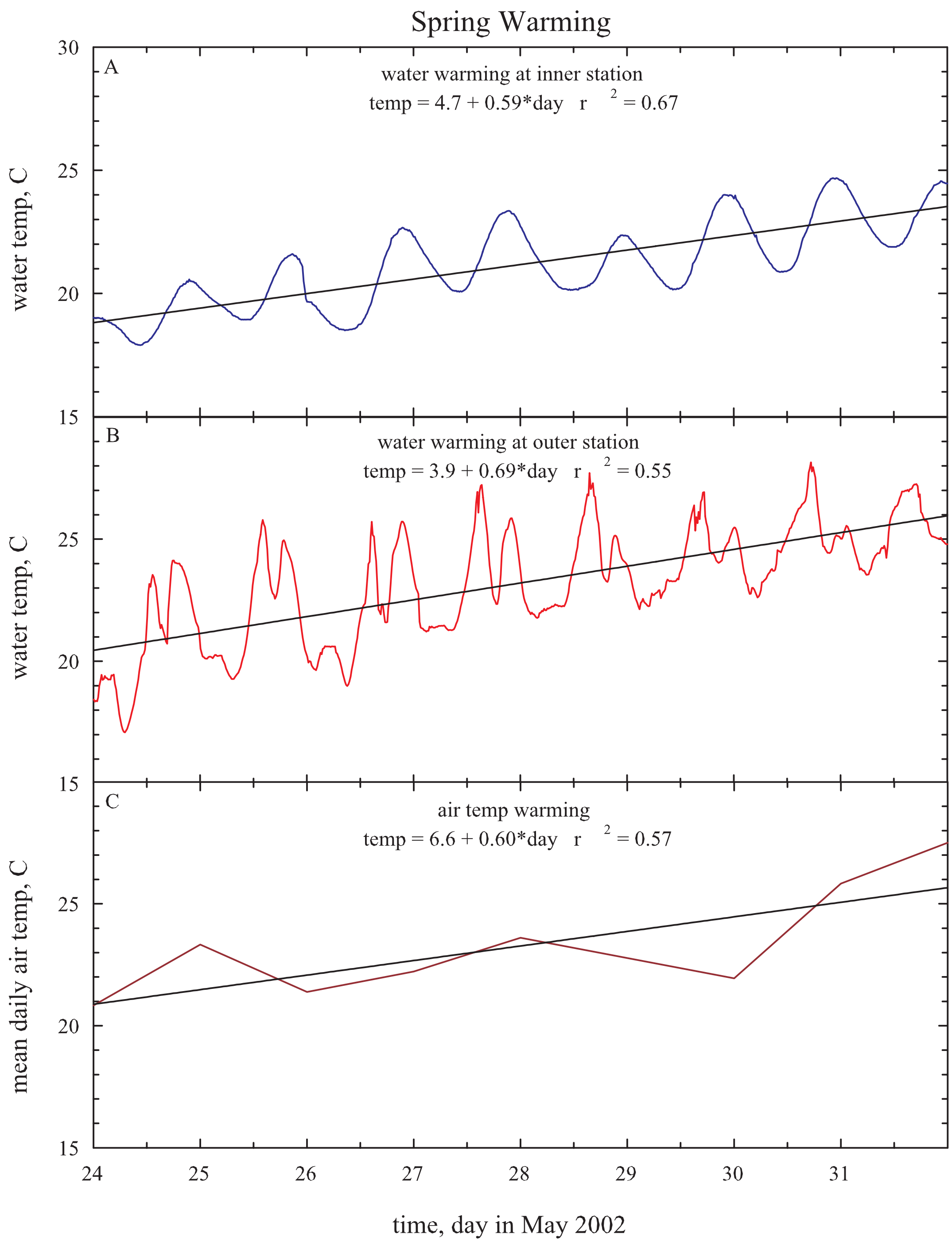

Spring Warming
A
water warming at inner station
temp = 4.7 + 0.59*day r 2 = 0.67
water temp, C
30
25
20
15
B
water warming at outer station
temp = 3.9 + 0.69*day r 2 = 0.55
water temp, C
25
20
15
C
air temp warming
temp = 6.6 + 0.60*day r 2 = 0.57
mean daily air temp, C
25
20
15
24
25
26
27
28
29
30
31
time, day in May 2002

Figure 2-10

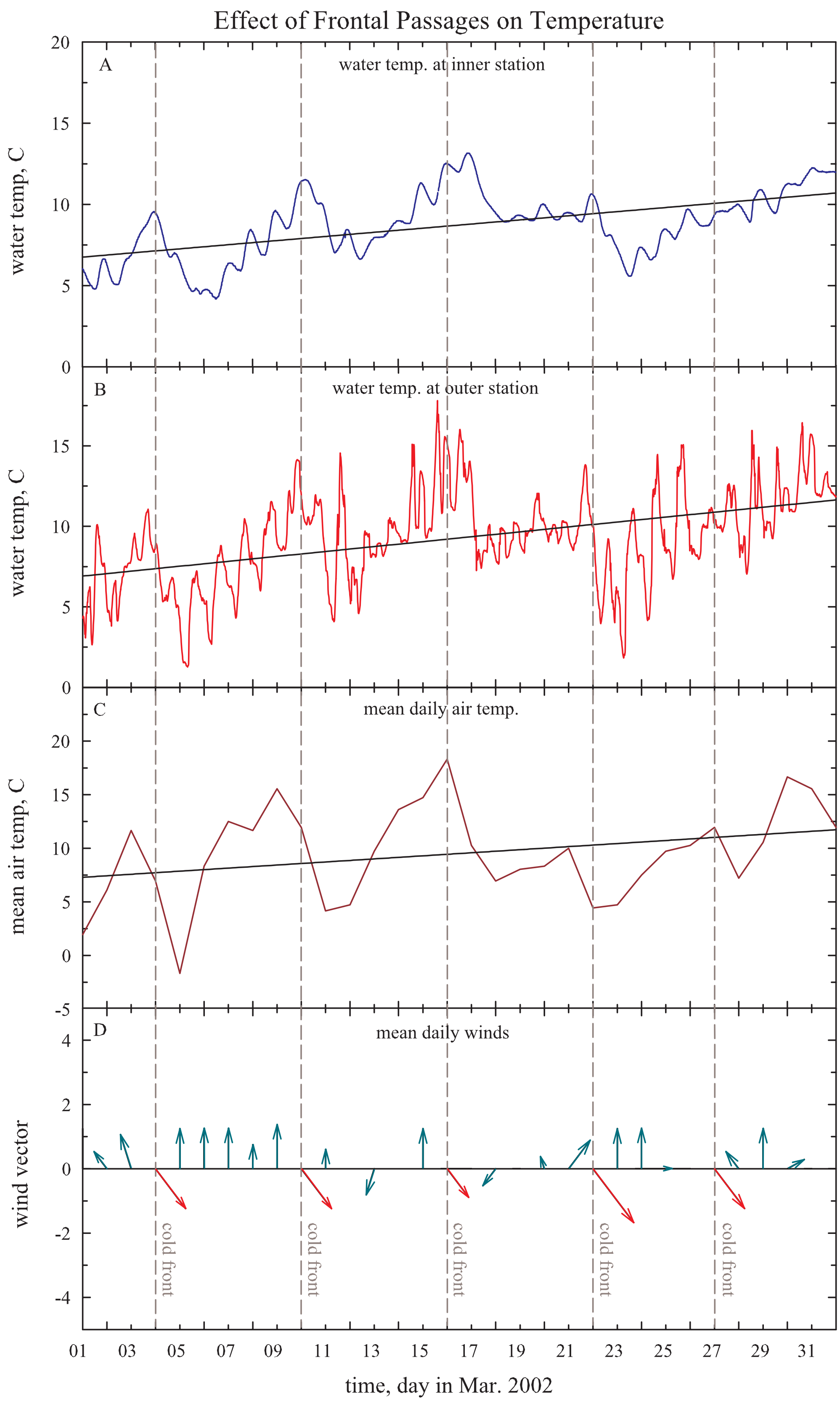

Effect of Frontal Passages on Temperature
A
water temp. at inner station
B
water temp. at outer station
C
mean daily air temp.
D
mean daily winds
water temp, C
water temp, C
mean air temp, C
wind vector
cold front
cold front
cold front
cold front
cold front
time, day in Mar. 2002

Figure 2-11

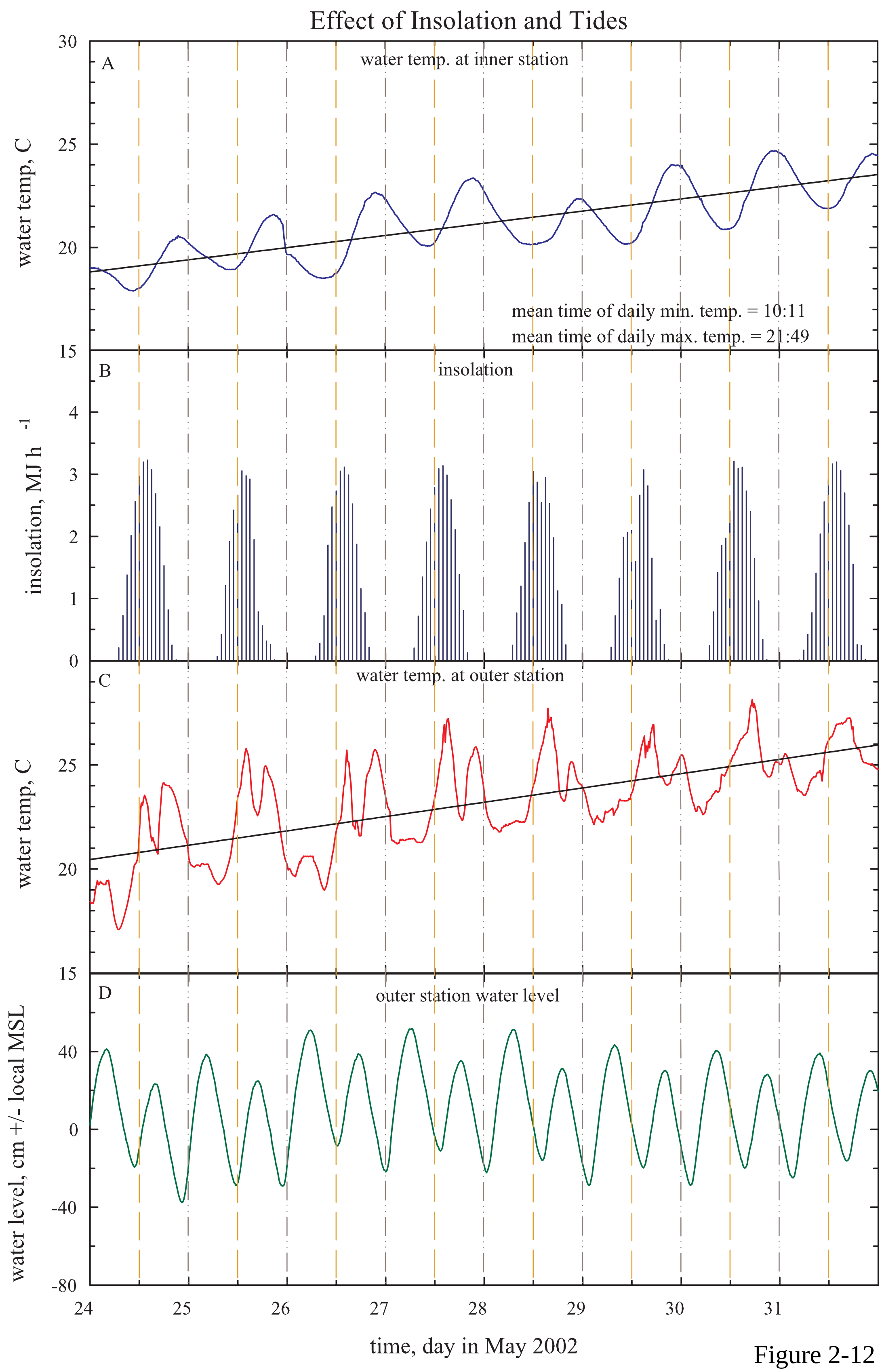
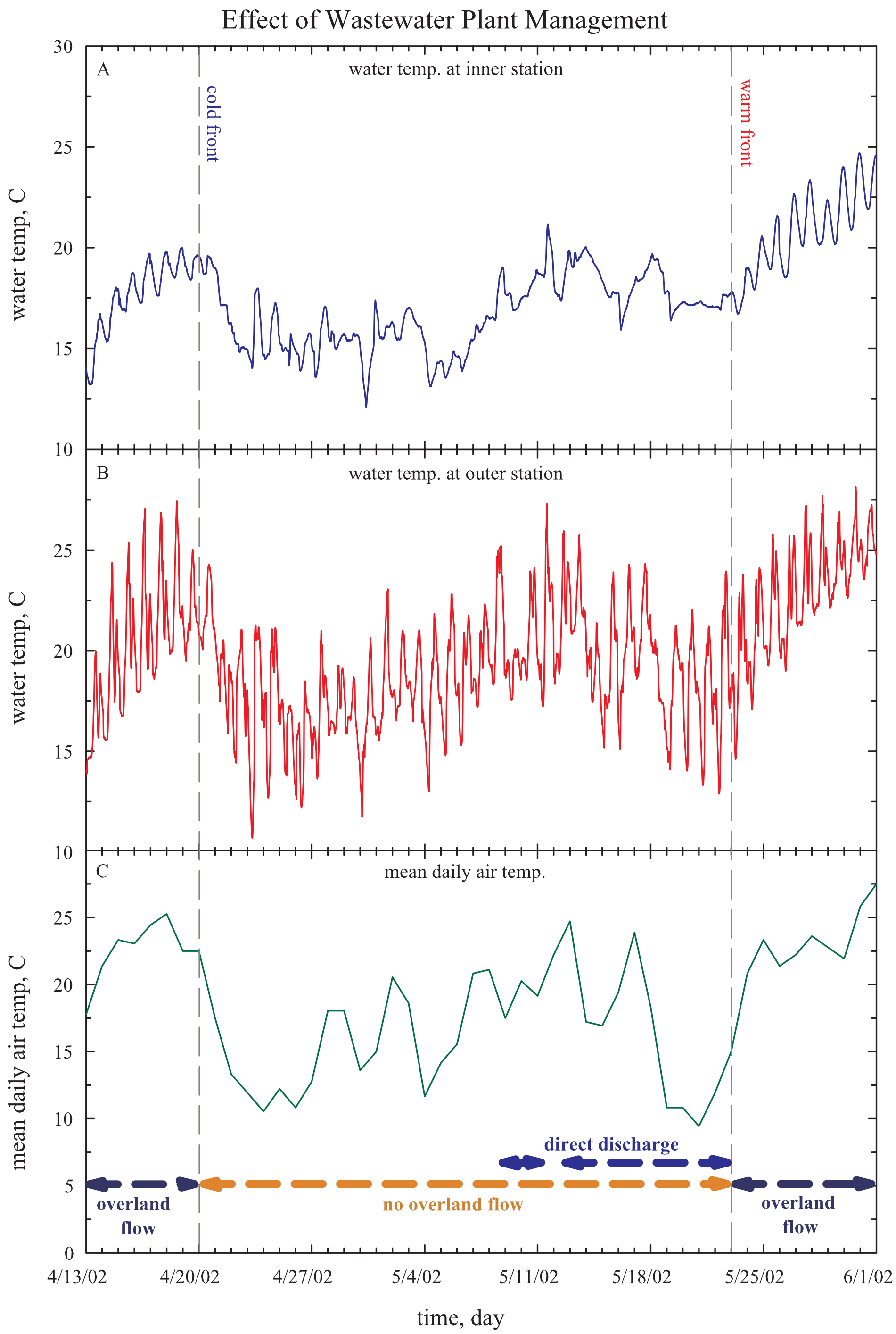

Effect of Insolation and Tides
A
water temp. at inner station
mean time of daily min. temp. = 10:11
mean time of daily max. temp. = 21:49
water temp, C
B
insolation
insolation, MJ h -1
C
water temp. at outer station
water temp, C
D
outer station water level
water level, cm +/- local MSL
time, day in May 2002

Figure 2-12



Effect of Wastewater Plant Management
A
water temp. at inner station
cold front
warm front
water temp, C
B
water temp. at outer station
water temp, C
C
mean daily air temp.
mean daily air temp, C
direct discharge
overland flow
no overland flow
overland flow
4/13/02
4/20/02
4/27/02
5/4/02
5/11/02
5/18/02
5/25/02
6/1/02
time, day

Figure 2-13

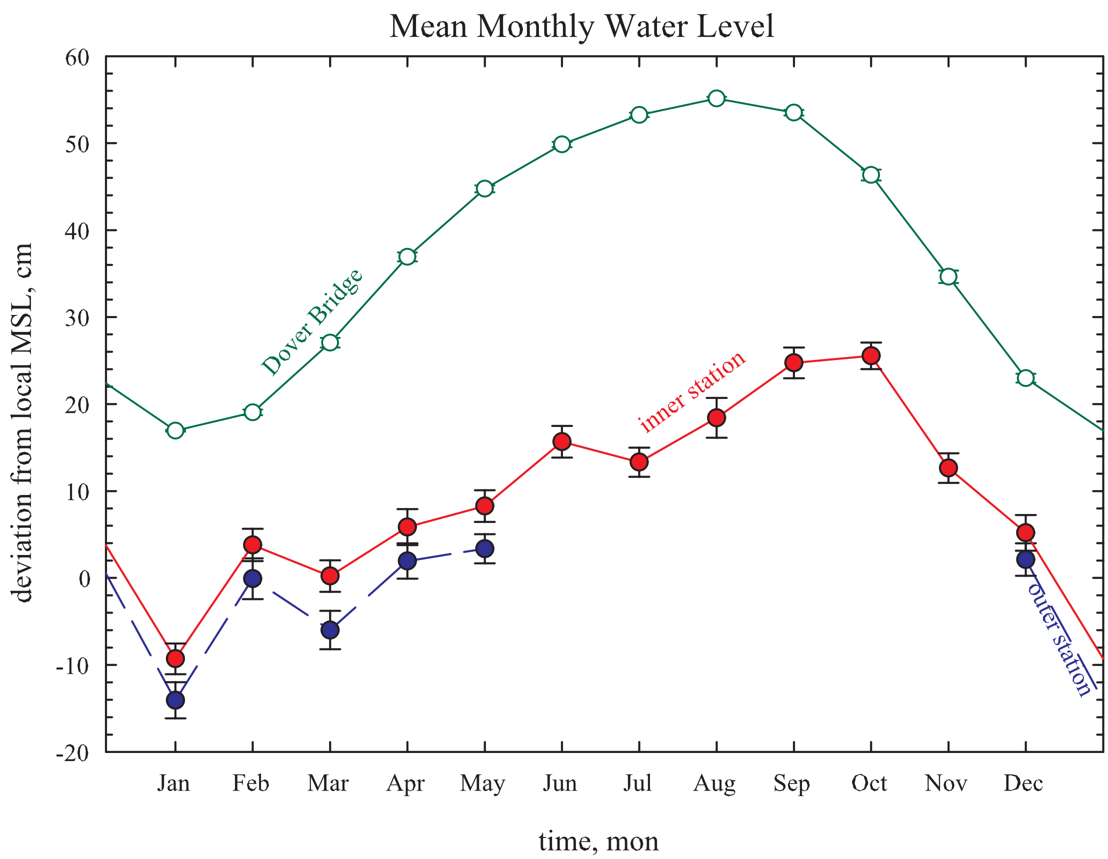
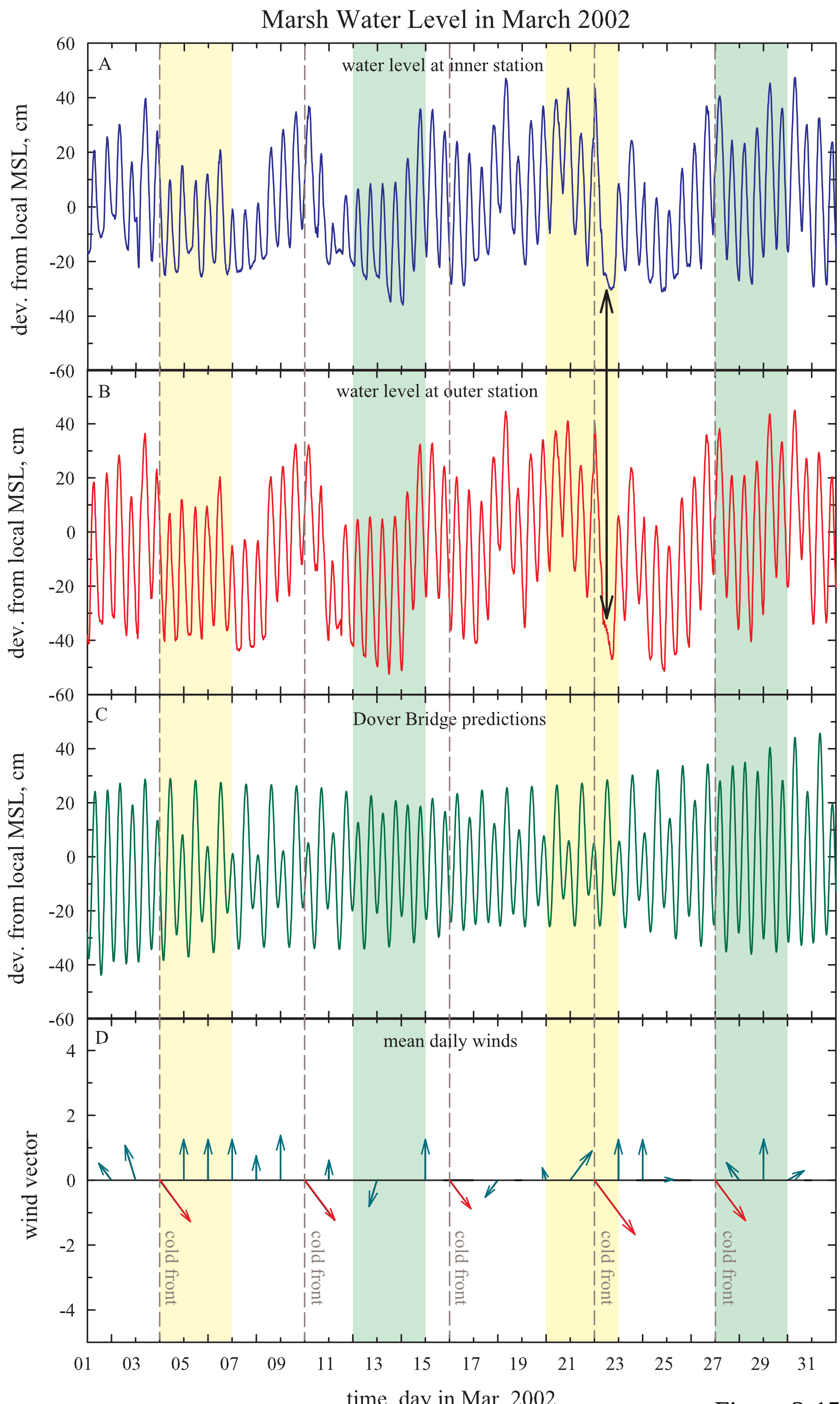

Mean Monthly Water Level
deviation from local MSL, cm
time, mon
Dover Bridge
inner station
outer station
60
50
40
30
20
10
0
-10
-20
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec

Figure 2-14



Marsh Water Level in March 2002
A
water level at inner station
B
water level at outer station
C
Dover Bridge predictions
D
mean daily winds
dev. from local MSL, cm
wind vector
60
40
20
0
-20
-40
-60
4
2
-2
-4
cold front
cold front
cold front
cold front
cold front
01
03
05
07
09
11
13
15
17
19
21
23
25
27
29
31
time, day in Mar. 2002

Figure 2-15

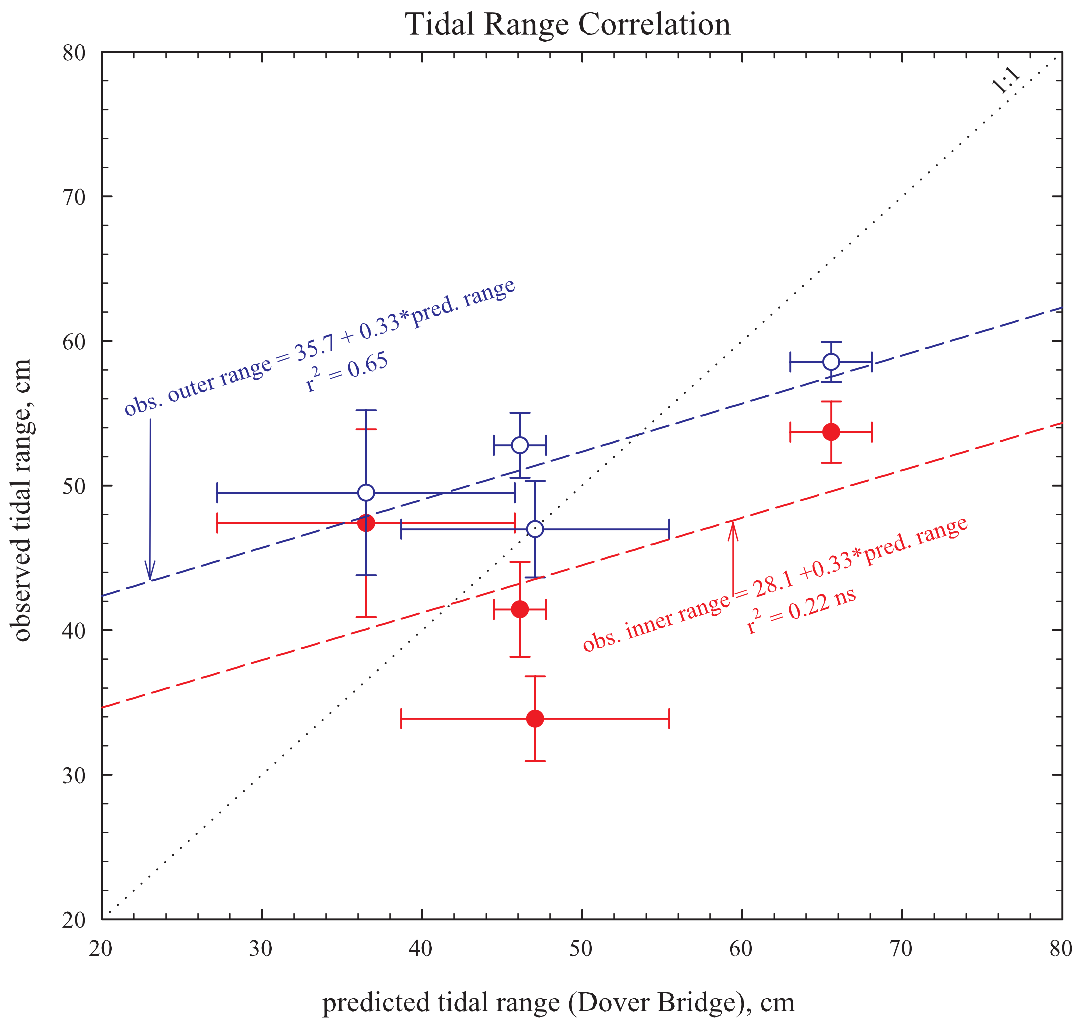
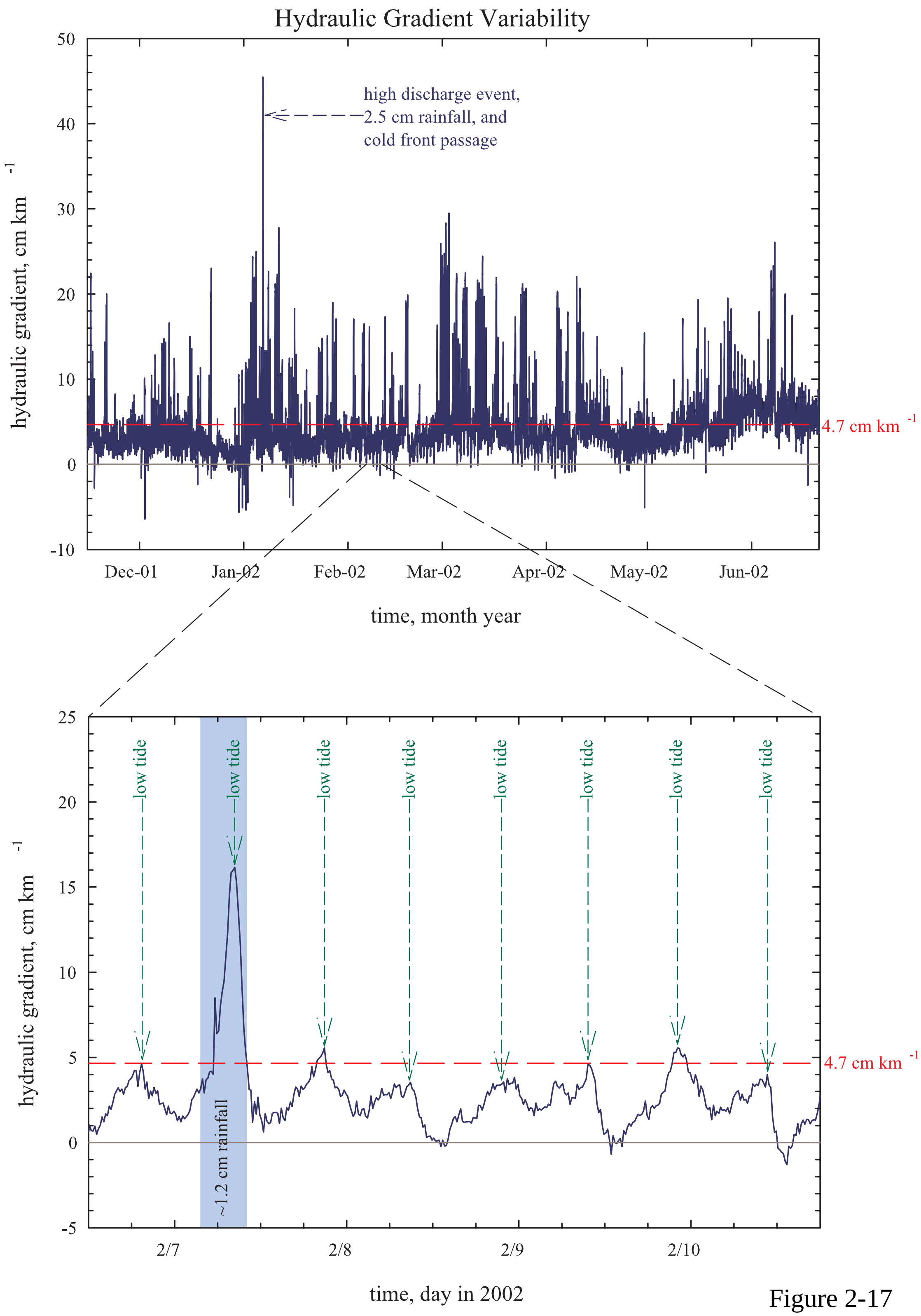

Tidal Range Correlation
observed tidal range, cm
predicted tidal range (Dover Bridge), cm
80
70
60
50
40
30
20
20
30
40
50
60
70
80
1:1
obs. outer range = 35.7 + 0.33*pred. range
r2 = 0.65
obs. inner range = 28.1 +0.33*pred. range
r2 = 0.22 ns

Figure 2-16



Hydraulic Gradient Variability
high discharge event,
2.5 cm rainfall, and
cold front passage
hydraulic gradient, cm km -1
4.7 cm km -1
Dec-01
Jan-02
Feb-02
Mar-02
Apr-02
May-02
Jun-02
time, month year
low tide
~1.2 cm rainfall
2/7
2/8
2/9
2/10
time, day in 2002

Figure 2-17

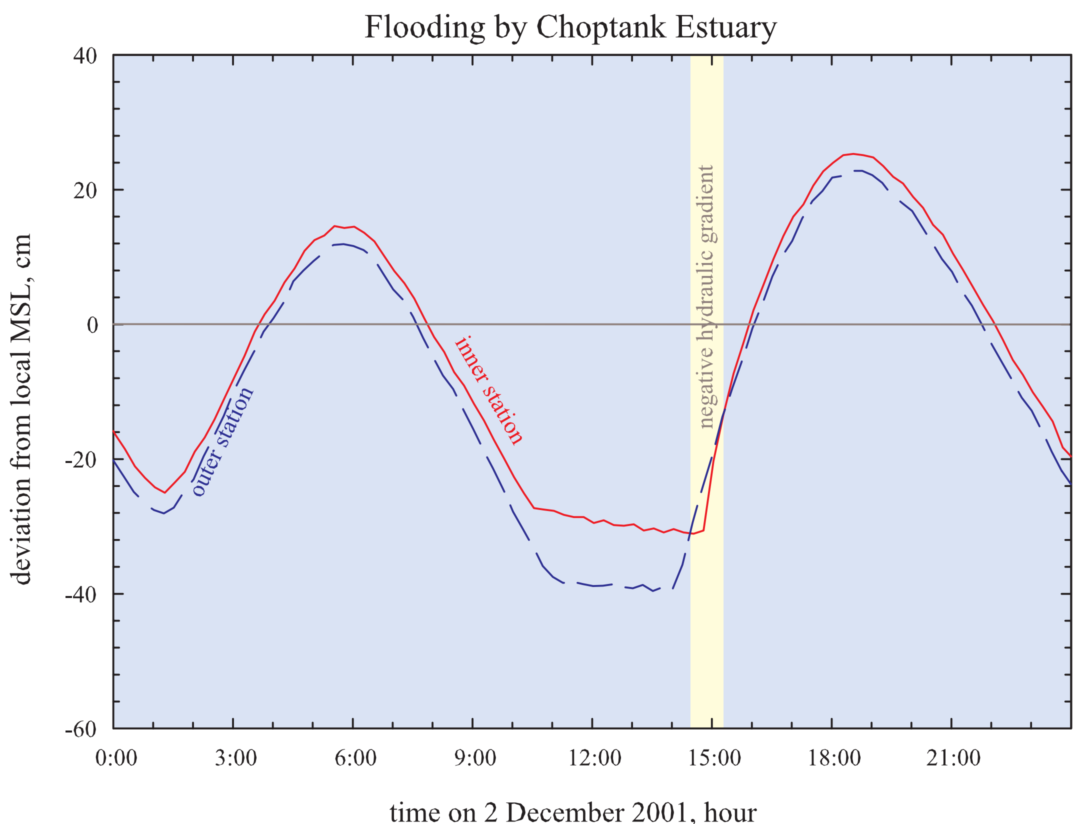

Flooding by Choptank Estuary
40
20
0
-20
-40
-60
deviation from local MSL, cm
0:00
3:00
6:00
9:00
12:00
15:00
18:00
21:00
time on 2 December 2001, hour
outer station
inner station
negative hydraulic gradient

Figure 2-18

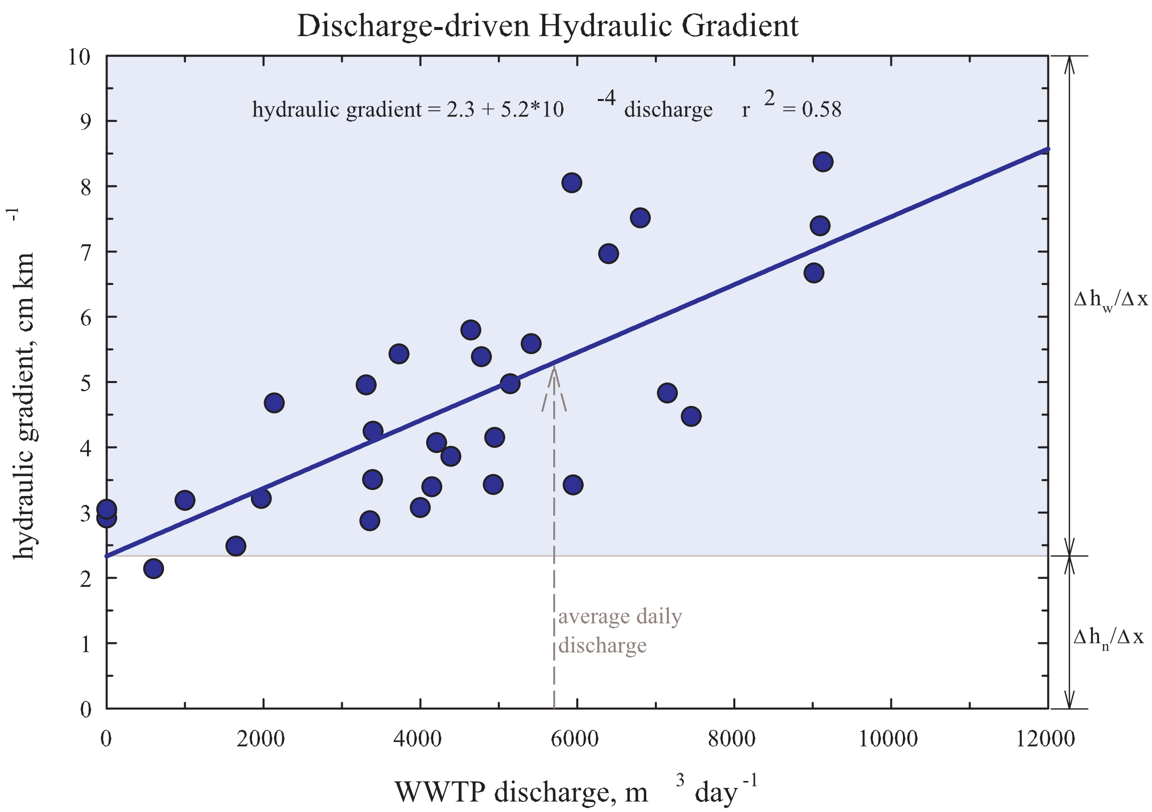

Discharge-driven Hydraulic Gradient
hydraulic gradient = 2.3 + 5.2*10 -4 discharge r 2 = 0.58
hydraulic gradient, cm km -1
WWTP discharge, m 3 day -1
average daily discharge
Δhw/Δx
Δhn/Δx
0
1
2
3
4
5
6
7
8
9
10
0
2000
4000
6000
8000
10000
12000

Figure 2-19

Chapter 3

PROCESSING OF WASTEWATER NUTRIENTS IN A NATURAL OLIGOHALINE MARSH

Abstract

Interception of anthropogenic nutrients by coastal wetlands can potentially contribute to water quality improvements. The large amounts of plant biomass and accumulated sediments can potentially trap N and P via plant uptake, sorption, denitrification, and burial. Wetlands receiving human wastewater face a steady flow of high nutrients, and the hypothesis tested here is that elevated N and P concentrations via wastewater discharged into Council Creek marsh are significantly attenuated via biological, chemical and physical processes occurring within the marsh. Council Creek marsh (0.16 km^2) on the Delmarva Peninsula receives an average wastewater flow of 5716 m^3 d^{-1} (1.53 MGD) from the town of Easton, MD. The wastewater plant, located within Council Creek basin (2.20 km^2), employs a 0.27 km^2 overland flow system to provide partial tertiary treatment prior to discharge into a stream, which flows into the marsh.

Council Creek marsh appears to function as an additional tertiary treatment component for wastewater N and a seasonal filter for wastewater P. Reductions in N concentrations of approximately 50% of N in wastewater effluent were seen during the height of the growing season due to dilution by Choptank River water during the seasonally higher tides. This reduction resulted in TN concentrations being discharged to the Choptank River that are less than 2 times higher than average Choptank River concentrations. During winter, some of the organic N was mobilized and exported to the

Choptank. The pattern for P was not as clear based on the low tide sampling. Total P concentrations at the creek mouth suggested net gains in P concentrations early in the growing season, perhaps due to the seasonally higher sea level, and little effect on wastewater P late in the growing season and into the fall. Furthermore, the transect data suggested that the TP concentrations are augmented by marsh P by late fall to early winter.

Introduction

Anthropogenic activity has significantly affected estuarine and coastal waters (Vitousek et al. 1997). Nitrogen (N) and phosphorus (P) inputs from both diffuse and point sources are one of the major causes of degrading water quality, because these nutrients stimulate the growth of algae in natural waters (Fisher et al. 1988, Valiela et al. 1990). Eutrophication resulting from these inputs has severe consequences including increased accumulation of phytoplankton, macroalgal, and epiphyte biomass, bottom water hypoxia/anoxia, and increased probability of fish kills (Malone et al 1988, Smith et al. 1999).

Point sources of nutrients from human wastewater have long been linked to eutrophication of lakes (e.g., Schindler et al. 1977) and coastal waters (e.g., D'Elia et al. 1986). In recent decades, treatment wetlands have been employed as a management strategy to reduce point source nutrient loading to estuaries (Kadlec and Knight 1996). The ultimate goal in using wetlands to reduce nutrient transport to estuarine waters is to use natural and more cost effective methods to reduce anthropogenic impacts on coastal resources rather than relatively expensive tertiary treatment. However, this use of marshes has been obscured by the continuing debate over the whether tidal marshes act as nutrient sources or sinks (Nixon 1980, Correll et al. 1992, Childers 2000).

An important point source is the discharge of municipal wastewater to coastal ecosystems. Disposal of human wastewater has a long history, dating as far back as Greek civilization when wastewater was used to irrigate orchards (Stoddard et al 2002). Increased human population densities in cities and towns and development of urban areas necessitated the development of the urban water cycle, creating water supply demands and consequently generating wastewater flows. Until about 100 years ago, the primary goal of wastewater disposal was removal from the municipality to reduce the spread of water-borne disease (Stoddard et al. 2002), and this typically meant that wastewater was discharged to rivers and estuaries. With continually rising human populations, degradation of the water supply of downstream municipalities soon became an issue (Stoddard et al. 2002). To address these problems, municipal wastewater treatment methods were developed and implemented for the purpose of reducing contaminants and pathogens. However, the increased costs of more effective yet increasingly complex wastewater treatment are important limitations on the widespread use of advanced treatment (Kadlec and Knight 1996).

Wastewater treatment methods are classified into three general categories (Kadlec and Knight 1996, Stoddard et al. 2002). In increasing complexity, effectiveness, and cost, the methods are primary, secondary, and tertiary treatment. There are, however, different approaches and varied levels of treatment within each group. Primary treatment methods employ settling and screening of wastewater to remove solids, but suspended organic matter, often with a high biological oxygen demand (BOD), is still discharged in effluent water. Secondary treatment of wastewater utilizes biological processes such as bacterial metabolism to metabolize this residual organic matter (remove the BOD), yet high concentrations of the original nutrients in the human waste (e.g. N and P) remain in solution. Tertiary or advanced wastewater treatment is aimed at removal of these wastewater nutrients prior to discharge into receiving water bodies. Tertiary or advanced wastewater treatment can be achieved through costly, high maintenance, chemical or biological processes, but utilization of natural physical, chemical, and biological processes can sometimes provide a cost-effective alternative (Smith et al. 2000).

The utilization of wetlands as an alternative method of wastewater treatment is becoming increasingly common (Kadlec and Knight 1996, Kadlec and Reddy 2001). Because they are highly productive ecosystems (Kadlec and Knight 1996, Mitsch and Gosselink 2000), wetlands have the potential to process and remove wastewater contaminants via macrophytic nutrient uptake, burial in sediments, and denitrification (Merrill and Cornwel 2000). Compared to more expensive alum additions for phosphorus removal or the oxic/anoxic steps required for biological nitrogen removal, wetlands used to remove the contaminants from human wastewater can potentially provide a low cost alternative to these or other tertiary wastewater treatment methods (Dolan et al. 1981, Kadlec and Knight 1996, Smith et al. 2000, Sartoris et al. 2000).

The focus of this chapter of my thesis is wastewater nutrient immobilization and processing as the wastewater moves through an oligohaline marsh. Council Creek marsh on the Choptank estuary receives an average of 5716 m^3 d^{-1} (1.53 MGD) of wastewater, transporting average concentrations of 732 μM N (10.3 mg l^{-1}) and 87 μM P (2.7 mg l^{-1}) from the town of Easton, Maryland on the Delmarva Peninsula (Figure 3-1). The plant uses primary and secondary treatment in a lagoon system, plus a partial tertiary treatment in a grassed overland flow system. I hypothesize that elevated N and P concentrations in wastewater discharged into Council Creek marsh are significantly attenuated via biological, chemical, and physical processes occurring within the marsh. Evidence to support or reject this hypothesis should emerge from two sets of water quality data collected within the Council Creek: (1) concentrations of N and P in water exiting the Creek in to the Choptank River at low tide after processing by the marsh and (2) the distribution of N and P in the creek as it exchanges tidally with the marsh.

Study Site Description

Council Creek is located in the Choptank River basin on the Delmarva Peninsula (eastern shore of the Chesapeake Bay, Figure 3-1). In the top panel of Figure 3-2, Council Creek basin (within the yellow polygon) has an area of 2.2 km^2 and flows through a 0.16 km^2 natural oligohaline marsh (green polygon in both panels of Figure 3-2) before flowing into the Choptank River. The town of Easton, Maryland is located approximately 6.3 km north/northwest of Council Creek (Figure 3-1), and the municipal wastewater treatment plant is located within the Council Creek watershed (Figure 3-2). Additionally, part of the Talbot County, Maryland landfill is located within the Council Creek watershed.

The treatment plant consists of three components. The first component (primary and secondary treatment) is the lagoon system, which is used to settle out solids and oxidize organic matter (Figure 3-2, top panel). Water retention time in the lagoons is approximately 90 days. The second component (partial tertiary treatment) is the overland flow system (Figure 3-2, top panel), which consists of 0.28 km^2 of grass terraces (Cronshaw et al. 1990). Water from the lagoons is spray-irrigated onto the terraces to allow plant and soil processes to remove nutrients from the wastewater, and the grass is periodically harvested to remove plant-accumulated nutrients. During this process the treatment plant has the ability to cease all discharge, or to by-pass the overland flow fields to allow them to dry.   The third component is the treatment facility (Figure 3-2, top panel), where water is first chlorinated to kill any remaining pathogens, de-chlorinated, and finally oxygenated before being discharged into Council Creek. The entire treatment system is designed to discharge a maximum of about 8900 m^{-3} day^{-1} (2.35 MGD) of treated wastewater (Cronshaw et al. 1990). After discharge into Council Creek, the wastewater flows 1.2 km through the meandering creek before entering the Choptank River (orange line, bottom panel of Figure 3-2). For comparison with a marsh receiving no direct wastewater flows, I also sampled in Little Creek marsh, directly across the Choptank River from Council Creek marsh (Figures 3-1 and 3-3).

Methods

Automated sampling of low tide discharge

An ISCO model 3700 automated sampler collected samples of water flowing out of the mouth of Council Creek on ebbing tides. The sampler was set up near the mouth of Council Creek approximately 13 m from the creek bank platform (Figure 3-2, bottom panel). The platform was constructed as a wooden frame (~1 m by 1.2 m) that was fitted around and secured to two plastic 55-gallon drums to allow the platform to float on high tides. To secure the platform, it was chained to two earth anchors augured into the marsh surface and stabilized with 2.5 m long treated lumber poles driven into the marsh surface approximately 0.75 m on each side of the platform.

The ISCO sampler, secured to the platform, was powered by a 12V battery charged by a PhotoComm Inc. DVM-12 photovoltaic cell. Samples were pumped through a 15 m long, 0.95 cm inside diameter, flexible, PVC tube secured to a 20 cm diameter schedule 80 PVC pipe installed in the creek bottom (see Chapter 2, Figure 2-4 for outer station pipe installation). To prevent algal build up and potential damage to the tubing, it was buried approximately 15-20 cm below the marsh surface and graded toward the creek to decrease the amount of water remaining in the tube between sample times.

Samples were taken 1-hour before every predicted low tide at nearby Dover Bridge from 25 May through 19 November 2002 (Eastern Standard Time, see Chapter 2 for tidal information). The goal of this sampling strategy was to collect the mix of wastewater and natural waters draining from the marsh, which had been exposed to biological and chemical processing by the marsh. Sample timing was determined by comparing 62 observed and predicted tides. Predicted low tide at Dover Bridge and observed low tide at the ISCO intake tube occurred on average at the same time with observed low tide occurring both before and after predicted low tide due to wind tide effects (see Chapter 2). Since the difference between the two was usually not more than 30 minutes, the sampler was set to take samples 1 hour before predicted low tide. At each sample time, samples were collected by pre-purging the entire tube twice and then pumping 400 ml of water into the appropriate bottle. Two consecutive ebb tide samples were composited in each bottle yielding on average one bottle per day. Bottles were collected on average every 7 days to minimize contamination and evaporative losses.

Samples were transported to Horn Point Laboratory and refrigerated at 4˚C until they could be analyzed for TN and TP, usually within 10 days. If analyses could not be completed within 10 days of collection, samples were frozen at –15˚C until analyses were completed, usually within 30 days. Conductivity was measured for each sample bottle using a Yokogawa model SC-82 conductivity meter calibrated using a 100 µmho cm^{-1} NaCl standard to measure conductivity and temperature for all samples. These values were then converted to conductivity standardized at 25˚C using an empirically determined temperature correction, hereafter referred to as cond (25) (Figure 3-4). Samples were also processed for TN and TP using the oxidation procedure described by Valderrama (1981). After persulfate digestion, the TP aliquots were analyzed using a colorimetric PO_4 procedure (Strickland and Parsons 1974), and the TN aliquots were analyzed for NO_3 on a four channel Technicon Auto Analyzer II by Horn Point Laboratory analytical services (Lane et al. 2000). Inorganic N and P were not measured (only TN and TP) because of the unrefrigerated storage in the ISCO sampler for up to one week between sample retrievals.

Transects of Council Creek

Ten water quality transects were collected in Council Creek from 18 January 2001 to 16 September 2002. Transects were collected to obtain detailed longitudinal descriptions of the distributions of N and P through Council Creek, and in particular to estimate marsh nutrient processing during different seasons, tidal stages, and WWTP discharge conditions. One comparative transect was also collected in Little Creek, directly across the Choptank from Council Creek (Figures 3-1 and 3-3) on 17 September

2002 to examine water quality in a marsh with no direct wastewater discharge.

Transects consisted of 12-15 grab samples of water collected in labeled, acid-washed, Nalgene polypropylene bottles. All sample locations were georeferenced using a Garmin GPS III-plus to <15 m accuracy, and points were later downloaded to a personal computer using MapSource software (v.3.02). For each transect, samples were collected from four fixed points and 8-11 varied locations down the creek. The fixed points included: (1) a sample collected from the stream above the WWTP oufall (upstream of tidal influence), (2) a sample at the wastewater outfall, (3) a sample at the mouth of Council Creek, and (4) a sample of Choptank River water (Figure 3-2, bottom panel). The remainder of the samples was collected from the centerline of Council Creek downstream of the WWTP and consisted of a mixture of streamflow, wastewater flow and Choptank River water (orange line in bottom panel of Figure 3-2). Samples were collected by canoeing from the WWTP to the Choptank River, where the first sample was collected after allowing a minimum of ten minutes for any sediment re-suspension from the canoe wake to settle. The Yokogawa conductivity meter was utilized to ensure that this sample was collected outside of the Council Creek wastewater plume. Subsequent samples were collected along Council Creek using the conductivity meter to balance sample spacing between distance and conductivity. All samples were immediately placed in a cooler until they were returned to the lab on the same day and refrigerated at 4˚C for nutrient analyses within three days.

All sample analyses were completed at the Horn Point Laboratory in Cambridge, Maryland. The Yokogawa conductivity probe was re-calibrated in the lab as described above to acquire a more accurate measure of conductivity for all samples, and these values were converted to cond (25) (Figure 3-4). Water samples were filtered through Whatman 47 mm GF/F filters, and manual colorimetric methods were employed for the determination of NH_4, PO_4, and SiO_4 within 3 days of sample collection using a Spectronic model 301 spectrophotometer following the procedures of Koroleff (1969) for NH_4 and Stickland and Parsons (1974) for PO_4, and SiO_4. Approximately 15 ml of the filtered samples were placed in labeled plastic scintillation vials within one day of sample collection and immediately frozen for $NO_2 + NO_3$ analysis on the four channel Technicon Auto Analyzer II (Lane et al. 2000). Unfiltered samples were also processed for TN and TP using the procedure described above. In addition to NH_4, $NO_2 + NO_3$, PO_4, SiO_4, TN, and TP analyses, the 16 September 2002 Council Creek and the 17 September 2002 Little Creek transects were analyzed for chlorophyll(*a*) as described by Lorenzen (1967) and Jeffery and Humphrey (1975) using a Turner Designs model 111 fluorometer.

Data Analysis

Data were analyzed using mixing diagrams to establish net marsh nutrient processing (Boyle et al. 1974, Loder and Reichard 1981, Sharp et al. 1982, Fisher et al. 1988, Rochelle-Newell and Fisher 2002). Under the assumption that conductivity, a measure of major ion concentrations, is a generally conservative measure of mixing of different water sources, nutrient data were plotted as a function of conductivity. The WWTP outfall and the Choptank River samples were used as end members (Figure 3-2, bottom panel) of conservative mixing lines (Figure 3-5). Only transects where wastewater discharge and Choptank River flow were stable for the week prior to sampling were used for interpretation. Stable flow was defined as values greater than

0.50 for the ratio of the minimum:maximum flow conditions for the week prior to sampling for both wastewater discharge and Choptank River flow. The latter was estimated by flow measured at the nearby USGS gauging station at Greensboro, MD. On sample dates when there was no wastewater flow, the sample at the stream above the WWTP outfall (bottom panel of Figure 3-2) was used as the upper end member. On both 17 December 2001 transects, the Choptank end member samples were unavailable; as a result, water quality data from the CISnet study were used to interpret these transects. The data used were averages of samples taken at the nearest station from two CISnet cruises temporally bracketing (27 November 2001 and 9-10 January 2002) the 17 December 2001 transects (Malone et al. 2003).

All mixing diagrams with stable flow were interpreted using the change in concentration (Δ conc.) method of Rochelle-Newell and Fisher (2002) (Figure 3-5). Linear functions were fitted to the end member concentrations to plot the expected conservative mixing lines as shown by the red dashed line in Figure 3-5. If all transect samples visually appeared to be distributed along the conservative mixing line, a linear function was fitted to the data, and a two-tailed t-test ($\alpha = 0.025$) was performed to determine any differences in the slopes of the two lines. No significant difference was interpreted as conservative behavior (black line Figure 3-5). If there were significant slope differences, and for those transects that were clearly not linear, a higher order polynomial function was fitted to the data. Positive deviations from the conservative mixing line resulted in positive Δ conc. (net accumulation in water passing through the marsh, orange line in Figure 3-5), while negative deviations from conservative mixing resulted in negative Δ conc. (net losses in water passing through the marsh, blue line in

Figure 3-5). The maximum difference in concentration between the conservative mixing line and the polynomial function was used to determine the Δ conc, which represents the maximum observed deviation from conservative mixing. When compared to the expected value of the conservative concentration, the Δ conc is an estimate of the magnitude of the net source or sink in the marsh.

Results

Low tide sampling at the mouth of Council Creek

From 25 May to 19 November 2002, 339 low tide samples were collected, of which 39 were lost due to instrument failure. Two consecutive low tide samples were composited in individual bottles of an ISCO autosampler, resulting in 150 near daily samples taken from the end of May to the end of November. Cond (25), TN, TP, and WWTP discharge are plotted as a time series for the entire low tide record in Figure 3-6. With the exceptions of several high flow days and five multi-day periods of WWTP shutdowns, discharge from the WWTP was relatively consistent throughout this 7-month period with an average ± se flow of 5476 ± 218 m^3 d^{-1} (1.447 ± 0.058 MGD, Figure 3-6 panel D). Despite the relative constancy of discharge, cond (25), TN, and TP of the water leaving the marsh and entering the Choptank River on low tides, exhibited different patterns throughout the time series (Figure 3-6 panels A, B, and C). Cond (25) was low at the start and end of the time series with a peak in late summer, while TN concentrations were high in the spring and late fall and at a minimum in late summer. Total P concentrations appeared to decrease linearly throughout the time series.

To remove some of the temporal variability from Figure 3-6, the low tide time series data were averaged over weekly intervals (Figure 3-7). This smoothed representation of the data more clearly showed that cond (25) exhibited a late summer/early fall peak with values at the creek mouth consistently higher than the average cond (25) of wastewater (Figure 3-7 panel A). This summer increase in cond (25) might be due decreased freshwater discharge from Council Creek basin above the WWTP outfall; however, when low tide cond (25) was compared to regional river flow, there was no significant inverse correlation ($p > 0.05$). Local evapotranspiration from the marsh and net river water inflow on higher summer tides (see Chapter 2) are the likely causes of the high summer cond (25).

Low tide TN concentrations at the creek mouth showed a pattern opposite that of cond (25). There was a minimum during late summer and higher values in spring and fall (Figure 3-7 panel B, red filled circles). Some of this decrease was due to seasonal changes in TN concentrations of wastewater (open blue triangles). The summer minimum in TN concentrations in WWTP water was only ~40% of the mean TN for winter (1090 µM or 15 mg N l^{-1} during November – March 2001, 2002), which shows the seasonal effect of N removal by the overland flow system (~700 µM or 9.8 mg N l^{-1}) decrease. However, during the summer, Council Creek marsh provided an additional decrease in TN concentrations of ~200 µM or 2.8 mg N l^{-1}. In August 2002, at the time of lowest N concentration in low tide discharge, TN was equivalent to or less than 2 times higher than Choptank River water. TN concentrations exiting the creek at low tide were weakly and inversely correlated with temperature ($r^2 = 0.23$, $p < 0.05$), and TN was also inversely correlated with cond(25), exhibiting an exponential decline in TN concentration as cond (25) increased ($r^2 = 0.63$, $p < 0.01$).

Unlike cond (25) and TN, low tide TP concentrations showed a linear decrease at the mouth of Council Creek over the entire study period (Figure 3-7 panel C). Mean winter wastewater TP concentration was approximately 87 μM or 2.7 mg P l^{-1}, and TP concentrations in WWTP effluent discharging into Council Creek remained generally close to this value (open blue triangles in Figure 3-7 panel C). During late spring and early summer, low tide P concentrations (filled red circles) were equivalent to those of effluent, and no marsh effect was apparent. However, from August through November, concentrations of TP declined by ~50%, reaching a minimum concentration of ~40 μM or 1.2 mg P l^{-1}. There are insufficient data to define the temporal extent of this period of decreased P concentrations in water exiting Council Creek marsh at low tide.

The low tide time series also enabled the estimation of water residence times in Council Creek marsh. During two periods of brief plant shutdowns that occurred during the low tide time series, there were large changes in low tide concentrations, which clearly illustrated marsh behavior as it equilibrated from steady state, high nutrient concentrations (i.e., stable wastewater discharge) to steady state, low nutrient concentrations (i.e., no wastewater discharge). The November 2002 plant shutdown (Figure 3-6) is shown in more detail in Figure 3-8. Wastewater discharge was relatively constant until 8 November 2002 when effluent flow was stopped for 3.5 days (Figure 3-8 panel D). A four parameter sigmoid model was fitted to cond (25), TN, and TP data at the creek mouth during this period (Figure 3-8 panels A, B, and C).

$$C = C_l + (C_h - C_l)/(1 + \exp((t_m - t)/b)) \qquad (1)$$

where C is cond(25) or concentration of TN or TP, C_l is the lower of the two cond(25) or concentration values at steady state, C_h is the higher of the two cond(25) or concentration values at steady state, t_m is the time of the mid or inflection point between C_l and C_h, and b is a scaling parameter, which inversely controls the rate at which C changes. For TN data, the difference between steady state concentrations with and without wastewater was 845 µM or 11.8 mg N l^{-1} (red arrow in Figure 3-8 panel B). The time to reach the point of inflection (midpoint between the two equilibria, black arrows in Figure 3-8 panel B) was 1.7 days from the time of shutdown, suggesting that marsh flushing in Council Creek requires approximately 3.4 days to approach Choptank River concentrations. Similar estimates were fitted to cond(25) and TP in Figure 3-8 and to data from a similar shutdown in July – August 2002 (Table 3-1). These data gave an average ± se estimated flushing time of 3.0 ± 0.3 days for Council Creek marsh (Table 3-1).

Transects of Council Creek

Ten transects were sampled from 17 January 2001 through 16 September 2002 to examine distributions of cond(25) and nutrients within Council Creek marsh (Figure 3-9 and Table 3-2). An 11th transect, was sampled in Little Creek. Transects were interpreted using mixing diagrams, which assume flow stability at the end members. Stable flow was defined as having a ratio of the minimum to maximum flow for the week prior to sampling of greater than 0.50 for both end members (wastewater discharge and Choptank River flow). Four transects violated these conditions, and therefore were not interpreted in terms of mixing of freshwater and Choptank River water (gray dashed lines in Figure 3-9 and italicized transect dates in Table 3-2). Although the 7 May 2002 transect was in slight violation of stable flow due to a rain event during the week prior to sampling, it was still utilized to illustrate marsh behavior in the absence of wastewater flow. The remaining five transects illustrate summer versus winter conditions in Council Creek, and the one transect from Little Creek is used for contrast with Council Creek.

Transects of Council Creek during summer showed evidence of net uptake of nutrients within the marsh, consistent with Figure 3-7. For example, net losses of nitrogen and phosphorus were seen during the 21 August 2001 transect (Figure 3-10 and 3-11). Ammonium and NO_3 exhibited net losses of 62 μM or 0.87 mg NH_4-N $l^{-1}$and 20 μM or 0.28 mg NO_3-N l^{-1}, while there was an apparent net loss of 96 μM TN (1.3 mg TN l^{-1}, Figure 3-10 panels A, B, and C). The net TN loss was only slightly greater than the combined losses of NH_4 + NO_3 (82 μM or 1.1 mg N l^{-1}). There was also evidence for a 13–14 μM or ~0.4 mg P l^{-1} net loss of PO_4 and TP (Figure 3-11 panels A and B), whereas the distributions of SiO_4 during this transect exhibited conservative behavior (Figure 3-11 panel C). In summer, Council Creek marsh clearly appears to be a small net sink for wastewater N and P (Figures 3-7, 3-8, and 3-9, Table 3-3 to 3-7), removing on average 39 ± 6 % of the TN (Table 3-5) and 25 ± 6% of the P (Table 3-7).

In contrast to summer conditions, marsh nutrient processing in Council Creek during winter exhibited more complex patterns. For example, the 17 December 2001 ebb tide transect shows an apparent net loss of 32 μM NH_4 (0.45 mg NH_4-N l^{-1}) in the inner marsh, balanced by a net gain of 32 μM NH_4 in the outer marsh (Figure 3-12 panel A). Although the spatial pattern for NO_3 was similar, the net loss of 74 μM (1.0 mg NO_3-N l^{-1}) at the inner marsh was outweighed by the larger 161 μM (2.3 mg NO_3-N l^{-1}) net gain at the outer marsh (Figure 3-12 panel B). There was no evidence for net loss of TN at the inner marsh; however, at the outer marsh there was a large net gain of 1000 μM TN (14 mg N l^{-1}), dominated by organic N probably from marsh plants or sediments. Phosphate, TP, and SiO_4 showed similar patterns of net gain in concentrations throughout the marsh, with the largest gains occurring at the outer marsh (Figure 3-13 panels A, B, and C). Although there was some evidence for net loss of NH_4 and NO_3, in the inner marsh, Council Creek marsh appeared to export large quantities both organic and inorganic nutrients during winter (250 ± 180 % N and 165 ± 50 % P increases, Tables 3-5 and 3-7), probably due to mobilization of degraded marsh plant tissue and sediments.

The 7 May 2002 transect was collected 17 days after the discharge from the wastewater treatment plant had stopped for management purposes (Figure 3-9). Since the marsh had adequate time for residual wastewater to drain (17 days ≈ 5x estimated flushing time in Table 3-1), this transect represented the behavior of the marsh when the large wastewater inflow was halted (Figures 3-14 and 3-15). During this transect, Council Creek marsh generally appeared to be a large net source of nutrients to the Choptank River; however, concentrations were ~10 less than during discharge periods. Evidence for apparent net gains in N was shown by the 39 μM (0.55 mg N l^{-1}) net gain of TN and a 19 μM net gain of NH_4 (Figure 3-14 panels A and C). However, these net gains are partially counterbalanced by the 18 μM (0.25 mg NO_3-N l^{-1}) net loss of NO_3, possibly via denitrification (Figure 3-14 panel B). Phosphate and TP during this transect also exhibited a pattern of marsh outwelling with large net gains of 15 and 21 μM (0.46 and 0.65 mg P l^{-1}), respectively (Figure 3-15 panels A and B). For SiO_4 there was a consistent net loss throughout the marsh with the maximum loss of 76 μM (2.1 mg Si l^{-1}) occurring near the mouth of Council Creek (Figure 3-15 panel C). The dashed lines in all panels of Figures 3-14 and 3-15 represent the spatial difference between the sample at the mouth of Council Creek and the Choptank end member. Although the distance between the two samples was only 84 m, the large increase in cond(25) left the polynomial function line unconstrained over this interval, and it was therefore removed. Data from this transect (Figures 3-14 and 3-15) show that Council Creek is at least partially equilibrated with nutrients from the WWTP, and that the marsh releases large quantities of N and P when the wastewater nutrient source is removed.

Data from the 17 September 2002 transect in Little Creek (with no WWTP effluent) were interpreted in the same manner as the Council Creek transects. This one example of nutrient processing in Little Creek marsh showed a pattern quite different from the Council Creek transects (Figures 3-16 and 3-17). Like Council Creek, nutrients in Little Creek appear to be influenced by landuse in the surrounding basin, but nutrient concentrations are an order of magnitude lower (compare Figures 3-8 and 3-9 with Figures 3-16 and 3-17). On 17 September 2002 a large algal bloom was occurring with a net gain of 770 $\mu g\ l^{-1}$ chlorophyll(*a*) concentration during this transect that was immediately downstream of both a housing development and a large poultry operation (Figure 3-16 inset on panel C). In contrast, there was a 16 $\mu g\ l^{-1}$ net loss of chlorophyll(*a*) in Council Creek during a 16 September 2002 transect, one day prior (data not shown).

The Little Creek algal bloom seemed to be the primary driver for nutrient behavior on this date. There was a 370 and 21 μM (5.2 and 0.65 $mg\ l^{-1}$) accumulation of TN and TP in the creek water, respectively, that followed the same general pattern as the net gain in chlorophyll(*a*) (Figure 3-16 and Figure 3-17 panel C). However, in Council

Creek on 16 September 2002, there was evidence for apparent net losses of –230 μM (3.2 mg N l^{-1} or -35%) TN and –19 μM (0.59 mg P l^{-1} or -16%) TP (Tables 3-4 and 3-6). The net gains of organic N and P in Little Creek were accompanied by complete depletion of inorganic N and a 56% net loss of PO_4 (Figure 3-16 panel A and B and Figure 3-17 panel A), and a smaller net loss of 39 μM SiO_4 (1.1 mg Si l^{-1}, Figure 3-17 panel C). Net losses of both NH_4 and PO_4 occurred in the vicinity of the bloom, whereas depletion of NO_3 occurred just downstream of the algal bloom. In contrast, in Council Creek marsh on 16 September there were small net losses of TN (-35%) and TP (-16%) and conservative behavior for SiO_4 (Table 3-3 to 3-8). Algal accumulation in Little Creek is potentially the result of nutrient inputs via development and agriculture and longer water residence times (in the absence of direct wastewater inflows).

The general pattern of N behavior in Council Creek marsh was for net losses during the growing season and net releases to the Choptank River in winter. Ammonium losses averaged –92 ± 30 μM (–1.3 ± 0.4 mg NH_4-N l^{-1}) for the three summer transects when wastewater was present (Table 3-3). Although apparent net losses were evident during the growing season, during winter and in the absence of wastewater discharge, Council Creek marsh released small amounts of NH_4 to the Choptank River (+11 ± 11 μM or +0.12 ± 0.12 mg NH_4-N l^{-1}). For NO_3 (Table 3-4), Council Creek averaged a net gain of 12 ± 18 μM (+0.17 ± 0.25 mg NO_3-N l^{-1}) with no clear seasonal pattern; however, scatter in the data from two of the transect dates resulted in ambiguous interpretations of the mixing diagrams. The 7 May 2002 transect did, however, show apparent net losses of NO_3. As for NH_4, there was evidence for net losses of TN (Table 3-5) during the growing season (-186 ± 45 μM or –2.6 ± 0.6 mg N l^{-1}), but Council Creek marsh appeared to release large quantities of organic N during the winter and in the absence of wastewater flow 658 ± 343 µM or +9.2± 4.8 mg N l^{-1}, Table 3-5). Processes potentially responsible for this strongly seasonal pattern of N behavior could be plant uptake and denitrification during the growing season and organic N release during senescence and decomposition in fall and winter. Although none of these processes were directly measured, each has been shown to be an important process in the retention or loss of N and P by intertidal marshes (Merrill 1999, Hsieh 1996, Nixon 1980, and Simpson et al. 1978)

The pattern of P behavior in Council Creek marsh was similar to that of N behavior (Tables 3-6 and 3-7). There were significant net P losses at the end of the growing season, with comparable P releases in the winter and in the absence of wastewater discharge. Average Δ conc for PO_4 was –18 ± 3.1 µM (-0.56 ± 0.10 mg P l^{-1}) for the summer transects when wastewater was being discharged, while average Δ conc for TP was –21 ± 5.3 µM (-0.65 ± 0.16 mg P l^{-1}, Table 3-6 and 3-7). Phosphate and TP were also re-mobilized during the winter as evidenced by the 21 ± 2.5 µM (0.65 ± 0.08 mg P l^{-1}) net gain in PO_4 and 27 ± 3.0 µM (0.84 ± 0.09 mg P l^{-1}) net gain in TP. The large net gains of both PO_4 and TP in the absence of wastewater flow occurred spatially near the transition from the inner marsh community to the middle marsh community and may be related to sediment P saturation in this area of the marsh (see Chapters 4 and 5). These comparable seasonal changes in withdrawal and release of TP (and TN) are indicative of seasonal nutrient storage but not of permanent removal by the marsh.

Silicate behavior in Council Creek marsh exhibited a pattern somewhat similar to that of N and P (Table 3-8). However, there was no evidence for net SiO_4 release when there was no wastewater discharge, probably due to the fact that the wastewater does not contain elevated SiO_4 concentrations. Early in the growing season there were small net losses of –16 ± 16 μM SiO_4 (-0.45 ± 0.45 mg SiO_4 l^{-1}) potentially due to diatom uptake or macrophyte uptake resulting from growth of silicate–rich *Phragmites australis* plant tissue (Tscharntke 1989). Later in the growing season when most of the macrophytes reached peak biomass, SiO_4 appeared to behave conservatively. Net gains of 80 ± 6.5 μM SiO_4 (2.2 ± 0.2 mg SiO_4 l^{-1}) during the winter could have been the result of the export of dead macrophyte material and other particulate matter to Council Creek water.

Discussion

An investigation of the integrated effect of all marsh processes acting on wastewater passing though the system revealed seasonal nutrient processing. Low tide sampling at the creek mouth yielded apparent marsh processing of both TN and TP. Although the treatment plant's overland flow system removes approximately 700 μM TN or 9.8 mg N l^{-1} from wastewater in summer (64%), Council Creek marsh appears to add an additional treatment component by reducing concentrations by another 200 μM or 2.8 mg N l^{-1} (Figure 3-7 panel B). The net summer effect of both the overland flow system and Council Creek marsh is to reduce wastewater TN concentrations exiting Council Creek marsh at low tide to approximately the same as or slightly higher than Choptank River concentrations. During the low tide time series, creek mouth TP concentrations were approximately equivalent to wastewater values, indicating little net TP retention by the marsh until about mid July (Figure 3-7 panel C). From late July to the end of the time series, low tide P concentrations decreased, indicating that Council Creek marsh appeared to be a net sink for wastewater TP. Marsh behavior during brief WWTP shutdowns indicated that this 0.16 km^2 marsh requires ~3 days to be flushed of residual wastewater.

Further evidence of the pattern seen in the low tide time series was shown in the transect data. Evidence of nutrient processing of NH_4, TN, PO_4, and TP was clearly seen in transects collected during the growing season (Figures 3-9, 3-10, Tables 3-2 to 3-6). However, over winter Council Creek marsh appeared to release nutrients back in to the water, which were exported to the Choptank River (Figures 3-11, 3-12, Tables 3-2 to 3-6). Although there seemed to be some SiO_4 removal early in the growing season, SiO_4 appeared to be conservative later in the growing season and was even exported to the Choptank during winter (Table 3-8). When wastewater discharge into the marsh was halted during the growing season, Council Creek marsh released significant concentrations of NH_4, TN, PO_4, and TP, but apparent net losses of NO_3 and SiO_4 were still observed (Figures 3-13 and 3-14).

Although apparent losses of TN and TP were observed in the low tide time series, one important consideration is dilution of the samples by Choptank River water. To account for this potential effect, the assumption of conductivity as a conservative indicator of mixing was again used. Low tide samples exiting Council Creek marsh were comprised of a mix of low conductivity freshwater from the landward end of Council Creek and higher conductivity water from the Choptank River. I used the conductivity values of low tide water samples to estimate the fractional volumes of river and freshwater using flow of the freshwater and Choptank end members, and estimated values of seasonal cond(25), TN, and TP in the end members. Predicted conservative TN and TP were then estimated from the fractional volumes and compared with observed TN and TP values in the low tide samples.

Wastewater and stream water were the major freshwater components. To estimate the volume-weighted, combined conductivity of the two freshwater sources ($cond_{ww+st}$), daily wastewater discharge records and seasonal changes in stream flow above the WWTP outfall (Figure 3-2, bottom panel) were calculated. Council Creek flow above the WWTP outfall was estimated from the drainage area above the WWTP outfall and estimated monthly water yields at Greensboro, MD (Fisher et al. 1998). Additionally, occasional measurements of wastewater and stream cond(25) (Figure 3-18 panels A and B) from the three transects collected during the low tide series were utilized in the freshwater mass balance equations:

$$cond_{ww+st} = f_{ww} * cond_{ww} + f_{st} * cond_{st} \qquad (2)$$

$$f_{ww} + f_{st} = 1 \qquad (3)$$

where f_{ww} is the fractional volume of wastewater (estimated from discharge records), $cond_{ww}$ is cond(25) of wastewater Figure 3-18 panel A), f_{st} or (1- f_{ww}) is fractional volume of the stream (estimated from regional streamflows), and $cond_{st}$ is cond(25) of the stream (Figure 3-18 panel B). There were only a few samples of Council Creek freshwater and WWTP effluent during this period (Figure 3-18 panel A and B), but the available data suggested little seasonal variation. I therefore assumed that the average conductivities of 645 and 677 µmho cm^{-1} represented WWTP and effluent and stream water, respectively. This approach gave an estimated combined freshwater flow of 4568 to 10397 m^3 d^{-1} and a combined $cond_{ww+st}$ of 645 to 653 µmho cm^{-1}, varying seasonally.

The seasonal trend in Choptank conductivity was also estimated from the transect data, and I assumed that the larger variations in available data represented the seasonal variations in cond(25) in the Choptank River water exchanging with the marsh (Figure 3-18 panel C). These conductivity values were substituted into the low tide mass balance equations:

$$cond_{lt} = f_{ww+st} * cond_{ww+st} + f_{Ch} * cond_{Ch} \quad (4)$$

$$f_{ww+st} + f_{Ch} = 1 \quad (5)$$

where $cond_{lt}$ is cond(25) of the low tide sample, $cond_{ww+st}$ is cond(25) of combined wastewater and stream water end members (equation 2), f_{Ch} is 1- f_{ww+st} or the fractional volume of Choptank water, and $cond_{Ch}$ is cond(25) of the Choptank River. Equations 4 and 5 were used to determine the fractional volume of the freshwater contained in the low tide samples (f_{ww+st}). By substituting eq. 5 into eq. 4 and solving for f_{ww+st}, equation 4 simplifies to:

$$f_{ww+st} = (cond_{lt} - cond_{Ch})/(cond_{ww+st} - cond_{Ch}) \quad (6)$$

Equation 6 enabled me to estimate the fraction of freshwater (WWTP effluent and Council Creek stream water) in each low tide sample (Figure 3-18 panel D).

There was a pronounced summer decline in the fractional volume of freshwater in the low tide samples (Figure 3-18 panel D). Although quite variable, on average the water discharging from Council Creek at low tide was ~90% freshwater (wastewater and stream water) in May (and November) and presumably through the winter; however, in summer low tide discharge was, on average, only ~50% freshwater, probably because of the higher sea level stands at this time of the year and greater high tides. This pattern of 50% dilution of freshwater was quite similar to the one observed in the low tide TN samples (Figure 3-7) and may be due to either dilution by river water or evapotranspiration within the marsh.

The effect of evapotranspiration (ET) between low tides was small. Evapotranspiration has been estimated by several other researchers, and ET values are summarized in Table 3-9. These values range from 0.15 – 0.50 mm h^{-1}, with an overall mean of 0.25 mm h^{-1}. Using the mean of the Chesapeake Bay ET values (0.40 mm h^{-1}) and an average marsh water depth of 15 cm, the maximum ET-related increase in cond(25) between successive low tides would be 5 mm or 3.2% of the estimated mean depth of 15 cm. This fractional loss of water is much less than the 300% increases in conductivity observed in summer (Figure 3-7 panel A). Consequently, cond(25) changes due to marsh ET were discounted as a minor effect, and the increases in conductivity were assumed to be the result of mixing of Choptank River water and freshwater within the marsh. In the middle of summer (July-August), water exiting Council Creek marsh at low tide appeared to be, on average, a 50:50 mixture of Choptank River water and the combined freshwater inflows from the WWTP outfall and the Council Creek basin above the outfall. The latter mixture varies in composition but is generally dominated by the wastewater flows (73.5 to 98.8%).

Seasonal changes in TN and TP concentrations of the wastewater, stream, and Choptank were estimated in the same manner as conductivity (Figure 3-19). However, in this case, there were significant seasonal variations in TN and TP in most end members.

The regressions in Figure 3-19 and equations similar to eq. 2 for Tn and TP were then utilized to determine predicted conservative low tide TN or TP (pred cons TN_{lt} or TP_{lt}) concentrations for each low tide sample:

$$\text{pred cons } TN_{lt} = f_{ww+st}*TN_{ww+st} + f_{Ch}*TN_{Ch} \quad (7)$$

$$\text{pred cons } TP_{lt} = f_{ww+st}*TP_{ww+st} + f_{Ch}*TP_{Ch} \quad (8)$$

The predicted conservative concentrations in the low tide samples (pred cons TN_{lt} or TP_{lt}) were those expected to be present on the basis of the estimated dilution of freshwater samples by Choptank River water (Figures 3-18 panel D), with no biological or chemical processing in the marsh. Plots of the observed low tide concentrations versus the conservative concentrations predicted from the conductivity changes are shown in Figures 3-20 and 3-21. In both plots the data were partitioned into blocks of time, which were (1) June, July and August, (2) September, and (3) October and November. For low tide TN, it appears that much of the apparent seasonal pattern of TN concentration reductions was due to simple dilution of the samples by Choptank River water (Figure 3-20). Although the regression line through the data is significantly > 1 ($p < 0.05$), and the intercept is significantly < 0 ($p < 0.05$), the points scatter about the 1:1 line. There are some samples that lie below the 1:1 line, indicating net TN losses, and there are others which lie above the 1:1 line, indicating net gains in TN. This pattern is similar to those observed in the longitudinal transects (e.g., Figures 3-10, 3-12, and 3-14), but on average the low tide samples exhibited conservative behavior influenced only by dilution. Additionally, there appeared to be no discernable pattern related to time.

In contrast, P concentrations showed a somewhat different pattern (Figure 3-21). Although lower TP values were scattered about the 1:1 line, many higher observed values of TP >100uM (3.1 mg P l^{-1}) fell above the 1:1 line, indicating net gains in TP by the water exiting the marsh during the summer months. In contrast, the observed values that fell below the 1:1 line, indicating net losses of TP, were largely restricted to the fall. This seasonal pattern of summer release and fall uptake helps explain the apparent linear decrease in TP concentrations in the low tide time series in Figure 3-7 despite the large seasonal changes in conductivity. Because of the scatter (Figure 3-21), the slope of the regression of predicted conservative TP and observed low tide TP data was not significantly different from the 1:1 line; however, the intercept was significantly > 0 ($p < 0.01$). This indicates that the majority of the observed concentrations significantly exceeded those predicted from the conductivity changes and that Council Creek marsh was exporting P to the Choptank River in this time series primarily in the summer. These small net increases in TP in summer were increased when WWTP flow stopped (Figure 3-15) and in winter (Figure 3-13).

Test of the hypotheses

The hypothesis to be tested in this chapter was that elevated N and P concentrations via wastewater discharged into Council Creek are significantly attenuated via biological, chemical and physical processes occurring within the associated marsh. To test this hypothesis, two data sets were collected (low tide samples at the creek mouth and longitudinal transects down the centerline of Council Creek). The low tide samples were collected to determine water quality patterns in water (mix of freshwater, wastewater, and Choptank water) exiting Council Creek that had been exposed to chemical and biological processing in the marsh, while transects were used to determine detailed descriptions of marsh nutrient processing during different seasons, nutrient loads, and tide stages.

These data only partially supported the hypothesis that Council Creek marsh attenuates wastewater N and P via biological, chemical and physical processing. Although individual transects in summer showed small net losses of N in the marsh (e.g., Figure 3-10), the decreases in N concentrations in Figure 3-7 appeared on average to be attenuated primarily by dilution with Choptank River water in the low tide sampling (Figure 3-20). Similarly, P concentrations appeared to be reduced in low tide discharge from mid July into late November (Figure 3-7), and several transects also showed this effect (Figure 3-11). However, on average, there was evidence for net export of P in summer and fall, and especially during the winter when no active plant growth was occurring and dead and decaying plant material from the previous year's growing season were potentially re-mobilized, as shown by the winter transects (Figures 3-12, 3-13). Additionally, during periods with no wastewater flow, Council Creek marsh appeared to export both N and P to the Choptank River (Figures 3-14, 3-15).

The data collected in this study provided insight into wastewater nutrient processing by a natural marsh. However, it should be cautioned that interannual variability in precipitation can play an important role in marsh nutrient retention (Correll et al. 1992). Reduction in N concentrations in water exiting the marsh at low tide seems to be a largely seasonal process of dilution by Choptank River water with maximum N reductions of approximately 50%. Most of this summer N concentration reduction is likely due to the higher water levels in summer (see Chapter 2), with potential mechanisms of N immobilization (removal) including denitrification and plant and microbial uptake. However, these appeared to be smaller (39%) than dilution (50%). In addition, during winter it appeared that Council Creek marsh exported organic N from the outer marsh, probably when dead plant material from the previous year's growth may be remobilized (Figure 3-12). Simpson et al. (1983) found a similar pattern of N retention during the growing season and N export late in the year in Delaware River marshes. Additionally, Nixon (1980) characterized coastal salt marshes as N transformers.

The pattern for P immobilization was clearly different. The low tide time series showed a linear decline in P concentrations at the creek mouth from May to November (Figure 3-7), and the processes responsible for this pattern appear to be dilution, seasonal desorption (spring/summer), and resorption (fall). This interpretation is largely based on conductivity calculations (eq. 8, Figure 3-21). Although there have been several studies and reviews on nutrient flux from marshes to estuaries, there is no agreement on patterns of P movement (Nixon 1980, Jordan et al. 1983, Wolaver et al. 1983, Childers et al. 2000). However, these studies have all addressed P removal from estuarine waters in contrast to this study, which has addressed upstream P inputs from wastewater. It is also important to note that the time series spanned only 6 months, and therefore this sampling did not capture the entire cycle. Mean winter wastewater effluent concentrations did not appear to be a different from the spring and early summer low tide discharges at the mouth of the creek, suggesting that the marsh P concentrations in early summer were increasing above those expected by dilution with Choptank River water (Figures 3-7 and 3-21). Following this time, the marsh became more neutral or resorbed P during the rest of the summer and well into fall when the sampling ended. Data from the December transects, however, suggested that the fall TP pattern reversed by mid December, as shown by Table 3-7 and Figure 3-15 panels A and B, when P was clearly leaking from the marsh.

The marsh response to cessation of N and P loading via WWTP shutdown was a large net release of NH_4, TN, PO_4 and TP. A possible reason for this marsh outwelling of nutrients may be the result of the gradient of potentially high nutrient pore water and sediment concentrations equilibrated with the previously high creek concentrations. When the creek concentrations fell to low N and P values, some of this adsorbed and dissolved N and P appeared to be mobilized, probably during high tide, and to exit the marsh with outflowing creek water. In contrast to the other nutrients, net losses of both NO_3 and SiO_4 occurred during the 7 May 2002 transect with no wastewater. However, the concentrations of NO_3 and SiO_4 in wastewater were usually not significantly different from those in the stream sample above the tidal influence, and consequently there was little gradient from pore water and sediment to creek water.

To summarize the results of this chapter, Council Creek marsh appears to function as a seasonal treatment component for wastewater N and a seasonal filter for wastewater P. Reductions in N concentrations of approximately 50% of N in wastewater effluent were seen during the height of the growing season primarily due to dilution by Choptank River water during the seasonally higher tides. This reduction results in TN concentrations being discharged to the Choptank River that are less than 2 times higher than average Choptank River concentrations. During winter, some of the marsh N is mobilized as organic material and exported to the Choptank. For P there is a more complex pattern of summer desorption, resorption and mobilization. The linear decrease in TP concentrations at the creek mouth suggest net gains in P concentrations early in the growing season, perhaps due to higher sea level, and little effect on wastewater P late in the growing season and into the fall. Furthermore, the transect data suggested that the TP concentrations were augmented by marsh P by late fall to early winter. Overall, Council Creek marsh appeared to process N and P seasonally, generally with retention in the warmer plant growing season. However, cooler winter months resulted in net losses to the Choptank River, probably with small net retention only as sediment accretion (see Chapter 5).

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Table 3-1: Summary of marsh draining at the cessation of WWTP discharge. Estimated model parameters for eq. 1 ($C = C_l + (C_h - C_l)/(1 + \exp((t_m - t)/b)))$ and correlation coefficients are indicated by x_o, y_o, a, b, and r^2. Time to inflection is the time in days from the cessation of wastewater discharge to the inflection point, and estimated flushing time is the estimated time required for the marsh to reach the low nutrient concentration steady state.

date	parameter	t_m	C_l	C_h	b	r^2	time to inflection, days	estimated flushing time, days
November 2002	cond (25)	313.1	1728	2591	+0.504	0.95	1.1	2.2
	TN	313.7	15.4	860	-0.937	0.90	1.7	3.4
	TP	313.6	12.4	54.3	-0.604	0.88	1.6	3.2
July-August 2002	TP	215.6	31.7	46.7	-0.506	0.89	1.6	3.2

Table 3-2: Flow conditions during water quality transects. End members are cond (25) values used to define the conservative mixing lines. Flow means are the mean values of the week prior to and including the transect dates. Flow stability values are the ratio of daily min:max flow values used to calculate the mean. Italicized transects are not interpretable because flow stability conditions were violated (stability < 0.50); however, the 7 May 2002 transect was used (Figures 3-13 and 3-14) as an example of nutrient distributions when WWTP discharge had been discontinued for 17 days. Choptank flow data are from the USGS gauging station at Greensboro, Maryland.

date, tide	inner end member cond, μmho cm^{-1}	outer end member cond, μmho cm^{-1}	mean WWTP flow, $m^3\ d^{-1}$	WWTP flow stability	mean Choptank flow, $10^6\ m^3\ d^{-1}$	Choptank flow stability	conditions
Council Creek							
17 Jan 01, ebb	*713*	*2747*	*4149*	*0.00*	*0.202*	*0.90*	*12-14 Jan-no discharge; 14-16 Jan-0.89 cm precip.*
18 Jul 01, ebb	558	2398	6196	0.60	0.194	0.61	semi-stable discharge; 18 Jul-0.74 cm precip
21 Aug 01, flood	557	2650	7911	0.68	0.108	0.75	semi-stable discharge; 14 Aug-3.68 cm precip
17 Dec 01, ebb	665	6735	6221	0.50	0.063	0.89	semi-stable discharge; 11&14 Dec 1.30 cm total precip
17 Dec 01, flood	669	6735	6221	0.50	0.063	0.89	semi-stable; 11&14 Dec-1.30 cm total precip
23 Apr 02 ebb	*638*	*7123*	*1661*	*0.00*	*0.152*	*0.83*	*3rd day of no discharge;19, 21, & 22 Apr-1.09 cm total precip*
7 May 02, ebb	*577*	*4675*	*0*	*1.00*	*0.523*	*0.40*	*17^{th} day of no discharge; 2&5 May-3.05 cm total precip*
28 May 02, ebb	*644*	*6068*	*8566*	*0.32*	*0.212*	*0.50*	*high discharge then no discharge 4 hours before sampling; no precip*
7 July 02, ebb	*683*	*8967*	*4691*	*0*	*0.041*	*0.49*	*1 Jul - no discharge; no precip*
16 Sep 02, ebb	609	10144	7843	0.70	0.052	0.67	semi-stable discharge; 15 & 16 Sep-1.93 cm total precip

Table 3-2: cont

date, tide	inner end member cond, μmho cm^{-1}	outer end member cond, μmho cm^{-1}	mean WWTP flow, $m^3 d^{-1}$	WWTP flow stability	mean Choptank flow, $10^6 m^3 d^{-1}$	Choptank flow stability	conditions
Little Creek							
17 Sep 02, ebb	804	9315	na	na	0.053	0.62	no wastewater; 15 & 16 Sep-1.93 cm total precip

Table 3-3: Summary of NH_4 processing during water quality transects. Poly order and r^2 refer to the polynomial order and correlation coefficient of the curve fitted to the data. Δ [NH_4] is the maximum change in concentration from conservative mixing followed by the % Δ [NH_4], which represents the % loss or gain from the conservative concentration. The inner and outer marsh Δ [NH_4] and % Δ [NH_4] values for the 17 Dec 02 transects were averaged to get the net effect on this day. Italicized transects were not interpretable due to violations of flow stability (see Table 3-1); however, the 7 May 2002 transect was used as an example of nutrient distributions when WWTP discharge had been discontinued for 17 days. The summer average values reported are mean±se of 18 July 2001, 21 August 2001, and 16 Sept 2002; winter average values reported are mean±se of 17 Dec 2001 ebb and flood transects.

date, tide	inner end member [NH_4], μM	outer end member [NH_4], μM	poly order	r^2	Δ [NH_4], μM	% Δ [NH_4]	interpretation
Council Creek							
17 Jan 01, ebb	*968*	*21.3*	*4*	*0.99*	--	--	*change in flow, not interpretable*
18 Jul 01, ebb	315	0.00	2	0.94	-152	-54	net loss
21 Aug 01, flood	198	0.00	3	0.97	-62	-60	net loss
17 Dec 01, ebb	203	8.89	3	0.95	0	+27	inner marsh: net loss/outer marsh: net gain
17 Dec 01, flood	190	8.89	3	0.98	+21	+6	inner marsh: net loss/outer marsh: net gain
23 Apr 02 ebb	*3.19*	*16.3*	*4*	*0.96*	--	--	*flow off (3 days), not interpretable*
7 May 02, ebb	*5.06*	*5.22*	*5*	*0.98*	*+19*	*+360*	*net gain*
28 May 02, ebb	*5.64*	*2.27*	*5*	*0.79*	--	--	*flow off (~4 hours), not interpretable*
7 July 02, ebb	*271*	*0.39*	*2*	*0.99*	--	--	*change in flow, not interpretable*
16 Sep 02, ebb	412	3.37	4	0.98	-63	-16	net loss
summer avg	**308±62**	**1.1±1.1**			**-92±30**	**-43±14**	**net loss**
winter avg	**197±7**	**8.9±0.0**			**11±11**	**17±11**	**net gain**
Little Creek							
17 Sep 02, ebb	17.0	8.61	7	0.96	-15	-100	net loss

Table 3-4: Summary of NO_3 processing during water quality transects. Poly order and r^2 refer to the polynomial order and correlation coefficient of the curve fitted to the data. Δ [NO_3] is the maximum change in concentration from conservative mixing followed by the % Δ [NO_3], which represents the % loss or gain from the conservative concentration. The inner and outer marsh Δ [NO_3] and % Δ [NO_3] values for the 17 Dec 02 ebb tide transect were averaged to get the net effect on this day. Shaded transects were not interpretable due to violations of flow stability (see Table 3-1); however, the 7 May 2002 transect was used as an example of nutrient distributions when WWTP discharge had been discontinued for 17 days. The summer average values reported are mean±se of 18 July 2001, 21 August 2001, and 16 Sept 2002; winter average values reported are mean±se of 17 Dec 2001 ebb and flood transects.

date, tide	inner end member [NO_3], µM	outer end member [NO_3], µM	poly order	r^2	Δ [NO_3], µM	% Δ [NO_3]	interpretation
Council Creek							
17 Jan 01, ebb	*28.0*	*78.8*	*4*	*0.98*	--	--	*change in flow, not interpretable*
18 Jul 01, ebb	26.8	57.3	2	0.57	+11	+36	net gain
21 Aug 01, flood	77.6	32.0	4	0.96	-20	-32	net loss
17 Dec 01, ebb	545	31.7	3	0.91	+44	+68	inner marsh: net loss/outer marsh: net gain
17 Dec 01, flood	563	31.7	?	?	ambiguous	ambiguous	ambiguous
23 Apr 02 ebb	*41.5*	*77.3*	*4*	*0.97*	--	--	*flow off (3 days), not interpretable*
7 May 02, ebb	*47.0*	*44.1*	*5*	*0.82*	*-18*	-39	*net loss*
28 May 02, ebb	*39.7*	*60.6*	*3*	*0.99*	--	--	*flow off (~4 hours), not interpretable*
7 July 02, ebb	*34.9*	*10.9*	*3*	*0.52*	--	--	*change in flow, not interpretable*
16 Sep 02, ebb	78.1	20.8	?	?	ambiguous	ambiguous	ambiguous
summer avg	**61±17**	**37±11**			**-5±16**	**2±34**	**net loss**
winter avg	**554±13**	**31.7±0.0**			**44**	**68**	**net gain**
Little Creek							
17 Sep 02, ebb	82.1	10.0	4	0.90	-60	-100	net loss

Table 3-5: Summary of TN processing during water quality transects. Poly order and r^2 refer to the polynomial order and correlation coefficient of the curve fitted to the data. Δ [TN] is the maximum change in concentration from conservative mixing followed by the % Δ [TN], which represents the % loss or gain from the conservative concentration. The inner and outer marsh Δ [TN] and % Δ [TN] values for the 17 Dec 02 flood tide transect were averaged to get the net effect on this date. Italicized transects were not interpretable due to violations of flow stability (see Table 3-1); however, the 7 May 2002 transect was used as an example of nutrient distributions when WWTP discharge had been discontinued for 17 days. The summer average values reported are mean±se of 18 July 2001, 21 August 2001, and 16 Sept 2002; winter average values reported are mean±se of 17 Dec 2001 ebb and flood transects.

date, tide	inner end member [TN], µM	outer end member [TN], µM	poly order	r^2	Δ[TN], µM	% Δ[TN]	interpretation
Council Creek							
17 Jan 01, ebb	*330*	*90.9*	*2*	*0.49*	--	--	*change in flow, not interpretable*
18 Jul 01, ebb	510	122	2	0.93	-233	-50	net loss
21 Aug 01, flood	410	92.6	3	0.98	-96	-32	net loss
17 Dec 01, ebb	1040	96.5	4	0.88	+1000	+430	net gain
17 Dec 01, flood	1310	96.5	3	0.67	+315	+70	inner marsh: net loss/outer marsh net gain
23 Apr 02 ebb	*75.2*	*144*	*4*	*0.91*	--	--	*flow off (3 days), not interpretable*
7 May 02, ebb	*86.3*	*97.9*	*4*	*0.70*	*+39*	*+43*	*net gain*
28 May 02, ebb	*71.0*	*113*	*4*	*0.73*	--	--	*flow off (~4 hours), not interpretable*
7 July 02, ebb	*529*	*61.4*	*2*	*0.90*	--	--	*change in flow, not interpretable*
16 Sep 02, ebb	775	50.3	5	0.95	-230	-35	net loss
summer avg	**565±109**	**88±21**			**-186±45**	**-39±6**	**net loss**
winter avg	**1175±135**	**97**			**658±343**	**250±180**	**net gain**
Little Creek							
17 Sep 02, ebb	189	83.4	7	0.97	+370	+230	net gain

Table 3-6: Summary of PO_4 processing during water quality transects. Poly order and r^2 refer to the polynomial order and correlation coefficient of the curve fitted to the data. Δ [PO_4] is the maximum change in concentration from conservative mixing followed by the % Δ [PO_4], which represents the % loss or gain from the conservative concentration. The inner and outer marsh Δ [NH_4] and % Δ [NH_4] values for the 17 Dec 02 transects were averaged to get the net effect on this day. Italicized transects were not interpretable due to violations of flow stability (see Table 3-1); however, the 7 May 2002 transect was used as an example of nutrient distributions when WWTP discharge had been discontinued for 17 days. The summer average values reported are mean±se of 18 July 2001, 21 August 2001, and 16 Sept 2002; winter average values reported are mean±se of 17 Dec 2001 ebb and flood transects.

date, tide	inner end member [PO_4], μM	outer end member [PO_4], μM	poly order	r^2	Δ [PO_4], μM	% Δ [PO_4]	interpretation
Council Creek							
17 Jan 01, ebb	*58.1*	*1.06*	*2*	*0.93*	--	--	*change in flow, not interpretable*
18 Jul 01, ebb	80.2	1.59	3	1.00	-24	-34	net loss
21 Aug 01, flood	79.2	1.51	3	1.00	-14	-26	net loss
17 Dec 01, ebb	64.3	0.64	4	0.99	+23	+190	net gain
17 Dec 01, flood	63.4	0.64	4	0.99	+18	+180	net gain
23 Apr 02 ebb	*0.89*	*1.03*	*2*	*0.99*	--	--	*flow off (3 days), not interpretable*
7 May 02, ebb	*1.38*	*0.94*	*4*	*0.99*	*+15*	*+1300*	*net gain*
28 May 02, ebb	*2.49*	*1.37*	*4*	*0.77*	--	--	*flow off (~4 hours), not interpretable*
7 July 02, ebb	*96.5*	*1.49*	*3*	*1.00*	--	--	*change in flow, not interpretable*
16 Sep 02, ebb	128	1.71	4	1.00	-16	-16	net loss
summer avg	**96±16**	**1.60±0.05**			**-18±3.1**	**-25±5**	**net loss**
winter avg	**64±0.5**	**0.64**			**21±3**	**185±5**	**net gain**
Little Creek							
17 Sep 02, ebb	4.58	1.59	4	0.97	-2.2	-56	net loss

Table 3-7: Summary of TP processing during water quality transects. Poly order and r^2 refer to the polynomial order and correlation coefficient of the curve fitted to the data. Δ [TP] is the maximum change in concentration from conservative mixing followed by the % Δ [TP], which represents the % loss or gain from the conservative concentration. The inner and outer marsh Δ [NH_4] and % Δ [NH_4] values for the 17 Dec 02 transects were averaged to get the net effect on this day. Italicized transects were not interpretable due to violations of flow stability (see Table 3-1); however, the 7 May 2002 transect was used as an example of nutrient distributions when WWTP discharge had been discontinued for 17 days. The summer average values reported are mean±se of 18 July 2001, 21 August 2001, and 16 Sept 2002; winter average values reported are mean±se of 17 Dec 2001 ebb and flood transects.

date, tide	inner end member [TP], µM	outer end member [TP], µM	poly order	r^2	Δ[TP], µM	% Δ[TP]	interpretation
Council Creek							
17 Jan 01, ebb	85.4	1.80	2	0.94	--	--	change in flow, not interpretable
18 Jul 01, ebb	97.2	4.15	3	1.00	-31	-36	net loss
21 Aug 01, flood	83.9	4.19	3	1.00	-13	-23	net loss
17 Dec 01, ebb	85.5	2.31	4	0.99	+30	+170	net gain
17 Dec 01, flood	86.4	2.31	4	0.99	+24	+160	net gain
23 Apr 02 ebb	*2.84*	*4.46*	*2*	*0.99*	--	--	*flow off (3 days), not interpretable*
7 May 02, ebb	*4.24*	*5.05*	*4*	*0.99*	*+21*	*+460*	*net gain*
28 May 02, ebb	*4.49*	*3.62*	*4*	*0.80*	--	--	*flow off (~4 hours), not interpretable*
7 July 02, ebb	*1.14*	*3.43*	*3*	*1.00*	--	--	*change in flow, not interpretable*
16 Sep 02, ebb	150	4.91	4	1.00	-19	-16	net loss
summer avg	**110±20**	**4.42±0.25**			**-21±5.3**	**-25±6**	**net loss**
winter avg	**86±0.45**	**2.31**			**27±3.0**	**165±5**	**net gain**
Little Creek							
17 Sep 02, ebb	6.85	4.48	5	0.86	+21	+330	net loss

Table 3-8: Summary of SiO_4 processing during water quality transects. Poly order and r^2 refer to the polynomial order and correlation coefficient of the curve fitted to the data. Δ [SiO_4] is the maximum change in concentration from conservative mixing followed by the % Δ [SiO_4], which represents the % loss or gain from the conservative concentration. The inner and outer marsh Δ [NH_4] and % Δ [NH_4] values for the 17 Dec 02 transects were averaged to get the net effect on this day. Italicized transects were not interpretable due to violations of flow stability (see Table 3-1); however, the 7 May 2002 transect was used as an example of nutrient distributions when WWTP discharge had been discontinued for 17 days. The summer average values reported are mean±se of 18 July 2001, 21 August 2001, and 16 Sept 2002; winter average values reported are mean±se of 17 Dec 2001 ebb and flood transects.

date, tide	inner end member [SiO_4], μM	outer end member [SiO_4], μM	poly order	r^2	Δ [SiO_4], μM	% Δ [SiO_4]	interpretation
Council Creek							
17 Jan 01, ebb	--	--	--	--	--	--	change in flow, not interpretable
18 Jul 01, ebb	259	78.3	2	0.99	-47	-19	net loss
21 Aug 01, flood	309	71.4	1	0.99	cons.	cons.	cons.
17 Dec 01, ebb	240	64.7	4	0.99	+86	+90	net gain
17 Dec 01, flood	246	64.7	4	0.99	+73	+79	net gain
23 Apr 02 ebb	*302*	*27.6*	*1*	*0.98*	--	--	*flow off (3 days), not interpretable*
7 May 02, ebb	*310*	*26.1*	*4*	*1.00*	*-76*	*-69*	*net loss*
28 May 02, ebb	*258*	*50.4*	*2*	*0.97*	--	--	*flow off (~4 hours), not interpretable*
7 July 02, ebb	*283*	*70.9*	*2*	*1.00*	--	--	*change in flow, not interpretable*
16 Sep 02, ebb	267	82.9	5	1.00	cons.	cons.	cons.
summer avg	**278±16**	**78±3**			**-16±16**	**-6.33±6.33**	**net loss**
winter avg	**243±3**	**65**			**80±7**	**85±6**	**net gain**
Little Creek							
17 Sep 02, ebb	323	89.7	2	1.00	-39	-19	net loss

Table 3-9: Summary of evapotranspiration estimates from other marshes.

location	time period	ET, mm hr^{-1}	reference
Chesapeake Bay	June - Sept	0.3-0.5	Hussey and Odum 1992
Wisconsin marshes	May - Oct	0.2-0.5	Lott and Hunt 2001
Florida coastal marshes	annual avg.	0.18	Sutula et al. 2001
Florida Everglades	annual avg.	0.15	Guardo 1999
Australian salt marshes	Jan - Apr	0.16	Hughes et al. 1998
overall mean		**0.25**	
Chesapeake Bay mean		**0.40**	

FIGURES LEGENDS

Figure 3-1: Upper panel: Location of the Choptank Basin relative to Chesapeake Bay. Main panel: Location of Easton, Council Creek, and Little Creek (see Figures 3-2 and 3-3)

Figure 3-2: Council Creek Basin Top panel: aerial photograph of Council Creek basin, the yellow polygon indicates the basin boundary; bottom panel: aerial photograph of Council Creek marsh. Orange dots indicate locations of fixed water quality sampling sites and the orange line indicates the centerline transect of Council Creek. The green polygons in both panels indicate the Council Creek marsh boundary.

Figure 3-3: Little Creek Basin Top panel: aerial photograph of Little Creek basin, the yellow polygon indicates the basin boundary; bottom panel: aerial photograph of Little Creek marsh. Orange dots and line indicate water quality sampling sites. The green polygons in both panels indicate the Little Creek marsh boundary.

Figure 3-4: Council Creek Temperature Coefficient: conductivity μmho cm^{-1} plotted as a function of temperature (˚C) for the determination of the temperature coefficient of conductivity specific to Council Creek.

Figure 3-5: Hypothetical mixing diagram examples: Red points and line are the endmembers and the conservative mixing line. Orange points and line show a pattern of net gain of nutrients, blue points and line show a pattern of net loss of nutrients, and black points and line show conservative behavior. Δ conc. Is the maximum difference between observed and predicted conservative concentration.

Figure 3-6: Low tide time series: Panels A, B, and C: Time series of low tide samples at the mouth of Council Creek for cond (25) (μmho cm^{-1}), TN and, TP (μM), respectively. Panel D: Discharge records from the Easton WWTP (m^3 d^{-1}). Gray dashed line represents the average ± se daily wastewater discharge.

Figure 3-7: Weekly average of low tide sampling: Panel A: Weekly averaged cond (25) (μmho cm^{-1}) values for the low tide sampling. The dashed line indicates the mean wastewater cond (25) value averaged from all transects where there was wastewater discharge. Dashed line in panels A indicates the mean cond(25) for the winter transects. Panel B: Weekly averaged TN (μM) values for the low tide sampling. Panel C: Weekly averaged TP (μM) values for the low tide sampling. Blue open triangles represent concentrations of wastewater at the outfall, and red filled circles represent concentrations at the mouth of the creek. Dashed lines in panels B and C indicate the study period mean values of discharge water averaged over the entire study period (November – March 2001, 2002).

Figure 3-8: Marsh flushing: Panel A: Cond(25) before, during, and after a WWTP shutdown event. Panel B: TN before, during, and after a WWTP shutdown event.

The red arrow indicates the concentration difference between high and low steady states, while the black arrows indicate the time from the cessation of discharge to reach inflection point. Panel C: TP before, during, and after a WWTP shutdown event. Panel D: WWTP discharge record. In the top three panels the solid black lines are the regression of the sigmoid model fitted to the data, and the blue dotted lines are estimated Choptank River concentrations.

Figure 3-9: Study period flow record: Panel A: Choptank River flow conditions (10^6 m^3 d^{-1}) at the USGS gauging station at Greensboro, MD. Panel B: Wastewater discharge (m^3 d^{-1}) into Council Creek. Vertical blue dashed lines indicate stable flow conditions (interpretable), vertical gray dashed lines indicate flow instability (not interpretable), and the vertical red dashed line indicates no wastewater flow.

Figure 3-10: N concentrations during the 21 August 2001transect: Panel A: Mixing diagram for NH_4. Panel B: Mixing diagram for NO_3. Panel C: Mixing diagram for TN. Open triangles represent the stream above the wastewater outfall. Open squares and dotted lines represent the end members and conservative mixing line. Filled circles and solid line represent all transect data and a polynomial curve fitted to the data.

Figure 3-11: P and SiO_4 concentrations during the 21 August 2001transect: Panel A: Mixing diagram for PO_4. Panel B: Mixing diagram for TP. Panel C: Mixing diagram for SiO_4. The polynomial regression was omitted in Panel C, since SiO_4 was conservative (slope through observed points not significantly different from conservative mixing line). Open triangles represent the stream above the wastewater outfall. Open squares and dotted lines represent the end members and conservative mixing line. Filled circles and solid line represent all transect data and the polynomial line fitted to the data.

Figure 3-12: N concentrations on the ebb tide transect of 17 December 2001: Panel A: Mixing diagram for NH_4. Panel B: Mixing diagram for NO_3. Panel C: Mixing diagram for TN. Open triangles represent the stream above the wastewater outfall. Open squares and dotted lines represent the end members and conservative mixing line. Filled circles and solid line represent all transect data and the polynomial line fitted to the data.

Figure 3-13: P and SiO_4 concentrations on the ebb tide transect of 17 December 2001: Panel A: Mixing diagram for PO_4. Panel B: Mixing diagram for TP. Panel C: Mixing diagram for SiO_4. Open triangles represent the stream above the wastewater outfall. Open squares and dotted lines represent the end members and conservative mixing line. Filled circles and solid line represent all transect data and the polynomial line fitted to the data.

Figure 3-14: N concentrations during the 7 May 2002 transect (no wastewater discharge for 17 days): Panel A: Mixing diagram for NH_4. Panel B: Mixing diagram for NO_3. Panel C: Mixing diagram for TN. Open triangles represent the stream

Choptank Basin

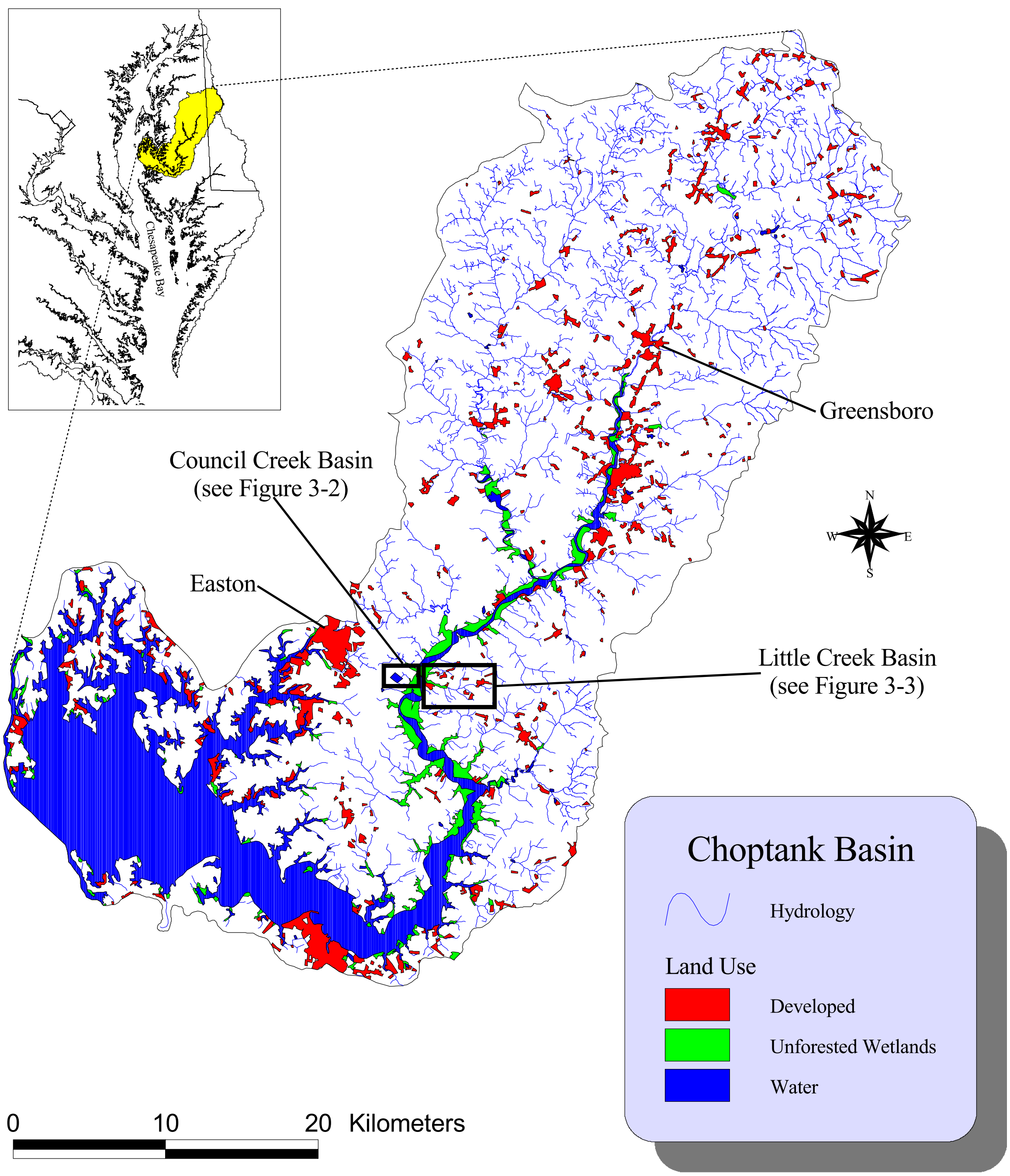


Figure 3-1

Council Creek Basin and Marsh

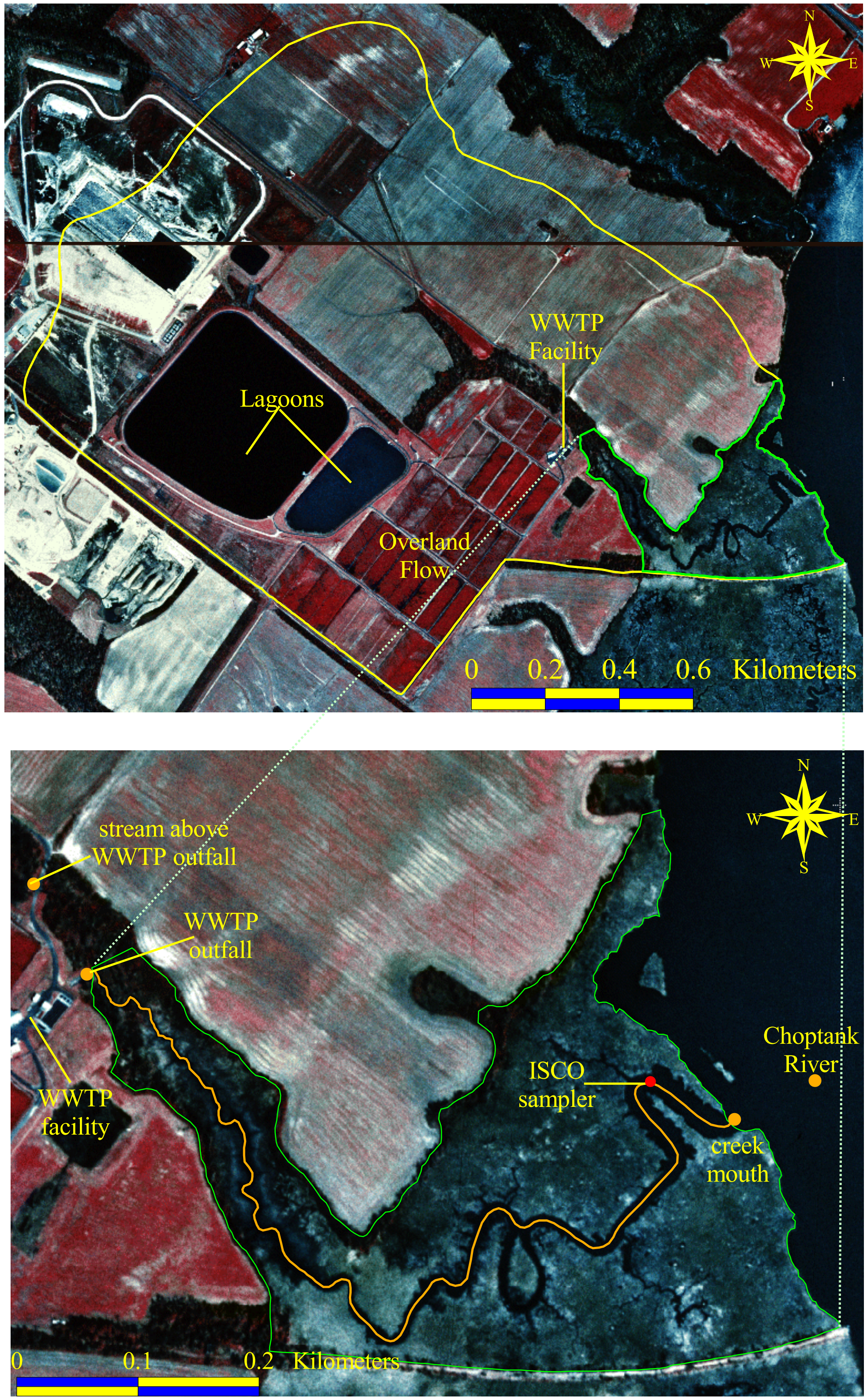


Figure 3-2

Little Creek Basin and Marsh

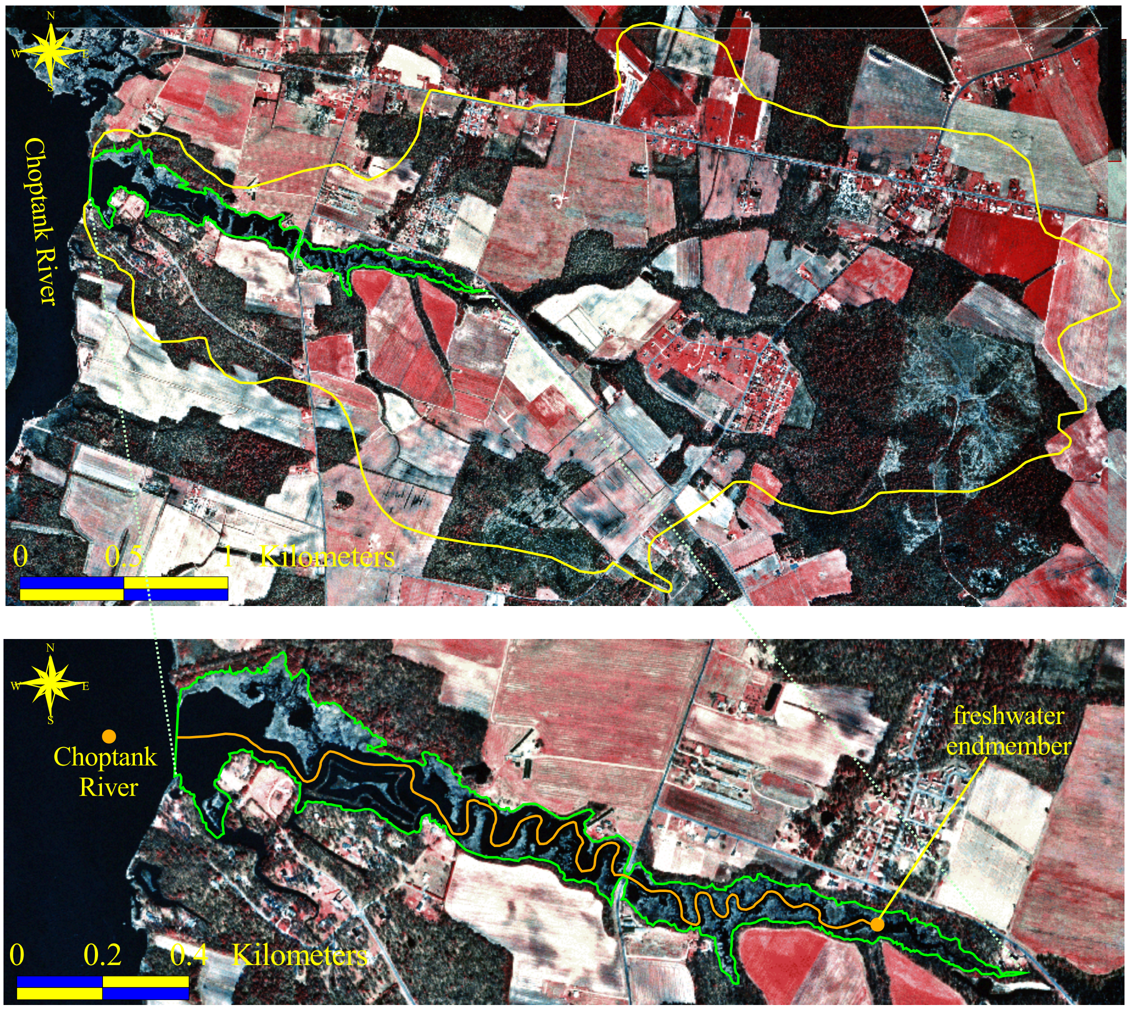
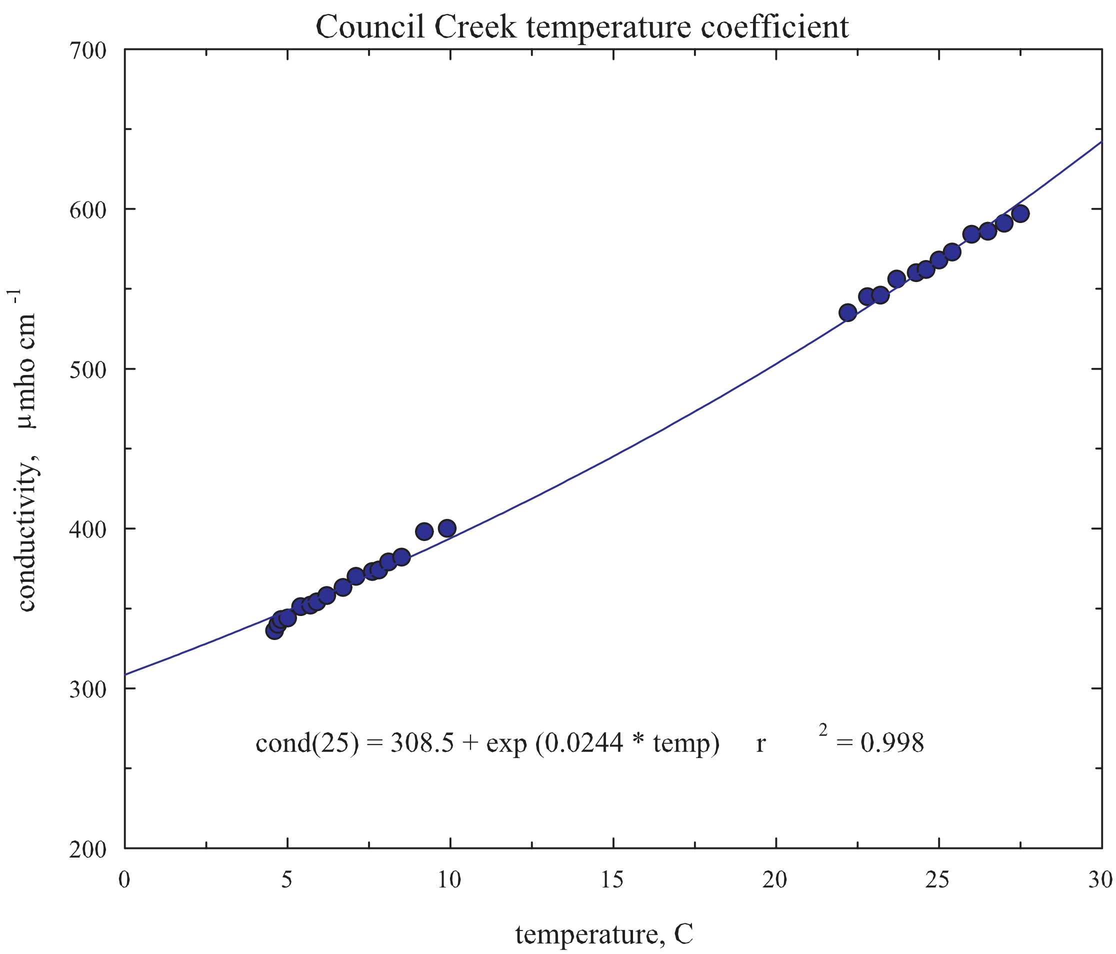


Figure 3-3



Council Creek temperature coefficient
conductivity, µmho cm -1
temperature, C
700
600
500
400
300
200
0
5
10
15
20
25
30
cond(25) = 308.5 + exp (0.0244 * temp) r 2 = 0.998

Figure 3-4

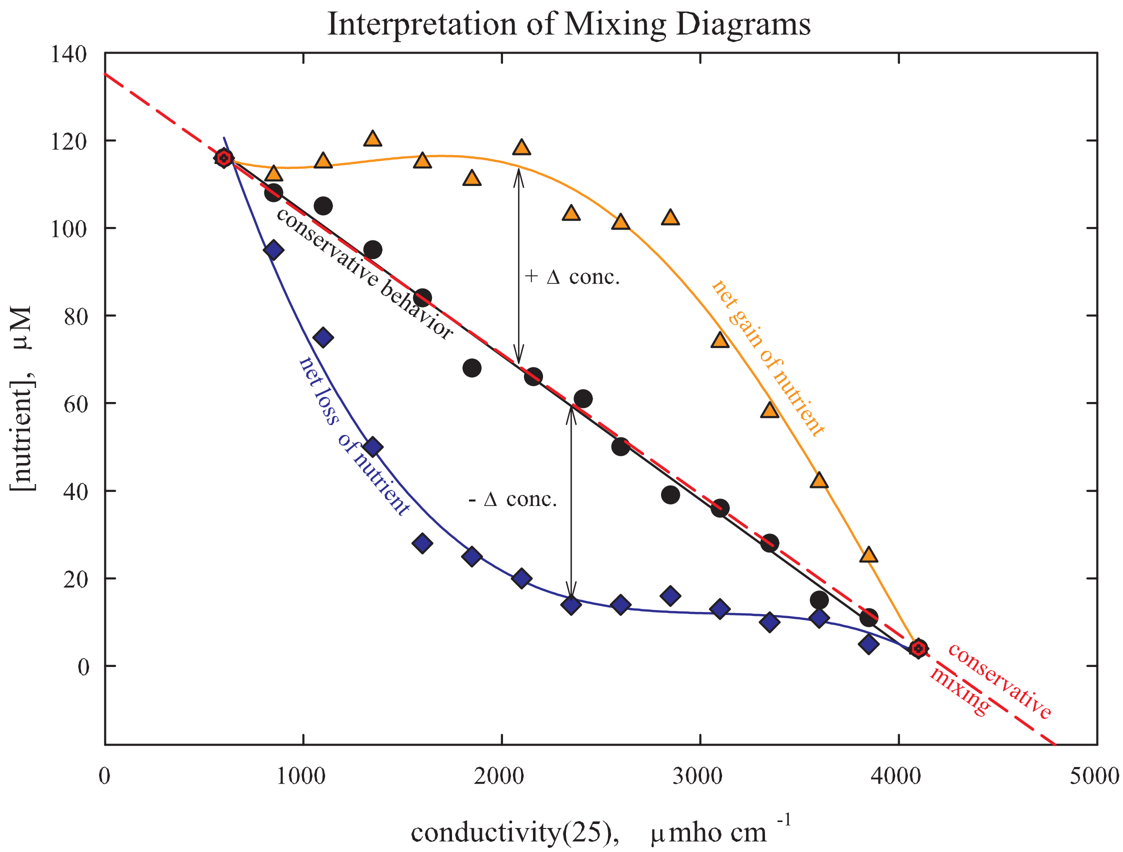
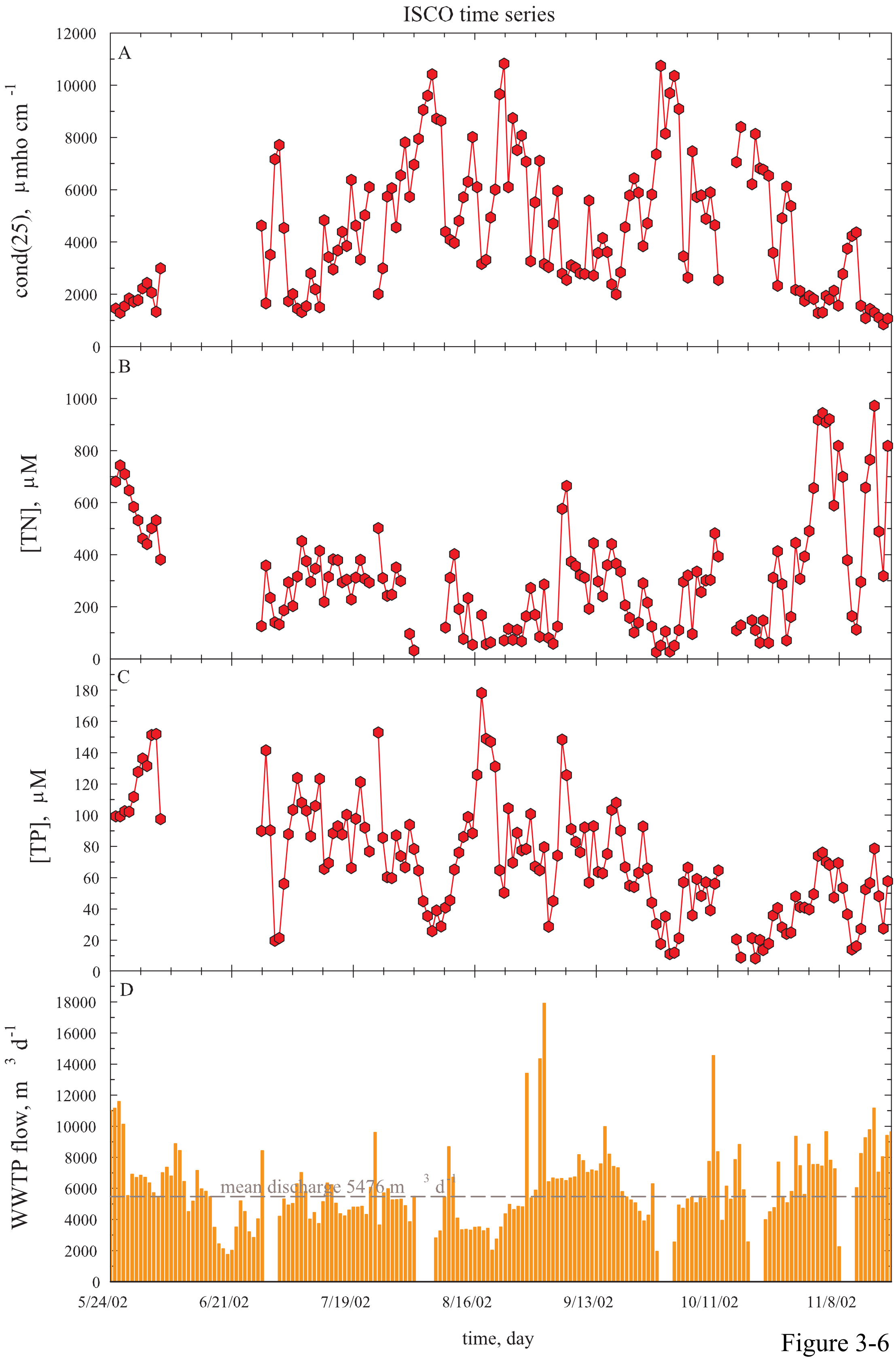

Interpretation of Mixing Diagrams
[nutrient], µM
conductivity(25), µmho cm $^{-1}$
conservative behavior
net gain of nutrient
net loss of nutrient
+ Δ conc.
- Δ conc.
conservative mixing
140
120
100
80
60
40
20
0
0
1000
2000
3000
4000
5000

Figure 3-5



ISCO time series
A
cond(25), μmho cm -1
B
[TN], μM
C
[TP], μM
D
WWTP flow, m 3 d-1
mean discharge 5476 m 3 d-1
5/24/02
6/21/02
7/19/02
8/16/02
9/13/02
10/11/02
11/8/02
time, day

Figure 3-6

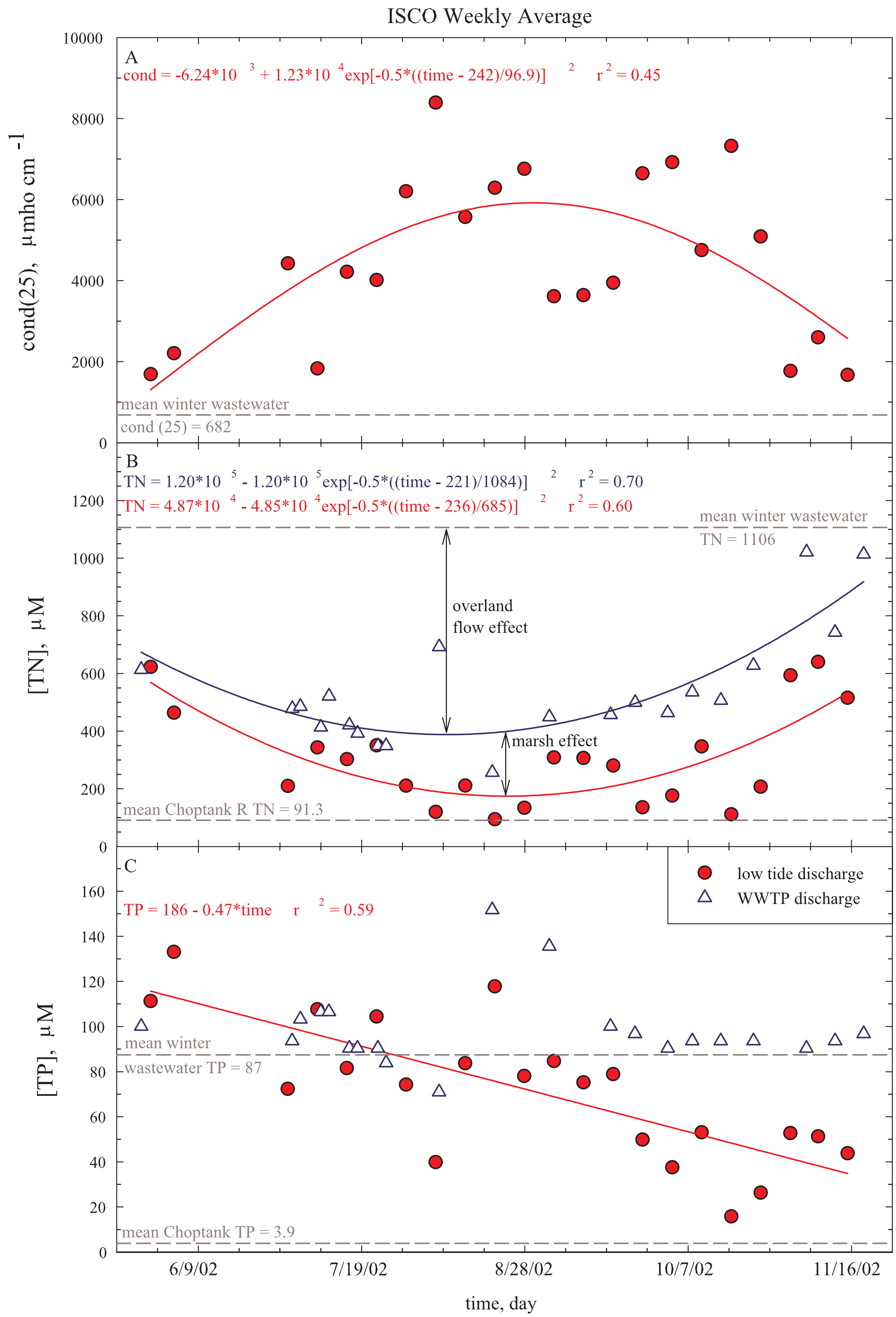

ISCO Weekly Average
A
cond = -6.24*10 3 + 1.23*10 4exp[-0.5*((time - 242)/96.9)] 2 r 2 = 0.45
cond(25), μmho cm -1
mean winter wastewater
cond (25) = 682
B
TN = 1.20*10 5 - 1.20*10 5exp[-0.5*((time - 221)/1084)] 2 r 2 = 0.70
TN = 4.87*10 4 - 4.85*10 4exp[-0.5*((time - 236)/685)] 2 r 2 = 0.60
mean winter wastewater
TN = 1106
overland
flow effect
marsh effect
mean Choptank R TN = 91.3
[TN], μM
C
TP = 186 - 0.47*time r 2 = 0.59
low tide discharge
WWTP discharge
mean winter
wastewater TP = 87
mean Choptank TP = 3.9
[TP], μM
6/9/02
7/19/02
8/28/02
10/7/02
11/16/02
time, day

Figure 3-7

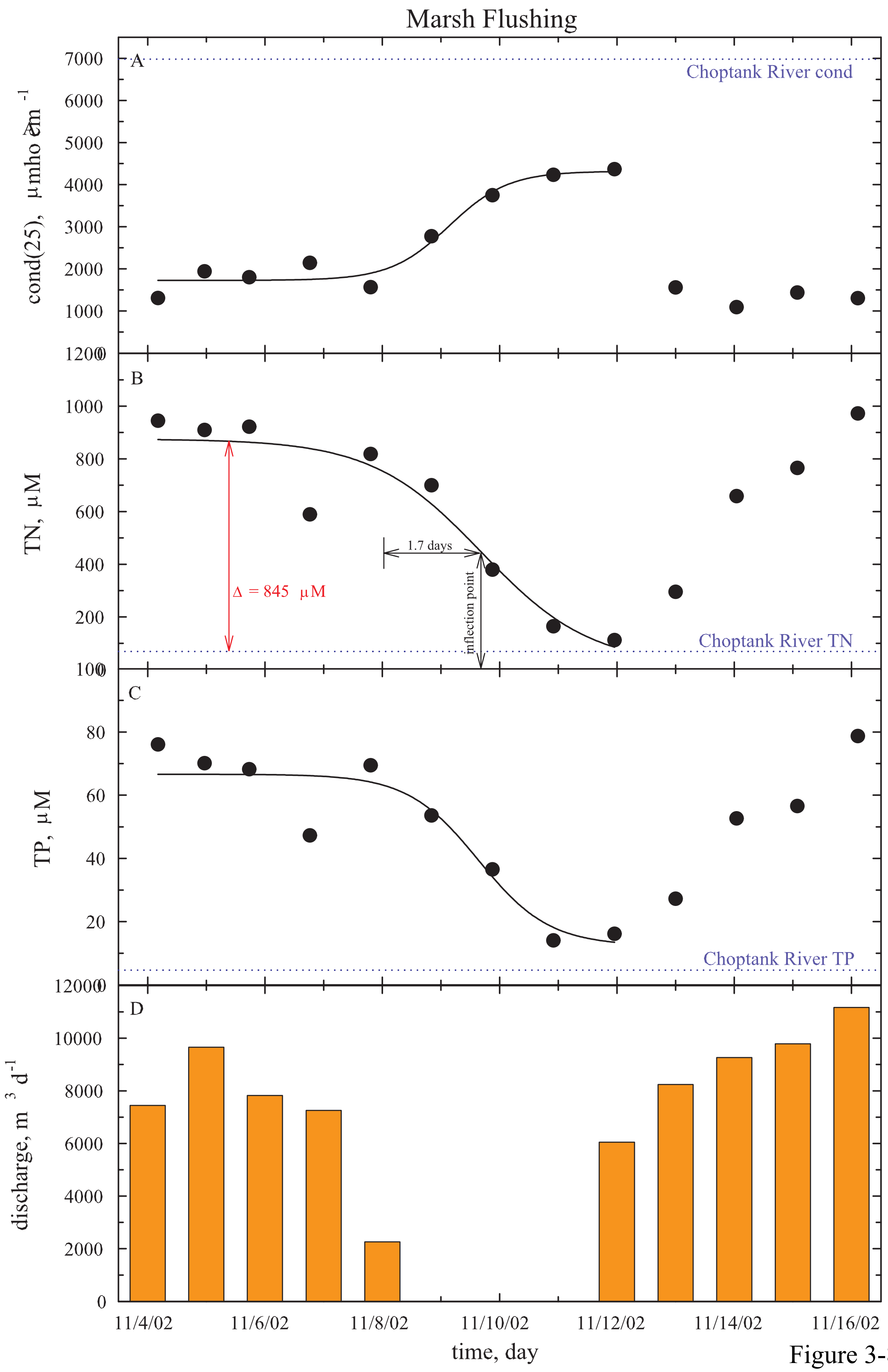

Marsh Flushing
A
Choptank River cond
cond(25), µmho cm⁻¹
B
TN, µM
Δ = 845 µM
1.7 days
inflection point
Choptank River TN
C
TP, µM
Choptank River TP
D
discharge, m³ d⁻¹
11/4/02
11/6/02
11/8/02
11/10/02
11/12/02
11/14/02
11/16/02
time, day

Figure 3-8

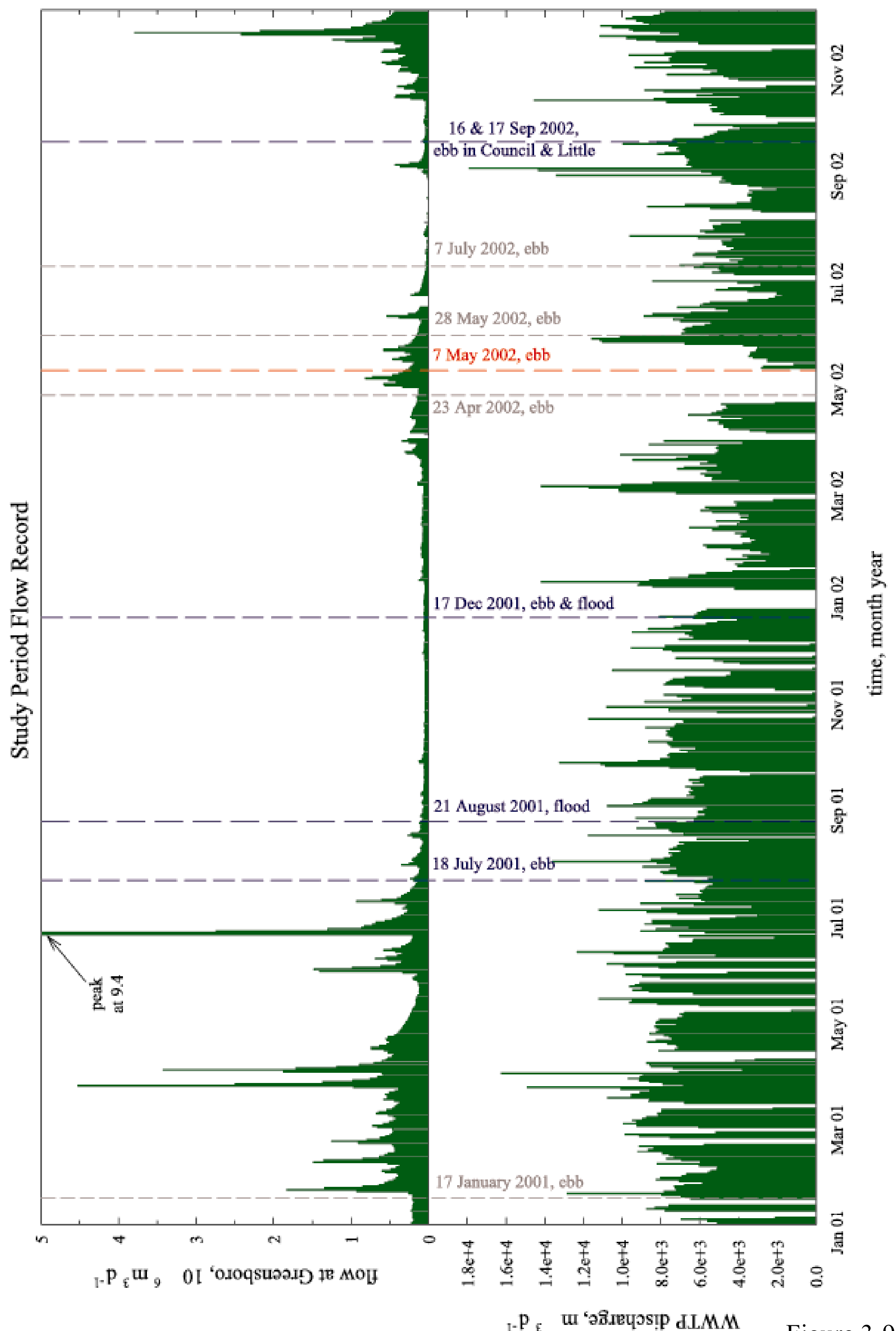

Study Period Flow Record
flow at Greensboro, 10 6 m 3 d-1
WWTP discharge, m 3 d-1
time, month year
peak at 9.4
17 January 2001, ebb
18 July 2001, ebb
21 August 2001, flood
17 Dec 2001, ebb & flood
23 Apr 2002, ebb
7 May 2002, ebb
28 May 2002, ebb
7 July 2002, ebb
16 & 17 Sep 2002, ebb in Council & Little
Jan 01
Mar 01
May 01
Jul 01
Sep 01
Nov 01
Jan 02
Mar 02
May 02
Jul 02
Sep 02
Nov 02
0.0
2.0e+3
4.0e+3
6.0e+3
8.0e+3
1.0e+4
1.2e+4
1.4e+4
1.6e+4
1.8e+4
0
1
2
3
4
5

Figure 3-9

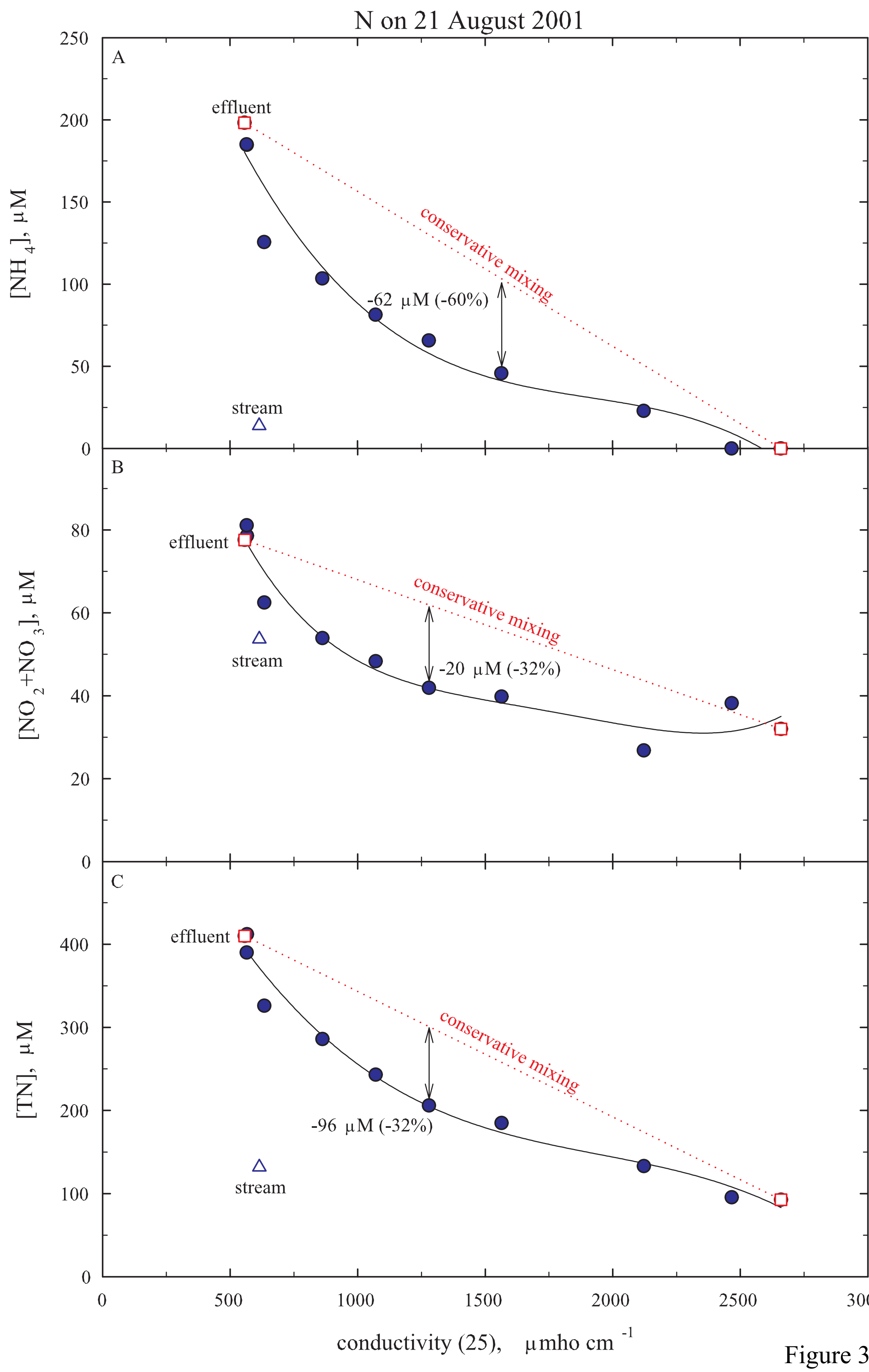
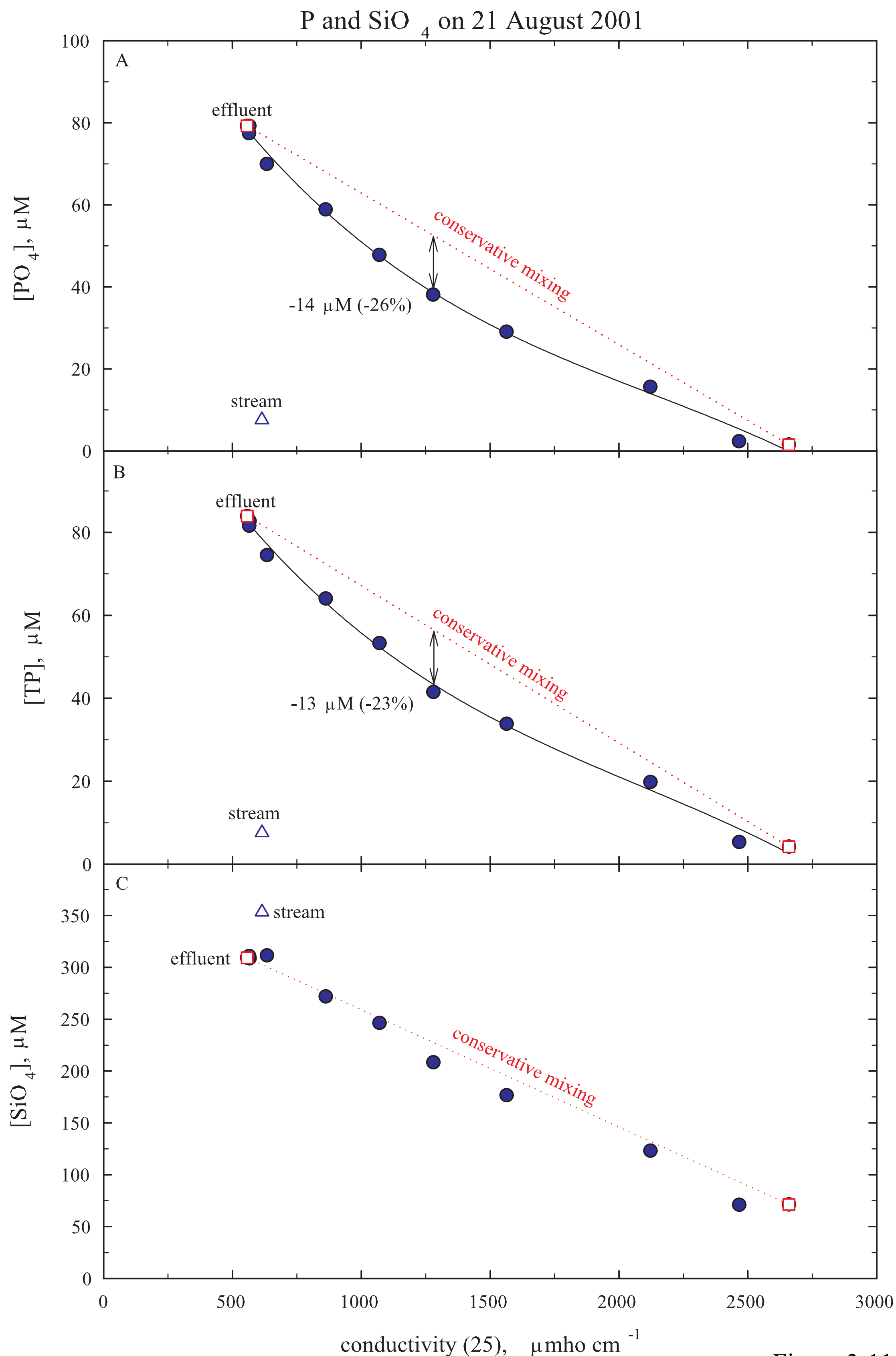
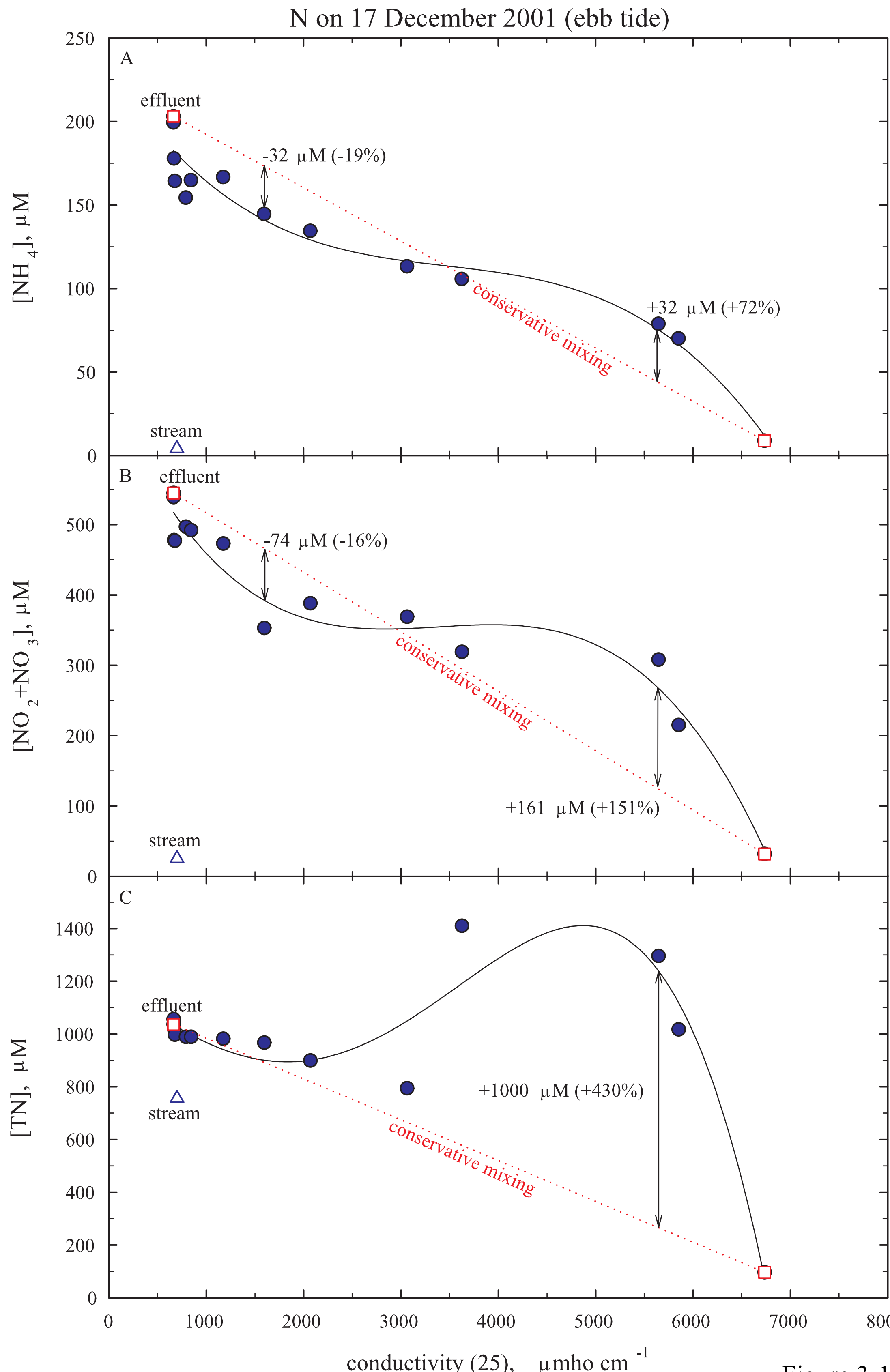

N on 21 August 2001
A
effluent
conservative mixing
-62 μM (-60%)
stream
[NH 4], μM
B
effluent
conservative mixing
stream
-20 μM (-32%)
[NO 2 +NO 3], μM
C
effluent
conservative mixing
-96 μM (-32%)
stream
[TN], μM
0
500
1000
1500
2000
2500
3000
conductivity (25), μmho cm -1

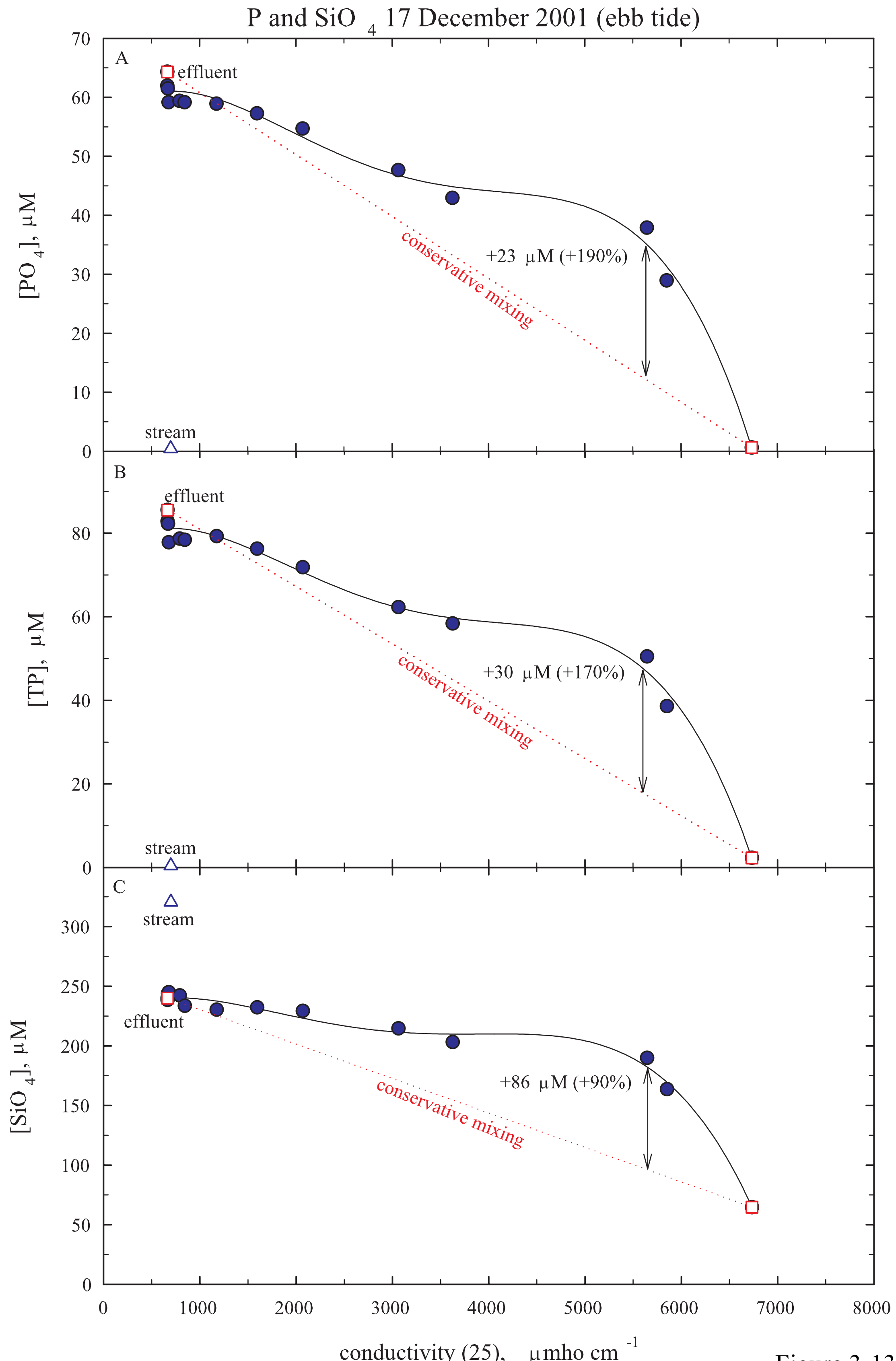
Figure 3-10



P and SiO 4 on 21 August 2001
A
effluent
conservative mixing
-14 µM (-26%)
stream
[PO 4], µM
B
effluent
conservative mixing
-13 µM (-23%)
stream
[TP], µM
C
stream
effluent
conservative mixing
[SiO 4], µM
conductivity (25), µmho cm -1

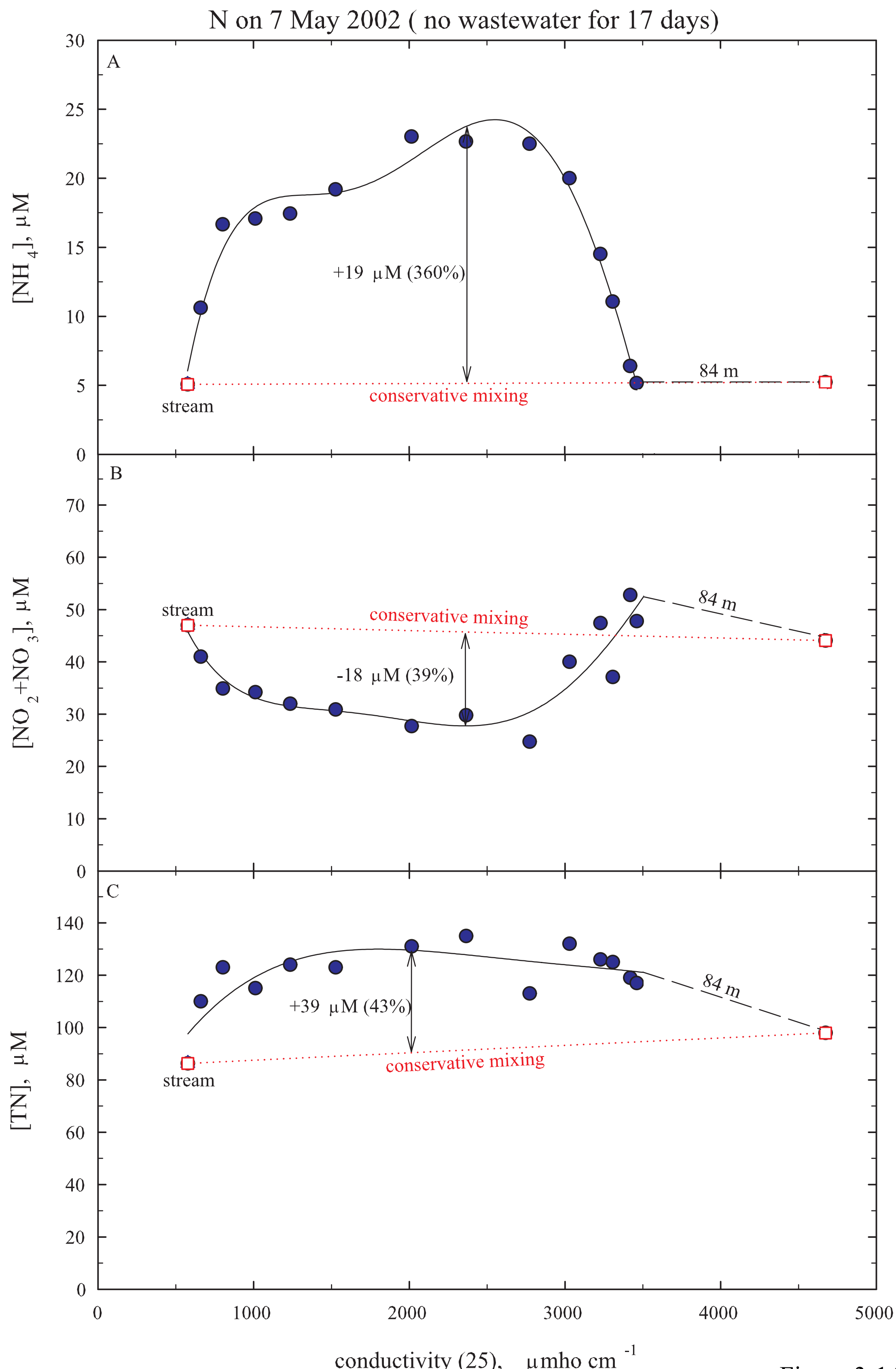
Figure 3-11


N on 17 December 2001 (ebb tide)
A
effluent
-32 μM (-19%)
conservative mixing
+32 μM (+72%)
stream
B
effluent
-74 μM (-16%)
conservative mixing
+161 μM (+151%)
stream
C
effluent
stream
+1000 μM (+430%)
conservative mixing
[NH 4], μM
[NO 2+NO 3], μM
[TN], μM
conductivity (25), μmho cm -1
0
50
100
150
200
250
300
400
500
600
800
1000
1200
1400
2000
3000
4000
5000
6000
7000
8000
Figure 3-12



P and SiO 4 17 December 2001 (ebb tide)
A
effluent
stream
+23 µM (+190%)
conservative mixing
[PO 4], µM
B
effluent
stream
+30 µM (+170%)
conservative mixing
[TP], µM
C
stream
effluent
+86 µM (+90%)
conservative mixing
[SiO 4], µM
conductivity (25), µmho cm -1

Figure 3-13



N on 7 May 2002 (no wastewater for 17 days)
A
+19 µM (360%)
84 m
stream
conservative mixing
[NH_4], µM
B
stream
conservative mixing
84 m
-18 µM (39%)
[NO_2+NO_3], µM
C
+39 µM (43%)
84 m
conservative mixing
stream
[TN], µM
conductivity (25), µmho cm^{-1}

Figure 3-14

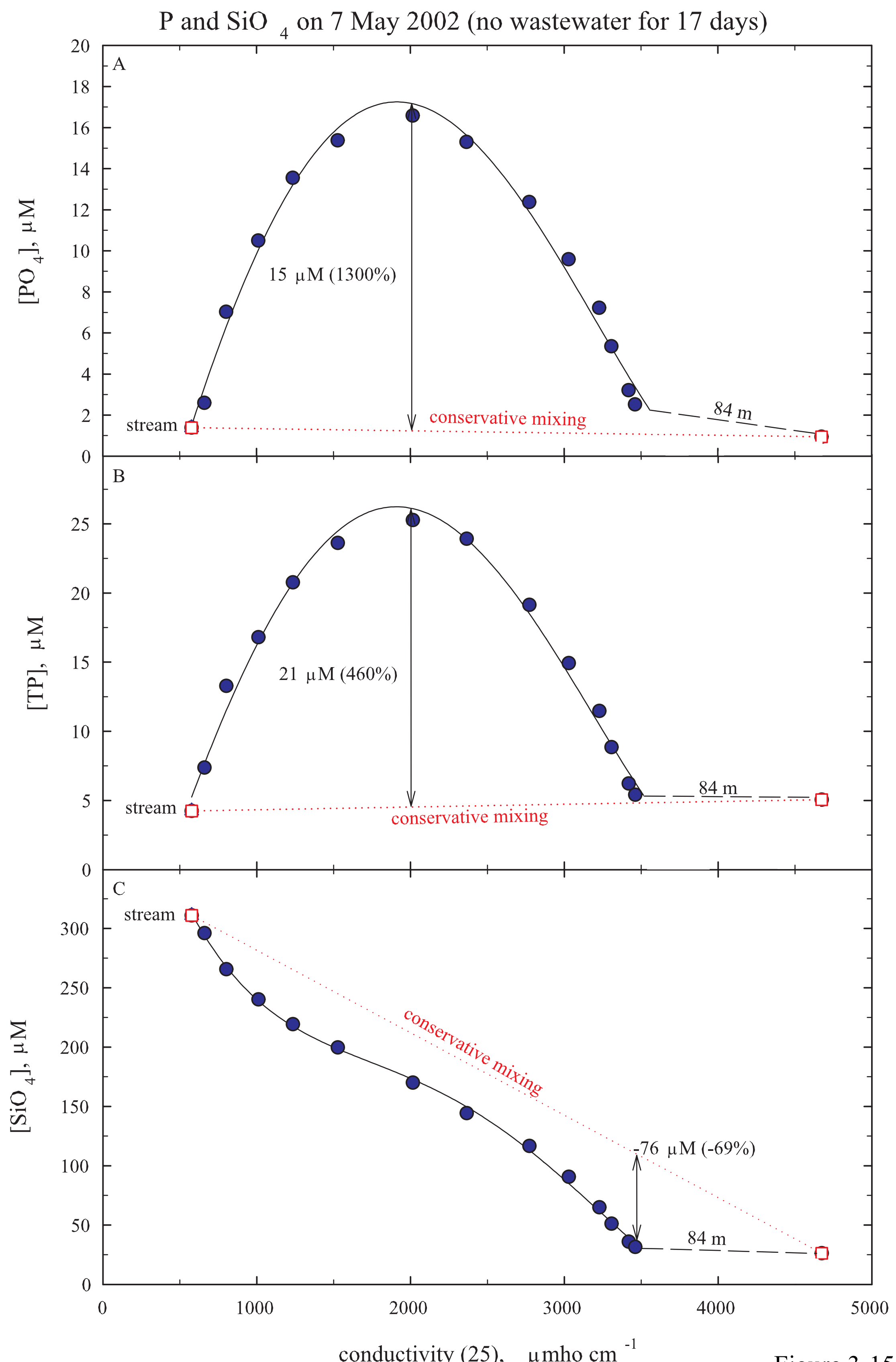
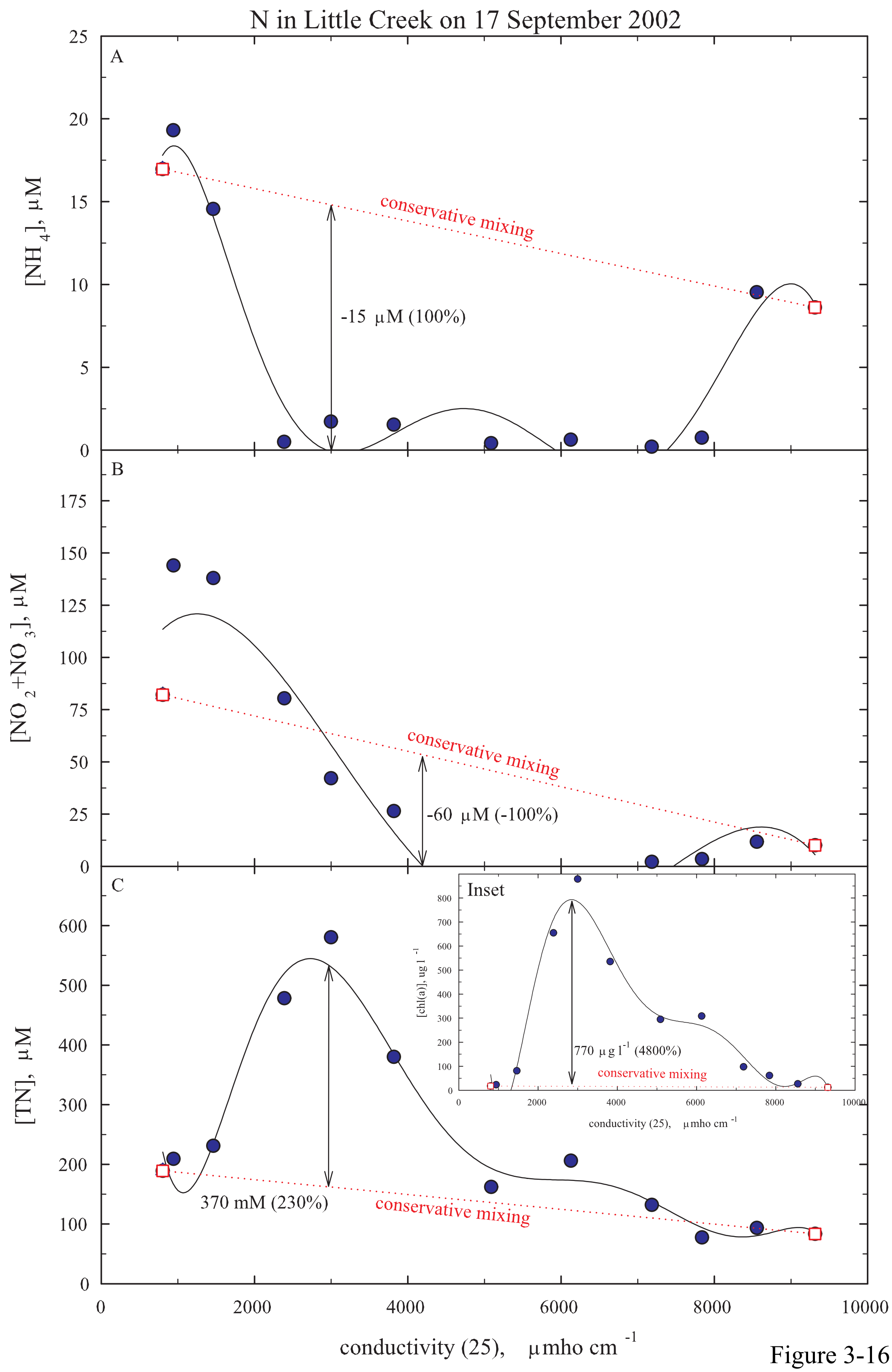
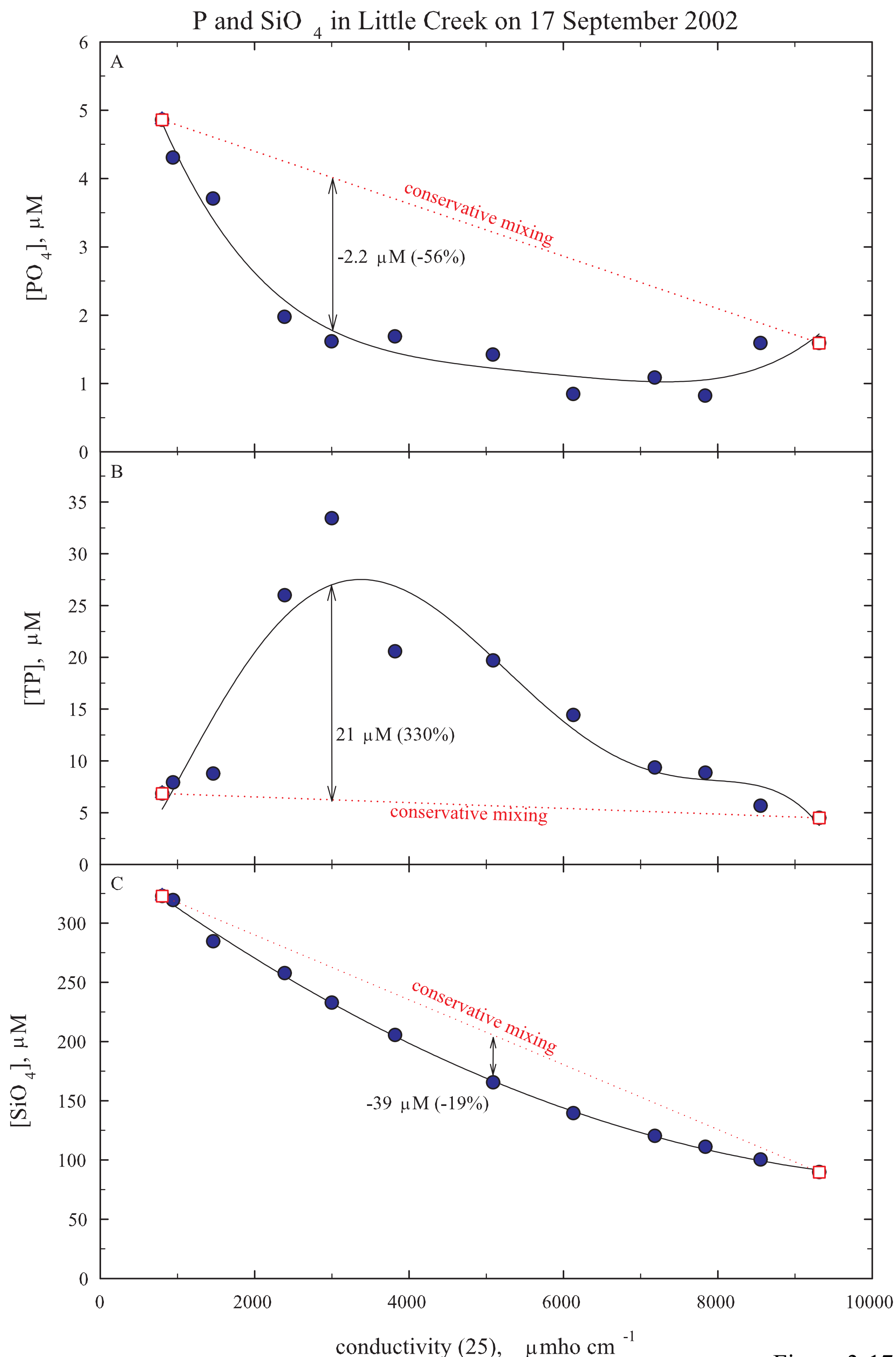

P and SiO 4 on 7 May 2002 (no wastewater for 17 days)
A
[PO4], µM
15 µM (1300%)
stream
conservative mixing
84 m
B
[TP], µM
21 µM (460%)
stream
conservative mixing
84 m
C
[SiO4], µM
stream
conservative mixing
-76 µM (-69%)
84 m
conductivity (25), µmho cm -1

Figure 3-15



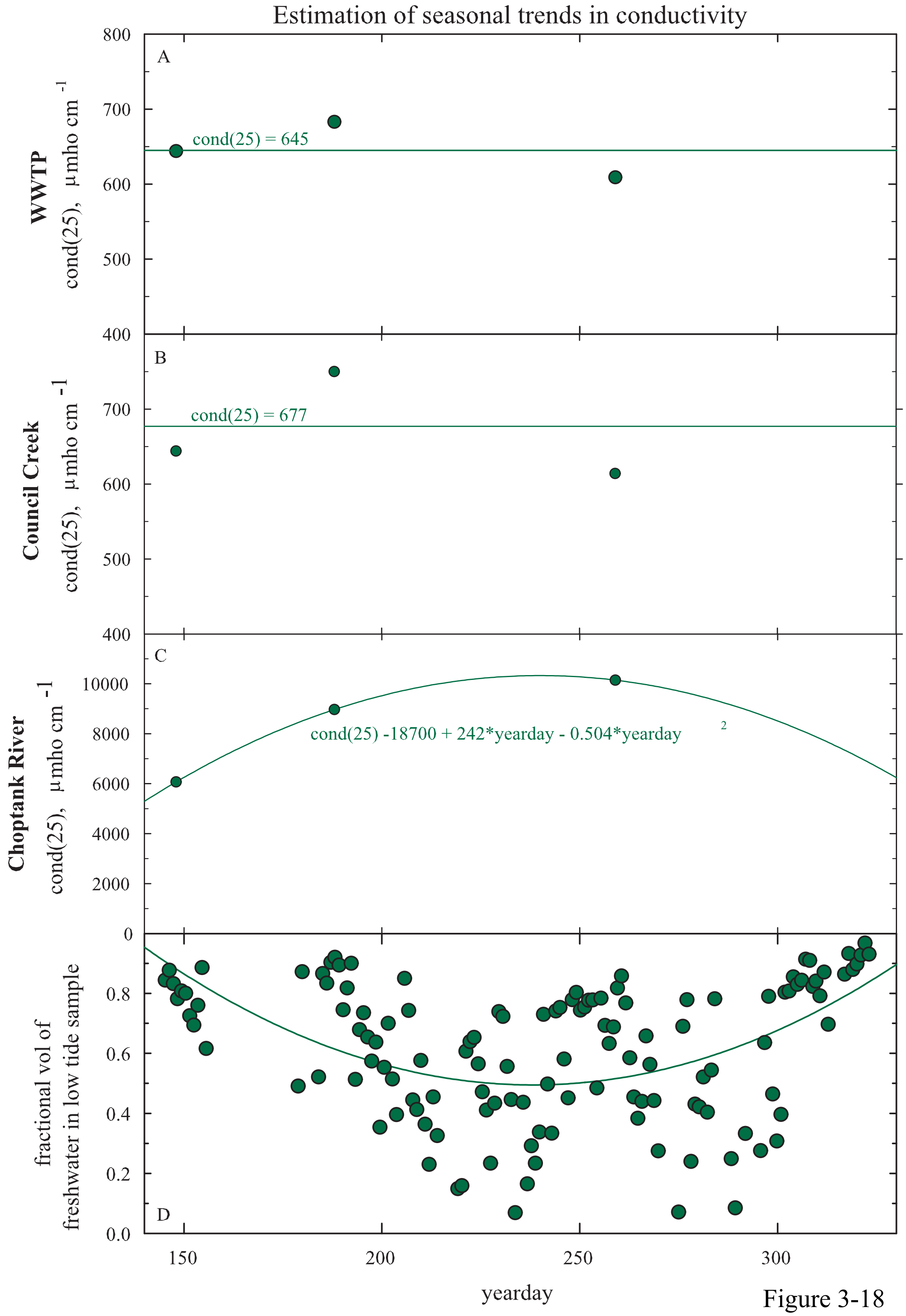
N in Little Creek on 17 September 2002
A
conservative mixing
-15 µM (100%)
$[NH_4]$, µM
B
conservative mixing
-60 µM (-100%)
$[NO_2+NO_3]$, µM
C
Inset
770 µg l^{-1} (4800%)
conservative mixing
[chl(a)], ug l^{-1}
conductivity (25), µmho cm^{-1}
370 mM (230%)
conservative mixing
[TN], µM
conductivity (25), µmho cm^{-1}

Figure 3-16



P and SiO 4 in Little Creek on 17 September 2002
A
[PO4], µM
conservative mixing
-2.2 µM (-56%)
B
[TP], µM
21 µM (330%)
conservative mixing
C
[SiO4], µM
conservative mixing
-39 µM (-19%)
conductivity (25), µmho cm -1
0
1
2
3
4
5
6
5
10
15
20
25
30
35
50
100
150
200
250
300
2000
4000
6000
8000
10000

Figure 3-17



Estimation of seasonal trends in conductivity
A
WWTP
cond(25), μmho cm -1
cond(25) = 645
B
Council Creek
cond(25), μmho cm -1
cond(25) = 677
C
Choptank River
cond(25), μmho cm -1
cond(25) -18700 + 242*yearday - 0.504*yearday 2
D
fractional vol of
freshwater in low tide sample
yearday

Figure 3-18

Estimation of seasonal trends in TN and TP

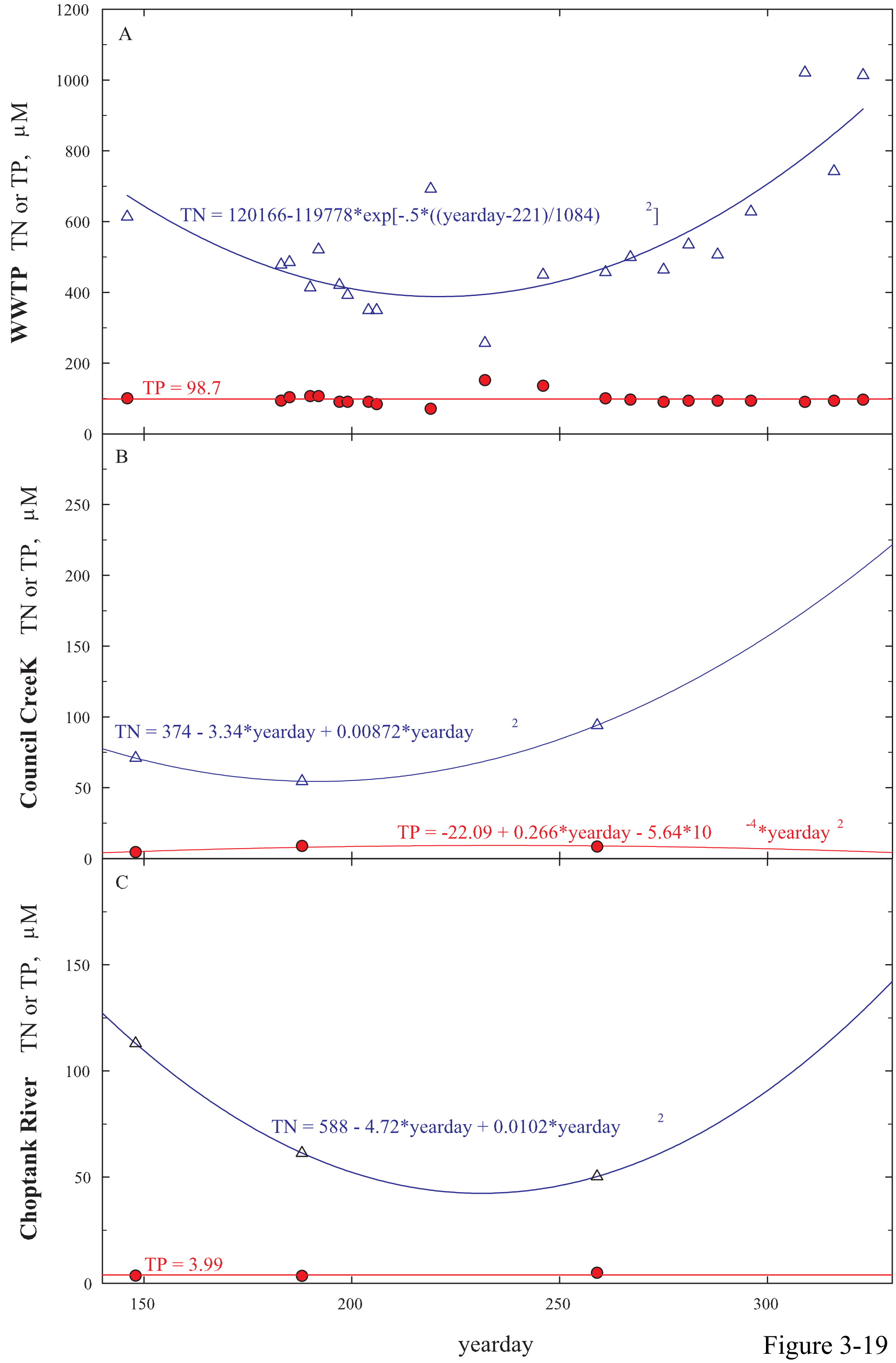
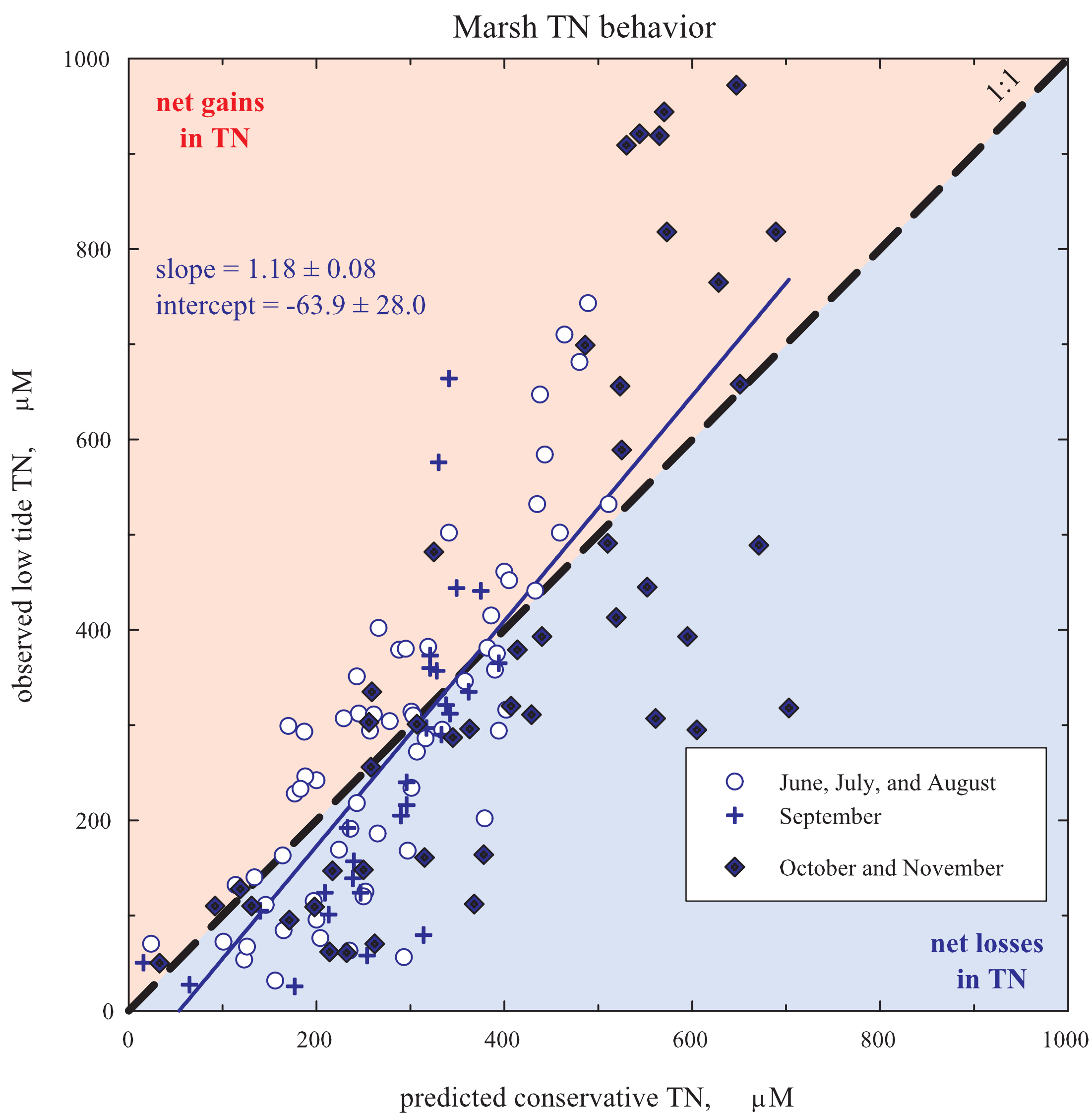
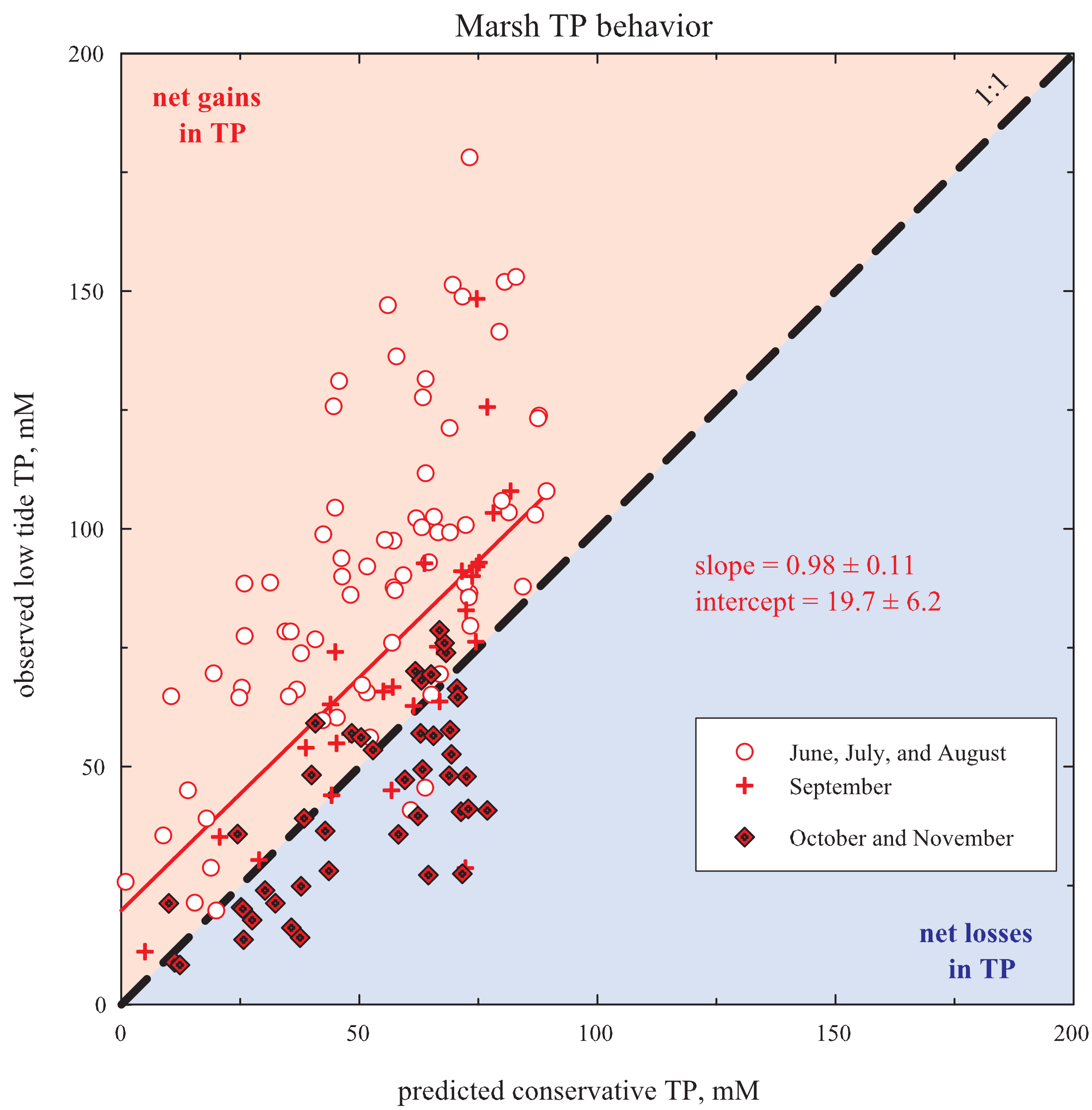


Figure 3-19



Marsh TN behavior
net gains
in TN
slope = 1.18 ± 0.08
intercept = -63.9 ± 28.0
1:1
June, July, and August
September
October and November
net losses
in TN
observed low tide TN, µM
predicted conservative TN, µM
0
200
400
600
800
1000

Figure 3-20



Marsh TP behavior
net gains
in TP
net losses
in TP
1:1
slope = 0.98 ± 0.11
intercept = 19.7 ± 6.2
June, July, and August
September
October and November
observed low tide TP, mM
predicted conservative TP, mM
0
50
100
150
200

Figure 3-21

Chapter 4

COUNCIL CREEK MARSH – COMPARISON OF MACROPHYTE BIOMASS AND NUTRIENT COMPOSITION BETWEEN TWO OLIGOHALINE MARSHES

Abstract

One of the factors contributing to high wetland productivity and nutrient assimilative capacity is the growth of macrophyte vegetation. In this chapter of my thesis, the hypotheses tested were (1) that high N and P loading from the wastewater effluent passing through the marsh increases the aboveground biomass and nutrient concentrations of macrophytes in an oligohaline marsh relative to a non-wastewater impacted control marsh and (2) that competitive exclusion of slower growing species by faster growing species under the nutrient-enriched conditions causes fewer plant species to be present in the wastewater-impacted marsh compared to a nearby marsh without wastewater. These hypotheses were tested in Council Creek marsh, 0.16 km^2 and Little Creek marsh, 0.27 km^2 in the Choptank Basin on the Delmarva Peninsula. The marsh basins have different land use characteristics with Council Creek being impacted by wastewater discharge and part of the Talbot County, MD landfill, while Little Creek basin is largely comprised of forest and agriculture.

Wastewater discharged into Council Creek marsh has a moderate impact on marsh vegetation, partially supporting the hypotheses. There was no evidence of significant differences in species richness between Council Creek marsh and Little Creek marsh, and aboveground plant biomass and carbon storage in Council Creek marsh were not significantly different. However, in 11 of 70 tests (16%), macrophytic vegetation in Council Creek was N and P enriched with only 1 test (1%) that supported higher

macrophyte nutrient concentrations in Little Creek marsh. Using the biomass data measured here, over the growing season (May – Sept), marsh plants are potentially capable of intercepting ~30-44% of the wastewater N and 11-17% of the wastewater P passing through Council Creek marsh.

Introduction

Wetlands are generally known to be highly productive ecosystems (Kadlec and Knight 1996, Mitsch and Gosselink 2000). Located at the boundary between terrestrial and aquatic ecosystems, wetlands have an adequate water supply to support high rates of biological activity and biogeochemical processing of material passing from land to water. With increased human populations and anthropogenic disturbance, wetlands also have the potential to store and/or intercept large quantities of terrestrially derived nutrients (nitrogen and phosphorus) of anthropogenic origin in biomass and in sediments.

One of the factors contributing to high wetland productivity is the growth of macrophytic vegetation. The vascular flora varies in species composition and diversity across wetland types; of particular note are the differences between coastal marsh types which are primarily driven by salinity differences (Odum 1988). Coastal salt marshes are typically low diversity wetlands that are composed of dense monotypic stands, such as *Spartina* spp. In contrast, tidal freshwater marshes have high spatial and temporal species diversity, often with 25 – 40 species present (Odum 1988). High species diversity in these marshes is the result of reduction in salt stress, elevation gradients, and seasonal species succession (Mitsch and Gosselink 2000, Perry and Hershner 1999). However, the natural high diversity of these marshes can be reduced by the encroachment of invasive species like *Phragmites australis* (Windham and Lathrop 1999), often under conditions of anthropogenic disturbance.

Tidal freshwater marshes can produce 1000-3500 g m^{-2} y^{-1} of macrophytic biomass (Whigham et al 1978, Mitsch and Gosselink 2000). Annual production in these marshes is composed of a range of species that are adapted to different spatial and temporal growth conditions (Whigham et al 1978, Perry and Hershner 1999, and Pasternak et al 2000). Two important factors in determining species composition in tidal freshwater marshes are elevation gradients and seasonal succession. Low elevation, regularly flooded areas in tidal freshwater wetlands are dominated by emergent perennial species such as *Peltandra virginica*, with relatively high turnover rates (production:biomass, P:B, ratios of up to 2.5 y^{-1}), peak biomass that occurs in early summer, and complete loss of aboveground plant material over winter (Perry and Hershner 1999, Neubauer et al. 2000). Species such as *Leersia oryzoides* and *Polygonum arifolium* reach peak biomass in late summer (Doumele 1981) and are often found in at mid-marsh elevations (Pasternack et al 2000). Plants such as *Typha* spp have peak biomass occurring in early to mid-summer and are often found in high marsh areas (Whigham et al 1978). As these are general patterns, it should be noted, however, that there can be considerable overlap within marsh plant associations.

Recognized for their high productivity and potential value as nutrient filters, wetlands are increasingly being used to treat wastewater (Kadlec and Knight 1996). In a literature survey of North American temperate wetlands by Bedford et al (1999), average nitrogen and phosphorus content of marsh plants was found to be 1.97% N (0.83-4.20%) and 0.29% P (0.10-0.64%). From the above rates of aboveground annual plant biomass production, wetland plants have the potential to immobilize 20-69 g N m^{-2} y^{-1} and 3-10 g

P m^{-2} y^{-1}. Additionally, increased nutrient availability can enhance macrophyte biomass production, and other processes such as microphytobenthos production sediment sorption and burial and degradation of high carbon content detritus all potentially contribute to more N and P per unit area.

Increased nutrient loading of natural marshes is not without consequences. For example, species richness (or species density, number of species per unit area, Odum 1983) may decline as a result of competitive exclusion and invasion of exotic species. Excessively high nutrient loading can also inhibit plant growth (Clarke and Baldwin 2002), and litter production at the end of the growing season increases the opportunity for export and mineralization of particulate nitrogen and phosphorus that are not translocated to belowground biomass. Increased N and P loading have also been found to reduce plant N and P efficiency of uptake (saturation effect), efficiency of recovery (nutrient translocation to belowground parts) and efficiency of use (increase in biomass resulting from increase in N an P availabilty, Shaver and Melillo 1984).

This chapter of my thesis focuses on the biomass and nutrient concentrations of marsh macrophytes in a wastewater-impacted oligohaline marsh. Council Creek marsh on the Choptank estuary receives an average of 5716 m^3 d^{-1} (1.53 MGD) of wastewater transporting an average of 59.1kg N d^{-1} and 15.6 kg P d^{-1} from the town of Easton, Maryland on the Delmarva Peninsula. I hypothesize that high N and P loading from the wastewater effluent passing through the marsh increases the aboveground biomass and nutrient concentrations of macrophytes in Council Creek marsh relative to a non-wastewater impacted control marsh directly across the Choptank River. Furthermore, I expect fewer plant species in Council Creek marsh because of competitive exclusion of slower growing species by faster growing species under the nutrient-enriched conditions. Evidence to support or reject this hypothesis should emerge from an examination of macrophyte data collected in both the control and wastewater-impacted marshes.

Study Site Description

Council Creek and Little Creek are oligohaline marshes located in the Choptank River basin on the Delmarva Peninsula on the eastern shore of the Chesapeake Bay (Figure 4-1). Council Creek basin, delineated by the yellow polygon in the top panel of Figure 4-2, has an area of 2.20 km^2 and marsh area of 0.16 km^2 delineated by the green polygons in the top panels of Figures 4-2 and 4-3. Little Creek basin, delineated by the yellow polygon in the bottom panel of Figure 4-2, has an area of 8.57 km^2 and marsh area of 0.27 km^2 delineated by the green polygons in the bottom panels of Figures 4-2 and 4-3. Council Creek marsh is designated as the treatment marsh due to nutrient enrichment from the upstream inputs of wastewater from the municipal treatment facility that serves the town of Easton, MD (Figure 4-1). Despite the Easton WWTP's partial tertiary treatment method (an overland flow system partially visible in the upper panel of Figure 4-3), there are still high concentrations of N and P in wastewater discharged nearly continuously into Council Creek (see Chapter 3). In contrast, Little Creek is the control marsh receiving no direct municipal wastewater inputs; however, there are several low-density urban areas with septic systems adjacent to Little Creek marsh (Figure 4-3, lower panel). Since Little Creek is located east of Council Creek directly across the Choptank River (Figure 4-1), both marshes are subjected to similar salinity regimes, weather conditions, and tidal cycling; therefore, differences in plant community structure, aboveground biomass, and nutrient composition should be primarily the result of nutrient enrichment.

However, the two basins differ in several aspects (Table 4-1). First, Council Creek marsh is only ~59% as large as Little Creek marsh, and Council Creek basin is only 26% of the area of Little Creek basin (note that the map scales are the same for both panels of Figures 4-2 and 4-3). Second, there are large differences in landuse between the two basins. Landuse in the Council Creek basin is largely composed of agriculture (40.3%) and WWTP property (35.3%) with part of the Talbot County landfill (14.1%) lying within the basin as well (Table 4-1). Forest is a minor land cover (3.3%). In contrast, Little Creek basin is mostly agriculture (47.9%) and forest (37.8%) with several developments (9.9%) and feedlots (1.5%). Finally, Council Creek basin has a relatively low population density (5.5 km^{-2}), while population density in Little Creek basin (81.4 km^{-2}) is 15 times higher due the presence of developments.

Methods

Aboveground plant biomass was collected in the first week of October 2001 in the two marshes. By sampling at the end of the growing season, I attempted to capture maximum aboveground biomass (McCormick and Somes 1982). Within both marshes, four randomly collected replicate 0.25 m^2 quadrats were collected in each of three plant communities. The inner marsh plant communities were dominated by *Peltandra virginica*, with other species such as *Zizania aquatica* and *Rumex verticilatus* being present in higher elevation areas. The middle marsh plant communities were composed of dense stands of *Phragmites australis*, while the outer marsh plant communities contained a diverse mixture of species such as *Leersia oryzoides*, *Spartina cynosuroides*, *Typha* spp, and others. See upper panel of Figure 4-3 for the approximate distribution of these three communities in Council Creek marsh. These estimations were based on the method described by Rice et al. (2000) using 1994 digital orthographic quarter quadrant aerial photographs from the Talbot County, Maryland USDA Natural Resources Conservation Services office as well as personal observations made in Council Creek marsh. Estimation of the community distributions in Little Creek marsh was not possible due to short amount of time spent in Little Creek marsh relative to Council Creek marsh.

Samples were collected by clipping all standing stems at sediment level that were within each 0.25 m^2 quadrat. All dead plant material and detrital matter on the sediment surface within each quadrat was also collected. Biomass from each quadrat was placed in labeled plastic bags in the field; upon returning to the lab, plant material was refrigerated at 5ºC until live, dead, and detrital material were sorted by species within 5 days.

Sample preparation for nutrient analysis required sorting, drying, weighing, and grinding all of the material. Biomass was sorted between live and dead material to the species level using Radford et al. (1968) and White (1989) as taxonomic guides. Plant material was classified as live if there was visible chlorophyll present on any part of the plant and dead if there were no visible green areas. All detritus and unidentifiable plant material was placed into a separate detritus category for each quadrat. Sorted material was washed in tap water to remove any excess sediment and dried in paper bags in an oven at 60ºC. Samples were dried to constant weight, to within 0.1 grams dry weight (gdw) over the course of 14 days. Upon removal from the drying oven, all material was immediately weighed on an OHAUS model 1500D balance to within 0.1 gdw to prevent any bias from plant equilibration with atmospheric humidity. For smaller samples (~0.2-

100 gdw), the entire content of each bag was then ground with a Thomas Wiley Intermediate Mill using a 20-mesh screen. For larger samples (~101-500 gdw), the entire content of each bag was cut into 2-4 cm pieces and ground with a Model 4 Wiley Mill using a 20-mesh screen. A few large samples (>500 gdw) required sub-sampling by choosing three similar sub-samples of approximately 50-100 gdw each that visually appeared to be representative of the whole sample. Each sub-sample was then ground in the same manner as the ~101-500 gdw whole samples. Several species were also split into duplicate sub-samples to determine within quadrat species variability. All ground samples were stored at room temperature in plastic zip bags until they could be analyzed for nutrient content.

All ground plant biomass samples were analyzed for carbon, nitrogen, and phosphorus content. Two instruments were used for CHN analysis. One was an Exeter Analytical CE-440 Elemental Analyzer that used acetanilide ($CH_3CONHC_6H_5$) as the standard; the other was a Leco CHN 2000 Carbon, Nitrogen, and Hydrogen Analyzer that used a Leco alfalfa standard. Samples run on the Leco CHN 2000 Analyzer were pre-weighed into tin capsules and sent to the University of Maryland College Park Soils Lab where they were analyzed. Six samples were analyzed on both instruments in duplicate to test for bias. Within these six cross-referenced samples, the range of absolute differences for %C and %N was 0.4-2.5 and 0.02-0.22, respectively, with mean absolute differences of 1.4%C and 0.1%N. Although these average absolute differences represent ~3-10% error, there was no consistent between-instrument bias in %C and %N measurements made with the two instruments (Figure 4-4 panels A and B). All samples were processed for phosphorus content using a modified version of the procedure used by

Anderson et al (1976). Plant material (5-15mg) was weighed into 70 mg aluminum weigh boats and placed in glass tubes. The glass tubes were ignited in a muffle furnace at 500ºC for 2 hours. The tubes were cooled, 10 ml of 1N HCl were added, and boiled for 15-minutes in a water bath. Samples were then diluted to 40 ml with de-ionized water and 4 ml 1N NaOH were added to adjust the pH. The diluted/adjusted solutions were then analyzed for orthophosphate using the procedure of Strickland and Parsons (1972). Standards and blanks were treated in the same manner as samples. For blanks, empty aluminum weigh boats were used placed in glass tubes, while standards were pipetted from 10mM ATP stock solution. Mean values are reported below for all sub-samples that were split into duplicate where the entire sample was ground. Means are also reported for triplicate sub-samples of the large samples (>500 gdw).

All plant C, N, and P data were adjusted to correct for hygroscopic water that the samples acquired in storage. To test for re-absorption of water, about 0.1 g of several dried plant samples were weighed in duplicate into 70 mg aluminum weigh boats and placed a drying oven at 60 ºC. Samples were re-weighed at approximately 2, 6, 9, and 24 hours to calculate the remaining percent of initial biomass at each time. A hyperbolic decay function was fitted to the resultant time series to determine the minimum required re-drying time for the somewhat hygroscopic, ground samples (Figure 4-5). Since ~8-hours seemed sufficient to bake out the re-absorbed water, all corrections were based on the mean of the 24-hour values (Table 4-2). To account for morphological plant variability in some species that affected the amount of hygroscopic water that was acquired in storage, more than one sample from that species was tested and a separate correction was used for each sample (see *Typha latifolia* in Table 4-2). There were no significant differences between plant species collected in both marshes (Figure 4-5).

Data were analyzed using SAS/STAT Software's MIXED Procedure with a mixed linear model that allows data to exhibit correlation and nonconstant variability (SAS 1996). The least-squares means (LSMEANS) statement with the TUKEY option ($p > 0.05$) was used for the comparison among groups (i.e., Council Creek marsh/Little Creek marsh, inner/middle, etc.) with unbalanced data (Bedford et al 1999), while the Bonferonni correction was applied where multiple comparisons of interaction effects between groups were made (i.e., Council Creek marsh inner/ Little Creek marsh inner, Council Creek marsh middle/ Little Creek marsh middle, etc.). The Bonferonni correction adjusts the α level to reduce the chances of making a type I error in multiple comparisons (i.e., accepting an incorrect hypothesis) by dividing the uncorrected α level by the number of pairwise comparisons (e.g., if the uncorrected α level is 0.05 and there are 5 pairwise comparisons, the corrected α level required is 0.01).

Results

Species Composition

Over 20 species of marsh macrophytes were identified in trips to the two marshes (January 2001 through November 2002). A more comprehensive list of species present in both marshes along with their approximate abundances (Table 4-3) revealed that the initial three-community structure was probably too simplistic to characterize these tidal freshwater marshes. Community structure is more complex with several plant associations that vary temporally and with elevation, as in Pasternack et al. (2000). In fact, in this list of 24 visually determined species, 22 are present in Council Creek marsh, whereas only 21 are present in Little Creek marsh. However, it should be noted that this difference may be the result of the number of times that I was at Council Creek marsh relative Little Creek marsh (>40 at Council Creek marsh versus 2 at Little Creek marsh).

Of the 24 visually observed species, only 15 were collected during sampling. In the Council Creek marsh quadrats, there were relatively fewer species (9) compared to 15 being present in the Little Creek marsh quadrats (Table 4-4, first 2 columns). The data in Table 4-4 show all of the species/categories present in each of the quadrats taken in the three communities of both marshes and the number of replicate quadrats in which the species/category was present.

The data in Figure 4-6 show species richness (no m^{-2}) plotted as a function of aboveground biomass (g m^{-2}). Despite the sampling differences in total number of species observed between the two marshes, a plot of species richness showed no significant differences between marshes ($p > 0.05$), as shown by the scatter in Figure 4-6; i.e., vertical overlap of circles (Council Creek) and triangles (Little Creek). Since there were no significant species richness differences, all quadrats from both marshes were pooled to compare richness versus biomass and fitted with a log normal curve (black line in Figure 4-6). The data exhibit a modal pattern with low species richness at very low and high biomass, while maximum richness (and high variability) occurred at low to intermediate biomass suggesting some optimum conditions that maximizes both biomass and species richness. Conversely, the low biomass and low species richness of the inner marsh communities may be limited by some environmental stressor, while the high biomass and low species richness of the middle marsh communities may be controlled by competitive exclusion by the high density monoculture of *Phragmites australis* (open circles and triangles).

Biomass

Total aboveground dry biomass m^{-2}, nutrient mass m^{-2}, and total quadrat nutrient ratios were compared to determine if there were significant differences between communities within each marsh. In Council Creek marsh, there were no significant differences between any of the three communities for aboveground dry biomass, gC, gN, or gP (top of Table 4-5). The inner marsh community had significantly lower N:P, C:N, and C:P than both the middle and outer marsh communities (top of Table 4-5). The middle marsh community had significantly higher N:P but significantly lower C:N than the outer marsh community (top of Table 4-5). In Little Creek marsh, the inner marsh community had significantly lower aboveground dry biomass, gC, gN, gP, N:P, C:N, and C:P than the middle marsh community and significantly lower N:P, C:N, and C:P than the outer marsh community (top of Table 4-5). Comparisons of the middle and outer communities of Little Creek marsh yielded significantly higher aboveground dry biomass, gC, gN, gP, and N:P in the middle marsh community (top of Table 4-5). Therefore, Council Creek marsh had few significant differences across the three communities, whereas the middle marsh community of Little Creek was largely distinct from the inner and outer communities.

Total aboveground dry biomass m^{-2}, nutrient mass m^{-2}, and total quadrat nutrient ratios were also compared to determine if there were significant differences between marsh communities of the same type. Mean ± se total gdw m^{-2}; total gC m^{-2}, gN m^{-2}, and gP m^{-2}; and total N:P, C:N, and C:P are plotted in Figures 4-7 and 4-8 with the p-value from statistical testing. Aboveground dry biomass and mass of carbon, nitrogen and phosphorus in Council Creek marsh did not significantly ($p > 0.01$) differ from Little Creek marsh in any of the communities (Figure 4-7 panel A, B, C, and D, respectively). Ratios of total N:P (Figure 4-8 panel A) and C:P (Figure 4-8 panel C) in Council Creek marsh were also not significantly ($p > 0.01$) different from Little Creek marsh in any of the communities. However, there was a significantly ($p < 0.01$) lower C:N ratio in the middle community of Council Creek marsh, but not in the inner and outer communities (Figure 4-8 panel B). Therefore, plant elemental composition was relatively uniform within both marshes.

Biomass comparisons within species

Mean ± se aboveground dry biomass and nutrient composition for all species/categories in each community at both marshes are reported in the bottom of Table 4-5. Aboveground dry biomass per square meter, nutrient content, and nutrient ratios for individual species were compared to test for differences between Council Creek marsh and Little Creek marsh. The species tested were *Peltandra virginica* to represent the inner community, *Phragmites australis*, to represent the middle community, and *Leersia oryzoides* to represent the outer community. The species were chosen for two reasons: (1) they were present in all of the replicate quadrats in both marshes, and (2) nutrient content comparisons are expected to be different in samples that have different species compositions (as shown in the bottom of Table 4-5).

Mean ± se species values are plotted in Figures 4-9 and 4-10. There were no significant ($p > 0.01$) differences between Council Creek marsh and Little Creek marsh in any of the three communities for aboveground dry biomass or percent C (Figure 4-9 panels A and B and bottom of Table 4-5). While *Peltandra virginica* (inner communities) and *Leersia oryzoides* (outer communities) did not significantly differ in % nitrogen, *Phragmites australis* (middle communities) percent N in Council Creek marsh was significantly ($p < 0.01$) higher (Figure 4-9 panel C and bottom of Table 4-5). Percent phosphorus for *Peltandra virginica* (inner communities) was also significantly ($p < 0.01$) higher in Council Creek marsh, but no significant differences were observed for the other two communities (Figure 4-9 panel D and bottom of Table 4-5). N:P ratios for all three species did not significantly differ ($p > 0.01$, Figure 4-10 panel A and Table 4-5). However, there was a significantly lower C:N ratio for *Phragmites australis* (middle communities) in Council Creek marsh (Figure 4-9 panel B and bottom of Table 4-5). C:P ratio was not significantly ($p > 0.01$) different for *Peltandra virginica* (inner communities) but was significantly lower in Council Creek marsh for both *Phragmites australis* (middle communities) and *Leersia oryzoides* (outer communities) (Figure 4-10 panel C and bottom of Table 4-5). This analysis indicates that there was significant enrichment in either N or P content or elemental ratios in selected species of Council Creek marsh.

Although there were no other categories in which all four quadrat replications were present in like communities at both marshes, seven additional species/categories comparisons were made. To meet the statistical criteria to be tested, the species/category were required to have at least two replicates present in both Council Creek marsh and Little Creek marsh. In addition to the three species tested above, these seven comparisons are summarized in the bottom of Table 4-5. There was significantly higher percent N in live and dead *Phragmites australis* (Figure 4-9 panel B) and detritus in both the middle and outer marsh communities of Council Creek. *Peltandra virginica* exhibited significantly higher percent P in both the inner (Figure 4-9 panel C) and outer marsh communities of Council Creek. Live and dead *Phragmites australis* (middle marsh communities, see also Figure 4-10 panel B), and dead *Spartina cynosuroides* (outer marsh communities) contained significantly lower C:N in Council Creek marsh. C:P was significantly lower in Council Creek marsh for live *Phragmites australis* in the middle marsh communities and *Leersia oryzoides* in the outer marsh communities. Comparison of C:N in *Polygonum punctatum* in the outer marsh communities was the only test that yielded results that indicated nutrient enrichment in Little Creek. Although only 12 of the 70 (17%) tests performed on these data resulted in significant marsh species/category differences, 11 of them indicate that Council Creek marsh is enriched in N and P relative to Little Creek marsh. However, significant enrichments were relatively small, ranging over 17% to 61%, and averaged ($\pm$ se) $41 \pm 4\%$.

Peltandra virginica samples were used to compare biomass and nutrient composition across communities within Council Creek marsh. This species was chosen as it was the only species that was present in all replicates of each Council Creek marsh community (Table 4-4). Statistical tests of all Council Creek biomass and compositional ratios revealed that only *Peltandra virginica* biomass and C:N ratio were significantly different. Biomass was significantly higher ($p < 0.01$) for the inner community than both the middle and outer communities. Although this difference was expected since *Peltandra virginica* was the dominant species at the inner marsh communities with other species dominating the middle and outer communities. In addition, *Peltandra virginica* C:N ratio in the middle community was significantly ($p < 0.05$) lower than that of the outer community. In the comparison of the middle and inner communities, C:N ratios were not significant at the $\alpha = 0.05$ level, although the difference was significant at the $\alpha = 0.10$ level.

Discussion

Several differences in plant nutrient composition were determined from a single end of the growing season sampling of aboveground macrophytic biomass in Council Creek and Little Creek marshes. There were, however, no significant differences in species richness between Council or Little Creek marsh and, species richness versus aboveground biomass data for both marshes exhibited a modal pattern with maximum species richness at intermediate community biomass (Figure 4-6, see below).

In both marshes, there are inter-community biomass differences (top of Table 4-5). Within-marsh total community biomass comparisons revealed that the middle communities of both marshes generally immobilized more nutrients m^{-2} than both the inner and outer marsh communities, probably due to the growth of the high biomass *Phragmites australis* (Figure 4-7, 4-8, and top of Table 4-5). Although the reasons for the colonization of only certain areas by *Phragmites australis* are unclear here, a potential suggestion could be differences in sediment properties (see Chapter 5). However, expansion of *Phragmites australis* into the lateral fringes of the inner marsh community of Council Creek was visually noted over the study period and may be continuing.

Total community mass and elemental composition comparisons by species yielded evidence for several significant but small N and P enrichments of macrophytes in Council Creek marsh. Between marsh total community biomass comparisons revealed that only C:N ratio at the middle community of Little Creek marsh was significantly higher than that of Council Creek marsh, but there were no significant differences between the inner and outer communities of the two marshes (Figure 4-7 and 4-8). Comparisons of elemental composition of individual species yielded several significant differences. Percent P of *Peltandra virginica* was significantly higher in Council Creek marsh than in Little Creek marsh at both the inner and outer communities, indicating P enrichment in the inner community of Council Creek (Figure 4-9 and bottom of Table 4-5). Additional evidence for P enrichment of the inner marsh community of Council Creek was clear from the significantly higher % P at the inner marsh over the outer marsh communities of Council Creek (bottom of Table 4-5). Further evidence for significant but small N and P enrichment of Council Creek marsh from comparisons of both live and dead *Phragmites australis* (middle marsh communities), detrital material from the middle and outer marsh communities of both marshes, and *Leersia oryzoides* and *Spartina cynosuroides* (outer marsh communities (Figures 4-9, 4-10, and Table 4-5). All of these species-specific elemental comparisons indicate that Council Creek marsh is enriched in N and P relative to Little Creek marsh. Although the reason for significantly higher C:N in *Polygonum punctatum* of Council Creek was unclear, it should be noted that only one of the two quadrats in which this species was present at the outer marsh community of Council Creek was significantly higher than the other five quadrat values used for this statistical comparison (2 quadrats in Council Creek marsh, 4 in Little Creek marsh, 6 total quadrats). As there was no evidence for nutrient limitation in either marsh (i.e., no significant differences in biomass), it seemed that luxury consumption of N and P by macrophytes in Council Creek marsh was occurring, due to the above evidence for N and P enrichment.

The hypothesis tested in this chapter was that high N and P loading via wastewater discharge in Council Creek would generally augment aboveground growth of macrophytic biomass, increase macrophytic elemental nutrient composition, and decrease species abundance relative to Little Creek. To test this hypothesis, the data obtained from the replicate quadrats of aboveground macrophytic biomass in each of three communities at both marshes were compared statistically.

The hypothesis that there was evidence for a wastewater signal found in marsh macrophytes was partially supported by these data. Although differences in species richness, aboveground dry biomass, and percent C between the two marshes were not significant, 13 of the 91 comparisons (14.3%) of mass and species-specific elemental composition tested as significantly different. Of these 13 significant results, all of them showed evidence that Council Creek marsh is enriched in N and P relative to Little Creek marsh (Figures 4-6 through 4-9 and bottom of Table 4-5). In addition, there was evidence that *Peltandra virginica* in the inner community was P-enriched relative to the outer community within Council Creek marsh. Thus, these data partially support the hypothesis, although the differences were relatively small, averaging (± se) 41 ± 4%.

Moderate impacts to Council Creek marsh relative to Little Creek marsh are evident from the macrophyte data collected in both marshes (Table 4-5). However, it is important to note that these data must be interpreted with caution for two reasons. First, tidal freshwater marshes are temporally diverse ecosystems. Some species such as *Peltandra virginica* can have high turnover rates and peak biomasses that occur at different times during the growing season. Therefore, a single sampling event cannot capture seasonal species succession and interannual variability. Second, the high spatial diversity in these marshes (compare Tables 4-3 and 4-4) require a more comprehensive sampling strategy than I was able to achieve with the available resources. The initial division of both marshes into three general plant communities proved to be too simplistic a perception of tidal freshwater marsh plant associations. Nevertheless, conclusions and implications can be drawn from these data.

This study of the impacts of nutrient enrichment on macrophytic vegetation in a natural oligohaline marsh has several implications. The modal pattern of aboveground biomass versus species richness (Figure 4-6) suggests some optimum growing conditions that maximize both biomass and species richness. Low biomass and low species richness at the inner communities seems to indicate the presence of some environmental stressor common to both communities. The lower elevation inner communities are likely to be low-biomass and species-poor communities because of more regular flooding. Also contributing to low-biomass may be the higher reported turnover rates of species such as *Peltandra virginica* (Neubauer et al. 2000). In the high-biomass, low-species richness found at the middle marsh community, *Phragmites australis* dominated the vegetation. This productive and invasive species produces such dense stands that it can competitively exclude other species. The pattern of biomass versus species richness seems to be in agreement with the abiotic and stress competitive exclusion discussed in Gough et al. (1994) and Mittelbach et al. (2001). However, biomass alone is not always a good predictor of species richness (Gough et al. 1994, Figure 4-6).

There was no indication of augmented aboveground macrophytic biomass or carbon storage in Council Creek marsh. However, there was evidence for elevated N and P concentrations in aboveground plant biomass as has also been reported in other studies of marshes exposed to wastewater. In a two-year study on freshwater marsh nutrient enrichment, Bayley et al. (1985) found that nutrient enrichment did increase aboveground biomass and P concentrations in one of the growing seasons, while N concentrations were significantly higher in both years. Whigham et al. (1980) found that aboveground plant N and P concentrations increased due to wastewater application to a New Jersey tidal freshwater marsh, although plant biomass did not, suggesting that their control sites may not have been nutrient limited. In the case of Little Creek marsh, it too may not be a nutrient-limited marsh since its plant biomass was not significantly different from that of Council Creek marsh. Although Little Creek marsh does not have a direct wastewater source, there may be significant N and P contributions due to the higher human population density, the presence of concentrated animal feedlots, and the greater amount of agriculture compared to Council Creek basin (Table 4-1).

Based on average community biomass and total areal coverage for each community in Council Creek, the total mass of N and P that is seasonally sequestered in aboveground plant biomass can be estimated (Table 4-6). Average biomass of N and P for each community was calculated in columns 4 and 8 of Table 4-6, respectively. These averages were all well below the range of expected values calculated from Mitsch and Gosselink (2000) and Bedford et al. (1999) except for N in the middle marsh community, which was just within the range (Table 4-6 and page 4). Grouped according to this study's community structure, Table 4-7 summarizes several studies, which address marsh macrophyte production. From this table, average community turnover (P:B) numbers were calculated. By applying these turnover estimates to the average sampled biomass of N and P for each community, average N m^{-2} became 14.4, 51.5, and 21.9 and average P

m^{-2} became 2.4, 3.8, and 2.3 for the inner, middle and outer communities, respectively. These values are all within or more reasonably close to the expected values.

Estimates of the areal coverage for each community were made from personal observations coupled with aerial photography of Council Creek marsh (Figure 4-3, top panel). Total macrophyte N and P immobilization in each community were calculated from area (second column of Table 4-6) and the average N and P m^{-2} (columns 4 and 8 of Table 4-6, respectively). The community estimates of P:B in Table 4-7 were also utilized in Table 4-6 to calculate total macrophyte N and P immobilization to include an estimation for plant turnover. A range of percent immobilization of wastewater N and P were then calculated that included both total N and P interception and total N and P interception using the average turnover in Table 4-7. These data suggest that marsh macrophytes immobilize the equivalent of 12-18% of the wastewater N and 4.7-6.9% of the wastewater P in aboveground biomass on an annual basis, primarily in the large outer marsh community (top of Table 4-6). Identical calculations were made in the bottom of Table 4-6, but the numbers were adjusted to calculate the equivalent macrophytic nutrient uptake during only a 5-month growing season (May-September). Although N and P plant immobilization is inflated to 30-44% of wastewater N and 11-17% of wastewater P for this period, use of these numbers should more accurately reflect marsh nutrient interception during the growing season. It should, however, be noted that there are potentially large errors involved these calculations. Sediment nutrient recycling almost certainly supplies some macrophytic nutrients, while the exclusion of belowground plant biomass certainly caused an underestimation of the vegetative contribution to nutrient immobilization. Marsh plants may also indirectly remove N and P from water because decomposing plant material has been shown to increase in nutrient content, probably due to microbial uptake during decomposition (Odum and Heywood 1978). However, unless these nutrients are buried in sediments, tidal flushing is likely to provide a mechanism for the export of dissolved and particulate nutrients to the downstream estuary (Nixon 1980 and Childers et al. 2000).

In summary, wastewater discharged in to Council Creek marsh has a moderate impact on marsh vegetation. There were no significant differences in species richness between Council Creek marsh and Little Creek marsh, and aboveground plant biomass and carbon storage in Council Creek marsh were not significantly different. However, in 11 of 70 tests (16%), macrophytic vegetation in Council Creek was N and P enriched (Figures 4-6 through 4-9 and Table 4-5). During the growing season (May – Sept), marsh plants are potentially capable of intercepting ~30-44% of wastewater N and 11-17% of wastewater P.

References

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Table 4-1: Descriptions of the study site basins. Areas have units of km^2, and population densities have units of number people km^{-2}.

marsh	marsh area	basin area	% landuse							1990 pop
			forest	agriculture	wetland	developed	feedlot	landfill	WWTP	
Council Creek	0.16	2.20	3.3	40.3	7.0	--	--	14.1	35.3	5.5
Little Creek	0.27	8.57	37.8	47.9	3.0	9.9	1.5	--	--	81.4

Table 4-2: Dry weight corrections for all plant species due to re-absorption of water after initial drying and grinding. See Figure 4-5 for examples of how corrections were obtained.

Category	% of initial mass
Echinochloa walteri	94.44%
Hibiscus moscheutos	93.50%
Hibiscus moscheutos (dead)	95.25%
Leersia oryzoides	95.12%
Mikana scandens	95.76%
Peltandra virginica	93.25%
Phragamites australis	95.71%
Phragmites australis (dead)	94.73%
Polygonum punctatum	93.08%
Pontederia cordata	96.81%
Rumex verticillatus	94.85%
Scirpus fluviatilis	96.17%
Scirpus fluviatilis (dead)	95.19%
Spartina cynosuroides	95.20%
Spartina cynosuroides (dead)	94.17%
Typha angustifolia	94.75%
Typha angustifolia (dead)	94.41%
Typha latifolia (seed spike)	90.12%
Typha latifolia	95.86%
Typha latifolia (senescing)	93.15%
Typha latifolia (dead)	94.82%
unidentified grass	93.75%
Zizania aquatica (dead)	95.44%
detritus and unidentifiable material	95.03%

Table 4-3: List of all species observed and their approximate abundances in Council Creek marsh and Little Creek marsh. "NO", "RARE", "COM", and "AB" indicate not observed, rare, common, and abundant, respectively

Species/category	Council Creek	Little Creek
Acer rubrum	RARE	COM
Amaranthus cannabinus	COM	COM
Hibiscus moscheutos	COM	COM
Impatiens capensis	RARE	RARE
Juncus effusus	RARE	RARE
Leersia oryzoides	COM	COM
Lythrum salicaria	COM	NO
Mikana scandens	COM	COM
Nuphar advena	NO	RARE
Peltandra virginica	AB	COM
Phragmites australis	AB	AB
Polygonum arifolium	COM	COM
Polygonum punctatum	COM	COM
Pontederia cordata	RARE	COM
Rumex verticillatus	COM	COM
Panicum hemitomum	COM	COM
Pluchea purpurascens	RARE	NO
Scirpus fluviatilis	COM	COM
Spartina cynosuroides	COM	COM
Typha angustifolia	COM	COM
Typha latifolia	COM	COM
unidentified grass	NO	RARE
Viburnam dentatum	RARE	NO
Zizania aquatica	RARE	RARE

Table 4-4: Mean species/category values for biomass, nutrient content, and nutrient ratios for all species/categories present in each community in both marshes. Reported values (mean ± se) are averages of quadrat replications. "(d)" indicates dead plant material, "unid" means unidentifiable, and "no" indicates the number of quadrats in the community in which the species was present.

Marsh/community	species/category	no	gdw m^{-2}	%C	%N	%P	N:P	C:N	C:P
Council/inner	*Peltandra virginica*	4	360 ± 17	39 ± 1.0	2.7 ± 0.1	0.5 ± 0.0	6.1 ± 0.41	15 ± 0.5	89 ± 4.8
Council/middle	*Leersia oryzoides*	1	70	39	1.4	0.20	7.0	27	190
Council/middle	*Peltandra virginica*	4	5.2 ± 1.8	38 ± 1.2	3.2 ± 0.5	0.5 ± 0.1	6.2 ± 0.5	12 ± 1.3	75 ± 7.1
Council/middle	*Phragmites australis*	4	2000 ± 750	45 ± 0.2	1.4 ± 0.0	0.1 ± 0.0	12 ± 0.9	32 ± 1.0	410 ± 24
Council/middle	*Phragmites australis* (d)	4	820 ± 510	45 ± 0.5	1.2 ± 0.2	0.1 ± 0.0	19 ± 2.2	40 ± 6.2	770 ± 140
Council/middle	*Scirpus fluviatilis* (d)	1	4.8	43	1.2	0.1	12	36	430
Council/middle	detritus and unid material	2	100 ± 12	37 ± 2.1	2.1 ± 0.1	0.2 ± 0.0	12 ± 1.1	18 ± 2.1	210 ± 4.4
Council/outer	*Leersia oryzoides*	4	66 ± 34	41 ± 0.5	1.3 ± 0.1	0.2 ± 0.0	7.5 ± 0.4	34 ± 3.4	250 ± 19
Council/outer	*Peltandra virginica*	4	42 ± 15	40 ± 1.0	2.6 ± 0.1	0.5 ± 0.0	5.1 ± 0.2	15 ± 0.2	79 ± 1.5
Council/outer	*Polygonum punctatum*	2	33 ± 13	46 ± 3.1	1.1 ± 0.2	0.2 ± 0.0	6.0 ± 0.5	43 ± 10	250 ± 38
Council/outer	*Scirpus fluviatilis*	2	43 ± 8.2	43 ± 0.8	0.6 ± 0.0	0.0 ± 0.0	14 ± 2.6	83 ± 1.8	1100 ± 230
Council/outer	*Scirpus fluviatilis* (d)	2	67 ± 18	48 ± 1.1	0.6 ± 0.1	0.0 ± 0.0	15 ± 2.7	76 ± 4.5	1200 ± 140
Council/outer	*Spartina cynosuroides*	4	730 ± 220	47 ± 1.1	0.7 ± 0.0	0.1 ± 0.0	11 ± 0.1	72 ± 3.7	820 ± 48
Council/outer	*Spartina cynosuroides* (d)	4	480 ± 78	46 ± 0.5	0.8 ± 0.1	0.1 ± 0.0	13 ± 1.1	58 ± 4.7	770 ± 130
Council/outer	*Typha angustifolia*	3	200 ± 41	47 ± 0.2	0.8 ± 0.1	0.1 ± 0.0	9.2 ± 0.2	59 ± 5.3	550 ± 60
Council/outer	*Typha angustifolia* (d)	1	230	47	0.6	0.1	9.0	77	690
Council/outer	*Typha latifolia*	1	160	46	1.4	0.2	7.7	32	250
Council/outer	*Typha latifolia* (d)	1	130	45	0.6	0.0	13	77	1000
Council/outer	detritus and unid material	3	120 ± 38	36 ± 2.0	1.6 ± 0.0	0.2 ± 0.0	8.9 ± 0.7	22 ± 1.2	200 ± 26
Little/inner	*Leersia oryzoides*	1	1.2	43	1.7	0.2	7.0	26	180
Little/inner	*Peltandra virginica*	4	350 ± 92	38 ± 0.9	2.8 ± 0.1	0.3 ± 0.0	8.1 ± 0.2	14 ± 0.2	110 ± 2.5
Little/inner	*Pontederia cordata*	1	12	37	2.6	0.4	6.3	14	90
Little/inner	*Rumex verticillatus*	1	160	45	1.0	0.2	6.3	45	280
Little/inner	*Scirpus fluviatilis* (d)	1	43	37	1.0	0.1	8.9	39	350
Little/inner	detritus and unid material	4	250 ± 68	28 ± 4.3	1.6 ± 0.1	0.2 ± 0.0	8.6 ± 0.4	17 ± 3.2	150 ± 30

Table 4-4: cont

Marsh/community	species/category	no	gdw m^{-2}	%C	%N	%P	N:P	C:N	C:P
Little/middle	*Hibiscus moscheutos*	1	220	47	0.7	0.1	5.2	63	330
Little/middle	*Hibiscus moscheutos* (d)	1	110	43	0.6	0.1	9.8	69	680
Little/middle	*Leersia oryzoides*	3	10 ± 3.8	40 ± 1.2	1.3 ± 0.1	0.1 ± 0.0	9.7 ± 1.4	33 ± 4.5	300 ± 6.3
Little/middle	*Mikana scandens*	1	50	45	1.3	0.2	8.7	35	300
Little/middle	*Phragmites australis*	4	2600 ± 360	46 ± 0.2	1.0 ± 0.0	0.1 ± 0.0	10 ± 0.3	47 ± 1.8	480 ± 7.0
Little/middle	*Phragmites australis* (d)	4	940 ± 180	47 ± 0.6	0.7 ± 0.0	0.0 ± 0.0	19 ± 1.1	64 ± 1.8	1200 ± 78
Little/middle	unidentified grass	1	6.4	44	2.0	0.4	5.9	22	130
Little/middle	detritus and unid material	4	75 ± 16	38 ± 2.2	1.5 ± 0.1	0.1 ± 0.0	11 ±1.0	25 ± 0.4	280 ± 21
Little/outer	*Echinochloa walteri*	1	270	43	1.0	0.2	6.3	43	270
Little/outer	*Hibiscus moscheutos*	2	130 ± 120	43 ± 3.3	0.7 ± 0.0	0.1 ± 0.0	10 ± 2.8	59 ± 6.7	570 ± 100
Little/outer	*Leersia oryzoides*	4	46 ± 20	41 ± 0.5	1.3 ± 0.1	0.1 ± 0.0	10 ± 1.3	33 ± 3.1	320 ± 24
Little/outer	*Peltandra virginica*	2	23 ± 21	40 ± 2.0	2.4 ± 0.4	0.3 ± 0.0	7.5 ± 1.2	17 ± 1.9	130 ± 5.4
Little/outer	*Polygonum punctatum*	4	89 ± 42	46 ± 0.7	1.3 ± 0.1	0.1 ± 0.0	11 ± 1.3	36 ± 2.6	400 ± 69
Little/outer	*Rumex verticillatus*	1	98	48	0.8	0.1	10	58	600
Little/outer	*Scirpus fluviatilis* (dead)	1	90	47	0.6	0.1	12	80	990
Little/outer	*Spartina cynosuroides*	2	670 ± 30	47 ± 0.8	0.6 ± 0.1	0.1 ± 0.0	9.4 ± 0.8	77 ± 7.5	720 ± 6.3
Little/outer	*Spartina cynosuroides* (d)	2	400 ± 170	48 ± 0.2	0.6 ± 0.0	0.0 ± 0.0	14 ± 1.6	83 ± 1.8	1170 ± 160
Little/outer	*Typha angustifolia* (d)	1	54	44	0.9	0.1	8.9	49	440
Little/outer	*Typha latifolia*	2	590 ± 290	46 ± 0.6	1.0 ± 0.1	0.1 ± 0.0	7.9 ± 0.7	45 ± 6.0	350 ± 15
Little/outer	*Typha latifolia* (d)	1	113	41	1.0	0.1	8.7	43	370
Little/outer	unidentified grass	2	1.2 ± 0.0	37 ± 0.9	1.9 ± 0.3	0.3 ± 0.0	6.5 ± 1.0	21 ± 4.2	130 ± 5.5
Little/outer	*Zizania aquatica* (dead)	2	32 ± 27	42 ± 0.5	1.1 ± 0.1	0.1 ± 0.0	8.5 ± 0.4	40 ± 4.8	340 ± 55
Little/outer	detritus and unid material	4	230 ± 53	34 ± 4.0	1.1 ± 0.1	0.2 ± 0.0	7.9 ± 0.8	30 ± 5.0	250 ± 57

Table 4-5: Statistical comparisons of elemental composition. Comparisons in the top rows are total quadrat comparisons, while comparisons in the bottom rows are species/category comparisons. ns = not significant, C = Council Creek, L = Little Creek, I = inner community, M = middle community, and O = outer community.

Marsh(es), community(ies)	species/category	gdw m^{-2}	gC	gN	gP	N:P	C:N	C:P
Council, inner vs middle	total quadrat	ns	ns	ns	ns	I<M	I<M	I<M
Council, middle vs outer	total quadrat	ns	ns	ns	ns	M>O	M<O	ns
Council, inner vs outer	total quadrat	ns	ns	ns	ns	I<O	I<O	I<O
Little, inner vs middle	total quadrat	I<M	I<M	I<M	I<M	I<M	I<M	I<M
Little, middle vs outer	total quadrat	M>O	M>O	M>O	M>O	M>O	ns	ns
Little, inner vs outer	total quadrat	ns	ns	ns	ns	I<O	I<O	I<O
Council vs Little, inner	total quadrat	ns	ns	ns	ns	ns	ns	ns
Council vs Little, middle	total quadrat	ns	ns	ns	ns	ns	C<L	ns
Council vs Little, outer	total quadrat	ns	ns	ns	ns	ns	ns	ns
Marsh(es), community(ies)	species/category	gdw m^{-2}	%C	%N	%P	N:P	C:N	C:P
Council vs Little, inner	*Peltandra virginica*	ns	ns	ns	C>L	ns	ns	ns
Council vs Little, middle	*Phragmites australis*	ns	ns	C>L	ns	ns	C<L	C<L
Council vs Little, middle	*Phragmites australis* (dead)	ns	ns	C>L	ns	ns	C<L	ns
Council vs Little, middle	detritus and unid material	ns	ns	C>L	ns	ns	ns	ns
Council vs Little, outer	*Leersia oryzoides*	ns	ns	ns	ns	ns	ns	C<L
Council vs Little, outer	*Peltandra virginica*	ns	ns	ns	C>L	ns	ns	ns
Council vs Little, outer	*Polygonum punctatum*	ns	ns	ns	ns	ns	C>L	ns
Council vs Little, outer	*Spartina cynosuroides*	ns	ns	ns	ns	ns	ns	ns
Council vs Little, outer	*Spartina cynosuroides* (dead)	ns	ns	ns	ns	ns	C<L	ns
Council vs Little, outer	detritus and unid material	ns	ns	C>L	ns	ns	ns	ns
Council, inner vs middle	*Peltandra virginica*	I>M	ns	ns	ns	ns	ns	ns
Council, middle vs outer	*Peltandra virginica*	ns	ns	ns	ns	ns	ns	ns
Council, inner vs outer	*Peltandra virginica*	I>O	ns	ns	ns	ns	ns	I>O

Table 4-6: Summary of potential interception of wastewater nutrients by macrophytes in Council Creek. Annual average WWTP discharge is $2.2*10^7$ g N y^{-1} and $5.7*10^6$ g P y^{-1} and discharge during the growing season is $8.9*10^6$ g N y^{-1} and $2.4*10^6$ g P y^{-1} P:B is the averaged production to biomass ratio from Table 4-7, avg N and P is the average N and P content per m^{-2}, total N and P is the total community mass of N and P based on only the sample numbers, total N and P w/ turnover is the total community mass of N and P including the P:B ratio, % of wastewater N and P is the percent of wastewater N and P discharged that is intercepted by marsh macrophytes. Calculations in the top lines are based on uptake on an annual basis, while the calculations in the bottom lines assume a 5-month growing season (May-September).

comm.	area, m^{-2}	P:B	avg N, g m^{-2}	total N, g	total N w/ turnover, g	% of wastewater N	avg P, g m^{-2}	total P, g	total P w/ turnover, g	% of wastewater P
Basis: annual (Jan. – Dec.)										
inner	$2.60*10^4$	1.5	9.6	$2.5*10^5$	$3.8*10^5$	1.1-1.7	1.6	$4.2*10^4$	$6.3*10^4$	0.7-1.1
middle	$2.53*10^4$	1.4	36.8	$9.3*10^5$	$1.3*10^6$	4.3-6.0	2.7	$6.8*10^4$	$9.5*10^4$	1.2-1.7
outer	$1.04*10^5$	1.5	14.6	$1.5*10^6$	$2.3*10^6$	7.0-10.6	1.5	$1.6*10^5$	$2.3*10^5$	2.7-4.1
total	$\mathbf{1.55*10^5}$	**1.5**	**20.3**	$\mathbf{2.7*10^6}$	$\mathbf{4.0*10^6}$	**12.5-18.4**	**1.9**	$\mathbf{2.7*10^5}$	$\mathbf{3.9*10^5}$	**4.7-6.9**
Basis: annual (May. – Sep.)										
inner	$2.60*10^4$	1.5	9.6	$2.5*10^5$	$3.8*10^5$	2.8-4.2	1.6	$4.2*10^4$	$6.3*10^4$	1.8-2.7
middle	$2.53*10^4$	1.4	36.8	$9.3*10^5$	$1.3*10^6$	10.4-14.5	2.7	$6.8*10^4$	$9.5*10^4$	2.9-4.0
outer	$1.04*10^5$	1.5	14.6	$1.5*10^6$	$2.3*10^6$	16.9-25.4	1.5	$1.6*10^5$	$2.3*10^5$	6.6-9.9
total	$\mathbf{1.55*10^5}$	**1.5**	**20.3**	$\mathbf{2.7*10^6}$	$\mathbf{4.0*10^6}$	**30.0-44.0**	**1.9**	$\mathbf{2.7*10^5}$	$\mathbf{3.9*10^5}$	**11.2-16.6**

Table 4-7: Literature summary of production:biomass for various freshwater/brackish marsh plant species. Sampling methods included: a-multiple harvest, b-single harvest at peak standing crop, c-paired plot harvest, and d-carbon gas flux.

species	peak standing crop (B), g m^{-2}	annual production (P), g $m^{-2}y^{-1}$	P:B	sample method	source
Peltandra/pontederia	686	888	1.3	a&b	Whigham et al 1978
Peltandra/pontederia	423	423	1.0	a	Doumele 1981
Peltandra/pontederia	677	1126	1.7	?	McCormick and Somes 1982
Peltandra/pontederia	--	--	2.5	?	Neubauer et al 2000
Peltandra/pontederia	--	--	1.0	?	Neubauer et al 2000
Peltandra/pontederia	671	888	1.3	?	Mitsch and Gosselink 2000
Peltandra/pontederia	--	--	2.0	d	Neubauer et al 2000
upper mean	**614**	**831**	**1.5**		
Phragmites	990	2318	2.3	a	Hopkinson et al. 1978
Phragmites	1380	1592	1.2	c	Linthurst and Reimold 1978
Phragmites	1850	1872	1.0	a&b	Whigham et al 1978
Phragmites	1451	1678	1.2	?	McCormick and Somes 1982
Phragmites	1493	2066		?	McCormick and Somes 1982
Phragmites	600-3500	1000-3500	1.6-1.7	?	Kadlec and Knight 1996
Phragmites	1850	1872	1.0	?	Mitsch and Gosselink 2000
Phragmites (natural)	2900	3200	1.1	a	Meuleman et al 2000
Phragmites (waste)	5000	7000	1.4	a	Meuleman et al 2000
middle mean	**2107**	**2650**	**1.4**		

Table 4-7 cont

species	peak standing crop (B), g m^{-2}	annual production (P), g $m^{-2}y^{-1}$	P:B	sample method	source
Scirpus sp.	600	713	1.2	?	Kadlec and Knight 1996
Spartina cyn	808	1355	1.6	a	Hopkinson et al. 1978
Spartina cyn	2176	5996	2.8	c	Linthurst and Reimold 1978
Spartina cyn	1113	1053	0.95	a&b	Whigham et al 1978
Spartina cyn	1207	1572	1.3	?	McCormick and Somes 1982
Spartina cyn	560	563	1.0	?	McCormick and Somes 1982
Spartina cyn	515	825	1.6	?	McCormick and Somes 1982
Spartina cyn	762	872	1.1	?	McCormick and Somes 1982
Spartina cyn	1242	2092	1.2	?	McCormick and Somes 1982
Typha sp.	1215	1420	1.2	a&b	Whigham et al 1978
Typha sp.	966	1868	1.9	?	McCormick and Somes 1982
Typha sp.	1199	1534	1.3	?	McCormick and Somes 1982
Typha sp.	430-2250	800-6100	1.9-2.7	?	Kadlec and Knight 1996
outer mean	**1054**	**1793**	**1.5**		

FIGURE LEGENDS

Figure 4-1: Upper panel: Location of the Choptank Basin relative to Chesapeake Bay. Main panel: Location of Easton, Council Creek, and Little Creek (see Figure 4-2)

Figure 4-2: Marsh basins: Top panel: composite of aerial photographs of Council Creek basin. Bottom panel: composite of aerial photographs of Little Creek marsh. Yellow polygons indicate basin boundaries, and green polygons indicate marsh boundaries. Black horizontal line and red vertical line are the seams between aerial photographs in the composite.

Figure 4-3: Marsh study sites: Top panel: aerial photograph of Council Creek marsh, bottom panel: aerial photograph of Little Creek marsh. Green polygons indicate marsh boundaries, and colored dots indicate locations of plant sampling quadrats for each community.

Figure 4-4: Comparison of %C and % N between University of Maryland College Park and Horn Point Laboratory CHN analyzers. Panel A: comparison of % C, Panel B: comparison of % N. Circles represent samples run on both instruments, solid line represents the regression of the points, dashed line is the 1:1 line. The slope of the regression was not significantly different from the 1:1 line in both panels.

Figure 4-5: Examples of dry weight corrections for hygroscopic, ground plant materials: Plot of two *Peltandra virginica* samples and two *Leersia oryzoides* samples dried for 24 hours at 60ºC. Samples were weighed at t = 0 and approximately t = 2, 6, 8, and 24 hours to determine adequate re-drying time of previously dried and ground plant samples.

Figure 4-6: Species richness/biomass: Species richness (no. m^{-2}) plotted as a function of total aboveground dry biomass (g m^{-2}) for all quadrats in both marshes. Circles represent Council Creek marsh, and triangles represent Little Creek marsh. Red indicates inner community, white indicates the middle community, and black indicates the outer community.

Figure 4-7: Total plant community comparisons between Council Creek and Little Creek. Means ± se (g m^{-2}) of replicate quadrats are plotted for each community in both marshes. Panel A: total quadrat aboveground dry biomass, Panel B: total carbon mass, Panel C: total nitrogen mass, and Panel D: total phosphorus mass. The p-values above each pair of bars are the result of statistical comparisons. There were no significant pair-wise differences.

Figure 4-8: Comparisons of total plant community nutrient ratios. Means ± se nutrient ratios of replicate quadrats are plotted for each community in both marshes. Panel A: N:P ratio, Panel B: C:N ratio, and Panel C: C:P ratio. The p-values above each pair of bars are the result of statistical analyses. The bold p-value for the middle marsh C:N indicates a significant difference; all others are not significant.

Figure 4-9: Comparisons of elemental composition by species. Means ± se of replicate quadrats are plotted for a selected species to represent the individual communities. *Peltandra virginica*, *Phragmites australis* and *Leersia oryzoides* represent the inner, middle, and outer communities, respectively. Panel A: aboveground dry biomass (g m^{-2}), Panel B: percent carbon, Panel C: percent nitrogen, and Panel D: percent phosphorus. The p-values above each pair of bars are the result of pair-wise statistical comparisons. The bold p-values for %P for *Peltandra virginica* and %N for *Phragmites australis* indicate a significant difference; all others are not significant.

Figure 4-10: Comparisons of elemental composition by ratios. Mean ± se of elemental nutrient ratios of replicate quadrats are plotted for a selected species from each community. *Peltandra virginica*, *Phragmites australis* and *Leersia oryzoides* were sampled within the inner, middle, and outer communities, respectively. Panel A: N:P ratio, Panel B: C:N ratio, and Panel C: C:P ratio. The p-values above each pair of bars are the result of pair-wise statistical comparisons. Bold p-values indicate significant differences.

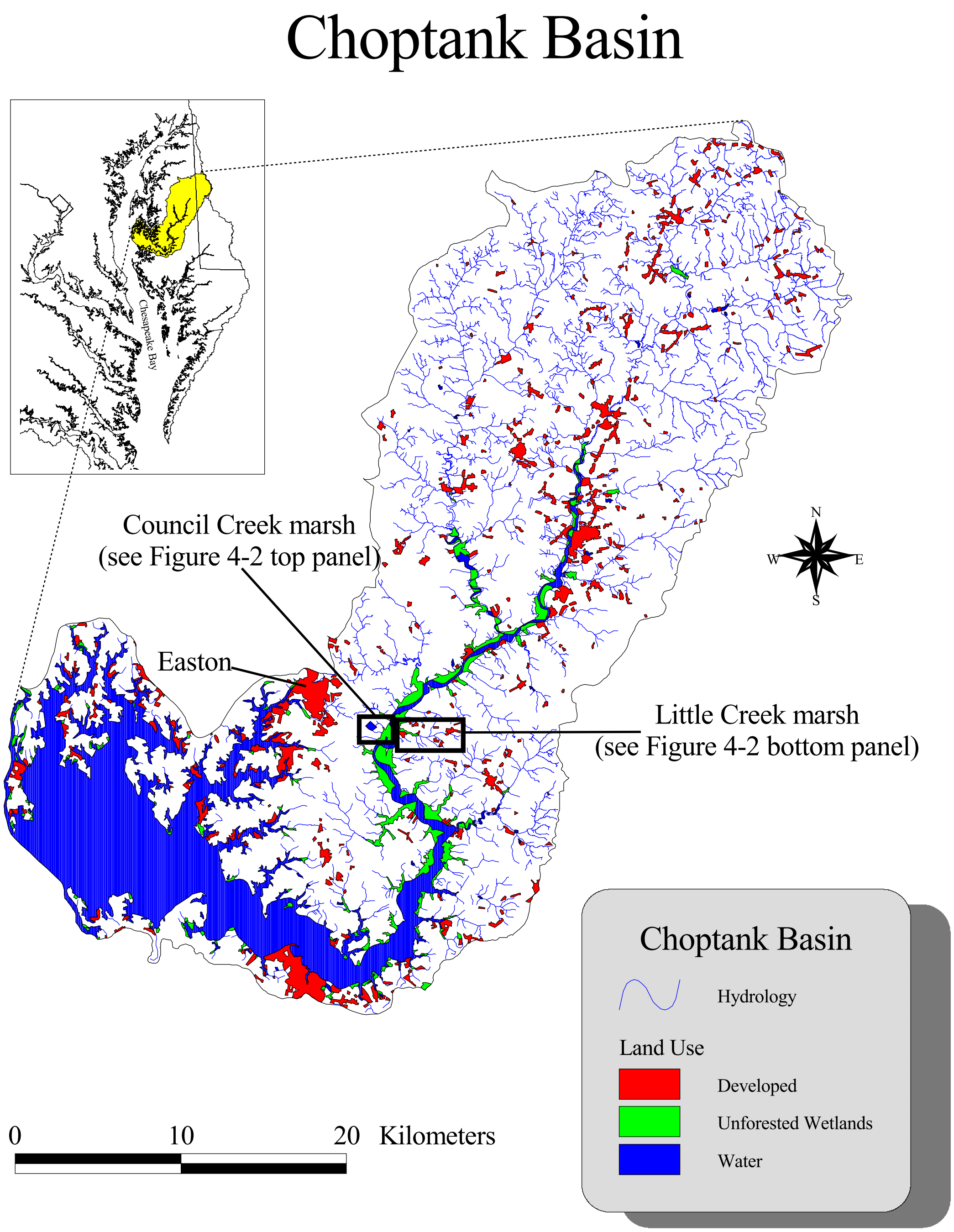

Choptank Basin
Chesapeake Bay
Council Creek marsh
(see Figure 4-2 top panel)
Easton
Little Creek marsh
(see Figure 4-2 bottom panel)
N
W
E
S
0
10
20
Kilometers
Choptank Basin
Hydrology
Land Use
Developed
Unforested Wetlands
Water

Figure 4-1

Marsh Basins

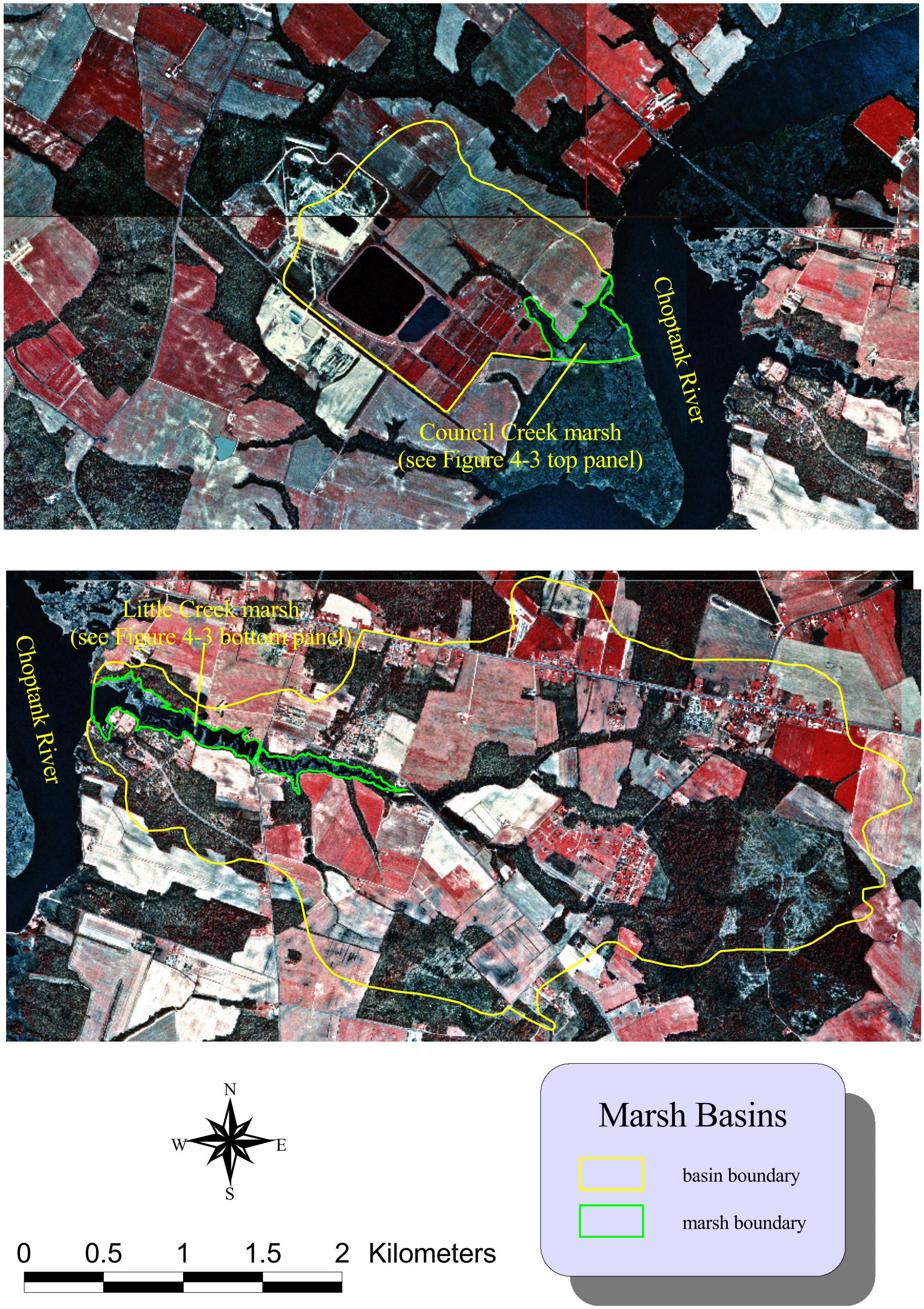


Figure 4-2

Marsh Study Sites

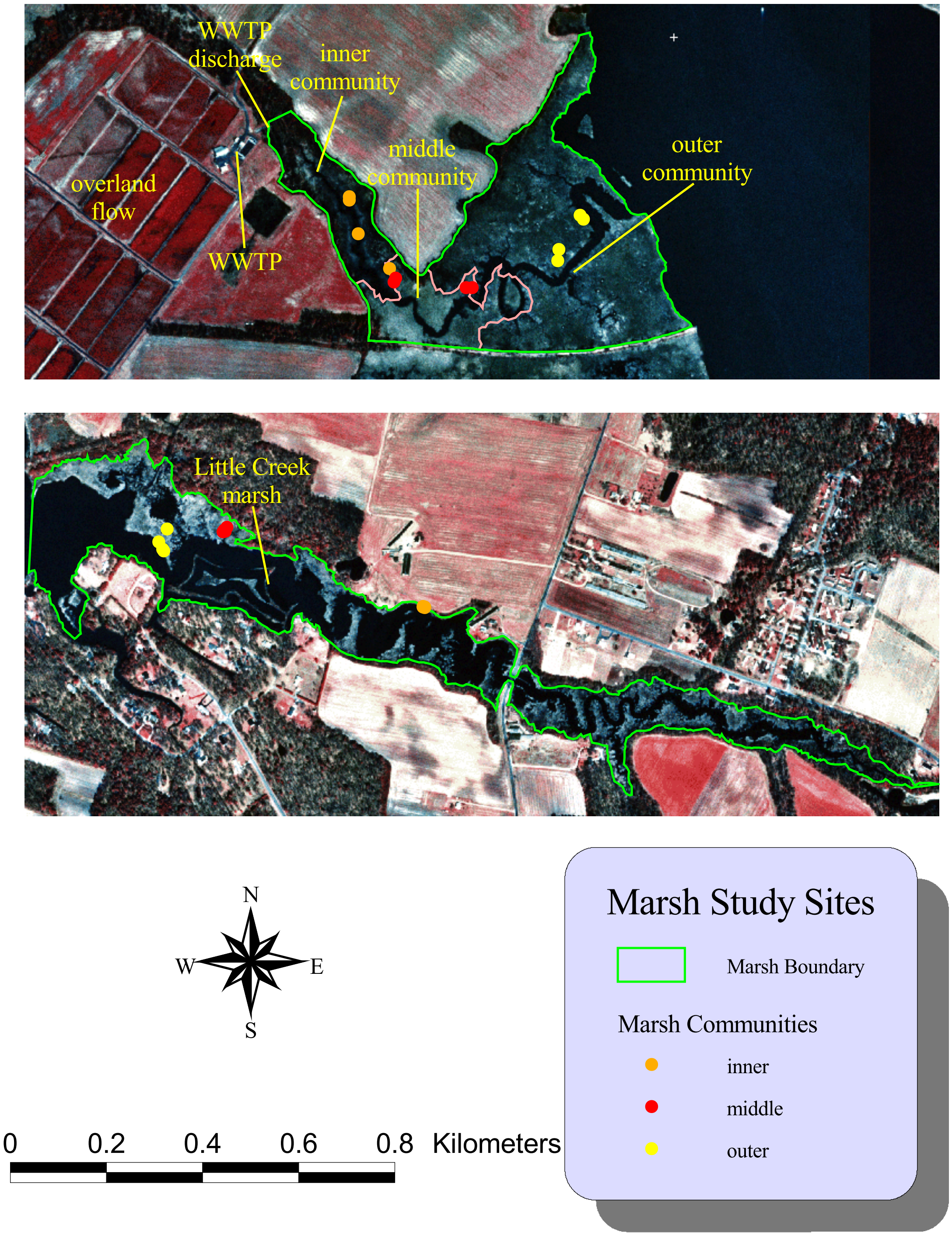
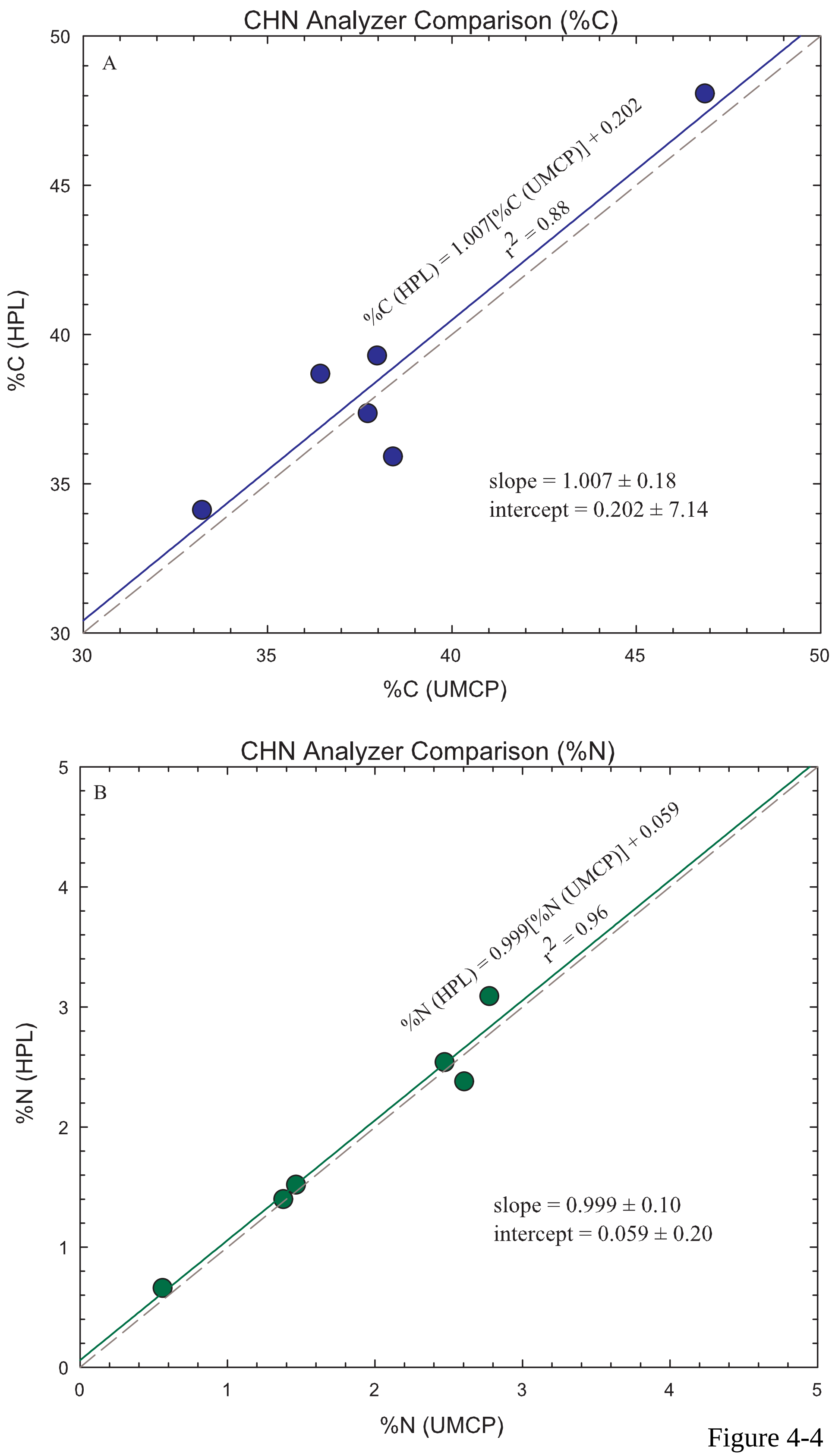
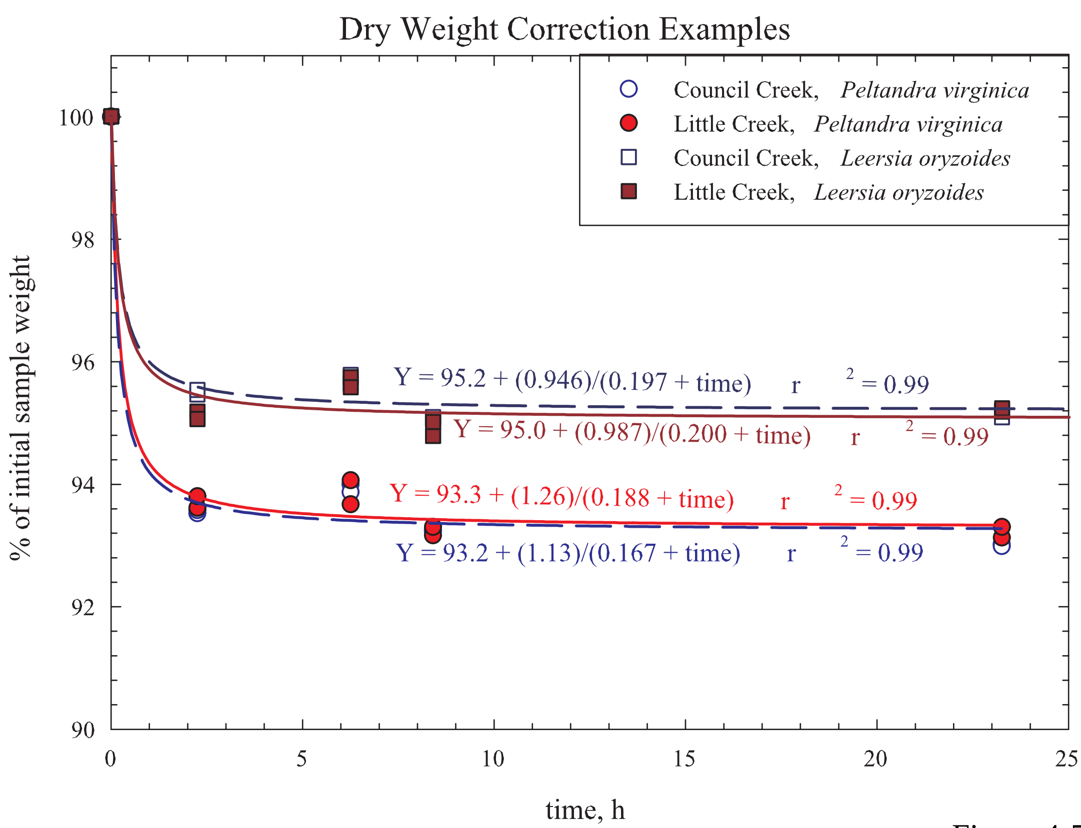


Figure 4-3



CHN Analyzer Comparison (%C)
A
%C (HPL) = 1.007[%C (UMCP)] + 0.202
r2 = 0.88
slope = 1.007 ± 0.18
intercept = 0.202 ± 7.14
%C (HPL)
%C (UMCP)
30
35
40
45
50
CHN Analyzer Comparison (%N)
B
%N (HPL) = 0.999[%N (UMCP)] + 0.059
r2 = 0.96
slope = 0.999 ± 0.10
intercept = 0.059 ± 0.20
%N (HPL)
%N (UMCP)
0
1
2
3
4
5

Figure 4-4



Dry Weight Correction Examples
Council Creek, Peltandra virginica
Little Creek, Peltandra virginica
Council Creek, Leersia oryzoides
Little Creek, Leersia oryzoides
Y = 95.2 + (0.946)/(0.197 + time) r 2 = 0.99
Y = 95.0 + (0.987)/(0.200 + time) r 2 = 0.99
Y = 93.3 + (1.26)/(0.188 + time) r 2 = 0.99
Y = 93.2 + (1.13)/(0.167 + time) r 2 = 0.99
% of initial sample weight
100
98
96
94
92
90
0
5
10
15
20
25
time, h

Figure 4-5

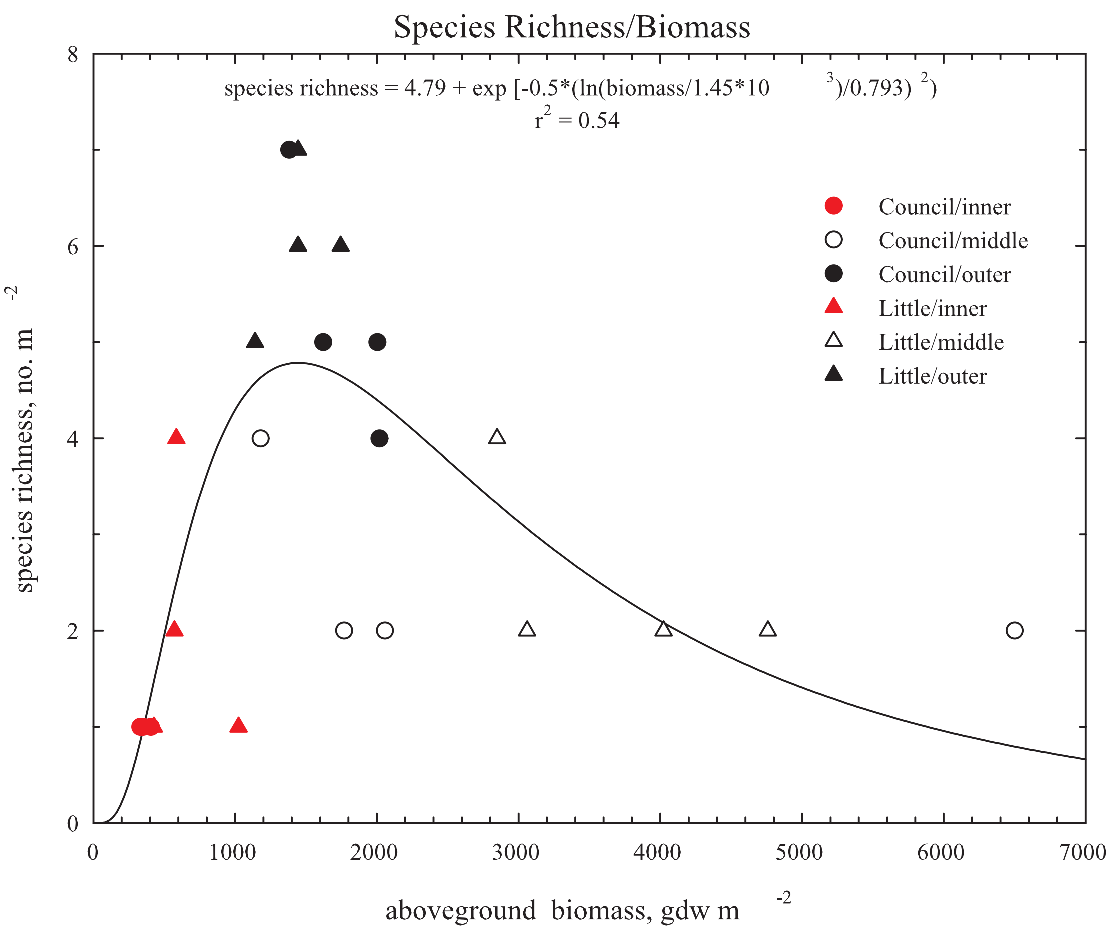
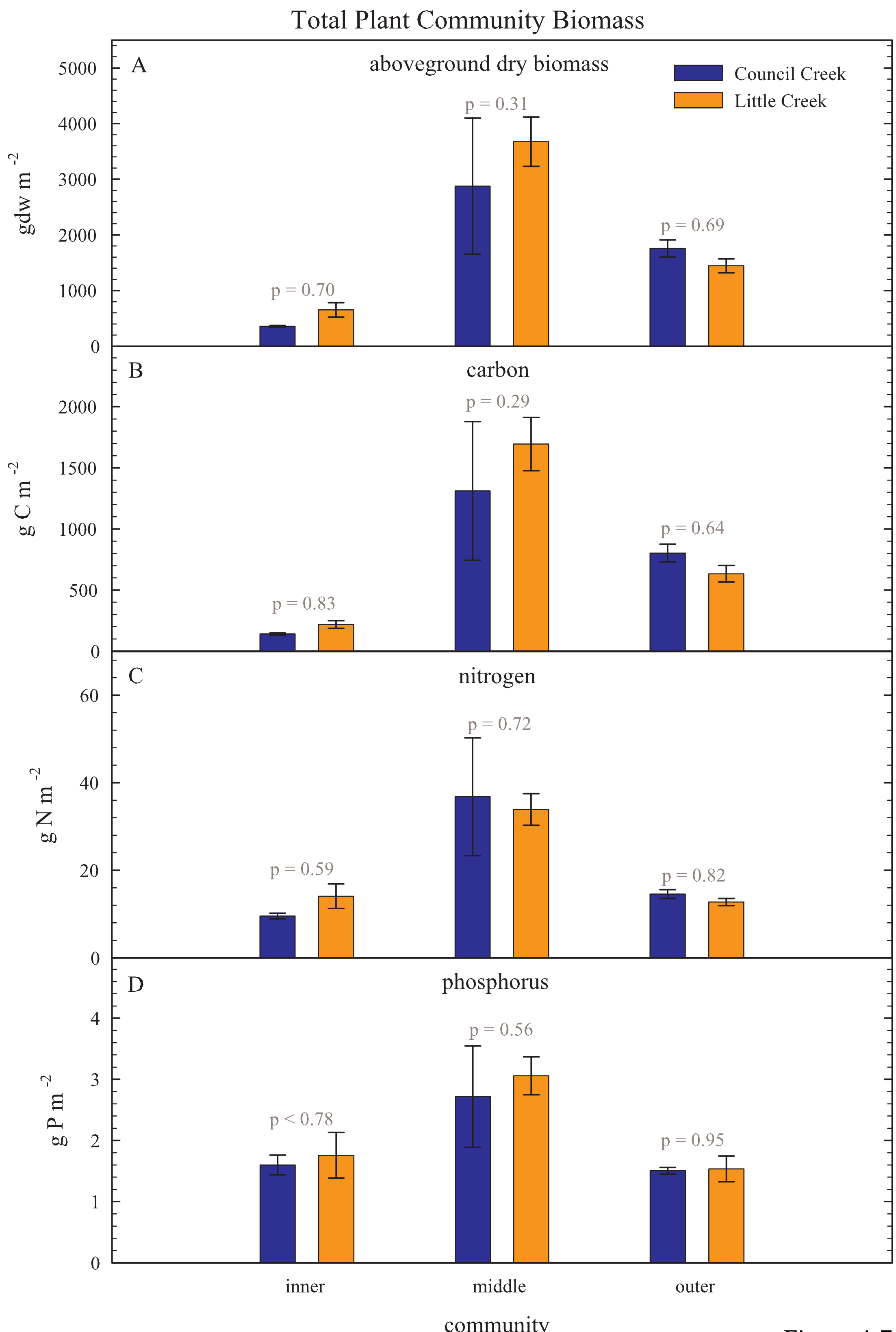

Species Richness/Biomass
species richness = 4.79 + exp [-0.5*(ln(biomass/1.45*10 3)/0.793) 2)
r2 = 0.54
Council/inner
Council/middle
Council/outer
Little/inner
Little/middle
Little/outer
species richness, no. m -2
aboveground biomass, gdw m -2
0
2
4
6
8
0
1000
2000
3000
4000
5000
6000
7000

Figure 4-6



Total Plant Community Biomass
A
aboveground dry biomass
Council Creek
Little Creek
gdw m -2
5000
4000
3000
2000
1000
0
p = 0.70
p = 0.31
p = 0.69
B
carbon
g C m -2
2000
1500
1000
500
0
p = 0.83
p = 0.29
p = 0.64
C
nitrogen
g N m -2
60
40
20
0
p = 0.59
p = 0.72
p = 0.82
D
phosphorus
g P m -2
4
3
2
1
0
p < 0.78
p = 0.56
p = 0.95
inner
middle
outer
community

Figure 4-7

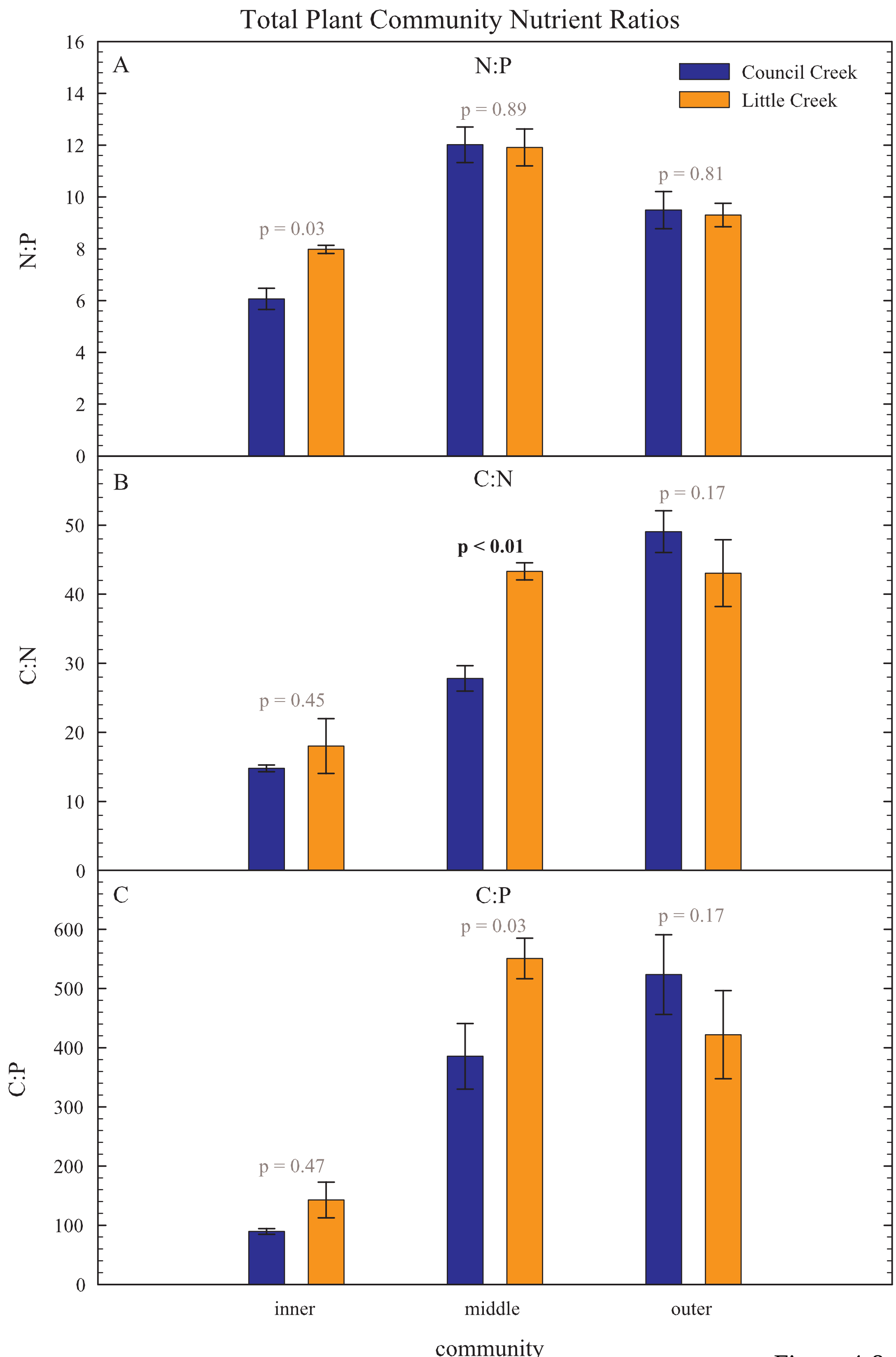
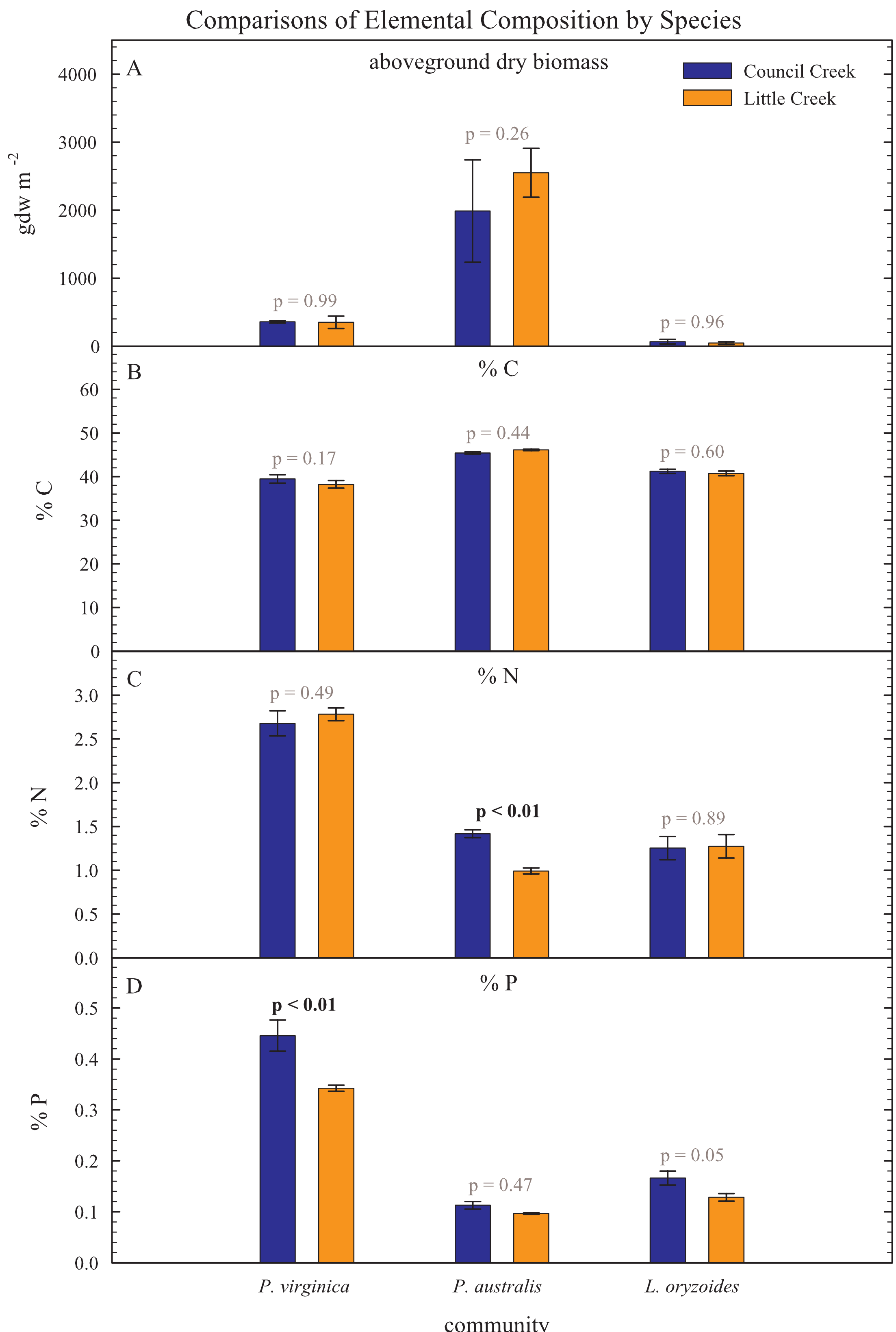
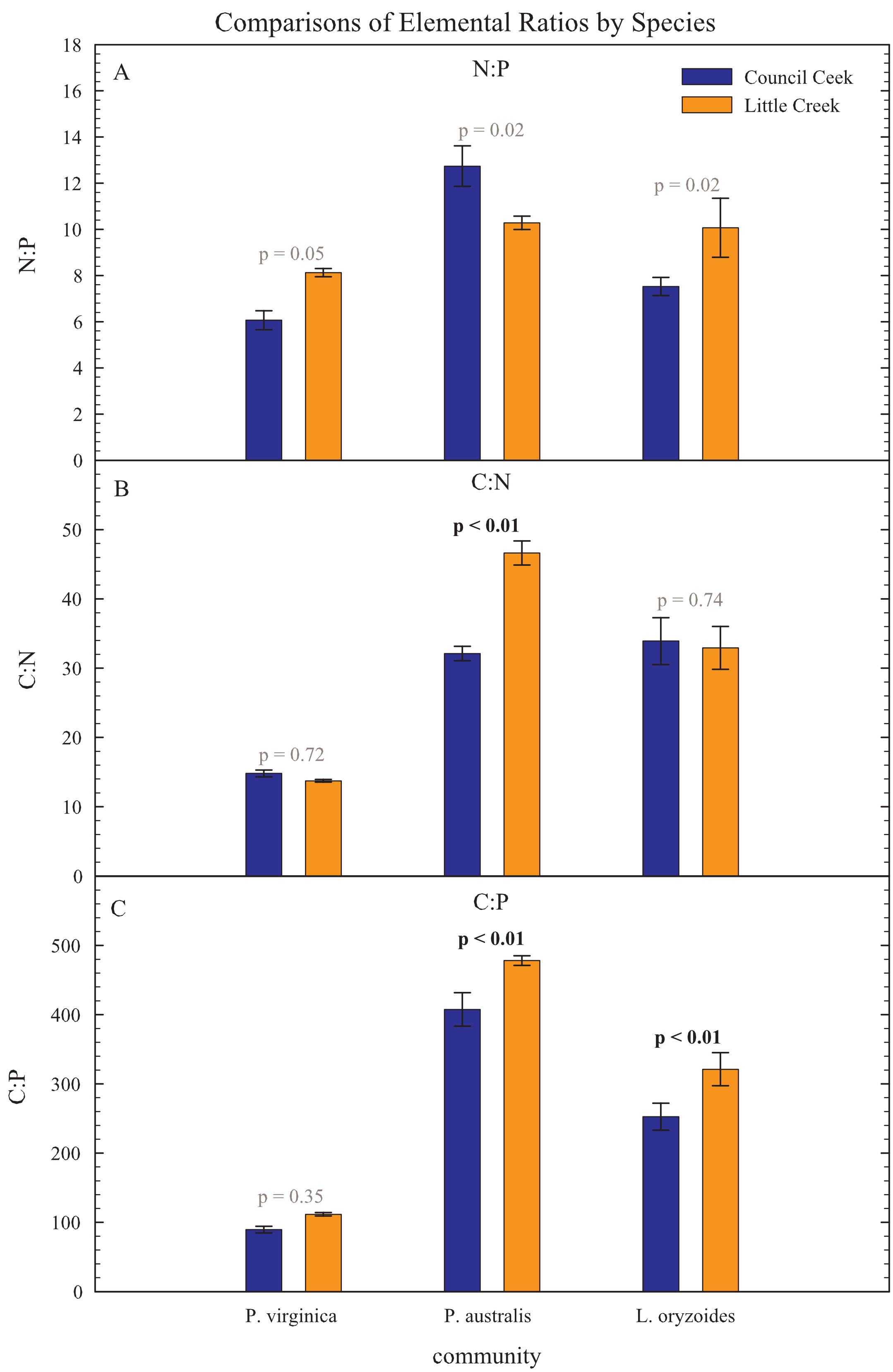

Total Plant Community Nutrient Ratios
A
N:P
Council Creek
Little Creek
p = 0.03
p = 0.89
p = 0.81
B
C:N
p = 0.45
p < 0.01
p = 0.17
C
C:P
p = 0.47
p = 0.03
p = 0.17
inner
middle
outer
community

Figure 4-8



Comparisons of Elemental Composition by Species
A
aboveground dry biomass
Council Creek
Little Creek
gdw m -2
4000
3000
2000
1000
0
p = 0.99
p = 0.26
p = 0.96
B
% C
60
50
40
30
20
10
0
p = 0.17
p = 0.44
p = 0.60
C
% N
3.0
2.5
2.0
1.5
1.0
0.5
0.0
p = 0.49
p < 0.01
p = 0.89
D
% P
0.5
0.4
0.3
0.2
0.1
0.0
p < 0.01
p = 0.47
p = 0.05
P. virginica
P. australis
L. oryzoides
community

Figure 4-9



Comparisons of Elemental Ratios by Species
A
N:P
Council Ceek
Little Creek
p = 0.05
p = 0.02
p = 0.02
N:P
18
16
14
12
10
8
6
4
2
0
B
C:N
p < 0.01
p = 0.74
p = 0.72
C:N
50
40
30
20
10
0
C
C:P
p < 0.01
p < 0.01
p = 0.35
C:P
500
400
300
200
100
0
P. virginica
P. australis
L. oryzoides
community

Figure 4-10

Chapter 5

SEDIMENT NUTRIENT BURIAL IN A WASTEWATER-IMPACTED OLIGOHALINE MARSH

Abstract

Nitrogen and phosphorus burial in wetland sediments is an important mechanism for long-term nutrient removal. Council Creek marsh is a 0.16 km^2 oligohaline wetland through which an average of 5716 m^3 d^{-1} (1.53 MGD) of passes from the small nearby town of Easton, MD (population ~12,000) on the Delmarva Peninsula. The hypotheses tested here are (1) the high N and P loading from the wastewater effluent passing through the marsh has increased nutrient burial in the marsh sediments relative to other nearby marshes with no wastewater and (2) there are gradients in sediment nutrient content with increasing distance downstream of the discharge point, with increasing distance from the creek bank, and vertically within the sediments.

The marsh sediment data presented here only partially supports these hypotheses. The data offered little support for the first hypothesis that increased N and P loading increased nutrient burial. Only average N content and N burial were significantly higher in Council Creek marsh than other nearby Choptank River marshes. However, the data do partially support the second hypothesis. There was both a longitudinal and lateral gradient in sediment iP and TP content, although there was no gradient in sediment N or oP content. Mehlich 3 P data also indicated that the sediments of the inner and particularly the middle marsh deposited in the last 40 years are in the high to very high-risk category of P-release. In contrast, the Mehlich 3 P data from the outer marsh core showed a profile similar to the two other Choptank marsh cores. Integrated over the

entire marsh surface, burial of N and P via sediment accumulation represents 27 % and 5.9 % of N and P wastewater inputs to the marsh.

Introduction

With increased anthropogenic activity in coastal river basins, estuarine and coastal waters are becoming increasingly degraded. Nitrogen (N) and phosphorus (P) inputs from both diffuse and point sources are one of the major causes of degrading water quality, because these nutrients stimulate the growth of algae in natural waters (Fisher et al. 1988, Valiela et al. 1990). Preventing these pollutants from reaching coastal waters has become an important point of research in recent decades (e.g., Carpenter et al. 1998).

Tidal wetlands are valued for their high productivity and potential to improve water quality (Kiker and Lynne 1997, Mitsch and Gosselink 2000). Macrophyte uptake, denitrification, and burial of N and P in wetland sediments are important mechanisms for nutrient removal. However, nutrient immobilization by vegetation is seasonal; uptake occurs during the growing season with potential release of nutrients in fall and winter (see Chapter 3 and 4). Direct denitrification measurements have only recently become readily available (Kana et al. 1994), and the seasonal and spatial variations are large and not well characterized (Merrill 1999). In contrast, nutrient burial in sediments is easily measurable with lead-210 (^{210}Pb) geochronology and can account for more permanent removal (Johnston et al. 1984).

Lead-210 (22.3 year half-life) is a naturally occurring Pb isotope that results from decay of radon (^{222}Rn) gas through the emission of alpha particles (Krishnaswami and Lal 1978). The terrestrially derived decay of ^{222}Rn results in background or supported ^{210}Pb activity (Merrill 1999). However, since ^{222}Rn is a gas, much of it escapes to the atmosphere. As atmospheric ^{222}Rn decays, ^{210}Pb sorbs to particles, which then fall back to Earth in wet and dry deposition, becoming trapped in marsh sediments as they accumulate, which results in excess or unsupported ^{210}Pb activity. From the known 22.3-year half-life of ^{210}Pb, sediment accretion rates can be accurately estimated.

It is the position of wetlands in the landscape that makes them such productive ecosystems and efficient at trapping materials. Located at the boundary between terrestrial and aquatic ecosystems, wetlands have an adequate water supply to support high rates of biological activity and biogeochemical processing of material passing from land to water (Kadlec and Knight 1996). However, this position also makes them a convenient point for disposal of municipal wastewater. Historically, the primary goal of wastewater disposal was removal from the municipality to reduce the spread of water-borne disease locally (Stoddard et al. 2002), and this typically meant that wastewater was discharged to rivers and estuaries. It soon became apparent that the receiving waters were becoming degraded and consequently affecting downstream municipalities. To address these problems, municipal wastewater treatment methods were developed and implemented for the purpose of reducing contaminants and pathogens. However, the increased costs of more effective, yet increasingly complex wastewater treatment are important limitations on the widespread use of advanced treatment (Kadlec and Knight 1996).

This chapter of my thesis focuses on sedimentary processes and long-term burial of wastewater N and P in sediments in an oligohaline marsh. Council Creek marsh on the Choptank estuary receives an average of 5716 $m^3\ d^{-1}$ (1.53 MGD) of wastewater, transporting an average of 59 kg N d^{-1} and 16 kg P d^{-1} from the town of Easton, Maryland on the Delmarva Peninsula. I hypothesize that high N and P loading from the wastewater effluent passing through the marsh has increased nutrient burial in the sediments of Council Creek marsh relative to other similar marshes in the Choptank basin. Furthermore, I expect to see gradients in sediment nutrient content with increasing distance downstream of the discharge point, with increasing distance from the creek bank, and vertically within the sediments. Evidence to support or reject this hypothesis should emerge from an examination of sediment cores collected in the marsh and from comparisons with cores collected in nearby marshes with no wastewater.

Study Site Description

Council Creek is located in the Choptank River basin on the Delmarva Peninsula (eastern shore of the Chesapeake Bay, Figure 5-1). In Figure 5-2, Council Creek basin within the yellow polygon has an area of 2.2 km^2 and discharges through a 0.16 km^2 natural oligohaline marsh before flowing into the Choptank River. The town of Easton, Maryland is located approximately 6.3 km north/northwest of Council Creek (Figure 5-1), and the municipal wastewater treatment plant is located within the Council Creek watershed (Figure 5-2). Additionally, part of the Talbot County, Maryland landfill is located within the Council Creek watershed.

Historically, Easton's wastewater treatment plant has undergone several upgrades (Cronshaw et al. 1990). Wastewater treatment began in 1911 with two primary treatment facilities located in the town of Easton. To meet the demands of a growing town, a new larger facility was constructed at the present location in 1938 with effluent being discharged directly into the Choptank River via a pipe following the road now forming the southern boundary of the basin (Figure 5-2). The presently used lagoon system was completed in 1962, at which time the discharge point moved from the Choptank River to Council Creek (upstream of its current location, Figure 5-2). In the late 1980's, the overland flow system was completed and the discharge point was moved to its current location (Figure 5-2).

Currently, the treatment plant consists of three components. The first component is the lagoon system, which is used to settle out solids and oxidize organic matter (Figure 5-2). Water retention time in the lagoons is approximately 90 days. The second component is the overland flow system (Figure 5-2), which consists of 0.28 km^2 of grass terraces (Cronshaw et al. 1990). Water from the lagoons is spray-irrigated onto the terraces to allow plant and soil processes to remove nutrients from the wastewater, and the grass is periodically harvested to remove plant-accumulated nutrients. During this process the treatment plant has the ability to cease all discharge, or to by-pass the overland flow fields to allow them to dry. The third component is the treatment facility (Figure 5-2), where water is first chlorinated to kill any remaining pathogens, de-chlorinated, and finally oxygenated before being discharged into Council Creek. The entire treatment system is designed to discharge a maximum of about 8900 m^{-3} day^{-1} (2.35 MGD) of treated wastewater (Cronshaw et al. 1990). After discharge into Council Creek, the wastewater flows 1.2 km through the meandering creek before entering the Choptank River. Once the wastewater is in the creek, the tides cause creek water to regularly flood over the surrounding marsh, where further removal of N and P occurs in an area of marsh that is equivalent to 57% of the grassed overland flow system (see Chapters 3 and 4).

Methods

Core Sampling

Sediment cores were collected during 2002 in Council Creek marsh using a McAuley corer. This corer collects a 100 cm by 7.6 cm half round core while minimizing vertical compaction of marsh sediments. Each core was sectioned in the field into thirteen 2.5–10 cm samples for analyses, and sections were transferred to labeled 50 ml Corning Snap-Seal containers, which were then placed in a cooler until returning to the lab. All core locations were georeferenced using a Garmin GPS III-plus to <15 m accuracy, and points were later downloaded to a personal computer using MapSource software (v 3.02). The sampling strategy was designed to define spatial patterns of burial in the marsh area by collecting cores longitudinally down the creek and laterally out from the stream bank (Figure 5-3).

Nutrient and iron analyses

Upon returning to the lab, the core sections were prepared for analyses. Assuming a constant (measured) weight ± se of 10.095 ± 0.002 g for the 50 ml containers, core sections and containers were weighed on a Mettler Toledo model PJ400 balance. The containers were filled to the 50 ml mark with deionized water and reweighed to obtain the core section volume. Samples were then dried at ~60 ˚C to constant weight (~14 days), reweighed and corrected for the container mass. Bulk density (dry weight volume^{-1}) was calculated by dividing corrected dry mass by the calculated volume of the core section. To account for sediment compaction, bulk densities at >30 cm were used for calculations. Core sections were ground in a large mortar and pestle and stored in labeled 20 ml plastic scintillation vials for analyses.

All sediment samples were analyzed for total N, total P, inorganic P (iP), and iron (Fe). All contents and rates are expressed per gram dry sediment (e.g., mg g^{-1}). Prior to analyses, samples were re-dried overnight at ~60 ˚C to evaporate any hygroscopic water that the sediments may have acquired in storage. Nitrogen content was determined using a Leco CHN 2000 Carbon, Nitrogen, and Hydrogen Analyzer that used a Leco alfalfa standard. Samples run on the Leco CHN 2000 Analyzer were pre-weighed into tin capsules and sent to the University of Maryland College Park Soils Lab where they were analyzed. Total and inorganic P were determined using the procedure described by Aspila et al. (1976); organic P (oP) was calculated by difference from TP and iP. Inorganic P extracts were also saved for later Fe analysis on a Varian Spectra AA-20 Atomic Absorption Spectometer, which was calibrated with deionized water blanks and Fe standards of up to 10 ppm. A 5 ppm standard was run between every 10 samples to insure that instrument drift was <5%; the instrument was recalibrated with a complete set of standards if drift was >5%.

Lead-210 geochronology

Subsamples (~1 g) of the core sections were analyzed for ^{210}Pb utilizing alpha spectroscopy. Lead-210 was counted using polonium-210 (^{210}Po), a daughter nuclide of ^{210}Pb, as a proxy for ^{210}Pb under the assumption of secular equilibrium between ^{210}Pb and ^{210}Po. The much shorter half-life (only 138 days ensuring secular equilibrium in < 1 year), emission of alpha particles, and strong sorption on silver plating makes ^{210}Po easier to measure than the long-lived, beta-emitting ^{210}Pb. A constant ratio of 1:1 Pb:Po was used to convert counts of ^{210}Po to ^{210}Pb.

Subsamples of the core sections were first digested in 10 ml of concentrated HNO_3 for one hour. Ten ml of concentrated HCl were then added and heated at 85°C for 1.5 hours. After samples had cooled, they were centrifuged to remove residual sediment. The supernatant was dried overnight at ~70-80ºC, re-dissolved in concentrated HCl twice, and evaporated to dryness to remove residual HNO_3, which can interfere with the plating procedure. Samples were then dissolved in 0.10N HCl and a small amount of ascorbic acid was added to prevent interference from iron. Silver plates (~ 1 cm^2) were added to the solution and allowed to sit overnight at 70ºC to sorb Po from solution. The plates with sorbed Po were then removed from the solution, rinsed in deionized water, and counted for ~24 h on a Canberra model 7404 four channel Alpha Spectrometer using Tennelec TC256 and Ortec 576A detectors.

Lead-210 calculations were completed using the constant initial concentration model, which assumes constant sedimentation rates (Merrill 1999). To utilize this model, unsupported (or excess) ^{210}Pb was calculated by subtracting measured total ^{210}Pb values from supported (or background) ^{210}Pb levels (from ^{222}Ra in sediments). This calculation yields an exponentially declining profile of excess ^{210}Pb (Figure 5-4 panel A). The natural logarithm of excess ^{210}Pb values was plotted as a function of cumulative mass down the length of the core (Figure 5-4 panel B), and the reciprocal decay constant ^{210}Po (y^{-1}) was then divided by the slope of the linear regression of natural logarithm of excess ^{210}Pb versus cumulative mass (g cm^{-2}) plot to yield a sedimentation rate (g m^{-2} y^{-1}) for each core:

$$\text{sedimentation} = (\text{reciprocal decay constant}/k) * (10^{4}\ \text{cm}^{2}\ \text{m}^{-2}) \qquad (1)$$

where reciprocal decay constant = -0.0311 y^{-1}, k (g cm^{-2}) is the slope of the regression of the natural logarithm of excess ^{210}Pb and cumulative mass (Figure 5-4). Marsh accretion rates (cm y^{-1}) were derived by dividing sedimentation rate (g m^{-2} y^{-1}) by >30 cm bulk density (g cm^{-3}):

$$\text{accretion rate} = \text{sedimentation rate}/\text{bulk density} * (10^{-4}\ \text{m}^{2}\ \text{cm}^{-2}) \qquad (2)$$

Nutrient burial rates (g m^{-2} y^{-1}) were calculated by multiplying sedimentation rate (g m^{-2} y^{-1}) by average nutrient content (>10 cm, mg g^{-1}).

$$\text{nutrient burial} = (\text{sedimentation rate} * \text{nutrient content}) * (10^{-3}\ \text{g mg}^{-1}) \qquad (3)$$

P-saturation

The Mehlich 3 P test is used to estimate soil saturation with respect to P (Sims et al. 2002). Compared to the established but lengthy acid oxalate P saturation method for determining Degree of P Saturation (DPS) in soils, Mehlich 3 is a more routine and faster soil P test, which Sims et al. (2002) found to be well correlated with DPS. Bioavailable P, Al, and Fe are extracted using a weakly acidic mixture of CH_3COOH, NH_4NO_3, NH_4F, HNO_3, and EDTA, and the resultant extracts are analyzed by inductively coupled plasma atomic absorption spectroscopy at the University of Delaware Soil Testing Laboratory.

A raw P Saturation Ratio (raw-PSR) is calculated as the molar ratio of P/[Al +Fe], and then converted to a DPS equivalent M3-PSR value yielding an index of a soil/sediment's likelihood of P release. An M3-PSR value of < 25 is a sediment that is at low to medium risk of P loss via runoff and/or leaching, an M3-PSR value of 25-50 is a sediment that is at high risk of P loss, and an M3-PSR value of > 50 is a sediment that is at very high risk of P loss (Sims et al. 2002).

Mehlich 3 P analysis was performed on five core sections (0-2.5, 5-7.5, 10-15, 30-35, and 50-60 cm) of four Council Creek marsh cores. For comparison, the Mehlich 3 P analysis was also run on two Choptank marsh sediment cores collected by Jeff Cornwell for the CISnet study of the Choptank River. One core was collected in a marsh just upstream of Council Creek (59-A) and the other core was collected in a marsh just downstream of Council Creek (55-1, Figure 5-1). To maintain consistency, all cores analyzed were located near the creek bank. For one of the cores (55-1), there was not enough sample for the 0-2.5 cm section, so it was vertically composited with the 2.5-5.0 cm section at a ratio of 1:1.

Results

Ten sediment cores were collected in Council Creek marsh. All ^{210}Pb data (dpm g^{-1}) resulted from the analysis of 10 of the 13 sections from each core to contrast variations in vertical profiles and sediment accretion rates (Figure 5-5). Lead-210 generally declined down the length of each core (Figure 5-5), and an average ± se asymptotic level of 1.67 ± 0.19 dpm g^{-1} was obtained for supported ^{210}Pb (Table 5-1). Although one to three outlier points were omitted from the regressions in five of the cores, the model generally fit the data well, and correlation coefficients varied from 0.43

to 0.98 (Table 5-1).

Three cores were collected in the upper plant community, two were collected in the middle plant community, and five were collected in the outer plant community (Figure 5-3, see Chapter 4 for plant communities). Core locations ranged from 128 to 1034 m down stream of the wastewater discharge point and 1.6 to 49.8 m from the creek bank (Table 5-2, Figure 5-5). Bulk densities (>30 cm) ranged from 0.15 to 0.60 g dry sediment (cm^{-3} core volume)$^{-1}$ (g cm^{-3}) with a mean ± se of 0.43 ± 0.05 g cm^{-3}. Bulk density exhibited no significant ($p > 0.05$) spatial relationship with either distance from the wastewater outfall or distance from the creek bank.

Marsh sedimentation (g m^{-2} y^{-1}) and accretion rates (cm y^{-1}) varied over an order of magnitude. Sedimentation rates computed with eq. 1 in the 10 cores from Council Creek marsh ranged from 745–7893 g m^{-2} y^{-1}, with an average ± se of 3095 ± 765 g m^{-2} y^{-1} (Table 5-2); the lowest rates (core H) occurred in the outer marsh, and highest rates (core E) of sedimentation occurred in the middle marsh. Accretion rates, computed with eq. 2, varied from 0.22–1.69 cm y^{-1}, with an average ± se of 0.70 ± 0.13 cm y^{-1} (Table 5-2). The lowest accretion rate occurred at the inner marsh (core J), while the highest occurred in the middle marsh (core E). Sedimentation showed a general pattern of higher rates closer to the creek bank (<15 m from bank) and lower rates at distances >15 m from the creek bank (Figure 5-6). With the exception of core J that exhibited a sedimentation rate of only 1117 g m^{-2} y^{-1}, the other five cores located < 15 m from the creek bank had sedimentation rates greater than 2000 g m^{-2} y^{-1}. Longitudinally, down the length of the creek, these five cores also showed a modal pattern with maximum sedimentation rates in and near the middle marsh community. Although accretion rates showed a pattern similar to sedimentation, as expected from eq. 2, there was greater scatter among the points (data not shown). Bulk density exhibited a significant exponential increase as a function of sedimentation rate, up to a maximum of about 0.56 g cm^{-3} ($r^2 = 0.53$, $p < 0.05$), but accretion rates did not. Although community averaged sedimentation and accretion rates were both highest in the cores taken in the *Phragmites australis* community (middle community), these averages were not significantly higher ($p > 0.05$) than the inner or outer community rates. However, cores near the creek bank (Figure 5-6) clearly showed a strong spatial pattern with highest rates in the middle marsh community.

Sediment nutrient content

Nutrient concentrations in the cores were highly variable. Average core N, iP, TP and oP are reported in Table 5-2, and sediment Fe analyses were utilized to determine an Fe to iP relationship. To account for post-depositional mobility (accumulation at or just below the redox boundary near the top of the cores), these values are averages of all core sections below 10 cm (hereafter referred to as average N or P content).

Nitrogen content generally decreased with increasing depth, especially in the top 10-20 cm, as shown by the N profiles of all marsh cores (Figure 5-7); however, cores H and G (Figure 5-7) deviated from this general pattern and were highly organic at the bottom of the core probably due to the presence of older deposits. Consequently, sediment N was high with an average ± se of 14.7 ± 1.8 mg N g^{-1} and variable with a range of 9.42 – 26.95 mg g^{-1}. Average sediment N content exhibited no spatial pattern longitudinally down the marsh, laterally across the marsh, or as a function of accretion.

Average sediment N, however, exhibited a significant inverse correlation with bulk density ($r^2 = 0.92$, $p < 0.0001$, Figure 5-8 top panel) and a significant exponential relationship with sedimentation rates, declining with increasing sedimentation ($r^2 = 0.59$, $p < 0.05$, Figure 5-8 bottom panel).

Sediment P content also generally decreased with depth, especially in the top 10-20 cm (Figure 5-9). The exceptions are cores F and E (see below) with large P accumulations at mid-depth. Average ± se TP was 0.75 ± 0.11 mg g^{-1}, with a minimum of 0.43 mg g^{-1} in core G and a maximum of 1.50 mg g^{-1} in core F (Table 5-2). This represents a similar range of variation as was observed for N content (~3). Average iP paralleled average TP, but with an average ± se of 0.39 ± 0.13 mg g^{-1} and a range of 0.06 – 1.34 mg g^{-1}. Both average TP ($r^2 = 0.44$, $p < 0.05$) and iP ($r^2 = 0.51$, $p < 0.05$) were significantly correlated with distance from the wastewater outfall (Figure 5-10 top panel), exhibiting an approximately linear decrease in P content with increasing distance from the discharge point. Additionally, average TP ($r^2 = 0.78$, $p < 0.01$) and iP ($r^2 = 0.87$, $p < 0.001$) content were both significantly and inversely related to distance from the creek bank (Figure 5-10 bottom panel), with P content decreasing exponentially within the first 10 m of the creek bank. In contrast, average oP content exhibited no spatial pattern and had an average ± se of 0.30 ± 0.04 mg g^{-1}, with a minimum of 0.15 mg g^{-1} in core H and a maximum of 0.57 mg g^{-1} in core I. No forms of P were significantly correlated with bulk density.

Cores F and E deviated from the general pattern of declining P content with increasing depth (Figure 5-9). These profiles exhibited a peak in P content related to iP's high affinity for binding with iron oxides. Core F showed a large peak in both TP and iP from about 20-40 cm depth due to high Fe content in these sections of the core (Figure 5-9 core F). A similar Fe/P peak was also seen from 5-20 cm in core E (Figure 5-9 core E). Inorganic P data core sections collected within 10 m of the creek bank exhibited a significant linear relationship to Fe content ($r^2 = 0.80$, $p < 0.0001$, Figure F-11). In contrast, core sections collected at locations >10 m from the creek bank showed no significant relationship with Fe content ($p > 0.05$). This pattern suggests that in sediments near the creek bank, virtually all P binding sites (Fe and Al, see P saturation ratio) are filled due to exposure to P-rich wastewater; however, sediments >10 m from the creek bank appeared to have available P binding sites.

There did not appear to any evidence of systematically elevated sediment N or P concentrations at depths shallower than 40 years old (Figures 5-6 and 5-8), despite 40 years of nutrient enrichment from the Easton wastewater plant. Using the calculated marsh accretion rates in Table 5-2, the dotted lines in Figures 5-6 and 5-8 indicate the approximate depths of the 40 year horizon (approximate start of wastewater discharge into Council Creek) and 80 year horizon. For N and P content of cores (Figure 5-7 and 5-9), there was little evidence of systematic enrichment above the 40-year horizon; however, at least in the case of P (Figure 5-9), changes in P content may be partially concealed by the post-depositional mobility of P. The primary spatial patterns of N and P content were related to distance from the outfall and creek bank (P, Figure 5-10) and sedimentation rates (N, Figure 5-8)

Sediment nutrient burial

Sediment nutrient burial in Council Creek marsh exhibited high spatial variability as shown in Table 5-2. N burial had an average ± se of 38.7 ± 8.1 g m^{-2} y^{-1}, with a range of 11.9 – 98.7 g m^{-2} y^{-1}. Inorganic and total P burial was 1.67 ± 0.73 and 2.60 ± 0.88 g m^{-2} y^{-1} and ranged over 0.06–6.64 and 0.50–8.94 g m^{-2} y^{-1}, respectively. Organic P burial averaged 0.86 ± 0.21 g m^{-2} y^{-1}, with a range of 0.10–2.31 g m^{-2} y^{-1}.

Sediment nutrient burial was correlated with sedimentation and accretion. Nitrogen burial exhibited no spatial relationship with distance from the wastewater outfall, distance from the creek bank, or bulk density. However, variability in N burial was largely explained by significant linear correlations (autocorrelations) between N burial and sedimentation ($r^2 = 0.98$, $p < 0.0001$) and accretion rates ($r^2 = 0.91$, $p < 0.0001$), as expected from eq. 1-3. Total P burial was also correlated with sedimentation ($r^2 = 0.77$, $p < 0.001$) and accretion ($r^2 = 0.70$, $p < 0.01$). Inorganic P burial was also correlated with sedimentation ($r^2 = 0.64$, $p < 0.01$) and accretion ($r^2 = 0.56$, $p < 0.05$). Organic P burial was significantly correlated with both sedimentation ($r^2 = 0.88$, $p < 0.0001$) and accretion rates ($r^2 = 0.83$, $p < 0.001$). These relationships are expected from eq. 1-3, but imply that sedimentation rates, not sediment nutrient content, are the primary determinant of burial rates.

P-saturation

Five sections of 4 near bank cores were analyzed to determine if there was a longitudinal gradient in P saturation (PSR). Analyses of sections of cores F, J, E, and A resulted in M3-PSR depth profiles for cores located 128 m to 856 m downstream from the wastewater outfall (Figure 5-12). In addition to the Council Creek data, M3-PSR data from two cores taken from marshes near Council Creek and in close proximity to the

Choptank River were included in the core A panel of Figure 5-12. The shaded colors in Figure 5-12 indicate the likelihood of P release as measured by MS-PSR. The entire profiles of both marsh cores taken outside Council Creek, as well as the outer core in Council Creek, lie within the category of low to medium risk of P loss by leaching. This is indicative of free Fe and Al sites to bind P. Core F, the closest to the wastewater discharge point in Council Creek marsh, was in the category of high risk of P release from the surface down to about 26 cm where the M3-PSR dropped to a low to medium risk (Figure 5-12). It therefore appeared that the M3-PSR was elevated from the approximate time at which wastewater began to be discharged into Council Creek. Although marsh accretion was much lower at core J (332 m downstream of the wastewater outfall), the same pattern of high to very high risk of P release occurred in sections from the past 40 years (Figure 5-12 core J). Core E (526 m downstream of the wastewater outfall) had the highest surface M3-PSR values; however, M3-PSR declined to low to medium risk at approximately 30 cm or 20 years before present (Figure 5-12 core E). Core A, the farthest core from the wastewater discharge point, exhibited a profile similar to the cores from the other marshes, with the entire M3-PSR profile lying within the low to medium risk category. This M3-PSR data suggests that there is a longitudinal pattern in P saturation in creek bank sediments with the highest risk of P release being just within and upstream of the *Phragmites* (middle marsh) community. In addition, it also appears that this pattern has largely occurred within the past 40 years, probably as a result of the wastewater discharge.

Discussion

Sediment properties across Council Creek marsh exhibited high variability. Bulk density was significantly correlated with sedimentation rate, potentially as a result of increased sediment compaction with increased sedimentation. However, bulk density exhibited no relationship with accretion rates, distance from the wastewater outfall, or distance from the creek bank. These data differ from other studies that have found that bulk densities generally decreased toward the interior of the marsh (Pasternack and Brush 1998, Merrill 1999). Average ± se sedimentation and accretion rates were 3095 ± 765 g m^{-2} y^{-1} and 0.70 ± 0.13 cm y^{-1}, respectively, with the highest rates near the creek bank within and in the vicinity of the middle community (Figure 5-11 and Table 5-2). The high variability of and burial of N and all forms of P was explained primarily by variations in sedimentation and accretion rates, with iP burial also being correlated with distance from the creek bank (Table 5-2). Similar to iP burial, average iP and TP content were inversely correlated with distance from the wastewater outfall and from the creek bank (Figure 5-10), while organic P content was not correlated any of the variables considered here. Average N content was correlated with sedimentation rate and bulk density (Figure 5-8). Mehlich 3 P profiles showed that relative to two other Choptank marsh cores, Council Creek marsh sediments within ~600 m of the wastewater discharge appear to be in the high to very high risk category of P release (Figure 5-12 panels A, B, and C). In contrast, core A (856 m from the wastewater discharge and closest to the Choptank River) showed an M3-PSR profile similar to that of the other Choptank marsh cores with no exposure to wastewater (low risk of P release, Figure 5-12 panel D).

Test of the hypotheses

This chapter tested two hypotheses. Evidence was presented to test whether (1) N and P loading from the wastewater effluent passing through the marsh increased nutrient burial in the sediments of Council Creek marsh relative to nearby Choptank River marshes, and (2) there is a decreasing gradient in sediment nutrient content with both increasing distance downstream of the discharge point, with increasing distance from the creek bank, and vertically within the cores. To test these hypotheses, accretion rates and N and P burial in 10 marsh sediment cores collected in Council Creek marsh were compared.

The marsh sediment data presented here only partially supported these hypotheses. The data offered little support for the first hypothesis that high N and P loading increased nutrient burial (Table 5-3). Except for average N content and N burial, which were significantly higher in Council Creek marsh, there were no other significant differences from values reported for other nearby Choptank River marshes (Malone et al. 2003). In fact, oP content was significantly lower in Council Creek marsh than other Choptank River marshes (Malone et al. 2003). Only maximum rates of N and P burial in Council Creek marsh exceeded those of the other data sets compared in Table 5-3.

The data do, however, partially support the second hypothesis. There was both a longitudinal and lateral gradient in sediment iP and TP content (Figure 5-10), although there was no vertical gradient in sediment N or oP content that could be consistently related to the 40-year time horizon. Average iP and TP content exhibited linear decreases with increasing distance from the wastewater outfall, while average iP and TP exhibited exponential decreases with increasing distance from the creek bank (Figures 5-9). The Mehlich 3 P data also indicated that the sediments of the inner and particularly the middle marsh deposited in the last 40 years are in the high to very high-risk category of P-release

(Figure 5-12 cores F, J, and E). In contrast, the Mehlich 3 P data from the outer marsh core showed a profile similar to the two other Choptank marsh cores (Figure 5-12 core A), indicating the spatial limit of P saturation in Council Creek marsh.

Council Creek marsh cores were compared to 11 other Choptank cores collected for the CISnet project (Malone et al. 2003). These cores were compared because the locations of the marshes in which they were collected were in close proximity to Council Creek. T-tests revealed that four of the eleven parameters in Table 5-2 (columns 5 – 16) were significantly different from the values for other cores taken outside Council Creek. Average N content, N burial, and >30 cm bulk density were significantly higher in Council Creek marsh ($p < 0.05$), while average oP content in Council Creek marsh was significantly lower than the other marshes ($p < 0.05$). However, most comparisons indicated that Council Creek marsh core properties were similar to those observed elsewhere in the Choptank.

Due to high nutrient loading in Council Creek marsh from wastewater, N burial does appear to be significantly greater than other nearby marshes; however, P burial does not. Since N is a good indicator of sediment organic content (Khan and Brush 1994), it would appear that sediments in Council Creek marsh have high organic content. This conclusion, however, conflicts with the higher average bulk densities and lower average oP content observed in Council Creek marsh. Increased decomposition of organic matter (i.e., marsh organic matter is decomposing near the surface before it can be buried) could explain the lower P values, but the reason for high N content is unclear.

Accumulation of N and P in the marsh sediments of Council Creek was quite variable, but exhibited ranges that overlapped N and P burial in other regional marshes

(Table 5-3). Jug Bay marshes in the Patuxent estuary were found to accumulate sediment in the range of 0.50 – 0.64 cm y^{-1} in the past 100 years with N burial of 7.5 – 19 g m^{-2} y^{-1} and P burial of 0.6 – 1.2 g m^{-2} y^{-1} (Khan and Brush 1994). These Jug Bay N and P burial values did not significantly differ ($p > 0.05$) from Council Creek burial rates; however, this is likely due to the small sample size of the Jug Bay study (2 marsh cores). In tidal freshwater marshes of the York River, Greiner and Hershner (1998) estimated sediment accumulation of 0.3 – 0.56 cm y^{-1} with P burial ranging from 0.07 – 1.01 g m^{-2} y^{-1}. Burial of P in the York River marshes was significantly lower ($p < 0.05$) than P burial in Council Creek. The ranges of burial for these other systems largely overlap with those of Council Creek marsh, with the highest values were obtained in Council Creek marsh. It is likely that spatial heterogeneity is obscuring higher N and P burial rates in Council Creek marsh as well.

Burial of N and P

On an annual basis, the Easton Wastewater Treatment Plant discharges approximately $2.2*10^7$ g N and $5.7*10^6$ g P. Based on these discharge rates and community averaged N and P burial rates, the percent of wastewater N and P that is buried in sediments can be estimated (Table 5-4). For each plant community, average N and P burial (g m^{-2} y^{-1}) was calculated and multiplied by the community area (m^2, see also Chapter 4). From the resulting total community N and P burial, percent of wastewater N and P buried was calculated (Table 5-4 columns 5, 8). These calculations indicate that sediment nutrient burial in Council Creek marsh appears to play a small role in wastewater nutrient processing, burying only 27% of wastewater N and 5.9 % of wastewater P annually. In addition, it appears that most of the P removal is occurring most intensively in the middle marsh (Table 5-4) and within 10 m of the creek bank (Figure 5-10). Overall it appears that the utilization of Council Creek marsh as an additional wastewater treatment component offers only marginal effectiveness with respect to long-term nutrient removal by burial in sediments.

This study examined the long-term removal of wastewater nutrients via burial in marsh sediments. Although there was no clear spatial pattern of N burial, the marsh is annually removing approximately 27% of the wastewater N. Phosphorus in sediments was highest near the wastewater discharge point and near the creek banks, annually burying approximately 5.9 % of the wastewater P. It also appears that marsh sediments near the wastewater discharge are becoming P saturated, increasing the likelihood of P release back into creek water (Figure 5-12 and Chapter 3 Figure 3-15). This was, in fact, observed when wastewater flows stopped for plant management reasons (see Chapter 3). It therefore appears that Council Creek marsh has little capacity to remove additional P from the wastewater flowing through it, although some N removal via denitrification is likely (see Chapter 3).

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Table 5-1: Summary of excess ^{210}Pb regressions. Values for slopes were used in the calculations of sedimentation rates (equation 1). Surface and background ^{210}Pb are in units of disintegrations per minute per gram.

community	core	surface ^{210}Pb, dpm g^{-1}	background ^{210}Pb, dpm g^{-1}	no. pts in reg	r^2	slope
inner	F	4.35	2.27	8	0.90	- 0.084
inner	B	6.28	2.57	9	0.43	- 0.062
inner	J	3.91	1.70	8	0.89	- 0.278
middle	I	10.84	1.44	9	0.92	- 0.224
middle	E	6.73	1.36	9	0.88	- 0.039
outer	C	9.39	1.62	9	0.95	- 0.184
outer	A	8.09	2.31	6	0.62	- 0.053
outer	H	7.70	0.53	8	0.91	- 0.418
outer	G	8.09	1.21	9	0.98	- 0.141
outer	D	9.86	1.71	8	0.94	- 0.229
mean ± se		**7.53 ± 0.71**	**1.67 ± 0.19**			

Table 5-2: Summary of data obtained from the sediment cores. Cores are list by increasing distance from the wastewater outfall. "Comm" indicates the plant community in which the core was located, bulk density represents the average bulk density of core sections at depths greater than 30 cm, and average N, iP, TP, and oP content represent averages of core sections at depths greater than 10 cm. Sedimentation, accretion, and burial were calculated using equations 1-3.

comm.	core	dist from outfall, m	dist from bank, m	sedimentation, g m^{-2} y^{-1}	bulk density	accretion, cm y^{-1}	avg N, mg g^{-1}	avg iP, mg g^{-1}	avg TP, mg g^{-1}	avg oP, mg g^{-1}	N burial, g m^{-2} y^{-1}	iP burial, g m^{-2} y^{-1}	TP burial, g m^{-2} y^{-1}	oP burial, g m^{-2} y^{-1}
inner	F	128	1.6	3711	0.60	0.61	9.68	1.34	1.50	0.17	35.91	4.96	5.58	0.62
inner	B	205	11.5	5016	0.60	0.84	9.42	0.49	0.76	0.27	47.24	2.44	3.81	1.37
inner	J	332	3.5	1117	0.50	0.22	10.66	0.27	0.57	0.30	11.91	0.30	0.63	0.33
middle	I	471	29.2	1387	0.24	0.58	20.89	0.26	0.83	0.57	28.97	0.36	1.15	0.79
middle	E	526	2.8	7893	0.47	1.69	12.50	0.84	1.13	0.29	98.67	6.64	8.94	2.31
outer	C	804	38.5	1687	0.52	0.33	13.09	0.19	0.56	0.38	22.08	0.32	0.95	0.64
outer	A	856	12.9	5824	0.57	1.02	10.80	0.21	0.46	0.25	62.87	1.24	2.67	1.43
outer	H	938	32.8	745	0.15	0.51	26.95	0.08	0.67	0.14	20.07	0.06	0.50	0.10
outer	G	947	7.2	2212	0.31	0.71	16.45	0.06	0.43	0.21	36.39	0.14	0.95	0.47
outer	D	1034	49.8	1357	0.30	0.45	16.99	0.18	0.61	0.43	23.06	0.25	0.83	0.58
mean± se				**3095±765**	**0.43±0.05**	**0.70±0.13**	**14.71 ±1.79**	**0.39± 0.13**	**0.75± 0.11**	**0.30± 0.04**	**38.72± 8.13**	**1.67± 0.73**	**2.60± 0.88**	**0.86± 0.21**

Table 5-3: Summary of marsh accretion rates and N and P burial in Council Creek marsh and other regional marshes. N.d indicates no data.

marsh	no. of cores	accretion, cm y^{-1}	N burial, g m^{-2} y^{-1}	avg N burial, g m^{-2} y^{-1}	P burial, g m^{-2} y^{-1}	avg P burial, g m^{-2} y^{-1}	reference
Council Creek, Choptank	10	0.22 – 1.69	12 – 99	38.7 ± 8.1	0.50 – 8.94	2.60 ± 0.88	this study
oligohaline, Choptank	11	0.32 – 1.69	7 – 44	18.1 ± 3.2	0.5 – 3.8	1.72 ± 0.38	Malone et al 2003
tidal freshwater, York	14	0.30 – 0.56	n.d.	n.d.	0.07 – 1.01	0.28 ± 0.07	Greiner and Hershner 1998
tidal freshwater, Patuxent	2	0.50 – 0.64	7.5 – 19	13.3 ± 5.8	0.6 – 1.2	0.90 ± 0.30	Khan and Brush 1994

Table 5-4: Summary of N and P burial in Council Creek marsh. Cores from each plant community were averaged, and estimated areas of each community (see Chapter 4) were used to calculate burial of the wastewater N and P, which averaged 2.2×10^7 g N y^{-1} and 5.7×10^6 g P y^{-1}.

community	area, m^2	avg N burial, g m^{-2} y^{-1}	total N burial, g y^{-1}	% of wastewater N	avg TP burial, g m^{-2} y^{-1}	total P burial, g y^{-1}	% of wastewater P
inner	26038	31.69	825.10	3.82	3.34	87.04	1.53
middle	25287	63.82	1613.84	7.48	5.05	127.60	2.24
outer	104160	32.89	3426.24	15.88	1.18	122.86	2.16
total	**155485**	**38.72**	**5865.18**	**27.19**	**2.60**	**337.50**	**5.93**

FIGURE LEGENDS

Figure 5-1: Upper panel: Location of the Choptank Basin relative to Chesapeake Bay. Main panel: Location of Easton, Council Creek (see Figure 5-2)

Figure 5-2: Council Creek basin: Composite of aerial photographs of Council Creek basin. Yellow polygon indicates basin boundary, and green polygon indicates marsh boundary. Black horizontal line and red vertical line are the seams between aerial photographs in the composite (see Figure 5-3).

Figure 5-3: Council Creek marsh: Aerial photograph of Council Creek marsh, Green polygon indicates marsh boundaries, and colored dots indicate locations of sediment cores within each plant community (see Chapter 4).

Figure 5-4: Example ^{210}Pb profile and excess ^{210}Pb plot. Panel A: Lead-210 (dpm g^{-1}) versus depth (cm) for core G. Solid blue line represents the exponential regression line fitted to the profile. Dashed vertical gray line indicates the background ^{210}Pb level, while the dashed horizontal blue arrow indicates excess ^{210}Pb. Panel B: Natural logarithm of excess ^{210}Pb as a function of cumulative mass (g cm^{-3}) down core.

Figure 5-5: Lead-210 profiles: Lead-210 (dpm g^{-1}) versus depth (cm) profiles for all sediment cores. Vertical gray dashed lines indicate the estimated backgound ^{210}Pb. Cores are arranged left to right and top to bottom according to increasing distance downstream from the wastewater discharge point as shown by the diagram in the lower right.

Figure 5-6: Sedimentation relationship: Sedimentation (g m^{-2} y^{-1}) versus distance from the wastewater outflow (m). Blue filled circles and blue line represent cores collected < 15 m of the creek bank, while the blue line represents the regression of the near bank data. Red open triangles represent cores collected > 15 m of the creek bank. Vertical gray dotted lines indicate the approximate transition points between marsh plant communities.

Figure 5-7: Nitrogen profiles: Nitrogen (mg g^{-1}) versus depth (cm) profiles for all sediment cores. Dotted horizontal lines indicate 40 and 80 years before present. Cores are arranged left to right and top to bottom according to increasing distance downstream from the wastewater discharge point as shown by the diagram in the lower right.

Figure 5-8: Top panel: Average N (mg g^{-1}) as a function of bulk density (g cm^{-3}) in all cores from Council Creek marsh. Bottom panel: Average N (mg g^{-1}) as a function of sedimentation (g m^{-2} y^{-1}) in all cores from Council Creek marsh.

Figure 5-9: Phosphorus profiles: Phosphorus (mg g^{-1}) versus depth (cm) profiles for all sediment cores. Filled circles and solid lines indicate total P profiles, while open circles and dashed lines indicate iP profiles. Dotted horizontal lines indicate 40 and 80 years before present. Shaded areas in cores F and E indicate locations of iron peaks. Cores are arranged left to right and top to bottom according to increasing distance downstream from the wastewater discharge point as shown by the diagram in the lower right.

Figure 5-10: Top panel: Average P (mg g^{-1}) as a function of distance from the wastewater outfall (m). Bottom panel: Average P as a function of distance from the creek bank. Filled circles and solid lines indicate total P data, while open circles and dashed lines indicate iP data.

Figure 5-11: Iron (mg g^{-1}) to iP (mg g^{-1}) correlation. Blue filled circles represent sections of cores collected < 10 m of the creek bank, while the blue line represents the regression of the near bank data. Red open triangles represent sections of cores collected > 10 m of the creek bank.

Figure 5-12: PSR depth profiles of four cores from Council Creek marsh (F, J, E, and A), as well as two cores from nearby marshes without wastewater influence. Filled circles and solid line indicates the Council Creek sediment data. Bar at the bottom indicates locations of the cores along the creek length. Filled triangles and dashed/dotted line indicate data from the marsh downstream of Council Creek, while open hexagons and dashed line indicate data from the marsh upstream of Council Creek. The horizontal dotted lines indicate 40 and 80 years before present.

Choptank Basin

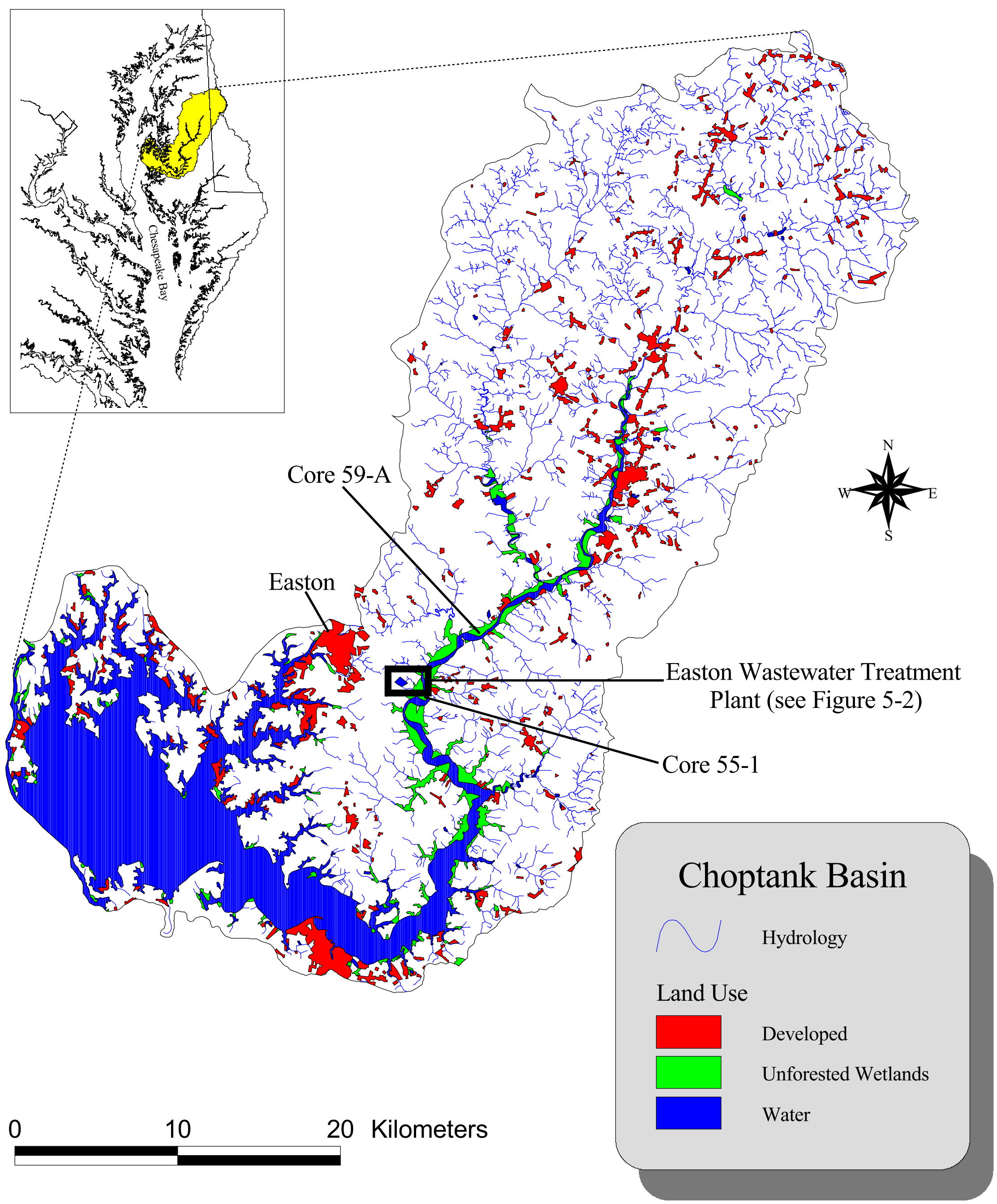


Figure 5-1

Council Creek Basin

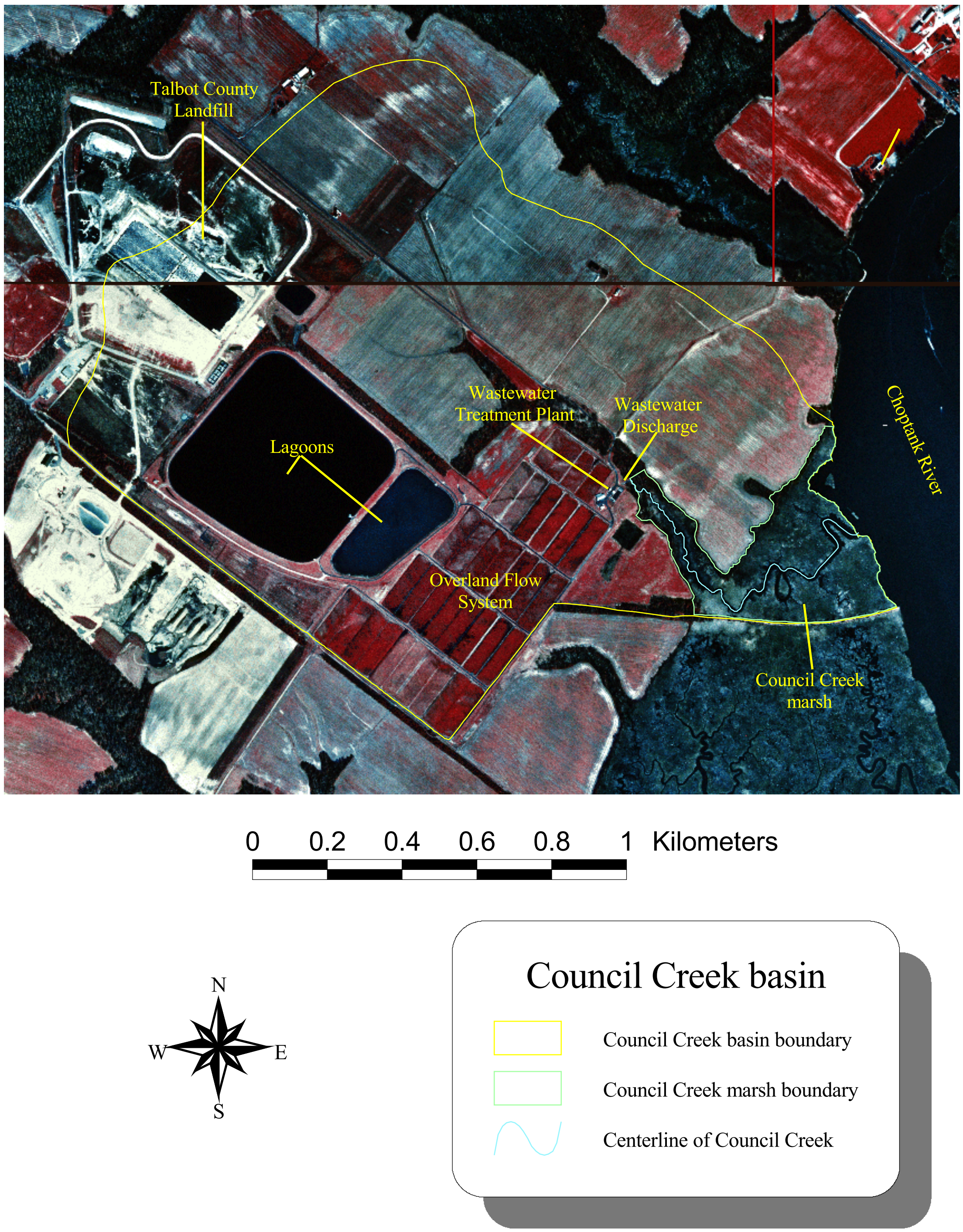


Figure 5-2

Council Creek marsh

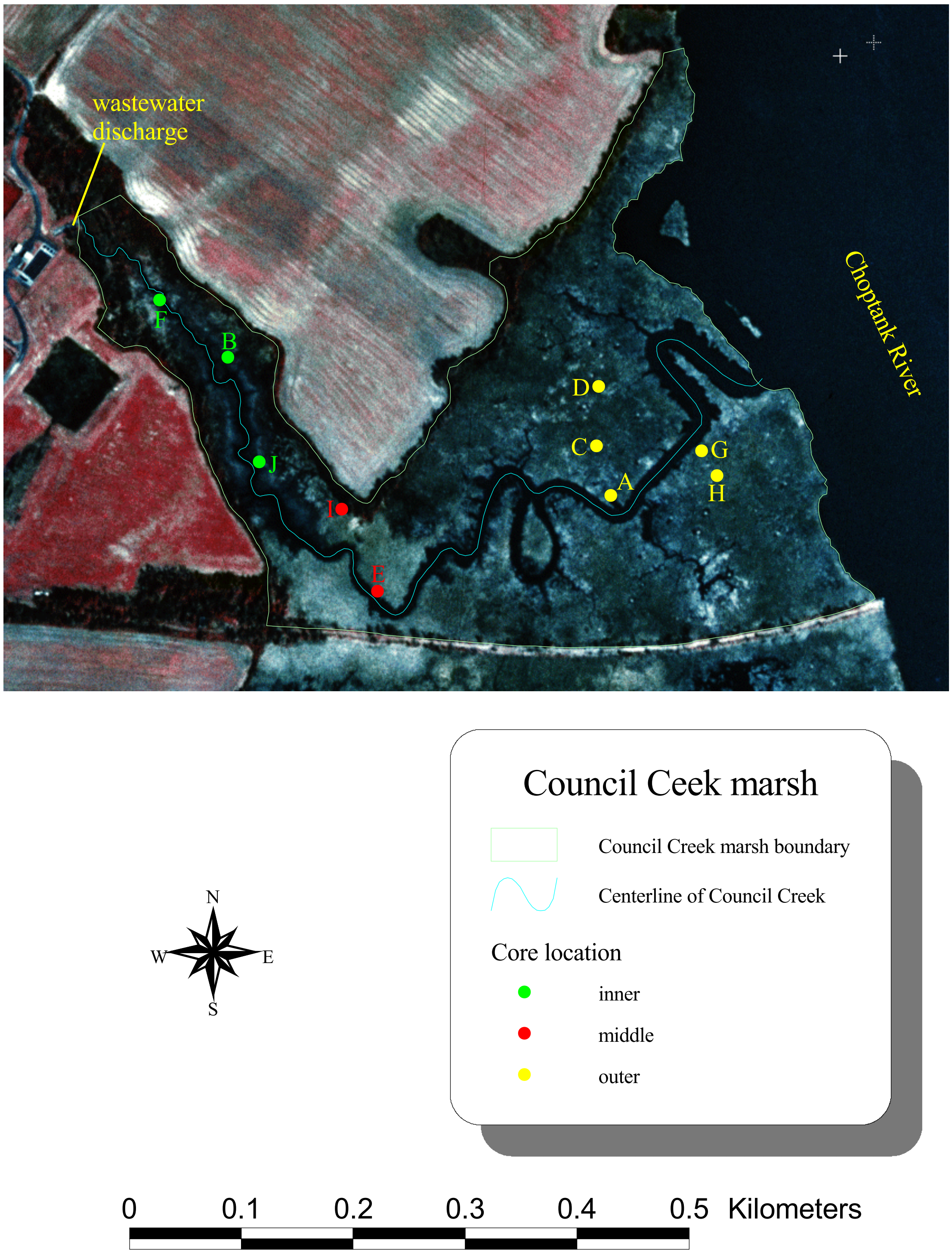
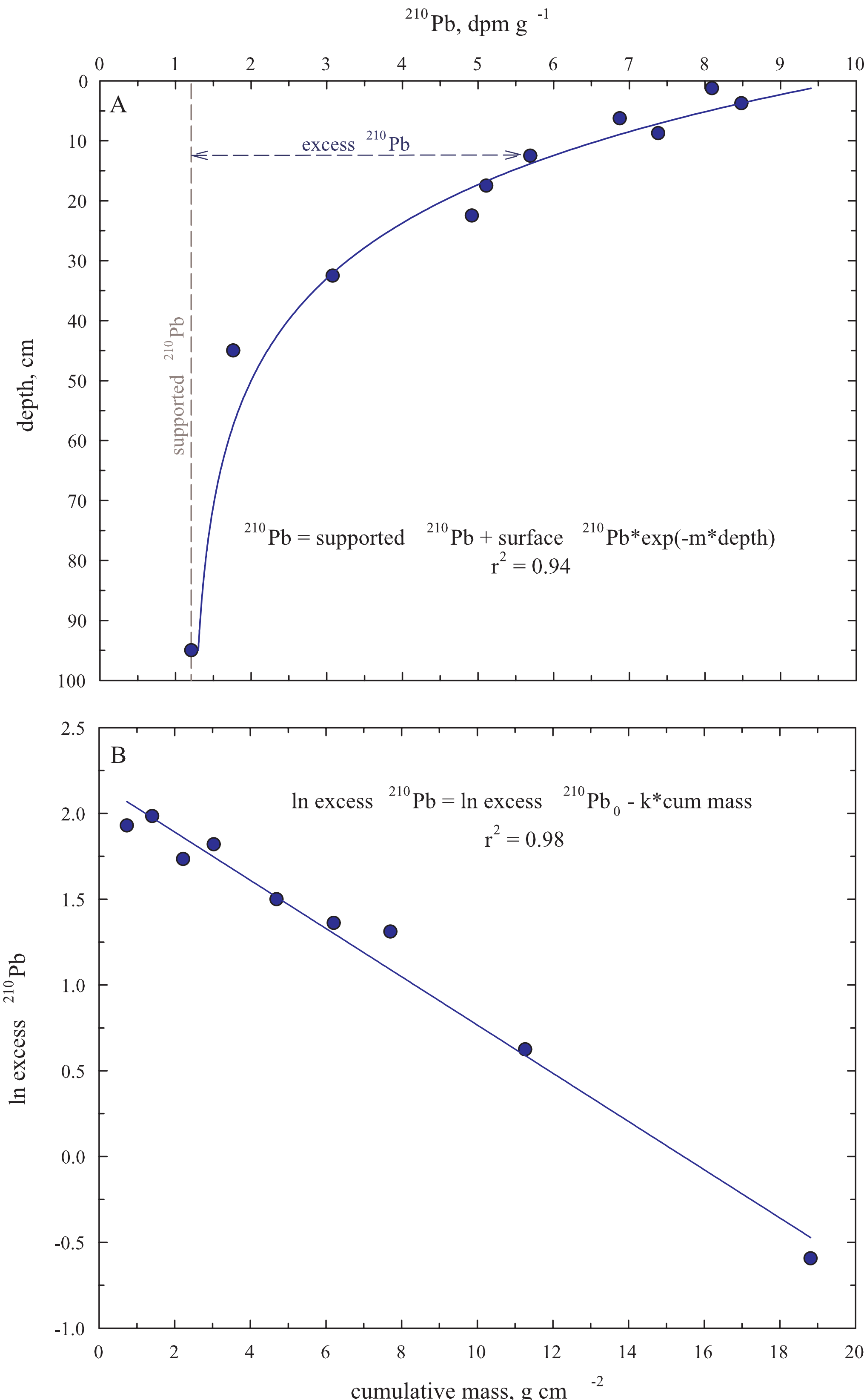
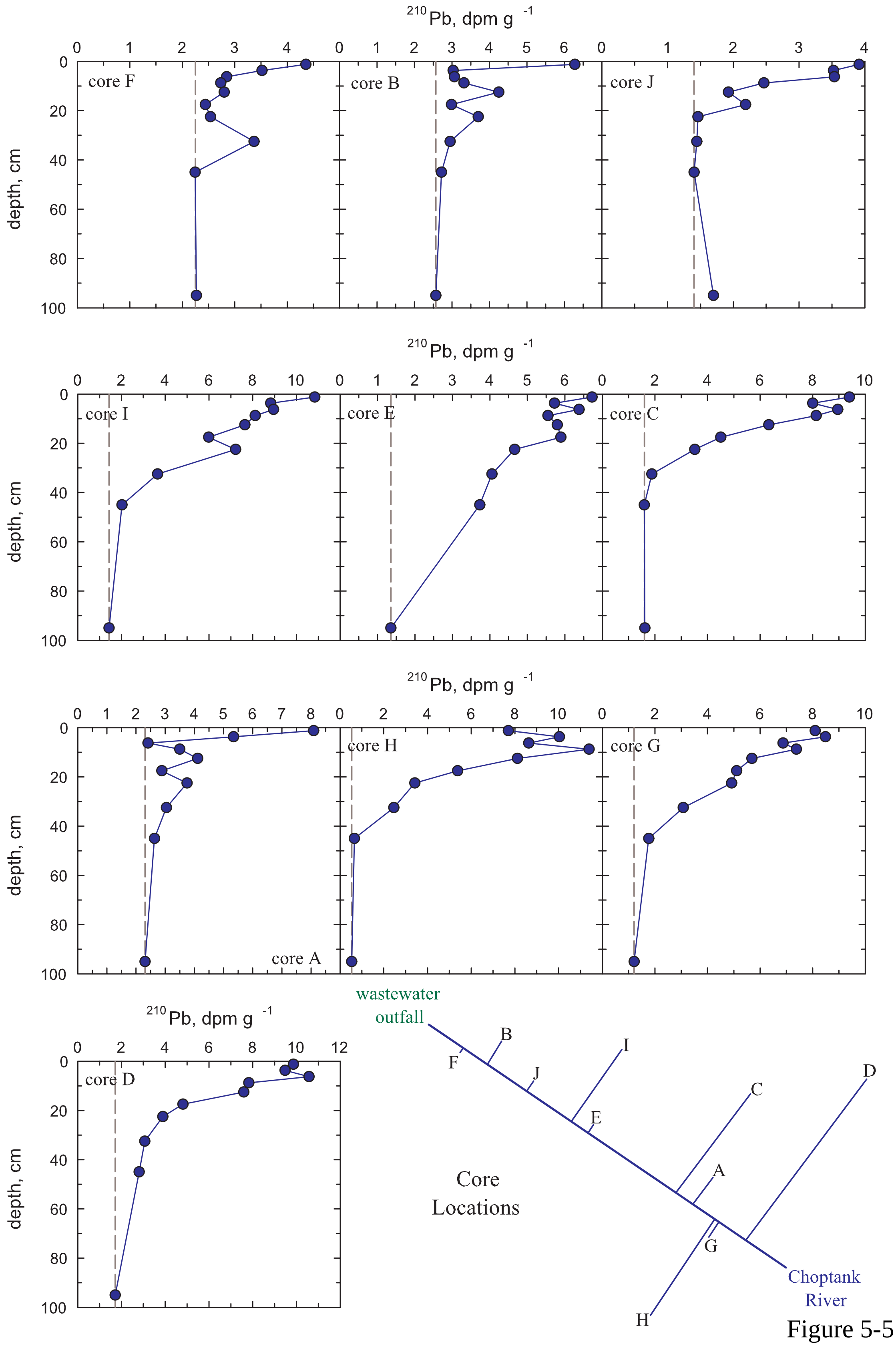


Figure 5-3



210Pb, dpm g -1
A
excess 210Pb
supported 210Pb
depth, cm
210Pb = supported 210Pb + surface 210Pb*exp(-m*depth)
r2 = 0.94
B
ln excess 210Pb = ln excess 210Pb0 - k*cum mass
r2 = 0.98
ln excess 210Pb
cumulative mass, g cm -2

Figure 5-4



210Pb, dpm g -1
depth, cm
core F
core B
core J
core I
core E
core C
core A
core H
core G
core D
wastewater
outfall
B
F
J
I
E
C
D
A
G
H
Core
Locations
Choptank
River

Figure 5-5

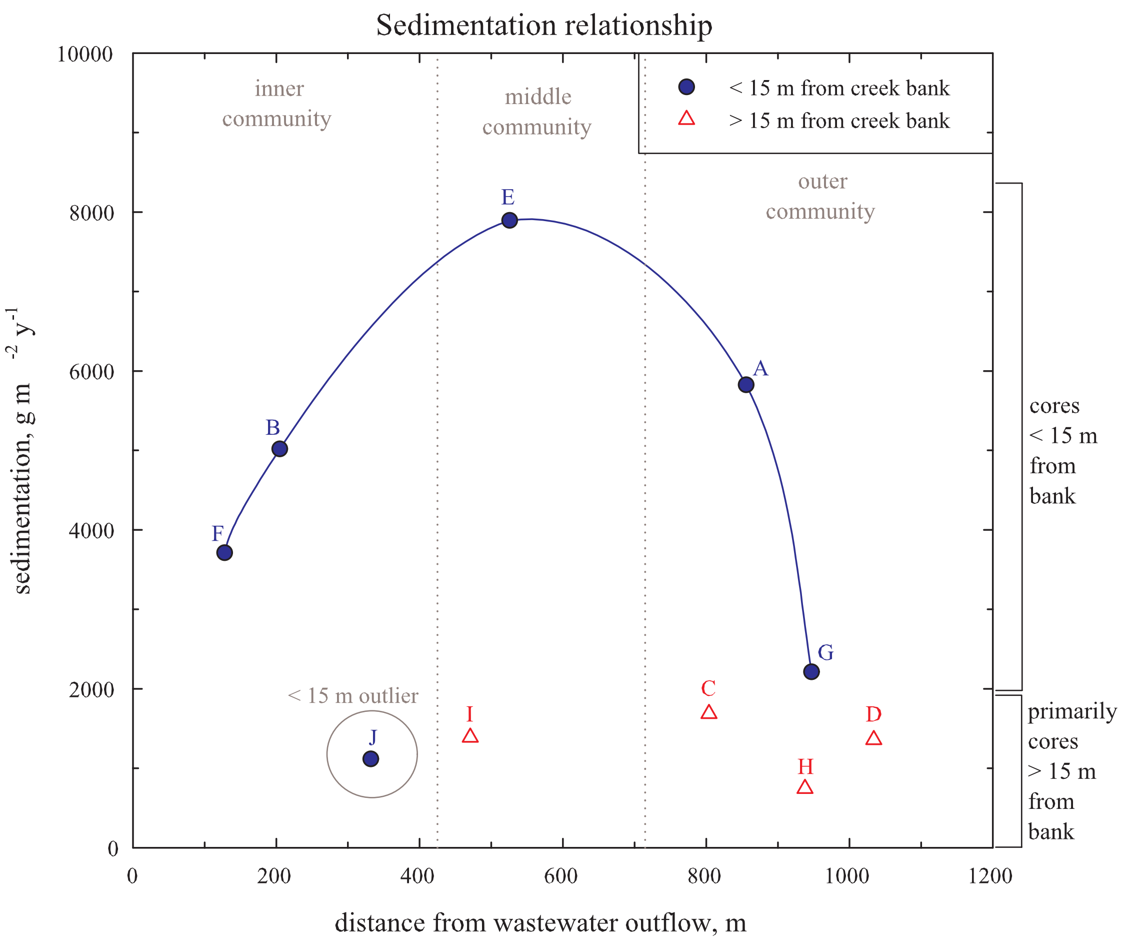

Sedimentation relationship
inner community
middle community
outer community
< 15 m from creek bank
> 15 m from creek bank
sedimentation, g m -2 y-1
distance from wastewater outflow, m
< 15 m outlier
F
B
E
A
G
J
I
C
H
D
0
2000
4000
6000
8000
10000
0
200
400
600
800
1000
1200
cores < 15 m from bank
primarily cores > 15 m from bank

Figure 5-6

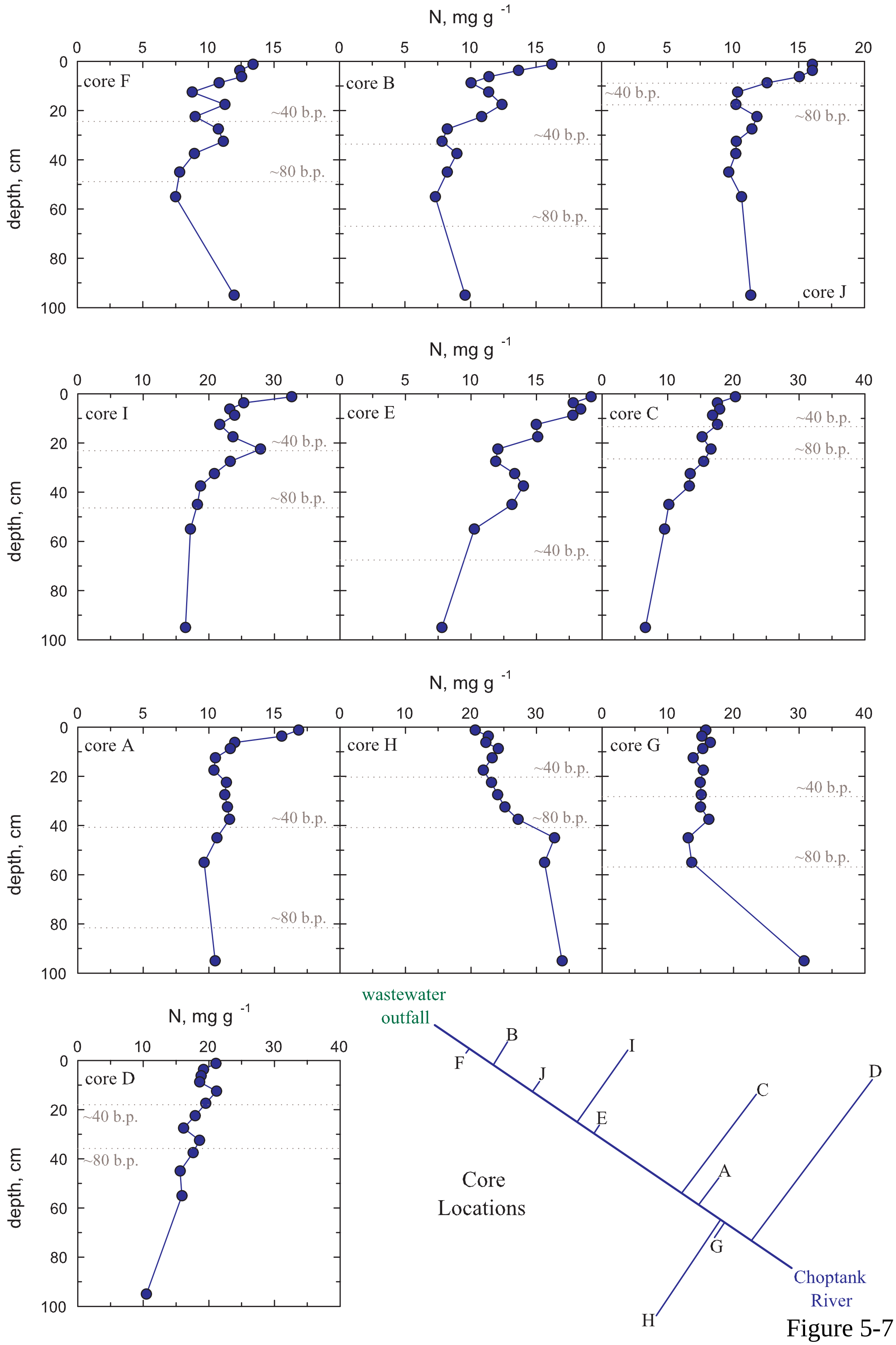

N, mg g -1
depth, cm
core F
core B
core J
core I
core E
core C
core A
core H
core G
core D
~40 b.p.
~80 b.p.
wastewater
outfall
Core
Locations
Choptank
River

Figure 5-7

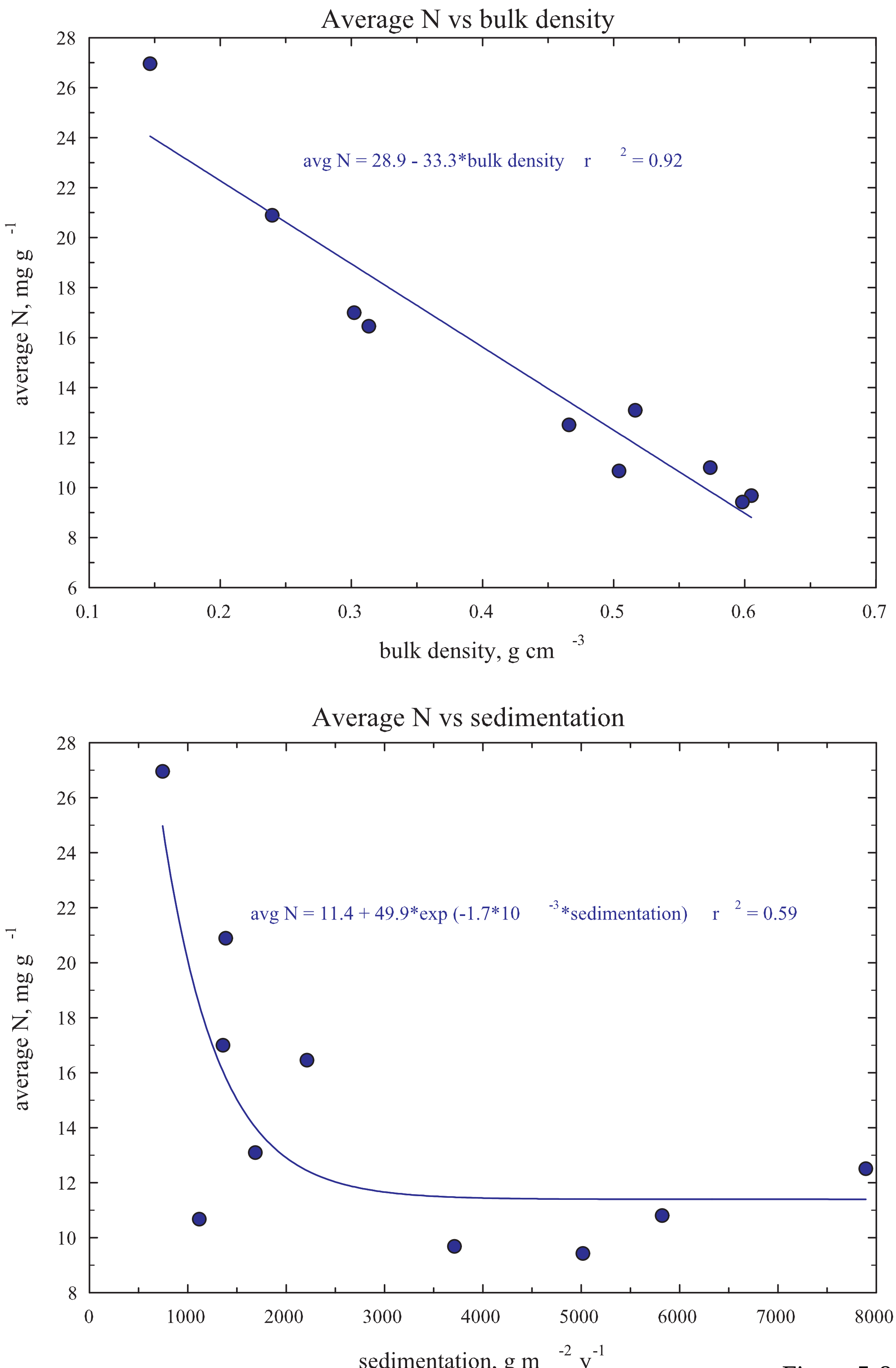

Average N vs bulk density
avg N = 28.9 - 33.3*bulk density r 2 = 0.92
average N, mg g -1
bulk density, g cm -3
28
26
24
22
20
18
16
14
12
10
8
6
0.1
0.2
0.3
0.4
0.5
0.6
0.7
Average N vs sedimentation
avg N = 11.4 + 49.9*exp (-1.7*10 -3 *sedimentation) r 2 = 0.59
average N, mg g -1
sedimentation, g m -2 y -1
28
26
24
22
20
18
16
14
12
10
8
0
1000
2000
3000
4000
5000
6000
7000
8000

Figure 5-8

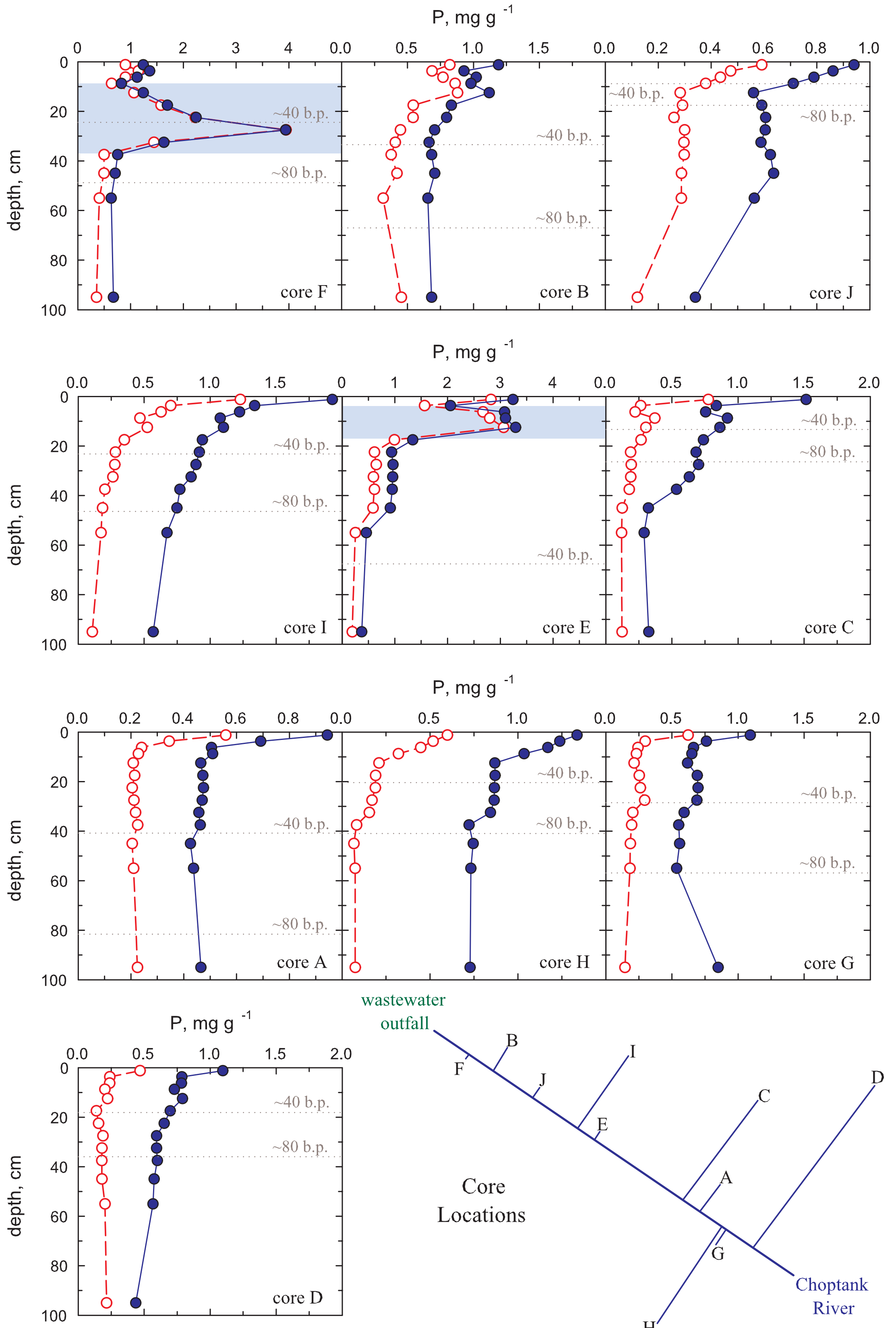
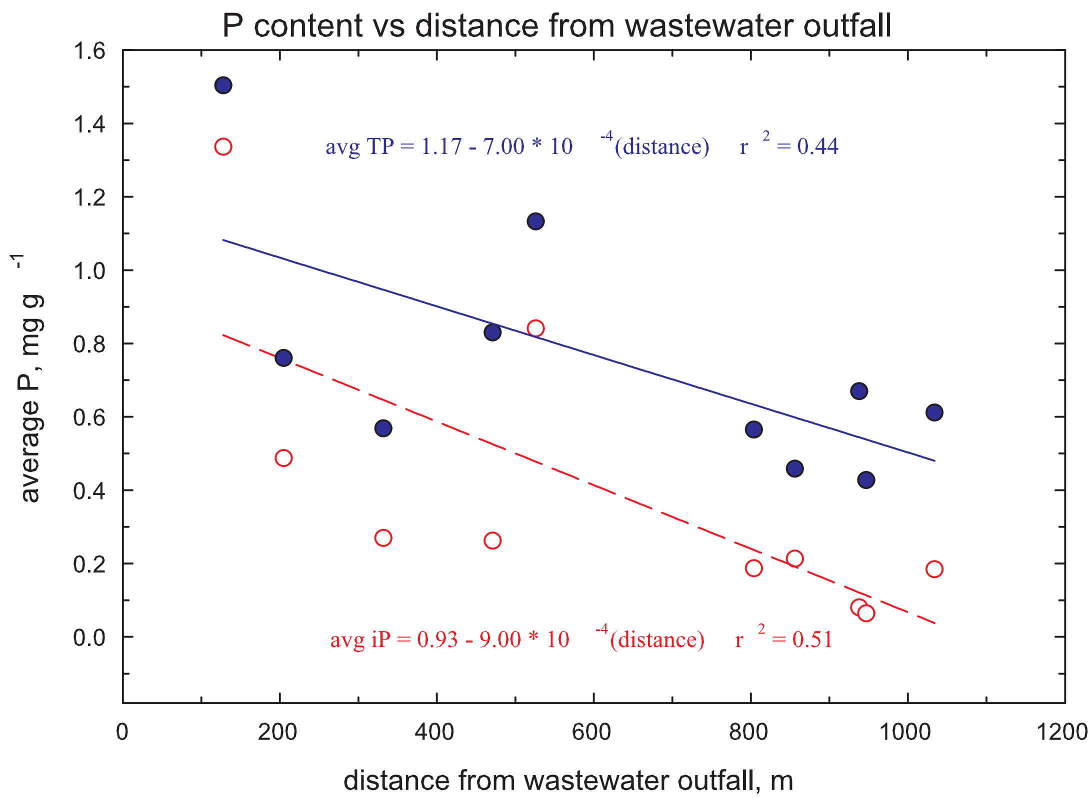

P, mg g -1
depth, cm
~40 b.p.
~80 b.p.
core F
core B
core J
core I
core E
core C
core A
core H
core G
core D
wastewater
outfall
Core
Locations
Choptank
River

Figure 5-9


P content vs distance from wastewater outfall
average P, mg g -1
distance from wastewater outfall, m
avg TP = 1.17 - 7.00 * 10 -4(distance) r 2 = 0.44
avg iP = 0.93 - 9.00 * 10 -4(distance) r 2 = 0.51
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
0
200
400
600
800
1000
1200

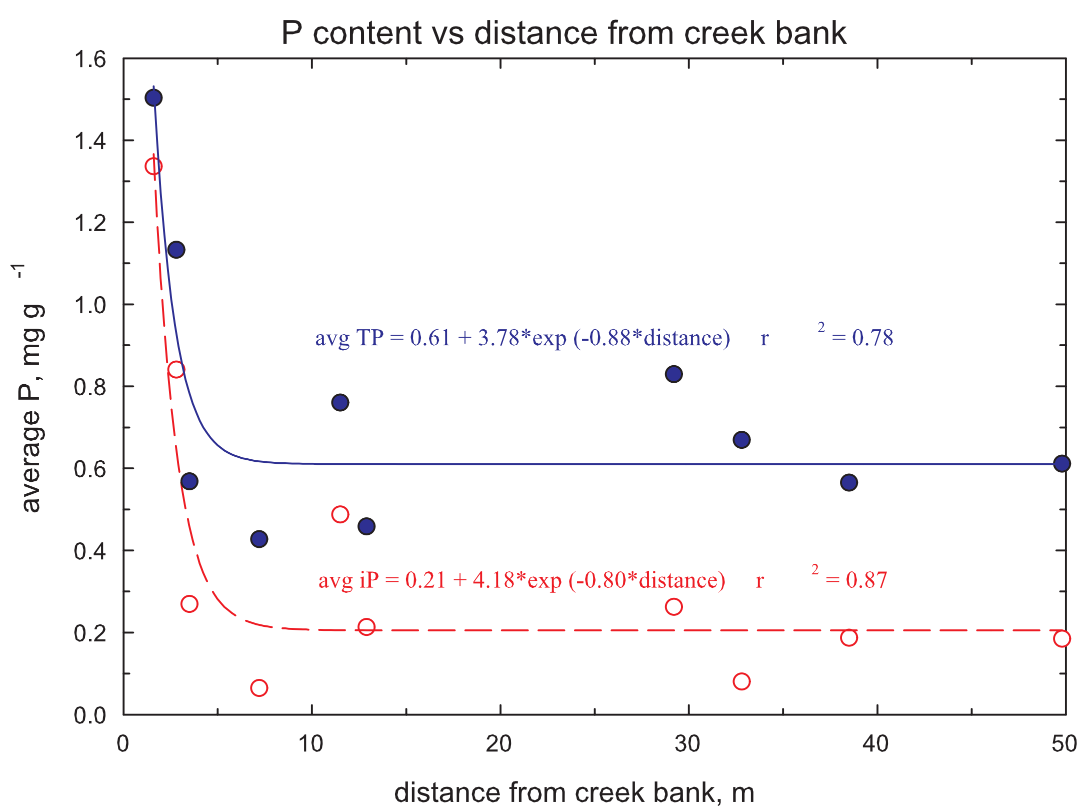
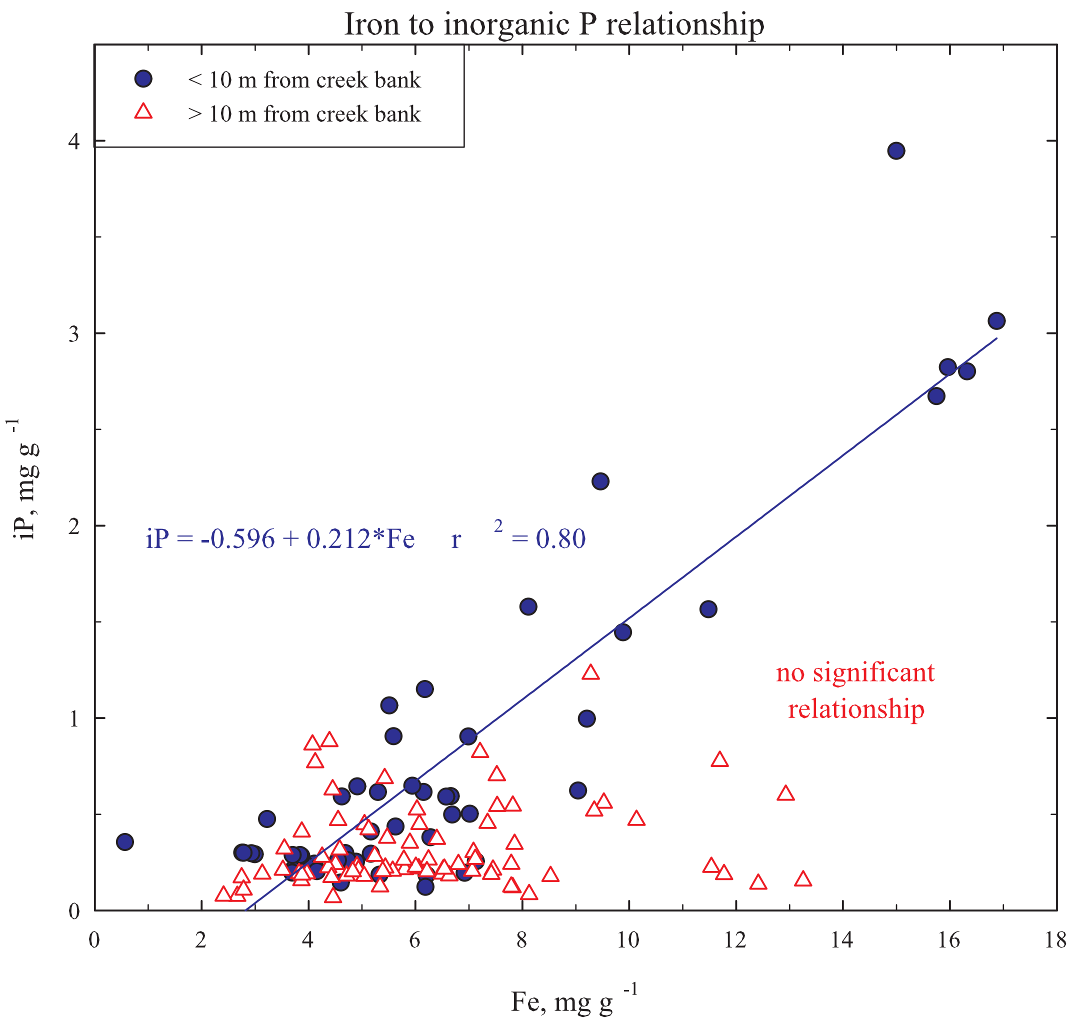

P content vs distance from creek bank
average P, mg g -1
distance from creek bank, m
avg TP = 0.61 + 3.78*exp (-0.88*distance) r 2 = 0.78
avg iP = 0.21 + 4.18*exp (-0.80*distance) r 2 = 0.87
1.6
1.4
1.2
1.0
0.8
0.6
0.4
0.2
0.0
0
10
20
30
40
50

Figure 5-10



Iron to inorganic P relationship
< 10 m from creek bank
> 10 m from creek bank
iP = -0.596 + 0.212*Fe r ^{2} = 0.80
no significant relationship
iP, mg g $^{-1}$
Fe, mg g $^{-1}$
0
1
2
3
4
0
2
4
6
8
10
12
14
16
18

Figure 5-11

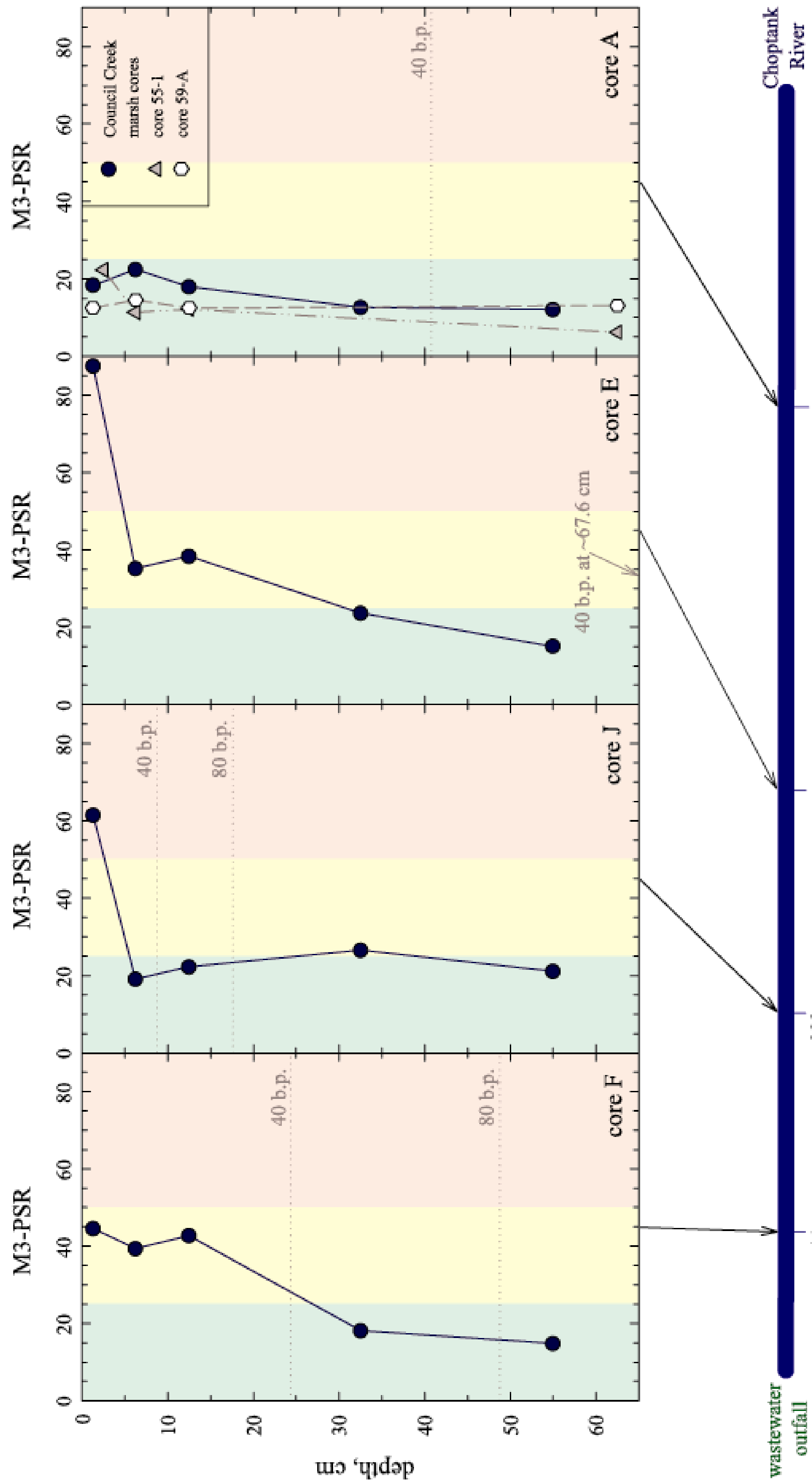

M3-PSR
core A
core E
core J
core F
Council Creek
marsh cores
core 55-1
core 59-A
40 b.p.
80 b.p.
40 b.p. at ~67.6 cm
depth, cm
Choptank River
856 m
526 m
332 m
128 m
wastewater outfall

Figure 5-12

Chapter 6

SYNTHESIS

The focus of this thesis was a comprehensive examination of the fate of wastewater as it passes through a natural oligohaline marsh. The physical influences of weather, tides, and wastewater discharge were investigated to aid in the interpretation of chemical and biological processing of wastewater N and P as it passed through Council Creek marsh. Marsh water quality was studied to determine the integrated effect of all marsh processes acting on wastewater. The marsh macrophyte study compared aboveground plant biomass, N and P content, and species richness in Council Creek marsh and Little Creek marsh, a marsh with no direct wastewater influence. In addition, Council Creek marsh's effectiveness at seasonal nutrient retention was considered. Finally, nutrient burial in sediments was investigated as mechanism of long-term N and P removal.

Weather, tides, and WWTP discharge were the main drivers of water temperature and water level in Council Creek marsh. Water temperature in Council Creek marsh exhibited rapid responses to these physical influences at several time scales. Rates of seasonal cooling and warming in marsh waters were similar to seasonal cooling and warming in air (Figure 2-8), and frontal passages were the major drivers of water temperature at weekly time scales (Figure 2-11). Water temperature at the inner station exhibited a cyclic diel pattern that is caused by solar heating and radiative cooling of the WWTP's overland flow system (Figure 2-12 panel A). Solar heating and radiative cooling of marsh water was clear at the outer station as well, but there was also an additional temperature influence from tidal exchanges with the more conservative thermal mass of the Choptank River that can result in temperature changes of up 10ºC in one tidal cycle (Figure 2-12 panel D). Seasonal (non-tidal) changes in mean water levels of approximately 30 cm resulted from thermal expansion of ocean waters during summer, with a minimum in winter (Figure 2-14). The augmentation of hydrologic inputs to Council Creek due to wastewater discharge significantly increased the hydraulic gradient across the marsh and reduced water retention time (Figure 2-17 top panel and 2-19).

Council Creek marsh appeared to function as an additional tertiary treatment component for wastewater N and a seasonal filter for wastewater P. Reductions in N concentrations of approximately 50% of N in wastewater effluent were seen during the height of the growing season due to dilution by Choptank River water during the seasonally higher tides (Figure 3-7 panel B). This dilution, plus the seasonally low values of N in wastewater due to effective removal in the overland flow system, results in TN concentrations being discharged to the Choptank River that were less than 2 times higher than average Choptank River concentrations. During winter, some of the marsh N was mobilized as organic material and exported to the Choptank (Figure 3-12). The pattern for P was not as clear based on the low tide sampling. The linear decrease in TP concentrations at the creek mouth suggested net gains in P concentrations early in the growing season, perhaps due to rising sea level, and little effect on wastewater P late in the growing season and into the fall (Figure 3-7 panel C). Furthermore, the transect data suggested that the TP concentrations are augmented by marsh P by late fall to early winter (Figure 3-13 panels A and B).

Wastewater discharged into Council Creek marsh has a moderate impact on marsh vegetation. There were no significant differences in species richness between

Council or Little Creek marsh, and species richness versus aboveground biomass data for both marshes exhibited a modal pattern with maximum species richness at intermediate community biomass (Figure 4-6). Within-marsh total community biomass comparisons revealed that the middle communities of both marshes generally immobilized more nutrients m^{-2} than both the inner and outer marsh communities, probably due to the growth of the high biomass *Phragmites australis* (Figure 4-7, 4-8, and top of Table 4-5). Although the reasons for the colonization of only certain areas by *Phragmites australis* were unclear here, a potential suggestion could be differences in sediment properties. However, expansion of *Phragmites australis* into the lateral fringes of the inner marsh community of Council Creek was visually noted over the study period and may be continuing.

Total community mass and elemental composition comparisons by species yielded evidence for several significant but small N and P enrichments of macrophytes in Council Creek marsh. Between marsh total community biomass comparisons revealed that only C:N ratio at the middle community of Little Creek marsh was significantly higher than that of Council Creek marsh, but there were no significant differences between the inner and outer communities of the two marshes (Figure 4-7 and 4-8). Comparisons of elemental composition of individual species yielded several significant differences. All of the species-specific elemental comparisons indicate that Council Creek marsh is enriched in N and P relative to Little Creek marsh (Figures 4-9, 4-10, and Table 4-5). As there was no evidence for nutrient limitation in either marsh (i.e., no significant differences in biomass), it seemed that luxury consumption of N and P by macrophytes in Council Creek marsh was occurring, due to the above evidence for N and

P enrichment, and in 11 of 70 tests (16%) macrophytic vegetation in Council Creek was N and P enriched (Figures 4-6 through 4-9 and Table 4-5). Annually, marsh macrophytes appeared to immobilize the equivalent of 12–18% of the wastewater N and 4.7–6.9% of the wastewater P in aboveground biomass, but during the growing season (May – Sept), they are potentially capable of intercepting ~30–44% of wastewater N and 11–17% of wastewater P. This calculation assumes that wastewater is the only N and P source and represents an upper limit of plant mobilization.

On an annual basis sediment N and P burial in Council Creek marsh was approximately equivalent to macrophyte uptake. Additionally, Council Creek marsh appeared to be burying significantly more N than other nearby Choptank River marshes. However, burial of iP and TP were not significantly different and oP burial was significantly lower in Council Creek marsh when compared with other Choptank marsh sediment data. Although there was no clear spatial pattern of N burial, the marsh is annually removing approximately 27% of the wastewater N. Phosphorus in sediments was highest near the wastewater discharge point and near the creek banks, annually burying approximately 5.9 % of the wastewater P. It also appears that marsh sediments near the wastewater discharge are becoming P saturated, increasing the likelihood of P release back into creek water (Figure 5-12 and Figure 3-15). This was, in fact, observed when wastewater flows stopped for plant management reasons (Figure 3-15). It therefore appears that Council Creek marsh has little capacity to remove additional P from the wastewater flowing through it, although denitrification is likely as suggested by Figure 3-14.

Overall, Council Creek marsh seems to be little affected by, and has a relatively small impact on, the wastewater stream flowing through it (Figure 6-1, Table 6-1). Physically, the additional water flow from wastewater raises mean water levels by as much as 3 cm, in addition to normal seasonal changes of 30 cm induced by thermal expansion and contraction of seawater, and solar heating of the wastewater causes distinct diel temperature oscillations. Floristically, the Council Creek marsh plant community is statistically indistinguishable from an adjacent marsh with no direct wastewater inputs, although there was some evidence for nutrient-enriched compositional ratios of some plants, perhaps as a result of luxury uptake. The substantial accumulation of plant biomass in the marsh during the growing season is equivalent to ~40% of the wastewater N and ~15 % of the wastewater P; however, some of the summer accumulation of organic N and P appeared to be exported during winter. Nutrients passing through the marsh experienced some transformations and losses, especially in summer (coupled nitrification/denitrification, PO_4 sorption/desorption, etc.), but the magnitude of these transformations was small (~40% N and ~25% P losses) compared to the dilution of the wastewater in the marsh by Choptank River water (>50%). During summer high sea level stands, wastewater exiting the marsh at low tide is diluted 25–75% by Choptank River water present in the marsh from previous high tides. Although some biological processing of the wastewater N and P occurs during the ~3 d flushing time of water in the marsh, dilution of the wastewater appears to be the dominant process. Finally, burial of N and P is the only long-term loss quantified here, and 27% of the N and 6 % of the P appear to be removed annually.

Table 6-1: Important physical, chemical, and biological process identified or inferred as important in Council Creek marsh. This list emphasizes interactions between normal marsh processes and the presence of high N and P wastewater.

Physical	Chemical	Biological
- raised water level	- summer storage of inorganic N and P in plant biomass	- nutrient-enriched compositional ratios in biomass
- reduced retention time	- winter loss of organic N and P	- inhibition of algal blooms
- diel temperature variations	- saturation of Fe binding sites for PO_4 in sediments	- coupled nitrification/denitrification of wastewater N
- summer dilution of low tide discharges	- burial of N and P	
- increased water turbidity		

FIGURE LEGENDS

Figure 6-1: Synthesis: Top panel: summary of processes occurring in Council Creek marsh during summer (growing season), bottom panel: summary of processes occurring in Council Creek marsh during winter. Arrows indicate the direction of N or P movement, and percentages indicate the percent of wastewater N and P.

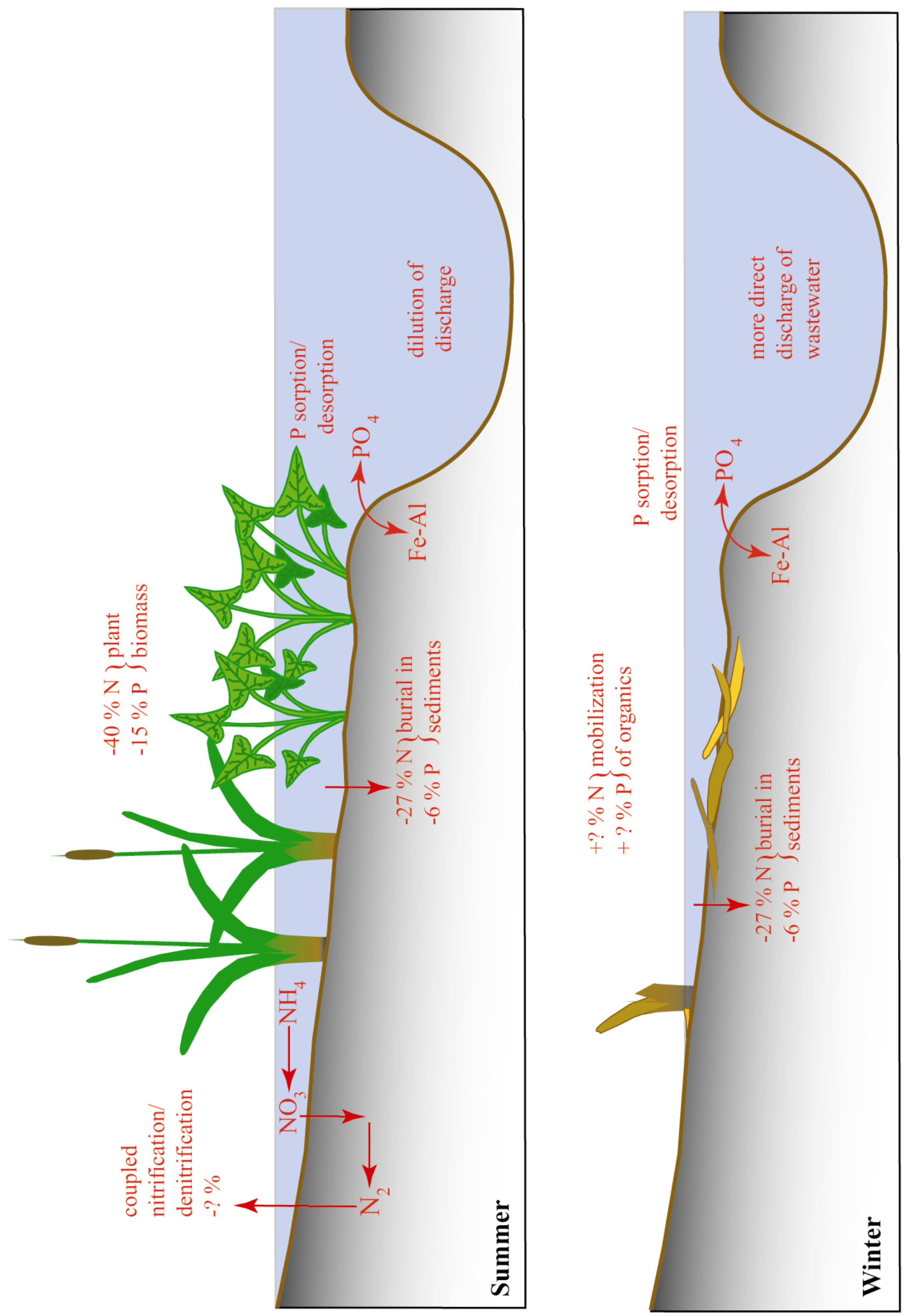

coupled nitrification/ denitrification -? %
NO_3
NH_4
N_2
-40 % N
-15 % P
plant biomass
-27 % N
-6 % P
burial in sediments
P sorption/ desorption
PO_4
Fe-Al
dilution of discharge
Summer
+? % N
+ ? % P
mobilization of organics
-27 % N
-6 % P
burial in sediments
P sorption/ desorption
PO_4
Fe-Al
more direct discharge of wastewater
Winter

Figure 6-1

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