ALTERATIONS IN ECOSYSTEM PROCESSES AS A RESULT OF ANTHROPOGENIC MODIFICATIONS TO STREAMS AND THEIR CATCHMENTS

by

CATHERINE A. GIBSON

(Under the direction of Judith L. Meyer)

ABSTRACT

Ecosystem services are the benefits human societies derive from ecosystem processes. Freshwater ecosystems provide many ecosystem services including clean water and waste assimilation. However, humans are altering freshwater ecosystems through practices such as hydrologic alteration and nutrient loading. The objective of this dissertation is to examine how anthropogenic alterations of streams and/or their catchments affect ecosystem processes and concomitantly the ability of streams to provide ecosystem services. I focus on effects of three human modifications: climate change, dam operation, and urbanization. Future climate scenarios can cause dramatic shifts in flow regimes in the Cle Elum and the Apalachicola-Chattahoochee-Flint River systems. These flow regime changes would result in decreased habitat availability and changes in food resources and demonstrate that these types of analysis are a starting point four understanding the ecological implications of future climate scenarios. The Chattahoochee River below Atlanta, GA is regulated by hydropower dams and receives most of Atlanta's wastewater treatment plant effluent. Processing of the effluent-derived fine particulate organic matter (FPOM) and nutrients is essential for maintaining downstream water quality. Hydropeaking reduces gross primary production. FPOM and phosphorus (P) loading increases respiration and overwhelms the effects of low flow discharge regime on respiration. Nutrient retention by the river is sporadic and temporary. P retention is controlled by sediment absorption, but retention is temporary as P is readily released from sediments. The study reaches are more likely to be sources of ammonium than sinks. Nitrate uptake is controlled by biological uptake associated with suspended FPOM. Overall, P and ammonium uptake velocities in the Chattahoochee River are lower than those in less modified systems. Urban headwater streams had lower uptake velocities and higher ambient nutrient concentrations than forest streams in the Etowah Basin, GA. P uptake is controlled chemical sorption to sediments. Ammonium uptake is controlled by biological processes. Conserving functioning riparian areas and channels in a conservation subdivison reduces some urbanization effects and maintains nutrient retention. Human alterations have diminished the capacity of these rivers and streams to provide ecosystem services, but reducing hydropeaking, decreasing nutrient loading, and maintaining riparian areas and stream channels could increase the ability of urban streams and rivers to provide the services that human communities value.

INDEX WORDS: Ecosystem services, Nutrient uptake, Ecosystem metabolism, Climate change, Flow regime, Urbanization, Streams

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Ecosystem services represent the benefits human societies derive, directly or indirectly, from ecosystem processes (Costanza et al. 1997, Daily 1997). Clean water, waste assimilation, and transportation are just a few of the services freshwater ecosystems provide to humans (Naiman and Turner 2000, Jackson et al. 2001, Baron et al. 2002). However, the demands for freshwater and the services it provides are increasing as a result of increased human population and changes to land-use and land cover (Postel et al. 1996, Naiman and Turner 2000). Currently many freshwater ecosystems are becoming degraded through such practices as hydrologic alteration and nutrient loading, and it is uncertain how this degradation will alter the ability of the ecosystem to function and provide these services (Benke 1990, Poff et al. 1997, Naiman and Turner 2000).

Human land-use alters the structure and functioning of ecosystems and the interactions between ecosystems (Vitousek et al. 1997). For instance, human population density within the catchment accounts for 76% of the variation in nitrate concentration in 42 rivers of the world (Peierls et al. 1991). In addition, nitrate export from 35 of the world's rivers is highly correlated with point-source inputs of nitrogen and human induced nitrogen loading in the watershed in the forms of fertilizer and N-deposition (Caraco and Cole 1999). Similarly, a simple model involving point and non-point source inputs of phosphorus explains 72% of the variation in soluble reactive phosphorus export of 32 of the world's rivers (Caraco 1995). In addition, streams in urban environments typically have higher nutrient and contaminant concentrations, flashier hydrographs, and decreased species richness (Paul and Meyer 2001). Although there is evidence for changes in freshwater ecosystems such as increased nutrient concentrations and changes in hydrology and species composition as a result of human modifications, there is less evidence regarding how these changes affect ecosystem functions such as nutrient uptake or metabolism. The objective of this dissertation is to examine how anthropogenic alterations of streams and/or their catchments affect ecosystem processes and concomitantly the ability of streams to provide ecosystem services.

Humans have modified the landscape in a variety of ways. Streams and rivers have been directly impacted by some of these alterations including dam construction, water diversion for agriculture, industrial, and municipal use, eutrophication as a result of excess nutrient loading from point and non-point source inputs, channelization, and burial or piping of stream systems. In addition, because freshwater systems are tightly linked with the surrounding terrestrial systems, streams and rivers are impacted by changes to the terrestrial landscape including land use change, creation of impervious surfaces, and removal of riparian zones. Therefore, I examine the impact of three human modifications, climate change, dam operation, and urbanization, on stream ecosystem processes by asking the following questions:

1. Do future climate scenarios result in changes in flow regime in two river basins (Cle Elum, WA and Apalachicola-Chattahoochee, AL, GA, FL), are these different from changes in flow regime caused by dams, and how would these changes affect ecosystem processes and biota in rivers? (Chapter 2)
2. Can flow regime explain the variation in ecosystem metabolism in the Chattahoochee River near Atlanta, GA, or are other physical, chemical, and biological factors needed? (Chapter 3)

3. Are nutrients from Atlanta wastewater treatment plants being retained by the Chattahoochee River, and what environmental parameters control nutrient uptake? (Chapter 4)
4. How does urbanization affect nutrient uptake in headwater streams in the Upper Etowah River Basin, GA? (Chapter 5)

Flow regime alterations affect the ecological integrity of river systems (Poff et al. 1997). Therefore, climatic changes that create shifts in the timing and magnitude of low or high flow events or change the magnitude of river flow at monthly, seasonal, or yearly time scales could result in dramatically altered river systems. Numerous studies have examined the potential impacts of future climate scenarios on aquatic ecosystems on a regional scale (Hauer et al. 1997, Melack et al. 1997, Mulholland et al. 1997). These studies give insight into what might be expected for aquatic ecosystems including changes in species ranges, nutrient delivery, temperatures and hydrology. Recently, there have been efforts to examine the potential impacts of climate change on hydrology at the watershed scale (Band et al. 1996, Poff et al. 1996, Wilby et al. 1999, Hay et al. 2000). In Chapter 2, I utilize available wastershed-scale atmospheric-hydrologic models to examine potential changes in flow regime and the implications of these changes for the riverine ecosystem in two river systems, the Cle Elum River, WA and the Apalachicola-Chattahoochee Rivers, GA, AL, and FL. This analysis is a starting point for understanding the ecological implications of future climate scenarios for riverine ecosystems at the watershed scale. I determine the potential impacts of future climate scenarios on flow regime in two study sites and compare these changes with what has been caused by dams in the Chattahoochee River. Then I examine the implications of these changes on fishes, macroinvertebrates, organic matter transport and storage, and riparian species.

Dams are one of the most prominent and pervasive forms of flow alteration on large and some smaller rivers and streams (Dynesius and Nilsson 1994). In the contiguous United States, there are only 42 rivers with greater than 200 river kilometers unregulated by major dams (Benke 1990). A number of studies have documented the impacts of dams on channel morphology (Fenner et al. 1985, Ligon et al. 1995, Stevens et al. 1995), fish (Freeman et al. 2001, Sparks 1995), habitat availability (Freeman et al. 2001, Pringle et al. 2000, Sparks 1995), and riparian species survival and recruitment (Jannson et al. 2000, Rood et al. 1995), but less is known about the impact of dams and flow regime on basic ecosystem processes such as nutrient uptake and metabolism. Metabolism has been shown to vary with high stream discharge in unregulated rivers as a result of bed-moving spates reducing both primary production and community respiration (Uehlinger 2000). Hydropeaking causes some regulated rivers to experience almost daily discharge spates that may influence both primary production and community respiration.

In Chapter 3, I examine whether the variation in net ecosystem metabolism (NEM), gross primary production (GPP), or respiration (R) in a regulated urban river is related to the variation in low flow discharge or if metabolism variation is better explained by other physical, chemical, or biological parameters. I examined this question in the Chattahoochee River downstream of Atlanta, GA. This river is regulated by both a large hydropower dam and a smaller re-regulation dam, plus it receives large amounts of point source inputs from Atlanta. Point source inputs from waste water treatment plants contain sewage-derived particulate organic matter (POM) which is a labile mixture of organic detritus and microorganisms such as bacteria and algae (deBruyn and Rasmussen 2002). While advances in technology have reduced the organic matter loading so that it typically does not cause oxygen depletion in rivers, the processing of this organic matter is critical to maintaining downstream water quality. Therefore, organic matter processing is a critical ecosystem service that is performed by the river and understanding what physical, biological, and chemical factors influence this service is important.

In rivers, the transport of nutrients is tightly linked with the physical movement of water. In flowing waters, nutrient cycles are longitudinally extended to become spirals (Webster and Ehrman 1996). The length of the spiral is primarily determined by uptake length, which is strongly correlated with stream velocity (Newbold et al. 1983, Meyer and Edwards 1990, Webster and Ehrman 1996). Therefore, flow can influence the movement of nutrients in a system, and changes in flow could affect the transport of nutrients to downstream ecosystems. Uptake in streams alters the form and amount of nutrients transported from terrestrial ecosystems to downstream ecosystems (Stream Solute Workshop 1990, Alexander et al. 2000). Small streams are efficient in retaining nutrients (Peterson et al. 2001). However, small streams receiving point-source inputs (such as waste water treatment plants) are typically less efficient in retaining both nitrogen and phosphorus inputs (Marti et al. 2004, Haggard et al. 2001, Pollock and Meyer 2001). Urban rivers, such as the Chattahoochee, typically receive large amounts of point-source inputs which they are depended upon to retain and transform in order to maintain downstream water quality. Empirical measures of nutrient uptake in medium and large rivers are scarce, and, therefore, there is little information on the ability of these larger systems to perform these services. Therefore, in Chapter 4, I quantify nutrient uptake in the Chattahoochee River to determine how it varies with discharge and other environmental parameters and to determine the level of this ecosystem service provided by the river.

Headwater streams are linked with both the terrestrial landscape and downstream ecosystems (Meyer and Wallace 2001, Gomi et al. 2002). With urbanization, many headwater streams are lost or significantly altered, as a result of channelization or piping (Meyer and Wallace 2001).

As stated earlier, streams in urban areas have flashy hydrographs, altered channel morphologies, increased nutrient and contaminant loads, and decreased species richness (Paul and Meyer 2001). However, there is less information about how urbanization and the concomitant changes to urban streams affect ecosystem processes such as nutrient uptake in these small streams. Previous research in 3rd and 4th order streams has shown that demand for nutrients was lower in urban streams compared to similarly sized forested streams (Meyer et al. submitted). However, material transport in headwater streams is different from that of larger network stream systems (Gomi et al. 2002). Therefore, it is important to determine how urbanization affects nutrient uptake in headwater streams and if it is similar to the effects seen in larger stream systems.

In Chapter 5, I examine the impacts of urbanization on nutrient uptake in headwaters streams by measuring nutrient uptake length and velocity in four forested and four urban streams during summer in both a wet and dry year. The urban streams are located within the city limits of two Atlanta suburbs and drain a mix of residential and industrial land-uses. One of the urban streams is located in a conservation subdivision. Thus, I also examine how different development strategies affected nutrient uptake in urban streams. Population in Atlanta has increased 97% from 1970-1995, and the amount of land occupied by urban areas has more than tripled (Frick et al. 1998). Therefore, determining how urbanization has affected nutrient uptake and if different types of development can buffer some of the effects of urbanization is necessary to understanding how downstream systems will be affected and what management strategies might help maintain water quality in these systems.

Freshwater ecosystems provide a variety of crucial resources. Therefore, the maintenance of functioning freshwater ecosystems is important. However, humans are altering these systems by altering flow regimes, nutrient loads, and drainage networks. The examination of how these alterations affect ecosystem processes should provide insight into the resiliency of freshwater ecosystems and management strategies to lessen the impacts of these human alterations.

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CHAPTER 2

FLOW REGIME ALTERATIONS UNDER CHANGING CLIMATE IN TWO RIVER BASINS: IMPLICATIONS FOR FRESHWATER ECOSYSTEMS[1]

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[1] Gibson, C.A., J.L. Meyer, N.L. Poff, L.E. Hay and A. Georgakakos. Submitted to River Research and Applications 4/10/03.

Abstract

We examined impacts of future climate scenarios on flow regimes and how predicted changes might affect river ecosystems. We examined two case studies: Cle Elum River, Washington and Chattahoochee-Apalachicola River Basin, Georgia and Florida. These rivers had available downscaled global circulation models (GCM) data and allowed us to analyze the effects of future climate scenarios on rivers with 1) different hydrographs, 2) high future water demands, and 3) a river-floodplain system. We compared observed flow regimes to those predicted under future climate scenarios to describe the extent and type of changes predicted to occur. Daily stream flow under future climate scenarios was created by either statistically downscaling GCMs (Cle Elum) or creating a regression model between climatological parameters predicted from GCMs and stream flow (Chattahoochee-Apalachicola). Flow regimes were examined for changes from current conditions with respect to ecologically relevant features including the magnitude and timing of minimum and maximum flows. The Cle Elum's hydrograph under future climate scenarios showed a dramatic shift in the timing of peak flows and lower low flow of a longer duration. These changes could mean higher summer water temperatures, lower summer dissolved oxygen, and reduced survival of larval fishes. The Chattahoochee-Apalachicola basin is heavily impacted by dams and water withdrawals for human consumption; therefore, we made comparisons between pre- large dam conditions, current conditions, current conditions with future demand, and future climate scenarios with future demand to separate climate change effects and other anthropogenic impacts. Dam construction, future climate, and future demand decreased the flow variability of the river. In addition, minimum flows were lower under future climate scenarios. These changes could decrease the connectivity of the channel and the floodplain, decrease habitat availability, and

potentially lower the ability of the river to assimilate wastewater treatment plant effluent. Our study illustrates the types of changes that river ecosystems might experience under future climates.

Introduction

The quantity and timing of river flow are critical components to water supply, water quality, and the ecological integrity of river systems (Poff et al., 1997). The five components of flow regime (magnitude, frequency, duration, timing and rate of change) influence the ecological dynamics of river systems directly and indirectly through their effects on other primary regulators (Karr, 1991; Poff et al., 1997). Alteration of any of these flow parameters can have dramatic effects on aquatic organisms, riparian species, energy flow in the system, sediment movement and floodplain interactions (Poff et al., 1997 and references therein). Hence, climatic changes that create shifts in the timing and magnitude of low or high flow events or change the magnitude of river flow at monthly, seasonal, or yearly time scales could result in dramatically altered river systems.

There have been numerous studies that have examined the potential impacts of future climate scenarios on aquatic ecosystems on a regional scale (e.g. Hauer et al., 1997; Melack et al., 1997; Moore et al., 1997; Mulholland et al., 1997; Meyer et al., 1999; Stone et al., 2001). These studies give insight to the type of changes that might be expected for aquatic ecosystems and their inhabitants under future climate scenarios including: changes in species ranges, nutrient delivery, temperatures, hydrology, and mixing regimes of lakes. In addition, many of these studies have noted that climate change is not occurring in a static landscape, but rather impacts of climate change and human disturbance are likely to interactively affect freshwater ecosystems (Moore et al., 1997; Meyer et al., 1999; Fox et al., 2001; Stone et al., 2001). Due to these interactions, some basins may be more sensitive to climate change than others. This sensitivity is partly a result of the magnitude and type of human disturbance that is occurring within the basin. For instance, regulated rivers in areas where future climate scenarios predict earlier snowmelt and more winter rain may not experience a shift in the timing of high flows compared to naturally flowing rivers in the area. However, in areas where future climate scenarios predict drier conditions for parts of the year, rivers that are used for water supply could experience even lower discharges, and thus potentially be more sensitive to climate change than rivers in the area that are not as heavily utilized for human water consumption.

Although large scale warming and cooling trends of the Earth occur naturally, current human-mediated climate changes have reached or exceeded the level of these natural fluctuations (Levine, 1992; Vitousek, 1992; Schindler et al., 1996). The fluctuations in the dynamics of the atmosphere and oceans are potentially of a magnitude to constitute global change. However, it is uncertain what these global changes will mean to the Earth's climate, especially at small spatial scales. The most commonly used tools to predict future climate conditions are Global Circulation Models (GCMs). By definition, these models are coarsely scaled, and their ability to predict localized changes in climate is questionable. Recently there have been efforts to examine the potential impacts of climate change on watershed hydrology by statistically downscaling these GCM models. Downscaling includes using correlations between synoptic regional scale climatic variables and local climate to simulate weather at a local scale (Giorgi et al., 1994; Wilby and Wigley, 1997; Wilby et al., 1999; Hay et al., 2000). Another approach uses smaller scale models to make watershed level predictions of the potential impacts of climate change on freshwater systems (Leavesley, 1994; Poff et al., 1996; Band et al., 1996; Clair and Ehrman, 1996; Wilby and Wigley, 1997; Wilby et al.,1999).

The objective of this study is to examine the potential impacts of future climate scenarios on river ecosystems at the watershed scale. We utilize available atmospheric-hydrologic models to examine potential changes in flow regime and then discuss the implications of these changes for the riverine ecosystem. We view this analysis as a starting point to understanding the ecological implications of future climate scenarios for riverine ecosystems at the watershed scale. We selected rivers with available scaled-down GCM data that adequately predicted observed data and that allowed us to analyze the effects of future climate scenarios on 1) two different hydrographs (snow+ rain, and rainfall), 2) a system with high future water demands, and 3) a floodplain river with available data on the flows needed to connect the floodplain and the channel. The two case study systems in which we examined the changes in flow regime under future climate scenarios are the Cle Elum River, Washington and the Chattahoochee-Apalachicola River Basin in Georgia and Florida. In addition, for the Chattahoochee-Apalachicola Basin we placed the potential changes in flow regime that result from future climatic scenarios in perspective by comparing the changes in the flow regime caused by direct human appropriation (e.g. dam construction) to those predicted by future climatic scenarios.

These two case studies should help illustrate the implications of future climate scenarios for river ecosystems in the Pacific Northwest and in the Southeastern United States and explore the range of potential changes in flow regime. Our study was limited by data availability, and different GCMs and different statistical downscaling techniques will be used for the two case studies. These differences enhance the range of possibilities that could occur under future climate scenarios.

Methods

We evaluated current and predicted future flow regimes using Richter *et al.*'s (1996) Indicators of Hydrologic Alteration (IHA). IHA was designed to examine changes in flow regimes caused by dams, but its ability to compare flow regimes makes it an ideal tool for this study. This program takes daily streamflow values and characterizes flow regime in terms of five ecologically significant factors: magnitude, duration, frequency, timing, and rate of change. By dividing the streamflow record into a pre- and post-impact period, the program calculates the percent change in several ecologically relevant statistics including: magnitude of monthly water conditions, magnitude and duration of annual extreme water conditions, timing of annual extreme water conditions, and frequency and duration of high and low pulses.

The IHA was designed primarily to test for differences in flow regime in distinct pre- and post-impact periods (e.g. as caused by dams). Because, we were assessing the more subtle impact of climate on flow regime, we decided to focus on a specific subset of the 32 statistics calculated by the program that were more likely to be sensitive to climate change. Specifically, we focused on mean monthly discharge for each month, magnitude of the 1, 3, 7, 30 and 90 –day maximum and minimum flows, and timing of the 1-day maximum and minimum flows. Future climate scenario data for the Chattahoochee-Apalachicola basin were calculated on a weekly time scale, thus we could only calculate the 7, 30, and 90-day maximum and minimum flows and we were not capable of calculating the timing of the 1-day maximum or minimum flows (see below). There are several other parameters calculated by IHA including rise rate, fall rate, number of reversals, number of low and high flow pulses, and duration of low and high flow pulses. However, these parameters were either unlikely to be affected by climate change (rise rate, fall rate, and number of reversals) or would not be well predicted by statistically downscaled GCM models (number of low and high flow pulses, duration of low and high flow pulses). Therefore, they were excluded from the analysis. We assessed the inter-annual variability of monthly flows by calculating the coefficient of variation for each month over the period of record or the period modeled.

After quantifying potential changes in flow regime as a result of climate change, we examined implications of these changes on stream functioning and the organisms living within and adjacent to the stream. Specifically, we evaluated potential implications for resident fish species, macroinvertebrates, and riparian plant species.

Case Study 1 – Cle Elum River

The Cle Elum River has a drainage area of 525 km^2 at the gauging station near Rosyln, WA (USGS site 12479000). The one large mainstem reservoir, Cle Elum Lake, in the drainage is managed to supplement flows from March to August for irrigation (Beckman et al., 2000). However, the flows we are examining have been naturalized to remove the effects of the dam (L.E. Hay personal communication). By looking at the flow regime without the influence of dams, the analysis gives a better illustration of the potential implications throughout the Pacific Northwest rather than just a particular river. Precipitation at Cle Elum, WA, slightly downstream of the Cle Elum River and Yakima River confluence, averages 56.62 cm per year. The river drains the Wenatchee Mountains, which are heavily forested in spruce (*Picea engelmannii*) and Douglas fir (*Pseudotsuga menziesii*) (Carter et al., 1996). This perennial river has a moderately seasonal runoff pattern characterized by a late spring (April – June) high flow period driven by snowmelt and a summer low flow period. The snowmelt signature, along with moderate values for flow predictability, flood predictability, and length of flood free season, suggests a Rain + Snow type stream (Poff and Ward, 1989; Poff, 1996). Rain + Snow streams are typically western montane streams that are influenced by both annual snowfall and winter rainfall patterns (Poff and Ward, 1989). Therefore, they are likely to have the characteristic snowmelt peak in spring, but warmer winters with heavy precipitation can lead to rain-on snow events that cause peak flows.

In order to assess the impact of climate change on flow regime, we conducted IHA analysis on three sets of daily stream flow values. First, we calculated IHA hydrologic parameters for observed daily stream flow from 1980-1995. Second, we calculated hydrologic conditions for the flow predicted by statistically downscaled GCMs for current climate 1980-1995. This analysis was used to test the sensitivity of statistically downscaled GCMs in predicting known daily streamflow records. Third, we calculated hydrologic parameters for the future flows predicted by statistically-downscaled GCMs for 2080-2095. We ran 20 simulations for each of the two statistically downscaled data sets (validation: 1980-1995 and future: 2080-2095). Coefficient of variation for observed data (1980-1995) was calculated for each month. For the two data sets resulting from model predictions, we first calculated an inter-annual coefficient of variation for each month for each simulation and then calculated the mean and standard error of the coefficient of variation from the 20 simulations.

For future climate scenarios, we used the U.K. Meteorological Office, Hadley Centre's coupled ocean/atmosphere model (HadCM2) forced by changes in combined CO_2 and a sulfate aerosol proxy (Johns et al. 1997; Mitchell and Johns, 1997; Wilby et al., 1999). In this model, CO_2 levels increase by 1% a year from 1990 to 2100 (Mitchell and Johns, 1997). For current climate modeling, we ran the same climate model but with observed values of the parameters, such as CO_2 (Wilby et al.,1999). For both climate models, statistical downscaling and hydrologic modeling were performed as described by Wilby et al. (1999) and Hay et al. (2000).

Briefly, with this approach daily temperature and precipitation from the downscaled GCM output were spatially distributed over the basin using monthly lapse rates calculated from observed data (Wilby et al., 1999). These spatially distributed temperature and precipitation data were then put into the PRMS (Precipitation-Runoff Modeling System) watershed model, which modeled daily stream flows (Leavesly et al., 1983; Leavesly and Stannard, 1995; Wilby et al., 1999; Hay et al., 2000).

Due to inherent variability associated with models, we ran 20 simulations of the downscaling model for both current and predicted future climate scenarios. Each simulation was independent, and we analyzed the flow regime resulting from each simulation separately using IHA. For each of the two statistically downscaled model types based on current or predicted future climate, we calculated the mean and standard deviation of each IHA hydrologic parameter to characterize model variability. To evaluate the impact of climate on flow regime, we compared the parameters calculated from the observed data (1980-1995) to parameters calculated from the GCMs with predicted future climate (2080-2095). To assess the ability of statistically downscaled GCMs to predict daily stream flow, we compared parameters calculated from the observed data (1980-1995) to those calculated from the statistically downscaled GCMs with current climate (1980-1995). The models were not capable of precisely replicating the observed data, so we attempted to determine the extent to which the model accurately represented the observed data. We defined a significant change in a parameter as differing from observed data by greater than 2 standard errors.

To differentiate between changes in flow regime resulting from modeling inefficiencies and changes resulting from predicted future climate scenarios, we performed similar comparisons between statistically downscaled GCMs with current climate and statistically downscaled GCMs with future climate scenarios. For instance, if modeled flow regimes based on current climate conditions resulted in values of a parameter that were higher than for the observed flow regime, but the modeled future climate scenario predicted a decrease in the same parameter, we had confidence that the change was a consequence of the future climate scenarios and not an artifact of the statistically downscaled model. However, if the magnitude and direction of changes in a flow regime parameter were similar between modeled current conditions and modeled future climate scenarios, then we concluded that there was not a detectable effect of climate change on that parameter.

The downscaled GCM under current climate conditions (1980-1995) captured the observed seasonal pattern of discharge, but underestimated magnitude of the flow from spring snowmelt (April – June) and overestimated fall and winter (October -February) low flows (Figure 2.1). The model significantly overestimated the 1, 3, and 7-day maxima (Figure 2.2) and the 1 and 90-day minima (Figure 2.2). The model was fairly accurate in predicting the timing of the 1-day minimum and maximum. In terms of inter-annual variability, the model slightly overestimated variability from January – May, was much higher than what was observed in September, and underestimated variability in October – December (Figure 2.3).

The hydrograph under predicted climate for 2080-2095 differed markedly from both the current conditions and the modeled current conditions. In addition, the changes in the hydrograph under predicted future climate were dramatically different from the changes attributable to model inadequacies in predicting current hydrographs. Peak monthly flow shifted from late spring / early summer (April – June) to winter (December – February) indicating a shift from precipitation falling primarily as snow and being stored, to precipitation falling as rain and immediately running off (Figure 2.1). Spring and summer flows were lower, and the length of summer low flow period was extended in the 2080-2095 hydrograph compared to observed conditions (Figure 2.1). Timing of the 1 day minimum was about 1 month later under future climate scenarios (Sept 28) compared to observed conditions (Aug 25), but the 1 day maximum was almost 6 months earlier under future climate scenarios (January 4) compared to observed conditions (May 28). In addition, the flow regime under predicted future climate scenarios will have an extended period of time with no floods (203 days) compared to observed (82 days) or modeled (81 days) current conditions.

There were no significant changes in the magnitude of the 1, 3, 7, 30, or 90-day maxima under predicted future climate compared to modeled current conditions (Figure 2.2). This suggests that, on average, the magnitude and duration of yearly floods did not differ between current conditions and those predicted under future climate scenarios. However, the magnitude of the 1, 3, 7, 30, and 90-day minima were all much lower under predicted future climatic scenarios than under current conditions (Figure 2.2). This suggests that, on average, the lowest water flows will be lower and low water periods will last longer under predicted future climatic scenarios. This is consistent with the monthly flow data, which predicted an extended summer low flow period and lower summer flows under simulated future climate scenarios (Figure 2.1).

We assessed variability by calculating the coefficient of variation of mean monthly discharge over the 15-year timespan. Thus, the coefficient of variation was a measure of inter-annual variation. For the most part, the predicted future coefficient of variation was similar or less than the current coefficient of variation (Figure 2.3). The exception was in September when the coefficient of variation was much higher under future climatic scenarios. However, it is likely that this large coefficient of variation is a consequence of the model because the coefficient of variation was also much higher in September in the model derived from current climate conditions. Downscaled GCMs are much better at predicting precipitation when large weather systems are dominant than when smaller convective systems are responsible for the majority of precipitation events, such as in the late summer (Wilby *et al*., 1999). This model artifact may have artificially inflated the coefficient of variation in both model outputs (Figure 2.3). In addition, the lowest mean flows occured in September, so if there was high variability due to the limited capabilities of the model to predict the small storms that dominate late summer precipitation, this would also lead to an inflation of the coefficient of variation.

Flow regime is directly related to several factors that affect biological communities and ecological processes in river systems (see Poff et al., 1997). For example, lower summer flows can lead to increases in water temperatures and reduced dissolved oxygen. Lower flows also indicate a reduced wetted perimeter, which would decrease habitat availability and impact lateral exchanges between the riparian zone and the stream. A shift in the timing of peak flow can alter the retention time of organic matter (Mulholland et al., 1997), disrupt the recruitment of riparian species that rely on appropriately timed high flows to disperse seeds on to the floodplain (Auble et al., 1994; Rood et al., 1995), and impact the survival of certain fish species whose larval emergence is timed to avoid high spring flows (Seegrist and Gard, 1972; Erman et al., 1988; Hauer et al., 1997).

Although we do not have biological data specific to the Cle Elum River, Cuffney *et al*. (1997) sampled several watersheds surrounding the Cle Elum that are also located in the Cascades Ecoregion and are tributaries of the Yakima River. For fishes, Cuffney et al. (1997) found that high elevation, cold-water streams with little agricultural influence, such as the Cle Elum, were typically populated by salmonids and sculpins (*Cottus* spp.). The salmonids included dolly varden (*Salvelinus malma*), non-native brook trout (*S. fontalis*), cutthroat trout

(*Oncorhynchus clarki*), coho salmon (*O. kisutch*), spring chinook salmon (*O. tshawytscha*) and rainbow trout (*O. mykiss*). All of these fishes are coldwater species that typically inhabit areas where summer water temperatures usually do not exceed 20 - 24° C (Eaton et al., 1995). Therefore, increases in summer water temperatures above this threshold could depress or extirpate populations of these fishes. Most of these fish species are either autumn or spring spawners. Therefore, the direction and extent of a shift in peak flow timing will affect different species in different ways. Autumn spawners spawn in the fall, and the fry emerge from the gravel before the spring snowmelt. Spring spawners spawn after the spring snowmelt, and the fry emerge during the summer or early fall. The future scenarios that we evaluated predicted a dramatic shift in peak flow from May to January (Figure 2.1). This degree of shift would likely adversely affect the autumn spawners (dolly varden, brook trout, and coho salmon) whose eggs could be scoured and whose fry would potentially be washed away and/or covered with sediment (Montgomery et al., 1999). Spring spawners (spring chinook and rainbow and cutthroat trout) are less likely to be adversely affected by this predicted shift in peak flow, although they would probably be affected by reduced habitat availability resulting from extended summer low flow conditions (see Figure 2.1). Sixty percent of the chinook salmon in the Yakima Basin spawn in the Cle Elum, so shifts in the hydrograph that might affect these species could result in dramatic population declines basin-wide (Beckman et al., 2000).

One likely consequence of the shift from snowmelt to winter rain flow regime is that flood frequency will increase because precipitation from individual winter storms will not be stored as snow with only one run-off event during snow melt. These earlier floods would reduce the retention time of organic matter, thereby reducing food availability for detritivorous macroinvertebrates. Cuffney et al. (1997) found that these high elevation streams were inhabited by insects (predominantly mayflies, stoneflies, and caddisflies) that utilized allochthonous organic matter (relatively high shredder richness) rather than autochthonous organic matter (relatively low scraper richness). Moreover, a longer low flow period in the summer could benefit algae because of the reduction in shear stress (Biggs, 2000) and the increased water temperatures. Therefore, the importance of algae in the benthic food web could become more significant. This could cause a switch in the macroinvertebrate community from shredder-dominated to scraper-dominated. In addition a shift in high flow timing from late spring to winter would have significant negative impacts on the abundance of many aquatic insects, especially those that grow in the winter and emerge prior to the predictable high spring flows. Work in California streams that have a Mediterranean climate (winter floods and low summer flows) have documented important changes in invertebrate community composition and food web structure when winter floods are stored by dams and released as high summer flows (Power et al., 1996, Wootton et al., 1996). This example provides a basis for asserting that seasonal shifts can have significant impacts on benthos.

Case Study 2 - Apalachicola-Chattahoochee-Flint River Basin

The Chattahoochee-Apalachicola River Basin has its headwaters in Georgia and Alabama and extends approximately 620 kilometers before emptying into Florida's Apalachicola Bay in the Gulf of Mexico. The basin has a drainage area of 50,764 km^2. Fifty eight percent of the basin is forest, 29 percent of the basin is agriculture, and 5 percent of the basin is urban and suburban land uses (Frick et al., 1998). Average precipitation ranges from 114 to 152 cm per year. There are 14 reservoirs on the mainstems of the two rivers. These dams and reservoirs are used for hydropower, recreation, water supply, and flood control. The first large dams built, J. Woodruff and Buford dams, were completed in 1957. The Chattahoochee River runs through the city of Atlanta and has been impacted by urbanization including water withdrawals, wastewater plant effluents, and sedimentation. The Apalachicola River is formed by the confluence of the Chattahoochee and Flint Rivers and is a wide meandering Coastal Plain river with extensive floodplains. The channel has been dredged since 1954 to allow for barge traffic and the resulting channel degradation plus channel degradation downstream from Woodruff Dam has reduced the connectivity between the river and its floodplain (Light et al., 1998).

We used one site from the Chattahoochee and one from the Apalachicola to assess the impact of dams, surface water demand schemes, and climate scenarios on flow regime. The Chattahoochee River site at Whitesburg, GA is downstream of Atlanta and one major reservoir (Lake Lanier formed by Buford Dam). The Apalachicola River site at Chattahoochee, FL is downstream of Lake Seminole Reservoir (formed by Woodruff Dam) and all the dams on the Chattahoochee and Flint Rivers. We chose these sites because of their location on the river and the length of the gauging station records available. In addition, Light et al. (1998) have extensive data on flows needed to connect the floodplain and the channel for the Apalachicola River at Chattahoochee, FL.

For future climate scenarios, we used the Canadian Center for Climate Modeling and Analysis CGCM1 model for the years 2018-2048. This model predicts a warmer and drier climate for the southeast. These scenarios are derived from coupled atmospheric-ocean global climate model experiments with transient greenhouse gas and sulfate aerosol forcing. This is a transient model, which means that the climate is changing gradually over time. The original data set for this model has a resolution of 3.5 x 3.5 degrees, but the Vegetation / Ecosystem Modeling and Analysis Project (VEMAP) transferred this to 0.5 x 0.5 degrees for the United States. To calculate stream flow, we applied a procedure using regression analysis between runoff and climatological parameters (Georgakakos and Yao, 2000). Therefore, to calculate future weekly stream flow, we used the climate parameters from the CGCM1 model downscaled by VEMAP with the regression model to predict weekly runoff and stream flow (Georgakakos and Yao, 2000). These weekly stream flow values were translated into daily stream flow values by assuming a linear relationship in flow from one week to the next.

The demand projections used in this model were created by the Alabama Office of Water Resources, the Environmental Protection Division of the Georgia Department of Natural Resources, the Northwest Florida Water Management District, and the Mobile District Corps of Engineers for the Apalachicola-Chattahoochee-Flint Comprehensive Study (USACE, 1998). They developed three levels of demand (low, moderate, and high), which varied depending on conservation measures and agricultural demand. We used the moderate demand level. In this scenario, water use projection for municipal users was based on 1990-1993 residential use patterns and projection for industrial use was based on 1990 non-residential water use patterns. In both cases, the moderate demand level assumed passive conservation measures. Projected changes in population, employment, and housing characteristics drove water demand projections in the basin. The agricultural water demand assumed a continuation of the current rate of economic and technological growth and no substantial change in national agricultural policy, price support, or agricultural export. The water demand projections for agriculture also assumed continued irrigation with no conservation mechanisms. The demand projections do not take into account ground water pumping, which is significant in the Coastal Plain. Therefore, the agricultural demand is underestimated and our analysis is limited to the effects of surface water demands and withdrawals.

In order to assess the impact of dams, future water demand scenarios, and future climate scenarios on the flow regime of the Chattahoochee-Apalachicola River Basin, we evaluated flow regimes under three different conditions using IHA. First, we calculated IHA parameters for observed daily flows prior to construction of the major large reservoir producing dams on the mainstem of the Chattahoochee (Buford, West Point, Walter F. George, and J. Woodruff) and compared them to IHA parameters calculated from current observed daily stream flows (1965–1995). There are several smaller dams on the Chattahoochee that were built earlier than the large dams listed above. However, there are not sufficient flow data for the time period prior to the construction of these small dams to conduct an IHA analysis. Therefore, our "pre-dam" data set is actually pre-large reservoir dams. For the Chattahoochee River at Whitesburg, we used available pre-dam flow records from 1938-1950 and for the Apalachicola River at Chattahoochee available pre-dam flows were from 1928-1950. Second, to evaluate the impact of water demand in the year 2050, we subtracted 2050 demand from current observed stream flow conditions (1965-1995), calculated the IHA parameters for current conditions with 2050 demand superimposed and compared the parameters to current conditions. Third, to assess the combined impact of future climate scenarios and water demand, we calculated IHA parameters for daily flows interpolated from weekly flows predicted by the model based on the climate scenario parameters-stream flow relationship described above with 2050 demand subtracted.

Unlike the Cle Elum River, the statistical downscaling procedure for the Chattahoochee-Apalachicola basin was only simulated once. Therefore, the variability and standard errors for these data sets represent inter-annual variability rather than inter-simulation variability. We considered a parameter calculated from the flow regime of the future climate scenario to be greater than the value calculated from the observed data, when the value of the parameter under future climate scenario was out of the range of variability typically occurring within the time frame of the observed data. For instance, if mean 7-day maximum flow under current conditions was less than the mean minus standard error of the 7-day maximum flow under pre-dam conditions, then the current maximum flow was considered lower than that typically experienced in the pre-dam time period.

Chattahoochee River at Whitesburg

Although only 12 years of pre-dam records were available at this site, flows before the construction of dams were typified by high winter and spring flows and lower summer and especially fall flows (Figure 2.4.). The construction of Buford dam on the Chattahoochee River upstream of Whitesburg resulted in an approximately 28 m^3s^{-1} increase in summer flows (Figure 2.4). However, expected increases in demand for water in 2050 would decrease summer and fall flows (June – October) even without any climate change (Figure 2.4). Management changes and expected demand increases would lead to decreased mean monthly flow in the summer and fall by almost 28 m^3s^{-1}. Under predicted future climate scenarios and 2050 demand, summer flows would decrease even further (Figure 2.4). Mean monthly flow for summer months (June-September) were predicted to decrease another 14-28 m^3s^{-1}. Under current climate and demand, mean flow in the Chattahoochee River at Whitesburg, GA for the summer months of June through September is 99 m^3s^{-1}. Under 2050 demand and future climate scenarios, average flow for these months was predicted to be 57 m^3s^{-1}.

The average 7, 30, and 90-day maximum flows were similar among the three water conditions (pre-dam construction, current conditions, and current climatic conditions but increased demand) (Figure 2.5). However, under future climate scenarios and expected demand, the 7, 30, and 90-day maxima were all lower than current conditions and current conditions with

2050 demand (Figure 2.5). The average 7, 30, and 90-day minima increased after construction of the dams (Figure 2.5). Demand increases expected in the future lowered the 30 and 90-day minima, but had little impact on the 7-day minimum (Figure 2.5). Future climate scenarios combined with expected demand increases were predicted to lower the minimum flow at all three time scales, 7, 30, and 90-days (Figure 2.5). However, these low flows are still higher than those experienced from 1938-1950.

Construction of Buford Dam decreased the variability of flow in the river during summer and winter and slightly increased flow variability in the spring (Figure 2.6). Increases in demand under current climate conditions would slightly increase the variability of the system in the summer. Future climate scenarios coupled with future demand would decrease variability of river flow throughout the year (Figure 2.6). The predicted decreased system variability under future climate scenarios and future demand appears to primarily be the result of the climate scenario since the increased demand under current conditions actually slightly increased system variability (Figure 2.6).

Apalachicola River at Chattahoochee, FL

Dam construction did not significantly alter mean monthly flow in the Apalachicola (Figure 2.7), although winter flows (January and February) were slightly higher under current conditions. Increased water demand would slightly lower mean monthly flow from late spring to late summer (April to September) (Figure 2.7). Future climate scenarios coupled with increased demand produced even lower flows from late winter through the fall (February to October) (Figure 2.7).

Seven, 30 and 90 day maxima for current conditions were similar to those experienced prior to large impoundments (Figure 2.8). Moreover, 2050 demand levels did not appear to alter these maxima (Figure 2.8). Future climate scenarios combined with 2050 demand levels would decrease the 7, 30, and 90 day maxima (Figure 2.8). Seven, 30 and 90 day minimum flows for current conditions were also similar to those experienced before large impoundments (Figure 2.8). However, when future water demands were imposed on current conditions, the 7 and 30 day minima decreased (Figure 2.8). When future climate scenarios and future water demands were used, the 7, 30, and 90 day minima were all lower than current conditions and current conditions with future demand (Figure 2.8).

In general, variability within the system declined after dam construction. Increased demand did not appear to significantly alter the variability of the system. However, under future climate scenarios, the variability in the system was predicted to be lower than under current conditions (Figure 2.9). The lowering of the variability appears to be driven by the future climate scenario rather than the increased demand, since the increased demand did not have a noticeable impact on system variability (Figure 2.9).

Seventy-three of the 91 species of fishes known to inhabit the Apalachicola River have been collected in river floodplains under a variety of hydrologic conditions (Light *et al*., 1998). Fishes in the Apalachicola River system primarily spawn from April through July, and therefore the extent of connected aquatic habitats during this time period represents the potential availability of spawning habitat (Light et al., 1998; Freeman, 1998). Under increased demand and future climate scenarios, the percentage of the floodplain connected to the river will decrease. For instance the predicted 90-day maximum flow under 2050 demand and climate is 935 m^3 s^{-1}, at which point about 50% of the floodplain is connected (Light et al., 1998). Under current conditions the 90-day maximum flow is 1167 m^3s^{-1} at which point about 68% of the floodplain is connected to the river (Light et al., 1998). Thus, in an average year 18% of the floodplain typically available for fish to forage and spawn during the critical spawning period under current conditions would be unavailable under 2050 demand and climate.

Currently, the movement of fish between the main channel and the tributaries requires a flow of 312 m^3s^{-1} at the Chattahoochee gauge (Light et al., 1998). Prior to 1954, there were no days from May to November below this flow level with restricted tributary access; however, after 1981 there was a median of 66 days below this flow threshold. Under 2050 demand and future climate scenarios, mean monthly flow for August, September, and October and the average 90-day minimum are all below the 312 m^3s^{-1} threshold, which means the fish could not move between the main channel and the tributaries.

The presence of specific habitat types is often dependent on the presence of certain flow levels. For instance, in the Apalachicola River near the Chattahoochee gage, the area of raceway and shallow pool habitat increases by more than 25% as flow increases from 240 to 325 $m^3\ s^{-1}$ (Freeman et al., 1997). Under 2050 demand and future climate scenarios, the area of this type of habitat could be drastically reduced as the mean monthly flow for August and September are not predicted to exceed 227 $m^3\ s^{-1}$. Raceway habitat makes up less than 3% of the available habitat at flows below 227 $m^3\ s^{-1}$ and shallow pool makes up around 10-15% of the available habitat (Freeman *et al.*, 1997). These percentages decline as discharge falls below 142 $m^3\ s^{-1}$ (Freeman et al., 1997).

One of the major concerns with the Chattahoochee River at Whitesburg is the ability to assimilate the large amount of wastewater treatment plant effluent it receives daily. Currently, during baseflow conditions 20% of the streamflow at Whitesburg is wastewater treatment plant effluent (USACE, 1998). Under the 2050 demand scenario with current climate conditions the percentage of baseflow that is effluent will increase to 33% (USACE, 1998). With the decreased mean monthly flow predicted by the future climate scenarios (Figure 2.4), percent effluent is predicted to increase further. In addition, decreased flow is often associated with increased temperature which, in turns, lowers dissolved oxygen. Thus, a higher effluent percentage could increase biological oxygen demand and combine with higher temperatures to potentially lead to low oxygen levels and problems for aquatic organisms.

Conclusion

These case studies illustrate some of the types of changes to river ecosystems under future climate scenarios. Changes in timing and form of precipitation can affect the timing of maximum and minimum flows as we saw in the Cle Elum, where future scenarios predicted more winter rains and less winter snows. In addition, human demands for freshwater can exacerbate the situation in areas that are predicted to have less rain under future climate scenarios. We saw this situation in the Chattahoochee-Apalachicola basin, where future climate scenarios coupled with increased human demand resulted in lower high and low flows. In both cases, the changes in the flow regime indicated changes in habitat availability during times of the year that are critical for the survival of fish and other organisms. Changes in the timing of the peak flows could affect survival of salmon species in the Cle Elum and longer low flows in the summer could lead to increased importance of autochthonous resources. Lower peak flows in the Apalachicola could further disconnect the floodplain from the channel which would dramatically reduce the availability of that habitat for fish spawning and for young-of-the-year fish. In addition, the lower flows could mean reductions in in-channel habitat such as raceways and shallow pools as a result of the wetted width of the channel being reduced. Changes in flow regime as a result of changes in climate will be gradual, unlike the abrupt changes that result from dam construction or increased water withdrawal.

Future climate scenarios can cause dramatic shifts in flow regimes, which influences ecological processes in aquatic ecosystems. Shifts in timing of flood events, changes in seasonal flow regimes, and changes in the magnitude of baseflows all can influence organisms within the streams and ecosystem processes. In addition, human alteration of rivers and flow regimes through channelization, dams, and water withdrawals can interact with these climate changes to further alter the functioning of ecosystems. These types of analyses provide scenarios of the type and magnitude of hydrologic changes under future climates and implications for freshwater organisms and ecosystem processes.

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Figure 2.1. Mean monthly flow of the Cle Elum River, WA using observed and downscaled GCM generated stream flow. Error bars are mean ± 1 standard error for the 20 GCM simulations. When not shown, error bars are smaller than the symbol.

Figure 2.2: 1, 3, 7, 30, and 90 day maxima (A) and minima (B) flows for the Cle Elum River, WA using observed and downscaled GCM predicted stream flows. Error bars are mean ± 1 standard error. When not shown, error bars are smaller than the symbol.

Figure 2.3. Coefficient of variation for mean monthly flow of the Cle Elum River, WA using observed and downscaled GCM generated stream flow. Error bars are mean ± 1 standard error for the GCM data. When not shown, error bars are smaller than the symbol.

Figure 2.4: Mean monthly flow of the Chattahoochee River at Whitesburg, GA under different management and climate scenarios: pre-dam conditions; current climate and management (1965-1995); current climate (1965-1995) and 2050 management ; predicted future climate (2018-2048) and 2050 water demand. Error bars are mean ± 1 standard error. When not shown, error bars are smaller than the symbol.

Figure 2.5: 7, 30, and 90 day maxima (A) and minima (B) flows of the Chattahoochee River at Whitesburg, GA under four different climate and management scenarios: 1) pre-dam conditions (1938-1950), 2) current climate and management (1965-1995), 3) current climate (1965-1995) and 2050 management, 4) predicted future climate (2018-2048) and 2050 management. Error bars are mean ± 1 standard error.

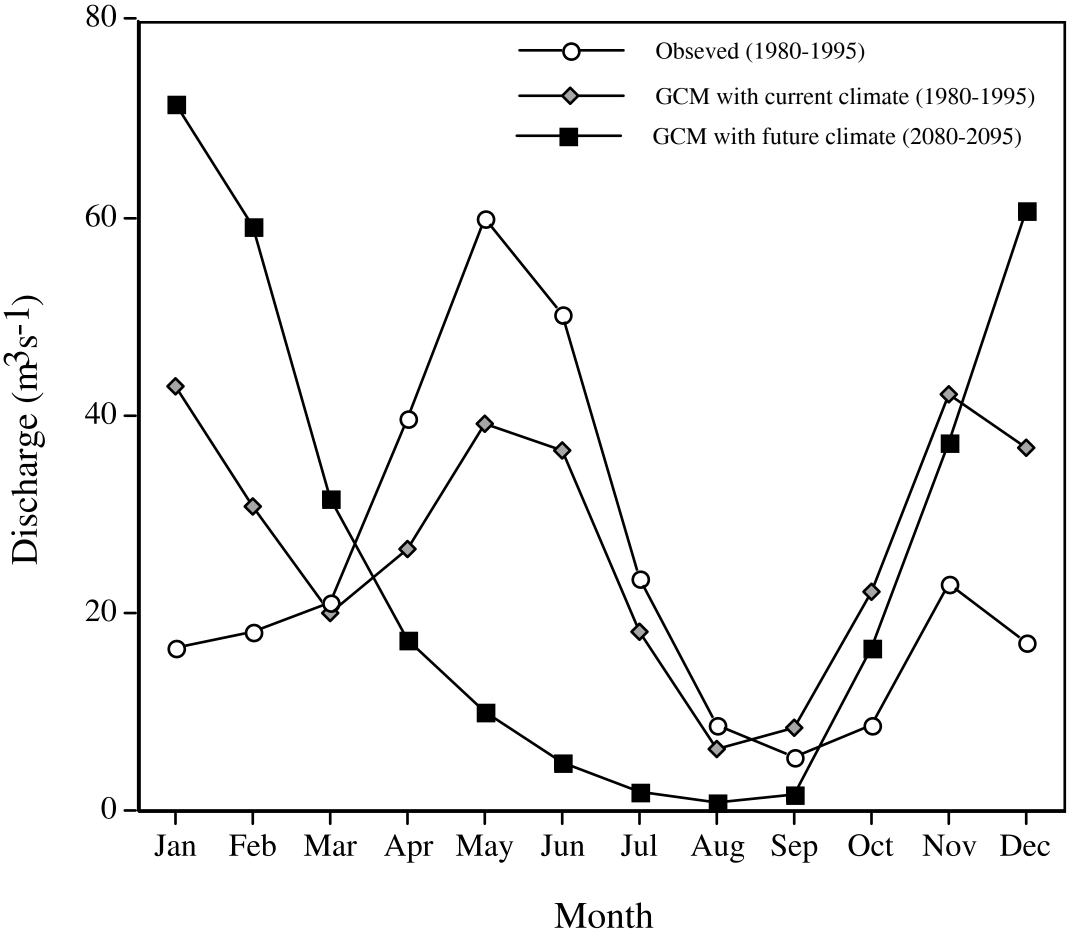
Figure 2.6. Coefficient of variation for mean monthly flow of the Chattahoochee River at Whitesburg, GA under four different climate and management scenarios: pre-dam conditions (1938-1950), current climate and management (1965-1995), current climate (1965-1995) and future management (2050), and predicted future climate (2018-2048) and management (2050).

Figure 2.7: Mean monthly flow of the Apalachicola River at Chattahoochee, FL under four different management and climate scenarios: pre-dam conditions(1928-1950), current climate and management (1962-1992), current climate (1962-1992) and 2050 management, and predicted future climate (2018-2048) and 2050 water demand. Error bars are mean ± 1 standard error.

Figure 2.8: 7, 30, and 90 day maxima (A) and minima (B) flows of the Apalachicola River at Chattahoochee, FL under four different climate and management scenarios: 1) pre-dam conditions (1928-1950), 2) current climate and management (1962-1992), 3) current climate (1962-1992) and 2050 management, 4) predicted future climate (2018-2048) and 2050 management. Error bars are mean ± 1 standard error.

Figure 2.9. Coefficient of variation for mean monthly flow of the Apalachicola River at Chattahoochee, FL under four different climate and management scenarios: pre-dam conditions (1928-1950), current climate and management (1962-1992), current climate (1962-1992) and future management (2050) and predicted future climate (2018-2048) and management (2050)

Figure 2.1.



80
60
40
20
0
Discharge (m^3s^{-1})
Obseved (1980-1995)
GCM with current climate (1980-1995)
GCM with future climate (2080-2095)
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
Month

Figure 2.2.

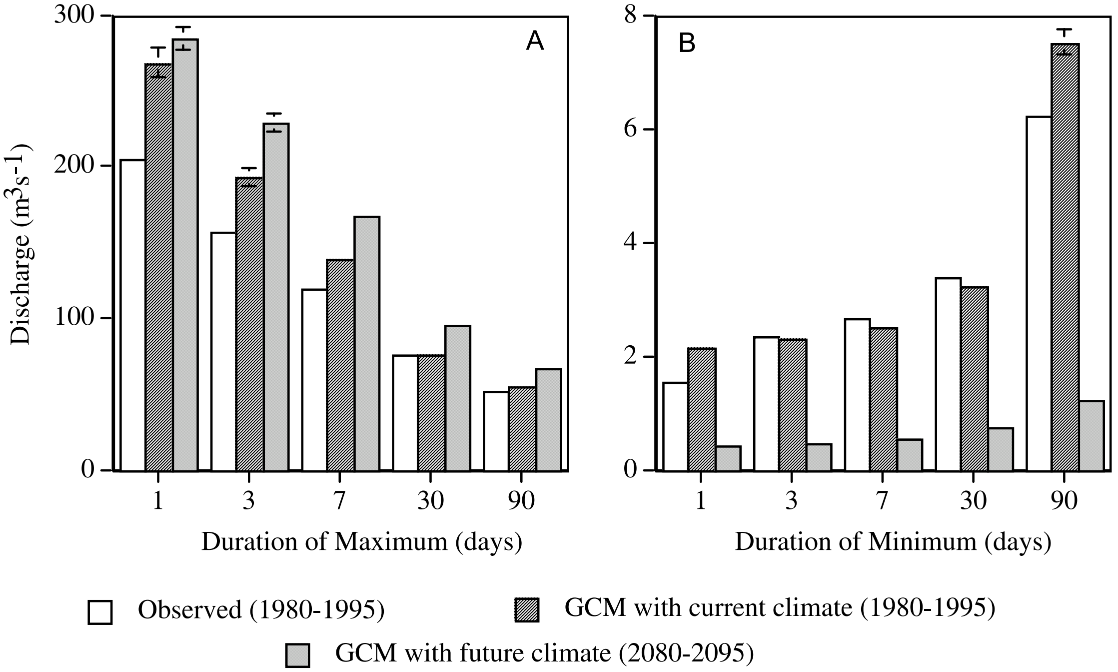
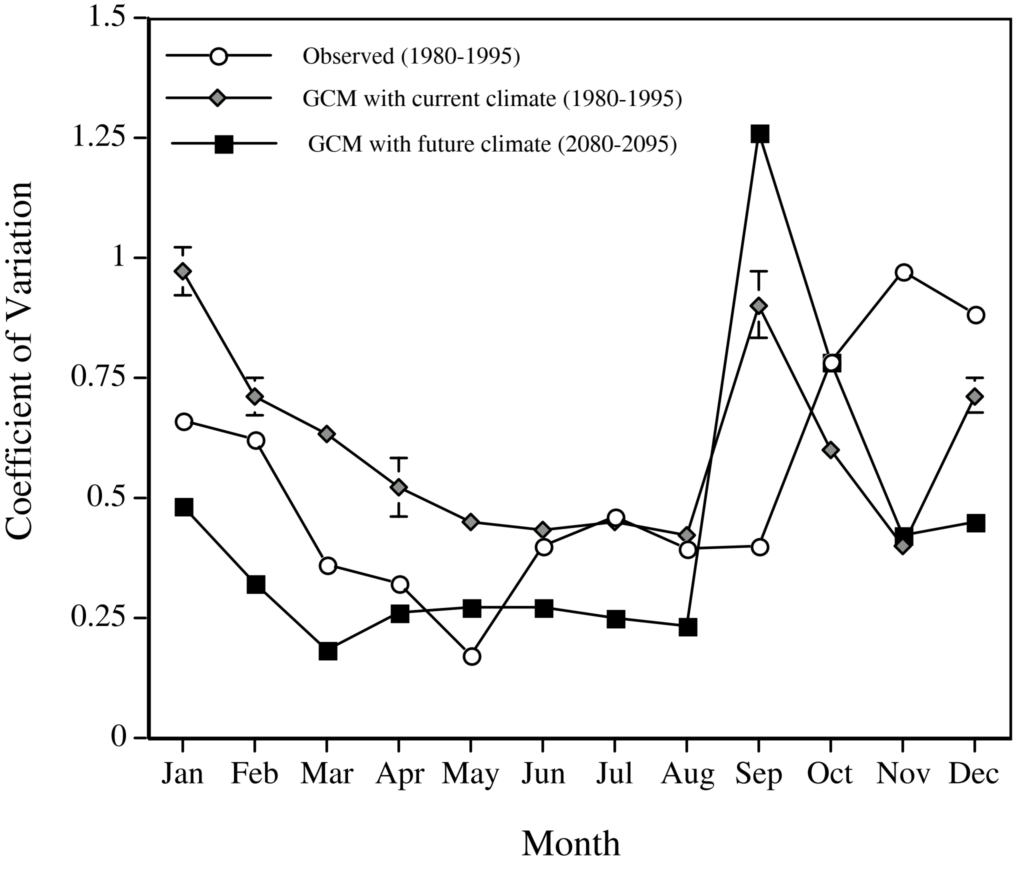
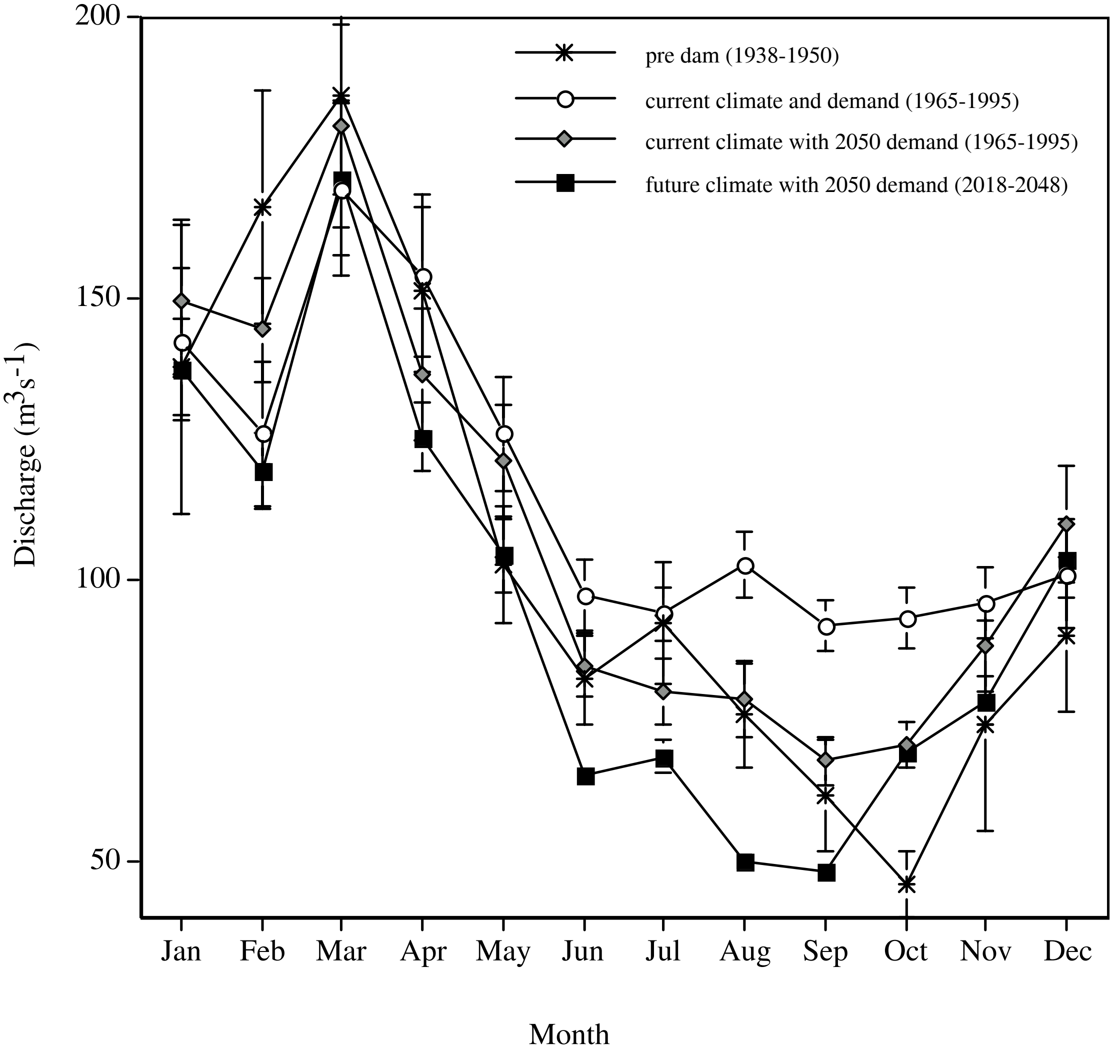

A
B
Discharge (m^3s^-1)
300
200
100
0
8
6
4
2
0
1
3
7
30
90
Duration of Maximum (days)
Duration of Minimum (days)
Observed (1980-1995)
GCM with current climate (1980-1995)
GCM with future climate (2080-2095)

Figure 2.3



Observed (1980-1995)
GCM with current climate (1980-1995)
GCM with future climate (2080-2095)
Coefficient of Variation
1.5
1.25
1
0.75
0.5
0.25
0
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
Month

Figure 2.4



200
150
100
50
Discharge (m^3s^{-1})
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
Month
pre dam (1938-1950)
current climate and demand (1965-1995)
current climate with 2050 demand (1965-1995)
future climate with 2050 demand (2018-2048)

Figure 2.5

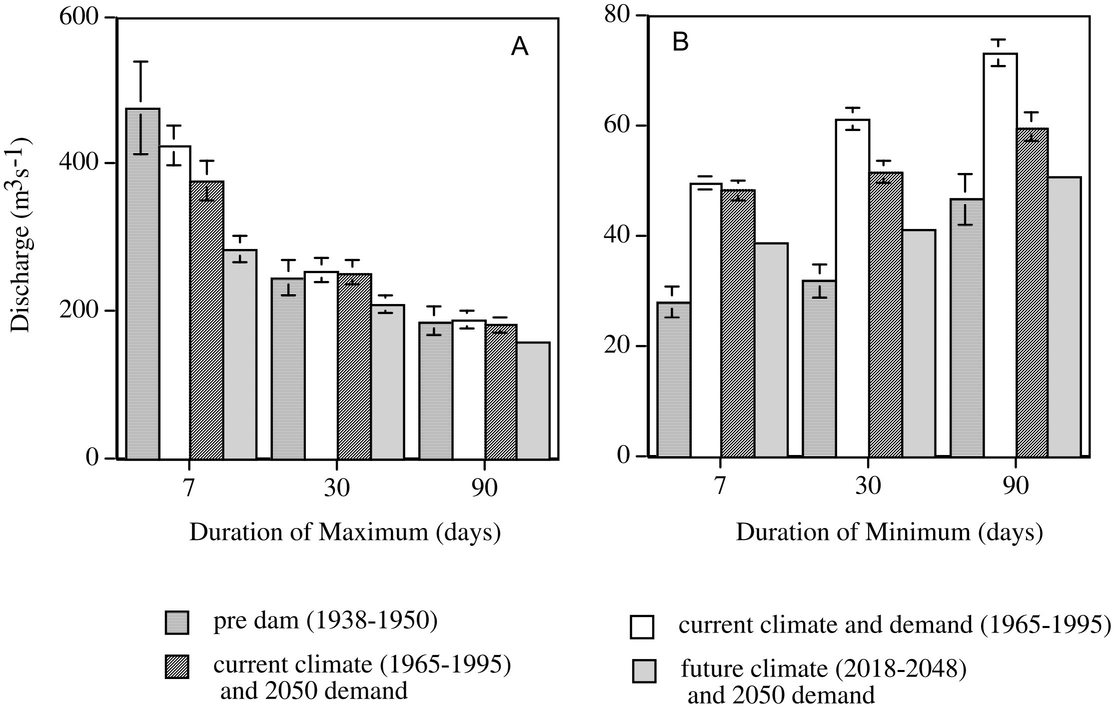

A
B
Discharge (m3s-1)
600
400
200
0
80
60
40
20
7
30
90
Duration of Maximum (days)
Duration of Minimum (days)
pre dam (1938-1950)
current climate (1965-1995) and 2050 demand
current climate and demand (1965-1995)
future climate (2018-2048) and 2050 demand

Figure 2.6

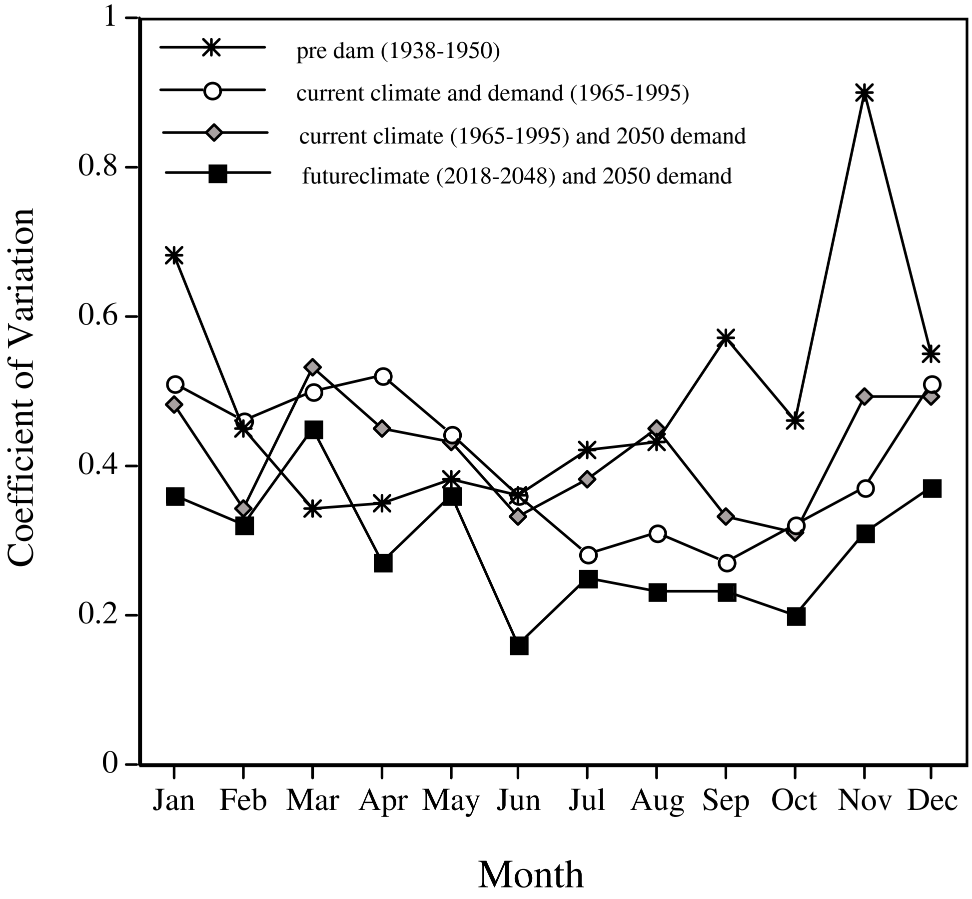
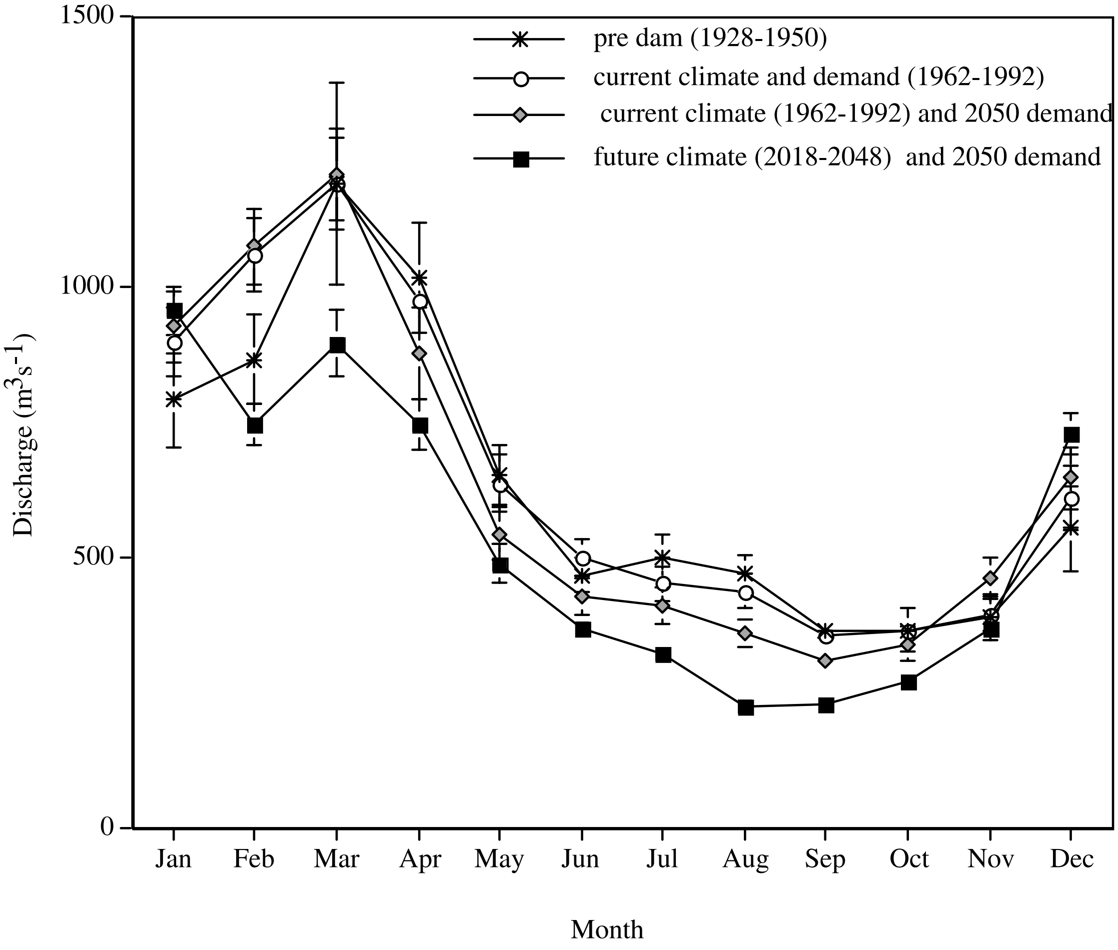

1
0.8
0.6
0.4
0.2
0
Coefficient of Variation
pre dam (1938-1950)
current climate and demand (1965-1995)
current climate (1965-1995) and 2050 demand
futureclimate (2018-2048) and 2050 demand
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
Month

Figure 2.7



1500
1000
500
0
Discharge (m^3s^{-1})
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
Month
pre dam (1928-1950)
current climate and demand (1962-1992)
current climate (1962-1992) and 2050 demand
future climate (2018-2048) and 2050 demand

Figure 2.8

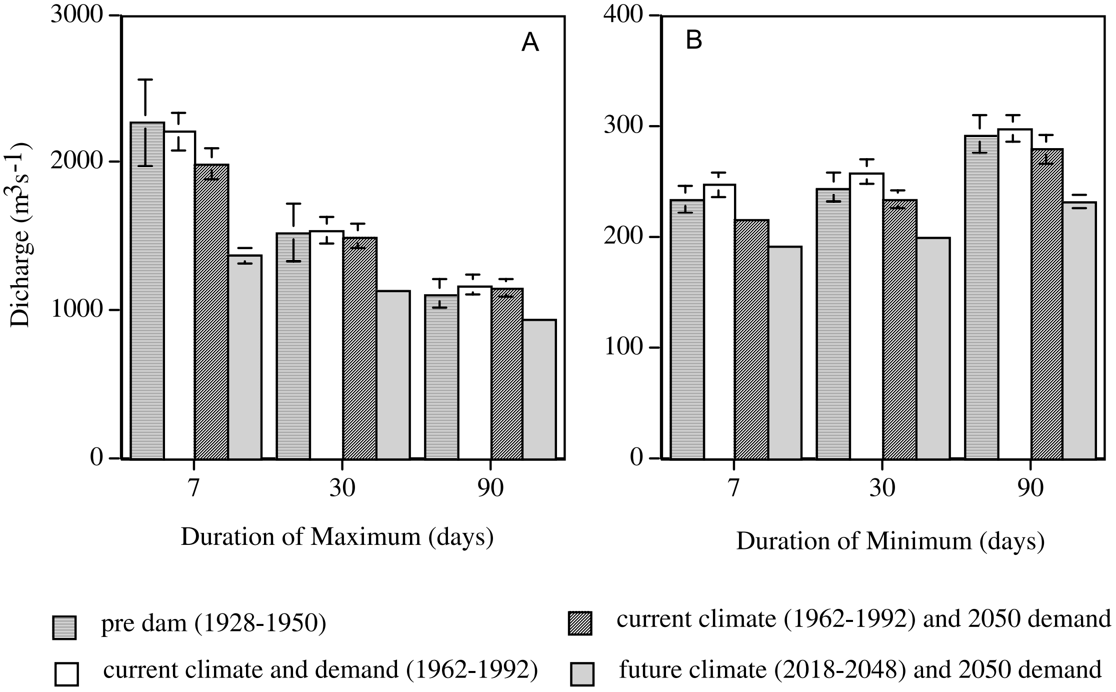
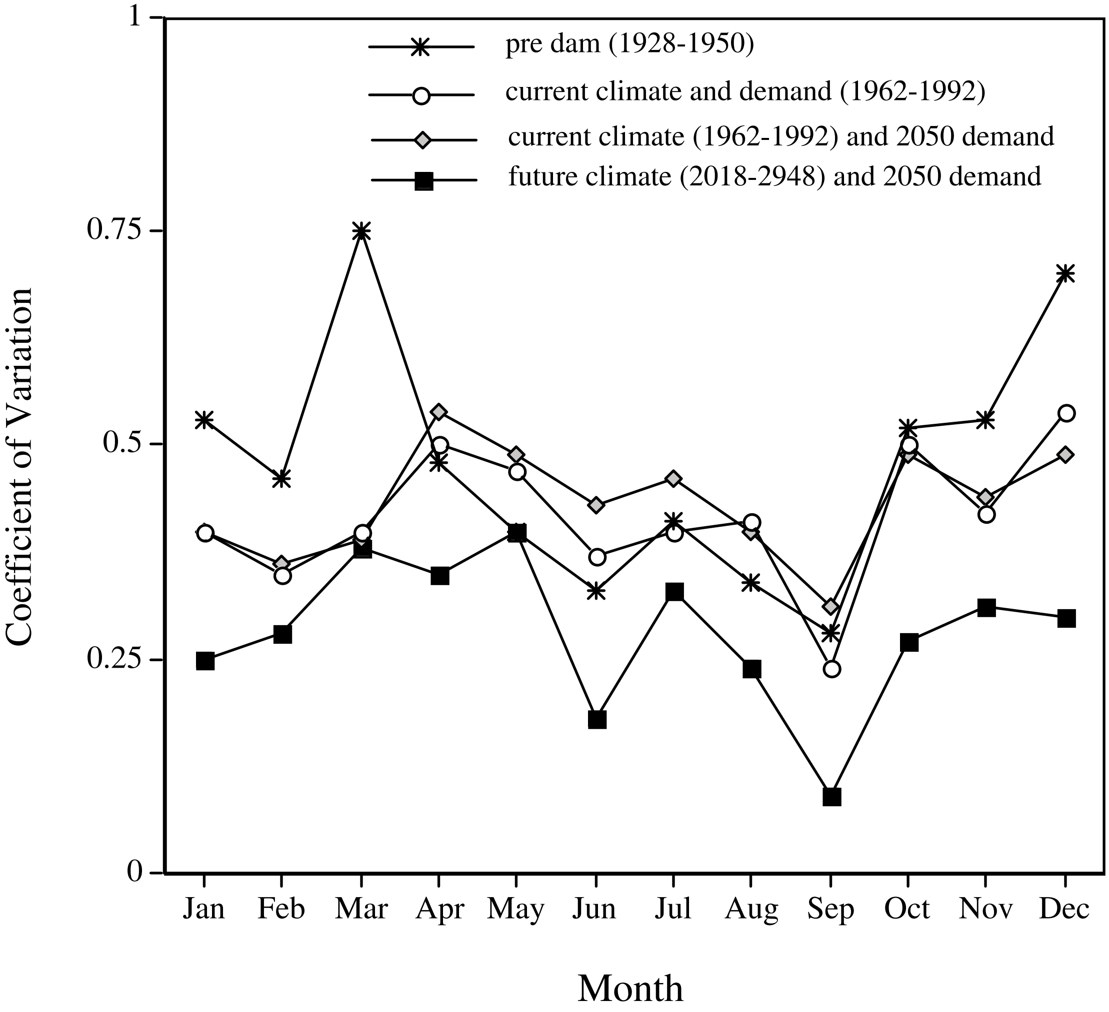

3000
2000
1000
0
A
Dicharge (m³s⁻¹)
7
30
90
Duration of Maximum (days)
400
300
200
100
0
B
7
30
90
Duration of Minimum (days)
pre dam (1928-1950)
current climate (1962-1992) and 2050 demand
current climate and demand (1962-1992)
future climate (2018-2048) and 2050 demand

Figure 2.9


pre dam (1928-1950)
current climate and demand (1962-1992)
current climate (1962-1992) and 2050 demand
future climate (2018-2948) and 2050 demand
Coefficient of Variation
1
0.75
0.5
0.25
0
Jan
Feb
Mar
Apr
May
Jun
Jul
Aug
Sep
Oct
Nov
Dec
Month

CHAPTER 3

RELATING ECOSYSTEM METABOLISM AND FLOW REGIME IN A REGULATED URBAN RIVER[2]

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[2] Gibson, C.A. and J.L. Meyer To be submitter Freshwater Biology

Abstract

Flow regime alterations resulting from river regulation by dams affect the ecological integrity of river systems. However, little is known about the impacts of changes in flow regime on basic ecosystem processes such as metabolism. In addition, the relationship between metabolism and low-flow condition is uncertain, especially in larger river systems. The objective of this study was to determine if mean 24-hour low-flow discharge regime could account for variation in ecosystem metabolism, or if other physical, biological, and chemical parameters were needed to explain the variation. We measured ecosystem metabolism on 14 dates in the Chattahoochee River, downstream of Atlanta, GA. Hydropower dams on the river result in discharge fluctuations that are typically much smaller on the weekend than during the week; therefore, all measurements were made on weekends. Net ecosystem metabolism was always negative, and respiration ranged from 12.5 to 39.2 g O_2 m^{-2} day^{-1}. Gross primary production (GPP) was variable and ranged from 0.4 to 4.7 g O_2 m^{-2} day^{-1}. Average maximum discharge for the five days prior to metabolism measurements was negatively correlated with GPP and accounted for 28% of the variation in GPP. Hydropeaking reduces GPP most likely as a result of higher flows disturbing snag and benthic periphyton communities. Both suspended fine particulate organic matter (FPOM) and total phosphorus concentration were positively correlated with respiration, explaining 72 % of the variation in respiration. Atlanta's wastewater treatment plants are likely a source of both FPOM and total P. Water column processes accounted for approximately one-third of whole system metabolism. Compared to other rivers of a similar size, the Chattahoochee River had lower GPP and higher respiration than might be expected. Therefore, it appears that hydropeaking decreases GPP, but organic matter and nutrient loading increases respiration in the Chattahoochee River. In addition, by processing sewage-derived particulate organic matter, the

Chattahoochee is helping to maintain downstream water quality; providing a critical ecosystem service to the human communities in this watershed. This study indicates how measures of ecosystem processes can help quantify ecosystem services that are being provided to human society by streams and rivers.

Introduction

The quantity and timing of river flow are critical attributes for water supply, water quality, and the ecological integrity of river systems (Poff et al. 1997). Although there are many different types of hydrologic and channel alterations that result in changes to the flow regime, dams are one of the most prevalent forms of flow alteration on large and some small rivers and streams (Dynesius and Nilsson 1994). In the contiguous United States, there are only 42 rivers with greater than 200 river kilometers unregulated by major dams (Benke 1990). Although there have been a number of studies on the impacts of dams on channel morphology (Fenner et al. 1985, Ligon et al. 1995, Stevens et al. 1995), fish (Sparks 1995, Freeman et al. 2001), habitat availability (Sparks 1995, Freeman et al. 2001), and riparian species survival and recruitment (Rood et al. 1995, Jannson et al. 2000), less is known about the impacts of dams and flow regime on basic riverine ecosystem processes such as metabolism.

Metabolism has been shown to vary with high stream discharge as a result of shifts in primary production (Uehlinger and Naegeli 1998). In an 18 month study of metabolism of a pre-alpine river, bed-moving spates decreased primary production, but had less of an effect on respiration (Uehlinger 2000). In addition, primary production recovered more quickly from the spates than respiration (Uehlinger 2000). Hydropeaking by power-generating dams causes large fluctuations in discharge for large stretches of the river both within and between days. For example, hourly flow data for the Tallapoosa River, AL below Harris Dam indicated that two to six-fold variations in discharge can occur as the result of hydropower generation within a 24 hour period (Irwin and Freeman 2002). Therefore, regulated rivers experience almost daily discharge spates that may influence both primary production and ecosystem respiration.

In the United States, dams are regulated to provide certain services such as flood control or navigation. Regulations typically require that low-flow levels be maintained above a threshold and result in altered low flow conditions in regulated rivers (Poff et al. 1997). In the Chattahoochee River, dam construction has increased summer discharge compared to pre-dam conditions (Chapter 2). Relationships between net ecosystem metabolism and low flow conditions are uncertain, and this is particularly the case in large rivers.

Urban streams are characterized by high nutrient concentrations and high sediment inputs (Paul and Meyer 2001). For larger urban streams, high nutrient concentrations are the result of non-point and point source inputs. In addition to nutrients, point source inputs from waste water treatment plants contain sewage-derived particulate organic matter (POM) which is a labile mixture of organic detritus and microorganisms such as bacteria and algae (deBruyn and Rasmussen 2002). While advances in technology have reduced the organic matter loading so that it typically does not cause oxygen depletion in rivers, the processing of this organic matter is critical to maintaining downstream water quality especially for the next reservoir downstream.

Small reservoirs in the southeast are typically monomitic with high densities of heterotrophic bacteria and anoxic conditions in the hypolimnion during summer months (Porter et al. 1996). In larger reservoirs, organic matter sedimentation from organic-rich catchments can also lead to anoxia in the hypolimnion (Cole and Hannan 1990). In these reservoirs, the anoxia typically starts in the transition zone (an area of high sedimentation) and can move up and downstream depending on the stability of the stratification and the amount of organic matter loading (Cole and Hannan 1990). High rates of microbial processing of the labile sewage-derived particulate organic matter in the reservoir rather than in the river could exacerbate anoxia and create further habitat limitations for both crustacean zooplankton and fish. In addition, increasing POM loading to the reservoir would increase turbidity, thereby decreasing light availability and most likely decreasing phytoplankton, an important basal food resource (Porter et al. 1996).

The objective of this study was to determine if variation in net ecosystem metabolism (NEM), gross primary production (GPP), or respiration (R) is related to the variation in low flow discharge in the Chattahoochee, or if metabolism variation is better explained by other physical, chemical, or biological parameters. In previous studies, GPP was frequently correlated with light availability (Mulholland et al. 2001, Young and Huryn 1996, 1999, Wiley et al. 1990). In streams draining agricultural catchments, light availability in the stream was mediated by turbidity (Young and Huryn 1996, 1990). Therefore, we predicted that GPP would decrease with increasing discharge as a result of increases in turbidity. Respiration has been shown to increase with transient storage zone size in several small streams (Fellows et al. 2001, Mulholland et al. 2001). Transient storage zones include side channel areas and sand bars, where water moves more slowly than in the thalweg. Therefore, we would expect R to increase with increasing discharge as more sand bars and side channel areas become inundated. In addition, in sandy-bottom streams in the Southeast, snags have been shown to be areas of high biological activity. The area of snags inundated would increase with increasing discharge and therefore likely result in higher R as well.

Methods

Study Site

We measured metabolism on the Chattahoochee River just downstream of Atlanta, Georgia. The Chattahoochee River provides drinking water for portions of Atlanta and receives the majority of its wastewater effluent (Frick et al. 1996). Nine municipal wastewater treatments plants upstream of this site discharge approximately 7.25 × 10^8 L day^{-1} of effluent in to the river (EPA Permit Compliance System 2001). Discharge is regulated by two upstream dams: Buford Dam, a large hydropower dam, and Morgan Falls Dam, a smaller re-regulation hydropower dam. The generation of hydropower causes large fluctuations in discharge within and between days. Discharge fluctuations are typically much smaller on weekend days (Saturday, Sunday, and sometimes Monday) than on week-days (Tuesday-Friday) so we measured metabolism Saturday to Monday.

During 2000, we measured metabolism over a 1.2 km reach just above the Highway 166 bridge (Figure 3.1). In 2001, this reach was not accessible due to low flows; therefore, we used a 0.75 km reach below the Highway 166 bridge. Both reaches had primarily sand and silt bottoms with each reach having one small shoal that contained large boulders and cobble. Large snags were present along the sides of the channel in both reaches. Both sides of the channel were constrained by high, steep banks that prevented connection between the river and the floodplain during all but the most extreme flood events.

In 2000 we sampled from June to September. In 2001 we sampled once in both April and May and more intensively from June through mid-October. Both 2000 and 2001 were drought years in Georgia. The 37-year average discharge at the Highway 166 bridge for the five month period of June to October is 82.7 m^3s^{-1} (USGS station # 02337170, Chattahoochee River at Fairburn, GA). In 2000 average discharge for this period was 66.1 m^3s^{-1}, and in 2001 average discharge was 51.3 m^3s^{-1}.

Metabolism

We determined metabolism using the upstream-downstream diurnal dissolved oxygen change technique (Marzolf et al. 1994, Young and Huryn 1998). Travel time between the stations was determined by floating oranges between them. We used the median orange velocity, and corrected for the difference between surface and mean velocity by multiplying by 0.85 (Gordon et al. 1992). We measured dissolved oxygen and temperature at 5-minute intervals for a 24-hour or 48-hour period using sondes (Hydrolab Corporation, Austin Texas 2000, YSI Corporation, Yellow Springs, Ohio 2001). Sondes were hung from snags in the thalweg of the channel. Oxygen exchange across the water surface was determined by multiplying average oxygen saturation deficit within the reach, the reaeration coefficient, and travel time (Young and Huryn 1998). Reaeration coefficients were calculated using the energy dissipation model, which is based on channel slope and water velocity (Tsivoglou and Neal 1976). Channel slope for this model was determined by using 1:24000 USGS topographic maps. We measured water velocity in the reach at a variety of different discharges using oranges, and determined the relationship between discharge and velocity using linear regression ($r^2 = 0.89$ $p < 0.01$ for 2000, $r^2 = 0.95$ $p< 0.01$ for 2001). We used this empirically derived relationship and discharge measurements from a USGS real time gauging station (station # 02337170, Chattahoochee River at Fairburn, GA) to estimate velocity at 15 min intervals. Net change in oxygen per time was calculated by determining the changes in oxygen concentration between the upstream and downstream stations corrected for changes due to reaeration at 15 min intervals. This concentration was multiplied by discharge to determine the change in mass flux of oxygen over the reach. A positive value indicates a net addition of oxygen over the reach, whereas a negative value indicates a net loss of oxygen over the reach.

Respiration (R) rate per day was determined by summing the change in mass flux of oxygen during the night and the extrapolated daytime R determined from the area under a two-point regression between dawn and dusk measurements (Mulholland et al. 2001). Gross primary production (GPP) was calculated by determining the difference between the observed change in mass flux of oxygen over the reach and extrapolated max flux of oxygen during the day if only R was occurring. We then summed these values over the daylight hours for GPP per day (Mulholland et al. 2001). Net ecosystem metabolism is the difference between GPP and R for the day.

We converted these reach-based measurements to rates per unit area by dividing by the channel area. Wetted channel width and depth was determined from 10 transects measured during a discharge that was in the middle of the range over which metabolism was measured. Due to the difficulty of making these measurements, we were only able to measure channel width on one date in each reach, and we used this as width for all metabolism measurements for that reach. The steep banks present throughout the reach prevent water from moving laterally with increases in discharge, therefore wetted width stays relatively constant over the range of discharges sampled during this study. Depth measurements were also only measured on one date, but we used the relationship between stage height and discharge to determine mean depth for each date.

To determine the importance of water column processes to whole-system metabolism, we determined water column GPP and R using light and dark bottles on 6 dates. We collected water from just below the surface of the thalweg in six 300 ml biological oxygen demand (BOD) bottles. Three of the bottles were clear and three of the bottles were covered with dark electrical tape. Oxygen concentrations were measure using a stirring probe designed for BOD bottles attached to an oxygen meter (YSI, Yellow Spring, OH). Bottles were hung just below the surface of the water from snags for one-hour during the late morning. Bottle GPP and R were calculated using the methods of Wetzel and Likens (2001). Due to turbidity, the photic zone in the Chattahoochee is less than the entire water column. Therefore, to estimate water column GPP we needed to multiply bottle GPP by the ratio of the photic zone to water column depth. In the Hudson River, NY, a large, moderately turbid river, the ratio of the photic zone to the water column depth varies from 0.5 to less than 0.2 (Cole et al. 1991). From 1987-1992, average summer (June-August) suspended solid concentration in the Hudson ranged from about 11-13 mg L^{-1} (Strayer et al. 1990). Based on these data, we multiplied bottle GPP by 0.35 (median value of the ratio of photic zone depth to water column depth in the Hudson River) and by the number of daylight hours to estimate water column GPP (g O_2 m^{-3} day^{-1}). Water column R (g O_2 m^{-3} day^{-1}) was estimated by multiplying bottle R by 24 hours. To determine the percentage of whole-system metabolism accounted for by water column processes, we converted water column metabolism measures from volumetic measures (g O_2 m^{-3} day^{-1}) to aeral measures (g O_2 m^{-2} day^{-1}) by multiplying by water depth on each date.

We measured discharge and a number of other physical, chemical, and biological characteristics to explore possible relationships with metabolism. Discharge was measured every 15 minutes at a USGS station (station # 02337170, Chattahoochee River at Fairburn, GA) immediately downstream (in 2000) or upstream (in 2001) of the metabolism reach. Using 15 minute discharge data, we averaged the maximum discharge for each of the five days prior to the metabolism measurement to determine average maximum discharge. Total solar radiation (KJ/m^2), precipitation for the week prior to metabolism measurements, and barometric pressure were obtained from a Georgia Automated Environmental Monitoring Site approximately 25 km from the site (Dunwoody station). We measured concentrations of total dissolved phosphorus and total dissolved nitrogen one day prior to the metabolism measurements using standard methods (Wetzel and Likens 2001). To capture variation in wastewater treatment plant discharge, we measured chloride concentration in the water column using standard methods (Wetzel and Likens 2001). Total suspended solids were also measured one day prior to metabolism measurements by filtering a known volume of water through pre-ashed and weighed glass fiber filters (Gelman A/E). The filters were dried at 60° C for at least 48 hours before being weighed again. Dissolved organic carbon was measured one day prior to metabolism using a total carbon analyzer (Shimadzu TOC, UGA Soil Analysis Laboratory).

Samples for water column bacteria concentration were taken one day prior to metabolism measurements and preserved in 5% formalin until direct counts using acridine orange could be performed (Kirchman 1993). Water column chlorophyll a measurements were made one day prior to metabolism measurements by filtering a known amount of water through glass fiber filters (Whatman GFF). On 5 dates in 2001 we collected algal samples for chlorophyll a analysis from 10 snags located in the metabolism reach. Snags were picked haphazardly. Biofilm was scraped from a known area using a brush, and placed on ice for transport back to the laboratory. The sample was then brought to a known volume and filtered on to glass fiber filters. Filters from both water column and snags were kept in the dark and frozen until analyzed by grinding for 30 s in 90% buffered acetone and measured either using a spectrophotometer (for 2000 samples, Wetzel and Likens 2000) or a fluorometer (2001 samples, Arar and Collins 1997). Snag chlorophyll a was multiplied by total snag surface area per m^2 of stream channel. We determined the surface area of snags in the channel using the methods of Wallace and Benke

(1984). Snag surface ranged from 3.8 to 4.2 m^2/ m of stream channel over the course of the study depending on discharge.

All statistical analyses were performed using metabolism measurements that were expressed per unit channel area (g O_2 m^{-2} day^{-1}). The distribution of each parameter was tested to see if it met normality assumptions using Shapiro-Wilk test (SAS Institute JMP 2002). Average maximum discharge and rainfall were not normally distributed, and we transformed these data using natural logarithms. We then created a correlation matrix with the environmental variables to test for autocorrelation among the physical, chemical, and biological variables. Variables used in the correlation matrix included: discharge, temperature, total phosphorus concentration, total nitrogen concentration, chloride concentration, DOC concentration, total solar radiation, total suspended solids, suspended fine particulate organic matter, water column chlorophyll a concentration, water column bacteria concentration, rainfall in the previous week, and average maximum discharge for the previous 5 days. If autocorrelation occurred ($p<0.05$) between two variables, only one of the variables would be included in the multiple regression analyses. To determine if there was a linear relationship between discharge and metabolism, we regressed GPP, R, and NEM against discharge. We used forward, backwards, and mixed stepwise multiple regression to determine the relationship between GPP, R, NEM and all of the physical, chemical, and biological characteristics (SAS Institute JMP 2001). However, all three types of multiple regression models produced the same results so only one set of results is presented.

Results

Water temperature during the study ranged from 18.4˚C in October to 29.3˚C in July, 2001. Dissolved oxygen concentrations were always below saturation level and the 24 hour average ranged from 5.98 to 9.03 mg/L or 77% to 96% of saturation. Nutrient concentrations were fairly high. Total P concentration ranged from 48 to 137 $\mu g\ L^{-1}$ and Total N concentrations ranged from 1.7 to 3.8 mg L^{-1} (Table 3.1). Total suspended solids were variable and ranged from 6 to 21 mg L^{-1} (Table 3.1). Net ecosystem metabolism was consistently negative, indicating heterotrophy. Respiration ranged from 12.5 to 39.2 g $O_2\ m^{-2}\ day^{-1}$ (Table 3.1). Gross primary production was more variable, ranging from 0.4 to 4.7 g $O_2\ m^{-2}\ day^{-1}$ (Table 3.1).

Metabolism was not linearly related to discharge. There were no significant linear correlations between discharge and GPP ($r^2 = 0.11$; $p = 0.24$), R ($r^2 = 0.13$; $p = 0.2$), or NEM ($r^2 = 0.16$; $p=0.15$). Stepwise multiple regression analyses indicated that GPP was negatively correlated with ln average maximum discharge for the previous five days ($r^2=0.28$ $p=0.05$, Figure 3.2, Table 3.2). Other environmental factors that did not meet the criteria to enter the model ($p<0.05$) included discharge, rainfall, total N and total P concentration, total suspended solids, total solar radiation, chlorophyll a concentration, and R. Temperature was correlated with solar radiation, chloride concentration was correlated with total N concentration, and suspended FPOM was correlated with total suspended solids; therefore, these parameters were not included.

A multiple regression model including both suspended FPOM and total P concentration explained 72 % of the variation in R (Table 3.2). We did not have total P concentration for one date; therefore, the multiple regression model included 13 out of the 14 measurements. When all 14 dates were included in a univariate analyses of R and suspended FPOM there was still a significant positive correlation ($r^2=0.49$ $p=0.005$ Figure 3.3). The environmental factors that did not meet the criteria ($p<0.05$) of entering the stepwise multiple regression model included: temperature, average maximum discharge for the previous 5 days, total N concentration, DOC concentration, water column bacteria concentration. In addition to the correlations described above, discharge was correlated with water column bacteria concentration.

Net ecosystem metabolism correlated with the same two parameters as R. A stepwise multiple regression model including suspended FPOM and total P concentration explained 71% of the variation in NEM (Table 3.2). Again we were missing total P concentration for one date, so we ran a univariate analysis of NEM and suspended FPOM and there was a significant negative correlation (r^2=0.45 p=0.009). The environmental parameters that did not meet the criteria to enter the model were the same as those listed in the GPP and R models combined.

Although NEM and R did not correlate with discharge in a linear fashion, the plots indicated that there may be a curvilinear relationship between the two measures of metabolism and discharge over the range of discharge sampled. We predicted that transient storage zone area would increase with discharge and result in increases in R. Two factors may have confounded our prediction. First, the range of 24-hour mean discharge over which we measured metabolism was lower than the range of discharge experienced during the week when hydropeaking is occurring. For example, we measured metabolism between 30 and 50 m^3s^{-1}, during a week in early July 2000, discharge ranged between 50 and 90 m^3s^{-1} on Monday, Tuesday and Wednesday. Therefore, we may not have sampled over a wide enough range of discharge to detect the relationship between mean daily discharge on the weekend and metabolism. In addition, as we discussed earlier, the channel is confined by high banks which results in narrow sand bars and prevents water from moving laterally as discharge increases. As a result, transient storage zone size will not continue to increase with increasing discharge once the side channel and sand bars areas have been inundated. Therefore, R may not continue to increase, but rather will plateau once the discharge is high enough to inundate all side channel and sand bar areas. The same phenomenon could occur with snag inundation as well. R will increase as long as increasing discharge inundates more snags. However, once all the snags have been inundated, R may no longer increase and reach a plateau instead. In addition, because the river is highly heterotrophic, we expected NEM to have similar relationship to discharge as R. Therefore, we tested to see if there was a curvilinear relationship between discharge and R and discharge and NEM.

We tested two curvilinear models. First, we determined if there was a logarithmic relationship between R and discharge using the following equation:

$$\mathrm{R} = a \times \ln(\mathrm{Q} - 30) + b \qquad (1)$$

Where R is respiration, Q is discharge, and a and b are constants. As discussed earlier, low flow conditions in regulated rivers are controlled and therefore flow is more likely to approach a boundary other than zero, we subtracted discharge by 30 m^3s^{-1} because the lowest mean daily discharge for the 65 year record at the gauge is 30.3 m^3s^{-1}. There was not a significant logarithmic relationship between R and discharge (r^2=0.22 p=0.09), or between NEM and discharge (r^2=0.26 p=0.06). If R plateaus, the data would be bounded on both sides. Therefore, we also tested a hyperbolic relationship using the following equation:

$$\mathrm{R} = b\left(\frac{(\mathrm{Q} - 30)}{(\mathrm{Q} - 30) + a}\right) \qquad (2)$$

Where R is respiration, Q is discharge, and a and b are constants. There was a significant hyperbolic relationship between R and discharge (r^2=0.37 p=0.02). However, the relationship did not explain as much of the variation in R as the stepwise multiple regression model. There was also a significant hyperbolic relationship between NEM and discharge (r^2=0.43 p=0.01), but again it did not explain as much of the variation as the stepwise model. However, the non-linear relationship between R and discharge does suggest that transient storage zone areas and snags may contribute to the control of R in this river.

The proportion of whole system GPP that could be attributed to water column GPP was highly variable ranging from 0.03 to 0.9 and averaged 0.38 for the six dates. The proportion of whole system R that could be attributed to water column R was also variable ranging from 0.04 to 0.69 and averaged 0.34 for the six dates. Therefore, water column processes accounted for a little more than 1/3 of whole-system metabolism. Variation in the proportion of ecosystem metabolism that could be attributable to water column processes was not related to any of the environmental parameters.

Chlorophyll a concentrations from snag biofilm ranged from 24 to 52 g m^{-2} of wood sampled. Converting water column concentrations from a volumetric measure to an aerial measure by multiplying by depth indicates that snag chlorophyll a concentrations are three orders of magnitude higher than water column concentrations (Table 3.1). Bacteria concentrations in the snag biofilm ranged from 1.8×10^{8} to 1.5×10^{9} cells cm^{-2} of wood sampled. Therefore, bacteria concentrations on snags are one to two orders of magnitude higher than water column concentrations (Table 3.1). Therefore, snags appear to be areas of high biological activity. We have measures of snag chlorophyll a and bacteria on only four dates with metabolism measurements. Therefore, we could not include snag chlorophyll a or bacteria standing crop in the multiple regression models.

Discussion

The Chattahoochee River below Atlanta is a highly heterotrophic system as indicated by the consistently negative NEM values. There was a ten-fold variation in GPP over the course of late spring, summer, and early fall. Contrary to our prediction, GPP was not correlated with low flow discharge or turbidity measures. Average maximum discharge for the five weekdays prior to metabolism measurements was negatively correlated with GPP and explained 28% of the variation in GPP. Average maximum discharge was not correlated with rainfall, and therefore appears to be a function of hydropeaking. A two year study of metabolism in an undamned gravel bed river in Switzerland indicated that primary production was reduced an average of 37% by bed-scouring floods (Uehlinger 2000). Due to the upstream hydropower dam and substratum dominated by fine sediments, the Chattahoochee River below Atlanta has bed-moving floods almost every week day. A doubling of discharge during a 24-hour period was common during the course of this study. These results suggest that hydropeaking reduces GPP and is likely the result of disturbance to benthic and snag periphyton community. Thus, discharge appears to affect GPP in this system, but it is the peak discharges resulting from power generation that reduce GPP, rather than the low-flow discharge regime.

Respiration was less variable than GPP, with only a three fold difference between the minimum and maximum values. The model that explained the greatest amount of variation in respiration was a function of both suspended FPOM and total P concentration. These results indicate that suspended FPOM is an important source of organic matter for respiration in this system. Because of the large inputs from wastewater treatment plants, much of the suspended FPOM is likely highly labile sewage-derived POM (deBuryn and Rasmussen 2002). In addition, the organic matter content of the benthic sediments (2%) is much lower than the organic matter content of the suspended sediments (average across all dates 60%). We did not measure the benthic organic matter standing crop and therefore cannot determine its role in this system. However, Webster et al. (1995) suggest that the effect of benthic organic matter storage on respiration rate in eastern U.S. rivers is not strong. Because R was highly correlated with POM in the water column, it appears that the river is processing organic matter from the wastewater treatment plants which helps maintain water quality and prevent organic matter loading to the reservoir downstream.

A multi-biome study of metabolism in less-modified small streams indicated that respiration rate was partially controlled by phosphorus concentrations and that nutrient limitation may be an important control on heterotrophic metabolism in streams (Mulholland et al. 2001). Other research in this system has demonstrated that Chattahoochee SRP uptake velocities are significantly lower than those found in less modified streams and there is a significant amount of biologically available phosphorus being stored in the sediments (Chapter 4). Because of these elevated concentrations, it is surprising to find a positive relationship between R and total P concentration. Regardless, organic matter and phosphorus loading appear to control respiration in the Chattahoochee River, which suggests that that the large volumes of wastewater treatment plant effluent are affecting ecosystem processes in this river.

Contrary to our hypothesis, respiration did not vary with discharge in a linear fashion. However, there was a significant hyperbolic relationship between R and low flow discharge (Figure 3.4). R increased sharply with increasing discharge at low discharges (below 35 m^3s^{-1}), but increased only slightly with discharge between 35 and 50 m^3s^{-1}. Previous research has found positive relationships between transient storage zone area and R (Mulholland et al. 2001, Fellows et al. 2001). The increase in R with increases in transient storage zone size is thought to be the result of greater storage of organic matter in the slow moving areas and increased surface area for heterotrophic microbes (Mulholland et al. 2001). Transient storage zone includes the hyporheic and surface areas such as side channels and eddies where water moves more slowly than in the main channel (Harvey et al. 1996). While we did not measure transient storage zone size on each sampling date, increased discharge resulted in increased inundation of debris dams and sand bars, which are areas that serve as surface transient storage zones. In addition, because the channel is confined by high banks, water cannot move laterally and therefore the increase in transient storage zone size as a result of increase in discharge is not likely to be a linear relationship. Therefore, these results suggest that R increases with transient storage zone size in the Chattahoochee River. However, the effect of discharge is not as strong as the effects of suspended FPOM and total P concentration. The effects of organic matter and phosphorus loading appear to overwhelm the effects of discharge on respiration in the Chattahoochee River for the range of discharge we measured.

Snags in the channel had high concentrations of bacteria and therefore are likely to be areas of high heterotrophic respiration. Over the range of discharges sampled during this study, there was a positive linear relationship between area of wood in the channel and discharge (r^2=0.98 $p<0.001$). Therefore, it does not appear that snags contribute to the hyperbolic relationship between discharge and R.

Respiration dominated net ecosystem metabolism (NEM), and as a result there was also a three-fold variation in NEM. The model that explained the largest amount of variation in NEM was a function of both suspended FPOM and total P concentration (Figure 3.3, Table 2). There was a hyperbolic relationship between discharge and NEM similar to the respiration model. The results of the net ecosystem metabolism model suggest that NEM decreases (becomes more negative) with discharge at low discharges (below 35 m^3s^{-1}), but then plateaus and decreases only slightly with increasing discharge between 35 and 50 m^3s^{-1}. However, the effects of suspended FPOM and total P concentration were stronger than the effect of discharge again demonstrating the importance of the point source inputs from wastewater treatment plants of ecosystem metabolism.

On average, water column processes accounted for almost 40% of GPP in the Chattahoochee. However, this proportion was highly variable (0.03 to 0.9). In the Ogeechee River, a sandy blackwater river, water column processes accounted for 49 % of metabolism on one date, but this proportion was likely highly variable due to changes in depth and turbidity (Edwards and Meyer 1987). The Chattahoochee River is well-mixed; therefore, the size of the photic zone is likely to be controlled by suspended solids, DOC, and depth. If variation in water column processes were responsible for the majority of variation in GPP, we would expect significant relationships between GPP and water column chlorophyll a, turbidity, and/or discharge. The lack of relationships between GPP and these variables suggest that, although the water column can be productive, it is not responsible for the majority of the variation in GPP over the course of this study. Instead, disturbance as a result of hydropeaking has a greater affect on GPP.

Water column respiration was also highly variable, but on average accounted for a little over one-third of whole-system respiration. However, the strong positive relationship between R and suspended FPOM suggests that water column processes contribute significantly to the variation in R. Water column R accounted for 26% of whole system R on one date in the Ogeechee River (Edwards and Meyer 1987). However, in the Ogeechee, benthic processes appeared to account for the majority of the variation in R (Edwards and Meyer 1987).

Snags appear to be locations of significant amounts of primary production and respiration. Although we cannot account for the effects of algae and bacteria in the snag biofilm on metabolism, the range of inundated snag surface area did not vary substantially during the course of the study (3.8 - 4.2 m^2 of wood/m stream channel). Therefore, for the narrow range of low-flow conditions we sampled, the importance of productivity and respiration associated with snags was probably similar across all dates. Variation in metabolism as a result of variation in snag inundation would be more pronounced across a wider range of discharge, or after periods of extreme high flows that alter the snag surface area in the reach through deposition, removal, or scouring of snags. Therefore, it seems likely that variation in the area of snags that are inundated and the amount of algal and bacterial biomass on the snags could account for inter-annual variation in metabolism or seasonal variation in metabolism. This is another reason why GPP correlated with average maximum discharge during five previous days (point made above).

Compared to other rivers of a similar size, the Chattahoochee River below Atlanta has a lower GPP and higher R than might be expected (Table 3.3). Mean summertime GPP in the Ogeechee River, GA and Vermillion River, IL, USA and mean annual GPP in the River Thur, Switzerland were twice the mean GPP in the Chattahoochee (Table 3.3). However, mean R for this study was three times higher than mean summertime R for the Ogeechee, the most downstream site on the Vermillion, and mean annual R for the River Thur, but similar to an upstream site on the Vermillion River (Table 3.3). The Vermillion is heavily impacted by agriculture and some small urban areas (Wiley et al. 1990); the River Thur receives waste water treatment plant effluent (Uehlinger 2000); and the Ogeechee has high DOC concentrations and receives large amounts of organic matter from floodplain swamps (Meyer and Edwards 1990). Therefore, it is surprising how much higher R is in the Chattahoochee River.

There are several potential reasons for the low GPP and high respiration in the Chattahoochee River including quality of organic matter, amount of wastewater effluent input and the sewage-derived POM concentration of the effluent. Comparing the relatively unpolluted streams to those streams receiving agricultural or point source inputs (Table 3.3) indicates that GPP was approximately doubled by agricultural and point source inputs (mean unpolluted = 2.5 g O_2 m^{-2} day^{-1}, mean modified = 5.5 g O_2 m^{-2} day^{-1}, Welch's $t = 2.00$ $p = 0.06$). Although algal biomass often increases with nutrients in low nutrient streams, light effects can be much stronger (Lowe et al. 1985). Therefore, the effects of agricultural and point-sources on GPP may not be as strong as expected due to light limitation by riparian trees in less-modified streams or high turbidity in urban and agricultural streams. Agricultural and point-source inputs had a stronger effect on R. R was almost three times higher in the modified streams compared to the less modified systems (mean unpolluted = 4.5 g O_2 m^{-2} day^{-1} mean modified = 12.1 g O_2 m^{-2} day^{-1}, Welch's t = 3.5 p = 0.003).

An objective of this study was to evaluate one type of ecosystem services provided by the Chattahoochee River. Although improved technology at wastewater treatment facilities typically prevents deoxygenation of the river as a result of sewage-derived POM inputs, oxygen concentrations are well below saturation on all dates. Therefore, increasing GPP would increase dissolved oxygen concentrations in addition to increasing algal-based food resources for aquatic insects and fish. GPP was negatively affected by hydropeaking presumably as a result of disturbance to algal mats in the sediments and on snags. Therefore, dam operations decreased primary production, and decreasing hydropeaking should increase GPP in the Chattahoochee River. R in the Chattahoochee River was higher than in most other river systems including the Ogeechee River, a blackwater river with high DOC concentrations are large organic matter inputs from the floodplain, and the River Thur, a gravel bed river also receiving wastewater treatment plant inputs. This indicates that the Chattahoochee is processing sewage-derived POM. High R also indicates that organic matter and nutrient loading are increasing R and oxygen demand in the river. However, by processing sewage-derived POM through respiration, the Chattahoochee River is helping to maintain downstream water quality and providing a critical ecosystem service to the human communities in this watershed.

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Figure Captions

Figure 3.1: Map of study reach in the Chattahoochee River below Atlanta, GA. Buford Dam is a large hydroelectric dam. Morgan Falls dam is a smaller re-regulation dam, but also produces hydroelectric power.

Figure 3.2: Results of the multiple regression analyses between environmental variables and GPP. GPP declined significantly with increasing average maximum discharge for the 5 day prior to metabolism measurements ($\mathrm{GPP} = -1.81(\ln Q_{max}) + 9.32$). Q_{max} appears to be a function of hydropeaking as it was not correlated with rainfall. GPP was not correlated with any of the other environmental parameters.

Figure 3.3: Significant correlations between respiration and suspended fine particulate organic matter (A) and net ecosystem metabolism (NEM) and suspended fine particulate organic matter (B).

Figure 3.4: Relationship between discharge and NEM (panel A) and discharge and R (panel B). A hyperbolic equation best described the relationship between NEM and discharge ($\mathrm{NEM} = -36.96\left(\frac{(Q-30)}{(Q-30)+3.67}\right)$, $r^2 = 0.43$ $p = 0.01$) and the relationship between R and discharge ($R = 36.47\left(\frac{(Q-30)}{(Q-30)+2.62}\right)$, $r^2 = 0.37$ $p = 0.02$).

Table 3.1: Summary of environmental conditions on all sampling dates. Q stands for discharge. Blanks indicate no data.

Date	Q (m^3s^{-1})	Temp (˚C)	Total P (µg/L)	Total N (mg/L)	Chloride (mg/L)	TSS (mg/L)	Total Radiation	DOC (mg/L)	Water col. Chl. a (µg/L)	Water col. Bacteria (#/ ml)	sus. FPOM (mg/L)	avg. max Q (m^3s^{-1})	GPP (g O_2 $m^{-2}d^{-1}$)	R (g O_2 m^{-2} d^{-1})
6/26/00	43.41	23.73	79	2.24	14.14	7	$2.35\cdot10^4$	2.62	7.5		6.4	79.95	1.84	15.28
7/3/00	32.11	24.92	48	2.93	15.46	9	$2.6\cdot10^4$	2.64	4	3.16×10^6	4.6	76.74	1.47	12.47
7/16/00	32.51	28.3	137	3.1	14.94	8.1	$2.85\cdot10^4$	2.53	10.2	1.89×10^6	4.9	77.94	1.79	21.08
8/7/00	36.84	28.2	98	1.70	16.04	16	$2.05\cdot10^4$	2.67	8.2	2.61×10^6	5.3	92.03	1.93	28.14
4/14/01	45.65	22	86	3.25	12.22	14	$1.81\cdot10^4$	2.77	4.2	5.58×10^6	9.1	49.18	2.37	39.17
5/5/01	37.38	25.67	81	1.79	15.8	7.5	$2.73\cdot10^4$	2.19	11.1	3.01×10^6	8.5	40.83	3.01	31.77
7/7/01	35.23	28.56	146	2.40	10.77	12.4	$2.23\cdot10^4$	3.18	4.06	2.91×10^6	6.5	73.41	0.43	31.91
7/14/01	33.87	29.25	40	2.22	18.5	21.2	$2.32\cdot10^4$	2.44	4.25	3.34×10^6	11.6	35.31	3.42	34.95
7/22/01	36.59	26.59		2.76	16.79	17.3	$2.53\cdot10^4$		2.67	3.03×10^6	7.4	46.32	2.92	38.79
8/25-26/01	40.76	26.81	99	2.97	16.73	16.6	$2.12\cdot10^4$	2.52	8.5	2.35×10^6	8	45.36	1.14	33.94
9/1/01	47.52	24.3	82	2.96	25.93	20	$9.77\cdot10^3$	2.35	10.95	3.64×10^6	10	54.77	1.11	32.11
9/15-16/01	36.95	23.89	78	3.06	24.36	12.7	$2.01\cdot10^4$	2.41	8.8	1.56×10^6	6.4	46.27	1.67	22.57
10/7/01	39.59	18.74	81	3.22	32.5	6	$1.94\cdot10^4$	2.44	9.35	4.72×10^6	4.2	37.44	1.67	22.88
10/20-21/01	32.35	18.38	75	3.84	24.49	10.3	$1.6\cdot10^4$	2.69	12.2	2.5×10^6	5.7	38.91	4.74	16.84

Table 3.2: Results of stepwise multiple regression analysis for rates of gross primary production (GPP), respiration (R), and net ecosystem production (NEM). (n=14 for GPP and n=13 for R and NEM). See text for environmental parameters that did not meet criteria for entering model.

Dependent Variable	Independent Variable	Parameter estimate (SE)		r^2	Prob>F
GPP	Intercept	9.323	(3.32)		0.016
	ln Q_{max}	-1.808	(0.83)	0.28	0.049
R	Intercept	-7.4	(7.4)		0.34
	Total P suspended	0.126	(0.05)	0.17	0.032
	FPOM	3.26	(0.66)	0.55	0.001
	Full Model			0.72	0.002
NEM	Intercept	10.85	(7.76)		0.19
	Total P suspended	-0.14	(0.05)	0.20	0.022
	FPOM	-3.24	(0.69)	0.50	0.001
	Full Model			0.70	0.002

Table 3.3: Comparison of metabolism in streams modified by agriculture and point-source inputs and less modified streams. We only used data from streams with discharge > 0.05 m^{3}s^{-1}.

Site Name	Catchment Area (km^2)	Discharge (m^3 s^{-1})	GPP (g O$_2$ m^{-2} day^{-1})	R Mean (g O$_2$ m^{-2} day^{-1})	Reference	human influence	notes
Powder Stream, New Zealand	9.9	0.04	0.6	5.4	Young and Huryn 1999	unpolluted	
Mack Creek, OR, USA		0.06	1.9	11	Mulholland et al. 2001	unpolluted	
Sutton Stream, New Zealand	8.8	0.23	0.8	4.6	Young and Huryn 1999	agriculture	
Eagle Creek, MI, USA		0.2	0.8	6.2	Mulholland et al. 2001	unpolluted	
Big Stream, New Zealand	12.8	0.12	1.5	2.3	Young and Huryn 1999	unpolluted	
Vermillion River, IL, USA	7-21		0.1 - 10.2	6.2 - 34	Wiley et al. 1990	agriculture	range of 3
Three O'clock stream, New Zealand	38.7	0.12	3.7	2.7	Young and Huryn 1999	agriculture	
Vermillion River, IL, USA	26-84		0.1 - 15.5	11.5 - 18.8	Wiley et al. 1990	agriculture	range of 6
Lee Stream, New Zealand	96.5	0.48	1.3	2	Young and Huryn 1999	agriculture	
Mechums River, VA, USA		0.8	4.4	8.7	Hornberger et al. 1977	point-source	
Rivanna River, VA, USA		2.4	6.4	15.5	Hornberger et al. 1977	point-source	
River Necker, Switzerland	126	4.6	0.05 - 12.6	0.7 - 10.3	Uehlinger and Naegeli 1998, Uehlinger 2000	point-source	
Little Tennessee River, NC, USA		7.4	0.9	3.6	McTammany et al. 2003		
Little Tennessee River, NC, USA		7.4	1	2.6	McTammany et al. 2003		
Vermillion River, IL, USA	126-378		8.6 - 44.2	8.5 - 41.6	Wiley et al. 1990	agriculture	range of 7
Eno River, Durham, NC USA			2.4	5.6	Hoskin 1959		
Little Tennessee River, NC, USA		14.8	2.2	2.8	McTammany et al. 2003		
Sycamore Creek, AZ, USA	505	0.03	15	8.4	Mulholland et al. 2001	unpolluted	
Vermillion River, IL, USA	473-890		5.5 - 17	10.6 - 18.4	Wiley et al. 1990	agriculture	range of 5
Little Tennessee River, NC, USA		18.4	3	3.2	McTammany et al. 2003		
Neuse River Smithfield, NC, USA			1.8	2.4	Hoskin 1959	little pollution	
Rappahannock River, VA, USA		22	3	4	Hornberger et al. 1977	unpolluted	
Neuse River, Kinston, NC, USA			0.3	1.7	Hoskin 1959	little pollution	
Black Creek, GA, USA	700	?	0.1-3.6	2.3-9.6	Meyer and Edwards 1990	unpolluted	
Vermillion River, IL, USA	1021		1.5	23.3	Wiley et al. 1990	point + agriculture	
Vermillion River, IL, USA	1130		7	10.7	Wiley et al. 1990	point + agriculture	
Middle Oconee River, GA, USA	1031	53.7	0.1 - 0.3	0.4 - 1.1	Nelson and Scott 1962	little pollution	
River Thur, Switzerland	1696	46.7	0.05 - 20.2	2.4 - 14.6	Uehlinger 2000	point-source	
Vermillion River, IL, USA	3505		4.3	8.3	Wiley et al. 1990	point + agriculture	
Chattahoochee River, GA, USA	5335	37.9	0.4 - 4.7	12.5 - 39.2	This study	point-source	
Ogeechee River, GA, USA	6860	66.4	0.5 - 14	3.7 - 11.5	Edwards and Meyer 1987	unpolluted	summer mean


Buford Dam
Morgan Falls Dam
Atlanta
Chattahoochee
Metabolism Reach
0 5 10 20 Kilometers
N

Figure 3.1

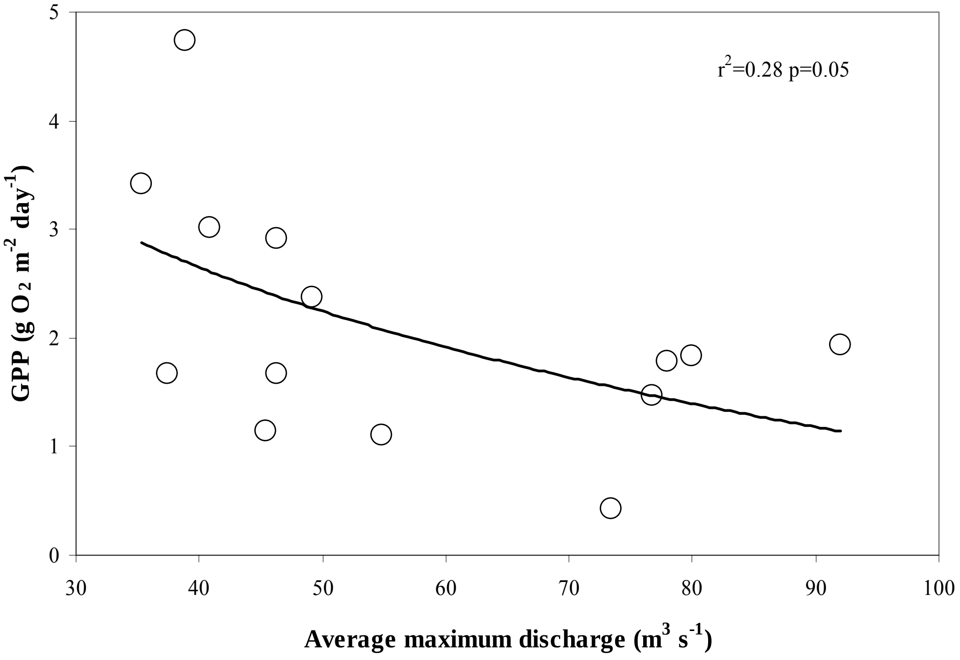
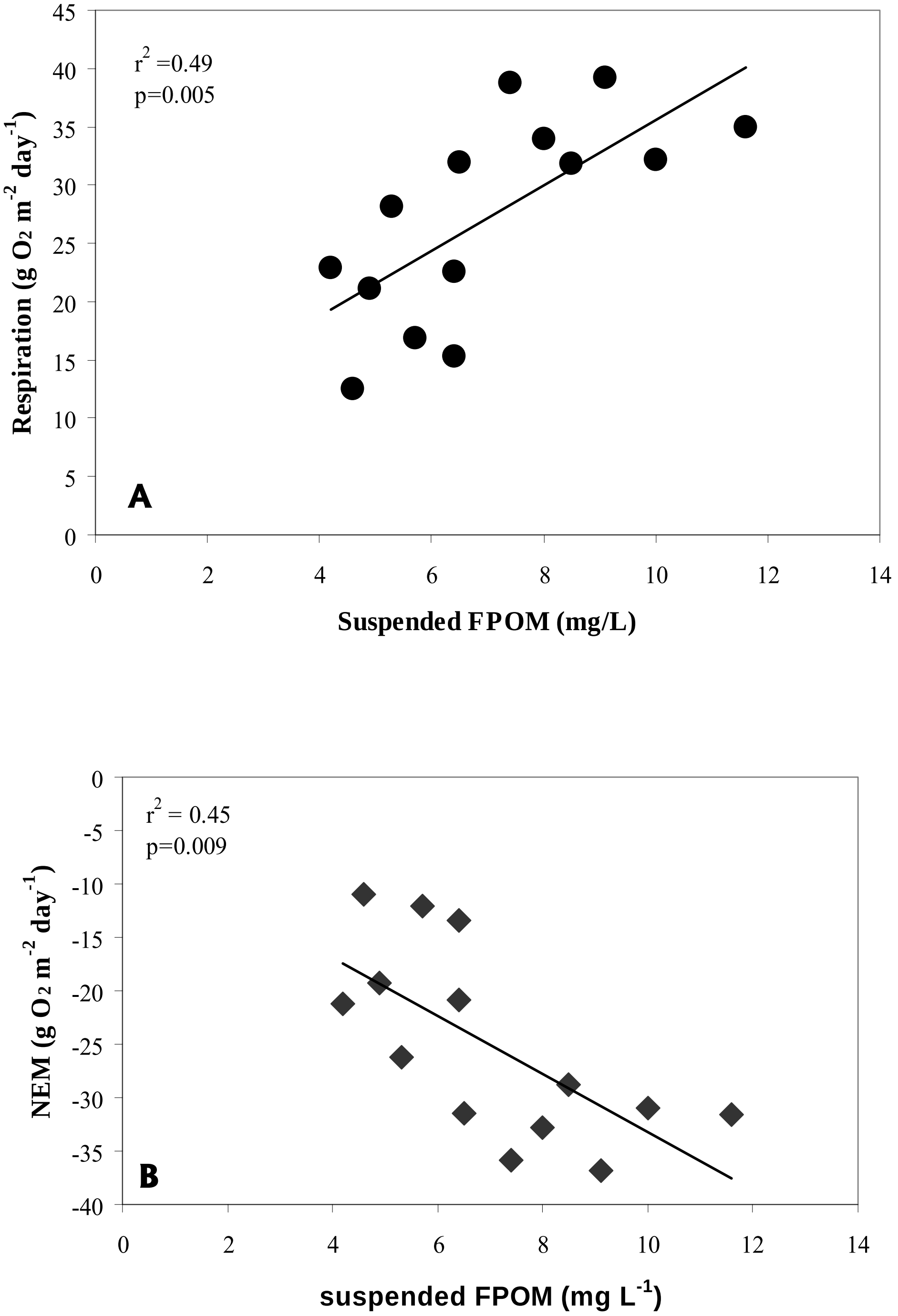

r^2=0.28 p=0.05
GPP (g O$_2$ m^{-2} day^{-1})
0
1
2
3
4
5
Average maximum discharge (m^3 s^{-1})
30
40
50
60
70
80
90
100

Figure 3.2



r2 =0.49
p=0.005
Respiration (g O2 m-2 day-1)
45
40
35
30
25
20
15
10
5
0
A
0
2
4
6
8
10
12
14
Suspended FPOM (mg/L)
r2 = 0.45
p=0.009
NEM (g O2 m-2 day-1)
0
-5
-10
-15
-20
-25
-30
-35
-40
B
0
2
4
6
8
10
12
14
suspended FPOM (mg L-1)

Figure 3.3

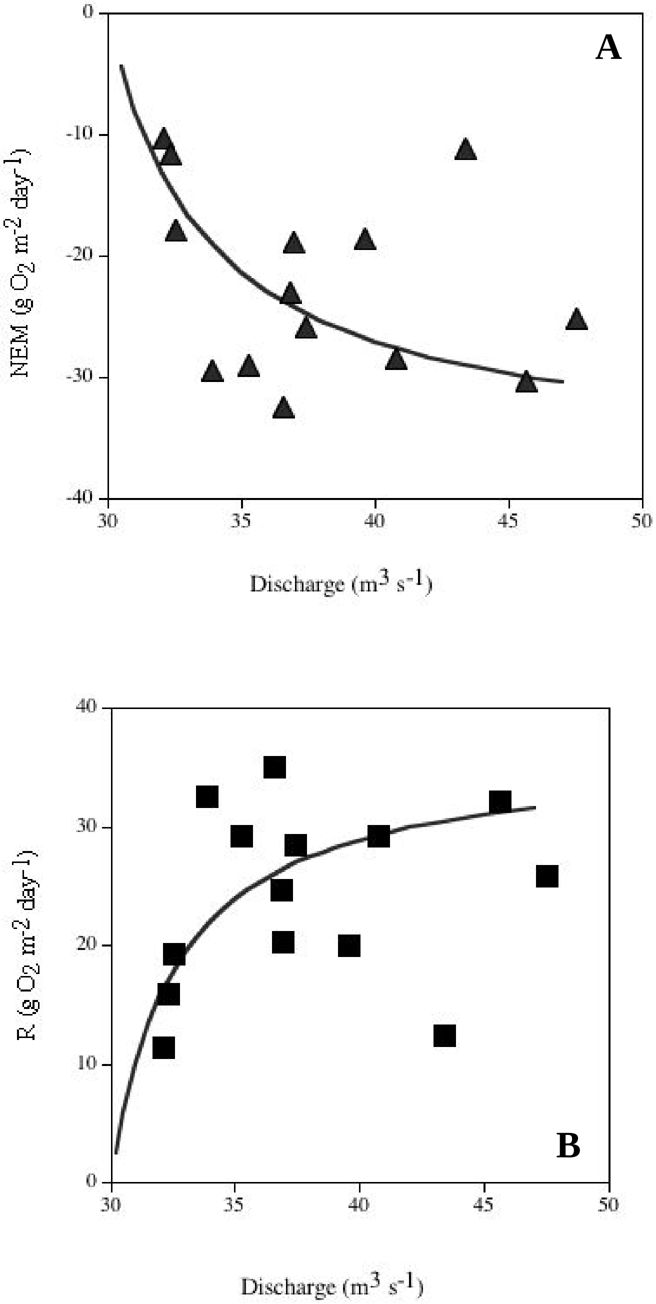

A
NEM (g O$_2$ m^{-2} day^{-1})
0
-10
-20
-30
-40
30
35
40
45
50
Discharge (m^3 s^{-1})
B
R (g O$_2$ m^{-2} day^{-1})
40
30
20
10
0
30
35
40
45
50
Discharge (m^3 s^{-1})

Figure 3.4

CHAPTER 4

NUTRIENT UPTAKE IN A REGULATED URBAN RIVER[3]

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[3] Gibson, C.A. and J.L. Meyer. To be submitted Journal of the American Water Resource Association

Abstract

Nutrient retention and transformation by streams is important in regulating downstream fluxes of nutrients and maintaining water quality. While small streams have been shown to be efficient in retaining nutrients, less is known about nutrient retention in larger rivers. We quantified nutrient uptake length in a regulated urban river to determine the river's ability to retain and transform large inputs of nutrients associated with wastewater treatment plant effluent. In addition, we examined potential environmental controls on nutrient uptake in this river. We measured net soluble reactive phosphorus (SRP), dissolved organic phosphorus, ammonium (NH_4), nitrate (NO_3), and dissolved organic nitrogen (DON) uptake in the Chattahoochee River downstream of Atlanta, GA by following the downstream decline of nutrients below a series of wastewater treatment plants in two sections of a 20.2 km reach on 10 dates under low flow conditions. We predicted that uptake length would increase with increasing discharge. Uptake of all the nutrients was sporadic, and uptake length did not increase with discharge for any of the nutrients. When uptake occurred, SRP uptake length was negatively correlated with total suspended solids and temperature. However, measurable uptake in the upper reach was typically followed by increases in concentration in the lower reach. Nitrogen uptake was less consistent than SRP uptake. On dates when NO_3 uptake occurred, NO_3 uptake velocity was positively correlated with suspended fine particulate organic matter indicating biological uptake of NO_3. There was evidence of DON mineralization on almost half of the dates. Chattahoochee SRP and NH_4 uptake velocities were lower than velocities in less-modified systems, but NO_3 uptake velocities were similar. These results demonstrate that despite severe modifications by humans, a large, regulated, urban river is capable of retaining and transforming nutrients although in a much diminished capacity.

Introduction

Humans depend on freshwater ecosystems to provide resources such as clean water and certain ecosystem services such as transportation and waste assimilation (Naiman and Turner 2000, Jackson et al. 2001). Increasing human population, water demand, and changes to land cover and use are causing the demand for freshwater and associated ecosystem services to increase (Postel et al. 1996, Naiman and Turner 2000). In addition, it is uncertain how modifications that are associated with human demands such as dam construction, increased water demand, and increased nutrient, contaminant, and organic matter loading affect the ability of freshwater ecosystems to provide these resources and services.

Streams alter the form and amount of nutrients transported from terrestrial ecosystems to downstream ecosystems through uptake and transformation (Stream Solute Workshop 1990, Alexander et al. 2000). This retention and transformation can be important in regulating downstream nutrient fluxes (Peterson et al. 2001). Anthropogenic nutrient loading of terrestrial and aquatic systems is occurring throughout the world (Caraco 1993, Caraco 1995, Vitousek et al. 1997, Caraco and Cole 1999, Bennet et al. 2001). However, the export of these nutrients from the watershed depends on a variety of factors including hydrology, geology, alterations of the landscape, and the extent of anthropogenic loading (Caraco and Cole 1999, Peterson et al. 2001). Small streams have been shown to be efficient in retaining nutrients across a variety of biomes (Peterson et al. 2001). Although not well-studied compared to their less-altered counterparts, small streams receiving point-source inputs (such as waste water treatment plants) are typically less efficient in retaining both nitrogen and phosphorus inputs (Marti et al. 2004, Haggard et al. 2001, Pollock and Meyer 2001). In addition, nitrogen removal in streams decreases with increasing channel size primarily as a result of increases in depth (Alexander et al. 2000). Urban rivers typically receive large amounts of point-source inputs, and society depends on rivers to retain and transform these inputs to maintain downstream water quality. Empirical measures of nutrient uptake in large rivers are scarce, and, therefore, there is limited information on the ability of larger rivers to perform these services.

In rivers, the transport and transformation of nutrients is tightly linked with the physical movement of water. In flowing water, nutrient cycles are longitudinally extended to become spirals (Webster and Ehrman 1996). Moreover, greater hydrologic variability can lead to large variations in solute turnover lengths (Meyer and Edwards 1990). Therefore, the movement of nutrients in a system can be affected by flow, and changes in flow will influence the transport of nutrients to downstream ecosystems. Although there are many different types of hydrologic and channel alterations that result in changes in the flow regime, dams are one of the most conspicuous and prevalent forms of flow alteration on large and some smaller rivers and streams (Dynesius and Nilsson 1994). In the contiguous United States, there are only 42 rivers with greater than 200 river km unregulated by major dams (Benke 1990). Despite the prevalence of dams and recent focus on the impacts of dams on freshwater ecosystems (see Poff et al. 1997 and references therein), there is little information about the effects of dams and concomitant changes in flow regimes on nutrient uptake.

We measured nutrient uptake in a regulated river downstream of metropolitan Atlanta, GA, USA. The objectives of this study were to: 1) Quantify nutrient retention in a regulated urban river receiving wastewater treatment plant effluent and relate this to the level of ecosystem services being provided by the river; 2) Determine how nutrient uptake lengths vary with discharge and other environmental parameters to examine what controls nutrient uptake in large urban rivers. We hypothesized that nutrient uptake lengths would increase with discharge as seen in many small streams (Butterini and Sabater 1998, Peterson et al. 2001, Hall et al. 2002).

Methods

Study Site

We measured nutrient retention of a 24 km reach of the Chattahoochee River in Fulton County, Georgia just below Atlanta, Georgia (Figure 4.1). Discharge is strongly influenced by two upstream dams, Buford Dam, a large hydropower dam, and Morgan Falls Dam, a smaller re-regulation hydropower dam. The generation of hydropower causes large fluctuations in discharge within and between days. The Chattahoochee River provides drinking water for portions of Atlanta and receives most of its wastewater effluent (Frick et al. 1996). Nine municipal wastewater treatments plants upstream of this site discharge approximately 7.25×10^8 L day^{-1} of effluent into the river (EPA Permit Compliance System 2001).

Substrate in the river consists primarily of sands and silts. One large shoal was present at the half-way point in the reach, where the substrate consisted of large boulders. The study reach was characterized by high, steep banks that prevented the connection of the river and the floodplain except during extremely high discharge. The steep banks confine the river so that wetted width remains fairly similar over the range of discharges in this study. There is an intact narrow riparian corridor present along the majority of the reach, and large debris dams were present throughout the reach. In 2001, snag surface area in a 1 km reach immediately downstream of the Highway 166 bridge ranged between 3.75 and 4.02 m^2/m of stream channel over the range of discharges in this study (Chapter 3).

Power generation at the upstream dam causes large fluctuations in discharge. A doubling of discharge over a 24 hour period was common during the period of this study. To avoid these large changes in discharge, we conducted most of our sampling on weekends because hydropower generation does not usually occur on week-ends. There were 5 larger and several smaller tributaries that entered into the reach, but due to drought conditions only the 5 larger tributaries were flowing at the start of the study in 2001 and only two were still flowing after the first 2 weeks of the study. Both 2000 and 2001 were drought years throughout Georgia. The 37-year average discharge at the beginning of the reach for the five month period of June to October is 82.7 m^3s^{-1} (USGS station # 02337170, Chattahoochee River at Fairburn, GA). In 2000 average discharge for this period was 66.1 m^3s^{-1}, and in 2001 average discharge was 51.3 m^3s^{-1}.

During summer 2000, we conducted a preliminary study of nutrient uptake by sampling at 3 points in the reach 0, 6.8, and 24.1 km downstream of the Highway 166 bridge crossing, and at a site upstream of Atlanta for background concentrations. The preliminary study suggested more extensive sampling was needed. Therefore, in 2001, we sampled the same background reference point and 14 points along the 24 km reach of the river. The start of the reach was moved 3.6 km downstream of the Highway 166 bridge to a point below a relatively small wastewater treatment plant (Figure 4.1). The average discharge of this plant is $1.9 \cdot 10^7$ L day^{-1} (EPA Permit Compliance System 2001). In addition, we divided the 24 km reach into two separate reaches above and below a sand pumping operation located 14.5 km downstream from the Highway 166 bridge crossing (Figure 4.1). The two tributaries that flowed throughout the study were located in the lower portion of the reach (Figure 4.1). Although the reach length is much longer than that used in typical nutrient uptake studies, the river is also much larger than previous studies. Streams receiving wastewater treatment plant effluent typically have longer uptake lengths than those not receiving effluent (Marti et al. 2004). Therefore, we considered it necessary to use a long reach.

Nutrient Uptake

Uptake lengths were determined by calculating the slope of the line relating nutrient concentration to downstream distance using the following equation:

$$\ln C_x = \ln C_0 - k_c x \quad (1)$$

Where C_x is the nutrient concentration (x) meters from the top of the reach, C_0 is the nutrient concentration at the top of the reach, and k_c is the per meter uptake rate of the nutrient (Newbold et al. 1981). Uptake length is equivalent to k_c^{-1} and represents the average downstream distance traveled by a nutrient atom before removal from the water column (Newbold et al. 1981). To correct for dilution, nutrient concentrations were divided by the concentration of the conservative tracer (Webster and Ehrman 1996). We used wastewater treatment plant effluent as the source of nutrients and conservative tracers, chloride and fluoride (Marti et al. 2004, Haggard et al. 2001, Pollock and Meyer 2001). We took one water sample from the thalweg at each sampling point and filtered all the samples in the field. We measured the uptake length of dissolved organic phosphorus (DOP), soluble reactive phosphorus (SRP), dissolved organic nitrogen (DON), nitrate (NO_3^-), and ammonium (NH_4^+) over a range of discharges in 2000 and 2001. Concentrations of total dissolved phosphorus, SRP, NO_3, and total dissolved nitrogen were determined using the methods of Wetzel and Likens (2000), NH_4^+ was determined either by the phenol hypochlorite method (2000 samples, Wetzel and Likens 2000) or fluorometrically (2001 samples, Holmes et al. 1999). Chloride was determined with an ion chromatograph (UGA Soil Ecology Lab) and fluoride was determined with an ion specific electrode method (adapted from Froelich et al. 1983).

Although we sampled both chloride and fluoride as conservative tracers, fluoride concentrations are less likely to be affected by non-point source inputs in the reach (Table 4.1, see below). Therefore, we used fluoride as the conservative tracer for calculating nutrient uptake. We calculated uptake for the upper 8.9 km (above the sand pumping operation) and the lower 11.6 km of the 24.1 km reach separately for all the 2001 dates.

We calculated uptake velocity (V_f, also referred to as mass transfer coefficients) to compare uptake in the Chattahoochee River with other streams. Uptake velocity corrects for differences in uptake length caused simply by differences in water velocity and is defined as the velocity at which a nutrient moves downward through the water column toward the sediments and is calculated as:

$$V_f = \frac{Q}{w\,(S)} \tag{2}$$

where Q is discharge (m^3min^{-1}), w is wetted width (m), and S is nutrient uptake length (m) (Stream Solute Workshop, Hall et al. 2002, Dodds et al. 2002). Due to the size of the river and the reach we were not capable of measuring wetted width throughout the entire channel. However, we were able to measure wetted width on one date at ten transects in the first one km of the reach. The channel is confined by high banks, and, therefore, width does not vary much with discharge. The first km is the one of the narrowest parts of the channel. Therefore, the uptake velocities are probably higher than if we had measured width throughout the channel. However, uptake velocity is still the best parameter to use for comparison of the Chattahoochee River with other systems.

In addition to nutrient concentration, we measured a number of other biotic and abiotic variables to determine what environmental parameters correlated with uptake length. A USGS gauging station is located at the top of the reach at the Highway 166 bridge and provided discharge measurements every 15 minutes. At each sampling point we measured total suspended solids (TSS) and suspended fine particulate organic matter (FPOM) by filtering a known volume of water onto pre-ashed and pre-weighed glass fiber filters (Gelman A/E). Filters were dried at 60˚C for 1 week, weighed, and then ashed for 3 hours at 500˚C and re-weighed. We measured water column chlorophyll a concentrations at each sampling point using the monochromatic method after extracting the pigments by grinding in buffered acetone (Wetzel and Likens 2000). Average daily temperature was measured at the beginning of the reach (Highway 166) using temperature loggers (YSI Sondes, Yellow Springs, OH). We sampled water column bacteria concentrations at each of the three bridge crossings and took the average for the reach. Bacteria densities were determined by direct counts after staining with 5% acridine orange, and counted based on the methods of Kirchman (1993). Rainfall total (mm) for the three days prior to sampling was obtained from a Georgia Automated Environmental Monitoring Site approximately 25 km from the Highway 166 bridge crossing (Dunwoody station).

Because clay is abundant in the bed sediments and previous research has shown strong correlations between suspended solids and total phosphorus concentrations in upstream areas of the Chattahoochee River Basin (Zeng and Rasmussen 2001), we determined the amount of biologically available phosphorus stored in the top 10 cm of the sediments. We sampled three 2.5 cm diameter cores of sediment from five areas at each of the three bridge crossings in summer 2003. We determined biologically available phosphorus using the iron oxide sink method of Meyers et al. (1995, 1997). Briefly, we placed approximately 3 g of wet sediment, 5.5 cm filters impregnated with iron oxide, and 80 mL of 0.01 M $CaCl_2$ in 125 mL glass jars. The jars were placed on a reciprocating shaker and shaken for 16 hrs. The filter was rinsed with deionized water, air dried, and the phosphorus was extracted with 50 ml of 0.2 M H_sSO_4 for one hour. We neutralized the solution using NaOH, and analyzed the soluble reactive phosphorus content colorimetrically (Wetzel and Likens 2000). Concentrations were corrected based on a control of the iron-oxide impregnated filter paper and 80 mL of 0.01 *M* $CaCl_2$. We converted all values to µg SRP cm^{-2} channel bottom per hour. We used the following equation to determine the potential increase in water column P concentration as a result of release from the sediment:

$$C_w = \frac{1}{Q}\left(\frac{C_s}{t} \times A\right) \qquad (3)$$

where C_w (µg L^{-1}) is the water column P concentration resulting from sediment release, C_s (µg cm^{-2}) is the biologically available P concentration in the sediments, *t* is length of time that samples were shaken (16 hrs.), A (cm^2) is the cross-sectional area of the channel, and Q is discharge (L hr^{-1}).

We used multiple regression to determine what environmental factors correlated with nutrient uptake length. All of the nutrients had several dates with no measurable uptake ($p > 0.1$ for regression of equation 1), or increases in concentrations downstream. For these nutrients we used t-tests to compare the environmental parameters on dates with and without measurable uptake, and to compare dates with declines or increases in concentrations. For dates with increases in concentrations downstream, we quantified the increase in concentration by determining the difference between the observed nutrient concentration and the nutrient concentration predicted based on dilution of the conservative tracer. We then regressed this against distance downstream to determine the rate of increase.

Results

Because few stations were sampled during the preliminary study, we do not have confidence in the uptake length calculations. However, by examining the frequency with which we were able to detect measurable uptake, we can get a general picture of what nutrients were being removed from river water during 2000. Measurable uptake of DOP occurred on 1 out of 5 dates and SRP uptake occurred on 3 out of 6 dates. Measurable uptake of NH_4 occurred on 2 of 6 dates, and there was never measurable uptake of NO_3.

In 2001, dilution in the reach was typically low except on 6 July in the upper reach and 13 July in both reaches (Table 4.2). Dilution in the lower reach was typically slightly higher than in the upper reach because the two largest tributaries that flowed throughout the study were located in the lower reach (Table 4.2). Overall dilution estimates based on chloride were similar to those based on fluoride. However, on three dates dilution based on fluoride calculations was higher than those based on chloride, suggesting that tributary inputs contained chloride (Table 4.2).

We were not able to detect nutrient uptake on 6 July in the upper reach or 13 July in both reaches because of high dilution from tributary inputs. Tributary inputs into the river were likely high in nutrients due to the urban setting and interfered with nutrient uptake measures. Therefore, we have excluded these dates from further analyses. We included the remaining data from both upper and lower reaches in multiple regression analyses.

Phosphorus Uptake

There was measurable SRP retention on the seven of eight dates in the upper reach and on three of nine dates in the lower reach (Table 4.3). SRP uptake lengths varied between 11 and 85 km (Table 4.3). Contrary to our hypothesis, SRP uptake length was not positively correlated with discharge, but rather SRP uptake length was negatively correlated with discharge (r^2 = 0.44 p=0.04 Figure 4.2a). This negative correlation was the result of two factors. First, discharge, total suspended solids and temperature were all positively correlated with one another (Q vs TSS, r^2=0.63, p=0.006, Q vs temp r^2=0.63 p=0.006. temp vs. TSS r^2=0.45 p=0.03). Second, SRP uptake length was negatively correlated with TSS (r^2=0.67 p=0.003 Figure 4.2b). Higher TSS and temperature at higher discharges would be expected to increase biotic and abiotic P removal resulting in the observed decrease in uptake length at higher discharges. Therefore, total suspended solids and temperature appear to be driving the negative correlation between SRP uptake length and discharge, but we cannot differentiate the effects of temperature and TSS. Similarly, SRP uptake velocity was positively correlated with TSS ($y = -0.58 + 0.11x$, $r^2=0.4$ $p=0.049$). Uptake lengths were similar in both reaches on the two occasions when there was measurable uptake in both reaches (Table 4.3). On six out of eight dates, SRP increased in the lower reach after measurable uptake or non-significant declines in the upper reach (Table 4.3). SRP increases ranged from 0.4 to 3.9 µg L^{-1} km^{-1} (Table 4.3).

In the upper reach, measurable uptake of organic phosphorus occurred on only one of the eight dates and declines in DOP concentrations occurred on two out of eight dates (Table 4.4). In the lower reach, there was no measurable uptake of DOP, but declines in concentration occurred on four out of nine dates. On the remaining five dates, DOP concentration increased downstream. Significant increase in DOP ranged between 1.1 and 3.9 µg L^{-1} km^{-1} (Table 4.3). Increases in DOP are either the result of the incorporation of SRP into the organic matter or release of organic phosphorus from the sediments.

At the Highway 166 bridge crossing, biologically available phosphorus concentrations in the sediment averaged 10 ± 1.5 µg P cm^{-2}, at the next bridge crossing 6.8 km downstream it averaged 6 ± 1.2 µg P cm^{-2}, and at the end of the study reach the average was 7 ± 2.5 µg P cm^{-2}. Using the range of biologically available phosphorus present in the sediments (6-10 µg P cm^{-2}), we calculate that there is $27.4 \cdot 10^3$ kg of biologically available P present in the sediments of the upper reach and $36.2 \cdot 10^3$ kg present in the sediments of the lower reach. In addition, using the measured range in biologically available P concentration (6-10 µg cm^{-2}) and discharge from dates with increases in SRP in the lower reach, we calculate that available phosphorus in the sediment can account for a water column increase of 0.6 to 3.0 μg P L^{-1} km^{-1}. Therefore, enough P is stored in the sediments to account for all but one of the observed increases in SRP concentration in the lower reach.

Nitrogen Uptake

Nitrogen uptake was much more variable than phosphorus uptake. Measurable ammonium uptake only occurred on two of eight dates in the upper reach and three of nine dates in the lower reach (Table 4.3). Measurable nitrate uptake occurred on four of eight dates in the upper reach and two of nine dates in the lower reach (Table 4.3). Neither ammonium uptake length nor nitrate uptake length was correlated with discharge (Figure 4.3). In fact, none of the environmental parameters correlated with either ammonium or nitrate uptake length. However, nitrate uptake velocity was positively correlated with suspended fine particulate organic matter concentration (FPOM) (r^2=0.81 p=0.014, Figure 4.4). There were no significant differences in discharge, temperature, total suspended solids, suspended FPOM, TN:TP ratios, water column chlorophyll a concentration, or water column bacteria concentration on dates with declines in NH_4 concentrations compared to dates with increases. Similarly, the environmental variables did not differ between dates with declines in NO_3 concentrations compared to those with increases. Significant increases of ammonium occurred on three of eight dates in the upper reach and on four of nine dates in the lower reach (Table 4.3). In both upper and lower reaches, increases in ammonium coincided with non-significant declines in dissolved organic nitrogen suggesting mineralization of nitrogen. Measurable uptake of DON occurred on two of eight dates in the upper reach and on three of nine dates in the lower reach (Table 4.3). On most dates, the two sections of the reach showed opposite trends in concentration for all 3 nitrogen species (Table 4.3).

Discussion

We used wastewater treatment plant effluent as the source of nutrients and conservative tracer because of the size of the river, and because we were interested in quantifying retention of nutrients associated with effluent (Marti et al. 2004). We measured uptake by following the decline in nutrients concentrations downstream from several point-sources. As a result, uptake measures represent net uptake because concentrations are affected by both uptake and release processes (Marti et al. 1997). Nutrient uptake is typically measured using short-term nutrient additions that assume that release of nutrients to the stream does not significantly affect nutrient concentrations during the experiment and therefore reflects gross uptake (Stream Solute Workshop 1990, Mulholland et al. 1990, Marti et al. 1997). An experiment comparing uptake lengths calculated from short-term nutrient additions to those calculated from natural nutrient declines showed that the latter were always longer (Marti et al. 1997). Therefore, we expect the uptake lengths measured by following the natural decline in nutrient concentrations downstream from a point-source discharge to reflect net uptake length and to be longer than what we would find if we were measuring gross uptake.

Nutrient uptake in the Chattahoochee River downstream of Atlanta is highly variable. The element that most consistently demonstrated measurable uptake was SRP. When uptake occurred, uptake length ranged from 11 to 85 km, meaning that half of the phosphorus present in the water column at the beginning of the reach was not removed from the water column until many km downstream. The distance between the Highway 166 bridge and the next downstream reservoir (West Point Lake) is approximately 120 km. Therefore, at the longer uptake lengths, and during periods of non-measurable uptake or increases in concentration, a significant portion of phosphorus added from Atlanta's wastewater treatment plants is being delivered to West Point

Lake. In addition, the shortest uptake lengths in the upper reach were followed with the largest increases in the lower reach (Table 4.3). Therefore, even when uptake is occurring in one portion of the reach, phosphorus is being released in another.

SRP uptake length was correlated with total suspended solid concentration suggesting that retention is the result of phosphorus binding to sediment particles suspended in the water column. By binding to sediments phosphorus will be less available to the biota, however, it is not a long-term sink. Phosphorus retention was temporary as indicated by measurable increases in phosphorus on 5 out of 8 dates in the lower reach and significant amounts of biologically available P released from sediments in our laboratory experiments.

The sediments are storing a large amount of phosphorus. On average, P is accumulating in the world's soil, and this is especially prevalent in U.S. agricultural soils (Carpenter et al. 1998, Bennett et al. 1999, Bennett et al. 2001). In the past 20 years, the highest 4-year average soil test phosphorus for Dane Co., WI, USA was 56 μg P/g soil (Bennett et al. 1999); our laboratory analysis indicated that biologically available P in the bed sediments of the Chattahoochee River averaged 58 μg P/g dry wt sediment. Therefore, P concentrations in Chattahoochee River sediments are similar to those found in fertilized agricultural soils. Phosphorus is readily mobilized from the sediments as the binding and release of P is an equilibrium reaction (Wetzel 2001). Therefore, binding of P to sediments is temporary and does not maintain downstream water quality by retaining P long-term in a form less available to biota. Instead, this P is likely to be mobilized quickly and potentially create water quality problems downstream.

Nitrogen retention was more variable than SRP retention. On the four occasions where there was measurable NH_4^+ uptake, the uptake lengths were relatively short (7 to 20 km). However, there were more dates with increases in NH_4^+ concentration (9) than there were dates with decreases (6). This suggests that these reaches are more likely to be sources of NH_4^+ than sinks. Increases in NH_4^+ concentration are likely to be the result of mineralization of organic nitrogen and subsequent ammonification. Almost half of the experiments (7 of the 16) showed declines in water column DON followed by increases in water column NH_4^+ as would be expected if mineralization was occurring. Other potential sources of organic nitrogen for mineralization include the sediments and suspended organic matter. Organic content of the suspended solids (average across dates 41%) is much greater than the organic content of the sediments (avg 2%), and, therefore, the suspended solids are a more likely source of organic nitrogen for mineralization.

Nitrate uptake was also highly variable, occurring on only 6 of 16 dates. Again, this suggests that the majority of the nitrate is not being retained by the river, but rather exported directly to the next reservoir. There was only one date in which declines in NH_4^+ were associated with increases in NO_3^- as would be expected if nitrification was occurring. Although nitrate uptake length was not correlated with any of the environmental parameters, nitrate uptake velocity was positively correlated with suspended FPOM. Uptake length measurements are confounded by discharge, with longer uptake lengths typically occurring at higher discharge. In addition, suspended FPOM increased with discharge, although not significantly. Therefore, it is not surprising that suspended FPOM correlated with uptake velocity and not uptake length because discharge increases may have offset the increases in suspended FPOM. The positive correlation between suspended FPOM and nitrate uptake velocity suggests that biological uptake of heterotrophs associated with the organic matter controlled nitrate uptake when it occurred. Biological control of nitrate uptake has been shown previously in relatively low nutrient streams (Hall and Tank 2003, Hall et al. 2003).

Contrary to our expectations, nutrient uptake lengths did not increase with discharge as previous studies have demonstrated (Butterini and Sabater 1998, Wollheim et al. 2001, Peterson et al. 2001, Hall et al. 2002). However, we did not sample over a large range of discharges and therefore it is possible that we just did not have the range of discharge needed to see this relationship.

SRP uptake velocities in the Chattahoochee River are significantly lower than those found in less-modified streams (mean less-modified = 4.6 mm/min; mean this study = 0.8 mm/min, Welch's t=6.62 $p<<0.001$; Figure 4.5). Ammonium uptake velocities are also significantly lower in the Chattahoochee River compared to less-modified streams (mean less-modified = 3.9 mm/min, mean this study = 1.0 mm/min, Welch's t = 4.64, df=31, $p<0.001$; Figure 4.6). However, there was no significant difference in NO_3 uptake velocities between less-modified stream systems and the Chattahoochee (mean less-modified = 1.2 mm/min, mean this study = 0.6 mm/min, Welch's t=1.45, df=48, p=0.15; Figure 4.7). Ammonium uptake velocity has been shown to decrease with increasing nutrient concentrations in a multibiome study of small streams (Dodds et al. 2002). The decline in uptake velocity with increasing concentration indicated a saturation of uptake and represents a decrease in the downward movement of nutrients from the water column per unit time (Dodds et al. 2002). Therefore, the lower uptake velocities for NH_4^+ and SRP in the Chattahoochee River compared to other systems indicates slower movement of nutrient from the water column to the sediments, and, potentially, uptake saturation. The range of reported nitrate uptake velocities is lower than the range of reported SRP and NH_4 uptake velocities (Figure 4.7). Therefore, despite differences in stream size and ambient concentration, the demand for nitrate was generally low compared to phosphorus and ammonium. Uptake of phosphorus and ammonium can occur through physiochemical processes (binding to sediments) and biological processes (uptake by autotrophs and/or heterotrophs). However, nitrate uptake can only occur through the biological processes of assimilatory and dissimilatory nitrate uptake. In addition, heterotrophic bacteria do not normally use NO_3^- as a preferred inorganic nitrogen source (Kirchman 1994). Since most of the streams included in the comparison are forested and mostly likely dominated by respiration, the preference of heterotrophic bacteria could explain the lower and less variable NO_3^- uptake velocities.

Previous research in a variety of stream systems has shown control of nitrogen uptake by ecosystem metabolism (Hall and Tank 2003, Hall et al. 2003, Meyer et al. submitted). We measured metabolism in a one km reach immediately below the Highway 166 bridge during the same time period as this study (Chapter 3). Therefore, we attempted to examine the relationship between ecosystem metabolism and nutrient uptake in the Chattahoochee River by regression of uptake velocity with gross primary production (GPP), community respiration (R), and total metabolism (GPP+R). The study reach lengths were 8.6 and 11.1 km for nutrient uptake measurements compared to 1 km for metabolism.

We had measures of both SRP uptake in the upper reach and metabolism on six dates. SRP uptake velocity was not related to either GPP ($r^2 = 0.2$ $p=0.4$), R ($r^2=0.13$ $p=0.5$), or total metabolism ($r^2=0.09$ $p=0.6$). However, given that SRP uptake in the Chattahoochee River is controlled by adsorption to sediments, it is not surprising that ecosystem metabolism does not control SRP uptake. Using data from the upper and lower reaches, we only had measures of NH_4^+ uptake and metabolism for three dates. There was a positive relationship between NH_4^+ uptake velocity and GPP ($r^2=0.8$ $p=0.2$), R ($r^2=0.8$ $p=0.2$) and total metabolism ($r^2=0.8$ $p=0.2$), but the relationships were non-significant with only 3 points. However, the date with the highest total metabolism also had the highest NH_4^+ uptake velocity. Research in headwater 3rd and 4th order streams showed relationships between NH_4^+ uptake velocity and total metabolism (Meyer et al. submitted, Chapter 5). However, NH_4^+ uptake was rarely observed in the Chattahoochee River, and the study reaches were more likely to be sources of NH_4^+ than sinks. Therefore, although it does appear that total metabolism may correlate with NH_4^+ uptake velocity, it is difficult to determine because significant NH_4^+ was so rare.

Using data from the upper and lower reaches, we had measures of NO_3^- uptake and metabolism on six dates. Nitrate uptake velocity was also not correlated with either GPP ($r^2 = 0.31$ $p=0.25$), R ($r^2=0.01$ $p=0.9$), or total metabolism ($r^2=0.05$ $p=0.7$). In the low N streams of Grand Tetons National Park, NO_3^- uptake was correlated with GPP suggesting autotrophs control NO_3^- uptake (Hall and Tank 2003). GPP in the Chattahoochee River is low compared to most other stream systems as a result of hydropeaking discharges (Chapter 3). Therefore, it is possible that with frequent disturbances to benthic and snag periphyton communites, the autotroph community can not contribute as much to nitrate uptake, especially when nitrate loading is elevated.

The ability of rivers and streams to retain and transform the excess nutrients present in wastewater treatment plant effluent is critical to the maintenance of water quality, and is worth an estimated 665 US dollars ha^{-1} yr^{-1} (Costanza et al. 1997). However, the significantly lower SRP and NH_4 uptake velocities present in the Chattahoochee River suggest that this ability can be altered. The Chattahoochee River is a regulated urban river and as a result has a severely modified hydrology, geomorphology, and chemical loading that is typical of both regulated and particularly urban streams and rivers (Poff et al. 1997, Paul and Meyer 2001). These modifications include large daily fluctuations in discharge, an incised channel that isolate the channel from the floodplain except at the highest discharges, and increased nutrient and other chemical loadings. As a result, nutrient uptake is sporadic at best. The results of this study demonstrate that, on occasion, some nutrients are being retained by the river. However, it was rare that both reaches showed significant retention for a particular solute, and all five solutes never demonstrated measurable uptake within a reach on the same date. In addition, when uptake did occur most of the uptake lengths were so long that a significant portion of the nutrients from the wastewater treatment plants were transported to West Point Lake.

Because all of the modifications interact, and because we could not determine why there was measurable uptake on some dates and not on others, it is not possible to say which of the modifications had the greatest effect on the ability of the river to retain nutrients. However, the decline in SRP and NH_4^+ uptake velocities compared to less modified systems and the findings of previous research showing that uptake velocities decline with increasing concentrations suggest that decreasing nutrient loading to the Chattahoochee River would help in the maintenance of water quality through both direct and indirect effects. Specifically, decreases in nutrient loading would improve the nutrient uptake capacity of the river.

These results demonstrate that despite severe modifications by humans, this urban river is still capable of retaining and transforming nutrients although in a much diminished capacity. In addition, the substantial reservoir of biologically available phosphorus in the sediments suggests that the response to a management strategy that decreases the loading of phosphorus to the river may have a significant lag time. Water quality in the Chattahoochee River has improved since the 1970's as a result of improvements to wastewater treatment plants and legislative action to control point-source loads of phosphorus (Frick et al. 1996). However, to further improve water quality and maintain downstream ecosystems, additional actions to enhance nutrient uptake in the river are needed. This study suggests that enhancement of nutrient uptake could be achieved by two actions. First, and most important, reducing nutrient loading would enhance nutrient uptake capacity and result in greater reductions in ambient nutrient concentrations than expected from the loading reductions alone. Increases in autotrophic communities would also potentially enhance nitrate uptake and could be accomplished by decreasing hydropeaking.

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Figure 4.1: Map of study reach. Bridge crossings are indicated by black triangles, sampling points for the upper reach are shown as black circles, sampling points for the lower reach are shown as gray squares. The Highway 166 bridge crossing is the most upstream triangle.

Figure 4.2: Panel A: Relationship between discharge and SRP uptake length ($y = 180.8 - 3.6x$). Notice the negative correlation is contrary to our hypothesis. Panel B: Relationship between total suspended solids and SRP uptake length ($y = 97.7 - 2.9x$). There is a significant positive correlation between discharge, total suspended solids, and temperature. Therefore, the negative relationship in Panel A is due to the relationship between SRP uptake length and TSS in Panel B and the correlation between discharge, TSS, and temperature. We could not differentiate the effects of temperature and TSS on uptake length.

Figure 4.3. Panel (A): The non-significant relationship between ammonium uptake length and discharge on dates with measurable ammonium uptake. Panel (B): The non-significant relationship between nitrate uptake length and discharge on dates with measurable nitrate uptake. Data for both reaches were combined for both panels.

Figure 4.4: Relationship between suspended fine particulate organic matter (FPOM) and nitrate uptake velocity (V_f) on dates with measurable uptake in nitrate. Data for both the upper and lower reaches were combined and the equation for the line is $y = -0.4 + 0.29x$.

Figure 4.5: Soluble reactive phosphorus uptake velocities from a variety of streams. Clear symbols depict less-modified streams, filled symbols depict streams receiving wwtp effluent,

with the Catalonia streams in gray and the Chattahoochee samples in black. Legend symbols are Hubbard Brook Experimental Forest (HBEF) (Hall et al. 2002), Coweeta and Oregon (Munn and Meyer 1990), Reira Major (Marti and Sabater 1996), Appalachian (Valett et al. 2002), Catalonia (Marti et al. 2004), Chattahoochee (this study). Chattahoochee uptake velocities were significantly lower than those from the less-modified streams (mean less-modified = 4.6 mean Chattahoochee = 0.8, Welch's t = 6.62, df = 59, p <<0.001).

Figure 4.6: Ammonium uptake velocities from a variety of streams. Clear symbols depict less-modified streams, filled symbols depict streams receiving wwtp effluent, with the Catalonia streams in gray and the Chattahoochee samples in black. Legend symbols are as follows: Hubbard Brook Experimental Forest (HBEF) (Hall et al. 2002), Grand Tetons National Park (GTNP) (Hall and Tank 2003) Reira Major (Marti and Sabater 1996), Lotic Intersite Nitrogen Experiement (LINX) short-term nutrient additions (Mulholland et al. 2003), Catalonia (Marti et al. 2004), Chattahoochee (this study). Chattahoochee uptake velocities were significantly lower than those from the less-modified streams (mean less-modified = 3.9 mean Chattahoochee = 1.0, Welch's t = 4.64, df = 31, p <0.001).

Figure 4.7: Nitrate uptake velocities from a variety of streams. Clear symbols depict less-modified streams; filled symbols depict streams receiving wwtp effluent, with the Catalonia streams in gray and the Chattahoochee samples in black. Legend symbols are as follows: Hubbard Brook Experimental Forest (HBEF) (Bernhardt et al. 2002), Grand Tetons National Park (GTNP) (Hall and Tank 2003), Konza (Dodds et al. 2002 smallest enrichment measurements), Idaho (Davis and Minshall 1999), Coweeta and Oregon (Munn and Meyer

1990), Catalonia (Marti et al. 2004), Chattahoochee (this study). Chattahoochee uptake velocities did not differ significantly from the less-modified streams (mean less-modified = 1.2 mean Chattahoochee = 0.6, Welch's t = 1.45, df = 48, p=0.14).

Table 4.1: Environmental variables on each sample date in 2001. Discharge and temperature were measured at the Highway 166 bridge crossing. TN:TP, TSS, and suspended FPOM are averages based on measurements taken at each sampling point in the reach. Water column chlorophyll a and bacteria concentrations are averages based on three samples taken at the bridge crossings. TN:TP is the molar ratio of the average total nitrogen concentration to the average total phosphorus concentration in the entire reach, TSS is total suspended solids, and suspended FPOM is suspended fine particulate organic matter.

Date	Q (m^3 s-^{1})	Temp. (˚C)	TN:TP	TSS (mg/L)	suspended FPOM (mg/L)	water column chl. a (µg/ L)	bacteria (#/mL)	rainfall 1 week prior (mm)
7/6/2001	36.9	28.6	35.2	21.1	13.0	4.1	$2.91 \cdot 10^6$	45.7
7/13/2001	34.7	29.3	57.6	21.2	6.4	4.3	$3.34 \cdot 10^6$	4.6
7/20/2001	41.2	26.6	71.9	17.3	6.1	2.7	$3.03 \cdot 10^6$	2.3
8/14/2001	86.5	24.4	59.7	72.4	14.7	4.0	$6.01 \cdot 10^6$	50.3
8/17/2001	44.3	26.1	63.6	30.7	7.0	6.0	$6.72 \cdot 10^6$	33.0
8/24/2001	40.8	26.7	79.0	16.6	4.0	8.5	$2.35 \cdot 10^6$	1.8
8/31/2001	38.1	24.3	63.7	20.0	4.3	11.0	$3.64 \cdot 10^6$	12.4
9/14/2001	36.7	24.8	80.4	14.0	4.9	8.8	$1.56 \cdot 10^6$	2.5
10/5/2001	30.6	18.7	117.9	8.7	8.0	9.4	$4.72 \cdot 10^6$	16.8
10/19/2001	31.9	18.0	90.8	9.5	6.6	12.2	$2.5 \cdot 10^6$	11.2

Table 4.2: Dilution based on two conservative tracers, fluoride and chloride for the 24.1 km study reach in 2001. Dilution on the first 2 dates was much higher than on subsequent dates. On most days, fluoride based measurements were similar to those based on chloride. Chloride inputs associated with tributaries result in lower dilution estimates for chloride than for fluoride. Due to these potential inputs, we used fluoride in calculations of nutrient uptake. Dashes represent dates lacking data.

Date	3.7 to 12.5 km		14.7 to 24.1 km	
	fluoride	chloride	fluoride	chloride
7/6/2001	14%	9%	< 1%	3%
7/13/2001	16%	14%	23%	14%
7/20/2001	2%	2%	2%	2%
8/14/2001	1%	1%	18%	2%
8/17/2001	6%	--	7%	--
8/24/2001	1%	1%	1%	5%
8/31/2001	2%	4%	7%	8%
9/14/2001	5%	3%	5%	8%
10/5/2001	6%	8%	3%	< 1%
10/19/2001	7%	2%	9%	3%

Table 4.3: Nutrient uptake lengths and increases in concentration for the upper and lower reaches in 2001. For a given date and a given element, the left column is the upper 8.8 km of the reach and the right column is the lower 11.6 km of the reach. Note that on July 6th dilution was too high in the upper reach to detect measurable uptake and increases. Arrows pointing down (↓) denote declines in nutrient concentrations, and numbers after the arrow are the nutrient uptake length in km. Arrows pointing up (↑) denote increases in nutrient concentration, and the number after the arrow is the increase in concentration per km (µg L^{-1} km^{-1}). ns denotes a non-significant increase or decline.

Date	**SRP uptake**		**Organic P uptake**		**NH_4 uptake**		**NO_3 uptake**		**DON uptake**	
7/6/2001	-	↓ 33	-	↑ ns	-	-	-	↓ 13	-	-
7/20/2001	↓ 50	↑ ns	-	-	↑ ns	↑ ns	↓ 60	↑ ns	↑ 285.3	↓ 5
8/14/2001	↓ ns	↑ 0.4	↓ ns	↑ ns	↓ ns	↑ 13	↓ ns	↑ ns	↑ ns	↓ 4
8/17/2001	↓ 19	↑ 1.9	↑ 3.9	↓ ns	↑ 43.2	↓ 12	↓ ns	↑ ns	↓ ns	↑ ns
8/24/2001	↓ 56	↓ 45	↑ 1.6	↓ ns	↓ 7	↑ 5.0	↓ 42	↓ ns	↓ 17	↓ ns
8/31/2001	↓ 11	↑ 1.1	↓ ns	↑ ns	↑ ns	↑ 3.2	↓ 23	↑ ns	↓ 5	↑ ns
9/14/2001	↓ 45	↓ 48	↑ 1.1	↑ 1.4	↑ 7.3	↓ 12	↓ ns	↓ ns	↑ ns	↓ ns
10/5/2001	↓ 75	↑ 0.5	↑ 1.9	↓ ns	↑ 13.4	↓ 20	↑ ns	↓ 22	↓ ns	↓ ns
10/19/2001	↓ 85	↑ 3.9	↓ 16	↓ ns	↓ ns	↑ 7.4	↓ 37	↑ ns	↓ ns	↓ 10

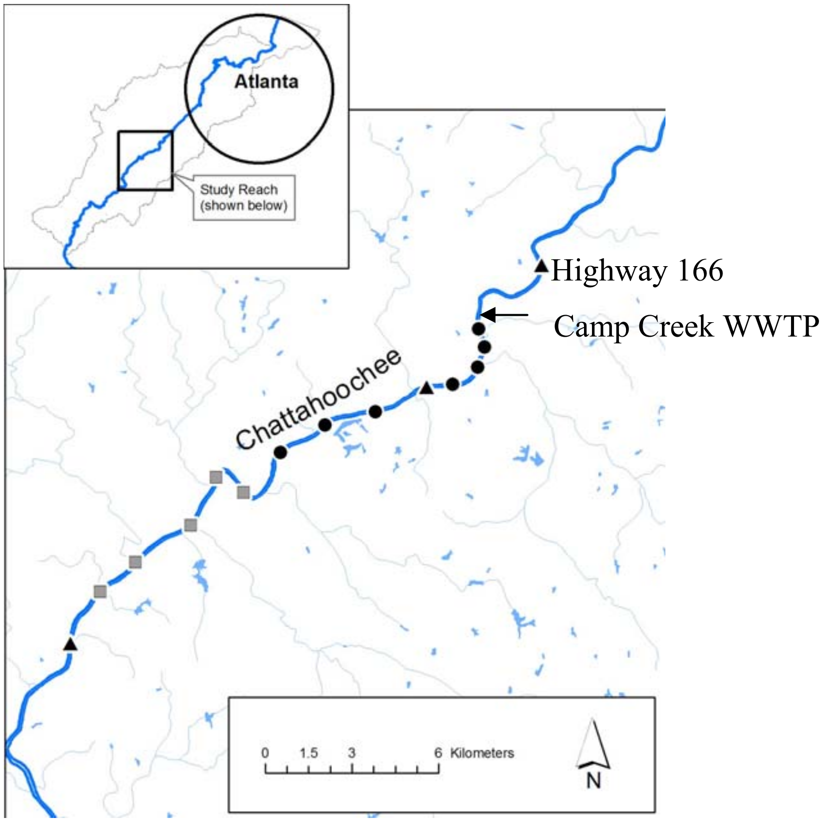
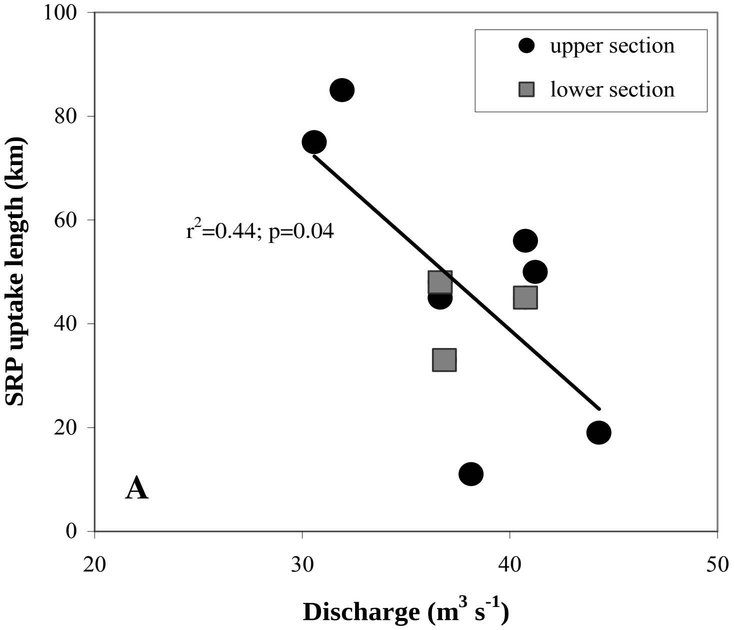

Atlanta
Study Reach
(shown below)
Highway 166
Camp Creek WWTP
Chattahoochee
0
1.5
3
6
Kilometers
N

Figure 4.1


100
80
60
40
20
0
20
30
40
50
SRP uptake length (km)
Discharge (m^3 s^{-1})
● upper section
■ lower section
r^2=0.44; p=0.04
A

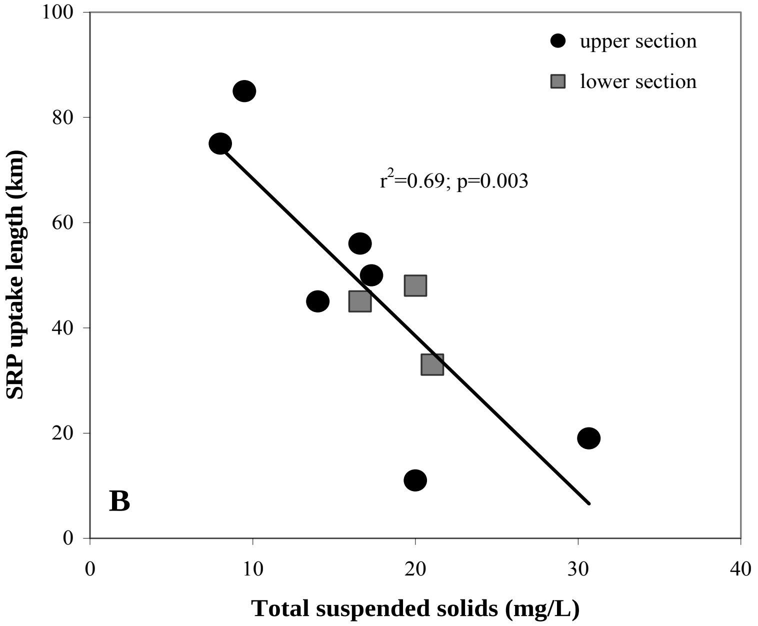

100
80
60
40
20
0
0
10
20
30
40
SRP uptake length (km)
Total suspended solids (mg/L)
● upper section
■ lower section
r^2=0.69; p=0.003
B

Figure 4.2

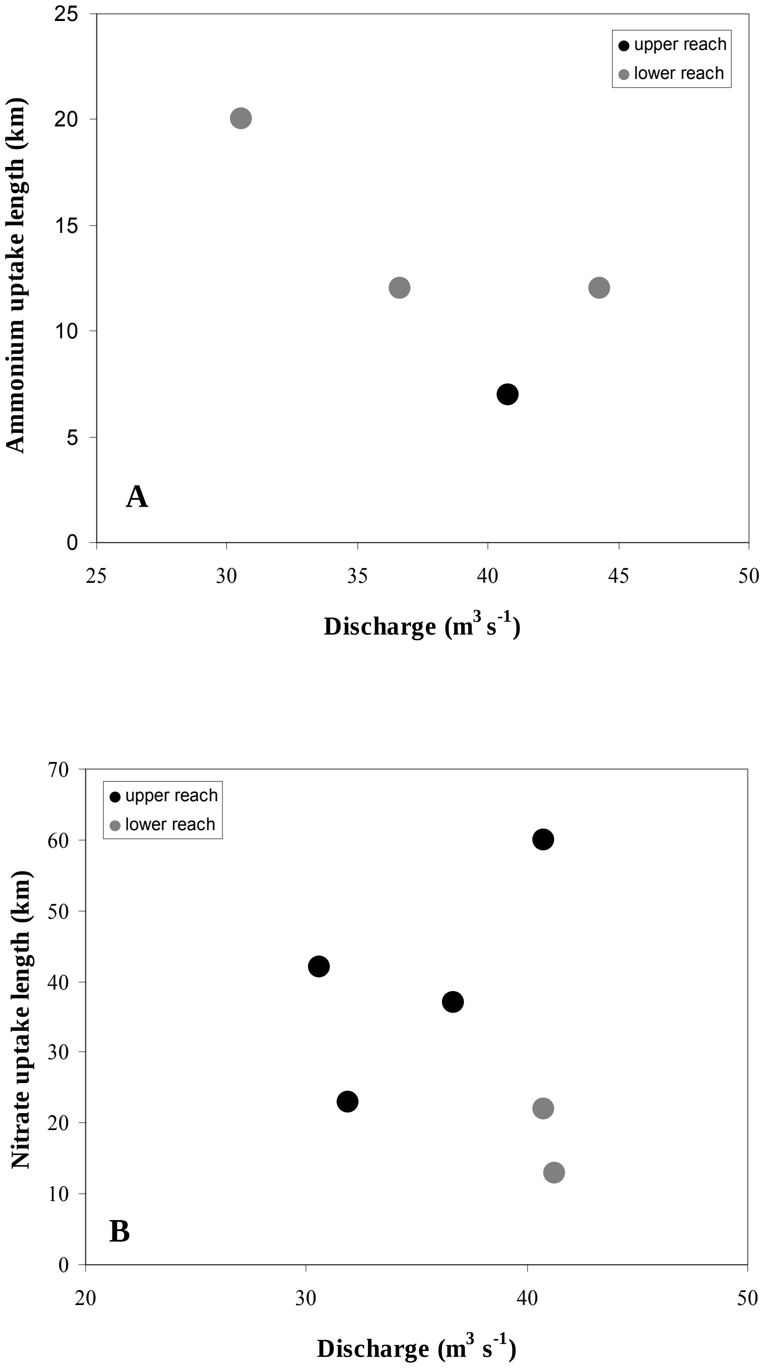

25
20
15
10
5
0
Ammonium uptake length (km)
● upper reach
● lower reach
A
25
30
35
40
45
50
Discharge (m^3 s^{-1})
70
60
50
40
30
20
10
0
Nitrate uptake length (km)
● upper reach
● lower reach
B
20
30
40
50
Discharge (m^3 s^{-1})

Figure 4.3

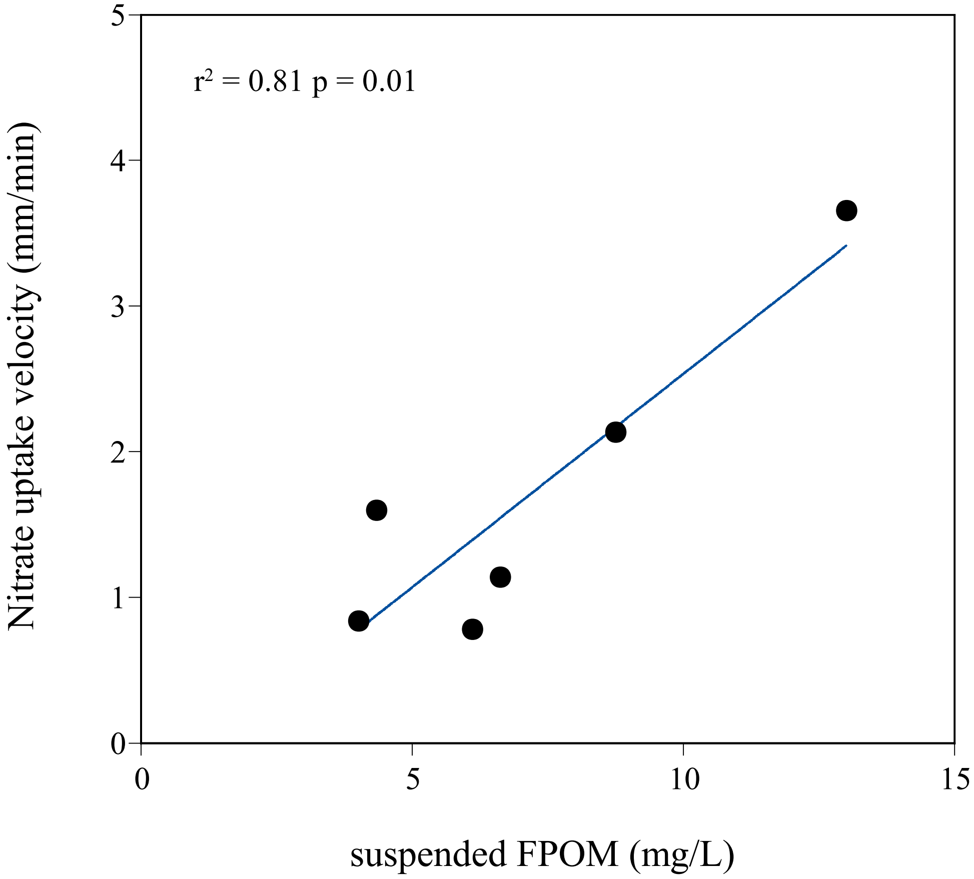
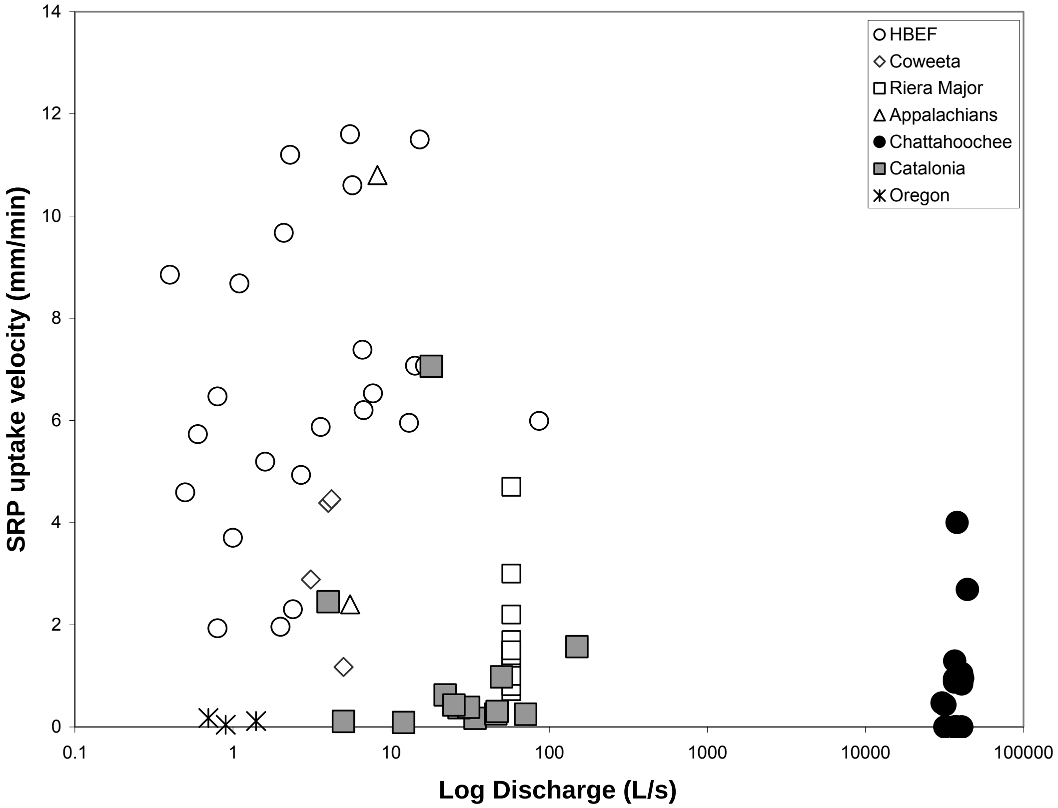

r² = 0.81 p = 0.01
Nitrate uptake velocity (mm/min)
suspended FPOM (mg/L)
0
1
2
3
4
5
0
5
10
15

Figure 4.4



SRP uptake velocity (mm/min)
Log Discharge (L/s)
0
2
4
6
8
10
12
14
0.1
1
10
100
1000
10000
100000
○ HBEF
◇ Coweeta
□ Riera Major
△ Appalachians
● Chattahoochee
▨ Catalonia
✳ Oregon

Figure 4.5

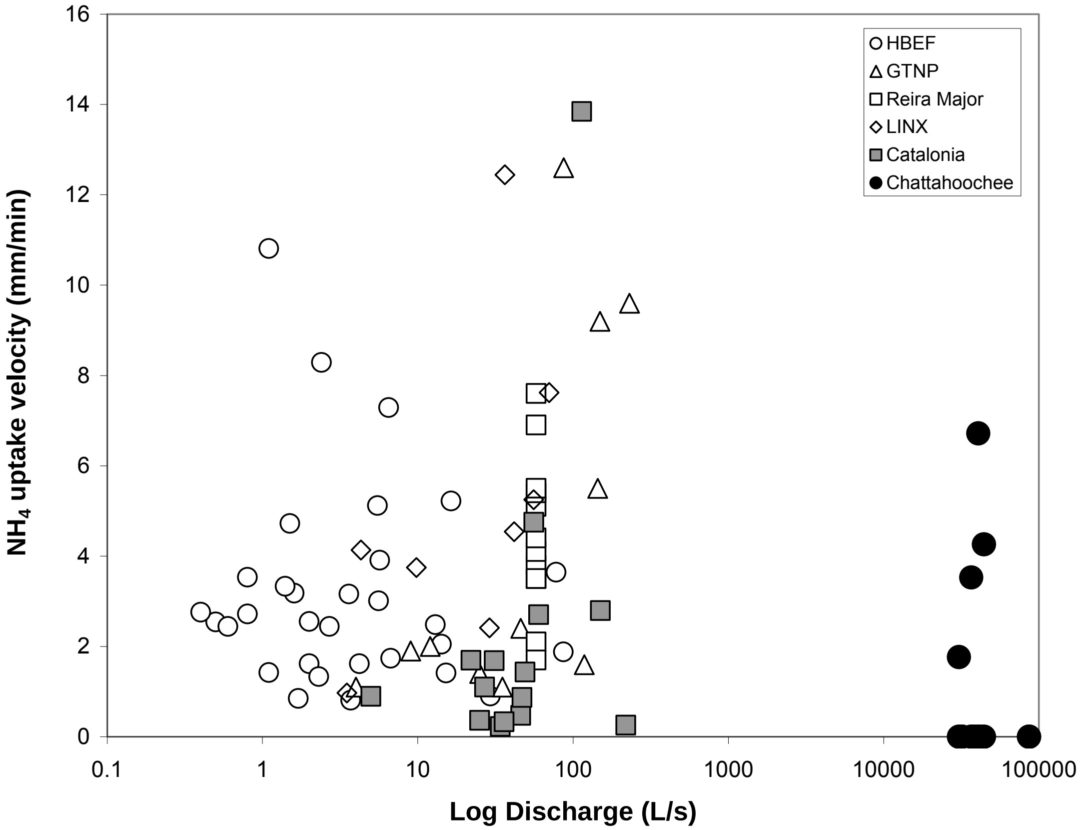
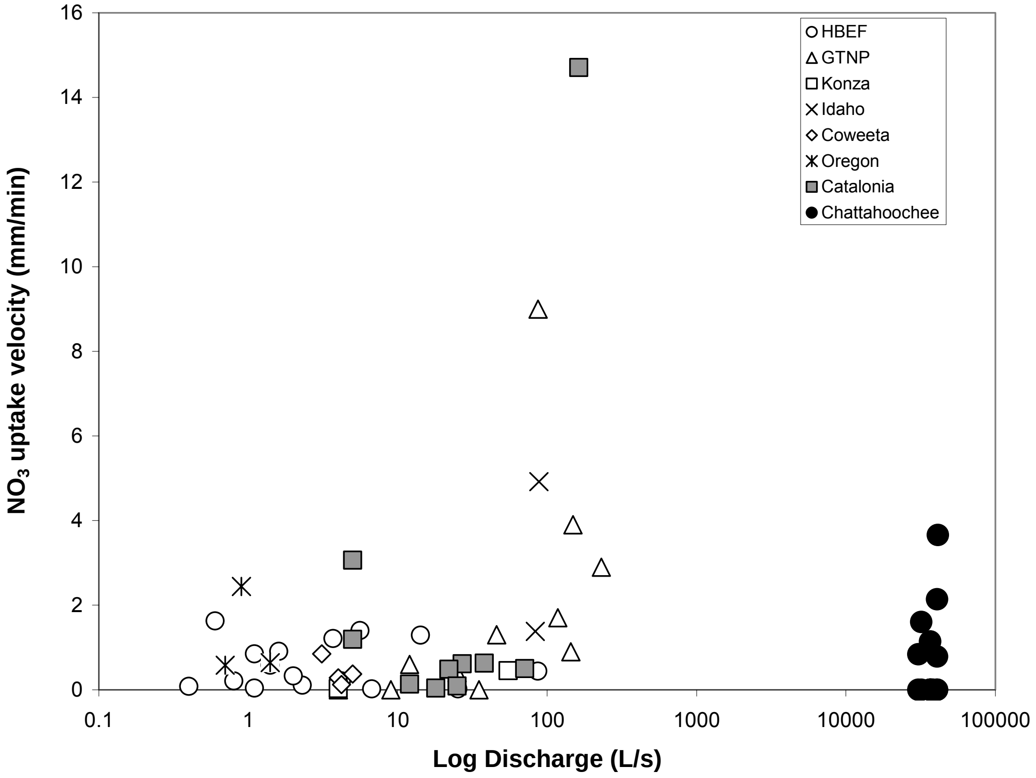

16
14
12
10
8
6
4
2
0
0.1
1
10
100
1000
10000
100000
NH_4 uptake velocity (mm/min)
Log Discharge (L/s)
○ HBEF
△ GTNP
□ Reira Major
◇ LINX
■ Catalonia
● Chattahoochee

Figure 4.6



16
14
12
10
8
6
4
2
0
NO3 uptake velocity (mm/min)
0.1
1
10
100
1000
10000
100000
Log Discharge (L/s)
○ HBEF
△ GTNP
□ Konza
✕ Idaho
◇ Coweeta
✳ Oregon
▨ Catalonia
● Chattahoochee

Figure 4.7

CHAPTER 5

NUTRIENT UPTAKE IN URBAN AND FOREST HEADWATER STREAMS[4]

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[4] Gibson, C.A. and J.L. Meyer. To be submitted Limnology and Oceanography

Abstract

We examined the impact of urbanization on nutrient uptake processes by measuring uptake lengths (S) and uptake velocity (V_f) of ammonium (NH_4) and soluble reactive phosphorus (SRP) in four forest and four urban headwater streams in the Upper Etowah River Basin, GA, USA. We performed the measures during dry and wet summers using short-term nutrient additions. One of the urban streams was located in a conservation subdivision, which made it possible to examine nutrient uptake in streams associated with this different type of development. Nutrient concentrations were higher in urban streams than in forest streams. Nutrient uptake lengths were similar between the two stream types, but nutrient uptake velocities were significantly higher in forest streams than in urban streams. In the conservation subdivision stream, nutrient uptake velocity was typically intermediate between the forest and urban streams. Ammonium concentrations were higher in the wet summer than in the dry summer, but total P concentrations were similar. Similarly, SRP V_f was similar in the two summers, but NH_4^+ V_f was lower in the wet summer than in the dry summer. SRP V_f was positively correlated with iron and aluminum content of the sediments and ln N:P ratio in the stream. These data indicate that SRP uptake was predominantly controlled by chemical processes, although biological processes played a role as well. NH_4^+ V_f was positively correlated with total metabolism (GPP + CR) indicating that biological processes controlled NH_4^+ uptake. The urban streams in this study have maintained the ability to retain nutrients, but urbanization has diminished this ability. In addition, it appears that conserving functioning riparian areas and natural channel structure can help buffer some of the effects of urbanization and maintain nutrient uptake.

Introduction

Headwater streams are tightly linked with both the terrestrial landscape and downstream ecosystems (Meyer and Wallace 2001, Gomi et al. 2002). Headwater streams retain large amounts of organic matter and nutrients as they slow the delivery of water to downstream ecosystems (Meyer and Wallace 2001). However, with urbanization, many headwater streams are lost or significantly altered, resulting in decreased drainage densities in urban areas (Meyer and Wallace 2001). In addition, streams in urban settings are characterized by flashy hydrology, high nutrient and contaminant loads, altered channel morphology, and lower species richness (Paul and Meyer 2001). It is likely that these physical, chemical, and biological changes affect ecosystem processes, but there is less information about how urbanization affects ecosystem processes in streams.

Streams alter the form and amount of nutrients transported from terrestrial ecosystems to downstream ecosystems through nutrient uptake and transformation (Stream Solute Workshop 1990, Alexander et al. 2000). This retention and transformation can be important in regulating downstream fluxes of nutrients. Numerous studies have shown that land-use can affect nutrient loading to streams (Caraco 1995, Carpenter et al. 1998, Caraco and Cole 1999). Human population within a watershed is related to mean annual nitrate transport (Peierls et al. 1991), and human sewage inputs and fertilizer runoff accounts for much of the variation in SRP transport (Caraco 1995). In a multi-biome study, headwater streams retained and transformed an average of 64% of inorganic nitrogen inputs from their watersheds (Peterson et al. 2001). Due to their large-surface to volume ratios, small streams play an essential role in determining downstream water chemistry (Alexander et al. 2000, Peterson et al. 2001).

Physical factors such as discharge, stream depth, and current velocity have been shown to be important factors in determining nutrient uptake (Haggard et al. 2001, Peterson et al. 2001, Webster et al. 2003). Therefore, changes in land-use that affect the physical characteristics of streams could influence uptake lengths. One such physical factor is transient storage zones.

Transient storage zones are areas where water moves more slowly than in the channel and include both hyporheic and surface areas such as backwaters and pools (Harvey et al. 1996, Lancaster and Hildrew 1993, Harvey and Wagner 2000). Transient storage areas slow the movement of water, which should facilitate nutrient uptake because contact between the water column and the benthos is increased. However, transient storage zone size only explained 35% of the variation in ammonium (NH_4^+) uptake velocities in one season and was not related to soluble reactive phosphorus (SRP) uptake in 13 streams at Hubbard Brook Experimental Forest (Hall et al. 2002). In addition, in a multi-biome study of small headwater streams, most NH_4^+ uptake occurred before the water even entered transient storage zone areas (Webster et al. 2003).

Ambient nutrient concentration has also been shown to influence nutrient uptake lengths and uptake rates (Wollheim et al 2001, Dodds et al. 2002). It is possible that with increased nutrient inputs, nutrient uptake can become saturated, which would lead to longer uptake lengths and lower uptake velocities (Dodds et al. 2002, Wollheim et al. 2001). For example, NH_4^+ and SRP uptake lengths were longer in streams receiving point-source discharge when compared to less-modified, but similarly sized streams (Marti et al. 2004). In addition, NH_4^+ and SRP uptake velocities in a medium-sized river receiving large amounts of point-source inputs were significantly lower than less-modified streams (Chapter 4). Therefore, physical and chemical changes as a result of urbanization are likely to affect nutrient uptake in streams. For example, a comparison of forest and urban 3rd and 4th order streams in Atlanta, Georgia showed that NH_4^+ and SRP uptake velocities were significantly lower in the urban streams and uptake velocity was correlated with measures of extent of urbanization (Meyer et al. submitted).

Material transport in headwater systems is dominated by processes from hillslopes to stream channels (Gomi et al. 2002). This leads to tight linkages between the terrestrial ecosystem and the stream. Material transport in larger downstream systems is affected by processes occurring in the headwater streams, the terrestrial catchment, and often the floodplain (Gomi et al. 2002). This indicates that nutrient transport to headwater streams in urban areas is likely to be dominated by non-point source inputs. Whereas, nutrient transport in larger streams is likely to be influenced by upstream channels, non-point sources from the surrounding watershed, and potentially point-source inputs. However, in both cases impervious surface areas may speed the delivery of nutrients, as these surfaces lack the capacity for retention of water, nutrients, or organic matter (Meyer and Wallace 2001). Given the influence of headwater streams on water chemistry in downstream systems (Peterson et al. 2001), it is important to determine (1) how urbanization affects nutrient uptake in headwater streams and (2) if it is similar to the effects seen in larger stream systems (Haggard et al. 2001, Marti et al. 2004, Meyer et al. submitted, Chapter 4). These are the objectives of this study.

We examined effects of urbanization on nutrient uptake by measuring NH_4^+ and SRP uptake in four forest and four headwater streams in the Upper Etowah River Basin, GA during summer (June-August) in both a dry and wet summer. We measured a variety of physical, chemical, and biological parameters to determine what controlled NH_4^+ and SRP uptake in these streams. We hypothesized that urban streams would have longer uptake lengths and lower uptake velocity due to higher ambient nutrient concentrations and less physical retention of water.

Methods

Study Sites

All streams were located in the piedmont portion of the Upper Etowah River Basin, Georgia. From the late 1800’s to the early 1900’s most of Georgia’s piedmont was used for cotton agriculture; therefore, it is likely that all 8 streams drained agricultural areas during this time period. All streams were unnamed first order streams on 1:24,000 USGS topographic maps and contained a mix of pool and riffle habitats. Surface sediments in the channels of all eight streams were a mix of bedrock, cobble and finer sands and silts (Table 5.1). The four forest streams were all located within the Dawson Forest Wildlife Management Area in Dawson, Co. Dawson Forest is a mix of pine and hardwood forests that are actively managed for timber production and recreation. Watershed area for the forest streams ranged from 0.3 to 1.8 km^2.

The four urban streams were located within the city limits of either Kennesaw or Acworth, GA and drained a mixture of residential and industrial land uses. Both towns are suburbs of Atlanta, GA. All four streams had incised banks that are typical of many urban streams (Paul and Meyer 2001), although the degree of incision varied. One of the urban streams was located in a conservation subdivision. Conservation subdivisions cluster houses on smaller lots in order to preserve a large forested or "green" space. In addition, this conservation subdivision maintained relatively wide riparian buffer zones (at least 30 m on each side of stream). Therefore, even though this stream drained a residential neighborhood with impervious surfaces, it was potentially buffered from some of these effects by the wide riparian zone. The other 3 urban streams had riparian zones that were typically one tree wide. Watershed area for the urban streams ranged from 0.4 to 1.8 km^2.

Nutrient Uptake

Nutrient uptake length was measured using the methods described by Webster and Ehrman (1996) and Stream Solute Workshop (1990). We performed short-term nutrient additions using sodium chloride as a conservative tracer during the summers of 2002 and 2003. The summer of 2002 was at the end of a 3 year drought in Georgia whereas precipitation in 2003 was significantly higher. Therefore, we designated 2002 as the "dry" summer and 2003 as the "wet" summer. Nutrients and salt were dissolved in water and simultaneously injected at a known constant rate using a peristaltic pump (Watson-Marlow). Reach length varied between 130 and 150 m in the forest streams and between 100 and 120 m in the urban streams depending on travel time and accessibility. Reaches were shorter in the urban streams due to longer travel times (see below). Prior to injection, we measured background conductivity and nutrient concentrations (NH_4^+, SRP, nitrate, total dissolved N and total dissolved P) at 5 stations spread across the reach. Target concentrations varied based on background concentrations, but we typically increased conductivity 30 μScm^{-2}, NH_4^+ concentrations by 25 $\mu g\ L^{-1}$, and increased SRP by 15 $\mu g\ L^{-1}$. We injected SRP and NH_4^+ concentrations simultaneously, but maintained the stream's ambient N:P ratio.

We used a recording multi-probe to measure change in relative conductivity with time at a point in the thalweg throughout the injection (YSI, Yellow Springs, OH). Once salt concentrations had reached a plateau, we sampled NH_4^+, SRP, and conductivity at 10 m intervals. For the nutrients, we took three grab samples across each transect and immediately filtered the sample using glass fiber filters (Whatman GFF). NH_4^+ samples were analyzed with a fluorometer (Holmes et al. 1999), and SRP samples were analyzed colorimetrically (Wetzel and Likens 2000).

We calculated two measures of nutrient uptake. First, we calculated nutrient uptake length using the equation:

$$\ln C_x = \ln C_0 - k_c x \tag{1}$$

Where C_x is the nutrient concentration (x) meters from the top of the reach, C_0 is the nutrient concentration at the top of the reach, and k_c is the per meter uptake rate of the nutrient (Newbold et al. 1981). Uptake length (S) is equal to k_c^{-1}. Second, uptake velocity (V_f) is the velocity at which a nutrient moves downward through the water column toward the sediments (Stream Solute Workshop 1990, Hall et al. 2002, Dodds et al. 2002). It is calculated using the following equation:

$$V_f = \frac{Q}{w\,(S)} \tag{2}$$

where Q is discharge ($m^3 min^{-1}$), w is wetted width (m), and S is nutrient uptake length (m) (Stream Solute Workshop 1990). Uptake velocity represents the demand for nutrients relative to concentration within the water column and is independent of depth (Hall et al. 2002).

In addition to nutrients, we measured a number of other biological and physical parameters to determine what controlled nutrient uptake in these streams. We scraped periphyton from a known area at three spots on each of 10 transects in the reach using a brush. Samples were kept on ice and in the dark, until they were sub-sampled and filtered onto glass fiber filters. We ground one set of filters in 90% buffered acetone to extract the pigments and determined chlorophyll a content based on the methods of Arar and Collins (1997). We also determined the organic content of the periphyton by drying another set of filters at 60˚C for at least 1 week, weighing, and then combusting the filters at 500˚C for 3 hours. We measured canopy cover of the channel using a spherical densiometer at each of the algal sampling locations. We measured width and depth at three points across a transect every 10 m. We also counted the number of debris dams present in each reach as a measure of potential surface-water retention. Because the iron, aluminum and organic content of sediments affects chemical sorption of phosphorus (Meyer 1979), we determined iron and aluminum content of sediments in the thalweg and channel edge in 2003. We combined ten samples from each habitat in each stream into a thalweg and a channel edge sample for each stream. Iron and aluminum content was analyzed with an flame atomic absorption spectrophotometer after a double acid extraction (UGA Soil Analysis

Laboratory). Iron and aluminum content in the thalweg and channel-edge sediments were highly correlated, so we only include results from the thalweg.

We estimated transient storage for each solute injection by estimating the parameters of a one-dimensional advection, dispersion, and transient storage model (Bencala and Walters 1983). The parameters were estimated by solving the model iteratively and by finding a least-squares best fit of the model output compared to the observed solute concentration curve (Hart 1995). The parameters included water velocity (m min^{-1}), exchange rate of water between the main channel and the transient storage zone (k_1, min^{-1}), and exchange rate of water between transient storage and the main channel (k_2, min^{-1}). Transient storage zone size equals k_1/k_2. To determine if we had the appropriate reach length for measuring transient storage zone parameters, we followed the method of Wagner and Harvey (1997) and calculated the Damkohler number (DaI) using the following equation:

$$\text{DaI} = \frac{[(k_1 + k_2) \times L]}{v} \tag{3}$$

where L is the length of stream reach (m). If the reach is too short ($\text{DaI} < 0.5$) than the parameters cannot be properly estimated because not enough water has entered transient storage zone areas. If the reach is too long ($\text{DaI} > 5$) than the parameters cannot be estimated because dispersion dominates the solute curve (Wagner and Harvey 1997). This eliminated one experiment out of eight in both forest and urban streams.

In summer 2003, we measured whole-stream metabolism in an 80 m reach of each stream using the upstream-downstream or single station diurnal dissolved oxygen change technique (Marzolf et al. 1994, Young and Huryn 1998, Bott 1996). The single station method was used in two of the forest streams because one probe was vandalized, and the two-station upstream-downstream technique was used in the remaining streams. Metabolism measurements were made between one day and one week after nutrient uptake measurements. We measured dissolved oxygen and temperature at 5-minute intervals for 36-hours using multi-probe sondes (YSI, Yellow Springs OH). We calculated metabolism based on the difference in O_2 concentration between the upstream and downstream station (two-station) or between time intervals (single station) while accounting for O_2 exchange across the water surface for 24-hours starting and ending at midnight (Marzolf et al. 1994, Bott 1996). Daily community respiration and gross primary production were determined by integrating these measurements over the 24-hour period following the methods of Mulholland et al. (2001). The oxygen reaeration coefficient was estimated at the beginning of the 36-hour period by measuring the decline in propane during a steady-state injection of propane and sodium chloride. Propane was added to the streams by bubbling at a constant rate of 3 L s^{-1} and simultaneously adding sodium chloride. We collected three samples of 45 mL of stream water in 60 mL syringes at 10-m intervals. We closed the syringes with 3-way stop-cocks and kept them on ice for transport back to the lab. We injected 15 ml of N_2 gas into the syringes to create a head space, and equilibrated the head space by spinning the samples on a rotary shaker for 30 minutes. We analyzed for propane in the head space on a gas chromatograph. The reaeration of propane was converted to oxygen using a factor of 1.39 (Rathbun et al. 1978).

We used t-tests assuming unequal variances (Welch's t) to test for differences in discharge, water travel time, width, algal biomass, canopy cover, and NH_4^+ and SRP uptake length and uptake velocity between the urban and forest streams. We tested for autocorrelation among the environmental variables by constructing a correlation matrix. If two variables were autocorrelated ($p<0.05$), we only included one of the variables in the stepwise multiple regression model. We used forward, backwards, and mixed stepwise multiple regression to determine what environmental parameters correlated with nutrient uptake velocity in all the stream sites combined and in forest and urban streams separately. First we only included environmental parameters that we collected in both dry and wet summers: transient storage zone size, epilithon organic matter, epilithon chlorophyll a, and N:P ratio. Then we analyzed just the wet summer in order to include metabolism, FBOM, and iron and aluminum data. All three types of stepwise multiple regression models produced the same results; therefore, only one set of results is included. To evaluate differences in nutrient uptake between a dry summer and a wet summer, we used paired t-tests to test for differences in total N and total P concentration and NH_4^+ and SRP uptake velocity.

Results

Discharge at the time of sampling did not differ between forest and urban streams (Table 5.1); however, discharge was higher in the wet summer than in the dry summer (Table 5.3). Urban streams were 1.5 times wider than forest streams (Table 5.1). Water residence time was greater in urban streams than in forest streams; however, transient storage zone size was similar in the two stream types (Table 5.1). Total N concentration was three times higher and total P concentration was two times higher in the urban streams (Table 5.1). Chlorophyll a concentration was significantly higher in the urban streams, but organic matter content of the periphyton was similar between the two stream types (Table 5.1).

Both NH_4^+ and SRP uptake lengths were similar in the two stream types (Table 5.1). Ammonium uptake velocity ranged from 0.48 to 2.22 mm min^{-1} in the forest streams (Table 5.2). During the dry summer, the conservation subdivision stream had the highest NH_4^+ uptake velocity at 2.08 mm min^{-1}, but, in the wet summer, the value was intermediate between the forest and urban streams at 0.34 mm min^{-1} (Table 5.2). Ammonium uptake in the remaining urban streams ranged from 0.07 to 0.7 mm min^{-1} (Table 5.2). When all streams were included, NH_4^+ uptake velocities did not differ significantly between the two stream types (Table 5.1). However, when the conservation subdivision stream was excluded, NH_4^+ uptake velocities were significantly lower in the urban streams (Table 5.1).

SRP uptake velocity ranged from 0.73 to 4.4 mm min^{-1} in forest streams (Table 5.2). SRP uptake velocity in the conservation subdivision was intermediate between forest and urban streams at 1.12 and 0.76 mm min^{-1} in the dry and wet summers respectively (Table 5.2). In the remaining urban streams, SRP uptake velocity ranged from 0.22 to 1.48 mm min^{-1} (Table 5.2). As with NH_4^+, when all streams were included SRP uptake velocity did not differ between forest and urban streams (Table 5.1). However, when the conservation subdivision stream was excluded, SRP uptake velocities were significantly lower in the urban streams (Table 5.1).

Across all sites and summers, transient storage zone size, epilithon organic matter and ln N:P ratio were not correlated with NH_4^+ uptake velocity. Epilithon chlorophyll a was correlated with ln N:P and therefore was not included in the model. In the wet summer when we measured metabolism, NH_4^+ uptake velocity was positively correlated with total metabolism (GPP +R) ($r^2 = 0.55$ $p = 0.03$, Figure 5.1). None of the other environmental parameters met the criteria ($p<0.05$) to enter the model. Contrary to expectations, log SRP uptake velocity was negatively correlated with transient storage zone size across all sites and summers ($\log SRP\ V_f = -0.8 (k_1/k_2) + 0.59$; $r^2 = 0.69$ $p=0.001$). However, iron and aluminum content of sediments was negatively correlated with transient storage zone size ($r^2=0.5$ $p=0.04$) and positively correlated with log SRP uptake velocity in the wet summer ($r^2=0.81$ $p=0.002$, Figure 5.2). Although we do not have iron and aluminum data for the dry summer, SRP uptake velocity appears to be controlled by iron and aluminum content in the sediments and the negative correlation with transient storage zone size is likely the result of the negative correlation between transient storage zone size and sediment iron and aluminum content.

When the urban and forest streams were analyzed separately, SRP uptake velocity was positively correlated with ln N:P molar ratio in both the forest and urban streams (Figure 5.3). However, the slopes of the two lines were different in the two stream types (Figure 5.3). None of the environmental parameters correlated with NH_4^+ uptake velocity in the forest or urban streams when they were analyzed separately.

Total N concentration was almost two times lower in the dry summer than in the wet summer (Table 5.3). Total P concentration was similar in the two summers (Table 5.3). Ammonium uptake velocity was two times greater in the dry summer than in the wet summer (Table 5.3). However, SRP uptake velocity was similar in the wet and dry summers (Table 5.3).

Discussion

Short-term nutrient additions over-estimate uptake length and under-estimate uptake velocities because increased nutrient concentrations can saturate uptake sites (Mulholland et al. 2002). We attempted to minimize this effect by limiting the amount of nutrients we added and decreasing the reach length in the urban streams to shorten travel times. In addition, uptake velocity and nutrient uptake length were not significantly correlated with added nutrient concentration. Therefore, although these uptake velocities are probably lower than if we had used stable- or radio-isotopes, they provide a useful tool for comparison among streams in this study and other studies that have used the same technique.

We predicted that urban streams would have longer uptake lengths and lower uptake velocities as a result of decreased physical retention of water and increased nutrient concentrations. In fact, water residence time was greater in urban streams than in the forest streams. Background nutrient concentrations were higher in the urban streams than in the forest streams, but nutrient uptake lengths were similar. Variation in uptake length can be caused by hydrologic, geomorphic, and biotic characteristics. Longer water residence times typically lead to shorter uptake lengths because increased contact time between the water column and the sediments should facilitate nutrient uptake (Valett et al. 1996, Butterini and Sabater 1998, Hall and Tank 2003). In addition, high algal abundance increases the demand for nutrients and decreases uptake lengths (Marti et al. 1997). Urban streams had both longer water residence times and greater algal biomass (epilithon chlorophyll a); therefore, we would have expected shorter uptake lengths and higher uptake velocities in the urban streams. However, urban streams had similar NH_4^+ and SRP uptake lengths and lower NH_4^+ and SRP uptake velocities than the forest streams.

Measurements of uptake length are often influenced by depth and velocity (Hall et al. 2002). However, the high ambient nutrient concentrations can lead to longer uptake lengths (Wollheim et al. 2001). Therefore, it appears that in the urban streams the effects of long water residence times and high algal biomass were balanced by the high ambient concentrations and resulted in uptake lengths that were similar to the forest streams.

Uptake velocity represents nutrient demand relative to its concentration in the water column; therefore, the demand for both NH_4^+ and SRP was lower in the urban streams. Nutrient uptake rates have been shown to saturate with increasing nutrient concentration (Dodds et al. 2002). With saturation of uptake rate, uptake velocities are expected to decrease because benthic uptake moves a lower proportion of nutrient molecules out of the water column per unit time (Dodds et al. 2002). Previous research in larger urban streams also demonstrated lower NH_4^+ and SRP demand in urban streams compared to similarly sized forest systems (Meyer et al. submitted). In addition, previous studies have shown that streams receiving wastewater treatment plant inputs also have lower demand for nutrients compared to less-modified systems (Marti et al. 2004, Chapter 4). Therefore, when ambient nutrient concentrations are high, demand relative to concentration will be lower, potentially as a result of saturation. However, ambient concentrations were high in the conservation subdivision stream and nutrient uptake velocities were more similar to those seen in the forest streams. Therefore, it appears that riparian zone conservation and the associated organic matter inputs can help maintain nutrient retention capacity (see below).

Uptake velocities for both the forest and urban streams in this study were at the low end of the range of velocities found in forest streams with similar discharge at Hubbard Brook Experimental Forest (Hall et al. 2002). Ammonium uptake velocities in Hubbard Brook streams ranged from 0.81 to 10.81 mm min^{-1} and SRP uptake velocities ranged from 1.93 to 11.6 mm min^{-1} (Hall et al. 2002). The lowest SRP uptake velocity in Hubbard Brook was higher than in any of the streams in this study. In addition, 25 of the 33 NH_4^+ uptake velocities in Hubbard Brook streams were higher than the highest NH_4^+ uptake velocity in any of the streams in this study. We expected uptake velocities in the forest streams to be similar to those found in other systems. The low uptake velocities in the forest streams may be the result of past land-use. As stated earlier, all of the streams in this study were likely previously affected by cotton agriculture. Dawson Forest is actively logged, and the substrate of some of the forest streams contained large proportions of fine sediments that may indicate sedimentation after logging (Table 5.1). In addition, given the high algal biomass and long water residence time, we would expect urban uptake velocities to be similar to those found in other systems as well. However, it appears that other aspects of urbanization, such as nutrient loading, overwhelmed the positive effects of water residence time and high algal biomass. This resulted in decreased nutrient uptake velocities in the urban streams.

SRP uptake velocity was correlated with iron and aluminum content of sediments across all sties. Control of phosphorus uptake by chemical sorption has been demonstrated in forest streams, and smaller sediment particles had higher phosphorus demand (Meyer 1979). In addition to chemical sorption, SRP uptake velocity was also related to ln N:P ratio in both the urban and the forest streams. Therefore, as N concentration in the water column increased, the demand for P increased as well. This suggests that, in addition to chemical sorption, biological processes played a role in regulating SRP uptake in both stream types. In slightly larger urban streams, Meyer et al. (submitted) found that SRP uptake velocity was positively correlated with standing crop of benthic organic matter, suggesting that nutrient demand by microbes associated with the organic matter influenced SRP uptake.

In the wet summer, NH_4^+ uptake velocity was positively correlated with total metabolism (GPP + CR) across all sites, suggesting demand for NH_4^+ by both autotrophs and heterotrophs controls NH_4^+ uptake in these streams. The relationship between metabolism and NH_4^+ uptake velocity links NH_4^+ demand with carbon fixation and organic matter decomposition. Previous research in forest and urban systems has shown correlations between nitrogen uptake and metabolism (Hall and Tank 2003, Meyer et al. submitted). In addition, the relationship between NH_4^+ uptake velocity and total metabolism is similar across a wide range of streams (Meyer et al. submitted).

The stream located in a conservation subdivision had the highest NH_4^+ uptake velocity of any of the streams in this study in the dry summer and the highest of the urban streams in the wet summer (Table 5.2). SRP uptake velocity in the conservation subdivision was the highest of the urban streams in the dry summer and the second highest in the wet summer (Table 5.2). Ambient total N and P concentrations were still high in the conservation subdivision stream (Table 5.2), but uptake velocities were more similar to those found in the forest streams compared to the other urban sites. Riparian zones in the conservation subdivision were much wider than those in the other urban systems and potentially helped ameliorate some of the effects of the surrounding urbanization. Riparian areas help to reduce the input of nutrients to the stream from the catchment (Lowrance et al. 1984), but ambient nutrient concentrations in the conservation subdivision were similar to the other urban streams. In addition, riparian areas provide organic matter inputs to the stream. Leaf and wood removal from a headwater stream in North Carolina led to increased nutrient uptake lengths compared to a stream containing natural amounts of allochthonous organic matter (Webster et al. 2001). The high respiraiton rates in the conservation subdivision stream compared to other urban streams are likely also a result of the organic matter inputs from the larger riparian zone. Therefore, by conserving riparian areas, the design of conservation subdivision appears to have helped maintain higher levels of nutrient demand in this stream compared to the other urban streams.

Unlike many urban streams, the urban streams in this study maintained complex channels that retained water and had substrates that were suitable for algal attachment. For example, water residence time in the urban streams was longer than in the forest streams and transient storage zone area was similar in the two stream types. In addition, epilithon chlorophyll *a* concentrations were higher in the urban streams than the forest stream. Headwater streams are frequently lost in urbanized areas, thereby reducing drainage density (Paul and Meyer 2001, Meyer and Wallace 2001). These urban streams are capable of retaining nutrients because they have maintained channel structure and in-channel substrates that have not been silted over and therefore still function as algal attachment sites. Our results demonstrate that urban headwater streams are functioning ecosystems that retain nutrients and suggest that urban design practices such as conservation subdivisions can increase nutrient retention. Therefore, the loss of headwater streams due to urbanization would likely cause increases in nutrient concentrations in downstream ecosystems and intensify the effects of urbanization on downstream ecosystems.

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Figure Captions

Figure 5.1: Results of stepwise multiple regression for NH_4^+ uptake velocity in forest (◆) and urban (○) streams. The variation in NH_4^+ uptake velocity was best explained by total metabolism, which accounted for 55% of the variation ($NH_4\ V_f = 0.21(GPP + CR) + 0.067$).

Figure 5.2: Relationship between iron and aluminum content of sediments and SRP uptake velocity in forest (◆) and urban (○) streams (log SRP V_f = 0.025 (Fe + Al) -1.23).

Figure 5.3: Results of mixed stepwise multiple regression for SRP uptake velocity in both urban (○) and forest (◆) streams. In both the urban and the forested stream, N:P ratio was the only environmental parameter that met the criteria ($p < 0.05$) to enter the model. N:P ratio was based on total N and total P concentrations. The ln N:P ratio explained 73% of the variation in forest SRP V_f (log SRP V_f = 0.33 (ln N:P) – 0.55). In urban streams, ln N:P ratio explained 67% of the variation in log SRP uptake velocity (log SRP V_f = 0.33(ln N:P) – 1.08). The stream in the conservation subdivision (△) was included in the urban regression, however we used a different symbol for this stream to highlight the fact that it fell in between the urban and forested streams.

Table 5.1: Summary of environmental parameters and nutrient uptake length and velocity in forested and urban streams. All t-tests were Welch's t-tests assuming unequal variances.

	Forest	Urban	t-stat	p-value
Watershed area (km^2)	0.78	1.11	0.65	0.49
Discharge (L/s)	4.9 (4.2)	2.7 (1.6)	1.34	0.2
Width (m)	1.6 (0.51)	2.53 (0.45)	3.83	0.002
Water residence time (min)	30 (10)	59 (29)	2.69	0.024
Total N (μg/L)	201 (145)	673 (304)	3.96	0.002
Total P (μg/L)	31 (10)	74 (33)	3.46	0.008
transient storage zone	0.72 (0.49)	0.98 (0.53)	0.9	0.4
epilithon chlorophyll *a* (mg/m^2)	7.0 (2.9)	18.2 (8.4)	3.57	0.006
epilithon organic matter	0.81 (0.7)	0.62 (0.3)	0.63	0.54
canopy cover (% cover)	70 (7)	47 (16)	2.65	0.57
FBOM (mg/m2)	477 (140)	603 (162)	1.18	0.28
NH_4 S (m)	193 (157)	256 (275)	0.56	0.6
SRP S (m)	109 (47)	114 (59)	0.19	0.8
NH_4 V_f (mm/min)	1.13 (0.56)	0.59 (0.64)	1.8	0.9
SRP V_f (mm/min)	1.72 (1.24)	0.69 (0.46)	2.19	0.06
NH_4 V_f without conserv. subdivision (mm/min)		0.38 (0.25)	3.34	0.007
SRP Vf without conserv. subdivion (mm/min)		0.59 (51)	2.29	0.045
No. of debris dams	6.9	1.6		

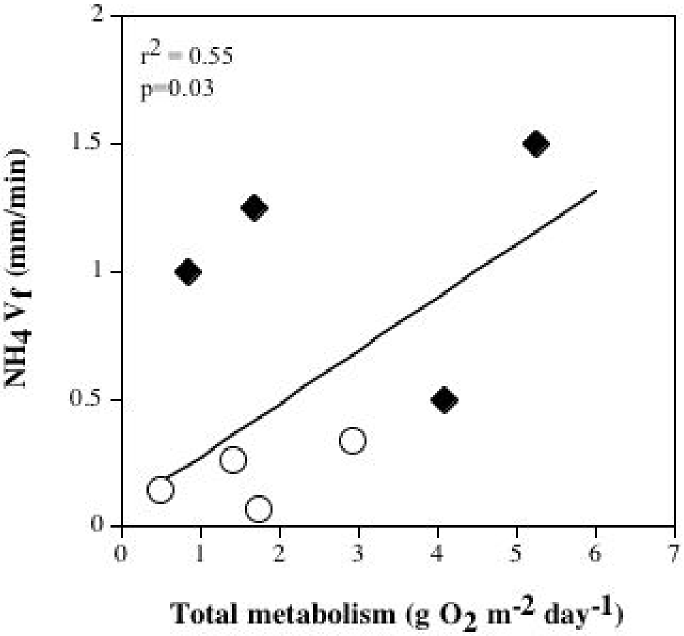
Table 5.2: Summary of results of nutrient addition experiments in forest and urban streams. For transient storage zone size * indicate too high or too low Damkohler numbers. Units for GPP and R are g O_2 m^{-2} day^{-1}. – indicates no data

Site	Discharge (L/s)		Width (m)		Total N (μg/L)		Total P (μg/L)		transient storage zone (k1/k2)		% composition of sediment		
	dry	wet	dry	wet	dry	wet	dry	wet	dry	wet	bedrock	cobble	sand/silt
Forest 1	1.2	2.6	1.05	1.03	136	111	28	33	0.84	1.65	17	28	55
Forest 2	2	4.2	1.24	1.68	123	282	52	32	0.73	0.79	3	65	42
Forest 3	3.3	13.4	1.68	2.56	165	151	36	28	0.67	0.16	40	38	22
Forest 4	3.2	9.3	1.6	1.99	109	533	19	23	*	0.23	27	33	40
Urban 1	1.8	3.6	2.15	2.83	420	1210	23	48	1.34	0.07	18	47	35
Urban 2	1	2.8	1.86	2.85	345	594	139	71	1.2	1.75	31	36	33
Urban 3	1.5	6	2.49	3.27	418	1001	90	73	1	0.68	20	55	25
Cons. Sub.	2	2.8	2.19	2.63	633	762	77	71	*	0.82	11	41	30

Site	NH_4 S (km)		SRP S (km)		NH_4 V_f (mm/min)		SRP V_f (mm/min)		GPP	R	epi lithon chlorophyll a (mg/m^2)		epilithon org. matter (mg/cm^2)	
	dry	wet	dry	wet	dry	wet	dry	wet	wet	wet	dry	wet	dry	wet
Forest 1	63	105	51	206	1.12	1.45	1.38	0.74	0.59	3.49	4.72	7.18	1.47	0.61
Forest 2	69	315	130	103	1.42	0.48	0.75	1.47	0.06	0.79	3.4	6.50	0.24	0.33
Forest 3	162	488	92	118	0.74	0.64	1.30	2.66	0.18	1.5	4.81	7.90	0.28	0.25
Forest 4	53	291	109	64	2.23	0.97	1.08	4.40	0.09	5.15	9.45	12.29	2.33	0.94
Urban 1	89	821		52	0.55	0.07		1.48	0.16	1.58	8.27	23.92	0.93	1.09
Urban 2	61	394	151	172	0.54	0.15	0.22	0.34	0.06	0.44	24.64	24.28		0.75
Urban 3	52	415	97	197	0.70	0.26	0.37	0.56	0.11	1.31	25.82	11.22	0.25	0.54
Cons. Sub.	26	189	48	85	2.08	0.34	1.12	0.76	0.06	2.86	21.93	5.54	0.45	0.36

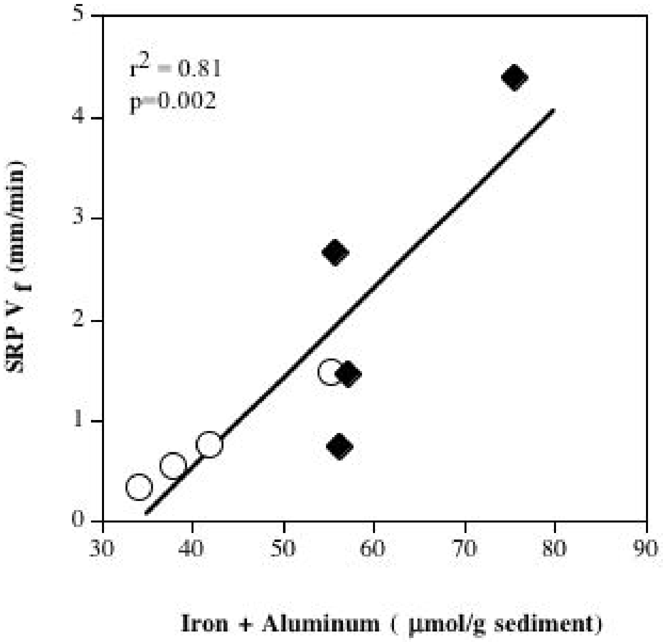
Table 5.3: Differences in ambient nutrient concentration and nutrient uptake velocity in the dry and wet summers. Paired t-tests were used to test for differences between the two summers.

	Dry (2002)	Wet (2003)	t-stat	p-value
Total N (μg/L)	293	580	2.79	0.03
Total P (μg/L)	58	47	1.11	0.3
NH_4^+ V_f (mm/min)	1.17	0.55	2.67	0.03
SRP V_f (mm/min)	0.89	1.55	1.2	0.23



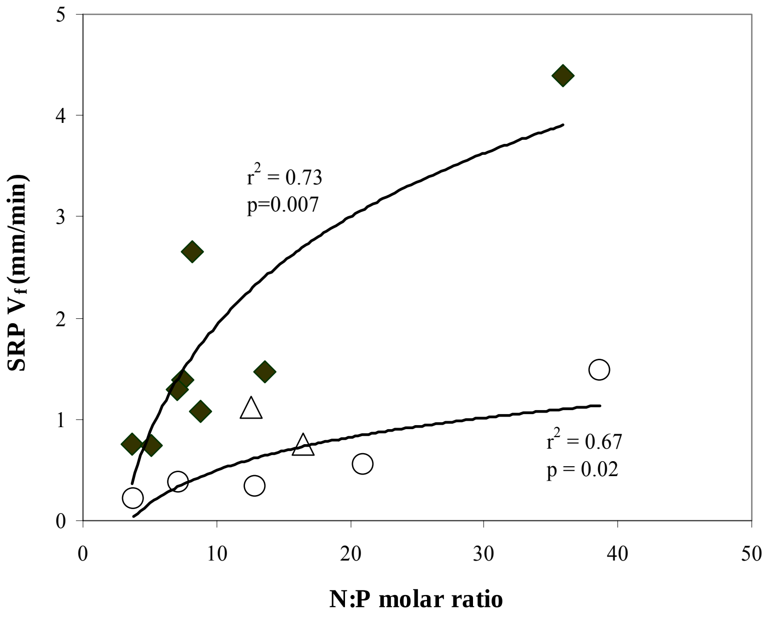
$r^2 = 0.55$
$p=0.03$
NH_4 V_f (mm/min)
Total metabolism (g O_2 m^{-2} day^{-1})
0
0.5
1
1.5
2
0
1
2
3
4
5
6
7

Figure 5.1



$r^2 = 0.81$
p=0.002
SRP V_f (mm/min)
Iron + Aluminum (µmol/g sediment)
0
1
2
3
4
5
30
40
50
60
70
80
90

Figure 5.2



SRP V_f (mm/min)
N:P molar ratio
r² = 0.73
p=0.007
r² = 0.67
p = 0.02
0
1
2
3
4
5
0
10
20
30
40
50

Figure 5.3

CHAPTER 6

CONCLUSION

Approximately 43% of the Earth's vegetated land has become degraded as a result of the impacts of land-use (Daily 1995). This degraded land has a diminished capacity to supply humanity with direct benefits such as agriculture and forestry as a result of soil degradation, desertification, and tropical moist forest degradation (Daily 1995). Ecosystem services are the benefits that human communities receive from ecosystem processes (Costanza et al. 1997, Daily 1997). Therefore, when vegetated land is degraded the ecosystem services provided are reduced. Aquatic ecosystems are also being severely altered or destroyed at a far faster rate than they are being restored (Baron et al. 2002). Human activities have altered the global-water cycle through the consumptive use of both ground water and surface water (Jackson et al. 2001). The most common impairment of surface waters in the United States is eutrophication caused by excessive inputs of nitrogen (N) and phosphorus (P) (Carpenter et al. 1998). In addition, there are only 42 rivers with greater than 200 river kilometers unregulated by major dams in the contiguous United States (Benke 1990). Despite the pervasiveness of these problems, there is little information about how anthropogenic modifications to streams and their catchments affect ecosystem processes and the ecosystem services provided by these processes in streams and rivers.

In this dissertation I have examined the impact of three human modifications, climate change, dam operation, and urbanization on stream ecosystem processes and the services they provide. Human-mediated climate changes have reached or exceeded the level of natural fluctuations in climate (Levine 1992, Vitousek 1992, Schindler et al. 1996). Urbanization is an important component of land-transformation world wide, and close to 80% of the US population is urban (Grimm et al. 2000). Urbanization leads to changes in the hydrology, geomorphology, nutrient and contaminant loading, and species richness of streams (Paul and Meyer 2001). As described above, dams are one of the most pervasive forms of flow alteration on large and some smaller rivers and streams (Dynesisus and Nilsson 1994). I examined the effects of these three perturbations by asking the following questions:

1. Do future climate scenarios result in changes in flow regime in two river basins (Cle Elum, WA and Apalachicola-Chattahoochee, AL, GA, FL), are these different from changes in flow regime caused by dams, and how would these changes affect ecosystem processes and biota in rivers? (Chapter 2)
2. Can flow regime explain the variation in ecosystem metabolism in the Chattahoochee River near Atlanta, GA, or are other physical, chemical, and biological factors needed? (Chapter 3)
3. Are nutrients from Atlanta wastewater treatment plants being retained by the Chattahoochee River, and what environmental parameters control nutrient uptake? (Chapter 4)
4. How does urbanization affect nutrient uptake in headwater streams in the Upper Etowah River Basin, GA? (Chapter 5)

Chapter 2 showed that future climate scenarios can cause dramatic shifts in flow regimes. Changes in timing and form of precipitation can affect the timing of maximum and minimum flows. For example in the Cle Elum River, future climate scenarios predicted more winter rains and less winter snows which shifts peak monthly flows from late spring/early summer to winter. A precipitation shift from snow to rain in the Cle Elum River would increase flood frequency because precipitation from individual winter storms would not be stored as snow. The earlier and more frequent floods would interfere with life-cycles of native fishes and reduce organic matter retention time, thereby reducing food availability for detritivorous macroinvertebrates. Human demands for freshwater can intensify the flow regime changes in areas that are predicted to have less rain under future climate scenarios. For example, the coupling of the future climate scenario with increased human demand resulted in lower high and low flows in the Apalachicola-Chattahoochee-Flint basin. These lower summer flows in the Apalachicola River would decrease the percentage of the floodplain connected to the river which would decrease organic matter inputs from the floodplain to the river as well as habitat for fish species. In addition, these flow regime changes would result in changes in habitat availability during critical times of the year for fish survival in both the ACF Basin and the Cle Elum River. Therefore changes in flow regime associated with climatic changes can alter organic matter delivery and storage as well as habitat for aquatic organisms.

In Chapter 3, the level of hydropeaking during the previous week was negatively correlated with gross primary production in a regulated urban river, the Chattahoochee River. This reduction is most likely the result of higher flows disturbing the benthic periphyton community. The Chattahoochee River receives a large amount of point-source inputs and sewage-derived particulate organic matter. Therefore, maintaining dissolved oxygen concentrations is of concern in this river. The reduction of primary production as a result of hydropeaking decreases the production of oxygen in the river as well as reducing algal-based food resources for fish and macroinvertebrates. In addition, reduction in GPP reduces capacity for nutrient uptake and retention. Community respiration in the Chattahoochee River was not strongly controlled by discharge. Instead, respiration was strongly correlated with suspended fine particulate organic matter and total P concentration. Sewage-derived particulate organic matter is a highly labile mix of organic detritus and microorganisms such as bacteria and algae (deBuryn and Rasmussen 2002). Organic matter processing in the river helps limits organic matter loading to downstream reservoirs. Organic matter sedimentation can lead to anoxia in the hypolimnion; creating habitat limitation for fish and zooplankton (Cole and Hannan 1990, Porter et al. 1996). Respiration in the Chattahoochee was higher than in most other river systems including the Ogeechee River, a blackwater river with high dissolved organic carbon concentrations and large organic matter inputs from the floodplain. Therefore it appears that flow regulation by hydropeaking decreases primary production, but organic matter and nutrient loading increases respiration in this regulated, urban river.

Chapter 4 demonstrated that nutrient uptake in the Chattahoochee River is highly variable with measurable NH_4^+ uptake only occurring on 4 out of 16 dates and measurable NO_3^- uptake only occurring on 6 out of 16 dates. Nutrient uptake length was not related to discharge, however we did not sample over a wide-range of discharges. The element that most consistently demonstrated measurable uptake was soluble reactive phosphorus (SRP). However, measurable uptake in the upper reach was almost always followed by measurable increases in the lower reach. Therefore, SRP retention was temporary at best. In addition, SRP uptake length was positively correlated with total suspended solid concentration in the water column suggesting SRP retention is controlled by P binding to the sediments. Phosphorus is readily mobilized from the sediments as the binding and release of P is an equilibrium reaction (Wetzel 2001). Sediment phosphorus concentrations were high in the Chattahoochee River and could account for the observed increases in concentration in the reach. Therefore, retention through the binding of P is temporary, and this P is likely to be mobilized quickly, creating water quality problems downstream.

Nitrogen retention was even more variable. These reaches were more likely to be sources of NH_4^+ than sinks because there were more dates with increases in NH_4^+ than there were dates with decreases. Nitrate uptake velocity was positively correlated with suspended fine particulate organic matter, suggesting that biological uptake of microorganisms associated with the organic matter controlled nitrate uptake when it occurred. Overall, demand for both NH_4^+ and SRP as measured by uptake velocity was significantly lower in the Chattahoochee River than in less modified systems. Therefore, although the river has maintained its ability to retain nutrients, its capacity has been diminished.

Ambient nutrient concentrations, water residence time, and epilithon chlorophyll a content were all higher in urban headwater streams than in forested headwater streams in Chapter 5. However, NH_4^+ and SRP uptake velocities were lower in the urban streams compared to the forested streams. Phosphorus uptake in these streams was controlled by chemical sorption to the sediments as demonstrated by the strong positive relationship between SRP uptake velocity and iron and aluminum content of the sediments. Ammonium uptake in these systems was controlled by biological processes as indicated by the positive correlation between NH_4^+ uptake velocity and the sum of gross primary production and respiration. Ammonium uptake velocities were lower in the wet year than in the dry year, but SRP uptake velocity was similar in the two years.

The urban streams in this study maintained complex channels that retained water and had substrates that were suitable for algal attachment. By maintaining function channels, these urban streams maintained the ability to retain nutrients, although in a diminished capacity. This result was even more evident in the urban stream located in a conservation subdivision that maintained wide riparian zones around streams. In the conservation subdivision stream, nutrient uptake velocity was intermediate between urban and forest, and in the dry year it had the highest NH_4^+ uptake velocity. The urban streams in this study have maintained the ability to retain nutrients, but urbanization has diminished the demand for nutrients in this system. In addition, it appears that by conserving functioning riparian areas and channels some of the effects of urbanization can be reduced and maintain nutrient retention.

Overall the results of this study demonstrate that streams and rivers are resilient ecosystems that maintain their ability to provide ecosystem services to human societies despite major alterations resulting from urbanization and flow regulation. However, for the most part, human alterations have diminished the capacity of these rivers and streams to provide these services. Therefore, reductions in hydropeaking, decreases in nutrient loading, and maintenance of riparian buffers and channel structure could increase the ability of urban streams and rivers to provide the ecosystem services that are of value to human communities.

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