

Providing a Restoration Framework for Regulated Rivers

Ryan Austin McManamay

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Donald J. Orth  
Charles A. Dolloff  
Emmanuel A. Frimpong  
Paul L. Angermeier  
Theresa M. Wynn  
Mark A. Cantrell

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## ABSTRACT

With over 800,000 dams occurring globally and the construction of thousands more being proposed, successful restoration of regulated rivers will depend on the creation of broadly applicable frameworks that provide management solutions by generalizing patterns in habitat and ecology. Based on the prevailing scientific literature, restoring natural stream flows in disturbed rivers is dependent upon developing quantitative, transferable stream flow-ecology relationships. The purpose of my dissertation was to apply a framework to regulated and unregulated streams within an eight-state region of the southeastern US to test its ability to generalize patterns in natural and altered stream flow and develop flow-ecology relationships. I created a simplified, 5-step version of the Ecological Limits of Hydrologic Alteration (ELOHA) framework (Poff et al. 2010). I carried out each of the steps in sequential order for unregulated and dam-regulated streams found in my region. The steps of my restoration framework are as follows:

- 1) Develop a natural flow classification of unregulated streams
- 2) Develop a tool that uses landscape characteristics to predict flow class membership
- 3) Use the predictive tool or pre-disturbance hydrologic information to classify regulated rivers to natural flow classes
- 4) Based on class membership, generalize patterns in hydrologic alteration
- 5) Relate ecological patterns to patterns in hydrologic alteration in relation to morphology, temperature, and landscape disturbance

Altogether, the results of steps 1-4 suggest that patterns in natural flow dynamics and hydrologic alterations can successfully be placed within a framework and generalized to provide the basis and context for environmental flow management; however, results of step 5 suggest that patterns in flow alteration were poorly related to fish assemblages

relative to channel morphology, habitat fragmentation, temperature, and substrate. Thus, the development of patterns in hydrologic alteration using the existing frameworks (including mine) may not be ecologically-relevant. My results suggest that current regulated river restoration should not be dependent upon the development of flow-ecology relationships alone, but the interaction between flow, morphology, and temperature within a landscape disturbance context. These relationships should be incorporated within a hierarchical framework to guide restoration efforts in regulated rivers in the future.

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## 1. Introduction

Currently, less than two percent of the US's rivers and less than 40 % of the world's large rivers run un-impounded from their headwaters to the ocean (Vitousek et al. 1997). In the US alone, there are over 42,000 dams greater than 7 m high, 6,300 of which are large dams (> 15 m) (USACE 2011). Globally, there are over 800,000 dams, 42,000 of which are large dams (McCully 1996, Rosenberg 2000). According to Richter (2011), "nearly 400 dams are planned or under construction in Central America, 200 in Brazil, and, in China, nearly 50 in the Yangtze River basin alone" in response to poor living conditions. In summary, "living with dams" is the new status quo (McCartney 2009).

The list of problems caused by damming river systems is quite extensive. The most obvious of these can be the complete loss of free flowing environments above the dam and the transformation of flowing environments below the dam. Peak flood magnitudes are diminished whereas minimum flow flows are inflated (Richter et al. 1996; Magillan and Nislow 2001 and 2005; Pyron and Neumann 2008; Poff et al. 2007). Massive reductions in the flow variability has resulted in a homogenization of regionally and continentally-distinct fluvial habitats across global scales (Poff et al. 2006a; Poff et al. 2007). Interestingly, the majority of other problems caused by dams are inter-related with flow.

Dams interrupt the primary function of river systems: to transport abiotic and biotic materials from upstream sources (Vannote 1980; Kondolf 1997). Sediment transport in regulated rivers is limited, if not stopped, by settling out within impoundments (Kondolf 1997). Dams capture approximately 25 % of total rill and sheet erosion from the US landscape (Renwick et al. 2005). The impacts of dams are not just a unidirectional upstream-to-downstream problem but also a lateral problem. Dams reduce the hydrologic connectivity of a river with its floodplain and alter the morphology of the channel that carries its flow. Dams alter the timing, magnitude, frequency, and duration of floods that pulse riparian areas important for organic and sediment inputs (Poff et al. 1997; Trush et al. 2000; Nislow et al. 2002). Dams tend to decrease bankfull area, decrease sinuosity, and increase riparian vegetation due to encroachment (Gordon and Meentemeyer 2006).

The thermal regime of river systems can also be substantially modified by dams (Pozo et al. 1997; Hamblin and McAdam 2003; Lessard and Hayes 2003; Krause et al. 2005; Olden and Naiman 2010). Releases from stratified layers of the impoundment can result in temperature regimes drastically different from natural conditions. For example, hypolimnetic releases can cause dramatic reductions in temperature (Pozo et al. 1997; Krause et al. 2005) whereas releases from the surface of the reservoir can lead to increases (Lassard and Hayes 2003; Caissie 2006). Interestingly, flow and temperature are related (Caissie 2006). For example, larger releases from dams results in higher thermal buffering capacity whereas reductions in discharge below reservoirs result in lower thermal buffering capacity and generally, higher annual temperatures (Caissie 2006).

Of the impacts of dams on biota, habitat fragmentation cannot be ignored. Obviously, dams block the migration and dispersal of organisms between populations and spawning habitats (Vaughn and Taylor 1999; Han et al. 2008; Hoagstrom et al. 2008; Reid et al. 2008). Dams have been the principal cause of the decline in the great Pacific salmon runs. Unfortunately, the fish, invertebrates, mammals, birds, reptiles, and humans that inhabit the water and banks of the world's rivers have had to bear the burden for dams. The US pays at least \$1 billion per year to restore stream habitats in the United States (Bernhardt et al. 2005). The annual cost of supporting Pacific salmon recovery alone is estimated at \$400 million (Katz et al. 2007).

### *Restoring Impounded Rivers*

The degradation of the world's freshwater habitats has led to global restoration efforts (Roni et al. 2008). Dam breaching has increased and the documented results seem positive (Doyle et al. 2003; Hart et al. 2002; Stanley and Doyle 2003); however, the majority of removals have targeted smaller dams. In addition, with current energy demands, the majority of large, hydroelectric facilities (those with the greatest impact) will remain in place for decades following relicensing. In the following section, I provide a brief overview of the existing dominant frameworks that have been used by managers in restoring various aspects of regulated river systems (later I will discuss their application to my objectives/chapters):

*Flow* - The majority of restoration efforts *in regulated river systems* have been focused on restoring flow (Tharme, 2003; Roni et al. 2008). This probably is due to two main reasons: 1) a river is not a river without flowing water and 2) the natural flow regime influences the majority, if not all, physical and chemical relationships in river systems and organizes the structure of aquatic and riparian communities (Poff et al. 1997). It is broadly accepted that a naturally variable flow regime is required to sustain river ecosystems (Richter et al. 1996; Poff et al. 1997; Bunn and Arthington 2002). However, understanding the ecological needs of the aquatic community and using that information to make environmental flow recommendations for regulated rivers is not an easy process. Two major frameworks were developed to aid in this process: 1) Instream-Flow-Incremental-Methodologies (IFIM) and 2) Outside-stream methods. IFIM approaches, include physical habitat simulation modeling (PHABSIM) and range from simple relations between hydrologic indices and aquatic habitats to more complex hydrodynamic models, which are generally very expensive, in terms of time and money, and may or may not be linked to any relevant components of the river ecosystem (Tharme, 2003; Anderson *et al.*, 2006).

The outside method basically includes a suite of simple to complex methods that range from comparing pre/post disturbance hydrologic information to creating regional comparisons of flow-ecology relationships. Richter et al (1996) developed the Indicators of Hydrologic Alteration (IHA) and the Range of Variability Approach (RVA). The methods are dependent upon the presence of pre and post-dam-regulation discharge data. Multiple ecologically-relevant hydrologic indices are calculated for pre and post-regulation periods of record and then compared. Comparisons yield information such as which aspects of the hydrograph should be the prioritized when proposing new dam operations. However, the IHA and RVA methods are limited not only in that they require pre and post-dam flow data but also in that they have no quantitative relationship to in-stream ecological needs. In addition, the IHA and RVA methods still operate on an individual river basis.

The need for standardized flow-ecology relationships provided the motivation for the creation of river flow classifications (Poff and Ward 1989, Poff 1996). Therefore, instead of managing for every individual river, classes of rivers with similar hydrologic

properties across regions can be used to develop standards for managing flow needs (Poff, 1996; Arthington *et al.*, 2006). These classes then form the foundation for assessing hydrologic alterations and flow-ecology relationships (Arthington *et al.* 2006). The ELOHA (Ecological Limits of Hydrologic Alteration) framework is the latest development of the classification-based management and provides a holistic framework for regulated river management (Poff *et al.* 2010) (Fig. 1.1). The framework includes 5 main steps: 1) building a hydrologic foundation of pre/post disturbance conditions, 2) classifying river types based on hydrology and “other” characteristics, 3) assessing flow alterations, 4) determining flow-ecology relationships for each river type, and 5) implementing policies based on the findings. Therefore, the ELOHA framework is a broad-scale mechanism that can be used to efficiently develop environmental flow policies for flow restoration and develop environmental flow policies that are sensitive to ecological needs.

*Temperature* - Although far less documented than flow, restoration efforts focused on improving altered temperature regimes have also been conducted below impoundments (Price and Meyer 1992, Krause *et al.* 2005). The standard protocol for IFIM methodologies includes modeling temperature with variable discharge releases (Bovee *et al.* 1998; Krause *et al.* 2005). Krause *et al.* (2005) simulated temperature scenarios using a hydrodynamic model coupled with temperature modeling to assess the influence of various flow releases on the thermal regime in a regulated river in Virginia. According to Olden and Naiman (2010) incorporating thermal regimes into environmental flow assessments is rarely done. They indicate that some of the challenges of incorporating assessments of thermal regimes into flow assessments are due to a lack of understanding of the impact of dams on temperature, the ecological consequences of an altered temperature, and limited knowledge of the availability and success of temperature management strategies. Sherman (2000) provides a review of various temperature mitigation strategies considering dam structure and the use of stratification layers to mimic natural temperature conditions. Frameworks for restoring temperature are not as prevalent as flow; however, Price and Meyer (1992) provide a guide to operational and structural water quality management techniques for reservoirs and tailwaters.



*Morphology* - Restoring bedload transport and spawning habitats below impoundments through gravel and sediment augmentation has been documented largely in salmonid rivers in the Western, US (Kondolf et al. 1996; Merz and Setka 2004; Merz and Chan 2005; Sarriquet et al. 2007) and to a lesser extent in the eastern US (McManamay et al. 2010). In addition, modeling sediment in relation to flow has been conducted considerably (Wilcock et al. 1996a; Wilcock et al. 1996b; Singer and Dunne 2006) and many sediment transport models are currently available (HECRAS, SHIRA, etc) (Bunte 2004). Bunte (2004) provides a detailed guide to gravel mitigation and augmentation below dams and provides a conceptual framework in determining the amount, location, and timing of gravel additions. The geomorphic consequences of dam regulation on channel morphology are well known (Ligon et al. 1995; Grant et al. 2003; Gordon et al. 2004); however channel morphological restoration, such as removing encroached riparian vegetation, creating bar habitats, and channel grading, in large regulated rivers is rarely documented in the literature (Trush et al. 2000). Broad frameworks for morphological restoration below dams are largely missing most likely the restoration needs is individual to every river system. However, Grant et al. 2003 created a conceptual and quantitative framework for assessing the effect of dams on morphology.

#### *Applying restoration frameworks to research*

The utility of frameworks used in conservation management is largely based on their broad applicability. Successful restoration depends on developing appropriate endpoints or goals and considering the context of each management situation (Roni et al. 2002). However, the development of appropriate goals will depend largely on the creation of measurable objectives (Tear et al. 2005). Frameworks that provide a template for creating measurable objectives and operate across larger scales are needed. In addition, frameworks should the context for restoring aspects of river systems considering that the dam is still intact.

Recent environmental flow management and potentially regulated river restoration at large, has been thwarted, at least to some degree, by the absence of quantitative, transferable flow-ecology relationships (Poff et al. 2010; Poff and Zimmerman 2010). The development of flow-ecology relationships will aid in

prescribing the environmental flow needs of rivers with regulated or substantially altered flow. The purpose of my dissertation was to use a broad framework to guide research concerning restoration in regulated rivers and to develop flow-ecology relationships. My framework is a simplified 5-step version of the ELOHA (Ecological Limits Of Hydrologic Alteration) conceptual model and includes ideas inspired by Arthington et al. (2006) (Fig. 1.2). My framework is based on the ELOHA model because it is currently, the most holistic and widely-accepted framework for regulated river flow management (based on literature, Poff et al. 2010) and it has become the framework used by The Nature Conservancy (Conservation Online 2011), one of the largest leading groups in the environmental flow arena. Unfortunately, the current ELOHA framework does not take into account morphological alterations or temperature alterations (Olden and Naiman 2010; Poff et al. 2010). In addition, to my knowledge, there are no channel morphology and temperature frameworks that are as widely-applicable and widely-used in managing regulated rivers. The fact that the framework structure largely ignores temperature and morphology in is an artifact of the lack of sufficient and readily accessible data at large spatial scales. In essence, flow, channel morphology, and temperature all interact to influence riverine ecology; thus, these aspects should be incorporated into a framework for assessing the influence of flow alterations on ecology. Otherwise, the majority of assessments will be very coarse (Carlisle et al. 2010b) and may fail to provide a basis for quantitative prescriptions for restoration applications. For example, in a review of 165 published papers on the ecological responses of altered flow regimes, Poff and Zimmerman (2010) conclude that “our analyses do not support the use of the existing global literature to develop general, transferable quantitative relationships between flow alteration and ecological response.”

The first step of my framework includes creating a classification of “natural” flow regimes of similar hydrologic properties. The flow classes then become management units that provide environmental flow recommendations. Secondly, flow classes are related to the landscape in order to understand what climate, soils, and topographical variables govern natural flow but also to provide a predictive watershed tool to classify streams without sufficient pre-dam-disturbance hydrologic information. In the third step, regulated rivers are classified to a natural flow class either by using pre-disturbance

hydrologic information or by using the watershed predictive tool. The fourth step includes generalizing patterns in how dams influence hydrology. It becomes evident that my framework differs from the ELOHA framework in the fifth step. Although the ELOHA framework does incorporate a geomorphic subunit classification in the river classification step, it does not account for *altered* morphology or *altered* temperature regimes when assessing flow-ecology relationships. I incorporate the effects of altered channel morphology and temperature in the last step of my framework in order to understand the relative importance of these factors in influencing fish assemblages relative to flow.

There are two ways that altered channel morphology and altered temperature regimes could conceivably be included in the framework. The first way, is by including channel morphology and temperature in the river classification step and then generalizing the effect of dams on channel morphology and temperature within classes. Although this would provide a robust approach to compare to fish assemblages, assessing morphology and temperature over large spatial scales and acquiring this information for every regulated river system would prove difficult. The second way (my method) is by accounting for altered channel morphology and temperature during the 5<sup>th</sup> step (when fish assemblages are related to flow alterations). The limitation of this method is that analyses relating altered flow, channel, and temperature to fish assemblages are confined to smaller spatial scales than that of the original flow classification because of the effort required to collect relevant variables. Once relationships are made, managers can decide whether flow, substrate variables, and even habitat connectivity (dam removal) are in need of restoration.

My hope is that the framework discussed herein can provide a broad approach to restoring flow and developing flow-ecology relationships in dam regulated systems. My target audience is managers who are engaged in relicensing agreements at the Federal Energy Regulatory Commission (FERC) roundtable, ecologists/biologists whose role is to provide statewide criteria for withdrawal permitting issues, or scientific investigators who plan to develop flow-ecology relationships. The burden of taking the little existing knowledge of the fluvial habitat needs of aquatic biota and applying that to an environmental flow standard setting rests on the shoulders of these individuals. My

framework will hopefully provide the context or actual tools to develop environmental flow standards and decrease the complexity of managing for disturbed river systems by providing 1) ecologically relevant management units (classes of streams that share similar hydrology), 2) tools to predict flow in basins with limited or no gage information, 3) generalizations in patterns of how dams influence flow (i.e. prioritize indices that show general and most substantial changes due to regulation). By evaluating patterns in fish assemblages, I also attempt to provide evidence that temperature and morphology are certainly needed if the goal of future investigations is to develop transferable, quantitative flow-ecology relationships.

For my dissertation, I follow the conceptual model in Figure 1.2. I focus my efforts on the southeastern United States because of the increasing water demand from multiple sources and the need for a framework to develop sustainable water management (Sun *et al.*, 2008). Secondly, channel morphological restoration in regulated rivers, is not as widespread in the southeast as in the western US (McManamay et al. 2010). My objectives are as follows: 1) develop a flow classification of unregulated streams in an 8-state region of the Southeast (Step 1), 2) relate flow classes to the landscape to develop a watershed predictive tool (Step 2), 3) classify regulated rivers to natural flow classes and generalize patterns in hydrologic alteration (Steps 3 & 4), and 4) determine the relative importance of flow alterations, channel alterations, and other factors (such as landuse and temperature) on influencing fish assemblages in regulated and unregulated rivers the upper Tennessee River basin. I conclude by applying my restoration template to observations made in the Cheoah River, a regulated system in western NC.

# ELOHA (Ecological Limits of Hydrologic Alteration)

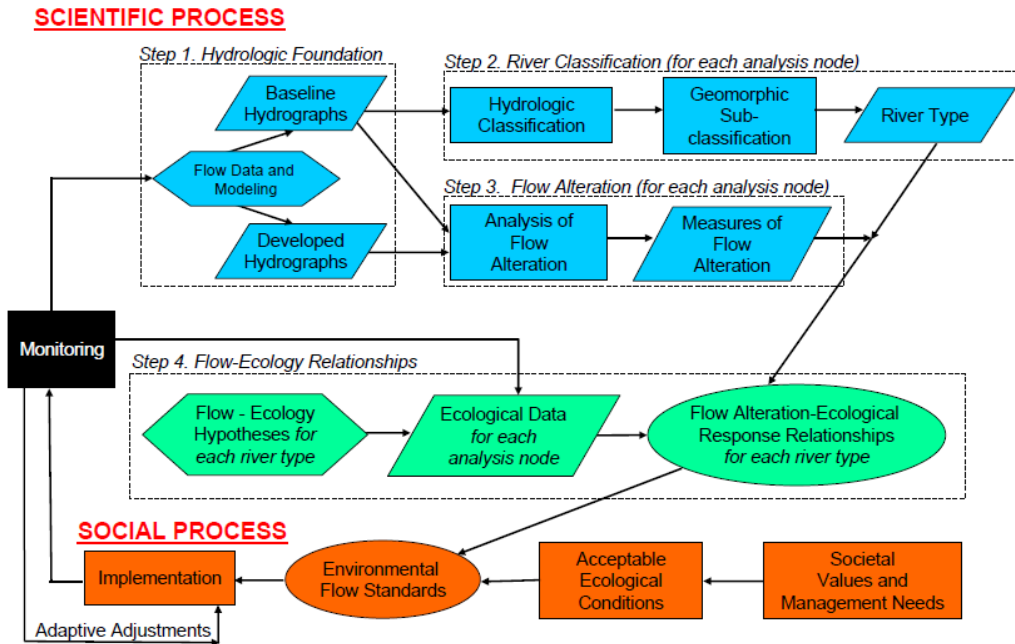


Figure 1.1 The Ecological Limits of Hydrologic Alteration (ELOHA) framework (taken directly from Poff et al. (2010)). The framework includes 5 main steps, which can be used to efficiently develop environmental flow policies for flow restoration and develop environmental flow policies that are sensitive to ecological needs. For more information on the ELOHA framework and process see Poff et al. (2010).

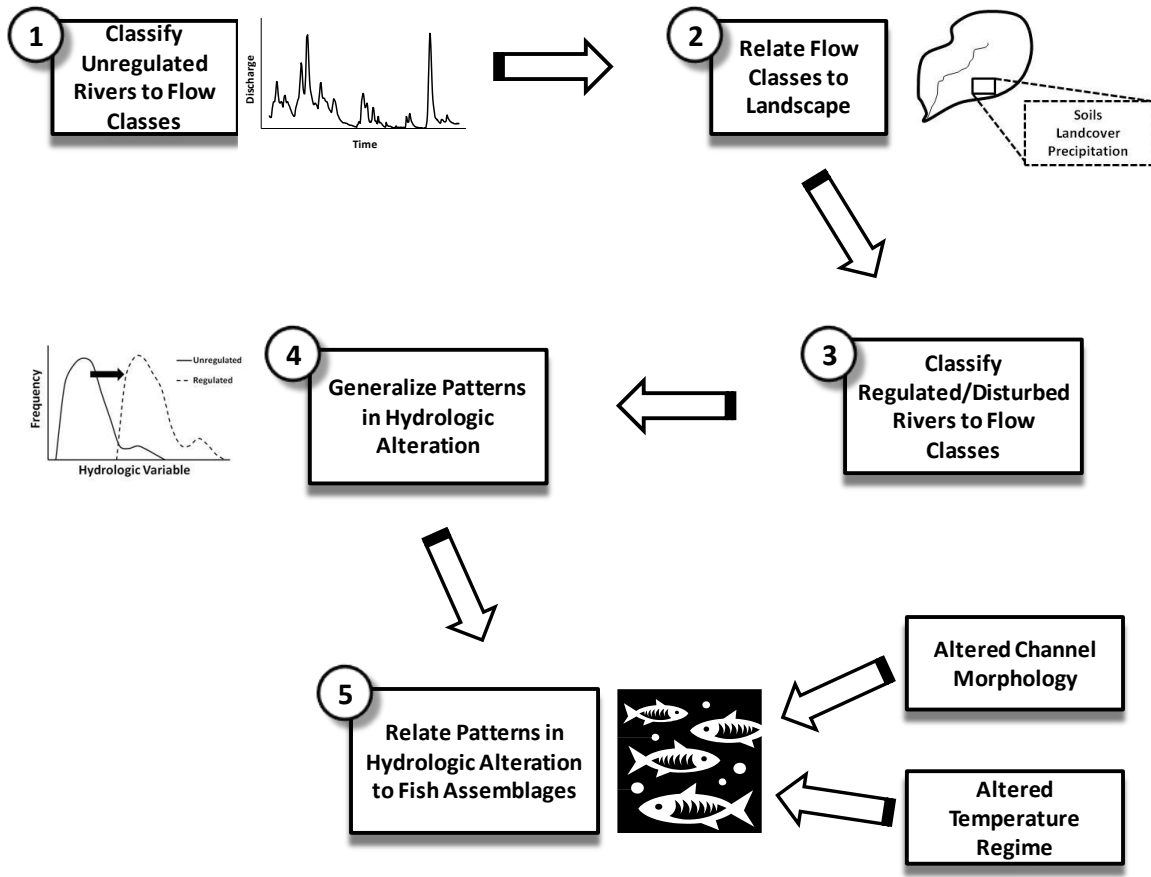


Figure 1.2 Conceptual model of the 5-step “Restoration Template” tested in this research. Ideas for the template were inspired by the ELOHA framework (Poff et al. 2010) and Arthington et al. (2006). Arrows indicate steps in the process and the flow of scientific investigations. The steps include 1) developing a classification of unregulated stream flows, 2) relating flow classes to the landscape to develop a watershed predictive tool, 3) using the landscape predictive tool to classify regulated rivers to natural flow classes, 4) generalize patterns in hydrologic alteration, and 5) determine the relationship between flow alterations and ecology relative to channel morphology and temperature in a landscape disturbance context.

## **2. A Regional Classification of Unregulated Stream Flows: Spatial Resolution and Hierarchical Frameworks**

### **Abstract**

River regulation has resulted in substantial losses in habitat connectivity, biodiversity, and ecosystem services. River managers are faced with a growing need to protect the key aspects of the natural flow regime. A practical approach to providing environmental flow standards is to create a regional framework by classifying unregulated streams into groups of similar hydrologic properties, which represent natural flow regime targets. Because spatial resolution can influence the structure of regional datasets, it may be advantageous to relate datasets created at different scales in order to establish hierarchical structure and to understand how the relative importance of variables change with regard to scale. The purpose of this study was to classify unregulated streams within an eight-state region into groups in order to provide environmental flow standards for managers and to relate that dataset to frameworks created at larger scales. Using USGS daily stream gauge information, I used 66 hydrologic statistics to classify 292 streams in groups of similar hydrologic properties. I isolated six flow classes in a sub-region of the Southeastern US that ranged from extremely stable to highly-variable to intermittent. I developed classification trees to reduce the number of hydrologic variables for future classifications. By comparing flow classes in our study to those of the entire US, I found that hierarchical structure did exist and that the divergence of flow classes will largely depend on the spatial resolution.

<sup>1</sup> Appears as:

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## 2.1 Introduction

River regulation has resulted in substantial losses in natural flow variability, habitat integrity, and consequently, species diversity (Poff et al. 1997; Vitousek et al. 1997; Pringle et al. 2000; Poff et al. 2007). The natural flow regime (magnitude, frequency, duration, timing, and rate of change in flow events) is essential for creating and maintaining habitat in river channels, transporting sediment, and connecting rivers and their floodplains (Poff et al. 1997). Hydrologic disturbances create and maintain habitat heterogeneity (Trush et al. 2000) and stabilize food webs (Wootton et al. 1996; Cardinale et al. 2005). Dams alter the frequency and duration of floodplain-inundation (Nislow et al., 2002), which decreases bankfull area and lateral migration while also increasing riparian encroachment (Gordon and Meentemeyer 2006). Substantial withdrawals have either left rivers without any water or dramatically reduced flows to the extent that river ecosystem function is lost (Poff et al. 2003). The “homogenization” of natural flow variability across geographic scales (Poff et al. 2007) has resulted in the decline of species whose life history strategies are adapted to the natural variation in flow regimes (Bunn and Arthington, 2002; Poff et al. 1997).

With over 82,000 dams in the US (USACE 2011) and water rights battles across the country (Poff et al. 2003), river managers are faced with a growing need to protect the key aspects of the natural flow regime. However, managing for the specific needs of every river and their associated biotic community is easier said than done. The interaction between social, economic, political, and finally, ecological demands results in simple and general flow rules that ignore the complexity of flow variability responsible for sustaining river systems (Arthington et al. 2006). One practical approach to providing environmental flow standards is to form classes of rivers with similar hydrologic properties across regions from which standards for managing flow needs can be developed (Poff 1996; Arthington et al. 2006). Each flow class then becomes a hydrologic unit for management rather than managing for the individuality of each and every river system. The assumption is that rivers within similar hydrologic units of the same zoogeographic region are also similar ecologically, in terms of community composition, functional groups, and responses to flow variability. Also, broad generalizations concerning the impacts of flow regulation on multiple groups of riverine



biota has only recently received attention (Poff and Zimmerman 2009); thus, a regional framework to evaluate biotic responses to flow regulations should be advantageous.

The other main approach to developing environmental flow standards is by using instream-flow-needs (IFN) techniques that relate flow to ecological targets. IFN approaches range from simple relations between hydrologic indices and aquatic habitats (i.e. weighted usable area) to more complex hydrodynamic models, which may or may not be linked to approaches that relate flow variability to many components of the river ecosystem (Tharme 2003; Anderson et al. 2006). PHABSIM models have been used extensively in the US and worldwide (Spencer and Hickley 2000; Tharme 2003) and have been used with success in providing insights into potential ecological responses to flow alterations (Gallagher and Gard 1999). However, these models are applicable to only the reach under study (Moir et al. 2005), biased by site location (Williams 2010), and are generally expensive in terms of time and money (Spencer and Hickley 2000). In addition, PHABSIM models only focus on instantaneous flow conditions rather than the entire flow regime and develop suitability criteria for one up to several target biota rather than the entire river community. Although holistic approaches may consider how a given flow regime may influence multiple components of a river ecosystem, Anderson et al. (2006) argues that these approaches generally do not incorporate process-driven ecological dynamics, especially internal feedback loops, into analyses of the influence of flow on aspects of the ecosystem. Ecological restoration should be founded upon the restoration of processes responsible for maintaining ecosystems (Ward et al. 2001). However, empirical information on the relationship between flow regimes and complex ecological dynamics is extremely limited (Poff and Zimmerman 2010). Furthermore, forming environmental flow standards by evaluating all the complex ecological dynamics (population, community, spatio-temporal) and then translating that scientific information into quantitative demands to policy makers is unrealistic for every regulated river.

Regional flow classifications based on unregulated rivers provide ecologically-relevant units, which are an organized and less-complex framework for developing environmental flow standards for management. Because flow standards are developed using the natural flow regime of unregulated rivers, prescriptions for flow regime alterations are applicable to the entire river ecosystem not just target biota. Future efforts

to create sustainability boundaries (Richter 2010) require acceptable regional flow classification. Flow classifications are also convenient because they can produce flow prescriptions quickly without time-intensive field work, the development of flow-ecology relationships, and high monetary costs. Lastly, the majority of current understanding of the impacts of flow alterations on biota is case-specific knowledge (Poff and Zimmerman 2010); thus, regional flow classifications may provide a framework to generalize patterns of disturbance as additional investigations proceed.

The classification of flows based on stream discharge alone is not unprecedented. Poff (1996) classified natural flow variability for 816 streams across the lower 48 states of the US into 10 flow categories using only flow records; however, because of the coarse scale of that study, only 3 flow classes were isolated for the area of interest of this study (GA, KY, MD, NC, SC, TN, VA, WV). Because of the variation in climate and watershed characteristics found at the scale of my study, a higher resolution classification is needed to adequately represent flow classes with distinct hydrologic properties. It is important to clarify that I am not campaigning for one classification over another. In contrast, I believe that coarse and fine resolution classifications are both essential to management and to facilitate the formation of hierarchical datasets of flow variability at different scales. In addition, relating datasets to existing regional frameworks can increase the understanding of how ecosystem dynamics are governed at multiple scales. For example, Hydrologic Landscape Regions (HLRs) were created using landscape characteristics that influence hydrology to provide a hydrological classification to stratify basins (Wollock et al. 2004). I discuss the relationship between HLRs and my flow classification further in Chapter 3.

I chose to focus my classification within the southeastern United States because of the increasing water demand from multiple sources and the need for a framework to develop sustainable water management in the Southeast (Sun et al. 2008). Secondly, for the states found in this region, the practice of making environmental flow recommendations has been to apply state-wide criteria, treating all classes of flow types in a similar way. Obviously, I find this inadequate for protecting the variability in flow regimes that support aquatic biodiversity.

The purpose of this study was to classify unregulated streams based on hydrologic data within an eight-state region of the southeastern US in order to provide environmental flow standards for regulated rivers. Classifications are important to management in that they consolidate large amounts of information into digestible units. Because large amounts of variables can be overwhelming, I also wanted to provide a reduced set of hydrologic indices useful to managers in classifying future streams. I also wanted to compare my dataset to Poff's US flow classification to determine the potential for scale-dependent hierarchical flow classes. Specifically, my objectives were to 1) classify unregulated streams within an 8-state region of the southeast into distinct flow classes important for environmental flow management using only available discharge information, 2) provide a reduced set of hydrologic variables that can be used as foundational indices for future classifications and flow management, and 3) determine the hierarchical structure in the divergence of flow classes between within this study's region and that of the US flow classification.

## **2.2 Methods**

I accessed the USGS Realtime Water Data for the Nation website (<http://waterdata.usgs.gov>) to find daily stream gauge data and to judge the extent of regulation due to impoundment or other hydrologic disturbances. Criteria for relatively undisturbed flow status was determined if the stream had: 1) at least 15 years of data, 2) no upstream impoundments (including tributaries), 3) no large diurnal fluctuations due to withdrawals, and 4) low urbanization and channelization within its drainage area.

I used a four-step process to determine relatively undisturbed status. First, I selected gauges with at least 15 years of total data (some gage records had missing data as long as at least 15 total years were represented). Kennard et al. (2010a) concluded that at least 15 years of record are suitable for estimating variables that are used to detect differences in the spatial variation in stream flows, such as flow classifications. The study also concluded that discharge records should be contained within a temporal window that allows 50% overlap across records. I used the entire period of record available to capture as much of the hydrologic variability possible. However, because some records were from different time periods, had some missing data, and may not have

sufficiently overlapped with other records, I examined whether the variability in hydrologic metrics due to period of record may have overwhelmed or masked the variation within and across classes. I discuss this further in the Temporal Analysis section. In the second step, I used USGS annual water reports to determine that there were no impoundments upstream of the gauge (including tributaries). For stream gauges with extensive records that had at least 15 years of pre-impoundment data, I selected data within periods of time that had no regulated flow to include in my analysis as “natural” conditions. Thirdly, I used USGS annual water reports to eliminate any gauges from my analyses that had large diurnal fluctuations due to withdrawals (wording in USGS reports). Some stream gauges were selected that had slight diurnal fluctuations caused by upstream withdrawals or small mills. However, I assumed that slight diurnal fluctuations would not influence the hydrologic statistics that I used, which are influenced by trends across days and months, not within a 24-hour period. To determine the extent of disturbance due to urban development and fragmentation, I used the hydrologic disturbance index (Falcone et al. 2010). The dataset includes 375 variables for 6785 USGS stream gauges with at least 20 years of continuous data in the US including gauge identification and location, basin morphology, climate, topography, soils, and anthropogenic disturbance factors (disturbance index, population density, and landuse). The disturbance index is a composite score for USGS gauged streams based on 8 factors for each entire basin: major dam density, change in reservoir storage from 1950 to 2006, freshwater withdrawal, artificial paths (canals, ditches, and pipelines), road density, distance to major NPDES (National Pollutant Discharge Elimination System) sites, and the fragmentation of undeveloped land. Because I had eliminated gages that had upstream impoundments, I adjusted the index to take into account only the other 6 disturbance factors by deleting major dam density and change in impoundment storage from the composite score. I then used natural breaks (Jenks 1967) to classify the score distribution into three categories: low (4-12), moderate (12-20), and high (20-27). I removed streams from the analysis that were in the high category (n=18), leaving 292 streams with low to moderate disturbance.

Mean daily and annual peak flow data for the 292 stream gauges were downloaded from the USGS Realtime Water Data for the Nation website. Hydrologic

statistics were calculated for each stream using the Hydrologic Index Tool (HIT) software available through the USGS (Hendriksen et al. 2006). Daily and peak flow gauge data were imported into the HIT software, which calculates the 171 hydrologic indices reported in Olden and Poff (2003). The indices are summaries of the entire period of record. The indices are grouped into five categories of flow: magnitude (n = 94), frequency (n = 14), duration (n = 44), timing (n = 10), and rate of change (n = 9) with each category having low, average, and high flow subcategories (Richter et al. 1996; Olden and Poff 2003). Because of the large amount of correlated variables, I reduced the dataset by evaluating correlation matrices among variables within each subcategory and removed variables with correlations of  $r > 0.75$  and  $r < -0.75$ . For each pair of correlated variables, I favored keeping variables that were in the Index of Hydrologic Alteration (Richter et al. 1997) or used in Poff (1996). If neither of these applied to the variables or both were favored, then variables were removed on the basis of order in the dataset, where variables listed later were removed. I then combined all subsections together to eliminate any other variables that were highly correlated across different subsections. The final dataset had 66 variables. I divided any variables related to magnitude by the median daily flow in order to ensure that flow groups were based upon trends in relative flow magnitude and flow variability rather than being confounded by river size. I divided all variables by their respective maximum value for all streams to normalize variables on a scale from 0 to 1. I did this to ensure that variables with greater variability did not overwhelm my analysis. All normalized variables were  $\log(x + 1)$  transformed. The dataset is freely available and can be obtained by contacting me.

I used a K-means cluster analysis to classify streams into groups of similar hydrologic properties. K-means cluster approaches require the investigator to specify a priori the number of clusters, to which streams are then assigned. Because I was uncertain of the appropriate number of flow clusters, I re-ran the cluster analysis using a series of iterations with a different number of clusters to determine the minimum number of clusters. The K-means method yields a cluster assignment for each stream but also a distance measurement of each stream to the centroid of its respective cluster (i.e. residual). I then calculated the sum-of-squares of the distances (SSD) for all streams and grouped that value against the number of clusters. The SSD will decline as the number of

clusters increases. I determined the minimal number of clusters at the point where the rate of the decline in the SSD is small. High numbers of variables ( $n = 66$ ) in relation to the sample size ( $n = 292$ ) may increase the dimensionality of the data set, which could lead to a higher number of clusters and erroneous conclusions. Thus, to ensure that the number of clusters was stable, I produced two reduced datasets followed by iterative K-means clustering procedures. I produced the first reduced dataset by conducting a principal components analysis on all 66 variables and isolating the first 15 principal components, which explained over 90% of the correlations. The second reduced dataset was formed by using forward stepwise variable selection in discriminant function analysis (DFA) to select 15 variables that explained the majority of variability in the clusters assigned with 66 variables.

Another potential problem of K-means procedures is that the cluster assignment can be sensitive to the order of samples (streams) in the dataset (SAS 2008). To attempt to determine the probability of cluster assignment for each stream, I randomized the dataset and re-ran the K-means procedure for 10 more iterations. Unlike other cluster procedures, K-means clusters do not have any spatial structure (multivariate space); thus, clusters may be labeled differently and may overlap with other clusters making it difficult to compare the results following each iteration. One way to avoid confusing labeling problems is to compare average values of variables between clusters of different analyses. I compared cluster means of 10 variables, chosen using DFA, in the baseline dataset to cluster means formed by each randomized dataset. Clusters that had the smallest difference in mean values of variables were assigned to similar clusters. Based on the result of 10 iterations, streams could then be assigned a dominant cluster and a probability of cluster assignment could be calculated.

#### *Temporal Analysis and Cluster Assignment*

To ensure that records from different time periods did not cause uncertainty in cluster assignment, I used 6 stream gages from different clusters with long term records ( $> 68$  years of record). For each stream gage, I split the records into 3-4 discrete 15-year time periods. I re-calculated all 66 hydrologic metrics using the HIT software for each 15-yr time period. I calculated the absolute difference between 15-year metrics to

metrics calculated using the entire time period (used for the classification). I then averaged all absolute differences between all 15-yr time periods and the entire time period. I compared those differences to the inter-quartile range (IQR) and entire range of each stream's respective cluster. I assumed that if the difference between metrics calculated for different time periods exceeded the IQR, then cluster assignment may be influenced.

### *Assessment of hydrologic properties*

I isolated 9 hydrologic indices supported by literature that are known to influence various life stages and occurrences of macroinvertebrates, fish, and riparian vegetation. I compared the average values and variation of the hydrologic indices among the different clusters in order to assign clusters ecologically-meaningful class names, which would be important for management. I also plotted each stream by its respective cluster on a map of the southeastern US with physiographic provinces in ARCmap 9.2. Physiographic provinces, originally mapped by Fenneman and Johnson (1946) were downloaded from the USGS website and used for mapping because of simplicity (only 5 provinces in my study area) and less ambiguity in representation. In addition, because physiographic provinces are regions that share common topography and geomorphological structure, flow classes may show spatial affiliation to provinces.

### Hydrologic Classification Tree

My goal in developing a hydrologic tree was to provide a subset of key variables responsible for the divergence of natural flow classes that can be used as an initial foundation for environmental flow management and a useful tool for classifying streams in the future without having to use a large suite of hydrologic indices. I used the rpart package in the program R to develop classification trees that can be used to classify a stream into a flow class. All 66 hydrologic variables along with their respective flow classes were imported into R. The rpart package in R uses recursive partitioning, which includes some of the same ideas developed in the CART software (Therneau et al. 2010). Trees are built in a two step procedure. The first step involves splitting the data on the initial node using the "best" variable that minimizes the risk of misclassification. This procedure continues throughout subsequent nodes until the subgroups reach a specified

minimal size or no further splits can be made (Therneau et al. 2010). Because trees can become very complex, the second step involves a pruning procedure that minimizes the number of nodes, the cost complexity factor, and the cross validation error. The cost complexity factor takes into account minimizing misclassification while also increasing the complexity of the tree. I then evaluated the cross validation versus tree size plot to determine how to prune the tree. The tree is pruned at the number of nodes that minimize the cross validation error to avoid over-fitting the data. After the trees were completed, I calculated a misclassification error to assess the accuracy to which the subset of variables could classify flow groups.

### Hierarchical Structure

One way to determine if there was a hierarchy among flow classifications created at different scales was to compare my clusters to those created for the entire US by Poff (1996). I obtained the US classification dataset through direct communication with Leroy Poff. The dataset contains 816 stream gauges, their respective flow classes, GPS locations, and the variables used to create the clusters. I isolated common gages between the two datasets using the USGS gage number. Because of differences in the number of total clusters represented and cluster sample sizes, using a statistical procedure to directly compare datasets “as is” would be uninformative. I decided to compare the datasets in two ways. First, I assumed that different classes that share similar hydrologic properties at the scale of my study may cluster together when the overall variability of the dataset increases (e.g. larger scales). Thus, I clumped similar flow classes in my study together and compared to the US classes by evaluating the percentage of streams misclassified. I also plotted my clumped classes and the US classes on separate maps to compare any similarities in geographical affiliation. The second way I wanted to compare datasets was to determine how well variables used in the US classification could predict the US flow classes relative to my classes using discriminant function analysis (linear, common covariance method). I assume that the datasets may share similar structure if variables used in US classification accurately predict my flow classes. I then plotted the first two canonical scores of streams to understand how my clusters may be embedded in the clusters created for the entire US in multivariate space. I also show the biplot rays of the



direction of variables in canonical space to show how the hierarchical relationship was governed by the hydrologic variables.

### **2.3 Results**

Altogether, 292 gages were used in my flow classification. Over 80% of my gages had records that spanned from 1969-2009 (40 yrs). After accounting for any missing data, 86% of the gages had 30 or more years of data and 60% had 50 or more years of record for the entire dataset. Less than 8% of the gages had chunks of missing data that comprised more than 30% of the entire record. Thirty-four of the 292 gages had pre-impoundment data that I used in the analysis including 24 gages with over 30 years of data.

#### *Cluster Analysis*

I found that for the original 66-variable dataset, the SSD minimized at 8 clusters (Fig. 2.1). I reduced the variables in the dataset using PCA and forward-stepwise DFA. The first 15 PCs isolated explained over 90% of the total variation in the dataset. The 15 variables isolated using stepwise DFA had a misclassification error of 7.6% (22/292 misclassified), which suggests that the variables were fairly accurate in explaining the majority of variability in clusters formed using all 66 variables. Interestingly, for both reduced datasets, the SSD minimized around 8 clusters, which was very similar to the analysis conducted with 66 variables (Fig. 2.1). Thus, I assumed that the 66 variables did not erroneously create “new” clusters but actually did a better job of describing and assigning streams to their appropriate flow group.

After randomizing the dataset and re-running the K-means procedure, I found that only 11 out of the 292 gauges (3.8% error rate) were assigned to a different class than the baseline K-means procedure. Six of the eight classes had greater than 22 streams (Fig. 2.2). Class B had only 2 streams and class D only had 1 stream. Thus, for all practical purposes, these streams in these classes could be lumped into larger classes that share similar hydrology. Results of the re-running the K-means procedure showed that seven of the eight classes had mean probability of class membership greater than 85%; however, flow class A had a mean class membership of 71%. Over 80% of all gauges

had greater than a 0.8 mean probability of class membership, suggesting that most gauges had a fairly strong affinity for their assigned flow class (Fig. 2.2).

#### Temporal Analysis and Cluster Assignment

I compared the inter-quartile range (IQR) and entire ranges of 15-year time periods of 6 streams to that of each stream's respective cluster. I found that, on average, the IQR for each cluster was 18 times the absolute difference between 15-yr metrics and metrics calculated for the entire time period. On average, less than 5 metrics per stream had differences in values that exceeded the IQR of the respective cluster. On average, the entire range for each cluster was 140 times that of the absolute difference between 15-yr metrics and metrics for the entire time period. Only one hydrologic index (RA6) in one stream had an average absolute difference that was greater than the range of its respective cluster, which was primarily due to its extremely low average value for the cluster (mean RA6 = 0.114).

#### *Assessment of hydrologic properties*

I assigned classes ecologically-relevant names by qualitatively evaluating differences in 9 key hydrologic variables (Table 2.1, Fig. 2.3). The eight flow classes differed in terms of the magnitude and variability in low flows, the frequency of high flow events, the duration of flow events, the predictability and constancy of flows, and the rate of change in flow (Fig. 2.3). Intermittent flashy (IF) streams had many zero-flow days per year, high variability, high frequency of high-flow events, low predictability, and fast rise rates. The coastal/swamp intermittent (CSI) flow class is characterized by some intermittency with 6 of the streams having many zero-flow days (mostly swamps). The majority of the class, however, has no zero-flow days, has low variability, low frequency of high-flow events, high duration of high-flow events, and very low rise rates. The Black River (BKR) near Tomahawk, NC (USGS gauge 02106500) has a very high seasonal predictability of non-flooding (not shown) or a very high proportion of each year consists of flows greater in magnitude than the 5-yr low-flow magnitude. Otherwise, the BKR is similar to CSI streams. Perennial runoff streams (PR1 and PR2) had moderate variability in daily flows, low to moderate baseflow levels, low duration of high flows, moderate predictability, and moderate rise rates.

Unpredictable perennial runoff streams (UPR) were similar to PR streams except that they had a large range in monthly flows and had low predictability. Stable high baseflow streams (SBF 1 and SBF2) had low variability in daily flows, higher baseflows and minimum flows, moderate frequency of high-flow events, high predictability, and moderate rise rates. SBF2 differs from SBF1 in that it has a lower duration of high-flow events.

### *Hydrologic Classification Tree*

I evaluated the cross validation plots to determine the appropriate size of the hydrologic and watershed classification tree. For the hydrologic tree, the cross validation minimized around 6 nodes (branches), or around a cp (cost complexity factor) of 0.05 (Fig. 2.6). The tree was pruned to that value and represented 6 of the 8 flow classes (Fig. 2.7). The BKR and UPR classes were not represented because the cost of representing 1 and 2 streams, respectively, was too high compared to the gain in variation explained. The hydrologic classification tree isolated five primary variables that accurately classified 85% of the gauges to their assigned flow class. The first four competing variables are also listed in order of accuracy under each primary splitting variable. Most classes were accurately assigned to their respective flow class (~ 80% or higher) except the SFB1 class, which only had 66% of gauges accurately assigned.

The vertical lengths of the branches are an indication of the amount of variability explained by each variables. Thus, the first two primary variables, mean September flows and minimum July flows explained a great deal of variation in the entire dataset. Lower September flows separated PR1 from the rest of the dataset and re-emerged as a primary splitting variable separating SBF1 and SBF2 stream classes. Higher minimum July flows separated SBF1 and SBF2 classes from the remainder of the dataset. Maximum November flows separated CSI streams from IF and PR2 streams, which were distinguished by daily flow variability.

### *Hierarchical Structure*

Eighty-five streams were found in both my dataset and that of the US classification. Six of my eight flow classes were represented in the dataset, excluding

UPR and BKR flow classes. Only three flow classes were represented in the US classification dataset: groundwater streams (GW), perennial runoff streams (PR), and one intermittent-runoff stream (IR). By comparing the grouping between my dataset with the US classification, I found that SBF1 and SBF2 streams tended to primarily be classified as GW streams while PR1, PR2, IF, and CSI streams tended to classify with the PR streams (Table 2.2). The one IR class grouped with the CSI streams. Initially, I combined the SBF1 and SBF2 classes into a 'GW' class and combined the PR1 and PR2 classes together into a 'PR' class while leaving the IF and CSI classes separate. After comparing the combined classes to the US classification, I found that 75% of the streams were grouped similarly, with 11% of the error coming from the CSI and IF classes. I then combined the CSI and IF classes with the PR class and compared the new classes to the US classification and found that 86% of the streams were grouped similarly. The map of my new flow classes and the US flow classes showed similar geographical affiliation of classes with similar hydrology (Fig. 2.8).

Discriminant function analysis showed that the 13 variables used in the US classification misclassified 4.76 % of streams to their actual US flow classes. The 13 variables misclassified 3.57 % of their streams to my 6 flow classes. The biplot of canonical scores showed that my 6 flow classes were embedded in multivariate space of their compatible US flow classes, suggesting hierarchical structure (Fig. 2.9). For example, SBF1 and SBF2 classes were centralized around the GW class where PR1, PR2, IF, and CSI classes were centralized around the PR class. my 6 flow classes also tended to capture a greater extent of the dimensionality of the dataset than the US classes at the scale of this analysis (85 streams). I show 7 of the 13 variables with the strongest loadings. The biplot rays showed that the SBF1 streams had a higher baseflow index but SBF2 streams had a lower low flow predictability (Fig. 2.9). IF streams diverged from PR streams based on daily variability whereas PR streams diverged from one another based on low flow and daily predictability. CSI streams diverged from PR streams based on flood duration and the number of zero-flow days.

## 2.4 Discussion

I isolated six flow classes (8 statistically) in the eight-state region that differ in the magnitude, frequency, duration, timing, and rate of change in flow and provide a reduced set of hydrologic indices to use in future classifications. Secondly, I show that datasets created at different scales can be related and may exhibit hierarchical structure. Thus, the divergence of flow classes may be relevant to the scale of management.

One of the challenges in managing for flow diversity is the inherent complexity and the variability of river systems (Poff et al. 2003; Arthington et al. 2006). My purpose is to not undermine the importance in the individuality of river systems. On the contrary, flow classifications should reduce the complexity of “managing” by providing a starting baseline from which regional flow standards and criteria can be developed. Flow classes provide ecologically-relevant management units that reduce the complexity of managing for the natural flow regime of every individual river while also protecting the key elements that make river flow distinct (Arthington et al. 2006). For example, flow classifications could provide a context for creating environmental flow standards during the dam-relicensing agreements that occur under the Federal Energy Regulation Commission (FERC). In addition, the flow classification could also be useful to statewide flow managers in providing different rules for allowable withdrawals within different regions of the state.

Flow classifications should make management more effective because they are efficient. For example, environmental flow standards can be developed rapidly and inexpensively through simple comparisons of the values of ecologically-relevant hydrologic indices from perturbed rivers to the range of values presented in a “natural” flow classes. The choice of hydrologic indices to focus attention may vary from case to case depending on the type of disturbance and depending on regional differences. Variables important to protecting the natural flow regime have been isolated, such as the 33 Indicators of Hydrologic Alteration (Richter et al. 1997) and subsets of variables particular to a given region (5 from this study, Fig. 2.7). Once key hydrologic indices have been chosen, comparisons can yield management recommendations that are based on the “natural” flow regime and not just instantaneous flow values and are relevant to particular regions. In essence, flow classes provide a mechanism to efficiently provide

holistic environmental flow recommendations. Using this framework also sets the stage for adaptive management. For example, after flow standards and criteria are developed and implemented based on “natural” flow classes, they also could be modified based on the individual needs of each river system. Therefore, treating river flows as ecosystem-scale experiments is a necessary aspect of river management (Poff et al. 2003) and a less-explored area of ecology (Palmer and Bernhardt 2006; Poff and Zimmerman 2010).

I found that eighty percent of stream gauges had probabilities of class membership greater than 0.8, which suggests that most gauges had a relatively strong affiliation to their designated flow class. Also, results of the three different clustering procedures indicated that the number of flow classes (8) was fairly stable. Considering that 80% of my gauges spanned the last four decades (i.e., substantial temporal overlap), I believe that assessing the uncertainty in using 15-year records is a worst-case scenario. Regardless, the time period assessment revealed that 15-year records of different time periods had substantially lower differences compared to the inter-quartile range and entire range of their respective clusters. Thus, my results suggest that different periods of record should not influence the results of the flow classification.

Comparisons of hydrologic variables showed a great deal of divergence among different flow classes. Thus, over-generalized, static flow rules are inappropriate for managing for the variability in flow of these ecosystems (Arthington et al. 2006). Because I observed strong divergence in flow classes, state agencies should re-consider existing approaches for recommending environmental flows in the permitting process, such as statewide 7Q10 policies for all river systems (7Q10 refers to the seven-day low flow with a 10-year recurrence interval). Management strategies for protecting environmental flows will require policy reform, which abandons static allocations to ranges of numbers that protect the inherent variability in river flows (Richter et al. 1997). Ultimately, managers and stakeholders must move away from “water-allocation” strategies and adopt “sustainable-boundary” approaches, which are similar to water quality protection in that they provide social goods and services and create a sustainable, long-lasting resource (Richter 2010).

Classifying streams with similar hydrologic properties has progressively received more attention in peer-reviewed literature. Stream classification has been conducted

across the US (Poff and Ward 1989; Poff 1996), Australia (Kennard et al. 2010b), and even at global scales (Poff et al. 2006b). Classifications have not just been limited to theoretical frameworks but have been implemented in management and flow policies. For example, with the recent advent of state-wide water use management plans, flow classifications have been developed for New Jersey, Missouri, and Oklahoma as standards for implementing flow rules (Kennen et al. 2007; Kennen et al. 2009; Turton et al. 2008). One area of needed research is to determine if rivers within similar flow classes respond to regulation similarly (Arthington et al. 2006).

#### Assessment of hydrologic properties

To provide names for flow classes, I chose a set of ecologically-relevant hydrologic variables that would readily distinguish groups and that would make intuitive sense to managers. Flow classes showed a great deal of divergence among the hydrologic variables. Stable high baseflow streams (SBF1 and SBF2) had characteristics of stable flow (low variability, high predictability and higher baseflows). However, they differed in terms of high flow duration. Intermittent –flashy streams (IF) were primarily classified based on the high number of zero-flow days (intermittency) and highly variable flows (flashy). Coastal/swamp intermittent (CSI) streams were stable and unresponsive (opposite of flashy), similar to SBF streams, but differed in that they had sustained lower flows. Of the CSI streams, swamps were the only water bodies showing intermittency. Swamps may be highly sensitive to drought conditions making them intermittent yet stable in all other aspects. Most intermittent flashy (IF) streams were located in the Piedmont and had small drainage areas, which suggests that small watersheds in combination with piedmont type soils may result in flashy, highly variable flows. The values of hydrologic variables of perennial runoff streams (PR1 & PR2) generally were moderate in comparison to the stable and highly variable extremes. For example, PR streams had higher variability and lower baseflow than stable streams yet had lower variability and higher baseflows than PF and IF streams. PR streams also had moderate predictability, constancy, and frequency of high flow events relative to the other classes. PR streams had broad geographical ranges across the five provinces. Interestingly, I found a higher density of PR streams in the northern section of the region. This suggests

that climate, evapotranspiration, and soil type may vary considerably with latitude within the same physiographic province and lead to differences in flow regimes.

#### *Hydrologic classification tree*

Providing flow rules from a large suite of hydrologic indices can be overwhelming. Thus, I wanted to provide a reduced set of hydrologic variables for managers that form two useful tools: 1) a set of variables to classify streams into one of the eight flow classes and 2) a foundation of key indices to develop environmental flow standards. The classification tree produced 5 variables that correctly classify 85 % of the gauges in this study.

Interestingly, monthly flows dominated the primary splitting variables in the hydrologic classification tree. However, in general, the tree supported the results of my qualitative assessment using hydrologic variables alone to describe flow classes. For example, stable high baseflow streams (SBF1 and SBF2) were separated from the other classes based on higher summer flows (minimum July flows). Although IF streams and PR2 streams were isolated from CSI streams primarily based on maximum November flows, rise rate (responsiveness) and variability were represented as competing variables that could separate the classes. Also, IF streams were separated from PR2 streams on the basis of variability, which makes intuitive sense. The error assessment also suggested that there was some overlap among flow classes (Fig. 2.7). For example, the SFB1 class was misclassified primarily as SFB2 streams and to some degree in the other classes, except IF streams. In general, this suggests that flow classes, in some cases, may share some attributes of other classes, which would make clear partitioning in a tree prone to error. Thus, flow classes should not be viewed by their mean values but the range of variability that they represent.

#### Hierarchical structure

The spatial scale at which flow regimes are evaluated greatly influences the resolution to which ecologically relevant differences can be isolated and then used in management. Poff and Ward (1989) and Poff (1996) conducted a flow classification of the entire US; however, only 3 flow classes were isolated in the region of interest. my flow classes formed 'sub-classes' within the US classes suggesting there was a hierarchical structure to flow variability. Furthermore, my flow classes encompassed



more of the dimensionality of the dataset suggesting that as the spatial resolution under consideration increases, the divergence of more classes may be needed to adequately represent variability at that scale.

The number of flow classes (e.g. clusters) is largely a statistical artifact and a balance between sample size and the variability of the entire dataset. In a clustering procedure, a stream that is found in the margins of a cluster boundary will obviously be more likely to join a different cluster if more streams of a similar flow regime are included in the dataset. In contrast, if the variability of the entire dataset is substantially increased (e.g. expanding the spatial scale across multiple regions), then classes at smaller scales may be lost. Thus, classifying flow regimes at various spatial scales can be useful in forming hierarchical classification systems (Poff et al. 2006b). The hierarchical classes may be advantageous in that managers can then prioritize management strategies based on the strengths of class divergence and the specific application. For example, flow classes that are divergent over large spatial scales should never be managed similarly whereas flow classes that are divergent at finer scales may or may not be managed similarly depending on the whether management is conducted over inter or intra-regional scales.

## **2.5 Conclusion**

I classified 292 streams of an eight-state region of the southeast into six distinct flow classes that represent unique flow regimes and differ in terms of the magnitude, variability, frequency, duration, and rate of change in flow events. My classification may be biased in that it is based only on unregulated streams, which generally, have smaller drainage areas and make up a small percentage of all rivers in my study area.

Nonetheless, I believe that these flow classes represent targets of natural flow variability that can be used as management units to develop environmental flow standards and criteria, providing guidelines for relicensing agreements and withdrawal permitting, while also setting the stage for adaptive management of flow restoration. The classes could provide a less-complex method of management since FERC relicensing agreements and withdrawal permits usually occur on a stream-by-stream basis. For example, rules could be developed for regions within states so that standards for a particular impaired

stream could be based upon geographical location and then adapted for specific stream needs.

I also provide a subset of hydrologic variables that can be used to classify streams in this region while also providing a reduced set of relevant indices that are building blocks in developing flow standards. I show that classifications conducted at different spatial resolutions display hierarchical structure and if underlying variables are available for analysis, hierarchical analyses can be informative and useful for management.

Table 2.1. Class ID's, class names, mean drainage area, and drainage area standard deviation (Dev.) for flow classes found in this study and those of Poff (1996) found in my study region.

| Class ID  | n  | Class Name                      | Drainage Area (km <sup>2</sup> ) | Standard Dev. |
|---|----|---------------------------------|----------------------------------|---------------|
| BKR   | 1  | Black River near Tomahawk, NC   | 1751                             | --            |
| CSI   | 22 | Coastal, Swamp and Intermittent | 2307                             | 2334          |
| IF  | 26 | Intermittent Flashy             | 228                              | 285           |
| PR1   | 84 | Perennial Runoff 1              | 2547                             | 5081          |
| PR2   | 72 | Perennial Runoff 2              | 673                              | 906           |
| SBF1  | 33 | Stable High Baseflow 1          | 1500                             | 3010          |
| SBF2  | 52 | Stable High Baseflow 2          | 1701                             | 3755          |
| UPR   | 2  | Unpredictable Perennial Runoff  | 70                               | 14            |
| US Flow Classes (Poff 1996) Found in Current Study Region |    |                                 |                                  |               |
| GW  | 22 | Groundwater                     | 775                              | 968           |
| IR  | 1  | Intermittent Runoff             | 751                              | --            |
| PR  | 62 | Perennial Runoff                | 885                              | 1040          |

Table 2.2. Comparison of the proportion of streams within each of the flow classes created in this study compared to classes created for the entire US by Poff (1996). n refers to number of streams within each flow class in this study. Shaded boxes refer to the dominant US class in which flow classes, created for the current study, were found. For class codes, see Table 2.1.

| US Classes | Flow Classes (This Study) |      |      |      |      |      |
|------------|---------------------------|------|------|------|------|------|
|            | CSI                       | IF   | PR1  | PR2  | SBF1 | SBF2 |
| n          | 9                         | 3    | 21   | 25   | 12   | 15   |
| GW         | 0.22                      | --   | --   | 0.04 | 0.66 | 0.73 |
| IR         | 0.11                      | --   | --   | --   | --   | --   |
| PR         | 0.66                      | 1.00 | 1.00 | 0.96 | 0.33 | 0.26 |

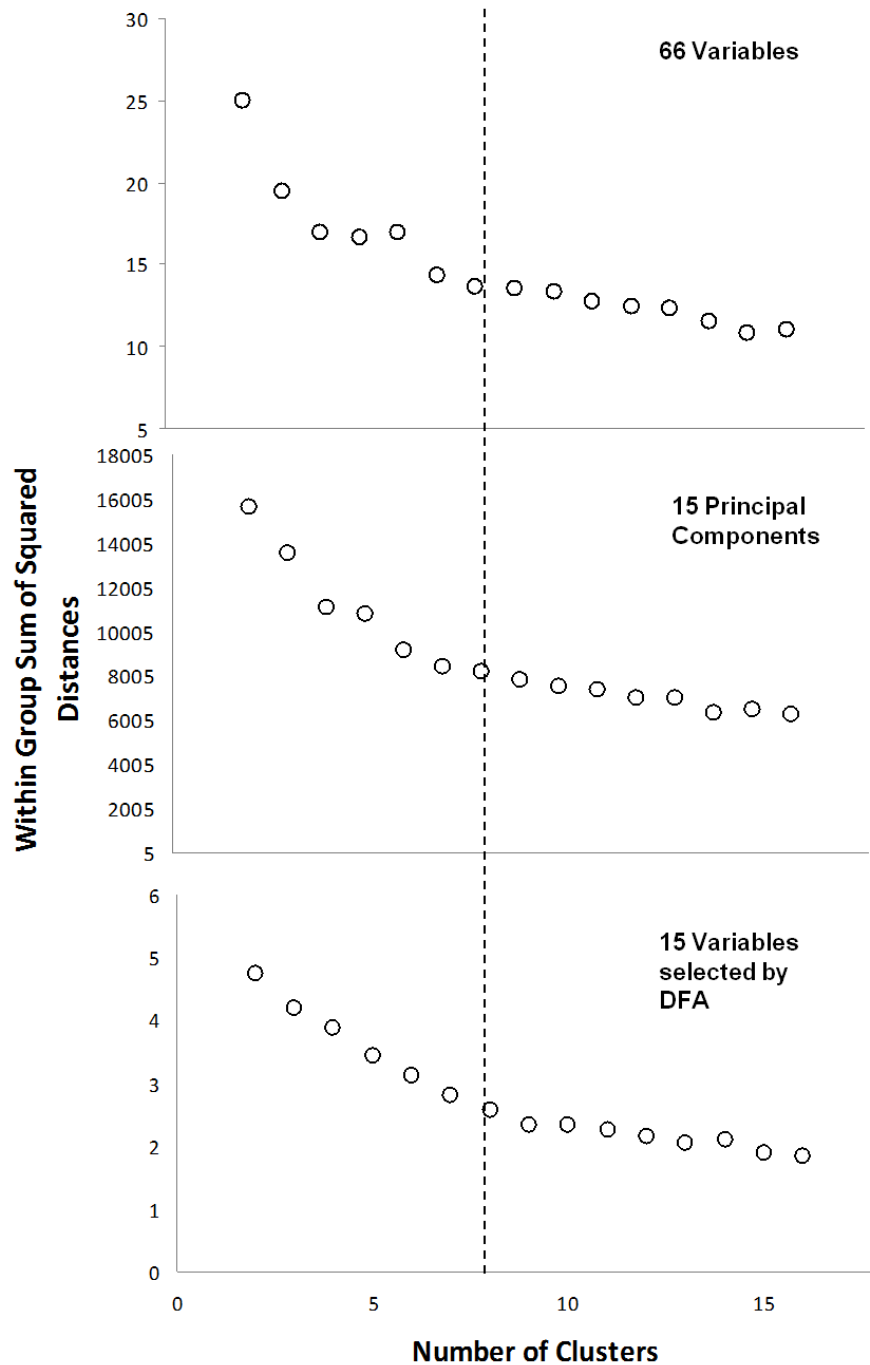


Figure 2.1 Comparisons of the sum-of-squared distances within groups relative to the number of clusters following iterative K-means clustering procedures to determine the appropriate number of clusters and whether variable number increased the dimensionality of the data set. Cluster analysis was conducted for all 66 variables, 15 principal components, and 15 variables selected using forward-stepwise procedure in discriminant function analysis.

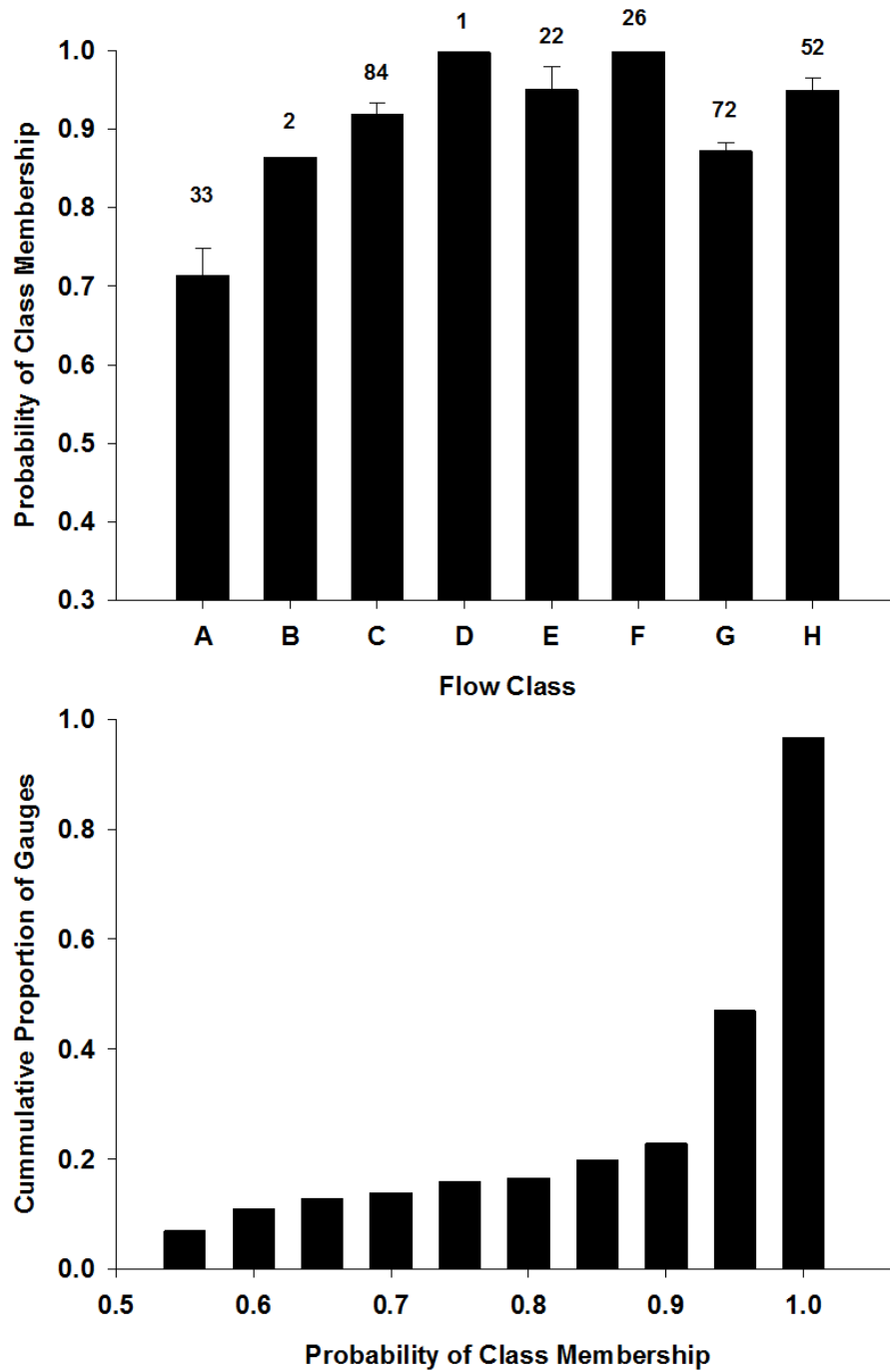


Figure 2.2 The mean probability of class membership for each flow class (cluster) and the cumulative proportion of gauges under various class membership probabilities. Numbers indicate the sample size in each flow class. Letters for each flow class are assigned ecologically relevant names, as indicated in Fig. 2.3.

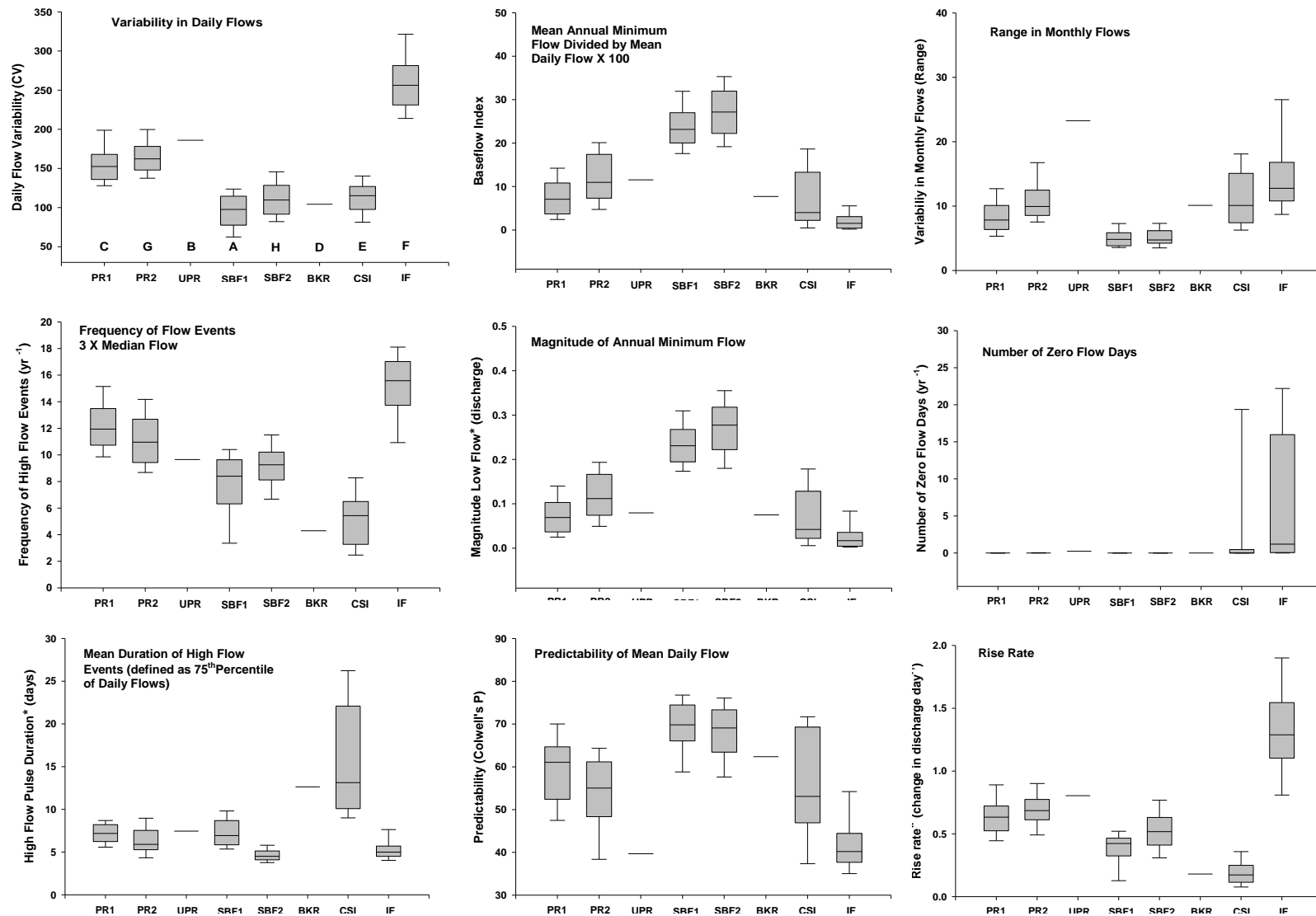


Figure 2.3 Comparisons of 9 ecologically-relevant variables among different flow classes. Letters in the first box and whisker plot are given in order to compare letters in Figure 2.2 with the flow-class names. \* indicates that variable was standardized by dividing by median daily flow. For class codes, see Table 2.1

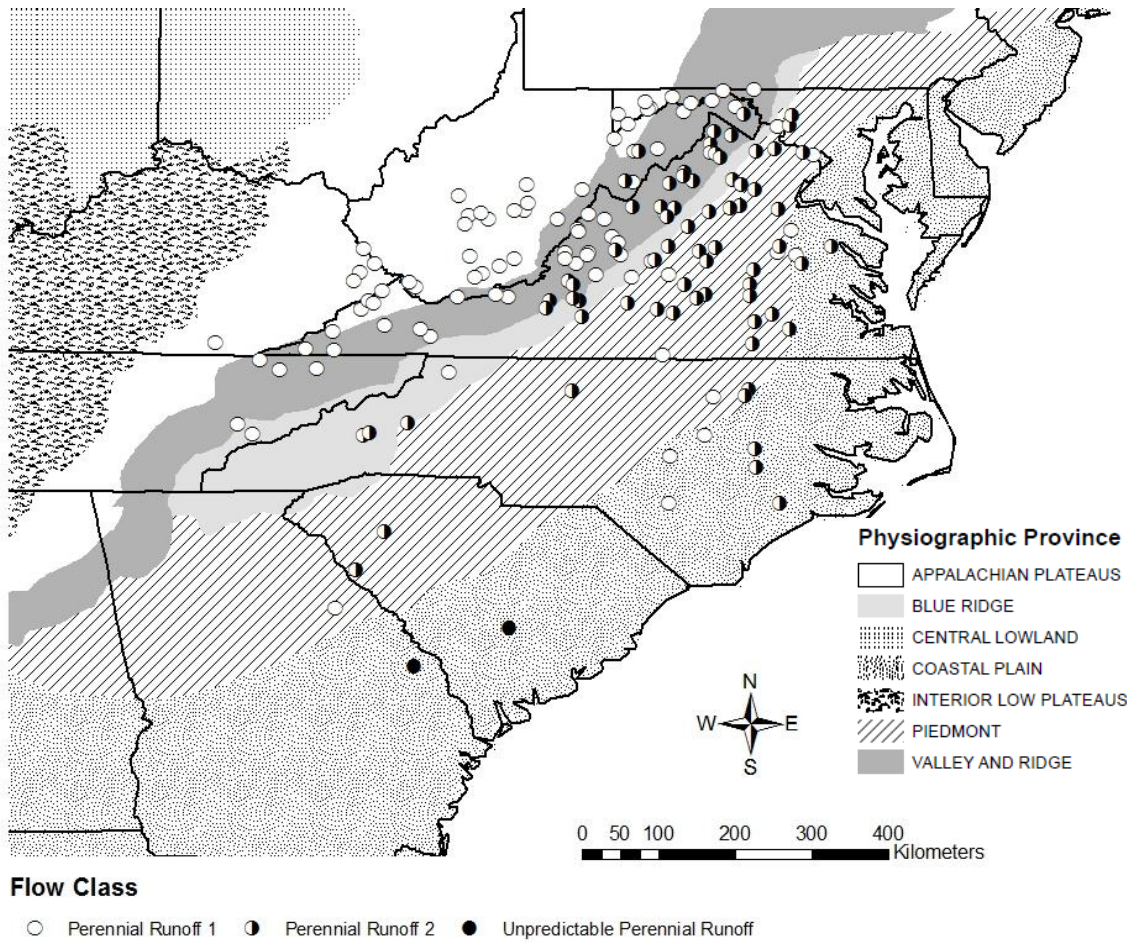


Figure 2.4 Geographic distribution of three of the eight flow classes across physiographic provinces in the eight-state region. Physiographic provinces created by Fenneman and Johnson (1946).

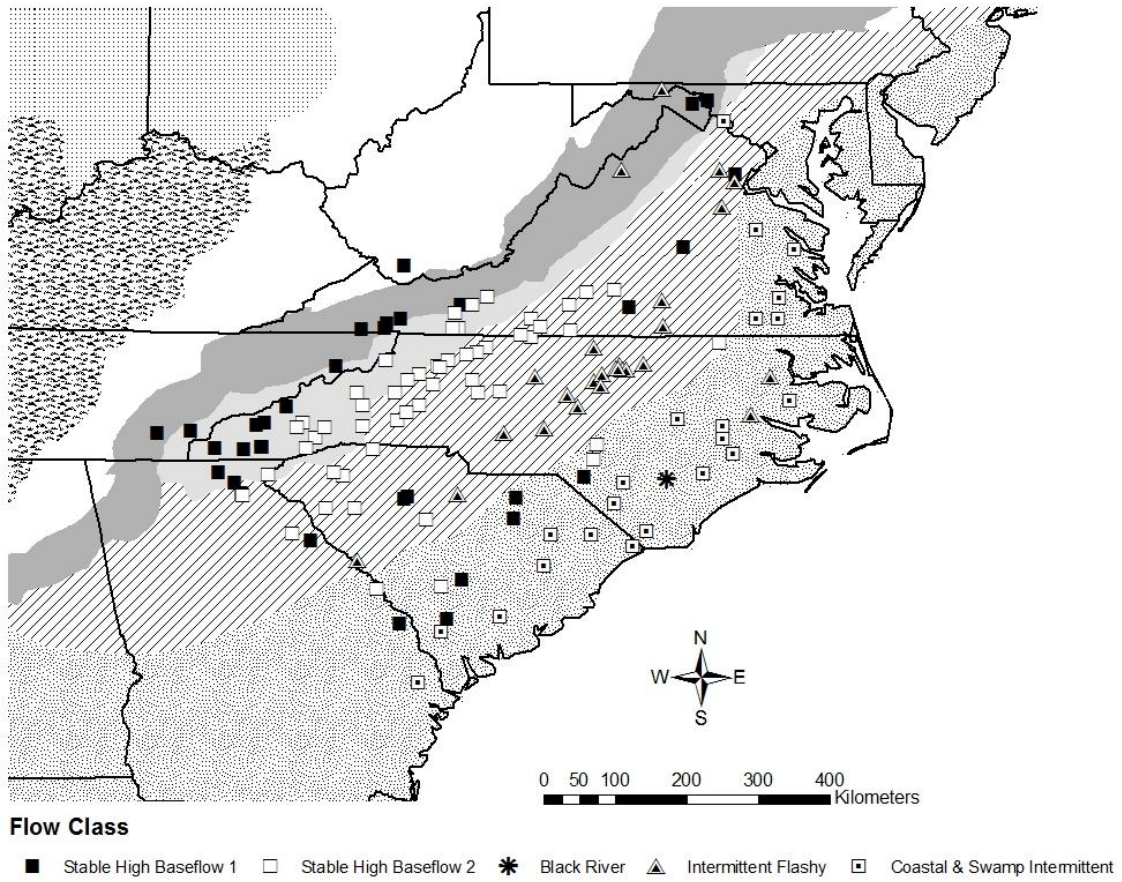


Figure 2.5 Geographic distribution of five of the eight flow classes across different physiographic provinces in the eight state region. Physiographic provinces created by Fenneman and Johnson (1946)



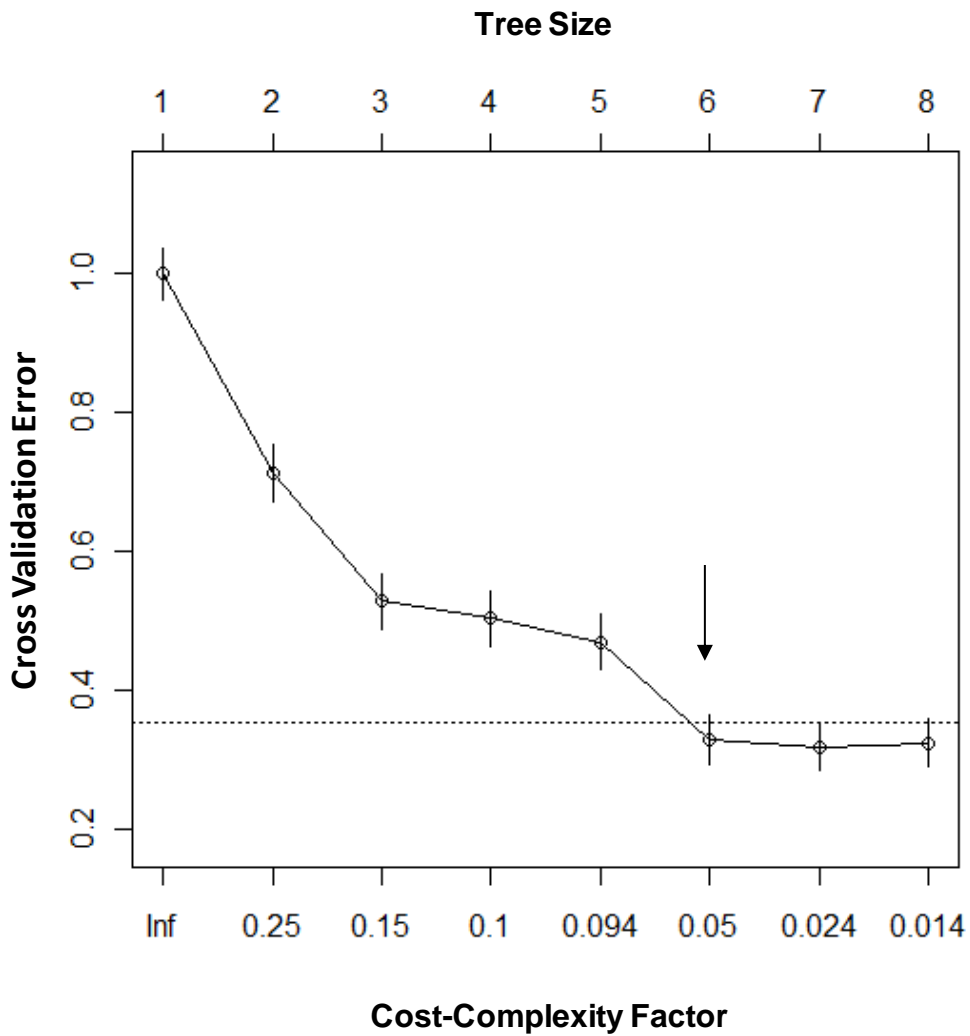
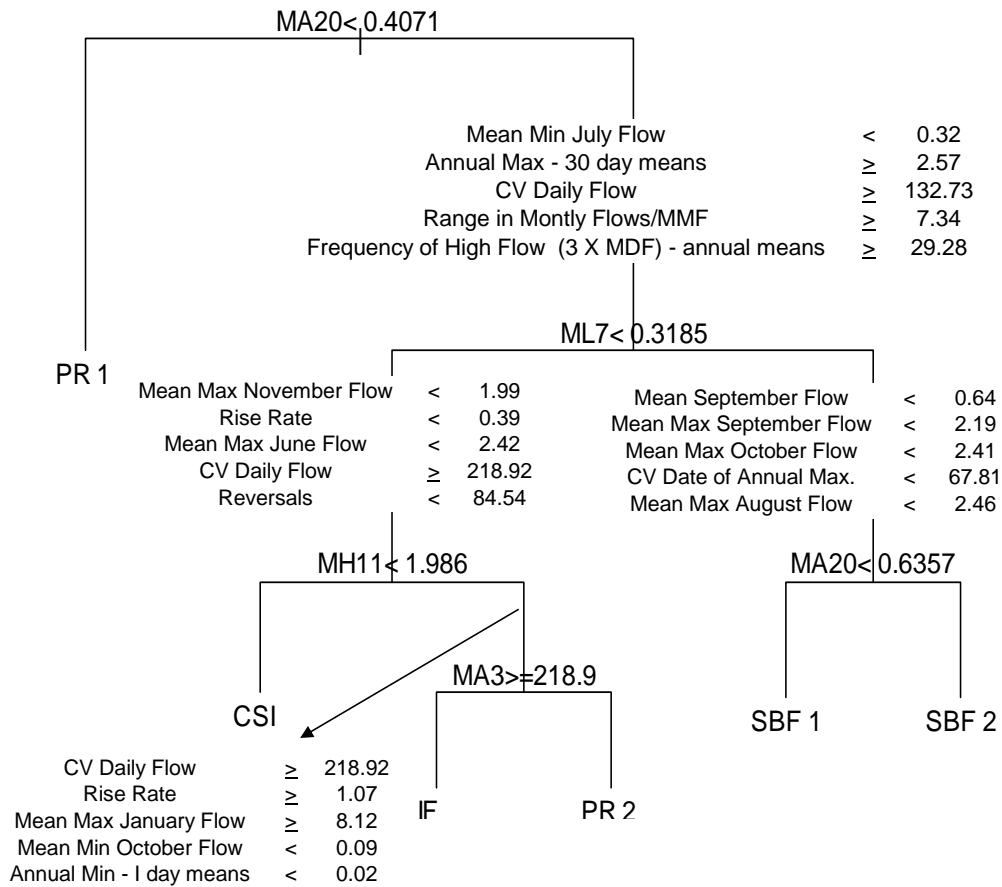


Figure 2.6 Cost-complexity plot for the hydrologic classification tree comparing the cross validation error to the tree size (number of nodes) in order to determine where the tree should be pruned. Trees are generally pruned at the cost-complexity factor where the minimum number of nodes also minimize the cross validation error (indicated by arrow). “Inf” refers to infinity.

Mean September Flow < 0.41  
 Mean Max September Flow < 2.63  
 Mean Min July Flow < 0.32  
 Annual Max - 30 day means  $\geq$  2.57  
 CV Date of Annual Max. < 63.88



|                 |       | Actual Classification |      |      |      |      |       |       |      |
|-----------------|-------|-----------------------|------|------|------|------|-------|-------|------|
|                 |       | BKR                   | CSI  | IF   | PR 1 | PR 2 | SBF 1 | SBF 2 | UPR  |
|                 | n     | 1                     | 22   | 24   | 84   | 72   | 33    | 52    | 2    |
| Tree Assignment | CSI   | 1.00                  | 0.95 | --   | 0.02 | --   | 0.06  | --    | 0.50 |
|                 | IF    | --                    | --   | 0.79 | --   | 0.01 | --    | 0.02  | --   |
|                 | PR 1  | --                    | --   | 0.13 | 0.86 | 0.01 | 0.09  | --    | 0.50 |
|                 | PR 2  | --                    | --   | 0.08 | 0.12 | 0.92 | 0.03  | 0.06  | --   |
|                 | SBF 1 | --                    | --   | --   | --   | 0.03 | 0.66  | --    | --   |
|                 | SBF 2 | --                    | 0.05 | --   | --   | 0.03 | 0.15  | 0.92  | --   |

Figure 2.7 Classification tree using 5 hydrologic metrics as primary splitting variables along with the 4 corresponding competing variables to classify 6 of the eight flow classes. The left branch meets the conditions of the equation on each node. The matrix below the tree shows the proportion of gauges in the actual flow class (columns) classified to each flow class using the tree (rows). The proportion of each actual flow class accurately assigned by the tree is shown in gray boxes. For class codes, see Table 2.1.

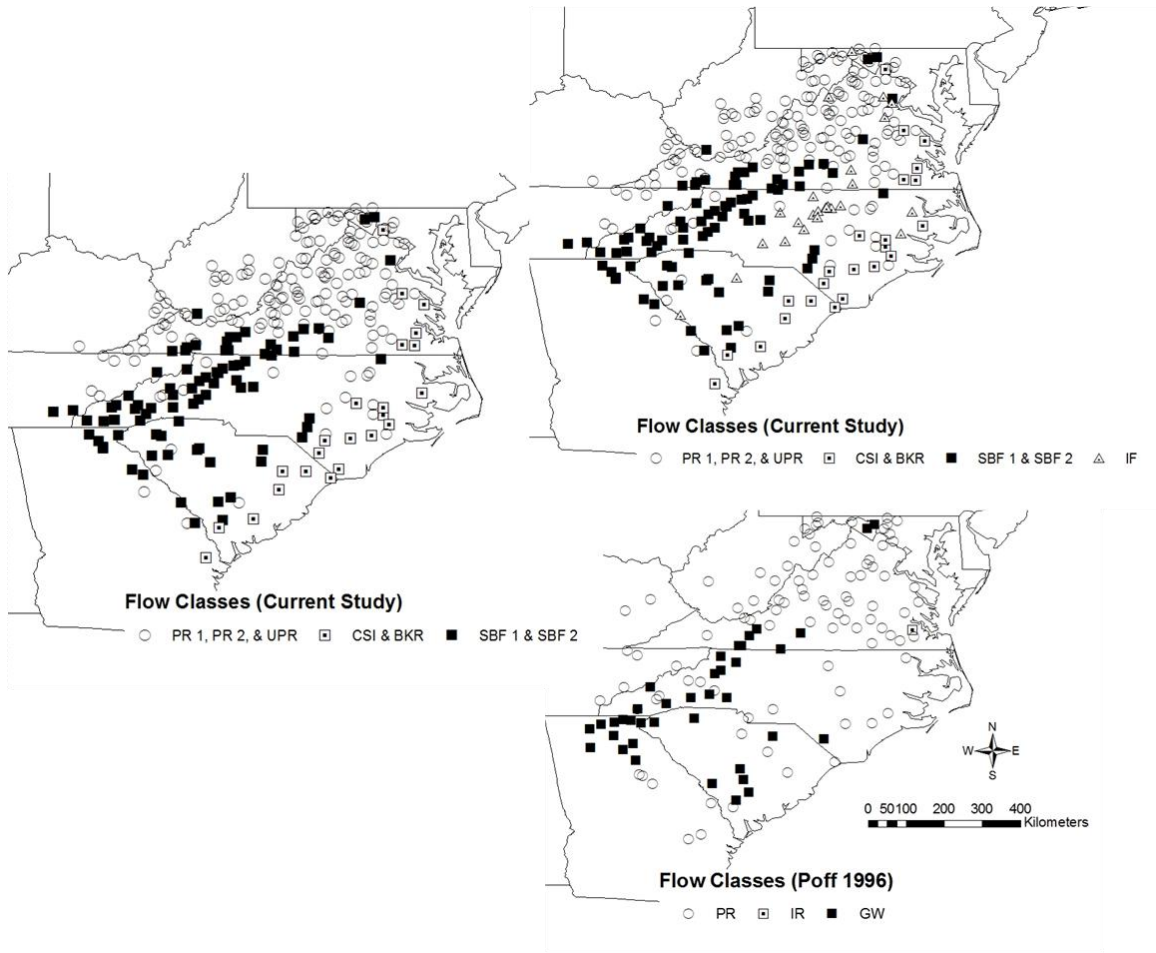


Figure 2.8 Geographic distribution of my combined flow classes and the US flow classes (Poff 1996) found within the eight state region. For class codes see Table 2.1.

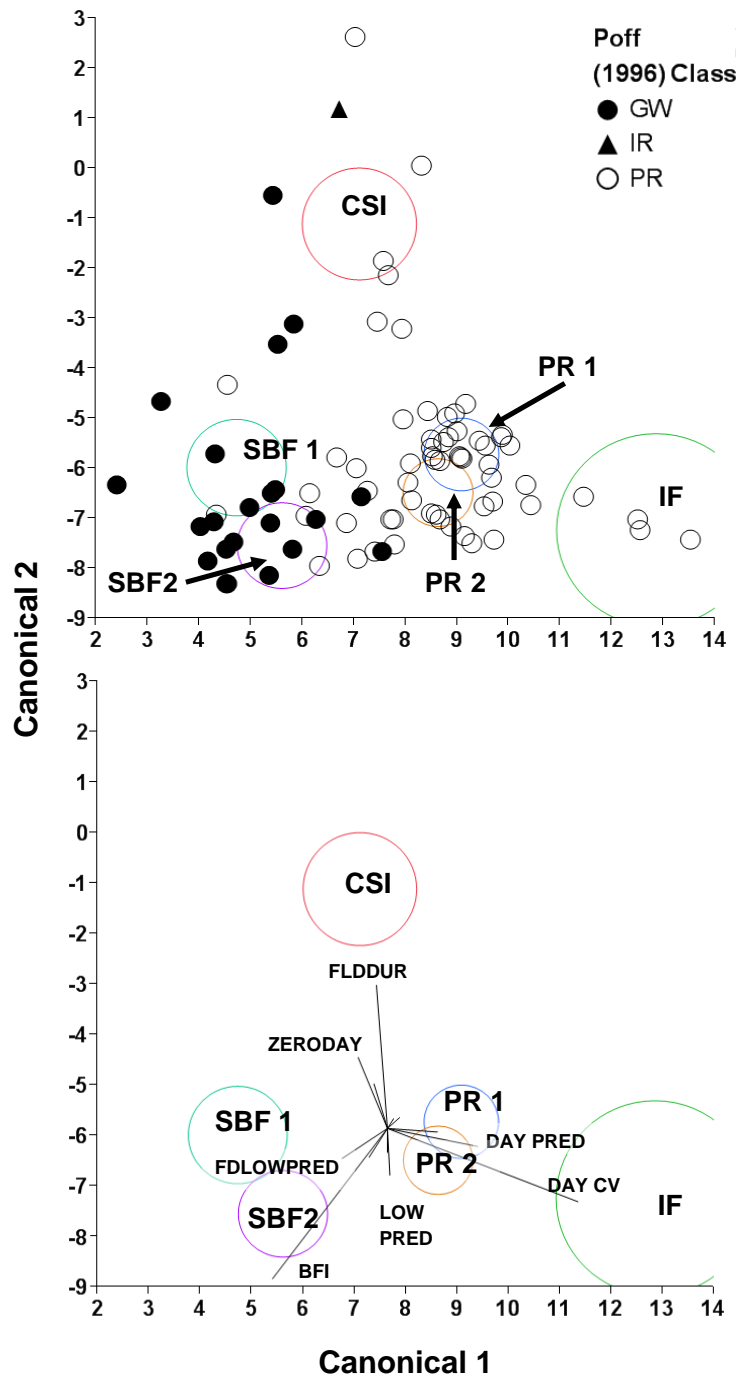


Figure 2.9 Canonical plot of the streams found in both my study and that of the US flow classification (Poff 1996). Plot is based on the 13 variables used in Poff (1996) to discriminate amongst the flow classes created in this study. Ellipses represent flow classes from this study and were created using the linear, common covariance discriminant method; they show the 95 % confidence region. Point markers represent US flow classes. Biplot rays show 7 variables that explain the most variability in flow classes (see Poff, 1996 for variable names). For class codes see Table 2.1.

### **3. Regional Frameworks Applied to Hydrology: Can Landscape-based Frameworks Capture the Hydrologic Variability?<sup>2</sup>**

#### **Abstract**

Regional frameworks have been used extensively in recent years to aid in broad-scale management. Widely-used landscape-based regional frameworks, such as Hydrologic Landscape Regions (HLRs) and physiographic provinces, may provide predictive tools of hydrologic variability. However, hydrologic-based regional frameworks, created using only streamflow data, are also available and have been created at various scales; thus, relating frameworks that share a common purpose can be informative. Secondly, identifying how the relative importance of variables change in governing stream flow with respect to scale can also be informative. The purpose of this study was to determine whether landscape-based frameworks could explain variation in stream flow classifications and in the hydrologic variables used in their creation. I also evaluated how climate and watershed-based variables govern the divergence of different flow classifications at two different scales. HLRs and physiographic provinces poorly predicted flow class affiliation within my study and for the entire US, although physiographic provinces explained more variability. I also found that HLRs explained very little variation in individual hydrologic parameters. Using variables summarized at the watershed scale, I found that climate played a larger role in influencing hydrology whereas at smaller scales, soils governed variation in hydrology. My results suggest that predictor variables, developed at the watershed scale, may be the most appropriate at explaining hydrology and that the variables used in creating regional landscape based frameworks may be more useful than the frameworks themselves. In addition, managers should be careful when using landscape-based regional classifications for stream management since the scale of their construction may be too broad to capture differences in flow variability.

<sup>1</sup> Appears as:

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### 3.1 Introduction

The development of regional frameworks for conservation, prioritization, or broad-scale management has increased substantially in recent years (McMahon et al. 2001; Snelder et al. 2004; Wollock et al. 2004; Sowa et al. 2007). Regional frameworks inform management by relating spatial patterns to ecological and physical variables at the landscape scale. However, their utility rests upon the ability to provide spatially explicit data, predictive tools, templates for categorization, a way of answering pressing management questions, and relating existing datasets to geographical information. Thus, as new regional frameworks are created, it may be important to understand how these datasets are related to other frameworks to inform management. This is especially true for hydrological frameworks since water is becoming a scarce resource (Sun et al. 2008; Vörösmarty et al. 2010).

Hydrology varies extensively across large scales and can change substantially depending on how humans alter the landscape (Poff et al. 1997; Poff et al. 2006a). Because of this obvious relationship, it makes intuitive sense that regional frameworks might be developed to relate hydrology with the landscape. Hydrologic Landscape Regions (HLRs) were developed by the USGS as a part of the National Water-Quality Assessment (NAWQA) Program in order to provide a regional framework for stratifying water quality study sites based on different hydrologic contexts (Wollock et al. 2004). HLRs were developed using variables that control hydrology (Wollock et al. 2004); thus, they have a potential to predict flow variability in streams. HLRs, or the specific variables that compose them, have been used to predict or model chemical concentrations in streams (Poor et al. 2008; Hoos and McMahon 2009), baseflow levels (Santhi et al. 2008), and fish assemblages (Frimpong and Angermeier 2010a). Another regional framework, physiographic provinces, was originally created by Fenneman and Johnson (1946) as regions that share common topography and geomorphological structure and history. Physiographic provinces have been used as a regional framework for regional channel morphology relationships (Johnson and Fecko 2008), the effects of different variables on hydrology (Mohamoud 2008; Morris et al. 2009), watershed classifications (Wardrop et al. 2005), and fish assemblages (Angermeier and Winston 1998). Thus,

similar to HLRs, physiographic provinces may have the ability to predict hydrologic variability.

Landscape-based approaches used to predict hydrology have primarily focused on flow-routing tools or complex hill storage models. Flow routing tools and hydrology models can accurately predict stream flow discharge across time (Easton et al. 2007; Gong et al. 2009; Matonse and Kroll 2009), with increasing accuracy as model complexity increases (Butts et al. 2004). However, flow routing tools are generally limited to small scales and individual basins (Arora et al. 2001; Shaw et al. 2005; Gong et al. 2009). Although landscape-based approaches are widely used as predictive tools of hydrology, datasets created to evaluate spatial variation in hydrology based on stream discharge alone (e.g. flow classifications) have been conducted for individual states (Kennen et al. 2007; Kennen et al. 2009; Turton et al. 2008), specific regions (McManamay et al. 2011), the entire US (Poff and Ward 1989; Poff 1996), Australia (Kennard et al. 2010b), and globally (Poff et al. 2006b). An important clarification is that spatial flow classifications, in comparison to flow routing tools, generally do not encompass temporal variability in streamflows. Nonetheless, within a spatial context, stream flow classifications may be useful in relating hydrologic variability to landscape-based regional frameworks over large scales. It also may be informative to understand how regional frameworks and landscape characteristics (including climate) influence the spatial variation in hydrology with respect to scale.

The overall purpose of this study was to determine if landscape-based hydrologic frameworks could explain variation in stream flow hydrology across different scales. I find this very important since, in the absence of sufficient hydrologic information, management strategies have and will continue to require landscape-based models to predict stream flow variability (Carlisle et al. 2010a). Specifically, I wanted to understand how flow classifications, conducted at different scales, may relate to existing and widely-used regional frameworks (HLR and physiographic provinces). Poff (1996) created 10 flow classes for 806 streams across the entire US based on hydrologic variables. In Chapter 2, I classified 292 streams into 8 flow classes in a sub-region of the southeastern US, which formed the basis for a hierarchical classification system (rationale for focus on southeastern US given in Chapter 2). Therefore, I wanted to determine how well HLRs

and physiographic provinces predict the affiliation of these two spatially-explicit flow classifications. It is important to clarify that I do not assess the relative importance of spatial autocorrelation in the predictive capability of HLRs or physiographic provinces. Rather, my sole purpose is to compare how well these two frameworks, which I expect to be sensitive to hydrology, predict natural flow classes. Predicting natural flow class membership, rather than the values of individual hydrologic indices, may help to consolidate the variability since flow classes represent a range of variability. However, using flow classifications to represent natural flow variability decreases the resolution of analyses and may lead to losses in information of specific important hydrologic indices. Because there is some bias in using one classification system to “predict” another classification system, I wanted to show how well HLRs can predict specific hydrologic variables that were used to create the US flow classes. Lastly, I wanted to determine what specific climate or watershed variables (topography, soils, etc) best explain flow variability at the sub-regional (8-state region of the southeastern US) and the continental US spatial scales.

## **3.2 Methods**

### *Regional Frameworks*

One of my goals was to compare the performance of existing regional frameworks in explaining the variation in the regional affiliation of flow classes at two scales. HLRs are small watersheds (approx. 200 km<sup>2</sup> each) that were categorized into 1 of 20 different classes of regions that differ in hydrologic conditions across the entire conterminous US. The development of HLRs were based on factors that govern the hydrologic cycle (precipitation, evapotranspiration, infiltration, groundwater flow, and overland flow) (Wollock et al. 2004). Physiographic provinces, on the other hand, were originally created to map regions of common geographical structure and topography (Fenneman and Johnson 1946), which may also explain differences in hydrology. I mapped the US flow classes and the sub-regional flow classes on HLRs and physiographic provinces in order to visually evaluate whether flow classes were geographically affiliated with different regional frameworks.



I also wanted to quantitatively determine the ability of regional frameworks to predict flow class affiliation. Flow classes were developed using discharge records for 292 US Geological Survey (USGS) stream flow gages. The dominant hydrologic landscape region within the basin of 273 of the 292 gages was available in the GAGES dataset created by Falcone et al. (2010). The dataset consists of 375 variables for 6785 USGS stream gauges across the US including basin morphology, climate, topography, soils, and anthropogenic disturbance factors (disturbance index, population density, and landuse). Each basin's dominant physiographic region, however, was not included in the dataset. Using the 'select by location' tool in ARCmap 9.2, I recorded the physiographic region where each gauge was located for all streams. Using the province represented by each site location rather than the basin-wide average may induce some bias in the analysis, especially in gauges that are very close to physiographic boundaries. However, my purpose is simply to evaluate, on a coarse level, the overall ability of these regional frameworks to predict flow class membership. I used  $r^2$  values calculated from -log-likelihood tests to compare the ability of HLRs and physiographic provinces to explain the variation in the grouping of flow classes. The  $r^2$  in -log-likelihood tests is calculated as the proportion of variation explained by the model relative to total uncertainty where:

$$\begin{aligned}
 & -\log \text{ likelihood Model} = \sum_{ij} n_{ij} \ln\left(\frac{p_{ij}}{p_j}\right) \\
 \text{and} \quad & -\log \text{ likelihood Total} = -\sum_{ij} n_{ij} \ln(p_j) \\
 \text{where} \quad & p_{ij} = \frac{n_{ij}}{N} \quad \text{and} \quad p_j = \frac{N_j}{N}
 \end{aligned}$$

and  $n_{ij}$  is the count for  $i^{th}$  factor and  $j^{th}$  response level,  $N_j$  is the total of the  $j$ th response level, and  $N$  is the total sample size (SAS 2008). Assessing the capability of regional frameworks to predict flow class grouping is biased because of a different number of flow classes and regional units. Therefore, to assess whether the number of flow classes made a difference, I used the US classification variables (13) to re-run a cluster procedure in order to create the same number of classes as regional units for each framework: HLRs ( $n=19$ ) and physiographic provinces ( $n=22$ ). I then re-ran the statistical tests to determine if there were increases in the predictive ability of the frameworks. I also compare the average proportion of gages in all classes that were

found within their dominant region since that value may be more comparable to the accuracy of classification trees (discussed later).

#### *Ability of HLRs to Predict Hydrologic Variables*

I hypothesized that regional frameworks, such as HLRs, may be more useful in explaining the variability of hydrologic variables that make up flow classes rather than the flow classes themselves. Secondly, there may be a great deal of bias when using one classification system to predict another, especially considering that frameworks may be created with totally different underlying variation. Thus, I wanted to determine how much variation HLRs explained in the individual hydrologic variables that were used to form the US flow classes. I conducted one-way analysis of variance tests (ANOVA) for the hydrologic variables among different HLRs. I also conducted ANOVA tests for the 15 variables among US flow classes to compare the amount of variability explained in hydrologic variables for each regional framework. I did not conduct this analysis for physiographic provinces because the dominant province was not summarized for each gage and the data were not readily available.

#### *Variables that Govern Flow at Different Scales*

Flow classifications conducted at different spatial scales may be governed by very different factors. It may be very informative to understand the relative importance of specific variables in predicting hydrology, which is not possible when using regional classes as predictors. Therefore, I wanted to determine what physical and climate variables at the watershed scale explained flow variability within and across regions. Watershed and climate variables for USGS gauges were also downloaded from the GAGES dataset (Falcone et al. 2010). I was primarily interested in how natural climate and watershed variables (climate, basin morphology, topography, and soils) could be used to classify relatively undisturbed flow classes; thus, anthropogenic disturbance variables were removed from the analysis. I also removed any categorical variables from the dataset because classification tree decisions are based upon only 2 outcomes. Many variables, primarily climate, were represented by a value for the entire basin or for the site where each gauge was located. I removed variables that were site-specific (at the

gauge location) assuming that flow dynamics are governed by variables that account for the entire basin. The dataset was reduced to 83 variables and each gauge was joined to its respective flow class. I joined my flow classes and the US classes to 83 watershed/climate variables forming two separate datasets. The joined dataset with my flow classes only had 273 streams (rather than 292 in the original) and the joined dataset with the US flow classes had 787 streams (rather than the 816 total). Missing streams were excluded primarily because they did not have 20 years of complete-year flow records from 1950-2007 or they watersheds that could not be accurately delineated (Falcone et al. 2010).

I used the rpart package in the program R to develop classification trees that can be used to classify a stream into a flow class using climate or watershed variables. The rpart package in R uses recursive partitioning, which includes some of the same ideas developed in the CART software (Therneau et al. 2010). Trees are built in a two step procedure. The first step involves splitting the data on the initial node using the “best” variable that minimizes the risk of misclassification. This procedure continues throughout subsequent nodes until the subgroups reach a specified minimal size or no further splits can be made (Therneau et al. 2010). Because trees can become very complex, the second step involves a pruning procedure that minimizes the number of nodes, the cost complexity factor, and the cross validation error. The cost complexity factor takes into account minimizing misclassification while also increasing the complexity of the tree. I then evaluated the cross validation versus tree size plot to determine how to prune the tree. The tree is pruned at the number of nodes that minimize the cross validation error to avoid over-fitting the data. After the trees were completed, I calculated a misclassification error to assess the accuracy to which the subset of variables could classify flow groups.

### **3.3 Results**

#### *Regional Frameworks*

Patterns emerged in the spatial grouping of flow classes that suggested that flow classes were regionally affiliated; however individual classes were found in multiple

HLRs and physiographic provinces (Fig. 3.1-3.2). Twelve HLRs were represented in the southeastern sub-region compared to only five physiographic provinces (Tables 3.1 and 3.2). Perennial run-off 1 and 2 classes (PR1 and PR2) were represented in over 9 of the HLRs. PR1 and stable high baseflow 1 classes (SBF1) were represented in all 5 physiographic provinces. Excluding the unpredictable perennial run-off and Black River flow classes, the average percentage of gauges of each class found in their dominant region was 35% for HLRs and 62% for physiographic regions in the sub-regional area (Table 3.1 and 3.2).

Nineteen HLRs and 22 physiographic provinces were represented across the US where flow classes were present (Table 3.3 and 3.4). For the US classification, perennial runoff streams (PR) were found in 18 HLRs and snow & rain and intermittent run-off streams were found in 14 HLRs. PR streams were found in 15 of the 22 provinces whereas groundwater and snow & rain streams were found across 13 provinces. On average, for the US flow classes, the percent of gauges in each class found in a dominant region was 31% for HLRs and 41% for provinces (Table 3.3 and 3.4).

For the sub-regional flow classes, chi-squared analysis revealed that flow-class grouping was not statistically independent from HLRs and physiographic provinces ( $X^2 = 119.2$ ,  $df = 55$ ,  $p < 0.0001$  and  $X^2 = 297.8$ ,  $df = 24$ ,  $p < 0.0001$ ). HLRs poorly predicted flow-class grouping ( $r^2 = 0.13$ ,  $-\log\text{likelihood} = 59.61$ ). Similarly, physiographic provinces poorly predicted flow-class grouping although its explanatory power was stronger than HLRs ( $r^2 = 0.30$ ,  $-\log\text{likelihood} = 148.9$ ). Flow class grouping for the US were also not statistically independent from HLRs and physiographic provinces ( $X^2 = 635.7$ ,  $df = 162$ ,  $p < 0.0001$  and  $X^2 = 907.28$ ,  $df = 189$ ,  $p < 0.0001$ ). HLRs explained more variation in flow class grouping across the US than in the Southeastern sub-region, however the relationship was still very weak ( $r^2 = 0.23$ ,  $-\log\text{likelihood} = 317.8$ ). Again, provinces explained slightly more variation in flow classes across the US and explained more variation in flow classes than HLRs ( $r^2 = 0.33$ ,  $-\log\text{likelihood} = 317.8$ ). Because different numbers of regions and classes may bias the analysis, I re-ran cluster analyses with the variables used in the US flow classification and specified the same number of clusters as regional units (19 HLRs and 22 provinces). Increasing the number of clusters did not substantially increase the predictive ability of the HLRs ( $r^2 = 0.25$ ,  $-\log\text{likelihood}$

= 491.7, df=324) or the physiographic provinces ( $r^2=0.34$ ,  $-\log\text{likelihood} = 664.3$ ,  $df=441$ ). Although not large, the percentage of gages in each class affiliated with a dominant region did increase and was 43% for HLRs and 58% for provinces.

#### *Ability of HLRs to Predict Hydrologic Variables*

Hydrologic Landscape Regions explained 7 to 39% of the variation in the hydrologic variables whereas US flow classes explained 9 to 87% of the variation in hydrologic variables using 1-way ANOVAs (Table 3.5). HLRs explained more variation than the US flow classes for only 1 variable, mean annual runoff (39% compared to 29%). For HLRs, only 2 variables explained more than 30% of the variation whereas for US flow classes 6 of the variables explained more than 60% of the variation.

#### *Variables that Govern Flow at Different Scales*

For the sub-regional flow classes, the watershed cross-validation plot minimized at 7 branches, with a  $cp = 0.028$ ; however, this caused some overfitting since there was only 6 classes (Fig. 3.3). Thus, I pruned the tree at 6 branches, with a  $cp = 0.0525$ . Five primary splitting variables, along with their corresponding competing variables, were isolated that accurately assigned 74% of the streams to their actual classes (Fig. 3.4). Soil and infiltration variables explained the majority of variation in the model, along with some variation explained by precipitation. PR1 streams were separated from the other streams primarily based on lower amounts of finer-sized soils and having shallower soils. PR2 streams were separated on the basis of northern latitude. Stable high baseflow streams were separated from coastal swamp & intermittent (CSI) and intermittent flashy (IF) streams by the subsurface flow contact time index, which estimates the days infiltrated water resides in the saturated zone before being discharged into the stream and is calculated using topography and soil properties (Falcone et al. 2010). CSI and IF streams were separated from one another on the basis of soil size (permeability). SBF1 differed from SBF2 streams in terms of higher soil bulk densities, soil components, and precipitation seasonality.

For the US flow classes, the watershed cross-validation plot minimized around a  $cp=0.025$ , or 6 branches (Fig. 3.3). Because 6 branches would have excluded 4 of the US

flow classes, I pruned the tree to a  $cp=0.015$ , which I felt was a compromise between overfitting the data and pruning the tree back to its barest form (Fig. 3.5). Because of the size of the tree, I do not display competing variables along with the primary splitting variables. However, I do compare results of the two pruning procedures (Nodes 1-5 indicate variables used in the barest tree). The 6-branch tree accurately assigned 62 % of the streams to their actual class. Increasing the tree size did not substantially increase accuracy (from 62% to 70%). The majority of variation was explained by climate variables, with only a small portion of the variation explained by soils. The 6-branch tree was completely based on climate variables.

### **3.4 Discussion**

Although there was some regional affiliation of flow classes, HLRs and physiographic provinces did not explain substantial the variation in the grouping of flow classes in different regions. Some of my analyses were biased because I used one set of classes, which were produced using landscape-based variables, to predict another set of classes, which were produced using only hydrologic variables. Thus, I assumed that a regional framework, such as HLRs, may explain more variation in the hydrologic variables that make up flow classes, rather than the flow classes themselves. However, I found that for the majority of hydrologic variables, HLRs explained less than 30% of the overall variability. I also found that, depending on scale, different variables will govern flow variability. Altogether, my results suggest that landscape-based regional frameworks (i.e. landscape classifications) should not be used to predict hydrology unless the relative importance of variables that comprise them is allowed to change with scale (Buttle 2006).

#### *Regional Frameworks*

Regional frameworks have increasingly been used in the development of predictive tools to aid in conservation (McMahon et al. 2001; Snelder et al. 2004; Wollock et al. 2004; Sowa et al. 2007; Frimpong and Angermeier, 2010a). I used two landscape-based frameworks to predict hydrologic variability at two spatial scales in

order to understand how scale can influence a framework's predictive ability and to discuss the applicability of using regional frameworks, given their underlying structure. Although Hydrologic Landscape Regions (HLRs) were developed with variables that govern hydrology and have the potential to predict flow variability in streams, they were not necessarily created as a predictive tool. Poor et al. (2008) found that HLRs did not improve predictions of nitrate concentrations beyond commonly-used metrics. However, Hoos and McMahon (2009) found that the incorporation of HLRs into their analysis gave their models spatial structure and improved the estimation of nitrogen loading. Frimpong and Angermeier (2010a) found that HLRs did a poor job of explaining fish assemblages alone, but explained significant additional variation when nested in other frameworks.

Physiographic provinces have not been tested as a predictive tool to the extent that HLRs have been used in scientific investigations. Provinces were created for mapping rather than predictive purposes; however, they have been used as a spatial framework in the development of other relationships. For example, Johnson and Fecko (2008) showed that the majority of regional curves for channel morphology relationships are similar within physiographic provinces. Physiographic provinces have been shown to govern how different variables control hydrology (Mohamoud, 2008; Morris et al. 2009). In addition, Frimpong and Angermeier (2010a) found that physiographic provinces explained a substantial amount of variation in fish assemblages when used in conjunction with zoogeographic regions and performed better than HLRs when used alone.

Because HLRs and physiographic provinces both are defined by factors that may influence flow in streams, I wanted to determine if these regional frameworks could explain the variability in the affiliations of flow classes, especially in relation to my watershed/climate trees. I assumed that the scale at which datasets are created will largely influence their predictive abilities depending on the scale of the response dataset. Hydrologic landscape regions and physiographic provinces explained only 13% and 30% of the variation in the sub-region flow class affiliation, respectively, and explained 22% and 33% of the variation in US flow class affiliation, respectively. As an accuracy assessment, I wanted to determine how many streams within a given flow class were affiliated with one dominant HLR or province. I found that within the sub-region flow classes, 35% to 62% of the streams, on average, were affiliated with only one dominant

HLR or province, respectively. Interestingly, within each of the US flow classes, 31% to 41% of the streams, on average, were affiliated with only one dominant HLR or province, respectively. Because, the number of clusters relative to the number of regional units may create some bias, I repeated a k-means procedure in order to create as many US flow classes as regional units. However, I found that there was not a substantial increase in the percentage of streams within a dominant flow class affiliated with only one dominant HLR or province (43% to 53%, respectively).

#### *Variables that Govern Flow at Different Scales*

Isolating key physical and climate variables that are responsible for the divergence in flow classes can be useful by providing a conceptual model that shows how flow dynamics are regulated at the watershed scale and by providing a means for classifying disturbed streams that lack sufficient pre-disturbance hydrologic data. Predicting hydrologic regimes from the landscape has become a reality in water resource management (Wollock et al. 2004); thus, it may be advantageous to understand how variables that govern flow dynamics change with spatial resolution. I show that at smaller spatial scales, soils and factors that influence infiltration may govern flow dynamics whereas at the scale of the entire US, climate may be responsible for governing flow variability.

For the sub-regional flow classes, I isolated five primary splitting variables along with their corresponding competing variables that accurately assigned 74% of the streams to their actual classes (Fig. 3.4). Soil properties, such as particle size, soil thickness, and the amount of soils in various hydrologic groups, influences permeability, infiltration capacity, and the response of watershed to precipitation events (Hewlitt and Hibbert 1963). In humid areas, the vast majority of the precipitation is yielded as subsurface flow, which is primarily influenced by soil and catchment properties (Hewlitt and Hibbert 1963). Climate played a smaller role in discriminating among flow classes; however, PR2 streams were separated on the basis of northern latitude, which certainly is related to climate, as indicated by the monthly precipitation competing variables, but also to potential evapotranspiration.



Stable high baseflow streams had a lower subsurface flow contact time than CSI or IF streams. Subsurface contact time is an estimate of the time that infiltrated water remains in contact with “saturated” soil before discharging into the stream. Initially, this seems contradictory considering that stable baseflow streams are sustained by slow draining soils, which suggests that saturated conditions would be extensive. However, humid mountain catchments, at least in western North Carolina, are characterized by deep soils with saturated areas primarily confined to aquifers along channels and saturated flow occurring only for short periods of time following precipitation events (Hewlitt and Hibbert 1963). In these areas, high baseflows and stability in SBF streams are most likely sustained due to deep soils with properties conducive to the slow migration of moisture downslope and extended drainage times (Hewlitt and Hibbert 1963). SBF1 and SBF2 streams were separated from one another by bulk density as the primary variable, which again, would influence soil permeability and infiltration rates, and in turn, flow variability. Because IF streams were separated from CSI streams on the basis of soil size (related to permeability), I conjecture that small drainage basins originating in piedmont soils may induce flashiness in flow dynamics.

For the US classification, climate variables explained the majority of variation in flow classes (Fig. 3.5). The pruned tree isolated 5 climate variables that accurately classified 62% of the streams to their respective flow classes. The partially-pruned tree accurately classified 70% of the streams, in which 8 of the 12 variables were climate variables. In a similar continental scale analysis, Kennard et al. (2010b) developed a classification tree using geographic and environmental variables to discriminate among 12 flow classes across Australia. The best model included catchment, soils, vegetation, and climate variables and accurately classified 62% of the streams in the study; however, climate was the dominant variable in the model and when used alone, it accurately assigned 58 % of the streams to their respective flow class.

*Can landscape-based frameworks capture the hydrologic variability?*

Ultimately, my results suggest that two widely-used, landscape-based classifications poorly predicted stream flow variability across the entire US and within a

sub-region of the US. I find this highly significant since landscape-based frameworks have currently been used to predict the natural flow regimes of rivers and inform management (Carlisle et al. 2010a). The poor performance of both frameworks most likely stems from the purpose and scale of their creation, the underlying variability of their classification, and the structure of their framework. Large-scale regional frameworks are currently being used to organize river conservation measures (McMahon et al. 2001, Snelder et al. 2004, Wollock et al. 2004, Sowa et al. 2007). In light of this, I wanted to provide some broadly applicable considerations for management. I provide three main reasons for the inability of the landscape-based classifications used in this study to accurately predict stream flow variability:

*1. The spatial resolution of continental-wide, landscape-based classifications is too coarse for predicting the flow variability of geographically-close river systems* - The ability of a regional dataset to predict hydrology is largely an artifact of the number and size of the regions represented and differences in variability between datasets. Fewer and larger physiographic provinces most likely allowed more clustering of streams within the region's boundaries as compared to HLRs. However, I found that arbitrarily increasing the number of classes (via cluster analysis) for the entire US did not substantially increase the predictive capability of either framework. All spatial frameworks are subject to spatial autocorrelation (Frimpong and Angermeier 2010a). Obviously, physiographic provinces explained more variation in flow class affiliation because provinces are spatially contiguous whereas HLRs are not. However, my purpose was not to test how much spatial autocorrelation is explained by various regional frameworks. In contrast, I simply wanted to determine how much variation in flow class affiliation each of these frameworks explain, since they were constructed using variables that influence hydrology.

I believe that the scale of HLRs and physiographic provinces was unable to accurately assess flow variability for three main reasons: First, flow at a given gauge represents the culmination of watershed processes from that point upstream regardless of the geographical location of the gauge. In the case of large river systems, this may include areas across multiple regions. Although this may be an obvious fact, most ecologists would agree that a stream in the lower piedmont looks quite different than a

stream in a mountainous environment. The tendency is to assume that because gradient, substrate, and channel geometry are far different, flow characteristics should follow suit. However, flow metrics are calculated from discharge, which is the volume of water per time, and not just velocity, depth, and channel profile alone. Thus, there is a tendency of a river to have “flow inertia”, that is the tendency to retain flow characteristics from upstream areas despite geographical location and reach characteristics. Secondly, smaller streams, whose watershed may be entirely contained within the given province, may be located in close proximity to larger rivers, whose watershed may span multiple provinces. Thirdly, a river’s flow regime is largely dictated by watershed characteristics and climate patterns, which may vary extensively within the same physiographic province. This suggests that flow regime should be related to watershed characteristics and not just geographical location alone.

*2. Landscape-based classifications may not incorporate layers of information or hierarchical structure* - Classifications are generally a way of consolidating variability. However, the construction of one framework may poorly predict another regional framework, if the underlying variability between the two datasets is very different. Using continuous variable descriptors rather than discrete classes will allow for flexibility in the relative importance of some variables in comparison to others. Also, allowing for hierarchical structure, such as nesting classes, may be informative and increase accuracy. For example, my watershed/climate trees suggest that at various scales the relative importance of variables may change; therefore, static classifications conducted at one scale may be inappropriate for applications of finer resolution.

Buttle (2006) argues that one limitation of Hydrologic Landscape Regions is that they do not identify the relative importance of different controls on hydrology nor do they indicate how the importance of those controls change depending on scale. Interestingly, Santhi et al. (2008) found that the variables used in the construction of HLRs could be used to accurately define groundwater flow. Variables used in the construction of HLRs are publicly available for each of the 43,931 small watersheds (approx 200 km<sup>2</sup>). I only used 19 HLRs as predictors; however, if I had isolated variables used in the construction of HLRs across each of my watersheds, they may have explained a great deal of variability in flow classes. This suggests that, depending on the application, the variables

that comprise regional frameworks may be more useful for predicting flow variability than the regional framework itself.

3. *The watershed is the appropriate scale to relate landscape characteristics to flow variability* -The spatial scale of frameworks will largely influence their ability to accurately predict some response variable. For example, although I used the dominant HLR in each watershed, watersheds that were delineated at each gauge location may have been composed of many different HLRs. My watershed/climate trees accurately assigned 70 – 74% of streams to their appropriate flow class compared to an average of 31 – 35% and 41 - 62% of streams affiliated with a dominant HLR or province, respectively. Although this comparison is somewhat biased due to key structural differences, I wanted to make a very obvious point: regional frameworks created as classes are mutually exclusive whereas watersheds are not. My watershed/climate trees were created with variables that were summarized at the watershed scale. HLRs and provinces, on the other hand, span extensive areas and may not relate to the scale at which flow is measured. Carlise et al. (2010) found that HLRs poorly predicted 13 stream flow indices and concluded that local basin-scale factors in addition to regional factors must be included in models used to predict natural flow variability. Because flow in rivers is the result of a culmination of hydrologic processes within a watershed, the watershed scale (delineated at the point where hydrology is measured) is the most appropriate at linking the landscape to flow dynamics. Furthermore, this scale continuously changes with drainage area.

### **3.5 Conclusion**

Landscape-based regional frameworks (at least classes) should be used with caution as independent predictive entities of hydrology, depending on their purpose, the scale at which they were produced, and the underlying variability of their classification. The classification of flow regimes based on hydrological data alone is important in the broader management context of river conservation. A general trend in current conservation management is developing regional frameworks to organize conservation objectives (McMahon et al. 2001, Snelder et al. 2004, Wollock et al. 2004, Sowa et al.

2007). In addition, current conservation strategies have and will continue to require landscape-based models to predict stream flow variability when sufficient hydrologic information is not available (Carlisle et al. 2010a). The development of many regional frameworks operates under the assumption that similar patterns in landscape-scale factors will be represented in either physical responses (i.e. hydrology) or biotic responses. I find this highly appropriate and very useful; however, I suggest that managers should be careful in selecting variables to use in river classification. The scale of regional frameworks may not explain ecological differences in geographically close river systems (Snelder et al. 2004). My data suggest that flow regimes can be quite different for streams occurring in the same physiographic province or hydrologic landscape region and gross classes may override important differences in the hydrologic regime of rivers. If landscape-based approaches are to be used to predict hydrology, I suggest that those frameworks should be built as predictive entities using existing (although limiting) hydrologic information rather than created on the basis of landscape factors alone. In addition, because flow classes consolidate variability, landscape-based frameworks may explain more variation when predicting flow classes rather than predicting individual hydrologic indices. Thus, in order to develop landscape based frameworks, I suggest that classifications based on hydrology should be conducted first. Furthermore, similar to Poff et al. (2006b), I suggest a hierarchical approach is appropriate when applying flow variability to a geomorphic context across multiple scales. Additionally, I suggest that managers use layers of information either by nesting classes or using the underlying variables of frameworks rather than classes. Also, I suggest that the appropriate scale for attributing flow dynamics to the landscape is the watershed.

I also suggest that regional framework datasets, including variables used in their construction, should be publicly available (Frimpong and Angermeier 2010a). Much of the comparisons in this study were possible because datasets were available through USGS (Wollock et al. 2004), Ecological Archives (Falcone et al. 2010), and through direct communication with an author (Poff 1996). The utility of regional frameworks is their ability to relate to other datasets. Obviously, the utility of those frameworks cannot be tested if they have limited access.

Table 3.1. Proportion of gauges in each flow class affiliated with each hydrologic landscape region (HLR) in the southeastern sub-region. Shaded boxes represent the dominant HLR for each flow class. --- indicates no gauges were found for that respective HLR. HLRs developed by the USGS (Wollock et al. 2004). Class Codes: BKR=Black River near Tomohawk, NC; CSI=Coastal Swamp & Intermittent, IF=Intermittent Flashy, PR1=Perennial Runoff 1, PR2=Perennial Runoff 2, SBF1 =Stable High Baseflow 1, SBF2=Stable High Baseflow 2, UPR=Unpredictable Perennial Runoff.

| HLR | Flow Classes |      |      |      |      |       |       |      |
|-----|--------------|------|------|------|------|-------|-------|------|
|     | BKR          | CSI  | IF   | PR 1 | PR 2 | SBF 1 | SBF 2 | UPR  |
| 1   | 1.00         | 0.24 | ---  | 0.01 | ---  | 0.03  | 0.04  | ---  |
| 2   | ---          | 0.10 | ---  | 0.03 | 0.04 | 0.06  | 0.06  | ---  |
| 4   | ---          | 0.14 | ---  | ---  | 0.01 | 0.03  | 0.02  | ---  |
| 7   | ---          | 0.14 | 0.37 | 0.10 | 0.17 | 0.09  | 0.10  | ---  |
| 9   | ---          | 0.05 | 0.05 | 0.20 | 0.20 | 0.31  | 0.08  | 1.00 |
| 11  | ---          | 0.05 | 0.21 | 0.04 | 0.07 | 0.03  | 0.02  | ---  |
| 12  | ---          | ---  | ---  | ---  | ---  | 0.03  | 0.02  | ---  |
| 15  | ---          | 0.10 | ---  | 0.16 | 0.01 | 0.03  | ---   | ---  |
| 16  | ---          | 0.14 | 0.26 | 0.27 | 0.40 | 0.19  | 0.52  | ---  |
| 18  | ---          | ---  | ---  | 0.01 | 0.01 | ---   | 0.04  | ---  |
| 19  | ---          | ---  | ---  | 0.13 | 0.01 | 0.16  | 0.04  | ---  |
| 20  | ---          | 0.05 | 0.11 | 0.05 | 0.06 | 0.03  | 0.04  | ---  |

Table 3.2. Proportion of gauges in each flow class affiliated with each physiographic province in the study area. Shaded boxes represent the dominant province for each flow class. “---“ indicates no gauges were found for that province. Class Codes: BKR=Black River near Tomohawk, NC; CSI=Coastal Swamp & Intermittent, IF=Intermittent Flashy, PR1=Perennial Runoff 1, PR2=Perennial Runoff 2, SBF1 =Stable High Baseflow 1, SBF2=Stable High Baseflow 2, UPR=Unpredictable Perennial Runoff.

| Physiographic Province | Flow Class |      |      |      |      |       |       |      |
|------------------------|------------|------|------|------|------|-------|-------|------|
|                        | BKR        | CSI  | IF   | PR 1 | PR 2 | SBF 1 | SBF 2 | UPR  |
| Appalachian Plateau    | ---        | ---  | ---  | 0.38 |      | 0.03  | ---   | ---  |
| Blue Ridge             | ---        | ---  | ---  | 0.04 | 0.08 | 0.30  | 0.29  | ---  |
| Coastal Plain          | 1.00       | 0.95 | 0.08 | 0.06 | 0.07 | 0.18  | 0.08  | 1.00 |
| Piedmont               | ---        | 0.05 | 0.92 | 0.08 | 0.51 | 0.21  | 0.60  | ---  |
| Valley and Ridge       | ---        | ---  | ---  | 0.44 | 0.33 | 0.27  | 0.04  | ---  |

Table 3.3. Proportion of gauges in each of the 10 US flow classes, created by Poff (1996), that are found within 19 Hydrologic Landscape Regions (HLR) across the entire US. Shaded boxes represent the dominant HLR for each flow class. “--“ indicates no gauges were found for that HLR. HLRs were developed by the USGS (Wollock et al. 2004). Class Codes: BKR=Black River near Tomohawk, NC; CSI=Coastal Swamp & Intermittent, IF=Intermittent Flashy, PR1=Perennial Runoff 1, PR2=Perennial Runoff 2, SBF1 =Stable High Baseflow 1, SBF2=Stable High Baseflow 2, UPR=Unpredictable Perennial Runoff.

| HLR | US Flow Class (Poff 1996) |      |      |      |      |        |      |      |      |      |
|-----|---------------------------|------|------|------|------|--------|------|------|------|------|
|     | GW                        | HI   | IF   | IR   | PF   | PR     | SN1  | SN2  | SR   | SS   |
| 1   | 0.04                      | --   | --   | 0.06 | --   | 0.05   | 0.02 | --   | 0.02 | 0.14 |
| 2   | 0.12                      | --   | --   | --   | 0.02 | 0.02   | 0.02 | 0.09 | 0.03 | 0.23 |
| 3   | --                        | 0.03 | --   | 0.02 | --   | 0.03   | --   | --   | 0.01 | --   |
| 4   | --                        | --   | --   | 0.04 | 0.02 | 0.02   | --   | --   | 0.02 | --   |
| 5   | --                        | --   | --   | --   | --   | < 0.01 | --   | --   | --   | --   |
| 6   | 0.02                      | 0.24 | 0.19 | 0.08 | 0.08 | 0.09   | --   | 0.27 | 0.03 | --   |
| 7   | 0.11                      | 0.03 | --   | 0.06 | 0.08 | 0.10   | 0.02 | 0.09 | 0.02 | 0.09 |
| 8   | --                        | 0.16 | 0.33 | 0.02 | 0.17 | 0.02   | --   | --   | --   | --   |
| 9   | 0.11                      | 0.03 | --   | --   | 0.08 | 0.12   | 0.02 | --   | 0.02 | 0.09 |
| 10  | 0.01                      | 0.03 | 0.10 | --   | 0.02 | 0.01   | --   | --   | 0.01 | --   |
| 11  | 0.06                      | --   | 0.24 | 0.18 | 0.36 | 0.11   | 0.02 | --   | 0.02 | 0.27 |
| 12  | 0.06                      | 0.03 | --   | 0.04 | --   | 0.02   | --   | --   | --   | --   |
| 13  | --                        | 0.16 | 0.14 | 0.12 | 0.09 | --     | --   | --   | --   | --   |
| 15  | --                        | 0.05 | --   | --   | --   | 0.04   | 0.04 | --   | 0.03 | --   |
| 16  | 0.28                      | --   | --   | 0.12 | --   | 0.28   | 0.13 | 0.27 | 0.20 | 0.05 |
| 17  | --                        | 0.08 | --   | 0.04 | 0.02 | 0.01   | --   | --   | --   | --   |
| 18  | 0.01                      | 0.16 | --   | 0.14 | 0.06 | 0.02   | 0.11 | 0.09 | 0.08 | --   |
| 19  | 0.11                      | --   | --   | 0.02 | --   | 0.03   | 0.02 | --   | 0.23 | 0.14 |
| 20  | 0.05                      | 0.03 | --   | 0.04 | 0.02 | 0.04   | 0.59 | 0.18 | 0.27 | --   |



Table 3.4. Proportion of gauges in each of the 10 US flow classes, created by Poff (1996), that are found within 22 physiographic provinces across the entire US. Physiographic provinces were originally created by Fenneman and Johnson (1946) and later digitized by USGS for GIS analysis. Shaded boxes represent the dominant HLR for each flow class. “---” indicates no gauges were found for that province. Class Codes: BKR=Black River near Tomohawk, NC; CSI=Coastal Swamp & Intermittent, IF=Intermittent Flashy, PR1=Perennial Runoff 1, PR2=Perennial Runoff 2, SBF1 =Stable High Baseflow 1, SBF2=Stable High Baseflow 2, UPR=Unpredictable Perennial Runoff.

| Physiographic Province    | US Flow Class (Poff 1996) |      |      |      |      |      |      |      |      |      |
|---------------------------|---------------------------|------|------|------|------|------|------|------|------|------|
|                           | GW                        | HI   | IF   | IR   | PF   | PR   | SN1  | SN2  | SR   | SS   |
| Adirondack                | --                        | --   | --   | --   | --   | --   | --   | --   | 0.02 | --   |
| Appalachian Plateaus      | --                        | 0.03 | --   | 0.06 | --   | 0.18 | --   | --   | --   | 0.05 |
| Basin and Range           | --                        | 0.03 | 0.05 | 0.04 | --   | 0.01 | 0.04 | 0.09 | 0.01 | --   |
| Blue Ridge                | 0.19                      | --   | --   | --   | --   | 0.03 | --   | --   | --   | --   |
| Cascade-Sierra Mountains  | 0.04                      | 0.03 | --   | 0.06 | 0.02 | 0.03 | 0.02 | 0.09 | 0.24 | 0.09 |
| Central Lowland           | 0.10                      | 0.42 | 0.76 | 0.18 | 0.66 | 0.18 | 0.02 | 0.36 | 0.09 | 0.50 |
| Coastal Plain             | 0.31                      | 0.11 | --   | 0.18 | 0.13 | 0.15 | --   | --   | 0.05 | 0.05 |
| Colorado Plateaus         | 0.01                      | 0.03 | --   | --   | --   | --   | 0.07 | --   | --   | --   |
| Columbia Plateau          | 0.01                      | 0.03 | --   | --   | --   | --   | 0.02 | --   | 0.04 | 0.05 |
| Great Plains              | --                        | 0.18 | 0.19 | 0.12 | 0.08 | 0.03 | 0.02 | 0.09 | 0.01 | 0.05 |
| Interior Lowland Plateaus | 0.01                      | --   | --   | 0.06 | 0.04 | 0.02 | --   | --   | --   | --   |
| Middle Rocky Mountains    | --                        | --   | --   | --   | --   | --   | 0.28 | 0.09 | 0.01 | --   |
| New England               | 0.02                      | --   | --   | --   | --   | 0.09 | 0.04 | --   | 0.02 | --   |
| Northern Rocky Mountains  | 0.01                      | --   | --   | --   | --   | --   | 0.28 | --   | 0.09 | --   |
| Ouachita                  | --                        | --   | --   | 0.12 | --   | 0.01 | --   | --   | --   | --   |
| Ozark Plateaus            | 0.05                      | --   | --   | 0.02 | --   | 0.03 | --   | --   | --   | --   |
| Pacific Border            | 0.01                      | 0.16 | --   | 0.14 | 0.08 | 0.02 | --   | --   | 0.39 | 0.05 |
| Piedmont                  | 0.19                      | --   | --   | --   | --   | 0.11 | --   | --   | --   | --   |
| Southern Rocky Mountains  | --                        | --   | --   | --   | --   | --   | 0.13 | --   | 0.01 | --   |
| Superior Upland           | --                        | --   | --   | --   | --   | 0.01 | --   | 0.09 | 0.02 | 0.18 |
| Valley and Ridge          | 0.05                      | --   | --   | --   | --   | 0.09 | --   | --   | --   | --   |
| Wyoming Basin             | --                        | --   | --   | --   | --   | --   | 0.07 | 0.18 | --   | --   |

Table 3.5. Results of one-way analysis of variance for 15 hydrologic variables among 19 different Hydrologic Landscape Regions and 10 US flow classes, created by (Poff, 1996). Hydrologic variables used in the clustering procedure of the 10 US flow classes. All one-way comparisons were significant ( $p < 0.0001$ ).

| Hydrologic Variables                    | Hydrologic Landscape Regions |                |                     | US Flow Classes |                |                     |
|---|------------------------------|----------------|---------------------|-----------------|----------------|---------------------|
|   | F                            | r <sup>2</sup> | r <sup>2</sup> Adj. | F               | r <sup>2</sup> | r <sup>2</sup> Adj. |
| Daily Mean Discharge                    | 4.33                         | 0.09           | 0.07                | 9.79            | 0.10           | 0.09                |
| Mean Annual Run-off                     | 28.87                        | 0.40           | 0.39                | 37.20           | 0.30           | 0.29                |
| Daily Flow Variability (CV)             | 27.13                        | 0.39           | 0.37                | 163.88          | 0.65           | 0.65                |
| Predictability of Flow                  | 21.54                        | 0.34           | 0.32                | 141.65          | 0.62           | 0.62                |
| Flood Variability                       | 8.42                         | 0.16           | 0.15                | 16.22           | 0.16           | 0.15                |
| Flood Frequency                         | 5.09                         | 0.11           | 0.09                | 12.85           | 0.13           | 0.12                |
| Flood Duration                          | 13.40                        | 0.24           | 0.22                | 56.35           | 0.39           | 0.39                |
| Seasonal Predictability of Flooding     | 11.32                        | 0.21           | 0.19                | 131.68          | 0.60           | 0.60                |
| Timing of Flooding                      | 5.99                         | 0.12           | 0.10                | 18.36           | 0.18           | 0.17                |
| Seasonal Predictability of Non-flooding | 14.50                        | 0.25           | 0.24                | 132.40          | 0.61           | 0.60                |
| Number of Zero Flow Days                | 8.10                         | 0.16           | 0.14                | 562.65          | 0.87           | 0.87                |
| Baseflow Index                          | 8.34                         | 0.16           | 0.14                | 164.98          | 0.65           | 0.65                |
| Seasonal Predictability of Low Flow     | 13.14                        | 0.24           | 0.22                | 41.23           | 0.32           | 0.32                |
| Timing of Low Flow                      | 6.28                         | 0.13           | 0.11                | 31.38           | 0.26           | 0.26                |
| Seasonal Predictability of Non-low Flow | 17.40                        | 0.29           | 0.27                | 63.42           | 0.47           | 0.47                |

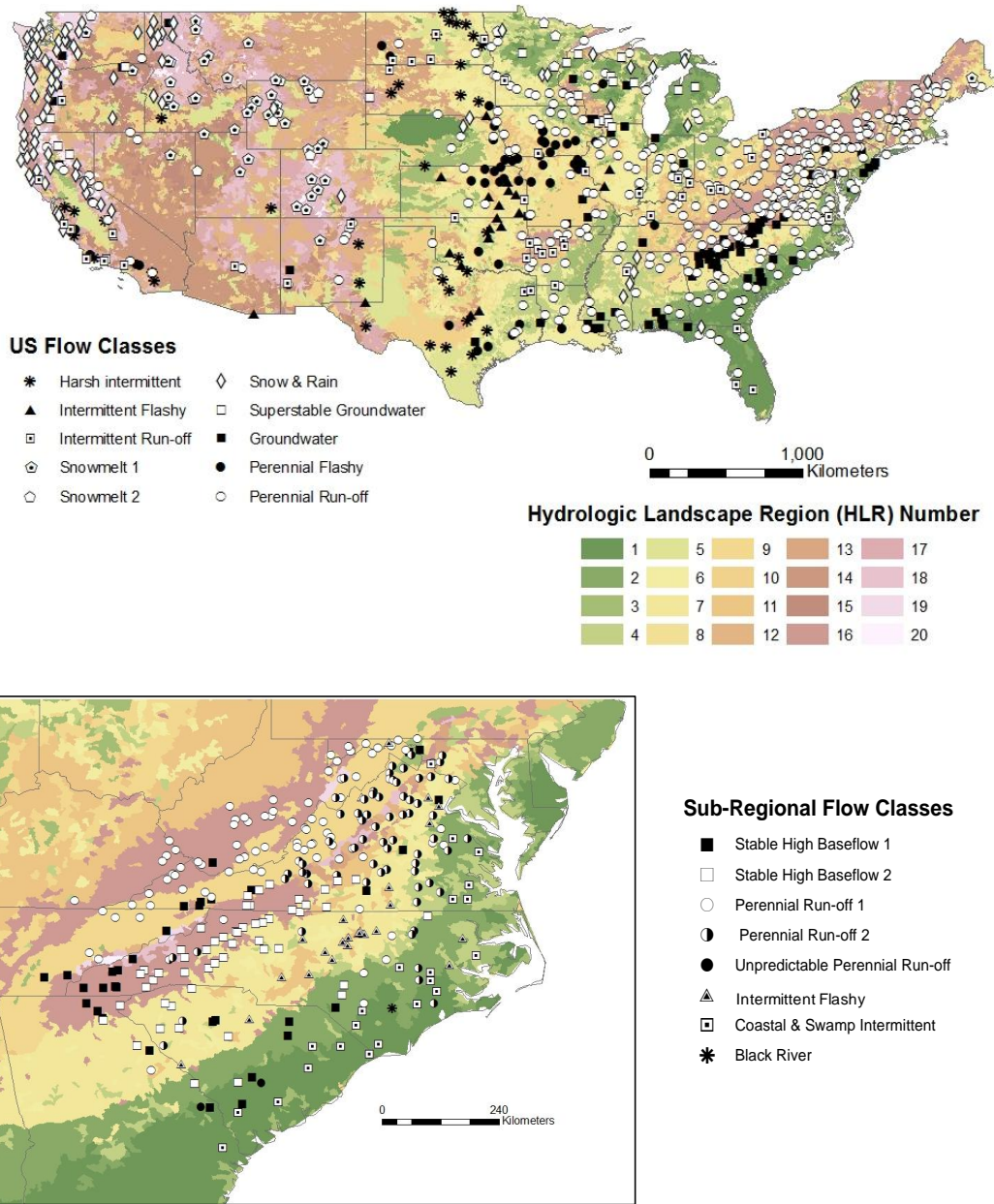


Figure 3.1. Geographical affiliation of flow classes for the US and the Southeastern sub-region across Hydrologic Landscape Regions.

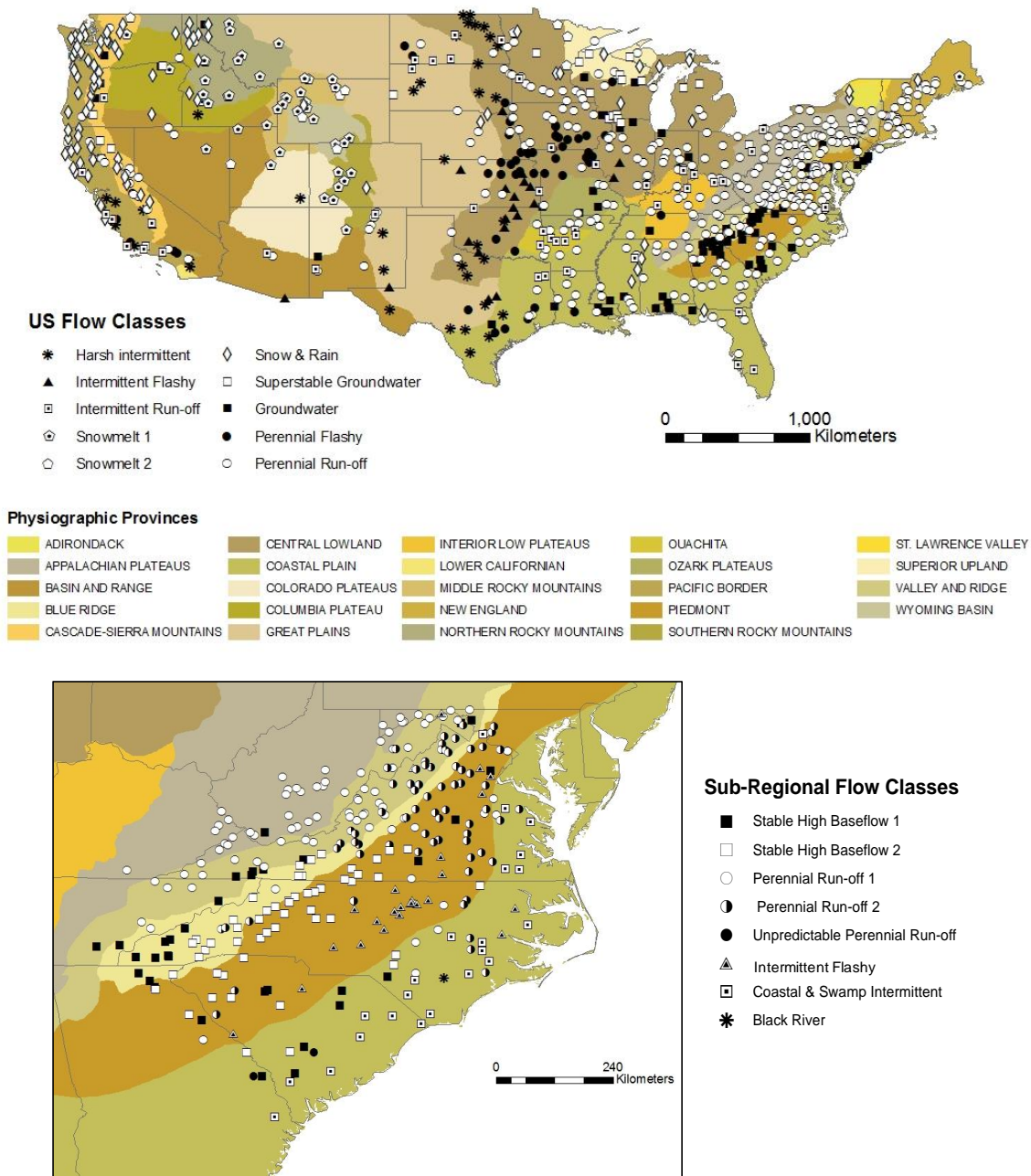


Figure 3.2. Geographical affiliation of flow classes for the US and the Southeastern sub-region across physiographic provinces.

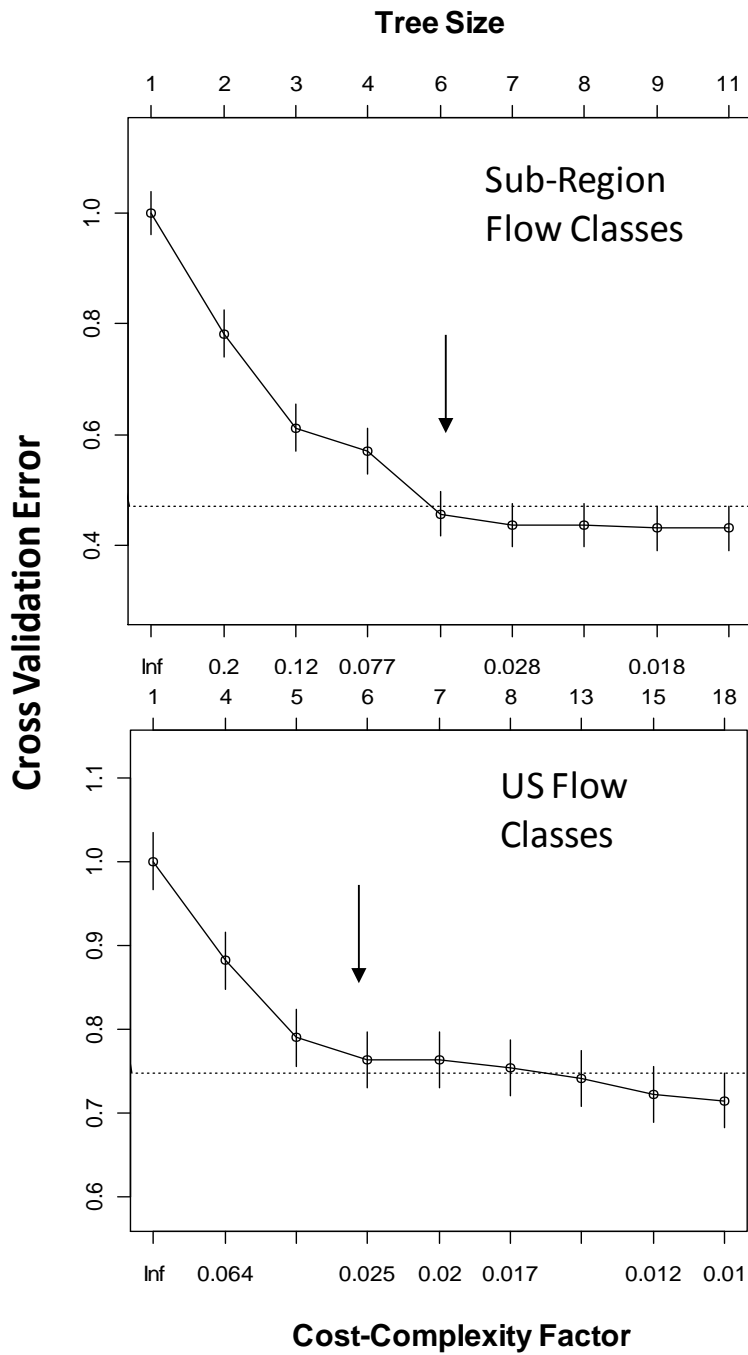
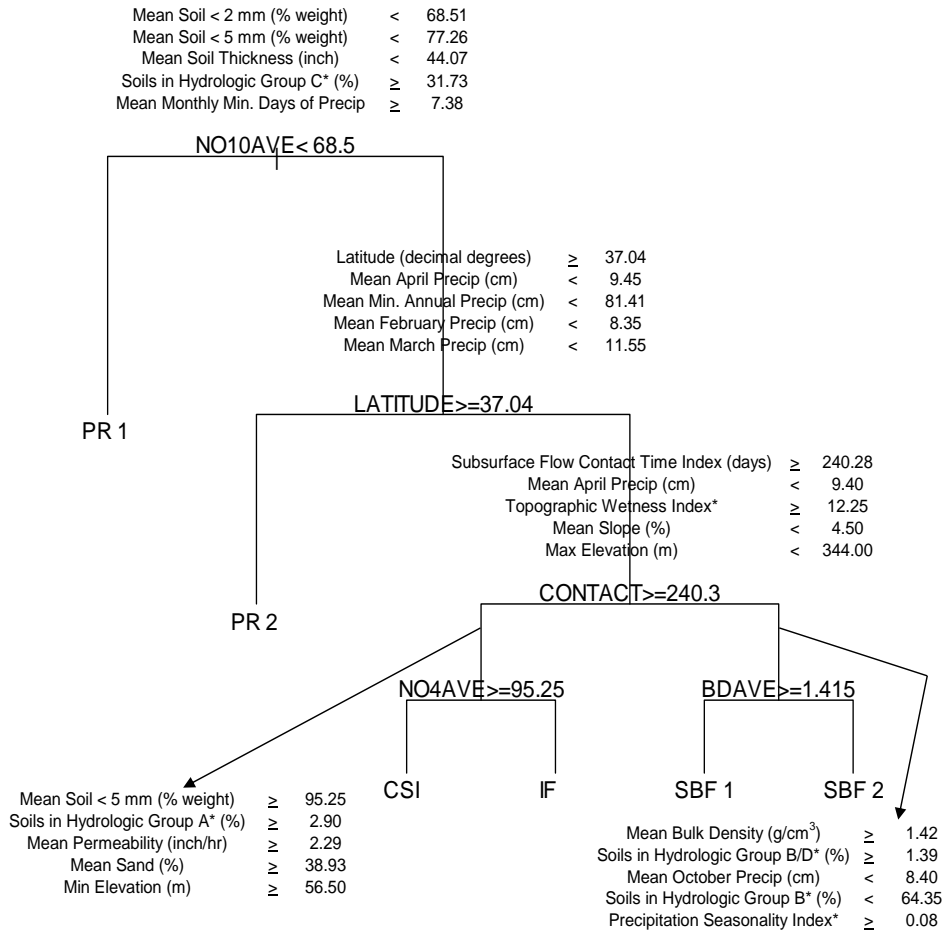


Figure 3.3 Plots for the hydrologic and watershed classification trees comparing the cross-validation error to the tree size (number of nodes) in order to determine where the tree should be pruned. Trees are generally pruned at the cost-complexity factor that minimizes the number of nodes and the cross-validation error. Arrows indicate the tree size that I plotted



**Actual Classification**

|       | BKR  | CSI  | IF   | PR 1 | PR 2 | SBF 1 | SBF 2 | UPR  |
|-------|------|------|------|------|------|-------|-------|------|
| n     | 1    | 21   | 19   | 79   | 70   | 32    | 48    | 2    |
| CSI   | 1.00 | 0.81 | 0.05 | --   | 0.13 | --    | --    | 0.50 |
| IF    | --   | 0.05 | 0.74 | 0.08 | --   | --    | 0.02  | --   |
| PR 1  | --   | --   | 0.05 | 0.78 | 0.10 | 0.03  | --    | --   |
| PR 2  | --   | 0.14 | 0.16 | 0.05 | 0.71 | 0.09  | 0.02  | --   |
| SBF 1 | --   | --   | --   | 0.01 | 0.01 | 0.59  | 0.13  | 0.50 |
| SBF 2 | --   | --   | --   | 0.08 | 0.04 | 0.28  | 0.83  | --   |

**Tree Assignment**

Figure 3.4 Classification tree using 5 climate/watershed metrics as primary splitting variables along with the 4 corresponding competing variables to classify 6 of the 8 flow classes in this study. The left branch meets the conditions of the equation on each node. The matrix below the tree shows the proportion of gauges in the actual flow class (columns) classified to each flow class using the tree (rows). The proportion of each flow class accurately assigned by the tree is shown in gray boxes. For class codes, see Figures 3.1 and 3.2.

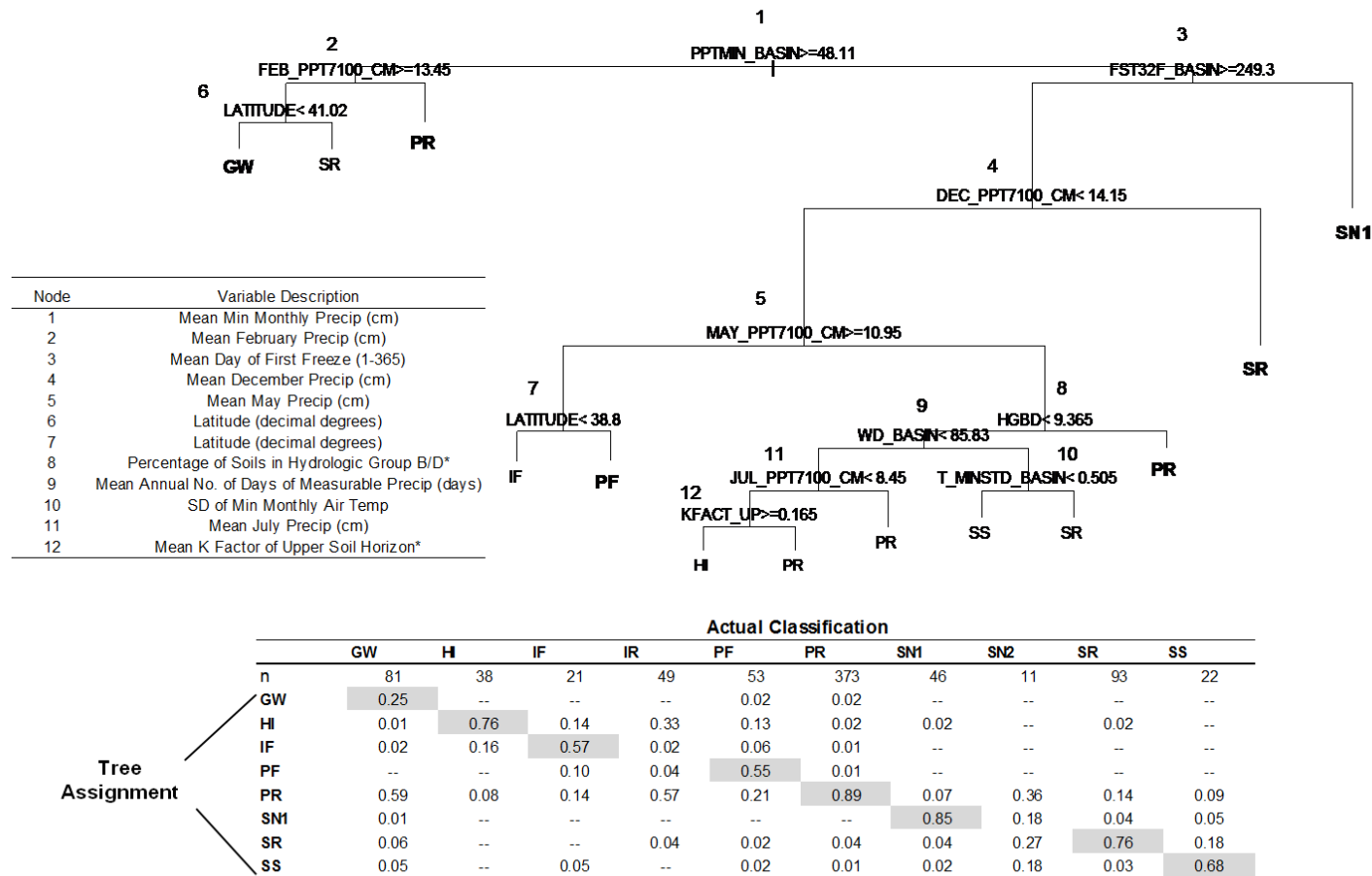


Figure 3.5 Classification tree using 5 climate/watershed metrics as primary splitting variables along with the 4 corresponding competing variables to classify 8 of the 10 US flow classes (Poff, 1996). Tree shown has been pruned to a  $cp=0.015$ . The first 5 nodes and the classes in larger, bold letters indicate the tree pruned to a  $cp=0.025$  (see methods). The left branch meets the conditions of the equation on each node. The matrix below the tree shows the proportion of gauges in the actual flow class (columns) classified to each flow class using the tree (rows). The proportion of each actual flow class accurately assigned by the tree is shown in gray boxes. For class codes, see Figures 3.1 and 3.2.

#### **4. Do Dams Affect Streams the Same? A Step Towards Generalizing Regulated Stream Flow Patterns.**

##### **Abstract**

For some time, ecologists have attempted to make generalizations concerning how disturbances influence natural ecosystems, especially river systems. Because the existing literature suggests that dams “homogenize” the hydrologic variability of rivers, a possible misconception is that dam regulation influences all rivers the same. In order to evaluate patterns in dam-regulated hydrology and associated ecological relationships, a broad framework is needed. Flow classes, or groups of streams that share similar hydrology, may provide a framework to evaluate the relative effects of dam regulation on natural flow dynamics. The purpose of this study was to use a regional flow classification as the foundation for evaluating patterns of hydrologic alteration due to dams and to determine if the response of rivers to regulation was specific to different classes. I used the US Geological Survey (USGS) database to access discharge information for 284 unregulated and 117 regulated flow records. For each record, I used the Hydrologic Index Tool to calculate 44 hydrologic statistics, including the Indicators of Hydrologic Alteration. I used a sub-regional flow classification created by McManamay et al. (2011) for eight states as a way to stratify unregulated and regulated streams into comparable units. For streams with at least 15 years of pre- and post-regulation information (n=46), I evaluated the percent change in 10 ecologically-relevant hydrologic variables following dam regulation for all streams and separately for individual flow classes. For each flow class, I built general linear models to predict the effect of dam regulation on hydrologic response variables while including the potential effects of urbanization, withdrawals, and fragmentation in each basin. I also conducted principal components analyses (PCA) on hydrologic variables for all streams and within each flow class to determine how dam regulation influences the overall variation of flow in multivariate space. Results of the linear models showed that dam regulation had the strongest effects on hydrologic indices; however, the effects of urbanization, withdrawals, and fragmentation, at times, were comparable or exceeded the effects of dam regulation depending on the hydrologic variable. In agreement with the existing literature, maximum flows, flow variability, and rise rates were lower whereas minimum flows and reversals were higher in dam regulated streams. However, the response of monthly and seasonal flows, flow predictability, and baseflows were variable depending on flow class membership.



The PCA showed that regulated streams occupied a larger multivariate space than unregulated streams, which suggests that dams may not homogenize river systems, but they may move them outside the bounds of “normal” river function. Ultimately, my results suggest that flow classes provide a suitable framework to generalize patterns in hydrologic alterations due to dam regulation.

#### **4.1 Introduction**

Of the many disturbances that negatively impact the integrity of aquatic ecosystems, dam regulation results in the most extensive damages (Vitousek et al. 1997). Dams diminish the hydrologic variability of river systems that is responsible for forming and maintaining the habitats to which river biota are adapted (Poff et al. 1997; Trush et al. 2000; Bunn and Arthington 2002). A general rule of thumb is that dams tend to homogenize river flows across geographical scales, leading to a loss of habitat variability (Poff et al. 2007). For example, the majority of studies show that following dam regulation, minimum flow magnitudes increase whereas maximum flow magnitudes decrease (Magillan and Nislow 2001 and 2005; Pyron and Neumann 2008; Poff et al. 2007). Other general patterns include decreased rise and fall rates of hydrographs and increases in reversals (positive or negative changes from one day to the next) (Magillan and Nislow 2001 and 2005; Pyron and Neumann 2008). However, there are exceptions to the rule that dams influence all rivers the same. This may be especially true for rivers in the southeastern US that differ in terms of climate and geomorphology. For example, the effect of dams on monthly flows, the frequency of flows, and the timing of flows can vary due to dam type, dam operations, and are specific to different regions (Richter et al. 1996; Magillan and Nislow 2005; Pyron and Neumann 2008).

One of the obvious and more robust approaches for determining the effect of dam regulation on stream flows is by comparison of pre- and post-regulation datasets, which has been conducted extensively (Richter et al. 1996; Magillan and Nislow, 2001; Magillan and Nislow, 2005; Poff et al., 2007; Pyron and Neumann, 2008; Gao et al. 2009). Using paired pre- and post-regulation data comparisons for individual drainages is a preferred method because it controls for differences in basin size and confounding factors. However, pre-post-designs do not control for temporal shifts in climate regimes. In addition, the criteria for selecting appropriate gages for such analyses can be quite strict (e.g. adequate pre- and post-regulation information, usually

regulated by no more than one dam, and basins are not disturbed by factors other than dam regulation). Confining analyses to these criteria results in reduced sample size, reduced spatial resolution, and ultimately, a loss of information. Thus, it appears there is a need for a general framework that provides an improved, alternative means of assessing the influence of dam regulation on stream hydrology. Such a framework could stratify basins into comparable units thereby eliminating the need, but allowing the inclusion, of pre- and post regulation information. However, the framework would also have to control for differences among basins, such as basin size and other hydrologic disturbances. Additionally, a similar framework would also be useful as structure for evaluating the effect of hydrologic alterations on aquatic biota, especially since there is limited information to develop quantitative relationships between flow alterations and ecological responses (Carlisle et al. 2010b; Poff and Zimmerman 2010).

Geographic or geomorphic settings provide some context for evaluating hydrologic variability (Poff et al. 2006b) or patterns in hydrologic alterations (Poff et al. 2007). For example, Poff et al. (2007) used 16 regions of the US to evaluate patterns in hydrologic alterations. Although the study did show a tendency of dams to “homogenize” natural flow variability, the magnitude and direction in which dams influence flow dynamics were region-specific. Another approach to stratify hydrologic alteration studies is by classifying streams into groups of similar flow characteristics (Arthington et al. 2006) (Fig 4.1). Classifications of rivers based on stream discharge data alone have been conducted at the continental scale (Poff and Ward 1989; Poff 1996; Kennard et al. 2010b), the regional scale (Chapter 2, McManamay et al. 2011), and for individual states (Kennen et al. 2007; Turton et al. 2008; Kennen et al. 2009). These flow classes can be used as ecologically-meaningful management units by which environmental flow standards are formed or as a framework for evaluating hydrologic alterations (Arthington et al. 2006). For example, a disturbed river can be classified to a group of streams that share similar unregulated or undisturbed flow characteristics either by using pre-disturbance hydrologic information or by using landscape characteristics (Fig. 4.1). Flow classes then become the organizational structure by which hydrologic alterations are measured (Fig. 4.1).

The purpose of this study was to use a regional flow classification as the foundation for evaluating patterns of hydrologic alteration due to dams. In order to evaluate patterns in hydrologic alterations, I used the 33 indicators of hydrologic alteration (IHA) because they explain the majority of variation in all 171 published hydrologic indices (Olden and Poff 2003).

In addition, I also use 3 environmental flow component (EFC) indices (Mathews and Richter 2007) and 8 indices used in the flow classification created by Poff (1996) because of their practical application and because they explain additional variability in the streams in my study region (McManamay et al. 2011). Using flow classifications can provide a framework for creating environmental flow standards, assessing patterns in hydrologic alterations, and forming relationships between altered hydrology and ecology (Poff et al. 2010). I used the flow classification created in Chapter 2, which isolated 6 distinct flow classes in an eight-state region of the southeastern US. Although broad generalizations may exist in how dams influence flow dynamics throughout my region of interest, hydrologic alterations due to dams may be class-specific. In addition, flow classes may control for some of the differences among basins thereby eliminating the necessity to use only gages with adequate pre- and post-regulation data. However, other disturbance factors, such as withdrawals and urbanization, should be considered. Therefore, I developed models to isolate the effects of dam regulation from other potential disturbance factors. Lastly, it may be informative to understand how dam regulation influences the overall variability in river systems in order to adequately judge whether there may be a “homogenizing” effect. My specific objectives were to 1) compare pre/post-regulation stream flow records as standard protocol to provide initial evidence for class-specific hydrologic alterations caused by dams and to warrant further investigation, 2) isolate the effects of dam regulation from other causes of hydrologic disturbance within flow classes using all stream gages, regardless of pre/post regulation hydrologic data, 3) evaluate how dam regulation influences the overall variability in river systems, and 4) determine which disturbance factors explain the most variability in hydrologic indices.

## **4.2 Methods**

I accessed the US Geological Survey (USGS) Realtime Water Data for the Nation website (<http://waterdata.usgs.gov>) to find daily stream gauge data and to judge the extent of regulation due to impoundments. I selected gages with at least 15 years of total data (some gage records had missing data as long as at least 15 total years were represented), which should be sufficient for detecting differences in hydrologic variables summarized across entire periods of record and not evaluating changes in hydrologic variables across temporal scales (Kennard et al. 2010a). I initially categorized gages as unregulated or regulated by evaluating the “remarks”

section of the USGS annual water reports. Water reports generally indicated what dams, mills, municipalities, or power plants, if any, caused substantial regulation of flows and the time periods in which flow has been regulated. I isolated “unregulated” gages as those with no remarks concerning regulation, with the exception of small diurnal fluctuations due to minor withdrawals. To further ensure that “unregulated” streams were not influenced by other variables such as urbanization and substantial withdrawals, I eliminated other streams that were disturbed according to similar methodology in Chapter 2. I determined “regulated” status as gages where dam regulation was mentioned as the primary regulation of river flows. For stream gages with extensive records that had at least 15 years of pre-impoundment data, I selected data from periods of time with no regulated flow to include in my analysis as “unregulated” conditions whereas periods of regulated flow were included with the “regulated” gages. I also used gages that had extensive pre- and post-regulation data in the *before and after* analysis. The majority of regulated gages were influenced by only one dam; however, I included gages influenced by multiple reservoirs in an attempt to generalize patterns in how dams affect river flow, despite differences in the amount of reservoir storage and the serial discontinuity of river systems (Ward and Stanford 1983). Secondly, because information was available on the amount of storage and density of dams in each watershed (Falcone et al. 2010), including a disturbance gradient of impounded rivers may be informative. Because other factors, such as withdrawals, urbanization, and power plants could influence flow in regulated rivers, I attempted to determine the relative importance of other disturbance variables in influencing flow dynamics, which I discuss in the *Hydrologic Alteration Model* section.

Mean daily and annual peak flow data were downloaded from the USGS Realtime Water Data for the Nation website for 401 stream gage records (284 unregulated and 117 regulated records). Of the 287 unregulated records, 49 of the records were pre-regulation information from gages that were currently regulated. Similarly, of the 117 regulated gage records, 49 of the records were post-regulation. Hydrologic statistics were calculated for each stream record using the Hydrologic Index Tool (HIT) software available through the USGS (Henriksen et al., 2006). Daily and peak flow gauge data were imported into the HIT software, which calculates the 171 hydrologic indices reported in Olden and Poff (2003). The indices are summaries of the entire period of record. The indices are grouped into five categories of flow: magnitude (n = 94), frequency (n = 14), duration (n = 44), timing (n = 10), and rate of change (n = 9) with each

category having low, average, and high flow subcategories (Richter et al., 1996; Olden and Poff, 2003). Because of the large amount of correlated variables, I reduced the dataset to 44 variables (Table 4.1), which included the 33 IHA statistics, 4 EFC variables, and 8 of the variables used in Poff (1996). All magnitude variables and any variables related to magnitude were divided by the median daily flow to standardize for differences in river size.

### *Assigning stream gages to flow classes*

The first step in forming generalizations of how dams affect flow dynamics is by assigning streams to a particular class (Fig. 4.1, Arthington et al. 2006) or to a particular region (Poff et al. 2007). The majority of unregulated gages and pre-regulation flow periods in my dataset were used in the flow classification in Chapter 2 and thus, were already assigned to 1 of 6 flow classes created for this study's region of interest. Post-regulation flow periods were assigned to the same class as their respective pre-regulation flow periods. However, for the remainder of the regulated gages, I used an existing classification framework to assign gages to appropriate flow classes. Hydrologic classification trees, based solely on hydrologic data, and watershed classification trees, based solely on landscape characteristics, were developed in Chapter 3 for the continental US and for a sub-region of the southeastern US in order to assign gages to particular flow classes. Watershed classification trees were built using the GAGES dataset (Falcone et al. 2010), which includes soils, topographic, and climate information summarized across the contributing watershed upstream of each gage. The GAGES dataset includes information for 6,785 USGS gages with at least 20 years of data, which included the regulated stream gauges used in my study. Because the majority of regulated gages had inadequate pre-regulation hydrologic data, I used the sub-region watershed classification tree, which consists of 4 soil-hydrology variables and latitude with a 74 % accuracy rate, to assign gages to 1 of 6 flow classes (Chapter 3). Although some gages may have been misclassified, misclassification generally results in gages being assigned to flow classes that share similar hydrology (Chapter 3).

### *Pre- and Post-Regulation Analysis*

My main purpose was to provide evidence that classes provide a structural framework from which generalizations can be made concerning the effect of dam regulation on streamflows.

Because pre/post analyses have dominated the literature, I wanted to use this analysis as a first step to provide some justification that flow classes can be used as a basis for evaluating hydrologic alterations. I calculated percent changes in 10 ecologically-relevant hydrologic variables following dam regulation for the 49 gauges with pre and post-regulation data. I used raw values for hydrologic variables rather than those standardized by the median daily flow for the before/after analysis. I sorted each pre-post analysis by flow class and evaluated percent changes among different flow classes using box plots.

### *Assembling the Hydrologic Disturbance Dataset*

Determining the influence of dams on natural flow dynamics by using only gages with adequate pre-regulation data may limit the sample size and may exclude important information in analyses, especially within particular regions. For example, only 49 of the 117 regulated gages in my study region had adequate pre-regulation data. Therefore, to increase the power in analyses testing how dams influence flow dynamics, I wanted to evaluate gross differences in hydrologic variables between regulated and unregulated streams. One limitation of this approach is that gross comparisons may not account for differences in watershed characteristics and other disturbance factors, besides dam regulation. Hence, to conduct gross comparisons between unregulated and regulated streams, accounting for other disturbance factors would be necessary.

The GAGES dataset was created to identify reference streams and the impacts of flow alterations on natural flow. The dataset contains many hydrologic disturbance variables, including 27 dam regulation variables, 11 hydrologic modification variables (e.g. withdrawals), and a large suite of land-use metrics. A subset of the disturbance variables was used to calculate a hydrologic disturbance index for all 6,785 streams in the dataset (Falcone et al. 2010), which can be used as a composite score to assess the relative impacts on flow. Each of the variables in the dataset represented a summary for each gage's entire basin and did not just represent values at each gage location.

One challenge that arose with my dataset was that the same hydrologic disturbance values were originally assigned to both pre- and post-regulation gages because the disturbance assessment was based on current conditions. For example, dam disturbance variables, such as total dam storage, were based on 2006 National Dam Inventory Data (USACE) whereas land-use variables, such as % urbanization and % fragmentation in the watershed, were based using the

2001 National Land-Cover Dataset (NLCD). Freshwater withdrawal estimates came from 1995 – 2000 county-level estimates from USGS datasets. Thus, to use hydrologic data from the pre-regulation time periods in my dataset, I had to correct for recent changes in water use, dam regulation, and land use. The GAGES dataset included changes in total dam storage and dam density for each gage in every decade since 1940 (National Dam Inventory), which allowed for easily correcting for differences in total storage in each basin. If the dam regulation time period was prior to 1930, then I assumed a value of 0 for both dam storage and dam density.

Historical values for withdrawals and land use were not as readily available as dam regulation information. I used the 2005 USGS national water report (Kenny et al. 2009) to evaluate changes in withdrawals over time. Trends in water withdrawals were available for each major water consumption category across the US since 1950 (public supply, domestic, irrigation, livestock, industrial, and thermoelectric) (Fig 4.2). Most categories showed general increases in water use since 1950 except for industrial uses (Fig 4.2). Thermoelectric and irrigation withdrawals peaked in 1980 followed by a small decrease in 1985 and have remained at a steady rate since. Industrial uses increased and peaked in the 1970s and then have shown decreases since. I developed linear regressions on percent changes in withdrawals according to year for all categories except industrial, in which I ran a second-order polynomial regression. Based on the results of the 2005 USGS national water report (Fig. 4.2), thermoelectric and irrigation withdrawals were assumed to be the same as current rates, unless the pre-regulation time period was prior to 1985 (which was the case for the majority of gages). The proportion of water use in each category was available for each state (Appendix A). Thus, the percent changes in withdrawals for each category were weighted by water use category depending on the location of each gage. I then applied regressions for each gage based on the year since dam regulation to correct withdrawal estimates. Thermoelectric withdrawals dominated water usage trends for most states and across the US; however, based on USGS annual water reports, thermoelectric usage is patchy and does not occur in every basin. Since accounting for thermoelectric usage in each basin could heavily influence my historical withdrawal estimates, I only included trends in thermoelectric usage for gages in which the USGS water reports mentioned some flow regulation due to power plants within the basin.

Historical trends in land use since 1950 were available for the southeastern US according to seven different level-3 ecoregions (Brown et al. 2005) (Appendix B). I ran regressions for

percent changes in each land-use category (% urban and % agriculture) for each ecoregion. I corrected for changes in % urban and % agriculture land cover types using the year since dam regulation depending on the ecoregion in which each gage was located. Watershed fragmentation is an index based upon the percentage of undeveloped land (non-urban and non-agricultural land – higher index values indicate more fragmentation (Falcone et al. 2010)). Because % agriculture showed relatively little change (0% to 21% decrease) compared to urbanization (33% to 103% increase), I used changes in % urban land cover to account for any changes in fragmentation.

One of the limitations in my analysis for correcting withdrawal and land-use estimates is that I assume patterns across the US and across entire ecoregions are representative of patterns within each basin. Also, my corrected withdrawal and land cover estimates were highly dependent upon current estimates (corrected using % change); thus, if withdrawal and urbanization is currently high within a particular basin, “pre-regulation” estimates should reflect current high conditions. However, I do not expect that slight inaccuracies in assessing historical estimates would not overwhelm my analyses since there are only 49 “pre-regulation” gages out of the 284 unregulated gages. Lastly, flow classes represent differences in hydrology that vary according to watershed, climate, and geography. Thus analyzing patterns of dam regulation within flow classes should control for some factors that may confound my analyses.

Regulated *and* unregulated rivers may show a large gradient of hydrologic alteration. Thus, the hydrologic disturbance index should provide some assessment of cumulative hydrologic disturbances within each basin and can be used as a first step in partitioning the various contributors to hydrologic alteration. After I assembled the hydrologic disturbance variables, I developed a new HDI for the study region. I chose a subset of the hydrologic disturbance variables that were pertinent to my analysis (freshwater withdrawals, total dam storage, major dam density, % urban landcover, and fragmentation). Fragmentation represents an index of undeveloped land and can be used as a surrogate for the combined disturbance caused by urbanization and agriculture. I did not include % agriculture as a disturbance variable because it was highly correlated with fragmentation, whereas % urbanization was not. Similar to Falcone et al. (2010), I calculated thresholds for each disturbance variable based on percentiles (10% increments). I then assigned scores of 1 to 10 for each variable and the sum of the scores were used to calculate a hydrologic disturbance index for each stream. I imported all gages, their



GPS location, and their hydrologic disturbance index values into ARC map 9.2. I used natural breaks (Jenks 1967) to categorize the hydrologic disturbance index into low, low to moderate, moderate, moderate to high, and high categories. I then plotted unregulated and regulated streams on maps to visualize overall hydrologic disturbance in the region. I compared the hydrologic disturbance index values in regulated and unregulated streams for all streams using a Mann-Whitney Test.

### *Hydrologic Alterations within Classes*

One of my objectives was to generalize patterns in how dam regulation influences hydrologic variables within classes. However, as mentioned previously, gross comparisons based on subjective delineations into “unregulated” and “regulated” categories may be confounded due to differences in other hydrologic disturbances or differences in drainage area, which can certainly influence flow dynamics (Poff et al. 2006a). I found that the mean drainage area was slightly larger for regulated rivers compared to unregulated rivers, which could influence analyses. Even after my hydrologic variables were standardized by the median daily flow, they still were related to drainage area. For example, the standardized rise rate of flow events has a negative relationship with drainage area. Thus, to control for differences in basin size, I ran regressions for each hydrologic variable drainage area for only unregulated streams and then calculated residuals for both regulated and unregulated streams. Using only unregulated streams to form regressions ensures that relationships between drainage area and hydrologic variables are natural and not biased due to regulated rivers with larger basins. I ran separate regressions for all streams and for each class (Table 4.2). All hydrologic variables and drainage area were  $\log(x+1)$  transformed prior to any analysis.

After I calculated residuals, I had to control for the influence of other hydrologic disturbances that may confound differences in hydrologic variables due to dam regulation. Thus, I built general linear models using dam regulation (regulated or unregulated) along with other hydrologic disturbance variables (withdrawal estimates, fragmentation index, and % urban landcover) to predict responses in the residuals of 40 hydrologic variables (Table 4.1). I built models for all streams and for each flow class. I then compared t-statistics calculated for the dam regulation parameter in each of the linear models constructed for all 40 hydrologic variables in all streams and in each flow class. It also may be informative to compare the magnitude and

direction of the t-statistics for dam regulation compared to other hydrologic disturbance parameters. Therefore, I compared the mean t-statistic for each of the hydrologic disturbance parameters (dam regulation, withdrawal, fragmentation, urbanization) for 9 hydrologic variables using the 6 flow classes as replicates ( $n=6$ ). Combining t-statistics across flow classes can also aid in generalizing patterns in dam regulation and in other causes of hydrologic alteration. Because I had used exclusive classes (regulated or unregulated) to represent dam regulation, I questioned whether a more continuous variable, such as total dam storage would be more powerful in a linear model. Furthermore, classifying streams as regulated or unregulated may be easier than calculating total dam storage. Thus, I re-ran the models for the 9 hydrologic variables using total dam storage (storage/drainage area) rather than the regulated-unregulated classes and compared their t-statistics. I used t-statistics rather than actual parameter estimates because their single value represents the directionality of each parameter with respect to the standard error as well as the significance level. Once again, all hydrologic response variable and hydrologic disturbance predictors were  $\log(x+1)$  transformed prior to any analysis.

#### *Overall Variation in Flow Dynamics of Regulated and Unregulated Streams*

Comparisons of single hydrologic variables can be very informative in evaluating how and to what extent dam regulation may alter hydrology. However, because streams cluster together (i.e. form classes) and because multiple hydrologic variables are correlated, the influence of hydrologic alterations on natural flow dynamics should be explored in multivariate space. I conducted principal component analyses (PCA) on correlations for regulated and unregulated rivers for all streams and within flow classes to examine how dam regulation may influence the overall variation of the hydrologic variables using JMP 8.0 software (SAS). I conducted a PCA on 38 rather than 40 variables because of missing values and the inclusion of other variables. Six of the variables were not calculated for all streams. Of the missing six variables, I replaced four with variables from Richter et al. (1996) and Poff (1996) as substitutes since these variables tended to exhibit some divergence between regulated and unregulated rivers. I ran PCAs on correlations since PCAs on covariance resulted in individual variables having the highest loadings on multiple components. I did not control for differences in land use, withdrawals, or drainage area because I wanted to see the overall existing correlation of streams

in multivariate space. Variables were standardized by subtracting each value by the sample mean and then dividing that value by the sample standard deviation prior to analysis.

I used the broken-stick method to determine how many principal components to retain because it is simple to calculate, accurately assesses dimensionality, and does not overestimate the number of interpretable components compared to other methods (Jackson 1993). The broken-stick rule involves comparing eigenvalues calculated from random data to eigenvalues from the actual data. The number of interpretable components is found where the eigenvalues from random data exceed those of the actual data (Jackson 1993). I manually calculated eigenvalues for random-data according to Jackson (1993) to find the number of interpretable components. For each component, I sorted variables in increasing order and then plotted the distribution of loadings to select outliers or “breaks” in order to interpret components. Because hydrologic variables can be highly correlated and lead to redundancy (Olden and Poff 2003), isolating single or multiple variables with the highest loadings on each principle component may be difficult. Most components had obvious outliers with strong negative or positive loadings. However, in components without obvious outliers, I manually chose up to a maximum of 5 variables on either side of the distribution to interpret components. I plotted regulated and unregulated streams on 3-dimensional scatter-plots using the first three components to visually evaluate the divergence of regulated and unregulated streams within Sigma Plot 9.0. I spun the principal components in order to display the most divergence between regulated and unregulated streams.

### *Hydrologic Alteration Model*

My last objective was to determine the relative influence of various hydrologic disturbance variables in explaining variation in hydrologic response variables with respect to classes. Altered hydrology may result from compounding or confounding disturbances. Hence, hydrologic alteration models should show the relative contribution of different variables in explaining variation in hydrologic variables but also should isolate the “best” predictor variables from a suite of potential candidates. However, flow classes are a reference or “starting point” from which hydrologic disturbances cause deviations in natural flow since they represent differences in climate, soils, and topography (Chapter 3). Furthermore, differences among flow classes may account for a large amount of the variation in hydrologic variables and should

possibly be controlled. Thus, I conducted 1-way analysis of variance (ANOVA) tests for each hydrologic variable among the 6 flow classes. I then controlled for flow classes by using the residuals from the 1-way ANOVA tests in the hydrologic alteration model. My 18 predictor variables were classified as three types: one natural variable (drainage area), 7 dam regulation variables including total dam storage, dam density, and distance to nearest dam, and 10 hydrologic disturbance variables including freshwater withdrawals, road density, fragmentation, urbanization, and agriculture.

I used all predictor variables to build linear models to predict the same 40 hydrologic variables used in the Hydrologic Alteration section for all 401 stream records. I ran all possible regressions using 1 to a maximum of 4 predictor variables for each hydrologic response variable. I then determined the “best model” for each number of different predictor variables using Akaike Information Criterion values (AICc) (Burnham and Anderson 2002). I used four predictors to be conservative since AICc values will tend to reduce with increases in predictors despite very small increases in the overall variation explained. In addition, I then compared the number of predictors to Mallows Cp value for each model to determine if the number of predictors should be reduced to less than four variables (Mallows 1973). The model with the number of predictors that approaches Mallows Cp is considered the most appropriate model (Mallows 1973). After isolating the specific predictor variables in each “best” model, I used stepwise regression to determine how much variation each predictor variable explained.

### **4.3 Results**

Regulated stream classes had fairly broad representation across the region of interest had a similar geographical distribution as the unregulated streams (Fig. 4.3). Gages with adequate pre- and post-regulation information, on the other hand, were not adequately represented across all 6 flow classes, had very limited sample size (less than half of the regulated streams), and were generally clustered to individual drainage basins (Fig 4.3).

#### *Pre- and Post-Regulation Analysis*

The response of hydrologic variables to regulation was substantially different among the flow classes represented (Fig. 4.4). For example, for the base flow index and the annual

minimum, perennial run-off 1 (PR1) streams and coastal swamp & intermittent streams (CSI) showed positive changes whereas the stable high baseflow streams (SBF) showed negative changes. Similarly, PR 1 and CSI streams showed positive changes in flow predictability whereas SBF streams showed negative changes. In addition, some variables, such as the flood interval, showed highly variable responses, whereas other variables, such as the rise rate, showed similar responses across all flow classes. The responses to regulation were also variable *within* some of the flow classes. For example, streams within the stable high baseflow class showed variable responses in the low flow pulse count, high flow pulse count, and the flood interval in response to regulation.

#### *Hydrologic Disturbance Dataset*

I plotted regulated and unregulated streams according to their hydrologic disturbance index (HDI) values to evaluate the degree of hydrologic alteration between unregulated and regulated streams and within regulated streams (Fig. 4.5). HDI values were significantly higher in regulated streams compared to unregulated streams (Mann-Whitney Test,  $X^2=93.73$ ,  $p < 0.0001$ ). Hydrologic disturbance indices (HDI) for unregulated streams averaged 13.9 (stdev=5.13) and were dominated by HDIs in the “low” to “moderate” categories. However, several unregulated streams had HDIs in the “moderate to high” and a few in the “high” categories. HDI values for regulated streams averaged 20.0 (stdev=4.79) and were dominated by streams in the “moderate” to “high” HDI categories, but some regulated streams had HDIs in the “low to moderate” category.

#### *Hydrologic Alterations within Classes*

I evaluated the effect of dam regulation along with 3 other hydrologic disturbance variables for 40 hydrologic indices using general linear models for all streams and individual flow classes after controlling for drainage area (Table 4.2). Drainage area explained 0 to 73 % of the variation in hydrologic indices for unregulated streams depending on flow class and the individual hydrologic index ( $r^2$  adj) (Table 4.2). After controlling for drainage area, disturbance models explained, on average, 8 to 32 % of the variation in hydrologic indices depending on flow class (average refers to mean  $r^2$  for all 40 hydrologic models). However, disturbance models ranged from 0 to 69 % depending on flow class and depending on the hydrologic response

variable (Table 4.2) . In general, dam regulation explained more variation than any other variable. However, withdrawal, fragmentation, and urbanization explained a substantial amount of the overall variation, which at times, was higher than the variation explained by dam regulation.

The direction and magnitude of the t-statistic values for dam regulation varied substantially among flow classes for some hydrologic indices whereas other hydrologic indices showed consistent patterns across all flow classes (Fig 4.6-4.7). For example, intermittent flashy and perennial run-off streams showed positive changes in the base flow index with regulation whereas the stable high base flow streams showed negative changes (Fig. 4.6-4.7). In addition, the magnitude and direction of changes in various monthly flows and minimum/maximum flows were class-specific. In contrast, flow variability, rise rate, and the number of reversals all showed consistent negative changes across all flow classes.

Since I included other other potential hydrologic disturbance variables in models, I was able to compare the relative influence of dam regulation in comparison to urbanization, fragmentation, and withdrawals. Dam regulation had the largest and most consistent t-statistics relative to the other disturbance variables across flow classes (Fig. 4.8). However, urbanization showed large t-statistic values that generally had similar directionality to dam regulation. In contrast, withdrawals and fragmentation had smaller mean t-statistics, but generally showed opposite directionality relative to dam regulation and urbanization. Comparisons of the directionality and magnitude of the t-statistics for dam storage (continuous variable) and dam regulation (categorical variable) parameter estimates were very similar (Fig. 4.8).

#### *Overall Variation in Flow Dynamics of Regulated and Unregulated Streams*

I retained the first four principal components for all stream classes because the eigenvalues from random data exceeded the eigenvalues from the actual data at four components (Fig. 4.9). I plotted the first three principal components for all streams and for each individual flow class. Unregulated streams showed very close clustering where classes filled a small multivariate niche (Fig. 4.10). Regulated streams, however, showed more of a random structure where some flow classes had migrated into the multivariate space of others. For all streams in my dataset, four of the five key aspects of the natural flow regime (magnitude, timing, frequency, and duration) were represented by hydrologic indices with high loadings in the first 3

PCs (Fig 4.10). Flow classes showed substantial differences in the hydrologic indices that had high loadings. The grouping of regulated and unregulated streams also differed depending on class. In some classes, the influence of regulation was observed along one component whereas in others, it was observed along all three components (Fig. 4.11). For example, unregulated and regulated stable high baseflow 2 streams seemed to show major divergence on the basis of seasonal flow predictability whereas stable high baseflow 1 streams showed major divergence on the basis of the number of reversals, flow frequency, flow magnitude, and flow variability. The PCA for individual classes also isolated obvious outlier streams that show the most divergence or “disturbance”.

#### *Hydrologic Alteration Model*

I used 1-way ANOVA to control for the effect of flow classes on hydrologic indices. Flow classes explained, on average, 39% of the variation in hydrologic indices and ranged from 0 to 65% (1-way ANOVA,  $R^2$  adj.). After controlling for flow class, models, on average, explained only 15% of the variation in hydrologic indices and ranged from 3% to 55% ( $R^2$  adj.). Dam variables explained up to 24% of the overall variation as individual variables in models. Twenty-three of the 40 hydrologic alteration models had dam variables that explained more variation than any other individual variable. Of the remaining models, 8 had urbanization variables and 7 had drainage area that explained more variation than any other individual variable. Because of the large number of models, I only display models that explained 15% or more of the variation in hydrologic indices, which corresponds to a  $p < 0.0001$  (Appendix C). Monthly flow indices had poor representation in Appendix C whereas duration flows (minimum and maximum of various durations) had the greatest representation. Timing indices were not represented at all. Flow predictability had the most variation explained (55%), the majority of which was explained by drainage area (52%). One-day maximum flows and rise rate also had a significant amount of variation explained by the models (36% and 31%, respectively).

#### **4. 4 Discussion**

My results suggest that flow classes provide a framework for forming generalizations in how dam regulation affects flow. In essence, the central tendency of flow classes provide the starting point from which deviations in the natural flow regime can be measured. Poff et al.

(2007) grouped streams into 16 “hydro-regions” of the US as a framework to evaluate the influence of dams on the natural flow regimes of river systems. The study found that although there was an overall ‘homogenization’ of regional flow dynamics, hydrologic alterations caused by dams were region-specific. Likewise, I found that the magnitude and direction of the effects of dams on river flows are strongly influenced by flow class membership. Flow classes, similar to hydro-regions, should reflect climate, geography, and landscape characteristics (Chapter 3) and provide the basis for evaluating hydrologic alterations.

One of the differences in my study is that I did not limit my analyses to only pre/post regulation data, which would have reduced the sample size and resolution of my study. However, because I compared various drainages in analyses, I had to consider other factors that may confound my analysis, including other hydrologic disturbances. I found that other hydrologic disturbances, especially urbanization, can have equally strong influences that may compound or counter the hydrologic effects of dams. Hence, it is apparent that to form broad generalizations concerning certain hydrologic disturbances, the source(s) of hydrologic alteration must be isolated.

#### *Pre- and Post-Regulation Analysis*

One of the observations of this study is the disparity in the number of gages with adequate pre- and post-regulation data relative to the total number of regulated gauges. Before/after regulation analysis has dominated the literature concerning the effects of dam regulation on natural flow dynamics (Richter et al. 1996; Magillan and Nislow, 2001; Magillan and Nislow, 2005; Poff et al., 2007; Pyron and Neumann, 2008; Gao et al. 2009). However, gages with adequate pre-regulation data composed less than 50% of the regulated gages dataset and did not have adequate representatives in all flow classes. This suggests that only using gages with pre-regulation data to form generalizations may under-represent the overall variability and may limit the analytical power of finer-resolution analyses. Although not all flow classes were represented in my pre/post analysis, the 4 flow classes that were represented showed that dams affect river systems differently depending on their pre-existing natural flow regime.

#### *Hydrologic Disturbance Dataset*



The hydrologic disturbance index (HDI) provided an assessment of the cumulative hydrologic disturbances within each basin. Interestingly, I found that both regulated and unregulated rivers showed a large gradient of hydrologic alterations. Also, “pre-regulation” gages showed a variety of HDI values, which suggests that even some pre/post analyses may be confounded if studies do not account for other hydrologic disturbances besides dam regulation. Although unregulated streams were dominated by lower HDIs, several unregulated streams had HDIs in the more disturbed categories. This suggests that these few streams were highly disturbed. However, since all variables that compose the disturbance index are weighted equally, there is no structure that influences the relative importance of various disturbance factors. Thus, it is very feasible for unregulated streams to have higher HDIs than regulated streams. For example, the James River, VA is considered regulated because flow in one of its tributaries, the Jackson River, is controlled by Gathright Dam. However, total dam storage along with withdrawal, urbanization, and fragmentation is fairly small in the James River watershed, which leads to a low HDI value. In addition, the HDI does not take into account dam operation type, which may have profound influences on stream hydrology, regardless of total dam storage. Therefore, HDIs may not adequately address the extent of disturbance.

I removed some of the most disturbed “unregulated” streams from the dataset at the outset of the study because my main purpose was to evaluate hydrologic alterations due to dam regulation (although I had to consider the confounding effects of other disturbances). Thus, the highest HDI scores do not reflect the most disturbed systems in my region, but only the most “disturbed” in my dataset. My purpose in calculating the HDI for my streams was not to show that unregulated rivers are less disturbed than regulated rivers. In contrast, my purpose was to provide some evidence that there is a gradient of cumulative hydrologic alterations that should be accounted for in analyses that attempt to isolate the effects of individual disturbances. Instead of dodging these “confounding” disturbances by removing streams from my dataset, appropriate analyses should be conducted to increase the knowledge of how hydrologic alterations influence flow dynamics.

### *Hydrologic Alterations within Classes*

To isolate the effects of dam regulation, I had to control for other factors that may also explain variability in hydrologic indices, such as drainage area and watershed disturbance factors

through general linear models. Drainage area explained 0 to 73 % of the variation in unregulated streams, depending on the hydrologic index and flow class (Table 4.2). This suggests that my analyses could have been very biased without controlling for drainage area, since regulated rivers tend to be larger rather than smaller systems. My disturbance models included dam regulation, withdrawals, urbanization, and fragmentation and explained 0 – 69 % ( $R^2$  adj.) of the variation in hydrologic indices for all streams depending on the individual hydrologic index and flow class (Table 4.2). The ranges in  $R^2$  values suggests that some patterns in hydrologic indices are either not easily generalized or are not influenced by my specific predictor variables whereas other indices showed stronger patterns. Furthermore, this suggests that some prioritization can be made concerning which hydrologic variables to focus attention in hydrologic alteration studies (Gao et al. 2009).

Disturbance models showed that the magnitude and the direction of the influence of dam regulation on hydrology vary quite differently depending on the individual hydrologic index and the flow class (Fig 4.6-4.7). Perennial run-off 1 streams and the stable high baseflow streams showed the strongest affects of dam regulation; however, this may be associated with higher sample sizes in each of these classes. Across all classes, maximum flows, flow variability, rise rates, low flow durations, and flood intervals *generally* showed decreases whereas low-flow pulse counts, high-pulse variability, and reversals showed increases, some of which are similar to findings in other studies (Magillan and Nislow 2001 and 2005; Pyron and Neumann 2008; Poff et al. 2007) (Table 4.3). Thus, there are some broad generalizations that can be made concerning the influence of dams on river systems, despite large pre-existing differences in those river systems (Table 4.3). Interestingly, minimum flows generally show increases following dam regulation (Magillan and Nislow 2001 and 2005; Pyron and Neumann 2008; Poff et al. 2007). However, I found that the effect of dams on minimum flows varied depending on class (Table 4.3). For example, intermittent-flashy and perennial run-off 1 streams showed positive effects of dam regulation on minimum flow. Yet, the other classes were either impartial or showed strong decreases in minimum flow (stable high baseflow 2). Additionally, the effect of dam regulation on baseflows, predictability, and average monthly flows showed inconsistent results across flow classes, but showed stronger results within flow classes. Again, this suggests that rivers may be influenced differently by dams according to their pre-existing natural flow regimes. The fact that flow regimes may be “homogenized” by dam regulation (Poff et al. 2007) does not insinuate that

dams affect all rivers similarly. Rather, “homogenization” of flow regimes suggests that dams moderate or negate the natural processes responsible for the divergence of flow regimes (Poff et al. 2007). For example, intermittent-flashy and perennial run-off streams showed increases in the annual minimum whereas stable high baseflow streams show decreases. The result is that, for some individual hydrologic indices, flows within very different river systems appear more similar following dam regulation.

When compared to other disturbance factors, dam regulation had the largest and most consistent effect on flow across classes (Fig. 4.8). In general, dam regulation decreased flow variability, 1-day maximum flows, flood intervals, and rise rates whereas the frequency of low flows and reversals showed increases. Dam storage gave very similar results to the “regulated” vs “unregulated” classification and thus, could be used as a surrogate for dam regulation in general. I also found that other disturbance factors, at times, explained the majority of the variation in hydrologic indices (Table 4.2) and had strong influences on hydrologic variables (Fig. 4.8). In addition, urbanization may compound the effects of dam regulation while withdrawals and fragmentation tend to counter them. Thus, I believe that it is very important to isolate individual disturbances within basins and in order to understand how each may alter flow.

#### *Overall Variation in Flow Dynamics of Regulated and Unregulated Streams*

The results of the PCA suggested that dam regulation pushed the flowing environment outside the “bounds” of normal river function rather than “homogenizing” river flows. Furthermore, this would also suggest that the cumulative effects of dams on the multi-dimensional fluvial habitats creates environments to which endemic riverine biota are maladapted, which is a substantiated claim (Poff et al. 1997; Bunn and Arthington 2002). I assumed that in a multivariate analysis, such as PCA, the effects of “homogenization” would be manifested by regulated streams showing higher correlative structure and occupying a smaller multivariate space (i.e. less divergence) than their unregulated counterparts. However, I found that unregulated streams were actually highly correlated and filled a more confined multivariate space relative to regulated streams, which occupied a larger multivariate area with more random spread. Thus, in the multivariate sense, stream hydrology does not appear to be homogenized by dams. However, this may reflect the fact that I used 38 variables in the PCA instead of a fewer number of dominant hydrologic indices that exert a larger relative influence on river function and

habitats. Thus, if those “dominant” hydrologic indices tend to be stabilized by dam regulation, as in the case of maximum flows and rise rates, then in an ecologically meaningful sense, rivers may be homogenized by dams.

Similar to my evaluation of individual hydrologic indices, dam regulation affected the overall variability in flow differently depending on flow class. Not surprisingly, different hydrologic indices had the highest loadings for different flow classes. Thus, in terms of providing environmental flow standards for altered systems, it may be important to evaluate different subsets of hydrologic variables that are relevant to each flow class (Olden and Poff 2003). Another interesting aspect was that the PCA showed that some regulated rivers were embedded in the multivariate space of unregulated rivers whereas others showed extensive divergence or “disturbance”. Examining the multivariate structure of flow dynamics may provide a framework to isolate systems that are the most altered, which should have implications for ecological relationships and management. For example, systems that show large hydrologic alterations may also display major shifts in fish or macroinvertebrate assemblages (Bunn and Arthington 2002). However, one limitation of my analysis is that the disturbances responsible for the divergence in some of the regulated streams may have been induced by other factors besides dam regulation. Thus, other appropriate analyses, such as model building, are needed to separate confounding effects of various watershed disturbances.

#### *Hydrologic Alteration Model*

Since many of the effects of hydrologic disturbances are specific to individual flow classes, the flow classes themselves can be used as a starting point from which deviations in hydrologic indices can be measured. Flow classes alone, on average, explained 39% of the variation in hydrologic indices. Thus, I controlled for flow class by using the residuals of 1-way ANOVAs of hydrologic indices among classes. Another option could have been to incorporate flow class as a variable in general linear models, which would have achieved similar results but this also would have resulted in complex models with a higher number of predictor variables. One limitation of my approach was that I did not include interactions between certain classes and hydrologic disturbance variables, which could differ greatly depending on flow class. For example, I used all 18 disturbance variables to predict the baseflow index and the model only explained 12% of the variation ( $R^2$  adj) (results not shown). I added a dam storage\*flow class

interaction term and the model explained 24% of the variation in baseflow index ( $R^2$  adj), 14% of which was explained solely by the interaction term. Many of the disturbance variables may show significant interaction effects; however, adding class interactions to 18 variables would result in very complex models. Another way of handling this would be to run models within each class; however, with a high number of predictors, the sample size within each class may limit such analyses. Nonetheless, my purpose was not to explore the intricacies of model construction or model complexity but to provide evidence that flow classes can be incorporated into model structure and may provide a foundation for generalizing how disturbances influence flow. One other limitation of my study was that I did not explore the influence of various types of dam regulation (e.g. flood control, hydropower, etc). This information is available for individual dams in the US through the National Dam Inventory database (USACE 2011) and would certainly be an informative for management relevant to dam operations.

My hydrologic alteration models were constructed from 18 total variables using only the ‘best’ variables. On average, my models only explained 15% of the variation in hydrologic indices (range: 3%-55%,  $R^2$  adj.) (Appendix C). Interestingly, in the “hydrologic alteration within classes” section, I developed linear models for each individual class to isolate the effects of dam regulation by using only four disturbance variables (dam regulation, withdrawals, urbanization, and fragmentation) (Table 4.1). The models for individual classes explained on average 9 to 32 % of the variation depending on class compared to 15%, on average, for the hydrologic alteration models. In my hydrologic alteration models, dam variables explained up to 24% of the overall variation as individual variables in models whereas dam regulation explained up to 39% of the overall variation in the individual class models (Table 4.2). Once again, this suggests that separating analyses by flow class or incorporating flow class membership into the structure of models can aid in generalizing patterns.

Although models explained substantial variation for some indices, the poor predictive ability disturbance model and hydrologic alteration models for other hydrologic indices may indicate that the model structure may have been inappropriate given the data (i.e. non-linear relationships). For example, Carlisle et al. (2010a) built regression trees using climate, geologic, soil, topographic, and geographic variables to predict hydrologic indices across the US. The trees showed very low error when predicted values were compared to observed values compared to static classifications, which suggested that hierarchical structure may have increased model

predictive power. However, linear regression models have been used predict hydrologic indices and have explained substantial amounts of variation in response variables in various regions (DeWalle et al. 2000; Sanbord and Bledsoe 2006; Mohamoud 2008; Zhu and Day 2009). Thus, another potential source of unexplained variability was the exclusion of factors that may have been important in predicting hydrologic indices. I used flow classes as a stratification to control for climate, geography, topography, and geomorphology and then developed disturbance linear models separately for each class. Similarly, Sandbord and Bledsoe (2006) stratified basins in Colorado by major differences in flow regime and geography and developed separate linear regressions predicting streamflow metrics for each stratification. However, climate, geomorphology, and soil factors may exert various localized controls on hydrology, depending on regional affiliation (Mohamoud 2008). Thus, including “natural” factors in hydrologic alteration models may have increased their predictive abilities. One limitation, however, was that the number of predictors in models were limited given the sample size in each class.

Patterns in hydrologic variables indicate that some disturbances influence some hydrologic variables more than others. Similar to dam disturbance, urbanization or impervious surfaces were also present in many of the hydrologic alteration models. Interestingly, low flows, such as July flows and minimum flows, were influenced more by urbanization disturbances whereas maximum flows, rate of change, flow durations, and flow variability were more influenced by dam disturbances. The ranges in the  $R^2$  values of hydrologic alteration models suggest that some patterns in hydrologic indices may be more easily generalized than others, especially those at the extremities of the hydrograph (minimum, maximum flows). Similarly, Gao et al. (2009) isolated a few representative indicators out of the 32 IHA variables that explained the majority of the variation in hydrologic alterations.

## **4.5 Conclusions**

The framework I have presented for generalizing patterns in hydrologic alteration may be more important than the results discussed herein. The approach that I have outlined could provide a framework to form accurate generalizations on the affect of dam regulation on stream flows because it is not limited to only pre- and post-regulation analyses. This approach also can provide environmental flow standards for regulated systems that lack sufficient pre-disturbance hydrologic information. In addition, the framework could be modified to predict hydrologic

alterations in basins without any discharge information. The ability to create frameworks that incorporate river systems with insufficient or no hydrologic information is critical to increasing the resolution of relationships between ecology and altered hydrology (Knight et al. 2008).

Scientists and managers benefit from organizing data into pieces of digestible information. Natural flow classes represent differences in flow that are the result of climate, geographical, and landscape-driven processes, which may provide a suitable framework for organizing information into more meaningful analyses (Chapters 2-3, McManamay et al. 2011). Similar to Poff et al. (2006b), I envision that this framework can be used as a hierarchical approach. For example, in analysis evaluating hydrologic alterations, flow classes may provide the primary structure, followed by a geomorphological stratification (Poff et al. 2006b), which is then further stratified into regulation type (dam or other hydrologic disturbance types).

Overall, dam regulation exerted the strongest and most consistent influences on flow dynamics compared to other disturbance variables. However, for some hydrologic indices, other disturbances in a basin may compound or counter the influences of dam regulation. Analyses should isolate the various contributors to overall hydrologic alteration. Information and large datasets concerning hydrology and landscape-scale variables are now more readily available than ever (e.g. Falcone et al. 2010, Wollock et al. 2004). Thus, there is great potential to understand general patterns in hydrologic alterations across the landscape.

Table 4.1. Hydrologic indices include the 33 Indicators of Hydrologic Alteration (IHA) (Richter et al. 1996), 3 Environmental Flow Component (EFC) indices (Mathews and Richter 2007), and 8 indices used in Poff (1996).

|                               |                                      |                             |  |
|-------------------------------|--------------------------------------|-----------------------------|--|
| Flow variability <sup>1</sup> | November flow                        | 7-day minimum               | Flood duration*                                      |
| January flow                  | December flow                        | 30-day minimum              | Flow predictability <sup>1</sup>                     |
| February flow                 | Minimum July flow <sup>2†</sup>      | 90-day minimum              | Seasonal flood predictability <sup>1</sup>           |
| March flow                    | Base flow index                      | Low flow duration           | Seasonal predictability (lowflow) <sup>1*</sup>      |
| April flow                    | Low pulse count                      | No. of zero flow days†      | Seasonal predictability (non-low flow) <sup>1*</sup> |
| May flow                      | Low pulse variability <sup>2†</sup>  | 1-day maximum               | Seasonal predictability (non-flooding) <sup>1*</sup> |
| June flow                     | High pulse count                     | 3-day maximum               | Date of annual minimum                               |
| July flow                     | High pulse variability <sup>2†</sup> | 7-day maximum               | Date of annual maximum                               |
| August flow                   | Flood frequency <sup>1†</sup>        | 30-day maximum              | Rise rate  |
| September flow                | 1-day minimum                        | 90-day maximum              | Fall rate  |
| October flow                  | 3-day minimum†                       | Flood interval <sup>1</sup> | Reversals  |

<sup>1</sup> Poff (1996); <sup>2</sup> EFC (Mathews and Richter 2007), † used in hydrologic alteration model but not PCA, \* used in PCA but not hydrologic alteration model.



Table 4.2. Results of the linear regressions to control for differences in drainage area size and results of the hydrologic disturbance models constructed for all streams and the six flow classes created by McManamay et al. (2011) for this study's region of interest. Disturbance models were constructed using dam regulation, withdrawal, fragmentation, and urbanization. Bold numbers indicate the average  $R^2$  or  $R^2$  adj value for all 40 hydrologic indices. Numbers in parentheses indicate the range in  $R^2$  or  $R^2$  adj values (single numbers represent the maximum and indicate that the minimum value was 0). For drainage area models, n refers to the number of unregulated streams in all flow classes or those found within each respective flow class. For disturbance models, n refers to the sample size of all unregulated and regulated streams found in all classes and within each respective class.

| Model                |           | All                          | Coastal &<br>Swamp<br>Intermittent | Intermittent<br>Flashy       | Perennial<br>Run-off 1 | Perennial<br>Run-off 2       | Stable High<br>Baseflow 1    | Stable High<br>Baseflow 2    |
|----------------------|-----------|------------------------------|------------------------------------|------------------------------|------------------------|------------------------------|------------------------------|------------------------------|
| Drainage Area        | n         | 284                          | 22                                 | 19                           | 80                     | 71                           | 37                           | 55                           |
|                      | $R^2$     | <b>0.04</b><br>(0.44)        | <b>0.29</b><br>(0.74)              | <b>0.06</b><br>(0.28)        | <b>0.11</b><br>(0.68)  | <b>0.04</b><br>(0.70)        | <b>0.05</b><br>(0.47)        | <b>0.10</b><br>(0.67)        |
|                      | $R^2$ adj | <b>0.04</b><br>(0.44)        | <b>0.26</b><br>(0.73)              | <b>0.03</b><br>(0.24)        | <b>0.10</b><br>(0.67)  | <b>0.03</b><br>(0.69)        | <b>0.04</b><br>(0.45)        | <b>0.08</b><br>(0.66)        |
| Disturbance Model    |           |                              |                                    |                              |                        |                              |                              |                              |
|                      | n         | 401                          | 26                                 | 30                           | 111                    | 84                           | 54                           | 96                           |
|                      | $R^2$     | <b>0.10</b><br>(0.01 - 0.26) | <b>0.23</b><br>(0.05 - 0.52)       | <b>0.41</b><br>(0.08 - 0.73) | <b>0.11</b><br>(0.34)  | <b>0.14</b><br>(0.01 - 0.38) | <b>0.23</b><br>(0.06 - 0.54) | <b>0.18</b><br>(0.02 - 0.41) |
|                      | $R^2$ adj | <b>0.09</b><br>(0.26)        | <b>0.11</b><br>(0.43)              | <b>0.32</b><br>(0.69)        | <b>0.08</b><br>(0.32)  | <b>0.09</b><br>(0.35)        | <b>0.17</b><br>(0.50)        | <b>0.15</b><br>(0.39)        |
| Individual Variables |           |                              |                                    |                              |                        |                              |                              |                              |
| Dam Regulation       | $R^2$     | <b>0.04</b><br>(0.18)        | <b>0.04</b><br>(0.19)              | <b>0.11</b><br>(0.34)        | <b>0.06</b><br>(0.22)  | <b>0.03</b><br>(0.23)        | <b>0.13</b><br>(0.40)        | <b>0.12</b><br>(0.39)        |
| Withdrawal           | $R^2$     | <b>0.02</b><br>(0.06)        | <b>0.04</b><br>(0.20)              | <b>0.05</b><br>(0.24)        | <b>0.01</b><br>(0.05)  | <b>0.03</b><br>(0.07)        | <b>0.04</b><br>(0.23)        | <b>0.03</b><br>(0.11)        |
| Fragmentation        | $R^2$     | <b>0.03</b><br>(0.13)        | <b>0.08</b><br>(0.35)              | <b>0.04</b><br>(0.28)        | <b>0.01</b><br>(0.09)  | <b>0.01</b><br>(0.05)        | <b>0.04</b><br>(0.16)        | <b>0.03</b><br>(0.12)        |
| Urbanization         | $R^2$     | <b>0.02</b><br>(0.06)        | <b>0.05</b><br>(0.20)              | <b>0.13</b><br>(0.34)        | <b>0.02</b><br>(0.16)  | <b>0.03</b><br>(0.10)        | <b>0.03</b><br>(0.11)        | <b>0.02</b><br>(0.05)        |

Table 4.3. General trends of the influence of dams on stream hydrology found in literature, across all streams found in this study, and specific to each class. Decrease and increase indicate the direction of the influence of dams on each hydrologic variable. All variables included had a t statistics with  $p < 0.05$ .

| Entire sample or Class                       | Decrease   | Increase   |
|--|--|--|
| Generalizations from literature <sup>1</sup> | maximum flows, flow variability, rise/fall rates   | minimum flows, reversals   |
| All classes (this study)                     | maximum flows, flow variability, rise rate, low flow duration, flood interval                  | reversals, low flow pulse counts, high pulse variability   |
| Intermittent-flashy                          | spring flows   | minimum flows, baseflow index, flow predictability, flood frequency  |
| Perennial Run-off 1                          | February flow, seasonal flood predictability   | minimum flows, baseflow index, fall flows, June flow, flood frequency, date of annual maximum, flow predictability |
| Perennial Run-off 2                          | low pulse variability, flow predictability   | high pulse count, seasonal flood predictability  |
| Stable High Baseflow 1                       | winter/spring flows, flow predictability   | summer/fall flows, flood frequency   |
| Stable High Baseflow 2                       | winter/spring flows, minimum flows, baseflow index, low pulse variability, flow predictability | high pulse count, flood frequency, seasonal flood predictability   |

<sup>1</sup> Richter et al. 1996; Magillan and Nislow 2001&2005; Pyron and Neumann 2008; Poff et al. 2007

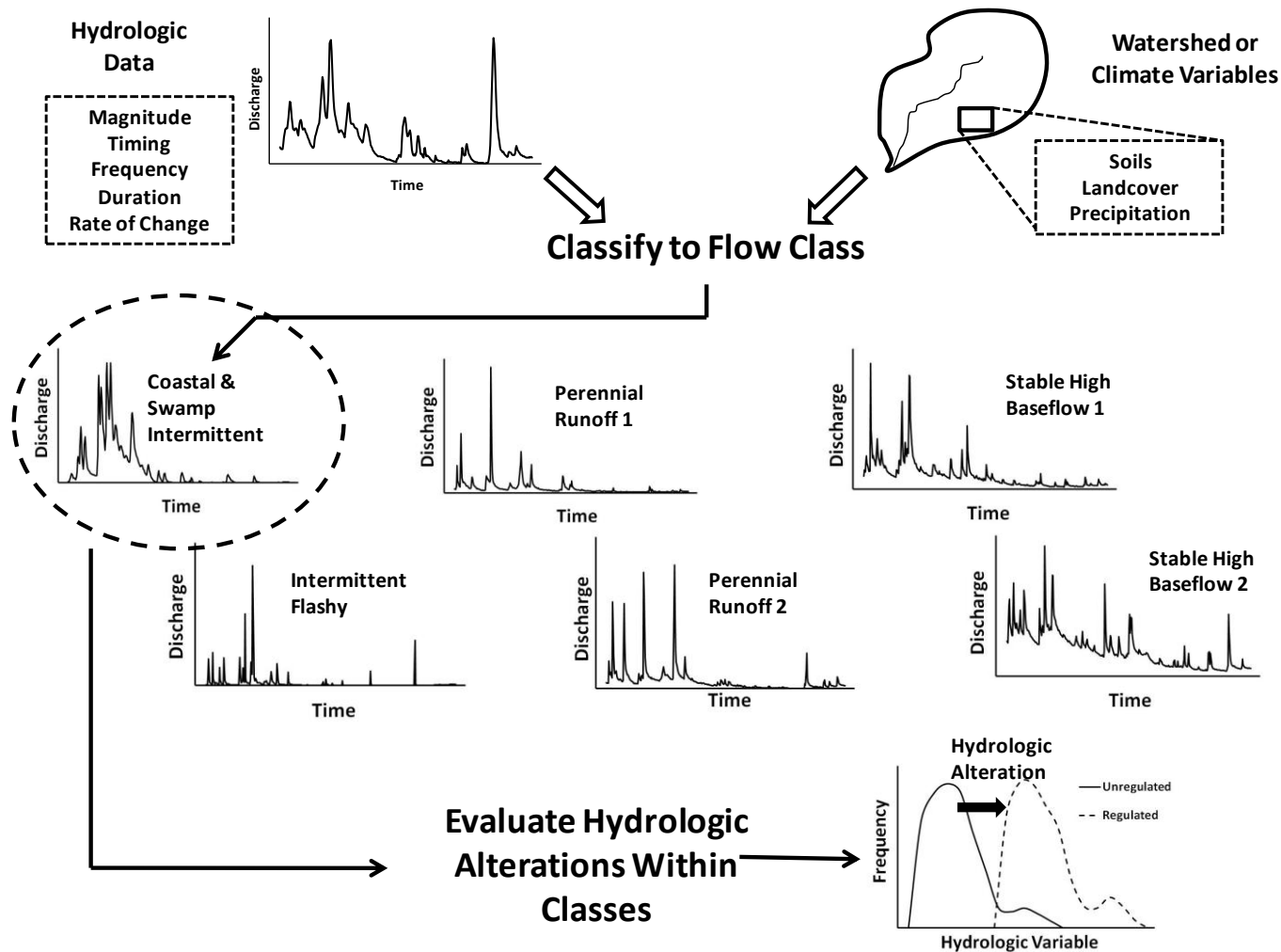


Figure 4.1 Conceptual model of a possible approach to assessing hydrologic alterations within basins. Arrows indicate steps in the approach. First, a disturbed river is classified to a particular flow class using pre-disturbance hydrologic data (if available) or landscape characteristics measured at the watershed scale. Once a regulated system has been ‘assigned’ to a flow class, its hydrology can then be compared to unregulated rivers in the same flow class. Thus, the magnitude and direction of hydrologic alterations are assumed to be relative to river flow-class membership. Basic principles and theory behind the conceptual model were taken from Arthington et al. (2006)

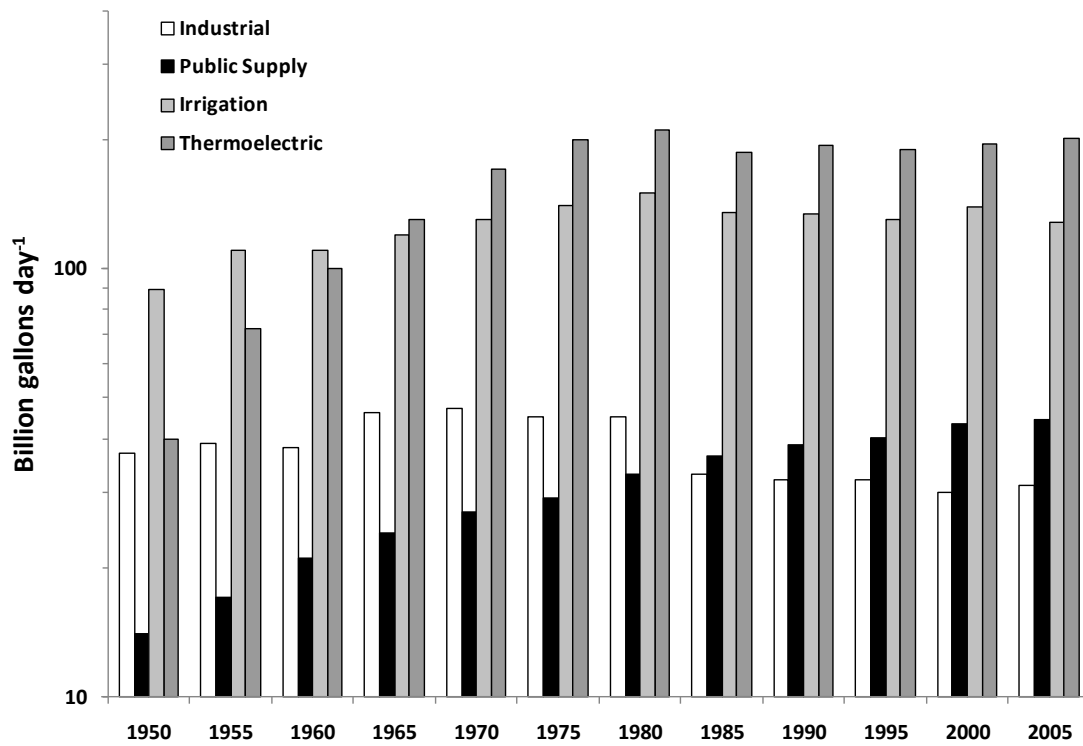


Figure 4.2 Freshwater withdrawals according to various consumption categories from 1950 to 2005. Data taken directly from the 2005 USGS Water-Use Report (Kenny et al. 2009)

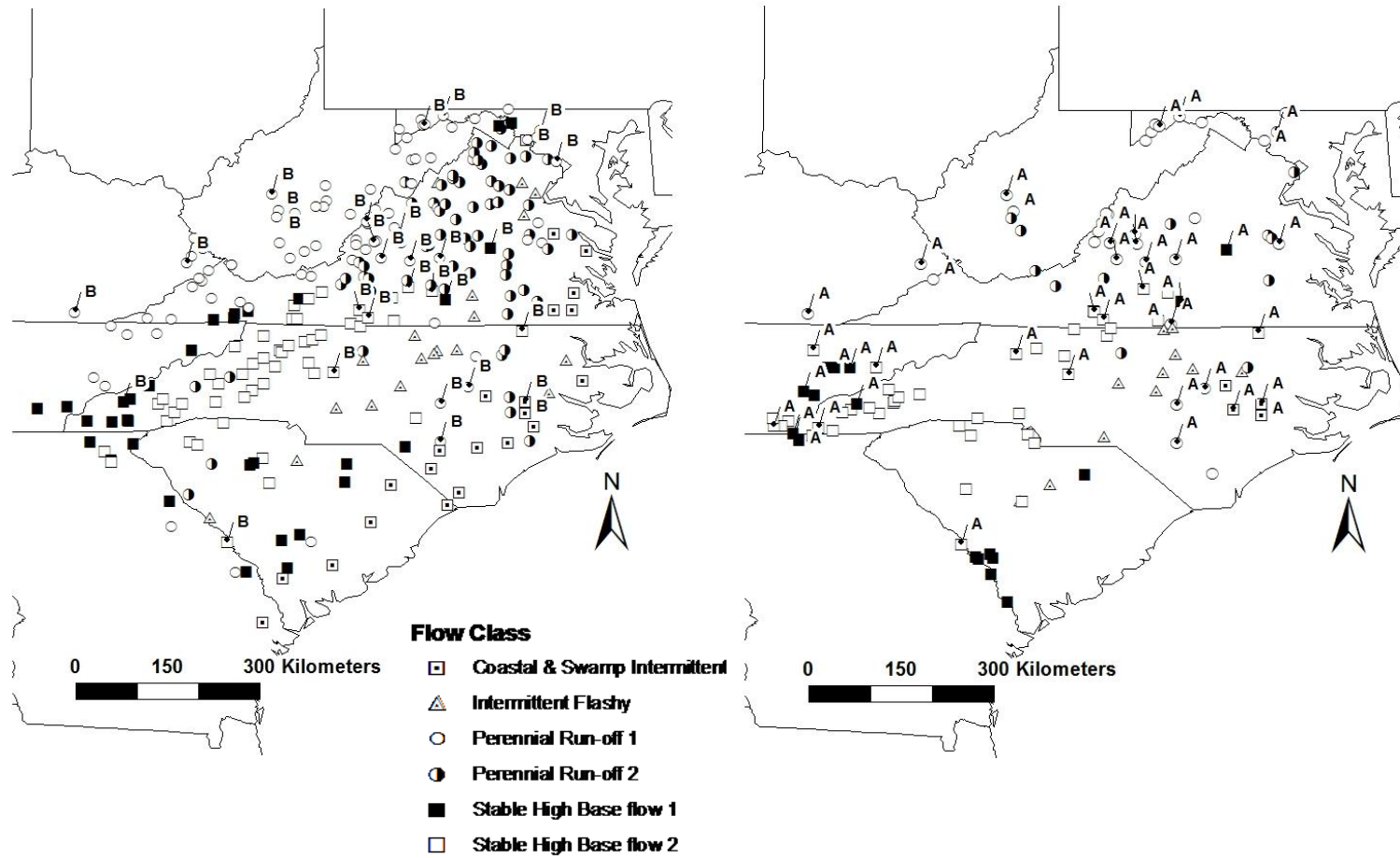


Figure 4.3 Unregulated and regulated rivers found in my study area in their respective natural flow classes created by McManamay et al. (2011) for the region of interest. Gages marked with “B” or “A”s represent gages that have pre- and post-regulation gage information. “B” indicates that the point represents gage data taken from pre-regulation time periods whereas “A” represents post-regulation gage data.

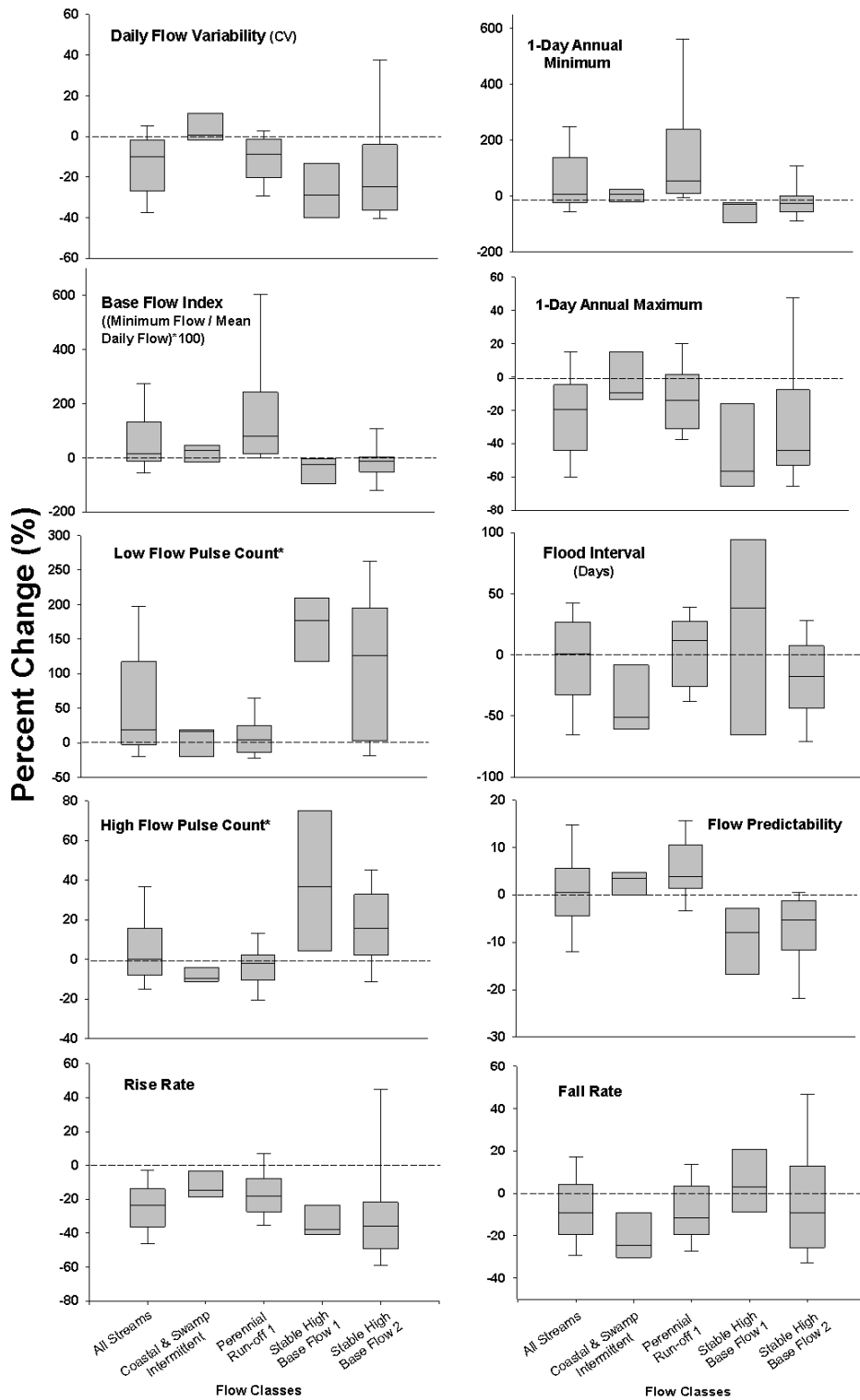


Figure 4.4 Percent changes in 10 hydrologic indices following dam regulation for all streams (n=49) and for streams within four of the six flow classes created by McManamay et al. (2011) for the 8-state study area. Sample sizes: Coastal & Swamp Intermittent (n=3), Perennial Run-off 1 (n=27), Stable High Base Flow 1 (n=7), Stable High Base Flow 2 (n=12).

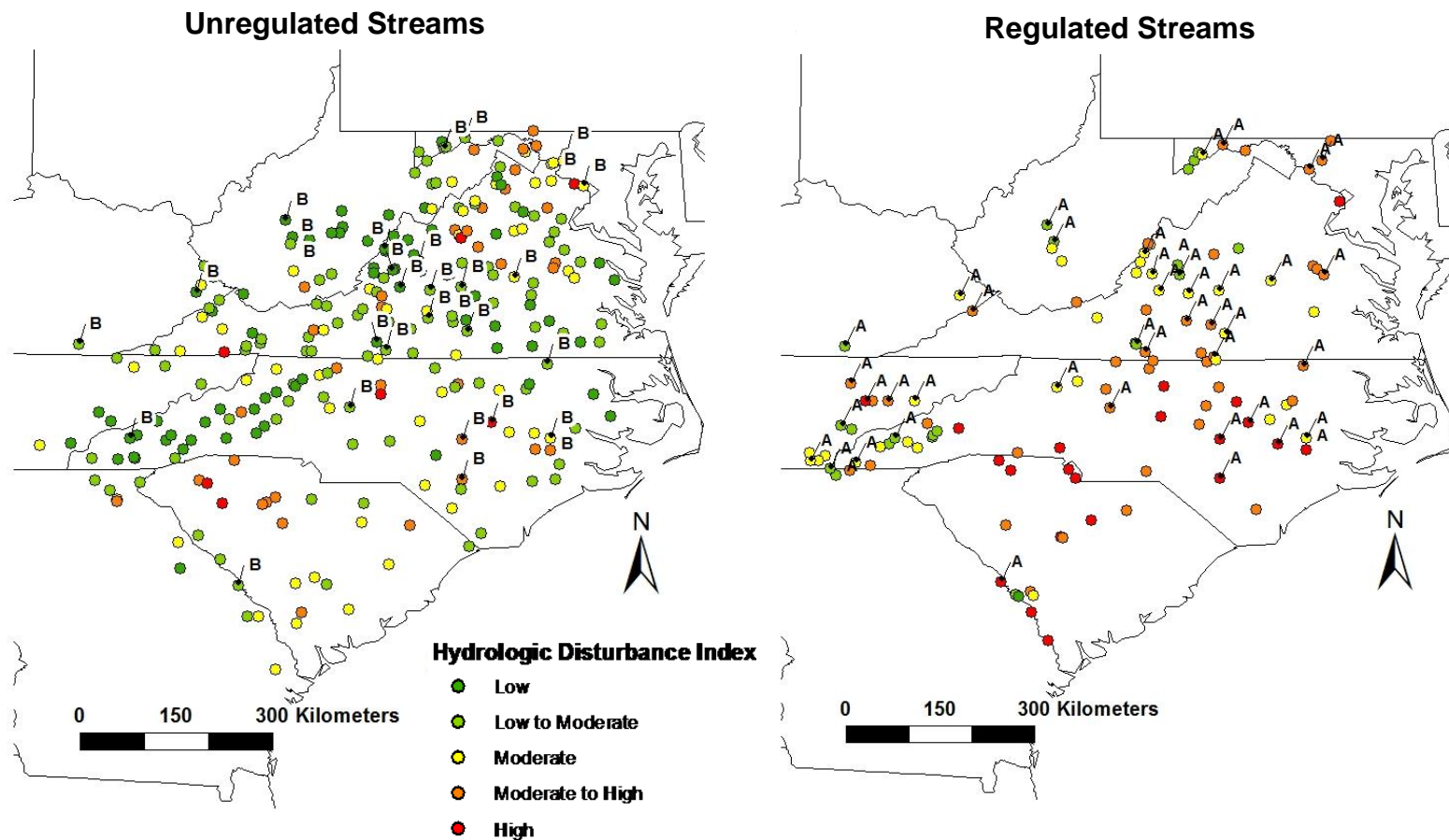


Figure 4.5 Hydrologic disturbance index of unregulated and dam-regulated streams found in the 8-state study area. The hydrologic disturbance index (HDI) was based on total dam storage, total freshwater withdrawals, urbanization, and fragmentation within each basin. Gages marked with a “B” indicates that the point represents data taken from pre-regulation time periods whereas “A” represents post-regulation data.

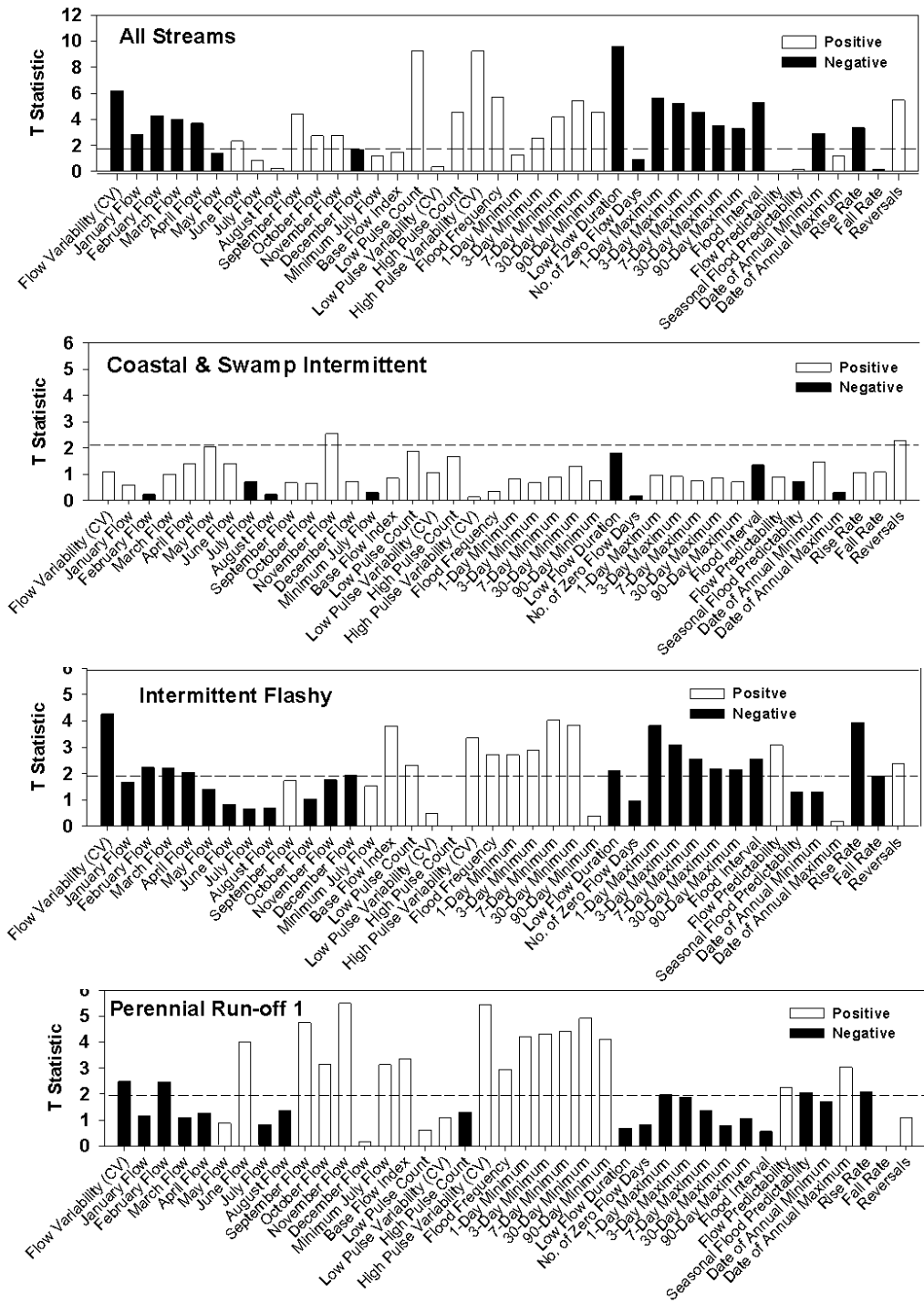


Figure 4.6 T statistics of the dam-regulation parameter estimate in 40 hydrologic alteration models for all streams and within three of the six flow classes created by McManamay et al. (2011). Hydrologic alteration models were general linear models constructed to predict hydrologic indices using four disturbance variables: dam regulation, withdrawals, urbanization, and fragmentation variables. Positive effects of dam regulation are represented in white bars whereas black bars represent negative effects of dam regulation. Dashed line indicates the significance of the t-statistic for each hydrologic index.



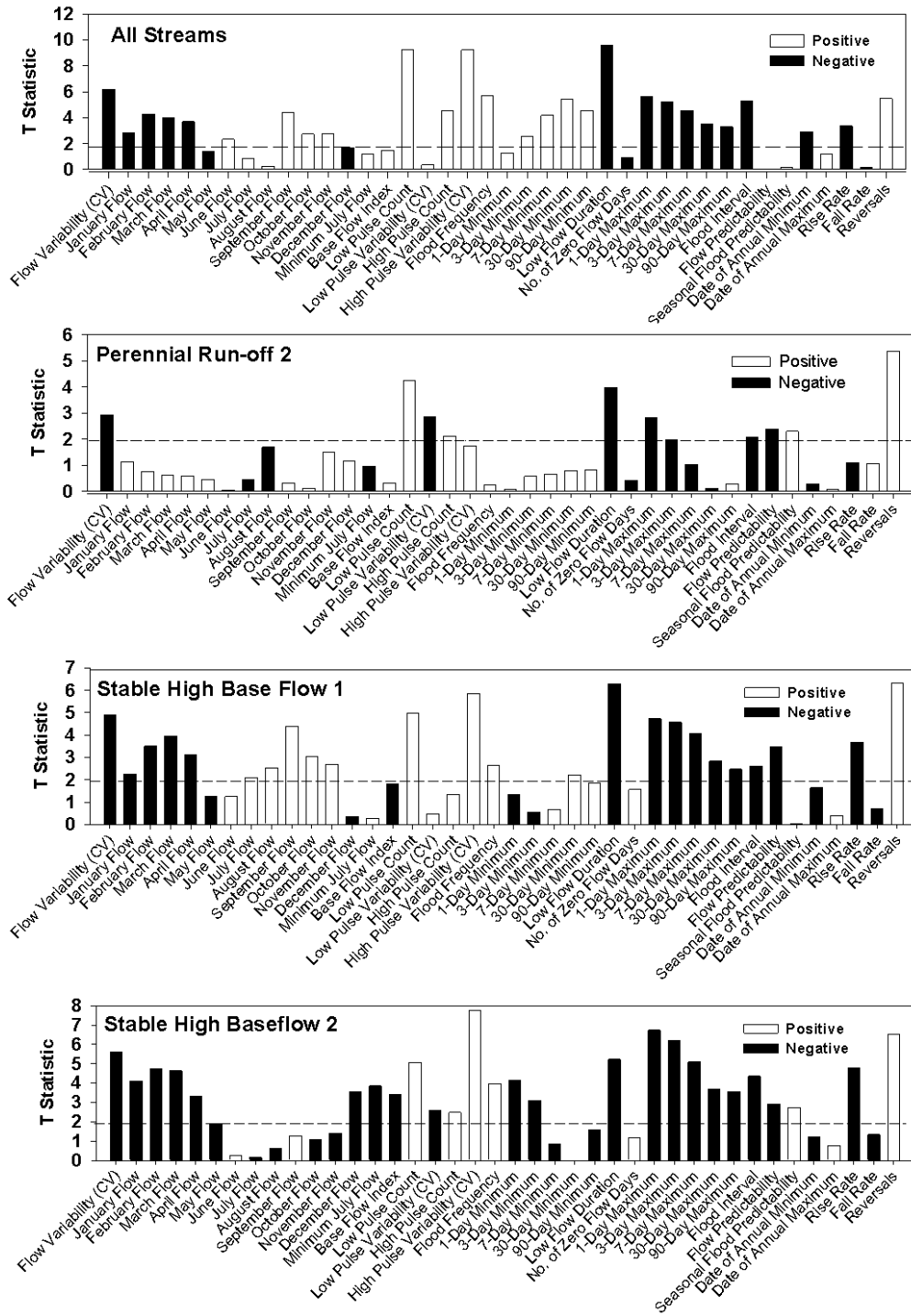


Figure 4.7 T statistics of the dam-regulation parameter estimate in 40 hydrologic alteration models for all streams and within three of the six flow created by McManamay et al. (2011). Hydrologic alteration models were general linear models constructed to predict hydrologic indices using four disturbance variables: dam regulation, withdrawals, urbanization, and fragmentation variables. Positive effects of dam regulation are represented in white bars whereas black bars represent negative effects of dam regulation. Dashed line indicates the significance of the t-statistic for each hydrologic index

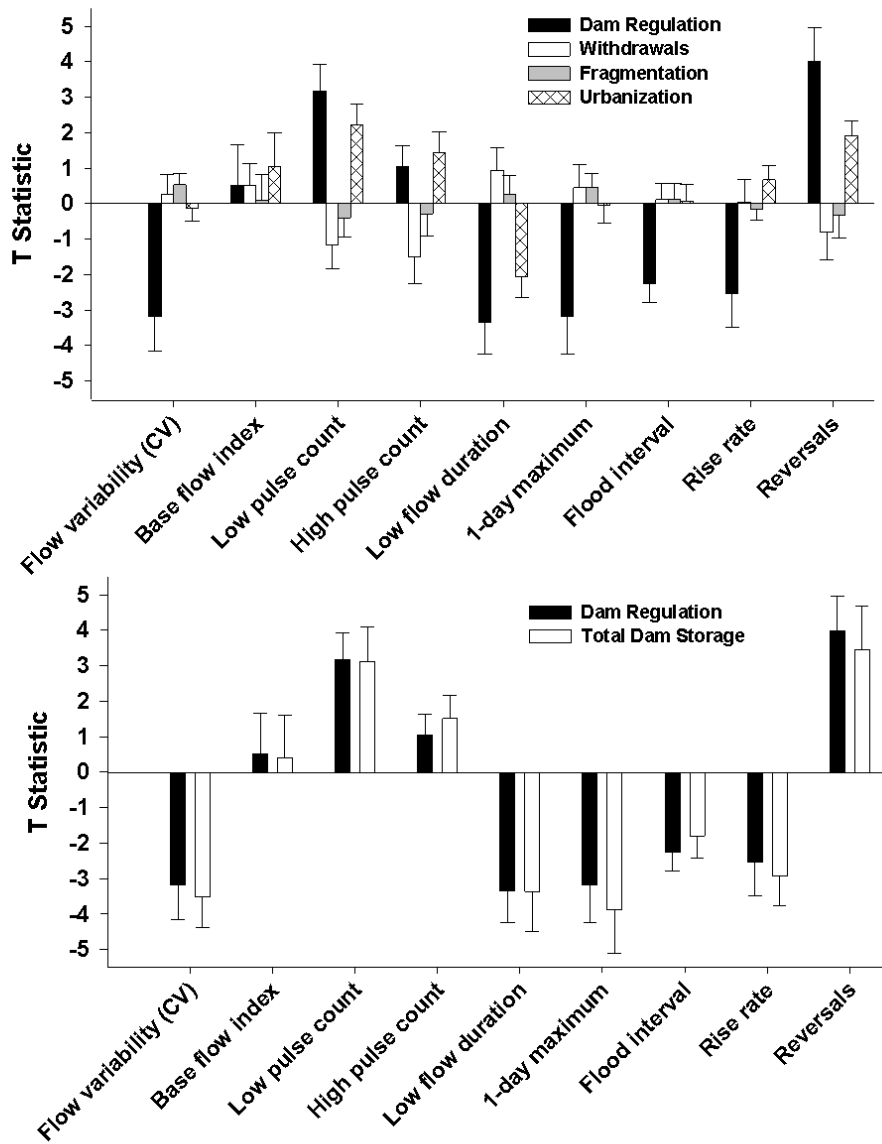


Figure 4.8 Comparisons of average t-statistics from hydrologic alteration models of parameter estimates for dam regulation, withdrawals, urbanization, and fragmentation averaged across flow classes created by McManamay et al. (2011) (top graph). Comparisons of the average t-statistics from hydrologic alteration models of parameter estimates for models run using dam regulation as a categorical variable or with total dam storage as a continuous variable across all flow classes (bottom graph). Positive t-statistic values indicate positive effects of each disturbance variable whereas negative values indicate negative effects of each disturbance variable. Error bars represent 1 standard error.

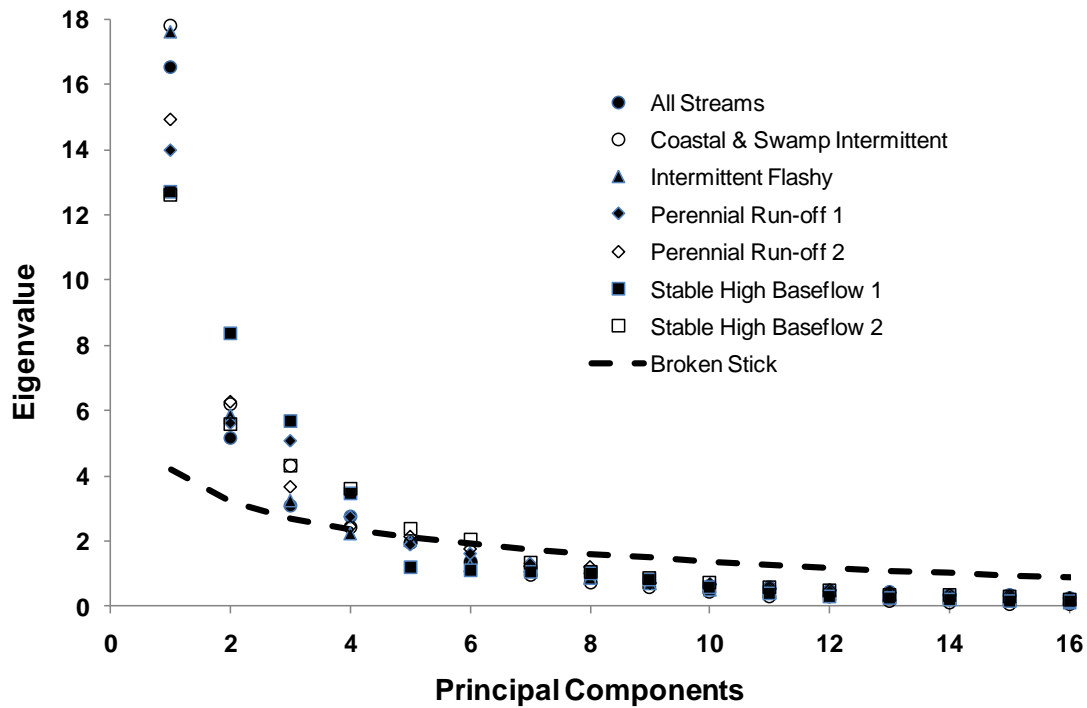


Figure 4.9 Scree plot of eigenvalues versus number of principal components for PCA analyses conducted for all streams and for each flow class and for the broken-stick model. Flow classes created by McManamay et al. (2011).

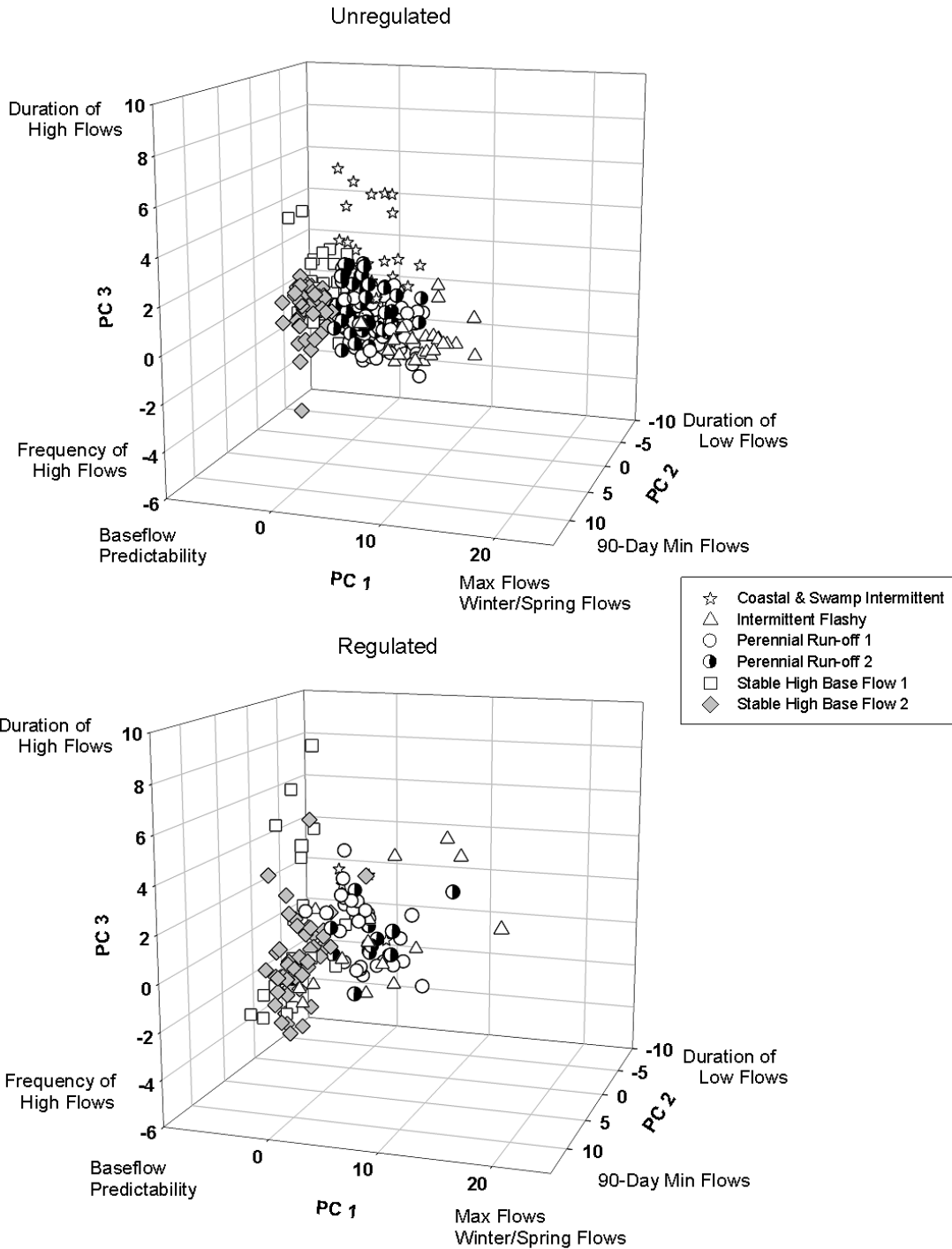


Figure 4.10 Three-dimensional scatterplots of unregulated and regulated streams and their respective flow classes, plotted along the first 3 principal components. Hydrologic indices with highest loadings are labeled on each of the principal component axes. Flow classes created by McManamay et al. (2011).

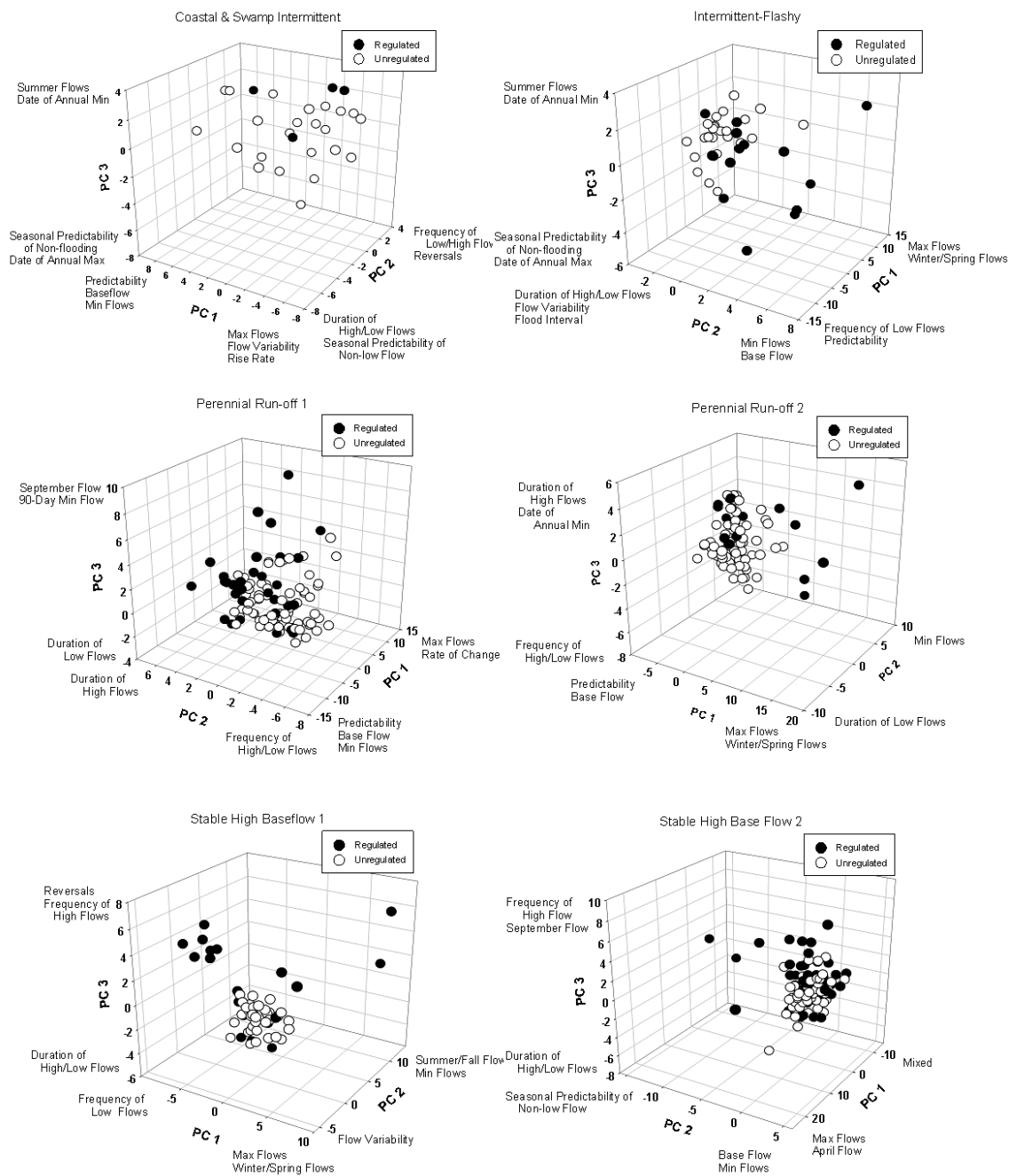


Figure 4.11 Three-dimensional scatterplots of principal component analyses for unregulated and regulated streams within each of the six flow classes created by McManamay et al. (2011). Streams were plotted along the first 3 principal components. Hydrologic indices with highest loadings are labeled on each of the principal component axes.

## **5. Providing a Flow and Morphology Restoration Template for Fish Assemblages in the Upper Tennessee River System.**

### **Abstract**

Restoring fish assemblages in regulated river systems should be based upon the current understanding of how flows, morphology, and temperature interact within a larger landscape-level disturbance context (i.e. habitat fragmentation). Recent scientific literature suggests that the development of quantitative and transferable flow-ecology relationships is needed in order to prescribe environmental flow standards for restoring disturbed river systems. However, the existing frameworks to develop these relationships do not take into account other potentially confounding factors such as temperature and morphology, all within the context of landscape disturbances. The purpose of this study was to determine the relative importance of flow, morphology, and temperature in explaining patterns in fish assemblages relative to landscape-scale disturbances (i.e. land-use, fragmentation) within the upper Tennessee River Basin. Fish assemblage sampling data was collected from 22 regulated and 28 unregulated sites within moderate to large, coolwater river systems. I assembled 19 predictor variables from several categories: landscape disturbance variables (e.g. land use, fragmentation) (5), channel morphology (4), substrate (3), flow (6), and temperature. I categorized regulated sites into 3 dam operation types to explore if any generalizations could be made concerning how specific operations affect flow, morphology, and temperature, and in turn, fish assemblages. I hypothesized that dam operation would influence numerous physical variables (flow in addition to channel morphology, substrate, and temperature), all of which could influence the structure of fish assemblages. Results of my analyses showed that diversion-bypass and bottom-release (hypolimnetic) operations had the most substantial influences on flow, morphology, substrate, and temperature and showed dramatic reductions in species richness relative to basin size compared to unregulated and other regulated systems. Linear models explained, on average, 48% of the variation in biotic response variables and ranged from 24% to 64%. Morphology, land-use, fragment length (river km distance from upstream to downstream dam), and substrate had the highest frequencies as predictors in linear models, in that order. Fragment length tended to explain the most variation in models followed by land-use, bankfull width, gradient, and temperature. Flow variables were poorly represented in models. My results suggest that if

current regulated river restoration is dependent upon the development of flow-ecology relationships, then the interaction of flow with morphology and temperature within a landscape disturbance context should be taken into account in future analyses.

## **5.1 Introduction**

River systems are a continuous matrix of lateral, longitudinal, and vertical processes that shape the habitats in which stream fish complete their life history requirements (Fausch et al. 2002). Viewing river habitats as the outcome of cumulative landscape processes influences how scientists and managers view species recovery and management (Ward et al. 2001). Because of the complexity of river systems, restoring the essential habitat requirements for fish within regulated systems will have to be based upon science conducted at scales that capture the spatial and temporal variability of riverscapes (Fausch et al. 2002).

Of the many impacts of dams on river systems, the most documented effects have been reductions in natural flow variability (Magillan and Nislow 2001 and 2005; Pyron and Neumann 2008; Poff et al. 2006a; Poff et al. 2007, Chapter 4). Dams have altered the magnitude, frequency, duration, timing, and rate of change in flow events, which has homogenized flow variability across geographic scales (Poff et al. 2007) and led to the loss of certain flow components that are responsible for sustaining native flora and fauna (Bunn and Arthington 2002). Flow variability has been shown to organize fish assemblages (Poff and Allan 1995; Herbert et al. 2003; Pyron and Lauer 2004). Not surprisingly, extensive literature suggests that hydrologic alterations can cause changes in endemic fish assemblages (Pringle et al. 2000; Roy et al. 2005; Freeman and Marcinek 2006; Poff and Zimmerman 2010; Carlisle et al. 2010b).

Although the impact of disturbances on flow variability has recently taken front stage (Poff et al. 2010), there are morphological consequences of dam regulation on river systems that should be considered. Dams tend to decrease floodplain-inundation (Nislow et al. 2002), decrease bankfull area, decrease lateral migration, and increase riparian encroachment (Gordon and Meentemeyer 2006). Dams also trap and store sediments within impoundments, leaving the downstream river channel gravel-starved (Petts 1979; Milhous 1982; Kondolf 1997; Renwick et al. 2005; Schmidt and Wilcock 2008). The lack of sediment inputs causes channel degradation and armoring (Konfolf 1997), which may destroy fish spawning and refuge habitat. However,

after significant sediment inputs from tributaries further downstream, the channel may aggrade. Similar to flow variability, morphological variables, such as gradient, channel morphology, and substrate, have also been shown to structure fish assemblages (Sutherland et al. 2002; Dauwalter et al. 2008; D’Ambrosio et al. 2009). However, very little literature directly links the morphological effects of dam regulation to fish assemblages.

The extensive degradation of dam regulation will require process-driven restoration in order for fish assemblage recovery to become a reality (Ward et al. 2001; Roni et al. 2008). Restoring aspects of the natural flow regime has shown some promise for biotic recovery (Richter et al. 2003). For example, increases in minimal flow or flow variability of impounded rivers have shown increases in the abundance of fish occupying fast-flowing habitats (eg. fluvial specialists, darters, minnows) and decrease the abundance of generalists (eg. centrarchid species) (Travnichek et al. 1995; Marchetti and Moyle 2001; Brown and Ford 2002; Lamouroux et al. 2006; Sabaton et al. 2008). Mimicking the amplitude of natural spring snowmelts in a regulated river has been shown to enhance native fish recruitment, thus increasing density (Propst and Guido 2004). Morphological restoration (i.e. substrate restoration) below dams has also benefitted fish. For example, gravel additions below dams have been conducted in the western US to enhance salmonid spawning habitat and have shown some positive responses (Kondolf et al. 1996; Merz and Setka 2004; Merz and Chan 2005). Interestingly in the eastern US, McManamay et al. (2010) found that gravel addition provided spawning habitat for river chubs (*Nocomis micropogon*) in a regulated river.

Landscape-scale patterns, such as habitat connectivity or fragmentation, and their relation to reach-scale processes ultimately determine the ability of many species to persist (Faush et al. 2002; Dieterman and Galat 2004; Reid et al. 2008). The natural flow regime influences reach-scale stream morphology (Poff et al. 1997) and the interaction between flow and morphology should shape fish assemblages. For example, the “bank-full” width is an indication that high flood pulses literally leave their mark upon the landscape (Gordon et al. 2004). Also, the slope, sediment regime, stream bed size, bedform size/shape, and the flow regime are all intricately related (Buffington and Montgomery 1999; Gordon et al. 2004). Roy et al. (2005) found that relationships between hydrological alterations and fish communities were complicated due to the sediment regime of streams and their interaction with gradient.



The prescription of ecologically-relevant environmental flows currently hinges upon the development of quantitative and transferable flow-ecology relationships (Poff et al. 2010; Poff and Zimmerman 2010). However, the existing framework to develop these relationships (i.e. ELOHA framework: Ecological Limits Of Hydrologic Alteration) currently does not take into account other potentially confounding factors such as temperature, morphology, and landscape disturbances, which the authors of ELOHA readily admit (Poff et al. 2010). However, I suggest that temperature and morphological alterations should be taken into consideration alongside hydrologic alterations to provide a context for developing flow-ecology relationships. For example, simply restoring flows that move sediment (i.e. channel maintenance) will be insufficient attempts at process-driven restoration if there is a change in sediment supply and size due to dam regulation. In a recent review, Jackson and Pringle (2010) argue that restoring hydrologic connectivity may lead to negative consequences such as destroying endemic species' habitat if the sediment regime of rivers is not considered. Thus, restoring fish assemblages by flow restoration alone may be futile if not accompanied by morphological restoration. For appropriate restoration to occur, a template is needed for understanding the relationship between stream flow and channel morphology in regulated rivers and how that influences fish assemblages. Roni et al. (2008) suggested that the failure of many restoration projects stems from a poor understanding of the processes that govern the outcome of a project but also lack monitoring at appropriate spatial and temporal scales. I have found very limited information on the relationship between dam disturbances, flow alterations, and morphological alterations and their overall influence on the structure of fish assemblages. However, Peterson et al. (2009) developed and used a morphological stream classification system to predict the responses of fish to changes in flow. In addition, Pool et al. (2010) compared the relative importance of dam disturbance, land use, and climate on native and non-native fish species as well as various life history strategies.

The purpose of this study was to answer one main question, "What is the relative importance of flow, channel morphology, and temperature in shaping the fish assemblages relative to landscape-scale disturbances (i.e. land-use, fragmentation) within the upper Tennessee River?" Because habitat connectivity is essential, the relative importance of habitat fragmentation and large-scale dam disturbance on fish assemblages and specific guilds must be taken into account to provide some realistic expectations from restoration. For example, for

migratory fishes like the redhorses (*Moxostoma*), the restoration of flows and addition of spawning sediments may be futile if fragment lengths are not sufficient to support yearly migrations (Reid et al. 2008). My study is centralized to rivers systems within the Little Tennessee, Hiwassee, and Pigeon basins because of the high diversity within these systems and the widespread dam regulation. In addition, I want to provide a restoration template (i.e. a framework to provide ecologically relevant flow, morphological, and temperature management actions) for ongoing restoration efforts in the Cheoah River, in western North Carolina (Little Tennessee Basin). I do not account for differences in productivity or evolutionary history. Although there may be some variation due to these factors, my analyses are limited to a fairly small geographic range and based upon guilds and larger taxonomic groups.

One way to simplify management is to organize sites into similar management units. Thus, I classify streams into groups of similar dam operations in order to understand whether systems within the same operational type will influence flow, morphology, and fish assemblages similarly (Anderson et al. 2006). I evaluate spatial patterns in flow variability rather than temporal patterns because I analyze site-to-site differences in morphology and fish assemblages. I consider stream morphology variables that structure fish assemblages to operate at the reach scale (Gordon et al. 2004)(defined in methods); thus, I evaluate the relative importance of gradient, channel morphology (e.g. entrenchment and bar habitat), and substrate size at this scale.

I focus attention on particular taxa and guilds because these categories are easily recognizable, widely-used, and make intuitive sense to managers. Substrate preference is also included as response variables because dam regulation directly and indirectly influences substrate availability and stability. Furthermore, the conservation of many endemic minnows is directly tied to substrate (Johnston 1999). My specific objectives are to 1) determine the affect of various dam operation types on flow, morphology, temperature, and species richness and 2) understand the relative importance of landscape variables (dam disturbance, fragmentation, land-use), flow variables, and morphological variables at explaining the variability in the richness, percent species composition (out of entire community), and the relative abundance of specific guilds and taxa.

## 5.2 Methods

### *Study Sites*

Fish assemblage sampling sites (n=50) were located within the Little Tennessee, Hiwassee, and Pigeon river drainages in the Blue Ridge Physiographic Province (Figure 5.1). One sampling location is on the Little River near Townsend, Tennessee, a direct tributary of the Tennessee River. Sites were chosen by Tennessee Valley Authority biologists primarily to compare fish assemblage characteristics in regulated rivers to unregulated rivers in order to assess the benefit of tailwater habitat improvements. In addition, sites were selected that were located within valleys with lower gradients rather than ridge systems. I chose additional sites that had comparable fish assemblage data within the Cheoah River watershed and the Abrams Creek watershed (Great Smoky Mountains National Park) because they could provide additional information. Samples generally consisted of fish sampling conducted at the “reach” scale (I defined “reach” as at least 5 times bankfull width- tending to be longer in smaller streams). However, the area sampled depended on whether any new species were captured after a number of consecutive runs (see fish assemblage sampling methods). Fish sampling at each site ensured that all habitats (riffle, run, pool) represented within each reach were sampled.

Twenty-eight of the sites were located in streams of unregulated flow (above dams) whereas the remaining 22 sites were located in streams whose flows were regulated. Drainage area ranged from 17 km<sup>2</sup> in Abrams Creek, TN to 3,193 km<sup>2</sup> in the lower Hiwassee River (mean=595 km<sup>2</sup>). Elevation ranged from 220 m to 818 m (mean=524), with the Pigeon River drainage sites generally at higher elevations (> 726 m). Gradient was generally low to moderate and ranged from 0.09% to 2.4% (mean=0.6%). Watersheds for each site had at least 70% or greater forested land coverage (mean = 88%).

### *Riverscape Characteristics and Dam Disturbance Variables*

Explanatory variables are listed in (Table 5.1). I accessed the National Dam Inventory Database (US Army Corps of Engineers) to assess information on dam storage capacity and purpose. I measured the distance upstream from sites to dams using digitized topographic maps in ESRI ARC map 9.2. I then measured the total stream length distance from the site upstream to the headwaters of the largest tributary to calculate the proportional distance of the nearest dam

upstream relative to stream network length. I used a proportional distance value to correct for differences in river size. Thus, I assigned all unregulated rivers a maximum value of 1. Fragment length was measured as the sum of the distance from the site upstream to the nearest dam (or, in the case of unregulated streams, upstream to the headwaters of the largest tributary) and downstream to the nearest dam. I searched the National Dam Inventory database to find all large impoundments upstream of each site. I calculated the total dam storage for each site by summing the storage capacity (megaliters) of all upstream reservoirs and then dividing that value by drainage area (km<sup>2</sup>) and avoided double counting storage capacity (Cantrell and Hill 2009).

### *Land-Use*

I assessed the percent agriculture and developed land for the watershed of each site to account for variation in fish assemblages due to land use (Table 5.1). I delineated the watersheds upstream of each site using 30-m digital-elevation models (DEMs) in ARC map 9.2. I then used National Land Cover Dataset (NLCD) to summarize each land-use category within the boundaries of each watershed. Forested areas included all forest types plus wetlands. Developed lands included all developed categories from low to high density to “open” developed land. Agriculture consisted of grassland/herbaceous, pasture/hay, and cultivated crop categories.

### *Reach-Scale Characteristics: Channel Morphology and Substrate*

I visited each site once to characterize the channel morphology and substrate conditions (Table 5.1). I considered the “reach” to be at least 5 times the bankfull width and summarized site characteristics at this scale. Generally, the length of the reach selected should include two sets of pools and two riffles (Gordon et al. 2004), which the majority of my reaches contained. Reaches have been defined anywhere from 12-50 times bankfull width (Gordon et al. 2004; Bisson et al. 2006); however, Dolloff et al. (1993) suggest that representative sections of a reach can be studied provided they contain all channel units found at the reach scale. I focused my measurements on riffle habitats because riffles, as compared to other habitat units, have the most homogenous cross-sections and generally, are areas of active transport (Gordon et al. 2004); thus, the bed composition of riffles should be characteristic of a river’s regular transport potential (Kappesser 2002). Secondly, focusing on one type of habitat unit can provide a simplified, standard protocol for comparing the morphologies of many different river systems. I isolated

stretches of uniform riffle habitat and avoided areas such as cascades, constrictions of the channel, and deeper riffle/run habitats. In smaller streams, I established transects in the center of four separate riffles; however in larger streams (> 60 m wide), riffle habitats were often separated by 300 m or more. Thus, in larger rivers, I isolated at least two different riffles and established two transects at equal distances from the center of each. The number of cross sections taken within habitat units in a given reach is generally related to the % of the total reach area occupied by each habitat unit (Bisson et al. 2006). Gordon et al. (2004) recommend that 3-5 total cross sections stratified by riffle/pool habitats are sufficient for estimating width/depth in a representative reach. Jowett (1993) was able to discriminate among habitat units using approximately 1 riffle cross section per 100 meters of stream. My main purpose was not to discriminate among habitat units within each reach, but to provide an assessment of gross reach characteristics among different river systems. Habitat unit sequences are typically repeated every 5-7 channel widths (Leopold 1994). Channel widths at my study sites ranged from 8 to 89 meters and whereas total study site reaches ranged from 129 to 636 meters. Because most stream reaches were fairly uniform, 2-4 riffle habitats should adequately represent channel morphology and substrate conditions at each site.

Along each transect, I established benchmarks from which bankfull width was measured, according to Harrelson et al. (1994). Entrenchment ratio is an indication of the connectivity of a river with its floodplain. I measured the entrenchment ratio as the width of the floodplain measured at 2 times the bankfull height divided by the bankfull width. Thus, high entrenchment ratios indicate higher connectivity between a river channel and its floodplain. Gradient was measured over at least the entire reach distance with a transit and stadia rod. I attempted to minimize any bias by measuring the change in elevation from the same point in riffles at the upstream and downstream ends of the reach. I measured the bar habitat index by estimating the total aerial coverage of point, lateral-alternating, and mid-channel bars (of various size material) and dividing that value by the total area surveyed within the reach. Bars are typically exposed above the water surface, but in some higher flow conditions the edges of bars may be inundated. Thus, I attempted to measure the entire surface area by looking for changes in material sizes due to deposition or the presence of active (non-high flow) hydrologic processes.

Substrate conditions were assessed by pebble counts along each of the four transects. Particles were randomly selected from the streambed at equal intervals along each transect and

measuring their intermediate axis with a ruler. I assumed that 200 pebble counts (800 total) would be adequate to assess substrate conditions; however, I questioned whether a smaller sample size (50 % smaller) would be sufficient to yield similar results while minimizing effort. A smaller sample size could be achieved by measuring 200 pebbles at 2 rather than 4 transects or to conduct 100 pebble counts at all 4 transects. I conducted a test using pebble count data collected at three relatively undisturbed sites: the Hiwassee River above Chatuge Lake, Georgia (a small, low-gradient stream), the upper section of Citico Creek, Tennessee (a small, high-gradient stream), and the Little Tennessee River near Iotla, North Carolina (a large, low gradient stream). I conducted 200 pebble counts along 4 transects at each site. For each transect, I then numbered the pebble counts from 1 to 200 and then split the data into two datasets by separating odd and even counts. I then calculated the % sand (< 2mm), % gravel (2mm-64mm), % cobble (64mm-256mm), % boulder (256mm-2046mm), and % bedrock (> 2046mm) for the 200-count dataset and each 100-count sub-dataset. I compared the percentages in two ways: 1) comparing the average percentages of all 4 transects calculated with 200 counts versus 100 counts each, and 2) comparing the average percentage of 200 counts conducted at all 4 transects versus 200 counts at 2 randomly selected transects. I calculated the absolute differences for each comparison of each size class and report the maximum absolute difference that I observed. For comparisons between 200 and 100 counts, I found that the maximum absolute difference in the overall average was 2 %, 2.3 %, and 0.6 % for the Hiwassee River, Citico Creek, and Little Tennessee River, respectively. For comparisons between 4 and 2 transects, I found that maximum absolute differences in the overall average was 4.6 %, 5.3 %, and 5.4 %, respectively. Results of the test suggested that 100 counts would yield very similar results to 200 counts at each transect; however, it also suggested that the choice of transect location could result in some bias. Thus, at the remainder of the sites, I conducted 100 pebble counts along 4 transects. I used pebble counts to calculate the average  $D_{25}$  (particle size of 25<sup>th</sup> percentile) and  $D_{50}$  (median particle size) for each site.

Although surface substrate conditions can represent differences in the sediment regime of river systems, sub-surface conditions also provide some assessment the excess sedimentation, the disturbance regime of deeper streambed layers, and embeddedness (Harrelson et al. 1994; Gordon et al. 2004), which may influence fish reproductive habitat. For example, greenfin darters (*Etheostoma chlorbranchium*), redline darters (*E. rufilineatum*), and wounded darters

(*E. vulneratum*) utilize the interstitial areas between large boulders as protected areas for spawning (Etnier and Starnes 1993). Females partially bury themselves in clean sand or fine gravel in these interstitial areas while males fertilize eggs. These darter species need a complex of sand and boulders to carry out their reproductive behavior, which may not be adequately measured from surface particle counts. Therefore, at 10 equally-spaced locations along each transect, I randomly chose a cobble or boulder particle, lifted it from the streambed, and determined whether or not there was sand or fine gravel underneath (2 – 4 mm). I calculated the sub-surface index as the percentage of 10 particles that had sand or fine gravel underneath.

### *Temperature*

Stream temperature can also be substantially influenced by dam operations, such as surface or bottom releases, and can have profound implications on fish assemblages. I did not directly collect temperature data during my single visit since diurnal fluctuations and variations in flow could cause bias in temperature estimates. (One exception is that I have collected continuous temperature data on 4 sites on the Cheoah River for 3 years). Instead, I acquired data from external sources that were collected continuously or intermittently from mid-June through mid-September (regardless of year). Although this assessment could also be biased by year-to-year differences, I wanted a coarse spatial estimate of temperature at the site level. I assessed temperature at all sites in two main ways: 1) Searching the USGS “field/lab water-quality sample” section of the gage information site and 2) querying North Carolina Wildlife Resources Commission, US Fish and Wildlife Service biologists, and Great Smoky Mountain National Park biologists for continuous or intermittent temperature data and temperature reports. Temperature data through the USGS gage information site was available for 26 sites whereas continuous and intermittent collections (including temperature reports) were available for 19 sites. Altogether, 45 of the 50 sites had temperature data collected continuously or intermittently during June through September. Twenty of the USGS field/lab sites had temperature data from the mid 1970’s through mid-1980’s. All other sites had data from the last decade. Since the majority of the watersheds in my region have very similar land-use coverage compared to the 1970’s (Brown et al. 2005), I assumed that my coarse temperature assessment was not substantially biased due differences in time period. Secondly, the purpose of my assessment is to quantify large deviations in temperature due to dams, all of which have been operating for at least the past 43

years in my region. I did not find temperature data for 5 of my 50 sites, all of which were smaller, unregulated systems. I developed a multiple linear model using elevation, drainage area, gradient, and bankfull width for 26 unregulated gages. The model explained 40% of the variation in site-level temperatures and was used to estimate temperature for the 5 remaining sites. Temperature values for each site represent a composite value of temperatures averaged from mid-June to mid-September.

### *Flow Assessment*

My flow assessment was based on spatial, rather than temporal patterns in flow dynamics. For example, the flow metrics I use summarize patterns across at least a 10-year period and do not represent year-to-year variability. In addition, because continuous discharge information is not available for all sites, this method allows for spatial extrapolation of natural flow indices to sites where discharge data may not be available. Thirty-four of my 50 sites were located on the main-stem of a river where a USGS gaging station was located. Twenty-three of those sites had reaches in which an actual USGS gage station was located (including 1 site directly below dam with daily spillage data). The remaining 16 sites were on unregulated streams that were located adjacent to watersheds with gages. Mean daily and annual peak flow data for the 22 stream gauges were downloaded from the USGS Water-Data Site Information for the Nation website. I ensured that discharge data from unregulated sites spanned the longest records available whereas discharge data from regulated sites consisted of only “post-regulation” data. Hydrologic statistics were calculated for each gage using the Hydrologic Index Tool (HIT) software available through the USGS (Henriksen *et al.*, 2006). Daily and peak flow gauge data were imported into the HIT software, which calculates the 171 hydrologic indices reported in Olden and Poff (2003). I chose 5 hydrologic indices that were representative indicators of natural flow variability for streams in the study region based on McManamay *et al.* (2011) (Table 5.1). For sites without gage information (small, unregulated watersheds), I used flow indices calculated for unregulated stream gages of similar size that were found in the nearest neighboring watershed. I assumed that streams found within close geographical proximity would share similar patterns in flow, especially since datasets created using HIT software are generally used for spatial analyses. For indices influenced by basin size (annual maximum and minimum flows), I used regional curves based on 93 stream gages from McManamay *et al.* (2011) to adjust



indices for differences in drainage size. Once I had assembled my flow indices dataset, I standardized the 2 magnitude variables by dividing by drainage area to control for differences in river size. However, I noticed that the standardized max and min flows were negatively related to drainage area (i.e. max flow per unit drainage area is higher in smaller streams). Therefore, I ran regressions for the unregulated streams only and then calculated residuals for both regulated and unregulated streams.

### *Fish Assemblage Sampling*

The Tennessee Valley Authority (TVA) modified IBI methodologies to assess fish assemblages below tailwaters during 1987 to 1994 (Saylor and Ahlsted 1990). TVA biologists required a standard protocol for assessing fish assemblages and required metrics whose scoring criteria were more appropriate for assessing whether improvements in tailwaters influenced fish assemblages (Saylor and Ahlsted 1990). By the late 1990's, TVA field crews were using the IBI methodology throughout many regulated and unregulated rivers in the Tennessee River system.

The sampling methodology employed multiple gear types including boat electroshockers, backpack electroshockers, seines, and dipnets to ensure sampling in all habitat types represented at the site (riffles, runs, pools, and shorelines). Sampling was conducted during a period from March to August and during minimal flows to ensure crew member safety and unbiased capture efficiencies. Generally, the shoreline and channels of deep pools were sampled by boatshocking in the downstream direction for 10-minute sampling efforts. The 10-minute sampling efforts continued until two consecutive runs failed to collect any new species.

Shallower habitats, such as riffles and runs, were sampled by backpack electrofishing units and seines. In moderate to fast currents, a 20-ft seine was held perpendicular to the current while one crew member backpack electrofished from 20 ft upstream towards the seine (1 sampling effort). Generally fish were stunned and drifted into the seine or were netted by the backpack operator. Sampling was conducted as to target all habitats represented with respect to current, depth, and substrate. Backpack shocking was also conducted in 5-minute sampling efforts along shorelines. In sluggish backwater and shallow pool habitats seine hauls were made. Sampling efforts continued in all habitat types until three successive runs failed to collect any new species. Fish were collected, identified, and inspected for abnormalities. YOY were

identified but only adults were enumerated, especially since sampling was conducted at various times of the year.

From 1997 to 2009, TVA field crews conducted fish assemblage sampling at 43 of my 50 sites on a 3- to 5-year rotation with the exception of a few sites that were sampled every year. Sites were sampled based on a randomized design that ensured every site was sampled at least once within a 5-yr window. The 7 sites not sampled by TVA consisted of 3 Abrams Creek sites and 4 Cheoah River sites. Abrams Creek sites are located within the Great Smoky Mountains National Park and were sampled by 3-pass depletion by the Park Service using backpack electroshockers and blocknets. Abrams creek sites were sampled on a yearly basis from 1993 to 2009. The Cheoah River sites were sampled sporadically by North Carolina Wildlife Resources Commission from 1993 to 2003 using 3-pass depletion and blocknets. From 2003 to 2009, the Cheoah sites were sampled almost every year using a variety of methods including IBI approaches in shallow habitats, snorkeling, and a combination of both. In the Cheoah, care was taken to sample all represented habitats and geomorphological units.

I used fish assemblage information for the latest two sampling occasions (if available) for each site. I summed the abundance for each species for both of the sampling occasions and then calculated a composite relative abundance value for each species for each site. This method minimizes the influence of individual-year outliers. I then used the FishTraits Database (Frimpong and Angermeier 2009) to access information for two main categories: 1) substrate preference (n=2) and 3) taxonomic group and guild (n=8) (Table 5.2). I chose traits/guilds that should either be directly influenced by flow, substrate, or indirect relationships (habitat stability, flow environment, disturbance regime, etc). I calculated summary values for each trait/guild for each site based on the relative abundance, richness, or the percent of species found that possessed each trait or within each guild (Table 5.2).

### *Fish Sampling Assessment*

Sampling conducted during different months could cause some bias in biotic response metrics. I isolated two sites, the Hiwassee River at Reliance Bridge, TN and the Little Tennessee River at Needmore, NC that were sampled for multiple years (7 and 11 years, respectively) and were sampled within at least 3 different months. For each year, I calculated richness for all species and richness and relative abundance for *Moxostoma* species, *Notropis* species, and Percid

species since their values may be influenced by sampling at different times of the year. For each site, I calculated the range in richness and relative abundance for the entire sampling period for the dominant sampling month. I then determined whether the richness and relative abundance values for non-dominant months fell outside of the ranges of the dominant sampling month. I also evaluated temporal patterns in richness and relative abundance to determine if non-dominant sampling months seemed to create “outliers.”

Since I used the latest two sampling occasions to calculate the biotic response metrics in my study, I wanted to determine the extent of overlap in the timing of sampling events across all sites. Thus, for the next-to-last sampling event, I sorted all sites by their sampling month (from earliest to latest) and plotted the results. I then plotted the month of the corresponding latest sampling event. I shaded the region between the 1<sup>st</sup> and 2<sup>nd</sup> sampling events to determine temporal overlap.

#### *Assessing patterns in dam regulation types, flow, morphology, and species richness*

Using information from the National Dam Inventory database and from field visits, I classified streams into groups of similar dam operations. Bypass or diversion sites consisted of reaches where water was diverted around the site leaving minimal flows punctuated by large spill events. Bottom-release sites consisted of baseflow-hypolimnetic releases and scheduled daily peaking. All other sites controlled by dam operations were classified as “Regulated” either because 1) dam storage was minimal, 2) upstream dams consisted of water-supply reservoirs that did not dramatically alter flows, or 3) they were far enough downstream and had sufficient tributary inflow such that operations did not substantially alter riverine habitat. In order to evaluate any operation-specific effects, I plotted all six flow metrics by regulation type versus dam storage. I also plotted entrenchment ratio, bar habitat index, D<sub>50</sub>, and temperature by regulation type versus distance to dam and dam storage. In both cases, I compared the range of values to values for unregulated streams. I also plotted benthic insectivore, percid, catostomid, and cyprinid richness versus drainage area according to regulation types to determine if operations had specific influences on fish assemblages.

#### *Constructing Linear Models*

In order to select the “best” explanatory variables for each response variable, I used “all possible models” in JMP (SAS) to select the best models from 1 variable to a maximum of 3 variables. Best models for 1 to 3 variables were chosen and ranked based on the corrected Akaike’s Information Criterion (AICc) (Burnham and Anderson 2002). Models were limited to three variables in order to be conservative and avoid over-fitting data. Alternative models with the lowest AICc score are considered “best” in terms of maximizing the overall amount of variability explained while minimizing the total number of variables (most parsimonious explanation). However, I also compared Mallows  $C_p$  value to the number of predictors to determine if I needed to reduce the number of variables in each model (Mallows 1973) where:

$$C_p = \left( \frac{SSE_p}{S^2} \right) - (N - 2p)$$

and  $SSE_p$  is the sum-of-square error for models with  $p$  variables,  $S^2$  is the mean square error,  $N$  is the sample size, and  $p$  is the number of predictors + 1. The model should be chosen where it minimizes the lack of fit (bias) or where  $C_p$  first approaches  $p$ .

I assorted variables into different categories (flow, channel morphology, etc) and evaluated the frequency in which they were used in all models. I then calculated the partial  $r^2$  values for all variables found in the best models and compared their values across different categories. All variables were transformed prior to analysis. Variables with count data were square root( $x+0.5$ ) transformed whereas variables in the form of proportions were *arcsin* square root transformed. All other variables were  $\log(x+1)$  transformed.

### 5.3 Results

#### *Fish Sampling Assessment*

Eighty-eight percent of all sites ( $n=43$ ) were sampled more than once whereas the remainder ( $n=7$ ) were sampled only once. This yielded a total of 93 sampling events that were used to summarize biotic response variables for each site. Although, samples ranged from 1997 to 2009, 80% of sites were sampled at least once since 2004 and 68% of sites were sampled at least once since 2006. At the Hiwassee River at Reliance Bridge, TN, sampling was conducted over 7 years during March, April, and May with March being the dominant sampling month. At the Little Tennessee at Needmore, NC site, sampling was conducted over 11 years during May,

June, and July with June being the dominant sampling month. At both sites and for all variables, the only consistent pattern was lower *Moxostoma* richness in April and May compared to March at the Hiwassee River site. Otherwise, most of the variation in variables was due to temporal trends and not the timing of sampling. For example, at the Little Tennessee site, most biotic variables, with the exception of *Notropis* relative abundance, did not seem to be influenced by the sampling month as much as longer-term temporal trends (Fig. 5.2).

By comparing the sampling month of the latest two sampling events, I found that there was considerable overlap in the timing of samples despite sampling being conducted from March through November (Fig. 5.3). Forty-two out of the 50 sites (84 %) were sampled at least once during the summer (June through August). However, my analysis showed that some sites were sampled at very different times of the year, which could have certainly still influence the composite values (last two sampling events) for my biotic response variables.

#### *Assessing patterns in dam regulation types, flow, morphology, and species richness*

Some flow variables tended to be related to dam storage whereas others were more influenced by regulation type. For example, following log transformation, minimum flows, maximum flows, and flood interval were negatively related to dam storage (linear regression,  $r^2=0.18$ ,  $p=0.049$ ,  $r^2=0.37$ ,  $p=0.003$ ,  $r^2=0.22$ ,  $p=0.020$ , respectively) whereas the number of reversals were positively related to dam storage ( $r^2=0.21$ ,  $p=0.030$ ). Bottom-release sites had the highest number of high flow events and reversals whereas diversion and bottom-release sites together had the lowest maximum and minimum flows and shortest flood intervals. Most regulated streams tended to fall within the range of unregulated rivers; however, diversion-bypass and bottom-release sites tended to fall outside of the range of unregulated rivers.

Similar to flow variables, other physical variables tended to related to distance to upstream dam and dam storage whereas others were “outliers” due to regulation type. Entrenchment ratio and temperature (June-September average) tended to be most dramatically influenced by either diversion or diversion and bottom-release sites, respectively. Entrenchment ratio and temperature was highest in diversion sites whereas temperature was lowest in bottom-release sites. The median particle size was positively related to dam storage and negatively related to distance from dam (log transformed,  $r^2=0.19$ ,  $p=0.042$  and  $r^2=0.26$ ,  $p=0.016$ , respectively). Bar habitat index was positively related to distance from dam (log transformed,

$r^2=0.34$ ,  $p=0.004$ ). Similar to flow variables, most physical variables for regulated streams fell within the range of unregulated streams with the exception of diversion-bypass sites and bottom-release sites. However, not all sites within a regulation type acted similarly (i.e. some were outliers whereas others were not) suggesting that the response of river systems to dam regulation was site-specific.

I also plotted benthic insectivore, percid, catostomid, and cyprinid richness versus drainage area according to regulation type (unregulated, regulated, bottom-release, diversion). Although each plot revealed a positive relationship between site richness and drainage area, the relationships were weak (transformed data, linear regression: benthic insectivores,  $r^2=0.14$ ,  $p=0.008$ ; percids,  $r^2=0.28$ ,  $p<0.001$ ; catostomids,  $r^2=0.29$ ,  $p<0.001$ ; cyprinids,  $r^2=0.10$ ,  $p=0.022$ ) (Fig. 5.6). However, the plots revealed a separation in richness-drainage area relationships between unregulated and all other regulated sites. Regulated sites had smaller richness values relative to drainage area than unregulated streams. In addition, bottom-release and diversion-bypass sites had substantially lower richness values relative to drainage area (Fig. 5.6).

### *Constructing Linear Models*

On average, models explained 48% of the variation in response variables and ranged from 24% to 64% ( $R^2$  adj.) (Tables 5.3-5.5). Linear models for richness explained, on average, 53% of the variability (Table 5.3) whereas relative abundance and percent species models both averaged 46% ( $R^2$  adj) (Tables 5.4-5.5). Morphology, land-use, fragment length, and substrate had the highest frequency as predictors in models (in that order) (Fig. 5.7A). Flow variables had the lowest frequency with the exception of distance to dam (Fig. 5.7A). On average, fragment length explained the most variation in models followed by temperature and land-use (Fig. 5.7B). Fragment length was found as the best single predictor in 7 of the models followed by land-use, bankfull width, temperature, and gradient serving as the best single predictors in fewer models, respectively (Tables 5.3-5.5).

In general, richness for individual taxa and guilds was governed by fragment length, river size (bankfull width), gradient, and substrate (Table 5.3). Richness variables tended to increase with fragment length (Table 5.3). In terms of relative abundance and % species composition, fragment length was positively related to percid relative abundance, percid % species composition, and sand/gravel spawner % species composition (Tables 5.4-5.5). Fragment length

was negatively related to rock/gravel spawner % species composition. Temperature explained significant variation in models for benthic insectivores, centrarchids, percids, and species preferring lentic habitats. Temperature had positive effects on all centrarchid variables, percid relative abundance, and the relative abundance of species preferring lentic habitats (Tables 5.3-5.5). However, temperature had negative effects on benthic insectivore relative abundance and % species composition. Centrarchid, species preferring lentic habitats, and catostomid variables were positively related to % development whereas percid and cyprinid variables were negatively related to % development. Catostomid richness and relative abundance (including *Moxostoma*) increased with bankfull width (river size) and smaller substrate sizes ( $D_{25}$ ). Flow variables were generally found in models for benthic insectivores and substrate size specialists (Tables 5.4-5.5). Benthic insectivore and rock/gravel spawner relative abundances were negatively related to the number of reversals. Sand/gravel spawners generally increased with lower minimum flows, higher maximum flows, and longer flood intervals (Table 5.4-5.5).

## 5.4 Discussion

Managing river systems for fish assemblages requires multi-faceted approaches that account for the interaction of flow, channel morphology, and landscape fragmentation at multiple spatial and temporal scales. I do not presume that my study can simplify the relationships between variables that influence fish assemblages of regulated and unregulated river systems. However, I believe that isolating factors with greatest explanatory power can help to prioritize management applications by narrowing the focus on relative characteristics.

The results of my linear models suggest that transferable flow-ecology relationships must take into account landscape, channel, substrate, and temperature disturbances. Although the dominant literature suggests that flow organizes fish assemblages in unregulated rivers and streams (Poff and Allan 1995; Herbert et al. 2003; Pyron and Lauer 2004), landscape disturbances (fragmentation and land use) and morphology (gradient, bar habitat, etc) have the highest explanatory power on fish taxonomic groups and guilds in the upper, regulated Tennessee River system. Flow variables were poorly represented in models, which suggests that in highly fragmented systems, other factors may “govern” fish assemblages. Morphology, land-use, fragmentation, and substrate had the highest frequency as significant predictors in models,

which suggest that these factors should be considered when managing for “environmental flows” in regulated systems.

Evaluating individual biotic responses can provide important management implications; however, approaches that attempt to generalize patterns in disturbance can provide insight into managing fish assemblages. For example, I organized sites according to different dam operations to evaluate patterns in flow, morphological, and temperature alterations. Dam operation types, specifically diversion and bottom-release projects, tended to effect flows, morphology, and temperature dramatically different than the other regulated systems, which suggests that different dam operations may influence the habitats that organize fish assemblages, differently. However, I did find that not all sites responded similarly under the same operation site. Some of the variables showed positive or negative relationships with dam storage and distance from dam. Regardless, there is evidence that generalizations concerning habitat alterations can be made.

#### *Assessing the relationships between dam regulation type, flow, morphology, and species richness*

Ultimately, different operation types influenced flow, morphology, and temperature. In addition, my results suggest that operation types also dramatically influenced taxa/guild richness. Predictor variables tended to also be influenced, in general, by dam storage and distance from dam. For example, minimum flows, maximum flows, flood interval, the number of reversals,  $D_{50}$ , and bar habitat were all related to dam storage or distance from dam (Fig. 5.4-5.5). This is an important conclusion considering that categorizing operation “types” may be difficult over large spatial scales given that proper classification may include field reconnaissance; however, spatial datasets that include dam storage are becoming more available (National Hydrography Dataset (USGS); Falcone et al. 2010) and distance to dam can easily be measured within a GIS framework.

Fish taxa and guild richness were influenced, in general, by dam regulation; however, different dam operations had variable effects on richness. For example, my results suggest that the Cheoah River sites (Diversion-Bypass) have 13 to 29 fewer species than unregulated sites with similar drainage areas. Similarly, the Hiwassee Diversion-Bypass sites have 12-20 fewer species than comparable unregulated sites. The diversion-bypass projects tended to have larger  $D_{50}$  values and little to no bar habitat. Altogether, this suggests that diversion projects (bypass



reaches) may have even more substantial influences on fish communities than typical dam-regulation projects and will require substantial restoration efforts. Although there is not a large literature base on the effect of dam diversion projects, evidence shows that bypass/diversion projects have substantial negative effects on biotic communities (Dessaix et al. 1995; Anderson et al. 2006). Both water diversions and dams have been shown to adversely affect fluvial geomorphology (Osterkamp and Hupp 2010); thus, diversion dams created a compounded effect of massive losses in flow volume and a substantially altered sediment regime. Also, the bottom-release sites had similar richness values as sites with drainage basins that were almost six times smaller, which suggests that temperature alterations will also dramatically affect fish assemblages.

Because of the large influence that diversion-bypass and bottom-release dam operations exerted on fish assemblages, I compared species common to the Upper Tennessee River to those in diversion-bypass and bottom-release sites to determine any common trends in the loss of species. I summarized the 40 most common species found across all 50 sites in the upper Tennessee River and then determined if those species were found in each of the operation types (Appendix D). I included sites found on Richland Creek (unregulated sites above Lake Junaluska), because they were disturbed from high urbanization and relatively short fragment lengths. Patterns emerged that showed common losses in species (Appendix D). For example, river redhorse (*Moxostoma carinatum*), golden redhorse (*M. erythrurum*), and shorthead redhorse (*M. macrolepidotum*) were missing or largely uncommon at all sites regardless of operation type. Interestingly, although mottled sculpin (*Cottus bairdii*) were found in 74% of all sites, they were missing from the Cheoah River and Richland Creek sites. Although warpaint shiners (*Luxilus coccogenis*) and river chubs (*Nocomis micropogon*) were found in 84 and 86% of the sites, respectively, they were missing from the Richland Creek sites and the Hiwassee Chatuge sites (bottom-release). In addition, many common *Notropis* and darter species were missing from these regulated/disturbed sites. Altogether, this suggests two conclusions: Some fish species will respond negatively to disturbances regardless of cause (land-use or dam regulation), whereas the presence/absence of other fish taxa will be related to the extent and type of disturbance (i.e. operational types). Although this may seem obvious, it highlights a larger point. The type and extent of different disturbances (e.g. dam operation type) will have to be

accounted for in spatial analyses to that relate habitat alteration to ecology, such as determining flow-ecology relationships.

### *Constructing Linear Models*

Models predicting guild and taxa richness typically included fragment length, bankfull width, gradient, and substrate (for catostomids) (Table 5.3). Fragment length was the dominant single best predictor and was present in many models including those for benthic insectivores, cyprinids (including *Notropis*), Percids, and substrate size specialists. Colonization potential is directly related to migratory movement rates, which may be unrelated to body size (Albanese et al. 2009). For example, darters have also been shown to migrate extensive distances (Neely and George 2006; Roberts and Angermeier 2007). Not surprisingly, the negative impacts of fragmentation on genetic structure, dispersal, and presence of darter species has been well documented (Quinn and Kwak 2003; Haponski et al. 2007; Beneteau et al. 2009; Kashiwagi and Miranda 2009). Other small-bodied benthic insectivores may also be influenced by dam fragmentation. Breen et al. (2009) documented mottled sculpin (*Cottus bairdii*) dispersing over 500 meters in a year.

Fragmentation has also shown negative impacts on the genetic structure and habitat connectivity of cyprinid species (Han et al. 2008; Skalski et al. 2008). Hoagstrom et al. (2008) found that the presence of the Pecos bluntnose shiner (*Notropis simus pecosensis*) was related to longer fragment lengths and increased tributary confluences (i.e. habitat connectivity). Barriers to migration are also considered one of the top threats to suckers species (Cooke et al. 2005). For example, Reid et al. (2008) reported that redhorse were absent upstream of major barriers in the Grand River watershed, Ontario and were completely absent from the highly-fragmented Speed River sub-watershed. Interestingly, fragment length was not found in any models for catostomids or *Moxostoma*. However, the Valley River, a small to medium-sized river along its course (52 to 282 km<sup>2</sup> at my sites), is undammed and unfragmented for its entire course until its junction with the Hiawassee River and has the greatest sucker richness in the region.

Because drainage area was poorly related to species richness, I plotted benthic insectivore, percid, catostomid, and cyprinid richness versus fragment length (Fig. 5.8). Significant positive relationships between taxa/guild-specific richness and fragment length were

stronger than relationships between richness and drainage area. For example, fragment length explained 50 % of the variation in percid richness (transformed,  $r^2=0.50$ ,  $p<0.0001$ ). Fragment length explained 31%, 29%, and 27% in benthic insectivore, castostomid, and cyprinid richness, respectively ( $p<0.001$ ,  $p=0.006$ ,  $p<0.001$ ).

Channel morphology and substrate will certainly influence richness and abundance of some species (Walters et al. 2003; Dauwalter et al. 2008). For example, Walters et al. (2003) found that fish assemblage structure was predicted best by reach-level geomorphic variables. Likewise, Dauwalter et al. (2008) found that channel morphology and stream size explained the most variability in fish species composition compared to other geomorphological and habitat variables (non flow variables). The results of my models showed that catostomid and *Moxostoma* richness were influenced by bankfull width and  $D_{25}$ . Bankfull width may be a better surrogate for the presence of redhorse species than basin size since dam regulation, especially diversions, generally decreases channel width. Walters et al. (2003) found that redhorse richness increased with basin area and baseflow width. Reid et al. (2008) found that channel morphology and substrate were only significantly different between occupied and unoccupied sites for 2 redhorse species. In addition, my results suggest that the presence of finer substrates (lower  $D_{25}$ ) will influence the abundance and richness of catostomids (Tables 5.3-5.5). Substrate variables were also found in models for benthic insectivores, sand/gravel spawners, and lentic habitat species (Tables 5.3-5.5) and has been shown to influence habitat generalists and benthic insectivores (Walters et al. 2003). Gradient was represented in models for centrarchids, cyprinids (including *Notropis*), benthic insectivores, lentic habitat specialists, and rock/gravel spawners.

Temperature alone explained 45% of the variation in benthic insectivore relative abundance (Table 5.5). Temperature was also found as a variable in models for centrarchids, percids, and species preferring lentic habitats. The reduction in temperatures and the biotic effects of hypolimnetic releases (bottom-releases) has been well documented (Hamblin and McAdam 2003; Krause et al. 2005). However, smaller dams and surface-release operations may increase temperatures leading to biological impairment (Lassard and Hayes 2003; Caissie 2006).

Flow variables explained a significant amount of variation in models for benthic insectivores and substrate size specialists (rock/gravel spawners and sand/gravel spawners). In general, dam regulation decreases maximum flows and increases minimum flows (Magillan and

Nislow 2001 and 2005; Pyron and Neumann 2008, Chapter 4). Reductions in hydrologic variability and creation of slack water environments generate habitats favorable to generalist species and unfavorable to fluvial specialists (Bunn and Arthington 2002; Poff et al. 1997). In a study across the US, Carlisle et al. (2010b) found that diminished flow magnitudes led to increases in habitat generalists and decreases in fluvial specialists. Similarly, Freeman and Marcinek (2006) and Kanno and Vakoun (2010) reported that diminished flow magnitudes led to decreases in fluvial specialists. In this study, I found that the relative abundance of sand/gravel substrate specialists decreased with lower 1-day maximum flow magnitudes (Table 5.4-5.5); however, I also found that the % of sand gravel substrate specialists in the fish assemblage was negatively related to minimum flow magnitude. The majority of sand/gravel substrate specialists in this study (17 species) were fluvial specialists (6 darters, 5 cyprinids (including dace, chub, and *Notropis* species), 2 lampreys, 2 catostomids, redbreast sunfish, and black bullhead). I speculate that sand/gravel specialists benefit from high peak flood pulses that deposit sediments scoured from riparian habitats (Trush et al. 2000). Because dams may increase minimum flows (excluding the bottom-release and diversion sites), lower minimum flows may provide a more natural flowing environment for sand and gravel substrate specialists.

Knight et al. (2008) reported that constancy, the frequency of moderate flood events, and the rate of streamflow recession explained a significant amount of variation in the fish community structure of streams in the Tennessee River Valley. Dam regulation increases the number of flow reversals (Magillan and Nislow 2001 and 2005; Pyron and Neumann 2008) but decreases the duration of low flows and the flood interval (Chapter 4, Table 4.3). I found that benthic insectivores and rock/gravel spawners were negatively associated with flow reversals (positive/negative changes in flow from one day to the next). In addition, sand/gravel spawners were negatively associated with shorter flood intervals (more frequent flood events). Similarly, Knight et al. (2008) showed that specialized insectivores were negatively associated with elevated streamflow recession rates (higher numbers of reversals) and more frequent flooding (shorter flood intervals).

Relative to fragmentation, land-use, channel morphology, and substrate, flow variables were poorly represented in the linear models in this study (5/23 models). I find this highly important since flow is generally considered the “governing” variable that organizes fish

assemblages in unregulated rivers and streams (Poff and Allan 1995; Herbert et al. 2003; Piron and Lauer 2004). In addition, management in hydrologically-altered river systems is dependent upon developing flow-ecology relationships to inform the creation of environmental flow standards (Poff et al. 2010). Although there might be a suite of plausible reasons for why flow variables were poorly represented, I provide three possible explanations:

1) The choice of biotic response variables in my study may not be sensitive to flow alterations. I chose taxa and guilds that are easily distinguishable and were relevant to management. Benthic insectivores are commonly used as a biotic indicator of habitat alteration (Knight et al. 2008). However, guilds such as fluvial specialists and habitat generalists are commonly used in association with flow-alteration studies (Roy et al. 2005; Freeman and Marcinek 2006; Kanno and Vokoun 2010). Other traits/guilds commonly used in flow-related studies are life history traits, such as periodic/equilibrium species (Frimpong and Angermeier 2010), morphological traits, such as swimming ability (Olden et al. 2006), and habitat preferences, such as lentic habitat or reproductive habitat (Carlisle et al. 2010b). Although I could have included more flow-sensitive biotic responses, the suite of biotic responses I chose should be relevant to flow management. For example, substantial literature suggests that percids and *Moxostoma* species are fluvial specialists that are usually negatively affected by dam regulation (Quinn and Kwak 2003, Cooke et al. 2005). In addition, I included preference for lentic habitat, benthic insectivores, and reproductive substrate preference as response variables: all of which have been shown relationships with flow alterations.

2) The resolution and spatial scale of my analysis may be inadequate to assess patterns in biotic responses due to flow alterations. According to Jackson et al. (2001), the spatial scale of an analysis will influence the overall ability for investigators to perceive the importance of what factors organize fish assemblage structure. Carlisle et al. (2010b) conducted a nationwide assessment of altered stream flow magnitudes and the potential influences on traits and found strong patterns. Large differences in flow variability occur across broad regions (Poff et al. 1996); thus, larger spatial analyses may be needed to assess the influence of flow alteration on fish assemblages. However, many analyses whose purpose is to assess the “effect” of flow alteration at large scales may, in actuality, be representing associations between biotic responses and other factors *correlated* with flow alteration, such as morphological and temperature alterations and even landscape disturbances (e.g. fragmentation). I evaluated spatial rather than

temporal patterns in flow. However, temporal evaluations of changes in flow at the same location may yield more specific patterns in flow-fish relationships, which may not be isolated by gross spatial differences. Again, Jackson et al. (2001) indicate that as the resolution of analyses increases from large-scale assessments to that of individual streams, the extent of local environmental extremities (i.e. changes in flow regime) will play a larger role in structuring fish assemblages.

3) Other factors may exert stronger influences on habitat formation than flow in streams in the upper Tennessee River. A river's substrate size, slope, and sediment regime are all related to its discharge (Gordon et al. 2004). Flow is considered the governing variable that controls the maintenance and formation of riverine habitats (Trush et al. 2000). Obviously, excess sediment loads relative to a given flow regime's carrying capacity will cause a channel to aggrade and lower its slope (Gordon et al. 2004). However, excess sediment is not generally an issue in my sites, since most unregulated systems did not suffer from extensive poor land-use practices (with the exception of upper Little Tennessee R.). In addition, a great deal of excess sediment is trapped within reservoirs. Thus, the presence and storage capacity of reservoirs control the amount of sediment available for transport, which leads to sediment starvation rather than excess sedimentation. The ability of a river channel to adjust its morphology (at least gradient) and habitats in relation to changes in flow will depend upon the alluvial character of the river (i.e. ability to aggrade, degrade). Interestingly, I observed bedrock outcrops at the majority of my sites (76%). This suggests that the river systems in the upper Tennessee are largely not alluvial but are confined by valley and geomorphic constraints, which may limit the overall influence of flow on channel morphology. Not surprisingly, channel morphology variables, such as gradient, explained more variation than flow in my models. Because of valley constraints, gradient may exert more influence on river habitats in the upper Tennessee River. Interestingly, Walters et al. (2003) found that stream gradient was the dominant factor that controlled stream habitats (substrate size, bed mobility, and tractive forces (shear stress)) in Piedmont streams; thus, gradient may govern stream habitat morphology, which in turn, will influence fish assemblages.

Restoring habitats for fish communities in the upper Tennessee River Valley, and rivers at large, will require multi-faceted management strategies. My results suggest that fragmentation, land-use, channel morphology, and substrate explained the majority of variation and had a high frequency of occurrence in models predicting fish assemblage variables.

Although less common in models, flow variables did explain significant variation in models for benthic insectivores and substrate-size specialists. Therefore, I suggest that in order to develop flow-ecology relationships, analyses should be placed in the context of landscape disturbance, morphology, and temperature regime. My results also suggest that assigning regulated rivers to dam operation types may be an informative way to assess flow, morphological, and biological effects of dam regulation. The obvious effects of bottom-release dams are dramatic decreases in temperature, which also have profound influences on fish assemblages. However, diversion dams (bypass reaches) seem to have substantial negative impacts on the flow, channel morphology, sediment, and temperature regime, all of which influence fish assemblages.

## **5.5 Conclusion**

The results of this study have regional as well as global implications for environmental flow management as a simple reminder of the complexities of attempting to make generalizations concerning disturbance-ecology relationships. Recent environmental flow management and potentially regulated river restoration at large, has been thwarted, at least to some degree, by the absence of quantitative, transferable flow-ecology relationships (Poff et al. 2010; Poff and Zimmerman 2010). The development of the ELOHA (Ecological Limits of Hydrologic Alteration) framework is the product of the consensus view of 19 international scientists and leaders in the field of environmental flow science (Poff et al. 2010). The basis of the ELOHA framework is the development of regional flow-ecology relationships from which environmental flow standards are formed. Although the authors of the ELOHA framework readily admit that “scientific uncertainty will exist in the flow alteration–ecological response” due to other “environmental determinants” besides flow, the ELOHA framework largely ignores landscape-scale disturbance, morphological alterations, or temperature alterations (Olden and Naiman 2010; Poff et al. 2010), all of which collectively influence fish assemblages (Fausch et al. 2002) and may confound relationships between altered hydrology and ecology. For example, Olden and Naiman (2010) suggest that temperature regimes should be included environmental flow assessments; however, it is rarely done. In addition, Jackson et al. (2001) indicate that spatial connectivity will govern the overall importance of other abiotic and biotic factors in determining fish community composition. I found that fragmentation, land-use, channel morphology, and substrate explained as much or more variation than flow variables in models,

which suggests that these variables cannot be ignored in the development of flow-ecology relationships. According to the ELOHA framework, once environmental flow standards are created, they are intended to be implemented in water policy (Poff et al. 2010). Implementing new flow regimes without considering and understanding the implications of potential responses in morphology and habitat connectivity can be detrimental to aquatic communities (Jackson and Pringle 2010).

Interestingly, the principles of the ELOHA framework are already being put into practice at regional scales. For example, the Southeastern Aquatic Resources Partnership (SARP) is a collaboration of natural resource agencies, scientists, conservation organizations, and private citizens to manage and conserve aquatic resources within the 14-state southeastern United States. The Southern Instream Flow Network (SIFN), an organization within SARP, adopted a research agenda in 2010 that includes 5 priority research topics, all of which are heavily based upon the ELOHA framework (SIFN 2010). The 5 priority research topics are as follows: 1) Develop a regional river classification system, 2) Identify commonalities in ecosystem responses to flow alterations, 3) Compile regional aquatic ecology data sets, 4) Develop hypotheses for regional ecological responses to flow alteration, and 5) Perform field studies to test ecological responses to altered flow regimes. These research topics are certainly needed and will influence flow management in the southeastern US. However, despite the brief mention of a geomorphic subclassification in topic 1 and the mention of potential relationships between flow and other physical factors topic 4, it is obvious that environmental flow standards for this entire region could be developed without the consideration of other relationships.

There are several ways that other factors besides flow could be incorporated into a broad framework. One way is to incorporate geomorphology, temperature, and disturbance (e.g. dam operation) into classifications similar to flow. These classifications can serve as a physical baseline similar to the hydrologic baseline developed from flow classifications. The physical baseline then serves as a foundation from which physical alterations are measured. Thus, in river systems, rather than just developing flow-ecology relationships, flow-geomorphic-temperature-ecology relationships could be developed.

Restoring river systems requires a multi-faceted approach that considers habitat connectivity and the spatial and temporal interactions between flow dynamics, channel morphology, and sediment. Based on my results, fragmentation and larger landscape



disturbances should provide a context for determining how physical disturbance may structure fish assemblages. Knight et al. (2008) concludes that managing one factor to restore habitat conditions may not “guarantee improvements in fish community health; it only allows for the potential response provided all other factors are restored.” Because river ecosystems consist of dynamic lateral and longitudinal interactions over multiple spatial and temporal scales, the development of frameworks to understand the relationships between physical, chemical, and biological properties and to simplify management has become a necessity for restoration efforts.

Table 5.1. Explanatory variables used in the analyses.

| Variables                 | Explanation  |
|---------------------------|--|
| <b>Dam Disturbance</b>    |  |
| Distance to Dam           | ratio (distance upstream to dam (km)/ total stream length (km))    |
| Dam Storage               | megaliters/km <sup>2</sup>   |
| Fragment Length           | km (distance from upstream dam to downstream dam)                  |
| <b>Land-Use</b>           |  |
| % Agriculture             | percent agriculture (NLCD)   |
| % Development             | percent developed land (NLCD)                                      |
| <b>Channel Morphology</b> |  |
| Gradient                  | gradient measured in field at 5-7 X bankfull width                 |
| Bankfull Width            | meters   |
| Entrenchment              | ratio: width of 2 X bankfull height/ bankfull width                |
| Bar Habitat Index         | ratio: area of bar coverage/stream reach area                      |
| <b>Substrate</b>          |  |
| D <sub>25</sub>           | 25 th percentile particle size                                     |
| D <sub>50</sub>           | median particle size   |
| Subsurface Index          | see methods  |
|                           | 0  |
| <b>Temperature</b>        |  |
| Temperature               | Celcius, average June-September Value                              |
| <b>Flow<sup>1</sup></b>   |  |
| 1-Day Maximum Flow        | residuals from flow vs drainage area curve for unregulated streams |
| 1-Day Minimum Flow        | residuals from flow vs drainage area curve for unregulated streams |
| No. of High Flows         | discrete number of high flow events per year                       |
| Low Flow Duration         | days per year where flow remains below 25 <sup>th</sup> percentile |
| Flood Interval            | annual median interval in days between floods                      |
| Reversals                 | no. of negative and positive changes in flow from 1 day to next    |

<sup>1</sup> All flow variables represent the average of all values generated for each year of the entire period of record

Table 5.2. Spawning substrate preference, taxonomic group, and guild response variables used in the study. Type of data represents the type of summary information used to calculate each response variable. “RA” indicates relative abundance, “%S” indicates percent species, and “Rich” indicates richness.

| <b>Fish Assemblage Response Variables</b>           | <b>Type of Data</b> |
|---|---------------------|
| <b>Spawning Strategy &amp; Substrate Preference</b> |                     |
| Sand and Gravel Spawners                            | RA, % S             |
| Gravel and Rock Spawners                            | RA, % S             |
| <b>Taxonomic Groups, Guilds, &amp; Other</b>        |                     |
| Benthic Insectivores                                | RA, Rich, % S       |
| Preference for Lentic Environment                   | RA, % S             |
| Centrarchidae                                       | RA, Rich, % S       |
| Percidae  | RA, Rich, % S       |
| Catostomidae  | RA, Rich, % S       |
| <i>Moxostoma</i>                                    | RA, Rich, % S       |
| Cyprinidae  | Rich                |
| <i>Notropis</i>                                     | Rich                |

Table 5.3. Best linear models for each richness (Rich) variable and the total and adjusted amount of variation explained ( $R^2$  and  $R^2$  adj, respectively). AICc refers to the corrected Akaike Information Criterion (Burnham and Anderson 2004) where the smallest value indicates the best model.  $C_p$  refers to Mallows  $C_p$  criterion for model selection. The model is chosen where  $C_p$  first approaches the number of model predictors (Mallows 1973). (+) or (-) refers to the direction of the effect of each predictor on the response variable (e.g. benthic insectivore richness is positively related to fragment length).

| Model   | $R^2$ | $R^2$ adj | AICc  | $C_p$ |
|---|-------|-----------|-------|-------|
| <b>Benthic Insectivores</b>                                 |       |           |       |       |
| Fragment Length(+), Entrenchment(-), Subsurface Index(+)    | 0.58  | 0.56      | 35.87 | 16.87 |
| Fragment Length(+), Entrenchment(-)                         | 0.49  | 0.46      | 43.38 | 27.89 |
| Fragment Length(+)  | 0.31  | 0.28      | 56.48 | 51.96 |
| <b>Centrarchid</b>  |       |           |       |       |
| Gradient(-), Temperature(+), % Development(+)               | 0.52  | 0.49      | 72.45 | 14.53 |
| Gradient(-), Temperature(+)                                 | 0.46  | 0.43      | 75.62 | 19.29 |
| Gradient(-)   | 0.31  | 0.28      | 85.33 | 34.57 |
| <b>Percidae</b>   |       |           |       |       |
| Fragment Length(+), Bankfull Width(+), Bar Habitat Index(+) | 0.66  | 0.64      | 50.23 | 21.88 |
| Fragment Length(+), Bankfull Width(+)                       | 0.61  | 0.58      | 54.90 | 29.69 |
| Fragment Length(+)  | 0.50  | 0.47      | 64.55 | 47.71 |
| <b>Catostomidae</b>   |       |           |       |       |
| Bankfull Width(+), D25(-)                                   | 0.53  | 0.50      | 52.16 | 3.66  |
| Bankfull Width(+)   | 0.33  | 0.30      | 67.46 | 21.87 |
| <b><i>Moxostoma</i></b>                                     |       |           |       |       |
| Bankfull Width(+), D25(-)                                   | 0.50  | 0.47      | 65.87 | -1.34 |
| Bankfull Width(+)   | 0.26  | 0.23      | 82.92 | 16.91 |
| <b>Cyprinidae</b>   |       |           |       |       |
| Fragment Length(+), Gradient(-), % Development(-)           | 0.54  | 0.51      | 50.25 | 21.48 |
| Fragment Length(+), Gradient(-)                             | 0.44  | 0.41      | 57.74 | 33.47 |
| Fragment Length(+)  | 0.27  | 0.24      | 68.51 | 54.77 |
| <b><i>Notropis</i></b>                                      |       |           |       |       |
| Fragment Length(+), Gradient(-), % Development(-)           | 0.55  | 0.52      | 44.54 | 23.24 |
| Fragment Length(+), Gradient(-)                             | 0.44  | 0.41      | 53.14 | 37.43 |
| Fragment Length(+)  | 0.30  | 0.27      | 61.80 | 55.51 |

Table 5.4. Best linear models for each % species (%S) variable and the total and adjusted amount of variation explained ( $R^2$  and  $R^2$  adj, respectively). (e.g. % species indicates % of species in fish assemblage that are benthic insectivores) AICc refers to the corrected Akaike Information Criterion (Burnham and Anderson 2004) where the smallest value indicates the best model. Cp refers to Mallows Cp criterion for model selection. The model is chosen where Cp first approaches the number of model predictors (Mallows 1973). (+) or (-) refers to the direction of the effect of each predictor on the response variable (e.g. % benthic insectivores is negatively related to temperature).

| Model  | $R^2$ | $R^2$ adj | AICc    | $C_p$ |
|--|-------|-----------|---------|-------|
| <b>Benthic Insectivores</b>                              |       |           |         |       |
| Temperature(-), Gradient(+), Reversals(-)                | 0.53  | 0.50      | -100.85 | 25.70 |
| Temperature(-), Gradient(+)                              | 0.45  | 0.42      | -95.43  | 35.27 |
| Temperature(-)   | 0.22  | 0.21      | -80.30  | 66.48 |
| <b>Centrarchidae</b>                                     |       |           |         |       |
| % Development(+), Temperature(+)                         | 0.53  | 0.50      | -64.30  | 3.77  |
| % Development(+), Temperature(+)                         | 0.43  | 0.39      | -56.94  | 12.03 |
| <b>Percidae</b>  |       |           |         |       |
| % Development(-), Fragment Length(+), Distance to Dam(-) | 0.49  | 0.46      | -93.23  | 2.06  |
| % Development(-), Fragment Length(+)                     | 0.40  | 0.37      | -88.00  | 7.40  |
| % Development(-)   | 0.24  | 0.21      | -78.35  | 19.36 |
| <b>Catostomidae</b>                                      |       |           |         |       |
| Entrenchment(-), % Agriculture(+)                        | 0.33  | 0.30      | -113.33 | 2.47  |
| Entrenchment(-)  | 0.23  | 0.20      | -108.66 | 7.49  |
| <b><i>Moxostoma</i></b>                                  |       |           |         |       |
| Bankfull Width(+), $D_{25}$ (-)                          | 0.41  | 0.38      | -67.47  | -4.85 |
| Bankfull Width(+)  | 0.24  | 0.21      | -56.80  | 4.83  |
| <b>Preference for Lentic Habitat</b>                     |       |           |         |       |
| % Development(+), Gradient(-)                            | 0.39  | 0.36      | -57.61  | 15.74 |
| % Development(+)   | 0.36  | 0.33      | -57.34  | 16.98 |
| <b>Rock and Gravel Spawners</b>                          |       |           |         |       |
| Gradient(+), Reversals(-), Bar Habitat Index(-)          | 0.61  | 0.58      | -373.86 | 19.90 |
| Gradient(+), Reversals(-)                                | 0.47  | 0.44      | -360.81 | 40.44 |
| Gradient(+)  | 0.32  | 0.29      | -350.81 | 62.14 |
| <b>Sand and Gravel Spawners</b>                          |       |           |         |       |
| Subsurface Index(+), Min Flow(-), Flood Interval(+)      | 0.58  | 0.55      | -324.23 | 31.16 |
| Subsurface Index(+), Min Flow(-)                         | 0.44  | 0.41      | -312.24 | 53.71 |
| Subsurface Index(+)                                      | 0.27  | 0.24      | -301.32 | 81.43 |

Table 5.5. Best linear models for each relative abundance (RA) variable and the total and adjusted amount of variation explained ( $R^2$  and  $R^2$  adj, respectively). AICc refers to the corrected Akaike Information Criterion (Burnham and Anderson 2004) where the smallest value indicates the best model. Cp refers to Mallows Cp criterion for model selection. The model is chosen where Cp first approaches the number of model predictors (Mallows 1973). (+) or (-) refers to the direction of the effect of each predictor on the response variable (e.g. benthic insectivores are negatively related to temperature).

| Model  | $R^2$ | $R^2$ adj | AICc    | Cp    |
|--|-------|-----------|---------|-------|
| <b>Benthic Insectivores</b>                          |       |           |         |       |
| Temperature(-), Bankfull Width(-)                    | 0.51  | 0.47      | -16.86  | 4.75  |
| Temperature(-)                                       | 0.45  | 0.41      | -13.57  | 8.59  |
| <b>Centrarchidae</b>                                 |       |           |         |       |
| Temperature(+), % Development(+)                     | 0.57  | 0.54      | -73.23  | 4.60  |
| Temperature(+)                                       | 0.31  | 0.28      | -52.84  | 30.62 |
| <b>Percidae</b>                                      |       |           |         |       |
| Fragment Length(+), % Development(-), Temperature(+) | 0.54  | 0.51      | -75.62  | 18.74 |
| Fragment Length(+), % Development(-)                 | 0.46  | 0.43      | -70.09  | 27.29 |
| Fragment Length(+)                                   | 0.39  | 0.36      | -66.27  | 34.69 |
| <b>Castostomidae</b>                                 |       |           |         |       |
| % Development(+), $D_{25}$ (-), Bankfull Width(+)    | 0.49  | 0.46      | -109.59 | 10.85 |
| % Development(+), $D_{25}$ (-)                       | 0.44  | 0.40      | -106.53 | 15.05 |
| % Development(+)                                     | 0.32  | 0.29      | -99.88  | 24.71 |
| <b><i>Moxostoma</i></b>                              |       |           |         |       |
| Bankfull Width(+), $D_{25}$ (-)                      | 0.51  | 0.48      | -122.55 | -8.35 |
| Bankfull Width(+)                                    | 0.27  | 0.24      | -104.67 | 7.44  |
| <b>Preference for Lentic Habitat</b>                 |       |           |         |       |
| Gradient(-), $D_{25}$ (-), Temperature(+)            | 0.36  | 0.33      | -101.96 | 6.84  |
| Gradient(-), $D_{25}$ (+)                            | 0.29  | 0.26      | -99.44  | 9.98  |
| Gradient(-)  | 0.25  | 0.22      | -98.39  | 11.80 |
| <b>Rock and Gravel Spawners</b>                      |       |           |         |       |
| Fragment Length(-), Low Flow Duration(-)             | 0.27  | 0.24      | -32.03  | 1.97  |
| Fragment Length(-)                                   | 0.19  | 0.16      | -29.22  | 4.99  |
| <b>Sand and Gravel Spawners</b>                      |       |           |         |       |
| Fragment Length(+), Max Flow(+), Min Flow(-)         | 0.60  | 0.57      | -57.33  | 28.80 |
| Fragment Length(+), Max Flow(+)                      | 0.47  | 0.43      | -45.20  | 50.83 |
| Fragment Length(+)                                   | 0.26  | 0.23      | -31.43  | 84.95 |

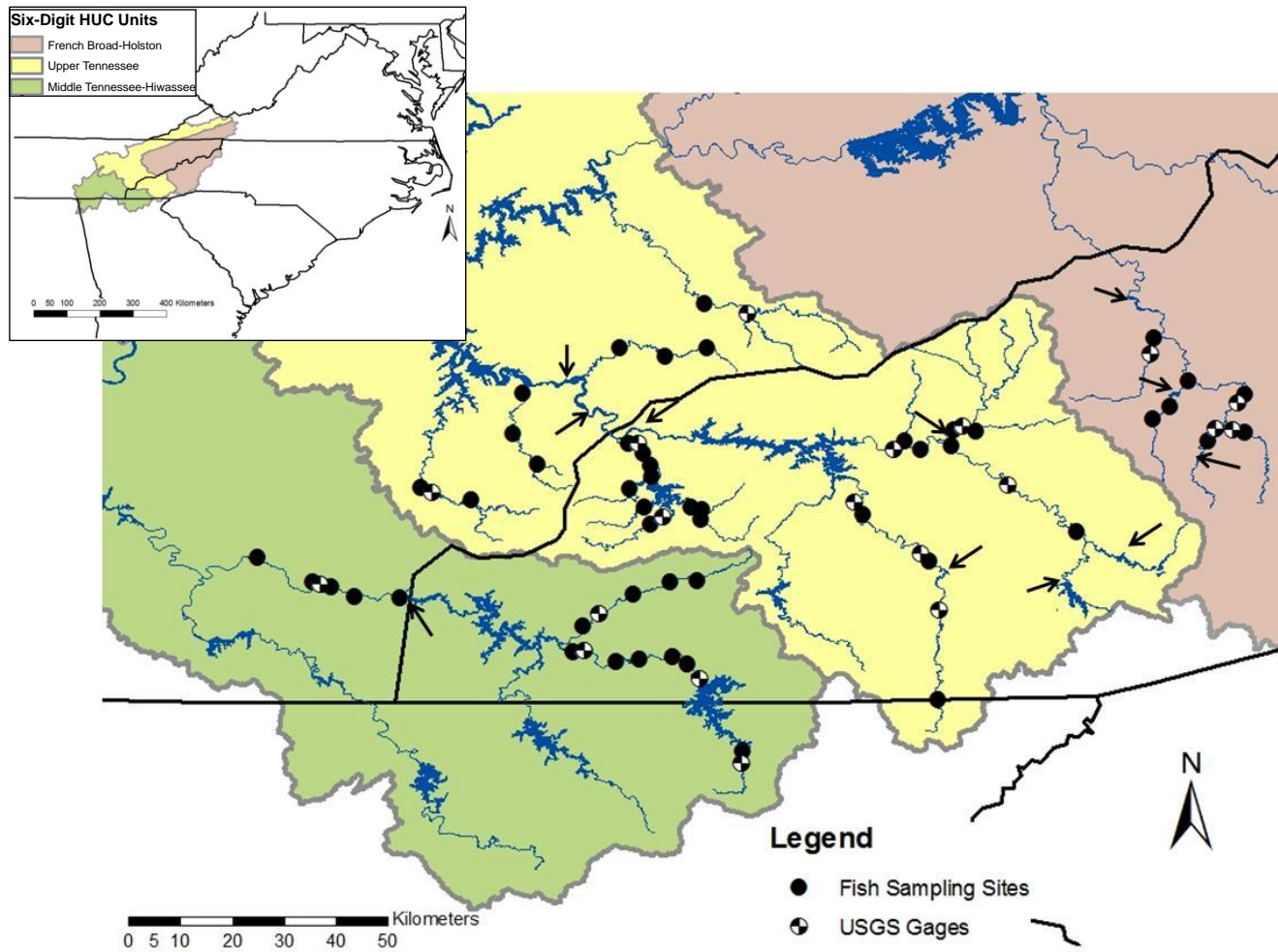


Figure 5.1 Sites and US Geological Survey gage locations in the study area in the Upper Little Tennessee, Upper French Broad, and Hiwasee drainages. Arrows indicate locations of dams that may not be obvious otherwise due to low reservoir storage. “HUC” indicates Hydrologic Unit Codes.

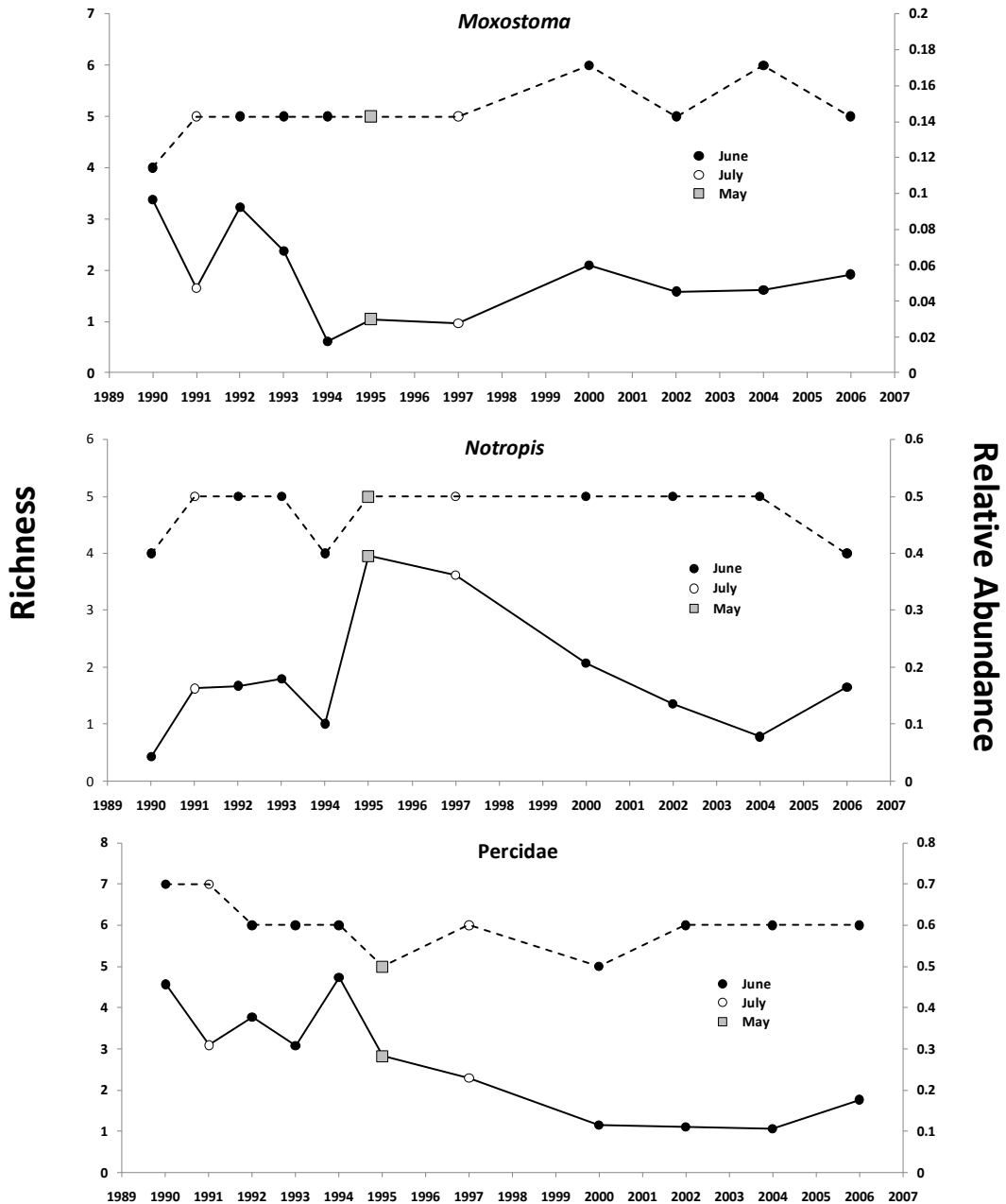


Figure 5.2 Temporal trends in the species richness and relative abundance of *Moxostoma*, *Notropis*, and Percidae found during sampling conducted in May, June, and July in the Little Tennessee River at Needmore, NC.



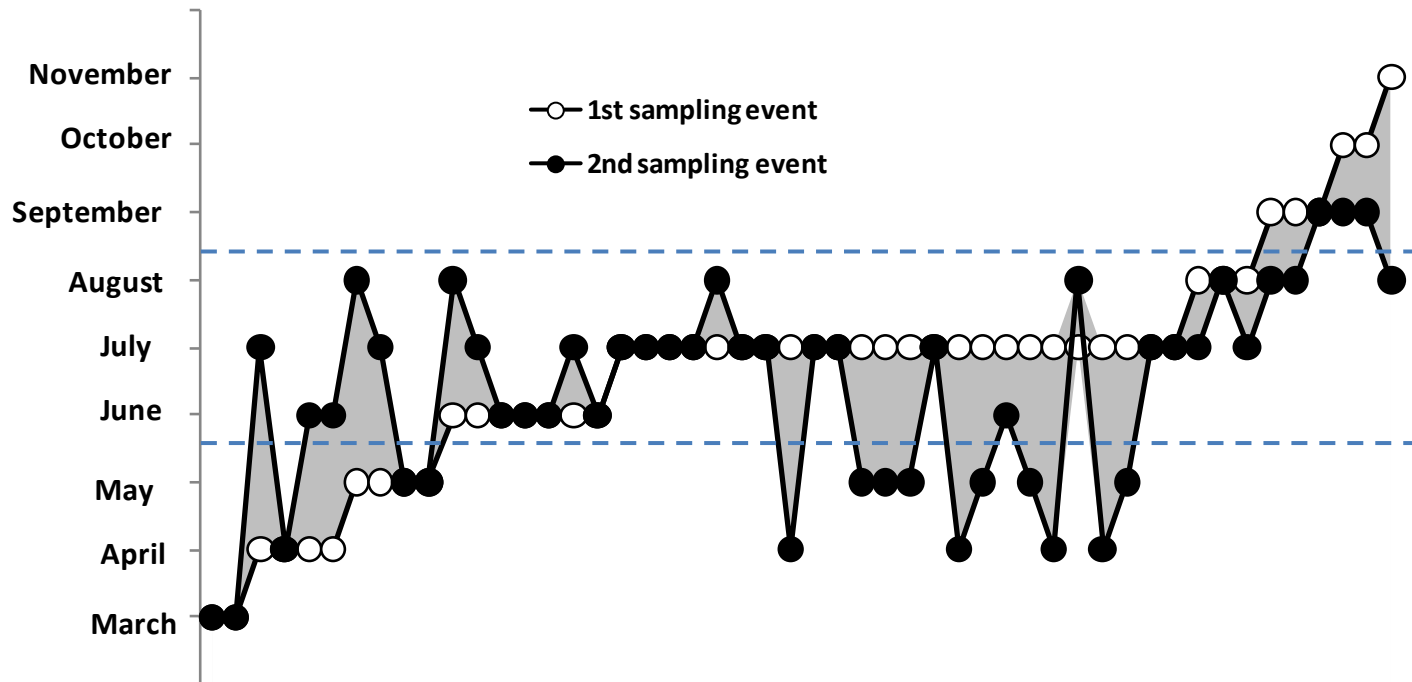


Figure 5.3 The month in which sampling was conducted for the latest two sampling occasions at all 50 sites in this study. Sites were sorted in order of the month in which the 1<sup>st</sup> sampling occurrence was conducted. The month in which the 2<sup>nd</sup> sampling occurrence was conducted was then overlaid over its corresponding 1<sup>st</sup> sampling event. The shaded region represents the area of overlap between sampling time periods. The area between the dashed lines represent the summer time period.

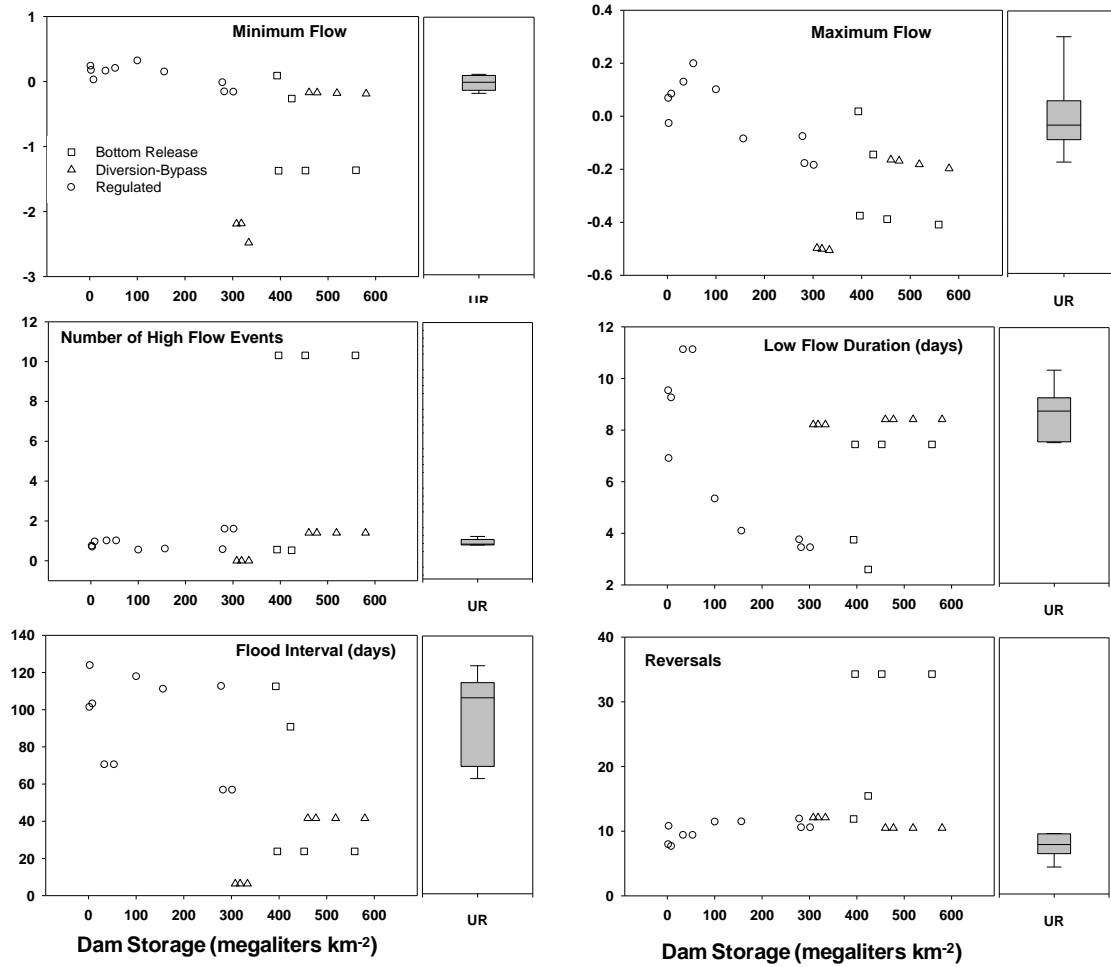


Figure 5.4 Relationships between six flow variables and dam storage for within three different dam operation types. Box and whisker plots represent unregulated stream values (“UR”) for each variable.

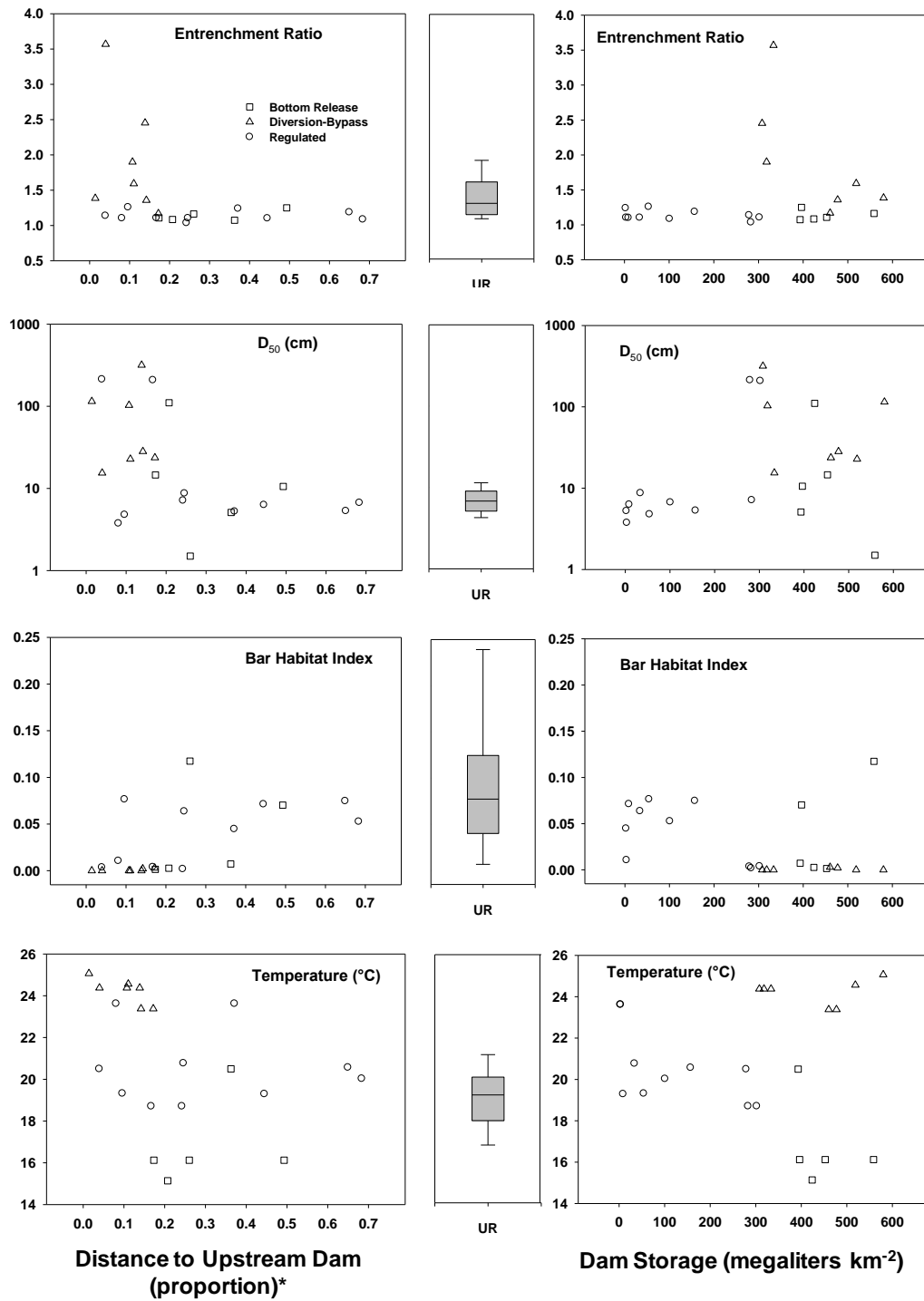


Figure 5.5 Relationships between physical variables (entrenchment ratio, median particle size ( $D_{50}$ ), bar habitat index (see methods), and temperature) and dam storage and distance from dam within three different dam operation types. Box and whisker plots represent unregulated stream values (“UR”) for each variable. \* proportion refers to the distance to nearest dam divided by total stream network distance (see methods).

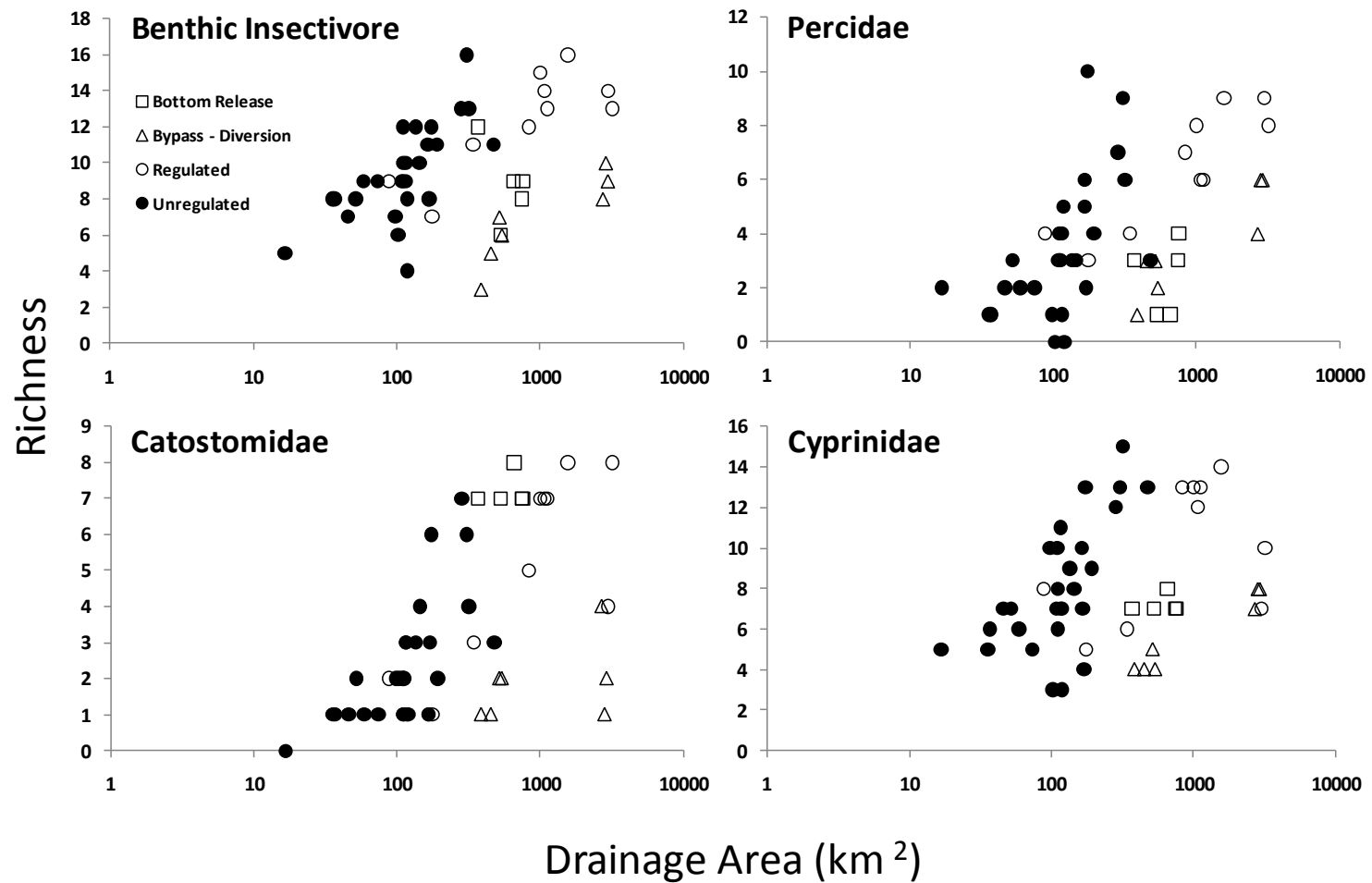


Figure 5.6 Benthic insectivore, Cyprinidae, Percidae, and Catostomidae species richness versus drainage area for unregulated and regulated sites. Sites were divided into unregulated (filled symbols) and regulated (open symbols). Regulated sites were further divided into dam operation types.

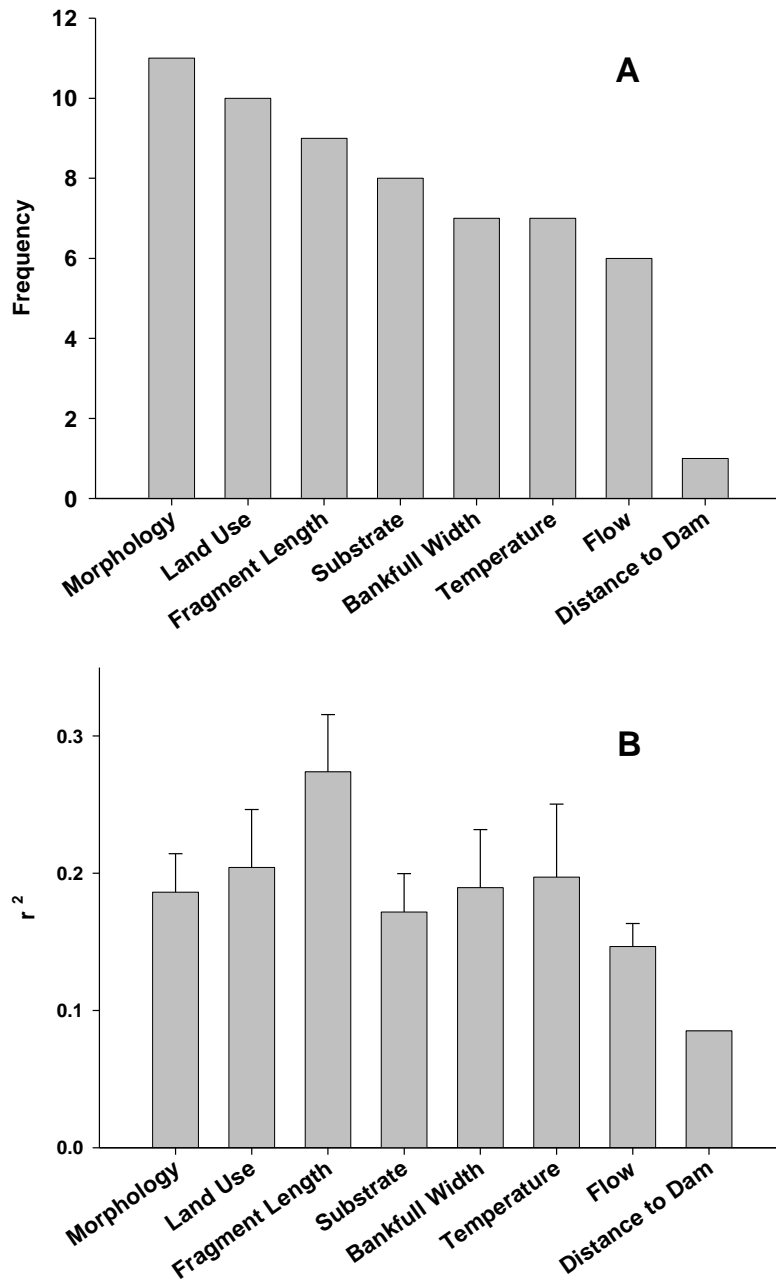


Figure 5.7 Frequency that various categories of variables were used in the linear models to predict biological response variables and the partial  $r^2$  values representing the average amount of variation explained by each variable in the models in which they were found.

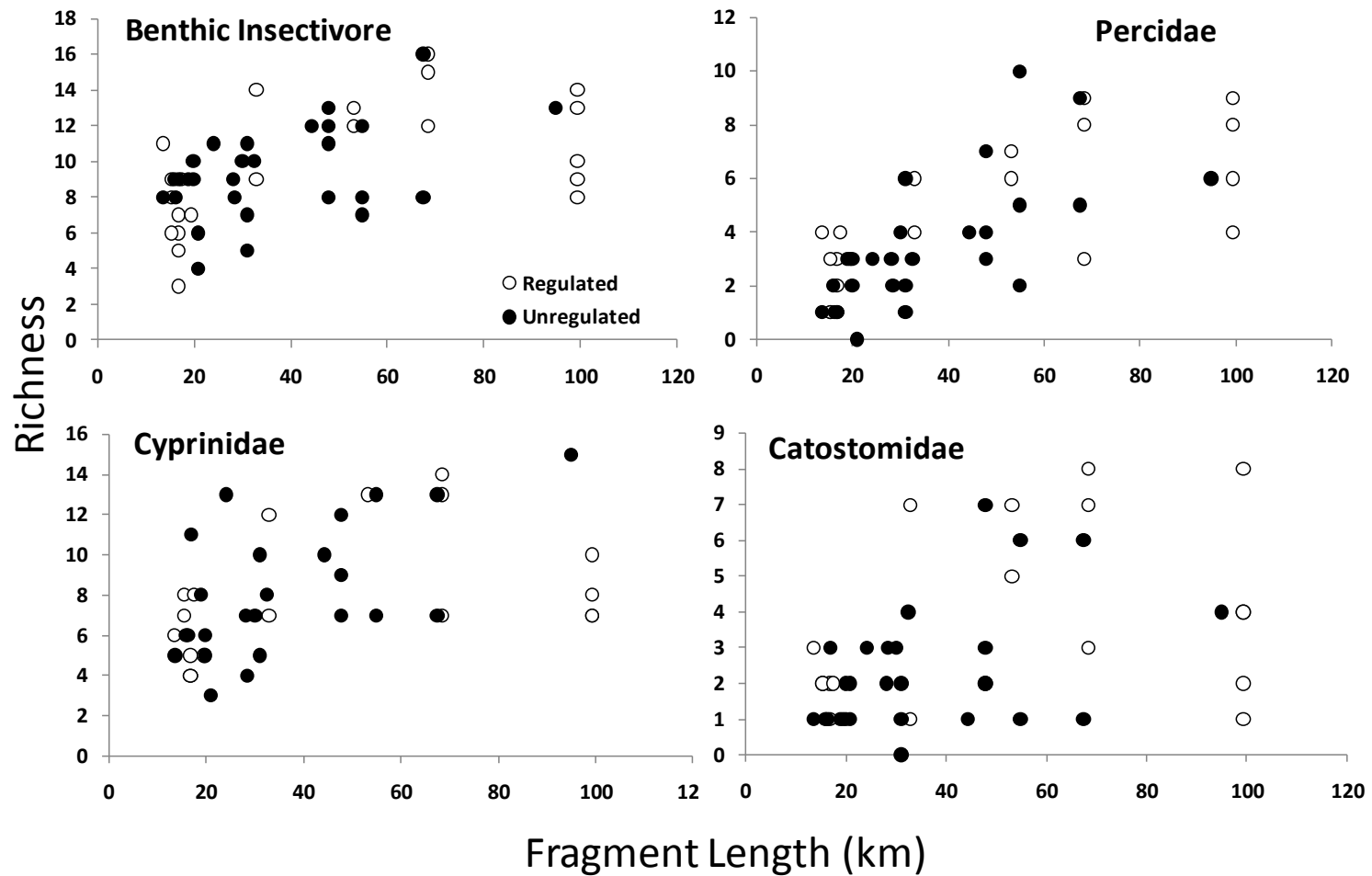


Figure 5.8 Benthic insectivore, cyprinidae, percidae, and catostomidae species richness versus fragment length for unregulated and regulated sites. Sites were divided into unregulated (filled symbols) and regulated (open symbols). Regulated sites were further divided into dam operation types.

## 6. Case Study: Application of the Restoration Framework to the Cheoah River, a Regulated System in the Little Tennessee River Basin.

### 6.1 Introduction

Many of the ideas in this dissertation, including the creation of restoration frameworks for regulated rivers, stemmed out of the frustrations of researching a river with very little pre-disturbance information. Insufficient knowledge of appropriate restoration endpoints (i.e. pre-disturbance state) can obviously be problematic when developing conservation objectives (Tear et al. 2005). The Cheoah River Case Study provides an excellent example of the hardships faced when conducting research to inform management, especially restoration applications that depend on defining appropriate endpoints through best available science. In 2005, a new flow regime and gravel enhancement plan was established through a Federal Energy Regulatory Commission (FERC) license and settlement agreement for the Cheoah River, a regulated system in the Little Tennessee basin (Tapoco Hydroelectric Project, FERC #2169: FERC 2005, FERC 2006). Since Santeetlah Dam's construction in 1927, the majority of flow in the Cheoah River was limited to leakage from the dam and input from tributaries. In addition, sediment transport into the system was eliminated by the dam, but large spill events still provided significant transport capacity (more background information discussed later). The central restoration objectives, if possible, were to restore the historical fish assemblage, macroinvertebrate community (primarily an endangered mussel), and riparian species through reintroductions and habitat enhancement (flow restoration and gravel additions) (Mark Cantrell (US FWS), Steve Reed (NC DWR), *personal communication*).

*Here I describe limitations of researching the Cheoah River and the main reasons that I developed the objectives contained in this dissertation:*

One of the top research priorities for Virginia Tech investigators concerning the Cheoah River was evaluating the influence of the new flow regime on the fish assemblage. Thus, it became important to understand what fishes were expected to be found in the Cheoah River to properly understand how the fish community may respond to changes in flow. However, for the Cheoah River, the earliest fish assemblage information was collected in 1981 (Bonner 1983) whereas Santeetlah Dam has been in operation since 1927. Fish assemblage sampling was conducted on an inconsistent basis starting in 1993 whereas fairly consistent (annual or bi-

annual) fish assemblage sampling was conducted from 2003-2009 (all using various sampling methodologies). In order to understand whether the flow restoration had improved specific aspects of the natural hydrograph, comparisons of pre- and post-dam hydrologic regimes were necessary since at least 20 years each of pre- and post-impoundment data is generally required (Richter et al. 1997). However, pre-impoundment hydrologic information is only available for 12, non-continuous years and was only limited to average daily discharge records (no peak flow records). In addition, current USGS records for the Cheoah River only span from 1999 to current. Thus, in both of these cases, the best available science (at least for my research) was limited due to insufficient historical (pre-disturbance) information and insufficient current information. However, the agencies within the relicensing processes used simulated hydrographs based on reservoir inflow, occasional spill events, and neighboring watersheds to reconstruct a hydrograph for the Cheoah River. Data limitations are the realities of restoration and conservation management. Matter of fact, the collaboration between agencies of the Cheoah Fund Board has resulted in management actions that have been the epitome of ecosystem management – adaptive, holistic, and relevant to time/spatial scales operating at the ecosystem scale (Christensen et al. 1996).

However, one main complication in the Cheoah River Case study was inadequate communication and short timeframes. The second main research objective was to evaluate the influence of gravel additions on fish spawning habitat and the macroinvertebrate community. The FERC deadline for the first gravel addition was March 1, 2008, which had been established as an extension. Virginia Tech investigators were unaware of the deadline until December 2007 at which time there was no hope for an additional time extension. Thus, prior to the first gravel addition, there was no time to plan or conduct extensive pre-restoration biotic surveys. In addition, because of the rapidly approaching FERC deadline, agency personnel had to quickly choose locations for gravel additions. Locations were limited to Alcoa-owned property because of strictly enforced permitting issues on Forest Service land. Thus, chosen locations for enhancing habitat for target biota (mussels, darters, catostomids) were compromised.

Short time frames influenced research and management options in indirect ways as well. During the relicensing process from 1999 to 2003, at least five different consultant firms were contracted to conduct research on various aspects of management in the Cheoah River. The work conducted by the contractors was hypothesis-driven, highly technical, extremely



informative, and consisted of some of the latest technologies. The research included instream-flow-incremental methodologies (IFIM), predicting bedload transport, and the development of sub-foot-resolution geospatial data layers. In my opinion, it was the best available science. However, the massive amount of information gathered was possibly too much information to be synthesized during the relicensing timeframe. Normandeau Associates conducted more work on the Cheoah River than any other consulting firm; thus, it is possible to suggest that their field staff and researchers spent the most time on the river prior to 2005. One of the leading scientists for Normandeau, Doug Neiman, had developed a 257-page “Stream Habitat Fragmentation Evaluation” draft that was never fully completed due to his illness and eventual passing. The report included complex evaluations of species distributions in relation to fragmentation and habitat connectivity, applications to metapopulation theory, and the application of fish traits to understand vulnerability to fragmentation. It includes some of the most novel approaches to conservation management in fragmented systems in which I have not observed in the leading scientific literature. Ultimately, Doug’s work had developed a template for recreating a realistic fish assemblage for the Cheoah River given the current constraints and a framework for a holistic long-term biological management plan. I believe that the time frame in which the FERC relicensing committee operated limited the ability of agencies to collectively digest the available information and develop a holistic picture of the major limitations in the Cheoah River system. In other words, 6 years (1999 to 2005) is not enough time to understand the complexities of a river system modified by 75 years of diversion. However, the new FERC license was issued for a 40-year timeframe.

Altogether, there were some major limitations imposed by the lack of historical data and insufficient time frames to foster appropriate scientific investigation. Therefore, I knew I had to alter my approach. Because I had only 3-4 years to develop a dissertation and because answering major questions concerning the Cheoah required substantially longer temporal scales, I focused efforts broadly across larger spatial scales. In other words, I had to look outside the watershed of the Cheoah River. Thus, my dissertation is focused on developing restoration frameworks by evaluating patterns in hydrology, morphology, and fish assemblages in disturbed and undisturbed river systems across various spatial scales. However, I still conducted field work to evaluate the influence of the flow restoration and gravel addition on morphological conditions in the river and on the fish assemblage. Therefore, since the Cheoah inspired

(actually forced) the development of this broader framework for restoration, it seems fitting that I would conclude my dissertation by applying the restoration template to the waters of the Cheoah River based on observations that I have made from 2008 to 2010.

I use the conceptual model, outlined in Figure 1.1, to provide a template for my observations of the Cheoah River. Because the Cheoah River has poor historical records on pre-dam disturbance flow conditions, I use the flow classification to provide some assessment of whether the flow restoration in the Cheoah matches the natural flow regime of its corresponding unregulated flow class. First, I use the watershed classification tree to assign the Cheoah to an appropriate flow class. I then compare the pre- and post-flow restoration regime of the Cheoah to unregulated counterparts in order to make recommendations for improved environmental flows.

Step 5 of the conceptual model (Fig 1.1) concerns the relationship between altered river hydrology and morphology and the implications for fish assemblages. I focus the majority of my discussion on the implications of altered hydrology and morphology since the interaction between these two factors may have imposed strong influences on the fish assemblage in the Cheoah River. However, I discuss other factors that impose greater limitations on restoration in the Cheoah River, such as habitat fragmentation, channel morphology, and the temperature regime. Ultimately, using the Cheoah River as a case study highlights both the strengths and the weaknesses of the restoration template.

## **6.2 Study Site Description**

The Cheoah River is a regulated system located in western North Carolina within the Blue Ridge physiographical province (Fig. 6.1). The Cheoah River is impounded by Santeetlah Lake, a 456 km<sup>2</sup> reservoir, and flows 14.6 km before emptying into the Little Tennessee River System downstream of Cheoah Reservoir. The 143 km<sup>2</sup>, predominately-forested watershed is primarily located within Nantahala National Forest. The area generally receives 150 to 230 cm of precipitation annually. The remaining Cheoah River is a high gradient system, falling from 533 m at the dam to less than 335 m over its length (slope~1.3 %). Valley relief is relatively steep, approximately 30% grade. Geology is dominated by gneiss, sandstone, and granite (Normandeau Associates 2002a). In general, the Cheoah River is constrained by its valley and underlying bedrock along with a steep and high embankment from the road leading to very little

lateral migration (Normandeau Associates 2002a). The upper 2 miles of the river are dominated by bedrock and large boulders ( $D_{50} = 370$  mm) and have a relatively low gradient (0.3 – 0.6%). The lower 7 miles generally has a steeper gradient (1 – 2%) and although gravel and cobble substrates tend to increase with distance from the dam; the streambed is still very coarse ( $D_{50} = 230$  mm) and sediment starved (R2 2003).

### *Background*

The completion of Santeetlah Dam in 1927 substantially altered the sediment supply and hydrology of the Cheoah River. Because of surface-release operations, sediment supply has been cut-off from entering the Cheoah River and is limited to tributary input and episodic landslides below Santeetlah Dam (Normandeau Associates 2002a, Dilts et al. 2003). Prior to 2005, flow from Santeetlah Dam into the lowermost 14.6 km of the Cheoah River was limited to leakage from the dam ( $< 0.002$  cms), inputs from tributaries, and occasional large pulses ( $> 24$  cms) from the reservoir. Because of low-flow conditions, riparian vegetation has encroached much of the upper Cheoah River, which has locked up finer substrate (Normandeau Associates, 2002a). The high gradient of the Cheoah River only intensified sediment-starved conditions below the dam. Altered hydrology and sediment supply led to degraded habitat for many aquatic biota, including the federally protected plant, Virginia spiraea (*Spiraea virginiana*), and the federally endangered Appalachian elktoe (*Alasmidonta raveneliana*) (USFWS 1994). Historically, the Cheoah River may have had over 40 fish species (Jenkins, R. and D. A. Etnier, *personal communication*). The Cheoah River, currently, is a cool/warm-water system with 19 species of fish (Years 2008 and 2009 combined, Table 6.2, Appendix D).

To remediate the effects of habitat degradation, a settlement agreement with natural resource agencies along with corresponding FERC orders in 2005 and 2006 required Alcoa Power Generating, Inc to 1) provide a seasonally variable streamflow regime punctuated by higher flow events, and 2) develop an adaptive management plan to add and monitor gravel on a biannual basis (FERC 2005, FERC 2006). The relicensing process was a collaborative effort between Alcoa Power, USDA Forest Service, US Fish and Wildlife Service, North Carolina Wildlife Resources Commission, NC Division of Water Resources-DENR, and many other interested parties. Consultant groups were contracted to assess existing conditions in the river and make recommendations for future management actions including environmental flow

prescriptions and substrate augmentation (Normandeau 2002a, R2 2002, Dilts et al. 2003).

The Federal Energy Regulatory Commission issued the new 40-year license in effect March 1, 2005 (FERC 2005). The license included requirements for seasonally variable base flows between 40 cfs and 100 cfs (1.13 to 2.83 cms) along with periodic high flow events (1000 cfs or 28.32 cms) to enhance aquatic diversity (FERC 2005). An evident “jump” in the mean monthly dam spillage indicated that flow in the upper river was heavily modified following the initiation of the new flow regime on September 1, 2005 (Fig. 6.2 top). However, flow at the USGS gage in the lower river displayed moderate increases in mean monthly flow following 2005 (Fig. 6.2 top). Likewise, annual minimum flows showed very little change at the USGS gage in the lower river following the initiation of the new flow regime in 2005 (Fig. 6.2 bottom).

The FERC license also specifically required that 1) 76.5 m<sup>3</sup> of gravel must be supplemented on a bi-annual basis to the lower river reaches and 2) that monitoring of the effects of flow and substrate enhancement should be initiated. For more information regarding consultant recommendations and FERC orders, refer to McManamay et al. (2010). In addition to physical restoration, the Conservation Fisheries Inc. propagated and then reintroduced wounded darters (*Etheostoma vulneratum*) and spotfin chub (*Cyprinella monacha*) into the Cheoah River in multiple areas and currently monitor their status (Rakes et al. 1999; Russ and Fraley, 2009; Petty et al., 2011).

### **6.3 Assessing hydrologic alterations– Steps 2-4**

The flow classification dataset (Chapter 2) is useful in that environmental flow standards and policies for states or regions can be based upon unique classes rather than defining “natural” baselines for each river. Secondly, recommendations can easily be developed by qualitative or quantitative comparisons between the flow regime of a particular river and that of its respective flow class. This approach is advantageous in situations where no flow records are available or they are insufficient to establish baseline conditions. If baseline data are unavailable, the watershed classification tree can be used to classify a river into an appropriate natural flow class. To provide an example of the utility of this dataset, I compare the current (post FERC relicensing agreement) and past (pre FERC relicensing agreement) flow regime of the Cheoah River to that of its respective flow class in order to form environmental flow recommendations. Fortunately, in the case of the Cheoah River, historical baseline data (although limited) was available prior to

regulation; thus, we include the pre-regulation flow data in the analysis as an accuracy assessment of the performance of environmental recommendations formed from the flow classification.

*Using the Watershed Classification Tree to Designate a Natural Flow Class* - When baseline flow conditions do not exist for a given river, the watershed classification tree (Chapter 3) can be used as a first step in designating an appropriate flow class. To simulate a realistic situation, I used the watershed classification tree to designate the appropriate flow class for the Cheoah River. I used the same data layers (e.g. STATSGO soil, PRISM climate data) as Falcone et al. (2010) to summarize variable values across the Cheoah's basin. I summarized the averages of each data layer's values (e.g. soil bulk density) across the entire drainage to use as values for primary splitting variables in the watershed classification tree. Using the watershed tree, the Cheoah River was classified as a stable high baseflow 1 (SBF1) stream.

*Comparing the Cheoah River to its Respective Natural Flow Class* – USGS daily and peak flow data for the Cheoah River gage (0351706800) was available from 1999 to present. Also, pre-impoundment daily flow data (1912-1918, 1920-1926) was available for the Cheoah River at the Johnson, NC gage (03517000). Although the data are from a slightly different gage locations, both are within the same reach and our hydrologic indices were standardized by the median daily flow; thus, maximum or minimum flows are relative to median conditions and can be comparable across and within watersheds. Secondly, the Johnson, NC gage's drainage area (458 km<sup>2</sup>) was very similar to that of the current gage for the Cheoah (533 km<sup>2</sup>). To evaluate the effectiveness of the FERC relicensing project, we divided the current data (1999-2010) into two datasets: pre-flow restoration (1999-2005) and post-flow restoration (2005-2010). Generally, at least 20 years of data is recommended to for pre/post comparisons (Richter et al. 1997). However, in many situations, extensive datasets may not be available, such as the 12 years of pre-impoundment data in this study. Hence, managers are forced to use the best knowledge at hand to make decisions. The Cheoah River, then, provides a realistic example managing complex systems using limited background information. Although not ideal, I assume that a 5-year post-restoration period is sufficient at least to provide an example of the utility of the dataset.

Pre- and post-restoration data and pre-impoundment data (Johnson gage) were imported into the Hydrologic Index Tool (HIT) software and 171 indices were calculated. Magnitude

variables in my dataset were standardized by dividing by the median daily flow, which has many advantages, such as ensuring classifications are unbiased with respect to river size or providing the ability to compare flow regimes of rivers despite differences in drainage area. However, examining flows relative to other flows (e.g. peak flow relative to median flow) could be potentially confusing and I urge caution in drawing broad conclusions prior to examining specific aspects of the dataset. Furthermore, using standardized variables isolates specific hydrologic problems that otherwise may not be obvious.

I compared hydrologic statistics of the pre-restoration, post-restoration, and pre-impoundment (Johnson gage) flow periods to that of the stable high baseflow 1 class using box-whisker plots (Fig. 6.3). In general, values of hydrologic statistics in the SBF 1 class overlapped very well with those calculated for the Johnson gage, suggesting that environmental flow standards developed from the flow class alone could be fairly accurate. Even in situations where the Johnson gage would be considered an outlier in the SBF 1 dataset, the direction of the environmental recommendation would still be accurate. For example, using solely the SBF 1 dataset, one flow recommendation would be to decrease the rise and fall rates (Fig. 6.3). Although rise and fall rates for the Johnson gage are higher than the SBF 1 class, the direction (decrease) of the recommendation is correct. In general, adjustments to the post-restoration flow regime are recommended, such as a decrease in several factors: daily flow variability, minimum July Flow, frequency of high flow events, rise and fall rates, and the annual minimum. Interestingly, the establishment of the variable baseflow in the Cheoah, may have led to baseflow levels that were too high relative to the median daily flow.

Interestingly, I observed what seemed to be a contradiction in patterns in the annual maximum flow. According to Fig. 6.3, the annual maximum flow standardized by the *median daily flow* is very high in the post restoration time period relative to the Johnson Gage and that of the SBF1 class (Fig. 6.3). However, I also observed that the annual maximum flow standardized by *catchment area* is very low for the post-restoration time period relative to the Johnson gage and falls well within the SBF1 class (Fig. 6.3). The contradiction, arises because the current magnitudes of the median and mean daily flows for the Cheoah are substantially lower than of the Johnson gage (Fig. 6.3, bottom middle), yet annual maximum flow magnitudes are still very high. Thus, the large disparity between high flow magnitudes and daily flow magnitudes results in a skewness in the distribution of flows (Fig. 6.3, bottom right and Fig. 6.4). For example, log

transformed daily discharges from the Johnson gage tend towards a normal or even bi-modal distribution, where the median and mean are close (Fig. 6.4). However, due to the establishment of a minimum baseflow, the daily flows from the Cheoah are highly skewed resulting from a higher frequency of lower magnitude flows (shifting the median to the left) but still very high peak flows (shifting the mean to the right) (Fig. 6.4). An obvious recommendation is that skewness should be adjusted by decreasing the disparity between median and high peak flows (i.e. increasing the frequency of moderate flows, lowering high peak flows, and lowering the annual minimum even further).

If magnitudes variables are restored to their unstandardized states, relationships between the magnitude of various flows and drainage area can then be developed for each class, from which recommendations of specific magnitudes can be based (Fig. 6.5). However, this will only work if sufficient volumes of water from the reservoir are available for releases, as in the case of some storage reservoirs and hydroelectric facilities. In the case of the Cheoah River, over 50 % of the water that historically would have flowed through the river is now diverted around it. Restoring the full magnitude of peak flows without restoring the magnitude of more frequent flows can lead to very large disparities between peak and lower magnitude flows (i.e. skewness). Ultimately, extreme changes in fluvial habitat result in disturbances to which aquatic biota may be poorly adapted (Bunn and Arthington 2002). Although less frequent peak floods are channel-forming flows, lower-magnitude peak flows that occur at a higher frequency are channel-maintenance flows (Trush et al. 2000). Thus, for many rivers, using relative magnitudes may be a way to ensure that flows are interrelated, which should be a premise of the natural flow regime. In the case of the Cheoah River, given the volume, a few less obvious recommendations would be that the frequency and magnitude of spring moderate flows could be increased, and the magnitude of summer minimum flows (summer baseflows) decreased, thereby decreasing skewness. Of course, a different flow regime will interact differently within historical channel and should certainly be a consideration in terms of fluvial habitat.

#### **6.4 Interactions between flow and morphology – Step 5**

Following impoundment, river channel migration and the magnitude of episodic flows generally decrease, both of which lead to the encroachment of riparian vegetation (Gordon and

Meentemeyer 2006). This can be intensified for diversion projects where large volumes of water are diverted around channels. For example, extreme low-flow conditions lead to riparian encroachment within the historical active channel, leading to dramatic decreases in active channel width, and the establishment of a new floodplain (Fig. 6.6).

### *Effects of flow restoration on morphology*

The restoration of natural flow variability (i.e. peak flows and a higher minimum base flow) should restore some of the processes of active channel maintenance, such as habitat formation and scouring encroaching vegetation. I compared photographs from 2001 (pre-flow restoration) to photographs taken in 2008 (post-flow restoration) at monuments that were established by Normandeau Associates (Normandeau 2002a) (Fig. 6.7). Clearly, large decreases in riparian vegetation were observed following flow restoration (Fig. 6.7). However, some of the large decreases in riparian vegetation were due to active riparian removal from NC state agencies in 2005 (*personal communication*, Steve Reed, NC DWR). Nonetheless, peak flows have removed vegetation and have inhibited re-establishment since 2005. For example, following a large peak flood event of 8200 cfs (232 cms) (Fig. 6.8), I observed overturned vegetated islands within the wetted channel (Fig. 6.9). The 8200-cfs flood event was the largest peak magnitude during the course of my study (August 2007 – Dec 2010) (Fig. 6.8). Prior to this large disturbance, I had observed very little break-up of encroached riparian vegetation despite some floods exceeding 6000 cfs (170 cms). Thus, discharges ranging from 6500 to 8000 cfs (184-226 cms) may be a threshold for the Cheoah River, which once exceeded, may surpass the critical shear stress needed for riparian removal.

Sediment supply, streambed particles, discharge, and channel slope are all intricately balanced in a river system (Gordon et al. 2004). Thus, rivers respond to reduced sediment supply by stream bed armoring below dams (coarsening of the stream bed surface particles) (Gordon et al. 2004). Riparian encroachment can intensify substrate conditions by locking up potentially mobile particles. Thus, the removal of encroached riparian vegetation (especially in-channel vegetation) should mobilize sediment, which is observed via streambed textural fining. Secondly, the inundation of new stream margins should also increase the amount of sediment in the channel via erosive processes. I compared pebble counts conducted at eight transects before (2002) and after (2008) flow restoration. I found that there were general decreases in the  $D_{50}$  in



median particle size following flow restoration in the first 9.5 km from the dam downstream (Table 6.1). However, after 11 km from the dam, I observed a coarsening of the stream bed. Although the sample size is limited, the data suggest that there has been a textural fining in the upstream reaches. However, in the downstream reaches, the increase in flow magnitude may have increased shear stress on the stream bed, which may have offset any new deposits of material and led to overall streambed coarsening.

### *Morphological restoration*

Although there are some morphological benefits of flow restoration in the Cheoah River, there is still room for improving channel conditions (either by time or active restoration). Gravel additions have occurred and will continue to occur in the Cheoah River (McManamay et al. 2010). McManamay et al. (2010) reported that following the initial augmentation, gravel transported into the river channel and was incorporated into the stream bed after a series of high flow events. However, we also reported that sand, gravel, and smaller cobble particle sizes were still very deficient in the Cheoah River relative to comparison streams and at least twice as much substrate is needed at each site in shorter recurrence intervals to meet the transport capacity needs of the river under the new flow regime (Fig. 6.10). Other morphological restoration is still needed. For example, riparian vegetation is currently found in the active stream channel in many upstream reaches of the river (Fig. 6.11). In addition, riparian vegetation is established on historically-active bars (Fig. 6.11), which once were a source of fine sediments and gravel to the channel. According to Chapter 5, the Cheoah River has very few active bars, which may provide habitat for endemic riparian species. Bars are generally formed during high flow events by deposition of sediment load after being thrust upwards by longitudinal undulations in the stream bed (i.e. riffles) (Leopold 1994). Thus, bars might also buffer the shear forces on the stream bed by absorbing or diverting water into the floodplain during peak flow events.

River channels below dams constructed decades ago are many times not the same channel they were prior to dam regulation. For example, I compared a photo of the Cheoah River channel just following dam construction (early 1930's) to a recent photo from 2009 (Fig. 6.12). The current active channel in the Cheoah is only 2/3 the width of the historical channel, which agrees with the conceptual model in Figure 6.6. Because of decades of riparian encroachment, streambed degradation, and road construction, the active channel resembles more

of a conduit or pipeline shape than that of a trapezoid (Fig. 6.12). Wider channels may provide more habitat for fluvial specialists and riffle-dwelling species whereas narrow channels increase depth and velocities. Therefore, the morphological consequences of dam regulation could explain how hydrologic alterations may result in habitat favorable to active swimmers (Carlisle et al. 2010b). Therefore, prior to restoring flows, it may be important to understand how current morphological conditions differ from historical conditions in order to appropriately design restoration applications.

## **6.5 What about the fish? – Step 5**

### *Overall patterns in the fish assemblage*

As mentioned earlier, narrower channels promote deeper habitats and deeper habitats should favor active swimmers (Dauwalter et al. 2007; Craven et al. 2010). Not surprisingly, active swimmers, such as whitetail shiner, warpaint shiner, and smallmouth bass, dominate the fish assemblage in the Cheoah River (68-80% of the relative abundance). Crevice spawners (whitetail shiners) have the highest relative abundance in the Cheoah River, especially in the upper reaches, compared to other streams in the region. This seems to be primarily due to a combination of high temperatures and coarse substrates (Chapter 5). Habitat generalists, such as centrarchids, tend to favor slack water environments and conditions below dams. Accordingly, centrarchids make up 21% of the fish assemblage in the Cheoah River compared to only 6% on average in other streams in the region. Benthic insectivores are less common in the Cheoah River averaging 8% of the relative abundance compared to an average of 59% at the other sites across the Upper Tennessee basin. Interestingly, *Notropis* species have never been found in the Cheoah River (Chapter 5). Species currently found in the Cheoah River are listed in Table 6.2. Another group of fish that is missing from the Cheoah River (and the majority of its tributaries below the dam) but is found in many neighboring rivers is *Cottus* species.

### *Effects of the flow restoration on the fish assemblage*

One of the key components of assessing the effectiveness or the success/failure of restoration is through monitoring. The flow re-regulation included in the FERC license was initiated September 1, 2005 and included a seasonally variable base flow between 40 and 100 cfs

(1.13-2.83 cms) along with periodic higher flow events (1000 cfs or 28.3 cms). To assess the effects of the flow restoration on the fish assemblage, I evaluated trends in occupancy for 6 species from 1993 to 2009. Fish assemblage monitoring was conducted sporadically in the Cheoah River from 1993 to 2003 and then on a more consistent basis from 2004 to 2009 (Table 6.3). Sampling was conducted by different agencies and methodology ranged from 3-pass depletion electrofishing methods, to IBI sampling methodology, to snorkeling (Table 6.3). Sampling locations and associated fish assemblage information was collected from agency personnel. Mesohabitats (riffle, run, pool, cascades, etc) were delineated by Forest One Consultants and Entrix Consultatns for the entire Cheoah River below Santeetlah Dam and were used as habitat units for calculating occupancy estimates for the 6 fish species for each year. Effort, in terms of the number of mesohabitats sampled each year, generally increased over time (Fig. 6.13).

One of my main goals was to evaluate trends in occupancy between the pre- and post-flow periods (1993-2005 and 2006-2009, respectively). However, four of the six species (Tuckasegee darter, tangerine darter, greenfin darter, and northern hogsucker) showed declines in occupancy after 2003, which did not correspond to the re-regulation (Fig. 6.14). One species, central stoneroller, was not found in the river until 2003, after which there were no apparent trends. The trends in occupancy observed following 2003 were most likely due to a massive flood event on May 6, 2003 whose magnitude was in excess of 15000 cfs (425 cms) (Fig. 6.8). USGS gage records begin in 1999; however, historic daily spillage records from Santeetlah Dam (1927-current) suggest that a flood of similar magnitude was unlikely (only average daily records available – not maximum). The only flood that may have rivaled that of 2003 could have occurred in the mid-1970s; thus, the 15000 cfs discharge was at least a 30-year flood event.

Only two species (of the six evaluated), northern hogsucker and black redhorse, seemed to be influenced at all by the flow re-regulation initiated in September 2005 (after the 2005 sampling event). Northern hogsucker showed positive trends after 2006. Black redhorse, on the other hand, was found in 1993 and not again until 2006. Schools of black redhorse were found in 2001 in Calderwood Reservoir by Normandeau Associates (Normandeau 2002b). Thus, they may have been temporarily using the Cheoah as spawning grounds (1993 sampling was conducted in June) and, following the increased minimum flow, they became permanent residents in the Cheoah. Elevated water depths in pools may have provided adequate feeding

habitat for black redhorse since all of the black redhorse since 2006 have been observed in deep-slow runs and pool habitats.

Altogether, the trends in occupancy suggest that the effects of the flow restoration were masked largely by the large flood disturbance in 2003. Secondly, higher minimum flows and peak flows occurred from 2003 to 2005 (Fig. 6.2). Thus, the re-regulation did not generate an obvious “treatment effect” in the river, at least at this point in time. Despite substantial increases in effort, the occupancy and abundance of Tuckasegee darters, a species endemic to the Little Tennessee basin, has drastically decreased. Although abundance is biased by sampling methodology, efficiency, and effort, I display abundance values to show that despite extensive efforts, I only observed 2 individuals during sampling in 2008 and 2009. I also display abundance to show variation from year-to-year within the same sampling methodologies. For example, during 3-pass depletion efforts in 1993 and 1999, crews collected over 100 greenfin darters in three 150-m reaches (Fig. 6.14). However, in August of 2003, immediately after the flood event, similar sampling methodology yielded only 23 greenfin darters at the same sites (Fig. 6.14). Similar trends were apparent in Tuckasegee darters, tangerine darters, and northern hogsuckers. Occupancy estimates may also be somewhat inflated during the 1993, 1999, and 2003 sampling occasions because the sampling was conducted in limited mesohabitats and little sampling was conducted close to the dam. Tangerine darter occupancy dramatically decreased after 2003 and prior to that time, they were found throughout the length of the Cheoah River. Currently, tangerine darters are only found in the upper reaches of the river; however, they generally occur in dense aggregations and their numbers seem to be growing.

Large disturbances, such as flood events, are generally considered beneficial to river systems because they form and maintain habitats (Trush et al. 2000), create environments unfavorable to habitat generalists (Poff et al. 1997; Bunn and Arthington 2002), and moderate diversity (Wootton et al. 1996; Cardinale et al. 2005). However, in highly fragmented systems, large disturbances may push populations beyond recovery if there are no source populations to support recolonization. In systems characterized by extreme physical disturbances, recolonization potential, reproductive success, and ability to find habitat refugia will become more important in structuring fish communities than other factors, such as smaller variations in flow (e.g. change in minimum flow) (Jackson et al. 2001). At larger spatial scales, the flow regime of streams may provide a template of environmental variability that structures fish

communities (Poff et al. 2007; Jackson et al. 2001). However, in terms of temporal patterns, the magnitude of the relative change in the flow regime (i.e. extent of disturbance) will govern the morphological and ecological processes that shape how fish communities response to that change (Poff and Ward 1990; Jackson et al. 2001). Thus, our ability to perceive the response of fish communities to changes in environmental variability will depend upon scale but also the extent of physical disturbance (Jackson et al. 2001).

Jackson and Pringle (2010) indicate there may be negative consequences to increasing hydrologic connectivity (e.g. establishing peak floods) in degraded environments. They cited Scheerer (2002) who made the case that restoring peak flows may be detrimental to the endangered Oregon Chub in the Willamette Valley, Oregon due to increased connectivity to habitats containing potentially competitive non-native species. Isolated populations of endemic species, many times, persist only on the basis of small patches of suitable habitat. Since large flood events have the potential to “form” habitats, they also have the potential to destroy any habitats necessary for the persistence of small, patchy populations. Jackson et al. (2001) indicate that spatial habitat connectivity in relation to physical disturbance will determine how other factors affect the structure of fish communities. Tuckasegee darters were found in Calderwood reservoir in 2001 and in Slickrock Creek, a tributary to Calderwood (Normandeau 2002b); however, they are not found in any tributaries of the Cheoah River below Santeetlah Dam. Similarly, there are no source populations of tangerine darters to support recolonization. Northern hogsucker and stonerollers are found in tributaries below Santeetlah Dam; however, greenfin darters are not. Thus, the future persistence or non-persistence of species in the Cheoah River may lie partially in the reintroduction and continued monitoring of endemic fish populations but also the restoration of other factors besides flow.

#### *Effects of morphological restoration on the fish assemblage*

According to Chapter 5, morphological conditions in the Cheoah River are very different than that of the surrounding river systems. Gravel is scarce in the Cheoah River, averaging 12% compared to 40% in other river systems (Chapter 5). In the Cheoah River, sand occurs in amounts less than half of that found in other drainages and bar habitat is almost non-existent. The subsurface index is the lowest in the Cheoah River compared to any other river. Degraded

morphological conditions continue to impose strong constraints on the potential fish assemblage restoration in the Cheoah River.

During February 21-23, 2008, washed gravel (10mm and 50mm), mined from drained floodplains of the Alabama River in a quarry near Montgomery, Alabama, and transported to the Cheoah River and dumped down the stream bank and into the channel at four locations. Embankments were fairly steep to promote gravel migration into the channel. For more information regarding the restoration, refer to McManamay et al. (2010). Newly added gravels provided spawning substrate for river chubs (Fig. 6.15, McManamay et al. (2010)); however, river chubs are not uncommon in the Cheoah River and have an occupancy close to 100%. Unfortunately, one of the main conclusions of McManamay et al.'s assessment was that gravel volumes and depths were inadequate to provide sufficient spawning substrates for catostomids, especially redhorse. The migration of gravel in the channel was extremely sparse and unstable because of low augmented volumes and finer substrates were not included in the augmentation (McManamay et al. 2010). A total of 100 yd<sup>3</sup> of gravel was added to all four sites; however, results of the assessment indicate that a minimum of 100 yd<sup>3</sup> should be added to each site. Because Santeetlah Dam has blocked bedload transport for over 80 years, extensive morphological restoration and larger amounts of supplemental bedload are needed to restore spawning and refuge habitat in the Cheoah River. Additionally, a wider range of substrates (sand, fine to large gravel, and smaller cobble) is needed to provide habitat for different species.

#### *Effect of temperature on the fish assemblage*

Temperature is higher in the Cheoah River compared to other comparable river systems, such as the Valley and Upper Pigeon Rivers (Chapter 5). In 2001 (pre-relicensing agreement), Normandeau Associates along with NC Wildlife Resource Commission Biologists conducted a temperature assessment throughout the Cheoah basin (Normandeau 2001). Fifteen temperature loggers were placed at a range of elevations in tributaries above Santeetlah Lake and four loggers were placed in the Cheoah River to determine the effect of surface release operations on temperatures in the lower river. Temperatures generally showed a linear increase with decreased elevation; however, temperature displayed a non-linear increase directly below Santeetlah Dam, especially in the late summer (Fig. 6.16). During that time, surface-releases were not a regular event; thus, elevated temperatures most likely were due to an active channel that was confined to

the center of a historically-wider channel and shallow, stagnant seepage, all of which allowed water to receive more direct solar radiation. Because the temperature assessment was conducted prior to the re-regulation, I questioned whether the established minimum flow would influence temperature. Using temperature data from the Cheoah River USGS gauge (0351706800), I compared average monthly summer temperatures (June – September) for the pre-relicensing agreement (1999-2005) to those after the relicensing agreement (2006-2010). Interestingly, average summer monthly temperatures were significantly higher following the flow restoration (Fig. 6.16, two tailed t-test,  $p=0.0004$ ). One explanation is that prior to implementation of the new flow regime, cold tributary inflow made up a larger proportion of the volume of water in the Cheoah River. Thus, releasing higher volumes of water from the surface of Santeetlah reservoir resulted in a larger volume of water with higher thermal capacity (Caisse 2006) that diluted the effect of tributary inflow. In essence, the flow restoration amplified warm temperature conditions in the Cheoah River. During the FERC relicensing studies, the instream flow incremental methodology (IFIM) process was conducted by contractors and was applied to the Cheoah River under a range of flows to determine the most appropriate discharges to restore physical habitats for targeted organisms (Normandeau 2002a). However, temperature modeling in conjunction with discharge variability was not explicitly conducted, though considered standard protocol for IFIM by the USGS (Bovee et al. 1998).

Evaluations of species distribution maps in the Cheoah drainage provide additional evidence of a temperature issue in the Cheoah River. For example, mottled sculpin are found in tributaries above and in one tributary below Santeetlah Dam and they are found in the frigid waters of Calderwood Reservoir ( $\geq 16^{\circ}\text{C}$ ) but are absent from the Cheoah River (Fig. 6.17). Similarly, only one individual longnose dace was found directly below the entrance of Yellow Creek in the Cheoah River but populations are found in all sampled tributaries above and below Santeetlah Dam (Fig. 6.17). In addition, Tennessee shiners are found in the tributaries above Santeetlah Dam but are not found within the main-stem Cheoah River. The dominant fish in Calderwood Reservoir is white sucker (Normandeau 2002b); however, they have never been found in the Cheoah River (Fig. 6.17). Maximum temperature tolerance reported for mottled sculpin is  $30.9^{\circ}\text{C}$  (Kowalski et al. 1978) whereas the maximum weekly average reported for mottled sculpin is considerably lower,  $24.3^{\circ}\text{C}$  (Eaton and Scheller 1996; Werlhy et al. 2003). The maximum weekly average temperatures (not maximum temperature tolerance) reported for

white sucker and longnose dace are 27.4°C and 26.5°C, respectively. Maximum summer temperatures reported from the USGS gage in the lower Cheoah River have reached 30.9°C. Mottled sculpin, longnose dace, and Tennessee shiners are common in many of the medium to large rivers in the upper Tennessee River basin. However, the maximum temperature tolerance is 30.8°C for northern hogsucker, which is found throughout the Cheoah River. Thus, some species may not inhabit the Cheoah River due to a temperature barrier in combination with other habitat factors. Although temperature maxima may be important to species distribution, the ability of species to acclimate to large diurnal fluctuations may be of greater significance and could be more pronounced in the Cheoah River due to surface releases from Santeetlah Reservoir. Comparisons of diurnal fluctuations in addition to average daily conditions in the Cheoah River relative to other streams in the region would be an informative analysis.

## **6.6 Conclusions**

The utility of the restoration template is that it provides a framework to 1) form environmental flow recommendations for regulated rivers, 2) assess ecologically relevant hydrologic alterations based on class membership, and 3) evaluate how fish assemblages respond to hydrologic alteration. The restoration framework outlined in Fig 1.2 is a simplified version of the ELOHA framework (Ecological Limits of Hydrologic Alteration) (Fig. 1.1) (Poff et al. 2010), which is currently being used extensively as the environmental flow management tool by The Nature Conservancy and the Southern Instream Flow Network (SIFN) of the Southeastern Aquatic Resources Partnership (SARP). Although the ELOHA framework indicates that the river classification can be based on “hydrology and other factors”, both frameworks largely ignore the effects of altered channel morphology, temperature, and even landscape patterns (i.e. fragmentation) on fish assemblages. The Cheoah River case study highlights the fact that the effect of dams will influence not only the flow, but channel morphology and temperature. Templates that guide restoration are only as good as the data that create them. Thus, the current template needs to be modified either by 1) including factors other than hydrology in the classification step or 2) including other factors as covariates with hydrology in determining ecological relationships. The fact that other factors besides flow are important to the management of fish communities is not a new concept.



To anyone “living with dams”, the impact of each dam on flow, water quality, and channel morphology is unique and dependent upon the geomorphic and climatic context (McCartney 2009). Channel morphology is obviously a factor that cannot be ignored. This is exemplified in the Trinity River restoration in California. Trush et al. (2000) discuss that the dynamic interactions between flow, channel morphology, and sediment budgets should provide a context for regulated alluvial river restoration and should be included in current water policy. Consideration of morphologic alterations is also included in the USGS’s IFIM protocol. “Restoring peak flows” has become the recent focus of environmental flow experts around the globe. However, Jackson and Pringle (2010) argue that reducing peak flows may be a better management action to prevent sediment starvation in streams. In the Apalachicola River system in Florida, Jackson and Pringle indicate that although the U.S. Fish and Wildlife Service recognizes that sediment starvation and altered channel morphology are critical problems to four endangered species, the recommendations focused on minimum flows and did not address channel morphology issues. The Cheoah River case study illustrates that stream flow is certainly linked to morphology and must be considered in terms of the sediment regime (gravel additions) and altered channel morphology. Although I have not discussed gradient extensively, I believe the high gradient conditions (at times > 2%) in the Cheoah River limits morphological restoration and will also limit the extent of substrate improvements because of the high transport capacity of the river. High gradient could also potentially inhibit fish immigration from various areas of the river, especially for non-active swimmers.

River managers should also consider the compound relationship between flow and temperature below impoundments. The relationships between flow and temperature are known and can be modeled accurately (Krause et al. 2005). Olden and Naiman (2010) indicate that temperature is “a key, yet poorly acknowledged component of environmental flows” and discuss some of the challenges for why temperatures are not included in environmental flow assessments. Once again, modeling temperature in conjunction with variable releases during IFIM studies is considered standard USGS protocol (Bovee et al. 1998). In my opinion, I consider temperature one of the most critical aspects of needed restoration in the Cheoah River. Increased discharges from surface layers of the reservoir may have amplified a temperature problem. I place temperature as higher priority over morphology and flow because temperature seems to be an obvious barrier to species already found throughout the drainage. In my opinion,

morphological conditions would not prevent an occasional rare specimen, such as mottled sculpin or longnose dace, from showing up in fish assemblage collections whereas temperature conditions might prevent such an occurrence. Although 12-14 species have been found as suitable fish hosts for the Appalachian elktoe (*personal communication*, Mark Cantrell USFWS), the best documented fish host species are mottled sculpin and banded sculpin (Gordon and Moorman 2001; Keller and Augspurger 2005). My results indicate that the temperature regime under current dam operations may be prevent sculpin from inhabiting the Cheoah River, thereby limiting the relict population of *A. raveneliana*. Mussel assemblages are largely influenced by the presence of a host fish species (Geist et al. 2006; Bogan 2008). Mussel communities in the upper Tennessee basin show signs of nestedness, where species found in degraded conditions are a subset of the diversity in the region (Rashleigh 2008). Largely, this is the result of dams, which can have devastating impacts on mussel communities (Vaughn and Taylor 1999; Babko and Kuzmina 2009). Although mussel species distributions were not related to the number of fish hosts in the upper Tennessee basin, mussel community nestedness may be driven by the abundance of available fish hosts (Rashleigh 2008). Unfortunately, the Cheoah River still suffers in terms of fish diversity and the abundance of available fish hosts. Vaughn and Taylor (1999) reported that temperature alterations caused by dams may limit the colonization potential of mussels. Variation in mussel assemblages in Lake Erie tributaries were influenced by multiple factors including altered channel morphology due to large flood events, sedimentation, and altered water quality (Krebs et al. 2010). Thus, multiple factors, including temperature, channel morphology, and availability of fish hosts, may interact to influence the relict population of *A. raveneliana* in the Cheoah River.

Habitat fragmentation is a major limitation to fish assemblages (and mussel communities) (Vaughn and Taylor 1999; Han et al. 2008; Hoagstrom et al. 2008; Reid et al. 2008). Stream network fragmentation can easily be estimated through GIS applications and be incorporated into large spatial frameworks. However, large spatial assessments of hydrologic alterations on fish and macroinvertebrate communities rarely consider fragmentation as a driver (Knight et al. 2008; Carlisle et al. 2010b; Kennen et al. 2010). For the most part, flow patterns in the Cheoah River resemble the majority of streams in the upper Tennessee River basin; however, the Cheoah River has among the shortest fragment lengths of all the fish assemblage sites sampled in Chapter 5.

Obviously, disconnectivity from source populations will continue to limit recovery in the Cheoah River and must be included in analyses.

Table 6.1. Percent changes in the median particle size (D50) of pebble counts conducted at 8 sites along the length of the Cheoah River. Pre (2002) data available from R2 (2003).

| <b>Transect</b> | <b>Distance<br/>from Dam (km)</b> | <b>Pre<br/>2002</b> | <b>Post<br/>2008</b> | <b>% change</b> |
|-----------------|-----------------------------------|---------------------|----------------------|-----------------|
| DC3             | 0.6                               | 762                 | 160                  | -79.00          |
| DC7             | 2.3                               | 1676.4              | 1000                 | -40.35          |
| CY3             | 5.5                               | 457.2               | 270                  | -40.94          |
| CY8             | 6.7                               | 304.8               | 160                  | -47.51          |
| YD2             | 8.5                               | 279.4               | 195                  | -30.21          |
| YD7             | 9.5                               | 914.4               | 350                  | -61.72          |
| DM2             | 11.0                              | 228.6               | 250                  | 9.36            |
| DM5             | 12.5                              | 203.2               | 257.5                | 26.72           |

Table 6.2. Records of collected fish species from fish assemblage sampling in the Cheoah River from 1993 to 2009. Agency code descriptions are given in Table 6.3.

|                                  | 1993 | 1999 | 2003 | 2004 | 2005 | 2006 | 2008 | 2009 |
|----------------------------------|------|------|------|------|------|------|------|------|
| Agency                           | NCW  | NCW  | NCW  | NCW  | NCW  | FS   | VT   | VT   |
| Total Species                    | 13   | 11   | 13   | 14   | 16   | 10   | 18   | 17   |
| Centrarchidae                    |      |      |      |      |      |      |      |      |
| <i>Amploplites rupestris</i>     | X    | X    | X    | X    | X    | X    | X    | X    |
| <i>Lepomis cyanellus</i>         |      |      |      | X    | X    |      |      | X    |
| <i>Lepomis macrochirus</i>       |      |      | X    | X    | X    |      | X    | X    |
| <i>Micropterus dolomieu</i>      | X    | X    | X    | X    | X    | X    | X    | X    |
| <i>Micropterus salmoides</i>     |      |      | X    | X    |      |      | X    | X    |
| <i>Micropterus punctatus</i>     |      |      |      |      | X    |      |      |      |
| Catostomidae                     |      |      |      |      |      |      |      |      |
| <i>Hypentelium nigricans</i> *   | X    | X    | X    | X    | X    | X    | X    | X    |
| <i>Moxostoma duquesnei</i>       |      |      |      |      |      | X    | X    | X    |
| <i>Moxostoma sp.</i>             | X    |      |      |      |      |      |      |      |
| Cyprinidae                       |      |      |      |      |      |      |      |      |
| <i>Campostoma anomalum</i> *     |      |      | X    | X    | X    |      | X    | X    |
| <i>Cyprinella galactura</i>      | X    | X    | X    | X    | X    | X    | X    | X    |
| <i>Luxilus coccogenis</i>        | X    | X    | X    | X    | X    | X    | X    | X    |
| <i>Nocomis micropogon</i> *      | X    | X    | X    | X    | X    | X    | X    | X    |
| <i>Pimephales promelas</i>       |      |      |      |      | X    |      |      |      |
| <i>Rhinichthys cataractae</i> *  |      |      |      |      | X†   |      | X†   | X†   |
| Ictaluridae                      |      |      |      |      |      |      |      |      |
| <i>Pylodictis olivaris</i>       | X    | X    |      |      |      |      | X†   |      |
| Percidae                         |      |      |      |      |      |      |      |      |
| <i>Etheostoma chlorbranchium</i> | X    | X    | X    | X    | X    |      | X    | X    |
| <i>Etheostoma gutselli</i>       | X    | X    | X    | X    | X    | X    | X    | X    |
| <i>Percina aurantiaca</i>        | X    | X    | X    | X    | X    | X    | X    | X    |
| <i>Sander vitreum</i>            |      |      |      |      |      |      | X†   |      |
| Salmonidae                       |      |      |      |      |      |      |      |      |
| <i>Onchorynchus mykiss</i> *     | X    | X    | X    | X    | X    | X    | X    | X    |
| <i>Salmo trutta</i> *            | X    |      |      |      |      |      | X    | X    |

\* species is present in tributaries below dam † only 1 specimen was represented in entire collection

Table 6.3. Year and agency responsible for fish assemblage sampling and the sampling methodology used.

| Year | Agency Code | Agency   | Methodology  |
|------|-------------|--|--|
| 1993 | NCW         | North Carolina Wildlife Resources Commission             | 3-pass depletion - up to 10 backpack electroshocker units        |
| 1999 | NCW         |  |  |
| 2003 | NCW         |  |  |
| 2004 | NCW         | North Carolina Wildlife Resources Commission             | IBI methods including seines and single backpack electroshockers |
| 2005 | NCW         |  |  |
| 2006 | USFS        | USDA Forest Service, Central Aquatic Technology Transfer | Snorkeling   |
| 2008 | VT          | Virginia Tech, Fisheries and Wildlife                    | IBI methods described above and snorkeling                       |
| 2009 | VT          | Department   |  |

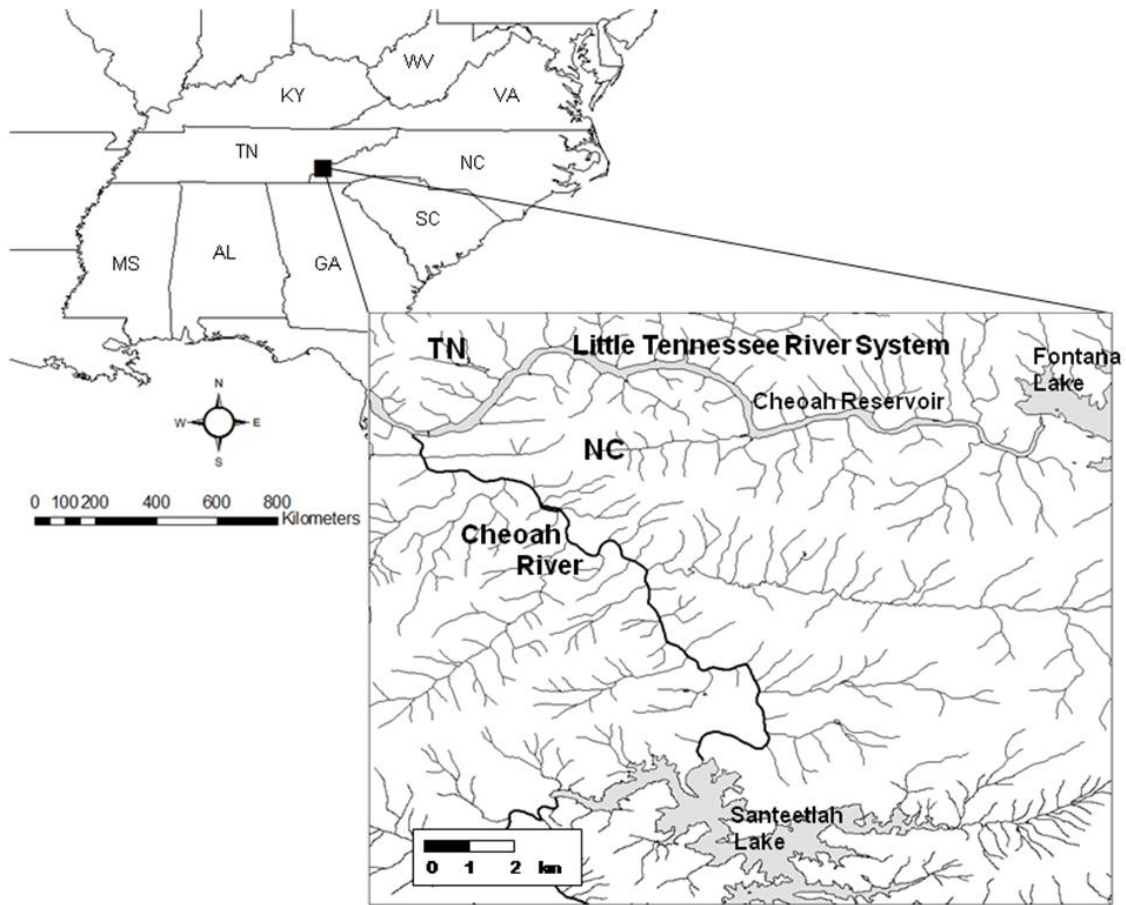


Figure 6.1 Map of the Cheoah River, a regulated tributary of the Little Tennessee River system in Graham County, North Carolina.

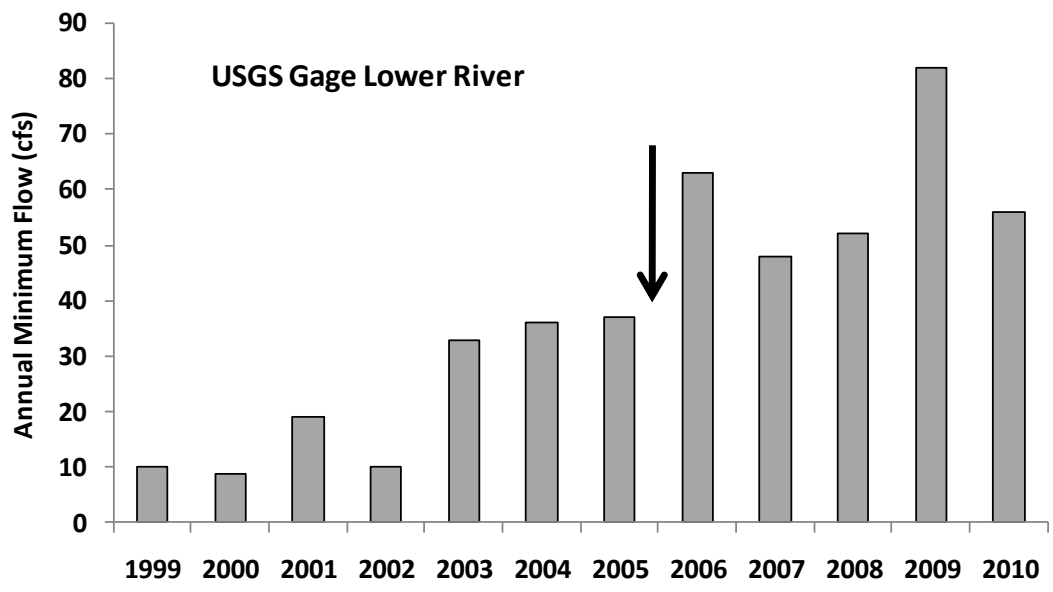
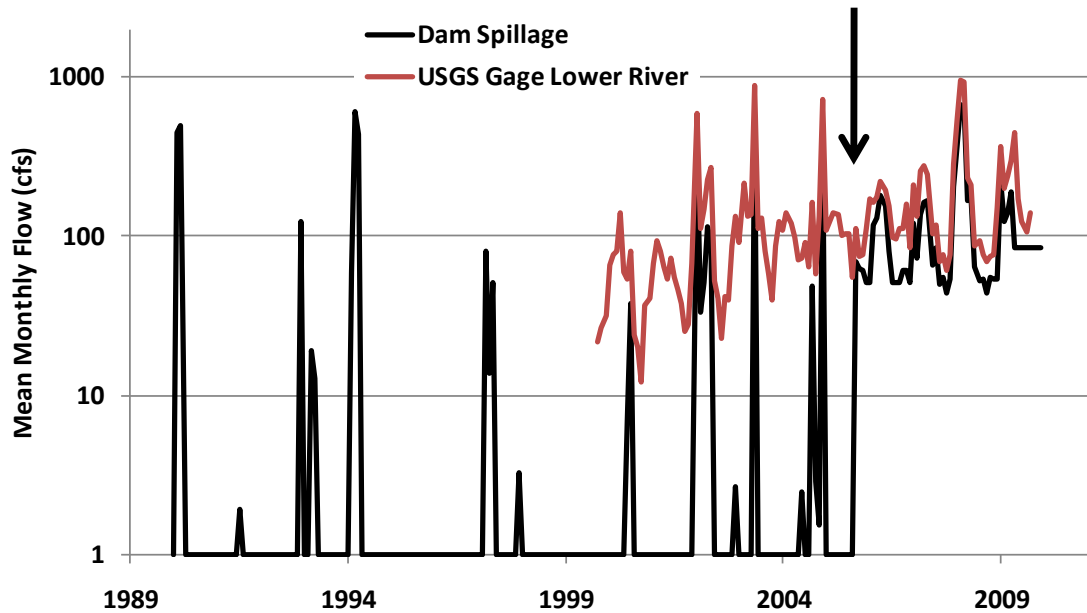


Figure 6.2 Mean monthly dam spillage and discharge at the USGS gauge in the lower Cheoah River (top). Differences in magnitudes reflect tributary inflow. One-day annual minimum flow from 1999 to 2010 at the USGS gauge in the lower Cheoah River (bottom). Arrows indicate the initiation of the new FERC mandated flow regime in both graphs.



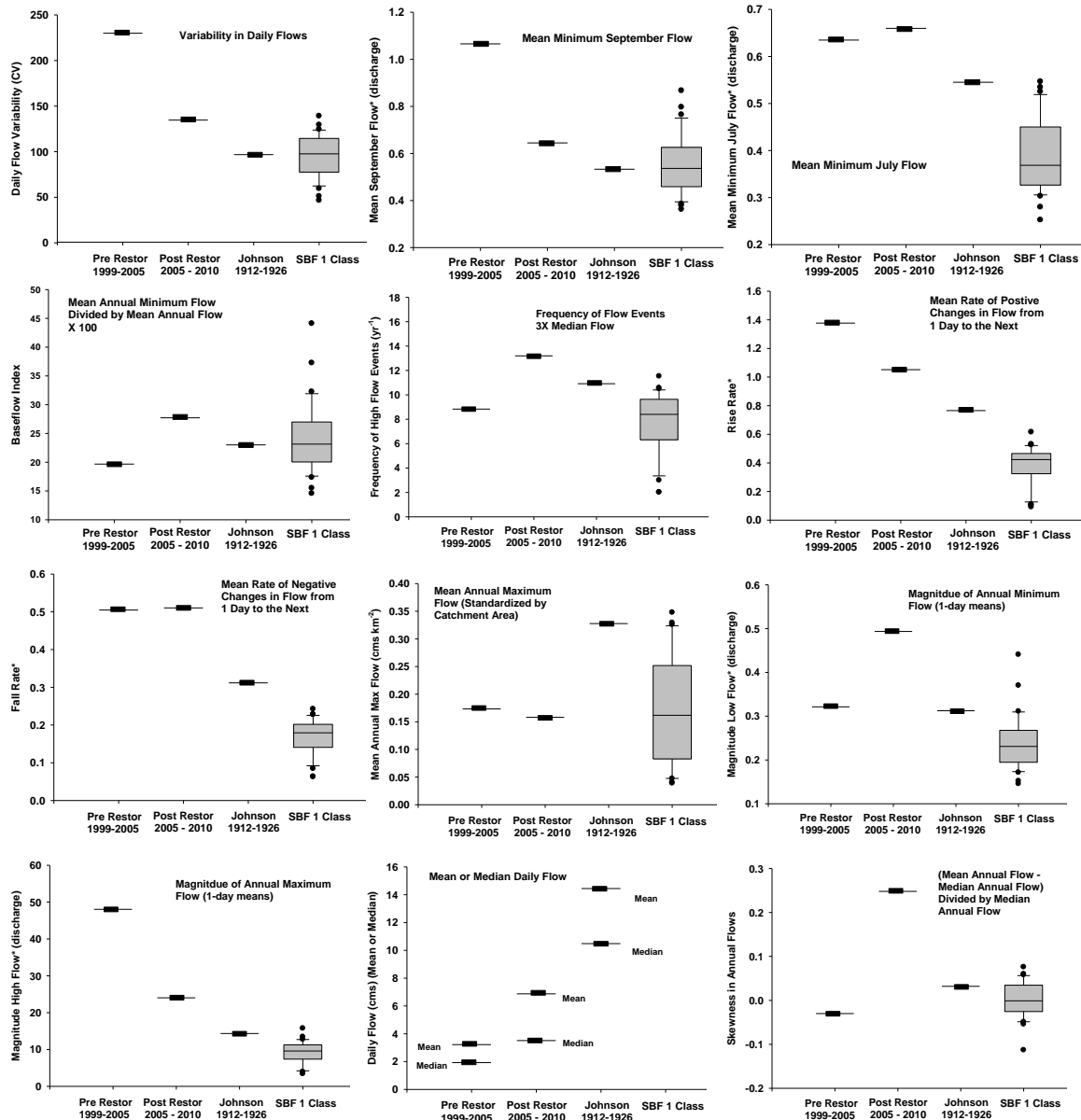


Figure 6.3 Comparison of pre-impoundment (Johnson 1912-1924), pre-restoration (1999-2005), and post-restoration (2005-2011) flow metrics of the Cheoah River with those of its appropriate flow class (stable high baseflow 1 or SBF1) in order to make environmental flow recommendations.

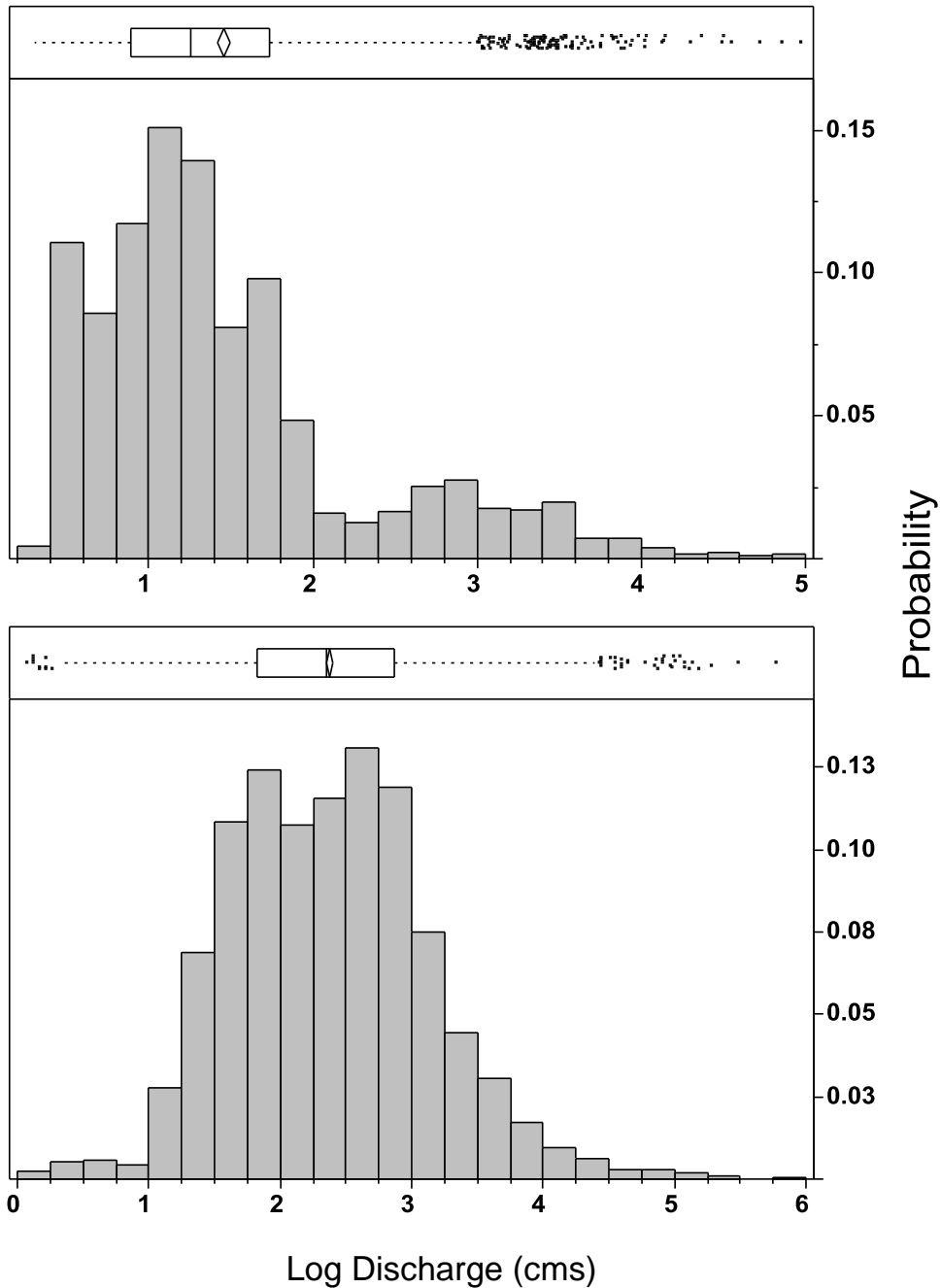


Figure 6.4 Log-transformed distributions and outlier box and whisker plots of daily flow values for the Cheoah River for the post-flow restoration time period (2005-2011) and the pre-impoundment time period (1912-1924). Vertical line in the box and whisker plot represent the median whereas the diamond represents the mean.

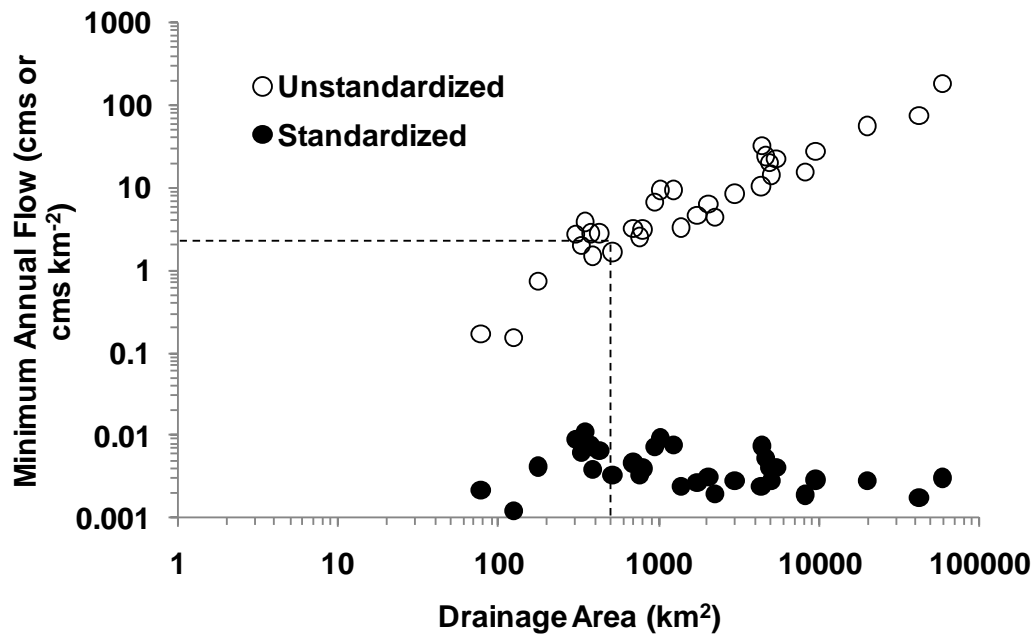
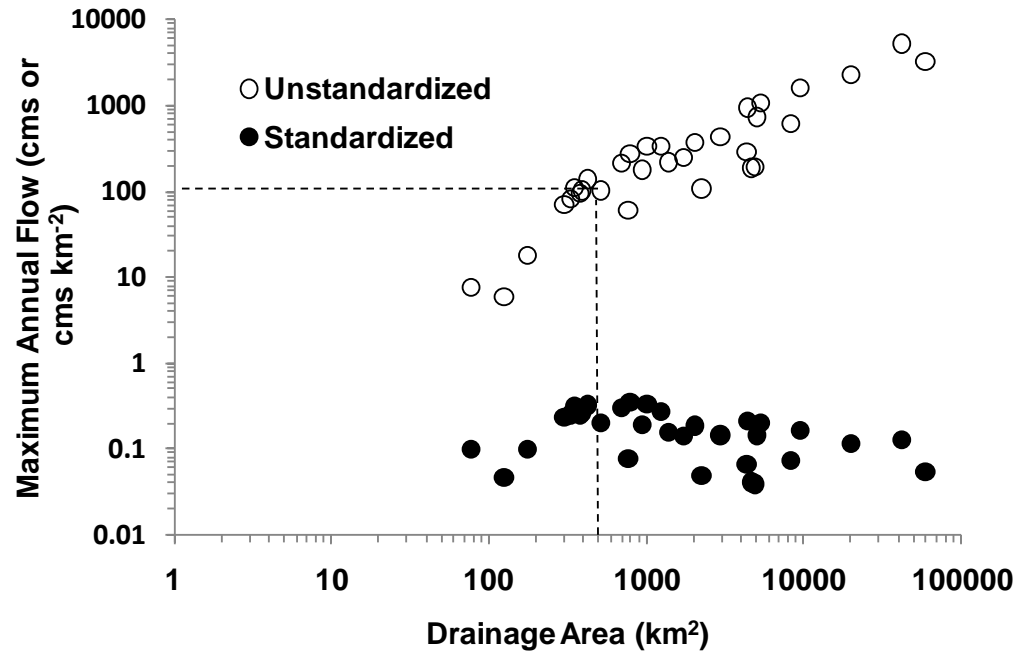


Figure 6.5 Relationship between drainage area size and specific flow magnitudes (annual maximum and annual minimum flows). Dashed line indicates the drainage area for the Cheoah River and its appropriate flow magnitudes.

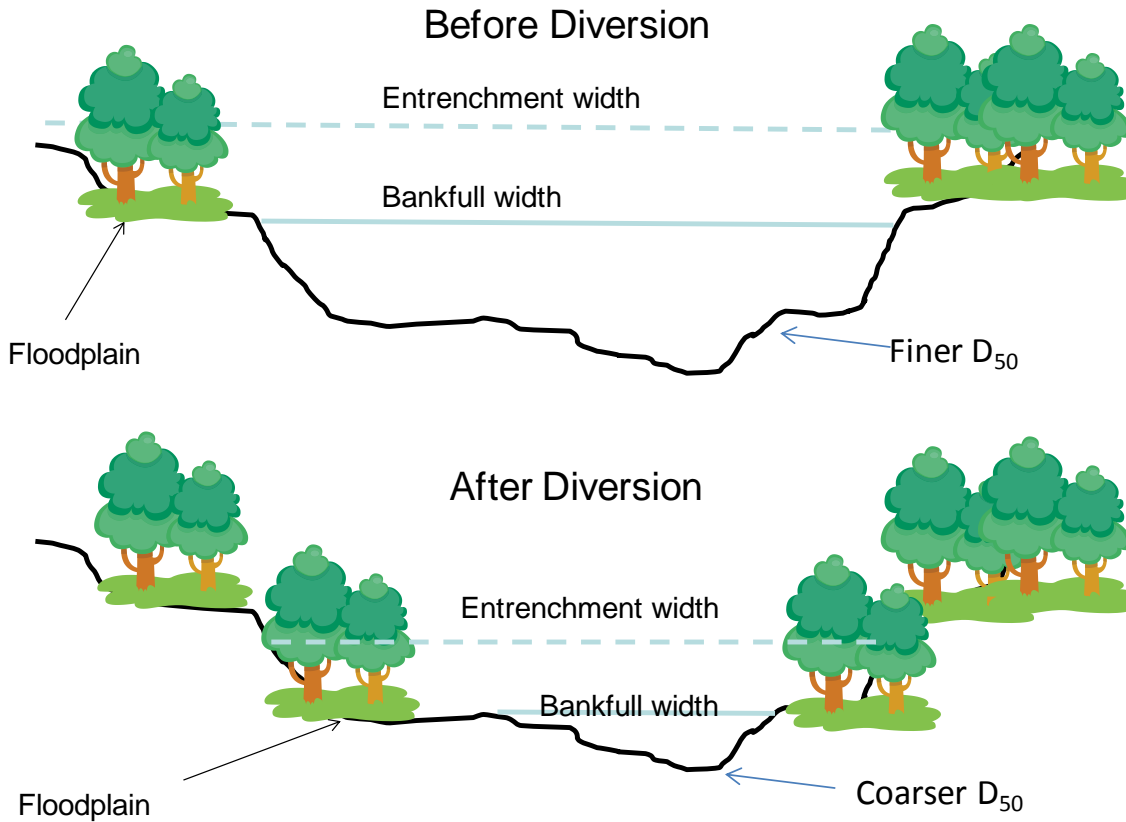


Figure 6.6 Conceptual model of the morphological changes in a channel following diversion. First riparian vegetation encroaches the historically-active channel and a new floodplain is established. The active channel is reduced to a smaller sub-section of the historical channel, with width/depth ratio, bankfull width, and entrenchment ratio far different than prior to diversion.



Figure 6.7 Serial photographs from bank monuments of riparian vegetation before (2001) and after (2009) the new flow regime initiated in the Cheoah River in 2005. Photographs from 2001 taken from Normandeau Associates (2002a).

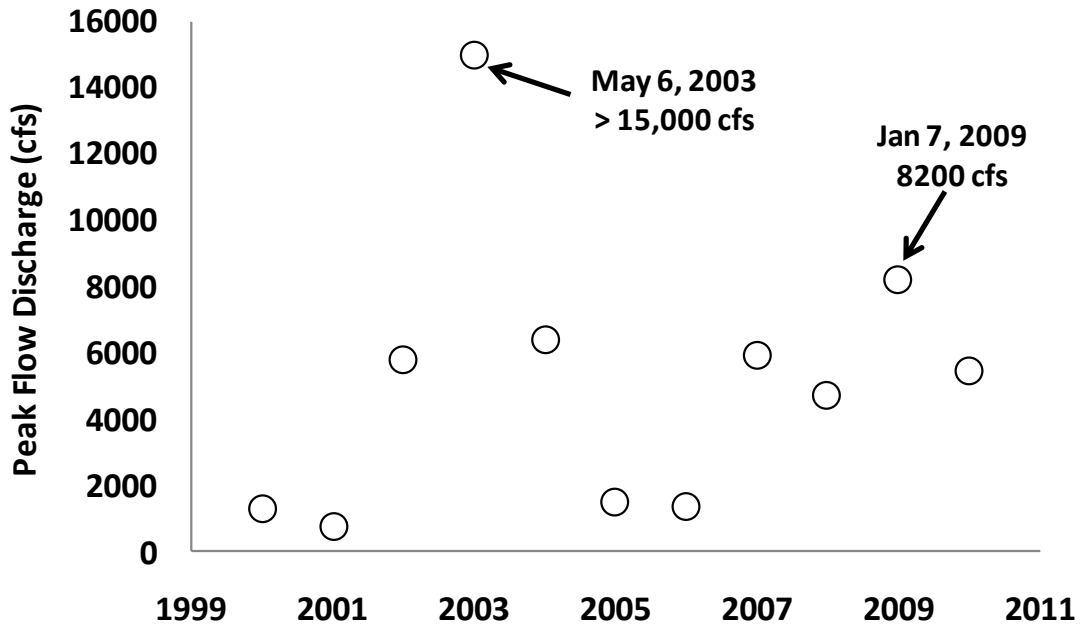


Figure 6.8 Peak flow discharges from 1999 to 2010 at the USGS gauge on the Cheoah River.



Figure 6.9 In-stream vegetated islands that were flipped as a result of a large spill event in excess of 8000 cfs (226 cms) in the Cheoah River. In the top photograph, stream bed particle sizes, representative of stream bed conditions prior to dam completion over 80 years ago, are uncovered by riparian vegetation removal.

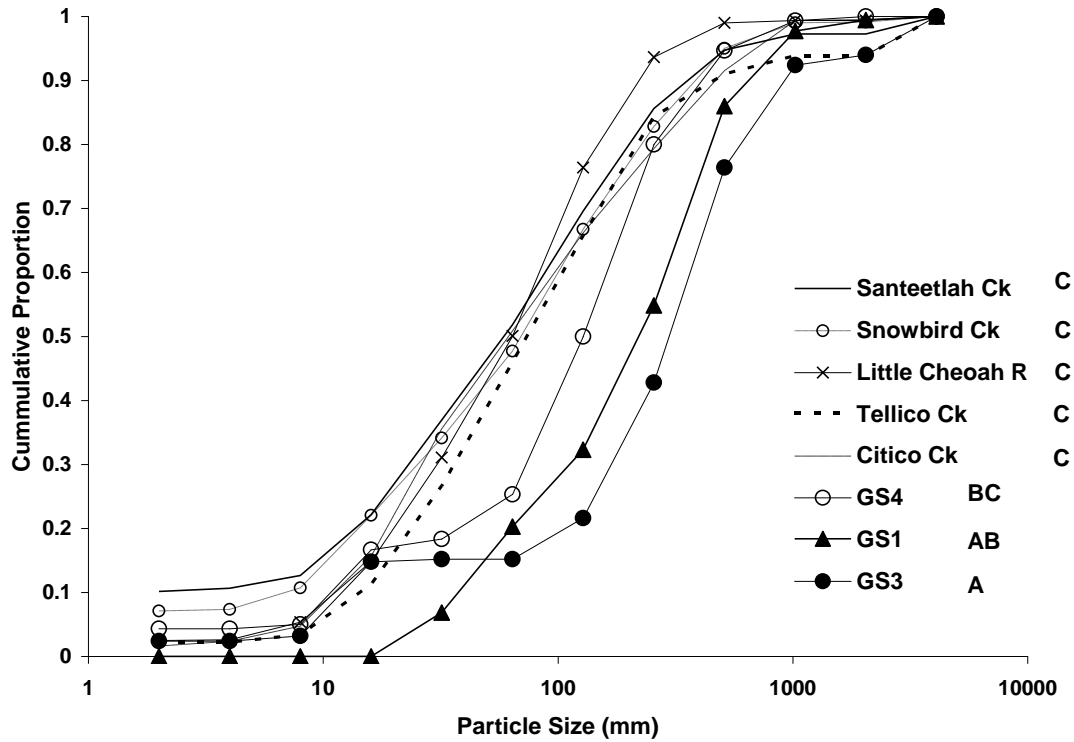


Figure 6.10 Comparison of particle size distributions at each gravel site (post gravel addition) and five reference stream reaches from rivers that share similar morphology to the Cheoah River. Letters indicate significant differences at the  $p = 0.05$  level in post hoc comparison (Tukey's test) using the D10 through D50 (increments of 5) size classes (Taken from McManamay et al. 2010).





Figure 6.11 Photographs taken in 2010 of vegetation clogging active channel in upper river reaches of the Cheoah River near the dam (top) and extensive riparian encroachment on a historically-active bar (bottom).



Figure 6.12 Postcard from the 1930's of the lower penstock crossing on the Cheoah River soon after the dam completion (1927) (top) (source: <http://www.tailofthedragon.com>). Serial photos of the channel under the penstock crossing in 1930's (middle) and in 2009 (bottom).

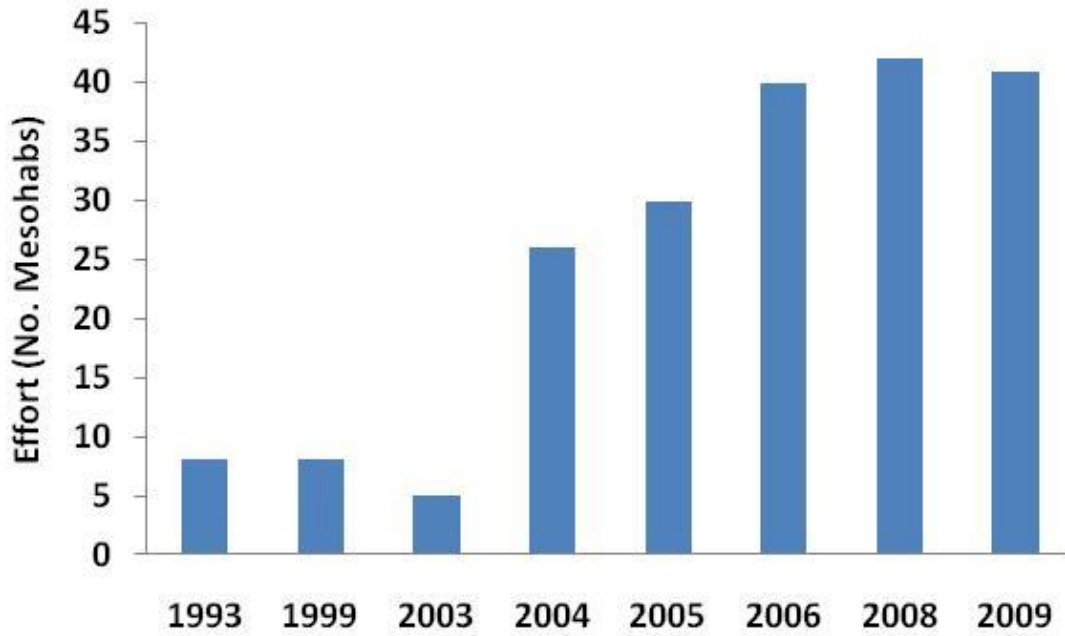


Figure 6.13 Fish assemblage sampling in the Cheoah River from 1993 to 2009 using a variety of fish sampling techniques, summarized as effort (number of mesohabitat units sampled per year).

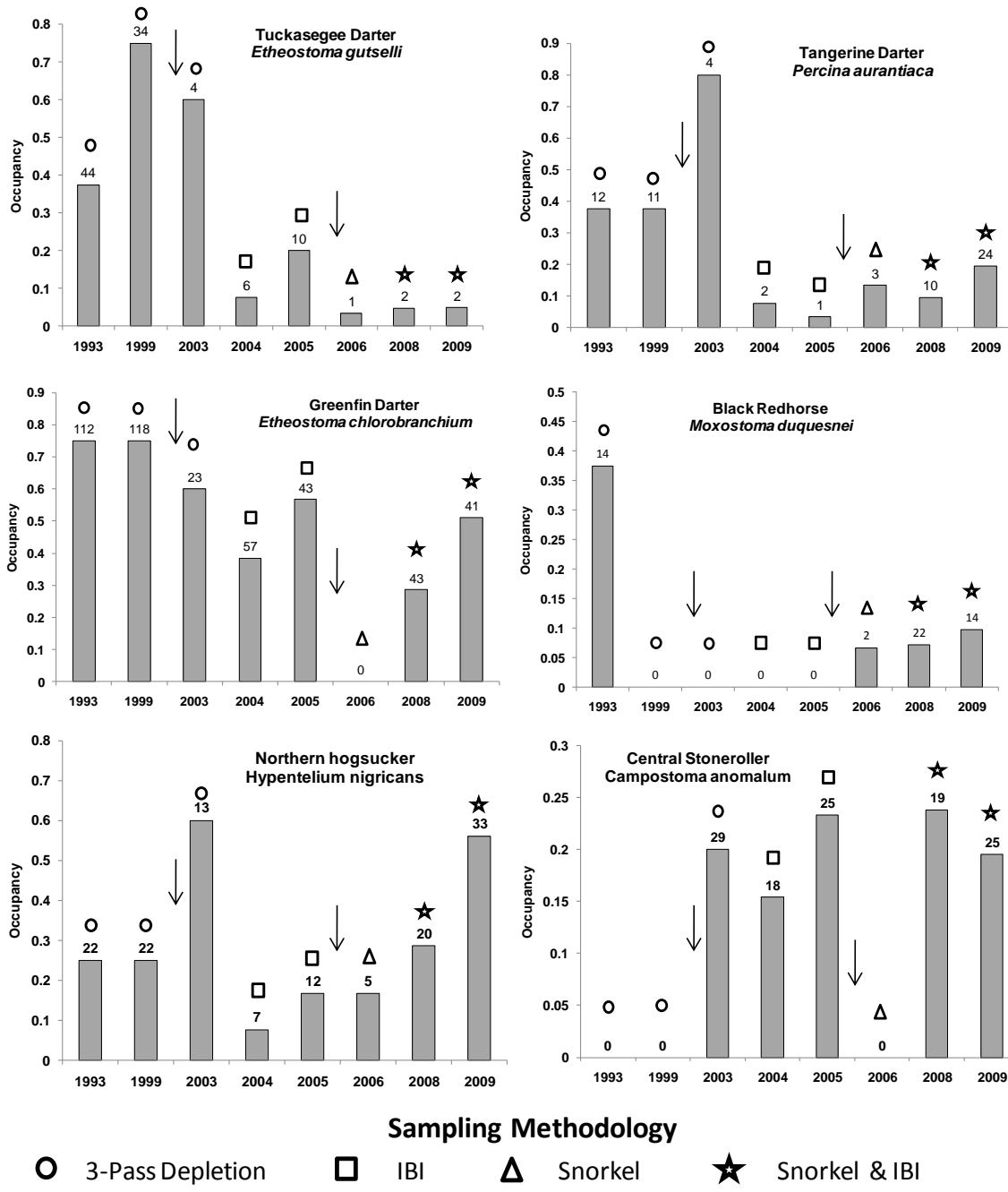


Figure 6.14 Trends in occupancy of 6 species from 1993 to 2009 according to various sampling methodologies in the Cheoah River. The first arrow indicates the timing of the 15,000-cfs (425 cms) peak flood on May 6, 2003 whereas the second arrow indicates the new, seasonally-variable flow regime initiated on September 1, 2005.



Figure 6.15 Photograph of a river chub (*Nocomis micropogon*) mound in the Cheoah River with newly augmented gravel (lighter particles) and native geologic material (darker particles). Taken from McManamay et al. (2010).

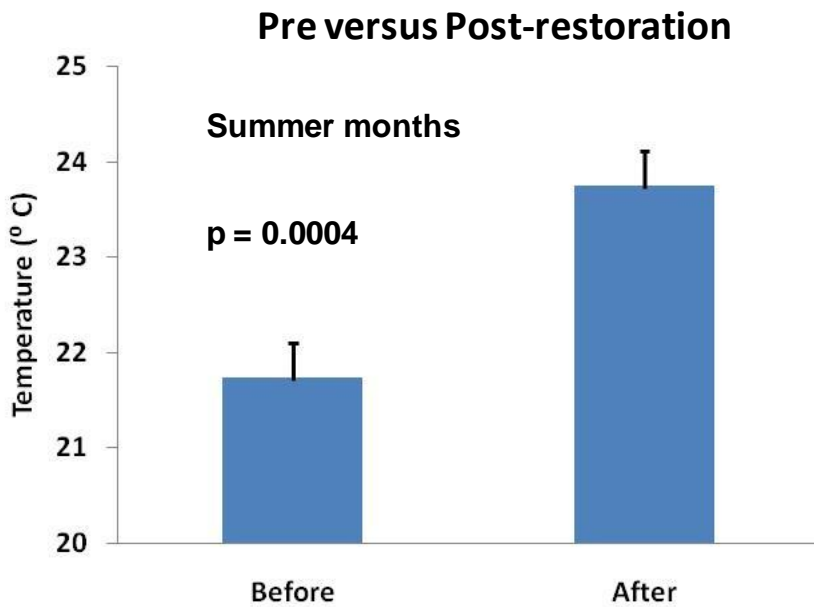
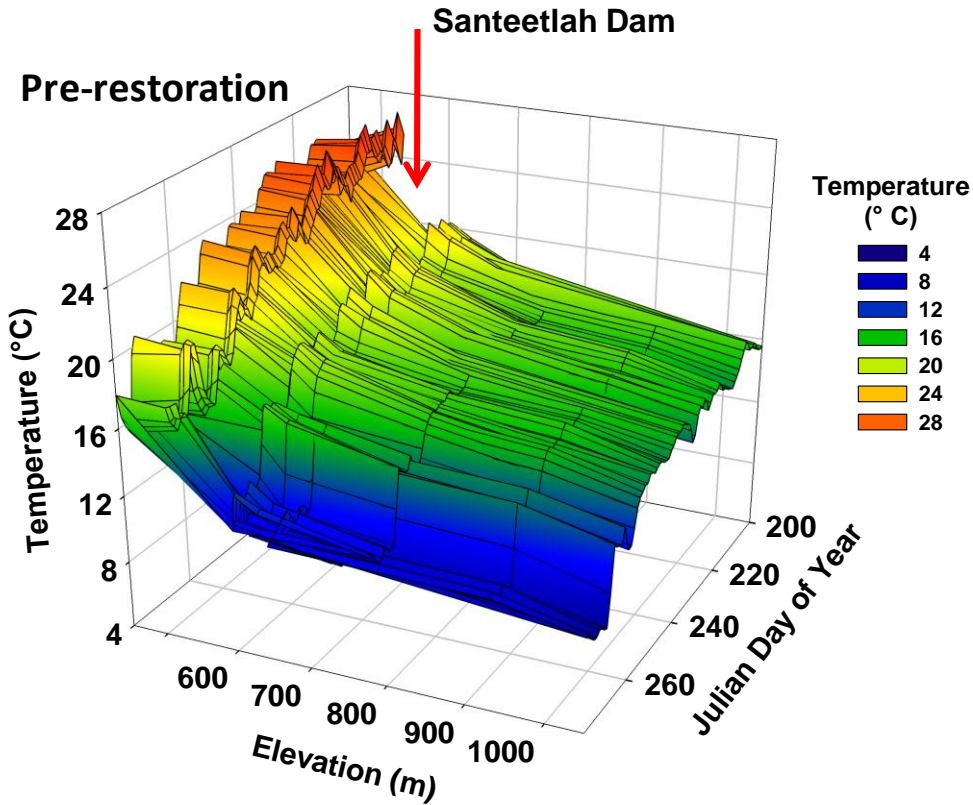


Figure 6.16 The influence of Santeetlah Dam on the relationship between temperature, elevation, and julian day of year (top) in the Cheoah River. Comparison of average summer monthly temperatures before and after the new seasonally-variable flow regime (bottom). Error bars represent 1 SE.

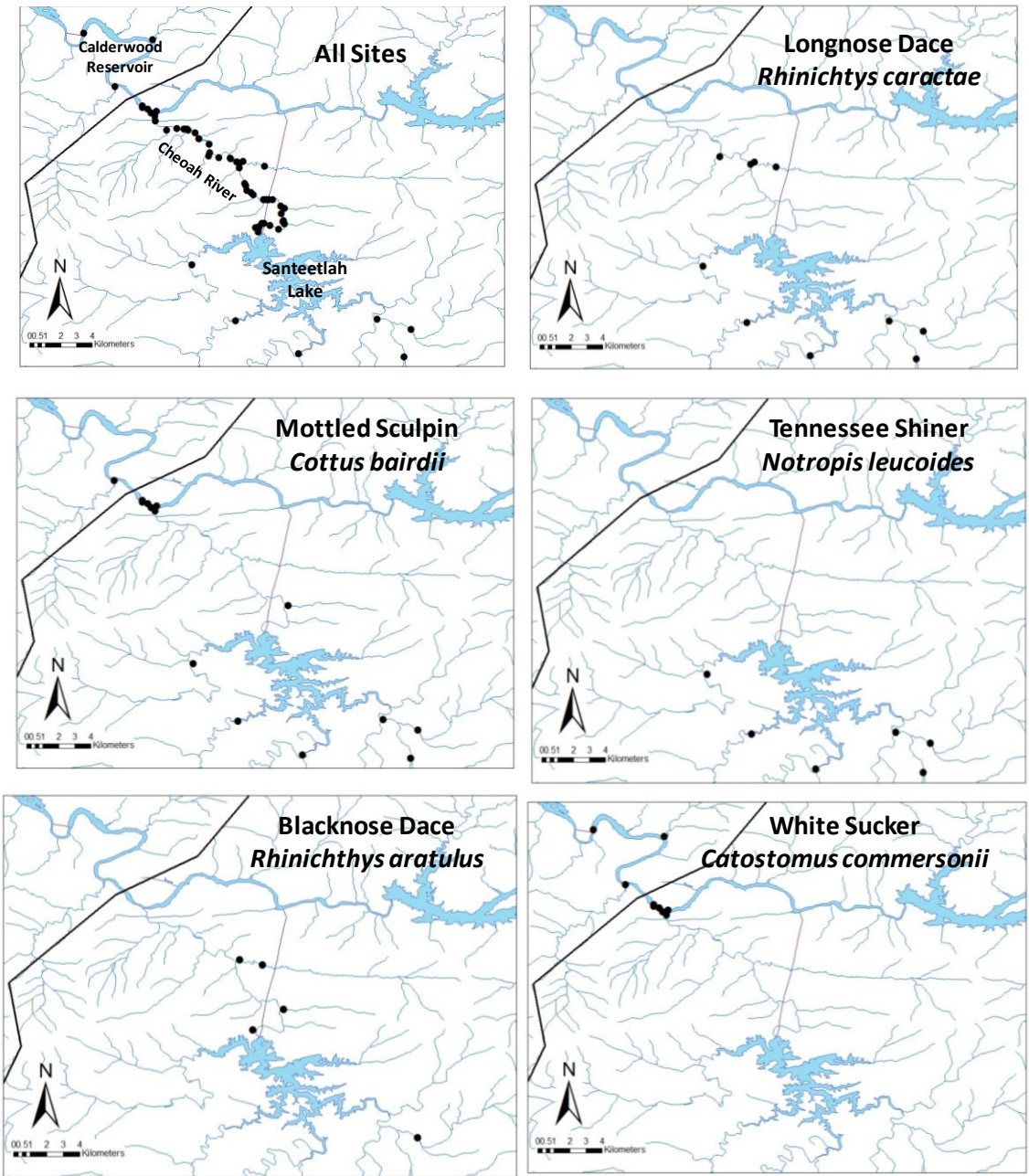


Figure 6.17 Distributions of sampling sites and 5 fish species across the Cheoah River watershed, including tributaries above Santeetlah Dam and within Calderwood Reservoir.

## 7. Conclusions

Less than two percent of the US's rivers run unimpounded (Vitousek et al. 1997). Given the current (and proposed) water demands and hydroelectric energy demands, the majority of large dams will remain in place. With over 82,000 dams in the US and the extensive damages to river systems impounded by those dams, providing management solutions on an individual-by-individual basis for every river system will be costly, inefficient, and, possibly, ineffective. Efficient and ecologically-relevant management will depend on the creation of frameworks that 1) operate at broad scales, 2) generalize patterns in habitat and ecology to provide a context for management solutions and standard protocols, and 3) focus management strategies to create measurable objectives.

Recent environmental flow management and potentially regulated river restoration at large, has been thwarted, at least to some degree, by the absence of quantitative, transferable flow-ecology relationships (Poff et al. 2010; Poff and Zimmerman 2010). The development of flow-ecology relationships should aid in prescribing the environmental flow needs of rivers with regulated or substantially altered flow. In response to these scientific needs, the Ecological Limits of Hydrologic Alteration (ELOHA) process was established as a framework for developing flow-ecology relationships and using that information to inform management and contribute to the social decision-making, process (Poff et al. 2010) (Fig. 1.1). Unfortunately, the ELOHA framework does not taken into account landscape-scale disturbance, morphological alterations, or temperature alterations when considering the development of flow-ecology relationships (Olden and Naiman 2010; Poff et al. 2010). However, the ELOHA process is the most holistic and widely-accepted framework for providing the basis for regulated river management at broad scales (e.g. regional); thus, I tested the utility of such a framework for providing environmental flow standards (basis for protocols) and the applicability at providing ecologically-relevant information.

Herein, I presented and used a 5-step restoration framework for establishing environmental flow standards and developing flow-ecology relationships as a guide for restoring regulated river flow (Fig. 1.2). My framework is a simplified version of the ELOHA process (Poff et al. 2010) and was inspired by the ideas of Arthington et al. (2006). The 5 step framework included: 1) developing a flow classification of unregulated stream flow, 2) relating flow classes to the landscape to develop a watershed predictive tool, 3) classifying regulated



rivers to natural flow classes, and 4) generalizing patterns in hydrologic alteration, and 5) evaluating the relationship between altered flow and ecology while considering the influences of channel morphology, substrate, and temperature within a landscape disturbance context (Fig. 1.2).

I applied steps 1-4 of my framework to streams in an 8-state region of the southeastern US. Altogether, my results suggest that patterns in natural flow dynamics and hydrologic alterations can successfully be placed within a framework to provide the basis and context for environmental flow management (Chapters 2-4). I successfully created flow classes from hydrologic information to provide the context for developing environmental flow standards and flow-regime targets. I also developed a classification tree that accurately classified 74% of the streams to their assigned class in the eight-state region using climate, precipitation, and soils data. I then used the classification tree to assign regulated rivers to natural flow classes to provide a context to assess and generalize hydrologic alterations. Interestingly, I was able to generalize patterns in how dams influence stream flows within and across different flow classes. Thus, classes provide the baseline for measuring hydrologic alteration or departures.

Step 5 of my framework differed than that of the ELOHA process because I included the influence of temperature, morphology, and landscape disturbance in my assessment of flow-ecology relationships (fish assemblages). Interestingly, the influence of altered flow on fish assemblages, at least at the scale of the upper Tennessee River basin, was overwhelmed by the influence of channel morphology, land-use, fragmentation, and substrate at unregulated and regulated sites. In summary, I found that developing flow-ecology relationships for these systems would prove difficult due to the confounding factors listed above. Furthermore, my results suggest that if habitat fragmentation remains the same (no dam removal), then restoring morphology and substrate conditions may be the most effective approach to restoring fish assemblages in regulated systems.

Overall, the utility of my 5-step framework and the ELOHA framework is that both frameworks provide the basis for providing environmental flow standards and assessing patterns in hydrologic alteration for regulated rivers at broad spatial scales (i.e. regions). However, the management standards developed through the frameworks may not be ecologically relevant, especially for regulated rivers since other factors besides flow, such as habitat fragmentation and temperature, are dramatically influenced by dams. These results are important in the larger

context of regional river management. The development of regional frameworks for conservation, prioritization, or broad-scale management of river systems has increased substantially in recent years (McMahon et al., 2001; Snelder et al., 2004; Wollock et al., 2004; Sowa et al., 2007). In addition, the ELOHA framework has been adopted by The Nature Conservancy, one of the largest leading groups in the environmental flow arena, as the primary driver in the development of in-stream flow guidelines (Conservation Online 2011). My results suggest that frameworks created to inform broad-scale river management and restoration should not be dependent upon the development of flow-ecology relationships alone, but the interaction between flow, morphology, and temperature within a landscape disturbance context. In the case of mitigating the effects of extensive withdrawals or water supply planning, the ELOHA process may provide an adequate framework to develop the context and protocols for management solutions. However, dam regulation does not only influence flow, but also sediment transport, channel morphology, temperature, and habitat connectivity. Thus, for regulated rivers and potentially all rivers at large, a framework is needed to incorporate these relationships. My framework incorporated these relationships during the final step. However, I advise that morphology and temperature should be incorporated as variables into a hierarchical classification within the initial step. Then patterns in hydrologic alteration, along with morphological and temperature alterations, can be generalized and related to ecology to inform management.

## 8. Appendices

**8.1 Appendix A.** Percentages of total estimated surface water withdrawals (freshwater) in each category for 7 states (taken from Kenny et al. 2009). The total 2005 withdrawal is measured in million gallons day<sup>-1</sup>. Numbers with bold lettering and shading represent the dominant category of withdrawal consumption whereas only bold letters indicate the secondary category.

| State          | Public Supply | Domestic | Irrigation | Livestock | Industrial  | Thermoelectric | Total 2005 |
|----------------|---------------|----------|------------|-----------|-------------|----------------|------------|
| Georgia        | <b>21.9</b>   | 2.2      | 14.0       | 0.5       | 11.5        | <b>49.8</b>    | 5380       |
| Kentucky       | <b>12.9</b>   | 0.8      | 0.4        | 1.1       | 5.6         | <b>79.2</b>    | 4330       |
| Maryland       | <b>52.5</b>   | 5.7      | 3.8        | 0.7       | 3.5         | <b>33.7</b>    | 1350       |
| North Carolina | 8.1           | 1.4      | 2.6        | 1.1       | <b>12.9</b> | <b>73.8</b>    | 11300      |
| Tennessee      | <b>8.4</b>    | 0.3      | 0.5        | 0.3       | <b>8.0</b>  | <b>82.5</b>    | 10800      |
| Virginia       | <b>13.8</b>   | 1.8      | 0.7        | 0.4       | <b>14.1</b> | <b>69.2</b>    | 7080       |
| West Virginia  | 3.9           | 0.7      | 0.0        | 0.1       | <b>21.5</b> | <b>73.8</b>    | 4810       |

**8.2 Appendix B.** Percent Land cover and overall % changes from 1950 to 2000 for 6 level III ecoregions found in the study region. Data for 1973 and 2000 taken directly from Brown et al. (2005), Table 2. Data for 1950 was estimated from linear trends between 1973 and 2000. Urban information for 1950 was only available for metropolitan areas and exurban (suburbs) and would not be comparable to overall % urban areas in watersheds (Brown et al. 2005). n refers to the number of stream gage records with sufficient “pre-regulation” information found in each ecoregion.

| <b>Land use</b> | <b>Year</b>  | <b>Southeastern<br/>Plains</b> | <b>Northern<br/>Piedmont</b> | <b>Piedmont</b> | <b>Blue<br/>Ridge</b> | <b>Ridge<br/>and<br/>Valley<sup>1</sup></b> | <b>Central<br/>Appalachians</b> |
|-----------------|--------------|--------------------------------|------------------------------|-----------------|-----------------------|---|---------------------------------|
| n               |              | 1                              | 1                            | 16              | 10                    | 11  | 8                               |
| Urban           | 1950         | 7.8                            | 18.8                         | 8.1             | 5.2                   | 5.2   | 1.0                             |
|                 | 1973         | 9.0                            | 22.7                         | 11.9            | 6.1                   | 6.1   | 1.3                             |
|                 | 2000         | 10.4                           | 27.3                         | 16.4            | 7.2                   | 7.2   | 1.7                             |
|                 | Overall<br>% | 33                             | 45                           | 103             | 39                    | 39  | 77                              |
| Agriculture     | 1950         | 27.1                           | 40.5                         | 25.5            | 13.7                  | 13.7  | 7.7                             |
|                 | 1973         | 24.5                           | 37.7                         | 24.4            | 13.7                  | 13.7  | 7.4                             |
|                 | 2000         | 21.5                           | 34.4                         | 23.1            | 13.7                  | 13.7  | 7.1                             |
|                 | Overall<br>% | -21                            | -15                          | -9              | 0                     | 0   | -7                              |
| Forest cover    | 1950         | 54.2                           | 38.2                         | 63.8            | 80.5                  | 80.5  | 88.1                            |
|                 | 1973         | 53.3                           | 36.9                         | 59.8            | 79.5                  | 79.5  | 87.4                            |
|                 | 2000         | 52.3                           | 35.4                         | 55.1            | 78.3                  | 78.3  | 86.6                            |
|                 | Overall<br>% | -3                             | -7                           | -14             | -3                    | -3  | -2                              |

<sup>1</sup>Data unavailable for Ridge and Valley ecoregion – assumed to be same as Blue Ridge ecoregion.

**8.3 Appendix C.** Hydrologic alteration models for 13 of the 44 hydrologic indices (Table 1) with  $R^2$  greater than 0.15 ( $p < 0.0001$ ). Each model represents the best model using only 4 variables. The number below each variable indicates the proportion of the overall variation explained by that variable. RMSE indicates root mean squared error.

| Model                         | Var 1                 | Var 2                   | Var 3                  | Var 4                        | $R^2$ | $R^2$ adj | RMSE  |
|-------------------------------|-----------------------|-------------------------|------------------------|------------------------------|-------|-----------|-------|
| <b>Flow Variability</b>       | Dam_Stor<br>0.190     | Drain_Area<br>0.046     | Ag_Main_St<br>0.034    | Pop_Den<br>0.006             | 0.279 | 0.272     | 0.087 |
| <b>July Flow</b>              | Imperv_Basin<br>0.049 | Ndams<br>0.042          | Artifpath<br>0.042     | Urban_Main_St<br>0.023       | 0.157 | 0.148     | 0.045 |
| <b>November Flow</b>          | Drain_Area<br>0.113   | Urban_Main_St<br>0.0311 | Dam_Stor<br>0.005      | Dam_dens<br>0.0191           | 0.169 | 0.160     | 0.048 |
| <b>Low Pulse Count</b>        | Dam_Stor<br>0.164     | Urban_Basin<br>0.016    | Dam_dens<br>0.037      | Ag_Main_St<br>0.020          | 0.244 | 0.237     | 0.129 |
| <b>High Pulse Variability</b> | Dam_Stor<br>0.162     | Dam_dens<br>0.042       | Urban_Main_St<br>0.016 | Dis_Nearest_Dam_Maj<br>0.018 | 0.238 | 0.230     | 0.080 |
| <b>30-Day Minimum</b>         | Imperv_Basin<br>0.109 | Dam_Stor<br>0.063       | Pop_Den<br>0.010       |                              | 0.182 | 0.176     | 0.031 |
| <b>90-Day Minimum</b>         | Imperv_Basin<br>0.127 | Urban_Main_St<br>0.029  | Ag_Basin<br>0.026      | Withdrawal<br>0.011          | 0.192 | 0.184     | 0.025 |
| <b>Low Flow Duration</b>      | Dam_Stor<br>0.174     | Ag_Main_St<br>0.017     | Road_Dens<br>0.018     | Dam_dens<br>0.046            | 0.254 | 0.246     | 0.122 |
| <b>1-Day Maximum</b>          | Dam_Stor<br>0.237     | Drain_Area<br>0.085     | Ag_Main_St<br>0.033    | Dam_dens_Maj<br>0.007        | 0.362 | 0.355     | 0.164 |
| <b>3-Day Maximum</b>          | Dam_Stor<br>0.162     | Drain_Area<br>0.033     | Ag_Basin<br>0.024      |                              | 0.219 | 0.213     | 0.147 |
| <b>Flow Predictability</b>    | Drain_Area<br>0.519   | Dam_dens<br>0.007       | Artifpath<br>0.021     | Ndams_Maj<br>0.007           | 0.554 | 0.549     | 0.052 |
| <b>Rise Rate</b>              | Drain_Area<br>0.247   | Dam_Stor<br>0.049       | Ag_Main_St<br>0.010    |                              | 0.306 | 0.301     | 0.084 |
| <b>Number of Reversals</b>    | Dam_Stor<br>0.144     | Urban_Main_St<br>0.025  | Dam_dens_Maj<br>0.018  | Ndams_Maj<br>0.019           | 0.206 | 0.198     | 0.068 |

**8.4 Appendix D.** The occurrence of the 40 most common species found in in the study region at sites within each disturbance type. “Prop Sites” indicates the proportion of all 50 study sites in which each species was found. “X” indicates that the species was found at all sites in each regulation type, “C” indicates common or found at more than 1 site, and “UC” indicates uncommon or found only at 1 site.

|  | River Sites Aggregated by Disturbance Type |                  |             |                                   |                    |
|--|--|------------------|-------------|-----------------------------------|--------------------|
|  | Prop Sites                                 | Cheoah Diversion | Richland Ck | Hiwassee Chatuge (Bottom Release) | Hiwassee Diversion |
| <b>No. Sites</b>                               |  | 4                | 2           | 3                                 | 3                  |
| <b>Drainage Area (km<sup>2</sup>)</b>          |  | 471              | 111         | 645                               | 2828               |
| <b>Fragment Length (km)</b>                    |  | 17               | 21          | 15                                | 99                 |
| <b>Total No. of Species</b>                    |  | 19               | 13          | 25                                | 32                 |
| <b>Catostomidae</b>                            |  |                  |             |                                   |                    |
| <i>Catostomus commersonii</i>                  | 0.28                                       |                  | C           |                                   |                    |
| <i>Hypentelium nigricans</i>                   | 1  | X                | X           | X                                 | X                  |
| <i>Moxostoma anisurum</i>                      | 0.16                                       |                  |             |                                   |                    |
| <i>Moxostoma carinatum</i>                     | 0.22                                       |                  |             |                                   |                    |
| <i>Moxostoma duquesnei</i>                     | 0.62                                       | C                |             | X                                 | C                  |
| <i>Moxostoma erythrurum</i>                    | 0.3  |                  |             |                                   | UC                 |
| <i>Moxostoma macrolepidotum</i>                | 0.2  |                  |             |                                   |                    |
| <b>Centrarchidae</b>                           |  |                  |             |                                   |                    |
| <i>Ambloplites rupestris</i>                   | 0.82                                       | X                |             | C                                 | X                  |
| <i>Lepomis auritus</i>                         | 0.62                                       |                  | X           | UC                                | X                  |
| <i>Lepomis cyanellus</i>                       | 0.4  | UC               | X           |                                   | C                  |
| <i>Lepomis macrochirus</i>                     | 0.74                                       | C                | X           | C                                 | X                  |
| <i>Micropterus dolomieu</i>                    | 0.66                                       | X                |             |                                   | X                  |
| <i>Micropterus punctulatus</i>                 | 0.24                                       |                  |             |                                   | UC                 |
| <i>Micropterus salmoides</i>                   | 0.48                                       | UC               | X           | C                                 | C                  |
| <b>Cottidae</b>                                |  |                  |             |                                   |                    |
| <i>Cottus bairdii</i>                          | 0.72                                       |                  |             | X                                 | X                  |
| <i>Cottus carolinae</i>                        | 0.24                                       |                  |             | UC                                | C                  |
| <b>Cyprinidae</b>                              |  |                  |             |                                   |                    |
| <i>Campostoma anomalum</i> & <i>oligolepis</i> | 0.98                                       | X                | X           | X                                 | X                  |
| <i>Clinostomus funduloides</i>                 | 0.18                                       |                  |             |                                   |                    |
| <i>Cyprinella galactura</i>                    | 0.76                                       | X                | X           | C                                 | X                  |
| <i>Hybopsis amblops</i>                        | 0.22                                       |                  |             | C                                 |                    |
| <i>Luxilus chrysocephalus</i>                  | 0.16                                       |                  |             |                                   | X                  |
| <i>Luxilus coccogenis</i>                      | 0.84                                       | X                |             |                                   | X                  |
| <i>Nocomis micropogon</i>                      | 0.86                                       | X                |             |                                   | X                  |
| <i>Notropis leuciodus</i>                      | 0.66                                       |                  |             |                                   | X                  |
| <i>Notropis photogenis</i>                     | 0.28                                       |                  |             | X                                 |                    |
| <i>Notropis spectrunculus</i>                  | 0.54                                       |                  |             | X                                 | X                  |
| <i>Notropis telescopus</i>                     | 0.36                                       |                  |             |                                   | UC                 |
| <i>Rhinichthys atratulus</i> or <i>obtusus</i> | 0.24                                       |                  |             | C                                 | UC                 |
| <i>Rhinichthys cataractae</i>                  | 0.64                                       | UC               | X           | X                                 |                    |
| <i>Semotilus atromaculatus</i>                 | 0.24                                       |                  |             | X                                 |                    |

**Appendix D. (continued)**

|  | River Sites Aggregated by Regulation Type |   |   |   |  |
|--|---|---|---|---|--|
|  | Prop Sites                                | Cheoah Diversion                                    | Richland Ck   | Hiwassee Chatuge (Bottom Release)   | Hiwassee Diversion   |
| <b>No. Sites</b>                                 |   | 4   | 2   | 3   | 3  |
| <b>Percidae</b>                                  |   |   |   |   |  |
| <i>Etheostoma blennioides</i> or <i>gutselli</i> | 0.74                                      | UC  |   | UC  | X  |
| <i>Etheostoma chlorbranchium</i>                 | 0.44                                      | C   |   |   |  |
| <i>Etheostoma rufilineatum</i>                   | 0.46                                      |   |   | UC  | X  |
| <i>Etheostoma simoterum</i>                      | 0.2                                       |   |   |   | UC   |
| <i>Etheostoma vulneratum</i>                     | 0.18                                      |   |   |   |  |
| <i>Etheostoma zonale</i>                         | 0.34                                      |   |   |   | X  |
| <i>Perca flavescens</i>                          | 0.28                                      |   |   | X   |  |
| <i>Percina aurantiaca</i>                        | 0.46                                      | C   |   |   | X  |
| <i>Percina evides</i>                            | 0.46                                      |   |   |   |  |
| <b>Petromyzontidae</b>                           |   |   |   |   |  |
| <i>Ichthyomyzon greeleyi</i>                     | 0.42                                      |   |   | X   |  |
| <b>Salmonidae</b>                                |   |   |   |   |  |
| <i>Oncorhynchus mykiss</i>                       | 0.76                                      | C   | C   | X   |  |
| <i>Salmo trutta</i>                              | 0.44                                      | UC  | X   | UC  |  |
| <b>Other Species found in each type</b>          |   | <i>Pylodictis olivaris</i><br><i>Sander vitreus</i> | <i>Pomoxis nigromaculatus</i><br><i>Salvelinus fontinalis</i> | <i>Ameiurus nebulosus</i><br><i>Lepomis gulosus</i><br><i>Notemigonus crysoleucas</i> | <i>Etheostoma camurum</i><br><i>Lepomis microlophus</i><br><i>Minytrema melanops</i><br><i>Percina caprodes</i><br><i>Pylodictis olivaris</i><br><i>Percina squamata</i> |

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