INFORMING FLOW MANAGEMENT DECISIONS IN THE MIDDLE OCONEE RIVER

by

STEVEN KYLE MCKAY

(Under the Direction of Alan P. Covich)

ABSTRACT

Whether justified by economic, environmental, or socio-cultural ends, there is a strong imperative for thoughtful management of freshwater resources. Meeting this need requires a framework for river management decisions that is transparent, fair, repeatable, and capable of explicitly stating trade-offs. Although the importance of "environmental" or "instream" flows is widely acknowledged, challenges arise in specifically identifying what flow regime is needed to obtain a desired ecological state. Taken as a whole, this dissertation sought to inform environmental flow decision making in the Middle Oconee River near Athens, Georgia.

Operated by a four-county authority, Bear Creek Reservoir is an off-channel, pump-storage reservoir, which withdraws water from the Middle Oconee River for municipal water supply. This dissertation applied a structured decision making framework to examine some of the economic and ecological trade-offs associated with alternative pumping schemes. In particular, the collective body of this dissertation addressed a single question: How can the water authority withdraw water from the Middle Oconee River with the least ecological impact?

Rather than recommending a flow regime, my goal was to inform decision-makers of the advantages and disadvantages of alternative withdrawal schemes. To do so, I developed a variety of techniques which incorporate novel dimensions not commonly addressed in environmental flow decisions (e.g., data visualization, hydrologic simulation, effectiveness analysis, value-laden decisions). As applied to the Middle Oconee River, these analyses indicate that current regulatory and management approaches are consistently outcompeted by novel environmental flow schemes relative to both withdrawal volumes and ecological endpoints of hydrologic similarity, fish recruitment, sediment transport, and organic matter transport. Although developed in the context of a single river, this dissertation provided a structured and rational approach which is broadly applicable to informing environmental flow decision making.

INDEX WORDS: Environmental Flows, Middle Oconee River, Structured Decision Making, Effective Discharge

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

Introduction

With freshwater biodiversity in sharp decline (Strayer and Dudgeon 2010, Collen et al. 2014) and over half of the world's large rivers dammed (Nilsson et al. 2005), the need for ecologically effective river management can hardly be overstated (Baron et al. 2002). A key component of conserving, managing, and restoring river ecosystems is the environmentally sensitive operation of water resources infrastructure such as locks and dams, and diversions (Freeman and Marcinek 2006, Richter et al. 2006).

As demand on freshwater increases, water and environmental managers must trade-off potentially conflicting uses of this resource, one of which is the maintenance of aquatic ecosystem integrity (Baron et al. 2002, Postel and Richter 2003, Arthington et al. 2006). In some cases, the process of analyzing trade-offs can identify solutions where numerous outcomes benefit (i.e., win-win scenarios, King and Brown 2010). Although the importance of "environmental" or "instream" flows is widely acknowledged, challenges arise in specifically identifying what flow regime is needed to obtain a desired ecological state (Richter et al. 1996).

"Environmental flows describe the quantity, timing, and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems" (Brisbane Declaration 2007). This consensus definition from environmental flow scientists and managers succinctly summarizes the challenges of trading-off ecological and socio-economic objectives in water management. However, environmental flow decision-making is further complicated by alternative flow regimes (Tharme 2003, Richter et al. 2011, Arthington 2012, McKay 2013), numerous ecological endpoints (Richter et al. 2006), various ecologically relevant components of a river's flow regime (Poff et al. 1997, Bunn and Arthington 2002, Matthews and Richter 2007), and hydrologic variability (Poff 2009).

Study Site: Middle Oconee River

This study was conducted in the Middle Oconee River near Athens, Georgia (Figure 1.1). The Middle Oconee River is a sixth-order tributary of the Altamaha River with a drainage basin of 1,031 km^2. Characteristic of the southern Appalachian Piedmont, watershed land use was historically altered by late 1800s cotton farming with poor sediment management (Jackson et al. 2005, Trimble 2008) and is currently altered by urban and suburban development (Grubaugh and Wallace 1995) with minor amounts of animal agriculture (Fisher et al. 2000).

Common to the Piedmont, river morphology alternates between low-slope, sandy reaches (i.e., "runs") and high-slope, bedrock reaches (i.e., "shoals"). Runs comprise greater than 90% of river length and are characterized by low velocities and shifting sandy substrate. Shoals are shallow, swift-flowing habitats with high velocity, relatively immobile substrate, and relatively high light penetration. These contrasting velocity, light, substrate, and temperature conditions induce uniquely different carbon and food web dynamics, and accordingly contain strikingly different ecological communities. Generally speaking, shoals of the Middle Oconee River are better studied than runs because of their importance as local biodiversity hotspots, susceptibility to drought impacts, and their importance in reaeration. Some ecological processes studied in Middle Oconee shoals include: secondary production (Nelson 1957, Nelson and Scott 1962, Grubaugh and Wallace 1995), aquatic macrophytes (Pahl 2009), benthic macroinvertebrates

(Katz 2009, W.G. McDowell unpublished data), and benthic fish population dynamics (R.A. Katz and M.C. Freeman unpublished data).

In 2002, the Upper Oconee Basin Water Authority constructed Bear Creek Reservoir to serve as a municipal water supply source for a four-county region (UOWBA 2005, GAWP 2011). Bear Creek is a tributary to the Middle Oconee River, and the off-channel reservoir is filled by pumping water from the mainstem of the Middle Oconee River during periods of sufficiently high flows (Campana et al. 2012). The reservoir is permitted to withdraw a maximum of 60 million gallons per day (MGD; Georgia EPD Permit Number 078-0304-05). Historically, state environmental flow regulations would have established minimum flow downstream of the intake corresponding to the annual 7-day low flow with a 10-year recurrence interval (i.e., the “7Q10”, GA DNR 2001) or 45 cubic feet per second (cfs) for the Middle Oconee River (Carter and Putnam 1978). However, since 2001, the state has applied a monthly minimum flow policy corresponding to the monthly 7Q10 (GA DNR 2001).

The U.S. Geological Survey operates a streamflow monitoring station downstream of the reservoir intake near Athens, Georgia (USGS Gage No. 02217500). Continuous, daily discharge records have been collected from 1938 to present. From 1938-2012, mean discharge was 496 cfs) and median discharge was 329 cfs. However, significant hydrologic variability exists at this site with observed daily minimum and maximum flows of 3.5 and 13,300 cfs, respectively.

The unified government of Athens-Clarke County (ACC) operates a separate withdrawal on the Middle Oconee River at Ben Burton Park. Here, I focus exclusively on the effects of withdrawals to Bear Creek Reservoir. In doing so, I examine the reach from the reservoir intake near Tallassee Shoals dam to Ben Burton Park (~7.5 river miles) and neglect the effects of the ACC withdrawal in the gage records.

Although off-channel reservoirs can reduce direct effects on fluvial habitat and water quality, intakes can also provide substantive changes in ecological communities due to shifts in the flow regime (Freeman and Marcinek 2006). The long record of pre-reservoir river discharge measurements provides an appropriate data set with which to examine potential withdrawal schemes and accompanying environmental flows relative to a minimally altered condition. Furthermore, extensive ecological studies on this river (highlighted above) provide the background knowledge and data to assess the ecological effects of alternative environmental flow regimes. Bear Creek Reservoir's off-channel location also provides an opportunity to examine the effects of flow regimes in isolation from the confounding effects of connectivity such as those of a large dam (Freeman and Marcinek 2006, Kanno and Vokoun 2010).

Project Overview

In this dissertation, I use a structured approach to examine environmental flow decision making in the Middle Oconee River. In particular, I apply the PROACT framework of Hammond et al. (1999). In this framework, PRoblems are first characterized, Objectives are set, Alternatives are developed, Consequences are examined, and Trade-offs are made between objectives. In addition to the core elements of this framework, uncertainty, risk tolerance, and connectedness among decisions are explicitly addressed in order to better clarify the decision context. The following paragraphs examine how this dissertation addresses each of these topics in relation to the Middle Oconee River.

Problem: The permitted withdrawal rate for Bear Creek Reservoir represents a substantial portion of river discharge (60 MGD = 92.8 cfs), particularly during the late summer months when flow rates are lowest (September mean discharge = 237 cfs). Although the water authority currently withdraws less than 20 MGD based on a daily average (GAWP 2011, Campana et al.

2012), conflicts over water allocation and withdrawal are most effectively addressed before they occur (Baron et al. 2002). The collective body of this dissertation addresses a single question: How can the water authority withdraw the most water from the Middle Oconee River with the least ecological impact?

At the crux of this problem is the issue of managing withdrawals on the Middle Oconee River under significant hydrologic variability. River ecosystems are inherently variable environments (Poff and Ward 1989, Poff et al. 2006), subject to multiple sources of periodic, stochastic, and catastrophic change (Sabo and Post 2008). Hydrologic variability is defined broadly as both predictable and stochastic changes in river discharge, stage, or other hydrologically mediated variables. Adequate understanding of hydrologic variability is imperative for river management issues such as flood risk management, early detection of drought, and environmental flow management (Poff et al. 2006). To better understand the magnitude of this problem, I applied a variety of techniques for visualizing time series data (Chapter 2). These methods are well-developed in the discipline of visual analytics (e.g., Aigner et al. 2011), but ecologists and hydrologists have not widely adopted these methods. These methods assisted in understanding the more than four orders of magnitude of river discharge observed in the Middle Oconee.

Objectives: Water withdrawal decisions on the Middle Oconee River were assessed relative to two primary objectives: (1) maximize withdrawal rates and (2) minimize environmental impact. Clearly, these simplified objectives only approximately represent the economic and environmental endpoints rather than a more nuanced discussion of environmental flows. However, these objectives are suitable for the purpose of a first-order analysis comparing the relative merits of environmental flow schemes. Future analyses could include other objectives for river management such as maximizing recreational opportunities and minimizing water treatment costs.

Alternatives: Development of environmental flow alternatives often requires a combination of social, physical, and life sciences due to the complex trade-offs involved in water management. As a testament to this complexity, over 200 techniques have been used to develop environmental flows in more than 44 countries (Tharme 2003). Here, I define environmental flow alternatives in the general sense of any plan for managing water for environmental objectives. I conducted a literature review of these alternative techniques (McKay 2013 and Chapter 3), and found that four general categories of alternative environmental flow regimes are typically highlighted (Jowett 1997, Tharme 2003, Arthington et al. 2003, Acreman and Dunbar 2004): hydrologic methods, hydraulic rating, habitat analysis, and holistic methodologies. Recently, two additional environmental flow methods have emerged from the literature and been applied extensively: optimization and regionalization.

While many methods exist, hydrologic methods relying on simple operational rules remain easier to implement and commonly applied (Arthington et al. 2006, Poff 2009, Richter 2010). In particular, minimum flow criteria are often still used as default methods for water regulation decisions across large spatial scales (Shiau and Wu 2004, Richter 2010, Snelder et al. 2011, Ouyang 2012). The deficiencies in minimum flow criteria are well-characterized and include issues such as lack of consideration of a river's entire flow regime (Poff et al. 1997), an ecologically inappropriate "one size fits all" approach (Poff 2009), "flat-lining" of hydrographs (Richter et al. 2011), and selection of ecologically arbitrary hydrologic thresholds (Arthington et al. 2006). Other simple hydrologic rules have been proposed such as "high flow skimming" where water withdrawals occur only during high flows (Richter and Thomas 2007), “sustainability boundaries” where withdrawals occur on the basis of a percentage of unmodified river discharge (Postel and Richter 2003, Richter 2010, Richter et al. 2011), and classification of a river’s flow regime into environmental flow components (Postel and Richter 2003, Matthews and Richter 2007).

Although other environmental flow methods have been encouraged over hydrologic rules (Arthington et al. 2006), the broad application and simple implementation of these methods justifies continued investigation (Richter 2010, Snelder et al. 2011, Ouyang 2012). In the Middle Oconee River, I examine trade-offs over a range of minimum flows and sustainability boundaries. This analysis is not meant to imply that these environmental flow methodologies are preferable to more complex analyses, but instead to add complexity and objectivity to the most common methods of environmental flow analysis.

Consequences: Quantitative metrics were developed for each objective and assessed for each alternative action. The long period of pre-reservoir discharge records provided an opportunity to examine the potential effects of alternative environmental flow schemes and associated withdrawal patterns. For the purpose of this analysis, only discharge data from 1938-1997 were applied. Although the reservoir was completed in 2002, 1997 was used as a temporal break point for this analysis because these data were available to regulators prior to reservoir operation (i.e., during the permitting process). Historical data can be limited with respect to legacy changes in the watershed or future challenges from non-stationary climate (Milly et al. 2008, Poff and Matthews 2013); however, long-term hydrologic data serves a valuable role in the relative assessment of environmental flow alternatives.

Long-term average withdrawal rates provided a useful measure of municipally available water yield (Vogel et al. 2007). Average annual withdrawal rate in millions of gallons per day was computed for each flow scenario. In some cases, this metric was normalized by dividing the average annual withdrawal rate by the maximum permitted withdrawal rate. The resulting metric provided a consistent zero to one scale for comparison with the environmental metrics described below.

The second objective (minimize environmental impact) was assessed relative to two metrics. In the first analysis (Chapter 4), I applied hydrologic similarity as a surrogate for environmental impact and assumed that ecological impacts increased as the flow regime diverged from the unaltered hydrograph. A common method for conceptualizing a hydrograph and accompanying environmental flows is to break down the "flow regime" into its constituent parts, namely: magnitude, frequency, duration, timing, and rate-of-change (Poff et al. 1997). Over 170 metrics have been proposed to characterize a river's flow regime (e.g., Richter et al. 1996, Henriksen et al. 2006), many of which are redundant (Olden and Poff 2003). Here, I applied the seven "fundamental daily streamflow statistics", which were selected based on an analysis of metrics that parsimoniously summarize flow regimes (Archfield et al. 2013): mean, coefficient of variation, skewness, kurtosis, the auto-regressive lag-one correlation coefficient, amplitude of the seasonal signal, and phase shift of the seasonal signal. Following computation, each metric was normalized using the baseline condition of no withdrawal, where one is identical to the unaltered flow regime (i.e., the no withdrawal scenario).

In the second analysis (Chapter 5), I examined the ecological endpoint of young-of-year fish production. To do so, I applied a flow-dependent model for fish recruitment (Craven et al. 2010) to each environmental flow scenario. To date, environmental flows have primarily centered on habitat provision for habitat for game species and imperiled species (Poff 2009). This analysis moves beyond habitat-based analyses (e.g., Bovee and Milhous 1978, Stalnaker et al. 1995, Hickey and Fields 2009, Milhous et al. 2012) to demographic processes (Shenton et al. 2012). I also applied effective discharge analysis from the discipline of geomorphology (Wolman and Miller 1960) to address both the magnitude and frequency of ecological response. This framework had been previously adapted to ecological processes (Doyle 2005, Doyle et al. 2005, Wheatcroft et al. 2010); however, its application was limited to instantaneous responses to discharge (i.e., ecological processes correlated with daily discharge). The application presented here extends this framework to incorporate flow timing, duration, and rate-of-change. The metrics derived using these two modeling approaches address the total quantity of young-of-year production expected for five focal taxa occurring in the Middle Oconee River. As before, each metric was normalized using the baseline condition of no withdrawal, where one is identical to the unaltered flow regime (i.e., the no withdrawal scenario).

Trade-offs: At its most basic level, a trade-off is giving up one thing to gain another (Yoe 2002). When metrics are similar, trade-off analysis may be straightforward. However, when metrics are dissimilar, trade-offs become less clear without additional analyses (McKay et al. 2012). Trade-offs in the Middle Oconee River were examined in three alternative ways: direct comparison, normalization and combination, and value-laden combination.

First, because there are few metrics (i.e., one for withdrawal, seven for hydrologic similarity, and one for each of five focal fish taxa), normalized environmental metrics were compared to withdrawal rates directly (Chapters 4 and 5).

Second, by normalizing quantities to a consistent scale (i.e., 0 to 1), environmental metrics were combined into a single hydrologic similarity metric and a single fish recruitment metric. These combined metrics were used to assess overall trade-offs between water withdrawal and environmental endpoints (Chapters 4 and 5).

Third, I examined the effects of alternative value judgments on trade-offs (Chapter 4). I combined the normalized water withdrawal rate and the normalized hydrologic similarity metric using a weighted average. In doing so, I was able to vary the relative weight on water supply and ecological endpoints. This overall "utility" score for each alternative allows the testing of alternative value judgments on decision making (e.g., if a stakeholder valued only municipal water supply). Using this integrated value assessment, I was able to find environmental flow alternatives and associated flow thresholds that maximize both objectives simultaneously.

Uncertainty: The outcome of any decision is somewhat uncertain, and uncertainties are particularly pervasive in environmental decision making (Ascough et al. 2010). The goal of any uncertainty analysis is not to eliminate, but instead clarify key sources of uncertainty (Hammond et al. 1999). In this spirit, I addressed uncertainty as follows.

In the analysis of fish recruitment (Chapter 5), I examined the sensitivity of the recruitment model (Craven et al. 2010) to input parameters and found that the relative model outcome was independent of a particularly uncertain model input, adult fish density. I also studied the sensitivity of trade-offs under three sets of model coefficients: expected values, lower 90% confidence interval, and upper 90% confidence interval (Craven et al. 2010). This analysis demonstrated consistency in decision recommendations regardless of parameterization.

Because of the novel application of effective discharge analysis to environmental flow decisions, I developed a separate analysis examining the sensitivity of effective discharge metrics (Chapter 6). The objective of this study was to quantify how uncertainty in estimated frequency distributions and rating curves propagate through to three metrics of discharge effectiveness (Wolman and Miller 1960, Doyle and Shields 2008, Klonsky and Vogel 2011). I focused on transport of sediment and organic matter due to the high applicability of the effectiveness framework and the ecological relevance of these processes (Doyle et al. 2005). Results indicated effectiveness metrics are more sensitive to uncertainties in the frequency distribution than the rating curve, which highlights the often-emphasized importance of adequate discharge records (Kennard et al. 2010). Sensitivity analysis is part of any rigorous model development process (Schmolke et al. 2010), and this analysis supported the adaptation of the effectiveness framework to novel applications in environmental flow decision making.

Risk Tolerance: Individuals vary significantly in their capacity to accept uncertainty and risk. For instance, an agricultural stakeholder using irrigation to reduce impacts of drought might be willing to accept a 0.1% chance of not meeting municipal water supply needs, while crossing this threshold may be unacceptable to the public utility manager. Rather than impose a particular risk-averse or risk-taking behavior, I examined the effects of different value judgments on overall outcomes (Chapter 4). While this approach does not explicitly address risk, it does inform water utility decision makers with the effects of alternative value judgments.

Linked Decisions: Often a decision influences what options are available in the future (Hammond et al. 1999). Environmental flows represent an interesting case of a decision that is both independent and dependent. From one view, water withdrawals are nothing more than a series of independent decisions made on a daily basis. On the other hand, by electing not to withdraw one day, a reservoir may require pumping on the next to maintain sufficient water levels. Likewise, an environmental flow recommendation could be considered a one-time decision (e.g., the establishment of a statewide regulatory threshold) or a sequential decision. Many authors have emphasized the value of framing environmental flows as sequential decisions, flow experiments, and opportunities for adaptive management (e.g., Konrad et al. 2011, Olden et al. 2014).

In the Middle Oconee River, the decision of when to declare drought is tightly linked with environmental flow decisions. Under drought declaration, water use restrictions are instituted which reduce overall water demand (UOBWA 2005). The decision of when to declare drought can then influence how much water is being used for municipal and environmental purposes. To this end, I examined the ability of the current drought declaration criteria and found them quite predictive relative to two competing models despite their relative simplicity (Appendix A).

Summary

Taken as a whole, this dissertation seeks to inform environmental flow decision making. I apply the PROACT framework (Hammond et al. 1999) for structured decision making to examine the consequences and trade-offs associated with multiple alternative flow regimes in the Middle Oconee River. Rather than recommending a flow regime, my goal is to inform decision-makers of the advantages and disadvantages of alternative withdrawal schemes. These analyses incorporate novel dimensions not commonly addressed in environmental flow decisions (e.g., effectiveness analysis, value-laden decisions); however, many more ecological processes and flow regime alternatives could be examined, and significant opportunities remain for future research (Chapter 7). Altogether, this dissertation provides a structured and rational approach to informing environmental flow decision making both in the Middle Oconee River and for rivers nationwide.

Figure 1.1. Middle Oconee River near Athens, Georgia.

CHAPTER 2

VISUALIZING LONG-TERM DATA: A CASE STUDY OF HYDROLOGIC VARIABILITY[1]

[1] S. Kyle McKay. Submitted to *Bioscience*, 02/10/2014.

Abstract

The era of "big data" has arrived, and the ecological community of practice must be prepared not only to store, share, and analyze these data but also visualize, understand, and communicate them. Rapid and meaningful interpretation of large data sets will only become more relevant as sensor capacity and memory increase and as massive data streams are generated in long-term and large-scale projects (e.g., National Ecological Observatory Network). Visual exploration of time series data can identify patterns not apparent from purely quantitative approaches, guide quantitative analyses, and effectively communicate findings. Although line graphs will continue to provide valuable insight, newer methods of presenting time series data are also available. This paper reviews visualization techniques for investigating long-term time series data relative to historical levels of variability at a single site. A case study of streamflow is presented to demonstrate the utility of modern visualization methods for observing patterns in hydrologic variability.

Introduction

Visual exploration of time series data is a powerful tool for increasing understanding of complex and long-term data sets (Helsel and Hirsch 2002, Keim et al. 2008, Fox and Hendler 2011). Data visualization methods are well-studied in fields of visual analytics, information visualization, computer graphics, and scientific communication (e.g., McCormick et al. 1987, Tufte 2001, Keim et al. 2008, Aigner et al. 2011). However, ecologists, environmental scientists, and hydrologists rarely take advantage of these powerful visualization methods.

Large data sets are increasingly available in ecology (e.g., U.S. Geological Survey stream gage network, National Ecological Observatory Network), and effective visualization techniques will be crucial to rapid and efficient communication of these observations (Michener and Jones

2012, McCulloch 2013). Visualization cannot substitute for more rigorous quantitative and statistical methods (Gabrecht and Fernandez 1994, Helsel and Hirsch 2002). However, visual exploration takes advantage of the powers of the human eye to rapidly detect and discern visual patterns, when presented effectively (McCormick et al. 1987, Keim et al. 2008, Healey and Enns 2012).

Although the classic line plot of a time series often communicates important components of long-term variability, over 100 techniques exist for presenting time series data (Aigner et al. 2011). In this paper, I aim to highlight some time series visualization methods that could be readily applied to ecological data sets. In particular, I emphasize long-term time series data, which are challenging to condense into meaningful figures because of the need to present the data of interest at high resolution and in historical context. While these visualization methods are more than a decade old, they have not yet been widely adopted by the hydrology or ecology communities-of-practice.

A Typology of Time Series Visualization

Because of varied applications of time series data, a myriad of techniques for visualization exist, which can be customized for a specific purpose. Aigner et al. (2007, 2011) provide a basic typology of time series visualization methods that relies on three criteria: (1) the characteristics of the time axis, (2) the data to be presented, and (3) how data are to be represented. This section briefly reviews these criteria relative to visualizing ecological time series.

The properties of the temporal data direct the form of time series visualization. Time can be presented in ordinal (e.g., first, second), discrete (e.g., monthly), or continuous (e.g., 12:02AM, 12:07AM, 1:08AM, 2:13AM) forms. Data can also be presented as discrete

Euclidean points or intervals with a specified duration (e.g., river discharge on Dec-18 at 9:08AM v. average-daily discharge on Dec-18, respectively). However, this distinction can become irrelevant as the scale of a plot increases relative to a single data point (e.g., a daily-averaged discharge shown with 10,000 sequential values resembles a Euclidean point). Time can proceed linearly from past to future or cyclically over a natural periodicity. Finally, time can be ordered to sequentially occur, or it may be branching to account for alternative trajectories (e.g., multiple forecasts of hurricane tracks or future climate scenarios).

The type of data is the second criteria distinguishing visualization methods. At the broadest scale, visualizations are quickly guided by the use of quantitative (e.g., 7 mg/L) versus qualitative (e.g., juvenile, adult) variables and univariate versus multivariate data. Data may also be distinguished based on their spatial representation, or lack thereof.

The final criteria guiding visualization is associated with how the time series will be represented. Time series data can be presented in both two- and three-dimensions depending upon the application of interest, although three-dimension figures risk obscuring relevant data (Strandhagen et al. 2006). In contrast to dynamic representations (i.e., animation, Muller and Schumann 2003), static visualizations must reflect passage of time in a single image. Visualizations can also incorporate user-interaction that provides for active exploration of the data (Andrienko and Andrienko 2005, Keim et al. 2008)

In this paper, a subset of time series visualization methods is presented that are particularly suited to presenting ecological data. Time is considered non-ordinal and sequential, but multiple examples are shown that treat time as a discrete, interval-based, periodic measurement and a continuous, point-based, serial measurement. While comparison between multiple variables (e.g., discharge, temperature) or locations (e.g., paired watersheds) is often required, techniques presented here focus on presentation of one data set at a single location. Only static, non-interactive, and two-dimensional data representations are presented here. Although beyond this paper's scope, additional techniques are available for other types of time, data, and representation conditions (e.g., Aigner et al. 2011, http://survey.timeviz.net/).

Case Study: Visual Exploration of Hydrologic Variability

Background

River ecosystems are inherently variable environments (Poff and Ward 1989, Poff et al. 2006), subject to multiple sources of periodic, stochastic, and catastrophic change (Sabo and Post 2008). In particular, hydrologic (i.e., flow) variability is a well-known driver of processes ranging from species adaptation (e.g., Lytle and Poff 2004) to community composition (e.g., Poff and Allan 1995) to ecosystem process rates (e.g., Doyle 2005). Fortunately, the U.S. Geological Survey currently operates more than 9,000 active streamflow gaging sites, many with long-term records exceeding 50 years. Thus, hydrologic variability provides an opportune case study for demonstrating the utility of time series visualization methods for ecological applications.

Hydrologic variability is defined broadly as both predictable and stochastic changes in river discharge, stage, or other hydrologically-mediated variables. Adequate understanding of hydrologic variability is imperative for river management issues such as flood risk management, early detection of drought, and environmental flow management (Poff et al. 2006). Numerous statistical methods have been applied to recognize, understand, and predict variability in hydrographs. Hydrographs are often summarized as ranges of flow, mean or median discharges, return intervals, and other statistical indices (Olden and Poff 2003), and a variety of software packages have been developed to facilitate these calculations (e.g., Indicators of Hydrologic Alteration, Richter et al. 1996; NATHAT, Henriksen et al. 2006; GeoTools, Bledsoe et al. 2007).

Probability distributions and flow duration curves are other common methods of condensing data into a more understandable and usable format (Bledsoe et al. 2007). Inter-annual variability has also been addressed by classifying hydrographs both among rivers and for a single river (Poff and Ward 1989, Booth et al. 2006). Although these and other methods are extremely useful, they often reduce data into summary formats (e.g., indices, distributions, classes), which can result in the loss of hydrographic information such as timing or duration of floods and droughts.

Middle Oconee River, Athens, Georgia

The Middle Oconee River in Athens, Georgia is a sixth-order tributary of the Altamaha River with a 398 mi^2 drainage basin. The U.S. Geological Survey operates a long-term gage at this site (USGS Gage number 02217500) with continuous daily discharge data from 1938-2012 and 15-minute discharge data from 1999-2012. Over the 75 year period of record, the mean annual discharge is 496 cfs with an observed daily minimum of 3.5 cfs and maximum of 13,300 cfs. In 2000, Bear Creek reservoir, an off-channel pump-storage reservoir, was constructed as the primary water supply for four surrounding counties. The reservoir is filled primarily by withdrawing water directly from the Middle Oconee river. This case study illustrates the value of visualization in understanding hydrologic variability in a watershed where drought could threaten water availability for municipal withdrawal.

In 2012, the region experienced widespread drought as indicated by the 2nd lowest mean and the 3rd lowest median annual discharges on record. The river was below the regulatory minimum flow of 45 cfs for 20 days, which prohibited withdrawals. Understanding the depth and severity of this drought relative to the four orders-of-magnitude of observed river discharge can inform water withdrawal decision-making and accompanying environmental flows. Here, visualization is applied to examine 2012 discharge data in the context of the historical streamflow variability.

Visualizing Hydrographs

As a first examination of the 2012 drought, daily discharge records are presented as a line plot, the simplest and most common form of graphical time series visualization (Figure 2.1). Three alternative scales are used to emphasize different components of hydrologic variability: (A) a linear axis, (B) a log-transformed axis, and (C) a normalized axis (Kelleher and Wagener 2011). Although log-transformed, Figure 2.1B is shown in units of discharge (rather than logarithm of discharge) to increase intuitive understanding. Figure 2.1C has been standardized by the mean and standard deviation (Garbrecht and Fernandez 1994). As a reference point, the long-term mean discharge has been displayed in each figure to provide context for the data and highlight the depth of the 2012 drought. This type of plot feature can also be used to emphasize thresholds important to ecological or management concerns, such as a decline in wetted area or a point where pumping can no longer occur (Michener and Jones 2012).

To provide the historical context of the 2012 drought, Figure 2.2 presents all streamflow data from 1938-2012. Although 2012 can be seen as an exceptionally dry year, individual data points are dwarfed by the sequential presentation of the additional 27,000+ data points and associated change in aspect ratio (Kelleher and Wagener 2011). Nevertheless, important observations can be gleaned from this figure. For instance, periodicity emerges as a trait of the Middle Oconee system (particularly in Figure 2.2B), and the extreme magnitude of high flows relative to low flows become more clear (particularly in Figure 2.2A and 2.2C).

Figure 3 takes advantage of annual periodicity using "envelope" plots (Andrienko and Andrienko 2005, DePhilip and Moberg 2010). Figure 2.3A presents each annual hydrograph as a separate line, and while the individual years and data points are obscured, the general trends relative to 2012 are easily observed. Figures 2.3B and 2.3C present 2012 data compared to summary statistics for each day of the year (minimum, mean, and maximum discharges and 10th, 50th, and 90th percentile discharges, respectively). These figures show that high discharge values are somewhat aseasonal and can occur year round, but low river levels show much more predictable, seasonal trends. The level of 2012 drought is also easily observed relative to the historical context provided by these long-term summary statistics.

Pixel-based figures have also emerged as a powerful visualization method for cyclic data sets (Keim 2000). These figures use consistent plot structure oriented to two temporal scales, and then present data (i.e., discharge) as colors (Figure 2.4A). Importantly, discharge data must be distributed into “bins” for this type of analysis (logarithmic bins were used here), and bin thresholds should be selected carefully. A rectangular version of these plots has been previously applied to hydrographic settings to demonstrate long-term effects of dams (Koehler 2004, Stranhagen et al. 2006). Spiral diagrams can also be used to present these data (Figure 2.4B), but care should be taken in selecting the period of the cycles as incorrect selection can dramatically alter the efficacy of a figure (Weber et al. 2001). Notably, spiral diagrams emphasize newer data with larger concentric rings, a quality which may be used to highlight recent data collection but could also over-emphasize the severity of recent conditions. In sharp contrast to Figures 2.1-2.3, multi-year droughts, seasonal trends, and the relative frequency of floods and droughts emerge quickly in both figures. Although the historic nature of the 2012 conditions is visible, it is not emphasized due to the relative size of the data points, but loss in resolution may be made up for by the sequential presentation of the data without statistical summary.

Figure 2.5 demonstrates the flexibility of pixel-based methods for different time scales. Figure 2.5A shows 15-minute discharge data for each day in a 3-month period. This type of high-resolution data collection is often challenging to visualize because of the massive number of data points generated (over 35,000 per year). On the other end of the temporal spectrum, Figure 2.5B presents monthly-averaged discharge for 1938-2012, which provides an interesting point of comparison to Figure 2.4A. Although the daily discharge figure shows 26,475 more data points, the monthly-averages provide a "less busy" version of the same seasonal trends and extent of the 2012 conditions (Kelleher and Wagener 2011).

The Utility of Multiple Visualizations

Here, multiple visualization techniques have been applied to understand the depths of the 2012 drought in the Middle Oconee River. Figure 2.1 provides minimal historical context (i.e., a mean), but also guides more sophisticated visualization methods (e.g., logarithmic axes and bins were applied in Figures 2.3 and 2.4). Figure 2.2 presents historical context and an observed periodic behavior, but 2012 data are not easily distinguished. Cyclic plots shown in Figures 2.3 and 2.4 facilitate comparison between 2012 data and sequentially-presented, long-term records. Figure 2.5 demonstrates how these visualization methods can be applied to data sets with alternative time scales that might also inform the analysis. Importantly, each method builds upon the previous, and together multiple visualizations summarize the varied aspects of 2012 drought conditions.

Selecting a Visualization Method

The vast array of options for time series visualization can make choosing a technique challenging. The following questions are proposed to help hydrologists and ecologists navigate these methods. These questions are merely intended to structure thinking on method selection and are presented in an approximate priority of importance based on the key issues in time series visualization (Aigner et al. 2007, 2011). Importantly, multiple methods can (and should) be applied simultaneously to explore a data set, emphasize specific components of a time series, and highlight different elements of variability.

(1) What is the best method for communicating a message? Purpose and objectives should always drive the selection of a visualization method (Kelleher and Wagener 2011). For instance, a simple line plot may be sufficient for communicating the time of sampling relative to recent hydrographic trends (e.g., Figure 2.1B). However, a pixel-based approach (e.g., Figure 2.4) might be more effective for communicating long-term trends such as drought periodicity to stakeholders. In particular, methods might change relative to the components of a data set that are of interest (e.g., floods vs. droughts, extremes vs. central tendencies).

(2) What are the relevant temporal scales? Relevant ecological time scales range from minutes to decades in both length and interval. For many ecological processes, valuable understanding can be gained from the presentation of data in the historical context of a long-term record. The time interval (e.g., 15-minute, hourly, daily, monthly, annual) relevant to a particular problem also influences the efficacy of a particular visualization method.

(3) How are data distributed? As evidenced by the Middle Oconee case study, environmental variability can occur over many orders of magnitude. Visualization can often be made more effective by rescaling data (e.g., Figure 2.1) or lumping data into frequency bins (e.g., Figure 2.4).

(4) What are the constraints of the visualization environment? Selection of methods is influenced by not only the need to present data effectively, but also the limits of the presentation medium (Aigner et al. 2011). The spatial extent, resolution, and use of color in a figure are often

limited by screen or page size and the medium of interest (e.g., journal article vs. presentation vs. interactive on-line infographic). This paper has focused on static presentation of a single time series, but the capacity to animate or interact with figures can increase visualization options significantly.

(5) What tools and expertise are available? A large variety of software is available to visualize time series data (e.g., Microsoft Excel, MATLAB, R Statistical Software, Indicators of Hydrologic Alteration). While all of these tools can visualize time series in multiple formats, not all visualizations are easily conducted in all programs (e.g., pixel-based methods). In addition to availability and capability of the software, there may be personnel limitations in terms of expertise or time availability, which should not be overlooked. Herein, the freely-downloadable, R statistical software package was used to develop all figures (R Development Core Team 2012), and code is provided in supplemental material to facilitate future applications.

Conclusions

Large data sets and long-term time series are becoming more common in ecology (Michener and Jones 2012, McCulloch 2013). Using readily-available long-term hydrologic data from the extensive U.S. Geological Survey stream gage network, this paper has highlighted a few (of many) techniques for visualizing time series data. Visualization cannot substitute for rigorous quantitative analyses, but it can inform the analyst and guide the analyses (Helsel and Hirsch 2002). As long-term data sets grow and statistical methods increase in sophistication, our ability to visualize data has also increased (Kelleher and Wagener 2011). However, broad adoption of novel visualization methods has lagged. Without first visualizing and understanding environmental variability, we will continue to struggle quantifying and managing it.

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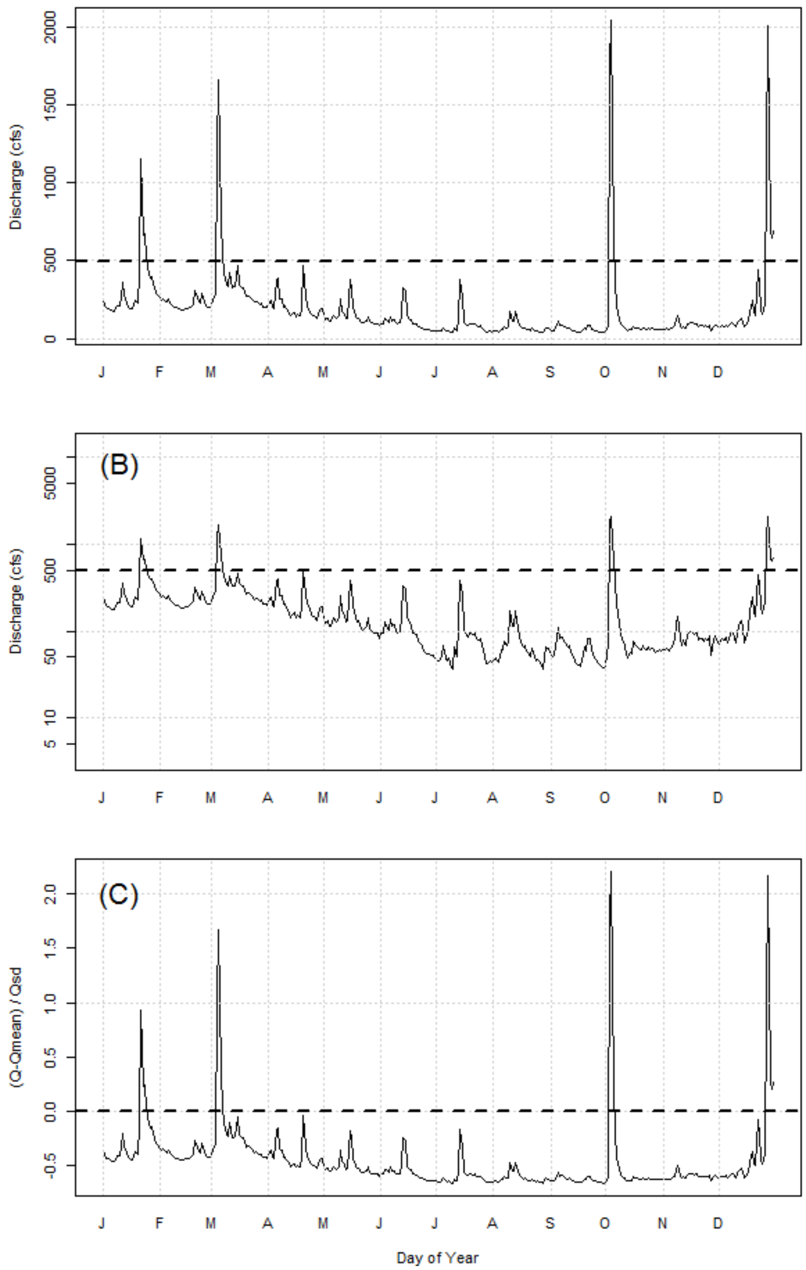

Figure 2.1. Streamflow hydrograph for 2012: (A) linear axis, (B) log-transformed axis, and (C) normalized axis. Mean discharge is shown as a dashed line in each figure.

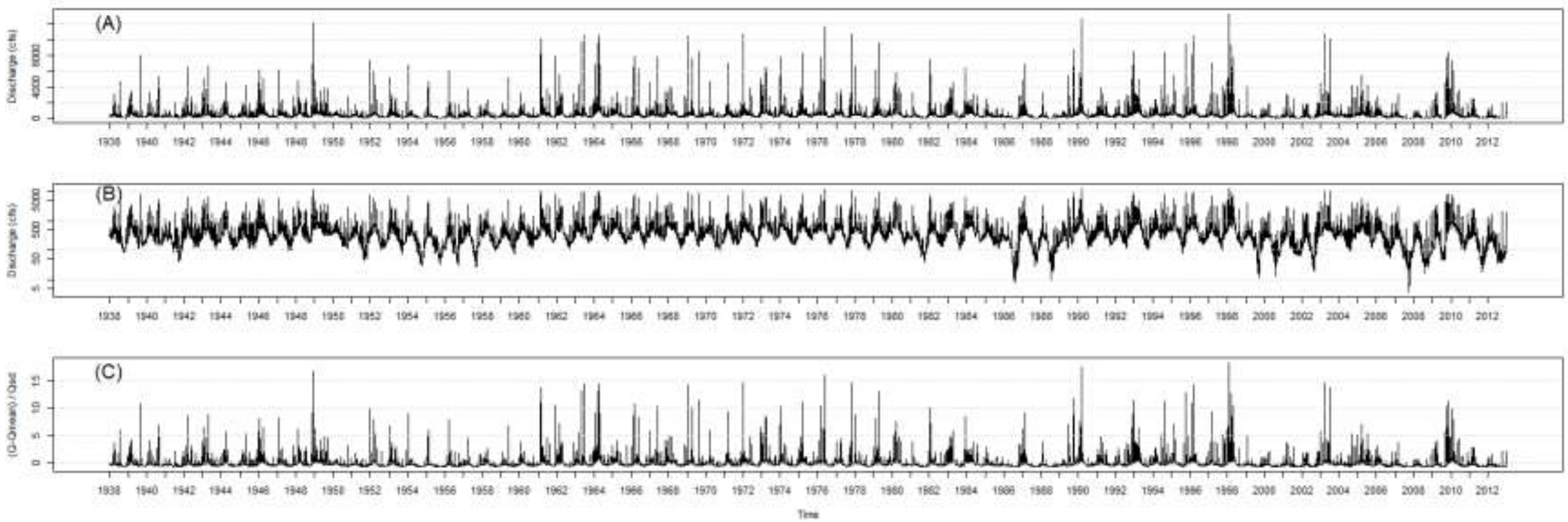

Figure 2.2. Streamflow hydrograph for the long-term period of record for: (A) linear axis, (B) log-transformed axis, and (C) normalized axis.

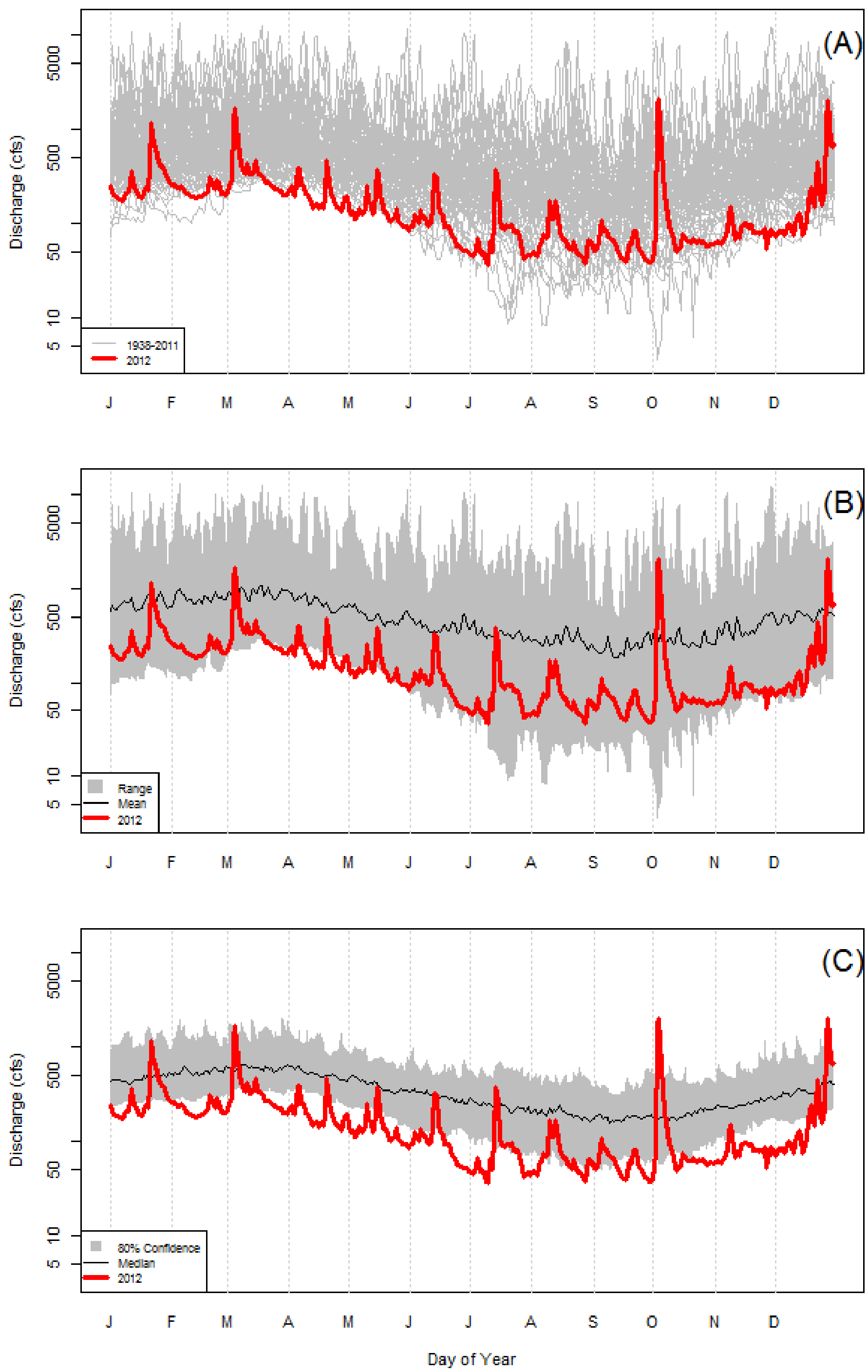

Figure 2.3. “Envelope” plots showing a hydrograph of interest (2012) relative to: (A) all observed historical hydrographs, (B) the range of observed hydrographs, and (C) the 80% confidence interval of observed daily discharge (i.e., the 10th and 90th percentile discharges).

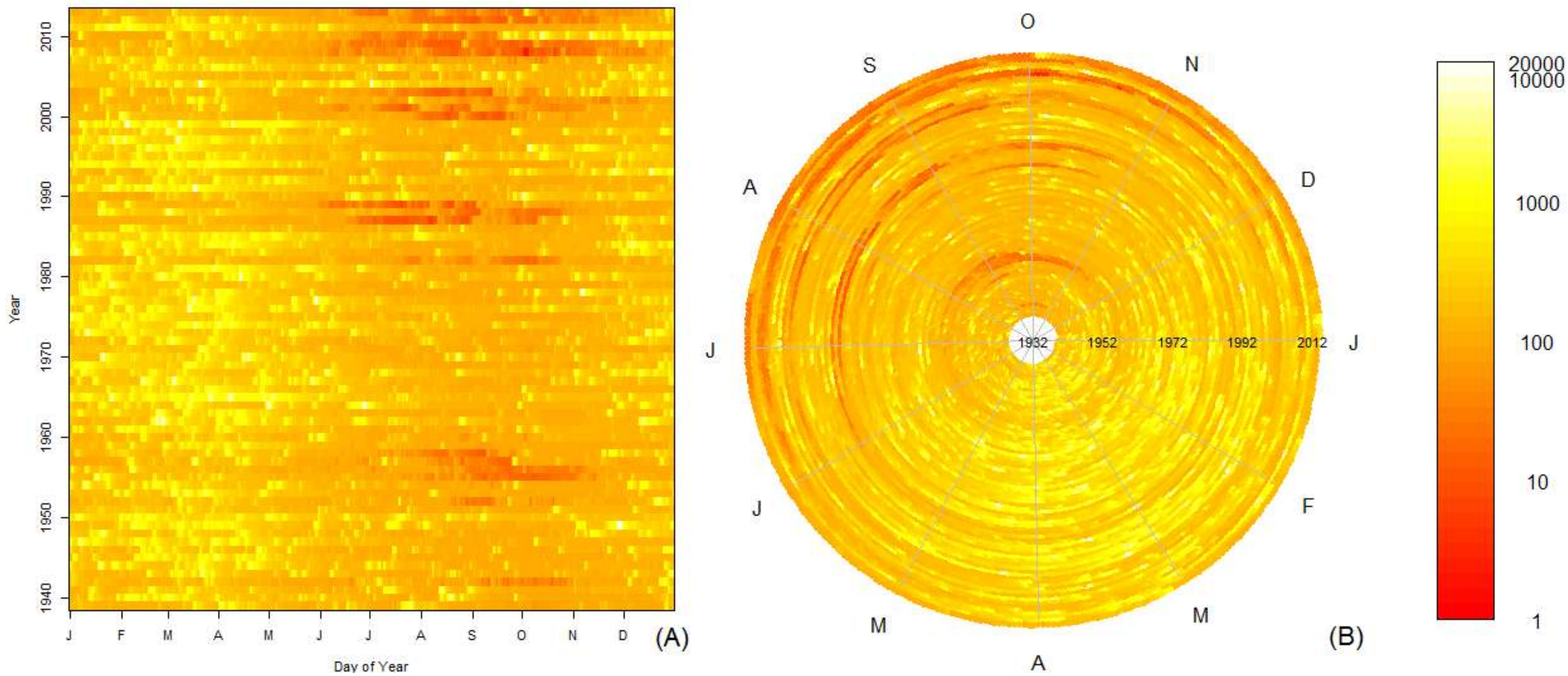

Figure 2.4. Pixel-based diagrams for long-term daily records in cubic feet per second: (A) raster-based and (B) spiral.

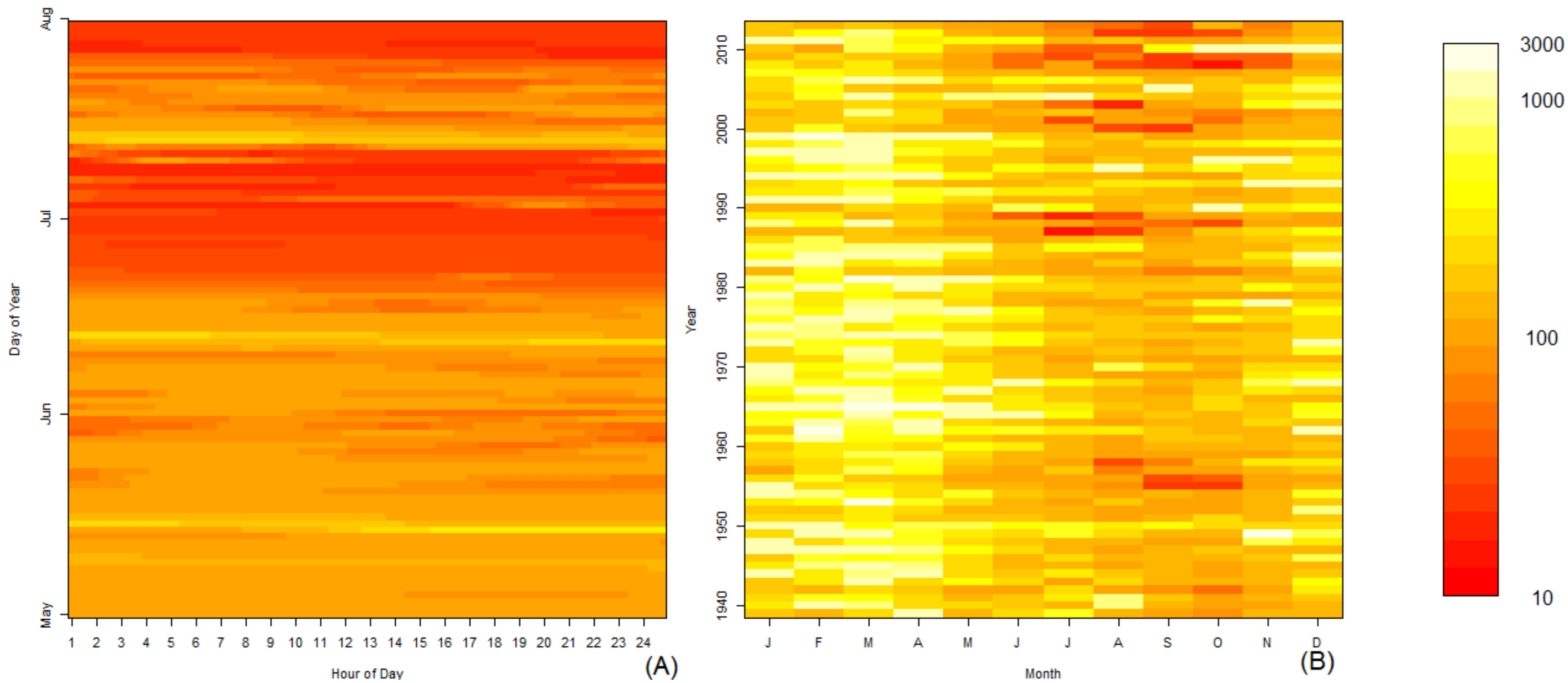

Figure 2.5. Raster hydrographs in cubic feet per second for: (A) 15-minute discharge data presented on a daily cycle from May-August 2012 and (B) monthly-averaged discharge presented on an annual cycle from 1938-2012.

CHAPTER 3

ALTERNATIVE ENVIRONMENTAL FLOW REGIMES IN THE MIDDLE OCONEE RIVER

Introduction

As demand on freshwater increases, water and environmental managers must trade off potentially conflicting uses of this resource, one of which is the maintenance of aquatic ecosystem integrity (Baron et al. 2002, Postel and Richter 2003, Arthington et al. 2006). Although the importance of "environmental" or "instream" flows is widely acknowledged, challenges arise in specifically identifying what flow regime is needed to obtain a desired ecological state (Richter et al. 1997). Furthermore, while flow regimes are crucial, other elements of water quality (e.g., sediment and temperature regimes) can also play disproportionately important roles in maintaining ecological integrity.

Different portions of a hydrograph may be of interest depending upon the objectives of a particular hydrologic analysis. For instance, a flood risk management project is likely interested in high stage events, whereas a municipal water supply may be interested in low flow conditions caused by drought. Likewise, ecological objectives are differentially addressed by hydrograph characteristics, and there is extensive documentation of the ecological importance of many components of a river's flow from natural low flows to floods (Bunn and Arthington 2002). For example, high velocity conditions may increase physical abrasion and breakdown of organic matter, while low velocity conditions are critical for residence times governing nutrient uptake. Conflicting social and environmental objectives have led to challenges in defining what is (and is not) an "environmental flow." To overcome this confusion, more than 750 scientists from 50 countries presented a consensus definition as: "Environmental flows describe the quantity, timing, and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems" (Brisbane Declaration 2007).

Development of environmental flow alternatives often requires input from social, physical, and life sciences due to the complex trade-offs involved in water management. As a testament to this complexity, over 200 techniques have been used to develop environmental flows in more than 44 countries (Tharme 2003). Here, environmental flow alternatives are defined in the general sense of any scheme or rubric for managing water for environmental objectives. These alternatives may vary from setting a minimum river flow (e.g., Tennant 1976) to identifying thresholds in ecological process from incremental investigation of discharge (e.g., Stalnaker et al. 1995) to expert panel recommendations (e.g., Richter et al. 2006).

While many methods exist, it remains unclear whether these methods produce similar recommendations for management. In this chapter, my objective is to develop environmental flow recommendations using a variety of methods in order to determine the relative utility of these approaches. To this end, this chapter will: (1) briefly review alternative methods presented in the literature, (2) apply a subset of these methods to a case study in the Middle Oconee River, and (3) qualitatively compare the different recommendations made by alternative methods.

Environmental Flow Alternatives

Environmental flow methods are commonly classified in four general categories of alternatives (Jowett 1997, Tharme 2003, Arthington et al. 2003, Acreman and Dunbar 2004, Kilgour et al. 2005, de Freitas 2008, Navarro and Schmidt 2012): (1) hydrologic methods, (2) hydraulic rating, (3) habitat analysis, and (4) holistic methodologies. Recently, two additional environmental flow methods have emerged from the literature and have been applied extensively: optimization and regionalization. The following sections describe each general class of alternatives, highlight selected techniques within that class, and provide references for further investigation.

Hydrologic Methods

The most commonly applied methods for environmental flow management have historically been simple hydrologic rules (Tharme 2003, Acreman and Dunbar 2004). These rules are typically based on hydrologic indices calculated from historically observed discharge data at daily, monthly, or annual time-scales (e.g., Tennant 1976). Often these methods are manifested in a minimum flow level such as the 7-day averaged low flow recurring on a 10-year frequency (7Q10; Evans and England 1995), the 2-year frequency 30-day averaged low flow (30Q2; Peterson et al. 2011), a percentage of mean annual flow (Tennant 1976), a percentage of a flow duration curve (Acreman and Dunbar 2004), or other criteria (Tharme 2003). In some cases, these minimum flows are specified for monthly, rather than annual, periods (Evans and England 1995). However, minimum flows only regulate low flow and do not address high flow, which is broadly acknowledged as a significant short-coming (Poff et al. 1997).

Minimum flows are by far the most common hydrologic method, but other simple operational rules are possible. For instance, a “peak shaving” approach encourages water withdrawal during high flows, when withdrawal is a smaller percentage of total river discharge (Richter and Thomas 2007); however, these approaches are often avoided because suspended materials are at their highest concentrations during peaks. Alternatively, Richter (2010) and Richter et al. (2011) propose a “sustainability boundary” approach where discharge is allowed to fluctuate within a specified range, expressed as a percentage of the natural flow.

Hydrologic methods are generally intended to provide a desktop technique for rapid application when extensive ecological data collection or analyses are not feasible. These methods are often used in conjunction with existing hydrologic and ecological data to understand the natural range of variability within the flow regime (Poff et al. 1997, Richter et al. 1997, 1998). The ecological significance of these rules should be well-established prior to application throughout a region (Tharme 2003, TNC 2009). Methods targeting multiple components of the flow regime (low flows, high flow pulses, floods, etc.) are generally preferred to those setting only minimum flows (Poff et al. 1997, Mathews and Richter 2007, Navarro and Schmidt 2012).

Hydraulic Rating

Hydrologic data may be translated into hydraulic parameters such as wetted perimeter, wetted cross-sectional area, hydraulic radius, velocity, depth, or shear stress using channel cross-sectional data and hydraulic analyses. When using hydraulic methods for environmental flows, these parameters are an implicit surrogate for overall habitat provision (Navarro and Schmidt 2012). The relationship between discharge and a hydraulic variable is often examined for critical thresholds or breakpoints, which become minimum flow recommendations (Gippel and Stewardson 1998). Hydraulic methods may also be used to examine sediment movement, water quality, or other hydraulically-mediated processes of interest (Navarro and Schmidt 2012). Although these methods are well-established, Tharme (2003) points out that these methods have shown few recent advances and are largely being used in conjunction with similar, yet more robust, habitat-based methods.

Habitat Analysis

Hydraulic analyses may be used in conjunction with the physical conditions required by plant or animal species to develop detailed discharge-habitat relationships. These relationships may be developed for a variety of focal taxa for a given study and are often considered in terms of "weighted useable area" (Acreman and Dunbar 2004). This suite of techniques includes the Physical Habitat Simulation (PHABSIM; Bovee and Milhous 1978, Gore and Nestler 1998) module of the Instream Flow Incremental Methodology (IFIM; Nestler 1993, Stalnaker et al. 1995), the Riverine Community Habitat Assessment and Restoration Concept (RCHARC; Peters et al. 1995), and a variety of other methods (Milhous et al. 2012). Habitat analyses have utilized a variety of output formats from steady-state techniques to flow duration approaches to time series of habitat availability. Habitat models have become more precise as resolution of hydraulic models has increased to two and three dimensions and spatial data becomes finer in scale. A number of numerical tools exist to facilitate these analyses by importing data from hydraulic programs such as the system for environmental flow analysis (SEFA; Milhous et al. 2012, Payne and Jowett 2013) or the spatially-explicit Ecosystem Functions Model (HEC-EFM; Hickey and Fields 2009). Because of the structured analytical approach, these techniques are often more repeatable than other methods (Acreman and Dunbar 2004). Implicitly embedded in many applications of these methods are the assumptions that changes in habitat are directly correlated to changes in a population (e.g., survival and recruitment rates) and that channels are static and fixed rather than dynamic and changing environments (e.g., sediment starvation below a dam). Importantly, habitat analyses do not explicitly develop a flow recommendation, but instead serve as an analytical platform to compare and trade-off the effects of flow recommendations.

Holistic Methods

The methods described thus far have focused on general hydrologic and hydraulic conditions and habitat for a specific set of target or focal taxa. However, many authors recommend a broader (i.e., more holistic) perspective of the aquatic ecosystem as a whole, and numerous structured approaches have been developed to address ecosystem-wide environmental flow recommendations (Tharme 2003, Arthington et al. 2003). Holistic methods typically consider many components of an aquatic ecosystem including geomorphology, hydraulic habitat, water quality, invertebrates, vertebrates (e.g., fish, mammals, amphibians, birds, and reptiles), adjacent riparian or floodplain vegetation communities, downstream estuarine communities in coastal rivers, and adjacent human settlements. Because many ecological and social endpoints may be involved, alternative analytical methods may vary from empirical, field study to theoretical, desktop modeling (Arthington et al. 2003).

When considering multiple ecological outcomes, environmental flow recommendations are typically constructed in one of two logical frameworks (Arthington et al. 2003). "Bottom-up" construction begins with a null scenario of zero river discharge and adds elements to the flow regime for each ecological process of interest (i.e., a restoration of flow). Some common examples of bottom-up approaches include: the Building Block Methodology (BBM; King et al. 2008), the Flow Restoration Methodology (FLOWRESM; Arthington et al. 1999), the "Wissey" method (Petts et al. 1999), and the "Savannah Process" (Richter et al. 2006). Conversely, "top-down" methods begin with the null scenario of an unaltered hydrograph and remove discharge until undesirable ecological outcomes are reached (i.e., a restriction management threshold). Some common examples of top-down approaches include: the benchmarking method (Brizga et al. 2001), Downstream Response to Imposed Flow Transformations (DRIFT; King et al. 2003, Arthington et al. 2003b), and the Ecological Limits of Hydrologic Alteration (ELOHA; Poff et al. 2010).

Optimization

Reservoir operation for non-environmental objectives historically has relied on rule curves which specify release targets throughout a given year. These curves may be written as a set of objective functions and constraints, which may then be subjected to optimization analyses. Recently a number of authors have extended these analyses to include environment flows as objectives, penalty functions, and constraints for optimization (Suen and Eheart 2006, Shiau and Wu 2007, Suen et al. 2009, Chen 2010, Yin et al. 2010, Yin et al. 2012). Although these methods show great promise, environmental flows must be written as a set of quantifiable objectives and/or constraints, which requires significant understanding of the system. In conjunction with increased computational capacity, development of more sophisticated optimization algorithms capable of addressing multiple reservoirs in series, multiple competing objectives, and stochastic processes will only lead to greater utility of these analyses for environmental flow management (Labadie 2004).

Regionalization

Recent environmental flow methods build from prior techniques to define flow recommendations at a regional scale. The Ecological Limits of Hydrologic Alteration (ELOHA) framework approaches environmental flow management from a holistic perspective which couples interacting scientific and social processes (Poff et al. 2010). The framework represents an internationally collaborative effort to present a consensus framework for environmental flow management, which can be applied to multiple rivers within a region simultaneously. The flexible approach emphasizes hypothesis testing and adaptive management and consists of four primary steps for scientific evaluation: (1) building a hydrological foundation of baseline and altered hydrographs, (2) classifying rivers into regionally distinct types for broad scale application, (3) analyzing flow alterations for each unique river class, and (4) developing flow-ecology relationships for each river type. Although quite new, this framework's broad consensus view and regional approach have already been applied in many locations across the United States (Apse et al. 2008, DePhilip and Moberg 2010, Kendy et al. 2012, USACE 2012), Spain (Belmar et al. 2011), China (Zhang et al. 2012), New Zealand (Snelder et al. 2011), and more are in development. In addition to the ELOHA framework, regional statistical methods have been applied to predict both streamflow (e.g., Archfield et al. 2009, Knight et al. 2012) and accompanying ecological responses (e.g., Knight et al. 2008, 2013).

Middle Oconee River Case Study

In this section, I apply a case study of the Middle Oconee River as a concrete, albeit simplified, example of the recommendations made by alternative environmental flow methods. The Middle Oconee River near Athens, Georgia is a middle order stream in the north Georgia Piedmont. Land use was historically agricultural, but is increasingly influenced by urban and suburban pressures. In 2002, a four county water authority completed Bear Creek Reservoir, which provides an off-channel storage facility for Middle Oconee River water. The authority is permitted to withdraw a maximum of 60 million gallons per day (MGD; Georgia EPD Permit Number 078-0304-05) from the river, but withdrawal rates are typically less than 20 MGD (Campana et al. 2012). The U.S. Geological Survey operates a streamflow monitoring station downstream of the withdrawal site (398 mi^2) with daily records from 1938 to present. This section presents simplified applications of competing environmental flow methods to this site with the intent of side-by-side comparison. I only examine hydrologic records from 1938-1997 (60 years) to avoid the confounding effects of existing reservoir operations. Furthermore, I do not apply optimization and regionalization methods due to their recent development and extensive data requirements.

Hydrologic Methods

The most common technique for developing environmental flow recommendations is the use of hydrologic rules. The multitude of methods (> 60, Tharme 2003) prohibits application of all methods simultaneously. Table 3.1 presents the flow recommendations associated with a subset of methods applied to the Middle Oconee based on recommendations (Evans and England 1995) and regulations (GA DNR 2001) in the state of Georgia. Notably, all of these methods are centered on developing minimum flows either on a monthly or annual basis. Table 3.1 also presents minimum flows developed through flow duration approaches applied in Florida (Ouyang 2012). Rather than examining a single minimum flow, I apply a range of potential annual and monthly minimum flows throughout this dissertation (scenarios below). The "peak shaving" approach is effectively a scenario with high minimum flow, and is thus captured by these scenarios. Additionally, the sustainability boundary approach was also applied (Richter et al. 2011). Specifically, I examine the following scenarios:

- Annual minimum flow (MFL): This method assigns a single, year-round flow threshold below which water may not be withdrawn. Although well-acknowledged as an imperfect method for environmental flow provision (Arthington et al. 2006, Freeman and Marcinek 2006, Poff 2009, Richter et al. 2011), minimum flows remain extensively applied in practice (Tharme 2003, Kanno and Vokoun 2010). MFL was varied from 0 to 1,000 cfs by 10 cfs.
- Monthly minimum flow (mMFL): This method assigns a monthly-varied flow threshold below which water may not be withdrawn. This common adjustment to the MFL

approach incorporates elements of flow timing not captured in annual minimum flows (Hughes and Mallory 2008). This method is also the current environmental flow approach for the state of Georgia (GA DNR 2001). mMFL was varied in 101 intervals from the minimum observed monthly-averaged flow to the maximum observed monthly-averaged flow for the 60-year record for each of the 12 months.

- Sustainability boundaries (SB): As a simple, first-order alternative to minimum flows, Richter (2009) and Richter et al. (2011) offer a percent-of-discharge approach, which they call sustainability boundaries and propose as the "presumptive standard" for simplistic environmental flow rules. SB, or the percent of daily discharge available for abstraction, was varied from 0 to 50% water abstraction by 0.5%.

Figure 3.1 shows the hydrologic implications of a few common hydrologically based environmental flow recommendations. The long-term Middle Oconee River hydrograph is provided for relative comparison.

Hydraulic Rating

Three cross-sectional surveys were collected in a 400 foot section of river downstream of the withdrawal location in Ben Burton Park. A normal flow hydraulic model was constructed in the R statistical software (R Core Development Team 2012). The model was executed for steady state discharges from 10 to 329 cfs (long-term median) with an average water surface slope of 0.00053 (Barnes 1967), a channel roughness of Manning's n = 0.04, and a floodplain roughness of Manning's n = 0.10. Figure 3.2A shows the change in wetted perimeter (a surrogate for bed area) for each cross-section. Following Gippel and Stewardson (1998), a threshold change was visually identified and a minimum flow corresponding to 200 cfs was selected.

Habitat Analysis

The hydraulic model described above also projected changes in cross-sectional area with discharge. Although habitat analyses often focus on more detailed assessment of instream habitat for focal taxa (e.g., PHABSIM, Bovee and Milhous 1978), I applied cross-sectional area as a surrogate for available habitat. The frequency of each discharge was computed, and area values were multiplied by frequency to obtain estimates of frequency-weighted habitat. I assume that the peak of this curve provides an ecologically important threshold, and the corresponding discharge of 225 cfs was used as a minimum flow threshold (Figure 3.2B). This analysis is far less complex than most habitat based methods, and certainly does not include the negotiation associated with the Instream Flow Incremental Method (IFIM, Stalnaker et al. 1995). However, this demonstrates a key limitation of habitat methodologies; these methods do not provide flow recommendations outright, but instead an analytical framework for comparing flow regimes.

Holistic Methods

Most holistic methods rely on a panel of local subject matter experts and water managers for assembling environmental flow recommendations. A simplified example of a holistic flow recommendation is shown in Table 3.2 for the Middle Oconee River. The effect of withdrawal on high flows is minimal due to pumping capacity limitations above 60 MGD (92.5 cfs). As such, these recommendations largely focus on low flows. Extreme low flows prohibit withdrawal consistent with state regulations (GA DNR 2001). However, low flows apply a strict sustainability boundary of 10% rather than a minimum flow due to the preference to maintain flow variability, but also protect the ecological community during particularly vulnerable periods. However, as flow increases, the sustainability boundary is increased to 15% to accommodate the reduced ecological risk. At very high flows, the effect of the withdrawal is minimal, and withdrawals are allowed to occur freely.

Comparing Environmental Flow Recommendations

The remainder of this dissertation focuses on techniques for comparing the consequences and trade-offs associated with alternative environmental flow regimes. However, here I compare the hydrologic effects of these different methods to highlight the relative benefits of alternative techniques for developing flow recommendations. Figure 3.3 presents hydrographs for four alternative environmental flow regimes relative to the unaltered hydrograph: annual minimum flow, monthly minimum flow, sustainability boundary, and holistic recommendations. Importantly, the scenarios shown all represent equal average annual withdrawal rates of approximately 29.5 MGD over the 60 years simulation period. Although the hydrographs exhibit some visible differences (e.g., "flat-lining" in May for Figure 3.3B), the effects are challenging to detect without additional analysis. Figure 3.3F present the percent change in the hydrograph on a given day relative to the unaltered condition. The differences in the methods become more apparent when presented in this light. For instance, annual and monthly minimum flows exhibit greater variability relative to the sustainability boundary and holistic methods.

In addition to site-specific comparisons, a number of authors have reviewed the relative merits of environmental flow methods (Jowett 1997, Arthington et al. 2003, Tharme 2003, Acreman and Dunbar 2004, Freeman 2005, Kilgour et al. 2005, de Freitas 2008, Navarro and Schmidt 2012, McKay 2013). No environmental flow management scheme is appropriate to all applications, and in many cases multiple techniques may be applied to a single environmental flow problem (e.g., informing initial decisions with a more basic approach as data are collected for more complex analyses). Table 3.3 presents a qualitative comparison of these methods based on prior reviews of the topic. Although this table provides a basis for side-by-side evaluation, methods are often applied together. For instance, a holistic method may be used to identify objectives and an optimization approach could be applied to numerically identify the flow targets. Alternatively, a habitat methodology could be used as the quantitative criteria for a holistic analysis. Distinguishing between techniques for establishing environmental flow targets, forecasting the effects of environmental flows, and implementing flows can be challenging given the intertwining nature of methods.

Although significant progress has been made, environmental flow science remains young, and many lingering questions persist, particularly with respect to designing flow regimes for desired ecological responses. This chapter has presented six alternative schemes for developing environmental flow recommendations. Simplified applications of these methods were presented for the Middle Oconee River, and future work should emphasize better characterizations of habitat, improved holistic recommendations based on a range of expert opinions, and the development of optimization and regionalization approaches. Furthermore, additional work to characterize the consequences and trade-offs of alternative flow regimes will be needed.

Regardless of the methods applied in the Middle Oconee (or elsewhere), environmental flow recommendations always contain uncertainty and should be considered hypotheses about the response of a stream ecosystem to a given driver (usually discharge). Environmental flows and their ecological outcomes should be monitored to validate hypotheses and adaptively managed accordingly (Poff et al. 2003, Richter et al. 2003, Postel and Richter 2003). In some cases, environmental flow releases may be experiments used to test a hypothesis (Konrad et al. 2011, Olden et al. 2014). Careful monitoring of any environmental flow method should be conducted to avoid crossing tipping points or irreversible thresholds.

Acknowledgements

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Table 3.1. Summary of minimum flow recommendations for the Middle Oconee River based on multiple hydrologic methods and a single hydraulic rating method using wetted perimeter.

Minimum flow regulation	**Minimum flow (cfs)**											
	Jan	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Georgia State Regulations (GA DNR 2001)												
7Q10 –Regulation Prior to 2001	37	37	37	37	37	37	37	37	37	37	37	37
Monthly 7Q10 – Current Regulation: Option 1	247	283	316	289	185	133	113	67	53	88	146	175
Current Regulation: Option 2	Site-specific analysis approved by Georgia’s Environmental Protection Division.											
Mean annual flow – Current Regulation: Option 3	156	156	156	156	156	156	156	156	156	156	156	156
Mean annual flow – Current Regulation: Option 3 with water supply modification	313	313	313	313	209	209	156	156	156	156	156	209
Other Minimum Flow Approaches (Evans and England 1995)												
30Q2 (Peterson et al. 2011)	158	158	158	158	158	158	158	158	158	158	158	158
Tennant Method (30% MAF)	156	156	156	156	156	156	156	156	156	156	156	156
Modified Tennant Method (20% MAF)	104	104	104	104	104	104	104	104	104	104	104	104
South Carolina Method	209	209	209	209	156	156	104	104	104	104	104	156
Arkansas Method (30% Monthly Flow)	221	242	277	224	161	124	101	92	71	89	122	158
New England Base Flow Method (August Median)	252	252	252	252	252	252	252	252	252	252	252	252
Modified New England Base Flow Method (September Median)	223	223	223	223	223	223	223	223	223	223	223	223
Minimum monthly mean (applied here)	140	251	285	233	162	64	25	41	47	42	113	123
Flow Duration Curve derived Minimum Flows (Ouyang 2012)												
Florida: Minimum infrequent high (95% ex)	111	111	111	111	111	111	111	111	111	111	111	111
Florida: Minimum frequent high (80% ex)	195	195	195	195	195	195	195	195	195	195	195	195
Florida: Minimum average (50% ex)	350	350	350	350	350	350	350	350	350	350	350	350
Florida: Minimum frequent low (20% ex)	632	632	632	632	632	632	632	632	632	632	632	632
Florida: Minimum infrequent low (5% ex)	1400	1400	1400	1400	1400	1400	1400	1400	1400	1400	1400	1400
Hydraulic Rating and Habitat Analysis Methods												
Discharge associated with a threshold in Wetted Perimeter (Evans and England 1995, Gippel and Stewardson 1998, Figure 2)	200	200	200	200	200	200	200	200	200	200	200	200
Frequency-weighted cross-sectional area (a surrogate for habitat provision)	225	225	225	225	225	225	225	225	225	225	225	225

Table 3.2. Holistic environmental flow recommendations for the Middle Oconee River using an “environmental flow components” approach (Matthews and Richter 2007).

Environmental Flow Component	Middle Oconee Recommendation
Extreme low flows (Less than the Monthly 7Q10)	State-regulated prohibition of withdrawals.
Low flows (Monthly 7Q10 to Monthly median)	Withdrawals should observe a 10% sustainability boundary to avoid undesirable changes to the ecosystem. This should be observed as an instantaneous threshold (rather than a daily average).
High flow pulses (Monthly median to 920 cfs)	Withdrawals should observe a 15% sustainability boundary as risk to the ecological community is reduced in this range. This should be observed as an instantaneous threshold (rather than a daily average).
Small and large floods (Greater than 920 cfs)	Withdrawals may occur freely and at the discretion of the water authority. The minimum of this range equates to approximately 10% of river discharge when pumps are operating at full capacity, 60MGD (92.5 cfs).

Table 3.3. Comparing alternative environmental flow methods. Developed following prior reviews (Jowett 1997, Arthington et al. 2003, Tharme 2003, Acreman and Dunbar 2004, Freeman 2005, Kilgour et al. 2005, de Freitas 2008, Navarro and Schmidt 2012, McKay 2013).

Scheme	**Strengths**	**Weaknesses**
Hydrologic	• Low resource requirements • Rapid application • Desktop approach • Broad spatial application is simple	• Often results in simplistic, inflexible, or low resolution outputs • Low ecological relevance • Not site-specific • Flow dynamism is seldom considered • Likely inappropriate for highly controversial decisions
Hydraulic	• Readily available tools and support • Rapid application	• Low ecological relevance • Proxy for habitat • Few recent developments
Habitat	• Repeatable • Predictive • Demonstrated legal precedent • Capacity to examine multiple focal taxa	• Habitat is not necessarily the endpoint of interest (populations are) • Focus on specific taxa rather than ecosystem health • (Often) Limited consideration of flow regime beyond flow magnitude • Significant uncertainty can be associated with suitability indices
Holistic	• Flexible and robust • Broad ecological basis and focus on the whole ecosystem • Multi-disciplinary input • Incorporates socio-economic endpoints • Scalable to data rich and data poor environments • Addresses multiple flow regime components	• (Often) Resource and time intensive • Reliant on expert judgment • Challenges in reconciling a vision for the river and conflicting judgments • High ecological data or knowledge requirements
Optimization	• Objective development of flow recommendations based on specification of objectives and constraints • Familiar to classical dam operation • Can be used in conjunction with holistic methods	• Numerical expertise required • Developing holistic, quantitative objectives (and a multi-objective combination algorithm) is challenging • “Optimality” may not exist due to incomplete specification of objectives
Regionalization	• Generates flow prescriptions for many rivers and streams in a region which accelerates implementation. • Holistic view of multiple components of the socio-ecological system • Broad spatial application to sites beyond those studied • Multi-disciplinary input • Emphasizes hypothesis-driven, adaptive management	• Regional development may be time and resource intensive • Requires significant expertise to facilitate the process (hydrologic foundation, classification, flow alteration, flow-ecology relationships) • Better suited to tributaries than to river mainstems. • For any individual site, it’s not as robust as site-specific assessment.

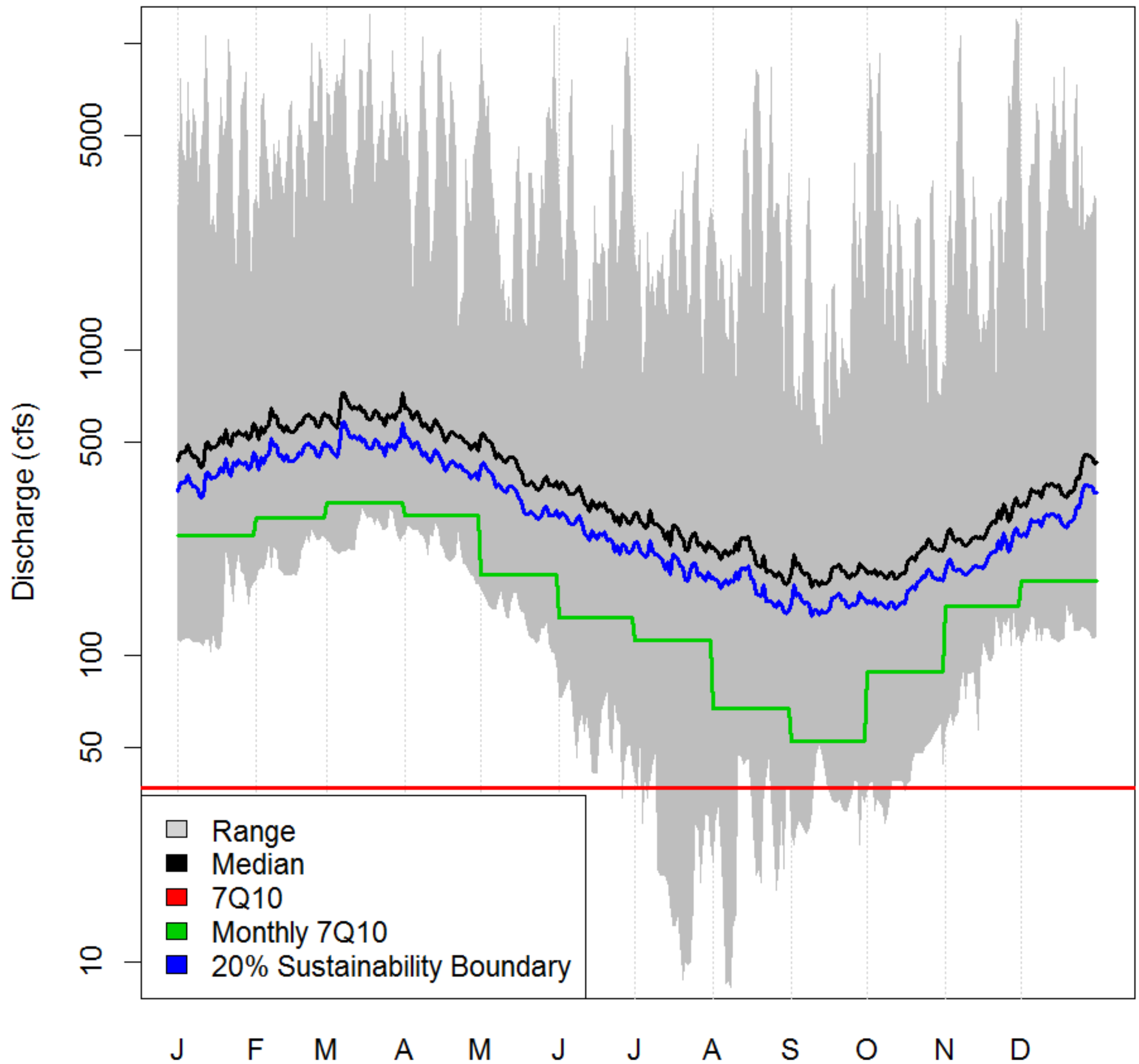

Figure 3.1. An envelope plot showing the Middle Oconee River hydrograph from 1938-1997. The range interval represents the minimum and maximum observed discharge for each day in the period of record. Select flow thresholds from hydrologic methods have been shown for relative comparison.

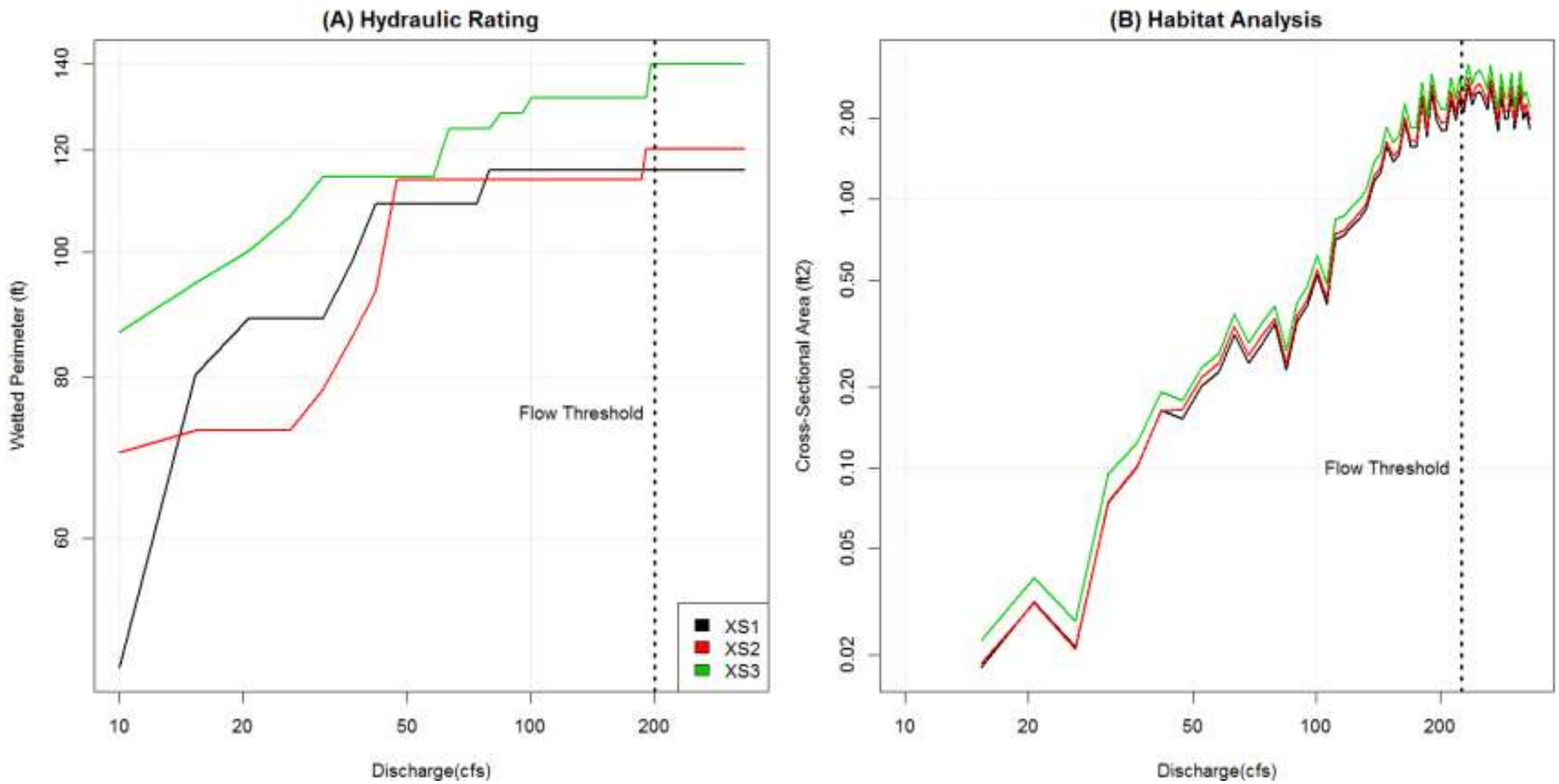

Figure 3.2. (A) Hydraulic rating based on the wetted perimeter method (Gippel and Stewardson 1998). (B) Frequency-weighted cross-sectional area as a surrogate for habitat.

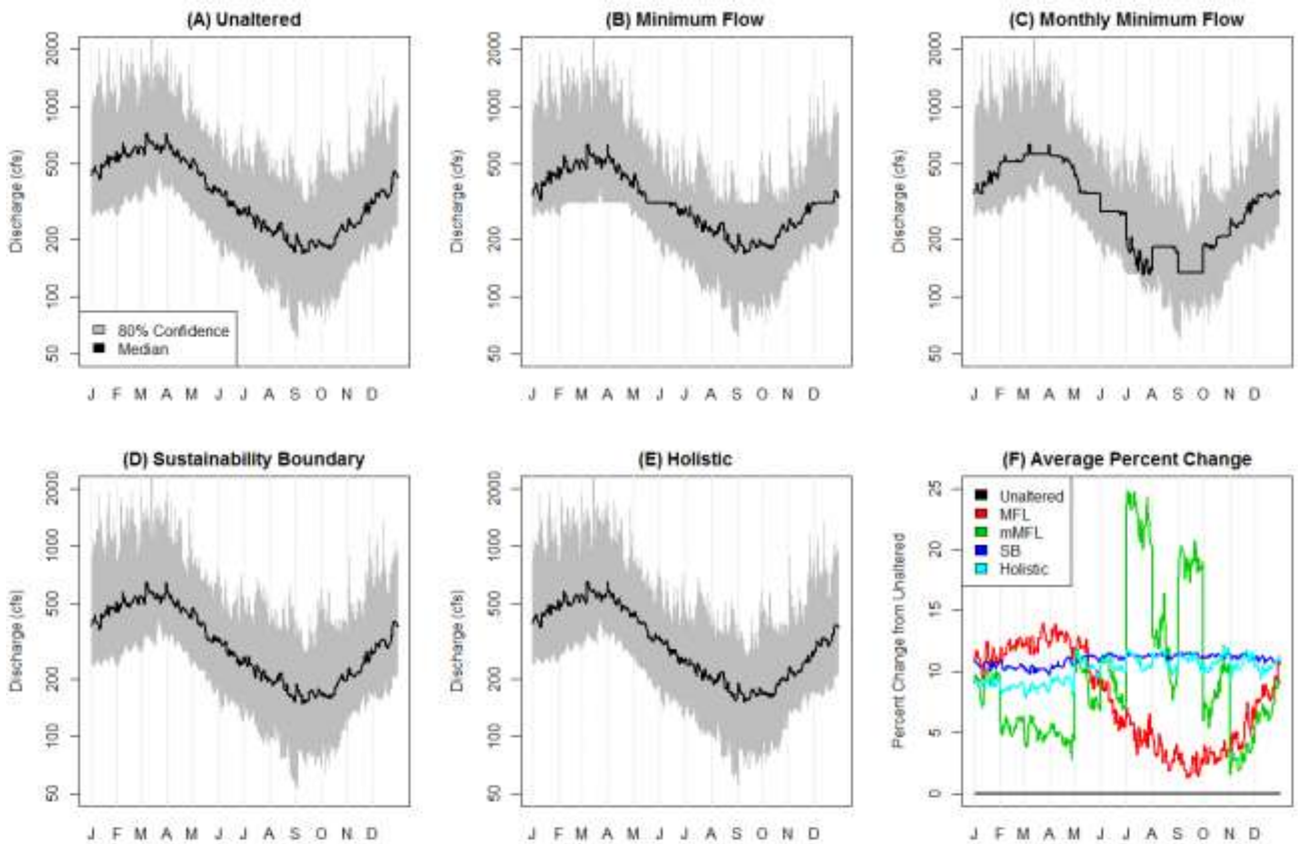

Figure 3.3. Hydrographic effects of alternative environmental flow schemes. (A) Envelope plot showing the median and 80% confidence intervals (10th and 90th percentiles) for the unaltered hydrograph from 1938-1997. (B-E) Envelope plots for environmental flow alternatives with approximately 30 MGD average annual withdrawal rate. Flow thresholds for each scenario are: MFL = 310 cfs, mMFL = $Q_{m,min} + 0.08 * (Q_{m,max} - Q_{m,min})$, SB = 11.5%, and holistic recommendations from Table 2. (F) Percent change in discharge relative to the unaltered flow regime.

CHAPTER 4

ADDING COMPLEXITY TO HYDROLOGIC ENVIRONMENTAL FLOW METHODOLOGIES BY EXPLICITLY INCORPORATING TRADE-OFFS[2]

[2] S. Kyle McKay. To be submitted to *Journal of the American Water Resources Association*.

Abstract

Freshwater management requires balancing and trading-off multiple objectives, many of which may be competing. Ecological needs for freshwater are often described in terms of environmental flow recommendations (e.g., minimum flows), and there are many techniques for developing these recommendations, which range from hydrologic rules to complex analyses supported by large teams of subject matter experts. Although hydrologic rules are well-acknowledged as overly simplified, these techniques remain the state-of-the-practice in many locations. This paper examines the hydrologic implications of these simple environmental flow methodologies. In particular, I seek to add complexity to the application of these techniques by studying the emergent properties of hydrologic environmental flow methodologies. Two hydrologic rules are applied: minimum flow criteria and sustainability boundaries. Objectives and metrics associated with withdrawal rate and similarity to natural flow regimes are used to trade-off economic and environmental needs, respectively. These trade-offs are examined for multiple environmental flow thresholds as well as varying value judgments. A case study of hypothetical water withdrawals on the Middle Oconee River near Athens, Georgia, U.S.A. is applied to demonstrate these techniques.

Introduction

As declared by more than 750 scientists from 50 countries in the Brisbane Declaration (2007), "Environmental flows describe the quantity, timing, and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems." At the crux of this definition is the balance of water provision between ecological and human needs, which can be synonymous (Baron et al. 2002, Richter 2010, Bryan et al. 2013). However, river flow regimes exhibit numerous sources of variability relative to the magnitude, frequency, timing, duration, and rate-of-change of river discharge or stage (Poff et al. 1997), which can result in numerous effects on ecological processes (Bunn and Arthington 2002, Arthington 2012). A primary challenge before river managers is to equitably balance and trade-off alternative needs for freshwater in light of hydrologic variability (Arthington et al. 2006, Grantham et al. 2013, Auerbach et al. 2012, Mubako et al. 2013).

Environmental flow methodologies attempt to facilitate trade-offs by structuring water management decision-making. Given the diversity of potential ecological endpoints (e.g., habitat for macroinvertebrates, population demographics of endangered fishes, phosphorous uptake) and perspectives on river management (e.g., municipal water manager, fisheries ecologist, river engineer), a large number and variety of methods exist to identify environmental flow requirements (Arthington 2012). These techniques are often placed into four categories (Jowett 1997, Tharme 2003, Acreman and Dunbar 2004, Arthington et al. 2006): hydrologic methods, hydraulic rating, habitat analysis, and holistic methodologies. However, recent environmental flow analyses are also utilizing optimization methods (e.g., Suen and Eheart 2006, Yin et al. 2012, Bryan et al. 2013) and regionalization approaches (e.g., Poff et al. 2010, Sanderson et al. 2011, Snelder et al. 2011, Kendy et al. 2012) as tools for identifying environmental flow recommendations (McKay 2013).

While many methods exist, hydrologic methods relying on straightforward operational rules remain easier to implement and are commonly applied (Arthington et al. 2006, Poff 2009, Richter 2010). In particular, minimum flow criteria are often still used as default methods for water regulation decisions across large spatial scales (Shiau and Wu 2004, Richter 2010, Snelder et al. 2011, Ouyang 2012). The deficiencies in minimum flow criteria are well-characterized and include issues such as lack of consideration of an entire river's flow regime (Poff et al. 1997), an ecologically inappropriate "one size fits all" approach (Poff 2009), "flat-lining" of hydrographs (Richter et al. 2011), and selection of ecologically arbitrary hydrologic thresholds (Arthington et al. 2006). Other hydrologic rules have been proposed such as "high flow skimming" where water withdrawals occur only during high flows (Richter and Thomas 2007), "sustainability boundaries" where withdrawals occur on the basis of a percentage of unmodified river discharge (Postel and Richter 2003, Richter 2010, Richter et al. 2011), and management based on environmental flow components summarizing a river's flow regime (Postel and Richter 2003, Matthews and Richter 2007).

Although other environmental flow methods have been encouraged over hydrologic rules (Arthington et al. 2006), the broad application of hydrologic methods justifies continued investigation (Richter 2010, Snelder et al. 2011, Ouyang 2012). The objective of this paper is to examine the hydrologic implications of simple operational rules for defining environmental flows. In particular, I examine trade-offs between water withdrawal and river levels over a range of minimum flows and sustainability boundaries. This analysis is not meant to imply that these environmental flow methodologies are preferable, but instead to add complexity and objectivity to the most common methods of environmental flow analysis that potentially provide for improved management decision making.

Methods

Study Site

The Middle Oconee River is a sixth-order tributary of the Altamaha River in northeast Georgia, U.S.A. This region was historically dominated by row crop agriculture, but suburban, urban, and pasture land uses are currently predominate (Grubaugh and Wallace 1994, Fisher et al. 2000). The U.S. Geological Survey operates a streamflow monitoring station near Athens, Georgia (398 mi^2 watershed, Gage No. 02217500). Continuous, daily discharge records have been collected from 1938 to present. Over this period of record, mean discharge is 496 cubic feet per second (cfs) and median discharge is 329 cfs. However, significant hydrologic variability exists at this site with observed daily minimum and maximum flows of 3.5 and 13,300 cfs, respectively.

As the population of the region has grown, increasing pressure has been placed on local rivers for municipal water supply (Campana et al. 2012). In 2002, a four-county authority constructed Bear Creek Reservoir to meet regional municipal water needs. This reservoir is an off-channel, pump-storage reservoir designed to extract and store Middle Oconee River water. Although off-channel reservoirs can reduce direct effects on fluvial habitat and water quality, intakes can still provide a substantive change in ecological communities due to changes in the flow regime (Freeman and Marcinek 2006). The reservoir's off-channel location provides an opportunity to examine alternative environmental flow regimes in isolation from the confounding effects of connectivity. The reservoir is permitted to withdraw a maximum of 60 million gallons per day (MGD; Georgia EPD Permit Number 078-0304-05). State environmental flow regulations establish a monthly minimum flow downstream of the intake corresponding to the 7-day low flow with a 10-year recurrence interval (i.e., the "7Q10") for each month (GA DNR 2001).

Hydrograph Modification

The long period of pre-reservoir discharge records provided an opportunity to examine the potential effects of alternative environmental flow schemes and associated withdrawal patterns. For the purpose of this analysis, only data from 1938-1997 were applied. Although the reservoir was completed in 2002, 1997 was used as a temporal break point for this analysis because these data were available to regulators prior to reservoir operation and this time period is not influenced by the withdrawal.

The 60 year discharge record was modified based on three simple environmental flow regimes: (1) an unconstrained minimum flow, (2) a constrained minimum flow, and (3) an unconstrained sustainability boundary approach. Unconstrained minimum flows (UMF) refer to the withdrawal of water at any time when the river is sufficiently high to meet minimum flow criteria (Equation 1). Constrained minimum flows (CMF) refer to withdrawal of water based on the environmental flow requirement and a common operational constraint (Richter and Thomas 2007), a preference for less turbid water (Equation 2). The water authority generally prefers to only withdraw water when suspended sediment is less than 50 parts per million (ppm) due to increased treatment cost (S.K. McKay, personal observation). Based on two alternative models, a maximum river discharge of 500 cfs is used as a ceiling for suspended sediment constraints (See Appendix). The final environmental flow regime is based on an unconstrained "sustainability boundary approach" (SB), which represents a percentage of river discharge that may be withdrawn (Equation 3, Richter 2010, Richter et al. 2011). Importantly, constrained operations associated with storage availability (i.e., reservoir volume) were not included in this analysis in order to focus on hydrologic effects rather than site-specific constraints.

UMF
$$Q_w = \begin{cases} Q_{w,\max} & Q_r > Q_{w,\max} + Q_{r,\min} \\ Q_r - Q_{r,\min} & Q_{w,\max} > Q_r > Q_{r,\min} \\ 0 & Q_{r,\min} > Q_r \end{cases} \tag{1}$$

CMF
$$Q_w = \begin{cases} 0 & Q_r > Q_{r,\max} \\ Q_{w,\max} & Q_{r,\max} > Q_r > Q_{w,\max} + Q_{r,\min} \\ Q_r - Q_{r,\min} & Q_{w,\max} > Q_r > Q_{r,\min} \\ 0 & Q_{r,\min} > Q_r \end{cases} \tag{2}$$

SB
$$Q_w = \begin{cases} Q_{w,\max} & \%_{SB} Q_r > Q_{w,\max} \\ \%_{SB} Q_r & \%_{SB} Q_r < Q_{w,\max} \end{cases} \tag{3}$$

Where Q_w is withdrawal rate, Q_r is the unmodified river discharge, $Q_{r,min}$ is the minimum flow standard, $Q_{w,max}$ is the maximum withdrawal capacity, $Q_{r,max}$ is the maximum acceptable river discharge for constrained operations, and $\%_{SB}$ is the sustainability boundary.

These environmental flow alternatives and accompanying withdrawal patterns were applied to each day in the 60 year period of record. For each alternative, daily, annual, and long-term average withdrawal rates were computed and converted to MGD for comparison with the permitted withdrawal rate of 60 MGD. The following range of environmental flow thresholds was considered:

- UMF: minimum flow varied from 0 to 750 cfs by 5 cfs
- CMF: minimum flow varied from 0 to 750 cfs by 5 cfs
- SB: percentage withdrawn was varied from 0 to 50% of river discharge by 0.33%

Decision Analysis

Alternative environmental flow regimes were compared based on two objectives: (1) maximize withdrawal rates and (2) maximize similarity to the natural flow regime. These objectives only partly represent the economic and environmental endpoints of what could be a more nuanced discussion of environmental flows. However, these objectives are suitable for a first-order analysis comparing the relative merits of alternative environmental flow methods.

Long-term (i.e., 60-year) average withdrawal rate ($Q_{w,avg}$) provides a useful measure of municipally available water yield (Vogel et al. 2007). Withdrawal rate was then normalized using the maximum permitted withdrawal rate ($Q_{w,max}$). The resulting metric ($Q_{w,norm}$) provides a consistent zero to one scale for comparison with the flow regime metric.

The second objective was measured using seven "fundamental daily streamflow statistics (FDSS)" (Archfield et al. 2013) calculated for the simulation period: mean, coefficient of variation, skewness, kurtosis, the auto-regressive lag-one correlation coefficient, amplitude of the seasonal signal, and (7) phase shift of the seasonal signal. Table 4.1 provides a summary of each statistic, its computation, and its relevance to elements of the natural flow regime. Although many other discharge statistics could be applied (Richter et al. 1996, Olden and Poff 2003, Shiau and Wu 2004, Vogel et al. 2007), these were selected based on their utility in parsimoniously summarizing flow regimes (Archfield et al. 2013). Following computation, each metric was normalized using the baseline condition of no withdrawal, where one is identical to the no withdrawal scenario (Equation 4).

$$Q_{i,norm} = 1 - \frac{\left|Q_i - Q_{i,u}\right|}{Q_{i,u}} \tag{4}$$

Where $Q_{i,norm}$ is the normalized flow metric, Q_i is one of the seven flow metrics examined, $Q_{i,u}$ is the unaltered value of the metric, and u denotes the unaltered condition with no withdrawal.

Normalization provides a relative scale for comparison between unaltered and managed conditions (Poff et al. 2010) and multiple metrics with different scales. Metrics were assumed to function as limiting factors given the importance of each metric in characterizing the flow regime (Archfield et al. 2013). As such, a summary metric for flow regime impacts was calculated as the minimum value of the seven FDSSs (Equation 5).

$$Q_{r,combined} = \min\left(Q_{1,norm}, Q_{2,norm}, \ldots, Q_{7,norm}\right) \tag{5}$$

Where $Q_{r,combined}$ is a combined flow metric summarizing hydrologic impact.

The ability of a given environmental flow regime to meet both objectives (i.e., maximize withdrawal rate without compromising the flow regime) was assessed using a combined metric capturing the overall utility of a plan. Utility was calculated as a weighted average using varying value judgments to reflect preferences for water use (Equation 6). Preferences were assessed based on a zero to one proportion of preferences for maximizing withdrawal (w_w) or similarity to the natural flow regime (w_r), where one weight is determined by the other (i.e., $w_w = 1 - w_r$). If a user wished to only maximize withdrawal rate, then $w_w = 1$ and $w_r = 0$. Conversely, if a user wished to only maximize hydrologic similarity to the unaltered hydrograph, then $w_w = 0$ and $w_r = 1$. Rather than imposing a value judgment, a range of weights (w_w = 0.00, 0.01,…, 1.00) was applied to examine sensitivity of decision making to values (Bryan et al. 2013).

$$U = w_w Q_{w,norm} + w_r Q_{r,combined} \quad (6)$$

Where U is the overall utility of an environmental flow scenario.

Results

Alternative environmental flow regimes uniquely altered river hydrographs and withdrawal volumes. Figure 4.1 demonstrates modification of river hydrographs by these methods for the year 1941, which was a modestly dry year (10^{th} lowest annual discharge on record). These figures demonstrate common elements of this analysis such as “flat-lining” of hydrographs for minimum flow regulations, large alteration of low flow conditions by the constrained minimum flow conditions, and maintenance of flow variability by sustainability boundary scenarios. For consistent comparison, environmental flow thresholds were selected which produce equivalent long-term withdrawal rates of 15, 35, and 55 MGD. These rates roughly correspond to water demand projections on the reservoir in 2010, 2040, and 2060 under a low usage scenario (Campana et al. 2012). Figure 4.1 also shows flow duration curves for each hydrograph presented, which serve as a useful comparative tool for the overall effects of a management scenario (Vogel et al. 2007, Snelder et al. 2011, Ouyang 2012). Flow duration curves for minimum flow scenarios again demonstrate “flat-lining” effects and large alterations relative to the unaltered condition. Conversely, sustainability boundaries result in flow duration curves similar to unaltered conditions, but shifted to lower flows. For both hydrographs and flow duration curves, the difference in environmental flow regimes becomes more pronounced as withdrawal rates increase, but notably, even for the 15 MGD scenario, the difference between constrained and unconstrained minimum flows is quite large. For instance, a 15 MGD withdrawal rate is possible with a 515 cfs UMF compared to a 270 cfs CMF.

Owing to natural variability in river discharge, the annual withdrawal rate of a single environmental flow scenario varied significantly from year-to-year. Figures 4.2A-4.2C show annual variability in water yield for each environmental flow scenario considered. For even the least stringent flow thresholds, the permitted withdrawal rate of 60 MGD cannot be obtained in all years. In fact, the permitted rate can only be obtained in 45 of 60 years for UMF = 0 cfs and 8 of 60 years for $\%_{SB}$=50%. When considering constrained operations to minimize treatment cost (CMF), the permitted withdrawal rate can never be obtained. Figure 4.2D shows average withdrawal rates across all years to compare the three alternative environmental flow methods.

Figures 4.3A-4.3G show the effects of increased withdrawal rate on the flow regime metrics for each environmental flow scheme. Each flow metric demonstrates a unique response to the environmental flow regimes, which indicates that even simple environment flow rules show unique flow regime responses across a diversity of hydrologic metrics. Figure 4.3H presents the combined flow regime metric for all environmental flow regimes. Without considering the utility analysis below, this could be used to assess the relative trade-offs between different environmental flow schemes (Bryan et al. 2013). For instance, at a given level of withdrawal, one would choose the environmental flow regime producing the least altered flow regime, and for a given level of flow regime alteration, one would choose the maximum withdrawal rate.

Although there is significant intra-annual variability in withdrawal rates, average conditions were used to assess the utility of alternative environmental flow thresholds and to examine the influence of preferences on decision-making. Figure 4.4 presents the utility for a single environmental flow method (e.g., UMF) across a range of flow thresholds (e.g., minimum flow = 200 cfs) for varying value judgments. For UMF and SB, maximum utility is obtained at different flow thresholds depending upon preferences. Examining the UMF method (Figure 4.4A), when withdrawal rate is preferentially valued, the maximum utility is obtained at the lowest minimum flow. Conversely, when river discharge is preferentially valued, the maximum utility is obtained at the highest minimum flow. These results are expected given these extreme scenarios. However, utility converges across value judgments at a UMF threshold of 130 cfs and SB threshold of 29.3%. In contrast to the clear trade-offs for UMFs and SBs, CMFs show diverging flow thresholds across varying value judgments.

Figure 4.5 presents the total utility of the three alternative environmental flow methods (i.e., UMF, CMF, SB) for equivalent withdrawal volumes of 15, 35, and 55 MGD. The sustainability boundary approach out-competes the minimum flow approaches for a majority of value judgments and withdrawal volumes. However, at low withdrawal rates (e.g., 15 MGD, Figure 4.5A) and extremely low valuation of ecological endpoints ($w_r < 0.20$), the minimum flow approaches emerge as preferable. Under very high abstraction scenarios (e.g., 55MGD,

Figure 4.5C), the hydrologic impacts of the environmental flow methods are indistinguishable and similarly detrimental.

Discussion

The objective of this paper is to examine the hydrologic implications of hydrologic rules for defining environmental flows. To do so, a case study of water withdrawals was developed that addresses two objectives: (1) maximize average withdrawal rates and (2) maximize hydrologic similarity to the natural flow regime. While this case study is overly-simplified, it serves as a useful heuristic model to make observations regarding simple hydrologic methods of environmental flows.

Inter-annual variability of river discharge proved to be an important factor affecting average annual withdrawal rates. Even when no environmental flow was considered (UMF = 0 cfs), the permitted withdrawal rate was obtainable in only 75% of years. Without explicit consideration of variability, an inappropriate flow regime could be recommended (Poff 2009). To minimize conflicts between multiple withdrawals or between ecologic and human needs, regulators issuing permits should minimally address this by planning for the worst case scenario (the driest year). This finding also indicates that the Middle Oconee River is already over allocated at this gage. To avoid future conflicts, regulators could either deny future permit applications or constrain the withdrawal permits relative to river discharge (e.g., only wet-period withdrawal, Eheart 2004). These potential conflicts could become more pronounced under long-term drought, which could be exacerbated by a changing climate (Campana et al. 2012, Patterson et al. 2012).

This analysis has also highlighted the importance of considering real-world operational constraints in environmental flow recommendations (Richter and Thomas 2007). For instance, the constraint of a maximum withdrawal due to minimize turbidity led to lower utility in all scenarios presented. Importantly, this restriction in withdrawal also prevented obtaining the permitted withdrawal rate in any year considered. Future efforts should also account for reservoir volume constraints, which were neglected here.

Comparisons between environmental flow methods also showed the importance of stakeholder preferences in recommending a flow regime (Poff et al. 2010, Arthington 2012, Bryan et al. 2013, Pahl-Wostl et al. 2013). Trade-offs were shown both within a single method (e.g., multiple minimum flow thresholds, Figure 4.4) and across methods (e.g., UMFLs v. SBs, Figure 4.5). Although the value-laden process presented was simplistic, more sophisticated methods for multi-criteria decision analysis and structured decision making are known to be useful for environmental decision making (Linkov and Moberg 2011, Conroy and Peterson 2013).

Incorporating value judgments also provides an additional mechanism for identifying flow thresholds based on scientific evidence. More sophisticated methods exist and have been applied extensively to environmental flow decision making such as habitat simulation methods (e.g., the Instream Flow Incremental Method, Bovee and Milhous 1978, Stalnaker et al. 1995), literature-based flow-ecology relationships (Sanderson et al. 2011), demographic modeling techniques (e.g., Peterson et al. 2011), and expert panel judgments (Richter et al. 2006). However, many minimum flow decisions are often made using simplistic thresholds such as the 7-day low flow with 10-year recurrence (7Q10), the 30-day low flow with 2-year recurrence (30Q2), 80% flow exceedence (Ouyang 2012), or 30% of mean annual flow (Tennant 1976). For the Middle Oconee River case study, the 7Q10 = 45 cfs, the 30Q2 = 155 cfs (Carter and Putnam 1978), 80% flow exceedence = 195 cfs, and 30% of mean annual flow = 149 cfs (USGS 2013).

The utility-based method presented here showed that a minimum flow of 130 cfs or a sustainability boundary of 29.33% maximized utility across value judgments. A sustainability boundary of 29.3% is larger than the "presumptive standard" of 20% proposed by Richter et al. (2011), but their standard was scoped to be conservative and precautionary with respect to ecological integrity. Although "no (scientifically credible) rule-of-thumb" can "satisfy environmental flow needs" (Richter 2010), the methods presented here provide a rational framework for identifying a minimum flow threshold or sustainability boundary beyond what can be ecologically arbitrarily hydrologic statistics.

This paper has examined the hydrologic implications of environmental flow methods that apply simplistic hydrologic rules. Many more methods exist for defining environmental flows that often include more ecological variables and nuanced analysis of flow regimes and environmental flow components (Tharme 2003, Acreman and Dunbar 2004, Arthington 2012, McKay 2013). Tools for examining alternative flow regimes are actively being developed (Vogel et al. 2007, Hickey 2012, Payne and Jowett 2013). However, simple hydrologic methods remain frequently utilized in practice (Arthington et al. 2006, Richter et al. 2011, Ouyang 2012). This analysis is not intended to endorse or discourage these methods, but instead to promote thoughtful application of these methods and the full exploration of the hydrologic effects of environmental flow recommendations.

Acknowledgements

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agency.   A prior draft was significantly improved from comments by Alan Covich, Mary Freeman, Rhett Jackson, Bobby McComas, Bruce Pruitt, and John Schramski.

Appendix 4A

Eighty-three observations of total suspended sediment have been made at the USGS streamflow gage on the Middle Oconee River near Athens, Georgia, U.S.A. (Gage Number 02217500). These suspended sediment concentrations (mg/L) were used in conjunction with instantaneous volumetric discharge estimates to develop a suspended sediment rating curve (Figure 4.A1). Following common practice (e.g., Julien 2010), a power function was fit to these data to provide a continuous estimate of suspended sediment with variable discharge (Equation A1). Using the inverse form of Equation A1, a withdrawal threshold of 466 cfs was estimated from a sediment concentration of 50 mg/L. Owing to variability in suspended sediment measurements at low discharges, a second approach to assess a withdrawal threshold was applied. The maximum observed discharge associated with suspended sediment less than 50 mg/L in the 83 observations was 884 cfs. A maximum withdrawal of 500 cfs was assumed to be a conservative threshold in line with typical operations of the water authority (S.K. McKay, personal observation), and thus was applied in the environmental flow analyses of the main text.

$$C = 1.06Q^{0.63} \quad R^2 = 0.42 \tag{A1}$$

Where C is the suspended sediment concentration in mg/L and Q is river discharge in cfs.

Table 4.1. Fundamental daily streamflow statistics (FDSS, Archfield et al. 2013) used in this study. All statistics are computed from daily-averaged discharge for a 60-year period of record (1938-1997) on the Middle Oconee River.

FDSS	Description	Components of Flow Regime[1]	Unregulated Values
1	Mean	M	521 cfs
2	Coefficient of variation	M	1.36
3	Skewness	M	6.60
4	Kurtosis	M	66.3
5	Auto-regressive lag-one (AR(1)) correlation coefficient of the daily discharge time series[2]	D, R	0.784
6	Seasonal amplitude Daily discharge was expressed as the addition of the sine and cosine functions as: $q_t = a\sin(2\pi y) + b\cos(2\pi y)$ Where q_t is discharge at day t, a and b are model coefficients, and y is the decimal year (1 day = 1 /365). Coefficients a and b were fit using linear regression. Seasonal amplitude was then computed as: $A = \sqrt{a^2 + b^2}$	T	303 cfs
7	Seasonal phase shift Coefficients a and b were computed as in the description of seasonal amplitude. Seasonal phase shift was then computed as: $\varphi = \tan^{-1}\left(\frac{a}{b}\right)$	T	-1.10

[1]As categorized and described by Archfield et al. (2013): M=magnitude, F=Frequency, D=Duration, T=Timing, and R=Rate-of-change.

[2]Archfield et al. (2013) deseasonalize and standardize data for the purpose of between site comparison. However, this was not conducted due to the single site application.

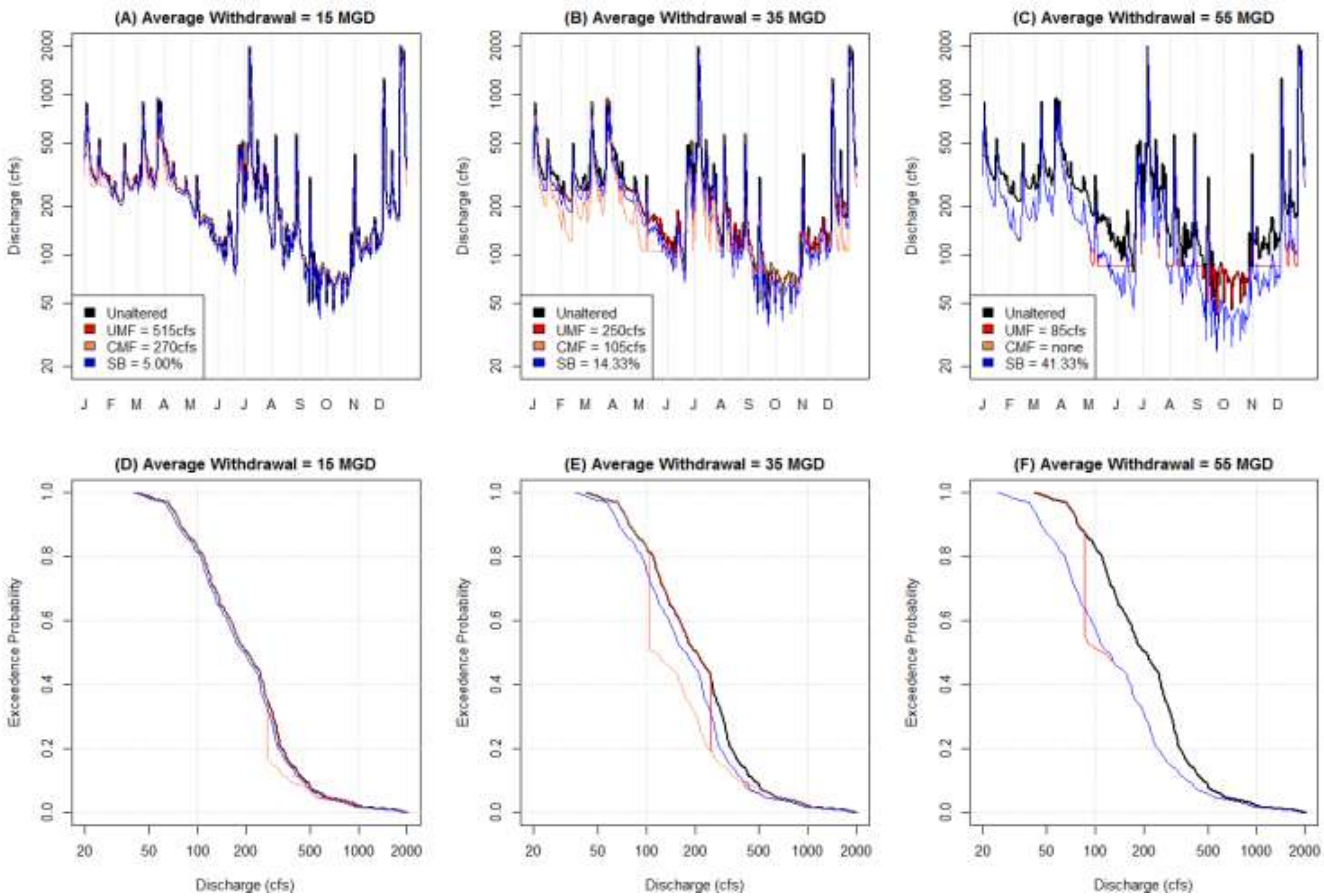

Figure 4.1. Effects of alternative environmental flow decisions on river hydrographs for the year 1941. These scenarios were selected to produce nearly equivalent long-term average withdrawal rates for each environmental flow regime of (A) 15, (B) 35, and (C) 55 MGD. Unaltered and altered flow duration curves long-term average withdrawal rates of (D) 15, (E) 35, and (F) 55 MGD.

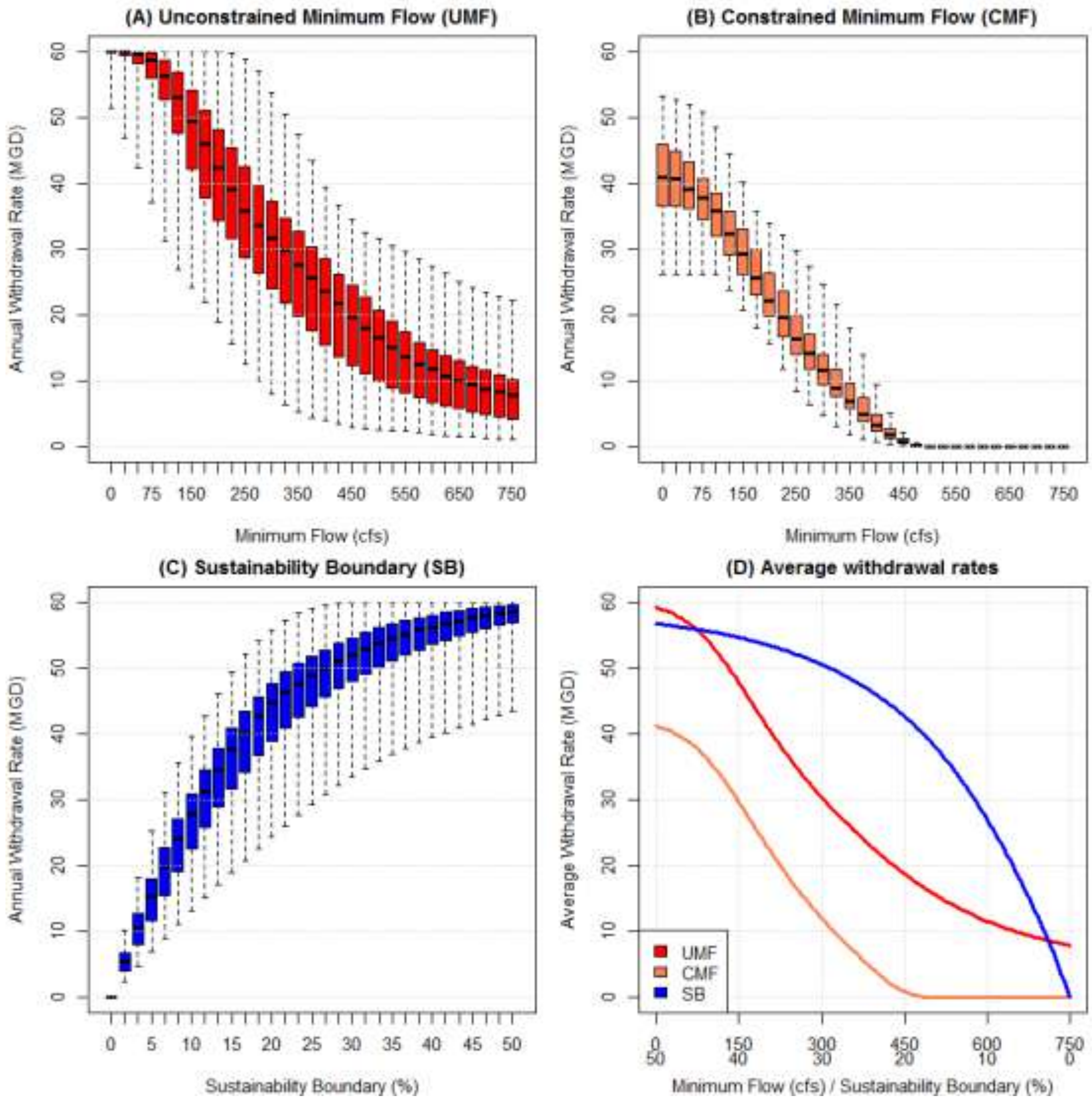

Figure 4.2. Annual variability in water yield for: (A) unconstrained minimum flows, (B) constrained minimum flows, and (C) sustainability boundaries. In each box plot, the thick black line is the median, box extremes are the 25th and 75th percentile, and whiskers are minimum and maximum observed points. Although a finer resolution of flow thresholds was carried through analyses (e.g., minimum flow = 0, 5,…, 750 cfs), every fifth data point is shown here (e.g., 0, 25,…, 750 cfs). Figure D presents average withdrawal rates across all years for direct comparison between alternative environmental flow regimes.

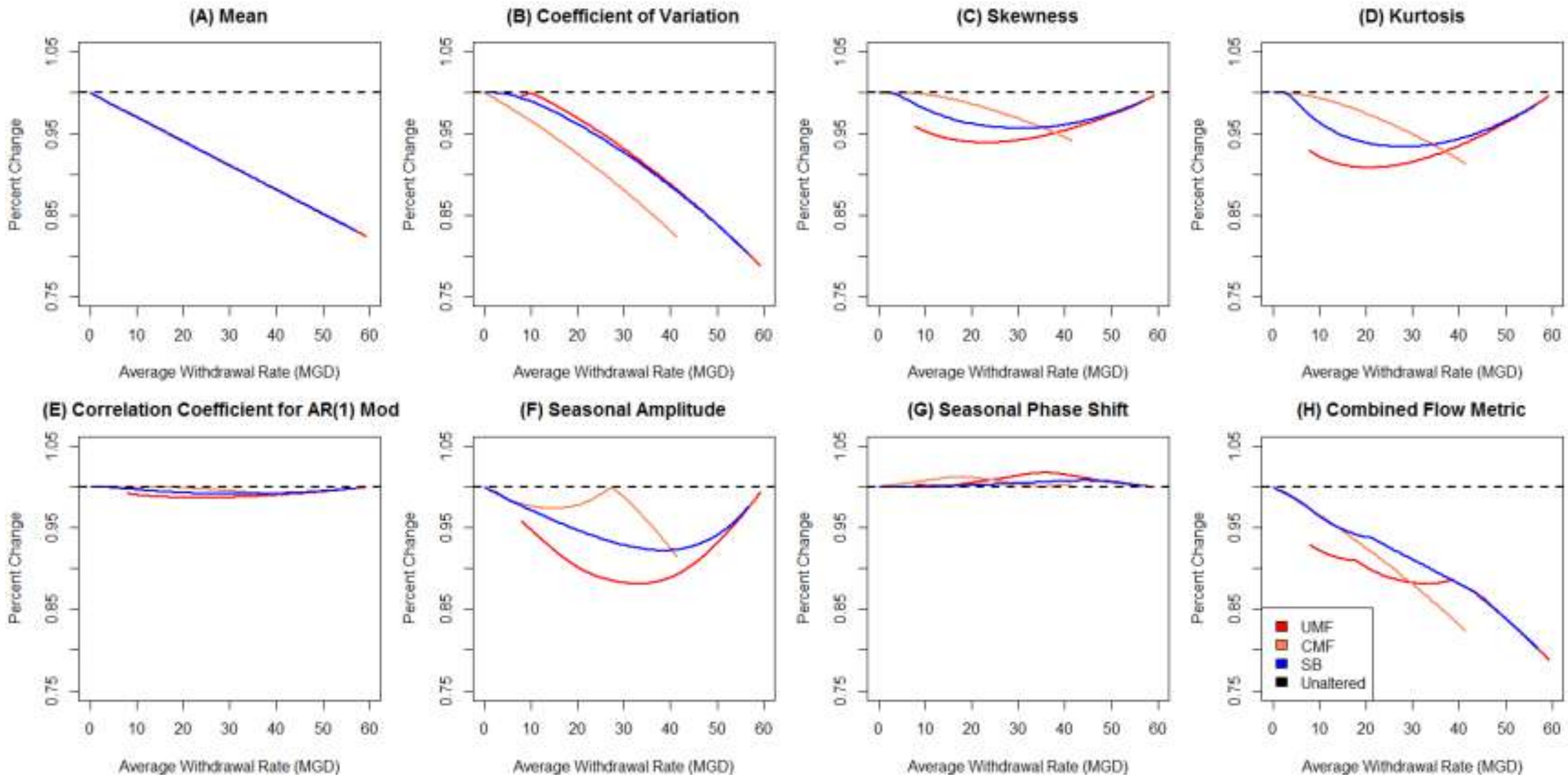

Figure 4.3. Hydrologic effects of environmental flow methodologies. (A-G) Seven fundamental daily stream statistics identified by Archfield et al. (2013) and described in Table 1. (H) A combined flow metric accounting for limiting hydrologic factors.

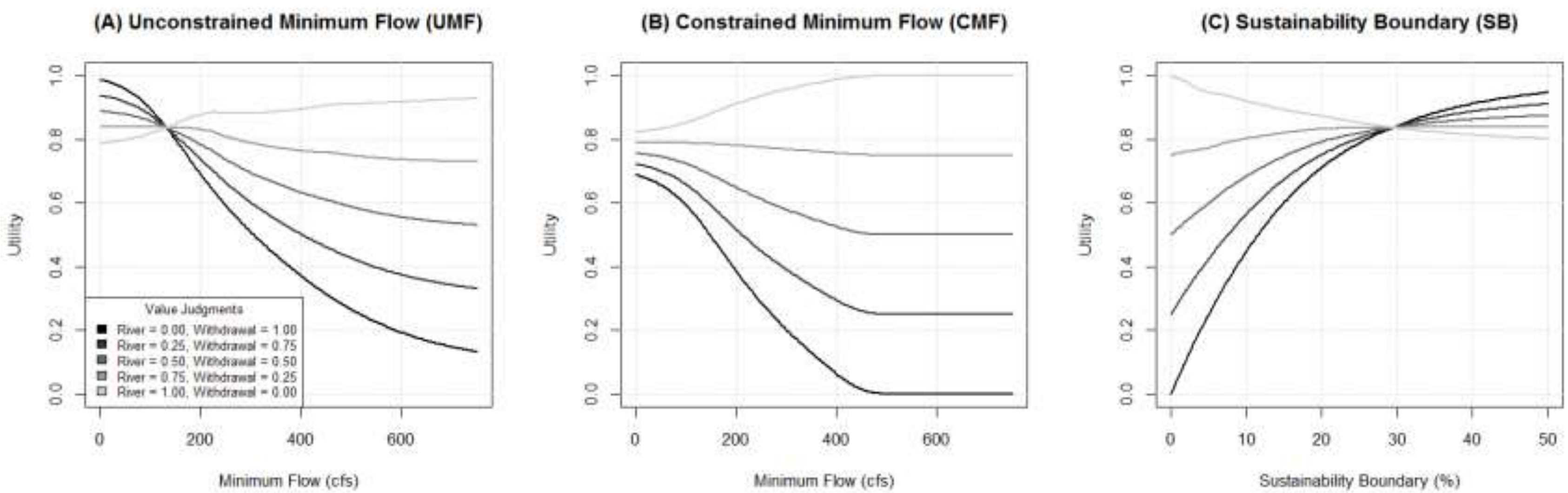

Figure 4.4. Comparing flow thresholds within an environmental flow method using total utility and varying value judgment.

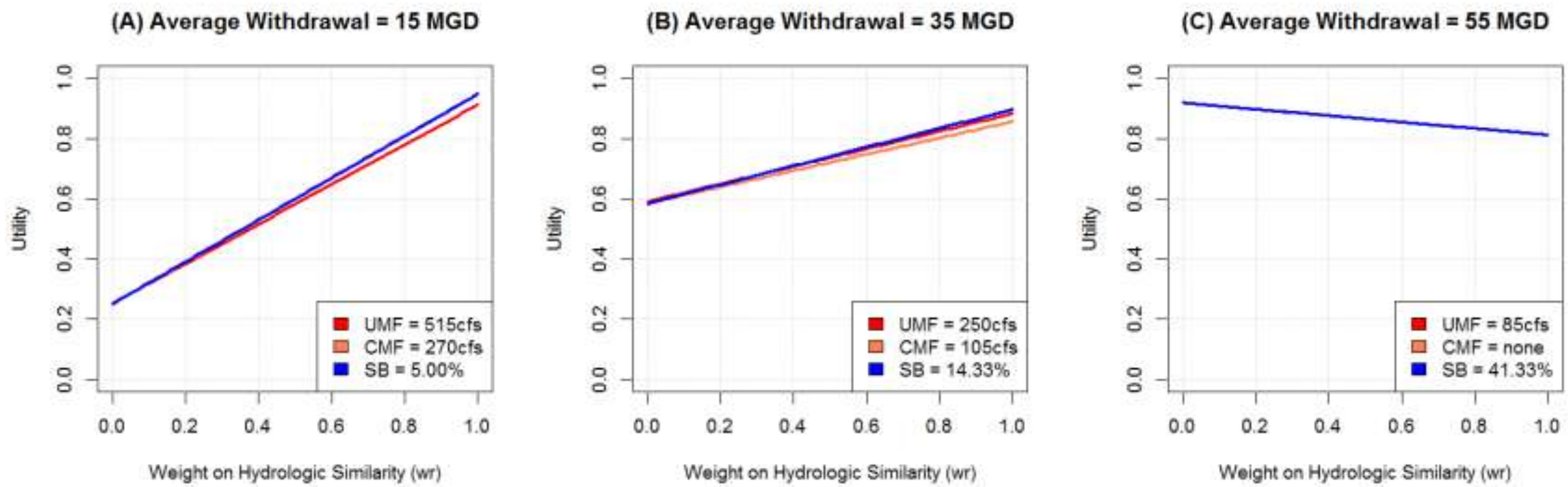

Figure 4.5. Comparing between environmental flow methods for scenarios with nearly equivalent long-term average withdrawal rates of (A) 15, (B) 35, and (C) 55 MGD.

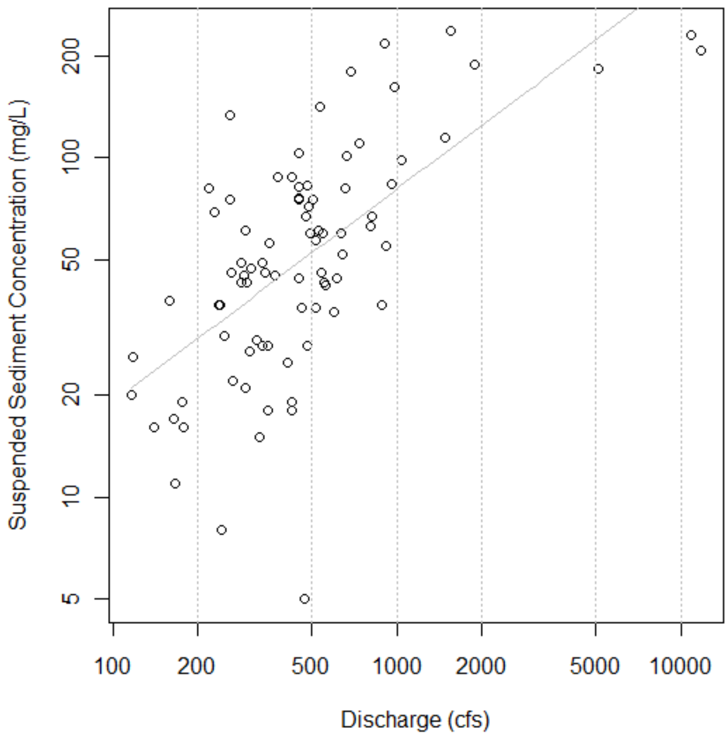

Figure 4.A1. Suspended sediment rating curve for the Middle Oconee River near Athens (n=83).

CHAPTER 5

APPLICATION OF EFFECTIVE DISCHARGE ANALYSIS TO ENVIRONMENTAL FLOW DECISION MAKING[3]

[3] S, Kyle McKay, Mary C. Freeman, and Alan P. Covich. To be submitted to *Environmental Management*.

Abstract

Well-informed river management decisions rely on an explicit statement of objectives, repeatable analyses, and a transparent system for assessing trade-offs. These components may then be applied to compare alternative operational regimes for water resources infrastructure (e.g., diversions, locks and dams). Intra- and inter-annual hydrologic variability further complicate these already complex environmental flow decisions. Effective discharge analysis (from the discipline of geomorphology) has shown promise as a powerful tool for integrating temporal variability and ecological consequences of flow magnitude. Here, we adapt this effectiveness framework to include multiple elements of the natural flow regime (i.e., timing, duration, and rate-of-change) and multiple independent flow variables. We demonstrate this analytical approach using a case study of environmental flow management in the Middle Oconee River near Athens, Georgia, USA. Specifically, we apply an existing model for estimating young-of-year fish recruitment based on flow-dependent metrics to an effective discharge analysis that integrates across hydrologic variability and multiple focal taxa. We then illustrate use of this effective discharge analysis to compare alternative environmental flow regimes.

Introduction

With freshwater biodiversity in sharp decline (Strayer and Dudgeon 2010, Cullen et al. 2014) and over half of the world's large rivers dammed (Nilsson et al. 2005), the need for ecologically effective river management is increasing (Baron et al. 2002, Poff and Mathews 2013). A key component of conserving, managing, and restoring river ecosystems is the environmentally sensitive operation of water resources infrastructure such as diversions, locks, and dams (Freeman and Marcinek 2006, Richter et al. 2006). "Environmental flows describe the quantity, timing, and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems" (Brisbane Declaration 2007). This consensus definition succinctly summarizes the potential for trade-offs between ecological and socio-economic objectives in water management. However, environmental flow decision-making is further complicated by multiple, feasible flow management regimes (Tharme 2003, McKay 2013), numerous ecological endpoints (Richter et al. 2006), various ecologically relevant components of a river's flow regime (Poff et al. 1997, Bunn and Arthington 2002, Matthews and Richter 2007), and hydrologic variability (Poff 2009).

Environmental variability is a well-known driver of ecological processes in rivers (Poff and Ward 1989, Poff et al. 2006, Sabo and Post 2008, Auerbach et al. 2012). Hydrologic variability is defined broadly as both predictable and stochastic changes in river discharge, stage, or other hydrologically mediated variables (McKay 2014). Hydrologic variability can serve as a "filter" for the adaptation of aquatic and riparian species (Lytle and Poff 2004), a driver of community composition (Poff and Allan 1995, Mims and Olden 2012), and a governing mechanism for ecosystem processing rates (Doyle 2005). Thus, for environmental flows to be effective, ecologists suggest that river managers must not only *manage variability*, but also *manage for variability* (Arthington et al. 2009, Poff 2009).

Effective discharge analysis (also referred to as effectiveness analysis) is a well-studied technique for coupling hydrologic variability and river processes. This analytical framework has a long history in geomorphology and river engineering (Woman and Miller 1960, Doyle et al. 2007, Meitzen et al. 2013), and is being increasingly extended to ecological processes (Doyle 2005, Doyle et al. 2005, Wheatcroft et al. 2010, Zarris 2010, Ensign et al. 2013). Effectiveness analysis combines the magnitude of a response to discharge with the probability of that discharge occurring. Multiple indices may then be computed to summarize the response, as described below (Vogel et al. 2003, Doyle and Shields 2008, Klonsky and Vogel 2011, McKay et al. 2014).

Effective discharge analysis has shown promise as a tool for integrating hydrologic variability and discharge-mediated ecological processes (Doyle et al. 2005). However, to date, it has not been applied beyond instantaneous response variables dependent on daily discharge (e.g., sediment and organic matter transport, habitat availability, nutrient uptake; Doyle 2005, Doyle et al. 2005, Wheatcroft et al. 2010, Zarris 2010, McKay et al. 2014). Our objectives are to: (1) adapt effectiveness analysis to incorporate elements of a flow regime beyond magnitude and frequency (i.e., to include timing, duration, and rate-of-change) and (2) demonstrate the application of effectiveness analysis to inform environmental flow decision-making.

Methods

Study Site

This study examines ecological and economic trade-offs associated with alternative environmental flow schemes for the Middle Oconee River near Athens, Georgia, USA. In 2002, the Upper Oconee Basin Water Authority constructed Bear Creek Reservoir to serve as a municipal water supply source for a four-county region. Bear Creek is a tributary to the Middle Oconee River, and the off-channel reservoir is filled by pumping water from the main stem of the Middle Oconee River (Campana et al. 2012). Since 1938, the U.S. Geological Survey has operated a streamflow monitoring gage downstream of the reservoir intake location (Gage Number 02217500). This long period of daily records prior to reservoir construction provides a sufficient data set with which to examine potential withdrawal schemes and accompanying environmental flows relative to a minimally altered reference condition (Stoddard et al. 2006).

Daily discharge records from 1938 to 1997 were used in the following analyses to represent the period of record available to planners and regulators prior to reservoir permitting and construction. Over this 60 year period, daily mean, median, minimum and maximum discharges were 521, 350, 8.2, and 12,600 cubic feet per second (cfs), respectively. The reservoir is permitted to withdraw a maximum of 60 million gallons per day (MGD; Georgia EPD Permit Number 078-0304-05) subject to meeting minimum flow criteria. Currently the reservoir typically withdraws less than 20 MGD (Campana et al. 2012), but the permitted rate represents a substantial portion of river discharge (60 MGD = 92.8 cfs), particularly during the late summer months when flow rates are lowest (September mean = 237 cfs). Although environmental flow trade-offs in the Middle Oconee River are not currently contentious, conflicts over water allocation and withdrawal are most effectively addressed before they occur (Baron et al. 2002).

Alternative Environmental Flows

In this study, we examine trade-offs between municipal water availability and an ecological endpoint, fish recruitment, under alternative river flow patterns. Four alternative flow regimes were examined, which may also be viewed as operational withdrawal schemes or environmental flows. For each method, the unaltered hydrograph was modified for the entire 60-year observational period (i.e., 1938-1997). Water was abstracted at a maximum rate of 60 MGD in accordance with existing pump capacity. Environmental flow thresholds were systematically varied across a wide range of values, as described below. Although previously acknowledged as operational constraints (Vogel et al. 2007, McKay 2014b), neither reservoir volume limitations nor increased water treatment costs due to turbidity of high flows were included in this analysis. The four examined scenarios of withdrawal and environmental flow requirements were:

1. Unaltered: A reference condition without any withdrawal was applied in this analysis as the best attainable ecological condition (Stoddard et al. 2006).
2. Annual minimum flow (MFL): This method assigns a single, year-round flow threshold below which water may not be withdrawn. Although well-acknowledged as an imperfect method for environmental flow provision (Arthington et al. 2006, Freeman and Marcinek 2006, Poff 2009, Richter et al. 2011), minimum flows remain extensively applied in practice (Tharme 2003, Kanno and Vokoun 2010). MFL was varied from 0 to 1,000 cfs by 10 cfs.
3. Monthly minimum flow (mMFL): This method assigns a monthly-varied flow threshold below which water may not be withdrawn. This common adjustment to the MFL approach incorporates elements of flow timing not captured in annual minimum flows (Hughes and Mallory 2008). Current regulations in the state of Georgia recommend monthly minimum flows associated with the 7-day low flow with a 10-year recurrence interval (i.e., the "7Q10") for each month (GA DNR 2001). For this study, mMFL was varied in 101 intervals from the minimum observed monthly-averaged flow to the maximum observed monthly-averaged flow for the 60-year record for each of the 12 months.
4. Sustainability boundaries (SB): As a simple, first-order alternative to minimum flows, Richter (2009) and Richter et al. (2011) offer a percent-of-discharge approach, which they call sustainability boundaries and propose as the "presumptive standard" for

simplistic environmental flow rules. In our study the percent of daily discharge available for abstraction (SB) was varied from 0 to 50% water abstraction by 0.5%.

Ecological Response Modeling

An existing model is applied to examine ecological response to changes in the flow regime. Craven et al. (2010) present a flow-dependent model for predicting young-of-year fish recruitment for multiple species with varying traits. This hierarchical linear model (Equation 1) incorporates flow regime variables for spawning and rearing periods as well as species traits pertaining to spawning strategy (broadcasting eggs v. non-broadcast) and locomotion morphology (cruiser v. non-cruiser). Craven et al.'s model represents the best-supported of multiple alternative models for variation in juvenile fish abundances over multiple years in three eastern U.S. rivers, based on species traits and flow characteristics. To generalize the model, Craven et al. (2010) present all flow metrics for species-specific spawning and rearing periods as values normalized by the long-term mean discharge.

$$\ln(YOY)_{i,j} = \beta_0 + \beta_1 \frac{Q_{sp,i,j}}{Q_{mean}} + \beta_2 B_j + \beta_3 \frac{Q_{sp,i,j}}{Q_{mean}} B_j + \beta_4 \frac{Q_{re,i,j}}{Q_{mean}} + \beta_5 C_j + \beta_6 \frac{Q_{re,i,j}}{Q_{mean}} C_j + \beta_7 D_{ad,i-1,j} \quad (1)$$

Where *YOY* is young of year density (no/ha), $\beta_{0\text{-}7}$ are model coefficients shown in Table 5.1, i is year, j is species, $Q_{sp,i,j}$ is the maximum 10-day average discharge observed during the spawning period in year i for species j, Q_{mean} is the mean discharge for the unaltered period of record (1938-1997), B_j is a binary variable denoting whether or not a species broadcast spawns (1=Yes, 0=No), $Q_{re,i,j}$ is the minimum 10-day standard deviation of discharge observed during the rearing period in year i for species j, C_j is a binary variable denoting whether or not a species has a cruising morphology (1=Yes, 0=No), and $D_{ad,i-1,j}$ is the density of adults and juveniles in the prior year.

More than 28 species of fish have been observed in a reach of the Middle Oconee River near the reservoir intake (R.A. Katz and M.C. Freeman, unpublished data). Five species were selected for this analysis representing a range of life histories and species traits observed locally (Table 5.2; Shenton et al. 2012). The Craven et al. model allowed us to predict juvenile density based on species traits and annual flow data, with the exception of the effect of prior year fish density (for which we lacked data). A sensitivity analysis was conducted to determine the model's dependence on this parameter, and a global value of 6 individuals per hectare was selected for all species (Appendix A). Furthermore, the time-dependent property of this parameter (i.e., sequencing and dependence on the prior year density) was neglected to simplify analyses. However, the objective of this analysis was not to estimate the absolute YOY density, but instead to provide a relative comparison between YOY densities under alternative flow regimes (Shenton et al. 2012).

Effectiveness Analysis

In a landmark paper for fluvial geomorphology (Meitzen et al. 2013), Wolman and Miller (1960) proposed and developed the concepts of dominant and effective river discharges. Dominant discharge is a simplifying theoretical concept that suggests there is a discharge or range of discharges disproportionately important to long-term river channel evolution. Effective discharge combines the rate of sediment transport at a given discharge (i.e., magnitude) and the probability of that discharge (i.e., frequency) to estimate the "effectiveness" of a given discharge over long time scales – a measure of geomorphic work done by flowing water. Effective discharge is calculated by multiplying the probability distribution of river discharge with a sediment rating curve to develop a sediment transport effectiveness curve; the peak of this curve is the "effective" discharge (Wolman and Miller 1960; Figure 5.1A). Doyle and Shields (2008) propose the functional-equivalent discharge as a second metric of discharge effectiveness, which represents the continuous discharge required to produce the long term sediment load (i.e., the area under the effectiveness curve; Figure 5.1B). Effective discharge analyses have been extensively developed and applied to geomorphic and sediment transport processes as evidenced by broad applications (Shields et al. 2003), guidelines for computation (Biedenharn et al. 2000), software (Bledsoe et al. 2007), and review in river morphology texts (Garcia 2008). Owing to its successful application in geomorphology, Doyle et al. (2005) proposed effective discharge analysis as a promising tool for assessing ecological endpoints. Effectiveness analysis has subsequently been applied to a variety of ecological processes including algal growth, macroinvertebrate drift, habitat availability (Doyle et al. 2005), organic matter transport (Doyle et al. 2005, Wheatcroft et al. 2010, McKay et al. 2014), nutrient retention (Doyle 2005), and denitrification (Ensign et al. 2013).

Here, we adapted effectiveness analysis for use with Craven et al.'s (2010) fish recruitment model described above. Flow metrics ($Q_{sp,i,j}$ and $Q_{re,i,j}$) were calculated for each species for each year (60 years). We then calculated a frequency distribution of these flow metrics using a nonparametric kernel density approach with a Gaussian kernel and 512 equally spaced discharge bins bound from 0 to 4,100 cfs for $Q_{sp,i,j}$ and 0 to 30 for $Q_{re,i,j}$. This approach for estimating frequency distributions maintains an empirical basis rather than assuming a theoretical distribution, and has proven more repeatable and objective than techniques applying user-specified bins (Klonsky and Vogel 2011, McKay et al. 2014). Because the model includes two flow metrics, a joint probability distribution was obtained by multiplying the probability of each spawning season discharge with the probability of each rearing season discharge. Craven et al.'s (2010) model was then applied to every combination of spawning and rearing season discharges as the ecological rating curve for each species. A three-dimensional effectiveness curve was computed as the product of the joint probability distribution and the rating curve. Figure 5.2 presents each of these steps graphically for all species under the unaltered flow regime.

Using this approach, effectiveness curves were computed for each species and flow regime. Although other metrics have been applied in effectiveness analysis such as effective, functional-equivalent, and "half-load" discharges (Wolman and Miller 1960, Vogel et al. 2003, Doyle and Shields 2008, Ferro and Porto 2012), we focus on an alternative metric due to its readily interpretable ecological meaning. We compute the volume under the effectiveness curve as the sum of effectiveness across all values of spawning and rearing season discharges (V_{eff}). This metric summarizes the total amount of ecological processing over the entire distribution of flows, for a particular alternative flow regime. In this case, the volume under the effectiveness curve represents a frequency-weighted estimate of total young-of-year fish recruitment, or a measure of young-of-year production. This variable is related to the functional-equivalent discharge, but is not transformed to discharge units via the rating curve (Doyle and Shields 2008).

This analysis resulted in an effectiveness metric (V_{eff}) for each combination of species and flow regime. To increase interpretability, these metrics were normalized from zero to one and combined. First, all flow regimes were normalized relative to the unaltered flow regime (Equation 2). Second, all species were combined by averaging the normalized values, which resulted in a single metric for each flow regime (Equation 3).

$$V_{norm,j,k} = 1 - \frac{\left| V_{eff,j,u} - V_{eff,j,k} \right|}{V_{eff,j,u}} \quad (2)$$

$$V_{norm,k} = \frac{1}{5}\sum_{j} V_{norm,j,k} \quad (3)$$

Where V_{eff} is the area under the effectiveness curve, j is a given species, k is a given flow regime, u is the unaltered flow regime, $V_{norm,j,k}$ is a normalized value of the effectiveness metric for each species and flow regime, and $V_{norm,k}$ is a normalized value representative of all species for a given flow regime.

In order to examine trade-offs with municipal water supply, flow regimes were compared relative to the average annual flow rate (in MGD) over 60 years. As described, $V_{norm,j,k}$ and $V_{norm,k}$ were computed for each species and flow regime combination. Three alternative parameterizations of the Craven et al. (2010) model were used to test the sensitivity of decision making to model uncertainty. Table 5.1 presents these parameters for the sensitivity scenarios as the upper confidence set, expected model, and lower confidence set.

All computations were performed in the R statistical software package (version 2.15.2; R Development Core Team 2012). In accordance with best practices in environmental modeling (Schmolke et al. 2010), code was error checked and annotated to the extent practicable. Code and data are available from the authors upon request.

Results

Effectiveness metrics were calculated for all combinations of species and flow regimes. An example of these computations (Figure 5.2) and associated output metrics (Table 5.3) are presented for the unaltered, baseline flow regime without water withdrawal. Recruitment estimates vary widely across species due to alternative traits and variable spawning and rearing seasons (Table 5.2). Furthermore, within a single species, a large range of recruitment estimates exists due to rating curve uncertainty (Table 5.3).

Four alternative flow regimes were considered (i.e., unaltered, MFL, mMFL, SB), each with multiple thresholds values. Scenarios with equal annual withdrawal rates are used to consistently compare the ecological effects of flow modification. For instance, the hydrologic effects of all four scenarios are shown for an average annual withdrawal rate of approximately 40 MGD for the year 1941 (Figure 5.3). An example taxa (*Lepomis* spp.) is also presented to demonstrate how hydrologic change can alter the joint probability distribution of flow metrics ($Q_{sp,i,j}$ and $Q_{re,i,j}$) over a time series including multiple years (Figure 5.3).

The two decisions metrics, annual withdrawal rate and young-of-year fish production, varied substantially over the range of flow thresholds considered (Figure 5.4). As expected, lower minimum flow and higher sustainability thresholds yield greater withdrawal rates (Figures 5.4A-5.4C). The five focal taxa each shows a unique response to changes in a river's hydrograph (Figures 5.4D-5.4F, Konrad et al. 2011). An unanticipated outcome involved the positive ecological response (i.e., higher similarity to the unaltered hydrograph) of annual and monthly minimum flow alternatives to extremely high withdrawal rates (and corresponding low minimum flow requirements). This result emerges as an artifact of the recruitment model construct. The model associates low levels of variability with high young-of-year densities, and high withdrawal rates coupled with minimum flow criteria reduce variability. This reduction is likely due to "flat-lining" of hydrographs during minimum flows, whereas sustainability boundaries reduce flows while maintaining natural levels of variability (Richter et al. 2011).

The ecological effects of alternative flow regimes and associated thresholds were compared on the basis of two metrics: the normalized effectiveness metric and average annual withdrawal rate (in MGD). Trade-off curves were developed to show a species-averaged view of the effectiveness metrics (i.e., $V_{norm,k}$) relative to withdrawal rates (Figure 5.5). Sustainability boundary approaches consistently outperform minimum flow approaches, particularly at high withdrawal rates. Results are consistent across the three model parameterizations (i.e., lower confidence set, best estimate, and upper confidence set), which lends confidence to the relative ranking of alternative flow regimes.

Discussion

The objectives of this paper have been to: (1) adapt effectiveness analysis to incorporate elements of a flow regime beyond magnitude and frequency; and (2) demonstrate the application of effectiveness analysis to environmental flow decision-making. The effective discharge framework has proven valuable in the field of geomorphology and increasingly is being applied to ecological processes (Doyle et al. 2005). To date, this analytical framework has been limited by its application only to ecological processes with instantaneous responses to discharge (i.e., those correlated with daily discharge such as organic matter transport and habitat availability). Here, we have extended the effectiveness framework to include additional elements of a river's flow regime. Craven et al.'s (2010) model of fish recruitment uses discharge metrics related to timing (i.e., spawning and rearing seasons), duration (i.e., 10-day flow windows), and rate-of-change (i.e., the standard deviation of discharge). We have applied these metrics within the effectiveness framework by calculating each on an annual basis and computing an associated frequency distribution. Moreover, we extend this framework to include multi-variate models with two independent variables ($Q_{sp,i,j}$ and $Q_{re,i,j}$) by using a joint probability approach. While not required to characterize sediment transport processes, multi-variate models are much more common in ecological processes where complex life histories may depend on multiple components of a flow regime.

In traditional sediment transport analyses, effective discharge metrics are commonly used in channel design or assessment of an alternative flow regime's capacity to shape a channel (Shields et al. 2003). Here, we have presented an analysis that applies an effectiveness metric as an integrative response variable rather than a design target. Effectiveness analysis is shown to be a useful framework for coupling ecological processes and hydrologic variability, which can then be applied to assess large scale changes to a river's flow regime (i.e., the crux of environmental flow decision-making).

The effectiveness framework provided a powerful analytical tool for comparing the effects of alternative environmental flow regimes on fish recruitment. Comparisons across species (Figure 5.4D-5.4F) could be used not only to assess sensitivities to flow regimes (Konrad et al. 2011), but also the potential for changes in community composition (Freeman and Marcinek 2006, Kanno and Vokoun 2010) or food-web dynamics (Cross et al. 2011). Comparisons across many scenarios (Figure 5.5) could allow decision-makers to assess trade-offs between ecological costs and economic benefits of alternative withdrawal schemes. How the decision-maker values these two endpoints could affect which decision may be preferable for implementation (Bryan et al. 2013, McKay 2014b). In our example, sustainability boundaries emerged as the preferred alternative for both objectives regardless of values. Interestingly, relative to fish recruitment, monthly minimum flows under-performed annual minimum flows for much of the withdrawal range examined, possibly as an artifact of effects on flow stability.

Habitat-based analyses have been applied broadly in environmental flow decision-making due to their repeatability, transparency, and capacity to inform trade-offs via incremental changes in flow regimes (Bovee and Milhous 1978, Jowett 1997). Although widely applied, these approaches have been criticized due to their inherent use of a few focal taxa (often game fish species), the assumption that habitat is indicative of population processes, the lack of biological processes such as competition and predation, assumptions of "optimal" flows rather than distributions of discharge, and a lack of consideration of flow timing (Orth 1987, Shenton et al. 2012). The framework presented here directly addresses several of these concerns and provides a quantitative set of techniques for explicitly incorporating demographic processes into incremental environmental flow decision-making (Shenton et al. 2012).

This analysis has examined a single ecological endpoint, fish recruitment. Even using one rating curve, ecological responses were highly dependent on the taxa of interest. If these analyses were applied to multiple ecological processes (e.g., nutrient retention, habitat availability, and fish recruitment), the range of responses would likely be even larger (Konrad et al. 2011). The issue of multiple effective discharges or ranges of discharges has been highlighted in geomorphology as well (e.g., Barry et al. 2008, Ferro and Porto 2010, Zarris 2010). We normalized our effectiveness metrics from zero to one to facilitate comparison across species. As expansion to more ecological processes continues, this normalization approach may facilitate comparison and combination of disparate ecological responses.

Owing to uncertainty in the rating curve, the effectiveness metric had a large range of outcomes even under a single flow regime (e.g., the unaltered condition shown in Table 5.3). This result is not unexpected given that ecological rating curves often exhibit significant uncertainty (Kanno and Vokoun 2010). While not the subject of this paper, sensitivity analysis can be a useful tool for bounding uncertainty in effectiveness metrics (McKay et al. 2014). Although effectiveness metrics varied widely for a single flow regime, the relative ranking of flow regimes remained the same across model parameterizations (Figure 5.5), which provides confidence in the environmental flow analyses considered here.

The expanded effectiveness framework presented here opens up additional future applications to any flow-dependent ecological process where a hydrologically mediated variable has sufficient data to develop a frequency distribution. Prior to this analysis, many rating curves could not have been considered because daily discharge is not the independent variable (e.g., Kiernan et al. 2012). In addition to expanded views of the flow regime, we encourage investigators to consider alternative time series variables influenced by hydrologic variability (Arthington et al. 2009, Olden and Naiman 2010, Davies et al. 2013). For instance, stage, velocity (Ensign and Doyle 2006), light (Julian et al. 2011), Froude Number (Statzner et al. 1988), or temperature (Olden and Naiman 2010) time series could be applied analogously with an accompanying ecological rating curve. Table 5.4 provides examples of literature-reported ecological rating curves that could be used in the effectiveness framework.

Conclusions

Accounting for hydrologic variability in river management is challenging (Poff 2009, Auerbach et al. 2012). Effective discharge analysis provides a unique and versatile tool for coupling ecological processes and hydrologic variability. Here, we have both expanded the ability of this tool to address all elements of a river's natural flow regime (Poff et al. 1997) and demonstrated its application in an example of environmental flow decision-making. As management decisions become more complex (e.g., more ecological processes, trade-offs among additional objectives), techniques that simplify outcomes will not only be needed, but will be increasingly helpful. The effectiveness framework may not only help us address these challenges, but also move beyond our focus on *managing variability* to *managing for variability*.

Acknowledgements

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Notation

$\beta_{0\text{-}7}$	Coefficients of the Craven et al. (2010) model shown in Table 5.1
B_j	Binary variable denoting whether or not a species broadcast spawns (1=Yes, 0=No)
cfs	Cubic feet per second
CI	Confidence interval (90% for all uses herein)
C_j	Binary variable denoting whether or not a species has a cruising morphology (1=Yes, 0=No)
$D_{ad,i-1,j}$	Density of adults and juveniles in the prior year.
i	Year
j	Species
k	Flow regime
MFL	Annual minimum flow
MGD	Million gallons per day
mMFL	Monthly minimum flow
SB	Sustainability boundary
Q	Volumetric river discharge

Q_m Monthly average discharge

Q_{mean} Mean discharge

$Q_{re,i,j}$ Minimum 10-day standard deviation of discharge observed during the rearing period in year *i* for species *j*

$Q_{sp,i,j}$ Maximum 10-day average discharge observed during the spawning period in year *i* for species *j*

u Unaltered flow regime

V_{eff} Area under the effectiveness curve

$V_{norm,j,k}$ Normalized value of the area under the effectiveness curve for each species and flow regime

$V_{norm,k}$ Normalized value of effectiveness for all species for a given flow regime.

YOY Young of year density (no/ha)

Appendix 5A: Rating curve sensitivity

Adult and juvenile fish density data do not exist for the species of interest in the Middle Oconee River (Table 5.2). As such, this value must be estimated for the environmental flow modeling presented in the main text. This appendix examines the sensitivity of YOY density estimates to changes in the adult density parameter of the model. Discharge metrics for the unaltered flow regime were computed for each year and species-specific spawning and rearing periods (i.e., $Q_{re,i,j}$ and $Q_{sp,i,j}$). For each species, every combination of spawning and rearing flow metric was computed (i.e., $Q_{re,i,j}$ for 60 years and $Q_{sp,i,j}$ for 60 years). Using these metrics and the associated species traits, adult density was varied from 1 to 10 by 1, and YOY density was calculated. Figure 5.A1 shows the variation in YOY density relative to adult density estimates with the boxplot capturing all combinations of flow indices.

Craven et al. (2010) present their model with confidence intervals for parameter estimates (Table 5.1). We computed 10,000 estimates of YOY density while randomly varying model coefficients based on a normal distribution. In these simulations, we applied the median of the unaltered flow regime metrics, species traits, and a density estimate of 6 individuals per hectare. Figure 5.A2 presents the variability in YOY densities for each species of interest.

Based on these two analyses, we concluded that (relative to hydrologic change and model parameters) the model is relatively insensitive to changes in the adult and juvenile density parameter. In light of this finding and a lack of density data, we apply a consistent adult density of 6 individuals per hectare in the main body of the text. Although not ideal for predicting absolute recruitment values, we deem these estimates to be sufficient for the purposes of relative comparison between flow management schemes (Shenton et al. 2012).

Table 5.1. Parameter estimates for hierarchical linear model predicting young-of-year fish density (Equation 1). All parameters are directly from Craven et al. (2010) and include the estimate, standard error, and 90% confidence intervals.

	β_0	β_1	β_2	β_3	β_4	β_5	β_6	β_7
Estimate	2.182	0.619	-1.063	-0.509	-2.536	1.115	-2.022	0.313
Standard Error	0.608	0.172	0.274	0.219	1.092	0.211	1.008	0.029
Lower CI	0.918	0.277	-1.601	-0.938	-4.722	0.699	-4.002	0.255
Upper CI	3.445	0.961	-0.525	-0.080	-0.350	1.530	-0.042	0.371

Table 5.2. Species traits for taxa examined in this analysis of the Middle Oconee River (following Craven et al. 2010).

Taxon	Broadcast Spawning	Cruising Morphology	Spawning Season: First Month	Spawning Season: Last Month	Rearing Season: First Month	Rearing Season: Last Month
Cyprinella spp.		X	4	7	5	8
Etheostoma inscriptum			4	5	5	8
Lepomis spp.			5	8	6	8
Micropterus spp.		X	4	5	5	8
Notropis hudsonius	X	X	4	7	5	8

Table 5.3. Effectiveness metrics for each species for the unaltered flow regime. Values represent the number of young-of-year individuals per hectare on a frequency-weighted basis. Values are presented with mean model coefficients and the 90% confidence intervals.

Guild	V_{eff} (Lower CI)	V_{eff} (Expected Value)	V_{eff} (Upper CI)
Cyprinella spp.	92	3,499	217,854
Etheostoma inscriptum	49	1,110	39,461
Lepomis spp.	42	777	23,635
Micropterus spp.	89	3,196	182,052
Notropis hudsonius	2	153	77,796

Table 5.4. Examples of ecological rating curves containing non-traditional hydrologic variables and flow-dependent ecological processes that could be adapted to effectiveness analysis.

Reference	Ecological Variable or Process	Time Series Variable
Gido and Propst (2012)	Native and nonnative fish density	Mean spring discharge, mean summer discharge, and days less than a threshold discharge
Hagler (2006)	Probability of occurrence of young-of-year amber darters	Number of days above a discharge threshold during spring
Hall et al. (2002), Doyle (2005), Ensign and Doyle (2006), Tank et al. (2008)	Nutrient uptake and retention	Discharge, velocity, depth, specific discharge (discharge / width), and/or cross-sectional area
Hart et al. (2013)	Periphyton coverage	Velocity
Hester and Doyle (2011)	Standardized growth, development, reproduction, and survival rates of invertebrates and fishes	Temperature
Julian et al. (2011)	Macrophyte coverage	Light
Kiernan et al. (2012)	Proportion of fish species native to the river	Mean spring discharge
Merritt et al. (2009)	Probability of occurrence of riparian plants	All elements of the natural flow regime
Mims and Olden (2012)	Life history composition of fish community	Annual coefficient of variation, high pulse count, flow predictability, and high pulse duration of discharge
Negishi et al. (2012)	Mussel growth rate	Mean daily water temperature
Peterson et al. (2011)	Mussel survival, recruitment, and capture probability	Median, minimum 10-day, and maximum 10-day discharge in spring, summer, and winter
Power et al. (1995), Schuwirth and Reichert (2013)	Food web models	Velocity, depth, light, temperature, width, nutrient concentration
Sakaris and Irwin (2010)	Young-of-year recruitment of flathead catfish	Number of spring discharge pulses
Statzner et al. (1988)	Benthic invertebrates	Velocity, depth, Reynolds number, and Froude number
Strayer (1999)	Mussel density	Depth, velocity, and grain size

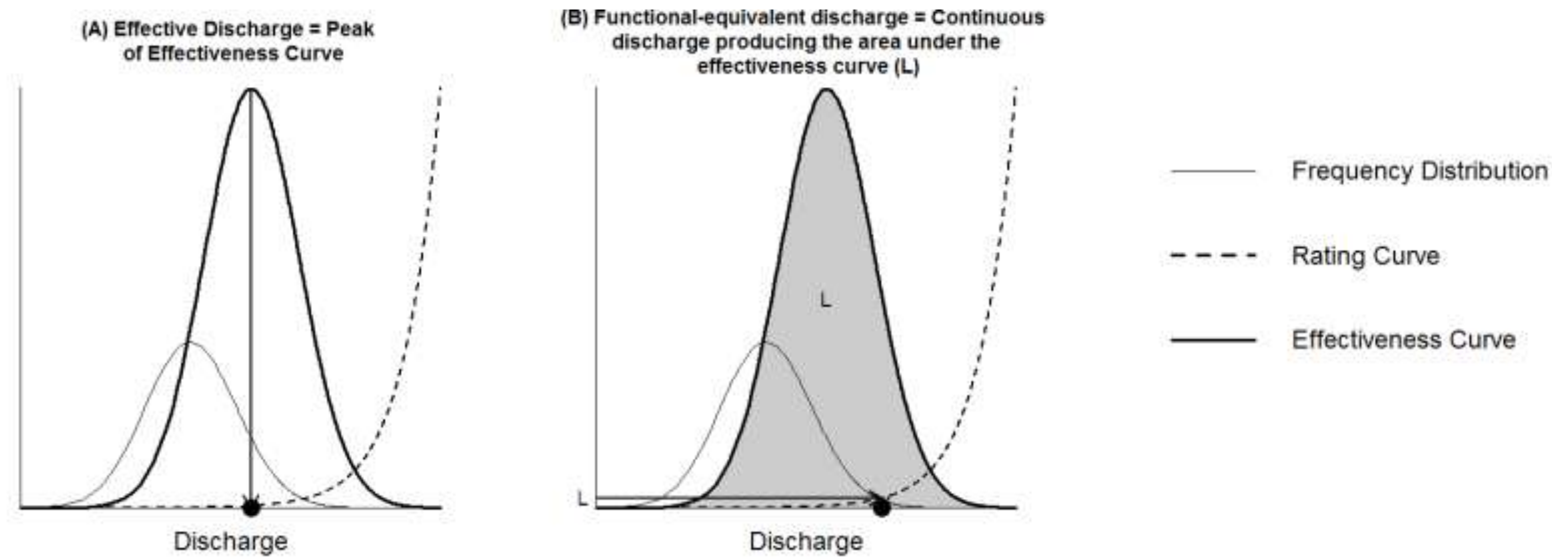

Figure 5.1. Schematic drawing of effectiveness metrics (after Doyle and Shields 2008).

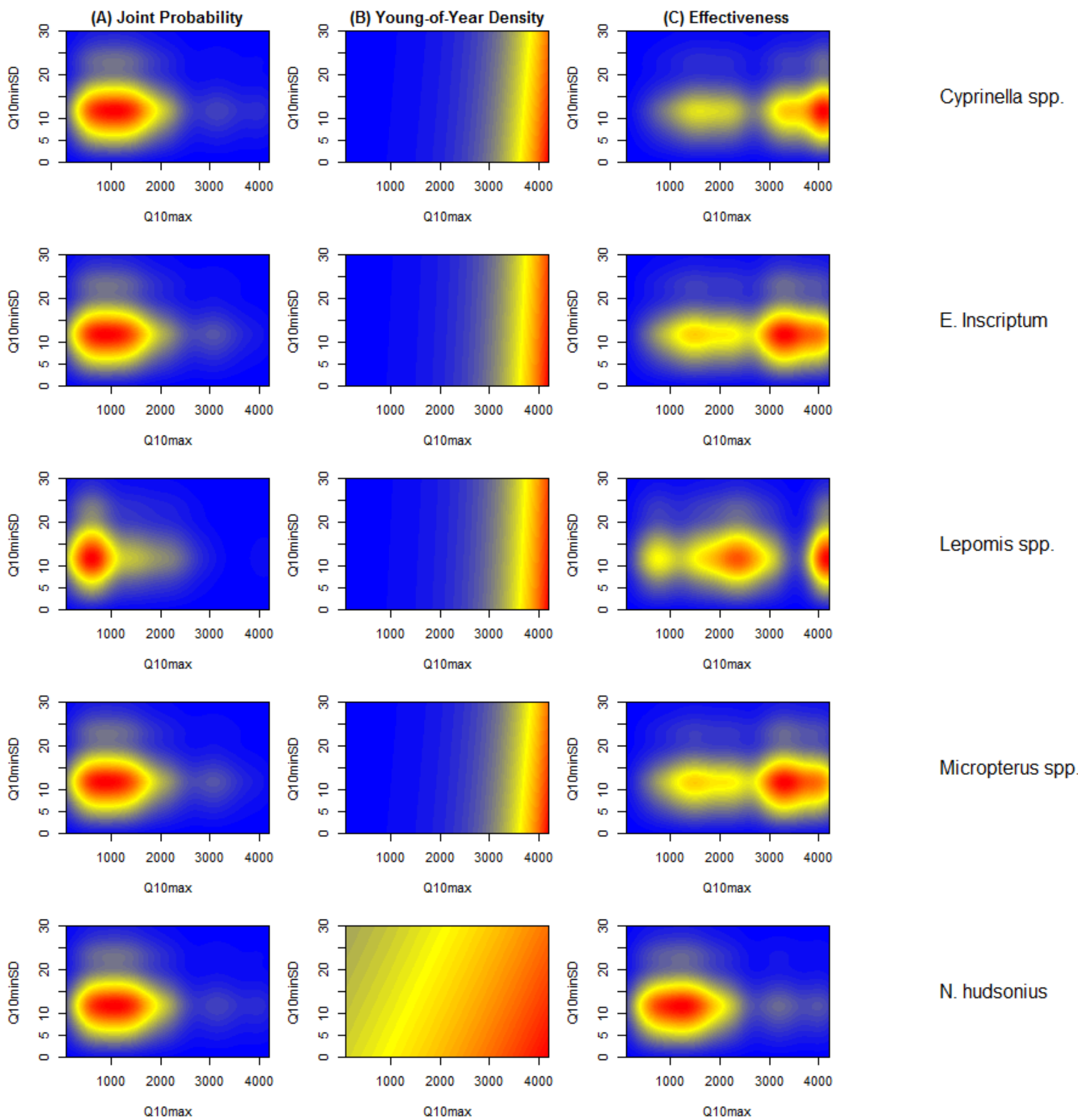

Figure 5.2. Effectiveness analysis for the five focal taxa for the unaltered flow regime estimated using the expected value of the model coefficients. Cool colors indicate low values of probability, density, or effectiveness, while warm colors indicate high values.

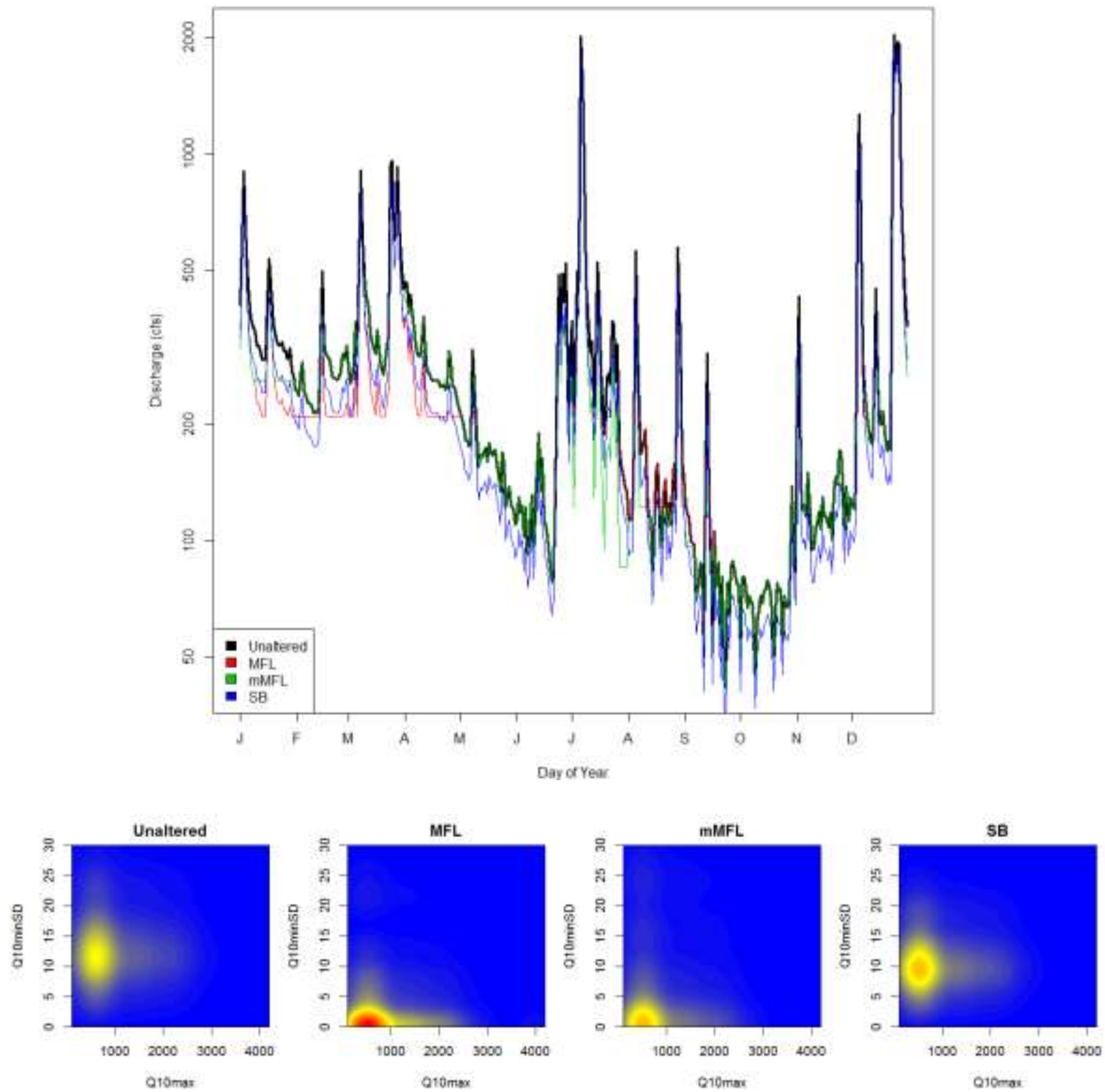

Figure 5.3. Hydrographic effects of water withdrawal for an equal-volume scenario of 40 MGD average annual withdrawal rate. For each scenario the flow thresholds are as follows: MFL = 210 cfs, mMFL = $Q_{m,min}$ + *0.08 ** ($Q_{m,max}$-$Q_{m,min}$), SB = 18%. (top) Flow modification for the year 1941, which was a moderately dry year (10th lowest mean annual discharge on record). (bottom) Effects on the joint probability distribution of flow metrics relative to *Lepomis* spp. spawning and rearing seasons.

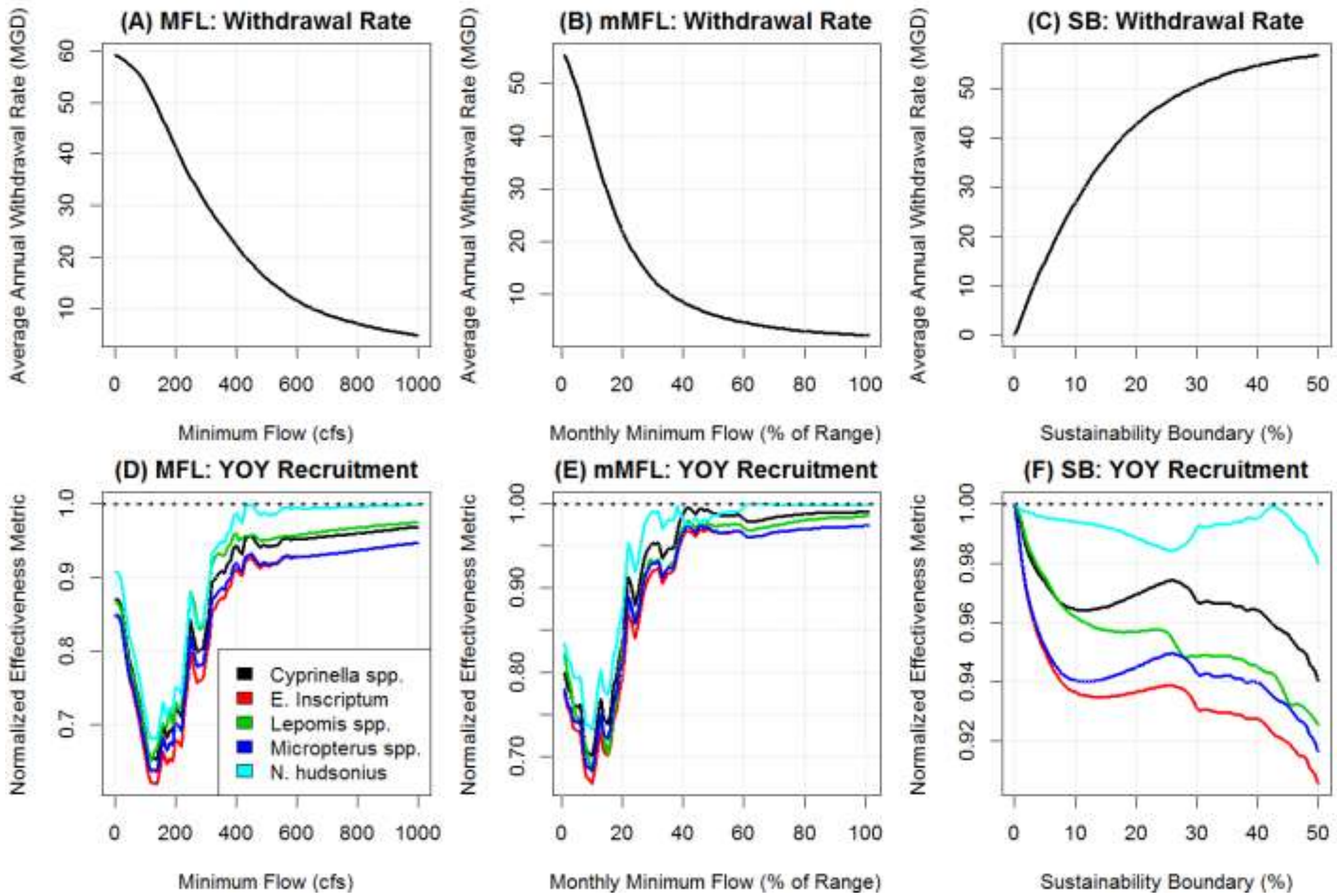

Figure 5.4. Comparison of alternative flow regimes based on the two metrics considered: (A-C) average annual withdrawal rates and (D-F) normalized young-of-year recruitment of five Middle Oconee River taxa. Note that alternative scales of the y-axis are used in 5.4D-5.4F to highlight inter-species differences within a single flow regime. The dashed line in Figures 5.4D-5.4F represents the unaltered flow regime for comparison.

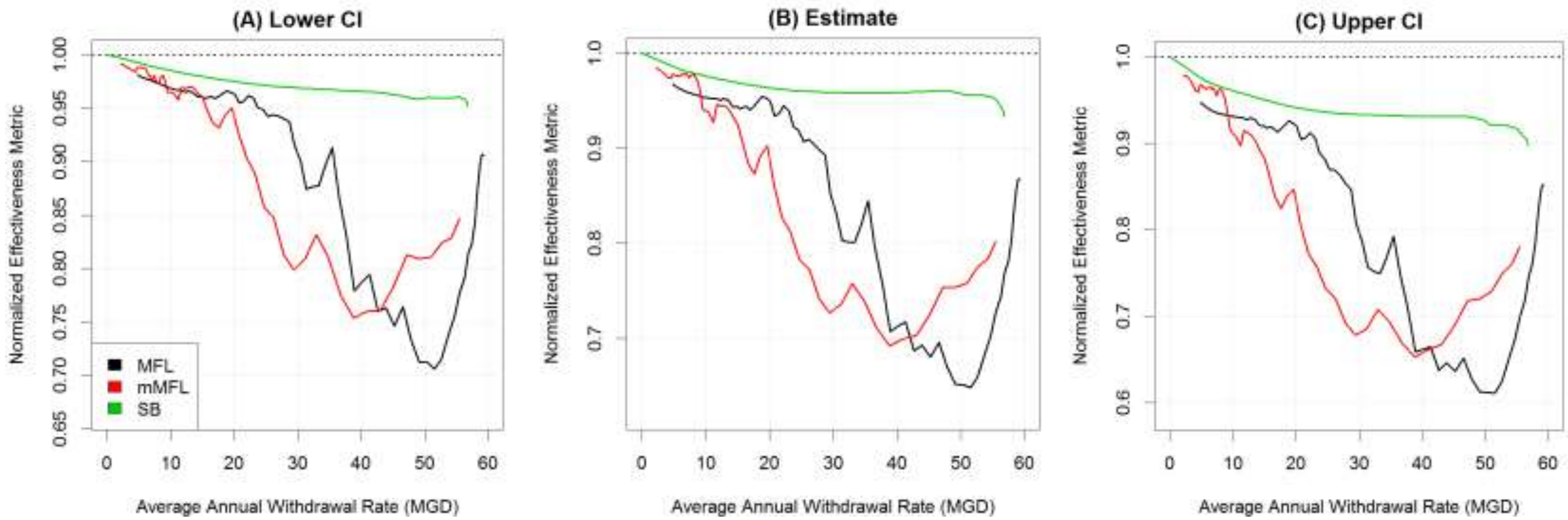

Figure 5.5. Trade-offs curves for alternative flow regimes in the Middle Oconee River. Ecological endpoints are represented by the Craven et al. (2010) model, which has been parameterized for the (A) lower confidence set, (B) best estimate, and (C) upper confidence set. Note that alternative scales of the y-axis are used in figures to highlight differences across parameterizations.

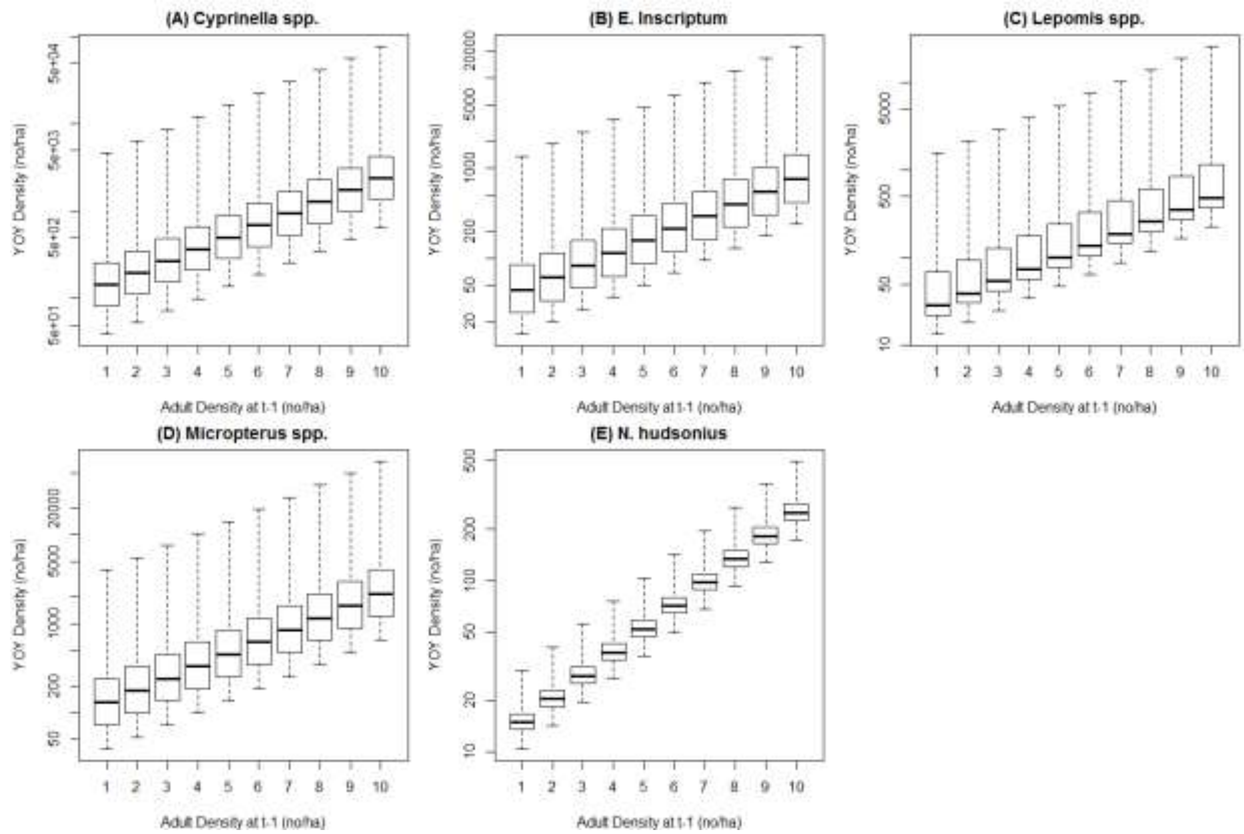

Figure 5.A1. Sensitivity of young-of-year density to estimates of adult/juvenile density and ranges of flow variables observed in the Middle Oconee River. In each box plot, the thick black line is the median, box extremes are the 25th and 75th percentile, and whiskers are minimum and maximum observed points.

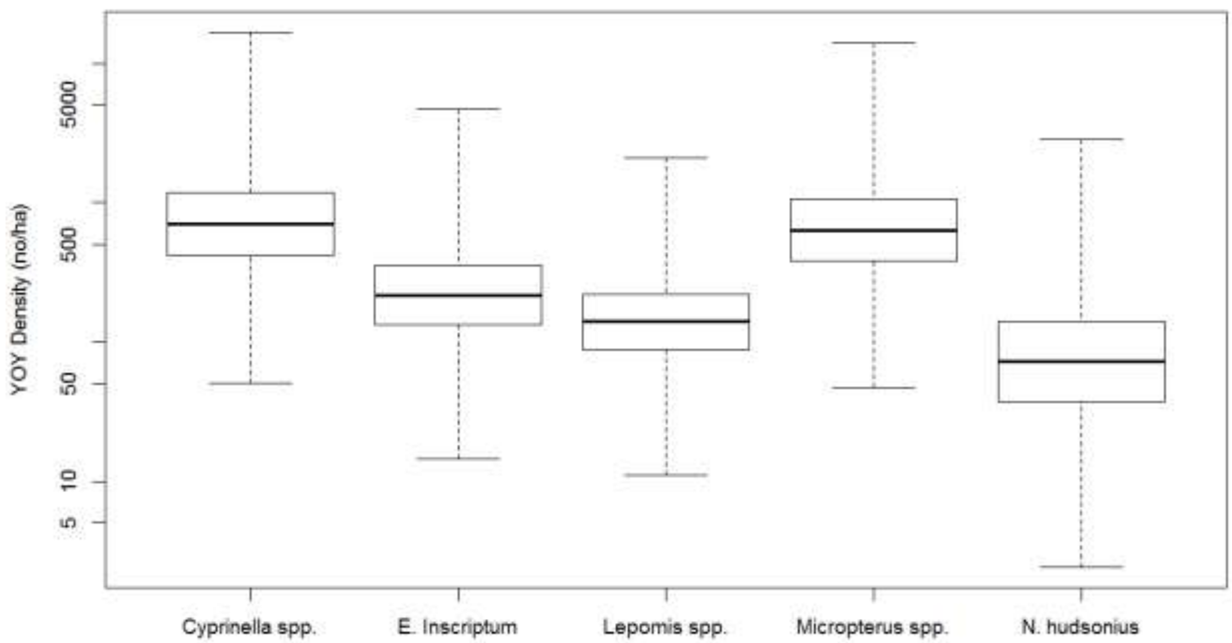

Figure 5.A2. Sensitivity of model estimates to 10,000 random draws from a normal distribution of the model coefficients. In each box plot, the thick black line is the median, box extremes are the 25th and 75th percentile, and whiskers are minimum and maximum observed points.

CHAPTER 6

SENSITIVITY ANALYSIS OF ECOLOGICALLY EFFECTIVE DISCHARGE METRICS[4]

[4] S. Kyle McKay, Alan P. Covich, Mary C. Freeman, and John R. Schramski. Submitted to *Hydrological Processes*, 01/20/2014.

Abstract

The quantity, quality, and timing of river discharge influences numerous physical, chemical, biological, and ecological processes in lotic ecosystems. Accordingly, river scientists have applied numerous, diverse techniques to quantify the link between hydrographic variability and process-based outcomes. One such technique, known as effectiveness analysis, combines the magnitude and frequency of sediment transport events into multiple metrics to estimate the range of discharges that transport the most sediment – a measure of geomorphic work done by flowing water – over long time scales. Here, we investigate the sensitivity of effectiveness metrics to variation in flow frequency distributions and parameter values of constituent rating curves. We demonstrate these analyses with applications to sediment and organic matter transport and apply three metrics of discharge effectiveness: effective, functional-equivalent, and half-load discharges (Q_{eff}, Q_{fed}, and Q_{half}, respectively). Results indicate effectiveness metrics are more sensitive to uncertainties in the frequency distribution than the rating curve, and half-load discharge is more sensitive than effective or functional-equivalent discharges. Recently, effectiveness analysis has been applied as a tool for addressing ecological outcomes of hydrologic variability. Ecological rating curves represent functional relationships between river discharge and ecological processes (e.g., organic matter transport, nutrient uptake, habitat availability), which often exhibit high levels of uncertainty. The sensitivity analyses presented here will not only support traditional, sediment transport applications of effectiveness analysis (e.g., for river restoration design), but will also be critical to ecological applications of this framework (e.g., for environmental flow management).

Introduction

River management is a multivariate, interdisciplinary problem requiring complex trade-offs among competing and often conflicting objectives such as maintaining municipal water supply, avoiding flood damage, generating hydropower, and sustaining ecosystem integrity (Poff et al. 2003, 2010, Vogel et al. 2007). Furthermore, physical, economic, or ecological processes may exhibit thresholds that are critical for implementing management actions (Groffman et al. 2006, Deitch and Kondolf 2012). Confounding these already complex analyses, river flow regimes contain elements of discharge magnitude, frequency, duration, timing, and rate of change that make management decisions increasingly difficult (Poff et al. 1997, Richter et al. 2011).

Owing to this complexity, river scientists and managers have devised techniques to simplify analyses by identifying significant components of a river's flow regime (Tharme 2003). For example, hydrologists, geomorphologists, and river engineers have focused on identifying a river's dominant, or channel-forming, discharge or range of discharges. Because dominant discharge is a simplifying theoretical concept to describe long-term morphologic processes, numerous techniques have been used as surrogates for this quantity, including: bankfull discharge, specified-recurrence interval discharge, and effective discharge (Leopold and Langbein 1962, Langbein and Leopold 1964, Copeland et al. 2000, Shields et al. 2003, Ferro and Porto 2012). Effective discharge is calculated by multiplying the probability distribution of river discharge with a sediment rating curve to develop a sediment transport effectiveness curve; the peak of this curve is the "effective" discharge (Q_{eff}; Wolman and Miller 1960, Figure 6.1). This concept has proven scientifically robust, analytically tractable, and repeatable (Shields et al. 2003) with extensive applications (Watson et al. 1999, Shields et al. 2003, Huang et al. 2010), guidelines for computation (Biedenharn et al. 2000, Emmett and Wolman 2001, Schmidt and Potyondy 2004), software (Orndorff and Whiting 1999, Bledsoe et al. 2007), and extensive review in river morphology texts (Knighton 1998, Garcia 2008, Soar and Thorne 2012).

Two additional metrics have also been developed as surrogates of discharge effectiveness. Doyle (2005) proposed and Doyle and Shields (2008) expanded an alternative measure of discharge effectiveness, functional-equivalent discharge (Q_{fed}, Figure 6.1), which represents the continuous discharge required to produce the long term sediment load. Vogel et al. (2003) and Ferro and Porto (2012) suggest a third measure of discharge effectiveness, half-load discharge, which represents the discharge above and below which 50% of the long-term sediment load is transported (Q_{half}, Figure 6.1). Whereas Q_{eff} identifies the most geomorphically efficient discharge, Q_{fed} and Q_{half} summarize long-term sediment transport by an entire flow regime. Herein, we define "effectiveness analysis" as any of the analytical procedures and associated metrics for combining the magnitude and frequency of events to estimate a range of discharges that may be disproportionately important to sediment transport over long time scales (Wolman and Miller 1960, Doyle 2005, Doyle et al. 2005).

Recently, effective discharge has also been applied to assess ecological response to river discharge. Doyle (2005) applied Q_{eff} and Q_{fed} to examine the role of hydrologic variability in nutrient retention. Doyle et al. (2005) formally developed effective discharge analysis as a generic analytical framework for ecological variables and computed ecologically effective discharge using geographically-dispersed data sets for organic matter transport, algal growth, flood transport of macroinvertebrates, nutrient transport and retention, and physical habitat availability. Wheatcroft et al. (2010) applied effective discharge analysis to assess delivery of particulate organic carbon to the ocean. These applications have both extended and demonstrated the utility of effectiveness analysis for coupling ecological processes and hydrologic variability.

In this paper, our objective is to quantify how uncertainty in estimated frequency distributions and rating curves propagate through to effectiveness metrics (i.e., Q_{eff}, Q_{fed}, and Q_{half}). We focus on transport of sediment and organic matter due to the high applicability of the effectiveness framework and the ecological-relevance of these processes (Doyle et al. 2005). Regardless of application to geomorphic or ecological processes, all frequency distributions and rating curves are inherently uncertain simplifications of observed phenomena. The need for sensitivity analysis is recognized in the effective discharge literature (Goodwin 2004, Doyle 2005, Barry et al. 2008). However, rating curve uncertainty is particularly important for ecological processes because of intricate and varying responses to a single physical variable (e.g., alternative reproductive responses to flow extremes by fishes exhibiting cruising and non-cruising morphologies; Craven et al. 2010). Sensitivity analysis is part of any rigorous model development process (Schmolke et al. 2010), and this step is particularly critical for uncertain models used in environmental decision making (Ascough et al. 2010).

Methods

Study Site

This study was conducted using the Middle Oconee River near Athens, Georgia. The Middle Oconee River is a sixth-order tributary of the Altamaha River with a drainage basin of 1,031 km^2. Characteristic of the southern Appalachian Piedmont, watershed land use was historically altered by late 1800s cotton farming with poor sediment management (Jackson et al. 2005, Trimble 2008) and is currently altered by urban and suburban development (Grubaugh and Wallace 1995) with minor animal agriculture (Fisher et al. 2000). This site was selected due to the long record of streamflow observations (75 years) as well as historical observations of sediment and organic matter transport, which are described in the section on Rating Curves.

Frequency distributions

The U.S. Geological Survey operates a long term streamflow gage on the Middle Oconee River near Athens, Georgia (75 year continuous record from 1938-2012; USGS Gage #02217500). Mean daily discharge at this site is 14.1 m^3/s with daily peaks as low as 0.099 m^3/s and as high as 377 m^3/s. Following Klonsky and Vogel (2011), a frequency distribution of daily discharge was computed using a nonparametric kernel density approach, which maintains an empirical basis rather than an assumed, theoretical frequency distribution (Nash 1994, Vogel et al. 2003, Goodwin 2004, Quader et al. 2008, Zarris 2010). This method provides a more repeatable and objective technique for frequency distribution calculation than traditional methods with user-specified bins (e.g., Searcy 1959, Biedenharn et al. 2000, Crowder and Knapp 2005, Lenzi et al. 2006, Barry et al. 2008). A frequency distribution for the entire record was computed with a Gaussian kernel with the distribution bound between the minimum and maximum observed flows (Klonsky and Vogel 2011). We applied 512 equally spaced discharge bins of 0.76 m^3/s based on prior observation that 256 bins resulted in variable estimates of Q_{eff} (Klonsky and Vogel 2011).

Rating curves

Suspended sediment and organic matter transport rating curves were developed from existing data sources available at the site. Suspended sediment concentration data were obtained from USGS sampling from 1963-1977 (n = 83, http://waterdata.usgs.gov/nwis/qwdata). Dissolved and particulate organic matter transport data were collected approximately weekly from October 1954 through December 1956 (n = 113; Nelson 1957). Data were converted to sediment and organic matter transport rates (Mg/day). Rating curves for suspended sediment transport (SS), dissolved organic matter transport (DOM), and particulate organic matter transport (POM) were computed using least squares regression and a power-law functional form as (Vogel et al., 2003; Wheatcroft et al., 2010):

$$Q_t = \alpha Q^{\beta} \tag{1}$$

Where Q_t is the transported load of suspended sediment in Mg/da, Q is volumetric discharge in m^3/s, and α and β are best-fit coefficients.

Effectiveness Analysis

Effectiveness analyses were undertaken for all three metrics, conceptually described in Figure 6.1. Effective discharge (Q_{eff}) was computed as the maximum of the effectiveness curve given a frequency distribution and rating curve. Functional-equivalent and half-load discharges were computed empirically without any model assumptions as follows (Vogel et al. 2003, Doyle and Shields 2008, Klonsky and Vogel 2011). Discharge data were sorted and transport loads were computed for each day with the process-specific rating curve. The cumulative transport rate was used to identify the half-load for the period of record, and the discharge nearest but not exceeding this load was identified (i.e., the half-load discharge, Q_{half}). The total load for the period of record (L) was averaged over the length of the record (L/n), and the functional-equivalent discharge (Q_{fed}) was calculated with the inverse form of each rating curve.

Sensitivity Analyses

Sensitivity analyses were conducted using three effectiveness metrics and three ecologically-relevant rating curves to examine the relative effects of uncertainty associated with both frequency distributions and rating curves. All computations were performed in the R statistical software package (version 2.15.2; R Development Core Team 2012). In accordance with best practices in environmental modeling (Schmolke et al. 2010), code was error checked and annotated to the extent practicable. Code and data are available from the authors upon request.

The discharge bins identified by the kernel density method for the total discharge data set (i.e., 1938-2012) were applied to determine frequency distributions for each five year interval (i.e., 1938-1942, 1943-1947,…). Effectiveness analyses were undertaken for each frequency distribution using rating curves obtained from least squares regression. Effectiveness metrics for each five-year subset were compared to those developed using the full period of record.

Sensitivity of effectiveness metrics was also assessed relative to changes in rating curve parameters. Quantile regression was used for bounding uncertainty associated with constituent rating curves (Cade and Noon 2003). Regression was conducted for data quantiles from 0.05 to 0.95 by 0.01 using the quantreg package in R (Koenker 2006). These alternative rating curves are intended to bracket uncertainties in effectiveness metrics associated with parameterizing simple power functions. The long-term frequency distribution was used for these analyses.

Results

Baseline effectiveness metrics were computed from the frequency distribution for the 75 year period of record and the best fit rating curve for each of the three transport constituents (Table 6.1). These values ranged as expected, with Q_{half} consistently much greater than Q_{eff} (Vogel et al. 2003) and Q_{fed} slightly larger than Q_{eff} (Doyle and Shields 2008).

Sensitivity analyses resulted in a broad range of outcomes for examining both frequency and rating curve uncertainty. Frequency distributions for five-year intervals showed significant variability across the period of record with peak frequency ranging from 2.3 to 12.6 m^3/s (Figure

6.2). Linear least squares rating curves (Figure 6.3) showed high predictive capacity for suspended sediment (SS, R^2 = 0.830) and dissolved organic matter (DOM, R^2 = 0.896), but lower model fit for particulate organic matter (POM, R^2 = 0.625). Accordingly, the range of quantile regressions was narrower for SS and DOM than POM (Figure 6.3).

Direct calculation of Q_{fed} and Q_{half} proved computationally efficient and avoided potential error associated with binning data (Vogel et al. 2003), but discharge binning influenced estimates of Q_{eff}. Effectiveness curves contained noise because of discontinuity in frequency distributions using only five years of data (i.e., discharge bins with zero observed flow events). Variability influenced the effective discharge estimate for a single case, suspended sediment from 1998-2002 (Figure 6.4). The effective discharge for this time period was identified as 377 m^3/s, which is the maximum discharge over the 75 year period of record. This value results from an insufficiently large period of record (i.e., 5 years) to adequately represent the frequency of this extreme event (i.e., 75-year event). This data point was discarded from further analyses to minimize the influence of assumed discharge bins on the conclusions of our sensitivity analysis. These discontinuities are a well-recognized problem associated with discharge binning (e.g., Ma et al. 2010), which we have attempted to minimize by using an objective procedure for calculating frequency distributions (Klonsky and Vogel 2011).

All three metrics were consistently sensitive to changes in frequency distributions across each process. On the basis of discharge, Q_{half} was more sensitive than Q_{eff} or Q_{fed} (Figures 6.5A-6.5C). However, when metrics were normalized by baseline values from Table 6.1, the metrics show similar sensitivity in the range of ±40% (Figures 6.5D-6.5F).

Conversely, effectiveness metrics were differentially sensitive to changes in rating curves. Metric values for each quantile show how alternative rating curves can influence metric estimates (Figures 6.6A-6.6C). The discontinuous properties of Q_{eff} arise from discharge binning, which was not used for the continuous Q_{fed} and Q_{half} calculations. For relative comparison, these values were again normalized by baseline values from Table 6.1 (Figures 6.6D-6.6F). Sensitivity to rating curve parameterization increased from Q_{eff} to Q_{fed} to Q_{half}. Sensitivity of effectiveness metrics to changes in the rating curve also increased as rating curve fit decreased (i.e., R^2 was lowest for the POM rating, which showed the most sensitivity). That is, uncertainty in rating curve fit was translated directly into uncertainty in effectiveness metrics.

Discussion

Previous analyses have focused on sensitivity of effectiveness metrics across multiple sites (e.g., Crowder and Knapp 2005, Doyle and Shields 2008). Investigators have identified the influence of rating curve slopes on effective discharge estimates (Vogel et al. 2003, Goodwin 2004, Doyle and Shields 2008, Quader and Guo 2009). Rating curve intercepts do not influence estimates of effective or half-load discharge obtained with analytical solutions (Vogel et al. 2003), but they do influence functional-equivalent discharge (Doyle and Shields 2008). Analytical (Vogel et al. 2003, Goodwin 2004) and empirical (Barry et al. 2008, Doyle and Shields 2008, Quader and Guo 2009, Klonsky and Vogel 2011) results to date have also highlighted the influence of discharge variability on effectiveness metrics. Consistent with our findings, Barry et al. (2008) observed increasing errors in effective discharge predictions with increasing rating curve uncertainties.

In contrast to these multi-site approaches, this study examined variability of multiple metrics at a single site. To do so, we were required to make two critical assumptions: (1) a five-year period of record is sufficient for estimating a frequency distribution and (2) limited observations of sediment and organic matter transport are sufficient to develop rating curves. A five-year period cannot fully represent long-term climatic fluctuations such as the Atlantic Multi-Decadal Oscillation (Johnson et al. 2013). A five-year interval represented a compromise between increasing the number of frequency distributions (n = 15 independent periods) and maintaining sufficient length of record (*n* = 1,825 days). Klonsky and Vogel (2011) showed that calculated effective discharge on the Eel River in California was stable beyond 1,000 days of record, and our five-year period (*n* = 1,825 days) is well beyond this threshold. Although other forms of uncertainty analysis have proven useful in estimating sediment transport from limited data (e.g., Bayesian inference; Schmelter et al. 2012), we deem quantile regression to be a useful compromise between insight building and speed of application. While somewhat acute, limited discharge records and minimal rating data are commonly encountered in many water management problems. Our analysis directly addresses these assumptions by examining the sensitivity of effectiveness metrics to this range of input uncertainties. In summary, we feel that our procedure for site-specific sensitivity testing is sufficient for examining the compounding effects of uncertainties common to effectiveness analysis.

Our sensitivity analysis was parameterized to identify extreme conditions for frequency distributions and rating curves. As explained above, frequency distributions varied widely, and the quantile approach to rating curve development could be interpreted as a 90% confidence interval over the observed transport values. Under these extreme perturbations, effectiveness metrics were more sensitive to changes in frequency distributions than rating curves. In particular, sensitivity to changing frequency distributions (Figure 6.5) was remarkably consistent across all three metrics and all three processes (ranging approximately from 60-140% of mean conditions). Conversely, even large perturbations to rating curves (Figure 6.6) resulted in relatively small changes in effective discharge (90-110% of mean conditions) and moderate changes in functional-equivalent discharge (90-130% of mean conditions). However, half-load discharge was consistently more sensitive to these changes in rating curves (70-260% of mean conditions).

Effectiveness metrics varied consistently across the three transport constituents considered, with SS > POM > DOM (Table 6.1). Higher effectiveness metrics for suspended sediment over organic matter are consistent with the findings of Wheatcroft et al. (2010). Higher thresholds for motion of particulate over dissolved constituents are a result of increasing rating curve slope with increasing constituent size. Doyle et al. (2005) also observed variability in effective discharge with size fraction of organic matter, although without consistent trending across their diverse sites. Effectiveness analysis seems highly appropriate for physical and ecological process for which discharge exerts “first-order control” (Wheatcroft et al. 2010) such as these transport-mediated processes (Doyle et al. 2005).

Conclusions

Effectiveness analysis provides a promising analytical framework for addressing ecological response to varying river discharge (Doyle et al. 2005), and the notion of identifying effective discharges corresponding to multiple physical and ecological processes holds great potential (Wheatcroft et al. 2010). For instance, river restoration projects could be designed not only around geomorphically effective discharges (Shields et al. 2003), but also around ecologically effective values. This framework could also be applied to determine ecological thresholds associated with river discharge that could then be translated into environmental flow recommendations (Vogel et al. 2007, Poff et al. 2010, Zarris 2010, Deitch and Kondolf 2012, Meitzen et al. 2013). Importantly, applications to ecological endpoints may require a shift in paradigms from a single design discharge to a suite of effective flows, which is already underway for sediment transport problems (Doyle et al. 2005, Barry et al. 2008, Soar and Thorne 2012). The analysis presented here demonstrates how uncertainties associated with the system can be included in effectiveness analyses and accounted for in these decisions.

Future Research

Additional development of the effectiveness framework is needed as applications to ecological endpoints continue to expand. First, ecological processes may not increase monotonically with discharge, and alternative forms of rating curves may be required (Doyle 2005, Doyle et al. 2005). As these develop, it is crucial to consider the limits of a functional form rather than statistical fit alone. For instance, the suspended sediment data presented here are best fit by a linear rating ($Q_t = 19.0Q - 168$, R^2 = 0.989). However, we did not use this function because it implies the physical impossibility of negative sediment transport when discharge is less than 8.8 m3/s. Because effectiveness metrics often occur near the extremes of rating curves, we cannot overemphasize the importance of realistic boundary conditions when selecting a rating.

Secondly, prior to wide application of ecological effectiveness analysis, additional elements of a river's flow regime (i.e., duration, timing, and rate-of-change; Poff et al. 1997) must be incorporated into the framework. For instance, discharge effectiveness could vary according to timing or seasonality (Doyle 2005). In fact, Doyle et al. (2005) observed somewhat large differences in effective discharge for organic matter transport at a single site across four seasons. Herein, we treated organic matter transport data as an aseasonal rating to maximize sample size (n = 113). However, these data have previously been shown to exhibit seasonal trends between winter and summer (Nelson and Scott 1962). Seasonality could have important implications for ecological processes such as abundance of filter feeding or shredding macroinvertebrates. This simple example illustrates the importance of extending effectiveness to incorporate other elements of a river's flow regime.

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Table 6.1. Baseline effectiveness metrics.

Transport Constituent	Rating Curve	Correlation Coefficient (R^2)	Q_{eff}	Q_{fed}	Q_{half}
Suspended Sediment	$0.857\ Q^{1.63}$	0.830	14.1	19.5	49.5
Dissolved Organic Matter	$0.0256\ Q^{1.18}$	0.896	12.6	15.4	22.9
Particulate Organic Matter	$0.00436\ Q^{1.48}$	0.625	13.4	18.0	37.1

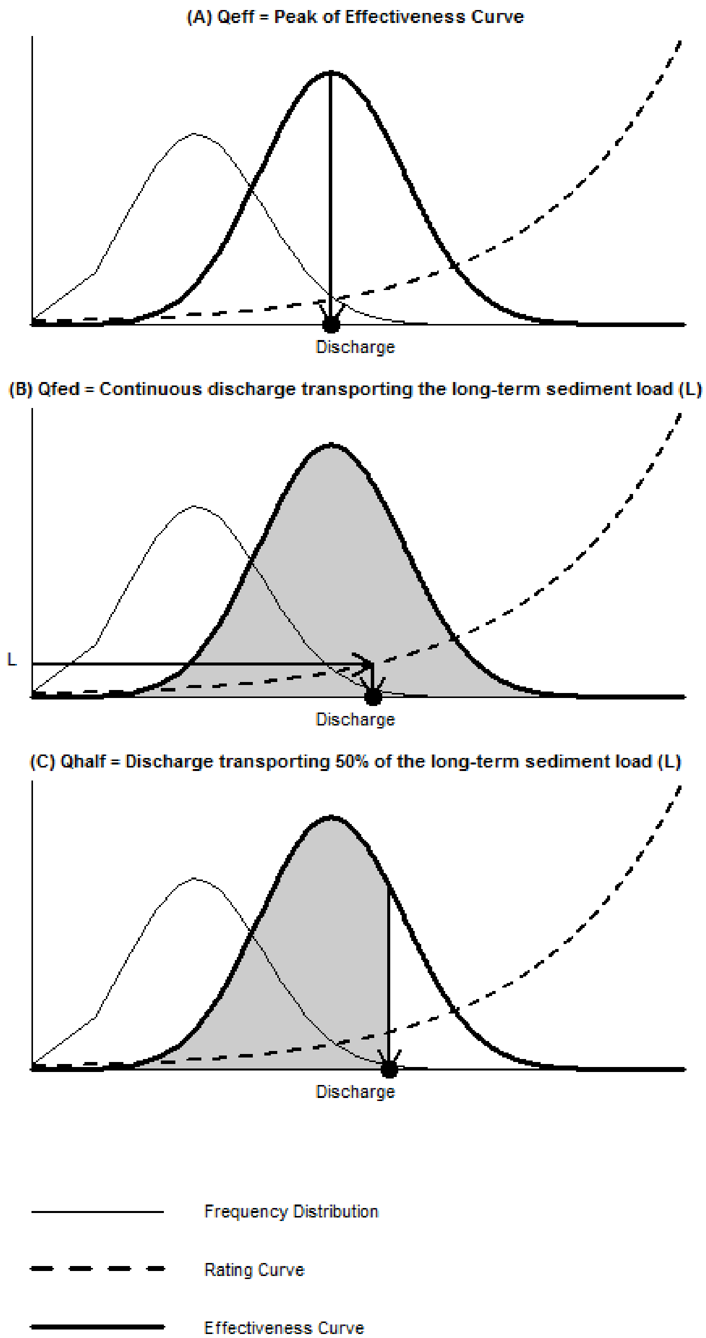

Figure 6.1. Conceptual schematic of effectiveness metrics: (A) effective, (B) functional-equivalent, and (C) half-load discharges (adapted from Wolman and Miller 1960, Doyle et al. 2005, Doyle and Shields 2008).

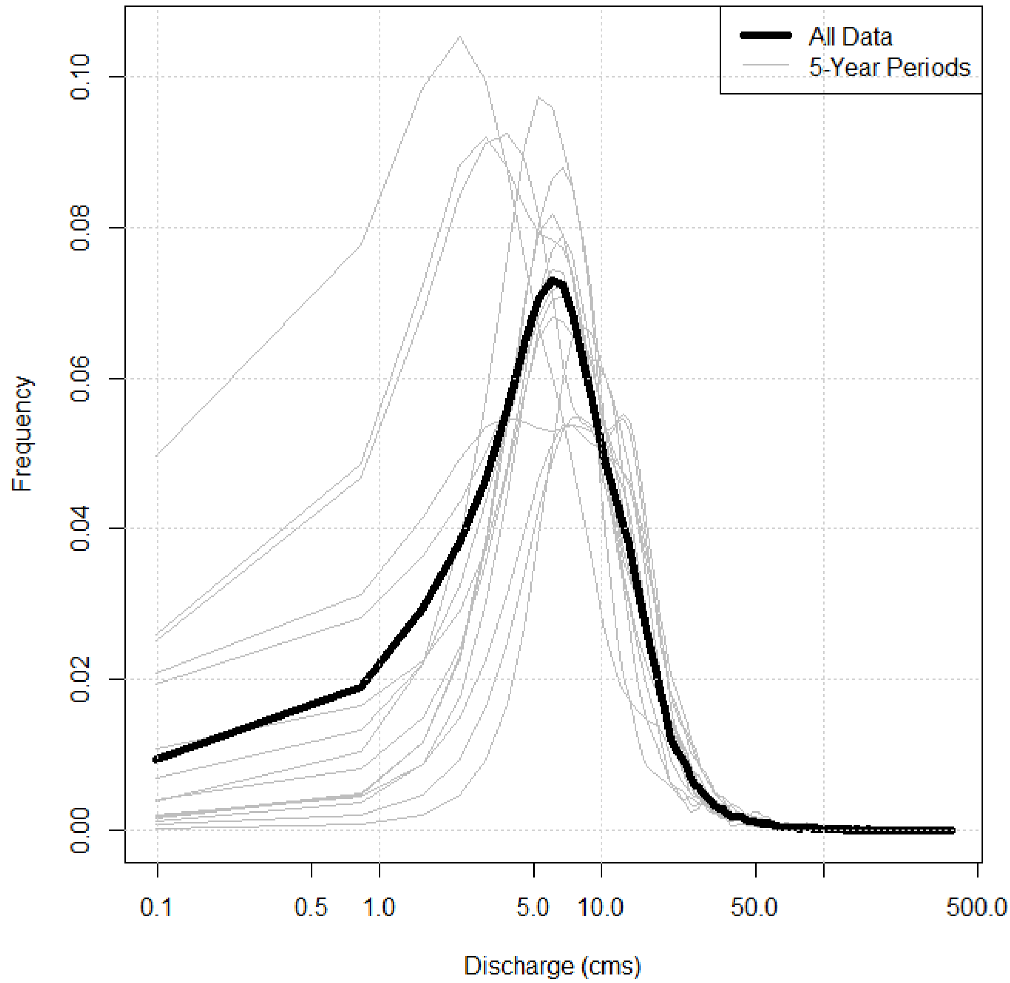

Figure 6.2. Uncertainty in frequency distributions for daily discharge records in the Middle Oconee River. The black line indicates the frequency distribution using the entire 75 year record (*n*=27,375), and the grey lines indicate individual 5 year segments of the record (*n*=1,825).

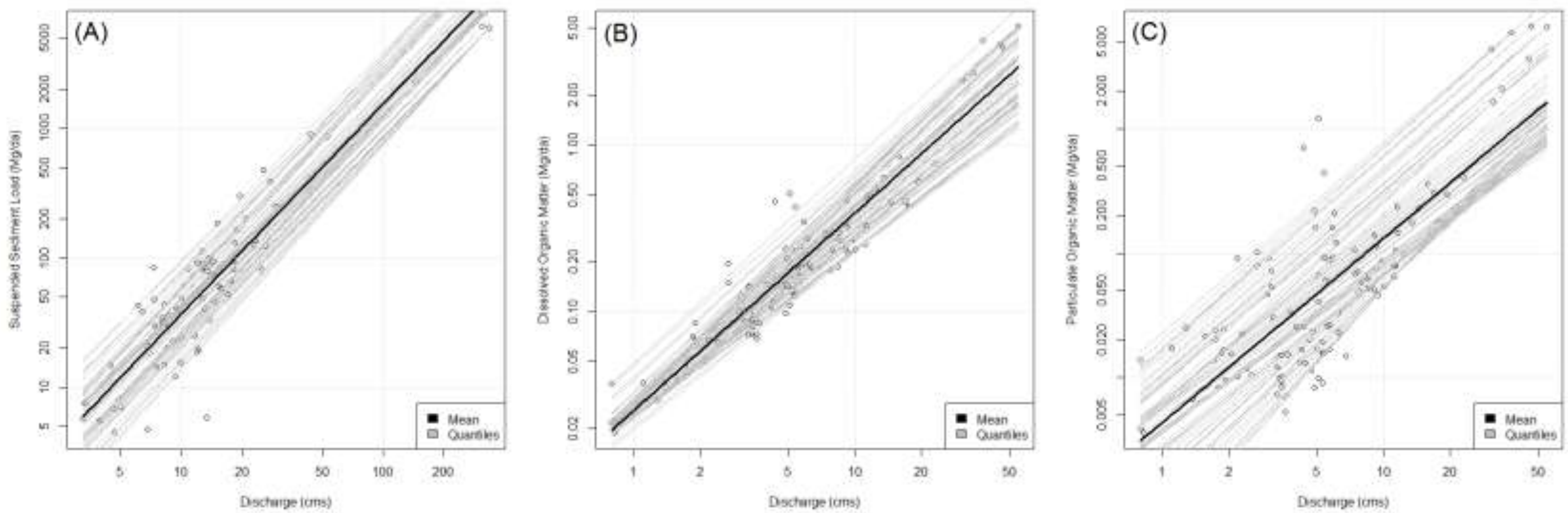

Figure 6.3. Uncertainty in rating curves using quantile regression: (A) suspended sediment, (B) dissolved organic matter, and (C) particulate organic matter. Linear least squares regressions shown in Table 6.1 are also presented for SS (R^2 = 0.830), DOM (R^2 = 0.896), and POM (R^2 = 0.625).

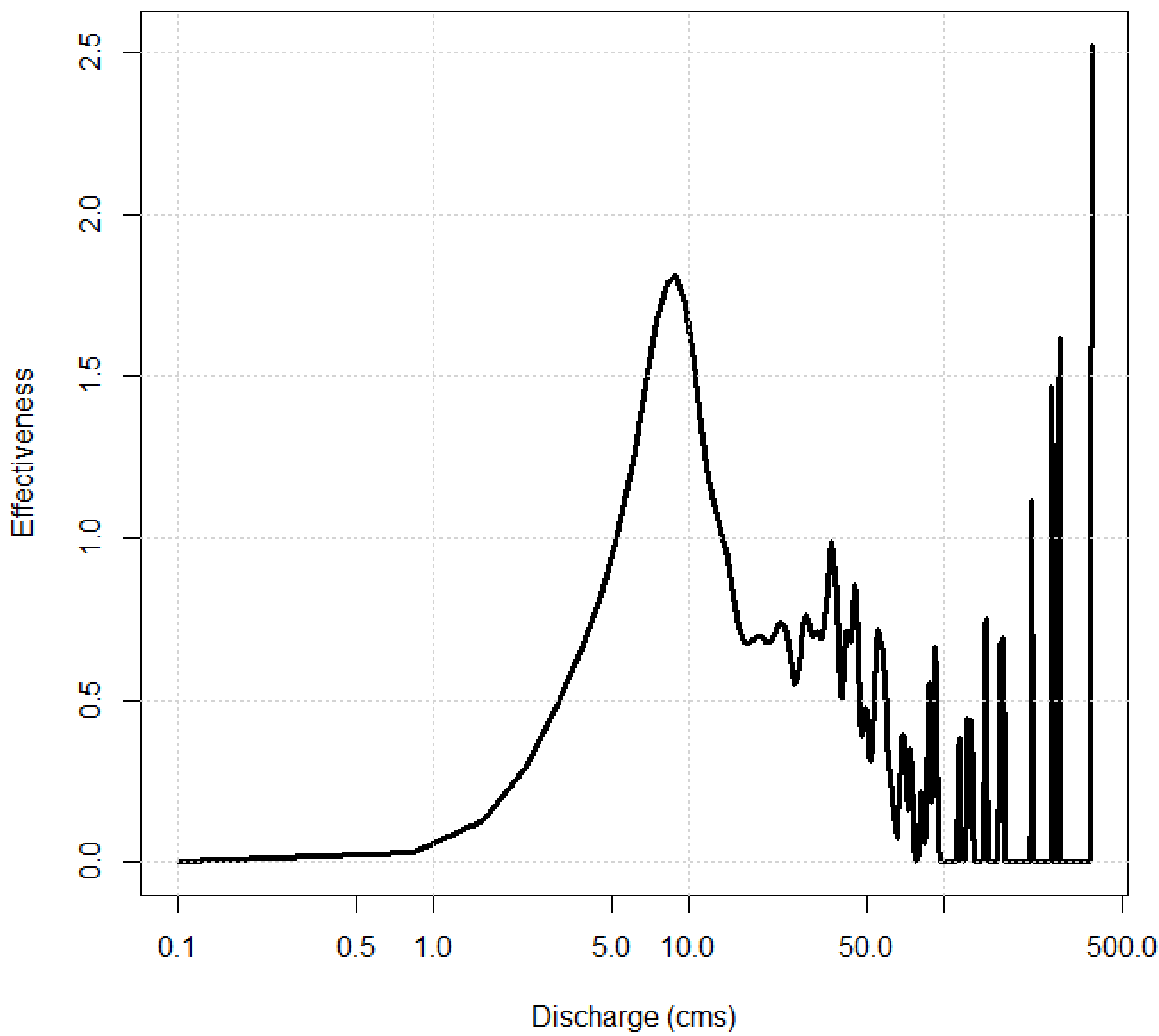

Figure 6.4. Suspended sediment effectiveness plot for 1998-2002 in Mg/da.

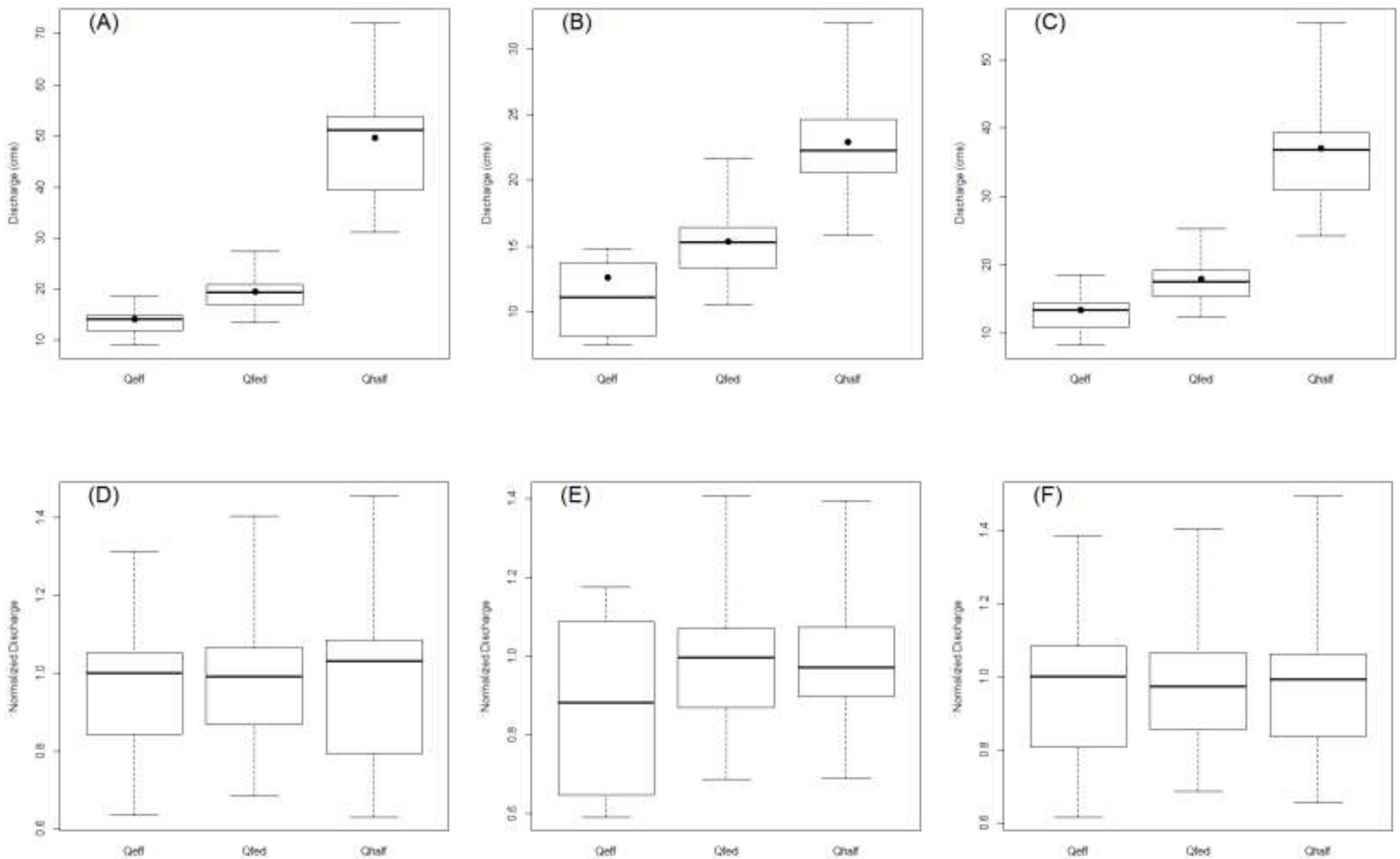
(A)
(B)
(C)
(D)
(E)
(F)
Discharge (cms)
Normalized Discharge
Qeff
Qfed
Qhalf

Figure 6.5. Sensitivity of effectiveness metrics to alternative frequency distributions from individual 5 year periods. (A/B/C) Discharge metrics in m^3/s. Black marker indicates the effectiveness metrics using the long term frequency distribution for the period of record. (D/E/F) Discharge metrics shown in A/B/C normalized by effectiveness metrics using the long term frequency distribution. In Figures 6.5A and 6.5D, a single estimate of effective discharge was removed as a computational outlier; see Results section and Figure 6.4 for additional explanation. In each box plot, the thick black line is the median, box extremes are the 25th and 75th percentile, and whiskers are minimum and maximum observed points.

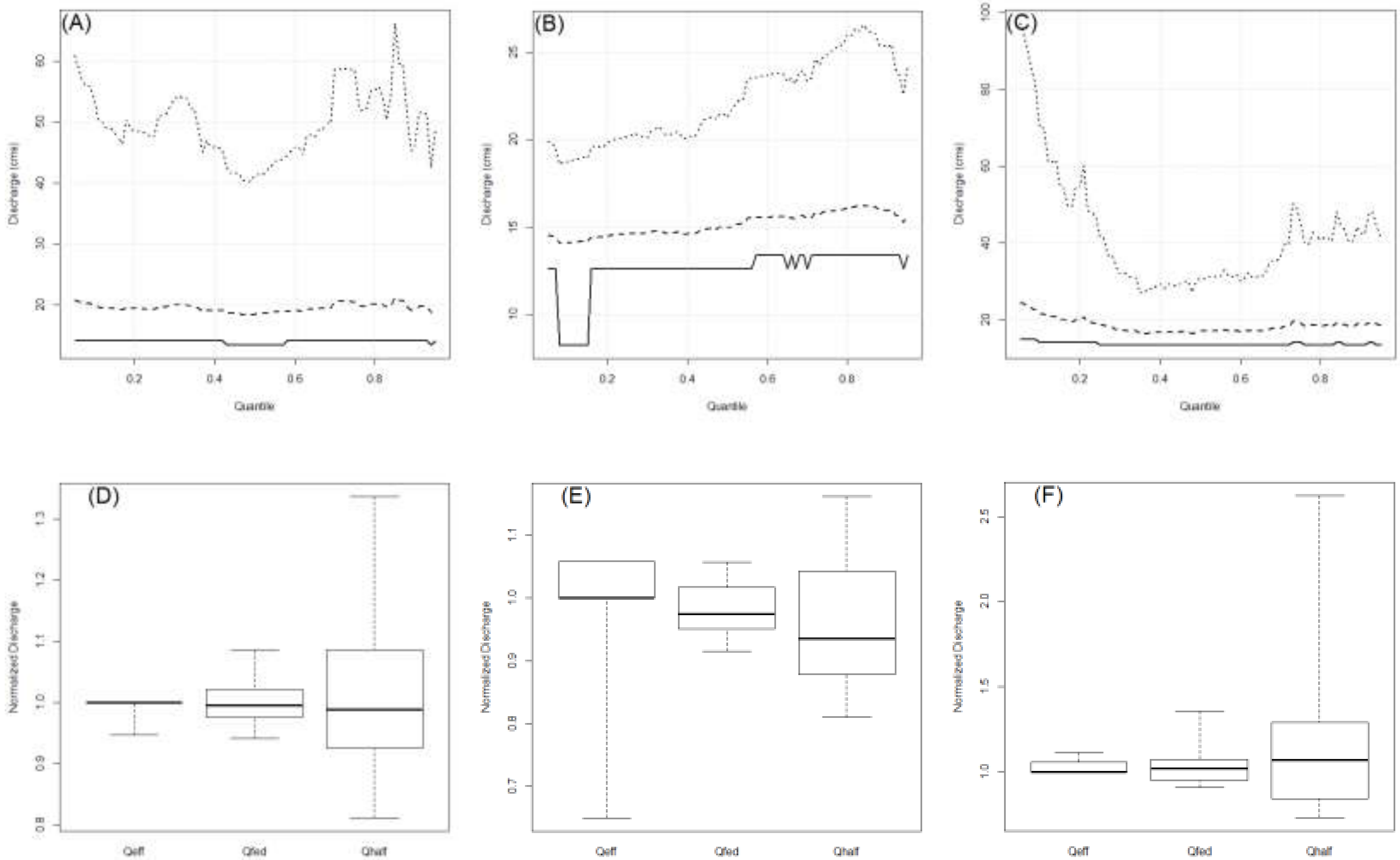
(A)
(B)
(C)
(D)
(E)
(F)
Discharge (cms)
Quantile
Normalized Discharge
Qeff
Qfed
Qhalf

Figure 6.6. Sensitivity of effectiveness metrics to alternative rating curves identified by quantile regression: (A) suspended sediment, (B) dissolved organic matter, and (C) particulate organic matter. Effectiveness metrics are denoted as Q_{eff} (———), Q_{fed} (– – – –), and Q_{half} (········). Discharge metrics normalized by effectiveness metrics using the long term frequency distribution and least squares regression: (D) suspended sediment, (E) dissolved organic matter, and (F) particulate organic matter. In each box plot, the thick black line is the median, box extremes are the 25th and 75th percentile, and whiskers are minimum and maximum observed points.

CHAPTER 7

CONCLUSIONS

Introduction

Whether justified by economic, environmental, or socio-cultural ends, there is a strong imperative for thoughtful management of freshwater resources (Baron et al. 2002). Meeting this need requires a framework for river management decisions that is transparent, fair, repeatable, and capable of explicitly stating trade-offs. This dissertation addressed these challenges through two primary tasks: (1) develop tools and techniques for informing environmental flow decision making; and (2) demonstrate the application of these methods to inform water withdrawal decisions in the Middle Oconee River near Athens, Georgia.

Tools for informing environmental flow decisions

Although developed in the context of the Middle Oconee River, this dissertation has produced a number of techniques and tools that are broadly applicable beyond the constraints of this case study. In Chapter 2, novel methods for visualizing time series were applied to river hydrographs to understand and communicate hydrologic variability. These methods were developed over a decade ago in the field of visual analytics (Ainger et al. 2011), but their adoption by river scientists has lagged. In Chapters 4 and 5, I used hydrologic simulation as a mechanism for comparing alternative environmental flow regimes. This type of analysis builds a strong "hydrologic foundation" for environmental flow decision making (Poff et al. 2010), and utilizes existing data sets in a novel fashion. In Chapter 5, effective discharge analysis was used to integrate ecological processes and hydrologic variability. Although magnitude-frequency analyses have been applied previously to ecological processes (e.g., Doyle et al. 2005), the applications here extended this framework to include other elements of the natural flow regime (i.e., timing, duration, and rate-of-change), multiple independent variables, and the use of effectiveness metrics as integrative response variables rather than design targets. In Chapter 6, a suite of techniques for assessing the sensitivity of effectiveness metrics were applied. These methods could not only be applied to environmental flow problems, but also to more traditional effective discharge analyses in the fields of geomorphology and river engineering.

Middle Oconee River case study

All of the techniques described above were applied in the context of a single case study on the Middle Oconee River near Athens, Georgia. In the Middle Oconee River, a four-county water authority is pumping water to an off-channel reservoir for municipal use. This dissertation has applied a structured decision making framework to examine some of the economic and ecological trade-offs associated with alternative pumping schemes. In particular, the collective body of this dissertation addressed a single question: How can the water authority withdraw the most water from the Middle Oconee River with the least ecological impact?

To answer this question, I applied the PROACT structured decision making framework (Hammond et al. 1999). In this framework, PRoblems are first characterized, Objectives are set, Alternatives are developed, Consequences are examined, and Trade-offs are made between objectives. Chapters 1 and 2 were intended to characterize the decision problem and understand the challenges of managing withdrawals over more than four orders of magnitude of river discharge in the Middle Oconee River. In Chapter 3, I reviewed alternative methods for providing environmental flows. In Chapter 4, I examined the consequences and trade-offs associated with the objectives of maximizing water withdrawal and maximizing similarity between altered and unaltered hydrographs. In Chapter 5, I studied the consequences and trade-offs associated with the objectives of maximizing water withdrawal and maximizing young-of-year fish recruitment. In Chapter 6, I studied the effective discharge associated with transport of suspended sediment, particulate organic matter, and dissolved organic matter. For the purpose of this concluding chapter, I also analyzed these three transport processes relative to the environmental flow alternatives presented in Chapters 4 and 5. In this analysis, the area under the effectiveness curve was used to summarize the total mass transported over a given flow regime. Figure 7.1 compiles the five "lines of evidence" into trade-off curves for these ecologically relevant processes in the Middle Oconee River. Ecological processes are all presented relative to four alternative environmental flow regimes and their accompanying water withdrawal rates: annual minimum flows, annual minimum flows constrained to avoid water treatment costs, monthly minimum flows, and sustainability boundaries. Figure 7.2 presents the mean of all five processes to synthesize the findings of this dissertation for simple interpretation. Both figures are presented as trade-off curves, where a decision-maker could apply individual value judgments about the degree of environmental change or the rate of water withdrawal.

The collective body of this dissertation leads to the following conclusions and recommendations for management of Middle Oconee River water withdrawals.

- Overall, sustainability boundaries (*sensu* Richter et al. 2011) provide more water for a given level of environmental impact and less impact for a given level of water withdrawal. Thus, sustainability boundaries are more robust than minimum flow approaches relative to both water withdrawal and environmental endpoints.
- In Chapters 4 and 5, a range of minimum flow thresholds were examined. A minimum flow associated with the 7Q10 of 45 cfs (Carter and Putnam 1978) corresponds to an

average annual withdrawal rate of 57.7 MGD. This minimum-flow threshold does not appear to be associated with any particular level of environmental impact, nor does it correspond to thresholds in environmental impact. Thus, I conclude that this hydrologic statistic is somewhat arbitrary, and more focused analyses need to be applied to determine minimum flow thresholds.

- Although often viewed as environmentally superior, these analyses also indicate that monthly minimum flows incur as much change in flow-dependent responses as annual minimum flow criteria.
- In Chapter 4, I found that the permitted withdrawal rate of 60 MGD can only be obtained in 75% of years even without a minimum flow regulations. Furthermore, if high flow withdrawals are avoided (as is often the case to minimize water treatment costs), the river cannot meet this 60 MGD permit in any years. Thus, the single permit associated with Bear Creek Reservoir over-allocates the capacity of the Middle Oconee River. This analysis ignored the effects of other permitted withdrawals (e.g., Athens-Clarke County's withdrawal at Ben Burton Park), which would further exacerbate this finding. This finding also indicates that other withdrawal permits should either be denied or strictly limited (e.g., to high flow seasons) to avoid future conflicts over limited water quantity.

Future Research

Here, I have developed a variety of new approaches to examining environmental flow problems. However, much work remains to be done both in the development of tools and in the assessment of the Middle Oconee River. The following items provide fertile ground for future research in both topics:

- Chapter 5 demonstrated the application of effective discharge analysis to a non-traditional ecological process. This framework could be extended to other ecological processes (e.g., metabolism, leaf decomposition). The framework could also be applied with time series variables other than discharge (e.g., depth, shear stress, light).
- Off-channel reservoirs are currently common alternatives being proposed to address water supply in Georgia. The techniques presented in this dissertation could be easily adapted and applied throughout this region to determine water yield, environmental flow thresholds, and environmentally sensitive operational approaches provided that there are sufficient discharge data and appropriate rating curves.
- Extension to the Middle Oconee River case study:
    - The analyses presented here were limited by neglecting reservoir volume as a constraint. This assumption should be addressed in future analyses to better reflect the potential water yield for municipal use. Furthermore, the existing size and pumping capacity was not addressed beyond the maximum pumping rate of 60 MGD. Current pumps are single speed, which could complicate provision of the environmental flow recommendations made above. Operational practicalities cannot be overlooked as they can affect the overall success of an environmental flow (Davies et al. 2013).
    - This study assumed that withdrawal rate was a reasonable proxy for economic benefits of freshwater resources. However, there likely exist thresholds in the economic advantage of a particular rate of water yield. Furthermore, there are other economic and socially relevant objectives that could be added to future

analyses such as the desires to minimize water treatment cost or maximize recreational opportunities (e.g., kayaking and fishing).

- This study examined five ecologically relevant endpoints (Figure 7.1). However, many other ecological endpoints have been studied in the Middle Oconee River, such as: secondary production (Nelson and Scott 1960, Grubaugh and Wallace 1995), aquatic macrophytes (Nelson 1957, Pahl 2009), benthic macroinvertebrates (Katz 2009), benthic fishes (R.A. Katz and M.C. Freeman, unpublished data), and filter-feeding invasive clams (W.G. McDowell, unpublished data). In addition to these existing data sets, analyses could be extended to include other ecological processes such as river metabolism, leaf decomposition, and habitat availability.
- This analysis also focused on a daily time scale. Due to large pumping capacity, the water authority withdraws much of their daily volume in a short period of time (e.g., 8 hours or less). These large pumps create an unnecessary environmental cost associated with "peaking" the hydrograph (Richter and Thomas 2007). Future analyses need to address intra-daily variation in flows.
- The Bear Creek withdrawal is part of a larger water supply system including withdrawals on the North and Middle Oconee Rivers in Athens Clarke County and other intakes in other counties. This broader water system must be included to rigorously understate the optimal operation of the Bear Creek withdrawal.

Concluding Remarks

This dissertation has sought to apply the best available science to inform environmental flow recommendations for the Middle Oconee River. However, the focus now must shift to the more challenging issue of implementing these recommendations. First, these recommendations were derived from models and simulations, and while these techniques are extremely useful, they are not reality. Environmental flows need to be analyzed, tested, and refined in an experimental or adaptive management framework (Konrad et al. 2011, Olden et al. 2014). Second, analyses do not make decisions, people do. Implementing environmental flows requires working with the operational specialists that are constrained by regulatory issues, infrastructure, and the need to balance objectives not considered here. Working through logistical and social complexities is a crucial part of developing and applying effective environmental flows (Poff et al. 2010).

The best environmental flow alternative is reduced demand. Environmental flows often focus on minimizing environmental impact for a specified level of water supply. However, that same environmental impact could potentially be addressed by reducing the demand on the water system. Reducing demand requires wide understanding by the general public as water users. For instance, if every citizen in the region serviced by Bear Creek Reservoir (~310,000; Campana et al. 2012) watered their lawn for five less minutes per day during the growing season (~120 days), there would be a nearly 10% decrease in water demand! These incremental, individual decisions can collectively lead to large reductions in demand for freshwater resources.

This dissertation has focused on trade-offs associated with water management. While there can be sources of conflict among competing uses, I remain hopeful that the Middle Oconee River can be managed to adequately meet both economic and ecological needs for freshwater. The best window to address water resources conflict is before contention arises (Baron et al. 2002). Bear Creek Reservoir currently withdraws less than one third of its permitted rate (~ 20 MGD of the 60 MGD permit). By analyzing and managing these problems now, we have an opportunity to avoid future conflicts over what all agree is a valuable resource and local treasure.

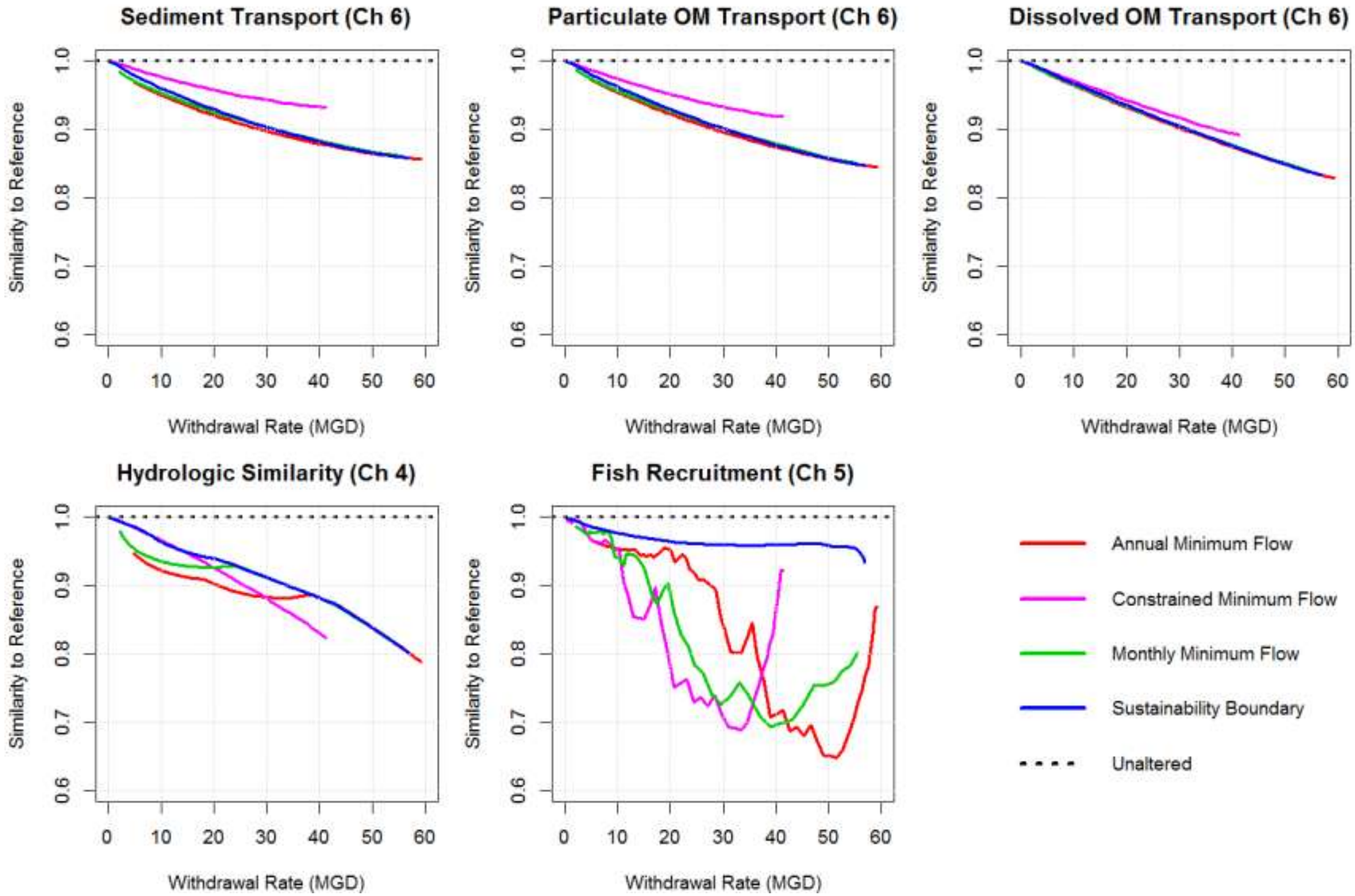

Figure 7.1. Trade-offs studied in the Middle Oconee River.

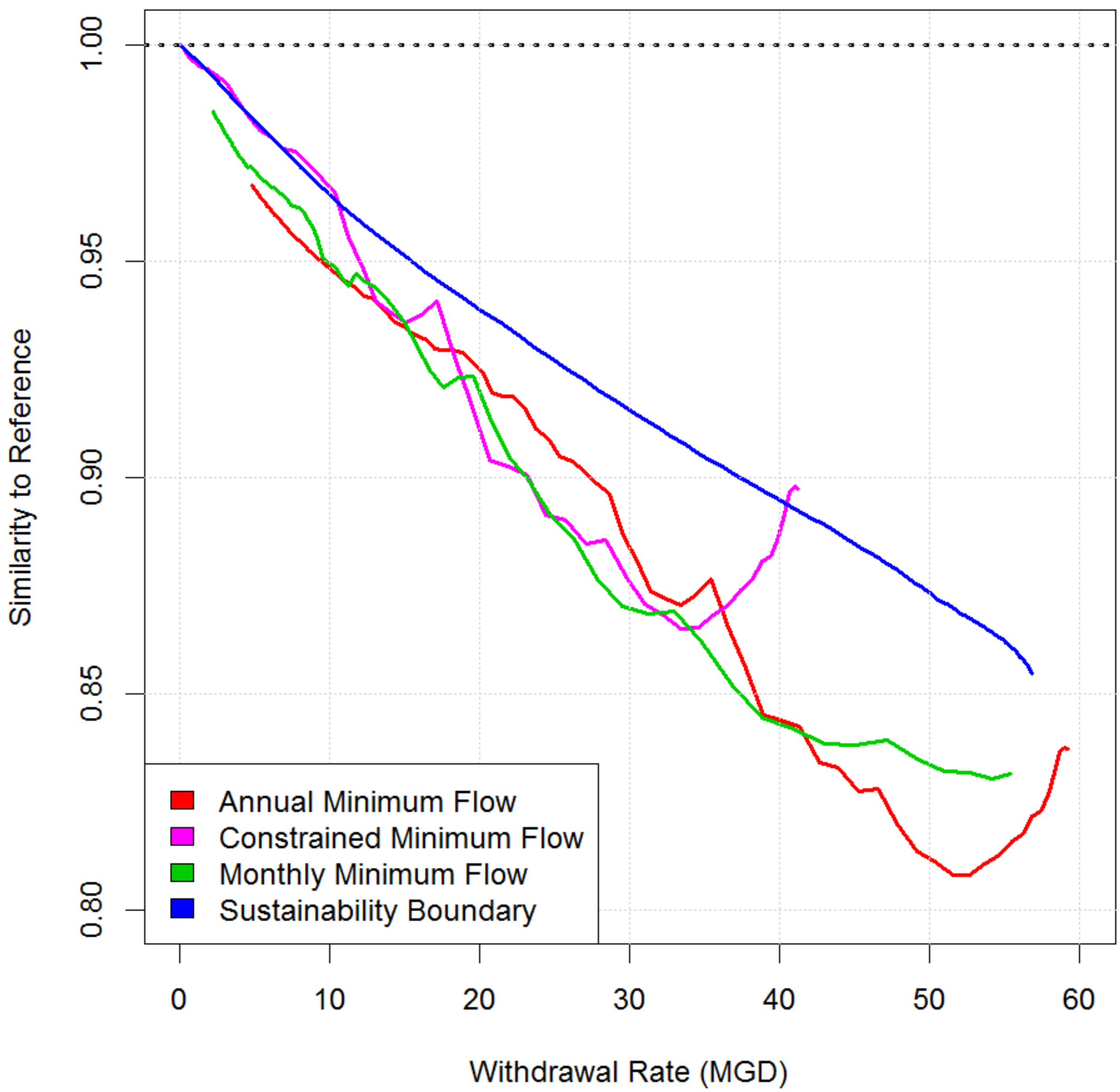

Figure 7.2. Trade-offs in the Middle Oconee River.

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APPENDIX A

DECLARING DROUGHT FOR EFFECTIVE WATER MANAGEMENT[5]

[5] S. Kyle McKay and Todd C. Rasmussen. 2013, *Proceedings of the Georgia Water Resources Conference*, Reprinted here with permission of publisher.

Abstract

Water managers are tasked with resolving conflicts between freshwater resource uses, which range from municipal water supply, to recreation, and to sustaining aquatic ecosystem integrity. Further complicating management, hydrologic processes experience numerous sources of periodic, quasi-periodic, and episodic variation. Water allocation trade offs are often most complex and contentious when availability is low. Drought is a “recurring extreme climatic event over land characterized by below-normal precipitation over a period of months to years” (Dai 2011).

Water managers often apply indicators of climatologic and hydrologic conditions to identify when drought conditions are reached (e.g., Palmer Drought Severity Index, streamflow, respectively). These indicators inform drought declarations, with associated drought responses such as watering restrictions.

Herein, we suggest techniques for predicting and declaring oncoming drought to improve the accuracy of drought declarations. We hypothesize that drought indicators in preceding months are predictive of future drought levels. Specifically, we develop predictive models using the Palmer Hydrologic Drought Index, a common drought indicator. We then demonstrate the utility of our model for drought declarations for the Middle Oconee River near Athens.

Introduction

Hydrology in the Southeastern United States under-goes numerous sources of variation, such as daily river fluctuations, seasonal climate dynamics, and extreme events. Drought is a common and normal component of this naturally fluctuating regime (Stooksbury 2003). Alt-hough the societal costs of recent droughts have been significant, more severe droughts have been observed in both instrumented and modeled history (Campana et al. 2012, Pederson et al. 2012).

Over twenty indicators of drought are commonly applied to measure and evaluate ambient and historical moisture conditions (Heim 2002, Dai 2011). These indicators help water managers identify, declare, and respond to drought conditions (Campana et al. 2012).
We address techniques for identifying oncoming drought conditions in order to declare drought and preemptively adjust water use and management schemes. Specifically, we develop an approach which relies on varied applications of the Palmer Hydrologic Drought Index for comparing alternative drought prediction techniques.

Study Site

The Middle Oconee River watershed is a 398 mi^2 basin in the rapidly developing Georgia Piedmont. In 1997, Barrow, Clarke, Jackson, and Oconee counties jointly constructed Bear Creek Reservoir under the auspices of the Upper Oconee Basin Water Authority (UOBWA 1997). This 500-acre off-channel reservoir is filled using water pumped from the Middle Oconee River and is a primary water source for these counties.

The four-county water authority (herein referred to as UOBWA) maintains a contingency plan to respond to — and mitigate the effects of — drought (UOWBA 2003). This plan specifies three primary drought indicators that are used to declare five levels of drought:

- Palmer Hydrologic Drought Index (*PHDI*),
- Middle Oconee River streamflow at Arcade GA, and
- Bear Creek reservoir levels.

For the purpose of this paper, we focus only on *PHDI* to evaluate our proposed method; however, similar analyses could be undertaken for drought declaration using other indicators.

The PHDI assesses long-term hydrologic conditions for the region on a non-dimensional scale from 6 (extremely wet) to -6 (extremely dry). The UOBWA plan identifies five drought levels based on *PHDI* values (Table A.1), with each level having specific water conservation targets.

Methods

Real-time drought indicators are rarely available due to data and computational burdens. Thus, water management decisions typically rely on data from a previous time period. For instance, August decisions may use July's *PHDI* data, or this week's decisions may use last week's streamflow.

Here, we demonstrate the application of simple time series models for predicting current *PHDI* values and associated drought levels. *PHDI* values are computed monthly by the National Climatic Data Center for the north-central region of Georgia (NOAA 2012; state code = 09, region code = 02).

Application of prior conditions to predict current (or future) conditions is generally referred to as time series analysis. We used standard time-series methods to analyze data from 1895-2011 (n=1404) for the purpose of demonstrating the prediction accuracy of three alternative *PHDI* models. For each model, *PHDI* was predicted at time t based on the following equations:

- Current method (UOBWA)
  - $PHDI_t = PHDI_{t-1}$
- Differenced auto-regressive model (Model 1)
  - $PHDI_t = PHDI_{t-1} + (PHDI_{t-1} – PHDI_{t-2})$
- Auto-regressive model (Model 2)
  - $PHDI_t = (PHDI_{t-1} + PHDI_{t-2}) / 2$

Drought levels were computed for each *PHDI* prediction as well as observed *PHDI* data at each time step. Because drought levels have associated management actions, models were evaluated based on their capacity to predict drought level, rather than PHDI.

The primary evaluation metric is the number of drought levels different from the observed drought level. For instance, if observed drought level at time t is 2, but a model predicted drought level is 3, then the departure is 1.

We summed the number of data points at each level of departure (ε_d) and normalized this by the total number of data points ($\varepsilon_p = \varepsilon_d / n$). The range of predictive error is bounded between $-4 \leq \varepsilon_p \leq 4$, because a prediction error cannot exceed four drought levels. Importantly, positive values indicate a Type-I error (false prediction of drought) and negative values indicate a Type-II error (drought occurred undetected).

To compare our three competing models, an overall score (S) was computed as a weighted average of the magnitude and probability of errors. A low value of this metric indicates high predictive capability, whereas high values indicate low predictive capability (perfect prediction occurs when the score is S = 0). This formulation penalizes larger over smaller errors.

Results

While all three models track *PHDI* values with reason-able certainty (Figure A.1A), large discrepancies between observed and predicted values may occur (Figure A.1B). In general, these *PHDI* predictions provide relatively accurate drought declarations, with all three models predicting over 66% of drought levels correctly and over 88% of values within one drought level (Figure A.2). However, dramatic 2-, 3-, and 4-drought level errors occasionally occur. Interestingly, the models do not demonstrate a bias toward Type-I (false positive) or Type-II (false negative) errors.

Although all models demonstrate similar efficacy, the scores favor the UOBWA method over models 1 and 2 with scores of 0.338, 0.489, and 0.409, respectively. This is attributable to the UOBWA method's low prevalence of drought-level differences of more than one level (5.6%).

Discussion

Early detection of dry conditions and accompanying drought declaration is critical to water management. This analysis demonstrates a simple method for comparing alternative detection methods using three simple time series models, and compared their utility in declaration decisions. In this analysis, the existing UOBWA model proved superior to the other two formulations. However, additional research is needed to reduce the number of declaration errors, including:

Time series models. Time series analysis has a wealth of applications, from tracking markets to weather forecasting. We present three simple models, but additional analyses may reveal that more complex models would better detect drought. For instance, Rugel et al. (2012) demonstrate predictability of river discharge in Georgia on time scales as long as six months.

Drought indicators. Drought detection is a complex multi-metric process, and we have only considered a single variable, the *PHDI*. The UOBWA also relies on river discharge at weekly and monthly time scales to make withdrawal decisions (UOBWA 2003). Furthermore, other indicators such as the Standard Precipitation Index (*SPI*) are also quite useful in drought decision-making (Campana et al. 2012).

The UOWBA declares drought levels based on an average of indicators from *PHDI*, discharge, and reservoir levels. The first two indicators are not influenced by local managers, and thus respond to ambient conditions. Reservoir levels, however, may be manipulated out-of-sync with ambient moisture levels, and are likely to be poor drought indicators.

Reservoir levels are response variables to management actions, and we recommend a more nuanced approach for their incorporation into drought declaration decisions. For instance, if drought levels based on *PHDI*, discharge, and reservoir levels were 4, 3, and 0, the overall drought level would be only 2, which is a clear understatement of the existing conditions.

What is needed is a metric that relates the reservoir volume to the predicted total demand for water for the remainder of the season. Thus, low water levels late in the season should be managed differently than low levels early in the season. The difference between the total reservoir volume (plus projected river withdrawals) and the predicted demand for the rest of the season would provide such a metric. In fact, the drought response could be managed so that conservation reduces the gap between available and needed supplies.

Methods such as those presented here help water man-agers preemptively respond to drought by making in-formed drought declaration decisions. Methods for accurate, reliable, and repeatable drought detection and declaration are challenging to develop. However, this topic is likely to become more important in light of increased demand for freshwater in the Georgia Piedmont and the relative wetness of the late-20th century (Pederson et al. 2012).

We have presented a simple method for analyzing alternative drought declaration schemes, and although we have focused on simple models, we believe this general framework is transferrable to other basins, drought indicators, and more complex models.

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Table A.1. Palmer Hydrologic Drought Index (PHDI) for determining UOBWA drought levels.

Drought Level	Palmer Hydrologic Drought Index (PHDI)			Water Use Reduction Goal (%)
0	-0.5 <	PHDI		0
1	-1 <	PHDI	≤ -0.5	2.5
2	-2 <	PHDI	≤ -1	5
3	-3 <	PHDI	≤ -2	10
4		PHDI	≤ -3	20

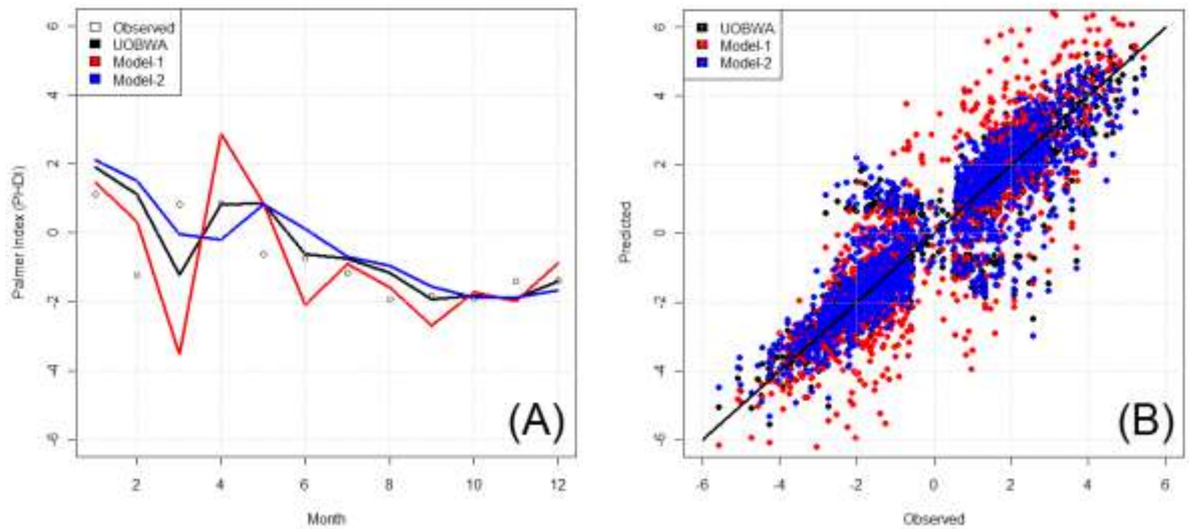

Figure A.1. Model evaluation: (A) model predictions over a sample time period in 2011, and (B) observed vs. predicted PHDI.

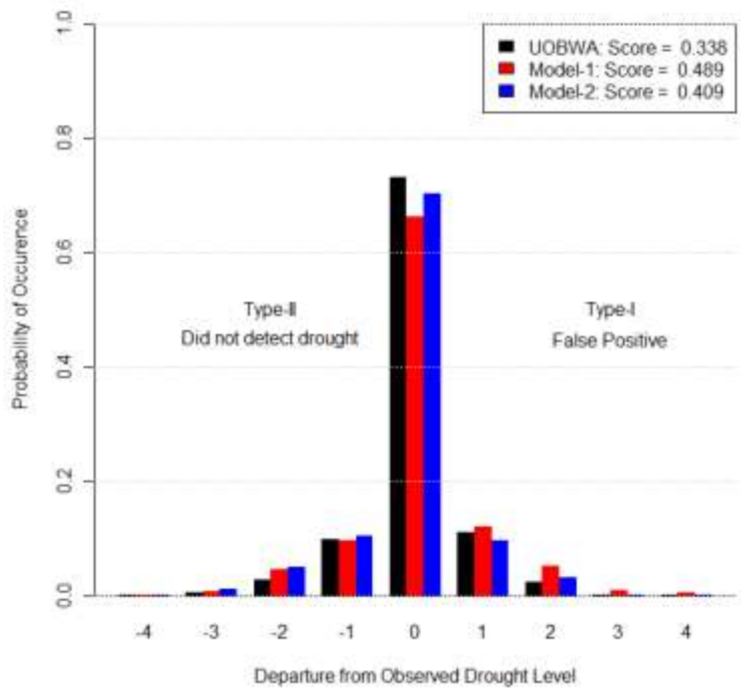

Figure A.2. Departures from observed drought levels.