QUANTIFYING STREAMFLOW NEEDS FOR RIVER ECOSYSTEMS IN THE CONTEXT OF WATER PLANNING AND MANAGEMENT

by

LAURA RACK

(Under the Direction of Mary Freeman)

ABSTRACT

Humans rely on freshwater ecosystems for a range of values including biodiversity, cultural, supporting, provisioning, and regulating services. Freshwater ecosystems have experienced significant changes to flow and nutrient regimes from human modifications. In addition, climate change has resulted in an increase in the frequency and severity of extreme weather events, like drought, resulting in reduced water availability for human and river ecosystem needs. It is critical to manage river ecosystems for both short-term human needs and long-term ecosystem services. In this dissertation, I examined how to manage for environmental outcomes when direct relationships between river flows and ecological outcomes are context dependent and policies and planning were built around municipal, industrial, and agricultural water needs. First, I examined environmental flow relationships for a submerged macrophyte, *Podostemum ceratophyllum*. I found an indirect effect of low river flows on *Podostemum* growth, mediated by algal and sediment accrual and grazing herbivores when water velocities were reduced. Next, I developed and integrated ecological indicators into Georgia's State-wide Water Planning Framework. A few key themes emerged for developing the framework and ecological metrics for planning and for using the approach in other settings. Context dependency was important both

for water resource decision-making and in selecting and evaluating environmental-flow outcomes for local systems. In addition, it was important to evaluate river ecosystem needs alongside other water uses and to provide the information to evaluate and interpret findings to planners or decision makers. In the final study, I evaluated management actions that could improve drought resilience in the Flint River in Georgia, U.S.A. I explored how increasing infiltration throughout the basin, augmenting flow with additional storage, and changing low-flow management operations could impact river flows. I found that management operations were the most critical factor for maintaining adequate river flows during drought periods, and if combined with additional storage would allow municipal utilities to meet short-term human needs without compromising long-term ecosystem services.

INDEX WORDS: water planning, drought, flow-ecology, macrophyte

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DEDICATION

I dedicate this dissertation to the mentors that inspired me.

In high school, Helen Connelly helped me discover my passion for the environment and the interaction between humans and the environment.

After college I had the opportunity to travel to Belize to volunteer with endangered primates, where I met the founders of the rehabilitation center, Zoe and Paul Walker. Through working with Zoe and the Sarteneja Alliance for Conservation and Development I witnessed the power of research and monitoring in practice and saw a glimpse of how you could make a career in conservation. That experience motivated me to pursue graduate school.

Once arriving at UGA I lucked into working with Mary Freeman, one of the most humble, knowledgeable, curious, and kind individuals anyone could work with. I credit Mary for much of my success, she gave me the space to learn on my own, encouraged me to gain new skills, but never failed to help me when I needed it.

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CHAPTER 1

INTRODUCTION AND LITERATURE REIVEW

Water is a complex resource, one that is necessary for human life, unevenly distributed compared to the global population, and physically difficult to transport (Salzman 2012). While the specific nature may differ, in general, water can be a physical, cultural, social, political, or economic resource for humans, often acting as more than one at a time (Salzman 2012). Humans have significantly impacted freshwater ecosystems across scales and continue to do so (Dudgeon 2019). At the widest scale, climate change has impacted the severity and frequency of extreme events, like floods and droughts. Direct modifications to landscapes, river channels, and flow regimes have severely altered river ecosystems, as have pollution, overexploitation, and direct or facilitated spread of invasive species (Dudgeon 2019). Freshwater ecosystems support disproportionally high biodiversity compared to terrestrial ecosystems and we are at a critical juncture to address biodiversity loss and the impacts to freshwater ecosystems which humans rely on so heavily for a range of critical resources (Tickner et al. 2020).

In the United States, surface water resources, rivers and lakes, provide the largest source and use for freshwater (USGS 2018). River ecosystems provide many benefits and values beyond drinking water including provisioning, regulating, cultural, and supporting services, which encompass broad categories of uses and values (Millennium Ecosystem Assessment 2005, Hanna et al. 2017, Vari et al. 2022). During times of drought water availability is reduced across all uses and values, but traditional management systems tend to be protective of human uses over broader ecosystem level protections. In drought times, human uses and river ecosystem water

needs can also be pitted against one another and viewed as a zero-sum game, rather than approaching the problem to optimize outcomes across uses and values.

Ecologists have made large strides in tackling research questions that are important for natural resource management (Poff et al. 2017). Water management decisions often center around water quantity and water quality. In river ecosystems, environmental flows (e-flows), have been developed to quantify and describe the range of river flows (e.g., magnitude, timing, duration, and frequency) needed to support geomorphic and biological processes (Arthington et al. 2018). Context dependency is a common theme that arises in the development of e-flow relationships, however, making it difficult to develop broadly generalizable e-flow guidance (Dewson et al. 2007, Power et al. 2008, Poff and Zimmerman 2010, Walters 2016). In addition, a lot of research has focused more on the development of methods and technologies, leaving much to discover about flow-driven ecological responses (Davies et al. 2014).

It is challenging to integrate e-flow findings into management, policy, and planning frameworks (Arthington et al. 2006, Arthington et al. 2024). Often environmental protection is not prioritized in decision making and water quantity for the environment is not well defined in the legal or regulatory space (Zellmer 2008, Richter 2009). Environmental flow research can be difficult to apply to the structure and pace of decision making in management, policy, and planning, and additional work is needed to translate results into a useful format for decision makers (Cartwright et al. 2017). Translational research approaches offer scientists an opportunity to develop research questions from the early stages that target the types of decisions being made (Enquist et al. 2017) and may be a useful step in integrating science into practice.

My dissertation builds from my master's research where I worked closely with the Upper Flint River Working Group (herein Working Group), started by Ben Emanuel of American Rivers

and Gordon Rogers of the Flint Riverkeeper, that brought together municipal water managers in the Upper Flint River Basin in Georgia, local conservationists, conservation organizations, and Atlanta's international airport sustainability staff. Starting in 2013, the Working Group has focused on ensuring water security for municipal water supply and a shared vision "of a river system healthy enough to maintain the many social, ecological, recreational and economic values that the Flint River system provides—values such as water supply, recreation, fisheries, property values and a healthy river ecosystem" (Emanuel 2019). My research has been shaped by close working relationships with members of the Working Group, members of the Upper Flint and Upper Oconee Regional Water Planning Councils, Georgia Environmental Protection Division, and collaborations with the Georgia Water Planning and Policy Center.

I approached the challenge of managing for environmental outcomes when direct relationships between river flows and ecological outcomes are context dependent and policies and planning are built around municipal, industrial, and agricultural water needs by developing local e-flow relationships, relevant for water planning, and creating metrics and methods for integration into Georgia's Water Planning framework. Much of this work took place within the context of state-wide water planning in Georgia in the Upper Oconee and Upper Flint Water Planning Regions. Georgia passed the Comprehensive State-wide Water Planning Act in 2004 and has made large strides in the information and structure for water planning, but the State's instream flow policy dates back to 1977, and is based on the seven-day, ten-year minimum flow. My work provides a framework for how we could update the way that instream flows are considered in a planning framework.

I started by exploring the relationship of flow-mediated drivers to growth of *Podostemum*, a plant that grows on stable substrates in swift flowing water (Chapter 2). My aim

was to increase our understanding of the mechanisms that may drive biomass loss during low-flow periods, which had been observed in previous studies. To do this I conducted a field study in the Middle Oconee River, near Athens, Georgia, and tracked *Podostemum* growth and cover using two types of data to investigate the hydrologic and hydraulic mediated drivers of the plant: cover data on each study unit (a rock covered with the plant) based on photographs and data for growth rate based on stem measurements. These findings helped to understand the different factors that impact *Podostemum* growth and cover and how they relate to hydrology and hydraulics.

Chapter 3 focuses on developing e-flow metrics in the context of water planning and the general themes that emerged from integrating ecological impacts into a planning framework. We adapted the functional flows framework (Yarnell et al. 2015) for application to Piedmont and Coastal Plain rivers. This framework was developed around the assumption that river flows have been modified by humans and that while we cannot return to an unimpaired flow regime, we can target specific flows that support key geomorphic or ecological process in river ecosystems. Within this framing, we developed five functional flows to represent the river ecosystem that could be evaluated for water planning. We expect that our approach will be useful to other locations where environmental protection is vaguely defined in the regulatory and management context and secondary to meeting other societal needs.

In Chapter 4, I bring together lessons learned about the water planning process and ecological metrics (Chapters 3 and 2) to evaluate alternative water management actions during times of drought for the upper Flint River basin. Using a basin-wide, flow-routing model, I simulate the impact of different drought management actions on low river flows, municipal water utility storage and operations, and *Podostemum* biomass during summer months. This

information was developed through the Georgia Environmental Protection Division grant program to improve information available for water planning and the findings touch on both impacts to the human and environmental dimensions of river ecosystems.

Overall, I aim to quantify ecological outcomes to inform value-laden decisions about water and highlight the connectedness between functioning aquatic ecosystems and societal benefits and services. Often there is a legal or regulatory setting for decisions around water resources, and we found traction by identifying the opportunities to bring ecological outcomes to the table as a quantifiable target to evaluate in decision making. This opened the conversation to consider what opportunities there were in the planning space to proactively manage water resources to meet multiple needs. It is difficult to make well-informed decisions in a reactionary setting, however the planning process provided an opportunity to explore a proactive holistic approach to the management of water resources in Georgia.

References

Arthington, A. H., A. Bhaduri, S. E. Bunn, S. E. Jackson, R. E. Tharme, D. Tickner, B. Young, M. Acreman, N. Baker, S. Capon, A. C. Horne, E. Kendy, M. E. McClain, N. L. Poff, B. D. Richter, and S. Ward. 2018. The Brisbane Declaration and Global Action Agenda on Environmental Flows (2018). Frontiers in Environmental Science 6.

Arthington, A. H., S. E. Bunn, N. L. Poff, and R. J. Naiman. 2006. The challenge of providing environmental flow rules to sustain river ecosystems. Ecological Applications 16:1311-1318.

Arthington, A. H., D. Tickner, M. E. McClain, M. C. Acreman, E. P. Anderson, S. Babu, C. W. S. Dickens, A. C. Horne, N. Kaushal, W. A. Monk, G. C. O'Brien, J. D. Olden, J. J. Opperman, A. G. Owusu, N. LeRoy Poff, B. D. Richter, S. A. Salinas-Rodríguez, B. Shamboko Mbale, R. E. Tharme, and S. M. Yarnell. 2024. Accelerating environmental flow implementation to bend the curve of global freshwater biodiversity loss. Environmental Reviews 32:387-413.

Cartwright, J., C. Caldwell, S. Nebiker, and R. Knight. 2017. Putting Flow–Ecology Relationships into Practice: A Decision-Support System to Assess Fish Community Response to Water-Management Scenarios. Water 9.

Davies, P. M., R. J. Naiman, D. M. Warfe, N. E. Pettit, A. H. Arthington, and S. E. Bunn. 2014. Flow–ecology relationships: closing the loop on effective environmental flows. Marine and Freshwater Research 65.

Dewson, Z. S., A. B. W. J. James, and R. G. Death. 2007. A review of the consequences of decreaed flow for instream habitat and macroinvertebrates. Journal of the North American Benthological Society 26:401-415.

Dudgeon, D. 2019. Multiple threats imperil freshwater biodiversity in the Anthropocene. Curr Biol 29:R960-R967.

Emanuel, B. (2019). Ensuring water security for people and nature: A status report from the upper flint river working group. American Rivers.

Enquist, C. A. F., S. T. Jackson, G. M. Garfin, F. W. Davis, L. R. Gerber, J. A. Littell, J. L. Tank, A. J. Terando, T. U. Wall, B. Halpern, J. K. Hiers, T. L. Morelli, E. McNie, N. L. Stephenson, M. A. Williamson, C. A. Woodhouse, L. Yung, M. W. Brunson, K. R. Hall, L. M. Hallett, D. M. Lawson, M. A. Moritz, K. Nydick, A. Pairis, A. J. Ray, C. Regan, H. D. Safford, M. W. Schwartz, and M. R. Shaw. 2017. Foundations of translational ecology. Frontiers in Ecology and the Environment 15:541-550.

Poff, N. L., R. E. Tharme, and A. H. Arthington. 2017. Evolution of Environmental Flows Assessment Science, Principles, and Methodologies. Pages 203-236 *in* A. Horne, J. A. Webb, M. J. Stewardson, B. D. Richter, and M. Acreman, editors. Water for the Environment. Academic Press.

Poff, N. L., and J. K. H. Zimmerman. 2010. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshwater Biology 55:194-205.

Power, M. E., M. S. Parker, and W. E. Dietrich. 2008. Seasonal reassembly of a river food web: floods, droughts, and impacts of fish. 78:263-282.

Richter, B. D. 2009. Re-thinking environmental flows: from allocations and reserves to sustainability boundaries. River Research and Applications 26:1052-1063.

Salzman, J. 2012. Drinking Water: A History. Overlook Duckworth, New York.

Tickner, D., J. J. Opperman, R. Abell, M. Acreman, A. H. Arthington, S. E. Bunn, S. J. Cooke, J. Dalton, W. Darwall, G. Edwards, I. Harrison, K. Hughes, T. Jones, D. Leclere, A. J. Lynch, P. Leonard, M. E. McClain, D. Muruven, J. D. Olden, S. J. Ormerod, J. Robinson, R. E. Tharme, M. Thieme, K. Tockner, M. Wright, and L. Young. 2020. Bending the Curve of Global Freshwater Biodiversity Loss: An Emergency Recovery Plan. BioScience 70:330-342.

USGS. 2018, Water Science School: Source and use of freshwater in the U.S. in 2015. https://www.usgs.gov/media/images/source-and-use-freshwater-us-2015#:~:text=The%20top%20row%20of%20cylinders,water%20used%20came%20from%20groundwater [accessed March 20, 2025].

Walters, A. W. 2016. The importance of context dependence for understanding the effects of low-flow events on fish. Freshwater Science 35:216-228.

Wood, J., and M. Freeman. 2017. Ecology of the macrophyte Podostemum ceratophyllum Michx. (Hornleaf riverweed), a widespread foundation species of eastern North American rivers. Aquatic Botany 139:65-74.

Yarnell, S. M., G. E. Petts, J. C. Schmidt, A. A. Whipple, E. E. Beller, C. N. Dahm, P. Goodwin, and J. H. Viers. 2015. Functional Flows in Modified Riverscapes: Hydrographs, Habitats and Opportunities. BioScience 65:963-972.

Zellmer, S. 2008. Legal Tools for lnstream Flow Protection. Pages 285-327 Integrated Approaches to Riverine Resource Stewardship: Case Studies, Science, Law, People, and Policy. The Instream Flow Council.

CHAPTER 2

EVALUATING FLOW-RELATED DRIVERS OF COVER AND GROWTH BY A SUBMERGED AQUATIC MACROPHYTE, *PODOSTEMUM CERATOPHYLLUM* [1]

[1] Rack, L. R., M. C. Freeman, S. K. McKay, R. M. Holdo, C. Conn, and S. J. Wenger. To be submitted to Freshwater Biology

Abstract

Macrophytes play important functional roles in freshwater ecosystems, from providing habitat to carbon and nutrient cycling. River discharge is an important driver of ecosystem processes, and macrophytes are both influenced by discharge and can alter hydraulic conditions. We conducted a field study in the Middle Oconee River, in Athens, GA, to investigate the flow-related dynamics of a submerged macrophyte, *Podostemum ceratophyllum*, which is widely distributed in rivers from the southeastern U.S.A. to southern Canada. From previous studies, we knew the plant could be vulnerable to grazing at low water velocities and to desiccation if exposed; however, other low flow mechanisms such as algal build-up or sedimentation could also drive plant survival or growth. Our objectives were to identify flow-related variables that could predict grazing, algal occurrence and accrual, and *Podostemum* growth rate. We found that hydraulic variables could predict occurrence of grazing and the accrual of algae once present, but we could not predict the onset of algal occurrence. The best predictor for growth rate was the proportion of algae or sediment covering *Podostemum*, rather than an individual hydrologic or hydraulic variable. Our findings help to understand how *Podostemum* may be vulnerable at low flows, in the context of a submerged macrophyte that could be a useful ecosystem indicator for water management.

Introduction

Macrophytes are an important part of aquatic ecosystems, contributing to carbon and nutrient cycling (Clarke 2002), influencing sediment dynamics (Clarke 2002, Cotton et al. 2006, (Gurnell 2013), providing habitat structure (Thomaz and Cunha 2010), and serving as a food source for grazing organisms (Bakker et al. 2016). Macrophytes can also restore water quality and clarity (Srivastava et al. 2008, Swe et al. 2021) and their integral role in freshwater

ecosystems have made them a useful indicator of ecological status (Poikane et al. 2018). These benefits extend to humans as well, through the supporting and regulating services as well as direct use as food and fiber, and spiritual and educational values (Thomaz 2021).

Because humans have significantly impacted the integrity of freshwater ecosystems, it is important to understand the vulnerabilities and drivers of macrophyte growth and persistence. In river ecosystems, streamflow is thought of as the dominant driver of many ecosystem processes that sustain aquatic organisms (Power et al. 1995). River macrophytes are impacted by river flows through effects on plant growth, dispersal, and biomass loss (Gurnell 2013). Light, temperature, nutrients, and carbon are drivers of growth in macrophytes, with specific needs that vary depending on the species. Senescence, herbivory, competition, burial, scour, and desiccation can result in partial or complete loss of a plant (Dietterich et al. 2024). Understanding what factors may be limiting for plants or which flow-mediated drivers have the greatest impact on plant growth and loss is important for predicting plant dynamics under future flow conditions altered by changes in climate and management.

Podostemum ceratophyllum, hereafter *Podostemum*, is a submerged macrophyte that is broadly distributed in rivers of the eastern U.S.A. and southern Canada, with populations as far south as Central America. *Podostemum* grows in swift flowing water on stable substrates (Philbrick and Novelo 2004) and can be abundant in shoal ecosystems in the southeastern US. (Nelson and Scott 1962, Everitt and Burkholder 1991, Grubaugh and Wallace 1995). The plant provides important habitat for river biota and may be associated with higher abundances of invertebrates and fishes than non-vegetated substrates (Nelson and Scott 1962, Hutchens et al. 2004, Argentina et al. 2010). While the plant is tolerant to some emersion, studies have demonstrated that it is vulnerable to low flows, drying, and desiccation (Pahl 2009). Previous

studies suggest plant dynamics are similar to those of bryophytes in the conceptual model by Suren and Riis (2010), in which stable substrates and higher velocity lead to a dominance of the plant whereas increased duration of low flows lead to a decline in plant biomass. Pahl's 2009 study of *Podostemum* during a severe drought, which dried portions of a southeastern river shoal, showed that the plant persisted in wetted areas but had biomass reduced by an order of magnitude compared with non-drought studies in the same river (Pahl 2009). Pahl further related month-to-month loss in biomass to exposure of the plant to water depths less than 5 cm. Subsequent research by Wood et al. (2019) implicated reduced water velocity that facilitated grazing by large-bodied herbivores as a driver of *Podostemum* loss. We are interested in exploring the mechanisms of how seasonally low and extended low-flow periods affect the plant's growth.

Podostemum's apparent vulnerability to declining streamflows and its importance as a producer and as habitat suggest that the plant could serve as an ecologically meaningful sentinel for streamflow management. However, managers and ecologists will be better able to predict responses of *Podostemum* or similar fluvial macrophytes to reduced streamflow if we understand the mechanisms of flow effects. We know that *Podostemum* is vulnerable to desiccation and potentially enhanced grazing at lower water velocities, but we do not understand the effects of declining streamflow on growth suppression. To investigate plant loss as streamflow declines, we tracked changes in *Podostemum* cover and plant growth on individual rocks, measured at three-week intervals across a 5-month, low-flow season, in rocky shoal habitats of a southeastern US river. Our specific objectives were to identify flow-related variables that could predict (1) plant removal by grazers, (2) overgrowth by epiphytic algae, and (3) variation in plant growth rate, during low-flow conditions. Our results help identify mechanisms relating declining

discharge to reduced *Podostemum* cover and growth, with relevance to defining macrophyte-flow relations for use in water management.

Methods

Study Area

We conducted our study in the Middle Oconee River, a sixth-order Piedmont river in Georgia, U.S.A, near Ben Burton Park, Athens, GA (33.961430, -83.441167). *Podostemum* grows abundantly on rocky substrates at this site, which has also been used in previous studies of *Podostemum* (Nelson and Scott 1962, Grubaugh and Wallace 1995, Pahl 2009, Wood et al. 2019). The Middle Oconee River has a legacy of high sediment loads from past land use (Jackson et al. 2005) and drains approximately 1010 km^2 at this site, with upstream land use comprising about 43% forest (deciduous, evergreen and mixed), 24% agriculture (including pasture and crops) and 27% developed (developed open space and low-, medium, and high-intensity development; USGS Streamstats, accessed online 27 February 2025). Mean annual discharge is approximately 500 cfs (14 m^3/s, water years 1929-2021, USGS Gage 02217500, located approximately 2.5 km downstream from the study site), with seasonally lower flows from July through October (289 cfs, 8.2 m^3/s, 4-month mean).

We selected two contrasting shoal locations at the study site to diversify the hydraulic conditions for our measurements of plant growth. The river channel at the upper shoal site is about 100 meters wide, with a substrate consisting of extensive bedrock, large boulders, cobble, and sand. The upper shoal location is the site of a former mill dam, and hydraulic conditions vary along the width of the river, with multiple channels during low flow conditions. The channel at the lower shoal site is about 30 meters wide and incised between banks that are over 3 meters high. The lower site has a well-defined thalweg, relatively uniform hydraulics, and substrate

primarily composed of cobble, gravel, and sand with scattered boulders. The shoal sites are separated by about 0.6 km.

Study Design

To evaluate the drivers of *Podostemum* growth, we tagged and revisited individual rocks colonized by the plant (herein referred to as rocks) from June to November 2022. We deployed 20 rocks in each of the two shoal locations (40 total). We initially trimmed stems in a 10x10 cm patch on one side of the rock to provide a standardized set of stems for estimating growth-rate with minimum effect of density dependence (e.g., slower growth induced by shelf shading as leaves mature). We attempted to cut stems within the patch to a length of approximately 5 mm and measured 10 to 13 of the trimmed stems to the nearest mm with a metal ruler to quantify this starting length. We photographed and tagged each rock with a unique combination of colored zip-ties. We placed the rocks haphazardly throughout each shoal site with the intent of capturing a range of velocity conditions. We measured the depth with a wading rod and velocity at 60% of depth with an electronic current meter at each rock placement. In the first deployment, we placed 10 rocks each in the upper and lower shoals and revisited the rocks one week later (Table 2.1). For each rock on each date, we measured up to 13 of the previously trimmed stems, re-measured depth and velocity at the rock location at the time of sampling, and then re-photographed and replaced the rock in the same location. We determined a 1-week interval was too short to capture changes in stem length, so for the remainder of the study we revisited rocks approximately every three weeks and measured up to 13 stems, depth, velocity, and photographed each rock.

To maintain a standardized set of stems for estimating growth rate, we re-cut stems approximately every 6 weeks or on the 2nd visit (Table 2.1). We measured and photographed the rocks before re-cutting and then measured and photographed again after cutting the stems. If a

rock had a buildup of algae or sediment covering the stems, we photographed the rock as it appeared when retrieved from the river and then rinsed and re-photographed the rock and took the stem measurements. Rocks were deployed at three different events. The first 20 rocks were deployed on the same date (June 29, 2022), with 10 in the upper shoal and 10 in the lower shoal. Six weeks later (August 10th, 2022), rocks 21-30 were deployed in the upper shoal and three weeks after that (September 1st, 2022), rocks 31-40 were deployed in the lower shoal (Table 2.1).

Image analysis

We used data from the photographs of tagged rocks on each sample date to evaluate evidence for the flow-mediated drivers of *Podostemum* growth rate. We processed all images in ImageJ version 1.54h (Rasband 2018). We analyzed photos of rocks from the initial deployment, from rocks before re-cutting, and from before and after rinsing rocks that were visually covered in sediment or algae. We thus analyzed up to two photographs for any rock on a single date.

For each photo we started by overlaying a grid (18,000 pixels^2). We selected the grid size that was small enough to capture a dominant cover type in a cell but was not too cumbersome for manual labeling. We used the 'multi-point' tool in ImageJ to assign a colored dot in each cell on the rock based on a cover category. We used 7 cover categories: leaf, stem, rock, algae, sediment, grazed (i.e., to the surface of the rock), and cut. Leaf was assigned for the leaf portion of the plant and stem was used for regrowth, runners, or very long stems without leaves. The category "rock" was used when bare rock was exposed with no plant growth. "Sediment" and "algae" were assigned when they occurred in a cell, covering either rock or plant. When sediment was trapped in algae, we labeled it algae. "Grazed" was used when there were clear signs of grazing on a rock by geese or other herbivores that reduced the stems to essentially absent with bare rock remaining. The "cut" category was used to signify when we were labeling the cut side of the

rock. We saved each color as an overlay and exported the overlay to a *csv.* file once we completed classifying the rock. We calculated the proportion of each cover category by dividing the number of cells for an individual category by the total number of cells categorized on a rock. To determine how many grid rows (horizontal) we needed to label on each photograph to accurately assess the cover categories on each rock, we labeled all cells on 10 rocks with varying types of cover and photos taken at different heights. We then compared the estimated proportion of cover for the seven categories based on alternative labeling schemes (every other cell, every other grid-row, every 3rd grid-row, etc.) and compared to the proportions we calculated based on all cells classified. We selected the most accurate and efficient method, which was labeling every cell on every third grid-row (horizontal) and analyzed the remaining photos using the 3rd grid-row method. We labeled some cells with more than one cover category, however we used another color to signify when cells were double labeled, so that we could still have a total cell count for each rock.

Model covariates

We calculated values for two shoal-wide covariates to estimate effects of hydrology and hydraulic conditions on *Podostemum* condition and growth during each interval between measurements (Table 2.2). We used gage data from USGS gage 02217500 (USGS 2025) to calculate the mean daily discharge 3 weeks prior to deployment for each set of rocks and for the intervals between subsequent measurements. We also used the velocity threshold of 0.4 m/s, hypothesized by Wood et al. (2019) to inhibit grazing on *Podosemtum*, to evaluate an effect of shoal-wide velocity conditions on plant growth, algal accumulation, and occurrence of grazing. For this covariate, we used three datasets to estimate the proportion of each shoal site with water velocities less than 0.4 m/s ("low shoal velocities") based on the mean daily discharge during the

interval. For the upper shoal site, we used data collected by Katz (2009) and Pahl (2009), who measured mean water-column velocity at 8-10 randomly chosen locations along a single 100-m transect across the river on 10 dates between December 2007 and October 2008 (total n= 92; range of mean daily discharge = 21-274 cfs). We combined these data with our velocity measurements taken at 1-m intervals along the same 100-m transect on 12 dates between June and December 2021 (total n= 1095; range of mean daily discharge = 115-296 cfs). We coded velocities as 1 if less than 0.4 m/s, and 0 otherwise, and fit a binomial general linear model (GLM) with a logit-link to estimate the proportion of the shoal with velocities below 0.4 m/s in relation to river discharge. We repeated this analysis for the lower shoal using data from C. Conn (UGA; unpublished), who measured mean water-column velocities at an average of 15 hard-substrate locations along 3 transects in the lower shoal on 16 dates between June 2016-November 2017 (total n= 213; range of mean daily discharge = 46-321 cfs). Finally, for models of algal occurrence (below), we repeated the logit-regressions to estimate proportion of each shoal with velocities < 0.20 m/s and <0.60 m/s.

We calculated rock-specific covariates using date- and rock-specific measurements of mean water-column velocity and water depth. We averaged point measurements of water-column velocity measurements for all dates to calculate rock-specific mean velocities, which represented the average velocity at a given rock during the flows when rocks were sampled. While velocity conditions likely change non-linearly as flow changes, we placed rocks across a range of lower and higher velocities which was represented through the rock-specific mean velocity. As an alternative to this average velocity condition, we also used date- and rock-specific velocity measurements and Froude number (calculated from depth and velocity measurements) as model covariates representing hydraulic conditions. For models of plant growth, we also included rock-

and date-specific algal cover and total cover (algae and sediment cover combined), estimated from the image analysis and expressed as proportion of the rock grid cells with algae present and with either algae or sediment present.

Models

We developed a series of candidate models using a Bayesian framework to estimate the effects of covariates on 1) the probability of occurrence for grazing events or algal presence, 2) the proportional cover by algae when present on a rock, and 3) plant growth rates. We tested a null model and covariates individually and in pairs of shoal-wide condition (i.e., discharge or estimated proportion of the shoal with velocity <0.40 m/s) and rock-specific hydraulic variables. All models included random effects for rock and interval (date) to account for repeated measurements on individual rocks and within intervals. Because values for all covariates had similar ranges (mostly 0 to about 1, ranging as high as 4.6 for discharge as cfs divided by 100), and to improve model interpretability, we did not further standardize covariates.

Occurrence of grazing and algae

We used binomial generalized linear mixed models (GLMM) to estimate the probability of grazing and algal occurrence with data from the image analysis. We compared models with up to two covariates to a null model:

$$Y_{r,i} \sim Bernoulli(x_{r,i}); logit(x_{r,i}) = \beta_0 + \varepsilon_r + \varepsilon_i$$

where $Y_{r,i}$ is coded as 1 if rock *r* was grazed or, in the algae occurrence model, had any visual algal cover on date *i*. For models of both variables, β_0 is an intercept and ε_r and ε_i are normally-distributed random effects with mean 0 for rock and interval identities.

For occurrence of grazing by macroconsumers (likely Canada Geese), we tested the hypothesis that grazers are excluded from higher velocity areas (Wood et al. 2019). We ran

candidate models with proportion of shoal < 0.4 m/s, rock- and date-specific velocity, mean rock velocity, and Froude number individually. Then we modeled grazing occurrence with the proportion of shoal < 0.4 m/s and either rock- and date-specific velocity or mean rock velocity to see if a combination of shoal-wide with individual rock conditions predicted grazing.

For visible occurrence of algae on rocks, we considered that flow can exert physical controls over algal growth and persistence through drag force, light attenuation (depth), and nutrient delivery. At the same time, once algal blooms begin, they may spread throughout the entire shoal. We selected the proportion of shoal <0.40 m/s to represent shoal-wide conditions, and Froude number to characterize rock-specific local hydraulic conditions. We tested these candidate models with covariates individually and together. In addition, we also tested the sensitivity to different velocity thresholds using the estimated proportion of the shoal with velocity < 0.20 m/s, which could facilitate algal accumulation, or proportion of the shoal with velocity <0.60 m/s, assuming higher velocities could limit algal accumulation.

Algal cover

We used a beta regression to estimate the proportion of algal cover on a rock when algae were present with data from the image analysis. We compared models to the null model with random effects, shown below:

$$Y_{r,i} \sim Beta(p_{r,i}, q_{r,i})$$
$$p_{r,i} = \mu_{r,i} * \tau;\ q_{r,i} = (1 - \mu_{r,i}) * \tau$$
$$logit(\mu_{r,i}) = \beta_0 + \varepsilon_r + \varepsilon_i$$

where $Y_{r,i}$ is the proportion of algal cover on rock *r* at the end of interval *i;* $p_{r,i}$ and $q_{r,i}$ are shape parameters of the beta distribution, and $\mu_{r,i}$ and τ are the mean and precision of estimated proportion. We expected that once algae were present on a rock, microhabitat variables may exert the most direct control on how much algae can accumulate. We used the same combination of

covariates for candidate models as the algal presence model, i.e., proportion of shoal with velocities <0.4 m/s and rock-specific Froude number, individually and combined.

Plant growth

We used an exponential model to estimate plant growth rate during sampling intervals, using the trimmed stem measurements on each sampling date. We used the 7 to 13 stem measurements on each rock and date to estimate the mean stem length for the growth model. We compared models to the null model with random intercepts for rock and interval, shown below:

$$stem_{k,j,i} \sim Lognormal(\mu_{k,j,i}, \tau)$$
$$\mu_{k,j,1} = \beta_{0\,j}$$
$$\mu_{k,j,i} = \mu_{[k,j,(i-1)]} + r_{k,i,j} * days_{j,(i-1)}$$
$$r_{k,j,i} = \beta_1 + \varepsilon_r + \varepsilon_i$$

where $stem_{k,j,i}$ is the k^{th} stem length on rock j in interval i; $\mu_{k,j,1}$ is the latent mean length of stem k on rock j on the date that stems were trimmed, and $\mu_{k,j,i}$ is the mean length of stems on subsequent dates. The daily growth rate, $r_{k,j,i}$ was modeled as a function of covariates hypothesized to reduce growth. We ran candidate models with individual covariates for the proportion of algal cover and total cover (algae or sediment) on each rock on each date, which represented the plant being covered. We then assessed the impact of hydrologic and hydraulic variables by running separate models with shoal-wide covariates, proportion of shoal with velocities <0.40 m/s and mean discharge, and rock- and date-specific velocity and Froude number to see if there were measurable, direct effects of flow on plant growth.

We fit all models using JAGS software (Plummer, 2003) in RStudio (Posit team 2024), with the *rjags* package (Plummer 2014). All models were run with 3 chains for 8000 iterations, with the first 1000 discarded and retained every 3rd iteration to minimize temporal dependence. We compared models using DIC. We used uninformative priors for model parameters in the

binomial GLMMs and beta regression. For the model of plant growth, we used the global average stem length on the first date as the prior for the average stem length for each rock on the first date and uninformative priors for remaining model parameters. We assessed model convergence using the Gellman-Rubin statistic and by inspecting trace plots of chains for estimated parameters.

Results

Of the 40 rocks deployed, we excluded four rocks that were lost after the first visit, leaving 36 rocks for the analysis that we revisited 3-7 times depending on deployment date and whether they were found each visit. We visited rocks a total of 199 times across all rocks, including deployment dates. We analyzed 197 photos of rocks for the cover analyses. There were two rocks at two dates excluded from analysis due to a missing photo or issue with the photo angle. For the grazed dataset, we analyzed the rinsed photos (if multiple photos were taken); we analyzed the pre-rinsed photos for the algae dataset.

Plant cover (including leaves and stems) on rocks did not vary substantially over the study period, ranging from 79% to 97% across all sampling intervals. In contrast, we observed changes in leaf cover throughout the study. Most new growth on cut stems came from leaf growth. Leaf cover varied among rocks and between dates. For example, the percentage of leaf cover on the first date for rocks 1-20 averaged 45% but ranged from 10% to 80%. Leaf cover declined consistently in the upper shoal after September, but patterns in the lower shoal were less obvious (Figure 2.1). Sediment accumulation was observed on 18 of 36 rocks and occurred throughout the study period. Six rocks had sediment present on three or more visits. The minimum sediment cover was 0.8% with a maximum of 87% cover.

There were 14 grazing events on 10 rocks, with 9 located in the upper shoal. Grazing occurred throughout the study period, with the first evidence observed on July 29th, and resulted in the total loss of the plant, exposing bare rock. Figure (2.2) shows one of the most extensive grazing events we captured on a rock.

Algae occurred on 33 of 36 rocks and was first detected on September 1st (Figure 2.3). Based on the rocks with algae present, the total mean algal cover was 60% and ranged from 5% to 100%. Figure 2.4 shows examples of high algal cover on two rocks, with 82% cover on rock 5 on September 29th and 96% cover on rock 35 on September 17th.

Model covariates

Average daily discharge ranged from 98 cfs to a max of 458 cfs during intervals prior to or between rock observations (Table 2.3). The average discharge for July-November, 243 cfs, was 16% lower than the long-term average for this four-month period, although average discharge for October 2022 was 63% lower than the long-term October average. The wider channel in the upper shoal compared with the lower shoal was reflected in the relationships we developed to estimate the proportion of shoal with velocity < 0.40 m/s (Figure 2.5). The average proportion of the upper shoal < 0.40 m/s during the study period was 0.63 with a maximum of 0.81 compared with an average of 0.33 with a maximum of 0.51 in the lower shoal (Table 2.3). The mean rock-specific velocity across all sampling intervals ranged from 0.05 m/s to 1.1 m/s (Figure 2.6). Froude number calculated for each rock on each date ranged from -0.06 (i.e., an upstream eddy current) to 0.99 (Figure 2.7).

All models converged based on trace plots and Gelman-Rubin statistic ($\hat{R}$<1.1) for model parameter estimates.

Occurrence of grazing and algae

The best-supported model for grazing occurrence included Froude number. The probability of grazing decreased with an increase in Froude number; the parameter estimate did not cross zero (Table 2.4, Figure 2.8).

The best supported model for the probability of algal occurrence included predictors for both the proportion of the shoal with velocities < 0.40 m/s and Froude number. The probability of algal occurrence increased with a larger proportion of the shoal < 0.40 m/s and a higher Froude number (Table 2.5; Figures 2.9 & 2.10). However, the credible intervals around both parameter estimates crossed zero (Table 2.5). We also compared alternative shoal-wide velocity thresholds for the probability of algal occurrence given that 0.40 m/s was established based on a grazing threshold for *Podostemum* (Wood et al. 2019). We found that the model fit was similar to that of the best-supported model (Table 2.5) and that the changes in the proportion of shoal habitat with water velocities below the three thresholds (< 0.4, 0.6, and 0.2) changed similarly in relation to discharge in the upper and lower shoals (Appendix A; Figure A1).

Algal cover

The best model for the proportion of algal cover, when algae was present, included the same parameters as the model for algal occurrence: the proportion of the shoal with velocities <0.40 m/s and Froude number. Algal cover increased with an increasing proportion of shoal < 0.40 m/s, but unlike algal occurrence, decreased with increasing Froude number (Table 2.6). The credible interval for the proportion of the shoal with velocities <0.40 m/s did not cross zero, but the credible interval for the effect of Froude number did include zero. The predictions from the algal cover model are more precise compared to the predictions of algal occurrence (Figures 2.11 & 2.12).

Plant growth

The best supported model for predicting the growth rate of *Podostemum* included a single covariate, the proportion of algal and sediment cover on a rock. Algae occurred more often on rocks once the bloom began, but sediment was present consistently on a few rocks, with some rocks having both algal and sediment cover (Appendix A; Figure A2). Plant growth rate decreased as algal and sediment cover increased and confidence intervals around the parameter estimate did not cross zero (Table 2.7, Figure 2.13). Estimated growth rate in the absence of algal or sediment cover was about 2.5 to 2.8% per day (Table 2.7). Complete cover by algae and sediment reduced growth rate by an estimated 61%, to about 1.1% per day. Algal cover alone similarly depressed growth rate, however credible intervals for effects of discharge, proportion of shoal with water velocities <0.4 m/s, rock-specific water velocity or Froude number all included zero (no effect; Table 2.7).

Discussion

Our investigation of *Podostemum* dynamics on individual rocks in a southeastern US Piedmont river revealed algal overgrowth and sediment deposition, along with grazing by large-bodied herbivores, as primary drivers of plant loss or reduced growth during seasonally low flows. Grazing events resulted in total removal of the plant on portions of about 27% of our tagged rocks and were more likely to occur as flow conditions resulted in lower water velocities. Overall, we found that plant cover (i.e., leaves and stems) did not change on individual rocks over our 5-month study period as much as leaf cover alone, which varied substantially among rocks and across sampling dates. Leaf cover notably declined in the lower-velocity portion of our study area when algal growth began to appear on rocks, midway through our study. Our model did not predict the onset of the algal bloom well; however, once present, algal cover was higher

on rocks with a lower Froude number and when a higher proportion of shoal water velocities were less than 0.40 m/s. In turn, *Podostemum* growth declined with increasing cover by algae and sediment. We estimated stems grew at about 2.5% per day, which was in line with previous estimates of growth rate (Pahl 2009, Wood et al. 2019), but growth declined by up to 61% as algae or sediment increased to cover the entire rock.

The use of photos and repeated observations of trimmed stems allowed us to capture patterns and trends in *Podostemum* growth that would have been difficult to interpret with measurements of untrimmed stems alone. We knew from preliminary studies of the plant (Rack unpublished) that stem length is a poor predictor of areal biomass without stem density. However, using non-destructive sampling methods allowed for repeated measures on the same rocks as habitat conditions varied, and became particularly important for characterizing the onset of the algal bloom in the shoals and quantifying the impact on stem growth. Our study benefited from the ability to remove the rock from the water to photograph *Podostemum* cover and condition. Pairing cover data with the stem measurements allowed us to assess the dynamics for the whole rock and test for mechanisms controlling growth and cover on the rocks.

Grazing is not often the focus of river macrophytes studies (Bakker et al. 2016); however, we observed grazing events that resulted in complete removal of *Podostemum*. Grazing occurrence was patchy but occurred mostly in the upper shoal, likely due to a greater proportion of the shoal velocity <0.40 m/s across flow levels. Grazing was more likely to occur under lower Froude number, driven by low velocity, i.e., the type of habitat that was more accessible to grazers hypothesized to include turtles, crayfishes, waterfowl, deer, muskrats, and beavers (Wood et al. 2019). In the study by Wood et al. (2019) stems were not always grazed down to bare rock, leading us to believe we observed one type of grazer, Canada goose (*Branta canadensis*). We

observed groups of a dozen or more geese grazing on *Podostemum* in our upper study site on multiple sample dates, but not in our lower site where water velocities were higher. Grazing can have large impacts on plant abundance and production (Bakker et al. 2016, Wood et al. 2017). The importance of grazing in regulating *Podostemum* biomass will likely vary depend on the identity and abundance of herbivores.

The conditions that can lead to agal proliferations are well studied, but it is difficult to predict the onset of an algal bloom (Biggs et al. 1998, Power et al. 2008, Suren and Riis 2010, Glibert et al. 2018). In the absence of light limitation, the combination of low, stable flows and high nutrients can result in algal blooms (Biggs and Price 1987, Suren et al. 2003). In addition, a release from grazing pressure can also contribute to algal proliferation (Biggs 1996, Power et al. 2008). We captured an algal bloom that began in September. Algal cover, primarily filamentous green algae and diatoms remained high in our upper shoal, but we observed a decline in algae in the lower shoal by the end of the study. Hydrologic predictors alone were not sufficient for predicting the onset of the bloom in our study but were useful for predicting algal cover on rocks once it was present. Our observation that algal cover tended lower on rocks situated in relatively higher velocities may not be general (Biggs and Price 1987, Biggs 1996, Suren et al. 2003, Power et al. 2008). For example, the relationship between Froude number and algae can depend on the type of algae present (Tonetto et al. 2014). In addition, different types of algae tend to be more successful at different temperature ranges, with diatoms tending to dominate at lower temperatures and cyanobacteria being competitive at higher temperatures (Davis et al. 2009). Flow effects on algal growth could also be indirect, e.g., through nutrient enhancement by excretion by large-bodied *Podostemum* grazers.

Algal and sediment cover were strong predictors and had a negative effect on *Podostemum* growth rate during our study. Others have also shown that algae can have negative impacts on macrophyte cover or biomass (Wilby et al. 1998, Wade et al. 2002). Algae could have affected *Podostemum* growth in a few ways: smothering, shading (Arthaud et al. 2012), direct competition for nutrients (Xie et al. 2013, Zhang et al. 2019, Kaijser et al. 2021) or a combination of these. We observed extensive algal cover on *Podostemum* that may have impeded CO_2 uptake or could have resulted in light limitation. We were not able to distinguish between potential physiochemical and physical factors that drove this relationship, but we observed some recovery by *Podostemum*, based on increasing proportion of leaf and decreasing proportion of algae, in the final dates in our lower shoal. Although we did not find a strong relationship between hydraulic variables and *Podostemum* growth, we did find that hydraulics could predict algal accumulation, and that algae and sediment cover was the most important low-flow driver of *Podostemum* growth.

Submerged aquatic macrophytes are good candidates for assessing effects of streamflow alteration because they are responsive to changes in flow conditions and ecologically important. In addition, the ability to signal potential challenges across shoal ecosystems during low flows would make them a good indicator for management purposes, as it provides a single quantifiable target (Pearson 1994, Burger 2006). Macrophytes provide habitat structure and support ecological processes in river ecosystems. Macrophytes mediate flow at the microhabitat scale though slowing current velocity, which can increase sedimentation (Gregg and Rose 1982, Madsen et al. 2001), provide habitat for filter-feeding macroinvertebrates and refugia for juvenile fishes (Hutchens et al. 2004). Macrophytes also play an important role in nutrient retention and cycling in river ecosystems (Clarke 2002). *Podostemum* in particular has been referred to as a

foundation species by Wood and Freeman (2017), due to its high biomass and productivity (Nelson and Scott 1962, Grubaugh and Wallace 1995), habitat importance for aquatic macroinvertebrates invertebrates and fishes (Hutchens et al. 2004, Argentina et al. 2010), and contributions to nutrient cycling (Nelson and Scott 1962, Wood and Freeman 2017). These attributes make *Podostemum* a promising indicator species for low river flows as it could also represent cascading effects with the loss of the plant.

Most decisions around water management in rivers are made in terms of water quantity or river discharge, so linking ecological responses to changing river flows is important for developing management-relevant information. Macrophytes have been used in numerous locations as ecosystem-level monitoring indicators due to their sensitivity to water quality conditions, provisioning of habitat structure, and potential to interact with algae, aquatic invertebrates, and fish populations (Kuhar et al. 2011, Umetsu et al. 2018, Szoszkiewicz et al. 2019, Bytyci et al. 2022). Macrophytes can be relatively easy to measure since they have limited dispersal ability to respond to changing flow conditions and may respond directly or indirectly to changes in flow conditions (Lázár et al. 2016). Identifying a mechanistic basis for how discharge drives macrophyte cover, abundance, or growth provides an opportunity to develop robust and management-relevant ecological information for water management.

Table 2.1 The deployment and sampling schedule for tagged rocks with *Podostemum* used to measure plant growth, June to November 2022. Stems on half of each rock were trimmed at deployment (D). Then rocks were visited every 3-weeks, except for the first visit one week later for rocks 1-20. At each visit stems were measured (M) and rocks were photographed and every 6-weeks stems were also re-cut, re-measured, re-photographed (R).

Rocks	June 29th	July 6th	July 29th	Aug 10th	Aug 16th	Sept 1st	Sept 8th	Sept 17th	Sept 28th	Oct 11th	Oct 18th	Nov 9th
1-20	D	M	M		R		M		R		M	M
21-30				D		M		R		M		M
31-40						D		M		R		M

Table 2.2. Covariates for modeling occurrence grazing, algal cover and plant growth of *Podostemum* on 36 tagged rocks in the Middle Oconee River, July-November 2022. Model inclusion (AC, algal cover; AO, algal occurrence; G, Grazing occurrence; SG, plant growth) and abbreviations used to report model results are listed for each covariate.

Covariate	Abbreviation	Models that included covariate
Mean daily discharge (cfs) over sampling interval, scaled by division by 100	discharge	SG
Mean estimated proportion of shoal with velocities <0.4 m/s	prop.40	G, AO, SG
Mean estimated proportion of shoal with velocities <0.2 m/s	prop.20	AO
Mean estimated proportion of shoal with velocities <0.6 m/s	prop.60	AO
Rock-specific mean water velocity	mean.velocity	G, AC, SG
Rock- and date-specific water velocity	rock.velocity	G, SG
Rock- and date-specific Froude number	Froude	G, AO, AC, SG
Proportion of rock with algal cover	algae	SG
Proportion of rock with algal or sediment cover	total cover (algae+sediment)	SG

Table 2.3. Time intervals and dates pre- and during deployment of tagged rocks in the Middle Oconee River near Athens, GA, July-November 2022. For each interval we calculated the mean discharge, the estimated proportion of the upper and lower shoal locations with velocities < 0.40 m/s and listed the rocks deployed or measured at the end of each interval. Rocks 1, 16, 18, and 20 were lost after the first or second visit and excluded from the study. Additional rocks between 1-20 were either lost or not found during a visit over the sampling period. We did not need a value for lower shoal < 0.40 m/s for interval 4 since rocks 21-30 were deployed in the upper shoal and the following interval we deployed rocks 31-40 in the lower shoal.

Interval	Date range	Mean discharge (cfs)	Upper shoal Proportion <0.40 m/s	Lower shoal Proportion <0.40 m/s	Rock numbers deployed or measured at end of interval
1	June 9th-29th	174	0.70	0.36	Deployed 1-20 on June 29
2	June 29th – July 6th	179	0.69	0.34	1-17;19
3	July 6th- 29th	458	0.38	0.14	2-15; 17; 19
4	July 18th- Aug 10th	311	0.50	NA	Deployed 21-30 on August10
5	July 29th – Aug 16th	258	0.56	0.22	2,3,5-10,12,13,15
6	Aug 10th – Sept 1st	231	0.61	0.26	21-30 Deployed 31-40 on September 1
7	Aug 16th – Sept 8th	268	0.55	0.24	2,3,5-10,12-15
8	Sept 1st – 17th	299	0.50	0.20	21-40
9	Sept 8th - 28th	182	0.69	0.34	2,3,5-10, 12-15
10	Sept 17th – Oct 11th	109	0.80	0.49	21-40
11	Sept 28th – Oct 18th	98	0.81	0.51	2,3,5-15
12	Oct 11th – Nov 9th	140	0.75	0.42	21-40
13	Oct 18th – Nov 9th	147	0.74	0.41	2,3,5-15

Table 2.4. Model comparisons for the occurrence of *Podostemum* grazing on tagged rocks in the Middle Oconee River, July-November 2022. Estimated parameter means (mu) and 95% Bayesian credible intervals on the logit scale are shown for model intercept (b0) and covariates (abbreviated as in Table 2.2) along with DIC for each model. All models include random effects for rock identity and for time interval. The best-supported model is highlighted in bold.

Model covariates	mu	2.5	97.5	DIC
b0	b0: -2.9363	-3.7472	-2.2117	114.8
b0 + b1*prop.40	b0: -4.6007 b1: 3.0327	-7.1016 -0.5072	-2.5247 6.9449	113.2
b0 + b1*rock.velocity	b0: -2.2809 b1: -2.1967	-3.3941 -5.0921	-1.3448 0.2113	113.2
b0 + b1*mean.velocity	b0: -2.0023 b1: -3.0770	-3.2170 -6.7368	-0.9047 -0.0604	112.0
b0 + b1*froude	**b0: -1.9708** **b1: -4.6808**	**-3.0353** **-9.1561**	**-1.00336** **-0.0975**	**108.9**
b0 + b1*prop.40 + b2*rock.velocity	b0: -3.7575 b1: 2.4351 b2: -1.7681	-6.4936 -1.2504 -4.7241	-1.3146 6.3407 0.7089	113.2
b0 + b1*prop.40 + b2*mean.velocity	b0: -3.4859 b1: 2.3993 b2: -2.5662	-6.2433 -1.2233 -6.1424	-1.0690 6.3836 0.3322	111.4

Table 2.5. Model comparisons for the occurrence of algae on tagged rocks in the Middle Oconee River, July-November 2022. Estimated parameter means and 95% Bayesian credible intervals on the logit scale are shown for model intercept (b0) and covariates (abbreviated as in Table 2.2) along with DIC for each model. All models include random effects for rock identity and for time interval. The best-supported model is highlighted in bold.

Model covariates	mu	2.5	97.5	DIC
b0	b0: -1.5740	-6.2535	2.6466	129.0
b0 + b1* prop.40	b0: -2.9721 b1: 2.9396	-8.0503 -0.9832	1.5226 7.6449	126.6
b0 + b1*Froude	b0: -1.7387 b1: 1.1873	-6.2305 -1.4576	2.4911 3.9360	129.0
b0 + b1* prop.40 + b2*Froude	b0: **-4.0018** b1: **3.7704** b2: **1.9349**	**-9.4731** **-0.4006** **-0.9103**	**0.9922** **8.4906** **5.0208**	**124.2**
b0 + b1* prop.60 + b2*Froude	b0: -4.5831 b1: 3.3358 b2: 1.9440	-10.7450 -0.4383 -0.8863	0.6998 7.8204 5.1064	127.3
b0 + b1*prop.20 + b2*Froude	b0: -3.6556 b1: 5.7653 b2: 1.9698	-8.7086 -0.4372 -0.8988	0.9036 13.0558 5.0216	125.0

Table 2.6. Model comparisons for the proportion of algae on tagged rocks, when algae was present, in the Middle Oconee River, July-November 2022. Estimated parameter means and 95% Bayesian credible intervals on the logit scale are shown for model intercept (b0) and covariates (abbreviated as in Table 2.2) along with DIC for each model. All models include random effects for rock identity and for time interval. The best-supported model is highlighted in bold.

Model covariates	mu	2.5	97.5	DIC
b0	b0: 0.2250	-0.5307	0.9838	23.8
b0 + b1 * prop.40	b0: -0.9713 b1: 2.1086	-2.1103 0.6295	0.1308 3.6500	1.1
b0 + b1 * Froude	b0: 0.4390 b1: -1.0098	-0.4411 -2.0065	1.2022 -0.6668	8.3
b0 + b1 * prop.40 + b2*Froude	**b0: -0.6387** **b1: 1.8351** **b2: -0.7443**	**-1.8491** **0.2930** **-1.7210**	**0.5579** **3.3923** **0.2583**	**-5.4**

Table 2.7. Model comparisons for the plant growth rate (*r, day^-1)* on tagged rocks in the Middle Oconee River, July-November 2022. Estimated parameter means and 95% Bayesian credible intervals are shown for model intercept (b0) and covariates (abbreviated as in Table 2.2) along with DIC for each model. All models include random effects for rock identity and for time interval. The best-supported model is highlighted in bold.

Model covariates	mu	2.5	97.5	DIC
b1	b1: 0.0223	0.0101	0.0376	11980.2
b1 + b2*algae	b1: 0.0252 b2: -0.0108	0.0166 -0.0171	0.0343 -0.0047	11972.4
b1 + b2*total cover (algae+sediment)	**b1: 0.0279 b2: -0.0169**	**0.0177 -0.0239**	**0.0377 -0.0098**	**11963.8**
b1 + b2*discharge (/100)	b1: 0.0108 b2: 0.0045	-0.0090 -0.0032	0.0287 0.0147	11980.1
b1 + b2*prop.40	b1: 0.0292 b2: -0.0139	0.0147 -0.0306	0.0408 0.0036	11977.9
b1 + b2*velocity	b1: 0.0231 b2: -0.0028	0.0148 -0.0105	0.0315 0.0047	11979.5
b1 + b2*Froude	b1: 0.0210 b2: -0.0011	0.0101 -0.0115	0.0306 0.0089	11983.9

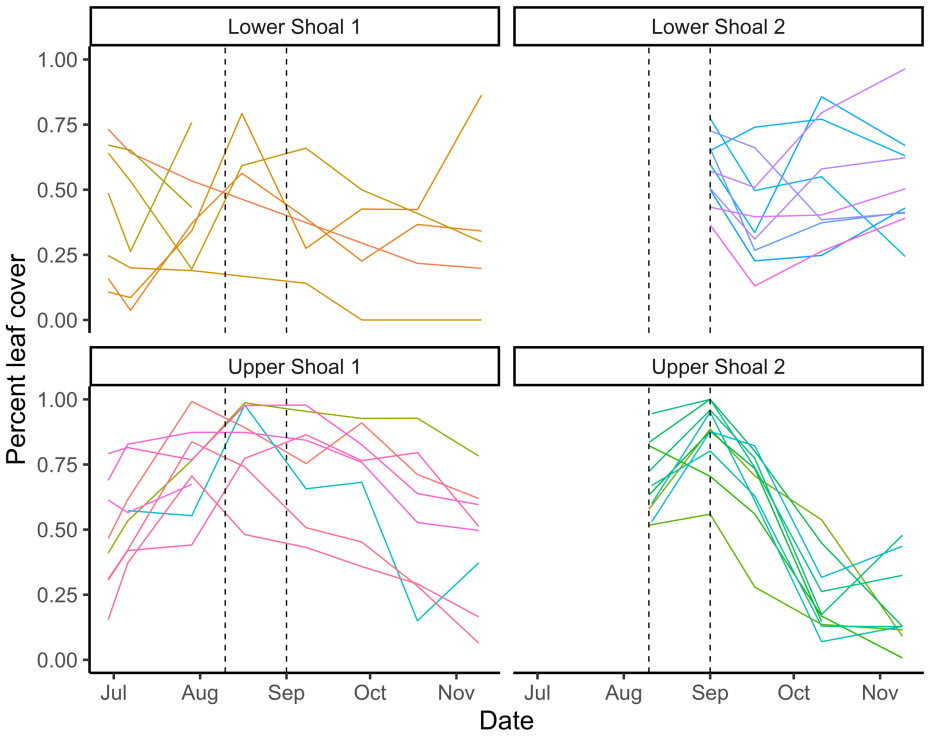

Figure 2.1. Percent *Podostemum* leaf cover on tagged rocks in the Middle Oconee River near Athens, GA, June 29th through November 9th, 2022. Rocks 1-20 were deployed on the same date, with 1-10 in the upper shoal ("Upper Shoal 1") and 11-20 in the lower shoal ("Lower Shoal 1"). About 6 weeks later rocks 21-30 were deployed in the upper shoal ("Upper Shoal_2") and three weeks after that the final 31-40 rocks were deployed in the lower shoal ("Lower Shoal 2"). Vertical dashed lines depict the second and third deployment dates. We observed a consistent decline in leaf cover in the upper shoal after September, but patterns were less clear for the lower shoal.

Figure 2.2. *Podostemum* loss from one of the most extensive grazing events observed on tagged rocks in the Middle Oconee River near Athens, GA, June-November 2022. Photographs show rinsed rock number 8 on July 29th (left) at the beginning of the interval before the grazing event and on August 17th (right) after a grazing event. The bare patch on the rock (left) is where we trimmed stems and observed sediment accumulation and was not attributed to grazing. In the photo on the right, we observed extensive grazing over the entire rock

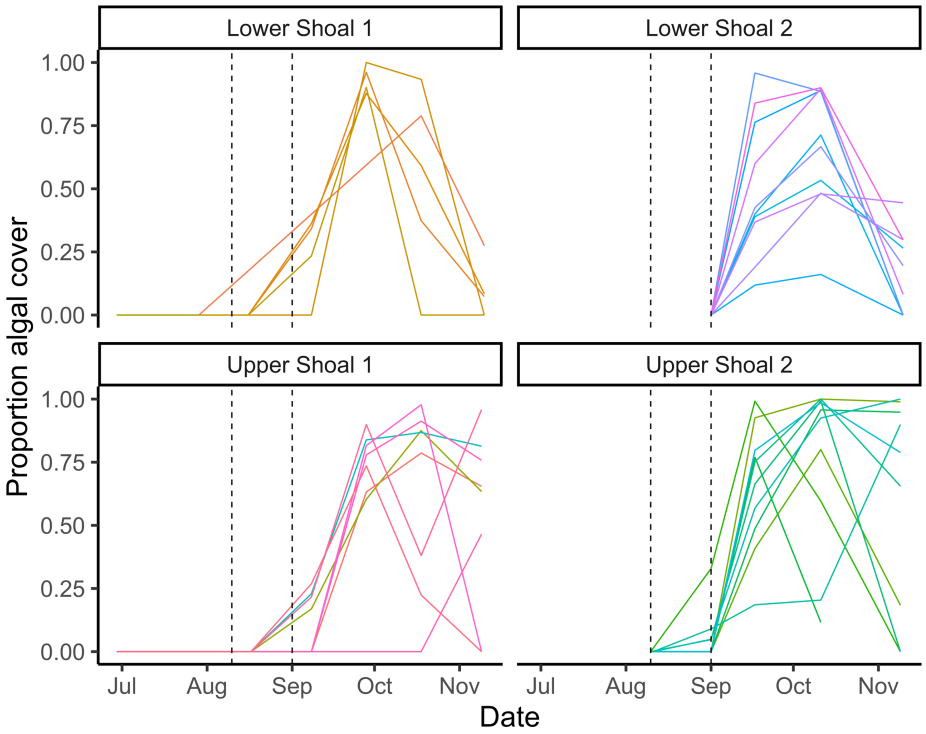

Figure 2.3. Proportion algal plotted for each rock on each date, 36 rocks total, in the upper and lower shoal locations in the Middle Oconee River near Athens, GA, June-November 2022. Algae was not detected until September 1st.

Figure 2.4. Two examples of high algal cover on tagged rocks in the Middle Oconee River near Athens, GA, June-November 2022. Algal cover on rock 5 (left) was the thickest observed buildup of algae, estimated as 82% cover on September 29th. Rock 35 (right) was photographed on September 17th and had 95% cover.

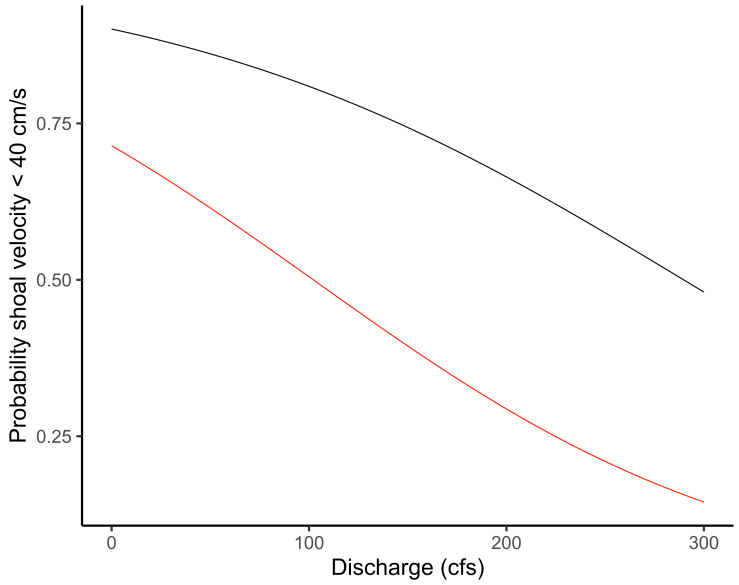

Figure 2.5. Estimated probability that shoal water velocities are below 0.40 m/s based on river discharge in two shoal locations (upper shoal,black, and lower shoal, red) on the Middle Oconee River near Athens, GA. The upper shoal is approximately three times wider than the lower shoal and has a higher proportion of the shoal water velocities < 0.40 m/s across discharge levels.

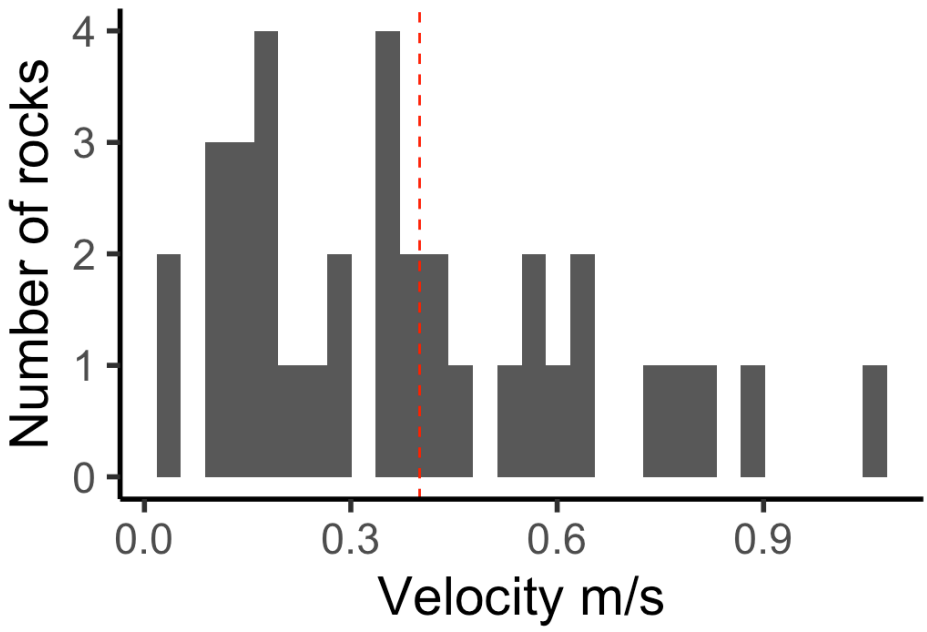

Figure 2.6. Mean velocity over the sampling period for each of 36 tagged rocks in the Middle Oconee River near Athens, GA, July-November 2022. Dashed line indicates 0.4 m/s, an hypothesized threshold for deterring grazing on *Podostemum*.

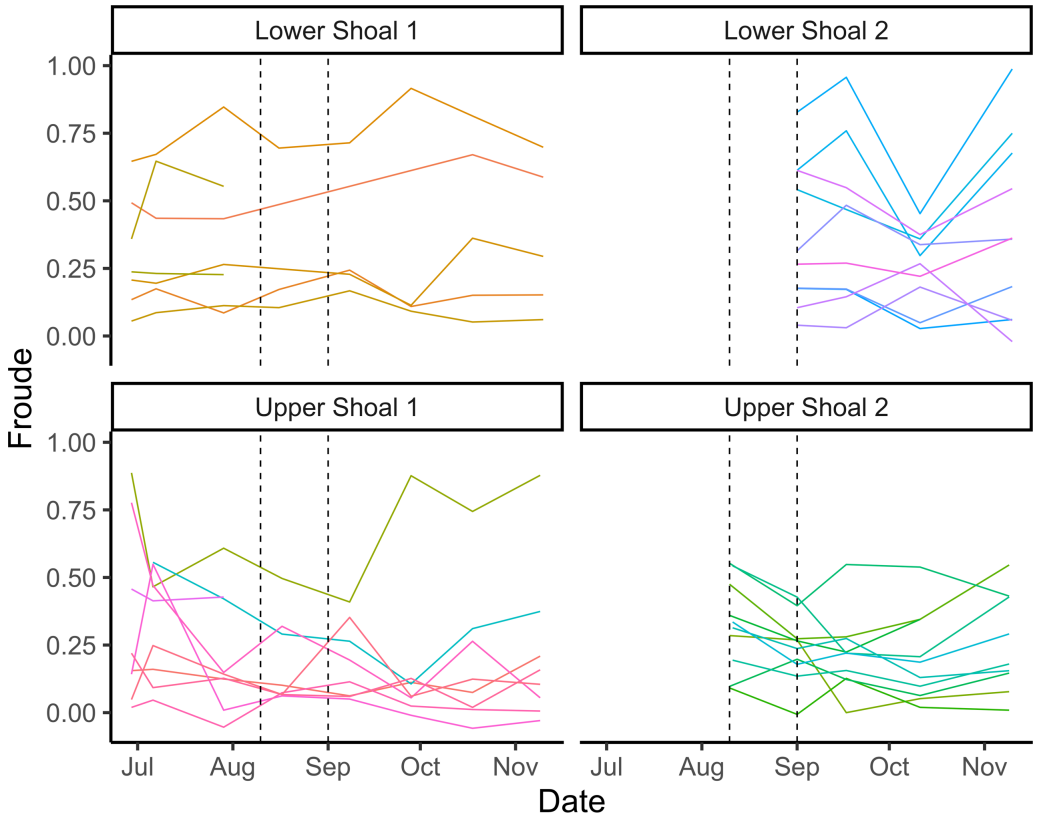

Figure 2.7. Froude number calculated for individual tagged rocks (shown in different colors) in the Middle Oconee River, July-November 2022, on each sampling date. Vertical dashed lines depict the second (rocks 21-30) and third (rocks 31-40) deployment dates.

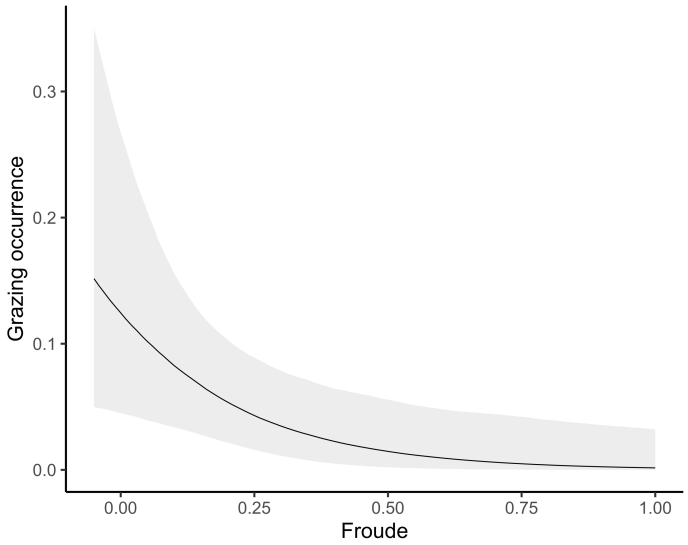

Figure 2.8. The predicted grazing occurrence on *Podostemum* attached to 36 tagged rocks in the Middle Oconee River, July-November 2022, declines as Froude number increases. Predictions are based on the best-supported grazing model and shading represents the 95% credible interval.

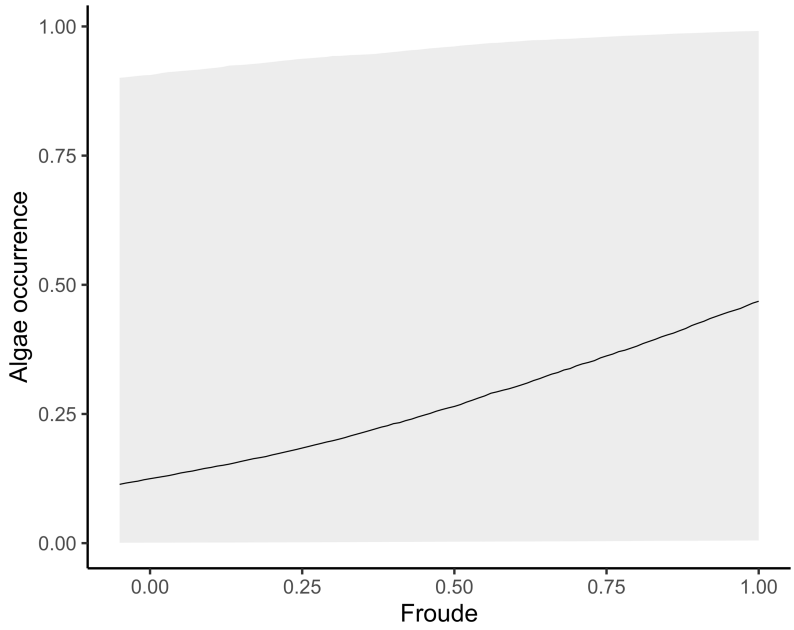

Figure 2.9. Predicted algal occurrence on 36 tagged rocks in the Middle Oconee River, July-November 2022, increases as Froude number increases. Predictions are based on the best-supported algal occurrence model and shading represents the 95% credible interval.

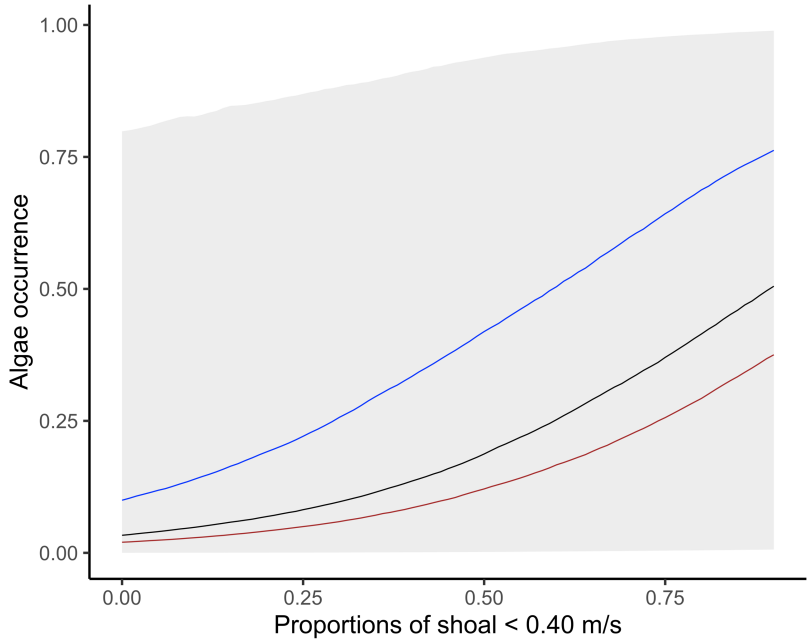

Figure 2.10. Predicted algal occurrence on 36 tagged rocks in the Middle Oconee River, July-November 2022, increases as the proportion of the shoal with water velocity <0.40 m/s increases. Froude number was held constant for three levels: mean Froude over study period (0.28; black), high Froude number (0.90; blue), and low Froude number (0; brown). Predictions are based on the best-supported algal occurrence model and shading represents the 95% credible interval.

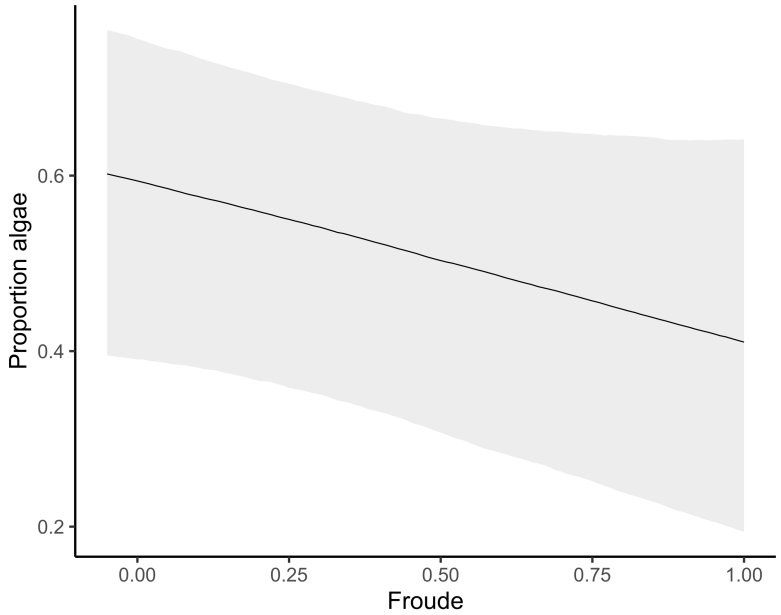

Figure 2.11. The predicted proportion cover by algae on 33 tagged rocks in the Middle Oconee River, July-November 2022, decreases in relation to Froude number; plotted line and credible interval is for the mean proportion (0.55) of shoal with velocities < 0.40 m/s.

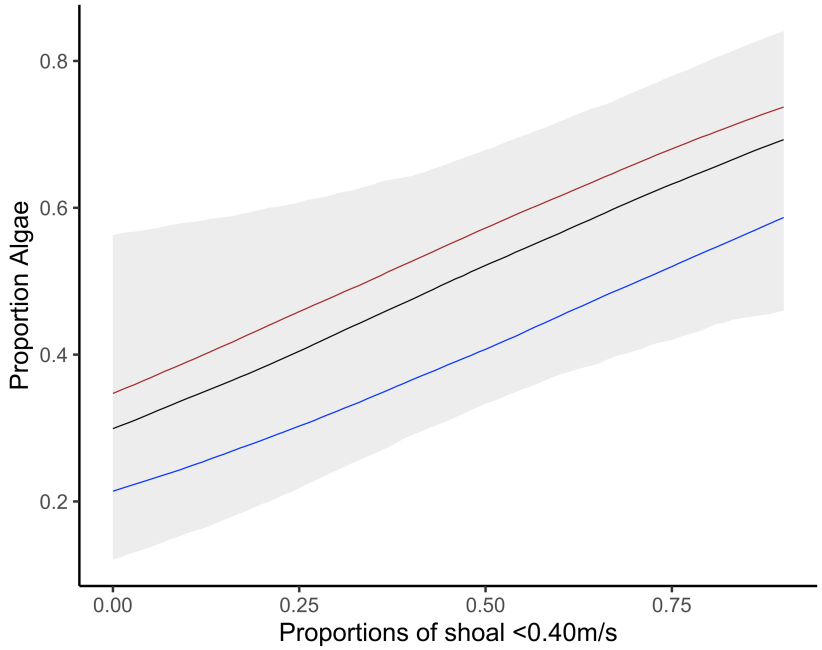

Figure 2.12. The predicted proportion cover by algae on 33 tagged rocks in the Middle Oconee River, July-November 2022, increases as the proportion of the shoal with water velocities< 0.40 m/s increases. Froude number was held constant for three levels: mean Froude over study period (0.47; black), high Froude number (0.90; blue), and low Froude number (0; brown). Predictions are based on the best-supported algal cover model and shading represents the 95% credible interval.

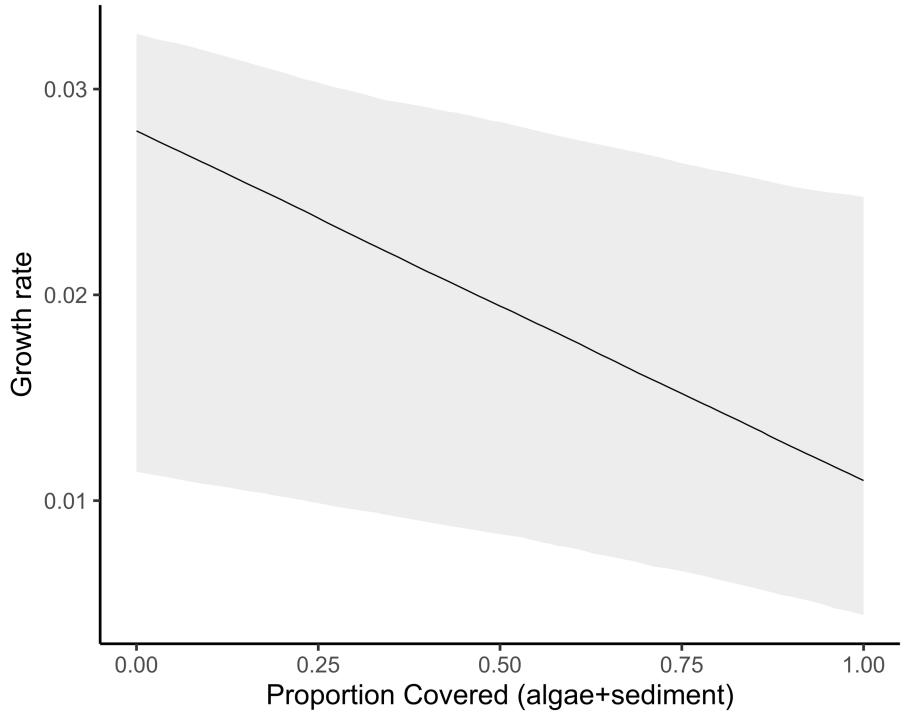

Figure 2.13. The predicted plant growth rate on 36 tagged rocks in the Middle Oconee River, July-November 2022, decreases as the total cover (algae + sediment) increases. Growth rate declines by up to 61%. Predictions are based on the best-supported plant growth model and shading represents the 95% credible interval.

References

Argentina, J. E., M. C. Freeman, and B. J. Freeman. 2010. The response of stream fish to local and reach-scale variation in the occurrence of a benthic aquatic macrophyte. Freshwater Biology **55**:643-653.

Arthaud, F., M. Mousset, D. Vallod, J. Robin, A. Wezel, and G. Bornette. 2012. Effect of light stress from phytoplankton on the relationship between aquatic vegetation and the propagule bank in shallow lakes. Freshwater Biology **57**:666-675.

Bakker, E. S., K. A. Wood, J. F. Pagès, G. F. Veen, M. J. A. Christianen, L. Santamaría, B. A. Nolet, and S. Hilt. 2016. Herbivory on freshwater and marine macrophytes: A review and perspective. Aquatic Botany **135**:18-36.

Burger, J. 2007. Bioindicators: Types, Development, and Use in Ecological Assessment and Research. Environmental Bioindicators **1**:22-39.

Biggs, B. J. F. 1996. Patterns in Benthic Algae of Streams. Pages 31-56 *in* R. J. Stevenson, M. L. Bothwell, and R. Lowe, editors. Algal Ecology: Freshwater Benthic Ecosystems. Academic Press, California.

Biggs, B. J. F., D. G. Goring, and V. I. Nikora. 1998. Subsidy and stress responses of stream periphyton to gradients in water velocity as a function of community growth form. Journal of Phycology **34**:598-607.

Biggs, B. J. F., and G. M. Price. 1987. A survey of filamentous algal proliferations in New Zealand rivers. **21**:175-191.

Bytyci, P., A. Shala-Abazi, F. Zhushi-Etemi, G. Bonifazi, M. Hyseni-Spahiu, O. Fetoshi, H. Cadraku, F. Feka, and F. Millaku. 2022. The Macrophyte Indices for Rivers to Assess the Ecological Conditions in the Klina River in the Republic of Kosovo. Plants (Basel) **11**.

Clarke, S. J. 2002. Vegetation growth in rivers: influences upon sediment and nutrient dynamcis. Progress in Physical Geography **26**:159-172.

Davis, T. W., D. L. Berry, G. L. Boyer, and C. J. Gobler. 2009. The effects of temperature and nutrients on the growth and dynamics of toxic and non-toxic strains of Microcystis during cyanobacteria blooms. Harmful Algae **8**:715-725.

Dietterich, L. H., S. O. Rosa, B. R. Charbonneau, and S. K. McKay. 2024. A systematic review of mechanistic models of riverine macrophyte growth. Aquatic Botany **190**.

Everitt, D. T., and J. M. Burkholder. 1991. Seasonal dynamics of macrophyte communities from a stream flowing over granite flatrock in North Carolina, USA. Hydrobiologia **222**:159-172.

Glibert, P. M., G. C. Pitcher, S. Bernard, and M. Li. 2018. Advancements and Continuing Challenges of Emerging Technologies and Tools for Detecting Harmful Algal Blooms, Their Antecedent Conditions and Toxins, and Applications in Predictive Models. Pages 339-357 *in* P. M. Glibert, E. Berdalet, M. A. Burford, G. C. Pitcher, and M. Zhou, editors. Global Ecology and Oceanography of Harmful Algal Blooms. Springer International Publishing, Cham.

Gregg, W. W., and F. L. Rose. 1982. THE EFFECTS OF AQUATIC MACROPHYTES ON THE STREAM MICRO- ENVIRONMENT. Aquatic Botany **14**:309-324.

Grubaugh, J. W., and J. B. Wallace. 1995. Functional structure and production of the benthic community in a Piedmont river: 1956-1957 and 1991-1992. Limnology and Oceanography **40**:490-501.

Gurnell, A. 2013. Plants as river system engineers. Earth Surface Processes and Landforms **39**:4-25.

Hutchens, J. J., J. B. Wallace, and E. D. Romaniszyn. 2004. Role of Podostemum ceratophyllum Michx. in structuring benthic macroinvertebrate assemblages in a southern Appalachian river. Journal of the North American Benthological Society **23**:713-727.

Jackson, C.R., Martin, J.K., Leigh, D.S. and West, L.T., 2005. A southeastern piedmont watershed sediment budget: Evidence for a multi-millennial agricultural legacy. Journal of Soil and Water Conservation, 60(6), pp.298-310.

Kaijser, W., A. W. Lorenz, S. Birk, and D. Hering. 2021. The interplay of nutrients, dissolved inorganic carbon and algae in determining macrophyte occurrences in rivers. Sci Total Environ **781**:146728.

Kuhar, U., M. Germ, A. Gaberščik, and G. Urbanič. 2011. Development of a River Macrophyte Index (RMI) for assessing river ecological status. Limnologica **41**:235-243.

Lázár, A. N., A. J. Wade, and B. Moss. 2016. Modelling Primary Producer Interaction and Composition: an Example of a UK Lowland River. Environmental Modeling & Assessment **21**:125-148.

Madsen, J. D., P. A. Chambers, W. F. James, E. W. Koch, and D. F. Westlake. 2001. The interaction between water movement, sediment dynamics and submersed macrophytes. Hydrobiologia **444**:71-84.

Nelson, D. J., and D. C. Scott. 1962. Role of Detritus in The Productivity of a Rock-Outcrop Community in a Piedmont Stream Limnology and Oceanography **7**:396-413.

Pahl, J. 2009. EFFECTS OF FLOW ALTERATION ON THE AQUATIC MACROPHYTE PODOSTEMUM CERATOPHYLLUM (RIVERWEED); LOCAL RECOVERY POTENTIAL AND REGIONAL MONITORING STRATEGY. Thesis - MS.

Pearson, D. L. 1994. Selecting Indicator Taxa for the Quantitative Assessment of Biodiversity. Philosophical Transactions: Biological Sciences **345**:75-79.

Philbrick, C. T., and A. Novelo. 2004. Monograph of Podostemumm (Podostemaceae). Systematic Botany Monographs **70**.

Poikane, S., R. Portielje, L. Denys, D. Elferts, M. Kelly, A. Kolada, H. Maemets, G. Phillips, M. Sondergaard, N. Willby, and M. S. van den Berg. 2018. Macrophyte assessment in European lakes: Diverse approaches but convergent views of 'good' ecological status. Ecol Indic **94**:185-197.

Power, M. E., M. S. Parker, and W. E. Dietrich. 2008. SEASONAL REASSEMBLY OF A RIVER FOOD WEB: FLOODS, DROUGHTS, AND IMPACTS OF FISH. **78**:263-282.

Power, M. E., A. Sun, G. Parker, W. E. Dietrich, and J. T. Wootton. 1995. Hydraulic Food-Chain Models. BioScience **45**:159-167.

Srivastava, J., A. Gupta, and H. Chandra. 2008. Managing water quality with aquatic macrophytes. Reviews in Environmental Science and Bio/Technology **7**:255-266.

Suren, A. M., B. J. F. Biggs, C. Kilroy, and L. Bergey. 2003. Benthic community dynamics during summer low-flows in two rivers of contrasting enrichment 1. Periphyton. New Zealand Journal of Marine and Freshwater Research **37**:53-70.

Suren, A. M., and T. Riis. 2010. The effects of plant growth on stream invertebrate communities during low flow: a conceptual model. Journal of the North American Benthological Society **29**:711-724.

Swe, T., P. Lombardo, A. Ballot, J.-E. Thrane, J. Sample, T. E. Eriksen, and M. Mjelde. 2021. The importance of aquatic macrophytes in a eutrophic tropical shallow lake. Limnologica **90**.

Szoszkiewicz, K., S. Jusik, K. Pietruczuk, and D. Gebler. 2019. The Macrophyte Index for Rivers (MIR) as an Advantageous Approach to Running Water Assessment in Local Geographical Conditions. Water **12**.

Thomaz, S. M. 2021. Ecosystem services provided by freshwater macrophytes. Hydrobiologia **850**:2757-2777.

Thomaz, S. M., and E. R. d. Cunha. 2010. The role of macrophytes in habitat structuring in aquatic ecosystems: methods of measurement, causes and consequences on animal assemblages' composition and biodiversity. Acta Limnologica Brasiliensia **22**:218-236.

Tonetto, A. F., R. Cardoso-Leite, M. C. Novaes, and R. Guillermo-Ferreira. 2014. The relationship between macroalgal morphological complexity and hydraulic conditions in stream habitats. Hydrobiologia **747**:33-41.

Umetsu, C. A., F. C. Aguiar, M. T. Ferreira, L. F. Cancian, and A. F. M. Camargo. 2018. Addressing bioassessment of tropical rivers using macrophytes: The case of Itanhaém Basin, São Paulo, Brazil. Aquatic Botany **150**:53-63.

Wade, A. J., P. G. Whitehead, G. M. Hornberger, and D. L. Snook. 2002. On modelling the flow controls on macrophyte and epiphyte dynamics in a lowland permeable catchment: the River Kennet, southern England. Science of the Total Environment **282-283**:375-393.

Wilby, R. L., L. E. Cranston, and E. J. Darby. 1998. Factors Governing Macrophyte Status in Hampshire Chalk Streams: Implications for Catchment Management. Water and Environment Journal **12**:179-187.

Wood, J., and M. Freeman. 2017. Ecology of the macrophyte Podostemum ceratophyllum Michx. (Hornleaf riverweed), a widespread foundation species of eastern North American rivers. Aquatic Botany **139**:65-74.

Wood, J. L., J. W. Skaggs, C. Conn, and M. C. Freeman. 2019. Water velocity regulates macro-consumer herbivory on the benthic macrophyte Podostemum ceratophyllum Michx. Freshwater Biology **64**:2037-2045.

Wood, K. A., M. T. O'Hare, C. McDonald, K. R. Searle, F. Daunt, and R. A. Stillman. 2017. Herbivore regulation of plant abundance in aquatic ecosystems. Biol Rev Camb Philos Soc **92**:1128-1141.

Xie, D., D. Yu, W. H. You, and L. G. Wang. 2013. Algae mediate submerged macrophyte response to nutrient and dissolved inorganic carbon loading: a mesocosm study on different species. Chemosphere **93**:1301-1308.

Zhang, L., L. Biyun, G. Fangjie, Q. Liu, Y. Zhang, Q. Zhou, D. Xu, and Z. Wu. 2019. Interspecific Competition for Nutrients between Submerged Macrophytes (Vallisneria natans, Ceratophyllum demersum) and Filamentous Green Algae (Cladophora oligoclona) in a Co-Culture System. Pol. J. Environ. Stud. **28**:1483-1494.

CHAPTER 3

QUANTIFYING RIVER ECOSYSTEM NEEDS: DEVELOPING ENVIRONMENTAL FLOWS AND METRICS FOR REGIONAL WATER PLANNING IN GEORGIA, USA[2]

[2]Rack, L. R., M. C. Freeman, G. Cowie, C. Yang, S. K. McKay, L. Craig, and S. J. Wenger. Submitted to PLOS Water 3/18/2025

Abstract

Effective water management requires the capacity to make trade-offs among diverse uses of water such as municipal water supply, irrigation, and ecological outcomes. Environmental flow management seeks to understand the relationships between river flows and ecosystems processes to evaluate the relative change in ecological outcomes associated with different strategies for river management. However, operationalizing ecological flow thresholds remains technically and administratively challenging, particularly at large scales. Here, we present a case study identifying environmental flow targets using the functional flows framework in the Oconee River Basin, Georgia, USA. Quantitative discharge thresholds are developed for five ecologically relevant flows addressing channel maintenance, floodplain connectivity, springtime pulses, reproductive season baseflows, and dry season baseflows. We demonstrate how these targets integrate ecosystem water needs into a broader state-level water planning process. Four themes emerge from this case study that are applicable in other geographies and contexts. First, environmental flow targets cannot be abstracted from their physical, ecological, and political geography, and context-specificity is critical to developing management-relevant flow targets. Second, quantitative environmental flow thresholds help establish ecological outcomes on equal footing with socio-economic uses of water in planning processes. Third, environmental flow frameworks should align with the management scope so that metrics align with the local context for implementation. Finally, decision makers should be provided with information to evaluate and interpret different outcomes for environmental flow targets alongside other water management targets. Despite these complexities, environmental flow analyses remain an essential tool to address the threats to freshwater ecosystems and biodiversity driven by human alteration, water use, and global change.

Keywords: flow-ecology, instream flows, river management, decision-making, trade-off analysis

Introduction

Societies depend on provisioning, regulating, cultural, and supporting services provided by freshwater ecosystems (Millennium Ecosystem Assessment 2005, Hanna et al. 2017, Vari et al. 2022). However, the maintenance of naturally functioning freshwater ecosystems often is omitted as an objective of water management, particularly for rivers (Richter et al. 2012). Most riverine ecosystems have undergone significant changes through damming, water withdrawal, channelization, impacts from development, altered flows of water, and inputs of nutrients and sediment (Fisher et al. 2000, Carlisle et al. 2011, Auerbach et al. 2014, Ferrazzi and Botter 2019). In addition, weather patterns are becoming more variable and extreme under climate change, leading to greater uncertainty around the quantity and timing of access to water resources (Palmer et al. 2009). There has been increased emphasis in the management realm to recognize impacts to biodiversity and freshwater ecosystems through development and implementation of natural infrastructure and river restoration (van Rees et al. 2023), but this necessitates foresight in both planning and management to address specific ecological outcomes.

A broadly accepted approach to preserving functioning river ecosystems is to maintain "environmental flows" (or "e-flows")—i.e., the quantity, timing, and volume of water needed to produce valued environmental services and outcomes, which may range from sustaining aquatic organisms to maintaining natural riverine geomorphic processes (The Brisbane Declaration 2007). Agencies charged with public water supply or regulating water resources often have a narrow scope of legal authority, complex institutional connections to other entities, or may lack specific targets for environmental protection (Prosser et al. 2015, Yaryan-Hall and Bledsoe 2023), defaulting to general notions of natural system sustainability. Prioritizing and

implementing e-flow frameworks often requires high-level coordination, such as in the Building Blocks Method, which was codified at the national level in South Africa (King et al. 2000). In cases without explicit environmental protections, the language of instream flow laws and institutional biases tend to favor human uses (Zellmer 2008). While there has been significant progress in approaches to develop and implement e-flow frameworks since the 1940's (Poff et al. 2017), adoption and prioritization of e-flows in water allocation decisions are often lacking.

The functional flows approach (Yarnell et al. 2015), which has similarities to the Building Blocks Method, was developed as a tool for use by state agencies, non-governmental organizations (NGOs), watershed groups, and others to develop e-flow relationships and recommendations based on aspects of the flow regime that support generalized ecological and geomorphic processes of a river system (California Environmental Flows Working Group 2021). The functional flows approach has the same underpinnings as many ecological-flow frameworks, namely that river ecosystems reflect adaptations to a flow regime (Power et al. 1995, Petts 1996, Poff et al. 1997) that humans have severely altered through damming, diversions, alterations to channel structure, and landscape-level changes to hydrology (Carpenter et al. 2011, Craig et al. 2017, Spinti et al. 2023). In response, ecologists have developed numerous hydrologic indicators (e.g., the Indicators of Hydrologic Alteration) and more complex metrics (e.g., Mathews and Richter 2007) that could be used to assess critical loss of ecological functions because of flow alteration. These approaches may involve dozens of individual metrics to fully describe natural flow regimes (see, e.g., TNC 2007, Olden and Poff 2003, George et al. 2021). However, to fit the context of water planning, we argue for a more practicable number of metrics (i.e., five or fewer) that broadly represent ecosystem functions. Planners assess water availability to meet specific municipal and industrial targets; creating analogous targets to meet ecosystem needs is a better

fit to the planning process (Acreman et al. 2014) than asking planners to assess degree of departure from a natural flow regime (e.g., McKay 2015). Furthermore, keeping the number of targets to five or fewer also aligns with research on the cognitive limits on decision-making (Yoe 2002, Retief et al. 2013).

We present a case study on the applicability of a functional flow approach to develop ecological metrics for use in basin-wide water planning. Our work is relevant to the challenge of specifically assessing water availability for ecological values when environmental protection is vaguely defined in the regulatory and management context and secondary to meeting other societal needs. The functional flows approach provides a path for specifying a small set of e-flows to support ecological functions. We illustrate how we used available data to develop functional flow targets for distinct portions of the basin, with the recognition that these represent a first set of quantitative targets based on the current data available and should be updated over the long-term as new information becomes available. We also show how we integrated ecosystem water needs into the planning process. We then discuss four themes that emerged from the work that may be broadly useful when applying e-flow in water management and planning.

Case Study: Developing Functional Flows for the Oconee River

Georgia's Comprehensive State Water Plan was adopted in 2008 by the Georgia General Assembly and was guided by state and federal statutes that protect environmental quality, support public health, and is intended to ensure water resource availability into the future (Water Council 2008). The plan established 10 regions, each with its own council that develops a regional water plan that is updated every five years (Georgia Water Planning). These regions are mostly drawn around river basins, though the boundaries follow county lines, such that each

local government only belongs to a single region. The regional water plans assess surface water availability, ground water availability, and surface water quality (primarily through the lens of assimilative capacity). As of the 2022 planning cycle, the Surface Water Availability Resource Assessment (herein "Resource Assessment") conducted by the Georgia Environmental Protection Division uses a hydrologic model that can represent withdrawals, discharges and other water demands at a spatial resolution of the stream reach to provide quantitative information to guide water management decisions. Water plans summarize the current municipal, energy, agricultural, and industrial water demands within a region and develop a forecast of future water demand. Based on the results of the demand forecasts and Resource Assessments, the plans highlight expected challenges in meeting water needs, along with management practices to address those challenges (Water Council 2008, Council 2023). However, regional water plans do not typically consider current or future gaps in surface water availability to meet environmental or recreational needs.

In Georgia, water rights are governed by the riparian doctrine and "reasonable use" as determined by impacts to downstream neighbors or other riparian landowners (Bowen 2001). The Environmental Protection Division has created permitting programs for wastewater, drinking water, water withdrawals, stormwater, and erosion and sedimentation. The typical minimum flow defined in permits is the one-in-ten year, 7-day low-flow (7Q10), a level that was intended for the protection of water quality, not to maintain aquatic communities (Board of Natural Resources 2001). The need for better information about environmental water needs has been expressed in reports produced by Georgia's regional water councils (e.g., ARCADIS 2019).

Our study specifically focused on the Oconee River Basin (Figure 3.1), which is almost entirely within the Upper Oconee Water Planning Region, located in east-central Georgia. This

water planning region spans 13 counties and has a population of around 620,000 (Council 2023). The watershed exhibits a suite of water management challenges common throughout the region, namely: a legacy of sediment in rivers from historic land uses (Jackson et al. 2005) and modern water supply and stormwater issues associated with urban development of the Piedmont region (Jackson et al. 2023). Most urbanization is in the northern part of the region, with the remaining basin dominated by agriculture, silviculture and low-density residential development. The basin supports municipal, industrial, energy, and agricultural water uses, with surface water as the main water source for the region (Council 2023).

The Oconee River drains 8,578 km^2 on the Atlantic Slope of Georgia and is a major tributary of the Altamaha River (Figure 3.1). The Oconee Basin has 34 surface water withdrawal permits (with 17 in the mainstem river and reservoirs), 90 National Pollution Discharge Elimination System (NPDES) permitted discharges, and three licensed hydropower projects (data as of 2020). One hydropower project is a small run-of-river dam (Tallassee Dam) on the Middle Oconee River in the Piedmont portion of the basin. The other two, Wallace and Sinclair dams, are situated on the Fall Line (the physiographic divide between the Piedmont and Coastal Plain), and impound large reservoirs (Lake Oconee, about 8000 ha and Lake Sinclair, about 6200 ha) that are jointly operated for pumped storage hydropower. Recreation includes motorized boating in the reservoirs and larger river reaches and non-motorized (e.g., canoeing, kayaking) boating throughout the basin. People also use areas along the river for hunting, fishing, and outdoor recreation on public and private lands.

The headwaters of the Oconee basin are in the Piedmont physiographic province where larger streams have rocky shoal habitats that support distinct aquatic communities, along with deeper-water pools and runs. As the river transitions into the Coastal Plain, river and floodplain

habitats include oxbow lakes, sand and gravel bars, deeper pools, snags (i.e., accumulations of large wood), and seasonally inundated floodplains. The river basin is home to at least 65 native species of fishes, 16 native mussel species, and 11 native crayfish species (Wildlife Resources Division 2021, Georgia Museum of Natural History 2021).

Developing Metrics for Water Planning

The functional flows approach explicitly accounts for the geomorphic and ecological processes supported by a river's flow regime (Yarnell et al. 2015). Rivers in the Piedmont and Coastal Plain Physiographic provinces of the southeastern US are perennial runoff systems (McManamay and DeRolph 2019), in contrast with the snowmelt dominated systems for which the functional flows approach was initially developed. We modified the functional flows framework to reflect the seasonally higher flows in the winter and spring and lower flows in the summer and fall and to accommodate the differences between the geologic context of the Piedmont and Coastal Plain. We developed five categories of functional flows (or functional flow components; Yarnell et al. 2020) for Georgia rivers (Table 3.1) to encompass a range of fundamental ecological processes driven by seasonal flow variation.

Our objective was to develop hydrologic environmental flow thresholds for the Upper Oconee Water Planning Region that could be evaluated with the Resource Assessment and could be used similarly to the metrics for water supply and wastewater treatment. To do this, we used information from local studies to develop threshold flow levels (river discharges) that supported an ecological or geomorphic process, or below which an ecological function was expected to decline. The intent was that managers and stakeholders could evaluate location-specific data for how long or how often (or both) flows would be above or below a threshold given future water

demands. Because Piedmont and Coastal Plain regions experience dramatically different physical, ecological, and social drivers, we developed separate flow recommendations for each region.

Each of our thresholds was based on a specific research study and linked with the nearest USGS gage (with at least a 20-year record) to the study location. To provide context for how the thresholds related to historical flows, we used historical gage data to calculate the flow percentile for each threshold based on the month or months that were specified in the metric for each functional flow component. For example, the flow percentile for the floodplain connectivity component is the percentile of the threshold level given all the daily flow values from November to March over the period of record.

We identified eight local studies that reported information useful for identifying river discharges that supported four of the five functional flow components at sites in the Piedmont portion of the Oconee River basin. Topics included a study on invertebrate movement between the river and floodplain, models of instream habitats and flows to support fish recruitment, and studies of shoal ecosystems or shoal biota during dry season conditions and a drought event. We extracted a flow level associated with active mayfly dispersal between the river channel and inundated floodplain (Galatowitsch and Batzer 2011) as a starting point for a site-specific flow that facilitates floodplain habitat connectivity for small organisms (functional flow category 2; Table 3.2). For spring pulse flows (functional flow category 3), we used a model of fish recruitment in relation to the flow regime for a Middle Oconee River site (McKay et al. 2016). This model combined the probability distribution for 10-d maximum flows during species-specific spawning seasons with a generalized, trait-based model of flow effects on juvenile fish recruitment (Craven et al. 2010) to estimate “effective discharges” (Doyle and Shields 2007) for

juvenile recruitment in five diverse genera of Oconee River fishes. We selected a 10-d maximum flow threshold that theoretically provided for recruitment by all five taxa. A hydraulic model for the same site in the Middle Oconee River (Bhattacharjee et al. 2019) estimated areal extent of three generalized habitat types (shallow,<35 cm deep, with low-velocity, < 35 cm/s; shallow with high-velocity, >55 cm/s; and "deep", ≥35 cm, with moderate velocity, >45 cm/s) in relation to river discharge. This model showed a sharp decline in availability of deep, moderate-velocity habitat at about 14.2 m^3s^{-1}, and we used this value as a threshold for reproductive season baseflow (functional flow category 4). Finally, for dry season baseflows (functional flow category 5), we identified two thresholds, one associated with maintaining some deep, moderate velocity habitat in Middle Oconee River shoals (Bhattacharjee et al. 2019, Wood et al. 2019), and a second extreme low-flow associated with decline in the biomass or abundance of aquatic organisms in those same shoals (Katz 2009, Pahl 2009). We did not find a study that could identify a threshold for channel maintenance flow (functional flow category 1), so we could not evaluate this function at any Piedmont site.

Habitat simulation models constructed for the Oconee River downstream from the Fall Line hydropower dams provided a basis for four functional flow metrics in the Coastal Plain portion of the basin (Table 3.2). As part of the relicensing process for the downstream-most hydropower project (Sinclair Dam), engineering consultants estimated high flows needed for channel maintenance (functional flow category 1; Table 3.2; also supported by Yearwood (2010)) and modeled relations between flow levels and floodplain habitat inundation, instream spawning habitat for a protected fish species, connectivity to oxbow lakes, and inundation of in-channel woody debris (i.e., snag habitat; EA Engineering 1994). Availability of different habitats varied among river locations but generally showed similar and consistent relations to increasing flow

levels. We used these previously constructed habitat models to identify flows below which functionally-defined habitats were projected to decrease substantially in one or more Coastal Plain reaches (Table 3.2). Specifically, we identified two thresholds for floodplain habitat connectivity (functional flow category 2), one in which low-lying floodplain and oxbow lakes were inundated and a higher flow level that inundated a substantial portion of the floodplain (Table 3.2). Reproductive season baseflow (functional flow category 4) was based on a flow that provided a variety of habitat conditions in oxbow lakes to support spawning and rearing. Because there was a state-listed fish species (Robust Redhorse, *Moxostoma robustum*) found below the dam, we also included a specific flow range that maintained spawning habitat availability for this species. Finally, for functional flow category 5, we identified summer and fall baseflows to support fish passage for small-bodied fishes between the river channel and oxbow lakes and a second, lower flow threshold estimated to inundate at least a third of in-channel woody debris, which supports insect production and can provide refugia for aquatic organisms during low-flows. We did not have information available to develop a spring flow pulse (functional flow category 3) in the Coastal Plain.

Evaluating Metrics for Water Planning

During the water planning process, metrics for water supply, wastewater assimilation, etc. are evaluated under current and future conditions and then compared to assess potential challenges in meeting future water needs. The historical flow record, or current conditions, is a useful basis to compare functional flow metrics, under the assumption that historical flows have mostly maintained riverine functions in the past. We recommend that planners or managers compare the functional flow metrics to see how future demand and planned water management

may increase the annual duration or the number of years that the functional flows thresholds are not met.

We used the historical time series from the Middle Oconee to illustrate a comparison of the frequency and duration of low-flow events. We split the 86-year period of record into three sequential periods and evaluated shifts in the two summer and fall dry-season baseflow thresholds (as referenced in Figure 3.4) through time. We found an increase in frequency and duration of flows below the summer and fall baseflow thresholds in the most recent period, 1998-2023 (Figure 3.2). During those years, there were a similar number of low-rainfall years as the previous 30-year period, but substantially higher failure to meet the summer and fall functional flows compared with both prior periods (Figure 3.2). The most recent period also corresponded with three multi-year droughts and increased water demand, including the 2002 completion of a pump-storage reservoir that provides water to multiple counties upstream of the study sites.

Similarly, in the Coastal Plain, we observed fewer days in the most recent period, 1998-2023, with river connectivity to the floodplain (USGS gage near Dublin GA 02223500; USGS 2024). We also observed a decline through the three time periods in the 80th percentile number of days with the high river flows that connect the river and floodplain (Figure 3.3).

Communication and Contextualization of Environmental Flow Thresholds

We can use the historical flow record as a reference for how long and how often functional flow thresholds may need to occur to support ecological or geomorphic processes. We used a hydrograph to depict how the functional flows relate to long-term averages and other water uses at the same location, for the Water Council members and state partners. The

hydrograph includes the long-term daily median flow (50th percentile daily flow) in the Middle Oconee River, near Athens, GA (Figure 3.4, USGS gage 02217500; USGS 2024). We displayed the year 2004 (the black line), a near-median annual flow year, to show the flow variability within a year (Figure 3.4).

Using the hydrologic context as the backdrop, we can then think about the relationship between functional flow thresholds and other socio-economic values. In this reach of the Middle Oconee River there is a municipal water supply withdrawal in the river. The purple dashed line represents the minimum permitted withdrawal level, below that flow level the utility is not allowed to withdrawal water from the river. The reach is also used for paddling; the range of flow conditions that support relaxed paddling is shown in the orange box (Georgia River Network undated, American White Water undated, Cook 2019). We define relaxed padding as the flow range that is safe and navigable for a novice paddler, above this level, higher river flows could be unsafe and below this level paddlers may have to drag their boat to pass through the shoals and shallow sections of the river. Thresholds for functional flow categories 4 and 5 (reproductive season baseflows and dry season baseflows) are shown in blue (Figure 3.4). The higher dry season baseflow (from June to October) is the point at which deep and swift water habitats are lost, and the lower threshold is the point at which drying in shoals occurs, leading to loss of an ecologically important aquatic plant. While we use a specific location in the Middle Oconee River to illustrate this relationship, the concept holds across locations. Visualizing the thresholds for multiple uses in the river helps to illustrate how river uses and functions overlap.

Discussion

Decisions around water are socially complex due to multiple uses and varying values of stakeholders that rely on water. Despite this complexity, there is a pressing need to address the threats to freshwater ecosystems and biodiversity driven by human alteration, water use, and changing weather patterns (Tickner et al. 2020). We highlight four themes that we found important to integrating ecological outcomes into a water planning framework: 1) Identifying and understanding the context for water resource decision-making; 2) Developing quantitative metrics and thresholds for evaluating river ecosystem needs alongside other water uses; 3) Adapting an e-flow framework from other geographies and evaluating recommendations in a local system; and 4) Providing the necessary information for decision-makers to evaluate and interpret ecological metrics alongside other water uses.

Vaguely defined laws and goals for environmental protection, and fragmentation of environmental responsibility among and within local, state, and national governments, make it challenging to manage directly for environmental outcomes (Zellmer 2008, Richter 2009, Pahl-Wostl et al. 2013). Regulatory agencies responsible for biological or environmental protections often do not have direct authority to allocate instream flows for aquatic species and habitats (Zellmer 2008, Wineland et al. 2021). Water utilities, or other entities responsible for municipal, industrial, or agricultural water supply, must prioritize meeting the needs of water users and complying with statutes for surface or groundwater management. In the U.S. and more broadly, without a regulatory directive, implementing protections for river ecosystems relies on coordination with the regulatory authority that manages water quantity, partnerships with water users and water providers, conservation organizations, and other relevant parties to build consensus around ecological outcomes (Arthington et al. 2024). The water planning process in

the state of Georgia was well established at the time of our project, had relatively high-resolution information on the water resources for the state, and offered a formalized setting to discuss water resources across sectors. Working in the water planning space, or a similar cross-boundary group, helped steer the conversation towards what opportunities there were to meet an ecological need across sectors rather than asserting pressure on one entity (Safford et al. 2017). The structure of the state's water planning process created an opening for building partnerships around environmental flow metrics. Though it takes time to build trust among participants, partnerships and collaborations are key to building a solid foundation and a shared vision for management that can be responsive to changes in research or policy (Barbour et al. 2016, Anderson et al. 2019, Iwanaga et al. 2021, Golladay et al. 2022, Rack et al. 2024).

In addition to understanding the context around water resource decisions, ecologists face the challenge of the typical sparsity of place-specific data to support environmental flow thresholds. Quantitative flow-ecology relationships are difficult to develop (Poff and Zimmerman 2010), and even when there is a signal of flow effects on aquatic communities, high variance (Knight et al. 2013) or context dependency (Walters 2016) may complicate the identification of flow thresholds for community change. Flow-habitat relations, like those developed to support instream flow assessments (Stalnaker et al. 1995, IFIM), are more deterministic but may not display obvious thresholds or have demonstrated relevance to population dynamics (Shenton et al. 2012, Lancaster and Downes 2010). Having to validate flow-ecology relationships for specific species or groups of organisms can force decisions about which organisms to manage for and when, which is complex for rivers like those in the southeastern U.S. that harbor high species diversity. Alternatively, leveraging locally available data to support hypothesized thresholds around broader ecological outcomes and river processes

(e.g., will fish be able to spawn? Will animals survive the summer?) may be more meaningful for water resource partners. In our case, we used available data for specific locations within the basin to develop functional flow thresholds and identified the locations and functions where data were not available to quantify a threshold. It is important to communicate early with partners that these relationships should be revised as new data become available, ideally through strategic monitoring and model-updating (Peterson and Freeman 2016). Nevertheless, defining a quantitative threshold, even if it is in the early stages of development, provides a tangible ecological outcome to discuss and compare alongside other water uses.

Building on the ecological flow thresholds, it is important to identify an e-flow framework that is appropriate in scale and scope ecologically and for the application. Strengths of the functional flows approach for water planning lie in the flexibility of application alone or alongside other e-flow frameworks (Yarnell et al. 2015). The approach was initially developed for highly regulated systems, where returning to a natural flow regime is not feasible. Instead, a few high-level aspects of the natural flow regime, or functional flow components via Yarnell et al. (2020), that support biological and geomorphic processes in river systems are the focus for managing river ecosystems. Our experience indicates that similar functional flow components can be useful in assessing ecological consequences of future water management in less highly regulated river systems. The functional flow components can be evaluated at the basin- or reach-scale, making the approach scalable depending on the application. There is no theoretical limit to the number of metrics that can be used to evaluate function flows, but in our experience the number must be kept manageable to avoid overwhelming decision-makers. We found that about five metrics, one or two for each functional flow component, were effective because they aligned with number of metrics used to evaluate other water uses. Presenting the functional flows as

thresholds to evaluate rather than a prescriptive flow regime can open conversation for what actions, both short and long-term, would be needed to move towards meeting an instream flow.

Ecological metrics for water availability need to be comparable to the metrics for other water needs, such as water supply, wastewater discharge, hydropower, etc. (Enquist et al. 2017, Vogel et al. 2015), but also require guidance for interpretation since ecological outcomes can depend on the magnitude, duration, and frequency of flow events. Planners in Georgia may assess water availability for municipal withdrawal as the proportion of all days under a future scenario when streamflow is projected to be insufficient to support permitted withdrawal rates (Council 2023). Using the same approach for ecological metrics is less informative, however. For example, our final period (1997-2024) had 9% more days than the next highest period with flows less than $7.5 m^3 s^{-1}$. This could be 33 more days each year or 10.5 months longer below $7.5 m^3 s^{-1}$ every 10 years, with either type of increase resulting in very different ecological consequences. Instead, comparing shifts in the annual frequency and duration of low-flow events could reveal ecologically meaningful changes to flow conditions. Guidance from scientists for how to evaluate and interpret ecological metrics (Enquist et al. 2017) can support dialogue between planners, utilities, scientists and other interested parties. Whether a given shift in an ecological flow metric (e.g., from <50% to >70% of years with extreme low-flow durations) is too much depends on social tolerance for risk, ecological understanding of aquatic community resilience to drying, and regulatory guidance (if available) for a given situation.

Conclusions

Integrating ecological information into water planning represents a first step in assessing ecological outcomes, however moving from information to action also requires prioritizing

environmental outcomes alongside other uses. Often environmental flows are given the lowest priority when water becomes scarce and are viewed only as protecting non-human benefits, rather than supporting river ecosystems that provide valuable services (Richter 2009). In addition, private water interests are often over-emphasized by traditional optimization methods or decision support tools used in the water planning process (Yang et al. 2023, Fletcher et al. 2022), making it difficult to represent public interest in the water planning process. Increasing representation in the water planning space for cultural uses, recreation, and ecological outcomes involves changes to the structure of decision making around water. Evaluating ecological conditions in water planning can start a dialogue about managing for ecological outcomes. Often decisions to address environmental impacts are made reactively; however, developing, evaluating, and interpreting e-flow metrics that fit the management context highlight risks to traditionally under-valued resources, and encourage exploration of management approaches to minimize those losses. This may help shift away from the idea of water for either humans or nature and towards a system that values ecosystem services of natural river ecosystems alongside other social and economic water uses.

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Table 3.1 Five functional flow components developed for application in regional water planning

Functional Flow	Description
1. Channel maintenance flows	Maintain the dynamic erosional and depositional forces that shape channel form and aquatic habitats
2. Floodplain connectivity flows	Inundate, connect and cue movements to diverse floodplain habitats, and support sediment and nutrient exchange between river and floodplain
3. Springtime pulse flows	Provide spawning cues for fishes or other organisms and flushing flows during the spawning season
4. Reproductive season baseflows	Provide adequate water for successful reproduction (e.g., spawning behaviors, egg-laying, and larval rearing) including availability of and connectivity among diverse habitats
5. Dry season baseflows	Maintain habitat connectivity and conditions for the survival of aquatic organisms during seasonal low-flows

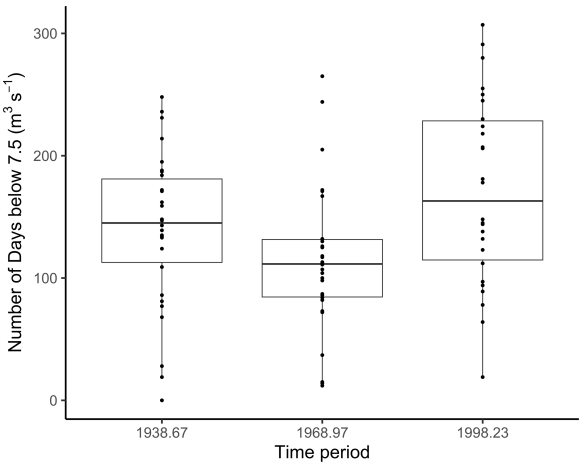
Table 3.2 Functions associated with each functional flow component and a general metric that could be evaluated with location-specific thresholds for the Piedmont and Coastal Plain. We reported thresholds for each physiographic province when data were available. We linked each threshold with a nearby USGS gage with at least a 20-year flow record. We reported the discharge values and flow percentile over the period of record, which was calculated from the long-term record using the calendar months associated with the metric.

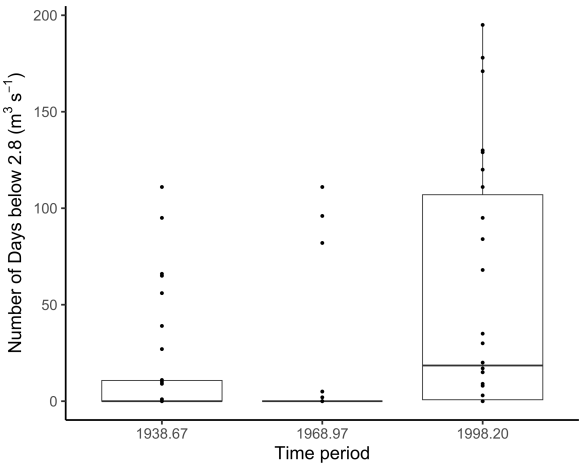
Functional Flow Component	***Function(s)***	***Metric***	***Location-specific thresholds***
Channel maintenance	Sediment transport and channel dynamics that maintain and create diversity of in-channel habitats	# years > *channel threshold level*	Piedmont: No data available for channel maintenance flow levels or frequencies Coastal Plain: flow levels that maintain channel migration and bank erosion processes. USGS gage: 02223000 340 m^3s^{-1}; 97%
Floodplain connectivity	Inundate and connect habitat for wetland dependent species (amphibians, aquatic insects, fishes, birds) Support seed dispersal for floodplain tree species, e.g. bald cypress and water tupelo Nutrient exchange between channel and floodplain	# days during November-March with flows > *floodplain threshold level*	Piedmont: flows that connect the river and floodplain, which supports connectivity and movement by organisms USGS gage: 02217770 30 m^3s^{-1}; 88% Coastal Plain: inundation of floodplain habitat and oxbow lakes; ranges from low elevation habitat inundation to full inundation of floodplain habitat USGS gage: 02223000 283 m^3s^{-1}; 93% USGS gage: 02223056 142 m^3s^{-1}; 80%

			USGS gage: 02223500 425 m^3s^{-1}; 93%
Springtime pulse flows	Flush fine sediment from fish spawning substrates (e.g., gravel, crevices, cavities)	# years with the maximum 10-day high flow in March-May > *spring pulse flow*	Piedmont: flushing flows maximize reproduction output for gravel-spawning fishes USGS gage: 02217500 34 m^3s^{-1}; 91% Coastal Plain: No data available for spring pulse flows
Reproductive season baseflows (spring and early summer)	Create and maintain conditions needed for animals to successfully reproduce, including habitat availability, preventing settling (broadcast- spawned) and siltation (gravel- and crevice-spawned) of eggs and larvae, providing oxygen to deposited eggs and larvae	# days during March-May with flow < *reproductive season threshold*	Piedmont: decline in availability of swift water habitats USGS gage: 02217500 14 m^3s^{-1}; 49% Coastal Plain: maintain spawning and rearing habitat for fishes USGS gage: 02223056 Oxbow habitat 85 m^3s^{-1}; 62% Robust redhorse *Consecutive days between*

			28 – 57 m^3s^{-1}; 14% - 41%
Dry season baseflows (summer and fall)	Support growth and survival of aquatic organisms Sustain higher velocity habitats Maintain habitat connectivity	# days during June-October with flow < *dry season threshold*	Piedmont: severe reduction in deep swift water habitat; loss of aquatic organisms at severe low flows USGS gage: 02217500 7.5 m^3s^{-1}; 62% 2.8 m^3s^{-1}; 15% Coastal Plain: loss of connectivity between channel and oxbow and decline in area of submerged woody debris USGS gage: 02223056 21 m^3s^{-1}; 7% 14 m^3s^{-1}; 2.5%

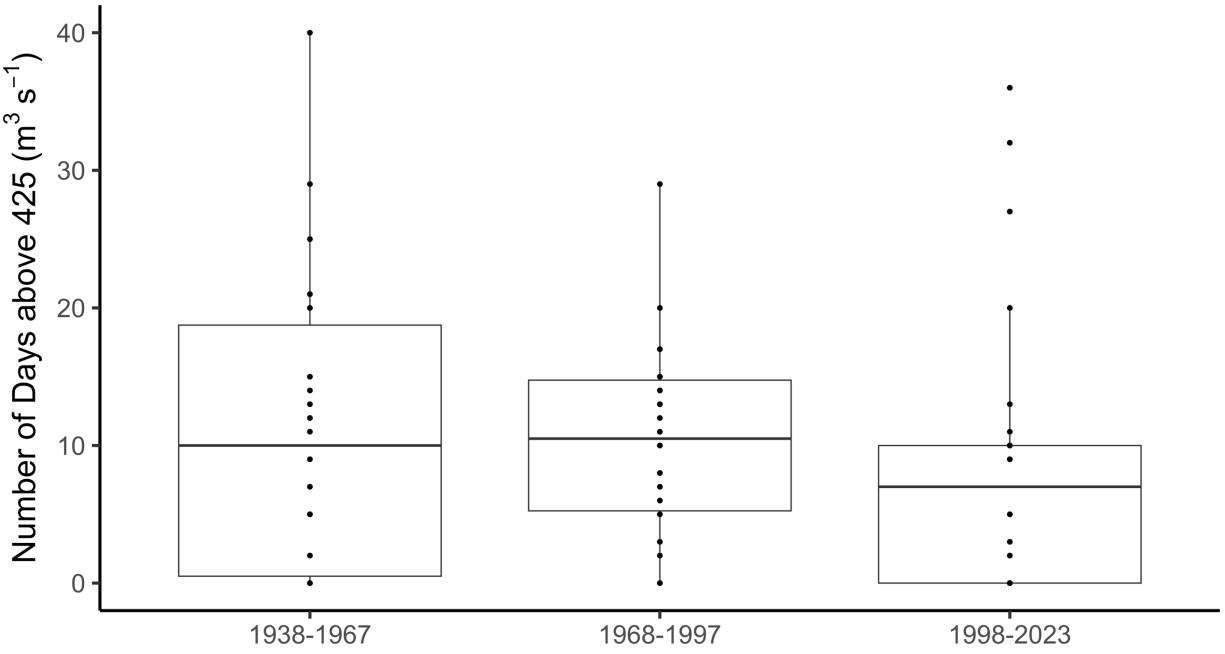
Figure 3.1. The Oconee River Basin is outlined in blue, and the Upper Oconee Regional Water Planning Region is in yellow and drawn based on county lines. The Fall Line denotes the transition from the Piedmont physiographic province (north) and the Coastal Plain physiographic province (south). The two largest hydropower dams are situated near the Fall Line. USGS gages are denoted with red points.





	1938-1967	1968-1997	1998-2023
TOTAL # YEARS	30	30	26
MEAN ANNUAL FLOW (m^3s^{-1})	14.15	15.38	12.66
MEAN ANNUAL RAINFALL (cm)	127.88	122.89	121.28
# YEARS <100 cm OF RAINFALL	2	5	6
# YEARS WITHOUT OCCURRENCE OF FLOWS:			
<7.5 m^3s^{-1}	1	0	0
<2.8 m^3s^{-1}	18	23	7
TOTAL % OF TIME BELOW FLOW:			
<7.5 m^3s^{-1}	38%	32%	47%
<2.8 m^3s^{-1}	4.5%	3.8%	15.1%
MEDIAN # DAYS JUNE - OCTOBER WITH FLOWS:			
<7.5 m^3s^{-1}	104.5	98	110
<2.8 m^3s^{-1}	0	0	18.5

Figure 3.2. Comparison of failure to meet dry-season functional flow thresholds during three time periods in the Middle Oconee River, Georgia, spanning 1938 to 2023. The most recent interval (1998 - 2023) had more years of June to October flows with longer durations under the thresholds for maintaining swift water habitat (7.5 m^3s^{-1} , left) and for drought survival (2.8 m^3s^{-1} , right). The majority of years during the two earlier periods (1938-1967, 1968-1997) did not experience June to October flows below the drought survival threshold, in contrast to an annual June to October median of 18.5 days below this threshold in the recent period (table), with durations extending three or more months in some years.



	1938-1967	1968-1997	1998-2023
TOTAL # YEARS	30	30	26
# YEARS NOVEMBER – MARCH > 425 m^3s^{-1}	22	26	19
MEAN ANNUAL FLOW (m^3s^{-1})	131.8	124.5	103.4
MEAN ANNUAL RAINFALL (cm)	117	120	118
MEDIAN # DAYS, NOVEMBER – MARCH > 425 m^3s^{-1}	10	10.5	7
80^{TH} PERCENTILE # DAYS, NOVEMBER – MARCH > 425 m^3s^{-1}	20.2	15.4	11

Figure 3.3. Comparison of attainment of floodplain habitat connectivity functional flow thresholds during three time periods in the Oconee River, Georgia. The most recent interval (1998 - 2023) had fewer days above the threshold for connecting the river to the floodplain. We also observed a decline in the 80^{th} percentile number of days connecting the river and floodplain (> 425 m^3s^{-1}) over the three time intervals, meaning that over time there were shorter events connecting the river and floodplain.

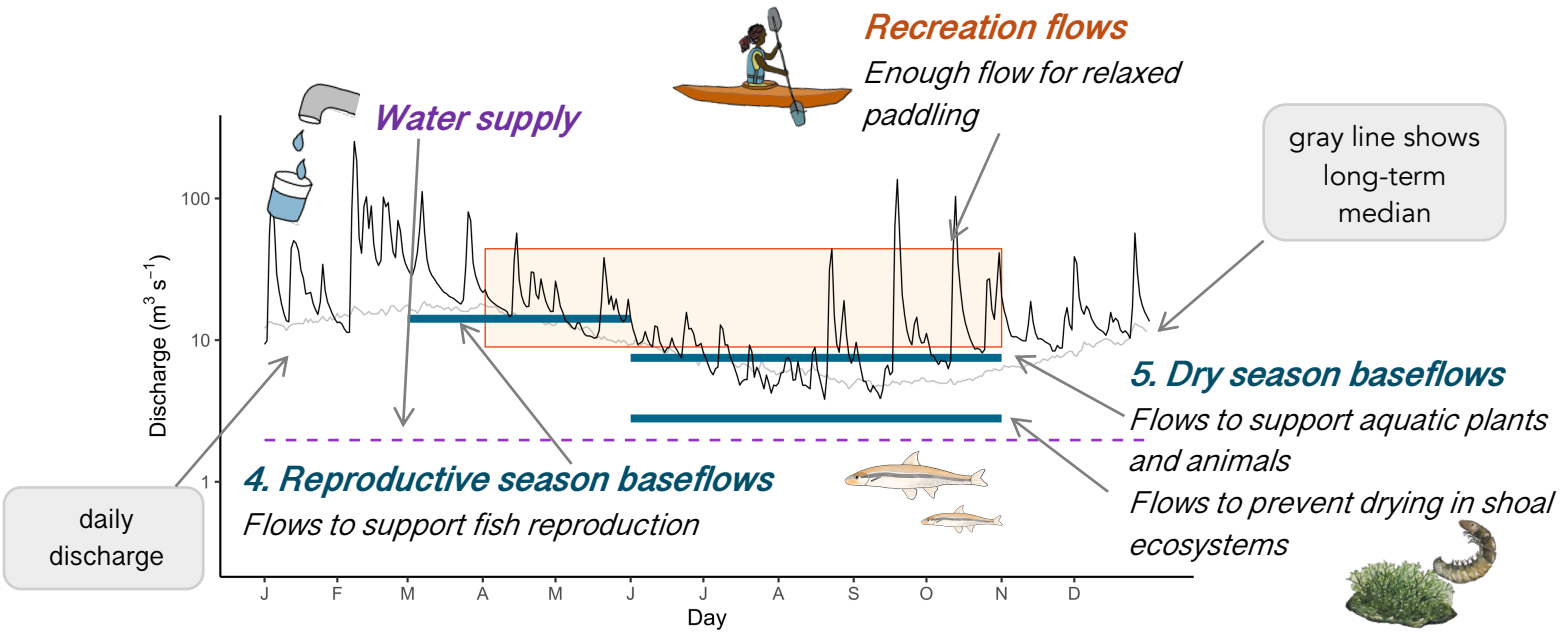


Figure 3.4. Multiple uses supported by the Middle Oconee River ecosystem in the Middle Oconee River, near Athens, GA (USGS gage 02217500; USGS 2024). We displayed the functional flow thresholds available at this reach of the Middle Oconee River alongside other uses quantified in this reach including water supply and flows that support paddling recreation. Visualizing thresholds for multiple uses helps to see how meeting a flow need can support multiple uses.

References

Acreman, M., and M. J. Dunbar. 2004. Defining environmental river flow requirements - a review. Hydrology and Earth System Sciences **8**:861-876.

Acreman, M., A. H. Arthington, M. J. Colloff, C. Couch, N. D. Crossman, F. Dyer, I. Overton, C. A. Pollino, M. J. Stewardson, and W. Young. 2014. Environmental flows for natural, hybrid, and novel riverine ecosystems in a changing world. Frontiers in Ecology and the Environment **12**:466-473.

American White Water. Georgia River List. https://www.americanwhitewater.org/content/River/view/river-index/state/USA-GEO [accessed February 13, 2025]

Anderson, E. P., S. Jackson, R. E. Tharme, M. Douglas, J. E. Flotemersch, M. Zwarteveen, C. Lokgariwar, M. Montoya, A. Wali, G. T. Tipa, T. D. Jardine, J. D. Olden, L. Cheng, J. Conallin, B. Cosens, C. Dickens, D. Garrick, D. Groenfeldt, J. Kabogo, D. J. Roux, A. Ruhi, and A. H. Arthington. 2019. Understanding rivers and their social relations: A critical step to advance environmental water management. WIREs Water **6**.

ARCADIS. 2019. Methods for flow regime evaluation, Ocmulgee River, Georgia. prepared for Georgia Environmental Protection Division

Arthington, A. H., D. Tickner, M. E. McClain, M. C. Acreman, E. P. Anderson, S. Babu, C. W. S. Dickens, A. C. Horne, N. Kaushal, W. A. Monk, G. C. O'Brien, J. D. Olden, J. J. Opperman, A. G. Owusu, N. LeRoy Poff, B. D. Richter, S. A. Salinas-Rodríguez, B. Shamboko Mbale, R. E. Tharme, and S. M. Yarnell. 2024. Accelerating environmental flow implementation to bend the curve of global freshwater biodiversity loss. Environmental Reviews **32**:387-413.

Auerbach, D. A., D. B. Deisenroth, R. R. McShane, K. E. McCluney, and N. LeRoy Poff. 2014. Beyond the concrete: Accounting for ecosystem services from free-flowing rivers. Ecosystem Services **10**:1-5.

Barbour, E. J., L. Holz, G. Kuczera, C. A. Pollino, A. J. Jakeman, and D. P. Loucks. 2016. Optimisation as a process for understanding and managing river ecosystems. Environmental Modelling & Software **83**:167-178.

Bhattacharjee, N., J. R. Willis, E. W. Tollner, and S. K. McKay. 2019. Habitat Provision Associated with Environmental Flows. EMRRP Technical Notes Collection. ERDC/TN EMRRP-SR-85., U.S. Army Engineer Research and Development Center, Vicksburg, Mississippi.

Board of Natural Resources. 2001. Water Issues White Paper. State of Georgia.

Bowen, B. L. 2001. Georgia Riparianism and Irrigation.*in* Proceedings of the 2001 Georgia Water Resources Conference. Institute of Ecology, University of Georgia.

California Environmental Flows Working Group. 2021. California Environmental Flows Framework FAQs. https://mywaterquality.ca.gov/monitoring_council/environmental_flows_workgroup/docs/CEFF_FAQs.pdf [accessed August 17, 2024]

Carlisle, D. M., D. M. Wolock, and M. R. Meador. 2011. Alteration of streamflow magnitudes and potential ecological consequences: a multiregional assessment. Frontiers in Ecology and the Environment **9**:264-270.

Carpenter, S. R., E. H. Stanley, and M. J. Vander Zanden. 2011. State of the World's Freshwater Ecosystems: Physical, Chemical, and Biological Changes. Annual Review of Environment and Resources **36**:75-99.

Cook, J. 2019. Oconee River User's Guide. University of Georgia Press, Athens, GA.

Council, 2023. Upper Oconee Regional Water Plan. Upper Oconee Regional Water Planning Council.

Craig, L. S., J. D. Olden, A. H. Arthington, S. A. Entrekin, C. P. Hawkins, J. J. Kelly, T. A. Kennedy, B. M. Maitland, E. J. Rosi, A. H. Roy, D. L. Strayer, J. L. Tank, A. O. West, and M. S. Wooten. 2017. Meeting the challenge of interacting threats in freshwater ecosystems: A call to scientists and managers. Elem Sci Anth **5**:1-15.

Craven, S. W., J. T. Peterson, M. C. Freeman, T. J. Kwak, and E. Irwin. 2010. Modeling the Relations Between Flow Regime Components, Species Traits, and Spawning Success of Fishes in Warmwater Streams. Environmental Management **46**:181-194.

Cvitanovic, C., A. J. Hobday, L. van Kerkhoff, S. K. Wilson, K. Dobbs, and N. A. Marshall. 2015. Improving knowledge exchange among scientists and decision-makers to facilitate the adaptive governance of marine resources: A review of knowledge and research needs. Ocean & Coastal Management **112**:25-35.

Davies, P. M., R. J. Naiman, D. M. Warfe, N. E. Pettit, A. H. Arthington, and S. E. Bunn. 2014. Flow–ecology relationships: closing the loop on effective environmental flows. Marine and Freshwater Research **65**.

Doyle, M. W., and C. A. Shields. 2007. An alternative measure of discharge effectiveness. Earth Surface Processes and Landforms **33**:308-316.

EA Engineering, S., and Technology, Inc.,. 1994. Sinclair Hydroelectric Project Relicensing Technical Studies (FERC Project No. 1951), Oconee River Instream Flow Study. Georgia Power Company, Atlanta, Georgia.

Enquist, C. A. F., S. T. Jackson, G. M. Garfin, F. W. Davis, L. R. Gerber, J. A. Littell, J. L. Tank, A. J. Terando, T. U. Wall, B. Halpern, J. K. Hiers, T. L. Morelli, E. McNie, N. L. Stephenson, M. A. Williamson, C. A. Woodhouse, L. Yung, M. W. Brunson, K. R. Hall, L. M. Hallett, D. M. Lawson, M. A. Moritz, K. Nydick, A. Pairis, A. J. Ray, C. Regan, H. D. Safford, M. W. Schwartz, and M. R. Shaw. 2017. Foundations of translational ecology. Frontiers in Ecology and the Environment **15**:541-550.

Ferrazzi, M., and G. Botter. 2019. Contrasting signatures of distinct human water uses in regulated flow regimes. Environmental Research Communications **1**.

Fisher, D. S., J. L. Steiner, D. M. Endale, J. A. Stuedemann, H. H. Schomberg, A. J. Franzluebbers, and S. R. Wilkinson. 2000. The relationship of land use practices to surface water quality in the Upper Oconee Watershed of Georgia. Forest Ecology and Management **128**:39-48.

Fletcher, S., A. Hadjimichael, J. Quinn, K. Osman, M. Giuliani, D. Gold, A. J. Figueroa, and B. Gordon. 2022. Equity in Water Resources Planning: A Path Forward for Decision Support Modelers. Journal of Water Resources Planning and Management **148**.

Galatowitsch, M. L., and D. P. Batzer. 2011. Benefits and costs ofLeptophlebia(Ephemeroptera) mayfly movements between river channels and floodplain wetlands. Canadian Journal of Zoology **89**:714-723.

George, R., McManamay, R., Perry, D., Sabo, J. and Ruddell, B.L., 2021. Indicators of hydro-ecological alteration for the rivers of the United States. *E*cological Indicators **120:** p.106908.

Georgia Museum of Natural History. 2021. Fishes of Georgia. http://fishesofgeorgia.uga.edu/.

Georgia River Network. Georgia Water Trails. https://garivers.org/allgeorgiawatertrails/ [accessed February 13, 2025]

Georgia Water Planning and Policy Center. “Waters of the Oconee River Basin: Uses, Benefits, and Flow-Related Metrics for Water Planning.” Albany State University, https://h2opolicycenter.org/projects/waters-of-the-oconee-river-basin/ [accessed July 17, 2024]

Georgia Water Planning. “Regional Water Planning.” State of Georgia, https://waterplanning.georgia.gov/water-planning [accessed July 17, 2024]

Golladay, S. W., L. S. Craig, A. D. DePalma-Dow, B. N. Emanuel, and S. G. Rogers. 2021. Building resilience into water management through public engagement. Freshwater Science **40**:238-244.

Grubaugh, J. W., and J. B. Wallace. 1995. Functional structure and production of the benthic community in a Piedmont river: 1956-1957 and 1991-1992. Limnology and Oceanography **40**:490-501.

Hanna, D. E. L., S. A. Tomscha, C. Ouellet Dallaire, E. M. Bennett, and D. Hooftman. 2017. A review of riverine ecosystem service quantification: Research gaps and recommendations. Journal of Applied Ecology **55**:1299-1311.

Iwanaga, T., H. H. Wang, S. H. Hamilton, V. Grimm, T. E. Koralewski, A. Salado, S. Elsawah, S. Razavi, J. Yang, P. Glynn, J. Badham, A. Voinov, M. Chen, W. E. Grant, T. R. Peterson, K. Frank, G. Shenk, C. M. Barton, A. J. Jakeman, and J. C. Little. 2021. Socio-technical scales in socio-environmental modeling: Managing a system-of-systems modeling approach. Environ Model Softw **135**:104885.

Jackson, C.R., Martin, J.K., Leigh, D.S. and West, L.T., 2005. A southeastern piedmont watershed sediment budget: Evidence for a multi-millennial agricultural legacy. Journal of Soil and Water Conservation, **60**(6), pp.298-310.

Jackson, C.R., Wenger, S.J., Bledsoe, B.P., Shepherd, J.M., Capps, K.A., Rosemond, A.D., Paul, M.J., Welch-Devine, M., Li, K., Stephens, T. and Rasmussen, T.C., 2023. Water supply, waste assimilation, and low-flow issues facing the Southeast Piedmont Interstate-85 urban archipelago. Journal of the American Water Resources Association, **59**(5):1146-1161.

Katz, R. A. 2009. Abundance and survival of common benthic biota in a river affected by water diversion during an hstoric drought. MS thesis, University of Georgia, Athens, GA.

Katz, R. A., and M. C. Freeman. 2015. Evidence of population resistance to extreme low flows in a fluvial-dependent fish species. Canadian Journal of Fisheries and Aquatic Sciences **72**:1776-1787.

King, J. M., R. E. Tharme, and M. S. de Villiers. 2000. Environmental Flow Assessments for Rivers: Manual for the Building Blocks Methodology. Pretoria, South Africa.

Knight, R. R., J. C. Murphy, W. J. Wolfe, C. F. Saylor, and A. K. Wales. 2013. Ecological limit functions relating fish community response to hydrologic departures of the ecological flow regime in the Tennessee River basin, United States. Ecohydrology **7**: 1262-1280

Lancaster, J., and B. J. Downes. 2010. Linking the hydraulic world of individual organisms to ecological processes: Putting ecology into ecohydraulics. River Research and Applications **26**:385-403.

Mathews, R., and B. D. Richter. 2007. Application of the Indicators of Hydrologic Alteration Software in Environmental Flow Setting. JAWRA Journal of the American Water Resources Association **43**:1400-1413.

McKay, S.K., 2015. Quantifying tradeoffs associated with hydrologic environmental flow methods. Journal of the American Water Resources Association, **51**(6):1508-1518.

McKay, S. K., M. C. Freeman, and A. P. Covich. 2016. Application of Effective Discharge Analysis to Environmental Flow Decision-Making. Environ Manage **57**:1153-1165.

McManamay, R. A., and C. R. DeRolph. 2019. A stream classification system for the conterminous United States. Sci Data **6**:190017.

Millennium Ecosystem Assessment, 2005. *Ecosystems and Human Well-being: Synthesis*. Island Press, Washington, DC.

Naiman, R. J., J. J. Latterell, N. E. Pettit, and J. D. Olden. 2008. Flow variability and the biophysical vitality of river systems. C. R. Geoscience **340**:629-643.

O'Keeffe, J. 2009. Sustaining river ecosystems: balancing use and protection. Progress in Physical Geography: Earth and Environment **33**:339-357.

Olden, J.D. and Poff, N.L., 2003. Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. River Research and Applications **19**(2):101-121.

Pahl, J. 2009. Effects of flow alteraton on the aquatic macrophyte *Podostemum ceratophylum* (Riverweed); local recovery potential and regiponal monitoring strategy. MS Thesis, University of Georgia, Athens, GA.

Pahl-Wostl, C., Arthington, A., Bogardi, J., Bunn, S.E., Hoff, H., Lebel, L., Nikitina, E., Palmer, M., Poff, L.N., Richards, K. and Schlüter, M., 2013. Environmental flows and water

governance: managing sustainable water uses. Current Opinion in Environmental Sustainability, **5**(3-4), pp. 341-351.

Palmer, M.A., Lettenmaier, D.P., Poff, N.L., Postel, S.L., Richter, B. and Warner, R., 2009. Climate change and river ecosystems: protection and adaptation options. Environmental Management **44**:1053-1068.

Peterson, J.T. and Freeman, M.C., 2016. Integrating modeling, monitoring, and management to reduce critical uncertainties in water resource decision making. Journal of Environmental Management **183:**361-370.

Petts, G. E. 1996. Water Allocation to Protect River Ecosystems. Regulated Rivers: Research & Management **12**:353-365.

Poff, N. L., D. J. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The Natural Flow Regime a paradigm for river conservation and restoration. BioScience **47**:769-784.

Poff, N. L., and J. K. H. Zimmerman. 2010. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshwater Biology **55**:194-205.

Poff, N. L., R. E. Tharme, and A. H. Arthington. 2017. Evolution of Environmental Flows Assessment Science, Principles, and Methodologies. Pages 203-236 *in* A. Horne, J. A. Webb, M. J. Stewardson, B. D. Richter, and M. Acreman, editors. Water for the Environment. Academic Press.

Portela, A. P., C. Vieira, C. Carvalho-Santos, J. Gonçalves, I. Durance, and J. Honrado. 2021. Regional planning of river protection and restoration to promote ecosystem services and nature conservation. Landscape and Urban Planning **211**.

Power, M. E., A. Sun, G. Parker, W. E. Dietrich, and J. T. Wootton. 1995. Hydraulic Food-Chain Models. **45**:159-167.

Prosser, T., Morison, P.J. and Coleman, R.A., 2015. Integrating stormwater management to restore a stream: perspectives from a waterway management authority. Freshwater Science **34**(3):1186-1194.

Rack, L. E., M. C. Freeman, B. N. Emanuel, L. S. Craig, S. W. Golladay, C. Yang, and S. J. Wenger. 2024. How low is too low? Partnering with stakeholders and managers to define ecologically based low-flow thresholds in a perennial temperate river. River Research and Applications **40**:1393-1416.

Retief, F., Morrison-Saunders, A., Geneletti, D. and Pope, J., 2013. Exploring the psychology of trade-off decision-making in environmental impact assessment. Impact Assessment and Project Appraisal **31**(1):13-23.

Richter, B. D. 2009. Re-thinking environmental flows: from allocations and reserves to sustainability boundaries. River Research and Applications **26**:1052-1063.

Richter, B.D., Davis, M.M., Apse, C. and Konrad, C., 2012. A presumptive standard for environmental flow protection. River Research and Applications **28**:1312-1321.

Rosenfeld, J. S. 2017. Developing flow-ecology relationships: Implications of nonlinear biological responses for water management. Freshwater Biology **62**:1305-1324.

Safford, H. D., S. C. Sawyer, S. D. Kocher, J. K. Hiers, and M. Cross. 2017. Linking knowledge to action: the role of boundary spanners in translating ecology. Frontiers in Ecology and the Environment **15**:560-568.

Shenton, W., N. R. Bond, J. D. Yen, and R. Mac Nally. 2012. Putting the "ecology" into environmental flows: ecological dynamics and demographic modelling. Environ Manage **50**:1-10.

Spinti, R. A., L. E. Condon, and J. Zhang. 2023. The evolution of dam induced river fragmentation in the United States. Nat Commun **14**:3820.

Stalnaker, C. B., B. L. Lamb, J. Henriksen, K. Bovee, and J. Bartholow. 1995. Instream Flow Incremental Methodology A Primer for IFIM. U.S. Department of the Interior, National Biological Service.

The Brisbane Declaration (2007). Environmental flows are essential for freshwater ecosystem health and human well-being. in *10th International River Symposium and International Environmental Flows Conference* (Brisbane, QLD).

TNC (The Nature Conservancy), 2007. User's Manual for the Indi- cators of Hydrologic Alteration (IHA) Software. The Nature Conservancy, Charlottesville, Virginia. http://www.nature.org/ initiatives/freshwater/conservationtools/art17004.html, accessed January 2007.

Tickner, D., J. J. Opperman, R. Abell, M. Acreman, A. H. Arthington, S. E. Bunn, S. J. Cooke, J. Dalton, W. Darwall, G. Edwards, I. Harrison, K. Hughes, T. Jones, D. Leclere, A. J. Lynch, P. Leonard, M. E. McClain, D. Muruven, J. D. Olden, S. J. Ormerod, J. Robinson, R. E. Tharme, M. Thieme, K. Tockner, M. Wright, and L. Young. 2020. Bending the Curve of Global Freshwater Biodiversity Loss: An Emergency Recovery Plan. BioScience **70**:330-342.

Trimble, S. W. 1969. Culturally Accelerated Sedimentation on the Middle Georgia Piedmont. University of Georgia, Athens, GA.

USGS. 2024. National Water Information System. U.S. Geological Survey (USGS) web interface, doi:10.5066/F7P55KJN. Available from https://nwis.waterdata.usgs.gov/nwis [accessed June 2024]

van Rees, C.B., S. Jumani, L. Abera, L. Rack, S.K. McKay, S.J. Wenger. 2023. The potential for nature-based solutions to combat the freshwater biodiversity crisis. PLOS Water 2(6): e0000126. https://doi.org/10.1371/journal.pwat.0000126

Vari, A., S. A. Podschun, T. Eros, T. Hein, B. Pataki, I. C. Ioja, C. M. Adamescu, A. Gerhardt, T. Gruber, A. Dedic, M. Ciric, B. Gavrilovic, and A. Baldi. 2022. Freshwater systems and ecosystem services: Challenges and chances for cross-fertilization of disciplines. Ambio **51**:135-151.

Vogel, R. M., U. Lall, X. Cai, B. Rajagopalan, P. K. Weiskel, R. P. Hooper, and N. C. Matalas. 2015. Hydrology: The interdisciplinary science of water. Water Resources Research **51**:4409-4430.

Water Council. 2008. Georgia Comprehensive State-wide Water Management Plan. Georgia Environmental Protection Division.

Webb, J. A., A. H. Arthington, and J. D. Olden. 2017. Models of Ecological Responses to Flow Regime Change to Inform Environmental Flows Assessments. Pages 287-316 *in* A. Horne, J. A. Webb, M. J. Stewardson, B. D. Richter, and M. Acreman, editors. Water for the Environment: From Policy and Science to Implementation and Managment. Elsevier Academic Press.

Wheeler, K., S. J. Wenger, and M. C. Freeman. 2018. States and rates: Complementary approaches to developing flow-ecology relationships. Freshwater Biology **63**:906-916.

Wildlife Resources Division. 2021. Georgia Rare Species & Natural Community Data. Georgia

Department of Natural Resources. https://georgiawildlife.com/conservation/species-of-concern.

Wineland, S. M., H. Başağaoğlu, J. Fleming, J. Friedman, L. Garza-Diaz, W. Kellogg, J. Koch, B. A. Lane, A. Mirchi, L. F. Nava, T. M. Neeson, J. P. Ortiz-Partida, S. Paladino, S. Plassin, G. Gomez-Quiroga, R. Saiz-Rodriguez, S. Sandoval-Solis, K. Wagner, N. Weber, J. Winterle, and A. M. Wootten. 2021. The environmental flows implementation challenge: Insights and recommendations across water-limited systems. WIREs Water 9.

Wood, J. L., J. W. Skaggs, C. Conn, and M. C. Freeman. 2019. Water velocity regulates macro-consumer herbivory on the benthic macrophyte Podostemum ceratophyllum Michx. Freshwater Biology **64**:2037-2045.

Yang, G., M. Giuliani, and A. Castelletti. 2023. Operationalizing equity in multipurpose water systems. Hydrology and Earth System Sciences **27**:69-81.

Yarnell, S. M., G. E. Petts, J. C. Schmidt, A. A. Whipple, E. E. Beller, C. N. Dahm, P. Goodwin, and J. H. Viers. 2015. Functional Flows in Modified Riverscapes: Hydrographs, Habitats and Opportunities. BioScience **65**:963-972.

Yarnell, S. M., E. D. Stein, J. A. Webb, T. Grantham, R. A. Lusardi, J. Zimmerman, R. A. Peek, B. A. Lane, J. Howard, and S. Sandoval-Solis. 2020. A functional flows approach to selecting ecologically relevant flow metrics for environmental flow applications. River Research and Applications **36**:318-324.

Yaryan Hall, H.R. and Bledsoe, B.P., 2023. Integrated Urban Riverscape Planning: Spatial Prioritization for Environmental Equity. ASCE OPEN: Multidisciplinary Journal of Civil Engineering, 1, p.04023004.

Yearwood, K. I. 2010. River planform change downstream of the Sinclair Dam, Oconee River, Georgia. PhD dissertation, University of Florida, Gainesville, FL.

Yoe, C. 2002. Trade-off analysis planning and procedures guidebook. IWR Report 02-R-2. Alexandria, VA: U.S. Army Institute for Water Resources. https://www.iwr.usace.army.mil/portals/70/docs/iwrreports/02-r-2.pdf.

Zellmer, S. 2008. Legal Tools for lnstream Flow Protection. Pages 285-327 Integrated Approaches to Riverine Resource Stewardship: Case Studies, Science, Law, People, and Policy. The Instream Flow Council.

CHAPTER 4

EVALUATING OPTIONS FOR IMPROVING DROUGHT RESILIENCE OF THE UPPER FLINT RIVER[3]

[3] Rack, L. R., M. C. Freeman, B. N. Emanuel, C. R. Jackson, and S. J. Wenger. Formatted as a grant report to be submitted to the Georgia Environmental Protection Division

Abstract

The upper Flint River is an important water source for multiple uses, including water supply for municipalities south of Atlanta, recreation, and supporting diverse aquatic ecosystems. Five droughts since the late 1990's highlighted the potential vulnerability of the river system to severe drought. Municipal water utilities and others as part of the Upper Flint River Working Group have worked to ensure water security. Working in collaboration with the Upper Flint Regional Water Planning Council and Upper Flint River Working Group, through a seed grant funded by a Georgia Environmental Protection Division, we explored alternative management actions for low-flow and drought resilience in the upper Flint River. We developed three scenarios for evaluation in the Flint Basin Environment Assessment Model (BEAM). In the first scenario we estimated the impact of increased stormwater infiltration. In the second, we simulated additional water storage in a retired quarry near the top of the basin to supplement river flows and in the third we simulated using a higher minimum flow and shifting the timing of water withdrawals from the river. Finally, we combined the stormwater infiltration, quarry storage, and modified operations scenarios to explore the collective impacts of all management actions. We found that the only way to meaningfully enhance river flow during drought events was to change low-flow operations by raising the minimum flow for withdrawal level during the summer and early fall. Altered low-flow operations also resulted in the lowest reservoir storage levels of the three scenarios, however when all scenarios were combined, the impact on reservoir storage was partially offset by releases from the repurposed quarry. Thus, the combination of actions showed the best potential to improve riverine ecological conditions while maintaining adequate water supplies for human needs.

Introduction

In freshwater ecosystems, environmental flows provide an important tool for sustainable water management to meet human needs without degrading river ecosystems – which is particularly important during drought periods when water availability becomes scarce across human and ecosystem needs. Environmental flows are defined as "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" (Arthington et al. 2018). The framing and implementation of the project presented here draw from key elements of holistic environmental flow approaches, which has been recommended for Georgia's Regional Water Planning process because the holistic approach incorporates social, economic, and environmental values (ARCADIS 2019).

The purpose of this project is to identify and evaluate short- and long-term management actions to improve ecological and water resource resilience in the upper Flint River Basin building on the Upper Flint Regional Water Plan and work by the Upper Flint River Working Group (herein "Working Group"). Since their start in 2013, the Working Group has sought to improve the security of water resources of the Flint River for people and nature (American Rivers 2019). Water utilities of the Working Group have implemented projects to return water to the river, upgrade water withdrawal infrastructure, update management practices, undertake proactive drought response and engender cross-jurisdictional communication during drought (American Rivers 2019). In addition, starting in 2018, the group started focusing on the ecological impacts of drought and low flows on Flint River shoal ecosystems – the shallow, rocky expanses that support diverse and abundant fish and wildlife as well as river recreation including boating, wading and angling. In this project we build on the ecologically based low-

flow thresholds developed by the Working Group (see "Guidance on Drought Resilience for People and Nature in the Upper Flint River Basin" presented to the Council in 2021).

The three main project objectives are to:

1. Develop and simulate short- and long-term drought management actions using EPD's Basin Environmental Assessment Model;
2. Predict the ecological consequences of droughts of different severity and duration; and
3. Evaluate how alternative drought-response management actions could mitigate ecosystem effects.

The findings are meant to be useful for the Flint Council in identifying potential actions or combination of actions that may align with the needs or values of basin users to build drought resilience. In addition, our findings are also meant to provide a starting point for the utilities of the Working Group to identify areas of opportunity within their operations to support river flows for ecosystem outcomes. This project represents a first step in evaluating the potential impacts of various management actions that could guide subsequent detailed studies to evaluate the feasibility, cost, and benefits of such actions.

In the following sections, we provide an overview of the three management scenarios evaluated, and detail methods for how we simulated increasing infiltration in the basin, augmenting river flows, and changing low-flow operations by the utilities in the Upper Flint Basin. For each scenario, and for a combination of all management actions, we evaluate ecological metrics, compare the predicted ecological response to a baseline condition, and evaluate modeled reservoir storage to understand how utility operations could be impacted by each management action.

Scenario Development

We leveraged Georgia's existing Flint River Basin Environmental Assessment Model (Flint BEAM) to evaluate how management actions could impact low-flow and drought resilience in the upper Flint River. The Flint BEAM is a linear routing model that simulates daily flows and provides location-specific data for water withdrawals, discharges, and reservoirs in the basin. Water is routed based on the permit limits for withdrawals and discharges and monthly average demand for municipal utilities, agricultural, and industrial permits. BEAM was developed as a long-term planning tool to assess water availability based on the operations in the basin and to evaluate challenges for meeting future demands. The inflows into the basin for Flint BEAM were based on streamflow data between 1938 and 2018, an 80-year time span that included multiple droughts. The simulation in BEAM used water withdrawal and discharge permit levels as of 2018. The water demand was set as the 2011 demand, as reported by utilities, and repeated each year of the scenario. The output of BEAM consisted of daily flows for 80 years at locations of permitted withdrawals in the basin and at long-term USGS gage sites, along with daily reservoir storage volumes. This type of model provides an important tool for planning and is not meant to simulate precisely the daily operations of each individual utility or daily system demand. Instead, it allows for a relative comparison of river flows in the basin based on permit levels and the system demand during a past drought year to identify if there will be challenges in meeting current or future water needs.

We developed three scenarios based on recommendations from the Upper Flint Regional Water Plan (2023) and actions discussed by the Upper Flint River Working Group (Emanuel 2019): (1) increased infiltration across the headwaters of the basin resulting from improved stormwater management in the Atlanta metro area, (2) conversion of a quarry to a water storage

reservoir that could augment flows in the river, and (3) changes to low-flow operations by water utilities. We also evaluated a combination scenario consisting of all management actions in scenarios 1-3.

Scenario 1: Increasing infiltration in the Upper Flint Basin

Rapid population growth in Georgia, and particularly in metropolitan Atlanta, has led to increased impervious surface area, which is associated with non-point source pollution, flooding, and degradation of waterways (Walsh et al. 2005, Jackson et al. 2023). In the Metropolitan North Georgia Water Planning District, which includes the headwaters of the Flint River, developed land was forecast to increase 40% from 2019 levels by 2040 with an 80 to 100% increase in run-off volume (Bell and Gurney 2022). Stormwater management that promotes infiltration into soils can greatly reduce the hydrologic and water quality impacts of impervious surface runoff on streams and rivers. Progressive stormwater regulations that have come about since approximately 2018 in Georgia, and cover most of the Flint River headwaters, will likely increase infiltration for new development, but the Upper Flint has large areas of impervious cover that predate these rules. We were interested in evaluating the effects of retrofitting stormwater management structures or other methods to improve stormwater management in the upper basin.

We used an empirical method to estimate the effect of three levels of increased stormwater infiltration across the basin: 30 mm (1.18 in), 22.5 mm (0.87 in), and 15 mm (0.59 in). River flows in the region typically display baseflow recession from spring through fall due to the drainage of groundwater from the landscape following winter groundwater recharge and high evapotranspiration rates during the summer. Using the fractional drop in monthly flows between April and October over the period of record, we developed representative month-to-month recession rates, excluding those periods where substantial spring/summer rainfall or severe

droughts rendered the recession rates negative or very large. We apportioned the basin-wide volume of increased winter stormwater infiltration depths across the months of April-October using these monthly recession rates, and we added the monthly volumes to the observed baseflows over the period of analysis. The three levels of infiltration we estimated represented modest additions to infiltration that could be achieved through broader application of existing stormwater management practices. To achieve 30 mm of increased winter infiltration, for example, we would need to infiltrate an overage of 1.36 mm (about $1/20^{th}$ of an inch) each for 22 winter storms.

Scenario 2: Augmenting river flows with quarry storage

Vulcan Materials Company operates a large rock quarry in the headwaters located just south of the Atlanta Hartsfield-Jackson International Airport, near the Flint River and Mud Creek confluence (CH2M Hill 2018). Some basin stakeholders have suggested the quarry could serve as a potential reservoir storage for the Upper Flint Basin that could provide water supply, low-flow augmentation, and/or flood control. Although the quarry does not have a decommissioning schedule and there is no formal plan to convert it to use for water storage, the concept has spurred interest and discussion in the basin for several years. Our objective was to estimate the effect of releasing water stored in the quarry on the shoal ecosystems downstream during low-flow periods. To simulate storage operations at Vulcan Quarry, we created a new "reservoir node" in the Flint BEAM (Reservoir Node 6050, Figure 4.1).

Due to the small size of the Flint River and Mud Creek at the quarry site, the quarry would likely need to be filled using a diversion structure rather than pumps (CH2M Hill 2018), however we simulated quarry operations in BEAM by setting bounds on pumping into the reservoir. When selecting the bounds for when to fill the quarry, our goal was to preserve the

median flows in the rivers. We therefore set a maximum daily pumping rate into the quarry and set a pass-by flow between nodes 6100 and 6120 to ensure we were not diverting all water to the quarry (Figure 4.1, Table 4.1). When setting rules for water release, we did not want to release extremely high flows into small channels. We therefore set a maximum daily release rate and set releases to occur when the USGS Carsonville gage in the BEAM scenario was less than 250 cfs, which represents a low summertime flow. All other operations were left the same as in the baseline BEAM with 2011 municipal, industrial, and agricultural demand. The scenario assumed no changes to operations, which meant that quarry releases were available to all downstream users for withdrawals.

Scenario 3: Changes to low-flow operations

In this scenario, our objective was to maintain greater instream flows in the river during low-flow periods. Once flows start declining in the river, the only way to maintain instream flows is to stop pumping water out, so we simulated an increase in the minimum flow level required for municipal water utility operations. We could not simulate drought response actions directly in BEAM because they are primarily based on demand reduction, and we did not have data available to estimate the reduced demand based on such actions. Furthermore, each water utility in the Upper Flint has unique triggers for drought response based on their infrastructure and operations, making drought responses complex to simulate in BEAM. Instead, we changed the low-flow withdrawal limit for each utility to the mean June-October 20^{th} percentile flow (Tables 4.2-4.4). We calculated the 20^{th} percentile flow from the inflow to the relevant reservoir or junction node in the BEAM baseline. It is important to note that these are not recommended changes to permits or operations; this scenario was intended solely as a first-order approximation of the potential effect of operational changes on the ecological outcomes in Fint River shoals. It

is also important to note that although water utilities account for the largest combined volume of water withdrawal in the system, we only changed the operations for water utilities – our scenario does not reflect equal changes to operations across the study area.

Combined Scenario

We wanted to examine the combined impact of all actions on river flows and ecological outcomes in Flint River shoals. We conducted a run in the Flint BEAM that included Vulcan Quarry operations and the changes to the low-flow operations for utilities operating withdrawals in the Upper Flint Basin. We then added the daily infiltration values to the flow values for all years of the model run.

Scenario Evaluation

We used two approaches to evaluate the impact of management actions on ecological outcomes. First, we assessed the number of days river flow was below 100 cfs and 200 cfs at the USGS Carsonville gage for each scenario and compared them to the BEAM baseline scenario. These environmental flow thresholds were developed for the aquatic macrophyte riverweed (see below) and presented to the Council in 2021 in the document "Guidance on Drought Resilience for People and Nature in the Upper Flint River Basin". The 100 cfs threshold represents a condition of "more rocks than water" in the river at Sprewell Bluff, i.e., significant drying in the shoals, and was evaluated in the 2023 Upper Flint Regional Water Plan. We also evaluated 200 cfs, the point at which about 50% of the shoal at Sprewell Bluff is exposed, to understand how actions were affecting low-flow levels that occurred more frequently. Second, we developed and applied a predictive model for day-to-day change in riverweed biomass based on flow conditions to the flow outputs from each scenario to simulate the resulting biomass in a typical Flint River shoal. Model development and evaluation are discussed in more detail in the following section.

We evaluated the effect of scenarios 2 and 3 on the water availability and reservoir storage levels of four utilities: Clayton County Water Authority, Fayette County Water System, Newnan Utilities, and City of Griffin. These utilities had water withdrawal operations on the upper Flint River or tributaries and were also members of the Upper Flint River Working Group. We reported the number of days each year reservoir storage was at or below drought level 2, based on levels identified in the utilities' Drought Contingency Plans, required by the Environmental Protection Division. We also solicited information about the individual reservoir or combined storage levels that were of concern for the utilities' operations and summarized the impact on reservoir storage.

Ecological Model Development

We developed an ecological model for Flint River shoal ecosystems using the response of the submerged aquatic plant riverweed (*Podostemum ceratophylum*), that grows abundantly in shoal ecosystems (Nelson and Scott 1962, Grubaugh and Wallace 1995). Riverweed grows in swift flowing water and provides habitat for aquatic invertebrates and fishes; it is also vulnerable to low flows and desiccation (Wood et al. 2019, Pahl 2009, Argentina et al. 2010). Riverweed has been referred to as a foundation species (Wood and Freeman 2017) and serves as a promising low-flow indicator for shoal ecosystem condition both due to its key ecological role and because past studies make it possible to develop quantitative relationships between flow variables and riverweed biomass.

We estimated growth rate of riverweed biomass in relation to shoal water velocities using data from the Middle Oconee River near Athens GA (Appendix B, Part 1), which is similar in size and geology to the Upper Flint and has been the site of four separate studies of monthly

changes in riverweed biomass. We then used the relationship between discharge and velocities in the Flint River shoals to estimate change in biomass of riverweed based on flow conditions. Our model had two components:

1. Daily flows for a scenario were used to project daily net change in riverweed biomass for the years 2009 to 2018. Daily net change in biomass was used to simulate biomass standing stock during each annual growing period, from an arbitrary beginning amount (e.g., 1000 mg ash-free dry mass per square meter).
2. Simulated standing stock biomass at the end of each autumn was multiplied by the lowest 30-day average proportion of shoal width estimated to maintain flow in the Flint River shoals at Sprewell Bluff. We assumed that drying for 30 days leads to complete loss of the plant (Pahl 2009), and so the smallest area of shoal that retained flow across the season (drying for less than 30 days) was the area that could support the simulated standing stock riverweed biomass.

We used the outputs at the USGS Carsonville gage for all scenarios from 2009 to 2018 to predict riverweed biomass. This time-period encompassed a one multi-year drought from 2011-2012 and a flash drought that occurred in the summer of 2016.

We include expanded methods and R and Jags code for the model in Appendix B, Part 1.

Scenario Results

The Flint BEAM does not simulate the daily actions by the permittees. We therefore used BEAM to evaluate the relative change between the baseline scenario and our three management action scenarios for ecological metrics and reservoir storage for utilities. In the baseline scenario, flows below 200 cfs occurred in 26 years of the 80-year model period, often during drought

periods. The annual duration ranged from one day to 160 days, with longer annual durations of flows below 200 cfs after the year 2000 compared to the previous years. Flows below 100 cfs occurred in 10 years of the 80-year model period, starting in 1986, with annual duration ranging from 6 to 80 days.

When we estimated the impact of increased infiltration in the basin (scenario 1), we saw the greatest contributions to baseflow in the early spring. Effects tapered off through summer and fall (Tables 4.5 and 4.6). Based on the median monthly discharge from the baseline scenario, 30 mm of increased infiltration annually at the Carsonville gage would result in a 40% increase for the median April flow and a 15% increase in October (Table 4.6). We chose not to evaluate this scenario using the 100 and 200 cfs thresholds at Carsonville since these severe low-flow levels are not appropriate metrics for the time of year (i.e., early spring) that infiltration has the greatest impact on baseflow. Because our estimated values for infiltration were added to BEAM outputs, we were unable to evaluate the impacts on water utilities (i.e., for meeting system demand and reservoir storage).

Scenarios 2, 3, and combined scenario

We present the results of scenarios 2, 3, and the combined scenario together since we could evaluate the interactions among river flows, ecological outcomes, and water allocation.

Ecological Outcomes

Changes to low-flow operations, scenario 3, had the greatest impact on river flows. We saw reductions in the number of days and years with flows below 100 cfs and 200 cfs (Figures 4.2 and 4.3). Quarry operations, scenario 2, also resulted in a decrease in the number of days below 200 cfs, but did not change the days below 100 cfs. This is because we did not change utility operations for the quarry scenario (scenario 2), so the additional water released from the

quarry was available for use by utilities, and the simulation indicated that most of the released water would be withdrawn. The combined scenario primarily reflects the changes to river flows from the low-flow operations. Our results reflect that once flows start to decline there are limited options to keep water in the river, so early actions that support reducing withdrawals have the largest impact on instream flows.

The duration, magnitude, and frequency of events are important factors to evaluate for ecological metrics. We evaluated the 80th and 90th percentile annual number of days below 200 cfs, which is a very low flow but one that occurs more often than 100 cfs during the period of record, for the 80-year run in BEAM. Since these events do not occur in most years, we wanted to compare how long events occur (the duration) when they happen. We observed that quarry operations resulted in shorter events (# of days per year) for the 80th percentile (but not the 90th percentile) number of days. The low flow operations (scenario 3) and all scenarios reduced both the 80th and 90th percentile annual number of days below 200 cfs by more than 10 days (Figure 4.4).

Similarly, 100 cfs is an even more extreme low flow in the Flint River and only occurred in 10 years of the 80-year run in BEAM. Changing the low flow operations (scenario 3) reduced the duration of low-flow events and eliminated some years with excursions below 100 cfs (Figure 4.5). Quarry operations did not change the number of years (frequency) or duration of events below 100 cfs (Figure 4.5).

Reservoir Storage and Utility Operations

The average monthly system demand, as reported by utilities in 2011, was met for the baseline, scenarios 2-3 and all scenarios combined. In the baseline scenario only Clayton County Water Authority combined reservoir storage was below drought level 2 during the model period;

the other utilities' storage did not drop below their drought level 2 threshold. The change in operations we simulated (scenarios 2 and 3) did not impact the ability to meet the volume of water demand at the utility locations we evaluated in the basin, however there were differences in reservoir storage levels between scenarios, with the quarry operations supplementing reservoir storage and the low-flow operations resulting in storage levels that would cause concern for utility operations.

There are often multiple factors, e.g., river flows, reservoir storage, previous rainfall conditions, etc., in municipal utility drought contingency plans used to trigger drought response. We compared the annual number of days at or below drought level 2 reservoir storage for the 80-year simulation to the baseline scenario for all utilities (Table 4.7). We found that the low-flow operations (scenario 3) increased the number of years reservoir levels were at or below drought level 2, but the number of days and storage volumes were offset by quarry releases in the combined scenario for Clayton County Water Authority, Fayette County Water System, and City of Griffin. Quarry operations did not impact the withdrawal location for Newnan Utilities. The degree of impact on reservoir storage varied by utility. Clayton County Water Authority experienced the greatest decline in reservoir storage with the change in low-flow operations, followed by Fayette County Water System, and the Heads Creek Reservoir for the City of Griffin.

Clayton County Water Authority

Clayton County combined reservoir storage was at or below 75% (drought level 2) in 7 years out of 80 years for the baseline scenario, 4 years for the quarry scenario, 19 years in the low-flow operations scenario, and 15 years in the combined scenario. Low flow operations increased the number of days and years under 75% and resulted in lower storage volumes in the

reservoir. Quarry operations reduced the number of years below 75% compared to baseline. The combined scenario reduced the number of years below 75% and increased the minimum reservoir storage compared to low-flow operations alone (Figure 4.6, Table 4.8).

Clayton County combined storage at or below 85% was of interest to the utility and followed a similar pattern to storage below 75%. Low-flow operations increased the number of days and years below 85%, but these were partially offset in the combined scenario with the quarry. Quarry operations reduced the number of days and years below 85% (Figure 4.7).

Fayette County Water System

We evaluated the storage volume in Lake Horton to represent the series of reservoirs operated by Fayette County. Reservoir storage in drought level 2, or 5 ft below reservoir pool level, occurred in three years in the low-flow operations (scenario 3) and two years in the combined scenario. There were zero years with days below 5ft for the baseline and quarry scenario (Figure 4.8).

Reservoir elevation for Lake Horton was 2 ft below pool in one year in the baseline scenario. The quarry scenario reduced this to zero years, whereas the low-flow operations scenario had 14 years below the threshold and the combined scenario had 10 years below the threshold (Figure 4.9).

We also tracked how pool levels changed in Lake Horton and Lake Kedron for all scenarios in BEAM (Figures 4.10-4.13). Monitoring how closely the reservoir levels tracked through the scenarios was of interest to the utility.

Newnan Utilities

There were zero years with reservoir storage below 70% for the baseline scenario and two years in the low-flow operations scenario (Figure 4.14). Newnan's operations in the Flint Basin are unaffected by the quarry releases.

City of Griffin

The City of Griffin's Still Branch Reservoir showed the smallest impact from the low-flow operations (scenario 2). Combined storage was not below the 60% or 70% storage level for baseline or any scenario in BEAM. The combined storage of the Still Branch and Heads Creek Reservoirs was also not below the 60% or 70% storage level for the baseline or any scenario in BEAM.

Heads Creek Reservoir was below the 60% storage for 6 years for the low-flow operations (scenario 3) scenario and 3 years during the combined scenario, and primarily occurred during the recent droughts in the 2000's (Figure 4.15). The results were similar for years below 70% storage, with 7 years for the low-flow operations (scenario 3) and 3 for the combined scenario (Figure 4.16). Heads Creek Reservoir was not below 60% or 70% for the baseline or quarry scenario (Figures 4.15 and 4.16).

Ecological Model Results

The ecological model provided a simulation of riverweed dynamics under the different management scenarios. Our model of Flint River riverweed biomass from 2009 to 2018 showed that biomass peaked in the winter and was lowest in the summer. The summer and winter biomass values were lower during drought years (2011, 2012, and 2016, Figure 4.17). These patterns were consistent across all scenarios and illustrated general growth dynamics, so we only displayed the daily biomass for the baseline scenario (Figure 4.17). For each scenario, we compared the annual minimum standing stock biomass of riverweed adjusted for the extent of

shoal drying (Figure 4.18). In our three drought years, we saw the greatest increase in riverweed biomass in 2016 from the low-flow operations (scenario 3) and the combined scenario as compared to the baseline, with a 54 and 50% increase respectively. We also saw some increases in non-drought years, such as in 2009 when the flows from scenario 3 and the combined scenario once again led to the greatest increase in biomass compared to the baseline. It is also worth noting that we see small increases in biomass for scenarios 1 and 2 in most years. This is likely due to their effects on moderately low flows, which were important for determining the extent of the shoal that was wet during the summer.

Conclusions

We evaluated the relative impact of management actions on aquatic ecosystems and water utility operations during low-flow and drought periods, when it is challenging to meet human water needs and support aquatic ecosystems. We found that each of the three scenarios, and the combined scenario, provided unique but often complementary outcomes for the Upper Flint River Basin. Changes to low-flow operations (Scenario 3) was the only scenario to mitigate extreme low flows in the Flint River, with reductions in number and duration of events under 200 and 100 cfs. This scenario also resulted in the lowest reservoir storage, but we found the storage declines were partially offset in the Combined Scenario due to augmentation to river flows from the quarry and stormwater infiltration. Quarry operations alone (Scenario 2) provided modest flow increases in the river, but our model indicated that most of the released water would be withdrawn by utilities, buffering their storage capacity. Increasing infiltration (Scenario 1) had the greatest impact on springtime river flows between April and June. While converting water to baseflow rather than runoff is important for the river ecosystem, we could not assess the impacts to utilities since the results were applied to BEAM outputs, rather than simulated within BEAM.

The development and implementation of the Flint BEAM for water planning allowed us to evaluate water quantity at the scale of individual water utilities and the relative difference in water availability under alternative management actions. BEAM was useful for investigating the relative difference of management actions on utility operations and ecological outcomes, and for identifying when it may be useful to conduct more detailed study on specific operations. We found BEAM was most useful for exploring how different management actions could impact streamflow and reservoir storage on average. However, to understand how specific operations impact river flow, reservoir storage levels, and system demand on a daily time step would require additional information to input into BEAM or an alternative model outside of BEAM. For example, BEAM currently routes water based on the permitted limits and the daily time step is based on the monthly data reported by utilities. To more closely reflect the operation by specific utilities, we would need to develop information with utilities about general day-to-day operations based on a combined reservoir and river level during different times of year, which could then be translated to operations in BEAM. This general approach could be useful to other regional planning councils to evaluate management alternatives that are specific to each basin (also see Appendix B, Part 2 for general guidance for evaluating ecological indicators in regional water planning).

Water utilities in the Working Group have seen growth in their customer base since the 2022 planning cycle and are expecting continued growth in certain areas of the Flint Basin. With this in mind, utilities are interested in updating the drought demand values (which are based on year 2011) to evaluate meeting water needs during low-flow periods. In addition, “flash droughts” or periods of high heat and a sudden lack of rainfall have become more common during the summers and have created a different type of management challenge. Recognizing

that this type of event will continue, Working Group participants are interested in exploring how these types of events may interact with regard to drought planning. In future iterations of Flint BEAM, it would also be helpful to investigate how the drought response plans could be simulated within the BEAM framework. Since the plans are utility-specific and partly based on actions that lead to reduced demand from customers, a combination of information may be needed to simulate changes to demand in BEAM alongside operations shifts.

Table 4.1. The operations set in BEAM to control the quarry filling and releases

Operations	Minimum flow	Maximum daily flow
Filling	*Set as a passby between nodes 6100 and 6120*: minimum flow is 10 cfs	*Tiered pumping operations:* No pumping when inflow to 6100 is 10 cfs Daily pumping can take 50% of inflow to 6100 when inflows are between 11-45 cfs, Daily pumping max is 20 cfs when inflows to 6100 are greater than 45 cfs
Releases	No minimum flow	Release 0 when Carsonville (node 7281) is greater than 250 cfs Release 50 cfs when Carsonville (node 7281) is less than 250 cfs

Table 4.2. Pass-by flows required for reservoirs as they are set up in BEAM; utilities can withdraw water if flows passing downstream are at least this level.

Junction/Reservoir Node	Name	Utility	Operation in baseline	Operation in Scenario 3
6260	Horton Creek Reservoir	Fayette	Passby flow: 30 cfs	Passby flow: 42.36 cfs
6300/6305	Heads Creek Reservoir	Griffin	Passby flow: 10 cfs	Passby flow: 46.94 cfs
6680	Still Branch	Griffin	Minimum passby flow when reservoir storage is below 70%: Monthly passby flow (cfs): June - 60 July - 60 August – 60 September – 60 October – 60 November - 60	Minimum passby flow when reservoir storage is below 60%: Monthly passby flow (cfs): June - 213 July - 171 August - 129 September - 122 October – 122 Nov - 187
6640	White Oak Creek withdrawal	Newnan	Passby flow: 1.9 cfs	Passby flow: 11.6 cfs

Table 4.3. Tiered passby flows based on pumping volumes and river flows; water can be withdrawn when flows are at or above this level.

Reservoir Node	Name	Utility	Operation in baseline	Operation in Scenario 3
6180	J. W. Smith Reservoir	Clayton	Tiered minimum based on pumping rates: 1st tier pumping 0-6MGD Passby flow: 12 cfs	Tiered minimum based on pumping rates: 1st tier pumping 0-6MGD Passby flow: 26.5cfs No change to 2nd and 3rd tiers
6340	Line Creek withdrawal	Newnan	Tiered pumping structure based on river flow: Withdrawal 0: 0 – 2 MGD in river Withdraw up to 50% of river flows: 2-24 MGD in river Withdrawal 12 MGD: 24 MGD and up	Tiered pumping structure based on river flow: Withdrawal 0: 0–3.7 MGD in river Withdraw up to 25% if river flows: 2-24 MGD in river Withdrawal 12 MGD: 24 MGD and up

Table 4.4. Required outflow from reservoir, this was set as the annual 15th percentile flow, and the release rate is the smallest value based on the inflow or the value in the table.

Reservoir Node	Name	Utility	Operation in baseline	Operation in Scenario 3
6260	Lake Horton	Fayette	2.6 cfs or natural inflow	4 cfs or natural inflow
6380	Lake McIntosh	Fayette	4.64 cfs or natural inflow	13 cfs or natural inflow
6440	Lake Kedron	Fayette	1.6 cfs or natural inflow	4 cfs or natural inflow

Table 4.5. The median monthly flow and the monthly additions to baseflow in CFS for the Carsonville and Molena gages based on additional storage of 30 mm, 22.5 mm, and 15 mm in the basin.

Site	April	May	June	July	Aug	Sept	Oct
Carsonville	2799	1542	1195	1090	906	659	668
30 mm	1228	737	442	309	247	124	99
22.5 mm	921	552	331	232	186	93	74
15 mm	614	368	221	155	124	62	49
Molena	1103	558	357	354	218	227	240
30 mm	640	372	192	111	78	40	27
22.5 mm	480	279	144	84	58	30	20
15 mm	320	186	96	56	39	20	14

Table 4.6. The percent increase in monthly median discharge with 30 mm of additional storage for the Carsonville and Molena gages.

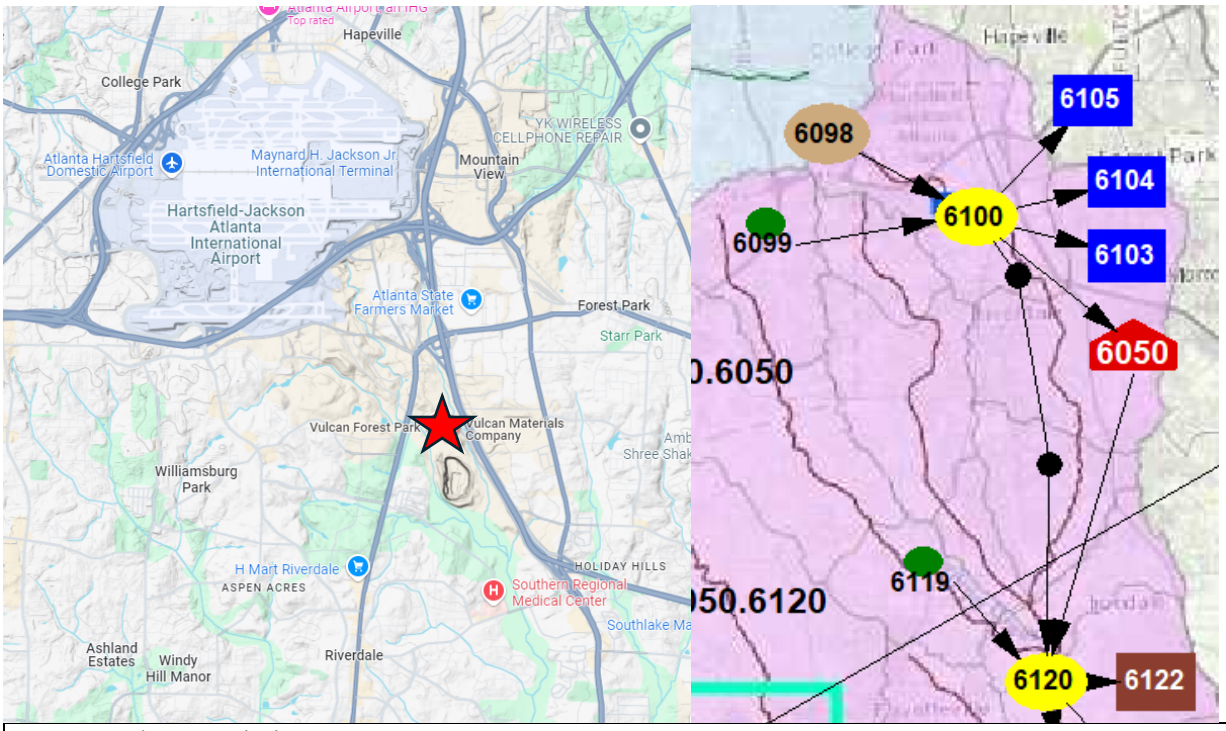
Site	April	May	June	July	Aug	Sept	Oct
Carsonville							
30 mm	40%	48%	37%	28%	27%	19%	15%
Molena							
30 mm	58%	67%	54%	31%	36%	18%	11%

Table 4.7. Reservoir storage volumes associated with drought level 2 for the four utilities with withdrawals in the upper Flint River.

Utility	Drought level 2 reservoir storage	Additional storage volumes to evaluate
Clayton County Water Authority	75%	85%
Fayette County Water System	5 ft	2 ft
Newnan utilities	70%	NA
City of Griffin	60%	70%

Table 4.8. The years Clayton County combined reservoir storage was under 75% and the minimum combined storage volume and percent storage in parentheses for each scenario in BEAM. The total number of years below 75% for a scenario is in parentheses.

Year	Baseline (7)	Quarry (4)	Low flow ops (19)	All scenarios (15)
1940			6601 (71%)	
1941	6485 (70%)		4437 (48%)	6307 (68%)
1954	3447 (37%)	6370 (68%)	404 (4%)	4342 (47%)
1955	4836 (52%)		1637 (18%)	5617 (60%)
1986			6629 (71%)	
1987			5541 (60%)	6838 (73%)
1988			5418 (58%)	5916 (64%)
1993			5584 (60%)	
1999			6212 (67%)	
2000			5165 (55%)	5579 (60%)
2001			4612 (50%)	5694 (61%)
2002			5407 (58%)	6491 (70%)
2007			4150 (45%)	4690 (50%)
2008			5580 (60%)	6122 (66%)
2011	5798 (62%)	6656 (71%)	3252 (35%)	5325 (57%)
2012	6151 (66%)	6259 (67%)	1899 (20%)	3100 (33%)
2013			4446 (48%)	5670 (61%)
2016	4754 (51%)	6001 (64%)	2760 (30%)	5360 (58%)
2017	6400 (69%)		4316 (46%)	6939 (75%)



Quarry Characteristics	
Location	33°35'49.9"N 84°23'45.8"W
Volume	5 billion gallons; 14,300 ac-ft capacity

Figure 4.1. Location of Vulcan Quarry in the Flint Basin (left) represented by the red star and as it was represented in BEAM (right) as a red pentagon. The Quarry is situated at the very top of the headwaters, just south of the Atlanta Airport, with a storage capacity of 5 billion gallons (Table).

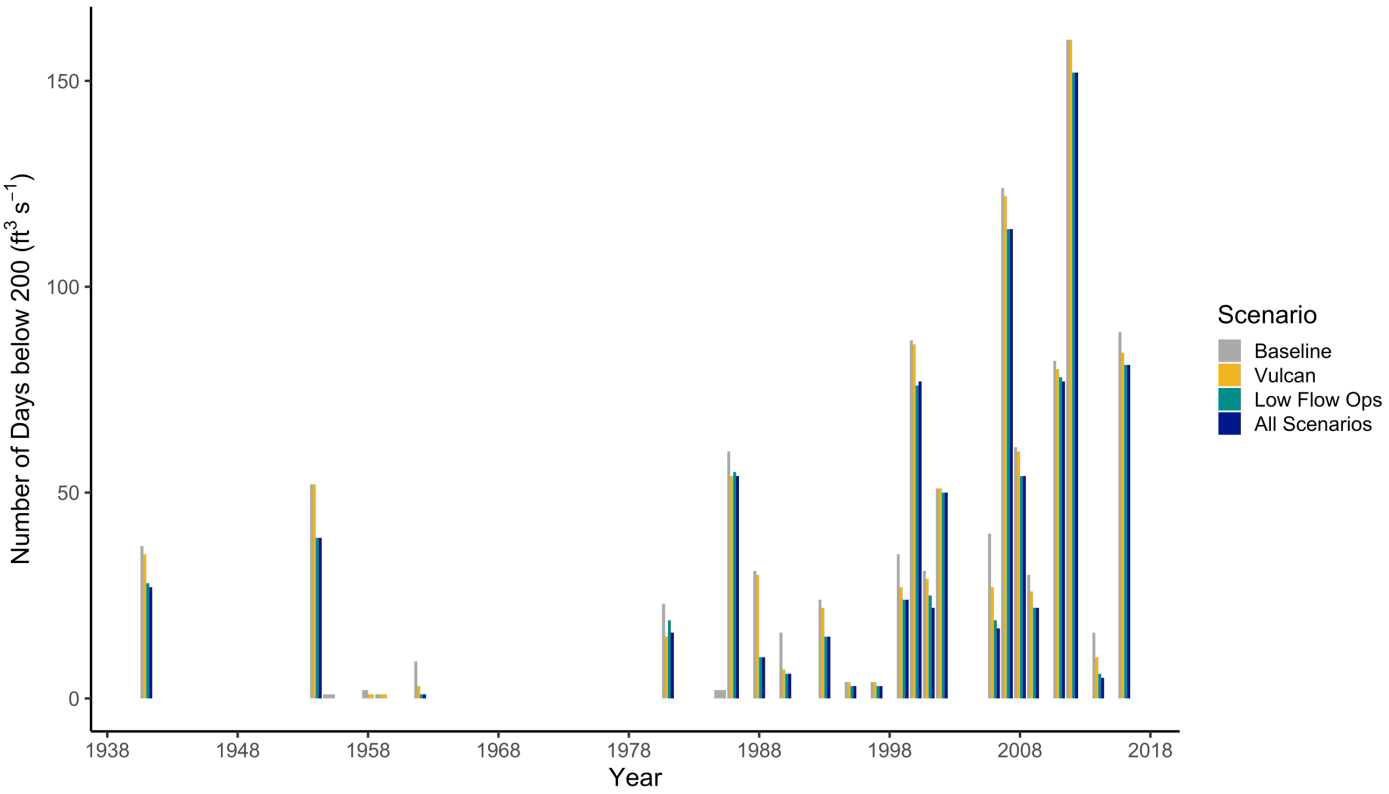


Figure 4.2. The number of days below 200 cfs at the Carsonville gage site in the baseline BEAM scenario and scenarios 2, 3 and all scenarios. Days with 200 cfs started in 1941 and we see the fewest days each year with the Low-Flow Ops (scenario 3) and All Scenarios.

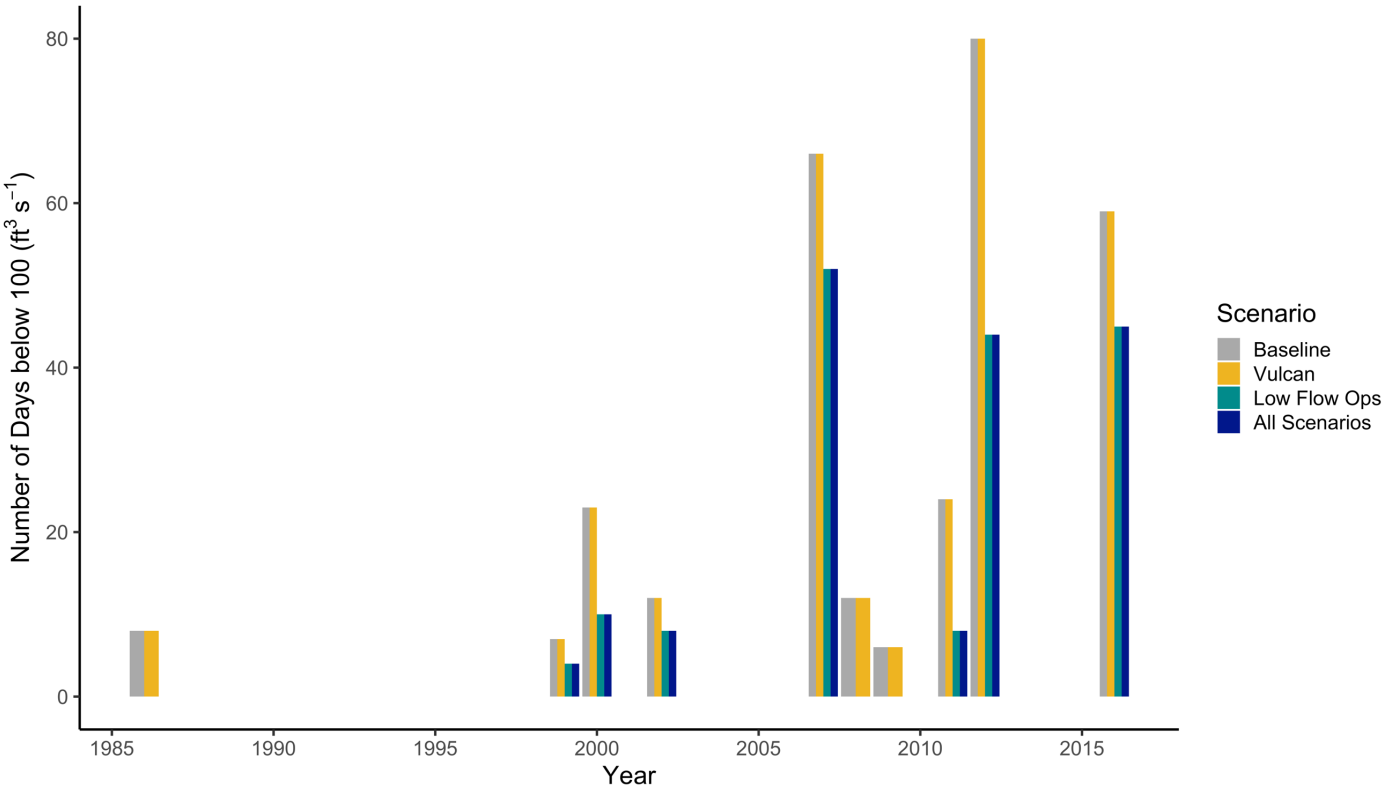


Figure 4.3. The number of days below 100 cfs (bottom) at the Carsonville gage site in the baseline BEAM scenario and scenarios 2, 3, and all scenarios. Days below 100 cfs did not begin until the mid 80's and again we see that the fewest days occurred under scenarios 3 and the combined scenario.

Scenario	Annual number of days below 200 cfs (80th percentile)	Annual number of days below 200 cfs (90th percentile)
Baseline	23.2	51.1
Vulcan	16.4	51.1
Low-Flow Ops	11	40.1
All scenarios	11	40.1

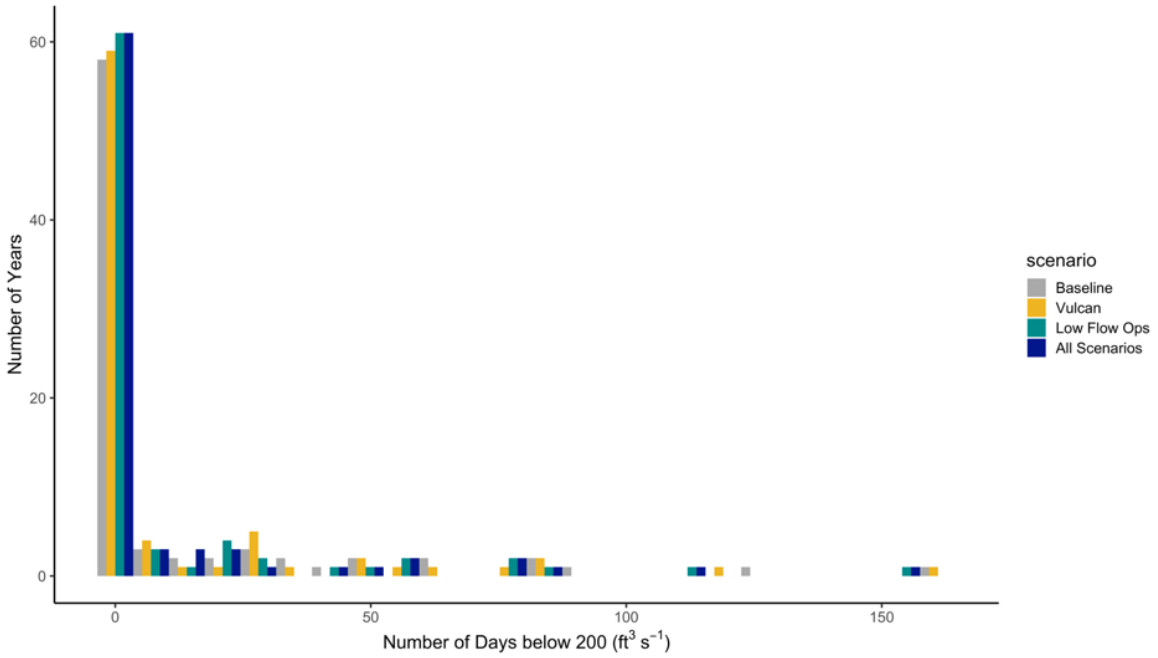


Figure 4.4. The 80th and 90th percentile annual number of days with flow below 200 cfs (Table) and histogram of the annual number of days below 200 cfs for each scenario. In most years flows below 200 did not occur. We observed the greatest reduction in the number of years and the number of days below 200 cfs from the low-flow operations and all scenarios. Vulcan quarry operations also reduced the number of days and years under 200 cfs.

Scenario	Annual number of days below 100 cfs (80^{th} percentile)	Annual number of days below 100 cfs (90^{th} percentile)
Baseline	0	7.1
Vulcan	0	7.1
Low-Flow Ops	0	0
All scenarios	0	0

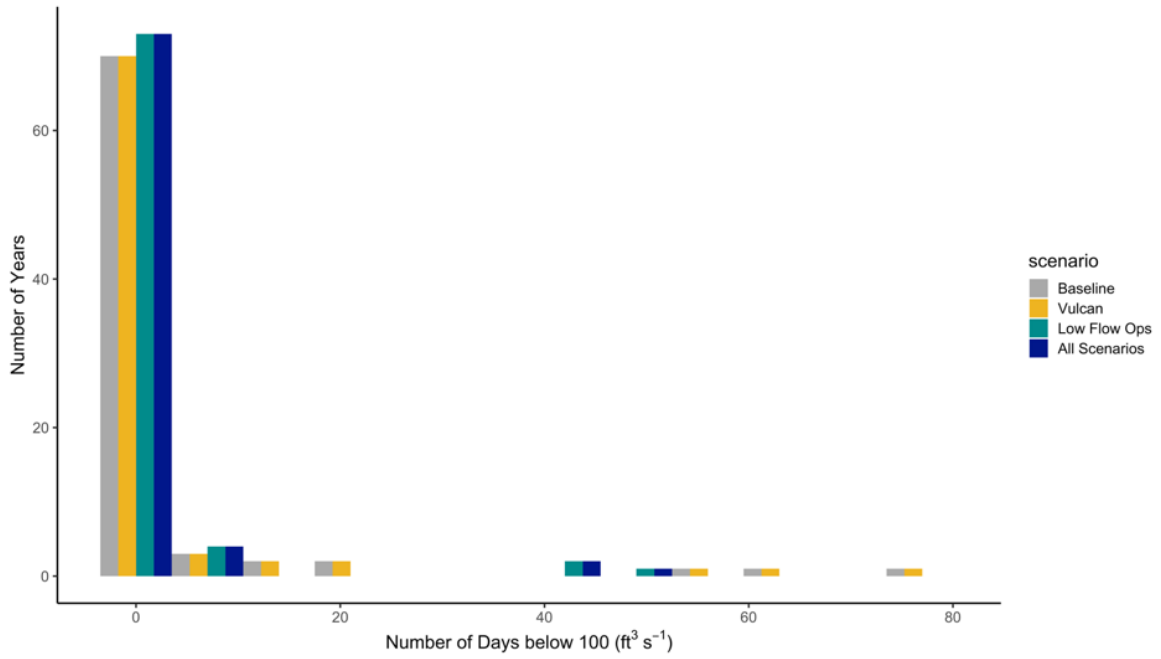


Figure 4.5. The 80^{th} and 90^{th} percentile annual number of days with flow below 100 cfs (Table) and histogram of the annual number of days below 100 cfs for each scenario. Flows below 100 cfs did not occur in most years. We saw a reduction in annual number of days and years under 100 cfs with low-flow operations and all scenarios, which eliminated more than 50 days of flow below 100 cfs. We did not see a change with Vulcan quarry.

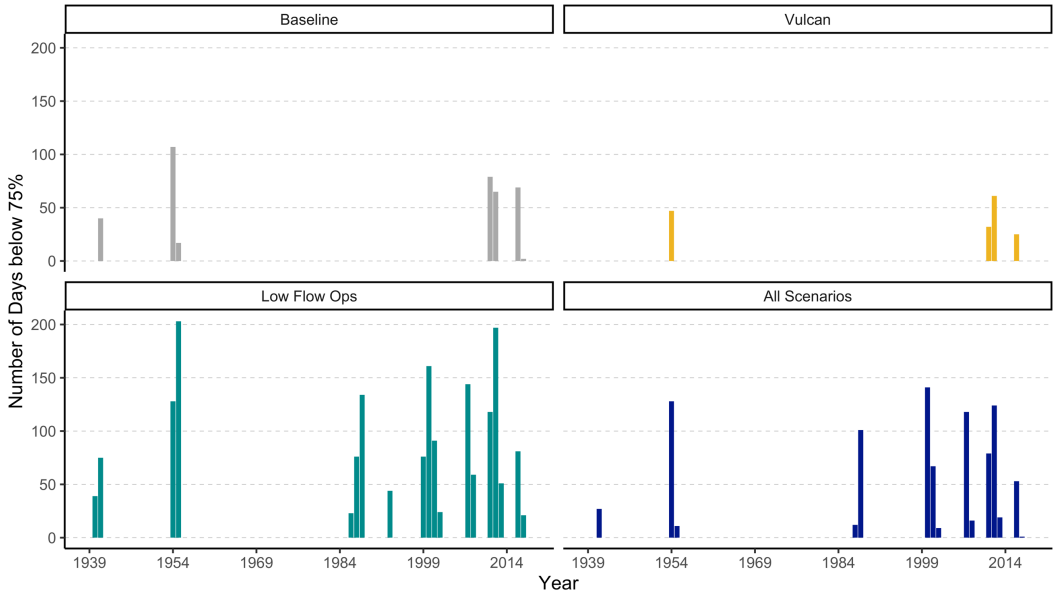
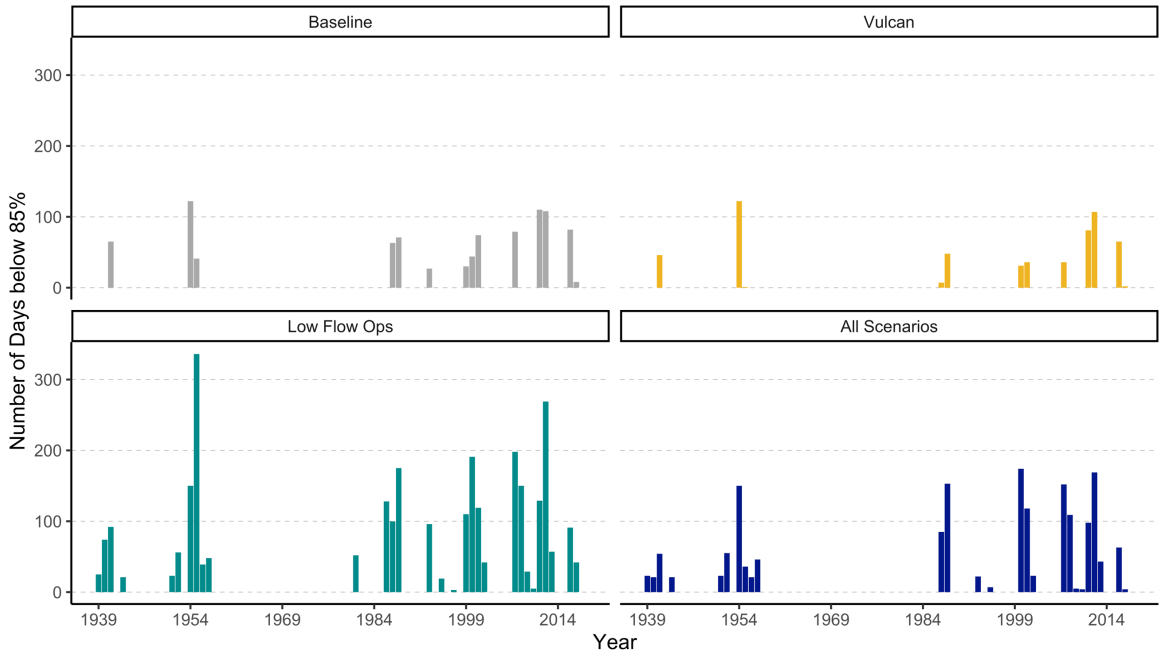
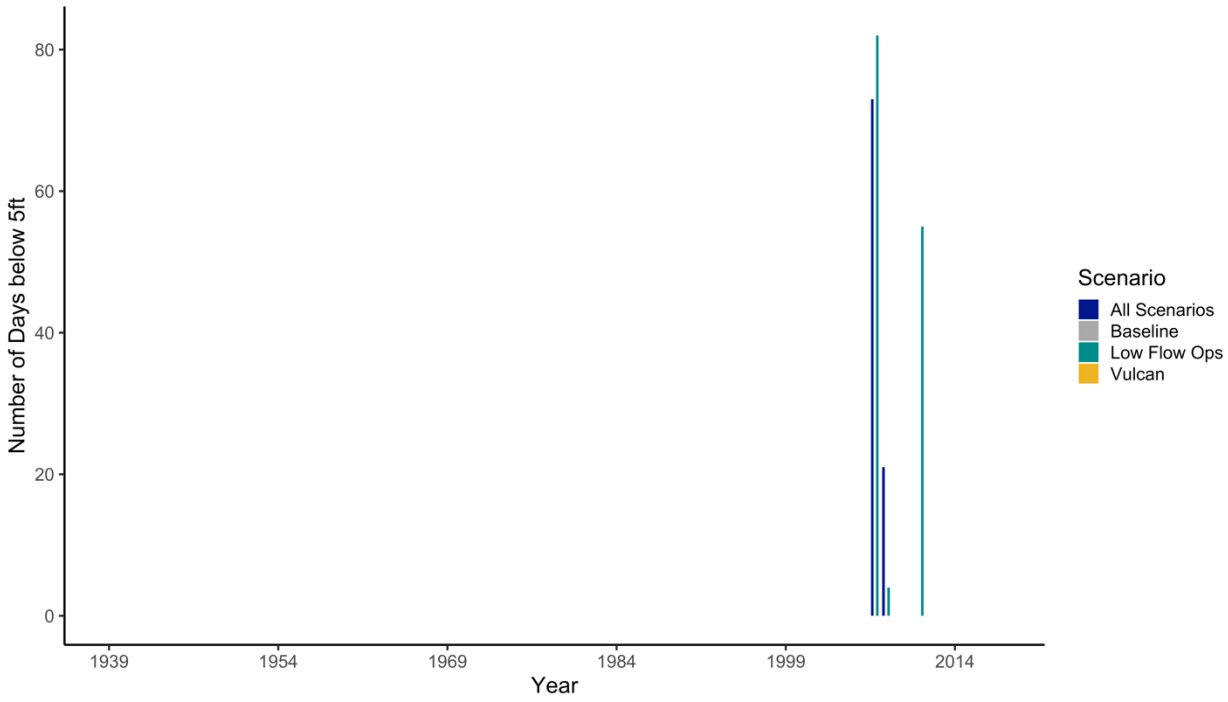


Figure 4.6. Number of days with combined reservoir storage below 75% for Clayton County reservoirs in each scenario in Flint BEAM.



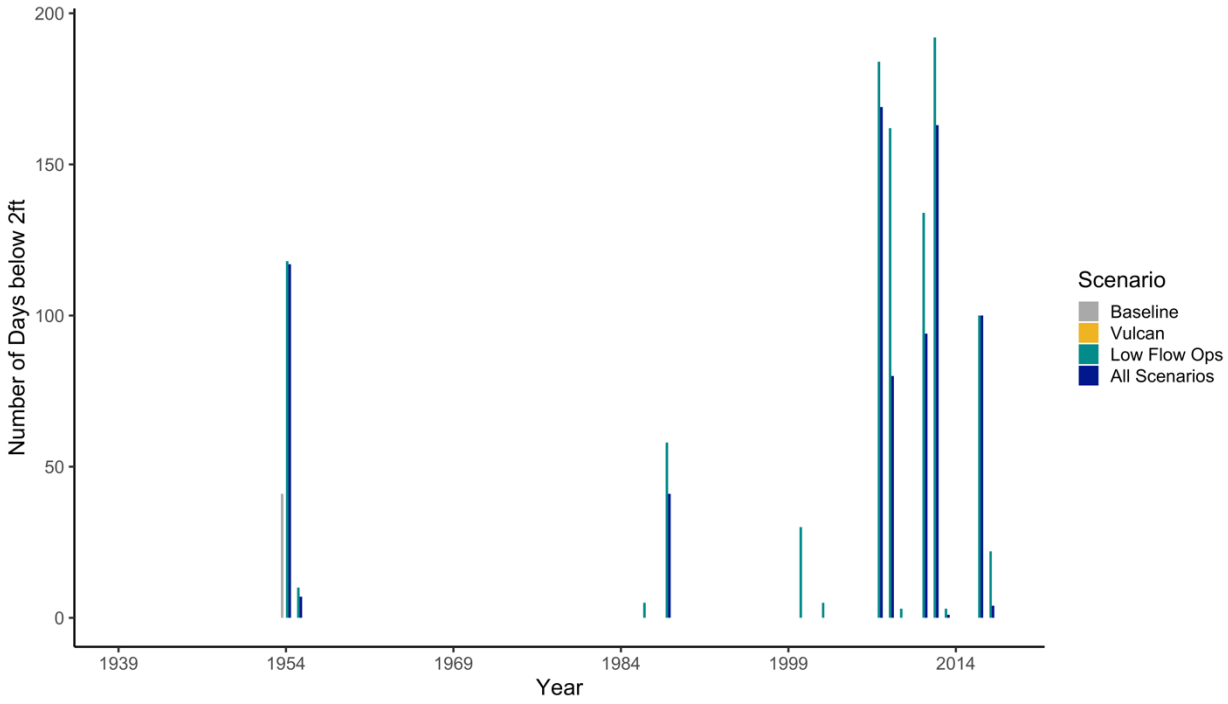
Scenario	Number of years below 85%
Baseline	14
Vulcan quarry	12
Low-flow ops	30
All scenarios	26

Figure 4.7. Number of days with combined reservoir storage below 85% for Clayton County reservoirs in each scenario in Flint BEAM. We summarized the total number of years below 85% combined storage for each scenario (Table).



Year	Baseline	Quarry	Low flow ops (3)	All scenarios (2)
2007			773 (57%)	771 (48%)
2008			774 (63%)	772 (53%)
2011			774 (63%)	

Figure 4.8. Number of days 5 ft below pool at Lake Horton reservoir in Fayette County Water System for each scenario in BEAM. The minimum pool elevation and percent storage in parentheses for each scenario in BEAM. The total number of years with 5 ft below pool for a scenario is in parentheses.



Year	Baseline (1)	Quarry (0)	Low flow ops (14)	All scenarios (10)
1954	777 (79%)		775 (68%)	775 (68%)
1955			777 (79%)	777 (79%)
1986			777 (79%)	
1988			776 (74%)	777 (79%)
2000			777 (79%)	
2002			777 (79%)	
2007			773 (57%)	771 (48%)
2008			774 (63%)	772 (53%)
2009			777(79%)	
2011			774 (63%)	776 (74%)
2012			775 (68%)	775 (68%)
2013			777 (79%)	777 (79%)
2016			775 (68%)	776 (74%)
2017			776 (74%)	777 (79%)

Figure 4.9. Number of days 2 ft below pool at Lake Horton reservoir in Fayette County Water System for each scenario in BEAM. The minimum reservoir elevation and percent storage in parentheses for each scenario in BEAM. The total number of years with pool below 2 ft for a scenario is in parentheses (Table).

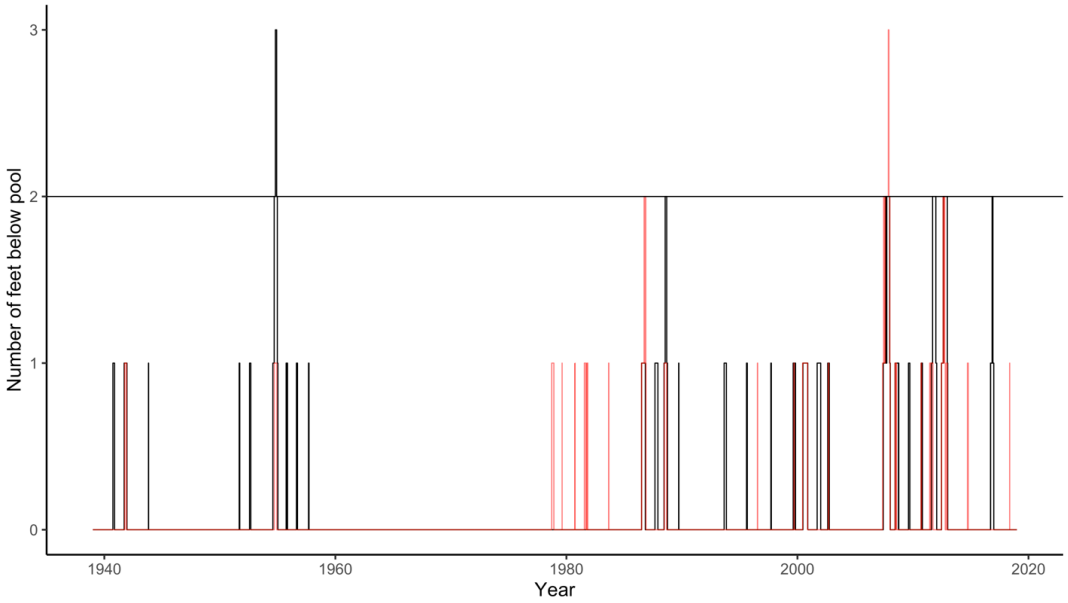


Figure 4.10. The number of feet below full pool for Lake Horton (black) and Lake Kedron (red) for the baseline scenario. The horizontal black line is at 2 feet below pool.

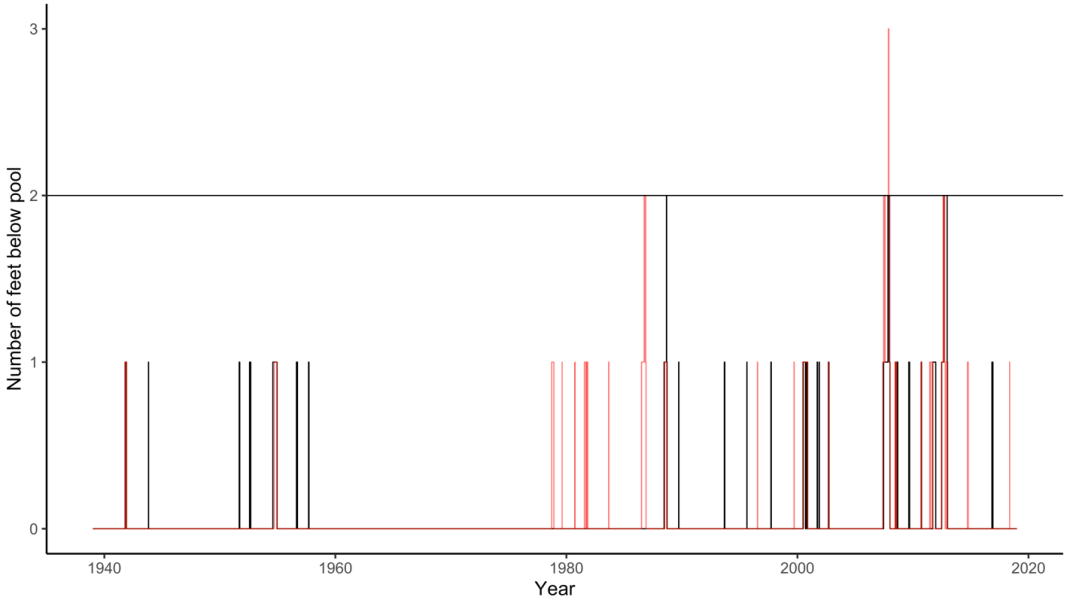


Figure 4.11. The number of feet below full pool for Lake Horton (black) and Lake Kedron (red) for the Vulcan quarry scenario. The horizontal black line is 2 feet below pool.

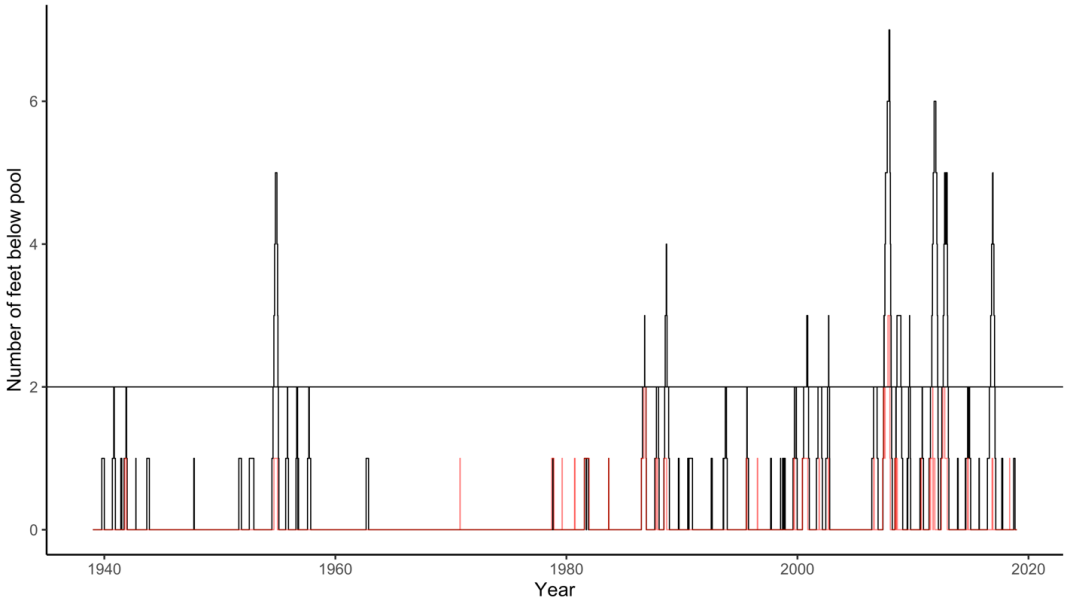


Figure 4.12. The number of feet below full pool for Lake Horton (black) and Lake Kedron (red) for the changing minimum flow scenario. The horizontal black line is 2 feet below pool.

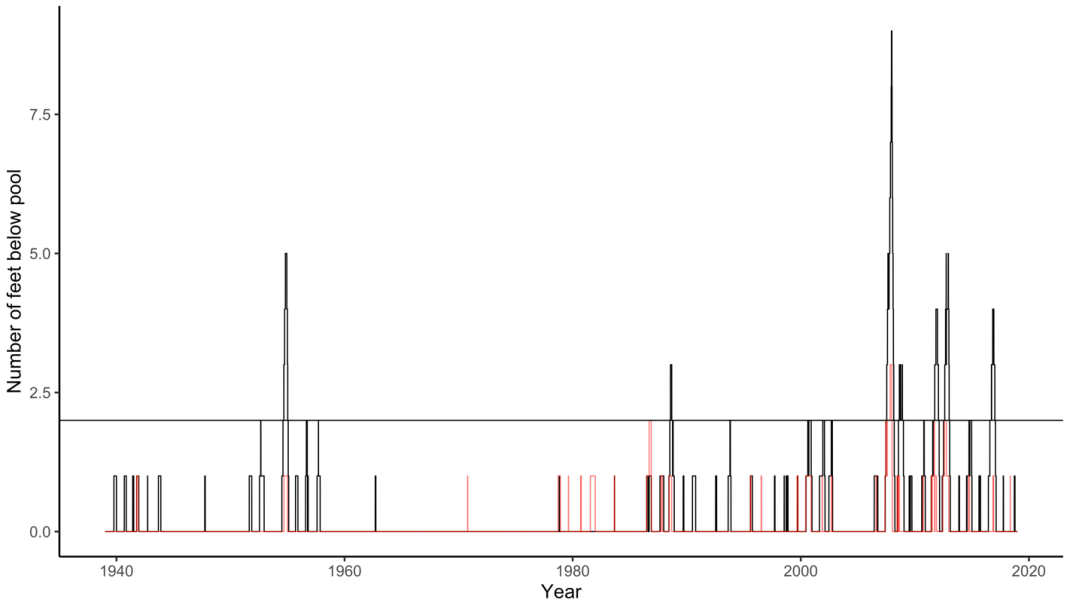
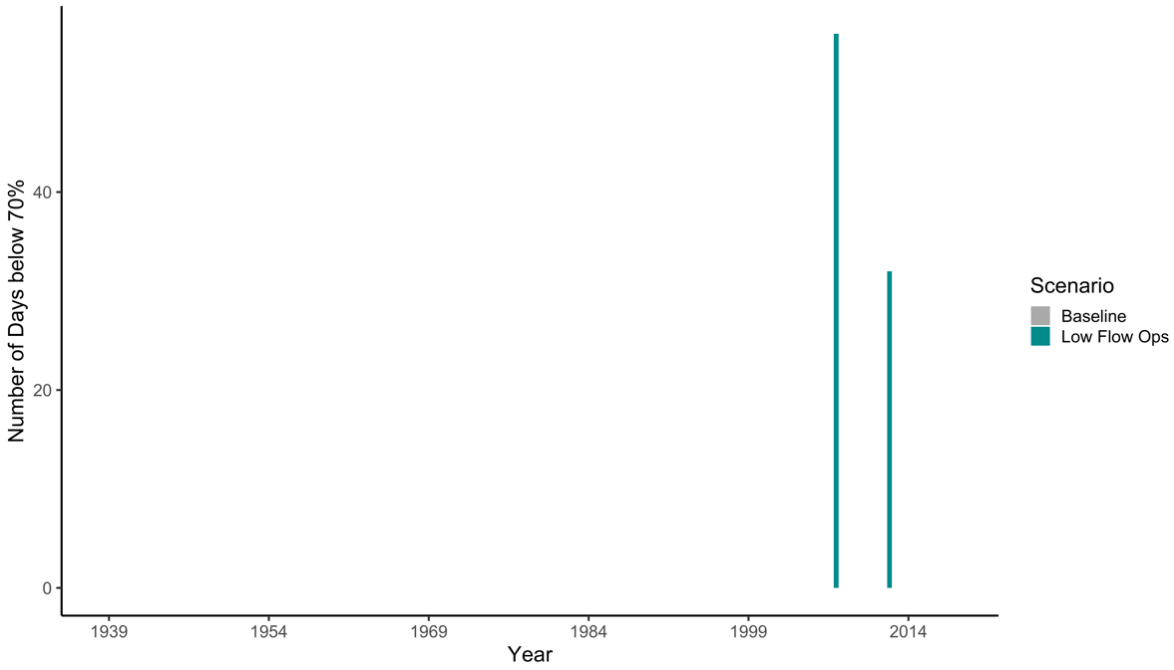
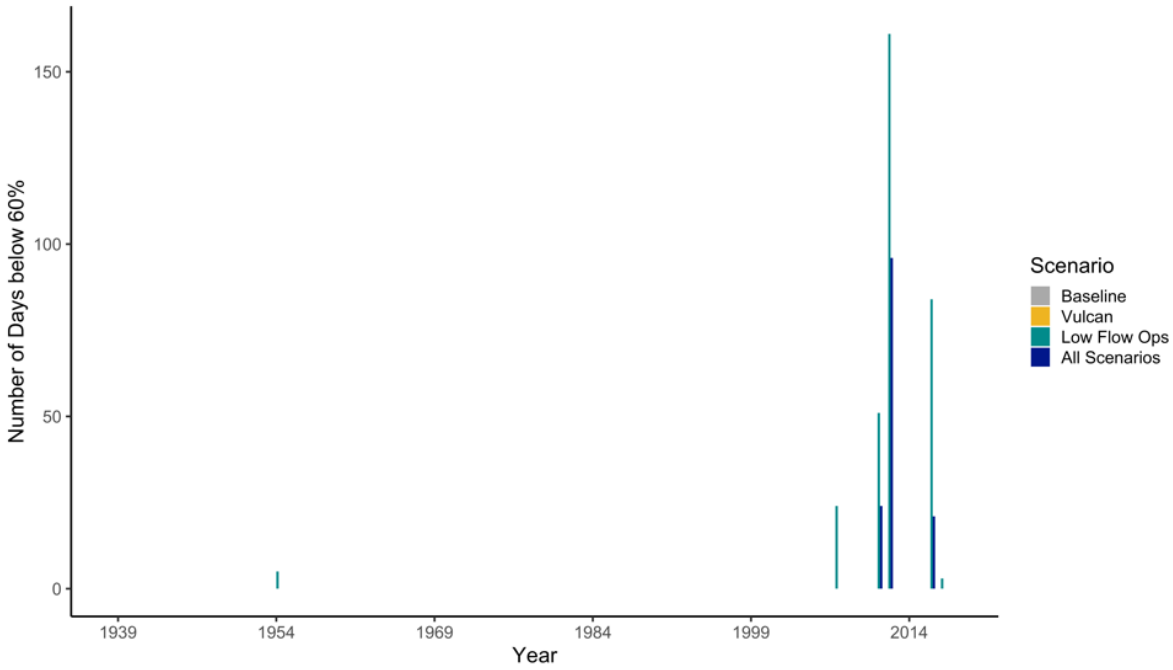


Figure 4.13. The number of feet below full pool for Lake Horton (black) and Lake Kedron (red) for the combined scenario. The horizontal black line is 2 feet below pool.



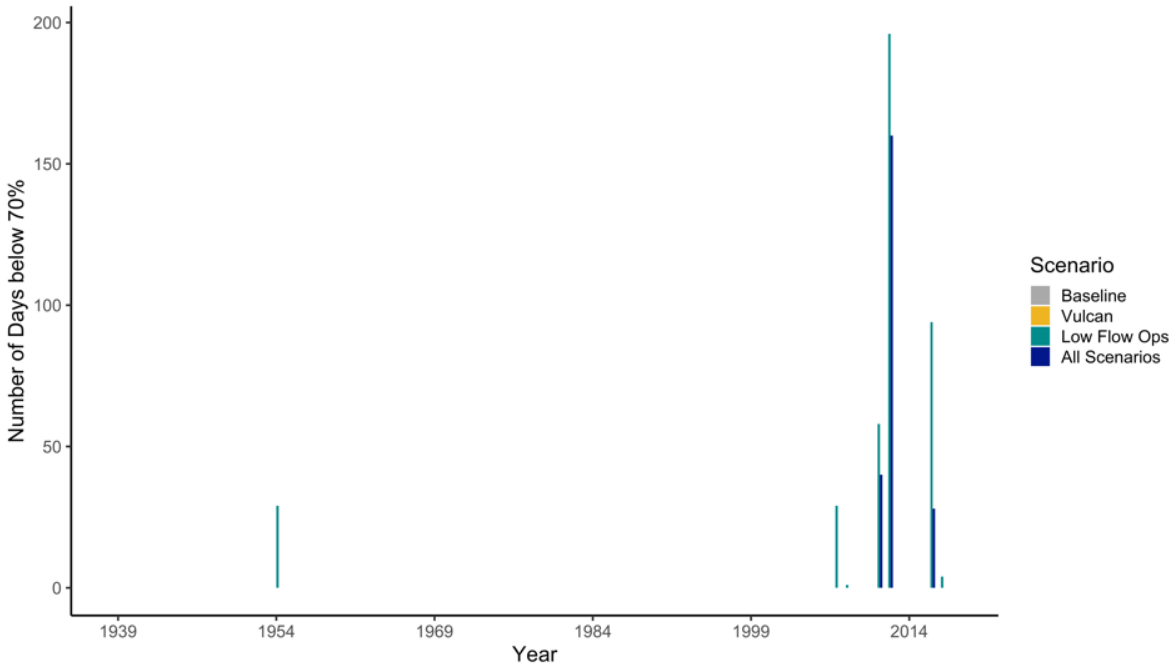
Year	Baseline	Low-flow ops
2007		2252 (58%)
2012		2485 (64%)

Figure 4.14. Number of days with combined reservoir storage below 70% for Newnan Utilities for each scenario in Flint BEAM. We summarized the reservoir storage for years below 70% combined storage (Table).



Year	Baseline	Quarry	Low-flow ops	All scenarios
1954			962 (58%)	
2007			671 (41%)	
2011			203 (12%)	587 (36%)
2012			115 (7%)	251 (15%)
2016			0	535 (32%)
2019			625 (38%)	

Figure 4.15. Number of days with Griffin Counties' Heads Creek Reservoir below 60% in each scenario in Flint BEAM. We summarized the minimum storage volume and percent storage for years with storage below 60% (Table).



Year	Baseline	Quarry	Low-flow ops	All scenarios
1954			962 (58%)	
2007			671 (41%)	
2008			1118 (68%)	
2011			203 (12%)	587 (36%)
2012			115 (7%)	251 (15%)
2016			0	535 (32%)
2019			625 (38%)	

Figure 4.16. Number of days with Griffin Counties' Heads Creek Reservoir below 70% in each scenario in Flint BEAM. We summarized the minimum storage volume and percent storage for years with storage below 70% (Table).

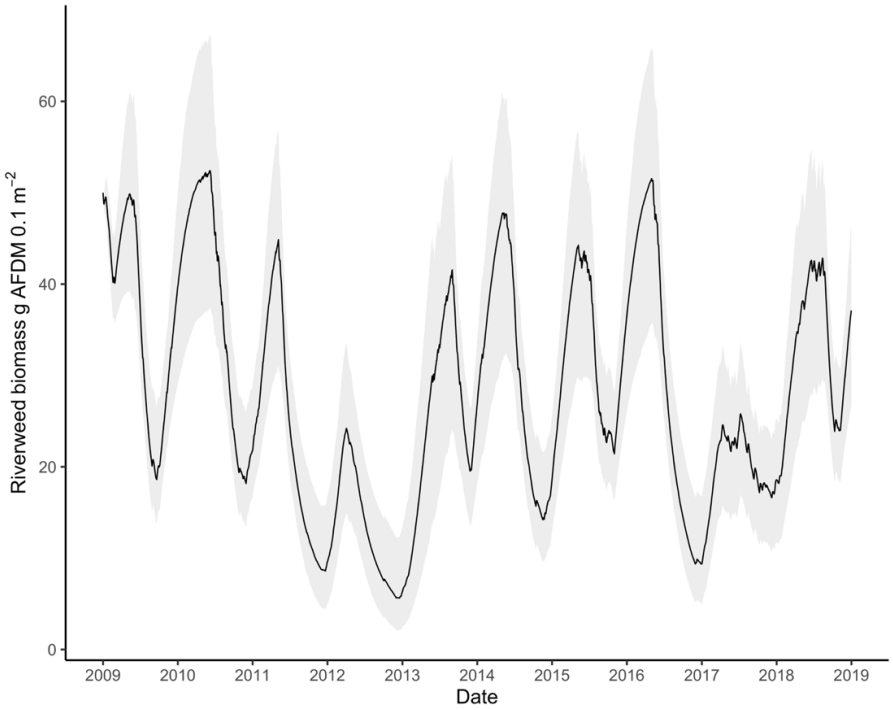


Figure 4.17. Change in riverweed biomass simulated at a daily time-step as driven by the discharge levels at the Carsonville gage site from the output of the Flint baseline scenario. Discharge was used to estimate the velocity conditions in the shoal which in turn influenced the growth rate of the plant. At lower velocities the plant was vulnerable to grazing, a mechanism of loss in our model.

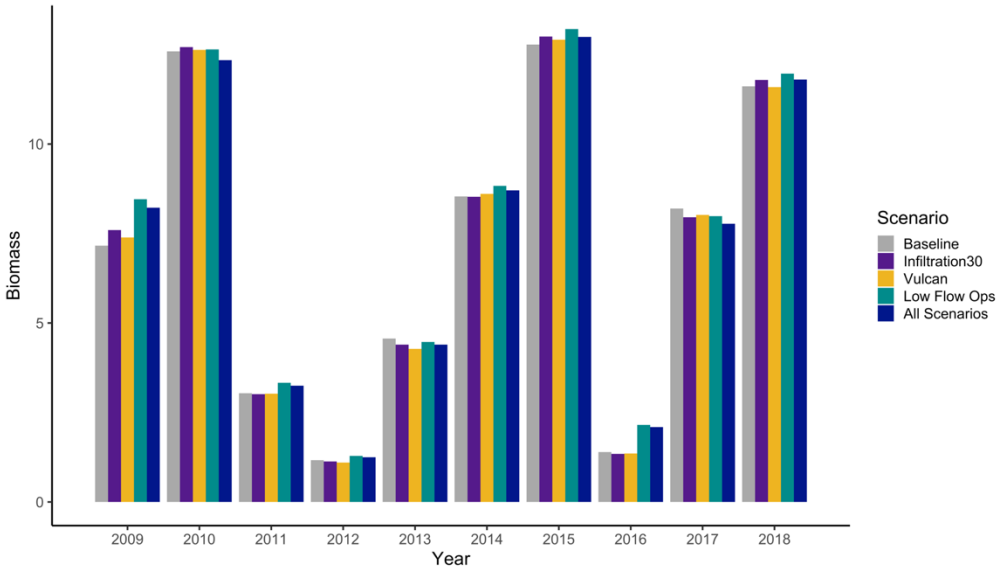


Figure 4.18. The minimum monthly riverweed biomass (g AFDM weighted by proportion of shoal habitat wetted for at least 30 d) for each scenario from 2009 to 2018. In most years, the scenarios with management actions showed slightly higher biomass than baseline, with the largest increase from the baseline seen during the 2016 drought.

References

ARCADIS. 2019. Methods for flow regime evaluation, Ocmulgee River, Georgia. Prepared for Georgia Environmental Protection Division

Argentina, J. E., M. C. Freeman, and B. J. Freeman. 2010. The response of stream fish to local and reach-scale variation in the occurrence of a benthic aquatic macrophyte. Freshwater Biology **55**:643-653.

Arthington, A. H., A. Bhaduri, S. E. Bunn, S. E. Jackson, R. E. Tharme, D. Tickner, B. Young, M. Acreman, N. Baker, S. Capon, A. C. Horne, E. Kendy, M. E. McClain, N. L. Poff, B. D. Richter, and S. Ward. 2018. The Brisbane Declaration and Global Action Agenda on Environmental Flows (2018). Frontiers in Environmental Science **6**.

Bell, D. B. and R. Gurney. 2022. Stormwater Forecast – Technical Approach and Results. Metropolitan North Georgia Water Planning District. Jacobs.

Grubaugh, J. W., and J. B. Wallace. 1995. Functional structure and production of the benthic community in a Piedmont river: 1956-1957 and 1991-1992. Limnology and Oceanography **40**:490-501.

Jackson, C.R., Wenger, S.J., Bledsoe, B.P., Shepherd, J.M., Capps, K.A., Rosemond, A.D., Paul, M.J., Welch-Devine, M., Li, K., Stephens, T. and Rasmussen, T.C., 2023. Water supply, waste assimilation, and low-flow issues facing the Southeast Piedmont Interstate-85 urban archipelago. Journal of the American Water Resources Association, 59(5), pp.1146-1161.

Nelson, D. J., and D. C. Scott. 1962. Role of Detritus in The Productivity of a Rock-Outcrop Community in a Piedmont Stream. Limnology and Oceanography **7**:396-413.

Pahl, J. 2009. Effects of flow alteration on the aquatic macrophyte *Podostemum ceratophyllum* (riverweed); local recovery potential and regional monitoring strategy. MS Thesis, University of Georgia.

Wood, J., and M. Freeman. 2017. Ecology of the macrophyte Podostemum ceratophyllum Michx. (Hornleaf riverweed), a widespread foundation species of eastern North American rivers. Aquatic Botany 139:65-74.

Emanuel, B. 2019. Ensuring water security for people and nature: A status report from the upper flint river working group. American Rivers.

CH2M Hill. 2018. Shoal Creek Water Reclamation Facility Decommissioning Feasibility Study, Task 1 of the Upper Flint Hydrology Study Project. Clayton County Water Authority

Walsh, C.J., Roy, A.H., Feminella, J.W., Cottingham, P.D., Groffman, P.M. and Morgan, R.P., 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society, 24(3), pp.706-723.

Wood, J.L., Skaggs, J.W., Conn, C. and Freeman, M.C., 2019. Water velocity regulates macro-consumer herbivory on the benthic macrophyte *Podostemum ceratophyllum* Michx. Freshwater Biology, 64(11), pp.2037-2045.

CHAPTER 5

CONCLUSIONS

A key factor shaping my work was the collaboration and willingness to listen to shared perspectives on how people are responsible for or view water resources across disciplines. My research included working with water utilities, conservation organizations, local and state government, and academic disciplines from ecology to engineering. Collaborators brought their knowledge and experience to the projects, and I listened and learned about a broad range of perspectives and dimensions of water resource management. I built a relationship and trust with the Working Group over seven years, which demonstrated the time and consistency it takes to build meaningful partnerships, but resulted in rewarding mutual learning and respect.

Humans rely on rivers and aquatic ecosystems for biodiversity, supporting, provisioning, regulating, and cultural services (Millennium Ecosystem Assessment 2005, Baron et al. 2002) and in turn it is important to understand how humans have changed and impacted aquatic ecosystems. Climate change is causing increasing strain on water resources through increases in temperature and increasing the frequency and severity of severe weather events such as floods and droughts (Scott 2016). Changes to weather patterns have implications for human infrastructure, such as stormwater, and natural ecosystems. Through this body of work, I have focused on river ecosystems through the lens of human dimensions and how we can broaden management to consider supporting and maintaining ecosystems alongside other water uses in management and planning.

Discussions of what and how to integrate ecological information into management has been in development since the 1940's (Poff et al. 2017). There are evolving tools and methods to move the field of environmental flows forward (Webb et al. 2017), with most approaches grounded in the basis that aquatic organisms are adapted to and have life history strategies that evolved within a river flow-regime (Petts 1996, Poff et al. 1997). While it is important to develop e-flow relationships for management in the context of discharge or relate it back to the lever that management can control, flow may not always be the best predictor of an ecological outcome. In addition, identifying a flow-related or driven mechanism for response is also important (Lancaster and Downes 2010). In chapter two I found that hydrology and hydraulics were not the best predictors of *Podostemum* growth rate, instead flow-mediated effects of algae or sediment cover and grazing had the greatest impact on growth rate or cover of *Podostemum*. I did find that hydraulics were useful in identifying vulnerability to grazing and in accrual of algae, however it was difficult to predict the onset of an algal bloom. This result was not surprising as algal blooms occur due to a series of interacting mechanisms that can allow algae to proliferate (Biggs and Price 1987, Suren et al. 2003, Power et al. 2008). We were able to identify and test important flow-related drivers of *Podostemum* and link those back to potential impacts of low-flow conditions in the river.

In chapter three, I adapted the functional flows framework to develop ecological indicators for Georgia's state-wide regional water planning. In Georgia, laws and statues around water quantity are based on the doctrine of riparian rights and "reasonable use," which is the case of all of the eastern USA (Bowen 2001, Zellmer 2008). Management is primarily based on water quality, with no direct mandate to manage river flows for ecosystem outcomes alone. I took the approach of developing indicators and metrics for river ecosystems that could be evaluated

alongside other water uses. Within this context, decisions can be made around options available for meeting river ecosystem needs, rather than siloing responsibility or trying to point out one responsible party.

Four themes emerged that were important for developing ecological indicators and metrics in the context of water planning. First, context is critical for management-relevant flow metrics. Selection and development of e-flow targets needs to be linked with the physical, ecological, and political geography in which they would be applied. Second, developing quantitative targets is particularly important if they will be used alongside quantitative targets for socio-economics water uses. Without a basis for comparison, it is very challenging to view the ecological indicators on the same footing as socio-economic uses. Third, the e-flow framework should be comparable to the management scope, so that the number of ecological indicators is similar to the number used for socio-economic outcomes. Fourth, information for managers and planners needs to include how to evaluate and interpret the ecological outcomes and how they may relate to the other management or planning targets. I kept these lessons learned in mind through chapter four, where I compared the relative impact of water management actions on ecological outcomes and reservoir storage.

In chapter four, I worked with the Upper Flint River Working Group and Upper Flint Regional Water planning council to develop and select management scenarios to simulate in the context of the regional water planning framework in Georgia. We used the Flint Basin Environment Assessment Model (BEAM), developed and shared by the Georgia Environmental Protection Division, which is a linear model that routes water through the Flint River Basin using the state level permitted limits for water withdrawal and discharge and simulates reservoir levels based on individual utility reservoir operations, and the reported demand by users for a set

period. This allowed me to simulate flow augmentation with a quarry at the top of the basin and see the impact of alternative low flow management (i.e., raising the minimum flow) on river flows in the Flint and its tributaries. We found the only way to meaningfully raise the lowest flow levels in the Flint River was to change management operations, however this resulted in lower reservoir storage, and reached levels that would be of concern for municipal utilities. We found that combining the augmentation of flows from the quarry with the alterative low flow operations reduced the impact on municipal water utility storage and still resulted in higher instream flow levels that would benefit *Podostemum* and shoal ecosystems. This type of information is a useful starting point for planners to identify the potential impact of management actions and direct resources on more detailed studies of these management options.

Overall, this work showed that there is opportunity to integrate ecosystem needs into planning and management frameworks that tend to be center around human needs. I found that quantifying ecosystem needs alongside other water uses in a formal setting can catalyze discussions around what management opportunities are worth further consideration. In addition, managing for river ecosystems and human needs is not a zero-sum game. Rather, there are opportunities for co-benefits for human needs and river ecosystems particularly if planning is proactive and the pinch-points, like severe drought, are investigated and evaluated early and often. While flow-ecology relationships can be complex to develop, we should use the best available information for decision making and update the information as ecological research continues.

While this work provided a framework and model for integrating ecological information into water planning in Georgia, it represents a first step in actively planning for aquatic ecosystems. I applied the approach in two of the eleven Water Planning Regions in the State so

there is opportunity for further adaptation and adoption in the other regions. In addition, if the framework is adopted in planning, there is flexibility to customize how the approach is used given the concerns and data availability in a water planning region. Our framework was developed with the intent that it would be updated as new information becomes available. The same premise applies to the development of environmental flow relationships. We used the information available to develop relationships, but there is a need for further research to better understand the limits of resilience in river ecosystems so that we can answer the question of how low is too low or how long is too long before we start to see irreversible changes the river ecosystems and organisms we value.

References

Baron, J.S., Poff, N.L., Angermeier, P.L., Dahm, C.N., Gleick, P.H., Hairston Jr, N.G., Jackson, R.B., Johnston, C.A., Richter, B.D. and Steinman, A.D., 2002. Meeting ecological and societal needs for freshwater. Ecological Applications, 12(5), pp.1247-1260.

Biggs, B. J. F., and G. M. Price. 1987. A survey of filamentous algal proliferations in New Zealand rivers. **21**:175-191.

Bowen, B. L. 2001. Georgia Riparianism and Irrigation.*in* Proceedings of the 2001 Georgia Water Resources Conference. Institute of Ecology, University of Georgia.

Lancaster, J., and B. J. Downes. 2010. Linking the hydraulic world of individual organisms to ecological processes: Putting ecology into ecohydraulics. River Research and Applications **26**:385-403.

Millennium Ecosystem Assessment, 2005. *Ecosystems and Human Well-being: Synthesis*. Island Press, Washington, DC.

Petts, G. E. 1996. Water Allocation to Protect River Ecosystems. Regulated Rivers: Research & Management **12**:353-365.

Poff, N. L., D. J. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The Natural Flow Regime a paradigm for river conservation and restoration. BioScience **47**:769-784.

Poff, N. L., R. E. Tharme, and A. H. Arthington. 2017. Evolution of Environmental Flows Assessment Science, Principles, and Methodologies. Pages 203-236 *in* A. Horne, J. A. Webb, M. J. Stewardson, B. D. Richter, and M. Acreman, editors. Water for the Environment. Academic Press.

Power, M. E., M. S. Parker, and W. E. Dietrich. 2008. Seasonal reassembly of a river food web: floods, droughts, and impacts of fish. **78**:263-282.

Suren, A. M., B. J. F. Biggs, C. Kilroy, and L. Bergey. 2003. Benthic community dynamics during summer low-flows in two rivers of contrasting enrichment 1. Periphyton. New Zealand Journal of Marine and Freshwater Research **37**:53-70.

Webb, J. A., A. H. Arthington, and J. D. Olden. 2017. Models of Ecological Responses to Flow Regime Change to Inform Environmental Flows Assessments. Pages 287-316 *in* A. Horne, J. A. Webb, M. J. Stewardson, B. D. Richter, and M. Acreman, editors. Water for the Environment: From Policy and Science to Implementation and Managment. Elsevier Academic Press.

Zellmer, S. 2008. Legal Tools for lnstream Flow Protection. Pages 285-327 Integrated Approaches to Riverine Resource Stewardship: Case Studies, Science, Law, People, and Policy. The Instream Flow Council.

APPENDIX A

SUPPLEMENTARY METHODS AND DATA: CHAPTER 2

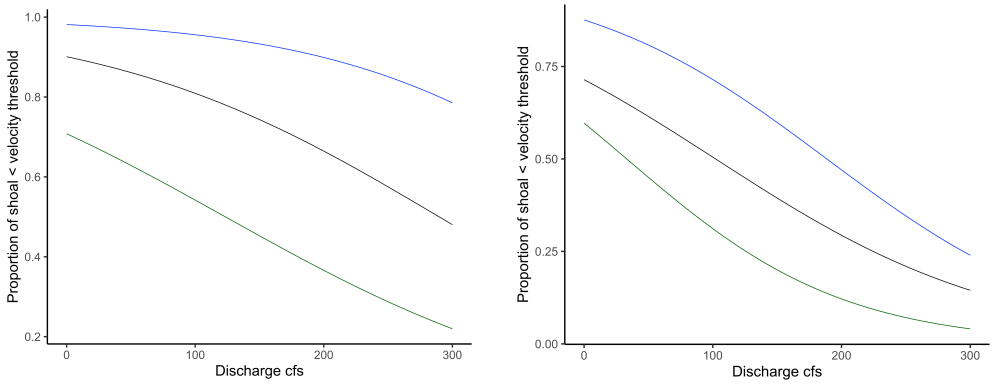


Figure A1. Proportion of shoal < 0.40 m/s (black), 0.60 m/s (blue), and 0.20 (green) m/s in the Middle Oconee River near Ben Burton Park for the upper shoal (left) and lower shoal (right). The proportion of the shoal below the three velocity thresholds changed similarly with discharge.

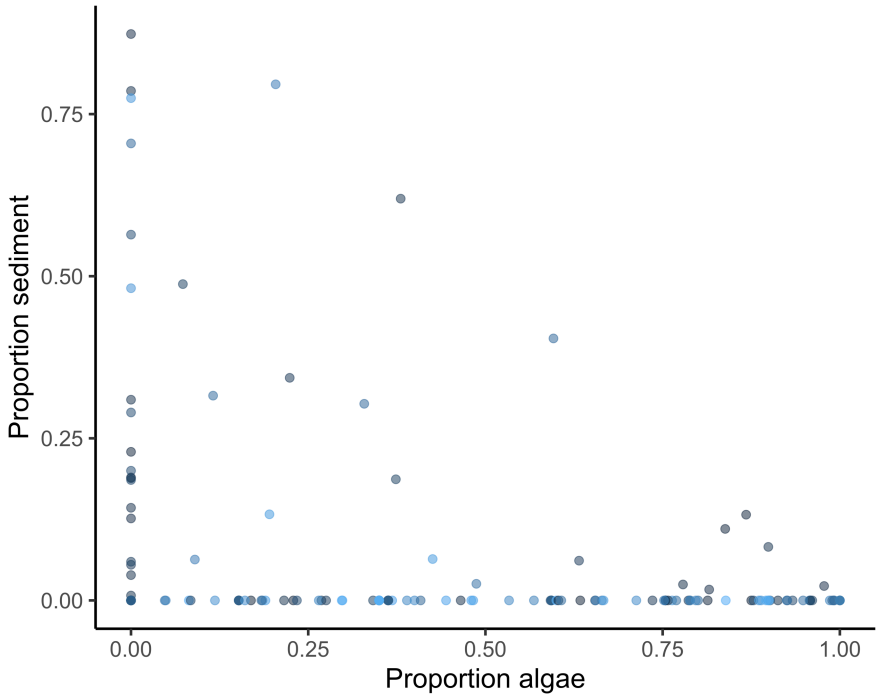


Figure A2. Proportion sediment cover versus proportion algal cover for each rock measured on 36 tagged rocks in the Middle Oconee River near Athens, GA, June-November 2022. Grid-cells were either labeled algae or sediment, and some rocks had both.

APPENDIX B

SUPPLEMENTARY MATERIAL: CHAPTER 4

PART 1: Podostemum simulation model for Flint River Shoals

Objective: Compare ecological outcomes among flow-scenarios in the upper Flint River.

Model components:

1. Daily flows for a scenario are used to simulate daily net change in *Podostemum* biomass for a one or more growing annual cycles. Daily net change in biomass is used simulate biomass standing stock across each annual cycle, from an arbitrary beginning amount (e.g., 1000 mg AFDM/m^2 at the beginning of the first cycle if simulation covers multiple years).

2. Simulated standing stock biomass at the end of each annual cycle is multiplied by the lowest 30-d average proportion of shoal width estimated to maintain flow during that year. We assume that drying for 30-d leads to complete loss of the plant (Pahl 2009), and so the smallest area of shoal that retains flow (i.e., drying for less than 30 d) across the season is the area that can support the simulated standing stock *Podostemum* biomass.

Methods:

1. Use measurements of water velocity in Flint River shoals (during 2001, 2002; Marcinek, UGA, unpublished) to estimate a relation between discharge at the USGS Carsonville gage and proportion of shoal habitat that has water velocity < 0.4 m/s ("low velocity extent").
2. For a flow scenario, project daily flows at the Carsonville gage
3. For each day in the scenario, estimate proportion of shoal habitat with velocity <0.4 m/s using the relation from step 1.
4. Use regression coefficients estimated from Middle Oconee River to translate daily proportion of shoal habitat with velocity <0.4 m/s to a daily *Podostemum* growth rate.
5. Use daily growth rate to estimate daily change in *Podostemum* biomass, over each annual cycle in the simulation period.
6. Finally, use a linear regression to estimate the minimum 30-d average percent wetted channel in Flint River shoal habitat for each annual cycle in the simulation period.
7. Assume that exposure for 30 days eliminates *Podostemum* (Pahl 2009). *Podostemum* biomass in shoal habitat at the end of each annual cycle is estimated by final, annual biomass estimate (from step 5) multiplied by the minimum 30-d average percent wetted channel (step 6).

Sections:

I. Estimating proportion of Flint River shoal habitat with velocity <0.4 m/s in relation to daily flow, using field observations in 2001 and 2002.
II. Overview: estimating *Podostemum* daily growth rate in relation to shoal habitat with velocity <0.4 m/s, using biomass time-series observations in the Middle Oconee River.
III. Estimating the proportion of shoal drying at Sprewell Bluff shoals based on three discharge and drying points
IV. Model code
V. References

I. Estimating proportion of Flint River shoal habitat with velocity <0.4 m/s in relation to daily flow, using field observations in 2001 and 2002.

Data comprise velocity measurements at randomly chosen locations within 17 Flint River shoals, made in conjunction with fish sampling during 2001 and 2002 (Marcinek 2003). Shoals are located between Gay-Flat Shoals Road and Pobiddy Road crossings of the Flint River and were randomly chosen to represent large (>100 m in length) and small (<100 m in length) shoals in the upper and lower halves of the study reach (Marcinek 2003). Water velocity was measured at 60% of the water depth (measured from the surface) with an electronic current meter and top-setting wading rod.

We tested two models (log-log and logistic) to relate the proportion of measurements that were <0.4 m/s ("slow velocity") to the flow at the Carsonville gage on the day measurements were made. Both models include a random effect for shoal identity ("site[i]"); three shoals were sampled in both years, one shoal in 2001 only, and 13 shoals in 2002 only. Number of velocity measurements per shoal visit ranged from 26 to 80 (median = 59). Note that on 18 of the 20 total visits, flow at the Carsonville gage was < 600 cfs, and that the majority of velocity measurements were <0.4 m/s.

Code for log-log model:

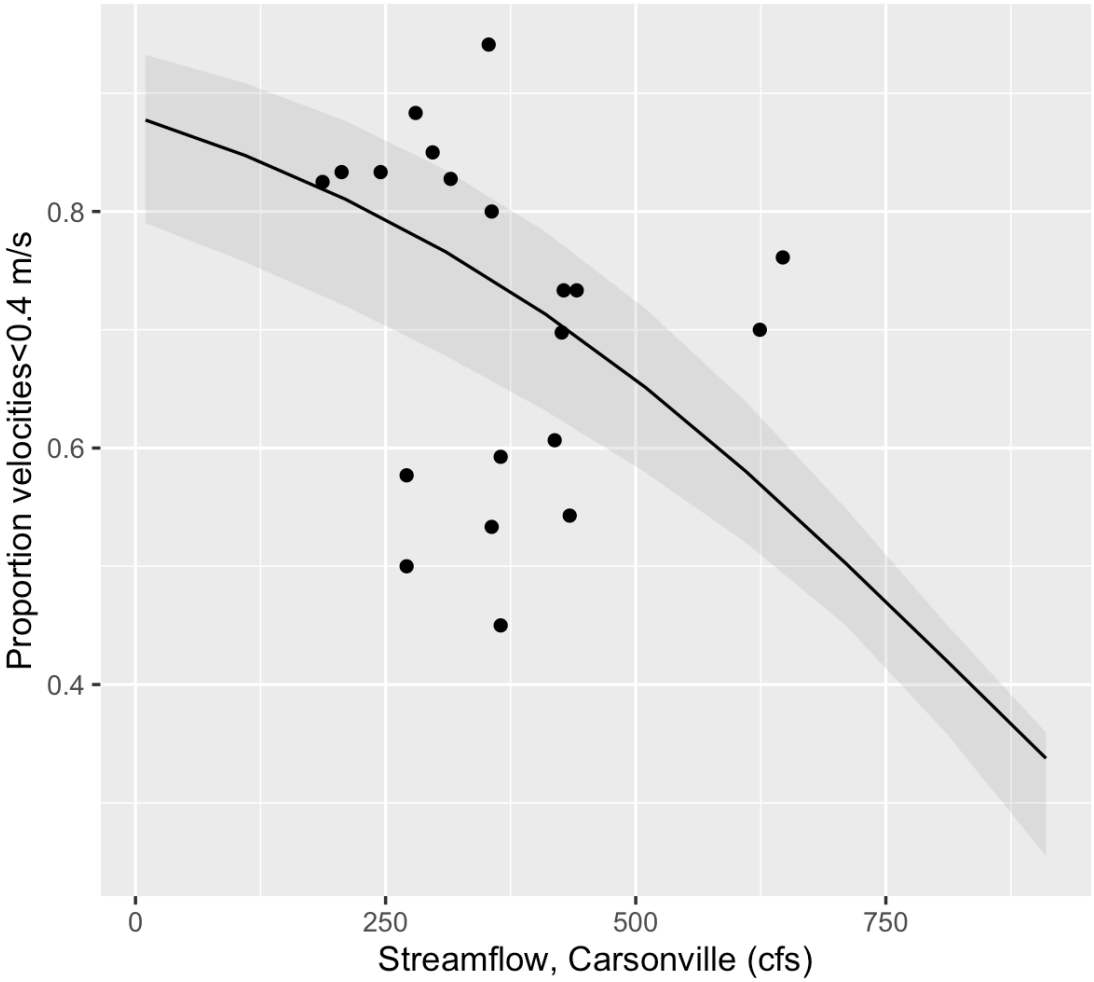
```
slow[i]~dbern(p.slow[i])
p.slow[i]<-exp(-exp(s[i]*100*(q[i]-c)))
s[i]<-s0 + epsilon[site[i]]
```

Code for the logistic model:

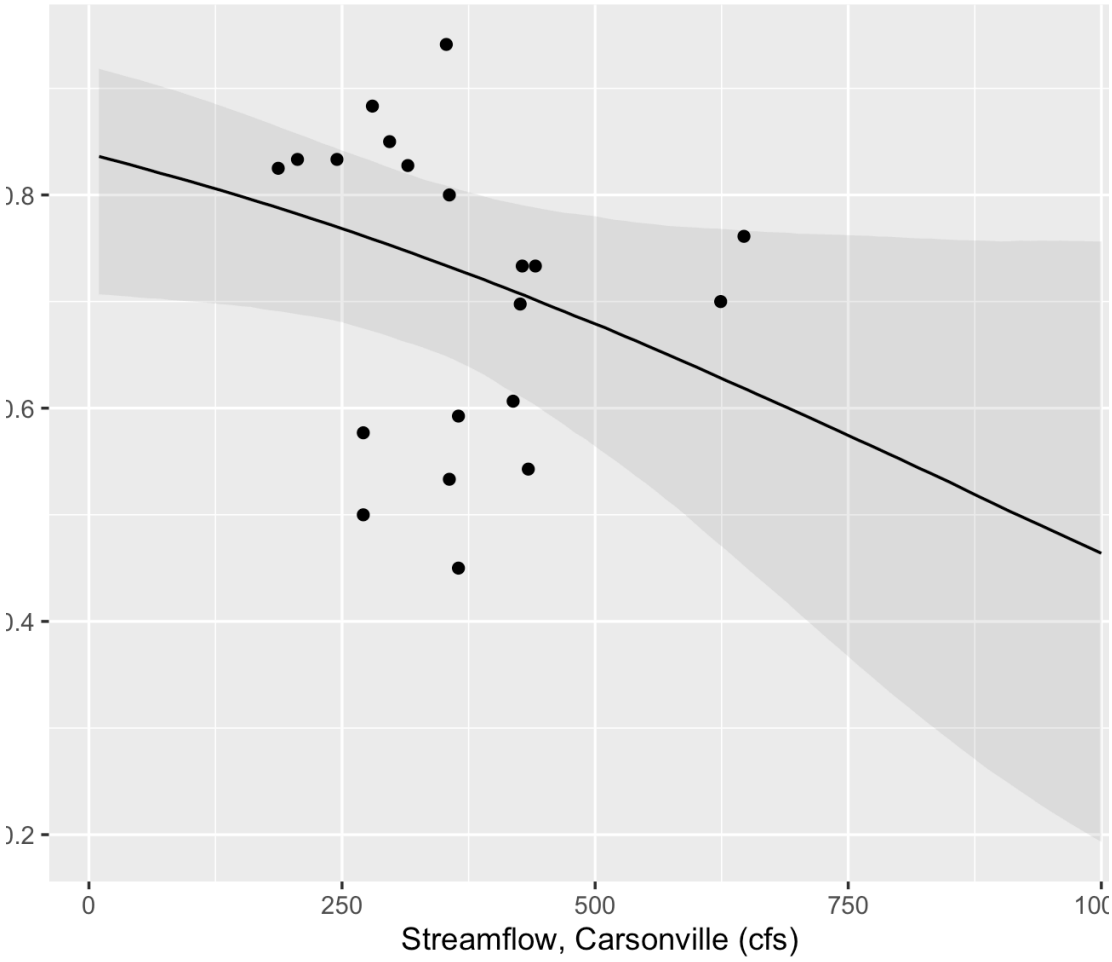
```
slow[i]~dbern(p.low[i])
logit(p.low[i])<-a0.1 + a1.1*q[i] + epsilon[site[i]]
```

Here, 'slow[i]' is an individual water velocity measurement at a particular shoal, coded as 1 if <0.4 m/s.

Resulting regressions based on 992 velocity measurements at 17 shoals are illustrated below. We used the log-log regression model in the Flint shoal application.



Log-log model of proportion 'slow velocity' v. Q at Carsonville. Regression line and 95% credible interval are based on individual velocity measurements; data summarized as proportion of measurements for each shoal visit are plotted as points.



Logistic regression model of proportion 'slow velocity' v. Q at Carsonville. Regression line and 95% credible interval are based on individual velocity measurements; data summarized as proportion of measurements for each shoal visit are plotted as points.

II. Overview: estimating *Podostemum* daily growth rate in relation to shoal habitat with velocity <0.4 m/s, using biomass time-series observations in the Middle Oconee River.

Data comprise four time-series of approximately monthly biomass estimates for *Podostemum* growing in four locations in the Middle Oconee River near Athens. Data were collected in 1956-1957 (Nelson and Scott 1962), 1991-1992 (Grubaugh and Wallace 1995), 2007-2008 (Pahl 2009), and 2016-2018 (Conn, unpublished) Time series were assembled into a 4 x 26 matrix of monthly mean biomasses as reported in each study, with missing values for months lacking measurements.

Model code:

```
biomass[i,j]~dlnorm(mu[i,j], tau.biomass)  # i =1 to 4 timeseries, j = 1 to 26 monthly biomasses
   mu[i,j]<-mu[i,(j-1)] + r[i,j]*days[i,(j-1)]    # days = number of days, j-1 to j
   r[i,j]~dnorm(mu.r[i,j], tau.r)                          # r is daily accumulation rate

mu.r[i,j]<-b0 + b1*(exp(mu[i,(j-1)]))+b2*grazing[i,(j-1)]
```

Here, daily growth rate (mu.r[i,j]) is influenced by a density-dependent term (exp(mu[i,(j-1)]), which relates to the biomass on date j-1 in g AFDM/0.1 m^2), and by the mean proportion of shoal area with velocities < 0.4 m/s, each interval j-1 to j ("grazing[i,(j-1)]"). This (vulnerable to) grazing term was estimated using logistic regression of velocity measurements in relation to streamflow for a range of low-flow conditions (Pahl, unpublished; Conn, unpublished; Rack, unpublished) in areas representing each biomass time-series.

Parameter estimates, mean (95% CI):

b0 0.17 (0.0058 - 0.029)
b1 -0.0003 (-0.0005 to -0.0001)
b2 -0.026 (-0.043 to -0.0066)

III. Estimating the proportion of shoal drying at Sprewell Bluff shoals based on three discharge and drying points

Estimating how much shoal habitat remains wetted and flowing during low-flow periods is key to understanding low-flow effects on riverweed (*Podostemum*). This is because our model assumes that riverweed can only persist in areas of a shoal that dry for less than 30 days (based on experimental evidence from Pahl 2009). To construct a preliminary relation between streamflow (as recorded at the Carsonville gage) and extent of rock exposure, we used three assumptions for the Sprewell Bluff shoal:

- when flow = 0 at Carsonville, the entire shoal lacks flow (although there may be wet areas)
- when flow = 200 cfs at Carsonville, 50% of the shoal is wetted and flowing (based on photos)
- when flow = 1000 cfs at Carsonville, the entire shoal is wetted and flowing.

For each simulated annual cycle, we interpolated the proportion of the shoal estimated to retain flow during the lowest 30-d average flow during that cycle.

This portion of the simulation model would be substantively improved with additional data on wetted area in upper Flint River shoals in relation to streamflow.

IV. Model code written in the software R and using packages rjags and R2jags.

This model:
(1) Uses four time-series of Middle Oconee River biomass measurements and estimates of extent of low to estimate effect of low velocity on *Podostemum* growth rate;
(2) uses 2001 & 2002 water velocity data from 17 Flint River shoals to relate proportion of shoal velocity measurements that are <0.4 m/s ("low-velocity extent") to discharge (q);
(3) calculates "low-velocity extent" in Flint River shoal habitat for each day using a daily flow time-series, and finally
(4), computes the *Podostemum* growth rate given that day's "low-velocity extent" using regression coefficients from a regression model for the four time-series of Middle Oconee River biomass measurements.

One can estimate proportion of shoal habitat with velocity <0.4 in relation to stream flow with a log-log model (as in model below) or with a logit model (see Appendix A).

```
model {
## estimate regression coefficients, growth v. time<0.4m/s, using Middle Oconee data;
## note biomasses are scaled to g AFDM/0.1 m2
for (i in 1:nseries){
   biomass[i,1]~ dlnorm(mu[i,1], tau.biomass) ## starting biomass, each time series
   mu[i,1]<-a0[i] #have 4 values, 1 for each time-series

   for (j in 2:26){
   biomass[i,j]~dlnorm(mu[i,j], tau.biomass)
   mu[i,j]<-mu[i,(j-1)] + r[i,j]*days[i,(j-1)]
   r[i,j]~dnorm(mu.r[i,j], tau.r) # r is daily accumulation rate

   ## density and grazing - 2 terms
  mu.r[i,j]<-b0 + b1*(exp(mu[i,(j-1)]))+b2*grazing[i,(j-1)]
    }}

# priors
for (i in 1:4){
a0[i] ~ dnorm(5, 0.01) #log scale

}
b0 ~ dnorm(0, 0.01) # mean daily growth rate
```

```
b1 ~ dnorm(0, 0.001) #adjustment for biomass
b2 ~ dnorm(0, 0.001)  #adjustment for low velocities

tau.biomass<-1 / sigma.biomass^2
sigma.biomass~dunif(0,10)

tau.r<-1 / sigma.r^2
sigma.r~dunif(0,1)

#### estimate time <0.4m/s for flint time series, log-log relation
for (i in 1:nobs){
   slow[i]~dbern(p.slow[i])
   p.slow[i]<-exp(-exp(s[i]*100*(q[i]-c)))
   s[i]<-s0 + epsilon[site[i]]
}

s0 ~ dnorm(0, 0.001)
c  ~ dunif(2,9)
for (i in 1:17){
  epsilon[i]~dnorm(0, tau.site)
}
   tau.site<-1 / sigma.site^2
   sigma.site~dunif(0,10)

## estimate prop of shoal with velocity <0.4, each day, 184 d growing season in this case

for (i in 1:184){
p.low.est[i]<-(exp(-exp(s0*100*(obs.q[i]-c))))
}

## estimate biomass each day using exponential growth rate;
   flint.biomass[1]<-100 #biomass, g AFDM/0.1 m2; starting value
   for (i in 1:183){
   flint.r[i]<-b0 + b1*(flint.biomass[i])+b2*p.low.est[i]
   flint.biomass[i+1]<-flint.biomass[i]*exp(flint.r[i])
}}

inits <- function(){list(a0 =c(3,4,1,4), b0 =0.02, b1=-0.0003, b2=-0.02, s0=0, c=6, sigma.site=1,
                  sigma.biomass = 1, sigma.r = 0.1)}
```

Data to run this code:

- nseries = 4
- biomass = a 4 x 26 matrix of monthly biomass estimates from the Middle Oconee River, scaled to g AFDM/0.1 m^2
- grazing = a 4 x 25 matrix of interval-specific, estimated mean proportion of the study area (for each Middle Oconee River data set) with velocity < 0.4 m/s

- days = a 4 x 25 matrix of the number of days between each ~ monthly Middle Oconee River biomass measurement
- nobs= 992
- slow = 992 observed velocities in Flint River shoals, coded as 1 if < 0.4 m/s, and 0 otherwise
- q= 992 observed streamflow values for the Flint River at Carsonville divided by 100, corresponding to 'slow' observations
- site = 992 coded site locales, corresponding to 'slow' observations
- obs.q = *n* daily flows for the Flint River at Carsonville divided by 100, for the simulation period

V. References

Grubaugh, J.W. and Wallace, J.B., 1995. Functional structure and production of the benthic community in a Piedmont river: 1956-1957 and 1991-1992. *Limnology and Oceanography*, *40*(3), pp.490-501.

Marcinek, P.A. 2003. Variation of fish assemblages and species abundances in the upper Flint River shoals, Georgia. MS thesis, University of Georgia, Athens, GA.

Nelson, D.J. and Scott, D.C., 1962. Role of detritus in the productivity of a rock-outcrop community in a Piedmont stream. *Limnology and Oceanography*, *7*(3), pp.396-413.

Pahl, J. P. 2009. Effects of flow alteration on the aquatic macrophyte *Podostemum ceratophyllum* (riverweed); local recovery potential and regional monitoring strategy. MS thesis, University of Georgia, Athens, GA.

PART 2: Guidance for evaluating ecological indicators in water planning

Introduction

Freshwater species are adapted to, and depend on, the full range of flows that a river system naturally experiences across seasons and among years to complete their life cycles and sustain populations. For this reason, managers and stakeholders need information on flow levels that support a range of ecosystem functions when assessing future water availability for river ecosystems.

Evaluating water availability to support river ecosystems requires a different approach than is currently used to evaluate gaps in water availability for other demands. During each 5-year cycle in Georgia's water planning process, planners compare a forecast of future water demand to current water availability. Gaps are expressed as the proportion of time during a model period (80-years) that a demand is not met, or that streamflow falls below the wastewater assimilation threshold. Ecological indicators, or attainment of functional flows, can be assessed using the same framework of current and future flow projections, however evaluation requires shifting from averaging over the entire model period to examining the occurrence and severity of ecologically stressful events.

Evaluating and Interpreting Ecological Indicators

The ecological outcome of an exceptional flow condition (such as an extreme low flow) will partly depend on how low (magnitude), how long (duration), and how often (frequency) stressful events occur. Therefore, it is most useful to evaluate flow thresholds (magnitudes) in the context of how long and how often they are exceeded with respect to current and future conditions.

For example, supporting survival of aquatic organisms is a key streamflow function that will be affected when flow falls below a 'dry-season threshold'. To evaluate whether future flows during the dry season are likely to compromise organism survival, it would be useful to compare the annual frequency and duration of flow events below the dry season threshold (e.g., during June-October) for the current and future scenarios. We show an example of this evaluation process below for the Upper Flint Regional Water Council.

Deciding how much change is too much may depend on a variety of factors, including risk tolerance (e.g., of utilities and resource managers), the availability of current or future options to minimize the change, and the ecological function of the flow being evaluated (e.g., flows necessary for survival across many groups of organisms versus seasonal connectivity to floodplain habitats for a subset of organisms). If the consequences of crossing a flow threshold in a future scenario are too great, the next step is to investigate management alternatives to prevent this outcome.

Example for the Upper Flint River Water Council

In the 2023 Upper Flint Regional Water Plan, the Council requested that metrics for recreation and ecological indicators be evaluated, based on flows levels provided in "Guidance on Drought Resilience for People and Nature in the Upper Flint River Basin" (Upper Flint River Working Group 2021). The streamflow metrics were evaluated at the Carsonville gage (Flint River at US 19, near Carsonville, USGS gage 02347500; USGS 2025) and comprised two flow levels: 100 cfs, representing a drying threshold where the river shoals were "more rocks than water", and 600 cfs, which is a generally accepted minimum flow for floating a kayak or canoe down Flint River shoals. This "paddling flow" is similar to a flow level (500 cfs) estimated to

sustain swift-water habitat in Flint River shoal ecosystems and can be used to evaluate outcomes for both recreation and shoal ecosystems.

The metrics were evaluated in the Regional Water Plan (RWP) as the total proportion of the 80-year model period during which flow at the Carsonville gage was below metric thresholds for the baseline demand (average demand from 2010-2018) and the baseline drought demand (2011; RWP, pages 3.6-3.10). The baselines were compared with future water availability to meet these metrics based on data from agricultural demand forecasts through 2060; results showed minimal differences between current and future conditions for either metric, since most agricultural growth was projected to occur downstream of the Carsonville gage.

Table 3-5: Surface Water Availability Streamflow Results

Carsonville Flow Summary	Streamflow Metric *cfs*	Scenario	
		Baseline	Baseline Drought
% Time Below Streamflow Metric	100	0.91%	1.02%
	600	23.6%	23.9%
**% Time is calculated as a proportion of the full model period (1939-2018).*			

*Results table from the 2023 Upper Flint RWP.

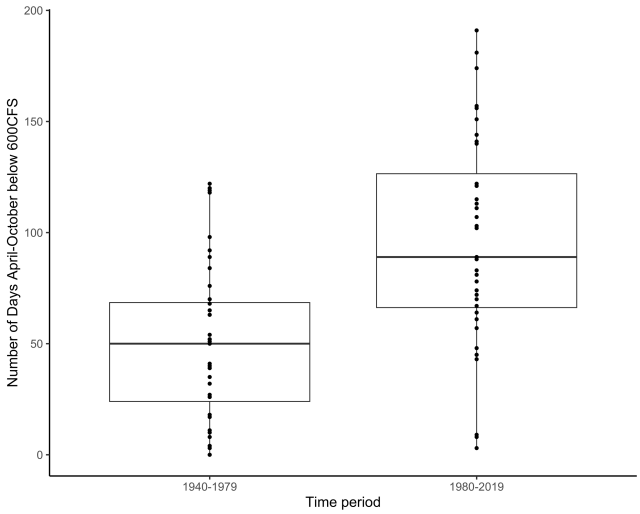
Interpreting these metrics as percent of total time exceeded presents a challenge. For example, 1% of time below 100 cfs (“more rocks than water” condition) could reflect annual events of 3-4 days each year of the 80-year period, or events lasting over a month once every 10 years. The ecological consequences of these scenarios could be substantially different, depending on an organism’s ability to withstand stagnant water or emersion. Similarly, the effect of flows below the river-recreation threshold may depend on whether those low flows occur as one “poor boating” year out of every four or represent three months of lost recreation during the period of highest demand every year. Thus, to interpret the ecological consequences or the impact on

recreation of flows under a given scenario, it is relevant to consider the seasonality, duration and frequency of individual flow excursions below ecological and recreational thresholds. Recreational paddling (best supported when flows exceed 600 cfs) is concentrated between April and October, which overlaps with the seasonally low flows that impact shoal habitat for aquatic organisms (Flint River flows are generally higher in winter and spring and lowest during summer and fall). Extreme low flows that lead to riverbed drying ("more rocks than water" condition; 100 cfs) are most likely to occur and overlap with potentially stressful, elevated water temperatures from June to October.

One can use the record for the Carsonville gage to evaluate the historic annual occurrences of seasonal flows below the thresholds for recreational boating (and shoal habitat) and river drying. Because we did not have the forecasted demand data available to compare historic and future scenarios, we split the historical record at the Carsonville gage into two 40-year periods to illustrate how one could evaluate changes in recreation and ecological metrics between time periods. In the context of water planning, one would compare the agreed-upon baseline or current conditions to a future scenario.

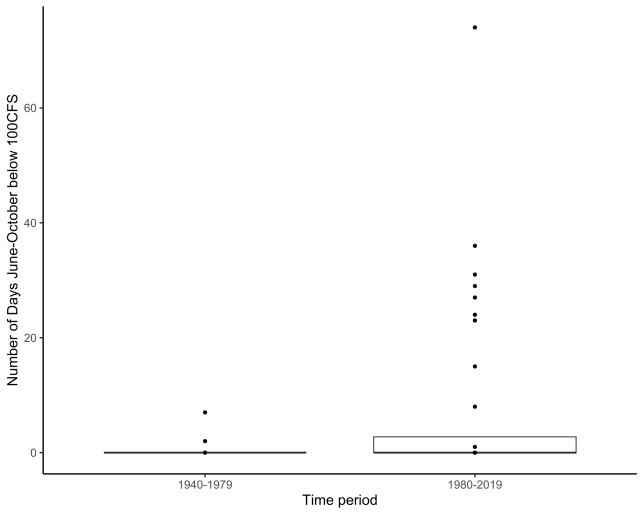
River flows recorded at the Carsonville gage were below 600 cfs and 100 cfs more often and for more days in the years 1980-2019 than in the earlier period, 1940-1979. These changes could be consequential. In the 1980-2019 period, the time that the river was below the paddling threshold almost doubled compared to the prior 40-year period, with nearly half of the years having unsuitable recreational flows for much or most of the season (Figure 1). Flows below 100 cfs rarely occurred between 1940-1979, but in the period 1980-2019 they occurred in about 25% of years and for up to 74 days (Figure 2).

Observing a shift like this in the summer and fall baseflow thresholds would raise a flag that river flows are trending lower for longer during the months evaluated. If these trends were to appear for a water planning scenario, it would be relevant to consider potential causes or evaluate alternative management actions that could mitigate the occurrence or length of these events.



	1940-1979	1980-2019
# YEARS	40	40
% OF TIME BELOW 600 (APRIL-OCTOBER)	23.4	44.8
# YEARS WITH OCCURRENCE OF FLOWS BELOW: 600 CFS	38	40
MEDIAN # DAYS APRIL-OCTOBER WITH FLOWS BELOW: 600 CFS	50	89
MAX # DAYS APRIL-OCTOBER WITH FLOWS BELOW: 600 CFS	122	191

Figure 1. Boxplot of the annual number of days between April and October that flows were below 600 cfs at the Carsonville gage on the Flint River. In the boxes, 25 percent of the data fall below the lower line, the middle line is the median, and 75 percent of the data are above the upper black line. This type of figure helps visualize the spread of the occurrence and duration of events below the 600 cfs threshold. The table summarizes the total percent of time and the median number of days each year, and the maximum number of days in one year, below 600 cfs during the recreational season.



	1940-1979	1980-2019
# YEARS	40	40
% OF TIME BELOW 100 (JUNE-OCTOBER)	0.15	4.8
# YEARS WITH OCCURRENCE OF FLOWS BELOW: 100 cfs	2	11
MAX # DAYS JUNE-OCTOBER WITH FLOWS BELOW: 100 cfs	7	74

Figure 2. Boxplot of the annual number of days between June and October that flows were below 100 cfs at the Carsonville gage on the Flint River. In the boxes, 25 percent of the data fall below the lower line, the middle line is the median, and 75 percent of the data are above the upper black line. This type of figure helps visualize the spread of the occurrence and duration of events below the 100 cfs "more rocks than water" threshold. The table summarizes the total percent of time, the number of years with occurrence (i.e., at least one day), and the maximum number of days in a single year with flows below 100 cfs during each time period.