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Prepared in cooperation with the Alabama Department of Conservation and Natural Resources, Alabama Power Company, U.S. Fish and Wildlife Service, and R.L. Harris Dam Adaptive Management Stakeholders

Adaptive Management of Flows from R.L. Harris Dam (Tallapoosa River, Alabama)—Stakeholder Process and Use of Biological Monitoring Data for Decision Making

Open-File Report 2019–1026

U.S. Department of the Interior
U.S. Geological Survey

Cover. Photograph showing collection of depth and velocity data at an electrofishing sample location. Taken at Hillabee Creek on May 26, 2010, by M. Martin, Alabama Cooperative Fish and Wildlife Research Unit.

Back cover. Photograph showing a prepositioned area electrofisher sample collection in action. Two workers hold a seine at the downstream end of the electrofisher and, after the electricity ceases, another worker (on right) visually inspects the area and kicks through the substrata to dislodge any fish that are entrained. Taken at Hillabee Creek on May 26, 2010, by M. Martin, Alabama Cooperative Fish and Wildlife Research Unit.

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Edited by Elise R. Irwin

Chapter A

Adaptive Management of a Regulated River—Process for Stakeholder Engagement and Consequences to Objectives

By Elise R. Irwin, Mary C. Freeman, James Peterson, Kathryn D.M. Kennedy, and M. Clint Lloyd

Chapter B

Long-Term Dynamic Occupancy of Shoal-Dwelling Fishes Above and Below a Hydropeaking Dam

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Chapter C

Macroinvertebrate Community Structure in Relation to Variation in Hydrology Associated with Hydropower

By Kristie M. Ouellette Coffman, Ely Kosnicki, M. Clint Lloyd, Tom Hess, and Elise R. Irwin

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U.S. Department of the Interior
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U.S. Department of the Interior
DAVID BERNHARDT, Secretary

U.S. Geological Survey
James F. Reilly II, Director

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Flotilla of boats parked on a gravel bar located in Hillabee Creek. Taken on October 26, 2005, by B. Martin, Alabama Cooperative Fish and Wildlife Research Unit.

Black redhorse (*Moxostoma duquenei*, 274 millimeters) captured and released from Hillabee Creek. Taken on September15, 2008, by M. Martin, Alabama Cooperative Fish and Wildlife Research Unit.

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Conversion Factors

International System of Units to U.S. customary units

Multiply	By	To obtain
	Length	
centimeter (cm)	0.3937	inch (in.)
millimeter (mm)	0.03937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
meter (m)	1.094	yard (yd)
	Flow rate	
cubic meter per second (m^3/s)	70.07	acre-foot per day (acre-ft/d)
cubic meter per second (m^3/s)	35.31	cubic foot per second (ft^3/s)

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$°F = (1.8 \times °C) + 32.$$

Temperature in degrees Fahrenheit (°F) may be converted to degrees Celsius (°C) as follows:

$$°C = (°F - 32) / 1.8.$$

Supplemental Information

A water year is the 12-month period from October 1 to September 30 and is designated by the year in which it ends; for example, water year 2015 was from October 1, 2014, to September 30, 2015.

Abbreviations

AMP	Adaptive Management Project
APC	Alabama Power Company
BBN	Bayesian Belief Network
CPT	conditional probability table
EPT	Ephemeroptera, Plecoptera, and Trichoptera
FERC	Federal Energy Regulatory Commission
FFG	functional feeding group
NMDS	nonmetric multidimensional scaling
PAE	prepositioned area electrofisher
PERMANOVA	permutational multivariate analysis of variance
PHABSIM	physical habitat simulation [model]
PMDI	Palmer’s Modified Drought Index
SIMPER	similarity percentage [analysis]
USGS	U.S. Geological Survey

Adaptive Management of Flows from R.L. Harris Dam (Tallapoosa River, Alabama)—Stakeholder Process and Use of Biological Monitoring Data for Decision Making

Edited by Elise R. Irwin[1,2]

Abstract

Adaptive management has been applied to problems with multiple conflicting objectives in various natural resources settings to learn how management actions affect divergent values regarding system response. Hydropower applications have only recently begun to emerge in the field, yet in the specific example reported herein, stakeholders invested in determining the best management alternatives for attainment of a suite of objectives outlined in a long-term adaptive management program below R.L. Harris Dam, a large, privately owned dam in Alabama. Stakeholders convened an objective-setting workshop to engage a governance structure and developed a decision support model to determine appropriate actions that optimized stakeholder values. The process led to implemented change in dam operation inclusive of incorporating hypothetical responses in system parameters to management. To account for the iterative loop of adaptive management, yearly monitoring of state variables that approximated many stakeholder objectives was performed from 2005 to 2016 and data collected were incorporated into the decision model. Specific analysis of fish and macroinvertebrate population responses indicated a less than satisfactory response for some stakeholders to the flow-management changes at the dam. Uncertainty regarding the best management to provide adequate hydrologic and thermal habitats for fauna and boatable days for recreationists still exists. The project led to a Federal Energy Regulatory Commission process for renewing the license to operate the dam (beginning in 2018); adaptive management could be a viable path forward to ensure stakeholder satisfaction related to new management options.

Introduction

Freshwater resources are a basic need of society and ecosystems. Because the management of water is multiobjective for diverse users, conflicts are inevitable when environmental concerns are pitted against economic interests. Adaptive management and decision analysis can account for multiple competing objectives identified by stakeholders, and these frameworks are applicable to water issues. Recently, there has been a call for large-scale flow manipulations in rivers to facilitate rapid learning so that theoretical frameworks can be transferred into knowledge of system function. Adaptive management is an iterative process that facilitates learning by making predictions relative to system uncertainty (decision support or other models), applying the scientific process to monitor effects of management actions that are applied to optimize resource objectives, and updating the knowledge base (relative to predicted responses) to improve future management.

Purpose and Scope

This report describes the implementation and long-term application of adaptive management in a regulated river in the southeastern United States (Tallapoosa River, Alabama; R.L. Harris Dam, owned by Alabama Power Company). The implementation process included stakeholder involvement, development of flow prescriptions, predictions regarding system response, and design and implementation of a long-term monitoring program. We present the decision network used to assist stakeholders in prescribing initial flow manipulations, and 12 years of model updates from the monitoring program including assessment of the fish and macroinvertebrate communities at sites regulated by the dam and at unregulated sites.

Chapter A

Stakeholder engagement was key to implementation and long-term monitoring of potential system change related to management actions. Hypotheses regarding how the system state would change with increased base flow, better spawning conditions, and improved thermal conditions formed the underpinnings of the decision model. The model (Bayesian Belief Network) was constructed to (1) predict the consequences of different combinations of management actions that

[1]U.S. Geological Survey.

[2]Alabama Cooperative Fish and Wildlife Research Unit.

changed flows from the dam on multiple stakeholder objectives, (2) assist in selecting the decision that best satisfied equally weighted stakeholder objectives, and (3) incorporate data collected yearly to update probabilities (that is, Bayesian updating) quantifying how management actions contributed to meeting stakeholder objectives. A total of three management decisions were linked in the Bayesian Belief Network; flows from the dam either maintained the status quo or provided flows that variably matched unregulated flows in the river upstream from the reservoir impounded by the dam. A total of two other decisions were modeled: the provision of spring and summer, spring only, or summer only flows for spawning conditions and the provision (or not) of October boating flows.

Data to inform the model were supplied by Alabama Power Company (management decision, lake levels), collected through direct monitoring (fish population, temperature), or incorporated from U.S. Geological Survey streamgage data (reservoir inflows, boatable days). The initial decision was to implement the Alabama Power Company flow option, coupled with spring and summer periods for spawning (10 day, no discharge over 7,000 cubic feet per second), and October boating flows (discharge was 250–1,500 cubic feet per second on weekends). The portfolio of actions was termed the "Green Plan" and was implemented in 2005 along with a stakeholder governance structure to guide the project engagement. As the model was updated with the 12 years of monitoring and case data, the optimal decision changed, indicating uncertainty regarding the management options that best satisfied stakeholder objectives. The flow management options predicted to satisfy stakeholders most often were flows that matched the U.S. Geological Survey Tallapoosa River near Heflin, Ala. (U.S. Geological Survey streamgage 024112000, hereafter, "Heflin," unregulated upstream), streamgage or the Green Plan, with the addition of spring and summer spawning flows and October boating flows. In practice, the Green Plan was always the flow decision, and with few exceptions, spring and summer spawning flow conditions were provided; however, October boating flows were never provided as a management option. Responses of fish populations were not positively related to changes in management contrary to model predictions. This finding further illustrated uncertainty regarding how management influenced stakeholder objectives and indicated that some stakeholder objectives were not satisfied.

Chapter B

A critical aspect of any adaptive management project is the monitoring of system response to the management protocols imposed. Based on the decision model, it was determined that the response of biological objectives to management was uncertain; reducing the uncertainty regarding the responses was an overall goal of the project. To reach this goal, we monitored dynamic occupancy (that is, two metapopulation processes: persistence and colonization) for 38 fish species (species for which we had more than 25 detections) at 25 sites above and below R.L. Harris Dam. Sites were selected in a stratified-random approach by surveying in shoal habitats in 2 unregulated reaches (upper Tallapoosa River and Hillabee Creek) and 3 regulated reaches (main stem Tallapoosa River; from R.L. Harris Dam to the Malone Bridge, from Malone Bridge to Wadley Bridge, and the Horseshoe Bend area). We sampled shoals for fishes (and macroinvertebrates; chapter C) twice a year (summer and fall) for 12 years (2005–16) using prepositioned area electrofishers, resulting in occurrence records of over 81,900 individuals of 46 species. In general, fish density and species richness were depressed in the regulated versus the unregulated reaches. We modeled the effects of river regulation, distance from the dam, power generation, and temperature on probability of both persistence (species present on a shoal from year to year) and colonization (species present on a shoal where it was absent the previous year). Fishes persisted and colonized at lower probabilities at flow-regulated sites (downstream from R.L. Harris Dam) than at the unregulated sites in the upper Tallapoosa (Heflin) main stem and in Hillabee Creek. Estimated rates varied considerably among sites and among years; however, the estimated mean effects (across all species) of locations downstream from the dam were to lower the persistence by 81 percent and the probability of colonization by 65 percent. The effects of a site being located in the flow-regulated reach also varied among species for both persistence and colonization; however, only a single species (*Ambloplites ariommus* [shadow bass]) had an estimated effect that was positive.

We found evidence that both flow instability and depressed temperatures influenced fish persistence and colonization at the flow-regulated sites. Averaged over all taxa, fish persistence increased with greater distance from the dam and decreased during years with more generation events. Colonization was lower in years with more generation events; however, no clear increases in colonization were observed at the sites farthest from the dam. Persistence was not clearly related to warmer thermal regimes but increased at sites farthest from the dam. Fish colonization increased during years with warmer water temperatures.

Chapter C

Rapid changes in river stage and temperature are known to affect benthic invertebrate assemblage structure; research indicates that hydropeaking can reduce diversity and increase biomass of certain species. Some have suggested that management decisions should consider sensitive benthic species; adaptive management frameworks have been called upon to determine water release scenarios below dams that enhance invertebrate production and diversity.

In 2005, adaptive management was implemented to determine effects of flow prescriptions on state variables of interest to stakeholders in the regulated part of the Tallapoosa River below R.L. Harris Dam. Thus far, decisions have been based on projections of fish population responses to management. By quantifying the relations between hydrology and the invertebrate assemblage structure in our study reaches, decisions regarding effective flow management could potentially be implemented more frequently because invertebrates should respond more rapidly than fishes to changes in management.

Since sampling began in 2005, macroinvertebrates were collected as part of the adaptive management monitoring protocol. In 2016, a macroinvertebrate team began to sort and identify macroinvertebrate specimens within the stored collections. The prioritization of sample processing was designed to select for maximum variation in natural hydrologic variables as indicated by Palmer's Modified Drought Index, a monthly metric available through the U.S. Geological Survey, to confirm macroinvertebrate communities display a response to alterations of hydrologic variables. We identified (to the lowest taxonomic level possible for each taxa) invertebrate specimens, calculated reach level community similarity metrics, and modeled the relations between faunal functional feeding group and habit, degree of river regulation, and yearly variation in hydrology.

Analyses based on the additive results of the identification of 4 samples from 3 shoals within each of 4 reaches (Main stem Tallapoosa River from R.L. Harris Dam to the Malone Bridge and from Malone Bridge to Wadley Bridge representing regulated reaches, and Upper Tallapoosa River and Hillabee Creek representing unregulated reaches) encompassing 5 years (2005, 2008, 2009, 2012, and 2014) indicate that the macroinvertebrate communities downstream from the R.L. Harris Dam have overall lower diversity, greater density driven by increased abundances of flow disturbance tolerant taxa, and the exclusion of some flow sensitive species from regulated reaches. Ordination of the macroinvertebrate community composition per shoal indicates a shift in community composition based on regulation, water year, and the interaction of year and regulation type (permutational multivariate analysis of variance results: probability [p] less than 0.001 for year, regulation type, and year×regulation type interaction). Among those taxa that contribute the most to the deviations in the similarities of the community composition as displayed on the ordination, flow-tolerant species display greater abundances within regulated reaches, and inclusions of large groups of these taxa are driving the similarities of closely ordinated regulated and unregulated sites. When analyzing the association of functional feeding group and habit with the ordination of the macroinvertebrate community, 4 of 6 functional feeding groups and 4 of 5 habits tested had a significant association with the ordination of macroinvertebrate shoal community composition; vectors for filterer/collectors, swimmers, and climbers ordinated towards regulated reaches, and vectors for predators, gatherer/collectors, clingers, and burrowers ordinated towards unregulated reaches, further illustrating the shift in community composition to flow-tolerant taxa in regulated reaches. Cluster analysis based on the presence/absence matrix of shoal community composition also indicated differences in the community composition between regulated and unregulated reaches, and a unique community composition in the downstream reach closest to the dam in all years analyzed, likely because of the regular absence of many flow sensitive species from regulated reaches. Results of the macroinvertebrate community composition in regulated and unregulated reaches of the mid-Tallapoosa River Basin suggest agreement with the standing published literature in which regulated reaches display a reduced richness and increase density of flow disturbance tolerant taxa. These results indicate that the Green Plan did not meet the stakeholder objective to restore and maintain macroinvertebrate community composition similar to unregulated reaches within the regulated portions of the river.

Peaking hydropower facilities that produce pulsed, high-velocity discharge create an unstable environment including adverse effects on hydraulic variables such as water velocity, depth, and temperature. The flow decision model and other analyses in this report indicate that stakeholder resource objectives, faunal communities, and fish and macroinvertebrate populations were negatively impacted by river regulation below the R.L. Harris Dam. Despite long-term (12 years) monitoring of fishes and macroinvertebrates in this adaptive management program, high levels of uncertainty regarding the ability to predict species-specific population response to management remain. One major impediment to defining specific flow and temperature regimes that would lessen the impacts of the dam on this socioecological system is engineering constraints in the design of the dam that make active adaptive management approaches difficult.

Implications for Adaptive Management

Despite potential obstacles, an adaptive management approach still holds substantial promise for improving the management of regulated rivers by allowing managers and scientists to address the uncertainty in predicting and measuring how river fauna will respond to flow-regime alterations. However, the recognition of uncertainty and the use of decision-analytic processes to reduce uncertainty attributed to flow management decisions with model updating through careful, replicated monitoring programs contribute to a viable path forward. Although monitoring data were used to assess stakeholder objectives using the decision model, the additional analysis to identify ecological responses to elements of dam operation could inform additional flow management regimes during relicensing. If flow and thermal alteration from the dam can be modified toward improving natural resource objectives, adaptive management processes and long-term monitoring could further reduce uncertainty related to biotic response to new Federal Energy Regulatory Commission licensing requirements.

Sampling site on the Tallapoosa River below R.L. Harris Dam. A prepositioned area electrofisher is visible in the lower right part of the photograph (white PVC frame). Taken on October 19, 2005, by B. Martin, Alabama Cooperative Fish and Wildlife Research Unit.

Chapter A

Adaptive Management of a Regulated River—Process for Stakeholder Engagement and Consequences to Objectives

By Elise R. Irwin,[1,2] Mary C. Freeman,[1,3] James Peterson,[1,4] Kathryn D.M. Kennedy,[2] and M. Clint Lloyd[2]

Introduction

Freshwater resources are a basic need for societal and ecosystem functions. Because the management of water involves multiple objectives and users, conflicts arise when environmental concerns are pitted against economic interests. Structured decision making and adaptive management can facilitate the resolution of conflicts by incorporating multiple stakeholder objectives into a framework relating system function and dynamics to water resource issues and decisions (Harwell, 1998; Irwin and Freeman, 2002; Poff and others, 2003; Raadgever and others, 2008; Irwin, 2014). Several high-profile aquatic socioecosystems in the United States have been identified as candidates for adaptive management. Examples include the Florida Everglades (Harwell, 1998; Gunderson and Light, 2006), the Colorado River (National Research Council, 1999), and southeastern regulated rivers (Irwin and Freeman, 2002; Richter and Thomas, 2007; Irwin, 2014). In a review article highlighting the decade-long Alabama-Florida-Georgia "water war," Poff and others (2003) call for large-scale river flow manipulations to facilitate rapid learning to improve management. Adaptive management is an iterative process that provides an appropriate context for meeting this call. Within the adaptive management framework, multiple hypotheses—each represented by a model (or set of models) weighted by a plausibility or probability—predict system response to management actions. The optimal management decision is then selected based on the current system state and a prediction of the expected future state, taking into account various sources of uncertainty. For dynamic decision-making situations, model probabilities are updated by comparing model-specific predictions to observed (actual) future conditions (t+1; where t is a time step such as months or years). The adjusted model probabilities then are used to predict future conditions and, as a result, future optimal management decisions. A cyclical feedback loop explicitly provides for learning through time with the possibility of resolving competing hypotheses.

[1]U.S. Geological Survey.

[2]Alabama Cooperative Fish and Wildlife Research Unit.

[3]Patuxent Wildlife Research Center.

[4]Oregon Cooperative Fish and Wildlife Research Unit.

Adaptive Management of Regulated Rivers

Approaches to regulated river management, especially with respect to instream flow needs for biota, have historically focused on one-time negotiated flow settlements often related to the requirements of the Federal Energy Regulatory Commission (FERC; see Russo, 1999). In addition, the tools used to arrive at flow requirements have generally been based on models that do not explicitly incorporate uncertainty in physical, biological, or social components of socioecosystems. Monitoring of system responses to flow management has been rare once a settlement has been negotiated; there is usually no flexibility (or desire) to change management options. More recently, others have advocated the wider use of adaptive management to improve instream flow management (Castleberry and others, 1996; Van Winkle and others, 1997; Walters, 1997; Johnson, 1999; Irwin and Freeman, 2002; Poff and others, 2003; Irwin, 2014; Poff, 2017; McManamay and others, 2016), including several examples of adaptive management associated with FERC licenses (see below). Flow settlements at a few western U.S. dams have required additional scientific studies to address uncertainties and to support future adjustments to flow requirements (Castleberry and others, 1996; Van Winkle and others, 1997), and a few FERC licenses (Pearsall and others, 2005; Podolak and Yarnell, 2015) include an adaptive management process associated with the licenses. Yoccoz and others (2001) recognized adaptive management as an efficient framework for explicitly integrating management objectives, actions, and monitoring data for the purpose of improving management decision making. Coupling science-based monitoring with flow manipulations implemented to meet regulatory or societal requirements will allow for improved understanding of the functional relations among system biota and flow regime components (Irwin and Freeman, 2002; Irwin, 2014). Adaptive management focuses on the achievement of management goals with learning through monitoring, and emphasis on the reduction of system uncertainties that affect decision making (Williams and others, 2007). However, the key to the success of adaptive management of trust resources is stakeholder involvement and agreement (McLain and Lee, 1996). Stakeholders must agree on management objectives and governance for the framework to operate successfully (Williams and Johnson, 1995; Williams and others, 2007; Conroy and Peterson, 2013). Decisions with multiple competing objectives, decision alternatives, and complex system dynamics have the potential to become overly technical, thereby excluding the involvement by stakeholders throughout the process; therefore, reaching a starting point to begin adaptive management is not an easy process.

To overcome the threat of stakeholder abandonment, managers need a transparent process for identifying stakeholder objectives and establishing a governance structure for making decisions. Ideally, the process would involve scientists only as technical experts; neutral parties that develop the models linking potential management actions and stakeholder objectives. An ideal process also would include, to the extent possible, decision support models that are transparent and generally understandable by nontechnical stakeholders to facilitate understanding and buy-in to the process. Unfortunately, there are few examples of stakeholder-driven processes that have led to the successful implementation of adaptive management (Gregory and others, 2012; Conroy and Peterson, 2013). Here, we describe the development, implementation, and sustained prescription of adaptive management in the Tallapoosa River, Alabama, a process that included substantial stakeholder involvement, the development of a decision model used to aid determination of initial flow prescriptions, and model updating over an 11-year period (2005–16) to inform future management. Although the specific application is regional, we believe that the framework and lessons learned are applicable to managed ecological systems worldwide.

Methods

The Tallapoosa River below R.L. Harris Dam is a strongly flow-regulated reach in the Piedmont region of east-central Alabama (fig. A1). The system under study is a 78-kilometer (km) reach of the Tallapoosa River beginning at R.L. Harris Dam and terminating in the headwaters of Lake Martin (not shown). R.L. Harris Dam was constructed primarily as a hydropower facility, with other potential benefits including flood control, recreational opportunities on the reservoir created by the dam, and economic growth associated with the reservoir. The generation capacity for the 2-turbine facility is 135 megawatts, which accounts for about 10 percent of the total capacity of the 11 privately owned hydropower dams in the eastern Mobile River Basin.

Management Context

Since going into service in 1983, R.L. Harris Dam has been operated primarily as a hydropeaking facility, such that water is released in pulses, usually 4–6 hours in duration, through one or two turbines, each with the capacity to pass 226 cubic meters per second (m^3/s). Historically, generation events were once or twice daily, 5 days a week, and usually included no generation on weekends (that is, “status quo” scenario in models below). As a result of the hydropeaking operation, the flow regime through the study reach typically fluctuated between extreme low flows and high flows corresponding to one- or two-turbine generation (fig. A2). Comparison of pre- and postdam hydrographs indicated changes in multiple aspects of the flow regime; for example, high flows were dampened, low flows were lower and more frequent, and seasonal shifts in flow magnitude were apparent (Irwin and Freeman, 2002). In addition to changes in the flow regime, the thermal regime below the dam has also been affected; during spring and summer months, temperature decreases as much as 10 degrees Celsius (°C) during generation events (Irwin and Freeman, 2002). During nongeneration periods, the FERC license for R.L. Harris Dam requires that flow as recorded at the U.S. Geological Survey (USGS) streamgage at Wadley, Ala. (22 km downstream from the dam; U.S. Geological Survey streamgage 02414500 [U.S. Geological Survey, 2018]), is not to fall below the predam historical record low flow of 1.27 m^3/s.

The study reach represents one of the longest and highest-quality segments of Piedmont river habitat remaining in the Mobile River Basin, one of the most biologically diverse river drainages in North America (Lydeard and Mayden, 1995; Freeman and others, 2005). Extensive areas of shoal habitat, river features that typically support high faunal diversity and that have been replaced by impoundments throughout much of the Southeast, are characteristic along this part of the river. The native fish assemblage includes at least 57 species, including at least 7 species endemic to the Tallapoosa River system. Of the fish species, several are considered “at risk” by the U.S. Fish and Wildlife Service; however, no fish species are on the list of threatened and endangered species regulated by the Endangered Species Act of 1973. A total of 1 fish species, 3 crayfish species, and 4 mussel species are listed as Greatest Conservation Need species by the State of Alabama (Wood, 2015). Management needs for State Greatest Conservation Need fish and invertebrate species were considered when defining management options. One Federally listed unionid mussel species (*Hamiota altilis* [finelined pocketbook]) may have historically been in the river reach below R.L. Harris Dam; its current perceived absence also affected management options discussed by stakeholders, specifically in regard to providing habitat for reintroduction. Before construction of R.L. Harris Dam, the study reach also supported productive sport fisheries for native black basses (*Micropterus* spp.) and catfishes (primarily *Ictalurus punctatus* [channel catfish], and *Pylodictis olivaris* [flathead catfish]), as well as river

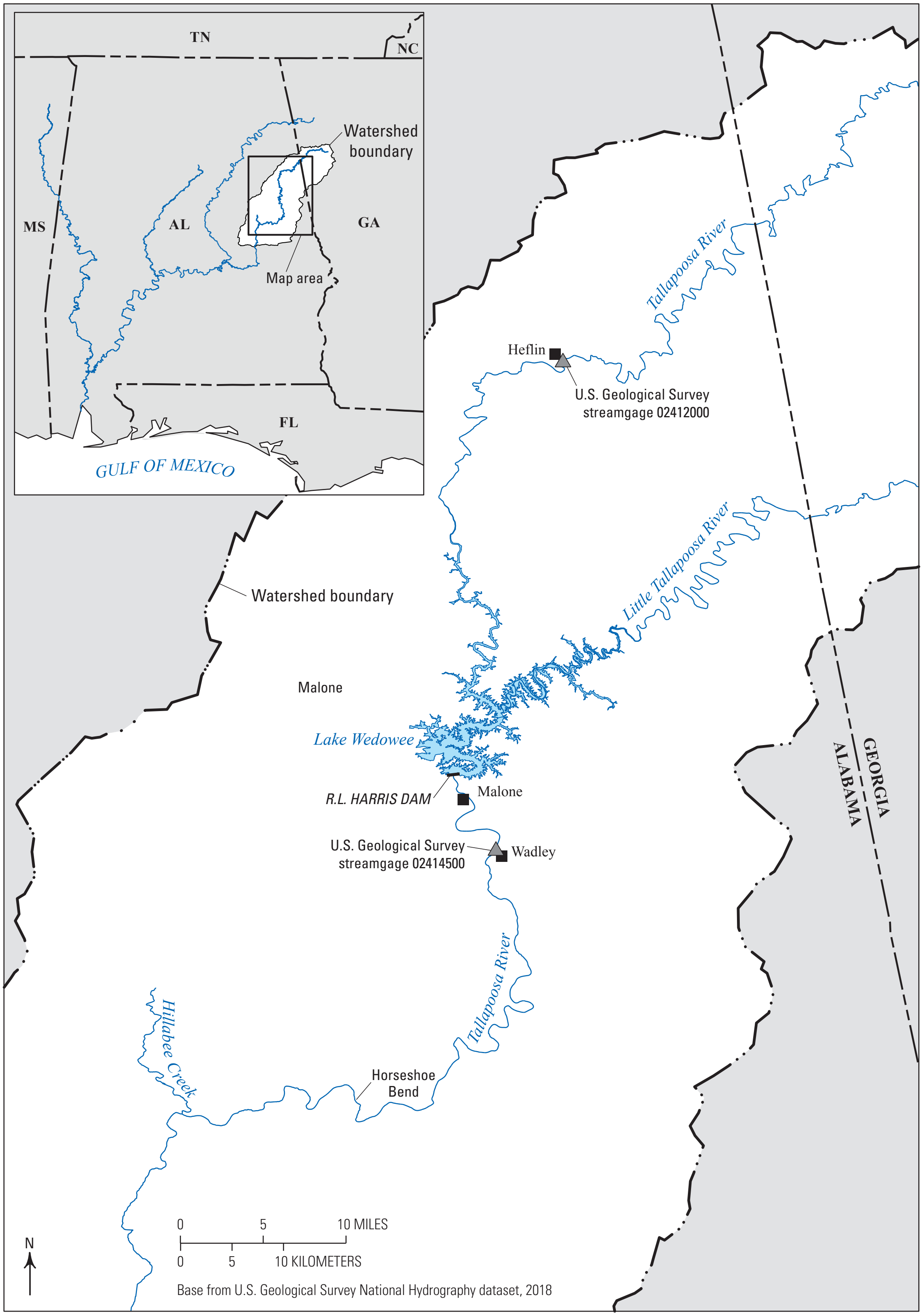

Figure A1. Location of study site in the Tallapoosa River Basin. The river is regulated below R.L. Harris Dam and unregulated above Lake Wedowee. U.S. Geological Survey streamgages are maintained at Heflin and Wadley, Alabama (see figure B1 for sampling reach locations).

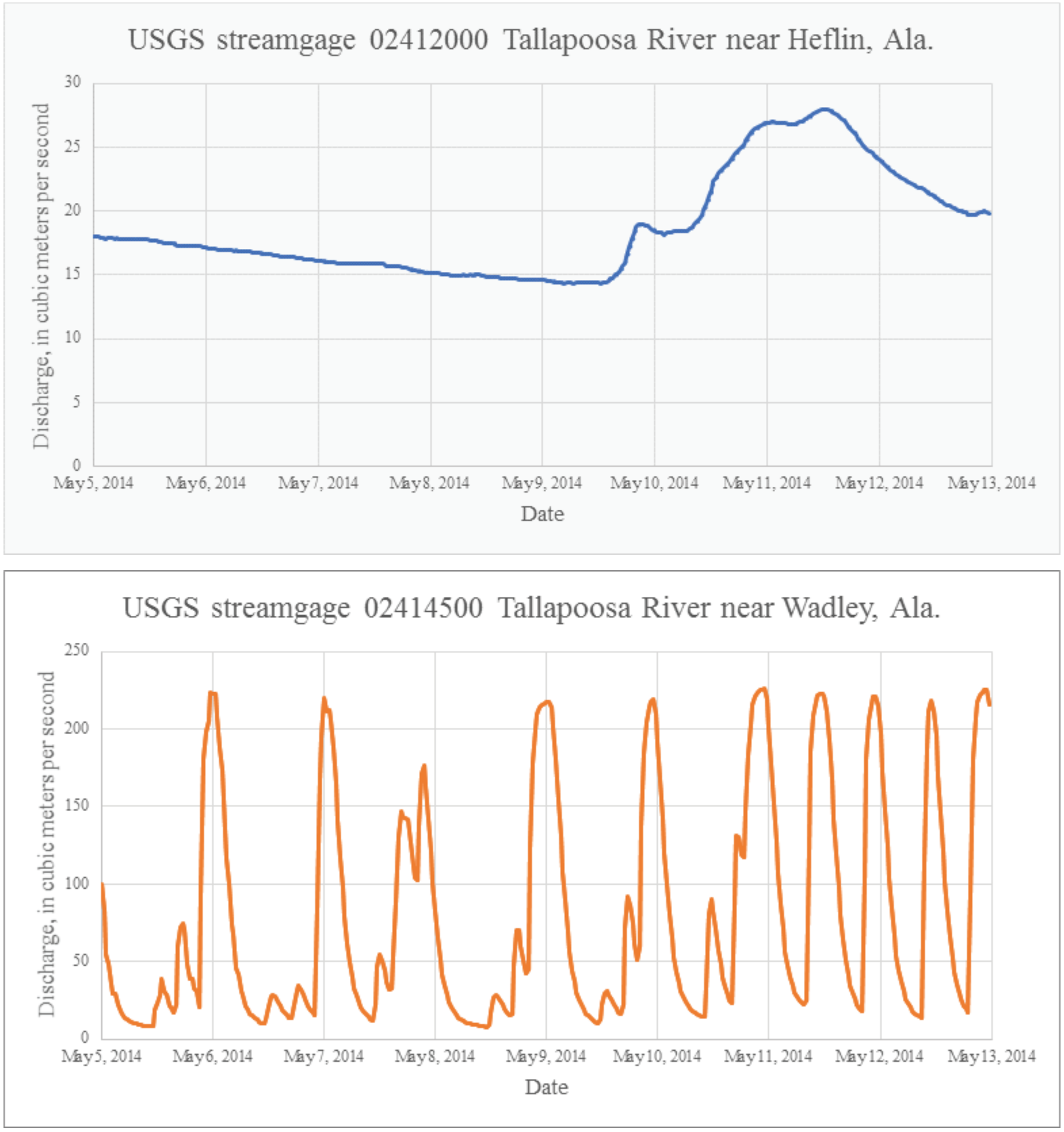

Figure A2. Tallapoosa River discharge measured at U.S. Geological Survey (USGS) streamgage 02412000 (top panel—naturally occurring flows) located near Heflin, Alabama, and USGS streamgage 02414500 (bottom panel—regulated by R.L. Harris Dam) located in Wadley, Alabama, 22 kilometers below the dam (May 5–12, 2014; U.S. Geological Survey, 2018; data are reported in cubic meters per second). [The Green Plan called for passing daily volumes of water equivalent to or greater than those recorded by the Heflin streamgage, which is located above the reservoir and measures flows that amount to about 50 percent of inflows into the reservoir.]

boating activities (D. Catchings, Alabama Department of Conservation and Natural Resources, oral commun., 2002). A decline in sport fish populations and the loss of access to the river because of changes in flow regime have been major stakeholder concerns since construction of R.L. Harris Dam. Conversely, altering the peaking operation could threaten the power utility's flexibility to provide and sell electricity on demand during periods of peak consumption. Changes in dam operation could also affect water levels and therefore values for users in the reservoirs, particularly at Harris Lake.

Management issues in the study reach below R.L. Harris Dam have therefore revolved around the effects of hydropower operation on values associated with power production needs; water availability for economic development, consumption, boating, angling and other recreational activities (upstream and downstream from the dam); and the general health of the Tallapoosa River ecosystem. Although these conflicting management objectives had been recognized for many years, the ability of stakeholders to reach consensus regarding implementation of changes in the flow regime had not been realized. During the 1990s, multiple stakeholders had communicated their desire for development of a plan of action. One option was to ask FERC to reopen the regulatory license and order an evaluation of the dam operation with respect to competing objectives. Reopening the license was not desirable to the power company, particularly in light of previous experiences where a reopened regulatory license resulted in a renegotiated flow regime developed without options to amend the license based on meeting (or not meeting) stakeholders' objectives. Formal discussions with stakeholders and the publication of Irwin and Freeman's (2002) framework provided a roadmap toward implementation of adaptive management in the system. The stakeholders recognized that quantification of system function during management would assist with reduction in uncertainty related to future FERC regulations—the license will be renewed in 2023.

Stakeholder Objectives

To begin the adaptive management process, we arranged a workshop to define stakeholder objectives. Our goal was to incorporate values associated with these conflicting objectives into a structured decision model to determine a starting point for adaptive management. From April 29 through May 1, 2003, we held a workshop open to all stakeholders in the middle Tallapoosa Basin to introduce the concept of adaptive management and to create an open discussion for building consensus on management objectives and values (see appendix A1 for transcripts of the workshop).

Following a series of presentations by experts in the field of adaptive natural resource management, professional facilitators (https://groupsolutions.us; under contract with USGS) convened an interactive session, beginning with an open forum for all workshop participants to suggest and discuss potential values and objectives. Suggested objectives were judged in an electronic poll by 1 representative from each of 23 participating stakeholder groups. From the poll results, a tentative list of objectives (Clemen, 1997) was drawn for discussion among stakeholders. Objectives identified by stakeholders were as follows (order does not imply rank):

1. Maximize economic development, primarily for municipalities.
2. Maximize diversity and abundance of native fauna and flora.
3. Minimize bank erosion downstream from R.L. Harris Dam.
4. Maximize water levels in the reservoir.
5. Maximize boating and angling opportunities downstream from R.L. Harris Dam.
6. Minimize total cost to the power utility.
7. Maximize power utility operation flexibility.
8. Minimize river fragmentation (that is, no new dams).
9. Minimize consumptive use of water resources by municipalities and other users (that is, agriculture).

Stakeholders ultimately agreed upon these objectives as complete and representative of the interests of all parties involved. These higher level objectives are objectives defined as what the stakeholders want to accomplish, not the way in which they will accomplish them (that is, means objectives). In addition, stakeholders agreed to adopt the concept of adaptive management as a framework for future discussions and management decisions.

Development of a Decision Support Model

Objectives established at the workshop were then used in the development of a decision model to assist stakeholders in making the complex decisions necessary to change the flow regime below R.L. Harris Dam. To conduct the decision analysis, we followed the basic steps outlined by Clemen (1997): (1) formed hypothesized relations between flow and system response, (2) constructed a basic model outlining these hypotheses, (3) parameterized the model, (4) determined the optimal decision from the model results, and (5) completed sensitivity analyses to determine which components of the model had the greatest effect on the decision.

Hypothesized Faunal Response

Using existing knowledge, expert opinion, and empirical data, we constructed hypothesized relations of faunal dependence on flow regime. Studies in the system have contributed to our knowledge of how fauna may respond to specific flow features in the system. Irwin and Freeman (2002) hypothesized that (1) depleted low flows, (2) flow instability, and (3) thermal-regime alteration were the features most likely to affect faunal response in the system.

Published findings indicate that hydrologic alteration (fig. A2) in the river has affected various biological processes. Irwin and Hornsby (unpublished data) repeated Swingle's 1951 rotenone survey (Swingle, 1954) in the regulated river near Horseshoe Bend and reported a major shift in community composition (from specialists to generalists) and declines in overall fish numbers and biomass. In 1951, the community consisted of catfishes (46 percent) and minnows (47 percent), and in 1996, the community was dominated by black basses and sunfishes (51 percent); whereas, catfishes and minnows comprised 22 and 5 percent of the community, respectively. With the exception of the recorded absence of *Lepomis megalotis* (longear sunfish [Irwin and Hornsby, unpublished data]; Andress, 2002; Martin, 2008), fish species composition seems to be stable; yet, Freeman and others (2001) reported that persistence of fishes in the flow-regulated section depended, in part, on periodically stable shoal habitat conditions that allowed reproduction and juvenile survival. Irwin and others

(1997), Andress (2002) and Martin (2008) reported disrupted spawning for sunfishes including *Micropterus punctulatus* (Alabama bass), *M. tallapoosae* (Tallapoosa bass), and *L. auritus* (redbreast sunfish). Nest success for redbreast sunfish was negatively related to both peaking power generation and depressed water temperatures (also caused by the dam; Andress, 2002; Martin, 2008). A list of some hypotheses related to flow features in the river is provided in table A1.

Model Structure

Once we established hypothesized relations between flow and biotic response, we incorporated these relations, along with various other management objectives, into an influence diagram (fig. A3) where relations among decision components are explicitly represented in graphical form (Clemen, 1997). In this figure, decision elements (or nodes) are represented by blue rectangles, consequence (or utility) nodes by pink hexagons, and uncertainty (or chance) nodes by yellow rectangles. Influence diagrams provide explicit representations of individual decision components and their dependencies.

State variables that were important for describing relations among outcomes representing high level objectives and flow regime were identified and incorporated into an influence diagram (state variables = uncertainty nodes, yellow rectangles; fig. A3) with casual relations (links) between management options (decision nodes, blue rectangles; fig. A3), state variables, and stakeholder values (utility nodes, pink hexagons; fig. A3). Using this structure, we developed a Bayesian belief network (Marcot and others, 2006), and a consequent Bayesian network model using Netica 1.12 (Norsys Software Corporation, 1998) to both quantify uncertainty regarding the response of the system to management actions and our understanding of system dynamics (hypotheses) relative to predicted response. A description of each model component (hereafter, “node”) represented in the model follows.

Decision Nodes

The decision alternatives included four differing primary flow regimes, the provision of spawning windows (periods during which flow fluctuations are minimized to allow for hypothesized increased spawning success), and increased weekend flows in October for recreational boating (table A2). Relations between flow and system response were modeled using probabilistic dependencies derived from long-term (1982–2004) empirical data from multiple projects and expert opinion; whereas, relations between system response and stakeholder satisfaction (that is, utility values) were based upon stakeholder opinion.

The primary set of decision alternatives concerned daily flow operations from the dam. Based on our hypotheses, increasing the flow level during nongeneration periods should have positive effects on the abundance, diversity, and growth of fishes (and likely other flow-dependent biota). Increased flow from the dam also should have positive benefits for river boaters and river landowners. We examined four alternatives to the primary decision set. The first was no change: keep the system at status quo (see details in description of study site above). The other three alternatives were based upon the concept of mimicking the flow regime recorded at the USGS streamgage in Heflin, at Wadley, 22 km below the dam. The Heflin streamgage measures flows in the unregulated upper portion of the Tallapoosa River (fig. A1); several stakeholders hypothesized that mimicking these flows at the dam would allow for some natural flow variability in the regulated portion of the river. The first of these alternatives was, in effect, modeled as a constant flow from the dam to maintain the Heflin target at Wadley (Heflin), which consisted of minimum flows plus any necessary generation flows. The second was similar, except the flow from the dam was to never reach levels below 8.5 m^3/s (Heflin 300). The third was an option proposed by the power utility, in which at least 75 percent of the Heflin target was maintained by 2–3 daily pulses, 1 at 0600 and 1 at 1200 (Alabama Power Company [APC]).

Table A1. Stated a priori hypotheses (not exclusive) regarding how fishes and habitat will respond to specific flow conditions.

[Developed using published literature and empirical data from the Tallapoosa River]

Hypothesized biotic response	Hypothesized flow linkage
Presence of fluvial specialists will be highest at unregulated sites.	Unregulated flows provide more stable conditions for habitat specialists.
Habitat persistence will be greatest at unregulated sites positively affecting recruitment processes.	Highly regulated flows negatively affect persistence of habitats critical for fauna because of rapidly changing stage.
Spawning success will be highest in years when spawning windows are provided.	Stable flows provided for spawning will increase recruitment of multiple fish species.
Filter-feeding invertebrate populations will respond positively to increased base flow.	Increased base flow dampens magnitude of disturbance on shoal habitats and increases flow through pools.

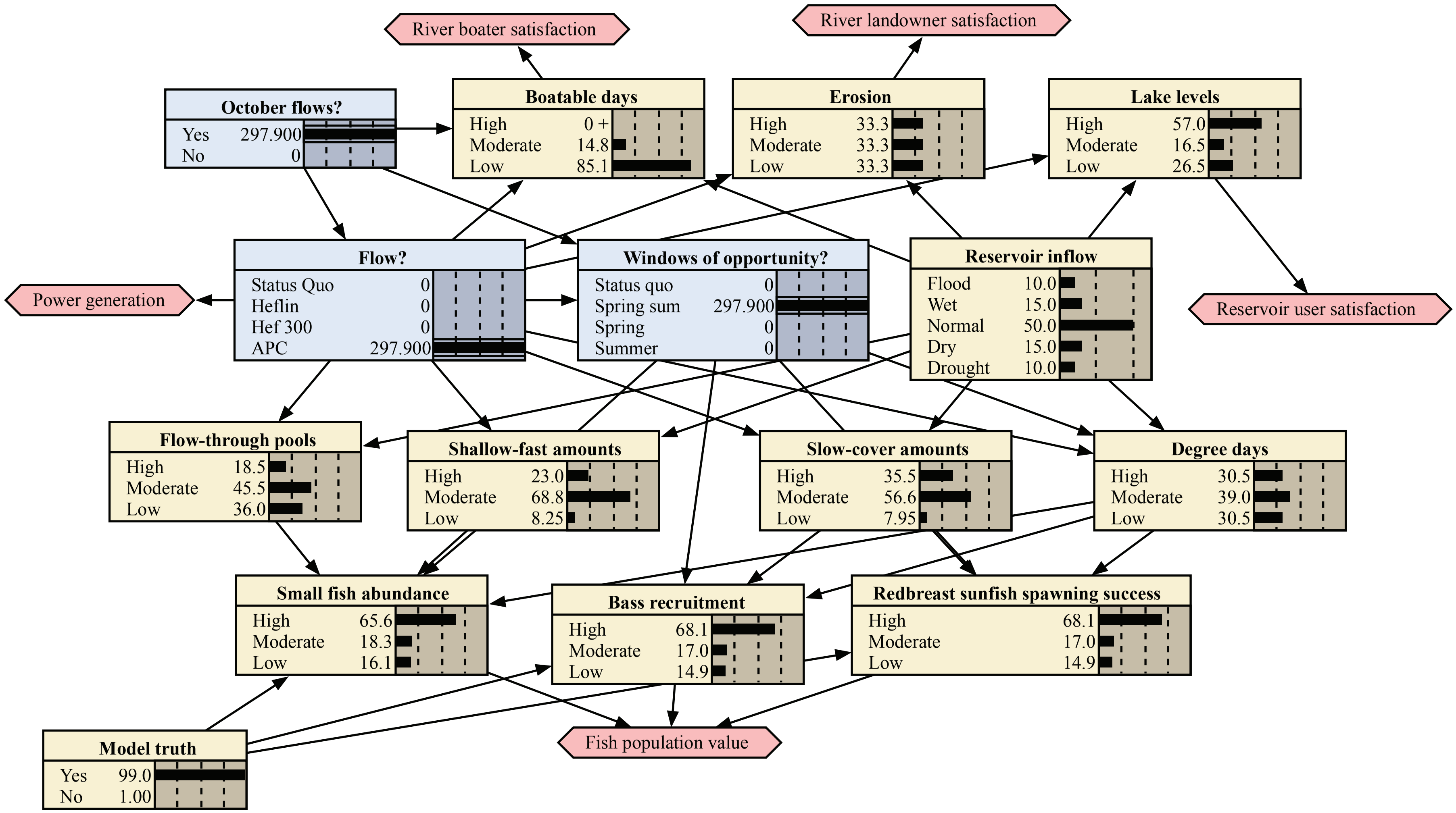

Figure A3. An influence diagram (with utility and decision nodes) showing the structure of the Bayes network and complexity of managing flows below R.L. Harris Dam. The blue rectangles are the decisions that were made about flow from the dam, the yellow rectangles are stakeholder objectives represented by measurable state variables, and the pink hexagons are consequence (utility) values of the different stakeholders. The initial flow management decision was the portfolio that maximized the satisfaction of the stakeholders (Alabama Power Company; sum of utility scores = 297.900).

Table A2. Description of the three decisions and their alternatives for the R.L. Harris Dam Adaptive Management Project.

[m^3/s, cubic meter per second; --, no data; APC, Alabama Power Company]

Flows from the dam	Description	Spawning conditions	Description	October flows	Description
Status quo	No change in current operations.	Status quo	No provision of spawning flows.	Yes	Boatable flows of 14.2–42.5 m^3/s were provided on weekends.
Heflin	Flows are constant to maintain the Heflin streamgage discharge at the Wadley streamgage.	Spring and summer	10-day window where flows did not exceed 198.2 m^3/s in both spring and summer.	No	No boatable flows were provided.
Heflin 300	Heflin scenario plus flow from dam did not drop below 8.5 m^3/s.	Spring	10-day window where flows did not exceed 198.2 m^3/s in spring only.	--	--
APC	75 percent of the Heflin flow was maintained through pulsing 2–3 times per day.	Summer	10-day window where flows did not exceed 198.2 m^3/s in summer only.	--	--

A second set of decision alternatives concerned spawning or "flow" windows. Based on our hypotheses, periods of stable flow without hydropeaking should increase opportunities for fish to spawn and larvae to develop successfully. Alternative decisions included no change (status quo), spawning windows in both spring and summer, spring windows only, and summer windows only. The third decision set was whether or not to provide recreational boating flows on weekends in October, a traditionally popular time to float the river. The combination of decisions is equivalent to 32 possible decision portfolios in the Bayesian Belief Network.

Uncertainty Nodes

Relations among the uncertainty nodes (yellow rectangles; fig. A3) were modeled using probabilistic dependencies derived from empirical data and expert opinion. For each uncertainty node, a conditional probability table (CPT; table A3) was populated with probabilities of causal links among associated nodes. CPTs were populated using both expert opinion and available empirical data (Marcot and others, 2006) In practice, each causal link to a node represents a column in a CPT and probabilities associated with the causal links must be entered (table A3). For example, the node "small fish abundance" has four causal links: "shallow-fast habitat," "degree days," "flow through pools," and the decision to provide "flow windows" (table A3). The probability of having high, moderate, or low small fish abundance was estimated for each of the 108 potential scenarios in this part of the BBN model and is represented on the associated CPT (table A3). For the "small fish abundance" node in our BBN, we had empirical data for two decisions (status quo and summer flow windows; Freeman and others, 2001). The other probabilities were estimated by expert opinion; for example, what is the probability of having low small fish abundance when conditions provide high amounts of shallow-fast habitat, high numbers of degree days, spring flow windows, and high flow through pools? As new data became available (for example, data associated with a new decision) from an associated monitoring program, CPTs were updated with new information that replaced the expert opinion. The nodes are described in detail below and the source and range for each are presented in table A4.

Reservoir inflow.—The input for reservoir inflow was based on the 10-percent, 25-percent, 75-percent, and 90-percent exceedance flows for the combined Heflin (main stem Tallapoosa River upstream from Harris Reservoir) and Newell (USGS streamgage 02413300, Little Tallapoosa River near Newell, Ala.) streamgages for the period of record. The main stem Tallapoosa River and the Little Tallapoosa River are the two primary sources of inflow into Harris Reservoir. Using the period of record, we assigned conditional probabilities to each inflow condition as follows: flood, 0.10; wet, 0.15; normal, 0.50; dry, 0.15; drought, 0.10. Thus, flood conditions were equated to flows with greater than (>) 48.1 m^3/s, wet conditions were flows between 42.5 and 48.1 m^3/s, normal conditions were flows between 28.3 and 42.5 m^3/s, dry conditions were flows between 17.0 and 28.3 m^3/s, and drought conditions had flows less than (<) 17.0 m^3/s.

Table A3. Conditional probability table (CPT) for small fish abundance. The top 42 of 108 possible scenarios related to the decision to provide flow windows, availability of shallow-fast habitat, number of degree days (temperature component), and the amount of flow through pool habitat are shown. Probabilities were derived from data collected by prepositioned area electrofishing grids, by physical habitat simulation models, and from expert opinion.

[Nodes are listed as included in the Netica software CPT; bins of high, moderate, and low for each node are described in table A4]

Shallow-fast habitat	Degree days (temperature)[1]	Decision—Flow windows?	Flow through pools	Small fish abundance		
				High	Moderate	Low
High	High	Status quo	High	0.3	0.5	0.2
High	High	Status quo	Medium	0.3	0.5	0.2
High	High	Status quo	Low	0.3	0.5	0.2
High	High	Spring and summer	High	1.0	0.0	0.0
High	High	Spring and summer	Medium	1.0	0.0	0.0
High	High	Spring and summer	Low	1.0	0.0	0.0
High	High	Spring	High	0.6	0.3	0.1
High	High	Spring	Medium	0.6	0.3	0.1
High	High	Spring	Low	0.6	0.3	0.1
High	High	Summer	High	0.6	0.3	0.1
High	High	Summer	Medium	0.6	0.3	0.1
High	High	Summer	Low	0.6	0.3	0.1
High	Moderate	Status quo	High	0.2	0.5	0.3
High	Moderate	Status quo	Medium	0.2	0.5	0.3
High	Moderate	Status quo	Low	0.2	0.5	0.3
High	Moderate	Spring and summer	High	0.9	0.1	0.0
High	Moderate	Spring and summer	Medium	0.9	0.1	0.0
High	Moderate	Spring and summer	Low	0.9	0.1	0.0
High	Moderate	Spring	High	0.4	0.4	0.2
High	Moderate	Spring	Medium	0.4	0.4	0.2
High	Moderate	Spring	Low	0.4	0.4	0.2
High	Moderate	Summer	High	0.4	0.4	0.2
High	Moderate	Summer	Medium	0.4	0.4	0.2
High	Moderate	Summer	Low	0.4	0.4	0.2
High	Low	Status quo	High	0.0	0.2	0.8
High	Low	Status quo	Medium	0.0	0.2	0.8
High	Low	Status quo	Low	0.0	0.2	0.8
High	Low	Spring and summer	High	0.6	0.2	0.2
High	Low	Spring and summer	Medium	0.6	0.2	0.2
High	Low	Spring and summer	Low	0.6	0.2	0.2
High	Low	Spring	High	0.2	0.6	0.2
High	Low	Spring	Medium	0.2	0.6	0.2
High	Low	Spring	Low	0.2	0.6	0.2
High	Low	Summer	High	0.2	0.6	0.2
High	Low	Summer	Medium	0.2	0.6	0.2
High	Low	Summer	Low	0.2	0.6	0.2
Moderate	High	Status quo	High	0.2	0.5	0.3
Moderate	High	Status quo	Medium	0.2	0.5	0.3

Table A3. Conditional probability table (CPT) for small fish abundance. The top 42 of 108 possible scenarios related to the decision to provide flow windows, availability of shallow-fast habitat, number of degree days (temperature component), and the amount of flow through pool habitat are shown. Probabilities were derived from data collected by prepositioned area electrofishing grids, by physical habitat simulation models, and from expert opinion.—Continued

[Nodes are listed as included in the Netica software CPT; bins of high, moderate, and low for each node are described in table A4]

Shallow-fast habitat	Degree days (temperature)[1]	Decision—Flow windows?	Flow through pools	Small fish abundance		
				High	Moderate	Low
Moderate	High	Status quo	Low	0.2	0.5	0.3
Moderate	High	Spring and summer	High	0.9	0.1	0
Moderate	High	Spring and summer	Medium	0.9	0.1	0
Moderate	High	Spring and summer	Low	0.9	0.1	0
Moderate	High	Spring	High	0.4	0.4	0.2

[1]Degree days are described in the text.

Table A4. Description of state variables, data sources, and ranges of values for the initial model parameterization (see fig. A3).

[Erosion* is an uninformed node based on lack of data and resources to collect data; reservoir inflows** have five response levels versus three; m³/s, cubic meter per second; USGS, U.S. Geological Survey; >, greater than; d/yr, day per year; <, less than; APC, Alabama Power Company; --, no data; PHABSIM, physical habitat simulation (model); °C, degrees Celsius; %, percent]

State variable	Brief description; source	Range		
Boatable days	Number of consecutive weekend days of discharge between 12.7 and 56.6 m^3/s; USGS streamgage data	High >70 d/yr	Medium 40–70 d/yr	Low <40 d/yr
Erosion*	No data/uninformed node	High	Moderate	Low
Lake levels	Number of days/year that lake levels fall below rule curve; APC	High <10 day	Moderate 11–20 day	Low >21 day
Reservoir inflows**	Exceedance flows (m^3/s) for reservoir tributaries combined; USGS data	Flood >48.1 m^3/s Wet 42.5–48.1 m^3/s	Normal 28.3–42.5 m^3/s -- --	Dry 17.0–28.3 m^3/s Drought >17.0 m^3/s
Flow through pools	Pool habitat percent with flow >20 m^3/s; expressed for different inflows using PHABSIM model	High Normal year >50%	Moderate Normal year 20–50%	Low Normal year <20%
Shallow-fast amounts	Shallow (<45 m^3/s)-fast (>45 m^3/s) habitat percent; expressed for different inflows using PHABSIM model	High Normal year 60–100%	Moderate Normal year 20–60%	Low Normal year <20%
Slow-cover amounts	Slow (<20 m^3/s)-cover (present) percent; expressed for different inflows using PHABSIM model	High Normal year 50–100%	Moderate Normal year 10–50%	Low Normal year <10%
Degree days	Number of 10-day periods where cumulative degree days exceeded 63 at 17.2 °C threshold; USGS data	High >120	Moderate 100–119	Low <99
Small fish abundance	Ratio of juvenile fish/100 samples×100 in regulated compared to unregulated sites; USGS data	High >50	Moderate 20–50	Low <20
Bass recruitment	Number of juvenile bass in 100 samples; USGS data	High >20	Moderate 20–10	Low <10
Redbreast sunfish spawning	Number of juvenile redbreast sunfish in 100 samples; USGS data	High >60	Moderate 30–60	Low <30

Lake levels.—These probabilities were derived from lake level data provided by APC and were tied to the number of days in a year that lake levels fell below the FERC rule curve. High lake levels were years that the lake fell below the rule curve less than 10 days, moderate lake levels were years when lake levels fell below the curve 11–20 days and low lake levels were greater than 21 days below the rule curve. For the period of record from 1983 to 2001, lake levels were high, moderate, and low for 57, 16.5, and 26.5 percent of the time, respectively. This node was dependent on reservoir inflow.

Boatable days.—Boatable days were based on the number of consecutive weekend days per year when flow was between 12.7 and 56.6 m^3/s at the Wadley streamgage for the period of record through September 1974. Weekends were considered Saturday and Sunday but also included Columbus Day, Memorial Day, the Friday or Monday closest to July 4 (if on a Wednesday, the day within the flow bounds was chosen; if both Friday and Monday were within the flow bounds, Monday was chosen), and Labor Day. A high number of days was >70 days/year, a medium number of days was 40–70 days/year, and a low number of days was <40 days/year. For the period of record before the dam, 80 percent of years had between 40 and 70 boatable days per year, 10 percent of years had more than 70 boatable days per year, and 10 percent of years had less than 40 boatable days per year.

Erosion.—The erosion node was parameterized with three levels: high, moderate, and low; however, because we had no data on erosion, we gave equal weight to the levels (all 33.3 percent). This parameter was important to stakeholders, and probabilities will be updated pending the collection of additional data.

Shallow-fast habitat.—This node was directly dependent upon reservoir inflow. The probabilities we used to link these variables were based upon both the flow record since the dam was built and physical habitat simulation (PHABSIM) models developed by the USGS (Bowen and others, 1998). PHABSIM models were constructed at the Wadley site and depth/flow measurements were recorded along transects at high and low flows. Therefore, we were able to estimate the percent of shallow-fast habitat (depth <45 centimeters [cm]; flow >45 centimeters per second [cm/s]) in the channel during a flood, wet, normal, dry, or drought year; we estimated habitat for April–July based on importance of habitat during spawning periods. For example, with all other variables unknown, in a normal year, there was a 20-percent chance of having high amounts (60–100 percent maximum habitat) of shallow-fast habitat, a 70-percent chance of having moderate amounts (20–60 percent maximum), and a 10-percent chance of having low amounts (<20 percent maximum).

Slow-cover habitat.—Again, this node was dependent on reservoir inflows and was calculated based on PHABSIM models. Percent slow-cover habitat (flow <20 cm/s; cover present) was estimated in the channel for flood, wet, normal, dry, and drought years, during April–July. For example, with all other variables unknown, in a normal year, there was a 30-percent chance of having high amounts of shallow-fast habitat (50–100 percent maximum habitat), a 60-percent chance of having moderate amounts (10–50 percent maximum), and a 10-percent chance of having low amounts (<10 percent maximum).

Flow through pools.—This variable refers to maintenance of flowing water through the deep portions of the river channel, which can become nearly stagnant during prolonged nongeneration periods at the dam. We hypothesized that maintaining flow through pools would enhance fish abundances by increasing production by current-dependent macroinvertebrates (such as filter-feeding insects). We used PHABSIM to estimate flow in pool habitats during flood, wet, normal, dry, and drought years. As with the other habitat variables, flow through pools was dependent on reservoir inflow. The flow through pools variable was considered to be those pool habitats that had flows >20 cm/s. With all other variables unknown, in a normal year, there was a 15-percent chance of having a high proportion of flow through pools (>50 percent of all pool habitat), a 60-percent chance of having a moderate proportion (20–50 percent of pool habitat), and a 25-percent chance of having a low proportion of flow through pools (<20 percent of pool habitat).

Degree days.—This node was included because thermal effects were thought to be important for reproduction and development of certain faunal groups. Conditional probabilities in this CPT were based on data from Andress (2002). We parameterized this node based on the number of 10-day periods where cumulative degree days were 63 or greater. A degree day was calculated as:

$$\text{Degree days} = [(\text{maximum daily temperature} + \text{minimum daily temperature})/2] - \text{lower threshold temperature} \quad (1)$$

The lower threshold temperature used was 17 °C based on bluegill development data (Nakamura and others, 1971). Cumulative degree days (daily degree days added for each day from eggs to swim-up stage of fry) represent the amount of heat energy necessary for redbreast sunfish development to swim-up fry. Cumulative degree days must be around 63 for redbreast sunfish development from egg to swim-up stage of fry.

Small fish abundance.—We used our long-term (6 years) prepositioned electrofishing (grid) data to parameterize this node. This node was directly linked to shallow-fast habitat, degree days, and flow through pools (that is, production of food). We considered abundance to be high if >50 individuals were captured in 100 grids. Moderate abundance was 20–50 individuals captured in grids, and low abundance was <20 individuals captured in grids. With all other variables unknown, in a normal year, the model predicted a 31.5-percent chance of having a high abundance of small fish, a 30.8-percent chance of having a moderate abundance of small fish, and a 37.7-percent chance of having a low abundance of small fish.

Black bass recruitment.—We used backpack electrofishing data and expert opinion to parameterize this node, which was linked to slow-cover habitat, degree days, and flow through pools. High bass recruitment was equal to more than 20 juveniles collected in a sample, medium bass recruitment was 10–19 juveniles, and low bass recruitment equaled <10 juveniles. With all other variables unknown, in a normal year, the model predicted a 39.1-percent chance of high recruitment, a 26.2-percent chance of medium levels of recruitment, and a 34.7-percent chance of low recruitment.

Redbreast sunfish spawning success.—Similar to black bass recruitment, we used backpack electrofishing data and expert opinion to parameterize this node, which was linked to slow-cover habitat, degree days, and flow through pools. High success was equal to more than 60 juveniles collected in a sample, medium levels of success were 30–60 juveniles, and low levels equaled <30 juveniles. With all other variables unknown, in a normal year, the model predicted a 38.9-percent chance of high redbreast sunfish spawning success, a 26.5-percent chance of moderate success, and a 34.6-percent chance of low success.

Utility Nodes

Following the example of Peterson and Evans (2003), the utility (or consequence) nodes were representative of the satisfaction of stakeholders involved in the decision. In this way, the model remains flexible as knowledge is gained and updated to determine stakeholder values related to objectives. In addition, stakeholder satisfaction provides a nonmonetary, equitable currency for optimizing the resource. We narrowed stakeholder satisfaction into five categories: river boater satisfaction, river landowner satisfaction, reservoir user satisfaction, fish population value, and power generation. The river boater, river landowner, and reservoir user satisfaction values were based upon feedback from individual stakeholders and ranged from 0 to 100 percent. River boaters were most satisfied with high numbers of boatable days (boaters were 100 percent satisfied with number of days similar to predam conditions), river landowners with low rates of lateral bank erosion (100 percent), and reservoir users with lake levels at or above the established rule curve (100 percent). Each was less satisfied with the opposite and lowest result (0 percent). Fish population value (multiple stakeholders are represented here) increased with high incidences of small fish abundance, bass recruitment, and redbreast sunfish spawning success (100 percent satisfied) and decreased with low values (0 percent satisfied) of these influencing variables. The values incorporated for power generation were estimates of flexibility provided by the power utility; values ranged from 100 percent satisfied (status quo) to 83.5 percent (flow mimics Heflin streamgage but does not drop below 8.5 m^3/s at the dam).

Sensitivity Analysis

The final step before deciding on a starting flow regime was to conduct a sensitivity analysis on the decision model. The sensitivity analysis examined the effect of model components on each utility value and, therefore, on the modeled decision(s). We used methods outlined in Clemen (1997) and used one-way sensitivity analyses and a one-way response profile sensitivity analysis. One-way sensitivity analyses was used to identify the components that have the greatest effect on the utility, and the one-way response profile sensitivity analysis was used to identify the components that had the greatest effect on the optimal decision. To address the influence of utility values on the decision, we developed an indifference curve by varying the weight of the power generation utility from 0.1 to 1 and plotted the results. See Conroy and Peterson (2013) for detailed methods.

Monitoring

The monitoring program was designed to collect data relative to predicted (or hypothesized) attainment of stakeholder objectives. Incorporation of new information collected over time was intended to reduce uncertainty with respect to the magnitude and direction of responses of objectives to management. A technical advisory group comprised of agency and power company biologists was appointed by the stakeholder board to determine the spatial and temporal array for data collection. Once it was developed, the monitoring plan was reviewed by several USGS scientists to ensure that the design was sound. Data regarding reservoir inflows and lake levels, number of boatable days, and provision of spawning conditions were calculated each year based on the analysis of the hydrology data provided by the USGS streamgages or collected by the power company as part of their FERC license requirement. Data on all state variables were collected each year for model updating; sources, nature, and range for data included in the initial model and the monitoring program are listed in table A4.

Biological sampling.—The response of the biological parameters was evaluated by design and implementation of monitoring to quantify variation in fish count data (ultimately, occupancy; see also chapter B) in relation to system covariates. The response of fish habitat variables was evaluated by seasonal application of the post management hydrograph to a PHABSIM model (Bowen and others, 1998) developed at two of the sampling sites. Sampling was executed twice yearly in spring/summer (May–August) and fall (September–November) from 2005 to 2016. We randomly selected five shoals per sampling reach; the most downstream reach below R.L. Harris Dam was sampled only once per year and usually in the fall (fig. A1). We used prepositioned area electrofishers (Bowen and others, 1998; Freeman and others, 2001) to sample fishes from 20 to 22 shoals per season. In 2005–7, we collected 20 prepositioned area electrofisher samples per shoal, and in 2008–16, we collected 10 prepositioned area electrofisher samples per shoal. The reduced effort was warranted based on an analysis of the data from the previous year and allowed us to spend more effort on field identification (see below).

Fishes.—In 2005–7, fishes were euthanized in MS–222, preserved in 10-percent formalin, and returned to the laboratory for processing. Larger specimens were field identified and released after total length (in millimeters) data were recorded. Fishes were transferred to 70-percent ethanol for long-term storage. In the laboratory, specimens were identified to species and total length (in millimeters) data were recorded. In 2008–16, specimens were field identified, measured, and released; small individuals that could not be identified without a microscope were preserved and returned to the laboratory as described above.

Model updates.—Bayesian updating of probability distributions was completed yearly from 2005 to 2016 in Netica to learn how the management regime affected stakeholder objectives. The stakeholders were apprised of the results periodically and formally through board meetings and through other methods such as publications, presentations, and individual stakeholder briefings. Our initial decision model was based on the best available information. Because data are finite, and no single model can faithfully represent full truth (Burnham and Anderson, 2002), we wanted to be able to track the reliability of the initial decision model through time using monitoring data. Thus, we added an additional node, model truth, to the decision model representing the reliability of the model to predict the changes in small fish abundance, bass recruitment, and redbreast sunfish spawning success. The node contained two states, yes and no. For the yes node, the predicted changes in fish abundance, bass recruitment, and redbreast sunfish spawning success were based on the monitoring data, whereas the predictions under the no state were uniform distributions indicating that the changes in the three fish nodes in response to water management were unknown. The initial probability of the yes node was set at 0.99 and the no node at 0.01.

The relative reliability of the decision model was evaluated by comparing the model predictions under the yes and no states of the model truth node to the actual outcomes (that is, monitoring data) and calculating the likelihood of the observed outcome under each state. The posterior probabilities of yes and no states were then calculated using the prior node probabilities (that is, yes=0.99 and no=0.01 for the first time step) and the likelihood of the observed state under each alternative was calculated using Bayes theorem. The node updating process was repeated for each year of monitoring data using the posterior probabilities for model truth in t−1 as the prior probabilities for year t. (see appendix A2).

Results

Once the decision elements, state variables, and utility functions were related in the network and the CPTs were populated, we used the Netica software to find the optimal decision. In addition, we conducted sensitivity analysis and over time we conducted model updating to estimate new probabilities by incorporating field monitoring data.

Modeling the Optimal Decision

The optimal decision (fig. A3) was determined by examining the expected value associated with each alternative decision, which was the sum of the probability-weighted utility values.

$$U_{total} = \Sigma_{j=1}^{5} \Sigma_{i=1}^{n_j} p_{ij} u_{ij} \tag{2}$$

where

- n is the number of states for node j,
- p is the probability of state i for node j, and
- u is the value of state i for node j.

The modeled decision with the highest value was considered the optimal decision; therefore, to provide flows as measured at the USGS Wadley streamgage (22 km below R.L. Harris Dam) that matched flows at the USGS Heflin streamgage (located above Harris Reservoir), via pulsing operations supported by peaking generation, and to supply spring and summer spawning windows and October recreational boating flows (see fig. A3).

Sensitivity Analysis

One-way sensitivity analysis of the impact of uncertainty nodes on the overall utility values indicated that erosion and boatable day nodes were the most influential components of the model and the habitat nodes were the least influential (fig. A4). The power generation utility values ranged from 83.5- to 100-percent satisfaction for the four flow decisions (see “Utility Nodes” section above) and indifference curves indicated that the optimal decision was unaffected by the relative value of the utility states (fig. A5). One-way response profile sensitivity analysis indicated that the erosion node illustrates stochastic dominance where the optimal decision does not change across the range of node states, and the reservoir inflow node illustrates an instance where the optimal decision changes four times across the range of node states (fig. A6; table A5).

The fish population value had several influencing variables in the model. One-way sensitivity analysis revealed sensitivity to the number of degree days (fig. A7). Similarly, the

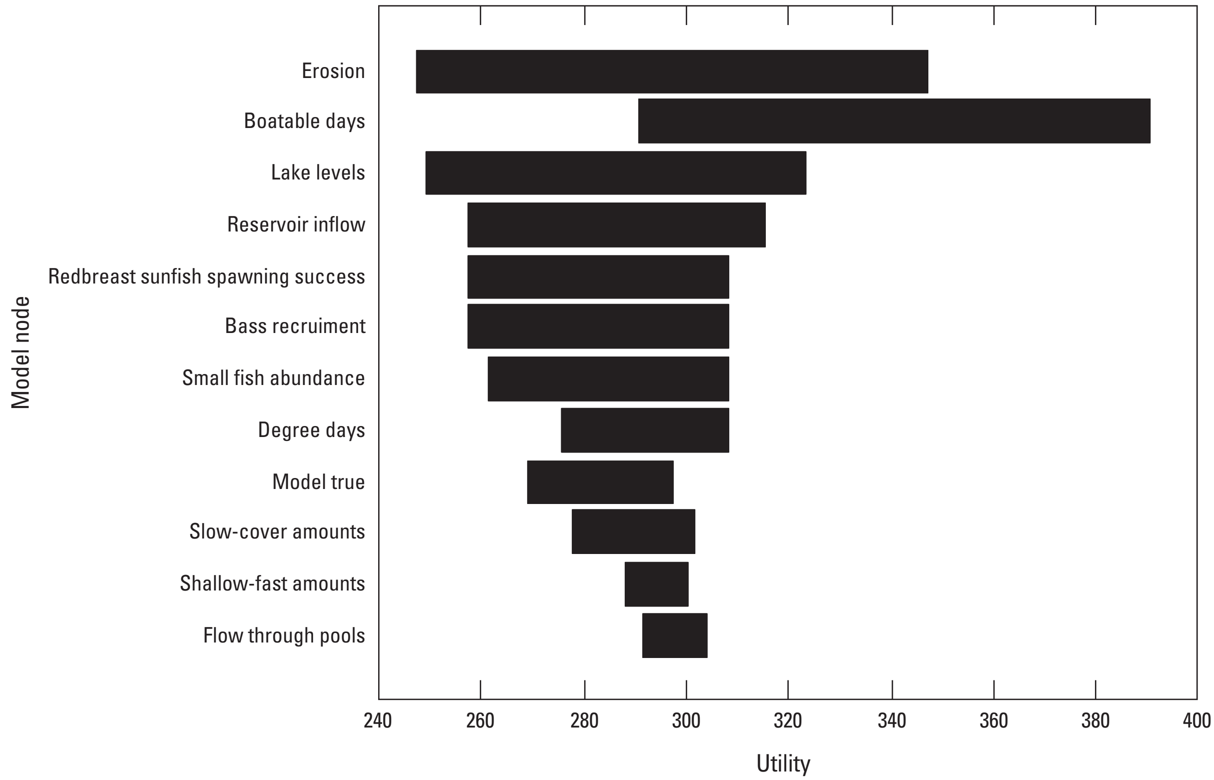

Figure A4. Tornado diagram from one-way sensitivity analysis of R.L. Harris Dam decision model. Each horizontal bar represents the range of utilities for optimal decision across a range of node states.

fish population value was sensitive to reservoir inflow, which is likely an influencing variable on degree days.

Model Updates

Bayesian updating of applied decisions and results of monitoring were incorporated into the decision and uncertainty nodes in Netica (table A6; appendix A2). The implemented decision each year was usually the same; APC (Green Plan) pulsed flows from the dam with added 10-day spring and summer periods for spawning where discharge was no greater than 1 unit from the dam (about 198.2 m^3/s). On three occasions (spring and summer 2005 and spring 2009), spawning periods of reduced generation were not provided. The third decision (provision of October boating flows) was never implemented. We incorporated cases of yearly results provided from APC (decision) and field collected and USGS streamgage data to update probabilities associated with the causal links in the Bayes network. See Conroy and Peterson (2013) for a description of the Bayesian paradigm that allowed us to use the rules of probability to make inference regarding the effects of management on objectives.

Overall, the highest utility scores for the best decision portfolios were associated with either the APC or Heflin flow regimes from the dam, along with spring and summer spawning periods and usually October boating flows (appendix A2). In general, utility scores did not change over time; however, probability distributions changed somewhat. The reservoir inflow distribution became more skewed toward drier conditions; however, lake levels had almost equal odds of being either high or low. The probabilities of small fish abundance, black bass recruitment, and redbreast sunfish spawning success converged to be nearly equal after probabilities were updated with 2008 field data.

Discussion

Use of adaptive management for solving resource dilemmas in multiple-use systems has become a goal for many agencies (Williams and Brown, 2011). However, procedures for implementation of the process are lacking; this project provided a template for adopting adaptive management for learning how multiple state variables representing stakeholder objectives in a regulated river system responded to manipulation of flow regimes. Historically, flow decisions have been made based on negotiations that often take years (for example, FERC relicensing), and although they are based on expert knowledge and data, one-time fixed flow regimes are often the net result. This approach does not allow for adjustment of management relative to system response and gained

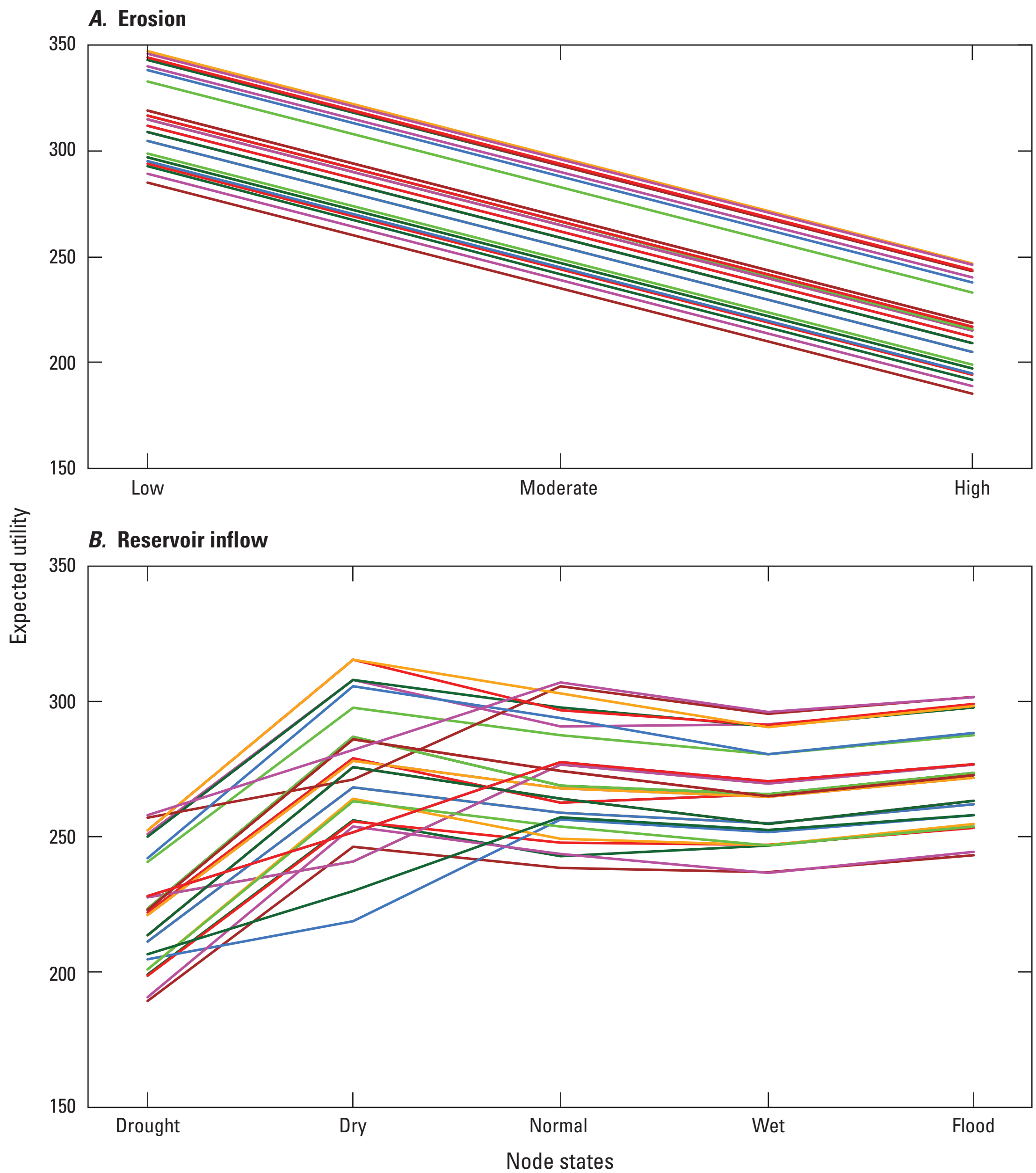

Figure A5. Example plots from one-way response profile sensitivity analysis with each line representing a decision alternative. The erosion node illustrates stochastic dominance where the optimal decision (top line) does not change across the range of node states, and the reservoir inflow node illustrates instances where the optimal decision changes four times across the range of node states.

knowledge, nor comparisons with a priori hypotheses (Irwin and Freeman, 2002). The development of an adaptive management template during this project was successful to the point that, based on the model developed by stakeholders, a decision was made to adjust flow at R.L. Harris Dam; adaptive management began in March 2005. The results of the decision model updates indicate that uncertainty regarding the fish population values has not been resolved. In addition, the decision portfolio that included the Green Plan (that is, APC pulsing from the dam) was not the decision portfolio that had the highest utility over time. Continuing adaptive management in tandem during the FERC relicensing process would be advantageous to include a specific assessment of long-term objectives of all stakeholders.

Failure to successfully implement adaptive management in natural systems is often caused by lack of stakeholder involvement (Johnson, 1999). Since 2000, we have attended dozens of meetings to engage stakeholders and discuss

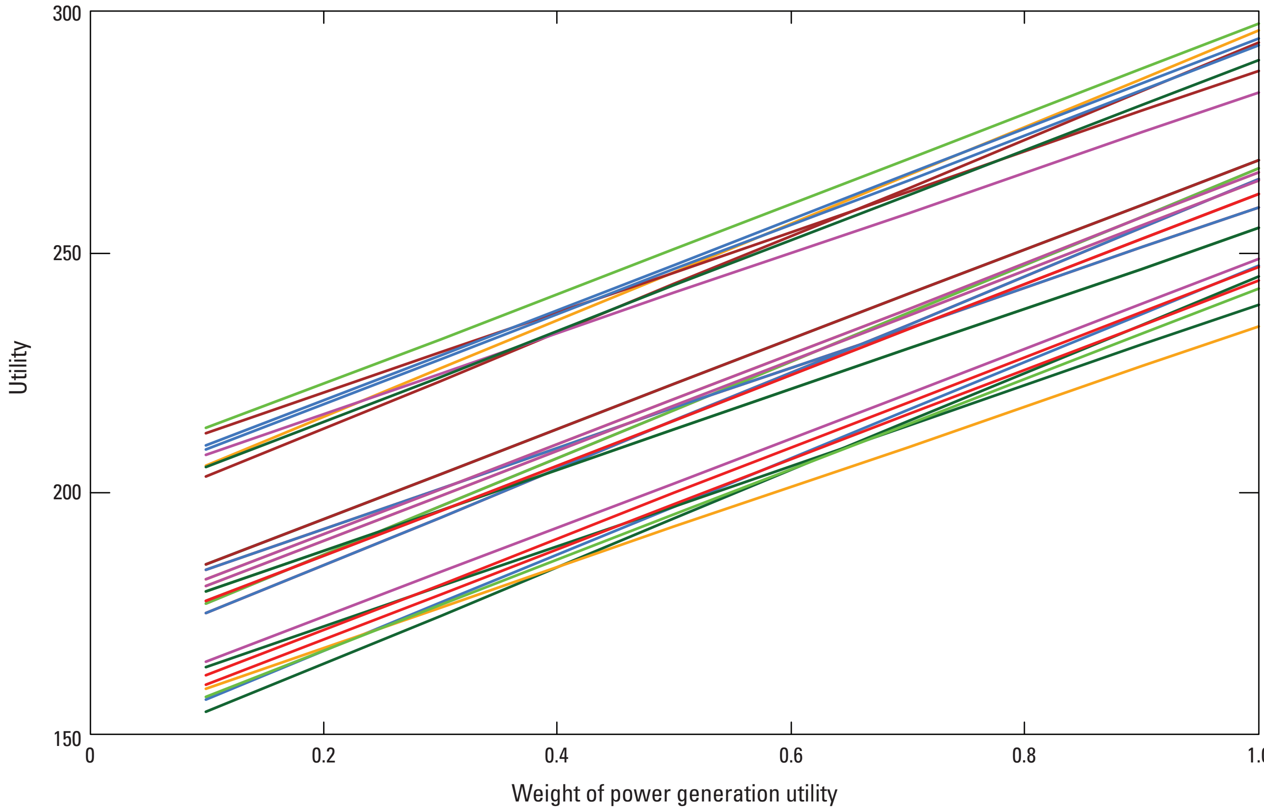

Figure A6. Indifference curves for power generation utility. The optimal decision (top line) does not change across the range of weights.

Table A5. Summary of one-way response profile sensitivity analysis with number of optimal decisions across the range of state values, the decisions where the utilities of alternatives were equal (indifferent), and the decisions that do not change across range of node states (stochastically dominant).

[--, no data]

Node	Optimal decisions	Indifferent decisions	Stochastically dominant decisions
Bass recruitment	2	--	October flow, yes; spawning window, spring and summer.
Boatable days	2	October flows	Flow, status quo; spawning window, spring and summer.
Degree days	2	--	October flow, yes; spawning window, spring and summer.
Erosion	1	--	October flow, yes; flow, Heflin; spawning window, spring and summer.
Flow through pools	2	--	October flow, yes; spawning window, spring and summer.
Lake levels	2	--	October flow, yes; spawning window, spring and summer.
Model true	2	Windows of opportunity	October flow, yes.
Redbreast sunfish spawning success	2	--	October flow, yes; spawning window, spring and summer.
Reservoir inflow	3	October flows	Spawning window, spring and summer.
Shallow-fast amounts	2	--	October flow, yes; spawning window, spring and summer.
Slow-cover amounts	2	--	October flow, yes; spawning window, spring and summer.
Small fish abundance	2	--	October flow, yes; spawning window, spring and summer.

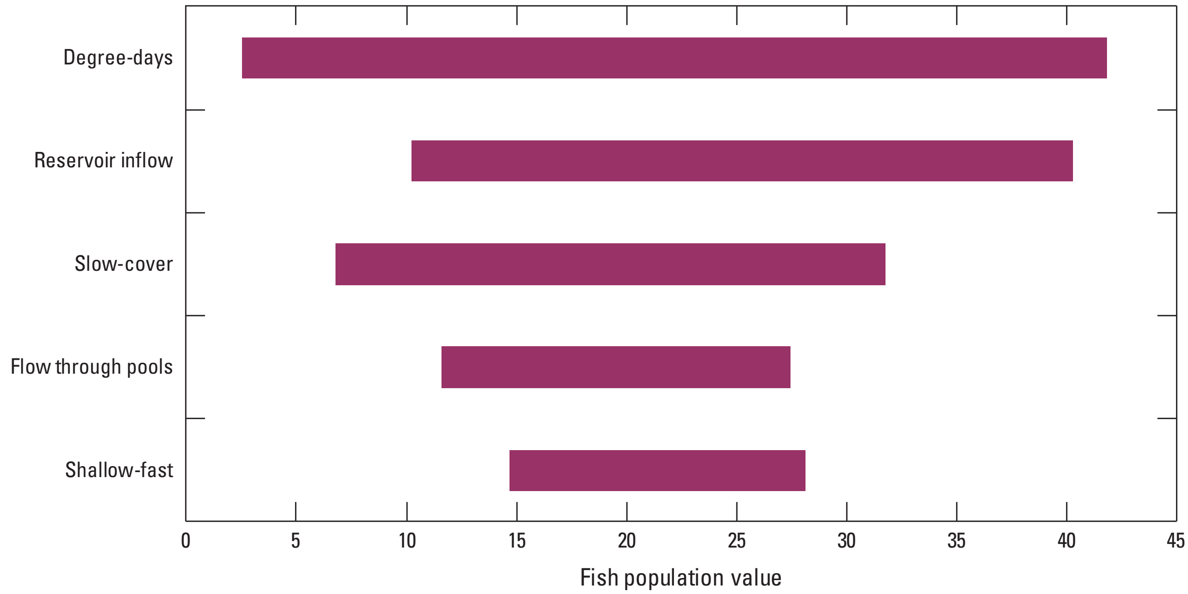

Figure A7. Tornado diagram of one-way sensitivity of fish population value to influencing variables.

objectives and implementation of the adaptive management process. A workshop for stakeholders in the Tallapoosa River system catalyzed participation by providing a formal forum that prompted the development of a governance structure (that is, The R.L. Harris Stakeholders Board; hereafter, “the Board”) and set a standard for future stakeholder involvement (see http://rivermanagement.org/). The stakeholders determined and agreed that the Board would consist of one voting member of each of the stakeholder groups. The Board adopted a charter (appendix A3) that defined purposes and principles, membership, rules of engagement, and decision-making processes. Examples of specifics outlined in the charter were a long-term commitment (5–7 years) to the process; election of a board chair, if needed; and determination that professional facilitators would guide meetings of the Board. One extraordinary aspect of the charter establishes that “progress will not regress do to the entry of new Members.” New stakeholders were expected to familiarize themselves with the process to date and contribute to the discussion from their point of entry. All previous minutes and presentations were posted on the web page. The process defined in the charter has been strictly adhered to primarily because of the professional facilitators. Over the course of the study, the Board was polled to determine if a change in flow management was desired. No changes were made based on apparent satisfaction with the process.

This project provided the framework for incorporating stakeholders’ objectives and values into decisions regarding flow modification in regulated river systems. During elicitation, stakeholders commented on the presence of commonalities in their objectives and values. In addition, objectives that were of value to individual stakeholders were included in the model and were transparent (that is, nodes were specific and visible), including objectives for which parameterization was not readily possible (for example, erosion). Stakeholders seemingly grasped the concept that information gained through monitoring would ultimately improve the effectiveness of the model for representation of system processes. The modeling process that we used was easy for the stakeholders to understand based on the visual nature of the Netica software. These characteristics of the process were important during stakeholder meetings where new knowledge was incorporated, thereby reducing system uncertainty. In the future, a reevaluation of stakeholder objectives will be important because stakeholder perspectives have likely evolved based on learning and (or) changes in institutional structure (Williams, 2011).

Freeman and others (2001, p. 189) called for flow manipulations in an adaptive management context, coupled with continued biological monitoring to “elucidate how hydrologic variation affects species persistence.” Although alterations in flow regimes affect fish populations and communities, functional relations among flow parameters (for example, frequency, duration, magnitude) and fish populations are not well defined. Key uncertainties relative to how dam operations affect aquatic communities still need to be resolved. Our hypotheses revolved around the following flow features: base flow (Travnichek and others, 1995), periods of stable flow (Freeman and others, 2001), and thermal regime (Andress, 2002). The governance structure that was developed by the stakeholders allowed for Technical Advisory Groups (appendix A3). The Board appointed a Science Technical Advisory Group and tasked them with developing a monitoring plan for the adaptive management project. These groups served in advisory capacity and reported to the Board so that appropriate actions were taken relative to decisions. The Science Technical Advisory Group developed a monitoring plan based on many of the uncertainty nodes in the decision support model and implemented it in spring 2005. Decisions regarding what

Table A6. Data used to conduct yearly model updates. See table A3 for a description of parameter calculations.

Parameter	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016
Lake levels bin	Low	Low	Low	High	High	High	High	Low	Moderate	Low	Low	Low
Inflows	Normal	Dry	Drought	Drought	Dry	Wet	Dry	Dry	Normal	Normal	Normal	Normal
Spawning window, spring	No	Yes	Yes	Yes	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Spawning window, summer	No	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Boatable days bin	Moderate	Low	Low	Low	Low	Low	Low	Low	Low	Low	Low	Low
Degree days bin	Low	Low	Moderate	Moderate	Low	Moderate	High	High	Low	Moderate	Moderate	High
Small fish	Low	Low	Low	Low	Low	Low	Moderate	Low	Low	Low	Low	Low
Bass recruitment	Low	Low	Moderate	Moderate	Low	Low	Low	Moderate	Low	Low	Low	Low
Redbreast sunfish recruitment	Low	Low	Low	Low	Low	Low	Moderate	Low	Low	Low	Low	Low

parameters to monitor were made dependent on (1) relevance to stakeholders' objectives and (2) ability of the scientists to collect adequate (that is, meaningful) and reliable (for example, bias minimized) data to use in updating model probabilities. Results of the monitoring and subsequent model updates indicated that the decision model updates (with one exception) appeared to reflect stakeholder values. The exception was the predicted responses of fish populations to flow management where the model predicted high fish abundances; whereas, when monitoring data were compiled, the fish population responses were overwhelmingly low (abundance and recruitment).

Specific analysis of colonization and persistence parameters (that is, in a metapopulation framework; Royle and Kéry, 2007) of fishes (see chapter B) indicated that thermal alteration and generation frequency negatively affected the occupancy of most fish species below the dam. Although degree days were included in the decision model, specific hydrology related to generation frequency was not. In addition, habitat conditions were predicted to be favorable for positive fish population responses under the Green Plan. Habitat availability for fishes increased under the Green Plan management (Irwin and others, 2011), but the improved conditions did not improve recruitment processes for species of interest. Continued monitoring to evaluate the effects of the flow and thermal regimes on aquatic communities is needed, and inclusion of other habitat parameters (for example, persistence) is potentially needed.

We used sensitivity analysis to assist the managers (that is, water and natural resource) relative to allocation of resources for monitoring because the analysis identified what variables affected stakeholder satisfaction. For example, under flow conditions that match the Heflin streamgage target, managers can expect the satisfaction of reservoir users to remain constant or even decline during a normal water year and to change consistent with changes in reservoir inflow. On the part of river recreationists, managers must not expect boaters to be satisfied to the degree they would have been under predam conditions, but they will likely experience positive feedback from this stakeholder group under the alternate flow conditions. Over the course of the study, boater satisfaction remained low; the provision of boating flows may be a management strategy for the future. Sensitivity analysis was also beneficial for fisheries management in that, given the empirical data, periods of stable flows appeared to be most beneficial for the integrity of the fish populations; however, the spawning windows that were provided during the study did not seem to improve recruitment processes. In addition, the habitat data that were included in the model were habitat availability data. Other work has indicated that habitat persistence may be a better predictor of fish population parameters in the Tallapoosa River (Freeman and others, 2001). Consequently, reexamination of the linkages among the model nodes and the specific data used to parameterize the conditional relations is warranted. In addition, the associated monitoring program collected data on the macroinvertebrate community structure at sites below the dam. Adding aspects of macroinvertebrate ecology into the decision framework may inform the development of different flow prescriptions or impact decisions already developed and analyzed. We hypothesized that enhancing flow through pools would benefit reproduction by unionid mussels (for example, *Hamiota altilis*); this aspect of mussel ecology could be defined and assessed with the existing decision model.

Although adaptive management was successfully implemented in the Tallapoosa River, other systems may have impediments preventing the process. One of the things that became apparent in the decision analysis process was inflexibility in the flow delivery mechanism. Specifically, the dam was not engineered to release finite amounts of water continuously, nor is there a mechanism to change water temperature. Consequently, delivery of base flow in the system is provided by pulsing flows from the turbines. When applying this template to other systems, decision elements regarding flow from dams will have to incorporate facility-specific engineering constraints. Because the flow management option did not result in the attainment of all stakeholder objectives, these constraints will need to be fully vetted to determine the next portfolio of adaptive management flows. In addition, the lack of delivery options and fine-scale control of flow and temperature regimes limited the experimental degree of the project and allowed for passive adaptive management only. Passive adaptive management focuses on achieving management objectives, and reducing uncertainty, and, therefore, learning is a byproduct of the approach (Williams and others, 2007). In the future, more emphasis on active adaptive management utilizing experimental learning could reduce uncertainty related to impacts of flow and thermal regimes on the natural resources in this system.

Overall, the development of the template for applying adaptive management and decision support was successful in the case of the Tallapoosa River, thus far. Key elements for success were (1) use of a professional and neutral facilitator to engage stakeholders in objective and value identification (see Belton and Jackson-Smith, 2010), (2) use of a visual decision support model that allowed for stakeholder input and optimization of values associated with various decisions, (3) development of a governance structure for future involvement and ownership in the process, and (4) recognition of a long-term commitment to learning the effects of management through system monitoring and adjustment of management regimes. The monitoring program associated with the project was labor intensive, both in terms of field and laboratory days, and is therefore costly. However, without the monitoring data, the stakeholders would not be informed regarding the lack of attainment of natural resource objectives. Cash and others (2003) describe processes to effectively manage how knowledge can be linked to action for sustainability. They state that institutional mechanisms (such as stakeholder groups) can improve communication, translation, and mediation toward understanding the science and technology that lend salience, credibility, and legitimacy to action. Future success during the FERC process will depend upon these elements and continued stakeholder involvement and support.

References Cited

Andress, R.O., 2002, Nest survival of *Lepomis* species in regulated and unregulated rivers: Auburn University, Auburn, Ala., master's thesis, 47 p.

Belton, L.R., and Jackson-Smith, D., 2010, Factors influencing success among collaborative sage-grouse management groups in the western United States: Environmental Conservation, v. 37, no. 3, p. 250–260. [Also available at https://doi.org/10.1017/S0376892910000615.]

Bowen, Z.H., Freeman, M.C., and Bovee, K.D., 1998, Evaluation of generalized habitat criteria for assessing impacts of altered flow regimes on warmwater fishes: Transactions of the American Fisheries Society, v. 127, no. 3, p. 455–468. [Also available at https://doi.org/10.1577/1548-8659(1998)127<0455:EOGHCF>2.0.CO;2.]

Burnham, K.P., and Anderson, D.R., 2002, Model selection and multimodal inference—A practical information-theoretic approach, 2d ed.: New York, Springer-Verlag, 488 p.

Cash, D.W., Clark, W.C., Alcock, F., Dickson, N.M., Eckley, N., Guston, D.H., Jager, J., and Mitchell, R.B., 2003, Knowledge systems for sustainable development: Proceedings of the National Academy of Sciences of the United States of America, v. 100, no. 14, p. 8086–8091. [Also available at https://doi.org/10.1073/pnas.1231332100.]

Castleberry, D.T., Cech, J.J., Jr., Erman, D.C., Hankin, D., Healey, M., Kondolf, G.M., Mengel, M., Mohr, M., Moyle, P.B., Nielsen, J.L., Speed, T.P., and Williams, J.G., 1996, Uncertainty and instream flow standards: Fisheries, v. 21, no. 8, p. 20–21. [Also available at https://doi.org/10.1577/1548-8446-21-8.]

Clemen, R.T., 1997, Making hard decisions—An introduction to decision analysis: Pacific Grove, Calif., Duxbury Press, 664 p.

Conroy, M.J., and Peterson, J.T., 2013, Decision making in natural resources—A structured, adaptive approach: Oxford, United Kingdom, Wiley-Blackwell, 456 p. [Also available at https://doi.org/10.1002/9781118506196.]

Freeman, M.C., Bowen, Z.H., Bovee, K.D., and Irwin, E.R., 2001, Flow and habitat effects on juvenile fish abundance in natural and altered flow regimes: Ecological Applications, v. 11, no. 1, p. 179–190. [Also available at https://doi.org/10.1890/1051-0761(2001)011[0179:FAHEOJ]2.0.CO;2.]

Freeman, M.C., Irwin, E.R., Burkhead, N.M., Freeman, B.J., and Bart, H.L., 2005, Status and conservation of the fish fauna of the Alabama River system: American Fisheries Society Symposium, v. 2005, no. 45, p. 557–585.

Gregory, R., Failing, L., Harstone, M., Long, G., McDaniels, T., and Ohlson, D., 2012, Structured decision making—A practical guide to environmental management choices: Chichester, West Sussex, United Kingdom, John Wiley & Sons, 299 p. [Also available at https://doi.org/10.1002/9781444398557.]

Gunderson, L., and Light, S.S., 2006, Adaptive management and adaptive governance in the everglades ecosystem: Policy Science, v. 39, p. 323–334. [Also available at https://doi.org/10.1007/s11077-006-9027-2.]

Harwell, M.A., 1998, Science and environmental decision making in south Florida: Ecological Applications, v. 8, no. 3, p. 580–590. [Also available at https://doi.org/10.1890/1051-0761(1998)008[0580:SAEDMI]2.0.CO;2.]

Irwin, E.R., 2014, Defining ecological and economical hydropower operations—A framework for managing dam releases to meet multiple conflicting objectives: Journal of Energy Challenges and Mechanics, v. 1, no. 3, p. 139–146.

Irwin, E.R., and Freeman, M.C., 2002, Proposal for adaptive management to conserve biotic integrity in a regulated segment of the Tallapoosa River, Alabama, U.S.A: Conservation Biology, v. 16, no. 5, p. 1212–1222. [Also available at https://doi.org/10.1046/j.1523-1739.2002.00514.x.]

Irwin, E.R., Freeman, M.C., and Isely, J.J., 1997, Effects of flow regime on recruitment and growth of juvenile basses (*Micropterus* spp.) in flow-regulated rivers: Arlington, Va., U.S. Fish and Wildlife Service, 32 p.

Irwin, E.R., Kennedy, K.M., Goar, T.P., Martin, B., and Martin, M.M., 2011, Adaptive management and monitoring for restoration and faunal recolonization of Tallapoosa River shoal habitats: Montgomery, Ala., Alabama Department of Conservation and Natural Resources, ALCFWRU Report 2011–1, 49 p.

Johnson, B.L., 1999, The role of adaptive management as an operational approach for resource management agencies: Conservation Ecology, v. 3, no. 2, article 8, accessed March 2017 at https://doi.org/10.5751/ES-00136-030208.

Lydeard, C., and Mayden, R.L., 1995, A diverse and endangered aquatic ecosystem of the southeast United States: Conservation Biology, v. 9, no. 4, p. 800–805. [Also available at https://doi.org/10.1046/j.1523-1739.1995.09040800.x.]

Marcot, B.G., Steventon, J.D., Sutherland, G.D., and McCann, R.K., 2006, Guidelines for developing and updating Bayesian belief networks applied to ecological modeling and conservation: Canadian Journal of Forest Research, v. 36, no. 12, p. 3063–3074. [Also available at https://doi.org/10.1139/x06-135.]

Martin, B.M., 2008, Nest survival, nesting behavior, and bioenergetics of redbreast sunfish on the Tallapoosa River, Alabama: Auburn, Ala., Auburn University, master's thesis, 91 p.

McLain, R.J., and Lee, R.G., 1996, Adaptive management—Promises and pitfalls: Environmental Management, v. 20, no. 4, p. 437–448. [Also available at https://doi.org/10.1007/BF01474647.]

McManamay, R.A., Brewer, S.K., Jager, H.I., and Troia, M.J., 2016, Organizing environmental flow frameworks to meet hydropower mitigations needs: Environmental Management, v. 58, no. 3, p. 365–385. [Also available at https://doi.org/10.1007/s00267-016-0726-y.]

Nakamura, N., Kasahare, S., and Yada, T., 1971, Studies on the usefulness of the bluegill sunfish, *Lepomis macrochirus* Rafinesque, as an experimental standard animal—II. On the developmental stages and growth from egg through one year: Journal of the Faculty of Fisheries and Animal Husbandry, Hiroshima University, v. 10, no. 2, p. 139–151. [Also available at https://doi.org/10.15027/41165.]

National Research Council, 1999, Downstream—Adaptive management of Glen Canyon Dam and the Colorado River ecosystem: Washington, D.C., National Academies Press, 242 p.

Norsys Software Corporation, 1998, Netica Application: Norsys Software Corporation web page, accessed October 2017 at https://www.norsys.com.

Pearsall, S.H., McCrodden, B.J., and Townsend, P.A., 2005, Adaptive management of flows in the lower Roanoke River, North Carolina, USA: Environmental Management, v. 35, no. 4, p. 353–367. [Also available at https://doi.org/10.1007/s00267-003-0255-3.]

Peterson, J.T., and Evans, J.W., 2003, Quantitative decision analysis for sport fisheries management: Fisheries (Bethesda, Md.), v. 28, no. 1, p. 10–21. [Also available at https://doi.org/10.1577/1548-8446(2003)28[10:QDAFSF]2.0.CO;2.]

Podolak, K., and Yarnell, S., 2015, Adaptive management in Federal Energy Regulatory Commission relicensing, *in* Lassiter, A., ed., Sustainable Water: Challenges and Solutions from California: Oakland, Calif., University of California Press, 389 p.

Poff, N.L., 2017, Beyond the natural flow regime? Broadening the hydro-ecological foundation to meet environmental flows challenges in a non-stationary world: Freshwater Biology, v. 63, no. 8, p. 1011–1021. [Also available at https://doi.org/10.1111/fwb.13038.]

Poff, N.L., Allan, J.D., Palmer, M.A., Hart, D.D., Richter, B.D., Arthington, A.H., Rogers, K.H., Meyer, J.L., and Stanford, J.A., 2003, River flows and water wars—Emerging science for environmental decision making: Frontiers in Ecology and the Environment, v. 1, no. 6, p. 298–306. [Also available at https://doi.org/10.1890/1540-9295(2003)001[0298:RFAWWE]2.0.CO;2.]

Raadgever, G.T., Mostert, E., Kranz, N., Interwies, E., and Timmerman, J.G., 2008, Assessing management regimes in transboundary river basins—Do they support adaptive management?: Ecology and Society, v. 13, no. 1, article 14, accessed May 2017 at https://doi.org/10.5751/ES-02385-130114.

Richter, B.D., and Thomas, G.A., 2007, Restoring environmental flows by modifying dam operations: Ecology and Society, v. 12, no. 1, article 12, accessed March 2017 at https://doi.org/10.5751/ES-02014-120112.

Royle, J.A., and Kéry, M., 2007, A Bayesian state-space formulation of dynamic occupancy models: Ecology, v. 88, no. 7, p. 1813–1823. [Also available at https://doi.org/10.1890/06-0669.1.]

Russo, T., 1999, The U.S. Federal Energy Regulatory Commission—Lessons learned in the last 78 years: U.S. Federal Energy Regulatory Commission, Contributing Paper to the World Commission on Dams, 27 p.

Swingle, H.S., 1954, Fish populations in Alabama Rivers and impoundments: Transactions of the American Fisheries Society, v. 83, no. 1, p. 47–57. [Also available at https://doi.org/10.1577/1548-8659(1953)83[47:FPIARA]2.0.CO;2.]

Travnichek, V.H., Bain, M.B., and Maceina, M.J., 1995, Recovery of a warmwater fish assemblage after the initiation of a minimum-flow release downstream from a hydroelectric dam: Transactions of the American Fisheries Society, v. 124, no. 6, p. 836–844. [Also available at https://doi.org/10.1577/1548-8659(1995)124<0836:ROAWFA>2.3.CO;2.]

U.S. Geological Survey, 2018, USGS water data for the Nation: U.S. Geological Survey National Water Information System database, accessed May 2018 at https://doi.org/10.5066/F7P55KJN.

Van Winkle, W., Coutant, C.C., Jager, H.I., Mattice, J.S., Orth, D.J., Otto, R.G.S., Railsback, F., and Sale, M.J., 1997, Uncertainty and instream flow standards: perspectives based on hydropower research and assessment: Fisheries, v. 22, no. 7, p. 21–22.

Walters, C.J., 1997, Challenges in adaptive management of riparian and coastal ecosystems: Conservation Ecology, v. 1, no. 2, article 1, accessed October 2018 at https://doi.org/10.5751/ES-00026-010201.

Williams, B.K., 2011, Adaptive management of natural resources—Framework and issues: Journal of Environmental Management, v. 92, no. 5, p. 1346–1353. [Also available at https://doi.org/10.1016/j.jenvman.2010.10.041.]

Williams, B.K., and Brown, E.D., 2011, Adaptive management—The U.S. Department of the Interior applications guide: Washington D.C., U.S. Department of the Interior, Adaptive Management Working Group, 136 p.

Williams, B.K., and Johnson, F.A., 1995, Adaptive management and the regulation of waterfowl harvests: Wildlife Society Bulletin, v. 23, p. 430–436.

Williams, B.K., Szaro, R.C., and Shapiro, C.D., 2007, Adaptive management—The U.S. Department of the Interior technical guide: Washington, D.C., U.S. Department of the Interior, 72 p.

Wood, T., 2015, Alabama’s Wildlife Action Plan 2015–2025: Montgomery, Ala., Alabama Department of Conservation and Natural Resources, 491 p.

Yoccoz, N.G., Nichols, J.D., and Boulinier, T., 2001, Monitoring of biological diversity in space and time: Trends in Ecology and Evolution. v. 16, no. 8, p.446–453. [Also available at https://doi.org/10.1016/S0169-5347(01)02205-4.]

Appendix A1. Transcripts from the Adaptive Management Workshop, April 30–May 1, 2003

This appendix contains transcripts developed by the R.L. Harris Stakeholders at a workshop (April 30–May 1, 2003). This document was prepared under contract to the U.S. Geological Survey and is owned by the U.S. Geological Survey. Appendix A1 is available at https://doi.org/10.3133/ofr20191026.

Appendix A2. Initial Bayesian Belief Network (2005), Training Cases and Learned Networks (2005–16)

Figures within depict the model probability updating during the 12-year project. The initial model suggested the best management option (blue rectangles) and monitoring data were used to inform the model. Monitoring data were entered as training cases and after data were incorporated, the learned network approximated updated probabilities for the next time step. Appendix A2 is available at https://doi.org/10.3133/ofr20191026.

Appendix A3. Charter of the R.L. Harris Stakeholders Board

This appendix contains the Charter developed by the R.L. Harris Stakeholders Board. This document was prepared under contract to the U.S. Geological Survey and is owned by the U.S. Geological Survey. Appendix A3 is available at https://doi.org/10.3133/ofr20191026.

Chapter B

Long-Term Dynamic Occupancy of Shoal-Dwelling Fishes Above and Below a Hydropeaking Dam

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Introduction

Because of the anthropogenic alteration of large river systems, faunal assemblages native to these systems mostly persist in river fragments that may be free flowing (that is, unimpounded) yet are variously affected by flow modification caused by upstream dams and reservoirs. This fragmentation and alteration of riverine habitat demand management options that explicitly address restoration and conservation of native aquatic biota and fisheries in flow-altered river reaches downstream from dams. Management options have usually been standard one-time negotiated flow plans without flexibility for change and generally begun and (or) governed by either the Federal Energy Regulatory Commission (FERC) or by the laws set forth in the Endangered Species Act. These historically one-time negotiated flow scenarios have hampered our ability to define relations between faunal processes (for example, population parameters) and system variability. Irwin and Freeman (2002) proposed that an adaptive approach could be used to manage riverine fish faunas in the southeast United States and elsewhere and specifically described the example of the Tallapoosa River, Alabama, where depleted low flows, altered thermal regimes, and flow instability were hypothesized to adversely affect fishes below the R.L. Harris Dam.

Adaptive management of river systems in the United States has been implemented in multiple river basins primarily in response to the protection of species under the Endangered Species Act (Jacobson and Galat, 2008; Cross and others, 2011). However, the value of adaptive management processes to non-Endangered Species Act management scenarios has been recognized to explicitly include multiple stakeholder objectives and to reduce uncertainty in the response of system parameters associated with management actions along with subsequent monitoring programs and decision updating (Williams and Brown, 2012 McGowan and others, 2011; Williams and Brown, 2014). For the purpose of this project, we adopted the National Research Council's (2004, p. 1–2) definition of adaptive management as a decision process with "flexible decision making that can be adjusted in the face of uncertainties as outcomes from management actions and other events become better understood. Careful monitoring of these outcomes both advances scientific understanding and helps adjust policies or operations as part of an iterative learning process." The process involves a planning phase to frame the problem, identify objectives and management options, construct predictive models that relate consequences of management to objectives, and define monitoring programs to capture project-relevant information to help improve decision making. The subsequent iterative phase begins at a decision point and uses monitoring data to update resource status relative to management during assessment (Williams and Brown, 2014). Adaptive management is well suited for situations where objectives are conflicting (even contentious) yet clearly defined by stakeholders, but uncertainty regarding how management actions affect those objectives is high.

The Tallapoosa River (Alabama) below R.L. Harris Dam was the subject of extreme stakeholder disagreement until spring 2005 when adaptive flow management changes were implemented at the dam (http://www.RiverManagement.org; Kennedy and others, 2006), consistent with the adaptive management framework described by Williams and Brown (2014). Leading up to the project, Irwin and Freeman (2002) provided a roadmap for applying adaptive decision making to address restoration and management of a strongly flow-regulated reach of the Tallapoosa River as an experimental system for determining the effects of managing for multiple stakeholder objectives including the conservation and management of populations of shoal-dwelling fish species below the dam. The deliberative or planning phase of the process (Williams and Brown, 2014) began by engaging stakeholders during a 3-day workshop (April 30–May 1, 2003) where they developed a set of objectives to define their values and desired outcomes associated with the management of the river and both reservoirs

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(upstream Lake Harris and downstream Lake Martin; see chapter A of this report and Irwin [2014]) for a more detailed description of stakeholder engagement). The stakeholders also developed a governance structure that consisted of a governing board with voting representation of all stakeholders. Next, flow management alternatives were identified based on hypothesized system responses (for example, periods of stable flows would increase recruitment for many fishes or flows between 14.2 and 34.0 cubic meters per second [m^3/s] and would provide conditions suitable for boating) and evaluated for their feasibility by technical teams appointed by the governing board; teams were usually comprised of engineers and biologists. Finally, because stakeholder objectives were often competing (for example, maximize power utility operation flexibility versus maximize diversity and abundance of native fauna and flora), a decision model that predicted the consequences of management portfolios on stated objectives was compiled and parameterized (chapter A; Irwin, 2014). Stakeholder objectives were equally weighted, and the management portfolio that was the most satisfactory to all the stakeholders was adopted as the starting point for adaptive management (portfolio hereafter termed the "Green Plan"; chapter A; Kennedy and others, 2006; Irwin, 2014).

During the iterative phase of the project, a long-term (12 years) monitoring program to quantify system responses to the Green Plan management changes of flow regimes began in 2005. The decision model was updated with monitoring data derived from multiple sources (chapter A); however, specific responses of fish populations to management were a primary interest to many stakeholders, and intensive fish monitoring began coincident with the Green Plan. The monitoring framework for fishes was a spatially and temporally replicated survey on randomly selected shoals (sites) that were either unregulated (that is, in free-flowing segments upstream from the dam or in direct tributaries) or variably regulated downstream from the dam (that is, most severe near the dam). This design allowed for the assessment of colonization and persistence (that is, 1-extinction) parameters in a metapopulation framework (Royle and Kéry, 2007). These vital rates were estimated for most of the fish species in regulated and unregulated river reaches of the middle Tallapoosa River Basin.

Temporal changes in metapopulation dynamics of fishes have been modeled to evaluate changes in persistence and colonization and (or) in relation to particular drivers in several southeastern river systems (Peterson and Shea, 2015; Shea and others, 2015; Freeman and others, 2017). Here, we are applying similar approaches to quantify differences in the processes underlying species assemblage composition in flow-regulated and unregulated sites and to evaluate evidence that flow and thermal regimes affect fish persistence and colonization in the flow-regulated reach below the dam. Metapopulations are spatial arrangements of distinct populations of species that occasionally intermix and differ with respect to their vital rates—specifically, extinction and colonization (Hanski and Simberloff, 1997). Dynamic occupancy models are hierarchical and are the product of submodels that account for latent detection/nondetection and latent occupancy process (Royle and Kéry, 2007). This class of models is beneficial because it estimates metapopulation rates through time and allows for the inclusion of random effects and covariates that may help explain temporal patterns in vital rates.

This chapter describes a multispecies assessment of persistence (that is, probability that a site remains occupied by a species from one time step to the next) and colonization (that is, exists at a site where it was previously absent) for a suite of 38 fish species over a 12-year period on shoals in the middle Tallapoosa River Basin. The monitoring and analytical framework were in support of the adaptive management of flows below a peaking hydropower dam and allowed for the estimation of detection probabilities to account for incomplete detection during monitoring (MacKenzie and others, 2003, Royle and Kéry, 2007). In addition, a long-term assessment of metapopulation dynamics on shoals in relation to temperature and hydrologic factors was completed. Specifically, we analyzed year-to-year variation in the probability that a species would persist or colonize against year-to-year variation in covariates that described different annual flow and thermal regimes in the basin. Because of the adaptive management project, the Tallapoosa River presented an opportunity to determine the effects of flow management on fish populations. Findings should be directly transferable to other similarly fragmented, flow-managed rivers that harbor shoal-dwelling Greatest Conservation Need species, such as the Coosa and (or) Tennessee Rivers.

The overall purpose of this study was to evaluate the effects of experimental flows on the persistence and colonization of shoal-dwelling fishes in the Tallapoosa River. Specific analyses were conducted (1) to evaluate evidence that fishes occupying shoals downstream from R.L. Harris Dam were less likely to persist or colonize from year to year than at sites with unregulated flow regimes and (2) to estimate the influences of flow fluctuations and of lowered water temperature on the persistence and colonization of shoal fishes downstream from R.L. Harris Dam. Based on a priori stated hypotheses, colonization rates should increase with the provision of spawning windows (that is, periods of semistable flows) below the dam, increase with distance from the dam, and be higher in unregulated reaches. Likewise, persistence rates should be lower at unregulated sites, be lower closest to the dam, and be unaffected by spawning windows. Hypotheses related to flow fluctuations included low expected persistence and colonization during years with more flow fluctuations than during years with fewer fluctuations. Hypotheses related to thermal effects predicted that colonization and persistence would increase downstream from the dam based on inflows of warmer water and dilution effects. The analysis provided for a community-wide assessment that included species of interest to stakeholders (that is, Greatest Conservation Need species [Wood, 2015] or sportfishes) and allowed for reduction in uncertainty regarding how management actions influenced stakeholder objectives related to fish populations outlined in the adaptive management project (chapter A, this report).

Methods

The regulated study reach, beginning at R.L. Harris Dam and terminating 78 kilometers (km) downstream in the headwaters of Lake Martin, represents one of the longest and highest quality segments of unimpounded Piedmont river habitat remaining in the Mobile River Basin (Lydeard and Mayden, 1995; Mettee and others, 1996; Neves and others, 1997; fig. B1). The native fish assemblage in the Piedmont section of the Tallapoosa River Basin includes at least 57 species including 7 species endemic to the Tallapoosa system (Mettee and others, 1996; Boschung and Mayden, 2004). During the adaptive management project, there were 2 main changes to flows from R.L. Harris Dam—increase in base flow and provision of spawning windows—and monitoring below the dam incorporated shoals in 3 reaches from near the dam (Malone, which was the closest at 2.6 km downstream) to shoals just upstream from Lake Martin (Horseshoe Bend, which was the farthest at 70.23 km downstream from the dam, and included sites at Peter's Island). The Wadley reach was in between these two reaches (beginning 11.98 km below the dam). A total of two unregulated river reaches (Hillabee Creek and the upper Tallapoosa River, hereafter, Heflin) were monitored to assess how persistence and colonization fluctuated independent of regulated flows (that is, under "natural" conditions).

Field Sampling Methods

Sampling was conducted in summer (May–June) and fall (September–November) 2005–16 in 5 sampling reaches in the Piedmont Tallapoosa River Basin; 5 shoals per sampling reach were randomly selected from all shoals in each reach (25 shoals total; 15 below the dam; hereafter, "sites"; fig. B1). We used prepositioned area electrofishers (PAEs; Bain and others, 1985; Bowen and others, 1998; Freeman and others, 2001) to sample fishes from sites. We deployed 1.5-meter (m) × 6-m PAEs in nearshore (less than [<] 1 m from bank) and offshore (greater than [>] 1 m from bank) shallow water habitats (<0.75 m in depth) during daylight hours. PAEs were electrified with alternating current by a 3,500 W generator and Smith-Root 7.5 generator powered pulsator unit. Before sampling, all PAEs were set and left undisturbed for 20 minutes (see Bain and others, 1985). Each PAE was oriented with its long side parallel to shore and to downstream flow; two workers held a 3- × 2-m seine (2-millimeter [mm] mesh size) at the downstream end to capture drifting stunned fish. One worker kicked through the PAE to dislodge any fish trapped in vegetation or substratum; then, PAEs were visually inspected for additional fish.

In 2005–7 we collected 20 PAE samples per site, and in 2008–16 we collected 10 PAE samples per site. The reduced effort was warranted based on the analysis of all data from the previous year indicating that 10 samples were sufficient for estimation of parameters of interest related to the response of fish populations to management. Prior to 2008, most fish were euthanized in MS–222, and fixed in 10-percent formaldehyde in the field; large specimens (>200 mm) were generally identified, measured [in millimeters total length] with a measuring board or soft measuring tape, and released live in the river. Specimens remained in formaldehyde for 2 weeks, then underwent several water soak and rinse cycles. Finally, specimens were stored in 70-percent ethanol and identified and measured (in millimeters total length) in the laboratory. Beginning in 2008, fish were identified and measured in the field, except where uncertainty regarding species identity was high (for example, juvenile Cyprinidae); those specimens were prepared using the above described protocol for pre-2008 samples.

Collections and ensuing count data from individual seasons, dates, sites, and PAEs were kept separate to facilitate the creation of detection/nondetection histories for each species in each year. The detection histories (0, 1) were incorporated into a matrix that included site, season, species, and year for 25 sites, 2 seasons (summer and fall), 38 species (see results below), and 12 years of data.

Analysis

We tested for differences in persistence and colonization of selected fish species between flow-regulated and unregulated sites within the study system. These vital rates represent two main aspects of the metapopulation dynamics of this fish community. Persistence was defined as the probability that a species that was present in a site during a given year will also be present in the next year. Colonization was defined as the probability that a species that is absent at a site in a given year will be present in the next year. To address our two objectives, we fit models to our sampling data, which consisted of annual observations for 12 years (2005–16) of fish species occurrences at as many as 25 sites (15 downstream from R.L. Harris Dam [3 reaches, Malone (upper, nearest the dam), Wadley (middle), and Horseshoe Bend (lowest)], 5 in the free-flowing Tallapoosa River main stem upstream from the dam and Lake Harris, and 5 in Hillabee Creek—a large, unregulated, and free-flowing tributary to the Tallapoosa River). Of the 52 fish species that we detected during monitoring for all years, 38 were the most common species and had >25 detections over all sites and years (table B1).

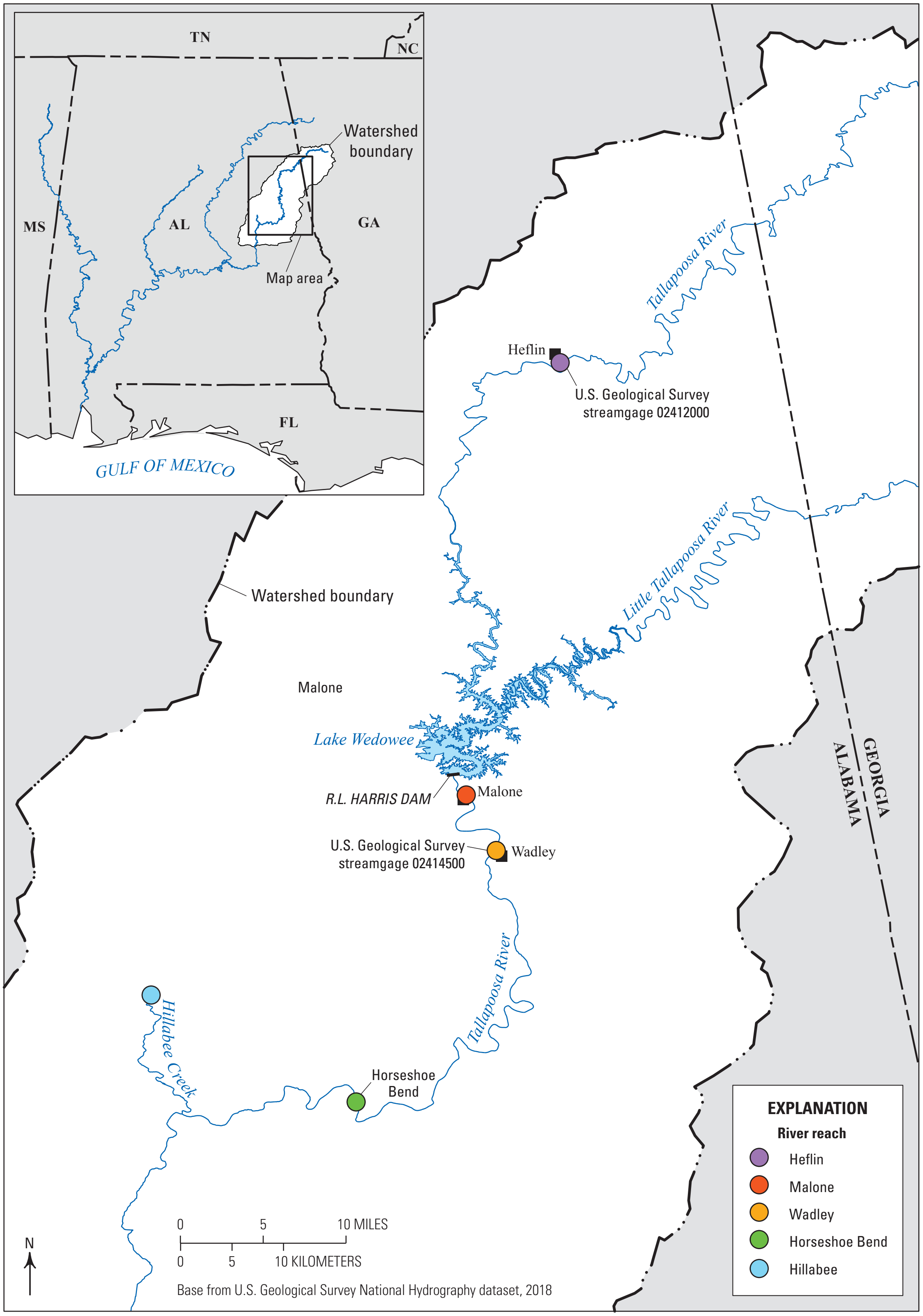

Figure B1. Locations of the 5 river reaches where fishes were collected from 5 shoals at each of the reaches between 2005–16.

Table B1. Count data for 38 species captured in prepositioned area electrofishers at 25 sites in 5 river reaches variably affected by R.L. Harris Dam (5 per reach; sampling years 2005–16).

[Horseshoe Bend is the lowest reach below the dam; Heflin and Hillabee are unregulated; PAE, prepositioned area electrofisher]

Species	Horseshoe	Wadley	Malone	Heflin	Hillabee
Ambloplites ariommus	29	48	49	24	61
Campostoma oligolepis	153	518	599	4,120	1,963
Cottus tallapoosae	1	4	5	69	5
Cyprinella callistia	2,568	3,196	602	3,115	4,444
Cyprinella gibbsi	280	117	20	766	2,989
Cyprinella venusta	165	210	128	516	1,075
Dorosoma petenense	2	2	124	0	0
Etheostoma chuckwachatte	941	3,064	1,750	3,031	5,114
Etheostoma stigmaeum	118	492	284	2,938	936
Etheostoma tallapoosae	183	296	498	314	217
Fundulus bifax	0	0	2	14	171
Fundulus olivaceus	0	2	3	112	33
Gambusia affinis	1	0	0	280	30
Hybopsis lineapunctata	1	2	3	3	22
Hypentelium etowanum	349	301	158	1,593	336
Ictalurus punctatus	67	86	43	121	64
Lepomis auritus	64	210	655	363	389
Lepomis cyanellus	0	4	16	5	3
Lepomis macrochirus	4	10	162	28	53
Luxilus chrysocephalus	7	54	48	138	290
Macrhybopsis aestivalis	1	0	0	0	401
Micropterus henshalli	88	29	45	76	84
Micropterus tallapoosae	74	32	33	10	48
Minytrema melanops	0	0	0	0	69
Moxostoma duquesnei	30	0	4	241	187
Moxostoma poecilurum	6	1	7	74	69
Notropis baileyi	2	116	20	6	3
Notropis stilbius	413	420	96	438	1,585
Notropis texanus	5	2	2	4	58
Noturus funebris	16	46	28	98	202
Noturus leptacanthus	26	25	13	434	145
Percina kathae	14	8	5	2	76
Percina palmaris	1,397	4,022	2,011	2,254	2,773
Percina smithvanizi	317	1,104	315	905	664
Phenacobius catostomus	8	0	2	247	165
Pimephales vigilax	32	0	0	3,014	2,234
Pylodictis olivaris	16	13	2	25	13
Semotilus atromaculatus	0	0	0	25	5
Total individuals	**7,378**	**14,434**	**7,732**	**25,403**	**26,976**
Richness	33	30	33	35	37
Total PAE samples	660	1,451	1,431	1,100	1,010
Mean number/PAE	11.18	9.95	5.40	23.09	26.71

For both objectives, we estimated persistence and colonization using hierarchical, dynamic occupancy models defined by Royle and Kéry (2007). These models were hierarchical in that they separated the ecological (occupancy) and observation (detection/nondetection) processes that result in the data (Kéry and Schaub, 2012) and therefore allowed for the possibility that a species may have been present at a site and yet not detected in a particular sample. We modeled occupancy through time at each site, beginning with the first sample, in which we allowed each species to have an (unknown) initial probability of occurrence. In the model comparing dynamics between flow-regulated and unregulated sites (objective 1), the initial species-specific occurrence was allowed to differ between sites downstream from the dam (flow-regulated sites), and sites in unregulated reaches:

$$z_{1ij} \sim \text{Bernoulii}\ (Psi_{i.regulated} \times Reg_j + Psi_{i.unregulated} \times (1 - Reg_j) \qquad (1)$$

where

z_{1ij} is 1 or 0 for presence or absence of species i at site j in year 1;

$Psi_{i.regulated}$ is the probability of species i occurrence in a flow-regulated site;

$Psi_{i.unregulated}$ is the probability of species i occurrence in an unregulated site; and

Reg_j is 1 if site j is flow regulated and 0 otherwise.

Then, in each subsequent year, a species was assumed to be present at a site with a probability that equaled either persistence (if the species was present in the previous year) or colonization (if the species was absent in the previous year). We modeled persistence and colonization as functions of objective-specific covariates, described below.

We accounted for imperfect detection by explicitly modeling observed detections and nondetections of each species in each site sample as a random draw from a species-specific probability of detection, conditional on the species being truly present; that is,

$$y_{tijl} \sim \text{Bernoulli}\ (p_{tij} \times z_{tij}) \qquad (2)$$

$$\text{logit}\ (p_{tij}) = p_i + \varepsilon_{tjl} \qquad (3)$$

where

y_{tijl} is the observed detection (1) or nondetection (0) in year t of species i in site j during replicate l (l=1 or 2, representing as many as 2 samples in each site and year);

p_{tij} is the probability of detection in year t of species i in site j;

z_{tij} is 1 if species i is present in year t in site j and is 0 otherwise;

p_i is the mean probability of detection of species i; and

ε_{tjl} is a normally distributed random effect with a mean of 0, accounting for variation in detection among surveys caused by unmodeled factors (for example, water level, weather).

We did not include explicit covariates on detection to simplify the models because identifying sources of variation in detection was not an objective. The observation process that resulted in the actual observed detection (1) or nondetection (0) of each species in each sample was represented as a random draw from a species-specific detection probability distribution, conditional on the species being truly present. To account for imperfect detection, sites were sampled as much as twice annually, and we used the summer and fall samples within a given year as replicates to estimate detection probabilities, assuming that any species detected in either sample was present and available to be detected in both samples. We also assumed that detection could differ among surveys (for example, because of differences in water level, instream habitat, and weather). Variation was represented as a survey-level random effect (normally distributed around a mean of 0) added to species-specific detection for each survey. Models were fit with a Markov chain Monte Carlo software JAGS (Plummer, 2003), run using package “rjags” in R (Plummer, 2014; R Core Team, 2014). All code is included in appendix B1.

To address the first objective (comparison of persistence and colonization between regulated and unregulated sites), we modeled persistence and colonization as additive functions of random effects representing variation among species, years, and sites and an additional potential effect of flow regulation:

$$\text{logit}\ ([persistence \text{ or } colonization]_{tij}) = b_0 + b_{i.reg} \times Reg_j + \varepsilon_t + \varepsilon_i + \varepsilon_j \qquad (4)$$

where

$persistence_{tij}$ and $colonization_{tij}$ are probabilities that species i persists (if present at t−1) and colonizes (if absent at t−1) site j in year t;

b_0 is the intercept;

$b_{i.reg}$ is the effect of site j being in a flow-regulated reach on persistence or colonization for species i; and

ε_t, ε_i, and ε_j are normally distributed random effects (each with a mean of 0) representing unmodeled variation in persistence and colonization rates among years, species, and sites, respectively.

We therefore assumed that fish species persistence and colonization in the study sites differed from year to year in response to unmeasured factors (such as regional variation in rainfall and temperature regimes; random year effect) and that local factors additionally affected year-to-year species turnover in individual sites (random site effect). We used this model to test for differences in species persistence, colonization, or both at sites downstream from R.L. Harris Dam compared to other sites (represented by the fixed effect, b_{reg}, for site location in the flow-regulated reach). We would expect fishes to have lower persistence, colonization, or both at sites downstream from R.L. Harris Dam if dam operations reduce fish survival or reproduction and, therefore, population sizes. The additive model allowed species to have differing average persistence and colonization rates (that is, random species effect on intercepts) and to differ in their responses to flow regulation (that is, random slopes), but the year effects were assumed to be similar at all sites (and, conversely, sites differed similarly in all years).

For our second objective (that is, relation of persistence/colonization [below dam only] to generation events and water temperatures), we fit a similar mixed-effects model to fish observation data for the 15 flow-regulated sites downstream from R.L. Harris Dam. As in the model using data for all sites, we included random effects on the intercepts for persistence and colonization that represented unexplained, additive variation among year, species, and sites. We additionally included three fixed effects in the model for persistence and colonization at the flow-regulated sites: annual hydropeaking frequency, seasonal cumulative water temperature relative to spawning requirements, and site-specific distance from R.L. Harris Dam. The first two variables specifically addressed our objective of estimating effects of dam operations on fish population dynamics through effects on flow instability (caused by hydropeaking) and on water temperatures (because released water typically is cooler than in unregulated sites during spring and summer, when fishes typically reproduce). The variable for distance from the dam tested the extent to which effects of dam operation on either persistence or colonization may ameliorate downstream through the regulated reaches. Because our objective was to estimate variable influences on metapopulation rates (rather than to identify the best-fitting model), and to simplify interpretation, we did not include variable interactions.

Covariates were calculated using the following methods. Distance from the dam was simply the linear distance (1.0 km) from the dam to a given site. The yearly number of generation events was derived by counting the number of days during the months of March–September that maximum daily discharge exceeded 198.2 m^3/s. Discharge data were obtained from the U.S. Geological Survey for the Tallapoosa River at Wadley, Ala., streamgage (U.S. Geological Survey streamgage 02414500; U.S. Geological Survey, 2018). The number of 10-day cumulative degree day periods that exceeded 63 cumulative degree day periods at a threshold of 17 degrees Celsius (°C) during March–September were enumerated for each year. This was based on the temperature requirements for successful hatching of a common centrarchid (*Lepomis auritus* [redbreast sunfish]; Andress, 2002). Temperature data were obtained for the Wadley reach using temperature logger data provided by the Alabama Power Company. Each covariate was scaled to a mean of 0 and standard deviation of 1 for analysis. We also plotted the relation between the number of generation events and cumulative degree day periods (fig. B2).

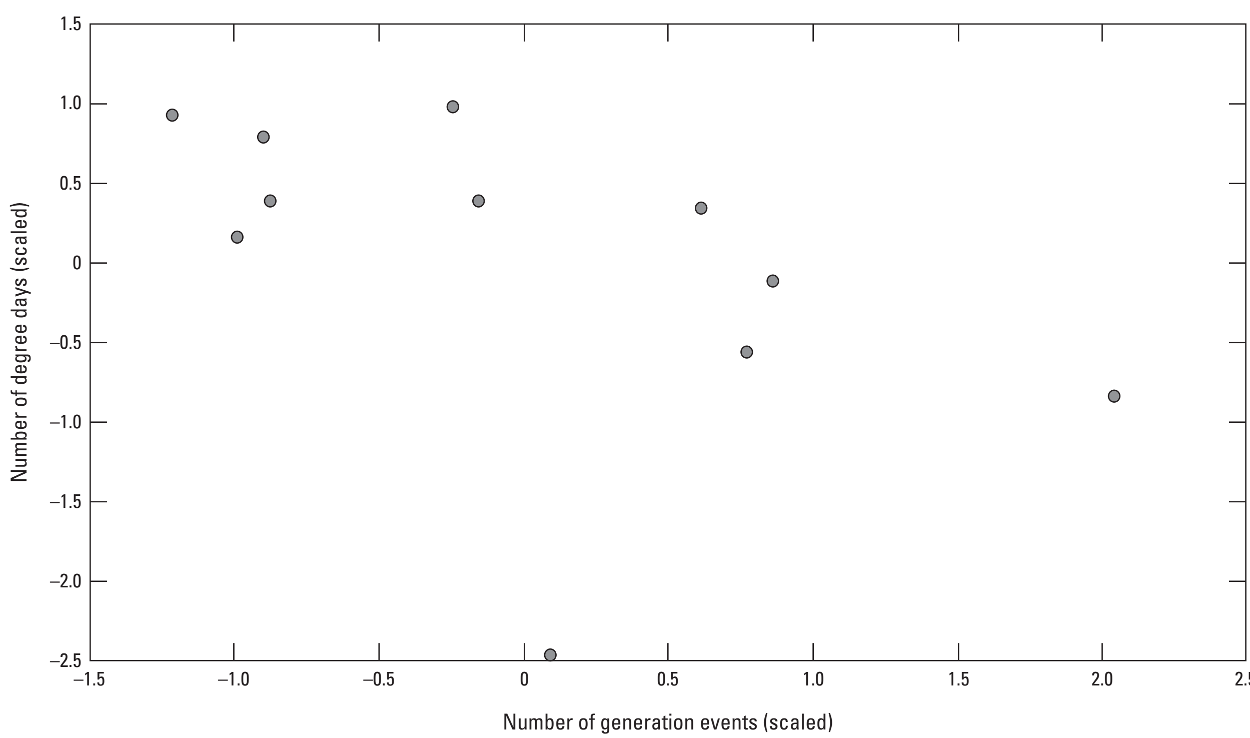

Figure B2. Relation between generation events and cumulative degree days; both variables were standardized to a mean of 0 and a standard deviation of 1. The outlier was 2006 when temperatures were cool and generation events were average. The correlation coefficient (*r*) is −0.49 and includes the outlier.

Results

We included 81,923 individual fish detections from 5,652 PAE samples taken in the 5 river reaches (25 sites total; 15 downstream from R.L. Harris Dam) over the 12 years. Overall, mean fish density (number of individuals/PAE) was lowest near the dam (5.40 fish/PAE, Malone) and highest in Hillabee Creek (26.71 fish/PAE; fig. B3; table B1). Although 46 species were captured during the study, we narrowed the list to 38 to include only species with greater than 25 detections. Overall richness was highest at the unregulated reaches, 35 and 37 for upper Tallapoosa (Heflin) and Hillabee Creek, respectively. The regulated reaches had the lower richness values; 33, 30, and 33 for Horseshoe Bend, Wadley and Malone, respectively. Many species at the regulated sites were represented by five or fewer specimens for the study period (table B1).

Fishes persisted and colonized at lower rates (that is, probabilities) at flow-regulated sites (downstream from R.L. Harris Dam) than at the unregulated sites in the upper Tallapoosa (Heflin) main stem and in Hillabee Creek. Estimated rates differed considerably among sites and among years; however, the estimated mean effects (across all species) of location downstream from the dam were to lower the probability of persistence by 81 percent (95-percent credible interval: −43 percent to −94 percent) and the probability of colonization by 65 percent (−18 percent to 86 percent; figs. B4 and B5; table B2). The effects of a site being in the flow-regulated reach also differed among species for both persistence and colonization. For persistence, 23.7 percent of species had an estimated positive effect; however, for colonization only two species (*Ambloplites ariommus* [shadow bass] and *Etheostoma chuckwachatte* [lipstick darter]) had an estimated effect that was positive (figs. B6 and B7).

We determined that both flow instability and depressed temperatures influenced fish persistence and colonization at the flow-regulated sites (tables B3 and B4; fig. B8). Models for the 15 sites downstream from R.L. Harris Dam resulted in somewhat higher estimates of persistence and colonization than when all 25 sites were modeled together, particularly persistence at the 5 sites that were located farthest (>60 km) downstream from the dam (figs. B9 and B10). This is likely due to removing the additive effect of flow regulation that was included when we compared unregulated to regulated sites. Averaged over all taxa, fish persistence increased with greater distance from the dam and decreased during years with more generation events (fig. B11). Colonization was lower in years with more generation events; however, no clear increases in colonization were observed at the sites farthest from the dam (fig. B12). Persistence was not clearly related to warmer thermal regimes but increased at sites farthest from the dam (fig. B13). Fish colonization increased during years with warmer water temperatures (fig. B14).

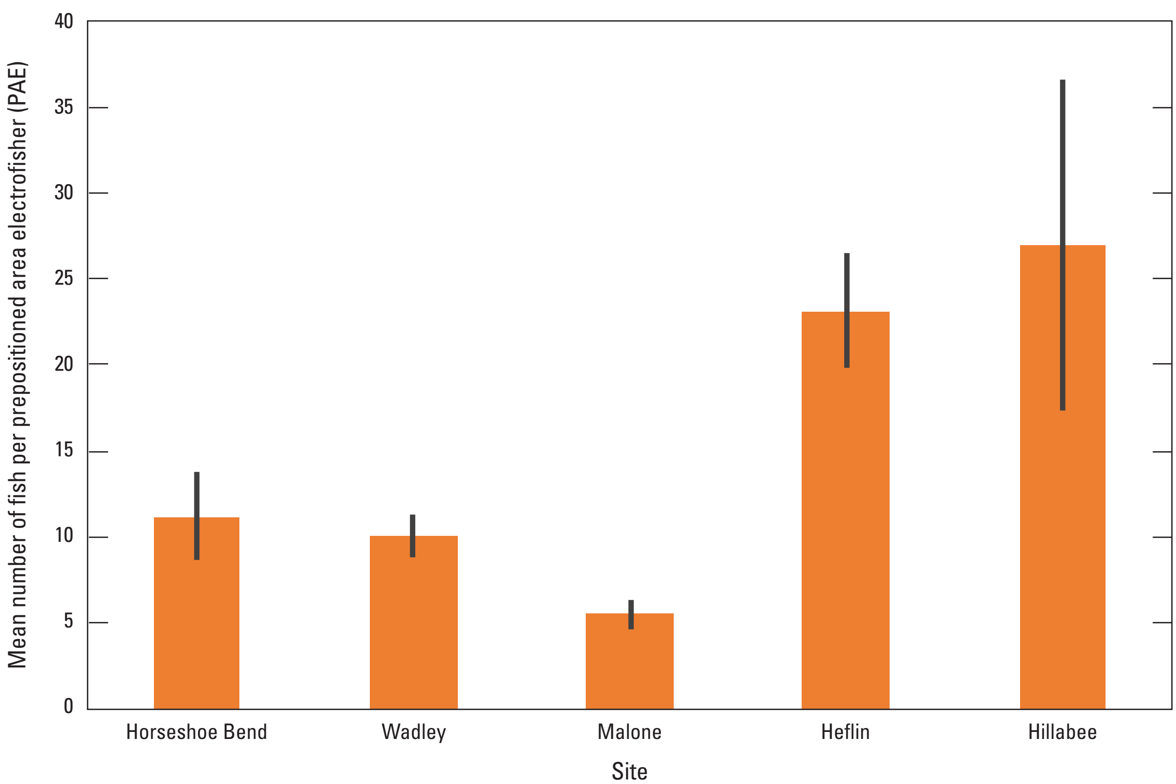

Figure B3. Average fish density (mean number of fish/PAE) by reach for all combined sites in a reach. Data from 2005 to 2016 were combined. High-low bars represent standard error.

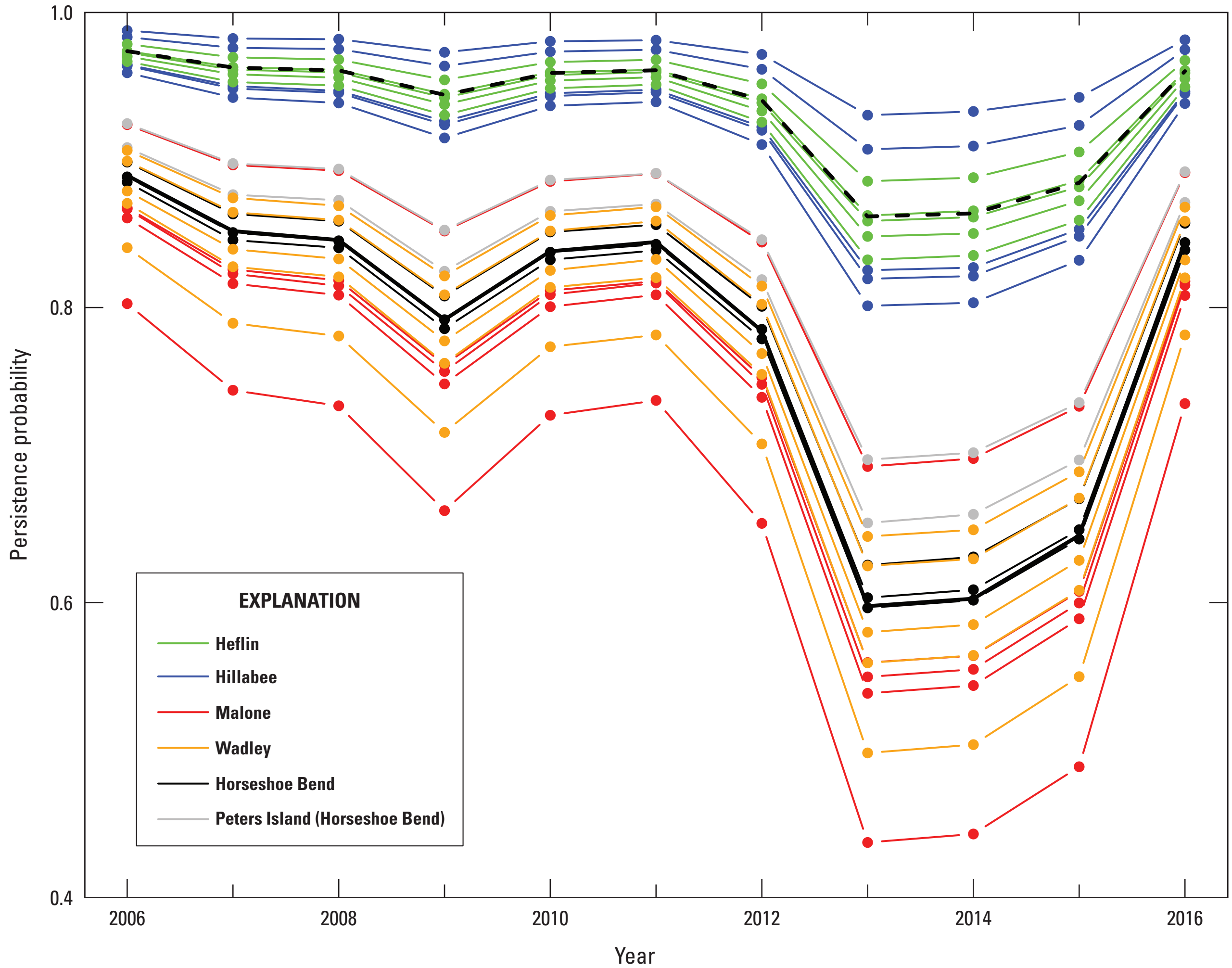

Figure B4. Estimated mean probability of persistence for 38 fish species at each of 25 sites in the Tallapoosa River system, 2005–16, modeled using dynamic, multitaxa occupancy models. Values for each year represent estimated mean species persistence from the previous year. [Sites with unregulated flow regimes are plotted in blue (Hillabee Creek) and green (Heflin). Flow-regulated sites are plotted in red (Malone), orange (Wadley), black (Horseshoe Bend), and gray (Peters Island). Horseshoe Bend and Peters Island together comprised the Horseshoe Bend reach. Estimated means for unregulated and regulated sites are plotted as bold dashed and solid lines, respectively.]

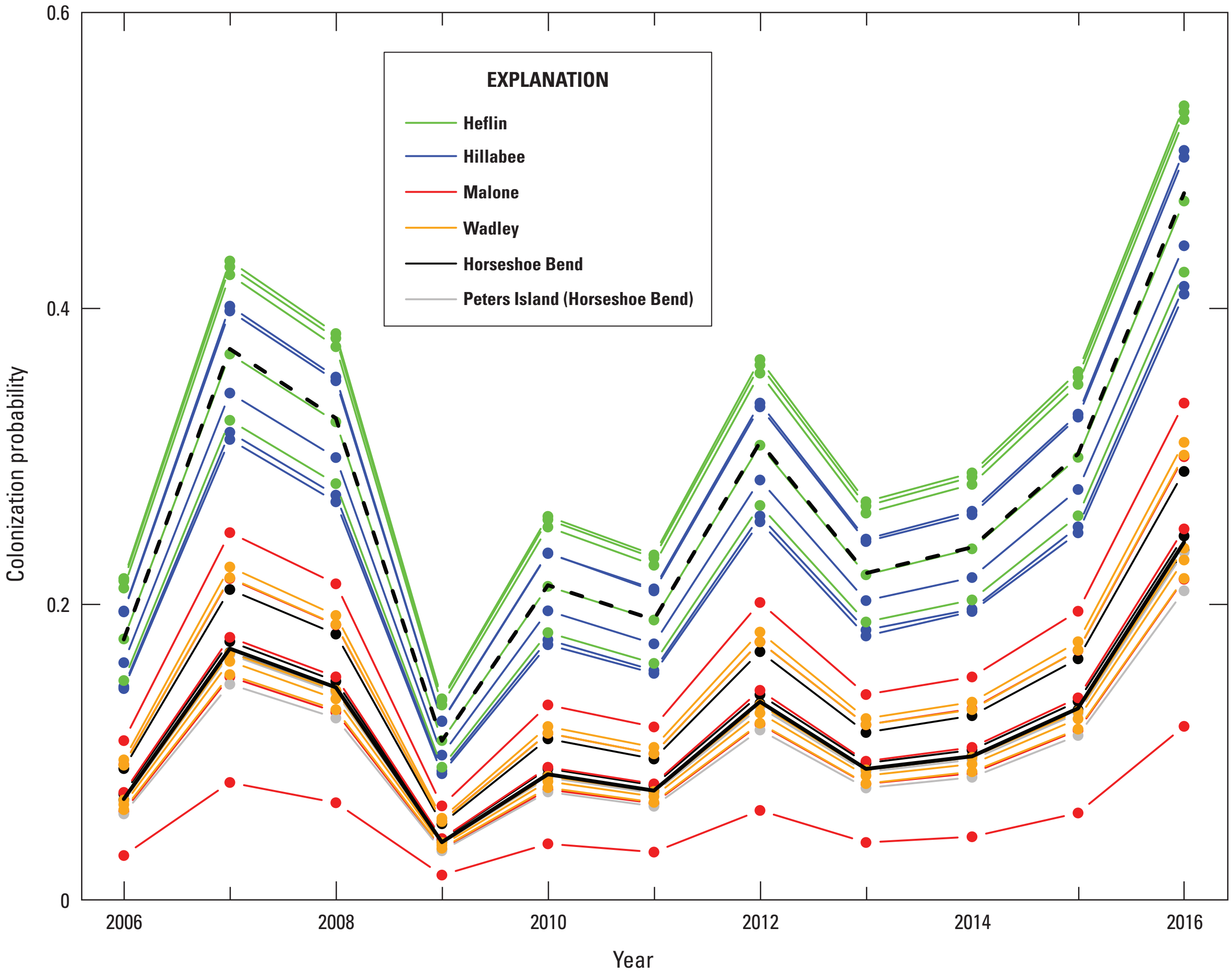

Figure B5. Estimated mean probability of colonization for 38 fish species at each of 25 sites in the Tallapoosa River system, 2005–16, modeled using dynamic, multitaxa occupancy models. Values for each year represent estimated mean species persistence from the previous year. [Sites with unregulated flow regimes are plotted in blue (Hillabee Creek) and green (Heflin). Flow-regulated sites are plotted in red (Malone), orange (Wadley), black (Horseshoe Bend), and gray (Peters Island). Estimated means for unregulated and regulated sites are plotted as bold dashed and solid lines, respectively.]

Table B2. Parameter estimates from dynamic occupancy model fit to 12 years (2005–16) of observations of 38 fish species at 25 Tallapoosa River system sites. Model includes additive fixed effect of site location downstream from R.L. Harris Dam (flow-regulation effect), and random effects for sites, taxa (intercept and slope), and years.

[Values are means on logit scale, with 95-percent credible intervals in parentheses. The **bold** numbers indicate that there was an effect measured for the parameter (the credible interval did not include zero).]

Parameter	Persistence	Colonization
Mean	2.84 (1.97, 3.77)	−1.16 (−2.18, −0.25)
Flow-regulation effect	**−1.64 (−2.75, −0.57)**	**−1.05 (−2.03, −0.20)**
	Variance terms	
Among sites	0.33 (0.03, 0.92)	0.21 (0.02, 0.59)
Among taxa (mean)	3.15 (1.27, 6.63)	3.42 (1.48, 7.11)
Among taxa (flow-regulation effect)	6.12 (2.44, 14.12)	2.52 (0.55, 6.47)
Among years	0.72 (0.17, 2.11)	0.70 (0.13, 2.38)

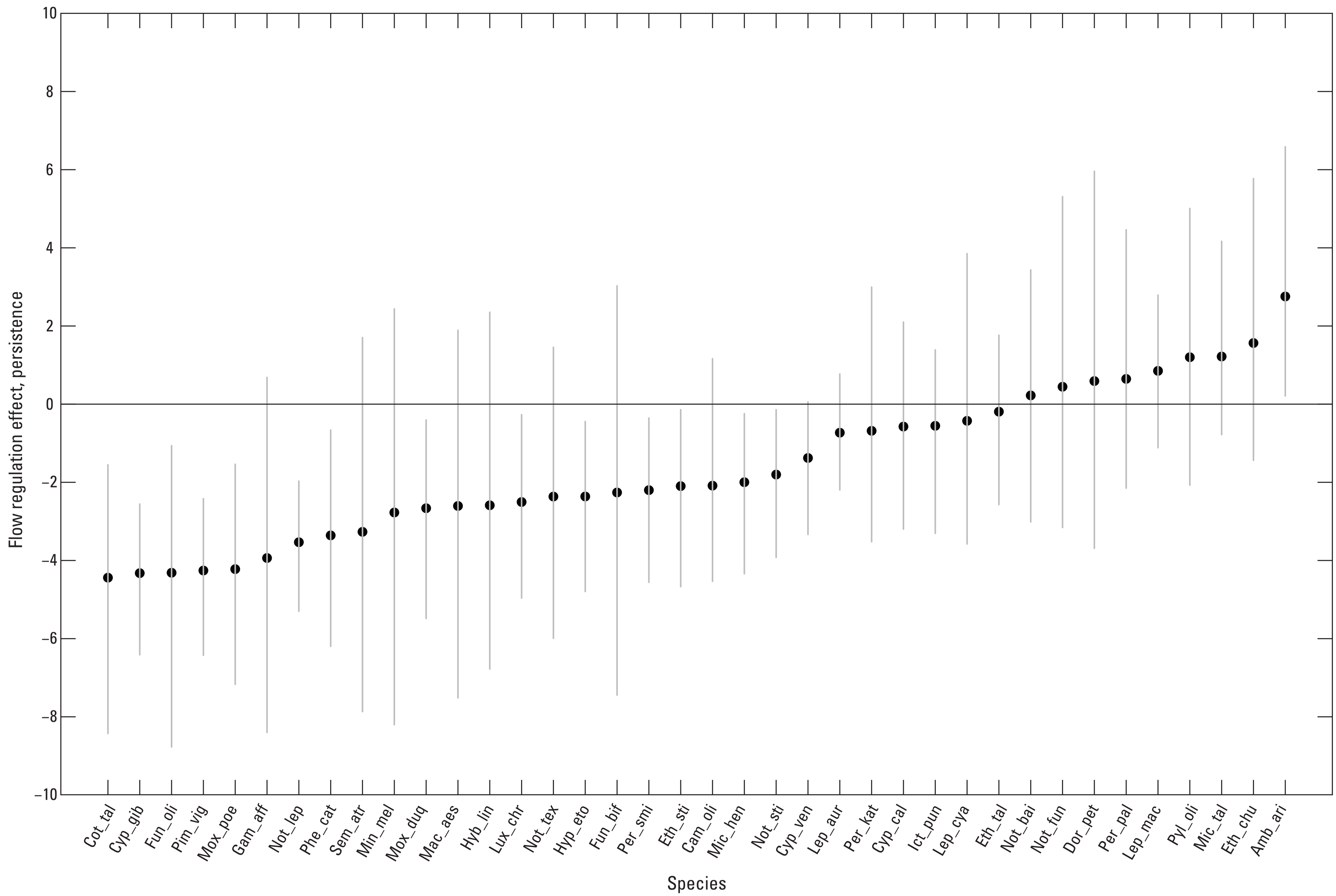

Figure B6. Estimated effects of location in the flow-regulated reach on species-specific persistence modeled using dynamic, multitaxa occupancy models applied to data for 25 sites in the Tallapoosa River system, 2005–16. Values are plotted on the logit scale and show 95-percent credible intervals. The horizontal line at 0 marks no effect; negative values indicate lower persistence in the flow-regulated reach compared to unregulated sites.

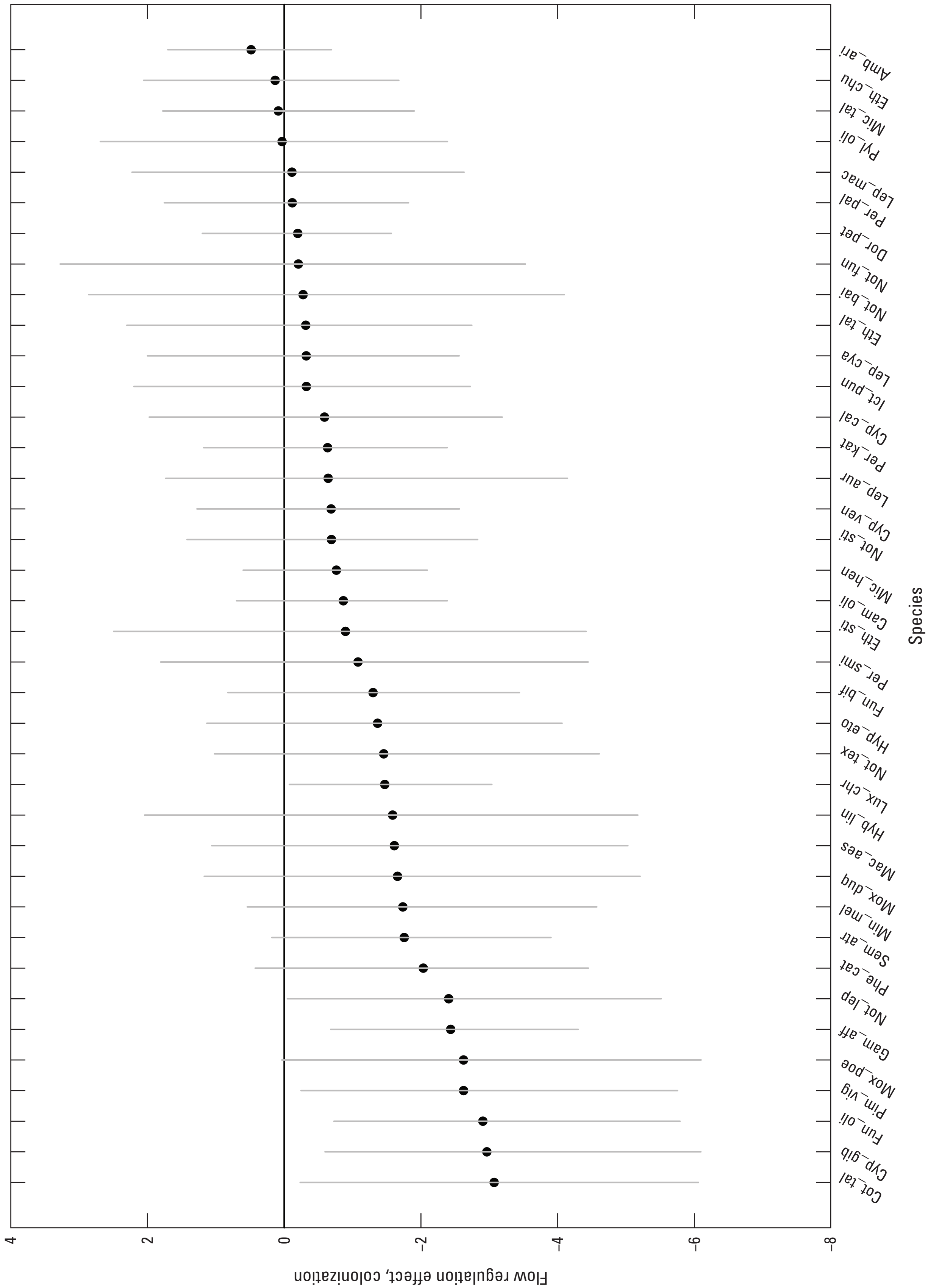

Figure B7. Estimated effects of location in the flow-regulated reach on species-specific colonization modeled using dynamic, multitaxa occupancy models applied to data for 25 sites in the Tallapoosa River system, 2005–16. Values are plotted on the logit scale and show 95-percent credible intervals. The horizontal line at 0 marks no effect; negative values indicate lower colonization in the flow-regulated reach compared to unregulated sites.

Table B3. Parameter estimates from dynamic occupancy model fit to 12 years (2005–16) of observations of 38 fish species at 15 flow-regulated sites. Model includes additive fixed effects of distance downstream from R.L. Harris Dam and annual number of generation events, an interaction between distance and number of generation events, and random effects for sites, taxa, and years.

[Values are means on logit scale, with 95-percent credible intervals in parentheses. The **bold** numbers indicate that there was an effect measured for the parameter (the credible interval did not include zero).]

Parameter	Persistence	Colonization
Mean	2.51 (1.23, 3.88)	−3.03 (−4.30, −1.62)
Distance from the dam	**0.94 (0.39, 1.71)**	**−0.95 (−1.94, −0.03)**
Annual number of generation events	−0.67 (−1.48, 0.16)	**−1.63 (−2.85, −0.25)**
Number of generation events × distance	−0.28 (−0.93, 0.36)	**−1.20 (−2.16, −0.18)**
Variance terms		
Among sites	0.19 (0.00, 0.82)	0.30 (0.00 1.09)
Among taxa	7.40 (2.66, 17.81)	6.59 (2.11, 15.90)
Among years	1.00 (0.02, 3.94)	1.34 (0.02, 5.77)

Table B4. Parameter estimates from dynamic occupancy model fit to 12 years (2005–16) of observations of 38 fish species at 15 flow-regulated sites. Model includes additive fixed effects of distance downstream from R.L. Harris Dam and annual number of days above a cumulative degree day threshold for fish reproduction (number of 10-day periods greater than [>] 63 cumulative degree days), an interaction between distance and number of days > threshold, and random effects for sites, taxa, and years.

[Values are means on logit scale, with 95-percent credible intervals in parentheses. The **bold** numbers indicate that there was an effect measured for the parameter (the credible interval did not include zero).]

Parameter	Persistence	Colonization
Mean	2.01 (0.69, 3.40)	−2.38 (−3.63, −1.12)
Distance from the dam	**0.78 (0.23, 1.52)**	−0.07 (−0.62, 0.48)
Annual number of days > degree day threshold	−0.24 (−1.20, 0.72)	**1.12 (0.07, 2.58)**
Number of days > threshold × distance	**−0.79 (−1.57, −0.17)**	0.16 (−0.72, 1.13)
Variance terms		
Among sites	0.19 (0.00, 0.77)	0.25 (0.00 1.10)
Among taxa	9.83 (3.67, 22.23)	6.44 (2.07, 16.87)
Among years	1.90 (0.20, 8.10)	2.15 (0.23, 8.51)

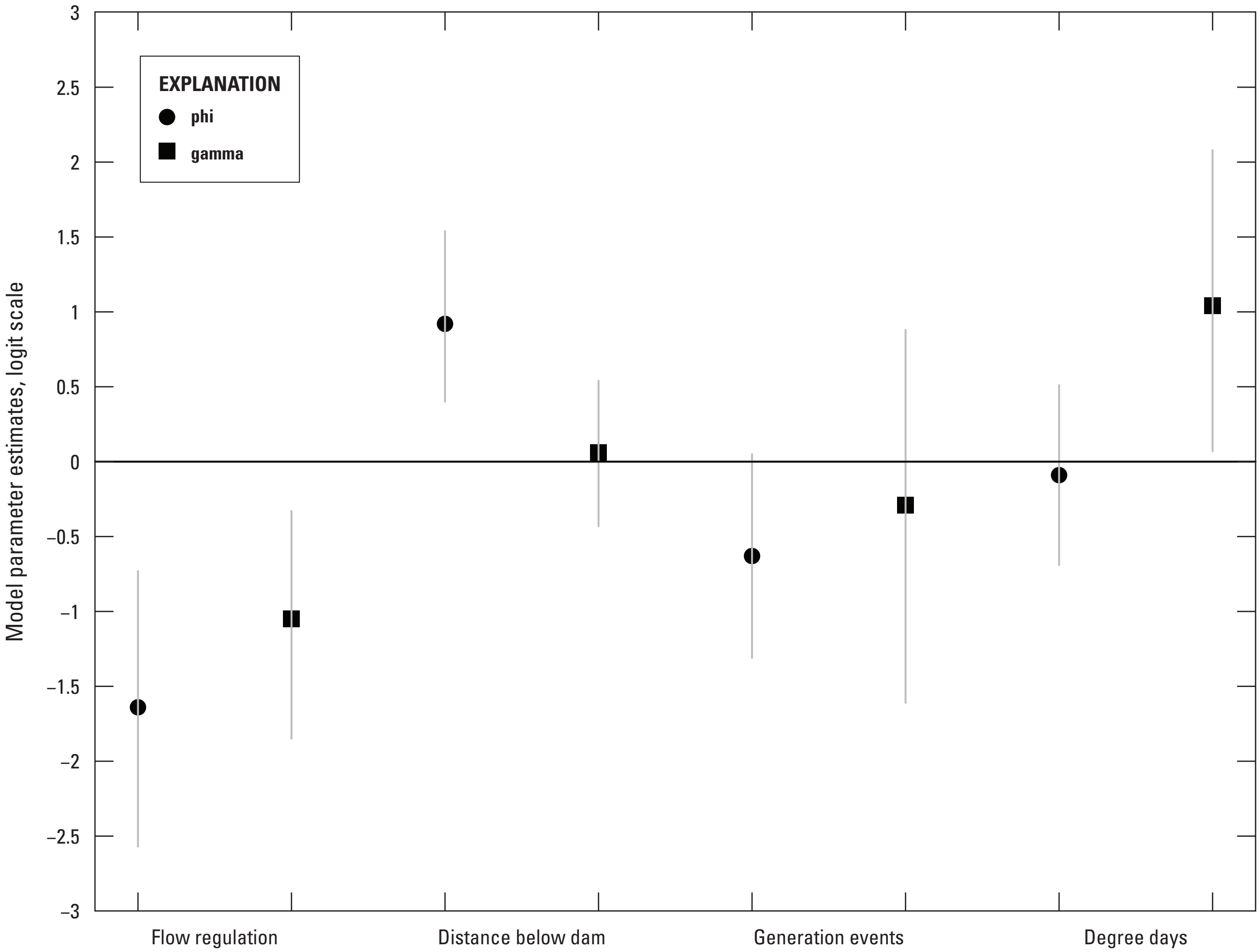

Figure B8. Estimated effects on flow regulation, distance downstream from R.L. Harris Dam, and annual frequency of hydropower generation (generation events) and cumulative time with temperatures meeting a threshold for spawning (degree days) on persistence and colonization for 38 fish species, modeled using dynamic, multitaxa occupancy models applied to data for 25 sites (flow regulation effect) or 15 sites (distance, generation, and degree days) in the Tallapoosa River system, 2005–16. Values are plotted on the logit scale and show 90-percent credible intervals. The horizontal line at 0 marks no effect; negative and positive values indicate, respectively, lower and higher persistence or colonization in relation to the covariate.

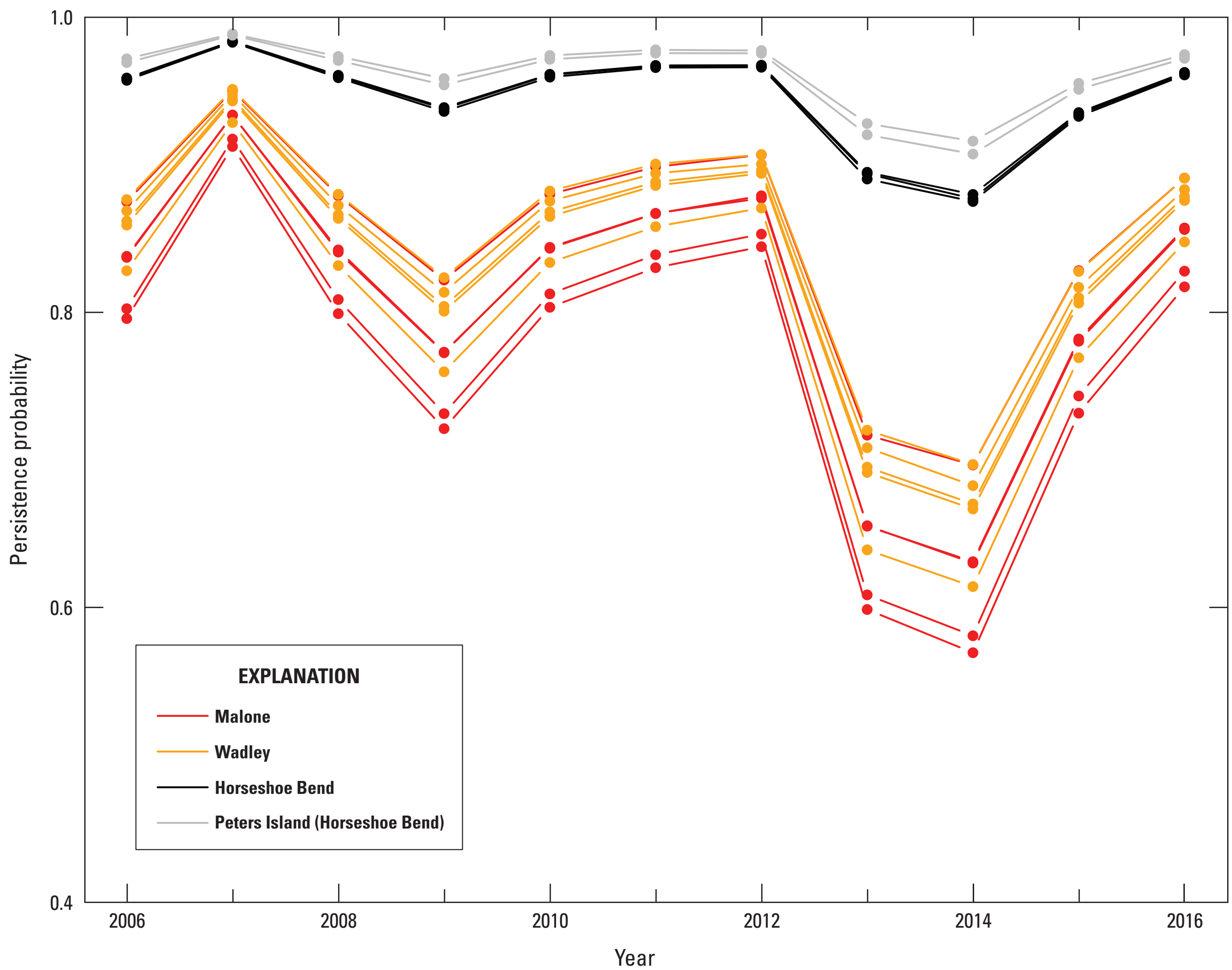

Figure B9. Estimated mean probability of persistence for 38 fish species at each of 15 flow-regulated sites in the Tallapoosa River system, 2005–16, modeled using dynamic, multitaxa occupancy models. Values for each year represent estimated mean species persistence from the previous year. [Sites farther from the dam are plotted in black (Horseshoe Bend) and gray (Peters Island); sites nearer the dam are plotted in red (Malone) and orange (Wadley).]

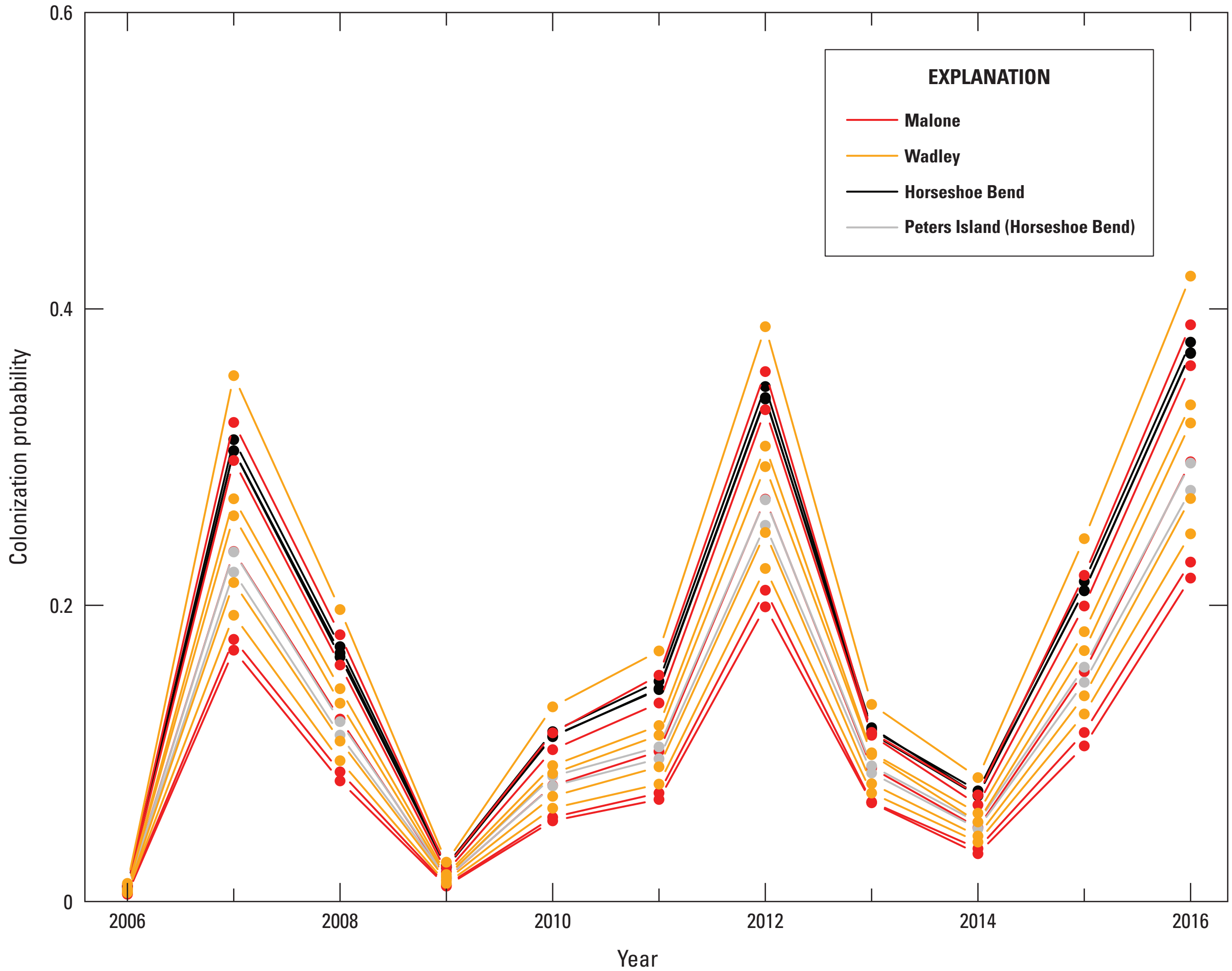

Figure B10. Estimated mean probability of colonization for 38 fish species at each of 15 flow-regulated sites in the Tallapoosa River system, 2005–16, modeled using dynamic, multitaxa occupancy models. Values for each year represent estimated mean species persistence from the previous year. [Sites farther from the dam are plotted in black (Horseshoe Bend) and gray (Peters Island); sites nearer the dam are plotted in red (Malone) and orange (Wadley).]

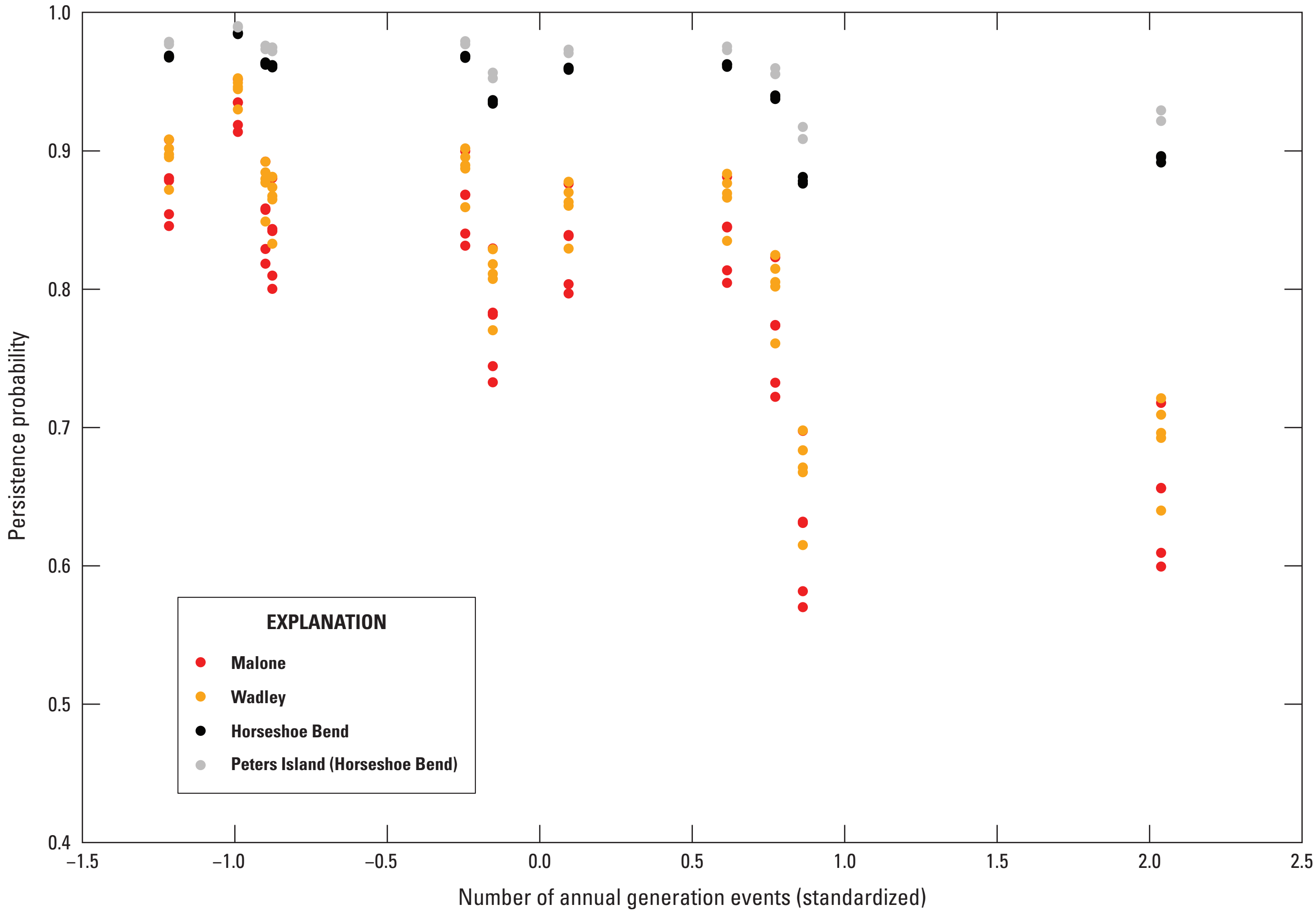

Figure B11. Estimated mean probability of persistence for 38 fish species at 15 flow-regulated sites in the Tallapoosa River system, 2005–16, modeled using dynamic, multitaxa occupancy models and plotted in relation to annual number of generation events (standardized to mean of 0 and standard deviation of 1). [Sites farther from the dam are plotted in black (Griffin Shoals) and gray (Peters Island); sites nearer the dam are plotted in red (Malone) and orange (Wadley).]

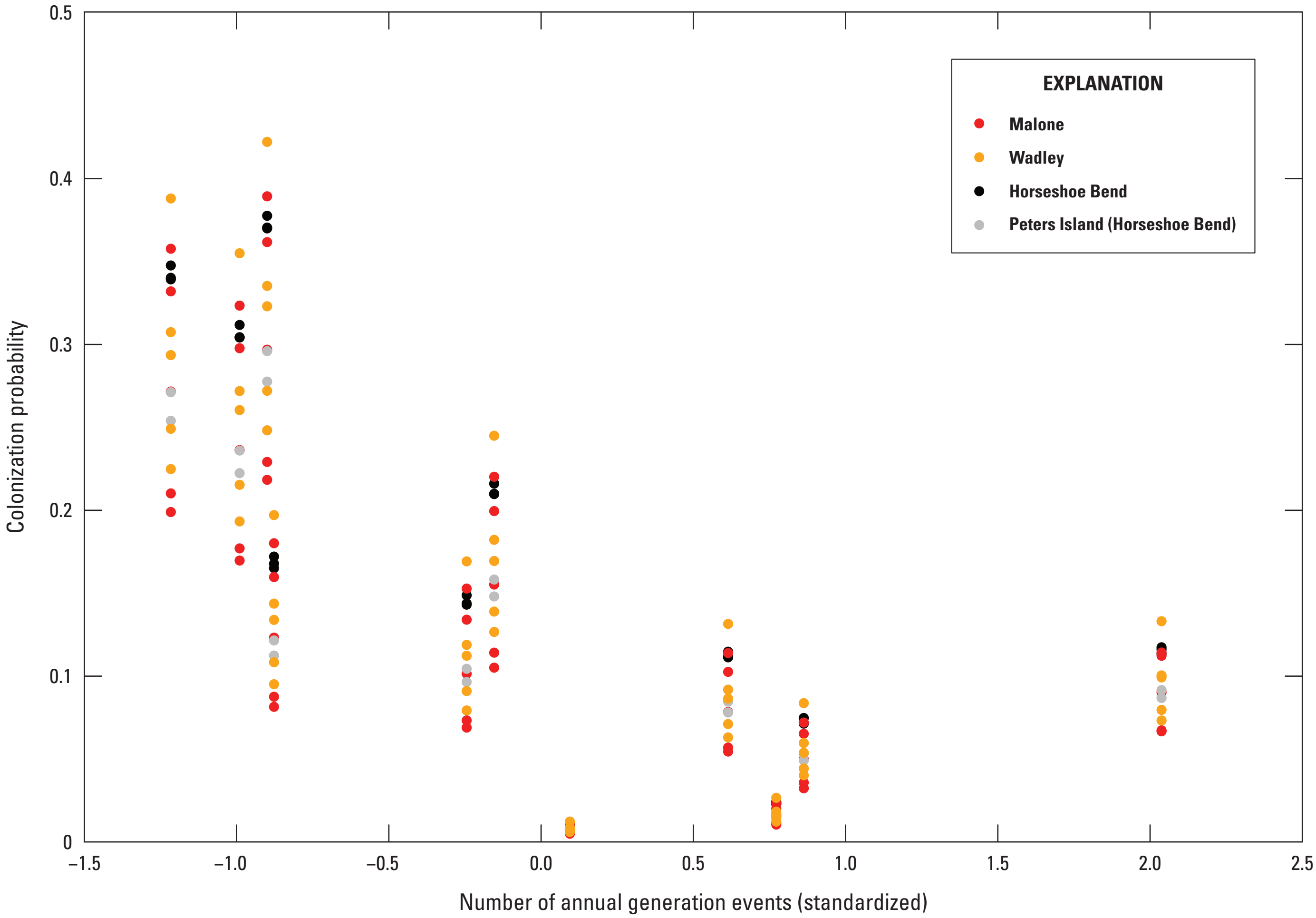

Figure B12. Estimated mean probability of colonization for 38 fish species at 15 flow-regulated sites in the Tallapoosa River system, 2005–16, modeled using dynamic, multitaxa occupancy models and plotted in relation to annual number of generation events (standardized to mean of 0 and standard deviation of 1). [Sites farther from the dam are plotted in black (Horseshoe Bend) and gray (Peters Island); sites nearer the dam are plotted in red (Malone) and orange (Wadley).]

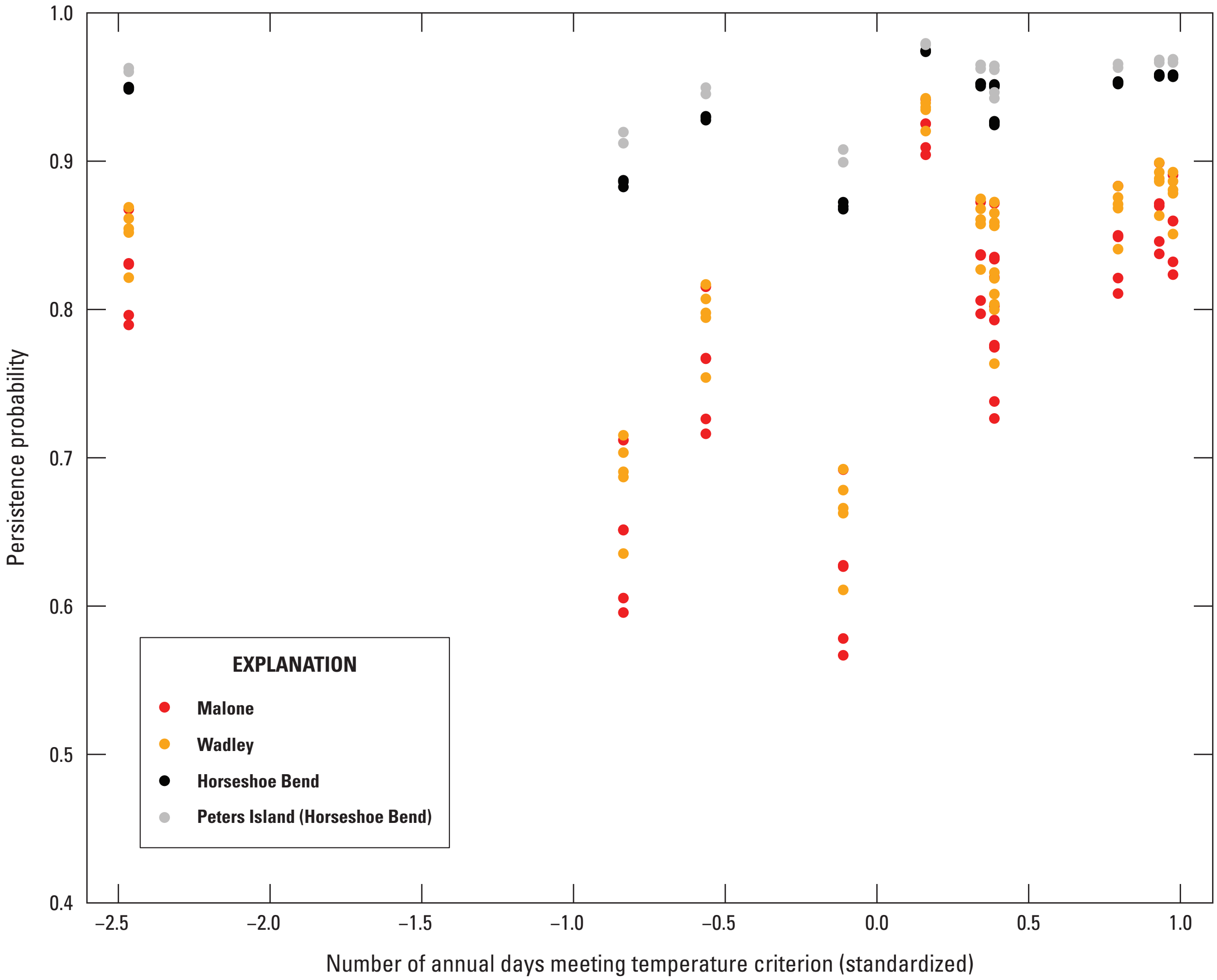

Figure B13. Estimated mean probability of persistence for 38 fish species at 15 flow-regulated sites in the Tallapoosa River system, 2005–16, modeled using dynamic, multitaxa occupancy models and plotted in relation to annual number of days meeting a temperature criterion for spawning (standardized to mean of 0 and standard deviation of 1). [Sites farther from the dam are plotted in black (Horseshoe Bend) and gray (Peters Island); sites nearer the dam are plotted in red (Malone) and orange (Wadley).]

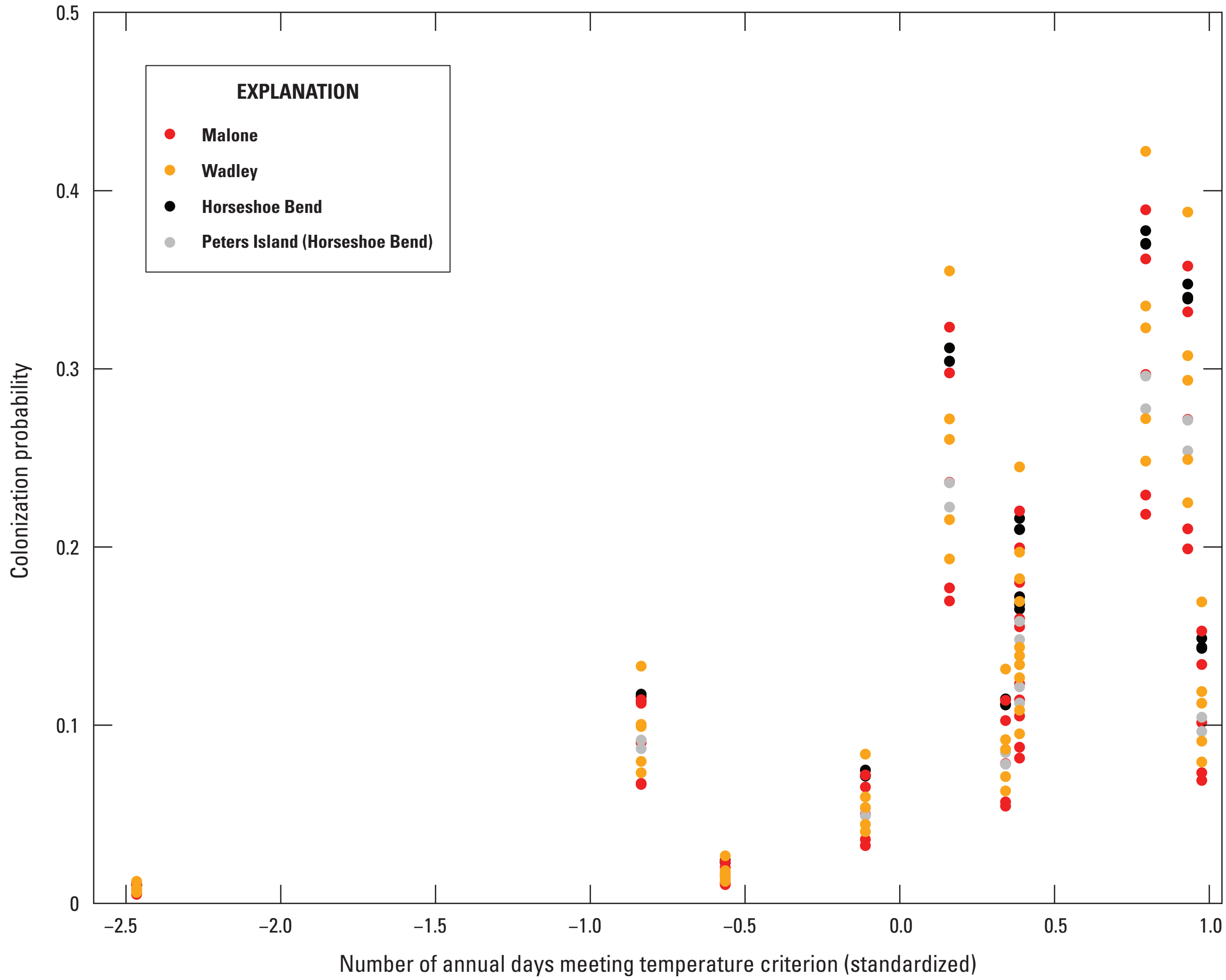

Figure B14. Estimated mean probability of colonization for 38 fish species at 15 flow-regulated sites in the Tallapoosa River system, 2005–16, modeled using dynamic, multitaxa occupancy models and plotted in relation to annual number of days meeting a temperature criterion for spawning (standardized to mean of 0 and standard deviation of 1). [Sites farther from the dam are plotted in black (Horseshoe Bend) and gray (Peters Island); sites nearer the dam are plotted in red (Malone) and orange (Wadley).]

Discussion

Maximizing conservation potential in free-flowing sections of rivers will require, at minimum, clear evidence for effects of the current and future alternative regulated flow regimes on river biota. Development of flow-ecology relations is a critical element in prescribing ecological flow regimes. Process rate approaches, such as the one used in our study, allow for a better understanding of underlying demographic mechanisms that define flow-ecology relations (Poff, 2017; Wheeler and others, 2018). The estimation of two demographic processes, persistence and colonization (that is, 1-extinction) probabilities, was robust and elucidated differences in vital rates in relation to river regulation, distance from the dam, and interannual differences in hydrology and thermal regimes. Monitoring of fauna over the 12-year study period has allowed us to quantify how metapopulation processes for many species responded to aspects of flow management in the Tallapoosa River downstream from R.L. Harris Dam. Although we observed increased density, persistence, and colonization along a downstream gradient from R.L. Harris Dam, richness did not differ appreciably at sites below the dam and all parameters were depressed relative the unregulated reaches. Additionally, densities below the dam were depressed relative to the unregulated sites, and usually, species contributed to richness values with less than five individuals over the course of the study. The alteration of fish diversity and abundance downstream from peaking hydroelectric dams has been reported for river systems, including our study site (Quinn and Kwak, 2003; Travnichek and Maceina, 1994). Kinsolving and Bain (1993) quantified downstream fish community changes below Thurlow Dam on the lower Tallapoosa River and reported that fluvial specialist populations were suppressed at sites nearest the dam prior to flow restoration. After increased base flows from the dam, fish species richness doubled and community composition shifted toward more fluvial specialists near the dam (Travnichek and others, 1995). In addition, positive temporal responses in persistence and colonization of fishes were not observed over the course of the study (see also Counihan and others, 2018; chapter A, this report), further indicating that the Green Plan flow management portfolio may not satisfy stakeholder objectives related to maximizing diversity and abundance of riverine fauna in the flow-regulated reaches.

Dams have altered flow and temperature regimes globally, potentially contributing to declines in abundance and early survival of fishes (Poff and others, 1997). Water released from hydroelectric dams is typically colder than surface temperatures, which alters downstream temperatures (Walker, 1985; Humphries and Lake, 2000). Although it has long been recognized that temperatures are altered below R.L. Harris Dam, specific inference regarding the influence on biotic processes has been lacking until this study, which clearly relates colonization rates (that is, recruitment of a species to a site) to increased thermal energy in the river. In addition, our data indicate that there is no downstream recovery for colonization processes such that colonization rates did not increase with distance from the dam. Conversely, persistence rates were higher at more downstream sites, indicating that instream forces that affect these metapopulation parameters may change variably with distance from the dam. Our state monitoring of juvenile fish density response to flow management at R.L. Harris Dam indicated that recruitment across species was low in most years (see chapter A, this report); however, findings were not consistent with predictions of positive responses related to increased base flows and periods of reduced generation to allow for spawning (chapter A, this report).

Abiotic factors (for example, physical habitat, physiological constraints) influence faunal distribution by changing population rates such as mortality or recruitment (Poff, 1997). Many studies have reported the negative influences of dams on recruitment and survival of early life-history stages of fishes caused by altered flow and temperature regimes (Poff and others, 1997; Connor and others, 2003 Clarkson and Childs, 2000; Humphries and Lake, 2000; Rolls and others, 2013). Matthews and others (1994) suggested that floods affected juvenile fish but not adults. In addition, short-term (hours, days) and long-term (seasonal) influences of river regulation are apparent in the ensuing flow (Poff and others, 1997) and thermal regimes (Caissie, 2006); local species distribution and biological responses to these regimes are also regulated by instream conditions (Poff, 1997; McManamay and Frimpong, 2015). Our data suggest that adults of most species can persist below R.L. Harris Dam; however, colonization rates may not result in growing populations under the Green Plan management regime (Rolls and others, 2013; chapter A).

Thermal regimes are extremely complex downstream from dams (Webb and Walling, 1997; Caissie, 2006) and are intrinsically related to discharge magnitude and duration and the thermal characteristics of the receiving water (that is, volume, tributary inflows; Toffolon and others, 2010). Despite the recognition that alterations of the hydrologic and thermal regimes are consequences of river regulation, the inherent relations between the two have received little attention in the assessment of environmental flows (Olden and Naiman, 2010; Poff, 2017). Downstream from R.L. Harris Dam, water temperatures have been measured to decrease as much as 10 °C during generation events (Irwin and Freeman, 2002). Thermal alteration below dams associated with dam releases have been implicated in lack of recruitment of fishes (Patton and Hubert, 1996; Rolls and others, 2013; Shea and others, 2015); however, changes in dam management have successfully mitigated for thermal effects to fishes (Shea and others, 2015). Rolls and others (2013) reported that the lack of recruitment of fish in their systems was related to the dual effect of colder temperatures and lack of prey availability because of prolonged increased base flow. They proposed that managing for low flow periods would increase recruitment of fish species (see also Freeman and others, 2001). Although we have not accounted for prey availability in this study, data regarding depressed growth rates of fishes (Nash and Irwin, 1999; Sakaris and others, 2006), growth abnormalities expressed on otoliths (Irwin and others, 1997; Goar, 2013), low abundance of forage fish (Freeman and others, 2001; Counihan and others, 2018; this study), and altered invertebrate communities (chapter C) have been quantified below R.L. Harris Dam relative to unregulated river reaches in the basin. Based on these studies, investigations of flow regimes and temperature on fish forage (abundance and availability) are warranted below R.L. Harris Dam.

Ecological responses of fishes to flow alteration are overwhelmingly negative with respect to diversity, abundance, or population life-history parameters (Poff and Zimmerman, 2010) and vital rates of metapopulations (this study; Shea and others, 2015). McManamay and others (2013, p. 30) reported that as little as 10-percent change in flow could result in “very large ecological responses.” However, theoretical underpinnings for development of flow-ecology hypotheses that can explicitly inform management are lacking (Poff and Zimmerman, 2010; Poff, 2017). Our long-term monitoring data are a step toward informing management at R.L. Harris Dam; however, the full range of flow alteration below the dam was not experienced during the study—flood and wet years were underrepresented and dry and drought years were more common during the study period. Freeman and others (2001) hypothesized that periods of lower flow consistent with droughts and years where inflows are low in the fall were conducive to recruitment of a suite of common fishes. In the current study, warmer thermal conditions increased colonization probability downstream from R.L. Harris Dam in some years; however, the overall probability of persistence and colonization was negatively affected by river regulation for almost all species. Analysis of temperature time-series data from below R.L. Harris Dam indicated poor thermal conditions for conspecifics of many warm-water species of fish below R.L. Harris Dam (Irwin and Freeman, 2002). Because of engineering constraints at the dam, it is unlikely that thermal conditions could be improved under any flow regime provided by pulsing at the dam to provide enhanced flows between generation events (for example, the Green Plan). Therefore, any investigation of thermal effects on fishes would need to be carefully considered relative to engineering constraints and the ability of the utility to provide for additional experimental flow portfolios below the dam. Toffolon and others (2010) assessed the thermal and hydrologic wave dynamics downstream from R.L. Harris Dam and inferred that the thermal and discharge waves did not decouple in the reach from below the dam to the headwaters of Lake Martin. Further analysis of complex hydrothermal data under different flow management regimes below R.L. Harris Dam is warranted. Olden and Naiman (2010) proposed that by viewing environmental flows together with various aspects of water quality (including temperature), the chances of long-term success in achieving ecologically sustainable water management are increased.

Disruption of riverine flows from impoundment and river regulation are primary reasons for high levels of imperilment of fishes and mollusks in Alabama and throughout the southeast. The primary cause of major extinction events for mollusks (mussels and gastropods) was the impoundment of shoal and riffle habitat (Neves and others, 1997). Similarly, 53 percent of fishes inhabiting medium-sized rivers and creeks in the southeast United States are in jeopardy (Etnier, 1997). This is likely because fish communities in these habitats are dependent on shoals and riffles for at least part of their life history (Etnier, 1997). One of the objectives of adaptive flow management from R.L. Harris Dam was to maximize the diversity of native fauna and flora downstream from the dam, and hypothesized responses of the Green Plan related increased shoal habitat persistence, decreased magnitude of disturbance, and increased flow stability (associated with spawning windows; Martin, 2008) to positive fish population responses. Irwin and others (2011) reported an increase in shoal habitat persistence associated with the Green Plan; however, positive population responses have not ensued (see chapter A). Bradford and others (2011) reported similar findings for a long-term study of Pacific salmon in British Columbia; despite predictions of increased fish abundance based on improved habitat conditions (habitat did improve), fish populations in terms of density did not respond to flow manipulation. Our long-term metapopulation data provide evidence that suggests broadscale negative influences of the dam on species persistence and colonization parameters. Specifically, generation frequency and cool thermal regimes negatively affected fish persistence and colonization, respectively. Shea and others (2015) reported similar findings for a suite of darters in the Elk River, Tennessee. In the case of the Elk River, river temperatures were increased by limiting or ceasing generation at the dam during spring and summer periods.

Historical and current data from below R.L. Harris Dam indicate that for many species, occupancy rates below R.L. Harris Dam have remained at low levels despite the continued persistence of species (Irwin and others, 2011; Kennedy, 2015; this study). Counihan and others (2018) analyzed differences in assemblage structure among most of the historical reaches used in this study and reported substantial differences among sites (that is, Upper Tallapoosa reach and the reaches below the dam). They reported that differences were greatest between the Upper Tallapoosa and the Malone reach (58.2 percent) and noted that Bullhead Minnow, Largescale Stoneroller, Alabama Shiner, and Speckled Darter each contributed greater than 10 percent of the dissimilarity between reaches. In addition, the two most abundant fishes, Lipstick Darter and Bronze Darter, contributed greater than 10 percent to dissimilarities between assemblages below the dam. They reported no temporal component to assemblage structure during the time series they analyzed (2005–13). However, historical fish assemblage structure has changed most likely in response to river regulation (from predam Ictalurid/Cyprinid dominant to postdam Centrarchid dominant; Swingle, 1954; E.R. Irwin, unpublished data). More specific analysis of temporal series of species-specific occupancy data may elucidate responses to interannual differences in flow and thermal regimes. In addition, species trait analysis may provide insights into predicted responses of species with similar life-history strategies to proposed future management (Freeman and others, 2001; Craven and others, 2010; McManamay and Frimpong, 2015; Poff, 2017).

Fish assemblage diversity presents a challenge for management of flow-regulated systems; therefore, further identification of the biotic and abiotic factors that influence demographic rates of these species may benefit conservation efforts below dams. Most studies of biotic responses to flow management are focused on a single species (salmonids) or a few species (for example, Shea and others, 2015) and fewer have been cited as “successful” (Poff, 2017), but when considering regulation and thermal impacts to the number of species covered in this report, the future flow and thermal prescription could likely be extremely complex and trend toward “designer flows” (Acreman and others, 2014). Recent publications have been realistic regarding the feasibility of this approach (Poff, 2017; Acreman and others, 2014); however, the use of decision frameworks and adaptive management with careful monitoring for learning and reducing uncertainty is recognized as the best way to resolve system complexities. If adaptive management is not an option, McManamay and others (2016) proposed a framework that focuses on modeling the consequences of flow management on multiple objectives in a proactive way to elucidate regulatory engineering and biological constraints in FERC licensing procedures.

When Irwin and Freeman (2002) described an iterative structured process for reducing uncertainty and learning how carefully imposed management influenced fish populations, they were unaware of any FERC regulated projects with adaptive management formally defined in the license. Since then, several FERC relicensing processes and subsequent licenses have adaptive management programs tied to the license to better understand how natural resources and other stakeholder values respond to the management of dams (see Pearsall and others, 2005; Jacobson and Galat, 2008; Cross and others, 2011; Podolak and Yarnell, 2015). Despite potential obstacles (Irwin and Freeman, 2002), an adaptive management approach holds substantial promise for improving management of regulated rivers by allowing managers and scientists to address the uncertainty in predicting and measuring how river fauna will respond to flow-regime alterations (Poff, 2017). Despite long-term monitoring of fishes in adaptive management programs, high levels of uncertainty regarding the ability to predict species-specific population response to management remain (Bradford and others, 2011; this study). However, the recognition of uncertainty and use of decision-analytic processes to reduce uncertainty attributed to flow management decisions with model updating through careful, replicated monitoring programs remains a viable path forward (Rolls and others, 2013). Although monitoring data were used to assess stakeholder objectives using a decision model (chapter A, this report), the additional data provided in this chapter can further be used to identify additional changes in dam operations to evaluate during relicensing and beyond. If flow and thermal alteration from the dam can be modified toward improving natural resource objectives, adaptive management processes and long-term monitoring could further reduce uncertainty related to biotic response to new FERC licensing requirements.

References Cited

Acreman, M., Arthington, A.H., Colloff, M.J., Couch, C., Crossman, N.D., Dyer, F., Overton, I., Pollino, C.A., Stewardson, M.J., and Young, W., 2014, Environmental flows for natural, hybrid, and novel riverine ecosystems in a changing world: Frontiers in Ecology and the Environment, v. 12, no. 8, p. 466–473. [Also available at https://doi.org/10.1890/130134.]

Andress, R.O., 2002, Nest survival of *Lepomis* species in regulated and unregulated rivers: Auburn University, Auburn, Ala., master’s thesis, 47 p.

Bain, M.B., Finn, J.T., and Booke, H.E., 1985, A quantitative method for sampling riverine microhabitats by electrofishing: North American Journal of Fisheries Management, v. 5, no. 3B, p. 489–493. [Also available at https://doi.org/10.1577/1548-8659(1985)5<489:AQMFSR>2.0.CO;2.]

Boschung, H.T., and Mayden, R.L., 2004, Fishes of Alabama: Washington, D.C., Smithsonian Books, 960 p.

Bowen, Z.H., Freeman, M.C., and Bovee, K.D., 1998, Evaluation of generalized habitat criteria for assessing impacts of altered flow regimes on warmwater fishes: Transactions of the American Fisheries Society, v. 127, no. 3, p. 455–468. [Also available at https://doi.org/10.1577/1548-8659(1998)127<0455:EOGHCF>2.0.CO;2.]

Bradford, M.J., Higgins, P.S., Korman, J., and Sneep, J., 2011, Test of an environmental flow release in a British Columbia river—Does more water mean more fish?: Freshwater Biology, v. 56, no. 10, p. 2119–2134. [Also available at https://doi.org/10.1111/j.1365-2427.2011.02633.x.]

Caissie, D., 2006, The thermal regime of rivers—A review: Freshwater Biology, v. 51, no. 8, p. 1389–1406. [Also available at https://doi.org/10.1111/j.1365-2427.2006.01597.x.]

Clarkson, R.W., and Childs, M.R., 2000, Temperature effects of hypolimnial-release dams on early life stages of Colorado River Basin big-river fishes: Copeia, v. 2000, no. 2, p. 402–412. [Also available at https://www.researchgate.net/publication/247708261_Temperature_Effects_of_Hypolimnial-Release_Dams_on_Early_Life_Stages_of_Colorado_River_Basin_Big-River_Fishes.]

Connor, W.P., Burge, H.L., Yearsley, J.R., and Bjornn, T.C., 2003, Influence of flow and temperature on survival of wild subyearling fall chinook salmon in the Snake River: North American Journal of Fisheries Management, v. 23, no. 2, p. 362–375. [Also available at https://doi.org/10.1577/1548-8675(2003)023<0362:IOFATO>2.0.CO;2.]

Counihan, T.D., Waite, I.R., Casper, A.F., Ward, D.L., Sauer, J.S., Irwin, E.R., Chapman, C.G., Ickes, B.S., Paukert, C.P., Kosovich, J.J., and Bayer, J.M., 2018, Can data from disparate long-term fish monitoring programs be used to increase our understanding of regional and continental trends in large river assemblages?: PLoS One, v. 13, no. 1, article e0191472, accessed March 2018 at https://doi.org/10.1371/journal.pone.0191472.

Craven, S.W., Peterson, J.T., Freeman, M.C., Kwak, T.J., and Irwin, E.R., 2010, Modeling the relations between flow regime components, species traits, and spawning success of fishes in warmwater streams: Environmental Management, v. 46, no. 2, p. 181–194. [Also available at https://doi.org/10.1007/s00267-010-9511-5.]

Cross, W.F., Baxter, C.V., Donner, K.C., Rosi-Marshall, E.J., Kennedy, T.A., Hall, R.O., Jr., Wellard Kelly, H.A., and Rogers, R.S., 2011, Ecosystem ecology meets adaptive management—Food web response to a controlled flood on the Colorado River, Glen Canyon: Ecological Applications, v. 21, no. 6, p. 2016–2033. [Also available at https://doi.org/10.1890/10-1719.1.]

Etnier, D.A., 1997, Jeopardized southeastern freshwater fishes—A search for causes, *in* Benz, G.W., and Collins, D.E., eds., Aquatic fauna in peril—The southeastern perspective: Decatur, Ga., Lenz Design and Communications, p. 87–92.

Freeman, M.C., Bowen, Z.H., Bovee, K.D., and Irwin, E.R., 2001, Flow and habitat effects on juvenile fish abundance in natural and altered flow regimes: Ecological Applications, v. 11, no. 1, p. 179–190. [Also available at https://doi.org/10.1890/1051-0761(2001)011[0179:FAHEOJ]2.0.CO;2.]

Freeman, M.C., Hagler, M.M., Bumpers, P.M., Wheeler, K., Wenger, S.J., and Freeman, B.J., 2017, Long-term monitoring data provide evidence of declining species richness in a river valued for biodiversity conservation: Journal of Fish and Wildlife Management, v. 8, no. 2, p. 418–435. [Also available at https://doi.org/10.3996/122016-JFWM-090.]

Goar, T.P., 2013, Effects of hydrologic variation and water temperatures on early growth and survival of selected age-0 fishes in the Tallapoosa River, Alabama: Auburn, Ala., Auburn University, Ph.D. dissertation, 91 p.

Hanski, I.A., and Simberloff, D., 1997, Metapopulation biology—Ecology, genetics, and evolution: Cambridge, Mass., Academic Press, 512 p.

Humphries, P.L., and Lake, P., 2000, Fish larvae and the management of regulated rivers: River Research and Applications, v. 16, no. 5, p. 421–432. [Also available at https://doi.org/10.1002/1099-1646(200009/10)16:5<421::AID-RRR594>3.0.CO;2-4.]

Irwin, E.R., 2014, Defining ecological and economical hydropower operations—A framework for managing dam releases to meet multiple conflicting objectives: Journal of Energy Challenges and Mechanics, v. 1, no. 3, p. 139–146.

Irwin, E.R., and Freeman, M.C., 2002, Proposal for adaptive management to conserve biotic integrity in a regulated segment of the Tallapoosa River, Alabama, U.S.A: Conservation Biology, v. 16, no. 5, p. 1212–1222. [Also available at https://doi.org/10.1046/j.1523-1739.2002.00514.x.]

Irwin, E.R., Freeman, M.C., and Isely, J.J., 1997, Effects of flow regime on recruitment and growth of juvenile basses (*Micropterus* spp.) in flow-regulated rivers: Arlington, Va., U.S. Fish and Wildlife Service, 32 p.

Irwin, E.R., Kennedy, K.M., Goar, T.P., Martin, B., and Martin, M.M., 2011, Adaptive management and monitoring for restoration and faunal recolonization of Tallapoosa River shoal habitats: Montgomery, Ala., Alabama Department of Conservation and Natural Resources, ALCFWRU Report 2011–1, 49 p.

Jacobson, R.B., and Galat, D.L., 2008, Design of a naturalized flow regime—An example from the lower Missouri River, USA: Ecohydrology, v. 1, no. 2, p. 81–104. [Also available at https://doi.org/10.1002/eco.9.]

Kennedy, K.D.M., 2015, Adaptive management and decision support analysis of resources associated with hydropower facilities: Auburn, Ala., Auburn University, Ph.D. dissertation, 204 p.

Kennedy, K.D., Irwin, E.R., Freeman, M.C., and Peterson, J., 2006, Development of a decision support tool and procedures for evaluating dam operation in the Southeastern United States: Atlanta, Ga., Science Support Partnership Program Project, Final Report to USGS and USFWS, 70 p.

Kéry, M., and Schaub, M., 2012, Bayesian population analysis using WinBUGS—A hierarchical perspective: Cambridge, Mass., Academic Press, 554 p.

Kinsolving, A.D., and Bain, M.B., 1993, Fish assemblage recovery along a riverine disturbance gradient: Ecological Applications, v. 3, no. 3, p. 531–534. [Also available at https://doi.org/10.2307/1941921.]

Lydeard, C., and Mayden, R.L., 1995, A diverse and endangered aquatic ecosystem of the southeast United States: Conservation Biology, v. 9, no. 4, p. 800–805. [Also available at https://doi.org/10.1046/j.1523-1739.1995.09040800.x.]

MacKenzie, D.I., Nichols, J.D., Hines, J.E., Knutson, M.G., and Franklin, A.B., 2003, Estimating site occupancy, colonization, and local extinction when a species is detected imperfectly: Ecology, v. 84, no. 8, p. 2200–2207. [Also available at https://doi.org/10.1890/02-3090.]

Martin, B.M., 2008, Nest survival, nesting behavior, and bioenergetics of redbreast sunfish on the Tallapoosa River, Alabama: Auburn, Ala., Auburn University, master's thesis, 91 p.

Matthews, W.J., Harvey, B.C., and Power, M.E., 1994, Spatial and temporal patterns in the fish assemblages of individual pools in a midwestern stream (U.S.A.): Environmental Biology of Fishes, v. 39, no. 4, p. 381–397.

McGowan, C.P., Smith, D., Sweka, J.A., Martin, J., Nichols, J.D., Wong, R., Lyons, J.E., Niles, L.J., Kalasz, K., Brust, J., Klopfer, M., and Spear, B., 2011, Multispecies modeling for adaptive management of horseshoe crabs and red knots in the Delaware Bay: Natural Resource Modeling, v. 24, no. 1, p. 117–156. [Also available at https://doi.org/10.1111/j.1939-7445.2010.00085.x.]

McManamay, R.A., Brewer, S.K., Jager, H.I., and Troia, M.J., 2016, Organizing environmental flow frameworks to meet hydropower mitigations needs: Environmental Management, v. 58, no. 3, p. 365–385. [Also available at https://doi.org/10.1007/s00267-016-0726-y.]

McManamay, R.A., and Frimpong, E.A., 2015, Hydrologic filtering of fish life history strategies across the United States—Implications for stream flow alteration: Ecological Applications, v. 25, no. 1, p. 243–263. [Also available at https://doi.org/10.1890/14-0247.1.]

McManamay, R.A., Orth, D.J., Kauffman, J., and Davis, M.D., 2013, A database and meta-analysis of ecological responses to flow in the South Atlantic Region: Southeastern Naturalist, v. 12, no. 5, p. 1–36. [Also available at https://www.researchgate.net/publication/255814051_A_Database_and_Meta-Analysis_of_Ecological_Responses_to_Stream_Flow_in_the_South_Atlantic_Region.]

Mettee, M.F., O'Neil, P.E., and Pierson, J.M., 1996, Fishes of Alabama and the Mobile Basin: Birmingham, Ala., Oxmoor House, 820 p.

Nash, M.K., and Irwin, E.R., 1999, Use of otoliths versus pectoral spines for aging adult flathead catfish, *in* Irwin, E.R., Hubert, W., Rabeni, C., Schramm, H., and Coon, T.G., eds., Catfish 2000—Proceedings of the 1st International Ictalurid Symposium: Bethesda, Md., American Fisheries Society Symposium 24, p. 309–316.

National Research Council, 2004, Adaptive management for water resources project planning: Washington, D.C., National Academies Press, 137 p.

Neves, R.J., Bogan, A.E., Williams, J.D., Ahlstedt, S.A., and Hartfield, P.D., 1997, Status of aquatic mollusks in the southeastern United States—A downward spiral of diversity, *in* Benz, G.W., and Collins, D.E., eds., Aquatic fauna in peril—The southeastern perspective: Decatur, Ga., Southeast Aquatic Research Institute, Special Publication no. 1, p. 43–85.

Olden, J.D., and Naiman, R.J., 2010, Incorporating thermal regimes into environmental flows assessments—Modifying dam operations to restore freshwater ecosystem integrity: Freshwater Biology, v. 55, no. 1, p. 86–107. [Also available at https://doi.org/10.1111/j.1365-2427.2009.02179.x.]

Patton, T.M., and Hubert, W.A., 1996, Water temperature affects smallmouth bass and channel catfish in a tailwater stream on the Great Plains: North American Journal of Fisheries Management, v. 16, no. 1, p. 124–131. [Also available at https://doi.org/10.1577/1548-8675(1996)016<0124:WTASBA>2.3.CO;2.]

Pearsall, S.H., McCrodden, B.J., and Townsend, P.A., 2005, Adaptive management of flows in the lower Roanoke River, North Carolina, USA: Environmental Management, v. 35, no. 4, p. 353–367.

Peterson, J.T., and Shea, C.P., 2015, An evaluation of the relations between flow regime components, stream characteristics, species traits, and meta-demographic rates of warm-water stream fishes—Implications for aquatic resource management: River Research and Applications, v. 31, no. 10, p. 1227–1241. [Also available at https://doi.org/10.1002/rra.2835.]

Plummer, M., 2003, JAGS—A program for analysis of Bayesian graphical models using Gibbs sampling, *in* Hornik, K., Leisch, F., and Zeileis, A., eds., Proceedings of the 3d International Workshop on Distributed Statistical Computing: Vienna, Austria, DSC 2003, 10 p.

Plummer, M., 2014, rjags—Bayesian graphical models using MCMC: R web page, R package version 3–14, accessed November 2017 at https://rdrr.io/cran/rjags/.

Podolak, K., and Yarnell, S., 2015, Adaptive management in Federal Energy Regulatory Commission relicensing, chap. 13 *of* Lassiter, A., ed., Sustainable water—Challenges and solutions from California: Berkley, Calif., University of California Press, p. 292–316.

Poff, N.L., 1997, Landscape filters and species traits—Towards mechanistic understanding and prediction in stream ecology: Journal of the North American Benthological Society, v. 16, no. 2, p. 391–409. [Also available at https://doi.org/10.2307/1468026.]

Poff, N.L., 2017, Beyond the natural flow regime? Broadening the hydro-ecological foundation to meet environmental flows challenges in a non-stationary world: Freshwater Biology, v. 63, no. 8, p. 1011–1021. [Also available at https://doi.org/10.1111/fwb.13038.]

Poff, N.L., Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., Sparks, R.E., and Stromberg, J.C., 1997, The natural flow regime—A paradigm for river conservation and preservation: Bioscience, v. 47, no. 11, p. 769–784. [Also available at https://doi.org/10.2307/1313099.]

Poff, N.L., and Zimmerman, J.K.H., 2010, Ecological responses to altered flow regimes—A literature review to inform the science and management of environmental flows: Freshwater Biology, v. 55, no. 1, p. 194–205. [Also available at https://doi.org/10.1111/j.1365-2427.2009.02272.x.]

Quinn, J.W., and Kwak, T.J., 2003, Fish assemblage changes in an Ozark River after impoundment—A long-term perspective: Transactions of the American Fisheries Society, v. 132, no. 1, p. 110–119. [Also available at https://doi.org/10.1577/1548-8659(2003)132<0110:FACIAO>2.0.CO;2.]

R Core Team, 2014, R—A language and environment for statistical computing: Vienna, Austria, R Foundation for Statistical Computing, accessed November 2017 at https://cran.r-project.org.

Rolls, R.J., Growns, I.O., Khan, T.A., Wilson, G.G., Ellison, T.L., Prior, A., and Waring, C.C., 2013, Fish recruitment in rivers with modified discharge depends on the interacting effects of flow and thermal regimes: Freshwater Biology, v. 58, no. 9, p. 1804–1819. [Also available at https://doi.org/10.1111/fwb.12169.]

Royle, J.A., and Kéry, M., 2007, A Bayesian state-space formulation of dynamic occupancy models: Ecology, v. 88, no. 7, p. 1813–1823. [Also available at https://doi.org/10.1890/06-0669.1.]

Sakaris, P.C., Irwin, E.R., Jolley, J.C., and Harrison, D., 2006, Comparison of native and introduced flathead catfish populations in Alabama and Georgia—Growth, mortality, and management: North American Journal of Fisheries Management, v. 26, no. 4, p. 867–874. [Also available at https://doi.org/10.1577/M05-135.1.]

Shea, C.P., Bettoli, P.W., Potoka, K.M., Saylor, C.F., and Shute, P.W., 2015, Use of dynamic occupancy models to assess the response of darters (Teleostei: Percidae) to varying hydrothermal conditions in a southeastern United States tailwater: River Research and Applications, v. 31, no. 6, p. 676–691. [Also available at https://doi.org/10.1002/rra.2766.]

Swingle, H.S., 1954, Fish populations in Alabama Rivers and impoundments: Transactions of the American Fisheries Society, v. 83, no. 1, p. 47–57. [Also available at https://doi.org/10.1577/1548-8659(1953)83[47:FPIARA]2.0.CO;2.]

Toffolon, M., Siviglia, A., and Zolezzi, G., 2010, Thermal wave dynamics in rivers affected by hydropeaking: Water Resources Research, v. 46, no. 8, article W08536, accessed November 2017 at https://doi.org/10.1029/2009WR008234.

Travnichek, V.H., Bain, M.B., and Maceina, M.J., 1995, Recovery of a warmwater fish assemblage after the initiation of a minimum-flow release downstream from a hydroelectric dam: Transactions of the American Fisheries Society, v. 124, no. 6, p. 836–844. [Also available at https://doi.org/10.1577/1548-8659(1995)124<0836:ROAWFA>2.3.CO;2.]

Travnichek, V.H., and Maceina, M.J., 1994, Comparison of flow regulation effects on fish assemblages in shallow and deep water habitats in the Tallapoosa River, Alabama: Journal of Freshwater Ecology, v. 9, no. 3, p. 207–216. [Also available at https://doi.org/10.1080/02705060.1994.9664888.]

U.S. Geological Survey, 2018, USGS water data for the Nation: U.S. Geological Survey National Water Information System database, accessed May 2018 at https://doi.org/10.5066/F7P55KJN.

Walker, K.F., 1985, A review of the ecological effects of river regulation in Australia: Hydrobiologia, v. 125, no. 1, p. 111–129. [Also available at https://doi.org/10.1007/BF00045929.]

Webb, B.W., and Walling, D.E., 1997, Complex summer water temperature behaviour below a UK regulating reservoir: River Research and Application, v. 13, no. 5, p. 463–477. [Also available at https://doi.org/10.1002/(SICI)1099-1646(199709/10)13:5<463::AID-RRR470>3.0.CO;2-1.]

Wheeler, K., Wenger, S.J., and Freeman, M.C., 2018, States and rates—Complementary approaches to developing flow-ecology relationships: Freshwater Biology, v. 63, no. 8, p. 906–916. [Also available at https://doi.org/10.1111/fwb.13001.]

Williams, B.K., and Brown, E.D., 2012, Adaptive management—The U.S. Department of the Interior Applications Guide: Washington D.C., U.S. Department of the Interior, Adaptive Management Working Group, 136 p.

Williams, B.K., and Brown, E.D., 2014, Adaptive management—From more talk to real action: Environmental Management, v. 53, no. 2, p. 465–479. [Also available at https://doi.org/10.1007/s00267-013-0205-7.]

Wood, T., 2015, Alabama's Wildlife Action Plan 2015–2025: Montgomery, Ala., Alabama Department of Conservation and Natural Resources, 491 p.

Appendix B

This appendix includes the R code used to conduct metapopulation analyses; it is separated into code specific to the two objectives described in the text. The R code can be downloaded from https://doi.org/10.3133/ofr20191026.

Sampling crew capturing fishes downstream from R.L. Harris Dam on the Tallapoosa River. Taken July 2, 2011, by E. Broder, Alabama Cooperative Fish and Wildlife Research Unit.

Chapter C

Macroinvertebrate Community Structure in Relation to Variation in Hydrology Associated with Hydropower

By Kristie M. Ouellette Coffman,[1] Ely Kosnicki,[1] M. Clint Lloyd,[1] Tom Hess,[1] and Elise R. Irwin[1,2]

Introduction

Hydropeaking dams can provide many valuable anthropogenic services, including flood and drought control, and the generation of electricity; however, hydrologic alteration from the daily operation of hydroelectric dams affects virtually every aspect of the riverine habitat because of changes to the natural flow and thermal regime. In general, river regulation changes the seasonal flow and temperature patterns of naturally flowing rivers by causing an increase in the number and frequency of flow events, an increase in the magnitude of normal daily thermal fluctuations, and a decrease in the variability and magnitude of flows (Graf, 2006; McMahon and Finlayson, 2003; Olden and Naiman, 2010). For hydropeaking dams, a dual shock of hydraulic sheer stress followed by thermal stress can be experienced by the downstream habitats, which can potentially take days to fully recover (Toffolon and others, 2010). Additionally, the availability and persistence of habitat area can be greatly altered by the daily scouring of substrate by hydropeaking releases, an overall reduction of median flow, and a general decrease in habitat heterogeneity (Freeman and others, 2001; Rehn, 2009; Tupinambás and others, 2016).

Over the last several decades the published literature has well documented the effects of changes in hydrology on the downstream communities, including aquatic macrophyte, macroinvertebrate, and fish communities (Bejarano and others, 2017; Poff and Zimmerman, 2010). Since the emergence of the River Continuum Concept (Vannote and others, 1980), the response of macroinvertebrate communities to changes in the environment have been well documented in the literature (Wallace and Anderson, 1996; Barbour and others, 1999; Poff and others, 2006). Additionally, with the development of the Serial Discontinuity Concept (Ward and Stanford, 1982), researchers began observing changes in the macroinvertebrate community because of anthropogenic effects on the natural ecosystem, including the effects of dams and reservoirs on downstream communities (Poff and Zimmerman, 2010). Whether it be through community analysis (for example, taxa; Rader and Ward, 1988; Holt and others, 2015), physiological traits (for example, functional feeding group [FFG] and habit; Kennedy and others, 2016; Tupinambás and others, 2016; White and others, 2017), or quantitative sensitivity metrics (for example, Index of Biotic Integrity; Gore and others, 2001; Rehn, 2009), there is a growing database of association of macroinvertebrate community changes with altered flow regimes.

Macroinvertebrates are useful for detecting immediate (minutes to days) and long-term (weeks to years) effects. Macroinvertebrates demonstrate sensitivity to changes in water chemistry and flow conditions by initiating drift upon reaching a tolerance threshold or leaving the habitat immediately by intentionally entering the water column and floating downstream from the disturbance (Wallace and Anderson, 1996). Additionally, many macroinvertebrates are sessile or weak swimmers that are unable to migrate back upstream immediately after initiating drift, so recolonization can take longer for macroinvertebrates versus fish, which are able to swim back into the habitat once the threat has passed (Barbour and others, 1999; Gore and others, 2001; Wallace and Anderson, 1996; Bruno and others, 2016; Miller and Judson, 2014; Timusk and others, 2016). Rapid decreases in flow rates and river height because of hydropeaking can cause an increase in mortality from stranding and desiccation (Kennedy and others, 2016; Meile and others, 2016). Additionally, shifts in annual flow and thermal maxima and minima can disrupt breeding cues for many species (Freeman and others, 2001; McManamay and Frimpong, 2015). The consensus has been a general decrease in richness and diversity in favor of generalist species that can thrive well in disturbed flow conditions.

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[2]U.S. Geological Survey.

R.L. Harris Dam

R.L. Harris Dam is a hydroelectric facility located on river kilometer 223.9 of the Tallapoosa River in Randolph County, Alabama (U.S. Army Corps of Engineers, 2015; fig. C1). Beginning operation on April 20, 1983, the facility contains two 65,500-kilowatt generators that release approximately 368.1 cubic meters per second (m^3/s) at best gate release (U.S. Army Corps of Engineers, 2015). The dam is generally operated on a daily hydropeaking schedule for electricity production in addition to meeting demands for water supply and flood control. In an effort to improve flow conditions in the section of the Tallapoosa River downstream from R.L. Harris Dam, the operators agreed to seek and test alternative flow plans that would better meet the objectives of the various stakeholders involved in the project, including ecological objectives of improving the conditions for downstream communities.

Since its induction, the R.L. Harris Dam Adaptive Management Project (AMP; “Harris AMP,” hereafter) has collected macroinvertebrate samples in conjunction with the collection of fish samples as part of the monitoring program. Thus far, reported faunal responses to flow management in the Tallapoosa River included a decrease in fish diversity below the dam potentially related to a reduction in shoal habitat persistence and a delay of thermal cues for breeding (Irwin and others, 2011). Although the biological emphasis of the Harris AMP has been to improve population persistence of fish species, the objective of maximizing the diversity and abundance of the native flora and fauna includes other faunal groups, such as macroinvertebrates. Understanding the response of the macroinvertebrate communities to river management is essential in terms of provision of productive food sources for fish populations (Gore and others, 2001).

Analysis of Historical Macroinvertebrate Collections

Historical macroinvertebrate collections from the Harris AMP were systematically analyzed to determine if the macroinvertebrate community of the middle Tallapoosa River demonstrated responses to regulation type under different natural hydrologic conditions and identify which group of taxa or suite of traits could be used to determine species or community responses to flow management. To meet these goals, our specific objectives were to (1) characterize the macroinvertebrate communities in regulated and unregulated reaches; (2) characterize the macroinvertebrate communities during extreme natural variation in hydraulic conditions, or water years preceded by prolonged drought or wet conditions; (3) examine the distribution trends for FFG and habit traits among regulated and unregulated reaches; and (4) identify taxa that may be useful for future monitoring efforts on the Tallapoosa River. Based on results of published literature, we hypothesize that (1) macroinvertebrate communities would demonstrate a shift in community composition to more generalist species in regulated reaches versus unregulated reaches, (2) macroinvertebrate communities would demonstrate a response to natural hydrologic conditions in unregulated reaches and that response may be muted in regulated reaches because of a regulated flow regime, (3) FFG and habits would reflect a generalist community in regulated reaches versus taxa specialized for burrowing or more stable flows found in unregulated reaches, and (4) taxa commonly noted within the literature will indicate disturbed flow conditions, such as Baetidae and Hydropsychidae having a greater association with disturbed reaches, whereas taxa favoring stable flows, such as burrowers and predators, will have a greater association with unregulated reaches.

Methods

Macroinvertebrate Surber sampling was completed concurrent with fish sampling on many occasions except where daylight or staffing hours may have reduced the daily field work hours available for safe working conditions. Sampling was completed in summer (May–July) and fall (September–November) of 2005–17 in 5 river reaches; 3 regulated by R.L. Harris Dam (Malone, Wadley, and Horseshoe Bend) and 2 unregulated reaches (Upper Tallapoosa [“Heflin,” hereafter] and Hillabee Creek [“Hillabee,” hereafter]; fig. C1). A total of five shoals (A–E) were randomly selected within each reach, and attempts were made to visit every shoal once during each sampling season. Refer to chapter B of this report for details on fish sampling methods and results.

Macroinvertebrate Field Sampling Methods

A total of 10 Surber samples were collected randomly from shoal habitat with a water depth of at least 27 centimeters (cm) and no more than 40 cm, proceeding from downstream to upstream to avoid collection of drifting invertebrates from upstream sampling in subsequent downstream samples. The Surbers were standard 500-micrometer (µm) mesh Surbers with a 30.5-cm x 30.5-cm base opening. After placing the Surber, large rocks and debris were cleared of macroinvertebrates before agitating the substrate 8-cm deep or to bedrock for about 30 seconds. Samples were promptly emptied into 5-gallon buckets about half full of clean river water. Again, large rocks and debris were visually inspected for macroinvertebrates before being returned to the shoal habitat. Remaining sample material was then elutriated through a 0.25-millimeter (mm) mesh sieve repeatedly until all organic material was removed from the bucket and no movement was noted upon close visual inspection (usually four or five times). Material left in the bucket after elutriation was returned to the shoal habitat. Specimens and material elutriated into the sieve were collected and stored in high-density polyethylene jars along with a printed label that recorded date, reach, shoal, and

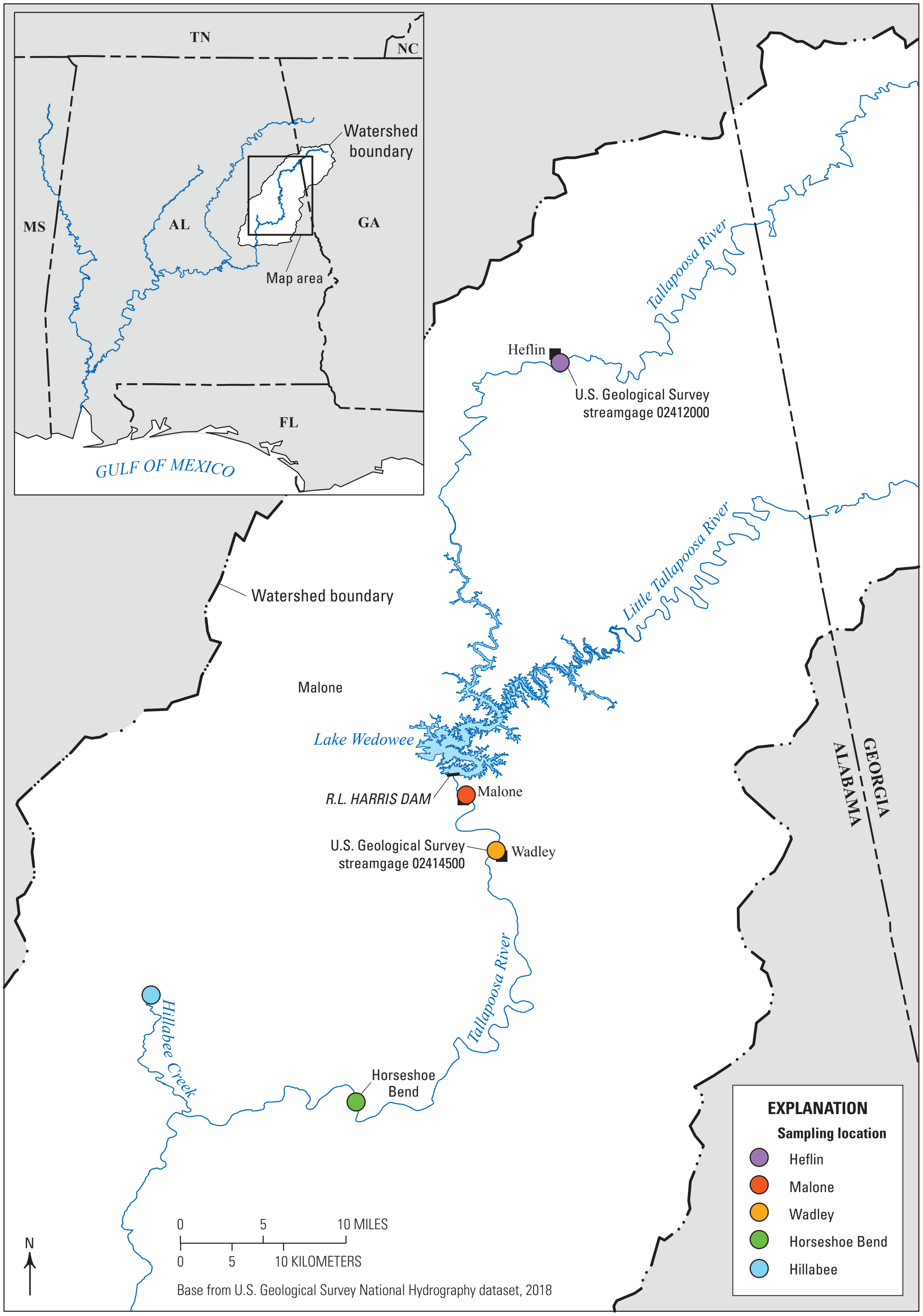

Figure C1. Sampling locations of the R.L. Harris Dam Adaptive Management Project in the Piedmont region of Alabama.

random (1–10) Surber number, then filled with 70-percent ethanol until sample material was completely covered. Samples were stored in fire-proof storage cabinets until samples were processed.

Sample Processing

The sorting procedures developed for this project are detailed in appendix C1. The objective of the protocol was to maximize the number of species sorted from a Surber sample within 2 hours. Samples were processed using a subsampling technique with a target count of 100–150 individuals, accompanied by a qualitative post hoc visual inspection of remaining unsorted material to account for additional taxa not detected in the timed sort (similar to the P2 protocol in Stark and others [2001]). The percentage of material that was sorted to meet the target count was recorded and used to estimate the total number of individuals in the sample based on sorted proportions. See appendix C1 for details. Sorted samples were placed in glass vials, labeled, and stored in 70-percent ethanol.

Identification

Specimens were identified using stereo microscopes (75–640×, Olympus SZH, Tokyo), and a compound microscope (40×; 100×; 400×; 1,000×; Motic BA210, China) for confirmation of smaller taxonomic features where required. All nondipteran insects were identified to the lowest taxonomic level (genus or species) using Epler (2010), Merritt and others (2012), and Morse and others (2017). All dipterans were identified to the family level at minimum, except for Chironomidae, which were identified to subfamily, using the guide by Epler (2001). *Cheumatopsyche* were identified to a select group of easily identified morphotypes as designated by Burington (2011): *Cheumatopsyche* A/B (no notch on the frontoclypeus and long setae on the pronotal margin), *Cheumatopsyche* D (small notch on the frontoclypeus with long pronotal setae), *Cheumatopsyche* E (large notch in the frontoclypeus with two large sclerites in the prosternum), and *Cheumatopsyche* (all others—short pronotal setae and various frontoclypeus shapes). Noninsects were identified to the order or family level using available guides (Kathman and Brinkhurst, 1998) and communication with local experts.

Taxonomists followed a “no head, no count” rule in which a present and identifiable head (or mouthpart for annelids) must be present (either attached or detached) for a specimen to be considered alive at the time of capture and counted. All terrestrial specimens (beetles, isopods) were excluded from the analysis. Aerial adults, including those of the orders Diptera, Ephemeroptera, Plecoptera, and Trichoptera, were not included in the analysis. Pupae, exuviae, empty shells, and cases were not counted. All zooplankton, including copepods and cladocerans, were considered microinvertebrates and excluded from the analysis.

Adult riffle beetles (Elmidae) were considered part of the general aquatic community assemblage and recorded as a separate taxon from larval Elmid beetles (for example, Elmidae_adult), but other adult aquatic beetles (including rarely caught surface skaters) were not included. Elmid beetles are prominent within the study system and larvae are commonly noted in large numbers. Larval and adult Elmidae have different morphologies and, therefore, have different diets, drift periods, and hydrologic and thermal tolerances (Elliott, 2008). Adult Elmidae beetles are still representative of the immediate habitat area because they have been known to travel short distances after pupation before returning to the water after which they will not leave the river gain for the rest of their lives, and some adults never fly at all and remain in the water after pupation (Elliott, 2008). For those reasons, we have included adult Elmidae beetles in the analysis, but they are noted as separate taxa from the Elmidae larvae.

Many samples contained large groups of early instar individuals, especially from the families Baetidae, Heptageniidae, Hydropsychidae, and Perlidae. During these early instar phases, many individuals lacked properly developed features that are typically used for identification (for example, gills of Heptageniidae and ocelli of Perlidae). These observations were recorded at the lowest possible identification level, generally at the family level. For analysis at the lowest taxonomic level, all early instar individuals are labeled as taxon* (for example, Hydropsychidae*).

Power Analysis

Although the established sampling protocol for the Harris AMP called for 10 samples to be collected at each shoal, there were several sampling occasions that did not have a full set of 10 samples available for that site. The availability of an appropriate Surber sampling area was generally determined by river conditions and was occasionally limited by excessive flood or drought conditions during sampling periods. Additionally, the number of samples available per shoal was diminished after storage because of desiccation and decay within some samples. With further restrictions on time and availability of trained personnel for macroinvertebrate identifications, it was decided that finding an appropriate sample size to represent the shoal community that would allow for even sampling effort across all shoals, reaches, and years would be the best option moving forward.

To determine adequate sample sizes for characterizing community structure on river shoals, six macroinvertebrate samples per shoal were randomly selected from available presorted material from the 2005 and 2014 sampling years, the first year sampled and the last year sampled on hand at the time. After samples were sorted, the data from these 2 years were analyzed to identify the minimum number of samples needed to represent macroinvertebrate communities at the shoal level. Fall samples from four reaches were used: the Heflin and Hillabee reaches represented unregulated reaches, and the Malone and Wadley reaches represented regulated reaches.

A power analysis is used to determine the number of samples needed to detect whether a result differs significantly from the null hypothesis given the parameters of power, or the significance criterion, and alpha (α), or the rate of rejecting a true null hypothesis (Cohen, 1988). Power analysis was completed on metrics derived from Barbour and others (1999) to determine the number of samples needed to distinguish among shoals. Replicate Surber samples were combined to represent a single shoal. Shoal-level power analysis (power=0.8, α=0.05) was completed on a 1:1 basis (1 regulated shoal versus 1 unregulated shoal), as well as with 3 shoals combined for several shoals (2005 shoals included Heflin E, Malone B, Hillabee A, and Wadley A; 2014 shoals included Heflin B, Wadley E, and Hillabee B). Species accumulation curves were also generated to support the results of the power analysis.

A total of 50 percent of the 12 power analyses indicated that 3 samples were sufficient to illustrate differences between regulated and nonregulated shoals; alternatively, the other 50 percent indicated 2 to 3 times the number would be required to demonstrate differences between regulated and nonregulated shoals (table C1). Subsequent species accumulation curves (fig. C2) indicated that additional samples would minimally increase species richness values; however, the objective of these analyses is to assess the structure of the macroinvertebrate community at a single shoal to estimate reach-level differences in the macroinvertebrate community, not to perform a complete biological survey of the macroinvertebrate community. Based on the results of the power analysis and species accumulation curves, it was determined four samples per shoal was adequate to detect reach-level differences in macroinvertebrate community composition.

Subsampling Procedure

Based on the power analysis, subsampling of macroinvertebrate samples collected during the Harris AMP proceeded by randomly selecting 3 shoals to represent each reach per year (see below) and randomly selecting 4 Surber samples per shoal. Fall samples were processed from four reaches; Heflin and Hillabee represented unregulated reaches, and Malone and Wadley represented regulated reaches.

Selection of Sampling Years Based on Natural Variation in Hydrologic Extremes

We prioritized identifying additional years based on maximizing differences in hydrologic regimes in the Tallapoosa Basin as measured by the National Oceanography and Atmospheric Administration. The National Oceanic and Atmospheric Administration maintains regional historical drought records for each State based primarily on Palmer's Drought Severity Index (Palmer, 1965). This method considers a water-balance measure generated with historical precipitation and

Table C1. Power analysis to estimate the number of Surber samples needed to test for a significant difference between regulated and nonregulated shoals of the R.L. Harris Dam study. Power, or the probability of accepting a false positive where the real difference is equal to the minimum effect size, was set to 0.8 and α, or the significance level at which the null hypothesis is rejected, was set to 0.05. The results of the power analysis estimate how many Surber samples are necessary to test for differences among shoals.

[Hef, Heflin; Mal, Malone; --, no data; Hill, Hillabee; Wad, Wadley]

Metrics	Year	Shoal 1	Shoal 2	Shoal 3	Power	Surbers
Percent burrower	2005	Hef E	Mal B	--	0.99	3
Percent burrower	2005	Hill A	Mal B	--	0.99	3
Percent clinger	2014	Hef B	Wad E	--	0.99	3
Percent clinger	2014	Hef B	Wad E	Hill B	0.99	3
Percent gatherer	2014	Hef B	Wad E	--	0.8	8
Percent gatherer	2014	Hill B	Wad E	--	0.778	13
Percent scraper	2014	Hef B	Wad E	--	0.86	7
Percent scraper	2014	Hill B	Wad E	Hef B	0.87	7
Richness	2005	Hill A	Wad A	Mal B	0.95	3
Richness	2005	Hill A	Wad A	--	0.91	3
Richness	2005	Hef E	Wad A	--	0.852	7
Richness	2005	Hef E	Wad A	--	0.78	6

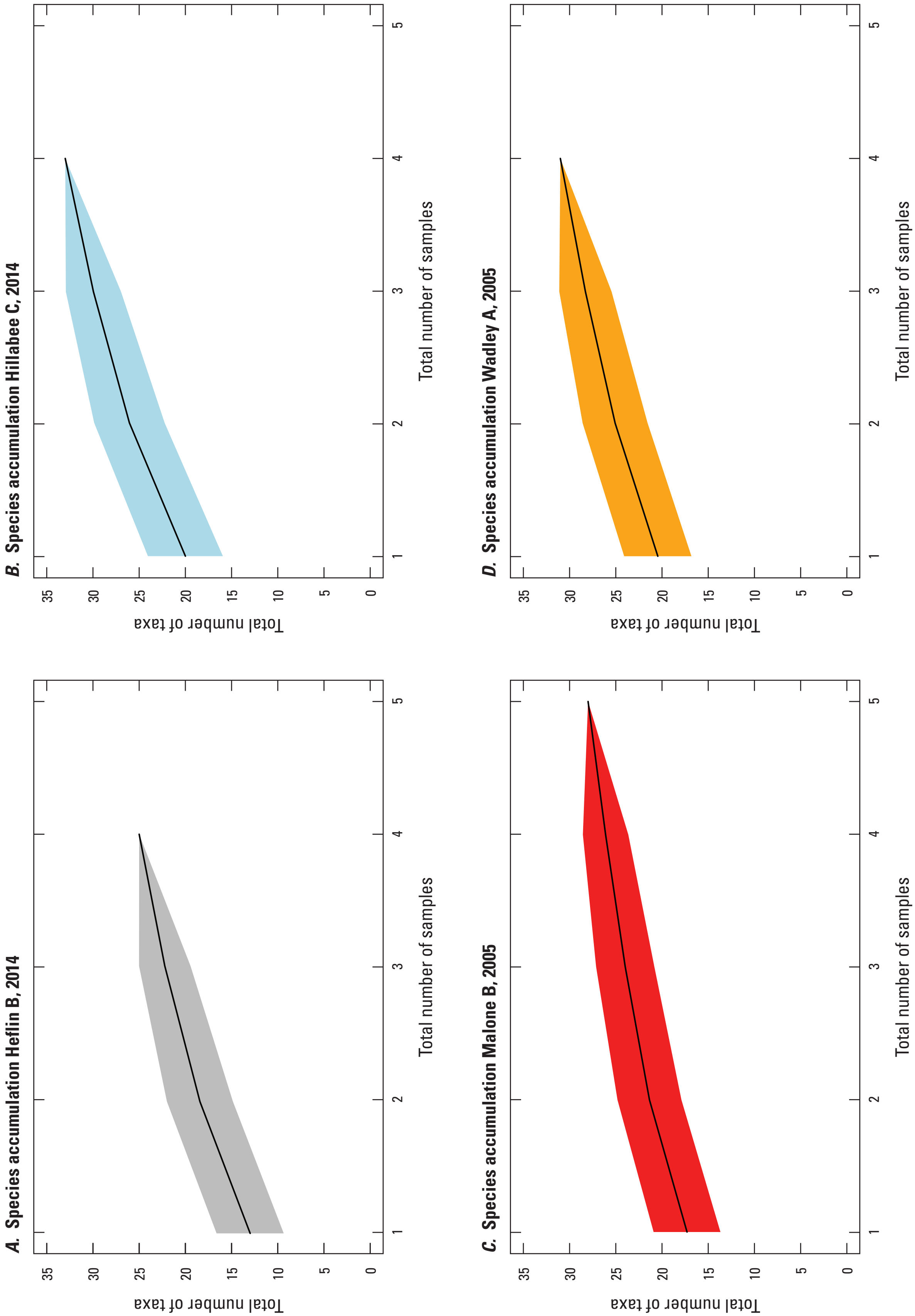

Figure C2. Species accumulation curves and associated standard deviation based on preliminary data for selected sites.

temperature data, soil moisture storage, potential evapotranspiration, and runoff to generate climate-dependent coefficients that assist in determining the deviation of the measured precipitation to the expected precipitation based on historical climate conditions while considering the cumulative water demands, which are normalized to values that are comparable across different climactic regions. Palmer's Modified Drought Index (PMDI) is the updated working model that considers the previous drought or wet period in addition to the current drought or wet period (Heddinghaus and Sabol, 1991).

Historical monthly PMDI values for the Piedmont Plateau Division (division 5) of Alabama were obtained online through the National Oceanic and Atmospheric Administration (National Climate Data Center, 2016). Historical discharge data were obtained through the U.S. Geological Survey online database for the two U.S. Geological Survey streamgages, Tallapoosa River at Wadley, Ala. (U.S. Geological Survey streamgage 05414500; "Wadley streamgage," hereafter; U.S. Geological Survey, 2018), and Tallapoosa River near Heflin, Ala. (U.S. Geological Survey streamgage 02412000; "Heflin streamgage," hereafter; U.S. Geological Survey, 2018), representing discharge for regulated and unregulated reaches of the Tallapoosa River, respectively. Monthly average values for PMDI and discharge for both the Wadley and Heflin streamgages were graphed in Microsoft Excel (Microsoft Office 365 ProPlus, version 1708; table C2, fig. C3). There were two extensive drought periods during the AMP: one from the beginning of 2006 to mid-2008, and another from mid-2010 to the beginning of 2013. Drought period 1 (2006–08) included the minimum PMDI value, which was recorded in September of 2007 (−5.48 monthly average) and low average discharge for the preceding winter (December 2006 through February 2007) and spring (March 2007 through May 2007) months during which the average discharge for Heflin was less than [<] 15.9 m^3/s and average discharge for Wadley was <76.4 m^3/s. Drought period 2 (mid-2010–12) was characterized by greater monthly average discharge during the winter and spring months (monthly averages for Heflin were <36.5 m^3/s and monthly averages for Wadley were <150.1 m^3/s during drought period 2) and a greater minimum PMDI (−4.18 for August 2011) compared to drought period 1. Conversely, a period of positive PMDI values was recorded from winter 2008–9 to mid-2010, with the greatest positive value for the PMDI (5.86) reported in December 2009. This period also contains the greatest monthly average discharge for the Wadley reach (monthly average for December 2009 was 238.5 m^3/s) but only the fourth greatest discharge value noted for the Heflin reach (after December 2013, July 2005, and December 2015 when average discharge was 196.1 m^3/s, 199.5 m^3/s, and 166 m^3/s, respectively) from the years 2005–15.

Additional years for sample processing and identification were selected based on the observations on the Piedmont PMDI values and availability of samples. The fall of 2008 samples were chosen to represent the culmination of drought period 1 as insufficient samples were available from the fall of 2007 because of extremely low water levels and inaccessibility of sampling sites (table C2). Fall of 2012 was selected as it represents the culminations of drought period 2 and sufficient samples were available. The fall of 2009 was chosen to represent a wet year from the 2009 to 2010 wet period.

A total of 228 Surber samples were included in the final analyses, with samples from each shoal summed to form 57 testable shoals from 5 years; 2005 (number of samples [*n*]=48) and 2014 (*n*=48) represent "bookend" years (the first and last year on hand when processing began), 2008 (*n*=44) and 2012 (*n*=40) represent "drought" conditions, and 2009 (*n*=48) represents "wet" conditions. Each shoal was represented by the sum of four randomly selected Surber samples from the available samples of sufficient quality (for example, no decay, no desiccation, and so on). Each reach is represented by 3 shoals, except for 2 reaches where sufficient sampling days did not take place that year: Heflin in 2008 (2 shoals) and Malone in 2012 (1 shoal). Because both of these occasions took place during drought years, it is likely that there were insufficient flows on most days of the designated sampling period to complete field work (see table C2).

Community Analyses

Analyses were completed on the sum of 4 Surbers from each of 3 randomly selected shoals from 4 reaches, 2 regulated (Malone and Wadley) and 2 unregulated (Heflin and Hillabee), for fall samples from 5 years (see above). Percent composition of order and family within each regulation type overall and percent composition of dominant orders and taxon for each year and regulation type were calculated using Microsoft Excel (Microsoft Office 365 ProPlus, version 1708). Density and richness metrics for the community, Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa, and all non-EPT taxa were calculated for the complete dataset using R statistical software (R Core Team, 2016) using functions included in the rich (Rossi, 2011) and vegan (Oksanen and others, 2017) packages.

Nonmetric multidimensional scaling (NMDS) utilizing Bray-Curtis dissimilarity scores (k=2 and try=999 where k is the number of dimensions and try is the minimum number of iterations the program is required to complete before returning a result) was used to visualize the dissimilarities in the macroinvertebrate density data for each shoal with regards to regulation type and years. A permutational multivariate analysis of variance (PERMANOVA) test was completed on the sample-taxa density matrix using Bray-Curtis dissimilarity indices (permutations=999 where permutations are the minimum number of iterations to complete before returning a result) to test the significance of the differences in community dissimilarity scores based on site, year, and site:year interactions (McArdle and Anderson, 2001; Reiss and others, 2010).

A similarity percentage (SIMPER) analysis was completed on the sample-taxa density matrix using Bray-Curtis dissimilarity indices (permutations=999), and the results were

Table C2. Monthly average discharge values, with standard deviations in parentheses, for two U.S. Geological Survey streamgages, the Tallapoosa River at Wadley, Alabama, and Tallapoosa River near Heflin, Ala. (U.S. Geological Survey streamgages 02414500 and 02412000, respectively), referred to as "Wadley" and "Heflin," respectively, within the table, and the monthly reported Palmer's Modified Drought Index (PMDI) values for the Piedmont Division of Alabama. Data are from U.S. Geological Survey (2018) and National Oceanic and Atmospheric Administration (National Climate Data Center, 2016).

Month	Average discharge and standard deviation of discharge, in cubic meters per second		PMDI
	Heflin	Wadley	
2005			
January	15.9 (5.9)	71.5 (90.2)	−0.02
February	21.1 (8.1)	118.2 (126.5)	−0.34
March	29.8 (19.3)	145.3 (133.4)	0.53
April	34.3 (31.7)	152.5 (151)	1.31
May	14.1 (6.7)	73.8 (97.4)	1.00
June	14.7 (7.6)	72.9 (101.8)	1.52
July	51.2 (53.3)	199.5 (199.9)	3.28
August	16.1 (7.2)	79.8 (103.4)	3.62
September	6.3 (2)	40.1 (67)	2.03
October	5 (1.7)	26.9 (45.9)	1.12
November	8.2 (9.4)	47 (74.7)	0.65
December	14.9 (8.1)	62.3 (79.8)	−0.29
2006			
January	26.9 (27.4)	102.1 (120.6)	−0.54
February	42.4 (41.2)	144.1 (142.3)	−0.55
March	29.4 (23)	117.6 (121.8)	−1.46
April	18.3 (3.7)	46.3 (58.7)	−1.93
May	12.6 (5)	61.7 (82.7)	−1.90
June	5.8 (2.2)	25.8 (41.9)	−2.27
July	2.9 (0.8)	22.4 (44.2)	−2.78
August	2.6 (0.8)	25.5 (48.5)	−3.19
September	4.4 (3.5)	13.6 (24.3)	−2.71
October	4.2 (4.1)	27.9 (45.9)	−1.44
November	14.6 (26.2)	74.3 (122)	−0.19
December	7.2 (2.6)	36.8 (60.2)	−1.24
2007			
January	15.9 (14.5)	76.4 (97)	−1.27
February	11.6 (4.8)	53.1 (76.7)	−2.27
March	10.5 (5.3)	41.5 (66.9)	−3.29
April	7 (1.2)	15.7 (10.3)	−3.45
May	3.5 (1.1)	10.8 (19.6)	−4.17
June	1.2 (0.4)	15.2 (39)	−4.50
July	2.3 (1.8)	18.5 (42)	−4.66
August	0.7 (0.5)	10.8 (26.2)	−5.36
September	0.5 (0.4)	10.6 (28.7)	−5.48
October	0.2 (0.2)	7.2 (18.1)	−5.20
November	0.7 (0.7)	5.2 (1.4)	−5.17
December	2.2 (2.3)	6.2 (2.4)	−4.95
2008			
January	5 (1.9)	8.5 (14.7)	−4.76
February	17.6 (16.9)	73.1 (102.5)	−3.67
March	17 (12.1)	69 (87.8)	−4.15
April	13.8 (8.2)	46.8 (67.6)	−3.99
May	10.2 (5.5)	50.8 (67.2)	−2.99
June	2.6 (1.4)	20.2 (41)	−3.49
July	2.1 (2.3)	18.4 (38.7)	−3.49
August	2.3 (3.2)	26.5 (44.6)	0.69
September	1.1 (0.5)	20.5 (44.7)	−0.37
October	1.1 (0.7)	18.8 (41.1)	1.09
November	1.5 (1.2)	13.2 (28.4)	0.80
December	12.7 (19.4)	61.7 (103.3)	1.04
2009			
January	14 (20.5)	62.7 (90.7)	0.39
February	11.6 (24.2)	51.3 (93.7)	0.26
March	27.6 (30.5)	127.1 (126.9)	1.09
April	19.4 (10)	97.3 (95.7)	0.61
May	18.1 (9.8)	103.6 (122.3)	2.00
June	5.2 (2.1)	33.9 (60.6)	0.69
July	3.4 (2.7)	27.7 (52.2)	0.51
August	6.5 (6.7)	18.6 (32.3)	1.36
September	15.9 (20.1)	87.7 (124.4)	2.70
October	25 (19.7)	155.3 (134.3)	4.67
November	38.2 (50)	182 (172.6)	4.93
December	48.9 (36.7)	238.5 (134.9)	5.86
2010			
January	39.2 (44)	176.9 (147.9)	5.51
February	43.3 (37)	187.8 (149.3)	4.67
March	47.6 (51.4)	211 (172.8)	4.02
April	21.3 (11.2)	55 (62.4)	2.13
May	24 (21.4)	106.6 (135.1)	2.58
June	11.1 (5.5)	60.7 (88.9)	1.34
July	5.7 (2.1)	32.1 (56.7)	−0.96
August	2.2 (0.8)	17.8 (36.1)	−2.01
September	0.7 (0.3)	12.8 (27.2)	−2.65
October	1.1 (1.4)	27.3 (53.6)	−2.53
November	2.1 (1.2)	35.5 (45.8)	−2.37
December	4.9 (2.7)	32.4 (39.6)	−2.74

Table C2. Monthly average discharge values, with standard deviations in parentheses, for two U.S. Geological Survey streamgages, the Tallapoosa River at Wadley, Alabama, streamgage and Tallapoosa River near Heflin, Ala. streamgage (U.S. Geological Survey streamgages 02414500 and 02412000, respectively), referred to as “Wadley” and “Heflin,” respectively, within the table, and the monthly reported Palmer’s Modified Drought Index (PMDI) values for the Piedmont Division of Alabama. Data are from U.S. Geological Survey (2018) and National Oceanic and Atmospheric Administration (National Climate Data Center, 2016).—Continued

Month	Average discharge and standard deviation of discharge, in cubic meters per second		PMDI
	Heflin	Wadley	
2011			
January	6.6 (2.5)	40.1 (51.5)	−3.00
February	11.3 (8.6)	62 (80.5)	−3.01
March	36.5 (44.8)	150.1 (141.5)	−1.58
April	18.5 (9.7)	79.9 (75.7)	−2.16
May	5.1 (1.5)	32.5 (39.6)	−2.88
June	2.8 (2)	25.1 (43.3)	−3.14
July	3.6 (2.7)	16.5 (25)	−3.11
August	1.9 (0.5)	16.3 (33.4)	−4.18
September	2 (1.4)	9.2 (10.1)	−3.15
October	1.2 (0.8)	23.3 (43.2)	−3.52
November	5.1 (7.2)	57.6 (81.4)	−3.05
December	8.9 (5.7)	56.4 (75.7)	−2.95
2012			
January	16.8 (22)	115.9 (135.7)	−2.40
February	9 (1.6)	65.8 (70)	−2.78
March	18.1 (18.8)	145.4 (134.7)	−3.09
April	6.4 (2.2)	13.1 (19)	−3.65
May	5.9 (3.7)	13.5 (15.4)	−3.26
June	2.2 (0.7)	18.9 (23.7)	−3.34
July	2.3 (2)	12.7 (12)	−3.67
August	1.6 (0.5)	7.9 (8.5)	−2.91
September	1.4 (0.5)	7.4 (7.7)	−3.19
October	1 (0.7)	12.5 (20.8)	−3.04
November	1 (0.2)	22.9 (39.9)	−3.46
December	8.5 (10.6)	60.9 (97.6)	−2.20
2013			
January	23.9 (30.7)	110.5 (123.3)	−1.73
February	35.6 (28.2)	159.5 (135.7)	1.05
March	20.7 (12)	112.8 (108.1)	0.07
April	25.1 (19.5)	84.2 (85.4)	0.40
May	24.7 (20.8)	160.5 (226.8)	1.35
June	12.2 (6)	79.7 (87.5)	1.75
July	23.6 (22.9)	127.7 (126.5)	2.74
August	16.7 (12)	78 (86.2)	3.17
September	12.9 (14)	44.7 (62.3)	2.67
October	6.6 (3.7)	58.7 (76.2)	1.98
November	6.9 (5.6)	56 (68.9)	1.14
December	52.1 (64)	196.1 (158.4)	2.95

Month	Average discharge and standard deviation of discharge, in cubic meters per second		PMDI
	Heflin	Wadley	
2014			
January	21.5 (17.1)	107 (114.2)	1.89
February	21.7 (7.5)	115.3 (90.9)	1.47
March	25.3 (11.6)	111.9 (96.3)	1.06
April	41.5 (43.7)	165.9 (176.9)	2.47
May	19.8 (7.3)	79.6 (77.2)	2.38
June	15.9 (7.9)	68.9 (74.6)	2.54
July	6.8 (3.8)	35 (43.2)	1.19
August	4.8 (3)	23.3 (29.8)	0.13
September	2.4 (0.9)	13.3 (15.3)	−0.87
October	3 (2.2)	23.6 (31.1)	−0.69
November	10.3 (13.5)	51.2 (74.9)	−0.87
December	18 (22.6)	87.9 (131.9)	−0.44
2015			
January	34.5 (55.1)	122.9 (142.4)	−0.76
February	17.9 (6.5)	82.9 (83.1)	−0.72
March	19 (6.6)	74.9 (71.2)	−1.38
April	41.4 (20.2)	125.2 (116.8)	−0.47
May	14.8 (4.6)	55.7 (60.6)	−0.27
June	10.6 (4.1)	31.5 (40.7)	−0.56
July	6.2 (3.3)	24.6 (36.6)	−1.34
August	2.9 (1.6)	16.9 (19.3)	−0.33
September	2.4 (0.7)	9.8 (11.2)	−1.14
October	4.1 (1.8)	26.2 (41.9)	−1.24
November	22 (16.5)	95.4 (99)	0.77
December	50.7 (68.8)	166 (215)	2.66

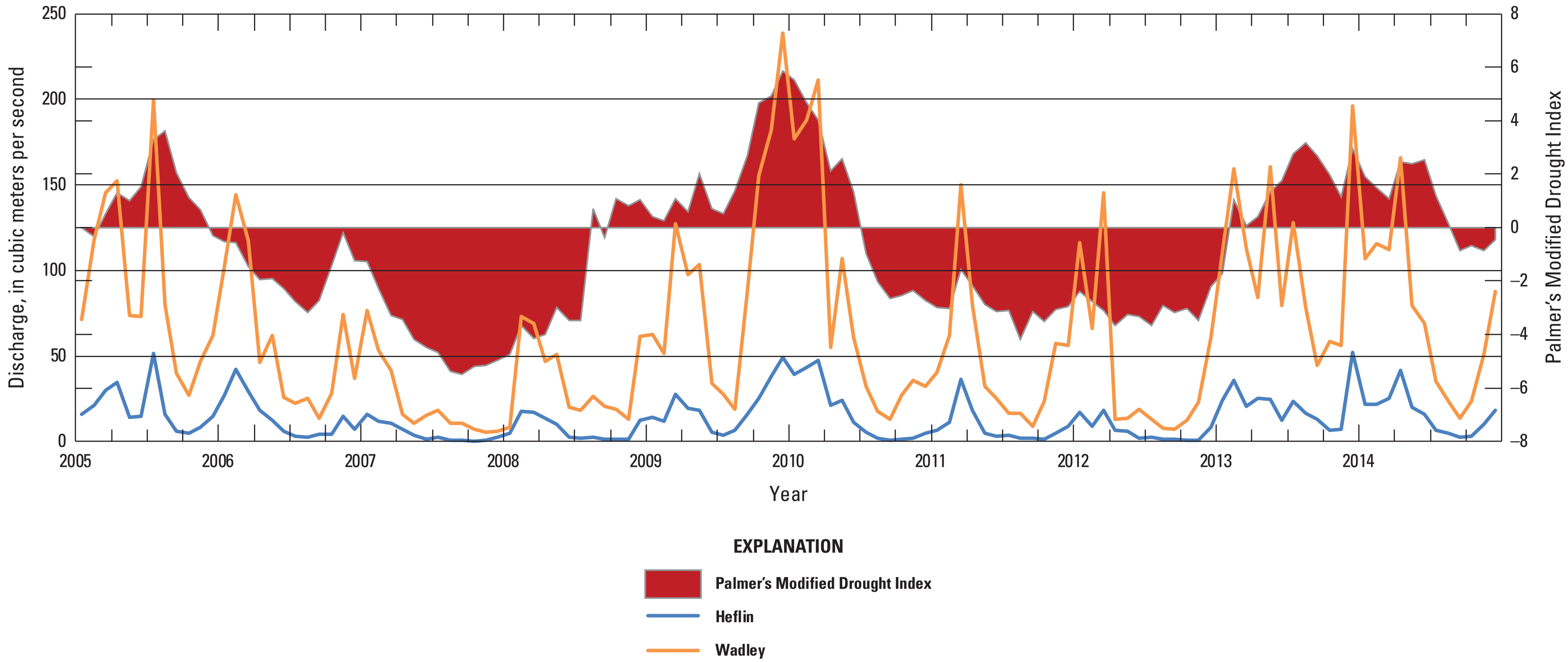

Figure C3. Monthly average discharge values for Heflin and Wadley streamgages (U.S. Geological Survey streamgages 02412000 and 02414500, respectively) and monthly average Palmer's Modified Drought Index (PMDI) score for the Piedmont Region (Piedmont Division of Alabama) from 2005 through 2016. Data are from U.S. Geological Survey (2018) and National Oceanic and Atmospheric Administration (National Climate Data Center, 2016).

used to determine which taxa contributed the most to the differences in the dissimilarity scores. To visualize taxa-specific responses to river regulation, the taxa that contribute the most to the cumulative deviation between regulation types were represented in the NMDS bubble plot using the estimated average density of each taxa as the size variable for the plotting function.

To determine if classes of FFG or habit had significant association with the NMDS results, the log transformed (log *x*+1) percent composition of the estimated average densities of each class for each site were fit on the NMDS ordination. Results were plotted over the NMDS results to visualize the associations of FFG and habit class vectors to the NMDS ordination of shoal over average estimated macroinvertebrate density.

To further illustrate similarities among macroinvertebrate communities at the reach scale, we completed a cluster analysis on the presence-absence taxa matrix of the taxa data using Ward's method of Hierarchical Clustering (Ward, 1963) with Euclidian distances. A SIMPER analysis of the sample-taxa presence-absence taxa matrix (permutations=999) was used to determine which taxa contribute the most to the dissimilarities between regulation type.

All statistical analyses as well as NMDS and cluster plots were completed using R statistical software (R Core Team, 2016) using functions included in the ggplot2 (Wickham, 2009), ggrepel (Slowikowski, 2016), reshape2 (Wickham, 2007), rich (Rossi, 2011), and vegan (Oksanen and others, 2017) packages.

Results

A total of 164 taxa were identified from the 230 samples selected from the fall sampling season for the years of 2005, 2008, 2009, 2012, and 2014 (appendix C1). Dipterans alone represented more than a quarter of all specimens observed (26.72 percent, table C3), whereas EPT orders represented 41.27 percent of all specimens observed (19.96 percent Ephemeroptera, 4.03 percent Plecoptera, and 17.28 percent Trichoptera). Mollusks accounted for 7.9 percent of all specimens (3.91 percent Neotaenioglossa, 2.94 percent Veneroida, and 1.05 percent Basommatophora), Tubificida 6 percent, and Trobidiformes 3.07 percent.

Table C3. Percent composition of each identified order or group for each regulation type (regulated and unregulated) and for all (total) observations, listed in order of greatest to least percent composition of the total observations. The number of families represents the number of families positively identified within each order or group (not counting groups of immature individuals).

[--, no data]

Order or group	Regulated, in percent	Unregulated, in percent	Total, in percent	Number of families
Diptera	21.85	31.16	26.72	13
Ephemeroptera	22.80	17.37	19.96	10
Trichoptera	21.85	13.13	17.28	10
Coleoptera	8.96	12.33	10.72	7
Tubificida	6.79	5.28	6.00	1
Plecoptera	1.91	5.97	4.03	7
Neotaenioglossa	3.93	3.89	3.91	2
Trombidiformes	3.24	2.92	3.07	1
Veneroida	2.55	3.30	2.94	2
Odonata	1.30	1.46	1.38	6
Nematoida	1.52	0.90	1.20	1
Basommatophora	1.07	1.04	1.05	3
Turbellaria	1.75	0.38	1.04	1
Megaloptera	0.46	0.69	0.58	2
Lepidoptera	0.00	0.14	0.07	2
Hirudinea	0.04	0.03	0.04	1
Total	**100**	**100**	**100**	**--**

Diptera was the most diverse order, representing 13 positively identified families (table C3). Overall, the EPTs were represented by 27 positively identified families. Regulated reaches lacked positive identifications for many macroinvertebrate families documented in this study, including Plecoptera (1 family identified in regulated reaches out of 7 total families documented in the study, or 1 of 7), Lepidoptera (0 of 2), Diptera (7 of 13) and Odonata (4 of 6), whereas unregulated reaches lacked 1 family of Trichoptera (9 of 10) and 2 Dipterans (11 of 13) out of all the macroinvertebrate families identified in this study.

Characterization of Regulated and Unregulated Communities

Regulated reaches were characterized by greater density (total density, EPT density, and non-EPT density), and unregulated reaches were characterized by greater richness (total richness and EPT richness; table C4). Except for the minimum EPT richness and the standard deviation of non-EPT richness, the unregulated reaches had greater mean, standard deviation, minimum, and maximum richness than regulated reaches. Except for minimum EPT density, minimum non-EPT density, and maximum non-EPT density, regulated reaches had greater mean, standard deviation, minimum, and maximum density values than unregulated reaches. The EPT density is notably larger in regulated reaches, with mean density more than three times greater than in unregulated reaches and maximum density more than five times greater than in unregulated reaches.

The community composition of regulated and unregulated reaches are different at the order, family, and taxon level (table C5). Regulated reaches were dominated by Ephemeroptera (29.75 percent), Diptera (27.57 percent), and Trichoptera (18.11 percent). The response for Ephemeroptera was largely driven by large abundances of Baetidae (19.82 percent) because of large abundance of *Iswaeon* (7.56 percent) observed in regulated samples. The response of Dipterans was driven by a large proportion of Chironomidae (22.82 percent), which was composed of nearly half Chironominae and half Orthocladiinae. The high rank for Trichoptera in regulated reaches was primarily because of large abundances of Hydropsychidae* (early instar, 6.97 percent), which made up more than half of the total observations of Hydropsychidae (13.01 percent).

Unregulated reaches were dominated by Diptera (40.15 percent), Ephemeroptera (14.44 percent), and Coleoptera (10.15 percent) orders (table C5). Most of the Dipterans observed were from the family Chironomidae (32.57 percent), which was composed of more Chironominae (17.82 percent) than Orthocladiinae (11.75 percent). Simuliidae also made up a large proportion of the Diptera specimens observed in unregulated reaches, accounting for 6.79 percent of both the family and taxa level observations. Coleoptera was the third ranking dominant order within unregulated reaches, driven by large abundances of the Elmidae (8.55 percent) family, of which more than half were *Stenelmis* (4.39 percent).

Macroinvertebrate Community Response to Natural Variation in Hydrology

The response of the community at the order and taxon level was assessed for each year within each regulation type (table C6). Regulated years from all years analyzed were dominated by Diptera, Ephemeroptera, and Trichoptera orders, with Diptera ranking first in the 2005 (32.25 percent) and 2009 (29.55 percent) years, and Ephemeroptera ranking first in 2008 (37.93 percent), 2012 (29.14 percent), and 2014 (38.83 percent). The high ranking of Diptera in regulated reaches was

Table C4. Values of selected metrics for regulated and unregulated reaches. Density is the average number of organisms per square foot per site (shoal). EPT are those taxa belonging to the Ephemeroptera, Plecoptera, and Trichoptera orders. Non-EPT are those taxa which are of the class Insecta but do not belong to the order Ephemeroptera, Plecoptera, or Trichoptera, which, in this study, are the orders Coleoptera, Diptera, Lepidoptera, Odonata, and Megaloptera. References for macroinvertebrate response to disturbance include Barbour and others (1999), Cortes and others (2002), Bednarek and Hart (2005), and Holt and others (2015).

[min, minimum; max, maximum; --, no data]

Metric name	Disturbance response	Regulated					Unregulated				
		Total	Mean	Standard deviation	Min	Max	Total	Mean	Standard deviation	Min	Max
Richness	Negative	84	30.04	6.98	10	44	112	32.07	7.28	18	46
EPT richness	Negative	47	14.89	3.70	9	24	58	15.41	5.05	7	25
Non-EPT richness	Positive	24	8.75	2.20	5	13	41	10.79	2.09	7	15
Density	Positive	--	375.78	335.49	58.00	1,613.00	--	217.04	203.00	34.50	1,009.00
EPT density	Positive	--	197.66	230.46	0.25	1,157.50	--	59.98	54.09	6.25	229.25
Non-EPT density	Positive	--	115.76	97.48	11.25	403.50	--	106.43	94.07	17.00	431.25

driven by large abundances of Chironominae, Simuliidae, and Orthocladiinae in 2005 (14.56 percent, 9.10 percent, and 6.91 percent, respectively), and Orthocladiinae and Chironominae in 2009 (14.02 percent and 10.18 percent, respectively). The high rank for Ephemeroptera in regulated reaches was driven by large abundances of both *Iswaeon* and *Acerpenna* in the 2008 (14.04 percent and 5.61 percent, respectively) and 2014 years (13.86 percent and 6.65 percent, respectively), whereas *Acerpenna* alone dominated the 2012 year (15.88 percent). Trichoptera ranked third in abundance within regulated reaches for all years except 2005, where it is ranked second. Hydropsychidae* is among the top five most abundant taxa in regulated reaches in the 2005 (7.14 percent), 2009 (9.28 percent), and 2012 (8.99 percent) years. Tubificida ranks fourth in regulated reaches in 3 of 5 years analyzed because of a large proportion of Naididae (5.99 percent to 11.92 percent).

The community composition in unregulated reaches indicates more diversity within the top five dominant orders and taxa across all years analyzed (table C6). Diptera was the dominant order in unregulated reaches for all years (32.86 percent to 55.62 percent) except 2014, where it is ranked second (15.51 percent) after Coleoptera (31.43 percent). Coleopterans are common in both regulated and unregulated reaches, ranking within the top five dominant taxa for both regulation types in 2005 and 2008 and unregulated reaches only in 2012 and 2014. The Coleoptera population was detected in much larger proportions in unregulated reaches in 2014, and these results were driven by the top-ranking taxon *Stenelmis* (17.60 percent) as well as fifth-ranking taxon *Optioservus* (5.72 percent). The same year Coleopterans dominated the unregulated reaches (2014) was also the only year in which Ephemeroptera was not among the five dominant orders in unregulated reaches. The 2005 year saw more than half of the individuals from the unregulated reaches were Dipterans (55.62 percent) because of a large proportion of Chironominae (27.26 percent) and Orthocladiinae (20.08 percent). Plecoptera were among the top five most common orders in unregulated reaches in 2005 (5.65 percent), 2012 (16.03 percent), and 2014 (10.64 percent), with Capniidae/Leuctridae complex driving results for this order in 2012 (9.84 percent) and *Neoperla* in 2014 (6.71 percent). Neotaenioglossa was also common in unregulated reaches, with large proportions in 2009 (8.25 percent) as a result of large proportions of Hydrobiidae (6.55 percent) and in 2014 (13.77 percent) as a result of large proportions of Pleuroceridae (10.04 percent). *Corbicula fluminea* were within the top five dominant taxa only in unregulated reaches during drought years (rank 5 at 5.14 percent in 2008 and rank 5 at 6.07 percent in 2012).

Table C5. List of the top five dominant taxa (order and taxon) in regulated and unregulated reaches based on percent of total observations.

Rank	Regulated		Unregulated	
	Name	Percent composition	Name	Percent composition
		Order		
1	Ephemeroptera	29.75	Diptera	40.15
2	Diptera	27.57	Ephemeroptera	14.44
3	Trichoptera	18.11	Coleoptera	10.15
4	Tubificida	7.58	Trichoptera	7.89
5	Coleoptera	5.27	Plecoptera	7.14
		Family		
1	Chironomidae	22.82	Chironomidae	32.57
2	Baetidae	19.82	Elmidae	8.55
3	Hydropsychidae	13.01	Baetidae	7.01
4	Naididae	7.58	Simuliidae	6.79
5	Heptageniidae	5.73	Naididae	6.06
		Taxon		
1	Chironominae	11.60	Chironominae	17.82
2	Orthocladiinae	10.48	Orthocladiinae	11.75
3	*Iswaeon*	7.56	Simuliidae	6.79
4	Naididae	7.56	Naididae	6.00
5	Hydropsychidae*	6.97	*Stenelmis*	4.39

*Indicates early instar individual.

Table C6. List of the top five dominant orders and taxa in regulated and unregulated reaches for each year.

Rank	Regulated		Unregulated		Regulated		Unregulated	
	Order	Percent composition	Order	Percent composition	Taxon	Percent composition	Taxon	Percent composition
2005								
1	Diptera	32.25	Diptera	55.62	Chironominae	14.56	Chironominae	27.26
2	Trichoptera	19.54	Ephemeroptera	14.58	Simuliidae	9.10	Orthocladiinae	20.08
3	Ephemeroptera	18.84	Trichoptera	9.89	Naididae	7.83	*Iswaeon*	4.95
4	Coleoptera	9.10	Plecoptera	5.65	Hydropsychidae*	7.14	*Cheumatopsyche*	4.08
5	Tubificida	7.90	Coleoptera	5.35	Orthocladiinae	6.91	Tanypodinae	3.64
2008								
1	Ephemeroptera	37.93	Diptera	32.86	*Iswaeon*	14.04	Chironominae	12.23
2	Diptera	22.43	Ephemeroptera	22.12	Chironominae	11.33	Naididae	7.78
3	Trichoptera	17.34	Coleoptera	8.59	Orthocladiinae	7.49	Simuliidae	7.63
4	Tubificida	5.98	Tubificida	8.09	Naididae	5.99	Orthocladiinae	7.40
5	Coleoptera	5.71	Trichoptera	7.06	*Acerpenna*	5.61	*Corbicula fluminea*	5.14
2009								
1	Diptera	29.55	Diptera	48.08	Orthocladiinae	14.02	Chironominae	18.30
2	Ephemeroptera	26.06	Ephemeroptera	14.29	Chironominae	10.18	Simuliidae	15.30
3	Trichoptera	18.82	Neotaenioglossa	8.25	Hydropsychidae*	9.28	Orthocladiinae	11.15
4	Tubificida	6.72	Tubificida	7.36	Naididae	6.68	Naididae	7.36
5	Trombidiformes	5.31	Trichoptera	7.12	*Acerpenna*	6.32	Hydrobiidae	6.55
2012								
1	Ephemeroptera	29.14	Diptera	33.26	*Acerpenna*	15.88	Chironominae	14.84
2	Diptera	27.87	Plecoptera	16.03	Chironominae	13.57	Orthocladiinae	12.72
3	Trichoptera	18.22	Ephemeroptera	11.83	Orthocladiinae	11.95	Capniidae/Leuctridae complex	9.84
4	Tubificida	11.92	Tubificida	9.49	Naididae	11.92	Naididae	9.47
5	Trombidiformes	5.17	Coleoptera	8.69	Hydropsychidae*	8.99	*Corbicula fluminea*	6.07
2014								
1	Ephemeroptera	38.83	Coleoptera	31.43	*Iswaeon*	13.86	*Stenelmis*	17.60
2	Diptera	25.54	Diptera	15.51	Orthocladiinae	11.06	Chironominae	10.94
3	Trichoptera	15.74	Neotaenioglossa	13.77	Chironominae	8.70	Pleuroceridae	10.04
4	Tubificida	6.47	Plecoptera	10.64	*Acerpenna*	6.65	*Neoperla*	6.71
5	Veneroida	3.59	Trichoptera	10.29	Naididae	6.47	*Optioservus*	5.72

*Indicates early instar individual.

Nonmetric Multidimensional Scaling

NMDS plots of the sample-taxa density matrix of macroinvertebrate taxa data ordinate on either side of the NMDS1 axis based on regulation type, with some exceptions (fig. C4, 2D stress=0.2247). All but one regulated shoal was ordinated towards the positive NMDS2 axis, whereas 23 of the 29 unregulated shoals were ordinated towards the negative NMDS2 axis. Regulated reaches cluster together with the exclusion of a notable outlier (2005_Malone_C). Unregulated reaches have a greater range for the ordination of abundance data on both the NMDS1 and NMDS2 axes, and, therefore, more regular deviation in the similarities of the community composition within unregulated reaches compared to regulated reaches.

Individual years show a great amount of overlap in the regulated reaches, with almost all shoals ordinated within the positive NMDS2 range (fig. C4). The regulated sites from the 2005 and 2008 years seem to ordinate towards the central NMDS1 axis (with the noted exception of the 2005_Malone_C outlier and 2008_Malone_A), whereas the 2012 and 2014 years are mostly ordinated in the negative NMDS1 direction, and the 2009 wet year ordinates throughout the regulated cluster. Individual years within the unregulated reaches also show some overlap as well as a distribution gradient along the NMDS1 axis. Most of the unregulated shoals from the 2012 and 2014 years seem to ordinate towards the positive NMDS1 axis and farthest from regulated reaches (with exceptions for 2012_Heflin_B and C), whereas shoals from the 2008 and 2008 years ordinate towards the center of the NMDS1 axis (with exceptions for 2005_Hillabee_A, 2005_Heflin_C and 2008_Heflin_E), and shoals from the 2009 year ordinate in the negative NMDS1 or positive NMDS2 directions.

Several regulated and unregulated sites are ordinated in such a manner as to seem to overlap regulation type. Specifically, 2009_Hillabee_C and D sites are the most positively ordinated unregulated sites, followed by 2014_Hillabee_C, 2005_Hillabee_D, and 2008_Hillabee_D. The regulated sites most positively ordinated on the NMDS1 axis are the 2008_Wadley_C and 2005_Wadley_D sites which ordinate between the aforementioned unregulated reaches that are positively ordinated on the NMDS2 axis. Inspection of the community composition at these sites reveals large abundances of Heptageniidae*, Hydrobiidae, and *Iswaeon* noted at all these sites.

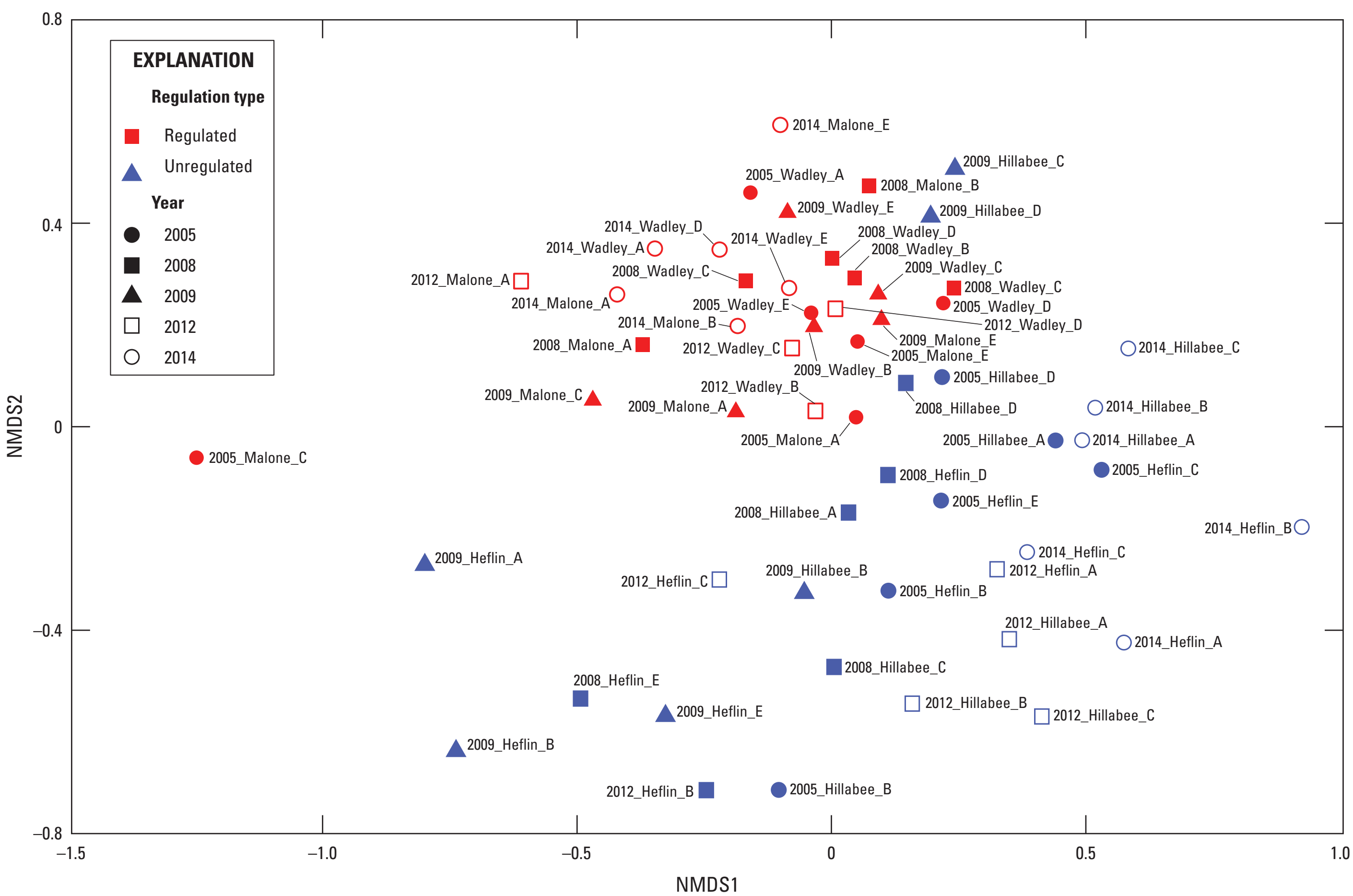

Figure C4. Nonmetric multidimensional scaling (NMDS) ordination of the average estimated sample abundance at each site. 2D stress=0.2247.

Permutational Multivariate Analysis of Variance

Results of the PERMANOVA indicated that the macroinvertebrate communities were different with regards to year (PERMANOVA, probability [p]<0.001), regulation (PERMANOVA, p<0.001), as well as the interaction between year and regulation (PERMANOVA, p<0.001) (table C7). The effect of year was significant (p<0.001) for both regulation types. The coefficient of determination (R^2) was greatest for the effect of year (R^2=0.12867), second greatest for the effect of regulation type (R^2=0.10855), and least for the interaction term (R^2=0.08569). The residual R^2 value is 0.6671.

Table C7. Results of permutational multivariate analysis of variance.

[DF, degrees of freedom; pseudo-F, variation in the distance matrix, analogous to the F statistic in an analysis of variance, or ANOVA (Reiss and others, 2010); R^2, coefficient of determination; p-value, probability value; reg, regulation type (regulated or unregulated); --, no data]

Metric	DF	Pseudo-F	R^2	p-value
Year	4	2.233	0.12867	<0.001
Reg	1	7.535	0.10855	<0.001
Year × reg	4	1.487	0.08569	<0.001
Residuals	47	--	0.67709	--

Density Similarity Percentage Analysis and Taxa Density Nonmetric Multidimensional Scaling Overlay

Results of the SIMPER analysis of macroinvertebrate taxa data indicated that 30 taxa out of 164 taxa analyzed contributed 84.8 percent of the variation in the macroinvertebrate communities (table C8). Chironominae contributed the greatest value to the percent contribution of differences in the SIMPER analysis and were found in larger abundances in regulated sites than unregulated sites. Graphing the previous NMDS with symbols indicating the abundance of the taxon, the shoals with a large abundance of Chironominae are ordinated towards the center of the NMDS1 axis, with shoals ordinated towards the extreme positive or negative NMDS1 axis having few or no Chironominae reported (fig. C5). Those unregulated sites from the 2012 and 2014 years that ordinated towards the positive NMDS1 axis show low abundances of Chironominae.

Another subfamily within the Chironomidae, Orthocladiinae, contributes the second most to the deviations in the SIMPER analysis of abundances (table C8). Graphing the abundance of Orthocladiinae as symbol size on the original NMDS ordination shows just a few of the sites, mostly from the regulated reaches, have large abundances of Orthocladiinae, especially from 2009_Malone_A and C sites, and 2008_Malone_A site (fig. C6).

The NMDS-abundance graph of the third-ranking taxon on the SIMPER analysis, *Iswaeon* (fig. C7), shows that all abundances greater than (>) 2,000 ordinate towards the positive NMDS2 axis. All unregulated reaches ordinated on the positive NMDS2 axis have notable *Iswaeon* population sizes (about [~] 2,000 individuals); whereas, among regulated reaches, all three Wadley reaches from 2008 and Malone_A_2008 have notably large population sizes (>3,000 individuals).

Large populations of Hydropsychidae* (SIMPER rank 4, fig. C8) are mostly within regulated reaches and are especially large in the 2009 and 2012 years but do not show a clear distribution pattern. In contrast, Simuliidae (rank 5, fig. C9) does show a distribution pattern, clearly driven by large abundances in the 2 unregulated reaches ordinated most towards the NMDS2 axis (2009_Hillabee shoals C and D) and, to a lesser extent, the 2 regulated reaches ordinated most towards the NMDS1 axis (2008_Wadley C and 2005_Wadley_D).

Functional Feeding Group and Habit Vectors on Nonmetric Multidimensional Scaling

Of seven FFG classes fit to the NMDS, four had significant association with the NMDS results (table C9): filterer/collectors (R^2=0.3269, p<0.001), gatherer/collectors (R^2=0.7023, p<0.001), predators (R^2=0.3297, p<0.001), and scrapers (R^2=0.5439, p<0.001). Plotting of the FFG vectors over the NMDS (fig. C10) indicated that filterer/collectors ordinated towards the NMDS2 axis. The scrapers vector ordinated towards the positive NMDS1 and positive NMDS2 axes, whereas gatherer/collectors were ordinated towards the negative NMDS1 and negative NMDS2 axes. The vector for predators was ordinated towards the center of the unregulated reaches in the positive NMDS1 and negative NMDS2 direction.

Of five habit vectors fit to the NMDS, four had significant association with the NMDS results (table C9): burrowers (R^2=0.5650, p<0.001), climbers (R^2=0.1073, p=0.047), clingers (R^2=0.4863, p<0.001), and swimmers (R^2=0.3603, p<0.001). Plotting of the vectors over the NMDS indicated swimmers and climbers have an association with regulated reaches as they both ordinate in similar positive NMDS2 and negative NMDS1 directions, with swimmers being the stronger of the two vectors (fig. C11). Burrowers ordinate in a negative NMDS1 and negative NMDS2 direction, like the gatherer/collector FFG vector (fig. C10), and clingers in a positive NMDS1 and negative NMDS2 direction with the unregulated sites, similar to the predator FFG vector.

Table C8. Top 30 results of the similarity percentage analysis of the Wisconsin square root transformed average estimated species abundance per shoal species matrix. The mean column shows the mean number observed in each regulation type, and the cumulative sum of deviation is the total deviation that taxon contributes to the overall similarity of the community observed at each site.

Rank	Taxon	Mean		Cumulative sum of deviation
		Regulated	Unregulated	
1	Chironominae	58.40793	53.03962	0.09489
2	Orthocladiinae	55.23785	30.36702	0.17857
3	*Iswaeon*	37.15852	9.45902	0.2369
4	Hydropsychidae*	40.18503	4.57040	0.29484
5	Simuliidae	19.75184	28.99499	0.35262
6	Naididae	32.97029	21.34779	0.40415
7	*Acerpenna*	37.93869	6.95384	0.45448
8	Acari	22.27980	8.59109	0.48468
9	Hydrobiidae	14.48402	12.49385	0.51472
10	*Stenelmis*	14.89303	9.89813	0.54413
11	*Corbicula fluminea*	6.22158	20.10891	0.57028
12	*Baetis*	18.41108	2.77428	0.59631
13	Heptageniidae*	15.85268	5.70873	0.62205
14	*Hydropsyche*	13.96072	2.65254	0.64398
15	*Cheumatopsyche*	17.31121	3.52870	0.6651
16	Pleuroceridae	6.65688	8.73726	0.68453
17	Baetidae*	10.71289	1.27670	0.70163
18	Tanypodinae	3.55907	10.41234	0.71789
19	*Maccaffertium*	13.83591	3.07144	0.73316
20	*Cheumatopsyche E*	11.99595	0.11533	0.74681
21	*Isonychia*	11.56421	3.42566	0.75951
22	Capniidae/Leuctridae complex	0.00000	6.68505	0.77175
23	*Neoperla*	2.01330	4.39741	0.78273
24	Ephemerellidae*	7.92330	0.27337	0.79363
25	*Microcylloepus pusillus*	6.20459	1.49031	0.80354
26	*Nectopsyche*	5.55427	2.08587	0.81318
27	*Dubiraphia*	0.08627	7.86792	0.82237
28	*Teloganopsis deficiens*	11.84292	0.05126	0.83111
29	*Optioservus*	2.00329	4.33796	0.83986
30	Turbellaria	5.39962	0.70763	0.84839

*Indicates early instar individual.

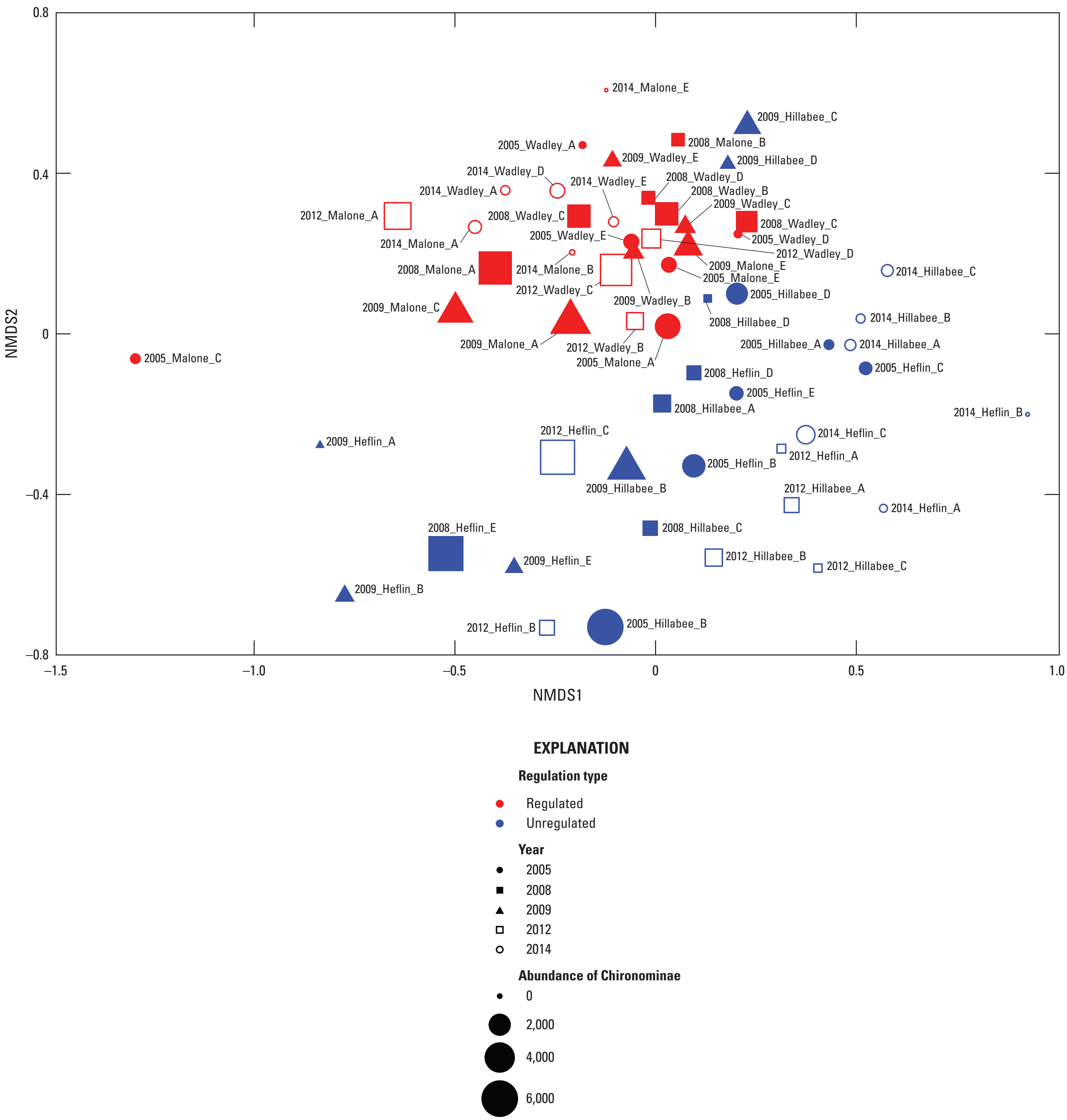

Figure C5. Nonmetric multidimensional scaling (NMDS) bubble plot of the abundance of Chironominae observed at each site (year, reach, and shoal) over the ordination of all data.

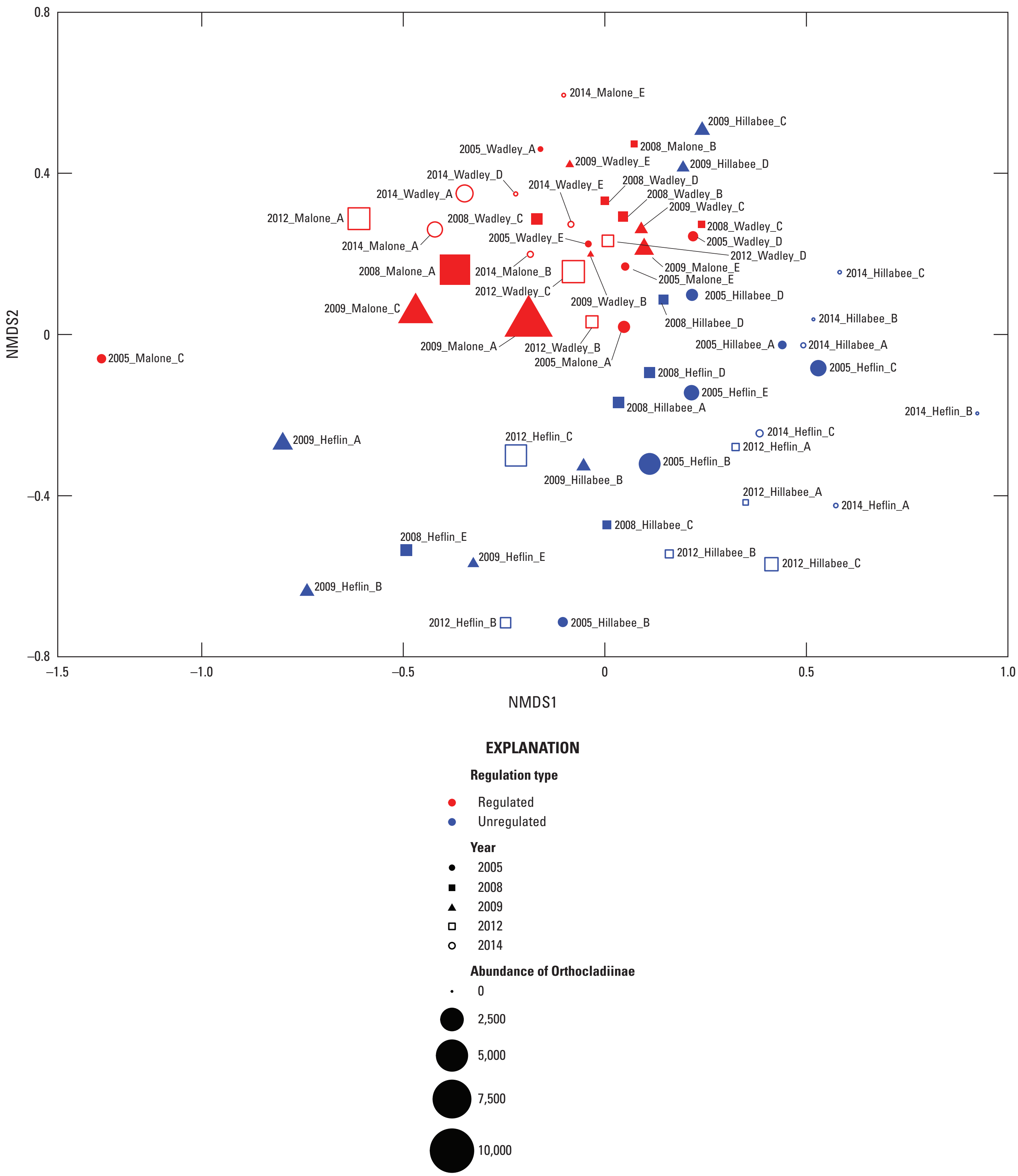

Figure C6. Nonmetric multidimensional scaling (NMDS) bubble plot of the abundance of Orthocladiinae observed at each site (year, reach, and shoal) over the ordination of all data.

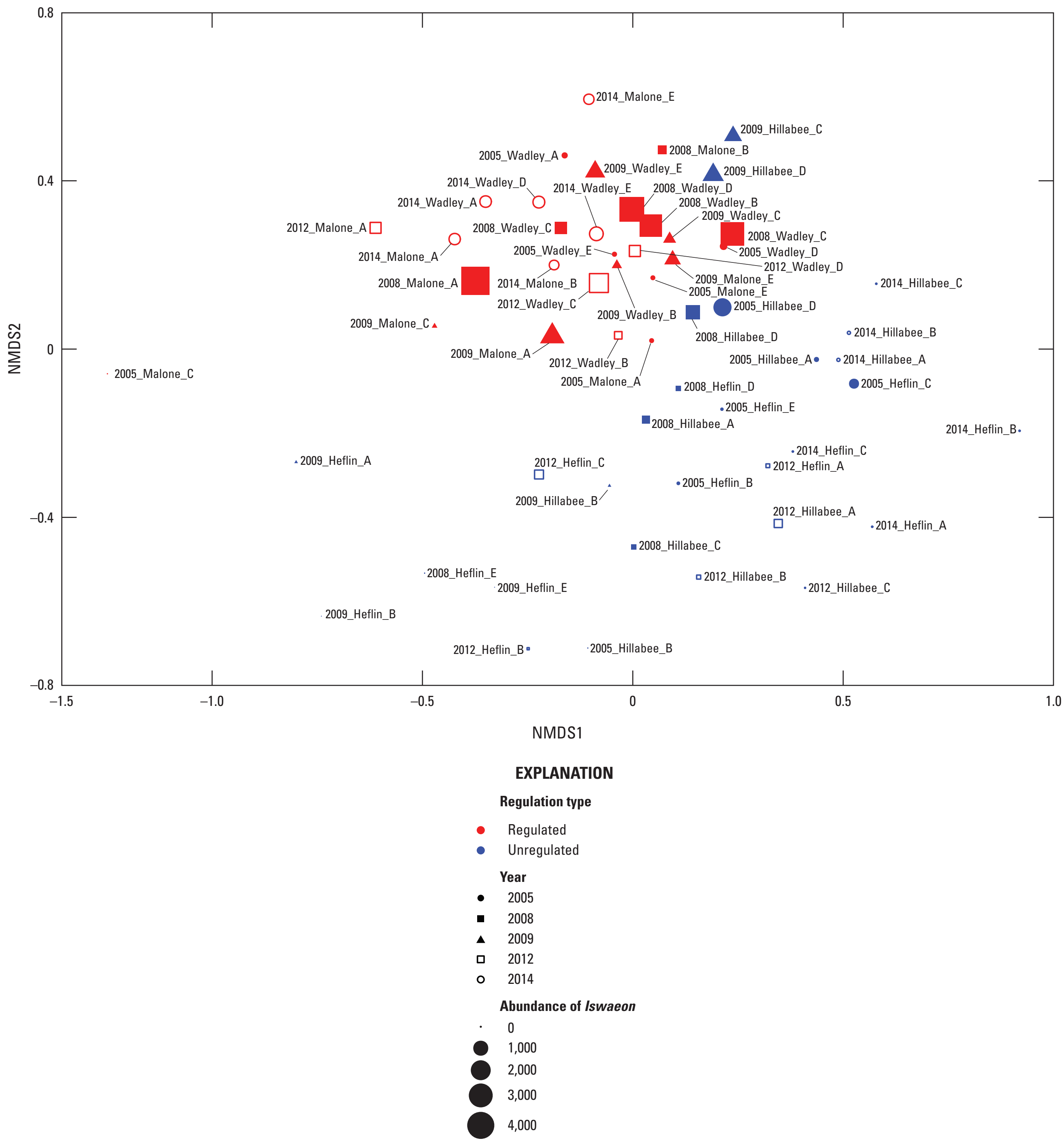

Figure C7. Nonmetric multidimensional scaling (NMDS) bubble plot of the abundance of *Iswaeon* observed at each site (year, reach, and shoal) over the ordination of all data.

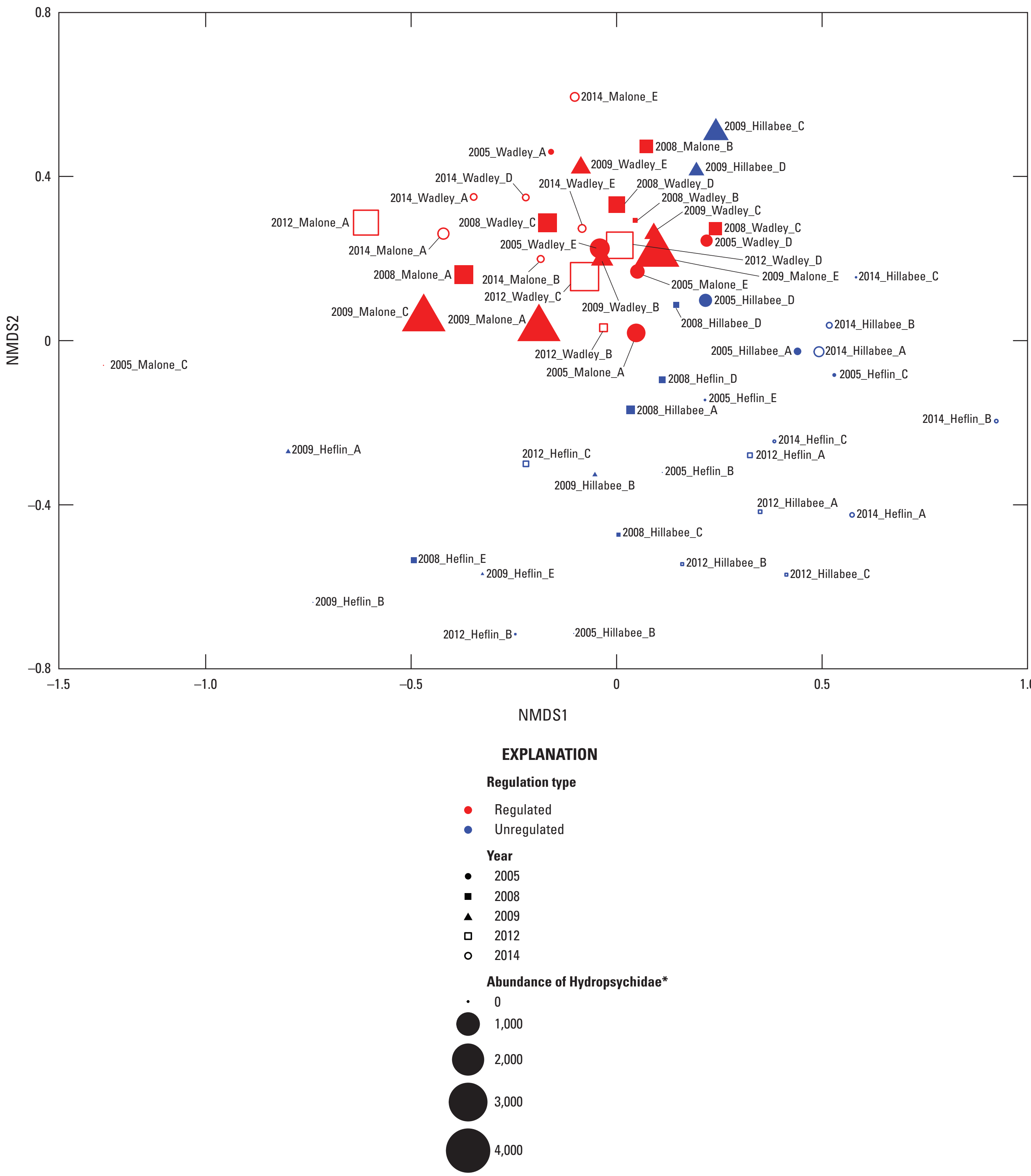

Figure C8. Nonmetric multidimensional scaling (NMDS) bubble plot of the abundance of Hydropsychidae* (* indicates early instar) observed at each site (year, reach, and shoal) over the ordination of all data.

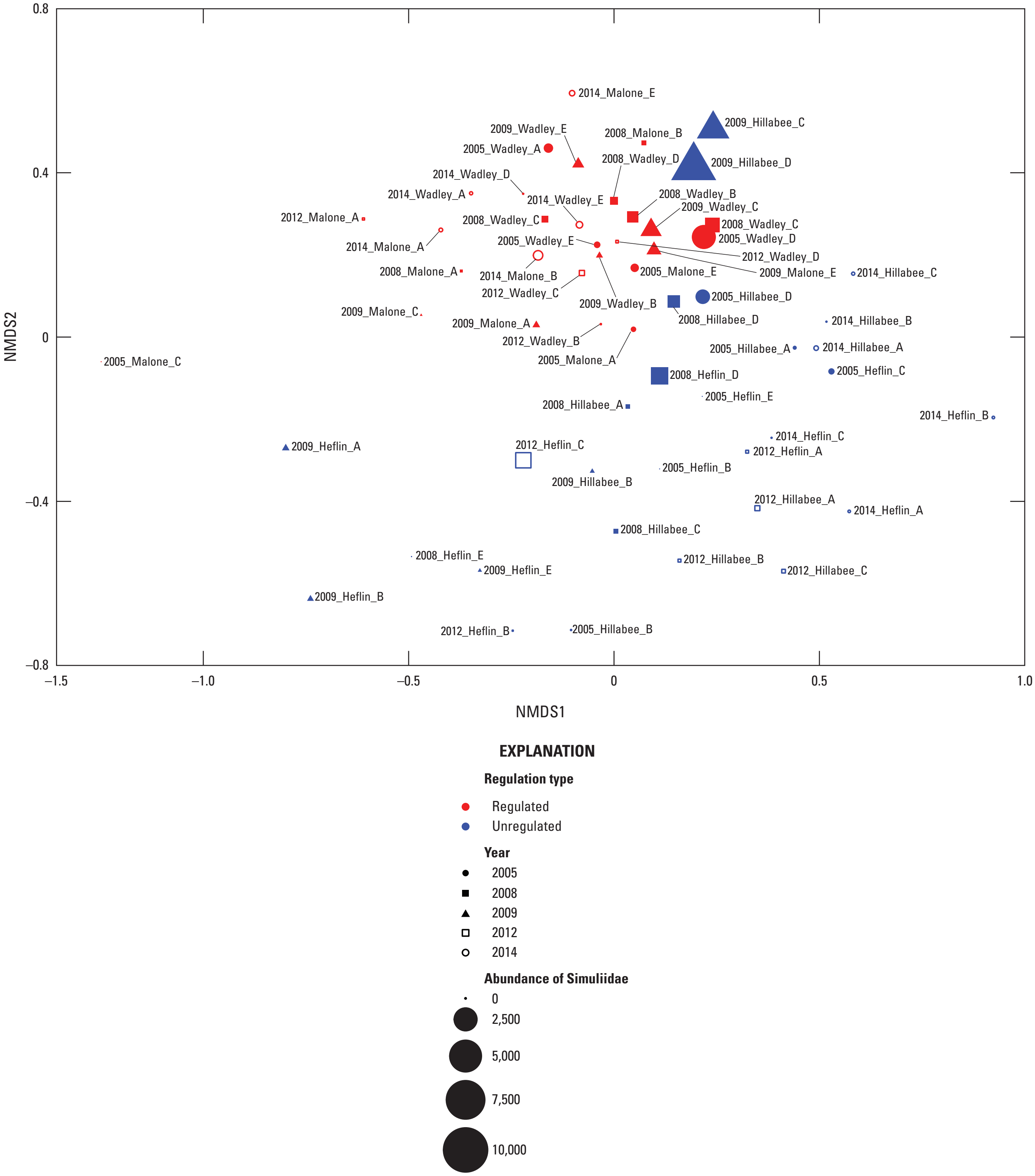

Figure C9. Nonmetric multidimensional scaling (NMDS) bubble plot of the abundance of *Simuliidae* observed at each site (year, reach, and shoal) over the ordination of all data.

Table C9. Results of analysis of fit for functional feeding group and habit class vectors on the initial nonmetric multidimensional scaling.

[R^2, coefficient of determination; *p*-value, probability value]

Class vector	R^2	p-value
Functional feeding group		
Filterer/collector	0.3269	<0.001
Gatherer/collector	0.7023	<0.001
Omnivore	0.0547	0.214
Predator	0.3297	<0.001
Scraper	0.5439	<0.001
Shredder	0.0171	0.644
Habit		
Burrower	0.5650	<0.001
Climber	0.1073	0.047
Clinger	0.4863	<0.001
Sprawler	0.0362	0.377
Swimmer	0.3603	<0.001

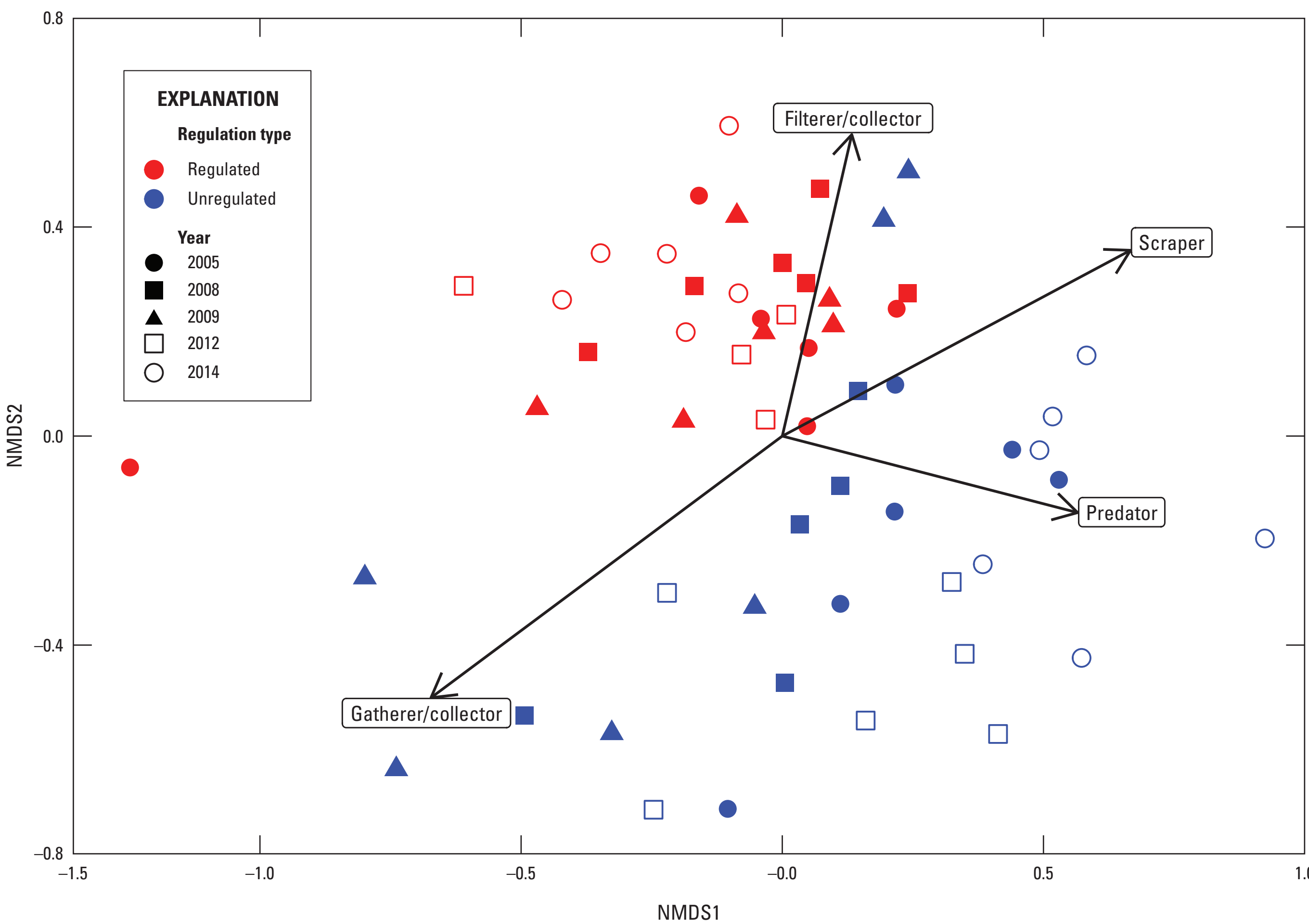

Figure C10. Nonmetric multidimensional scaling (NMDS) ordination with significant functional feeding group class vectors indicating the relation between the ordination of macroinvertebrate community composition and percent abundance of each class of functional feeding group.

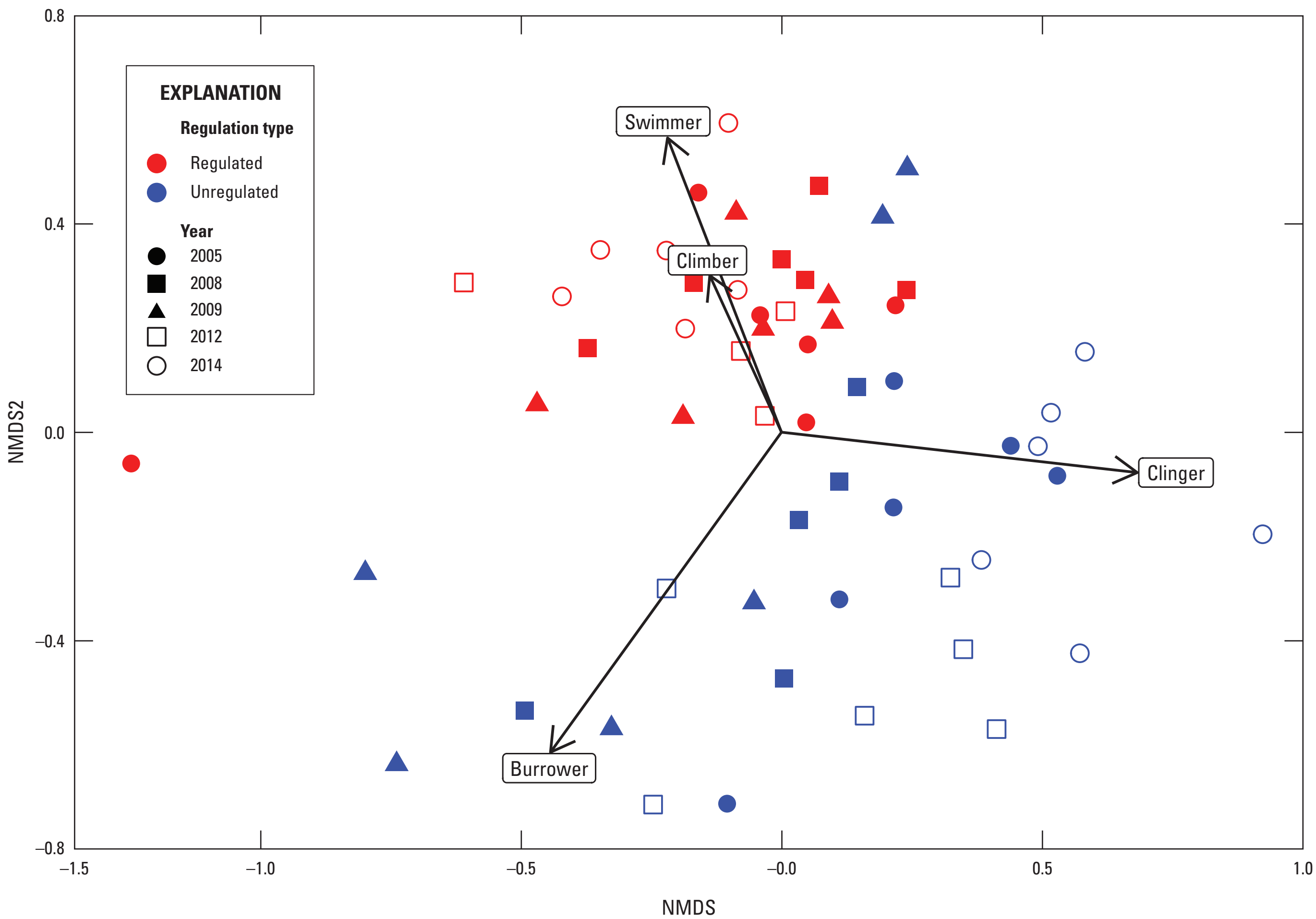

Figure C11. Nonmetric multidimensional scaling (NMDS) ordination with significant habit class vectors indicating the relation between the ordination of macroinvertebrate community composition and percent abundance of each class of habit.

Cluster Analysis

Cluster analysis of the presence-absence taxa matrix (fig. C12) clearly shows two groupings divided primarily by regulation type after the first cluster divide, with a few exceptions for each: 2008_Hillabee_D and 2014_Hillabee_C cluster with the regulated reaches, and 2005_Malone_C and 2009_Malone_C cluster amongst the unregulated reaches. Among the regulated reaches, all observations of Malone_A (shoal closest to the dam) clustered on their own branch after the second split, along with the 2005_Malone_E and 2008_Malone_B shoals, indicating a unique taxa composition in the Malone_A reach even amongst all regulated sites, especially in the 2012 and 2014 years. Similarly, all 2009_Heflin sites, along with the 2009_Heflin_E and 2005 and 2009 Malone_C sites, cluster on the opposite end of the graph after the third split, indicating a somewhat unique composition amongst unregulated sites. Most unregulated sites from 2012 share a branch after the fourth split (except 2012_Heflin_A).

Presence-Absence Similarity Percentage Analysis

Results of the SIMPER analysis of the presence-absence taxa matrix of the taxa analyzed within this study indicated that, of the 164 taxa identified, 30 taxa contributed to 43 percent of the variation between the communities of each regulation type (table C10). Among those taxa contributing the most variation in the presence-absence taxa matrix of the data, those most associated with regulated reaches were *Cheumatopsyche* morphotype E, Turbellaria, Hydrobiidae, and *Lepidostoma*, among others; and those taxa most associated with unregulated reaches were *Hexagenia*, Ceratopogonidae, *Neoperla*, *Oulimnius*, and Perlidae*, among others (see table C10).

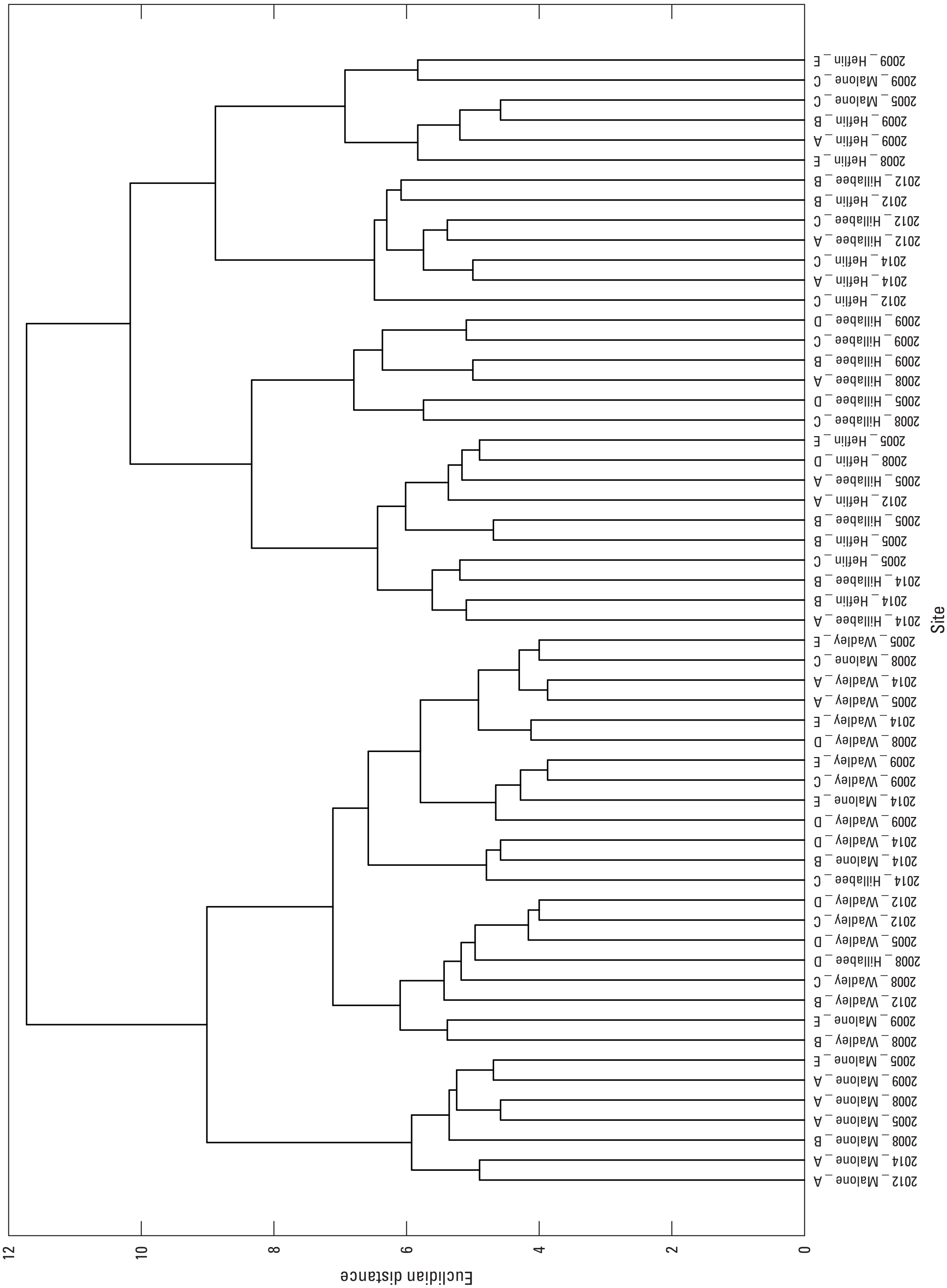

Figure C12. Cluster analysis of the presence-absence taxa matrix for all sites (year-reach-shoal average) analyzed.

Table C10. Top 30 results of the similarity percentage analysis of the presence-absence taxa matrix. The mean column shows the mean number of positive identifications for that taxon for each regulation type, and the cumulative sum of deviation is the total deviation that taxon contributes to the overall similarity of the community observed.

Rank	Family	Mean		Cumulative sum of deviation
		Regulated	Unregulated	
1	*Cheumatopsyche* E	0.85714	0.06897	0.02204
2	Turbellaria	0.78571	0.24138	0.04029
3	*Hexagenia*	0.00000	0.58621	0.05669
4	Ceratopogonidae	0.25000	0.65517	0.07273
5	*Neoperla*	0.39286	0.75862	0.08831
6	Hydrobiidae	0.71429	0.41379	0.10346
7	*Oulimnius*	0.46429	0.82759	0.11845
8	Perlidae*	0.46429	0.82759	0.13338
9	*Lepidostoma*	0.57143	0.20690	0.14797
10	*Micrasema*	0.42857	0.65517	0.16253
11	Ephemerellidae*	0.57143	0.27586	0.17706
12	Leptoceridae*	0.57143	0.34483	0.19151
13	*Microcylloepus pusillus*	0.67857	0.48276	0.20575
14	Empididae	0.57143	0.34483	0.21999
15	Elmidae*	0.46429	0.51724	0.23395
16	*Stenelmis*_adult	0.53571	0.51724	0.24788
17	*Oecetis*	0.60714	0.55172	0.26167
18	*Optioservus*	0.53571	0.72414	0.27542
19	Ferrissia	0.25000	0.48276	0.28892
20	*Acerpenna*	0.78571	0.55172	0.30232
21	*Baetis*	0.82143	0.55172	0.31571
22	*Tricorythodes*	0.42857	0.37931	0.32903
23	Tipulidae	0.32143	0.41379	0.34208
24	Baetidae*	0.92857	0.55172	0.35511
25	*Cheumatopsyche* A/B	0.42857	0.34483	0.36802
26	Nematoida	0.78571	0.58621	0.38078
27	*Cheumatopsyche*	0.67857	0.68966	0.39329
28	Capniidae/Leuctridae complex	0.00000	0.44828	0.40576
29	*Nectopsyche*	0.82143	0.62069	0.41808
30	*Hydropsyche*	0.96429	0.58621	0.43033

*Indicates early instar individual.

Summary of Results

- Most families are identified in samples from both regulation types; however, regulated reaches lack a substantial part of the Plecoptera and Coleoptera diversity detected in unregulated reaches, as well as several Odonata and Dipteran taxa, whereas unregulated reaches lack a single Ephemeroptera and two Dipteran families detected in the regulated reaches.
- Regulated reaches were characterized by greater density, whereas unregulated reaches were characterized by greater richness. These results reflect those commonly reported in the literature (Fuller and others, 2010; Robinson, 2012; Holt and others, 2015).
- Ephemeroptera dominate regulated reaches, driven by the large abundance of *Iswaeon*, and Chironomidae dominate unregulated reaches because of large abundances of the taxa Chironominae (all years) and Orthocladiinae (all except 2014; table C5).
- Regulated reaches were dominated by Ephemeroptera in most years because of a large abundance of *Iswaeon* and *Acerpenna*, whereas unregulated reaches were dominated by Dipterans in most years, driven by a large presence of Chironominae (within the top five for all years), Orthocladiinae (all but 2014), and Simuliidae (2008 and 2009).
- Based on total observed density of macroinvertebrates, many of the dominant macroinvertebrate families are shared between regulated and unregulated reaches, as would be expected for geographically similar sites (table C6); however, Plecoptera (especially Capniidae/Leuctridae complex in 2012 and *Neoperla* in 2014) and Coleoptera (notably *Stenelmis* in 2014) were much more common in unregulated reaches, indicating habitat conditions that are unique to unregulated reaches that allow these taxa to thrive.
- NMDS ordination of the species over site data (2D stress=0.2247, fig. C4) shows a tight clustering of regulated reaches towards the positive NMDS2 axis and unregulated reaches towards the negative NMDS2 axis, highlighting consistent differences in the composition of the macroinvertebrate community between regulated and unregulated reaches.
- NMDS ordination shows a much smaller range for the scattering of regulated sites, indicating much less variation in community composition among sites and years.
- A gradient of water years along the NMDS1 axis is present for unregulated years, with the latter 2012 and 2014 years ordinated towards the positive NMDS1 axis and other years ordinated more towards the center of the NMDS1 axis. Regulated reaches have more overlap within water years, indicating homogeneity of community composition across most samples.
- The points that ordinate closer to the opposite regulation type are the regulated sites 2008_Wadley_C and 2005_Wadley_D, which are the regulated sites most positively ordinated on the NMDS1 axis, and the unregulated sites 2009_Hillabee_C and 2009_Hillabee_D, which are the most positively ordinated unregulated sites along the NMDS2 axis. This ordination pattern is likely driven by abnormally large abundances of Simuliidae documented at these sites (fig. C9).
- The PERMANOVA results indicated significant differences among the Euclidian distances of the taxa composition among regulation type, year, as well as the response of water years in each regulation type (p<0.001 for all parameters).
- SIMPER analysis indicated that 30 of 164 taxa contributed to 84.8 percent of the variation in the Euclidian distances of the Wisconsin square root transformed taxa density between regulated and unregulated sites.
- The taxa Chironominae, Orthocladiinae, *Iswaeon*, Hydropsychidae*, and Simuliidae contribute the most to the cumulative deviation between regulated and unregulated sites (table C8). Whereas the graph of Chironominae site abundance (fig. C5) is somewhat unclear, Orthocladiinae (fig. C6) and Simuliidae (fig. C9) have clearly been detected in large abundances in a handful of sites each. *Iswaeon* (fig. C7) and Hydropsychidae* (fig. C8) show a preference for regulated reaches.
- Fitting FFG and habit classes to NMDS ordination, 4 of 7 FFGs (filterer/collectors, gatherer/collectors, omnivores, and scrapers) and 4 of 5 habits (burrowers, climbers, clingers, and swimmers) indicate a significant association with the ordination of samples over macroinvertebrate space, indicating that biological traits may be useful in describing broad changes within the community composition.
- The vector for swimmers and climbers ordinates towards the positive NMDS2 axis along with the regulated sites (fig. C11); whereas gatherer/collectors, predators, burrowers, and clingers ordinate to the negative NMDS2 axis (figs. C10 and C11).
- Cluster analysis shows that the underlying taxonomic assemblage detected is different between regulated and unregulated sites, with regulated and unregulated reaches mostly dividing on the first cluster split with a few exceptions among each group. Among the regulated samples, all five Malone A years plus two other

Malone sites separated on the second split, indicating a unique taxonomic assemblage at Malone A (site closest to the dam) among all the regulated shoals. Similarly, all 2009_Heflin sites share a branch with a couple other sites after the fifth split, indicating a somewhat unique taxonomic assemblage at these sites.

- SIMPER analysis utilizing macroinvertebrate presence-absence data indicated 30 taxa contributed to 43 percent of the variation in Euclidian distances between regulated and unregulated macroinvertebrate communities.
- The taxa *Cheumatopsyche* morphotype E and Turbellaria are more common in regulated sites and contribute the most to the cumulative deviation in the Euclidian distances between the regulated and unregulated communities (table C10). Alternatively, *Hexagenia*, Ceratopogonidae, *Neoperla*, *Oulimnius*, and Perlidae* are more common in unregulated reaches versus regulated reaches.

Discussion

This study has demonstrated, through the analysis of taxonomic assemblage, NMDS of macroinvertebrate density data with supporting PERMANOVA tests, and cluster analysis of the presence-absence taxa matrix, that the macroinvertebrate communities downstream from R.L. Harris Dam are significantly different from communities that are not subject to flow regulation. Macroinvertebrate communities in reaches subject to flow regulation have regularly been characterized in the literature by lower richness (Barbour and others, 1999; Robinson, 2012; Holt and others, 2015), greater overall density (Barbour and others, 1999; Cortes and others, 2002; Holt and others, 2015), and more tolerant taxa (Cortes and others, 2002; Bednarek and Hart, 2005) versus unregulated reaches. Observations of the mid-Tallapoosa River system reflect these results (table C4), indicating a disturbed community composition downstream from R.L. Harris Dam as a result of flow regulation.

Macroinvertebrate Community Composition

Samples from all reaches had large proportions of their populations dominated by the Diptera, Ephemeroptera, and Trichoptera orders. Many samples from regulated reaches had a larger proportion of early instar Ephemeroptera, especially from the genera *Iswaeon* and *Acerpenna*, and Trichoptera, especially from the family Hydropsychidae, whereas unregulated reaches generally had larger proportions of Dipterans. These taxa are common in riverine systems, maturing quickly and reproducing several times per year, especially in the warmer regions of the southern United States (Mackay, 1992; Wallace and Anderson, 1996). They are also commonly noted as some of the first taxa to colonize riverine habitats subject to flow disturbances (Clifford and others, 1992; Mackay, 1992; Robinson, 2012) and have been recorded as present in both regulated and unregulated reaches (Camargo and Voelz, 1998; Bruno and others, 2016; Ellis and Jones, 2016).

These taxa are equipped with some adaptations to aid in coping with spates of high flow. Baetidae are commonly cited as strong swimmers and can potentially find refuge during spates of high flow (Clifford and others, 1992; Mackay, 1992; Robinson, 2012; Timusk and others, 2016; Vinson, 2001). Hydropsychidae and some Chironomidae (for example, *Rheotanytarsus*) use silken retreats that allow them to hold fast during turbulent flow conditions (Clifford and others, 1992; Mackay, 1992); furthermore, Hydropsychidae prefer smooth stones free of algae for oviposition, and Baetidae have been reported to feed on thin epilithic films (Mackay, 1992). These strategies are likely the reason that these taxa dominate in reaches that are regularly exposed to scouring flows.

Macroinvertebrate Community Response to Natural Variation in Hydrology

Daily hydropeaking activities in regulated reaches can restrict many taxa from establishing in downstream reaches, resulting in large populations of generalists (Clifford and others, 1992). In the neighboring Chattahoochee River system (not shown), Holt and others (2015) reported no difference in the macroinvertebrate community composition among water years in reaches directly below the Buford Dam. Our results agree with their generalization, with Ephemeroptera, Diptera, and Trichoptera orders dominating regulated reaches consistently in all water years, whereas unregulated reaches had more variability in dominant orders, including some years with large proportions of Coleoptera, Plecoptera, and Neotaenioglossa.

Dipterans dominated both regulated and unregulated reaches in 2005 and 2009, led primarily by large abundances of Chironominae except in 2009 where Orthocladiinae composed 14 percent of regulated samples. The 2005 year included inclement weather events, including Hurricanes Dennis and Katrina affecting the Gulf Coast, which may have increased flow events, input large trees and other debris into the system, and temporarily increased pollutant runoff into the river system, potentially increasing the composition of Chironominae and Orthocladiinae in the unregulated reaches. The 2009 year saw the beginning of a wet period (fig. C3) where increased hydrologic stress from flooding events may have excluded many predators and allowed both Chironominae and Simuliidae to thrive. The increased observations of these taxa were not permanent, and we would need to analyze additional samples for the 2006 and 2007 years to begin to describe the mechanisms in colonization and persistence parameters during these years.

Nonmetric Multidimensional Scaling

Ordination of the estimated average macroinvertebrate densities indicates grouping of the data based on regulation type (fig. C4). Regulated reaches have a much smaller range for the ordination of points on the NMDS, reflecting the homogenization and predictability of flows in reaches subject to regulation. These data support the idea of the antidrought condition proposed by McMahon and Finlayson (2003), as well as increased predictability of flows (Graf, 2006; Olden and Naiman, 2010) where extreme conditions, such as drought or flood, are not experienced in flow regulated reaches because of the practice of mitigation of flow conditions that prevent these extreme flow events (except where mitigation measures require emergency release or retention of flows from the dam to meet upstream and downstream water demands). Unregulated reaches, on the other hand, are subject to the natural variation in hydrology, including annual or seasonal droughts and floods, and, therefore, can support greater variability within the macroinvertebrate community composition.

Several sites ordinate in a manner that indicates similarity in the communities at these regulated and unregulated sites. It is notable that the unregulated sites are all from the Hillabee reach, which has a higher water table than the other unregulated reach (Heflin), and that the two sites ordinated most in the positive NMDS2 directions are from the wet (2009) year. Upon closer inspection of the communities at these sites, a common feature among all these sites are large abundances of *Iswaeon* (fig. C7), Simuliidae (fig. C9), Heptageniidae*, and Hydrobiidae. Based on the distribution of abundances of *Iswaeon* and Simuliidae, these ordinations could be primarily driven by observations of large groups of these taxa. As noted in previous sections, the Baetidae are strong swimmers; however, the affinity of *Iswaeon* to regulated reaches indicates a particularly successful life-history strategy for dealing with spates of high flow used by this taxon as opposed to other Baetidae taxa. Simuliidae are also commonly noted as successful during high flow events because they utilize hooks and silk on the posterior abdominal segment to maintain attachment to the substrate (Merritt and others, 2012). The Baetidae and Simuliidae taxa are also commonly noted as drifting during extreme flow conditions (Bruno and others, 2016; Timusk and others, 2016), indicating that conditions were acceptable for breeding in the days before sampling to create these large clusters of individuals, and no catastrophic drift events had taken place in the several days that would have encouraged these individuals to drift.

Distribution Trends in Macroinvertebrate Functional Feeding and Habit Groups

The ordination of the FFG and habit vectors agreed with many generalizations reported in the literature. Gatherer/collectors are commonly generalist species that are typically noted as the first taxa to colonize after hydrologic disturbances (Clifford and others, 1992). Our findings agreed as these taxa ordinated strongly towards wet years, indicating that these taxa were able to thrive and dominate in the wet years.

Although it seems contradictory for burrowers to also be strongly associated with flood years as scouring flows generally eliminate burrowing substrate from the habitat, the persistence of *Hexagenia*, a burrowing mayfly that requires fine substrate for burrowing, indicated that flood years may not have affected the substrate and habitat composition in unregulated reaches the same way as daily pulses from a hydropeaking dam affect the regulated reaches. Additionally, many of the observations for the burrowing group were of the common and quickly reproducing taxa Orthocladiinae, which are known to thrive in regulated streams and are resistant to flood disturbances (Rader and Ward, 1988; Mackay, 1992). Many of these taxa are also strongly associated with drought years in the regulated reaches, which follows as burrowing taxa are more likely to survive spates of dewatering by retreat into the sediments for cooler temperatures and wetter conditions. Although burrowers were not eliminated from regulated reaches (that is, Orthocladiinae, *Corbicula fluminea*), taxa requiring fine burrowing sediments (that is, *Hexagenia*) were not observed in the regulated reaches. The absence of these taxa and the dominance of generalist feeders may reflect the change in substrate composition below the dam, where regular sheer stress from hydropeaking releases likely eliminates suitable habitat and potential foraging resources for other functional feeding and habit groups from the reaches.

Scrapers and clingers were strongly ordinated towards the positive x-axis. Scrapers include the gastropods, Elmidae, Baetidae, and Heptageniidae, among others. These taxa are prevalent in both regulated and unregulated reaches, reflecting the ordination of the scraper vector. The clinger vector, on the other hand, is more closely associated with unregulated reaches. Many taxa in the clinger group have adaptations to handle spates of high flows but may begin catastrophic drift during serious disturbances such as flood (Wallace and Anderson, 1996; Robinson, 2012). Many clinger taxa are Heptageniidae, Elmidae, and Simuliidae taxa, which are well known to thrive in disturbed conditions, whereas other taxa (Cortes and others, 2002; Robinson, 2012; Timusk and others, 2016), such as Perlidae and Elmidae do not thrive as well under high sheer stress conditions (Gore and others, 2001; Fuller and others, 2010). With many sensitive taxa composing the clinger habit, the clinger vector is more closely associated with “normal” water conditions in unregulated reaches versus regulated reaches or disturbed years for unregulated reaches.

Sprawlers and predators have a strong ordination towards the negative NMDS2 axis and unregulated drought/normal years (fig. C10 and fig. C11). Many sprawlers are from the Baetiscidae, Caenidae, and Capniidae/Leuctridae complex families, and predators include the water mites, Megaloptera, Odonata, Plecoptera, and a few Trichoptera genera. These taxa are not noted as resistant to flow disturbances, likely because of lacking adaptations to successfully thrive in reaches that experience frequent pulses, or because of longer lifespans and fewer opportunities to reproduce compared to multivoltine taxa.

Climbers and swimmers were both strongly associated with regulated reaches. Climbers commonly observed included the disturbance intolerant Trichoptera families Leptoceridae and Lepidostomatidae. Several large clusters of these taxa (>10 individuals) were observed, mostly in the 2012 and 2014 years (drought and bookend years, respectively), after periods of low flow with few or no peaking events in the weeks before sampling. These taxa could be taking advantage of available habitat area in regulated reaches during spates of low flow through either egg deposition, drift, or migration, and it is likely these large groups detected within regulated reaches are driving the ordination of the climber vector. Swimmers, such as Baetidae, are well adapted for thriving in regulated reaches (see discussion above). These results reflect the dominance of Baetidae and flow resistant taxa in the regulated reaches, and the inaccessibility of potential habitat in regulated reaches to disturbance sensitive taxa during normal peaking operations.

Cluster Analysis

The results of our cluster analysis of macroinvertebrate presence-absence data indicated that regulation was one of the primary factors influencing community composition. The analysis also indicates a unique community composition at the Malone_A shoal even among regulated reaches. The Malone_A shoal is closest to the dam and experiences the highest flow ramping rates because of hydropeaking (Meile and others, 2011; Toffolon and others, 2010). Samples from Malone_A were generally restricted to a large number of a few taxa because of daily catastrophic pulses eliminating many individuals from the immediate habitat area (Meile and others, 2011; Timusk and others, 2016). The restriction in community composition is likely the primary cause for regulated and unregulated reaches forming individual clusters at the first split, as well as for Malone_A reaches clustering together early in the dendrogram.

Identification of Indicator Taxa for the Mid-Tallapoosa River System

The results from the SIMPER analysis of the average estimated species abundance per shoal matrix identified that half of the deviation between regulation types could be explained by less than 10 taxa (table C8). Most of these taxa include those commonly noted by taxonomists as increasing in response to disturbances, such as Chironomidae, Baetidae, and Hydropsychidae taxa (Clifford and others, 1992; Mackay, 1992; Holt and others, 2015). The bubble plot NMDS clearly shows greater abundance within the regulated reaches for *Iswaeon* (fig. C7) and Hydropsychidae* (fig. C8), where these taxa are commonly detected in large groups of early instar individuals. Observations for Orthocladiinae (fig. C6) and Simuliidae (fig. C9) indicate a few sites with large abundances of these taxa and are likely responsible for driving the ordination of these sites further from their group. For example, the large proportion of Simuliidae in the 2009_Hillabee_C and D shoals is likely the reason these two points ordinated so strongly to the positive NMDS2.

The results of the SIMPER analysis of the presence-absence taxa matrix list the taxa that are least similar with regards to their presence or absence in different regulation types and highlight where some taxa can thrive, whereas other taxa may be excluded because of habitat constraints. The top-ranking taxon for deviation in similarities of sites based on the presence-absence taxa matrix (table C10), *Cheumatopsyche* morphotype E (associated with *C. etrona*; Burington, 2011), is detected in greater proportions in regulated reaches versus unregulated reaches (table C8, rank 20). *Cheumatopsyche* morphotype E is also one of the easiest *Cheumatopsyche* morphotypes to identify as it has unique morphological characteristics (large vetromental plates and a large excision from the frontoclypeus) not seen in any other *Cheumatopsyche* morphotype or *Hydropsyche* genus. Similar to *Cheumatopsyche* morphotype E, *Cheumatopsyche* morphotypes A/B (table C10, rank 25) and *Hydropsyche* sp. (table C10, rank 30) were also more prevalent in regulated reaches. Other *Cheumatopsyche* sp. were identified in similar proportions in regulated reaches and unregulated reaches (table C10, rank 27); however, both *Hydropsyche* and *Cheumatopsyche E* were detected in greater abundances (table C8, ranks 14 and 20, respectively) and proportions (table C10, ranks 30 and 1, respectively) in regulated reaches versus unregulated reaches. Our results indicated that many Hydropsychidae taxa were more associated with regulated reaches both in terms of presence-absence and density and would be useful indicators of habitat quality.

Hexagenia, the third ranking taxa contributing to deviations between the similarities of regulation types based on the presence-absence taxa matrix (table C10), are burrowers that may be excluded from regulated reaches because of lack of favorable habitat of fine or sandy substrate. Habitat in the regulated reaches can be characterized as large patches of scoured bedrock with occasional pockets of vegetation and gravel that lack deep layers of sediment, sand, gravel, and coarse rocks that are more favorable for burrowing taxa. Similarly, *Neoperla* (table C10, rank 5), Perlidae* (table C10, rank 8) and other sprawlers are less common in regulated reaches, perhaps because of the lack of consistent flows generally favored by these long-living taxa (Cortes and others, 2002). Turbellaria (table C10, rank 2) and Hydrobiidae (table C10, rank 6) were both present in more samples from regulated reaches than unregulated reaches, perhaps because of high flow tolerance, decreased exposure during low flow periods, increased feeding opportunities, or decreased predation (Fisher and LaVoy, 1972; Richards and others, 2014).

Summary

The R.L. Harris Dam Adaptive Management Project began in 2005 as a management method that would continually monitor the outcome of the agreed upon new flow management plan, referred to as "the Green Plan," report any findings of changes in the macroinvertebrate or fish populations as a result of the new flow management scheme, and suggest management actions to consider as future management alternatives. The team set forth to establish a protocol for analyzing the historical macroinvertebrate collections that included identifying the minimum number of samples required to reflect the macroinvertebrate community at each shoal to estimate reach-level differences in the macroinvertebrate community, establishing a subsampling protocol for processing macroinvertebrate samples in the laboratory, and selecting several years and seasons to analyze in which the potential maximum variability of macroinvertebrate communities was based on natural variation in water availability. The goal was to characterize the macroinvertebrate communities in regulated and unregulated reaches of the Tallapoosa River and Hillabee Creek tributary, characterize the macroinvertebrate community in both regulation types during different natural hydraulic conditions, examine the distribution trends in macroinvertebrate functional feeding and habit groups, and identify taxa that will be useful for monitoring changes in the macroinvertebrate community in both regulated and unregulated reaches of the Tallapoosa River.

The results of this study indicate that the macroinvertebrate community downstream from the R.L. Harris Dam in the regulated river reaches near Malone, Alabama, and Wadley, Ala., of the Tallapoosa River are different from those macroinvertebrate communities observed in both the unregulated Tallapoosa River reaches upstream from R.L. Harris Dam in Heflin, Ala., and the unregulated Hillabee Creek Tributary that intersects with the Tallapoosa River south of the Wadley reaches. Although the effects of changes to flow and temperature as a result of hydropeaking are less severe as distance from the dam increases, our results indicate that flow and temperature remain in a nonnatural state in regulated reaches of the Tallapoosa River from R.L. Harris Dam downstream to Wadley, Ala., as a result of hydropeaking that the community in regulated reaches show many dissimilarities to communities from unregulated river reaches. Fewer dissimilarities between the macroinvertebrate community composition in the regulated reaches from the first and last years analyzed than between the regulated and unregulated reaches of the last year to be analyzed indicate that the Green Plan did not improve the quality of flow conditions to shift the macroinvertebrate community in regulated reaches to more closely reflect the community composition found in unregulated river reaches.

References Cited

Barbour, M.T., Gerritsen, J., Snyder, B.D., and Stribling, J.B., 1999, Rapid bioassessment protocols for use in streams and wadeable rivers—Periphyton, benthic macroinvertebrates, and fish, 2d ed.: Washington, D.C., U.S. Environmental Protection Agency, Office of Water, EPA 841–B–99–002, 337 p.

Bednarek, A.T., and Hart, D.D., 2005, Modifying dam operations to restore rivers—Ecological responses to Tennessee River dam mitigation: Ecological Applications, v. 15, no. 3, p. 997–1008. [Also available at https://doi.org/10.1890/04-0586.]

Bejarano, M.D., Jansson, R., and Nilsson, C., 2017, The effects of hydropeaking on riverine plants—A review: Biological Reviews, v. 93, no. 1, p. 658–673. [Also available at https://doi.org/10.1111/brv.12362.]

Bruno, M.C., Cashman, M.J., Maiolini, B., Biffi, S., and Zolezzi, G., 2016, Responses of benthic invertebrates to repeated hydropeaking in semi-natural flume simulations: Ecohydrology, v. 9, no. 1, p. 68–82. [Also available at https://doi.org/10.1002/eco.1611.]

Burington, Z.L., 2011, Larval taxonomy, phylogeny and historical biogeography of the genus Cheumatopsyche (Trichoptera—Hydropsychidae) in North America: Clemson, S.C., Clemson University, Master's thesis, 44 p., accessed November 2016 at https://tigerprints.clemson.edu/cgi/viewcontent.cgi?article=2077&context=all_theses.

Camargo, J.A., and Voelz, N.J., 1998, Biotic and abiotic changes along the recovery gradient of two impounded rivers with different impoundment use: Environmental Monitoring and Assessment, v. 50, no. 2, p. 143–158. [Also available at https://doi.org/10.1023/A:1005712024049.]

Clifford, H.F., Casey, R.J., and Saffran, K.A., 1992, Short-term colonization of rough and smooth tiles by benthic macroinvertebrates and algae (chlorophyll a) in two streams: Journal of the North American Benthological Society, v. 11, no. 3, p. 304–315. [Also available at https://doi.org/10.2307/1467650.]

Cohen, J., 1988, Statistical power analysis for the behavioral sciences, 2d ed.: Hillsdale, N.J., L. Erlbaum Associates, 567 p.

Cortes, R.M.V., Ferreira, M.T., Oliveira, S.V., and Oliveira, D., 2002, Macroinvertebrate community structure in a regulated river segment with different flow conditions: River Research and Applications, v. 18, no. 4, p. 367–382. [Also available at https://doi.org/10.1002/rra.679.]

Elliott, J.M., 2008, The ecology of riffle beetles (Coleoptera: Elmidae): Freshwater Reviews, v. 1, no. 2, p. 189–203. [Also available at https://www.fba.org.uk/journals/index.php/FRJ/article/viewFile/107/36.]

Ellis, L.E., and Jones, N.E., 2016, A test of the serial discontinuity concept—Longitudinal trends of benthic invertebrates in regulated and natural rivers of northern Canada: River Research and Applications, v. 32, no. 3, p. 462–472. [Also available at https://doi.org/10.1002/rra.2861.]

Epler, J.H., 2001, Identification manual for the larval Chironomidae (Diptera) of North and South Carolina—A guide to the taxonomy of the midges of the southeastern United States, including Florida: Palatka, Fla., St. Johns River Water Management District Special Publication SJ2001–SP13, 526 p.

Epler, J.H., 2010, The water beetles of Florida—An identification manual for the families Chrysomelidae, Curculionidae, Dryopidae, Dytiscidae, Elmidae, Gyrinidae, Haliplidae, Helophoridae, Hydraenidae, Hydrochidae, Hydrophilidae, Noteridae, Psephenidae, Ptilodactylidae and Scirtidae: Tallahassee, Fla., Florida Department of Environmental Protection, 414 p.

Fisher, S.G., and LaVoy, A., 1972, Differences in littoral fauna due to fluctuating water levels below a hydroelectric dam: Journal Fisheries Research Board of Canada, v. 29, no. 10, p. 1472–1476.

Freeman, M.C., Bowen, Z.H., Bovee, K.D., and Irwin, E.R., 2001, Flow and habitat effects on juvenile fish abundance in natural and altered flow regimes: Ecological Applications, v. 11, no. 1, p. 179–190.

Fuller, R.L., Griego, C., Muehlbauer, J.D., Dennison, J., and Doyle, M.W., 2010, Response of stream macroinvertebrates in flow refugia and high-scour areas to a series of floods—A reciprocal replacement study: Journal of the North American Benthological Society, v. 29, no. 2, p. 750–760. [Also available at https://doi.org/10.1899/09-107.1.]

Gore, J.A., Layzer, J.B., and Mead, J., 2001, Macroinvertebrate instream flow studies after 20 years—A role in stream management and restoration: River Research and Applications, v. 17, no. 4–5, p. 527–542. [Also available at https://doi.org/10.1002/rrr.650.]

Graf, W.L., 2006, Downstream hydrologic and geomorphic effects of large dams on American rivers: Geomorphology, v. 79, no. 3–4, p. 336–360. [Also available at https://doi.org/10.1016/j.geomorph.2006.06.022.]

Heddinghaus, T.R., and Sabol, P., 1991, A review of the Palmer Drought Severity Index and where do we go from here? *in* Conference on Applied Climatology, 7th, Salt Lake City, Utah, 1991, Proceedings: Salt Lake City, Utah, American Meteorological Society, p. 242–246.

Holt, C.R., Pfitzer, D., Scalley, C., Caldwell, B.A., and Batzer, D.P., 2015, Macroinvertebrate community responses to annual flow variation from river regulation—An 11-year study: River Research and Applications, v. 31, no. 7, p. 798–807. [Also available at https://doi.org/10.1002/rra.2782.]

Irwin, E., Kennedy, K.M., Goar, T.P., Martin, B., and Martin, M.M., 2011, Adaptive management and monitoring for restoration and faunal recolonization of Tallapoosa River shoal habitats: Montgomery, Ala., Alabama Cooperative Fish and Wildlife Research Unit, 49 p., accessed June 2016 at https://www.outdooralabama.com/sites/default/files/Research/SWG%20Reports/Tallapoosa%20Shoals%20Final%20Report.pdf.

Kathman, R.D., and Brinkhurst, R.O., 1998, Guide to the freshwater oligochaetes of North America: College Grove, Tenn., Aquatic Resources Center, 264 p.

Kennedy, T.A., Muehlbauer, J.D., Yackulic, C.B., Lytle, D.A., Miller, S.W., Dibble, K.L., Kortenhoeven, E.W., Metcalfe, A.N., and Baxter, C.V., 2016, Flow management for hydropower extirpates aquatic insects, undermining river food webs: Bioscience, v. 66, no. 7, p. 561–575. [Also available at https://doi.org/10.1093/biosci/biw059.]

Mackay, R.J., 1992, Colonization by lotic macroinvertebrates—A review of processes and patterns: Canadian Journal of Fisheries and Aquatic Sciences, v. 49, no. 3, p. 617–628. [Also available at https://doi.org/10.1139/f92-071.]

McArdle, B.H., and Anderson, M.J., 2001, Fitting multivariate models to community data—A comment on distance-based redundancy analysis: Ecology, v. 82, no. 1, p. 290–297.

McMahon, T.A., and Finlayson, B.L., 2003, Droughts and anti-droughts—The low flow hydrology of Australian rivers: Freshwater Biology, v. 48, no. 7, p. 1147–1160. [Also available at https://doi.org/10.1046/j.1365-2427.2003.01098.x.]

McManamay, R.A., and Frimpong, E.A., 2015, Hydrologic filtering of fish life history strategies across the United States—Implications for stream flow alteration: Ecological Applications, v. 25, no. 1, p. 243–263.

Meile, T., Boillat, J.L., and Schleiss, A.J., 2011, Hydropeaking indicators for characterization of the Upper-Rhone River in Switzerland: Aquatic Sciences, v. 73, no. 1, p. 171–182. [Also available at https://doi.org/10.1007/s00027-010-0154-7.]

Merritt, R.W., Cummins, K.W., and Berg, M.B., eds., 2012, An introduction to the aquatic insects of North America, 4th ed.: Dubuque, Iowa, Kendall Hunt Publishing, 1,158 p.

Miller, S.W., and Judson, S., 2014, Responses of macroinvertebrate drift, benthic assemblages, and trout foraging to hydropeaking: Canadian Journal of Fisheries and Aquatic Sciences, v. 71, no. 5, p. 675–687. [Also available at https://doi.org/10.1139/cjfas-2013-0562.]

Morse, J.C., McCafferty, W.P., Stark, B.P., and Jacobus, L.M., 2017, Larvae of the southeastern USA mayfly, stonefly, and caddisfly species: Clemson, S. Car., Clemson University Public Service Publishing, 482 p.

National Climate Data Center, 2016, November, CDO Divisional Select: National Oceanic and Atmospheric Administration web page, accessed August 2016 at https://www7.ncdc.noaa.gov/CDO/CDODivisionalSelect.jsp.

Oksanen, J., Blanchet, F.G., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.H., Szoecs, E., and Wagner, H., 2017, vegan—Community ecology package: R package version 2.4–5, accessed November 2017 at https://cran.r-project.org/web/packages/vegan/index.html.

Olden, J.D., and Naiman, R.J., 2010, Incorporating thermal regimes into environmental flows assessments—Modifying dam operations to restore freshwater ecosystem integrity: Freshwater Biology, v. 55, no. 1, p. 86–107. [Also available at https://doi.org/10.1111/j.1365-2427.2009.02179.x.]

Palmer, W.C., 1965, Meteorological drought: Washington, D.C., U.S. Weather Bureau, Office of Climatology, research paper no. 45, 56 p., accessed December 3, 2016, at https://www.ncdc.noaa.gov/temp-and-precip/drought/docs/palmer.pdf.

Poff, N.L., Olden, J.D., Vieira, N.K.M., Finn, D.S., Simmons, M.P., and Kondratieff, B.C., 2006, Functional trait niches of North American lotic insects—Traits-based ecological applications in light of phylogenetic relationships: Journal of the North American Benthological Society, v. 25, no. 4, p. 730–755. [Also available at https://doi.org/10.1899/0887-3593(2006)025[0730:FTNONA]2.0.CO;2.]

Poff, N.L., and Zimmerman, J.K.H., 2010, Ecological responses to altered flow regimes—A literature review to inform the science and management of environmental flows: Freshwater Biology, v. 55, no. 1, p. 194–205. [Also available at https://doi.org/10.1111/j.1365-2427.2009.02272.x.]

R Core Team, 2016, R—A language and environment for statistical computing: Vienna, Austria, R Foundation for Statistical Computing, accessed November 2017 at https://www.R-project.org/.

Rader, R.B., and Ward, J.V., 1988, Influence of regulation on environmental conditions and the macroinvertebrate community in the upper Colorado River: River Research and Applications, v. 2, no. 5, p. 597–618. [Also available at https://doi.org/10.1002/rrr.3450020505.]

Rehn, A.C., 2009, Benthic macroinvertebrates as indicators of biological condition below hydropower dams on west slope Sierra Nevada streams, California, USA: River Research and Applications, v. 25, no. 2, p. 208–228. [Also available at https://doi.org/10.1002/rra.1121.]

Reiss, P.T., Stevens, M.H.H., Shehzad, Z., Petkova, E., and Milham, M.P., 2010, On distance-based permutation tests for between-group comparisons: Biometrics, v. 66, p. 636–643. [Also available at https://doi.org/10.1111/j.1541-0420.2009.01300.x.]

Richards, R.R., Gates, K.K., and Kerans, B.L., 2014, Effects of simulated rapid water level fluctuations (hydropeaking) on survival of sensitive benthic species: River Research and Applications, v. 30, no. 8, p. 954–963. [Also available at https://doi.org/10.1002/rra.2692.]

Robinson, C.T., 2012, Long-term changes in community assembly, resistance, and resilience following experimental floods: Ecological Applications, v. 22, no. 7, p. 1949–1961. [Also available at https://doi.org/10.1890/11-1042.1.]

Rossi, J.P., 2011, rich—An R package to analyse species richness: Diversity, v. 3, no. 1, p. 112–120. [Also available at https://doi.org/10.3390/d3010112.]

Slowikowski, K., 2016, ggrepel—Automatically position non-overlapping text labels with 'ggplot2': R package ver. 0.5, accessed September 2017 at https://cran.r-project.org/web/packages/ggrepel/index.html.

Stark, J.D., Boothroyd, I.K.G., Harding, J.S., Maxted, J.R., and Scarsbrook, M.R., 2001, Protocols for sampling macroinvertebrates in wadeable streams: New Zealand Macroinvertebrate Working Group Report no. 1, 65 p.

Timusk, E.R., Smokorowski, K.E., and Jones, N.E., 2016, An experimental test of sub-hourly changes in macroinvertebrate drift density associated with hydropeaking in a regulated river: Journal of Freshwater Ecology, v. 31, no. 4, p. 555–570. [Also available at https://doi.org/10.1080/02705060.2016.1193064.]

Toffolon, M., Siviglia, A., and Zolezzi, G., 2010, Thermal wave dynamics in rivers affected by hydropeaking: Water Resources Research, v. 46, no. 8, article W08536, accessed September 2017 at https://doi.org/10.1029/2009WR008234.

Tupinambás, T.H., Cortes, R.M.V., Hughes, S.J., Varandas, S.G., and Callisto, M., 2016, Macroinvertebrate responses to distinct hydrological patterns in a tropical regulated river: Ecohydrology, v. 9, no. 3, p. 460–471. [Also available at https://doi.org/10.1002/eco.1649.]

U.S. Army Corps of Engineers, 2015, Alabama-Coosa-Tallapoosa river basin water control manual—Appendix I: Mobile, Ala., U.S. Army Corps of Engineers, South Atlantic Division, Mobile District, 153 p.

U.S. Geological Survey, 2018, USGS water data for the Nation: U.S. Geological Survey National Water Information System database, accessed May 2018 at https://doi.org/10.5066/F7P55KJN.

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R., and Cushing, C.E., 1980, The river continuum concept: Canadian Journal of Fisheries and Aquatic Sciences, v. 37, no. 1, p. 130–137. [Also available at https://doi.org/10.1139/f80-017.]

Vinson, M.R., 2001, Long-term dynamics of an invertebrate assemblage downstream from a large dam: Ecological Applications, v. 11, no. 3, p. 711–730. [Also available at https://doi.org/10.1890/1051-0761(2001)011[0711:LTDOAI]2.0.CO;2.]

Wallace, J.B., and Anderson, N.H., 1996, Habitat, life history, and behavioral adaptations of aquatic insects, chap. 5 *of* Merritt, R.W., and Cummins, K.W., eds., An Introduction to the Aquatic Insects of North America, 3d ed.: Dubuque, Iowa, Kendall Hunt Publishing Company, p. 41–73.

Ward, J.H., 1963, Hierarchical grouping to optimize an objective function: Journal of the American Statistical Association, v. 58, no. 301, p. 236–244. [Also available at https://doi.org/10.2307/2282967.]

Ward, J.V., and Stanford, J.A., 1982, Thermal responses in the evolutionary ecology of aquatic insects: Annual Review of Entomology, v. 27, no. 1, p. 97–117. [Also available at https://doi.org/10.1146/annurev.en.27.010182.000525.]

White, J.C., Hannah, D.M., House, A., Beatson, S.J.V., Martin, A., and Wood, P.J., 2017, Macroinvertebrate responses to flow and stream temperature variability across regulated and non-regulated rivers: Ecohydrology, v. 10, no. 1, article e1773. [Also available at https://doi.org/10.1002/eco.1773.]

Wickham, H., 2007, Reshaping data with the reshape package: Journal of Statistical Software, v. 21, no. 12, p. 1–20. [Also available at https://doi.org/10.18637/jss.v021.i12.]

Wickham, H., 2009, ggplot2—Elegant graphics for data analysis: New York, Springer-Verlag New York, 90 p.

Appendix C1. Standard Operating Procedures—Sorting Protocol

Introduction

Benthic macroinvertebrates are diverse and ubiquitous organisms documented in many freshwater habitats and are commonly used as indicators of habitat quality and stability. The objective of sorting macroinvertebrates is to obtain a representative of the sample that can be used to characterize the community structure from which the sample was taken without spending an excessive amount of effort in picking out every specimen. By the end of your training, you should be able to (1) determine how long a sample should take to process, (2) estimate the proportion of the sample that needs to be picked to effectively characterize the sample community, (3) identify higher orders of select macroinvertebrates, and (4) understand how to properly label and document your work effort.

Sorting Objectives

Our target count for the number of specimens per sample is 100–150 if that many specimens are in the sample. It is likely that some samples may not contain half as many specimens, and for those samples, our goal will be to pick out every specimen. In addition to specimen counts, our goal is to pick our target number of specimens (or the whole sample) within 2 hours. This protocol should help guide you in making the decisions that will enable you to achieve these objectives. You should plan on spending about 3 hours for each sample. Keep in mind, once you start a sample, you should try to finish it rather than leaving it unfinished for another time.

Materials

- Sorting pans—2
- Sieve (250 micrometer)
- Spoon and (or) putty knife
- Petri dishes—6
- Stereo scope
- Light source
- Benthic freshwater macroinvertebrate identification sheet
- Sorting record sheet
- Pencil
- 75 percent ethanol
- Waste bucket
- Funnel
- Magnifying lens
- Shell vials and caps
- Data labels
- Timer

Detailed Procedures

Locate a sample from a crate in the main laboratory to be sorted. Wash the sieve and pans with water and a brush. Also wash the Petri dishes if they look like they are not clean. Place the funnel in the waste bucket and place the sieve on top of the funnel. Unscrew the sample cap and pour the contents of the sample into the sieve while ensuring that the sieve is secure by holding onto it. Remove the sieve from the funnel and place over a large sorting pan. Use water to wash the remaining contents of the sample cup through the sieve. Turn the water on before aiming it into the sample cup to make sure that the water does not flow out too fast, causing material and specimens to splash outside of the sieve. Also rinse the sample cap over the sieve and check both the sample cup (especially around the lip) and cover to ensure that no specimens remain. Once all the material from the cup is washed into to the sieve, wash the contents through the sieve with water to remove very small particles. During this part of the process, locate the sample label that was inside the sample cup, wash it with water over the sieve to remove excess material, and record its information on the sorting record sheet, benthic freshwater macroinvertebrate identification sheet, and vial labels for your sort of this sample. On the sorting record sheet, record today's date (date picked), the sample date, reach, sample, start time, and scientist (your initials). The rest of the information will be filled out at the end of the sort. On the benthic freshwater macroinvertebrate identification sheet, record the collection date (same as the sample date), reach, sample, sorted by (your initials), and sort date. Additional information will be filled out at the end of your sort. Find the premade label representing the collection reach of your sample and record the sample/collection date and sample number (for example, B05) for six labels. On each of the six labels, include one of the following: "EP" for Ephemeroptera and Plecoptera, "T" for Trichoptera, "Chi" for chironomids and other dipterans, "Gen" for general insects that do not fit into the previous categories, "Other" for taxa that are not insect but included in the benthic fauna (for example, mollusks), and "No Count" for specimens that may be counted but either are not identifiable in their current condition or lack an identifiable head or mouthpart. Place each label into a separate vial with the written information facing out.

After you have washed the sample, place all the contents into a sorting tray using as little water as possible. Spread the sample material out so that it is "homogenized." Use the putty knife or spoon to place a manageable amount of material into a Petri dish. A manageable amount should consist of about a single layer of material or less. Place the same amount of material into the remaining Petri dishes. If the sample contains an amount material that will fill more than 6 Petri dishes, divide the homogenized material into equal portions within the sorting tray so that 1 portion could fit into 4–6 Petri dishes.

Set your timer for 15 minutes and begin sorting material from one of the Petri dishes under the stereoscope. Systematically sort through material as instructed so that you can estimate the proportion of material sorted from the Petri dish after 15 minutes has expired. Use the counter to record each specimen that you pick as belonging to the groups represented by each label (you will receive additional training to recognize different aquatic invertebrate groups). After 15 minutes expire, use the number of specimens you sorted and the proportion of the Petri dish and number of Petri dishes to estimate the total number of specimens expected from this sample (tables C1.1–C1.3); however, if your specimens consist mostly of mollusks, do not include their numbers to determine the amount of material to subsample. For example, if you have split the sample into 5 Petri dishes, and you sorted 10 specimens ("bugs") from 40 percent of your first Petri dish, then you have sorted 8 percent of the sample in the first 15 minutes (table C1.1), and the estimated number of specimens for the sample is 125 (table C1.2), which falls in our target range of 100–150 specimens, indicating that you should be prepared to sort all the material from the sample (table C1.3). At this point, finish picking specimens from this Petri dish and continue sorting all contents of each Petri dish or the number of Petri dishes for the estimated proportion of material needed to reach our target count. If 100 specimens are picked before all Petri dishes are

Table C1.1. Estimate of sample sorted in 15 minutes. Find the estimated "percent of sample" sorted after 15 minutes based on the portion sorted in one Petri dish and total number of Petri dishes the sample is divided into.

[%, percent]

Portion of dish picked in 15 minutes	Number of Petri dishes you split the sample into									
	1	2	3	4	5	6	7	8	9	10
5%	5%	3%	2%	1%	1%	1%	1%	1%	1%	1%
10%	10%	5%	3%	3%	2%	2%	1%	1%	1%	1%
15%	15%	8%	5%	4%	3%	3%	2%	2%	2%	2%
20%	20%	10%	7%	5%	4%	3%	3%	3%	2%	2%
25%	25%	13%	8%	6%	5%	4%	4%	3%	3%	3%
30%	30%	15%	10%	8%	6%	5%	4%	4%	3%	3%
35%	35%	18%	12%	9%	7%	6%	5%	4%	4%	4%
40%	40%	20%	13%	10%	8%	7%	6%	5%	4%	4%
45%	45%	23%	15%	11%	9%	8%	6%	6%	5%	5%
50%	50%	25%	17%	13%	10%	8%	7%	6%	6%	5%
55%	55%	28%	18%	14%	11%	9%	8%	7%	6%	6%
60%	60%	30%	20%	15%	12%	10%	9%	8%	7%	6%
65%	65%	33%	22%	16%	13%	11%	9%	8%	7%	7%
70%	70%	35%	23%	18%	14%	12%	10%	9%	8%	7%
75%	75%	38%	25%	19%	15%	13%	11%	9%	8%	8%
80%	80%	40%	27%	20%	16%	13%	11%	10%	9%	8%
85%	85%	43%	28%	21%	17%	14%	12%	11%	9%	9%
90%	90%	45%	30%	23%	18%	15%	13%	11%	10%	9%
95%	95%	48%	32%	24%	19%	16%	14%	12%	11%	10%
100%	100%	50%	33%	25%	20%	17%	14%	13%	11%	10%

Table C1.2. Estimate of total number of specimens in the sample. Use the "percent of sample" from table C1.1 and the number of specimens (bugs) picked during the 15-minute sort to estimate the total number of specimens in the sample.

[no., number; %, percent]

No. of bugs	Percent of sample																								
	1%	2%	3%	4%	5%	6%	7%	8%	9%	10%	11%	12%	13%	14%	15%	16%	17%	18%	19%	20%	21%	22%	23%	24%	25%
5	500	250	167	125	100	83	71	63	56	50	45	42	38	36	33	31	29	28	26	25	24	23	22	21	20
10	1,000	500	333	250	200	167	143	125	111	100	91	83	77	71	67	63	59	56	53	50	48	45	43	42	40
15	1,500	750	500	375	300	250	214	188	167	150	136	125	115	107	100	94	88	83	79	75	71	68	65	63	60
20	2,000	1,000	667	500	400	333	286	250	222	200	182	167	154	143	133	125	118	111	105	100	95	91	87	83	80
25	2,500	1,250	833	625	500	417	357	313	278	250	227	208	192	179	167	156	147	139	132	125	119	114	109	104	100
30	3,000	1,500	1,000	750	600	500	429	375	333	300	273	250	231	214	200	188	176	167	158	150	143	136	130	125	120
35	3,500	1,750	1,167	875	700	583	500	438	389	350	318	292	269	250	233	219	206	194	184	175	167	159	152	146	140
40	4,000	2,000	1,333	1,000	800	667	571	500	444	400	364	333	308	286	267	250	235	222	211	200	190	182	174	167	160
45	4,500	2,250	1,500	1,125	900	750	643	563	500	450	409	375	346	321	300	281	265	250	237	225	214	205	196	188	180
50	5,000	2,500	1,667	1,250	1,000	833	714	625	556	500	455	417	385	357	333	313	294	278	263	250	238	227	217	208	200

sorted, finish sorting through your current Petri dish and stop (even if your estimate was to do more Petri dishes). Record the total proportion of material you sorted from the sample and record the number of specimens that were picked on the sorting record sheet and the benthic freshwater macroinvertebrate identification sheet. The sorting process should take about 2 hours.

Table C1.3. Subsample estimator. Based on the total number of specimens estimated in table C1.2, determine how much of the sample you will likely need to sort.

[>, greater than]

Estimated number of bugs in sample	Suggested proportion
<150	1
150–300	3/4
300–450	1/2
450–600	1/4
600–750	1/6
750–1,000	1/8
1,000+	1/10

If you reached your target count without finishing the sample, make a new data label with the sample site information and name it "last pick." Combine the remaining unsorted material into a sorting pan and use the magnifying glass to search for as many different specimens as you can find within 10 minutes (in addition to the 2 hours used for the timed sort). It is acceptable to pick several specimens that look the same, but do not spend all your time picking the same type of specimen. For example, if a sample contains hundreds of snails, pick about 5–10 of these during your "last pick" but try to spend more time looking for other invertebrates, especially ones that look different from what you have already picked during the timed pick. Place these specimens into the "last pick" vial and do not count them.

After you are finished sorting, place all the contents back into the original sample cup with a new inside label and enough 75 percent ethanol to cover more than 150 percent of the material. Place all the sample vials and the original sample label into a Ziploc bag with the reach, date, and sample information written on the outside of the bag with a Sharpie pen. Place the sample bag into the bin labeled with the same reach and place the benthic freshwater macroinvertebrate identification sheet in the in the appropriate bin. Wash the sieve, Petri dishes, and sorting pans with water, using a brush.

Outline of Procedures

1. Locate the sample cup and sorting materials; make sure Petri dishes, trays, and sieve are clean.
2. Pour contents into the sieve and wash contents.
3. Locate the sample label and record information on the sorting record sheet, benthic freshwater macroinvertebrate identification sheet, and vial labels.
4. Divide the sample evenly into several Petri dishes or into proportions that can accommodate a set of Petri dishes.
5. Complete the 15-minute sort and use tables C1.1 to C1.3 to estimate how much of the sample needs to be sorted.
6. Sort material until you reach 100 specimens; you must complete the Petri you are sorting even if it ends up being more than 100 specimens.
7. Record the number of specimens and proportion of sample sorted.
8. If necessary, perform a 10-minute "last pick" on the proportion of material that was not sorted with the magnifying glass.
9. Place the sorted material back into the original sample container with a new label.
10. Place the sorted specimens into a labeled Ziploc bag with the original sample label.

Appendix C2. Macroinvertebrate Data

The sum of total observations for each macroinvertebrate taxon at all sites, listed alphabetically by class, order, family, and taxon, is provided in table C2.1 (available at https://doi.org/10.3133/ofr20191026).

Table C2.1. Sum of total observations for each macroinvertebrate taxon at all sites, listed alphabetically by class, order, family, and taxon.

Sampling site on Hillabee Creek. Taken July 1, 2011, by B. Martin, Alabama Cooperative Fish and Wildlife Research Unit.

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