Theses and Dissertations

12-9-2022

Assessing the legacy of erosion and flood control management efforts on the fish assemblages and physical conditions of Yazoo Basin bluff hill streams

Nicky M. Faucheux
nmhfaucheux@gmail.com

Recommended Citation

Assessing the legacy of erosion and flood control management efforts on the fish assemblages

and physical conditions of Yazoo Basin bluff hill streams

By

Nicky Marie Hahn Faucheux

Approved by:

Leandro E. Miranda (Major Professor)
W. Todd Slack (Committee Member)
Beth Baker (Committee Member)
Michael E. Colvin (Committee Member)
Kevin M. Hunt (Committee Member & Graduate Coordinator)
L. Wes Burger (Dean, College of Forest Resources)

A Dissertation
Submitted to the Faculty of
Mississippi State University
in Partial Fulfillment of the Requirements
for the Degree of Doctor of Philosophy
in Forest Resources
in the Department of Wildlife, Fisheries, and Aquaculture

Mississippi State, Mississippi

December 2022

Name: Nicky Marie Hahn Faucheux

Date of Degree: December 9, 2022

Institution: Mississippi State University

Major Field: Forest Resources

Major Professor: Leandro E. Miranda

Title of Study: Assessing the legacy of erosion and flood control management efforts on the fish assemblages and physical conditions of Yazoo Basin bluff hill streams

Pages in Study: 79

Candidate for Degree of Doctor of Philosophy

The hills of Yazoo Basin have a long history of land use modification and subsequent erosion and flood control issues. In response, federal actions were taken to address these issues beginning after the Mississippi River flood of 1927. Four major flood control reservoirs were built in 1932-1957, and instream low-drop grade control structures (GCS) were installed beginning in the 1980s. The objective of my dissertation was to ascertain the long-term effects of these efforts on stream fish assemblages and channel morphology. To assess whether the reservoirs affected upstream fish assemblages as barriers to recolonization by fluvial fishes or as source population for invasion by lentic generalist fishes (Chapter 1), I used stream data collected 43-61 years after the rivers were impounded to test for differences in fish assemblages between sites upstream and downstream of the reservoirs. Analysis of catch per effort and diversity metrics displayed little influence of the reservoirs, but trait-based analysis revealed marginal increases in planktivores, herbivores, detritivores, and generalists in upstream assemblages. After determining that potential effects of reservoirs would not confound further analysis, I assessed the effects of GCS on channel morphology (Chapter 2) and fish assemblages (Chapter 3) 30 years post-installation. To assess GCS effects on channel morphology, stream

cross-sections were used to calculate Bank Height Ratio, Width/Depth Ratio, and Entrenchment Ratio, while point estimates made along the transects were used to calculate the average sediment size distribution. Analyses revealed that the GCS were successful in checking channel incision moving headward in the streams: sites upstream of the GCS were less incised and had greater accumulations of fine substrates compared to downstream sites and sites on streams lacking erosion control structures. The GCS could potentially affect fish assemblages through habitat modification or by selectively filtering the assemblages as a barrier to upstream migration. Analysis of beta diversity revealed that diversity was driven by species replacement rather than nestedness, which indicates GCS were not acting as filters on the assemblages. Analysis of catch per effort data confirmed differences in assemblage structure that echoed the instream habitat differences revealed in Chapter 2.

ACKNOWLEDGEMENTS

I'd like to thank both my family and the WFA grad students for their love and support, without which I doubt I could have finished. Finally, a huge thank you to my husband, Mitch, for his patience and willingness to help out however needed.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER I

REEVALUATING EXPECTATIONS FOR IMPOUNDMENT EFFECTS ON

STREAM FISH ASSEMBLAGES

Introduction

Because of its unique biogeographic history, the southeastern US is an epicenter of fish

biodiversity (McAllister et al. 1986; Jelks et al. 2008). Yet, many streams in the region have

experienced significant habitat degradation due to anthropogenic watershed alterations including

deforestation, channelization, and dam construction (Warren et al. 2002; Bennet and Rhoton

2009). Dams are commonly used to control the flow regimes of rivers, depending on both small

impoundments and large flood control reservoirs to mitigate flooding caused by the large amount

of precipitation the southeast receives annually. Over 17,000 dams (and their associated

reservoirs) have been constructed in the Tennessee, Lower Mississippi, and South Atlantic Gulf

river basins (Graff 1999; U.S. Army Corps of Engineers 2018). Reservoirs may influence stream

fish assemblages through changes in habitat, hydrology, and isolation effects, although

disentangling reservoir effects from other factors that influence patterns in fish assemblage

structure remains a challenge.

Reservoirs change stream conditions throughout impounded watersheds. Immediately

upstream of the impoundment, the rivers and tributaries gain lentic properties: flow is reduced or

eliminated, width and depth increase, and suspended sediment settles out of the water column.

Downstream of the impoundment, the sequence of flooding events often becomes decoupled

from the natural flow regime due to reservoir releases that, dependent on how water storage and the outlet are engineered and managed, may alter temporal flows and physicochemical properties of the water such as turbidity, temperature, and dissolved oxygen levels (Hannan 1979). Because the outlet of the dam is constrained within a defined bed, the river downstream of the reservoir is often channelized. The decrease in riverbed elevation caused by channelization causes headcuts (i.e., downcutting of the stream bed in an upstream direction) to degrade tributaries (Whitten and Patrick 1981; Simon 1989; Biedenharn et al. 1997). All these physical changes can drastically change the type, amount, and quality of aquatic habitat available for fish assemblages. The streams become incised, and instream habitat progressively changes due to cycles of scouring and aggradation (Schumm 1984; Simon 1989, 1994). During these cycles, the riffle-run-pool sequence common in streams is converted to channelized runs, resulting in homogenous habitat conditions that reduce the number of functional niches for stream fish. As the stream banks become steeper, connection to the floodplain and associated wetlands is reduced, which can also have implications for the biotic community (Junk et al. 1989).

In addition to changes in habitat, reservoirs can play a direct role in altering fish assemblages by altering connectivity between populations. Winston et al. (1991) reported four native fluvial specialist cyprinids absent upstream of Altus Dam on the Red River in Oklahoma despite their presence elsewhere in the drainage. They theorized that as the intermittent streams desiccated during dry seasons, the cyprinid species migrated into the reservoir, where they became naïve prey to piscivores often abundant in reservoirs. The loss of longitudinal connectivity caused by the dam prohibited recolonization from downstream populations. Moreover, species with drifting larvae may require large reaches of free-flowing river habitat and are unable to persist when their larvae or eggs drift into a reservoir and are preyed on or settle to

2

the substrate prematurely. This shift in fish assemblage has been supported by multiple studies who have reported extirpations, higher number of generalists, and shifts in fish assemblages upstream of reservoirs (e.g., Herbert and Gelwick 2003; Falke and Gido 2006; Guenther and Spacie 2006; Kashiwagi and Miranda 2009; Sá-Oliveira et al. 2015; Hedden et al. 2018; Reuter et al. 2019). Isolation effects and community shifts have also been documented for other aquatic species, including crayfishes (Barnett et al. 2021).

These trends are common in impounded river basins and may need to be factored in when investigating environmental drivers that structure fish assemblages in impounded basins. The Upper Yazoo Basin in Northern Mississippi typifies a southeastern watershed: deforestation and conversion to row-crop agriculture in the 1800s, followed by decades of topsoil erosion. The streams in the Upper Yazoo Basin were dredged and channelized to remove excess water and sediment, and four large flood control reservoirs were constructed on the mainstem rivers (Dabney et al. 2012). The tributary streams became extremely incised as the gradient change downstream caused scouring of the tributaries. In response, several federal projects reforested parts of the basin and installed low-drop grade control structures (GCS) to prevent headcuts from moving further upstream (Duffey and Ursic 1991).

I investigated the degree to which fish assemblages in the Yazoo Basin had been impacted by the large flood control reservoirs. Based on the literature, I expected that the reservoirs altered native fish assemblages by reducing network connectivity and changing habitat quality above and below reservoirs. Specifically, I expected that fish assemblages in tributaries upstream of a reservoir would include greater representation of tolerant lentic generalists, especially piscivorous species common in southern reservoirs, while the assemblages in tributaries downstream from the influence of the tailrace of the dam would retain a fluvial

3

assemblage complete with a greater emphasis on intolerant stream specialists. Moreover, I

expected some stream species to be absent above the reservoirs as occasional droughts, siltation,

or changes in other environmental conditions following over half a century of impoundment may

have produced localized extirpations due to the lack of recolonization from downstream reaches

blocked by dams.

Methods

Study area

I analyzed existing fish data from streams of the bluff hill region in the eastern Yazoo

Basin in north Mississippi (Figure 1.1). This area consists of six subbasins that discharge into the

Yazoo River. Five of the rivers in these subbasins (the Coldwater, Little Tallahatchie, Yocona,

Skuna, and Yalobusha) are impounded by four flood-control reservoirs (Arkabutla, Sardis, Enid,

and Grenada reservoirs) ranging from 4,800-14,500 ha in size that were built between 1938 and

1956. Annual discharge of the five rivers upstream of the impoundments ranges 17 to 51 m^3s^{-1}

(U.S. Geological Survey 2021). Streams in the region have a long history of erosion and

sedimentation due to the highly erodible loess soils and the conversion of the native hickory-oak

forests into agricultural lands in the mid to late 1800s (Hilgard 1860; Dabney et al. 2012).

Federal programs were introduced in the 1940s to reduce overland erosion and have been largely

successful, and other programs were initiated in the 1980s to reduce instream erosion (Williston

1988; Bledsoe et al. 2002). The regional species pool includes over 50 fish species, including

two endemics, the Yazoo Shiner *Notropis rafinesquei* and the Yazoo Darter *Etheostoma raneyi*

(Knight and Cooper 1987).

Site Selection:

I used data collected in the summers of 1999 and 2000 to represent fish assemblage status 43-61 years after the four reservoirs were impounded. Sites were selected from a pool of 49 sites that had been sampled by Arkansas State University and the USDA-ARS National Sediment Laboratory. Although a robust analysis would include a balanced representation of sites upstream and downstream of the reservoirs, the majority of available sites in the impounded subbasins (Coldwater River, Little Tallahatchie River, Yocona River, and Yalobusha River subbasins) were located upstream of the reservoirs. To bolster the representation of the downstream fish assemblages, I included sites from two neighboring un-impounded subbasins (Tallahatchie River and Upper Yazoo River) that contain tributary streams that flow directly into the Yazoo River, which is formed by the confluence of the impounded tributary rivers. Within these limits, sites were further selected such that (1) no two sites were on the same stream, (2) each site was at least 2.5 km downstream of instream GCS (Biedenharn et al. 1997), with preference given to sites on streams without instream GCS, and (3) sites were located at least 50 m upstream of a road crossing to avoid potential confounding effects associated with the road. In all, 26 sites were retained for study. Fourteen of the sites were located on tributary streams 1-81 km upstream of the four reservoirs. Twelve sites were located on streams that joined the mainstem rivers at least 3 km downstream of the reservoir tailwaters and positioned 2-31 km above the confluence of the tributary and the river discharging from the reservoir.

Fish Sampling

Fish collections consisted of single-pass backpack electrofishing of a 200-m reach, with a target electrofishing time of 20 minutes (mean = 23 min, SD = 14 min). Sampling coincided with baseflow conditions during June-September 1999-2000. At each site, specific conductance was checked before sampling, and the voltage on the backpack unit was adjusted to maintain a

relatively constant power. Sampling was conducted in a zig-zag pattern in an upstream direction

and covered all habitat types present. Two netters accompanied the backpack electrofisher to

retrieve fish affected by the electric field. Fish larger than 10 cm in total length were identified

to species and returned to the stream after electrofishing was completed, while smaller fish were

anesthetized in a solution of MS-222, preserved in 10% buffered formalin, and transported to

Arkansas State University for later identification.

Analysis

The analysis used historical fish collections that were designed to document stream fish

assemblages to assess the barrier effect of the reservoirs on fish assemblages while accounting

for differences in river subbasin and stream size. Fish data were standardized as catch per unit of

effort (CPE) defined as the number of individuals of a given taxa collected in a single pass

divided by the time needed to complete the pass. I applied a multivariate analysis of covariance

to test if CPEs (multiple continuous variables) differed between tributaries upstream and

downstream of the reservoirs (categorical variable), while controlling for subbasin (random

categorical variable to control for potential differences in species pool) and catchment size

(continuous variable to account for differences in stream size). Subbasins in the region could

potentially support different fish assemblages, and catchment size influences discharge and

stream volume, as well as a multiplicity of other accompanying physical stream characteristics

that shape fish assemblages (Rodriguez-Iturbe and Rinaldo 1997; Altermatt 2013). Catchment

size for each site was obtained using the StreamStats program (U.S. Geological Survey 2020)

and transformed $\log_{10}$ for linearity since catchment tends to increase exponentially. The

multivariate analysis of covariance was run with a permutation MANCOVA (PERMANCOVA)

applied to an among-sites similarity matrix computed with the Bray-Curtis similarity index

implemented on the transformed CPE values. Species CPE were fourth-root transformed to reduce right-skewness. Non-metric multidimensional scaling (NMDS) was also applied to the resemblance matrix to interpret graphically the results of the PERMANCOVA.

Research has suggested that analysis of assemblages organized by functional group rather than taxonomic group can highlight relationships between fish assemblages and environmental conditions (Hoeinghaus et al. 2007), especially changes in stream conditions due to dams (Smith et al. 2017). To assess my hypotheses regarding higher CPE of tolerant, lentic generalists in the upstream tributaries and higher lotic specialists in the downstream tributaries, I repeated the multivariate PERMANCOVA and NMDS analyses outlined above using alternative functional group descriptors including physicochemical tolerances, habitat preferences, and trophic guilds (Table 1.1). Tolerances (i.e., intolerant, moderate, tolerant) were assessed according to Meador and Carlisle (2007), habitat preferences (generalist, lentic, lotic) according to Frimpong and Angermeier (2009), and trophic guilds (detritivore, herbivore, invertivore, parasite, piscivore, planktivore) according to Goldstein and Simon (1999). For all three of these functional categorizations, species CPE values were summed across samples according to each category, then category CPEs were standardized to percentage composition across each sample. These values were then square-root transformed to reduce right-skewness. For each functional categorization, an among-sites matrix was constructed with the Bray-Curtis similarity index. Log_{10} transformed catchment sizes was included as a covariate because species differences in trophic guilds, tolerances, and habitat preferences are expected to change with increasing stream size (Vannote et al. 1980), but subbasin was not included as a covariate since analysis of functional guilds generalizes species' identity to their traits, which allows for community

7

comparisons between areas that are geographically distant (Troia and McManamay 2019). PRIMER-E version 7 software (Plymouth, UK) was used for all analyses.

Because one of the primary ways reservoirs can affect stream fish assemblages is by acting as barriers to recolonization after stochastic extinction events, I hypothesized tributaries upstream of the reservoir would have a lower species richness than downstream tributaries. Species richness at each site was estimated using individual-based rarefaction or Chao 1 extrapolation to facilitate comparisons between sites with unequal sample sizes (Chao et al. 2014). As noted by Colwell et al. (2012), extrapolation provides reliable estimates only up to roughly double the size of a sample, so I rarified or extrapolated all samples to a sample size of 50 fish (smallest catch was 27 fish). Rarefaction and extrapolation estimates were calculated using the *iNEXT* package in program *R* (Hsieh et al. 2016; R Core Team 2021). Potential differences in species richness of sites above and below reservoirs were assessed using a permutational analysis of covariance (PERANCOVA) applied to an among-sites similarity matrix computed with Euclidean distance, with $\log_{10}$ transformed catchment size as a covariate (Anderson 2017). The PERANCOVA was applied using PRIMER-E version 7 software (Plymouth, UK).

Results

Overall, 4,483 fish representing 58 species were collected in the 26 study sites (Table 1.1). Except for White Bass *Morone chrysops* and Smallmouth Buffalo *Ictiobus bubalus*, the assemblage was composed mostly of non-migratory fishes (Table 1.1). On average, 13 species were captured at each site with a mean of 172 (SD = 126) individuals per site. One upstream site was considered an outlier because the lab identification data were missing, leading to an extremely low sample size (N = 2 fish) and was removed from further analyses. Approximately

40% (24) of the species in my analysis were detected at only one or two sites. Relative to habitat preferences, 43% of the species were lotic, 14% lentic, and 43% generalists. Concerning tolerance, 57% of species were tolerant, 26% moderately tolerant, and 17% intolerant. Most of the species were either invertivores (69%) or piscivores (17%).

Contrary to expectations influenced by my literature review, there were no significant differences between the species composition of fish assemblages upstream and downstream of reservoirs (Pseudo $F = 1.5$, $P = 0.16$). Although the fish assemblage did change with increasing catchment size (Pseudo $F = 3.4$, $P = 0.0003$), there were no significant differences attributable to subbasin (Pseudo $F = 1.01$, $P = 0.45$), nor was the interaction between subbasin and sample location relative to upstream or downstream from the reservoir significant (Pseudo $F = 1.1$, $P = 0.36$). The lack of segregation between upstream and downstream sites is made apparent by the large overlap of fish composition among sites illustrated with an NMDS plot (Figure 1.2). Although the upstream sites seemed more dissimilar as a set (i.e., greater dispersion in Figure 1.2), their distribution overlapped entirely with the distribution of the downstream sites. Eight species found in the downstream sites were not captured from the upstream sites, and 19 species present in upstream sites were not captured from downstream sites; however, only three of these 27 species were present in more than two sites (Table 1.1). The majority of species present at more than two sites were common species found both upstream and downstream of the reservoirs, thus preventing them from being separated statistically.

No significant differences were detected between upstream and downstream sites for physicochemical tolerances (Pseudo $F = 0.96$, $P = 0.37$) or habitat preferences (Pseudo $F = 0.66$, $P = 0.48$), and similar to the species composition, sites above the reservoirs tend to have more variability in ordination space (Figure 1.3a, c). However, a marginal difference was apparent

when fish were grouped according to trophic guild (Pseudo F = 2.8, P = 0.04). The ordination (Figure 1.3b) displays some separation of upstream from downstream sites mostly due to a higher representation of planktivores, herbivores, and detritivores in upstream sites while downstream sites were dominated by insectivores. In general, both upstream and downstream sites were dominated by moderately tolerant to tolerant lotic specialists and by generalists, although a few upstream sites had a greater percentage of planktivores, herbivores, and detritivores.

The species richness analysis concurred with the species composition analysis. Overall, species richness was not markedly different in sites upstream or downstream from the reservoirs (Pseudo F = 0.62, P = 0.45). Species richness estimated at 50 individuals ranged from 7-14 species for the majority of sites (Figure 1.4), but in general upstream sites showed more variability. The confidence intervals for all estimates were quite narrow (typically within two species) indicating relatively good fits for both the rarefaction and extrapolation estimates (Chao et al. 2014).

Discussion

Contrary to expectations, I did not observe a compelling difference in fish assemblage in tributaries upstream or downstream of impoundments within my study region. This result represents an anomaly in light of the published literature (i.e., Winston et al. 1991; Herbert and Gelwick 2003; Falke and Gido 2006; Guenther and Spacie 2006; Kashiwagi and Miranda 2009; Oliveira et al. 2015; Hedden et al. 2018; Reuter et al. 2019). I present four hypotheses to account for this anomaly: (1) the history of land use in the Yazoo Basin, (2) the size of the tributaries investigated, (3) the effects of gear bias, and (4) a fish assemblage consisting mostly of non-migratory species. I consider each of these hypotheses below and argue that the effects of

impoundments on fish assemblage may be region-specific. I suggest that the observed anomaly can assist in refining expectations about fish assemblages and stream fish conservation in impounded river basins.

The Yazoo Basin, originally hickory-oak hardwood forest, was cleared and converted to agriculture starting in the mid-1800s. By 1940, over 60% of the forests in the basin had been cleared (U.S. Army Corps of Engineers 1981). As a result, the area has experienced significant soil loss and erosion, which in turn has led to degradation of the streams. By 1900, many of the streams in the region were completely aggraded with sand and silt (Williston 1988). In response, local communities dredged and channelized the streams (Leech and Biedenharn 2012). These instream alterations initiated a cycle of channelization, incision, and aggradation that has left a lasting legacy on area streams, despite federal programs to reduce overland erosion beginning in the 1940s and instream erosion in the 1980s (Leech and Biedenharn 2012). Based on this history, it is possible that species richness was reduced, and the fish assemblage was transformed and homogenized before faunal surveys were implemented, in such a way that any effect of the reservoirs on the fish assemblages are overshadowed by the "ghost of land use past" (Harding et al. 1998). This hypothesis is supported by the dominance of tolerant fish species captured during the study (Table 1). Sixty percent of the species are considered tolerant of most physicochemical stream conditions, while another twenty-five percent are considered moderately tolerant. Only nine species captured in this study, all classified as intolerant, are generally considered indicators of healthy streams (Shields et al. 1995). The representation of tolerant species in my study falls outside the range reported in national assessments. Barbour et al. (1999) reported tolerances for 266 species of which 10% were tolerant, 62% moderately tolerant, and 28% intolerant. Similarly, Meador and Carlisle (2007) reported that in a sample of 105 species in streams across the U.S.,

24% were tolerant, 60% moderately tolerant, and 16% intolerant. Also pertinent, the regionally endemic Yazoo Shiner *Notropis rafinesquei*, which typically is collected in large schools, contributed less than 1% of the total abundance. Feasibly, historical changes to landscapes in the basin could have reduced stream fish assemblages to tolerant and generalist species resilient to impounding.

Most of the studies reporting changes in fish assemblages above reservoirs have been conducted in smaller basins supporting smaller reservoirs filled by intermittent or low-order influents (e.g., Winston 1991; Kashiwagi and Miranda 2009; Hedden et al. 2018; Reuter et al. 2019). The 4,800-14,500 ha reservoirs included in this study had larger tributaries, many of them perennial, which may provide adequate refuge to fish assemblages during droughts or other physicochemical disturbances. The similar estimates for species richness of upstream and downstream sites support this hypothesis. As additional reinforcement for this argument, Adams and Warren (2005) studied the recolonization rate for Yazoo Basin streams that became desiccated during an extreme (occurrence <1 in 50 years) drought. All but two of the streams included in my study have a catchment area that is over an order of magnitude larger than the largest catchment area of their desiccated sites suggesting that the streams included in this study rarely if ever become desiccated. Therefore, the fragmentation caused by dams in the Yazoo Basin may not noticeably degrade fish assemblages as the larger tributaries above the reservoirs may provide sufficient populations for recolonization after rare desiccation events at headwater sites. A recent study (Hubbell et al. 2020) also tested for the influence of Sardis Reservoir (one of the reservoirs included in this analysis) on headwater fish assemblages, and no association was detected between either instream habit or fish assemblage and proximity to Sardis Reservoir. My study did include a few low order tributaries that drained directly into a reservoir; in fact the

three lowest estimates from the species accumulation curves in Figure 4 represent sites on streams that all drain directly into Arkabutla Reservoir. It is possible that these tributaries show isolation effects from periodic extirpations and may be contributing to the higher variation in dissimilarity between upstream tributaries, but that the signal from these smaller tributaries is being overridden by the larger tributaries I included.

Single pass electrofishing is one of the most commonly used methods for characterizing a fish assemblage, although it can be biased by differences in detection probabilities across sites and species (Rabeni et al. 2009; Price and Peterson 2010). Despite the fact that capture efficiency from backpack electrofishing using dipnets to collect the stunned fishes (as used in my study) tends to increase with fish size, which can result in under-sampling smaller bodied species (Mahon 1980; Rabeni et al. 2009; Wagner et al. 2019), this method typically captures a greater number of species compared to other single-gear stream sampling methods (Poos et al. 2007). Benthic fishes (e.g., darters), in particular, have a low capture efficiency due to both their typical small size and their lack of a gaseous swim bladder that prevents them from floating to the surface when stunned. Backpack electrofishing is particularly effective in collecting centrarchid species that use large woody debris for cover (Price and Peterson 2010). Minnows may be under-sampled due to both the difficulty in capturing large schools in a dipnet and the reduced effectiveness of electrical fields on small-bodied fish (Mahon 1980; Reid et al. 2009). These issues combined with the high levels of turbidity following the rain events that are common to northern Mississippi summers may have biased fish collection by preferentially selecting larger species with intact swim bladders that would be visible near the surface. This hypothesis is supported by the dominance of larger species such as Bluegill *Lepomis macrochirus* and Green Sunfish *L. cyanellus* and the rarity (low occurrence at sites) of almost half of the species

13

represented in my dataset (Table 1). Most of the fishes captured in two or fewer sites are benthic or have a small body size. In addition to bias regarding differential collection of species, quantifying the effort of electrofishing using only time may also cause fish assemblages in large streams (i.e., larger wetted widths) to be under-sampled compared to smaller streams. Moving forward, using seines rather than dipnets to collect stunned fishes may be more effective in turbid streams since fish would be collected from the whole water column rather than limited by those that are visible and close to the surface (Adams et al. 2004; Poos et al. 2007; Haden and Wagner 2021). Standardization of effort by area and time would also address potential differences in effort between sites.

Dams acting as barriers to potamodromous migrations is another major mechanism that can cause fish assemblage changes in impounded systems. Aadland et al. (2005) documented the near total absence of potamodromous fishes upstream of dams that were not equipped for fish passage in the Red River of the North. These findings are supported by similar studies of migratory fishes in Brazil and Puerto Rico (Agostinho et al. 2008; Cooney and Kwak 2013; Sá-Oliveira et al. 2015; Agostinho et al. 2016). In my study region, the species pool included only two migratory species, White Bass and Smallmouth Buffalo. Conceivably, a mostly non-migratory fish assemblage may not be conspicuously affected by the fragmentation created by reservoirs. Alternatively, potamodromous species may have dwindled in the region over the half century the subbasins have been impounded, removing the portion of the assemblages that would normally distinguish fish assemblages in reaches upstream and downstream from reservoirs.

The only apparent difference between sites upstream and downstream from the reservoirs was a marginal significantly higher representation of planktivores, herbivores, and detritivores in upstream sites. All but two of the species that formed these guilds were classified as lentic or

14

generalist, commonly found in southeastern reservoirs. Their representation was irregular across sites, with different species represented at different sites, and when present they generally occurred in low numbers. Given their low representation and abundance, these common reservoir species did not have a major influence on the species composition or habitat preference analyses yet were highlighted by the trophic guild analysis. Hoeinghaus et al. (2007) similarly found that using functional groups to describe fish assemblages displayed patterns associated with local habitat conditions irrespective of subbasin while analysis of taxonomic descriptions highlighted only regional geographic patterns. Thus, while taxonomic identities (e.g., species) are generally suitable for representing aquatic assemblages, my study further supports using functional classifications to provide alternative interpretations and contrasts, especially over regional scales where the species pool may differ from subbasin to subbasin (Salmaso et al. 2015; Troia and McManamay 2019).

My rather unanticipated results suggest the need for refining expectations about similarities in fish assemblages upstream and downstream from impoundments. Typically, the expectation is that impoundments impact upstream fish assemblages through mechanisms such as change in habitats and loss of longitudinal connectivity. My study suggests that the impacts of these mechanisms may not be universal as the severity of the effects may be nuanced by the regional species pool, the history of stream conditions in the watershed, and the resistance of the streams to periodic disturbances. Additionally, this study highlights how examining stream fish assemblages from several organization perspectives can give insight to different mechanisms working in the system. By examining fish collections beyond taxonomic composition, I gained additional understanding of how riverine species assemblages respond to impoundments.

Tables

Table 1.1   List of species caught in the 26 study sites.

Scientific Name	Common Name	Upstream Sites	Downstream Sites	Habitat Preference	Trophic Guild	Tolerance
Ichthyomyzon castaneus	Chestnut Lamprey	1	0	lotic	parasite	intolerant
Atractosteus spatula	Alligator Gar	1	0	lentic	piscivore	tolerant
Lepisosteus oculatus	Spotted Gar	1	4	lentic	piscivore	tolerant
Dorosoma cepedianum	Gizzard Shad	2	0	generalist	herbivore	tolerant
Dorosoma petenense	Threadfin Shad	2	0	generalist	planktivore	tolerant
Campostoma anomalum	Central Stoneroller	0	2	lotic	herbivore	moderate
Cyprinella camura	Bluntface Shiner	6	10	lotic	invertivore	moderate
Cyprinella lutrensis	Red Shiner	1	2	generalist	invertivore	tolerant
Cyprinella venusta	Blacktail Shiner	9	9	lotic	invertivore	moderate
Cyprinus carpio	Common Carp	2	0	generalist	detritivore	tolerant
Luxilus chrysocephalus	Striped Shiner	2	4	lotic	invertivore	moderate
Lythrurus fumeus	Ribbon Shiner	0	1	lotic	invertivore	intolerant
Notemigonus crysoleucas	Golden Shiner	3	0	generalist	invertivore	moderate
Notropis ammophilus	Orangefin Shiner	2	3	lotic	invertivore	intolerant
Notropis atherinoides	Emerald Shiner	4	0	generalist	planktivore	tolerant
Notropis buchanani	Ghost Shiner	1	0	generalist	invertivore	tolerant
Notropis rafinesquei	Yazoo Shiner	1	3	lotic	invertivore	intolerant
Opsopoeodus emiliae	Pugnose Minnow	1	0	generalist	detritivore	moderate
Pimephales notatus	Bluntnose Minnow	6	9	generalist	detritivore	tolerant
Pimephales vigilax	Bullhead Minnow	3	5	generalist	invertivore	tolerant
Semotilus atromaculatus	Creek Chub	8	8	lotic	invertivore	tolerant
Erimyzon oblongus	Creek Chubsucker	3	4	lotic	invertivore	moderate
*Ictiobus bubalus**	Smallmouth Buffalo	0	4	generalist	invertivore	tolerant
Moxostoma erythrurum	Golden Redhorse	1	0	generalist	invertivore	tolerant
Moxostoma poecilurum	Blacktail Redhorse	0	5	lotic	detritivore	intolerant
Ameiurus melas	Black Bullhead	1	0	generalist	invertivore	tolerant
Ameiurus natalis	Yellow Bullhead	9	11	lotic	invertivore	tolerant
Ameiurus nebulosus	Brown Bullhead	0	1	lentic	invertivore	tolerant
Ictalurus punctatus	Channel Catfish	1	7	generalist	piscivore	tolerant
Noturus gyrinus	Tadpole Madtom	2	0	lotic	invertivore	tolerant
Noturus hildebrandi	Least Madtom	1	0	lotic	invertivore	intolerant
Noturus nocturnus	Freckled Madtom	2	0	lotic	invertivore	tolerant
Pylodictis olivaris	Flathead Catfish	0	2	lotic	piscivore	tolerant
Esox americanus	Redfin Pickerel	1	0	lentic	piscivore	moderate
Aphredoderus sayanus	Pirate Perch	4	0	lentic	invertivore	moderate
Labidesthes sicculus	Brook Silverside	1	1	lentic	planktivore	tolerant
Fundulus chrysotus	Golden Topminnow	1	1	lentic	invertivore	moderate
Fundulus notatus	Blackstripe Topminnow	1	7	lotic	invertivore	tolerant

16

Table 1.1 (continued)

Scientific Name	Common Name	Upstream Sites	Downstream Sites	Habitat Preference	Trophic Guild	Tolerance
Fundulus olivaceus	Blackspotted Topminnow	10	10	lotic	invertivore	moderate
Gambusia affinis	Western Mosquitofish	4	3	generalist	invertivore	tolerant
*Morone chrysops**	White Bass	1	0	generalist	piscivore	tolerant
Lepomis cyanellus	Green Sunfish	12	11	generalist	invertivore	tolerant
Lepomis gulosus	Warmouth	4	1	generalist	invertivore	tolerant
Lepomis macrochirus	Bluegill	12	12	generalist	invertivore	tolerant
Lepomis megalotis	Longear Sunfish	4	10	generalist	invertivore	tolerant
Lepomis miniatus	Redspotted Sunfish	1	1	generalist	invertivore	moderate
Micropterus punctulatus	Spotted Bass	2	6	generalist	piscivore	tolerant
Micropterus salmoides	Largemouth Bass	6	9	generalist	piscivore	tolerant
Pomoxis annularis	White Crappie	2	0	lentic	piscivore	tolerant
Etheostoma artesiae	Redspotted Darter	1	2	lotic	invertivore	moderate
Etheostoma histrio	Harlequin Darter	1	0	lotic	invertivore	intolerant
Etheostoma lynceum	Brighteye Darter	0	2	lotic	invertivore	intolerant
Etheostoma parvipinne	Goldstripe Darter	1	1	lotic	invertivore	intolerant
Etheostoma swaini	Gulf Darter	1	1	lotic	invertivore	intolerant
Percina caprodes	Logperch	0	1	lotic	invertivore	moderate
Percina maculata	Blackside Darter	2	4	lotic	invertivore	tolerant
Percina sciera	River Darter	2	6	lotic	invertivore	moderate
Aplodinotus grunniens	Freshwater Drum	3	3	generalist	piscivore	tolerant

Values represent the number of sites where each species was detected. Asterisks denote migratory species. Migration habits and habitat preferences were assessed using Frimpong and Angermeier (2009). Trophic guild was assessed using Goldstein and Simon (1999). Tolerance was assessed using Meador and Carlisle (2007).

Figures
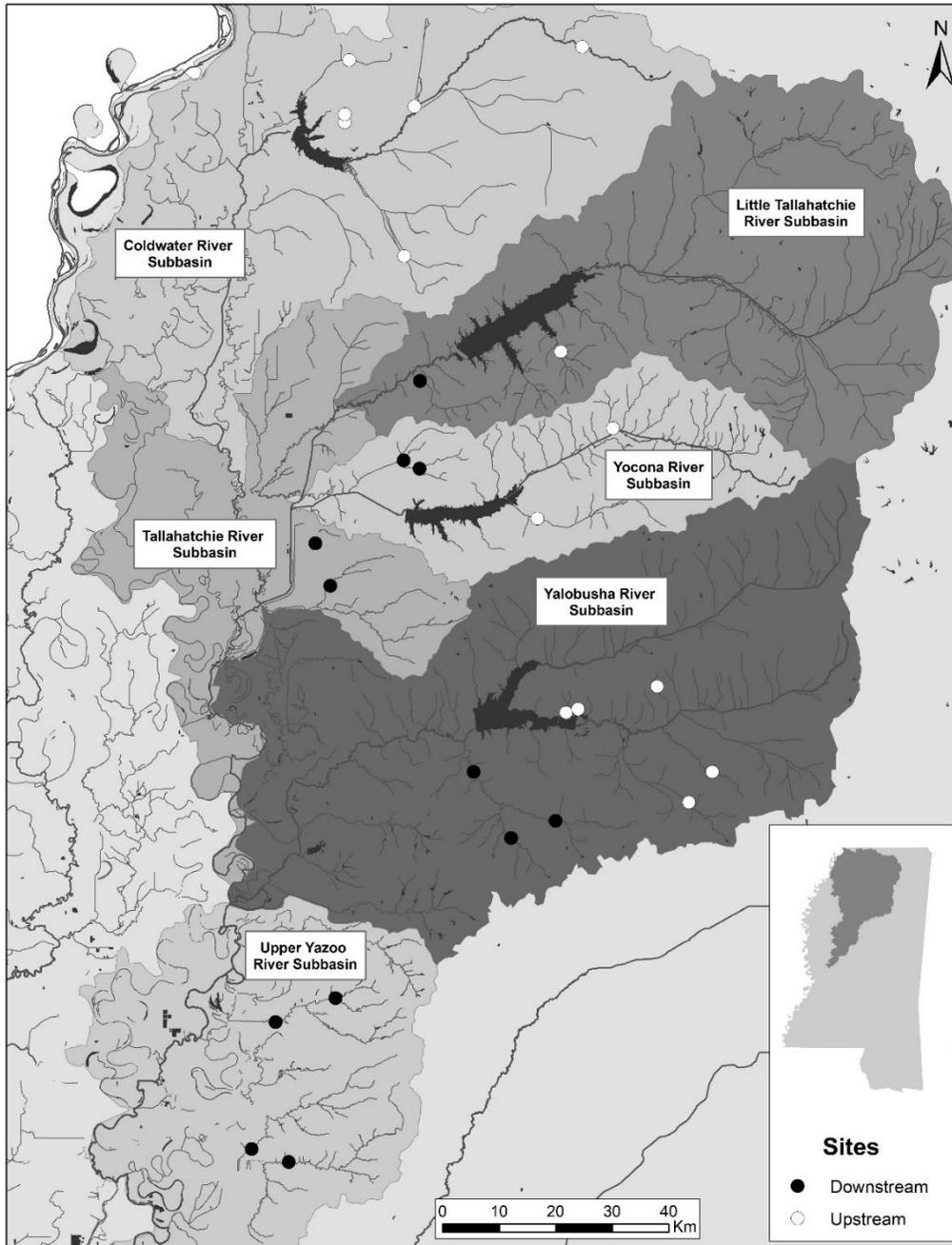
Figure 1.1   Map of eastern Yazoo River Basin

Sites sampled by Arkansas State University 1999-2000, coded by color. Upstream sites are white, while downstream sites are black.
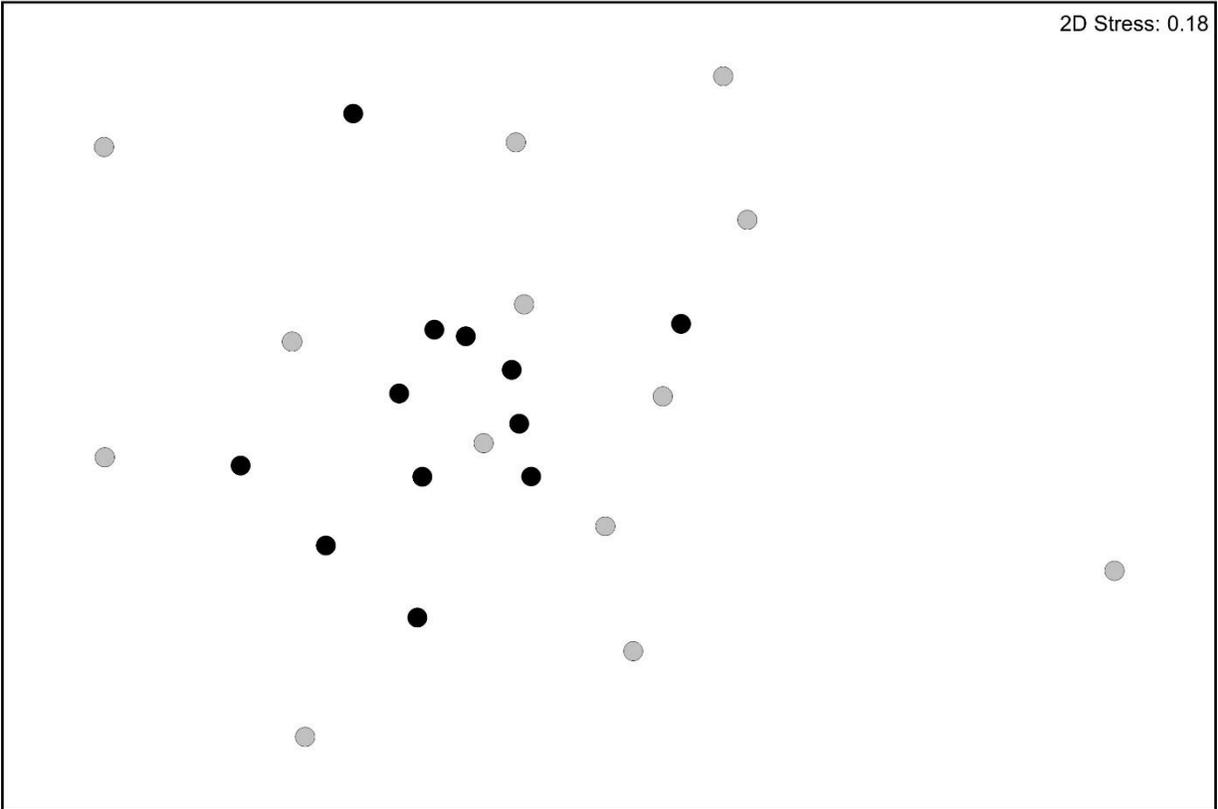
Figure 1.2   NMDS plot of CPUE data in streams of the eastern Yazoo Basin in North
Mississippi

Sites upstream and downstream of dams are coded by color. Grey symbols represent upstream
sites while black symbols represent downstream sites.
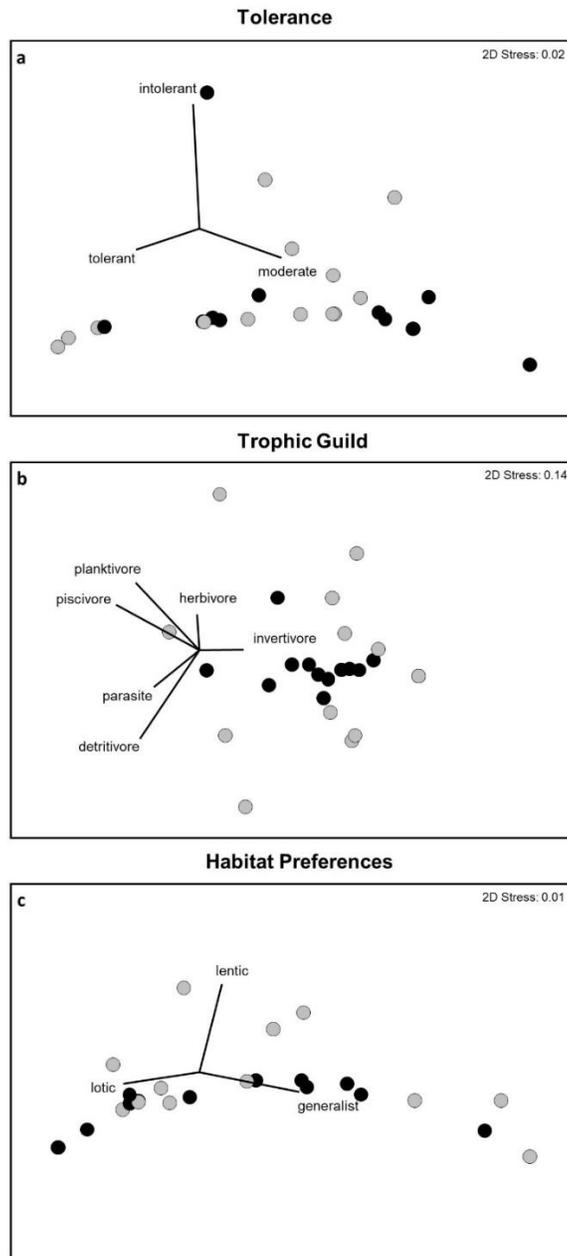
Figure 1.3   NMDS plots showing site similarity based on species tolerances, species trophic
levels and species habitat preferences in streams of the eastern Yazoo Basin in
North Mississippi.

Sites upstream and downstream of dams are coded by color. Grey symbols represent sites
upstream of a reservoir while black symbols represent sites downstream from a reservoir.

20
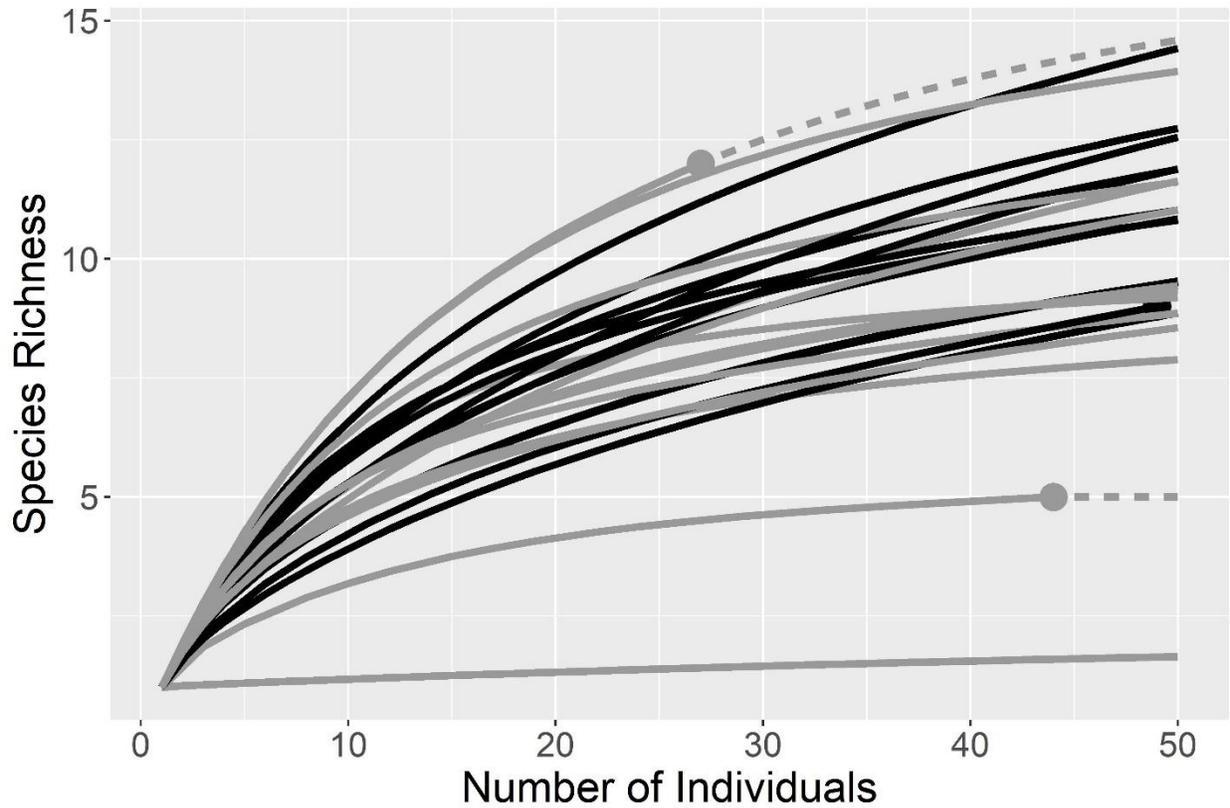
Figure 1.4   Species accumulation curves for sites in streams of the eastern Yazoo Basin in North Mississippi.

Sites upstream and downstream of dams are coded by color. Grey curves represent upstream sites while black curves represent downstream sites. Solid lines represent rarefied estimates while dashed lines represent estimates extrapolated using Chao 1 estimator. Points on the curve represent the sampled values. Y-axis denotes species richness; X-axis denotes number of individuals in a sample.

References

Aadland, L. P., T. M. Koel, W. G. Franzin, K. W. Stewart, and P. Nelson. 2005. Changes in fish assemblage structure of the Red River of the North. Pages 293-321 *in* J. R. Rinne, R. M. Hughes, and B. Calamusso, editors. Historical changes in large river fish assemblages of the Americas, American Fisheries Society, Symposium 45, Bethesda, Maryland.

Adams, S. B., and M. L. Warren. 2005. Recolonization by warmwater fishes and crayfishes after severe drought in Upper Coastal Plain hill streams. Transactions of the American Fisheries Society 134: 1173-1192.

Adams, S. B., M. L. Warren, and W. R. Haag. 2004. Spatial and temporal patterns in fish assemblages of upper coastal plain streams, Mississippi, USA. Hydrobiologia 521: 45-61.

Agostinho, A. A., L. C. Gomes, N. C. L. Santos, J. C. G. Ortega, and F. M. Pelicice. 2016. Fish assemblages in Neotropical reservoirs: colonization patterns, impacts and management. Fisheries Research 173: 26-36.

Agostinho, A. A., F. M. Pelicice, and L. C. Gomes. 2008. Dams and the fish fauna of the Neotropical region: impacts and management related to diversity and fisheries. Brazilian Journal of Biology 68(4): 1119-1132.

Altermatt, F. 2013. Diversity in riverine metacommunities: a network perspective. Aquatic Ecology 47: 365-377.

Anderson, M. J. 2017. Permutational Multivariate Analysis of Variance (PERMANOVA). *Wiley StatsRef: Statistics Reference Online*. Available at URL https://onlinelibrary.wiley.com/doi/10.1002/9781118445112.stat07841. Accessed 15 September 2021.

Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates, and fish, second edition. EPA 841-B-99-002. U.S. Environmental Protection Agency, Office of Water, Washington, D.C.

Barnett, Z. C. and S. B. Adams. 2021. Review of dam effects on native and invasive crayfishes illustrates complex choices for conservation planning. Frontiers in Ecology and Evolution 8: 621723. https://doi.org/10.3389/fevo.2020.621723.

Bennet, S. J. and F. E. Rhoton. 2009. Linking upstream channel instability to downstream degradation: Grenada Lake and the Skuna and Yalobusha river basins, Mississippi. Ecohydrology 2: 235-247.

Biedenharn, D. S., C. M. Elliott, and C. C. Watson. 1997. *The WES stream investigation and streambank stabilization handbook*. US Army Engineer Waterways Experiment Station.

Bledsoe, B. P., C. C. Watson, and D. S. Biedenharn. 2002. Quantification of incised channel evolution and equilibrium. Journal of the American Water Resources Association 38(3): 861-870.

Chao, A., N. J. Gotelli, T. C. Hsieh, E. L. Sander, K. H. Ma, R. K. Colwell, and A. M. Ellison. 2014. Rarefaction and extrapolation with Hill numbers: a framework for sampling and estimation in species diversity studies. Ecological Monographs 84(1): 45-67.

Clark, K. R. and R. N. Gorley. 2015a. Primer v7: User Manual/Tutorial. Primer-E: Plymouth.

Colwell, R. K., A. Chao, N. J. Gotelli, S. Y. Lin, C. X. Mao, R. L. Chazdon, and J. T. Longino. 2012. Models and estimators linking individual-based and sample-based rarefaction, extrapolation and comparison of assemblages. Journal of Plant Ecology 5: 3–21.

Cooney, P. B. and T. J. Kwak. 2013. Spatial extent and dynamics of dam impacts on tropical island freshwater fish assemblages. Bioscience 63: 176-190.

Dabney, S. M., F. D. Shields, R. L. Binger, R. A Kuhnle, and J. R. Rigby. 2012. Watershed management for erosion and sedimentation control case study Goodwin Creek, Panola County, Pages 539-568 *in* R. Lal and B. A. Stewart, editors. Advances in Soil Science, Soil Water and Agronomic Productivity, Taylor and Francis, Boca Raton, FL.

Duffy, P. D., and S. J. Ursic. 1991. Land rehabilitation success in the Yazoo basin, USA. Land Use Policy 8(3): 196-205.

Falke, J. A. and K. B. Gido. 2006. Effects of reservoir connectivity on stream fish assemblages in the Great Plains. Canadian Journal of Fisheries and Aquatic Sciences 63: 480-493.

Frimpong, E. A. and P. L. Angermeier. 2009. FishTraits: A database of ecological and life-history traits of freshwater fishes of the United States. Fisheries 34(10): 487-495.

Goldstein, R. M. and T. P. Simon. 1999. Toward a united definition of guild structure for feeding ecology of North American freshwater fishes, Pages 123-202 *in* T. P. Simon, editor. Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Boca Raton, Florida.

Graf, W. L. 1999. Dam nation: a geographic census of American dams and their large-scale hydrologic impacts. Water Resources Research 35(4): 1305-1311.

Guenther, C. B., and A. Spacie. 2006. Changes in fish assemblage structure upstream of impoundments within the Upper Wabash River Basin, Indiana. Transactions of the American Fisheries Society 135: 570-583.

Haden, L. L. and M. D. Wagner. 2021. Status, distribution, and length-frequency for the Frecklebelly madtom (*Noturus munitus*) in the Tombigbee River Drainage in Mississippi. Southeastern Naturalist, 20(1): 162-177.

Hannan, H. H. 1979. Chemical modifications in reservoir-regulated streams. Pages 75-94 in J. V. Ward and J. A. Stanford, editors. The Ecology of Regulated Streams. Springer, Boston, MA.

Harding, J. S., E. F. Benfield, P. V. Bolstad, G. S. Helfman, and E. B. D. Jones III. 1998. Stream biodiversity: the ghost of land use past. Proceedings of the National Academy of Science 95: 14843-14847.

Hedden, S. C., E. A. Renner, K. B. Gido, and K. J. Hase. 2018. Impacts of small impoundments on an intermittent headwater stream community. Southwestern Naturalist 63(1): 34-41.

Herbert, M. E., and F. P. Gelwick. 2003. Spatial variation of headwater fish assemblages explained by hydrologic variability and upstream effects of impoundment. Copeia 2003(2): 273-284.

Hilgard, E.W. 1860. *Report on the geology and agriculture of the state of Mississippi*. E. Barksdale, State Printer.

Hoeinghaus, D. J., K. O. Winemiller, and J. S. Birnbaum. 2007. Local and regional determinants of stream fish assemblage structure: inferences based on taxonomic vs. functional groups. Journal of Biogeography 34: 324-338.

Hubbell, J. P., J. F. Schaefer, P. Flood, M. L. Warren Jr., and K. A. Sterling. 2020. Fragmentation alters ecological gradients and headwater fish assemblage composition relative to land use in a dendritic river system. Canadian Journal of Fisheries and Aquatic Sciences 77(8): 1281-1291. https://doi.org/10.1139/cjfas-2019-0080.

Hsieh, T. C. K. H. Ma, and A. Chao. 2016. iNEXT: an R package for rarefaction and extrapolation of species diversity (Hill numbers). Methods in Ecology and Evolution 7: 1451-1456.

Jelks, H. L., S. J. Walsh, N. M. Burkhead, S. Contreras-Balderas, E. Diaz-Pardo, D. A. Hendrickson, J. Lyons, N. E. Mandrak, F. McCormick, J. S. Nelson, S. P. Platania, B. A. Porter, C. B. Renaud, J. J. Schmitter-Soto, E. B. Taylor, and M. L. Warren Jr. 2008. Conservation status of imperiled North American freshwater and diadromous fishes. Fisheries 33(8): 372-407.

Junk, W. J., P. B. Bayley, and R. E. Sparks. 1989. The floodpulse concept in river-floodplain systems. Canadian Special Publication of Fisheries and Aquatic Sciences 106(1): 110-127.

Kashiwagi, M. T. and L. E. Miranda. 2009. Influence of small impoundments on habitat and fish communities in headwater streams. Southeastern Naturalist 8(1): 23-36.

Knight, S. S., and C. M. Cooper. 1987. Fishes of Otoucalofa Creek, Mississippi prior to major channel modifications. Journal of the Mississippi Academy of Sciences (USA).

Leech, J. R., and D. S. Biedenharn. 2010. Sediment management at the watershed level. Report given to U.S. Army Corps of Engineers Engineer Research and Development Center.

Mahon, R. (1980) Accuracy of catch-effort methods for estimating fish density and biomass in streams. Environmental Biology of Fishes 5: 343-60.

McAllister, D. E., S. P. Platania, F. W. Schueler, M. E. Baldwin, and D. S. Lee. 1986. Ichthyofaunal patterns on a geographic grid. Pages 17-52 *in* C. H. Hocutt and E. O. Wiley, editors. The Zoogeography of North American Freshwater Fishes. John Wiley and Sons. New York.

Meador, M. R. and D. M. Carlisle. 2007. Quantifying tolerance indicator values for common stream fish species of the United States. Ecological Indicators 7: 329-338.

Poos, M. S., N. E. Mandrak, and R. L. McLaughlin. 2007. The effectiveness of two common sampling methods for assessing imperiled freshwater fishes. Journal of Fish Biology 70: 691-708.

Price, A. L and J. T. Peterson. 2010. Estimation and modeling of electrofishing capture efficiency for fishes in wadeable warmwater streams. North American Journal of Fisheries Management 30: 481-498.

Rabeni, C. F., J. Lyons, N. Mercado-Silva, and J. T. Patterson. 2009. Warmwater fish in wadeable streams. Pages 43-58 *in* S. A. Bonar, W. A. Hubert, and D. W. Willis, editors. Standard methods for sampling North American freshwater fishes. American Fisheries Society, Bethesda, Maryland.

Reid, S. M., G. Yunker, and N. E. Jones. 2009. Evaluation of single-pass backpack electrofishing for stream fish community monitoring. Fisheries Management and Ecology 16: 1-9.

Reuter, C. N., M. D. Kaller, C. E. Walsh, and W. E. Kelso. 2019. Fish assemblage response to altered dendritic connectivity in the Red River Basin, Central Louisiana. American Midland Naturalist 181: 63-80.

Rodriguez-Iturbe, I., and A. Rinaldo. 1997. Fractal river networks: chance and self-organization. Cambridge University Press, New York.

R Core Team. 2021. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Salmaso, N., L. Naselli-Flores, and J. Padisák. 2015. Functional classifications and their application in phytoplankton ecology. Freshwater Biology 60: 603-619.

Sá-Oliveira, J. C., J. E. Hawes, V. J. Isaac-Nahuum, and C. A. Peres. 2015. Upstream and downstream responses of fish assemblages to an Eastern Amazonian hydroelectric dam. Freshwater Biology 60: 2037-2050.

Schumm, S. A., Harvey, M. D., and C. C. Watson. 1984. Incised channels: morphology, dynamics, and control. Water Resources Publications.

Shields, F. D., S. S. Knight, and C. M. Cooper. 1995. Use of the index of biotic integrity to assess physical habitat degradation in warmwater streams. Hydrobiologia 312(3): 191-208.

Simon, A. 1989. A model of channel response in disturbed alluvial channels. Earth Surface Processes and Landforms 14(1): 11–26.

Simon, A., 1994. Gradation processes and channel evolution in modified West Tennessee streams: process, response and form. U.S. Geological Survey Professional Paper 1470, Washington, DC.

Smith, S. C. F., S. J. Meiners, R. P. Hastings, T. Thomas, and R. E. Columbo. 2017. Low-head dam impacts on habitat and the functional composition of fish communities. River Research Applications 33: 680-689.

Troia, M. J. and R. A. McManamay. 2019. Biogeographic classification of streams using fish community- and trait-environment relationships. Diversity and Distributions 26: 108-125.

U.S. Army Corps of Engineers. 1981. A comprehensive summary report on streambank erosion control demonstration projects constructed in eight watersheds of the Yazoo River basin. Appendix F. in the Final Report to Congress: The Streambank Erosion Control Evaluation and Demonstration Act of 1974 Section 32, Public Law 93-251.

U.S. Geological Survey. 2020. The StreamStats program v.4.3.11, accessed 06 February 2020 at URL http://streamstats.usgs.gov.

U.S. Army Corps of Engineers. 2018. National inventory of dams: U.S. Army Corps of Engineers, accessed 18 August 2021 at URL https://nid.usace.army.mil.

U.S. Geological Survey. 2021. National Hydrography Dataset (ver. USGS National Hydrography Dataset Best Resolution (NHD) for Hydrologic Unit (HU) 4 – 0803 (published 202110102)), accessed 05 January 2021 at URL https://www.usgs.gov/core-science-systems/ngp/national-hydrography/access-national-hydrography-products.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37: 130-137.

Wagner, M. D., D. A. Schumann, and B. J. Smith. 2019. Gear effectiveness and size selectivity for five cryptic madtom species (Noturus spp.). Journal of Applied Ichthyology 35: 673–682. https://doi.org/10.1111/jai.13892

Warren Jr., M. L., W. R. Haag, and S. B. Adams. 2002. Forest linkages to diversity and abundance in lowland stream fish communities. *in* Holland, M. M., M. L. Warren Jr., and J. A. Stanturf, eds. Proceedings of a conference on sustainability of wetlands and water resources: how well can riverine wetlands continue to support society into the 21st century? General Technical Report SRS-50. Ashville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station 191 p.

Whitten, C. B., and D. M. Patrick, 1981. Engineering geology and geomorphology of stream-bank erosion; Report 2, Yazoo River Basin Uplands, Mississippi. Geotechnical Laboratory, US Army Corps of Engineers.

Williston, H. L. 1998. The Yazoo-Little Tallahatchie Flood Prevention Project: A history of the Forest Service's role. Forestry report R8-FR-US Department of Agriculture, Forest Service, Southern Region (USA).

Winston, M. R., C. M. Taylor, and J. Pigg. 1991. Upstream extirpation of four minnow species due to damming of a prairie stream. Transactions of the American Fisheries Society 120: 98-105.

CHAPTER II

EFFECTS OF LOW-DROP GRADE CONTROL STRUCTURES ON CHANNEL

EVOLUTION OF YAZOO BASIN STREAMS

Introduction

Degradation of instream habitat is a leading cause of fish assemblage alterations.

Anthropogenic modifications to river systems such as channelization, channel shortening

through cutoffs, and removal of large woody debris through snagging can accelerate channel

incision that drastically alters existing stream habitat. Such habitat instability often leads to a

homogenization of the fish assemblage, which becomes dominated by generalist species with

wide physicochemical and habitat tolerances (Schlosser 1987; Hitt et al. 2020). For example, in

Bayou Pierre in southwestern Mississippi, channel incision and subsequent habitat alteration

have resulted in reduced diversity, where fish assemblages shifted to species with larger body

sizes, higher fecundity, and shorter age at maturity- all traits consistent with species adapted to

dynamic systems (Stearman and Schaefer 2022). Raborn and Schramm (2003) also found that

both channelized and incised reaches of a stream demonstrated lower species richness and

significantly different habitat conditions compared to an unaltered reach. In contrast, stable

stream conditions typically result in high degrees of niche separation, as species become

specialized to take advantage of specific habitat resources (Winemiller and Rose 1992; Hitt et al.

2020). Thus, channel stability can have far-reaching influence on the stream fish assemblage.

Stability in alluvial streams is difficult to define, largely because stream channels are dynamic systems that move laterally and change courses through time. Stability then, is a state that cannot exist at either very short or very long timescales, but over historical periods (i.e., a few decades to a few centuries), a general equilibrium state may be defined (Schumm 1985). An indicator of stream stability is Lane's equilibrium, where sediment supply is equal to sediment transport capacity (Lane 1955). Lane's equilibrium describes the relationship between the four parameters that largely determine stability: bed material discharge and particle size of bed material (hereafter sediment size) which together make up sediment supply, and discharge and channel slope, which together determine sediment transport capacity. For example, if discharge or channel slope increases, the stream will degrade because the transport capacity exceeds the available sediment supply; conversely, if bed material discharge were to increase, the channel would aggrade since the sediment supply exceeds the transport capacity. Thus, a stream is considered stable, or in an equilibrium state, when sediment supply and sediment transport capacity are roughly equal and the channel is neither degrading nor aggrading, although localized scour or aggradation may occur due to lateral movement of channel meanders.

Streams can become unstable when sediment transport capacity is altered either through natural processes or through anthropogenic changes to either slope or discharge. Unstable streams gradually readjust and follow a series of defined stages of channel evolution as the stream reaches a new equilibrium state. Schumm et al. (1984) developed a five-stage channel evolution model (CEM) that described channel response to naturally occurring disturbances that was later modified by Simon (1989, 1994) to incorporate a distinct disturbance regime by including channelization as a distinct stage (Figure 2.1). Stage I is a stable, sinuous stream. Stage II describes channelized conditions where the channel is usually deepened and

29

manipulated to a trapezoidal shape. In stage III, the channel begins to deepen, usually resulting

from a disturbance to sediment supply/transport equilibrium (Simon and Rinaldi 2006). As the

area of scouring moves upstream through the reach, the reach slope increases. Throughout stage

IV, the channel continues to deepen and begins to widen as the banks become over-steepened,

undercut, and eventually fail. Stage V describes a period of aggradation and widening as the

banks continue to fail, widening the channel and introducing large amounts of sediment into the

stream, which combined with sediment from upstream, begins to deposit in the channel as the

slope decreases. Finally, stage VI describes a quasi-equilibrium state where the stream is

reforming a sinuous channel between vegetated berms within the original top of bank elevations,

which have now become terraces.

When instability is triggered through lowering the base level of a stream system (usually

due to channelization or meander removal of a mainstem), incision moves upstream though the

migration of a headcut (knickpoint), which is a sharp change in gradient. As the headcut moves

upstream, it initiates the CEM process. Thus, headcuts cause bank instability, which can result

in the loss of massive amounts of sediment and permanent alterations to the morphology of the

stream channel. For example, Goodwin Creek, a fourth order stream in northwestern Mississippi,

produced an estimated sediment yield of 1,000 t/km^2 because of channel incision (Dabney et al.

2012). Over a period of 60 years, the cross-sectional area of the Blackwater River in Missouri

experienced an increase of 1173% resulting from incision induced by channelization downstream

(Emerson 1971). Similar incision along the Homochitto River in southwestern Mississippi

caused widening of the channel and the collapse of numerous bridges from 1945 to 1974,

totaling $1,863,115 in damages (not adjusted for inflation) (Wilson 1979). Channel incision is a

widespread problem (Ramser 1930; Daniels et al. 1960; others), especially in river basins that
have an extensive history of channel alterations.

Since channel incision causes large ecological and economic consequences, prevention of
continued migration of headcuts has been the focus of many stream engineering projects. An
effective and widely used method of controlling headcut migration has been the installation of
low-drop grade control structures (GCS), which are designed to halt the headcutting process by
mediating changes in channel gradient up to 1.8 m (6 ft) (Figure 2.2). The GCS consists of a
concrete or steel sheet-pile weir constructed either at grade or slightly above the current grade to
encourage sediment deposition (Abt et al. 1992; Biedenharn and Hubbard 2001). Below the
weir, a large riprap reinforced stilling basin with a baffle plate serves to dissipate the energy of
the stream before it enters the channel downstream of the structure. GCS are installed at or
slightly upstream of an active knickpoint (Biedenharn and Hubbard 2001), which often resulted
in multiple GCS installed in series on the same stream since multiple base-lowering events can
produce numerous waves of headcutting that move sequentially through a stream system.

Monitoring efforts to capture the effectiveness of these GCS after installation often only
last three to five years post installation through analysis of sediment load measurements, but the
results have been variable (Leech and Biedenharn 2012). Suspended sediment analysis for 11
stream gages in the Yazoo Basin, Mississippi found decreasing trends at eight sites, but only one
was statistically significant, while three others were statistically inconclusive (Rebich 1995;
Biedenharn and Watson 2012). Decades might be necessary to estimate the effects of the GCS
due to time lags in sediment production, storage, and delivery subsystems (Shields et al. 1995;
Martin et al. 2010). Modeling the results of the GCS into the future predicted significant
reductions in bed load as well as fewer indications of CEM progression upstream of the GCS

(Simon and Darby 2002; Martin et al. 2010; Leech and Biedenharn 2012). To measure the true effects of these structures, long term monitoring is advisable (Martin et al 2010; Leech and Biedenharn 2012).

The goal of this study was to examine the long-term (> three decades) effectiveness of GCS to mitigate headcuts by comparing channel morphology and sediment distribution of upstream and downstream sites across streams with varying numbers of structures. If effective, I expected the reaches downstream of the GCS would show signs of headcutting and mass wasting (deep, wide channel, steep banks, large aggradations of gravel and other coarse sediments) while the upstream reaches would be relatively unaffected by erosion (i.e., remain shallower and narrower with low sloped banks). From a sediment perspective, I expected that sites upstream of the GCS would have more fine sediments due to the way the GCS are constructed above grade, while downstream would have more large substrates and/or clay bedrock because of scouring as the stream attempts to regrade. The control streams also were expected to show signs of active or past erosion consistent with CEM stages III-VI, depending on how long ago a headcut passed through the system.

Methods

Study Area

The Yazoo Basin in Northwestern Mississippi is a tributary basin of the Mississippi River, and consists of two main regions: the large, flat alluvial plain and the loess bluff hills that drain into the alluvial plain. The soils in the loess bluff hills consist of a layer of yellow, loamy loess, wind carried sediments that were deposited during the Pleistocene, overlying slightly older sandy, gravelly layers (Whitten and Patrick 1981; Grissenger and Murphey 1983). These soils are highly erodible and have allowed channel incision resulting from widespread changes in land

32

use in both regions and channelization of the larger rivers to run unchecked though most of the tributary streams in the basin. Multiple channelization projects ranging from the early 1900s to the 1960s combined with a reduction in sediment supply due to reforestation in the 1970s have resulted in multiple waves of headcuts moving through the stream systems. Over 160 GCS (Figure 2.3) were installed in 1989-1995 to prevent further incision through the upstream movement of headcuts as part of the federal Demonstration Erosion Control project (DEC; Watson et al. 1999).

I surveyed sites on 15 streams balanced across five subbasins in the loess bluff hills (Figure 2.3) to examine the long-term effects of the GCS on stream morphology. Five streams had two to three GCS in series, five had more than four GCS in series, and five with no GCS were included to serve as a degraded control. Nearly all the streams in the Yazoo Basin show some extent of channel incision, so degraded controls are not acting as true reference streams for the best-case scenario but rather as an example of unchecked incision. Streams were chosen to maximize the opportunity to survey stream reaches accessible from road crossings that had been surveyed previously over the course of the DEC project to allow for future studies of temporal change in stream profiles. To assess the effects of GCS on channel morphology, a pair of sites were designated on each stream: one upstream from the GCS and one downstream. For the control streams (i.e., no structure), the paired sites were distanced similarly to the treatment streams. Distance between paired sites averaged 8.5 river kilometers. To avoid surveying areas where the channel morphology was influenced by the bridge or culvert design associated with the road crossing, surveyed reaches were delineated upstream of the crossing whenever possible, so that the downstream end occurred within the tree line (upstream from the cleared right of way associated with the road or highway).

Field Methods

At each site, stream geomorphology, channel dimensions, and substrate were measured in cross-sectional transects spaced every 20 m (Table 1). At each transect, channel width, width at bankfull, and wetted width were measured to the nearest 0.1 m. Bankfull stage was visually identified using several indicators, including lowest extent of perennial vegetation, abrupt changes in particle size, and breakpoints in bank angle. Within the wetted width, a minimum of three points along each transect were selected to measure water depth and current velocity (measured at 0.6 water depth). In uniformly U-shaped stream beds, the points were chosen to represent the deepest point, and the two points equidistant between the deepest point and the edge of wetted width. For transects with complex bed profiles, the deepest point was always measured, and other points were positioned to record prominent changes in transect depth, using additional points as necessary to reflect complexity of the transect. The dominant substrate size was visually estimated using a modified Wentworth scale (bedrock, boulder, cobble, gravel, coarse sand, fine sand, silt, clay; Blair and McPherson 1999) at every point, including endpoints of the width measurements. Riprap introduced for bank stabilization was also included as a potential substrate type to distinguish it from the similarly-sized native substrate, cobble.

The transect data were processed using stream modeling software, HEC-RAS 6.1 (USACE 2022), to calculate cross-sectional area (A_{bkf}) at bankfull conditions. Calculated A_{bkf} values were compared to those derived from a regional power curve for the loess streams of the Yazoo Basin to validate my field delineations (Hadadin 2017). From these cross-sectional areas and transect measurements (Table 1), I calculated three ratios to index the shape of the channel: width to depth ratio (WDR), bank height ratio (BHR), and entrenchment ratio (ER; Figure 2.4). High WDRs indicate streams that are actively eroding due to bank failures (e.g., CEM stage III

and IV); high BHRs indicate banks that are close to failure (CEM stage II and III); low ERs indicate disconnection with the floodplain. Incised streams generally have high WDR, BHR much greater than one, and low ER; unincised streams tend to have low WDR, BHR approximately one, and high ER.

Analysis

Because measures of stream morphology and sediment distribution tend to be highly correlated, I used multivariate analyses to test for the effects of GCS on stream morphology and sediment distribution. GCS effects of stream morphology were tested using ER, BHR, and WDR as the dependent variables since these ratios allow for easy comparison of morphology across streams of varying sizes. For the sediment analysis, percent area within bankfull of each particle size category of each transect were averaged for each site for use as dependent variables. Since land use and riparian vegetation can impact bank stability, basin-scale covariates that could interact with the GCS to temper or intensify the effects of channel incision were calculated using GIS. Catchment area upstream of each site was obtained from USGS StreamStats (USGS 2021). Landcover percentages were calculated for each watershed using the 2016 Landsat National Land Cover Dataset (NLCD; available at https://www.mrlc.gov/data). Deforestation in the watersheds over the life of the DEC project (1986-2020) was also calculated for each watershed using the Forest Disturbance dataset (available at https://www.mrlc.gov/data), which documents the amount of area deforested between each NLCD timestep, approximately every two to three years from 1986 to 2019. The extent of riparian vegetation at each reach was indexed by calculating the average percent canopy cover of an area extending from 100 m upstream to 100 m downstream of the reach endpoints and 30 m wide centered on the thalweg of the stream. A width of 30 m was chosen to align with Natural Resources Conservation Service

recommendations for riparian buffer width in perennial streams in forested areas (Bentrup 2008).

Percent canopy cover data from the 2016 NLCD dataset was resampled to 5 m pixels before

calculating the mean to ensure even coverage within the measured area (Coulston et al. 2012).

Because identification of bankfull elevations can be complicated by stream instability

(Doyle et al. 2007; Duncan et al. 2011), I validated the A_{bkf} calculations with an analysis of

covariance that compared the measured values with values predicted using the Yazoo Basin

regional regressions for stable (Simon CEM V and VI) and incised streams (Simon CEM III and

IV) reported in Hadadin (2017). The regional regressions were in the form

$$A_{bkf} = a(DA)^b \tag{2.1}$$

where A_{bkf} is cross-sectional area at bankfull stage (m^2), DA is drainage area (km^2), and a and b

are regression coefficients. I used Hadadin's (2017) a and b coefficients to predict A_{bkf} values

for my sites, then compared them using the linear regression

$$A_{bkf} = \beta_0 + \beta_1 Type \tag{2.2}$$

where *Type* specifies whether the A_{bkf} estimate was field calculated or predicted from incised or

stable regional regressions (Hadadin 2017).   Validation was conducted in Program R (R core

team 2022).

After validating the stream transects calculations, I conducted a Permutational Analysis

of Variance (PERMANOVA) to test for any effects the GCS may have on the channel

morphology. A Euclidean distance matrix was created from the sampling sites by channel

morphology (WDR, BHR, and ER) matrix. The categorical predictor variables were site location

(upstream/downstream) and number of GCS (control, two to three, four or more). Percent

riparian cover was included as a covariate because the root systems of mature riparian vegetation

can work to stabilize banks, which could reduce stream bank erosion expected through the headcutting process and thus complicate the expected patterns (Shields et al. 1995). A Principal Components Analysis (PCA) was used to graphically explain the results of the PERMANOVA.

Analysis of sediment distributions paralleled the stream morphology analysis. Using the visual point counts of sediment size, the percentage per size class for each channel cross-section was computed. Fine sand and coarse sands were combined into a single 'sand' category and silt and clay were combined into a 'fines' category to reduce observer bias. The percentages were then normalized (mean = 0; SD = 1) across variables and a resemblance matrix was created using Euclidean distance applied to the sampling sites by size class percentage category matrix, with transects acting as within-site replicates. This resemblance matrix was used to conduct a PERMANOVA analysis, where the predictors were site location and number of structures. Because surface erosion can also be a large contributor to sediment load in streams, I included as covariates percent agricultural land use and percent deforested land use of the watersheds upstream of the sites. I also included percent riparian cover for each reach as a covariate since the state of the riparian zone can influence both bank stability and the amount of sediment runoff that enters the stream. Results of the PERMANOVA analysis were visually represented by a bar plot showing proportions of sediment types across site categories. Both the stream morphology and sediment multivariate analyses were conducted in Primer-e7 (Plymouth, UK).

Results

The empirical measurements A_{bkf} agreed with those predicted using Hadadin's regional regressions for the Yazoo Basin (Figure 2.5). The global test for the validation regression indicated that the model did not fit the data better than the intercept only model, (F statistic; 0.598, P: 0.55). There was no difference (t: -0.825; P: 0.411) between the empirical A_{bkf} and the

predicted A_{bkf} for incised streams (β_2), or stable streams (β_3; t:-1.034; P: 0.304; Table 2.2). This result confirmed accuracy in identifying bankfull stage across the whole range of watershed sizes included in my dataset.

Overall, WDRs ranged from 9.5 to 56.8, BHRs from 3.1 to 21.9, and ERs from 1.1 to 1.8 indicating that most of the sites show some degree of channel incision when compared to national standards (Table 2.3). In the morphology analysis, I found significant differences due to both GCS (Pseudo F: 11.4; P: 0.0001) and site location (Pseudo F: 8.9; P: 0.0004) after blocking for percent riparian cover. The interaction of GCS by site location was also significant (Pseudo F: 5.6; P: 0.0001). The post-hoc pairwise PERMANOVA confirmed differences in channel morphology between upstream and downstream sites in streams with two to three GCS (t: 5.3; P: 0.0001) and four or more GCS (t: 1.8; P: 0.0391). Channel morphology in control streams was similar between upstream and downstream sites (t: 0.55; P: 0.8035). These results are illustrated by the PCA ordination (Figure 2.6). PC axis 1 explained 69.3% of the variation in ER, WDR, and BHR values and describes an incision gradient where sites that are relatively unincised with high ER have negative scores on axis 1 and highly incised sites with high WDR have positive scores. PC axis 2 explains an additional 17.6% of the variation and describes a gradient in BHR, where sites with high scores have high BHR and vice versa. Upon examination of site scores by site category, it is evident that in streams with two to three structures, upstream sites had a much greater ER, while downstream sites had greater BHR and WD values. Streams with four or more GCS in general show similar patterns, although the dispersion is greater. In control streams, there was no clear separation between upstream and downstream sites.

Across all sites, stream substrates were predominantly sand (mean of 59% of channel area). The next most common size class was the fines category (16%), followed by hard clay

bedrock (12%). Cobble was only found at four sites. Twelve sites included riprap substrate

introduced to stabilize banks or guard against scour from local features such as gas and water

pipelines or drainage ditch outlets. Results of the sediment analysis agreed with the morphology

analysis: after blocking for percent riparian cover, percent agricultural, and percent deforested

land use, number of GCS (Pseudo *F*: 1.9; *P*: 0.0390) and site location (Pseudo *F*: 2.6; *P*: 0.0221)

significantly affected the sediment distribution, and the interaction between the two was also

significant (Pseudo *F*: 3.6; *P*: 0.0001). The post-hoc pairwise PERMANOVA revealed that

upstream sites on both streams with two to three GCS (*t:* 2.81; *P*: 0.0001) and four or more GCS

(*t*: 2.36; *P*: 0.0002) differed significantly from the downstream sites. Upstream and downstream

sites on control streams were not significantly different (*t:* 1.3; *P:* 1629). The bar plot (Figure

2.7) illustrates sites upstream of the GCS have more fine substrate and less large substrate

(gravel, cobble, and hard clay bedrock) compared to the downstream sites. Sites on the control

stream display the opposite trend on average, although the trend was not statistically significant.

Control sites had limited fine substrates upstream but large percentages of large substrates, while

downstream had limited hard substrate.

Discussion

Overall, the GCS seem to have interrupted the upward migration of headcuts in the

Yazoo Basin. Upstream of the structures, the channels were narrower, less incised, and had

greater access to the floodplain when compared to sites downstream of the structures; however,

all the sites had a BHR greater than 3.0, which indicates that all are severely incised compared to

national standards (Rosgen 1994). This is unsurprising given the long history of erosion,

sedimentation, channelization, and incision in the Yazoo Basin. Many of the navigable rivers, as

well as their larger tributaries have repeatedly been channelized to remove excess sediment and

reduce flooding. Each channelization event lowered the base level, which in turn instigated a new series of headcuts moving through the basin (Dabney et al. 2012). Unsurprisingly, some DEC GCS were not as successful at curbing CEM advancement because prior headcuts might have already been upstream of the GCS (Simon and Darby 2002). However, results of this basin-wide analysis indicate that the GCS were largely effective in halting the progression of headcuts resulting from base lowering due to channelization that occurred in the 1960s and 1970s.

The results of the sediment analysis suggest the GCS have "frozen" the streams in CEM time. Upstream of the structures, large amounts of finer sediments (clays and silts) have been deposited, while the reaches downstream remain fairly sediment starved, evident from the high percentage of hard clay bedrock and gravels. In comparison, the control streams give examples of CEM in progress. Several of the control streams show signs of active incision, such as greater BHR and coarser substrates (CEM stage III) at the upstream sites. The downstream sites show evidence of CEM stages IV and V including higher WDR, greater ER, and greater deposition of fine substrates.

Given that the GCS seem to have frozen the streams in CEM time, I would expect the fish community to respond. Multiple studies have identified instream physical habitat characteristics to be a primary influence on fish assemblage composition, especially in modified streams (Miltner and Rankin 1998; Caskey and Frey 2009; Sanders et al. 2020). Thus, upstream of the structures, I would expect assemblages consistent with headwater stream assemblages, particularly those found in slackwater pool habitat (Harvey and Stuart 1991). Downstream from the structures, I can expect species indicative of large systems with fast, clear water, such as larger-bodied cyprinids, darters, and redhorses. In the control streams where CEM is continuing,

I would expect assemblages dominated by species adapted for dynamic conditions (Schlosser 1987). These expectations are investigated in the next chapter.

To further evaluate the success of different types of erosion control methods, future work should focus on comparisons of modern cross-sections with the pre-DEC profiles to analyze degree of change and compare with the model predictions of Martin et al. (2010) and Simon and Darby (2002). A basin-wide comparison of stream profiles would also be an excellent way to assess the success of the DEC program as a whole, since GCS were only one of the erosion control measures installed throughout the life of the DEC project.

Tables

Table 2.1　　　Stream geomorphology parameters measured or calculated for each transect.

Parameter	Abbreviation	Unit	Definition	Range of measured values
Bankfull Width	W_{bkf}	m	Width of the stream measured at bankfull stage	2-27.75
Width at Top of Banks	W_{tob}	m	Width of the channel measured at the elevation of the lowest bank	8.23-76.54
Cross-sectional Area at Bankfull	A_{bkf}	m^2	The area of the stream profile calculated at the bankfull stage	0.5-21.85
Maximum Depth at Bankfull	D_{max}	m	Difference in elevation measured between bankfull stage and the deepest point in the stream profile	0.14-1.6
Mean Depth at Bankfull	D_{bkf}	m	Average difference in elevation between bankfull stage and the stream bed; calculated as A_{bkf}/W_{bkf}	0.12-1.09
Depth of Floodplain Area Stage	D_{fpa}	m	Difference in elevation between the stage at which the stream is connected to its floodplain and the stream bed; defined as $2*D_{max}$	0.28-3.2
Width of Floodplain Area	W_{fpa}	m	Width of the stream measured when stage = D_{fpa}	3.32-39.45

Table 2.2   ANOVA table for the validation regression. Test statistic for the global test was an
             F test on 2 and 87 degrees of freedom. Other test statistics are t-tests between
             means within Type.

Coefficients	Estimate	St. Error	Test statistic	P value
Global Test		2.215	0.598	0.552
Intercept	4.3264	0.4044	10.698	2^{-16}
Type: incised	-0.4721	0.5719	-0.825	0.411
Type: stable	-0.5911	0.5719	-1.034	0.304
Residual		2.215		

Table 2.3   Comparison of the ER, BHR, and WDR in study sites compared to reference
             values presented in Harman et al. 2012.

Ratio	Yazoo Sites	Reference	Moderate Incision	Severe Incision
ER	1.1-1.8	> 2.2	1.41-2.28	1.0-1.4
BHR	3.1-21.9	1.1-1.2	1.2-2.0	>2.0
WDR	9.5-56.8			

Figures

h_c = CRITICAL BANK HEIGHT

I

$h < h_c$

SINUOUS, PREMODIFIED

II

$h < h_c$

FLOODPLAIN

h

CHANNELIZED

III

$h < h_c$

h

DEGRADATION

IV

$h > h_c$

TERRACE

h

DEGRADATION AND WIDENING

V

$h < h_c$

TERRACE

h

AGGRADATION AND WIDENING

VI

$h < h_c$

TERRACE

BERM

Q BERMFULL

h

QUASI EQUILIBRIUM

II

PRIMARY NICKPOINT

PRECURSOR NICKPOINT

PLUNGE POOL

III

TOP BANK

IV

DIRECTION OF FLOW

V

AGGRADED MATERIAL

VI

OVERSTEEPENED REACH

SECONDARY NICKPOINT

AGGRADATIONAL ZONE

Figure 2.1   Diagram of the CEM stages following Simon (1989, 1994).

Modified from Shields et al. 1998, with permission from Springer (license number 5360240034801).

44

Figure 2.2   Low-Drop Structure on Eskridge Creek, typical of GCS installed in the Yazoo
              Basin as part of the DEC project.

Photo taken in April 2022, about 30 years post-installation. Note the sheet-pile weir and the
circular stilling basin. Riprap revetment remains on the right descending bank but has been
eroded from the left bank. The baffle plate (steel structure in center of stilling basin) remains.
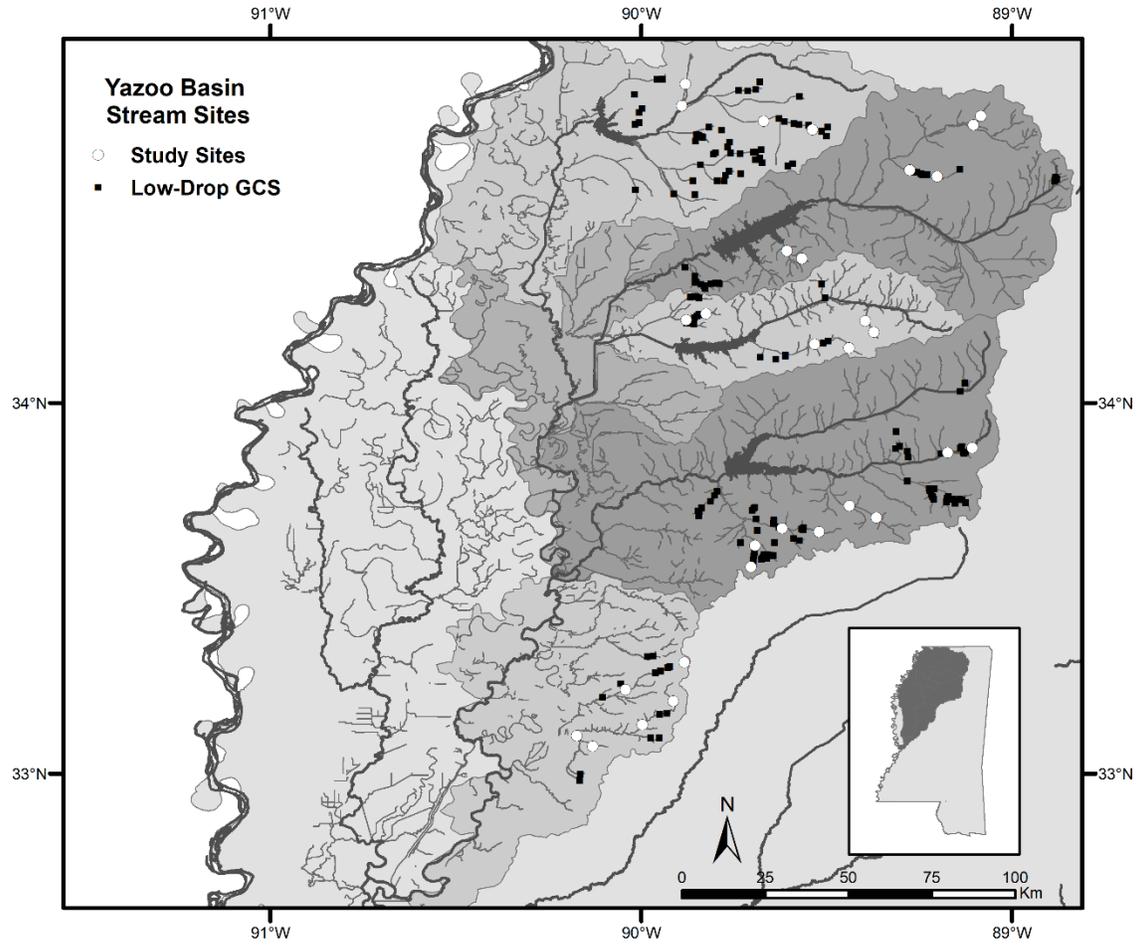
Figure 2.3   Map showing distribution of GCS in the Yazoo Basin.

Sites are coded as white circles, and low drop GCS are coded as black squares. Inset shows location of Yazoo Basin in Northwest Mississippi.
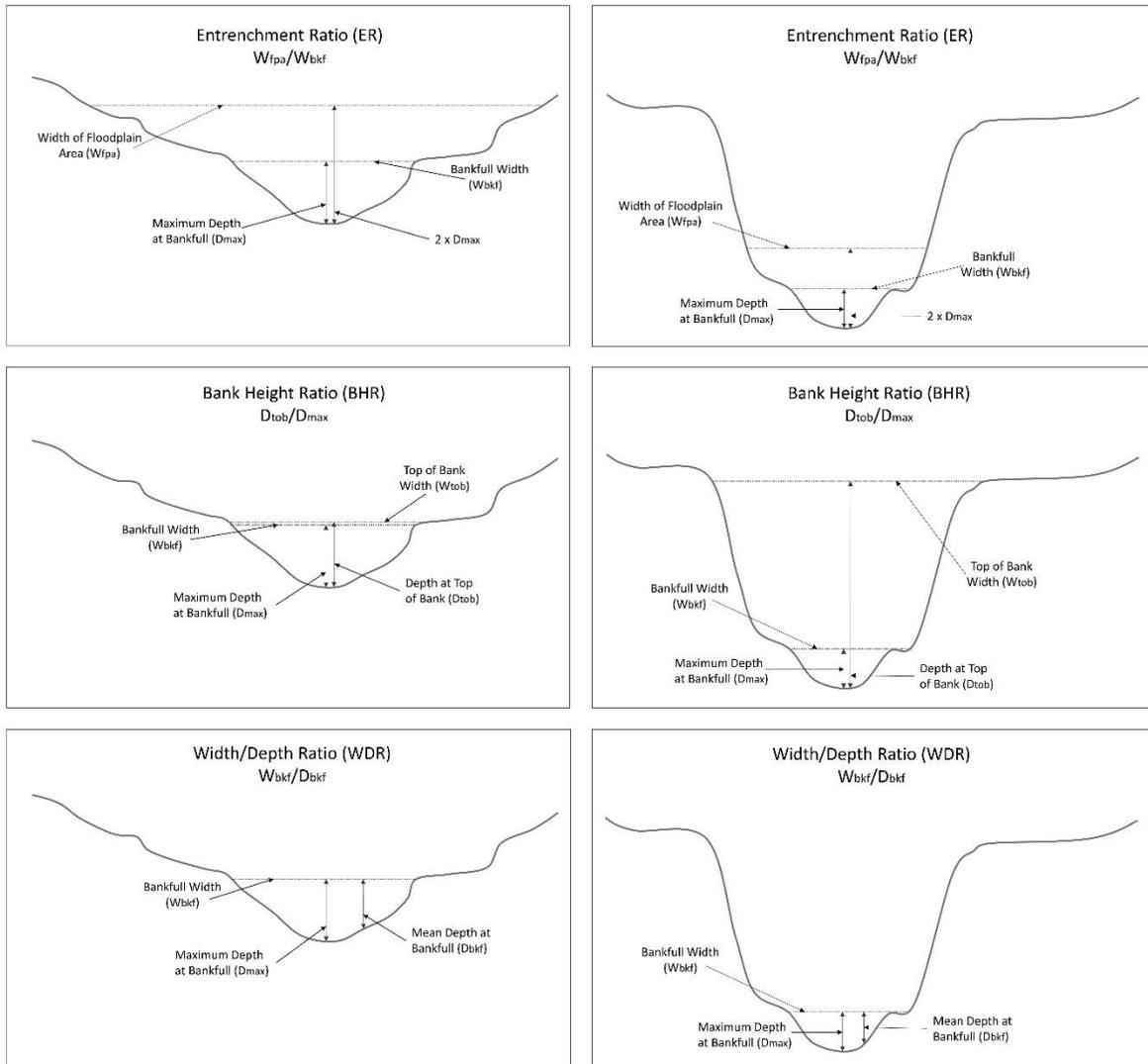
Figure 2.4   Illustrations of the three ratios used to quantify stream erosion.

The left column represents a stable stream while the right column illustrates the same measurements in an incised stream.
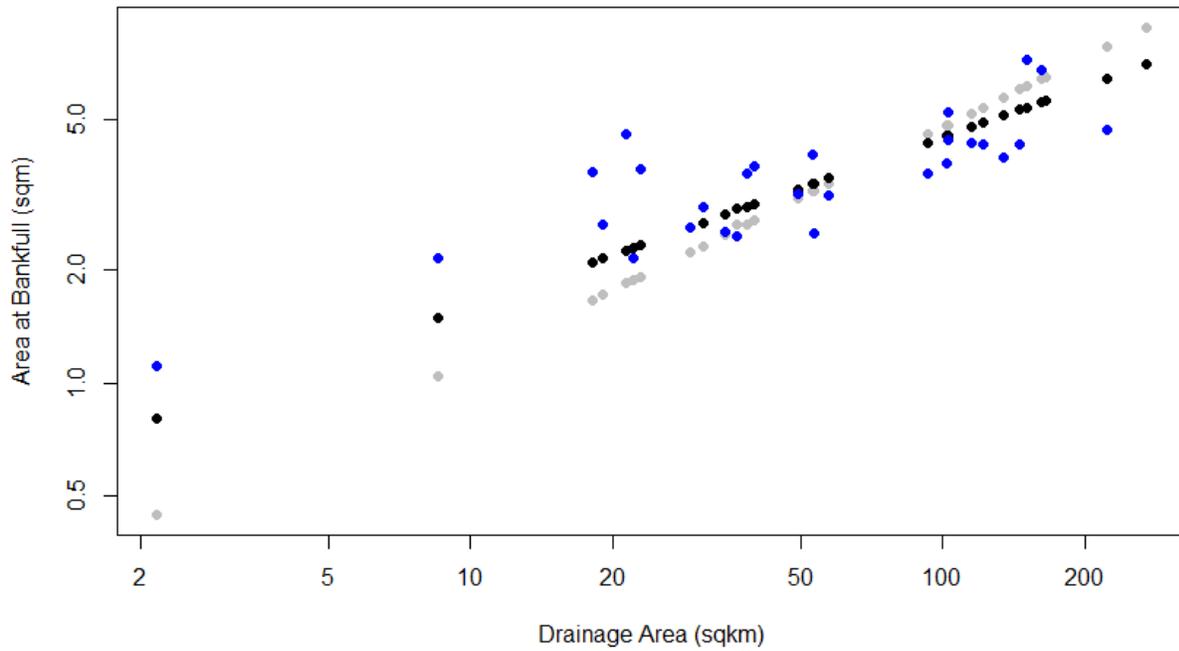
Figure 2.5   Scatter plot comparing field measured A$_{bkf}$ values to predicted values from regional regressions.

Symbols represent A$_{BKF}$ values. Blue points represent field measured values, grey points represent values predicted from the regional regression for incised streams, while black points represent values predicted from the regional regression for stable streams. Plot demonstrates relationship between A$_{BKF}$ and Drainage Area. Note that both axes are displayed on a log scale.
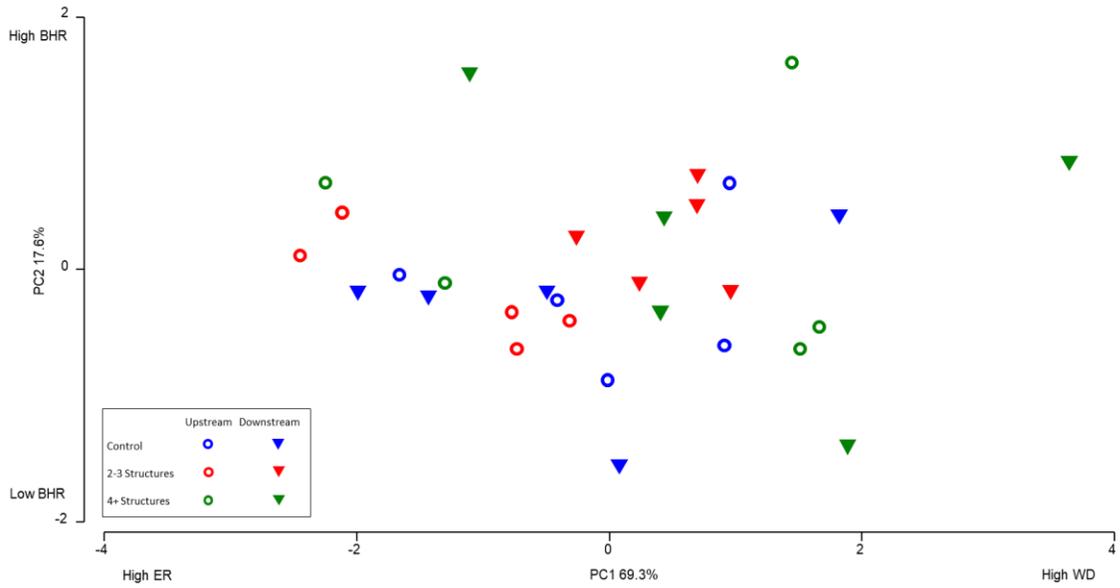
Figure 2.6   Principal component analysis ordination of the study sites according to ER, WDR, and BHR.

Sites are coded by GCS category and site location. Control sites are blue, sites on streams with 2-3 GCS are red, and sites on streams with 4+ GCS are green. Upstream sites are coded as open circles, while downstream sites are coded as triangles.
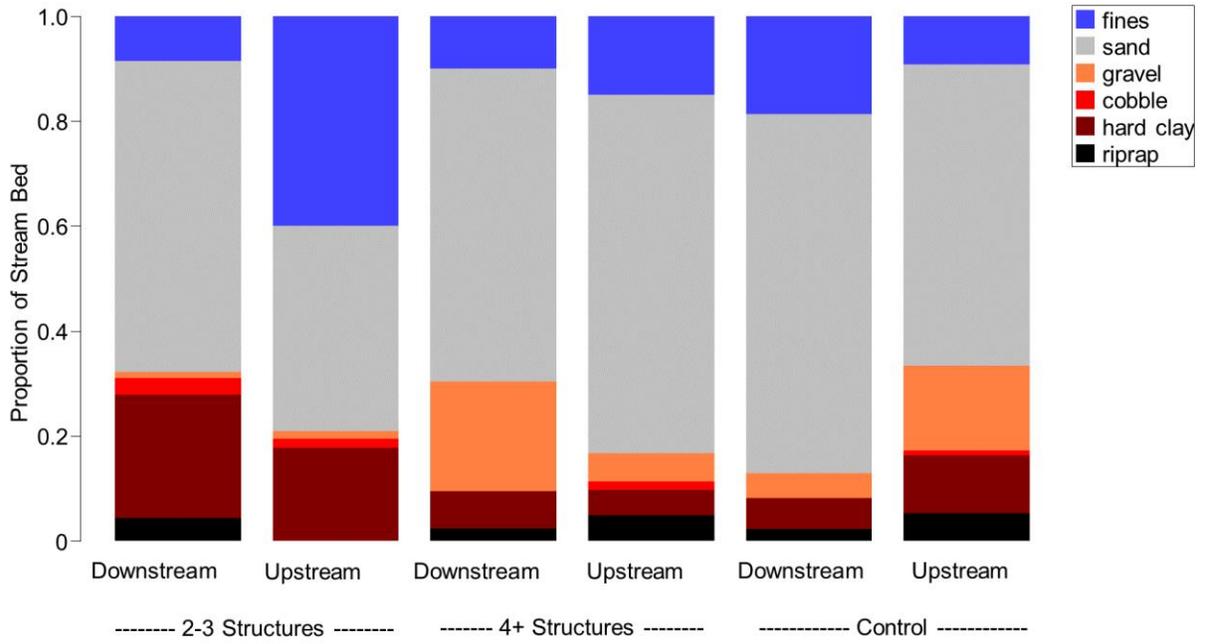
Figure 2.7   Bar plots representing the average proportions of sediment types for the stream beds by site category.

References

Abt, S. R., M. R. Peterson, C. C. Watson, and S. A. Hogan. 1992. Analysis of ARS low-drop grade-control structure. Journal of Hydraulic Engineering 118: 1424-1434.

Bentrup, G. 2008. Conservation Buffers: design guidelines for buffers, corridors, and greenways. General Technical Reports SRS-109. Ashville, NC: Department of Agriculture, Forest Service, Southern Research Station. 110 p.

Biedenharn, D. S. and L. C. Hubbard. 2001. Design considerations for siting grade control structures. Coastal and Hydraulics Engineering Technical Note CHETN-VII-3 U.S. Army Research and Development Center, Vicksburg, MS. http://chl.wes.army.mil/library/publications/chetn/.

Biedenharn, D. S. and C. C. Watson. 2012. The Delta Headwaters Project: A Review. Report given to Vicksburg District, US Army Corps of Engineers.

Blair, T. C., and J. G. McPherson. 1999. Grain-size and textural classification of coarse sedimentary particles. Journal of Sedimentary Research. 60(1): 6-19. https://doi.org/10.2110/jsr.69.6

Caskey, B.J., and Frey, J.W., 2009, Biological-community composition in small streams and its relations to habitat, nutrients, and land use in agriculturally dominated landscapes in Indiana and Ohio, 2004, and implications for assessing nutrient conditions in Midwest streams: U.S. Geological Survey Scientific Investigations Report 2009–5055, 21 p.

Coulston, J. W., G. G. Moisen, B. T. Wilson, M. V. Finco, W. B. Cohen, and C. K. Brewer. 2012. Modeling percent tree canopy cover: a pilot study. Photogrammetric Engineering and Remote Sensing 78: 715-727.

Dabney, S. M., F. D. Shields, R. L. Binger, R. A Kuhnle, and J. R. Rigby. 2012. Watershed management for erosion and sedimentation control case study Goodwin Creek, Panola County, Pages 539-568 *in* R. Lal and B. A. Stewart, editors. Advances in Soil Science, Soil Water and Agronomic Productivity, Taylor and Francis, Boca Raton, FL.

Daniels, R. B. 1960. Entrenchment of the willow drainage ditch, Harrison, County, Iowa. American Journal of Science 258: 161-176.

Doyle, M. W., D. Shields, K. F. Boyd, P. B. Skidmore, and D. Dominick. 2007. Channel-forming discharge selection in river restoration design. Journal of Hydraulic Engineering 133: 831-837. https://doi.org/10.1061(ASCE)0733-9429(2007)133:7(831).

Duncan, W. W., R. B., Goodloe, J. L. Meyer, and E. S. Prowell. 2011. Does channel incision affect in-stream habitat? Examining the effects of multiple geomorphic variables in fish habitat. Restoration Ecology 19(1): 64-73. https://doi.org/10.1111/j.526-100X.2009.00534.x.

Emerson, J. W. 1971. Channelization: a case study. Science 173: 325-326.

Hadadin, N. 2017. Variation in hydraulic geometry for stable versus incised streams in the Yazoo River Basin – USA. International Journal of Sediment Research 32: 121-126. https://doi.org/10.1016/j.ijsrc2016.03.003.

Harman, W., R. Starr, M. Carter, K. Tweedy, M. Clemmons, K. Suggs, and C. Miller. 2012. A function-based framework for stream fish assessment and restoration projects. US Environmental Protection Agency, Office of Wetlands, Oceans, and Watersheds, Washington D.C. EPA 843-K-12-006.

Harvey, B. C., and A. J. Stewart. 1991. Fish size and habitat depth relationships in headwater streams. Oecologia 87; 336-342.

Hitt, N. P., K. M. Rogers, Z. A. Kelly, J. Henesy, and J. E. Mulligan. 2020. Fish life history trends indicate increasing flow stochasticity in an unregulated river. Ecosphere 11(2): se03026.10.1002/ecs2.3026.

Lane, E.W., 1955. Importance of fluvial morphology in hydraulic engineering. Proceedings (American Society of Civil Engineers) 81: paper no. 745.

Leech, J. R. and D. S. Biedenharn. 2010. Sediment management at the watershed level. Report given to the U.S. Army Corps of Engineers Engineer Research and Development Center.

Martin, M., D. S. Biedenharn, C. Little, K. Smith, and C. C. Watson. 2010. Analysis of three delta headwaters project (DHP) streams using the sediment impact analysis method (SIAM) model. Proceedings from the 2nd Joint Federal Interagency Conference. Las Vegas, NM. 27 Jun - 01 Jul 2010.

Miltner, R. J., and E. T. Rankin. 1998. Primary nutrients and the biotic integrity of rivers and streams. Freshwater Biology 40: 145-158.

Raborn, S. W. and H. L. Schramm Jr. 2003. Fish assemblage response to recent mitigation of a channelized warmwater stream. River Research and Applications 19: 289-301. https://doi.org/10.1002/rra.704.

Ramser, C. E. 1930. Erosion and silting of dredged drainage ditches. Technical Bulletin 184, U.S. Department of Agriculture, Washington D.C.

Rebich, R. A. 1995. Estimation of sediment-discharge reduction for two sites of the Yazoo River Basin Demonstration Erosion Control Project, North-Central Mississippi, 1985-94. Water-Resources Investigations Report 95-4198. U.S. Geological Survey, Washington D.C.

Rosgen, D. L. 1994. A classification of natural rivers. Catena 22(3): 169-199.

Sanders, K. E., P. C. Smiley Jr., R. B. Gillespie, K. W. King, D. R. Smith, E. A Pappas. 2020. Conservation implications of fish-habitat relationships in channelized agricultural headwater streams. Journal of Environmental Quality 49: 1585-1598.

Schlosser, I. J. 1987. A conceptual framework for fish communities in small warmwater streams. p. 17-24 in W. J. Matthews and D. C Heins, eds. Community and evolutionary ecology of North American stream fishes. University of Oklahoma Press, Norman, OK.

Schumm, S. A., Harvey, M. D., and C. C. Watson. 1984. Incised channels: morphology, dynamics, and control. Water Resources Publications.

Schumm, S. A. 1985. Patterns of alluvial rivers. Annual Review of Earth and Planetary Sciences 12, 5-27.

Shields, F. D., S. S. Knight, and C. M. Cooper. 1995. Use of the index of biotic integrity to assess physical habitat degradation in warmwater streams. Hydrobiologia 312(3): 191-208.

Simon, A. 1989. A model of channel response in disturbed alluvial channels. Earth Surface Processes and Landforms, 14(1): 11–26.

Simon, A., 1994. Gradation processes and channel evolution in modified West Tennessee streams: process, response and form. U.S. Geological Survey Professional Paper 1470, Washington, DC

Simon, A., and S. E. Darby. 2002. Effectiveness of grade-control structures in reducing erosion along incised river channels: the case of Hotophia Creek, Mississippi. Geomorphology 42: 229-254.

Simon, A. and M. Rinaldi. 2006. Disturbance, stream incision, and channel evolution: the roles of excess transport capacity and boundary materials in controlling channel response. Geomorphology 79: 361-383. https://doi.org/10.1016/j.morph.2006.06.037.

Stearman, L. W. and J. F. Schaefer. 2022. Long-term minnow community trait shifts and metacommunity dynamics in a geomorphically unstable river. Ecology of Freshwater Fish 00: 1-13. https://doi.org/10.1111/eff/12660.

U.S. Geological Survey. 2021. The StreamStats program v.4.3.11, accessed 07 December 2021 at URL http://streamstats.usgs.gov.

Watson, C. C., D. S. Biedenharn, and C. R. Thorne. 1999. Demonstration Erosion Control Design Manual. U.S. Army Corps of Engineers, Vicksburg District.

Whitten, C. B., and Patrick, D. M. (1981). Engineering geology and geomorphology of stream-bank erosion; Report 2, Yazoo River Basin Uplands, Mississippi. Geotechnical Laboratory, US Army Corps of Engineers.

Wilson, K. V. 1979. Changes in channel characteristics, 1938-74, of the Homochitto River and tributaries, Mississippi. No. 70-554. U.S. Geological Survey.

Winemiller, K. O. and K. A. Rose. 1992. Patterns of life-history diversification in North American fishes: implications for population regulation. Canadian Journal of Fisheries and Aquatic Sciences 49: 2199-2218.

CHAPTER III

LONG-TERM EFFECTS OF LOW-DROP GRADE CONTROL STRUCTURES

ON FISH BETA DIVERSITY

Introduction

Streams are dynamic systems which typically change over periods of time ranging from

decades to centuries. In stable streams, channel geometry, sediment size, and slope remain

steady over long time periods as the stream lies in an equilibrium state between sediment

transport capacity and sediment supply (Leopold et al. 1964). Although the stream may shift

back and forth over the landscape reflecting local sediment aggradation patterns, in southeastern

streams, most stable streams produce run-riffle-pool habitat types (Jowett 1993). This

heterogeneity in habitat supports a diverse, relatively stable fish assemblage (Schlosser 1987).

Changes in climate or land use can, however, alter the discharge, sediment supply, and/or

gradient of the streams, which often leads to degradation and subsequent aggradation as the

streams attempt to rebalance sediment transport capacity with sediment supply. This process of

degradation and aggradation goes through a defined series of stages, known as the Channel

Evolution Model (CEM; Schumm et al. 1984). As the channel degrades, valuable instream

habitat such as gravel/cobble riffles and deep pools may disappear as they become covered in

depositional sediment. The transition to more homogenous shallow runs over finer sediments

can lead to a change in the fish assemblage. The lentic and riffle specialists (e.g., centrarchids

and percids) decline and are replaced by generalist or run-specific species (cyprinids) often resulting in reduced diversity (Rabeni and Smale 1995; Waters 1995).

In terms of stream stabilization and restoration, millions of federal and state dollars are spent annually to reduce erosion and return natural ecosystem services to degraded streams. In the process of stream restoration, stabilization structures and stream habitats are meticulously designed and implemented. These management efforts are usually monitored for only a short period (1-10 years) although the temporal scale needed to detect biotic results may be much longer (Lepori et al. 2005; Nilsson et al. 2015). Langford et al. (2009) suggests that the timescale for macroinvertebrate recolonization of rehabilitated stream reaches may extend as long as 30 years, especially in reaches that are farther removed from source populations of colonizers. Harding et al. (1998) found that the strongest predictor for diversity of fish and invertebrates in two North Carolina river basins was the 'Ghost of Land Use Past,' quantified as the land use of the catchment 50 years prior to sample collection. Thus, a major research need in this field involves examining the long-term effects of stream rehabilitation projects on stream stability, habitat conditions, and the biotic community that resides in them.

The loess bluff streams in the eastern half of the Yazoo Basin in Northern Mississippi present an ideal locale to investigate these effects. This area has experienced over two centuries of land use change and subsequent erosion. In the 1800s, the native hickory-oak forests were cleared, and the land was converted to agriculture. After the vegetative cover was removed, erosion of fine-grained loess topsoil from fields into streams increased, leaving deep gullies and ravines in its wake, as the highly erodible sandy-gravel layers were exposed. As a result, the tributary streams in the Yazoo Basin experienced vast changes in channel geomorphology caused by erosion and anthropogenic channelization efforts. The federal government attempted to

mediate this erosion by restoration efforts at the landscape scale beginning in 1945 (Yazoo-Little

Tallahatchie Project; Dabney et al. 2012) and at the channel scale in the 1980s (Demonstration

Erosion Control, DEC; Biedenharn and Watson 2012). When these projects were implemented,

the focus was to curb the physical effects of erosion. The DEC project used experimental

watersheds to examine the effectiveness of bed and bank control structures and hydraulic control

structures. After the initial evaluation period, the most successful methods were applied widely

over the basin. One of the most applied structures was low-drop grade control structures (GCS),

which were designed to halt the headcutting process by mediating changes in channel gradient up

to 1.8 m (6 ft) (Figure 3.1). GCS consist of a concrete or steel sheet-pile weir constructed either

at grade or slightly above the current grade to encourage sediment deposition (Abt et al. 1992;

Beidenharn and Hubbard 2001). Below the weir, a large riprap reinforced stilling basin with a

baffle plate serves to dissipate the energy of the stream before it enters the channel downstream

of the structure. The overarching goal of my study is to examine how large temporal scale

restoration efforts, such as the widespread application of GCS, have affected the biotic stream

community.

Objectives

The objective of this study was to determine if the GCS produced long-term effects on

the fish assemblage. The GCS could influence the fish assemblages through two major

processes. Process 1 involves altering stream morphology and instream habitat, whereas process

2 involves altering connectivity by reducing upstream fish passage over the structure. The GCS

were designed to prevent the upward movement of headcuts, and thus halt the progress of

channel evolution, which would result in different types of instream habitat upstream vs

downstream of the structures. Hence, if the installations were successful, upstream of the

structures, the channel should be shallower, with more large woody debris due to greater access

to the floodplain while downstream of the structures, the channel should be wider, more incised,

with greater aggradations of sediment. By changing the mesohabitat types available in a reach, I

would expect different species to replace those that once inhabited it, known as species

replacement (Schlosser 1987). Species replacement is often the result of such landscape filtering

mechanisms (Baselga 2010; Zbinden and Matthews 2017). In contrast, the GCS may be altering

upstream fish passage since periods of higher discharge are required to raise water surface

downstream of the GCS to match the upstream water surface. If this affecting the assemblage, I

would expect downstream species unable to swim over the GCS in high flows to be missing from

the upstream sites, resulting in a simplified assemblage that is nested in the downstream

assemblage (Taylor and Warren 2001; Roberts and Hitt 2010). Considering the overall beta

diversity of each stream, for process 1, I expect species replacement to contribute more to any

differences between assemblages, while for process 2, I would expect nestedness to contribute

more to beta diversity between upstream and downstream sites (Figure 3.2; Edge et al. 2017).

Methods

Study area

The study was conducted on first to third order streams in the bluff hills in the eastern

half of the Yazoo River Basin. The soils in the area consist of a layer of yellow, loamy loess,

wind carried sediments that were deposited during the Pleistocene, overlying slightly older

sandy, gravelly layers (Whitten and Patrick 1981; Grissenger and Murphey 1983). These soils

are highly erodible and have allowed channel incision to run unchecked though most of the

tributary streams in the basin. GCS (Figure 3.1) were installed in 1989-1995 to prevent further

incision through the upstream movement of headcuts. GCS were installed at or slightly upstream

of an active knickpoint (Biedenharn and Hubbard 2001), which in some cases resulted in

multiple GCS installed in series on the same stream. Over 160 GCS were installed throughout

the basin (Watson et al. 1999).

I selected fifteen streams balanced across five subbasins (Figure 3.3) for sampling the

fish assemblage and stream conditions. Five streams had two to three GCS in series, five

streams had more than four GCS in series, and five streams without GCS were included to serve

as a degraded control. Nearly all the streams in the Yazoo Basin show some extent of channel

incision, so these streams are not acting as true reference streams for the best-case scenario but

rather reflect the condition where channel incision remains unchecked. A pair of sites were

designated on each stream: one upstream from the GCS and one downstream. For the control

streams (i.e., no structure), the paired sites were distanced similarly to the treatment streams.

Distance between the paired sites averaged 8.5 river kilometers.

Sites at road crossings that had been sampled previously over the course of the DEC

project were used to allow for detection of long-term trends in later studies. To minimize the

effects of the bridge or culvert design on the stream morphology, the reaches sampled were

designated upstream of the crossing whenever possible, with the downstream end occurring

within the tree line (upstream from the cleared right of way associated with the road or highway.)

Field methods

Fish Sampling

Fish were collected with a combination of electrofishing and seining. At each site, a 100-

m reach was blocked at both ends with weighted 6x1.5 m, 5-mm delta mesh, seines held in place

with rebar. Three-pass electrofishing was conducted within the enclosure using a Smith-Root

LR-24 backpack electrofishing unit; to reduce visibility bias associated with turbid streams,

stunned fish were collected using a 3x1.5 m, 5-mm delta mesh, seine with a zinc-plated steel

chain added to the lead line for extra weight (Adams et al. 2004; Haden and Wagner 2021).

Before each pass, the amperage of the unit was adjusted for conductivity to reach an average of

25 watts of power output. For each pass, one person operated the electrofisher and was

accompanied by another person who carried a dip net and bucket to collect stunned fish. The

seine was crewed by two additional people. Passes began at the downstream end of the reach

and worked upstream to avoid increasing the turbidity of the unsampled portion of the reach.

Approximately 30 minutes were spent on each pass to standardize effort across sites. In small

streams (wetted width <4 m), all habitats were thoroughly sampled. In larger streams, the stream

was visually divided longitudinally into approximate thirds, and with each haul, the thirds were

alternated in a diagonal pattern as the crew moved upstream. Each pass began on a different

third, such that all the available habitats were sampled after completing three passes.

Mesohabitats within the 100-m reach were sampled following Price and Peterson (2010).

Riffles were sampled using short (~3 m long) 'kick-sets' where the seine was posted below the

riffle, and the electrofishing team moved downstream from the head of the riffle, shocking and

disturbing the substrate (kicking) to dislodge stunned fish into the seine. Depending on depth

and current velocity, runs were sampled by electrofishing in an upstream direction for the length

of the run, with the seine following about 2 m behind the electrofishing crew (shallow or slower

runs) or were sampled in a series of kick-sets when the current was too swift or the stream too

deep to prevent efficient upstream movement of the seine. To sample pools, the seine was

generally posted with one brail against the bank while the other end angled upstream around the

pool to prevent fish escape while the electrofisher applied current to the pool from the upstream

end. After current was applied, the seiners completed a haul through the pool to the bank to

collect the stunned fish. If the pool was too deep for safe electrofishing (depth > 0.8 m), only

seining was applied. Structures such as large woody debris, undercut banks, or instream trash

(e.g., discarded tires) were encircled with the seine, then electricity was applied. The current was

often enough to push stunned fish into the seine, but dipnets were also used to retrieve stunned

fish. Some of the sites (3 downstream sites; 5 upstream sites) had extremely low conductivities

(i.e., < 90 microSiemens/cm) or stream conditions (i.e., sudden deep pockets and slick clay

substrates, combined with turbid water) that hampered the use of electrofishing. Fish at these

sites were collected with 3-pass seining as described above except for runs, which were sampled

in long downstream hauls where the seine was moving faster than the current.

All fish smaller than about 15 cm were placed into the bucket after each haul, while

larger fish were identified to species, measured for total length, and released outside of the

sampling reach. At the end of each pass, fish were euthanized using MS-222 and preserved in

10% buffered formalin. Each pass was preserved separately. Fish were later identified in the

laboratory using Robison and Buchanan (1988), Etnier and Starnes (1993), Ross et al. (2001),

and Boschung and Mayden (2004) with current names using Page et al. (2013).

Environmental Sampling

Prior to each fish sampling pass, temperature, specific conductivity, dissolved oxygen,

and turbidity were measured using a YSI ProDSS handheld multimeter. Measurements were

taken at the downstream end of the reach to assess changes in temperature, conductivity, and

turbidity between passes. Stream geomorphology (run, riffle, pool), channel dimensions, and

substrate were measured across stream transects spaced every 20 m. At each transect, channel

width, width at bankfull, and wetted width were measured to the nearest 0.1 m. Within the

wetted width, a minimum of three points along each transect were selected to measure water

depth and current velocity (measured at 0.6 water depth). In uniformly U-shaped stream beds, the points were chosen to represent the deepest point, and the two points equidistant between the deepest point and the edge of wetted width. For transects with complex bed profiles, the deepest point was always measured, and other points were positioned to record salient changes in transect depth. I used additional points as necessary to reflect transect complexity. The dominant substrate size was visually estimated using a modified Wentworth scale (bedrock, boulder, riprap, cobble, gravel, coarse sand, fine sand, silt, clay; Blair and McPherson 1999) at every point of the width measurements, including endpoints. Large woody debris was qualitatively assessed by visually estimating the percentage of the area of each transect (including 5 m up- and downstream of the transect line) covered by woody debris.

Basin-scale variables were calculated using GIS. Catchment area upstream of each site was obtained from USGS Streamstats (USGS 2021). Percent riparian cover was calculated using a 30 m buffer centered on the stream flowline to agree with Natural Resources Conservation Service recommendations for perennial streams in forested areas (Bentrup 2008) using Percent Canopy Cover data from the 2016 NLCD dataset (Coulston et al. 2012).

Analysis

Because I did not meet the assumptions (i.e., catching fewer fish on each successive pass) for estimating abundance using three-pass depletion, the abundance data were converted to catch per unit effort (CPE) as counts of individuals of each species summed across all three passes/total minutes fished in the standard 100 m sampling reach. The CPE values were then used to create an among-sites Bray-Curtis resemblance matrix. CPE data were square-root transformed to reduce right-skewness. To test for treatment effects, I applied a multivariate analysis of variance to test if the difference in CPE (multiple continuous variables) between the

upstream and downstream sites (categorical variable) differed between streams with two to three GCS, four or more GCS, or control (no GCS) (categorical variable). Gear (i.e., electrofishing and seining vs seining only) was also included as a categorical variable to account for any effect they may have on catch. The multivariate analysis of covariance was run with a permutation MANOVA (PERMANOVA) applied to the resemblance matrix. Non-metric multidimensional scaling () was also applied to each resemblance matrix to interpret graphically the results of the PERMANOVA. To investigate associations between the fish assemblage and habitat parameters, habitat parameters were correlated to the NMDS plots for visual interpretation.

To examine the effect of the GCS on diversity, I used Hill's effective numbers of species to allow for comparisons of species richness (H0), Shannon diversity (H1), and Simpson's index (H2) across sites with different sample sizes (Jost 2007; Ellison 2010; Roswell et al. 2021). Hill's numbers 0-2 for each site were estimated using Chao 1 extrapolation or individual-based rarefaction to facilitate comparisons between sites with unequal sample sizes (Chao et al. 2014). As noted by Colwell et al. (2012), extrapolation provides reliable estimates only up to roughly double the size of a sample, so I rarified or extrapolated all samples to a sample size of 115 fish (smallest catch was 59 fish). Rarefaction and extrapolation estimates were calculated using the iNEXT package in program *R* (Hsieh et al. 2016; R Core Team 2021). Potential differences in the diversity estimates at upstream and downstream sites across control streams, two to three GCS streams, and four or more GCS streams were assessed for each Hill number separately using a permutational analysis of covariance (PERANCOVA) applied to an among-sites similarity matrix computed with Euclidean distance, with log10 transformed catchment size as a covariate (Anderson 2017). The PERANCOVA was applied using PRIMER-E version 7 software (Plymouth, UK).

To assess whether potential differences in fish assemblages are caused by the GCS altering instream habitat or altering fish passage, I partitioned pairwise beta diversity into its species replacement and nestedness components (Baselga 2010; Legendre 2014). Following Zbinden and Matthews (2017) I used the betapart package in program *R* to calculate pairwise beta diversity values for the whole basin, then extracted the values between sites on the same stream (Baselga and Orme 2012). Lastly, I calculated the overall mean in the basin and the mean values for each GCS category.

Results

In total, 8790 fish from 58 species representing 14 families of freshwater fish were captured from 30 sites. Median catch per site was 224 fish (range 59-945) and 15 species (range 7-23). Catch was overwhelmingly dominated by cyprinids (79%), followed by percids (5.1%) and centrarchids (4.8%). Wetted width at the sites averaged 7.7 m, with predominately sandy substrates and sufficient dissolved oxygen (Table 3.1; Chapter 2 results).

The results of the CPE PERMANOVA indicated that GCS influenced the fish assemblage. Number of GCS (Pseudo *F*: 2.42; *P*: 0.001) and site location (Pseudo *F*: 2.32; *P*: 0.019) significantly affected assemblage similarity between sites, while gear type had no effect (Pseudo *F*: 1.20; *P*: 0.286), which is illustrated in the NMDS plot (Figure 3.4a). Examination of the NMDS plot (Figure 3.4b) revealed distinct differences between upstream and downstream sites for streams with GCS while streams without GCS lack such a distinction. I followed up the initial PERMANOVA with a pairwise PERMANOVA (permutational equivalent of a Tukey's HSD test) assessing similarity between site locations within GCS category. This analysis confirmed that the upstream assemblages of streams with both two to three GCS and four or more GCS differed significantly from the downstream assemblages (2-3: *t*: 1.52; *P*: 0.020; 4+: *t*:

1.51; *P*: 0.027), while the assemblages of upstream and downstream sites on the control streams were similar (*t*: 0.71, *P*: 0.838).

After extrapolating or rarefying samples to 115 fish, sample coverage for all sites ranged 0.936-0.990, indicating that at each site, I accounted for the species that represent at least 93.6% of the individuals possible to sample. Despite differences in the assemblages as reflected in the CPE PERMANOVA, there were no significant differences between the upstream and downstream sites for any GCS category for all three Hill diversity numbers. This could indicate that differences in the assemblages are due to differing species subsets rather than species missing upstream of the structures; i.e. species turnover is being driven by species replacement rather than nestedness. This interpretation is further supported by the analysis of species turnover: species replacement contributes more (73.2%) to the overall beta diversity than nestedness (26.8%) between the upstream and downstream sites on the same stream. In comparing streams with and without structures, streams with two to three GCS and greater than four GCS were both dominated by replacement rather than nestedness (Table 3.1). In comparison, the control streams, while still being dominated by replacement, show a much higher nestedness fraction.

As illustrated in Figure 3.4c, fish assemblages were dominated by three sets of species. The first set, coded in shades of blue, dominated the assemblages at downstream sites located in the upper right quadrant of the plot. These species are typically large-bodied and select for deeper water with swift current. The sites in the lower right quadrant of the plot are dominated by species indicative of sandy stream communities, coded in shades of red and orange. These species have wide temperature and conductivity tolerances, consistent with survival in unstable streams where the conditions can change rapidly. Finally, the upstream sites in the left half of

65

the plot are dominated by species (coded in green) indicative of small streams with pools, in-stream structure, and plenty of slack-water habitat. The environmental variables were correlated with the NMDS plots and support the species observations (Figure 3.4b). The upstream sites tended to have greater riparian cover, deeper pools, and more large woody debris GCS. The downstream sites in the upper right quadrant (most of them downstream from structures) were correlated with higher current velocities, larger substrates, and lower turbidity, which are characteristic of streams with a history of dredging and channelization, as many of the Yazoo streams. The downstream sites in the lower right quadrant of Figure 3.4b, mostly control sites, were correlated with large wetted-widths, small substrates, higher temperatures and higher specific conductivities indicative of streams currently in the widening and aggradational stages of channel evolution, which follow the progression of a knickpoint.

Discussion

Overall, the GCS seem to have influenced the stream assemblages upstream of the structures. Assemblages upstream of the GCS are correlated with deeper pools, higher riparian cover, and more large woody debris than downstream. Assemblages downstream of the GCS are correlated with two types of stream habitat. The first assemblage type represents large-bodied darters and minnows, as well as other rheophilic species that associate with fast current velocities, large substrates, and high dissolved oxygen, which are representative of streams that are actively downcutting their channels (CEM stage II). The second assemblage type is correlated with wide, sandy channels with little cover and high temperature and conductivities. The shape of these channels is reflective of channels that are in the widening and aggradation stages of channel evolution (CEM stages III and IV). This second assemblage type tends to be dominated by species extremely tolerant of dynamic conditions.

66

In comparison to the streams with GCS installed, the control stream assemblages were similar between upstream and downstream sites. Some control streams, such as Sabougla Creek Canal and Camp Creek Canal, were channelized in the 1960s and 1970s and have assemblages reflective of eroded channels. Other control streams show few signs of incision and are only in CEM stages I or II, such as Toby Tubby Creek and Potlockney Creek. The assemblages of these control creeks were more similar to the upstream sites of the GCS treatment streams but showed no discernable difference between upstream and downstream sites.

Examination of beta diversity can lend insight into the processes driving differences in fish assemblages. Nestedness is often a large driver of assemblage structuring in streams due to the longitudinal nature of streams and their physical characteristics (Edge et al. 2017); however, when disturbances interrupt these gradients and cause discrete changes in habitat characteristics, species replacement may become the dominant process (Balsega 2010; Zbinden and Matthews 2017). This seems to be what is occurring in the Yazoo Basin, given the abrupt differences in stream habitat upstream of the structures. A recent study conducted on headwater streams in the Little Tallahatchie subbasin (Hubbell et al. 2020), also in the Yazoo Basin, confirms this interpretation since these authors also found the beta diversity to be driven almost entirely by species replacement. Subsequent analysis of the assemblage composition revealed that changes in instream habitat represents a large portion of the variance in fish community.

Although the assemblage differences are most likely due to habitat differences, fish passage over GCS could potentially affect stream assemblages in the Yazoo Basin. Passage over a GCS is dependent on the frequency of connectivity, flow conditions during that connectivity, and the swimming performances of fish. Because weirs such as the ones constructed as part of the GCS may only be passable during elevated stage conditions, some streams experience

reduced connectivity since downstream water levels seldom overtop the weirs. For example, weir construction on the Condamine and Murray Rivers in Australia resulted in an estimate of only two days of passage/year in some cases (Keller et al. 2012). This is unlikely in my study basin, however, because the Yazoo Basin receives an average of 127-152 cm of rain a year, often as brief but severe thunderstorms in the summer. For example, over a ten-year period, the Yalobusha River at Vardaman, Mississippi (one of the study sites with historical gage information) averages 25 days/year where stage was greater than 2 m over base flow (GCS are designed for a drop height less than 1.8 m) (USGS gage 07281960). Thomas et al. (2009, 2011) found that steeply-sloped grade control structures can hamper the passage of slackwater fishes such as Channel Catfish *Ictalurus punctatus*, Bullhead Catfishes *Ameiurus sp.*, and Creek Chubs *Semotilus atromaculatus*. While I did not capture Channel Catfish upstream of the structures, I caught numerous Creek Chubs and Yellow Bullheads *A. natalis*. I also captured species such as Bluntnose Minnow *Pimephales notatus* and Creek Chubsucker *Erimyzon oblongus* that migrate into small tributaries to spawn. My results might indicate that the regional species pool has adjusted to a flashy hydraulic regime.

I confirmed the locations of over 160 GCS installed throughout the Yazoo Basin (Figure 3.3. Similar structures have been used to address headcutting in other basins. Over 970 GCS have been successfully used to stabilize streams in the Loess Hills of Iowa (Litvan et al. 2007; Thomas et al. 2009). GCS have also been installed in Nevada, China, and Taiwan (Nelson 2011; J. Wang et al. 2018; Y. K. Wang et al. 2018), reflecting the ubiquity of streams facing stabilization issues. Thus, the results presented here can inform the long-term view of stream-stabilization projects worldwide.

Tables

Table 3.1   Summary statistics for physical characteristics and water quality parameters of Yazoo Basin streams.

	Minimum	**Maximum**	**Average**
Wetted Width (m)	2.06	19.29	7.71
Pool Depth (m)	0.21	1.89	0.72
Average Water Depth (m)	0.13	0.72	0.40
Average Current Velocity (m/sec)	0.01	0.25	0.09
Maximum Current Velocity (m/sec)	0.03	0.50	0.14
Temperature (°C)	18.00	33.13	25.47
Specific Conductivity (mSiemens/cm)	0.034	0.265	0.125
Dissolved Oxygen (mg/L)	5.84	11.03	7.87
Turbidity (NTU)	5.27	175	39.72

Values calculated across sites. When multiple measurements were takes at each site, values were averaged before computing study-wide values.

Table 3.2   Mean beta diversity values between the paired sites for the entire basin and for each of the stream categories.

	β	β_{rep}	β_{nes}
Overall	0.686	0.502 (73.2)	0.184 (26.8)
Control	0.648	0.394 (60.8)	0.254 (39.2)
2-3 structures	0.731	0.558 (76.4)	0.173 (23.6)
4+ structures	0.679	0.555 (81.7)	0.124 (18.3)

β represents mean beta diversity, calculated as the Bray-Curtis dissimilarity index, β_{rep} represents the fraction of beta diversity that represents species replacement, and β_{nes} represents the fraction of beta diversity that represents nestedness. These fractions are represented as percentages of the mean beta diversity in parentheses.

Figures

Figure 3.1   Low-Drop Structure on Eskridge Creek, typical of GCS installed in the Yazoo
             Basin as part of the DEC project.

Photo taken in April 2022, about 30 years post-installation. Note the sheet-pile weir and the circular stilling basin. Riprap revetment remains on the right descending bank but has been eroded from the left bank. The baffle plate (steel structure in center of stilling basin) remains.
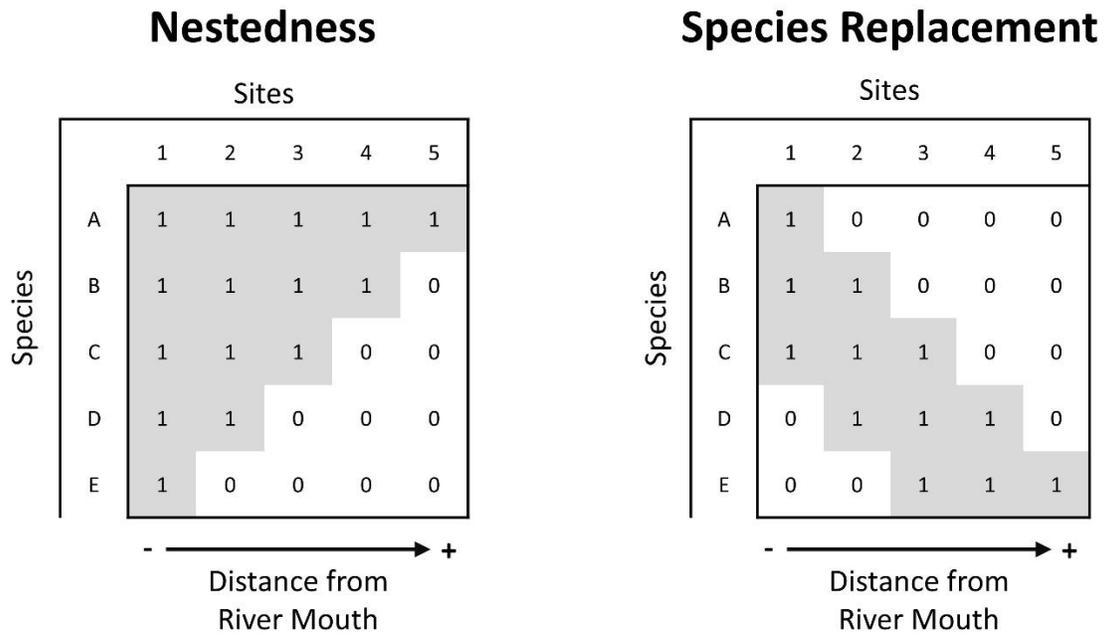
Figure 3.2   Diagram of expected partitioning of beta diversity.

Figure modified from Ulrich and Almeida-Neto (2012) to reflect theoretical responses in species occupancy to site location on a river distance gradient.
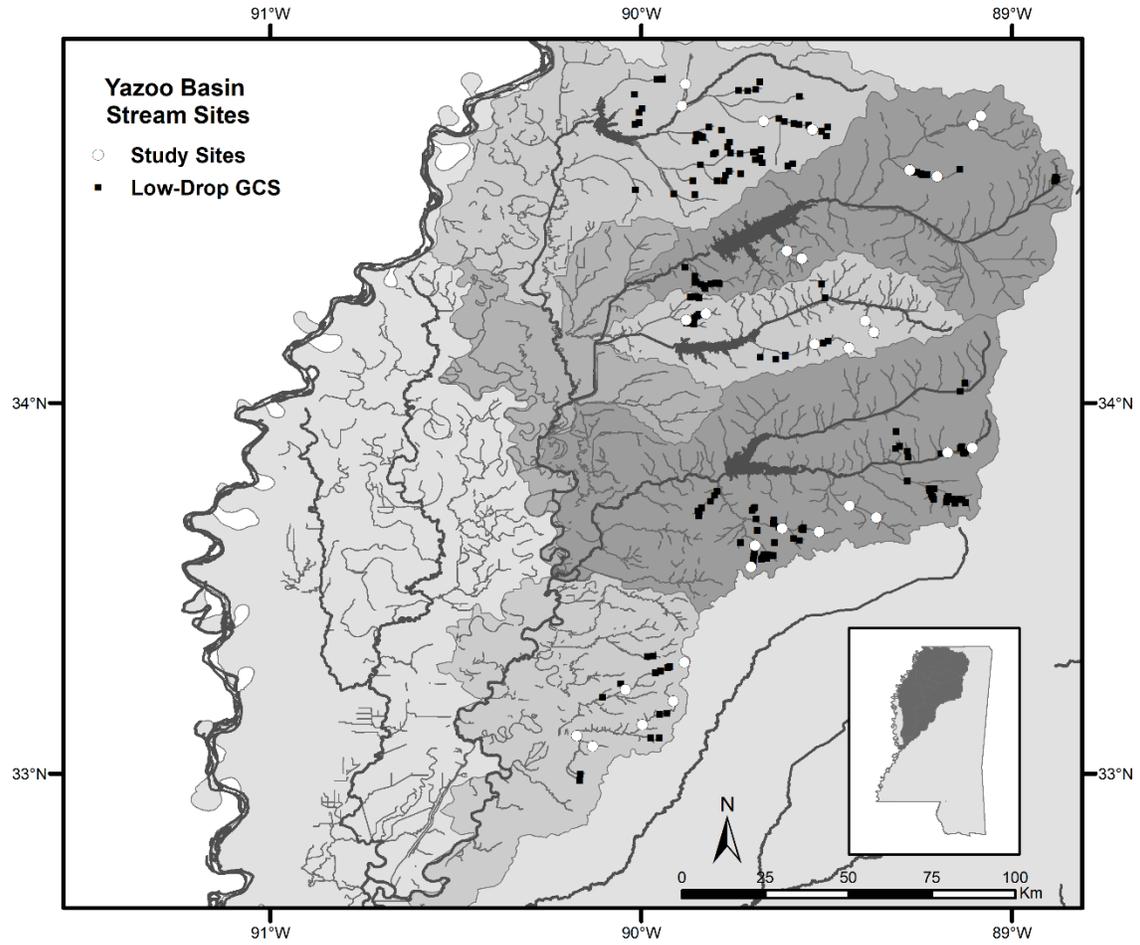
Figure 3.3   Map showing distribution of GCS in the Yazoo Basin.

Sites are coded as white circles, and low drop GCS are coded as black squares. Inset shows location of Yazoo Basin in Northwest Mississippi.
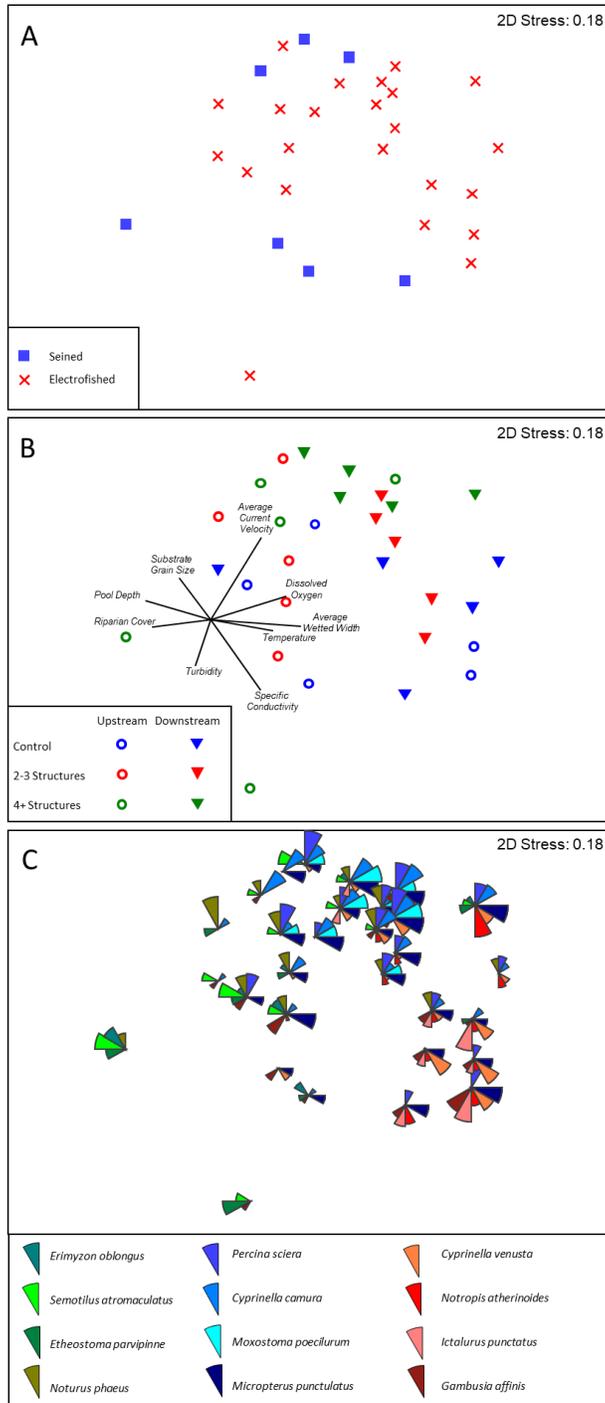
Figure 3.4   A series of NMDS plots illustrating results of the CPE PERMANOVA.

All plots show the sites in identical placement. Plot A illustrates the lack of differences between seined sites (blue squares) and electrofished (red Xs) sites. Plot B illustrates the differences in fish assemblages between upstream and downstream sites. Upstream sites are coded as hollow circles, downstream sites are solid triangles. Stream type is coded by color: control streams are

blue, stream with two to three GCS are red, and stream with four or more GCS are green. Vectors show the strength (length of ray) and direction of correlation between the environmental variables and the fish assemblage. Plot C illustrates species indicative of the differences in assemblages seen in Plot B. Green species represent slack water and riffle specialists common in headwater streams, blue species represent species adapted to larger streams with faster current and large substrates, and the red species are indicative of wide, braided sandy streams with little riparian cover.

References

Abt, S. R., M. R. Peterson, C. C. Watson, and S. A. Hogan. 1992. Analysis of ARS low-drop grade-control structure. Journal of Hydraulic Engineering 118; 1424-1434.

Adams, S. B., M. L. Warren, and W. R. Haag. 2004. Spatial and temporal patterns in fish assemblages of upper coastal plain streams, Mississippi, USA. Hydrobiologia 528; 45-61.

Anderson, M. J. 2017. Permutational Multivariate Analysis of Variance (PERMANOVA). Wiley StatsRef: Statistics Reference Online. https://:doi.org/10.1002/9781118445112.stat07841

Baselga, A. 2010. Partitioning the turnover and nestedness components of beta diversity. Global Ecology and Biogeography 19: 134-143. https://doi.org/10.1111/j.1466-8238.2009.00490.x

Baselga, A. and D. L. Orme. 2012. Betapart: an R package for the study of beta diversity. Methods is Ecology and Evolution 3: 808-812. https://doi.org/10.1111/j.2041-210X.2012.00224.x.

Biedenharn, D. S. and L. C. Hubbard. 2001. Design considerations for siting grade control structures. Coastal and Hydraulics Engineering Technical Note CHETN-VII-3 U.S. Army Research and Development Center, Vicksburg, MS. http://chl.wes.army.mil/library/publications/chetn/.

Biedenharn, D. S. and Watson, C.C. 2012. The Delta Headwaters Project: A Review. Report given to Vicksburg District, US Army Corps of Engineers.

Bentrup, G. 2008. Conservation buffers: design guidelines for buffers, corridors, and greenways. Gen. Tech. Rep. SRS-109. Asheville, NC: Department of Agriculture, Forest Service, Southern Research Station. 110 p.

Blair, T. C. and J. G. McPherson. 1999. Grain-size and textural classification of coarse sedimentary particles. Journal of Sedimentary Research. 60(1), 6-19. https://doi.org/10.2110/jsr.69.6

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

Chao, A., N. J. Gotelli, T. C. Hsieh, E. L. Sander, K. H. Ma, R. K. Colwell, and A. M. Ellison. 2014. Rarefaction and extrapolation with Hill numbers: a framework for sampling and estimation in species diversity studies. Ecological Monographs 84(1): 45-67. https://doi.org/10.1890/13-0133.1

Colwell, R. K., A. Chao, N. J. Gotelli, S. Y. Lin, C. X. Mao, R. L. Chazdon, and J. T. Longino. 2012. Models and estimators linking individual-based and sample-based rarefaction, extrapolation, and comparison of assemblages. Journal of Plant Ecology 5(1): 3-21. https://doi.org/10.1093/jpe/rtr044

Coulston, J. W., G. G. Moisen, B. T. Wilson, M. V. Finco, W. B. Cohen, and C. K. Brewer. 2012. Modeling percent tree canopy cover—A pilot study. Photogrammetric Engineering and Remote Sensing 78: 715-727. https://doi.org/10.14358/PERS.78.7.715.

Dabney, S. M., F. D. Shields, R. L. Binger, R. A Kuhnle, and J. R. Rigby. 2012. Watershed management for erosion and sedimentation control case study Goodwin Creek, Panola County, Pages 539-568 *in* R. Lal and B. A. Stewart, editors. Advances in Soil Science, Soil Water and Agronomic Productivity, Taylor and Francis, Boca Raton, FL.

Edge, C. B., M. Fortin, D. A. Jackson, D. Lawrie, L. Stanfield, and N. Shrestha. 2017. Habitat alternation and habitat fragmentation differentially affect beta diversity of stream fish communities. Landscape Ecology 32: 647-662. https://doi.org/10.1007/s10980-016-0472-9

Ellison, A. M. 2010. Partitioning Diversity. Ecology 91: 1962-1963.

Etnier, D. A. and W. C. Starnes. 1993. The Fishes of Tennessee. The University of Tennessee Press, Knoxville, TN.

Haden, L. L. and M. D. Wagner. 2021. Status, Distribution, and Length-Frequency for the Frecklebelly Madtom (Noturus miurus) in the Tombigbee River drainage in Mississippi. Southeastern Naturalist 20: 162-177. https://doi.org/10.1656/058.020.0120

Harding, J. S., E. F. Benfield, P.V. Bolstad, G. S. Helfman, and E. B. D. Jones III. 1998. Stream biodiversity: the ghost of land use past. Proceedings of the National Academy of Science 95: 14843-14847. https://doi.org/10.1073/pnas.95.25.14843

Hsieh, T. C., K. H. Ma, and A. Chao. 2016. iNext: an R package for rarefaction and extrapolation of species diversity (Hill numbers). Methods in Ecology and Evolution 7: 1451-1456. https://doi.org/10.1111/2041-210X.12613

Hubbell, J. P., J. F. Schaefer, P. Flood, M. L. Warren Jr., and K. A. Sterling. Fragmentation alters ecological gradients and headwater fish assemblage composition relative to land use in a dendritic river system. Canadian Journal of Fisheries and Aquatic Science 77: 1281-1291. https://doi.org/10.1139/cjfas-2019-0080.

Jowett, I. G. 1993. A method for objectively identifying pool, run, and riffle habitats from physical measurements. New Zealand Journal of Marine and Freshwater Research 27: 241-248. https://doi.org.10.1080/00288330.1993.9516563,

Jost, L. 2007. Partitioning diversity into independent alpha and beta components. Ecology 88: 2427-2439. https://doi.org/10.1890/06-1736.1

Keller, R. J., C. J. Peterken, and A. P. Berghuis. 2012. Design and assessment of weirs for fish passage under drowned conditions. Ecological Engineering 48: 61-69. https://doi.org/10.1016/j.ecoeng.2011.06.037

Langford, T. E. L., P. J. Shaw, A. J. D. Ferguson, and S. R. Howard, S. R. 2009. Long-term recovery of macroinvertebrate biota in grossly polluted streams: Re-colonization as a constraint to ecological quality. Ecological Indicators 9: 1064-1077. https://doi.org/10.1016/j.ecolind.2008.12.012

Legendre, P. 2014. Interpreting the replacement and richness difference components of beta diversity. Global Ecology and Biogeography 23: 1324-1334. https://doi.org/10.1111/geb.12207

Leopold, L. B., M. G. Wolman, and J. P. Miller. 1964. Fluvial Processes in Geomorphology. W. H. Freeman and Company. San Francisco, CA.

Lepori, F., D. Palm, E. Brännäs, and B. Malmqvist. 2005. Does restoration of structural heterogeneity in streams enhance fish and macroinvertebrate diversity? Ecological Applications 15: 2060-2071. https://doi.org/10.1890/04-1372

Litvan, M. E., T. W. Stewart, C. L. Pierce, and C. J. Larson. 2007. Effects of grade control structures on the macroinvertebrate assemblage of an agriculturally impacted stream. River Research and Applications 24: 218-233.

Nelson, S. M. 2011. Response of stream macroinvertebrate assemblages to erosion control structures in a wastewater dominated urban stream in the southwestern US. Hydrobiologia, 663: 51-69. https://doi.org/10.1007/s10750-010-0550-y

Nilsson, C., L. E. Polvi, J. Gardström, E. M. Hasselquist, L. Lind, and J. M. Sarneel. 2015. Riparian and in-stream restoration of boreal streams and rivers: success or failure? Ecohydrology 8: 753-764. https://doi.org/10.1002/eco.1480

Page, L. M., H. Espinosa-Pérez, L. T. Findlay, C. R. Gilbert, R. N. Lea, N. E. Mandrak, R. L. Mayden, and J. S. Nelson. 2013. Common and scientific names of fishes from the United States, Canada, and Mexico, 7th edition. American Fisheries Society special publication 34, Bethesda, MD.

Price, A. L. and J. T. Peterson. 2010. Estimation and modeling of electrofishing capture efficiency for fishes in wadeable warmwater streams. North American Journal of Fisheries Management 30: 481-498. https://doi.org/10.1577/M09-122.1

R Core Team. 2021. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Rabeni, C. F. and M. A. Smale. 1995. Effects of siltation on stream fishes and the potential mitigating role of the buffering riparian zone. Hydrobiologia 303; 211-219. https://doi.org/10.1007/BF00034058.

Roberts, J. H. and N. P. Hitt. 2010. Longitudinal structure in temperate stream fish communities: evaluating conceptual models with temporal data. American Fisheries Society Symposium 73, 281-299.

Robison, H. W. and T. M. Buchanan. 1988. Fishes of Arkansas. The University of Arkansas Press, Fayetteville, AR.

Ross, S. T., W. M. Brenneman, W. T. Slack, M. T. O'Connell, and T. L. Peterson. 2001. The Inland Fishes of Mississippi. University Press of Mississippi, Jackson, MS.

Roswell, M., J. Dushoff, and R. Winfree. 2021. A conceptual guide to measuring species diversity. Oikos 130: 321-338. https://doi.org/10.1111/oik.07202

Schlosser, I. J. 1987. A conceptual framework for fish communities in small warmwater streams. Pages 17-24 *in* W. J. Matthews and D. C Heins, eds. Community and evolutionary ecology of North American stream fishes. University of Oklahoma Press, Norman, OK.

Schumm, S. A., Harvey, M. D., and Watson, C. C. 1984. Incised channels: morphology, dynamics, and control. Water Resources Publications.

Taylor, C. M. and M. L. Warren. 2001. Dynamics in species composition of stream fish assemblages: environmental variability and nested subsets. Ecology, 82(8), 2320-2330. https://doi.org/10.1890/0012-9658(2001)082[2320:DISCOS]2.0.CO;2

Thomas, J. T., M. E. Culler, D. C. Dermiss, C. L. Pierce, A. N. Papanicolaou, T. W. Stewart, and C. J. Larson. 2011. Effects of grade control structures on fish passage, biological assemblages and hydraulic environments in western Iowa streams: a multidisciplinary review. River Research Applications 29; 389-398. https://doi.org/10.1002/rra.1600

Thomas, J. T., A. N. Papanicolaou, C. L. Pierce, D. C. Dermiss, and M. E. Litvan. 2009. Fish passage and abundance around grade control structures on incised streams. Natural Resource Ecology and Management Conference Papers, Posters, and Presentations 16. https://doi.org/10.1061/41036(342)312

Ulrich, W. and M. Almeida-Neto. 2012. On the meanings of nestedness: back to the basics. Ecography 35:865-871.

U.S. Geological Survey. 2021. The StreamStats program v.4.3.11, accessed 07 December 2021 at URL http://streamstats.usgs.gov.

Wang, J. M., X. G. Yang, H. W. Zhou, X. Lin, R. Jiang, and E. Q. Lv, 2018. Bed morphology around various solid and flexible grade control structures in an unstable gravel-bed river. Water 10: 822. doi:10.3390/w10070822

Wang, Y. K., C. C. Huang, C. H. Fuh, K. J. Jong, and S. T. Tsai, 2018. Effects of stream grade control structures on benthic algal communities in a subtropical stream. Inland Waters 8: 60-69. https://doi.org/10.1080/20442041.2018.1428423

Waters, T. F. 1995. Sediment in streams. American Fisheries Society, Monograph 7.

Watson, C. C., D. S. Biedenharn, and C. R. Thorne. 1999. Demonstration Erosion Control Design Manual. U.S. Army Corps of Engineers, Vicksburg District.

Whitten, C. B., and Patrick, D. M. 1981. Engineering geology and geomorphology of stream-bank erosion; Report 2, Yazoo River Basin Uplands, Mississippi. Geotechnical Laboratory, US Army Corps of Engineers.

Zbinden, Z. D. and W. J. Matthews. 2017. Beta diversity of stream fish assemblages: partitioning variation between spatial and environmental factors. Freshwater Biology 62: 1460-1471. https://doi.org/10.1111/fwb.12960