

Disentangling the ecological role of gizzard shad in Kansas small impoundment food webs:
changes in limnology, fish communities, and predator diets after rotenone manipulation

by

Elizabeth Anne Renner

B.A., Augustana University, 2017

AN ABSTRACT OF A DISSERTATION

submitted in partial fulfillment of the requirements for the degree

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Abstract

Benthic-feeding fishes can exert direct and indirect effects on multiple trophic levels in lentic ecosystems by increasing water column nutrient concentrations, stimulating phytoplankton abundances, and increasing turbidity. One benthic-feeding fish that has been the target of selective biomanipulations is the gizzard shad (*Dorosoma cepedianum*), a freshwater clupeid. In reservoirs across the United States, gizzard shad often dominate fish assemblages in terms of relative abundances and proportions of total fish biomass, especially in eutrophic to hypereutrophic systems. I tested the influence of gizzard shad on lentic ecosystems by sampling key limnological parameters in 12 small impoundments across a range of gizzard shad relative abundances over four years. I hypothesized that gizzard shad relative abundance would be positively associated with algal abundance, cyanobacterial abundance, and nutrient concentrations. Higher gizzard shad relative abundances were associated with elevated total nitrogen (TN) and total phosphorus (TP) during certain months and with a general reduction in zooplankton densities. Regardless of the apparent influence of gizzard shad abundance on top-down (zooplankton) and bottom-up (nutrient) processes that might affect primary producers, I found no relationship between gizzard shad relative abundance and cyanobacteria or total phytoplankton abundance. In a manipulative experiment, I tested whether selectively eradicating gizzard shad from food webs via replicated low-dose rotenone applications could precipitate responses in abiotic and biotic variables. I collected baseline measurements of limnological and fish community parameters from eight impoundments with gizzard shad for two years (2017 and 2018) before rotenone was applied to four treatment lakes in December 2018 and then monitored food web components for two years posttreatment (2019 and 2020). TN and TP decreased in treatment impoundments relative to control impoundments following shad removal, but I found

little evidence that gizzard shad removal had a substantial impact on limnological properties or fish populations. Finally, because gizzard shad are thought to be important forage for piscivorous fishes in reservoirs, I characterized the importance of gizzard shad in largemouth bass diets in 12 Kansas small impoundments and examined how piscivory varied across bass length classes, months, and a gradient of gizzard shad relative abundances, hypothesizing that largemouth bass would selectively prey upon gizzard shad compared to other forage fishes. Bass length class was a significant factor influencing the frequency of occurrence of fish (piscivory) in bass diets in our regression model, but piscivory was not associated with sampling month or gizzard shad relative abundance. Gizzard shad eradication did not alter the frequency of occurrence of fish in largemouth bass diets following rotenone treatment. Since the contribution of gizzard shad prey to bass diets was low relative to the contribution of centrarchids, cannibalism, and crayfish, selective shad removal via rotenone biomanipulation might not meaningfully alter largemouth bass diet composition. Overall, gizzard shad effects were weak and mixed, and our ability to detect responses was greatly limited by our small sample size of impoundments. My dissertation work sheds light on the ecological role of gizzard shad in small impoundments and illustrates the complexity of studying highly stochastic novel ecosystems managed for angling recreation.

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Approved by:

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Table of Contents

List of Figures	ix
List of Tables	xii
Acknowledgements	xiv
Dedication	xviii
Preface	xix
Chapter 1 - Gizzard shad mediate nutrient cycling in lentic food webs	1
Chapter 2 - Associations between gizzard shad (<i>Dorosoma cepedianum</i>) relative abundances and limnological parameters in small impoundments	13
Chapter 3 - Characterizing changes to lentic food webs in response to low-dose rotenone manipulations in Kansas small impoundments	44
Chapter 4 - Seasonal and ontogenetic variation in largemouth bass (<i>Micropterus salmoides</i>) diets across a gradient of gizzard shad abundance: bass piscivory does not change with shad eradication via low-dose rotenone	75
Chapter 5 - Conclusions	115
References	119
Appendix A - Supplemental Materials for Chapter 2	120
Appendix B - Supplemental Materials for Chapter 3	124
Appendix C - Supplemental Materials for Chapter 4	126

List of Figures

Figure 2.1 Map of 12 study impoundments in Kansas, USA (county borders are shown for reference). SFL = State Fishing Lake. All impoundments are managed for angling recreation by the Kansas Department of Wildlife & Parks (KDWP).	39
Figure 2.2 Photomicrograph of a phytoplankton sample under 400X magnification overlaid with a 10 x 10 μm counting grid. An index of relative biomass was estimated by the number of grids occupied by each natural unit or algal cluster.....	40
Figure 2.3 Plot of gizzard shad CPUE versus TN, TP, and molar N:P ratios. Gizzard shad were sampled with daytime shoreline electrofishing and quantified as CPUE (catch per unit effort, # shad/minute electrofishing). Gizzard shad densities were natural log ($x + 0.05$) transformed and TN and TP concentrations were natural log ($x + 0.01$) transformed. N:P ratios were square root transformed. Colors indicate month sampled. TN and TP were quantified using surface water grab samples collected concurrently with electrofishing. Trend lines were included if effects were significant at $\alpha = 0.05$	41
Figure 2.4 Gizzard Shad CPUE versus cyanobacteria abundance, green algae abundance, diatom abundance, and total phytoplankton abundance. Phytoplankton grab samples were collected from two stations (riverine and lacustrine) in June, August, and October 2017-2020. Gizzard shad were sampled with daytime shoreline electrofishing and quantified as CPUE (catch per unit effort, # shad/minute electrofishing). Gizzard shad densities were natural log ($x + 0.05$) transformed. Colors indicate month sampled. Trend lines were included if effects were significant at $\alpha = 0.05$	42
Figure 2.5 Gizzard Shad CPUE versus cladoceran density, calanoid copepod density, cyclopoid copepod density, rotifer density, nauplii density, and total zooplankton density. Three replicate 3-m zooplankton tows were collected at each of three stations (riverine, transition, lacustrine) in June, August, and October 2017-2020. Gizzard shad were sampled with daytime shoreline electrofishing and quantified as CPUE (catch per unit effort, # shad/minute electrofishing). Gizzard shad densities were natural log ($x + 0.05$) transformed and zooplankton densities were natural log transformed. Trend lines were included if effects were significant at $\alpha = 0.05$	43

Figure 3.1 Pre-treatment Gizzard Shad catch per unit effort (CPE, #/minute electrofishing) before rotenone manipulations (2017-2018). Relative abundances were obtained using daytime shoreline boat electrofishing in each impoundment.....	70
Figure 3.2 TN (left) and TP (right) concentrations before and after Gizzard Shad eradication. Pelagic nutrients were sampled at three stations in each impoundment in June, August, and October from 2017 to 2020. Stations were positioned in limnetic habitats in riverine zones, mid-lake transition zones, and lacustrine zones. Replicate 25 mL surface water grab samples were collected for Total N and P from a depth of 0.5 m from each station and analyzed for TN and TP with an Alpkem rapid flow analyzer.	71
Figure 3.3 Calanoid copepod densities before (left) and after (right) Gizzard Shad eradication in control and rotenone impoundments. Pelagic zooplankton communities were sampled via three replicate 3-m Wisconsin plankton net tows per station in June, August, and October 2017-2020. Vertical tows were performed at stations greater than 3.5 m deep and horizontal tows were performed at stations less than 3.5 m deep with approximate tow speeds of 1 m/s. Zooplankton were classified under one of seven taxonomic categories: Cladocera, Calanoida, Cyclopoida, Harpacticoida, Rotifera, Nauplii, and Culicomorpha (Chaoboridae). Mean zooplankton densities were calculated and reported as the average counts of each group per L.....	72
Figure 3.4 Relative abundance of Largemouth Bass < 150 mm TL before and after Gizzard Shad eradication. Data were collected via spring electrofishing in each impoundment by KDWP. Relative abundances were log-transformed.	73
Figure 3.5 Relative abundance of Bluegill greater than 100 mm TL (left) and less than 100 mm TL (right) before and after Gizzard Shad eradication. Bluegill were sampled via trap netting in each impoundment by KDWP from 2017-2020. Relative abundances were log-transformed.	74
Figure 4.1 Map of 12 study impoundments sampled for largemouth bass diet information from 2017-2020. SFL = State Fishing Lake. All impoundments are managed for angling recreation by the Kansas Department of Wildlife & Parks (KDWP).	109
Figure 4.2 Mean frequency of occurrence of prey items in largemouth bass diets by length class. Largemouth bass were collected from three length classes: individuals < 150 mm TL (top),	

individuals between 150 - 250 mm TL (middle), and individuals > 250 mm TL (bottom).	110
Figure 4.3 Mean frequency of occurrence of prey items in largemouth bass diets by month.	
Largemouth bass were collected from three months: June (top), August (middle), and October (bottom).....	111
Figure 4.4 Largemouth bass fish prey items. Fish prey were identified to the lowest possible taxon. Fish matter UNID was any fish tissue or structures that were too degraded or incomplete for positive identification to family.....	112
Figure 4.5 Frequency of occurrence of fish in largemouth bass diets summarized by length class.	
Largemouth bass were collected via daytime shoreline electrofishing and seining in each impoundment in June, August, and October from 2017-2020 from three length classes...	113
Figure 4.6 Frequency of occurrence of fish in largemouth bass diets before (left) and after (right) Gizzard Shad eradication. Largemouth bass were collected via daytime shoreline electrofishing and seining in each impoundment in June, August, and October from 2017-2020.....	114

List of Tables

Table 2.1. Linear mixed effect model type III ANOVA outputs. Gizzard shad were sampled with daytime shoreline electrofishing and quantified as CPUE (catch per unit effort, # shad/minute electrofishing). Shad relative abundances were either natural log (x + 0.01)- or natural log (x + 0.05)-transformed. All models followed the format: lmer(log (TN) ~ log (shadpermin + 0.05) * month * (1 lakecode)). Asterisks denote statistical significance at $\alpha = 0.05$	36
Table 2.2. Linear mixed effect model type III ANOVA outputs for reduced models disaggregated by month. Gizzard shad were sampled with daytime shoreline electrofishing and quantified as CPUE (catch per unit effort, # shad/minute electrofishing). Shad relative abundances were log (x + 0.01) transformed. All models followed the format: lmer(log(TN + 0.01) ~ log(shadpermin + 0.05) * (1 lakecode)). Asterisks denote statistical significance at $\alpha = 0.05$	38
Table 3.1 Length groups of Largemouth Bass and Bluegill collected for aging from Kansas small impoundments.	66
Table 3.2 Linear fixed effect model outputs for responses of abiotic parameters to rotenone treatments in Kansas small impoundments. Asterisks denote statistical significance at $\alpha = 0.10$	66
Table 3.3 Linear fixed effect model outputs for responses of phytoplankton abundances to rotenone treatments in Kansas small impoundments. Asterisks denote statistical significance at $\alpha = 0.10$	67
Table 3.4 Linear fixed effect model outputs for responses of zooplankton abundances to rotenone treatments in Kansas small impoundments. Asterisks denote statistical significance at $\alpha = 0.10$	68
Table 3.5 Linear fixed effect model outputs for responses of Largemouth Bass and Bluegill relative abundances in Kansas small impoundments. Largemouth Bass were sampled via daytime shoreline electrofishing and Bluegill were sampled via trap netting for relative abundance estimates from each impoundment in October 2017-2020. Asterisks denote statistical significance at $\alpha = 0.10$	69

Table 4.1 Summary of largemouth bass diet composition collected from Kansas small impoundments across all length classes and sampling months.....	94
Table 4.2 Frequency of occurrence of diet items in largemouth bass diet samples of individuals less than 150 mm TL. Samples are organized by impoundment, month, and year. Gizzard shad catch per unit effort (#/minute electrofishing) was calculated based on data from daytime shoreline electrofishing collected concurrently with diet samples.	95
Table 4.3 Frequency of occurrence of diet items in largemouth bass diet samples of individuals between 150 and 250 mm TL. Samples are organized by impoundment, month, and year. Gizzard shad catch per unit effort (#/minute electrofishing) was calculated based on data from daytime shoreline electrofishing collected concurrently with diet samples.....	99
Table 4.4 Frequency of occurrence of diet items in largemouth bass diet samples of individuals greater than 250 mm TL. Samples are organized by impoundment, month, and year. Gizzard shad catch per unit effort (#/minute electrofishing) was calculated based on data from daytime shoreline electrofishing collected concurrently with diet samples.....	103
Table 4.5 Generalized linear mixed effects model fit by maximum likelihood output to predict FO _{Fish} for largemouth bass across size classes, months, and impoundments with different relative abundances of gizzard shad.....	107
Table 4.6 Linear fixed effect model outputs for the frequency of occurrence (FO) of fish in largemouth bass diets in response to rotenone treatments. Asterisks denote statistical significance at $\alpha = 0.1$	108

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It takes a village to raise an aquatic ecologist and I'm blessed to have been raised by a tightly knit nuclear and extended family that's been my anchor and safe harbor for 27 years. Mom, Dad, Dan, Abby, Grandma Mary, Grandpa Gary, Grandma Gail, Grandpa Bob, Grandma Nancy, Uncle Erik, Uncle John, and all the rest of my wonderful circle of aunts, uncles, and

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I’ll close by thanking my focal study organism, the gizzard shad, for being a source of inspiration for me over the past four and a half years. Anglers overlook you as nothing more than catfish bait. Fisheries biologists joke about your susceptibility to handling stress. Folks underestimate you because they mistake your ostensible vulnerability for weakness, but they fail to recognize that vulnerability and resilience are your hidden strengths. You bounce back rapidly in the face of adversity, you grow and adapt to new habitats quickly, and you recognize that it’s always better to swim with others than to swim alone. Studying you has richly blessed my life, slime and all. *Ad astra per aspera*, my clupeid comrades.

Dedication

I dedicate this dissertation to my late grandfather, Don Renner, who passed away during my time as a PhD student but whose passion for conservation and love of the natural world lives on in me. You are missed, Gramps. It is also dedicated to the late Richard Sanders, who passed away unexpectedly in 2020 after working tirelessly to advance Kansas fisheries for 37 years with the KDWP. I had the blessing of collaborating with Richard for just a few years, but he was always kind and helpful to me whenever I had questions. Finally, I dedicate this dissertation to the healthcare workers, emergency responders, virologists, vaccine developers, lab workers, and public health officials on the frontlines of the COVID-19 pandemic, and the five million lives cut short by SARS-CoV-2 around the world at the time of this writing.

Preface

Chapters 2 - 4 of this dissertation were prepared in submission format and include coauthors. Chapter 2 is formatted for submission to *Freshwater Biology*, Chapter 3 is formatted for submission to *North American Journal of Fisheries Management*, and Chapter 4 is formatted for submission to *Ecology of Freshwater Fish*. Chapters 2, 3 and 4 include appendices that will be included in the submissions as supplemental material. Chapters 2, 3 and 4 will be submitted for publication following the defense.

Chapter 1 - Gizzard shad mediate nutrient cycling in lentic food webs

Introduction

Benthic-feeding fishes can exert direct and indirect effects on multiple trophic levels in lake ecosystems through several mechanisms, such as increasing water column nutrient concentrations (Lamarra 1975; Braband et al. 1990), stimulating phytoplankton abundances (Lamarra 1975; Schaus and Vanni 2000; Yu et al. 2020), increasing turbidity (Meijer et al. 1990; Breukelaar et al. 1994; Tarvainen et al. 2005), and damaging macrophyte beds (Miller and Crowl 2006; Weber and Brown 2009). These effects may stem from bioturbation, which physically disturbs the benthos (Vanni 2002; Matsuzaki et al. 2007; Adámek and Maršálek 2013), or fish excretion, which translocates nutrients from the sediments to the water column and stimulates primary production (Schaus et al. 1997; Schaus et al. 2010), fundamentally shifting lake trophic states. In certain scenarios, nutrient excretion by benthivorous fishes can exceed allochthonous nutrient sources like external loading or other autochthonous sources like nutrient release from lake sediments (Braband et al. 1990; Schindler et al. 2001) and can intensify along gradients of increasing fish biomass (Griffiths 2006) and eutrophication (Drenner et al. 1996, 1998; Vanni et al. 2006). Therefore, eutrophication mitigation measures may be more fruitful if reservoir managers take into consideration not only external nutrient inputs but also the cascading effects of benthic-feeding fishes, such as nutrient translocation from the benthos to the water column and resultant stimulation of the phytoplankton (Schaus et al. 2010).

Reservoir and lake managers often use fish removal as a tool to improve water quality because the approach both removes nutrients sequestered in fish tissues and prevents the bioturbation and nutrient excretion that would have occurred had the benthivorous fish biomass

not been removed from the ecosystem (Schaus et al. 2010). Many biomanipulations have documented significant improvements in water quality and growth of game fishes (Moss et al. 1996; Søndergaard et al. 2008), although in some cases these improvements have been less effective and more short-lived than intended (Persson et al. 1993; Hansson et al. 1998), especially in systems in which the harvested species increases in abundance (e.g., Kim and DeVries 2000; Irwin et al. 2003).

One such benthic-feeding fish that has been the subject of biomanipulations is the gizzard shad (*Dorosoma cepedianum* LeSueur). In reservoirs and lakes across the United States, the gizzard shad, a freshwater clupeid, often dominates fish assemblages in terms of relative abundances and proportions of total fish biomass (Aumen et al. 1992; Stein et al. 1995; Gido and Matthews 2000), especially in eutrophic to hypereutrophic systems (Bachmann et al. 1996; DiCenzo et al. 1996). The gizzard shad is facultatively omnivorous and can attain high biomass in small impoundments through positive feedback mechanisms (Vanni et al. 2005), resulting in negative consequences to economically important game fishes and water quality. They feed primarily on zooplankton and phytoplankton (Cramer and Marzolf 1970; Mundahl 1988; Yako et al. 1996) but can switch to feed extensively on benthic detritus (Mundahl and Wissing 1987; Gido 2001; Higgins et al. 2006), allowing them to maintain high biomass even when zooplankton are scarce. Gizzard shad influence small impoundment food webs both directly by depleting and restructuring zooplankton communities (Aday et al. 2003), and indirectly through sediment bioturbation during detrital foraging that reduces water clarity (Miner and Stein 1996) and re-suspends inorganic nutrients, subsidizing higher trophic levels (Vanni et al. 2005). Consequently, gizzard shad are key benthic-pelagic couplers that drive consumer-mediated nutrient cycling in lentic ecosystems.

As one of the most manipulated forage fishes within reservoirs managed for angling recreation, gizzard shad have been introduced into many small lakes and impoundments to provide forage to bolster the growth of game fishes such as largemouth bass (*Micropterus salmoides*) (DeVries and Stein 1990; Fincel et al. 2017; Michaletz 2020). Once established, gizzard shad populations can dominate fish communities and resist management or eradication efforts through their facultative detritivory, high fecundity, and rapid growth rates that allow age-0 gizzard shad to escape predation by gape-limited piscivores (Vanni et al. 2005). Many of these introductions have yielded negative impacts on bluegills (*Lepomis macrochirus*), but some researchers reported short-term improvements in growth of largemouth bass (Aday et al. 2003; DeVries and Stein 1990). Variation in gizzard shad abundance, reproductive success, and spawn timing can create unpredictable food web dynamics that result in imbalanced fish communities (Vanni et al. 2005).

A survey of KDWP district fisheries biologists revealed that hyperabundant gizzard shad are a growing management concern in Kansas small impoundments (Neely et al. 2016) due to their perceived contributions to decreased growth of littoral sport fishes such as bluegill as well as exacerbated water quality issues and harmful algae blooms. Managers were generally interested in exploring the use of rotenone, a piscicide commonly used for biomanipulation, to selectively kill gizzard shad but were concerned that reduction of gizzard shad could be deleterious to populations of largemouth bass and crappies. Rotenone has been employed to reduce or eliminate gizzard shad populations with variable success (Zeller and Wyatt 1967; Kirk et al. 1986; Kim and DeVries 2000; Catalano et al. 2010; Flammang 2014; Stewart 2015). In an experiment evaluating the efficacy of low concentration rotenone application (7.5 ppb) to reduce or extirpate gizzard shad from a small impoundment, Neely et al. (2018) reported a 99.8%

reduction of shad relative abundance but did not observe changes to bass or bluegill growth rates following manipulation. Observed improvements to age-0 bluegill and largemouth bass relative abundances and body condition were ephemeral for reasons not well understood.

For my dissertation, I sought to characterize the top-down and bottom-up effects of gizzard shad on food web components in Kansas small impoundments as part of a four-year, replicated whole lake manipulation experiment in a before-after-control-impact (BACI) design. A critical first step in understanding how gizzard shad regulate ecosystem processes in lentic food webs is to examine how gizzard shad abundances co-vary with key limnological variables, so in Chapter 2, I tested the influence of gizzard shad on lentic ecosystems by sampling key limnological parameters in 12 small impoundments across a range of gizzard shad relative abundances over four years. I used linear mixed models to test for associations between shad relative abundance and limnological characteristics, including total nitrogen (TN) and total phosphorus (TP), zooplankton densities, and phytoplankton abundances, hypothesizing that gizzard shad relative abundance would be positively associated with phytoplankton abundance and cyanobacterial abundance.

Next, in Chapter 3 I used a BACI design to test whether selectively eradicating gizzard shad from food webs via replicated low-dose rotenone applications precipitated responses in key abiotic and biotic parameters in lentic ecosystems. In 2017, I identified eight Kansas impoundments with gizzard shad that had similar fish communities and morphometry. I collected baseline measurements of all limnological and fish community parameters for two years (2017 and 2018) before rotenone was applied to four treatment lakes in December 2018 and then monitored food web components for two years after treatment (2019 and 2020). I fit linear models to evaluate how these food web components responded to rotenone treatment.

Gizzard shad are thought to be important forage for open-water piscivorous fishes in reservoir ecosystems (Storck 1986), but the extent to which they are preyed upon by gape-limited piscivores in Kansas small impoundments is unclear. There is variation in the prey base of Kansas small impoundment food webs, potentially driven by the presence or absence of gizzard shad. However, the impact the relative abundance of gizzard shad has on largemouth bass, an important predator in these systems, is poorly understood. I sought to describe general patterns in largemouth bass diet composition in 12 Kansas small impoundments and examine how the frequency of occurrence of fish (piscivory) in diets varied across bass size classes, months, and a gradient of gizzard shad relative abundances, hypothesizing that largemouth bass would selectively prey upon gizzard shad compared to other forage fishes. I used regression models to evaluate 1) seasonal variation in the frequency of occurrence of piscivory in largemouth bass diets across months, 2) ontogenetic variation in piscivory (i.e., variation in diets across different length classes of largemouth bass), and 3) variation in relation to gizzard shad relative abundance. I hypothesized that piscivory would increase with largemouth bass length, be highest in August in conjunction with the highest age-0 fish abundances, water temperatures, and metabolic rates, and increase with increasing gizzard shad relative abundance. Finally, I tested whether selective gizzard shad removal via low-dose rotenone manipulation influenced the frequency of occurrence of fish in largemouth bass diets, hypothesizing that piscivory would decline in treatment impoundments following shad removal due to reduced food availability. Results from my dissertation work might shed light on the ecological role of gizzard shad in Kansas small impoundments and advance the management of fisheries and water quality in these novel ecosystems.

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Chapter 2 - Associations between gizzard shad (*Dorosoma cepedianum*) relative abundances and limnological parameters in small impoundments

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Abstract

1. Hyperabundant gizzard shad (*Dorosoma cepedianum*) are thought to contribute to phytoplankton blooms by translocating nutrients from sediments to the water column where they can be readily taken up by phytoplankton and by consuming herbivorous zooplankton that might control phytoplankton.
2. To assess the influence of gizzard shad on lentic ecosystems, we sampled 12 small impoundments across a range of gizzard shad relative abundances over four years. We used linear mixed models to test for associations with limnological characteristics, including total nitrogen (TN) and total phosphorus (TP), zooplankton densities, and phytoplankton abundances.
3. Higher gizzard shad relative abundances were associated with elevated TN and TP during certain months and with a general reduction in zooplankton densities. Regardless of the apparent influence of gizzard shad abundance on top-down (zooplankton) and bottom-up (nutrient) processes that might affect primary producers, we found no relationship between gizzard shad relative abundance and cyanobacterial abundance or total phytoplankton abundance.

4. The lack of a relationship between gizzard shad relative abundances and phytoplankton abundances was thought to be obscured by the discontinuous polymictic mixing regimes of the study impoundments, the rapid turnover of phytoplankton, and other drivers, such as nutrient runoff that was highly variable across our study impoundments.

Keywords: cyanobacteria, harmful algal blooms, trophic decoupling, small impoundments, eutrophication

Introduction

Harmful algae blooms (cyanoHABs) present a growing public health threat to humans and livestock. Climate change and cultural eutrophication are likely to further exacerbate these blooms as they become more frequent, intense, and widespread (Paerl et al. 2011; O’Neil et al. 2012; Lürling et al. 2018). Recent advancements in regional and continental-scale satellite monitoring of cyanobacteria have highlighted the widespread prevalence of annual cyanobacterial blooms, although few studies have quantified long-term trends in the frequency or magnitude of cyanobacterial blooms at these scales across the United States (Coffer et al. 2021). In the United States, cyanoHABs adversely affect environmental, plant, and animal health, generating over two billion dollars in damage to recreational, drinking, and agricultural water resources annually (Codd et al. 2005; Backer 2002; Dodds et al. 2009).

Feeding and excretion by fishes play an important and frequently overlooked role in nutrient cycling (Sharitt et al. 2021), and several studies have analyzed the role of benthic-feeding fishes in reducing water quality and facilitating cyanobacterial blooms. One such benthivorous fish that can precipitate cascading ecosystem effects is the gizzard shad, *Dorosoma cepedianum*. Gizzard shad are one of the most ubiquitous forage fishes within reservoirs managed for recreational angling across the United States. Through positive feedback

mechanisms, gizzard shad can attain high biomass in small impoundments (Vanni et al. 2005), with potentially negative consequences to economically important fishes and water quality. Gizzard shad feed primarily on zooplankton and phytoplankton (Cramer and Marzolf 1970; Mundahl 1988; Yako et al. 1996) but can switch to feed extensively on benthic detritus (Mundahl and Wissing 1987; Gido 2001; Higgins et al. 2006; Smoot and Findlay 2010), which allows populations to maintain high biomass even when zooplankton are scarce. Gizzard shad can affect primary productivity (and therefore water quality) both directly via phytoplankton grazing and indirectly through zooplankton grazing and nutrient recycling (Watson et al. 2003). They directly influence small impoundment food webs as prey for higher trophic levels (Miner and Stein 1996) and via competition for zooplankton with zooplanktivores (Aday et al. 2003). Zooplanktivory by gizzard shad can occur at various life stages and influence the structure of zooplankton communities (Dettmers and Stein 1996). Therefore, at higher gizzard shad abundances, we would expect to see a zooplankton community dominated by smaller taxa such as copepods and rotifers rather than larger cladocerans that are more vulnerable to planktivory (Brooks and Dodson 1965; Hall et al. 1976). Moreover, as facultative detritivores, gizzard shad can remain abundant when zooplankton abundance is low; such trophic decoupling may allow gizzard shad to structure reservoir food webs by overconsumption of zooplankton, which in turn may constrain recruitment of co-occurring fishes with zooplanktivorous larvae (Dettmers and Stein 1996).

Benthic feeding and resulting excretion by gizzard shad translocate nutrients (nitrogen and phosphorous; N and P) from benthic habitats into the water column in dissolved inorganic forms (Schaus et al. 1997; Schaus and Vanni 2000; Vanni and Headworth 2004). The relative proportion of cyanobacteria in epilimnetic phytoplankton communities is dependent on the ratio

of total nitrogen to total phosphorus available, and cyanoHABs tend to occur when epilimnetic N:P ratios fall below roughly 29:1 (Smith 1983; Harris et al. 2014). Thus, excretion by gizzard shad may shift water column N:P ratios towards conditions favorable to cyanoHAB development. Schaus et al. (1997) hypothesized that the low N:P ratio of gizzard shad excretion [mean molar N:P = 16.75 (± 0.89 SE)] may shift phytoplankton communities in favor of cyanobacteria but did not directly quantify community composition to assess this association. Sharitt et al. (2021) quantified the proportion of phytoplankton P demand met by gizzard shad excretion in a eutrophic reservoir for 15 years and found that shad supported 7-27% of demand, and that interannual variation in young-of-year gizzard shad biomass significantly altered nutrient supply. Filter-feeding by gizzard shad can modify phytoplankton communities by suppressing populations of large organisms like *Ceratium* spp. and enhancing the biomass of phytoplankton and bacteria between 2 and 4 μm diam (Drenner et al. 1984). Bloom-forming cyanobacterial species such as *Microcystis aeruginosa* and *Dolichospermum spiroides* are ingested by gizzard shad in some eutrophic lakes, although the extent to which these cyanobacteria are digested and assimilated relative to other phytoplankton is variable and was the subject of past debate (Kutkuhn 1957; Velásquez 1939). As such, gizzard shad can influence nutrient dynamics and food web properties through a wide variety of trophic interactions at multiple trophic levels. However, gizzard shad population dynamics and resultant effects on ecosystem properties can vary with latitude (Kim and DeVries 2000; Wuellner et al. 2008; Fincel et al. 2013) and impoundment size (Michaletz 2017), and potentially lake mixing regime, suggesting that these trophic interactions do not necessarily vary predictably across regions.

In addition, to our knowledge most of the past work on gizzard shad trophic interactions has occurred in dimictic and warm monomictic ecosystems. Many Kansas small impoundments

are discontinuous polymictic lakes (Harris, unpublished data), meaning that they do not develop a full seasonal stratification but rather experience alternating periods of mixing and stratification lasting for periods of several days to weeks, with irregular interruption by mixing events driven strongly by meteorological conditions (Lewis 1983; Woolway et al. 2017). Polymictic lakes are thought to be particularly susceptible to mixing regime shifts due to climate change (Kirillin 2010; Woolway et al. 2021). A thorough understanding of mechanisms contributing to cyanoHABs in the context of stressor gradients and seasonal effects (Richardson et al. 2019) is needed to anticipate and mitigate against adverse effects of climate change on water quality and recreational angling in Kansas small impoundments, prompting this study.

Our objective was to test for associations between gizzard shad relative abundances and limnological parameters including zooplankton density, phytoplankton abundance, and N and P concentrations, in Kansas, USA, small impoundments, where more than 100 cyanoHABs have occurred over the past decade (Kansas Dept. of Health and Environment 2021). We hypothesized two mechanisms by which higher gizzard shad relative abundances would lead to increased algal biomass. First, TN and TP would both be positively associated with gizzard shad relative abundance, but P would increase at a faster rate, resulting in a decline in N:P with increasing shad relative abundance. These effects on nutrient availability should stimulate algal production, particularly of N-fixing cyanobacteria, because of reduced N:P. Second, zooplanktivory should increase with shad abundance, resulting in both a shift towards smaller taxa and reduced rates of herbivory leading to a further increase in algal biomass.

Study Area and Experimental Design

The project was conducted across 12 small, eutrophic to hypereutrophic impoundments (Fig 1) ranging in surface area from 7 to 166 ha (Appendix A.1) and managed for public angling

by the Kansas Department of Wildlife and Parks. All 12 impoundments contained typical Midwestern USA warmwater fish communities dominated by largemouth bass (*Micropterus salmoides*) and bluegill (*Lepomis macrochirus*). Green sunfish (*Lepomis cyanellus*), redear sunfish (*Lepomis microlophus*), white crappie (*Pomoxis annularis*), black crappie (*Pomoxis nigromaculatus*), channel catfish (*Ictalurus punctatus*), flathead catfish (*Pylodictis olivaris*), and common carp (*Cyprinus carpio*) were also common in most impoundments. Gizzard shad were present in eight of the impoundments at the start of the study (Appendix A.3). Two impoundments without gizzard shad (Cowley and Douglas) experienced unplanned shad introductions in 2018, while two impoundments with shad (Pottawatomie #1 and Shawnee) experienced successful selective gizzard shad eradications via low-dose rotenone in 2018. Two other impoundments with shad (Neosho and Washington) experienced fleeting reductions in gizzard shad relative abundance following unsuccessful rotenone manipulations in 2018. Thus, abundance of gizzard shad was highly variable both within and among small impoundments over the course of this study (Appendix A.3).

Field Methods for Assessment of Limnological Parameters

Limnological parameters were sampled at three stations in each impoundment in June, August, and October from 2017 to 2020. Stations were positioned in limnetic habitats in 1) riverine zones near an impoundment's inflow, 2) mid-lake transition zones, and 3) lacustrine zones near an impoundment's dam. Samples collected at each station were used to quantify water chemistry (Total N and P) and zooplankton abundance and composition. Phytoplankton abundance and composition were quantified at two stations per impoundment: riverine and lacustrine. Visual water clarity (vertical visual extinction) was measured at each station with a Secchi disc (Steel and Neuhausser 2002).

Water Nutrient Analysis

Replicate surface water grab samples (25 mL) were analyzed for TN and TP from a depth of 0.5 m from each sampling station. Samples were immediately placed on ice in coolers and then frozen until analysis. Samples were thawed in a refrigerator overnight prior to analysis. A 1 to 10 mL water sample was digested with potassium persulfate in an autoclave and then analyzed using an Alpkem rapid flow analyzer (Alpkem Corporation, Clackamas, Oregon, USA) for TN and TP (Hosomi and Sudu 1986; Nelson 1987) by the Kansas State University Soil Testing Lab.

Phytoplankton Collection and Enumeration

Phytoplankton were collected by taking one surface water grab sample in 25 mL vials at a depth of 0.5 m at each of two limnological stations (riverine and lacustrine) per impoundment. Samples were immediately preserved with 25 drops (approx. 1 mL) of Lugol's iodine and stored in opaque glass vials on ice in a dark cooler to halt photosynthesis and degradation before being stored in a dark setting. We settled 5 mL subsamples for a minimum of 48h in 10 mL polycarbonate tubes affixed with cover slips before placing on an inverted microscope. Images of 50 fields of view at 400X magnification were captured for each sample with a digital camera, and phytoplankton were counted and measured from images. We classified phytoplankton into broad taxonomic categories (green, cyanobacteria, diatom, dinoflagellate). Phytoplankton were placed into size bins by superimposing a 10 x 10 μm grid onto images and counting the number of grids occupied by each "natural unit" or algal cluster as an approximation of the surface area of each unit (Fig. 2). We report surface area per 50 fields of view of the different algal groups (number of grids occupied per number of fields counted) for each sample as a metric of

phytoplankton abundance (Moheimani et al. 2013; Guillard and Sieracki 2005). We considered this an index of relative biomass (more specifically photosynthetic area).

Zooplankton Collection and Enumeration

Pelagic zooplankton communities were sampled via three replicate 3-m tows of a Wisconsin plankton net (Wildco #623-1130, 80- μ m mesh, 12.7 cm diam, 6.35 cm radius; Yulee, FL, USA) per station. Vertical tows were performed at sampling stations greater than 3.5 m deep and horizontal tows were performed at stations less than 3.5 m deep with approximate tow speeds of 1 m/s. Net contents from three replicate tows were washed and pooled into a single Nalgene bottle and immediately fixed with 4% buffered sucrose formalin (Haney and Hall 1973) and transported on ice. Samples were diluted to a known volume. We used a Hensen-Stempel pipette (Wildco #3-1806-D52, Yulee, FL, USA) to transfer 2 to 10 mL of sample at a time to a Ward counting wheel depending on sample densities (Wildco #3-1806-D52, Yulee, FL, USA). All zooplankton counted were classified under one of seven taxonomic categories: Cladocera, Calanoida, Cyclopoida, Harpacticoida, Rotifera, Nauplii, and Culicomorpha (Chaoboridae). Mean zooplankton densities were calculated and reported as the average counts of each group per L (Mack et al. 2012; deBernardi 1984; Beaver et al. 2018).

Quantifying Gizzard Shad Relative Abundances

Shad relative abundances for each impoundment were based on boat electrofishing on the same day as collection of limnological data. Each lake was sampled with daytime pulsed DC electrofishing (60 Hz, 25% duty cycle, ETS Electrofishing). Shad relative abundances were reported as catch per unit effort (CPUE), the count of shad per minute electrofishing (Appendix 3). Electrofishing was typically conducted for shock times of 40-70 minutes total per sampling event (mean = 46 ± 14 SD minutes electrofishing).

Data Analysis

We fit a suite of linear mixed effects models to evaluate relationships between gizzard shad relative abundances and our dependent variables of interest using the function “lmer” of the package *lme4* (Bates et al. 2015). Gizzard shad relative abundance, month, and the interaction between shad relative abundance and month were treated as fixed effects, and impoundment was treated as a random effect. For each model with a significant interaction, we ran post hoc models for each month to interpret effects across the three months. To account for heterogeneity of variances across samples, gizzard shad relative abundances were either natural log ($x + 0.01$)- or natural log ($x + 0.05$)-transformed; TN and TP were natural log ($x + 0.01$)-transformed; N:P ratios were square root-transformed; phytoplankton abundances and Secchi depths were natural log-transformed; cladoceran, calanoid copepod, cyclopoid copepod, copepod nauplii, and rotifer densities were square root-transformed; and total zooplankton densities were natural log ($x + 1$)-transformed. For all models we calculated 95% confidence intervals for each fitted parameter (function “confint”) with the *stats* package (R Core Team 2021). We assessed the importance of random effects by comparing marginal R^2 coefficients of determination (variance explained by fixed effects only) to conditional R^2 coefficients of determination (variance explained by both fixed and random effects) for each model (function “r.squaredGLMM”) using the *MuMIN* package (Bartón 2020). All analyses were performed in R version 4.1.0 (R Core Team 2021).

Results

As predicted, there was a significant interaction between shad relative abundance and month for TN ($p < 0.001$) (Figure 2.3, Table 2.1), with a positive relationship between TN and shad abundance in August ($p = 0.003$; Table 2.2) but a negative relationship in October ($p = 0.012$; Table 2.2). There also was a significant interaction between shad relative abundance

and month for TP ($p = 0.011$) (Figure 2.3; Table 2.1). In this case, reduced models indicated that the association was stronger in August ($p < 0.001$; Table 2.2) and June ($p < 0.001$) than in October ($p = 0.945$). There was a significant interaction between shad relative abundance and month ($p = 0.033$) for N:P ratios, with no significant changes across shad abundances in June or August samples but significant ($p = 0.008$) declines in N:P in October with increasing shad relative abundance (Figure 2.3). For water clarity, there was a significant negative relationship between gizzard shad relative abundance and Secchi depth ($p = 0.047$), but no significant interaction with month ($p = 0.189$; data not shown).

Contrary to predictions, there was no relationship between gizzard shad relative abundance and total phytoplankton abundance ($p = 0.851$) or cyanobacteria abundance ($p = 0.635$) (Table 2.1). Shad relative abundance was not related to green algal abundance ($p = 0.267$) or diatom abundance (0.747), and interactions between shad abundance and month were not significant for any phytoplankton taxonomic group (Figure 2.4).

Total zooplankton densities decreased with increasing gizzard shad abundance ($p < 0.001$) as predicted, and this pattern was consistent across months. Cladoceran ($p = 0.013$), calanoid copepod ($p < 0.001$), cyclopoid copepod ($p < 0.001$) and rotifer ($p < 0.001$) densities were negatively associated with increasing gizzard shad abundance (Figure 2.5), but the interaction between shad abundance and month was not significant (Table 2.1).

The random effect of impoundment explained the majority of the variation in our statistical models, particularly for nutrients. Conditional R^2 values ranged from 0.201 (N:P ratios) to 0.538 (TP), suggesting the effects of gizzard shad relative abundance were minor relative to other factors that vary spatially across these systems.

Discussion

We hypothesized gizzard shad relative abundance would be associated with algal abundance or cyanobacterial abundance but the data failed to support that hypothesis. However, the positive relationship with TN and TP concentrations indicates the potential for bottom-up shad effects on primary producers, whereas decreased zooplankton densities with higher shad abundances indicates the potential for top-down shad effects through reduced algal grazing by zooplankton. In addition, we observed lower N:P ratios at higher gizzard shad relative abundances (Figure 2.3), a pattern consistent with that described by Schaus et al. (1997). These observed bidirectional effects on zooplankton density and nutrient concentrations were consistent with the literature (Stein et al. 1995; Vanni et al. 2005; Michaletz 2017). Phytoplankton communities generally display much faster turnover rates and responses to changes in biotic and abiotic variables due to their shorter generation times relative to zooplankton or fish communities (Harris et al. 1980; Carpenter et al. 1987; Hansen and Carey 2015), so any intermittent effect of gizzard shad relative abundance may have been too weak or fleeting for us to detect by sampling only three snapshots in time per growing season (Sharitt et al. 2021). Moreover, the ratio of marginal to conditional R^2 in our mixed models was large, indicating variation among impoundments was more pronounced than the fixed effects of gizzard shad relative abundance and month. Kansas small impoundments are highly variable across the landscape, depending on watershed land use, size, mixing regime, morphology, macrophyte cover and biotic communities (Neely et al. 2020). Thus, identifying the role of gizzard shad across this range of conditions is confounded by the high signal to noise ratio of measured variables.

Evidence for strong effects of gizzard shad on phytoplankton abundance from the literature is mixed, and responses of limnological variables to shad abundance or biomass are

highly context dependent. Several studies document the relationship between increasing gizzard shad abundance and lake trophic state, although some studies model gizzard shad population characteristics as drivers of limnological properties (Michaletz 2017), while others treat hyperabundant shad as a symptom of eutrophic conditions, modeling shad abundance as a response variable (Bachmann et al. 1996; DiCenzo et al. 1996; Hoxmeier and DeVries 1998; Allen et al. 2000; Bremigan and Stein 2001; Michaletz 1999; 2017). Watson et al. (2003) tested the ability of gizzard shad to influence primary productivity in a series of mesocosm and pond experiments and observed that shad reduced zooplankton density and indirectly enhanced chlorophyll-a and primary productivity. However, they found no evidence for differences in zooplankton and phytoplankton communities between ponds with and without gizzard shad, speculating that high densities of zooplanktivorous larval bluegill were regulating phytoplankton abundance through top-down effects. Schaus and Vanni (2000) manipulated sediment access in mesocosm experiments and found that gizzard shad stimulated phytoplankton only when they could feed directly on lake sediments and not when benthic feeding was prevented by a mesh netting; in our study impoundments, shad might have sufficient planktonic forage to reduce the necessity for benthic feeding. It is also possible that land use in catchments above these impoundments subsidizes nutrients (Lenat and Crawford 1994; Fraterrigo and Downing 2008; Lee et al. 2009; Carney 2009), confounding the effect of gizzard shad on phytoplankton dynamics (percent agriculture in the watershed above impoundments was correlated with TN; $r^2 = 0.307$). This scenario is illustrated by one of our study impoundments in western Kansas, which does not have gizzard shad but lies within a watershed impacted heavily by intensive row-crop agriculture (71% of watershed land cover; Appendix A.1) and has consistently high

phytoplankton abundances (Appendix A.2) and several cyanoHAB events (D. Spalsbury, Kansas Department of Wildlife and Parks, personal communication).

One caveat with our study was the difficulty in quantifying gizzard shad relative abundances with electrofishing. Gizzard shad are schooling fishes that occupy open-water habitat, so accurately quantifying their relative abundances without the use of hydroacoustic surveys or trawling is difficult (Johnson et al. 2019). However, studies employing hydroacoustic surveys have also ground-truthed their data with shoreline electrofishing (Vanni et al. 2006). Although Van Den Avyle et al. (1995) noted that gill netting does not index age-1 shad well in slow-growing populations, we did note a concordance between estimated gizzard shad CPUE from our daytime shoreline electrofishing and gizzard shad CPUE from the Kansas Department of Wildlife and Parks long-term fall gill netting surveys ($r = 0.604$; Appendix A.3). Furthermore, our models were grounded with some impoundments with no gizzard shad, thus anchoring our relationships.

Hyperabundant gizzard shad may be a convenient scapegoat for growing water quality issues, but our findings suggest that other factors such as variation in mixing regimes or watershed conditions might play a greater role than gizzard shad in influencing cyanobacterial blooms and governing limnological processes in Kansas small impoundments. While gizzard shad might weakly stimulate eutrophication issues in small impoundments, watershed-level factors regulating allochthonous nutrient inputs likely play a much larger role in driving nutrient patterns. That said, our observations that gizzard shad influenced nutrients and zooplankton suggest the potential to contribute to harmful algal blooms under some conditions. Experimental manipulations of gizzard shad populations might help elucidate the role of gizzard shad within impoundments with different biotic and abiotic properties.

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Tables

Table 2.1. Linear mixed effect model type III ANOVA outputs. Gizzard shad were sampled with daytime shoreline electrofishing and quantified as CPUE (catch per unit effort, # shad/minute electrofishing). Shad relative abundances were either natural log ($x + 0.01$)- or natural log ($x + 0.05$)-transformed. All models followed the format: $\text{lmer}(\log(\text{TN}) \sim \log(\text{shadpermin} + 0.05) * \text{month} * (1|\text{lakecode}))$. Asterisks denote statistical significance at $\alpha = 0.05$.

Model	Fixed Effect or Interaction Term	R^2 Marginal; R^2 Conditional	Denominator df	F value	p-value
Total Nitrogen	Shad Abundance	0.073; 0.487	385.99	2.165	0.142
	Month		375.86	24.587	< 0.001*
	Shad Abundance x Month		376.07	8.255	< 0.001*
Total Phosphorus	Shad Abundance	0.029; 0.534	381.07	0.384	0.536
	Month		375.74	0.301	0.740
	Shad Abundance x Month		375.79	4.580	0.011*
N: P Ratio	Shad Abundance	0.067; 0.201	318.05	1.098	0.296
	Month		377.37	7.637	< 0.001*
	Shad Abundance x Month		377.77	3.435	0.033*
Secchi Depth	Shad Abundance	0.012; 0.716	385.71	3.972	0.047*
	Month		379.24	1.594	0.205
	Shad Abundance x Month		379.29	1.675	0.189
Total Phytoplankton Abundance	Shad Abundance	0.047; 0.033	58.95	0.036	0.851
	Month		120.03	1.197	0.306
	Shad Abundance x Month		121.28	1.503	0.227
Green Algae Abundance	Shad Abundance	0.041; 0.045	44.71	1.266	0.267
	Month		117.53	1.619	0.203
	Shad Abundance x Month		119.18	1.678	0.191
Cyanobacteria Abundance	Shad Abundance	0.048; 0.048	128.00	0.226	0.635
	Month		128.00	0.953	0.388
	Shad Abundance x Month		128.00	1.593	0.207
Diatom Abundance	Shad Abundance	0.041; 0.120	85.34	0.105	0.747

	Month		118.71	0.407	0.666
	Shad Abundance x Month		119.38	2.169	0.119
Total Zooplankton Density	Shad Abundance	0.053; 0.111	194.39	13.266	< 0.001*
	Month		379.17	0.184	0.832
	Shad Abundance x Month		380.06	1.488	0.227
Cladoceran Density	Shad Density	0.062; 0.172	308.26	6.240	0.013*
	Month		374.76	0.339	0.713
	Shad Abundance x Month		376.62	2.520	0.082
Rotifer Density	Shad Density	0.064; 0.155	254.12	12.031	< 0.001*
	Month		378.14	0.716	0.489
	Shad Abundance x Month		378.74	2.639	0.073
Calanoid Density	Shad Density	0.023; 0.045	110.26	5.226	0.024*
	Month		380.84	0.423	0.656
	Shad Abundance x Month		382.44	0.718	0.488
Cyclopoid Density	Shad Density	0.039; 0.090	171.66	10.955	0.001*
	Month		379.12	0.753	0.471
	Shad Abundance x Month		380.18	1.312	0.270
Copepod Nauplii Density	Shad Density	0.041; 0.045	89.42	0.255	0.615
	Month		381.06	2.029	0.133
	Shad Abundance x Month		382.50	0.461	0.361

Table 2.2. Linear mixed effect model type III ANOVA outputs for reduced models disaggregated by month. Gizzard shad were sampled with daytime shoreline electrofishing and quantified as CPUE (catch per unit effort, # shad/minute electrofishing). Shad relative abundances were $\log(x + 0.01)$ transformed. All models followed the format: $\text{lmer}(\log(\text{TN} + 0.01) \sim \log(\text{shadpermin} + 0.05) * (1|\text{lakecode}))$. Asterisks denote statistical significance at $\alpha = 0.05$.

Model	Month	<i>R² Marginal; R² Conditional</i>	<i>df</i>	Estimate	Std. Error	95% CI	t value	p-value
Total Nitrogen	June	0.003; 0.466	123.76	0.028	0.045	-0.060; 0.117	0.622	0.535
	August	0.069; 0.417	131.44	0.129	0.043	0.044; 0.213	2.999	0.003*
	October	0.039; 0.545	128.38	-0.076	0.030	-0.135; -0.018	-2.556	0.012*
Total Phosphorus	June	0.055; 0.741	120.80	-0.155	0.040	-0.233; -0.073	-3.828	< 0.001*
	August	0.072; 0.643	131.08	0.176	0.046	0.085; 0.268	3.796	< 0.001*
	October	< 0.001; 0.375	128.25	-0.004	0.063	-0.127; 0.121	-0.070	0.945
N: P Ratio	June	0.030; 0.322	116.88	0.365	0.193	-0.051; 0.751	1.886	0.062
	August	0.014; 0.255	123.92	-0.231	0.181	-0.597; 0.124	-1.271	0.206
	October	0.060; 0.097	68.43	-0.591	0.216	-1.069; -0.137	-2.729	0.008*

Figures

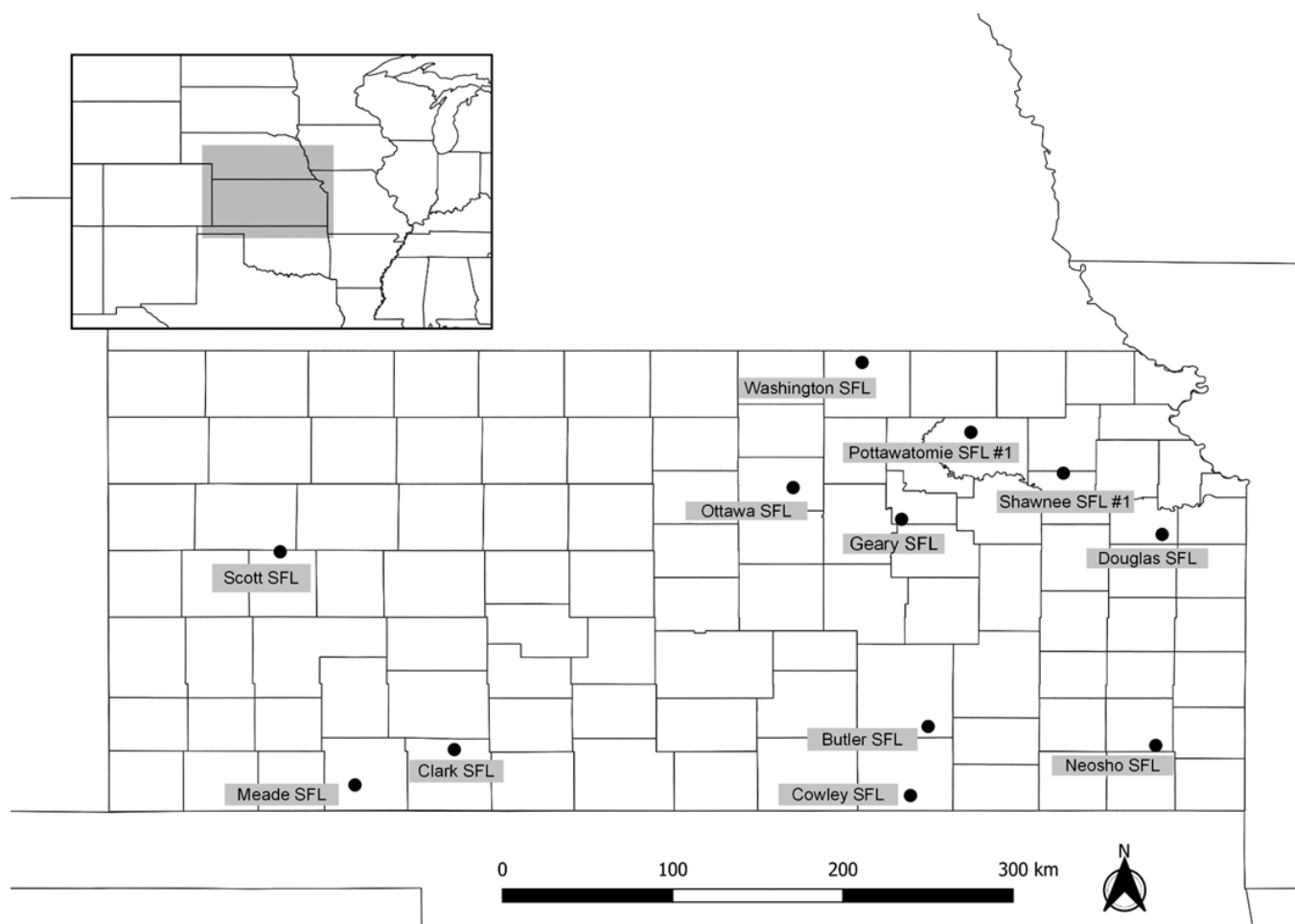


Figure 2.1 Map of 12 study impoundments in Kansas, USA (county borders are shown for reference). SFL = State Fishing Lake. All impoundments are managed for angling recreation by the Kansas Department of Wildlife & Parks (KDWP).

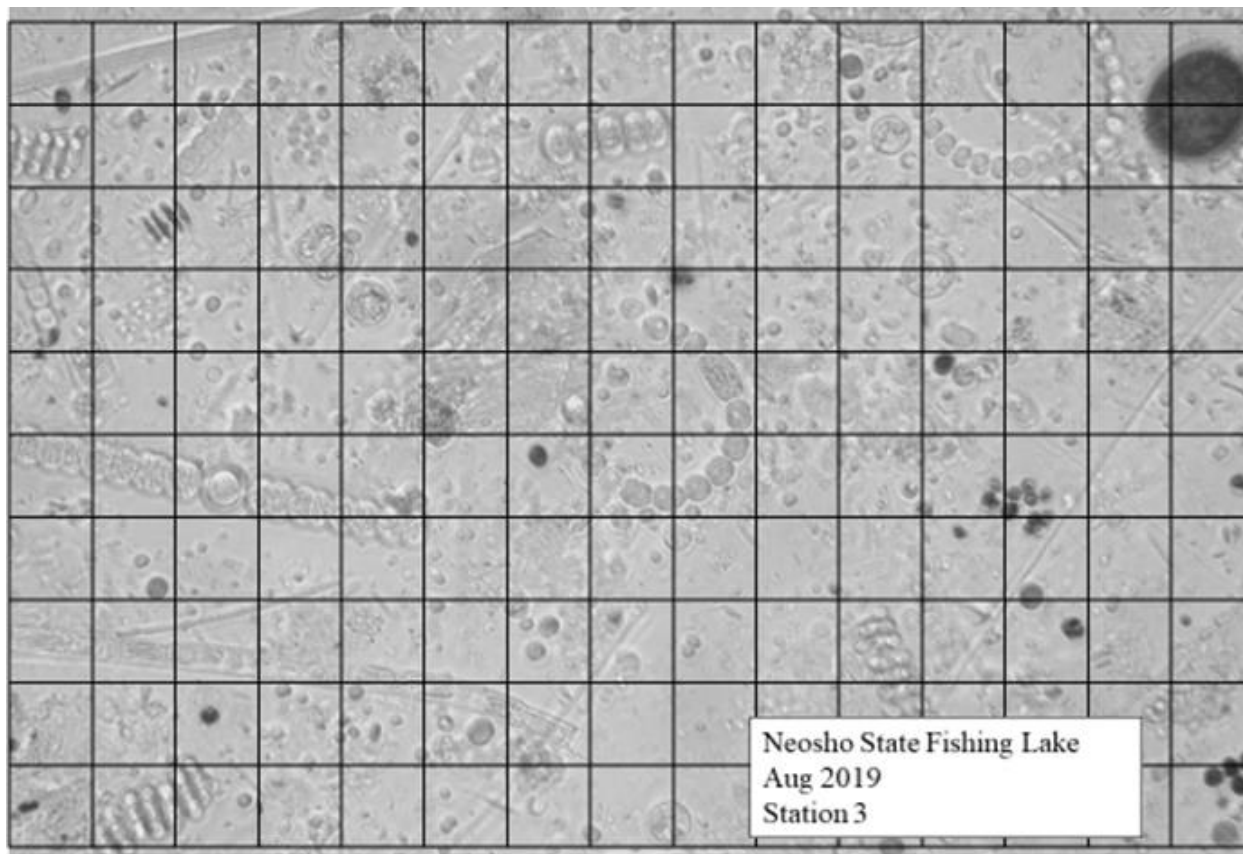


Figure 2.2 Photomicrograph of a phytoplankton sample under 400X magnification overlaid with a 10 x 10 μm counting grid. An index of relative biomass was estimated by the number of grids occupied by each natural unit or algal cluster.

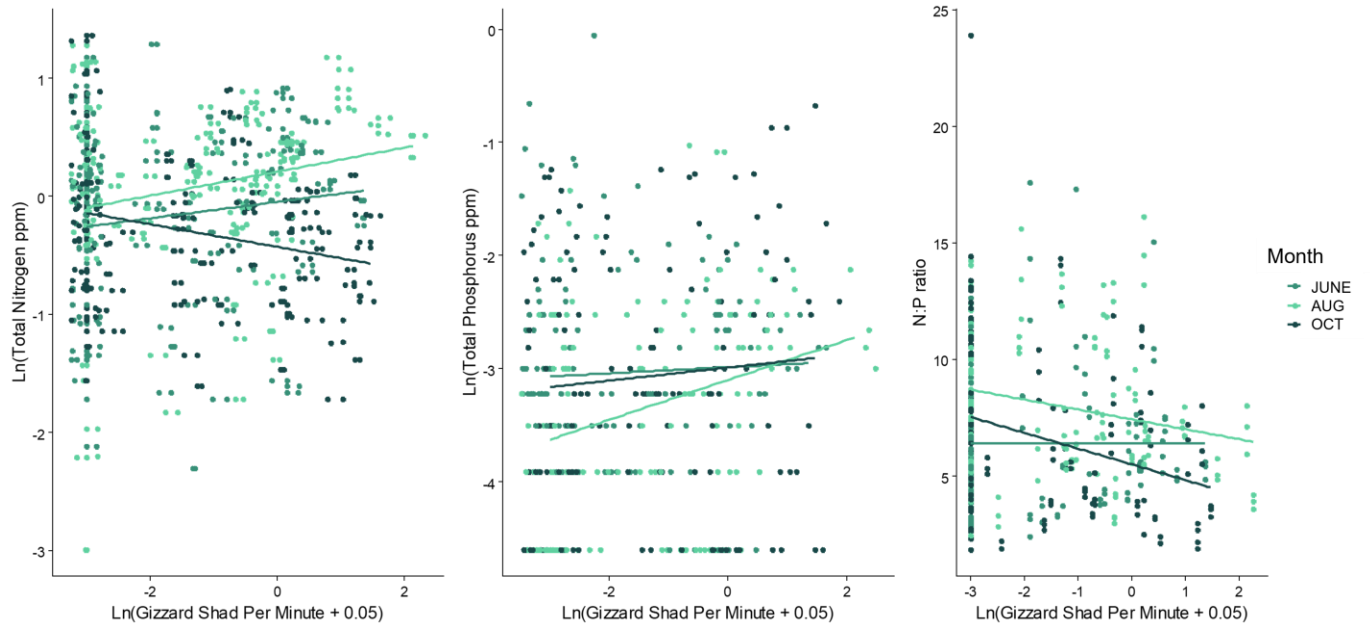


Figure 2.3 Plot of gizzard shad CPUE versus TN, TP, and molar N:P ratios. Gizzard shad were sampled with daytime shoreline electrofishing and quantified as CPUE (catch per unit effort, # shad/minute electrofishing). Gizzard shad densities were natural log ($x + 0.05$) transformed and TN and TP concentrations were natural log ($x + 0.01$) transformed. N:P ratios were square root transformed. Colors indicate month sampled. TN and TP were quantified using surface water grab samples collected concurrently with electrofishing. Trend lines were included if effects were significant at $\alpha = 0.05$.

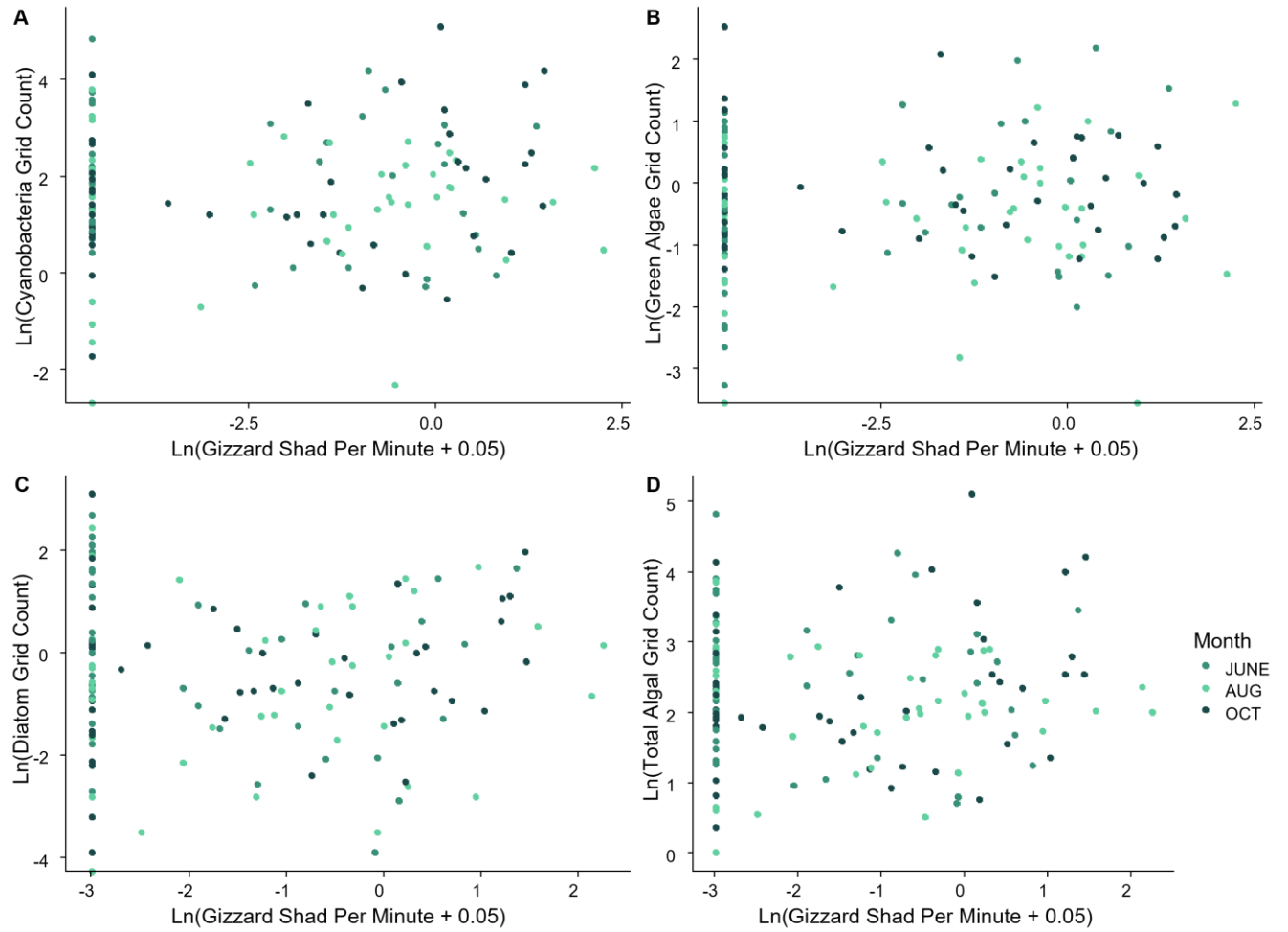


Figure 2.4 Gizzard Shad CPUE versus cyanobacteria abundance, green algae abundance, diatom abundance, and total phytoplankton abundance. Phytoplankton grab samples were collected from two stations (riverine and lacustrine) in June, August, and October 2017-2020. Gizzard shad were sampled with daytime shoreline electrofishing and quantified as CPUE (catch per unit effort, # shad/minute electrofishing). Gizzard shad densities were natural log ($x + 0.05$) transformed. Colors indicate month sampled. Trend lines were included if effects were significant at $\alpha = 0.05$.

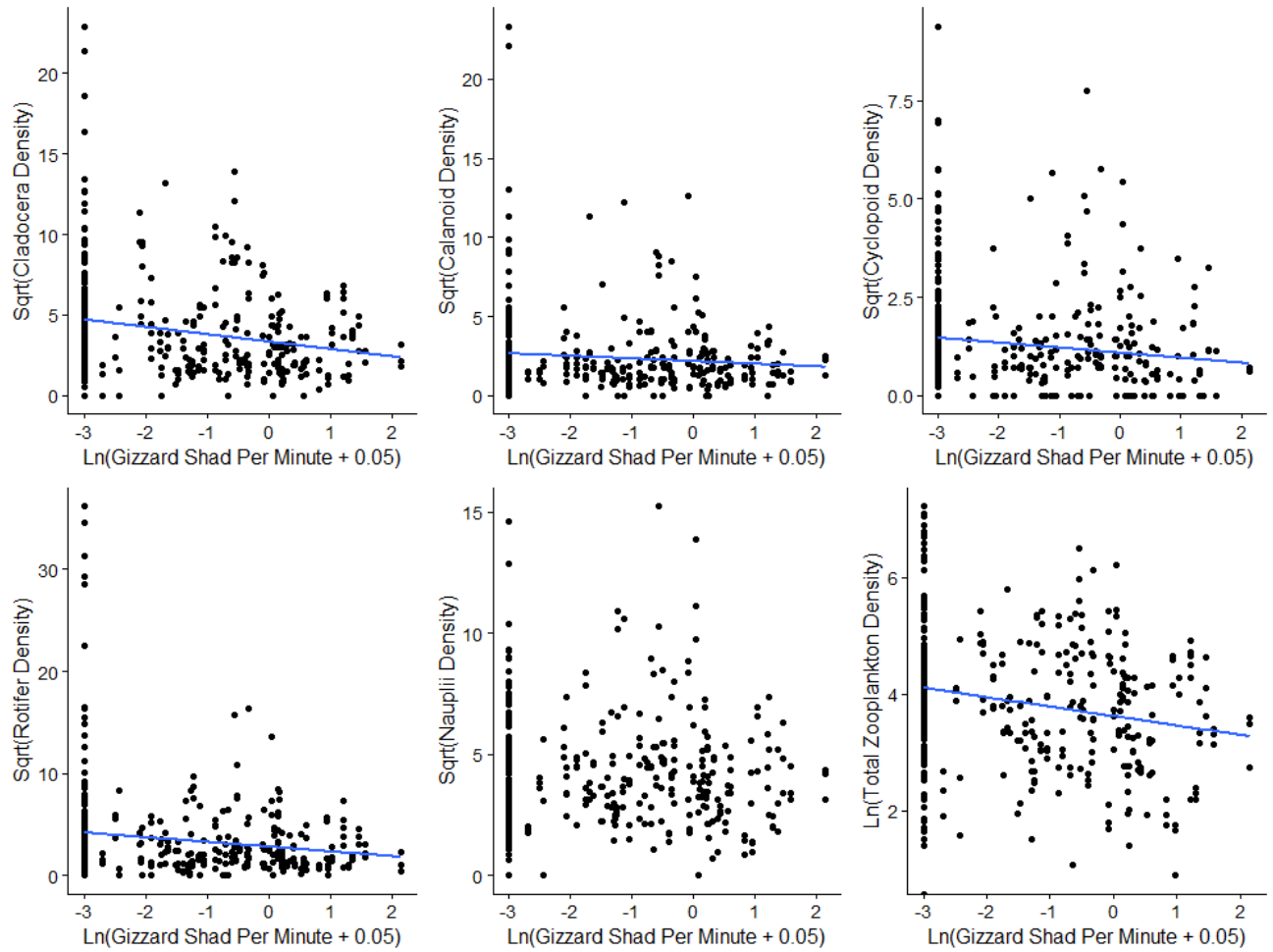


Figure 2.5 Gizzard Shad CPUE versus cladoceran density, calanoid copepod density, cyclopoid copepod density, rotifer density, nauplii density, and total zooplankton density. Three replicate 3-m zooplankton tows were collected at each of three stations (riverine, transition, lacustrine) in June, August, and October 2017-2020. Gizzard shad were sampled with daytime shoreline electrofishing and quantified as CPUE (catch per unit effort, # shad/minute electrofishing). Gizzard shad densities were natural log ($x + 0.05$) transformed and zooplankton densities were natural log transformed. Trend lines were included if effects were significant at $\alpha = 0.05$.

Chapter 3 - Characterizing changes to lentic food webs in response to low-dose rotenone manipulations in Kansas small impoundments

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Abstract

Gizzard Shad (*Dorosoma cepedianum*), a facultatively detritivorous fish, can attain high biomass in small impoundments through positive feedback mechanisms, potentially resulting in negative consequences to economically important fishes and water quality. Their influence on small impoundment food webs can occur directly via competition for zooplankton and indirectly through bioturbation of sediments during detrital foraging that reduces water clarity and re-suspends inorganic nutrients, triggering phytoplankton blooms. Low-dose rotenone applications that selectively kill Gizzard Shad from small impoundments might improve water quality and growth rates and relative abundances of co-occurring fishes, but responses are often limited and ephemeral (1-2 years post treatment) for reasons not fully understood. In 2017, we identified eight Kansas impoundments that had similar fish communities and morphometry. We collected baseline measurements of all limnological and fish community parameters for two years (2017 and 2018) before applying rotenone to four treatment lakes in December 2018. Although we found a slight decline in total nitrogen (TN) and total phosphorus (TP) in treatment impoundments relative to control impoundments following shad removal, overall, Gizzard Shad removal had little impact on limnological parameters or fish populations in small Kansas impoundments.

Introduction

Theoretical and empirical research underscore the importance of species interactions in regulating food web structure and ecosystem function. Gizzard Shad (*Dorosoma cepedianum*), a facultatively detritivorous fish (Vanni et al. 2005), can attain high biomass in small impoundments through positive feedback mechanisms, resulting in negative consequences to economically important fishes and water quality (Stein et al. 1995). They influence food webs directly via competition for zooplankton with zooplanktivores and indirectly through bioturbation of sediments during detrital foraging. This foraging reduces water clarity and re-suspends inorganic nutrients, triggering phytoplankton blooms and subsidizing higher trophic levels (Schaus et al. 1997; Schaus and Vanni 2000; Aday et al. 2003). Once established, Gizzard Shad populations can dominate fish communities and resist management or eradication efforts through their facultative detritivory, high fecundity, and rapid growth rates that allow age-0 Gizzard Shad to escape predation by gape-limited piscivores (Vanni et al. 2005).

Gizzard Shad are one of the most commonly manipulated forage fishes within reservoirs managed for recreational angling. They have been introduced into many small lakes and impoundments to provide forage to improve the growth of recreationally important fishes such as Largemouth Bass (*Micropterus salmoides*) (DeVries and Stein 1990). Many of these introductions have yielded negative impacts on Bluegill (*Lepomis macrochirus*), but some researchers reported short-term improvements in growth of Largemouth Bass following Gizzard Shad introductions (Aday et al. 2003; DeVries and Stein 1990). Variation in Gizzard Shad abundance, reproductive success, and spawn timing can create unpredictable food web dynamics resulting in imbalanced fish communities (Vanni et al. 2005). A 2016 survey of Kansas Department of Wildlife and Parks (KDWP) district fisheries biologists revealed that hyperabundant Gizzard Shad are a growing management concern in small impoundments (Neely

et al. 2016). Specific concerns cited by managers included decreased growth of littoral sport fishes such as Bluegill as well as exacerbated water quality issues and cyanobacteria blooms. Managers were generally interested in exploring the use of low-dose rotenone to selectively kill Gizzard Shad but were concerned that their removal could be detrimental to populations of Largemouth Bass and crappies.

Predicting how dominant species influence ecosystem processes presents challenges in complex natural systems. A fruitful approach to understanding the importance of key interactions has been to conduct replicated experiments removing a component of the system of interest (Mittelbach 2012). Small impoundments provide excellent systems for such investigations because they are well replicated on the landscape (Downing et al. 2006) and can be experimentally manipulated. The piscicide rotenone ($C_{23}H_{22}O_6$) at low doses (5.5-7.5 ppb) can selectively reduce or eliminate gizzard shad populations with variable success (Zeller and Wyatt 1967; Kirk et al. 1986; Kim and DeVries 2000; Catalano et al. 2010; Flammang 2014). Thus, replicated, large-scale, whole-lake experimental removals of Gizzard Shad are feasible. This allows the tracking of ecosystem responses in whole systems allowing realism not attainable in smaller-scale experiments. Results from studies using this approach have found that improvements to water quality parameters and co-occurring fishes' growth rates, body condition, and relative abundances are often limited and ephemeral (1-2 years post treatment) for reasons not fully understood (Kirk et al. 1986; Flammang 2014; Neely et al. 2018).

Study Objectives

Broadly, we sought to assess the effect of selective Gizzard Shad removal on food web components via replicated whole-lake low-dose rotenone manipulations in Kansas small impoundments in a paired before-after-control-impact (BACI) study design. Our first objective

was to test for changes in limnological response variables following Gizzard Shad removal from small impoundment ecosystems and compare them to control lakes with similar fish communities and limnological parameters. Our second objective was to test for changes to small impoundment fish community dynamics, with an emphasis on testing for changes in Largemouth Bass and Bluegill relative abundances and growth rates.

Predictions and Hypothesized Mechanisms

There are several mechanisms by which Gizzard Shad removal could induce both bottom-up and top-down responses to small impoundment food webs. We hypothesized that Gizzard Shad removal would elicit reductions in pelagic nutrient concentrations (TN and TP) because benthic foraging by Gizzard Shad on organic detritus and the resulting excretion resuspends inorganic nutrients into the water column. However, we expected greater reductions in TP resulting in an increase in N:P ratio following Gizzard Shad removal. Next, we hypothesized two mechanisms by which Gizzard Shad removal would lead to changes in algal abundance. First, lower concentrations of TN and TP and higher N:P should limit algal production, particularly of N-fixing cyanobacteria (Schaus et al. 1997; Harris et al 2014). Second, we hypothesized that because Gizzard Shad are zooplanktivorous, rates of zooplanktivory would decline following Gizzard Shad removal, resulting in increases in cladoceran and calanoid copepod densities and reduced densities of smaller-bodied rotifers. Reductions in zooplankton herbivory following shad removal might lead to an increase in phytoplankton abundance (Carpenter et al. 2001). Thus, it is possible these influences would cancel each other out and Gizzard Shad removal would not alter phytoplankton abundance.

For fish community responses, we predicted that relative abundances of planktivorous fishes such as Bluegill (*Lepomis macrochirus*) and young-of-year Largemouth Bass

(*Micropterus salmoides*) would increase following Gizzard Shad removal because of reduced interspecific competition for zooplankton. One alternative hypothesis is that growth and relative abundances of Largemouth Bass > 150 mm TL would decrease following shad removal because of lack of available prey fish, but that growth rates of Largemouth Bass < 150 mm TL would increase because of more abundant zooplankton resources.

Methods

Experimental Design

BACI designs are frequently used to determine the impacts of natural or experimentally induced ecosystem perturbations (Eberhardt 1976; Green 1979; Stewart-Oaten et al. 1986; Conner et al. 2016; Michaletz 2020; Reynolds and Aldridge 2021). In 2017, we identified eight Kansas impoundments that had similar fish communities and morphometry. We collected baseline measurements of all limnological and fish community parameters for two years (2017 and 2018) before applying rotenone to four treatment lakes in December 2018. Rotenone applications did not successfully eradicate Gizzard Shad in two of the four treatment impoundments, and a dam failure and low water level impeded our ability to sample one of the control impoundments, resulting in a reduced sample size of five impoundments: three controls and two treatment impoundments. We tracked all five impoundments for two years after rotenone application (2019 and 2020) to assess ecosystem responses to biomanipulation.

Water Nutrient Analysis

Replicate surface water grab samples (25 ml) taken from a depth of 0.5-m from each sampling station were analyzed for TN and TP. Samples were immediately placed on ice in coolers and then frozen until analysis. Samples were thawed in a refrigerator overnight prior to analysis. A 1 to 10 mL water sample was digested with potassium persulfate in an autoclave and

then analyzed using an Alpkem rapid flow analyzer (Alpkem Corporation, Clackamas, Oregon, USA) for TN and TP (Hosomi and Sudu 1986; Nelson 1987) by the Kansas State University Soil Testing Lab.

Phytoplankton Collection and Enumeration and Measurement of Water Clarity

Phytoplankton were collected by taking one surface water grab sample in 25 mL vials at a depth of 0.5 m from two limnological stations (up-lake and down-lake) per impoundment. Samples were immediately preserved with 25 drops (approx. 1 mL) of Lugol's iodine and stored in opaque glass vials on ice in a dark cooler to halt photosynthesis and degradation before being stored in a dark setting. We settled 5 mL subsamples for a minimum of 48h in 10 mL polycarbonate tubes affixed with cover slips before placing on an inverted microscope. Images of 50 fields of view at 400X magnification were captured for each sample with a digital camera, and algae were counted and measured from images. We classified algae into broad taxonomic categories (green, cyanobacteria, diatom, dinoflagellate). Algae were placed into size bins by superimposing a 10 x 10 μm grid onto images and counting the number of grids occupied by each natural unit (e.g., filament or colony) as an approximation of the surface area of each unit. We report surface area per 50 fields of view of the different algal groups (number of grids occupied per number of fields counted) for each sample as a metric of algal abundance (Moheimani et al. 2013; Guillard and Sieracki 2005). We also measured water clarity (vertical visual extinction) at each station with a Secchi disc (Steel and Neuhausser 2002).

Zooplankton Collection and Enumeration

Pelagic zooplankton communities were sampled via three replicate 3-m tows of a Wisconsin plankton net (Wildco #623-1130, 80- μm mesh, 12.7 cm diam, 6.35 cm radius; Yulee, FL, USA) per station. Vertical tows were performed at sampling stations greater than 3.5 m

deep and horizontal tows were performed at stations less than 3.5 m deep with approximate tow speeds of 1 m/s. Net contents from three replicate tows were washed and pooled into a single Nalgene bottle and immediately fixed with 4% buffered sucrose formalin (Haney and Hall 1973) and transported on ice. Samples were diluted to a known volume. We used a Hensen-Stempel pipette (Wildco #3-1806-D52, Yulee, FL, USA) to transfer 2 to 10 mL of sample at a time to a Ward counting wheel depending on sample densities (Wildco #3-1806-D52, Yulee, FL, USA). All zooplankton counted were classified under one of seven taxonomic categories: Cladocera, Calanoida, Cyclopoida, Harpacticoida, Rotifera, Nauplii, and Culicomorpha (Chaoboridae). Mean zooplankton densities were calculated and reported as the average counts of each group per L (Mack et al. 2012; de Bernardi 1984; Beaver et al. 2018).

Quantifying Littoral Fish Community Structure

We conducted daytime littoral seining with a 4.5 x 1.8 m seine (3.2-mm mesh) at three fixed stations per impoundment concurrent with limnological sampling events once per month in June, August, and October 2017-2020: up-lake (close to the inlet), mid-lake, and down-lake (as close to the dam as possible). We identified and measured all fishes >30 mm to species. Juveniles < 30 mm were excluded from our analysis because their abundance was highly variable and identification was questionable. We recorded surface area seined for each sampling station to give densities of different species.

Catch Rates of Largemouth Bass and Bluegill from Standard Sampling Procedures

We used data collected between 2010 and 2021 by the Kansas Department of Wildlife and Parks (KDWP) standard sampling (Bonar et al. 2009) in our study impoundments to quantify relative abundances for Largemouth Bass and Bluegill. Largemouth Bass were sampled using daytime boat electrofishing each spring during their spawning period when water temperatures

were 15.6 - 21.1 °C (Koch et al. 2014). Electrofishing surveys were conducted at a minimum of six randomly selected shoreline stations for 10 min (Miranda and Boxrucker 2009). Bluegills were collected with trap nets between October 1 to November 30 when water temperatures were 20 °C or below. Trap nets consisted of 0.9 x 1.8 m rectangular frames, a 10-cm opening in the frame, four 0.74 m-diameter hoops with a funnel attached to the first and third hoops, with 30.5 m leads constructed of 1.3-cm bar mesh (Pope et al. 2009). Trap nets were deployed in the afternoon at randomly selected locations in water 0.9-4.9 m deep and retrieved the following morning so sampling encompassed two crepuscular periods (Koch et al. 2014).

Collecting Age Structures to Estimate Largemouth Bass and Bluegill Length-at-Age

Total length (hereafter TL) for all Largemouth Bass and Bluegill from electrofishing surveys were measured and sagittal otoliths were removed from a target of ten individuals per length-group (Table 1) for each species per sampling event in 2018 and 2020. Otoliths were aged to compare length-at-age before and after rotenone manipulation (see methods outlined in Koch et al. 2019; Klein et al. 2017; Long and Schill 2017). Largemouth Bass and Bluegill otoliths were mounted in epoxy (Koch and Quist 2007) and sectioned with a Buehler Isomet low-speed saw (Buehler, Lake Bluff, IL). Two independent readers viewed sectioned and whole otoliths with reflected light on a stereoscope (10-40X magnification) linked to a television monitor and assigned ages to Bluegill and Largemouth Bass otoliths. Discrepancies were resolved by consensus reading. None of the readers had knowledge of fish lengths for any specimen and all readers had extensive experience enumerating annuli of Largemouth Bass and Bluegill prior to the study. We considered only age-1 and age-2 fish in our analysis because of small sample sizes for older fish. Mean length of age-0 Bluegill was estimated using lengths of littoral seine fish data from October. Mean length of age-0 Largemouth Bass could not be

estimated due to low and imbalanced sample size. Using minimum length-at-age-1 data for aged Bluegill, we considered only Bluegill collected from October sampling that were < 72 mm TL as age-0 in this analysis.

Data Analysis

We fit linear fixed effects models using our BACI design to evaluate how Gizzard Shad eradication affected abiotic and biotic response variables using the “lm” function in the *stats* package (R Core Team, 2021), with impoundment treatment (rotenone vs. control), time period (before vs. after rotenone manipulation), and sampling month as fixed effects. We assumed that there was interannual variation within each impoundment, so our design focused on quantifying the relative differences between responses in our treatment and control impoundments. If the interaction between treatment and time period was significant at $\alpha = 0.10$, that interaction indicated a significant treatment effect of the rotenone manipulation. This alpha level was used to identify potentially important effects regardless of the low power of our experiment associated with having to remove several replicates as described above. All analyses were performed in R version 4.1.0 (R Core Team, 2021).

Results

We observed a significant interaction between treatment and time period for both TN ($p = 0.048$) and TP ($p = 0.080$) (Figure 3.2, Table 3.2). As predicted, TN decreased by 41% after rotenone manipulation compared to a 14% decline in control impoundments, and TP decreased 37% in rotenone treatments after shad removal compared to the 9% decrease between pre- and post-treatment concentrations in control impoundments. However, contrary to our predictions we did not observe changes to N:P ratios in treatment impoundments relative to control impoundments in response to rotenone. There was no significant interaction between treatment

and time period for Secchi depth ($p = 0.399$) (Table 3.2). We did not observe a significant effect of Gizzard Shad removal on phytoplankton communities, including cyanobacteria abundance (treatment x time interaction $p > 0.37$) (Table 3.3). The only zooplankton taxonomic group to respond to the rotenone manipulation was calanoid copepods (Figure 3.3, Table 3.4), which increased 6% in response to rotenone manipulation in treatment impoundments ($p = 0.047$).

The only size class of Largemouth Bass to respond to rotenone manipulations were those < 150 mm TL, which increased in relative abundance by 16% in response to rotenone manipulation ($p = 0.043$) (Figure 3.4, Table 3.5). Bluegill relative abundances declined following rotenone manipulation relative to controls for both size classes ($p < 0.006$) (Figure 3.5, Table 3.5), decreasing 84% among individuals > 100 mm TL and 76% among individuals < 100 mm TL after rotenone treatments compared to pre-treatment abundances. There was no difference in length-at-age for any size class of Bluegill (Appendix B.6) or Largemouth Bass (Appendix B.7) post-treatment.

Discussion

We found little evidence that Gizzard Shad removal had a substantial impact on limnological parameters or fish populations in small Kansas impoundments following experimental manipulations. One exception was an observed decrease in epilimnetic TN and TP concentrations following Gizzard Shad eradication, a finding in concordance with our hypotheses and consistent with our results in Chapter 2. In addition, we had hypothesized that removing Gizzard Shad and the associated low N:P ratio with which they excrete nutrients would shift phytoplankton communities in favor of nitrogen-fixing cyanobacteria and chlorophytes (Schaus et al. 1997; Schaus and Vanni 2000) but found no evidence that Gizzard Shad eradication meaningfully affected phytoplankton community composition, N:P ratios, or Secchi

depths. Similarly, a biomanipulation experiment by Catalano et al. (2010) which reduced total Gizzard Shad population biomass by 32% via gillnetting found no significant short-term changes to total phosphorus concentration, Secchi depth, or chlorophyll *a* concentration following the reduction.

With the exception of a 9% increase in mean-length of age-0 Largemouth Bass, growth of Largemouth Bass and Bluegill (i.e., length-at-age or mean length of age-0 fish) did not exhibit marked responses to rotenone manipulations. We did observe an unexpected pattern of decreased Bluegill relative abundance following rotenone treatments, a decline potentially driven by incidental non-target mortality following rotenone application. This finding aligns with the incidental eradication of non-target Bluegill at a rotenone concentration of 9.5 ug/l reported by Flammang (2014), an issue in conflict with the Iowa DNR's desired management outcome of game fish preservation. Flammang (2014) fit logistic models to predict Bluegill mortality rates at different rotenone concentrations and suggested that treatment levels of 8.0 µg/l would result in a 40% reduction in Bluegill densities compared to a 20% reduction at a concentration of 7.0 mg/l, illustrating the need for accurate lake volume estimates and rotenone concentration verification prior to treatments. In contrast to our findings, a low-dose rotenone manipulation study by Neely et al. (2018) found that relative abundance of age-0 Bluegills and Largemouth Bass immediately increased following treatment in 2014 but decreased to pretreatment levels in 2015 and 2016. Unfortunately, it is difficult to parse out the effects of non-target Bluegill or Largemouth Bass mortality on changes to relative abundances following rotenone manipulations. We observed some incidental mortality of Bluegill following the rotenone treatment, but many of the fish sank to the bottom of the impoundment, prohibiting our attempts to quantify mortality.

Our study highlights some of the challenges in successfully executing whole-lake manipulation experiments, especially in small impoundments managed for angling recreation. Rotenone manipulations did not successfully eradicate Gizzard Shad from two of the four treatment impoundments: Washington State Fishing Lake (SFL) and Neosho SFL. Washington SFL has complex habitat structure around the impoundment's inflow that may have provided refugia, precluding complete Gizzard Shad eradication. Given the high levels of fecundity Gizzard Shad can achieve (Stein et al. 1995) anything less than complete eradication can be quickly outpaced by increased recruitment due to compensatory mechanisms (Catalano et al. 2010; Flammang 2014). Next, even if complete Gizzard Shad eradication in Neosho SFL had been successfully achieved, historically high spring flooding throughout the Neosho River Basin in 2019 might have created sufficient connectivity between the impoundment and upstream farm ponds with Gizzard Shad to allow for rapid recolonization and recruitment less than 6 months post-treatment. Additionally, one of our four paired "control" impoundments with Gizzard Shad experienced a partial dam collapse following a heavy rain event in November 2018 and had to be dewatered for concrete repairs and then re-stocked. The resultant small sample size left us with lower statistical power, making it difficult to draw strong inference from our data.

Baseline Gizzard Shad relative abundances before rotenone in the four treatment impoundments (Figure 3.1) also might not have been high enough to be significantly influencing limnological processes or fish communities on a scale that could be detected with our experimental design. For example, pre-treatment shad CPE in Pott #1 SFL was particularly low, potentially limiting a response in that impoundment. Gizzard Shad are facultatively omnivorous and can undergo biomass-dependent diet shifts from zooplankton resources to benthic detritus when zooplankton are reduced, so the magnitude of their effects on limnological variables is

strongly linked to their total biomass and size distribution. Schaus et al. (2002) used conventional gut content and stable isotope analysis in Acton Lake, OH to demonstrate that when Gizzard Shad biomass is higher (35 kg/ha), individuals > 35-mm standard length (SL) obtain most of their C from benthic detritus, but when biomass is low (15 kg/ha), zooplankton biomass increases and all sizes of Gizzard Shad obtain most of their C from zooplankton. Zooplanktivorous age-0 Gizzard Shad grow 3x more rapidly than detritivorous shad, illustrating how diet quality and growth rates are strongly linked to shad biomass and how such biomass-dependent feedback loops may sustain high interannual Gizzard Shad biomass in eutrophic systems like Acton Lake. Zooplanktivorous Gizzard Shad recycle nutrients within the water column whereas detritivorous shad transport nutrients from sediments to the water column, so these biomass-induced diet shifts modulate the intensity of their effects on phytoplankton through both top-down and bottom-up pathways (Schaus et al. 2002). Thus, the baseline shad densities in our study impoundments may have been too low to necessitate high rates of detritivory, which may have reduced their relative impact on nutrient levels and plankton communities.

Is selective eradication via low-dose rotenone manipulations a viable solution for Gizzard Shad management? Whole-lake low-dose rotenone manipulations are costly, labor-intensive, and difficult to implement (Flammang 2014; Michaletz 2020), and these endeavors may only be warranted for limited cases depending on specific management objectives. Gizzard Shad have undergone significant range expansion in recent decades (Fuller et al. 2021; Vatland and Budy 2007) due to a combination of factors, including purposeful introductions by agencies to bolster prey availability for piscivores in reservoirs (Finzel et al. 2017; Michaletz 2020), illegal or incidental introductions by anglers releasing live bait (Flammang 2014), and flooding generating

connectivity across the landscape between public waterways and private ponds (Jackson and Pringle 2010). Two of four impoundments without Gizzard Shad that were also monitored during our four-year study experienced unanticipated Gizzard Shad introductions in 2019, which underscores just how easily and often Gizzard Shad invade waters managed for angling recreation. Altogether, our work adds to a growing body of literature suggesting that low-dose rotenone manipulations may occasionally serve as bandages for temporarily controlling hyperabundant Gizzard Shad in small impoundments with imbalanced fisheries or water quality issues rather than a long-term solution (Neely et al. 2018; Michaletz 2020). As multiple stressors continue to impact small impoundment food webs, multi-pronged approaches combining several methods (e.g., repeated rotenone manipulations, mechanical removal, reductions in nutrient loading, and piscivore stocking) might be necessary to prevent hyperabundant benthic-feeding fishes like Gizzard Shad and Common Carp from rebounding to pretreatment levels and effectively mitigate against their adverse effects (Michaletz 2020; Kim and DeVries 2000; Weber and Brown 2009; Flammang 2014).

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Tables

Table 3.1 Length groups of Largemouth Bass and Bluegill collected for aging from Kansas small impoundments.

Species	Length Group I	Length Group II	Length Group III
Largemouth Bass (<i>Micropterus salmoides</i>)	< 150 mm	150 – 250 mm	> 250 mm
Bluegill (<i>Lepomis macrochirus</i>)	< 100 mm	> 100 mm	

Table 3.2 Linear fixed effect model outputs for responses of abiotic parameters to rotenone treatments in Kansas small impoundments. Asterisks denote statistical significance at $\alpha = 0.10$.

Model	Fixed Effect or Interaction Term	Estimate	Std. Error	t-value	95% CI	p-value
Total Nitrogen	Treatment	-0.751	0.196	-3.829	-1.138; -0.363	< 0.001*
	Time Period	-0.007	0.124	-0.057	-0.252; 0.238	0.954
	Treatment x Time Period	0.486	0.244	1.998	0.005; 0.968	0.048*
Total Phosphorus	Treatment	-1.500	0.198	-7.596	-1.890; -1.110	< 0.001*
	Time Period	0.279	0.175	1.601	-0.065; 0.624	0.111
	Treatment x Time Period	0.489	0.277	1.764	-0.058; 1.036	0.080*
Secchi Depth	Treatment	0.865	0.159	5.426	0.550; 1.179	< 0.001
	Time Period	0.131	0.145	0.904	-0.155; 0.416	0.367
	Treatment x Time Period	-0.192	0.227	-0.846	-0.639; 0.256	0.399
N:P Ratio	Treatment	0.611	0.285	2.172	0.087; 1.137	0.032*
	Time Period	-0.222	0.181	-1.379	-1.035; 0.592	0.170
	Treatment x Time Period	0.205	0.355	0.280	-0.461; 0.848	0.780

Table 3.3 Linear fixed effect model outputs for responses of phytoplankton abundances to rotenone treatments in Kansas small impoundments. Asterisks denote statistical significance at $\alpha = 0.10$.

Model	Fixed Effect or Interaction Term	Estimate	Std. Error	t-value	95% CI	p-value
Cyanobacteria Abundance	Treatment	0.691	0.268	2.574	0.153; 1.229	0.013*
	Time Period	-0.179	0.244	-0.736	-0.667; 0.309	0.465
	Treatment x Time Period	0.382	0.382	0.901	-0.421; 1.109	0.372
Green Phytoplankton Abundance	Treatment	-0.032	0.255	-0.125	-0.544; 0.480	0.901
	Time Period	0.211	0.232	0.912	-0.253; 0.676	0.366
	Treatment x Time Period	0.076	0.363	0.208	-0.652; 0.804	0.836
Diatom Abundance	Treatment	-0.438	0.482	-0.908	-1.403; 0.528	0.368
	Time Period	-0.810	0.437	-1.852	-1.686; 0.066	0.069
	Treatment x Time Period	0.098	0.685	0.143	-1.275; 1.471	0.887

Table 3.4 Linear fixed effect model outputs for responses of zooplankton abundances to rotenone treatments in Kansas small impoundments. Asterisks denote statistical significance at $\alpha = 0.10$.

Model	Fixed Effect or Interaction Term	Estimate	Std. Error	t-value	95% CI	p-value
Cladoceran Density	Treatment	-0.693	0.358	-1.934	0.294; 1.113	0.055*
	Time Period	-1.398	0.326	-4.289	-1.583; -0.839	< 0.001*
	Treatment x Time Period	0.766	0.508	1.509	-0.993; 0.167	0.133
Calanoid Density	Treatment	-0.662	0.426	-1.553	-1.502; 0.179	0.122
	Time Period	-1.597	0.387	-4.124	-2.361; -0.833	< 0.001*
	Treatment x Time Period	1.21	0.603	2.005	0.019; 2.401	0.047*
Cyclopoid Density	Treatment	0.637	0.480	1.327	-0.310; 1.583	0.186
	Time Period	-1.414	0.436	-3.242	-2.275; -0.553	0.001*
	Treatment x Time Period	-0.053	0.679	-0.078	-1.395; 1.288	0.938
Rotifer Density	Treatment	1.484	0.482	3.079	0.533; 2.437	0.002*
	Time Period	-1.188	0.439	-2.709	-2.053; -0.322	0.007*
	Treatment x Time Period	-0.773	0.683	-1.131	-2.122; 0.576	0.260
Copepod Nauplii Density	Treatment	-0.018	0.263	-0.069	-0.537; 0.500	0.845
	Time Period	-0.550	0.239	-2.303	-1.022; -0.079	0.023*
	Treatment x Time Period	-0.024	0.372	-0.064	-0.759; 0.711	0.949

Table 3.5 Linear fixed effect model outputs for responses of Largemouth Bass and Bluegill relative abundances in Kansas small impoundments. Largemouth Bass were sampled via daytime shoreline electrofishing and Bluegill were sampled via trap netting for relative abundance estimates from each impoundment in October 2017-2020. Asterisks denote statistical significance at $\alpha = 0.10$.

Model	Fixed Effect or Interaction Term	Estimate	Std. Error	t-value	95% CI	p-value
E-fish Largemouth Bass CPUE >250 mm TL	Treatment	0.262	0.314	0.835	-0.355; 0.879	0.404
	Time Period	0.750	0.201	3.735	0.355; 1.145	< 0.001*
	Treatment x Time Period	0.563	0.351	1.603	-0.127; 1.253	0.110
E-fish Largemouth Bass CPUE 150 – 250 mm TL	Treatment	1.321	0.322	4.102	0.688; 1.954	< 0.001*
	Time Period	0.093	0.206	0.453	-0.312; 0.499	0.651
	Treatment x Time Period	-0.011	0.361	-0.031	-0.720; 0.698	0.975
E-fish Largemouth Bass CPUE < 150 mm TL	Treatment	0.447	0.297	1.505	-0.137; 1.031	0.133
	Time Period	-0.494	0.190	-2.597	-0.869; -0.120	0.010*
	Treatment x Time Period	0.675	0.333	2.029	0.021; 1.329	0.043*
Trap Net Bluegill CPUE > 100 mm TL	Treatment	0.082	0.164	0.497	-0.242; 0.405	0.620
	Time Period	0.123	0.111	1.104	-0.096; 0.342	0.271
	Treatment x Time Period	0.527	0.182	2.900	0.169; 0.885	0.004*
Trap Net Bluegill CPUE < 100 mm TL	Treatment	-0.114	0.109	-1.047	-0.328; 0.100	0.296
	Time Period	0.013	0.073	0.173	-0.132; 0.158	0.863
	Treatment x Time Period	0.332	0.120	2.766	0.095; 0.569	0.006*

Figures

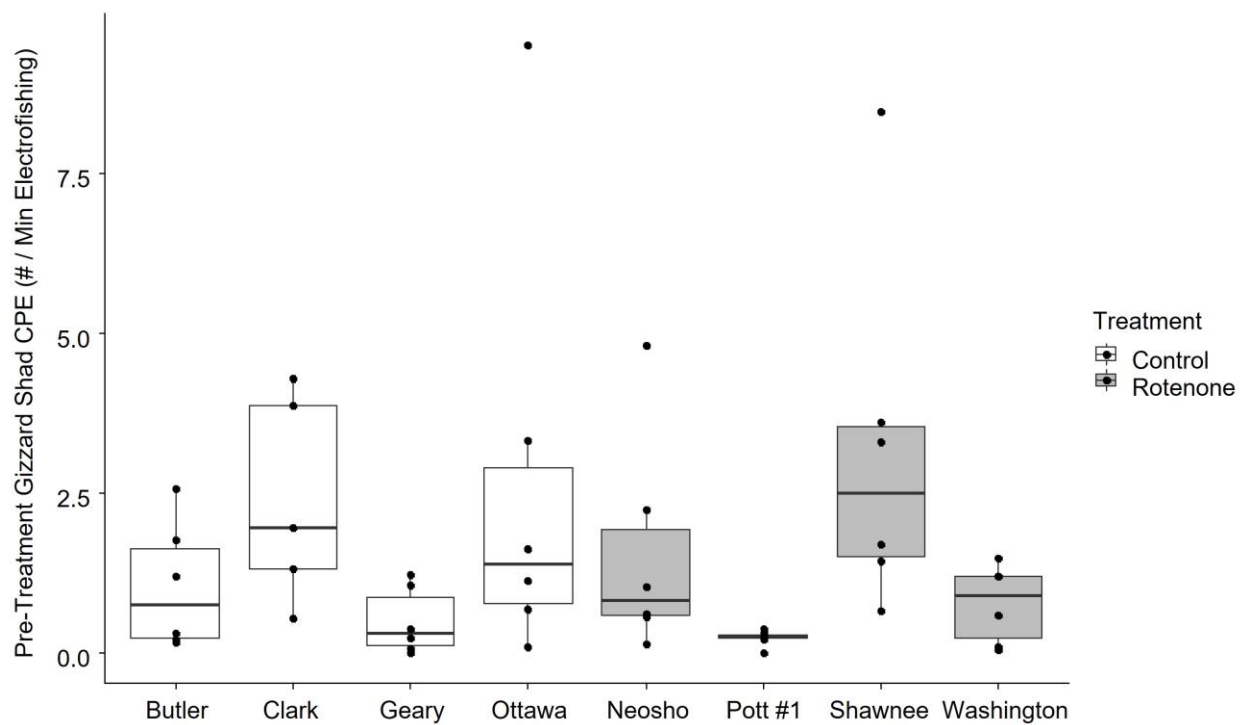


Figure 3.1 Pre-treatment Gizzard Shad catch per unit effort (CPE, #/minute electrofishing) before rotenone manipulations (2017-2018). Relative abundances were obtained using daytime shoreline boat electrofishing in each impoundment.

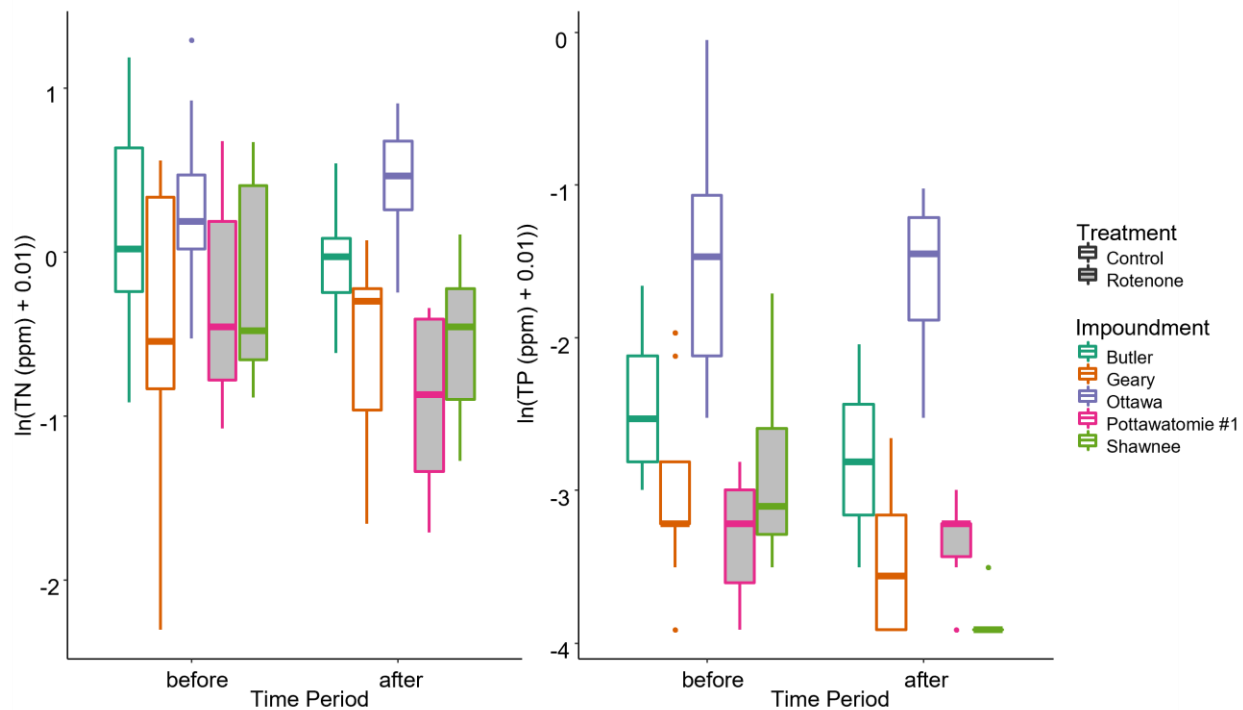


Figure 3.2 TN (left) and TP (right) concentrations before and after Gizzard Shad eradication. Pelagic nutrients were sampled at three stations in each impoundment in June, August, and October from 2017 to 2020. Stations were positioned in limnetic habitats in riverine zones, mid-lake transition zones, and lacustrine zones. Replicate 25 mL surface water grab samples were collected for Total N and P from a depth of 0.5 m from each station and analyzed for TN and TP with an Alpkem rapid flow analyzer.

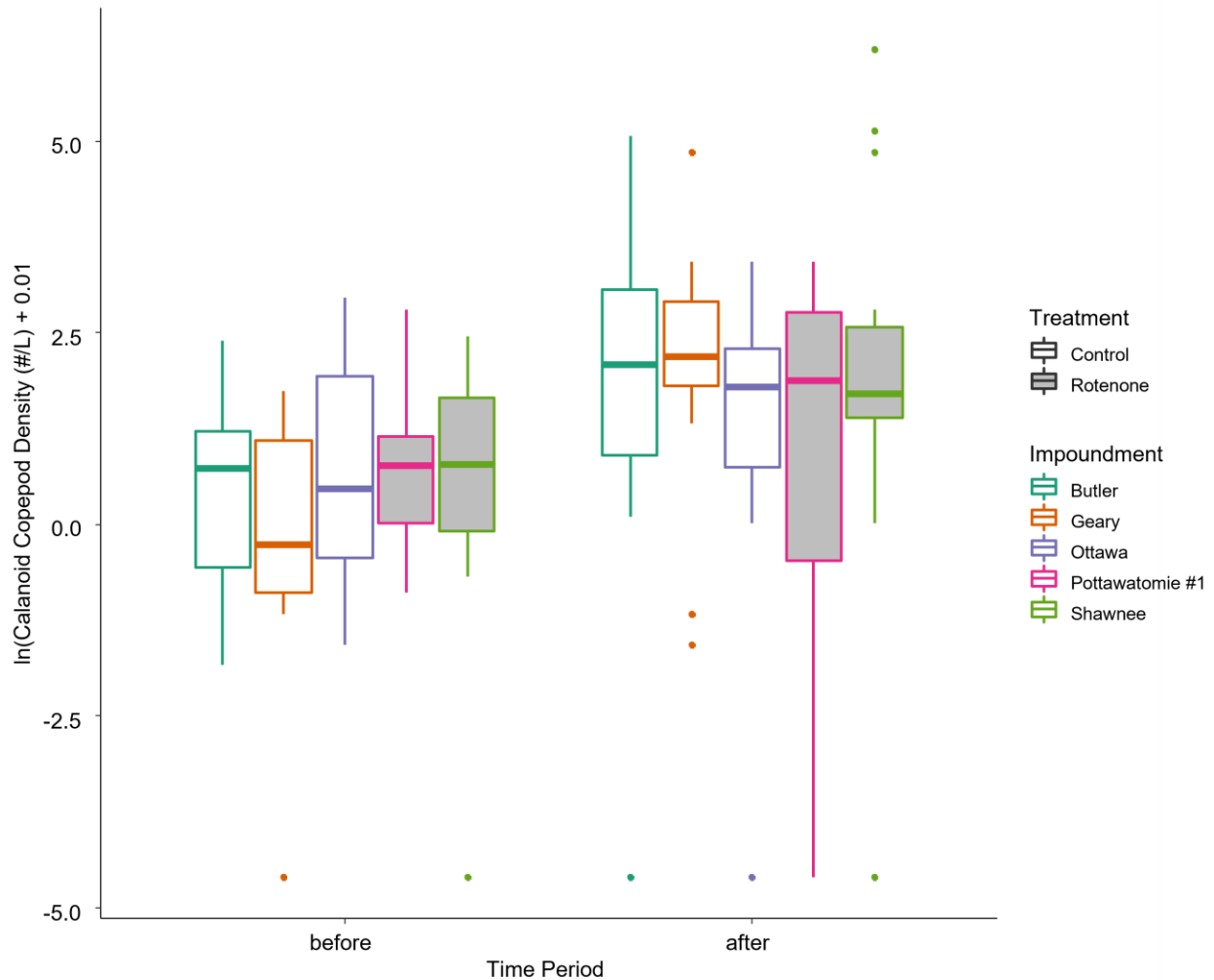


Figure 3.3 Calanoid copepod densities before (left) and after (right) Gizzard Shad eradication in control and rotenone impoundments. Pelagic zooplankton communities were sampled via three replicate 3-m Wisconsin plankton net tows per station in June, August, and October 2017-2020. Vertical tows were performed at stations greater than 3.5 m deep and horizontal tows were performed at stations less than 3.5 m deep with approximate tow speeds of 1 m/s. Zooplankton were classified under one of seven taxonomic categories: Cladocera, Calanoida, Cyclopoida, Harpacticoida, Rotifera, Nauplii, and Culicomorpha (Chaoboridae). Mean zooplankton densities were calculated and reported as the average counts of each group per L.

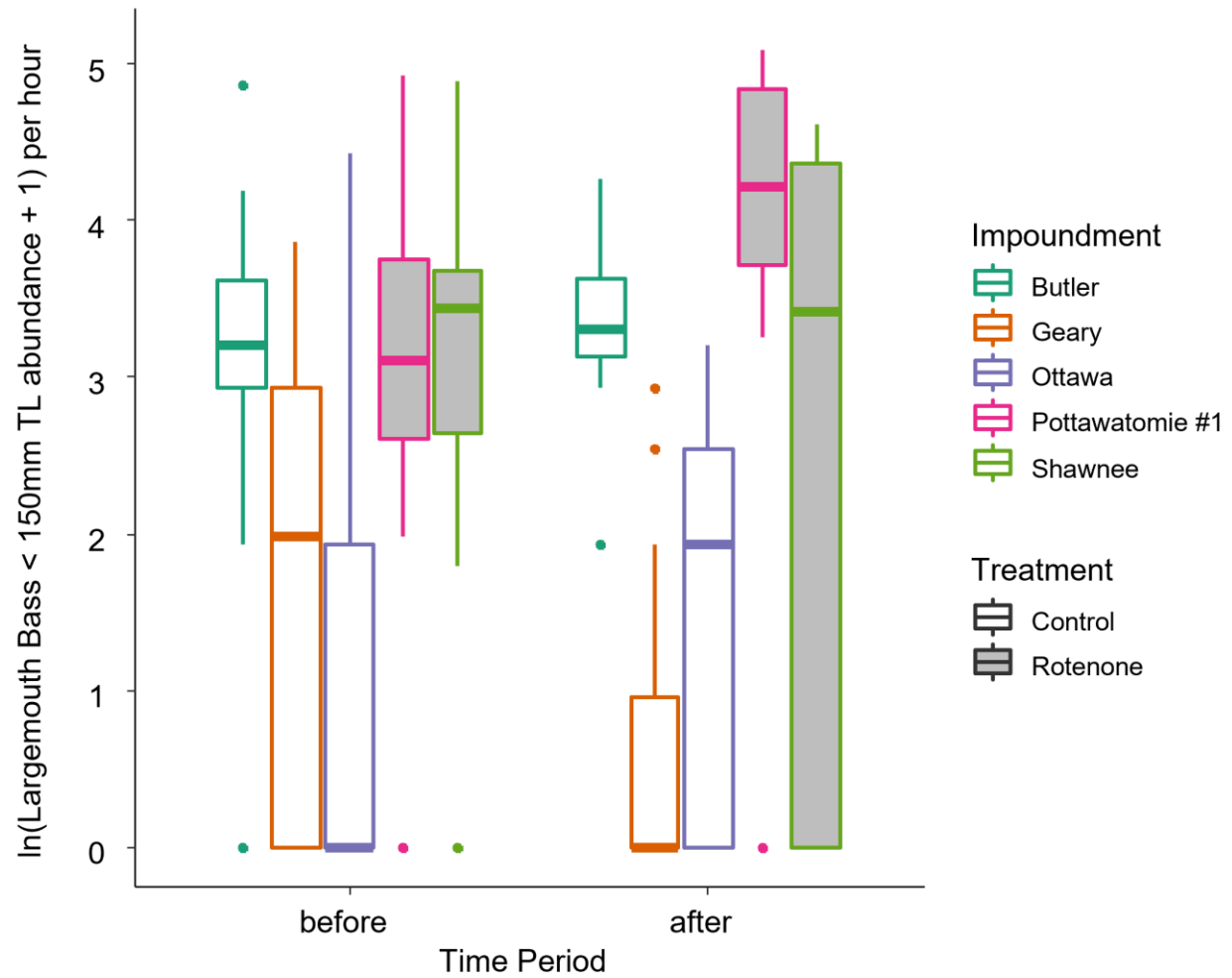


Figure 3.4 Relative abundance of Largemouth Bass < 150 mm TL before and after Gizzard Shad eradication. Data were collected via spring electrofishing in each impoundment by KDWP. Relative abundances were log-transformed.

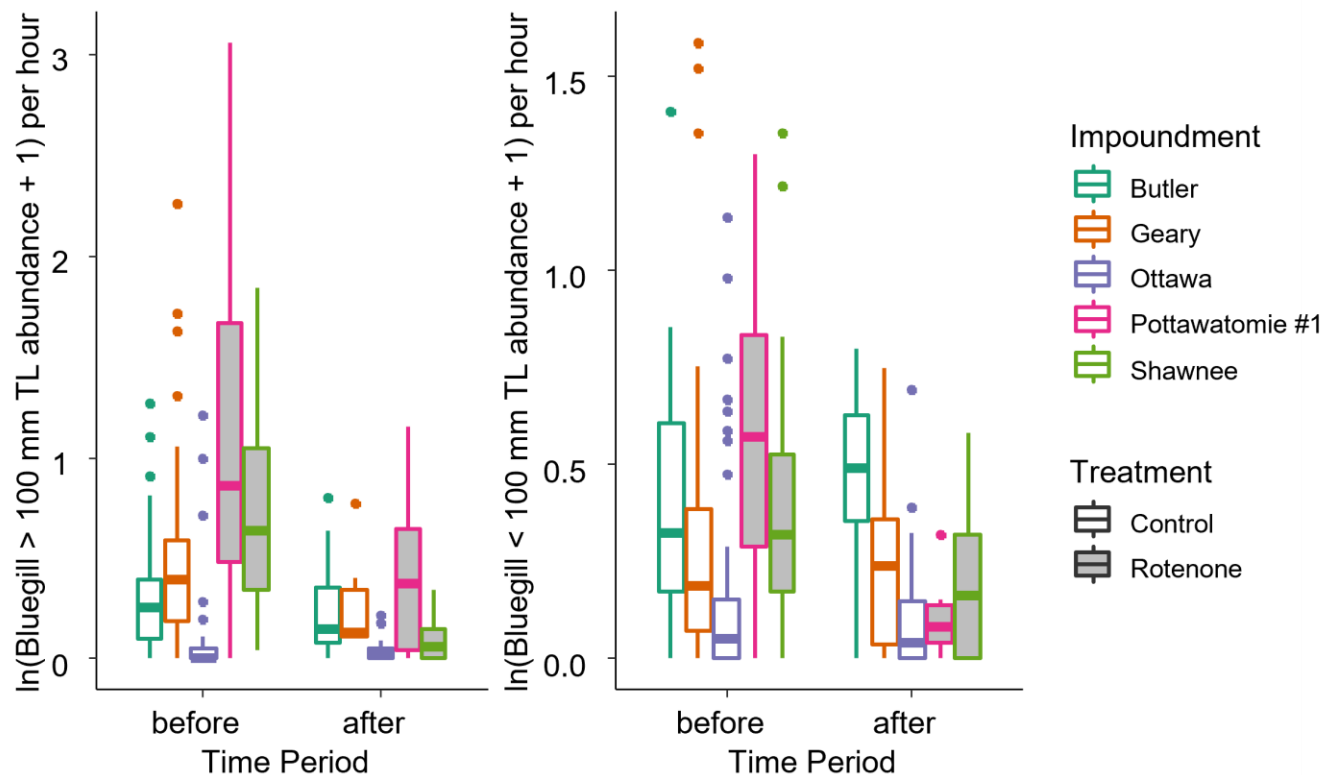


Figure 3.5 Relative abundance of Bluegill greater than 100 mm TL (left) and less than 100 mm TL (right) before and after Gizzard Shad eradication. Bluegill were sampled via trap netting in each impoundment by KDWP from 2017-2020. Relative abundances were log-transformed.

**Chapter 4 - Seasonal and ontogenetic variation in largemouth bass
(*Micropterus salmoides*) diets across a gradient of gizzard shad
abundance: bass piscivory does not change with shad eradication
via low-dose rotenone**

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Abstract

Variation in the prey base of Kansas small impoundment food webs may be driven by densities and size structure of gizzard shad (*Dorosoma cepedianum*), but the impact of shad abundance on largemouth bass (*Micropterus salmoides*) diet composition is poorly understood. We sought to describe complete diets of largemouth bass in 12 Kansas small impoundments and examine how they varied across bass size classes, months, and a gradient of gizzard shad relative abundances. Specifically, we hypothesized that frequency of occurrence of fish in the diet (piscivory) would increase with gizzard shad abundance and largemouth bass size and be highest in August in conjunction with seasonal peaks in age-0 prey fish abundance and water temperatures and metabolic rates. We used regression models to evaluate 1) seasonal variation in the frequency of occurrence of fish (piscivory) in largemouth bass diets across months, 2) ontogenetic variation in piscivory (i.e., variation in diets across different length classes of largemouth bass), and 3) variation in relation to gizzard shad relative abundance. We also tested whether selective gizzard shad removal via low-dose rotenone manipulation influenced largemouth bass piscivory, hypothesizing that piscivory would decline in treatment impoundments following shad removal due to reduced prey fish availability. Bass length class

was a significant factor influencing the frequency of occurrence of fish in largemouth bass diets ($P = 0.004$), but piscivory was not associated with sampling month or gizzard shad relative abundance. Accordingly, gizzard shad eradication did not alter the frequency of occurrence of fish in largemouth bass diets ($P = 0.390$) following rotenone manipulation. Since the occurrence of gizzard shad prey in bass diets was low compared to centrarchids, cannibalism, and crayfish, selective removal of gizzard shad via rotenone biomanipulation in Kansas small impoundments might not meaningfully alter largemouth bass diet composition.

Keywords: piscivory, largemouth bass, gizzard shad, small impoundments, rotenone, biomanipulation, diet composition

Introduction

Gizzard shad (*Dorosoma cepedianum*) are often thought to be the primary prey for open-water, gape-limited piscivorous fishes in reservoir ecosystems (Noble, 1981; Storck, 1986; DeVries & Stein, 1990), but the extent to which they are preyed upon by largemouth bass (*Micropterus salmoides*) in Kansas small impoundments is unclear. At small sizes, gizzard shad can be preferred prey for piscivorous fishes (Hartman & Margraf, 1992; Garvey & Stein, 1998) and age-0 gizzard shad abundance can be one of the primary drivers of the growth of predatory game fishes (Michaletz, 1997). However, shad can rapidly grow too large and deep-bodied to be consumed by gape-limited piscivores (Storck, 1986; Garvey & Stein, 1998; Stein et al., 1995; Michaletz et al., 1997; Irwin et al., 2003) because they only ingest prey they can swallow whole (Hambricht, 1991). Largemouth bass exhibit an ontogenetic diet shift from zooplanktivory to piscivory as juveniles (Mittelbach & Persson, 1998; Einfalt et al., 2015; Yasuno et al., 2012), and this transition is a critical niche shift because an early transition may increase growth and decrease mortality, thereby enhancing lifetime fitness (Olson, 1996; Post, 2003).

There is notable variation in the prey base among Kansas small impoundments associated with the presence or absence of gizzard shad (Appendix C.1). Gizzard shad populations can proliferate under certain conditions, resulting in high abundances of the age-0 and subsequent year classes (Vanni et al., 2005). An understanding of how predator diets are affected by variation in predator-prey size relationships, seasonality, and variation in prey abundance can be used to improve the management of these fisheries. However, it is not yet clear what impact the relative abundance of gizzard shad has on largemouth bass, an important predator in these ecosystems. Although gizzard shad may provide an important source of forage for piscivorous fishes in reservoirs (Michaletz, 1997), hyperabundant gizzard shad present a growing management challenge in Kansas small impoundments (Neely et al., 2016) because of their potential to negatively influence the growth and survival of littoral sport fishes such as largemouth bass.

Rotenone ($C_{23}H_{22}O_6$), a plant-derived piscicide, has been used at low doses (5.5-7.5 ppb) to selectively reduce or eliminate gizzard shad populations with variable success (Zeller and Wyatt, 1967; Kirk et al., 1986; Kim & DeVries, 2000; Catalano et al., 2010; Flammang, 2014). Results from studies using this approach have found that improvements to water quality parameters and game fish growth rates, body condition, and relative abundances small impoundments are often limited and ephemeral (1-2 years post treatment) for reasons not fully understood (Kirk et al., 1986; Flammang, 2014; Neely et al., 2018). However, it is unclear whether shad removal would generate negative consequences for largemouth bass diets and growth, and to our knowledge no study has evaluated changes to largemouth bass diet composition following experimental gizzard shad eradication from a reservoir. Therefore, we sought to characterize the relative importance of gizzard shad as prey components of largemouth

bass diets in Kansas small impoundments and assess whether the frequency of occurrence of gizzard shad changed in response to the elimination of a prey resource.

Study Objectives, Questions and Hypothesized Mechanisms

Our first objective was to quantify the diets of largemouth bass across 12 Kansas small impoundments and examine how different diet categories and fish prey species varied across size classes, months, and a gradient of gizzard shad relative abundances. We asked whether gizzard shad were the most important prey fish for bass, hypothesizing that largemouth bass would selectively prey upon gizzard shad compared to other forage fishes and therefore the frequency of occurrence of gizzard shad in largemouth bass diets would be higher than that of centrarchid fish prey. For our second objective we used regression models to evaluate associations between the frequency of occurrence of fish prey in bass diets and gizzard shad abundance, largemouth bass size, and sampling month. Specifically, we asked whether there was 1) seasonal variation in the frequency of occurrence of fish (piscivory) in largemouth bass diets across months, 2) ontogenetic variation in piscivory (i.e., variation in diets across different length classes of largemouth bass), and 3) variation in relation to gizzard shad relative abundance. We hypothesized that piscivory would vary across months and be highest in August coinciding with the highest age-0 prey fish abundances, water temperatures and metabolic rates, and that piscivory would increase with largemouth bass length such that the longest bass would have the highest frequency of occurrence of fish in their stomach contents. Finally, many studies have documented how the relative importance of a preferred prey species in a predator's diet increases as the abundance of that prey increases (Adams et al., 1982; Knight et al., 1984; Storck, 1986; Hartman & Margraf, 1992) and one study found that age-0 shad production was positively related to the species' importance within predator diets (Michaletz, 1997). Thus, we

hypothesized that gizzard shad would be preferred prey for largemouth bass and as such, piscivory would increase with increasing gizzard shad relative abundance. Our third objective was to test if the selective removal of gizzard shad via low-dose rotenone manipulation influences the frequency of occurrence of fish in largemouth bass diets. We hypothesized that piscivory by largemouth bass would decline in treatment impoundments following gizzard shad removal due to reduced food availability after a prey species is eliminated from the food web.

Methods

Study Impoundments and Experimental Design

For our first and second objectives we sampled largemouth bass diets from 12 small, eutrophic to hypereutrophic impoundments ranging in surface area from 6.9 to 165.7 ha and managed for angling recreation by the Kansas Department of Wildlife and Parks (Figure 4.1). All impoundments contained typical Midwestern USA warmwater fish communities dominated by largemouth bass (*Micropterus salmoides*) and bluegill (*Lepomis macrochirus*). Green sunfish (*Lepomis cyanellus*), redear sunfish (*Lepomis microlophus*), white crappie (*Pomoxis annularis*), black crappie (*Pomoxis nigromaculatus*), channel catfish (*Ictalurus punctatus*), flathead catfish (*Pylodictis olivaris*), and common carp (*Cyprinus carpio*) were also common in most impoundments. Gizzard shad were present in eight of the impoundments at the start of the study and variation in gizzard shad relative abundances was high both within and across impoundments between sampling events.

For our third objective we identified eight of the 12 impoundments that had gizzard shad present and similar morphometry and collected baseline largemouth bass diet samples for two years (2017 and 2018) before applying rotenone to four treatment lakes in December 2018. Rotenone applications did not successfully eradicate gizzard shad in two of the four treatment

impoundments, and a dam failure and low water level impeded our ability to sample one of the control impoundments, resulting in a reduced sample size of five impoundments: three controls and two treatment impoundments. We tracked all five impoundments for two years (2019 and 2020) to assess responses in largemouth bass diet composition to experimental removal of gizzard shad.

Quantification of Gizzard Shad Relative Abundances

Gizzard shad relative abundances for each impoundment were estimated based on boat electrofishing concurrent with sampling for largemouth bass diets. Each lake was sampled with daytime pulsed DC electrofishing (60 Hz, 25% duty cycle, ETS Electrofishing). Shad relative abundances were reported as the count of shad captured per minute electrofishing (CPUE; Appendix C.1). Electrofishing was typically conducted for shock times of 40-70 minutes total per sampling event (mean = 46 ± 14 SD minutes electrofishing).

Fish Diet Sample Collection Field Methods

To obtain diet information, largemouth bass were collected via shoreline electrofishing and seining in June, August, and October from 2017 -2020. Electrofishing was conducted between 8:00 and 11:30 AM to reduce temporal variation in stomach contents and capture diet composition just after the dawn feeding period. Netted individuals were brought into live wells and measured for total length (TL). An attempt was made to collect 10 individuals of each of three length groups: < 150, 150 to 250, and > 250 mm. Specimens were sacrificed and immediately placed on ice in coolers to slow digestion and reduce regurgitation (Doxtater, 1963). Whole alimentary canals were then removed in the field and stored in a 10% formalin solution until laboratory identification (Garvey & Chipps, 2012).

We seined at three fixed stations per impoundment concurrent with shoreline electrofishing to capture juvenile largemouth bass once per month in June, August, and October 2017-2020: up-lake (close to the inlet), mid-lake, and down-lake (as close to the dam as possible) in each impoundment with a 4.5 x 1.8 m seine (3.2-mm mesh). We collected a maximum of 10 specimens from each station for gut content analysis. Whole samples were immediately preserved in 10% formalin and kept on ice to slow digestion.

Quantification of Fish Diet Composition

Prey items were identified to the lowest possible taxonomic resolution (usually species or genus but often family depending on extent of digestion) with a dissecting light microscope (4X-10X magnification). The total surface area of each prey item was recorded by counting the number of 1×1 -mm squares covered by each prey item. Prey items were counted, measured, and classified as one of 15 prey categories: empty, algae, anthropogenic material (e.g., fishing tackle, microplastics, trash), crayfish, detritus and sediment, fish, aquatic macroinvertebrate, macrophyte, mollusk, oligochaete, ostracod, tadpole, terrestrial invertebrate, terrestrial plant, and zooplankton. We calculated the frequency of occurrence of fish in the diet (FO_{Fish}) for each length class of largemouth bass for each sampling month within each impoundment (Baker et al., 2014).

Statistical Analysis

For our first objective, we summarized the frequency of occurrence of diet items for the three size classes of largemouth bass across impoundment and months to develop a general description of largemouth bass diets, and we compared the frequency of occurrence of gizzard shad to that of other fish prey in bass diets. For our second objective, we fit a generalized linear mixed effects model to predict the FO_{Fish} across all individual bass diets with month, length

class, and gizzard shad relative abundance as fixed effects and impoundment as a random effect using a binomial distribution with the “glmer” function from the R package *lme4* (Bates et al., 2015). We used the *MuMIN* package to calculate and compare the marginal coefficient of determination (R^2), the proportion of the total variance explained by the fixed effects, to the conditional R^2 , the proportion of the total variance explained by both fixed effects and the random effect of impoundment (Bartón, 2020). For our third objective, we fit a linear model to evaluate how gizzard shad eradication affected FO_{Fish} , with impoundment treatment (rotenone vs. control) and time period (before vs. after rotenone manipulation) as fixed effects. We assumed that there was interannual variation within each impoundment, so our design focused on quantifying the relative differences between responses of largemouth bass piscivory in our treatment and control impoundments. Similar to other before-after-control-impact (BACI) designs (Stewart-Oaten et al., 1986; Conner et al., 2016; Michaletz, 2020; Reynolds & Aldridge, 2021), a significant ($\alpha = 0.10$) interaction between treatment and time period suggests a treatment response to the rotenone manipulation. All analyses were performed in the R statistical environment (version 4.1.0; R Core Team, 2021).

Results

Over four years and 135 sampling events, 2,715 largemouth bass stomach samples were collected and analyzed. Largemouth bass consumption across all length classes and months was dominated by fish, aquatic macroinvertebrates, and crayfish (Table 4.1). Out of all stomach samples collected, 934 (34%) contained fish; 866 (32%) contained aquatic macroinvertebrates, 525 (19%) contained crayfish; and 717 (26%) were empty. Juvenile bass less than 150 mm TL primarily consumed macroinvertebrates (40% FO) and fish (22% FO) (Table 4.2; Figure 4.2A), while diet composition of intermediate length bass was dominated by fish (43% FO), crayfish

(16% FO), and macroinvertebrate matter (11% FO; Table 4.3; Figure 4.2B). Adult bass greater than 250 mm TL consumed crayfish (38% FO), fish (32% FO), and macroinvertebrates (11% FO; Table 4.4; Figure 4.2C). Largemouth bass fish prey largely consisted of bluegill (5.01 % FO_{Fish}), cannibalism of smaller bass (3.17% FO_{Fish}), centrarchid and lepidomid prey (114 total occurrences for a combined 4.20% FO_{Fish}), and gizzard shad (1.22% FO_{Fish}).

The frequency of occurrence of fish in largemouth bass diets was not associated with gizzard shad relative abundance ($P = 0.572$; Table 4.5). However, length class was a significant factor influencing the frequency of occurrence of fish in bass diets in our regression model ($P = 0.004$) with greater piscivory at intermediate lengths (Figure 4.5). In contrast to our hypothesis, sampling month did not affect the frequency of occurrence of piscivory (Table 4.5; Figure 4.3). Finally, gizzard shad eradication did not alter the frequency of occurrence of fish in largemouth bass diets ($P = 0.390$) following rotenone manipulation (Table 4.6; Figure 4.6).

Discussion

Largemouth bass diets in Kansas small impoundments were dominated by fish, crayfish, and macroinvertebrate prey across length categories and months. Of the fish prey consumed, gizzard shad were only a minor component (33 occurrences = 1.22% FO_{Fish}) compared to the frequency of occurrence of bluegill (136 occurrences = 5.01 % FO_{Fish}) or cannibalism (86 occurrences = 3.17% FO_{Fish}) (Figure 4.4). Our regression model indicated that largemouth bass length class was the single greatest factor mediating the FO_{Fish} , and piscivory was greatest for the intermediate size class (Figure 4.5). Body size and predator gape-limit constraints govern predator-prey interactions in aquatic ecosystems because predator-prey interactions among fishes are generally size structured (Werner, 1974; Mittelbach & Persson, 1998; Persson et al., 1996; Gaeta et al., 2018), so it is logical that largemouth bass length was related to the frequency of

occurrence of fish prey in bass diets. However, in contrast with our hypotheses, we observed the highest frequency of occurrence of fish in intermediate length bass between 150-250 mm TL, a finding similar to diet patterns reported by Olson and Young (2003). These intermediate year classes are growing rapidly and require a high caloric diet composed of energetically rich fish prey to sustain their fast growth, whereas larger bass with slower growth rates might be able to meet their energetic demands with crayfish. While largemouth bass diets can exhibit predictable temporal patterns of variation in piscivory rates interannually (Michaletz, 1997) and past studies have reported a greater contribution of gizzard shad to largemouth bass diets in late summer and early fall compared to the spring (Irwin et al., 2003), we did not observe substantial variation in piscivory across months or higher piscivory in August, contrary to our hypotheses.

Given the low frequency of occurrence of shad in bass diets, it is not surprising that our regression model indicated gizzard shad relative abundance was not related to the FO_{Fish} in largemouth bass diets (Table 4.5). Gizzard shad are thought to provide important forage for open water piscivorous fishes such as largemouth bass in reservoir ecosystems (Storck, 1986). Contrary to our hypotheses, we found scant evidence that adult largemouth bass are preying upon shad more frequently than other co-occurring prey fishes such as bluegill, green sunfish, and juvenile largemouth bass. This finding was concordant with diet composition data from five Ohio reservoirs reported by Garvey et al. (2000), which suggested that gizzard shad represented only a relatively large contribution to largemouth bass diets across several reservoirs in one year of their study (potentially due to size-dependent vulnerability and opportune timing of a single abundant year class) but less important than the contributions of other prey fishes such as sunfishes *Lepomis* spp. and minnows *Pimephales* spp. in other years.

In contrast with our hypothesis that shad removal with rotenone would decrease piscivory by largemouth bass, we did not observe a change in the frequency of occurrence of fish in bass diets post-treatment in our rotenone treatment impoundments (Table 4.6). Since the relative contribution of gizzard shad prey to bass diets across all four years of data collection was low compared to the consumption of bluegill or cannibalism on smaller bass (Figure 4.3), fisheries managers in Kansas small impoundments may conclude that selective removal of gizzard shad via rotenone biomanipulation will not meaningfully alter largemouth bass diet composition. An unanticipated result from our diet characterization was that largemouth bass >250 mm TL consumed crayfish prey more frequently (38% FO) than fish prey (32% FO; Table 4.3). We did not directly quantify larval gizzard shad abundances via trawling or crayfish abundances via trapping in this study, which limited our ability to draw inferences on prey selection relative to prey availability in Kansas small impoundments. Crayfish are important members of freshwater stream and lake food webs due to their omnivorous feeding and habitat-modifying behavior, but their trophic role in reservoirs is still poorly resolved (Veselý et al., 2020). Our findings indicate that crayfish can be key components of largemouth bass diets, though the extent to which they are assimilated into tissue compared to fish prey remains unclear and warrants additional examination at a time when non-native crayfishes are rapidly expanding outside their native ranges (Smith et al., 2019).

A large portion (17.97 % FO_{Fish}) of the fish prey consumed by largemouth bass in this study was too digested or too incomplete for positive identification to families or genera. Since different organisms and life stages are digested and evacuated more readily than others (Bromley, 1994; Sutela & Huusko, 2000; Kim & DeVries, 2001), potentially energetically beneficial prey may be underestimated (Bromley, 1994). Indeed, studies have documented high

predation rates by adult fish on fish larvae despite finding insubstantial amounts of fish larvae in adult diet samples (Bromley, 1994), likely because fish larvae are rapidly digested by piscivores (often losing all identifiable characteristics 30 to 60 min post-ingestion; Schooley et al., 2008; Legler et al. 2010) and therefore underestimated in the field, so our methods could underestimate the true contribution of age-0 gizzard shad to largemouth bass diets. Gizzard shad digest faster than bluegill because the heavy bones and scale structure of bluegill can hinder physical access of pepsin to fish proteins (Brugger, 1992); thus, gizzard shad could have been less likely to be positively identified than bluegill. Fish consuming small, particulate meals evacuate their guts more rapidly, allowing them to feed continuously (Garvey & Chipps, 2012), whereas a piscivore may take 24 hours or more to evacuate a large prey fish (Gerking, 1994), and evacuation rates can be dependent on a variety of factors like water temperature (Hedden et al., 2020; Volkoff & Rønnestad, 2020). Furthermore, consumption does not necessarily lead to assimilation, so stomach contents that are high in indigestible carbohydrates such as chitin or cellulose (e.g., crayfish) may pass through the gut more slowly but with a smaller contribution to the energy budget than a prey fish (Garvey & Chipps, 2012). As such, we acknowledge that the examination of diets collected at several points in time and space provides a coarse snapshot of diet information because consumption and evacuation rates can vary.

Despite these caveats, the descriptive bass diet data we present may still provide useful information about the relative contribution of different prey items to the diets of piscivorous fishes such as largemouth bass in Kansas small impoundments. Our data suggest that gizzard shad might be less important prey fish for largemouth bass than has been reported in previous bass diet literature from impoundments in Illinois (Storck, 1986), Missouri (Michaletz, 1997), or Alabama (Irwin et al., 2003), an incongruence that could stem from a range of factors such as

climatic differences that affect the timing of shad spawning and growth, or differences in habitats or food web structure that dampen the intensity of bass predation on gizzard shad. Future predation studies both in natural ecosystems and in experimental foraging arenas (mesocosms) that quantify the extent to which prey selection by predators like largemouth bass is influenced by interactions between prey availability and prey movement at multiple spatial scales (sensu Bruckerhoff et al. 2021) may help further elucidate the drivers of variation in largemouth bass diets and foraging behavior across individuals and populations.

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Table 4.1 Summary of largemouth bass diet composition collected from Kansas small impoundments across all length classes and sampling months.

Prey Type	Number of Stomachs with Occurrences	% FO
Fish	934	34.40
Aquatic Macroinvertebrate	866	31.90
Crayfish	525	19.34
Zooplankton	232	8.55
Detritus & Sediment	167	6.15
Macrophyte	98	3.61
Algae	57	2.09
Terrestrial Invertebrate	20	0.74
Anthropogenic Material	18	0.66
Terrestrial Plant	11	0.41
Mollusk	10	0.37
Ostracod	10	0.37
Oligochaete	6	0.22
Tadpole	1	0.00
Empty Stomach	717	26.41
Total Number of Bass Stomachs Collected	2,715	

Table 4.2 Frequency of occurrence of diet items in largemouth bass diet samples of individuals less than 150 mm TL. Samples are organized by impoundment, month, and year. Gizzard shad catch per unit effort (#/minute electrofishing) was calculated based on data from daytime shoreline electrofishing collected concurrently with diet samples.

Impoundment	Gizzard Shad Catch per Unit Effort (#/ minute electrofishing)	Month	Year	Empty	Algae	Anthropogenic Material	Crayfish	Detritus & Sediment	Fish	Macroinvertebrate	Macrophyte	Mollusca	Oligochaeta	Ostracod	Tadpole	Terrestrial Invertebrate	Terrestrial Plant
Butler	0.20	June	2017	0.42	0.08	--	--	--	0.33	0.33	--	--	--	--	--	--	--
Butler	1.78	June	2018	--	0.07	--	0.04	--	0.26	0.70	--	--	--	--	--	--	--
Butler	0.50	June	2019	--	--	--	--	--	--	0.25	--	--	--	--	--	--	--
Butler	0.87	June	2020	0.17	--	--	--	--	--	0.33	--	--	--	--	--	--	--
Butler	2.58	August	2017	0.17	--	--	--	--	0.25	0.50	--	--	--	--	--	--	--
Butler	0.30	August	2018	0.33	--	--	--	--	--	0.33	--	--	--	--	--	--	--
Butler	0.95	August	2019	1.00	--	--	--	--	--	--	--	--	--	--	--	--	--
Butler	0.28	August	2020	--	--	--	--	--	1.00	--	--	--	--	--	--	--	--
Butler	1.20	October	2017	0.33	--	--	--	--	--	0.67	--	--	--	--	--	--	--
Clark	3.87	June	2017	0.50	--	--	--	--	0.50	--	--	--	--	--	--	--	--
Clark	NA	June	2018	0.30	--	--	--	--	--	0.40	--	--	--	--	--	--	--
Clark	1.31	August	2017	0.33	--	--	--	--	0.50	0.17	--	--	--	--	--	--	--
Clark	0.54	August	2018	0.33	--	--	--	0.17	0.25	0.33	--	--	--	--	--	0.08	--
Clark	4.28	October	2017	--	--	--	--	--	0.33	0.67	--	--	--	--	--	--	--
Clark	1.96	October	2018	0.20	--	--	--	--	0.40	0.80	--	--	--	--	--	--	--
Cowley	0	June	2017	0.22	--	--	0.09	0.17	0.17	0.39	0.04	--	0.04	--	--	--	--
Cowley	0	June	2018	0.23	--	--	--	--	0.31	--	--	--	--	--	--	--	--
Cowley	0	June	2019	--	0.17	--	0.67	--	0.33	0.50	--	--	--	--	--	--	--
Cowley	0.40	June	2020	0.09	--	--	--	--	0.09	0.82	--	--	--	--	--	--	--

Cowley	0	August	2017	0.33	--	--	--	0.11	0.37	0.19	--	--	--	--	--	--	0.04
Cowley	0	August	2018	0.09	--	--	--	--	0.09	0.82	--	--	--	--	--	--	--
Cowley	0	August	2019	0.14	--	--	--	--	0.29	0.57	--	--	--	--	--	0.07	--
Cowley	0	October	2017	0.50	--	--	--	--	--	0.33	--	--	--	--	--	--	--
Cowley	0.02	October	2018	0.20	--	--	--	0.20	0.40	0.40	--	--	--	--	--	--	--
Cowley	0	October	2019	0.50	--	--	--	--	0.25	0.13	--	--	--	--	--	--	--
Cowley	0.65	October	2020	0.27	--	--	--	--	--	0.45	0.18	--	--	--	--	--	--
Douglas	0	June	2017	0.30	--	--	0.10	--	0.10	0.50	--	--	--	0.10	--	--	--
Douglas	0	June	2018	--	--	--	--	--	0.10	0.10	--	--	--	--	--	--	--
Douglas	0	June	2019	0.27	--	--	--	--	0.45	0.09	0.09	--	--	--	--	--	--
Douglas	1.11	June	2020	0.38	--	--	--	--	0.38	0.38	--	--	--	--	--	--	--
Douglas	NA	August	2017	0.09	--	--	--	--	--	0.73	--	--	--	--	--	--	--
Douglas	0	August	2018	0.08	0.08	--	--	0.08	0.31	0.54	--	--	--	--	--	--	--
Douglas	0.47	August	2019	0.33	--	--	--	--	0.50	0.08	0.08	--	--	--	--	--	--
Douglas	NA	October	2017	0.50	--	--	--	--	--	0.50	--	--	--	--	--	--	--
Douglas	1.11	October	2019	0.18	--	--	--	0.27	0.27	0.27	0.18	--	--	--	--	--	--
Geary	0.23	June	2017	0.67	--	--	--	--	0.33	--	--	--	--	--	--	--	--
Geary	0.37	June	2018	0.33	--	--	--	0.33	0.67	0.33	--	--	--	--	--	--	--
Geary	0	June	2019	0.44	--	--	--	--	0.11	--	--	--	--	--	--	--	--
Geary	0.14	June	2020	--	--	--	--	--	0.25	0.13	--	--	--	--	--	--	--
Geary	1.23	August	2017	0.21	--	--	--	--	--	0.71	--	--	--	--	--	0.07	--
Geary	0.08	August	2018	0.14	--	--	--	--	--	0.86	--	--	--	--	--	--	--
Geary	0.88	August	2019	0.29	--	--	--	0.07	0.29	0.18	--	--	--	--	--	--	--
Geary	0	October	2017	0.31	--	--	--	--	0.13	0.50	0.06	--	--	--	--	--	--
Geary	2.77	October	2019	0.25	--	--	0.08	0.17	0.17	0.33	0.08	--	--	--	--	--	--
Meade	0	June	2017	--	0.14	--	0.14	--	0.14	0.71	--	--	--	--	--	--	--
Meade	0	June	2018	--	--	--	--	--	0.06	0.50	0.06	--	--	--	--	--	0.06
Meade	0	June	2019	0.50	--	--	--	--	--	--	--	--	--	--	--	--	--
Meade	0	June	2020	--	--	--	--	--	1.00	--	--	--	--	--	--	--	--
Meade	0	August	2018	0.50	--	--	--	--	--	0.50	--	--	--	--	--	--	--
Meade	0	August	2019	0.22	--	--	--	--	0.22	0.38	--	--	--	--	--	0.11	--
Meade	0	October	2017	0.30	--	--	--	--	0.10	0.50	--	--	--	--	--	--	--

Meade	0	October	2018	1.00	--	--	--	--	--	--	--	--	--	--	--	--	--
Meade	0	October	2019	0.20	--	--	--	0.20	0.40	0.20	--	--	--	--	--	--	--
Neosho	2.23	June	2017	0.08	0.08	--	--	--	--	0.58	--	--	--	0.08	--	--	--
Neosho	1.03	June	2018	0.33	--	--	--	0.08	0.50	--	--	--	--	0.08	--	--	--
Neosho	0	June	2019	0.14	--	--	--	0.14	0.29	0.57	--	--	--	--	--	--	--
Neosho	0.88	June	2020	0.27	--	--	--	--	0.09	0.36	--	--	--	--	--	--	--
Neosho	0.57	August	2018	--	--	--	--	--	0.10	0.90	--	--	--	--	--	--	--
Neosho	1.00	August	2019	0.50	--	--	--	0.25	--	0.25	--	--	--	--	--	--	--
Neosho	2.51	August	2020	0.27	--	--	--	--	0.64	0.09	--	--	--	--	--	--	--
Neosho	0.15	October	2017	--	--	--	--	--	--	1.00	--	--	--	--	--	--	--
Neosho	0.62	October	2018	0.05	0.11	--	--	--	0.05	0.68	0.05	--	--	--	--	--	--
Neosho	0.21	October	2019	0.21	--	--	--	--	0.50	0.43	0.07	--	--	--	--	--	--
Neosho	4.20	October	2020	1.00	--	--	--	--	--	--	--	--	--	--	--	--	--
Ottawa	9.51	June	2017	0.32	--	--	--	--	--	0.16	--	--	--	--	--	--	--
Ottawa	1.12	June	2018	0.33	--	--	--	0.17	--	0.83	--	--	--	--	--	--	--
Ottawa	0	June	2019	--	--	--	--	--	--	1.00	--	--	--	--	--	--	--
Ottawa	0.55	June	2020	--	--	--	--	--	--	1.00	--	--	--	--	--	--	--
Ottawa	9.51	August	2017	--	--	--	--	--	0.20	0.80	--	--	--	--	--	--	--
Ottawa	3.33	August	2019	--	--	--	--	0.20	--	0.80	--	--	--	--	--	--	--
Ottawa	1.12	October	2017	--	--	--	--	--	--	1.00	--	--	--	--	--	--	--
Ottawa	0.45	October	2019	--	--	--	--	--	--	0.90	--	--	--	--	--	0.50	--
Pott #1	0	June	2017	0.33	--	--	--	0.14	0.29	0.38	--	--	--	0.14	--	--	--
Pott #1	0.23	June	2018	0.07	--	--	--	--	0.29	0.64	--	--	--	--	--	--	--
Pott #1	0	June	2019	0.09	--	--	--	0.27	0.18	0.27	--	--	--	--	--	--	--
Pott #1	0	June	2020	0.30	--	--	--	0.30	0.20	0.20	0.10	--	--	--	--	--	--
Pott #1	0.23	August	2017	0.33	--	--	--	--	0.25	0.42	--	--	--	--	--	--	--
Pott #1	0.22	August	2018	0.58	0.08	--	--	0.17	0.25	0.17	--	--	--	--	--	--	--
Pott #1	0	August	2019	0.06	--	--	0.13	--	0.50	0.44	--	--	--	--	--	--	--
Pott #1	0	August	2020	0.14	--	--	--	--	0.36	0.57	0.07	--	--	--	--	--	--
Pott #1	0.37	October	2017	0.17	--	0.08	--	0.25	0.33	0.08	0.17	--	--	--	--	--	--
Pott #1	0	October	2019	0.50	--	--	--	--	0.50	--	--	--	--	--	--	--	--
Scott	0	June	2017	0.29	--	--	--	--	--	0.36	--	--	--	--	--	--	--

Scott	0	June	2018	0.11	0.05	--	--	--	0.26	0.63	--	--	--	--	--	--	--
Scott	0	June	2019	0.20	--	--	--	0.30	--	0.60	--	--	--	--	--	--	--
Scott	0	June	2020	--	--	--	--	--	--	1.00	--	--	--	--	--	--	--
Scott	0	August	2017	0.54	--	--	--	--	0.38	0.08	--	--	--	--	--	--	--
Scott	0	August	2018	--	--	--	--	--	--	1.00	--	--	--	--	--	--	--
Scott	0	August	2019	0.40	--	--	--	0.20	--	0.40	--	--	--	--	--	--	--
Scott	0	October	2017	0.40	--	--	--	--	0.20	0.30	0.10	--	--	--	--	--	--
Scott	0	October	2018	--	--	--	1.00	--	--	--	--	--	--	--	--	--	--
Scott	0	October	2019	0.44	--	--	--	--	0.33	0.33	--	--	--	--	--	--	--
Shawnee	1.70	June	2017	0.07	--	--	0.13	0.07	0.07	0.47	--	--	0.07	--	--	--	--
Shawnee	1.43	June	2018	0.14	--	--	--	--	0.71	0.29	--	--	--	--	--	--	--
Shawnee	0	June	2020	--	--	--	0.08	0.25	0.42	0.25	--	--	--	--	--	--	--
Shawnee	8.47	August	2017	0.20	--	--	--	--	0.60	0.40	--	--	--	--	--	--	--
Shawnee	0.66	August	2018	0.50	--	--	--	--	--	0.25	--	--	--	--	--	--	--
Shawnee	0	August	2019	0.40	0.20	--	--	--	0.20	0.60	--	--	--	--	--	--	--
Shawnee	0	August	2020	--	--	--	--	--	1.00	--	--	--	--	--	--	--	--
Shawnee	3.30	October	2017	0.50	--	--	--	--	0.50	--	--	--	--	--	--	--	--
Shawnee	0	October	2019	0.31	--	--	0.06	0.06	0.13	0.38	0.13	--	--	--	--	--	--
Shawnee	0	October	2020	1.00	--	--	--	--	--	--	--	--	--	--	--	--	--
Washington	0.60	June	2017	0.67	--	--	0.17	--	0.17	--	--	--	--	--	--	--	--
Washington	0.10	June	2018	--	0.08	--	--	0.08	0.46	0.38	0.08	0.08	--	--	--	0.15	--
Washington	0	June	2019	0.30	--	--	0.20	0.10	0.20	0.30	--	--	--	--	--	--	--
Washington	0.08	June	2020	0.50	--	--	0.25	--	0.25	0.25	--	--	--	--	--	--	--
Washington	1.20	August	2017	--	--	--	--	--	0.27	0.91	--	--	--	--	--	--	--
Washington	1.19	August	2018	0.18	--	--	--	--	0.09	0.73	--	--	--	--	--	--	--
Washington	0	August	2019	0.25	--	--	0.25	--	0.25	0.25	0.25	--	--	--	--	--	--
Washington	0.04	October	2017	0.25	0.25	--	0.25	--	0.25	--	--	--	--	--	--	--	--
Washington	1.49	October	2018	--	--	--	--	--	--	1.00	--	--	--	--	--	--	--
Washington	0	October	2019	0.43	--	--	0.14	--	0.57	--	--	--	--	--	--	--	--
Mean Frequency of Occurrence				0.25	0.01	--	0.03	0.04	0.22	0.40	0.02	--	--	--	--	0.01	--

Table 4.3 Frequency of occurrence of diet items in largemouth bass diet samples of individuals between 150 and 250 mm TL. Samples are organized by impoundment, month, and year. Gizzard shad catch per unit effort (#/minute electrofishing) was calculated based on data from daytime shoreline electrofishing collected concurrently with diet samples.

Impoundment	Gizzard Shad Catch per Unit Effort (#/ minute electrofishing)	Month	Year	Empty	Algae	Anthropogenic Material	Crayfish	Detritus & Sediment	Fish	Macroinvertebrate	Macrophyte	Mollusca	Oligochaeta	Ostracod	Tadpole	Terrestrial Invertebrate	Terrestrial Plant
Butler	0.20	June	2017	--	--	--	1.00	--	--	--	--	--	--	--	--	--	--
Butler	1.78	June	2018	0.11	--	0.11	0.33	--	0.33	0.22	--	--	0.11	--	--	--	--
Butler	0.50	June	2019	0.25	--	--	0.25	--	0.50	--	--	--	--	--	--	--	--
Butler	2.58	August	2017	0.50	--	--	0.17	--	0.17	0.33	--	--	--	--	--	--	--
Butler	0.30	August	2018	0.40	--	--	--	0.20	0.20	0.20	--	--	--	--	--	0.20	--
Butler	0.95	August	2019	0.40	--	--	--	--	0.40	0.20	0.20	--	--	--	--	--	--
Butler	0.28	August	2020	0.36	--	--	0.18	0.18	0.27	--	--	--	--	--	--	--	--
Butler	1.20	October	2017	--	--	--	--	0.33	0.33	--	--	--	--	--	--	--	--
Butler	0.18	October	2019	--	--	--	0.50	--	0.50	--	--	--	--	--	--	--	--
Clark	3.87	June	2017	0.57	--	--	0.07	--	0.14	0.36	--	--	--	--	--	--	--
Clark	1.31	August	2017	0.55	--	--	--	--	0.45	--	--	--	--	--	--	--	--
Clark	0.54	August	2018	0.14	--	--	--	--	0.71	0.14	0.14	--	--	--	--	--	--
Clark	4.28	October	2017	--	--	--	--	0.14	0.43	0.57	--	--	--	--	--	--	--
Clark	1.96	October	2018	0.38	--	--	0.13	0.25	0.38	0.25	--	--	--	--	--	--	--
Cowley	0	June	2017	0.13	0.13	--	0.25	0.13	0.38	--	0.13	--	--	--	--	--	--
Cowley	0	June	2018	0.33	--	--	0.33	--	0.33	--	--	--	--	--	--	--	--
Cowley	0	June	2019	0.57	--	--	--	0.29	0.29	--	--	--	--	--	--	--	--

Cowley	0.40	June	2020	0.50	--	--	0.50	0.50	0.50	--	--	--	--	--	--	--	--
Cowley	0	August	2017	0.22	--	--	0.33	--	0.44	--	--	--	--	--	--	--	--
Cowley	0	August	2018	0.71	0.14	--	--	--	0.14	--	--	--	--	--	--	--	--
Cowley	0.12	August	2019	0.40	--	--	--	0.20	0.20	--	0.20	--	--	--	--	--	--
Cowley	0.53	August	2020	--	--	--	--	0.67	0.67	--	0.33	--	--	--	--	--	--
Cowley	0	October	2017	0.36	--	--	0.09	--	0.36	0.27	--	--	--	--	--	--	--
Cowley	0.02	October	2018	0.50	--	--	--	--	0.50	--	--	--	--	--	--	--	--
Cowley	0	October	2019	0.17	--	--	--	--	0.83	--	0.17	--	--	--	--	--	--
Cowley	0.65	October	2020	0.78	--	--	--	--	0.11	0.11	--	--	--	--	--	--	--
Douglas	0	June	2017	0.40	--	--	--	--	0.60	--	0.20	--	--	--	--	--	--
Douglas	0	June	2018	--	--	--	--	1.00	1.00	0.14	0.43	--	--	--	--	--	--
Douglas	0	June	2019	--	--	--	--	0.14	0.86	--	--	--	--	--	--	--	--
Douglas	1.11	June	2020	--	--	--	0.13	--	0.88	0.13	--	--	--	--	--	--	--
Douglas	0	August	2017	0.13	--	--	0.13	--	0.75	--	--	--	--	--	--	--	--
Douglas	0.47	August	2019	0.22	--	--	--	--	0.67	0.22	--	--	--	--	--	--	--
Douglas	0.45	August	2020	0.40	--	--	--	0.20	0.40	--	0.40	--	--	--	--	--	--
Douglas	0	October	2017	0.50	--	--	--	0.13	0.50	--	--	--	--	--	--	--	--
Douglas	0.24	October	2018	--	--	--	--	0.50	0.50	0.50	--	--	--	--	--	--	--
Douglas	1.11	October	2019	0.29	--	--	--	0.29	0.57	--	0.14	--	--	--	--	--	--
Geary	0.23	June	2017	0.25	--	--	0.75	--	--	--	--	--	--	--	--	--	--
Geary	0.37	June	2018	0.33	--	--	0.33	--	0.33	--	--	--	--	--	--	--	--
Geary	0.14	June	2020	--	--	--	--	1.00	--	--	--	--	--	--	--	--	--
Geary	0.08	August	2018	0.33	0.33	--	--	0.33	--	0.33	--	--	--	--	--	--	--
Geary	0	October	2017	0.33	--	--	--	--	0.67	--	--	--	--	--	--	--	--
Geary	2.77	October	2019	--	--	--	0.33	--	0.50	0.17	0.17	--	--	--	--	--	--
Geary	0	October	2020	0.20	--	--	0.10	--	0.70	0.20	--	0.20	--	--	--	--	--
Meade	00	June	2017	0.17	--	--	0.17	0.17	0.33	0.17	--	--	--	--	--	--	--
Meade	0	June	2018	--	--	--	--	--	1.00	--	--	--	--	--	--	--	--
Meade	0	June	2019	--	--	--	1.00	--	--	--	--	--	--	--	--	--	--
Meade	0	June	2020	--	--	--	0.50	--	0.50	--	--	--	--	--	--	--	--
Meade	0	August	2018	0.13	--	--	0.25	--	0.63	--	--	--	--	--	--	--	--
Meade	0	August	2019	0.25	--	--	--	--	0.50	--	0.25	--	--	--	--	--	--

Meade	0	October	2017	0.67	--	--	--	--	0.33	--	--	--	--	--	--	--	--
Meade	0	October	2019	0.50	--	--	--	--	0.50	--	--	--	--	--	--	--	--
Neosho	2.23	June	2017	0.44	--	--	0.11	--	0.44	--	--	--	--	--	--	--	--
Neosho	1.03	June	2018	--	0.50	--	--	1.00	1.00	--	0.50	--	--	--	--	--	--
Neosho	0	June	2019	0.50	--	--	--	--	0.50	--	--	--	--	--	--	--	--
Neosho	0.88	June	2020	--	--	--	0.22	0.11	0.56	0.22	--	--	--	--	--	--	--
Neosho	4.80	August	2017	0.33	0.11	--	--	0.11	0.56	--	--	--	--	--	--	--	--
Neosho	0.57	August	2018	--	--	--	--	1.00	--	--	--	--	--	--	--	--	--
Neosho	1.00	August	2019	0.60	--	--	--	0.20	0.20	--	0.40	--	--	--	--	--	--
Neosho	2.51	August	2020	--	--	--	--	--	1.00	--	0.14	--	--	--	--	--	--
Neosho	0.15	October	2017	0.67	--	--	--	--	0.33	--	--	--	--	--	--	--	0.17
Neosho	0.21	October	2019	0.44	--	--	0.22	0.11	0.22	--	--	--	--	--	--	--	--
Ottawa	0.10	June	2017	--	--	--	1.00	--	--	--	--	--	--	--	--	--	--
Ottawa	1.12	June	2018	0.33	0.17	--	0.33	--	0.33	0.83	--	--	--	--	--	--	0.17
Ottawa	0.68	August	2018	--	--	--	--	--	1.00	--	--	--	--	--	--	--	--
Ottawa	0.03	August	2019	--	--	--	1.00	--	--	--	--	--	--	--	--	--	--
Ottawa	3.33	October	2017	--	--	--	--	--	1.00	--	--	--	--	--	--	--	--
Ottawa	0.45	October	2019	--	--	--	--	--	1.00	--	--	--	--	--	--	--	--
Pott #1	0	June	2017	0.33	--	--	0.17	--	0.50	0.17	--	--	--	--	--	--	--
Pott #1	0.30	June	2018	--	--	0.20	0.20	--	0.60	0.20	0.40	--	--	--	--	--	--
Pott #1	0	June	2019	0.64	--	--	--	--	0.27	0.27	--	--	--	--	--	--	--
Pott #1	0	June	2020	0.30	--	--	--	0.20	0.40	0.10	--	--	--	--	--	--	--
Pott #1	0.23	August	2017	0.57	--	--	0.14	--	0.29	--	--	--	--	--	--	--	--
Pott #1	0.22	August	2018	0.63	0.13	--	0.13	0.13	--	--	0.13	--	--	--	--	--	--
Pott #1	0	August	2019	0.21	--	--	0.07	0.07	0.50	0.14	0.07	--	--	--	--	--	--
Pott #1	0.37	October	2017	0.33	0.11	--	--	0.22	0.44	--	--	--	--	--	0.11	--	--
Pott #1	0	October	2019	0.22	--	--	--	0.11	0.44	0.22	--	--	--	--	--	--	--
Scott	0	June	2017	0.43	--	--	0.14	--	0.43	--	--	--	--	--	--	--	--
Scott	0	June	2018	--	--	--	--	--	1.00	--	--	--	--	--	--	--	--
Scott	0	June	2019	--	--	--	0.25	--	0.75	0.75	--	--	--	--	--	--	--
Scott	0	June	2020	0.50	--	--	--	--	--	0.50	--	--	--	--	--	--	--
Scott	0	August	2017	0.17	--	--	--	--	0.83	--	--	--	--	--	--	--	--

Scott	0	August	2018	0.33	--	--	0.67	--	--	--	--	--	--	--	--	--	--
Scott	0	October	2017	0.62	--	--	--	0.08	0.15	0.23	--	--	--	--	--	--	--
Scott	0	October	2018	0.25	--	--	--	--	0.25	0.50	--	--	--	--	--	--	--
Scott	0	October	2019	0.29	--	--	--	0.14	0.29	0.43	--	--	--	--	--	--	--
Scott	0	October	2020	0.50	--	--	--	0.25	0.25	--	--	--	--	--	--	--	--
Shawnee	1.70	June	2017	0.60	--	--	0.20	--	--	0.20	--	--	--	--	--	--	--
Shawnee	1.43	June	2018	0.14	0.14	--	0.29	--	0.86	--	0.14	0.14	--	0.14	--	--	--
Shawnee	0	June	2020	0.31	--	--	0.38	0.15	0.15	0.31	0.08	--	--	--	--	--	--
Shawnee	8.47	August	2017	--	--	--	--	--	0.67	0.33	--	--	--	--	--	--	--
Shawnee	0.66	August	2018	0.50	--	--	--	--	0.50	--	--	--	--	--	--	--	--
Shawnee	0	August	2019	--	--	--	0.50	--	0.50	--	0.50	--	--	--	--	--	--
Shawnee	0	August	2020	0.40	--	--	0.20	--	0.50	--	--	--	--	--	--	--	--
Shawnee	3.30	October	2017	0.40	--	--	--	--	0.60	--	--	--	--	--	--	--	--
Shawnee	0	October	2019	0.43	--	--	0.43	0.14	--	--	--	--	--	--	--	--	--
Shawnee	0	October	2020	--	--	--	0.50	--	0.50	--	--	--	--	--	--	--	--
Washington	0.60	June	2017	0.50	--	--	0.33	0.17	--	--	--	--	--	--	--	--	--
Washington	0.10	June	2018	--	0.60	--	--	0.20	0.40	0.80	--	--	--	--	--	--	0.20
Washington	0	June	2019	0.10	--	--	0.50	--	0.20	0.20	--	--	--	--	--	0.10	--
Washington	0.08	June	2020	0.18	--	--	0.18	--	0.18	0.45	0.09	--	--	--	--	--	--
Washington	1.19	August	2018	0.80	--	--	--	--	0.20	--	--	--	--	--	--	--	--
Washington	0	August	2019	0.20	--	--	0.40	--	0.40	--	--	--	--	--	--	--	--
Washington	0.25	August	2020	--	--	--	0.50	--	0.50	--	--	--	--	--	--	--	--
Washington	0.04	October	2017	0.40	--	--	0.40	--	0.40	0.20	--	--	--	--	--	--	--
Washington	0	October	2019	0.33	--	--	--	--	0.50	0.17	--	0.17	--	--	--	--	--
Mean Frequency of Occurrence				0.27	0.02	--	0.16	0.11	0.43	0.11	0.05	--	--	--	--	--	0.01

Table 4.4 Frequency of occurrence of diet items in largemouth bass diet samples of individuals greater than 250 mm TL. Samples are organized by impoundment, month, and year. Gizzard shad catch per unit effort (#/minute electrofishing) was calculated based on data from daytime shoreline electrofishing collected concurrently with diet samples.

Impoundment	Gizzard Shad Catch per Unit Effort (#/ minute electrofishing)	Month	Year	Empty	Algae	Anthropogenic Material	Crayfish	Detritus & Sediment	Fish	Macroinvertebrate	Macrophyte	Mollusca	Oligochaeta	Ostracod	Tadpole	Terrestrial Invertebrate	Terrestrial Plant
Butler	0.20	June	2017	0.38	--	--	0.52	0.05	0.19	--	--	--	--	--	--	--	--
Butler	1.78	June	2018	0.17	--	--	0.67	--	0.17	0.17	--	--	--	--	--	--	--
Butler	0.50	June	2019	0.44	--	--	0.44	0.11	0.33	0.11	--	--	--	--	--	--	--
Butler	2.58	August	2017	0.25	--	--	0.75	0.13	--	0.13	--	--	--	--	--	--	--
Butler	0.30	August	2018	0.40	--	--	0.40	0.10	0.50	0.30	--	--	--	--	--	--	--
Butler	0.95	August	2019	0.20	--	--	0.60	--	0.40	0.20	--	--	--	--	--	--	--
Butler	0.28	August	2020	0.33	--	0.11	0.33	--	0.11	0.11	--	--	--	--	--	--	--
Butler	1.20	October	2017	0.40	--	--	0.40	--	0.30	--	--	--	--	--	--	--	0.10
Butler	0.18	October	2019	0.10	--	--	0.30	--	1.00	--	--	--	--	--	--	--	--
Clark	3.87	June	2017	0.15	--	0.08	0.62	0.08	0.08	0.15	--	--	0.08	0.08	--	--	--
Clark	1.31	August	2017	0.18	--	--	0.36	--	0.45	0.09	--	--	--	--	--	0.09	--
Clark	0.54	August	2018	0.22	--	--	0.11	--	0.67	--	--	--	--	--	--	--	--
Clark	4.28	October	2017	0.63	--	--	0.13	--	0.25	--	--	--	--	--	--	--	--
Clark	1.96	October	2018	1.00	--	--	--	--	0.50	0.50	--	--	--	--	--	--	--
Cowley	0	June	2017	0.27	0.07	0.07	0.60	--	0.27	--	--	--	--	--	--	--	--
Cowley	0	June	2018	0.70	0.10	--	0.20	0.10	0.60	0.30	--	--	--	--	--	--	--
Cowley	0	June	2019	0.33	--	--	0.50	0.08	0.25	--	--	--	--	--	--	--	--

Cowley	0.40	June	2020	0.50	--	--	0.50	--	--	--	--	--	--	--	--	--	--
Cowley	0	August	2017	0.33	--	--	0.42	--	0.25	--	--	--	--	--	--	0.08	--
Cowley	0	August	2018	0.27	0.09	--	0.36	0.09	0.45	0.09	0.09	--	--	--	--	--	--
Cowley	0.12	August	2019	0.29	--	--	0.50	0.14	0.21	--	--	--	--	--	--	--	--
Cowley	0.53	August	2020	0.20	--	--	0.60	0.10	--	0.10	0.30	--	--	--	--	--	--
Cowley	0	October	2017	0.30	--	--	0.30	--	0.50	0.10	--	--	--	--	--	0.10	--
Cowley	0.02	October	2018	0.50	0.13	--	0.25	--	0.25	0.13	0.13	--	--	--	--	--	--
Cowley	0	October	2019	0.27	--	--	0.45	0.09	0.55	--	--	--	--	--	--	--	--
Cowley	0.65	October	2020	0.55	--	--	0.27	--	0.18	--	--	--	--	--	--	--	--
Douglas	0	June	2017	0.26	--	0.04	0.57	--	0.26	0.04	--	--	--	--	--	--	--
Douglas	0	June	2018	--	0.10	0.10	0.40	0.80	0.30	0.60	0.20	0.10	--	--	--	--	0.10
Douglas	0	June	2019	0.33	--	--	0.25	0.33	0.25	--	0.08	--	--	--	--	--	--
Douglas	1.11	June	2020	--	--	--	0.36	--	0.64	0.09	0.27	--	--	--	--	--	--
Douglas	0	August	2017	0.30	--	--	0.40	--	0.20	--	0.10	--	--	--	--	0.10	--
Douglas	0.47	August	2019	0.27	--	0.09	0.55	--	0.09	--	--	--	--	--	--	--	--
Douglas	0.45	August	2020	0.36	--	--	0.27	0.09	0.27	0.09	0.18	--	--	--	--	--	--
Douglas	0	October	2017	0.30	--	--	0.10	--	0.60	--	--	--	--	--	--	--	--
Douglas	0.24	October	2018	0.50	0.33	--	0.50	--	0.17	--	--	--	--	--	--	--	--
Douglas	1.11	October	2019	0.18	0.09	--	0.36	0.18	0.36	0.09	--	--	--	--	--	--	--
Geary	0.23	June	2017	0.11	--	0.05	0.84	--	0.11	--	--	--	--	--	--	--	--
Geary	0.37	June	2018	0.30	--	--	0.60	0.20	0.10	--	--	--	--	--	--	--	0.10
Geary	0.14	June	2020	0.30	--	--	0.70	--	0.30	--	--	0.10	--	--	--	--	--
Geary	1.23	August	2017	1.00	--	--	--	--	--	--	--	--	--	--	--	--	--
Geary	0.08	August	2018	0.25	0.13	0.13	0.50	--	--	--	--	--	--	--	--	--	--
Geary	0	October	2017	0.33	--	--	0.33	--	0.33	--	--	--	--	--	--	--	--
Geary	2.77	October	2019	0.40	0.20	--	0.20	--	0.20	--	0.20	--	--	--	--	--	--
Geary	0	October	2020	0.22	--	--	0.22	--	0.67	--	--	--	--	--	--	--	--
Meade	0	June	2017	0.29	--	0.07	0.07	--	0.57	--	--	--	--	--	--	--	--
Meade	0	June	2018	0.67	--	--	--	0.22	0.11	0.11	--	--	--	--	--	--	--
Meade	0	June	2019	0.33	--	--	--	--	0.67	0.33	--	--	--	--	--	--	--
Meade	0	June	2020	0.20	--	--	0.50	0.10	0.40	--	--	--	--	--	--	--	--
Meade	0	August	2017	0.40	--	--	0.20	--	0.40	--	--	--	--	--	--	--	--

Meade	0	August	2018	0.45	--	--	0.45	--	0.18	--	--	--	--	--	--	--	--
Meade	0	August	2019	0.50	--	--	0.33	--	0.50	--	--	--	--	--	--	--	--
Meade	0	October	2017	0.30	--	--	0.40	--	0.30	--	--	--	--	--	--	--	--
Meade	0	October	2019	0.25	--	--	--	--	0.75	--	--	--	--	--	--	--	--
Neosho	2.23	June	2017	0.27	0.13	0.13	0.27	0.07	0.27	0.13	--	--	--	0.07	--	--	--
Neosho	1.03	June	2018	0.20	0.30	--	0.30	0.40	0.50	--	--	--	--	--	--	--	--
Neosho	0	June	2019	0.25	--	--	0.25	--	0.50	--	0.13	--	--	--	--	--	--
Neosho	0.88	June	2020	0.20	--	--	0.50	--	0.50	--	--	--	--	--	--	--	--
Neosho	4.80	August	2017	0.50	0.10	--	--	--	0.40	--	--	--	--	--	--	--	--
Neosho	0.57	August	2018	0.70	--	--	--	--	0.50	0.80	--	--	--	--	--	--	--
Neosho	1.00	August	2019	0.23	--	--	--	0.23	0.31	--	0.38	--	--	--	--	--	--
Neosho	2.51	August	2020	0.10	0.10	--	--	--	0.50	0.20	0.40	--	--	--	--	--	--
Neosho	0.15	October	2017	0.27	--	--	0.18	0.09	0.55	--	--	--	--	--	--	--	--
Neosho	0.21	October	2019	0.22	--	--	0.22	--	0.67	0.11	--	--	--	--	--	--	--
Ottawa	0.10	June	2017	0.13	--	--	0.63	--	0.25	0.13	--	--	--	--	--	--	--
Ottawa	1.12	June	2018	0.05	0.05	--	0.58	0.11	0.21	0.26	--	--	--	--	--	0.05	--
Ottawa	0.55	June	2020	0.44	--	--	0.56	--	0.11	--	--	--	--	--	--	--	--
Ottawa	0.68	August	2018	1.00	--	--	--	--	--	--	--	--	--	--	--	--	--
Ottawa	0.03	August	2019	--	--	--	--	--	--	1.00	--	--	--	--	--	--	--
Ottawa	0.45	October	2019	0.33	--	--	0.67	--	0.17	--	--	--	--	--	--	--	--
Pott #1	0	June	2017	0.80	--	--	0.20	--	--	--	--	--	--	--	--	--	--
Pott #1	0.30	June	2018	0.09	--	0.09	0.27	--	0.45	0.55	0.18	--	--	--	--	--	--
Pott #1	0	June	2019	0.30	0.10	--	0.40	--	0.20	0.20	0.10	0.10	--	--	--	--	--
Pott #1	0	June	2020	0.10	--	0.10	0.40	--	0.40	0.10	--	--	--	--	--	--	--
Pott #1	0.23	August	2017	0.54	0.08	--	--	0.08	0.15	0.23	--	--	--	0.08	--	--	--
Pott #1	0.22	August	2018	0.56	--	--	0.22	--	0.22	--	--	--	--	--	--	--	--
Pott #1	0	August	2019	0.20	--	--	0.50	--	0.10	--	0.20	--	--	--	--	--	--
Pott #1	0.37	October	2017	0.45	--	--	0.18	0.18	0.27	--	--	--	--	--	--	--	--
Pott #1	0	October	2019	0.25	--	--	0.38	--	0.25	--	0.25	--	--	--	--	--	--
Scott	0	June	2017	0.25	--	--	0.38	0.04	0.42	0.21	--	--	--	--	--	--	--
Scott	0	June	2019	0.09	0.09	--	0.55	--	0.64	0.64	0.09	--	0.09	--	--	--	--
Scott	0	June	2020	0.20	--	0.10	0.30	--	0.40	0.10	0.10	--	--	--	--	--	--

Scott	0	August	2017	0.50	--	--	--	--	0.50	--	--	--	--	--	--	--	--
Scott	0	August	2018	0.14	0.14	--	0.43	0.14	0.57	0.14	0.29	--	--	--	--	--	--
Scott	0	October	2017	0.50	--	--	0.40	--	0.20	0.10	--	--	--	--	--	--	--
Scott	0	October	2018	--	0.14	--	0.57	--	0.43	0.29	0.14	--	--	--	--	--	--
Scott	0	October	2019	--	--	--	--	--	1.00	--	--	--	--	--	--	--	--
Scott	0	October	2020	0.10	--	--	0.30	--	0.70	--	0.10	--	--	--	--	--	--
Shawnee	1.70	June	2017	0.31	0.08	--	0.46	0.15	0.31	0.15	0.08	--	--	--	--	--	--
Shawnee	1.43	June	2018	--	0.13	--	0.75	0.25	0.25	0.13	--	--	--	--	--	0.13	0.13
Shawnee	0	June	2020	0.09	--	--	0.91	--	0.09	--	--	--	--	--	--	--	--
Shawnee	8.47	August	2017	0.20	--	--	0.60	--	0.20	--	--	--	--	--	--	--	--
Shawnee	0.66	August	2018	0.63	--	--	0.25	--	0.25	--	--	--	--	--	--	--	--
Shawnee	0	August	2019	--	--	--	1.00	--	--	--	--	--	--	--	--	--	--
Shawnee	0	August	2020	0.33	--	--	0.44	--	0.22	--	--	--	--	--	--	--	--
Shawnee	3.30	October	2017	0.44	--	--	0.33	--	0.22	--	--	--	--	--	--	--	--
Shawnee	0	October	2019	--	--	--	0.88	0.13	0.13	0.13	--	--	--	--	--	--	--
Washington	0.60	June	2017	0.25	--	--	0.42	0.08	0.33	0.42	0.08	--	--	--	--	--	0.08
Washington	0.10	June	2018	--	0.13	--	0.13	--	0.25	0.75	0.13	--	--	--	--	0.13	--
Washington	0	June	2019	0.33	--	--	--	0.33	0.33	--	--	--	--	--	--	--	--
Washington	0.08	June	2020	0.13	--	--	0.38	0.13	0.13	0.50	0.13	--	--	--	--	--	--
Washington	1.20	August	2017	--	--	--	--	--	1.00	--	--	--	--	--	--	--	--
Washington	1.19	August	2018	0.43	--	--	0.57	--	--	--	0.14	--	--	--	--	--	--
Washington	0	August	2019	--	--	--	--	--	0.50	0.50	--	--	--	--	--	--	--
Washington	0.25	August	2020	--	--	0.14	1.00	--	0.29	--	--	--	--	--	--	--	--
Washington	0.04	October	2017	0.70	--	--	0.10	--	0.10	0.10	--	--	--	--	--	--	--
Washington	1.49	October	2018	--	--	--	--	--	--	1.00	--	--	--	--	--	--	--
Washington	0	October	2019	0.14	--	--	0.43	0.14	0.71	--	0.29	--	--	--	--	--	0.14
Washington	0.13	October	2020	--	--	--	0.43	--	0.43	0.14	--	--	--	--	--	--	--
Mean Frequency of Occurrence				0.28	0.02	0.01	0.38	0.05	0.32	0.11	0.04	--	--	--	--	0.01	0.01

Tables

Table 4.5 Generalized linear mixed effects model fit by maximum likelihood output to predict FO_{Fish} for largemouth bass across size classes, months, and impoundments with different relative abundances of gizzard shad.

Model	<i>R</i> ² Marginal; <i>R</i> ² Conditional	Fixed Effect or Interaction Term	Estimate	95% CI	Std. Error	df	t value	p-value
lmer(FO _{Fish} ~ month + length class + shad CPE + (1 impoundment))	(0.032; 0.043)	Month - June	0.004	-0.062; 0.069	0.033	319.0	0.113	0.910
		Month - Oct	0.005	-0.022; 0.117	0.035	321.7	1.341	0.181
		Length Class	0.005	0.016; 0.082	0.017	317.8	2.926	0.004*
		Gizzard Shad CPE	< 0.001	-0.015; 0.028	0.012	1.910	0.566	0.572

Table 4.6 Linear fixed effect model outputs for the frequency of occurrence (FO) of fish in largemouth bass diets in response to rotenone treatments. Asterisks denote statistical significance at $\alpha = 0.1$.

Model	Fixed Effect or Interaction Term	Estimate	95% CI	Std. Error	df	t value	p-value
FO _{Fish} ~ Treatment * Time Period	Treatment	0.027	-0.102; 0.157	0.067	128.43	0.408	0.683
	Time Period	-0.044	-0.159; 0.073	0.061	4.93	-0.713	0.508
	Treatment x Time Period	0.078	0.099; 0.254	0.091	128.10	0.862	0.390

Figures

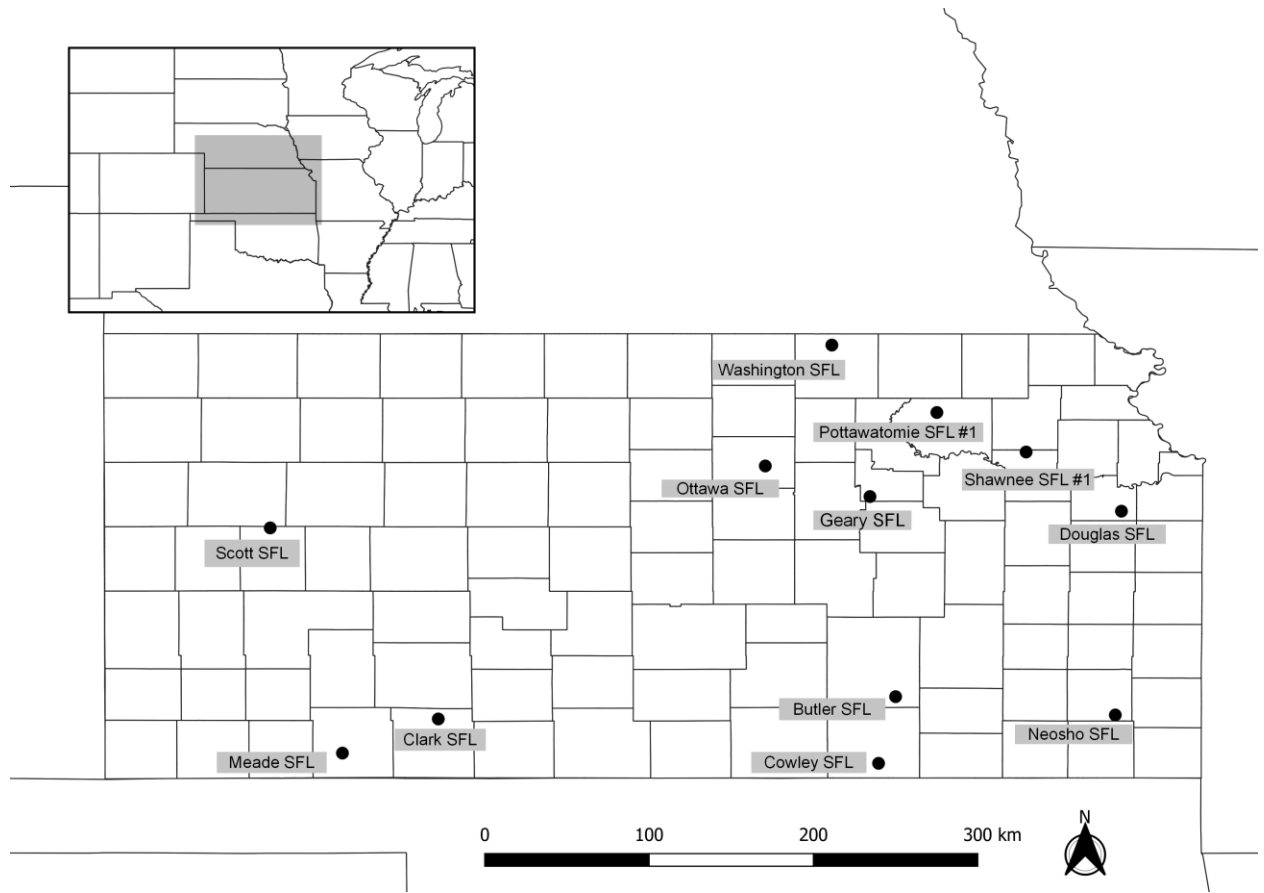


Figure 4.1 Map of 12 study impoundments sampled for largemouth bass diet information from 2017-2020. SFL = State Fishing Lake. All impoundments are managed for angling recreation by the Kansas Department of Wildlife & Parks (KDWP).

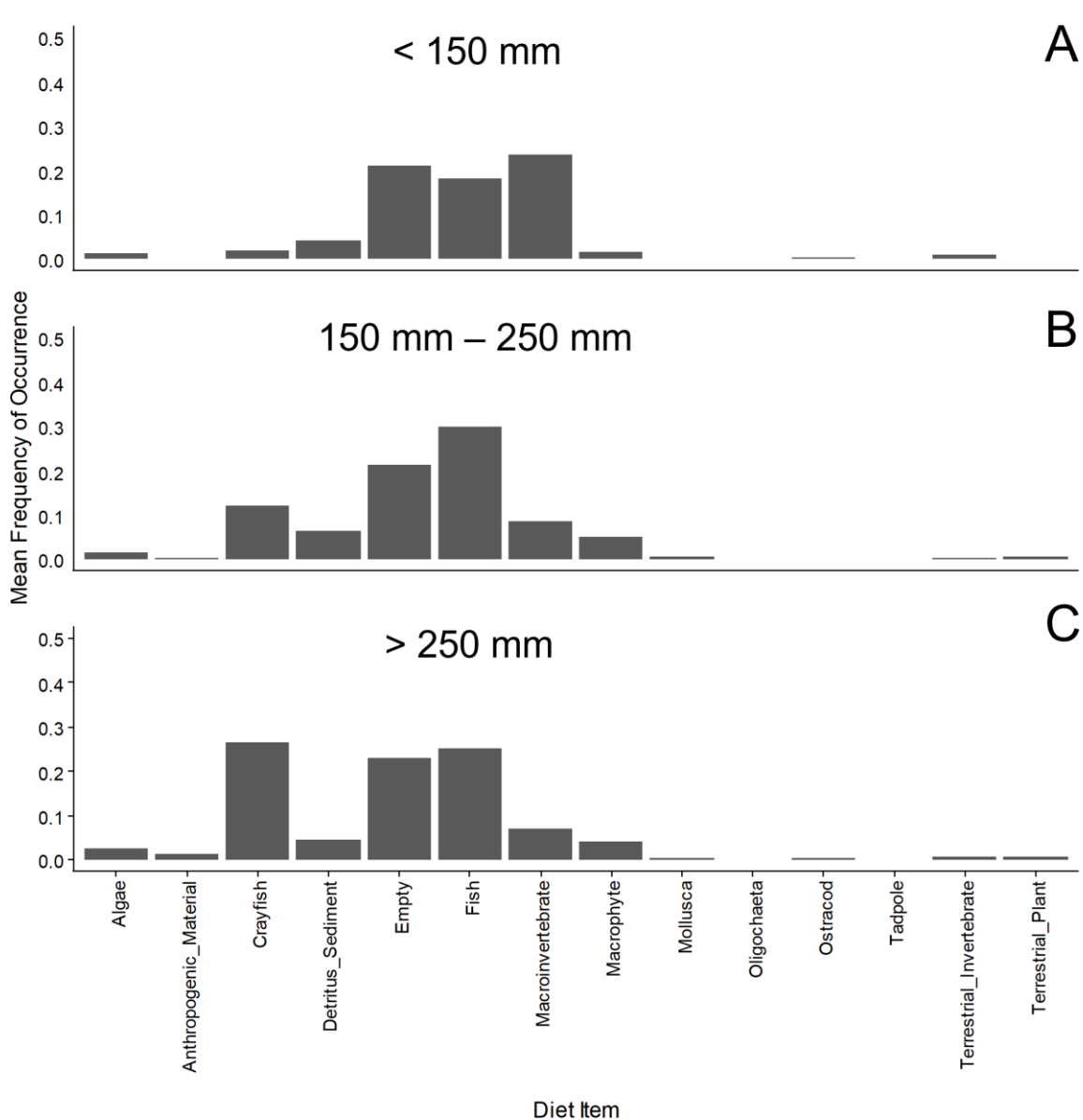


Figure 4.2 Mean frequency of occurrence of prey items in largemouth bass diets by length class. Largemouth bass were collected from three length classes: individuals < 150 mm TL (top), individuals between 150 - 250 mm TL (middle), and individuals > 250 mm TL (bottom).

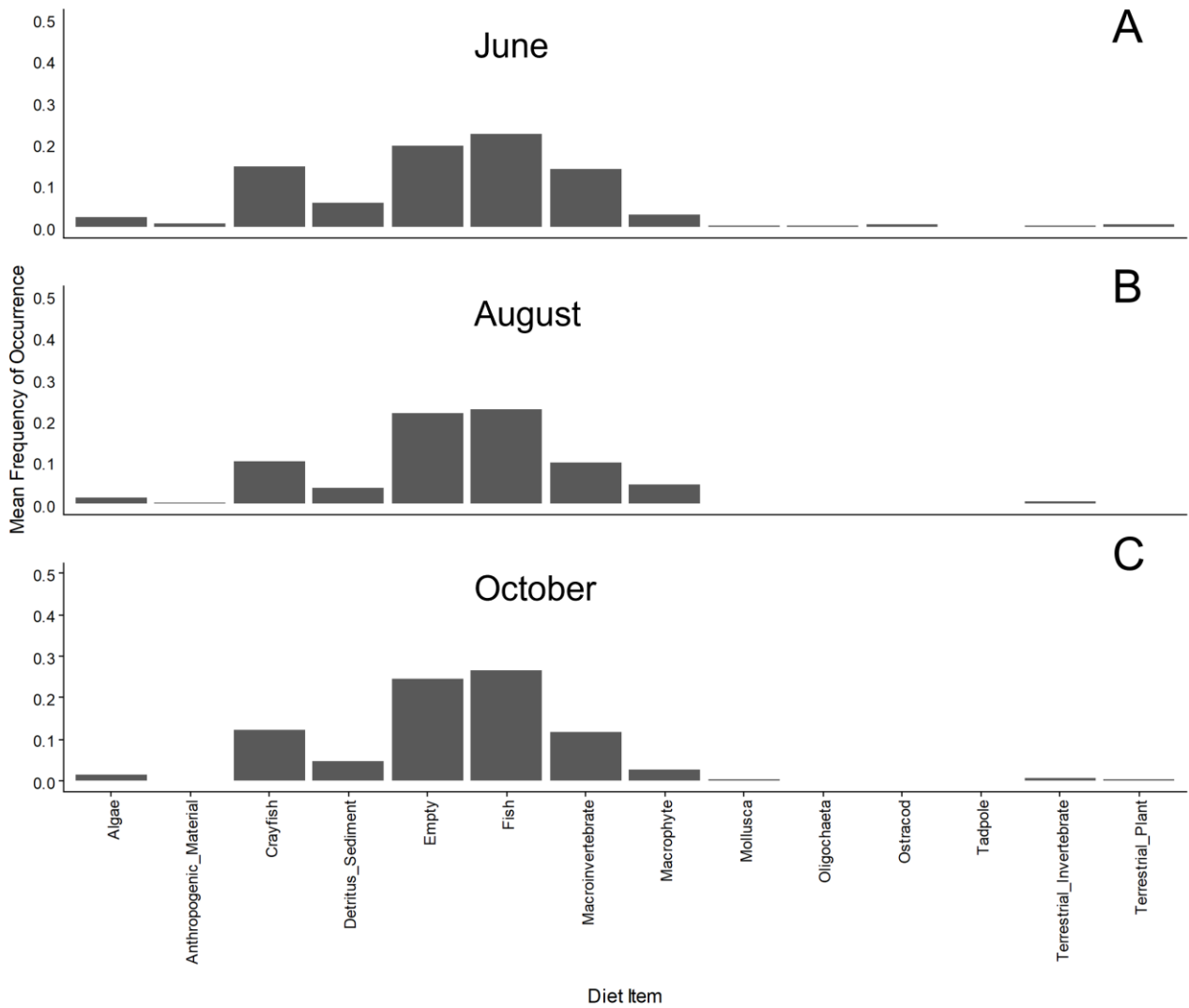


Figure 4.3 Mean frequency of occurrence of prey items in largemouth bass diets by month. Largemouth bass were collected from three months: June (top), August (middle), and October (bottom).

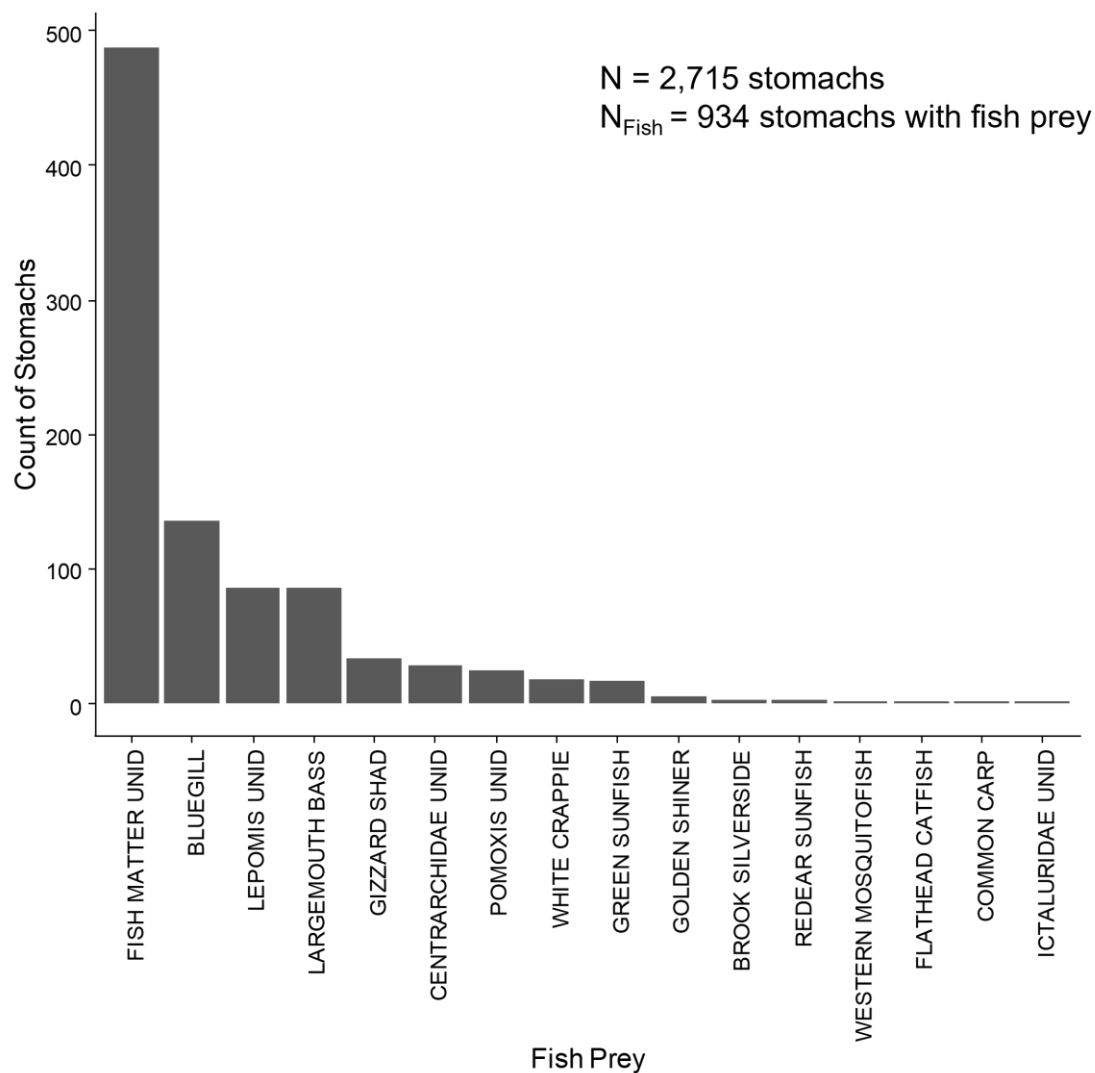


Figure 4.4 Largemouth bass fish prey items. Fish prey were identified to the lowest possible taxon. Fish matter UNID was any fish tissue or structures that were too degraded or incomplete for positive identification to family.

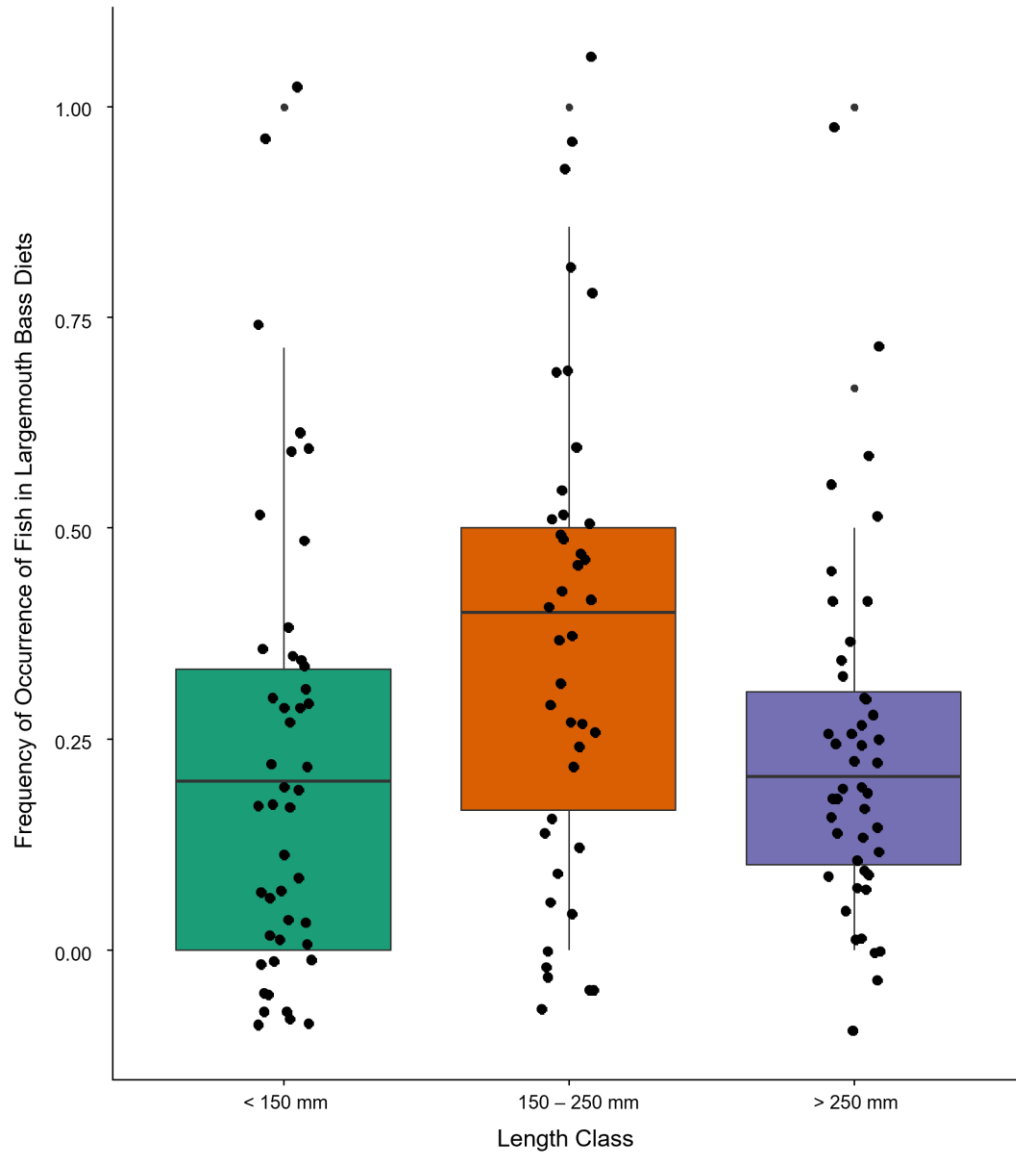


Figure 4.5 Frequency of occurrence of fish in largemouth bass diets summarized by length class. Largemouth bass were collected via daytime shoreline electrofishing and seining in each impoundment in June, August, and October from 2017-2020 from three length classes.

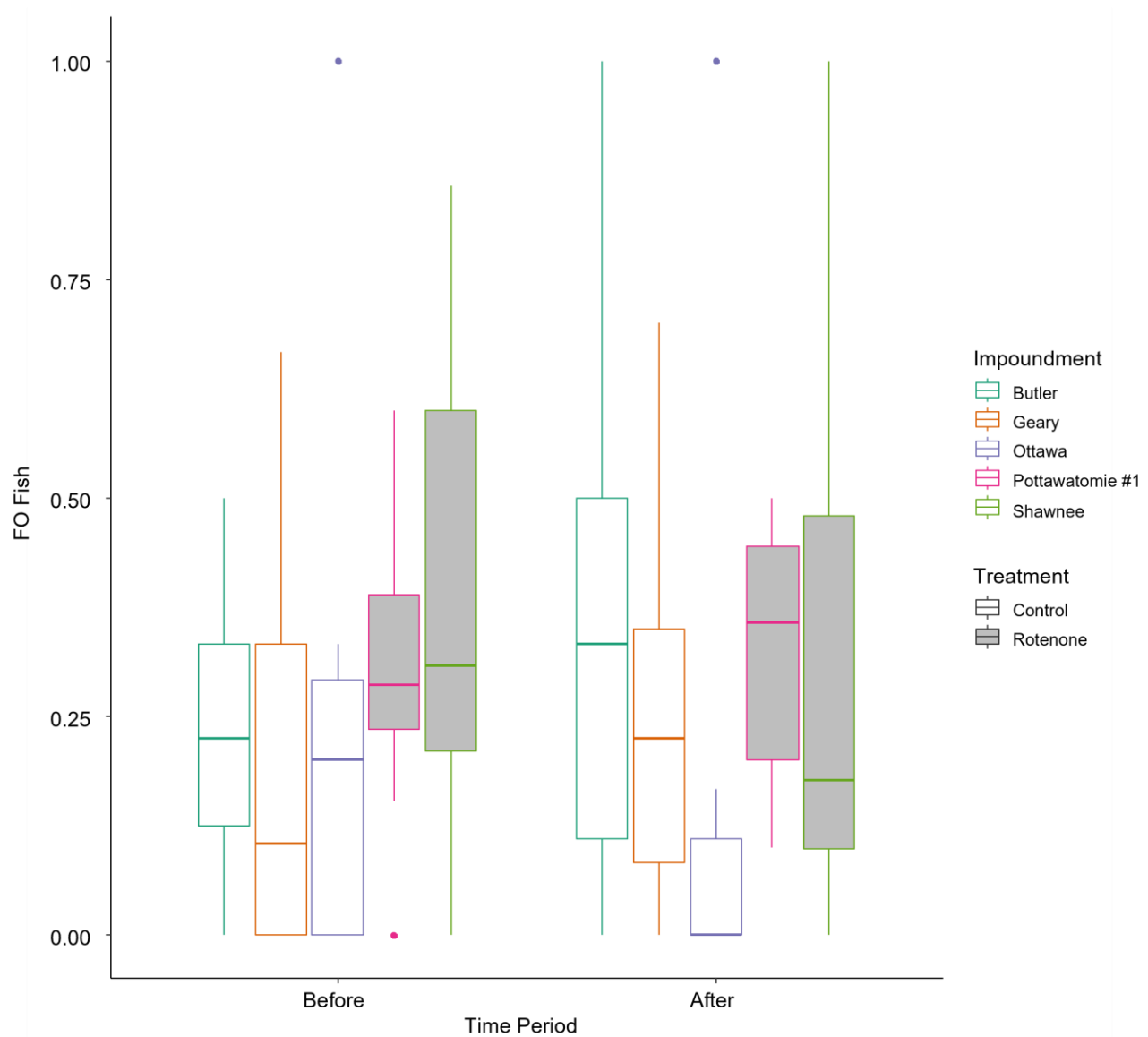


Figure 4.6 Frequency of occurrence of fish in largemouth bass diets before (left) and after (right) Gizzard Shad eradication. Largemouth bass were collected via daytime shoreline electrofishing and seining in each impoundment in June, August, and October from 2017-2020.

Chapter 5 - Conclusions

Gizzard shad are thought to be important components of reservoir food webs, but their trophic interactions in Kansas small impoundments are poorly understood. A key first step in understanding how gizzard shad regulate ecosystem processes in lentic food webs is to examine how gizzard shad abundances co-vary with key limnological variables. In Chapter 2, I demonstrated how gizzard shad can exert top-down effects on lower trophic levels and bottom-up effects via altered nutrient dynamics in small impoundment ecosystems. I tested the influence of gizzard shad on lentic ecosystems by sampling key limnological parameters in 12 small impoundments across a range of gizzard shad relative abundances over four years. I hypothesized that gizzard shad relative abundance would be associated with algal abundance or cyanobacterial abundance but the data did not support that hypothesis. However, the observed positive relationships between shad abundance and total nitrogen (TN) and total phosphorus (TP) concentrations indicated the potential for bottom-up shad effects on primary producers, whereas decreased zooplankton densities with higher shad abundances suggested the potential for top-down shad effects through reduced algal grazing by zooplankton. In addition, we observed lower N:P ratios at higher gizzard shad relative abundances, a pattern consistent with that described in prior literature (Schaus et al. 1997). My study illustrated how watershed-level factors such as agricultural land use and physical factors such as temperature, wind, and precipitation might play a role in regulating impoundment mixing regimes, thereby complicating efforts to quantify gizzard shad effects on limnological parameters in small impoundments.

In Chapter 3, I used a BACI design to test whether gizzard shad eradication influenced key biotic and abiotic parameters in small impoundment food webs. I collected baseline measurements of all limnological and fish community parameters for two years (2017 and 2018)

before applying rotenone to four treatment lakes in December 2018 and then monitored food web components for two years after treatment (2019 and 2020). I fit linear models to evaluate how these food web components responded to rotenone treatment. Although I found a slight decline in TN and TP in treatment impoundments relative to control impoundments following shad removal, overall, there was little evidence that gizzard shad removal had a substantial impact on limnological parameters or fish populations.

In Chapter 4, I sought to describe general patterns and characterize the importance of gizzard shad in largemouth bass diets in 12 Kansas small impoundments and examined how they varied across bass size classes, months, and a gradient of gizzard shad relative abundances, hypothesizing that largemouth bass would selectively prey upon gizzard shad compared to other forage fishes. Bass length class was a significant factor influencing the frequency of occurrence of fish in our regression model, supporting our hypothesis, but FO_{Fish} was not associated with sampling month or gizzard shad relative abundance. Gizzard shad eradication did not alter the frequency of occurrence of fish in largemouth bass diets following rotenone manipulation. Since the relative contribution of gizzard shad prey to bass diets was low compared to the contribution of centrarchids, cannibalism, and crayfish, selective removal of gizzard shad via rotenone biomanipulation in Kansas small impoundments might not meaningfully alter largemouth bass diet composition.

Altogether, it appears as though at high relative abundances gizzard shad may exert meaningful top-down effects on zooplankton communities through their zooplanktivory and interactions with co-occurring fishes, as well as bottom-up effects on nutrient dynamics through their benthic feeding and excretion. The gizzard shad, a freshwater clupeid much like marine clupeids, seems to occupy an intermediate trophic level and exert middle-out effects in small

impoundment food webs. Small impoundments might be thought of as wasp-waist ecosystems, in which an intermediate trophic level controls the abundance of predators through a bottom-up interaction and the abundance of prey through a top-down interaction (Fauchald et al. 2011). The ecosystem effects of gizzard shad at high biomass are substantial enough to merit more monitoring and management attention as part of long-term standardized sampling programs across impoundments, but they might not be the most ecologically pressing issue of concern in a time when Kansas small impoundments are facing multiple stressors such as anthropogenic climate change, cultural eutrophication, invasive species introductions, sedimentation due to natural reservoir aging processes, plastic and chemical pollution, and more. The Kansas Department of Wildlife and Parks collects data on gizzard shad relative abundances, but some districts lack information on shad length, weight, or body condition data across impoundments to estimate gizzard shad dynamic rates to characterize interannual variation in shad recruitment, growth, or mortality. Collecting some of these data as part of existing standardized sampling schemes may prove invaluable for further assessment of interactions between co-occurring forage fishes in small impoundments.

Northern range expansion of gizzard shad associated with climate change (VanDeHey et al. 2012) and illegal introductions by anglers is expected to continue and present management challenges into the future. In the second year of the study, two of the twelve impoundments I monitored (Cowley and Douglas SFL) experienced unanticipated gizzard shad introductions which led to the establishment of recruiting populations, illustrating the propensity of this species to colonize and thrive in new habitats, even in the face of competition with other fishes with overlapping niches. If gizzard shad continue to be problematic through their native or expanded range, their populations can be effectively controlled with low concentrations of rotenone (Neely

et al. 2018), although these remedial actions may be costly, temporary solutions at best in a time of unprecedented ecological change. Overall, gizzard shad effects were weak and mixed, and our ability to precisely quantify responses was limited by our small sample size of impoundments. My dissertation work sheds light on the ecological role of gizzard shad in small Kansas impoundments and illustrates the complexity of studying highly stochastic, novel ecosystems managed for angling recreation. I hope my contribution to advancing our understanding of the ecology of gizzard shad in Kansas small impoundments may inform improved management and monitoring of this species in the Anthropocene.

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Appendix A - Supplemental Materials for Chapter 2

Appendix A.1. Morphometric characteristics of 12 Kansas small impoundments based on Neely et al. 2020. SFL = State Fishing Lake.

Impoundment	Surf Area (ha)	Watershed Area (ha)	Watershed % Agriculture	Mean Depth (m)	Max Depth (m)	Lake Volume (L ³)	% Shallow (< 2 m depth)	Shoreline Length (m)	Watershed Surface Area Ratio	Shoreline Development Index
Butler SFL	59.6326	1775.69	87.33	2.195	5.73	1029957	0.52	7530	29.77	4.89
Clark SFL	165.7284	39930.18	89.09	5.121	11.064	6676837	0.22	11103	240.94	4.31
Cowley SFL	41.7249	1727.75	1.65	3.719	8.687	1234715	0.33	6488	41.41	5.02
Douglas SFL	95.7454	1008.99	27.64	4.115	10.089	3024497	0.37	8370	10.54	4.28
Geary SFL	51.4256	1440.64	6.65	5.395	12.68	2087051	0.23	6836	28.01	4.77
Meade SFL	37.7489	23954.09	80.05	1.707	2.835	472424	0.54	3287	634.56	2.67
Neosho SFL	33.0665	825.32	77.05	1.951	6.645	611807	0.63	5586	24.96	4.86
Ottawa SFL	60.9636	5267.46	16.28	1.311	2.987	579736	0.79	8411	86.40	5.39
Pottawatomie SFL #1	10.5651	323.91	6.76	2.896	5.395	198591	0.28	1996	30.66	3.07
Scott SFL	58.9109	256631.64	71.45	1.615	4.359	709252	0.66	10782	4356.27	7.024
Shawnee SFL	66.9979	635.98	35.66	3.688	9.114	1900796	0.33	6348	9.49	3.88
Washington SFL	36.3546	1666.44	15.13	2.438	5.243	678415	0.44	4661	45.84	3.87

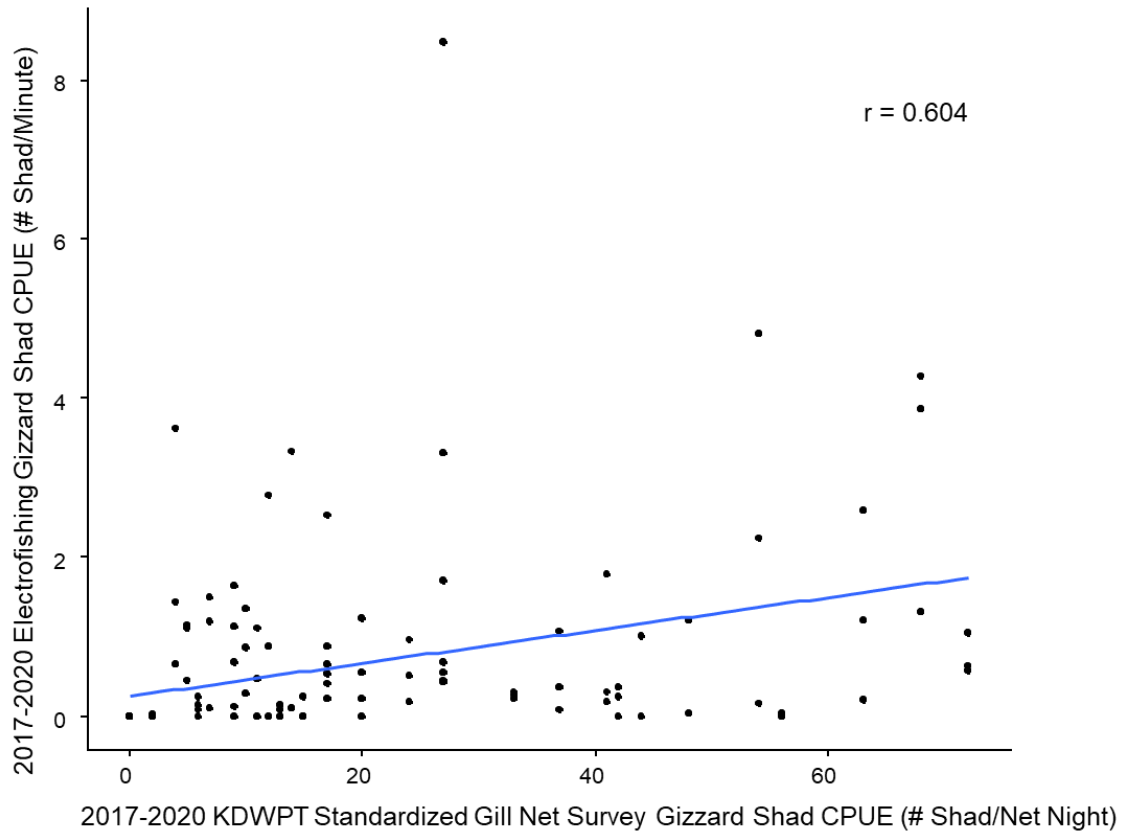
Appendix A.2. Mean and standard deviation of selected abiotic and biotic characteristics measured across 12 Kansas small impoundments. SFL = State Fishing Lake.

Impoundment	Secchi Depth (m)	TN (ppm)	TP (ppm)	Cyanobacteria Abundance (# sq/field)	Diatom Abundance (# sq/field)	Green Algae Abundance (# sq/field)	Total Phytoplankton Abundance (# sq/field)	Cladocera Density (#/L)	Calanoid Copepod Density (#/L)	Cyclopoid Copepod Density (#/L)	Rotifer Density (#/L)	Total Nauplii Density (#/L)	Total Zooplankton Density (#/L)
Butler SFL	0.43 ± 0.15	1.19± 0.65	0.07± 0.04	11.10±14.17	0.91±1.46	1.46 ± 2.16	14.28± 16.45	17.93±23.84	16.81±37.93	4.82±7.97	4.65±8.06	22.82±23.42	67.04±67.72
Clark SFL	0.83 ± 0.20	0.92± 0.34	0.04± 0.03	21.35 ± 24.92	2.11±2.07	2.03±1.52	25.94±24.36	12.02±6.28	3.40±2.71	1.94±2.89	6.12±6.57	14.95±11.79	38.48±24.25
Cowley SFL	1.08 ± 0.24	0.61± 0.32	0.03± 0.03	13.85±18.20	3.85±7.03	1.09±0.74	18.81±18.77	18.90±40.54	10.34±21.45	3.11±10.43	15.09±40.94	24.43±42.22	71.88±128.81
Douglas SFL	1.39 ± 0.57	0.58± 0.33	0.01± 0.03	6.83±7.75	1.95±2.24	0.76±0.66	9.58±9.64	7.05±6.79	5.18±6.05	1.05±1.26	16.58±25.57	15.30±15.37	45.21±41.89
Geary SFL	1.63 ± 0.39	0.71± 0.46	0.03± 0.03	20.89±45.21	0.55±1.13	1.06±1.02	22.54±45.24	27.82±44.52	9.50±21.77	1.28±2.46	8.24±11.31	14.19±12.30	61.08±69.98
Meade SFL	0.43 ± 0.12	1.09± 0.32	0.03± 0.04	10.64±10.35	1.78±3.18	0.65±0.62	13.08±12.48	41.56±32.82	7.75±7.53	3.26±5.19	11.41±17.11	20.34±19.58	84.42±56.54
Neosho SFL	0.54 ± 0.19	1.53± 0.51	0.06± 0.04	7.95±14.27	1.21±1.93	0.68±0.60	9.94±14.96	23.85±22.94	14.94±11.23	6.62±6.36	22.82±34.21	34.48±39.73	102.77±95.42
Ottawa SFL	0.27 ± 0.12	1.54± 0.63	0.24± 0.16	10.93±14.18	0.98±0.95	1.26±1.01	13.20±14.76	24.97±26.12	6.03±7.26	2.31±5.87	23.19±51.47	21.75±19.42	80.18±86.73
Pottawatomie SFL #1	1.77 ± 0.50	0.70± 0.42	0.02± 0.01	14.80±34.35	3.15±4.17	1.48±3.53	19.43±33.69	21.70±21.49	6.21±8.40	5.21±9.88	183.78±371.59	18.55±18.06	235.49±369.45
Scott SFL	0.55 ± 0.59	2.21± 0.85	0.11± 0.12	6.85±9.85	1.55±1.48	1.07±1.22	9.62±10.44	82.63±132.95	34.47±95.45	7.34±17.26	29.52±59.27	30.84±46.93	184.89±276.37
Shawnee SFL	0.85 ± 0.26	0.71± 0.48	0.02± 0.04	20.25±20.78	2.15±1.44	1.85±2.50	24.27±20.49	16.60±24.27	25.36±85.14	3.76±8.31	32.17±68.44	23.08±22.00	100.97±143.84
Washington SFL	0.64 ± 0.33	1.29± 0.74	0.07± 0.06	8.29±9.35	1.76±1.13	0.95±1.05	11.04±10.44	21.82±25.32	5.69±6.49	2.07±2.73	16.59±24.42	24.80±26.18	71.01±57.61

Appendix A.3. Gizzard shad daytime shoreline electrofishing CPUE (catch per unit effort, # gizzard shad / minute) by month sampled.
SFL = State Fishing Lake.

Impoundment	June 2017	August 2017	October 2017	June 2018	August 2018	October 2018	June 2019	August 2019	October 2019	June 2020	August 2020	October 2020	Overall Mean ± Std Dev
Butler SFL	0.2	2.58	1.2	1.78	0.3	0.17	0.5	0.95	0.18	0.87	0.28	1.34	0.86 ± 0.76
Clark SFL	3.87	1.31	4.28	NA	0.54	1.96	NA	NA	NA	NA	NA	NA	2.39 ± 1.62
Cowley SFL	0	0	0	0	0	0.02	0	0.12	0	0.4	0.53	0.65	0.14 ± 0.24
Douglas SFL	0	0	0	0	0	0.24	0	0.47	1.11	1.11	0.45	1.15	0.50 ± 0.50
Geary SFL	0.23	1.225	0	0.37	0.08	1.05	0	0.88	2.77	0.14	0.07	0	0.56 ± 0.82
Meade SFL	0	0	0	NA	0	0	0	0	0	0	0	0	0
Neosho SFL	2.23	4.8	0.15	1.03	0.57	0.62	0	1	0.21	0.88	2.51	4.2	1.52 ± 1.59
Ottawa SFL	0.1	9.51	3.33	1.12	0.68	1.64	0	0.03	0.45	0.55	0.68	0.43	1.54 ± 2.67
Pottawatomie SFL #1	0	0.23	0.37	0.3	0.22	0.27	0	0	0	0	0	0	0.12 ± 0.15
Scott SFL	0	0	0	NA	0	0	0	0	0	0	0	0	0
Shawnee SFL	1.7	8.47	3.3	1.43	0.66	3.62	0	0	0	0	0	0	1.60 ± 2.53
Washington SFL	0.6	1.2	0.04	0.1	1.19	1.49	0	0	0	0.08	0.25	0.13	0.42 ± 0.55

Appendix A.4. Comparison of gizzard shad catch per unit effort (CPUE) from daytime shoreline electrofishing (y-axis, # shad captured / minute) and gizzard shad CPUE from standardized fall gill net surveys (x-axis, # shad/net night). SFL = State Fishing Lake.



Appendix B - Supplemental Materials for Chapter 3

Appendix B.6. Linear fixed effect models for Bluegill length-at-age. Age-0 Bluegill were collected via littoral seining and age-1 and age-2 Bluegill were collected via daytime shoreline electrofishing from each impoundment in October 2017-2020. Asterisks denote statistical significance at $\alpha = 0.10$.

Model	Fixed Effect or Interaction Term	Estimate	Std. Error	t-value	95% CI	p-value
Bluegill mean length-of-age 0	Treatment	-0.381	5.884	-0.065	-12.921; 12.160	0.949
	Time Period	10.097	5.262	1.919	-1.119; 21.314	0.074*
	Treatment x Time Period	-4.247	8.727	-0.487	-22.848; 14.353	0.634
Bluegill length-at-age 1	Treatment	-4.000	4.501	4.501	-15.013; 7.013	0.408
	Time Period	-8.405	4.025	4.025	-1.445; 18.255	0.082*
	Treatment x Time Period	-4.151	6.365	6.365	-19.725; 11.424	0.539
Bluegill length-at-age 2	Treatment	-8.368	4.090	-2.046	-16.47; -0.267	0.043*
	Time Period	-17.638	3.861	-4.568	-25.28; -9.991	< 0.001*
	Treatment x Time Period	2.561	7.454	0.344	-12.20; 17.32	0.732

Appendix B.7. Linear fixed effects models for Largemouth Bass length-at-age. Age-1 and age-2 Largemouth Bass were sampled via daytime shoreline electrofishing in October 2017-2020. Asterisks denote statistical significance at $\alpha = 0.10$.

Model	Fixed Effect or Interaction Term	Estimate	Std. Error	t-value	95% CI	p-value
Largemouth Bass mean length-at-age-1	Treatment	-42.580	22.160	-1.922	-86.82; 1.65	0.059
	Time Period	-52.470	17.210	-3.049	-86.81; -18.12	0.003*
	Treatment x Time Period	35.510	28.260	1.257	-20.89; 91.9	0.213
Largemouth Bass mean length-at-age-2	Treatment	-37.009	8.663	-4.272	-54.14; -19.88	< 0.001*
	Time Period	-3.383	11.287	-0.300	-25.70; 18.94	0.765
	Treatment x Time Period	-6.699	15.232	-0.440	-36.82; 23.42	0.661

Appendix C - Supplemental Materials for Chapter 4

Appendix C.1. Gizzard shad daytime shoreline electrofishing CPUE (catch per unit effort, # gizzard shad / minute) by month sampled. SFL = State Fishing Lake.

Impoundment	June 2017	August 2017	October 2017	June 2018	August 2018	October 2018	June 2019	August 2019	October 2019	June 2020	August 2020	October 2020	Overall Mean ± Std Dev
Butler SFL	0.2	2.58	1.2	1.78	0.3	0.17	0.5	0.95	0.18	0.87	0.28	1.34	0.86 ± 0.76
Clark SFL	3.87	1.31	4.28	NA	0.54	1.96	NA	NA	NA	NA	NA	NA	2.39 ± 1.62
Cowley SFL	0	0	0	0	0	0.02	0	0.12	0	0.4	0.53	0.65	0.14 ± 0.24
Douglas SFL	0	0	0	0	0	0.24	0	0.47	1.11	1.11	0.45	1.15	0.50 ± 0.50
Geary SFL	0.23	1.225	0	0.37	0.08	1.05	0	0.88	2.77	0.14	0.07	0	0.56 ± 0.82
Meade SFL	0	0	0	NA	0	0	0	0	0	0	0	0	0
Neosho SFL	2.23	4.8	0.15	1.03	0.57	0.62	0	1	0.21	0.88	2.51	4.2	1.52 ± 1.59
Ottawa SFL	0.1	9.51	3.33	1.12	0.68	1.64	0	0.03	0.45	0.55	0.68	0.43	1.54 ± 2.67
Pottawatomie SFL #1	0	0.23	0.37	0.3	0.22	0.27	0	0	0	0	0	0	0.12 ± 0.15
Scott SFL	0	0	0	NA	0	0	0	0	0	0	0	0	0
Shawnee SFL	1.7	8.47	3.3	1.43	0.66	3.62	0	0	0	0	0	0	1.60 ± 2.53
Washington SFL	0.6	1.2	0.04	0.1	1.19	1.49	0	0	0	0.08	0.25	0.13	0.42 ± 0.55