Small water, big potential: farm ponds and their potential for native fish conservation

by

Peter Joel Pfaff

B.S., Concordia University Texas, 2009
M.S., Texas State University, 2019

AN ABSTRACT OF A DISSERTATION

submitted in partial fulfillment of the requirements for the degree

DOCTOR OF PHILOSOPHY

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Abstract

Small impoundments, particularly farm or agricultural ponds, are often overlooked as potential habitat for native species despite their ubiquity across many human altered landscapes. Farm ponds intermittently connect to streams during floods allowing species exchanges to and from these novel habitats. In 2014, a farm pond on Tallgrass Prairie National Preserve (TAPR) was found to contain federally endangered Topeka Shiner (*Notropis topeka*). This finding provided a unique opportunity to study how farm ponds can be used in native species conservation in conjunction with translocation. I used field and experimental studies to; (1) assess farm pond fish communities on a broad scale and investigate the factors structuring them, (2) quantify the impact of predator presence on the translocation success of a native minnow species, and (3) quantify the survival, movement, and reproduction of Topeka Shiner in select ponds and a stream on TAPR. I sampled 100 farm ponds across central Kansas in 2021 and more than half of sampled farm ponds contained stream fish to assess farm pond fish communities. Amphibians and crayfish dominated small, less permanent ponds, while larger ponds were dominated by stocked sportfish. Distribution modeling revealed a negative correlation between stocked fish and other community components and suggested that native stream fishes are most prevalent in intermediate size ponds with few or no sportfish. To look for mechanisms driving these correlations, I directly investigated interaction between sportfish and stream fish in farm ponds. I conducted a controlled experiment to quantify the influence of piscivorous Largemouth Bass (*Micropterus salmoides*) on the survival of translocated Bluntnose Minnow (*Pimephales notatus*). I translocated 1600 Bluntnose Minnow (*Pimephales notatus*) into replicate treatment ponds with and without Largemouth Bass. Each minnow was implanted with a passive integrated transponder (PIT) tag. Translocated populations were monitored using stationary and mobile PIT antennas and estimates of apparent survival and probability of detection for each pond were derived from open population mark-recapture models. Apparent survival was nearly two times higher in ponds without bass, suggesting that predation by bass could lead to higher mortality. Additionally, probability of detection was nearly 10 times higher in ponds without bass, suggesting reduced movement of translocated minnows when bass were present. While the direct effect of mortality impacts translocated populations, the indirect effect of altered behavior may heavily influence translocation success. These results suggest that Largemouth Bass could limit the success of translocated or naturally colonizing minnow species. Finally, I assessed the efficacy of using translocation as a conservation tool for the federally endangered Topeka Shiner. From 2019 to 2022, Topeka Shiner were captured from a naturally-colonized pond population on the TAPR and translocated to two nearby ponds and a nearby stream. A portion of all translocated Topeka Shiner were implanted with PIT tags, and some tagged individuals were returned to the source population. Overall apparent survival (mortality and emigration) estimates between six-week sampling intervals varied across ponds and the stream but were always greater than 0.5. We observed emigration and immigration in pond habitats during flooding and physically captures young-of-year Topeka Shiner in all habitats indicating successful reproduction. Translocation of wild individuals from stable populations to ponds, as in this study, is a viable and low-cost conservation strategy to both bolster populations and repatriate areas from which Topeka Shiner were extirpated. These findings may also have implications for other species of threatened and endangered minnows or amphibians that inhabit headwater streams across the Great Plains, particularly in more xeric regions and in light of climate change.

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Table of Contents

List of Figures

List of Tables

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I thank all the members of the KState Fish Ecology Lab over the years. Their support and friendship were excellent and without them I wouldn’t have been able to do this work. The same goes for all the other graduate students and undergrads I worked, took classes, and taught with. My time here at KState was a real joy and that was due to all the interactions I had with the excellent people here.

Finally, I would like to thank my family for their constant love and support over the years. I especially thank my wife, Ryoko, and our daughter, Kiyoka. They are the light of my life. 本当にどうもありがとう。

Dedication

I dedicate this dissertation to my late grandfather and grandmother, Bernie and Joyce Lees, who passed away before I went back to school, and my late grandmother, Lois Lees, who passed away during my time as a PhD student. I love and miss you all. It was the experiences I had with you all in Minnesota that first got my interested in fish and aquatic systems.

Preface

Chapter 2, 3, and 4 of this dissertation are in a format designed for submission to a peer-reviewed journal and include co-authors. Chapter 2 is formatted in Harvard style while Chapters 1, 3, 4, and 5 are formatted in APA style. Chapter 2 is adapted from Canadian Science Publishing, Canadian Journal of Fisheries and Aquatic Sciences, Pfaff et al. 2022.

Chapter 1 - Novel Ecosystems on the Prairie Landscape

As humans spread across the globe, so did their impacts on the landscape. Among the greatest and most widespread impact was the effect of agriculture. As agricultural practice spread to arid or intermittently wet areas, the need for reliable water sources became more important. Small farm ponds have been constructed for irrigation, watering livestock, and recreation for thousands of years (Upex 2004). Within North America, the widespread construction of farm ponds began in the 19th century as America expanded westward (McMurray 2020). Today there are up to 9 million constructed ponds in the United States (Renwick et al. 2005) with a majority being less than two hectares in size and varying in water permanency (Chumchal et al. 2016). Despite being so widespread across the landscape, farm ponds are often overlooked in terms of their interaction with the landscape by biotically and abiotically (Morden et al. 2020). Native stream fishes naturally colonize farm ponds during periods of intermittent connectivity (Austin et al., 1996; Tennessee Wildlife Resources Agency, 2009; Tiemeier, 1954), though little is known about pond fish communities. Existing studies are often limited in geographical extent but have highlighted the use of farm ponds by native species in North America (Hedden et al. 2021), Japan (Tsunoda and Mitsuo 2012; Yonekura, et al. 2004), and Europe (Copp et al. 2007; Tarkan et al. 2011) and show the importance of dispersal in determining community composition (Mitsuo et al. 2011). In 2014, a population of endangered Topeka Shiner was found in a farm pond located on the Tallgrass Prairie National Preserve (TAPR) (Portofee et al. 2018). This species has experienced considerable decline in Kansas (Schrank et al. 2001) and this finding presented an opportunity to assess if ponds might be valuable in Topeka Shiner conservation. I identified three key issues to investigate that would help inform the conservation value of farm ponds: (1) a broad scale assessment of existing farm pond communities and factors influencing species composition; (2) assessing potential biotic interactions in shaping farm pond communities; (3) determining if existing communities influence the survival of colonizing or translocated minnows.

Farm pond communities and the factors influencing them

Both abiotic and biotic factors can shape communities. Among farm ponds, connectivity to other ponds, water permanency, and pond size are abiotic factors that influence community structure (Hedden et al. 2021; Mitsuo et al. 2011; Swartz and Miller 2019; Tonn et al. 1990). Additionally, farm ponds may be stocked with sport fish (Dauwalter and Jackson 2005) which can negatively impact other species, such as native minnows (Jackson 2002). The second chapter of this dissertation examines macroconsumer (fishes, amphibians, and crayfish) community structure in farm pond in central Kansas. I developed a conceptual framework where communities respond to and change along a pond size and water permanency gradient which is further influenced by interactions among species. To do this, I sampled the communities and gathered local abiotic and landscape-scale data for 100 farm ponds along ecological gradients in central Kansas. I assessed associations among species and abiotic factors to assess which factors are important in predicting community structure and assessed potential interactions among species. In the third chapter, I test whether the presence of a predator, Largemouth Bass (*Micropterus salmoides*) influences the survival and behavior of a translocated Bluntnose Minnow (*Pimephales notatus*). Bluntnose Minnow were translocated to ponds with and without Largemouth bass and monitored using implanted passive integrated transponder (PIT) tags. This study directly investigates mechanisms driving species assemblage patterns in chapter 2. Moreover, it provides important information to managers when using farm ponds in conservation or the propagation of minnows.

The role of farm ponds in conservation

Farm ponds, despite being anthropogenic alterations, may be of conservation benefit to some species. They are used for reproduction by many species of amphibians (Swartz and Miller 2019) and have been recognized as habitat for threatened and endangered fish (Copp et al. 2007; Thompson and Berry 2009). In the fourth chapter of this dissertation, I directly assess the efficacy of using translocation to novel habitats in the conservation of the endangered Topeka Shiner. Topeka Shiner from a farm pond on Tallgrass Prairie National Preserve were translocated to nearby pond and stream habitats and monitored using implanted PIT tags. I assessed the apparent survival and detection of Topeka Shiner in the source and translocated populations, monitored their movement to and from ponds and within the stream, and assessed their ability to reproduce in each habitat. This study highlights the potential of using farm ponds as a habitat for an endangered species to both reproduce in and disperse from.

Together, the research presented in these chapters can fill key gaps in knowledge concerning the structure of farm pond communities, the abiotic and biotic factors influencing them, and how farm ponds can provide a tool for native species conservation. Even some fundamental information, such has how and when fish disperse to and from farm ponds, is largely unknown. Additional goals are to provide baseline information on the survival of translocated minnows, provide a proof-of-concept of the use of PIT tag technology to monitor fish populations in ponds, and to quantify the impact of stocked fish on native species.

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Chapter 2 - Community assembly of prairie farm ponds: Build it and they will come, stock it and they won't

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Abstract

Dam construction affects freshwater ecosystems worldwide. While there is much focus on large impoundments, farm ponds are overlooked despite their near-ubiquity across human-altered landscapes. Within the Great Plains of North America, there are millions of farm ponds, yet little is known about the fish communities and factors structuring them. We propose a conceptual model where fish, amphibian, and crayfish abundances differ along a pond size and water permanency gradient and are further influenced by interactions among species. In the summer of 2021, we sampled 100 farm ponds across central Kansas, primarily on private land. Pond size and permanency explained community structure with smaller and less permanent ponds being dominated by amphibians and crayfish while larger ponds were dominated by stocked sportfish. Distribution modeling revealed a negative correlation between stocked fish and other community components indicating potential interactions. If we are to conserve headwater stream species, especially those that are threatened or endangered, strategies that integrate farm ponds seem necessary given their prevalence on the landscape.

Keywords: Headwater Steams, Great Plains, Fish, Amphibian, Crayfish, Dams, Conservation

Introduction

Freshwater rivers and streams suffer from alterations caused by dam construction (Bonner and Wilde, 2000; Gao et al., 2010; Quist et al., 2005). However, small impoundments, particularly farm or agricultural ponds, are often overlooked as potential habitat for native species despite their near ubiquity across human altered landscapes. Within the United States, there are estimated to be between 2.6 and 9 million impoundments (Renwick et al., 2005) with one study estimating over 500,000 in the southeastern Great Plains alone (Chumchal et al., 2016). By default, farm ponds are constructed on low order stream channels with small catchments. Thus, communities in ponds are likely to interact with headwater stream communities through immigration and emigration. Farm ponds fragment streams, alter natural geomorphic processes, and also serve as a fountainhead for the dispersal of invasive species (Collins et al., 2020; Hedden et al., 2021; Kashiwagi and Miranda, 2009; Mbaka and Wanjiru Mwaniki, 2015). Additionally, the large number and broad distribution of ponds relative to that of large impoundments means that, while potentially less impactful individually, their cumulative effects on hydrology or biodiversity may be greater than that of large impoundments (Morden et al., 2021).

Despite high density of ponds on the landscape, little is known about factors structuring their fish communities. One obvious factor influencing these communities is fish stocking. There are numerous management guides that prescribe stocking recommendations for both recreational fishing [e.g., Largemouth Bass (*Micropterus salmoides*), Sunfish (*Lepomis spp.*), and Channel Catfish (*Ictalurus punctatus*)] and water quality management [e.g., triploid Grass Carp (*Ctenopharyngodon idella*)] of ponds (Dauwalter and Jackson, 2005). Farm pond management guides have referenced the "nuisance" of native stream fishes (or rough fishes as they are often termed pejoratively) moving into farm ponds from nearby streams (e.g., Austin et al., 1996; Tennessee Wildlife Resources Agency, 2009; Tiemeier, 1954). Likewise, fish introduced to ponds for sportfishing can move out of ponds, potentially influencing the local stream communities (Hedden et al., 2018; Perkin et al., 2016). To date, few studies have surveyed the entire fish communities of farm ponds and those that have are often limited in scope to a few ponds or specific areas such as nature preserves (Arruda, 1979; Hedden et al., 2021; Holmgren et al., 2019).

Although human engineered, farm ponds might provide habitat for native fish species colonizing these habitats. Studies in Japan found that native fish readily use irrigation ponds, though their use was negatively impacted by invasive species with native abundances being up to three time higher in the absence of invasive species (Tsunoda and Mitsuo, 2012; Yonekura, et al., 2004). Other studies have found correlations between species richness and measures of connectivity to nearby streams (Mitsuo et al., 2011). Urban ponds in Taiwan were found to harbor both native and near-threatened species (Huang et al., 2021). Similarly, ornamental estate and forest ponds in England are used for conservation of native Crucian Carp, *Carassius carassius* (Copp et al., 2007; Tarkan et al., 2009; Tarkan et al., 2011). Farm ponds can also provide ecosystem services such as breeding habitat for amphibians (Knutson et al., 2004; Leja, 1998; Swartz and Miller, 2019).

To better understand these systems and their conservation potential, an extensive and broadscale assessment of farm pond communities is needed. Based on a review of existing literature, we developed a conceptual model of how macroconsumer (fish, larval amphibian, and crayfish) communities change along a water permanency and size gradient along with interactions among community members (Hedden et al., 2021; Mitsuo et al., 2011; Swartz and

Miller, 2019; Tonn et al., 1990; Figure 2.1). Many farm ponds periodically dry and rewet yearly or during droughts and, over longer time scales, are transient due to sedimentation or deconstruction with an estimated 30 to 90% of ponds present in the 1950s being gone by 2000 (Renwick and Andereck, 2006). Farm ponds that are prone to regular drying may be more readily used by temporary residents such as amphibians due to lack of competition from fish that require greater water permanency (Swartz and Miller, 2019). We hypothesize the smallest and most ephemeral farm ponds will be dominated by amphibians and crayfish. As pond size and water permanency increase, they will provide more suitable habitat for stocked or native stream fish, which might displace or prey on amphibians and crayfish. Further, the largest and most hydrologically stable ponds are most likely to be stocked with predatory sport fish and asymmetrical competition or predation will lead to their dominance. To evaluate our conceptual framework, we surveyed farm pond communities, related those communities with landscape and abiotic factors, and assessed potential species interactions shaping these communities. We sampled farm ponds in the Flint Hills region of central Kansas, USA and primarily focused on ponds owned by private landowners, but also some that occur on conservation easements.

Methods

Functional Definition of Farm Ponds

Small impoundments are often defined as constructed water bodies up to 40 ha (Willis, Lusk and Slipke, 2010). This definition is functional since larger impoundments tend to have greater diversity in both habitat availability and predator-prey dynamics (Willis and Neal, 2012). This category can be further broken down with ponds being defined as a body of water less than 2 ha in size that vary in water permanency (Collinson et al., 1995; Davies et al., 2007). For the purpose of this study, we focused on ponds 2 ha or less. Within the North American Great Plains many farm ponds are constructed by excavating an area within a watershed, building an earthen dam, or a combination of both. These impoundments are often constructed to provide a service to a landowner such as watering cattle, irrigation, flood and erosion control, or providing recreational opportunities. Depending on location within the watershed and overall pond size, inflowing ephemeral or intermittent streams can be present. Typically, farm ponds have two outflows: a standpipe that maintains a constant maximum water level during normal conditions and a higher spillway over the dam which allows water to quickly leave the pond during flooding, while maintaining the structure of the dam (Deal et al., 1997). Because ponds typically occur on low-order streams, exchanges of stream organisms between ponds and nearby streams are most likely to occur during flooding over the spillway.

Study Area

Ponds were surveyed within the Kansas and Neosho River basins in the Flint Hills region of central Kansas. The National Hydrology Database (NHD) lists 216,466 ponds or reservoirs within the state of Kansas with a vast majority (211,454) being two hectares or less (http://nhd.usgs.gov/). Our target area contained 40,605 ponds or reservoirs with 39,961 being less than two hectares (Figure 2.2A) and was comprised of eight 8-digit Hydrologic Catalog Units (HUC; Figure 2.2B). Ponds were selected to represent a range of watershed characteristics using a random-stratified sampling design developed from a previous study that visited stream sites in this region (Bruckerhoff et al., 2021). Pond surface areas were calculated using the National Hydrology Database. Our size criteria were a minimum of 0.25 ha and maximum of 2.0 ha. Ponds meeting these criteria were randomly selected for landowner contact. If landowners did not grant permission to sample a pond, multiple reselections were made until we were able to gain land access. Additionally, many landowners had multiple ponds meeting our criteria and, time permitting, we generally sampled all ponds within our criteria to which landowners provided access. In total, we sampled 100 ponds (Figure 2.2B).

Date Collection and Sampling

Ponds were sampled a single time in the summer of 2021 (June to August) using a standard seine (4.6 × 1.8 m, 3.2-mm mesh) and gillnet (50 m x 1.5m, 2.54 cm mesh). We chose these techniques to maximize detection of fish species present based on a previous study (Hedden et al., 2021), but also report larval amphibian and crayfish, which were vulnerable to these methods. Sampling effort included a minimum of four and maximum of eight, 20 m seine hauls that covered all available habitat types, including pond inlets, side-banks, and dams. Dominant substrate types were estimated visually and by feel and recorded using a modified Wentworth scale for each seine haul (Cummins, 1962). Vegetation cover percentage and depth were also recorded for each seine haul. All macroconsumers (fishes, amphibians, and crayfish) were identified and enumerated in the field with at least the first 60 individuals of each species being measured for total length. A gillnet was deployed for 0.5 to 2.0 hours in each pond prior to seining. Additional data taken in the field included size and functionality of the standpipe, cattle access, and spillway construction and erosion including the presence and height of any knickpoints. A knickpoint is a sharp, steep change in stream channel slope such as a waterfall or cascade which can pose a challenge to fish movement. Finally, maximum pond depth was measured using HOBO pressure logger data processed with HOBOware software (HOBO U20L-01, Onset, Bourne, Massachusetts). The logger was set to record depth at one second intervals and was cast multiple times into the middle of each pond using a fishing rod and reel. The logger was allowed to settle on the bottom for a minimum of five seconds and then slowly reeled in ensuring that it settled on the bottom every 2 m.

Catchment area of each pond was estimated with 10m digital elevation models (DEMs) using QGIS and SAGA software (Conrad et al., 2015; QGIS Development Team, 2021). Catchment area shapefiles were then used in conjunction with the National Land Cover Database to estimate catchment land cover for each pond (Homer et al., 2015). Distance from pond to stream, nearest stream order, and gradient from pond to nearest stream were all calculated using 10 m DEMs and NHS flowline shapefiles in QGIS. Pond areas and perimeter were measured individually over a series of four time periods (2009, 2012, 2014, and 2019 imagery) using Google Earth Pro (https://www.google.com/intl/en-GB/earth/versions/). Finally, water permanency was calculated by dividing the minimum pond area by the maximum pond area from the four-year time series of imagery. These four years represented a range of conditions from a drought year (2012) to one of the wettest years on record (2019). A full list of environmental variables with summary statistics can be found in Table A.1.

Statistical Analyses

Program R version 4.1.1 (R Core Team, 2021) was used for all statistical analyses. First, Moran spectral randomization (MSR; Crabot et al. 2019) was performed using the "msr" function in the adespatial package (Dray et al. 2018) to detect spatial autocorrelation among ponds and communities rather than the Mantel test due to its low power when working with spatial data (Legendre et al. 2015). A straight-line distance matrix between all ponds was prepared and used to test if distance explained observed variation in farm pond community structure. To relate pond communities with landscape and abiotic factors, we used canonical correspondence analysis (CCA) and bidirectional stepwise selection of variables. Parameters were added or removed from the model at each step to improve AIC, selection stopped once AIC could not be improved further. Prior to model selection, parameters with a variance inflation factor (VIF) > 10 were removed (Borcard et al., 2011). Variables with high VIFs included metrics of size and water permanency (area and water permanency were removed), metrics of knickpoint characteristics (total knickpoint height and maximum knickpoint height were removed), and landcover variables (crop landcover was removed; Pearson correlations to visualize associations are presented in Figure A.1). Species present in two or fewer ponds were removed from this analysis. Permutational ANOVA was used to test the significance of CCA axes and environmental variables. CCA was conducted using the "cca" function in the vegan package (Oksanen et al., 2013). Because we predicted potentially modal associations between log abundance of community components (amphibians and crayfish, stream fish, and stocked fish; Figure 2.1) and pond size (log perimeter), we used both linear and polynomial regression tests. Null, linear, second-degree polynomial, and third-degree polynomial models were fitted for each community component. Model selection was performed using Akaike's information criteria corrected for small sample size (AICc) using the "model.sel" function in the MuMIn package (Barton and Barton, 2015). The model with lowest ΔAICc was selected.

Joint species distribution modeling (JSDM) was used to assess potential species interactions shaping communities. Joint species distribution models are an emerging statistical approach that uses latent variables from Bayesian Markov Chain Monte Carlo (MCMC) estimation via the JAGS software (Plummer 2003) to describe species occurrences as a function of the environment while also considering latent variables which can be seen as residual correlation patterns (Wilkinson et al., 2021). The latent variables derived from MCMC estimation are modeled correlations between communities with covariates (i.e., environmental variables) in a correlated response model (CRM) and communities without covariates in a pure latent variable model (LVM). In this study, JSDM was used to control for environmental variables and estimate residual correlations among species which is displayed as a correlation matrix as estimated in the CRM. Important environmental variables as identified in the CCA were included in the JSDM to maintain continuity between these analyses. Default model parameters, a negative binomial distribution on counts data to avoid overdispersion (Hui, 2016), and the use of weakly informative priors (normal distribution with mean of 0 and variance of 10; Gelman et al., 2008) were used in the models. We ran 30000 iterations with a burn-in of 10000 and a thinning factor of 50. JSDM was conducted using the "boral" function in the boral package (Hui, 2016). Model convergence was confirmed using the Geweke diagnostic included in the "boral" output (Geweke, 1992) where 95% of the Z statistics for the monitored parameters fall short of the cut-off. The relative influence of environmental variables on species co-occurrence patterns in the JSDM can be assessed using the proportional difference in trace score between the latent variable model and the model including environmental factors. We recognize these correlations do not necessarily infer interaction from co-occurrences (Blanchet et al., 2020), but they provide a window into co-occurrence patterns that are not explained by the spatial and environmental variables.

Results

Pond Communities, Landscape, and Abiotic Factors

Of the 100 ponds sampled, 82 contained fish and 67 contained amphibians or crayfish. Only a single pond did not contain any macroconsumers. Further, 56 contained one or more stream fish species while 66 contained one or more stocked species. Overall, 20 fish, three amphibian, and one crayfish species were observed in sampled farm ponds (Table 2.1). The most common species were Green Sunfish (*Lepomis cyanellus*; present in 49 pond), Largemouth Bass (*Micropterus salmoides*; 45), and Virile Crayfish (*Faxonius virilis*; 41). There was no relationship between spatial distribution of ponds and community structure in MSR ($p = 0.833$), suggesting pond communities independently responded to biotic and abiotic factors. CCA axis I (p-value = 0.006) explained a gradient of stream distance (p-value = 0.009), vegetation cover (p-value < 0.001), pond perimeter (p-value < 0.001), and catchment area (p-value = 0.013) while axis II (p-value = 0.002) was primarily a gradient of pond depth (p-value = 0.018) and perimeter (Figure 2.3). Individual species associations showed distinct grouping. Positive values on CCA axis II corresponded to ponds with amphibians and crayfish species. Values near zero or negative on CCA II and negative on CCA axis I largely corresponded to ponds with stocked or sportfish species such as Bluegill (*Lepomis macrochirus*) and Largemouth Bass. Finally, species negative on CCA axis II and positive on CCA axis I were largely ponds with stream species such as Orangespotted Sunfish (*Lepomis humilus*) and Topeka Shiner (*Notropis topeka*). Model selection revealed the top model for crayfish and amphibian log abundances was linear with log perimeter ($r^2 = 0.071$, $F_{(1, 98)} = 8.533$, $p = 0.004$; Table 2.2; Figure 2.4; Figure A.2 for individual species densities). Similarly, the top model for log stocked fish abundance was the linear model ($r^2 = 0.142$, $F_{(1, 98)} = 17.44$, $p < 0.001$). The top model for stream fish was the second-degree polynomial model ($r^2 = 0.041$, $F_{(1, 97)} = 3.123$, $p = 0.048$) suggesting a peak in abundance in intermediate-sized ponds.

Interaction between Stocked and Stream Fish

We verified MCMC convergence for all models (96.21% for LVM and 96.89% for CRM). The proportional difference in trace score for the LVM (193.008) and CRM (161.232) indicates that environmental variables explained 16.5% of the variation in community structure of farm ponds. The latent variable model showed a broad positive environmental correlation across many stocked and stream fish species (Figure 2.5A). Parsing out the effects of environmental variables, residual correlations suggest a distinct separation between many stocked and stream species (Figure 2.5B). There were nine significant positive residual correlations among stream fish species such as Bluntnose Minnow (*Pimephales notatus*), Topeka Shiner, Green Sunfish, and Orangespotted Sunfish and between stream fish and amphibians and crayfish. The second group with ten significant positive residual correlations was comprised of stocked species such as Largemouth bass, Bluegill (*Lepomis macrochirus*), Redear sunfish (*Lepomis microlophus*), and Channel Catfish and one stream species, Black Bullhead (*Ameiurus melas*). Further, these two groups had 19 significant negative residual correlations between them. The average residual correlation between amphibians and crayfish and stocked fish was -0.366 while the average residual correlation between stream fish (excluding Black Bullhead) and stocked fish was -0.893, further supporting asymmetrical competition or predation in shaping communities.

Discussion

Data from our surveys supported our conceptual model of how pond communities varied across gradients of pond size and water permanency. Specifically, we saw a linear decrease in amphibian and crayfish abundance and a linear increase in stocked fish abundance as pond size increased. Stream fish followed a second-degree polynomial curve as pond size increased. Additionally, CCA indicated that one of the main factors influencing community structure was pond size and depth, supporting our predictions that smaller and less permanent ponds would be dominated by amphibians while larger and more permanent ponds would be dominated by fishes. The polynomial response in stream fish abundance across a pond size gradient indicated that there may be an intermediate “sweet spot” in pond size (between 150 and 400 m perimeter) where stream fish are able to colonize and thrive without being suppressed by stocked fishes, which are more likely to be stocked in larger ponds. Observed responses may differ if ponds > 2.0 ha were included in our study, including a potentially non-linear response of amphibians and crayfish due to an extended “tail” of low abundance in large ponds or a plateau of stocked fish abundance in larger ponds. Interestingly, only a single pond did not contain any macroconsumers. That pond was among the smallest and shallowest that we sampled.

This study found 20 fish species including 11 stream fish occupying farm ponds, which is nearly one third of the regional fish species pool that was found in the same study area (61 species; Bruckerhoff et al., 2021). Major groups of fishes from Bruckerhoff et al. (2021) that were not found in farm ponds include species of sucker, darter, madtom, and some large-bodied fish such as gar. Within these groups, species that were highly represented in streams such as Longear Sunfish (*Lepomis megalotis*), Central Stoneroller (*Campostoma anomalum*), and Sand Shiner (*Notropis stramineus*) were underrepresented or absent from ponds. This suggests regional and local processes limit community assembly in ponds (e.g., Smith and Powell, 1971; Tonn et al., 1990). Abiotic conditions and habitat stability likely constrain which species, based on life history characteristics, are able to colonize and survive in pond environments and might explain the absence of riffle or riverine specialists such as darters or suckers. At a finer scale, local processes such as habitat complexity and biotic interactions are more likely to shape communities (Tonn et al., 1990). Lentic habitat can provide breeding habitats for a variety of stream species. For example, beaver ponds provide natural, and potentially important habitat to some stream fishes (Schlosser, 1995; Schlosser, 1998; Snodgrass and Meffe, 1999) and amphibians (Collen and Gibson, 2000; Cunningham et al., 2007; Stevens et al., 2007). Further research is necessary to untangle the relative importance of stream-to-pond connectivity as well as abiotic filters in limiting farm pond communities.

CCA indicated that groupings of stocked and stream fishes both occupied larger and more permanent bodies of water but separated out along a gradient of vegetation and distance from stream. Increased vegetation favored the occurrence of stocked fish. Top-down control of benthic or planktivorous species by stocked piscivores might increase water clarity, favoring the growth of aquatic vegetation (Carpender and Kitchell. 1996; Jeppesen et al., 2012; Potthoff et al., 2008). For example, Potthoff et al. (2008) found that stocked Walleye (*Sander vitreus*) resulted in declines in Bluntnose Minnow which in turn influenced water clarity and vegetation in shallow lakes. Increased occurrence of stream fish with distance from nearest stream seems counterintuitive, suggesting that lower connectivity is associated with presence of stream fish and contradicts other studies reporting that fish species richness was positively correlated with connectivity (i.e., size of inflow channel; Mitsuo et al., 2011). The observed pattern of increasing abundance of stream fish with distance from a stream might be a symptom of increased difficulty stocking ponds with predatory sportfish higher in the watershed. Farm ponds that were built relatively far from streams and primarily used for cattle watering are also not readily accessible to anglers, thus the desire to stock fish is lower in these ponds. Perhaps the reduced probability of colonization of ponds further from a stream is offset by the absence of predatory sport fishes. It is likely the stochastic process that govern colonization and stocking contributed to the relatively high amount of unexplained variation in our CCA. Further, other variables including pond age or stocking history, which we hypothesize is a driver of community structure, could not be included in the model because they are unknown for a majority of the ponds that were sampled.

We found evidence that species interactions are likely influencing pond fish community structure. Studies focusing on amphibians have found that water permanency and fish presence can limit amphibian richness and successful breeding (Cunningham et al., 2007; Sih et al., 2000; Swartz and Miller, 2019). In our study, Western Tiger Salamander (*Ambystoma mavortium*) were never captured alongside fish, but there was overlap between fishes and other amphibians. Residual correlations from the latent variable model suggest negative co-occurrences between many stocked and stream species are independent of abiotic variables (Figure 2.5B). While residual correlations do not necessarily equate to a measure of species interaction, the observed pattern of generally positive co-occurrences within stream or stocked fish but negative co-occurrences between these two groups is compelling. This pattern is not surprising for stocked species such as Largemouth Bass, which can have strong impacts on aquatic communities including reductions in stream fish occurrence and movement (Power et al., 1985; Schlosser, 1988) and crayfish abundance (Hill and Lodge, 1995). Past studies of farm ponds in Japan reported that introduced Largemouth bass were negatively correlated with the presence of native species (Maezono and Miyashita, 2003; Tsunoda and Mitsuo, 2012). A similar dynamic was found with populations of introduced Largemouth Bass in Korean streams (Jang et al., 2006). Conversely, Largemouth Bass presence in stream reaches in our study region was positively associated with overall fish richness after controlling for abiotic variables (Bruckerhoff et al., 2021). This difference likely reflected long-term turnover of stream fish assemblages, including an increase in species adapted to lentic habitats and co-occurrence with Largemouth Bass (Hedden et al., 2022). The present study corroborated Bruckerhoff et al. (2021) and Hedden et al.

(2022) in finding a positive correlation between Largemouth Bass and Bullhead species, suggesting avoidance behavior or prey defenses (e.g., spines) allow bullheads to coexist with Largemouth Bass.

Our survey identified three reproductive populations of federally endangered Topeka Shiner, indicating that ponds might be important for the conservation of native and endangered fish species. In all three cases the communities in ponds with Topeka shiner were dominated by stream fish including Green Sunfish (3 ponds), Orangespotted Sunfish (2), Fathead Minnow (*Pimephales promelas*; 2), and Black Bullhead (2). The co-occurrences with these species are similar to those found in previous studies (Bakevich et al., 2013; Campbell et al., 2016; Minckley and Cross, 1959; Simpson et al., 2019; Thomson et al., 2005), suggesting this suite of species might have a general affinity for lentic habitats, such as livestock watering holes (i.e., dugouts) or stock ponds. Such habitats are known to harbor reproductive populations of Topeka Shiner (Campbell et al., 2016; Thomson et al., 2005). Reproductive pond populations of Topeka Shiner and other species could provide important source populations (Thomson and Berry, 2009) that maintain marginal or sink populations in downstream stream habitats.

Conclusion

We found that a gradient of pond size and permanency results in predictable change in community structure, but negative interactions between stocked and stream fishes can limit suitability for native species. Thus, “Build it and they will come” holds true for farm ponds, the smallest and least permanent were used by amphibians and crayfish while larger and more hydrologically stable ponds were dominated by fish. However, “Stock it and they won’t” also holds true, thus if we are to conserve headwater stream species, especially those that are threatened or endangered, methods to control predatory stocked fishes would be necessary to manage these systems as reproductive source populations.

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Figures

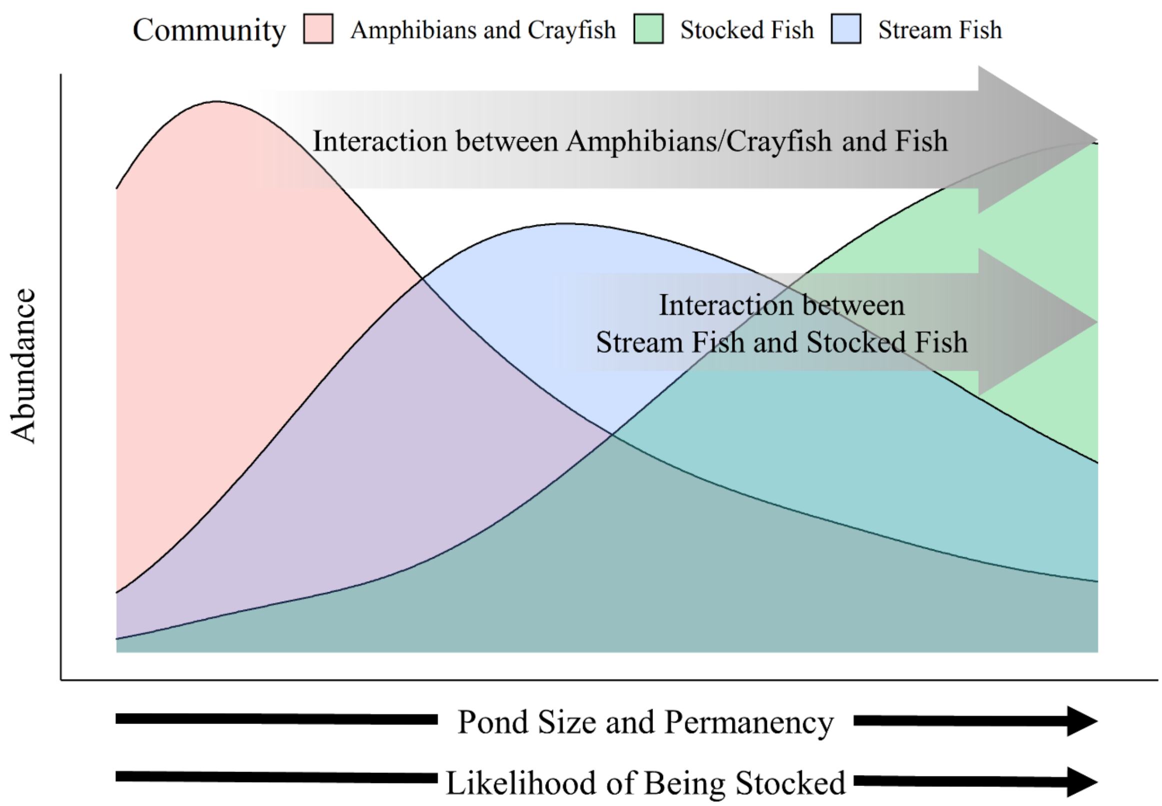

Figure 2.1 Conceptual model of farm pond macroconsumer communities. Pond size and permanency represent a gradient from the smallest and most ephemeral ponds dominated by amphibian and crayfish to large and completely permanent ponds dominated by stocked fish. As pond size and permanency increase so does the likelihood of being stocked. Shaded grey arrows represent expected interactions, with the prediction that interactions among community types increase with pond size, permanency, and likelihood of being stocked.

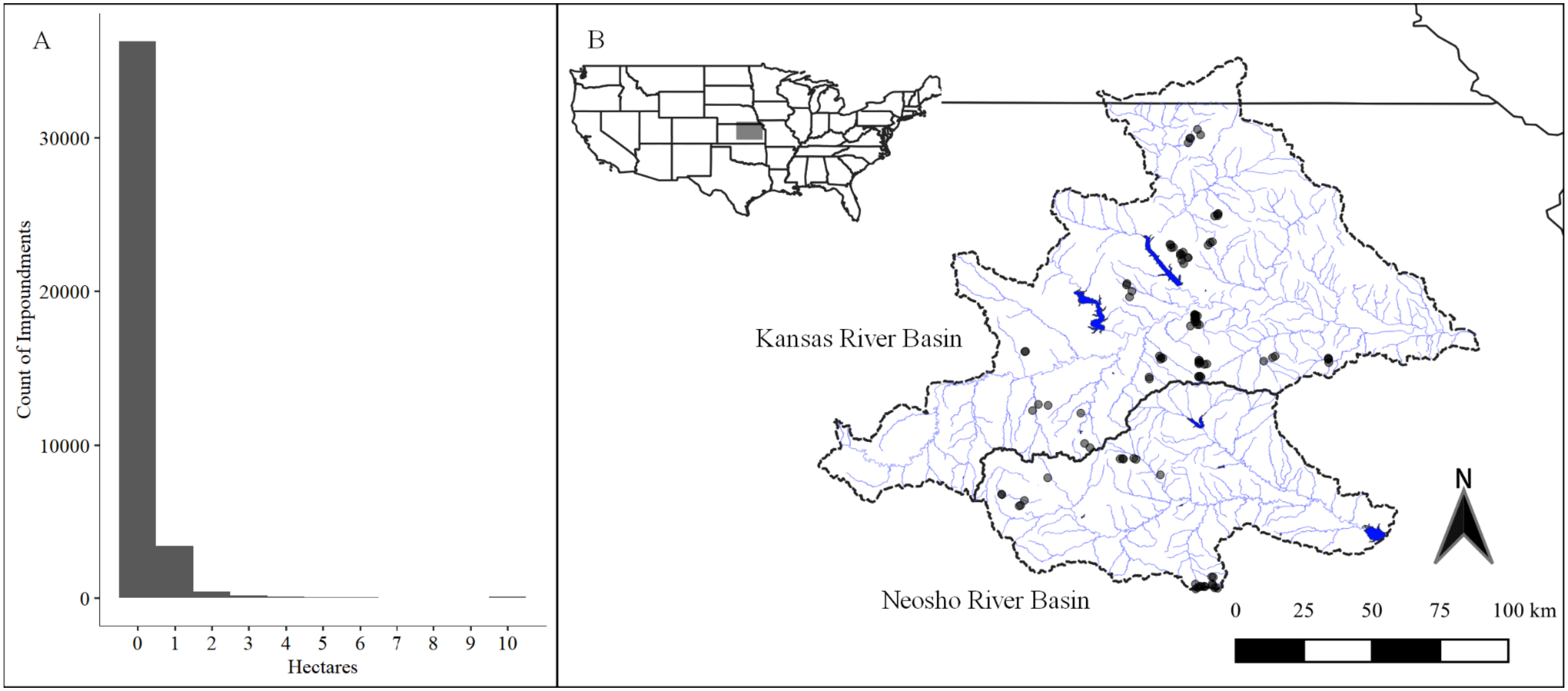

Figure 2.2. (A) Count of impoundments in target area within Kansas based on surface area (hectares). (B) Map of the study area and relative location of sampled farm ponds in the Flint Hills of Kansas, USA. Transparent circles represent sampled farm ponds. The dashed line represents the target area for this study and major river basin boundaries. Blue lines and polygons represent 3rd order streams or higher and major reservoirs. Waterbody shapefile data were accessed from the National Hydrography Dataset (U.S. Geological Survey, 2021a; accessed 13 May 2021 at https://www.usgs.gov/national-hydrography/national-hydrography-dataset). Watershed boundaries were accessed from the Watershed Boundary Dataset (U.S. Geological Survey, 2021b; accessed 16 April 2021 at https://www.usgs.gov/national-hydrography/watershed-boundary-dataset). The map projection is NAD83.

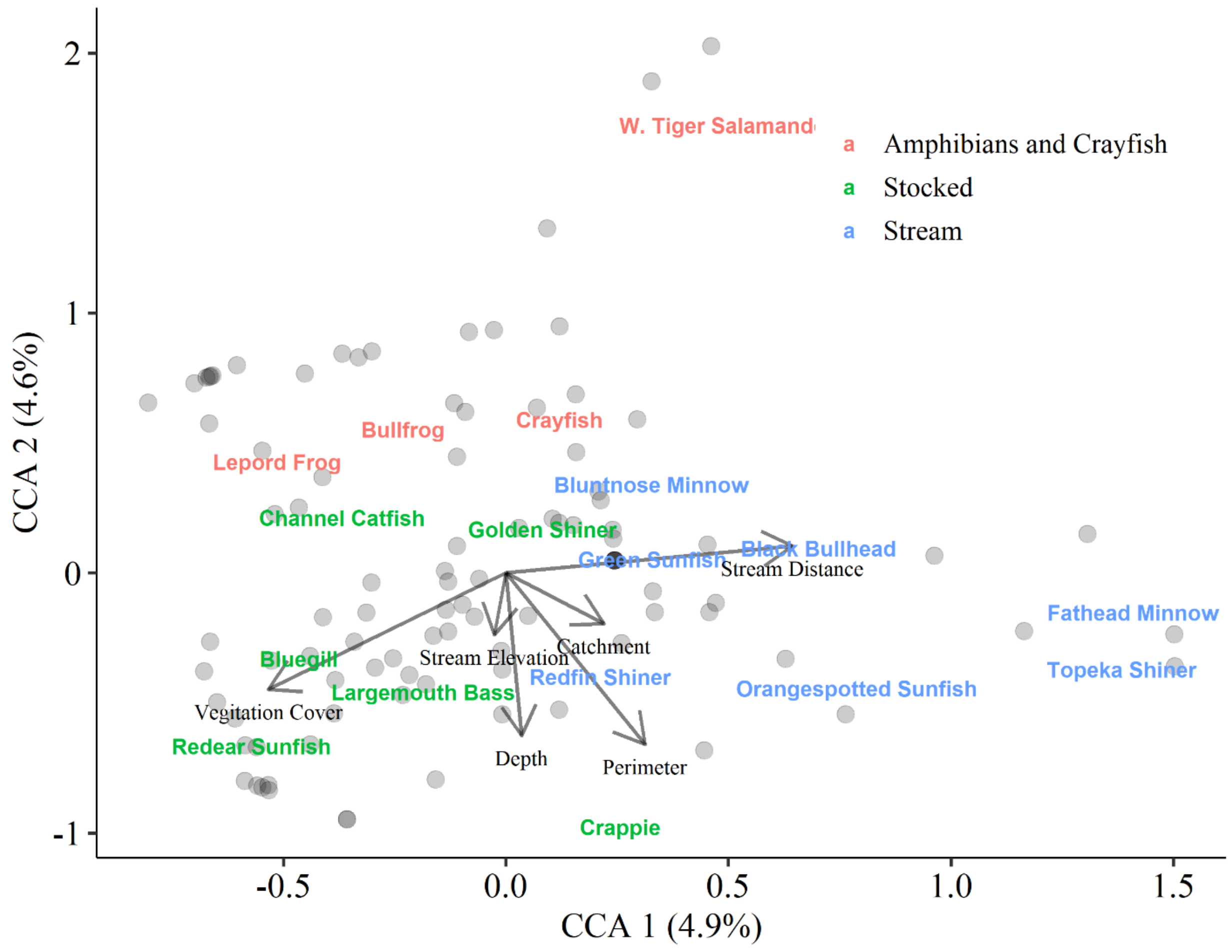

Figure 2.3. Canonical correspondence analysis (CCA) illustrating species-habitat association within farm ponds located in the Flint Hills region of Kansas, USA. Axes I and II were both significant ($P < 0.05$). Explanatory environmental variables are represented as blue arrows and were chosen via model selection. Transparent circles represent each site.

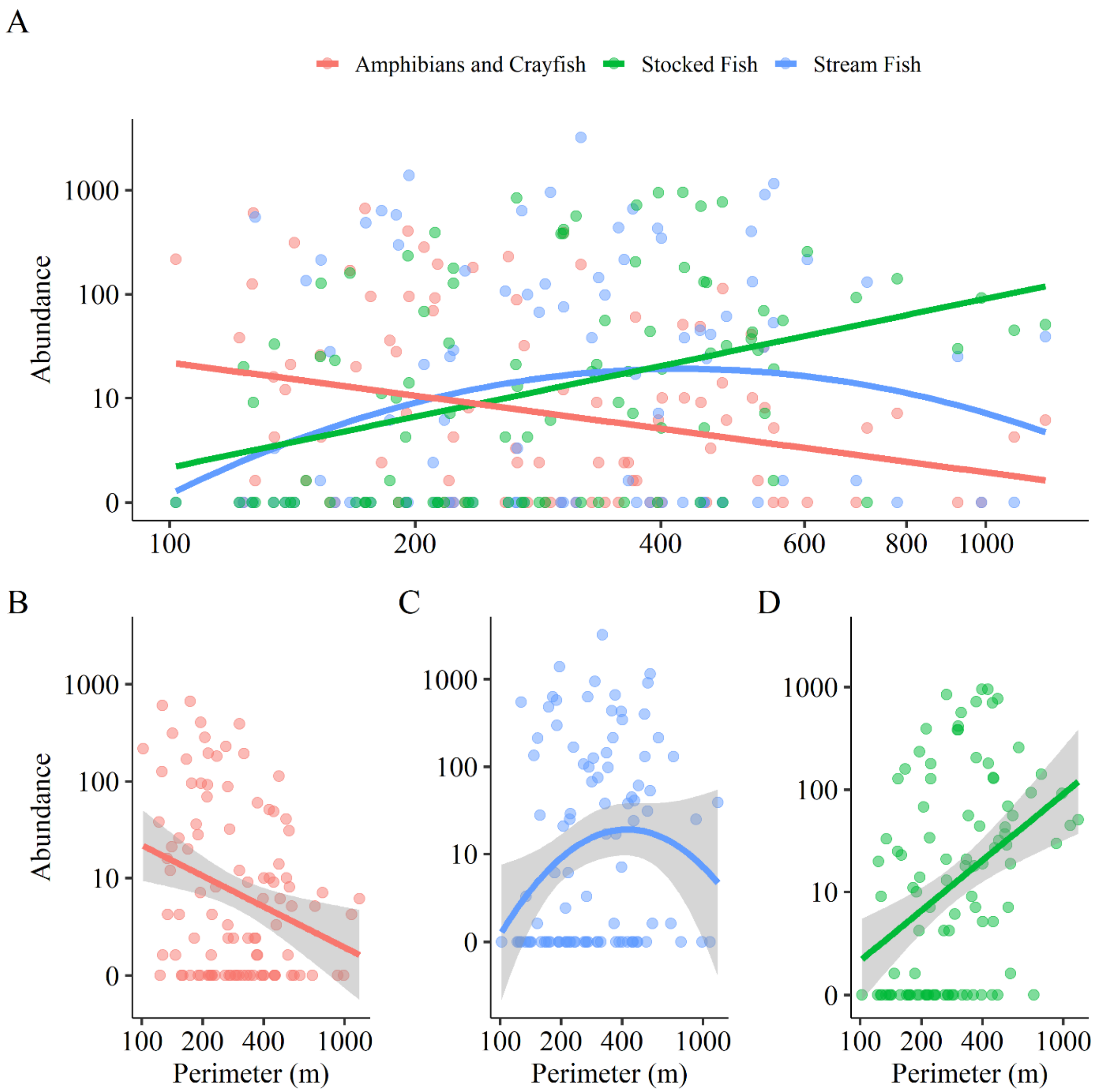

Figure 2.4. Regression plots of all species categories (A), amphibian/crayfish only (B), stream fish only (C), and stocked fish only (D) abundances with pond perimeter. Fit lines were selected and generated from a model selection process and error bars represent standard error. Both axes are represented in a log scale.

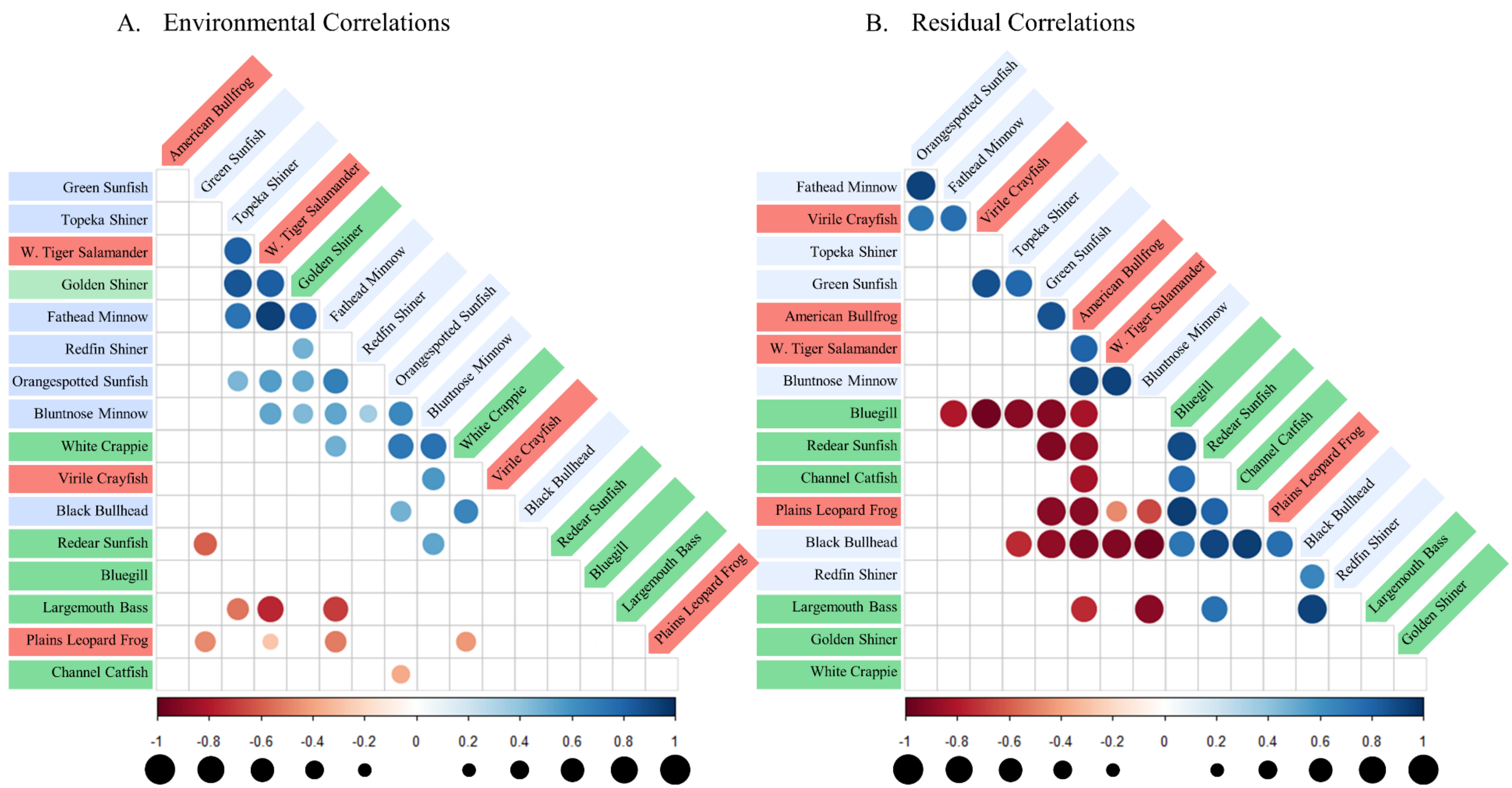

Figure 2.5. Correlation plots of Pearson coefficients between species due to environmental factors (A) and residual correlation (B) based on the joint species distribution model (JSDM). Only significant correlations, based on 95% credible intervals, are displayed. Red indicated negative correlation while blue indicates positive correlations. Circle size indicates strength of the correlation. Species labels are color coded by classification with amphibians and crayfish being red, stream fish being blue, and stocked fish being green. Environmental factors included in the environmental correlations are displayed in Figure 2.3.

Tables

Table 2.1. Species occurrence in sampled ponds in the Flint Hills region of Kansas, USA. Classification represents fish species that might naturally colonize ponds, fish that were stocked, or other macroconsumers that might have colonized ponds.

Species	Common Name	Number of Ponds	Classification
Lithobates catesbeianus	American Bullfrog	38	Amphibian
Lithobates blairi	Plains Leopard Frog	30	Amphibian
Ambystoma mavortium	Western Tiger Salamander	3	Amphibian
Faxonius virilis	Virile Crayfish	41	Crayfish
Lepomis cyanellus	Green Sunfish	49	Stream
Ameiurus melas	Black Bullhead	15	Stream
Lythrurus umbratilis	Redfin Shiner	10	Stream
Lepomis humilis	Orangespotted Sunfish	9	Stream
Pimephales promelas	Fathead Minnow	7	Stream
Pimephales notatus	Bluntnose Minnow	4	Stream
Notropis topeka	Topeka Shiner	3	Stream
Fundulus notatus	Blackstripe Topminnow	2	Stream
Campostoma anomalum	Central Stoneroller	1	Stream
Cyprinella lutrensis	Red Shiner	1	Stream
Semotilus atromaculatus	Creek Chub	1	Stream
Micropterus salmoides	Largemouth Bass	45	Stocked
Lepomis macrochirus	Bluegill	38	Stocked
Notemigonus crysoleucas	Golden Shiner	17	Stocked
Ictalurus punctatus	Channel Catfish	12	Stocked
Pomoxis annularis	White Crappie	9	Stocked
Lepomis microlophus	Redear Sunfish	4	Stocked
Gambusia affinis	Western Mosquitofish	2	Stocked
Pomoxis nigromaculatus	Black Crappie	2	Stocked
Cyprinus carpio	Common Carp	1	Stocked

Table 2.2. Model comparisons for the effect of log pond perimeter on log abundance of community components including amphibians and crayfish, stream fish, and stocked fish. Null intercept-only model, linear, second-degree polynomial, and third-degree polynomials models were fitted for each response. The best fit model is denoted in bold. AICc, ΔAICc, residual standard error (SE), adjusted r^2, and *p* value are displayed for each overall model.

Community Component	Model	AICc	ΔAICc	Residual SE	Adjusted r^2	p value
Amphibians and Crayfish	Intercept-only	424.4	6.22	1.989	-	< 0.001
	Linear	**418.2**	**0**	**1.917**	**0.071**	**0.004**
	2^{nd} Degree Polynomial	419.3	1.16	1.917	0.070	0.011
	3^{rd} Degree Polynomial	421.4	3.20	1.925	0.063	0.027
Stream Fish	Intercept-only	475.2	1.94	2.564	-	< 0.001
	Linear	474.6	1.39	2.542	0.016	0.106
	2^{nd} Degree Polynomial	**473.2**	**0**	**2.510**	**0.041**	**0.048**
	3^{rd} Degree Polynomial	475.4	2.21	2.523	0.031	0.110
Stocked Fish	Intercept-only	450.3	14.25	2.264	-	< 0.001
	Linear	**436.0**	**0**	**2.096**	**0.142**	**< 0.001**
	2^{nd} Degree Polynomial	437.3	1.23	2.097	0.141	<0.001
	3^{rd} Degree Polynomial	439.3	3.30	2.106	0.134	<0.001

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Chapter 3 - Predator presence influences survival and behavior of translocated stream fish in ponds

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Abstract

Small artificial impoundments such as farm ponds have recently been recognized as potential habitat for threatened native fish species. However, factors influencing translocation or colonization success into these environments, including connectivity to stream networks and interactions with existing fish community, are largely unknown. In this study we conducted a controlled experiment to quantify the influence of piscivorous Largemouth Bass (*Micropterus salmoides*) on the survival of a translocated native minnow species. We translocated a total of 1600 Bluntnose Minnow (*Pimephales notatus*) into replicate treatment ponds with and without Largemouth Bass in the summers of 2020 and 2021. Each minnow was implanted with a passive integrated transponder (PIT) tag. Translocated populations were monitored using stationary and mobile PIT antennas and estimates of apparent survival and probability of detection for each pond were derived from open population mark-recapture models. Apparent survival was nearly two times higher in ponds without bass suggesting predation by bass leads to higher mortality. Additionally, probability of detection was nearly 10 times higher in ponds without bass, suggesting reduced movement of translocated minnows when bass were present. While the direct effect of mortality impacts translocated populations, the indirect effect of altered behavior may

also be impactful on translocation success. These results suggest that Largemouth Bass limit the success of translocated or naturally colonizing minnow species.

Keywords: Endangered Species, Predator-Prey Interaction, Fish, Passive Integrated Transponder Tags, Mark-Recapture, Cormack-Jolly-Seber

Introduction

In many areas of the world, especially ranching or agricultural areas, freshwater ecosystems have been altered by the construction of small impoundments or farm ponds. Past studies have linked farm ponds to declines in native species, primarily through the stocking and subsequent escape and spread of predatory sport fish (Shrank et al. 2001; Taylor et al. 2001; Perkin et al. 2016; Hedden et al. 2018). Slowing these declines will require novel conservation strategies. One such approach in species conservation in regions with high densities of farm ponds might be the translocation or repatriation of threatened species to these environments coupled with control of sportfish communities, thereby establishing self-sustaining populations that buffer losses due to anthropogenic alterations (Copp et al. 2007; Thomson and Berry 2009). Translocation of wild individuals from stable populations can be a useful tool to increase the redundancy of populations and prevent local extirpation while maintaining genetic diversity (Minckley 1995; Fischer and Lindenmayer 2000; Seddon et al. 2007).

The concept of our study arose from our efforts to conserve the federally endangered Topeka Shiner (*Notropis topeka*) in the Flint Hills region of Kansas. Topeka Shiner inhabit small, headwater streams, however, they also use in-channel and near-channel natural (e.g., oxbows) or anthropogenically created (e.g., excavated ponds also known as dugouts and impoundments) habitats in the Flint Hills (Hedden et al. 2021) and other parts of their range

(Thomson and Berry 2009, Pierce et al. 2019). Topeka Shiner often nest-associate with sunfish, particularly Orangespotted Sunfish (*Lepomis humilis*) that might improve their nesting success (Campbell et al. 2016). Further, when pond environments are periodically connected to streams, Topeka Shiners might escape farm ponds and supplement stream populations (Hedden et al. 2021).

A major hurdle to establishing populations of native fish in ponds is due to negative interactions with stocked predators. Topeka Shiner rarely coexist with predators such as Largemouth Bass (*Micropterus salmoides*; Shrank et al. 2001, Mammoliti 2002, Campbell et al. 2016) and avoid them in an experimental setting (Knight and Gido 2005). However, the impact of predator-prey relationships varies across environmental conditions and spatial and temporal scales. Bass presence in lakes negatively impacts the abundance of some cyprinid species (MacRae and Jackson 2001; Jackson 2002) and can cause local extirpations (Kimberg et al. 2014; Van Der Walt et al. 2016; Pereira and Vitule 2019). In stream environments, bass presence can also decrease overall species richness (Van Der Walt et al. 2016), though not in all cases (Bruckerhoff et al. 2021). These impacts can result from mortality from direct consumption (Schlosser 1988; Harvey 1991; Steinmetz et al. 2008; Marsh-Matthews et al. 2013) or non-consumptive or indirect effects such as alterations in behavior which influences survival (Skalski and Gilliam 2002, Asaeda et al. 2007) or reproduction (Fraser and Gilliam 1992; Sheriff et al. 2020). Indirect effects may impart a stronger interaction than direct effects such as consumption (Holomuzki et al. 2010). Translocation of native fishes to farm ponds may be of great conservation benefit for some species, but limitations on translocation success have not been thoroughly tested and are based on mostly correlative data (Lamothe and Drake 2019).

Experiments that track the success of translocated native fish into ponds are necessary to better understand factors limiting their success in these habitats. Although we were interested in testing limitations in translocation success of federally endangered Topeka Shiner, we conducted experiments with Bluntnose Minnow (*Pimephales notatus*) as a surrogate because its populations in the region are more stable. Both species are known to naturally colonize ponds (Hedden et al. 2021, P. Pfaff, unpublished data) and share similar feeding (Kansas Fishes Committee 2014) and reproductive life histories, including nest association (Stark et al. 2002). Additionally, Bluntnose Minnow show similar responses to predator presence as Topeka Shiner (Knight and Gido 2005) and other native minnow species (Bruckerhoff et al. 2021) in controlled mesocosm experiments.

The specific goal of this study was to quantify the impact predator presence has on translocation success of Bluntnose Minnow. Although we suspected mortality associated with predation would be high, we wanted to first quantify apparent survival of translocated minnows between ponds with and without bass. Secondly, we tested if behavior of translocated minnows differs between bass and non-bass ponds by means of probability of detection. Four possible outcomes from our experiment could infer the impact of predators on translocation success of native minnows. (1) If survival and probability of detection do not differ between treatments then presence of bass does not increase mortality or alter behavior of translocated fish. (2) If survival is lower in bass treatments while probability of detection does not differ among treatments then the presence of bass leads to mortality of translocated fish while not altering their behavior. (3) If survival does not differ among treatments while probability of detection is lower in bass treatments then the presence of bass does not lead to mortality of translocated fish but alters behavior. (4) If survival and probability of detection are both lower in bass ponds, then the presence of bass leads to both mortality and altered behavior of translocated fish. We predicted that bass presence would impact both apparent survival and probability of detection of translocated Bluntnose Minnow.

Methods

This study took place in 2020 and 2021 at the Tallgrass Prairie National Preserve (TAPR, as designated by the National Park Service) located in central Kansas. TAPR is located within the Upper Cottonwood River basin and contains 24 farm ponds. Ten of the 24 ponds contain populations of Largemouth Bass. In 2020, we used four ponds that contained bass (ponds 3, 10, 15, and 19) and four control ponds that did not contain bass (ponds 5, 8, 9, and 22; Figure 3.1) to test if bass affected survival and behavior of Bluntnose Minnow. Two of the control ponds completely dried (ponds 4 and 8) and one pond nearly went dry (pond 9) during summer 2020. We replicated the experiment the following year using the same bass ponds while substituting two non-bass ponds more likely to maintain water permanency (ponds 4 and 11 were included while 5 and 8 were no longer used). However, pond 9 experienced total drying resulting in loss of all individuals in 2021. As such, this pond was excluded from analysis. Ponds included in this study were of similar size and habitat structure. Control ponds averaged 0.7 ha (standard deviation 0.2) while bass ponds averaged 0.64 ha (0.3). All ponds were vegetated around the margin with the level of vegetation increasing though the summer. No ponds contain wood or have overhanging trees. Additionally, all ponds are primarily silt substrate with pockets of courser substrate near the dam. All ponds except for pond 11 contain Green Sunfish (*Lepomis cyanellus*). Pond 11 contained Black Bullhead (*Ameiurus melas*) while ponds 3 and 22 also contained Bluegill (*Lepomis macrochirus*). All ponds except pond 10 were in grazed areas and accessed by cattle.

Each pond was stocked with 100 Bluntnose Minnows captured from a robust and naturally occurring population (pond 11) and implanted with 8 mm passive integrated transponder (PIT) tags (Mini HPT8 Pit Tag, Biomark, Boise, Idaho). PIT tags were implanted following standard procedures that yield high survivorship and tag retention on individuals with a total length ≥ 46 mm (Pennock et al. 2016, Schumann et al. 2020). Implanted individuals were monitored for one hour and individuals that did not survive were replaced. Overall, survival of minnows from tagging to release over the two-year study was 98.6 percent.

Two weeks after translocation or release of minnows, we began to monitor populations with submersible PIT antennas to quantify detections over time. The first year we used a single circular (1 m diameter) submersible PIT tag antenna (RM310, Biomark, Boise, Idaho) with a read range of approximately 20 cm with 8 mm PIT tags. Antennas were placed at approximately 1 m depth near the corner of the dam in each pond. Antennas were deployed in four of the eight ponds for one week and rotated to the next four ponds for one week, this process repeated after four weeks for a total of three sampling periods. Sampling occurred between May and August 2020. Because of low detections in bass ponds, we extended the duration of submersible antenna deployment for all ponds in 2021 to two weeks and included four sampling periods such that each antenna was deployed every six weeks from May to September. Additionally, we built a kayak-mounted PIT antenna that was designed to actively detect fish in small ponds. We did this to explore the possibility of fish being present but not active within ponds. Rather than using a floating antenna, as is common in other studies (Zentner et al. 2021), our antenna swivels from the front of the kayak allowing it to dip down through the water column and sample a depth of up to 2 m (Figure B.1). The antenna was constructed using stranded wire and enclosed in a frame of 2.54 cm PVC tubing. The antenna was composed of two separate inductance loops each measuring 2 m by 0.66 m and held together with a 0.66 m long PVC tube segment. This gave us an effective size of a 2 x 2 m antenna though the use of two inductance loops made it much easier to detect PIT tags moving through the middle of the antenna. The antenna was powered using a Biomark IS1001 PIT tag reader and 24-volt lithium-ion battery fitted into a custom waterproof enclosure. The read range of the antenna using 8 mm PIT tags was approximately 25 cm depending on tag orientation. Each pond was sampled using the kayak antenna prior to submersible deployment in the first three sampling periods.

Differences in the number of fish detected between treatments and ponds were visualized with accumulation curves. We also used Cormack-Jolly-Seber mark-recapture models (Pledger et al. 2003) using Program MARK (White and Burnham 1999) to estimate apparent survival and probability of detection across redetection periods. The Cormack-Jolly-Seber model is an open population live recaptures model allowing for changes in population size through mortality and emigration. We selected this model because apparent survival, through mortality or emigration, and probability of detection were the estimates of interest in this study. Model selection in Program Mark is done using Akaike Information Criterion corrected for small sample size (AICc). Prior to model selection, goodness-of-fit (GOF) was tested on the full model using the median c-hat function in Program Mark to estimate overdispersion, which may indicate model assumptions are violated (Cooch and White 2010). C-hat values between one and three indicate minor overdispersion, which require the use of a quasi-likelihood adjusted AICc (QAICc; Cooch and White 2010). Two non-bass ponds experienced total or partial drying in 2020 leading to no Bluntnose Minnow being detected in pond 5 in periods 2 and 3 and no detection in pond 7 in period 3. We were unable to model apparent survival and probability of detection in 2020 due to a lack of replication and overall detections. While our systems are open and periodically connected to the stream when pond level is high enough, no such overflow occurred during the second year of the study. Therefore, we believe the estimate of apparent survival to be indicative of true survival. Additionally, we are treating all detections as live individuals. Using stationary antennas, the only way to detect a “ghost tag” (a shed tag or a dead individual) would be if the tag was near the antenna and would be continually detected until the antenna was moved. We did not encounter such an event during this study. Apparent survival and probability of detection were modeled as being constant or differing between treatment as per our hypotheses. We assumed that declines in apparent survival in bass ponds would reflect predation and lower probability of detection would reflect reduced movement in the presence of bass.

We calculated the amount of time an individual Bluntnose Minnow was detected at an antenna within a single day to further explore if behavior was different among treatments. PIT antennas were set to detect unique individuals every 60 seconds providing a fine scale temporal resolution of fish presence around the antennas. Thus, a greater number of detections of a single individual within a day should reflect less activity because the individual stayed in the same location (i.e. close to the antenna) for a greater amount of time. All plots were created in program R version 4.1.1 (R Core Team 2021).

Results

The total number of detected individuals in non-bass ponds was an average of 20 times greater than bass ponds across both years (Figure 3.2). Only two Bluntnose Minnows, both in the first redetection period, were detected in bass ponds in 2020. In contrast, 73 individuals were detected in control ponds across all time periods. In 2021, 15 individual Bluntnose Minnow, 12 in the first period and three in the fourth period, were detected in bass ponds. In contrast, 252 individual Bluntnose Minnow were detected in control ponds across all time periods including 51 in the fourth period alone.

Median c-hat estimation of the full model (1.88) indicated minor overdispersion and QAICc was used for model comparison. The top selected Cormack-Jolly-Seber model included apparent survival and probability of detection differing by treatment (Table 3.1). The model estimates indicate that apparent survival and probability of detection were both higher in control ponds than in bass ponds (Figure 3.3). Apparent survival for control ponds (76%) was approximately double that of bass ponds (44%). Probability of detection in control ponds (42%) was ten times that of bass ponds (4%). The 95% confidence intervals in both cases were non-overlapping between treatments.

Bluntnose Minnow were more active in control ponds than in bass ponds. Activity was strongly bimodal in bass ponds, with individuals averaging 165.5 minutes per day within detection range of the antenna on days they were detected (Figure 3.4A). In control ponds, activity was more uniformly distributed averaging only 5.9 minutes per day within detection range of the antenna. Excluding the first redetection period, when individuals may have still been acclimatizing to the ponds, the contrast between treatments increases to 436.0 minutes in bass ponds and decreases to 4.7 minutes in control ponds (Figure 3.4B).

The mobile antenna detected only 16 individuals across all pond and time periods, 8 individuals in bass ponds and 8 individuals in control ponds. Patterns of detections from the mobile antenna, albeit much less successful than stationary antennas, were consistent and suggested less minnow activity in bass ponds. Of the eight individuals detected in bass ponds with the mobile antenna, we only detected one on a stationary antenna while in control ponds only one of the eight detected individuals was not detected on a stationary antenna. These data support the hypothesis that lower detections in bass ponds is at least partially due to reduced activity leading to lower detection probability.

Discussion

Our results indicate that survival of translocated Bluntnose Minnow was nearly twice as high in ponds that do not contain bass (76% in control ponds and 44% in bass ponds). Although direct predation of cyprinids by bass can strongly influence cyprinid survival (Schlosser 1988; Harvey 1991; Van Der Walt et al. 2016), relatively few studies have directly assessed true survival or apparent survival rates of translocated individuals and none in treatments of habitats with or without predators. Apparent survival in control ponds was higher than other studies with suckers (between 0 and 12% over a six-month period; Webber and Haines 2014) and chub (between 22 to 41% estimated annual apparent survival; Spurgeon et al. 2015). These studies took place in lotic systems and over varying lengths of time and did not control for presence or absence of predators which could account for the lower apparent survival rates.

In this study, we assessed only the influence of predator presence on apparent survival; however, observed differences among control ponds, including permanency, highlight the importance of abiotic factors to consider when selecting ponds for translocation. For example, pond 9 was the smallest among control ponds in 2021 and went dry resulting in death of the translocated population by redetection period four. In 2020, two of the control ponds went dry resulting in the loss of all translocated individuals. Other inter-pond factors that may influence survival or continued fitness include habitat and water quality. Yates et al. (2019) found that temperature and pH can limit fitness of small, translocated fish populations. Similarly, of 148 Gila topminnow (*Poeciliopsis occidentalis*) translocations, habitat of translocation site was one of the most important factors predicting persistence time of these populations (Sheller et al. 2006). While translocation site habitat factors may still play an important role in translocation success, direct predation influenced minnow survival in this study and differences between treatments were greater than that within treatments when excluding ponds that dried.

Our results also suggest that predator presence might influence the behavior of translocated minnows. Bass presence led to lower probability of detection compared with control ponds. Additionally, detected individuals in bass ponds spent 28 times longer within antenna range than those in control ponds indicating less total movement. This pattern intensified over the course of the study. Excluding the first redetection period, individuals detected in bass ponds spend nearly 100 times longer within antenna range than those in control ponds. Further, the redetection distribution in bass ponds was bimodal. No individuals redetected in period one for less than 10 minutes per day were redetected in subsequent periods. Moreover, fish detected on mobile antennas were much less likely to be captured on stationary antennas in bass ponds relative to control ponds. Those individuals that survived past the first time period may have modified their behavior to remain sheltered and less active. This finding is consistent with past literature. Knight and Gido (2005) found significant differences in habitat use and response to bass by four cyprinid species, including both Bluntnose Minnow and Topeka Shiner, in a mesocosm experiment. Further, they found that when cover was present, Bluntnose Minnow would become inactive and stay within cover exclusively. Similarly, Bruckerhoff et al. 2021 found significant reductions in movement of Red Shiner (*Cyprinella lutrensis*) and Bluntnose Minnow in mesocosms when bass were present. Changes in behavior when predators are present can influence overall movement and foraging and spawning success (Fraser and Gilliam 1992; Katano and Aonuma 2002; Divino and Tonn 2007; Peterson and Kitano 2021). Thus, mortality and altered behavior present a challenge to any minnow population that is translocated into or naturally colonizes a pond containing bass.

The mobile antenna was more difficult to glean useful information from because it was less efficient at detecting fish and because it might detect tags that were from dead fish (i.e. ghost tags). Regardless, a majority of individuals detected in bass ponds were not detected on stationary antennas (seven of eight) while the same pattern was reversed in control ponds with a majority of individuals also being detected on stationary antennas (seven of eight). This could be due to detecting ghost tags in bass ponds due to higher mortality and a greater number of such tags being present. Given the relatively narrow read range of our antenna and silty substrate in the ponds, we do not suspect this is the case, as those tag would likely settle into the sediment at the bottom of the pond. Alternatively, it could be due to having a higher number of unique detections in control ponds, thus more mobile detections would be represented in the stationary antenna data. This, coupled with lower minnow movement in bass ponds, would lead to the observed pattern. Lentic pond environments may not be conducive to mobile PIT antenna monitoring because fish are able to move away from the antenna in any direction. Mobile PIT antennas are commonly used in lotic systems that are by nature more constrained, though detection rates can be highly variable across studies (Musselman et al. 2017; Zentner et al. 2021). PIT tag studies in lentic estuary environments tend to have low probability of detection: 1.0 to 4.0% (Ledgerwood et al. 2006), 3.3 to 4.8% (Morris et al. 2015), 0.9 to 3.5% (Morris et al. 2018), 0.7 to 2.2% (Holcombe et al. 2019). Additionally, sampling efficiency in ponds is likely low due difficulty in moving the mobile antenna over and through vegetation cover. Overall, this method was not effective in lentic systems but did, nonetheless, provide some support for our hypothesized behavioral differences between ponds.

Another limitation of the present study is that we only consider interaction with a single predator and do not consider other predatory species such as Green Sunfish (*Lepomis cyanellus*), bullhead, or avian predators that might have been present in or around these ponds. Among the control ponds, Pond 4 and Pond 22 contained Green Sunfish while Pond 11 contained Yellow Bullhead (*Ameiurus natalis*). While Green Sunfish may influence survival rates of some cyprinids (Marsh-Matthews et al. 2013), ponds 4 and 22 had the highest apparent survival rates among all ponds, including fishless ponds (5, 7, and 9). In addition to predation from other fish species, avian predators act as structuring mechanism in shallow habitats (Power 1984, Power et al. 1989). The number of fish removed by avian predators accounted for more than those consumed by piscivorous fish in shallow eutrophic lakes (Winfield 1990). Bird exclusion experiments indicate that avian predators may even be the top predators in a variety of habitats including shallow lentic systems (Steinmetz et al. 2003). Further, altered behavior due to avian predation threat can lead to less foraging and slower growth in cyprinids (Allouche and Gaudin 2001). All ponds included in this study, however, were of similar depth and likely had similar rates of avian predation. The shallowest ponds included in this study (5, 7, and 9) would likely have the greatest risk of avian predation, though habitat loss through drying poses a greater risk to translocation failure as we observed.

This study suggests the presence of bass negatively impacted translocated minnow apparent survival and behavior and suggests translocation efforts or natural colonization of ponds with Largemouth Bass are not likely to be successful. In addition to mortality through direct predation, altered behavior might change habitat use, feeding behavior, and spawning success, all of which can negatively influence the persistence of translocated populations. These indirect effects may be stronger and more impactful to translocated population survival than direct effects through predation. While it is important to take predator presence into account when selecting translocation sites, other factors such as habitat stability also need to be considered. Translocation is a powerful conservation tool and selecting optimum habitats is key in determining translocation and conservation success for fish species such as Topeka Shiner. Further, removing existing bass from small ponds may increase the habitat available to Topeka Shiner for unassisted colonization, thereby further helping in the conservation of this species.

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Figures

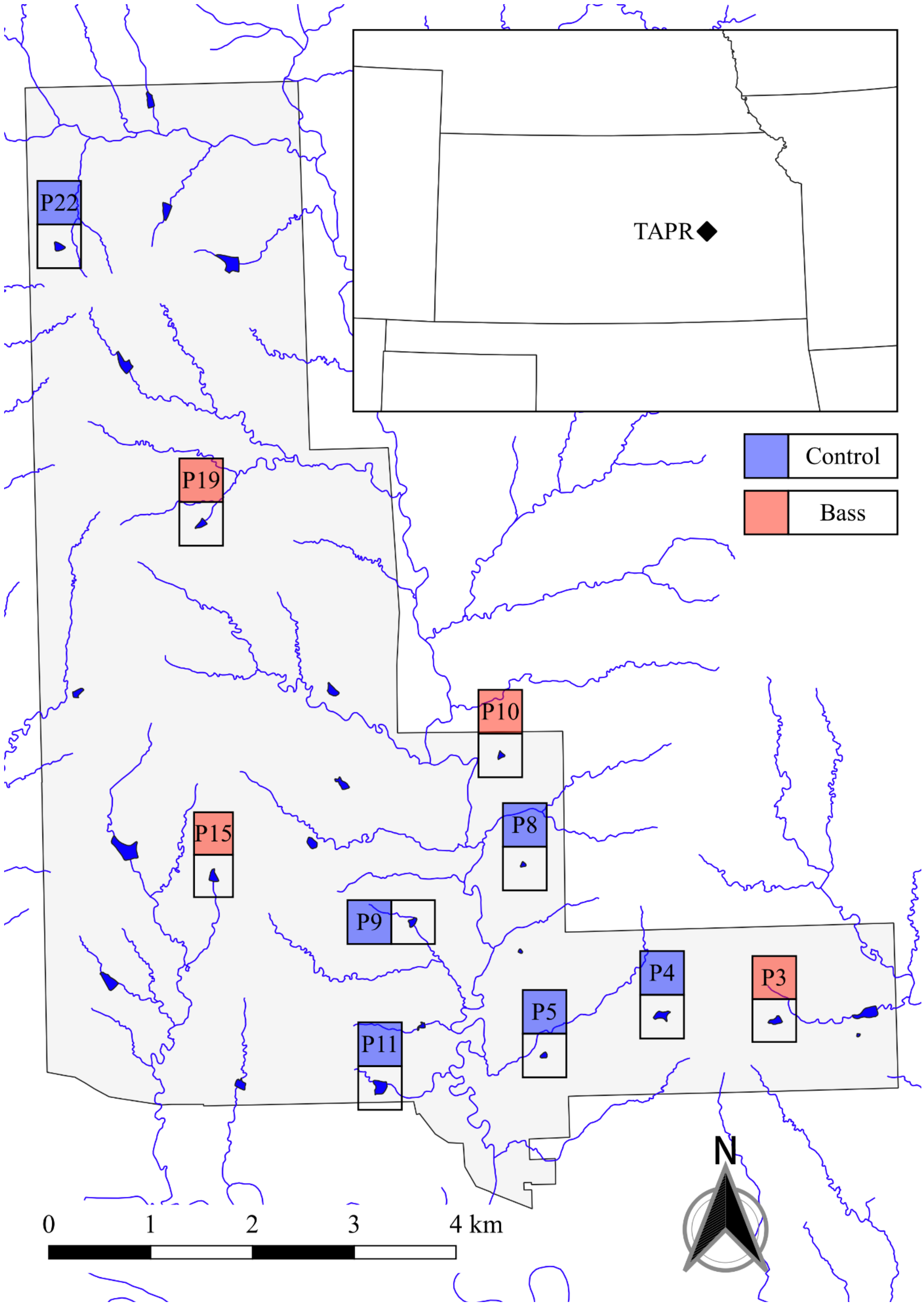
TAPR
Control
Bass
P22
P19
P10
P15
P8
P9
P11
P5
P4
P3
N
0
1
2
3
4 km

Figure 3.1. Map of ponds on Tallgrass Prairie National Preserve (TAPR) and its location within Kansas (subset). Each pond used in this study is outlined with a black box and labeled with pond number. Red coloration indicated ponds that contain bass while blue indicates a control pond.

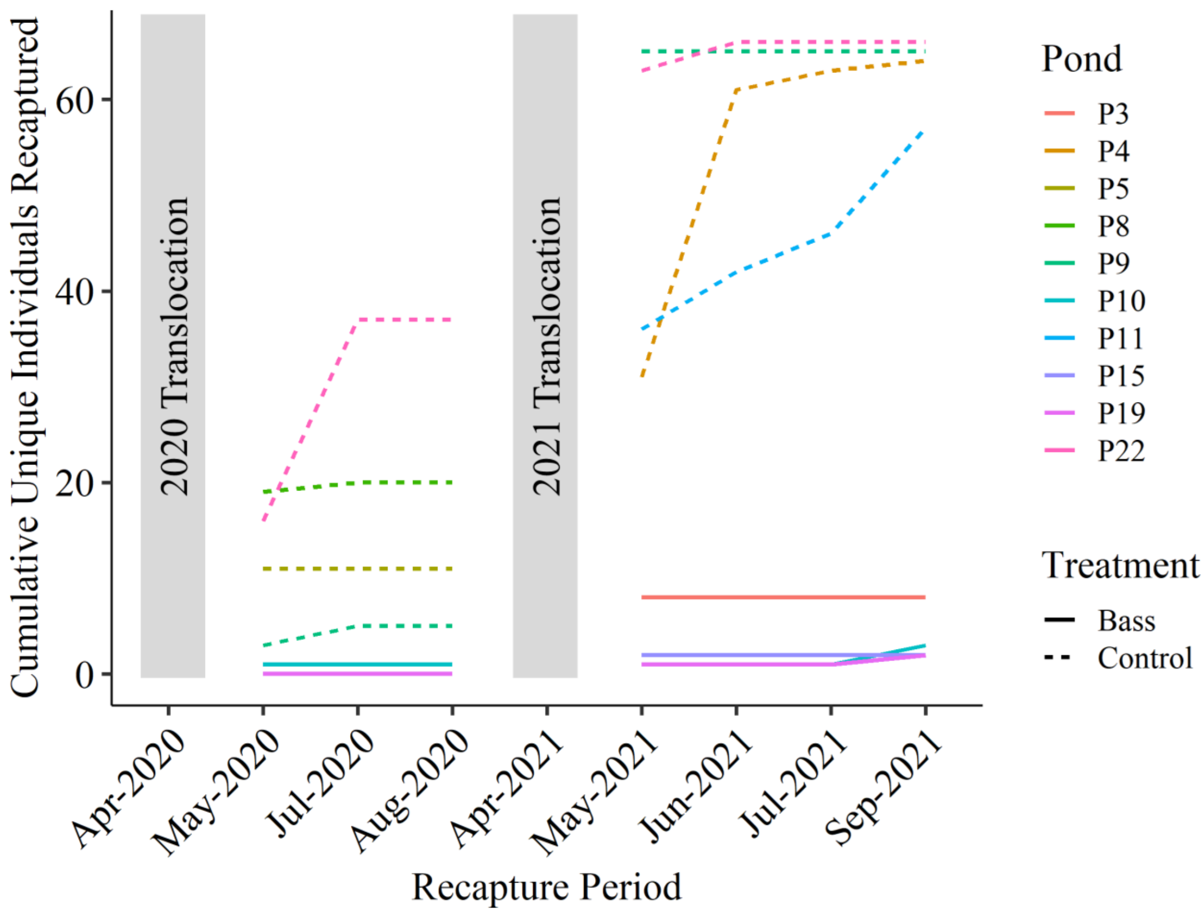

Figure 3.2. Cumulative unique individual detections of Bluntnose Minnow by pond, treatment, and year across detection periods. 2020 had three detection periods and 2021 had four. Relative timing of yearly translocations are represented with gray bars. Color indicates pond number while line type indicates treatment.

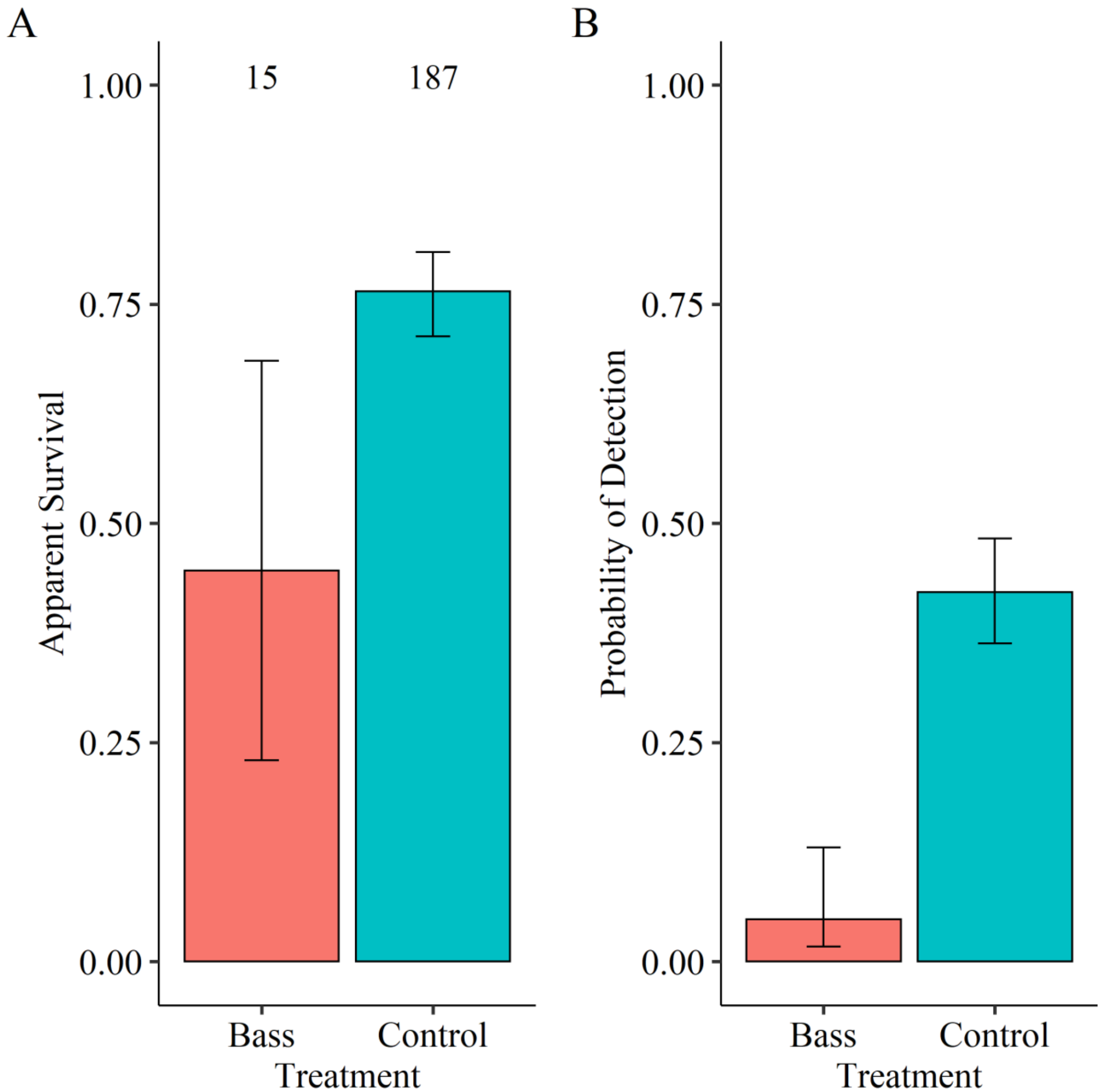

Figure 3.3. Estimates and 95% confidence intervals of (A) apparent survival and (B) probability of detection for treatments in 2021 from the top selected Cormack-Jolly-Seber model. Total unique redetections in treatments are represented at the top of plot A.

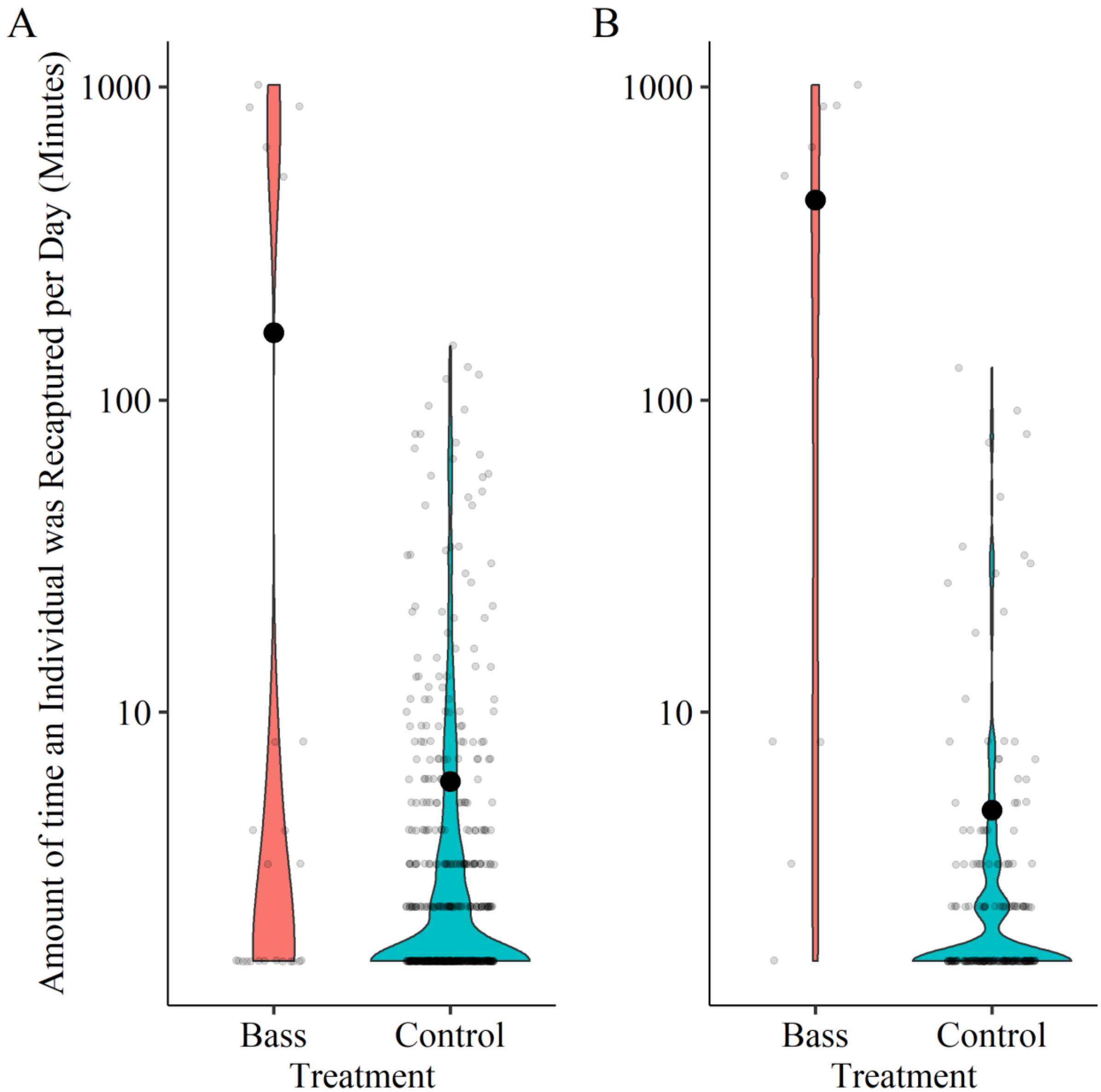

Figure 3.4. Violin plots of the amount of time individual Bluntnose Minnows were redetected on an antenna per day. Plots distinguish different treatments and panels represent (A) all redetection periods and (B) redetection periods 2, 3, and 4. The large black dot represents the average. The small transparent dots represent the raw data for each individual on each day.

Tables

Table 3.1. Results of model outputs ranked by QAICc used to determine estimates of apparent survival probability (AS) and probability of detection (Detection) of Bluntnose Minnow in treatment ponds with or without Largemouth Bass present.

Model	ΔQAICc	Model Likelihood	Number of Parameters	Deviance
AS (treatment) Detection (treatment)	0	1	4	216.358
AS (null) Detection (treatment)	2.1872	0.335	3	220.562
AS (treatment) Detection (null)	19.2555	0.0001	3	237.631
AS (null) Detection (null)	317.9875	0	2	538.375

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Chapter 4 - Survival, reproduction, and dispersal of translocated Topeka Shiner (*Notropis topeka*) in prairie streams and ponds

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Abstract

Translocation can be an important tool for conservation, allowing managers to bolster the resilience of existing population or repatriate populations into formerly occupied areas. Prior to broad scale translocation efforts, it is essential to understand factors associated with establishment success. Species residing in headwater prairie streams within the Great Plains of North America are particularly vulnerable to extirpation and might benefit from translocation efforts. This region also includes a large number of small ponds, which might support threatened populations of stream animals. In this study, we assessed the efficacy of using translocation as a conservation tool for the federally endangered Topeka Shiner (*Notropis topeka*). From 2019 to 2022, Topeka Shiner were captured from a naturally-colonized pond population on the Tallgrass Prairie National Preserve in central Kansas and translocated to two nearby ponds and a nearby stream. A portion of all translocated Topeka Shiner were implanted with a passive integrated transponder (PIT) tag, and some tagged individuals were returned to the source population (Pond S). Apparent survival estimates between our 6-week sampling intervals were 88% in renovated Pond PFT (predator-free translocation pond) compared to 71% in Pond T (translocation pond) and the source population. Apparent survival of fish translocated in Schoolhouse Creek was 54%. Declines in apparent survival between sampling intervals with high-flow events suggest emigration from ponds and the redetection of a tagged fish indicated immigration into ponds during high-flow events. Finally, we verified reproduction of translocated fish by the occurrence of juveniles in years following translocation. Because translocated Topeka Shiner were able to survive, reproduce in, and disperse from ponds and stream habitats, our results suggest this approach is a viable and low-cost conservation strategy to both bolster populations and repatriate areas from which Topeka Shiner were extirpated.

Keywords: Headwater Steams, Great Plains, Fish, Endangered Species

Introduction

Translocation of endangered species can bolster existing populations or repatriate them to previously occupied or restored habitats. Translocating wild individuals from stable populations is preferred since they are acclimated to a natural environment and can establish new populations that maintain genetic diversity (Minckley 1995; Fischer and Lindenmayer 2000; Seddon et al. 2007). Freshwater systems, especially intermittent and headwater streams and the species living there, are frequent targets of translocation events where populations have experienced declines or extirpation (Sheller et al. 2006; Groce et al. 2012; Schumann et al. 2012; Archdeacon and Reale 2020; Hickerson et al. 2021). Moreover, translocation is a means to mitigate dispersal limitations and associated problems in fragmented systems (Hugueny et al. 2011, Perkin and Gido 2012). Using translocated fish to increase the number of populations and thus decrease the extinction risk of the species as a whole (Fagan et al. 2009) is especially important in hydrologically variable habitats, such as headwater streams that are prone to periodic drying (Whitney et al. 2016; Hedden and Gido 2022).

Translocation of endangered species might also occur in novel habitats without natural occurrence of those species. Specifically, farm ponds in the Great Plains of North American provide habitat for native stream fish that colonize during flood events or are stocked, intentionally or unintentionally (Pfaff et al. 2022). Recently, managers have begun to look into the conservation value of small impoundments for fish (Copp et al. 2007; Tarkan et al. 2009; Tarkan et al. 2011) and amphibian (Knutson et al. 2004; Swartz and Miller 2019) populations. The use of farm ponds might have broad conservation potential because they occur across much of the agrarian areas of the world. Millions of ponds are broadly distributed across the North American Great Plains (Renwick et al. 2005) and a majority are less than a hectare in size (Chumchal et al. 2016). While landowners stock many ponds with predatory sport fish, many are left fallow and used primarily for cattle watering or irrigation and might be suitable targets for conservation.

Topeka Shiner (*Notropis topeka*) are native to the Great Plains of North America and were federally listed as endangered in 1998 (Tabor 1998). Historically, Topeka Shiner was present across portions of eight states within the Mississippi River basin. They were thought to primarily inhabit clear and cool low-order prairie streams, especially pools (Minckley and Cross 1959; Fishes of Kansas Committee 2014), but recently populations were reported from lentic habitats that periodically connect with streams or rivers to facilitate dispersal (Thompson et al. 2009; Campbell et al. 2016; Osterhaus et al. 2022). Fragmentation and predators (e.g., Largemouth Bass, *Micropterus salmoides*) have negatively affected Topeka Shiner in streams (Deacon 1961; Tabor 1997; Shrank et at. 2001). Conservation strategies for Topeka Shiner include oxbow rehabilitation, which involves dredging of built-up sediments to increase water depth and connectivity to groundwater (Kenney 2013, Simpson et al. 2019). These efforts are aimed at providing suitable habitat and rely on natural colonization; however, stocking Topeka Shiners in rehabilitated oxbows has been proposed (Simpson et al. 2019). Within Kansas, captive breeding populations in experimental ponds has been successful (S. Collins, Kansas Biological Survey, personal communication), but no repatriation efforts have occurred until the effort reported here. Topeka Shiner can naturally colonize and reproduce in farm ponds and the use of these habitats has conservation potential especially in light of translocation and establishing source populations (Pfaff et al. 2022). However, little is known about how well fish translocated from ponds might perform in natural stream habitats or other ponds.

The objectives of this study were to quantify: (1) the survival of Topeka Shiner translocated to pond and stream environments, (2) movement into or out of ponds during high flow events, (3) dispersal after translocation to stream environments, and (4) reproduction in stream or pond environments following translocation. To accomplish these objectives, we conducted a three-year translocation survival and detection study on Tallgrass Prairie National Preserve using an existing and stable population of Topeka Shiner located on the preserve. Our results will help inform managers and prioritize conservation options for threatened species occupying dynamic prairie streams that are subjected to numerous stressors ranging from land conversion to altered hydrologic regimes associated with many factors including climate change. If farm ponds can serve as viable habitats for refuge populations of Topeka Shiner and translocations of these fish into nearby streams is successful, these efforts would likely increase populations sizes and stabilize declining populations across the range of this species.

Methods

Sampling and translocation sites were on the Tallgrass Prairie National Preserve (TAPR) located in the central flint hills region of Kansas (Figure 4.1). TAPR is 44.1 km^2 preserve owned by The Nature Conservancy and managed by the National Parks Service as a remnant of historically widespread tallgrass prairie in the Great Plains of North America. TAPR is located within the Cottonwood River basin and contains 25 farm ponds. These ponds support 16 species of fishes including seven cyprinids, two ictalurids, six centrarchids, and one percid. One monitored population of Topeka Shiner historically occurred in Schoolhouse Creek; a small tributary of Fox Creek located in the central portion of TAPR (Figure. 4.1; Whitney et al. 2016). Following a severe drought in 2012, the population was greatly diminished and by 2014 only seven individuals were captured. In 2016 a single individual was captured and, because none were captured in the following years (K. Gido, unpublished data), they were presumed to be extirpated from this creek. In 2014, a farm pond on TAPR, Pond S (source population), was found to contain Topeka Shiner (Portofee et al. 2018) and provided an opportunity to re-establish a population in Schoolhouse Creek, as well as other ponds in the watershed. Common species in Schoolhouse Creek include Central Stoneroller (*Campostoma anomalum*), Cardinal Shiner (*Luxilis cardinalis*), Redfin Shiner (*Lythrurus umbratilis*), Bluntnose Minnow, Creek Chub (*Semotilus atromaculatus*), Green Sunfish, Longear Sunfish (*Lepomis megalotis*), and Orangethroat Darter (*Etheostoma spectabile*). Additionally, Largemouth Bass have been present at Site 5 and downstream since 2014. Target ponds for translocation included Ponds T (translocation pond) and PFT (predator-free translocation pond; Fig. 1). Pond T is near the source population and flows into the same small stream, South Headquarters Intermittent Tributary. Pond T supports Red Shiner (*Cyprinella lutrensis*), Bluntnose Minnow, Fathead

Minnow, Green Sunfish, and Orangespotted Sunfish (*Lepomis humilis*). Pond PFT is located in the Schoolhouse Creek watershed, and, in November 2020, rotenone was applied to remove Channel Catfish prior to stocking Topeka Shiner along with three Orangespotted Sunfish. All three ponds have similar depth, vegetation, substrate, and size (0.7 to 0.8 ha) and are permanent. Additionally, they all share a similar depth (about 2 meters), vegetation cover, and silty substrates with pockets of courser substrates near the dam. Ponds T and S are occasionally accessed by cattle while Pond PFT is occasionally accessed by bison.

Topeka Shiner were implanted with an 8mm passive integrated transponder (PIT) tag (Biomark, Boise, Idaho) to assess survival and movement of the source and translocated populations. PIT tags were implanted following standard procedures on individuals 46 mm or longer (Pennock et al. 2016, Schumann et al. 2020). During the course of this study, 430 Topeka Shiner were implanted with PIT tags and the survival of individuals from tagging to release was 100%. Only relatively small numbers of Topeka Shiner (up to 200 per year) were translocated due to concerns of depleting the source population. Tagging and translocation took place in March or April of each year except for 2019 where individuals were translocated in November.

To estimate the survival of translocated Topeka Shiner, tagged individuals were monitored using submersible (RM310, Biomark) and permanent solar-powered PIT receivers (IS1001, Biomark). Duplicate permanent solar-powered receivers were maintained in the downstream portions of Schoolhouse Creek and South Headquarters Intermittent Tributary to detect individuals that emigrated from these tributaries toward Fox Creek (Figure 4.1). These antennas were running nearly constantly from March to October or November each year. Occasionally high flow events pushed antennas to the stream margin, but they were promptly returned to their original locations. To estimate survival within ponds and potential dispersal, each pond was monitored using submersible antennas to detect PIT tagged individuals remaining in these habitats. We could not continuously monitor each habitat because of a limited number of antennas; thus, two antennas were deployed at standard locations in each pond for two weeks and those antennas were rotated to Schoolhouse Creek for two weeks. Antennas were deployed in ponds for three, two-week periods in 2020 and four, two-week periods in 2021 and 2022. Schoolhouse Creek was monitored with submersible receivers at 10 sites in 2020, but we reduced this to 6 sites in 2021 (excluded sites 1 to 4) and 6 in 2022 (excluded sites 2 to 4 and 10). These differences were due to submersible receiver availability and stocking locations. All receivers were set to detect unique individuals once every minute and had a read range on 8 mm tags of 20 to 25 cm.

To assess survival of Topeka Shiner translocated to pond and stream environments, Cormack-Jolly-Seber open mark-recapture models (CJS; Pledger et al. 2003) were fit using Program MARK (White and Burnham 1999). CJS models estimate apparent survival, which includes populations losses due to either mortality or emigration, and probability of detection. Models were fit for fish translocated to ponds. Apparent survival and probability of detection in ponds were modeled as being constant over time, changing over time, differing between ponds, differing between cohorts (individuals translocated in the same year), and each combination of these parameters. Models were also fit for fish translocated to Schoolhouse Creek by combining redetections for all sites within one period, as there were not sufficient numbers of detections to consider each site independently. For Schoolhouse Creek, apparent survival and probability of detection were each modeled as being constant over time, changing over time, differing between cohorts, and changing over time and differing between cohorts. Models were assessed using Akaike Information Criterion corrected for small sample size (AICc). Goodness-of-fit (GOF) was tested on the full model for each group of models using the median c-hat function in Program Mark. GOF testing is used to estimate overdispersion, which can occur when model assumptions are violated (Cooch and White 2010). A c-hat value of one or less indicates no overdispersion while a value between one and three indicates minor overdispersion which can be accounted for by adjusting the likelihood yielding a quasi-likelihood adjusted AICc (QAICc; Cooch and White 2010). Additionally, apparent survival parameters were assessed from models where apparent survival and probability of detection were constant across time for each pond and for Schoolhouse Creek to compare how overall apparent survival differed among habitats.

To assess pond connectivity and Topeka Shiner dispersal from pond or stream environments in response to high flow events, we first used time-lapse cameras to quantify overflow events in Ponds T and S that might allow dispersal in or out of these ponds. CJS models over time were used in conjunction with overflow events to assess if apparent survival declined between periods separated by an overflow event, indicating potential emigration. Finally, recapturing or detecting individuals in the stream or a different pond from the one where they were released would be direct evidence of movement from the pond, presumably during high flow events.

Finally, to assess if translocated Topeka Shiner reproduce in stream or pond environments, we sampled ponds and streams with a 1.8 x 4.5, 3.1-mm mesh seine in fall each year to evaluate size structure of populations and identify the presence and abundance of young-of-year individuals through the use of length-frequency histograms. Each pond was sampled with four 20 m seine hauls at set location including the pond inlet, side-banks, and both shallow and deep transects parallel to the dam. Sites in Schoolhouse Creek were sampled with one or more seine hauls until all available habitats were sampled. In Kansas, Topeka Shiner typically reproduce in late may or early June (Kerns and Bonneau 2002). Typical young-of-year size range by fall is 25mm while age-one average 40mm. The smallest size of translocated individuals in 2021 and 2022 was 46mm. As such, we will focus on physically captured individuals 45mm or less to assess reproduction in those years.

Results

Redetection and CJS Models

Redetection numbers indicate that 57.5% of released PIT tagged Topeka Shiner in ponds were redetected (Table 4.1). Some individuals persisted and were continually redetected for up to two years. This was most evident in Pond PFT where 10 of 50 PIT tagged individuals were redetected in period 4 of 2022. Median c-hat estimation of the full pond model (1.06) indicated minor overdispersion and QAICc was used. The top CJS model for ponds from model comparison indicated that apparent survival differs over time and among ponds while probability of detection only differed among ponds (Table 4.2). Model estimates showed that Pond PFT had higher apparent survival than the other two ponds (Figure 4.2A). Further, each pond followed a similar trend with a gradual decrease in estimated number of individuals until the end of summer (T4). This pattern held true for individuals from the previous cohort where they generally persist until T4 the following year and then were no longer detected. Pond PFT had a higher probability of detection (0.92, 95% CI 0.88- 0.95) than Ponds T (0.44, 0.37-0.52) and S (0.39, 0.31-0.18) as indicated by non-overlapping 95% confidence intervals (Figure 4.2B).

Redetection numbers in Schoolhouse Creek were lower than ponds (43.5%) despite the use of additional PIT antennas and being an in a more constrained space (Table 4.3). Sixteen of 51 translocated individuals from the 2019/20 cohort were redetected and a few individuals persisted and were redetected for a year and a half. Nineteen of 25 from the 2021 cohort and 22 of 55 from the 2022 cohort were redetected in subsequent periods. No individuals from the 2021 cohort were redetected in 2022 and no individuals from the 2022 cohort were redetected by period 4 in 2022. Median c-hat estimation of the full stream model (0.91) indicated no overdispersion. The top CJS model for Schoolhouse Creek from model selection indicates that apparent survival differed across time periods in which antennas were deployed while probability of detection was constant among cohorts and over time (Table 4.4). Apparent survival estimates indicate that the number of individuals from the 2019/20 cohort remained relatively constant for nearly two years while the number of individuals from the 2021 and 2022 cohorts decreased sharply after the first redetection period in both years (Figure 4.3A). Probability of detection (95% CI) was estimated to be 0.55 (0.43 - 0.68) (Figure 4.3B).

Overall apparent survival estimates were highest in Pond PFT (estimate 0.88, lower 95% confidence interval 0.84, upper 95% confidence interval 0.91), similar in Ponds T (0.71, 0.65, 0.76) and S (0.71, 0.66, 0.76), and lowest in Schoolhouse Creek (0.54, 0.46, 0.61) (Figure 4.4).

Overflow Events and Movement in and out of Ponds

Ponds T and S experienced overflows in July 2020 and June 2022. The 2020 event occurred during a single heavy rainstorm where 14.2 cm of rain fell at the MESOWEST weather station on TAPR in a 24-hour period. The 2022 overflow occurred on three consecutive evenings during a multi-day period of sustained rainfall with 12.8 cm of rainfall over a week. Both events led to increased discharge in the Cottonwood River (Figure C.1). The overflow in 2020 coincided with a very low apparent survival rates in ponds T (0.25, 0.12-0.46) and S (0.14, 0.05-0.33) between periods 4 and 5. The effects of the overflow in 2022 on apparent survival was more difficult to assess directly due to the low numbers of 2019/20 and 2021 cohort individuals pond T and S cohorts estimated to be remaining. The apparent survival estimates for Pond S 2022 cohort between release and period 1 was 0.43 (0.24-0.63) which was lower than immediate post stocking estimates for all other Pond S cohorts. Pond PFT did not overflow, with little change in apparent survival following this event.

One Topeka Shiner that was PIT tagged and released in Pond S in 2022 was detected in Pond T following the overflow in June 2022. Shortly after its release, TAPR experienced several days of sustained rainfall that caused Ponds T and S to connect to the stream. Pond S connected periodically for three consecutive days. The exact duration of connection cannot be verified as overflow typically began at dusk and continued overnight. The exact time water receded was not captured on the time-lapse cameras. During this period of connection, the individual was able to move from Pond S into Pond T and was redetected in two subsequent periods on our submersible PIT antenna.

Movement within Stream

Data from PIT antennas deployed at set locations along Schoolhouse Creek suggested restricted movement of fish within the first few weeks following translocation. In the first redetection period following translocation, 90% of redetected individuals (57 of 63) were redetected at release locations. One individual from the 2019/20 cohort and three individuals from the 2022 cohort moved downstream from release location with one of these individuals being detected on the downstream antennas near the confluence of Schoolhouse Creek and Fox Creek. One individual from each cohort moved upstream from release location. No matter release location, nearly all individuals that were redetected in more than one period were detected at Site 7 in Schoolhouse Creek. This one location had more redetections than all other sites combined.

Schoolhouse Creek experienced two large floods in July 2020 and June 2022. Apparent survival estimates from the 2019/20 and 2021 cohorts changed very little between the periods surrounding flooding in both 2020 (0.87, 0.35, 0.99) and 2022 (0.71, 0.38 0.90) (Figure 4.3A).

Physical Capture and Reproduction

In 2021, Topeka Shiner were physically captured in Pond PFT (101 total, 1 PIT-tagged individual), Pond S (227, 1), and Pond T (6; 0). No Topeka Shiner were physically captured in Schoolhouse Creek in 2021. In 2022, Topeka Shiner were physically captured in all habitats: Pond PFT (84, 0), Pond S (43, 1), and Pond T (16, 0), and Schoolhouse Creek (17; 0). Analysis of total lengths of physically captured individuals shows that a large proportion of total physical captures were young-of-year or age-1 (<45 mm; Figure 4.4).

Discussion

Our results indicate that Topeka Shiner translocated from a pond population were able to survive, reproduce, and disperse from and to ponds through the stream network. Apparent survival from Pond T and 26 were similar or higher than that of the source population, suggesting the potential to establish stable, translocated populations. Specifically, Pond T and S had very similar overall apparent survival estimates and contain similar species compositions. Predators were removed from Pond PFT prior to stocking and saw a higher overall apparent survival estimate than the other ponds. Although we are unaware of other studies reporting apparent survival of fish in farm ponds, in a companion study, we found similar apparent survival rates (0.76) of Bluntnose Minnow translocated to ponds not containing Largemouth Bass on Tallgrass Prairie National Preserve (P. Pfaff unpublished data). The estimated number of remaining individuals as calculated from apparent survival estimates diminished with time and declines in Ponds T and S coincided with overflow events suggesting some individuals dispersed from ponds during periods of overflow. Further, no decline was seen in Pond PFT following the overflow which occurred in only Ponds T and S. One individual was detected moving from Pond S to Pond T during the overflow in 2022 confirming that Topeka Shiner are able to both emigrate from and immigrate to ponds through connected streams. Collectively, these results suggest translocations are not only a viable method to establish pond populations, but those populations might "seed" nearby streams and other ponds after establishment.

Apparent survival from ponds was generally higher than that of Topeka Shiner translocated to Schoolhouse Creek, though in both habitat types some tagged individuals from different cohorts survived for over one year following translocation. This indicates that either mortality in the stream is higher than the ponds or that greater connectivity allows individuals to disperse more readily thereby lowering apparent survival through emigration. Only four individuals stocked in Schoolhouse Creek were detected dispersing more than two sites upstream or downstream from the release location, indicating a majority of fish did not move far from their release site. The rare movement of tagged fish from the release site is typically of tagging studies of stream fishes (Skalski and Gilliam 2000). All individuals that persisted in Schoolhouse Creek for more than two periods were present at Site 7, either being translocated to or dispersing to that site, suggesting some affinity for the habitat at this site. Accordingly, 19 of 25 PIT tagged individuals released in 2021 at Site 9 were redetected in the release pool during the first redetection period. By period 2, only two individuals were redetected at Site 9 while three were detected at Site 7 where they persisted during periods 3 and 4 indicating a downstream movement by few individuals. Contrary to apparent survival rates in ponds following overflow or high flow events, no sudden decrease was observed in Schoolhouse Creek.

Young-of-year Topeka Shiner were detected in Ponds PFT and T in both 2021 and 2022 and in Schoolhouse Creek in 2022 indicating successful reproduction. While the present study represents just two ponds and one stream, successful reproduction at all sites is higher than other studies assessing reproduction in translocated populations of Plains Topminnow (*Fundulus sciadicus*) (35% of sites; Schumann et al. 2017) and Arkansas Darter (*Etheostoma cragini*) (11% of sites; Groce et al. 2012). Stock populations of Topeka Shiner in small artificial ponds or impoundments readily reproduce (Campbell et al. 2016), however, there are no direct reports of reproduction of Topeka Shiner naturally occurring in lentic water. Simpson et al. (2018) reported juveniles being more abundant within oxbows than other habitats, indicating oxbows are favorable for reproduction or juvenile survival. Restored oxbow might provide great breeding conditions for Topeka Shiner and are likely sources of new individuals in the system (D. Osterhaus, New Mexico State University, personal communication). Topeka Shiner are known to reproduce in stream pools in association with Green Sunfish and Orangespotted Sunfish (Cross and Collins 1995, Stark et al. 2002) and are thought to require clean, coarse substrates such as gravel (Fishes of Kansas Committee 2014). The three ponds in this study are primarily silt bottomed with limited coarse substrates and might require sunfish nest associations, so eggs are not smothered in silt (Mammoliti 2002). Moreover, the three Orangespotted Sunfish stocked in Pond PFT successfully reproduced: 145 individuals were captured while seining in 2021 and 51 in 2022.

Applying our findings to other regions or species might consider the limitations of our study, foremost being a lack of replication across ponds and streams. Tracking multiple translocation events in different contexts such streams of different sizes or community members would help identify factors that either lead to successful or failed translocation efforts. That was not possible in this study due to the nature of translocating an endangered species and permitting restrictions. Additionally, our data on dispersal downstream was equivocal and we suspect there was a low probability of detecting fish that may have moved downstream during high flow events. During such events, antennas are often pushed aside or buried in sediments, which is compounded with the stream increasing in depth and width, further decreasing the probability of detection. Construction of larger and replicated PIT antennas downstream would better quantify rates of emigration downstream. This is most evident with the 25 fish from the 2022 cohorts of Topeka Shiner released into Schoolhouse Creek at site 1, which were never redetected and were presumed to have moved downstream.

Farm ponds hold much promise for establishing source populations of Topeka Shiner in regions with where natural populations are low or extirpated. The main limitations for the use of farms ponds include connectivity, the presence of predators, and getting buy-in from landowners since a majority of farm ponds are on private land. Farm ponds are only directly connected to streams during overflow events following heavy rain. In this study, two such events occurred over a three-year period. Further research is needed to directly quantify farm pond and stream connectivity and how that might influence dispersal. Dispersal ability is important in understanding colonization potential in a metapopulation framework. The presence of predators can influence the survival and behavior of minnows translocated to pond environments (P. Pfaff unpublished data). Within Kansas, Largemouth Bass and other predators such as Black Crappie

(*Pomoxis nigromaculatus*) commonly inhabit farm ponds (Pfaff et al. 2022) and removal efforts may be necessary prior to Topeka Shiner translocation or other conservation efforts. Our results suggest newly restored ponds might obtain the highest survival rates for Topeka Shiner. Finally, working with landowners and potentially incentivizing the use of their ponds for conservation is likely important to ensure a broad spatial distribution of ponds for conservation.

Our study highlights the potential of using ponds as fountainheads of dispersal for Topeka Shiner. Establishing replicated populations in ponds or other off-channel habitats connected to streams with diminished or extirpated populations holds promise to stabilize declining populations, restore populations in its former range and potentially help recover this species (Osterhaus et al. 2022). Moreover, farm pond populations might provide readily accessible sources for translocations given they are easily sampled, and abundance are high. Translocating wild individuals from nearby populations to ponds to increase redundancy is a low-cost strategy that may be the best option to conserve this and potentially other species that can survive in pond environments.

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permit No. TE067729-4, and Kansas State University Institutional Animal Care and Use Committee permit No. 4494.

Figures

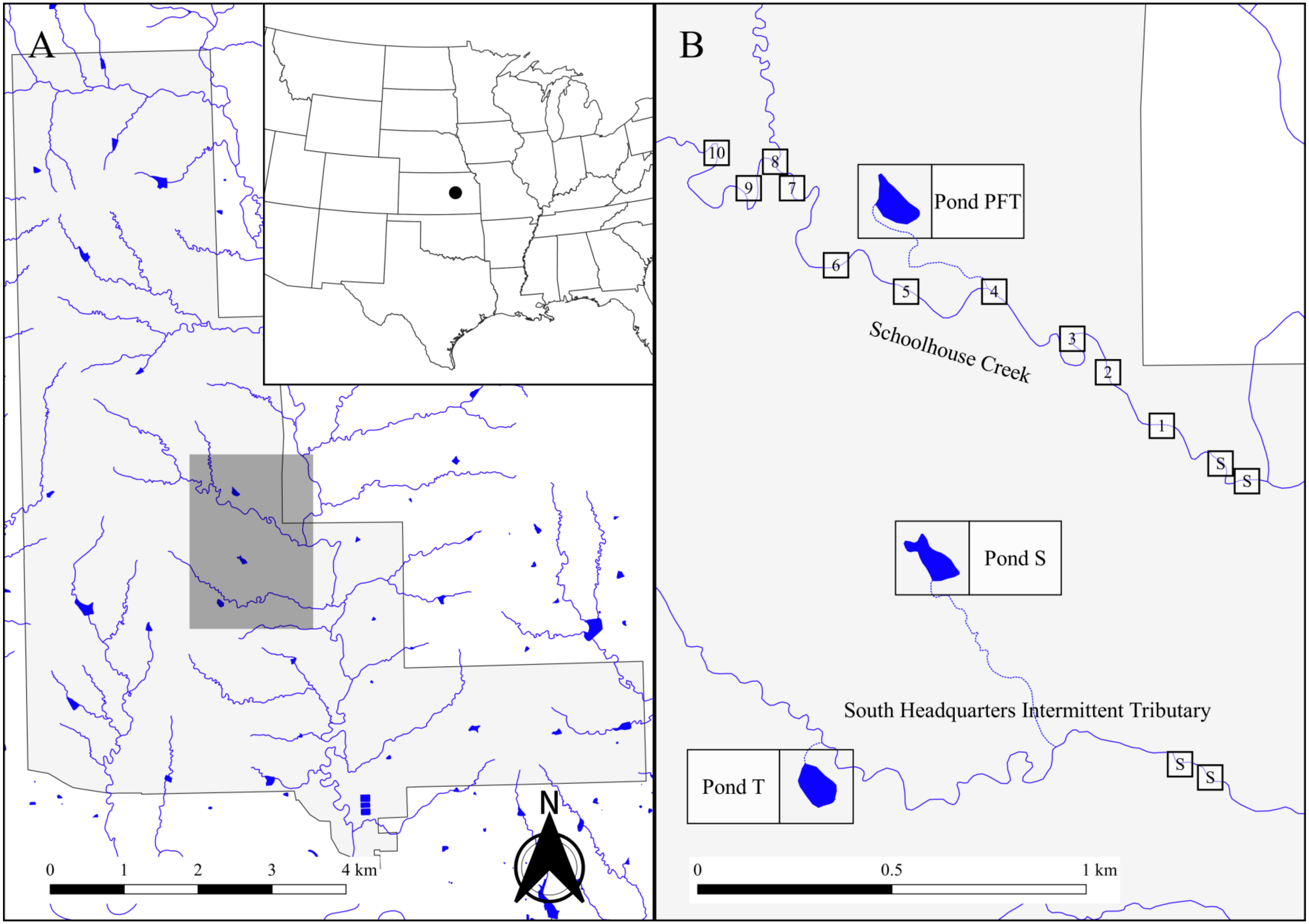
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Figure 4.1. (A) Map of Tallgrass Prairie National Preserve with shaded box outlining the area of the detailed map in panel B. Position within the United States represented by a black dot (insert). (B) Location of study ponds, stream along with PIT tag antenna locations. Schoolhouse Creek (top) and South Headquarters Intermittent Tributary (bottom) are represented with blue lines. Ponds PFT, T, and S are outlined and labeled with dashed blue lines showing intermittent spillway connection to stream. Solar-powered permanent antennas are labeled as “S”. Sites 1 through 10 are labeled with numbers on Schoolhouse Creek.

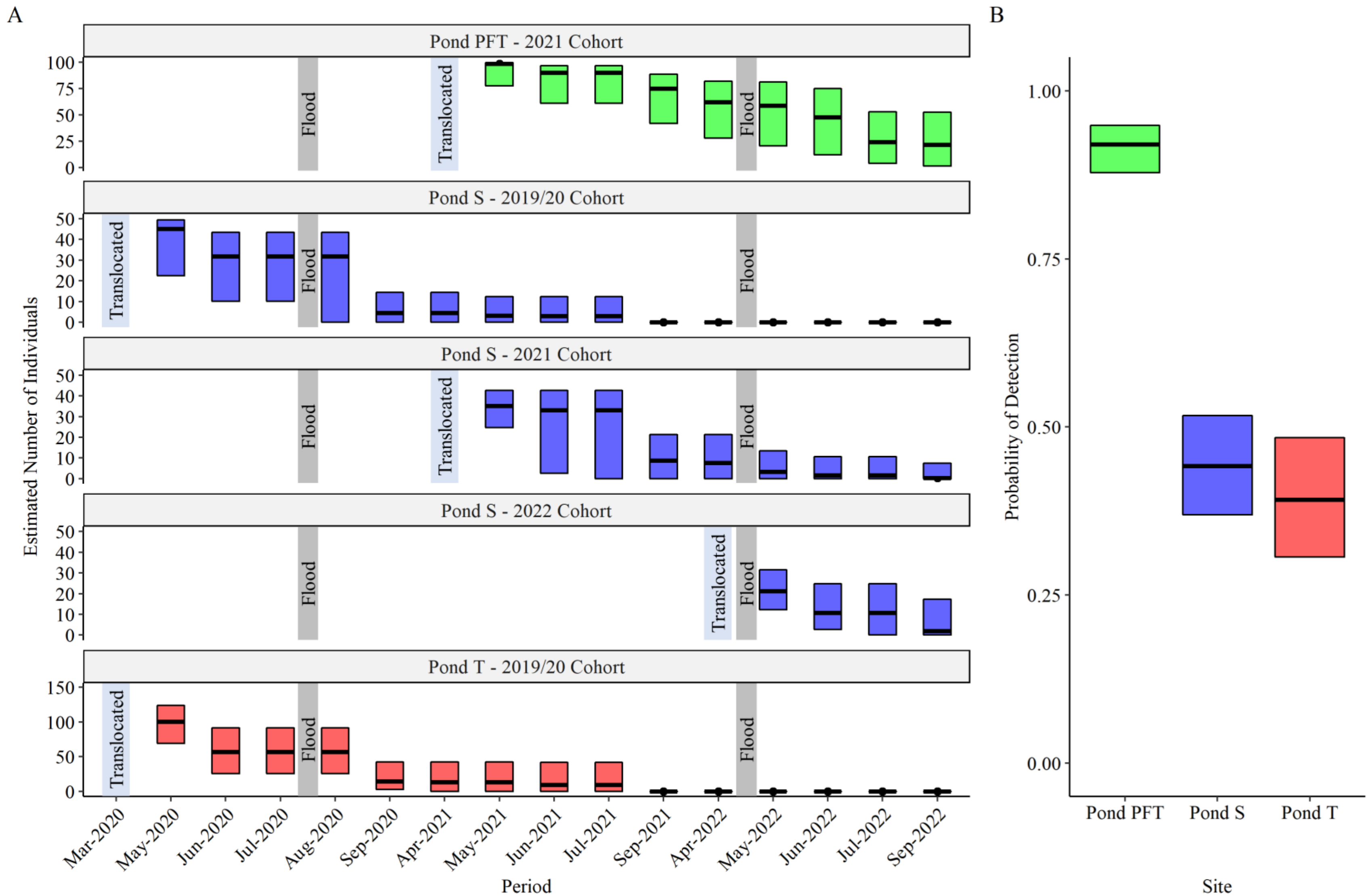

Figure 4.2. (A) Estimated number of translocated (for Ponds PFT and T) or tagged (Pond S) Topeka Shiner in ponds on Tallgrass Prairie National Preserve from apparent survival estimates (note, the scale on the y-axis is different for each pond). (B) Probability

detection estimates among ponds. Crossbars represent 95% confidence intervals. Relative timing of pond overflow is labeled with grey bars. The relative timing of translocation for each cohort is labeled with light blue bars. Y-axis maximum values represent the number of translocated or tagged number of individuals per cohort.

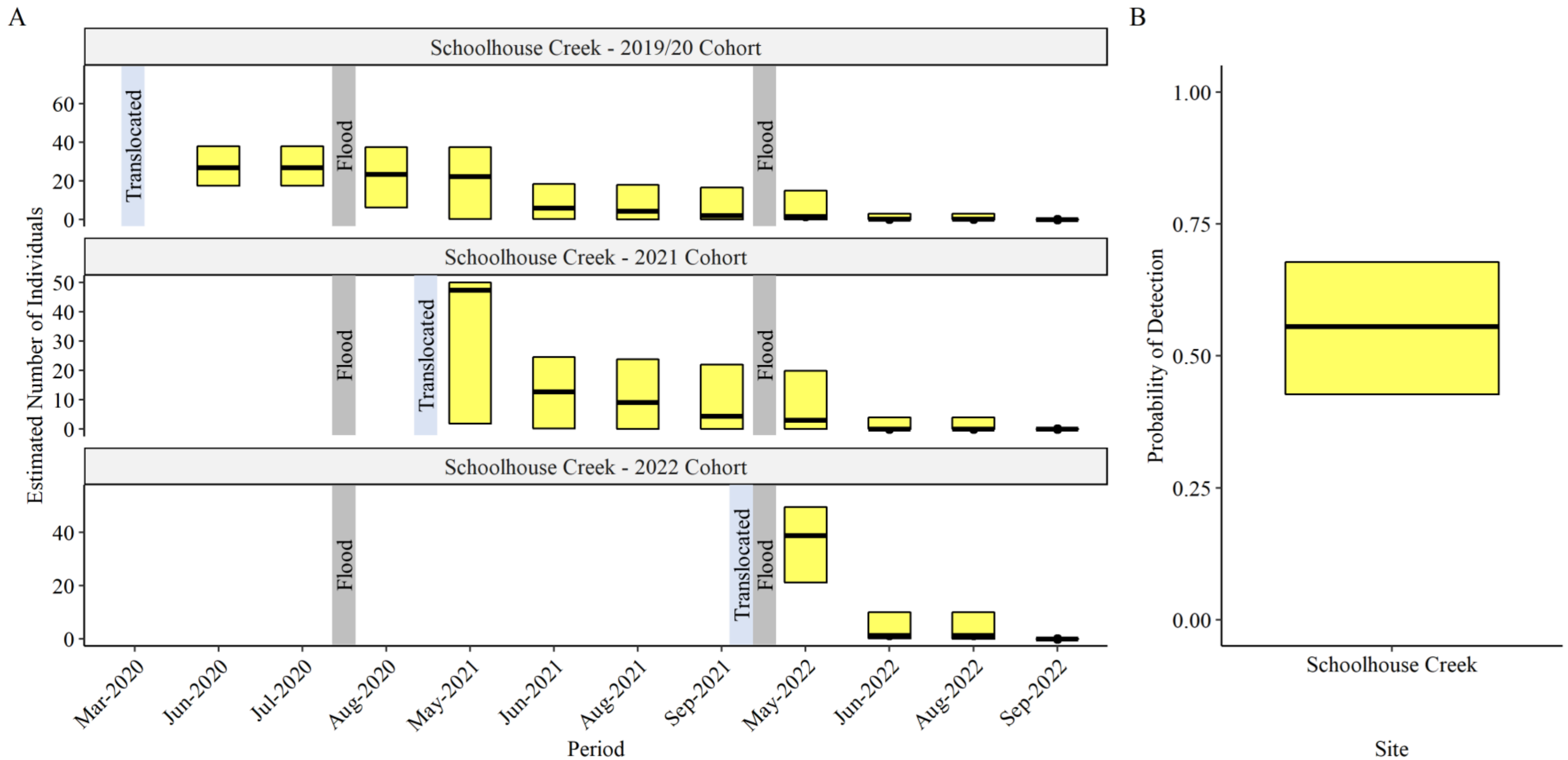

Figure 4.3. (A) Estimated number of translocated Topeka Shiner in Schoolhouse Creek on Tallgrass Prairie National Preserve from apparent survival estimates (note, the scale on the y-axis is different for each pond). (B) Probability detection estimates for Schoolhouse Creek. Crossbars represent 95% confidence intervals. Relative timings of pond overflow events are labeled with grey bars. The relative timing of translocation for each cohort is labeled with light blue bars. Y-axis maximum values represent the number of translocated or tagged number of individuals per cohort.

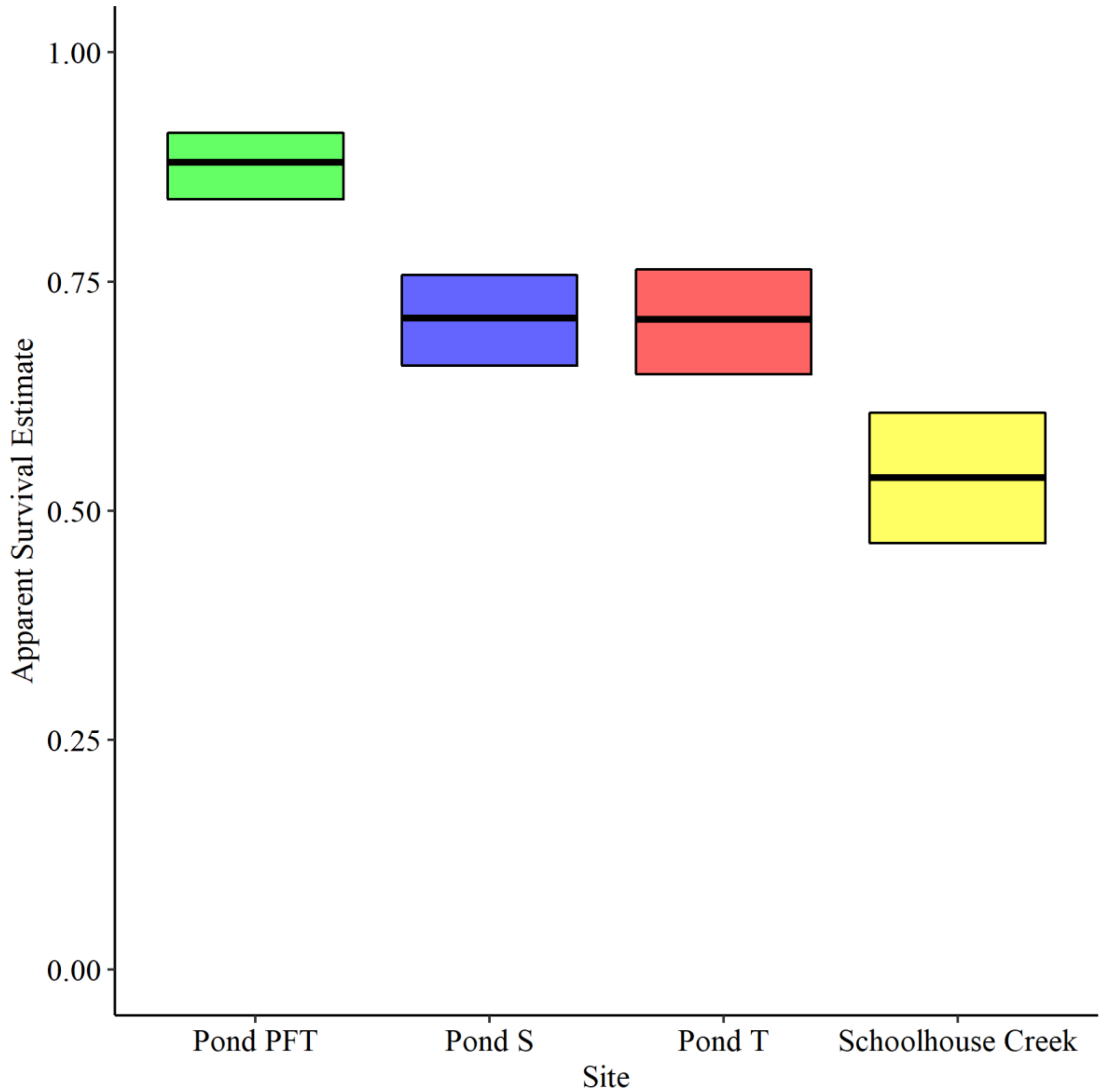

Figure 4.4. Overall apparent survival estimates for PIT-tagged Topeka Shiner translocated into Ponds PFT, T, S, and Schoolhouse Creek on the Tallgrass Prairie National Park. Crossbars represent 95% confidence intervals.

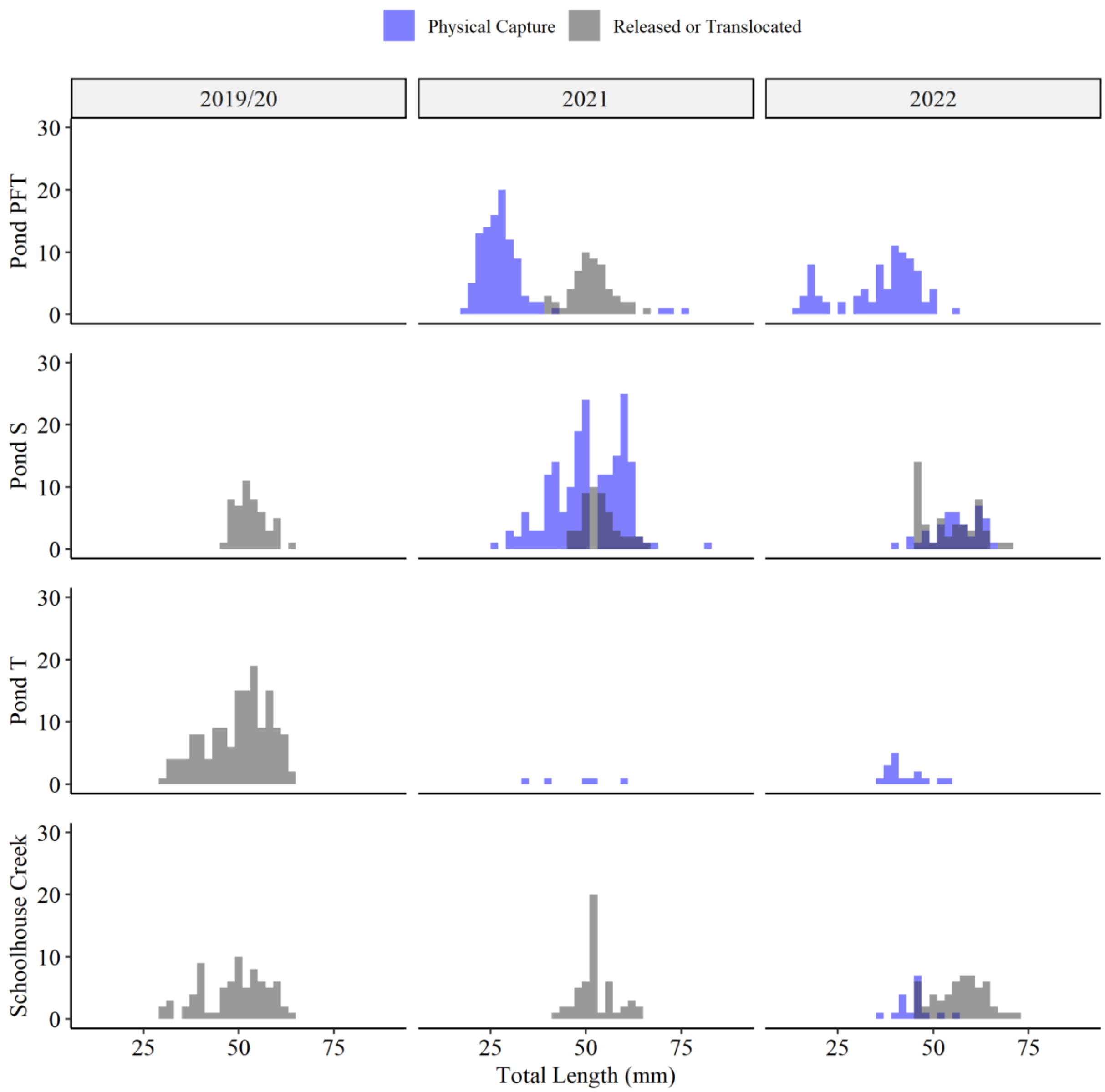

Figure 4.5. Histograms showing the total length (mm) count of captured (blue) and translocated or released (gray) Topeka Shiner by year from ponds PFT, T, S, and Schoolhouse Creek on Tallgrass Prairie National Preserve.

Tables

Table 4.1. Record of PIT-tagged and non-tagged translocated or released Topeka Shiner on Tallgrass Prairie National Preserve and unique individual redetections of PIT-tagged Topeka Shiner by release pond and cohort. Redetection columns are separated by year and redetection time period and total unique individuals detected in each cohort. All translocated fish were taken from Pond S.

				2019-20					2021					2022					
Pond	Year	PIT Tagged	Non-tagged	May	Jun	Jul	Aug	Sep	Apr	May	Jun	Jul	Sep	Apr	May	Jun	Jul	Sep	Total Unique Individuals
Pond T	2019/20	99	50	31	12	12	16	2	2	4	4	3	0	0	0	0	0	0	45
	2022 *	-	-	-	-	-	-	-	-	-	-	-	-	-	1	1	0	0	1
Pond S	2019-20	50	-	21	6	19	17	1	3	3	0	2	0	0	0	0	0	0	36
	2021	50	-	-	-	-	-	-	-	16	16	13	4	0	1	3	3	0	31
	2022	50	-	-	-	-	-	-	-	-	-	-	-	-	8	3	4	1	10
Pond PFT	2021	50	50	-	-	-	-	-	-	45	44	44	31	30	25	23	10	10	49

* Redetected individual that moved from Pond S to Pond T.

Table 4.2. Model comparison tables ranked by QAICc for PIT tagged Topeka Shiner in ponds on Tallgrass Prairie National Preserve from 2020 to 2022. Only the full model, null model, and models with 1% or greater model likelihood are included.

Apparent Survival Parameters	Detection Parameters	QAICc	ΔQAICc	Model Likelihood	Parameters	Deviance
Pond * Time	Pond	1517.87	0	1	45	295.30
Pond * Time	Pond * Cohort	1520.66	2.79	0.25	47	293.57
Time	Pond * Cohort	1522.84	4.97	0.08	19	356.90
Time	Pond	1533.66	15.79	0.01	17	371.92
Pond * Cohort * Time	Pond * Cohort * Time	1694.30	176.43	0	145	210.79
Null	Null	1814.58	296.71	0	2	683.62

Table 4.3. Record of PIT-tagged and non-tagged Topeka Shiner translocated to Schoolhouse Creek on Tallgrass Prairie National Preserve and unique individual redetections by translocation cohort (e.g., T1) and total unique individuals detected in each cohort. PS is the redetection period immediately prior to release of the year's respective cohort. All translocated fish were taken from Pond S.

			2020			2021				2022				
Stocking Year	Number of PIT Tagged	Number of Non-tagged	T1	T2	T3	T1	T2	T3	T4	T1	T2	T3	T4	Total Unique Individuals
2019/20	51	25	7	8	8	4	3	2	0	0	0	0	0	16
2021	25	25	-	-	-	19	5	3	2	0	0	0	0	19
2022	55	0	-	-	-	-	-	-	-	22	0	1	0	22

Table 4.4. Model comparison tables ranked by AICc for PIT tagged Topeka Shiner in Schoolhouse Creek on Tallgrass Prairie National Preserve from 2020 to 2022. Only the full model, null model, and models with 1% or greater model likelihood are included.

Apparent Survival Parameters	Detection Parameters	AICc	ΔAICc	Model Likelihood	Parameters	Deviance
Time	Null	369.90	0	1	12	60.34
Time	Time	374.54	4.64	0.10	21	43.76
Null	Time	375.12	5.22	0.07	13	63.30
Cohort	Time	379.36	9.46	0.01	15	62.94
Null	Null	410.80	40.90	0	2	122.73
Cohort * Time	Cohort * Time	515.09	145.19	0	69	26.90

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Chapter 5 - Conclusions

Farm pond communities and the factors influencing them

Fish communities are shaped by a combination of abiotic and biotic factors. In chapter 2, I presented and tested a conceptual framework of how farm pond communities respond to a pond size and water permanency gradient and are further shaped by interactions among species. Over half of the 100 farm ponds sampled contained stream fish indicating that they are frequently able to colonize these habitats. Amphibian and crayfish had a linear decrease in abundance in relation to a pond size and permanency gradient while stocked fish abundance had a linear increase. Stream fish were most common in ponds of intermediate size (between 150 and 400 m perimeter) and permanency where these species were able to colonize and persist without being suppressed by stocked fish species. This finding was corroborated using canonical correspondence analysis, which revealed distinct species groupings along a pond size, depth, and connectivity. We used distribution modeling to take into account environmental variables and revealed potential species interactions between stocked fish and other species. Further, only a portion of the regional species pool were found to inhabit farm ponds (Bruckerhoff et al. 2021). These results suggest regional and local processes limit community assembly in ponds (Smith and Powell 1971; Tonn et al. 1990). This survey also identified two previously unrecorded populations of federally endangered Topeka Shiner (*Notropis topeka*), indicating that lentic habitats, such as farm ponds, may be important in the conservation of this species as they are in other areas of Topeka Shiner distribution (Thomson et al., 2009, Osterhaus et al. 2022).

In Chapter 3, I used an experimental field study to quantify the influence of piscivorous Largemouth Bass (*Micropterus salmoides*) on the survival of a translocated Bluntnose Minnow (*Pimephales notatus*), a species found to naturally colonize farm ponds in Chapter 2. I translocated 1600 passive integrated transponder (PIT) implanted Bluntnose Minnow into replicate treatment ponds with and without Largemouth Bass. Translocated populations were monitored using PIT antennas and redetections were used in open population mark-recapture models were used to estimate apparent survival and probability of detection for each pond. Ponds without bass had an apparent survival nearly double and a probability of detection 10 times higher than that of ponds containing bass. These results indicate that a direct interaction via predation can have a strong influence on the survival of minnow species as seen in other studies (Schlosser 1988; Harvey 1991; Van Der Walt et al. 2016). Additionally, probability of detection estimates suggests that there is an indirect interaction between bass presence and minnow species via altered behavior. Changes in behavior can influence overall movement, such as seen in this study, as well as foraging and spawning success (Fraser and Gilliam 1992; Peterson and Kitano 2021). The direct and indirect interactions observed in this study present a challenge to minnow species successfully establishing a population, either through natural colonization or translocation, in farm ponds that contain bass.

Chapters 2 fills an important gap in knowledge on existing farm pond communities and the biotic and abiotic factors influencing them. Chapter 3 directly investigates the biotic interaction suggested from distribution modeling and the conceptual framework presented in Chapter 2. Further, two previously unrecorded reproductive populations of Topeka Shiner were found in farm ponds highlighting their potential role in conservation.

The role of farm ponds in conservation

In Chapter 4, I assessed the efficacy of using translocation to pond and stream habitats in Topeka Shiner conservation with a combination of remote mark-recapture techniques and physical capture to estimate survival, movement, and reproduction. Translocated Topeka Shiner were able to survive and reproduce in both pond and stream habitats. I found that they had the highest apparent survival in a rehabilitated pond and the lowest in a stream while the source population and a third pond containing a similar species composition to the source population had an apparent survival in the middle. However, in all cases, translocated individuals were able to persist for over a year. In this study, I also observed Topeka Shiner immigration and emigration to and from pond environments during periods of connectivity, demonstrating their natural ability to colonize ponds and disperse from them as observed in Chapter 2. These results show that translocated Topeka Shiner, even a relatively small numbers, are able to persist and reproduce in new habitats, especially rehabilitated ponds. The main limitation for the use of farms ponds by Topeka Shiner, either naturally or via translocation, is likely connectivity and the presence of predators as explored in Chapter 3. Translocation is a low-cost strategy that could bolter existing populations or repatriate other populations of Topeka Shiner in areas of their range within Kansas.

Together, these studies demonstrate how fish communities within farm ponds respond to biotic and abiotic factors. Further, they demonstrate that native species are able to utilize farm ponds given the right conditions and that they are, despite being anthropogenic alterations, important habitats for some species. Quantifying immigration and emigration to and from ponds proved difficult, but I was able to record both occurring and tie that to a specific pond overflow event. Additionally, I accurately estimate survival and detection of translocated minnows using PIT tag technology can readily be used to assess future conservation efforts. Finally, I demonstrated that stocked fish interact directly and indirectly with stream fish species in farm ponds which may limit successful colonization or translocation efforts to these habitats. If we are to conserve headwater stream species, especially those that are threatened or endangered, strategies that integrate farm ponds seem necessary given their prevalence on the landscape.

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Appendix A - Chapter 2 supplemental tables and figures

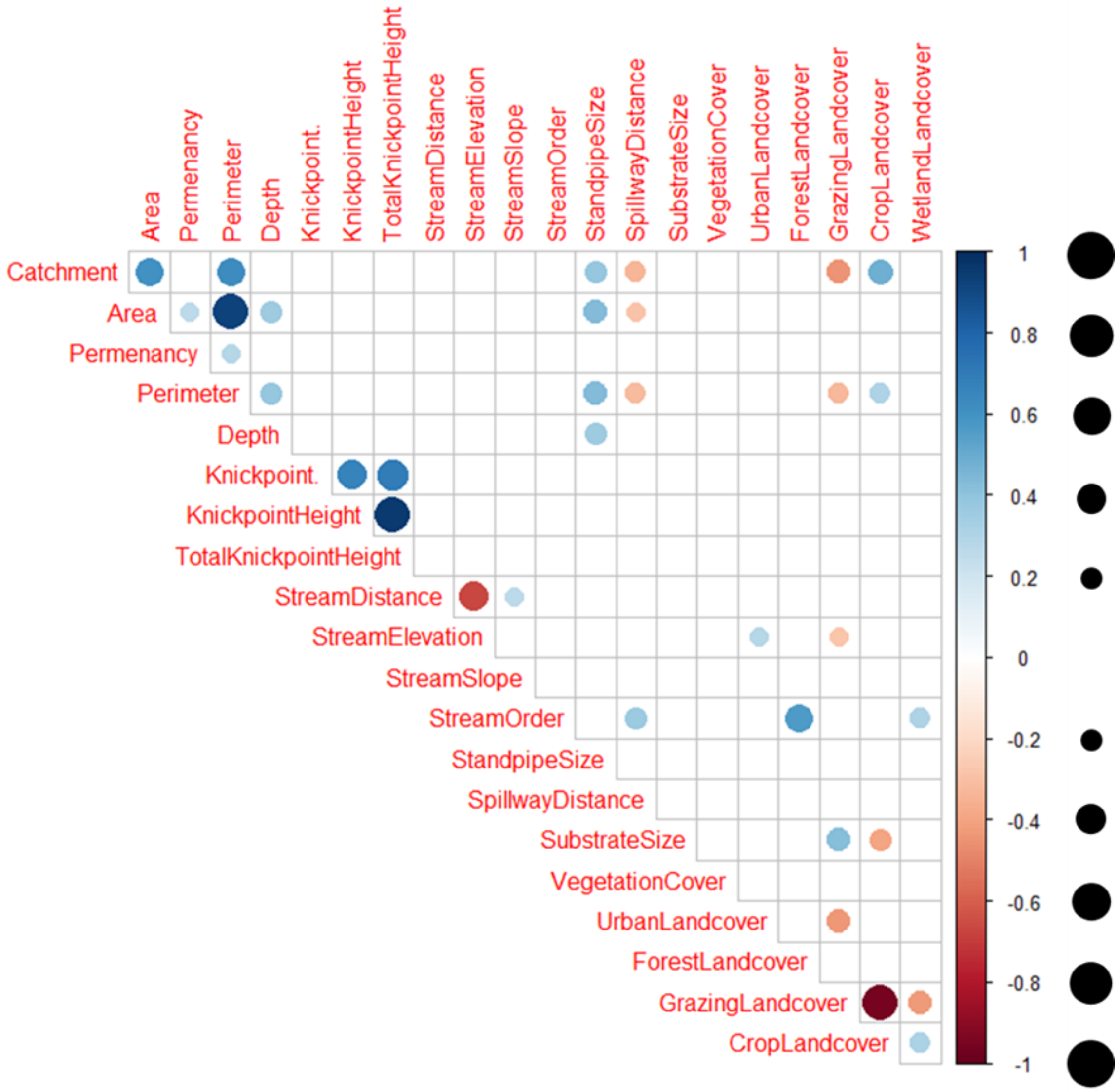

Figure A.1. Graph of abiotic and landscape variables pairs that had significant Pearson correlations. Circles represent significant associations with color ramp indicating the direction of the association with blue being positive and red being a negative, and circle size indicating the strength of correlation.

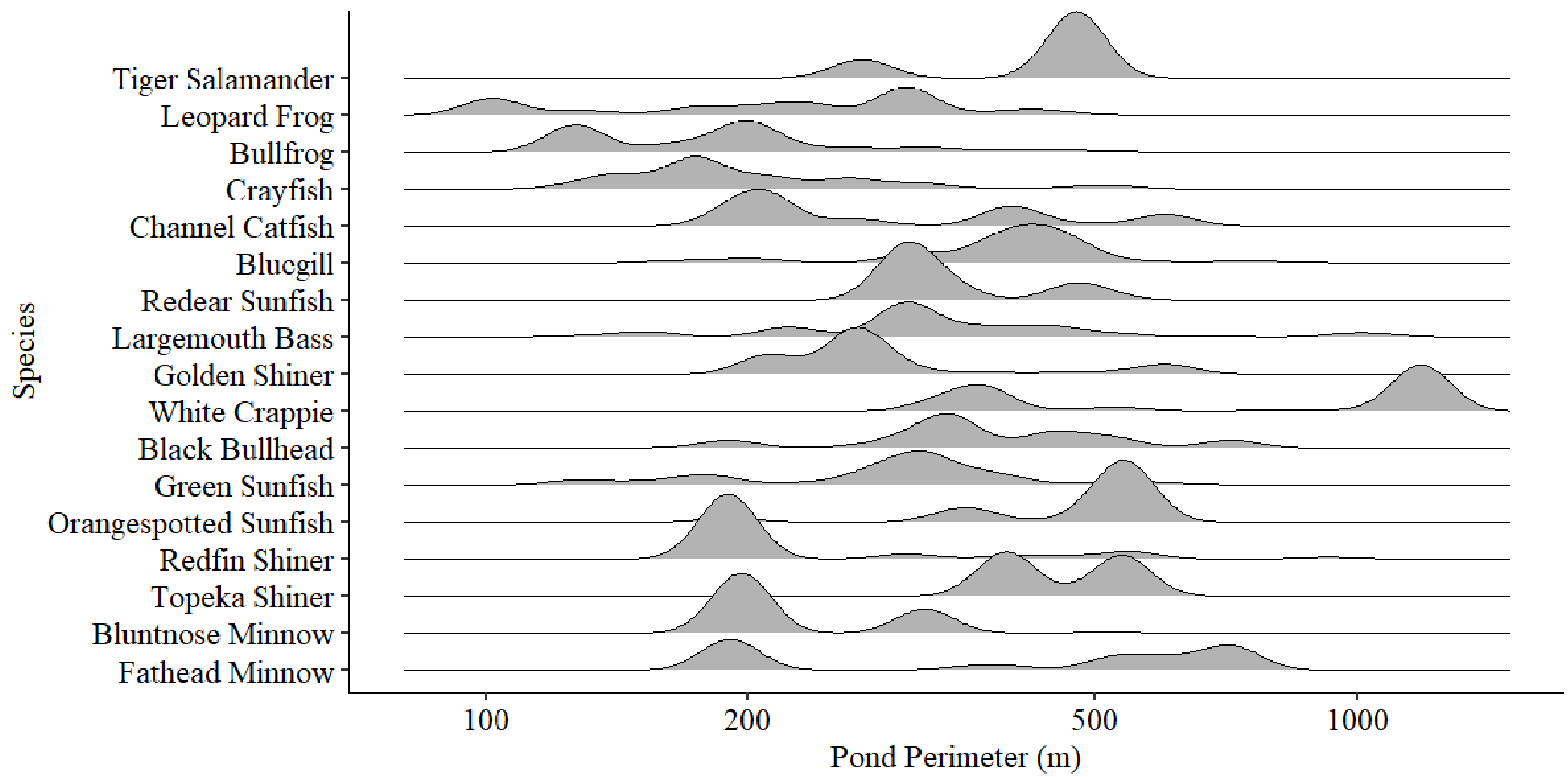

Figure A.2. Density plots of all species (three or more occurrences) captured from 100 farm ponds sampled across a gradient of pond perimeters (represented on a log scale).

Table A.1. Summary of abiotic and landscape parameters from 100 farm ponds sampled in the Flint Hills, Kansas, USA. Each parameter is summarized by mean, median, minimum, maximum, and standard deviation.

Parameter	Mean	Median	Minimum	Maximum	Standard Deviation
Catchment (m^2)	362046	272685	25107	1873600	347272.84
Average Area (m^2)	6053.36	3891.37	228.00	46623.72	6843.00
Permanency	0.62	0.64	0.00	0.99	0.23
Average Perimeter (m)	337.78	279.26	101.77	1184.00	207.72
Maximum Depth (m)	2.08	1.99	0.39	4.49	1.04
Spillway Knickpoint Count	0.27	0.00	0.00	4.00	0.68
Largest Knickpoint Height (m)	0.23	0.00	0.00	5.00	0.67
Total Knickpoint Height (m)	0.33	0.00	0.00	9.00	1.06
Distance to Stream (m)	394.49	332.00	68.00	2118.00	279.52
Elevation Change to Stream (m)	-15.80	-13.40	-0.48	-49.10	10.29
Slope to Stream	-0.07	-0.05	-0.73	0.00	0.11
Order of Stream	1.11	1	1	4	0.42
Standpipe Diameter (in)	7.68	8.00	0.00	24.00	6.16
Spillway Distance (m)	132.07	52.39	1.00	712.44	153.30
Substrate Size (Wentworth Scale)	1.43	1.22	0.52	3.30	0.58
Vegetation Cover Percentage	32%	26%	0%	100%	0.29
Urban Landcover Percentage	5%	2%	0%	30%	0.06
Forest Landcover Percentage	2%	0%	0%	25%	0.04
Grazing Landcover Percentage	77%	90%	3%	100%	0.28
Crop Landcover Percentage	15%	0%	0%	85%	0.26
Wetland Landcover Percentage	0%	0%	0%	5%	0.01

Appendix B - Chapter 3 supplemental figure

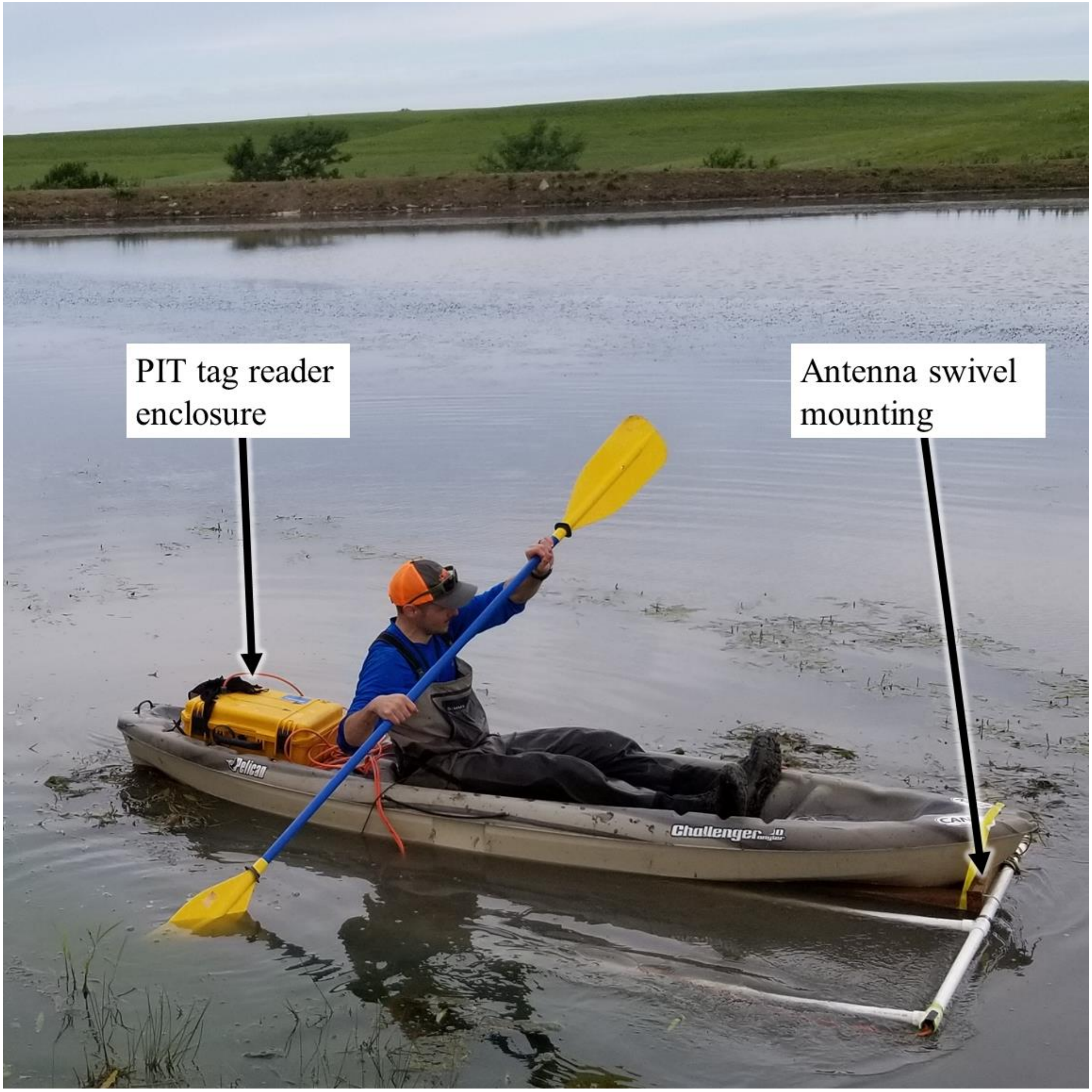

Figure B.1. Picture of mobile kayak-mounted PIT antenna. Not the forward antenna attachment which allows the rear-portion of the antenna to dip down to a depth of two meters.

Appendix C - Chapter 4 supplemental figure

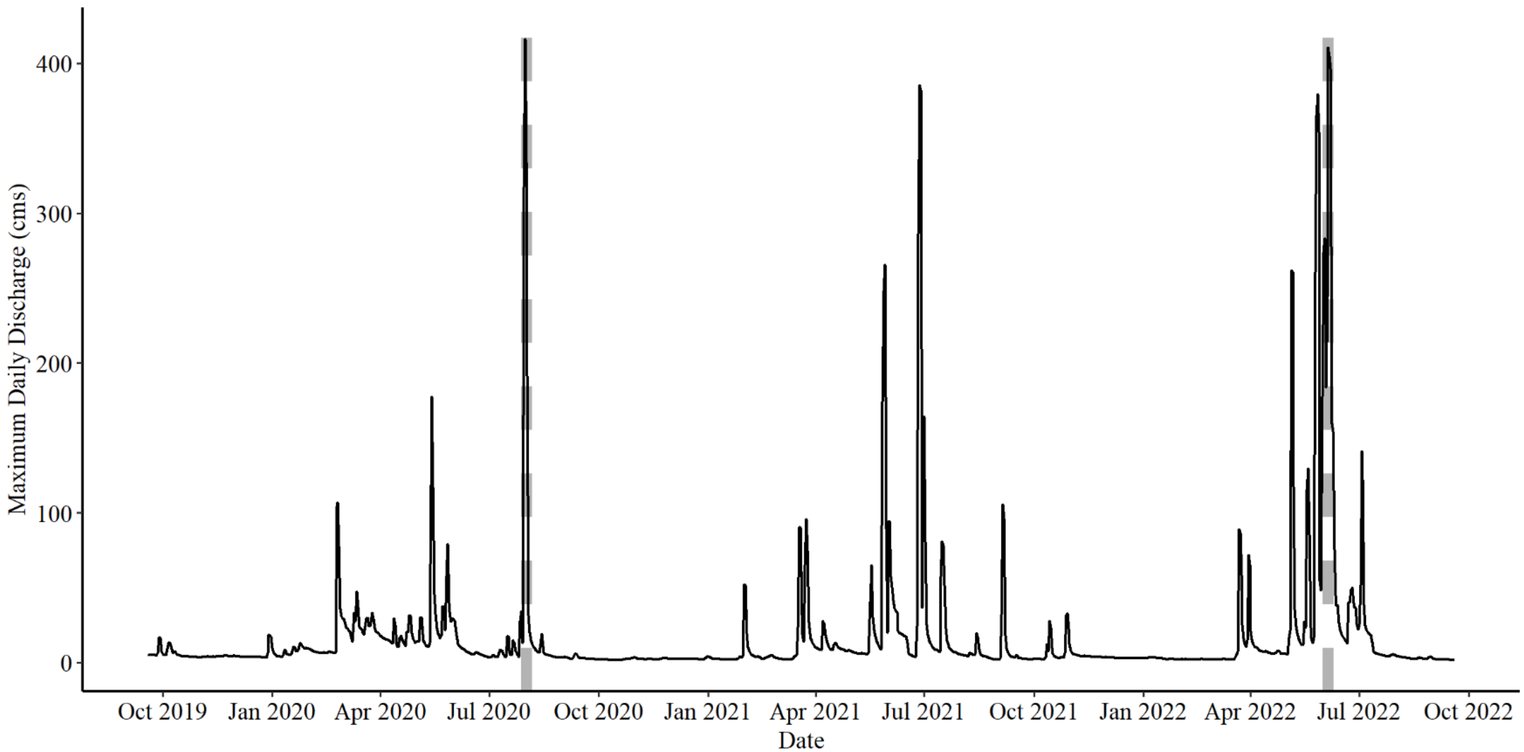

Figure C.1. Cottonwood River daily maximum discharge (cms) from the USGS gage located at Cottonwood Falls, Kansas (07182000) from September 19th, 2019, to September 19th. 2022. Pond overflow events are represented with dashed gray lines.