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USE OF ASSEMBLAGE STRUCTURE, ABUNDANCE, AND YEAR-CLASS STRENGTH TO ASSESS FISH ASSEMBLAGE RESPONSES TO WATER LEVEL FLUCTUATIONS

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by

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in the field of Zoology

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Water level fluctuations can influence lateral connectivity of a water body; during high water levels, fish gain access to off-channel and floodplain habitats that can provide refuge for smaller fish and provide access to abundant aquatic and terrestrial prey. Water level dynamics play a critical role in ecosystem productivity and can have varying effects on biota. Modifications to water flow in Buttonland Swamp, located near the headwaters of the Lower Cache River in southern Illinois, were put in place to improve drainage and restore historical water levels; however, these alterations have disrupted the natural hydrologic dynamics and flood pulse processes, which could influence fish assemblage dynamics. The fish assemblage in Buttonland Swamp was sampled using electrofishing, fyke nets, and mini fyke nets during 2020 and 2021 to evaluate effects of water level dynamics on fish abundance, assemblage structure, and year-class strength. Random stratified sampling was used to select sampling sites each month. Four macrohabitats (Cache River channel, Buttonland Swamp main channel, side channels, and Eagle Pond) within the swamp were surveyed monthly. Open water, nearshore vegetated, and offshore vegetated habitats within each of the macrohabitats were also surveyed.

The first chapter of this study evaluated whether assemblage structure and abundance of fishes in Buttonland Swamp differed across seasons, years (using historic data), and habitats over varying water levels and assessed associations between particular species (especially species of conservation concern) and habitats across seasons, years, and water levels. Each gear was analyzed separately, although electrofishing most effectively sampled all habitats, so analyses of fish assemblage structure across seasons, habitats, and years only used electrofishing data. Non-metric multidimensional scaling (NMDS) and analysis of similarities (ANOSIM) were used to evaluate spatiotemporal patterns in assemblage structure in relation to water level. Indicator Species Analysis (ISA) was used to identify prominent species among habitats, seasons, years (using historical data from Illinois Department of Natural Resources (IDNR) sampling in Buttonland Swamp), and datasets (from IDNR and SIU sampling). Repeated measures using mixed models were used to distinguish significant differences in catch per unit effort (CPUE) (catch per minute using electrofishing; catch per net night for fyke nets and mini fyke nets) among macrohabitats, microhabitats, and seasons. There was a significant difference in assemblage structure among macrohabitats, microhabitats, years, and seasons; however, the R-value was so low among macrohabitats and years that the significant P-values were disregarded since this could have been by chance since the sample size was large. Microhabitats had significantly different assemblage structures, with Silver Carp, Shortnose Gar, Threadfin Shad, and Gizzard Shad being associated with open water, whereas Taillight Shiner, Orangespotted Sunfish, Golden Shiner, Largemouth Bass, Redear Sunfish, Bluegill, Warmouth, Black Crappie, and Smallmouth Buffalo were associated with vegetated habitats. Using IDNR data, there were no significant differences in assemblage structure among years, although there was a significant difference in assemblage structure between the IDNR and SIU data; species that were associated with vegetated habitat were associated with IDNR data, whereas SIU data were more associated with species that were associated with open water. Additionally, some microhabitats and macrohabitats had significant differences in assemblage structure among seasons. ISA had similar results to NMDS and ANOSIM, but also showed prominent species for macrohabitats

and years. Additionally, depending on the gear used, CPUE was significantly different among macrohabitats, microhabitats, and seasons. Species that were associated with vegetated habitats may be more impacted by water level fluctuations because at very low water levels these species may have less shallow water vegetated habitat to utilize or may be left stranded as the water level recedes.

The second chapter assessed associations between historic water level data, air temperature data, and historical catch per unit effort (CPUE) data of predator and competitor fish species from Buttonland Swamp with year-class strength indices of Silver Carp, Bluegill, and Gizzard Shad to evaluate relationships between hydrology, temperature, and other species interactions with fish recruitment. Random stratified sampling was used to select sixteen electrofishing transects each month during September through November 2020 and 2021. Among fish collected, a subsample of Bluegill, Gizzard Shad, and Silver Carp were aged by otolith and cleithra annuli counts. An age-length key was developed to assign ages to unaged fish. Year-class strength indices were determined using the residual method from catch curve regressions. Bluegill and Gizzard Shad had weak year-class strength in 2017, which coincided with a low average water level that year. Water level during fall and spawning was positively associated with year-class strength for Bluegill and Gizzard Shad. Gizzard Shad and Bluegill year-class strength was positively associated with minimum winter temperature. Bluegill and Gizzard Shad year-class strength had a negative association with catch per unit effort (CPUE) of Common Carp and Largemouth Bass. Other predator and competitor fish species CPUE were also significantly associated with year-class strength of Bluegill, Gizzard Shad, and Silver Carp. Only a few age-classes of Silver Carp were caught, representing a boom and bust pattern of recruitment. Since Silver Carp could have migrated into the study area, Buttonland Swamp water level may not be an important factor influencing year-class strength of this species. Water level, along with other abiotic and biotic factors, appear to be influencing year-class strength of fishes within Buttonland Swamp. High water level throughout the year, especially from May-June and October-December, was associated with strong year-class strength of Bluegill and Gizzard Shad. Additionally, maintaining high water level in the winter could allow for deep water refuges for cold intolerant fishes to avoid winter mortality and maintain their abundance. This information contributes to both understand and management of hydrologically impaired systems and can also facilitate future assessments of how current and future hydrologic management regimes will affect the fish community.

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CHAPTER 1

ASSESSING FISH ASSEMBLAGE STRUCTURE RELATIVE TO HABITAT CHARACTERISTICS IN BUTTONLAND SWAMP, ILLINOIS

INTRODUCTION

Water level dynamics often control the temporal and spatial distribution of freshwater communities and the overall productivity of the ecosystem (Puckridge et al. 2010). Annual hydrologic dynamics (depth and flow variation) have species-specific effects depending on fish life history strategies (periodic, opportunistic, and equilibrium strategists) and microhabitat associations (Bice et al. 2014). Frequent water level fluctuations can produce unstable aquatic habitat that can threaten the livelihood of fish associated with particular microhabitats (Bain et al. 1988). Consistently low water levels can lead to habitat homogeneity and a resulting decline in species richness from a loss of habitat variability (Midwood and Chow-Fraser 2012) and can increase the risk of predation (Rolls et al. 2012), whereas greater water depths can promote habitat stability, which can be associated with higher dissolved oxygen and high species richness (Winemiller et al. 2000). Water level fluctuations can influence lateral connectivity of a water body; during high water levels, fish gain access to off-channel and floodplain habitats that can provide refuge for smaller fish and provide access to abundant aquatic and terrestrial prey (Junk et al. 1989, Lyon et al. 2010, Crook et al. 2020). Fish often use floodplain habitats seasonally when there is a flood pulse connecting main channel habitats to the floodplain (Junk et al. 1989). These long flood pulses are important since native fish depend on them, while non-native fish can tolerate and even thrive in highly variable water level and constantly low water levels (Koel and Sparks 2002). Water level dynamics play a critical role in ecosystem productivity and can have varying effects on biota.

Anthropogenic alterations to riverine systems and climate variability have impacted the hydrology of many watersheds, producing a wide range of water levels over time, which can affect the structure and function of these riverine systems, thus contributing to loss of biodiversity (Demissie et al. 1990, Bunn and Arthington 2002, IPCC 2007). Dams are known to degrade habitat and water quality and decrease connectivity by fragmenting river systems and contribute to the loss of flood pulse dynamics, which can reduce fish abundances, species richness, and diversity (Koel and Sparks 2002, Santucci et al. 2005, Agostinho et al. 2008, Slawski et al. 2008, Helms et al. 2011). Additionally, hydrologic dynamics of rivers in the Midwest have been altered due to channelization to improve drainage for agricultural purposes (Karr et al. 1985). Buttonland Swamp located near the headwaters of the Lower Cache River in southern Illinois has been impacted by these anthropogenic changes, which has led to major shifts in its hydrology (Demissie et al. 1990).

Channelization of streams, construction of levees, and agriculture conversion are the most influential human alterations that have been implemented in the Cache River basin (Demissie et al. 1990). The Post Creek Cutoff is a ditch that divides the watershed into the Upper and Lower Cache River watersheds (Demissie et al. 1990). The Post Creek Cutoff caused Buttonland Swamp to dry for extended periods of the year. To restore historical water levels within Buttonland Swamp Diehl Dam (on the west side of the swamp), an in-stream weir (located east of the swamp by Route 37), and the Karnak levee (farther to the east) were constructed to inundate the swamp to historical levels (Middleton 2000, Fidler 2014). The hydrology of Buttonland Swamp has been managed by the Illinois Department of Natural Resources (IDNR) for nearly 40 years using Diehl Dam (Brian Metzke, personal communication). Because water regulation structures are present on the swamp's east and west sides, Buttonland Swamp replicates characteristics similar to a shallow lake rather than a floodplain (Bennet et al. 2001). These human alterations have shifted the hydrologic dynamics in Buttonland Swamp, where flows can move east towards the Cache River levee or west into the Mississippi River depending on flows from tributaries and water-surface elevations in the river channel in Buttonland Swamp and further downstream (Demissie et al. 1990). Modifications to water flow in the Lower Cache River were put in place to improve drainage and restore historical water levels; however, these alterations have disrupted the natural hydrologic dynamics and flood pulse processes (Gough 2005), which could influence fish assemblage dynamics (Ropke et al. 2015).

Riverine Bald Cypress swamps in the Midwest historically had flood pulses in which water levels increased in the winter and decreased in the summer (Wharton et al. 1982). Modifications in Buttonland Swamp have altered the direction of water flow, timing, frequency, velocity, and volume (Gough 2005, Demissie et al. 2008, Demissie et al. 2010). Flood pulses can play a critical role in the behavior of biota (Junk et al. 1989) and the timing and duration of these pulses can influence fish movement and behavior (King et al. 2003, Ropke et al. 2015). Fish benefit from inundation, whereas wetland plant communities in swamps depend on high water levels in the winter and low water levels in the spring and summer for growth and regeneration (Dicke and Toliver 1990). There has been evidence within Buttonland Swamp of declining plant diversity and species richness (Middleton and McKee 2004). Bald Cypress (*Taxodium distichum*) and Water Tupelo (*Nyssa aquatica*) trees within the swamp have been negatively affected by high sedimentation rates, low dissolved oxygen from stagnant water, and consistent inundation, all of which impact recruitment (Middleton 2000). There is concern that this prolonged inundation has altered the wetland's productivity, since plant species that are dependent on these natural flood pulses can shift further upland or become extirpated from the area (U.S. Fish and Wildlife Service 2016). Although consistent inundation has negatively impacted Bald Cypress recruitment, this may not be problematic for fishes, whereas low water conditions that benefit Bald Cypress recruitment could have negative impacts on the fish community; low water level conditions could cause issues including low dissolved oxygen, greater temperature fluctuations, limited deep water refuge, or loss of habitat area for feeding and spawning, which could influence year-class strength and recruitment of fishes. Therefore, to balance benefits for biota in Buttonland Swamp, water level restoration efforts and management may require compromises.

The IDNR has led basin-wide fish surveys in the Cache River since 1992 (Brian Metzke, personal communication); 86 fish species have been found in the Lower Cache River region with 59 species having been recorded within Buttonland Swamp (Brian Metzke, personal communication). These surveys are limited since typically the same location was sampled once per year, few if any habitat characteristics were recorded, and spring and winter seasons were not included. Within the last thirty years, records and collections of species occurrences within Buttonland Swamp have been made by the Illinois Natural History Survey's fish, amphibian, and reptile museum collections and the IDNR Natural Heritage Database. These data, although limited, provide a history of what fish species have been found in Buttonland Swamp. A better understanding of how water level dynamics over time and space affect the fish assemblage can provide the IDNR with baseline data that can be used to modify and improve hydrologic management to benefit biota. The objectives of this study were to evaluate whether assemblage structure and abundance of fishes in Buttonland Swamp differed across seasons, years (using historic data), and habitats over varying water levels and to assess associations between particular species (especially species of conservation concern) and habitats across seasons, years, and water levels.

METHODS

Study Area

The study area encompassed 176.44 hectares (ha) of a Cypress-Tupelo wetland known as Buttonland Swamp which is part of the Lower Cache River Land and Water Reserve within Johnson and Pulaski counties, Illinois. In the 1940s, most of the original swamp was converted to agriculture, except for Buttonland Swamp. The swamp contains portions of open water and extensive patches of Buttonbush interspersed with an open canopy of Bald Cypress and Water Tupelo trees. Tributaries include Big, Cypress, and Mill creeks along with Ketchell and Limekiln sloughs, which can influence water flow and sedimentation within the swamp (Allgire and Cahill 2001, Heglund et al. 2016). The water level in Buttonland Swamp depends on rainfall and inflow from tributaries and agricultural ditches (Demissie et al. 1990, Allgire and Cahill 2001). The IDNR uses Diehl Dam to control the water level within Buttonland Swamp; the dam is opened during flood stages and closed if water levels are at 100.1 meters above sea level (MASL (historical water level elevation)) or lower to ensure the swamp stays inundated (Christina Feng, IDNR, personal communication). Upstream tributaries deposit high amounts of sediment into Buttonland Swamp. Because of the high rate of sedimentation, regular dredging of the Cache River channel adjacent to Buttonland Swamp is planned to maintain water depth (Christina Feng, IDNR, personal communication).

The Cache River watershed provides critical habitat making up 91% of Illinois' remaining forested swamp (Illinois Department of Natural Resources 1997). This critical habitat is used by 42% of all native fish species in Illinois (Burr 1992, Bennett et al. 2001). Buttonland Swamp is considered the northernmost Cypress-Tupelo wetland in the United States and provides habitat for eleven state-threatened, endangered, or imperiled fish species (Bennett et al.

2001, Metzke et al. 2012, Vandermyde and Shults 2015, Hannah Holmquist personal observation). The lower portion of the Cache River basin includes the Cache River State Natural Area (CRSNA) and Cypress Creek National Wildlife Refuge (CCNWR). Buttonland Swamp is valued for the unique habitat it provides and is considered a National Natural Landmark, an Illinois Land and Water Reserve, and a Wetland of International Importance (Ramsar Convention 2009).

Fish Sampling

The swamp was separated into four macrohabitats (Cache River channel, Buttonland Swamp main channel, side channels, and Eagle Pond) to compare fish abundance and assemblage structure among the different habitats (Figure 1). To determine what habitats fish were associated with in each macrohabitat, macrohabitats were divided into microhabitats (open water, nearshore, and offshore vegetated habitat). Open water was defined as an area without emergent vegetation that had a consistent channel width of at least 21.3 m. Nearshore vegetated habitat was defined as habitat where emergent vegetation was within 15.2 m of land (land being the bank or ground that was above water during a survey) and adjacent to land; this included areas that were islands, swamp margin, and other areas that were not necessarily considered swamp bank edges. Offshore vegetated habitat was considered habitat where emergent vegetation was more than 15.2 m from land. The presence of land varied in each macrohabitat depending on the water elevation within Buttonland Swamp. Only microhabitats that were present and accessible during each survey were sampled. Open water and nearshore vegetated habitat in the side channels were limited and nearshore habitat was limited and inaccessible in shallow water in the main channel, so these microhabitats were not sampled during every survey period. Nearshore vegetated habitat in the main swamp was only sampled once and in the side channels open water was only sampled five times and nearshore vegetated habitat was sampled four times.

Fish were sampled from June 2020 through November 2021. Fish were targeted using boat electrofishing in deep/open areas and around the periphery of the swamp (four 15-minute runs per macrohabitat) and fyke (three per macrohabitat) and mini fyke nets (three per macrohabitat) in shallow areas that were vegetated and around the swamp margin. All fish that were seen were caught during the electrofishing transect. For electrofishing a Smith Root control box using direct current with 60 pulses per second and the high range outlet was used. The number of volts were adjusted (ranged from 15 – 60 volts) so the electrical current output was consistently around 7 amps. The fyke nets had frame dimensions of 1.8 x 0.9 m with 1.3 cm square mesh, four steel hoops, and a shortened lead line of 4.57 m. The mini fyke nets had frame dimensions of 0.6 x 0.6 m with 0.6 cm square mesh, three 61-cm steel hoops, and a lead line of 3 m. Fish were sampled monthly from June to October 2020 and March to October 2021; fish were not sampled in March through May 2020 due to Covid-19 restrictions. In 2020, winter sampling was conducted in November by using electrofishing, fyke net, and mini fyke net sampling methods. In 2021, winter sampling was conducted in January and November using the same gears. Fish sampling in the winter provided data to assess habitat use, specifically focused on deep water areas as overwintering habitat for fishes in Buttonland Swamp.

Nets were checked daily, and fish were identified (to species), total length measured (to the nearest mm), and released. Endangered and threatened (E&T) species and Species in Greatest Need of Conservation (SGNC) were also identified. SGNC are defined by the Illinois Wildlife Action Plan as species that have rare and declining populations and habitats, and are highly localized or endemic. Collected E&T species were photographed, if possible, before being released and occurrence records were submitted to the IDNR Division of Natural Heritage Database.

Habitat and Water Quality Sampling

Habitat cover was estimated 6.1 m left, right, and behind the lead line of fyke and mini fyke nets, unless nets were connected to bank habitat, in which case habitat cover was estimated from the net frame to the bank edge. Habitat cover was averaged in the area within 6.1 m on both sides of the boat throughout electrofishing transects. At each site, the percentage of Buttonbush and Bald Cypress trees were estimated to determine if there was an association between fish and Bald Cypress trees as habitat cover. Stratified random sampling was used to randomly select four electrofishing, three fyke net, and three mini fyke net sample sites within each habitat type strata each month. For each electrofishing run, water depth was measured at the beginning and end of each run and averaged, while depth at each fyke or mini fyke net was recorded at the frame of the net using a measuring tape tied to a secchi disk. The width of each macrohabitat was measured at various points in Buttonland Swamp and measurements of how far vegetation extended out from the bank were made to quantify habitat characteristics (open water, nearshore, offshore). Additional habitat attributes recorded at each site included substrate type, aquatic habitat refugia type (percentage of open water, macrophyte/vegetation, stick/log, Bald Cypress, other trees, Buttonbush, and other bushes/shrub coverage by surface area), and vegetation type (percentage of emergent, submergent, overhanging, floating, and algae by surface area). Bank composition (percent of bare, rock, herbaceous/shrubs, woody/logs, trees, and grass by surface area), riparian cover (surface area shaded at noon), and floodplain quality (agricultural field, swamp/forest, developed/disturbed) were recorded when applicable. All habitat attributes were visual estimates, which have also been used in previous studies (Bice et al. 2014, Vandermyde and Shults 2015, Holmquist et al. 2022).

Water elevation was recorded at the Long Reach Road water gauge to determine the overall water level fluctuations in Buttonland Swamp during the time frame of the study. Environmental parameters that were measured included secchi depth (meters (m); to measure transparency), water/air temperature (°C), conductivity (µS), and dissolved oxygen (mg/L), which were measured in the afternoon during each sampling trip. Conductivity and water/air temperature were measured using an Oakton EcoTestr CTS1 Pocket conductivity, salinity, and TDS meter, while dissolved oxygen was measured using a YSI 550A dissolved oxygen meter.

Historical IDNR Data

The IDNR has been conducting fish surveys at Buttonland Swamp, using boat electrofishing, at least once annually during the summer (June - September) since 1992. This annual survey was conducted once at the same location every year; however, from 2011-2015 nine additional locations were sampled in different habitats (open water channel and vegetation [primarily Buttonbush]) along the Cache River section of Buttonland Swamp. Minimal to no habitat data were taken and sampling did not occur year around. Only IDNR fish catch per unit effort (CPUE) data from 1992 to 2020 were used in this study. The IDNR stocked Bluegill (*Lepomis macrochirus*) from 2014-2017 and Redear Sunfish (*Lepomis microlophus*) in 2014, 2016, and 2017 (Jana Hirst, personal communication). The IDNR has also collected water elevation data at least once monthly from 2010 - present. This gave us an idea of how water level is changing seasonally and yearly, although since water elevation was not taken every day there is a chance of missing a brief flooding or low water event.

Data Analysis

Data from each sampling gear, deployed by SIU personnel, were combined to calculate total count, species richness, Shannon-Wiener Index (H'), and proportion of different species to evaluate patterns among years, macrohabitats, and microhabitats. The IDNR data were only used to calculate yearly proportions of species and to evaluate assemblage structure. Proportion of species was calculated using data collected by the IDNR for each year from 1992 to 2020 and SIU data (2020 - 2021). Species were then grouped into families; suckers, carps, sunfishes, minnows, shads, gars, and "other" ("other" included Black Bullhead (*Ameiurus melas*), Bluntnose Darter (*Etheostoma chlorosoma*), Bowfin (*Amia calva*), Brook Silverside (*Labidesthes sicculus*), Brown Bullhead (*Ameiurus nebulosus*), Channel Catfish (*Ictalurus punctatus*), Flathead Catfish (*Pylodictis olivaris*), Freshwater Drum (*Aplodinotus grunniens*), Grass Pickerel (*Esox americanus*), Johnny Darter (*Etheostoma nigrum*), Mosquitofish (*Gambusia affinis*), Mud Darter (*Etheostoma asprigene*), Paddlefish (*Polyodon spathula*), Pirate Perch (*Aphredoderus sayanus*), Sauger (*Sander canadensis*), Slough Darter (*Etheostoma gracile*), Tadpole Madtom (*Noturus gyrinus*), Walleye (*Sander vitreus*), White Bass (*Morone chrysops*), and Yellow Bullhead (*Ameiurus natalis*)). When comparing SIU data with the IDNR data, only SIU electrofishing data collected from the same time of year as the IDNR data (June through September) were used. IDNR historical water elevation data (2010 – 2021) was averaged among seasons per year.

Each gear was analyzed separately, although electrofishing most effectively sampled all habitats, so analyses of fish assemblage structure across seasons, habitats, and years only used electrofishing data. For all assemblage structure analyses, CPUE was calculated, and $\log_{10}(1 + \mathrm{CPUE})$ transformed to reduce the influence of dominant species and decrease extreme catches observed in the CPUE calculations. Species that represented less than 1% of the dataset were excluded in assemblage structure analyses to minimize the impact of rare species on results. When analyzing seasons, winter was considered November and January (based on water temperature), spring was considered March, April, and May, summer was considered June, July, and August, and fall was considered September and October.

Non-metric multidimensional scaling (NMDS) and one-way analysis of similarities (ANOSIM) were used to evaluate changes in assemblage structure of fishes in SIU data across seasons, years, hydrologic variation, macrohabitats, and microhabitats. The IDNR data were also used in NMDS and ANOSIM analyses to evaluate changes in assemblage structure between the IDNR data and SIU data, between gear types used by the IDNR (AC vs DC boat electrofishing), and among years. The NMDS analysis used the Bray-Curtis dissimilarity index to evaluate and graphically depict associations between environmental conditions and fish assemblage across seasons, years, hydrologic variation, macrohabitats, and microhabitats using 999 random starts and maximum iterations (Holland 2008). For environmental condition analyses, values were standardized to range from 0 to 1. Stress was assessed for 1 – 7 dimensions using 999 maximum runs. Shepard plots were used to assess goodness of fit and a stress value around 0.1 to infer whether NMDS ordination distances reflected the observed dissimilarity. Ellipses explained 95% of the assemblage structure variation. R version 4.1.2 was used to run analyses using the function 'ordisurf' from the package 'vegan' to overlay site depth contour lines on the NMDS abundance plot (Oksanen et al. 2009, R Development Core Team, 2020).

ANOSIM was used to determine whether there were significant spatial and temporal trends in fish assemblage structure and environmental conditions by analyzing group similarities using $P < 0.05$ and R-values (Clarke and Warwick 1994). R-values range from -1 to 1 with values close to 0 showing little to no difference in groupings and values close to 1 showing differences in groupings (Chapman and Underwood, 1999). The Bray-Curtis distance metric was used to measure the dissimilarity among observations (9999 permutations, $\alpha = 0.05$).

Since NMDS is just a visualization Indicator Species Analysis (ISA) was also used to confirm what species were driving significant differences in assemblage structure. ISA was used to identify prominent species in each macrohabitat, microhabitat, season, year, and in SIU and IDNR sampling efforts. Indicator values were evaluated by a Monte Calo randomization test (9999 permutations, $\alpha = 0.05$). Indicator values range from 0 to 1, where 0 indicates a species has no correlation with a habitat, whereas 1 indicates perfect correlation (McCune and Grace 2002). R version 4.1.2 was used to run analyses using packages 'vegan' and 'indicspecies' (De Cáceres and Legendre 2009, Oksanen et al. 2009, R Development Core Team 2020).

Endangered, threatened, SGCN, and imperiled species were recorded and the number of times they were caught in a certain macrohabitat or microhabitat were recorded to assess associations between these species and habitat types.

Repeated measures using mixed models (Gurevitch and Chester 1986) were used to distinguish significant differences in mean water depth among macrohabitats and microhabitats over seasons and years (using SIU data) to evaluate differences in water level. In these models macrohabitats and microhabitats were fixed effects, while season or year or the interaction between season and year were random effects. Multiple model combinations were analyzed (Appendix C1 and C2), but only the X^2 and P-values from the model with the lowest AIC were reported in the results. Repeated measures using mixed models were also used to distinguish significant differences in mean water elevations among seasons and years (using IDNR data) to evaluate differences in water elevation. In these models, year was the fixed effect and season was the random effect when looking at how elevation varied among years. However, when looking at how elevation varied among seasons, season was the fixed effect and year was the random effect. Multiple model combinations were analyzed (Appendix D1 and D2), but only the X^2 and P-values from the model with the lowest AIC were reported in the results. The assumptions of normality of residuals and homogeneity of variances were met. To account for temporal autocorrelations an autoregressive-1 (AR1) variance-covariance matrix was used, implemented using the <gls> function from the NLME package (Pinheiro et al. 2007). The package <emmeans> was used to conduct post-hoc pairwise comparisons of treatment means (Searle et al. 1980).

Repeated measures using mixed models (Gurevitch and Chester 1986) were also used (using SIU data) to distinguish significant differences in CPUE per minute (using electrofishing) and per net night (using fyke nets and mini fyke nets) among macrohabitats, microhabitats, and seasons to evaluate differences in abundance. For each macrohabitat and microhabitat using electrofishing data, CPUE was also calculated in the winter to determine whether fish were associated with deep water for winter refuge. Deep water was not targeted using fyke and mini fyke net gears, so data collected using these gears could not be used in determining CPUE for each habitat in the winter. In these models, macrohabitats and microhabitats were fixed effects and year, season, and the interaction between year and season were random effects. Multiple model combinations were analyzed (Appendix E1, E2, E2, F1, F2, F3, G1, G2, and H1), but only the X^2 and P-values from the model with the lowest AIC were reported in the results. When necessary, data were $\log_{10}(1 + \text{CPUE})$ transformed to meet assumptions of normalized distributions and homogenized variances. To account for temporal autocorrelations an autoregressive-1 (AR1) variance-covariance matrix was used, implemented using the <gls>

function from the NLME package (Pinheiro et al. 2007). Models that included the AR1 variance-covariance structure always had the lowest AIC scores, except for the model comparison involving assessment of differences in CPUE among macrohabitats during winter (Appendix H). However, the model that included the autoregressive (AR1) variance-covariance structure had an AIC score within 2 units of the top model (which did not include AR1), indicating equivalent support for the model that included the AR1 variance-covariance structure (Sakamoto et al. 1986, Burnham and Anderson 1998). Thus, models that included the AR1 variance-covariance structure were selected for all comparisons of factors affecting CPUE. The package <emmeans> was used to conduct post-hoc pairwise comparisons of treatment means (Searle et al. 1980).

RESULTS

Species Proportion, Richness, and Diversity

From June 2020 through November 2021, SIU captured a total of 12,328 fish from 56 species and one hybrid, belonging to 34 genera, 18 families, and 12 orders. The most abundant species were Bluegill (42% of total fish caught), Gizzard Shad (8% of total fish caught; *Dorosoma cepedianum*), Redear Sunfish (6% of total fish caught), Black Crappie (6% of total fish caught; *Pomoxis nigromaculatus*), and Shortnose Gar (5% of total fish caught; *Lepisosteus platostomus*). Collectively, these species made up 67% of all fish sampled. Many species were only found a few times; 25 of the 56 species made up <1% of the total catch.

Family proportions have stayed relatively consistent in Buttonland Swamp from 1992 to 2021 (Figure 4). Historically, suckers, shads, and sunfishes have been the dominant families sampled in the swamp. Relatively similar proportions of families were seen from SIU 2020 data, compared to the IDNR 2020 data, although in SIU 2020 data there were greater proportions of shads and carps and smaller proportions of suckers, gars, and fish in the “other” category.

There were differences in species richness and diversity among habitat types in SIU data (Table 1). With all gears combined, the side channels and Cache River had the highest fish species richness with both having 47 species, then the main swamp with 41 species, and Eagle Pond with 40 species (Table 1). Nearshore and offshore vegetated habitats had similar species richness (51 and 48 species), while open water had lower richness (39 species; Table 1). The alpha diversity was similar among the macrohabitats, with the Cache River having the highest (H'=2.47) and Eagle Pond having the lowest (H'=2.23; Table 1). Open water had the highest diversity (H'=2.63), while offshore diversity was H'=2.41 and nearshore diversity was H'=2.10 (Table 1). Macrohabitats had similar proportions of fish species, while microhabitats varied (Figures 2 and 3).

Interannual, Seasonal, and Spatial Patterns of Fish Assemblage Structure

ANOSIM and NMDS plots (Stress<0.15, 3 dimensions) showed significant differences in fish assemblage structure among microhabitats (Figure 5; ANOSIM: R=0.32, P<0.001), macrohabitats (Figure 6; ANOSIM: R=0.06, P<0.001), years (Figure 7; ANOSIM: R=0.07, P=0.003), seasons (Figure 8; ANOSIM: R=0.13, P<0.001), seasons within some macrohabitats and microhabitats (Figure 9, 10, and 11), between the IDNR data and SIU data (Figure 12; ANOSIM: R=0.29, P<0.001), and gear types. However, the R-value was so low among macrohabitats and years that the significant P-value was disregarded since this could have been by chance since the sample size was large (Clarke and Warwick 2001). Microhabitats had significantly different assemblage structures, with the NMDS1 axis being driven by Warmouth (*Lepomis gulosus*), Black Crappie (*Pomoxis nigromaculatus*), Bluegill, Redear Sunfish, Largemouth Bass (*Micropterus salmoides*), Taillight Shiner (*Notropis maculatus*), and Orangespotted Sunfish on the right and Shortnose Gar (*Lepisosteus platostomus*), Silver Carp

(*Hypophthalmichthys molitrix*), Threadfin Shad (*Dorosoma petenense*), and Gizzard Shad on the left, which seemed to be separating open water habitat from vegetated habitat.

Assemblage structure was significantly different among seasons overall and seasons within some of the macrohabitats and microhabitats. When analyzing seasons with habitats pooled together, seasons had significantly different assemblage structure, with the NMDS2 axis being driven by Spotted Gar and Brook Silverside on the top, and Silver Carp, Shortnose Gar, and Gizzard Shad on the bottom, which seem to be separating winter and summer from spring and fall (Figure 8). When analyzing seasons within all habitats separately, only the main swamp, side channels, and offshore vegetated habitat had significantly different assemblage structures among seasons. Within the side channels, seasons had significantly different assemblage structure, with the NMDS1 axis being driven by Taillight Shiner, Brook Silverside, and Spotted Gar (*Lepisosteus oculatus*) on the right, which seemed to be separating winter from the other seasons (Figure 9; ANOSIM: $R=0.26$, $P<0.001$). In the main swamp, seasons had significantly different assemblage structure, with the NMDS1 axis being driven by Brook Silverside and Spotted Gar on the left, which seemed to be separating winter from the other seasons (Figure 10; ANOSIM: $R=0.24$, $P<0.001$). In the offshore vegetated habitat seasons had significantly different assemblage structure, with the NMDS1 axis being driven by Taillight Shiner, Spotted Gar, and Brook Silverside on the right, which seemed to be separating winter from the other seasons (Figure 11; ANOSIM: $R=0.20$, $P<0.001$). There were no significant differences in assemblage structure among years using IDNR data from 1992-2020 (ANOSIM: $R=0.03$, $P=0.38$). However, assemblage structure was significantly different between IDNR and SIU data (ANOSIM: $R=0.29$, $P<0.001$), with the NMDS2 axis being driven by Brook Silverside, Spotted Bass (*Micropterus punctulatus*), Redear Sunfish, Freshwater Drum, and Threadfin Shad on the top, which were associated with SIU data, and Bluntnose Minnow (*Pimephales notatus*), Warmouth, Largemouth Bass, Bullhead Minnow (*Pimephales vigilax*), Mosquitofish, Common Carp (*Cyprinus carpio*), and Spotted Gar on the bottom, which were associated with IDNR data (Figure 12). In 2011, IDNR switched from using AC electrofishing current to DC; however, there were no significant differences in assemblage structure between these two types of boat electrofishing sampling (ANOSIM: R=0.01, P=0.43).

In the ISA analysis, although most correlations were weak, species correlations differed among macrohabitats, microhabitats, seasons, years, and data sources (IDNR vs SIU data) (Table 2). Using SIU data, ISA trends were consistent with NMDS groupings with Shortnose Gar, Silver Carp, Gizzard Shad, and Threadfin Shad being indicator species for open water. ISA showed Warmouth, Black Crappie, and Smallmouth Buffalo being indicator species for nearshore vegetated habitat and Taillight Shiner, Bluegill, Redear Sunfish, Largemouth Bass, Golden Shiner, and Orangespotted Sunfish being indicator species for nearshore and offshore vegetated habitat (Table 2). Threadfin Shad was an indicator species for 2020 using SIU data, since the majority were seen in 2020 and only one was caught in 2021. With data pooled together Warmouth was an indicator species of spring, Spotted Gar, Black Crappie, Bowfin, Brook Silverside, Taillight Shiner, and Spotted Gar were indicator species for winter, along with other species that were associated with multiple seasons (Table 2). Within the side channels, Spotted Gar and Brook Silverside were indicator species of winter, and Taillight Shiner was an indicator species for spring and winter (Appendix B1). Within the main channel, Spotted Gar was an indicator species for winter (Appendix B2). Within offshore vegetated habitat, Spotted Gar, Taillight Shiner, Brook Silverside, and Black Crappie were indicator species for winter (Appendix B3). ISA found other species associated with years, and macrohabitats, along with interactions within each and interactions within seasons (Table 2). Comparing SIU data with the IDNR data, SIU data were associated with fewer species (Redear Sunfish, Brook Silverside, and Silver Carp) while the IDNR data had 21 species associated with it (Table 3). SIU caught seven species (Taillight Shiner, Yellow Bass (*Morone mississippiensis*), Bluntnose Darter (*Etheostoma chlorosoma*), Mimic Shiner (*Notropis volucellus*), Saugar (*Sander canadensis*), Tadpole Madtom (*Noturus gyrinus*), Bantam Sunfish (*Lepomis symmetricus*)) that the IDNR did not catch, while the IDNR caught nine species (White Bass (*Morone chrysops*), Longnose Gar (*Lepisosteus osseus*), Mud darter (*Etheostoma asprigene*), Slough Darter (*Etheostoma gracile*), Bluntnose Minnow (*Pimephales notatus*), Redfin Shiner (*Lythrurus umbratilis*), Silver Chub (*Macrhybopsis storeriana*), Blackside Darter (*Percina maculata*), Flathead Catfish (*Pylodictis olivaris*)) that SIU did not catch from June – September 2020 and 2021; these species were not included in the NMDS analysis since they each represented less than 1% of the catch.

Spatial Variation of Environmental Conditions

NMDS and ANOSIM plots (Stress=0.13, 3 dimensions) showed environmental conditions differed significantly among microhabitats (Figure 13; R=0.53, P<0.001) and macrohabitats (Figure 14; R=0.08, P<0.001). However, the R-value was so low among macrohabitats that the significant P-value was disregarded since the significant P-value could have been by chance since the sample size was large (Clarke and Warwick 2001). Nearshore and offshore vegetated habitats seemed to be associated with vegetation structure, while open water habitats were associated with deeper water and the amount of open water, although there was a slight overlap among microhabitat environmental conditions (Figure 13). The NMDS1 axis was being driven by the percent of open water and water depth on the right and the percent of overhanging vegetation, Buttonbush, emergent vegetation, Bald Cypress, sticks and logs, riparian cover, detritus, and other trees on the left, which seemed to be separating open water habitat from vegetated habitat (Figure 13). The amount of open water, silt and clay, riparian cover, detritus, and Buttonbush could be driving differences in assemblage structure among microhabitats since vector lengths for environmental variables indicates the strength of association (Figure 15). The NMDS1 axis was being driven by water temperature, algae, water depth, open water, and silt and clay on the left and Buttonbush, Bald Cypress, detritus, sticks and logs, and riparian cover on the right, which seemed to be separating open water habitat from vegetated habitat (Figure 15). The NMDS2 axis was being driven by secchi depth and dissolved oxygen on top and water temperature on bottom (Figure 15). Most species were associated with shallow water depths (< 0.9 m). However, Silver Carp and Shortnose Gar were associated with water depths around 1.5 m (Figure 16).

Endangered, Threatened, SGNC, and Imperiled Species

A state endangered fish species (Taillight Shiner), one state threatened fish species (Bantam Sunfish), and four Species in Greatest Need of Conservation (SGNC; Flier (*Centrarchus macropterus*), Brown Bullhead (*Ameiurus nebulosus*), Banded Pygmy Sunfish (*Elassoma zonatum*), and Pugnose Minnow (*Opsopoeodus emiliae*)) were found in the swamp. A total of 93 Taillight Shiner, 6 Bantam Sunfish, 10 Flier, 9 Brown Bullhead, 1 Banded Pygmy Sunfish, and 14 Pugnose Minnow were collected. The majority of Taillight Shiners were found in vegetated habitat, with 73% in offshore vegetated and 25% in nearshore vegetated habitat. Of the Taillight Shiners caught, 53% were found in Eagle Pond. The majority of Flier were found in vegetated habitat, 70% in nearshore vegetated areas and 20% in offshore vegetated habitat. Paddlefish (*Polyodon spathula*) were only found in deep open water in the Cache River. Bantam

Sunfish were only caught in nearshore vegetated areas. Brown Bullhead and Pugnose Minnow were mostly found in vegetated habitats.

Water Depth and Elevation

Habitats had significantly different water depths. Macrohabitats differed significantly in mean depth (X^2=14.03, P=0.003), with Eagle Pond (mean: 0.85 m $\pm$ 0.03 SE) having significantly greater mean water depth than the side channels (mean: 0.64 m $\pm$ 0.02 SE; Figure 17). Microhabitats differed significantly in mean water depth (X^2=309.2, P<0.001), with open water (mean: 1.22 m $\pm$ 0.06 SE) having a significantly deeper mean water depth than nearshore (mean: 0.59 m $\pm$ 0.02 SE) and offshore vegetated (mean: 0.76 m $\pm$ 0.02 SE) habitats, while offshore vegetated was significantly deeper than nearshore vegetated habitats (Figure 17). Mean depth varied for microhabitats in each macrohabitat (Figure 17).

Using SIU data, seasons and years did not have significantly different water elevation; however, using IDNR long term data, water elevation differed among seasons and years. Using SIU data from 2020-2021, seasons had nearly significant differences in water elevation (X^2=7.97, P=0.051), with spring (mean: 100.41 MASL $\pm$ 0.03 SE) having a greater mean water elevation than fall (mean: 100.03 MASL $\pm$ 0.0.01 SE). Winter mean water elevation (mean: 100.07 MASL $\pm$ 0.01 SE) was not significantly different than fall water elevation. Looking at differences in water elevation between years, 2020 (mean: 100.17 MASL $\pm$ 0.01 SE) and 2021 (mean: 100.12 MASL $\pm$ 0.02 SE) did not have significantly different water elevation (X^2=0.04, P=0.833; Figure 17). Using IDNR historical water data (2010 - 2021), seasons had significantly different water elevation (X^2=10.97, P=0.012), with the average spring water elevation (mean:100.43 MASL $\pm$ 0.04 SE) being significantly higher than during fall (mean: 99.99 MASL $\pm$ 0.02 SE). Summer water elevation (mean: 100.22 MASL $\pm$ 0.04 SE) was not significantly different than winter (mean: 100.34 MASL $\pm$ 0.03 SE) and spring, and fall was not significantly different than winter, spring, and summer (Figure 18). Mean water elevation was significantly different among years from 2010 – 2021 (X^2=31.335, P=0.001), although when doing pairwise comparisons, years did not have significantly different water elevation from each other, which was likely due to lower statistical power of pairwise comparisons compared to the overall model. The average water elevation from 2010-2021 was 100.28 MASL (Figure 18). The water elevation in 2020 and 2021 was typical compared to the water elevation in the other years.

CPUE Spatiotemporal Differences

CPUE was significantly different among macrohabitats and microhabitats (depending on the gear used). Using electrofishing data, CPUE differed significantly among macrohabitats (X^2=17.15, P=0.001); the Cache River (mean CPUE: 3.37 $\pm$ 0.34 SE) had significantly higher CPUE than the main channel (mean CPUE: 1.99 $\pm$ 0.15 SE) and side channels (mean CPUE: 1.74 $\pm$ 0.13 SE; Table 1). CPUE differed significantly among microhabitats (X^2=39.79, P<0.001); nearshore (mean CPUE: 3.46 $\pm$ 0.42 SE) and offshore (mean CPUE: 2.44 $\pm$ 0.13 SE; Table 1) vegetated habitats had significantly higher CPUE than open water (mean CPUE: 1.76 $\pm$ 0.27 SE). CPUE differed significantly among seasons (X^2=16.69, P<0.001); winter (mean CPUE: 3.70 $\pm$ 0.53 SE) and fall (mean CPUE: 2.50 $\pm$ 0.17 SE) had significantly higher CPUE than summer (mean CPUE: 1.75 $\pm$ 0.13 SE). CPUE did not differ significantly between years from 2020 to 2021 using SIU data (X^2=0.22, P=0.637). CPUE differed significantly among macrohabitats in the winter (X^2=20.45, P<0.001); Cache River (mean CPUE: 5.63 $\pm$ 1.23 SE) and Eagle Pond (mean CPUE: 5.96 $\pm$ 1.18 SE) had significantly higher CPUE than the side channels (mean CPUE: 1.65 $\pm$ 0.47 SE), and Eagle Pond had significantly higher CPUE than the main channel (mean CPUE: 1.58 $\pm$ 0.29 SE). CPUE did not significantly differ among microhabitats in the winter; X^2=1.53, P=0.465.

Using fyke net data, CPUE differed significantly among macrohabitats (X^2=18.153, P<0.001); Cache River (mean CPUE: 13.02 $\pm$ 2.20 SE) and Eagle Pond (mean CPUE: 7.35 $\pm$ 1.03 SE) had significantly higher CPUE than the side channels (mean CPUE: 4.18 $\pm$ 0.77 SE), whereas using mini fyke net data there were no significant differences among macrohabitats (X^2=6.65, P=0.084). Using fyke net data, CPUE did not significantly differ among microhabitats (X^2=3.22, P=0.073); offshore vegetated habitat mean CPUE was 6.97 $\pm$ 0.76 SE and nearshore vegetated habitat mean CPUE was 10.35 $\pm$ 1.68 SE. Using mini fyke net data, CPUE differed significantly among microhabitats (X^2=20.59, P<0.001); nearshore vegetated habitat (CPUE: 11.90 $\pm$ 2.75 SE) had significantly higher CPUE than offshore vegetated habitat (CPUE: 2.34 $\pm$ 0.34 SE). Using fyke net data, CPUE was not significantly different among seasons (X^2=7.54, P=0.057) or between years (X^2=0.07, P=0.788). Using mini fyke net data, CPUE was significantly different among seasons (X^2=10.47, P=0.015); winter (mean CPUE: 22.72 $\pm$ 7.40 SE) had significantly higher CPUE than spring (mean CPUE: 3.19 $\pm$ 0.70 SE). Using mini fyke net data, CPUE was not significantly different between years (X^2=2.14, P=0.144).

DISCUSSION

Spatiotemporal Differences in Fish Assemblage Structure

Microhabitats had significantly different assemblage structure, similar to previous studies (Bice et al. 2014, Winemiller and Rose 1992), with open water being associated with different species than vegetated habitat. Shortnose Gar, Silver Carp, Gizzard Shad, and Threadfin Shad were indicator species for open water habitat, whereas Taillight Shiner, Warmouth, Redear Sunfish, Largemouth Bass, Bluegill, Golden Shiner, Orangespotted Sunfish, Black Crappie, and

Smallmouth Buffalo were indicator species for vegetated habitat. These trends make sense since Shortnose Gar, Silver Carp, Gizzard Shad, and Threadfin Shad are often found in deep, open water (Pflieger 1997, Miller et al. 2018). Additionally, Warmouth are known to be associated with vegetation and habitat structure (Pflieger 1997). Taillight Shiner, Redear Sunfish, Largemouth Bass, Bluegill, Golden Shiner, and Black Crappie are also often found near and thrive in vegetated or structural habitat (Pflieger 1997, Wheeler and Allen 2003, Robinson and Buchanan 2020). It is unlikely gear bias caused the difference in assemblage structure among the microhabitats since the electrofishing gear effectively shocks depths up to 1.5 m deep (Yeager et al. 1990) and there were only 29 occurances out of 252 electrofishing sites where sites had a water depth >1.5 m. Additionally, all microhabitats in the Cache River, Eagle Pond, and main swamp macrohabitats had instances where site depths were >1.5 m.

Fish assemblage structure differed seasonally in some habitats in Buttonland Swamp, consistent with seasonal habitat use in other floodplain ecosystems. There is evidence that fishes in floodplain lakes in the Amazon River and Orinoco River floodplain inhabit different habitats across seasons depending on their life history, biotic, and abiotic influences (Fernandes 1997, Rodriguez and Lewis 1997). Assemblage structure was significantly different in the side channels, main channel, and offshore vegetated habitat among seasons. Using electrofishing data, in the winter the Cache River and Eagle Pond had significantly higher CPUE than the side channels, and Eagle Pond had significantly higher CPUE than the main channel, which suggests fish are utilizing the deepest parts of the swamp for refuge in the winter since Eagle Pond and Cache River had the greatest mean water depths. Water levels were the highest in the spring and lowest in the fall, with water level fluctuating 0.91 m. Assemblage structure was significantly different among seasons through this hydrologic variation, where uncommon species (Spotted

Gar, Brook Silverside, and Taillight Shiner) were often associated with winter in the NMDS analysis. In the winter species are often easier to catch since cooler water temperature slows down the fish’s response/swimming ability (Brett and Glass 1973, Parsons and Smiley 2003) and species congregate to deeper areas for refuge from the cold, so seasonal bias on the electrofishing gear could have influenced this pattern. Previous studies also found water level fluctuations and season influenced fish assemblage structure in the Atchafalaya River Basin and Poyang Lake part of the Yangtze River (Bennet and Kozak 2016, Zhang et al. 2021).

There were species associated with different years most likely based on their life history and environmental tolerances. Gizzard Shad and Threadfin Shad were indicator species for 2020 since the majority were caught in 2020, their CPUE was higher in 2020, and only one Threadfin Shad was caught in 2021. The low shad CPUE for 2021 could have been caused by low winter temperatures in January of 2021 since these species are intolerant of cold temperatures (Griffith 1978, Fetzer et al. 2011). Gizzard Shad mortality typically increases when the water temperature drops below 8 to 4 ºC (Griffith 1978, Fetzer et al. 2011) and during this study the minimum water temperature recorded was 2.3 ºC.

Patterns of Species Richness Among Habitats

Species richness was highest in the side channels (and the Cache River, among macrohabitats) and vegetated habitat (among microhabitats), similar to previous studies (Sylvester and Broughton 1983, Koel 2004, Miller et al. 2018), highlighting these habitats as potentially important for maintaining fish species richness in Buttonland Swamp. In the upper Mississippi River system that is constrained by dams, side channels have been reported to be associated with high species richness and essential habitat that provides refuge for fishes (Sylvester and Broughton 1983, Koel 2004). Open water had lower species richness and CPUE (using electrofishing data) than vegetated habitats, which could partly be explained by vegetation potentially providing refuge for small fishes from predation or increased food availability such as invertebrates (Werner et al. 1983a, Werner et al. 1983b, Rozas and Odum 1988). In Lake Erie, although a much larger system than Buttonland Swamp, there is also higher species richness and CPUE in vegetated habitat and a different fish assemblage structure in vegetated habitat compared to open water (Miller et al. 2018). Using fyke net and mini fyke net data, nearshore habitats in Buttonland Swamp had higher CPUE than offshore habitats. This may have been due to higher capture efficiency of fyke nets in nearshore locations. Nearshore areas may also have had more allochthonous nutrients and tended to have more physical habitat complexity (detritus, sticks and logs, etc.) than offshore or open water microhabitats (Duncan and Kubecka 1995).

Comparisons of Species Proportions and CPUE Among Habitats

The macrohabitats had similar proportions of species, while microhabitats varied with open water varying in proportion and had more diversity compared to vegetated habitats. Bluegill made up over 40% of the total catch, made up most of the proportion of species caught in vegetated habitat, and were associated with vegetated habitat, which could have contributed to open water having more diversity compared to vegetated habitats, since Bluegill comprised a much lower percentage of the catch in open water habitat.

In contrast to the similar proportional contributions of fish species to the assemblage in macrohabitats, macrohabitats had significantly different CPUE. The Cache River had significantly higher CPUE than the main swamp and side channels. This similarity in proportions among macrohabitats is reflected in the NMDS, where there were no significant differences in the assemblage structure. However, although there were similar proportions of species in each macrohabitat, the Cache River had more fish overall (and the highest diversity) since although there are dams and levees, the Cache River channel is the primary entry point for some fishes (Paddlefish, Silver Carp, Black Carp) coming into the swamp. Fish species may congregate in the Cache River since fish tend to use the main channel of rivers as a migration corridor to access floodplains, side channels, etc. (Junk et al. 1989). This suggests that the Cache River has high habitat heterogeneity and is associated with more habitat characteristics than other macrohabitats, which could influence high diversity and CPUE (Sylvester and Broughton 1983). Eagle Pond is located farthest from the Cache River channel and is more isolated, which could contribute to its low diversity (Ward et al. 1999).

Comparison of IDNR and SIU data

Species assemblage structure did not differ yearly using the IDNR data, but the IDNR and SIU data had significantly different assemblage structure. Using the IDNR data, proportion of families did not seem to change much over time from 1992 to 2020, which could be due to fragmentation of the system since dams and levees on either side of the swamp may be limiting the movement of fishes into and out of the swamp, although some species are able to pass through the dams (Paddlefish, Silver Carp, Black Carp). Many species (Silver Carp, Brook Silverside, Shortnose Gar, Threadfin Shad) that were associated with SIU data in the NMDS analysis were also associated with open water, whereas most species associated with the IDNR data were also associated with nearshore or offshore vegetation (Largemouth Bass, Spotted Gar, Common Carp, Black Buffalo, Longear Sunfish, Bigmouth Buffalo, Warmouth, Black Crappie, White Crappie, Bowfin, etc.). Most sites the IDNR surveyed were near Buttonbush or nearshore vegetated habitat, and almost no sites sampled solely open water. There was also a difference in effort, where the IDNR usually sampled once a year, whereas SIU performed 16 electrofishing transects per month. However, species that the IDNR data were associated with were still found in SIU data although they were either not strongly associated with SIU data or were found in another season that the IDNR did not sample, so those data were excluded. SIU and IDNR data had similar abundances of common species (Gizzard Shad, buffalos, crappies, Bluegill), although rare species or species that vary year-to-year were different in proportions (Silver Carp, Threadfin Shad, Spotted Bass, Redear Sunfish from SIU data, and Green Sunfish and Mosquitofish from IDNR data)) possibly because of interannual changes in abundances or microhabitats that were sampled, or the amount of time sites were sampled. Additionally, the IDNR caught more darter species than SIU did, which could be because the location the IDNR sample every year is the same, which is by a bridge that has artificial rocks where darters would more likely be found.

The IDNR data showed that the proportion of families has not changed much in the past 20 years. This shows that it is likely that the proportion of families will stay relatively the same in Buttonland Swamp in the future. However, since SIU data had significantly different assemblage structure compared to IDNR data, it is likely that there will be different proportions of fish species than the historic IDNR data if sampling throughout Buttonland Swamp, in different habitats, year-round, since IDNR caught more species associated with vegetated habitat, while SIU caught more species associated with open water.

Endangered, Threatened, SGNC, and Imperiled Species

A state endangered fish (Taillight Shiner), state threatened fish (Bantam Sunfish), and SGNC (Flier, Brown Bullhead, Banded Pygmy Sunfish, Pugnose Minnow) were found in the swamp and the majority were found in vegetated habitat. Paddlefish have only been found in deep open water (Pflieger 1997) in the Cache River. Paddlefish have nomadic tendencies where their movement is variable, and they have been found to move freely among rivers (Tripp et al.

2019). The Taillight Shiner was associated with the amount of Buttonbush which is consistent with prior studies of habitat use for this species (Cowell and Barnett 1974, Robinson and Buchanan 2020). Extensive year-round fish surveys are necessary in river floodplain habitats to acquire a basis of what species are present; The Taillight Shiner was thought to be extirpated in Illinois and had not been seen in over 30 years until these surveys rediscovered them.

Implications for Water Level Management

Buttonland Swamp acts as a shallow lake more than a floodplain and has lost its natural flood pulse dynamics that many biota depend on because of low head dams on either side of the swamp (Welcomme 1995, Bennet et al. 2001). Water level fluctuations influence what amount of habitats are inundated and accessible for fishes, therefore influencing habitat and food availability, which can affect food web structure, fish assemblage structure, and ecosystem productivity (Crook et al. 2020, Magoulick et al. 2021, Silva et al 2021). As water levels lower, this limits the amount of habitat that fish can utilize, forcing fish into a smaller area, possibly increasing predation, competition, and densities (Resh et al. 1988, Smith et al. 2005). Our data do not allow us to predict how water level management specifically intended to increase Bald Cypress recruitment might affect fish, because water levels fluctuated naturally during the course of this study and may not have fluctuated as much as would be required to promote Bald Cypress recruitment. The associations between many species (including E&T) and shallow, vegetated areas suggest that these species might be most affected by a prolonged water level drawdown since shallow areas would be expected to be the first places to dry up at low water levels and could leave fish stranded. Buttonland Swamp provides refuge for many rare, threatened, and endangered fish species, so it is important that water level management takes these species into account. Also, fish were associated with deeper areas in the winter, drawdown during that time of year might limit the amount of deep water habitat for fish to use. Additionally, Koel and Sparks (2002) suggest habitat specialists may be more influenced by water level fluctuations since they depend on access to inundated bank areas and harder substrate to make nests. Bald Cypress recruitment depends on flood pulses and has since been limited to the perimeter of the swamp since seeds cannot germinate while inundated (Bennet et al. 2001). Although water level management under the current configuration, using Diehl Dam to manipulate the water level, may be more realistic, removing the low-head dams and Post Creek Cutoff could potentially restore the pulse flood and benefit biota by restoring habitat connectivity (Burroughs et al. 2010) and increasing access to spawning habitat for migratory fishes (Shafroth et al. 2002). Continuing to manage Buttonland Swamp under the current water management plan using Diehl Dam would not involve a risk to fishes due to the water level that Buttonland Swamp is maintained at year-round. However, this could eventually lead to the loss of Bald Cypress within the swamp, which would eliminate a key plant species from this area. The loss of Bald Cypress could have negative implications for fish if other vegetation does not provide equivalent habitat. However, it is unknown how fish will be impacted by water level that is beneficial to Bald Cypress growth, so conducting an experimental draw-down that is specifically designed (length, magnitude of water level reduction) to promote Bald Cypress recruitment may be necessary to directly evaluate effects of water draw-down on fish. This would address a key limitation of this study in that water levels low enough to determine whether a Cypress-recruitment-promoting drawdown would negatively affect fishes were not observed. However, this experiment would also carry some risks to fishes that are strongly associated with shallow habitat areas since water level drawdown can negatively influence fish size-structure (Paller 1997, Fischer and Ohl 2005), abundance (Paller 1997), species richness (Paller 1997), survival (early and rapid drawdowns can result in winterkills (Gaboury and Patalas 1984)), and/or spawning and reproduction (Gaboury and Patalas 1984, Fischer and Ohl 2005).

Habitat	Caught (total count; electrofishing only)	CPUE (per minute; electrofishing only)	Species Richness (gears combined)	Diversity (H'; gears combined)
Cache River	3183	3.37	47	2.47
Eagle Pond	2650	2.80	40	2.23
Main Channel	1909	2.02	41	2.37
Side Channels	1649	1.70	47	2.38
Nearshore Vegetated	2592	3.46	51	2.10
Offshore Vegetated	5158	2.44	48	2.41
Open Water	1641	1.79	39	2.63

Table 1. Number of fish caught and CPUE for SIU (using only electrofishing), and species richness and diversity (with gears combined) for each macrohabitat and microhabitat in Buttonland Swamp within the Cache River watershed in 2020-2021.

Species (*Genus species*)	**Habitat/Season/Year**	**Indicator Value**	**P**
Smallmouth Buffalo (*Ictiobus bubalus*)	Cache River	0.280	<0.001
Orangespotted Sunfish (*Lepomis humilis*)	Cache River	0.265	<0.001
Bigmouth Buffalo (*Ictiobus cyprinellus*)	Cache River	0.252	<0.001
Silver Carp (*Hypophthalmichthys molitrix*)	Cache River	0.243	<0.001
White Crappie (P*omoxis annularis*)	Cache River	0.197	0.010
Taillight Shiner (*Notropis maculatus*)	Eagle Pond	0.182	0.021
Spotted Gar (*Lepisosteus oculatus*)	Main Channel	0.197	0.010
Shortnose Gar (*Lepisosteus platostomus*)	Cache River + Eagle Pond	0.241	<0.001
Black Crappie (*Pomoxis nigromaculatus*)	Cache River + Eagle Pond	0.227	0.003
Bowfin (*Amia calva*)	Cache River + Eagle Pond	0.216	0.002
Black Buffalo (*Ictiobus niger*)	Cache River + Eagle Pond	0.208	0.010
Brook Silverside (*Labidesthes sicculus*)	Main Channel + Side Channel	0.175	0.030
Warmouth (*Labidesthes sicculus*)	Cache River + Eagle Pond + Main Channel	0.196	0.003
Warmouth (*Labidesthes sicculus*)	Nearshore Vegetated	0.330	<0.001
Black Crappie (*Pomoxis nigromaculatus*)	Nearshore Vegetated	0.233	0.002
Smallmouth Buffalo (*Ictiobus bubalus*)	Nearshore Vegetated	0.230	0.002
Shortnose Gar (*Lepisosteus platostomus)*	Open Water	0.388	<0.001
Silver Carp (*Hypophthalmichthys molitrix*)	Open Water	0.383	<0.001
Threadfin Shad (*Dorosoma petenense*)	Open Water	0.184	0.014
Gizzard Shad *Dorosoma cepedianum*)	Open Water	0.176	0.024
Bluegill (*Lepomis macrochirus)*	Nearshore + Offshore Vegetated	0.535	<0.001
Redear Sunfish (*Lepomis microlophus*)	Nearshore + Offshore Vegetated	0.343	<0.001
Orangespotted Sunfish (*Lepisosteus platostomus*)	Nearshore + Offshore Vegetated	0.284	<0.001

Largemouth Bass (*Micropterus salmoides)*	Nearshore + Offshore Vegetated	0.247	0.001
Golden Shiner (*Notemigonus crysoleucas)*	Nearshore + Offshore Vegetated	0.212	0.003
Taillight Shiner (*Notropis maculatus*)	Nearshore + Offshore Vegetated	0.194	0.013
White Crappie (*Pomoxis annularis*)	Open Water + Nearshore Vegetated	0.167	0.041
Brook Silverside (*Labidesthes sicculus*)	Open Water + Offshore Vegetated	0.204	0.006
Warmouth (*Labidesthes sicculus*)	Spring	0.402	<0.001
Spotted Gar (*Lepisosteus oculatus*)	Winter	0.468	<0.001
Black Crappie (*Pomoxis nigromaculatus*)	Winter	0.304	<0.001
Bowfin (*Amia calva*)	Winter	0.292	<0.001
Brook Silverside (*Labidesthes sicculus*)	Winter	0.251	0.001
Taillight Shiner (*Notropis maculatus*)	Winter	0.231	0.003
Bluegill (*Lepomis macrochirus)*	Fall + Spring	0.215	0.007
Threadfin Shad (*Dorosoma petenense*)	Fall + Summer	0.211	0.006
Golden Shiner (*Notemigonus crysoleucas)*	Fall + Summer	0.206	0.007
Black Buffalo (*Ictiobus niger*)	Fall + Winter	0.244	0.002
White Crappie (*Pomoxis annularis*)	Fall + Winter	0.177	0.033
Bigmouth Buffalo (*Ictibus cyprinellus*)	Fall + Winter	0.174	0.033
Freshwater Drum (*Aplodinotus grummiens*)	Fall + Spring + Summer	0.179	0.034
Gizzard Shad *Dorosoma cepedianum*)	Fall + Summer + Winter	0.188	0.020
Threadfin Shad (*Dorosoma petenense*)	2020	0.274	<0.001
Gizzard Shad (*Dorosoma cepedianum*)	2020	0.158	0.013
Black Buffalo (*Ictiobus niger*)	2021	0.225	<0.001
Redear Sunfish (*Lepomis microlophus*)	2021	0.222	0.001
Bluegill (*Lepomis macrochirus)*	2021	0.172	0.010

Warmouth (*Labidesthes sicculus*)	2021	0.134	0.043

Table 2. Indicator Species Analysis using SIU electrofishing data from Buttonland Swamp within the Cache River watershed for each macrohabitat, microhabitat, season, and year in 2020-2021 ($\alpha = 0.05$).

Species	**Data Source**	**Indicator Value**	**P**
Redear Sunfish (*Lepomis microlophus*)	SIU	0.488	<0.001
Brook Silverside (*Labidesthes sicculus*)	SIU	0.276	<0.001
Silver Carp (*Hypophthalmichthys molitrix*)	SIU	0.197	0.017
Largemouth Bass (*Micropterus salmoides)*	IDNR	0.531	<0.001
Gizzard Shad *Dorosoma cepedianum*)	IDNR	0.475	<0.001
Spotted Gar (*Lepisosteus oculatus*)	IDNR	0.463	<0.001
Common Carp (*Cyprinus carpio)*	IDNR	0.416	<0.001
Black Buffalo (*Ictiobus niger*)	IDNR	0.386	<0.001
Longear Sunfish (*Lepomis megalotis*)	IDNR	0.370	<0.001
Bigmouth Buffalo (*Ictibus cyprinellus*)	IDNR	0.363	<0.001
Warmouth (*Labidesthes sicculus*)	IDNR	0.324	<0.001
White Crappie (*Pomoxis annularis*)	IDNR	0.320	<0.001
Bowfin (*Amia calva*)	IDNR	0.313	<0.001
Black Crappie (*Pomoxis nigromaculatus*)	IDNR	0.299	<0.001
Smallmouth Buffalo (*Ictiobus bubalus*)	IDNR	0.280	<0.001
Orangespotted Sunfish (*Lepisosteus platostomus*)	IDNR	0.246	0.002
Green Sunfish (*Lepomis cyanellus*)	IDNR	0.217	0.001
Mosquitofish (*Gambusia affinis*)	IDNR	0.210	<0.001
Pugnose Minnow (*Opsopoeodus emiliae*)	IDNR	0.201	0.002
Flier (*Centrarchus macropterus*)	IDNR	0.200	0.006

Slough Darter (*Etheostoma gracile*)	IDNR		0.168	0.021
Golden Shiner (*Notemigonus crysoleucas)*	IDNR		0.168	0.044
Channel Catfish (*Ictalurus punctatus*)	IDNR		0.154	0.020
Spotted Sucker (*Minytrema melanops*)	IDNR		0.152	0.005

Table 3. Indicator Species Analysis comparing SIU electrofishing data (SIU; 2020-2021) and IDNR data (IDNR; 1992-2020) from Buttonland Swamp within the Cache River watershed within June - September ($\alpha = 0.05$).

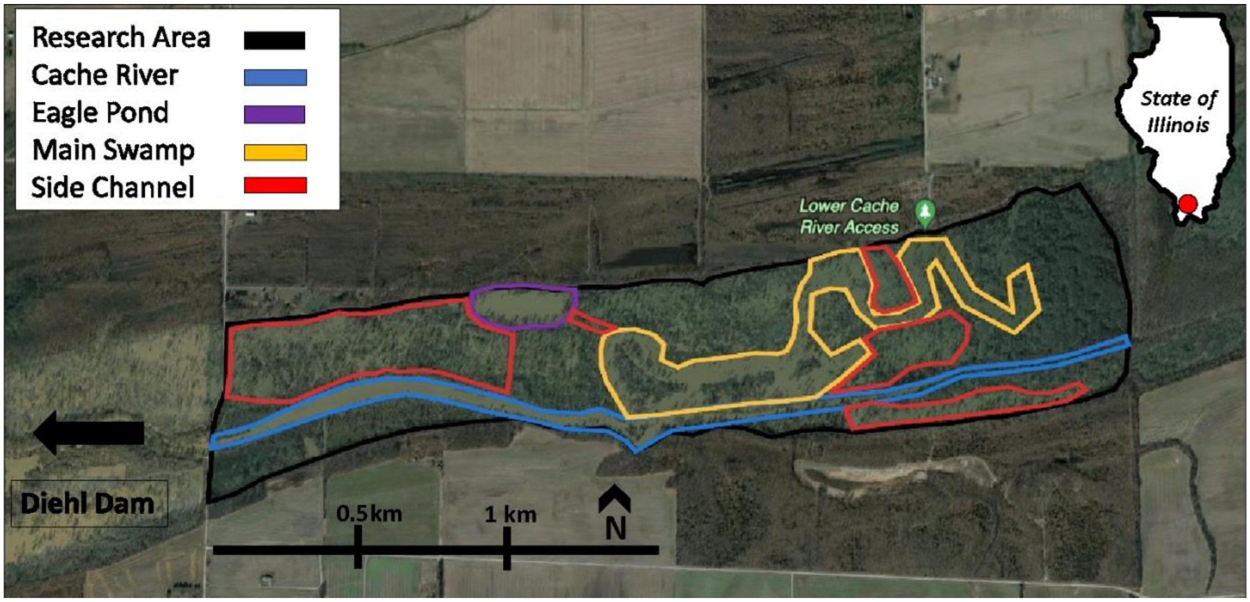

Figure 1. Map showing Buttonland Swamp within the Cache River watershed separated into four macrohabitats sampled for this study from 2020-2021.

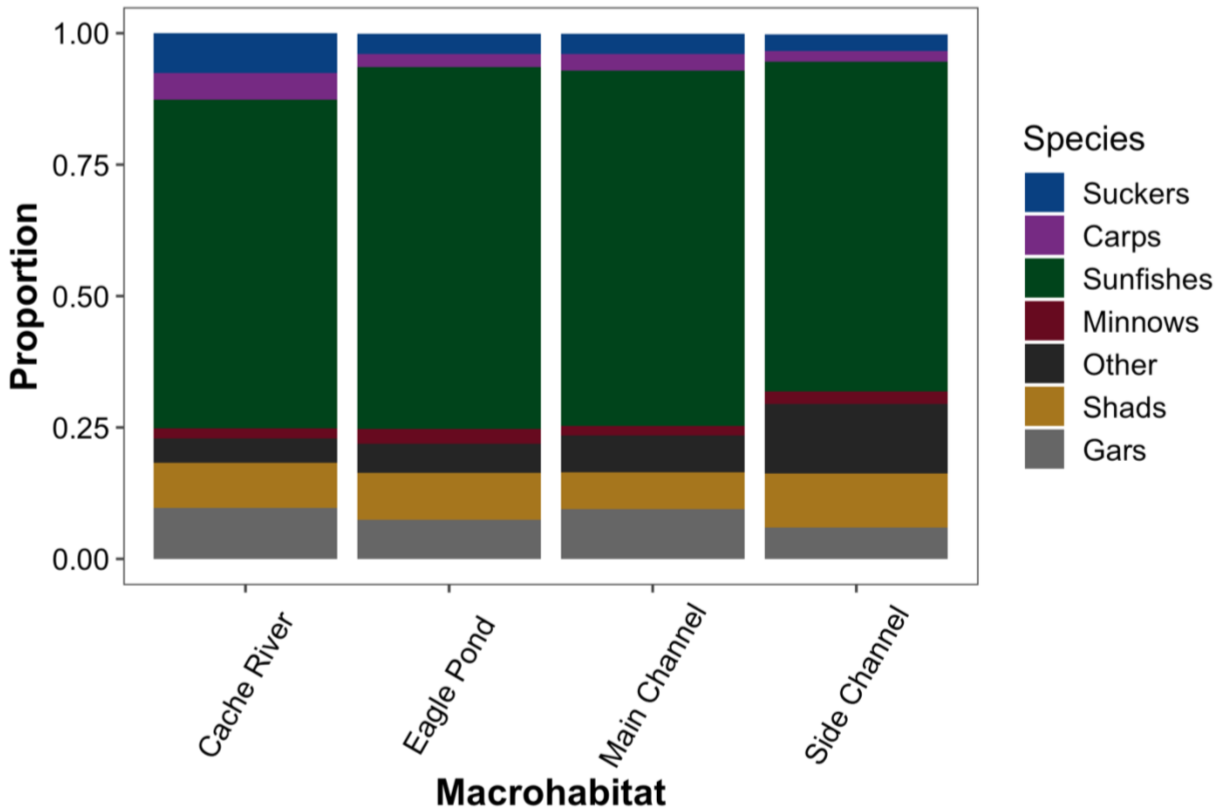

Figure 2. Proportion of species in each macrohabitat with fish species grouped into seven categories using SIU data from Buttonland Swamp within the Cache River watershed in 2020-2021 using all gears. The “other” category includes Black Bullhead (*Ameiurus melas*), Bluntnose Darter (*Etheostoma chlorosoma*), Bowfin (*Amia calva*), Brook Silverside (*Labidesthes sicculus*), Brown Bullhead (*Ameiurus nebulosus*), Channel Catfish (*Ictalurus punctatus*), Flathead Catfish (*Pylodictis olivaris*), Freshwater Drum (*Aplodinotus grunniens*), Grass Pickerel (*Esox americanus*), Johnny Darter (*Etheostoma nigrum*), Mosquitofish (*Gambusia affinis*), Mud Darter (*Etheostoma asprigene*), Paddlefish (*Polyodon spathula*), Pirate Perch (*Aphredoderus sayanus*), Sauger (*Sander canadensis*), Slough Darter (*Etheostoma gracile*), Tadpole Madtom (*Noturus gyrinus*), Walleye (*Sander vitreus*), White Bass (*Morone chrysops*), and Yellow Bullhead (*Ameiurus natalis*)).

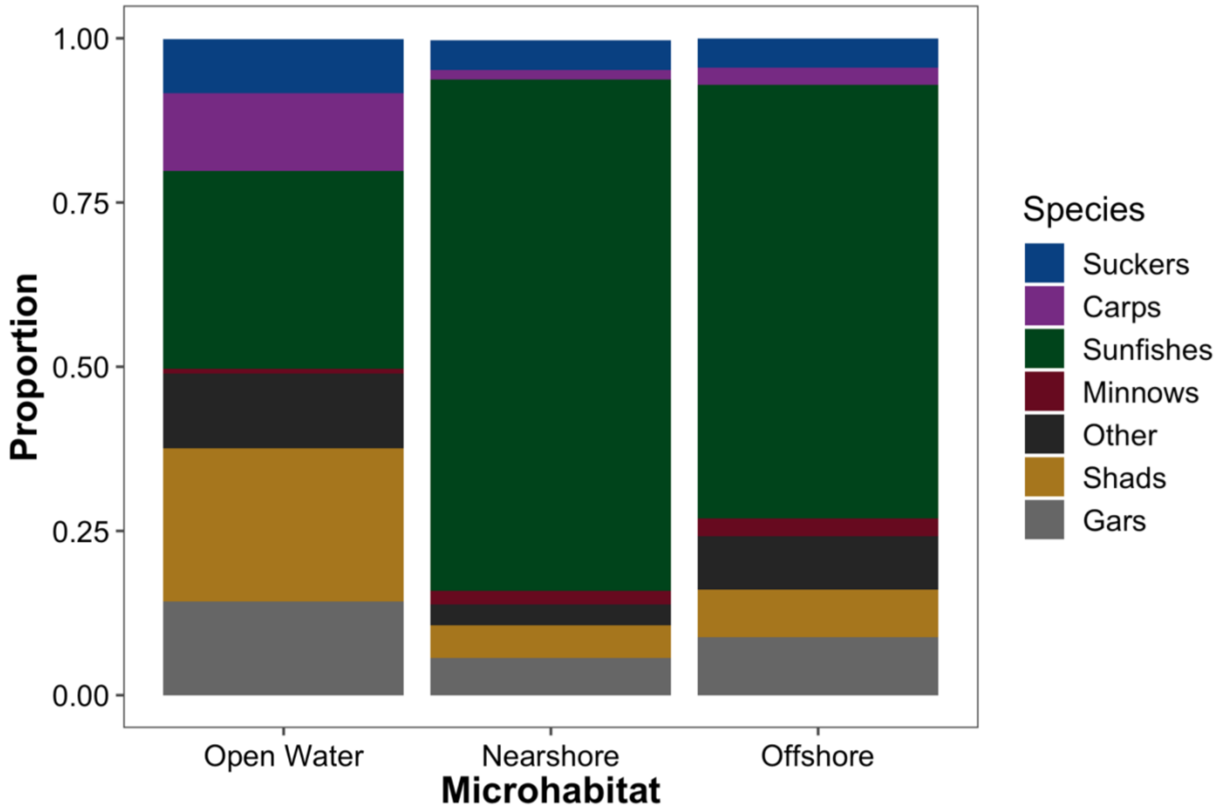

Figure 3. Proportion of species in each microhabitat with fish species grouped into seven categories using SIU data from Buttonland Swamp within the Cache River watershed in 2020-2021 using all gears. The "other" category includes Black Bullhead (*Ameiurus melas*), Bluntnose Darter (*Etheostoma chlorosoma*), Bowfin (*Amia calva*), Brook Silverside (*Labidesthes sicculus*), Brown Bullhead (*Ameiurus nebulosus*), Channel Catfish (*Ictalurus punctatus*), Flathead Catfish (*Pylodictis olivaris*), Freshwater Drum (*Aplodinotus grunniens*), Grass Pickerel (*Esox americanus*), Johnny Darter (*Etheostoma nigrum*), Mosquitofish (*Gambusia affinis*), Mud Darter (*Etheostoma asprigene*), Paddlefish (*Polyodon spathula*), Pirate Perch (*Aphredoderus sayanus*), Sauger (*Sander canadensis*), Slough Darter (*Etheostoma gracile*), Tadpole Madtom (*Noturus gyrinus*), Walleye (*Sander vitreus*), White Bass (*Morone chrysops*), and Yellow Bullhead (*Ameiurus natalis*)).

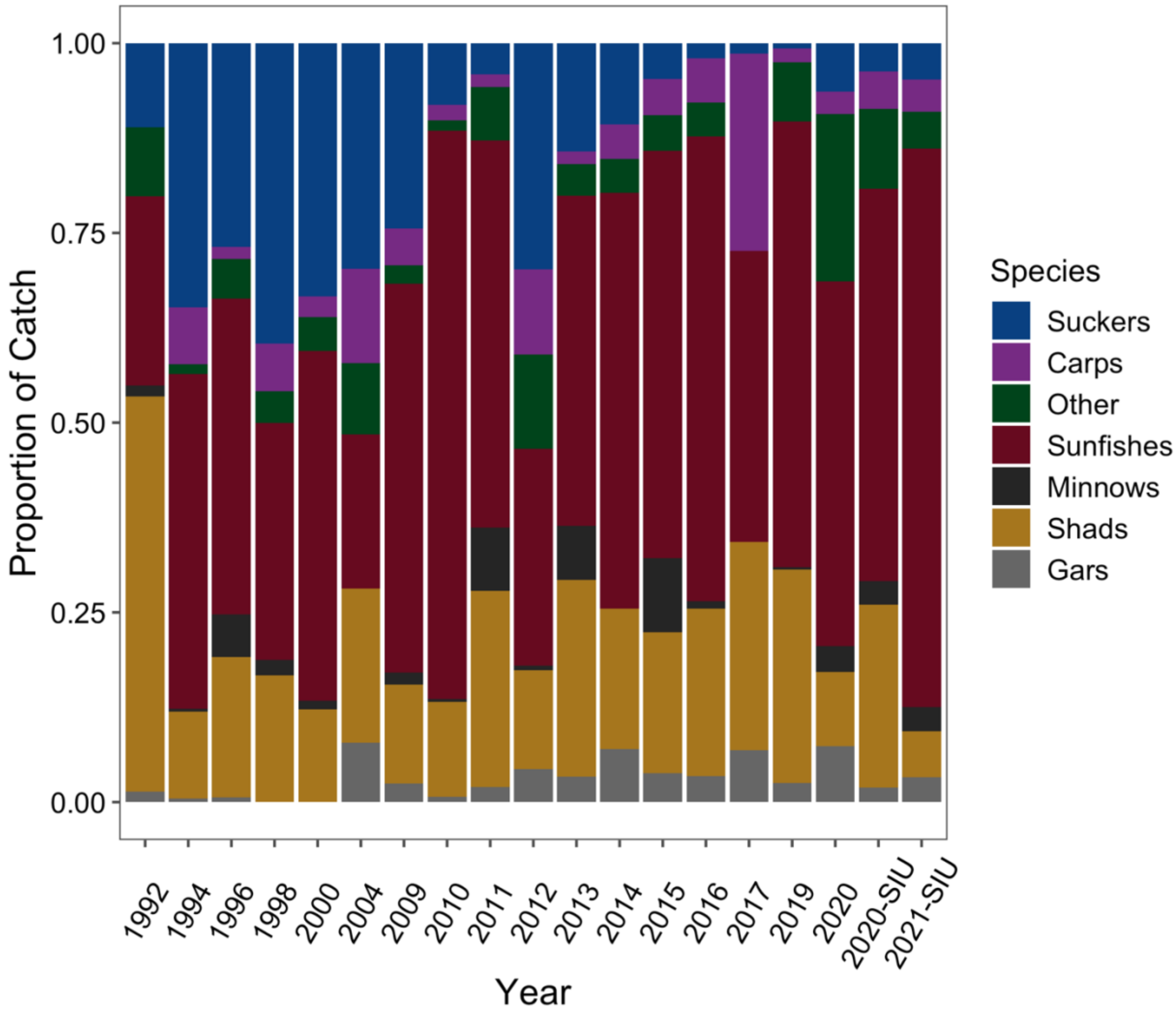

Figure 4. Using historical IDNR data (1992-2020) proportion of fish species was calculated for each year in Buttonland Swamp, including SIU data (2020-SIU, 2021-SIU), within the Cache River watershed (2020-2021) using just electrofishing data from June - September. The “other” category includes Black Bullhead (*Ameiurus melas*), Bluntnose Darter (*Etheostoma chlorosoma*), Bowfin (*Amia calva*), Brook Silverside (*Labidesthes sicculus*), Brown Bullhead (*Ameiurus nebulosus*), Channel Catfish (*Ictalurus punctatus*), Flathead Catfish (*Pylodictis olivaris*), Freshwater Drum (*Aplodinotus grunniens*), Grass Pickerel (*Esox americanus*), Johnny Darter (*Etheostoma nigrum*), Mosquitofish (*Gambusia affinis*), Mud Darter (*Etheostoma asprigene*), Paddlefish (*Polyodon spathula*), Pirate Perch (*Aphredoderus sayanus*), Sauger (*Sander canadensis*), Slough Darter (*Etheostoma gracile*), Tadpole Madtom (*Noturus gyrinus*), Walleye (*Sander vitreus*), White Bass (*Morone chrysops*), and Yellow Bullhead (*Ameiurus natalis*)).

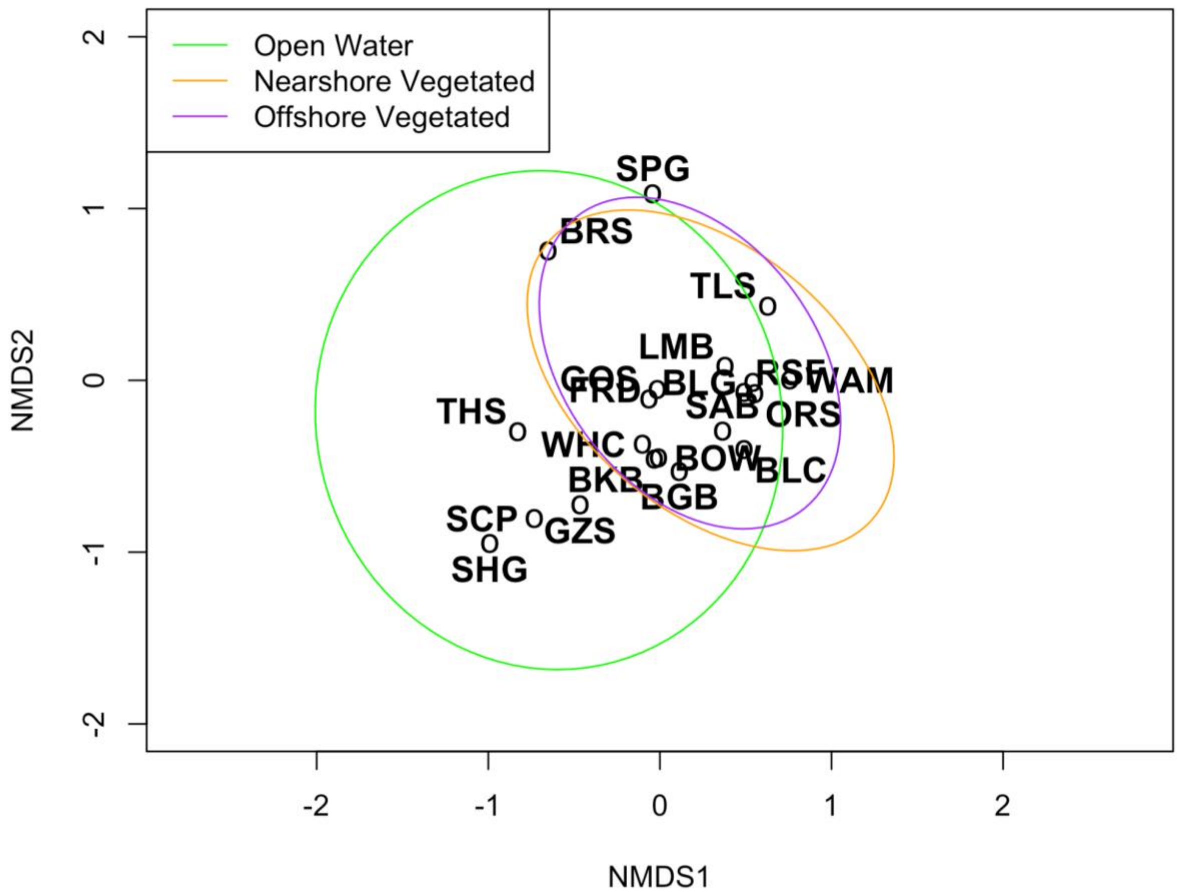

Figure 5. NMDS ordination of fish assemblage structure within microhabitats using SIU electrofishing data from Buttonland Swamp within the Cache River watershed in 2020-2021 (Stress: 0.148). Ellipses encompassed 95% of the variation and the small black circles represent the species codes position to reduce overlap of the codes. Fish include Bigmouth Buffalo (BGB; *Ictibus cyprinellus*), Black Buffalo (BKB; *Ictiobus niger*), Black Crappie (BLC; *Pomoxis nigromaculatus*), Bluegill (BLG; *Lepomis macrochirus*), Bowfin (BOW; *Amia calva*), Brook Silverside (BRS; *Labidesthes sicculus*), Freshwater Drum (FRD; *Aplodinotus grummiens*), Golden Shiner (GOS; *Notemigonus crysoleucas*), Gizzard Shad (GZS; *Dorosoma cepedianum*), Largemouth Bass (LMB; *Micropterus salmoides*), Orangespotted Sunfish (ORS; *Lepomis humilis*), Redear Sunfish (RSF; *Lepomis microlophus*), Shortnose Gar (SHG; *Lepisosteus platostomus*), Silver Carp (SCP; *Hypophthalmichthys molitrix*), Smallmouth Buffalo (SAB; *Ictiobus bubalus*), Spotted Gar (SPG; *Lepisosteus oculatus*), Taillight Shiner (TLS; *Notropis maculatus*), Threadfin Shad (THS; *Dorosoma petenense*), Warmouth (WAM; *Lepomis gulosus*), White Crappie (WHC; *Pomoxis annularis*).

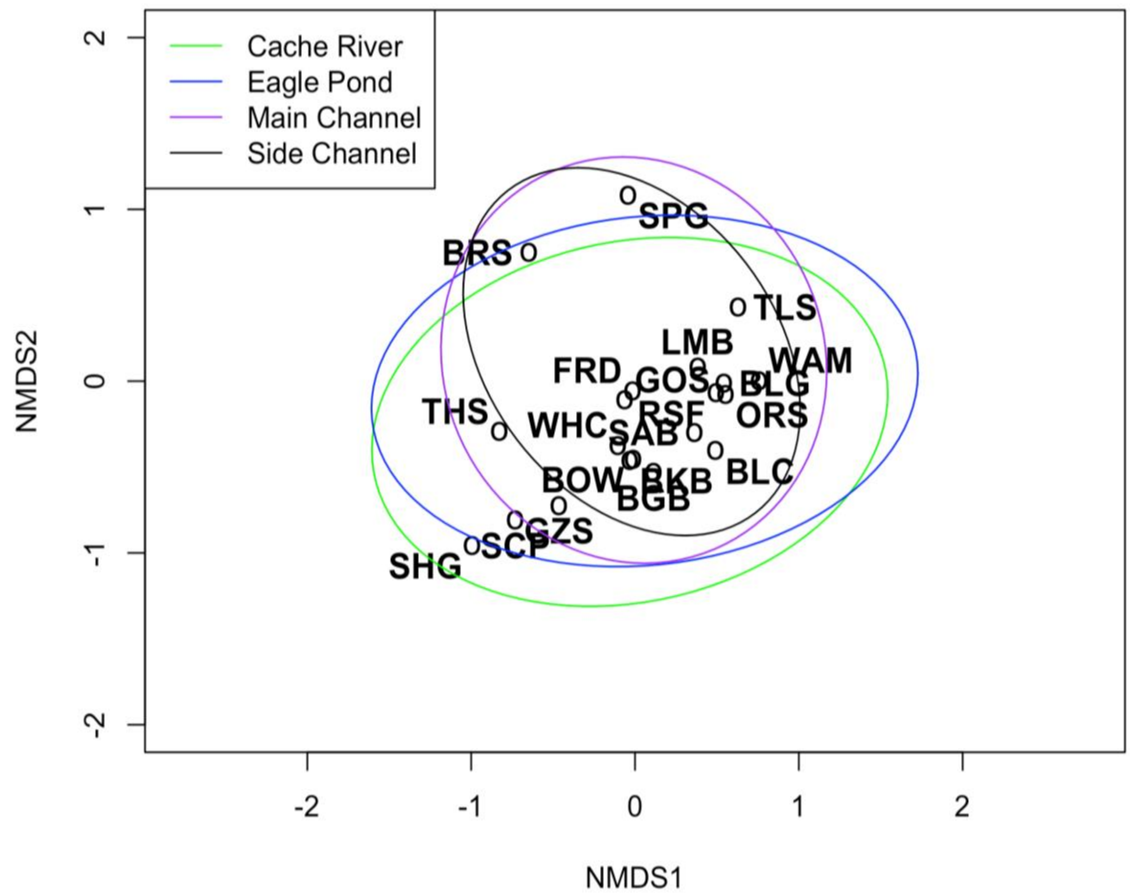

Figure 6. NMDS ordination of fish assemblage structure within macrohabitats using SIU electrofishing data Buttonland Swamp within the Cache River watershed in 2020-2021 (Stress: 0.148). Ellipses encompassed 95% of the variation and the small black circles represent the species codes position to reduce overlap of the codes. Fish include Bigmouth Buffalo (BGB; *Ictibus cyprinellus*), Black Buffalo (BKB; *Ictiobus niger*), Black Crappie (BLC; *Pomoxis nigromaculatus*), Bluegill (BLG; *Lepomis macrochirus*), Bowfin (BOW; *Amia calva*), Brook Silverside (BRS; *Labidesthes sicculus*), Freshwater Drum (FRD; *Aplodinotus grummiens*), Golden Shiner (GOS; *Notemigonus crysoleucas*), Gizzard Shad (GZS; *Dorosoma cepedianum*), Largemouth Bass (LMB; *Micropterus salmoides*), Orangespotted Sunfish (ORS; *Lepomis humilis*), Redear Sunfish (RSF; *Lepomis microlophus*), Shortnose Gar (SHG; *Lepisosteus platostomus*), Silver Carp (SCP; *Hypophthalmichthys molitrix*), Smallmouth Buffalo (SAB; *Ictiobus bubalus*), Spotted Gar (SPG; *Lepisosteus oculatus*), Taillight Shiner (TLS; *Notropis maculatus*), Threadfin Shad (THS; *Dorosoma petenense*), Warmouth (WAM; *Lepomis gulosus*), White Crappie (WHC; *Pomoxis annularis*).

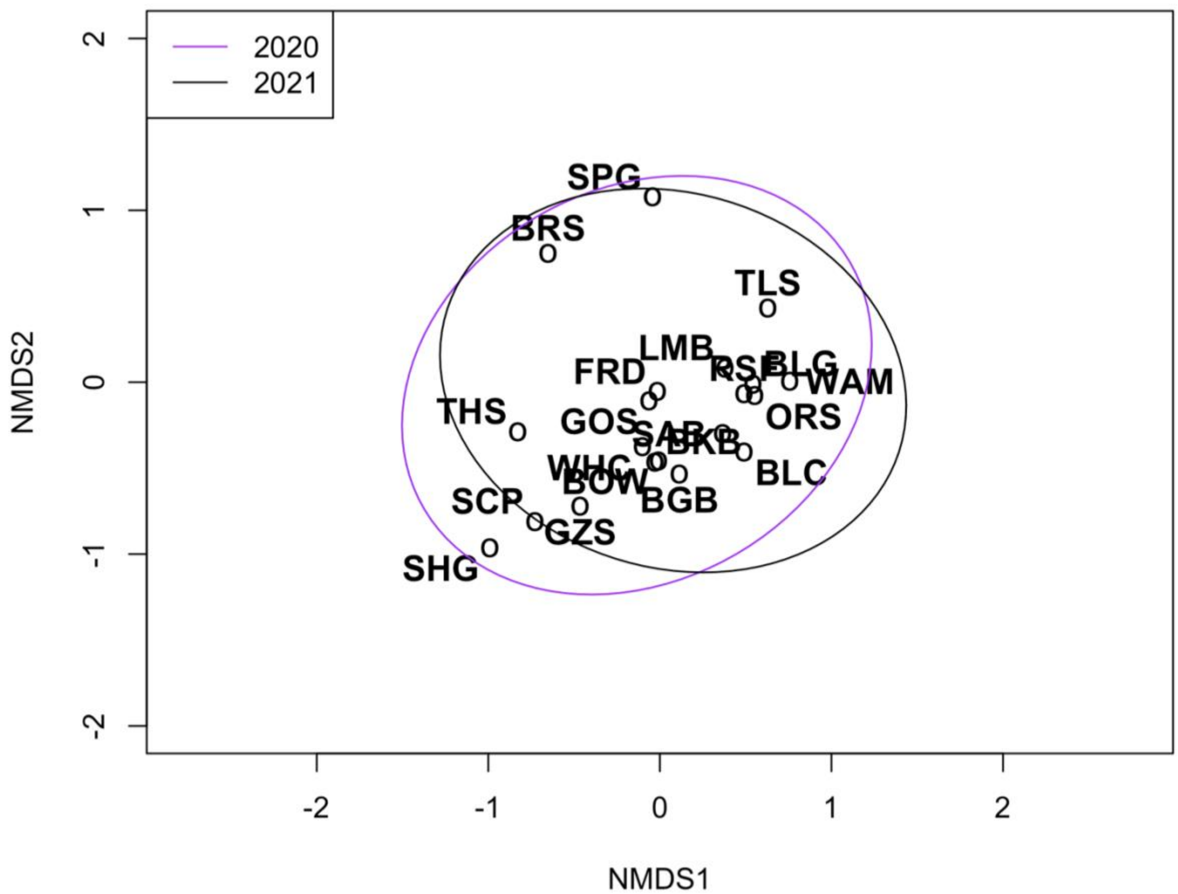

Figure 7. NMDS ordination of fish assemblage structure within years using SIU electrofishing data Buttonland Swamp within the Cache River watershed in 2020-2021 (Stress: 0.148). Ellipses encompassed 95% of the variation and the small black circles represent the species codes position to reduce overlap of the codes. Fish include Bigmouth Buffalo (BGB; *Ictibus cyprinellus*), Black Buffalo (BKB; *Ictiobus niger*), Black Crappie (BLC; *Pomoxis nigromaculatus*), Bluegill (BLG; *Lepomis macrochirus*), Bowfin (BOW; *Amia calva*), Brook Silverside (BRS; *Labidesthes sicculus*), Freshwater Drum (FRD; *Aplodinotus grummiens*), Golden Shiner (GOS; *Notemigonus crysoleucas*), Gizzard Shad (GZS; *Dorosoma cepedianum*), Largemouth Bass (LMB; *Micropterus salmoides*), Orangespotted Sunfish (ORS; *Lepomis humilis*), Redear Sunfish (RSF; *Lepomis microlophus*), Shortnose Gar (SHG; *Lepisosteus platostomus*), Silver Carp (SCP; *Hypophthalmichthys molitrix*), Smallmouth Buffalo (SAB; *Ictiobus bubalus*), Spotted Gar (SPG; *Lepisosteus oculatus*), Taillight Shiner (TLS; *Notropis maculatus*), Threadfin Shad (THS; *Dorosoma petenense*), Warmouth (WAM; *Lepomis gulosus*), White Crappie (WHC; *Pomoxis annularis*).

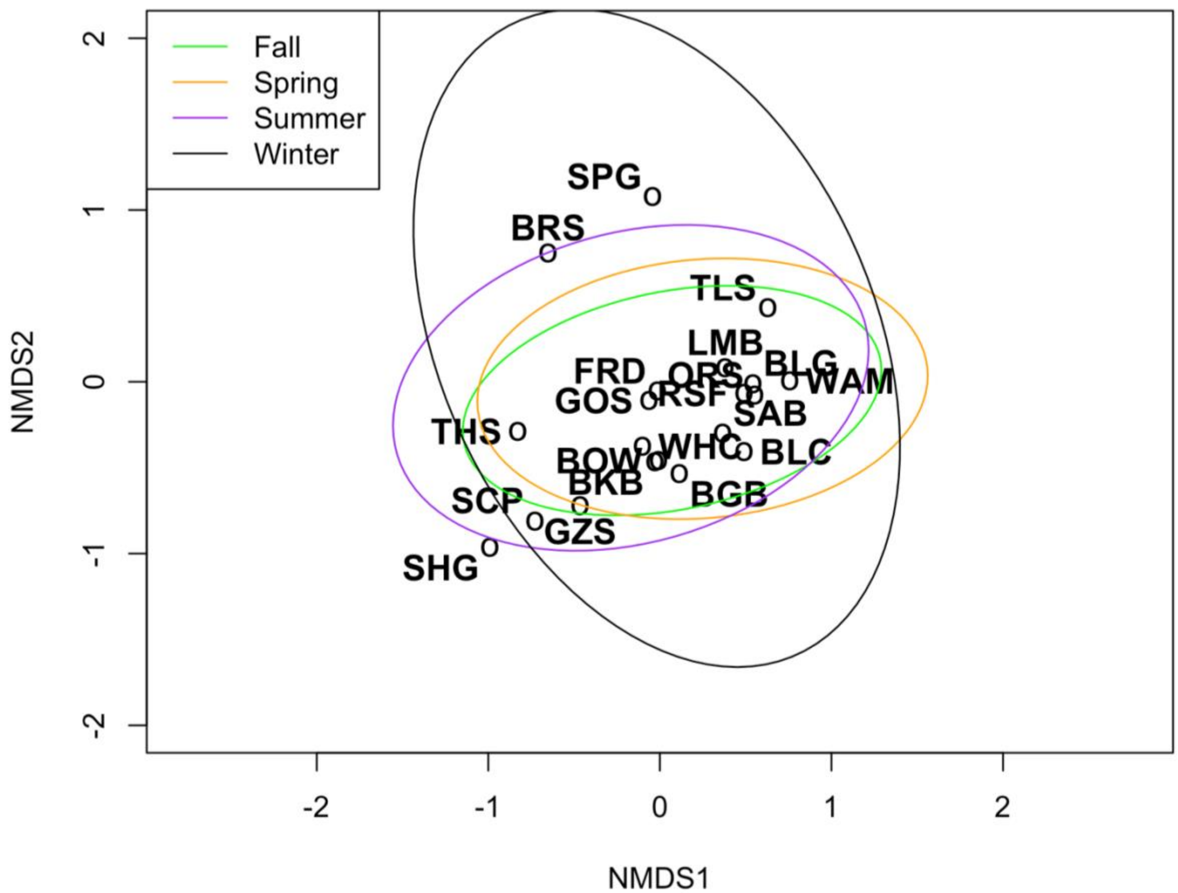

Figure 8. NMDS ordination of fish assemblage structure within seasons, with all data pooled together, using SIU electrofishing data Buttonland Swamp within the Cache River watershed in 2020-2021 (Stress: 0.148). Ellipses encompassed 95% of the variation and the small black circles represent the species codes position to reduce overlap of the codes. Fish include Bigmouth Buffalo (BGB; *Ictibus cyprinellus*), Black Buffalo (BKB; *Ictiobus niger*), Black Crappie (BLC; *Pomoxis nigromaculatus*), Bluegill (BLG; *Lepomis macrochirus*), Bowfin (BOW; *Amia calva*), Brook Silverside (BRS; *Labidesthes sicculus*), Freshwater Drum (FRD; *Aplodinotus grummiens*), Golden Shiner (GOS; *Notemigonus crysoleucas*), Gizzard Shad (GZS; *Dorosoma cepedianum*), Largemouth Bass (LMB; *Micropterus salmoides*), Orangespotted Sunfish (ORS; *Lepomis humilis*), Redear Sunfish (RSF; *Lepomis microlophus*), Shortnose Gar (SHG; *Lepisosteus platostomus*), Silver Carp (SCP; *Hypophthalmichthys molitrix*), Smallmouth Buffalo (SAB; *Ictiobus bubalus*), Spotted Gar (SPG; *Lepisosteus oculatus*), Taillight Shiner (TLS; *Notropis maculatus*), Threadfin Shad (THS; *Dorosoma petenense*), Warmouth (WAM; *Lepomis gulosus*), White Crappie (WHC; *Pomoxis annularis*).

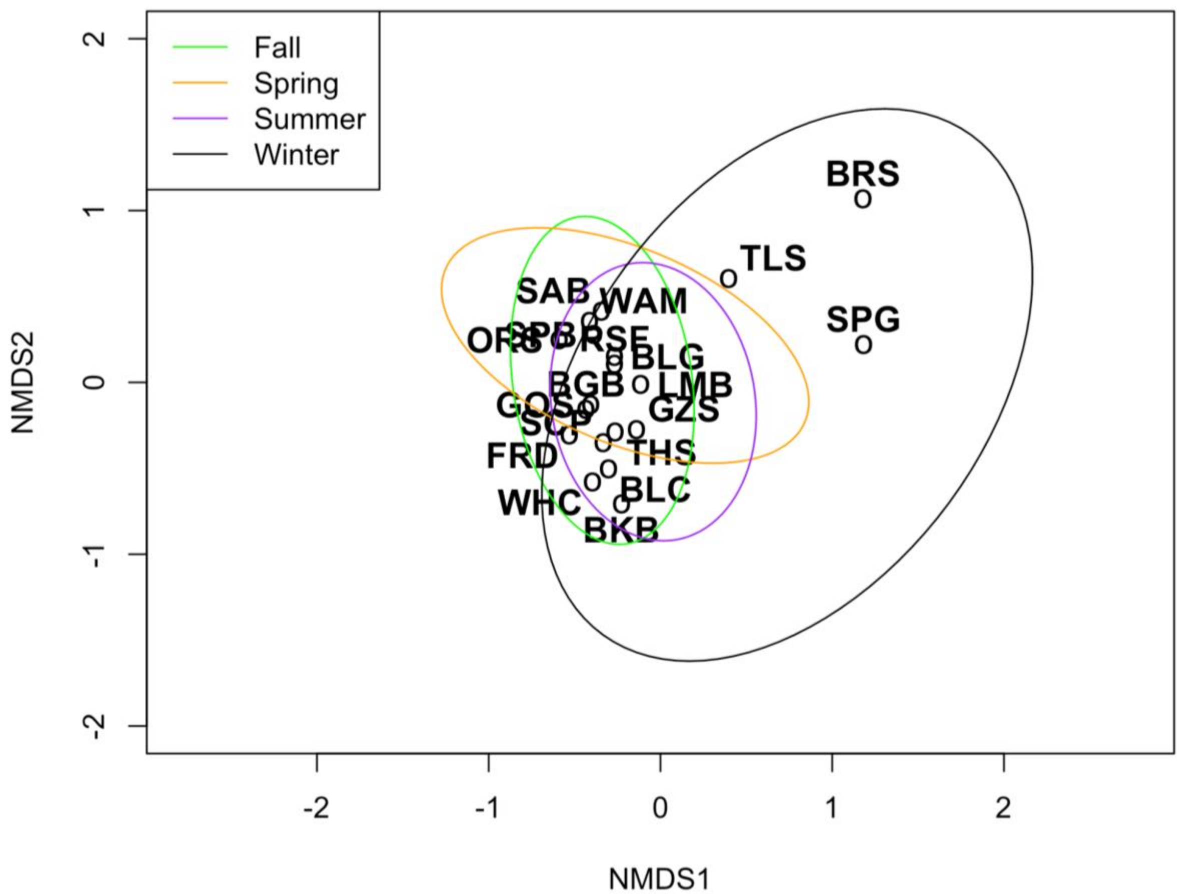

Figure 9. NMDS ordination of fish assemblage structure in the side channel habitat among seasons using SIU electrofishing data collected from Buttonland Swamp within the Cache River watershed in 2020-2021 (Stress: 0.13). Ellipses encompassed 95% of the variation and the small black circles represent the species codes position to reduce overlap of the codes. Fish include Bigmouth Buffalo (BGB; *Ictibus cyprinellus*), Black Buffalo (BKB; *Ictiobus niger*), Black Crappie (BLC; *Pomoxis nigromaculatus*), Bluegill (BLG; *Lepomis macrochirus*), Brook Silverside (BRS; *Labidesthes sicculus*), Freshwater Drum (FRD; *Aplodinotus grummiens*), Golden Shiner (GOS; *Notemigonus crysoleucas*), Gizzard Shad (GZS; *Dorosoma cepedianum*), Largemouth Bass (LMB; *Micropterus salmoides*), Orangespotted Sunfish (ORS; *Lepomis humilis*), Redear Sunfish (RSF; *Lepomis microlophus*), Silver Carp (SCP; *Hypophthalmichthys molitrix*), Smallmouth Buffalo (SAB; *Ictiobus bubalus*), Spotted Bass (SPB; *Micropterus punctulatus*), Spotted Gar (SPG; *Lepisosteus oculatus*), Taillight Shiner (TLS; *Notropis maculatus*), Threadfin Shad (THS; *Dorosoma petenense*), Warmouth (WAM; *Lepomis gulosus*), White Crappie (WHC; *Pomoxis annularis*).

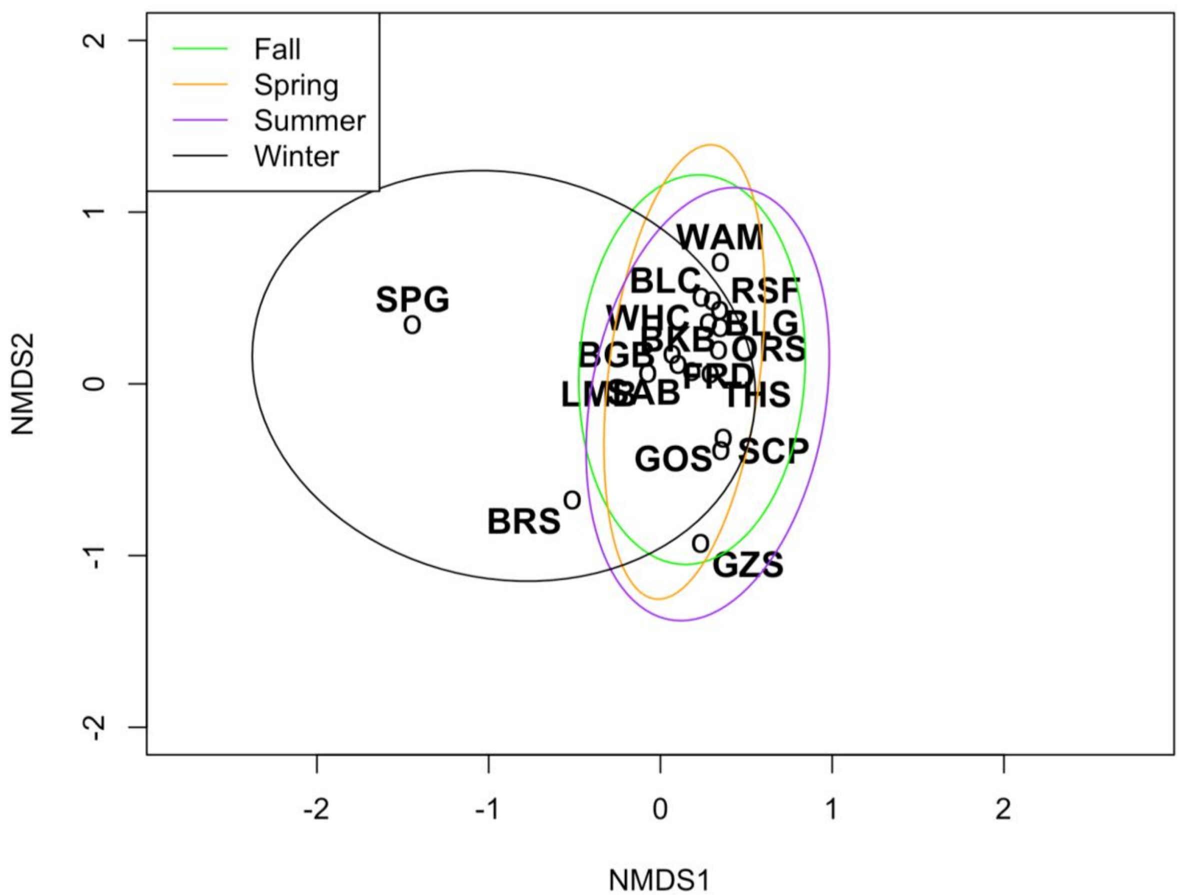

Figure 10. NMDS ordination of fish assemblage structure in the main channel habitat among seasons using SIU electrofishing data collected from Buttonland Swamp within the Cache River watershed in 2020-2021 (Stress: 0.13). Ellipses encompassed 95% of the variation and the small black circles represent the species codes position to reduce overlap of the codes. Fish include Bigmouth Buffalo (BGB; *Ictibus cyprinellus*), Black Buffalo (BKB; *Ictiobus niger*), Black Crappie (BLC; *Pomoxis nigromaculatus*), Bluegill (BLG; *Lepomis macrochirus*), Brook Silverside (BRS; *Labidesthes sicculus*), Freshwater Drum (FRD; *Aplodinotus grummiens*), Golden Shiner (GOS; *Notemigonus crysoleucas*), Gizzard Shad (GZS; *Dorosoma cepedianum*), Largemouth Bass (LMB; *Micropterus salmoides*), Orangespotted Sunfish (ORS; *Lepomis humilis*), Redear Sunfish (RSF; *Lepomis microlophus*), Silver Carp (SCP; *Hypophthalmichthys molitrix*), Smallmouth Buffalo (SAB; *Ictiobus bubalus*), Spotted Gar (SPG; *Lepisosteus oculatus*), Threadfin Shad (THS; *Dorosoma petenense*), Warmouth (WAM; *Lepomis gulosus*), White Crappie (WHC; *Pomoxis annularis*).

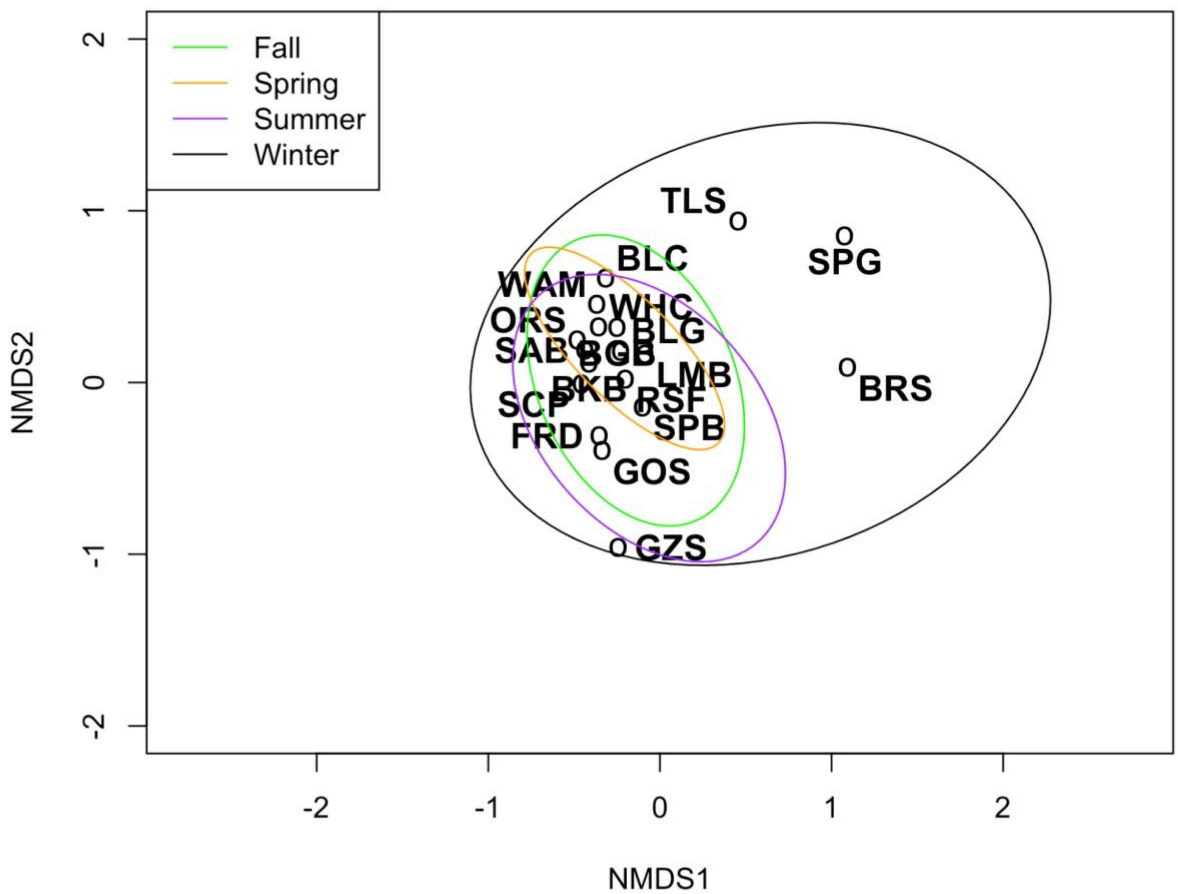

Figure 11. NMDS ordination of fish assemblage structure in offshore vegetated habitat among seasons using SIU electrofishing data collected from Buttonland Swamp within the Cache River watershed in 2020-2021 (Stress: 0.14). Ellipses encompassed 95% of the variation and the small black circles represent the species codes position to reduce overlap of the codes. Fish include Bigmouth Buffalo (BGB; *Ictibus cyprinellus*), Black Buffalo (BKB; *Ictiobus niger*), Black Crappie (BLC; *Pomoxis nigromaculatus*), Bluegill (BLG; *Lepomis macrochirus*), Brook Silverside (BRS; *Labidesthes sicculus*), Freshwater Drum (FRD; *Aplodinotus grummiens*), Golden Shiner (GOS; *Notemigonus crysoleucas*), Gizzard Shad (GZS; *Dorosoma cepedianum*), Largemouth Bass (LMB; *Micropterus salmoides*), Orangespotted Sunfish (ORS; *Lepomis humilis*), Redear Sunfish (RSF; *Lepomis microlophus*), Silver Carp (SCP; *Hypophthalmichthys molitrix*), Smallmouth Buffalo (SAB; *Ictiobus bubalus*), Spotted Bass (SPB; *Micropterus punctulatus*), Spotted Gar (SPG; *Lepisosteus oculatus*), Taillight Shiner (TLS; *Notropis maculatus*), Warmouth (WAM; *Lepomis gulosus*), White Crappie (WHC; *Pomoxis annularis*).

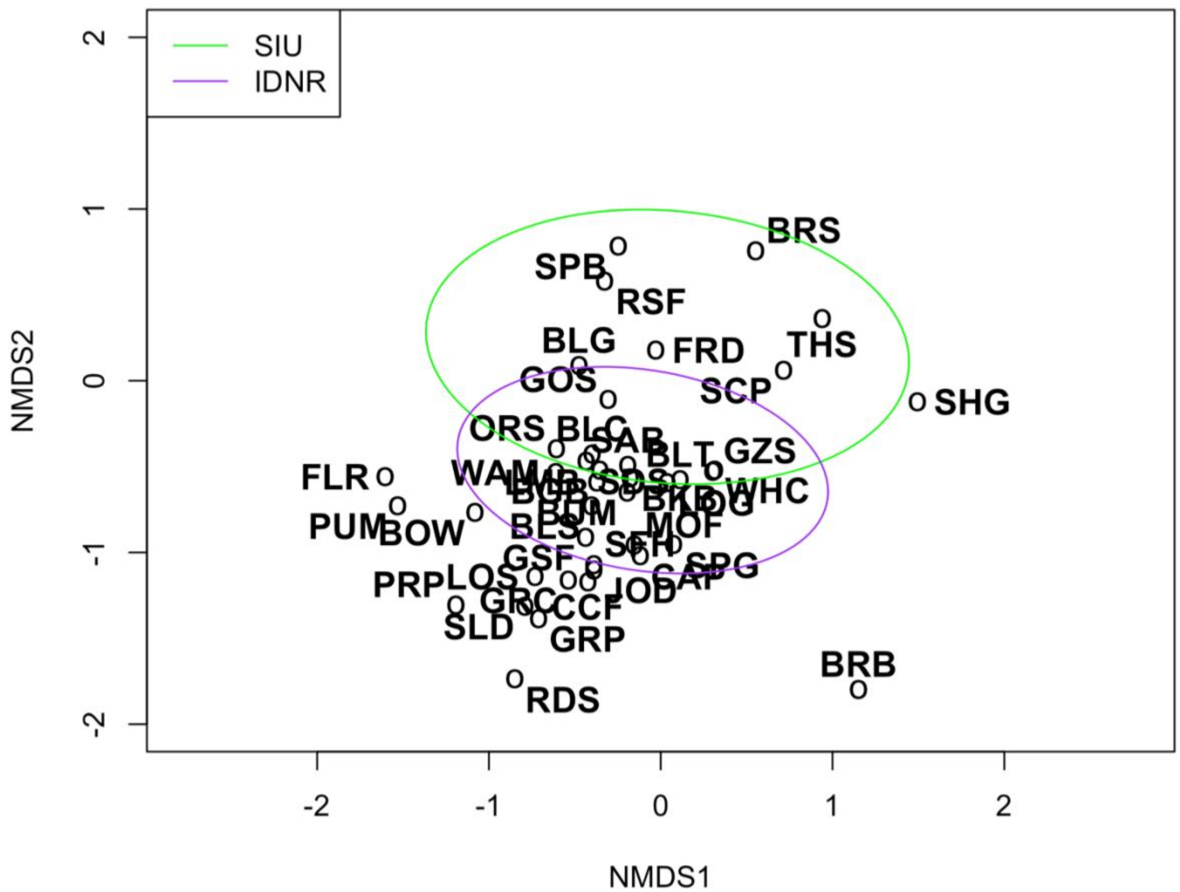

Figure 12. NMDS ordination of fish assemblage structure between IDNR (1992-2020) and SIU (2020-2021) electrofishing data from Buttonland Swamp within the Cache River watershed from June - September (Stress: 0.15). Ellipses encompassed 95% of the variation and the small black circles represent the species codes position to reduce overlap of the codes. Fish include Bigmouth Buffalo (BGB; *Ictibus cyprinellus*), Black Buffalo (BKB; *Ictiobus niger*), Black Crappie (BLC; *Pomoxis nigromaculatus*), Bluegill (BLG; *Lepomis macrochirus*), Bluntnose Minnow (BLS; *Pimephales notatus*), Blackstripe Topminnow (BLT; Fundulus notatus), Bowfin (BOW; *Amia calva*), Brown Bullhead (BRB; *Ameiurus nebulosus*), Brook Silverside (BRS; *Labidesthes sicculus*), Bullhead Minnow (BUM; *Pimephales vigilax*), Common Carp (CAP; *Cyprinus carpio*), Channel Catfish (CCF; *Ictalurus punctatus*), Flier (FLR; *Centrarchus macropterus*), Freshwater Drum (FRD; *Aplodinotus grummiens*), Golden Shiner (GOS; *Notemigonus crysoleucas*),Grass Carp (GRC; *Ctenopharyngodon idella*), Grass Pickerel (GRP; *Esox americanus*), Green Sunfish (GSF; *Lepomis cyanellus*), Sunfish Hybrid (SFH), Gizzard Shad (GZS; *Dorosoma cepedianum*), Johnny Darter (JOD; *Etheostoma nigrum*), Largemouth Bass (LMB; *Micropterus salmoides*), Longnose Gar (LOG; Lepisosteus osseus), Longear Sunfish (LOS; *Lepomis megalotis*), Mosquitofish (MOF; *Gambusia affinis*), Orangespotted Sunfish (ORS; *Lepomis humilis*), Pirate Perch (PRP; *Aphredoderus sayanus*), Pugnose Minnow (PUM; *Opsopoeodus emiliae*), Redfin Shiner (RDS; *Lythrurus umbratilis*), Redear Sunfish (RSF; *Lepomis microlophus*), Smallmouth Buffalo (SAB; *Ictiobus bubalus*), Silver Carp (SCP; *Hypophthalmichthys molitrix*), Spotted Sucker (SDS; *Minytrema melanops*), Shortnose Gar (SHG; *Lepisosteus platostomus*), Slough Darter (SLD; *Etheostoma gracile*), Spotted Bass (SPB;

Micropterus punctulatus), Spotted Gar (SPG; *Lepisosteus oculatus*), Threadfin Shad (THS; *Dorosoma petenense*), Warmouth (WAM; *Lepomis gulosus*), White Crappie (WHC; *Pomoxis annularis*).

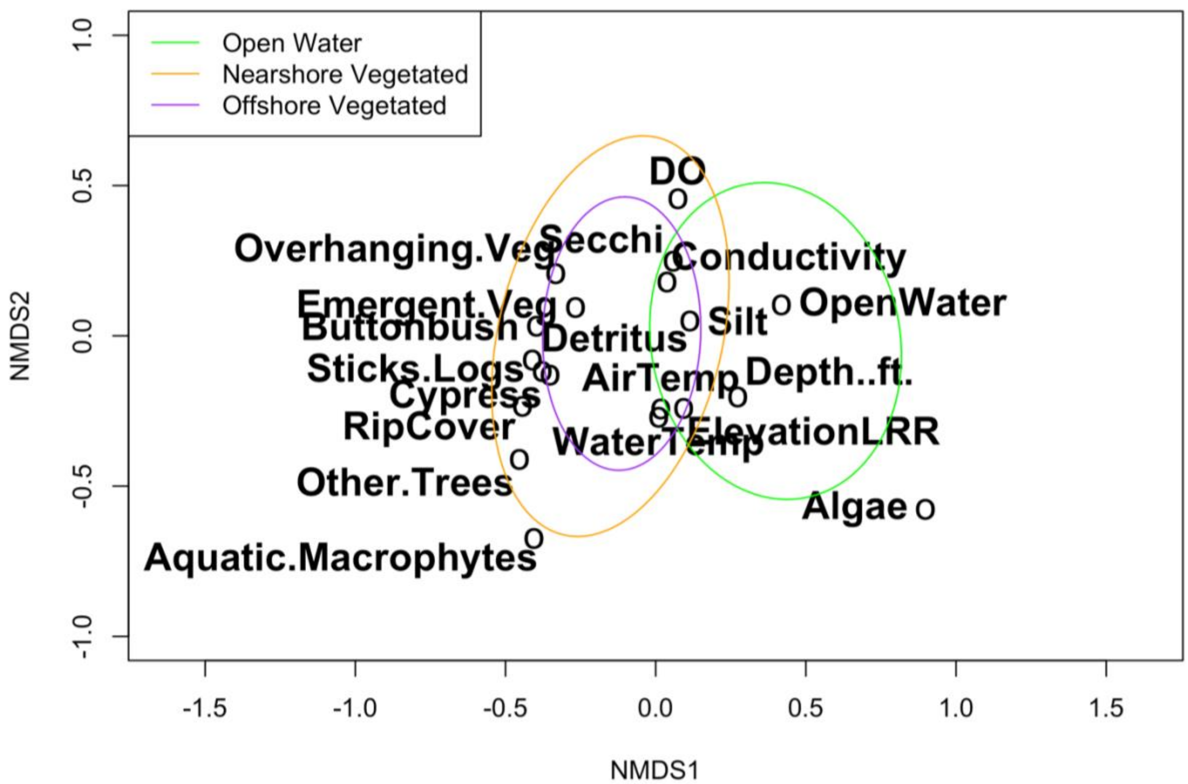

Figure 13. NMDS ordination of environmental variables within microhabitats using SIU electrofishing data collected from Buttonland Swamp within the Cache River watershed in 2020-2021 (Stress: 0.133). Ellipses encompassed 95% of the variation and the small black circles represent the environmental variable codes position to reduce overlap of the codes. DO = dissolved oxygen; Overhanging.Veg = overhanging vegetation; Emergent.Veg = emergent vegetation; RipCover = riparian cover; ElevationLRR = elevation at Long Reach Road.

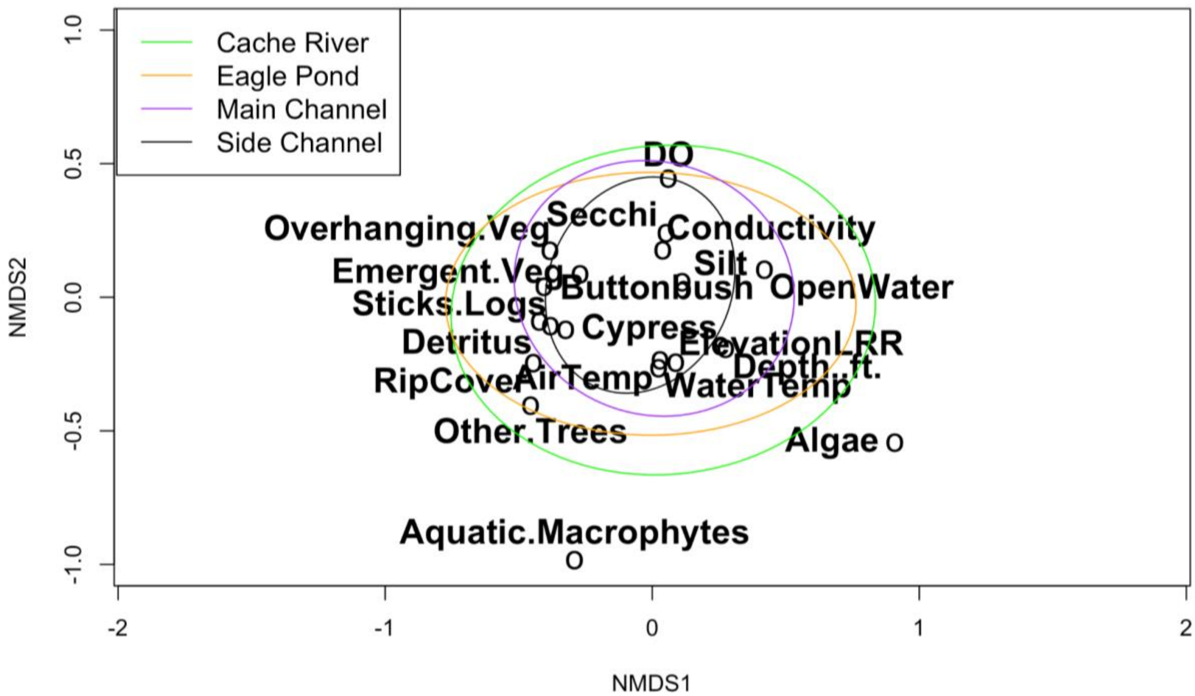

Figure 14. NMDS ordination of environmental variables within macrohabitats using SIU electrofishing data collected from Buttonland Swamp within the Cache River watershed in 2020-2021 (Stress: 0.133). Ellipses encompassed 95% of the variation and the small black circles represent the environmental variable codes position to reduce overlap of the codes. DO = dissolved oxygen; Overhanging.Veg = overhanging vegetation; Emergent.Veg = emergent vegetation; RipCover = riparian cover; ElevationLRR = elevation at Long Reach Road.

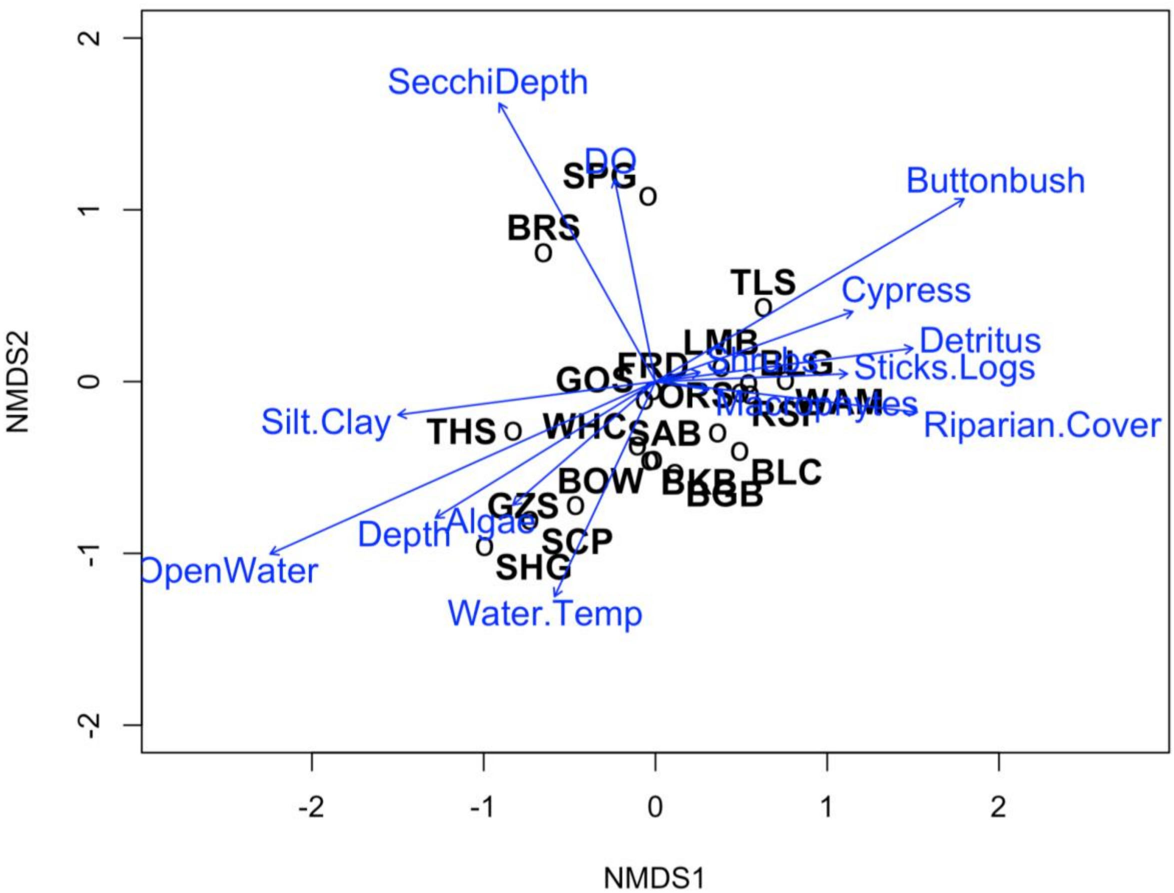

Figure 15. NMDS ordination of species spatial distribution associated with environmental variables using SIU electrofishing data collected from Buttonland Swamp within the Cache River watershed in 2020-2021 (Stress: 0.148). The small black circles represent the species codes position to reduce overlap of the codes. Vector lengths for environmental variables indicate the strength of association. Fish include Bigmouth Buffalo (BGB; *Ictibus cyprinellus*), Black Buffalo (BKB; *Ictiobus niger*), Black Crappie (BLC; *Pomoxis nigromaculatus*), Bluegill (BLG; *Lepomis macrochirus*), Bowfin (BOW; *Amia calva*), Brook Silverside (BRS; *Labidesthes sicculus*), Freshwater Drum (FRD; *Aplodinotus grummiens*), Golden Shiner (GOS; *Notemigonus crysoleucas*), Gizzard Shad (GZS; *Dorosoma cepedianum*), Largemouth Bass (LMB; *Micropterus salmoides*), Orangespotted Sunfish (ORS; *Lepisosteus platostomus*), Redear Sunfish (RSF; *Lepomis microlophus*), Shortnose Gar (SHG; *Lepisosteus platostomus*), Silver Carp (SCP; *Hypophthalmichthys molitrix*), Smallmouth Buffalo (SAB; *Ictiobus bubalus*), Spotted Gar (SPG; *Lepisosteus oculatus*), Taillight Shiner (TLS; *Notropis maculatus*), Threadfin Shad (THS; *Dorosoma petenense*), Warmouth (WAM; *Lepomis gulosus*), White Crappie (WHC; *Pomoxis annularis*).

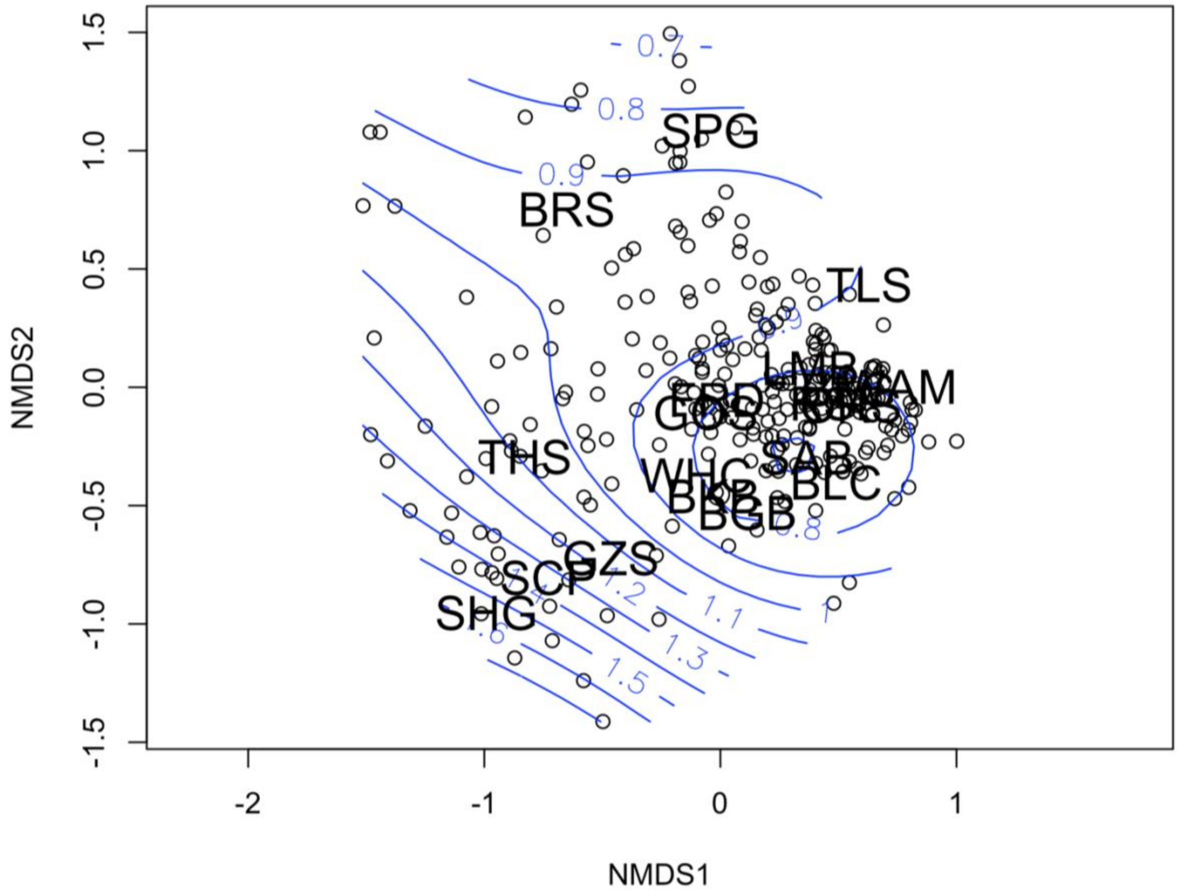

Figure 16. NMDS ordination of species spatial distribution associated with site depth using SIU electrofishing data collected from Buttonland Swamp within the Cache River watershed in 2020-2021 (indicated on contour line; Stress: 0.148). The small black circles represent sites species assemblages, where dots that are closer together have a similar assemblage structure than those dots that are further away. Fish include Bigmouth Buffalo (BGB; *Ictibus cyprinellus*), Black Buffalo (BKB; *Ictiobus niger*), Black Crappie (BLC; *Pomoxis nigromaculatus*), Bluegill (BLG), Bowfin (BOW; *Amia calva*), Brook Silverside (BRS; *Labidesthes sicculus*), Freshwater Drum (FRD; *Aplodinotus grummiens*), Golden Shiner (GOS; *Notemigonus crysoleucas*), Gizzard Shad (GZS; *Dorosoma cepedianum*), Largemouth Bass (LMB; *Micropterus salmoides*), Orangespotted Sunfish (ORS; *Lepisosteus platostomus*), Redear Sunfish (RSF; *Lepomis microlophus*), Shortnose Gar (SHG; *Lepisosteus platostomus*), Silver Carp (SCP; *Hypophthalmichthys molitrix*), Smallmouth Buffalo (SAB; *Ictiobus bubalus*), Spotted Gar (SPG; *Lepisosteus oculatus*), Taillight Shiner (TLS; *Notropis maculatus*), Threadfin Shad (THS; *Dorosoma petenense*), Warmouth (WAM; *Lepomis gulosus*), White Crappie (WHC; *Pomoxis annularis*).

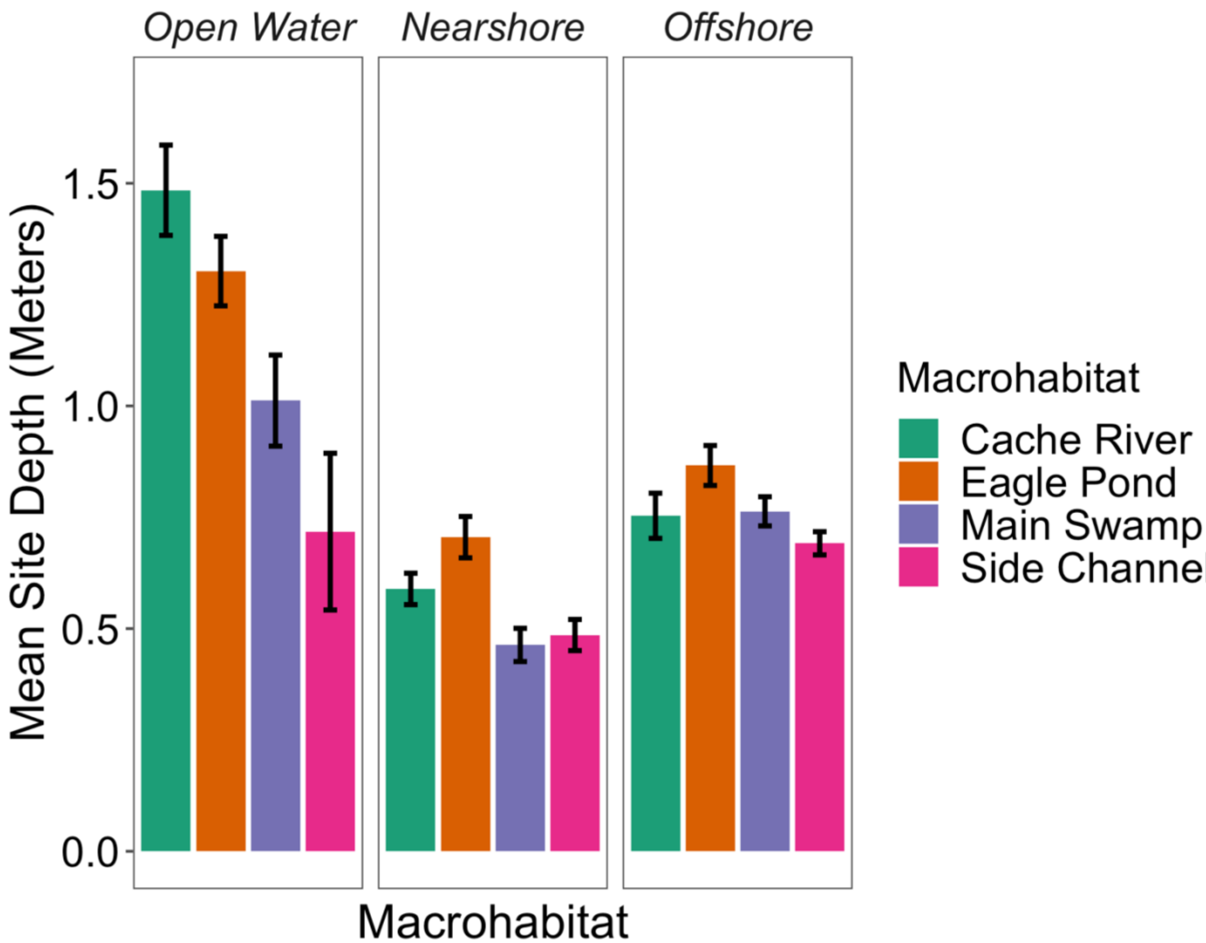

Figure 17. Mean depth of each microhabitat (open water, and vegetated habitats) sampled within each macrohabitat using all gears (electrofishing, fyke, and mini fyke net) using SIU data collected within Buttonland Swamp, in the Cache River watershed 2020-2021. Depth comparisons across macrohabitats occur only within microhabitats.

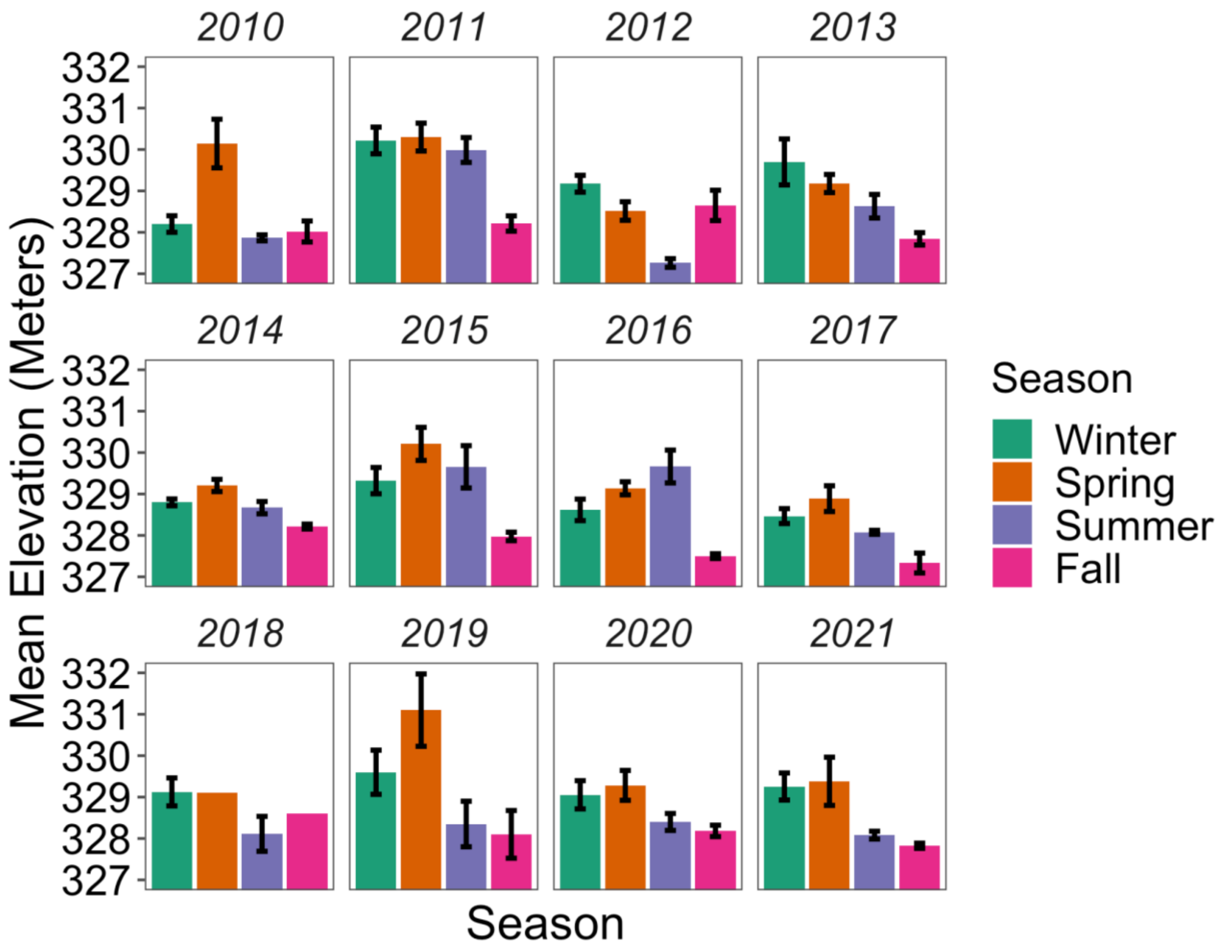

Figure 18. Mean elevation in meters over each season (fall, spring, summer, winter) within each year (2010-2021) using the IDNR historical water data collected within Buttonland Swamp, in the Cache River watershed.

CHAPTER 2

EVALUATING RELATIONSHIPS BETWEEN BUTTONLAND SWAMP HYDROLOGY AND FISH RECRUITMENT

INTRODUCTION

Year-class strength estimates how many fish were recruited to the population each year for a particular species. Knowing how many fish are recruited to a population each year can facilitate understanding of biotic and abiotic factors that are influencing survival of early life stages. There are many biotic and abiotic factors that can influence fish recruitment (Forney 1971, Houde 1987, Prout et al. 1990, Welker et al. 1994, Ludsin and DeVries 1997, Koel and Sparks 2002, Santucci and Wahl 2003). These factors may differ among species with different habitat requirements and life histories and within species among locations. Understanding the factors that influence fish recruitment could inform potential management actions, including water level manipulations, in managed freshwater ecosystems.

Wetland inundation can enhance fish recruitment (Michaletz 1997, Agostinho et al. 2004, Bonvechio and Allen 2005, Phelps et al. 2008, Dembkowski et al. 2016) by providing increased habitat availability for refuge from predators, feeding (Junk et al. 1989, Pusey and Bradshaw 1996, Crook et al. 2020), and spawning (Michaletz 1997), thereby contributing to a stronger year-class (Kahl et al. 2008). Fish recruitment has been positively associated with flow, which is generally linked to higher water levels (Bonvechio and Allen 2005). Water level fluctuations have influenced recruitment and year-class strength of fish species differently in lakes, reservoirs, and rivers, depending on fish life history characteristics and timing of water level and flow fluctuations (Beam 1983, Gaboury and Patalas 1984, Bonvechio and Allen 2005). Bonvechio and Allen (2005) found that in Florida lakes and rivers, impacts of water level on year-class strength of *Micropterus* and *Lepomis* species were stronger in rivers than in lakes. Additionally, *Micropterus* and *Lepomis* species year-class strength can be positively or negatively related to flow rates depending on the time of year (Bonvechio and Allen 2005). In a Tennessee reservoir, Spotted Bass (*Micropterus punctulatus*) year-class strength had no relationship with water level, while *Pomoxis* spp., White Bass (*Morone chrysops*), and Saugeye (*Sander canadensis x vitreus*) year-class strength were positively related to high water levels (Sammons and Bettoli 2000). Beam (1983) found in a Kansas reservoir that White Crappie (*Pomoxis annularis*) year-class strength was not related to an increase in water level but may have been influenced by other environmental variables.

Abiotic factors other than water level can affect fish recruitment and may modify the influence of water level on year-class strength. Air and water temperature, habitat, water quality, and weather during spawning season can co-influence recruitment along with water level (Hassler 1970, Mitzner 1991). In South Dakota reservoirs, Northern Pike (*Esox lucius*) year-class strength was associated with high water levels, high temperatures, flooded vegetation, and calm weather (Hassler 1970). Air and water temperature affect the growth of larval fish by influencing metabolism and consumption, and in turn impact their survival from predation (Michaletz 1997) because fish mortality is inversely related to body size (Wootton 1990). Temperature can also impact prey abundance as well, which can influence fish year-class strength (Michaletz 1997). Warm spring temperatures are known to produce stronger fish year-classes compared to cooler spring temperatures, which could be due to larval fishes growing slower at lower temperatures and being more susceptible to predation (Michaletz 1997, Cowx 2000, Grenouillet et al. 2001, Tomcko and Pierce 2005). Additionally, the magnitude and duration of cold temperatures during the first winter of a fish's life can result in higher mortality risk (Santucci and Wahl 2003), although winter mortality is size dependent and tends to primarily affect smaller fish in a cohort (Oliver et al. 1979, Post and Evans 1989, Miranda and Hubbard 1994, Shoup and Wahl 2008, Shoup and Wahl 2011).

Biotic factors may influence fish recruitment and potentially override the influence of water level on year-class strength. Predation is a crucial factor in regulating recruitment because larval and juvenile fishes face high mortality by piscivorous fish (Santucci and Wahl 2003, Quist et al. 2003). In four Kansas reservoirs, Walleye (*Sander vitreus*) recruitment is controlled by White Crappie predation, which overrides all abiotic variables (spring storage ratios, water level, temperature, water clarity; Quist et al. 2003). Largemouth Bass (*Micropterus salmoides*) are known to prey on larval Bluegill (*Lepomis macrochirus*) and can influence their recruitment and growth (Santucci and Wahl 2003, Tomcko and Pierce 2005). Additionally, fish that are smaller in size (Werner and Gilliam 1984, Post and Prankevicius 1987) or have reduced swimming capability (Rice et a. 1987) may have an increased risk of predation. Competition could also influence year-class strength. Bigheaded Carps (*Hypophthalmichthys molitrix, Hypophthalmichthys nobilis*) have been found to negatively impact planktivorous fish since they compete for resources, and as Bigheaded Carps densities increase, planktivorous fish biomass (Ivan 2020), abundance, and diversity of commercially important fishes have decreased (Sugunan 1997, Petr 2002). Thus, Bigheaded Carps have the potential to negatively impact year-class strength of fishes due to competition for zooplankton prey. Gizzard Shad (*Dorosoma cepedianum;* Stein et al. 1995) and other planktivorous fishes (Partridge and DeVries 1999) can negatively affect Bluegill recruitment by interspecific competition; Yellow Perch (*Perca flavescens*) can cause lower growth rate and survival of Bluegill from interspecific competition

(Kaemingk et al. 2014). Therefore, the abundance of predators and competitors could influence year-class strength of other fish species.

Water level fluctuations may be particularly important in influencing year-class strength of fishes in riverine ecosystems where natural flow and water level regimes have been altered by human activities (Sammons and Bettoli 2000, Hudon et al. 2010). Buttonland Swamp is a managed wetland that is a part of the Lower Cache River in southern Illinois. The Post Creek Cutoff is a ditch that separated the Cache River into the Upper and Lower Cache rivers. When the Post Creek Cutoff was constructed in the early 1900s it caused Buttonland Swamp to dry for extended periods of the year. To restore historical water levels within Buttonland Swamp Diehl Dam (on the west side of the swamp), an in-stream weir (located east of the swamp by Route 37), and Karnak levee (farther to the east) were constructed to inundate the swamp to historical levels (Middleton 2000, Fidler 2014). The hydrology of Buttonland Swamp has been managed by the Illinois Department of Natural Resources (IDNR) for nearly 40 years using Diehl Dam (Brian Metzke, personal communication). These human alterations have shifted the hydrologic dynamics in Buttonland Swamp, where flows can move east towards the Cache River levee or west into the Mississippi River depending on flows from tributaries and water-surface elevations in the river channel in Buttonland Swamp and further downstream (Demissie et al. 1990). Modifications to water flow in the Lower Cache River were put in place to improve drainage and restore historical water levels; however, these alterations have disrupted the natural hydrologic dynamics and flood pulse processes (Gough 2005), which could influence fish assemblage dynamics (Ropke et al. 2015).

Modifications in Buttonland Swamp have altered the direction of water flow, timing, frequency, velocity, and volume (Gough 2005, Demissie et al. 2008, Demissie et al. 2010). Flood pulses can play a critical role in the behavior of biota (Junk et al. 1989) and the timing and duration of these pulses can influence fish movement and behavior (King et al. 2003, Ropke et al. 2015). Fish benefit from inundation, whereas wetland plant communities in swamps depend on high water levels in the winter and low water levels in the spring and summer for growth and regeneration (Dicke and Toliver 1990). There is concern that this prolonged inundation has altered the wetlands' productivity, since species that are dependent on these natural flood pulses can shift further upland or become extirpated from the area (U.S. Fish and Wildlife Service 2016). Although consistent inundation has negatively impacted Bald Cypress recruitment, this may not be problematic for fishes, whereas low water conditions that benefit Bald Cypress recruitment could have negative impacts on the fish community; low water level conditions could cause issues including low dissolved oxygen, greater temperature fluctuations, limited deep water refuge, or loss of habitat area for feeding and spawning, which could influence year-class strength and recruitment of fishes. It is unknown whether water levels under the current management regime affects fish recruitment in the swamp or if recruitment is affected by other biotic and abiotic factors. Therefore, there is a need to assess associations between recruitment and naturally fluctuating water levels in Buttonland Swamp to evaluate whether low water level may be associated with weaker year-classes and whether relationships between water level and fish recruitment may be modified by biotic or other abiotic interactions. The objective of this study was to assess associations between historic water level data, air temperature, and historical predator/competitor catch per unit effort (CPUE) data from Buttonland Swamp with year-class strength indices of Silver Carp, Bluegill, and Gizzard Shad (from our data (SIU data) and from historical IDNR data) to evaluate relationships between hydrology, temperature, and other species interactions with fish recruitment.

METHODS

Study Area

The study area, within Johnson and Pulaski Counties, Illinois, encompassed 176.44 hectares (ha) of the northernmost Cypress-Tupelo wetland in the United States known as Buttonland Swamp. Because of its unique habitat and high species diversity it is an Illinois Land and Water Reserve and a National Natural Landmark (Ramsar Convention, 2009). Buttonland Swamp is a remnant swamp, as most of the original swamp was converted to agricultural lands in the 1940s. Buttonbush covers most of the swamp with Bald Cypress and Water Tupelo trees scattered throughout the swamp and some areas of open water. Rainfall and inflow from tributaries and agricultural ditches influence the water level in Buttonland Swamp (Demissie et al. 1990, Allgire and Cahill 2001). Water level within Buttonland Swamp is controlled by the IDNR using Diehl Dam; the dam is opened during flood stages and closed if water levels are at 100.1 meters above sea level (MASL (historical water level elevation)) or lower to ensure the swamp stays inundated (Christina Feng, IDNR, personal communication).

Historical Water Level and Air Temperature Data

Water level data have been recorded from 2010 to present by the IDNR in Buttonland Swamp. Water level has ranged from 99.36 to 101.80 MASL (meters above sea level; 0 m to 2.44 m in water depth); the monthly average peaks around March and is lowest in September, although each year varies. Historical water level data were used to assess changes in mean water level among years and to determine whether water level was associated with year-class strength of Bluegill, Gizzard Shad, and Silver Carp.

Long term water temperature data have not been recorded in Buttonland Swamp. The IDNR has conducted fish surveys typically at least once annually since 1992 in Buttonland Swamp and has taken water temperature during those surveys; however, one water temperature reading per year is insufficient to assess interannual trends. Daily average air temperature (°C) from the National Oceanic and Atmospheric Administration (NOAA) gauge in Bear Ridge, Illinois was used to evaluate whether air temperature was associated with year-class strength for each of the three fish species. The gauge was about 27 km away from Buttonland Swamp but was the closest location to Buttonland Swamp with available air temperature data for years in which fish were collected.

Fish Sampling

Adult and older juvenile fish were targeted using boat electrofishing in deep, open areas and around the periphery of the swamp (sixteen 15-minute runs per month) and fyke (12 per month) and mini fyke (12 per month) nets in shallow areas that were vegetated and around the swamp margin. Bluegill, Gizzard Shad, and Silver Carp were targeted since they were among the most abundant fish species in Buttonland Swamp. A subset of Bluegill, Gizzard Shad, and Silver Carp were euthanized with MS-222 and used for aging. An annual limit of 100 individuals (in 10 mm length-groups) of each species were euthanized in September, October, and November 2020 and 2021; this sample size was sufficient to develop age-length keys (Ricker 1975). Data from this study are referred to as SIU data and historical data collected by IDNR are referred to as IDNR data.

Using the IDNR historical data (1992-2020) CPUE was calculated for species that were considered predators to larval fish (*Lepisosteus* spp. (Lagler et al. 1942), Largemouth Bass (Pflieger 1997, Santucci and Wahl 2003), Bullheads (Becker 1983, Pflieger 1997), Warmouth (Pflieger 1997), Crappies (Pflieger 1997)) and species that could potentially have interspecific competition with larval and adult Bluegill, Gizzard Shad, and Silver Carp (Common Carp (Moen

1953, Pflieger 1997), *Ictiobus* spp. (Minckley et al. 1970, Starostka and Applegate 1970, Pflieger 1997), and other adult Bluegill (density dependent competition; Latta and Merna 1977), Gizzard Shad (Stein et al. 1995), Silver Carp (Sugunan 1997, Petr 2002, Ivan 2020), Threadfin Shad (Sammons et al. 1998)).

Calcified Structure Processing and Fish Age Estimation

Otoliths from Bluegill, Gizzard Shad, and Silver Carp were used to estimate age of individuals and estimate year-class strength for each species. Otoliths were used since age estimates from otoliths are typically more precise than other aging structures (Hoxmeier et al. 2001, Maceina and Sammons 2006, Oele et al. 2015, Tyszko and Pritt 2017). Sagittal otoliths were removed from Bluegill and Gizzard Shad and dried before they were aged. Lapilli otoliths were removed from Silver Carp since they provide the most precise and reliable ages from Silver Carp (Seibert and Phelps 2013). The post-cleithrum were removed to provide additional estimates of Silver Carp age. After otoliths dried, they were embedded in epoxy (Epo-Fix, Electron Microscopy Sciences, Hatfield, PA) and placed in a desiccator for 24 hours for the epoxy to harden. The otoliths were then sectioned into 1 mm thick sections in the transverse plane with a low-speed IsoMet saw, sanded with 500A grit sandpaper, dried, and polished using lapping film. A Leica MC170 HD microscope was used to photograph otoliths and post-cleithra to be aged. Small post-cleithra, with diameter smaller than 4 mm, were processed similar to otoliths (set in epoxy and sectioned) but were not sanded or polished. Large post-cleithra, with diameter of 4 mm or larger, were not set in epoxy but were sectioned since they were larger than the width of the epoxy mold. Fish ages were estimated by two independent readers, with a third reader to age otoliths and post-cleithra when fish age was disagreed upon, using annuli counts to provide precise age estimates (Maceina 1988, Long and Fisher 2001). If ages from a Silver Carp lapilli otolith and post-cleithrum differed, after age estimates were agreed upon, age would be recorded from the structure where annuli were the clearest. Whole otoliths were aged if otoliths were too small, smaller than 3 mm, to be sectioned (particularly in young Bluegill and Gizzard Shad); most of these fish were age zero.

Data Analysis

A subsample of Bluegill, Gizzard Shad, and Silver Carp were selected from each 10-mm length interval for age estimation and to develop an age-length key for each species (Ricker 1975). Since these species vary in maximum adult length, the larger the maximum length of a species, fewer fish were vouchered per 10-mm length interval since the limit was 100 individuals vouchered per year per species. Vouchered Silver Carp, per interval for both years combined, varied per interval since they have a boom and bust reproduction pattern; averaged six fish per interval that Silver Carp were caught in. Vouchered Bluegill, per 10-mm length interval for both years combined, averaged ten fish per interval. Vouchered Gizzard Shad, per 10-mm length interval for both years combined, averaged seven fish per interval. The developed age-length keys were used to assign ages to unaged fish using the semi-random method described in Isermann and Knight (2005). The age-length keys made from SIU data were applied to the IDNR data for Bluegill, Gizzard Shad, and Silver Carp as well. The residual method (Maceina 1997) was used to calculate year-class strength indices using the collected age structure data (Ogle 2016). Residuals (difference between predicted and observed) from the catch curve regression were used to assess year-class strength indices for associations with water level dynamics. Only age-classes fully recruited to the sampling gear were used in recruitment estimates, where fully recruited year-classes were considered when a young age-class had more fish abundance than an older age-class since numbers should decline over age-classes from natural mortality (Allen and Hightower 2010).

Year-class strength (determined from residuals from the catch curve regression) associations with yearly mean water level, air temperature/winter severity, and predator and competitor abundance were evaluated. Winter severity was calculated by averaging the daily minimum air temperature for each month in December through February since those months had the coldest temperatures. Winter severity was considered the lowest average daily air temperature during the first winter (December - February) each year-class endured. For example, for fish hatched in 2017, winter severity would be calculated by averaging the minimum daily average air temperatures in December 2017 and January and February 2018. Year-class strength associations with air temperature, winter severity, water level, and predators/competitors were also made with IDNR data to see if patterns in SIU data were also seen in IDNR data. Using SIU data, insufficient numbers of all three species were captured by fyke nets or mini fyke nets to allow assessment of year-class strength for fish caught using those gears, so age-length keys were only applied to fish caught using electrofishing gear. Additionally, not enough Gizzard Shad from 2021 sampling, and Silver Carp from 2020 sampling were recruited to the gear using electrofishing since at most three year-classes were fully recruited, and at least four year-classes are needed to calculate year-class strength. Therefore, using SIU data, only Bluegill (from 2020 and 2021 data), Gizzard Shad (from 2020 data), and Silver Carp (from 2021 data) collected using electrofishing gear were used in analyses of associations between biotic and abiotic factors and year-class strength. Additionally, not enough Silver Carp were recruited to the gear using IDNR data, so Silver Carp year-class strength was not estimated from IDNR sampling data. Using IDNR data, Gizzard Shad (from 2019 and 2011 data) and Bluegill (from 2020 and 2011 data) year-class strength were estimated and then used to evaluate associations between year-class and water level, air temperature/winter severity, and competitor/predator CPUE. These associations using IDNR sampling data were then compared with associations detected from SIU sampling data.

Average water level and air temperature/winter severity (daily average temperature/daily minimum temperature) were separated into four seasonal periods (Maceina and Stimpert 1998) for each year based on Gizzard Shad and Bluegill reproductive biology and hydrologic cycles: (1) January-April was considered the pre-spawning/winter period; (2) May-June was considered the spawning/spring period (Gizzard Shad (Willis 1987, Pflieger 1997, Wuellner et al. 2008) and Bluegill (Pflieger 1997) spawning peaks in late May or early June); (3) July-September was considered the post-spawn/summer period; and (4) October-December was considered the fall period. Since Silver Carp reproductive biology differs from Gizzard Shad and Bluegill, only three seasonal periods were defined: (1) January-March was considered the pre-spawning/winter period; (2) April-October was considered the spawning period based on water temperature in Buttonland Swamp (18-30 °C; Verigin et al. 1978) since spawning is temperature dependent (Verigin et al. 1978) (although Silver Carp have been found to spawn in April-August in the Illinois River (DeGrandchamp et al. 2007) and June-August in the Missouri River (Deters et al. 2013)); (3) November-December was considered the post-spawn/fall period. Competitor/predator species yearly average CPUE was calculated from the IDNR sampling data. The IDNR did not conduct sampling in 2018, so when running associations of predator/competitor CPUE and year-class strength of fishes, only CPUE of predator/competitors in years the IDNR collected data were used. Therefore, year-class strength of Bluegill, Gizzard Shad, and Silver Carp in 2018 were also excluded for these associations. All statistical analyses were conducted in R version 4.1.2 (R Development Core Team 2020) with a significance level of 0.05.

Repeated measures, mixed model analyses of variance (Gurevitch and Chester 1986) were used to test for significant effects of water level, air temperature/winter severity, and competitor/predator CPUE on year-class strength of Bluegill, Gizzard Shad, and Silver Carp (Appendix J1, J2, K1, K2, L1, and L2). In these models, water level, air temperature/winter severity, or competitor/predator CPUE were fixed effects and year was a random effect. The assumptions of normality of residuals and homogeneity of variances were met. To account for temporal autocorrelation, an autoregressive-1 (AR1) variance-covariance matrix was used, implemented using the <gls> function from the NLME package (Pinheiro et al. 2007). Models were also run without accounting for temporal autocorrelation. However, even though the AIC was 2 units lower for the models without AR1, the models chosen were the ones accounting for AR1 since AIC scores around 2 units of each other are essentially equivalent in AIC strength (Sakamoto et al. 1986, Burnham and Anderson 1998).

RESULTS

Historical Water Level and Air Temperature Data

The mean annual water level from 2010 to 2021 ranged from 100.04 to 100.54 m, with particularly high water levels in 2011 and 2019 and particularly low water levels in 2012 and 2017 (Figure 19). The mean yearly air temperature from 2010 to 2021 has ranged (using the average daily temperature) from 13.07 ℃ (in 2014; $\pm$ 0.55 SE) to 16.13 ℃ (in 2012; $\pm$ 0.48 SE) (Figure 20). Pooling years together from 2010 to 2021 for each month, the highest overall average monthly air temperature (using the average daily temperature) was 25.83 ℃ ($\pm$ 0.13 SE) in July and lowest average monthly air temperature was 1.47 ℃ ($\pm$ 0.31 SE) in January. The maximum daily average air temperature was in June 2012 (32.22 ℃) and the minimum daily average temperature was in January 2014 (-16.67 ℃). However, 2014 had the lowest average yearly air temperature (13.07 ℃ $\pm$ 0.55 SE) and 2012 had the highest average yearly air temperature (16.13 ℃ $\pm$ 0.48 SE). The average yearly winter severity (using the mean daily minimum temperature during December through February) was lowest in 2010 (-3.83 ℃ $\pm$ 0.56 SE) and highest in 2012 (1.35 ℃ $\pm$ 0.56 SE), while the lowest daily minimum temperature was in January 2014 (-18.89 ℃) and the highest minimum temperature was in 2011 (27.22 ℃)

Year-Class Strength

Using SIU data, Bluegill were more abundant than Gizzard Shad and Silver Carp and the length-frequency distribution showed at least two distinct year-classes for Bluegill and Gizzard Shad, while Silver Carp year-classes showed at least three distinct year-classes (Figure 21). Bluegill had a mean length of 100 mm, Gizzard Shad had a mean length of 195 mm, and Silver Carp 519 mm. Bluegill age ranged from zero to four years, with a total of 1764 fish caught (Figure 22). Gizzard Shad age ranged from zero to three years, with a total of 290 fish caught (Figure 22). Silver Carp age ranged from one to five years; with a total of 126 fish caught (Figure 22). More Bluegill, Gizzard Shad, and Silver Carp were caught in 2020 than 2021 (Figure 22). Silver Carp mean length at age increased quicker than Bluegill and Gizzard Shad (Appendix I).

Using SIU data, Silver Carp (from 2021 data) had strong negative residuals showing a poor year-class in 2016, followed by strong positive residuals showing a relatively strong year-class in 2017, and near-average year-classes in 2018 and 2019 (Figure 23). Bluegill (from 2020 and 2021 data) and Gizzard Shad (from 2020 data) showed similar trends to each other; both species had a poor year-class in 2017 and 2020, a relatively strong year-class in 2018, and an average year-class in 2019 (Figure 24).

Water Level

Bluegill, Gizzard Shad, and Silver Carp year-class strength all had an association with some water level seasonal period. Yearly average water level did not have a significant association with Bluegill year-class strength using SIU 2020 ($X^2 = 1.410$, P=0.235) and 2021 ($X^2 = 1.825$, P=0.177) data (Figure 25). Average fall water level had a significant positive association with Bluegill year-class strength using SIU 2020 ($X^2 = 83.639$, $P < 0.001$) and 2021 ($X^2 = 124.860$, $P < 0.001$) data (Figure 26a) and IDNR 2020 data ($X^2 = 24.272$, $P < 0.001$; Figure 26b)

Yearly average water level did not have a significant association with Gizzard Shad year-class strength using SIU data ($X^2 = 1.751$, P = 0.186; Figure 25). Average fall water level had a significant positive association with Gizzard Shad year-class strength using SIU data ($X^2 = 23.718$, $P < 0.001$; Figure 26a). Average spawn water level had a significant positive association with Gizzard Shad year-class strength using IDNR 2011 data ($X^2 = 7.881$, P = 0.005; Figure 27).

Average spawn water level had a significant negative association with Silver Carp year-class strength using SIU data ($X^2 = 47.809$, $P < 0.001$).

Air Temperature/Winter Severity

Bluegill, Gizzard Shad, and Silver Carp year-class strength were all negatively associated with some air temperature seasonal period, whereas Bluegill and Gizzard Shad were positively associated with the average minimum winter air temperature (winter severity). Yearly average air temperature had a significant negative association with Bluegill year-class strength from SIU 2021 data ($X^2 = 34.072$, $P < 0.001$; Figure 28a) and IDNR 2011 ($X^2 = 5.093$, P = 0.024) and 2020

($X^2 = 72.365$, $P < 0.001$) data (Figure 28b). Average pre-spawn air temperature had a significant negative association with Bluegill year-class strength from SIU 2020 ($X^2 = 29.965$, $P < 0.001$) and 2021 ($X^2 = 20.965$, $P < 0.001$) data (Figure 29a) and IDNR 2020 data ($X^2 = 47.928$, $P < 0.001$; Figure 29b). Average minimum winter temperature (winter severity) had a significant positive association with Bluegill year-class strength from SIU 2021 data ($X^2 = 4.145$, $P = 0.042$; Figure 30).

Yearly average air temperature had a nearly significant negative association with Gizzard Shad year-class strength from SIU data ($X^2 = 3.753$, $P = 0.053$; Figure 28a). Average pre-spawn air temperature had a significant negative association with Gizzard Shad year-class strength from SIU data ($X^2 = 9.381$, $P = 0.002$; Figure 29a). Average spawn air temperature had a significant negative association with Gizzard Shad year-class strength from IDNR 2019 data ($X^2 = 7.363$, $P = 0.007$). Average fall air temperature had a significant negative association with Gizzard Shad year-class strength from SIU data ($X^2 = 10.959$, $P < 0.001$). Average minimum winter temperature (winter severity) had a significant positive association with Gizzard Shad year-class strength from SIU data ($X^2 = 16.899$, $P < 0.001$; Figure 30).

Average pre-spawn air temperature had a significant negative association with Silver Carp year-class strength from SIU data ($X^2 = 6.334$, $P = 0.012$; Figure 29a). Average spawn air temperature had a significant negative association with Silver Carp year-class strength from SIU data ($X^2 = 14.410$, $P < 0.001$). Average post-spawn air temperature had a significant negative association with Silver Carp year-class strength from SIU data ($X^2 = 5.531$, $P = 0.019$).

Predation/Interaction with Other Fishes

Bluegill, Gizzard Shad, and Silver Carp all had positive and/or negative associations with the CPUE of predator or competitor species. Largemouth Bass CPUE had a significant negative association with Bluegill year-class strength from SIU 2020 ($X^2 = 61.440$, $P < 0.001$) and 2021 ($X^2 = 28.904$, $P < 0.001$) data (Figure 31a) and IDNR 2020 data ($X^2 = 1542.600$, $P < 0.001$; Figure 31b). Common Carp CPUE had a significant negative association with Bluegill year-class strength from SIU 2020 ($X^2 = 3549.700$, $P < 0.001$) and 2021 ($X^2 = 182.980$, $P < 0.001$) data (Figure 32a) and IDNR 2020 data ($X^2 = 53.748$, $P < 0.001$; Figure 32b). Bluegill CPUE had a significant positive association with Bluegill year-class strength from SIU 2020 ($X^2 = 7.021$, $P = 0.008$) and 2021 ($X^2 = 5.072$, $P = 0.024$) data and IDNR 202 data ($X^2 = 22.332$, $P < 0.001$). Warmouth CPUE had a significant positive association with Bluegill year-class strength from SIU 2020 ($X^2 = 4.19$, $P = 0.041$) and 2021 data ($X^2 = 5.677$, $P = 0.017$). White Crappie CPUE had a significant negative association with Bluegill year-class strength from SIU 2020 ($X^2 = 14.205$, $P < 0.001$) and 2021 ($X^2 = 9.332$, $P = 0.002$) data and IDNR 2020 data ($X^2 = 86.679$, $P < 0.001$). Black Crappie CPUE had a significant positive association with Bluegill year-class strength from IDNR 2020 data ($X^2 = 86.679$, $P < 0.001$). Spotted Gar CPUE had a significant negative association with Bluegill from IDNR 2020 data ($X^2 = 4.501$, $P = 0.034$). Silver Carp CPUE had a significant negative association with Bluegill year-class strength from SIU 2020 ($X^2 = 12.84$, $P < 0.001$) and 2021 ($X^2 = 8.570$, $P = 0.003$) data and IDNR 2020 data ($X^2 = 69.007$, $P < 0.001$).

Largemouth Bass CPUE had a significant negative association with Gizzard Shad year-class strength from IDNR 2019 data ($X^2 = 322.740$, $P < 0.001$; Figure 31b). Warmouth CPUE had a significant positive association with Gizzard Shad year-class strength from SIU 2020 ($X^2 = 217.38$, $P < 0.001$). Black Crappie CPUE had a significant positive association with Gizzard Shad year-class strength from IDNR 2019 data ($X^2 = 35.749$, $P < 0.001$). Bigmouth Buffalo CPUE had a significant negative association with Gizzard Shad from SIU data ($X^2 = 5.592$, $P =$

0.018). Gizzard Shad CPUE had a significant positive association with Gizzard Shad year-class strength from IDNR 2011 data (X^2 = 981.64, P < 0.001).

White Crappie CPUE had a significant positive association with Silver Carp year-class strength from SIU data (X^2 = 46.599, P < 0.001). Redear Sunfish CPUE had a significant positive association with Silver Carp year-class strength from SIU data (X^2 = 75.67, P < 0.001). Gizzard Shad CPUE had a significant positive association with Silver Carp year-class strength from SIU data (X^2 = 14.93, P < 0.001).

DISCUSSION

This study found that water level, along with other abiotic (air temperature) and biotic factors (predator/competitor CPUE), were associated with Bluegill, Gizzard Shad, and Silver Carp year-class strength. These results are similar to previous research that showed year-class strength can be influenced by multiple biotic and biotic factors (Hassler 1970, Santucci and Wahl 2003, Quist et al. 2003, Bonvechio and Allen 2005, Kaemingk et al. 2014).

Water Level

Silver Carp year-class strength was negatively associated with water level, while Bluegill and Gizzard Shad year-class strength were positively associated with water level. Silver Carp often have erratic recruitment since they typically have boom bust year-classes tied to flood years, which has been seen in the Missouri River tributaries (Hayer et al. 2014), Mississippi River (Garvey et al. 2006, Sullivan 2018), and Illinois River confluence (Garvey et al. 2006). Previous studies in large rivers, where spawning occurs, have found Silver Carp year-class strength positively associated with water levels (Garvey et al. 2006, Hayer et al. 2014, Sullivan 2018). However, this study found the opposite, that as water level increased during the spawning/spring water level period, Silver Carp year-class strength decreased. This may be a spurious association since Silver Carp initially entered Buttonland Swamp from either the Ohio or Mississippi rivers, so their recruitment would not be related to water levels within the swamp, but rather the water levels in the Ohio or Mississippi rivers. Additionally, other environmental variables could have been contributing to Silver Carp year-class strength; strong sustained discharge and temperatures above 17 ℃ have also lead to strong Silver Carp year-classes (Garvey et al. 2006, DeGrandchamp et al. 2007, Lohmeyer and Garvey 2009, Hayer et al. 2014, Stuck et al. 2015, Sullivan 2018).

Water levels were significantly positively associated with year-class strength of Bluegill and Gizzard Shad during the fall period (using SIU and IDNR data) and Gizzard Shad during the spawning/spring water level period (using IDNR data). Other studies have found high spring water levels associated with strong cohorts of Bluegill and Gizzard Shad (Michaletz 1997, Raibley et al. 1997, Dattilo et al. 2021). Other studies have also shown young-of-year cohort abundance declines during low water level (Hakala and Hartman 2004, Riley et al. 2009). A decline in water level can decrease habitat and food accessibility (Elliot et al. 1997) and can increase fish densities, which can increase competition and predation (Smith et al. 2005), whereas high water levels can increase habitat and food accessibility which can reduce competition and predation (Resh et al. 1988). Additionally, water level fluctuations can indirectly effect fishes by influencing the depth distribution of macrophytes (Rowe et al. 2003). As water levels increase during and shortly after spawning this gives larval and juvenile fish more access to nursery habitat to take refuge in (Nunn et al. 2012) and provides access to more aquatic and terrestrial prey (Junk et al. 1989, Lyon et al. 2010, Crook et al. 2020).

Air Temperature/Winter Severity

Along with water level, there are other abiotic factors, like air temperature, that can influence year-class strength of fishes. Yearly average, pre-spawn, spawn, post-spawn, and fall air temperature had significant negative associations with year-class strength of Bluegill, Gizzard Shad, and/or Silver Carp from SIU and/or IDNR data. However, other studies have found positive associations between air temperature and Bluegill (Tomcko and Pierce 2005), Gizzard Shad (Michaletz 1997), and Silver Carp (Chapman and George 2011, George and Chapman 2013) year-class strength. Additionally, there are other studies that found positive associations with water temperature and year-class strength (Cowx 2000, Grenouillet et al. 2001). Negative associations between year-class strength and temperature may be spurious and potentially masking other environmental factors that may be influencing year-class strength, such as food availability, competition (Prout et al. 1990, Welker et al. 1994, Ludsin and DeVries 1997), hatch date (Santucci and Wahl 2003), disease and predation (Forney 1971, Houde 1987), physical habitat, weather (Kramer and Smith 1962, Svardson and Molin 1973, Summerfelt 1975, Aggus 1979, Sammons et al. 1998), dissolved oxygen (Moore 1942), or turbidity (Campbell and Branson 1978, Koel and Sparks 2002).

A previous study found that Gizzard Shad in Missouri reservoirs have increased survival with increasing winter water temperatures (Michaletz 2010). We found similar results that average minimum winter air temperature is associated with Gizzard Shad year-class strength; as minimum winter air temperatures increased so did Gizzard Shad year-class strength. Gizzard Shad are intolerant to cold water temperatures, their mortality typically increases when the water temperature drops below 4 to 8 °C (Fetzer et al. 2011). In this study, using the NOAA air temperature gauge, the minimum winter air temperature dropped lower than 4 °C, indicating that the winter average water temperature in Buttonland Swamp (that was typically a few degrees higher than air temperature, based on SIU data) was around or below 4 °C. Therefore, it makes sense that Gizzard Shad year-class strength was positively associated with the minimum winter air temperature since the water temperature dropped low enough for winter kills to occur. Bluegill year-class strength was also found to be positively associated with the average minimum winter air temperature. In previous studies other biotic factors, like predation, hatch date (Santucci and Wahl 2003), fish size, and food accessibility (Shoup and Wahl 2011), have been influencing Bluegill survival after their first winter. Shoup and Wahl (2011) took Bluegill from an Illinois Lake and in lab subjected them to simulated harsh cold/cold (4 or 9 ± 1°C) winter water treatments and food/no food treatments and found that juvenile Bluegill that had no food in the warmer treatment had more cumulative mortality compared to the harsh cold treatment with no food, suggesting food accessibility may be influencing first winter mortality more than temperature.

Temperature, along with directly influencing survival, can also influence fish growth (Michaletz 1997). White et al. (2020) found that Illinois lakes that were heated by electrical power plants had Bluegill with faster first-year growth, shorter lifespans, and greater rates of maturation at small body sizes , only living 3-5 years. The lifespan of Silver Carp in Buttonland Swamp reached 5 years old, which was similar to the Illinois River where Silver Carp can reach 6 to 8 years old, whereas Silver Carp in the Mississippi River and Wabash River can reach ages that are older (up to 13 years old; Seibert et al. 2015, Stuck et al. 2015). Bluegill (lived to 4 years old) and Gizzard Shad (lived to 3 years old) in Buttonland Swamp were also found with shorter lifespans, since Bluegills in Illinois have been seen to reach 7 years old (Hoxmeier et al. 2001) and Gizzard Shad in Alabama reservoirs have been seen to reach 14 years old (DiCenzo et al. 1996). Buttonland Swamp's maximum water temperature was 31.7 °C, which was higher than the baseline lakes used in White et al. (2020), and more representative of temperatures of lakes that were heated by electrical power plants, which could explain the shorter lifespans of Bluegill, Gizzard Shad, and Silver Carp in Buttonland Swamp compared to other systems. The water temperature in Buttonland Swamp likely rose higher than 31.7 °C since water temperature was not recorded every day or at the hottest point in the day. Additionally, DiCenzo et al. (1996) found that in oligo-mesotrophic reservoirs Gizzard Shad were less abundant and lived longer (14 years old) compared to those in eutrophic reservoirs (7 years old). Since Buttonland Swamp is a eutrophic system, trophic state could also be influencing the shorter lifespans of Bluegill, Gizzard Shad, and Silver Carp.

Predator/Competitor Interactions

Along with abiotic interactions, biotic interactions can also influence fish year-class strength. In this study, all predator and competitor species that had significant associations had negative associations with Bluegill, Gizzard Shad, or Silver Carp year-class strength. Largemouth Bass are known to prey on Bluegill (Swingle 1950, Guy and Willis 1990, Otis et al. 1998, Tomcko and Pierce 2005) and Gizzard Shad (Irwin et al. 2003). Largemouth Bass predation is a large source of mortality for young Bluegill (Santucci and Wahl 2003) and Gizzard Shad (Irwin et al. 2003), which can decrease their abundance (Irwin et al. 2003) and influence size-structured interactions between fishes (Shoup and Wahl 2008). This study found similar results, where Largemouth Bass CPUE had significant negative associations with Bluegill and Gizzard Shad year-class strength from IDNR data. These same patterns were also seen using Bluegill from SIU data, showing that these patterns were not isolated incidences. White Crappie and Spotted Gar CPUE were also found to be negatively associated with Bluegill year-class strength, which could be due to the predation of White Crappie and Spotted Gar on Bluegill (Ellison 1984, Ostrand et al. 2004).

Indirect competition can also influence larval recruitment. Silver Carp have been known to affect native planktivores, for example, impacting body condition of Gizzard Shad, by outcompeting them (Sampson et al. 2009). However, in this study there were no associations with Gizzard Shad year-class strength and Silver Carp CPUE. Although Silver Carp CPUE had significant negative associations with Bluegill year-class strength using SIU and IDNR data, showing that these patterns were not isolated incidences. Freedman et al. (2012) found Silver Carp have similar stable isotope signatures to Bluegill, suggesting high diet overlap and thus competition for resources, which could explain why as Silver Carp CPUE increased, Bluegill year-class strength decreased. Bigmouth Buffalo CPUE had significant negative associations with Gizzard Shad year-class strength (using SIU data), which are both planktivores and have high diet overlap and thus compete for resources (Sampson et al. 2008). Additionally, Common Carp CPUE had significant negative associations with Bluegill year-class strength from SIU and IDNR data, showing that these patterns were not isolated incidences. Previous studies have found Common Carp reduces growth and survival of juvenile Bluegill (Wolfe et al. 2009), and negatively influences Bluegill abundance (Egertson and Downing 2004, Jackson et al. 2010), possibly from outcompeting Bluegill. Common Carp also eat other fishes' eggs (Miller and Beckman 1996), and could limit the foraging success of larval Bluegill, that are visual feeding fish, caused by increased turbidity from Common Carp (Miner and Stein 1993), which could in turn decrease year-class strength of Bluegill. Additionally, Common Carp may be indirectly influencing Bluegill year-class strength by uprooting and reducing macrophytes that Bluegill are associated with and rely on, especially at the larval stage, for refuge and spawning habitat (Paukert et al. 2002).

Bluegill CPUE had positive significant associations with Bluegill year-class strength. As Bluegill CPUE increases, this is representing more Bluegill being recruited in the population and is reflected by this positive association. This same pattern occurred for Gizzard Shad; Gizzard Shad CPUE had positive significant associations with Gizzard Shad year-class strength. Correlations between CPUE and year-class strength within a species may be expected since year-class strength is itself a measure of relative abundance. Warmouth CPUE had positive significant associations with Bluegill year-class strength. Warmouth and Bluegill have similar reproductive habits (both nest builders and spawn during the same months; Pflieger 1997) whereby if Bluegill recruitment is doing well there must be sufficient nest building habitat for Warmouth as well.

Along with water level, there are other abiotic and biotic variables that influence year-class strength of fishes. These water level, air temperature, and predator/competitor CPUE association analyses with year-class strength of Bluegill, Gizzard Shad, and Silver Carp gives a preliminary look at these associations, however more data are needed to determine whether these associations persist across periods of several years. The limitations of having only a few year classes present in each year's samples and only having a few years of data are that these patterns shown could be by chance and the associations are weak because of this lack of data. These associations could be interacting with other biotic and abiotic factors, so more data over many years are needed to examine possible interactions.

Water Management Implications

High water level throughout the year, especially from May-June (spawning/spring period) and October-December (fall period), is associated with strong year-class strength of Bluegill and Gizzard Shad. Water level management should focus on maintaining high water levels around these months to achieve strong recruitment of Bluegill and Gizzard Shad, that many sport fish rely on for food. Additionally, high water level in the winter could allow for deep water refuges for cold intolerant fishes, like Gizzard Shad and Threadfin Shad, to potentially experience lower winter mortality and maintain their abundance. Lastly, high water level throughout the year, especially during the spawning and fall period, has the potential to reduce predation pressure (from Largemouth Bass) on larval fish by providing larval fish more access to habitat and food (Resh et al. 1988).

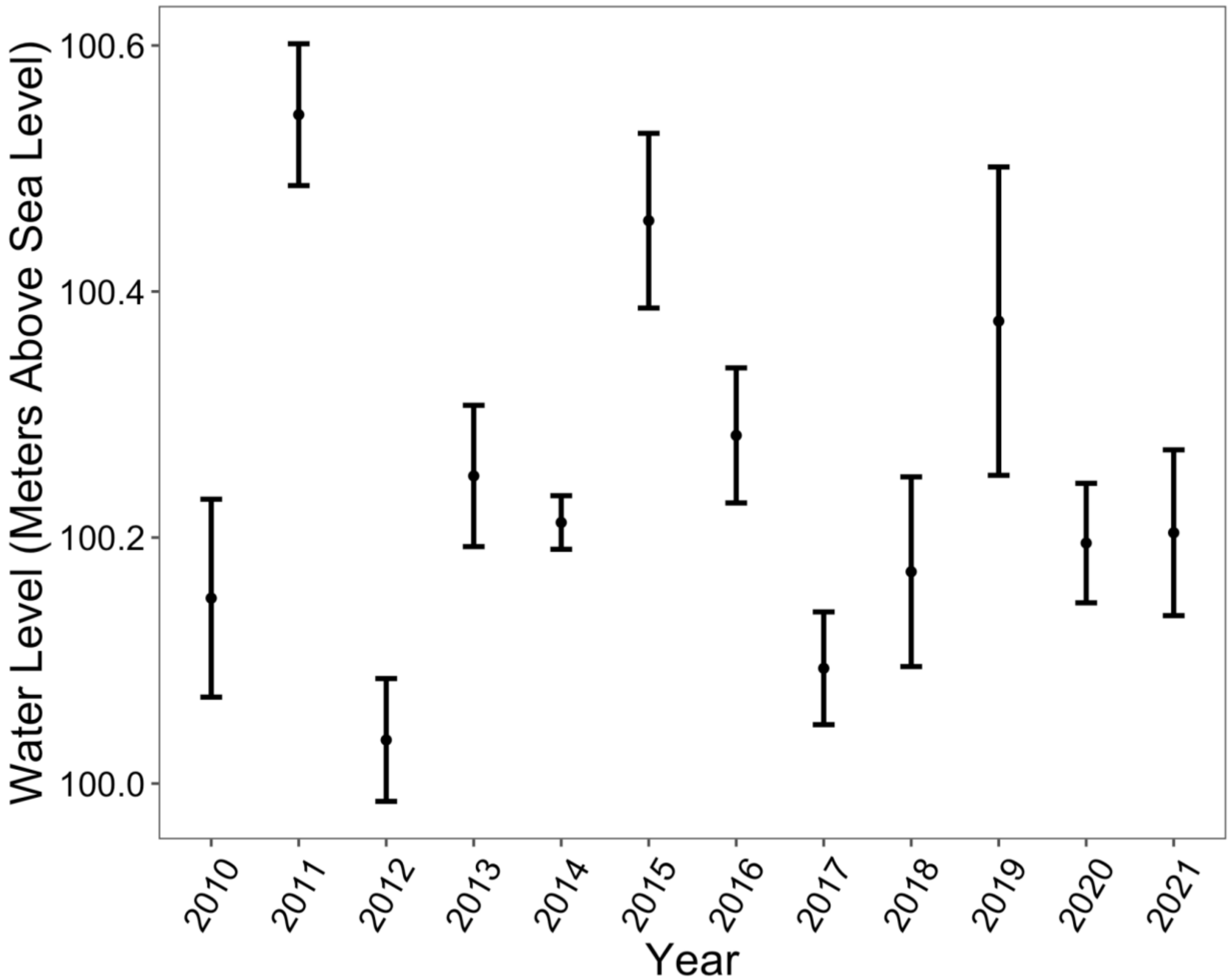

Figure 19. Average yearly water level (meters above sea level) in Buttonland Swamp in the Cache River wetlands, Illinois (mean $\pm$ standard deviation).

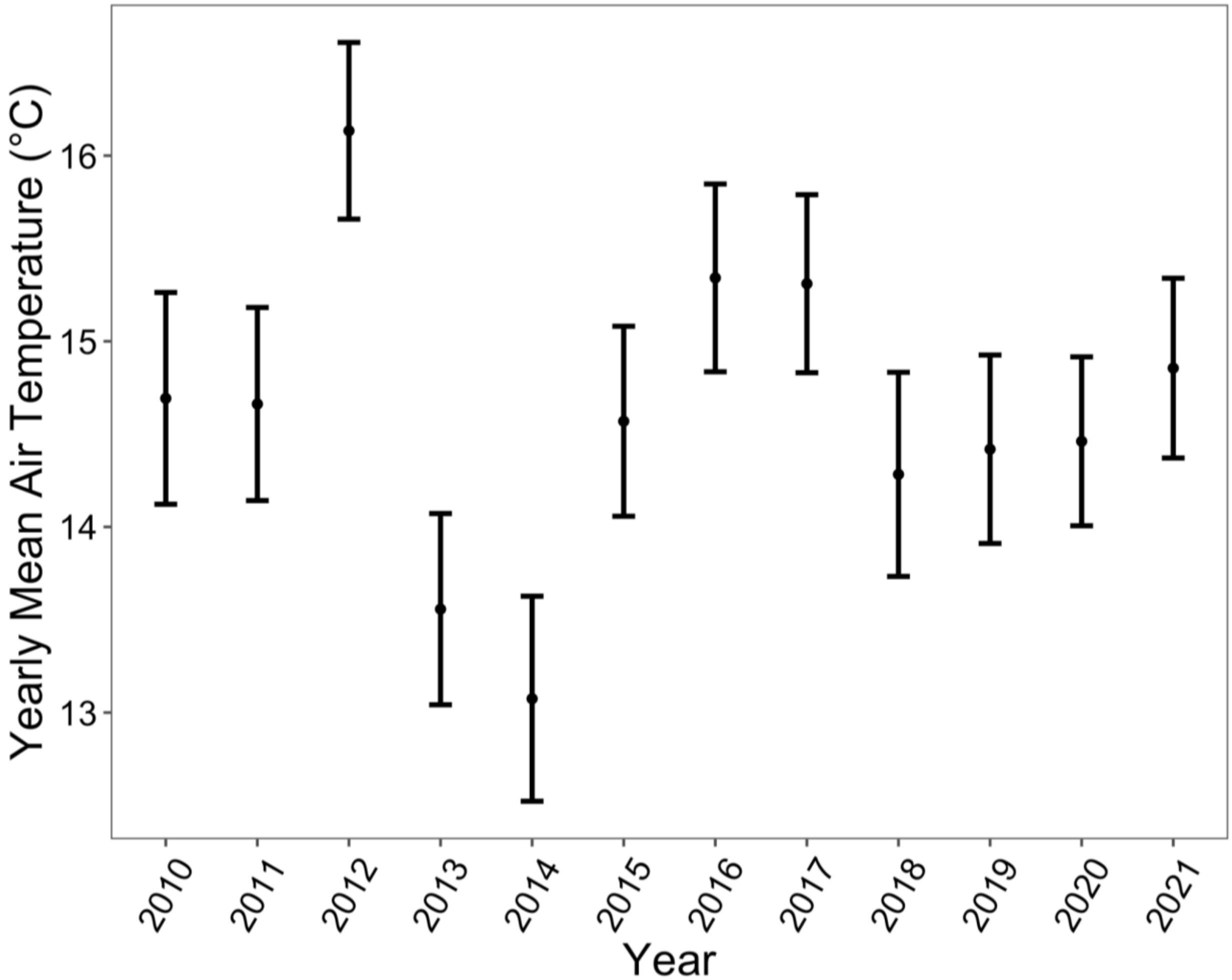

Figure 20. Average yearly air temperature (℃) taken from the NOAA gauge in Bear Ridge, Illinois (mean ± standard deviation).

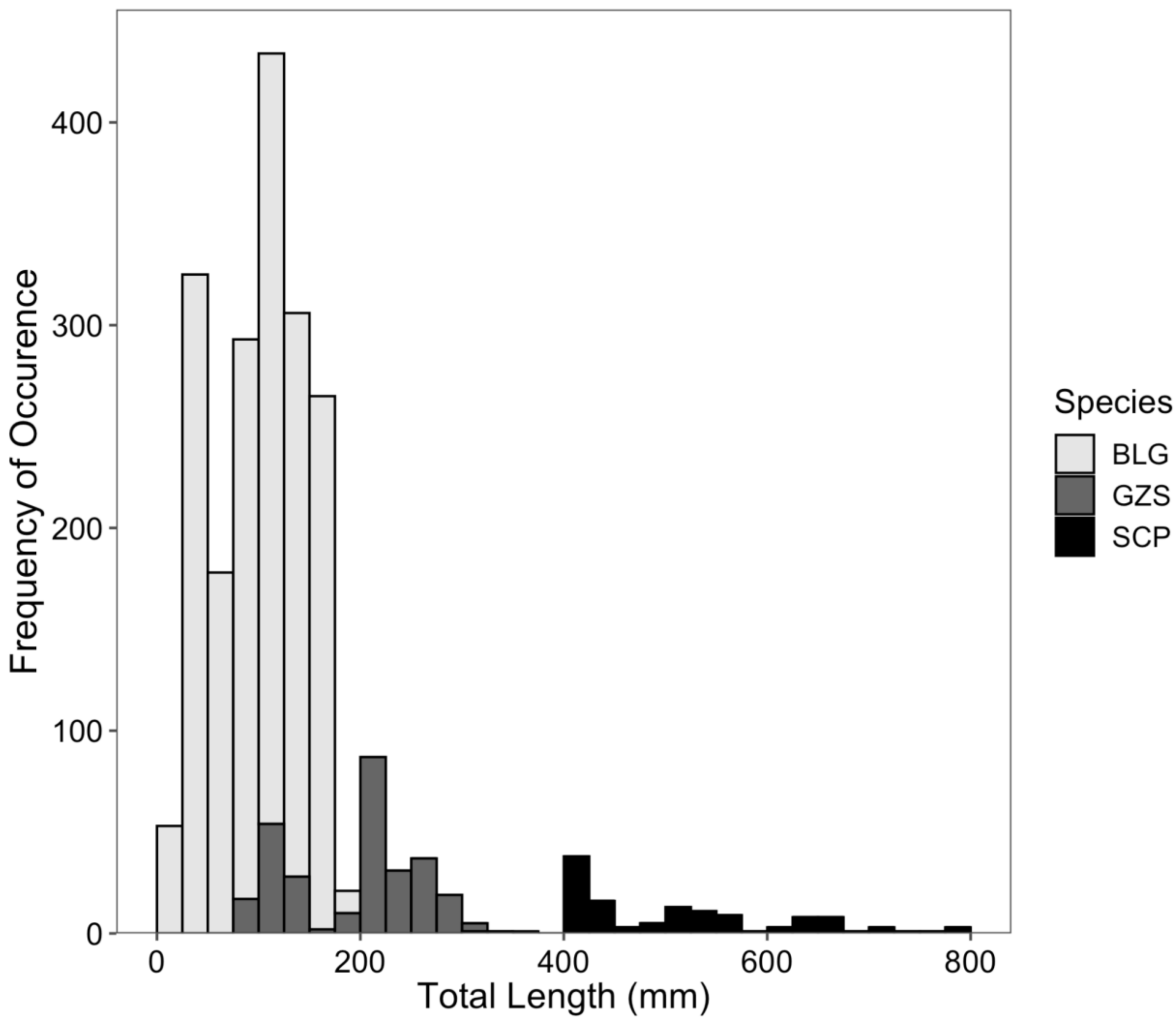

Figure 21. Length-Frequency distribution for Bluegill, Gizzard Shad, and Silver Carp collected in Buttonland Swamp in the Cache River wetlands in 2021 and 2020. Mean Bluegill length: 100 mm; mean Gizzard Shad length: 195 mm; mean Silver Carp length: 519 mm.

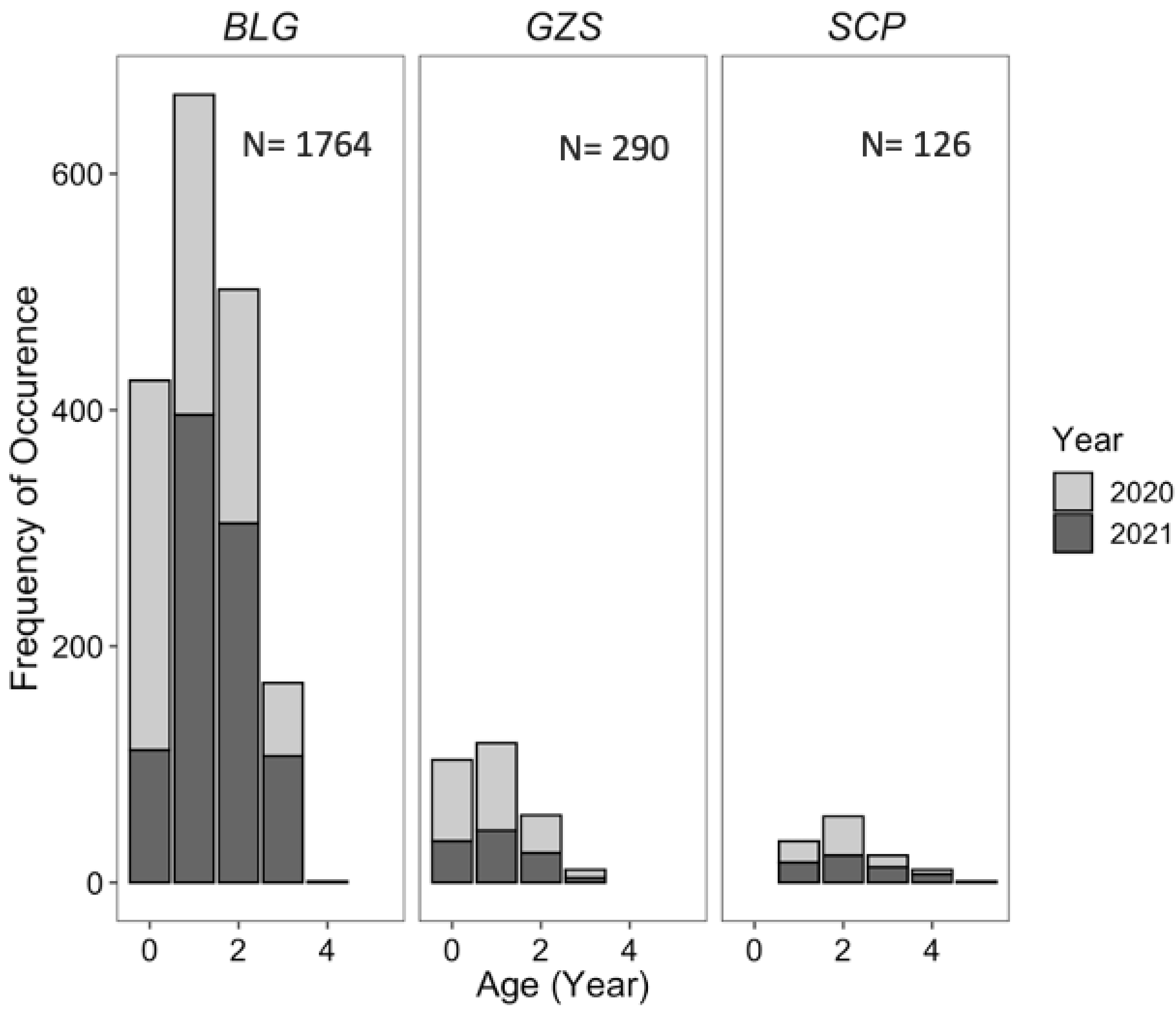

Figure 22. Age-Frequency histograms for Bluegill (BLG), Gizzard Shad (GZS), and Silver Carp (SCP) collected in Buttonland Swamp in the Cache River wetlands in 2020 and 2021.

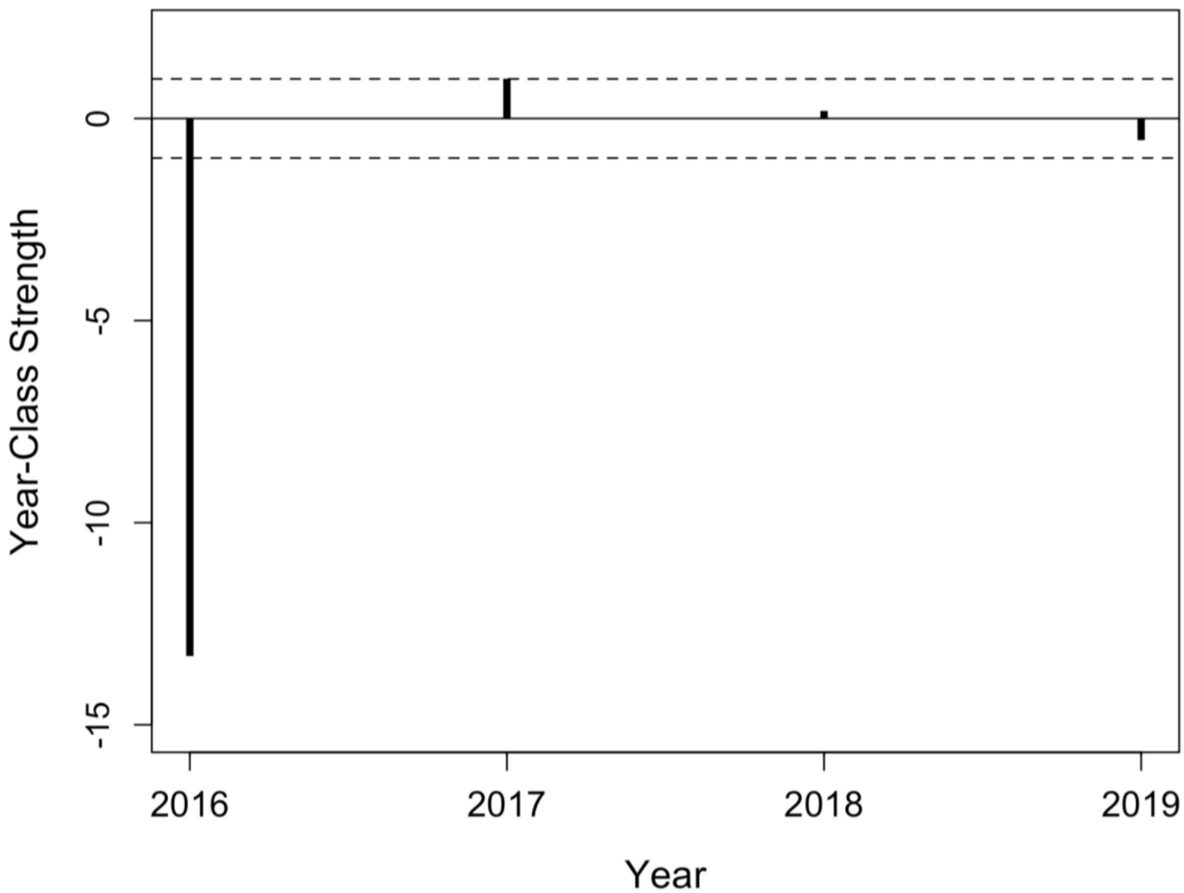

Figure 23. Year-class strength for Silver Carp sampled in 2021 in Buttonland Swamp in the Cache River wetlands. Catch curve residuals for estimating year-class strength with dashed lines representing 20th and 80th percentiles of the t distribution (year-class above upper dotted line are considered “strong” and year-class below lower dotted line are “weak”).

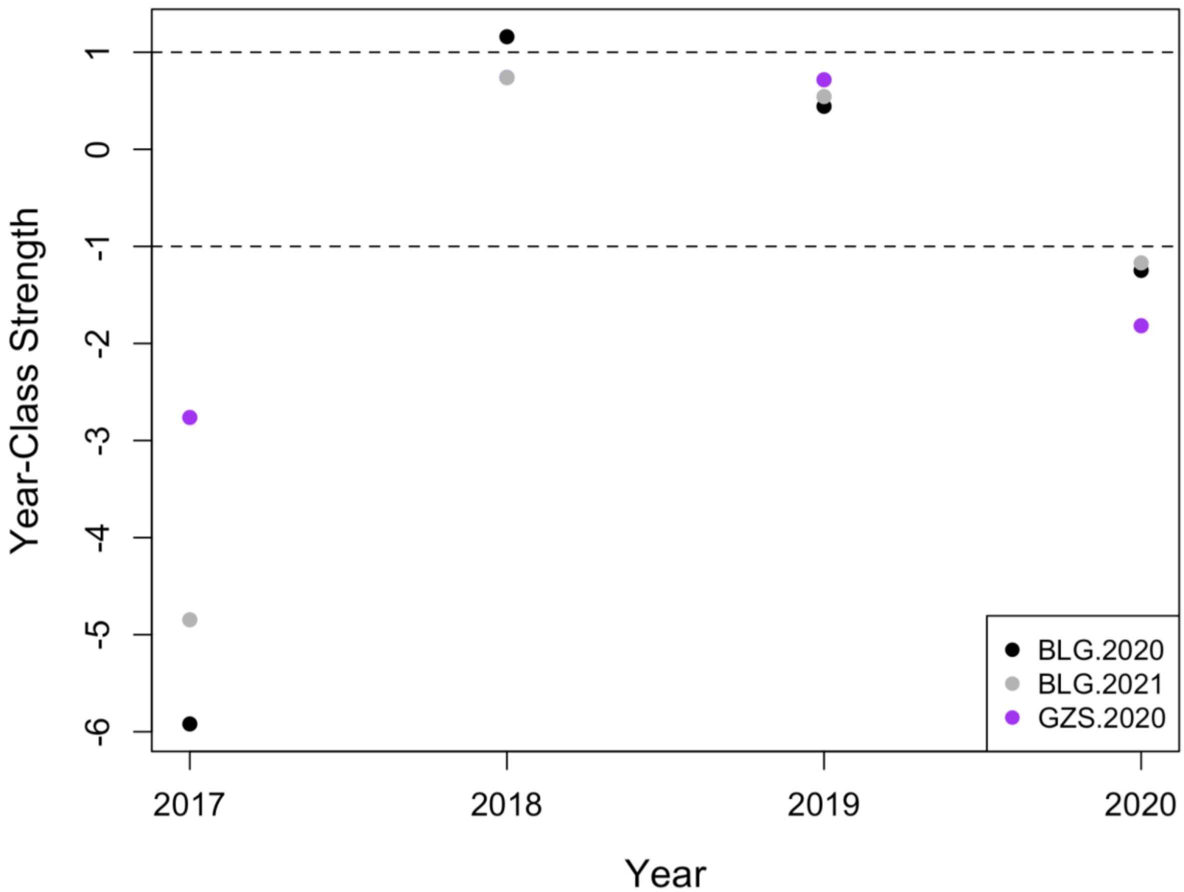

Figure 24. Year-class strength for Bluegill sampled in 2020 (in black) and 2021 (in gray), and Gizzard Shad sampled in 2020 (in purple) in Buttonland Swamp in the Cache River wetlands. Catch curve residuals for estimating year-class strength with dashed lines representing 20th and 80th percentiles of the t distribution (year-class above upper dotted line are considered “strong” and year-class below lower dotted line are “weak”).

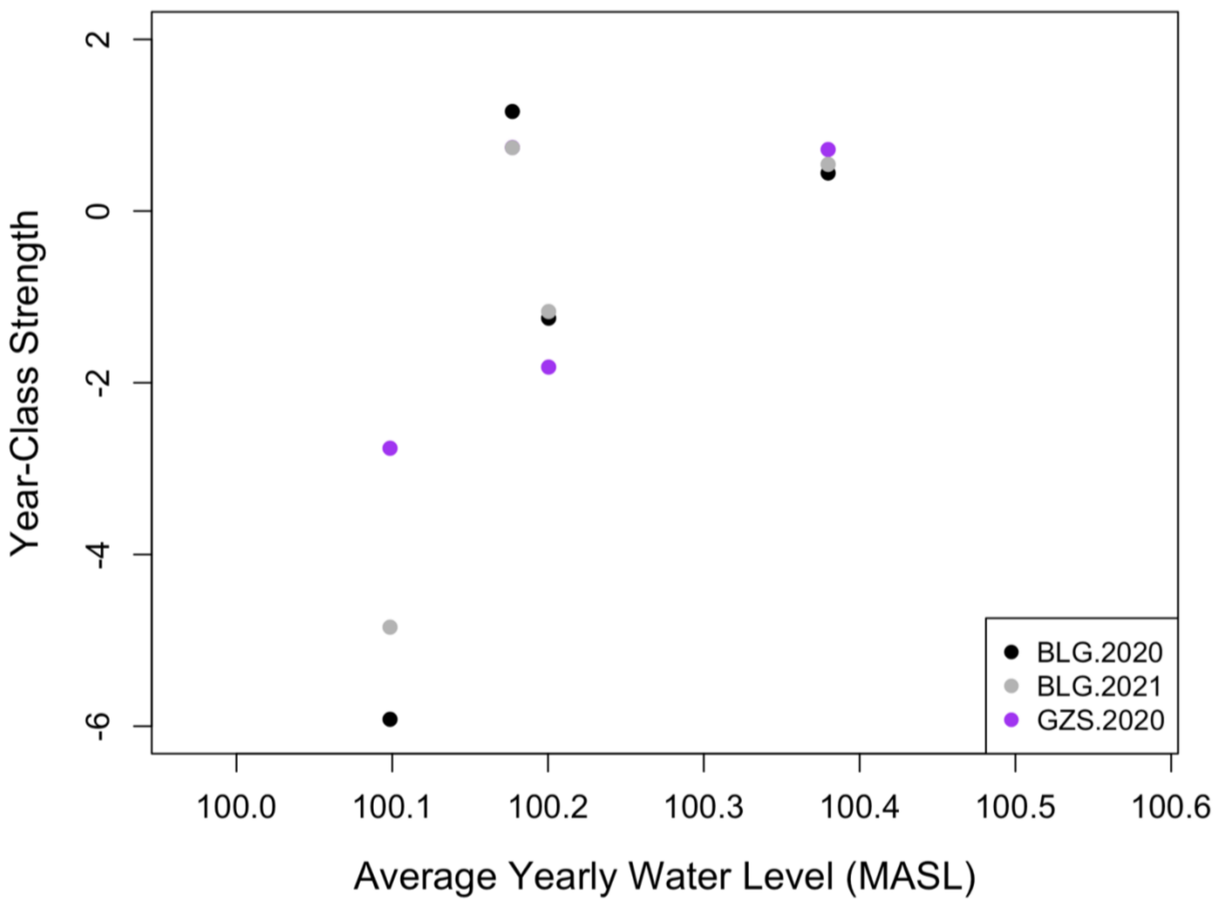

Figure 25. Catch curve residuals showing year-class strength and average yearly water level associations (2017-2020) for Bluegill from 2020 sampling (in black; $X^2 = 1.410$, P=0.235) and 2021 sampling (in gray; $X^2 = 1.825$, P=0.177) and Gizzard Shad from 2020 sampling in Buttonland Swamp in the Cache River wetlands (in purple; $X^2 = 1.751$, P = 0.186).

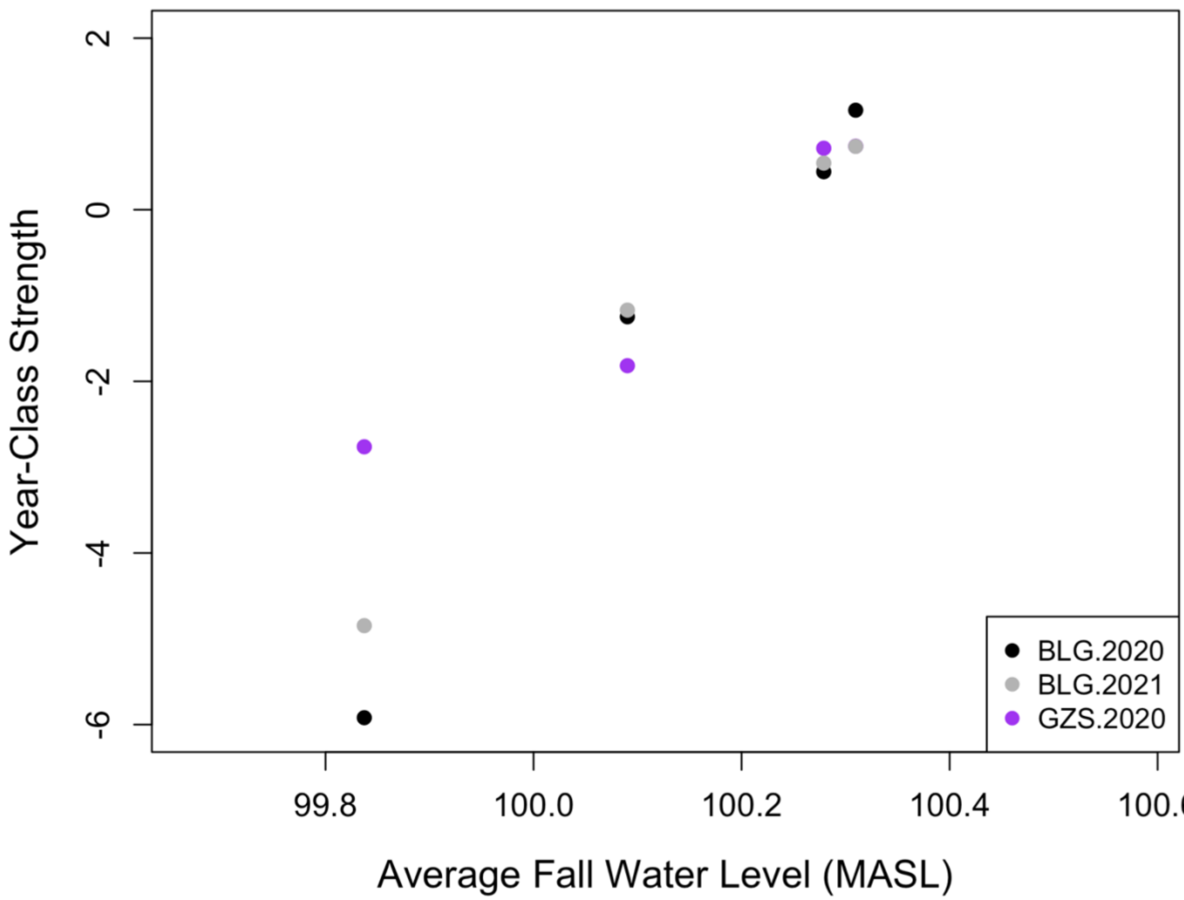

Figure 26a. Catch curve residuals showing year-class strength and average fall (October-December) water level associations (2017-2020) for Bluegill from 2020 sampling (in black; $X^2 = 83.639$, $P < 0.001$) and 2021 sampling (in gray; $X^2 = 124.860$, $P < 0.001$) and Gizzard Shad from 2020 sampling in Buttonland Swamp (in purple; $X^2 = 23.718$, $P < 0.001$).

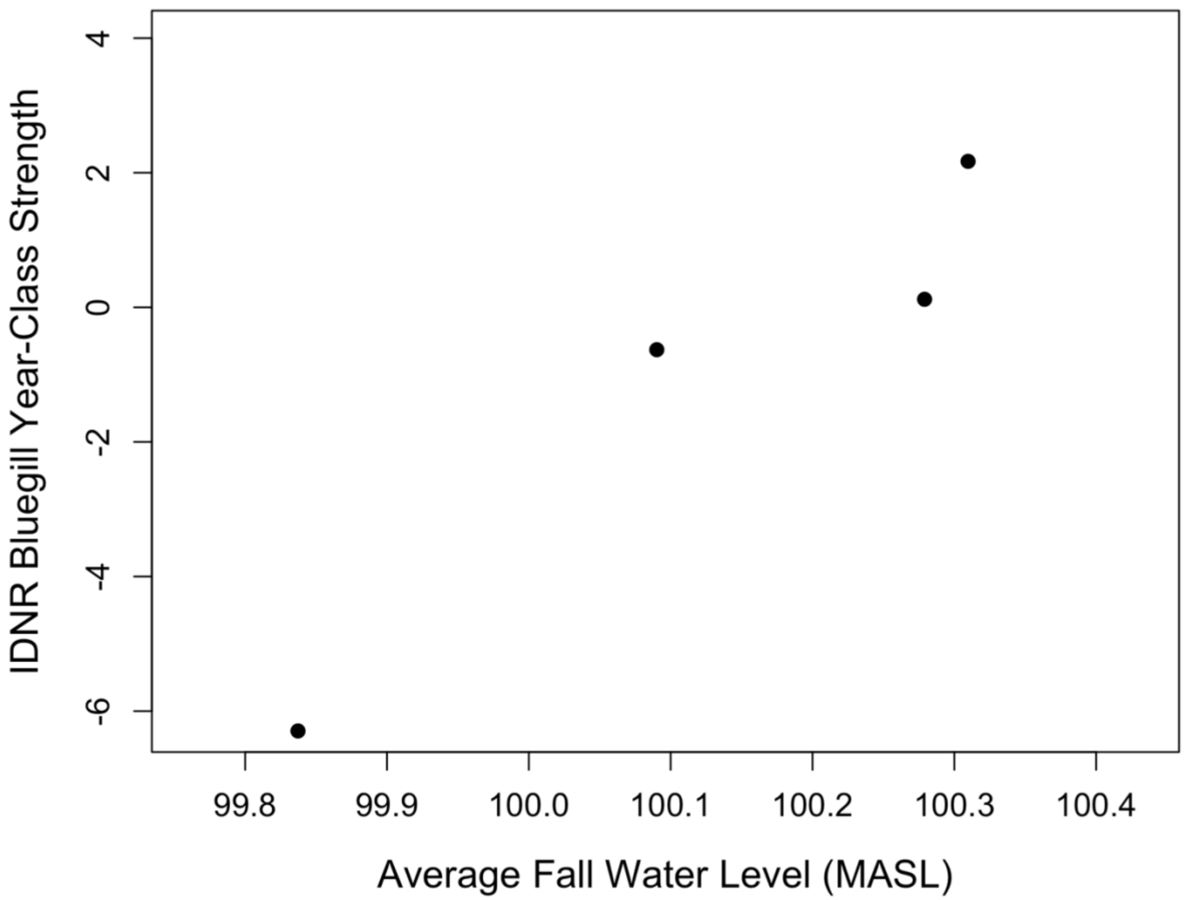

Figure 26b. Catch curve residuals showing year-class strength and average spawn (May-June) water level associations (2017-2020) for Bluegill from IDNR 2020 sampling in Buttonland Swamp; $X^2 = 24.272$, $P < 0.001$.

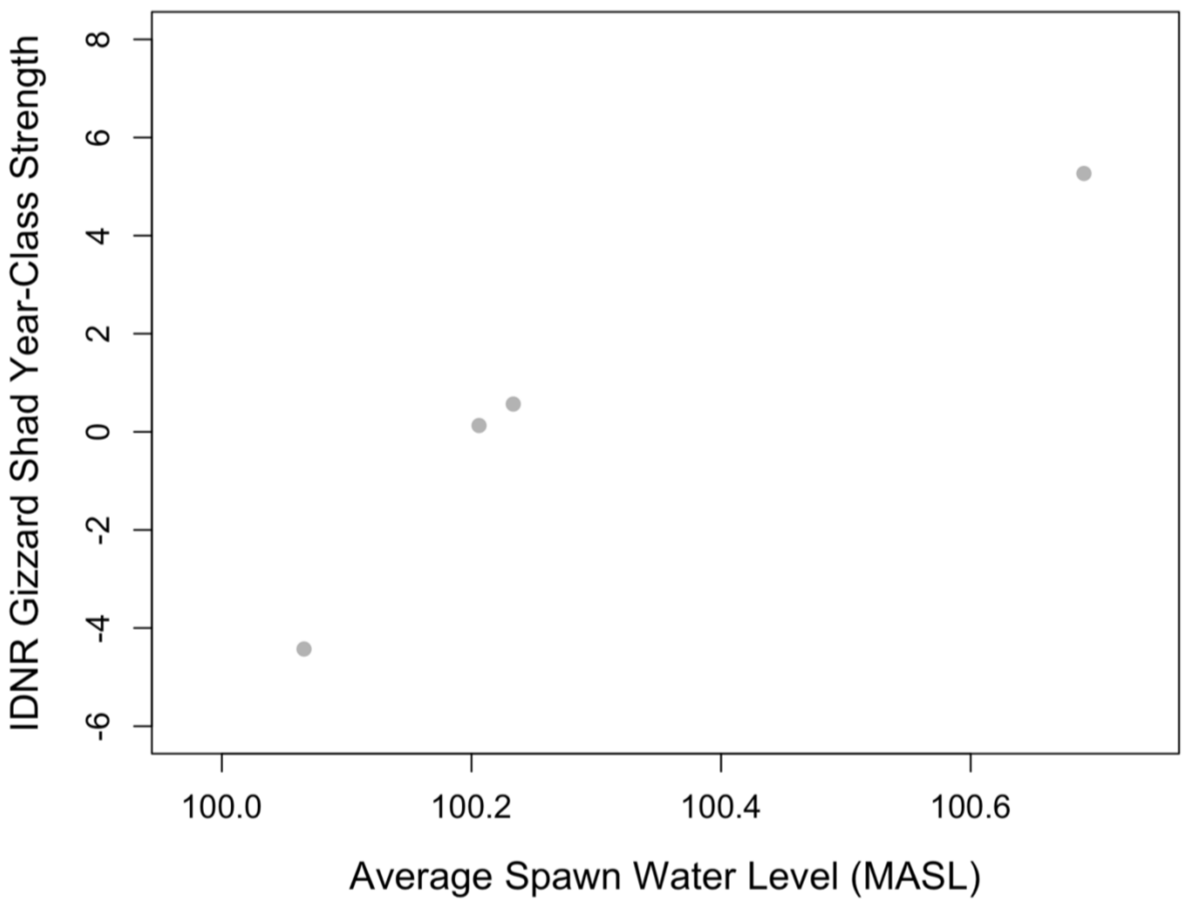

Figure 27. Catch curve residuals showing year-class strength and average spawn (May-June) water level associations (2008-2011) for Gizzard Shad from IDNR 2011 sampling in Buttonland Swamp; X^2 = 7.881, P = 0.005.

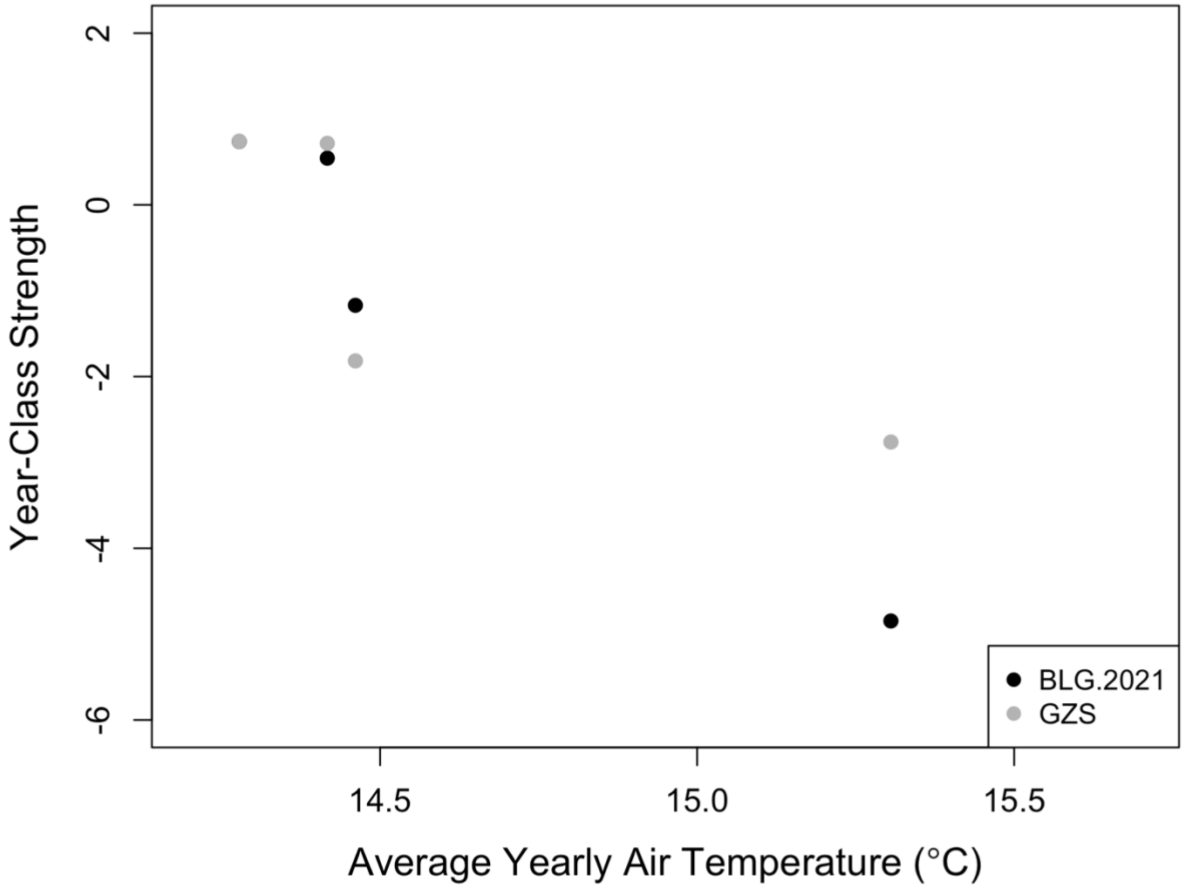

Figure 28a. Catch curve residuals showing year-class strength and average yearly air temperature (℃) associations (2017-2020) for Bluegill from 2021 (in black; $X^2 = 34.072$, $P < 0.001$) and Gizzard Shad from 2020 (in gray; $X^2 = 3.753$, $P = 0.053$) data in Buttonland Swamp.

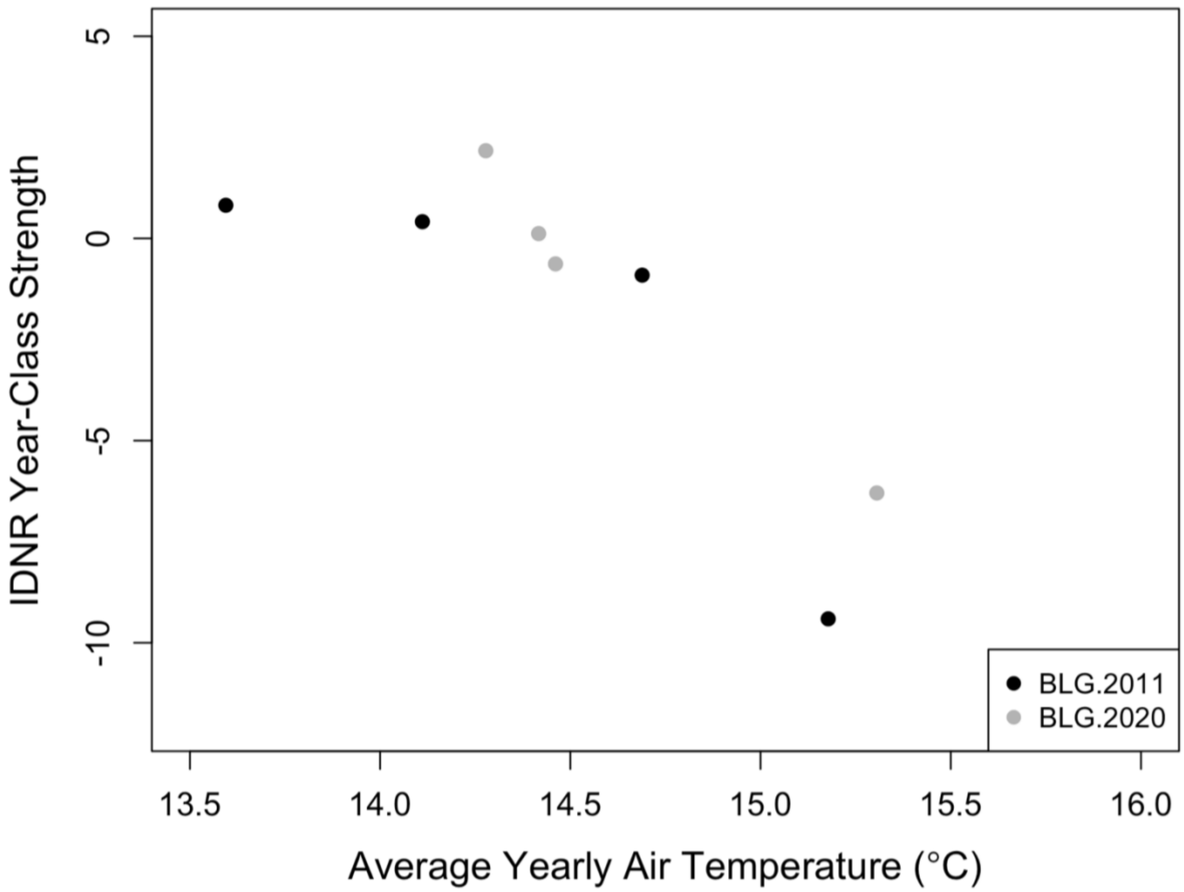

Figure 28b. Catch curve residuals showing year-class strength and average yearly air temperature (℃) associations (2017-2020) for Bluegill from 2011 (in black; $X^2 = 5.093$, $P = 0.024$) and from 2020 (in gray; $X^2 = 72.365$, $P < 0.001$) from IDNR 2020 data in Buttonland Swamp.

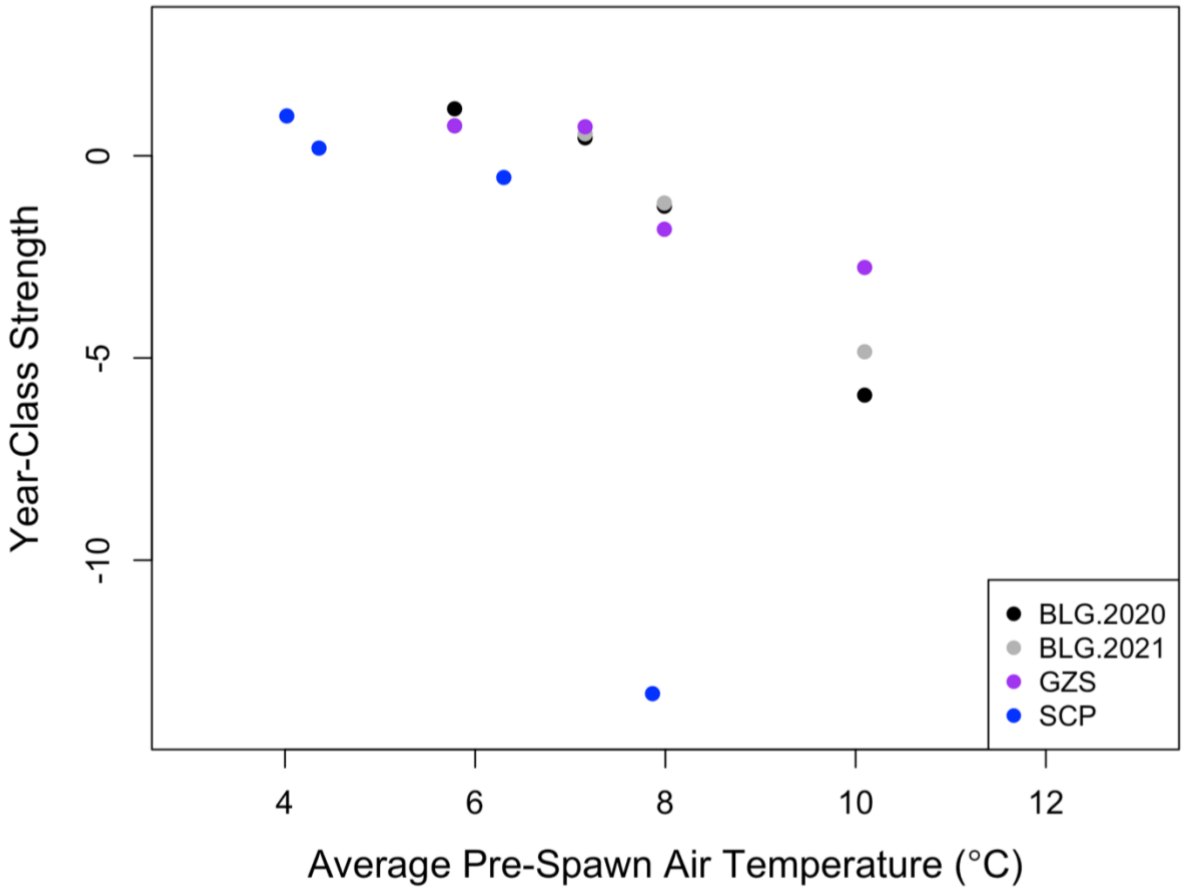

Figure 29a. Catch curve residuals showing year-class strength and pre-spawn/winter (January-April) average air temperature (°C) associations (2017-2020) for Bluegill from 2020 (in black; $X^2 = 29.965$, $P < 0.001$) and 2021 (in gray; $X^2 = 20.965$, $P < 0.001$), Gizzard Shad from 2020 (in purple; $X^2 = 9.381$, $P = 0.002$), and Silver Carp from 2021(in blue; associations from 2016-2019; $X^2 = 6.334$, $P = 0.012$) sampling in Buttonland Swamp from SIU data.

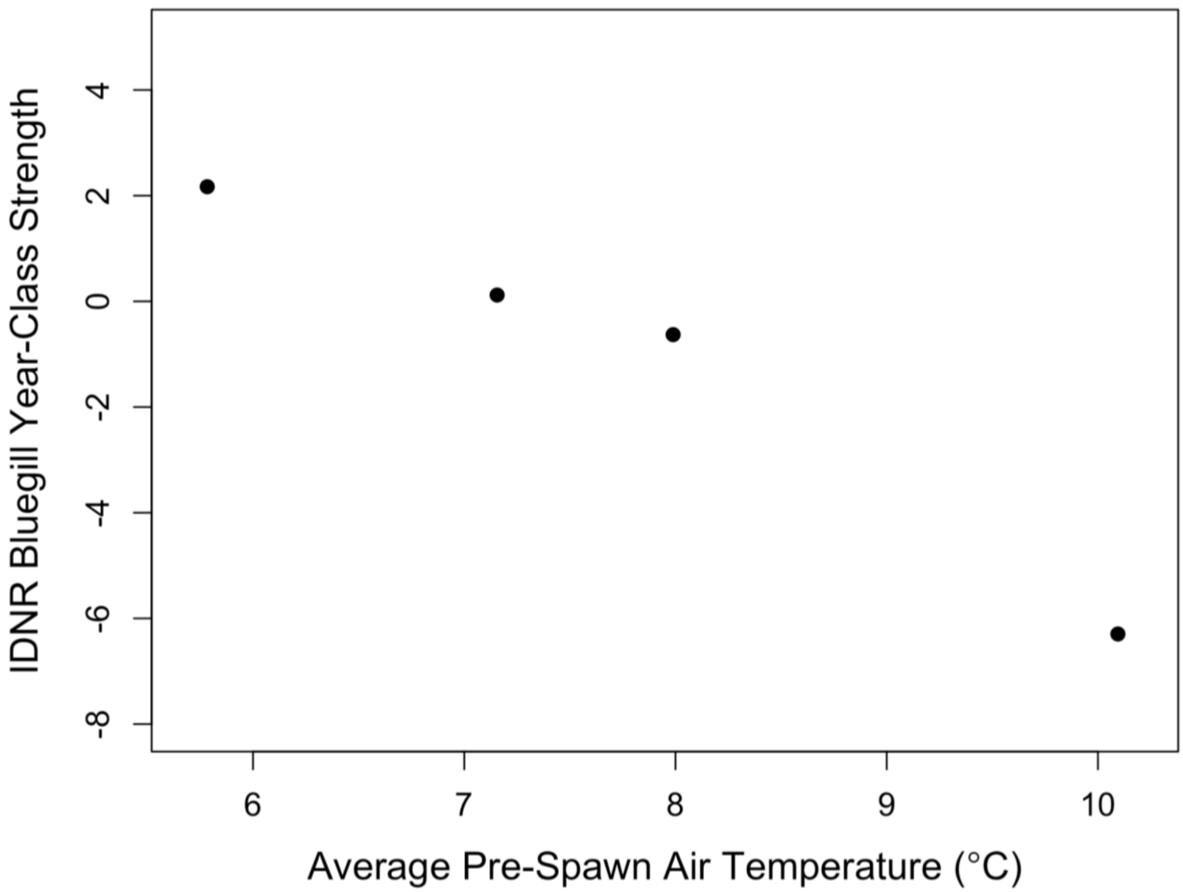

Figure 29b. Catch curve residuals showing year-class strength and pre-spawn/winter (January-April) average air temperature (°C) associations (2017-2020) for Bluegill from IDNR 2020 sampling in Buttonland Swamp; $X^2 = 47.928$, $P < 0.001$.

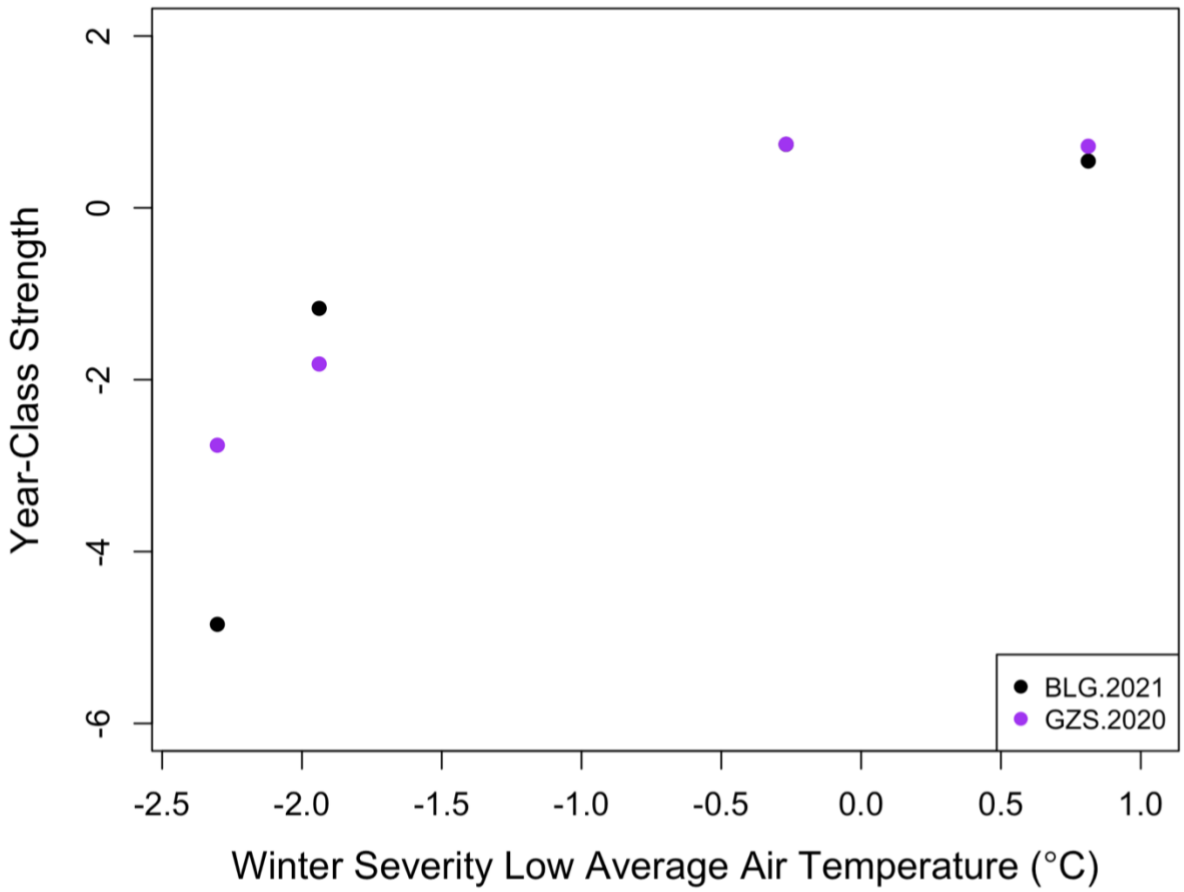

Figure 30. Catch curve residuals showing year-class strength and winter severity (December-February) low average air temperature (℃) associations (2017-2020) for Gizzard Shad (from SIU 2020 sampling; $X^2 = 16.899$, $P < 0.001$) and Bluegill (from SIU 2021 sampling; $X^2 = 4.145$, $P = 0.042$) in Buttonland Swamp.

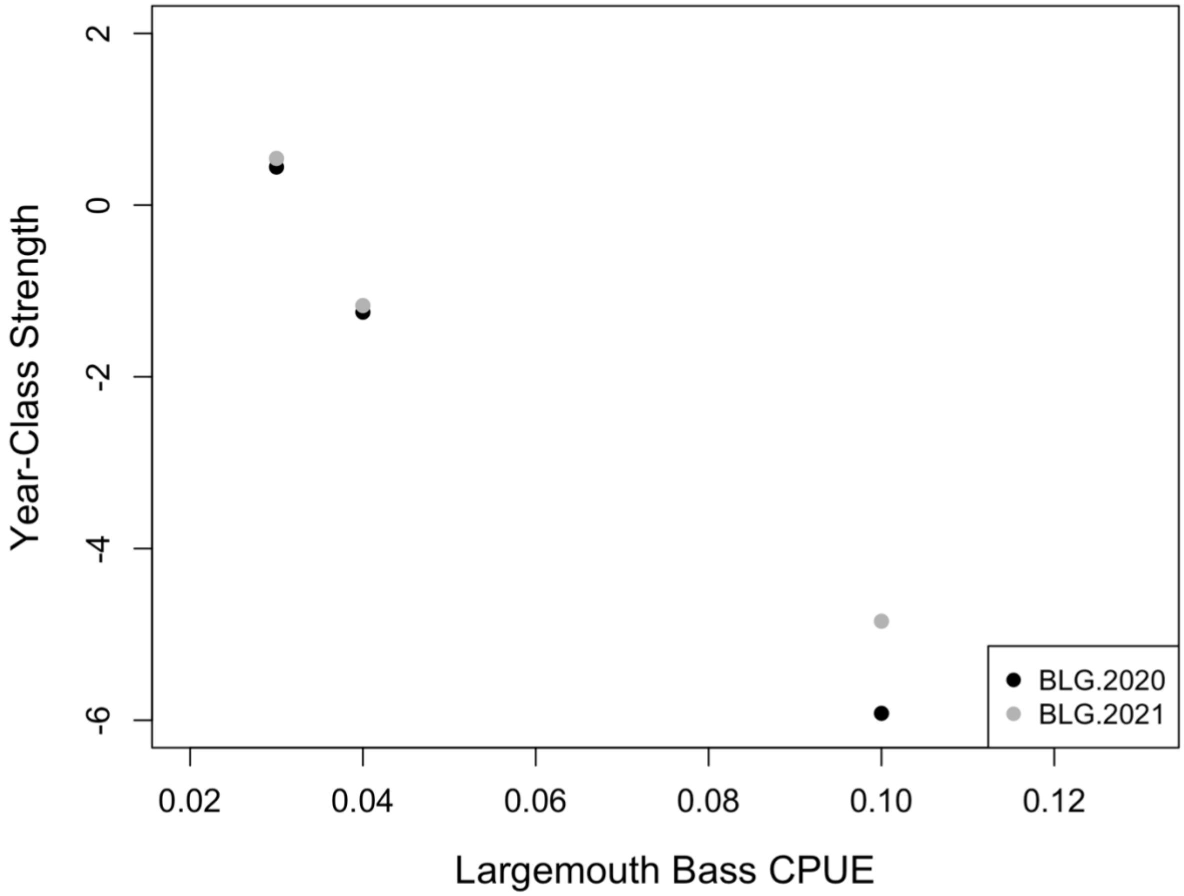

Figure 31a. Catch curve residuals showing year-class strength and yearly average historical Largemouth Bass CPUE associations (2017, 2019, 2020) for Bluegill from SIU 2020 sampling (in black; $X^2 = 61.440$, $P < 0.001$) and 2021 sampling (in gray; $X^2 = 28.904$, $P < 0.001$) in Buttonland Swamp.

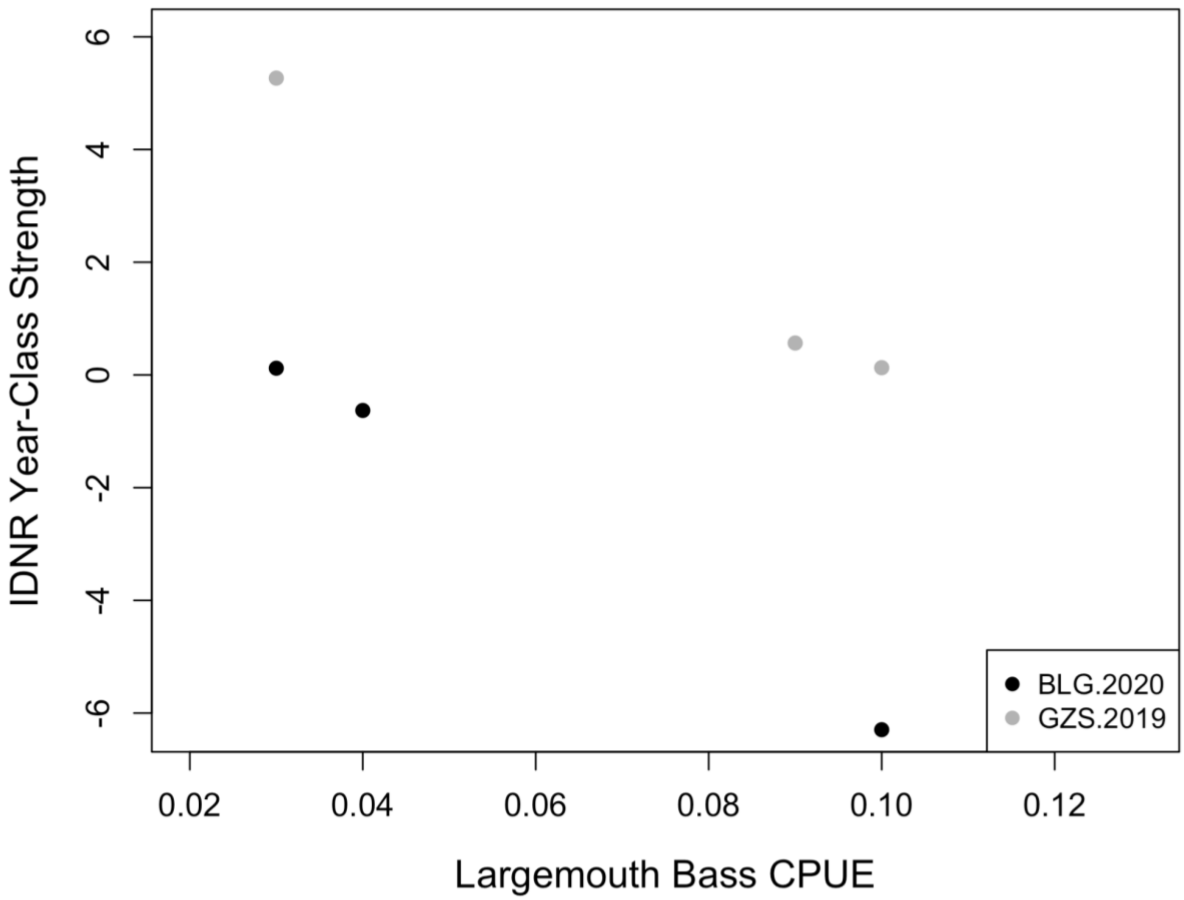

Figure 31b. Catch curve residuals showing year-class strength and yearly average historical Largemouth Bass CPUE associations (2017, 2019, 2020) for Bluegill from IDNR 2020 sampling (in black; X^2 =1542.600, P < 0.001) and Gizzard Shad from IDNR 2019 sampling in Buttonland Swamp (in gray; X^2 =322.740, P < 0.001).

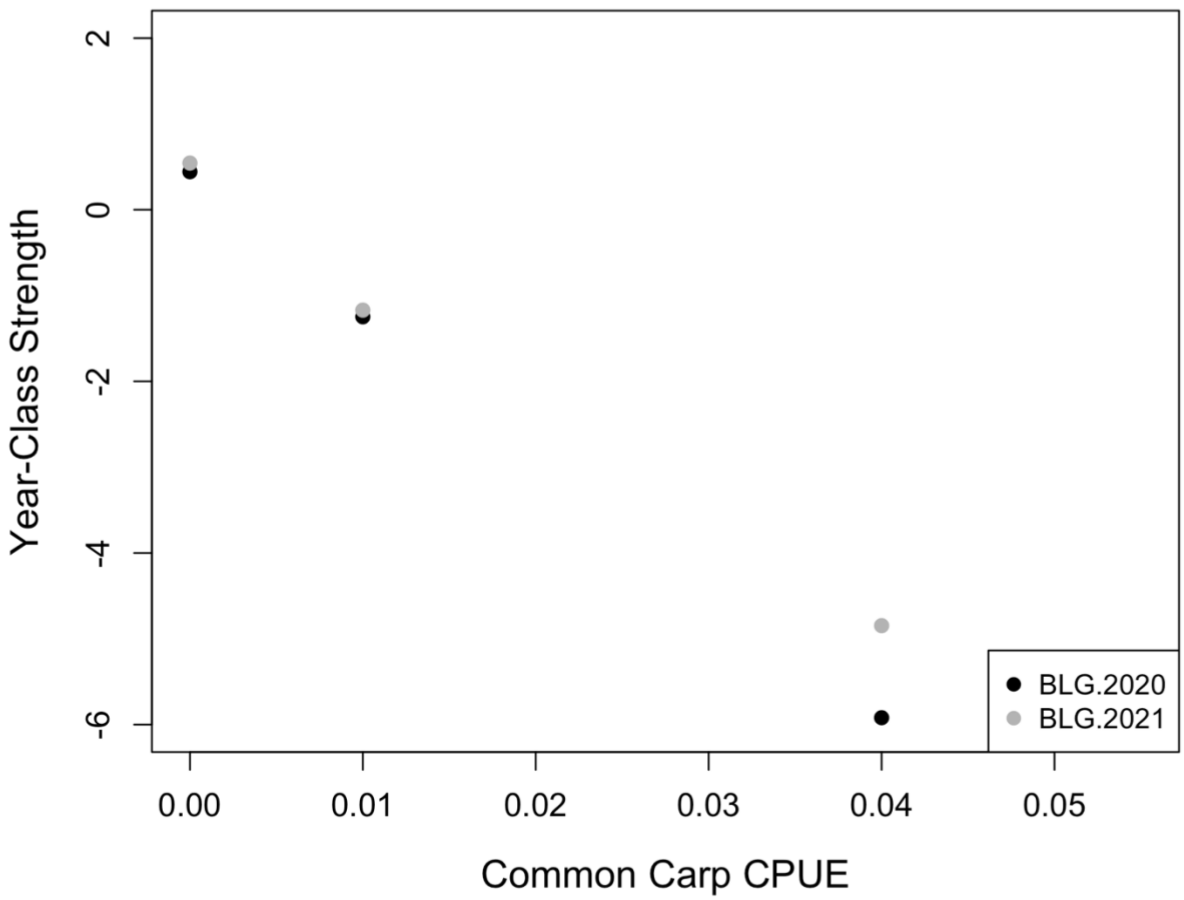

Figure 32a. Catch curve residuals showing year-class strength and yearly average historical Common Carp CPUE associations (2017, 2019, 2020) for Bluegill from SIU 2020 sampling (in black; $X^2 = 3549.700$, $P < 0.001$) and 2021 sampling (in gray; $X^2 = 182.980$, $P < 0.001$) in Buttonland Swamp.

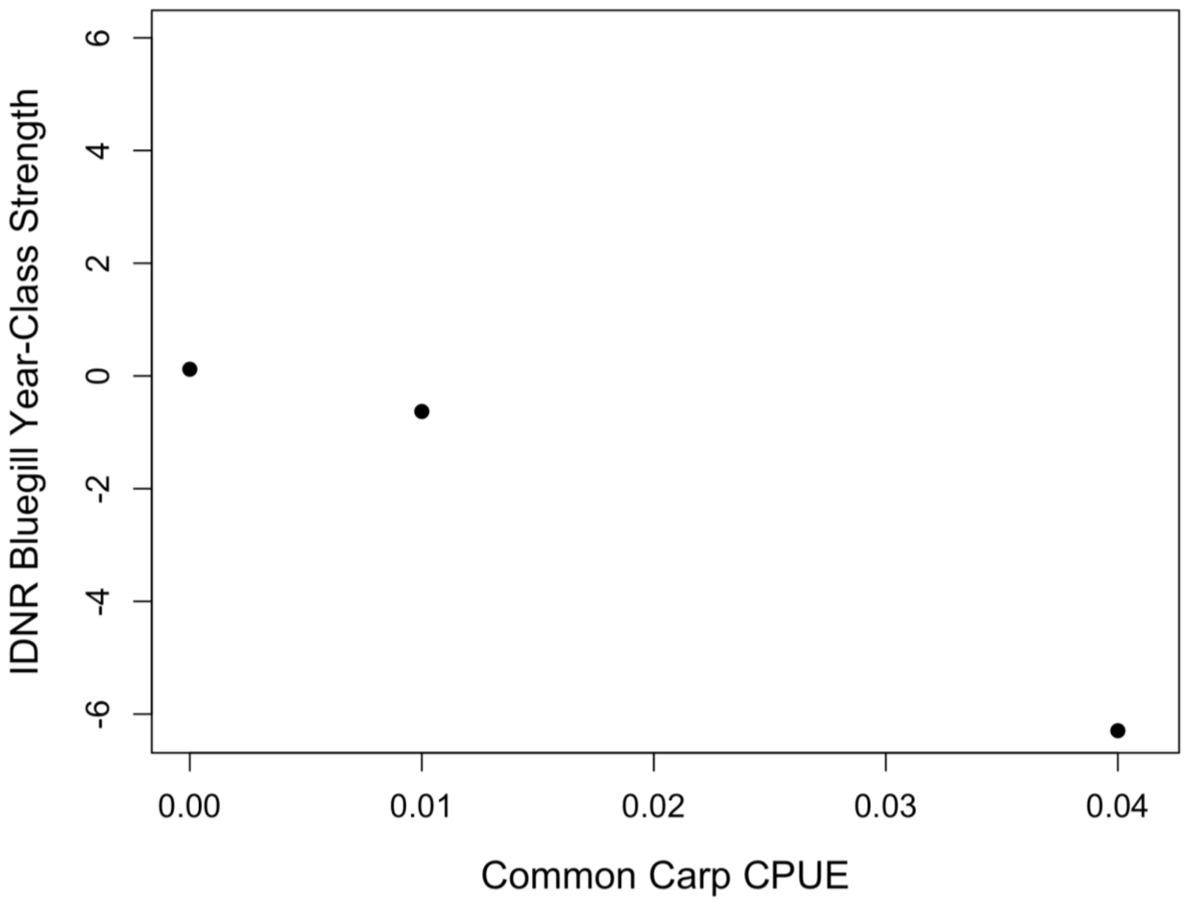

Figure 32b. Catch curve residuals showing year-class strength and yearly average historical Common Carp CPUE associations (2017, 2019, 2020) for Bluegill from IDNR 2020 sampling (in black) in Buttonland Swamp; $X^2 = 53.748$, $P < 0.001$.

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APPENDIX A

MEAN ENVIRONMENTAL VARIABLES WITHIN EACH HABITAT

Habitat	Depth (m)	EmergVeg (%)	Submerg (%)	Float (%)	Algae (%)	OpenWtr (%)	Macro (%)
Cache River	0.75	78.27	0.52	3.01	2.81	54.68	0.41
Eagle Pond	0.85	87.45	0.02	0.82	2.53	53.90	0.00
Main Channel	0.75	94.36	0.00	0.10	2.11	43.52	0.04
Side Channel	0.64	95.38	0.00	0.41	3.10	49.4	0.09
Nearshore Vegetated	0.59	91.00	0.39	0.90	1.38	54.79	0.15
Offshore Vegetated	0.76	97.17	0.01	0.83	1.78	39.93	0.15
Open Water	0.37	35.24	0.00	3.13	11.63	94.06	0.00

Habitat	Sticks.Logs (%)	Cypress (%)	OtherTrees (%)	Buttonbush (%)	SiltClay (%)	Gravel (%)
Cache River	8.86	5.91	3.03	25.60	74.81	0.00
Eagle Pond	7.53	8.43	5.85	24.12	64.35	0.00
Main Channel	6.46	6.43	0.76	41.65	81.28	1.49
Side Channel	6.24	5.65	1.61	36.90	79.53	0.29
Nearshore Vegetated	11.02	4.52	4.39	23.71	54.16	1.33
Offshore Vegetated	6.06	8.47	2.37	42.56	84.10	0.00
Open Water	1.55	3.08	0.05	1.25	93.55	0.00

Habitat	Cobble (%)	Detritus (%)	CC (%)	DO (%)	Wtemp (%)	Conductivity (%)	Secchi (m)
Cache River	0.16	25.03	24.60	4.62	20.79	153.85	0.35
Eagle Pond	0.00	35.65	30.68	4.41	19.88	139.88	0.44
Main Channel	0.84	16.38	9.19	4.74	20.26	155.16	0.45
Side Channel	0.13	20.05	14.46	4.84	19.80	150.77	0.39
Nearshore Vegetated	0.85	43.67	33.28	4.86	19.17	149.85	0.40
Offshore Vegetated	0.00	15.90	14.48	4.51	20.79	149.69	0.41
Open Water	0.00	6.45	3.85	4.74	20.17	150.90	0.42

Table A1. Mean environmental variables for each macrohabitat and microhabitat with all gears combined using Southern Illinois University (SIU) data collected from Buttonland Swamp within the Cache River watershed in 2020-2021.

APPENDIX B

INDICATOR SPECIES ANALYSIS WITHIN SEASONS FOR HABITATS THAT HAD SIGNIFICANTLY DIFFERENT ASSEMBLAGE STRUCTURE AMONG SEASONS

Species (*Genus species*)	**Season**	**Indicator Value**	**P**
Black Buffalo (*Ictiobus niger*)	Fall	0.348	0.041
Warmouth (*Labidesthes sicculus*)	Spring	0.425	0.007
Spotted Gar (*Lepisosteus oculatus*)	Winter	0.510	<0.001
Brook Silverside (*Labidesthes sicculus*)	Winter	0.395	<0.010
Bluegill (*Lepomis macrochirus)*	Fall + Spring	0.573	<0.001
Orangespotted Sunfish (*Lepisosteus platostomus*)	Fall + Spring	0.532	0.001
Smallmouth Buffalo (*Ictiobus bubalus*)	Fall + Spring	0.436	0.005
Taillight Shiner (*Notropis maculatus*)	Spring + Winter	0.517	<0.001

Table B1. Indicator Species Analysis using SIU electrofishing data from the side channels within Buttonland Swamp (2020-2021) for each season ($\alpha = 0.05$).

Species (*Genus species*)	**Season**	**Indicator Value**	**P**
Threadfin Shad (*Dorosoma petenense*)	Fall	0.343	0.027
Warmouth (*Labidesthes sicculus*)	Spring	0.351	0.033
Spotted Gar (*Lepisosteus oculatus*)	Winter	0.738	<0.001
Bluegill (*Lepomis macrochirus)*	Fall + Spring + Summer	0.435	0.005
Redear Sunfish (*Lepomis microlophus*)	Fall + Spring + Summer	0.390	0.015

Table B2. Indicator Species Analysis using SIU electrofishing data from the main channel within Buttonland Swamp (2020-2021) for each season ($\alpha = 0.05$).

Species (*Genus species*)	**Season**	**Indicator Value**	**P**
Warmouth (*Labidesthes sicculus*)	Spring	0.470	<0.001
Redear Sunfish (*Lepomis microlophus*)	Spring	0.253	0.025
Gizzard Shad *Dorosoma cepedianum*)	Summer	0.232	0.036
Spotted Gar (*Lepisosteus oculatus*)	Winter	0.600	<0.001
Taillight Shiner (*Notropis maculatus*)	Winter	0.420	<0.001
Brook Silverside (*Labidesthes sicculus*)	Winter	0.325	0.001
Black Crappie (*Pomoxis nigromaculatus*)	Winter	0.223	0.050
Bluegill (*Lepomis macrochirus)*	Fall + Spring	0.352	0.001
Spotted Bass (*Micropterus punctulatus)*	Fall + Spring	0.272	0.015
Orangespotted Sunfish (*Lepisosteus platostomus*)	Fall + Spring	0.224	0.049
Freshwater Drum (*Aplodinotus grummiens*)	Fall + Spring + Summer	0.241	0.032

Table B3. Indicator Species Analysis using SIU electrofishing data from the offshore vegetated habitat within Buttonland Swamp (2020-2021) for each season ($\alpha = 0.05$).

APPENDIX C

MODEL COMBINATIONS USED TO DETERMINE IF WATER DEPTH VARIED SIGNIFICANTLY AMONG HABITATS

Fixed Effect	**Random Effect**	**Interaction**	**AR1**	**AIC**	**P-Value**
**Water Depth ~ Macrohabitat	Season	None	Yes	1801.34	0.003
Water Depth ~ Macrohabitat	Year	None	Yes	1801.35	0.003
Water Depth ~ Macrohabitat	Year	Macrohabitat and Year	Yes	1801.93	0.950
Water Depth ~ Macrohabitat	Season	Macrohabitat and Year	Yes	1802.23	0.946
Water Depth ~ Macrohabitat	Year	Macrohabitat and Season	Yes	1807.04	0.916
Water Depth ~ Macrohabitat	Season	Macrohabitat and Season	Yes	1808.91	0.918
Water Depth ~ Macrohabitat	None	Macrohabitat and Season	No	2022.02	0.128
Water Depth ~ Macrohabitat	None	Macrohabitat and Year	No	2026.51	0.633
Water Depth ~ Macrohabitat	None	None	No	2048.63	<0.001

Table C1. Using SIU water depth data, model combinations of repeated measures using mixed models were used to see how water depth changed over seasons and years among macrohabitats in Buttonland Swamp within the Cache River watershed from 2020 - 2021($\alpha = 0.05$). The model chosen was the one with the lowest AIC. **model chosen with the lowest AIC.

Fixed Effect	Random Effect	Interaction	AR1	AIC	P-Value
**Water Depth ~ Microhabitat	Year	None	Yes	1570.36	<0.001
Water Depth ~ Microhabitat	Season	None	Yes	1571.21	<0.001
Water Depth ~ Microhabitat	Year	Microhabitat and Year	Yes	1571.25	0.300
Water Depth ~ Microhabitat	Season	Microhabitat and Year	Yes	1572.93	0.033
Water Depth ~ Microhabitat	Season	Microhabitat and Year	Yes	1573.28	0.349
Water Depth ~ Microhabitat	Season	Microhabitat and Season	Yes	1574.30	0.032
Water Depth ~ Microhabitat	None	Microhabitat and Season	No	1892.31	0.141
Water Depth ~ Microhabitat	None	Microhabitat and Year	No	1904.76	0.915
Water Depth ~ Microhabitat	None	None	No	1929.56	<0.001

Table C2. Using SIU water depth data, model combinations of repeated measures using mixed models were used to see how water depth changed over seasons and years among microhabitats in Buttonland Swamp within the Cache River watershed from 2020 - 2021($\alpha = 0.05$). The model chosen was the one with the lowest AIC. **model chosen with the lowest AIC.

APPENDIX D

MODEL COMBINATIONS USED TO DETERMINE IF WATER ELEVATIONS VARIED SIGNIFICANTLY AMONG SEASONS AND YEARS

Fixed Effect	**Random Effect**	**Interaction**	**AR1**	**AIC**	**P-Value**
**Water Elevation ~ Season	Year	None	Yes	1141.19	0.012
*Water Elevation ~ Year	Season	None	Yes	1162.51	0.001
Water Elevation ~ Season	None	Season and Year	No	1309.41	<0.001
Water Elevation ~ Year	None	Year and Season	No	1309.41	<0.001
Water Elevation ~ Season	None	None	No	1397.38	<0.001
Water Elevation ~ Year	None	None	No	1400.85	<0.001

Table D1. Using Illinois Department of Natural Resources (IDNR) water elevation data, model combinations of repeated measures using mixed models were used to see how water elevation changed over seasons and years in Buttonland Swamp within the Cache River watershed from 2010 - 2021($\alpha = 0.05$). The model chosen was the one with the lowest AIC. **model chosen with the lowest AIC to see water elevation differences among seasons. *model chosen with the lowest AIC to see water elevation differences between years.

Fixed Effect	**Random Effect**	**Interaction**	**AR1**	**AIC**	**P-Value**
**Water Elevation ~ Year	Season	None	Yes	-30.34	0.833
*Water Elevation ~ Season	Year	None	Yes	-24.15	0.051
Water Elevation ~ Season	None	None	No	15.11	<0.001
Water Elevation ~ Year	None	None	No	23.71	0.439

Table D2. Using SIU water elevation data, model combinations of repeated measures using mixed models were used to see how water elevation changed over seasons and years in Buttonland Swamp within the Cache River watershed from 2020 – 2021 ($\alpha = 0.05$). The model chosen was the one with the lowest AIC. **model chosen with the lowest AIC to see water elevation differences between years. *model chosen with the lowest AIC to see water elevation differences among seasons.

APPENDIX E

MODEL COMBINATIONS USED TO DETERMINE IF CATCH PER UNIT EFFORT VARIED SIGNIFICANTLY AMONG MACROHABITATS

Fixed Effect	**Random Effect**	**Interaction**	**AR1**	**AIC**	**P-Value**
**CPUE ~ Macrohabitat	Season	None	Yes	-51.94	<0.001
CPUE ~ Macrohabitat	Year	None	Yes	-49.25	<0.001
CPUE ~ Macrohabitat	None	None	No	-46.70	<0.001
CPUE ~ Macrohabitat	None	Macrohabitat and Season	No	-36.86	<0.001
CPUE ~ Macrohabitat	Year	Macrohabitat and Season	Yes	-34.88	<0.001
CPUE ~ Macrohabitat	Season	Macrohabitat and Year	Yes	-33.08	0.280
CPUE ~ Macrohabitat	None	Macrohabitat and Year	No	-29.08	0.165

Table E1. Using SIU CPUE electrofishing data, model combinations of repeated measures using mixed models were used to see how CPUE changed over seasons and years among macrohabitats in Buttonland Swamp within the Cache River watershed ($\alpha = 0.05$). Each of these combinations was used for each of the gear types and the same model was significant, with the lowest AIC, for each gear type. The model chosen was the one with the lowest AIC. CPUE = Catch Per Unit Effort per minute or per net night. **model chosen with the lowest AIC.

Fixed Effect	Random Effect	Interaction	AR1	AIC	P-Value
**CPUE ~ Macrohabitat	Year	None	Yes	167.60	0.002
CPUE ~ Macrohabitat	Season	None	Yes	168.88	0.001
CPUE ~ Macrohabitat	Season	Macrohabitat and Year	Yes	181.50	0.198
CPUE ~ Macrohabitat	Year	Macrohabitat and Season	Yes	200.80	0.843
CPUE ~ Macrohabitat	None	None	No	213.74	<0.001
CPUE ~ Macrohabitat	None	Macrohabitat and Season	No	223.44	0.040
CPUE ~ Macrohabitat	None	Macrohabitat and Year	No	228.48	0.489

Table E2. Using SIU CPUE fyke net data, model combinations of repeated measures using mixed models were used to see how CPUE changed over seasons and years among macrohabitats) in Buttonland Swamp within the Cache River watershed ($\alpha = 0.05$). Each of these combinations was used for each of the gear types and the same model was significant, with the lowest AIC, for each gear type. The model chosen was the one with the lowest AIC. CPUE = Catch Per Unit Effort per minute or per net night. **model chosen with the lowest AIC.

Fixed Effect	Random Effect	Interaction	AR1	AIC	P-Value
**CPUE ~ Macrohabitat	Season	None	Yes	208.78	0.084
CPUE ~ Macrohabitat	Year	None	Yes	211.99	0.112
CPUE ~ Macrohabitat	None	None	No	218.93	0.484
CPUE ~ Macrohabitat	Season	Macrohabitat and Year	Yes	230.54	0.057
CPUE ~ Macrohabitat	None	Macrohabitat and Year	No	238.11	0.310
CPUE ~ Macrohabitat	Year	Macrohabitat and Year	Yes	239.51	0.891
CPUE ~ Macrohabitat	None	Macrohabitat and Season	No	247.82	0.648

Table E3. Using SIU CPUE mini fyke net data, model combinations of repeated measures using mixed models were used to see how CPUE changed over seasons and years among macrohabitats in Buttonland Swamp within the Cache River watershed ($\alpha = 0.05$). Each of these combinations was used for each of the gear types and the same model was significant, with the lowest AIC, for each gear type. The model chosen was the one with the lowest AIC. CPUE = Catch Per Unit Effort per minute or per net night. **model chosen with the lowest AIC.

APPENDIX F

MODEL COMBINATIONS USED TO DETERMINE IF CATCH PER UNIT EFFORT VARIED SIGNIFICANTLY AMONG MICROHABITATS

Fixed Effect	**Random Effect**	**Interaction**	**AR1**	**AIC**	**P-Value**
**CPUE ~ Microhabitat	Season	None	Yes	-77.77	<0.001
CPUE ~ Microhabitat	Year	None	Yes	-71.71	<0.001
CPUE ~ Microhabitat	Season	Microhabitat and Year	Yes	-64.59	0.083
CPUE ~ Microhabitat	Year	Microhabitat and Season	Yes	-62.99	<0.001
CPUE ~ Microhabitat	None	None	No	-60.44	<0.001
CPUE ~ Microhabitat	None	Microhabitat and Season	No	-55.96	<0.001
CPUE ~ Microhabitat	None	Microhabitat and Year	No	-47.50	0.120

Table F1. Using Southern Illinois University's (SIU) catch per unit effort (CPUE) electrofishing data, model combinations of repeated measures using mixed models were used to see how CPUE changed over seasons and years among microhabitats in Buttonland Swamp within the Cache River watershed ($\alpha = 0.05$). Each of these combinations was used for each of the gear types, although only models using electrofishing data were able to run, so the best model chosen above is only using electrofishing data. The model chosen was the one with the lowest AIC. CPUE = Catch Per Unit Effort per minute or per net night. **model chosen with the lowest AIC.

Fixed Effect	Random Effect	Interaction	AR1	AIC	P-Value
**CPUE ~ Microhabitat	Year	None	Yes	167.58	0.073
CPUE ~ Microhabitat	Season	None	Yes	171.05	0.134
CPUE ~ Microhabitat	Season	Microhabitat and Year	Yes	179.84	0.360
CPUE ~ Microhabitat	Year	Microhabitat and Season	Yes	184.32	0.449
CPUE ~ Microhabitat	None	None	No	223.62	0.045
CPUE ~ Microhabitat	None	Microhabitat and Season	No	231.76	0.847
CPUE ~ Microhabitat	None	Microhabitat and Year	No	233.03	0.994

Table F2. Using SIU CPUE fyke net data, model combinations of repeated measures using mixed models were used to see how CPUE changed over seasons and years among microhabitats in Buttonland Swamp within the Cache River watershed ($\alpha = 0.05$). Each of these combinations was used for each of the gear types, although only models using electrofishing data were able to run, so the best model chosen above is only using electrofishing data. The model chosen was the one with the lowest AIC. CPUE = Catch Per Unit Effort per minute or per net night. **model chosen with the lowest AIC.

Fixed Effect	Random Effect	Interaction	AR1	AIC	P-Value
**CPUE ~ Microhabitat	Season	None	Yes	186.77	<0.001
CPUE ~ Microhabitat	Year	None	Yes	188.41	<0.001
CPUE ~ Microhabitat	Season	Microhabitat and Year	Yes	191.52	0.402
CPUE ~ Microhabitat	Year	Microhabitat and Season	Yes	203.75	0.635
CPUE ~ Microhabitat	None	Microhabitat and Year	No	205.00	0.193
CPUE ~ Microhabitat	None	None	No	206.23	<0.001
CPUE ~ Microhabitat	None	Microhabitat and Season	No	214.19	0.755

Table F3. Using SIU CPUE mini fyke net data, model combinations of repeated measures using mixed models were used to see how CPUE changed over seasons and years among microhabitats in Buttonland Swamp within the Cache River watershed ($\alpha = 0.05$). Each of these combinations was used for each of the gear types, although only models using electrofishing data were able to run, so the best model chosen above is only using electrofishing data. The model chosen was the one with the lowest AIC. CPUE = Catch Per Unit Effort per minute or per net night. **model chosen with the lowest AIC.

APPENDIX G

MODEL COMBINATIONS USED TO DETERMINE IF CATCH PER UNIT EFFORT VARIED SIGNIFICANTLY AMONG SEASONS AND YEARS

Fixed Effect	**Gear**	**Random Effect**	**Interaction**	**AR1**	**AIC**	**P-Value**
**CPUE ~ Season	EF	Year	None	Yes	-48.76	<0.001
CPUE ~ Season	EF	None	None	No	-45.99	<0.001
**CPUE ~ Season	FN	Year	None	Yes	171.26	0.057
CPUE ~ Season	FN	None	None	No	221.79	<0.001
**CPUE ~ Season	MF	Year	None	Yes	207.33	0.015
CPUE ~ Season	MF	None	None	No	218.34	<0.001

Table G1. Using SIU CPUE electrofishing (EF), fyke net (FN), and mini fyke net (MF) data, model combinations of repeated measures using mixed models were used to see how CPUE changed over seasons for each gear (electrofishing, fyke nets, and mini fyke nets) in Buttonland Swamp within the Cache River watershed ($\alpha = 0.05$). The model chosen for each gear was the one with the lowest AIC. CPUE = Catch Per Unit Effort per minute or per net night. **model chosen with the lowest AIC.

Fixed Effect	**Gear**	**Random Effect**	**Interaction**	**AR1**	**AIC**	**P-Value**
**CPUE ~ Year	EF	Season	None	Yes	-49.31	0.637
CPUE ~ Year	EF	None	None	No	-40.29	0.290
**CPUE ~ Year	FN	Season	None	Yes	172.84	0.788
CPUE ~ Year	FN	None	None	No	230.45	0.427
**CPUE ~ Year	MF	Season	None	Yes	203.14	0.144
CPUE ~ Year	MF	None	None	No	223.83	0.035

Table G2. Using SIU CPUE electrofishing (EF), fyke net (FN), and mini fyke net (MF) data, model combinations of repeated measures using mixed models were used to see how CPUE changed over years for each gear (electrofishing, fyke nets, and mini fyke nets) in Buttonland Swamp within the Cache River watershed ($\alpha = 0.05$). The model chosen for each gear was the one with the lowest AIC. CPUE = Catch Per Unit Effort per minute or per net night. **model chosen with the lowest AIC.

APPENDIX H

MODEL COMBINATIONS USED TO DETERMINE IF CATCH PER UNIT EFFORT VARIED SIGNIFICANTLY AMONG MACROHABITATS IN THE WINTER

Fixed Effect	**Random Effect**	**Interaction**	**AR1**	**AIC**	**P-Value**
**CPUE ~ Macrohabitat	Year	None	Yes	29.70	<0.001
CPUE ~ Macrohabitat	None	None	No	28.00	<0.001
CPUE ~ Macrohabitat	None	Macrohabitat and Year	No	35.45	0.043

Table H1. Using SIU CPUE electrofishing data within the winter, model combinations of repeated measures using mixed models were used to see how CPUE changed over years among macrohabitats in Buttonland Swamp within the Cache River watershed ($\alpha = 0.05$). CPUE = Catch Per Unit Effort per minute. **model chosen with the lowest AIC.

APPENDIX I

MEAN LENGTH AT AGE OF BLUEGILL, GIZZARD SHAD, AND SILVER CARP

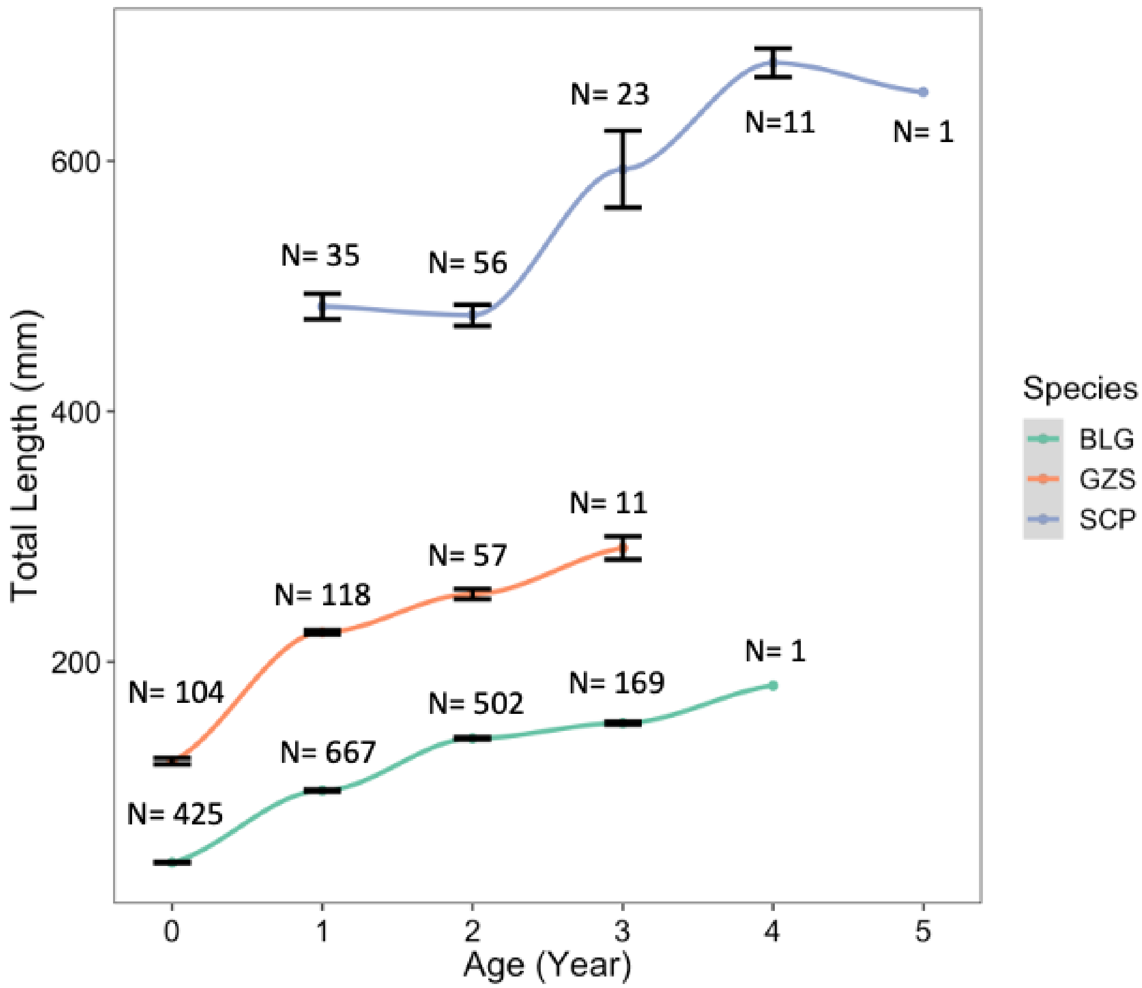

Figure I. Mean length at age (± standard deviation) of Bluegill, Gizzard Shad, and Silver Carp from Buttonland Swamp in 2020 and 2021.

APPENDIX J

MODEL COMBINATIONS OF WATER LEVEL AND YEAR-CLASS STRENGTH ASSOCIATIONS

Fixed Effect	Random Effect	AR1	Species (Year)	AIC	P-Value
YCS ~ Fall Water Level	none	No	BLG (2021)	9.80	<0.001
YCS ~ Fall Water Level	Year	Yes	BLG (2021)	11.80	<0.001
YCS ~ Fall Water Level	none	No	GZS (2020)	11.51	<0.001
YCS ~ Fall Water Level	Year	Yes	GZS (2020)	13.51	<0.001
YCS ~ Spawn Water Level	none	No	SCP (2021)	12.11	<0.001
YCS ~ Spawn Water Level	Year	Yes	SCP (2021)	14.11	<0.001

Table J1. Using SIU data, combinations of repeated measures using mixed models were used to see if there were any association of water level with year-class strength for Bluegill, Gizzard Shad, and Silver Carp in Buttonland Swamp within the Cache River watershed from 2020 - 2021($\alpha = 0.05$). Models included above are only the combination of models used for the models that were significant. The model chosen was the one with the lowest AIC. **model chosen with the lowest AIC for each model combination.

Fixed Effect	Random Effect	AR1	Species (Year)	AIC	P-Value
YCS ~ Spawn Water Level	none	No	GZS (2011)	25.49	0.005
YCS ~ Spawn Water Level	Year	Yes	GZS (2011)	27.49	0.005
YCS ~ Fall Water Level	none	No	BLG (2020)	14.29	<0.001
YCS ~ Fall Water Level	Year	Yes	BLG (2020)	16.29	<0.001

Table J2. Using IDNR data, combinations of repeated measures using mixed models were used to see if there were any association of water level with year-class strength for Bluegill, Gizzard Shad, and Silver Carp in Buttonland Swamp within the Cache River watershed from 2020 - 2021($\alpha = 0.05$). Models included above are only the combination of models used for the models that were significant. The model chosen was the one with the lowest AIC. **model chosen with the lowest AIC for each model combination.

APPENDIX K

MODEL COMBINATIONS OF AIR TEMPERATURE AND YEAR-CLASS STRENGTH ASSOCIATIONS

Fixed Effect	**Random Effect**	**AR1**	**Species (Year)**	**AIC**	**P-Value**
YCS ~ Yearly Air Temp	None	No	BLG (2021)	12.64	<0.001
**YCS ~ Yearly Air Temp	Year	Yes	BLG (2021)	14.64	<0.001
YCS ~ Yearly Air Temp	None	No	GZS (2020)	16.83	0.053
**YCS ~ Yearly Air Temp	Year	Yes	GZS (2020)	14.83	0.053
YCS ~ Pre-Spawn Air Temp	None	No	BLG (2020)	16.42	<0.001
YCS ~ Pre-Spawn Air Temp	Year	Yes	BLG (2020)	18.42	<0.001
YCS ~ Pre-Spawn Air Temp	None	No	BLG (2021)	16.25	<0.001
YCS ~ Pre-Spawn Air Temp	Year	Yes	BLG (2021)	18.25	<0.001
YCS ~ Pre-Spawn Air Temp	None	No	GZS (2020)	16.17	0.002
YCS ~ Pre-Spawn Air Temp	Year	Yes	GZS (2020)	18.17	0.002
YCS ~ Pre-Spawn Air Temp	None	No	SCP (2021)	22.12	0.012
YCS ~ Pre-Spawn Air Temp	Year	Yes	SCP (2021)	24.12	0.012
YCS ~ Spawn Air Temp	None	No	SCP (2021)	17.18	<0.001
YCS ~ Spawn Air Temp	Year	Yes	SCP (2021)	19.18	<0.001
YCS ~ Post-Spawn Air Temp	None	No	SCP (2021)	20.64	0.019
YCS ~ Post-Spawn Air Temp	Year	Yes	SCP (2021)	22.64	0.019
YCS ~ Fall Air Temp	None	No	GZS (2020)	13.47	0.001
YCS ~ Fall Air Temp	Year	Yes	GZS (2020)	15.47	0.001
YCS ~ Winter Severity	None	No	BLG (2021)	17.28	0.042
YCS ~ Winter Severity	Year	Yes	BLG (2021)	19.28	0.042
YCS ~ Winter Severity	None	No	GZS (2020)	13.55	<0.001
YCS ~ Winter Severity	Year	Yes	GZS (2020)	15.55	<0.001

Table K1. Using SIU data, combinations of repeated measures using mixed models were used to see if there were any association of air temperature with year-class strength for Bluegill, Gizzard Shad, and Silver Carp in Buttonland Swamp within the Cache River watershed from 2020 - 2021($\alpha = 0.05$). Models included above are only the combination of models used for the models that were significant. The model chosen was the one with the lowest AIC. **model chosen with the lowest AIC for each model combination.

Fixed Effect	**Random Effect**	**AR1**	**Species (Year)**	**AIC**	**P-Value**
YCS ~ Yearly Air Temp	None	No	BLG (2020)	12.54	<0.001
**YCS ~ Yearly Air Temp	Year	Yes	BLG (2020)	14.54	<0.001
YCS ~ Yearly Air Temp	None	No	BLG (2011)	19.15	0.024
**YCS ~ Yearly Air Temp	Year	Yes	BLG (2011)	21.15	0.024
YCS ~ Pre-Spawn Air Temp	None	No	BLG (2020)	16.04	<0.001
YCS ~ Pre-Spawn Air Temp	Year	Yes	BLG (2020)	18.04	<0.001
YCS ~ Spawn Air Temp	None	No	GZS (2019)	19.15	0.007
YCS ~ Spawn Air Temp	Year	Yes	GZS (2019)	21.15	0.007

Table K2. Using IDNR data, combinations of repeated measures using mixed models were used to see if there were any association of air temperature with year-class strength for Bluegill, Gizzard Shad, and Silver Carp in Buttonland Swamp within the Cache River watershed from 2020 - 2021($\alpha = 0.05$). Models included above are only the combination of models used for the models that were significant. The model chosen was the one with the lowest AIC. **model chosen with the lowest AIC for each model combination.

APPENDIX L

MODEL COMBINATIONS OF PREDATOR/COMPETITOR CATCH PER UNIT EFFORT AND YEAR-CLASS STRENGTH ASSOCIATIONS

Fixed Effect	**Random Effect**	**AR1**	**Species (Year)**	**AIC**	**P-Value**
YCS ~ BLG CPUE	none	No	BLG (2020)	5.61	0.008
**YCS ~ BLG CPUE	Year	Yes	BLG (2020)	7.61	0.008
YCS ~ BLG CPUE	none	No	BLG (2021)	5.53	0.024
**YCS ~ BLG CPUE	Year	Yes	BLG (2021)	7.53	0.024
YCS ~ LMB CPUE	none	No	BLG (2020)	3.03	<0.001
YCS ~ LMB CPUE	Year	Yes	BLG (2020)	5.03	<0.001
YCS ~ LMB CPUE	none	No	BLG (2021)	3.40	<0.001
YCS ~ LMB CPUE	Year	Yes	BLG (2021)	5.40	<0.001
YCS ~ WAM CPUE	none	No	BLG (2020)	5.08	0.041
YCS ~ WAM CPUE	Year	Yes	BLG (2020)	7.08	0.041
YCS ~ WAM CPUE	none	No	BLG (2021)	4.47	0.017
YCS ~ WAM CPUE	Year	Yes	BLG (2021)	6.47	0.017
YCS ~ WAM CPUE	none	No	GZS (2020)	0.13	<0.001
YCS ~ WAM CPUE	Year	Yes	GZS (2020)	2.13	<0.001
YCS ~ CAP CPUE	none	No	BLG (2020)	-2.21	<0.001
YCS ~ CAP CPUE	Year	Yes	BLG (2020)	-0.21	<0.001
YCS ~ CAP CPUE	none	No	BLG (2021)	0.39	<0.001
YCS ~ CAP CPUE	Year	Yes	BLG (2021)	2.39	<0.001
YCS ~ WHC CPUE	none	No	BLG (2020)	0.68	<0.001
YCS ~ WHC CPUE	Year	Yes	BLG (2020)	2.68	<0.001
YCS ~ WHC CPUE	none	No	BLG (2021)	0.70	0.002
YCS ~ WHC CPUE	Year	Yes	BLG (2021)	2.70	0.002
YCS ~ WHC CPUE	none	No	SCP (2021)	3.84	<0.001
YCS ~ WHC CPUE	Year	Yes	SCP (2021)	5.84	<0.001
YCS ~ RSF CPUE	none	No	SCP (2021)	1.23	<0.001
YCS ~ RSF CPUE	Year	Yes	SCP (2021)	3.23	<0.001
YCS ~ GZS CPUE	none	No	SCP (2021)	5.80	<0.001

YCS ~ GZS CPUE	Year	Yes	SCP (2021)	7.80	<0.001
YCS ~ SCP CPUE	none	No	BLG (2020)	6.79	<0.001
YCS ~ SCP CPUE	Year	Yes	BLG (2020)	8.79	<0.001
YCS ~ SCP CPUE	none	No	BLG (2021)	6.80	0.003
YCS ~ SCP CPUE	Year	Yes	BLG (2021)	8.80	0.003

Table L1. Using SIU data, combinations of repeated measures using mixed models were used to see if there were any association of predator/competitor Catch per Unit Effort per minute (CPUE) with year-class strength for Bluegill, Gizzard Shad, and Silver Carp in Buttonland Swamp within the Cache River watershed from 2020 - 2021($\alpha = 0.05$). Models included above are only the combination of models used for the models that were significant.

Fixed Effect	Random Effect	AR1	Species (Year)	AIC	P-Value
YCS ~ BLG CPUE	none	No	BLG (2020)	4.66	<0.001
**YCS ~ BLG CPUE	Year	Yes	BLG (2020)	6.66	<0.001
YCS ~ LMB CPUE	none	No	BLG (2020)	-0.06	<0.001
YCS ~ LMB CPUE	Year	Yes	BLG (2020)	1.94	<0.001
YCS ~ LMB CPUE	none	No	GZS (2019)	1.09	<0.001
YCS ~ LMB CPUE	Year	Yes	GZS (2019)	3.09	<0.001
YCS ~ SPG CPUE	none	No	BLG (2020)	3.76	0.034
YCS ~ SPG CPUE	Year	Yes	BLG (2020)	5.76	0.034
YCS ~ BLC CPUE	none	No	BLG (2020)	5.15	0.034
YCS ~ BLC CPUE	Year	Yes	BLG (2020)	7.15	0.034
YCS ~ BLC CPUE	none	No	GZS (2019)	2.07	<0.001
YCS ~ BLC CPUE	Year	Yes	GZS (2019)	4.07	<0.001
YCS ~ CAP CPUE	none	No	BLG (2020)	2.09	<0.001
YCS ~ CAP CPUE	Year	Yes	BLG (2020)	4.09	<0.001
YCS ~ WHC CPUE	none	No	BLG (2020)	-0.95	<0.001
YCS ~ WHC CPUE	Year	Yes	BLG (2020)	1.05	<0.001
YCS ~ GZS CPUE	none	No	GZS (2011)	5.51	<0.001
YCS ~ GZS CPUE	Year	Yes	GZS (2011)	7.51	<0.001
YCS ~ SCP CPUE	none	No	BLG (2020)	5.30	<0.001
YCS ~ SCP CPUE	Year	Yes	BLG (2020)	7.30	<0.001

Table L2. Using IDNR data, combinations of repeated measures using mixed models were used to see if there were any association of predator/competitor Catch per Unit Effort per minute (CPUE) with year-class strength for Bluegill, Gizzard Shad, and Silver Carp in Buttonland Swamp within the Cache River watershed from 2020 - 2021($\alpha = 0.05$). Models included above are only the combination of models used for the models that were significant.

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Use of Assemblage Structure, Abundance, and Year-Class Strength to Assess Fish Assemblage Responses to Water Level Fluctuations

Major Professor: Gregory W. Whitledge