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Fish communities respond to hydrology and elevation in restored western Kentucky wetlands

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Fish communities respond to hydrology and elevation in restored western Kentucky wetlands

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Abstract

The Wetlands Reserve Program (WRP) is designed to restore wetland ecological function and wildlife benefits; however, these projects rarely monitor biological responses. The objectives of our study were to (1) identify environmental factors that were the most important in determining wetland fish community composition, (2) examine the influence of the environment on specific groups of fish (large-river fishes and Kentucky Species of Greatest Conservation Need (KY-SGCN) wetland fishes), and (3) compare fish community composition and diversity in wetlands of different condition (e.g., degraded, restored, and reference). From April 2019 to August 2020, monthly electrofishing surveys were performed in degraded (n=2), restored (n=8), and reference (n=2) wetlands throughout western Kentucky, USA. Over 12,500 individual fish accounting for 53 species were collected. Non-metric multidimensional scaling was used to visualize fish community composition and vector analysis was used to determine which environmental factors most affected fish community composition. Our findings suggest that fish community composition was most affected by environmental factors that were related to the influence of the Mississippi River. Vector analysis identified that large-river fishes displayed a strong positive association with wetlands that were more hydrologically influenced by the Mississippi River, whereas KY-SGCN wetland fishes displayed a strong positive association with wetlands less hydrologically influenced by the Mississippi River. ANOSIM and Hill diversity (Hill-Shannon; q=1) were used to quantify differences in fish community composition and diversity among wetland condition. Furthermore, our results indicate western Kentucky WRP restorations have not created unique community composition but have rapidly fostered levels of fish diversity similar to reference wetlands. Current and future WRP restoration managers may need to consider potential tradeoffs between wetland wildlife communities and wetland function to best promote restoration goals.

Introduction

Wetlands are crucially important ecosystems for both humans and wildlife. Wetlands provide humans with numerous ecosystem services that are valued at approximately 35 trillion USD a year (Costanza et al. 2014). Wetlands are critical for wildlife because they are productive and habitat-rich ecosystems that foster the existence of diverse assemblages of biota (Mitsch & Gosselink 2015). Despite their value to humans and wildlife, wetlands have suffered large-scale global losses (Dahl & Allord 1996; Mitsch & Gosselink 2015). Many of the United States' non-coastal wetland resources exist as floodplain wetlands throughout the Mississippi River Alluvial Valley (MAV). Once supporting nearly 10 million ha of bottomland hardwood forest, the MAV has experienced radical alterations to its regional hydrology and large-scale land use conversion to agriculture, which has led to a dramatic loss of wetlands, wildlife benefits, and ecosystem function (Semlitsch 2000; Frederickson 2005; Rewa 2005; King et al. 2006; Faulkner et al. 2011; Kleunder et al. 2015; USDA-NASS 2017).

In response to wetland loss throughout the United States, the Wetlands Reserve Program (WRP), by 2013, had restored over 1 million ha of wetlands, of which approximately 250,000 ha were located throughout the MAV (Natural Resources Conservation Service 2013a). WRP focuses on using a combination of reforestation and hydrologic restoration techniques to foster ecosystem processes that allow wetland restoration (King & Keeland 1999; Hayes & Egan 2004; Rewa 2005). Special emphasis is placed on restoring hydrology. Hydrology has been found to be the critical driver of wetland processes that, when restored, most quickly returns wetland function and wildlife benefits (Bedford 1996; Brinson & Rheinhardt 1998; Rewa 2000; Zedler 2000; Haynes & Egan 2004; Rewa 2005; Brauman et al. 2007; Hunter et al. 2008; Faulkner et al. 2011; King & Keim 2019). Through wetland restoration, WRP aims to return lost wetland function and wildlife benefits (Natural Resources Conservation Service 2013a).

To assess wetland restoration success, WRP restoration practitioners employ a variety of post-restoration monitoring studies (see Osmond et al. 2012). Typically, these studies focus directly on the response of wetland function, water quality, or use vegetation response as a proxy for wetland function (Rewa 2005; King et al. 2006; Faulkner et al. 2011). Post-restoration monitoring efforts, however, are sometimes misleading as not all restorations are monitored and restoration success criteria are not always clearly defined (Zedler 2000; Stanturf et al. 2001;

Ruiz-Jaen & Aide 2005; USDA OIG 2008; Moreno-Mateos et al. 2012). Additional problems may arise if only wetland function is considered because wetland function and wildlife usage have not always been found to be maximized at the same wetland (Zedler 2000). Even though one goal of WRP is to return lost wildlife benefits, relatively few studies exist that quantify the response of wildlife communities (Rewa 2005). Considering the goals of the WRP, wildlife community responses should also be measured post-restoration.

The utility of evaluating post-restoration success using biological indicator species such as Species of Greatest Conservation Need (SGCN) have been published (Benson et al. 2018). Biological indicator species provide information on the functionality of a particular ecosystem, as species within an assemblage vary in their environmental requirements and their sensitivity to disturbance. To our knowledge, a paucity of studies exist that utilize fish community response as a measurement of WRP post-restoration success (but see Leao 2005; Rewa 2005; Benson et al. 2018). The use of fish community response to WRP restorations throughout the MAV is well-founded because hydrology is a major determinant of wetland function and fish community structure. MAV wetland function and fishes depend on a natural flood-pulse cycle where water, sediments, and nutrients are supplied to wetlands through seasonal connections to the floodplain (Junk et al. 1989; Faulkner & Patrick 1992). For both large-river and SGCN wetland fishes, seasonal connections to floodplain wetlands determines the diversity, community composition, and overall success of these fishes (Welcomme 1985; Junk et al. 1989; Caldwell et al. 2011; Zeug et al. 2005) by providing abundant energy resources (e.g., zooplankton) and granting suitable spawning habitat (Welcomme 1985; Winemiller and Rose 1992, King & Keeland 1999; Baber et al. 2002; Kluender et al. 2015). Hydrology and water quality in MAV floodplain wetlands, however, can differ greatly among seasons as wetlands are more influenced by the river channel in the winter and spring and more influenced by local climatic events during the summer and fall (Junk et al. 1989; Mitsch & Gosselink 2015). Individual species of fish display different responses to specific wetland environmental conditions (Jester et al. 1992; Miranda & Lucas 2005; Dembkowski & Miranda 2012). For example, large-river fishes depend on seasonal access to floodplain wetlands during flooding events while SGCN wetland fishes utilize the floodplain's shallow depths, sluggish flow, dense vegetation, and soft substrate year-round (Welcomme 1985; Junk et al. 1989; Petts 1989; Aarts et al. 2004; Hohausova et al. 2010; Beesley et al. 2014; Kluender et al. 2015; Eisenhour et al. 2018; Simpson et al. 2021). Differing responses of fishes, therefore, underline the importance to quantify the response of different groups of taxa (Benson et al. 2018).

Due to the suitability of utilizing fishes as biological indicators and to address knowledge gaps related to fish usage of WRP restored floodplain wetlands in the MAV, our study examined fish response to environmental conditions, i.e., hydrology, water quality, and biota in restored wetlands. The objectives of this research were to (1) determine the relationships between fish community composition and environmental conditions in western Kentucky wetlands, to (2) determine the relationships between specific groups of fish (large-river fishes and Kentucky SGCN (KY-SGCN) wetland fishes) and environmental conditions, and to (3) examine if differences in wetland condition (e.g., degraded, restored, and reference) led to differences in fish community structure. It was thought that wetland hydrology would be most important in determining fish community composition, that the relative abundance of large-river fishes and KY-SGCN wetland fishes would be determined by differences in hydrology, that fish community composition in WRP restored wetlands would differ from those in reference and degraded wetlands, and that WRP fish diversity would be intermediate between that of degraded and reference wetlands. Hopefully, the fish/environment relationships found in our wetlands can be used to inform future wetland restoration management decisions and ensure the most effective management decisions (Merkle et al. 2019).

Methods

Study Area

Our study was conducted in the Mississippi Alluvial Plain and Mississippi Valley Loess Plain ecoregions of western Kentucky (Omerink 1987). Historically, wetland resources in this region were characterized by bottomland hardwood forests and stream floodplains that experienced dramatic but predictable hydroperiods seasonally influenced by the upper Mississippi River watershed (King et al. 2006; Mitsch & Gosselink 2015). A substantial amount of wetland resources in this region still exist and are located along three major tributaries (Mayfield Creek, Obion Creek, and Bayou du Chien Creek) that drain approximately 2,800 km^2 into the Mississippi River. These wetlands are highly fragmented and suffer from degradation from intensive agriculture and regional hydrologic modifications (Frederickson 2005, King et al. 2006). Current land use of this region of western Kentucky is dominated by cultivated crops (64%) and forested floodplain wetlands (25%) (Dewitz 2019). Despite these anthropogenic influences, the region retains some features of a large-river floodplain; for example, seasonally high discharges reconnect the floodplain in the winter and spring (Mitsch & Gosselink 2015).

Wetland Selection

Eight wetlands restored by WRP in western Kentucky (Figure 1) were sampled. Restored wetlands ranged in size from one to 20 ha and in age since hydrologic restoration from one to 13 years. WRP employed a variety of engineering techniques to restore local hydrology on the selected wetlands (Personal communication, NRCS). Additionally, some restored wetlands were planted with tree saplings to initiate reforestation and currently managed following Compatible User Agreements, which include food plot planting, mowing, and water level management (Personal communication, NRCS). Before wetland selection, pertinent WRP information from the National Resources Conservation Service (NRCS) (including landowner contact, restoration type, restoration age, restoration plans) was obtained. After obtaining restored wetland information, the following criteria were used to select restored wetlands: (1) location on one of the regional tributaries to the Mississippi River (Figure 1) and (2) similarity of hydrogeomorphic wetland class (riverine following Brinson et al. 1993), hydroperiod (semi-permanent to permanent following Cowardin et al. 1995) (Figure 2), and size (1-20 ha).

Poor condition (degraded wetlands; n=2) and good condition (reference wetlands; n=2) wetlands were also sampled to compare with WRP restored wetlands (Figure 1). Degraded wetlands were once natural wetlands that have experienced dramatic local hydrologic alterations for agricultural purposes, but still exhibit some wetland characteristics. Both of our degraded wetlands were positioned in active agricultural fields. Reference wetlands have not been subjected to local hydrologic alterations, but ultimately still exist within a highly altered landscape. One of our reference sites was a forested wetland located on a KY Wildlife Management Area; the other site was a bottomland hardwood swamp positioned on an upstream portion of one of our WRP easements.

Fish sampling

From April 2019 to August 2020, monthly electrofishing surveys were conducted at each wetland. A 24-volt battery powered Smith-Root LR-24 backpack electrofisher was used to collect fish from accessible wetland shoreline. Each site was sampled for 600-750 seconds at 400-500 volts and at 30 Hz on a 25% duty cycle. Collected fishes were measured, identified to species using Etnier & Starnes (1993) and Pflieger (1997), enumerated, and then returned. Specimens that could not be identified in the field were anesthetized with clove oil, preserved in a 10% formalin solution, and later identified. For each sample, the abundance of each fish species was divided by the effort (minutes) to determine catch per unit effort (CPUE) (individuals per minute). CPUE was then averaged across all monthly sampling events to yield a single value representing the CPUE of a specific fish species for each wetland.

Environmental influences on fishes

Twenty-eight environmental predictor variables (see below for explanations; Table 1) were quantified to examine the influence on fish community composition and the relative abundance of large-river and KY-SGCN wetland fishes. Large-river fishes were classified using species descriptions from Etnier and Starnes (1993) and consisted of *Cycleptus elongatus* (blue sucker), *Ictalurus punctatus* (channel catfish), *Aplodinotus grunniens* (freshwater drum), *Dorosoma cepedianum* (gizzard shad), *Ictiobus bubalus* (smallmouth buffalo), *Sander canadensis* (sauger). KY-SGCN wetland fishes were classified using Kentucky's Comprehensive Wildlife Conservation Strategy report (2013) and consisted of *Umbra limi* (central mudminnow), *Hybognathus hayi (cypress minnow)*, *Lepomis marginatus* (dollar sunfish), *Fundulus chrysotus* (golden topminnow), *Erimyzon sucetta* (lake chubsucker), *Lepomis miniatus* (redspotted sunfish), and *Notropis maculatus* (taillight shiner).

Hydrology

Depth was recorded in each wetland from March 2019 to August 2020 using water level loggers (HOBO® U20-001-04, Onset Computer Corporation). One logger was deployed in the deepest wadeable location of each wetland. Depth was recorded every 15 minutes. Water level data was averaged per day and daily depths were then used to calculate mean depth (m), hydroperiod (days), and 13 indicators of hydrologic alteration (IHA) following Richter et al. (1996) (Table 1). The percent of time each wetland experienced connectivity to its associated stream was estimated by determining the stream depth at which each wetland was connected to the stream and then by calculating the percent of time the stream was greater than that depth. For wetlands on upstream portions of tributaries, stream depth was determined by using the United States Geological Survey stream gauge 07024000 (USGS 2019-2020) on the Bayou du Chien and for wetlands located on the Mississippi River floodplain stream depth was determined by using the United States Geological Survey stream gauge 07022000 (USGS 2019-2020) on the Mississippi River.

Hydrologic connectivity

Four landscape variables were calculated to indirectly quantify hydrologic connectivity between wetlands and other permanent bodies of water. Distance to Main Channel was quantified by measuring the straight-line distance from the center of each wetland to the main channel of the nearest major stream in ArcGIS Pro (Version 2.7, Esri Inc.). Topography was calculated by using 10-m U.S. Geological Survey (2017) 1/3 arc-second Digital Elevation Models (DEMs) to find the mean slope inside a 1-km buffer around each wetland. The “elevation profile” tool in USGS Stream Stats was used to delineate each wetland’s boundary and assign elevation (m) values for 50 different locations within each wetland. Elevation was calculated by taking the median of each wetland’s 50 elevation values. Waterway Distance to the Mississippi River was calculated by using the “flow-path” tool in USGS Stream Stats to find the shortest waterway distance (km) from each wetland to the main channel of the Mississippi River. Waterway distance was considered as a proxy for the hydrologic influence the Mississippi River exerted on each wetland; greater values imply lower hydrologic influence.

Water quality

Changes in water temperature (°C) and dissolved oxygen (DO) (mg/L) were recorded in each wetland from March 2019 to August 2020 using multi-parameter sondes (YSI® EXO2, Xylem Incorporated). One sonde was suspended in each wetland approximately midway in the water column in the deepest wadeable location. Sondes recorded data at 15-minute intervals. Measurements were averaged per day and then used to calculate minimum temperature, maximum temperature, mean temperature, mean DO, and minimum DO for each wetland.

Biotic variables

Zooplankton communities were sampled monthly from April 2019 through August 2020 at each wetland using a 9-cm diameter littoral sampling tube following Pennak (1962). Each sample consisted of three replicates averaged together, one taken from open water, wetland edge, and dense vegetation. Samples were poured into a volumetric container, volume (L) was recorded, rinsed through a 43-µm sieve, and preserved in 4% buffered formalin solution. Later, samples were subsampled to a maximum of 1/8 using a Folsom Plankton Splitter (WILDCO, Saginaw, MI) so that at least 50 zooplankters were found per subsample. Cladocerans and copepods were enumerated but rotifers were excluded. After scaling back up to 100% from the subsampled fraction, density was calculated by dividing the abundance of each sample by the original volume of water.

Aquatic vegetation was sampled monthly at each site between April 2020 and August 2020, which corresponded with the region's growing season, using a 1m-by-1m quadrat. Each sample consisted of nine replicates averaged together: three each taken from open water, wetland edge, and dense vegetation. Percent cover of three aquatic vegetation groups (aquatic emergent, aquatic submergent, and aquatic floating) was estimated using six cover categories (1 = 0-10%, 2= 11-20%, 3 = 21-40%, 4 = 41-60%, 5= 61-80%, 6 = 81-100%). The midpoint of the cover categories was used for percent cover calculations.

The Kentucky Wetland Rapid Assessment Method (KY-WRAM) (Kentucky Division of Water, 2016) was used to assess general wetland integrity and function. The KY-WRAM is used to evaluate overall wetland function and ecological integrity and was developed for use by regulatory agencies in wetland permitting decisions. The KY-WRAM assumes that wetlands with high ecological integrity also have high wetland function. The KY-WRAM metrics capture a full range of potential disturbances to wetland integrity and fall into six categories: size and distribution; buffer and intensity of surrounding land uses; hydrology; habitat alteration and habitat structure development; special wetlands; and vegetation, interspersion, and habitat features. Each category is subdivided into additional metrics. The special wetlands category was omitted because we were unable to gain access to the required information (all wetlands received a score of 0 for this category). The final KY-WRAM score is the sum of all the metric scores and ranges from 0 (very poor condition) to 90 (reference condition). Out of the metrics, 10 were evaluated using ArcGIS Pro® software (Version 2.7, Esri Inc.) and high resolution orthoimages (USDA 2018), and 10 were assessed in the field during a one-hour site visit per wetland during July - September 2020.

Statistical Analyses

Statistical analyses were performed using R statistical software (version 4.0.5) (R core team 2021) and the ‘Vegan’ software package (Oskansen et al. 2013). Nonmetric multidimensional scaling (NMDS) was used to visualize variation in fish community composition among wetland condition. Experimental units in ordinations were defined as the average CPUE of each fish species present at each wetland in multivariate space. No fish species were removed prior to ordination because we were interested in rare and uncommon taxa. Ordinations were constructed with Bray–Curtis distances, run for the maximum number of iterations (300), and chosen for minimum stress values. Dimensional solutions, stepping down from six to one, were tested and determined by the use and examination of individual scree plots (McCune & Grace 2002).

Analysis of similarity (ANOSIM) was used to test for differences in fish community composition among *a priori* groups of wetland condition. The Gower distance was used in ANOSIM and run for 9999 maximum iterations generating r-values between -1 and 1. Positive values indicate differences among groups and significance was tested at $\alpha = 0.05$.

The influence of environmental predictor variables (Table 1) on fish community composition, large-river fishes, and KY-SGCN wetland fishes was examined using vector fitting analysis with the function ‘ENVFIT’ on NMDS ordinations. Each variable’s association with each experimental unit was indicated by the vector’s direction while the strength of each variable’s association was indicated by the vector’s length (McCune & Grace 2002). Vector significance ($\alpha = 0.05$) was estimated using 999 random permutations of the data (Faith & Norris 1989). For ease of interpretation, the hydrologic variables 1-Day Maximum (m), 7-Day Maximum (m), 30-Day Maximum (m), and 90-Day Maximum were grouped into one vector labeled High Water Magnitude on ordination plots because of high correlation (overlapping angle and vector magnitude) among variables (Flinn et al. 2008).

Vectors representing the relative abundance of large-river and KY-SGCN fishes were also placed onto NMDS ordinations. Vectors representing these groups of fishes were intended to help explain their association with the environment and were not used to predict fish community composition.

Diversity

Hill diversity was calculated for each wetland based on condition. Hill diversity was chosen due to its numerous advantages over other diversity indices (see Chao et al. 2014) and was calculated according to Hsieh et al. (2016), and Chiu & Chao (2014). See Jost (2006) for a more thorough review of the benefits of Hill diversity in relation to other indices or Roswell et al. (2021) for a current consensus within the ecological community.

Before calculating Hill diversity, samples were standardized by 95% coverage to account for uneven sampling effort (Chao & Jost 2012; Chao et al. 2014; Roswell et al. 2021; R package iNEXT). Coverage is a relatively new method of sample standardization in ecology that measures sample completeness and accounts for the abundance of species in the sampled community. Coverage estimates the proportion of individuals in the community that belong to species present in a sample (Roswell et al. 2021). For example, achieving coverage of 95% means that 5% of individuals in the community were not sampled.

Hill diversity varies based on the choice of exponent used. Ecologists most commonly use $q = 0$ (species richness), $q = 1$ (Hill-Shannon diversity), and $q = 2$ (Hill-Simpson diversity). Hill-Shannon diversity ($q = 1$) was used because it results in all individuals being considered equally as it counts species proportionately to their abundance or incidence (Chao et al. 2014). Hill-Shannon diversity was calculated for each wetland using the iNEXT package which provides functions to compute the most widely used Hill numbers ($q = 1$, $q = 2$, $q = 3$) for individual-based abundance data or sampling-unit based incidence data. Incidence data was used because it suitably represents timed surveys, e.g., timed electrofishing surveys, and because Colwell et al. (2012), Chao et al. (2014), and Chao & Colwell (2017) demonstrated that incidence data allows for biological inference just as powerful as abundance-based approaches.

Our input data for the iNEXT package consisted of species-specific incidence data from each sample from each wetland which was categorized by each sample's wetland condition.

To describe patterns in Hill-Shannon diversity, sample and coverage-based rarefaction and extrapolation curves were generated using the "estimate d" function (R package, iNEXT) to determine how diversity increases with increasing sampling effort and completeness. Rarefaction and extrapolation of Hill-Shannon diversity were conducted according to Hsieh et al. (2016) and further discussed by Colwell et al. (2012), Chao & Jost (2012), and Chao et al. (2014). Sample-based curves evaluated the number of individuals in a sample by plotting diversity estimates in relation to the number of sampling units. Coverage-based curves were plotted against rarefied sample completeness to illustrate diversity estimates in relation to sample coverage. All extrapolation curves were plotted using a doubling in sample size and 500 bootstrap replicates were used to estimate 95% confidence intervals. Confidence intervals, a known alternative to standard statistical testing (Magurran 1988; Colwell, Mao, & Chang 2004), were used to determine if differences between wetland condition were statistically significant. Nonoverlapping 95% confidence intervals, associated with rarefied or extrapolated curves, indicate possible significant differences at $\alpha = 0.05$ (Chao & Jost 2012; Chao et al. 2014).

Results

Fish Sampling

All wetlands had fish present. 12,518 fish from 17 families, 37 genera, and 53 species were collected across all wetlands. The mean CPUE (individuals per minute) in restored wetlands was 10.95 ± 1.91 (SE), 7.41 ± 1.94 (SE) in degraded wetlands, and 4.33 ± 0.42 (SE) in reference wetlands. Degraded wetlands had 22 of the 53 recorded species, 47 species were found in restored wetlands, and 38 species were found in reference wetlands. Golden topminnow was only found in degraded wetlands, blacktail shiner (*Cyprinella venusta*), blue sucker, channel catfish, common carp (*Cyprinus carpio*), freshwater drum, gizzard shad, grass carp (*Ctenopharyngodon idella*), logperch (*Percina caprodes*), silver carp (*Hypophthalmichthys molitrix*), sauger, and yellow bass (*Morone mississippiensis*) were only found in restored wetlands, and bullhead minnow (*Pimephales vigilax*), pugnose minnow (*Opsopoeodus emiliae*), red shiner (*Cyprinella lutrensis*), and taillight shiner were only found in reference wetlands. Overall community composition of all wetlands was dominated by Centrarchidae (sunfishes) (54% of all individuals collected) and Poeciliidae (livebearers) (18%). Across all wetlands, the relative abundance of large-river fishes was 1.7% of all individuals collected whereas the relative abundance of KY-SGCN wetland fish was 3.7%.

Environmental influences on fish community composition

Final NMDS solutions consisted of two dimensions with a low final stress value (0.102) and high interpretability compared to alternative solutions (Figure 3). Eight environmental variables were found to be significant in determining fish community composition and were overlaid as vectors onto NMDS ordination (Figure 3, Table 2). Environmental vectors displayed a strong horizontal gradient: increasing Waterway Distance to the Mississippi River and Zooplankton Density were associated with each other and were negatively associated with increasing High Water Magnitude, Duration of Connectivity, and Low Water Duration. The horizontal gradient, at least in part, helped explain the fish community composition of many of our wetlands based on condition. Environmental vectors did not describe a vertical gradient well and fish community composition of wetland condition, in many cases, had at least some vertical aspect to it. Rise Count was an exception to this pattern as it lied in between vertical and horizontal gradients: increasing Rise Count was more positively associated with increasing

Zooplankton Density and Waterway Distance to the Mississippi River and more negatively associated with High Water Magnitude, Duration of Connectivity, and Low Water Duration. Positive and negative associations existed between Rise Count and fish community composition in many of our wetlands based on condition.

Environmental influences on river and KY-SGCN wetland fishes

The environmental variables High Water Magnitude, Duration of Connectivity, and Low Water Duration had strong positive association with the relative abundance of large-river fishes (Figure 3). Rise Count had strong positive association with the relative abundance of KY-SGCN wetland fishes (Figure 3). Zooplankton Density and Waterway Distance to the Mississippi River had at least some positive association with the relative abundance of KY-SGCN wetland fishes (Figure 3). The vectors representing relative abundance of large-river fishes and relative abundance of KY-SGCN wetland fishes had opposing relationships to one another (Figure 3).

Influence of wetland condition on fish community composition

NMDS ordination based on electrofishing CPUE data revealed little separation of fish community composition by wetland condition (Figure 3). Results of ANOSIM analysis confirmed that degraded, restored, and reference wetland fish community composition were not significantly different (r = -0.182, p = 0.732).

Influence of wetland condition on fish diversity

Hill-Shannon diversity in restored wetlands was 28.6, 95% CI [27.7, 29.9] which was not significantly different from Hill-Shannon diversity in reference wetlands 26.6, 95 % CI [24.9, 28.7] (Figures 4, 5). Hill-Shannon diversity in restored and reference wetlands were significantly greater than Hill-Shannon diversity in degraded wetlands 18.1, 95% CI [17.4, 19.6] (Figures 4, 5). The estimated curve patterns of Hill-Shannon diversity accumulation per sampling unit for degraded, restored, and reference wetlands were approaching asymptotic (plateau), suggesting that the sampling strategy was sufficient in revealing true patterns of fish diversity associated with these three wetland types. Coverage-based rarefaction and extrapolation further indicated that sample completeness was consistent across wetland type as coverage values were all greater than 95% (Figure 5).

Discussion

Our results indicated that hydrologic influence from the Mississippi River was more important in determining fish community composition than wetland condition as differences in hydrologic influence from the Mississippi River determined the relative abundances of large-river and KY-SGCN wetland fishes. Hydrologic influence from the Mississippi River led to differences in hydrology between wetlands with short and long waterway distances to the river. Wetlands with short waterway distances were strongly influenced by the river as they experienced direct lateral connectivity with the Mississippi River during its high magnitude depth long-lasting spring flood pulse but seldom reconnected with the river after the spring. Wetlands with long waterway distances were less influenced by the river as they did not experience direct lateral connectivity with the Mississippi River and, therefore, experienced lower magnitude depth shorter-lasting flooding and were more likely to flood after the spring pulse. Even though restored wetlands did not have unique community composition, restored wetlands had levels of diversity greater than those of degraded wetlands and was comparable to reference wetlands. With these results our study has demonstrated that wetland restoration may promote fish diversity and hydrologic influence is an important factor to consider regarding specific groups of fishes.

<u>Influence of the Mississippi River</u>

Our study suggests that each wetland's waterway distance to the Mississippi River determined the influence of the river's hydrology: differences in influence created contrasting hydrologic conditions in short and long waterway distance wetlands (see above) (Figure 3). Contrasting differences in hydrology most likely determined the relative abundance of large-river and KY-SGCN wetland fishes and led to the contrasting associations between the two fish groups. The hydrologic influence from the Mississippi River in wetlands with short waterway distances was exacerbated due to the exceptional winter and spring flooding of 2019 and 2020. According to the National Weather Service's (NWS) river gage at Cairo, IL, the Mississippi River stayed above flood stage (40 ft) for 146 consecutive days from February 8th, 2019 to July 13th, 2019 and, during this time, it reached its third greatest height ever recorded (56.5 ft). Although not as extreme as 2019, the Mississippi River still experienced exceptional flooding in 2020 as the NWS Cairo, IL gage reported 96 consecutive days above flood stage from February 8th, 2020 to May 12th, 2020, during which, the river reached its 21st greatest height ever recorded (52.6 ft). Mississippi River flood events, like these, may become more likely as winter and spring precipitation throughout the upper Mississippi River watershed is expected to increase 20-30 % by the year 2100 (Wuebbles & Hayhoe 2004). The association between zooplankton density and waterway distance to the Mississippi River (Figure 3) help support this claim, as previous studies have found that greater riverine influence decrease water residence times and maximize dilution effects leading to lessened zooplankton densities (Pace et al. 1992; Bozelli et al. 2015; Godfrey et al. 2020).

Influence of the Mississippi River may have also been responsible for the long periods of low water in wetlands with short waterway distances to the river (Figure 3). Watershed size determines how local water levels respond to precipitation (USGS Watershed Science School 2019). Precipitation in a small watershed can drastically increase stream levels and induce overbank flooding. The opposite is true of the Mississippi River as local water levels are determined by precipitation that has occurred upstream and throughout its entire watershed. Once water levels dropped in the Mississippi River after spring flooding, local precipitation was unlikely to increase water levels enough to reconnect short waterway distance wetlands to the river. Conversely, precipitation after spring flooding allowed wetlands further from the Mississippi River to periodically receive stream inputs and maintain water levels throughout the year. Water levels of the Mississippi River may have also affected the floodplain's water table and affected drying. Flood-stage river levels in high order streams have been found to impede floodplain water drainage (Berkowitz et al. 2020). After river levels dropped, the floodplain's water table may have begun to rapidly drain leading to eventual drying in wetlands with short waterway distances.

Environmental influences on large-river and KY-SGCN wetland fishes

Depth is a well-studied environmental factor that is important in structuring fish communities (Rodriguez & Lewis Jr. 1997). Generally, when increased, depth has been found to be beneficial for fishes as it provides habitat heterogeneity, environmental stability, and refugia from poor water quality (Zeug et al. 2005; Shoup & Wahl 2009; Miranda 2011; Dembkowski & Miranda 2012). For example, deeper depths increase habitat suitability for large river fishes (i.e., access to the floodplain) and serve as a trigger for reproduction (Welcomme 1985; Copp & Penaz 1988; Junk et al. 1989; Winemiller & Rose 1992; Beesley et al. 2014, Kluender et al. 2015). Benefits provided to fishes by increasing depth may reach a threshold, however, and depths exceeding that threshold may become detrimental to some fishes. High magnitude depth can act as a measure of environmental stress for wetland fishes by decreasing floodplain habitat suitability, i.e., creating lotic conditions or greatly inundating shallow littoral areas (Resh et al. 1988; Richter et al. 1996). When decreased, depth can limit habitat heterogeneity (Dembkowski & Miranda 2012), can impose foraging limitations on fishes (Thomasz et al. 1997), increase chance of predation, lead to poor water quality (Zeug et al. 2005), and increase likelihood of drying, all of which may lead to depauperate fish assemblages or cause die-offs (Zeug et al. 2005; Shoup & Wahl 2009; Miranda 2011; Dembkowski & Miranda 2012).

Our study observed contrasting associations that existed between different groups of fishes and wetland depth (Figure 3). Large-river fishes benefited from high magnitude depths as it granted floodplain access. KY-SGCN wetland fishes, however, were likely negatively affected by high magnitude depths because they require shallow littoral areas (Simpson et al. 2021) that also coincide with predictable water levels and lentic conditions (Etnier & Starnes 1993; Eisenhour et al. 2018). Relative abundance of KY-SGCN wetland fishes was negatively associated with low water events (Figure 3), however, this association was probably due to the influence of the Mississippi River, i.e., KY-SGCN wetland fish are less likely to utilize wetlands strongly influenced by the Mississippi River. Even though large-river fishes were able to utilize wetlands more influenced by the Mississippi River, prolonged low water events during the summer of 2019 prevented year-round survival (e.g., samples were fishless until the spring of 2020) as fishes became trapped and were subjected to poor water quality, predation, and eventual drying (Figure 6). Absence of prolonged low water and drying in wetlands less influenced by the Mississippi River benefited KY-SGCN by providing year-round habitat.

Lateral connectivity is important for fishes as it is one of the most influential components of floodplain ecosystem dynamics (Junk et al. 1989) that can affect water quality, primary productivity (Knowlton & Jones 1997; Galat et al. 1998), fish metacommunity dispersal, colonization, habitat utilization, and refugia from other adverse wetland conditions (Thomasz et al. 1997; Baber et al. 2002; Miranda 2005; Zeug et al. 2005; Zeug & Winemiller 2008; Shoup & Wahl 2009; Miyazono et al. 2010; Beesley et al. 2014). Conversely, a lack of lateral connectivity may be detrimental to fishes as it can lead to stranding and mortality (Richter et al. 1996). Lateral connectivity was present in wetlands with short and long waterway distances to the Mississippi River. However, lateral connectivity experienced in our wetlands behaved differently as lateral connectivity in wetlands with short waterway distances to the river were long-lasting and seasonal while lateral connectivity between nearby streams and wetlands with long waterway distances were shorter lasting, occurred more frequently, and occurred throughout the year (i.e., Rise Count, Table 1; Figure 3). Greater relative abundance of large-river fishes in wetlands highly influenced by the Mississippi River suggests that long-lasting lateral connectivity with the river allowed these fishes to disperse from the river and utilize floodplain habitat. Similarly, other studies have found that large-river fishes were the primary colonizers of floodplain habitat next to large rivers that experienced direct lateral connectivity between the river and floodplain (Miranda 2005; Zeug et al. 2005; Zeug & Winemiller 2008; Miyazono et al. 2010). Smaller frequent pulses resulting in year-round lateral connectivity most likely benefited KY-SGCN wetland fishes as they are less tolerant of high magnitude flooding and lotic conditions. Additionally, frequent pulses likely mitigated adverse water quality conditions associated with the summer and fall which allowed year-round survival of fishes. KY-SGCN wetland fishes require shallow, vegetated, lentic environments to complete their life cycles (Etnier & Starnes 1993; Eisenhour et al. 2018) and source of lateral connectivity may be meaningful. Habitat requirements for KY-SGCN wetland fishes were absent during periods of lateral connectivity with the Mississippi River, which brought upon lotic conditions, colder water temperatures, and exceptional depths. Additionally, a lack of vegetation was observed when sampling after flooding, which may have been due to plants being inundated too long or from scour.

Influence of wetland condition on fish community composition

The community composition of biota, including fishes, have been found to be more similar during floods (Miranda 2005). Similarities in fish community composition occur because annual seasonal floods homogenize floodplain environments by promoting lateral connectivity which allows for the exchange of fish between the river and floodplain habitats (Hamilton & Lewis 1990; Thomasz et al. 1997; Miranda 2005). Conversely, distinct fish community composition is a common occurrence in wetlands with less lateral connectivity and is driven by biotic interactions (e.g., predation and competition amongst fishes) and adverse water quality (Gawlik et al 2002; Henning et al. 2007; Faulkner et al. 2011). Historic seasonal flooding (see above) experienced in our wetlands likely drove similarities in community composition by greatly promoting lateral connectivity, which allowed for greater dispersal of fishes. Sunfishes and livebearers were dominant in our wetlands and these species are recognized as very common floodplain dispersers capable of quickly colonizing recently flooded areas (Gkenas et al. 2011; Alfermann & Miranda 2013). Many of our wetlands experienced long-lasting hydroperiods which may have also been responsible for driving similarities in fish community composition as these conditions have been found to ensure sunfish survival (Kushlan 1976; Hohausova et al. 2010; Alfermann & Miranda 2013) and possibly lead to competitive exclusion among other species for food resources and prime available habitat (Carrara et al. 2012; De Bie et al. 2012). The shared wetland geomorphic setting among our wetlands (i.e., riverine) may have created similarities in fish communities as many our wetlands shared source pools of fish (e.g., Bayou du Chien Creek, Mayfield Creek, Obion Creek, and the Mississippi River) during flooding. Additionally, many wetlands were proximate to one another or located on the same easement (Figure 1) allowing colonization from identical source pools of fish.

Influence of wetland condition on fish diversity

Even though fish community composition among wetland condition was similar, the effect of wetland restoration and its influence on fish communities was evident during our study as restored wetlands harbored levels of fish diversity comparable to reference wetlands and greater than diversity in degraded wetlands (Figures 4, 5). Previous studies have also found levels of diversity similar between reference and WRP restored wetlands (Juni & Berry 2001; Benson et al. 2018). Diversity in our restored wetlands reached reference levels quickly as wetlands were relatively young (i.e., 1-13 years). Similarly, Moreno-Mateos et al. (2012) found that diversity in restored wetlands quickly reached reference wetland conditions (0-5 years) if the wetland was in a warm climate and had a riverine geomorphic setting due to increased biogeochemical functioning. The humid sub-tropical climate (i.e., hot summers and mild winters) throughout our study area paired with the riverine geomorphic classification of our restored wetlands probably influenced diversity in our wetlands. Restored wetlands' association with two unique groups of fish (i.e., both large-river and KY-SGCN wetland fishes) (Figure 4) may have also helped drive high levels of diversity in restored wetlands. Low levels of diversity in degraded wetlands may have been due to lessened duration of lateral connectivity (Figure 3) and/or unmitigated alterations to wetland hydrology (King et al. 2006).

Management and conservation implications

The influence of hydrology on the relative abundances of large-river and KY-SGCN wetland fishes was evident throughout our study. Therefore, there may be utility in using hydrologic conditions as conservation management tools to influence community composition to better promote the presence of specific fish taxa in restored wetlands. Ultimately, restoration managers will have to consider tradeoffs in fish communities associated with the hydrologic conditions they hope to achieve through restoration. For example, if restoration managers aim for wetland hydrology to be influenced by the Mississippi River, large-river fishes may be promoted but at the expense of wetland obligates. Promotion of large-river fishes may be important as some species, like channel catfish and *Ictiobus* sp. (buffalo), have commercial and recreational value. Riverine influence on wetland hydrology may, however, lead to a greater likelihood of drying, presence of lotic conditions, or high magnitude depths. Drying may trap and cause mortality amongst large-river fishes utilizing these wetlands. Large scale die-offs may also be beneficial for other taxa such as wading birds, amphibians, and wetland plants. If restoration managers desire wetland hydrology to be less influenced by the Mississippi River, wetland obligates may be promoted but not necessarily large-river fishes. Promotion of wetland obligate fishes is important because many of these species are KY-SGCN, which, although not recreationally or commercially important, are of important conservation concern. Restoration of wetlands less influenced by the Mississippi River may improve the conservation status of these fishes by restoring environments that fulfill their specific habitat requirements (Eisenhour et al. 2018). Conservation of KY-SGCN wetland fishes is important because of their intrinsic value and specific habitat requirements which may indicate proper wetland ecosystem functioning (Benson et al. 2018; Simpson et al. 2021). Regardless of wetland restoration managers' goals, future wetland restorations throughout the lower Ohio River tributary ichthyofaunal region of Kentucky may benefit KY-SGCN wetland fishes as this area harbors high quality source pools of KY-SGCN wetland fishes (Eisenhour et al. 2018; Personal communication, KDFWR).

Limitations to our study

Patterns in floodplain fish community composition and diversity often vary seasonally because of hydrologic variation due to flooding and drying events (De Graaf 2003; Arrington & Winemiller 2006; Tedesco et al. 2008). Our study did not quantify seasonal patterns of community composition and diversity because we did not have adequate statistical inference due to a lack of samples taken during the winter and spring at several wetlands because of seasonal flooding. This lack of samples may have led to under representations in fish community composition and diversity because these samples may have been the most robust as flooding would have thoroughly mixed river and floodplain metacommunities. The methods used in our study were limited to data collected by backpack electrofishing. Utilizing multiple sampling gears may lead to better estimations of community composition and diversity. Further research in western Kentucky, as well as throughout the MAV, will improve our understanding of how fish respond to wetland restoration and to wetland environmental conditions. Further research should include studies that examine the effects of different wetland restoration techniques on fish communities and environmental conditions to inform managers which restoration techniques best promote the desired fish communities. Additionally, future studies may consider choosing degraded and reference sites that are also strongly influenced by the Mississippi River to determine if wetland condition determines fish communities in wetlands with similar hydrology.

Conclusions

Wetland hydrology had the greatest influence on fish communities from the variables examined. Therefore, wetland restoration practitioners may consider focusing on specific aspects of wetland hydrology to promote desired communities or increase the presence of specific taxa. The influence of wetland restoration throughout the MAV and other large-river floodplain ecosystems may have region-wide implications on fish communities as our study found high levels of fish diversity in restored wetlands. Knowledge gaps associated with a lack of post-restoration monitoring, however, exist with large wetland restoration programs and, therefore, quantifying fish communities post-restoration provides wetland restoration programs with insight and direction for future restoration and management efforts. Undoubtedly, restoration practitioners will face and must consider tradeoffs associated with wetland restoration practices and, hopefully, these results better inform future recommendations and restoration projects. Regardless, the need remains for future studies that span across larger temporal and spatial scales to better understand how wetland restoration practices can influence the entire MAV regions.

Literature Cited

Aarts, B. G., Van Den Brink, F. W., & Nienhuis, P. H. (2004). Habitat loss as the main cause of the slow recovery of fish faunas of regulated large rivers in Europe: the transversal floodplain gradient. *River research and Applications*, *20*(1), 3-23.

Alfermann, T. J., & Miranda, L. E. (2013). Centrarchid assemblages in floodplain lakes of the Mississippi alluvial valley. Transactions of the American Fisheries Society, 142(2), 323-332.

Arrington, D. A., & Winemiller, K. O. (2006). Habitat affinity, the seasonal flood pulse, and community assembly in the littoral zone of a Neotropical floodplain river. *Journal of the North American Benthological Society*, *25*(1), 126-141.

Baber, M. J., Childers, D. L., Babbitt, K. J., & Anderson, D. H. (2002). Controls on fish distribution and abundance in temporary wetlands. Canadian Journal of Fisheries and Aquatic Sciences, 59(9), 1441-1450.

Balcer, M. D., N. L. Korda, and S. I. Dodson. 1984. Zooplankton of the Great Lakes: A guide to the identification and ecology of the common crustacean species. Journal of Great Lakes Research 10:334.

Bedford, B. L. 1996. The need to define hydrologic equivalence at the landscape scale for freshwater wetland mitigation. Wiley 6:57–68.

Beesley, L. S., Gwinn, D. C., Price, A., King, A. J., Gawne, B., Koehn, J. D., & Nielsen, D. L. (2014). Juvenile fish response to wetland inundation: how antecedent conditions can inform environmental flow policies for native fish. *Journal of Applied Ecology*, *51*(6), 1613-1621.

Benson, C. E., Carberry, B., & Langen, T. A. (2018). Public–private partnership wetland restoration programs benefit Species of Greatest Conservation Need and other wetland-associated wildlife. *Wetlands Ecology and Management*, *26*(2), 195-211.

Berkowitz, J. F., Johnson, D. R., & Price, J. J. (2020). Forested Wetland hydrology in a large Mississippi river tributary system. Wetlands, 40(5), 1133-1148.

Bozelli, R. L., Thomaz, S. M., Padial, A. A., Lopes, P. M., Bini, L. M. 2015: Floods decrease zooplankton beta diversity and environmental heterogeneity in an Amazonian floodplain system. Hydrobiologia 753, 233–241.

Brauman, K., G. C. Daily, T. K. Duarte, and H. A. Mooney (2007), The nature and value of ecosystem services: An overview highlighting hydrologic services, Annu. Rev. Environ. Resour., 32, 67–98.

Brinson, M. M., and R. D. Rheinhardt. 1998. Wetland functions and relations to societal values. Pages 29–48 in M. G. Messina and W. H. Conner, editors. Southern forested wetlands ecology and management. Lewis Publishers, Boca Raton, Florida, USA.

Caldwell, P. V., Vepraskas, M. J., Gregory, J. D., Skaggs, R. W., & Huffman, R. L. (2011). Linking plant ecology and long-term hydrology to improve wetland restoration success. *Transactions of the ASABE*, *54*(6), 2129-2137.

Carrara, F., F. Altermatt, I. Rodriguez-Iturbe & A. Rinaldo, 2012. Dendritic connectivity controls biodiversity patterns in experimental metacommunities. Proceedings of the National Academy of Sciences of the United States of America 109: 5761–5766.

Chao, A., & Colwell, R. K. (2017). Thirty years of progeny from Chao's inequality: Estimating and comparing richness with incidence data and incomplete sampling. *SORT-Statistics and Operations Research Transactions*, 3-54.

Chao, A., & Jost, L. (2012). Coverage-based rarefaction and extrapolation: standardizing samples by completeness rather than size. Ecology, 93(12), 2533-2547.

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

Colwell, R. K. et al. 2012. Models and estimators linking individual-based and sample-based rarefaction, extrapolation and comparison of assemblages. – J. Plant Ecol. 5: 3–21.

Colwell, R. K., Mao, C. X., & Chang, J. (2004). Interpolating, extrapolating, and comparing incidence-based species accumulation curves. *Ecology*, *85*, 2717–2727. https://doi.org/10.1890/03-0557.

Copp, G. H., & Peňáz, M. (1988). Ecology of fish spawning and nursery zones in the flood plain, using a new sampling approach. *Hydrobiologia*, *169*(2), 209-224.

Costanza, R., De Groot, R., Sutton, P., Van der Ploeg, S., Anderson, S. J., Kubiszewski, I., ... & Turner, R. K. (2014). Changes in the global value of ecosystem services. *Global environmental change*, *26*, 152-158.

Cowardin, L. M., & Golet, F. C. (1995). US Fish and Wildlife Service 1979 wetland classification: A review. Classification and inventory of the world's wetlands, 139-152.

Dahl TE., & Allord GJ (1996) History of wetlands in the conterminous United States. In: Fretwell JD, Williams JS, Redman PJ (eds) National water summary on wetland resources: water supply paper 2425. Government Printing Office, Washington DC, pp 19–26.

De Bie, T., L. De Meester, L. Brendonck, K. Martens, B. Goddeeris, D. Ercken, H. Hampel, L. Denys, L. Vanhecke, K. Van der Gucht, J. Van Wichelen, W. Vyverman & S. A. J. Declerck, 2012. Body size and dispersal mode as key traits determining metacommunity structure of aquatic organisms. Ecology Letters 15: 740–747.

De Graaf, G. (2003). The flood pulse and growth of floodplain fish in Bangladesh. *Fisheries Management and Ecology*, *10*(4), 241-247.

Dembkowski, D. J., & Miranda, L. E. (2012). Hierarchy in factors affecting fish biodiversity in floodplain lakes of the Mississippi Alluvial Valley. *Environmental Biology of Fishes*, *93*(3), 357-368.

Dewitz J (2019) National Land Cover Database (NLCD) 2016 Products: United States Geological Survey data release, https://doi.org/10.5066/P96HHBIE (accessed 16 February 2021).

Eisenhour, D. J., Thomas, M. R., Culp, J. J., Compton, M. C., Brandt, S. L., & Pierce, R. (2018). Updated Distributional Records of Selected Kentucky Fishes. In *Southeastern Fishes Council Proceedings* (Vol. 1, No. 58, p. 1).

Etnier, D. A., & Starnes, W. C. (1993). *The fishes of Tennessee*. Newfound Press.

Faith, D. P., and R. H. Norris. 1989. Correlation of environmental variables with patterns of distribution and abundance of common and rare freshwater macroinvertebrates. Biological Conservation 50:77–98.

Faulkner, S. P., & Patrick Jr, W. H. (1992). Redox processes and diagnostic wetland soil indicators in bottomland hardwood forests. *Soil Science Society of America Journal*, *56*(3), 856-865.

Faulkner, S., Barrow Jr, W., Keeland, B., Walls, S., & Telesco, D. (2011). Effects of conservation practices on wetland ecosystem services in the Mississippi Alluvial Valley. *Ecological Applications*, *21*(sp1), S31-S48.

Flinn, M. B., Adams, S. R., Whiles, M. R., & Garvey, J. E. 2008. Biological responses to contrasting hydrology in backwaters of Upper Mississippi River Navigation Pool 25. *Environmental management*, *41*(4), 468.

Fredrickson, L. 2005. Contemporary bottomland hardwood systems: structure, function and hydrologic condition resulting from two centuries of anthropogenic activities. *Ecology and management of bottomland hardwood systems 10*, 19-35.

Galat, D. L., Fredrickson, L. H., Humburg, D. D., Bataille, K. J., Bodie, J. R., Dohrenwend, J., ... & Semlitsch, R. D. (1998). Flooding to restore connectivity of regulated, large-river wetlands: natural and controlled flooding as complementary processes along the lower Missouri River. *BioScience*, *48*(9), 721-733.

Gawlik, D. E. (2002). The effects of prey availability on the numerical response of wading birds. Ecological monographs, 72(3), 329-346.

Gkenas, C., Oikonomou, A., Economou, A., Kiosse, F., & Leonardos, I. (2012). Life history pattern and feeding habits of the invasive mosquitofish, Gambusia holbrooki, in Lake Pamvotis (NW Greece). Journal of Biological Research, 17.

Godfrey, P. C., Pearson, R. G., Pusey, B. J., & Arthington, A. H. (2020). Drivers of zooplankton dynamics in a small tropical lowland river. *Marine and Freshwater Research*, *72*(2), 173-185.

Hamilton, S. K., Lewis, W. M., & Sippel, S. J. (1992). Energy sources for aquatic animals in the Orinoco River floodplain: evidence from stable isotopes. *Oecologia*, *89*(3), 324-330.

Haynes, R. J., & Egan, D. (2004). The development of bottomland forest restoration in the Lower Mississippi River Alluvial Valley. *Ecological Restoration*, *22*(3), 170-182.

Henning, J. A., R. E. Gresswell, and I. A. Fleming. 2007. Use of seasonal freshwater wetlands by fishes in a temperate river floodplain. Journal of Fish Biology 71:476–492.

Hohausová, E., R. J. Lavoy, and M. S. Allen. 2010. Fish dispersal in a seasonal wetland: Influence of anthropogenic structures. Marine and Freshwater Research 61:682–694.

Hsieh, T. C., Ma, K. H., & Chao, A. (2016). iNEXT: An R package for interpolation and extrapolation of species diversity (Hill numbers). Methods in Ecology and Evolution: Under revision. 7(12): 1451-1456. https://doi. org/10.1111/2041 210X.12613.

Hunter, R. G., Faulkner, S. P., & Gibson, K. A. (2008). The importance of hydrology in restoration of bottomland hardwood wetland functions. *Wetlands*, *28*(3), 605-615.

Jester, D. B. (1992, February). The fishes of Oklahoma, their gross habitats, and their tolerance of degradation in water quality and habitat. In *Proceedings of the Oklahoma Academy of Science* (pp. 7-19).

Jost, L. (2006). Entropy and diversity. Oikos, 113, 363–375. https://doi.org/10.1111/oik.2006.113.issue-2.

Juni, S., Berry, C. R., & Fish, S. D. C. (2001). A biodiversity assessment of compensatory mitigation wetlands in eastern South Dakota. In *Proceedings of the South Dakota Academy of Science* (Vol. 80, pp. 185-200).

Junk, W. J., Bayley, P. B., & Sparks, R. E. (1989). The flood pulse concept in river-floodplain systems. Canadian special publication of fisheries and aquatic sciences, 106(1), 110-127.

Kentucky Division of Water. 2016. Guidance Manual for KY-WRAM, Version 3.0. https://eec.ky.gov/Environmental-Protection/Water/Monitor/Pages/KYWRAM.aspx (accessed 1 July 2020).

King, S. L., & Keeland, B. D. (1999). Evaluation of reforestation in the lower Mississippi River alluvial valley. *Restoration Ecology*, *7*(4), 348-359.

King, S. L., & Keim, R. F. (2019). Hydrologic modifications challenge bottomland hardwood forest management. *Journal of Forestry*, *117*(5), 504-514.

King, S. L., D. J. Twedt, and R. R. Wilson. 2006. The Role of the Wetland Reserve Program in conservation efforts in the Mississippi River Alluvial Valley. Wildlife Society Bulletin 34:914–920.

Kluender, E. R., Adams, R., & Lewis, L. (2017). Seasonal habitat use of Alligator Gar in a river–floodplain ecosystem at multiple spatial scales. *Ecology of Freshwater Fish*, *26*(2), 233-246.

Knowlton, M. F., & Jones, J. R. (1997). Trophic status of Missouri River floodplain lakes in relation to basin type and connectivity. *Wetlands*, *17*(4), 468-475.

Kushlan, J. A. (1976). Environmental stability and fish community diversity. *Ecology*, *57*(4), 821-825.

Leao, M. (2005). *Fish utilization and diversity associated with created wetlands within the Lower White River Watershed, Arkansas* (Doctoral dissertation, University of Arkansas at Pine Bluff).

Magurran, A. E. (1988). Why diversity? *Ecological diversity and its measurement* (pp. 1–5). Dordrecht, Netherlands: Springer Netherlands. https:// doi.org/10.1007/978-94-015-7358-0_1.

McCune B, Grace JB (2002) Analysis of ecological communities. MjM Software Design, Oregon.

Merkle, J. A., Anderson, N. J., Baxley, D. L., Chopp, M., Gigliotti, L. C., Gude, J. A., ... & VanBeek, K. R. (2019). A collaborative approach to bridging the gap between wildlife managers and researchers. *The Journal of Wildlife Management*, *83*(8), 1644-1651.

Miranda, L. E. (2011). Depth as an organizer of fish assemblages in floodplain lakes. *Aquatic Sciences*, *73*(2), 211-221.

Miranda LE, Lucas GM (2005) Determinism in fish assemblages of floodplain lakes of the vastly disturbed Mississippi Alluvial Valley. Trans Am Fish Soc 133:358–370.

Mitsch, W. J., & Gosselink, J. G. (2015). *Wetlands*. John Wiley & Sons.

Miyazono S, Aycock JN, Miranda LE, Tietjen TE (2010). Assemblage patterns of fish functional groups relative to habitat connectivity and conditions in floodplain lakes. Ecol Freshw Fish 19:578–585.

Moreno-Mateos, D., Power, M. E., Comín, F. A., & Yockteng, R. (2012). Structural and functional loss in restored wetland ecosystems. *PLoS Biol*, *10*(1), e1001247.

Natural Resources Conservation Service (2013a) Migratory Bird Habitat Initiative. https://prod.nrcs.usda.gov/wps/portal/nrcs/detailfull/national/ programs/farmbill/initiatives/?cid=steldevb1027669/ (accessed 27 June 2013)

Omernik, J. M. (1987). Ecoregions of the conterminous United States. *Annals of the Association of American geographers*, *77*(1), 118-125.

Oskansen J, Blanchet FG, Friendly M, Kindt R, Legendre P, McGlinn D, Minchin PR, O'Hara RB, Simpson GL, Solymos P, Stevens MHH, Szoecs E, Wagner H (2013) vegan: Community ecology package. R package version 2.0-7. https://CRAN.R-project.org/package=vegan Accessed 4 June 2021.

Pace, M.L., S.E.G. Findlay, and D. Links. 1992. Zooplankton in advective environments: the Hudson River community and a comparative analysis. Can. J. Fish. Aquat. Sci. 49: 1060-1069.

Pennak, R. 1962. Quantative zooplankton sampling in littoral vegetation areas. Limnology and Oceanography.

Petts GE (1989) Perspectives for ecological management of regulated rivers. Pages 3–24 In: Gore JA, Petts GE (eds) Alternatives in regulated river management. CRC Press, Inc., Boca Raton, Florida.

Pflieger, W. L., Taylor, L., & Sullivan, M. (1975). The fishes of Missouri.

R Core Team (2015). *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing.

Resh, V. H., Brown, A. V., Covich, A. P., Gurtz, M. E., Li, H. W., Minshall, G. W., ... & Wissmar, R. C. (1988). The role of disturbance in stream ecology. *Journal of the North American benthological society*, *7*(4), 433-455.

Rewa, C. 2000. Biological responses to wetland restoration: Implications for wildlife habitat development through the Wetlands Reserve Program. Pages 95–116 in W. L. Hohman and D. J. Halloum, editors. A comprehensive review of Farm Bill contributions to wildlife conservation, 1985–2000. U.S. Department of Agriculture, Natural Resources Conservation Service, Wildlife Habitat Management Institute, Technical Report USDA/NRCS/WHMI-2000.

Rewa, C. A. (2005). Wildlife Benefits of the Wetlands Reserve Program.

Richter, B. D., Baumgartner, J. V., Powell, J., & Braun, D. P. (1996). A method for assessing hydrologic alteration within ecosystems. *Conservation biology*, *10*(4), 1163-1174.

Rodríguez, M. A., & Lewis Jr, W. M. (1997). Structure of fish assemblages along environmental gradients in floodplain lakes of the Orinoco River. *Ecological monographs*, *67*(1), 109-128.

Roswell, M., Dushoff, J., & Winfree, R. (2021). A conceptual guide to measuring species diversity. Oikos, 130(3), 321-338.

Ruiz-Jaen, M. C., & Mitchell Aide, T. (2005). Restoration success: how is it being measured? *Restoration ecology*, *13*(3), 569-577.

Semlitsch, R. D. (2000). Size does matter: the value of small, isolated wetlands. *National Wetlands Newsletter*, *22*(1), 5-6.

Shoup, D. E., & Wahl, D. H. (2009). Fish diversity and abundance in relation to interannual and lake-specific variation in abiotic characteristics of floodplain lakes of the lower Kaskaskia River, Illinois. *Transactions of the American Fisheries Society*, *138*(5), 1076-1092.

Simpson, N. T., Bybel, A. P., Weber, M. J., Pierce, C. L., & Roe, K. J. (2021). Factors associated with distributions of six fishes of greatest conservation need in streams in midwestern USA. *Aquatic Conservation: Marine and Freshwater Ecosystems*.

Stanturf, J. A., Schoenholtz, S. H., Schweitzer, C. J., & Shepard, J. P. (2001). Achieving restoration success: myths in bottomland hardwood forests. *Restoration Ecology*, *9*(2), 189-200.

Tedesco, P. A., Hugueny, B., Oberdorff, T., Dürr, H. H., Merigoux, S., & De Mérona, B. (2008). River hydrological seasonality influences life history strategies of tropical riverine fishes. *Oecologia*, *156*(3), 691-702.

Thomaz, S. M., Bini, L. M., Bozelli, R. L. 2007: Floods increase similarity among aquatic habitats in river-floodplain systems. Hydrobiologia 579, 1–13.

United States. (2017) U.S. Census of Agriculture, accessed June 3, 2021, at URL http://www.agcensus.usda.gov.

USDA OIG [U.S. Department of Agriculture, Office of Inspector General]. 2008. Natural Resources Conservation Service Wetlands Reserve Program wetlands restoration and April 2011 WETLAND CONSERVATION PRACTICES S4729 compliance. Audit report number 10099-4-SF. USDA, Office of Inspector General Washington, D.C.

U.S. Geological Survey, 2020, National Water Information System data available on the World Wide Web (USGS Water Data for the Nation), accessed May 25, 2021, at URL https://waterdata.usgs.gov/nwis/inventory?agency_code=USGS&site_no=07024000.

USGS National Map 3DEP Downloadable Data Collection: U.S. Geological Survey. https://www.sciencebase.gov/catalog/item/4f70aa9fe4b058caae3f8de5. Accessed 30 September 2021.

USGS Watershed Science School, 2019, Streamflow and the Water Cycle, accessed June 27, 2022, at URL https://www.usgs.gov/special-topics/water-science-school/science/streamflow-and-water-cycle.

Welcomme, R. L. (1985). *River fisheries* (No. 262). FAO.

Winemiller KO, Rose KA (1992) Patterns of life-history diversification in North American fishes: implications for population regulation. Canadian Journal of Fisheries and Aquatic Sciences 49:2196–2218.

Zedler, J. B. (2000). Progress in wetland restoration ecology. *Trends in ecology & evolution*, *15*(10), 402-407.

Zeug, S. C., Winemiller, K. O., & Tarim, S. (2005). Response of Brazos River oxbow fish assemblages to patterns of hydrologic connectivity and environmental variability. Transactions of the American Fisheries Society, 134(5), 1389-1399.

Zeug, S.C. & Winemiller, K.O. 2008. Relationships between hydrology, spatial heterogeneity, and fish recruitment dynamics in a temperate floodplain river. River Research and Applications 24: 90–102.

Figures and Tables

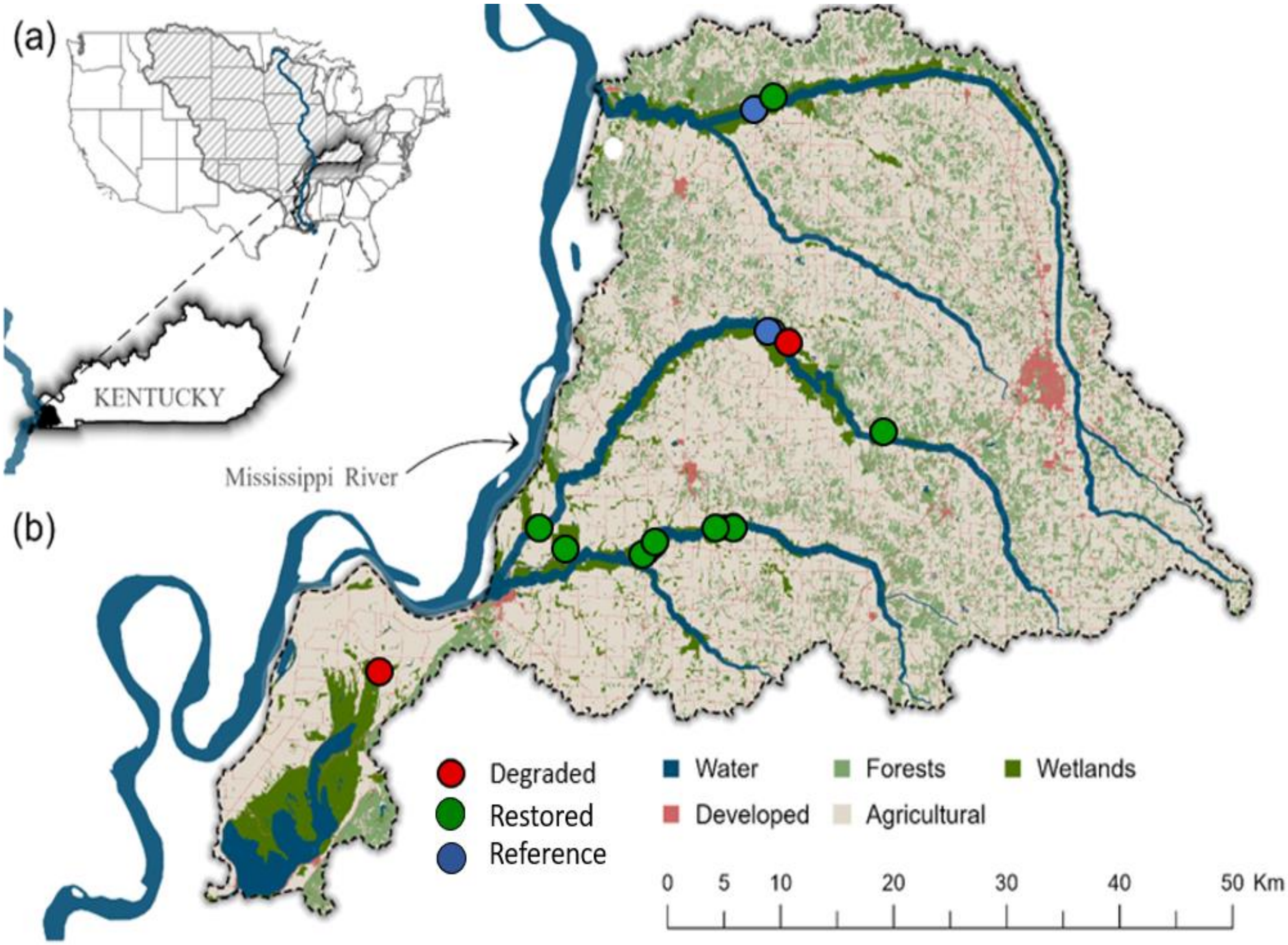

Figure 1. (A) Location of study area in western Kentucky and part of the Mississippi River watershed, USA. (B) Twelve study wetlands include degraded (n=2), restored (n=8), and reference wetlands (n=2) (land use classifications from 2018 USDA National Imagery Program).

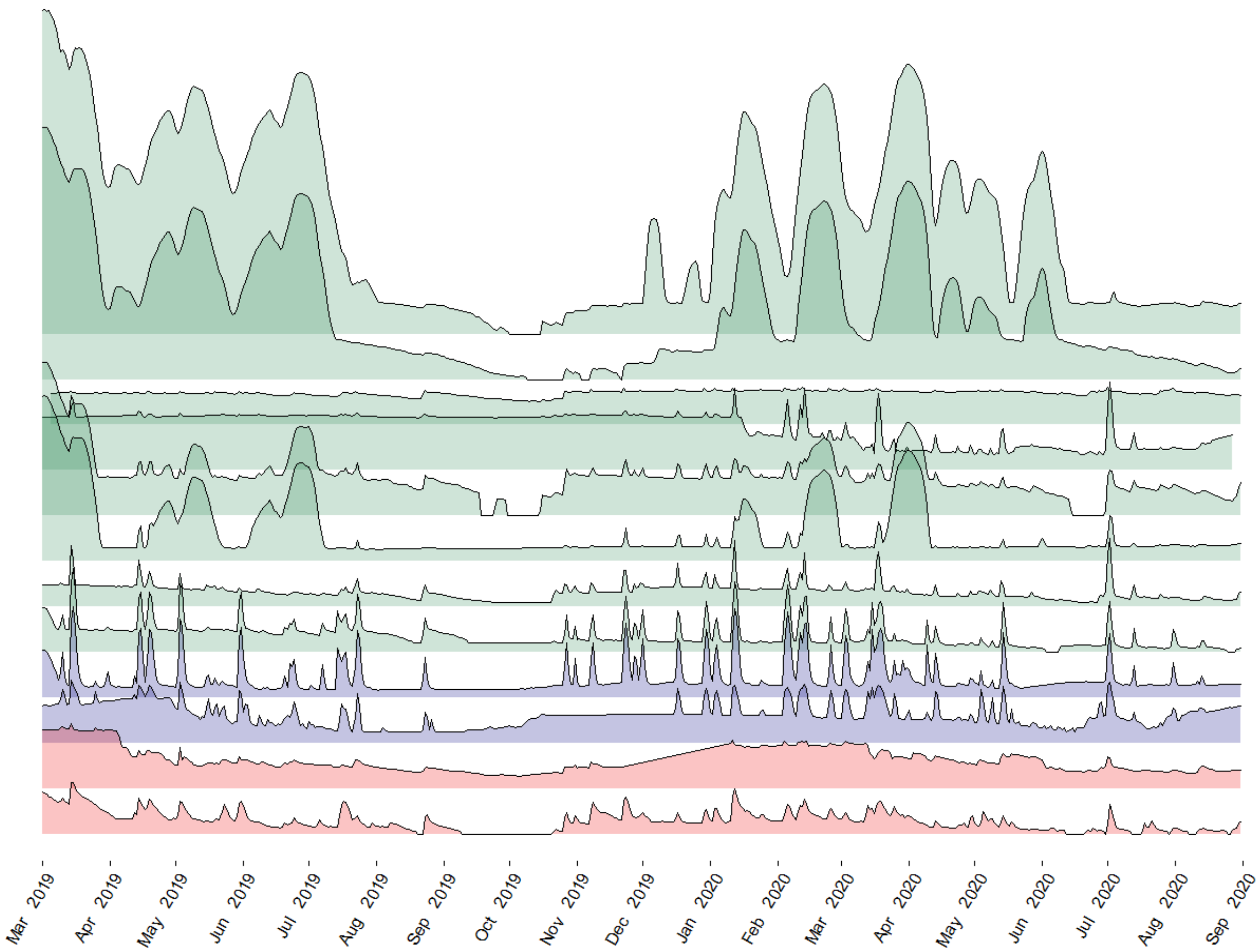

Figure 2: Hydrographs of degraded (red, n=2), restored (green, n=8), and reference wetlands (blue, n=2) in western Kentucky, USA. Wetland depth was collected at 15 min intervals from March 2019 to September 2020.

Table 1: Environmental metrics (28) considered for vector analysis in NMDS ordinations measured in wetlands throughout western Kentucky, USA. Each metric was calculated using data collected over the entire sampling period for each wetland in our study. Indicators of hydrologic alteration following Richter et al. (1996) are denoted with the abbreviation 'IHA'.

Metric	**Definition**	**Measures**
Hydrologic metrics		
Mean Depth (m)	The average wetland depth.	Magnitude
Hydroperiod	The number of days a wetland had water.	Duration
Duration of Connectivity	The percent of time a wetland exhibited lateral connectivity to its nearest stream (USGS-NWIS).	Connectivity
1-Day Maximum (m) (IHA)	The maximum single day depth recorded.	Magnitude/Duration
1-Day Minimum (m) (IHA)	The minimum single day depth recorded.	Magnitude/Duration
7-Day Maximum (m) (IHA)	The maximum 7-day rolling average recorded.	Magnitude/Duration
7-Day Minimum (m) (IHA)	The minimum 7-day rolling average recorded.	Magnitude/Duration
30-Day Maximum (m) (IHA)	The maximum 30-day rolling average recorded.	Magnitude/Duration
30-Day Minimum (m) (IHA)	The minimum 30-day rolling average recorded.	Magnitude/Duration
90-Day Maximum (m) (IHA)	The maximum 90-day rolling average recorded.	Magnitude/Duration
90-Day Minimum (m) (IHA)	The minimum 90-day rolling average recorded.	Magnitude/Duration
Minimum Date (day of the year) (IHA)	The date the lowest single recorded depth occurred.	Timing
Maximum date (day of the year) (IHA)	The date the greatest single recorded depth occurred.	Timing
Low Water Duration (days) (IHA)	The number of consecutive days depths stayed below the 25th percentile.	Duration
High Water Duration (days) (IHA)	The number of consecutive days depths stayed above the 75th percentile.	Duration
Rise Count (IHA)	The number of occurrences that wetland depth rose from the previous day.	Frequency/Rate of change
Water quality metrics		
Mean Dissolved Oxygen (DO) (mg/L)	The average DO measurement.	Magnitude
Minimum Dissolved Oxygen (DO) (mg/L)	The minimum single day DO recorded.	Magnitude
Mean Temperature (C°)	The average temperature recorded.	Magnitude
1-Day Maximum (C°)	The maximum single day temperature recorded.	Magnitude
1-Day Minimum (C°)	The minimum single day temperature recorded.	Magnitude

Biotic metrics		
Vegetation	The average aquatic vegetation scores.	Habitat
Zooplankton Density (individuals/L)	The average zooplankton density.	Food resources
KY –WRAM Score	The KY-WRAM score indication ecological integrity (0-100) (100 indicates highest quality).	Wetland quality
Hydrologic connectivity		
Distance to Main Channel (m)	The distance to the main channel of the nearest stream.	Connectivity
Topography	The mean slope inside a 1km buffer around wetland.	Connectivity
Waterway Distance to Mississippi River (km)	The shortest waterway distance from each wetland to the main channel of the Mississippi River (USGS Stream Stats).	Hydrologic influence of Mississippi River
Elevation (m)	The median of elevation (n=50) (USGS Stream Stats).	Connectivity

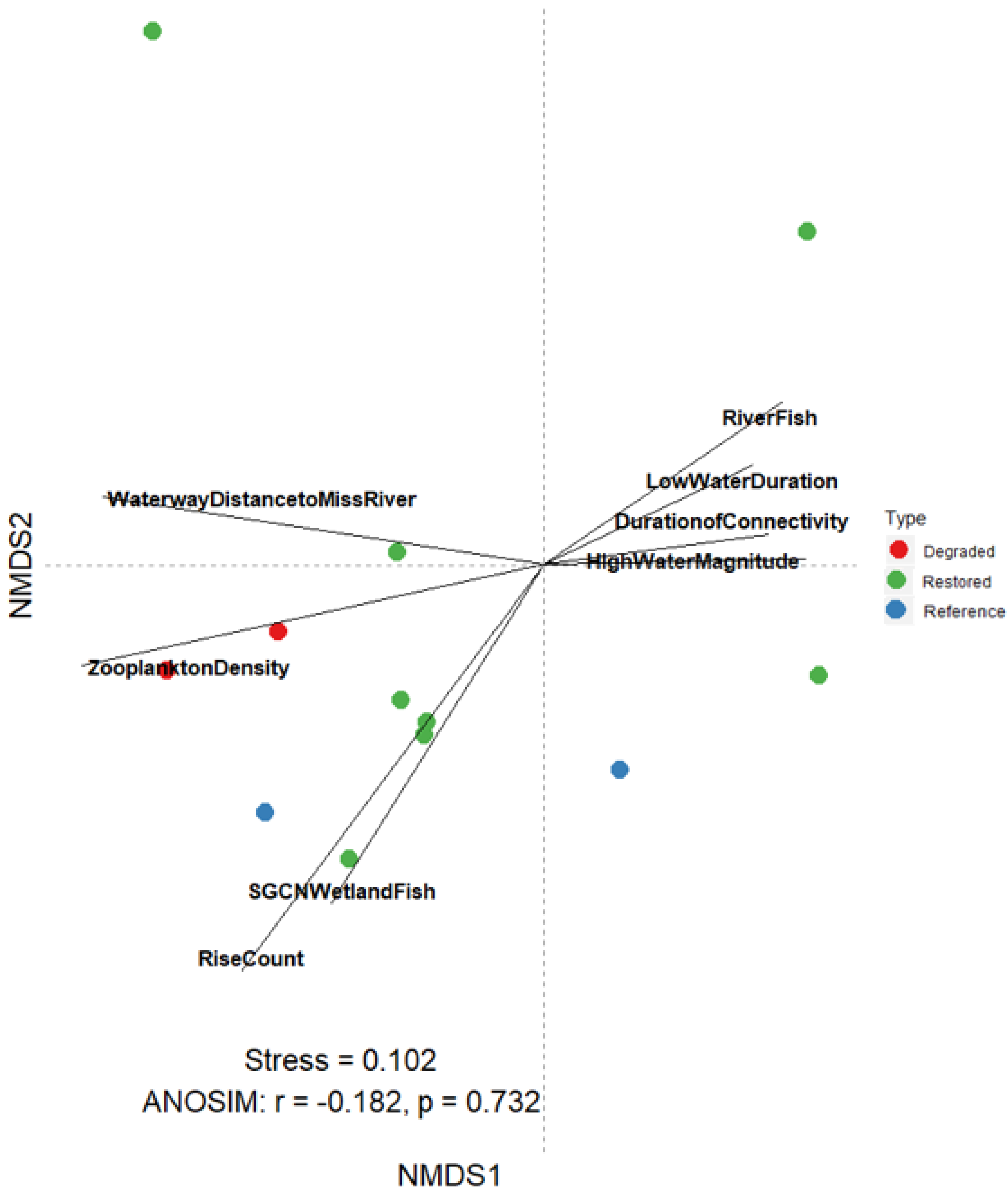

Figure 3: NMDS ordination of fish community composition from wetlands in western Kentucky, USA. Ordination is based on per taxa CPUE from electrofishing that occurred monthly from April 2019 to August 2020. Symbol colors indicate level of wetland condition (degraded, restored, reference). All variables included in Table 1 were tested and only significant variables were placed onto ordination as vectors. Vectors representing the relative abundance of river fish and KY-SGCN wetland fish were also placed onto ordination. The vector High Water Magnitude is a combination of the metrics 1-Day Maximum (m), 7-Day Maximum (m), 30-Day Maximum (m), and 90-Day Maximum (m). See Table 2 for definitions of variables used as vectors.

Table 2: Correlation coefficient and p-values associated for vectors placed onto NMDS ordination that had significant associations with fish community composition.

Vector	Correlation Coefficient	P-Value
High Water Magnitude	0.742	0.009
Zooplankton Density	0.620	0.015
Low Water Duration	0.599	0.025
Waterway Distance to Miss River	0.579	0.026
Rise Count	0.545	0.029
Duration of Connectivity	0.609	0.043
River Fish	0.784	0.013
SGCN Wetland Fish	0.358	0.128

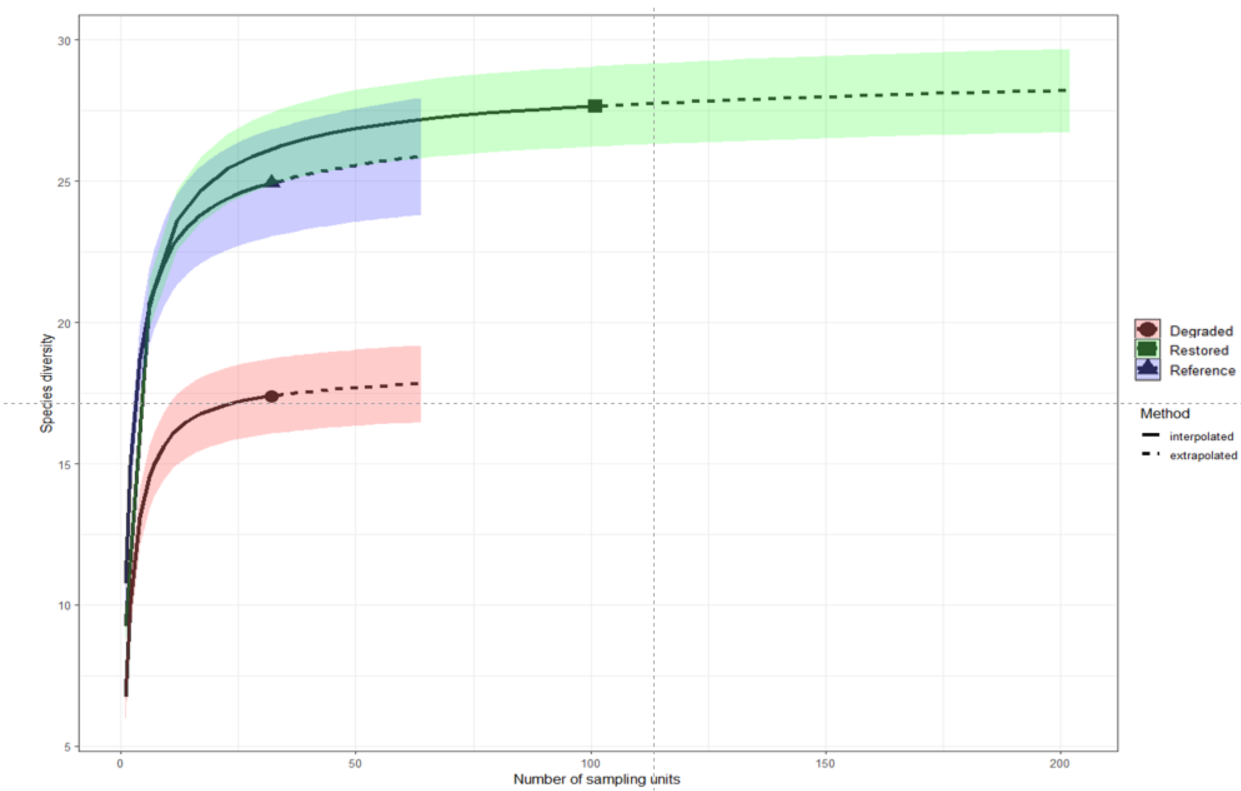

Figure 4: Hill-Shannon diversity estimates of wetland fish communities by wetland condition using incidence-based rarefaction and extrapolation. Curves are based on electrofishing data collected from April 2019 to August 2020 in western Kentucky, USA. All abundance-based extrapolation curves were plotted to achieve 95% coverage.

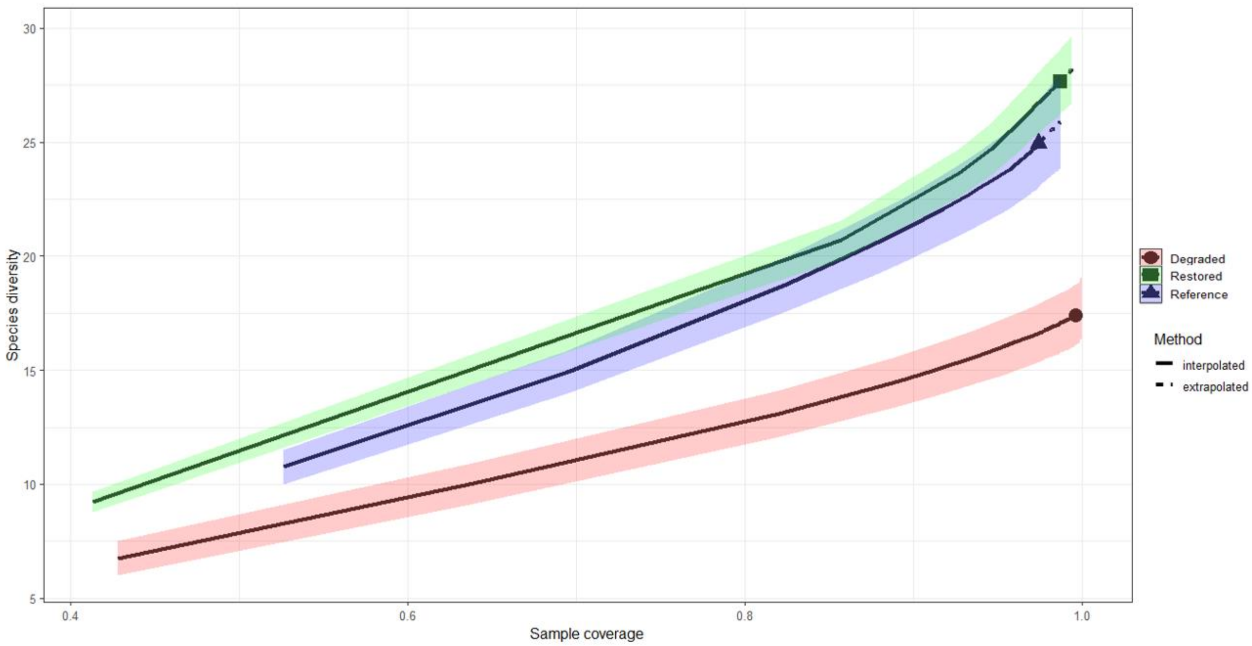

Figure 5: Hill-Shannon diversity estimates of wetland fish communities by wetland condition using coverage-based rarefaction and extrapolation. Curves are based on electrofishing data collected from April 2019 to August 2020 in western Kentucky, USA. All abundance-based extrapolation curves were plotted to achieve 95% coverage.

Figure 6: Photograph of a low water induced fish kill in a wetland with a short waterway distance to the Mississippi River (strongly influenced by the river) in western Kentucky, USA. Photograph was taken during August of 2020.

Chapter 2

Wetland elevation is an important determinant of larval fish community composition

Abstract

A major goal of the Wetlands Reserve Program (WRP) is to create wildlife benefits through wetland restoration. WRP, however, may overlook wetland elevation in their selection criteria which can have important implications on biological communities. The objectives of our study were to (1) examine if differences in wetland elevation led to variation in larval fish communities, (2) determine the relationships between larval fish community composition and wetland environmental conditions, and (3) determine the influence of differences in wetland condition (e.g., degraded, restored, and reference) on larval fish communities. From March through August 2020, we performed monthly dipnet surveys for larval fishes in lowland (n=3), transitional (n=2), and upland (n=7) wetlands throughout western Kentucky. Analysis of variance using distance matrices (ADONIS) and analysis of similarity percentages (SIMPER) were used in conjunction with non-metric multidimensional scaling to visualize larval fish community composition, quantify differences in composition amongst wetland elevation, and determine which species significantly contributed to differences in composition. Vector analysis was used to determine which environmental factors most affect larval fish community composition. Furthermore, larval fish diversity was quantified using Hill diversity (Hill-Shannon; q=1) and compared using 95% confidence intervals. Our findings suggest that differences in wetland elevation were characterized by differences in hydrologic conditions, which led to differences in larval fish community composition. Differences in community composition were driven by greater abundances of <u>Hypophthalmicthys</u> (*bighead carp*) and <u>Pomoxis</u> (*crappie*) *in lowland wetlands when compared to upland wetlands. Wetland elevation did not lead to differences in diversity. Furthermore, wetland condition did not lead to differences in larval fish community composition or diversity. Our study demonstrated that wetland elevation and hydrology are important factors for wetland restoration managers to consider when selecting sites for future wetland restorations.*

Introduction

Many of the United States non-coastal wetland resources exist as riverine wetlands throughout the Mississippi Alluvial Valley (MAV). Riverine wetlands are floodplain areas that are periodically inundated by direct overland flow, backwater flooding from adjacent rivers or streams, or surface runoff from precipitation (Brinson 1993; Hunter et al. 2008). As in other wetlands, the quantity and timing of hydrology define the structure and function of riverine wetlands (Faulkner & Patrick 1992; Corstanje & Reddy 2004). The hydrology of riverine wetlands throughout the MAV is largely influenced by the Mississippi River due to its seasonal flood pulse (winter and spring) and by local climate after flood waters recede (summer and fall) (Junk et al. 1989; Mitsch & Gosselink 2015). Like the biogeochemical processes and vegetation communities that rely on the hydrology of these wetlands, so do wildlife communities.

The timing, duration, and magnitude of the MAV's hydrology is especially important to floodplain fishes as it fulfills a variety of life history requirements. The MAV's hydrology is flood pulsed in nature which promotes high levels of connectivity between the river channel and nearby wetlands and coincides with the seasonally increased energetic needs required for reproduction, i.e., spawning and healthy eggs (Welcomme 1985; Junk et al. 1989; Bayley 1991; Winemiller & Rose 1992; King & Keeland 1999; Tocker et al. 2000; Baber et al. 2002; Kluender et al 2015). The timing of connectivity, however, must directly correspond with optimal spawning temperatures to ensure reproductive success (Lubinksi et al. 1991; Sparks et al. 1998). Many demersal spawners (i.e., those that require structure to lay eggs), including main channel obligates (e.g., blue sucker), require the habitat-rich littoral areas in riverine wetlands to successfully lay their eggs (Adams et al. 2006). Once hatched, many larval fish utilize food-rich (e.g., zooplankton) littoral areas to acquire energy required to grow and develop before emigrating back into their population (Chick & McIvor 1997; Dettmers et al. 2001; Adams et al. 2008; Kluender et al. 2015).

Despite the overwhelming importance of riverine wetlands for floodplain fishes throughout the MAV, the existence of these wetlands is in danger. Since European colonization, the MAV has lost 70-84% of its wetlands which once spanned across a vast 10 million ha (Haynes & Egan 2004; Frederickson 2005; Faulkner et al. 2011). Radical alterations to regional hydrology (e.g., dams, ditches, levees, and tile drains) paired with direct land-use conversion to agriculture have largely been responsible for these losses (Semlitsch 2000; Haynes & Egan 2004; King et al. 2006; Faulkner et al. 2011). Changes in hydrology and conversion to agriculture throughout a river-floodplain system are considered the most serious and pervasive anthropogenic threats to the system’s ecological integrity because it separates the river from its floodplain (Tockner et al. 2000; Poff et al. 2007). Loss of lateral (e.g., river to floodplain) connectivity greatly diminishes riverine wetland function and wildlife benefits (Zedler 2000; Rewa 2005; Hunter et al. 2008; Moreno-Mateos et al. 2012; King & Keim 2019). Dramatic wetland loss throughout the MAV, however, has not gone unnoticed and the past 30 years have seen the implementation of large-scale wetland restorations performed at state and federal levels.

The Wetlands Reserve Program (WRP) is one such large-scale federal-level restoration program. Implemented by the Natural Resources Conservation Service (NRCS), the WRP conducts wetland restorations throughout the United States. The goals of the WRP are to restore wetland ecological function and wildlife benefits (Natural Resources Conservation Service 2013a). The WRP focuses on using a combination of reforestation and hydrologic restoration techniques to restore historic wetland function (King & Keeland 1999; Hayes & Egan 2004; Rewa 2005). Special emphasis is placed on hydrologic restoration as hydrology drives wetland function and wildlife benefits (Bedford 1996; Brinson & Rheinhardt 1998; Zedler 2000; Haynes & Egan 2004; Rewa 2005; Brauman et al. 2007; Hunter et al. 2008; Faulkner et al. 2011; King & Keim 2019).

When selecting a site for wetland restoration, WRP ranking criteria are extensive and many hydrologic aspects are taken into consideration. Elevation of a wetland, however, is not always considered in selection criteria (NRCS WRP Ranking Criteria 2008). WRP typically employs restoration in riverine wetlands to maximize ecosystem function. Wetland elevation may result in large differences in environmental conditions, i.e., hydrology and water quality, even if hydrogeomorphic classifications are similar (Brinson 1993; Acreman & Holden 2013). For example, a riverine wetland located in the Mississippi River's floodplain may experience greater magnitude and duration of flooding during seasonal flooding events due to its massive watershed, while another riverine wetland located on an upstream tributary of the Mississippi River, i.e., smaller watershed, experiences relatively smaller magnitude and shorter duration flooding as its hydrology is less influenced by the seasonal flooding of the Mississippi River and more by inputs from local precipitation (Euliss et al. 2004; Acreman & Holden 2013). The effects of elevation on local wetland hydrology and water quality may have implications on the response of wildlife to wetland restoration as fish communities have been shown to be strongly influenced by these environmental conditions (Winter 2001; Euliss et al. 2004; Miranda & Lucas 2004; Miranda 2005; Miranda 2010; Dembkowski & Miranda 2012).

Understanding the factors that structure floodplain wetland larval fish communities is important because larval fish are reliant on specific hydrologic conditions and their recruitment is important in maintaining floodplain fish communities Therefore, choosing wetland restoration sites that promote larval fish usage may indicate proper hydrologic function and ultimately benefit floodplain wildlife. The primary goals of this research were to (1) examine if differences in elevation led to differences in larval fish communities, (2) determine the relationships between larval fish community composition and environmental conditions, i.e., hydrology, water quality, and biota, in wetlands, and (3) examine if differences in wetland condition (e.g., degraded, restored, and reference) influenced larval fish communities. Wetland elevation was predicted to lead to differences in larval fish community composition and diversity, that differences in larval fish communities would reflect differences in environmental conditions and lastly, that larval fish communities would differ based on wetland condition. Hopefully, larval fish/environment relationships from this research can inform future wetland restoration management decisions and ensure the most effective management decisions (Merkle et al. 2019).

Methods

Study Area

Our study was conducted in the Mississippi Alluvial Plain and Mississippi Valley Loess Plain ecoregions of western Kentucky (Omerink 1987). Historically, wetland resources in this region were characterized by bottomland hardwood forests and stream floodplains that experienced dramatic but predictable hydroperiods seasonally influenced by the upper Mississippi River watershed (King et al. 2006; Mitsch & Gosselink 2015). A substantial amount of wetland resources in this region still exist and are located along three major tributaries (Mayfield Creek, Obion Creek, and Bayou du Chien Creek) that drain approximately 2,800 km^2 into the Mississippi River. These wetlands are highly fragmented and suffer from changes to surrounding land use and regional hydrologic modifications (Frederickson 2005, King et al. 2006). Despite these anthropogenic influences, the region retains some features of a large river floodplain; for example, seasonally high discharges reconnect the floodplain in the winter and spring (Mitsch & Gosselink 2015). Current land use of this region of western Kentucky is dominated by cultivated crops (64%) and forested floodplain wetlands (25%) (Dewitz 2019).

Wetland Selection

Twelve wetlands in far western Kentucky (Figure 1) were sampled. Wetland hydrology exhibited differences based on elevation, measured in meters above sea level (MASL) (Figure 2). Following differences in hydrology based on elevation, wetlands were classified as lowland (n=3) if their elevation was < 91 (MASL); transitional (n=2) if their elevation was > 91 but < 97 MASL; and upland (n=7) if their elevation was > 97 MASL.

Eight of our wetlands were restored by the WRP in western Kentucky and were used in analyses to compare differences in wetland elevation and condition (Figure 1). Before wetland selection, pertinent WRP easement information was obtained from the National Resources Conservation Service (NRCS) (including landowner contact, restoration type, restoration age, restoration plans). After obtaining easement information, the following criteria were used to select restored wetlands: (1) location on one of the regional tributaries to the Mississippi River (Figure 1), (2) similarity of hydrogeomorphic wetland class (riverine following Brinson et al. 1993), (3) hydroperiod (semi-permanent to permanent following Cowardin et al. 1995), and (4) size (1-20 ha). Additionally, low (degraded; n=2) and high-quality wetlands (reference; n=2) were sampled to compare WRP restored wetlands with (Figure 1). Degraded wetlands were once natural wetlands that have experienced dramatic local hydrologic alterations for agricultural purposes, but still exhibit some wetland characteristics. Both of our degraded wetlands were in active agricultural fields. Reference wetlands were not subjected to local hydrologic alterations, but ultimately still exist within a highly altered landscape. One of our reference sites was a forested wetland located on a KY Wildlife Management Area; the other site was a bottomland hardwood swamp positioned on an upstream portion of one of our WRP easements.

Larval fish sampling

Larval fish were sampled monthly at all study sites from March 2020 through August 2020 using a 20 jab dipnet (30.5 cm x 25.4 cm x 55.9 cm, 500μm) survey from all available areas (open water, vegetation, woody debris). Larval fish were anesthetized using clove oil, preserved in a 10% formalin solution, and were later enumerated and identified to genus using Auer

(1982). Dipnet surveys were standardized by the number of jabs and catch per unit effort (CPUE) was considered as the abundance of each taxon captured from each survey. Monthly CPUE of each taxon was then averaged to give one CPUE value to be used for NMDS ordinations.

Environmental influences on larval fish community composition

Twenty-six environmental predictor variables (see below for explanations; Table 1) were quantified to examine their influence on larval fish community composition. Environmental predictor variables were to examine the influence of hydrology, water quality, landscape, and biota on community composition.

Hydrology

Surface water level changes were recorded in each wetland from March 2019 to August 2020 using water level loggers (HOBO® U20-001-04, Onset Computer Corporation). One logger was deployed in each wetland in the deepest wade-able location. Depth was recorded every 15 minutes. Water level data was averaged per day and daily depths were then used to calculate mean depth, hydroperiod, and 13 indicators of hydrologic alteration (IHA) based on Richter et al. (1996) (Table 1). The percent of time each wetland experienced connectivity to its associated stream was estimated by determining the stream depth at which each wetland was connected to its stream and then calculating the percent of time the stream was greater than that depth. For wetlands on upstream portions of tributaries, stream depth was determined by using the United States Geological Survey stream gauge 07024000 (USGS 2019-2020) on the Bayou du Chien and for wetlands located on the Mississippi River floodplain stream depth was determined by using the United States Geological Survey stream gauge 07022000 (USGS 2019-2020) on the Mississippi River.

Hydrologic connectivity

Two landscape variables were calculated to indirectly quantify hydrologic connectivity between wetlands and other permanent bodies of water. Distance to Main Channel was quantified by measuring the straight-line distance from the center of each wetland to the main channel of the nearest major stream in ArcGIS Pro (Version 2.7, Esri Inc.). The "elevation profile" tool in USGS Stream Stats was used to delineate each wetland's boundary and assign elevation (m) values for 50 different locations within each wetland. Elevation was calculated by taking the median of each wetland's 50 elevation values.

Water quality

Changes in water temperature (°C) and dissolved oxygen (DO) (mg/L) were recorded in each wetland from March 2019 to August 2020 using multi-parameter sondes (YSI® EXO2, Xylem Incorporated). One sonde was suspended in each wetland approximately midway in the water column in the deepest accessible location. Sondes recorded data at 15-minute intervals. Water temperature and DO were averaged per day and then used to calculate minimum temperature, maximum temperature, mean temperature, minimum DO, and mean DO for each wetland.

Biotic variables

Zooplankton communities were sampled monthly from April 2019 through August 2020 at each wetland using a 9-cm diameter littoral sampling tube following Pennak (1962). Each sample consisted of three replicates averaged together, one each taken from open water, edge of wetland, and dense vegetation. Samples poured into a volumetric container where volume (L) was recorded, rinsed through a 43-μm sieve, and preserved in 4% buffered formalin solution. In the laboratory, samples were subsampled to a maximum 1/8 using a Folsom Plankton Splitter

(WILDCO, Saginaw, MI) so that at least 50 zooplankters were found per sample. Cladocerans and copepods were enumerated but rotifers were excluded. After scaling back up to 100% from the subsampled fraction, density was calculated by dividing the abundance of each sample by the original volume of water.

Aquatic vegetation was sampled monthly at each site between April 2020 and August 2020, which corresponded with the region’s growing season, by using a 1m-by-1m quadrat. Each sample consisted of nine replicates averaged together: three each taken from open water, wetland edge, and dense vegetation. Percent cover of three aquatic vegetation groups (aquatic emergent, aquatic submergent, aquatic floating) was estimated using six cover categories (1 = 0-10%, 2= 11-20%, 3 = 21-40%, 4 = 41-60%, 5= 61-80%, 6 = 81-100%). The midpoint of the cover categories was used for percent cover calculations.

The Kentucky Wetland Rapid Assessment Method (KY-WRAM) (Kentucky Division of Water, 2016) was used to assess for general wetland integrity and function. The KY-WRAM is used to evaluate overall wetland function and ecological integrity and was developed for use by regulatory agencies in wetland permitting decisions. The KY-WRAM assumes that wetlands with high ecological integrity also have high wetland function. The KY-WRAM metrics were developed to capture a full range of potential disturbances to wetland integrity and fall into six categories: size and distribution; buffer and intensity of surrounding land uses; hydrology; habitat alteration and habitat structure development; special wetlands; vegetation, interspersion, and habitat features. Each category is subdivided into additional metrics. The special wetlands category was omitted because we were unable to gain access to the required information (all wetlands received a score of 0 for this category). The final KY-WRAM score is the sum of all the metric scores and ranges from 0 (very poor condition) to 90 (reference condition). Out of the metrics, 10 were evaluated using ArcGIS Pro® software (Version 2.7, Esri Inc.) and high resolution orthoimages (USDA 2018), and 10 were assessed in the field during a one-hour site visit per wetland during July - September 2020.

Statistical analyses

Statistical analyses were carried out using R statistical software (version 4.0.5) (R core team 2021) and the 'Vegan' software package (Oskansen et al. 2013). Nonmetric multidimensional scaling (NMDS) was used to visualize variation in larval fish community composition among wetland elevation (lowland, transitional, upland) and wetland condition (degraded, restored, reference). Experimental units in ordinations were defined as the average CPUE of all sampling events at each wetland in multivariate space. The input data for NMDS were defined by the CPUE of each larval fish taxa present from each wetland. No fish species were removed prior to ordination. Ordinations were constructed with Bray–Curtis distances, run for the maximum number of iterations (300), and chosen for minimum stress values. Dimensional solutions, stepping down from six to one, were tested and determined by the use and examination of individual scree plots (McCune & Grace 2002).

Analysis of variance using distance matrices (ADONIS) was used to statistically test larval fish community composition for differences among *a priori* groups of wetland elevation and condition. The Bray-Curtis distance was used in ADONIS and run for 9999 maximum iterations generating *r*-values between -1 and 1. Positive values indicate differences among groups and significance was tested at $\alpha = 0.05$.

Additionally, analysis of similarity percentages (SIMPER; Clarke 1993) was performed to make pairwise comparisons amongst wetland elevations. SIMPER assesses the contribution of individual species to the dissimilarity between objects in a Bray-Curtis dissimilarity matrix. P-values were considered significant at the $\alpha = 0.05$ level This allows the identification of species that are likely to be major contributors to differences between groups detected by methods such as ADONIS (Clarke and Warwick 2001).

The influence of environmental predictor variables (Table 1) was examined on larval fish community composition using vector fitting analysis with the function ‘ENVFIT’ (R package ‘vegan’) on NMDS ordinations. Each variable’s association with each experimental unit was indicated by the vector’s direction while the strength of each variable’s association was indicated by the vector’s length (McCune & Grace 2002). Vector significance ($\alpha = 0.05$) was estimated using 999 random permutations of the data (Faith & Norris 1989). For ease of interpretation, the hydrologic variables 1-Day Maximum (m), 30-Day Maximum (m), and 90-Day Maximum were grouped into one vector labeled High Water Magnitude on ordination plots because of high correlation (overlapping angle and vector magnitude) among variables (Flinn et al. 2008).

Diversity

Hill diversity was calculated for each wetland. Hill diversity was chosen due to its numerous advantages over other diversity indices (see Chao et al. 2014) and were calculated according to Hsieh et al. (2016), and Chao et al. (2014). See Jost (2006) for a more thorough review of the benefits of Hill diversity in relation to other indices or Roswell et al. (2021) for a current consensus within the ecological community.

Before calculating Hill diversity, samples were standardized by 90% coverage (Chao & Jost 2012; Chao et al. 2014; Roswell et al. 2021; R package iNEXT). Coverage is a relatively new method of sample standardization in ecology that measures sample completeness and accounts for the abundance of species in the sampled community. Coverage estimates the proportion of individuals in the community that belong to species present in a sample (Roswell et al. 2021). For example, achieving coverage of 90% means that 10% of individuals in the community were not sampled.

Hill diversity varies based on the choice of exponent used. Ecologists most commonly use $q = 0$ (species richness), $q = 1$ (Hill-Shannon diversity), and $q = 2$ (Hill-Simpson diversity). Hill-Shannon diversity ($q = 1$) was used because it results in all individuals being considered equally as it counts species proportionately to their abundance of incidence (Chao et al. 2014). Genus-level richness was used as opposed to species-level richness because of limitations in identification. Hill-Shannon diversity was calculated using the iNEXT package which provides functions to compute the most widely used Hill numbers ($q = 1$, $q = 2$, $q = 3$) for individual-based abundance data or sampling-unit based incidence data. Incidence data was used because it suitably represents our sampling methods and because Colwell et al. 2012, Chao et al. (2014), and Chao & Colwell (2017) demonstrated that incidence data allows for biological inference just as powerful as abundance-data based approaches. Our input data for the iNEXT package consisted of genus-specific incidence data from each sample from each wetland which was categorized by each sample's wetland elevation and condition.

To describe patterns in Hill-Shannon diversity, sample- and coverage-based rarefaction and extrapolation curves were generated using the "estimate d" function (R package iNEXT) to determine how diversity increases with increasing sampling effort and completeness. Rarefaction and extrapolation of Hill-Shannon diversity were conducted according to Hsieh et al. (2016) and further discussed in Colwell et al. (2012), Chao & Jost (2012), and Chao et al. (2014). Sample-based curves evaluated the number of individuals in a sample by plotting diversity estimates in relation to the number of sampling units. Coverage-based curves were plotted against rarefied sample completeness to illustrate diversity estimates in relation to sample coverage. All extrapolation curves were plotted using a doubling in sample size, and 500 bootstrap replicates were used to estimate 95% confidence intervals. Confidence intervals, a known alternative to standard statistical testing (Magurran 2004; Colwell et al. 2004), were used to determine if differences between wetland elevation and condition were statistically significant. Nonoverlapping 95% confidence intervals, whether rarefied or extrapolated, were considered to indicate significant differences at $\alpha = 0.05$ (Chao & Jost, 2012; Chao et al., 2014).

Results

Larval fish sampling

Larval fish were present at each wetland. In total, 2081 larval fish from 11 families and 16 genera were collected across all wetlands. The mean CPUE (abundance of taxa per survey) at lowland wetlands was 98 ± 76.3 (SE), 10 ± 3.00 (SE) at upland wetlands, and 8 ± 2.28 (SE) at transitional wetlands. Twelve of the 16 genera were collected in uplands wetlands, 10 genera in lowland wetlands, and 9 genera in transitional wetlands. *Dorosoma* (shads), *Ictiobus* (buffalo), *Lepisosteus* (gars), and *Pomoxis* (crappies) were only found in lowland wetlands. *Ameiurus* (bullheads), *Erimyzon* (chubsuckers), and *Umbra* (mudminnows) were only found in upland wetlands. The mean CPUE at restored wetlands was 43 ± 24.7 (SE), 8 ± 2.03 (SE) at reference wetlands, and 6 ± 3.33 (SE) at degraded wetlands. Fifteen of the 16 genera were collected in restored wetlands, 10 genera were collected in reference wetlands, and 5 genera were collected in degraded wetlands. Bullheads, shads, *Hypophthalmicthys* (bighead carp: invasive genus), buffalo, gars, and crappies were only found in restored wetlands. Mudminnows were only found in reference wetlands. *Lepomis* (true sunfish) was by far the most ubiquitous genus across wetland elevation and condition. Overall community composition of all wetlands was dominated by bighead carp (64%) and true sunfish (27%) with all other genera contributing ≤ 2% each.

<u>Differences in larval fish community composition among wetland elevation and condition</u>

NMDS ordination based on larval fish CPUE data revealed considerable separation by wetland elevation. Final NMDS solutions consisted of two dimensions with a relatively low final stress values (0.137) and high interpretability compared to alternative solutions (Figure 3). Results of ADONIS analysis confirmed that larval fish community composition among lowland, transitional, and upland wetlands was significantly different (r^2 = 0.282, p = 0.047).

SIMPER analysis identified genera that led to differences among wetland elevation (Table 2a, b, c). Bighead carp (54.2 %) and true sunfish (9.3%) contributed the most to the differences between lowland and transitional wetlands, although no genera's contribution was significant ($p > 0.05$) (Table 2a). True sunfish (28.1%) and *Elassoma* (pygmy sunfish) (8.5%) contributed the most to the differences between upland and transitional wetlands although no genera's contribution was significant ($p > 0.05$) (Table 2b). Bighead carp (50.2%) and true sunfish (25.8%) contributed the most to differences between lowland and upland wetlands. Bighead carp (p = 0.048) and crappies (p = 0.013) were found to be more important in lowland wetlands (Table 2c) and their contribution was significant to the differences between lowland and upland wetlands.

NMDS ordination based on larval fish CPUE data revealed little separation by wetland condition. Results of ADONIS confirmed that degraded, restored, and reference wetlands were not significantly different (r^2 = 0.109, p = 0.955).

<u>Environmental influences on larval fish community composition</u>

Significant environmental variables that helped explain differences in larval fish community composition were overlaid as vectors onto NMDS ordination (Figure 3, Table 3). Environmental vectors displayed a strong vertical gradient: increasing High Water Magnitude, Low Water Duration, and Percent Time Connected were all associated with each other and were negatively associated with Elevation, which also displayed a vertical gradient. Larval fish community composition based on wetland elevation displayed more of a horizontal gradient than vertical gradient, but still, followed the vertical gradient of Elevation. Larval fish community composition found in lowland wetlands were most associated with High Water Magnitude, Low Water Duration, and Percent Time Connected and negatively associated with Elevation. Larval fish community composition in upland wetlands had varying, but mostly negative association with these vectors. Transitional wetlands had little to no association with any vectors.

Influence of elevation on fish diversity

Rarefaction curve analysis did not detect significant differences in Hill-Shannon diversity among wetland elevation as overlap existed in the 95% confidence intervals among lowland 11.1, CI [7.3, 18.7], transitional 6.5, CI [5.5, 8.9], and upland wetlands 7.4, CI [6.6, 9,1] as the number of sampling units increased (Figure 4). The estimated curve patterns of Hill-Shannon diversity accumulation per sampling unit for transitional and upland wetlands were approaching asymptotic plateau, suggesting that the sampling strategy was sufficient in revealing true diversity associated with these wetlands. The estimated curve patterns of Hill-Shannon diversity accumulation per sampling unit for lowland wetlands, however, did not approach asymptotic plateau suggesting that the sampling strategy was insufficient in revealing the true diversity, likely leading to an underrepresentation of diversity. Coverage-based rarefaction and extrapolation further indicated that sample completeness was sufficient for transitional and upland wetlands as coverage values were greater than 90% (92% and 95% respectively) but was insufficient for lowland wetlands as coverage values (74%) were less than 90%.

Influence of wetland condition on diversity

Rarefaction curve analysis did not detect significant differences in Hill-Shannon diversity of larval fish communities among wetland condition as overlap existed in the 95% confidence intervals among degraded 7.8, CI [4.0, 16.0], restored 12.5, CI [9.4, 17.2], and reference wetlands 10.2, CI [7.4, 14.6] as the number of sampling units increased (Figure 5). The estimated curve pattern of Hill-Shannon diversity accumulation per sampling unit for restored wetlands was approaching asymptotic plateau, suggesting that the sampling strategy was sufficient in revealing true diversity associated with these wetlands. The estimated curve patterns of Hill-Shannon diversity accumulation per sampling unit for degraded and reference wetlands, however, did not approach asymptotic plateau suggesting that the sampling strategy was insufficient in revealing the true diversity associated with these wetlands leading to an under representation of degraded and reference wetland larval fish diversity. Underrepresentation of true diversity was also apparent in coverage-based rarefaction and extrapolation estimates, which indicated that sample completeness was insufficient for degraded (54%), restored (87.5%), and reference (83%) wetlands.

Discussion

Our results indicated that wetland elevation was an important factor in influencing larval fish community composition, likely via differences in wetland hydrology along elevation gradients. Other studies have found that elevation is important in determining wetland hydrology (Brinson 1993; Euliss et al. 2004). Lowland wetlands were greatly affected by the Mississippi River during its spring seasonal flooding due to their similar elevations (Figure 3). During periods of direct connectivity to the Mississippi River, lowland wetlands experienced high magnitude long-lasting flooding and most likely had lotic conditions during flooding. Upland wetlands experienced flooding but were never connected to the Mississippi River and, therefore, experienced lower magnitude and shorter duration flooding and were likely more lentic. Even though conditions in upland wetlands may lead to distinct communities, it was most likely high magnitude long-lasting flooding present in lowland wetlands drove the community differences we observed.

Bighead carp receive spawning cues from increased discharge typically associated with spring seasonal flooding (Hintz et al. 2017; but see Coulter et al. 2013). This association likely led to bighead carp's 50.2% contribution to the dissimilarity between lowland and upland wetlands (p = 0.048, Table 2c). It is unlikely that the bighead carp larvae found were the result of spawning that took place in lowland wetlands, as their eggs require flowing water to develop and may drift downstream over 100 km before hatching (George & Chapmann 2013; George et al. 2017). If spawning directly occurred in lowland wetland sites, eggs and larvae would surely drift much further downstream. Therefore, spawning likely occurred upstream of lowland wetlands, eggs drifted downstream, and eventually hatched in our wetlands. Once hatched, however, lowland wetlands likely provided larvae with suitable nursery habitat. Varble et al. (2007) found that floodplain environments are commonly utilized by bighead carp larvae as floodplains are productive environments that offer abundant food and warm temperature which result in fast growth rates. The pervasiveness of bighead carp throughout the Mississippi River is well documented (Chick & Pegg 2001; Pongruktham et al. 2010; Sass et al. 2010) and lowland wetlands may inherently be at greater risk of bighead carp invasion solely due to their high levels of connectivity to the river.

Crappie, which significantly contributed to the dissimilarity between lowland and upland wetlands (p = 0.013, Table 2c), are typically associated with lentic conditions, but populations are common in large river systems (Etnier & Starnes 1993). Despite many studies describing a strong dependence of many fishes on increased depth and lateral connectivity during seasonal flooding (Welcomme 1985; Junk et al. 1989; Zeug et al. 2005; Dembkowski & Miranda 2012), other studies have found that crappie are at best weakly correlated with increased depth and connectivity and instead are strongly correlated with shallower disconnected floodplain lakes (Miyazono et al. 2010; Alfermann & Miranda 2013). Our results suggest that crappie may be positively associated with the high connectivity present in lowland wetlands (Figure 3, Table 2c), which was most likely due to riverine populations utilizing floodplain habitat to spawn. Riverine crappie populations are commonly known to utilize the floodplain during spawning because they are demersal spawners, i.e., they require structure such as submergent aquatic or flooded terrestrial vegetation that the main channel does not afford (Phelps et al. 2009; Miranda et al. 2015). Recruitment of age-zero crappie has been found to suffer with deeper depths (Dagel & Miranda 2012). Even if spawning was successful in lowland wetlands, exceptional depths during seasonal flooding may have imposed negative implications on larval crappie recruitment.

Lowland wetlands experienced prolonged low water and eventually dried during the summer and fall (Figure 6) because they only flooded from the Mississippi River's seasonal flood pulse (winter and spring) and water levels were little affected by local precipitation events throughout the rest of the year (USGS Watershed Science School 2019; Berkowitz et al. 2020). Drying has obvious negative consequences on fishes such as physical stress, predation, and mortality. Additionally, Dembkowski & Miranda (2012) found that shallow depth is associated with harsh environmental conditions, i.e., low dissolved oxygen, high temperatures, which may cause depauperate fish assemblage composition and prevent larval fish recruitment (Beesley et al. 2012). During the summer of 2020, lowland wetlands experienced drying before they could hydrologically reconnect with the river causing local extinctions and prevented recruitment.

Lotic conditions serve as spawning cues for many fishes and certain species' eggs, or larvae require flows to drift downstream while developing (Welcomme 1985; Junk et al. 1989; Kluender et al 2015). It is probable that lotic conditions in lowland wetlands allowed other riverine species to access suitable spawning habitat on the floodplain. Lack of river fishes in our samples may have resulted from our sampling strategy or these fish may have emigrated from our wetlands before sampling occurred.

Similarities found in larval fish community composition among wetland condition may have resulted from exceptional spring flooding experienced throughout our study area during the spring of 2020. Community composition of fishes is more similar during floods than during low water periods because floods promote high levels of lateral connectivity which allows the exchange of fishes among river and floodplain habitat (Hamilton & Lewis 1990; Thomasz et al. 1997; Miranda 2005). Lack of differences in larval fish community composition among wetland condition may have also resulted from the identical fish species from similar source pools (e.g., Bayou du Chien Creek, Mayfield Creek, and Obion Creek) utilizing our wetlands as spawning habitat during spring flooding. For example, Centrarchidae are common in Mississippi River tributaries and are prolific floodplain dispersers during flooding (Alfermann & Miranda 2013). Furthermore, many of our sites were located very near one another, which most likely led to similarities in colonizing species.

The lack of differences in larval fish diversity amongst wetland elevation and condition (Figures 5, 6) were most likely due low taxonomic resolution and small sample sizes resulting in an underrepresentation of diversity. Sample sizes could have been increased by incorporating bi-monthly sampling or by sampling multiple spawning seasons over two or more years.

Management implications

Wetland elevation's influence on hydrologic conditions and its consequent influences on larval fish communities was evident throughout our study. When choosing sites for wetland restoration, restoration managers must consider meaningful tradeoffs that result from wetland elevation. The elevation of a restored wetland plays a large role in determining local wetland hydrology which is likely to in turn affect fish communities. Lowland elevation wetlands may be more strongly influenced by the Mississippi River, experience lotic conditions during seasonal floods, and dry more often. These conditions can promote spawning habitat for riverine species and important sport fish, i.e., crappie, but may also promote species invasions or prevent fish recruitment back into the Mississippi River. If managers are less concerned with providing benefits for fish, drying and subsequent large-scale die-offs in lowland wetlands may be beneficial to other wildlife or vegetation communities (Gawlik 2002; Benbow et al. 2020). Restoring wetlands at higher elevations may minimize the influence of the Mississippi River on wetland hydrology, which may promote spawning habitat for lentic fishes, nest, or demersal spawners. Lack of drying can help prevent large-scale die offs of larval fish and allow them to successfully recruit back into their respective populations during periods of higher hydrologic connectivity.

Limitations

Low abundances of larval fish were collected and may have been due to only employing dipnet surveys. Greater larval fish abundances could be achieved in the future by employing multiple sampling gears (e.g., dipnet and light trap surveys). Similar studies in the future may consider collecting data over multiple years to allow inter-year comparisons and provide greater inference into larval fish community patterns throughout the study area. Additionally, future studies may consider increasing the number of wetlands sampled and employing a balanced data set, i.e., equal number of wetlands based on elevation or condition categories.

Conclusions

The clear influence of elevation on larval fish communities found in this study may have implications for wetland restorations throughout the MAV and other large river floodplain ecosystems. Knowledge gaps associated with the consideration of landscape-level factors exist within large wetland restoration programs and therefore, quantifying the influence of wetland elevation on larval fish communities may provide wetland restoration managers with insight and direction when choosing sites for future restoration. Furthermore, wetland restoration managers may want to focus on wetland environmental conditions if wildlife response is a goal as our study found that hydrologic conditions associated with lowland wetlands may have assisted in promoting unique larval fish community composition. Even though wetland restoration did not lead to clear community differences when compared with degraded or reference wetlands, our study provides wetland restoration managers with important criteria to consider when wildlife response is a goal of restoration. Certainly, restoration practitioners will face and must consider tradeoffs associated with wetland restoration practices and, hopefully, our results may better inform future recommendations and restoration projects. Regardless, the need for future studies that span across multiple temporal and spatial scales to better understand how wetland restoration practices can influence the entire MAV regions still exists.

Literature Cited

Acreman, M., & Holden, J. (2013). How wetlands affect floods. Wetlands, 33(5), 773-786.

Adams, S. R., Flinn, M. B., Burr, B. M., Whiles, M. R., & Garvey, J. E. (2006). Ecology of larval blue sucker (Cycleptus elongatus) in the Mississippi River. Ecology of Freshwater Fish, 15(3), 291-300.

Alfermann, T. J., & Miranda, L. E. (2013). Centrarchid assemblages in floodplain lakes of the Mississippi alluvial valley. Transactions of the American Fisheries Society, 142(2), 323-332.

Auer, N. A. (Ed.). (1982). Identification of larval fishes of the Great Lakes basin with emphasis on the Lake Michigan drainage. Great Lakes Fishery Commission.

Baber, M. J., Childers, D. L., Babbitt, K. J., & Anderson, D. H. (2002). Controls on fish distribution and abundance in temporary wetlands. Canadian Journal of Fisheries and Aquatic Sciences, 59(9), 1441-1450.

Balcer, M. D., N. L. Korda, and S. I. Dodson. 1984. Zooplankton of the Great Lakes: A guide to the identification and ecology of the common crustacean species. Journal of Great Lakes Research 10:334.

Bayley, P. B. (1991). The flood pulse advantage and the restoration of river-floodplain systems. Regulated Rivers: Research & Management, 6(2), 75-86.

Bedford, B. L. 1996. The need to define hydrologic equivalence at the landscape scale for freshwater wetland mitigation. Wiley 6:57–68.

Benbow, M. E., Receveur, J. P., & Lamberti, G. A. (2020). Death and decomposition in aquatic ecosystems. Frontiers in Ecology and Evolution, 8, 17.

Berkowitz, J. F., Johnson, D. R., & Price, J. J. (2020). Forested Wetland hydrology in a large Mississippi river tributary system. Wetlands, 40(5), 1133-1148.

Brauman, K., G. C. Daily, T. K. Duarte, and H. A. Mooney (2007), The nature and value of ecosystem services: An overview highlighting hydrologic services, Annu. Rev. Environ. Resources., 32, 67–98.

Brinson, M. M. (1993). A hydrogeomorphic classification for wetlands.

Brinson, M. M., and R. D. Rheinhardt. 1998. Wetland functions and relations to societal values. Pages 29–48 in M. G. Messina and W. H. Conner, editors. Southern forested wetlands ecology and management. Lewis Publishers, Boca Raton, Florida, USA.

Chao, A., & Jost, L. (2012). Coverage-based rarefaction and extrapolation: standardizing samples by completeness rather than size. Ecology, 93(12), 2533-2547.

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

Chick, J. H., & Mclvor, C. C. (1997). Habitat selection by three littoral zone fishes: effects of predation pressure, plant density and macrophyte type. Ecology of Freshwater Fish, 6(1), 27-35.

Chick, J. H., & Pegg, M. A. (2001). Invasive carp in the Mississippi River basin. Science, 292(5525), 2250-2251.

Clarke KR (1993) Non-parametric multivariate analyses of changes in community structure. Aust J Ecol 18:117–14.

Clarke KR, Warwick RM (2001) Change in marine communities: an approach to statistical analysis and interpretation. PRIMER-E, Plymouth.

Colwell, R. K. et al. 2012. Models and estimators linking individual-based and sample-based rarefaction, extrapolation and comparison of assemblages. – J. Plant Ecol. 5: 3–21.

Colwell, R. K., Mao, C. X., & Chang, J. (2004). Interpolating, extrapolating, and comparing incidence-based species accumulation curves. Ecology, 85, 2717–2727. https://doi.org/10.1890/03-0557.

Corstanje, R. and K. R. Reddy. 2004. Response of biogeochemical indicators to a drawdown and subsequent re-flood. Journal of Environmental Quality 33:2357–66.

Coulter, A. A., Bailey, E. J., Keller, D., & Goforth, R. R. (2016). Invasive Silver Carp movement patterns in the predominantly free-flowing Wabash River (Indiana, USA). Biological Invasions, 18(2), 471-485.

Cowardin, L. M., & Golet, F. C. (1995). US Fish and Wildlife Service 1979 wetland classification: A review. Classification and inventory of the world's wetlands, 139-152.

Dagel, J. D., & Miranda, L. E. (2012). Backwaters in the upper reaches of reservoirs produce high densities of age-0 crappies. North American Journal of Fisheries Management, 32(4), 626-634.

Dembkowski, D. J., & Miranda, L. E. (2012). Hierarchy in factors affecting fish biodiversity in floodplain lakes of the Mississippi Alluvial Valley. Environmental Biology of Fishes, 93(3), 357-368.

Dettmers, J. M., Wahl, D. H., Soluk, D. A., & Gutreuter, S. (2001). Life in the fast lane: fish and foodweb structure in the main channel of large rivers. Journal of the North American Benthological Society, 20(2), 255-265.

Dewitz J (2019) National Land Cover Database (NLCD) 2016 Products: United States Geological Survey data release, https://doi.org/10.5066/P96HHBIE (accessed 16 February 2021).

Euliss, N. H., LaBaugh, J. W., Fredrickson, L. H., Mushet, D. M., Laubhan, M. K., Swanson, G. A., ... & Nelson, R. D. (2004). The wetland continuum: a conceptual framework for interpreting biological studies. Wetlands, 24(2), 448-458.

Faith, D. P., and R. H. Norris. 1989. Correlation of environmental variables with patterns of distribution and abundance of common and rare freshwater macroinvertebrates. Biological Conservation 50:77–98.

Faulkner, S. P., W. H. Patrick, Jr, and R. P. Gambrell. 1992. Field techniques for measured wetland soil parameters. Journal of Environmental Quality 53:883–90.

Faulkner, S., Barrow Jr, W., Keeland, B., Walls, S., & Telesco, D. (2011). Effects of conservation practices on wetland ecosystem services in the Mississippi Alluvial Valley. Ecological Applications, 21(sp1), S31-S48.

Flinn, M. B., Adams, S. R., Whiles, M. R., & Garvey, J. E. 2008. Biological responses to contrasting hydrology in backwaters of Upper Mississippi River Navigation Pool 25. Environmental management, 41(4), 468.

Fredrickson, L. 2005. Contemporary bottomland hardwood systems: structure, function and hydrologic condition resulting from two centuries of anthropogenic activities. Ecology and management of bottomland hardwood systems 10, 19-35.

Gawlik, D. E. (2002). The effects of prey availability on the numerical response of wading birds. Ecological monographs, 72(3), 329-346.

George, A. E., & Chapman, D. C. (2013). Aspects of embryonic and larval development in bighead carp Hypophthalmichthys nobilis and silver carp Hypophthalmichthys molitrix. PloS one, 8(8), e73829.

George, A. E., Garcia, T., & Chapman, D. C. (2017). Comparison of size, terminal fall velocity, and density of bighead carp, silver carp, and grass carp eggs for use in drift modeling. Transactions of the American Fisheries Society, 146(5), 834-843.

Haynes, R. J., & Egan, D. (2004). The development of bottomland forest restoration in the Lower Mississippi River Alluvial Valley. Ecological Restoration, 22(3), 170-182.

Hintz, W. D., Glover, D. C., Szynkowski, B. C., & Garvey, J. E. (2017). Spatiotemporal reproduction and larval habitat associations of nonnative Silver Carp and Bighead Carp. Transactions of the American Fisheries Society, 146(3), 422-431.

Hsieh, T. C., Ma, K. H., & Chao, A. (2016). iNEXT: An R package for interpolation and extrapolation of species diversity (Hill numbers). Methods in Ecology and Evolution: Under revision. 7(12): 1451-1456. https://doi. org/10.1111/2041 210X.12613.

Hu, S., Niu, Z., Chen, Y., Li, L., & Zhang, H. (2017). Global wetlands: Potential distribution, wetland loss, and status. Science of the Total Environment, 586, 319-327.

Hunter, R. G., Faulkner, S. P., & Gibson, K. A. (2008). The importance of hydrology in restoration of bottomland hardwood wetland functions. Wetlands, 28(3), 605-615.

Jost, L. (2006). Entropy and diversity. Oikos, 113, 363–375. https://doi.org/10.1111/oik.2006.113.issue-2.

Junk, W. J., Bayley, P. B., & Sparks, R. E. (1989). The flood pulse concept in river-floodplain systems. Canadian special publication of fisheries and aquatic sciences, 106(1), 110-127.

Kentucky Division of Water. 2016. Guidance Manual for KY-WRAM, Version 3.0. https://eec.ky.gov/Environmental-Protection/Water/Monitor/Pages/KYWRAM.aspx (accessed 1 July 2021).

King, S. L., & Keeland, B. D. (1999). Evaluation of reforestation in the lower Mississippi River alluvial valley. Restoration Ecology, 7(4), 348-359.

King, S. L., D. J. Twedt, and R. R. Wilson. 2006. The Role of the Wetland Reserve Program in conservation efforts in the Mississippi River Alluvial Valley. Wildlife Society Bulletin 34:914–920.

King, S. L., & Keim, R. F. (2019). Hydrologic modifications challenge bottomland hardwood forest management. Journal of Forestry, 117(5), 504-514.

Kluender, E. R., Adams, R., & Lewis, L. (2017). Seasonal habitat use of Alligator Gar in a river–floodplain ecosystem at multiple spatial scales. Ecology of Freshwater Fish, 26(2), 233-246.

Lubinski, K. S., Carmody, G., Wilcox, D., & Drazkowski, B. (1991). Development of water level regulation strategies for fish and wildlife, Upper Mississippi River System. Regulated Rivers: Research & Management, 6(2), 117-124.

Magurran, A. E. (1988). Why diversity? Ecological diversity and its measurement (pp. 1–5). Dordrecht, Netherlands: Springer Netherlands. https:// doi.org/10.1007/978-94-015-7358-0_1.

Mapes, R. L., DuFour, M. R., Pritt, J. J., & Mayer, C. M. (2015). Larval fish assemblage recovery: a reflection of environmental change in a large, degraded river. Restoration Ecology, 23(1), 85-93.

McCune B, Grace JB (2002) Analysis of ecological communities. MjM Software Design, Oregon.

Merkle, J. A., Anderson, N. J., Baxley, D. L., Chopp, M., Gigliotti, L. C., Gude, J. A., ... & VanBeek, K. R. (2019). A collaborative approach to bridging the gap between wildlife managers and researchers. *The Journal of Wildlife Management*, *83*(8), 1644-1651.

Miranda LE (2005) Fish assemblages in oxbow lakes relative to connectivity with the Mississippi River. Trans Am FishSoc 134:1480–1489.

Miranda LE (2010) Depth as an organizer of fish assemblages in floodplain lakes. Aquat Sci 73:211–221.

Miranda LE, Lucas GM (2004) Determinism in fish assemblages of floodplain lakes of the vastly disturbed Mississippi Alluvial Valley. TransAmFish Soc 133:358–370.

Miranda, L. E., Dagel, J. D., Kaczka, L. J., Mower, E. B., & Wigen, S. L. (2015). Floodplains within reservoirs promote earlier spawning of white crappies Pomoxis annularis. Environmental biology of fishes, 98(1), 469-476.

Mitsch, W. J., & Gosselink, J. G. (2015). Wetlands. John Wiley & Sons.

Miyazono, S., Aycock, J. N., Miranda, L. E., & Tietjen, T. E. (2010). Assemblage patterns of fish functional groups relative to habitat connectivity and conditions in floodplain lakes. Ecology of Freshwater Fish, 19(4), 578-585.

Moreno-Mateos, D., Power, M. E., Comín, F. A., & Yockteng, R. (2012). Structural and functional loss in restored wetland ecosystems. PLoS Biol, 10(1), e1001247.

Omernik, J. M. (1987). Ecoregions of the conterminous United States. Annals of the Association of American geographers, 77(1), 118-125.

Oskansen J, Blanchet FG, Friendly M, Kindt R, Legendre P, McGlinn D, Minchin PR, O'Hara RB, Simpson GL, Solymos P, Stevens MHH, Szoecs E, Wagner H (2013) vegan: Community ecology package. R package version 2.0-7. https://CRAN.R-project.org/package=vegan Accessed 4 June 2021.

Osmond, D. L., Meals, D. W., Hoag, D. L., Arabi, M., Luloff, A. E., Jennings, G. D., ... & Line, D. E. (2012). Synthesizing the Experience of the 13 National Institute of Food and Agriculture-Conservation Effects Assessment Project Watershed Studies: Present and Future. How to Build Better Agricultural Conservation Programs to Protect Water Quality: The National Institute of Food and Agriculture-Conservation Effects Assessment Project Experience. Soil and Water Conservation Society, Ankeny, IA.

Ostrand, K. G., & Wilde, G. R. (2004). Changes in prairie stream fish assemblages restricted to isolated streambed pools. Transactions of the American Fisheries Society, 133(6), 1329-1338.

Pennak, R. 1962. Quantative zooplankton sampling in littoral vegetation areas. Limnology and Oceanography.

Phelps, Q. E., Lohmeyer, A. M., Wahl, N. C., Zeigler, J. M., & Whitledge, G. W. (2009). Habitat characteristics of black crappie nest sites in an Illinois impoundment. North American Journal of Fisheries Management, 29(1), 189-195.

Poff, N. L., Olden, J. D., Merritt, D. M., & Pepin, D. M. (2007). Homogenization of regional river dynamics by dams and global biodiversity implications. Proceedings of the National Academy of Sciences, 104(14), 5732-5737.

Pongruktham, O., Ochs, C., & Hoover, J. J. (2010). Observations of silver carp (Hypophthalmichthys molitrix) planktivory in a floodplain lake of the lower Mississippi River basin. Journal of Freshwater Ecology, 25(1), 85-93.

Rewa, C. A. (2005). Wildlife Benefits of the Wetlands Reserve Program.

R Core Team (2015). *R: A Language and Environment for Statistical Computing*. Vienna: R Foundation for Statistical Computing.

Richter, B. D., Baumgartner, J. V., Powell, J., & Braun, D. P. (1996). A method for assessing hydrologic alteration within ecosystems. Conservation biology, 10(4), 1163-1174.

Roswell, M., Dushoff, J., & Winfree, R. (2021). A conceptual guide to measuring species diversity. Oikos, 130(3), 321-338.

Sass, G. G., Cook, T. R., Irons, K. S., McClelland, M. A., Michaels, N. N., Matthew O'Hara, T., & Stroub, M. R. (2010). A mark-recapture population estimate for invasive silver carp (Hypophthalmichthys molitrix) in the La Grange Reach, Illinois River. Biological Invasions, 12(3), 433-436.

Semlitsch, R. D. (2000). Size does matter: the value of small, isolated wetlands. National Wetlands Newsletter, 22(1), 5-6.

Sparks, R. E., Nelson, J. C., & Yin, Y. (1998). Naturalization of the flood regime in regulated rivers: the case of the upper Mississippi River. BioScience, 48(9), 706-720.

Tockner, K., Malard, F., & Ward, J. V. (2000). An extension of the flood pulse concept. Hydrological processes, 14(16-17), 2861-2883.

USGS National Map 3DEP Downloadable Data Collection: U.S. Geological Survey. https://www.sciencebase.gov/catalog/item/4f70aa9fe4b058caae3f8de5. Accessed 30 September 2021.

U.S. Geological Survey, 2020, National Water Information System data available on the World Wide Web (USGS Water Data for the Nation), accessed May 25, 2021, at URL https://waterdata.usgs.gov/nwis/inventory?agency_code=USGS&site_no=07024000.

USGS Watershed Science School, 2019, Streamflow and the Water Cycle, accessed June 27, 2022, at URL https://www.usgs.gov/special-topics/water-science-school/science/streamflow-and-water-cycle.

Varble, K. A., Hoover, J. J., George, S. G., Murphy, C. E., & Killgore, K. J. (2007). Floodplain wetlands as nurseries for silver carp, Hypophthalmichthys molitrix: A conceptual model for use in managing local populations. Engineer Research and Development Center Vicksburg MS.

Welcomme, R. L. (1985). River fisheries (No. 262). FAO.

Winemiller, K. O., & Rose, K. A. (1992). Patterns of life-history diversification in North American fishes: implications for population regulation. Canadian Journal of Fisheries and aquatic sciences, 49(10), 2196-2218.

Winter, T. C., Rosenberry, D. O., Buso, D. C., & Merk, D. A. (2001). Water source to four US wetlands: implications for wetland management. Wetlands, 21(4), 462-473.

Zedler, J. B. (2000). Progress in wetland restoration ecology. Trends in ecology & evolution, 15(10), 402-407.

Zeug, S. C., Winemiller, K. O., & Tarim, S. (2005). Response of Brazos River oxbow fish assemblages to patterns of hydrologic connectivity and environmental variability. Transactions of the American Fisheries Society, 134(5), 1389-1399.

Figures and tables

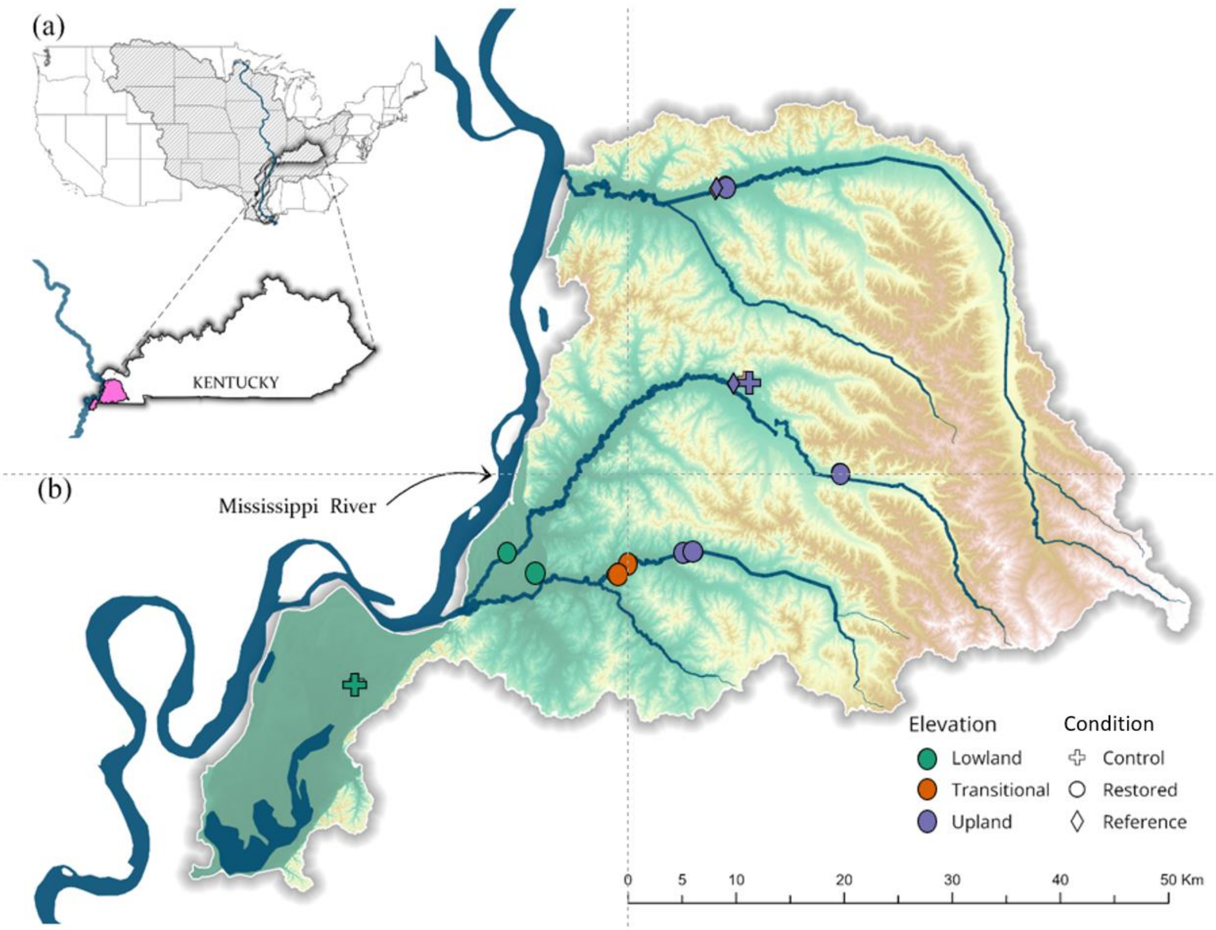

Figure 1. (A) Location of study area in western Kentucky and part of the Mississippi River watershed, USA. (B) Twelve study wetlands were classified by wetland elevation (lowland (n=3), transitional (n=2), upland (n=7)) denoted by symbol color and wetland condition denoted by shape (degraded (n=2), restored (n=8), reference (n=2)) (land use classifications from 2018 USDA National Imagery Program).

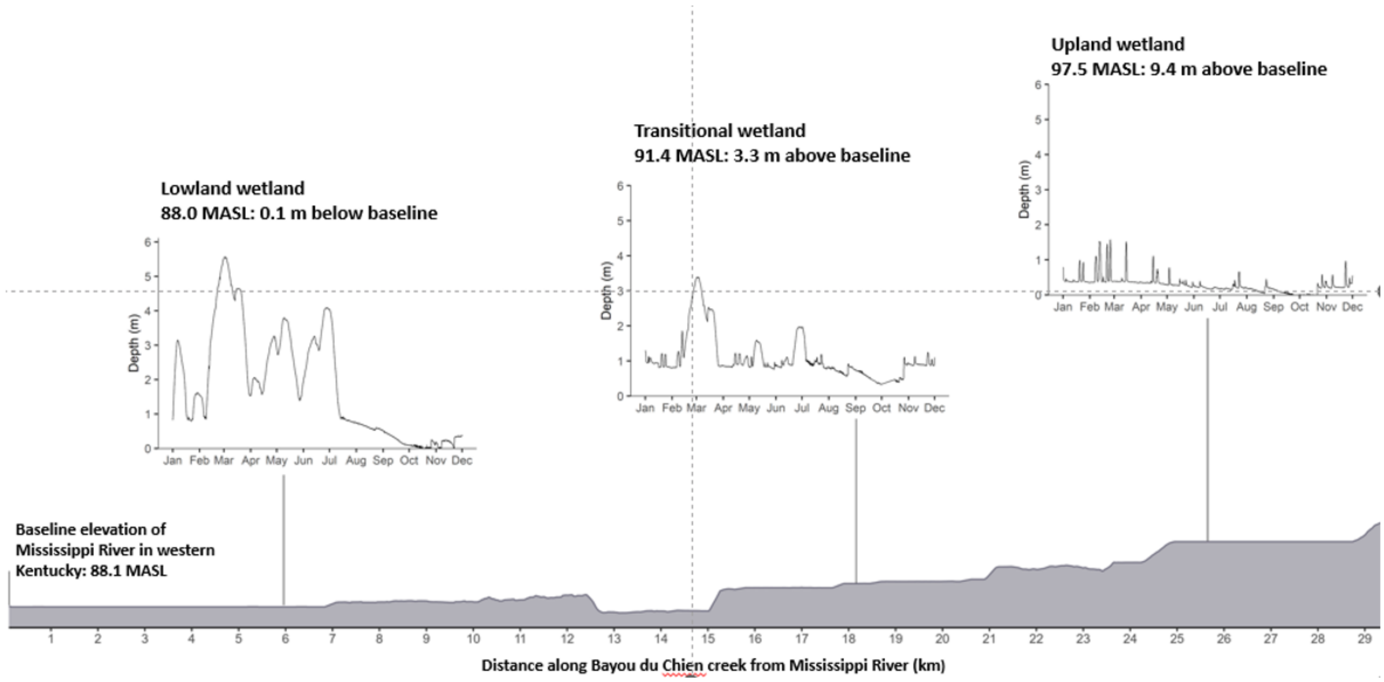

Figure 2: Location of a lowland (left, elevation < 91 MASL), transitional (middle, elevation > 91 but < 97 MASL), and upland (right, elevation > 97 MASL) along the elevation gradient of a tributary (Bayou du Chien) to the Mississippi River, USA. Distance from the Mississippi River (km) is measured along the tributary. One year of mean daily surface water depth (m) readings (taken during 2020) are pictured above each wetland location.

Table 1: Environmental metrics (26) considered for vector analysis in NMDS ordinations measured in wetlands throughout western Kentucky, USA. Each metric was calculated using data from the entire sampling period for each wetland in our study. Indicators of hydrologic alteration following Richter et al. (1996) are denoted with the abbreviation 'IHA'.

Metric	Definition	Measures
Hydrologic metrics		
Mean Depth (m)	The average wetland depth.	Magnitude
Hydroperiod	The number of days a wetland had water.	Duration
Percent Time Connected	The percent of time a wetland exhibited lateral connectivity to its nearest stream (USGS-NWIS).	Connectivity
1-Day Maximum (m) (IHA)	The maximum single day depth recorded.	Magnitude/Duration
1-Day Minimum (m) (IHA)	The minimum single day depth recorded.	Magnitude/Duration
7-Day Maximum (m) (IHA)	The maximum 7-day rolling average recorded.	Magnitude/Duration
7-Day Minimum (m) (IHA)	The minimum 7-day rolling average recorded.	Magnitude/Duration
30-Day Maximum (m) (IHA)	The maximum 30-day rolling average recorded.	Magnitude/Duration
30-Day Minimum (m) (IHA)	The minimum 30-day rolling average recorded.	Magnitude/Duration
90-Day Maximum (m) (IHA)	The maximum 90-day rolling average recorded.	Magnitude/Duration
90-Day Minimum (m) (IHA)	The minimum 90-day rolling average recorded.	Magnitude/Duration
Minimum Date (day of the year) (IHA)	The date the lowest single recorded depth occurred.	Timing
Maximum Date (day of the year) (IHA)	The date the greatest single recorded depth occurred.	Timing
Low Water Duration (days) (IHA)	The number of consecutive days depths stayed below the 25th percentile.	Duration
High Water Duration (days) (IHA)	The number of consecutive days depths stayed above the 75th percentile.	Duration
Rise Count (IHA)	The number of occurrences that wetland depth rose from the previous day.	Connectivity
Water quality metrics		
Mean Dissolved Oxygen (DO) (mg/L)	The average DO measurement.	Magnitude
Minimum Dissolved Oxygen (DO) (mg/L)	The minimum single day DO recorded.	Magnitude
Mean Temperature (C°)	The average temperature recorded.	Magnitude

1-Day Maximum (C°)	The maximum single day temperature recorded.	Magnitude
1-Day Minimum (C°)	The minimum single day temperature recorded.	Magnitude
<u>Biotic metrics</u>		
Vegetation	The average aquatic vegetation scores.	Habitat
Zooplankton Density	Average of each wetland's zooplankton density	Food resources
KY-WRAM Score	The KY-WRAM score indication ecological integrity (0-100) (100 indicates highest quality).	Disturbance
<u>Hydrologic connectivity</u>		
Distance to Main Channel (m)	The distance to the main channel of the nearest stream.	Connectivity
Elevation (m)	The median of elevation (n=50) (USGS Stream Stats).	Connectivity

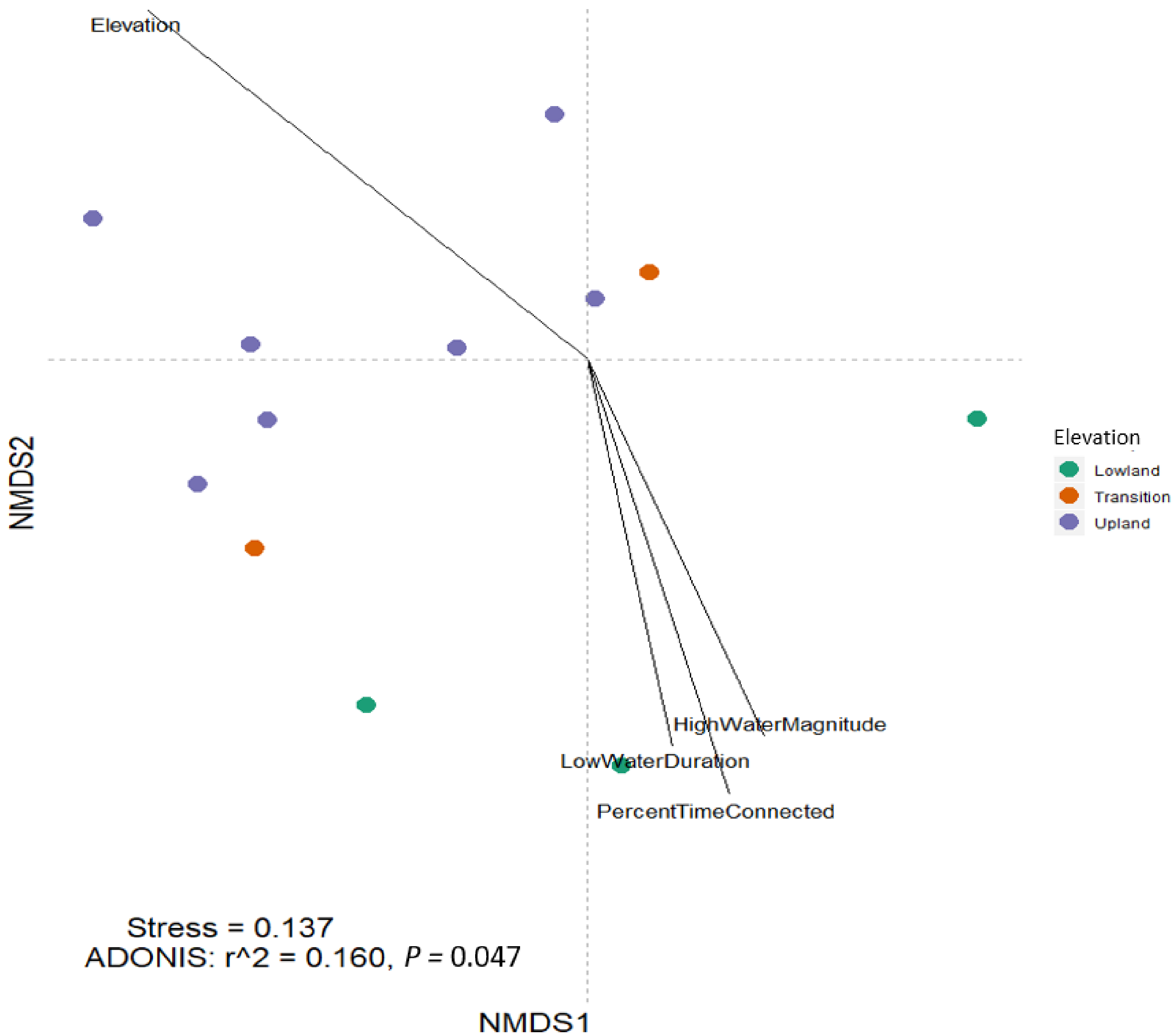

Figure 3: NMDS ordination of larval fish community composition in western Kentucky, USA wetlands. Ordination is based on per taxa CPUE from dipnet surveys that occurred monthly from March 2020 to August 2020. Symbol colors indicate wetland elevation (lowland, transitional, upland). All variables from Table 1 were tested and only significant variables were placed onto ordination as vectors. The vector High Water Magnitude is a combination of the metrics 1-Day Maximum (m), 30-Day Maximum (m), 90-Day Maximum (m). See table 1 for definitions of other variables used as vectors.

Table 2: Summary of SIMPER results for each fish genera from pairwise comparisons amongst (a) lowland vs transitional wetlands, (b) upland vs transitional wetlands, and (c) lowland vs upland wetlands: average abundance of individual species from each wetland, their average contribution (%) to overall dissimilarity, and each species associated p-value. P-values were considered significant at the α = 0.05 level.

(a) Lowland vs transitional	Lowland average abundance	Transitional average abundance	Contribution	P-value
Hypopthalimicthys	109.7	0.67	54.2	0.068
Lepomis	1.08	3.33	9.3	0.936
Elassoma	0.11	1.75	6.9	0.222
Notemigonus	0.67	0.42	2.1	0.233
Micropterus	0.42	0.00	1.5	0.775
Pomoxis	0.14	0.00	0.9	0.060
Fundulus	0.00	0.17	0.8	0.449
Esox	0.00	0.17	0.8	0.348
Aphredoderus	0.00	0.08	0.4	0.798
Dorosoma	0.08	0.00	0.3	0.262
Ictiobus	0.25	0.00	0.1	0.288
Lepisosteus	0.17	0.00	0.1	0.288
Umbra	0.00	0.00	0.0	1.00
Ameiurus	0.00	0.00	0.0	1.00
Erimyzon	0.00	0.00	0.0	1.00

(b) Upland vs transitional	Upland average abundance	Transitional average abundance	Contribution	P-value
Lepomis	11.7	3.33	28.1	0.476
Elassoma	0.48	1.75	8.5	0.081
Hypopthalimicthys	0.00	0.67	3.8	0.958
Notemigonus	0.21	0.42	2.4	0.107
Micropterus	0.55	0.0	2.0	0.650
Aphredoderus	0.29	0.08	1.6	0.419
Ameiurus	0.59	0.0	1.5	0.410
Fundulus	0.95	0.17	1.2	0.202
Esox	0.17	0.17	1.0	0.099
Erimyzon	0.07	0.0	0.4	0.356
Umbra	0.24	0.0	0.2	0.379
Pomoxis	0.00	0.0	0.0	1.00
Dorosoma	0.00	0.0	0.0	1.00
Lepisosteus	0.00	0.0	0.0	1.00
Ictiobus	0.00	0.0	0.0	1.00

(c) Upland vs lowland	Upland average abundance	Lowland average abundance	Contribution	P-value
Hypopthalimicthys	0.00	109.7	50.2	0.048*
Lepomis	11.7	1.08	25.8	0.602
Micropterus	0.55	0.42	1.98	0.642
Elassoma	0.47	0.11	1.48	0.943
Aphredoderus	0.29	0.00	1.29	0.630
Ameiurus	0.56	0.00	1.01	0.759
Notemigonus	0.21	0.67	1.00	0.911
Pomoxis	0.00	0.14	0.80	0.013*
Esox	0.17	0.00	0.04	0.974
Fundulus	0.10	0.00	0.04	0.849
Erimyzon	0.07	0.00	0.04	0.561
Dorosoma	0.00	0.08	0.03	0.091
Umbra	0.24	0.00	0.02	0.552
Ictiobus	0.00	0.25	0.01	0.212
Lepisosteus	0.00	0.17	0.01	0.212

Table 3: Correlation coefficients and p-values for vectors placed onto NMDS ordination that had significant associations with larval fish community composition.

Vector	Correlation Coefficient (r^2)	P-Value
Percent time connected	0.736	0.001
Elevation	0.777	0.002
High water magnitude	0.669	0.009
Low water duration	0.529	0.032

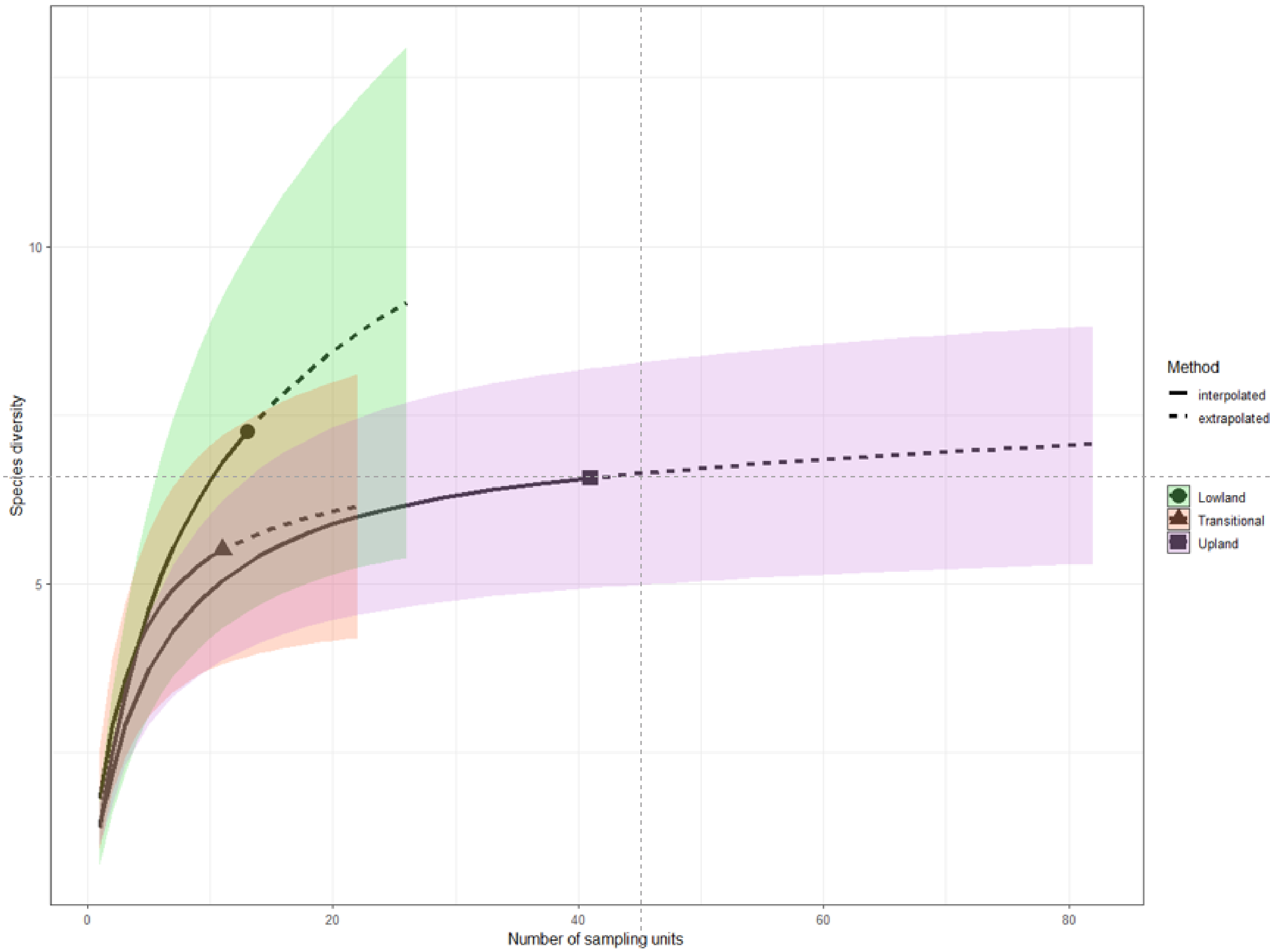

Figure 4: Hill-Shannon diversity estimates of wetland larval fish communities by wetland elevation using incidence-based rarefaction and extrapolation. Curves are based on larval fish dipnet survey data collected from March 2020 through August 2020 in western Kentucky, USA. All extrapolation curves were plotted to achieve 90% coverage.

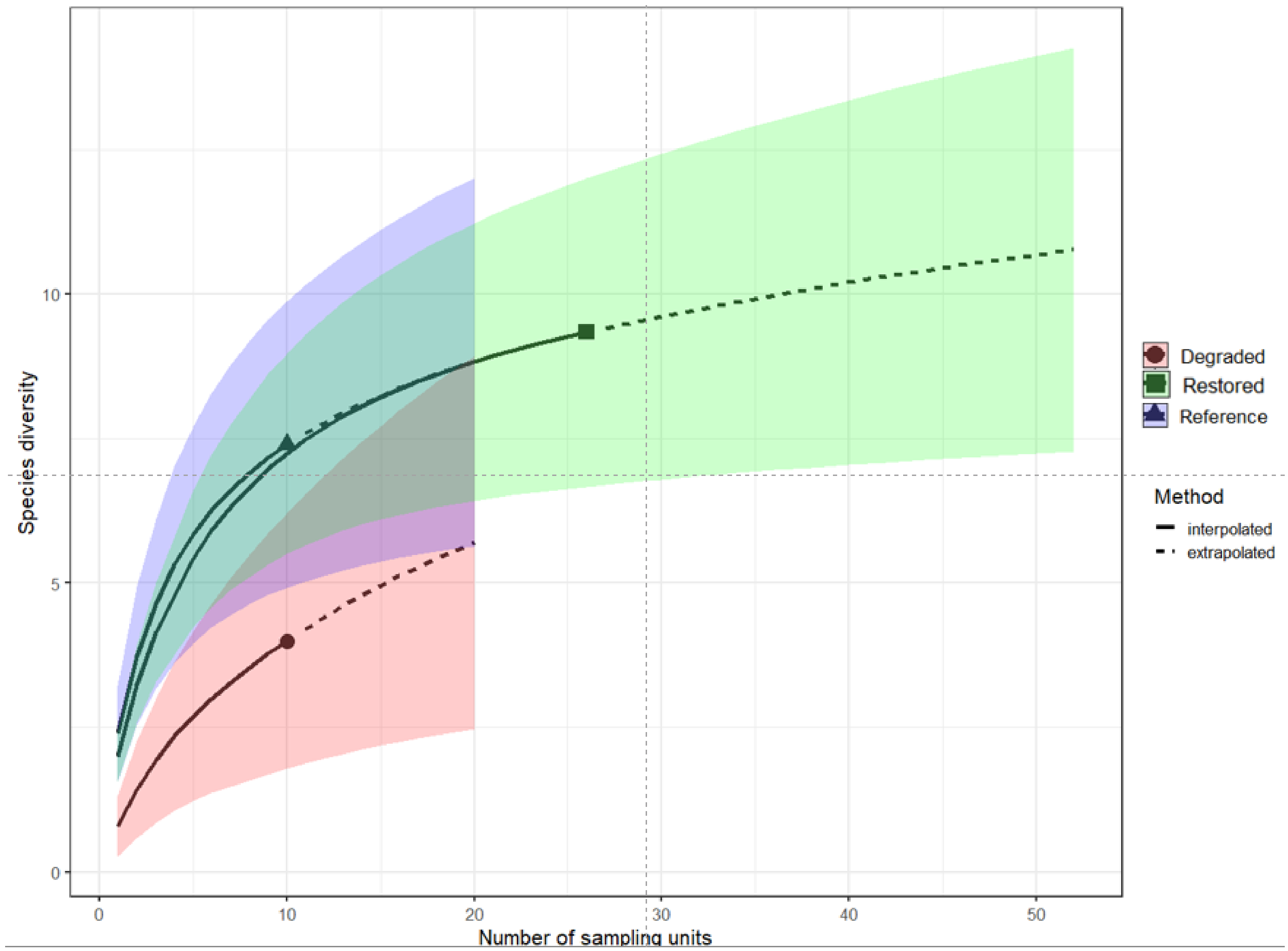

Figure 5: Hill-Shannon diversity estimates of wetland larval fish communities by wetland condition using incidence-based rarefaction and extrapolation. Curves are based on larval fish dipnet survey data collected from March 2020 through August 2020 in western Kentucky, USA. All extrapolation curves were plotted to achieve 90% coverage.

Figure 6: Photograph of dramatic low water/drying in a lowland wetland in western Kentucky, USA. Photograph was taken in August 2020.

Appendix

Supplemental tables

Table 1: Site name, wetland condition, wetland elevation, county, easement acreage, and wetland acreage of wetland sampled in western Kentucky, USA.

Site	Wetland condition	Wetland elevation	County	Acreage	Wetland acreage
ALEN	Restored	Upland	Hickman	68	49
BCYP	Degraded	Lowland	Fulton	NA	2
COFY	Restored	Lowland	Fulton	251	34
GDMN	Restored	Lowland	Fulton	115	24
GUTH	Restored	Upland	Graves	141	2
HEST	Restored	Transitional	Hickman	39	20
HOPK	Restored	Upland	Hickman	44	4
HWST	Restored	Transitional	Hickman	35	19
OBOT	Reference	Upland	Carlisle	NA	72
OWMA	Degraded	Upland	Carlisle	NA	5
SARC	Reference	Upland	Carlisle	NA	2
SWAN	Restored	Upland	Carlisle	784	9

Table 2: Presence/absence of every taxon collected at each wetland. White rows represent species-level presence/absence data of adult fish community collected with backpack electrofishing. Green rows represent genus-level presence/absence data of larval fish community collected with dipnet surveys. Orange row represents family-level presence/absence data (due to difficulty in identification) of larval fish community collected with dipnet surveys. 'X' indicates species was present at a wetland. '*' indicates larval genus was present at a wetland. '***' indicates special note: identification of *Hybognathus hayi* is pending on verification from Kentucky's state ichthyologist; the *Lepomis marginatus* we collected within the Bayou du Chien watershed were the first species records collected within that watershed (verified by Kentucky's state ichthyologist).

Genus/Species	ALEN	BCYP	COFY	GDMN	GUTH	HEST	HOPK	HWST	OBOT	OWMA	SARC	SWAN
Ameiurus												*
Ameiurus melas	X	X		X		X	X	X		X	X	X
Ameiurus nebulosus		X				X		X				
Ameiurus natalis	X			X	X	X	X	X		X		X
Amia												
Amia calva	X	X				X	X	X	X	X	X	X
Aphredoderus					*	*			*		*	*
Aphredoderus sayanus	X	X	X	X		X	X	X	X	X	X	X
Aplodinotus												
Aplodinotus grunniens				X								
Centrarchus												
Centrarchus macropterus	X	X		X		X	X	X	X	X	X	X
Ctenopharyngodon												
Ctenopharyngodon idella			X	X								
Cycleptus												
Cycleptus elongatus				X								
Cyprinella												
Cyprinella lutrensis											X	
Cyprinella venusta						X						
Cyprinidae			*	*		*					*	
Cyprinus												
Cyprinus carpio			X	X		X	X					X

Genus/Species	ALEN	BCYP	COFY	GDMN	GUTH	HEST	HOPK	HWST	OBOT	OWMA	SARC	SWAN
Dorosoma			*									
Dorosoma cepedianum			X	X		X	X	X				X
Elassoma	*	*				*	*	*	*		*	*
Elassoma zonatum	X	X			X	X	X	X	X	X	X	X
Erimyzon									*			*
Erimyzon sucetta	X					X	X	X	X	X		X
Esox							*	*		*	*	*
Esox americanus	X	X				X	X	X	X	X	X	X
Etheostoma												
Etheostoma asprigene						X					X	
Etheostoma chlorosomum			X			X					X	
Etheostoma gracile					X	X	X	X	X	X	X	X
Fundulus						*					*	
Fundulus chrysotus		X										
Fundulus olivaceus			X			X		X	X		X	X
Gambusia												
Gambusia affinis	X	X	X	X	X	X	X	X	X	X	X	X
Hybognathus												
*Hybogntahus hayi****											X	X
Hypophthalmichthys			*	*		*						
Hypophthalmichthys molitrix						X						
Ictalurus												
Ictalurus punctatus				X								
Ictiobus				*								
Ictiobus bubalus			X	X		X	X	X		X	X	X
Labidesthes												
Labidesthes sicculus	X					X					X	

Genus/Species	ALEN	BCYP	COFY	GDMN	GUTH	HEST	HOPK	HWST	OBOT	OWMA	SARC	SWAN
Lepisosteus				*								
Lepisosteus oculatus	X					X		X	X		X	
Lepisosteus osseus			X	X		X					X	
Lepomis	*	*	*	*	*	*	*	*	*	*	*	*
Lepomis cyanellus	X	X	X	X	X	X	X	X	X	X	X	X
Lepomis humilis			X	X			X				X	
Lepomis macrochirus	X	X	X	X		X	X	X	X		X	X
Lepomis marginatus	X		X	X		X	X	X	X	X		
Lepomis megalotis			X	X		X					X	
Lepomis microlophus	X			X							X	
Lepomis miniatus	X										X	X
Lepomis symmetricus	X	X		X		X	X	X	X	X	X	X
Lepomis gulosus	X	X	X	X	X	X	X	X	X	X	X	X
Menidia												
Menidia beryllina				X								
Micropterus	*		*				*				*	*
Micropterus salmoides	X	X	X	X		X	X	X	X	X	X	X
Minytrema												
Minytrema melanops	X					X			X		X	
Morone												
Morone mississippiensis				X								
Notemigonus						*			*	*	*	*
Notemigonus crysoleucas	X	X	X	X		X	X	X	X	X	X	X
Notropis												
Notropis maculatus											X	
Noturus												
Noturus gyrinus				X					X			

Genus/Species	ALEN	BCYP	COFY	GDMN	GUTH	HEST	HOPK	HWST	OBOT	OWMA	SARC	SWAN
Opsopoeodus												
Opsopoeodus emiliae											X	
Percina												
Percina caprodes			X									
Pimephales												
Pimephales vigilax											X	
Pomoxis		*	*									
Pomoxis annularis			X			X	X	X			X	X
Pomoxis nigromaculatus	X	X	X	X		X	X	X			X	X
Sander												
Sander canadensis			X									
Semotilus												
Semotilus atromaculatus						X					X	
Umbra									*			
Umbra limi	X	X			X	X	X	X	X	X	X	

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Table 3a, b, c, d, e, f, g, h, i, j, k, l: CPUE (individuals per minute) of each species collected with backpack electrofishing surveys from every sampling event at each wetland in western Kentucky, USA.

Table 3a:

ALEN

Species	2019-05-29	2019-06-25	2019-07-24	2019-08-27	2019-10-24	2020-02-01	2020-03-01	2020-04-01	2020-05-01	2020-06-02	2020-07-06	2020-08-05
Ameiurus melas	—	—	—	—	—	—	0.09	0.32	0.24	0.08	0.09	—
Ameiurus natalis	—	—	—	—	—	0.12	—	—	—	—	—	0.11
Ameiurus nebulosus	—	—	—	—	—	—	—	—	—	—	—	—
Amia calva	—	—	—	—	—	0.72	0.60	0.32	0.16	0.24	0.26	0.11
Aphredoderus sayanus	1.68	0.24	—	—	0.18	0.60	0.09	—	—	0.08	0.09	—
Aplodinotus grunniens	—	—	—	—	—	—	—	—	—	—	—	—
Centrarchus macropterus	—	—	—	—	0.06	0.96	1.11	0.24	0.24	—	0.34	0.11
Ctenopharyngodon idella	—	—	—	—	—	—	—	—	—	—	—	—
Cycleptus elongatus	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella lutrensis	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella venusta	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinus carpio	—	—	—	—	—	—	—	—	—	—	—	—
Dorosoma cepedianum	—	—	—	—	—	—	—	—	—	—	—	—
Elassoma zonatum	0.48	0.84	0.10	0.45	1.26	2.16	4.20	0.40	0.40	—	0.09	—
Erimyzon sucetta	4.20	—	0.10	—	—	—	0.09	0.08	—	0.08	—	0.55
Esox americanus	0.24	—	—	—	—	—	—	—	—	—	—	—
Etheostoma asprigene	—	—	—	—	—	—	—	—	—	—	—	—
Etheostoma chlorosoma	—	—	—	—	—	—	—	—	—	—	—	—
Etheostoma gracile	—	—	—	—	—	—	—	—	—	—	—	—
Fundulus chrysotus	—	—	—	—	—	—	—	—	—	—	—	—

ALEN

Species	2019-05-29	2019-06-25	2019-07-24	2019-08-27	2019-10-24	2020-02-01	2020-03-01	2020-04-01	2020-05-01	2020-06-02	2020-07-06	2020-08-05
Fundulus olivaceus	—	—	—	—	—	—	—	—	—	—	—	—
Gambusia affinis	—	2.04	2.00	1.35	0.48	1.20	3.60	—	1.04	0.72	4.54	0.11
Hybognathus hayi	—	—	—	—	—	—	—	—	—	—	—	—
Hypophthalmichthys molitrix	—	—	—	—	—	—	—	—	—	—	—	—
Ictalurus punctatus	—	—	—	—	—	—	—	—	—	—	—	—
Ictiobus bubalus	—	—	—	—	—	—	—	—	—	—	—	—
Labidesthes sicculus	—	—	—	—	—	—	0.17	0.08	—	—	—	—
Lepisosteus oculatus	0.12	—	—	—	—	0.24	0.77	—	0.08	0.32	—	—
Lepisosteus osseus	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis cyanellus	—	—	—	—	—	—	—	—	0.16	0.08	—	—
Lepomis gulosus	—	—	0.60	0.60	0.54	2.88	2.14	2.64	2.00	1.52	0.17	0.22
Lepomis humilis	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis macrochirus	—	—	1.00	0.30	0.06	1.80	2.91	3.28	3.44	1.92	0.77	0.76
Lepomis marginatus	—	—	—	—	0.06	—	—	0.08	0.88	0.24	0.09	—
Lepomis megalotis	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis microlophus	—	—	—	—	—	0.12	0.34	0.08	0.08	—	—	—
Lepomis miniatus	—	—	—	—	—	0.72	—	—	—	—	—	—
Lepomis symmetricus	0.12	0.60	0.10	—	0.12	8.88	0.60	0.56	1.04	0.64	0.09	0.33
Menidia beryllina	—	—	—	—	—	—	—	—	—	—	—	—
Micropterus salmoides	—	—	—	—	—	0.72	0.60	0.24	0.32	0.08	0.43	0.22
Minytrema melanops	—	—	—	—	—	0.12	—	0.32	—	—	—	—
Morone mississippiensis	—	—	—	—	—	—	—	—	—	—	—	—
Notemigonus crysoleucas	0.36	0.12	0.10	—	0.06	0.96	1.80	0.16	0.80	0.40	—	0.55
Notropis maculatus	—	—	—	—	—	—	—	—	—	—	—	—
Noturus gyrinus	—	—	—	—	—	—	—	—	—	—	—	—
Opsopoeodus emiliae	—	—	—	—	—	—	—	—	—	—	—	—

ALEN

Species	**2019-05-29**	**2019-06-25**	**2019-07-24**	**2019-08-27**	**2019-10-24**	**2020-02-01**	**2020-03-01**	**2020-04-01**	**2020-05-01**	**2020-06-02**	**2020-07-06**	**2020-08-05**
Percina caprodes	—	—	—	—	—	—	—	—	—	—	—	—
Pimephales vigilax	—	—	—	—	—	—	—	—	—	—	—	—
Pomoxis annularis	—	—	—	—	—	—	—	—	—	—	—	—
Pomoxis nigromaculatus	—	—	—	—	—	0.12	0.17	0.08	—	—	—	—
Sander canadensis	—	—	—	—	—	—	—	—	—	—	—	—
Semotilus atromaculatus	—	—	—	—	—	—	—	—	—	—	—	—
Umbra limi	—	—	—	—	—	—	0.09	—	—	—	0.09	—

Table 3b:

Species	BCYP															
	2019-04-29	**2019-05-31**	**2019-06-25**	**2019-07-24**	**2019-09-06**	**2019-10-03**	**2019-11-06**	**2019-12-13**	**2020-01-08**	**2020-02-06**	**2020-03-13**	**2020-03-31**	**2020-05-11**	**2020-06-10**	**2020-07-14**	**2020-08-18**
Ameiurus melas	—	—	—	—	—	—	—	0.10	0.09	—	—	—	—	—	—	—
Ameiurus natalis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ameiurus nebulosus	—	—	—	0.22	—	—	0.20	0.20	0.09	—	—	—	—	0.17	—	—
Amia calva	—	—	—	—	—	0.33	0.20	0.10	—	0.09	0.26	0.09	—	0.09	—	—
Aphredoderus sayanus	—	—	—	—	—	0.22	0.10	—	—	—	—	0.09	0.08	0.26	0.40	—
Aplodinotus grunniens	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Centrarchus macropterus	—	—	—	—	—	—	—	0.10	—	0.09	0.09	0.09	0.56	0.09	0.66	0.24
Ctenopharyng-don idella	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cycleptus elongatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella lutrensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella venusta	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinus carpio	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Dorosoma cepedianum	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Elassoma zonatum	—	—	0.12	—	—	4.58	—	—	—	—	—	—	—	—	—	—
Erimyzon sucetta	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Esox americanus	—	0.12	—	0.11	0.10	0.55	0.10	—	0.09	0.09	—	—	0.16	0.34	—	—
Etheostoma asprigene	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Etheostoma chlorosoma	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—

BCYP

Species	**2019-04-29**	**2019-05-31**	**2019-06-25**	**2019-07-24**	**2019-09-06**	**2019-10-03**	**2019-11-06**	**2019-12-13**	**2020-01-08**	**2020-02-06**	**2020-03-13**	**2020-03-31**	**2020-05-11**	**2020-06-10**	**2020-07-14**	**2020-08-18**
Fundulus chrysotus	—	—	—	0.11	—	0.11	—	—	—	—	—	—	—	—	—	—
Fundulus olivaceus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Gambusia affinis	0.60	0.24	0.72	0.22	—	45.37	—	—	—	—	—	—	0.64	—	2.77	12.95
Hybognathus hayi	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Hypophthalmicthys molitrix	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictalurus punctatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictiobus bubalus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Labidesthes sicculus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepisosteus oculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepisosteus osseus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis cyanellus	0.12	—	—	—	—	0.22	—	—	—	—	—	—	—	—	—	—
Lepomis gulosus	—	0.24	0.12	0.11	—	—	0.20	0.40	0.09	0.09	—	0.09	0.32	0.69	0.66	0.48
Lepomis humilis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis macrochirus	0.72	—	—	1.42	0.60	0.11	2.90	2.90	1.37	0.17	0.86	0.43	0.08	0.51	0.40	0.24
Lepomis marginatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis megalotis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis microlophus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis miniatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—

BCYP

Species	2019-04-29	2019-05-31	2019-06-25	2019-07-24	2019-09-06	2019-10-03	2019-11-06	2019-12-13	2020-01-08	2020-02-06	2020-03-13	2020-03-31	2020-05-11	2020-06-10	2020-07-14	2020-08-18
Menidia beryllina	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Micropterus salmoides	—	—	—	0.11	0.30	0.44	0.30	0.10	—	0.17	0.09	—	—	—	—	—
Minytrema melanops	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Morone mississippiensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Notemigonus crysoleucas	0.24	—	0.36	0.55	0.20	—	—	1.60	0.60	0.17	—	0.17	0.08	0.34	1.98	2.40
Notropis maculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Noturus gyrinus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Opsopoeodus emiliae	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Percina caprodes	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pimephales vigilax	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pomoxis annularis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pomoxis nigromaculatus	—	—	—	—	—	—	—	0.20	0.09	—	—	—	—	—	—	—
Sander canadensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Semotilus atromaculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Umbra limi	—	—	—	—	—	0.11	—	—	—	—	0.09	—	—	0.09	—	—

Table 3c:

			COFY				
Species	**2019-08-02**	**2019-09-03**	**2019-12-13**	**2020-05-18**	**2020-06-16**	**2020-07-14**	**2020-08-18**
Ameiurus melas	—	—	—	—	—	—	—
Ameiurus natalis	—	—	—	—	—	—	—
Ameiurus nebulosus	—	—	—	—	—	—	—
Amia calva	—	—	—	—	—	—	—
Aphredoderus sayanus	0.34	0.10	—	0.24	0.08	—	—
Aplodinotus grunniens	—	—	—	—	—	—	—
Centrarchus macropterus	—	—	—	—	—	—	—
Ctenopharyngodon idella	—	—	—	—	—	—	0.11
Cycleptus elongatus	—	—	—	—	—	—	—
Cyprinella lutrensis	—	—	—	—	—	—	—
Cyprinella venusta	—	—	—	—	—	—	—
Cyprinus carpio	—	—	—	—	—	—	0.22
Dorosoma cepedianum	0.60	—	—	0.24	—	—	0.33
Elassoma zonatum	—	—	—	—	—	—	—
Erimyzon sucetta	—	—	—	—	—	—	—
Esox americanus	—	—	—	—	—	—	—
Etheostoma asprigene	—	—	—	—	—	—	—
Etheostoma chlorosoma	0.43	0.10	—	—	—	—	—
Etheostoma gracile	—	—	—	—	—	—	—
Fundulus chrysotus	—	—	—	—	—	—	—
Fundulus olivaceus	0.09	—	0.20	—	—	0.08	1.20
Gambusia affinis	—	1.00	0.20	—	—	—	—
Hybognathus hayi	—	—	—	—	—	—	—
Hypophthalmichthys molitrix	—	—	—	—	—	—	—
Ictalurus punctatus	—	—	—	—	—	—	—
Ictiobus bubalus	0.17	0.20	0.70	—	—	—	—

COFY

Species	**2019-08-02**	**2019-09-03**	**2019-12-13**	**2020-05-18**	**2020-06-16**	**2020-07-14**	**2020-08-18**
Labidesthes sicculus	—	—	—	—	—	—	—
Lepisosteus oculatus	—	—	—	—	—	—	—
Lepomis cyanellus	0.09	—	—	—	—	—	—
Lepomis gulosus	0.17	—	—	0.08	0.08	—	—
Lepomis humilis	0.09	—	—	0.08	—	—	—
Lepomis macrochirus	0.09	0.30	0.30	0.40	0.08	0.32	0.33
Lepomis marginatus	—	—	0.20	—	—	—	—
Lepomis megalotis	0.09	0.50	—	0.08	0.08	—	0.22
Lepomis microlophus	—	—	—	—	—	—	—
Lepomis miniatus	—	—	—	—	—	—	—
Lepomis symmetricus	—	—	—	—	—	—	—
Menidia beryllina	—	—	—	—	—	—	—
Micropterus salmoides	—	—	—	0.08	0.32	0.64	—
Minytrema melanops	—	—	—	—	—	—	—
Morone mississippiensis	—	—	—	—	—	—	—
Notemigonus crysoleucas	—	—	—	—	0.08	0.88	0.11
Notropis maculatus	—	—	—	—	—	—	—
Noturus gyrinus	—	—	—	—	—	—	—
Opsopoeodus emiliae	—	—	—	—	—	—	—
Percina caprodes	0.09	—	—	—	—	—	—
Pimephales vigilax	—	—	—	—	—	—	—
Pomoxis annularis	0.17	0.10	—	—	—	—	—
Pomoxis nigromaculatus	0.26	1.00	0.60	0.08	0.56	0.16	0.11
Sander canadensis	—	0.10	—	—	—	—	—
Semotilus atromaculatus	—	—	—	—	—	—	—
Umbra limi	—	—	—	—	—	—	—

1865

Table 3d:

	GDMN						
Species	**2019-07-18**	**2019-09-03**	**2019-09-24**	**2020-05-14**	**2020-06-16**	**2020-07-14**	**2020-08-18**
Ameiurus melas	—	—	—	—	—	4.80	1.85
Ameiurus natalis	—	0.60	—	0.09	0.09	—	—
Ameiurus nebulosus	—	—	—	—	—	—	—
Amia calva	—	—	—	—	—	—	—
Aphredoderus sayanus	—	—	—	0.17	0.17	—	—
Aplodinotus grunniens	0.10	—	—	—	—	—	—
Centrarchus macropterus	—	0.40	—	—	0.34	—	—
Ctenopharyngodon idella	—	—	—	—	0.09	—	—
Cycleptus elongatus	—	—	—	0.09	—	—	—
Cyprinella lutrensis	—	—	—	—	—	—	—
Cyprinella venusta	—	—	—	—	—	—	—
Cyprinus carpio	—	—	0.15	—	1.20	1.37	0.87
Dorosoma cepedianum	0.20	—	—	—	—	—	—
Elassoma zonatum	—	—	—	—	—	—	—
Erimyzon sucetta	—	—	—	—	—	—	—
Esox americanus	—	—	—	—	—	—	—
Etheostoma asprigene	—	—	—	—	—	—	—
Etheostoma chlorosoma	—	—	—	—	—	—	—
Etheostoma gracile	—	—	—	—	—	—	—
Fundulus chrysotus	—	—	—	—	—	—	—
Fundulus olivaceus	—	—	—	—	—	—	—
Gambusia affinis	—	—	1.05	—	—	1.89	3.93
Hybognathus hayi	—	—	—	—	—	—	—
Hypophthalmichthys molitrix	—	—	—	—	—	—	—
Ictalurus punctatus	—	0.10	0.75	—	—	—	—
Ictiobus bubalus	—	—	10.20	—	1.11	—	—

GDMN

Species	**2019-07-18**	**2019-09-03**	**2019-09-24**	**2020-05-14**	**2020-06-16**	**2020-07-14**	**2020-08-18**
Labidesthes sicculus	—	—	—	—	—	—	—
Lepisosteus oculatus	—	—	—	—	—	—	—
Lepomis cyanellus	—	0.60	0.60	0.09	1.20	0.86	—
Lepomis gulosus	—	0.20	—	0.34	—	—	0.55
Lepomis humilis	—	—	—	0.09	—	0.09	—
Lepomis macrochirus	—	0.40	0.30	—	0.17	0.86	0.76
Lepomis marginatus	—	—	—	0.26	—	—	—
Lepomis megalotis	0.10	1.10	0.45	0.43	0.09	0.60	0.33
Lepomis microlophus	—	—	0.15	—	—	—	—
Lepomis miniatus	—	—	—	—	—	—	—
Lepomis symmetricus	—	—	0.15	—	—	—	—
Menidia beryllina	—	—	—	—	—	—	0.11
Micropterus salmoides	—	—	—	—	0.26	0.17	—
Minytrema melanops	—	—	—	—	—	—	—
Morone mississippiensis	0.10	—	—	—	—	—	—
Notemigonus crysoleucas	—	—	—	—	0.17	—	—
Notropis maculatus	—	—	—	—	—	—	—
Noturus gyrinus	—	0.10	0.15	—	—	—	—
Opsopoeodus emiliae	—	—	—	—	—	—	—
Percina caprodes	—	—	—	—	—	—	—
Pimephales vigilax	—	—	—	—	—	—	—
Pomoxis annularis	—	—	—	—	—	—	—
Pomoxis nigromaculatus	—	0.50	—	—	—	0.69	—
Sander canadensis	—	—	—	—	—	—	—
Semotilus atromaculatus	—	—	—	—	—	—	—
Umbra limi	—	—	—	—	—	—	—

1867

Table 3e:

	GUTH																
Species	**2019-04-09**	**2019-05-08**	**2019-06-03**	**2019-07-01**	**2019-08-01**	**2019-09-05**	**2019-10-08**	**2019-11-05**	**2019-12-12**	**2020-01-08**	**2020-02-07**	**2020-03-04**	**2020-04-01**	**2020-05-07**	**2020-06-11**	**2020-07-09**	**2020-08-04**
Ameiurus melas	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ameiurus natalis	—	—	—	—	—	0.10	—	—	—	—	—	—	—	—	—	—	—
Ameiurus nebulosus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Amia calva	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Aphredoderus sayanus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Aplodinotus grunniens	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Centrarchus macropterus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ctenopharyngo-don idella	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cycleptus elongatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella lutrensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella venusta	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinus carpio	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Dorosoma cepedianum	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Elassoma zonatum	—	—	0.12	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Erimyzon sucetta	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Esox americanus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Etheostoma asprigene	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Etheostoma gracile	—	0.84	0.24	—	—	0.30	0.24	—	—	—	—	0.24	0.12	—	—	—	—
Fundulus chrysotus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Fundulus olivaceus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Gambusia affinis	—	0.12	1.08	0.60	19.09	10.20	8.52	0.90	0.36	3.12	0.18	1.68	—	1.73	1.47	5.82	7.68
Hybognathus hayi	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Hypophtalmichthys molitrix	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—

GUTH

Species	2019-04-09	2019-05-08	2019-06-03	2019-07-01	2019-08-01	2019-09-05	2019-10-08	2019-11-05	2019-12-12	2020-01-08	2020-02-07	2020-03-04	2020-04-01	2020-05-07	2020-06-11	2020-07-09	2020-08-04
Ictalurus punctatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictiobus bubalus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Labidesthes sicculus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepisosteus oculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepisosteus osseus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis cyanellus	0.12	0.48	9.24	9.84	31.09	22.90	50.06	25.00	34.81	43.10	24.47	36.37	32.29	28.00	40.13	11.54	16.21
Lepomis gulosus	—	—	—	—	—	0.10	0.24	0.10	0.12	—	0.18	0.12	0.48	—	—	—	—
Lepomis humilis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis macrochirus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis marginatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis megalotis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis miniatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis symmetricus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Menidia beryllina	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Micropterus salmoides	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Minytrema melanops	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Morone mississippiensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Notemigonus crysoleucas	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Notropis maculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Noturus gyrinus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Opsopoeodus emiliae	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—

GUTH

Species	**2019-04-09**	**2019-05-08**	**2019-06-03**	**2019-07-01**	**2019-08-01**	**2019-09-05**	**2019-10-08**	**2019-11-05**	**2019-12-12**	**2020-01-08**	**2020-02-07**	**2020-03-04**	**2020-04-01**	**2020-05-07**	**2020-06-11**	**2020-07-09**	**2020-08-04**
Percina caprodes	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pimephales vigilax	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pomoxis annularis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pomoxis nigromaculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Sander canadensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Semotilus atromaculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Umbra limi	—	—	—	—	0.12	—	—	—	—	—	—	—	—	—	—	—	—

1869

Table 3f:

	HEST												
Species	**2019-05-31**	**2019-07-18**	**2019-08-28**	**2019-09-24**	**2019-12-11**	**2020-01-06**	**2020-02-03**	**2020-03-06**	**2020-04-15**	**2020-05-11**	**2020-06-10**	**2020-07-15**	**2020-08-18**
Ameiurus melas	—	0.30	0.80	52.55	—	0.10	0.32	0.09	—	0.28	0.27	0.32	—
Ameiurus natalis	—	—	1.50	28.83	—	—	—	—	—	—	—	0.24	0.87
Ameiurus nebulosus	—	—	0.30	4.20	—	—	—	—	—	—	—	—	—
Amia calva	—	—	0.10	—	—	0.10	—	0.09	0.16	—	—	—	—
Aphredoderus sayanus	0.48	1.10	0.10	0.60	0.36	0.20	0.56	0.26	—	—	1.47	—	0.22
Aplodinotus grunniens	—	—	—	—	—	—	—	—	—	—	—	—	—
Centrarchus macropterus	—	0.20	0.30	0.30	0.24	0.10	0.24	—	0.08	—	1.60	0.32	0.44
Ctenopharyngodon idella	—	—	—	—	—	—	—	—	—	—	—	—	—
Cycleptus elongatus	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella lutrensis	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella venusta	—	—	—	—	—	—	—	0.17	—	—	—	—	—
Cyprinus carpio	0.12	0.20	0.80	—	—	—	—	—	—	—	0.67	—	0.76
Dorosoma cepedianum	—	0.50	0.10	—	—	—	—	0.09	—	—	—	—	0.22
Elassoma zonatum	0.96	0.20	—	—	0.12	0.10	0.48	0.17	0.16	0.09	0.27	—	—
Erimyzon sucetta	—	—	—	—	—	—	—	0.09	—	—	2.40	—	0.33
Esox americanus	0.72	—	0.20	—	0.12	0.20	0.08	0.09	0.64	0.55	1.60	—	0.11
Etheostoma asprigene	—	—	—	—	—	—	—	—	—	—	0.13	—	—
Etheostoma chlorosoma	—	—	—	—	—	—	—	0.09	—	—	—	—	—
Etheostoma gracile	0.36	0.10	—	—	—	—	0.64	0.09	—	0.18	0.53	—	0.11

HEST

Species	2019-05-31	2019-07-18	2019-08-28	2019-09-24	2019-12-11	2020-01-06	2020-02-03	2020-03-06	2020-04-15	2020-05-11	2020-06-10	2020-07-15	2020-08-18
Fundulus chrysotus	—	—	—	—	—	—	—	—	—	—	—	—	—
Fundulus olivaceus	—	—	0.10	—	0.48	0.10	0.08	0.17	—	0.09	—	—	0.44
Gambusia affinis	0.24	0.40	0.10	69.37	0.48	0.10	0.16	0.17	0.40	0.65	15.33	0.16	1.64
Hypophthalmichthys molitrix	—	—	—	—	0.12	—	—	—	—	—	—	—	—
Ictalurus punctatus	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictiobus bubalus	4.32	0.10	0.20	0.60	—	—	—	0.77	—	—	—	—	—
Labidesthes sicculus	—	—	—	—	—	—	—	—	—	—	—	—	0.33
Lepisosteus oculatus	—	0.10	—	—	—	—	—	—	—	—	—	—	—
Lepisosteus osseus	—	—	—	0.30	—	—	—	—	—	—	—	—	—
Lepomis cyanellus	—	—	—	—	—	—	—	0.17	0.08	0.09	—	0.08	—
Lepomis gulosus	0.72	0.10	0.10	0.90	0.60	0.30	0.08	0.34	0.40	0.46	0.40	—	0.11
Lepomis humilis	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis macrochirus	0.36	—	—	—	0.48	0.90	0.16	0.34	0.24	0.28	—	0.24	0.44
Lepomis marginatus	—	—	—	—	—	—	—	0.09	—	—	—	—	—
Lepomis megalotis	—	—	—	—	—	—	—	—	—	—	—	0.08	0.11
Lepomis microlophus	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis miniatus	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis symmetricus	0.24	0.20	0.10	0.60	—	—	0.16	0.09	0.40	0.18	—	—	0.76

HEST

Species	2019-05-31	2019-07-18	2019-08-28	2019-09-24	2019-12-11	2020-01-06	2020-02-03	2020-03-06	2020-04-15	2020-05-11	2020-06-10	2020-07-15	2020-08-18
Menidia beryllina	—	—	—	—	—	—	—	—	—	—	—	—	—
Micropterus salmoides	0.24	—	0.30	—	0.12	0.30	0.16	0.43	—	—	—	—	0.11
Minytrema melanops	—	—	—	—	0.12	—	—	—	—	—	—	—	—
Morone mississippiensis	—	—	—	—	—	—	—	—	—	—	—	—	—
Notemigonus crysoleucas	3.24	1.90	—	—	0.12	0.10	—	0.09	0.08	0.18	4.13	0.16	0.33
Notropis maculatus	—	—	—	—	—	—	—	—	—	—	—	—	—
Noturus gyrinus	—	—	—	—	—	—	—	—	—	—	—	—	—
Opsopoeodus emiliae	—	—	—	—	—	—	—	—	—	—	—	—	—
Percina caprodes	—	—	—	—	—	—	—	—	—	—	—	—	—
Pimephales vigilax	—	—	—	—	—	—	—	—	—	—	—	—	—
Pomoxis annularis	—	0.20	—	—	—	—	—	—	—	—	—	—	—
Pomoxis nigromaculatus	—	0.20	0.40	0.60	—	0.10	—	0.09	—	—	—	—	—
Sander canadensis	—	—	—	—	—	—	—	—	—	—	—	—	—
Semotilus atromaculatus	—	—	—	—	—	—	—	—	—	—	—	0.24	—
Umbra limi	—	—	—	—	—	—	—	—	—	—	—	—	0.22

1871

Table 3g:

	HOPK														
Species	**2019-05-29**	**2019-06-25**	**2019-07-24**	**2019-08-27**	**2019-09-19**	**2019-10-24**	**2019-12-11**	**2020-01-06**	**2020-02-01**	**2020-03-01**	**2020-04-01**	**2020-05-01**	**2020-06-02**	**2020-07-06**	**2020-08-05**
Ameiurus melas	0.24	—	—	—	—	—	0.20	0.28	0.48	0.17	—	—	0.18	0.17	—
Ameiurus natalis	—	—	—	—	0.12	0.13	—	—	—	—	—	—	—	—	—
Ameiurus nebulosus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Amia calva	—	—	—	—	0.12	—	—	—	—	—	0.08	—	—	—	—
Aphredoderus sayanus	—	—	0.12	—	0.12	—	0.10	0.18	0.08	—	—	—	—	—	—
Aplodinotus grunniens	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Centrarchus macropterus	—	—	—	—	0.24	0.13	0.10	0.18	0.08	0.34	0.08	—	—	—	—
Ctenopharyngodon idella	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cycleptus elongatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella lutrensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella venusta	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinus carpio	—	—	0.12	0.10	—	—	—	—	—	—	—	—	—	—	—
Dorosoma cepedianum	—	—	0.06	—	—	—	—	—	—	—	—	—	—	—	—
Elassoma zonatum	—	—	—	0.10	—	0.13	—	0.09	0.40	1.54	1.28	—	—	—	—
Erimyzon sucetta	—	0.12	—	0.10	0.12	—	0.20	0.18	0.96	0.69	—	—	—	—	—
Esox americanus	—	—	—	—	0.36	0.13	0.20	0.28	0.48	0.26	0.16	0.09	—	—	—
Etheostoma asprigene	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Etheostoma chlorosoma	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Etheostoma gracile	—	—	—	—	—	—	—	—	0.48	0.69	—	—	—	—	—

HOPK

Species	2019-05-29	2019-06-25	2019-07-24	2019-08-27	2019-09-19	2019-10-24	2019-12-11	2020-01-06	2020-02-01	2020-03-01	2020-04-01	2020-05-01	2020-06-02	2020-07-06	2020-08-05
Fundulus chrysotus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Fundulus olivaceus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Gambusia affinis	0.12	0.24	0.24	—	—	0.13	—	0.09	0.08	—	0.16	1.94	0.09	—	—
Hybognathus hayi	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Hypophtalmichthys molitrix	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictalurus punctatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictiobus bubalus	—	—	—	—	0.24	—	—	—	—	—	—	—	—	—	—
Labidesthes sicculus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepisosteus oculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepisosteus osseus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis cyanellus	—	0.12	—	—	0.72	—	—	—	—	—	—	0.18	0.09	—	0.10
Lepomis gulosus	0.36	0.72	0.18	0.40	3.48	1.60	0.20	0.28	0.24	0.86	0.88	0.92	0.37	—	0.10
Lepomis humilis	—	—	—	—	—	—	—	0.09	—	—	—	—	—	—	—
Lepomis macrochirus	1.20	1.08	0.18	0.20	1.44	0.67	0.60	0.92	0.72	0.77	0.56	0.37	0.09	0.69	1.15
Lepomis marginatus	0.24	0.72	0.18	0.10	1.20	0.80	—	0.09	0.16	0.43	0.32	0.28	—	—	0.10
Lepomis megalotis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis microlophus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis miniatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis symmetricus	0.24	1.32	0.12	0.10	1.68	1.20	—	—	0.32	0.34	0.96	0.28	0.09	0.17	0.10

HOPK

Species	**2019-05-29**	**2019-06-25**	**2019-07-24**	**2019-08-27**	**2019-09-19**	**2019-10-24**	**2019-12-11**	**2020-01-06**	**2020-02-01**	**2020-03-01**	**2020-04-01**	**2020-05-01**	**2020-06-02**	**2020-07-06**	**2020-08-05**
Menidia beryllina	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Micropterus salmoides	—	0.12	—	0.10	—	0.13	0.10	—	—	—	—	—	—	0.34	—
Minytrema melanops	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Morone mississippiensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Notemigonus crysoleucas	0.24	0.12	0.36	1.60	1.08	0.13	0.10	—	0.16	0.17	—	0.28	0.09	0.51	0.31
Notropis maculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Noturus gyrinus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Opsopoeodus emiliae	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Percina caprodes	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pimephales vigilax	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pomoxis annularis	—	—	0.06	0.20	0.12	—	—	—	—	—	—	—	—	—	—
Pomoxis nigromaculatus	—	—	0.06	—	0.48	—	—	—	—	—	—	—	—	—	—
Sander canadensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Semotilus atromaculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Umbra limi	—	0.12	—	—	0.12	0.27	—	—	0.16	—	—	—	—	—	—

Table 3h:

	HWST												
Species	**2019-05-31**	**2019-07-18**	**2019-08-28**	**2019-09-24**	**2019-10-14**	**2019-12-11**	**2020-01-06**	**2020-02-03**	**2020-03-06**	**2020-04-15**	**2020-05-11**	**2020-06-10**	**2020-07-15**
Ameiurus melas	—	—	—	—	—	0.26	0.26	0.17	0.40	0.16	0.08	—	—
Ameiurus natalis	—	0.24	—	—	—	—	0.17	—	—	—	—	—	—
Ameiurus nebulosus	—	—	—	0.08	—	0.09	—	—	—	—	—	—	—
Amia calva	—	—	—	—	0.09	—	0.09	—	—	0.16	0.16	—	—
Aphredoderus sayanus	0.48	0.24	—	0.15	—	0.26	0.26	0.51	0.32	—	—	—	—
Aplodinotus grunniens	—	—	—	—	—	—	—	—	—	—	—	—	—
Centrarchus macropterus	—	—	1.29	0.69	0.77	0.26	0.60	0.43	0.08	—	1.28	0.90	—
Ctenopharyngodon idella	—	—	—	—	—	—	—	—	—	—	—	—	—
Cycleptus elongatus	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella lutrensis	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella venusta	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinus carpio	—	—	—	—	—	—	—	—	—	—	—	—	—
Dorosoma cepedianum	—	—	—	0.08	—	—	—	—	—	—	—	0.60	—
Elassoma zonatum	0.12	—	—	—	0.09	0.69	1.03	0.51	1.12	0.40	—	—	0.16
Erimyzon sucetta	0.24	—	—	0.08	—	0.09	0.09	—	0.24	0.16	0.24	0.15	0.16
Esox americanus	—	—	0.18	—	—	—	—	0.17	0.32	1.04	—	—	—
Etheostoma asprigene	—	—	—	—	—	—	—	—	—	—	—	—	—
Etheostoma chlorosoma	—	—	—	—	—	—	—	—	—	—	—	—	—
Etheostoma gracile	—	—	—	—	—	—	0.09	0.09	0.08	—	—	—	—

HWST

Species	2019-05-31	2019-07-18	2019-08-28	2019-09-24	2019-10-14	2019-12-11	2020-01-06	2020-02-03	2020-03-06	2020-04-15	2020-05-11	2020-06-10	2020-07-15
Fundulus chrysotus	—	—	—	—	—	—	—	—	—	—	—	—	—
Fundulus olivaceus	—	—	0.09	0.08	—	—	0.09	—	0.08	0.16	—	—	—
Gambusia affinis	0.84	0.84	—	0.08	—	0.09	0.09	—	0.32	0.48	—	—	0.08
Hybognathus hayi	—	—	—	—	—	—	—	—	—	—	—	—	—
Hypophthalmichthys molitrix	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictalurus punctatus	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictiobus bubalus	0.48	0.24	0.37	0.31	0.17	0.09	—	—	—	0.08	0.08	0.75	—
Labidesthes sicculus	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepisosteus oculatus	—	—	—	—	0.09	—	—	—	—	—	—	—	0.08
Lepisosteus osseus	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis cyanellus	—	—	0.09	0.15	—	0.09	0.17	0.17	0.24	—	—	—	—
Lepomis gulosus	0.24	0.12	1.20	0.23	0.60	1.29	2.06	1.54	0.96	0.32	1.04	—	0.16
Lepomis humilis	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis macrochirus	0.12	0.36	—	0.23	0.26	0.43	0.26	0.17	0.16	0.72	0.56	2.10	0.08
Lepomis marginatus	—	—	0.46	—	—	0.09	0.09	—	—	—	0.08	0.60	—
Lepomis megalotis	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis microlophus	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis miniatus	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis symmetricus	—	0.24	0.83	0.15	0.43	0.69	0.94	0.60	0.80	0.96	0.40	0.60	—
Menidia beryllina	—	—	—	—	—	—	—	—	—	—	—	—	—

HWST

Species	2019-05-31	2019-07-18	2019-08-28	2019-09-24	2019-10-14	2019-12-11	2020-01-06	2020-02-03	2020-03-06	2020-04-15	2020-05-11	2020-06-10	2020-07-15
Micropterus salmoides	0.12	0.12	0.46	0.38	0.43	0.51	0.34	0.17	0.08	0.08	0.08	0.75	0.16
Minytrema melanops	—	—	—	—	—	—	—	—	—	—	—	—	—
Morone mississippiensis	—	—	—	—	—	—	—	—	—	—	—	—	—
Notemigonus crysoleucas	3.60	3.00	—	—	—	—	—	0.26	—	—	—	—	0.48
Notropis maculatus	—	—	—	—	—	—	—	—	—	—	—	—	—
Noturus gyrinus	—	—	—	—	—	—	—	—	—	—	—	—	—
Opsopoeodus emiliae	—	—	—	—	—	—	—	—	—	—	—	—	—
Percina caprodes	—	—	—	—	—	—	—	—	—	—	—	—	—
Pimephales vigilax	—	—	—	—	—	—	—	—	—	—	—	—	—
Pomoxis annularis	—	0.24	—	—	—	—	—	—	—	—	—	—	—
Pomoxis nigromaculatus	—	—	—	0.46	0.26	0.43	0.09	0.26	—	—	0.16	0.15	—
Sander canadensis	—	—	—	—	—	—	—	—	—	—	—	—	—
Semotilus atromaculatus	—	—	—	—	—	—	—	—	—	—	—	—	—
Umbra limi	—	—	0.28	—	0.60	0.60	0.09	0.17	—	0.16	0.08	—	—

Table 3i:

	OBOT															
Species	**2019-04-09**	**2019-05-08**	**2019-06-03**	**2019-07-01**	**2019-08-01**	**2019-10-08**	**2019-11-05**	**2019-12-12**	**2020-01-08**	**2020-02-07**	**2020-03-04**	**2020-04-01**	**2020-05-07**	**2020-06-11**	**2020-07-09**	**2020-08-11**
Ameiurus melas	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ameiurus natalis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ameiurus nebulosus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Amia calva	—	0.12	—	0.09	0.12	—	—	0.08	—	—	—	—	0.08	0.08	0.16	0.13
Aphredoderus sayanus	—	0.12	0.24	0.28	—	0.09	—	0.15	0.43	0.06	0.34	—	1.28	0.40	1.28	1.33
Aplodinotus grunniens	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Centrarchus macropterus	—	0.84	0.24	—	0.12	—	0.09	—	0.17	—	—	—	1.04	2.16	2.08	2.80
Ctenopharyngodon idella	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cycleptus elongatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella lutrensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella venusta	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinus carpio	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Dorosoma cepedianum	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Elassoma zonatum	0.20	0.24	—	—	0.12	0.17	0.18	0.15	0.34	0.30	0.34	0.44	0.08	—	0.08	—
Erimyzon sucetta	—	—	0.60	0.28	—	0.26	0.09	—	0.09	0.12	0.17	0.22	0.48	0.48	0.40	0.53
Esox americanus	—	—	0.12	—	—	0.09	—	0.08	—	—	—	0.22	0.24	0.24	0.48	—
Etheostoma asprigene	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Etheostoma chlorosoma	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Etheostoma gracile	—	—	—	—	—	—	—	—	—	—	—	—	—	—	0.08	—
Fundulus chrysotus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—

OBOT

Species	**2019-04-09**	**2019-05-08**	**2019-06-03**	**2019-07-01**	**2019-08-01**	**2019-10-08**	**2019-11-05**	**2019-12-12**	**2020-01-08**	**2020-02-07**	**2020-03-04**	**2020-04-01**	**2020-05-07**	**2020-06-11**	**2020-07-09**	**2020-08-11**
Fundulus olivaceus	—	0.48	—	—	—	0.09	0.09	0.23	0.17	—	—	—	—	0.08	0.08	—
Gambusia affinis	0.30	2.04	0.24	1.20	1.56	1.71	1.66	0.60	0.69	0.24	0.69	1.09	0.24	0.24	0.96	—
Hybognathus hayi	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Hypophthalmichthys molitrix	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictalurus punctatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictiobus bubalus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Labidesthes sicculus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepisosteus oculatus	—	—	—	—	—	—	—	—	—	—	—	—	0.08	—	—	—
Lepisosteus osseus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis cyanellus	—	—	0.12	—	—	—	—	—	—	0.06	—	—	0.08	—	—	—
Lepomis gulosus	0.10	0.36	—	—	0.60	—	0.18	0.08	—	0.06	0.17	0.22	1.52	0.64	0.40	—
Lepomis humilis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis macrochirus	—	0.36	0.12	—	—	0.09	0.18	0.15	0.17	0.06	0.26	0.22	0.08	—	0.48	0.53
Lepomis marginatus	—	0.72	1.80	0.28	—	0.43	0.28	0.08	0.09	0.06	—	0.87	2.96	1.68	0.24	—
Lepomis megalotis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis microlophus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis miniatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis symmetricus	0.30	1.80	1.20	0.37	0.12	1.63	0.92	0.98	0.43	0.54	1.29	0.98	5.04	0.72	1.68	0.40
Menidia beryllina	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Micropterus salmoides	—	—	0.12	—	—	—	—	0.08	—	0.06	—	—	—	0.16	0.08	—

OBOT

Species	2019-04-09	2019-05-08	2019-06-03	2019-07-01	2019-08-01	2019-10-08	2019-11-05	2019-12-12	2020-01-08	2020-02-07	2020-03-04	2020-04-01	2020-05-07	2020-06-11	2020-07-09	2020-08-11
Minytrema melanops	—	—	—	—	—	—	—	—	0.09	—	—	—	—	—	—	—
Morone mississippiensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Notemigonus crysoleucas	—	—	0.24	—	—	—	—	0.08	—	—	—	—	—	—	—	—
Notropis maculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Noturus gyrinus	—	—	—	—	—	—	—	—	—	—	—	0.11	—	—	—	—
Opsopoeodus emiliae	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Percina caprodes	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pimephales vigilax	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pomoxis annularis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pomoxis nigromaculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Sander canadensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Semotilus atromaculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Umbra limi	—	—	—	—	—	0.09	—	—	0.09	—	—	0.33	0.08	—	0.24	—

Table 3j:

Species	OWMA 2019-04-09	2019-05-08	2019-06-03	2019-07-01	2019-08-01	2019-09-05	2019-10-08	2019-11-05	2020-01-08	2020-02-07	2020-03-04	2020-04-01	2020-05-07	2020-06-11	2020-07-09	2020-08-11
Ameiurus melas	0.36	—	—	2.28	—	—	—	1.35	—	—	—	—	1.80	2.20	0.60	—
Ameiurus natalis	—	—	—	—	3.67	3.91	10.88	0.75	—	—	—	—	0.40	—	—	—
Ameiurus nebulosus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Amia calva	—	0.36	—	—	—	0.14	—	—	—	—	—	—	—	—	0.15	—
Aphredoderus sayanus	—	0.12	0.20	—	—	0.56	0.16	0.15	—	—	—	—	0.50	4.30	1.65	1.20
Aplodinotus grunniens	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Centrarchus macropterus	—	0.48	0.40	0.36	0.14	—	—	0.75	0.12	—	—	—	—	0.20	0.30	0.53
Ctenopharyngodon idella	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cycleptus elongatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella lutrensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella venusta	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinus carpio	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Dorosoma cepedianum	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Elassoma zonatum	—	—	—	—	—	—	—	—	—	—	0.09	—	0.10	—	—	—
Erimyzon sucetta	0.48	—	—	—	0.14	0.28	0.32	0.15	—	0.08	—	0.88	—	—	—	—
Esox americanus	0.48	1.68	0.10	—	—	0.42	0.16	0.15	—	0.08	0.09	0.32	0.50	0.60	0.30	0.13
Etheostoma asprigene	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Etheostoma chlorosoma	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—

OWMA

Species	**2019-04-09**	**2019-05-08**	**2019-06-03**	**2019-07-01**	**2019-08-01**	**2019-09-05**	**2019-10-08**	**2019-11-05**	**2020-01-08**	**2020-02-07**	**2020-03-04**	**2020-04-01**	**2020-05-07**	**2020-06-11**	**2020-07-09**	**2020-08-11**
Etheostoma gracile	0.60	—	—	0.12	—	—	—	—	—	—	0.09	—	—	—	—	—
Fundulus chrysotus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Fundulus olivaceus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Gambusia affinis	—	0.36	0.60	1.44	3.25	1.26	1.28	—	0.12	—	—	—	1.50	—	—	—
Hybognathus hayi	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Hypophthalmic-hthys molitrix	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictalurus punctatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictiobus bubalus	—	—	—	—	—	0.14	—	—	—	—	—	—	—	—	—	—
Labidesthes sicculus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepisosteus oculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepisosteus osseus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis cyanellus	0.36	0.48	0.80	0.12	0.56	0.14	0.64	0.45	—	—	—	0.40	0.10	—	—	—
Lepomis gulosus	0.24	—	0.10	0.12	—	0.42	0.48	0.30	—	—	—	—	—	—	—	—
Lepomis humilis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis macrochirus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis marginatus	—	—	0.10	—	—	0.14	0.16	—	—	—	—	—	—	—	—	—
Lepomis megalotis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis microlophus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—

OWMA

Species	**2019-04-09**	**2019-05-08**	**2019-06-03**	**2019-07-01**	**2019-08-01**	**2019-09-05**	**2019-10-08**	**2019-11-05**	**2020-01-08**	**2020-02-07**	**2020-03-04**	**2020-04-01**	**2020-05-07**	**2020-06-11**	**2020-07-09**	**2020-08-11**
Lepomis miniatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis symmetricus	—	—	0.60	—	—	—	—	—	—	—	—	—	0.60	0.90	1.65	0.53
Menidia beryllina	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Micropterus salmoides	—	—	—	—	—	0.14	—	—	—	—	—	—	—	—	—	—
Minytrema melanops	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Morone mississippiensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Notemigonus crysoleucas	0.12	—	0.10	3.24	14.83	14.78	16.48	5.10	—	—	—	0.24	0.60	1.50	2.55	2.13
Notropis maculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Noturus gyrinus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Opsopoeodus emiliae	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Percina caprodes	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pimephales vigilax	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pomoxis annularis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pomoxis nigromaculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Sander canadensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Semotilus atromaculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Umbra limi	—	0.48	—	—	—	—	—	—	—	0.08	0.17	0.32	—	—	—	—

1879

Table 3k:

	SARC															
Species	**2019-04-01**	**2019-05-07**	**2019-06-04**	**2019-06-28**	**2019-07-26**	**2019-09-04**	**2019-10-01**	**2019-12-05**	**2020-01-07**	**2020-02-03**	**2020-03-05**	**2020-04-02**	**2020-05-05**	**2020-06-03**	**2020-07-08**	**2020-08-04**
Ameiurus melas	—	—	—	—	—	—	—	—	—	—	—	—	—	—	0.08	—
Ameiurus natalis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ameiurus nebulosus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Amia calva	—	—	—	—	—	—	—	—	0.17	0.08	—	—	0.08	0.08	—	—
Aphredoderus sayanus	—	—	—	—	—	0.60	0.17	—	0.26	0.08	0.09	0.08	0.32	0.08	0.16	—
Aplodinotus grunniens	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Centrarchus macropterus	—	0.10	0.50	0.12	0.10	—	0.09	—	—	0.16	—	—	—	—	—	—
Ctenopharyngodon idella	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cycleptus elongatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella lutrensis	—	—	—	—	—	—	—	—	—	—	0.17	—	—	—	—	—
Cyprinella venusta	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinus carpio	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Dorosoma cepedianum	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Elassoma zonatum	—	—	—	—	—	0.20	0.09	1.33	1.97	1.36	1.63	0.08	0.16	—	0.08	—
Erimyzon sucetta	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Esox americanus	—	1.00	0.20	—	—	—	—	—	—	—	0.17	0.08	0.16	—	—	0.08
Etheostoma asprigene	—	—	—	—	—	0.10	0.17	—	—	0.08	—	—	—	—	—	—
Etheostoma chlorosoma	—	0.10	—	0.96	0.10	0.50	0.34	—	—	0.16	—	0.16	—	—	0.08	0.08

SARC

Species	2019-04-01	2019-05-07	2019-06-04	2019-06-28	2019-07-26	2019-09-04	2019-10-01	2019-12-05	2020-01-07	2020-02-03	2020-03-05	2020-04-02	2020-05-05	2020-06-03	2020-07-08	2020-08-04
Etheostoma gracile	—	—	0.10	0.84	—	0.40	0.77	—	0.34	0.24	—	—	—	—	—	0.08
Fundulus chrysotus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Fundulus olivaceus	—	0.10	—	0.60	0.80	0.40	0.09	0.53	0.51	0.16	0.17	0.08	—	0.16	0.24	0.72
Gambusia affinis	0.30	0.30	—	0.48	0.70	0.20	0.69	1.47	0.34	0.48	0.43	—	0.24	0.32	0.24	0.16
Hybognathus hayi	—	—	—	0.12	—	—	—	—	—	—	—	—	0.16	—	—	—
Hypophthalmic-hthys molitrix	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictalurus punctatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictiobus bubalus	—	—	—	—	—	0.10	—	—	—	—	—	—	—	—	—	—
Labidesthes sicculus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	0.08
Lepisosteus oculatus	—	—	0.10	—	—	—	—	—	—	—	—	—	0.08	—	—	—
Lepisosteus osseus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	0.08	—
Lepomis cyanellus	—	—	—	—	—	0.20	0.09	—	—	—	—	—	—	—	—	—
Lepomis gulosus	0.40	1.00	0.20	0.24	0.10	0.40	0.69	0.80	0.17	0.48	0.77	0.80	0.48	0.32	0.08	0.40
Lepomis humilis	—	—	—	0.24	0.10	—	—	—	—	—	—	0.08	0.08	0.08	0.08	0.08
Lepomis macrochirus	0.70	2.10	0.70	0.12	0.60	1.10	0.60	0.27	0.09	0.16	1.29	1.36	1.04	0.64	0.80	2.08
Lepomis marginatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis megalotis	—	—	—	—	—	0.10	—	—	—	—	—	—	—	—	0.24	0.64
Lepomis microlophus	—	0.10	0.10	—	—	—	—	—	—	—	0.09	—	0.08	—	—	—

SARC

Species	**2019-04-01**	**2019-05-07**	**2019-06-04**	**2019-06-28**	**2019-07-26**	**2019-09-04**	**2019-10-01**	**2019-12-05**	**2020-01-07**	**2020-02-03**	**2020-03-05**	**2020-04-02**	**2020-05-05**	**2020-06-03**	**2020-07-08**	**2020-08-04**
Lepomis miniatus	—	0.10	—	—	0.10	—	—	—	—	—	0.09	0.08	0.08	0.16	—	—
Lepomis symmetricus	0.30	0.20	0.10	—	0.40	—	—	0.53	0.09	0.32	0.34	0.40	0.24	—	0.08	0.08
Menidia beryllina	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Micropterus salmoides	—	0.10	—	0.12	0.10	0.20	—	—	—	—	0.09	0.24	0.08	—	0.32	0.24
Minytrema melanops	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	0.08
Morone mississippiensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Notemigonus crysoleucas	—	—	—	—	0.10	—	—	—	—	—	—	—	—	—	—	0.08
Notropis maculatus	—	—	—	0.24	0.10	0.20	0.09	—	—	—	0.09	0.24	0.16	0.16	0.16	0.40
Noturus gyrinus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Opsopoeodus emiliae	—	—	0.60	—	—	—	—	—	—	—	—	—	0.08	0.08	0.16	—
Percina caprodes	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pimephales vigilax	—	—	—	—	—	—	—	—	—	—	—	—	0.40	0.08	—	—
Pomoxis annularis	—	—	—	—	0.20	—	—	—	—	—	—	—	—	—	—	—
Pomoxis nigromaculatus	—	—	—	—	—	0.10	0.17	0.13	—	—	—	0.08	0.16	—	—	—
Sander canadensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Semotilus atromaculatus	0.10	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Umbra limi	—	—	—	—	—	—	—	—	—	—	—	—	—	—	0.08	—

Table 3l:

Species	2019-04-01	2019-05-07	2019-06-04	2019-06-28	2019-07-26	2019-09-04	2019-10-01	2019-12-05	2020-01-07	2020-02-03	2020-03-05	2020-04-02	2020-05-05	2020-06-03	2020-07-08	2020-08-04
	SWAN															
Ameiurus melas	—	—	—	—	—	—	—	0.43	—	0.26	0.30	—	0.08	—	—	—
Ameiurus natalis	—	—	—	—	0.20	0.20	0.18	0.26	0.26	—	—	—	—	—	0.16	—
Ameiurus nebulosus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Amia calva	—	0.13	0.10	—	—	0.10	0.18	—	—	—	—	—	—	0.15	—	—
Aphredoderus sayanus	—	—	—	0.24	—	0.10	0.90	0.17	—	—	—	—	—	—	0.16	—
Aplodinotus grunniens	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Centrarchus macropterus	—	—	0.60	1.44	0.40	0.10	3.23	—	0.43	0.17	0.10	0.40	0.16	0.30	0.32	—
Ctenopharyngodon idella	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cycleptus elongatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella lutrensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinella venusta	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Cyprinus carpio	—	—	—	—	0.10	—	—	—	—	—	—	—	—	—	—	—
Dorosoma cepedianum	—	—	—	—	—	0.20	—	—	—	—	—	—	—	—	—	—
Elassoma zonatum	0.10	—	0.20	0.24	—	—	2.51	0.26	0.34	0.17	1.40	0.16	—	—	0.16	0.13
Erimyzon sucetta	0.10	—	—	0.12	—	—	0.54	—	—	—	—	—	—	—	0.16	0.27
Esox americanus	—	0.13	0.20	0.48	0.10	0.10	1.08	—	0.09	—	0.10	0.16	0.32	0.75	—	—
Etheostoma asprigene	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Etheostoma chlorosoma	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—

SWAN

Species	**2019-04-01**	**2019-05-07**	**2019-06-04**	**2019-06-28**	**2019-07-26**	**2019-09-04**	**2019-10-01**	**2019-12-05**	**2020-01-07**	**2020-02-03**	**2020-03-05**	**2020-04-02**	**2020-05-05**	**2020-06-03**	**2020-07-08**	**2020-08-04**
Etheostoma gracile	—	—	—	—	—	—	—	—	0.17	—	0.50	0.08	—	—	—	—
Fundulus chrysotus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Fundulus olivaceus	—	—	—	—	—	—	—	0.17	0.17	0.09	—	0.16	—	—	—	—
Gambusia affinis	—	—	0.20	0.12	0.10	—	2.87	0.17	—	0.17	0.30	—	0.40	3.45	2.24	0.27
Hybognathus hayi	—	—	—	—	—	—	0.36	—	—	—	0.10	0.08	—	—	—	—
Hypophthalmic-hthys molitrix	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictalurus punctatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Ictiobus bubalus	—	—	—	0.12	0.20	0.10	—	—	—	—	—	—	—	—	—	—
Labidesthes sicculus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepisosteus oculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepisosteus osseus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis cyanellus	0.10	0.13	—	0.24	—	—	—	0.09	—	0.09	0.10	0.16	0.32	—	—	—
Lepomis gulosus	0.30	0.13	—	0.24	0.10	0.30	1.79	0.69	0.17	0.94	0.60	0.16	0.16	0.60	—	0.13
Lepomis humilis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis macrochirus	0.80	—	—	—	—	—	0.36	3.43	2.49	2.83	1.00	0.96	—	0.15	—	0.93
Lepomis marginatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis megalotis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Lepomis microlophus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—

SWAN

Species	2019-04-01	2019-05-07	2019-06-04	2019-06-28	2019-07-26	2019-09-04	2019-10-01	2019-12-05	2020-01-07	2020-02-03	2020-03-05	2020-04-02	2020-05-05	2020-06-03	2020-07-08	2020-08-04
Lepomis miniatus	—	—	—	—	—	—	—	0.17	—	—	—	0.08	—	—	—	—
Lepomis symmetricus	—	—	—	0.36	0.40	0.70	8.42	—	—	—	0.10	—	0.64	0.90	0.96	0.67
Menidia beryllina	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Micropterus salmoides	0.20	—	—	—	0.30	0.20	1.08	0.26	0.26	0.09	—	—	—	—	0.32	—
Minytrema melanops	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Morone mississippiensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Notemigonus crysoleucas	—	—	0.10	0.72	0.60	0.60	4.84	—	0.09	0.09	0.10	0.80	0.08	0.75	0.48	0.40
Notropis maculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Noturus gyrinus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Opsopoeodus emiliae	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Percina caprodes	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pimephales vigilax	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Pomoxis annularis	—	—	—	—	0.10	—	—	—	—	—	—	—	—	—	—	—
Pomoxis nigromaculatus	—	—	—	—	—	0.30	1.61	0.17	0.09	0.09	0.30	0.32	—	—	—	—
Sander canadensis	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Semotilus atromaculatus	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—
Umbra limi	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—

1883

1884