



Effects of Sucker Gigging on Fish Populations in Oklahoma Scenic Rivers

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Executive Summary

Suckers (Catostomidae) are ecologically important, and some support popular fisheries, despite not being considered ‘sport fish’ in most states. Gigging suckers is a popular and culturally significant pastime in the Ozark Highlands, but little is known about the effect of gigging harvest on population dynamics of suckers. Therefore, research is needed to determine safe levels of sucker harvest that ensure sustainability of sucker gigging and protect overall ecosystem function. The objectives of this study were to: 1) determine the spatial distribution of common sucker species during spawning season (when sucker gigging is most effective), 2) determine the population size, age structure, and total mortality rate for common sucker species, and 3) model the effects of different harvest rates on sucker populations to determine the harvest rate at which growth overfishing and recruitment overfishing begin. Suckers were sampled using electrofishing, modified fyke netting, gillnetting, hoop netting, and seining and marked with passive integrated transponder (PIT) tags to provide information about population size, demographics, and coarse-scale movement patterns. A subset of fish sampled using the above gears and additional fish collected during gigging tournaments in 2017-2019 and 2021-2022 (no tournament was held in 2020) were used for age analyses. Tournament data collected prior to the initiation of this project were obtained from the state agency. Data from gigging tournaments indicated Golden Redhorse *Moxostoma erythrurum*, Black Redhorse *M. duquesnei*, White Sucker *Catostomus commersonii*, and Spotted Sucker *Minytrema melanops* were vulnerable to gigging harvest. Selection by giggers for larger individuals was apparent for all species except Golden Redhorse in 2019. Spotted Suckers constituted most fish harvested, but the proportion of each species harvested still varied among years. A total of 943 fish were aged from samples obtained from 2017 to 2022 and results from subsequent analyses indicated a high degree of variation in growth rates within and among species. Over 4,700 suckers were tagged with PIT tags and over 400 recaptures of these tagged fish were made since autumn 2018. Preliminary analyses indicate survival was consistent across samples and species, and detection rates varied by sampling event (3-month periods). Our most likely top multistrata model suggested that a large portion of fish within the upper Spavinaw, lower Spavinaw, and reservoir sections remain in these locations year-round (means: 0.46 –

0.67). Despite this, transition probabilities are still high for movement from upper Spavinaw to lower Spavinaw (mean: 0.32) and from lower Spavinaw to upper Spavinaw (mean: 0.38). Likewise, transition probabilities were high for movement from lower Spavinaw to the reservoir (mean: 0.15) and from the reservoir to lower Spavinaw (mean: 0.32). Transition probabilities between upper Spavinaw and the reservoir were low in both directions (means < 0.01). Population sizes, growth trajectories and length-weight relationships varied among species. Preliminary harvest models suggest species-specific regulation may be scientifically appropriate; however, it may be difficult for giggers to identify species while gigging. Based on our model results, there appears to be little risk of recruitment or growth overfishing for any species at current exploitation levels.

OBJECTIVES:

1. Determine the spatial distribution of common sucker species during spawning season (when sucker gigging is most effective).
2. Determine the population size, age structure, and total mortality rate for common sucker species.
3. Model the effects of different harvest rates on sucker populations to determine the harvest rate at which growth overfishing and recruitment overfishing begin.

Study area

Our sampling was conducted in the Spavinaw Creek catchment upstream of Lake Eucha Dam in northeast Oklahoma and northwest Arkansas. This system includes one reservoir (Lake Eucha) and several interconnected streams (Figure 1). This stream-reservoir complex is located within the Ozark Highlands ecoregion. This area is primarily forest, woodland, and pasture and characterized by karst topography and numerous springs (Nigh and Shroeder 2002; Woods et al. 2005). At full pool, Lake Eucha is a 902-ha reservoir that was created by damming Spavinaw Creek in 1952 (Jackson 1957). An Oklahoma Water Resources Board survey conducted in December of 1999 suggested the reservoir had a mean depth of 8.2 m and a maximum depth of 25.6 m; however, the surface area of the reservoir was 1,136 ha at the time suggesting it was above the spillway level (i.e., depths may not be representative of average conditions; Oklahoma Water Resources Board 2002). The streams of this ecoregion have a predominantly

coarse bed (i.e., average substrate diameter > 6.5 mm) and carry low sediment loads during baseflow (Nigh and Schroeder 2002).

Objective 1. Determine the spatial distribution of common sucker species during spawning season (when sucker gigging is most effective).

Background

Many catostomids in Oklahoma spawn between February and May (e.g., Black Redhorse *Moxostoma duquesnei*, Bowman 1970; White Sucker *Catostomus commersonii*, Corbett and Powles 1983; Northern Hogsucker *Hypentelium nigricans*, Matheney and Rabeni 1995). During this period, catostomids present in lentic habitats migrate into or near lotic systems (Green et al. 1966; Manny et al. 2010). Catostomids occupying lotic system remain in moving water; however, migrations within (e.g., downstream to upstream) or between (e.g., main stem to tributary) lotic systems may occur (Bowman 1970; Kwak and Skelly 1992). During spawning, catostomids congregate on or near lithic shoals (Mcswain et al. 1972; Curry and Spacie 1984; Kwak and Skelly 1992). Males are usually the first to arrive on spawning shoals and the last to leave, sometimes agonistically defending their locations (Mcswain et al. 1972; Kwak and Skelly 1992; Reid 2006). Females arrive later and stay for shorter periods of time (Kwak and Skelly 1992; Reid 2006). Following spawning, catostomids migrate away from these spawning areas (Raney and Webser 1942; Reid 2006).

Suckers may be particularly vulnerable to harvest by gigging during the spawning season given their behavior of aggregating around lotic shoal habitats (Raney and Webster 1984; Beckman and Howlett 2013). Sucker gigging is a popular and culturally significant pastime in the Ozark Highlands (Sarvis 2002). Though fishing effort has been quantified for some nontraditional fishing methods (e.g., darkhouse spearing, Pierce and Cook 2000), no information is available on susceptibility of fishes to gigging. Estimating susceptibility of fishes to these harvest methods is increasingly important, especially in regions where these methods are popular (e.g., Ozark Highlands).

Approach for Determining Spawning Locations

The primary means of capturing suckers during sampling (2019-2021) was electrofishing. Electrofishing gears included backpack (Smith-Root LR-24 and LR-20B), barge (Midwest Electrofishing Systems MLES Infinity box), and boat mounted electrofishing (Midwest Electrofishing Systems MLES Infinity box, Smith-Root 5.0 GPP). All three electrofishing gears were used within lotic waters due to variable water depth. When the electric field emitted by a single backpack electrofishing unit could effectively cover the entire stream width only the LR-24 was used, otherwise both units were operated in tandem. In sections where stream widths could not be effectively covered two backpack electrofishers, the barge electrofishing system was used. The barge electrofishing system consisted of an MLES Infinity box placed onto a Stealth Mini-Boat (Midwest Lake Management, Polo, Missouri) that acted as the cathode with two independently operatable anode poles. Backpack and Barge electrofishing was done moving upstream and always started at the downstream end of a sampling site (Rabeni et al. 2009). In stream sections where the barge could not be operated due to water depth, the boat mounted electrofishing system was used. Only the MLES Infinity box mounted electrofishing was used within the streams. The MLES boat electrofishing system used the aluminum hull of the boat as the cathode and one MLES collapsible boom-mounted anode. In streams, the electrofishing boat was operated from downstream to upstream targeting structure (e.g., large woody habitat) and generally operated from deep-to-shallow to “push” suckers to shallower areas where they could be captured by the dip netter. Both the MLES Infinity box and Smith-Root electrofishing systems were used on Lake Eucha as prior work suggested there was no statistical difference in catch rate for suckers (Zentner et al., in press). The Smith-Root electrofishing system used the aluminum hull of the boat as the cathode and two Smith-Root SAA-6 boom-mounted anodes. In the reservoir, electrofishing was conducted by maneuvering the boat towards and away from the shoreline, taking special care to sample submerged structures (e.g., emergent macrophytes, boulders). All electrofishing outputs were adjusted based on the ambient conductivity ($\mu\text{S}/\text{cm}^3$) of the water to standardize peak power to the fish based on the tables present in Miranda (2009).

To help reduce any bias in our mark-recapture estimates due to electrofishing gear bias, suckers were also collected with modified fykenets, hoopnets, gillnets, and seines. All netting gears except seines were used in both lotic and lentic systems. Seines were only used in lotic systems and were 1.3-m tall and 7.9-m wide and made of fine mesh (~1 mm delta). Seines were either pulled in a downstream direction within runs and pools or placed at the end of a riffle and fish were herded towards the seine via kicking and disturbing substrates (Rabeni et al. 2009). Modified fyke nets were made of 10-mm bar mesh and had a 19.2-m lead attached to a 0.9×1.8 -m (height, width) heart ending in four 0.8-m diameter hoops with one crowfoot-style throat on the first hoop. Modified fyke nets were placed perpendicular to the shoreline in both lotic and lentic systems and fished overnight. Hoop nets were 2.8 m long with 25.4-mm bar mesh and three hoops that narrowed in diameter from 0.7 to 0.5 m from heart to cod end. Hoop nets were placed with the opening of the net facing downstream in lotic systems (Guy et al. 2009). In the reservoir, hoop nets were deployed parallel to the shore with the opening facing either up- or down-reservoir randomly decided. Hoop netting within the reservoir was unsuccessful at capturing suckers and was discontinued in 2019. Gill nets were 2 m deep and 7.62 m long and contained eight panels of 12.7-, 15.9-, 19.1-, 25.4-, 38.1-, 50.8-, 63.5-, 76.2-mm bar mesh. Gillnets were used differently in lotic systems and Lake Eucha. In lotic systems the tag ends of the gillnet (i.e., float and lead line ends) were affixed to polyvinyl chloride pipe and the gillnet was pulled like a seine through pools to capture fish. In Lake Eucha gillnets were used passively with anchors attached to the lead line and buoys attached to the floatline to stretch the mesh. Passive gillnet sets were fished for approximately one hour prior to retrieval. Gillnetting was discontinued on Lake Eucha in 2019 due to low catch rates (i.e., $n < 10$ suckers) and the majority of captured suckers appeared to be too stressed to survive if tagged and released.

A subset of suckers captured during mark-recapture sampling during the spawning season and fish carcasses collected from annual gigging tournaments were measured to the nearest mm total length (TL) then evaluated for maturation based on the presence or absence of developed gonads, assessed via dissection (Meyer 1962; Trippel and Harvey 1991). Maturity curves were estimated using a binomial model via the “glm” function in program R version 4.1.3 (R core team 2022). The response variable

was maturity coded as a dummy variable (i.e., 1 = mature, 0 = immature) and the predictive variable was TL. To determine if maturity curves varied significantly ($\alpha = 0.05$) between sexes, preliminary binomial models were also fit with a sex (male or female) variable. Maturity curves were fit separately for sexes if the sex variable was significant, otherwise a single maturity curve was fit for both sexes. Maturity curves were used to determine the TL of each species at 50% and 95% maturation. Length data from field samples were then evaluated considering these maturation sizes to estimate where mature fish were located during the spawning season. Because these data provided little information regarding spawning areas in the Eucha-Spavinaw stream-reservoir complex (i.e., mature fish were found throughout the system), multistrata models (Hestbeck et al. 1991; Brownie et al. 1993) were constructed using tagging data to determine how suckers moved seasonally at the population level.

Multistrata models allow estimates of coarse spatial movements between defined units. Our system was broken into three distinct spatial units: the reservoir, lower Spavinaw Creek, and upper Spavinaw Creek. Strata for the model were defined based on the environmental characteristics of Spavinaw-Eucha stream-reservoir complex (Figure 1). The Lake Eucha strata began at the uppermost part of the reservoir (defined by reduction of flowing water and termination of a visible stream channel) and terminated at the dam. Lake Eucha was selected as a stratum due to the fact it represents a lentic to semi-lentic system and is frequently used by giggers (primarily the upper-end and lower portions). The lower Spavinaw Creek stratum began at the uppermost part of the reservoir and terminated a little over one third of the way up the system (specifically: 36.33 and -94.63 UTM). Whereas the upper strata of Spavinaw Creek began where the lower section of Spavinaw Creek terminated and continued to the headwaters of the system. Spavinaw Creek was divided into two separate strata due to the upper section being intermittently disconnected from the lower section during periods of low flow and, to the best of our knowledge, giggering only occurring in the lower section.

Sampling for Multistrata models was conducted seasonally in autumn (September, October, November), winter (December, January, February), spring (March, April, May [focused around spawning]), and summer (June, July, August). Within each of the defined strata (upper Spavinaw, lower Spavinaw, Lake Eucha), nine sites were sampled

using one or multiple gears (gears were defined *a priori* based on depths). Due to the parameter-heavy nature of multistrata models, additional recapture information was obtained with mobile passive integrated transponder (PIT) antennas to produce as many detections as possible. Mobile PIT antennas were made of 31.8- or 50.8-mm diameter polyvinyl chloride piping and measured either 3.2 by 1.5 m or 1.9 by 1.5 m and contained 12-AWG (American wire gauge) insulated stranded wire (Southwire, Carrollton, Georgia; Zentner et al. 2021). In lotic systems, PIT antennas were operated following Zentner et al. (2021). In Lake Eucha PIT antennas were operated by placing the booms perpendicular to an electrofishing boat and attaching the antennas with rope to the leading end of the electrofishing booms to limit interference from the hull of the boat.

Multistrata models allowed us to determine if fish ≥ 250 mm TL (minimum size tagged) exhibited coarse spatial movement patterns across our strata while accounting for survival (specifically apparent survival [Φ]) and capture probability (p). Coarse spatial movement estimates came in the form of transition probabilities (ψ). Transition probabilities represent the likelihood of a tagged fish from each stratum either staying in the stratum or transitioning to one of the other two defined strata. To determine the most likely coarse-movement pattern within the system while still accounting for factors influencing apparent survival and capture probability, a candidate model set was constructed. Multiple models were constructed in which apparent survival and capture probability were either allowed to remain constant throughout the sampling period or to vary by sample season, strata, or species. Transitional probabilities were also tested in different models where they either remained constant regardless of species or season or either varied by season or species. All Multistrata models were fit in program MARK (White and Cooch 2016).

An information theoretic approach was used to determine if either apparent survival (ϕ), capture probability (p), or transition probabilities (ψ) varied by season, strata, and species, or were invariant for these variables across our samples. Candidate models used for this approach contained all possible combinations of parameter types (i.e., parameters varied by a single covariate or held constant) as well as a fully saturated model (i.e., all additive effects of season, strata, and species as coefficients for each

parameter). Initially candidate models were ranked using Akaike information criterion corrected for small sample size (AICc; Hurvich and Tsai 1989). However, overdispersion ($\hat{c} > 1$) was apparent in several of our models. Therefore, quasi-AICc (QAICc) estimates were used to rank models (Williams et al. 2002). Models within 2.0 Δ QAICc of the top candidate model were considered to have similar likelihoods (Burnham and Anderson 2002). To help confirm the 2.0 Δ QAICc cutoff value, QAICc weights (w_i) were estimated and used to obtain evidence ratios via Kullback–Leibler information theory (Royall 1997; Burnham and Anderson 2002) relative to the top-ranked model. A final Bayesian formulation of the top ranked multistrata model was then used to better estimate confidence intervals around parameter estimates (Dyer and Brewer 2020). We used the rjags package (Plummer et al. 2019) to fit the multistrata models by using a Bayesian approach with Gibbs sampling in JAGS (Plummer 2003). The final model was fit using uninformative uniform priors for survival and capture probability and a multinomial prior for transition probabilities approximated via uninformative normal priors on the logit scale. Each mark–recapture model consisted of three Markov chain Monte Carlo (MCMC) iterations with 20,000 burn-in samples and 500,000 post-burn-in samples at a thinning rate of 100. MCMC chains were assessed for convergence post simulation by using visual diagnostic plots and a Gelman–Rubin diagnostic test (convergence was confirmed at $\hat{r} \leq 1.1$; Albert 2009).

Approach for Estimating Susceptibility to Harvest

Information about the spatial distribution of catostomids during the spawn was combined with harvest information collected from giggers to determine the susceptibility of these fishes to harvest during the spawning season. Harvest was also estimated by measuring the proportion of previously PIT-tagged suckers that were harvested at gigging tournaments. Suckers were PIT tagged from 2018-2021 and the 2019, 2021, and 2022 gigging tournaments were monitored for tag returns (and for objective 2 as described below). The 2020 tournaments were canceled due to the COVID-19 pandemic. Gigging tournaments took place in the spring over the course of two weekends. The first tournament occurred over two nights and giggers harvested the maximum number of fish possible over the course of two hours each night. The second

tournament for youth giggers (< 18 years old) was conducted in a single night by separating into two groups and allowing each group to harvest as many fish as possible for an hour. The exploitation rate was estimated by dividing the number of PIT-tagged suckers harvested at the tournaments by the number of PIT tags at large at that time. Exploitation estimates based on tournament PIT tag returns were adjusted using estimates of survival and tag loss (Miranda et al. 2002). Survival was estimated using a weighted catch curve and tag loss using the discrete method (Chapman et al. 1965; Wetherall 1982).

To determine if the sizes of suckers harvested in gigging tournaments were similar to those observed in the population, we compared TL from harvested suckers with TL of fish susceptible to our gear in 2019 and 2021 (only years with both samples available) using a Kolmogorov–Smirnov test (Kolmogorov 1933; Smirnov 1939). We used a Bonferroni-adjusted α value of 0.005 for all Kolmogorov–Smirnov tests as multiple comparisons ($n = 10$) were made for each species in each year (Bonferroni 1936). Because the Kolmogorov–Smirnov test can be influenced by sample size, we also compared the distributions using a distribution-free estimate of overlap ($\hat{\eta}$), which estimated the amount of overlap between the two distributions based on their estimated density functions (Pastore and Calcagni 2019). Means and ranges of $\hat{\eta}$ were estimated via 10,000 bootstrap replicates. The same approach was used to determine if the sizes of suckers harvested varied among years using harvest data from 2017, 2018, 2019, 2021, and 2022 tournaments. The frequency that each species was harvested in the tournaments was also compared using a χ^2 -test ($\alpha = 0.05$, Pearson 1900). Following this, 10,000 bootstrap replicates were used to estimate Cramér’s V statistic (Acock and Stavig 1979) to determine if there was any relationship between the frequency with which each species was harvested and tournament year. We interpreted Cramér’s V based on its relationship to the w-statistic when a table has four rows, as suggested by Cohen (1988). Therefore, $V = 0.06$, 0.17 , and 0.29 were the thresholds for low, moderate, and strong association between categories, respectively.

Objective 2. Determine the population size, age structure, and total mortality rate for common sucker species.

Background

Demographic information from fishes offers insight into their ecological roles and provides quantitative data that can be used for modeling and the development of best management practices. Although demographic information has been published for some sucker species (e.g., *Moxostoma* spp., Reid 2009), quantitative estimates vary among species' ranges (e.g., maximum body size, Rypel 2014). Therefore, it is important to derive system-specific estimates of key parameters to inform models used to manage specific species (e.g., growth coefficients for harvest models, Guy and Brown 2007; Slipke and Maceina 2010). From an ecological standpoint, differences in demographic information offer insight into life-history strategies that allow ecologically similar species to co-exist (e.g., Savannah River catostomids, Grabowski et al. 2008). No information regarding these life-history strategies currently exists for sympatric catostomids of the Ozark Highlands ecoregion.

Methods

Fish Tagging

We tagged 4,772 suckers (Black Redhorse, Golden Redhorse *M. erythrurum*, White Sucker, and Spotted Sucker *Minytrema melanops*) with 23-mm half duplex (HDX) passive integrated transponder (PIT) tags placed into the abdominal cavity. All fish were weighed (g) and measured (TL, mm) but only fish larger than 250 mm TL were tagged, as tagging was considered too stressful for smaller individuals. All fish also received a pectoral fin clip so discrete tag loss (Chapman et al. 1965; Wetherall 1982) could be estimated. Based on recaptures from the fall of 2018 to the spring of 2022 (maximum time at large = 984 days), we estimated discrete tag retention to be 87% for all suckers tagged over that 4.5-year period. Interestingly, retention varied by species (retention was estimated as 100% for White Sucker, Black Redhorse, Golden Redhorse, and Northern Hogsucker and 83% for Spotted Sucker). We built a nil-recapture model (Friedenberg et al. 2018) to estimate minimum population sizes for each species. The nil-recapture model was selected because it allowed us to attain relative estimates of population size even for

species for which no recaptures were observed, based on the amount of sampling effort expended and the number of fish tagged at large. This allowed us to estimate the population size of Golden Redhorse, the most cryptic of our species of interest, for which we only had one physical recapture case (antenna detections cannot be used for population estimation). Additional population estimates were made for Spotted Sucker, Black Redhorse, and Northern Hogsucker because we had enough recaptures to estimate population size via a Jolly-Seber estimator (Jolly 1965; Seber 1965) in program MARK (White and Cooch 2016). Apparent survival (ϕ), capture probability (p), and the entry parameter (Δ) were held constant in these models to make Jolly-Seber estimates theoretically comparable to nil-recapture estimates.

Age Estimation

Age estimates for each species were determined via pectoral rays and otoliths (sagittal, and asteriscus). When possible, otoliths or pectoral rays were extracted from suckers collected at sucker-gigging tournaments. Additional fish for age analysis were collected in conjunction with capture-recapture sampling for length classes that were not well represented in tournament harvest. Ages were estimated from fish using either pectoral fin rays or otoliths (sagittal or asteriscus; Lackmann et al. 2019). Fin rays were sectioned incrementally until a maximum age was obtained. To determine a maximum age from fin rays we took multiple sections, starting at the pectoral process working our way to the distal end of the fin in 0.5 mm increments until the next two sections produced lower age estimates. Sagittal and asteriscus otoliths were set-in two-part epoxy and thin sectioned via a transverse cut around the nucleus (Sylvester and Berry 2006) using an Isomet slow-speed saw (Secor et al. 1992; Thompson and Beckman 1995). Otoliths were wet sanded on 1,000 to 2,000 grit sandpaper and covered with a drop of baby oil to increase clarity of annuli. Both fin rays and otoliths were viewed under a variable power microscope. Age estimates were obtained by counting annuli that were identified as transparent bands adjacent to opaque bands (Sylvester and Berry 2006).

Population Demographic Characteristics

The population size for each species was estimated using fishery-independent capture-recapture sampling. Subsets of fish captured for population-size estimation were also measured (TL) and weighed (g) to develop length-weight relationships (Le Cren 1951). An exploratory ANOVA was built using all the species' length-weight data with species as a fixed factor in the analysis to determine if catostomid species in this system exhibited different weight-length relationships ($\alpha = 0.05$). We used the program R version 4.1.3 (R Core Team 2022) to fit the von Bertalanffy growth models using a nonlinear-least squares approach and estimated confidence intervals using 10,000 bootstrap iterations. Weighted catch curves were used to obtain mortality estimates for Spotted Sucker, Black Redhorse, and Northern Hogsucker. Weighted catch curves were fit by natural log transforming the number of fish caught from each age and regressing it against estimated ages to produce weights used to estimate total mortality via a weighted catch curve (Maceina and Bettoli 1998) in program R via the FSA package (Ogle et al. 2023). Weighted catch-curves were estimated using the first "fully recruited" age class, which were defined as the first fully recruited age class at peak catch (Smith et al. 2012). Ages 5 to 18 were used for Spotted Sucker, ages 5 to 11 were used for Black Redhorse and ages 6 to 13 were used for Northern Hogsucker. We obtained a mortality estimate for White Sucker by first estimating annual survival using the Chapman-Robson (Chapman and Robson 1960; Robson and Chapman 1961) estimator then converting survival estimates to mortality estimates. The Chapman-Robson estimate of survival was fit using all ages after the peak catch (Smit et al. 2012), resulting in ages 9 to 16 being used for White Sucker.

Preliminary estimates of apparent survival and capture probabilities were obtained for our species using a Cormack-Jolly-Seber model (CJS; Cormack 1964; Jolly 1965; Seber 1965, 1982). We used the rjags package (Plummer et al. 2019) to fit the CJS models using a Bayesian approach with Gibbs sampling in JAGS (Plummer 2003). All of the models were fit using uninformative uniform priors. Each mark-recapture model consisted of three Markov chain Monte Carlo (MCMC) iterations with 30,000 burn-in samples and 65,000 post-burn-in samples at a thinning rate of 40. MCMC chains were assessed for convergence post simulation by using visual diagnostic plots and a Gelman-

Rubin diagnostic test (convergence was confirmed at $\hat{r} \leq 1.1$; Albert 2009). An information theoretic approach was used to determine if either apparent survival (ϕ) or capture probability (p) varied based on sample event, species, or were invariant for species across our samples. Model likelihood was ranked using the deviance information criterion (DIC), with the lowest DIC value selected as the top model (Spiegelhalter et al. 2002; Albert 2009; Zentner et al. 2021).

Objective 3. Model the effects of different harvest rates on sucker populations to determine the harvest rate at which growth overfishing and recruitment overfishing begin.

Background

Proactive fisheries management offers better strategies relative to retroactive management (Bakelaar et al. 2004) for maintaining fisheries and protecting aquatic ecosystems (Murawski 2000; Morrison et al. 2001). This is especially true for harvest, as growth or recruitment overfishing can degrade stock quality for extended time periods (e.g., Atlantic Cod *Gadus morhua*, Rose 2004). Current statistical software allows for a preemptive estimation of safe harvest levels based on estimated population-level responses to angler harvest (Allen et al. 2013). Harvest estimates such as these are useful for successful management of fisheries.

Gigging tournaments are important social events drawing large crowds and can result in high harvest levels. Fish are typically harvested at night from specially rigged jon boats with bow railing and mounted lights (Turner 2014). Population-level effects of gigging are unknown, but evidence suggests harvest is inversely correlated with abundance of catostomids. For example, a 250% increase in fishing pressure reduced the relative abundance of the Northern Hogsuckers by 16% over seven years in Missouri (personal communication cited in Matheney and Rabeni 1995). Therefore, it is critical to understand the relationship between harvest intensity and population demographics so gigging can be sustainably managed.

Methods

Harvest Modeling

We developed a yield-per-recruit model for each sucker species to evaluate the expected effects of different harvest levels. The current model used demographic information from the von Bertalanffy growth curves, weight-length relationships, mortality information, and exploitation from the fishery to investigate the potential for growth overfishing for each species. Growth overfishing was defined as the point at which the average size at harvest is smaller than the size at which the maximum yield per recruit is realized (Allen and Hightower 2010). The yield-per-recruit model simulated recruitment using a constant recruitment of 1,000 age-0 individuals. This approach was selected as we were unable to reliably index recruitment for any of our sucker species. To estimate the average TL of each age class we used the mean parameter estimates from the von Bertalanffy growth curve for each species:

$$L_a = L_\infty (1 - e^{-k(a-t_0)})$$

Where L_a is length at age, L_∞ is asymptotic maximum size, k is the Brody growth coefficient, and t_0 is theoretical age at zero length. To approximate weight at each age we estimated average weight from length via a length-weight regression for each species:

$$W = aL^b$$

Where W represents weight, a represents the intercept of the regression, L represents length, and b represents the slope of the regression.

We then established vulnerability of each age class was established. Vulnerability (V) was estimated as knife-edge selectivity curve starting at either 205 (minimum size harvested) or 250 (majority of harvest) mm TL based on observed TLs from gigging harvest across tournaments. Survivorship per recruit at each age in the fished condition (lx_a) was then defined using the following equations:

$$lx_a = lx_{a-1}S_0(1 - u V_{a-1}); \text{ if } a > 1$$
$$lx_a = 1; \text{ if } a = 1$$

Where lx_{a-1} is survivorship from the previous age, S_0 is survival from natural mortality, u is the exploitation rate, and V_{a-1} is vulnerability (i.e., a combination of anglers' ability and desire to harvest fish of a certain size) of the previous age (Allen and Hightower 2010).

Once these relationships were established the vulnerable biomass per recruit was (Φ_{VB}) estimated using:

$$\Phi_{VB} = \sum_{a=1}^{a_{max}} lx_a W_a V_a$$

Where a_{max} represents the maximum estimated age for each species, W_a represents weight at age, and V_a represents vulnerability at age. Yield per recruit (Y) was then estimated via:

$$Y = u \times R \times \Phi_{VB}$$

Where u represents the exploitation rate and R represents the number of simulated recruits at age one.

Finally, we estimated Mean TL of fish harvested (μ_{TL}) using:

$$\mu_{TL} = \frac{(u \times R \times (\sum_{a=1}^{a_{max}} lx_a L_a V_a))}{(u \times R \times (\sum_{a=1}^{a_{max}} lx_a V_a))}$$

To determine how yield and average size at harvest would change, fishing exploitation rates ranging from 0.05 (higher than current tournament estimates) to 0.90 were then applied to vulnerable age classes. Parameters used for this model are available in Table 1.

A Length-Based Spawning Potential Ratio Growth-Type-Group (LBSP-GTG; Hordyk et al. 2016) model was used to investigate the potential for recruitment overfishing. Recruitment overfishing was defined as the point at which the population spawning potential ratio (SPR; Goodyear et al. 1993) drops below 30%. SPR is a ratio of harvested:unharvested population-level fecundity, where fecundity is proportionate to fish weight. An SPR of 30% was selected as it would allow for a more conservative estimate of recruitment overfishing as only tournament harvest data were used to fit the

model. The LBSP-GTG model was selected as it was designed for investigating the potential for recruitment overfishing in data poor fisheries (Hordyk et al. 2015, 2016). This model used the information presented above (i.e., weight-length relationships, growth parameters, mortality estimates) as well as a parameter to account for length-based susceptibility to harvest and maturity. Susceptibility to harvest was estimated based on TL using harvest data collected from 2017-2019 and 2021-2022. Maturity schedules were estimated by fitting length-at-maturity curves to data collected from each species during the spring spawning period (Figure 2). Spotted Sucker and Black Redhorse exhibited significantly different length at maturity curves for males and females. Therefore, female TL at 50 and 95% maturity was used for Spotted Sucker and Black Redhorse to generate the most conservative estimate of harvest.

The LBSP-GTG model estimates were fit via the LBSPR package (Hordyk 2021) in program R. The LBSP-GTG model assumes vulnerability using logistic selectivity with the following equation:

$$V_L = \frac{1}{1 + e^{(-\ln(19)\left[\frac{L-L_{50}}{L_{95}-L_{50}}\right])}}$$

Where V_L represents the vulnerability of an individual at length L , L_{50} represents the length where 50% of fish are vulnerable, and L_{95} represents the length where 95% of the fish are vulnerable.

The LBSP-GTG (Hordyk et al. 2016) model allows a standardized (i.e., sum to 1) estimate of the cumulative density of animals across length classes in recursive fashion using:

$$\tilde{D}_{L+dL,g} = \frac{\frac{1}{\theta L} (N_{L,g} - N_{L+dL,g})}{\sum_L \frac{1}{\theta L} (N_{L,g} - N_{L+dL,g})}$$

Where $\tilde{D}_{L+dL,g}$ represents the expected proportion of the population in each growth-type-group (g) per small-length increment (dL) in terms of asymptotic maximum length (L_∞), instantaneous mortality (Z), and the Brody growth coefficient (k). The θL parameter is the represents the ratio of length specific mortality over the Brody growth coefficient. The $N_{L,g}$ and $N_{L+dL,g}$ parameters represent the number of individuals alive per length (L)

at growth-type-group (g) before and after a change in small-length increment (dL) estimated recursively using:

$$N_{L+dL,g} = N_{L,g} \left(\frac{L_{\infty,g} - L - dL}{L_{\infty,g} - L} \right)^{\frac{Z_{L,g}}{k}} \text{ for } L < L_{\infty}$$

Where $L_{\infty,g}$ and $Z_{L,g}$ are derived estimates of the asymptotic maximum length and instantaneous mortality for the growth-type-groups approximated from the values given to the LBSPR package (Hordyk 2021). The initial condition for $N_{L+dL,g}$ is given by:

$$N_{L=0,g} = R_0 p_g$$

Where $R_0 = 1$ for the per-recruit formulation and p_g represents the fraction of recruits to each group (g) that is proportional to the normal probability density function:

$$(\mu = \bar{L}_{\infty}, \sigma = \sigma_{L_{\infty}}^2,)$$

These estimates can then be used to estimate SPR (i.e., proportion of reproduction in the fished and unfished state) using the following equation:

$$SPR = \frac{\sum_g \sum_L \frac{1}{\left(\left[\frac{M}{k} \right]_{L,g} + V_L \left[\frac{F}{M} \right] \left[\frac{M}{k} \right]_{L,g} \right)} (\tilde{D}_{L,g} - \tilde{D}_{L+gL,g}) Fec_{L,g}}{\sum_g \sum_L \frac{1}{\left(\left[\frac{M}{k} \right]_{L,g} \right)} (\tilde{D}_{L,g} - \tilde{D}_{L+gL,g}) Fec_{L,g}}$$

Where F represents the estimated instantaneous fishing mortality rate, M represents the estimated natural instantaneous mortality rate, and $Fec_{L,g}$ represents the estimated fecundity at size given by:

$$Fec_{L,g} = Mat_{L,g} L^{\beta}$$

Where $\beta = 0$ and $Mat_{L,g}$ represents the estimated maturity schedule based on following relationships:

$$L_{50,g} = \frac{\bar{L}_{50}}{\bar{L}_{\infty}} L_{\infty,g}$$

$$L_{95,g} = \frac{\bar{L}_{95}}{\bar{L}_{\infty}} L_{\infty,g}$$

Where \bar{L}_{50} and \bar{L}_{95} represent the mean length at 50 and 95% maturity, respectively.

The instantaneous fishing and natural mortality rates were not estimable from our data. Therefore, the instantaneous natural mortality rate (M) was approximated using:

$$M = 4.118k^{0.73}L_{\infty}^{-0.33}$$

as recommended by Then et al. (2015). The instantaneous fishing mortality rate was then estimated by rearranging equations present in Miranda and Bettoli (2007) to obtain:

$$F = \frac{\mu M}{(M(1 - e^{-z})/z)}$$

Where μ represents the interval fishing mortality estimated from the gigging tournaments based on tag-return data. Parameters used for this model are available in Table 1.

RESULTS AND DISCUSSION

Objective 1. Determine the spatial distribution of common sucker species during spawning season (when sucker gigging is most effective).

A total of 4,772 suckers ≥ 250 mm TL were tagged in the Spavinaw-Eucha complex between the fall of 2018 and the summer of 2021 and a total of 305 recaptures were obtained (Table 2, Figure 3). Tag-return data from the 2019, 2021, and 2022 gigging tournaments document that individual suckers undergo large migrations (i.e., ≥ 10 km) in a month or less and suggest tributaries to Lake Eucha are important spawning grounds. Our information theoretic approach indicated the most likely multistrata model to describe sucker movements within the river-reservoir complex was a model with constant transition probabilities between states (i.e., not varying by season; Table 3) for tagged suckers ≥ 250 mm TL (all species, N tagged = 4,772). This most likely model also suggested constant apparent survival and that capture probabilities varied by strata. Mean apparent survival was 0.57 (95% confidence interval (CI) = 0.38 – 0.82). Mean capture probability was 0.01 (95% CI = 0.01 – 0.01) in upper Spavinaw, 0.45 (95% CI = 0.01 – 0.99) in lower Spavinaw, and 0.57 (95% CI = 0.07 – 0.96) in Lake Eucha. Mean estimates from the top multistrata model suggested that most fish within the upper

Spavinaw section are sedentary (mean: 0.65), though roughly a third of the population moved to lower Spavinaw (mean: 0.32), and an extremely small proportion moved to the reservoir (mean: <0.01). Within lower Spavinaw, on average less than half the fish remain within the section (mean: 0.46), with over one-third of the fish moving to upper Spavinaw (mean: 0.38), and a smaller portion transitioning to the reservoir (mean: 0.15). Within the reservoir, most suckers remain within the section (mean: 0.67), with almost one-third of the fish moving to lower Spavinaw (mean: 0.32), and an extremely small portion transitioning to upper Spavinaw (mean: <0.01). These transitions appear highly variable (based on 95% credible interval) for all state-based transitions except the transition between the reservoir and upper Spavinaw (Figure 4). This suggests a large amount of variation in movements between states occurred throughout our study, but this variation was not related to season or species (based on w_i and evidence ratios; Table 3). This could also indicate we did not attain a large enough number of recaptures to properly estimate the difference in transition probabilities between species or seasons.

During the 2019 gigging tournaments, 11 tagged Spotted Sucker were harvested (N at-large, tagged fish = 372) and 1 tagged redhorse species was harvested (N at-large tagged fish = 147). No tagged White Sucker or Northern Hogsucker were harvested during the 2019 gigging tournaments; however, 2 untagged White Sucker and 28 untagged Northern Hogsucker were observed. During the 2021 gigging tournaments, 29 tagged Spotted Sucker were harvested (N at-large, tagged fish = 3,376), 8 tagged Redhorse spp. were harvested (N at-large tagged fish = 713), and 1 tagged Northern Hogsucker was harvested (N at-large tagged fish = 191). No tagged White Sucker were harvested during the 2021 gigging tournaments. During the 2022 gigging tournaments, 10 tagged Spotted Sucker were harvested (N at-large, tagged fish = 3,778), 2 tagged Redhorse spp. were harvested (N at-large, tagged fish = 714), and 1 tagged Northern Hogsucker was harvested (N at-large tagged fish = 195). No tagged White Sucker were harvested during the 2022 gigging tournaments; however, 17 untagged White Sucker were harvested. Using these data, exploitation estimates ranged from 0.3 to 2.4% (all species pooled) after adjusting for survival and tag loss. Species specific exploitation estimates ranged from 0.0% (White Sucker [no recaptures]) to 3.6% (Spotted Sucker).

For all species, harvest at the gigging tournament did not reflect the same size structure we observed in our fishery-independent sampling, except for Golden Redhorse in 2019 (Kolmogorov-Smirnov tests, significance assessed as $p < 0.05$; Figure 5). In 2019, giggers were more likely to harvest larger individuals (typically > 300 mm TL) from the population (Figures 5 and 6) and there was little-to-moderate overlap (Figure 6). In 2021, though giggers appeared to continue to harvest larger individuals (Figures 7 and 8), the overlap increased between size distributions from gigging and our fishery-independent sampling for each species except Golden Redhorse. This suggests that, on average, the fish harvested in 2021 were slightly larger than in our electrofishing samples, but the difference was not as large as it was in 2019. No fishery-independent data were available for 2018 or 2022, and no gigging tournament was held in 2020, so 2019 and 2021 are the only years where these comparisons could be made.

Both significance testing (p-values) and effect size estimates ($\hat{\eta}_{mean}$, $\hat{\eta}_{range}$) suggest larger Spotted Sucker, Northern Hogsucker, and Black Redhorse were more vulnerable to gigging harvest than smaller fish. We are unable to confirm relationships for White Sucker and Golden Redhorse as few fish were harvested at tournaments in 2019 and 2021 ($N \leq 9$). Kolmogorov-Smirnov tests comparing the size distribution of harvest from 2017-2019 and 2021-2022 also showed there was inter-annual variability in harvest (Figure 9).

Total lengths of Spotted Sucker harvested during all five years were significantly different from one another ($p < 0.005$; Bonferroni adjusted to maintain $\alpha = 0.05$ across multiple comparisons; Figure 9; Table 4). Total lengths of redhorse species harvested in 2018, 2019, 2021, and 2022 were all statistically different from one another and those harvested in 2017 were all statistically different from 2019 (Bonferroni-adjusted $p < 0.005$; Figure 9; Table 4). All other TL comparisons from harvested species between years were similar (Bonferroni-adjusted $p > 0.005$; Figure 9; Table 4). Despite the varying significance between sizes of fish harvested in different years, most statistically different samples exhibited at least moderate overlap ($\hat{\eta}_{mean}$ and $\hat{\eta}_{range}$) suggesting these differences, while significant, were not extreme (Table 4).

Spotted Sucker constituted most fish harvested during tournaments (Figure 10). Comparisons between the frequency ratios of each species harvested in tournaments

significantly varied between years ($p < 0.05$). Though frequency ratios were significantly different, there was only a small-to-moderate relationship with tournament year ($V_{\text{range}} = 0.14-0.18$). Therefore, it is probable the significant difference in frequency of species harvested each year is the result of the variation within the less abundant species (i.e., White Sucker, redhorse species, Northern Hogsucker).

Size-distribution data confirm that tagging suckers with a TL ≥ 250 mm allowed us to estimate parameters for the harvested population as only 26 individuals under 250-mm TL were harvested during the 5 tournament years (total harvest for those years was 6,283 fish). This constitutes $< 0.01\%$ of the harvested population over this time period.

Objective 2. Determine the population size, age structure, and total mortality rate for common sucker species.

Age and growth data were collected from 943 fish obtained at the gigging tournaments held in the spring from 2017-2019 and 2022 and sampling events conducted from 2019-2021. In total 676 Spotted Sucker, 153 Black Redhorse, 56 Northern Hogsucker, 49 White Sucker, and 7 Golden Redhorse were aged. Age estimates ranged from 0-18 years. Maximum age varied by species, with Golden Redhorse appearing to have the shortest longevity (maximum age observed was 9 years old) and Spotted Suckers being the oldest (maximum age observed was 18 years old). Maximum ages of Northern Hogsucker (max age observed 13 years old), Black Redhorse max age observed 11 years old, and White Sucker (max age observed 16 years old) fell between these species. All species exhibited slow and variable growth at older ages, with fish lengths similar from age 3 – maximum age (Figure 11). These growth models suggested fish reach typical adult sizes in about 3 – 4 years and grow little over the next 7 – 14 years.

All sucker species exhibited different weight-length relationships (Table 5). Independent length-weight curves were fit for all species. Spotted Sucker grow “plumper” as they increase in length (i.e., positive allometric growth, $\beta_{\text{estimate}} = 3.09-3.18$; $t = 5.85$, $df = 501$, $p < 0.01$). Interestingly, Black Redhorse, Northern Hogsucker, White Sucker and Black and Golden Redhorse did not become “plumper” or “thinner” as they grew, but instead appeared to maintain their body forms (i.e., isometric growth with $\beta_{\text{estimate}} \approx 3.0$, $t \text{ range} = 0.656 - 1.967$, $df \text{ range} = 11 - 480$, all $p \text{ range} = 0.05 - 0.52$).

Given the current sample size for Golden Redhorse length-weight data ($n = 13$), growth patterns for this species were inconclusive. Positive allometric growth for Spotted Sucker may be a result of the reservoir environment they primarily inhabit within the Spavinaw-Eucha complex, where drag is less of a factor than in lotic environments.

Using data from 2020-2022, we estimated all sucker populations reach 50% sexual maturity when they are between 233 – 333-mm TL and reach 95% sexual maturity between 304 – 385-mm TL (Figure 3). Northern Hogsucker reached 50 and 95% maturity the fastest, and female Black Redhorse the slowest. Maturity curves were significantly different (z range = 1.12 – 4.70, all $p < 0.05$) between male and female Spotted Sucker and Black Redhorse (Figure 3). Interestingly, male Spotted Sucker matured slower than male Black Redhorse, but the opposite was true for females. Maturity curves were similar (z range = -0.02 – 0.59, p range = 0.56 – 0.99) between males and females for Northern Hogsucker and White Sucker. We were unable to estimate a maturity curve for Golden Redhorse due to insufficient data.

During fishery-independent sampling, >4,700 individuals were tagged. Using mark-recapture data from our tagged population, we were able to generate estimates of apparent survival and detection probability for our tagged population (i.e., all species tagged). The top model suggested that apparent survival did not vary across seasons but capture probability did (Table 6, Figure 12). The median apparent survival estimate was 0.79. Mean annual mortality for our species ranged from 11 to 28%, with Northern Hogsucker exhibiting the lowest annual mortality and White Sucker exhibiting the highest (Table 7). Despite variation in mean estimates our 95% CIs for mortality overlapped for all species. We were unable to estimate a catch curve for Golden Redhorse due to insufficient age data.

Minimum population estimates of fish ≥ 250 mm TL (from nil-recapture model) varied for each species (Table 8) and were generally in agreement with the estimates from the Jolly-Seber open population model. Spotted Sucker had the largest estimated minimum population size (>60,000) and Golden Redhorse had the lowest estimated population size (>40). Though the nil-recapture model allowed us to estimate the minimum population sizes, estimates for Golden Redhorse and White Sucker were sufficiently imprecise that they only indicate that the population is likely quite small

(Table 8, Figure 13). The Jolly-Seber open population model estimated mean abundance of Spotted Sucker was 78,930 individuals (95% CI = 61,343 – 101,851), mean abundance of Black Redhorse was 6,132 individuals (95% CI = 4,566 – 8,318), and mean abundance of Northern Hogsucker was 6,616 individuals (95% CI = 1,483 – 31,194).

Objective 3. Model the effects of different harvest rates on sucker populations to determine the harvest rate at which growth overfishing and recruitment overfishing begin.

Our yield-per-recruit model suggested yield would be maximized at exploitation rates of 0.48, 0.50, and 0.62 for Spotted Sucker, Black Redhorse, and Northern Hogsucker, respectively based on a selectivity threshold of 250 mm TL (i.e., where the majority of harvest occurs; Figure 14). However, at a selectivity threshold of 205-mm TL (i.e., the smallest sucker occurring in harvest) exploitation rates of 0.27, 0.30, and 0.33 would maximize yield for Spotted Sucker, Black Redhorse, and Northern Hogsucker, respectively. Interestingly, yield was maximized at an exploitation level of 0.78 for White Sucker for both selectivity thresholds (Figure 14). In general, an increase in exploitation reduced the average size of fish harvested (based on mean TL, Figure 15). This phenomenon was more pronounced at a selectivity threshold of 250-mm TL for all species except White Sucker. This result indicates harvesting smaller fish would reduce the effects of increased harvest on the average size of fish captured for all species except White Sucker. Average size of fish harvested appeared to decrease fastest for Black Redhorse relative to the other four species in response to increased exploitation (Figure 15). Despite this, current levels of exploitation would have to increase ~10 to 15 times their current level to induce growth overfishing based on a selectivity threshold of 250 mm TL (i.e., where the majority of harvest is occurring) and ~5 to 15 times their current level to induce growth overfishing based on a selectivity threshold of 205 mm TL (i.e., most conservative estimate based on the smallest fish observed). Therefore, despite the differences where yield is maximized between species regulation of the giggering fishery is likely not necessary at current levels of harvest. It is likely that regulations would have little discernible effects on these populations until harvest caused recruitment overfishing (see below).

The LBSP-GTG Model suggested current SPR was > 0.82 for all species (based on the 2019, 2021, and 2022 exploitation estimates; Figure 16). This suggests recruitment overfishing (i.e., $SPR < 30$) is not occurring under the current level of harvest observed on the fishery. We also estimated SPR across a range of simulated fishing mortalities. This simulation suggested instantaneous fishing mortality estimates of 0.54 to 0.71 would result in recruitment overfishing (Figure 16). We also plotted SPR and Relative Yield against the ratio of instantaneous fishing (F) to natural (M) mortality (Figure 17). These plots indicated that recruitment overfishing was likely to occur and yield would be reduced when fishing mortality was over 80% of natural mortality. However, the effect of increased F/M ratios was more prominent for Black Redhorse and Spotted Sucker than for Northern Hogsucker and White Sucker.

RECOMMENDATIONS

Our data from the tournaments indicate giggers primarily harvest the most abundant species (Spotted Sucker) with the harvest of rarer species (e.g., White sucker, Golden Redhorse) being less common and more variable among years. Giggers appear to harvest larger fish, on average, than those present within the system. However, the size of harvested fish also varies among years. Current levels of harvest appear to be quite low and our current model predictions indicate the effects of harvest on size structure are quite small. It is possible that fishing pressure could eventually get high enough to lead to meaningful growth or even recruitment overfishing, but harvest would need to become much higher than the exploitation estimates we observed from the giggering tournaments.

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REFERENCES

- Acock AC, Stavig RG. 1979 A measure of association for nonparametric statistics. *Social Forces* 57:1381-1386.
- Albert, J. 2009. Bayesian computation with R. Springer Science+Business Media, New York, New York.
- Allen, M. S., R. N. M. Ahrens, M. J. Hansen, R. Arlinghaus. 2013. Dynamic angling effort influences the value of minimum-length limits to prevent recruitment overfishing. *Fisheries Management and Ecology* 20:247-257.
- Allen, M. S., and J. E. Hightower. 2010. Fish population dynamics: Mortality, growth, and recruitment. Pages 43–79 in W. A. Hubert and M. C. Quist, editors, *Inland fisheries management in North America*, third edition. American Fisheries Society, Bethesda, Maryland.
- Bakelaar, C. N., P. Brunette, P. M. Cooley, S. E. Doka, E. S. Millard, C. K. Minns, and H. A. Morrison. 2004. Geographic information systems applications in lake fisheries. Pages 133–152 in W. L. Fisher and F. J. Rahel, editors, *Geographic information systems in fisheries*. American Fisheries Society, Bethesda, Maryland.
- Beckman, D. W., and D. T. Howlett. 2013. Otolith annulus formation and growth of two redhorse suckers (*Moxostoma*: *Catostomidae*). *Copeia* 3:390-395.
- Bonferroni, C. E. 1936. Teoria statistica delle classi e calcolo delle probabilità. *Pubblicazioni del R Istituto Superiore di Scienze Economiche e Commerciali di Firenze* 8:1-62. (translated from Italian)
- Bowman, M. L. 1970. Life History of the Black Redhorse, *Moxostoma duquesnei* (Lesueur), in Missouri. *Transactions of the American Fisheries Society* 99:546-559.

- Brownie, C., J. E. Hines, J. D. Nichols, K. H. Pollock, and J. B. Hestbeck. 1993. Capture-recapture studies for multiple strata including non-Markovian transitions. *Biometrics* 49: 1173-1187.
- Burnham, K. P., and D. R. Anderson. 2002. *Model Selection and multimodel inference a practical information-theoretic approach*, second edition. Springer-Verlag, New York, New York.
- Chapman, D. G., B. D. Fink, and E. B. Bennett. 1965. A method for estimating the rate of shedding of tags from Yellowfin Tuna. *InterAmerican Tropical Tuna Commission Bulletin* 10:333-352.
- Chapman, D. G. and D. S. Robson. 1960. The analysis of a catch curve. *Biometrics* 16:354-368.
- Cohen, J. 1988. *Statistical power analysis for the behavioral sciences*. 2nd ed. New Jersey: Lawrence Erlbaum Associates.
- Corbett, B., and P. M. Powles. 1983. Spawning and Early-Life Ecological Phases of the White Sucker in Jack Lake, Ontario. *Transactions of the American Fisheries Society* 112:308-313.
- Cormack, R. M. 1964. Estimates of survival from the sighting of marked animals. *Biometrika* 5:429-438.
- Curry, K. D., and A. Spacie. 1984. Differential use of stream habitat by spawning catostomids. *American Midland naturalist* 111:267-279.
- Dyer, J. J., and S. K. Brewer. 2020. Seasonal movements and tributary-specific fidelity of Blue Sucker *Cyprinus elongatus* in a Southern Plains riverscape. *Journal of Fish Biology* 2020:1-14.
- Friedenberg, N. A., J. J. Hoover, K. Boysen, K. J. Killgore. 2018. Estimating abundance without recaptures of marked Pallid Sturgeon in the Mississippi River. *Conservation Biology* 32:457-465
- Goodyear C. P. Smith S. J., Hunt J. J., Rivard, D. 1993. Spawning stock biomass per recruit in fisheries management: foundation and current use. In: S.J. Smith, J.J. Hunt, D. Rivard (Eds.), *Risk evaluation and biological reference points for fisheries management*, Canadian Special Publications of Fisheries Aquatic Sciences, Canadian Special Publications of Fisheries Aquatic Sciences 120:67-81.
- Grabowski, T. B., N. L. Ratterman, and J. J. Isely. 2008. Demographics of the spawning aggregations of four catostomid species in the Savannah River, South Carolina and Georgia USA. *Ecology of Freshwater Fish* 17:318-327.

Green, G. H., T. G. Northcote, G. F. Hartman, and C. C. Lindsey. 1966. Life histories of two species of catostomid fishes in Sixteenmile Lake, British Columbia, with particular reference to inlet stream spawning. *Journal of the Fisheries Research Board of Canada* 23:1761-1788.

Guy, C. S., P. J. Braaten, D. P. Herzog, J. Pitlo, and R. S. Rogers. Warmwater fish in rivers. Pages 13–28 in S. A. Bonar, W. A. Hubert, and D. W. Willis, editors. *Standard methods for sampling North American freshwater fishes*. American Fisheries Society, Bethesda, Maryland.

Guy, C. S., and M. L. Brown, editors. 2007. *Analysis and interpretation of freshwater fisheries data*. American Fisheries Society, Bethesda, Maryland.

Hestbeck, J. B., J. D. Nichols, and R. A. Malecki. 1991. Estimates of movement and site fidelity using mark-resight data of wintering Canada Geese. *Ecology* 72:523-533.

Hordyk, A. 2021. LBSPR: Length-Based Spawning Potential Ratio. R package version 0.1.6 <https://CRAN.R-project.org/package=LBSPR>

Hordyk, A., K. Ono, J.D. Prince, and C.J. Walters. 2016. A simple length-structured model based on life history ratios and incorporating size-dependent selectivity: application to spawning potential ratios for data-poor stocks. *Canadian Journal of Fisheries and Aquatic Sciences*. 13: 1– 13.

Hordyk, A.R., K. Ono, S.R. Valencia, N.R. Loneragan, and J.D. Prince. 2015. A novel length-based empirical estimation method of spawning potential ratio (SPR), and tests of its performance, for small-scale, data-poor fisheries. *ICES Journal of Marine Science* 72: 217-231.

Hurvich CM, Tsai C. 1989. Regression and time series model selection in small samples. *Biometrika* 76:297-307.

Jackson, SW. 1957. Summary of a three-year creel census on Lake Eucha and Spavinaw Lake, Oklahoma, with comparisons of other Oklahoma reservoirs. *Proceedings of the Oklahoma Academy of Science* 1957:146-154.

Jolly, G. M. 1965. Explicit estimates from capture–recapture data with both death and immigration–stochastic model. *Biometrika* 52:225–247

Kolmogorov A. N. 1933. Sulla determinazione empirica di una legge di distribuzione. *Giornale dell’Istituto Italiano degli Attuari* 4:83-91. (translated from Italian)

Kwak, T. J., and T. M. Skelly. 1992. Spawning habitat, behavior, and morphology as isolating mechanisms of the Golden Redhorse, *Moxostoma erythrurum*, and Black Redhorse, *M. duquesnei*, two syntopic fishes. *Environmental Biology of Fishes* 34: 127-137.

- Lackmann, A. R., A. H. Andrews, M. G. Butler, E. S. Bielak-Lackmann, and M. E. Clark. 2019. Bigmouth Buffalo *Ictiobus cyprinellus* sets freshwater teleost record as improved age analysis reveals centenarian longevity. *Communications Biology*, 2. <https://doi.org/10.1038/s42003-019-0452-0>
- Le Cren, E. D. 1951. The length-weight relationship and seasonal cycle in gonad weight and condition in the perch (*Perca flavescens*). *Journal of Animal Ecology* 20:201–219
- Maccina, M. J., and P. W. Bettoli. 1998. Variation in Largemouth Bass recruitment in four mainstream impoundments on the Tennessee River. *North American Journal of Fisheries Management* 18:998–1003.
- Manny, B. A., G. W. Kennedy, J. C. Boase, J. D. Allen, E. F. Roseman. 2010. Spawning by Walleye (*Sander vitreus*) and White Sucker (*Satostomus commersoni*) in the Detroit River: Implications for spawning habitat enhancement. *Journal of Great Lakes Research* 36:490-496.
- Matheney IV, M. P., and C. F. Rabeni. 1995. Patterns of movement and habitat use by Northern Hog Suckers in an Ozark stream. *Transactions of the American Fisheries Society* 124:886-897.
- Mcswain, L. E., and R. M. Gennings. 1972. Spawning behavior of the Spotted Sucker *Minytrema melanops* (Rafinesque). *Transactions of the American Fisheries Society* 101:738-740.
- Meyer, W. H. 1962. Life history of three species of redhorse (*Moxostoma*) in the Des Moines River, Iowa. *Transactions of the American Fisheries Society* 91:412–419.
- Miranda, L. E., R. E. Brock, and B. S. Door. 2002. Uncertainty of exploitation estimates made from tag returns. *North American Journal of Fisheries Management* 22:1358-1363.
- Miranda, L. E. 2009. Standardizing electrofishing power for boat electrofishing. Pages 223-230 in Bonar SA, Hubert WA, Willis DW, editors. *Standard methods for sampling North American freshwater fishes*. Bethesda, Maryland: American Fisheries Society.
- Miranda, L. E., and P. W. Bettoli. 2007. Abundance, biomass, and production. Pages 229–277 in C. S. Guy and M. L. Brown, editors. *Analysis and interpretation of freshwater fisheries data*. Bethesda, Maryland: American Fisheries Society.
- Morrison, H. A., C. A. Minns, and J. F. Koonce. 2001. A methodology for identifying and classifying aquatic biodiversity investment areas: Application in the Great Lakes Basin. *Aquatic Ecosystem Health and Management* 4:1–12.
- Murawski, S. A. 2000. Definitions of overfishing from an ecosystem perspective. *ICES Journal of Marine Science* 57:649-658.

- Nigh, T. A., and W. A. Schroeder. 2002. Atlas of Missouri ecoregions. Missouri Dept. of Conservation, Jefferson City, MO.
- Ogle, D. H., J. C. Doll, A. P. Wheller, A. Dinno. 2023. FSA: Simple fisheries stock assessment methods version 0.9.4. Available: <https://fishr-core-team.github.io/FSA/>
- Oklahoma Water Resources Board. 2002. Water quality evaluation of the Eucha/Spavinaw lake system. Oklahoma Water Resources Board. Oklahoma City, OK.
- Pastore M, Calcagni A. 2019. Measuring distribution similarities between samples: A distribution-free overlapping index. *Frontiers in Psychology* 10(1089). available: <https://www.frontiersin.org/articles/10.3389/fpsyg.2019.01089/full> (December 2020).
- Pierce, R. B., and M. F. Cook. 2000. Recreational darkhouse spearing for Northern Pike in Minnesota: Historical changes in effort and harvest and comparisons with angling. *North American Journal of Fisheries Management* 20:239-244.
- Plummer, M. 2003. JAGS: a program for analysis of Bayesian graphical models using Gibbs sampling. In K. Hornik, F. Liesch, and A. Zeileis, editors. *Proceedings of the 3rd international workshop on distributed statistical computing*. R Foundation for Statistical Computing, Vienna.
- Plummer, M., A. Stukalov, and M. Denwood. 2019. rjags: Bayesian graphical models using MCMC version 4.10. Available: <http://mcmcjags.sourceforge.net>
- Quinn, S. P., and M. R. Ross. 1985. Non-annual spawning in the White Sucker, *Catostomus commersoni*. *Copeia* 3:613-618.
- R Core Team. 2022. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. <http://www.R-project.org>.
- Rabeni, C. F., J. Lyons, M. Mercado-Silvia, and J. T. Peterson. 2009. Warmwater fish in wadeable streams. Pages 13–28 in S. A. Bonar, W. A. Hubert, and D. W. Willis, editors. *Standard methods for sampling North American freshwater fishes*. American Fisheries Society, Bethesda, Maryland.
- Raney, E. C., and D. A. Webster. 1942. The spring migrations of the common White Sucker, *Catostomus c. commersonii* (Lacépède), in Skaneateles Lake inlet, New York. *Copeia* 1942:139-148.
- Reid, S. M. 2006. Timing and demographic characteristics of redhorse spawning runs in three Great Lakes basin rivers. *Journal of Freshwater Ecology* 21:249-258.

- Reid, S. M. 2009. Age, growth and mortality of Black Redhorse (*Moxostoma duquesnei*) and Shorthead Redhorse (*M. macrolepidotum*) in the Grand River, Ontario. *Journal of Applied Ichthyology* 25:178-183.
- Robson, D. S. and D. G. Chapman. 1961. Catch curves and mortality rates. *Transactions of the American Fisheries Society* 90:181–189
- Rose, G. A. 2004. Reconciling overfishing and climate change with stock dynamics of Atlantic Cod (*Gadus morhua*) over 500 years. *Canadian Journal of Fisheries and Aquatic Sciences* 61:1553-1557.
- Royall, R. M. 1997. *Statistical evidence: A likelihood paradigm*. Chapman and Hall, London, England.
- Rypel, A. L. 2014. The cold-water connection: Bergmann's rule in north American freshwater fishes. *The American Midland Naturalist* 183:147-156.
- Sarvis, W. 2002. A difficult legacy: Creation of the Ozark national scenic riverways. *The Public Historian* 24:31-52.
- Seber, G. A. F. 1965. A note on the multiple-recapture census. *Biometrika* 52:249–259.
- Seber, G. A. F. 1982. *The estimation of animal abundance and related parameters*, 2nd edition. Macmillan, New York, New York.
- Secor, D. H., J. M. Dean, and E. H. Laban. 1992. Otolith removal and preparation for microstructural examination. Pages 19–57 in D. K. Stevenson and S. E. Campana, editors. *Otolith microstructure and analysis*. Canadian Special Publication of Fisheries and Aquatic Sciences 117.
- Slipke, J. W., and M. Maceina. 2010. *Fishery analysis and modeling simulator (FAMS)*. Auburn University, Department of Fisheries and Applied Aquacultures, Agricultural Experiment Station, Auburn, AL.
- Smirnov NV. 1939. Estimate of derivation between empirical distribution function in two independent samples. *Bulletin of Moscow University* 2:3-16 (Translated from Russian)
- Smith, M. W., A. Y. Then, C. Wor, G. Ralph, K. H. Pollock, and J. M. Hoenig. 2012. Recommendations for catch-curve analysis. *North American Journal of Fisheries Management* 32:956–967.
- Spiegelhalter, D. J., N. G. Best, B. P. Carlin, and A. van der Linde. 2002. Bayesian measures of model complexity and fit. *Journal of the Royal Statistical Society Series B: Statistical Methodology* 4:583–639.

- Sylvester, R. M., and C. R. Berry. 2006. Comparison of White Sucker age estimates from scales, pectoral fin rays, and otoliths. *North American Journal of Fisheries Management*, 26:24–31.
- Then, A. Y., J. M. Hoenig, N. G. Hall, and D. A. Hewitt. 2015. Evaluating the predictive performance of empirical estimators of natural mortality rate using information on over 200 fish species. *ICES Journal of Marine Science* 72:82-92.
- Thompson, K. R., and D. W. Beckman. 1995. Validation of age estimates from White Sucker otoliths. *Transactions of the American Fisheries Society* 124:637–639.
- Turner, A. 2014. Fish Giggling: an Ozark tradition. *Missouri Conservationist Magazine*. Missouri Department of Conservation, Jefferson City.
- Trippel, E. A., and H. H. Harvey. 1991. Comparison of methods used to estimate age and length of fishes at sexual maturity using populations of White Sucker (*Catostomus commersoni*). *Canadian Journal of Fisheries and Aquatic Sciences* 48:1446-1459.
- Wetherall, J. A. 1982. Analysis of double-tagging experiments. U.S. National Marine Fisheries Service Fishery Bulletin 80:687–701.
- White, G., and E. Cooch. 2016. Program MARK, Fort Collins, Colorado: Colorado State University. Available: www.phidot.org/software/mark/background/index.html
- Williams, B. K., J. D. Nichols, and M. J. Conroy. 2002. Analysis and management of animal populations. Academic Press. San Diego, California.
- Woods, A. J., J. M. Omernik, D. R. Butler, J. G. Ford, J. E. Henley, B. W. Hoagland, D. S. Arndt, and B. C. Moran. 2005. Ecoregions of Oklahoma. Color poster with map, descriptive text, summary tables, and photographs. USGS, Reston, VA. Map scale 1:1,250,000.
- Zentner, D. L., J. J. Spurgeon., S. E. Lochmann, C. L. Graham. 2021a. Tag type and location-dependent retention impart varied levels of bias on mark-recapture parameter estimates. *North American Journal of Fisheries Management* 41:474-483.
- Zentner, D. L., S. L. Wolf, S. K. Brewer, D. E. Shoup. 2021b. A review of factors affecting PIT tag detection using mobile arrays and use of mobile antennas to detect PIT-tagged suckers in a wadeable Ozark stream. *North American Journal of Fisheries Management* 41:697-710.
- Zentner, D. L., S. K. Brewer, D. E. Shoup. In Press. Effects of environmental factors on sucker catch rate, size structure, species composition, and precision from boat

electrofishing. Journal of Fish and Wildlife Management. DOI:
<https://doi.org/10.3996/JFWM-22-052>

Tables

Table 1. Species parameters used in the yield-per-recruit and length based spawning potential ratio growth type group models. Parameter symbols denote the asymptotic maximum length (L_{inf}), time at age zero length (t_0), Brody growth coefficient (k), instantaneous mortality (Z), annual instantaneous mortality (A), slope of the log10 transformed length-weight regression (W_b), intercept of the log10 transformed length-weight regression, length at 50% selectivity (SL_{50}), length at 95% selectivity (SL_{95}), length at 50% maturity (M_{50}), and length at 95% maturity (M_{95}) estimated based on data from the Spavinaw-Eucha river-reservoir complex.

Parameter	Spotted Sucker	Black Redhorse	Northern Hogsucker	White Sucker
L_{inf}	368.763	384.568	322.800	354.327
t_0	-0.236	-0.396	-0.378	0.222
k	0.828	0.554	0.762	0.903
Z	0.119	0.216	0.182	0.314
A	11.236	0.194	0.166	0.275
W_b	3.042	3.097	3.193	3.083
W_a	5.025	5.234	5.384	5.188
SL_{50}	218.000	218.000	218.000	218.000
SL_{95}	245.000	245.000	245.000	245.000
M_{50}	285.000	306.000	245.000	257.000
M_{95}	341.000	349.000	310.000	322.000

Table 2. The number of each sucker species tagged, recaptured during sampling events, recaptured by antennas using both active and passive integrated transponders (PIT), and total recaptures (sampling and PIT antenna combined) from the Eucha-Spavinaw stream-reservoir complex in northeastern Oklahoma and northwestern Arkansas.

Species	Tagged	Sampling Recaptures	PIT Antenna Recaptures	Total Recaptures
Spotted Sucker	3,778	121	1	122
White Sucker	61	5	16	21
Northern Hogsucker	195	10	81	91
Black Redhorse	714	19	50	69
Golden Redhorse	24	1	1	2

Table 3. Candidate model rankings based on quasi-Akaike information criterion adjusted for small sample sizes (QAIC_c) from Multistrata models estimating apparent survival, capture probabilities, and transition probabilities for sucker species in the Eucha-Spavinaw stream-reservoir complex in northeastern Oklahoma and northwestern Arkansas. Apparent survival (Φ) and capture probability (p) were allowed remain constant or vary by species, strata, season, or all three (i.e., species, strata, and season [Full]). Transition probabilities (ψ) were allowed to remain constant or vary by season or by each species. Also given for each candidate model are the number of estimated parameters, the difference between the top candidate model (Δ QAIC_c), the relative likelihood of each model (ω QAIC_c) when compared to the other models (closer to 1 is better), and the evidence ratio (ER). Inestimable values are denoted (-). Refer to Figure 1 for a description of the strata.

Apparent Survival	Capture Probability	Transition Probability	Estimated Parameters	QAIC _c	Δ QAIC _c	ω QAIC _c	ER
Constant	Strata	Constant	10	161.68	0.00	0.79	1
Constant	Strata	Season	10	167.96	6.29	0.03	23.19
Season	Strata	Season	11	168.30	6.62	0.03	27.37
Constant	Season	Constant	9	168.91	7.23	0.02	37.15
Constant	Season	Season	9	169.00	7.32	0.02	38.89
Constant	Constant	Constant	8	169.68	8.00	0.01	54.57
Season	Strata	Constant	11	169.72	8.04	0.01	55.71
Constant	Constant	Season	8	169.77	8.09	0.01	57.12
Strata	Strata	Constant	12	169.85	8.18	0.01	59.68
Season	Season	Constant	10	170.92	9.24	0.01	101.46
Season	Season	Season	10	171.01	9.33	0.01	106.21
Strata	Strata	Season	12	171.03	9.36	0.01	107.52
Season	Constant	Constant	9	171.60	9.92	0.01	142.79
Season	Constant	Season	9	171.69	10.01	0.01	149.48
Strata	Season	Constant	11	172.67	10.99	<0.01	243.91
Strata	Season	Season	11	172.76	11.09	<0.01	255.34
Strata	Constant	Constant	10	173.28	11.60	<0.01	330.52
Strata	Constant	Season	10	173.37	11.69	<0.01	346.00
Species	Strata	Season	14	175.35	13.67	<0.01	930.20
Constant	Species	Constant	12	175.42	13.74	<0.01	964.50
Constant	Species	Season	12	175.45	13.78	<0.01	981.59
Species	Species	Constant	14	176.73	15.05	<0.01	1855.87
Species	Species	Constant	13	176.82	15.15	<0.01	1944.33
Species	Species	Season	13	176.91	15.24	<0.01	2035.23

Table 3 continued.

Season	Season	Constant	13	177.20	15.53	<0.01	2354.03
Season	Season	Season	13	177.30	15.62	<0.01	2464.30
Species	Species	Constant	12	177.48	15.80	<0.01	2700.19
Species	Species	Season	12	177.57	15.89	<0.01	2826.68
Strata	Strata	Constant	14	178.45	16.77	<0.01	4389.91
Strata	Strata	Season	14	178.54	16.87	<0.01	4595.54
Species	Species	Constant	16	183.35	21.68	<0.01	50966.65
Species	Species	Season	16	183.51	21.83	<0.01	55038.58
Constant	Strata	Species	34	208.16	46.48	<0.01	>6,000.00
Season	Strata	Species	35	209.53	47.86	<0.01	>6,000.00
Strata	Strata	Species	36	211.34	49.66	<0.01	>6,000.00
Constant	Season	Species	33	215.73	54.05	<0.01	>6,000.00
Species	Strata	Species	38	216.20	54.52	<0.01	>6,000.00
Constant	Constant	Species	32	216.47	54.80	<0.01	>6,000.00
Season	Season	Species	34	217.79	56.12	<0.01	>6,000.00
Season	Constant	Species	33	218.42	56.74	<0.01	>6,000.00
Strata	Season	Species	35	219.50	57.82	<0.01	>6,000.00
Strata	Constant	Species	34	220.10	58.42	<0.01	>6,000.00
Full	Full	Full	46	220.38	58.70	<0.01	>6,000.00
Constant	Species	Species	36	222.29	60.61	<0.01	>6,000.00
Species	Season	Species	37	223.77	62.09	<0.01	>6,000.00
Season	Species	Species	37	224.11	62.44	<0.01	>6,000.00
Species	Constant	Species	36	224.37	62.69	<0.01	>6,000.00
Strata	Species	Species	38	225.38	63.71	<0.01	>6,000.00
Species	Species	Species	40	230.39	68.72	<0.01	-

Table 4. Comparison of total lengths (mm) obtained from each sucker species **harvested** at giggering tournaments on Spavinaw Creek, Oklahoma from 2017-2022 (no tournament was held in 2020). Significant differences were determined via Kolmogorov–Smirnov tests (D and p -value below) and effect sizes were estimated via distributional overlap tests ($\hat{\eta}_{mean}$ and $\hat{\eta}_{range}$). Because multiple independent significant tests were conducted, α was Bonferroni-adjusted to 0.005. Significant test statistics were bolded for interpretation and NA indicates values that were unquantifiable due to sample size.

Species	Years compared	D	p -value	$\hat{\eta}_{mean}$	$\hat{\eta}_{range}$
Spotted Sucker	2017 vs 2018	0.66	<0.005	0.23	0.11-0.38
Spotted Sucker	2017 vs 2019	0.28	<0.005	0.53	0.33-0.78
Spotted Sucker	2017 vs 2021	0.47	<0.005	0.39	0.27-0.51
Spotted Sucker	2017 vs 2022	0.88	<0.005	0.10	0.01-0.27
Spotted Sucker	2018 vs 2019	0.45	<0.005	0.41	0.23-0.66
Spotted Sucker	2018 vs 2021	0.39	<0.005	0.51	0.32-0.70
Spotted Sucker	2018 vs 2022	0.58	<0.005	0.36	0.08-0.77
Spotted Sucker	2019 vs 2021	0.23	<0.005	0.62	0.41-0.86
Spotted Sucker	2019 vs 2022	0.72	<0.005	0.19	0.03-0.50
Spotted Sucker	2021 vs 2022	0.75	<0.005	0.20	0.03-0.52
White Sucker	2017 vs 2018	1.00 ^a	0.25 ^a	NA	NA
White Sucker	2017 vs 2019	0.67	0.25	0.23	0.00-0.62
White Sucker	2017 vs 2021	0.50	0.42	0.26	0.00-0.60
White Sucker	2017 vs 2022	0.56	0.05	0.36	0.01-0.78
White Sucker	2018 vs 2019	1.00 ^a	0.50 ^a	NA	NA
White Sucker	2018 vs 2021	0.50 ^a	1.00 ^a	NA	NA
White Sucker	2018 vs 2022	0.96 ^a	0.14 ^a	NA	NA
White Sucker	2019 vs 2021	0.5	0.66	0.25	0.00-0.90
White Sucker	2019 vs 2022	0.44	0.56	0.42	0.03-0.80
White Sucker	2021 vs 2022	0.46	0.33	0.35	0.00-0.88

^aOnly one White Sucker was giggered at the 2018 tournament.

Table 4 continued.

Northern Hogsucker	2017 vs 2018	0.64	0.14	0.35	0.00-0.83
Northern Hogsucker	2017 vs 2019	0.4	0.75	0.41	0.00-0.93
Northern Hogsucker	2017 vs 2021	0.25	>0.99	0.46	0.00-0.97
Northern Hogsucker	2017 vs 2022	0.26	0.93	0.45	0.03-0.90
Northern Hogsucker	2018 vs 2019	0.44	0.42	0.46	0.00-0.94
Northern Hogsucker	2018 vs 2021	0.51	0.13	0.40	0.00-0.92
Northern Hogsucker	2018 vs 2022	0.48	0.04	0.46	0.06-0.86
Northern Hogsucker	2019 vs 2021	0.35	0.77	0.5	0.04-0.94
Northern Hogsucker	2019 vs 2022	0.36	0.60	0.47	0.06-0.84
Northern Hogsucker	2021 vs 2022	0.19	0.96	0.53	0.05-0.90
Redhorse spp.	2017 vs 2018	0.82	0.04	0.01	0.00-0.48
Redhorse spp.	2017 vs 2019	0.93	<0.005	0.41	0.00-0.93
Redhorse spp.	2017 vs 2021	0.79	0.01	0.19	0.03-0.38
Redhorse spp.	2017 vs 2022	0.59	0.17	0.3	0.06-0.47
Redhorse spp.	2018 vs 2019	0.47	<0.005	0.46	0.00-0.93
Redhorse spp.	2018 vs 2021	0.74	<0.005	0.12	0.02-0.36
Redhorse spp.	2018 vs 2022	0.76	<0.005	0.11	0.01-0.31
Redhorse spp.	2019 vs 2021	0.33	<0.005	0.5	0.28-0.78
Redhorse spp.	2019 vs 2022	0.62	<0.005	0.27	0.14-0.43
Redhorse spp.	2021 vs 2022	0.36	<0.005	0.48	0.26-0.68

Table 5. ANOVA results obtained from exploratory analysis of pooled species data used to determine if intercepts and slopes of length-weight regressions are significantly different between all sucker species sampled in the Eucha-Spavinaw stream-reservoir complex in northeastern Oklahoma and northwestern Arkansas.

Variable	df	Sum of Square Error	Mean Square Error	<i>F</i> -statistic	p-value
Log ₁₀ (Length)	1	3315.2	3315.2	927569.4	<0.01
Species	5	54.4	10.9	3044.7	<0.01
Log ₁₀ (Length) × Species	4	0.2	>0.0	>0.0	<0.01

Table 6. Cormack-Jolly-Seber capture-recapture population modeling results for sucker populations in the Eucha-Spavinaw stream-reservoir complex in northeastern Oklahoma and northwestern Arkansas. Candidate models are ranked based on deviance information criterion (DIC) for each model with a measure of difference between the top candidate model included (Δ DIC). Parameters indicates the number of parameters included in the model. Φ estimates apparent survival and p estimates capture probability. A period next to the parameter indicates it was a constant whereas a subscript “species” or “sample” indicates it varied by species or time (sampling event), respectively, in the model.

Model	Parameters	DIC	Δ DIC
$\phi.$, p_{sample}	11	1191.64	0.00
ϕ_{species} , p_{sample}	15	1195.16	3.52
$\phi.$, p_{species}	6	1218.47	26.84
ϕ_{sample} , p_{species}	15	1218.48	26.85
ϕ_{sample} , p_{sample}	20	1218.65	27.01
ϕ_{species} , p_{species}	10	1222.50	30.87
ϕ_{sample} , $p.$	11	1234.39	42.75
ϕ_{species} , $p.$	6	1235.10	43.46
$\phi.$, $p.$	2	1235.57	43.94

Table 7. Means and 95% confidence intervals (CI) for instantaneous (Z) and annualized (A) mortality of sucker species from the Eucha-Spavinaw stream-reservoir complex in northeastern Oklahoma and northwestern Arkansas, estimated from a weighted catch curve analysis for Spotted Sucker, Black Redhorse and using the Chapman-Robson method for White Sucker. We have included 95% confidence intervals for all estimates.

Species	Mean (Z)	95% CI	Mean (A)	95% CI
Northern Hogsucker	0.18	0.00-0.56	17	0-43
Spotted Sucker	0.12	0.09-0.15	11	9-14
White Sucker	0.31	0.21-0.42	28	14-41
Black Redhorse	0.22	0.11-0.33	19	10-28

Table 8. Minimum estimated population sizes (estimated population) attained for sucker species in the Eucha-Spavinaw stream-reservoir complex in northeastern Oklahoma and northwestern Arkansas under different probability levels [$\Pr(\text{Population} \geq \text{Estimated Population})$] and prior constraints (Uniform, Jeffery's) from nil-recapture models. Estimated population sizes are interpreted as the minimum population size expected for each species with 99, 95, or 75% confidence, meaning there is a 1, 5, or 25% chance the true population size falls below these estimates, respectively.

Species	$\Pr(\text{Population} \geq \text{Estimated Population})$	Uniform	Jeffery's
Northern Hogsucker	0.99	901	935
Northern Hogsucker	0.95	1,078	1,123
Northern Hogsucker	0.75	1,421	1,490
Black Redhorse	0.99	8,539	8,778
Black Redhorse	0.95	9,973	10,277
Black Redhorse	0.75	12,637	13,075
White Sucker	0.99	115	123
White Sucker	0.95	146	157
White Sucker	0.75	212	234
Golden Redhorse	0.99	42	58
Golden Redhorse	0.95	64	100
Golden Redhorse	0.75	138	290
Spotted Sucker	0.99	66,679	67,088
Spotted Sucker	0.95	71,8329	72,297
Spotted Sucker	0.75	80,155	80,696

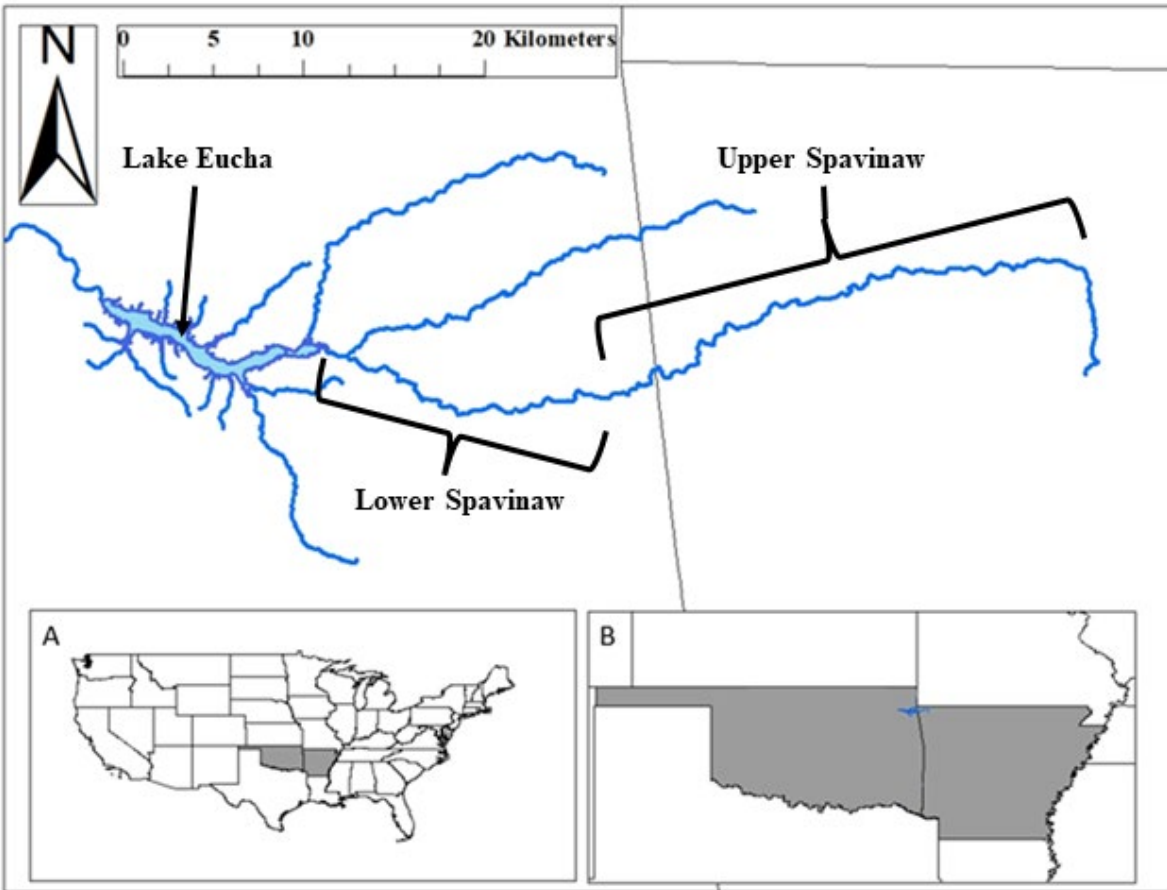


Figure 1. Lake Eucha stream-reservoir complex, including streams that were the focus of catostomid sampling efforts. Named sections denote the strata defined for the multistrata model used to estimate probability of movement between sections. Inset A shows the states which contain the Lake Eucha stream-reservoir complex in reference to the contiguous United States. Inset B shows the location of the of the Lake Eucha stream-reservoir complex within Oklahoma and Arkansas.

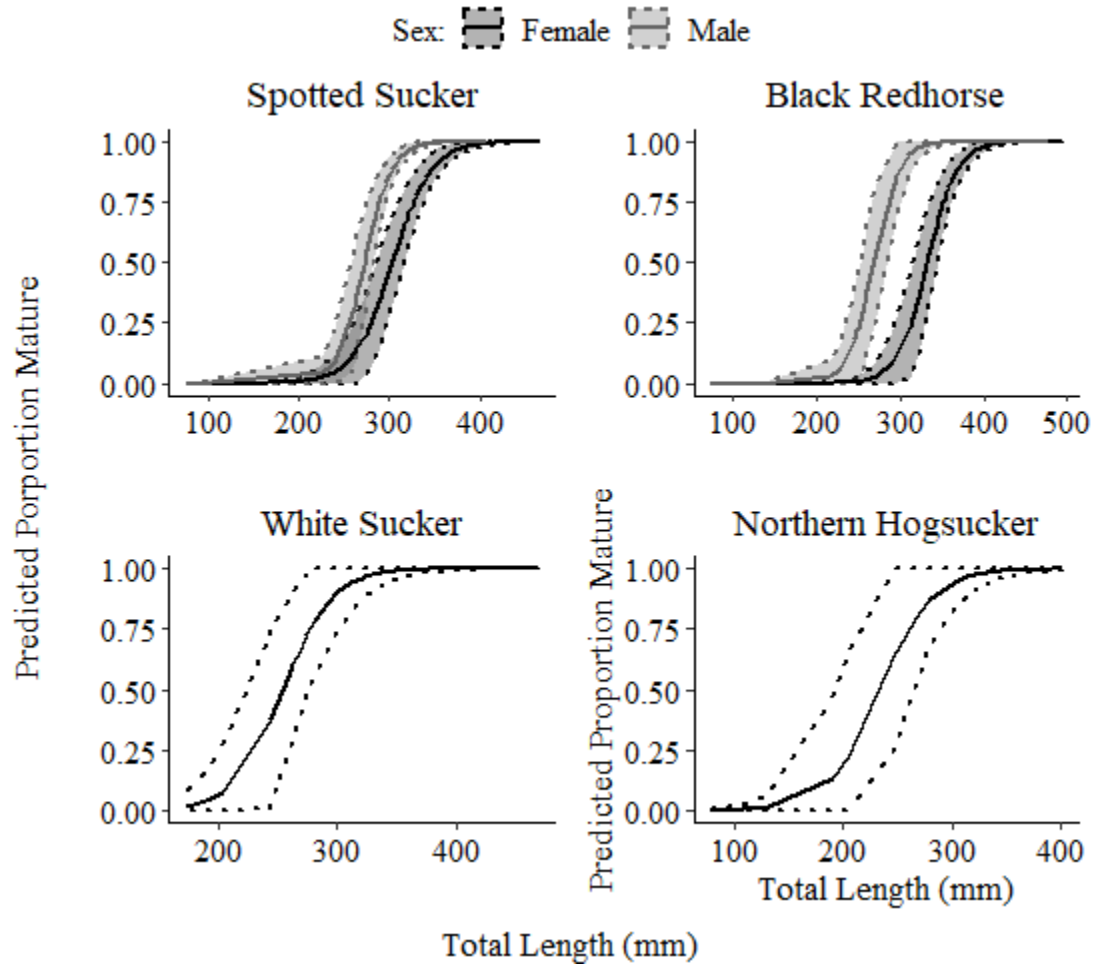


Figure 2. Predicted maturity curves for Spotted Sucker, Black Redhorse, White Sucker, and Northern Hogsucker in the Eucha-Spavinaw stream-reservoir complex in northeastern Oklahoma and northwestern Arkansas. Maturity curves were fit separately for male and female Spotted Sucker and Black Redhorse as a significant difference between sexes was observed for these species. Dotted lines indicate the 95% confidence interval. We were unable to fit maturity curves for Golden Redhorse due to small sample size.

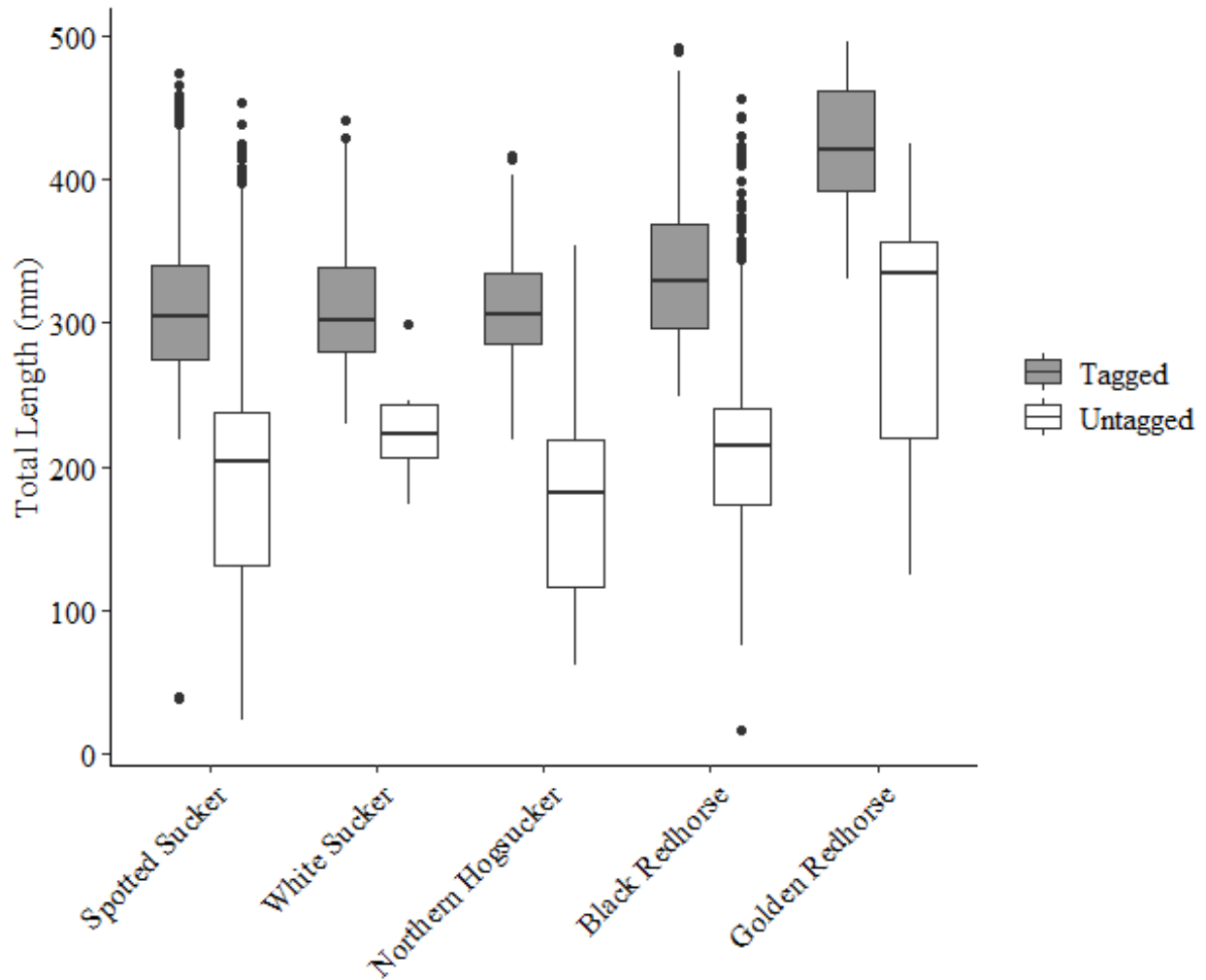


Figure 3. Boxplots showing the size distribution of tagged and untagged catostomid species sampled from 2018-2021 in the Eucha-Spavinaw stream-reservoir complex in northeastern Oklahoma and northwestern Arkansas. Horizontal line indicates the median, the box indicates the interquartile range, vertical line indicates $1.5 \times$ the interquartile range, and dots indicate suspected outliers. The minimum tagging size for this study was 250 mm (to prevent high mortality of small fish), which accounts for the difference in size of tagged and untagged individuals.

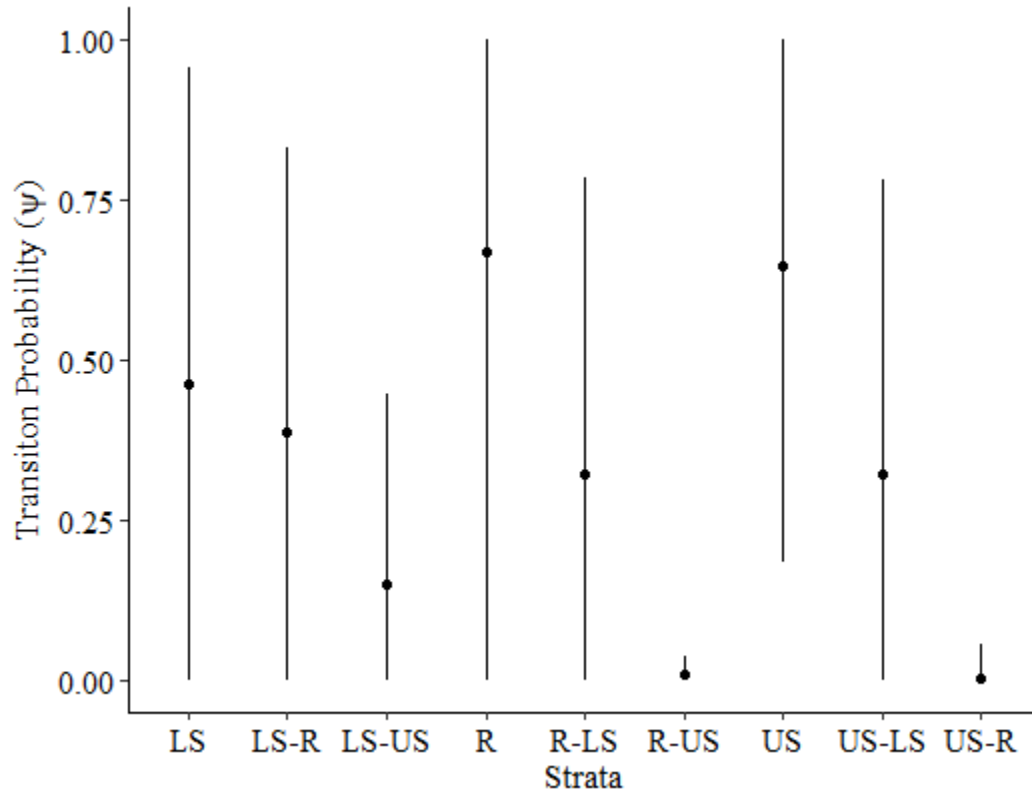


Figure 4. Mean transition probabilities (dots) and associated 95% credible intervals (vertical lines) estimated from tagged suckers (n tagged = 4,772), in the Spavinaw-Eucha river-reservoir complex. Transition probabilities represent the probability of each tagged fish within either upper Spaviaw (US), lower Spavinaw (LS), or the reservoir (R) remaining (denoted by single state [e.g., LS]) or transitioning to an alternative state (denoted as from state-to-state [e.g., LS-R is lower Spavinaw to reservoir]) between any sampling period within our study.

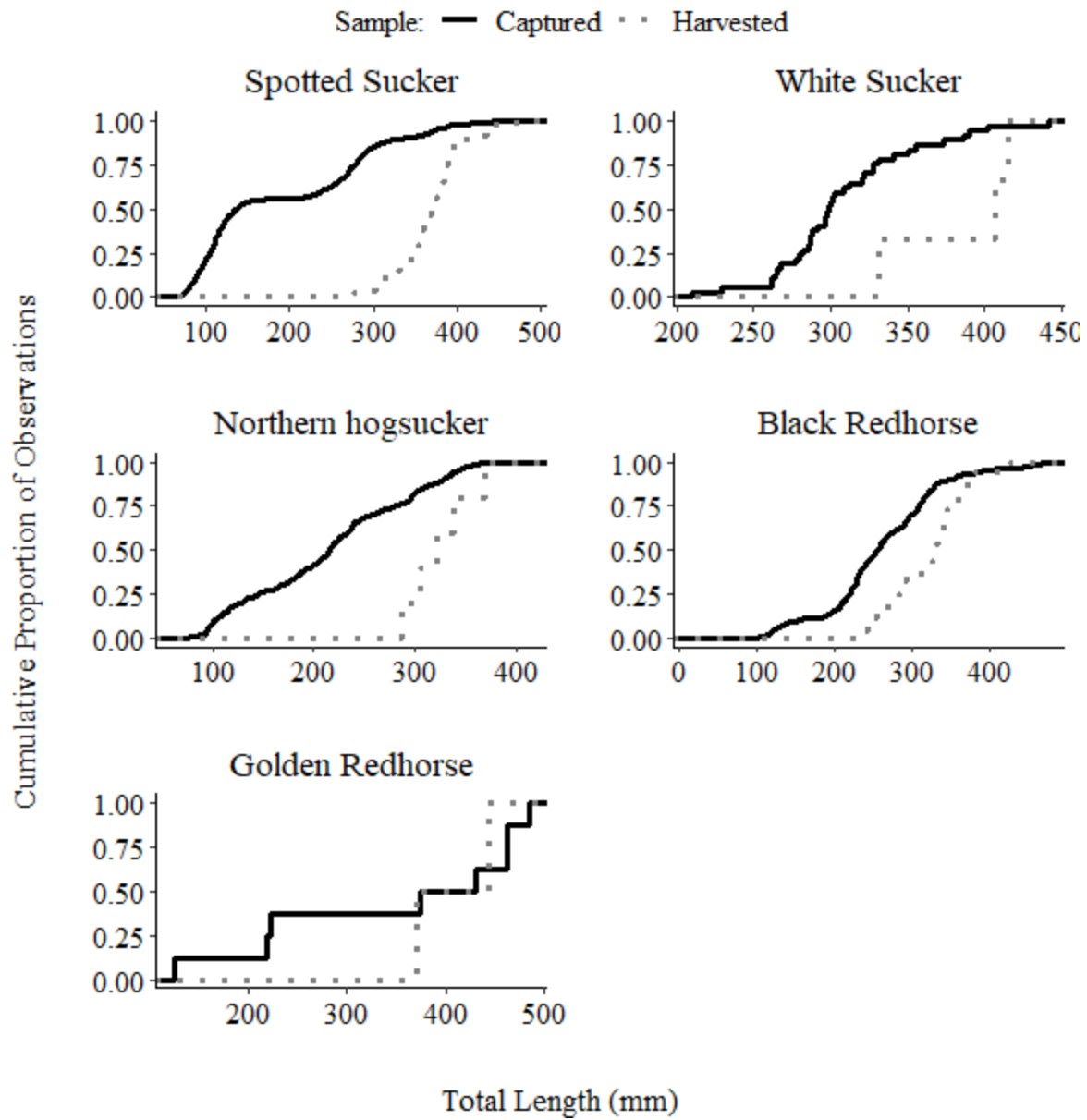


Figure 5. Cumulative length distributions of sucker species sampled prior to the 2019 gigging tournaments (Captured) and fish gigged during the 2019 gigging tournaments (Harvested) for each species of catostomid present within the Eucha-Spavinaw stream-reservoir complex, Oklahoma.

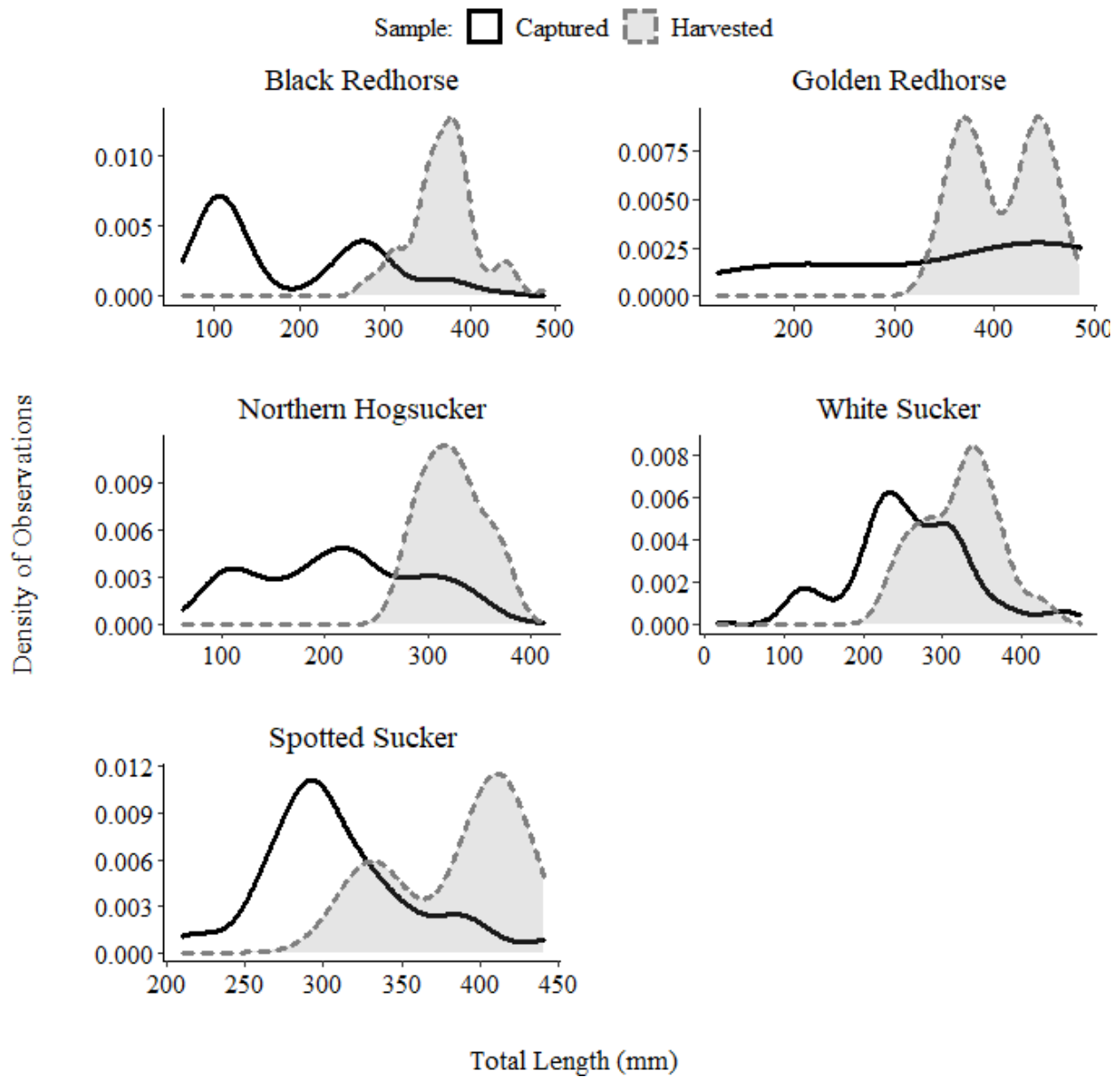


Figure 6. Density of total length observations for sucker species sampled prior to the 2019 gigning tournaments (Captured) and fish gigned during the 2019 gigning tournaments (Harvested) for each speecis of catostomid present within the Eucha-Spavinaw stream-reservoir complex, Oklahoma.

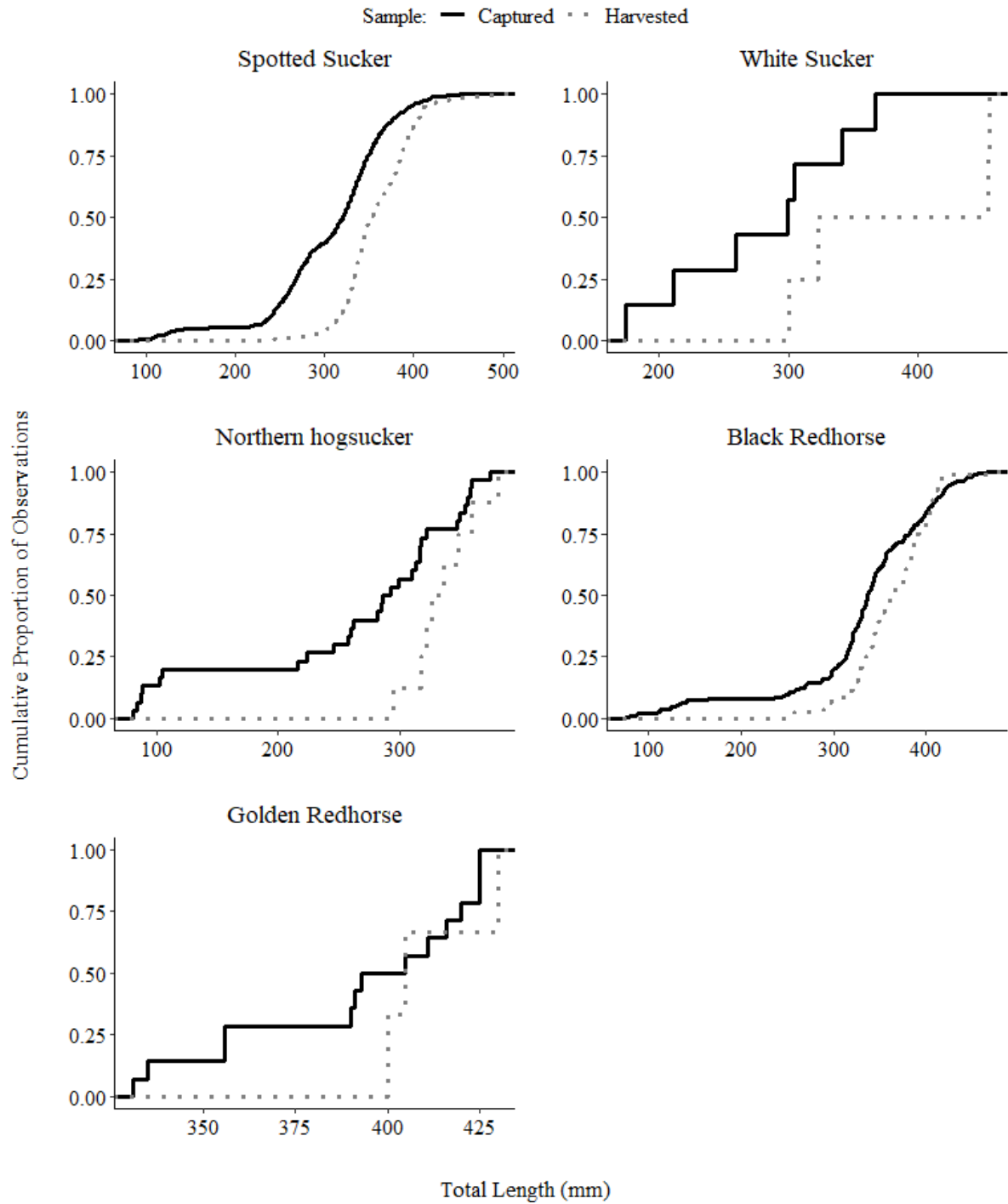


Figure 7. Cumulative size distributions of sucker species sampled prior to the 2021 gigging tournaments (Captured) and fish gigged during the 2021 gigging tournaments (Harvested) for each species of catostomid present within the Eucha-Spavinaw stream-reservoir complex, Oklahoma.

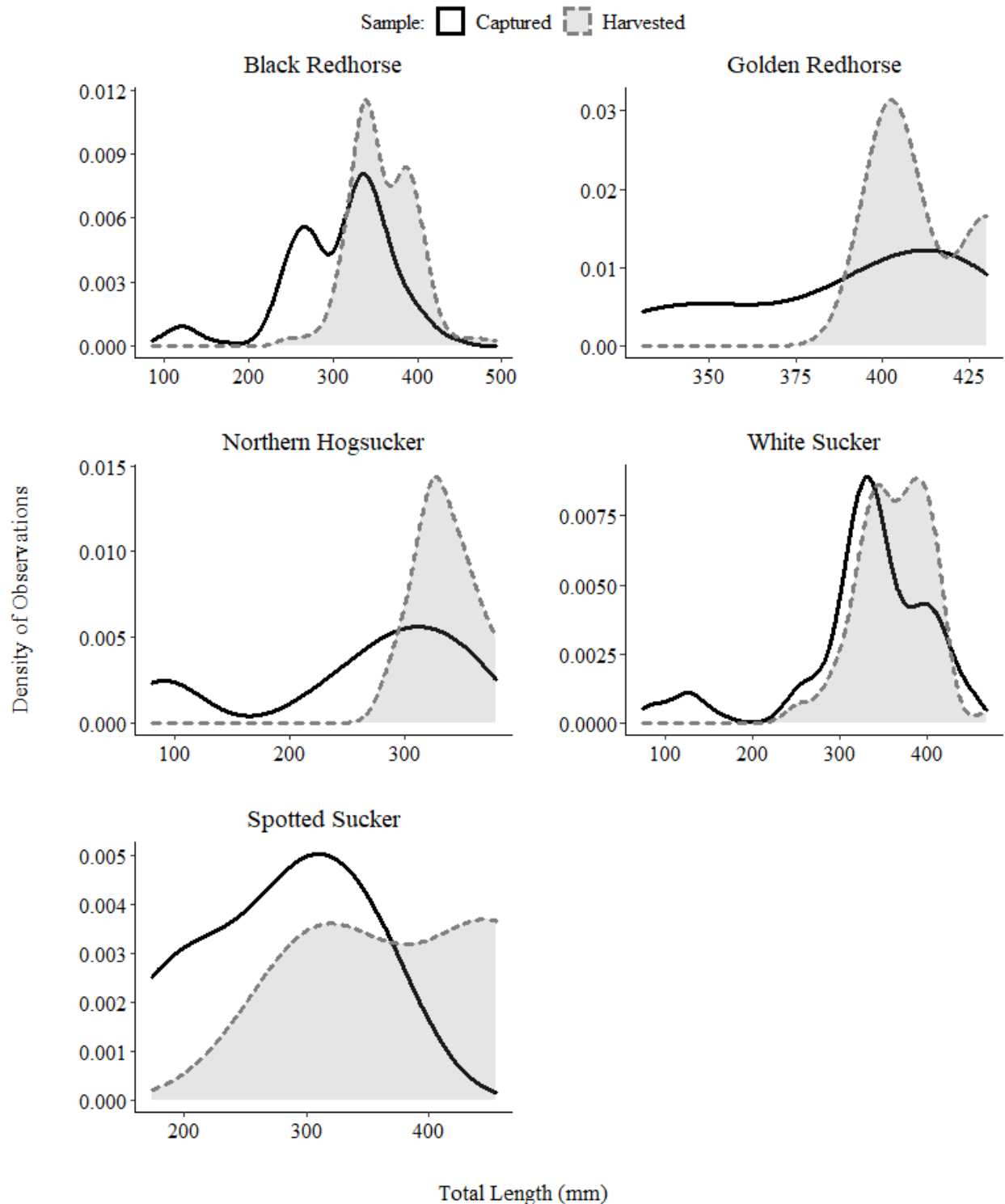


Figure 8. Density of total length observations for sucker species sampled prior to the 2021 gigging tournaments (Captured) and fish gigged during the 2021 gigging tournaments (Harvested) for each species of catostomid present within the Eucha-Spavinaw stream-reservoir complex, Oklahoma.

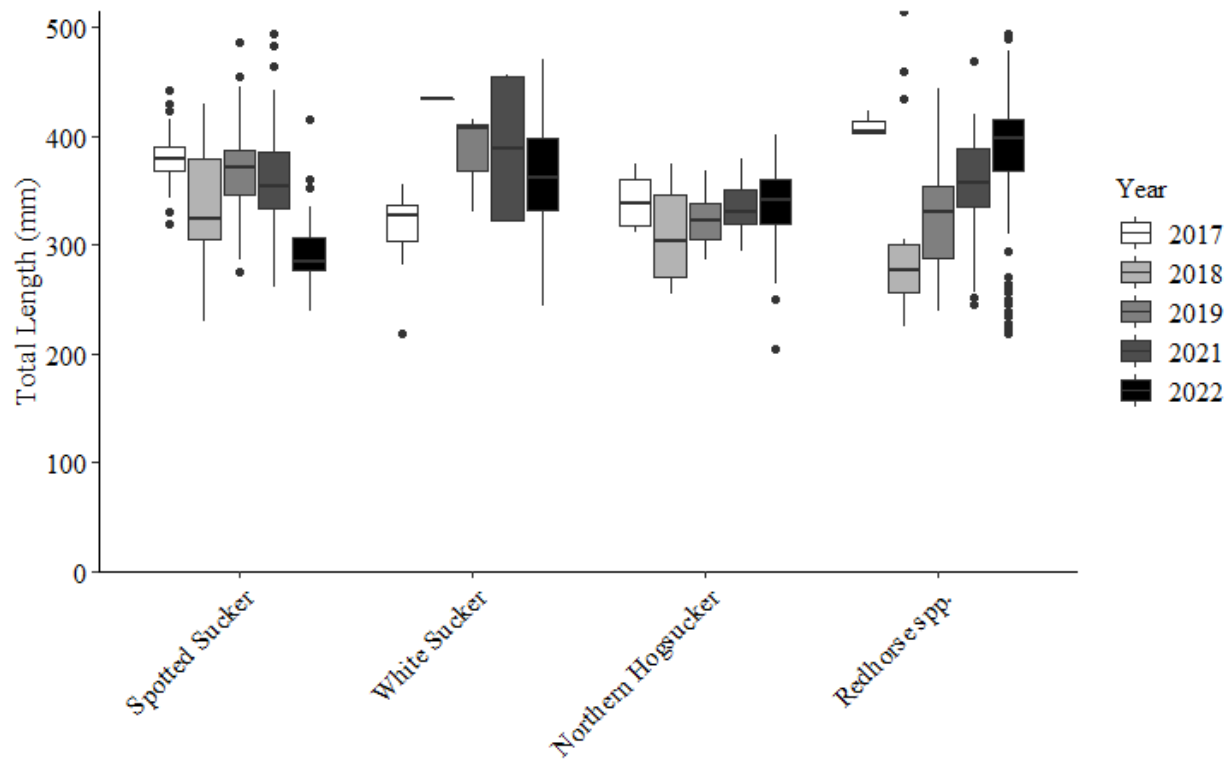


Figure 9. Boxplots of fish lengths for catostomid species harvested from gigging tournaments held in 2017, 2018, 2019, 2021 and 2022 in the Eucha-Spavinaw stream-reservoir complex in northeastern Oklahoma and northwestern Arkansas. Horizontal line indicates the median, the box indicates the interquartile range, vertical line indicates $1.5 \times$ the interquartile range, and dots indicate suspected outliers. Note: the 2020 gigging tournament was cancelled due to the COVID-19 pandemic.

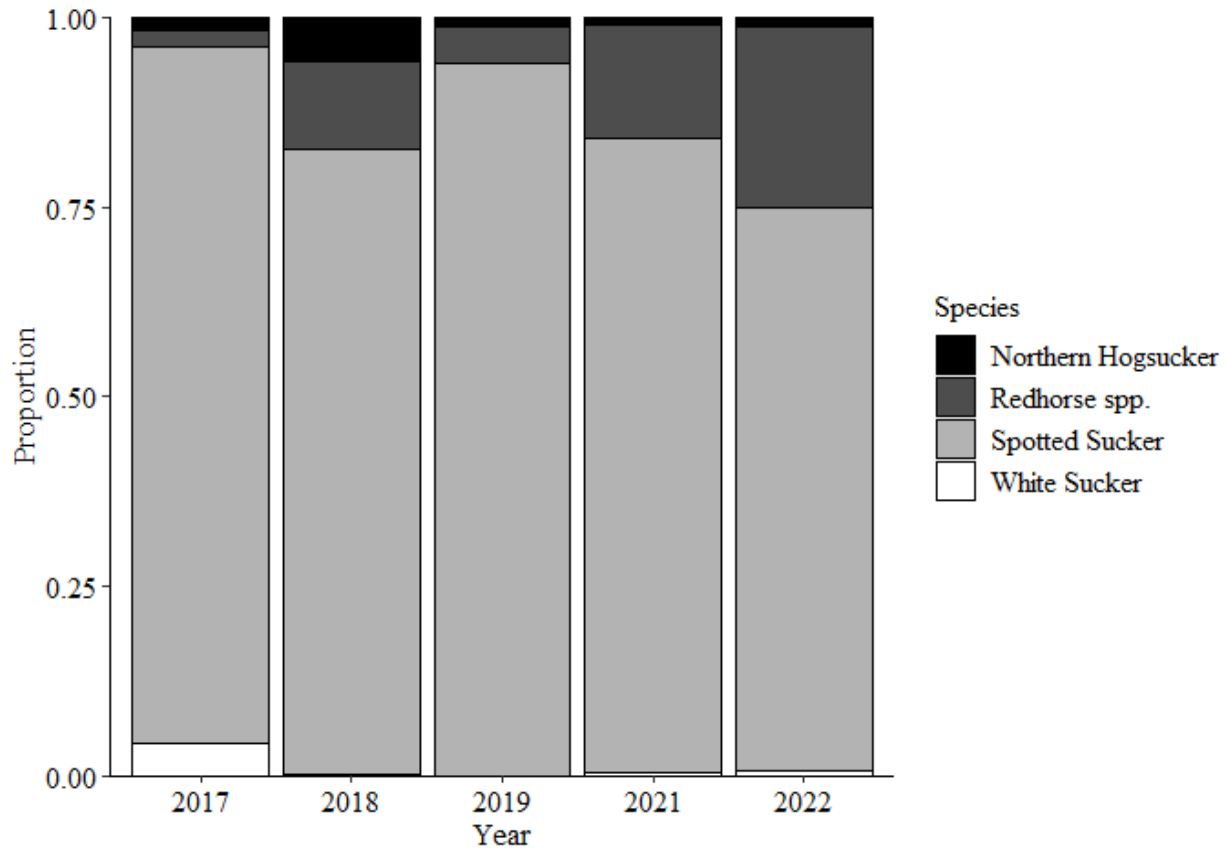


Figure 10. Proportion of gigging harvest comprised of each species from tournaments held in 2017, 2018, 2019, 2021 and 2022 in the Eucha-Spavinaw stream-reservoir complex in northeastern Oklahoma and northwestern Arkansas. Note: the 2020 gigging tournament was cancelled due to the COVID-19 pandemic.

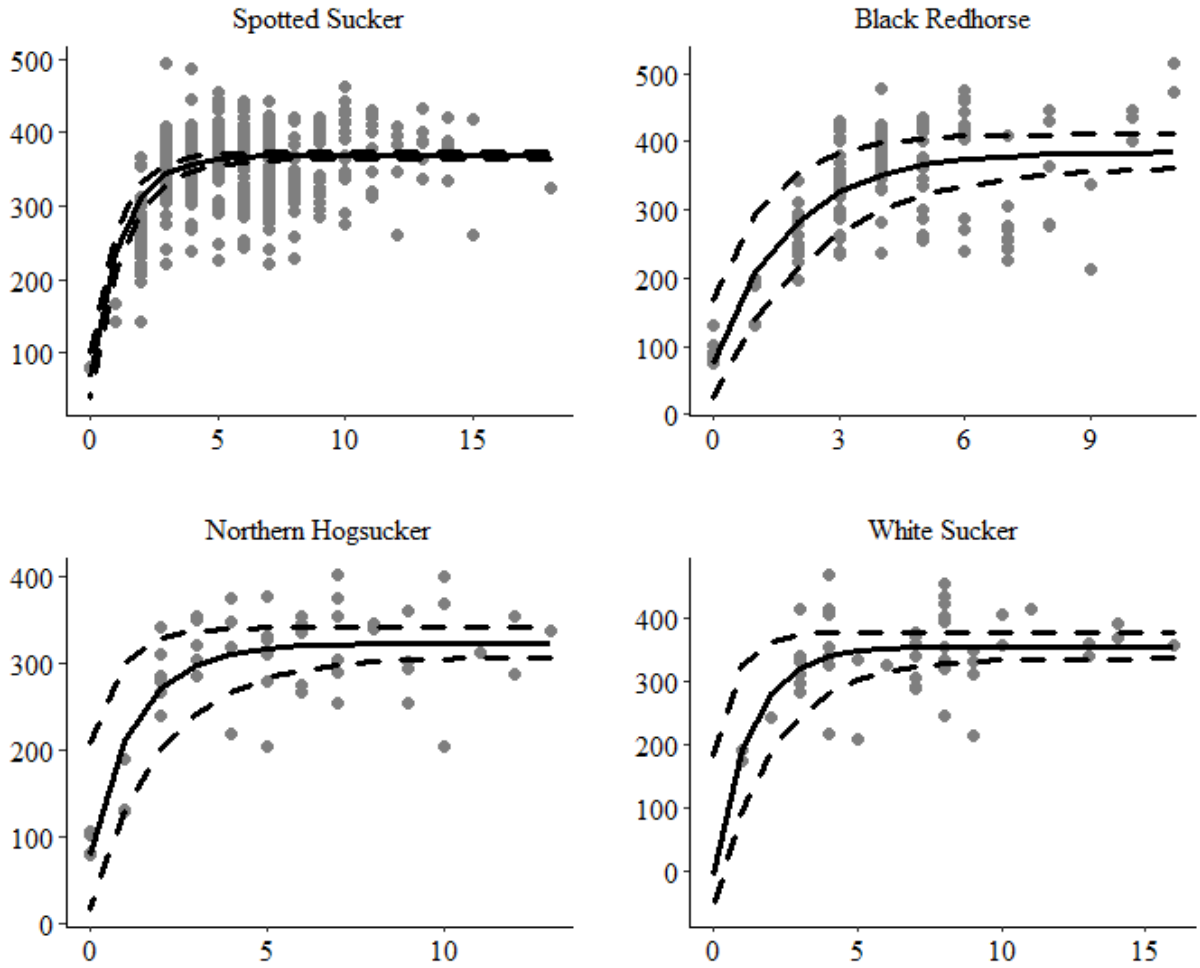


Figure 11. Age estimates obtained from fin rays and otoliths for Spotted Sucker, White Sucker, Black Redhorse, and Northern Hogsucker collected from the Eucha-Spavinaw stream-reservoir complex in Oklahoma from 2017-2022 (grey circles). Included are mean von Bertalanffy growth estimates (solid line) and associated 95% confidence intervals (dashed lines) for each species. We were unable to estimate a growth curve for Golden Redhorse due to low sample size of this rare species.

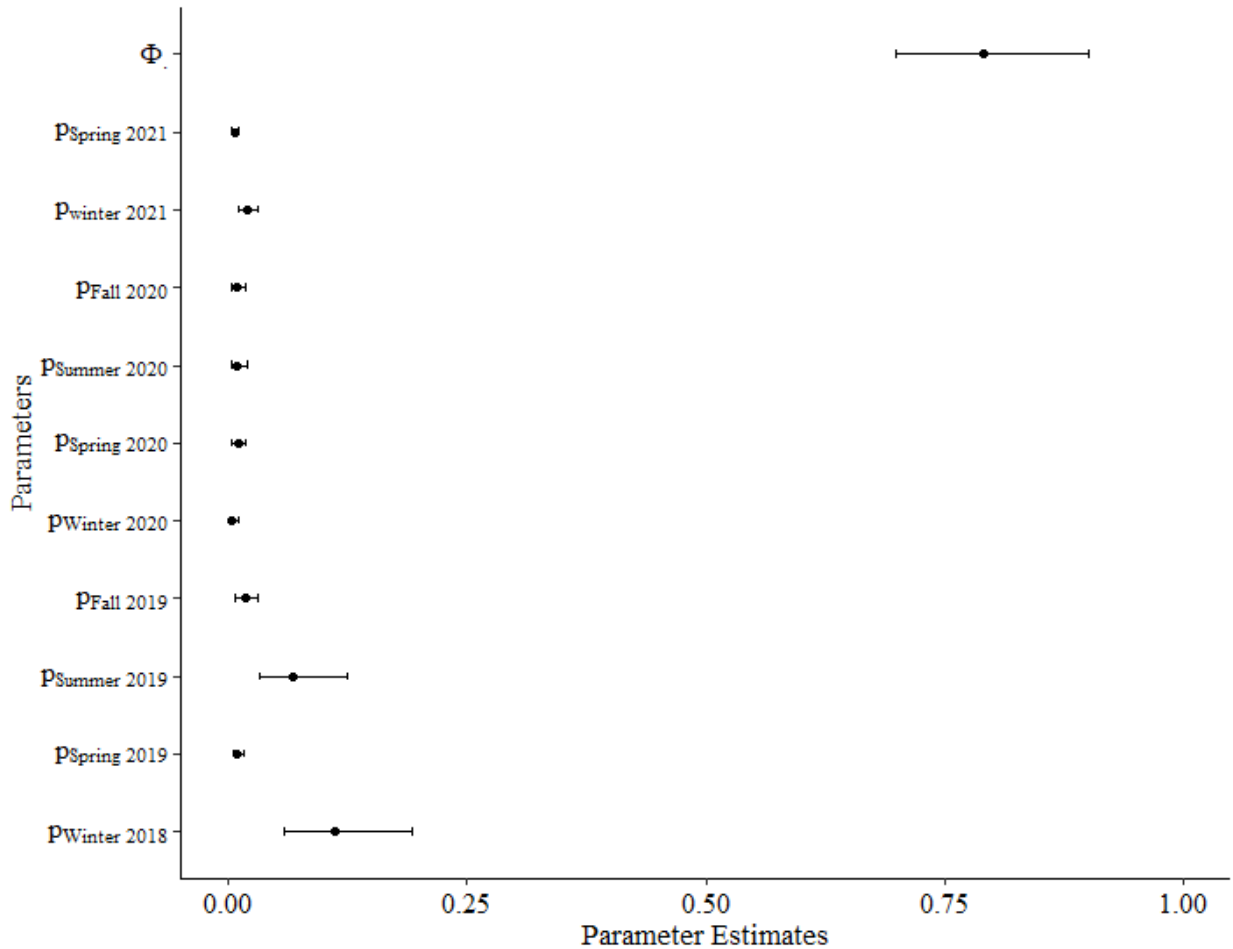


Figure 12. Apparent survival (ϕ) and detection (p) probabilities from the top candidate Cormack-Jolly-Seber models for the Eucha-Spavinaw stream-reservoir complex, Oklahoma. Apparent survival was best described as a fixed parameter across sucker species and sample events and detection as varying between sampling events. Circles represent medians and lines represent 95% credible intervals. Note: Spotted Sucker only exhibited 72.6% tag retention; therefore, apparent survival is lower than actual survival for this species due to tag loss.

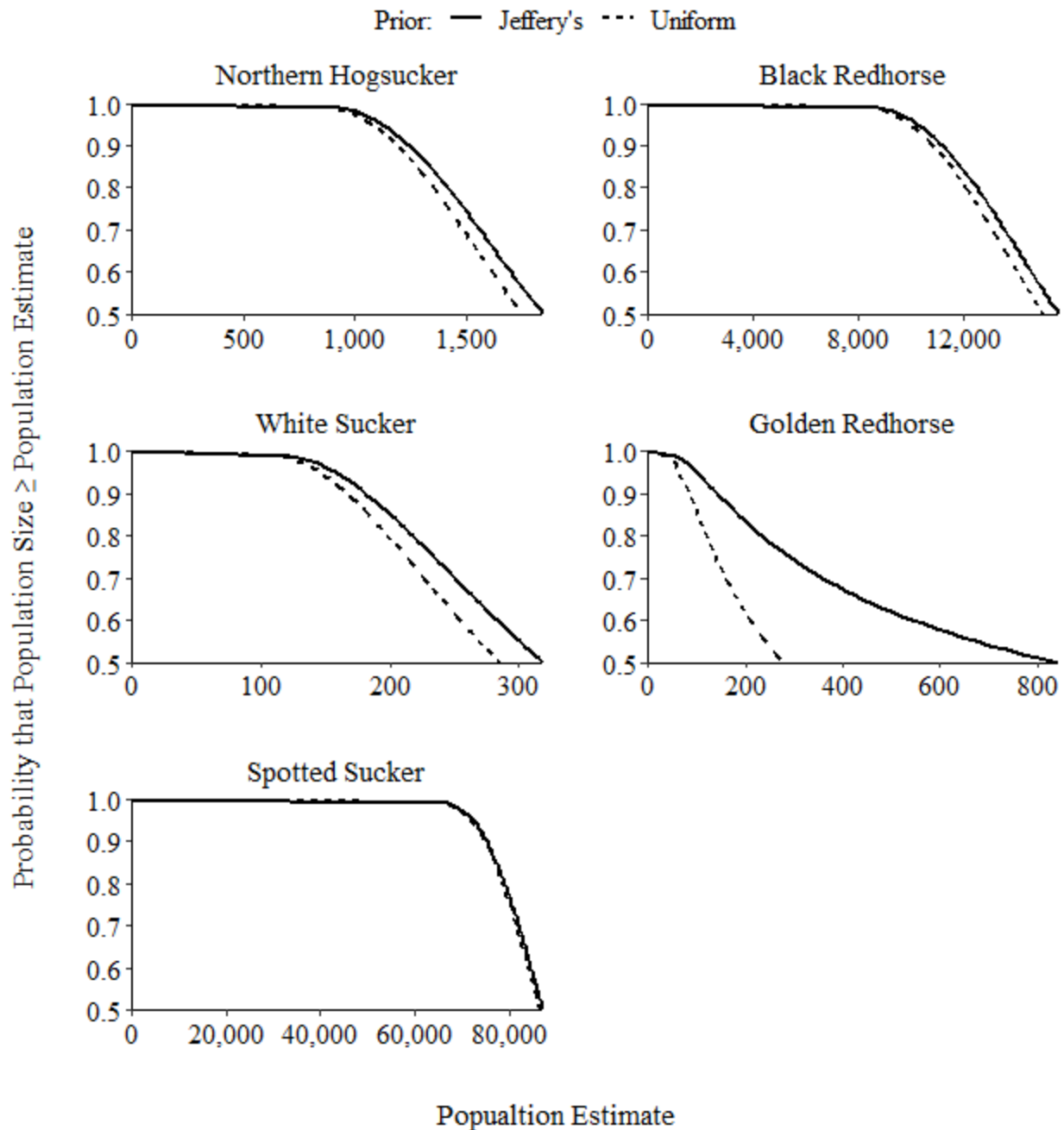


Figure 13. Minimum estimated population sizes (population estimate) attained for each sucker species sampled from the Eucha-Spavinaw stream-reservoir complex in northeastern Oklahoma and northwestern Arkansas at probability levels ranging from 1.0 to 0.5 under two different prior constraints (Jeffery's and uniform).

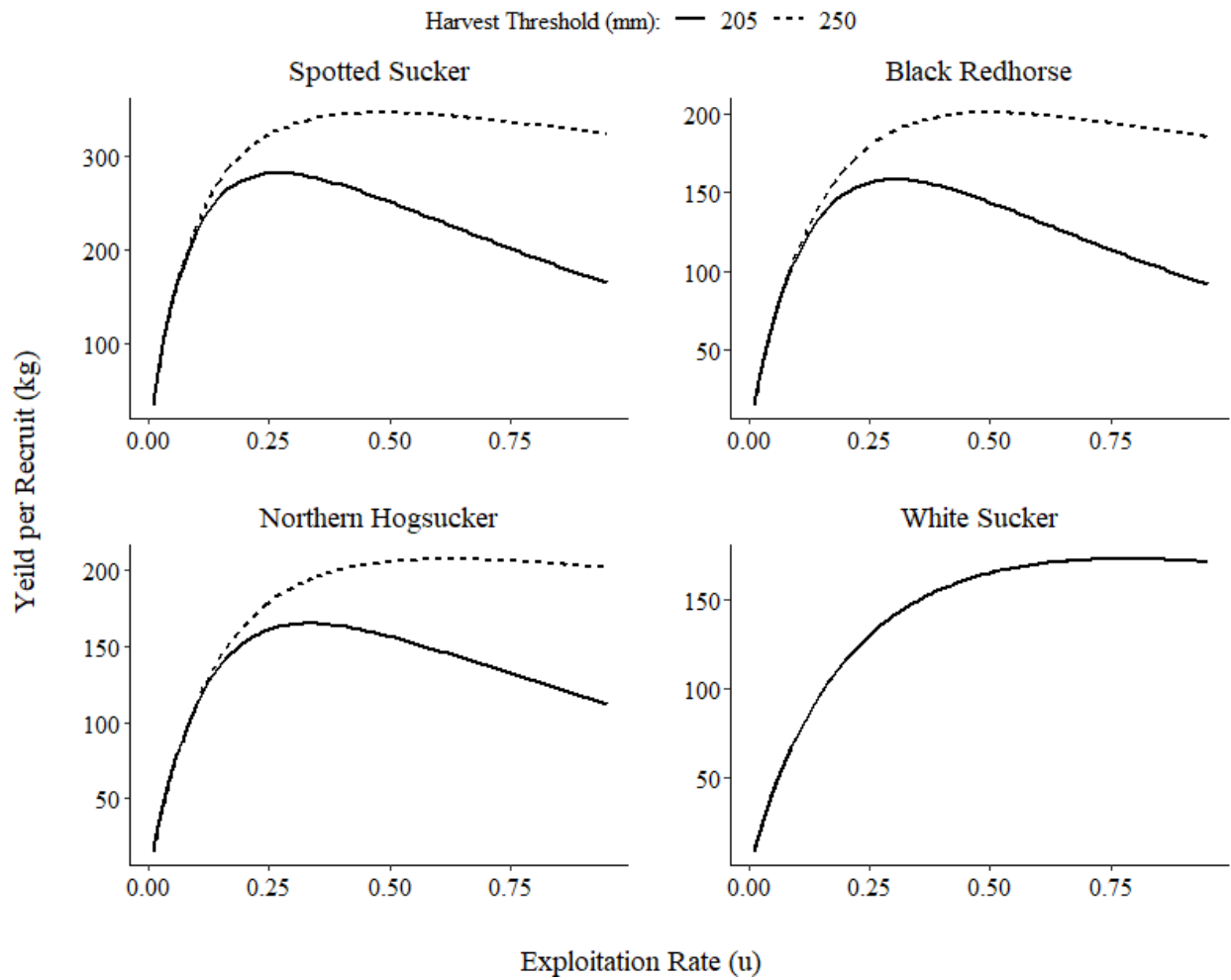


Figure 14. Yield-per-recruit of harvested suckers from Eucha-Spavinaw stream-reservoir complex, Oklahoma across different exploitation rates. Yield-per-recruit was estimated based on the minimum size of sucker sampled across gigging tournaments (205 mm) and the size at which the majority (>99%) of harvest occurred (250 mm). Note: There was not enough information to estimate YPR or mean total length of harvested Golden Redhorse at any exploitation rate.

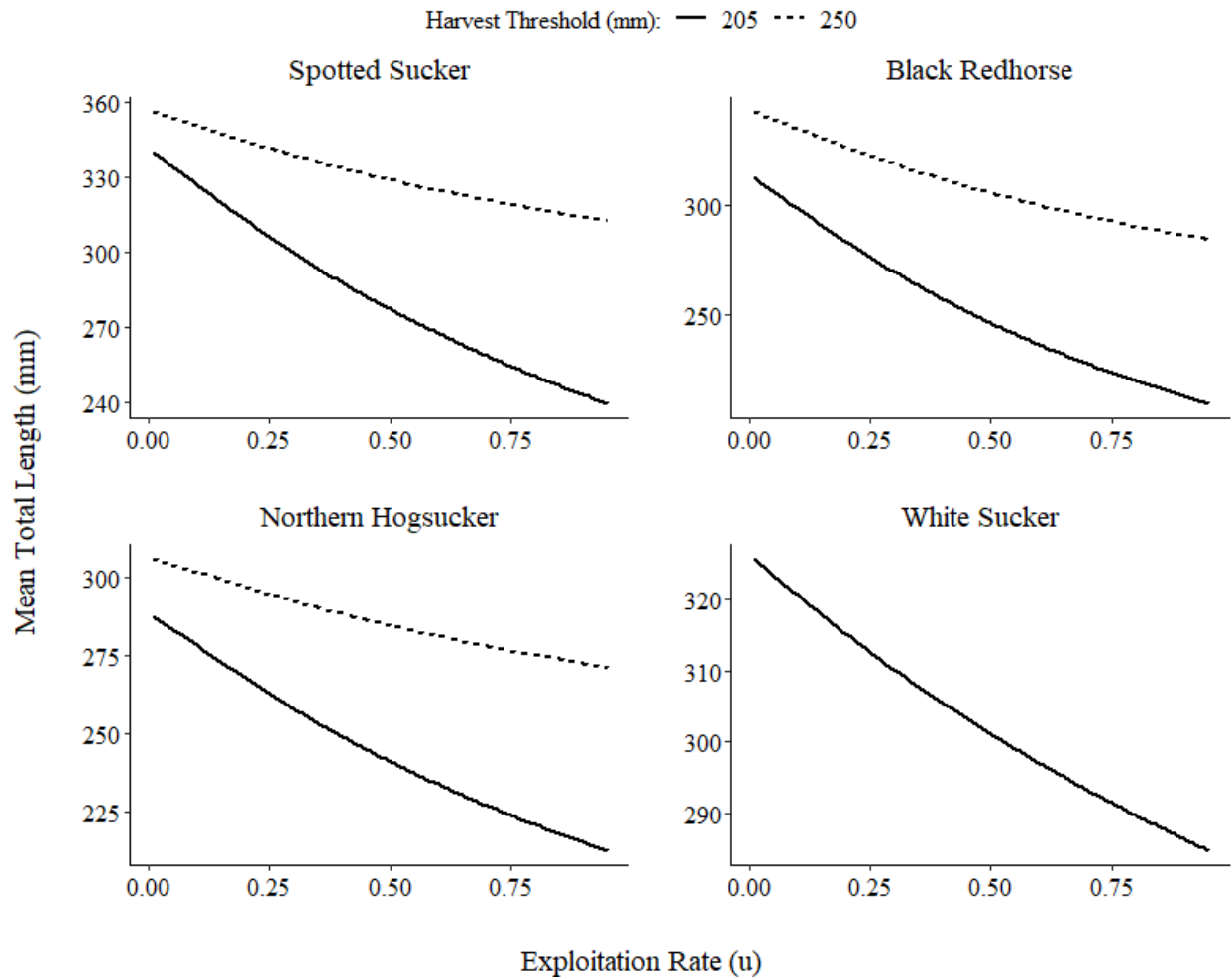


Figure 15. Mean total length (mm) of harvested suckers from Eucha-Spavinaw stream-reservoir complex, Oklahoma across different exploitation rates. Mean total length was estimated based on the minimum size of sucker sampled across gigging tournaments (205 mm) and the size at which the majority (>99%) of harvest occurred (250 mm). There was not enough information to estimate YPR or mean total length of harvested Golden Redhorse at any exploitation rate.

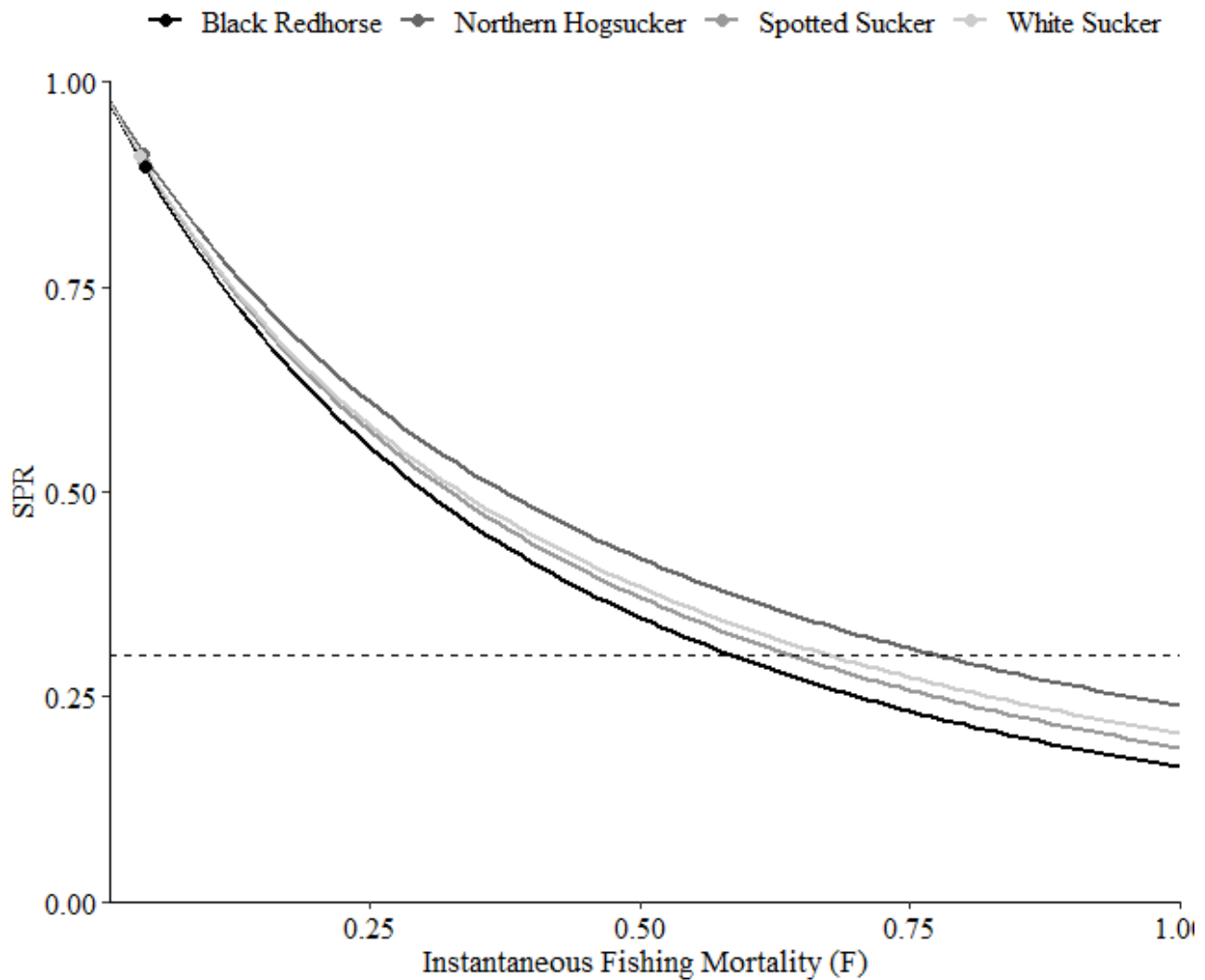


Figure 16. Estimated spawning potential ratio (SPR) at various levels of instantaneous fishing mortalities (F) for sucker species sampled in the Eucha-Spavinaw stream-reservoir complex, Oklahoma. Points represent the estimated SPR at the current level of F (estimated from gigging tournaments). Curves represent estimated SPR across levels of F ranging from 0.01 to 0.99. The dotted line indicates recruitment overfishing (defined as $SPR = 0.30$). There was not enough information to estimate SPR or F for Golden Redhorse.

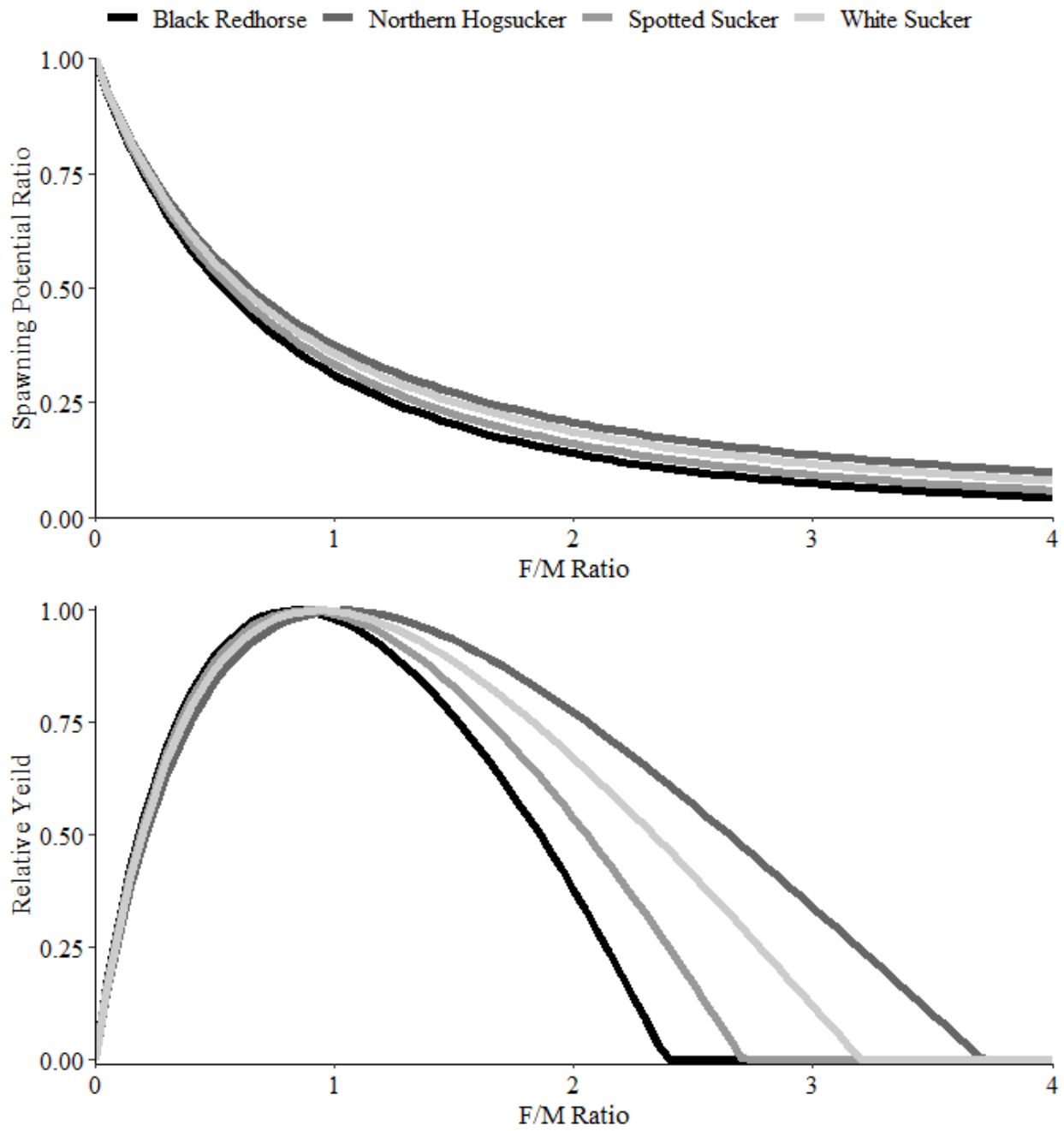


Figure 17. Estimated spawning potential ratio (SPR) and relative yields at various ratios of instantaneous fishing (F) to instantaneous natural (M) mortality for sucker species sampled in the Eucha-Spavinaw stream-reservoir complex, Oklahoma. There was not enough information to estimate SPR or F/M for Golden Redhorse.