# HABITAT SUITABILITY AND DETECTION PROBABILITY OF LONGNOSE DARTER (*PERCINA NASUTA*) IN OKLAHOMA

By

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HABITAT SUITABILITY AND DETECTION

PROBABILITY OF LONGNOSE DARTER (*PERCINA*

*NASUTA*) IN OKLAHOMA

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Abstract: North America has more than 700 species of freshwater fish that are considered imperiled and at least 27 species of North American fish have gone extinct within the last century. In Oklahoma, the Longnose Darter (LND) is a state-endangered fish species that was presumed extirpated from much of its range in Oklahoma for almost 70 years. Efforts to translocate this species in the 1990s went unassessed and their current distribution in Oklahoma is largely unknown. My objectives were to create ecological niche models at two spatial scales to identify potentially suitable habitat for LND and to sample two streams thought to contain LND in an occupancy modeling framework to estimate detection probability. The program Maxent was used to estimate probability of habitat suitability throughout the historical range of LND using a presence-only approach. This model identified several streams in Oklahoma with high probability of habitat suitability that have not previously been targeted for sampling for LND. After selecting 32 sites in Lee Creek and Blackfork Creek in Oklahoma, darter species were sampled with a backpack electrofisher in the summer of 2017 and spring 2018. No LND were found during the sampling in Blackfork Creek where LND were translocated in the 1990s. In Lee Creek, Longnose Darters were still extant, but had the lowest detection probability (5-10%) among co-occurring darter species depending on season. Compared to summer, the detection probability for LND in spring was not only higher, but more LND were found and at more sites. Presumably, the increased catches and detection of LND in spring was related to spawning activity, when the darters are shallower and more concentrated. Future monitoring for this state-endangered species can take advantage of this information and should be conducted in a way that considers imperfect detection. The range-wide ecological niche model results can be used to target new streams in Oklahoma for sampling for LND and the results from Lee Creek suggest that these surveys would maximize the potential for documenting this species if sampling were conducted in the spring when detection rates are higher.

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# CHAPTER I

## MULTI-SCALE ENVIRONMENTAL NICHE MODELING TO ESTIMATE HABITAT SUITABILITY OF LONGNOSE DARTER (*PERCINA NASUTA*) IN OKLAHOMA

### Introduction

Anthropogenic influences on freshwater aquatic ecosystems have led to the decline of many freshwater fish species (Allan and Flecker 1993; Ricciardi and Rasmussen 1999). In North America alone, more than 700 species of freshwater fish are considered imperiled (Jelks et al. 2008) and at least 27 species of North American fish have gone extinct within the last century (Miller et al. 1989). Threats to freshwater biodiversity include overexploitation, anthropogenic habitat degradation and flow modification, and competition with invasive or introduced species (Dudgeon et al. 2006; Helfman 2007). Freshwater habitat degradation through flow modification is not only one of the greatest threats to aquatic biodiversity, but also one of the most widespread (Nilsson et al. 2005), sometimes leading populations to local extinction (Wilcox and Murphy 1985).

State and federal agencies are tasked with conserving rare, threatened, and endangered species (McMullin and Pert 2010) and documenting their trends in abundance and range extent are critical for this task. Robust surveys to document species presence are important because there have been several cases of species being rediscovered after many years of assumed extirpation or extinction, even in areas with highly modified environments (Hammer et al. 2015). For example, the Black Kokanee (*Oncorhynchus kawamurae*), a deepwater trout native to Japan, was recently rediscovered within its historic range after 70 years of assumed extinction (Nakabo et al. 2011). Also, the Robust Redhorse (*Moxostoma robustum*) was rediscovered within its historic range after more than 120 years without a sighting (Hendricks 1998). After its rediscovery in 1991, several unsuccessful surveys for Robust Redhorse were conducted within the historic range in an attempt to find additional individuals (Nichols 2003). A considerable amount of effort is often dedicated towards finding rare fish species and a more refined sampling approach may reduce the costs of these surveys and time spent sampling (Guisan et al. 2006).

In Oklahoma, many fish species have suffered range-loss and could become extinct, including Arkansas River Shiner (*Notropis girardi*), Neosho Madtom (*Noturus placidus*), Leopard Darter (*Percina pantherina*), and Longnose Darter (LND; *Percina nasuta*). The first three species are currently listed as federally endangered and as such, federal agencies are tasked with monitoring these species (McMullin and Pert 2010). Currently, the Longnose Darter (Bailey 1941) is one of 404 species from the Southeastern United States petitioned for listing under the Endangered Species Act (Center for Biological Conservation 2010). However, little current information on LNDs exists to assist in this determination. In Oklahoma, LNDs are designated as state-endangered, a Tier 1 species of concern (ODWC 2016), and considered “threatened” throughout its entire range (Jelks et al. 2008). One of Oklahoma’s rarest fish species (Robison 1992; Miller and Robison 2004), the LND likely suffered population declines after the completion of Wister Dam in 1949. For over 60 years attempts to capture it in the Poteau

River system (PRS) failed (Cross and Moore 1952; Lindsey et al. 1983; Wagner et al. 1985), suggesting it had been extirpated from this system. However, in 2015 a LND was collected from the Poteau River upstream of Wister Lake (OWRB 2015). This record highlights how little we know about Longnose Darters' detectability and their distribution in Oklahoma. Longnose Darters are benthic insectivores that inhabit gravel riffles and shallow pools of upland streams within the Arkansas River drainage of Arkansas, Missouri, and Oklahoma (Miller and Robison 2004). Historically, LND were only known to definitively occur in Oklahoma in two river systems, Lee Creek and the PRS (multiple occurrences in the Poteau River and 1 occurrence in Brazil Creek), and only the population in Lee Creek is known to consistently persist (Burns & McDonnell Engineering Company 1990; Gatlin and Long 2011).

A potentially viable population of translocated LND may exist in Blackfork Creek, a tributary of the Poteau River. When dam construction on Lee Creek was proposed by the city of Fort Smith, Arkansas in 1989, concern grew that LND might be extirpated from Lee Creek. Translocation efforts in 1991-92 resulted in ~164 individuals moved from Lee Creek into three locations of Blackfork Creek to create a refuge population (Burns & McDonnell Engineering Company 1989) (Figure 1). Surveys conducted prior to translocation failed to collect any LND in Blackfork Creek (Burns & McDonnell Engineering Company 1990), although this stream is within the historic range of the species (Cross and Moore 1952). Blackfork Creek has not been surveyed for LND since the translocation in 1992 and the success of these efforts is unknown.

Ecological niche models (ENM) have been applied to a wide range of species and geographic areas (Elith and Leathwick 2009). These types of models have been used to prioritize conservation areas (Urbina-Cardona and Flores-Villela 2010) and to identify new populations of endemic habitat specialist fishes (Rhoden et al. 2017). Their use has helped the search for many species of conservation concern by identifying areas with suitable habitat, but these techniques have largely been applied to plants and terrestrial animals (Fois et al. 2018). Several studies that have used ENMs on aquatic species have successfully identified previously undiscovered populations (Rhoden et al. 2017) and possible protection areas with the highest species richness (Castillo-Torres et al. 2017). Aquatic ecosystems are one of the most impacted ecosystems by climate change and other anthropogenic influences such as flow modification and habitat degradation (Dudgeon et al. 2006). Multi-scale ecological niche models could aid management agencies in their task of monitoring and conserving areas for species of conservation concern.

Ecological niche models that rely on presence-only information, such as Maxent, provide a mechanism to estimate potential locations where LND may occur, especially at sites that have either not been sampled or not sampled extensively. Maxent uses species occurrence information and environmental predictors to infer habitat suitability in areas without occurrence data (Peterson et al. 2002, 2011). At a broad, river-segment scale, large stream segments from across its historical range that have potentially suitable habitat for LND can be identified. But, such a large scale is difficult to use later to select sampling locations to search for rare species. A finer-scale model that incorporates local environmental variables is better suited to identify areas to survey within the stream segments predicted suitable with the range-wide model. Thus the objectives of this study are to aid future sampling efforts where the goal is to detect LND by using ecological niche models at two different spatial scales to (1) estimate habitat suitability in streams throughout the historical range of Longnose Darter and (2) estimate habitat suitability at the stream reach scale in Lee Creek and Blackfork Creek.

## Methods

*Study Area*

The historical range of Longnose Darter extends across three states and four EPA level III ecoregions. The Ozark Highlands is partly covered by oak-hickory forest and is underlain by karst and dolomite features (Woods et al. 2005). The geology of the Boston Mountains ecoregion is characterized Pennsylvanian-age sandstone and shale. The Arkansas Valley ecoregion is a synclinal and alluvial valley that lies between the Boston Mountains and Ouachita Mountains ecoregion; it is also characterized by Pennsylvanian-age sandstone and shale geology. The Ouachita Mountains are structurally different from the Boston Mountains and are lithologically distinct from the Ozark Highlands. This ecoregion is largely forested and is comprised of oak, hickory, and pine trees. The streams within these four ecoregions flow primarily into the Arkansas River and White River drainages.

The Poteau River flows within the Arkansas Valley ecoregion of Arkansas and Oklahoma, originating in Arkansas and flowing west into Oklahoma where it turns abruptly north towards its confluence with the Arkansas River (Figure 1). Constructed in 1949, Wister Dam impounds the Poteau River where it meets Fourche Maline Creek. Upstream from Wister Lake, the Poteau River and its tributaries are characterized by boulder-gravel substrate typical of this region (Cross and Moore 1952; Lindsey et al.

1983). Downstream of Wister Lake, the Poteau River and its tributaries transition into more turbid, lowland rivers characterized by muddy riverbeds and occasional shale outcroppings. Historically, 89 species of fish inhabited the Poteau River system (Cross and Moore 1952), but more recent surveys have shown a declining abundance of a number of fish species (Lindsey et al. 1983). A general survey of the Poteau River failed to capture LND in 1974 (Lindsey et al. 1983) and until recently LND were assumed extirpated (Wagner et al. 1985, OWRB 2015). However, because of the unassessed translocations into Blackfork Creek, a LND population may persist in this tributary of the Poteau River.

Lee Creek originates within the Boston Mountains ecoregion of Arkansas and flows into Oklahoma before turning back east towards its confluence with the Arkansas River (Figure 1). Lee Creek is designated as one of Oklahoma's six "scenic rivers" (OSRC 2016) and historically had as many as 78 fish species (Funk 1979; FERC 1987). Longnose Darters were first documented in Lee Creek in 1886 (Jordan and Gilbert 1886) and this stream hosts the last known persistent populations of Longnose Darters in Oklahoma (Wagner et al. 1985; Gatlin and Long 2011).

*Modeling*

Ecological niche models created with the program Maxent were used to identify potentially suitable habitat for LND throughout its range particularly in Oklahoma, including Lee Creek and the PRS. Maxent is a machine-learning algorithm that uses presence-only occurrence data to produce a map of locations with probability of habitat suitability (Phillips et al. 2006; Elith et al. 2011). Habitat suitability probabilities can then be divided into strata of suitability categories, which can be used to guide future sampling efforts by natural resource agencies (Guisan et al. 2006). Rare species in particular are detected imperfectly when sampling, leading to difficulties in making inferences about their populations (MacKenzie et al. 2005). Although imperfect detection can bias inferences and predictions in a presence-background approach like Maxent, the resulting niche models can be used to prioritize areas for future sampling by ranking sites in terms of habitat suitability, but not to estimate actual probabilities of site occupancy (Lahoz-Monfort et al. 2014). To model Longnose Darter habitat suitability throughout their entire range while also incorporating more local habitat characteristics, models were created at two scales in this study: a stream segment, range-wide scale and at a more local, stream reach scale. The range-wide model identified potentially suitable stream segments within the species range, as well as identified suitable areas for finer-scale modeling. I used Lee Creek and Blackfork Creek for an assessment of habitat suitability in streams where this species has been recently documented or translocated. Longnose Darters were long presumed extirpated within their historic range in the Poteau River (Cross and Moore 1952; Lindsey et al. 1983; Burns & McDonnell Engineering Company 1990), but are presumed to still occur in Lee Creek (Gatlin and Long 2011) and could likely occur in Blackfork Creek (Burns & McDonnell Engineering Company 1990). It is possible that remnant populations of Longnose Darters exist in streams that have not been previously sampled for this species; determining probability of habitat suitability throughout their range would benefit natural resource management agencies in their search for undiscovered populations.

*Stream Segment Model* — The range-wide model was completed prior to the stream reach scale modeling, and was conducted in a team setting that included myself, faculty, and graduate students. This model was based on a presence-only LND occurrence dataset from online databases (e.g., BISON and Vertnet; Table 1) and coarse-scale environmental variables at the stream segment resolution. When historical records lacked precise GPS coordinates, the program Geolocate was used to assign coordinates based upon textual locality descriptions (Rios and Bart 2010). Occurrence records were then associated with stream segments (flowlines) in the Arkansas River basin drainage (USGS hydrologic region 11) of the National Hydrography Dataset (NHD) that intersect with EPA level III ecoregions in Missouri, Arkansas, and Oklahoma, encompassing the range of LND (i.e., Ouachita Mountains, EPA level III ecoregion 36; Arkansas Valley, 37; Boston Mountains, 38; and Ozark Highlands, 39).

After joining occurrence records to the selected NHDPlus flowlines in ArcMap (ESRI 10.3.1), duplicate records were removed and a single occurrence was retained for that segment. Of the ecoregions selected, those with the majority of LND records were selected as the model training area. The remaining ecoregions represented the model testing area to evaluate fit and performance. Eleven coarse-scale hydrologic and geological habitat covariates that quantify the natural abiotic and streamflow characteristics of the study area were included in the model (Albanese et al. 2014, Taylor et al. 2018; Table 2). Briefly, depth-to-bedrock is a metric quantifying the how deep the bedrock is under the soil, soil permeability is a measure of the ease with which air and water move through the soil, and rock fragment volume is a measure of particles >2 mm in the soil. Within the NHDPlus dataset, variables were selected that represent both the climate of the region (e.g, annual precipitation and maximum elevation) and streamflow characteristics (e.g., unimpacted mean annual discharge and slope). Total drainage area was removed because it had a Pearson's correlation coefficient of $|r| > 0.7$ with unimpacted discharge (Table 3). After an initial model run, predictor variables that contributed <2% to model accuracy gain were removed to avoid model overfitting. I selected the logistic output option in the Maxent algorithm because it reports values ranging from 0.0-1.0, which can be used to rank sites according to habitat suitability (Elith et al. 2011). When producing the logistic output, Maxent assumes and sets the prevalence ($\tau$) of a species at typical presence locations at 0.5 by default. Little prior data exists for LND prevalence so this default value was used. The number of background points was increased to 100,000. The jackknife option for variable importance was selected to determine the degree to which each environmental variable contributed to predicted habitat suitability. The minimum training presence (MTP) value was applied as a threshold to the logistic output values, where the lowest suitability value associated with a presence location became the cutoff for determining suitable and non-suitable stream segments. Values greater than the MTP threshold were separated into four categories of increasing probability of habitat suitability (Table 4). All other settings were left as default in the Maxent software. The resulting model estimates environmental habitat suitability for all stream segments in the study area, many of which have not been sampled for Longnose Darters in the past.

*Stream Reach Model* — Fine-scale habitat and LND occurrence data on Lee Creek exists from previous studies (Burns & McDonnell Engineering Company 1989; Gatlin 2013). I used this data to model Lee Creek at the reach scale and projected estimated probability of habitat suitability into Blackfork Creek, where LND have been translocated. The LND occurrence records available for this scale represent sites from across a variety of habitats along the entirety of Lee Creek in Oklahoma and include general substrate composition and channel unit information. Burns & McDonnell Engineering Company (1989) specifically targeted LNDs in their sampling of Lee Creek during FERC licensing of Lee Creek Dam. Their sampling design targeted riffles and raceway areas, including habitat descriptions for each site where LNDs were captured. Gatlin (2013) surveyed channel units in a probabilistic fashion for fish communities and recorded channel unit and substrate information, including at sites where LNDs were encountered. These two surveys within Lee Creek were sampled in relation to the available channel units, reducing sampling bias often found with museum records (Phillips et al. 2009). However, these data are also biased towards sampling in shallow water. The stream reach niche model was created using stream reach polygons (riffle-to-riffle) in the samples-with-data (SWD) format in Maxent. Of the predictor variables summarizing reach channel unit and substrate type composition, % Pool, % Bedrock, and % Glide were selected because they are not highly correlated (Table 5). The logistic output and jackknife options in Maxent were used as in the range-wide model. A 10-fold model cross-validation was applied to this model and provided mean values and error estimates from the 10 model replicates. In cross-validation, samples are evenly divided into folds, and during each of the specified number runs, each fold is left out of the model once as a testing subset (Merow et al.

2013). The model was trained and tested on Lee Creek occurrence records and projected into Blackfork Creek.

The environmental variables to be used in this ENM (i.e., substrate composition, channel unit composition, and reach area) were selected based on their presumed biological significance for fluvial benthic specialist species at a local scale (e.g., Tangerine Darter *Percina aurantiaca* and Goldline Darter *Percina aurolineata*; Leftwich et al. 1997; Albanese et al. 2014). Substrate and channel unit classifications for Lee Creek from Gatlin (2013) were used in this analysis and applied to Blackfork Creek. Gatlin (2013) visually categorized channel units into four categories (i.e., riffle, run, pool, and glide; Arend 1999) during side-scan sonar surveys. To account for changes in river structure and channel unit location in the time since the surveys were conducted, adjustments were made to channel unit boundaries while viewing Lee Creek via Google Earth imagery; this procedure assumes no changes in substrate composition and no field validation took place. To define the stream reaches in Lee Creek, riffle-to-riffle channel unit polygons were merged together to create reaches (i.e., any given reach only contains one riffle). The side-scan sonar surveys from Gatlin (2013) were also used to classify substrate composition (i.e., bedrock, fine, rocky, and boulder). The four substrate types were consolidated into three categories. The rocky and boulder categories were merged into one rocky-boulder category because he reported low accuracy in differentiating these two types. I calculated the proportion of each reach comprised of each substrate type and the resulting polygons were used in the stream reach scale ENM.

*Habitat Mapping* – To map habitat throughout Blackfork Creek to match that identified in Lee Creek, I followed procedures employed by Gatlin (2013), using a Humminbird 998c SI side-scan unit, in concordance with methods outlined by Kaeser and Litts (2010) to survey selected river segments identified as potentially suitable by the range-wide niche model. Because it was not feasible to completely scan Blackfork Creek, a protocol for side-scan sonar surveys was developed to limit sonar work to suitable areas from the range-wide ENM. To represent the different suitability categories (Table 4), side scan sonar surveys were conducted in Blackfork Creek in a disproportionate stratified mapping approach (Kalton and Anderson 1986). In this approach, the number of samples (sites per strata) is not distributed by strata size, but by suitability. Probability of habitat suitability was separated into five categories of increasing suitability and the amount scanned in each suitability category depended on the length of stream segment available. A 5 km threshold was selected where segments <5 km in a suitability category were scanned in its entirety. For areas >5 km, up to 25% of the stream segments in that category were scanned (Table 6).

After the side-scan sonar surveys were conducted, the geo-referenced mosaic of side-scan images were processed in SonarTRX Pro and exported to ArcMap. I manually delineated underwater substrate types into polygon shapeflies in ArcMap 10.3.1, as outlined by Kaeser and Litts (2010), using field notes taken concurrently with the sonar surveys. Substrates were categorized into three coarse types: sand, rocky-boulder, and bedrock (Table 7). Geomorphic channel units (i.e., riffle, run, pool, and glide) were visually determined using methods outlined by Arend (1999).

## Results

*Stream Segment Model* - The Maxent model performed well in the range-wide model. High area-under-the-curve (AUC) values for testing (0.92) and training (0.98) subsets indicated the model accurately predicted occurrences from random background points (Table 8). The MTP testing omission error rate, or false negative rate, was slightly elevated at 0.23. Of the 63 spatially unique occurrence records, 50 were used for training the model and 13 for testing. Using the minimum training presence threshold (0.09), all 12 stream segments with occurrence records in the Arkansas Valley and Ouachita Mountains were predicted present, although they occurred in stream segments of varying suitability (Figure 2). Of the 55,831 stream segments included in the range-wide model, 95% were below the MTP and classified as not suitable for LND (Table 9; Figure 3). The 2,763 stream segments with habitat suitability above the MTP are distributed between the other suitability categories and represent 3,820 km of streams.

Unimpacted mean annual discharge was the highest contributing variable to the range-wide model (Table 10; Figure 4). Depth to bedrock contributed the second highest amount of each of the variables with a percent contribution of 8.5%. Maximum elevation contributed 6.9% to the model gain. The mean monthly temperature contributed 6.5%, lithologic type contributed 5.2%, and annual precipitation contributed 4.5% to the model. The range-wide model predicted several streams in Oklahoma to have high probability of habitat suitability that do not have LND occurrences. A 10-km threshold of suitable habitat was used to highlight a number of streams with “large” amounts of predicted habitat suitability (Table 11; Figure 5).

*Habitat Mapping* – In total, 21.7 km of Blackfork Creek was scanned using side-scan sonar. The areas surveyed on Blackfork Creek were comprised mostly of run channel units and rocky-boulder substrates with some exposed bedrock (Table 12). This contrasts with Lee Creek, which was comprised mostly of pool channel units and almost entirely rocky-boulder substrates.

*Stream Reach Model* — The Maxent model performed well at the stream reach scale in Lee Creek, as the training and testing area-under-the-curve (AUC) values were high (>0.75) (Table 8; Figure 6). Of the 17 spatially unique occurrence records used in the model, the mean number of records used to train the model was 15.3 and the mean to test the model was 1.7. The MTP threshold from the logistic output from the 10-fold cross validation was 0.18. The mean MTP omission was 0%, while the mean MTP test omission rate was 10%. The jackknife measure of variable importance showed reach area to be the highest contributing variable in each of the model permutations with a percent contribution of 85.4% (Table 13; Figure 7). Percent pool composition contributed the second highest amount at 14.4%. Percent substrate composition contributed very little to the model.

## Discussion

The stream segment scale, range-wide ENM helped identify areas within the known range of LND that may harbor potentially suitable habitat. Specifically, in Oklahoma the model highlighted several streams that have not been sampled for Longnose Darter (e.g., San Bois Creek and Sallisaw Creek). Also, at this scale, both of the streams of interest, Lee Creek and Blackfork Creek, are areas of high estimated habitat suitability. This offers support to the idea that a population of reintroduced LND plausibly exists in a tributary of the Poteau River system after the translocation efforts in 1991-92.

The highest contributing variable in the stream segment ENM was unimpacted mean annual discharge, which is not the standard metric of measuring stream discharge. Unimpacted discharge is a back-calculated metric of stream discharge under conditions without impoundments that is included in the NHDPlus dataset. The standard metric of stream flows (i.e., impacted) was not used because I sought to model the broadest representation LNDs habitat suitability throughout its historical range. Also, because the occurrence records used in this model range from 1939–2014 (Table 2), impacted discharge may not have accurately represented the flow characteristics when these individuals were sampled. Because the objective of this model was to aid in the detection of this species, using unimpacted flows and occurrence records from both dammed and undammed periods results in a model that provides a liberal estimate of LND habitat suitability.

The stream reach scale ENM further differentiated suitable stream reaches in Lee Creek and Blackfork Creek by incorporating local habitat characteristics. Stream segments in these systems vary widely in size and any future sampling performed at the reach scale will allow more thorough sampling of the selected sites. By modeling at this finer scale, fewer occurrences records could be assigned specific stream reach locations. The 17 LND occurrence records that could be assigned to specific stream reaches were helpful in estimating suitable habitat at the reach scale in Lee Creek and by projecting this model into the portions of Blackfork Creek that were mapped with side-scan sonar.

A shortcoming of presence-only modeling is that without the use of absence data, we only have information about locations where a species was detected. Because these models do not have information of species frequency of occurrence, they provide relative indices of habitat suitability (Elith et al. 2006). In fact, presence-only ENMs might only be modeling where a species is more likely to be detected (Lahoz-Monfort et al. 2014) and researchers must decide if this is useful. Given the objectives of these ENMs were to aid future sampling efforts that might seek to detect this species in novel locations, estimating sites with high probability of habitat suitability is still meaningful information. If locations where this species is more easily detected are also areas of suitable habitat, the results of the stream segment ENM can be used by fisheries management agencies interesting in detecting LND.

To highlight how difficult LND can be to detect LND, the Oklahoma Water Resources Board captured a Longnose Darter in 2015 during routine sampling on the Poteau River upstream of Wister Lake, just west of the Arkansas border (OKRM-1023). This specimen was the first documented LND occurrence record in the PRS since Wister Lake was built in 1949. Like the Black Kokanee and the Robust Redhorse (Hendricks 1998, Nakabo et al. 2011), the Longnose Darter has been rediscovered in a portion of its native range after decades of believed extirpation. The range-wide ENM predicted habitat suitability for the stream segment the LND was captured in is 0.42. After being found in the PRS while presumed extirpated, Longnose Darter populations could plausibly exist in other streams in Oklahoma that have never been targeted specifically for LND sampling.

When using occurrence records collected from other databases we rely on the correct species identification. If misidentifications are made or a species known distribution is updated, many times past occurrence records or museum inventory data are not updated very well with respect to taxonomy. Robison (1992) discusses LND occurrences from the Spring River in Missouri that were determined to be Slenderhead Darter. This report to the US Forest Service is not widely accessible and some databases still list a LND occurrence record in the Spring River.

Similarly, a record of LND occurrence in the Kiamichi River of Oklahoma was determined to be a Slenderhead Darter in a genetic analysis (Robison et al. 2014), but this record has not been updated in online databases. There are a total of three occurrence records of LNDs in the Kiamichi River and a disjunct population may occur here (Holley and Long 2018). As Fourche Maline Creek is a tributary of the Poteau River, it may be possible that a remnant population of LNDs persists there after Wister Lake was built in 1949. The other four streams (i.e., Little Lee Creek, Coal Creek, Sans Bois Creek, and Sallisaw Creek) are outside of the PRS and have not been sampled specifically for Longnose Darters.

Both the stream segment and reach scale ENM identify suitable areas that support what is known about LNDs habitat preferences. Considering the results of both modeling scales, moderately large streams with abundant and large pool habitats appear to be broadly suitable. Longnose Darters have frequently been captured in low abundances across a broad range of local habitat types (Burns & McDonnell Engineering Company 1989; Gatlin and Long 2011), from shallow, swift-flowing riffle habitats to deep (>2m) low-flow pools. Wagner et al. (1985) reported seining LND from a 1.5m deep pool. This range in habitat documented by the occurrence records may be because of seasonal changes in habitat use, which has been long suspected (Thompson 1977, Robison 1992).

It appears that there are two different habitat types used by LND throughout the year. In the spring, spawning LND move into riffle habitats near swift runs up to three feet deep (Robison 1992). During the non-spawning season LNDs have been found in deeper quieter pool regions. This behavior, coupled with limited occurrences records, could mean that while rarely encountered they are a habitat generalist within stream reaches they occupy.

Little is known about the ecology and life history of the Longnose Darter. In 1975, the Fishes of Missouri field guide stated, “nothing is known about the biology of this fish” (Pflieger et al. 1975). Since then, more information about the life history of LND has been discovered (Thompson 1977, Robison and Buchanan 1988), but relatively little is still known about this species. It has been speculated that habitat fragmentation and degradation from the construction of dams has contributed to LNDs decline in distribution in Oklahoma (Wagner et al. 1985). However, several valid reports of Longnose Darter from Lake Wappapello, Missouri and Lake Nimrod, Arkansas suggest that this species may tolerate reservoir environments at least for short periods of time (Robison 1992). The research presented here will hopefully add some insights to what we know about this rare species and will aid future management efforts to monitor their status.

# CHAPTER II

# DETECTION PROBABILITIES OF LONGNOSE DARTER (*PERCINA NASUTA*) IN RELATION TO OTHER DARTERS IN LEE CREEK AND BLACKFORK CREEK OF OKLAHOMA

## Introduction

The Longnose Darter (Bailey 1941) is a species native to upland streams in the Arkansas River drainage of Arkansas, Missouri, and Oklahoma (Miller and Robison 2004). It is a benthic insectivore that inhabits swift gravel riffles and raceway areas during spawning season and moves into deeper pools later in the year (Thompson 1977; Robison 1992). Longnose Darters (LND; *Percina nasuta*) are listed as a state-endangered, tier I species of conservation concern in Oklahoma (ODWC 2016) and are considered "threatened" throughout its entire range (Jelks et al. 2008). Currently, LND is one of 404 species from the southeastern United States petitioned for listing under the Endangered Species Act (Center for Biological Conservation 2010). However, little current information on LNDs exists to assist in this determination.

Historically, Longnose Darters were only known to definitively occur in Oklahoma in two river systems, Lee Creek and the Poteau River system (multiple occurrences in the Poteau River and 1 occurrence in Brazil Creek). One of Oklahoma's rarest fish species (Robison 1992; Miller and Robison 2004), the LND likely suffered population declines after the completion of Wister Dam in 1949. Currently, only the population in Lee Creek is known to consistently persist (Burns & McDonnell Engineering Company 1990; Gatlin and Long 2011). This species was thought extirpated from much of its range in Oklahoma because for 68 years not a single individual was reported from the Poteau River system (PRS) (Cross and Moore 1952; Lindsey et al. 1983; Wagner et al. 1985). However, in 2015 a LND was collected from the Poteau River upstream of Wister Lake (OWRB 2015). This record highlights how little we know about Longnose Darters' detectability when sampling. Our ability to detect this species hinders our ability to properly document their distribution.

A potentially viable population of translocated LND may exist in Blackfork Creek, a tributary of the Poteau River. When dam construction on Lee Creek was proposed by the city of Fort Smith, Arkansas in 1989, concern grew that LNDs might be extirpated from Lee Creek. Translocation efforts in 1991-92 resulted in ~164 individuals moved from Lee Creek into three locations of Blackfork Creek to create a refuge population (Burns & McDonnell Engineering Company 1989) (Figure 1). Although surveys conducted prior to translocation failed to collect any LND in Blackfork Creek (Burns & McDonnell Engineering Company 1990), it is within the historic range of the species (Cross and Moore 1952). Blackfork Creek has not been surveyed for LND since the translocations in 1991-92 and the success of these efforts is unknown. Surveying Blackfork Creek in a framework where detection probability can be estimated would shed some light onto this unassessed population.

Imperfect species detection is an important consideration for species of conservation concern (Kéry and Schmidt 2008). The dynamic nature of stream environments further complicates the issues of species detection (Poff and Zimmerman 2010). Occupancy modeling is a technique that separates the ecological process of species occurrence from the detection process (MacKenzie et al. 2002), allowing the estimation of detection probability under different environmental sampling conditions. Detection probability can not only vary by species, but also by habitat and sampling conditions (Bailey et al. 2004; Mollenhauer et al. 2018). This is why it is important to sample under a variety of environmental conditions to establish species-environment relationships (Gwinn et al. 2016). When tasked with monitoring rare species, it is important that we are informed about heterogeneity in detection (MacKenzie et al. 2004b, 2005). For example, if the objective is to conduct targeted sampling surveys for a rare species, the study design can be adjusted to account for this difficulty (e.g., more sites, less surveys) (MacKenzie and Royle 2005). However, if the detection probability of a species is low, more surveys is the optimal choice. This creates an interesting challenge if a species is both rare and difficult to detect when sampling.

Lee Creek is thought to host the last robust population of LND in Oklahoma (Wagner et al. 1985; Gatlin and Long 2011). Since no post-stocking surveys were conducted after translocating LNDs into Blackfork Creek, it is currently unknown whether a population has been established in this tributary of the PRS. To determine how much effort is required to detect LND and to investigate how detection probably varies by stream and season, I surveyed Lee Creek and Blackfork Creek in the summer of 2017 and Lee Creek again in the spring of 2018. Therefore, the objective of this study was to sample these systems thought to contain Longnose Darter in an occupancy modeling framework to estimate detection probability and make comparisons about this rare species in relation to other co-occurring darters.

## Methods

*Study Area*

Lee Creek originates within the Boston Mountains ecoregion of Arkansas and flows into Oklahoma before turning back east towards its confluence with the Arkansas River (Figure 1). The geology of this region is characterized by Pennsylvanian-age sandstone and shale (Woods et al. 2005). Lee Creek is designated as one of Oklahoma's six "scenic rivers" (OSRC 2016) and had as many as 78 historically occurring species (Funk 1979; FERC 1987). Longnose Darters were first documented in Lee Creek in 1884 (Jordan and Gilbert 1886) and this stream currently hosts the last known persistent populations of LNDs in Oklahoma (Wagner et al. 1985; Gatlin and Long 2011).

Blackfork Creek is a tributary of the Poteau River within the Ouachita Mountains ecoregion of Oklahoma, originating in Arkansas and flowing west into Oklahoma where it meets the Poteau River upstream of Wister Lake (Figure 1). The geology of this region is distinct from other tributaries of the Poteau River because it is almost entirely contained within the Ouachita Mountains ecoregion, whereas the rest of the Poteau River system (PRS) is in the Arkansas Valley ecoregion. The geology of this region is characterized by low mountains comprised of folded, sandstone-capped ridges and shale valleys (Woods et al. 2005). Blackfork Creek is characterized by cobble, boulder, and bedrock substrate typical of this region (Cross and Moore 1952, Woods et al. 2005). Longnose Darters had never been documented in Blackfork Creek prior to the translocations in 1991-92 (Cross and Moore 1952, Lindsey et al. 1983), although the species assemblage and water quality are similar to Lee Creek (Burns & McDonnell 1990). However, because no follow up surveys on Blackfork Creek were conducted after the translocations, a LND population may persist in this tributary of the Poteau River.

*Site Selection*

Lee Creek and Blackfork Creek were selected as the target streams for this survey because they host populations of LND (Wagner et al. 1984, Gatlin and Long 2011), or plausibly could because of translocations (O'Donnell 1991, 1992). Stream reaches (hereafter referred to as sites) were selected based on the results of both the stream segment and stream reach scale ecological niche models (ENM) from Chapter 1. To conduct a targeted sampling survey of LND in Lee Creek and Blackfork Creek, only stream segments and stream reaches from the top two suitability categories from the ENMs were considered (Chapter 1; Table 4). Sixteen sites in each stream were selected haphazardly from reaches within the top two suitability categories of the stream reach ENM that lie within segments in the top two suitability categories of the stream segment niche model (Figure 2, Figure 3).

*Sampling*

To determine how much effort is required to detect LND in Lee Creek and Blackfork Creek, sampling was conducted in an occupancy modeling framework to estimate and account for imperfect species detection. Probability of detecting a species when it is present is almost never perfect and occupancy modeling uses species encounter histories over repeat surveys to estimate this probability (MacKenzie et al. 2002, 2003;

Gu and Swihart 2004). Varying species detection probability when sampling under different environmental conditions allows factors associated with detection probability to be determined and is an especially important consideration when investigating rare species (MacKenzie et al. 2005).

All sites were sampled in an occupancy modeling framework with spatially-replicated surveys. Spatial replication is useful because all of the repeat surveys at a site can be completed in a single day (Albanese et al. 2014; Mollenhauer et al. 2018). A summer field season (June – August) was conducted on Lee Creek and Blackfork Creek in 2017. Lee Creek was sampled again in the spring of 2018 (March – April), when many darter species are spawning in shallow, swift riffle habitats (Thompson 1977; Aadland 1993; Brewer et al. 2006). Sites were accessed via canoe travel between both private and public access locations.

To conduct the surveys on Lee Creek and Blackfork Creek a backpack electrofisher was used. Backpack electrofishing has been used to capture Longnose Darters before (Gatlin and Long 2011) and can be used in a range of habitat types. Ten 20-m transects (hereafter referred to as surveys) were conducted by traveling upstream in wadeable areas (<1 m deep) at each site with a Smith-Root ® LR-20B. The backpack electrofisher settings were adjusted to maintain the standard 2-5 amps of electrical output that is suitable in streams with moderate levels of conductivity (Rabeni et al. 2009). The person operating the backpack electrofisher held a fine-mesh dip net in one hand and a standard hoop anode pole in the other. The anode was moved in a sweeping pattern while traveling upstream. A technician accompanied the person operating the backpack electrofisher and held another fine-mesh dip net to maximize capture of stunned fish. All collected fish were kept in a bucket and after being counted were returned to the water downstream from each survey. A single LND was collected and preserved in 70% ethanol for use as a voucher specimen; all other LNDs encountered were placed in a photo tank (Wild Fish Conservancy ®) where photos were taken of the lateral and dorsal surface of each individual. The LNDs were then released at the site of capture.

Surveys were conducted in haphazardly selected areas at each site. To maintain independence between the spatially-replicated surveys, at least 5-m of stream width was maintained between adjacent surveys and 10-m of linear stream between from the end of any given survey to the beginning of another (Figure 2). Similar techniques have been applied to maintain independence of spatially-replicated surveys (Albanese et al. 2011, Mollenhauer et al. 2018).

Environmental variables were recorded after conducting each backpack electrofishing survey. To determine how environmental factors affected detectability (MacKenzie et al. 2006), survey-specific covariates were recorded during each sampling occasion (Table 1). The survey-specific covariates were recorded at the beginning and end points of each survey and averaged. Water clarity was assessed with a 120-cm Secchi tube (Forestry Suppliers ®) after allowing the water to settle. Water velocity was measured at 0.6 water depth and depth was recorded using a Global Water Flow Probe (model FP111) in the same locations at each survey. Proportion of cobble substrate was recorded using a 0.5m x 0.5m quadrat and estimates represent proportion of substrate within quadrat above 64-mm in size. These covariates were selected because they have been used to model the detection probabilities of similar species (Albanese et al. 2011, 2014; Anderson et al. 2012; Dextrase et al. 2014) and are presumed relevant to LNDs based on similar ecology (Miller and Robison 2004). To account for any spatial autocorrelation of surveys within sites, drainage area ($km^2$) for each reach was used to account for spatial position. Drainage area was assigned as a non-temporally variant detection variable for each survey (NHDPlus version 2; McKay et al. 2012).

*Detection Probabilities*

Backpack electrofishing detection probability was quantified by using the occupancy modeling framework described by MacKenzie et al. (2002). Occupancy models utilize repeat surveys to separate the ecological process and the detection process (MacKenzie et al. 2006). Each of these can be modeled as a function of covariates to investigate how they change across sites, surveys, and environmental conditions. Because an objective of this study was to determine how much effort is required to detect Longnose Darters, I focused on the detection process.

To compare detection probabilities of the darter communities within a season and between systems, multi-species detection models (MacKenzie et al. 2004a) were developed that included stream as a categorical variable. Banded Darter was the reference species used and coefficients of other darters are interpreted in relation to this species. Banded Darter was selected as the reference species because it was widely encountered in both streams. To compare detection probabilities of a darter community within a system and between seasons, Lee Creek was sampled in both the summer and spring field seasons. A set of candidate detection models for LND (N= 14) and a set of candidate multi-species detection models (N= 22) were developed with varying levels of complexity (Table 2; Table 3). Water clarity was not used in any analysis because variation was minimal across all sites in Lee Creek and Blackfork Creek. Water depth, water velocity, and substrate size were included as continuous variables. Species, channel unit, and stream name were included as categorical variables. A Pearson's correlation $|r| > 0.7$ was used as the threshold for highly correlated variables. All continuous variables were natural log-transformed because of right-skewed distributions. The datasets were then standardized such that each continuous variable had a mean of 0 and a variance of 1. Species-specific detection probabilities were examined by incorporating species as a categorical variable and including interaction terms between species and the continuous covariates (see Mollenhauer et al. 2018). Detection models were fit using the package "unmarked" (Fiske and Chandler 2011) in the statistical software R (version 3.4.2; R. Core Team 2017).

A goodness-of-fit test (MacKenzie and Bailey 2004) was applied to the most complex multi-species detection models (n = 300 bootstraps) and indicated that these models were not overdispersed (i.e., $\hat{c} \leq 1$, Summer: $\hat{c} = 0.99$; Spring: $\hat{c} = 0.86$). Models were ranked using Akaike information criterion corrected for small sample size ($AIC_c$; Burnham and Anderson 2002). A trap response (i.e., a change in detection probability after the first detection) was added to the top model from each season, where a decrease in $AIC_c$ indicates non-independence among the spatially-replicated surveys.

## Results

*Sampling*

In total, 2130 darters among 9 species were sampled between the two field seasons (Table 4). Of the 9 species encountered, all 9 were detected in Lee Creek and 7 species were detected in Blackfork Creek. Longnose Darter and Sunburst Darter were not sampled from any of the sites on Blackfork Creek. Sunburst Darter (*Etheostoma mihileze*), which was recently split from the Stippled Darter (*Etheostoma punctulatum*; Mayden 2010), is an Ozark endemic species and was not expected to be in Blackfork Creek. Of the 16 sites in Blackfork Creek, 3 were at the same locations as the 1991-92 translocation sites (Figure 3). Despite 160 surveys at these 16 sites, Longnose Darters were not detected in Blackfork Creek (Table 5).

Environmental sampling conditions varied between the two field seasons. In the summer field season, water velocity was on average lower in Blackfork Creek compared to Lee Creek (Table 6). The surveys conducted on Blackfork Creek also had on average deeper water and a larger proportion of cobble substrate. Both average water depth and average water velocity increased on Lee Creek in the spring compared to the summer field season.

Overall, detection probabilities increased for the darters in Lee Creek in the spring field season compared to the summer field season (Table 7). Out of the 9 species encountered in Lee Creek, detection increased for 7 species. Detection probabilities decreased for Greenside Darter and Sunburst Darter in the spring. Longnose Darter had the lowest detection probability (5%) out of the co-occurring species in summer 2017. However, when Lee Creek was sampled again in spring 2018, the detection probability for LND increased to 10% (Table 7).

*Analysis*

There were no highly correlated (|r| >0.7) detection covariates in the summer 2017 or the spring 2018 field season (Table 8). The multi-species candidate models were ranked using AICc and a trap response was applied to each top model to see if this lowered the AICc value. In both model sets, the trap response showed an improvement over the top models (Table 9, 10). This indicates that the spatially-replicated samples were not totally independent at each site in both the summer and spring field seasons.

The top multi-species model for the 2017 summer field season includes interaction effects between species and water velocity and depth, an additive relationship with channel unit and the trap response, and additive quadratic relationships with depth and velocity (Table 3). The additive quadratic effects of depth and velocity are not species-specific, whereas the interaction between species, depth, and velocity allow the slopes to vary by species. There is a positive linear relationship between detection probability and increasing water depth for Logperch, LND, and Sunburst Darter (Figure 4), while holding all other variables at mean values. However, these estimates do have a fair amount of uncertainty around them, as Figure 5 shows for Logperch. The 95% confidence intervals for the other species of darters with water depth overlap with zero and were not plotted (Table 9). In Blackfork Creek, Logperch, Channel, Orangethroat, and Redfin Darters exhibit a negative relationship with increasing water velocity, while holding all other variables at mean values (Figure 6). Fantail Darters exhibit a slight positive relationship with detection probability and water velocity.

The top multi-species model for the spring field season that includes only Lee Creek is model 12 with the trap response (Table 2.) This model includes only an interaction between species and quadratic water velocity plus an additive trap effect. All 9 darter species encountered in Lee Creek in the summer were detected again in the spring field season. Of these, only the 95% confidence intervals around the coefficient estimates for Logperch, Redfin, and Sunburst Darters did not overlap zero (Table 10, Figure 7). There are varying species-specific quadratic water velocity relationships, but these estimates also have uncertainty around them (Figure 8). Figure 7 shows Logperch, Redfin Darter, and Sunburst Darter and illustrates their varying species-specific relationships. The detection probability for these three species is highest with water velocities >0.8 m/s and <1.2 m/s.

The Longnose Darter only candidate models for both field seasons were no better than the null model. Longnose Darters were likely not detected enough to model a relationship between their encounter histories and the environmental covariates.

## Discussion

Longnose Darter detection probability was the lowest of all the species sampled during the summer field season. While no LND were encountered in Blackfork Creek, a detection probability of 0.05 in Lee Creek suggests likely twice the survey effort (20 surveys) is necessary to detect a Longnose Darter given the species is present. In the spring field season, LND detection probability in Lee Creek increased to 0.10. Only 8 individuals were encountered in the summer and 22 individuals were detected in the spring season. Future surveys for Longnose Darters in Lee Creek and in other streams with potentially suitable habitat should therefore be conducted in the spring to both increase the odds of detecting the species and to decrease the amount of resources used in the field. Gatlin (2013) performed a community assemblage survey of Lee Creek, sampling a wide variety of habitats and locations, and detected LND at four locations. My targeted sampling for Longnose Darter on Lee Creek resulted in detecting Longnose Darters in eight different sites; LND were found at three of the four sites from Gatlin (2013) and at five new sites (Table 11).

The detection probability for Sunburst Darters in Lee Creek ranged from 0.22 in the summer field season to 0.07 in the spring field season. Sunburst Darter (*Etheostoma mihileze*) is a species of greatest conservation need in Oklahoma (ODWC 2016). To my knowledge, detection probability has never been quantified for this species, or for the species it was recently split from, the Stippled Darter (Mayden 2010). This was one of only two decreases in species detection probability from summer to spring of the nine darter species sampled. If routine monitoring for this species should become a priority, knowing detection rates are higher in the summer for Sunburst Darters in Lee Creek will aid natural resource managers.

The detection estimates discussed here are in relation to a backpack electrofishing survey. Backpack electrofishing is commonly used in streams and can be applied where a tow-barge is impractical (Rabeni et al. 2009). A limitation of backpack electrofishing is that you are limited to wadeable depths (<1m). As Longnose Darters are thought to occupy pool habitats in the non-spawning season (Robison 1992), using a gear that can sample these deeper areas may be preferred. However, because darters are benthic species with absent or greatly reduced air bladders (Evans and Page 2003), it can be difficult to sample them in these areas. Albanese et al. (2011) estimated darter detection probabilities using snorkel surveys, but did not include deep-slow pools. Another concern is that occupancy modeling assumes there are no false species identifications. Backpack electrofishing allowed the handling of each LND and insured positive identification through use of a photo tank. However, there are only a few ways to identify a Longnose Darter from other similar species and there can be conflicting identifications (Holley and Long 2018).

My findings add to existing knowledge about Longnose Darter and can be used in future sampling efforts to conserve this species because, while only two streams were sampled in this study, this is the first time detection probability has been quantified for LND. Any future targeted sampling of LNDs on Lee Creek should be conducted in the spring because detection rates are higher. The increased attention imperfect species detection has received in the literature in recent years (Kellner and Swihart 2014) is well deserved because it affects nearly all species (Kéry and Schmidt 2008). By accounting for our imperfect surveys hopefully we can improve conservation efforts for imperiled species like the Longnose Darter.

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## APPENDIX A: Tables for Chapter 1

Table 1. Data sources for Longnose Darter occurrence records collected from online databases and used in the range-wide niche model.

| Data Sources | Collection Dates |
|---|---|
| Arkansas Department of Environmental Quality | 1963–2014 |
| Cornell University Museum of Vertebrates | 1951–1967 |
| Illinois Natural History Survey | 1948 |
| Louisiana Museum of Natural History | 1984 |
| Mississippi Museum of Natural Science | 1992 |
| Tulane University Museum of Natural History | 1955–1974 |
| University of Alabama | 1991 |
| University of Arkansas Collections Facility | 1962-1963 |
| University of Kansas Biodiversity Institute | 1947–1973 |
| University of Michigan Museum of Zoology | 1939–1955 |
| University of Texas | 1972 |
| University of Tulsa | 1962–1972 |
| Yale University Peabody Museum | 1984–2010 |

Accessed March 2016

Table 2. Environmental predictor variables used in the range-wide ecological niche model for Longnose Darter at a stream-segment scale.

| Variable | Unit | Source[1] | Development Date |
|---|---|---|---|
| Lithologic type | Category | CONUS -SOIL | 1998 |
| Depth to bedrock | cm | CONUS-SOIL | 1998 |
| Soil permeability | cm/hr | CONUS-SOIL | 1998 |
| Rock fragment volume | % | CONUS-SOIL | 1998 |
| Annual precipitation | mm*100 | NHDPlusV2 | 2015 |
| Mean monthly temperature | °C*100 | NHDPlusV2 | 2015 |
| Total drainage area | $km^2$ | NHDPlusV2 | 2015 |
| Maximum elevation | cm | NHDPlusV2 | 2015 |
| Slope | km/km | NHDPlusV2 | 2015 |
| Unimpacted mean annual discharge | $m^3/s$ | NHDPlusV2 | 2015 |
| Unimpacted mean annual flow velocity | m/s | NHDPlusV2 | 2015 |

[1]CONUS-SOIL = Conterminous United States Multi-Layer Soil Characteristics Dataset, available online http://www.soilinfo.psu.edu/index.cgi, accessed March 2016; NHDPlusV2 = National Hydrography Dataset Version 2, available online https://nhd.usgs.gov/, accessed March 2016.

Table 3. Correlation matrix for the predictor variables used in the stream segment ecological niche model. Pearson's |r| >0.7 was used as the threshold for highly correlated variables.

| | Depth-to-bedrock | Soil permeability | Rock fragment volume | Annual precipitation | Mean monthly temperature | Total drainage area | Maximum elevation | Slope | Annual discharge | Annua flow velocit |
|---|---|---|---|---|---|---|---|---|---|---|
| Depth-to-bedrock | 1 | 0.37 | -0.27 | -0.31 | -0.34 | -0.05 | 0.02 | -0.16 | -0.05 | 0.00 |
| Soil permeability | 0.37 | 1.00 | 0.15 | 0.08 | -0.05 | 0.03 | 0.05 | -0.03 | 0.01 | 0.07 |
| Rock fragment volume | -0.27 | 0.15 | 1.00 | 0.05 | -0.43 | -0.14 | 0.45 | 0.22 | -0.12 | 0.07 |
| Annual precipitation | -0.31 | 0.08 | 0.05 | 1.00 | 0.34 | -0.02 | 0.09 | 0.26 | -0.02 | 0.03 |
| Mean monthly temperature | -0.34 | -0.05 | -0.43 | 0.34 | 1.00 | 0.06 | -0.61 | -0.15 | 0.05 | -0.09 |
| Total drainage area | -0.05 | 0.03 | -0.14 | -0.02 | 0.06 | 1.00 | -0.09 | -0.04 | 0.81 | 0.28 |
| Maximum elevation | 0.02 | 0.05 | 0.45 | 0.09 | -0.61 | -0.09 | 1.00 | 0.46 | -0.11 | -0.01 |
| Slope | -0.16 | -0.03 | 0.22 | 0.26 | -0.15 | -0.04 | 0.46 | 1.00 | -0.05 | -0.01 |
| Annual discharge | -0.05 | 0.01 | -0.12 | -0.02 | 0.05 | 0.81 | -0.11 | -0.05 | 1.00 | 0.42 |
| Annual flow velocity | 0.00 | 0.07 | 0.07 | 0.03 | -0.09 | 0.28 | -0.01 | -0.01 | 0.42 | 1.00 |

Table 4. Categories of increasing probability of suitability for the Longnose Darter at the stream segment and stream reach-scale estimated with niche models. The minimum training presence (MTP) values were used as the suitability threshold for the unsuitable category (0) and the category of lowest suitability (1) at each scale. Categories 1–4 represent suitability categories above the MTP.

| Maxent Output Values | | |
|---|---|---|
| Suitability | Segment | Reach |
| 0 | 0.00 – 0.09 | 0.00 – 0.18 |
| 1 | 0.09 – 0.20 | 0.18 – 0.20 |
| 2 | 0.20 – 0.50 | 0.20 – 0.50 |
| 3 | 0.50 – 0.80 | 0.50 – 0.71 |
| 4 | 0.80 – 1.00 | |

Table 5. Correlation matrix for the predictor variables used in the stream reach ecological niche model. Pearson's |r| >0.7 was used as the threshold for highly correlated variables.

| | Reach Area | % Riffle | % Run | % Glide | % Pool | % Rocky-boulder | % Bedrock | % Fine |
|---|---|---|---|---|---|---|---|---|
| Reach Area | 1.00 | -0.48 | -0.39 | -0.17 | 0.60 | -0.39 | 0.39 | 0.36 |
| % Riffle | -0.48 | 1.00 | 0.38 | -0.18 | -0.73 | 0.22 | -0.22 | -0.11 |
| % Run | -0.39 | 0.38 | 1.00 | -0.24 | -0.83 | 0.19 | -0.19 | -0.08 |
| % Glide | -0.17 | -0.18 | -0.24 | 1.00 | -0.07 | 0.19 | -0.19 | -0.03 |
| % Pool | 0.60 | -0.73 | -0.83 | -0.07 | 1.00 | -0.32 | 0.32 | 0.13 |
| % Rocky-boulder | -0.39 | 0.22 | 0.19 | 0.19 | -0.32 | 1.00 | -1.00 | -0.03 |
| % Bedrock | 0.39 | -0.22 | -0.19 | -0.19 | 0.32 | -1.00 | 1.00 | 0.03 |
| % Fine | 0.36 | -0.11 | -0.08 | -0.03 | 0.13 | -0.03 | 0.03 | 1.00 |

Table 6. An example of the disproportionate stratified mapping approach applied to the Poteau River system, shown for Blackfork Creek.

| Stream | Suitability Category | Length (km) | Above 5 km Threshold | Proposed Length (km) | % Mapped |
|---|---|---|---|---|---|
| Blackfork Creek | 0 | 0.0 | No | 0.0 | 0 |
| Blackfork Creek | 1 | 8.1 | Yes | 2.0 | 25 |
| Blackfork Creek | 2 | 4.0 | No | 4.0 | 100 |
| Blackfork Creek | 3 | 1.4 | No | 1.4 | 100 |
| Blackfork Creek | 4 | 35.1 | Yes | 8.8 | 25 |

Table 7. Classification scheme and descriptions used to categorize substrate types from side-scan sonar images in the Poteau River system of Oklahoma; adapted from Kaeser and Litts (2010) and Gatlin (2013).

| Class | Description |
|---|---|
| Fine | >75% of area composed of particles < 2-mm diameter (sand, silt, clay, or fine organic detritus) |
| Rocky-boulder | >75% of area comprised of rocks > 64-mm diameter with scattered boulders > 500-mm (gravel or cobble) |
| Bedrock | >75% of area comprised of large, smooth sheets of exposed bedrock or outcropping |

Table 8. Maxent model outputs for the stream segment and stream reach scale Longnose Darter ecological niche models.

| Metric | Stream Segment | Stream Reach |
|---|---|---|
| Training samples | 50 | 15.3 |
| Testing samples | 13 | 1.7 |
| Training AUC | 0.98 | 0.79 |
| Testing AUC | 0.92 | 0.77 |
| MTP logistic threshold | 0.09 | 0.18 |
| MTP training omission | 0.00 | 0.00 |
| MTP test omission | 0.23 | 0.10 |

Table 9. Number of stream segments and total stream length within the suitability categories from the range-wide ecological niche model of Longnose Darter.

| Suitability Category | Stream Segments | Total Stream Length (km) |
|---|---|---|
| 0 | 53068 | 94084.2 |
| 1 | 1012 | 1463.4 |
| 2 | 984 | 1444.9 |
| 3 | 717 | 840.6 |
| 4 | 50 | 71.0 |
| Total | 55831 | 97904.1 |

Table 10. Environmental variable contribution to model accuracy gain for the range-wide, stream segment scale ecological niche model.

| Variable | Percent Contribution |
| --- | --- |
| Unimpacted Mean Annual Discharge | 68.4 |
| Depth to Bedrock | 8.5 |
| Maximum Elevation | 6.9 |
| Mean Monthly Temperature | 6.5 |
| Lithologic Type | 5.2 |
| Annual Precipitation | 4.5 |

Table 11. Streams in Oklahoma with at least 10 km of potentially suitable habitat in the suitability categories for Longnose Darter based on the stream segment scale, range-wide ecological niche model.

| | | Length (km) in Habitat Suitability Category | | | |
|---|---|---|---|---|---|
| Ecoregion | Stream | 1 | 2 | 3 | Total Length |
| Arkansas Valley | Canadian River | 2.8 | 5.6 | 2.0 | 10.5 |
| Arkansas Valley | Coal Creek | 10.3 | 1.7 | 0.0 | 12.0 |
| Arkansas Valley | Fourche Maline | 22.9 | 0.7 | 0.0 | 23.6 |
| Arkansas Valley | Poteau River | 8.4 | 3.8 | 3.8 | 16.0 |
| Arkansas Valley | Sans Bois Creek | 7.8 | 13.1 | 0.0 | 20.9 |
| Boston Mountains | Lee Creek | 0.0 | 12.3 | 14.2 | 26.4 |
| Boston Mountains | Little Lee Creek | 5.2 | 18.7 | 0.0 | 24.0 |
| Boston Mountains | Sallisaw Creek | 0.0 | 11.2 | 2.2 | 13.4 |
| Ouachita Mountains | Big Eagle Creek | 7.0 | 10.0 | 0.0 | 16.9 |
| Ouachita Mountains | Blackfork Creek | 0.0 | 11.9 | 0.0 | 11.9 |
| Ouachita Mountains | Buffalo Creek | 9.0 | 37.6 | 0.0 | 46.6 |
| Ouachita Mountains | Glover River | 9.3 | 6.6 | 0.0 | 15.9 |
| Ouachita Mountains | Jackfork Creek | 3.1 | 11.5 | 2.1 | 16.7 |
| Ouachita Mountains | Mountain Fork | 5.5 | 31.0 | 7.9 | 44.3 |
| Ouachita Mountains | West Fork Glover River | 14.1 | 0.0 | 0.0 | 14.1 |
| Ozark Highlands | Baron Fork | 44.1 | 0.6 | 0.0 | 44.7 |
| Ozark Highlands | Flint Creek | 0.0 | 9.1 | 1.4 | 10.5 |
| Ozark Highlands | Illinois River | 68.1 | 4.8 | 17.5 | 90.4 |
| Ozark Highlands | Spavinaw Creek | 19.7 | 3.7 | 4.0 | 27.4 |
| Ozark Highlands | Spring Creek | 11.1 | 0.0 | 0.0 | 11.1 |

Table 12. A comparison of Lee Creek and Blackfork Creek using results of side-scan sonar surveys. The Lee Creek summaries come from Gatlin's (2013) surveys.

| Variable | Lee Creek | Blackfork Creek |
|---|---|---|
| Reach Area ($m^2$) | 11789 | 15000 |
| % Glide | 9.7 | 4.2 |
| % Pool | 47.2 | 26.7 |
| % Riffle | 21.2 | 26.7 |
| % Run | 21.8 | 42.3 |
| % Rocky-boulder | 96.8 | 73.6 |
| % Bedrock | 3.2 | 22.5 |
| % Fine | 0 | 3.7 |

Table 13. Environmental variable contribution to model accuracy gain for the stream reach scale ecological niche model in Lee Creek, Oklahoma.

| Variable | Percent Contribution |
|---|---|
| Reach Area | 85.4 |
| % Pool Channel Unit | 14.4 |
| % Bedrock Substrate | 0.2 |
| % Glide Channel Unit | 0.0 |

## APPENDIX B: Figures for Chapter 1

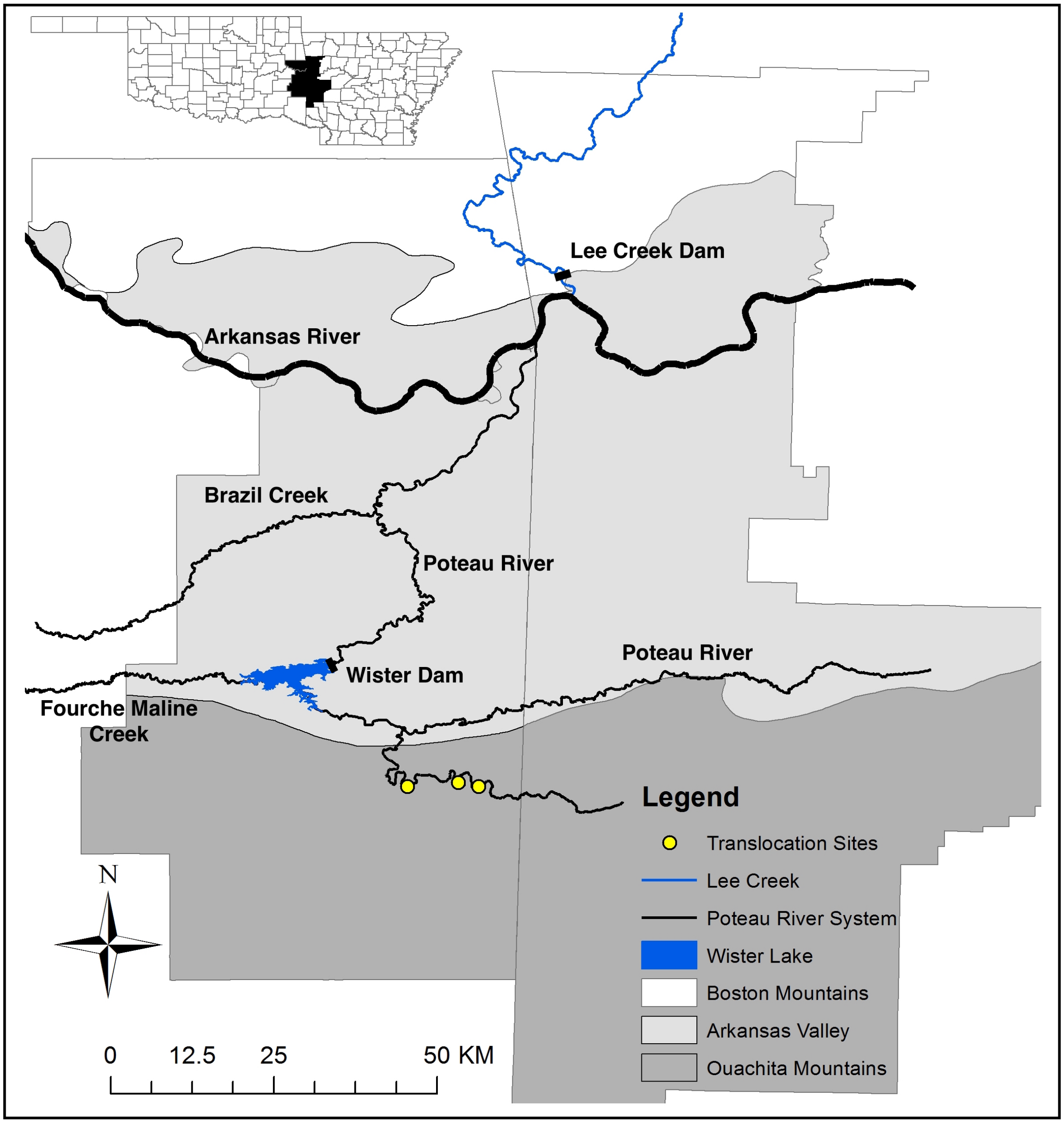


Figure 1. Map of Lee Creek and the Poteau River system in eastern Oklahoma and western Arkansas shown with ecoregions and Longnose Darter translocation sites in Blackfork Creek.

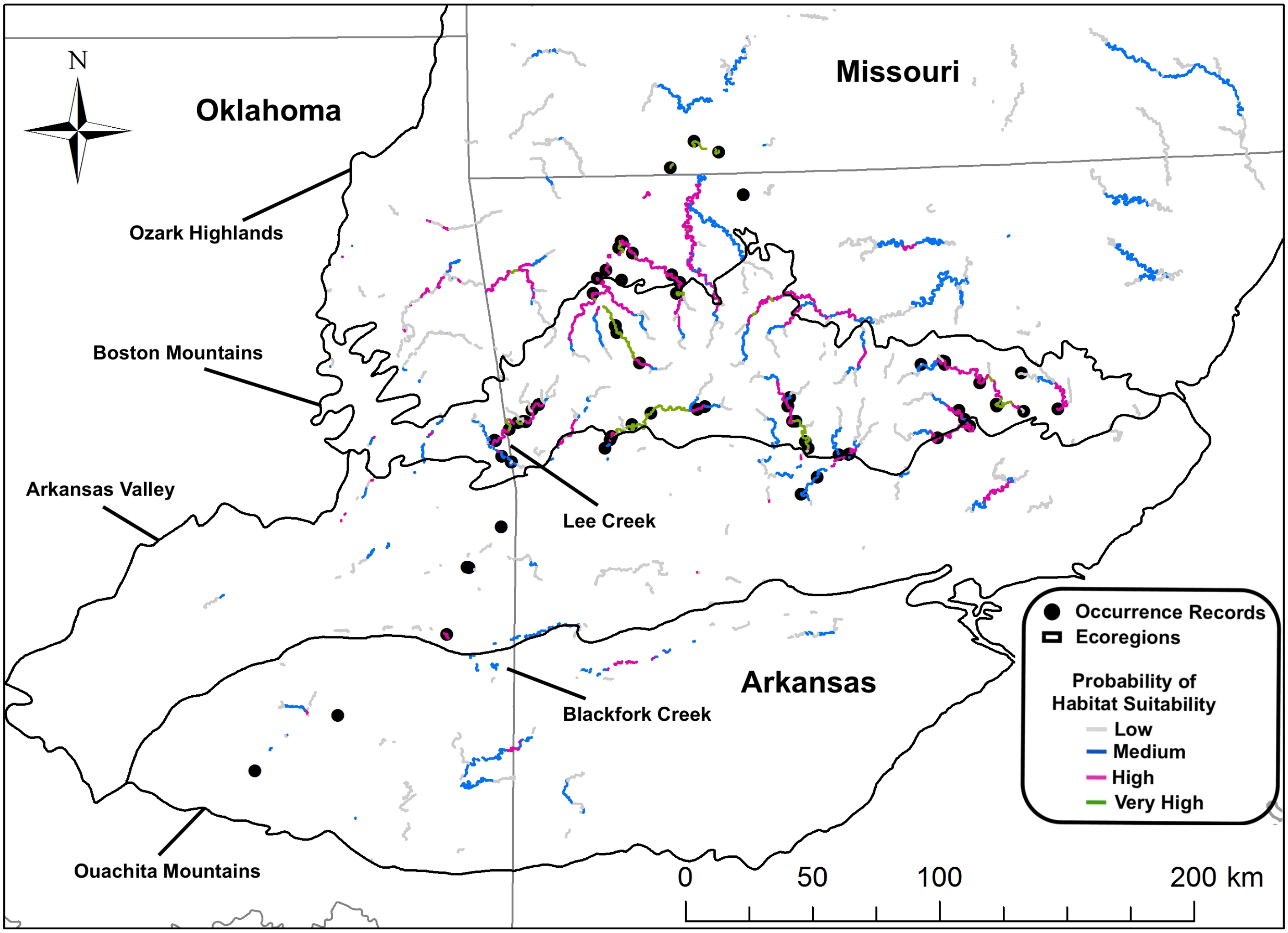


Figure 2. Map of the occurrence records used in the range-wide ecological niche model and the estimated probability of habitat suitability for stream segments in the study area above the minimum training presence threshold (Table 4).

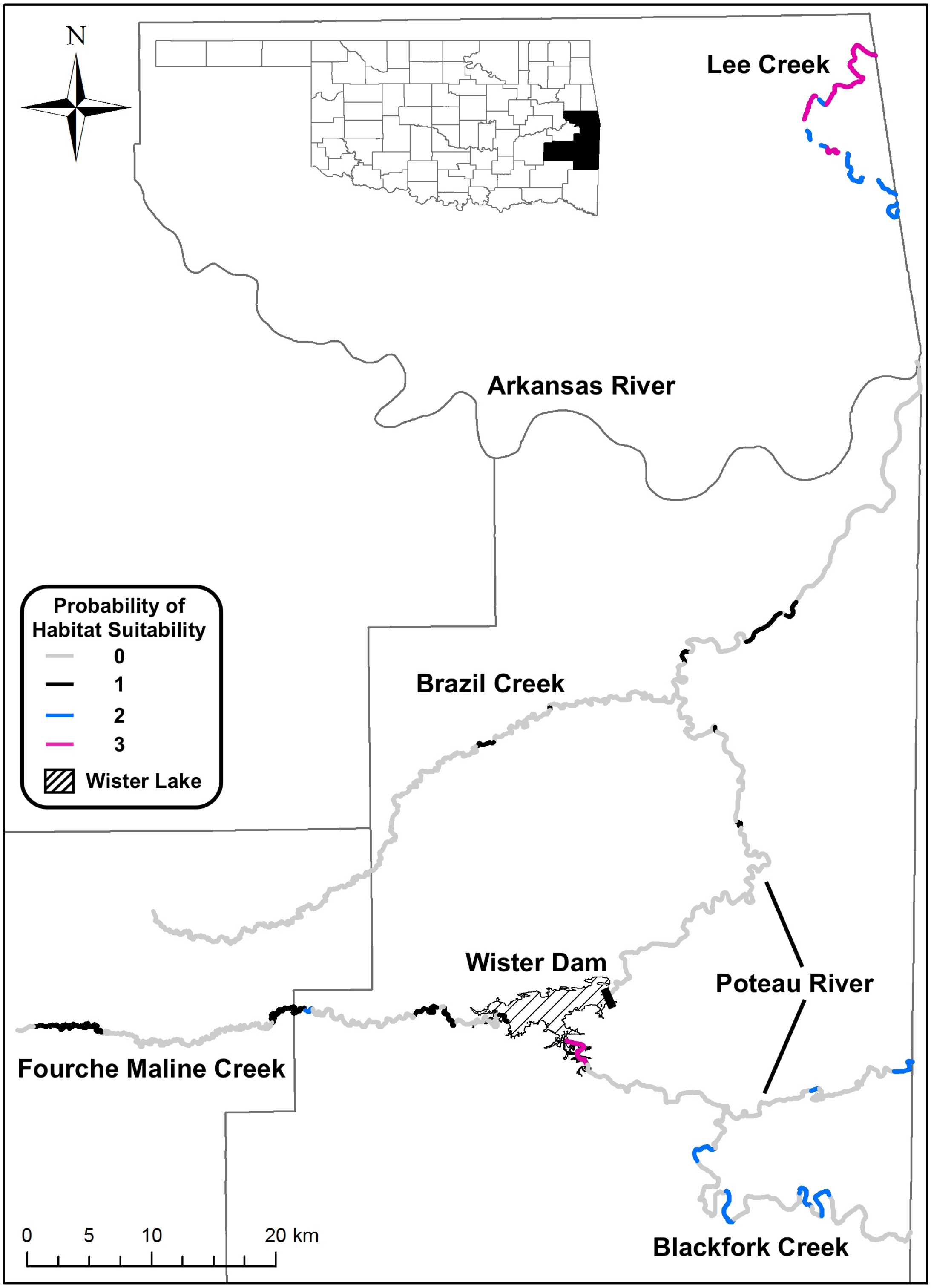


Figure 3. The Poteau River system and Lee Creek shown with outputs from the range-wide niche model for the Longnose Darter in categories of increasing probability of habitat suitability (Table 4).

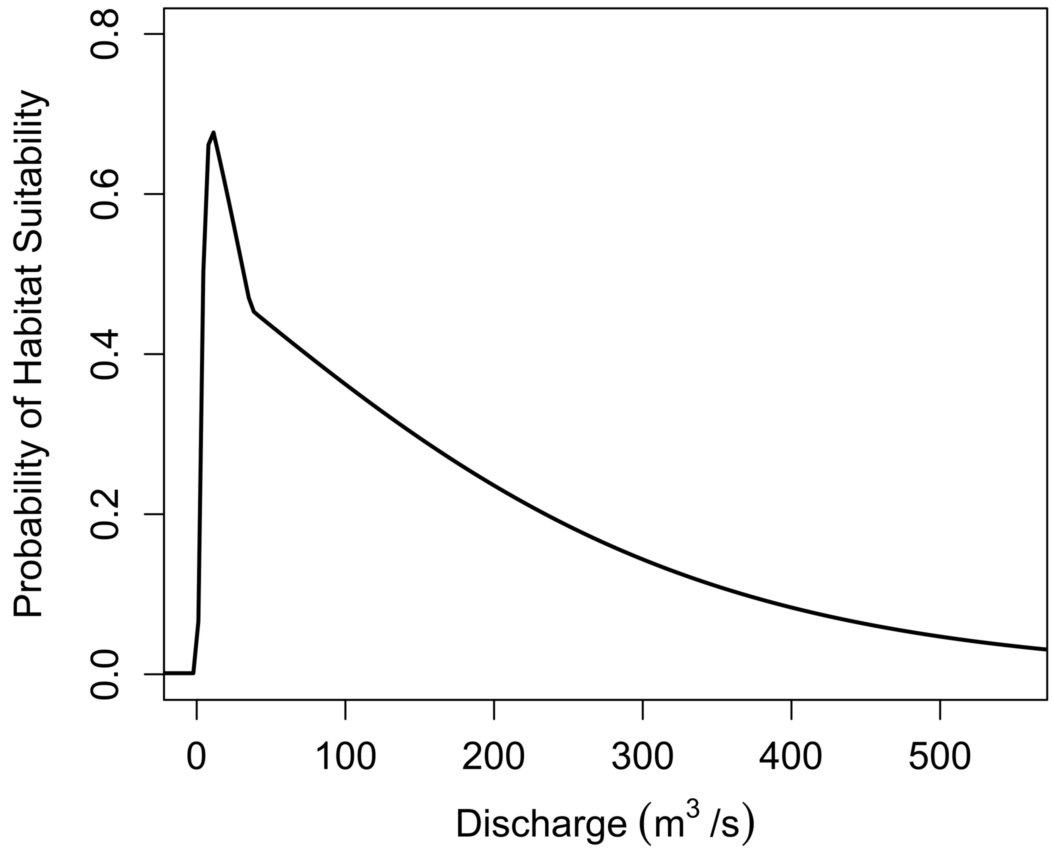


Figure 4. Environmental variable response curve for unimpacted mean annual discharge, the highest contributing variable from the stream segment scale, range-wide ecological niche model.

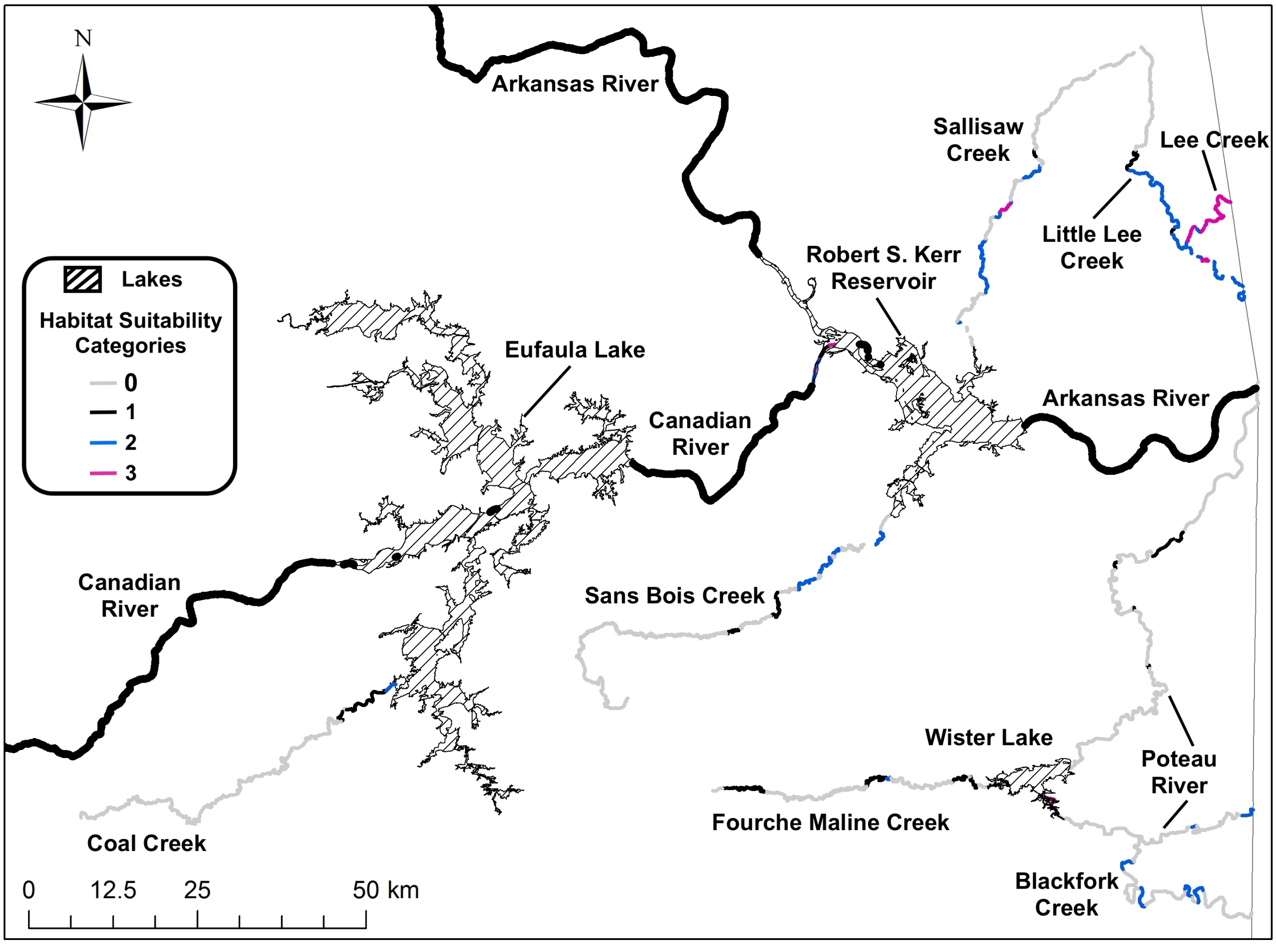


Figure 5. Streams in Oklahoma with >10 km of high probability of habitat suitability for Longnose Darter from the stream segment, range-wide ecological niche model (Table 11).

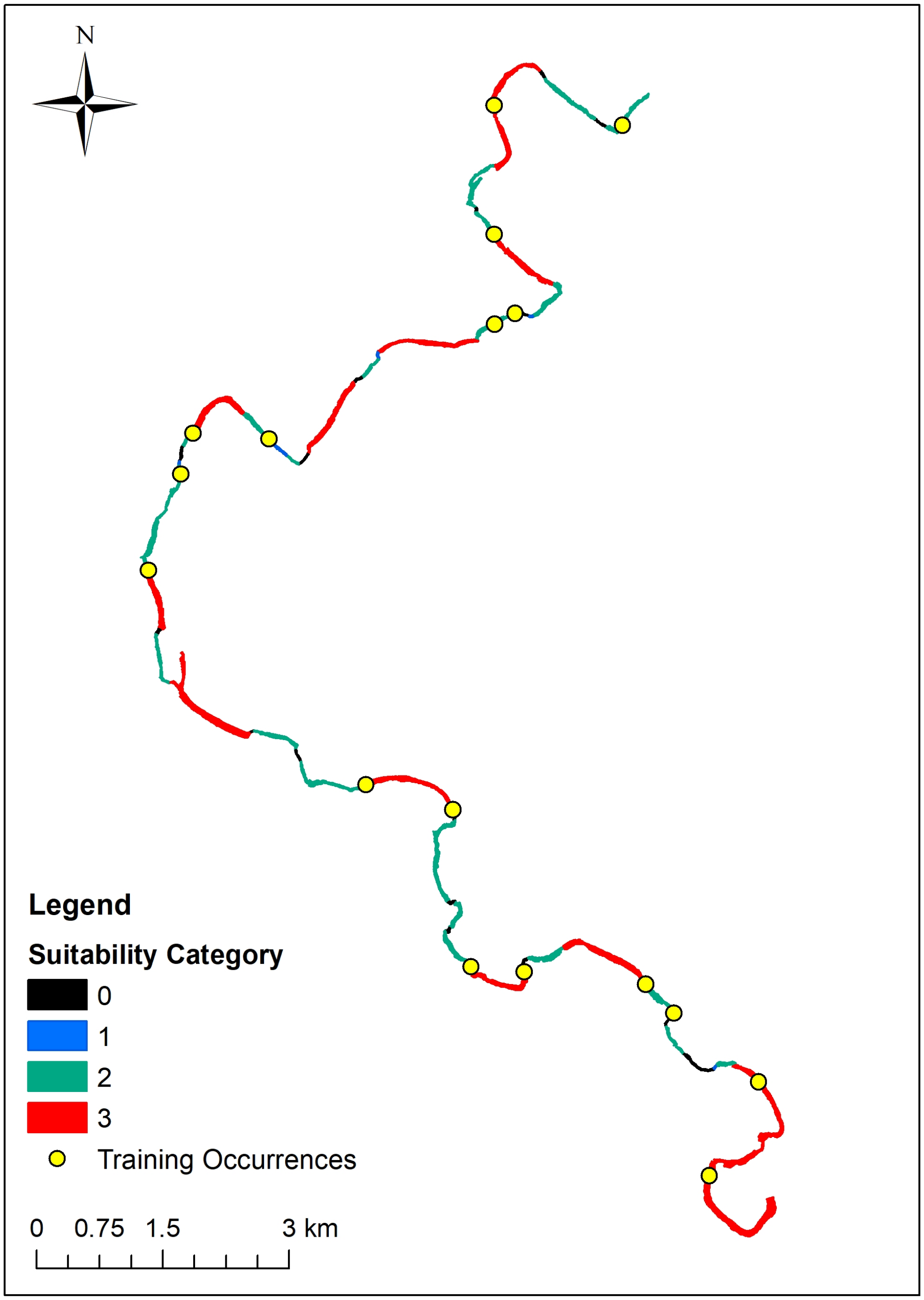


Figure 6. Lee Creek shown with outputs from the reach scale niche model for Longnose Darter in categories of increasing probability of habitat suitability (see Table 4), shown with Longnose Darter occurrences that were used to train the model (N = 17).

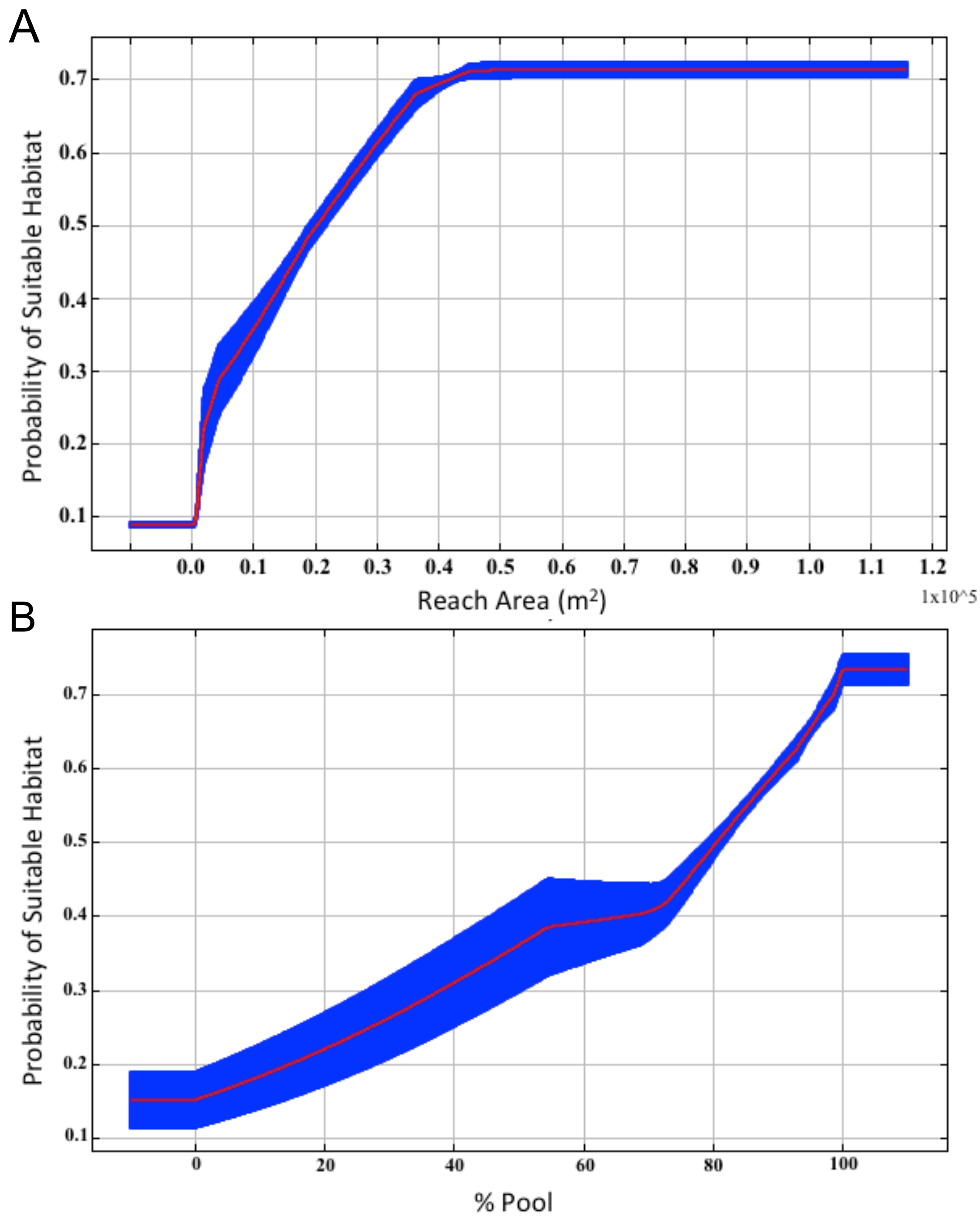


Figure 7. Environmental variable response curves from the stream reach scale ecological niche model. Panel A shows the response curve for the continuous variable Reach Area ($m^2$) and panel B shows the response curve for the continuous variable % Pool.

## APPENDIX C: Tables for Chapter 2

Table 1. Environmental covariates used to estimate detection probability, shown with metrics and methods of calculation or measurement.

| Covariate | Method |
|---|---|
| Water Clarity (m) | Secchi tube, 120-cm |
| Water Velocity (m/sec) | Measured at 0.6 depth at endpoints of surveys |
| Depth (m) | Measured at endpoints of surveys |
| Proportion of cobble substrate | Proportion of 0.5m x 0.5m quadrat comprised of cobble substrate (>64-mm) at endpoints of surveys |

Table 2. List of candidate models (n=22) for the spring 2018 multi-species detection models of nine darter species in Lee Creek, Oklahoma. Ψ is occupancy, *p* is detection probability, *K* is the number of parameters in the model. "da" is for drainage area. * denotes the top model.

| # | Model | *K* | AICc |
|---|---|---|---|
| null | Ψ(~1),*p*(~1) | *2* | 1214.099 |
| 1 | Ψ(da),*p*(species) | 11 | 1090.961 |
| 2 | Ψ(da),*p*(species + depth + depth$^2$) | 13 | 1083.041 |
| 3 | Ψ(da),*p*(species + velocity + velocity$^2$) | 13 | 1056.949 |
| 4 | Ψ(da),*p*(species + substrate) | 25 | 1090.776 |
| 5 | Ψ(da),*p*(species + channel unit) | 14 | 1065.130 |
| 6 | Ψ(da),*p*(species + depth + depth$^2$ + velocity + velocity$^2$) | 15 | 1054.993 |
| 7 | Ψ(da),*p*(species + depth + depth$^2$ + velocity + velocity$^2$ + substrate + channel unit) | 19 | 1039.521 |
| 8 | Ψ(da),*p*(species*depth + depth$^2$) | 21 | 1085.358 |
| 9 | Ψ(da),*p*(species*velocity + velocity$^2$) | 21 | 1035.223 |
| 10 | Ψ(da),*p*(species*substrate) | 20 | 1069.656 |
| 11 | Ψ(da),*p*(species*(depth + depth$^2$)) | 29 | 1098.961 |
| 12 | Ψ(da),*p*(species*(velocity + velocity$^2$)) | 29 | 1020.691 |
| *12_trap | Ψ(da),*p*(species*(velocity + velocity$^2$) + trap) | 30 | 1016.326 |
| 13 | Ψ(da),*p*(species*(depth + depth$^2$ + substrate)) | 38 | 1081.465 |
| 14 | Ψ(da),*p*(species*(velocity + velocity$^2$ + substrate)) | 38 | 1026.398 |
| 15 | Ψ(da),*p*(species*(depth + velocity) + depth$^2$ + velocity$^2$) | 31 | 1046.730 |
| 16 | Ψ(da),*p*(species*(depth + velocity) + depth$^2$ + velocity$^2$ + channel unit) | 34 | 1052.067 |
| 17 | Ψ(da),*p*(species*(depth + depth$^2$ + velocity + velocity$^2$)) | 47 | 1074.326 |
| 18 | Ψ(da),*p*(species*(depth + velocity + substrate) + depth$^2$ + velocity$^2$ + channel unit) | 43 | 1061.349 |
| 19 | Ψ(da),*p*(species*(depth + depth$^2$ + velocity + velocity$^2$) + channel unit) | 50 | 1087.231 |
| 20 | Ψ(da),*p*(species*(depth + depth$^2$ + velocity + velocity$^2$ + substrate)) | 56 | 1103.298 |
| 21 | Ψ(da),*p*(species*(depth + depth$^2$ + velocity + velocity$^2$ + substrate) + channel unit | 59 | 1126.567 |

Table 3. List of candidate models (n=22) for the summer 2017 multi-species detection models of nine darter species in Lee Creek and Blackfork Creek, Oklahoma. Ψ is occupancy, *p* is detection probability, *K* is the number of parameters in the model. "da" is for drainage area. * denotes the top model.

| # | Model | *K* | AICc |
|---|---|---|---|
| null | Ψ(~1),*p*(~1) | 2 | 2574.026 |
| 1 | Ψ(stream + da),*p*(species) | 12 | 2440.464 |
| 2 | Ψ(stream + da),*p*(species + depth + depth$^2$) | 14 | 2380.080 |
| 3 | Ψ(stream + da),*p*(species + velocity + velocity$^2$) | 14 | 2385.137 |
| 4 | Ψ(stream + da),*p*(species + substrate) | 13 | 2437.276 |
| 5 | Ψ(stream + da),*p*(species + channel unit) | 15 | 2367.637 |
| 6 | Ψ(stream + da),*p*(species + depth + depth$^2$ + velocity + velocity$^2$) | 16 | 2370.496 |
| 7 | Ψ(stream + da),*p*(species + depth + depth$^2$ + velocity + velocity$^2$ + substrate + channel unit) | 20 | 2360.546 |
| 8 | Ψ(stream + da),*p*(species*depth + depth$^2$) | 22 | 2308.660 |
| 9 | Ψ(stream + da),*p*(species*velocity + velocity$^2$) | 22 | 2286.232 |
| 10 | Ψ(stream + da),*p*(species*substrate) | 21 | 2435.144 |
| 11 | Ψ(stream + da),*p*(species*(depth + depth$^2$)) | 30 | 2320.952 |
| 12 | Ψ(stream + da),*p*(species*(velocity + velocity$^2$)) | 30 | 2284.216 |
| 13 | Ψ(stream + da),*p*(species*(depth + depth$^2$ + substrate)) | 39 | 2307.551 |
| 14 | Ψ(stream + da),*p*(species*(velocity + velocity$^2$ + substrate)) | 39 | 2285.108 |
| 15 | Ψ(stream + da),*p*(species*(depth + velocity) + depth$^2$ + velocity$^2$) | 32 | 2255.935 |
| 16 | Ψ(stream + da),*p*(species*(depth + velocity) + depth$^2$ + velocity$^2$ + channel unit) | 35 | 2244.591 |
| *16_trap | Ψ(stream + da),*p*(species*(depth + velocity) + depth$^2$ + velocity$^2$ + channel unit + trap) | 36 | 2235.262 |
| 17 | Ψ(stream + da),*p*(species*(depth + depth$^2$ + velocity + velocity$^2$)) | 48 | 2273.172 |
| 18 | Ψ(stream + da),*p*(species*(depth + velocity + substrate) + depth$^2$ + velocity$^2$ + channel unit) | 44 | 2245.685 |
| 19 | Ψ(stream + da),*p*(species*(depth + depth$^2$ + velocity + velocity$^2$) + channel unit) | 51 | 2262.708 |
| 20 | Ψ(stream + da),*p*(species*(depth + depth$^2$ + velocity + velocity$^2$ + substrate)) | 57 | 2275.278 |
| 21 | Ψ(stream + da),*p*(species*(depth + depth$^2$ + velocity + velocity$^2$ + substrate) + channel unit | 60 | 2268.949 |

Table 4. Sampling catch summary for nine species of darters from Lee Creek and Blackfork Creek in Oklahoma from the summer 2017 and spring 2018 field seasons.

| Scientific Name | Common Name | Blackfork Creek (Summer '17) | Lee Creek (Summer '17) | Lee Creek (Spring '18) |
|---|---|---|---|---|
| *Etheostoma blennioides* | Greenside Darter | 155 | 113 | 45 |
| *Etheostoma flabellare* | Fantail Darter | 247 | 213 | 154 |
| *Etheostoma mihileze* | Sunburst Darter | 0 | 29 | 19 |
| *Etheostoma spectabile* | Orangethroat Darter | 54 | 139 | 173 |
| *Etheostoma whipplei* | Redfin Darter | 101 | 66 | 68 |
| *Etheostoma zonale* | Banded Darter | 48 | 172 | 196 |
| *Percina caprodes* | Logperch | 21 | 18 | 26 |
| *Percina copelandi* | Channel Darter | 13 | 3 | 27 |
| *Percina nasuta* | Longnose Darter | 0 | 8 | 22 |
| | Total | 639 | 761 | 730 |

Table 5. A summary of the number of sites and surveys each darter species was detected in by season and stream.

| Season | Stream | Species | Sites w/ detection | Surveys w/ detection |
|---|---|---|---|---|
| Summer | Blackfork Creek | Banded Darter | 13/16 | 31/160 |
| Summer | Blackfork Creek | Channel Darter | 8/16 | 13/160 |
| Summer | Blackfork Creek | Fantail Darter | 16/16 | 70/160 |
| Summer | Blackfork Creek | Greenside Darter | 16/16 | 73/160 |
| Summer | Blackfork Creek | Logperch | 9/16 | 18/160 |
| Summer | Blackfork Creek | Orangethroat Darter | 13/16 | 35/160 |
| Summer | Blackfork Creek | Redfin Darter | 16/16 | 63/160 |
| Summer | Lee Creek | Banded Darter | 15/15 | 67/143 |
| Summer | Lee Creek | Channel Darter | 2/15 | 2/143 |
| Summer | Lee Creek | Fantail Darter | 13/15 | 42/143 |
| Summer | Lee Creek | Greenside Darter | 15/15 | 50/143 |
| Summer | Lee Creek | Logperch | 7/15 | 11/143 |
| Summer | Lee Creek | Longnose Darter | 4/15 | 5/143 |
| Summer | Lee Creek | Orangethroat Darter | 15/15 | 60/143 |
| Summer | Lee Creek | Redfin Darter | 14/15 | 45/143 |
| Summer | Lee Creek | Sunburst Darter | 8/15 | 19/143 |
| Spring | Lee Creek | Banded Darter | 12/12 | 73/116 |
| Spring | Lee Creek | Channel Darter | 5/12 | 12/116 |
| Spring | Lee Creek | Fantail Darter | 12/12 | 57/116 |
| Spring | Lee Creek | Greenside Darter | 10/12 | 33/116 |
| Spring | Lee Creek | Logperch | 7/12 | 15/116 |
| Spring | Lee Creek | Longnose Darter | 8/12 | 12/116 |
| Spring | Lee Creek | Orangethroat Darter | 12/12 | 68/116 |
| Spring | Lee Creek | Redfin Darter | 12/12 | 45/116 |
| Spring | Lee Creek | Sunburst Darter | 7/12 | 8/116 |

Table 6. Summary statistics for the detection covariates during the summer 2017 field season in Lee Creek and Blackfork Creek and the spring 2018 field season in Lee Creek.

| Season | Stream | Parameter | Mean | SD | Range |
|---|---|---|---|---|---|
| Summer 2017 | Lee Creek | Water Depth (cm) | 35.07 | 14.45 | 7.5 - 82 |
| | | Water Velocity (m/s) | 0.23 | 0.26 | 0.0 - 1.25 |
| | | Proportion of cobble substrate (% >64 mm) | 58.02 | 23.56 | 0 - 100 |
| Summer 2017 | Blackfork Creek | Water Depth (cm) | 42.79 | 15.67 | 11- 83.5 |
| | | Water Velocity (m/s) | 0.10 | 0.17 | 0 - 0.9 |
| | | Proportion of cobble substrate (% >64 mm) | 68.60 | 19.25 | 10 - 100 |
| Spring 2018 | Lee Creek | Water Depth (cm) | 38.35 | 15.36 | 11.5 - 87 |
| | | Water Velocity (m/s) | 0.35 | 0.31 | 0.0 - 1.50 |
| | | Proportion of cobble substrate (% >64 mm) | 49.27 | 21.17 | 0 - 100 |

Table 7. Comparison of null detection probabilities estimates for nine species of darter from Lee Creek and Blackfork Creek between seasons. *p* is detection probability. n is the number of individuals detected.

| | Summer 2017 | | Spring 2018 | | |
|---|---|---|---|---|---|
| Species | *p* | n | *p* | n | % Change (Summer - Spring) |
| Greenside Darter | 0.41 | 268 | 0.34 | 45 | -17 |
| Fantail Darter | 0.38 | 460 | 0.49 | 154 | 29 |
| Redfin Darter | 0.36 | 167 | 0.39 | 68 | 8 |
| Banded Darter | 0.35 | 220 | 0.63 | 196 | 80 |
| Orangethroat Darter | 0.31 | 193 | 0.59 | 173 | 90 |
| Sunburst Darter | 0.22 | 29 | 0.07 | 19 | -68 |
| Logperch | 0.14 | 39 | 0.19 | 26 | 36 |
| Channel Darter | 0.10 | 16 | 0.22 | 27 | 120 |
| Longnose Darter | 0.05 | 8 | 0.10 | 22 | 100 |

Table 8. Correlation matrix for the continuous variables used in the candidate detection probability models. A threshold of Pearson's |r| >0.7 was used for highly correlated variables.

| Season | | Average water velocity | Average water depth | Average proportion cobble substrate |
|---|---|---|---|---|
| Summer 2017 | Average water velocity | 1.00 | -0.66 | -0.34 |
| | Average water depth | -0.66 | 1.00 | 0.20 |
| | Average proportion cobble substrate | -0.34 | 0.20 | 1.00 |
| Spring 2018 | Average water velocity | 1.00 | -0.43 | -0.22 |
| | Average water depth | -0.43 | 1.00 | 0.21 |
| | Average proportion cobble substrate | -0.22 | 0.21 | 1.00 |

Table 9. Model summary showing coefficients, standard error (SE) and 95% confidence intervals (CI) from the top model for the nine darter species in Blackfork Creek and Lee Creek in the summer of 2017.

| Parameter | Coefficient ± SE | 95% CI |
|---|---|---|
| Intercept | -0.58 ± 0.29 | -1.15, -0.01 |
| Water Depth | -0.38 ± 0.17 | -0.71, -0.05 |
| Water Velocity | 0.79 ± 0.20 | 0.40, 1.18 |
| Water Depth$^2$ | -0.09 ± 0.05 | -0.19, 0.01 |
| Water Velocity$^2$ | -0.06 ± 0.16 | -0.37, 0.25 |
| Pool CU | -0.81 ± 0.24 | -1.28, -0.34 |
| Riffle CU | 0.10 ± 0.24 | -0.37, 0.57 |
| Run CU | -0.42 ± 0.2 | -0.81, -0.03 |
| Trap | 0.42 ± 0.12 | 0.18, 0.66 |
| Channel Darter | -2.05 ± 0.37 | -2.78, -1.32 |
| Fantail Darter | 0.37 ± 0.22 | -0.06, 0.80 |
| Greenside Darter | 0.53 ± 0.21 | 0.12, 0.94 |
| Logperch | -1.36 ± 0.30 | -1.95, -0.77 |
| Longnose Darter | -2.81 ± 0.74 | -4.26, -1.36 |
| Orangethroat Darter | 0.26 ± 0.22 | -0.17, 0.69 |
| Redfin Darter | 0.44 ± 0.21 | 0.03, 0.85 |
| Sunburst Darter | -0.81 ± 0.33 | -1.46, -0.16 |
| Channel Darter, depth | 0.84 ± 0.44 | -0.02, 1.70 |
| Fantail Darter, depth | -0.42 ± 0.24 | -0.89, 0.05 |
| Greenside Darter, depth | 0.19 ± 0.22 | -0.24, 0.62 |
| Logperch, depth | 0.74 ± 0.34 | 0.07, 1.41 |
| Longnose Darter, depth | 1.75 ± 0.77 | 0.24, 3.26 |
| Orangethroat Darter, depth | -0.20 ± 0.23 | -0.65, 0.25 |
| Redfin Darter, depth | 0.03 ± 0.22 | -0.40, 0.46 |
| Sunburst Darter, depth | 0.68 ± 0.34 | 0.01, 1.35 |
| Channel Darter, velocity | -1.43 ± 0.39 | -2.19, -0.67 |
| Fantail Darter, velocity | -0.61 ± 0.25 | -1.10, -0.12 |
| Greenside Darter, velocity | -0.36 ± 0.25 | -0.85, 0.13 |
| Logperch, velocity | -1.57 ± 0.32 | -2.20, -0.94 |
| Longnose Darter, velocity | -1.03 ± 0.52 | -2.05, -0.01 |
| Orangethroat Darter, velocity | -1.19 ± 0.25 | -1.68, -0.70 |
| Redfin Darter, velocity | -1.31 ± 0.24 | -1.78, -0.84 |
| Sunburst Darter, velocity | -1.23 ± 0.34 | -1.90, -0.56 |

Table 10. Model summary showing coefficients, standard error (SE) and 95% confidence intervals (CI) from the top model for the nine darter species in Lee Creek in the spring of 2018.

| Parameter | Coefficient ± SE | 95% CI |
|---|---|---|
| Intercept | -0.29 ± 0.41 | -1.09, 0.51 |
| Water Velocity | 1.95 ± 0.67 | 0.64, 3.26 |
| Water Velocity$^2$ | 0.61 ± 0.39 | -0.15, 1.37 |
| Trap | 0.52 ± 0.18 | 0.17, 0.87 |
| Channel Darter | -1.49 ± 0.74 | -2.94, -0.04 |
| Fantail Darter | -1.83 ± 0.66 | -3.12, -0.54 |
| Greenside Darter | -1.87 ± 0.65 | -3.14, -0.60 |
| Logperch | -0.66 ± 0.61 | -1.86, 0.54 |
| Longnose Darter | -1.61 ± 0.69 | -2.96, -0.26 |
| Orangethroat Darter | 0.47 ± 0.55 | -0.61, 1.55 |
| Redfin Darter | 0.07 ± 0.55 | -1.01, 1.15 |
| Sunburst Darter | -1.24 ± 0.68 | -2.57, 0.09 |
| Channel Darter, velocity | 0.17 ± 3.20 | -6.10, 6.44 |
| Fantail Darter, velocity | 2.35 ± 1.13 | 0.14, 4.56 |
| Greenside Darter, velocity | 0.13 ± 0.96 | -1.75, 2.01 |
| Logperch, velocity | -4.35 ± 1.17 | -6.64, -2.06 |
| Longnose Darter, velocity | -2.25 ± 1.10 | -4.41, -0.09 |
| Orangethroat Darter, velocity | -1.56 ± 0.88 | -3.28, 0.16 |
| Redfin Darter, velocity | -2.72 ± 0.89 | -4.46, -0.98 |
| Sunburst Darter, velocity | -4.35 ± 1.39 | -7.07, -1.63 |
| Channel Darter, velocity$^2$ | -5.57 ± 4.40 | -14.19, 3.05 |
| Fantail Darter, velocity$^2$ | 1.16 ± 0.66 | -0.13, 2.45 |
| Greenside Darter, velocity$^2$ | 0.32 ± 0.57 | -0.80, 1.44 |
| Logperch, velocity$^2$ | -1.87 ± 0.65 | -3.14, -0.60 |
| Longnose Darter, velocity$^2$ | -0.98 ± 0.65 | -2.25, 0.29 |
| Orangethroat Darter, velocity$^2$ | -0.71 ± 0.51 | -1.71, 0.29 |
| Redfin Darter, velocity$^2$ | -1.03 ± 0.51 | -2.03, -0.03 |
| Sunburst Darter, velocity$^2$ | -2.02 ± 0.76 | -3.51, -0.53 |

Table 11. Coordinates of Longnose Darters detected while sampling Lee Creek in the summer and spring field seasons.

| Season | Stream | Latitude | Longitude |
|---|---|---|---|
| Summer | Lee Creek | 35.587703 | -94.489627 |
| Summer | Lee Creek | 35.570027 | -94.529561 |
| Summer | Lee Creek | 35.532428 | -94.493939 |
| Summer | Lee Creek | 35.522764 | -94.474328 |
| Spring | Lee Creek | 35.612079 | -94.487881 |
| Spring | Lee Creek | 35.590276 | -94.484929 |
| Spring | Lee Creek | 35.576803 | -94.526946 |
| Spring | Lee Creek | 35.561575 | -94.532158 |
| Spring | Lee Creek | 35.532428 | -94.493939 |
| Spring | Lee Creek | 35.529938 | -94.493869 |
| Spring | Lee Creek | 35.519613 | -94.482183 |
| Spring | Lee Creek | 35.522764 | -94.474328 |

**APPENDIX D: Figures for Chapter 2**

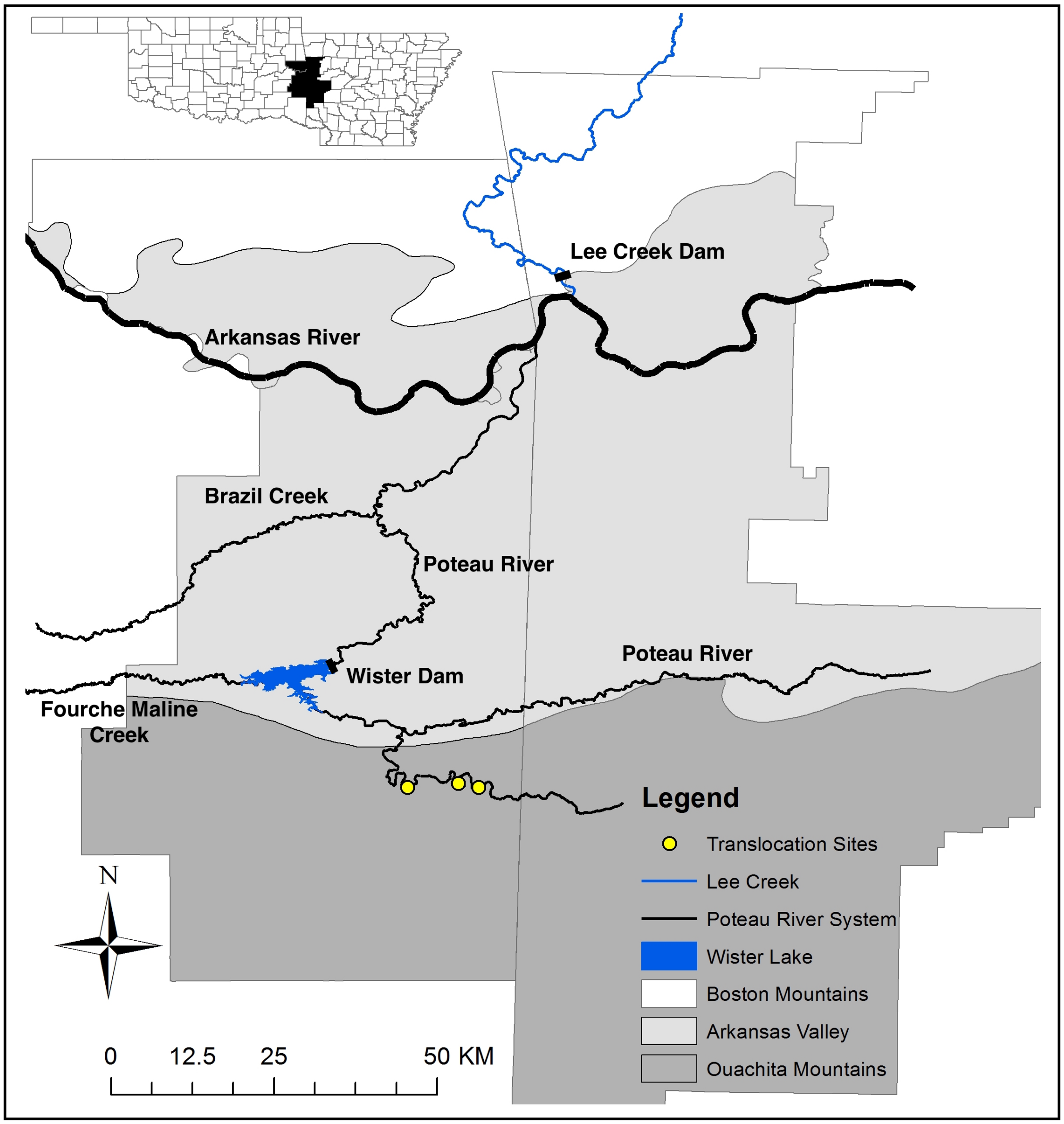


Figure 1. Map of Lee Creek and the Poteau River system in eastern Oklahoma and western Arkansas shown with ecoregions and Longnose Darter translocation sites in Blackfork Creek.

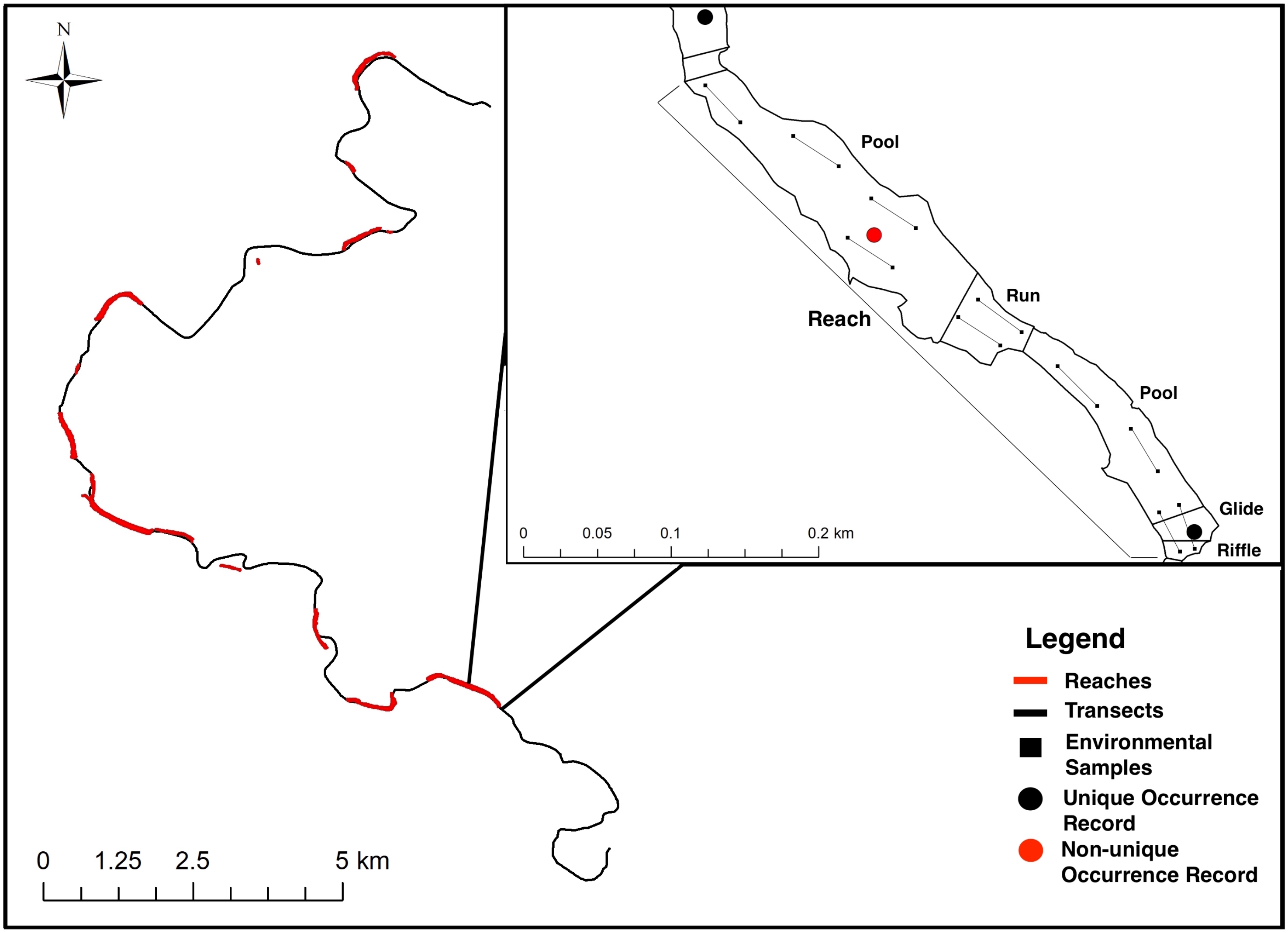


Figure 2. Sampling sites (16 stream reaches) in Lee Creek in Oklahoma shown with example surveys.

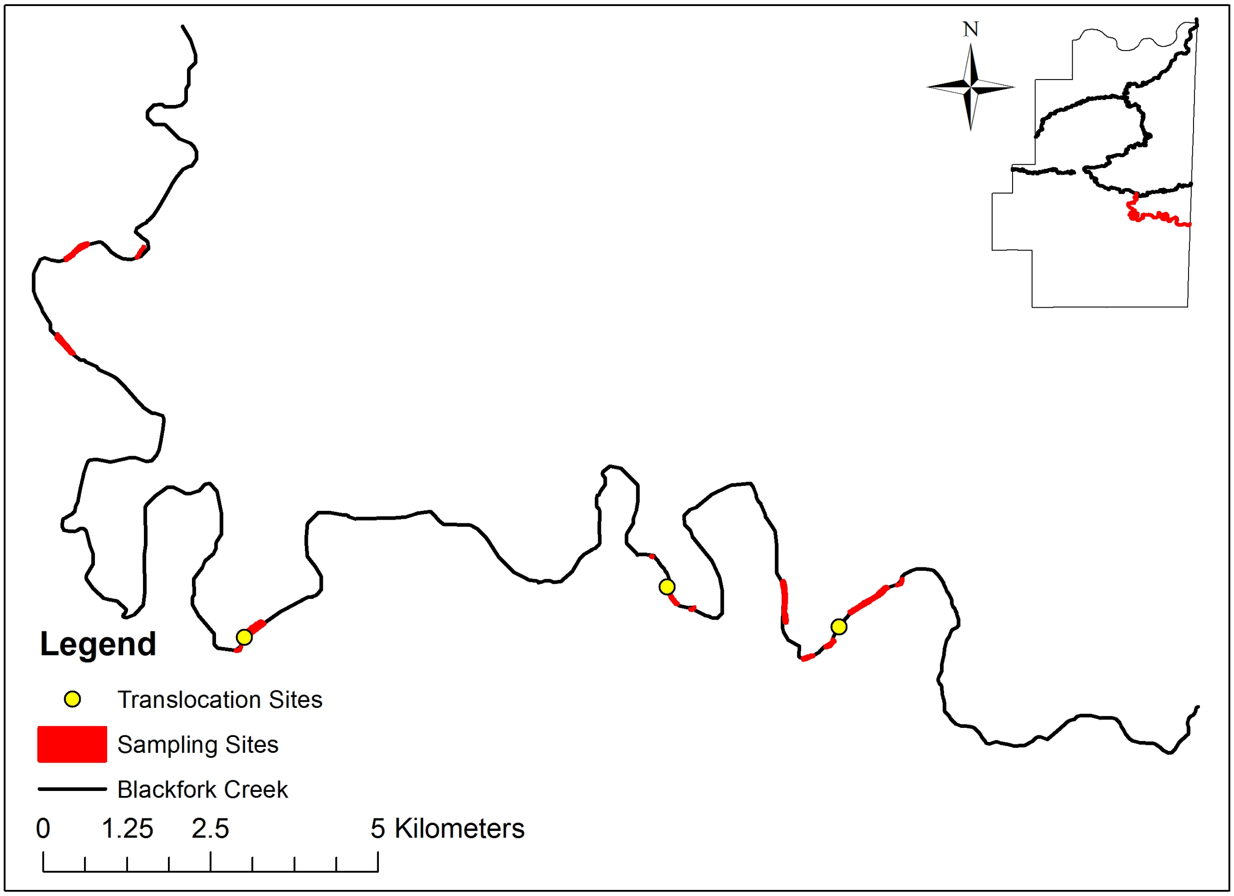


Figure 3. Sampling sites (16 stream reaches) in Blackfork Creek in Oklahoma shown with 1991-92 Longnose Darter translocation sites.

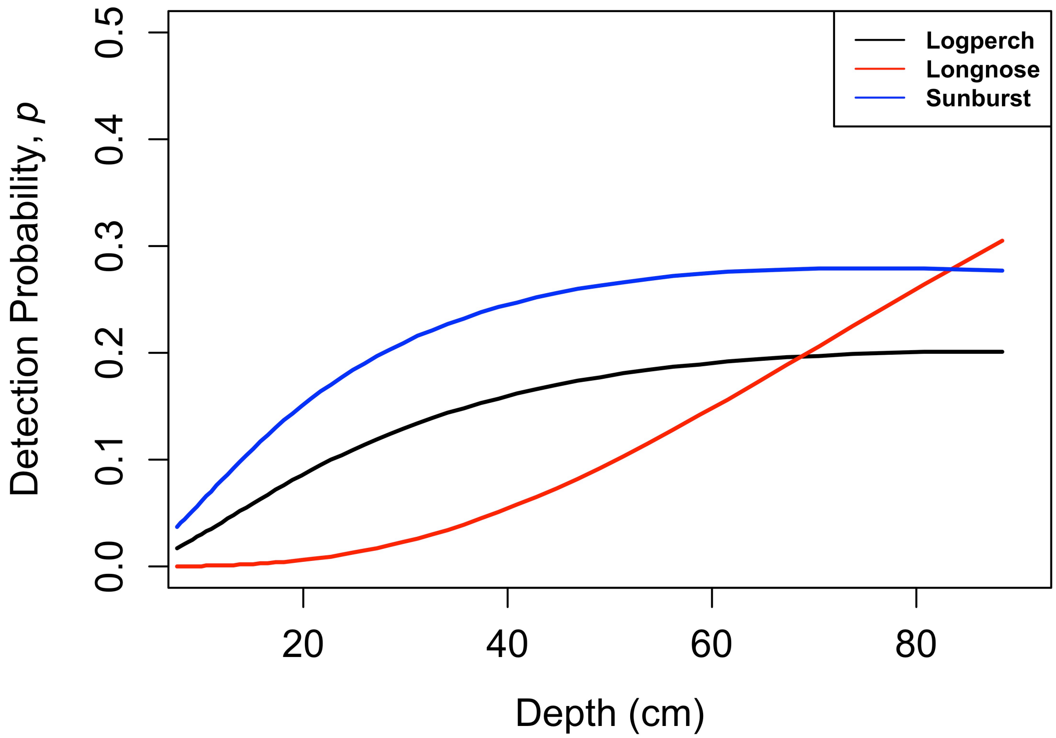


Figure 4. Detection probabilities by water depth for Logperch, Longnose Darter, and Sunburst Darter from the top model in the summer 2017 field season. The y-axis is scaled to 0.5 detection probability.

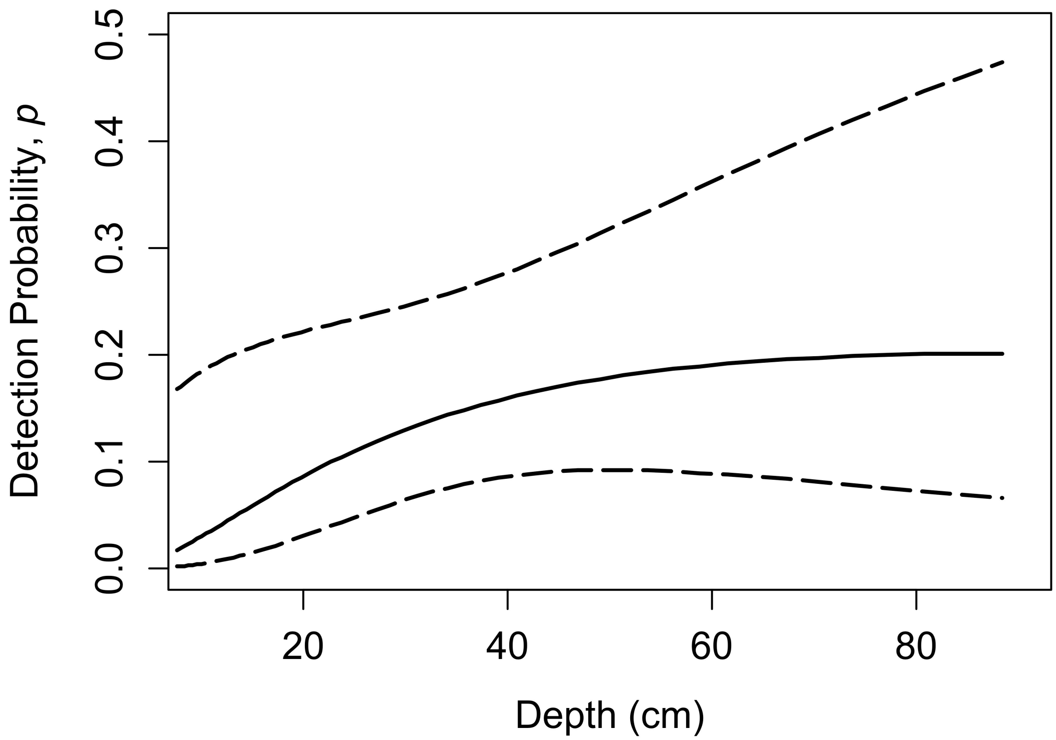


Figure 5. Detection probabilities by water for Logperch from the top model in the summer 2017 field season. Dashed lines represent the 95% confidence interval. The y-axis is scaled to 0.5 detection probability.

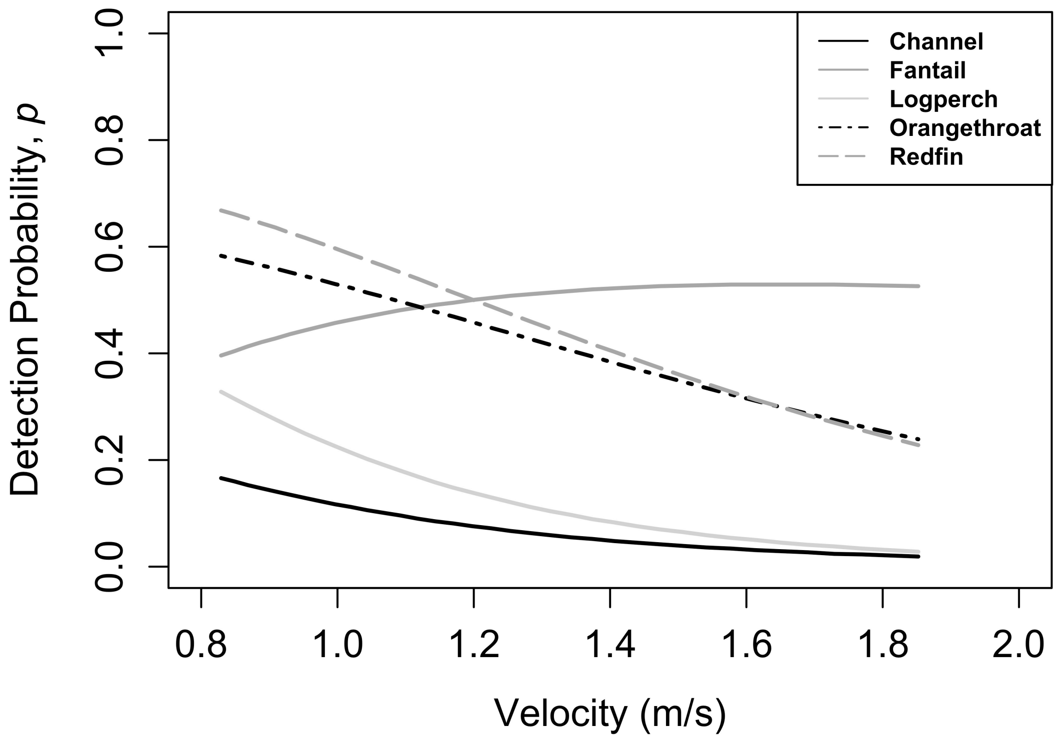


Figure 6. Detection probabilities by water velocity for the darter community in Blackfork Creek from the top model in the summer 2017 field season.

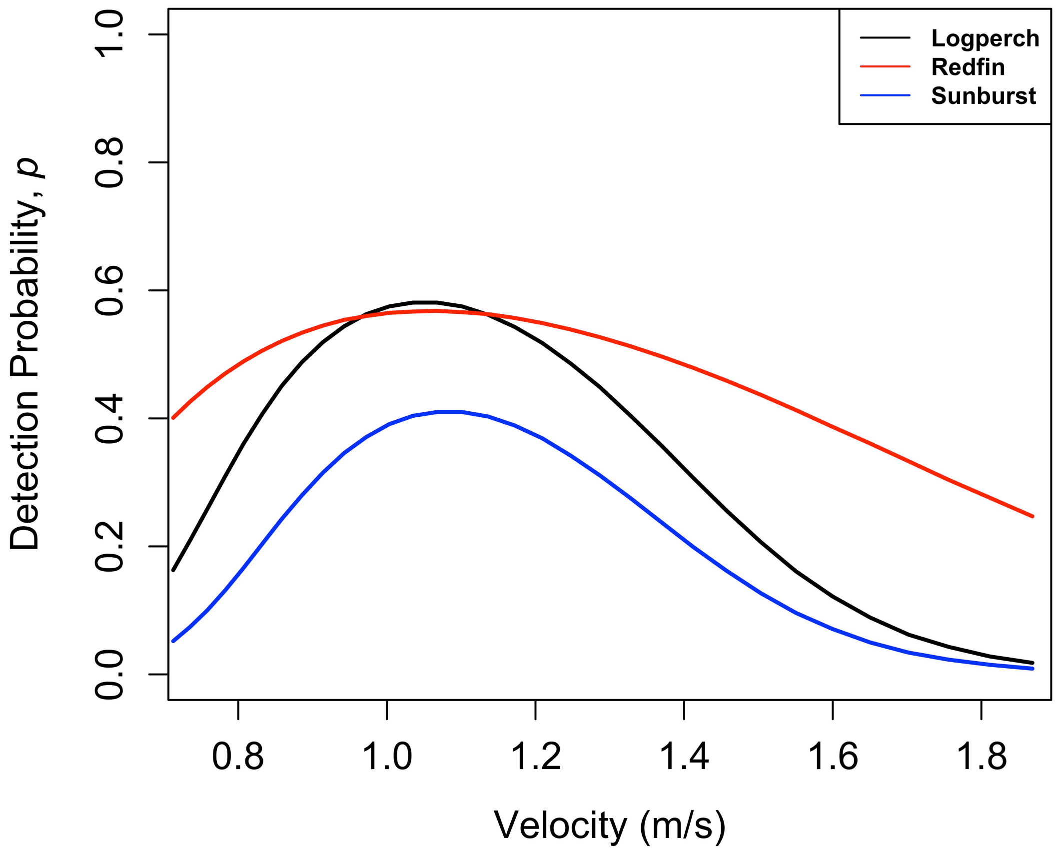


Figure 7. Detection probabilities by water velocity for Logperch, Redfin Darter, and Sunburst Darter in Lee Creek, OK from the top model of the spring 2018 field season.

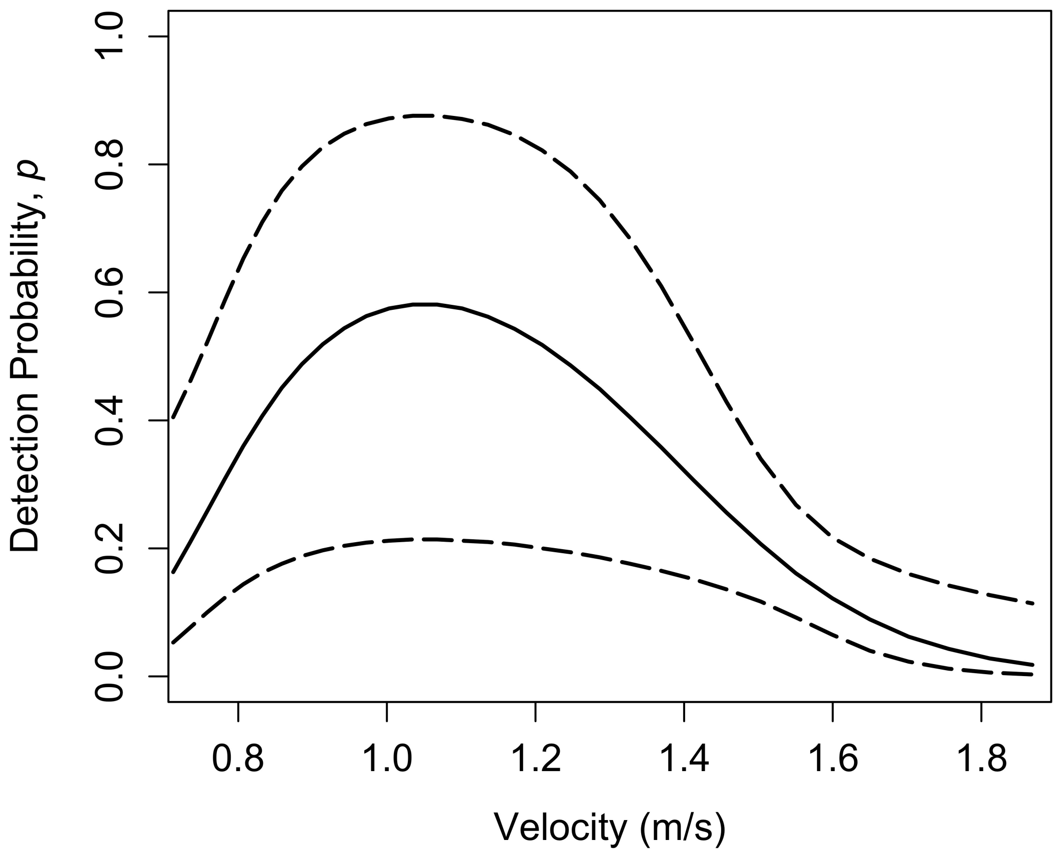


Figure 8. Detection probability by water velocity for Logperch in Lee Creek, OK from the top model of the spring 2018 field season. Dashed lines represent the 95% confidence interval.

VITA

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