

MULTI-SCALE RESOURCE SELECTION OF
NIANGUA DARTERS IN THE OSAGE RIVER BASIN, MISSOURI

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**Multi-scale resource selection of Niangua darters in the Osage River basin,
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TABLE OF CONTENTS

ACKNOWLEDGMENTS	ii
LIST OF TABLES	v
LIST OF FIGURES	vi
ABSTRACT.....	vii
DESCRIPTION OF CHAPTERS	x
CHAPTER 1: FACTORS AFFECTING THE SPATIAL DISTRIBUTION OF NIANGUA DARTERS IN THE OSAGE RIVER BASIN, MISSOURI	1
ABSTRACT.....	1
INTRODUCTION	2
METHODS	5
Study Area	5
Fish Sampling	6
Resource Sampling	7
Analysis.....	10
RESULTS	12
DISCUSSION	14
LITERATURE CITED	22
CHAPTER 2: SEASONAL MICROHABITAT SELECTION OF NIANGUA DARTERS	39
ABSTRACT.....	39
INTRODUCTION	40

METHODS	42
Study Area	42
Fish Sampling	43
Microhabitat Variables.....	44
Analysis.....	45
RESULTS	48
DISCUSSION.....	50
LITERATURE CITED	57

LIST OF TABLES

CHAPTER 1

Table 1: Parameters used in Niangua darter resource selection models at the macrohabitat-, reach-, and watershed- scale in sampled streams of the Osage River Basin, Missouri, 2010-2011.....	31
Table 2: Number of parameters (K), AICc (corrected for small samples size), change in AIC _C , and Akaike weight for models hypothesized to influence resource selection of Niangua darters at the macrohabitat-, reach-, and watershed-scale (see Table 1) in sampled streams of the Osage River Basin, Missouri, 2010-2011.....	33
Table 3: Coefficients and standard error for parameters contained in top-ranked models at the macrohabitat-, reach-, and watershed-scale where Niangua darters were detected or not detected within sampled streams of the Osage River Basin, Missouri, 2010-2011... ..	34
Table 4: Mean, standard error (SE), and range for continuous variables at the macrohabitat-, reach-, and watershed-scale (see Table 1) where Niangua darters were detected or not detected within sampled streams of the Osage River Basin, Missouri, 2010-2011.....	35

Table 5: Number of parameters (K), AICc (corrected for small samples size), change in AIC _C , and Akaike weight for models predicting the presence of Niangua darters within reaches where we obtained one year of water temperature data (n=31), within the Osage River Basin, Missouri, 2010-2011.....	36
---	----

CHAPTER 2

Table 1: Microhabitat parameters measured at used and available Niangua darter locations in studied reaches of the Little Niangua River and Starks Creek, Missouri, 2011-2012.	63
---	----

Table 2: A priori resource selection models for adult Niangua darters in the Little Niangua River and Starks Creek, Missouri, 2011-2012.	64
---	----

Table 3: Mean and standard error for depth (cm) and velocity (m s ⁻¹) at 60% depth of random available locations within studied reaches of the Little Niangua River and Starks Creek, Missouri, 2011-2012.	65
---	----

Table 4: AICc ranking of models hypothesized to influence microhabitat selection of adult Niangua darters in studied reaches of the Little Niangua River and Starks Creek, Missouri, 2011-2012.	66
--	----

Table 5: Coefficients and standard error for parameters contained in top-ranked models of adult Niangua darters selection among seasons within studied reaches of the Little Niangua River and Starks Creek, Missouri, 2011-2012. Category E4 (76-100%) and Deep Pool categories are excluded as the reference.	67
--	----

LIST OF FIGURES

CHAPTER 1

Figure 1: Sampled and unsampled reaches within the Osage Ecological Drainage Unit, Missouri, 2010-2011. Black triangles indicate randomly selected reaches where Niangua darters were detected and white triangles where they were not detected. Reaches sampled by the Missouri Department of Conservation are indicated by grey triangles. Unsamped reaches are depicted in black (dry streambed), grey (denied access), or white circles (underwater visibility <1 m). The transparent grey polygons indicate the current range of the Niangua darter.....	37
Figure 2: Relative probability of occurrence of Niangua darters based on macrohabitat depth in sampled streams of the Osage Ecological Drainage Unit, Missouri, 2010-2011.	38

CHAPTER 2

Figure 1: Number of locations collected for (a) adult and (b) juvenile Niangua darter during bi-monthly samples within studied reaches of the Little Niangua River and Starks Creek, Missouri, 2011-2012.	68
Figure 2: Mean and standard error for depth (cm) at locations used by adult Niangua darters and randomly selected locations that represent availability within studied reaches of (a) the Little Niangua River and (b) Starks Creek, Missouri, 2011-2012.	69
Figure 3: Resource selection functions of top-ranked models indicating the relative probability of selection among seasons: spring (top), summer (middle), and fall (bottom), across the range of depths used and available to adult Niangua darters in studied reaches of the (left) Little Niangua River and (right) Starks Creek, Missouri, 2011-2012.	70
Figure 4: Number of locations used by adult Niangua darter used and the number of randomly selected locations that represent availability for each habitat type during fall, within 500 m reach of Starks Creek, Missouri, 2011-2012.	71

**MULTI-SCALE RESOURCE SELECTION OF
NIANGUA DARTERS IN THE OSAGE RIVER BASIN, MISSOURI**

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ABSTRACT

Understanding temporal and spatial habitat relationships is important to the recovery of the federally threatened Niangua darter *Etheostoma nianguae*, and other imperiled freshwater fishes. We evaluated resource selection of Niangua darters at four spatial scales within the Osage River Basin in Missouri. Resource selection of Niangua darters was determined at the macrohabitat-, reach-, and watershed-scale in third to sixth order streams within the Osage Ecological Drainage Unit (EDU) in Missouri during summer in 2010 and 2011. Seventy-four reaches containing 651 macrohabitats were sampled using snorkeling from mid-June through August. We examined support for logistic regression models containing variables we believe to influence the presence of Niangua darters at each scale. At the macrohabitat- and reach-scale, depth and velocity had the greatest influence on Niangua darter presence. Niangua darters occupied macrohabitats with depths of 41.4 cm and velocities of 0.20 m s^{-1} and reaches with depths of 38.5 cm and water velocities of 0.26 m s^{-1} . Depth and velocity selection for lotic species is likely a tradeoff among numerous factors, and for Niangua darters may largely be determined by energetics and biotic interaction. At the watershed-scale, soil and geology characteristics had the greatest influence on the presence of Niangua darters. Streams originating from watersheds dominated by: limestone and dolomite bedrock geology, soils with relatively greater infiltration rates and lower transmission rates, and greater relief have the greatest likelihood of containing Niangua darters. We believe that this model is indicative of

watershed characteristic which create stable, coarse, silt-free instream habitats that are favorable for Niangua darters. Our findings suggest that currently anthropogenic land use does not play a dominant role in the presence of Niangua darters however water temperature does. Predicted increases in water temperature in the Ozarks of Missouri of 2-4 °C is likely to have negative effects on populations of Niangua darters, unless thermal refugia is available.

We also evaluated microhabitat-scale selection of adult Niangua darters among seasons in two Missouri Ozark streams. Niangua darters were located every other month from July 2011 to May 2012 to determine selection related to six instream habitat variables believed to be important to Niangua darter. We collected 196 locations which we fit to univariate discrete choice models. Overall Niangua darters selected relatively shallower depths during summer compared to spring and fall. Substrate embeddedness and habitat type also fell within the 90% confidence set of candidate models along with depth in spring and fall, respectively, however results differed between streams. Adult Niangua darters in Little Niangua River selected locations with less embedded substrate during spring but in Starks Creek showed no selection for substrate embeddedness. During fall in Starks Creek, adult Niangua darters selected deep runs and pools, the latter having stronger selection, but used all habitats in proportion to their availability in the Little Niangua River. We were unable to detect Niangua darters during winter as they may be using subsurface refugia or migrating outside of sample reaches, the latter being less likely due to their sedentary nature. We collected few juvenile Niangua darters, but the ones collected used habitats with shallower depths and slower velocities consisting of fine or gravel substrates that were greater than 50% embedded.

Across spatial and temporal scales, our results suggest that populations of Niangua darters may benefit from watershed- and reach-scale management that increase habitat diversity (depth and velocity) and buffer against increases in water temperature are likely to benefit population of Niangua darters. For example, reducing soil erosion and increasing/maintaining riparian corridors can: reduce inputs of finer substrate particles into streams, limit stream bank erosion, increase habitat diversity, increase substrate stability, and buffer stream water temperatures. These findings will aid in recovery efforts by helping managers to better define and evaluate conservation criteria, leading to a better informed recovery effort.

DESCRIPTION OF CHAPTERS

The follow chapters were written as independent manuscripts for submission into peer-reviewed journals. Subsequently, some material is repeated across chapters and each chapter contains an independent literature cited.

CHAPTER 1: FACTORS AFFECTING THE SPATIAL DISTRIBUTION OF NIANGUA DARTERS IN THE OSAGE RIVER BASIN, MISSOURI

ABSTRACT

Understanding spatial habitat relationships is important to the recovery of the federally threatened Niangua darter *Etheostoma nianguae*, and other imperiled freshwater fishes. Resource selection of Niangua darters was determined at the macrohabitat-, reach-, and watershed-scale in third to sixth order streams within the Osage Ecological Drainage Unit (EDU) in Missouri 2010-2011. Seventy-four reaches containing 651 macrohabitats were sampled using snorkeling from mid-June through August. We examined support for logistic regression models containing variables we believe to influence the presence of Niangua darters at each scale. At the macrohabitat- and reach-scale, depth and velocity had the greatest influence on Niangua darter presence. Niangua darters occupied macrohabitats with depths of 41.4 cm and velocities of 0.20 m s^{-1} and reaches with depths of 38.5 cm and water velocities of 0.26 m s^{-1} . Depth and velocity selection for lotic species is likely a tradeoff among numerous factors, and for Niangua darters may largely be determined by energetics and biotic interaction. At the watershed-scale, soil and geology characteristics had the greatest influence on Niangua darter presence. Streams originating from watersheds dominated by limestone and dolomite bedrock geology, soils with relatively greater infiltration rates and lower transmission rates, and greater relief, have the greatest likelihood of containing Niangua darters. We believe that this model is indicative of watershed characteristic which create stable, coarse, silt-free instream habitats that are favorable for Niangua darters. Our findings suggest that currently anthropogenic land use does not play a dominant role in the presence of Niangua darters.

however water temperature does. Predicted increases in water temperature in the Ozarks of Missouri of 2-4 °C is likely to have negative effects on populations of Niangua darters, unless thermal refugia is available. Watershed- and reach-scale management that increase habitat diversity (depth and velocity) and buffer against increases in water temperature are likely to benefit population of Niangua darters. For example, reducing soil erosion and increasing/maintaining riparian corridors can: reduce inputs of finer substrate particles into streams, limit stream bank erosion, increase habitat diversity, increase substrate stability, and buffer stream water temperatures. These findings will aid in recovery efforts by helping managers to better define and evaluate conservation criteria, leading to a better informed recovery effort. Significant improvement to this research would include; a larger sample, especially presence sites, accounting for the potential of imperfect detection, and linking population vital rates to selected resources.

INTRODUCTION

Lotic systems are characterized by the surrounding environment at multiple spatial scales (Ross, 1963; Hynes, 1975; Vannote et al., 1980; Fausch et al., 2002). The geology, vegetation, and climate of a watershed determine the morphology, hydrology, water chemistry, and biological community within the stream (Frissell et al., 1986; Poff et al., 1997; Knighton, 1998; Dauwalter et al., 2010). Changes across the landscape, both natural and anthropogenic, affect the quantity and quality of stream habitat, resulting in changes in distribution and abundance of many species (Schlosser, 1991; Allan & Johnson, 1997; Poff et al., 2002; Hitt & Angermeier, 2008). For example, agricultural development of a landscape has shown to increase erosion and runoff into streams, leading to decreased water quality, altered hydrology, and decreased biological diversity

(Cooper, 1993; Johnson et al., 1997; Allan, 2004; Dudgeon et al., 2006). Furthermore, warmer water temperatures, due to climate change, have the potential to greatly reduce and fragment thermal habitat thus influencing the abundance and distribution of some species (Eaton & Scheller, 1996; Xenopoulos et al. 2005, Ficke et al. 2007; Pease & Paukert, 2014). For Missouri's Ozarks, the predicted change in climate has been estimated to increase water temperatures by 2-4 °C by 2060 (The Nature Conservancy, 2009; Pease & Paukert, 2014).

When fish encounter unfavorable changes in habitat, species have two options, adapt to the new conditions or migrate to more suitable habitat. Given the rate and scale of anthropogenic change, species may not be able to adapt or acclimate (Peters, 1989; Lammert & Allan, 1999; Poff et al., 2002). For species with narrower tolerances and/or distributions, like some darter species, migration may also be hindered by existing biological or geographical barriers (Eaton & Scheller, 1996; Keleher & Rahel, 1996; Poff et al., 2002; Palmer et al. 2009). In either case, understanding factors that influence the distribution of suitable habitat for threatened and endangered species, such as the Niangua darter, will be crucial for directing future habitat management and/or translocation in response to land conversion and climate change (Griffith et al., 1989; Schlosser & Angermeier, 1995; Compton & Taylor; 2013).

The Niangua darter, *Etheostoma nianguae* Gilbert & Meek, is a relatively large, slender darter (6-9 cm total length) that is endemic to north-flowing streams of the Osage River Basin in south-central Missouri, U.S.A. (Pflieger, 1978). Since its discovery in 1884, ten populations have been documented and two extirpated, with additional populations likely extirpated prior to sufficient documentation of the species' distribution

(USFWS, 1989). In 1985 the Niangua darter was listed as federally threatened due to low densities, limited distribution, and numerous threats: reservoir construction, destruction of stream habitat, degrading water quality, and introduction of non-native species (USFWS, 1989; Pflieger, 1997). Currently eight of ten known populations remain, but detecting trends in abundance and distribution over time has been difficult (Figure 1; Novinger & Decoske, 2013).

Previous research has examined resource selection of Niangua darters (Mattingly & Galat, 2002; Sowa et al., 2007). However neither study examined the influence of land use and water temperature which may, like many other fish, play a pivotal role in understanding the distribution of Niangua darters (Allan et al., 1997; Johnson et al., 1997; Allan, 2004). Sowa et al. (2007) created models to predict the potential distributions of hundreds of fish, crayfish, and mussels throughout the state, using historical occurrence records and the same set of predictor variables for all species. The methods used were appropriate for predicting distributions of hundreds of species using limited habitat variables. However, these methods may not be completely applicable for understanding the distribution of a rare species with limited occurrence records. Last, Mattingly & Galat (2002) examined a single stream, the Little Niangua River, which makes up only 15% of the total range of the species. Therefore extending conclusion across the range of the Niangua darter from these limited studies may be problematic (Minshall 1988; Schlosser & Angermeier 1995; Leftwich et al. 1997; Millspaugh et al. 2009).

The objective of our research was to determine in-stream and landscape factors, at multiple spatial scales, affecting the distribution of Niangua darters throughout its range. Adopting a hierarchical framework with multiple spatial scales has shown to yield a more

robust understanding of how changes in the environment affect the abundance and distribution of fishes (Frissell et al., 1986; Sowa & Rabeni, 1995; Rabeni & Sowa, 1996; Angermeier & Winston, 1998; Kirsch & Peterson, 2014). To achieve this objective we examined the resource selection of Niangua darters in relation to macrohabitat, reach-, and watershed-scale metrics we believed to influence their presence. This approach will not only allow us to determine which variables were most important but also which scale is most influential on the presence of Niangua darters.

This information, along with previous research, will allow managers to proactively protect and restore current and future habitat which will hopefully increase the resilience of these systems and increase the probability of persistence of the species. This information will also provide crucial information for locating suitable habitat which may support undiscovered Niangua darter populations or be potential locations for future translocation. The application of our study will allow managers to develop actions that cover a larger proportion of the species' habitat and have greater probability of directly impacting populations, particularly as changes in climate and land use continue to alter freshwater systems (Strakosh et al., 2003).

METHODS

Study Area

Our study occurred in the Osage Ecological Drainage Unit (EDU; Figure 1) in the Osage River Basin, Missouri, which is the only EDU where Niangua darters occur. Ecological Drainage Units represent zoogeographic subunits within freshwater systems which are evolutionarily isolated and contain similar taxonomic composition (Sowa et al. 2007).

The Osage EDU encompasses approximately 380 km of the Osage River, from the mouth

of the Sac River to the confluence with the Missouri River draining roughly 17,500 km². The Osage EDU contains four major reservoirs constructed for flood control and power generation: Harry S. Truman, Lake of the Ozarks, Stockton, and Pomme de Terre (<http://extra.mdc.mo.gov/fish/watershed/wosage/> contents/). This region is characterized by highly dissected rolling hills dominated by oak-hickory forests underlain by carbonate bedrock (Nigh & Schroeder, 2002). Many streams are spring-fed, carrying very little suspended solids, and channel substrates that are dominated by chert (Rabeni & Jacobson, 1993). The land cover in this region consists of 51.3 % forest/woodland, 39.3 % crop/grassland, and 5.5% urban land use (Fry et al., 2011)

Fish Sampling

We systematically sampled third to sixth order streams within the Osage EDU of Missouri, to determine resource selection of Niangua darters at each of the three spatial scales (Strahler, 1957; Sowa et al., 2007). We focused sampling effort to only third to sixth order streams because Niangua darters only occur in this range of stream sizes (Pflieger, 1978; Mattingly & Galat, 2002). Initially 50 reaches were selected using a Generalized Random Tessellation Stratified design on the 1:24,000-scale National Hydrography Dataset (NHD; U.S. Geological Survey, 2009). This design yields an unbiased and spatially balanced sample (Stevens & Olsen, 2004). Reaches were approximately 300 m in length and did not span tributaries to avoid drastic changes in physical habitat (Frissel et. al, 1986). Reaches within 500 m of low water crossing were excluded, as these structures impact local stream habitat and thus have the potential to influence our results (Jackson 2003; Gibson et al. 2005). In addition, reaches in ownership where we were denied access or those in which water clarity was unacceptable

for snorkeling (i.e., lateral Secchi disk visibility <1 m), were replaced by reaches drawn from an oversample, created using the same methods as the initial sample (Figure 1). If underwater, lateral visibility was less than 1 m we returned on two subsequent dates to reassess water clarity. If conditions did not improve the reach was excluded.

Selected stream reaches were randomly sampled using snorkeling from mid-June through August in 2011 when river stage and subsequent water clarity permitted.

Snorkeling was selected because it is the most efficient and least invasive method for sampling Niangua darters (Mattingly & Galat, 2002). At each reach two observers snorkeled between the bank and the midline of the stream, while moving upstream in a zigzag progression at approximately $2\text{-}3 \text{ m min}^{-1}$. Samples were completed between one hour after sunrise and one hour before sunset. Deep pool habitats, with depths greater than one meter, were not sampled in their entirety because Niangua darters have not been found to occupy depths greater than one meter during summer months (Novinger & Decoske, 2010). In areas too shallow to snorkel (< 10 cm) the observer visually scanned the stream with polarized sunglasses, while walking slowly upstream. We supplemented our sampling with data from 23 reaches sampled during the 2010 annual population monitoring by the Missouri Department of Conservation (MDC) from June through September using the same methods as the 2011 sampling.

Resource Sampling

Within each reach, macrohabitats, or hydraulic channel units, were delineated as riffle, run, shallow pool, deep pool, and pool edge (Arend, 1999). For each macrohabitat, we collected variables that we believed to influence presence of Niangua darters (Table 1).

Depth (cm), water velocity (m s^{-1}), and wetted width (m) was measured in the thalweg at

three equidistant points using a Rickly top-setting wading rod (Ricky Hydrological Co., Columbus, Ohio, U.S.A.), a model 2000 Marsh-McBirney flow meter (Marsh-McBirney Inc., Frederick, Maryland, U.S.A), and a model RX-1000i Leupold digital laser rangefinder (Leupold & Stevens Inc., Beaverton, OR, U.S.A), respectively. These three equidistant measurements were then averaged for each macrohabitat. The percentage of shoreline occupied by emergent macrophytes was calculated by measuring the length of contiguous vegetation on each shoreline using a model RX-1000i Leupold digital laser rangefinder and dividing by the total length of shoreline for each macrohabitat.

For each reach, we collected both instream and riparian variables believed to influence presence of Niangua darters (Table 1). For many instream variables; mean depth, mean velocity, and mean wetted width, we summarized data from macrohabitats contained within each reach. From the macrohabitat-scale data we also calculated the percentage of each reach as riffle and run hydraulic channel units. To examine the influence of water temperature on Niangua darter presence, we deployed 50 HOBO Pendant[®] temperature loggers at the same random stream reaches, or nearby access points, prior to fish sampling and continued to collect water temperature for at least one year (Onset Computer Cooperation, Bourne, MA, U.S.A.). Using these data we calculated a maximum yearly temperature for each reach.

To examine characteristic of the riparian corridor we created 150 m buffers around each reach using Buffer in ArcGIS (Environmental Systems Research Institute, Redlands, CA, U.S.A.) to quantify riparian land use. Using these buffers we then extracted percent land use/cover for three categories: urban and agricultural from the National Land Cover Database (NLCD) (Fry et. al, 2011). Developed open space and

low, medium, and high intensity developed areas were aggregated as urban. The agricultural category consisted of grassland/herbaceous, pasture/hay, and cultivated crop areas.

At the watershed-scale we examined factors that we believe to influence the presence of Niangua darters at each reach (Table 1). To quantify watershed land use we created catchments, or segment-sheds, for each stream segment within the 1:24,000 NHD using a 30 m Digital Elevation Model (DEM), ArcHydro in ArcGIS, and Rivex (Gesch, 2007; Environmental Systems Research Institute, Redlands, CA, U.S.A.; RivEx, Southampton, UK). For each reach we aggregated all catchments upstream to represent the upstream watershed. Within each watershed we extracted the percent area as three categories of land use/cover from the NLCD (see above). The percent of each watershed as limestone and dolomite was calculated based on the 2006 Missouri Department of Natural Resources' (MDNR) 1:500,000 Bedrock dataset (MDNR, 2006). We summed percent area of hydrologic soil groups A and B, representing soils with high to moderate infiltration rates and low to moderate runoff potential, from the Natural Resources Conservation Service's Soil Survey Geographic database (SSURGO; NRCS, 2012). From this database we also calculated area-weighted mean K factor, which represents the susceptibility of soil to sheet and rill erosion by water (NRCS, 2012). For each watershed the number of springs per square kilometer was calculated based on MDNR's Springs dataset (MDNR, 2010). Relief was calculated as the difference between maximum and minimum elevation within each watershed based on a 30 m DEM. We also calculated gradient ($m\ km^{-1}$) and elevation of each reach using a 30 m DEM (Gesch, 2007). Link

magnitude was calculated using RivEx, as a representation of stream size (RivEx, Southampton, UK; Shreve, 1966).

Analysis

We used logistic regression to evaluate the influence of watershed, reach, and macrohabitat-scale factors on resource selection of Niangua darters. Logistic regression predicts a binary response variable, such as presence/absence, and allows both continuous and categorical independent variables. The general form of the logistic model is:

$$\pi = \frac{e^{(\beta_0 + \beta_1 x_1 + \dots + \beta_p x_p)}}{1 + e^{(\beta_0 + \beta_1 x_1 + \dots + \beta_p x_p)}}$$

where π is the probability of selection. We used an information theoretic approach to determine the suite of models that best fit the data, allowing us to test multiple hypotheses simultaneously (Burnham & Anderson, 2002). *A priori* models were developed at each spatial scale to examine the influence of collected variables on the presence of Niangua darters (Table 2). In addition to examining candidate models at the reach- and watershed-scale independently, we also examined support in a combined candidate model-set to determine which scale is more influential on the presence of Niangua darters. We were unable to collect water temperature data at each reach therefore water temperature was included in a separate candidate model set for only those reaches where water temperature data existed.

Categorical variables were coded as dummy variables with one category excluded from each model as a reference. We excluded the riffle habitat type and the 0-25% emergent vegetation as reference categories at the macrohabitat-scale. For continuous variables we examined plots of data prior to model fitting in order to determine whether

evidence existed to support nonlinear relationships. We verified support for these forms by using an approach similar to Franklin et al. (2000) in which we fit linear, quadratic, and log forms of each continuous variable to the dataset and examined the relative support of each form using Akaike's Information Criterion for small sample sizes (AIC_c ; Franklin et al., 2000; Burnham & Anderson, 2002). For all variables, the form observed from plotting received the lowest AIC_c value and was retained for model selection. From this we found a quadratic relationship existed for mean depth, mean velocity, mean wetted width, maximum annual water temperature, and spring density, indicating a parabolic response, as well as a log relationship for limestone/dolomite geology, indicating a pseudo-threshold response. We assessed multicollinearity among covariates in each model using a cutoff tolerance value of 0.4 (Allison, 1999). If collinearity existed, affected covariates were excluded from the model, starting with the variable that had the least influence on Niangua darter presence. We also examined cross-scale correlation at the reach- and watershed-scale, excluding the macrohabitat-scale due to differences in sample unit and thus sample size. Cross-scale correlations have the same influence on conclusions as do within-scale correlations and failure to evaluate cross-scale correlation in multi-scale habitat studies can lead to erroneous conclusions in habitat associations (Battin & Lawler, 2006). To examine cross-scale correlation we examined the correlation matrix of reach- and watershed-scale variables from top-ranked models.

We fit each candidate model set using PROC LOGISTIC in SAS[®] (SAS Institute Inc., Cary, NC, U.S.A.). Models were compared using Akaike's Information Criterion corrected for small sample size (AIC_c) and Akaike weights (w_i) to rank each model in the candidate set (Burnham & Anderson, 2002). All models within the 90% confidence set of

candidate models, based on Akaike weights, were considered the best-fit model(s) (Burnham & Anderson, 2002). We used odds ratios and their 95% confidence intervals to examine the influence of explanatory variables on Niangua darter selection. We estimated relative selection probabilities of explanatory variables by holding all other variables constant at their mean and varying the explanatory variable (Hosmer & Lemeshow, 2004).

To evaluate the performance of top-ranked logistic regression models, or “goodness of fit”, we calculated the area under the receiver operating characteristic (ROC) curve (Metz, 1978; Cumming, 2000), which evaluates correctly and incorrectly classified predictions across the entire range of threshold values, as opposed to threshold dependent methods (Pearce & Ferrier, 2000). A model with an AUC value of 0.5 has no predictive power as this value could be reached by chance, while a perfect model would have an AUC value of 1.0.

RESULTS

We sampled a total of 651 macrohabitats (mean length 32.9 ± 1.0 m) across all reaches from May to September in 2010 and 2011. Of the 651, only 51 were occupied by Niangua darters. All samples were conducted between 0800 and 1830 h at a net upstream rate of 3.3 ± 0.2 m min⁻¹ while mean lateral visibility was 1.8 ± 0.1 m. The macrohabitat model containing depth and velocity was most important for selection by Niangua darters ($w_i = 0.99$), based on AIC_c ranking, with no other model falling into the 90% confidence set of candidate models (Table 2; Table 3). The area under the ROC curve of our top-ranked macrohabitat-scale model was 0.73. Depths between 40-60 cm and velocities between 0.3-0.8 m s⁻¹ had the greatest probability of selection (Figure 2). Niangua darters

were detected in macrohabitats with mean depths of 41.4 ± 2.1 cm (range= 20-97) and mean velocities of 0.20 ± 0.03 m s⁻¹ (range= 0-0.77) compared to 28.1 ± 0.7 cm and 0.21 ± 0.01 m s⁻¹, respectfully, in macrohabitats where they were not detected (Table 4).

We sampled a total of 74 reaches (mean length 314.3 ± 2.3 m) from May to September in 2010 and 2011. Of the 74 reaches, Niangua darter were detect in only 25. Depth and velocity was most important for reach-scale selection by Niangua darters ($w_i=0.99$), based on AIC_c ranking, with no other model falling into the 90% confidence set of candidate models (Table 2; Table 3). The area under the ROC curve of our top-ranked reach-scale was 0.87. However, we were unable to estimate the probability of selection for this model as quasi-complete separation inhibited computation of the maximum likelihood estimate. For those reaches where we had water temperature data (detected= 8, not detected= 23), maximum annual water temperature was within the 90% confidence set of candidate models ($w_i=0.17$; Table 5) indicating water temperature may influence the presence of Niangua darters. Niangua darters were detected in reaches with 1) mean depths of 38.5 ± 2.1 cm (range= 15.5-64.7) compared to 24.2 ± 1.2 cm (range= 10.7-47.3) in non-detected reaches 2) mean velocities of 0.26 ± 0.03 m s⁻¹ (range= 0.03-0.68) compared to 0.17 ± 0.02 m s⁻¹ (range= 0-0.47) in non-detected reaches and 3) maximum annual water temperature of 34.0 ± 0.3 °C (range= 32.8-35.4) compared to 32.0 ± 0.6 °C (range= 24.8-37.1) in non-detected reaches (Table 4).

We examined characteristics of watersheds upstream of sampled reaches (195.5 ± 25.5 km²) and found soil and geology were most influential on the presence of Niangua darters ($w_i=0.99$), with no other models falling into the 90% confidence set of candidate models (Table 2). The area under the ROC curve of our top-ranked watershed-scale

model was 0.85. Niangua darters were detected in reaches originating from watersheds with significantly greater relief (mean=158.2 ± 6.7 m) and percent of the watershed as limestone/dolomite bedrock (mean=82.5 ± 2.7), with little difference in percent of watershed as soil groups A and B and area-weight mean K factor among watersheds (Table 4). Once again we were unable to estimate the probability of selection for this model due to quasi-complete separation, likely due to small sample size. In the candidate model-set including models from both reach- and watershed-scale, soil and geology characteristics at the watershed-scale were most important ($w_i = 0.89$), with no other models falling into the 90% confidence set of candidate models (Table 2). Cross-scale correlation indicated watershed relief was significantly correlated with mean depth at the reach-scale ($r=0.57$, $P<0.001$) and mean velocity at the reach-scale ($r=0.66$, $P<0.001$). In addition reach mean velocity was significantly correlated with hydrologic soil groups A and B ($r=0.47$), and strongly correlated with limestone and dolomite bedrock geology ($r=0.23$).

DISCUSSION

Depth and velocity had the greatest influence on presence of Niangua darters at the macrohabitat and reach-scale. Niangua darters were detected in deeper macrohabitats and reaches when compared to those where they were not detected. Mattingly & Galat (2002) also found depth to be important in the Little Niangua River, with Niangua darters selecting relatively deeper microhabitats. Our results indicate Niangua darters selected even deeper depth than Mattingly and Galat (2002), but may vary due differences in scale and spatial extent of sampling between studies. Depth and velocity are important resource attributes for darters and for coexisting darter species, are often spatially segregating

factors, with each species retaining morphological or life history traits to best utilize their niche (Chipps et al., 1994; Wood & Bain, 1995; Stauffer et al., 1996; Skyfield & Grossman, 2008; Davis & Cook, 2010). The Appalachia darter, *Percina gymnocephala*, a large darter much like the Niangua darter, has also been found to occupy deeper habitats (Chipps et al., 1994), while smaller darters, such as the orangethroat darter, *Etheostoma spectabile*, utilize shallower, swifter habitats (Musselman & Brewer, 2009). Like many other lotic fishes, depth and velocity selection is likely tied to energetics for darters. Shallow, swift habitats may provide greater density and diversity of benthic invertebrates however biotic interaction between darters and other lotic fishes may make such habitats less desirable (Chipps et al., 1994; Doisy & Rabeni, 2001; Mattingly & Galat, 2002; Ashton & Layzer, 2010; Davis & Cook, 2010). Large, cryptic darters, such as the Niangua darter, may utilize deeper habitats than other small bodied darters due to their vulnerability to aerial predation in shallower habitats (Power, 1987; Schlosser, 1988; Greenberg, 1991; Chipps et al., 1994). In addition, their large body and presumably better swimming ability may allow them to escape piscivorous predation in deeper habitats (Power, 1987; Schlosser, 1988; Greenberg, 1991; Chipps et al., 1994). Therefore, the selection of depth and velocity is likely attributed to multiple factors linked to energetics, food availability, and predator avoidance.

Maximum annual water temperature appeared to influence the presence of Niangua darters. Though thermal tolerance of Niangua darters is unknown, we hypothesized, based on thermal tolerances of other darters, that Niangua darters would be present in reaches with moderate water temperatures, avoiding extreme hot or cold streams that may hinder growth, survival, and reproduction (Smale & Rabeni, 1995;

Strange et al., 2002). Contrary to our hypothesis, Niangua darters were detected in reaches with relatively high annual maximum water temperature (mean= 34.0 ± 0.3 °C), at least in the 31 sites where we recorded water temperature. Furthermore, many of the reaches where Niangua darters were detected consistently exhibited water temperatures approaching or above critical thermal tolerances of other darters (Hlohowskyj & Wissing, 1985; Smale & Rabeni, 1995; Strange et al., 2002). For example, Hlohowskyj & Wissing (1985) found the greenside darter, *Etheostoma blennioides*, a relatively large bodied percid, has a critical thermal maximum of 34.5 °C. Should water temperature become unfavorable in streams occupied by Niangua darters, they may be able to utilize thermal refugia near spring outflows or in habitats with groundwater seepage during periods of elevated water temperatures (Schaefer et al., 2003). To the same end, given significant heterogeneity exists in groundwater-fed streams in the Ozarks (Westhoff & Paukert 2014), our reach-scale analysis may not fully reflect water temperature selection of Niangua darters. Alternatively, temperatures in streams where Niangua darters were detected may not be particularly stressful as some darters, such as the Arkansas darter, *Etheostoma cragini*, have relatively high critical thermal maximum, at 38.4°C (Smith & Fausch, 1997). However, more research is warranted to determine the vulnerability of Niangua darters (i.e. thermal tolerance) to increases in water temperature.

Soil and geology were most important for selection of Niangua darters at the watershed-scale. Our results suggested that streams originating from watersheds dominated by limestone and dolomite bedrock geology, soils with relatively greater infiltration rates and lower transmission rates, and greater relief, had the greatest likelihood of containing Niangua darters. Flow regime, temperate, channel morphology

and bed composition, and water chemistry are all directly influence by underlying soils and geology (Hynes, 1975; Frissell et al. 1986; Richards et al. 1996). We believe this model is indicative of watershed characteristics which create stable, coarse, silt-free instream habitats that are favorable for Niangua darters, through multiple mechanisms (Mattingly & Galat, 2002). Watersheds dominated by limestone and dolomite geology are very common within the Osage EDU. This geology permits the formation of numerous springs and underground flow, which feed into local streams, creating relatively stable base flows and thermal conditions throughout the year (Sowa et al. 2007; Westhoff & Paukert, 2014). Poff and Allan (1995) found that hydrologic stability significantly increased the presence of large, benthic invertivorous fish, like the Niangua darter. Stable lotic ecosystems often enable specialized feeding and life history strategies, like that of the Niangua darter, to prevail (Pfleiger, 1978; Poff & Allan, 1995).

Furthermore, these watershed characteristics help to minimize suspension and deposition of sediments, as well more immobile substrata which are favorable for benthic invertivore and/or lithophilous spawning fishes (Pfleiger, 1978; Mattingly & Galat, 2002). Furthermore, streams originating from soils with greater infiltration rates are less susceptible to erosional processes and produce less instream sedimentation with channel beds dominated by larger, less-embedded substrates (Turner, 2009). Stable habitats containing relatively coarse, silt-free substrates formed by these watershed characteristics are favorable for other benthic fishes, and likely so for Niangua darters, which often utilize these habitats for feeding and reproduction (Poff & Allan, 1995; Rabeni & Smale, 1995; Pfleiger, 1978; Mattingly & Galat, 2002). Alternatively, our macrohabitat and reach-scale metrics of embeddedness received little support although a large percentage

of macrohabitats and reaches occupied by Niangua darters consisted of minimally embedded substrates. Little variation existed in embeddedness among macrohabitats and reaches, with only 20% and 18% of all sampled macrohabitats and reaches, respectively, containing substrates greater than 25% embedded, leaving little room for discrimination. Last, embeddedness was only present in univariate models at both scales but may play a secondary role in habitat selection at these spatial scales.

In the face of climate and land use change, the viability of Niangua darter populations is uncertain. Our findings did not suggest that anthropogenic land use has a significant influence on Niangua darter populations, under current conditions. However, watersheds within the Osage EDU have relatively little anthropogenic land use and thus may be below a threshold where a response would be observed (Compton & Taylor, 2013). Furthermore, anthropogenic land use alteration has been shown to increase erosion and sedimentation in streams which we believe is a significant driver for Niangua darter populations (Cooper, 1993; Johnson et al., 1997; Allan, 2004; Dudgeon et al., 2006). Watershed features that create hydrologically stable conditions with coarse, silt-free substrates are favorable for Niangua darters. Our findings indicate that the presence of Niangua darters is likely to be influenced predicted increases in water temperature, due to climate change, in the Ozarks of Missouri of 2-4 °C, as many Niangua darters currently exists in reaches with water temperatures approaching or above critical thermal tolerances of other darters (Hlohowskyj & Wissing, 1985; Smale & Rabeni, 1995; Strange et al., 2002; The Nature Conservancy, 2009; Pease & Paukert, 2014; Westhoff et al., 2014). However, more research is warranted to determine the vulnerability of Niangua darters (i.e. thermal tolerance) to increases in water temperature.

Contrary to our hypothesis, reach-scale factors were less influential on the distribution of Niangua darters than watershed-scale factors. Currently the dominant hypothesis is that watersheds with less anthropogenic alteration, like the Osage River Basin which is over 50% forest and only 5% urban land use (Fry et al., 2011), are influenced more by local factors (Wang et al., 2003; Johnson et al., 2007). Others have also found results contradicting this hypothesis (Roth et al., 1996; Allan et al., 1997; Esselman & Allan, 2010). Perhaps, regardless of disturbance levels, factors at large spatial scales play a greater role in determining the ecological integrity of rivers and streams, while the influence of local-scale factors is secondary, yet complementary (Roth et al., 1996; Allan et al., 1997). Others have suggested study design, scale dependency of metrics, heterogeneity of catchments, and/or cross-scale correlations may confound the influence of local scale factors within a landscape of relatively low anthropogenic alteration (Allan et al., 1997; Battin & Lawler, 2006; Esselman & Allan, 2010), and may lead to erroneous conclusions in habitat associations (Battin & Lawler, 2006). Cross-scale correlation existed among variables in this study and therefore the specific mechanisms and spatial scale of resource selection of Niangua darters is difficult to discern.

Management actions that increase channel complexity, specifically water depth, may be beneficial for populations of Niangua darters. Although land use in the Ozark EDU is currently dominated by forest, future landscape alteration has the potential to negatively affect Niangua darter populations. Especially since watershed-scale factors have a greater influence in the Osage EDU, and other stream ecosystems (Alan et al., 1997). Increased sediment inputs and altered, flashy hydrology due to land use change

will likely create less stable, embedded bedforms that will influence channel complexity, specifically intermediate depths examined herein, as well as prey availability and reproductive success of Niangua darters (Poff & Allan, 1995; Sutherland et al., 2002; Tipton et al., 2004). Therefore, limiting anthropogenic impacts to stream systems and balancing stream conservation with human use may benefit Niangua darters. The amount of change that can be sustained will be variable among systems or within systems. In other systems, responses have been observed at > 50 % area for agricultural (Wang et al., 1997) and 10 – 20 % area for urban land use (Yoder et al., 1999), as the effect of urban land use is greater per capita.

Management to increase or maintain riparian corridors, can limit streambank erosion, further reduce fine sediment inputs, and reduce water temperature (Sullivan & Adams, 1989; Rutherford et al., 2004). Additionally, riparian corridors provide woody debris to stream system which has also been shown to increase local instream habitat complexity (Schlosser, 1991). Greater habitat complexity and buffered water temperature will likely improve population stability by reducing physiological stress and providing refuge during taxing periods (Schlosser, 1991). Increasing or maintaining the availability of habitats with intermediate depths with relatively coarse, silt-free substrates will likely benefit populations of Niangua darters. Necessary dimensions and connectivity of riparian corridors are not well studied (Barton et al., 1985), therefore exact prescriptions are not available. At a minimum, corridor width of 10 m, in areas devoid of riparian corridor, would likely improve instream conditions for Niangua darter. Currently thermal tolerances of Niangua darters are unknown however maintaining maximum water

temperatures below 34° C, conservatively (thermal tolerance of greenside darters, *Etheostoma blennioides*), would likely benefit Niangua darters.

Our research provides a better understanding of multi-scale factors that influence the distribution of the Niangua darter, and may also be used to evaluate sites for translocations if deemed a suitable management action by resource management agencies. Understanding habitat requirements of species prior to translocation will likely improve success of such efforts (Armstrong & Seddon, 2008; Spurgeon et al. In press). These findings will aid in recovery efforts by helping managers to better define and evaluate conservation criteria, leading to a better informed recovery effort.

Small sample sizes in this study reduced our inference and caused quasi-complete separation at the reach- and watershed-scale (Allison, 2004). Furthermore, due to the prevalence of non-detections in our initial random sample we used samples collected from the Missouri Department of Conservation annual Niangua darter sampling which were not a true random sample and may not accurately represent the scope of variability in Niangua darter populations. However, our findings are similar to other research on Niangua darters (Mattingly & Galat; 2002) and other lotic fishes (Wood & Bain, 1995; Poff & Allan, 1995; Skyfield & Grossman, 2008; Davis & Cook, 2010). Additional research is warranted to examine the intricacies of Niangua darter habitat preference, in which case these data could be valuable to refine sampling and increase effectiveness. Significant improvement to this research would include; a larger sample, especially presence sites, accounting for the potential of imperfect detection, linking population vital rates to selected resources, and incorporating known physiological characteristics of the species.

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Table 1: Parameters used in Niangua darter resource selection models at the macrohabitat-, reach-, and watershed- scale in sampled streams of the Osage River Basin, Missouri, 2010-2011.

Scale	Parameter	Description
Macrohabitat	DEP	Mean water depth (cm) measured at three equidistant points within each macrohabitat
	VEL	Mean water velocity ($m s^{-1}$) at 60% depth measured at three equidistant points within each macrohabitat
	WW	Mean of wetted widths (m) measured at three equidistant points within each macrohabitat
		Visual classification of percentage of shoreline (%) occupied by emergent macrophytes
	V1	0 to 25 percent occupied
	V2	26 to 50 percent occupied
	V3	51 to 75 percent occupied
	V4	76 to 100 percent occupied
		Visual classification of hydraulic channel unit
	RIF	Shallow with swift water velocity and considerable turbulence and gradient
	RUN	Deep with an even flow that lacked pronounced turbulence
	SP	Shallow with little or no water velocity
	PE	Margin of deep pool with gentle slope and little or no water velocity
Reach	RURB	Percent of 150 m riparian corridor as developed open space and low, medium, & high intensity
	RAGR	Percent of 150 m riparian corridor as grassland/herbaceous, pasture/hay, and cultivated crops
	RWW	Mean wetted width (m)
	RDEP	Mean depth (cm)
	RVEL	Mean velocity ($m s^{-1}$) at 60% depth
	RV2	Percent of reach as V2 described above in macrohabitat
	RV3	Percent of reach as V3 described above in macrohabitat
	RV4	Percent of reach as V4 described above in macrohabitat
	PRN	Percent of reach as RUN described above in macrohabitat
	PRF	Percent of reach as RIF described above in macrohabitat
	TEM	Maximum annual water temperature (C)

Table 1 Continued:

Scale	Parameter	Description
Watershed	WURB	Percent of upstream catchment as developed open space and low, medium, & high intensity developed land use/cover
	WAGR	Percent of upstream catchment as grassland/herbaceous, pasture/hay, and cultivated crops land use/cover
	AB	Percent of upstream catchment as hydrologic soil groups A and B, having relatively greater infiltration rates and lower runoff potential
	KFA	Area weighted mean K factor; susceptibility of soil to sheet and rill erosion
	REL	Difference between maximum and minimum elevation (m) within the upstream catchment
	LD	Percent of upstream catchment having limestone and dolomite bedrock geology
	LIM	Link magnitude of each reach
	GRA	Gradient of each reach ($m\ km^{-1}$)
	SDE	Density of springs within the upstream catchment (springs km^{-2})
	ELV	Elevation of each reach (m)

Table 2: Number of parameters (K), AIC_c (corrected for small samples size), change in AIC_C, and Akaike weight for models hypothesized to influence resource selection of Niangua darters at the macrohabitat-, reach-, and watershed-scale (see Table 1) in sampled streams of the Osage River Basin, Missouri, 2010-2011.

Model	K	AIC _c	ΔAIC_c	w _i
MACROHABITAT (n= 651)				
$\beta_1(\text{DEP}) + \beta_2(\text{DEP}^2) + \beta_3(\text{VEL}) + \beta_4(\text{VEL}^2)$	5	331.40	0	0.9998
$\beta_1(\text{RUN}) + \beta_2(\text{SP}) + \beta_3(\text{PE})$	4	349.24	17.83	0.0001
$\beta_1(\text{WW}) + \beta_2(\text{WW}^2)$	3	351.54	20.14	0.0000
$\beta_1(\text{V2}) + \beta_2(\text{V3}) + \beta_3(\text{V4})$	4	355.83	24.43	0.0000
REACH (n= 74)				
$\beta_1(\text{RDEP}) + \beta_2(\text{RDEP}^2) + \beta_3(\text{RVEL}) + \beta_4(\text{RVEL}^2)$	5	73.03	0.00	0.9830
$\beta_1(\text{RV2}) + \beta_2(\text{RV3}) + \beta_3(\text{RV4})$	4	81.19	8.16	0.0167
$\beta_1(\text{RWW}) + \beta_2(\text{RWW}^2)$	3	89.06	16.03	0.0003
$\beta_1(\text{PRN}) + \beta_2(\text{PRF})$	3	98.26	25.23	0.0000
$\beta_1(\text{RURB}) + \beta_2(\text{RAGR})$	3	99.02	25.98	0.0000
WATERSHED (n= 74)				
$\beta_1(\text{AB}) + \beta_2(\text{KFA}) + \beta_3(\text{REL}) + \beta_4(\log(\text{LD}))$	5	68.52	0	0.9956
$\beta_1(\text{LIM}) + \beta_2(\text{GRA})$	3	79.37	10.85	0.0044
$\beta_1(\text{SDE}) + \beta_2(\text{SDE}^2) + \beta_3(\text{ELV})$	4	94.06	25.54	0.0000
$\beta_1(\text{WURB}) + \beta_2(\text{WAGR})$	3	96.87	28.36	0.0000
REACH & WATERSHED (n=74)				
$\beta_1(\text{A}) + \beta_2(\text{KFA}) + \beta_3(\text{REL}) + \beta_4(\log(\text{LD}))$	5	68.47	0.00	0.8926
$\beta_1(\text{RDEP}) + \beta_2(\text{RDEP}^2) + \beta_3(\text{RVEL}) + \beta_4(\text{RVEL}^2)$	5	73.03	4.56	0.0913
$\beta_1(\text{LIM}) + \beta_2(\text{GRA})$	3	76.74	8.27	0.0143
$\beta_1(\text{RV2}) + \beta_2(\text{RV3}) + \beta_3(\text{RV4})$	4	80.91	12.43	0.0018
$\beta_1(\text{RWW}) + \beta_2(\text{RWW}^2)$	3	89.06	20.59	0.0000
$\beta_1(\text{SDE}) + \beta_2(\text{SDE}^2) + \beta_3(\text{ELV})$	4	94.06	25.58	0.0000
$\beta_1(\text{WURB}) + \beta_2(\text{WAGR})$	3	98.26	29.79	0.0000
$\beta_1(\text{PRN}) + \beta_2(\text{PRF})$	3	98.27	29.80	0.0000
$\beta_1(\text{RURB}) + \beta_2(\text{RAGR})$	3	100.44	31.97	0.0000

Table 3: Coefficients and standard error for parameters contained in top-ranked models at the macrohabitat-, reach-, and watershed-scale where Niangua darters were detected or not detected within sampled streams of the Osage River Basin, Missouri, 2010-2011.

Scale	Parameter	β	SE	Wald's 95% CI		X^2	Pr > X^2
				Lower	Upper		
Macrohabitat (n=651)	Intercept	-2.432	0.223	-2.869	-1.995	119.00	<.0001
	DEP	0.074	0.016	0.044	0.105	22.45	<.0001
	DEP ²	-0.001	0.000	-0.002	0.000	8.26	0.0040
	VEL	0.628	0.905	-1.147	2.402	0.48	0.4882
	VEL ²	-0.960	1.834	-4.555	2.635	0.27	0.6007
Reach (n=74)	Intercept	-1.212	0.485	-2.162	-0.262	6.25	0.0124
	DEP	0.186	0.053	0.082	0.291	12.28	0.0005
	DEP ²	0.000	0.004	-0.007	0.008	0.00	0.9518
	VEL	-2.252	2.824	-7.787	3.282	0.64	0.4251
	VEL ²	7.872	10.562	-12.830	28.574	0.56	0.4561
Reach (n=31)	Intercept	-2.309	-5.095	0.478	1.422	2.64	0.1044
	TEM	2.495	-1.085	6.075	1.827	1.87	0.1720
	TEM ²	-0.656	-1.740	0.428	0.553	1.41	0.2358
Watershed (n=74)	Intercept	-24.724	10.529	-45.360	-4.087	5.51	0.0189
	AB	-0.081	0.047	-0.173	0.012	2.91	0.0880
	KFA	-4.569	10.942	-26.014	16.877	0.17	0.6763
	Relief	0.062	0.016	0.031	0.094	14.91	0.0001
	LD	9.794	4.795	0.396	19.192	4.17	0.0411

Table 4: Mean, standard error (SE), and range for continuous variables at the macrohabitat-, reach-, and watershed-scale (see Table 1) where Niangua darters were detected or not detected within sampled streams of the Osage River Basin, Missouri, 2010-2011.

Scale	Parameter	Detected		Not Detected	
		\bar{x} (SE)	Range	\bar{x} (SE)	Range
Macrohabitat		(n = 51)			(n = 600)
	DEP	41.4 (2.1)	20-97	28.1 (0.7)	2-111
	VEL	0.20 (0.03)	0.00-0.77	0.21 (0.01)	0.00-1.45
	WW	10.1 (0.7)	3.0-24.0	8.0 (0.2)	0.5-39.3
Reach		(n = 25)			(n = 49)
	RURB	2.4 (0.7)	0-13.2	4.5 (1.4)	0-62.8
	RAGR	34.4 (3.4)	5.9-75.2	37.4 (3.3)	0-88.2
	RWW	10.6 (0.6)	3.8-16.3	8.3 (0.5)	2.8-17.7
	RDEP	38.5 (2.1)	15.5-64.7	24.2 (1.2)	10.7-47.3
	RVEL	0.26 (0.03)	0.03-0.68	0.17 (0.02)	0-0.47
	RV2	6.9 (1.8)	0-26.9	9.1 (2.1)	0-52.8
	RV3	5.8 (2.5)	0-44.8	2.7 (1.2)	0-53.4
	RV4	16.5 (4.8)	0-92.5	1.2 (0.5)	0-15.6
	PRN	58.6 (4.4)	12.7-89.9	66.2 (2.5)	29.7-94.9
	PRF	24.8 (2.8)	6-71.9	21.1 (2.1)	0-57.4
	(n=8)			(n=23)	
	TEM	34.0 (0.3)	32.8-35.4	32.0 (0.6)	24.8-37.1
Watershed		(n = 25)			(n = 49)
	WURB	4.6 (0.3)	1.5-7.4	5.6 (0.5)	2.2-18.9
	WAGR	42.8 (1.5)	19.6-55.1	48.3 (2.4)	15.6-75.3
	LIM	365.6 (58.4)	19.0-807.0	98.0 (18.5)	6.0-592.0
	ELV	239.3 (11.3)	168.2-343.7	257.3 (6.6)	170.9-366.0
	LD	82.5 (2.7)	63.3-99.6	75.2 (3.7)	0.7-100.0
	AB	21.4 (1.6)	7.1-36.4	20.4 (1.7)	1.8-56.1
	KFA	0.3 (0.0)	0.2-0.3	0.3 (0.0)	0.2-0.4
	SDE	0.0 (0.00)	0-0.1	0.0 (0.0)	0-0.2
	REL	158.2 (6.7)	107.3-212.1	111.7 (4.7)	49.1-198.9
	GRA	2.1 (0.3)	0-6.2	3.9 (0.7)	0-25.8

Table 5: Number of parameters (K), AIC_c (corrected for small samples size), change in AIC_C, and Akaike weight for models predicting the presence of Niangua darters within reaches where we obtained one year of water temperature data (n=31), within the Osage River Basin, Missouri, 2010-2011.

Model	K	AIC _c	ΔAIC_c	w _i
$\beta_1(\text{RDEP}) + \beta_2(\text{RDEP}^2) + \beta_3(\text{RVEL}) + \beta_4(\text{RVEL}^2)$	5	32.71	0.00	0.3633
$\beta_1(\text{TEM}) + \beta_2(\text{TEM}^2)$	3	33.50	0.79	0.2441
$\beta_1(\text{RV2}) + \beta_2(\text{RV3}) + \beta_3(\text{RV4})$	4	34.00	1.29	0.1907
$\beta_1(\text{RWW}) + \beta_2(\text{RWW}^2)$	3	34.06	1.35	0.1846
$\beta_1(\text{RURB}) + \beta_2(\text{RAGR})$	3	39.35	6.64	0.0131
$\beta_1(\text{PRN}) + \beta_2(\text{PRF})$	3	41.66	8.95	0.0041

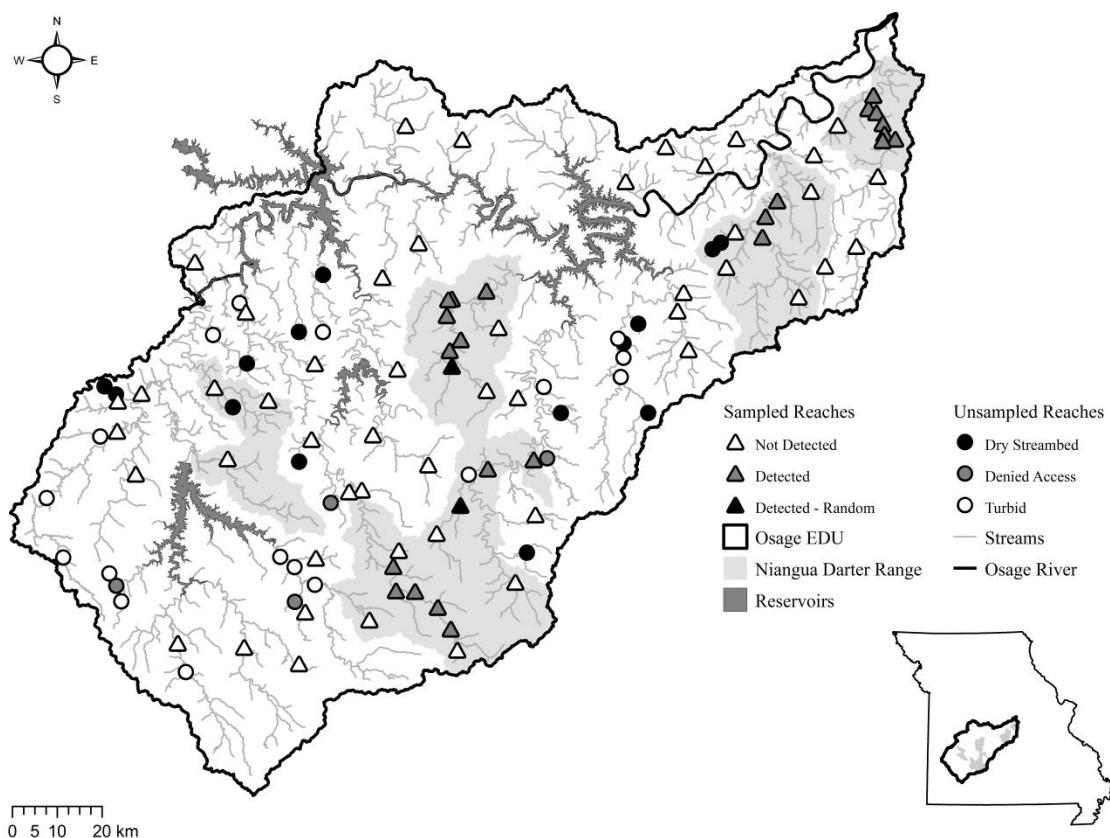


Figure 1: Sampled and unsampled reaches within the Osage Ecological Drainage Unit, Missouri, 2010–2011. Black triangles indicate randomly selected reaches where Niangua darters were detected and white triangles where they were not detected. Reaches sampled by the Missouri Department of Conservation are indicated by grey triangles. Unsampled reaches are depicted in black (dry streambed), grey (denied access), or white circles (underwater visibility <1 m). The transparent grey polygons indicate the current range of the Niangua darter.

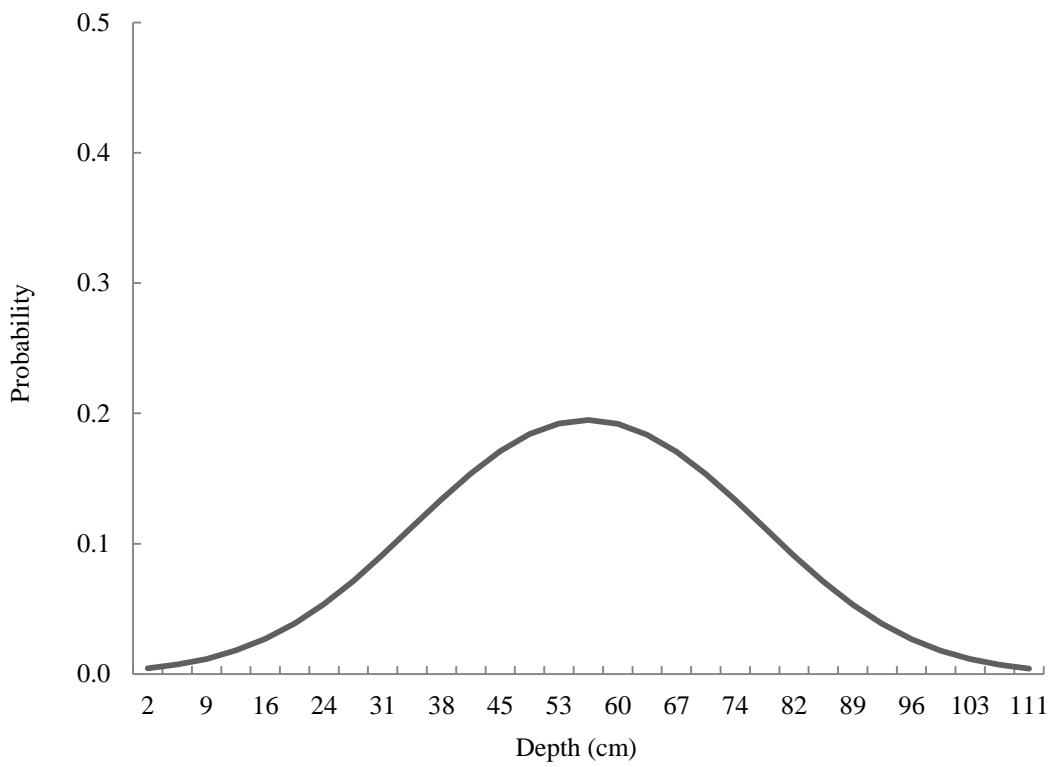


Figure 2: Relative probability of occurrence of Niangua darters based on macrohabitat depth in sampled streams of the Osage Ecological Drainage Unit, Missouri, 2010-2011.

CHAPTER 2: SEASONAL MICROHABITAT SELECTION OF NIANGUA DARTERS

ABSTRACT

Understanding temporal and spatial habitat relationships is important to the recovery of the federally threatened Niangua darter *Etheostoma nianguae*, and other imperiled freshwater fishes. We evaluated seasonal microhabitat selection of adult Niangua darters in two Missouri Ozark streams. Niangua darters were located every other month for one year to determine selection related to six instream habitat variables believed to be important to Niangua darter. We collected 196 locations of adult Niangua darters which we fit to univariate discrete choice models. Overall depth was our top-ranked model, with Niangua darters selecting relatively shallower depths during summer compared to spring and fall. Embeddedness and habitat type also fell within the 90% confidence set of candidate models along with depth in spring and fall, respectively, however results differed between streams. Adult Niangua darters in Little Niangua River selected less embedded locations during spring but in Starks Creek showed no selection for embeddedness. During fall in Starks Creek, adult Niangua darters selected deep runs and pools, the latter having stronger selection, but used all habitats in proportion to their availability in the Little Niangua River. We were unable to detect Niangua darters during winter as they may be using subsurface refugia or migrating outside of sample reaches, the latter being less likely due to their sedentary nature. Of the few juvenile Niangua darters locations collected, most were found in habitats with: shallower depths, slower velocities, and fine or gravel substrates that were greater than 50% embedded. Management that maintains or improves habitat diversity, specifically variability in

depth, and limits sediment input in streams occupied by Niangua darters should aid in the recovery of the species.

INTRODUCTION

Degradation of habitat is a main driver in the reduction of native stream fish abundance and diversity, specifically threatened and endangered fishes. Alterations to flow patterns, water chemistry, and/or channel morphology often challenge the longevity of biotic communities within streams (Allan & Flecker, 1993; Riccardi & Rasmussen, 1999). In order to conserve these imperiled fishes, it is advantageous to understand spatial and temporal ecological requirements of species' and how they are influenced by habitat and/or ecosystem changes (Schlosser & Angermeier, 1995; Smithson & Johnston, 1999; Musselman & Brewer, 2009).

Behavioral choice, or habitat selection, by an organism occurs across multiple spatial scales (Johnson, 1980). For lotic species, selection likely occurs across a classification gradient ranging from the smallest microhabitat to the largest drainage basin unit, typically a watershed (Frissel et al., 1986). In addition, selection often varies seasonally or ontogenically (Ross et al., 1992; Schlosser & Angermeier, 1995; Musselman & Brewer, 2009). Rosenberger (2002) found that Roanoke logperch, *Percina rex*, occupied deep microhabitats with slower bottom and mean water velocities, exposed gravel and cobble substrate, and less embedded substrate in winter months compared to summer months in Virginia. Musselman and Brewer (2009) found greater densities of orangemouth darters, *Etheostoma spectabile*, in deep pool habitats during winter months in Missouri. Ross et al. (1992) found that bayou darters, *Etheostoma rubrum*, used similar microhabitats throughout the year but were associated with cover, particularly boulders

and logs, and deeper habitats during winter months in southwest Mississippi. We believe that, like other darter species, Niangua darter *Etheostoma nianguae* Gilbert & Meek select habitat differently among seasons and across life history stages. Due to the sedentary nature of the Niangua darter we believe these variations in selection occur at a relatively small spatial scale (Pflieger, 1978).

The Niangua darter is a relatively large slender darter (6-9 cm total length) that is endemic to north-flowing streams of the Osage River Basin in south-central Missouri, U.S.A. (Pflieger, 1978). Since its discovery in 1884, ten populations have been documented and two extirpated, with additional populations likely extirpated prior to sufficient documentation of the species' distribution (USFWS, 1989). In 1985 the Niangua darter was listed as federally threatened due to low densities, limited distribution, and numerous threats: reservoir construction, destruction of stream habitat, degrading water quality, and introduction of non-native species (USFWS, 1989; Pflieger, 1997). Currently eight of ten known populations remain, but detecting trends in abundance and distribution over time has been difficult (Novinger & Decoske, 2013). Previous research has examined habitat preferences during summer months but we have no knowledge of habitat association during cooler months (Mattingly & Galat, 2002). Our objective was to determine seasonal and/or ontogenetic variation in habitat selection of Niangua darters at the microhabitat-scale.

Understanding habitat requirements of imperiled fishes, such as the Niangua darters, and how they may vary spatially and temporally will improve future efforts to maintain or improve habitat which in-turn will likely increase viability of such populations. For the Niangua darter this information may be crucial to understanding

their distribution and may assist in discovering or establishing additional viable populations to aid in recovery.

METHODS

Study Area

Our study occurred in three stream reaches approximately 500 m in length, two on the Little Niangua River, separated by approximately 25 river km, and one on Starks Creek, Missouri. We selected reaches with historically high densities of Niangua darters, based on previous sampling by the Missouri Department of Conservation (unpubl. data), in an attempt to attain sufficient sample sizes, given the rarity of the species.

The Little Niangua River and Starks Creek are centrally located within the distribution of the Niangua darter in the Osage River Basin in Missouri, USA. Drainages of these two streams fall within the Ozark Highlands Ecological Section which is characterized by highly dissected rolling hills dominated by oak-hickory forests underlain by carbonate bedrock (Nigh & Schroeder, 2002). Many streams in this region are spring-fed, carrying very little suspended solids, and dominated by chert substrates (Rabeni & Jacobson, 1993). The Little Niangua River is a fifth order stream that flows northeast 80 km, draining 830 km^2 , before entering Lake of the Ozarks. The Little Niangua River watershed consists of approximately 62 % forest/woodland, 31 % crop/grassland, and 4 % urban land use (Fry et al., 2006). Starks Creek is a third order stream that flows northeast 25 km, draining 93 km^2 , before its confluence with the Little Niangua River. Starks Creek consists of approximately 58 % forest/woodland, 36 % crop/grassland, and 4 % urban land use (Fry et al., 2006).

Fish Sampling

We systematically located Niangua darters within each of the three reaches every other month between July 2011 and May 2012 using snorkeling, which is the most efficient and least invasive method for sampling Niangua darters (Mattingly & Galat, 2002; Ashton & Layzer, 2010; Davis & Cook, 2010). Samples were completed between one hour after sunrise to one hour before sunset, while underwater, lateral visibility (m) was greater than one meter, measured with a Secchi disk (Carolina Biological Supply Co., Burlington, NC, U.S.A.). Samples were timed to assure equal effort among samples. Each reach was sampled by a single observer moving upstream in a zigzag progression at approximately $2\text{-}3 \text{ m min}^{-1}$. In addition to visually searching for Niangua darters, the observer moderately disturbed the substrate by overturning large substrates ($> 15 \text{ cm}$) and probing the stream bottom with their hands to dislodge individuals (Chipp et al., 1994; Davis & Cook, 2010). We believed this would maximize detection, as Niangua darters have been observed taking refuge underneath large substrates, presumably during periods of inactivity (Pfleger, 1978). In areas too shallow to snorkel ($< 10 \text{ cm}$), the observer visually scanned the stream with polarized sunglasses while walking slowly upstream. Upon sighting a Niangua darter, the observer visually designated its size class and marked its exact location with a weighted fluorescent marker. The size of each individual was classified to be juvenile [age 0; $< 60 \text{ mm}$ total length (TL)] or adult (age 1-4; $> 60 \text{ mm}$ TL; Pfleger, 1978; Mattingly & Galat, 2002). Niangua darters exhibited very little flight response to the observer, who was able to coax each fish downstream to assure they were not resampled. Niangua darters in close proximity to one another were treated as independent observations as we believed individuals were selecting locations based on

habitat attributes and not due to the presence of conspecifics. After sampling the length of the reach we returned to marked locations within 6 h to measure microhabitat variables.

Microhabitat Variables

To quantify microhabitat selection of Niangua darters, we defined the available choice set to be habitat within 100 m upstream or downstream of a used location, based on species' annual movement rates (Cooper & Millspaugh 1999). Niangua darters, like other darter species (Roberts et al., 2008; Skyfield & Grossman, 2008; Dammeyer et al., 2013), are fairly sedentary, with most individuals moving less than 100 m in a 12-month period (Pfleiger, 1997; McCleary, 2010). For each used location we randomly selected three locations within the available choice set based on a random distance (nearest m) upstream or downstream, along the shoreline, and a random percentage of the total wetted width at that distance, measured with a model RX-1000i Leupold digital laser rangefinder (Leupold & Stevens Inc., Beaverton, OR, U.S.A.).

At each used and available location, we measured six variables that we hypothesized to influence Niangua darter selection (Table 1). Habitat type was visually classified into hydraulic channel units modified after Arend (1999); high gradient riffle, low gradient riffle, shallow run, deep run, shallow pool, and deep pool. Depth (cm) and water velocity (m s^{-1}) at 60% depth (hereafter velocity) was measured using a Rickly top-setting wading rod (Ricky Hydrological Co., Columbus, Ohio, U.S.A.) and a model 2000 Marsh-McBirney flow meter (Marsh-McBirney Inc., Frederick, Maryland, U.S.A.).

Substrate and cover metrics were measured within a 0.25 m^2 circle, centered on each fish location. Dominant substrate was visually classified based on modified Wentworth (1922) size categories: fines (< 2 mm), gravel (2-15 mm), pebble (16-63 mm), cobble

(64-256 mm), boulder (> 256 mm), and bedrock. Embeddedness was visually classified among four percentage ranges, representing the percent to which larger particles were saturated by fine sediments (Platts et al., 1983). Cover density was measured as the number of physical structures with a surface axis ≥ 15 cm within the circle, capable of concealing an adult Niangua darter.

Analysis

We used discrete choice modeling to evaluate seasonal resource selection of adult Niangua darters. The discrete choice model yields the relative probability of a resource being selected based on its ‘utility’ or presumed satisfaction derived from the resource. The form of the discrete choice model is similar to the logistic model:

$$\pi = \frac{e^{(\beta_0 + \beta_1 x_1 + \dots + \beta_p x_p)}}{\sum e^{(\beta_0 + \beta_1 x_1 + \dots + \beta_p x_p)}}$$

however for discrete choice analysis each pair of used and available resources occur within a choice set, within which one used resource is compared to one or more available resources using iterative maximum likelihood methods. This method offers a more accurate estimation of resource selection as it accounts for the limited mobility of the species as well as spatial and temporal changes in available habitat that are common in lotic systems (Cooper & Millspaugh, 1999).

We fit univariate models for each of the six microhabitat variables to examine the relative importance of each variable simultaneously (Table 2). Although we hypothesized complex relationships between presence and habitat metrics, small sample sizes limited the number of variables used in our models (Figure 1). Furthermore, we were unable to fit models separately for each reach, incorporating a month interaction, to examine

seasonal variation in selection while acknowledging that availability may vary among reaches. Instead, samples were combined from both Little Niangua River reaches and months were pooled into three seasons; spring, summer, and fall. We felt that combining data from Little Niangua reaches, separated by only 25 river km, was justifiable as available microhabitat (Table 3) and stream size were comparable between reaches. However, Starks Creek is a substantially smaller stream which differs in available microhabitat (Table 3), particularly in average velocity, compared to Little Niangua River reaches. We combined samples from March and May to represent spring, as both samples occurred while conditions were favorable for spawning and Niangua darters could be expected to select microhabitat similarly (Pflieger, 1978; Mattingly et al., 2003). Limited biological information existed to define seasons outside of spring; however we combined samples from September and November as fall and retained July as summer. We were unable to locate adult Niangua darters during January and juveniles were excluded from formal seasonal analyses due to small sample sizes (n= 11; Figure 1).

Categorical variables were coded as dummy variables with one category excluded from each model as a reference. We excluded the deep pool category for habitat type, the fines category for dominant substrate, and the 75-100% category for embeddedness as reference categories. For continuous variables we examined plots of Niangua darter used and available data prior to model fitting in order to determine whether evidence existed to support nonlinear relationships. We verified support for these forms by using an approach similar to Franklin et al. (2000) in which we fit linear, quadratic, and log forms of each continuous variable to the dataset and examined the relative support of each form using Akaike's Information Criterion for small sample sizes (AIC_c; Burnham & Anderson,

2002). For all variables, the form observed from plotting received the lowest AIC_c value and was retained for model selection. From this we found a quadratic relationship existed for depth and velocity, indicating a parabolic response, as well as a log form of cover density, indicating a pseudo-threshold response (Table 2).

Models were fit separately for each combination of stream and season (hereafter model-set) using PROC MDC in SAS[®] (SAS Institute Inc., Cary, NC, U.S.A.) with each choice set consisting of one used and three available locations. Models were compared using Akaike's Information Criterion corrected for small sample size (AIC_c) and Akaike weights (w_i) to rank each model in the candidate set (Burnham & Anderson, 2002). All models within the 90% confidence set of candidate models, based on Akaike weights, were considered the best-fit model(s) (Burnham & Anderson, 2002). We used odds ratios and their 95% confidence intervals to examine the influence of explanatory variables on Niangua darter microhabitat selection (Hosmer & Lemeshow, 2004). We estimated relative selection probabilities of explanatory variables by holding all other variables constant at their mean and varying the explanatory variable.

To evaluate the performance of our top-ranked models we used a modified K-fold cross-validation in which we randomly separated the data into training and testing datasets (Boyce et al., 2002). The training dataset contained 80% of the original choice sets and was used to refit top-ranked models. Parameter estimates of the newly fit models were then used to estimate the relative probability of selection of used and available locations in the remaining 20% of the original choice sets, or testing dataset. Used locations were said to be predicted when they received the highest relative probability within the choice set. We repeated this process five times and determined performance of

top-ranked models based on the percentage of used locations correctly classified, averaged across all five replications.

RESULTS

We collected a total of 196 location used by adult Niangua darter across all reaches (mean length 498.2 ± 5.8 m) and months between July 2011 and May 2012, which we used to evaluate seasonal microhabitat selection. All samples were conducted between 0800 and 1200 h at a net upstream rate of 2.9 ± 0.1 m min⁻¹ while mean lateral visibility was 2.5 ± 0.2 m. Mean daily discharge on the Little Niangua River was 0.88 ± 0.35 m³ s⁻¹ during our study, slightly above base flow, no gage data was available for Starks Creek (USGS gage 06925250, Little Niangua River near Macks Creek). In all but two model-sets, depth was most important for microhabitat selection of adult Niangua darters ($w_i = 0.95-1.0$), based on AIC_c ranking (Table 4; Table 5). For the Little Niangua River, spring and Starks Creek, fall model-sets, model selection uncertainty existed with embeddedness and habitat type, respectively, falling into the 90% confidence set of candidate models along with depth.

During spring, adult Niangua darters used locations with depths of 42.7 ± 3.5 cm (range = 20-131) in the Little Niangua River compared to 52.8 ± 5.9 cm (range = 12-131) in Starks Creek (Figure 2). In the Little Niangua River, adult Niangua darters selected depths between 30 and 70 cm, with depths of 50 cm having 25 times the probability of selection relative to depths of 1 m (Figure 3). In Starks Creek, adult Niangua darters selected depths between 50 and 120 cm, with depths of 50 cm to 1 m having similar probabilities of selection. For the Little Niangua River, embeddedness was also an important microhabitat attribute for adult Niangua darters during spring. Substrates that

were 26-50% embedded had the greatest odds of selection (eight-fold), compared to 76-100% embedded substrates (Table 5).

During summer, adult Niangua darters used locations with depths of 23.1 ± 1.4 cm (range = 12-40) in the Little Niangua River compared to 49.7 ± 2.6 cm (range = 18-79) in Starks Creek (Figure 2). In the Little Niangua River, adult Niangua darters selected depths between 10 and 35 cm, with depths of 20 cm having 96 times the probability of selection relative to depths of 40 cm. In Starks Creek, adult Niangua darters selected depths between 35 and 80 cm, with depths of 40 cm having 9 times the probability of selection relative to depths of 20 cm (Figure 3).

During fall, adult Niangua darters used locations with depths similar to spring with an average depth of 39.2 ± 1.7 cm (range = 21-69) in the Little Niangua River compared to 58.8 ± 4.5 cm (range = 15-125) in Starks Creek (Figure 2). In the Little Niangua River, adult Niangua darters selected depths between 20 and 70 cm, with depths of 50 cm having 16 times the probability of selection relative to depths of 20 cm. In Starks Creek, adult Niangua darters selected depths between 35 and 125 cm, with depths of 90 cm having 8 times the probability of selection relative to depths of 40 cm (Figure 3). For Starks Creek, habitat type was also important for resource selection of adult Niangua darters during fall. However quasi-complete separation in the data inhibited computation of the maximum likelihood estimate for this model, likely because habitat type correctly classifies a large proportion of the data, in conjunction with small sample size (Allison, 2004; Table 5). In general, adult Niangua darters in Starks Creek selected locations in deep pool habitats, with relatively deep depths and little or no water velocity

(Figure 4). Explicitly, 22 of 33 (67%) Niangua darter locations were located in deep pool habitats while deep pools only made up 18 of 99 (18%) available locations.

Validation of our top-ranked model within each model-set showed that most models correctly predicted used locations 27-47% of the time, compared to 25% expected by random chance alone. However, for the Starks Creek, summer model-set, our depth model correctly classified 70% of used locations. Last, of our embeddedness model for the Little Niangua River, spring model-set correctly predicted used locations 18% of the time. We were unable to validate our habitat type model for the Starks Creek, fall model-set as quasi-complete separation prevented computation of the maximum likelihood estimate and therefore precluded calculations of relative probabilities.

We sampled a total of 11 juvenile Niangua darter locations across all reaches and months (Figure 1). Across all samples juvenile Niangua darters predominately used locations in shallow pool habitats ($n=8$) with either fine ($n=5$) or gravel ($n=5$) substrates that were greater than 50% embedded ($n=11$). Juvenile Niangua darters used locations with depths of 21.6 ± 2.9 cm (range = 5-43) and velocities of 0.03 ± 0.02 m s⁻¹ (range = 0-0.2). Last, Juvenile Niangua darters predominately used locations where cover was absent, with only one juvenile using a location with a cover density greater than zero.

DISCUSSION

Water depth was most important for microhabitat-scale selection of adult Niangua darters of the Little Niangua River and Starks Creek in spring, summer, and fall. Although Mattingly & Galat (2002) did not examine Niangua darter microhabitat selection seasonally, they too found depth to be important during summer, and our results build on their conclusion to better identify seasonal habitat selection. Depth is important for

numerous other darters, often varying seasonally and ontogenically (Ross et al., 1992; Chipps et al., 1994; Rosenberger & Angermeier, 2003; Davis & Cook, 2010). Depth selection for lotic species is likely a tradeoff among numerous factors, and for Niangua darters may be largely determined by energetics, biotic interaction, and reproduction (Schlosser, 1987). During summer, when energy requirements are likely greatest due to warmer water temperature, Niangua darters selected relatively shallower depths which may have greater prey availability. Doisy & Rabeni (2001) found that richness and density of benthic invertebrates, which are common prey of Niangua darters, significantly decreased with depth. However given their large body size, Niangua darters may have avoided extremely shallow depths to evade wading predators (Power, 1987; Schlosser, 1988; Harvey & Stewart, 1991; Rosenberger & Angermeier, 2003). Similar to our findings, Mattingly & Galat (2002) also found Niangua darters in the Little Niangua River to select depths of 20 to 40 cm during summer. During cooler months, when energy requirements are low Niangua darters selected relatively deeper depths, but may have avoided the deepest habitats due to risk of piscivorous predation (Power, 1987; Schlosser, 1988; Harvey & Stewart, 1991; Doisy & Rabeni, 2001). A similar seasonal change in selected water depth has been observed in other darters (Ross et al., 1992; Skyfield & Grossman, 2008), while others show no difference among seasons (Rosenberger & Angermeier, 2003; Schofield & Ross, 2003).

During spring and fall adult Niangua darters selected similar depths, however in fall the range of selected depths was narrower than in spring. Wider ranges of depths in spring may be attributed to a disparity in spawning behavior as some individuals may have been selecting microhabitats suitable for spawning while others were not. Mattingly

et al. (2003) found that, like many other Etheostomid darters, reproductive Niangua darters used relatively shallow depths, compared to depths used during periods of reproductive inactivity (Winn, 1958; Collette et al., 1977). Therefore our findings may represent a combination of two separate behaviors. Disparity in behavior, or asynchronous spawning, likely also caused differences in top-ranked models between streams during spring. In addition to depth, adult Niangua darters in Little Niangua River selected less embedded locations during spring but showed no selection for embeddedness in Starks Creek. Most likely, we observed a greater proportion of adults selecting spawning sites in the Little Niangua River compared to Starks Creek, leading to disparity in top-ranked models during spring. Lithophilous spawners like the Niangua darter need coarse, water-swept substrates to deposit their eggs, keeping them oxygenated and free of silt (Balon et al., 1977). Furthermore discrete water temperatures, taken prior to each sample, indicate water temperature in Starks Creek was cooler during spring (13°C), compared to Little Niangua River (upper reach= 18°C , lower reach= 17.5°C). Therefore adult Niangua darters in Starks Creek may have spawned later in the season when water temperatures were more favorable (Mattingly et al., 2003).

Niangua darter habitat selection differed between streams during fall. Adult Niangua darters in Starks Creek selected for deep runs and deep pools, the latter having stronger selection, but used all habitats in proportion to their availability in the Little Niangua River. Adult Niangua darters used significantly deeper depths in Starks Creek in both summer and fall compared to Little Niangua River, likely due to differences in fish community, and subsequent biotic interaction, within streams. Starks Creek, although not significantly different with respect to depth and velocity, is classified as a third order

stream and has significantly narrower mean wetted width and presumably fewer, smaller piscivorous species (Vannote et al., 1980; Schlosser, 1982). The paucity of large predators may lead darters, and other lotic fishes, to exhibit divergent selection patterns than they would otherwise in the presence of large predators (Schlosser, 1987; Rahel & Stein, 1988). Niangua darters may have selected deeper habitats in Starks Creek due to the paucity of large predator (*Micropterus* species) and subsequent limited risk of predation making these habitats more preferable than in the Little Niangua River.

No single variable accurately predicted the presence of adult Niangua darters in our study and thus it is likely that other variables play a secondary or tertiary role in selection, suggesting multiple factors such as depth, velocity, substrate, and/or cover influence resource selection (Wood & Bain, 1995; Musselman & Brewer, 2009; Davis & Cook, 2010; Anderson et al., 2012). Mattingly & Galat (2002) found depth, substrate, and siltation to be important for summer microhabitat selection of Niangua darters throughout the Little Niangua River. The presence and distance to cover was not a significant factor in adult Niangua darters selection in our analysis which contradicts other studies which found that Niangua darters are often observed in great densities in habitats with large, silt-free, pebble-cobble substrates and/or with dense stands of water willow (*Justicia* spp.; Pflieger, 1978; Novinger & Decoske, 2013). Our results could be due to ineffectiveness of our metrics to capture this pattern. Furthermore, stream level below base flow in 2011 may have limited the availability of water willow and other emergent macrophytes.

We were unable to determine winter habitat use of Niangua darters. In January, we failed to detect adult Niangua darters and only found one juvenile. Lack of detection

was not likely due to sampling bias, as sampling conditions and methods were similar in winter compared to other months, with only temperature varying markedly. In addition to standardized samples we also returned to reaches on multiple occasions during winter to search more extensively but again failed to detect Niangua darters. Although we disturbed the substratum we believe that Niangua darters were utilizing local subsurface refuge that was difficult to observe. This behavior has been documented in small bodied salmonids and a few benthic lotic fishes but to our knowledge not previously documented in darters (Cunjak, 1996). Niangua darters may have migrated outside of sampled reaches but given the sedentary nature of the species (Pfleiger, 1997; McCleary, 2010), and other darter species (Roberts et al., 2008; Skyfield & Grossman, 2008; Dammeyer et al., 2013), we believe this was unlikely. Like other darters, Niangua darters have been found to move less than 100 m in a 12-month period (Pfleiger, 1997; McCleary, 2010).

In general, juvenile Niangua darters used habitats with shallower depths and slower velocities compared to adults. Other large darter species have exhibit similar ontogenetic shifts in habitat selection largely related to predator-prey interactions (Rosenberger & Angermeier, 2003). Juvenile Niangua darters may use shallower, slower habitats as nursery habitats where they may escape piscivorous predation but are at lower risk from avian detection due to small body size (Power, 1984). Small bodied juveniles may be energetically taxed in higher velocity environments (Mann & Bass, 1997). The limited number of juvenile observations, which precluded statistical analysis, could be attributed to their smaller body size, making there cryptic coloration more difficult to discern against the cherty substrate backdrop. In addition, juvenile Niangua darters generally inhabited extremely shallow habitats that are difficult to snorkel and thus were

examined aerially, likely decreasing detection. Last, large spatial and temporal fluctuations in the presence of juveniles have been observed in Niangua darters, and other lotic fishes, and thus our sample may have occurred during a period of low abundance (Schlosser & Toth, 1984; Schlosser & Angermeier, 1995; Novinger & Decoske, 2013).

Management actions that increase channel complexity and water depth may be beneficial for populations of Niangua darters. Although land use within the range of the Niangua darter currently dominated by forest, future landscape alteration has the potential to negatively affect Niangua darter populations since watershed-scale factors have been shown to have a greater influence on stream ecosystems (Alan et al., 1997). Increased sediment inputs and altered, flashy hydrology due to land use change will likely create less stable, embedded bedforms that will influence channel complexity, specifically shallow and intermediate depths, as well as prey availability and reproductive success of Niangua darters (Poff & Allan, 1995; Sutherland et al., 2002; Tipton et al., 2004). Fish communities may respond to alterations that increase agriculture to over 50% or 10-20% urban land use in the watershed (Wang et al. 1997; Yoder et al 1999). Therefore, limiting anthropogenic impacts to stream systems and balancing stream conservation with human use may benefit Niangua darters.

Management to increase or maintain riparian corridors, can limit streambank erosion, further reduce fine sediment inputs, and reduce water temperature (Sullivan & Adams, 1989; Rutherford et al., 2004). Additionally, riparian corridors provide woody debris to streams which has also been shown to increase local instream habitat complexity (Schlosser, 1991). Greater habitat complexity and buffered water temperature will likely improve population stability by reducing physiological stress and providing

refuge during taxing periods (Schlosser, 1991). Increasing or maintaining the availability of habitats with shallow and intermediate depths, for various life stages, with relatively coarse, silt-free substrates will likely benefit populations of Niangua darters (Rosenberger & Angermeier, 2003). Necessary dimensions and connectivity of riparian corridors are not well studied (Barton et al., 1985), but a corridor width of at least 10 m in areas devoid of riparian corridor may improve instream conditions for Niangua darter.

This research provides a better understanding of temporal requirement of the Niangua darter and insight on how small scale instream habitat may influence their distribution. However, further research on winter/seasonal ecology, movement, and home range is warranted to better understand Niangua darter biology and life history. This information, coupled with an understanding of larger-scale factors, will aid in recovery efforts by helping managers to better define and evaluate conservation criteria, leading to a better informed recovery effort. If efforts should fail to maintain and/or improve habitat or climate change and land use practice preclude Niangua darters from their current range, this information may be used to assist in evaluating sites for translocations. Understanding habitat requirements of species prior to translocation will likely improve success of such efforts (Armstrong & Seddon, 2008; Spurgeon et al. In press).

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Table 1: Microhabitat parameters measured at used and available Niangua darter locations in studied reaches of the Little Niangua River and Starks Creek, Missouri, 2011-2012.

Parameters	Description
DEP	Water depth (cm)
VEL	Water velocity ($m s^{-1}$) at 60% depth
CD	Density of cover items within $0.25m^2$ circle
	Visual classification of hydraulic channel unit modified after Arend (1999)
HGR	High gradient riffle; shallow with swift water velocity and considerable turbulence and gradient
LGR	Low gradient riffle; shallow with moderate water velocity, turbulence and gradient
DR	Deep run; deep with an even flow lacking pronounced turbulence
SR	Shallow run; shallow with an even flow lacking pronounced turbulence
DP	Deep pool; deep with little or no water velocity
SP	Shallow pool; shallow with little or no water velocity
	Visual classification of dominant substrate particle size modified after Wentworth (1922)
FI	Fines; particle intermediate axis less than 2 mm
GR	Gravel; particle intermediate axis 2-15 mm
PE	Pebble; particle intermediate axis 16-63 mm
CO	Cobble; particle intermediate axis 64-256 mm
BO	Boulder; particle intermediate axis greater than 256 mm
BE	Bedrock; solid rock substrate
	Visual classification of the percent to which larger particles (> 2 mm) were saturated by fine sediments (Platts et al., 1983)
E1	0 to 25 percent embedded
E2	26 to 50 percent embedded
E3	51 to 75 percent embedded
E4	76 to 100 percent embedded

Table 2: A priori resource selection models for adult Niangua darters in the Little Niangua River and Starks Creek, Missouri, 2011-2012.

Model abbr.	Model*
Depth	$= \beta_1(\text{DEP}) + \beta_2(\text{DEP}^2)$
Velocity	$= \beta_1(\text{VEL}) + \beta_2(\text{VEL}^2)$
Cover	$= \beta_1[\log_{10}(\text{CD})]$
Ch. Unit	$= \beta_1(\text{HGR}) + \beta_2(\text{LGR}) + \beta_3(\text{DR}) + \beta_4(\text{SR}) + \beta_5(\text{SP})$
Subst.	$= \beta_1(\text{GR}) + \beta_2(\text{PE}) + \beta_3(\text{CO}) + \beta_4(\text{BO}) + \beta_5(\text{BE})$
Embed.	$= \beta_1(\text{E1}) + \beta_2(\text{E2}) + \beta_3(\text{E3})$

*parameter names according to table 1

Table 3: Mean and standard error for depth (cm) and velocity (m s^{-1}) at 60% depth of random available locations within studied reaches of the Little Niangua River and Starks Creek, Missouri, 2011-2012.

	Little Niangua (upper) (n = 81)		Little Niangua (lower) (n = 213)		Starks Creek (n = 294)	
	\bar{x}	SE	\bar{x}	SE	\bar{x}	SE
Depth (cm)	35.8	3.3	41.9	2.5	32.8	1.9
Velocity (m s^{-1})	0.15	0.03	0.19	0.02	0.08	0.01

Table 4: AICc ranking of models hypothesized to influence microhabitat selection of adult Niangua darters in studied reaches of the Little Niangua River and Starks Creek, Missouri, 2011-2012.

Spring									
Little Niangua River					Starks Creek				
Model*	K	AIC _c	ΔAIC _c	w _i	Model*	K	AIC _c	ΔAIC _c	w _i
Depth	2	109.72	0.00	0.8096	Depth	2	109.13	0.00	0.9504
Embed.	3	112.68	2.96	0.1844	Ch. Unit	5	115.24	6.10	0.0449
Subst.	4	119.56	9.84	0.0059	Subst.	4	121.17	12.04	0.0023
Cover	1	129.53	19.81	0.0000	Cover	1	122.39	13.25	0.0013
Velocity	2	131.39	21.67	0.0000	Embed.	3	123.61	14.48	0.0007
Ch. Unit	5	131.93	22.20	0.0000	Velocity	2	124.49	15.36	0.0004
Summer									
Depth	2	40.26	0.00	0.9833	Depth	2	29.24	0.00	0.9978
Embed.	3	49.42	9.16	0.0101	Subst.	3	42.11	12.87	0.0016
Ch. Unit	5	50.28	10.02	0.0066	Embed.	3	44.92	15.68	0.0004
Subst.	4	60.03	19.77	0.0000	Ch. Unit	5	46.20	16.96	0.0002
Velocity	2	65.73	25.47	0.0000	Velocity	2	55.92	26.68	0.0000
Cover	1	65.81	25.55	0.0000	Cover	1	60.04	30.81	0.0000
Fall									
Depth	2	51.49	0.00	1.0000	Ch. Unit	5	48.42	0.00	0.8535
Embed.	3	73.03	21.54	0.0000	Depth	2	51.94	3.53	0.1464
Velocity	2	75.83	24.34	0.0000	Embed.	3	66.29	17.87	0.0001
Subst.	3	81.49	29.99	0.0000	Subst.	4	89.10	40.69	0.0000
Cover	1	82.42	30.93	0.0000	Velocity	2	91.29	42.88	0.0000
Ch. Unit	5	84.51	33.01	0.0000	Cover	1	93.52	45.11	0.0000

*Model abbreviation according to table 2

Table 5: Coefficients and standard error for parameters contained in top-ranked models of adult Niangua darters selection among seasons within studied reaches of the Little Niangua River and Starks Creek, Missouri, 2011-2012. Category E4 (76-100%) and Deep Pool categories are excluded as the reference.

Stream	Season	Parameter	β	SE	Wald's 95% CI		X^2	Pr > X^2
					Lower	Upper		
Little Niangua River	Spring	DEP	0.008	0.010	-0.012	0.027	0.61	0.4352
		DEP ²	-0.001	0.000	-0.002	-0.001	12.23	0.0005
		E1	1.685	0.623	0.464	2.905	7.32	0.0068
		E2	2.092	0.531	1.051	3.132	15.52	<.0001
	Summer	E3	1.321	0.531	0.280	2.361	6.19	0.0129
		DEP	-0.299	0.153	-0.599	0.000	3.83	0.0502
	Fall	DEP ²	-0.017	0.007	-0.031	-0.002	5.15	0.0233
		DEP	0.087	0.030	0.029	0.145	8.51	0.0035
Starks Creek	Spring	DEP ²	-0.004	0.001	-0.007	-0.001	8.18	0.0042
		DEP	0.033	0.009	0.015	0.051	12.63	0.0004
	Summer	DEP ²	0.000	0.000	-0.001	0.000	6.68	0.0097
		DEP	0.104	0.032	0.040	0.167	10.23	0.0014
	Fall	DEP ²	-0.002	0.001	-0.004	0.000	4.40	0.0359
		HGR	-55.264	11590.500	-22772.300	22661.800	0.00	0.9962
		LGR	-19.641	3944.400	-7750.600	7711.300	0.00	0.9960
		DR	-1.548	1.127	-3.756	0.660	1.89	0.1695
		SR	-37.013	5751.100	-11308.900	11234.800	0.00	0.9949
		SP	-54.695	8603.500	-16917.200	16807.900	0.00	0.9949
		DEP	0.088	0.022	0.045	0.130	16.44	<.0001
		DEP ²	-0.001	0.000	-0.001	0.000	8.82	0.0030

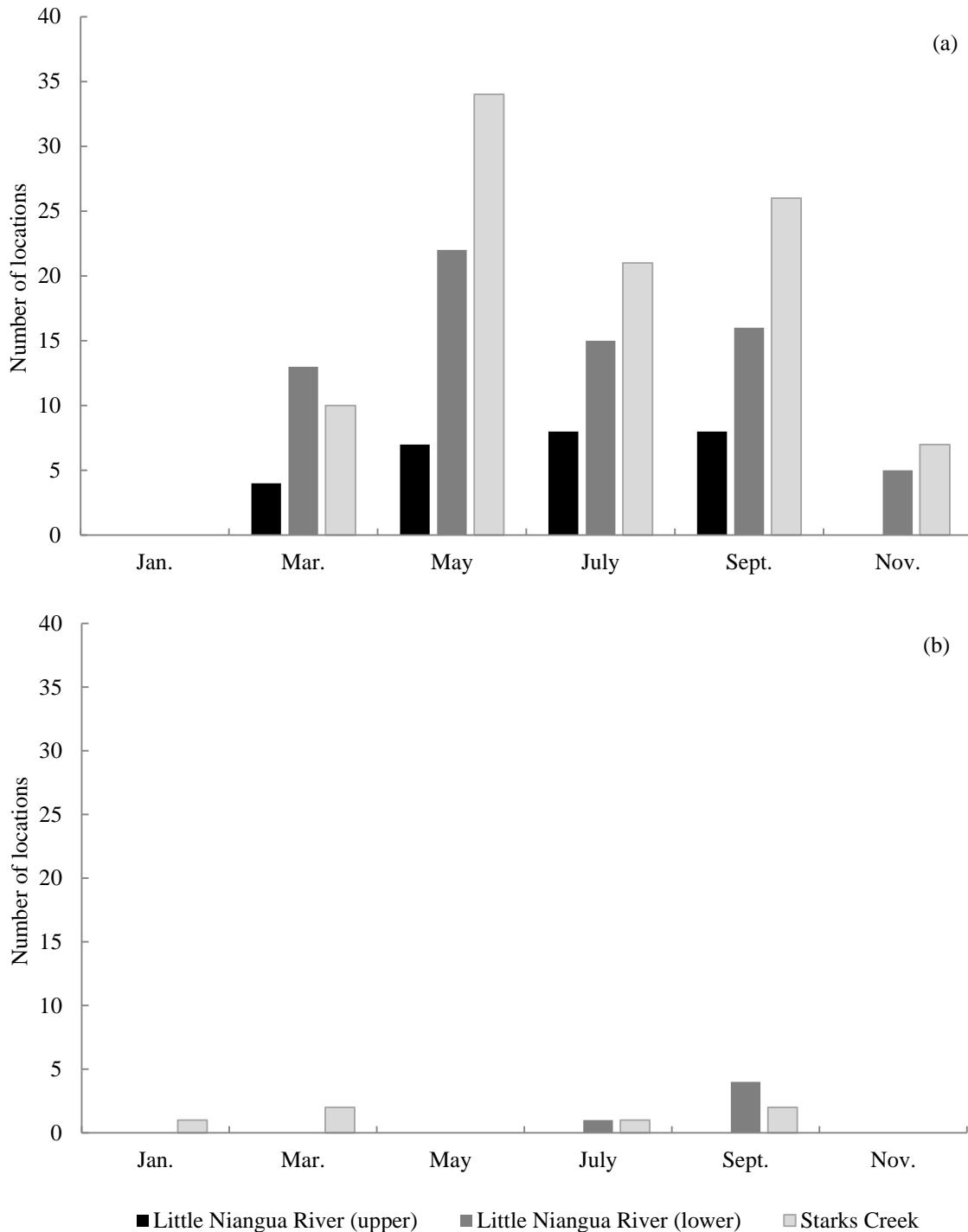


Figure 1: Number of locations collected for (a) adult and (b) juvenile Niangua darter during bi-monthly samples within studied reaches of the Little Niangua River and Starks Creek, Missouri, 2011-2012.

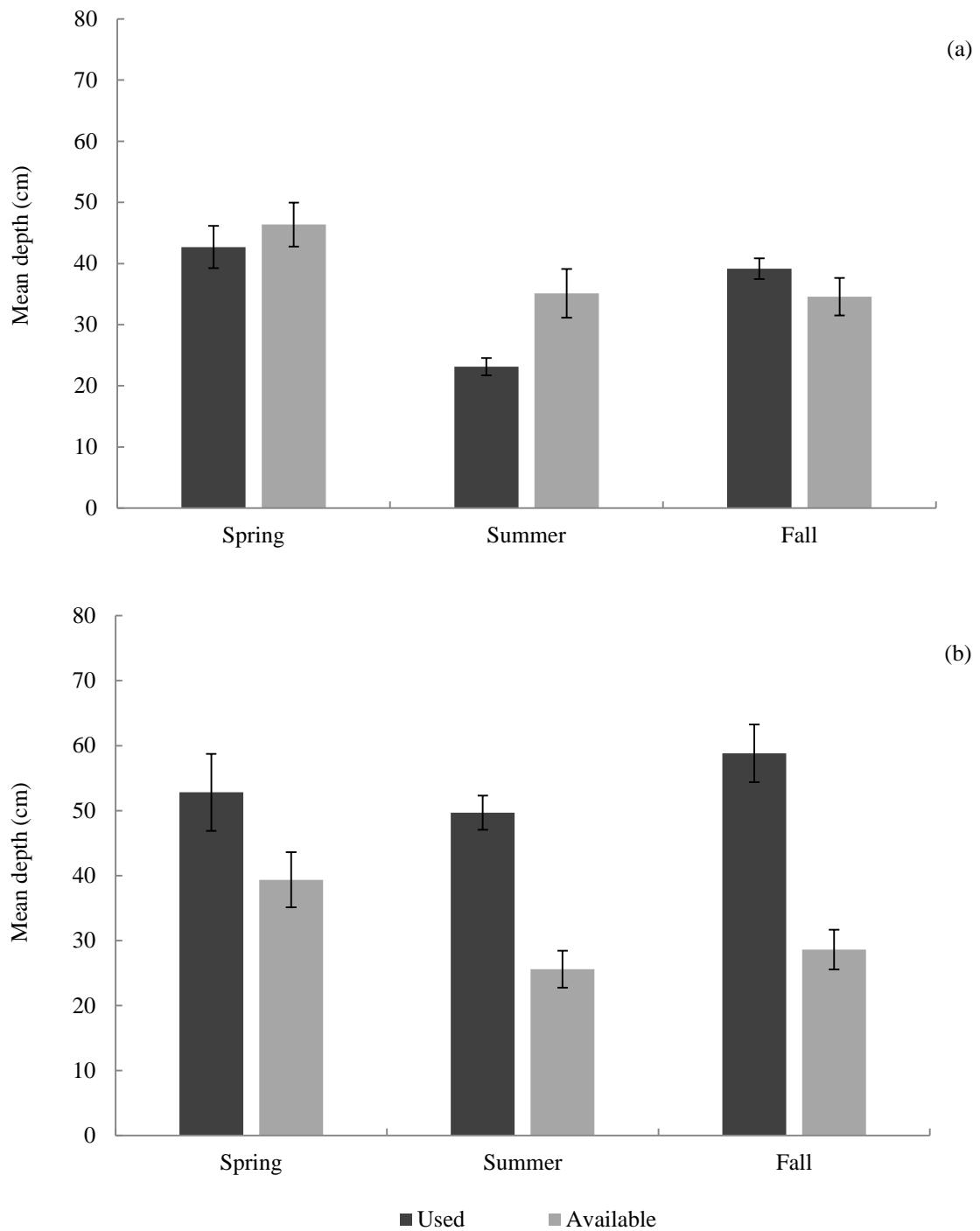


Figure 2: Mean and standard error for depth (cm) at locations used by adult Niangua darters and randomly selected locations that represent availability within studied reaches of (a) the Little Niangua River and (b) Starks Creek, Missouri, 2011-2012.

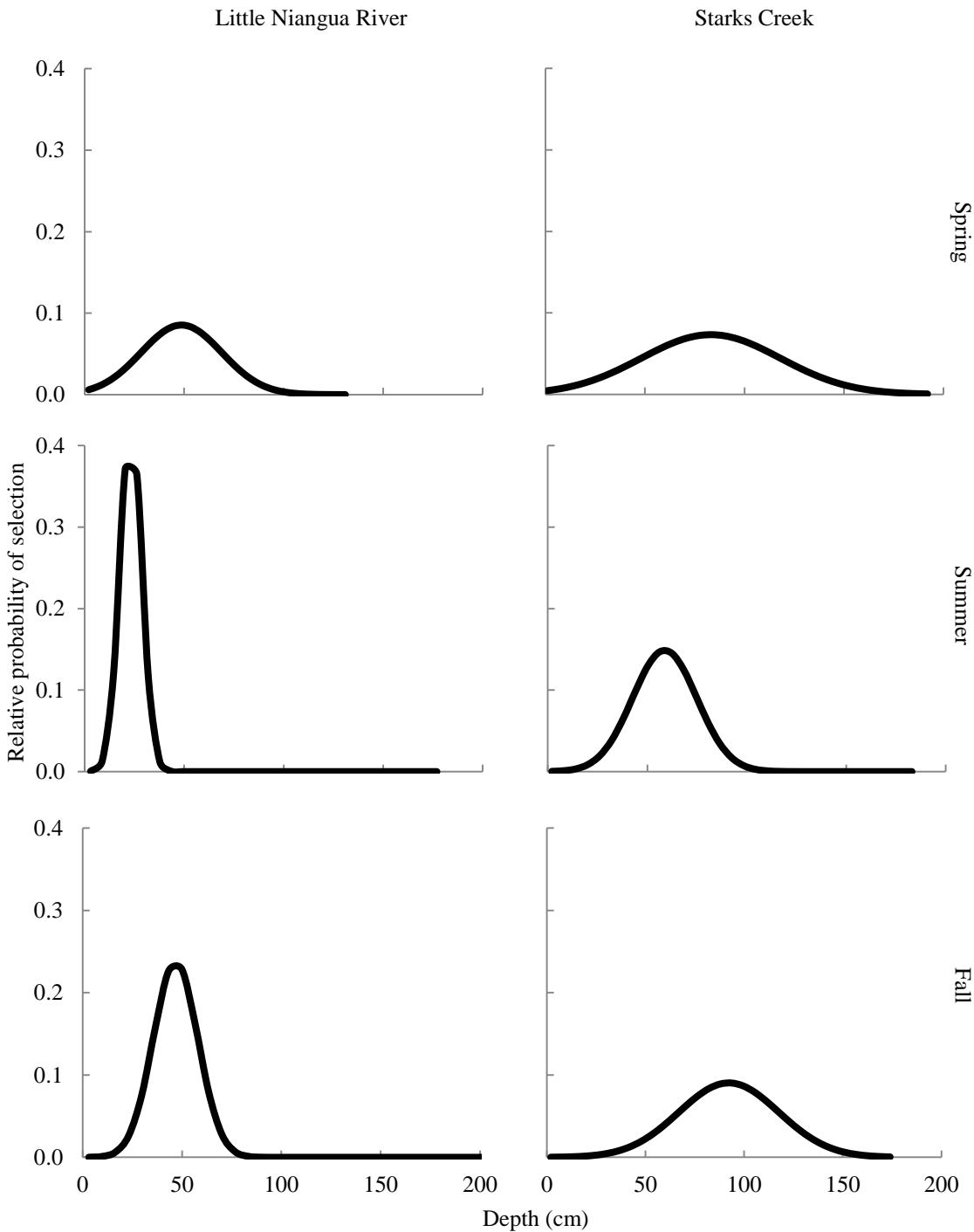


Figure 3: Resource selection functions of top-ranked models indicating the relative probability of selection among seasons: spring (top), summer (middle), and fall (bottom), across the range of depths used and available to adult Niangua darters in studied reaches of the (left) Little Niangua River and (right) Starks Creek, Missouri, 2011-2012.

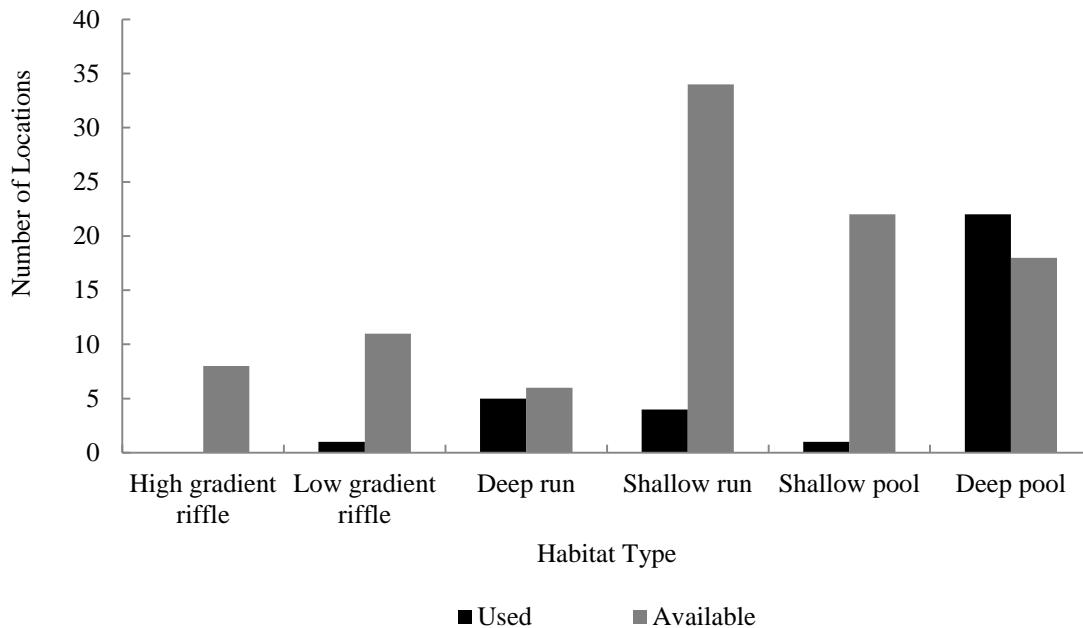


Figure 4: Number of locations used by adult Niangua darter used and the number of randomly selected locations that represent availability for each habitat type during fall, within 500 m reach of Starks Creek, Missouri, 2011-2012.