Modeling the Historic Distribution of Bluntface Shiner (*Cyprinella camura*) and Estimating Contemporary Detection and Occupancy Probabilities in Their Historic Range within Oklahoma

By

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Modeling the Historic Distribution of Bluntface Shiner (*Cyprinella Camura*) and Estimating Contemporary Detection and Occupancy Probabilities in Their Historic Range within Oklahoma

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Abstract

Bluntface Shiner (*Cyprinella camura*; BFS) is a fish that is endemic to the Arkansas, Lower- Mississippi, and Tennessee river basins, but exists as disjunct populations east and west of the Lower Mississippi River. Evidence suggests that BFS are experiencing declines across their known range, however the drivers of their decline are mostly unknown and knowledge of life history and habitat requirements of BFS remains limited. We aimed to address these knowledge gaps using species distribution modeling (SDM) and detection and occupancy modeling techniques. In chapter 1, we used SDMs to estimate the historic distribution of BFS across the entirety of its range, identify landscape scale factors which underly their distribution, and compare environmental conditions between the disjunct ranges. Our results revealed a naturally fragmented distribution in either BFS range, and that populations to the east of the Lower-Mississippi occupy streams with broadscale environmental conditions which differ from those to the west. In chapter 2, we focused on the historic distribution of BFS within Oklahoma, and we used a single season occupancy modeling framework while accounting for imperfect detection to estimate contemporary detection and occupancy probabilities for BFS. Our aims were to identify best gears (seine versus backpack electrofishing) for detecting BFS and the environmental conditions which influence gear effectiveness, as well as environmental factors which influence occupancy of the species in wadable streams of their historic range in Oklahoma. We found that detection of BFS varied with gear type depending on environmental conditions present at our sites, but seine produced overall higher detection probabilities. Our occupancy estimates were low despite our search within the historic distribution within Oklahoma. This suggests that BFS are rare and have seemingly declined across large portions of its range. Our findings call for focused conservation and management efforts for BFS across their entire range

Using species distribution modeling to estimate the historic distribution of Bluntface Shiner (*Cyprinella camura*)

Abstract

Bluntface Shiner (*Cyprinella camura*; BFS) is endemic to the Arkansas, Lower-Mississippi, and Tennessee river basins but exists as disjunct populations east and west of the Lower-Mississippi River. There is evidence that BFS are experiencing declines across their range; however, the drivers of their decline are unknown and knowledge of life history and habitat requirements of BFS remain limited. We aimed to address these knowledge gaps with species distribution modeling to 1) estimate the historic distribution of BFS; 2) identify landscape scale factors that underly their distribution; and 3) compare environmental conditions between the disjunct ranges. Our results revealed a naturally fragmented distribution both east and west of the Mississippi River, but populations to the east of the Lower-Mississippi occupy streams with broadscale environmental conditions that differ from those to the west. These findings provide a foundation for future research and conservation efforts toward BFS.

Introduction

Globally, freshwaters are remarkably biodiverse ecosystems with over 12,000 fish species existing in lakes and rivers despite these waters making up less than 0.01% of Earth's water supply (Dudgeon et al., 2006; Nelson, 2006). North American riverine ecosystems are hotspots for aquatic biodiversity with over 1,050 freshwater fish species representing 175 genera and 32 families (Lundberg et al., 2000; Masters et al., 1998). Yet, freshwater fauna are disproportionately impacted by human activities with extinction rates that exceed even the most impaired terrestrial systems (Ricciardi and Rasmussen, 1999). For example, nearly 46% of known freshwater fishes in North America are at risk of extinction, and the extinction rate has increased 8-fold in past decades. Therefore, freshwater fishes are among the most threatened taxa in the world (Jelks et al., 2008; Burkhead, 2012; Dudgeon et al., 2006). In light of unprecedented extinction rates, these animals require concerted research and conservation efforts now more than ever to maintain global biodiversity and provide economic and social value to humankind (Walsh et al, 2009).

Habitat alteration, degradation, and destruction by human activities are major threats to river ecosystems because they result in changes to natural stream hydrology, physiology, and water quality (Dudgeon et al., 2006; Paul and Meyer, 2001; Allan, 2004; Walsh et al., 2005). Streams are especially vulnerable given their innate connection to the surrounding landscape (Hynes, 1975). However, because natural processes of rivers are a function of the longitudinal (i.e., upstream to downstream), lateral (i.e., the floodplain and riparian zone), vertical (i.e., groundwater), and time dimensions (Ward, 1989), there is inherent difficulty in identifying causal relationships between the landscape, riverscape, and what factors are driving losses of biota. The challenge lies in determining the relevant spatial scale to investigate river-landscape interactions, measuring the influence of factors on the system, and gathering meaningful data across large areas (Wang et al., 2006). In recent years, advancements in geographic information technologies, accessibility to regional data bases, and the incorporation and application of landscape ecology to river ecosystems have helped alleviate these challenges (Wang et al., 2006; Brenden et al., 2006).

Species distribution models (SDMs; also known as ecological niche models [ENMs]) are valuables tools for incorporating species' niches into an understanding of large-scale distribution patterns. With SDMs, the niche of a species can be estimated by relating environmental covariates to occurrence records to estimate habitat suitability across a study area (Guisan and Thuiller, 2005). A widely used SDM called Maxent (Philips et al., 2017) follows the same principles, and it is advantageous for analyses utilizing museum data that is sparse in coverage or lacking absence data (Philips et al., 2006; Elith et al., 2011; Franklin, 2010). Maxent has been adapted for use in aquatic settings to identify important environmental associations of a species, elucidate the potential drivers of their decline, and to provide a basis to assess the extent of decline (e.g., Allen et al., 2022; Huang and Frimpong, 2015; McGarvey et al., 2021; Key et al., 2021; Taylor et al., 2018; Laman et al., 2018; Bouska et al., 2015; Labay et al., 2011; Pont et al., 2015). Distribution models can also be projected into novel spatial and temporal extents, providing inferences on the effects of environmental change or invasion into novel geographic regions (e.g., Sunblad et al., 2009, Bartnicki, 2021; Huang et al., 2016).

In this investigation, we used various Maxent packages within the R programming environment (R Core Team, 2021) to estimate the historic distribution of a river-dwelling fish, Bluntface Shiner (BFS; *Cyprinella camura),* which currently exists east and west of the Lower-Mississippi River in two disjunct regions. We were interested in characterizing the historic distribution within each disjunct range independently and with both ranges combined. Furthermore, we wanted to project the disjunct range models from the western and eastern ranges onto opposing areas to test similarity in important environmental variables and characterize overlap in suitable range. Therefore, our goals were to 1) estimate historic BFS range using environmental variables and location data; 2) identify important contemporary environmental variables associated with BFS distribution; and 3) compare estimated historic ranges of BFS to projections across the disjunct distribution of the species, as different regions are included or withheld. Our models would fill current knowledge gaps related to the environmental requirements of this understudied species and provide a reference for assessing the decline of this species across the entirety of its range.

Materials and Methods

Study Species

BFS currently exist in two disjunct regions on either side of the Lower-Mississippi River, an eastern segment in the Lower-Mississippi and Tennessee river basins and a western segment in the Arkansas River basin. According to the International Union for Conservation of Nature's global population assessment, BFS are of Least Concern, but the population trend is unknown. Populations are stable and secure in Kansas and Mississippi; however, BFS are susceptible to local declines (e.g., Cross and Braasch, 1968) and there is evidence of declines in several states. For example, in Louisiana, the BFS has not been captured in recent years (Robby Maxwell, Pers. Comm.). In Oklahoma, the majority of historic BFS records exist in tributaries of the lower Arkansas River; however, field surveys in this area in 2022 by the Oklahoma Department of Wildlife yielded detections of only two individual BFS (Anthony Rodger, Pers. Comm.). A number of streams that historically held BFS in Missouri and Oklahoma flow into the state of Arkansas, yet no BFS have been documented since the late 1960s (Robinson and Buchanan, 2020). As such, BFS are considered at-risk of extirpation in Oklahoma, Missouri, and Louisiana (Oklahoma Department of Wildlife Conservation, 2016; Missouri Department of Conservation, 2021, Louisiana Wildlife and Fisheries, 2022) and extirpated from Arkansas (Arkansas Game and Fish Commission, 2017). Given the ostensibly limited range and disjunct populations, any further declines are of great concern.

Early investigators of BFS phylogeny identified some apparent morphological differences between east and west populations, though genetic tests conducted at that time did not support subspecies designation (Gibbs, 1961; LeDuc, 1984). Though the populations occupy distant areas, there are some similarities in habitat. In both ranges, BFS are often associated with medium to large sized streams with riffles and runs (Wilkinson and Edds, 2001; Cross, 1954; Etnier and Starnes, 1993; Farr, 1996). In the lower-Mississippi River Basin, BFS occupy upland, headwater tributaries with swift flowing water over sand, mud, and gravel substrates (Johnston, 1999; Mayden, 1989; Farr, 1996; Ross and Brenneman, 2001; LeDuc, 1984). In the Arkansas River Basin, BFS occupy high gradient streams with flowing waters over gravel and rubble substrates (Fuselier and Edds, 1996; Metcalf et al., 2010), but are less abundant in lowland streams with sand and mud substrates (Cross and Calvin, 1971; Metcalf et al., 2010; Wilkinson and Edds, 2001). For either range, spawning occurs during spring and summer months (Distler, 2014; Robinson and Buchanan, 2020; Miller and Robinson, 2004; Ross and Brenneman, 2001; Etner and Starnes, 1993). As crevice spawners, BFS require streams possessing larger substrates (i.e., gravel) for successful reproduction (Mayden 1989; Johnston, 1999). Many BFS occurrence records are within larger, mainstem rivers with nearby access to smaller, tributary streams. In rivers of the Arkansas River Basin where drought conditions are common (Matthews, 1988), access to larger sized streams may be important for BFS survival and reproduction.

Study Area

The study extent was United States Geological Survey Hydrologic Regions Arkansas-Red-White (HUC-11), Lower-Mississippi (HUC-6), and Tennessee (HUC-8) (Figure 1A). Our models explored varying combinations of these regions. We created a model that treated the disjunct BFS populations as one with HUC-11, HUC-6 and HUC-8 joined together (ALL range). We also modeled the Arkansas-Red-White (ARW) and the Lower-Mississippi and Tennessee (LMT) ranges separately. The study grain was individual stream segments within NHDPlusV2 (McKay et al., 2012).

Arkansas-Red-White (ARW)

The ARW spans from the Great Continental Divide to the Mississippi River across eight states (Colorado, New Mexico, Kansas, Oklahoma, Texas, Arkansas, Missouri, and Louisiana). The mountainous western portion of the ARW reaches elevations up to 4000 m, and the streams flow over igneous and metamorphic geology (Cain, 1987; Kilsby et al., 1999). Within the central ARW are prairie streams, which are relatively low-elevation flatlands with bedrock and sedimentary-rock geology (Cain, 1987). Nearer to the Mississippi River basin are the Ozark Plateau and Ouachita Mountains, which have rolling hills and mountains that reach elevations from 150 to 800 m with limestone, sandstone, and sedimentary substrates (Arkansas Soil and Water Conservation Commission, 1990; Kilsby et al., 1999). In the mountainous western extremes of the ARW, hydrology is driven in large part by snowmelt runoff during spring-thaw, whereas in the prairie and Ozark regions, stream hydrology is primarily driven by rainstorm runoff in the spring and summer (Kilsby et al., 1999; Arkansas Natural Resource Commission, 2014). Within these areas, intermittent and ephemeral streams are common (Matthews, 1998).

Lower Mississippi and Tennessee (LMT)

The LMT spans from the Appalachian Mountains to the Gulf of Mexico across 10 states (Virginia, North Carolina, Kentucky, Tennessee, Georgia, Alabama, Missouri, Arkansas, Mississippi, and Louisiana). The Lower-Mississippi River Basin consists of meandering rivers abundant with tributaries, oxbows, and backwaters that ebb and flow across the Alluvial Plain from the confluence of the Ohio River toward the Gulf coast. Elevations range from sea level to 200 meters with uplands that surround the fluvial valley (Rittenour et al., 2007; Etneir and Starnes, 1993). The streams flow over fine sediments of sand, clay, silt, and some gravel deposits. The hydrology is driven by precipitation (Fisk, 1951; Agriculture Research Service, 2013) and flood conditions are common (Etneir and Starnes, 1993).

The Tennessee River Basin has the most geological complexity of the latter basins with highlands, plateaus, and mountainous expanses across the range. To the east is the Blue Ridge province that reaches elevations greater than 1,800 m, while to the west the Highland Region averages around 300 m elevation (Etnier and Starnes, 1993). In the low to moderate gradient streams, sand, gravel, and bedrock substrates of limestone, chert, sandstone, and shale predominate, whereas the high gradient mountains streams flow over bedrock and boulder substrates with some sand and gravel deposits (Rodgers, 1953; Etnier and Starnes, 1993). Winter snowfall and summer rain provide ample precipitation throughout the year (Federal Energy Regulatory Commission, 1981).

Data Collection and Preparation

To collect existing records of BFS, we queried online databases GBIF (https://www.gbif.org), Fishnet2 (http://www.fishnet2.net), IDigBio (https://www.idigbio.org), iNaturalist (https://www.inaturalist.org), and BISON (https://bison.usgs.gov). Given a marked paucity of public records for BFS in Oklahoma, we also requested records from Oklahoma state agencies (Oklahoma Department of Wildlife Conservation and Oklahoma Conservation Commission) and natural history museums (University of Oklahoma Sam Noble Museum and Oklahoma State University). We cleaned the dataset by removing replicates (199 records) and records that lacked both assigned coordinates and locality descriptions (30 records). For records with only locality descriptions, we used GEOLocate (http://www.geo-locate.org) to geo-reference 126 records. Using ArcMap v10.8.2 (ESRI, 2011), we spatially joined all occurrence records to NHDPlusV2 stream-flowlines to validate that the coordinates fell upon the closest stream segment, and if applicable, that the occurrence record was joined to a stream stated in the locality description. We removed records with an invalid locality and replicate georeferenced sites (54 records). In total, 762 occurrence records of BFS remained for model analysis (Figure 1B).

Environmental Variable Selection

We selected environmental variables from NHDPlusV2, StreamCat (Hill et al., 2016), and the Stream Classification System (SCS; McManamay and DeRolph, 2019) to inform our distribution models. NHDPlusv2 contains environmental information for streams of the conterminous United States at a 1:100,000 scale. StreamCat and the SCS were built upon the NHDPlusv2 stream network and provide a suite of additional environmental variables. Based on existing literature on BFS life history, we selected elevation, percent sand, percent clay, rock depth, slope, divergence type, and valley confinement at the catchment level, while at the watershed scale, we included total drainage area (Table 1). We joined environmental covariate data to their respective stream segments within the study area using a common identifier (COMID) among all datasets. To avoid issues arising from multicollinearity, we retained any covariate with Pearson's correlation coefficient $r < 0.7$ (Dormann et al. 2013).

Distribution modeling

We first employed the package *ENMeval2.0* (v2.0.4; Kass et al., 2021) to produce a suite of Maxent models and select the best performing model for further analysis. ENMeval2.0 allows for user-defined partitioning of occurrence records for spatial cross validation (Veloz, 2009). We structured our dataset in samples-with-data (SWD) format to represent each stream segment. Prior to modeling, we omitted any stream segments that were represented more than once (as an occurrence or background record) and removed stream segments that lacked complete coverage of environmental covariates. We specified the models to run with the "maxent.jar" implementation, regularization multipliers (*rm*) 1, 2, 3, 4, and 5, and four distinct feature class (*fc*) combinations: linear (L); linear + quadratic (LQ); linear + quadratic + hinge (LQH); and hinge (H). To gauge model performance, we used a k-fold cross-validation with user-defined spatial partitions. We partitioned occurrence and background groups by USGS HUC-8 (Figure 1C). In these ways, we balanced model complexity with potential for overfitting.

We selected the top *ENMeval2.0* models based on a ΔAICc score of zero (Warren and Seifert, 2011) and proceeded with further analysis in R package *dismo* (Hijmans et al., 2017) using the "maxent.jar" implementation. This package allowed for more detailed exploration of the top models, as well as projection of models into opposing ranges. The background number was set to 250,000, which allowed all background segments to be used in the AWR and LMT models but required a random subsampling of background for the ALL model. We used jackknife tests to measure variable importance. The raw outputs of suitability were cloglog transformed and joined to NHDPlusV2 stream segments for visualization. For the purposes of depicting these results, we adopted the following scale: Values of 0 – Minimum Training Presence (MTP; e.g., 0.218115 for the ALL model) as unsuitable, MTP - 0.500000 as low suitability, 0.500001 – 0.750000 as moderate suitability, and 0.750001 – 1.0000 as high suitability.

Results

Model inputs

Of the 762 total occurrence records, 431 records located in the ARW, while 331 were in the LMT. The earliest BFS record was 1911 and the latest record was 2018. Environmental variable value ranges differed (save for the categorical variables) between the ARW and LMT (Table 2). The ALL included 508 occurrence stream segments (repeat records at the same location were treated as one stream segment) with 391,361 background stream segments. The ARW included 225 occurrence stream segments with 185,507 background stream segments. The LMT included 283 occurrence stream segments with 197,240 background stream segments.

The best fit model for the ALL range involved a *fc* combination of LQH, and an *rm* of 4. The best fit model for the AWR range involved a *fc* combination of LQH and *rm* of 3. The best fit model for the LMT involved a *fc* combination LQH and *rm* of 2 (Table 3). Each model had good evaluation metrics with area under the curve (AUC) values close to one and omission rates not much higher than the 10% target, meaning that there was good fit between the models and the data, without overfitting.

ALL Distribution

The ALL model demonstrated an expansive distribution that spanned from the central watersheds of the Arkansas-Red-White to the coastal watersheds of the Lower-Mississippi and the far eastern watersheds of the Tennessee River Basin (Figure 2). Several watersheds had stretches of moderate to highly suitable stream segments separated by many low to unsuitable stream segments. The Red Headwaters, Red-Washita, Red-Sulphur, Upper-Cimarron, Lower-Cimarron, North-Canadian, Lower-Canadian, Middle-Arkansas, Lower-Arkansas, and Upper-White in the west, and the Louisiana Coastal, Lower-Red-Ouachita, Lower-Mississippi, Boeuf-Tensas, the Middle-Tennesse-Hilwassee, and the upper Tennessee watersheds in the east fit this description. There were watersheds that represented areas with noticeably high suitability. The Arkansas-Keystone, Neosho-Verdigris in the west, and Lake Maurepas, Pearl, Big Black, the Yazoo, Hatchie, St. Francis, Lower-Tennessee, and Middle-Tennessee- Elk in the east had long stretches of moderate to highly suitable stream segments across most of the watershed.

ARW Distribution

The ARW model had a distribution confined to the central portion of the range (Figure 3). Like the ALL model, the Neosho-Verdigris had moderate to highly suitable stream segments across the entire watershed. The Red-Headwaters, Red-Washita, Red-Sulphur, Lower-Canadian, North-Canadian, Lower-Cimarron, Middle-Arkansas, Arkansas-Keystone, Lower-Arkansas, and Upper-White had some highly suitable stream segments but were mostly low suitability or unsuitable. The western most watersheds and large portions of the watersheds nearest to the Lower-Mississippi River Basin were unsuitable for BFS.

LMT Distribution

The LMT model had a narrow distribution within the range (Figure 4). The most suitable watersheds were the Hatchie, Yazoo, Big Black, Lake Maurepas, and Pearl. Within the Yazoo, only streams to the east of the watershed were suitable, and within Lake Maurepas, only stream segments in the northern parts were suitable. The watersheds St. Francis, Lower-Red-Ouachita, and Louisiana Coastal were mostly unsuitable or low suitability stream segments with some moderate to highly suitability stream segments. Coastal watersheds, portions of watersheds along the Lower-Mississippi River, and the majority of the Tennessee River Basin were unsuitable areas.

Projected ranges

We transferred the ARW and LMT models onto opposing ranges to test how BFS suitability would compare between the disjunct populations if projected onto each other's ranges. In both cases there were considerable shifts in suitable stream segments compared to the historic ranges. When the LMT model was projected onto the western range, there were few suitable stream segments across the entire range (Figure 5). When compared to the historic western distribution, the range shifted along the eastern edge of the range. There were some branches of low suitability stream segments that stretched into the center of the range; however, most of the higher suitability stream segments were along the border with the Lower-Mississippi River Basin. When the ARW model was projected onto the eastern range, the suitable streams shifted to the extremities of the range (Figure 6). The suitable stream segments were confined to small areas to the northeast of the Lower-Mississippi River Basin and unlike the historic eastern range, the unsuitable watersheds of the Tennessee River Basin now had expansive stretches of highly suitable stream segments. Save for the coastal watersheds and along the Lower-Mississippi River, the suitable habitat ranges became inverted when the ARW was projected onto the eastern range.

Response Curves

There were eight environmental variables (ElevCat, SandCat, ClayCat, RckDepCat, Slope, Divergence, Confinement, and TotDA) that informed our models. The top model for each study range had a slightly different combination of variables with highest percent contribution and permutation importance (Table 3). In the ALL model, TotDA was the most influential variable, followed by ElevCat, ClayCat, Divergence, Slope, SandCat, RckDepCat, and Confinement (Table 4). In the ARW model, TotDA was the top variable, followed by ElevCat, ClayCat, SandCat, RckDepCat, Slope, Divergence, and Confinement respectively. In the LMT model, TotDA was again the top variable, followed by ClayCat, RckDepCat, Divergence, ElevCat, Slope, Confinement, and SandCat. Hereafter, we compare variables of the top models that had a percent contribution or permutation importance value above 10.

In each of our models, total drainage area was the variable with the highest percent contribution. In the ALL and LMT ranges, there was a gradual increase in suitability with increased total drainage sizes. In ARW range, the response curve was unimodal with the highest suitability at ~100,000 km^2 (Figure 7A). Elevation was the second highest contributing variable for the ALL and ARW ranges, though it was not of importance in the LMT. In the ALL and ARW ranges, the response curves had slight increases at low elevations before gradually decreasing at higher elevations. In the LMT range, the response curve had its highest suitability at low elevations before sharply decreasing in suitability at higher elevations (Figure 7B). Clay percentage within the catchment was the third highest contributing variable for the ALL and ARW ranges and was the second highest contributing variable for the LMT range. In the ALL range, the response curve was unimodal with the highest suitability at ~20%. The ARW range response curve was also unimodal with suitability that peaked at ~40%. In the LMT range, suitability had a sharp decline as percent clay reached ~40% (Figure 7C). Rock depth was the third highest contributing variable for the LMT range while contributing little to the ALL and ARW ranges. In the LMT range, the response curve had a sharp increase at rock depths greater than 120 cm, whereas the ALL range had a gradual increase in suitability as rock depth increased. Meanwhile, the ARW range had a unimodal response curve with suitability increasing to its maximum at ~ 130 cm before declining at greater depths (Figure 7D).

Discussion

The Bluntface Shiner is an under studied species that may be experiencing declines across large portions of its range. Existing knowledge is limited to microhabitat use, so an improved understanding of this species' broadscale habitat associations and life history is needed to inform conservation efforts (Matthews, 1998; Cooke et al., 2012). We used species distribution models to address knowledge gaps on BFS habitat requirements and to characterize the historic distribution of BFS. Our results revealed the historic distribution in two disjunct areas – east and west of the Mississippi River – that differed in environmental conditions at the landscape scale. Within each of these areas, BFS habitats varied in size and were patchily distributed across the riverscape. These findings raise further questions regarding the currently known BFS distribution and its conservation status.

Our visualization of the historic BFS distribution provided a unique view of the species distribution, and we were able to identify several consistently suitable and unsuitable watersheds for BFS in both their ranges. Watersheds that were estimated to be suitable were those along the eastern side of the Lower Mississippi River in the LMT and the central-eastern watersheds in the ARW. The Lower Mississippi River presented a major division between suitable habitats in the east and west, while coastal watersheds (for the LMT) and upland areas to the farthest extents in the LMT (to the east) and ARW (to the west) were consistently unsuitable. These patterns of suitable and unsuitable stream segments were most explained by various catchment- and watershed-scale environmental conditions, which differed between the LMT and the ARW. From these results, we estimated that these disjunct ranges lack suitable habitats that would have linked the two populations. These findings are potentially valuable for biogeographic and phylogenetic investigations of BFS.

A glaring feature of the estimated historic distribution was that suitable streams for BFS were disjunct and relatively limited in range within the LMT and ARW. This patchy matrix of suitable habitats was not entirely surprising given that riverine ecosystems naturally vary in physical, hydrological, and chemical conditions as they flow from headwaters to the river mouth (Vannote et al., 1980; Hynes, 1975), and that segments along the stream's length may have distinct physical conditions from adjacent segments which together form a highly heterogenous mosaic of habitats across the stream network (Poole, 2002). However, the naturally fragmented distribution may suggest that BFS are sensitive to declines (e.g., Fagan et al., 2002; 2005), particularly in the context of the contemporary landscape. The ARW and LMT have experienced major alteration over the past 100 years because of anthropogenic activities involving the construction of dams, levees, and reservoirs accompanied by increasing urbanization and agricultural land use (Simon et al., 2020; Remo et al., 2009; 2018; Turner and Rabalias, 2003; Yang et al., 2023; Pennock, 2017). These factors influence species distributions through physical blockage of movement or by deleterious changes to natural conditions, which together harm sensitive species like BFS (Jester et al., 1992) and likely fragment the distribution of BFS more than our estimates.

Our models captured various stream-network features that are influential to fish occurrence and distribution and potentially important for BFS population dynamics. For example, the highest contributing variable across the ALL, LMT, and ARW models was increasing total drainage area, which resulted in higher suitability for BFS. This variable may be important for various reasons given that drainage area is associated with factors including water volume and discharge, stream depth, size, length, and various physicochemical properties that change with increasing size (Allan et al., 2021; Matthews, 1998; Matthews and Robinson, 1988). Our results concur with previous reports that BFS are most associated with medium to large-sized streams (Wilkinson and Edds, 2001; Cross, 1954; Etnier and Starnes, 1993; Farr, 1996). Likewise, in a separate study where we investigated occupancy of BFS in the Arkansas River Basin, we found that occupancy probability of BFS increased with total drainage area (see Chapter 2). In that investigation, the result was attributed to larger drainage areas providing BFS with greater access to important habitats (e.g., spawning, rearing, refuge) and opportunities for dispersal and recolonization.

Here, our results expanded upon this idea by including BFS populations from the LMT range, but also by revealing various physical stream-network attributes relevant to metapopulation dynamics within the estimated distributions. Consider the spatial arrangement and organization of the estimated distributions, wherein the most suitable stream segments were generally elongate, moderately branched (relative to the LMT) and expansive in the ARW, while the most suitable stream segments in the LMT were generally shorter, highly branched, and compact. These distribution patterns captured varying degrees of drainage size, compactness, density, and connectivity between suitable tributaries and mainstems, which are important to stream-fish populations (Benda et al., 2004; Walters et al., 2003; Osborn and Wiley, 1992; Campbell-Grant et al., 2007; Smith and Kraft, 2005) for dispersal among habitats (Brown and Swan, 2010; Eros and Campbell-Grant, 2015; Hugueny et al., 2010; Shao et al., 2019; Altermatt, 2013). It is conceivable that the spatial arrangement of the suitable streams is influential to the distribution of BFS, particularly in the context of natural disturbances that are common within the range of this species (Matthews, 1988; Bryant, 2010).

Projected models

The disjunct BFS range presented an interesting opportunity to test model performance when transferred to opposing ranges. We expected some overlap in suitable conditions between the LMT and ARW given similarities in important habitat covariates which explain the distributions; however, the projected models resulted in severe mismatches in the forecasted and known distributions. For example, when the ARW model was projected onto the LMT, the estimated distribution shifted dramatically toward the Tennessee River Basin. Likewise, the LMT model projected onto the ARW resulted in major shifts toward the Lower-Mississippi River Basin. These results likely reflect considerable differences in broadscale environmental conditions between the ARW and LMT (Table 2). Though BFS from either range are said to have similar microhabitat associations, our results suggest that BFS east and west of the Lower-Mississippi occupy different habitats at larger spatial scales. Another explanation for the mismatch in models is the spatial-scale of our analysis wherein catchment- and watershed-level covariates did not reflect fine-scale environmental factors, which more directly act at the grain of stream segment (e.g., Luoto et al., 2002). More generally, our results emphasize the need for caution when extrapolating models across vast distances (Werkowska et al., 2017; Elith and Leathwick, 2009).

Conclusion

Knowledge of species distributions and habitat-relationships are fundamental to better ecological understanding and conservation of species (Elith et al., 2006), yet many threatened river fishes suffer from lack of information regarding these key aspects (Cook et al., 2012). Species distribution models are particularly useful for addressing such knowledge gaps, and our models demonstrated this utility.

Current knowledge of BFS life history is restricted to microhabitat associations and while this information is valuable, effectiveness of conservation and management strategies are maximized when incorporating processes acting at the landscape-scale (Roni et al., 2011). Along these lines, our analysis included both disjunct distributions of BFS, and we revealed important landscape-scale features that characterized the historic distributions in both ranges; therefore, these findings have considerable applications for conservation and management of BFS. For example, the distribution maps can be used as a baseline to gauge the extent of range loss in areas where BFS have experienced declines and identify at-risk populations most in need of monitoring and management. Impaired watersheds, which theoretically hold suitable habitats, present opportunities for investigating drivers of decline. Such investigations should consider anthropogenic land-use and river-modifications (e.g., agriculture and urbanization, dams and impoundments), as these are pervasive features across the contemporary distribution of BFS. In this circumstance, the historic distributions are a useful reference to assess how anthropogenic features are spatially related within BFS native range. Additionally, the distribution maps revealed hotspots of suitability. These areas could help narrow down search efforts for BFS. Surveys within suitable streams where few or no records of BFS have been documented are worthwhile and finding them in these areas would be encouraging for BFS conservation. Likewise, determining locations with healthy BFS populations provide great opportunities to investigate habitat factors required to sustain those populations.

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Table 1. Environmental variables related to Bluntface Shiner biology selected from various datasets (NHDPlusv2, StreamCat, Stream Classification System) and linked to stream segments in the study area.

Environmental variables					
Abreviation	Description	Unit	Scale	Source	Data Type
ElevCat	Elevation	m	Catchment	StreamCat	Continuous
SandCat	Percent sand	%	Catchment	StreamCat	Continuous
ClayCat	Percent clay	%	Catchment	StreamCat	Continuous
RckDepCat	Rock depth	cm	Catchment	StreamCat	Continuous
Divergence	Stream divergence	-	Segment	SCS	Factor
Slope	Slope	-	Segment	SCS	Continuous
Confinement	Valley confinement	-	Segment	SCS	Factor
TotDA	Total drainage area	km^2	Watershed	NHDPlusv2	Continuous

Table 2. Summary of environmental variables for the ALL, ARW, and LMT ranges, reported as the mean (minimum - maximum).

Variable	ALL	ARW	LMT
ElevCat (m)	1,431 (-374 - 4,355)	1,991 (-374 - 4355)	873 (-177 - 1923)
SandCat (%)	42.4 (-6.04 - 90.4)	43.6 (-3.2 - 90.4)	41.4 (-6.04 - 88.3)
ClayCat (%)	37.3 (-1.68 - 80.3)	34.1 (-1.68 - 69.8)	40.5 (0.741 - 80.2)
RckDepCat (cm)	97.5 (30.1 - 165)	97.4 (31.4 - 163)	97.5 (30.1 - 165)
Divergence	3.5 (-0.7 - 7.7)	3.5 (-0.7 - 7.7)	3.5 (-0.7 - 7.7)
Slope	0.77 (-0.302 - 3.32)	0.02 (-0.004 - 0.044)	1.51 (-0.3 - 3.32)
Confinement	1.5 (-0.3 - 3.3)	1.5 (-0.3 - 3.3)	1.5 (-0.3 - 3.3)
TotDA (km^2)	882,702 (-313,339 - 3,446,726)	198,710 (-39,742 - 437,163)	1,566,694 (-313,339 - 3,446,726)

Table 3. Model metrics for the first five models for the ARW and LMT. Model parameters are represented by feature class (fc) combinations, regularization multiplier (rm), and coefficients (ncoef). Top models were selected based on Akaike Information Criterion (AIC), but other model performance metrics used as references included: area under curve training values (auc.train), the average difference (auc.diff.avg), and ten percent omission rate (or.10p).

fc	rm	auc.train	auc.diff.avg	or.10p.avg	or.10p.sd	or.mtp.avg	or.mtp.sd	AICc	ΔAICc	w.AIC	ncoef
ALL											
LQH	4	0.90103	0.0689	0.14686	0.22269	0.0024706	0.01251	11960	0.0000	1.00	83
LQH	1	0.91364	0.06351	0.17416	0.25387	0.0024706	0.01251	11975	15.803	0.00	144
LQH	2	0.9097	0.06443	0.17493	0.25353	0.0016026	0.011103	11982	22.144	0.00	127
H	1	0.91362	0.06338	0.175	0.25389	0.0024706	0.012514	12000	40.071	0.00	150
LQH	3	0.90565	0.06692	0.17306	0.25427	0.0024706	0.012514	12001	41.256	0.00	115
ARW											
fc	rm	auc.train	auc.diff.avg	or.10p.avg	or.10p.sd	or.mtp.avg	or.mtp.sd	AICc	ΔAICc	w.AIC	ncoef
LQH	3	0.94154	0.08145	0.17676	0.29905	0.00644	0.02270	4621.2	0.0000	1.0000	54
H	3	0.94370	0.07902	0.19949	0.30394	0.05189	0.21297	4641.8	20.537	0.0000	60
LQH	4	0.93844	0.08215	0.20165	0.30439	0.00644	0.02270	4658.1	36.878	0.0000	54
LQH	5	0.93653	0.08096	0.20018	0.30421	0.00644	0.02270	4666.1	44.910	0.0000	47
H	4	0.94172	0.07850	0.20165	0.30439	0.05189	0.21297	4708.8	87.590	0.0000	69
LMT											
fc	rm	auc.train	auc.diff.avg	or.10p.avg	or.10p.sd	or.mtp.avg	or.mtp.sd	AICc	ΔAICc	w.AIC	ncoef
LQH	2	0.93847	0.03538	0.13957	0.22086	0.00641	0.03269	6064.7	0.0000	1.0000	64
H	3	0.93670	0.03579	0.13573	0.21327	0.00641	0.03269	6090.3	25.607	0.0000	65
H	2	0.93818	0.03544	0.14198	0.22175	0.00641	0.03269	6091.7	27.066	0.0000	72
LQH	5	0.93315	0.03786	0.12788	0.20833	0.00641	0.03269	6099.7	35.034	0.0000	52
LQH	3	0.93703	0.03620	0.13188	0.20725	0.00641	0.03269	6109.2	44.483	0.0000	70

Table 4. Highest ranked environmental variables determined by percent contribution (Contr. %) and permutation importance (Perm. imp.). Variables with values greater than 10 in at least one model highlighted with an asterisk.

	ALL		ARW		LMT	
Variable	Contr. %	Perm. imp.	Contr. %	Perm. imp.	Contr. %	Perm. imp.
TotDA*	54.4	48.3	47.1	46.2	36	37.8
ElevCat*	20.1	17.3	19.3	34.6	6.7	10.6
ClayCat*	12	21.1	15.7	8.1	28.8	34.2
RckDepCat*	0.1	0.1	6.1	2.6	13.9	6.4
SandCat	3.8	3.7	6.3	5.5	1.4	1.8
Slope	3.8	4.7	4.9	2.2	3.9	2.6
Divergence	5.8	4.7	0.5	0.2	7.4	5.5
Confinement	0	0	0.2	0.6	1.8	1.1

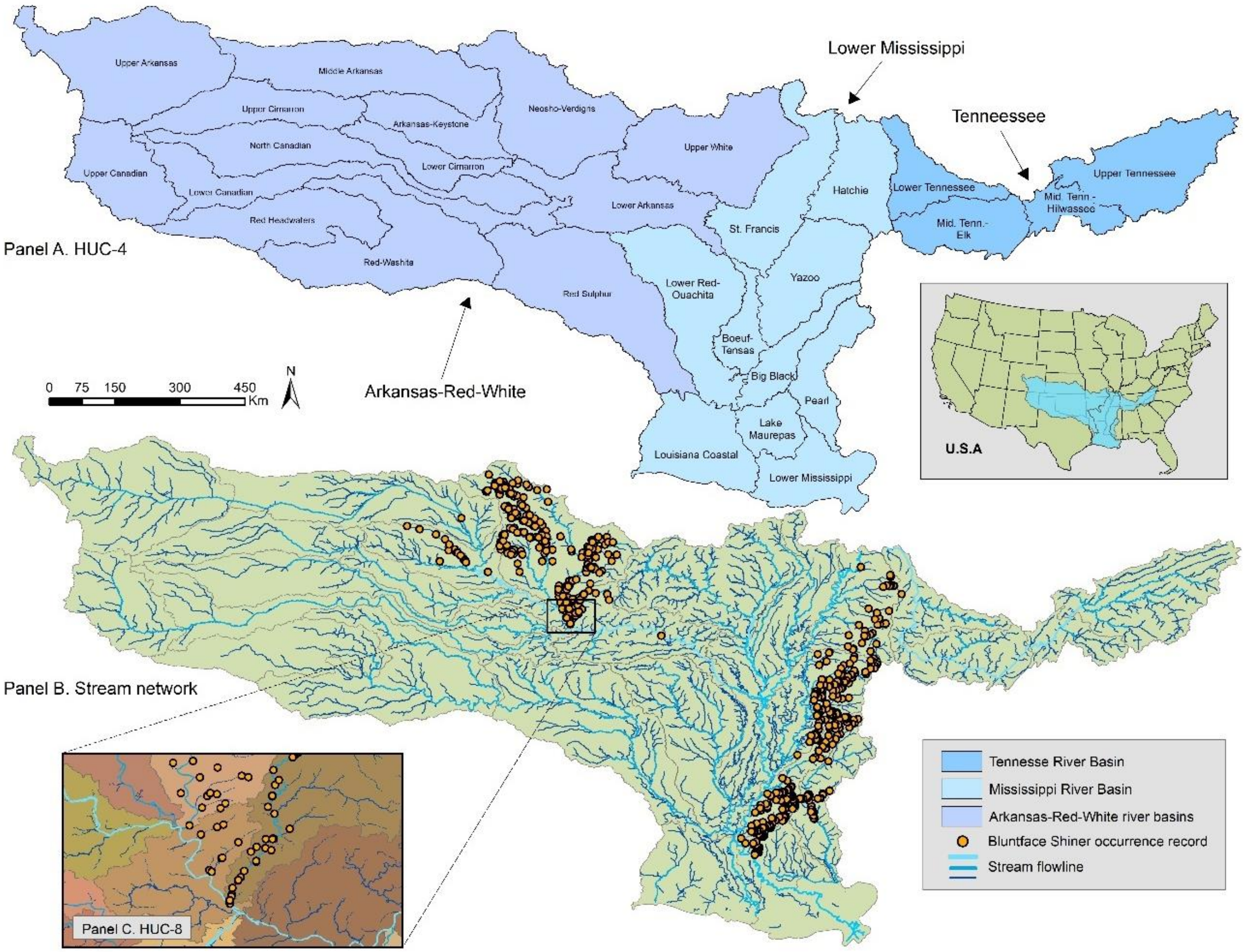

Figure 1. Study area encompassing watersheds of the Arkansas-Red-White, the Lower-Mississippi, and the Tennessee river basins (Panel A). The distribution models were produced with streamflow-lines (stream orders 1-4 not shown here) and occurrence records (Panel B). The occurrence and background records were divided by HUC-8 for k-fold cross-validation (Panel C).

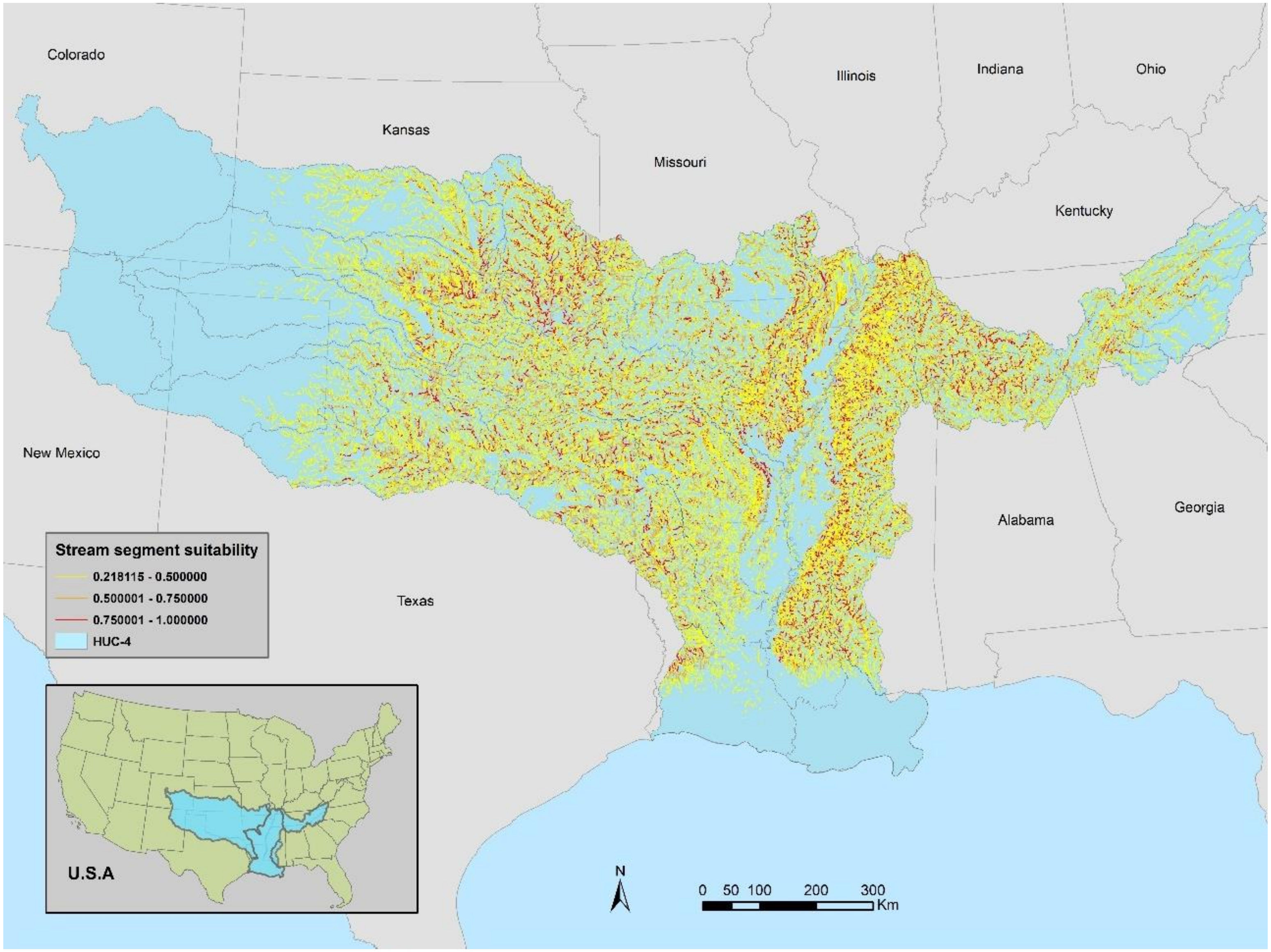

Figure 2. ALL model estimated stream segment suitability with occurrence records of Bluntface Shiner and environmental variables from the Arkansas-Red-White, Lower-Mississippi, and Tennessee river basins

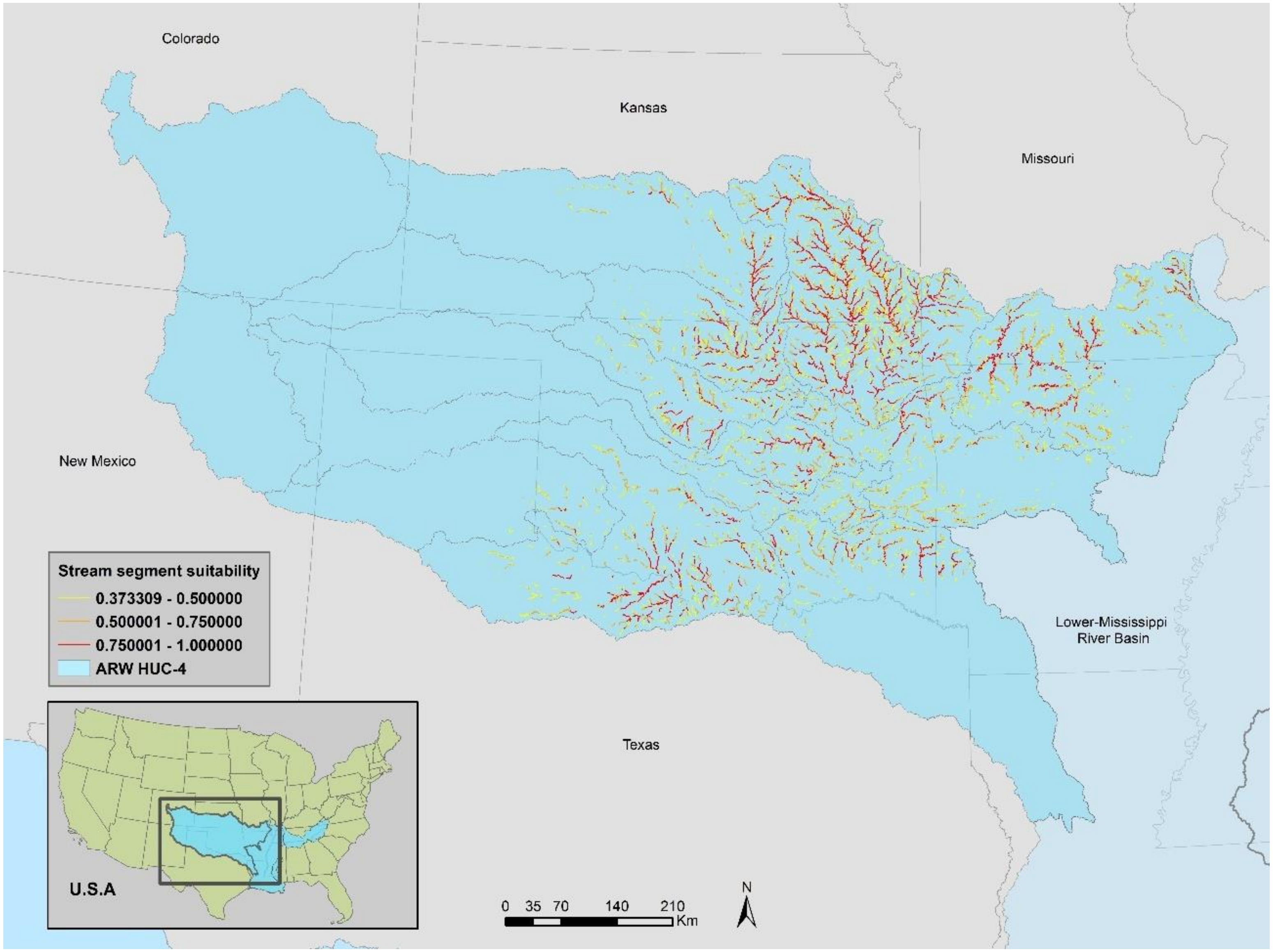

Figure 3. ARW model estimated stream segment suitability with occurrence records of Bluntface Shiner and environmental variables of the Arkansas-Red-White river basin.

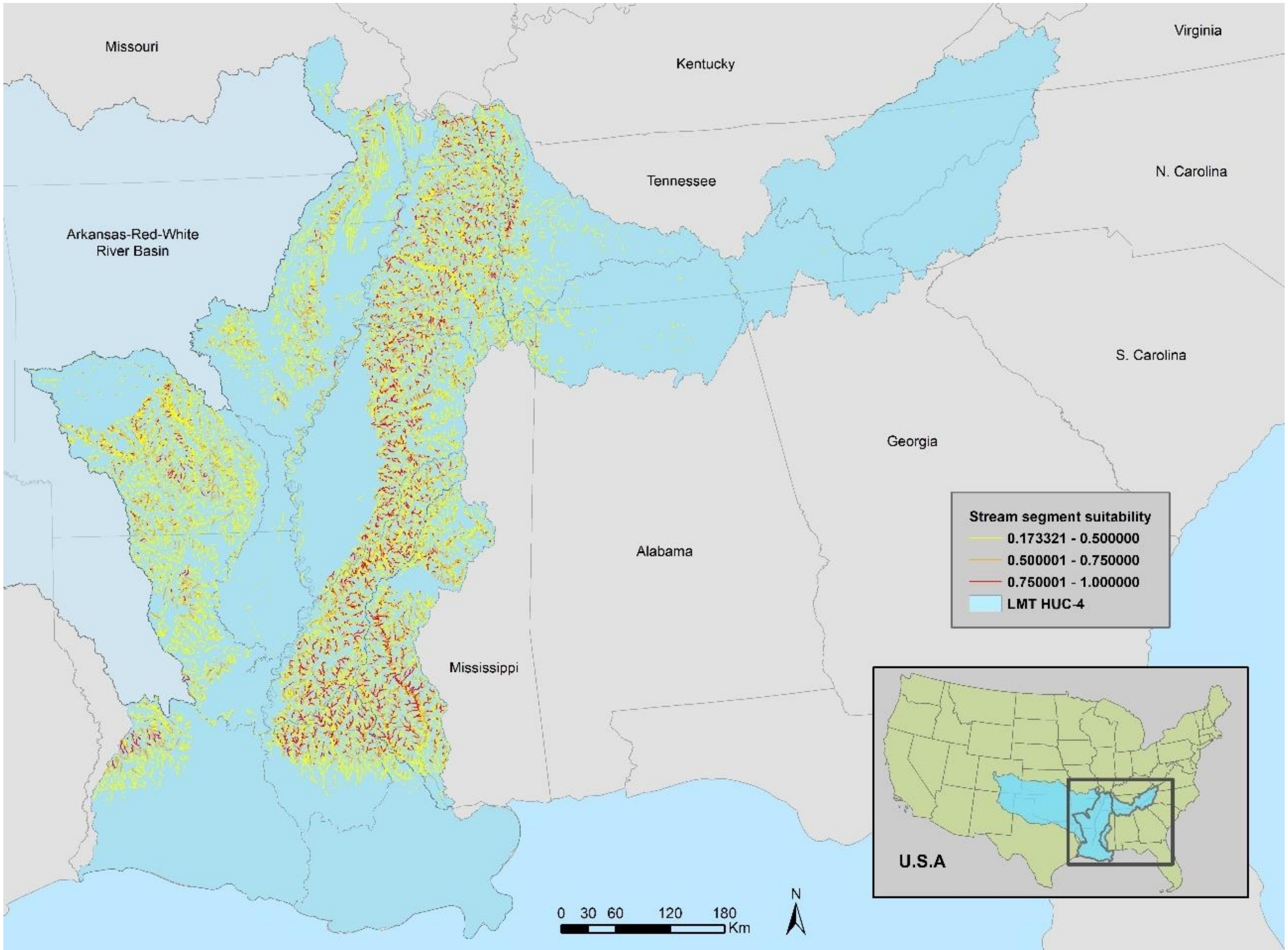

Figure 4. LMT model estimated stream segment suitability with occurrence records of Bluntface Shiner and environmental variables of the Lower-Mississippi and Tennessee river basins.

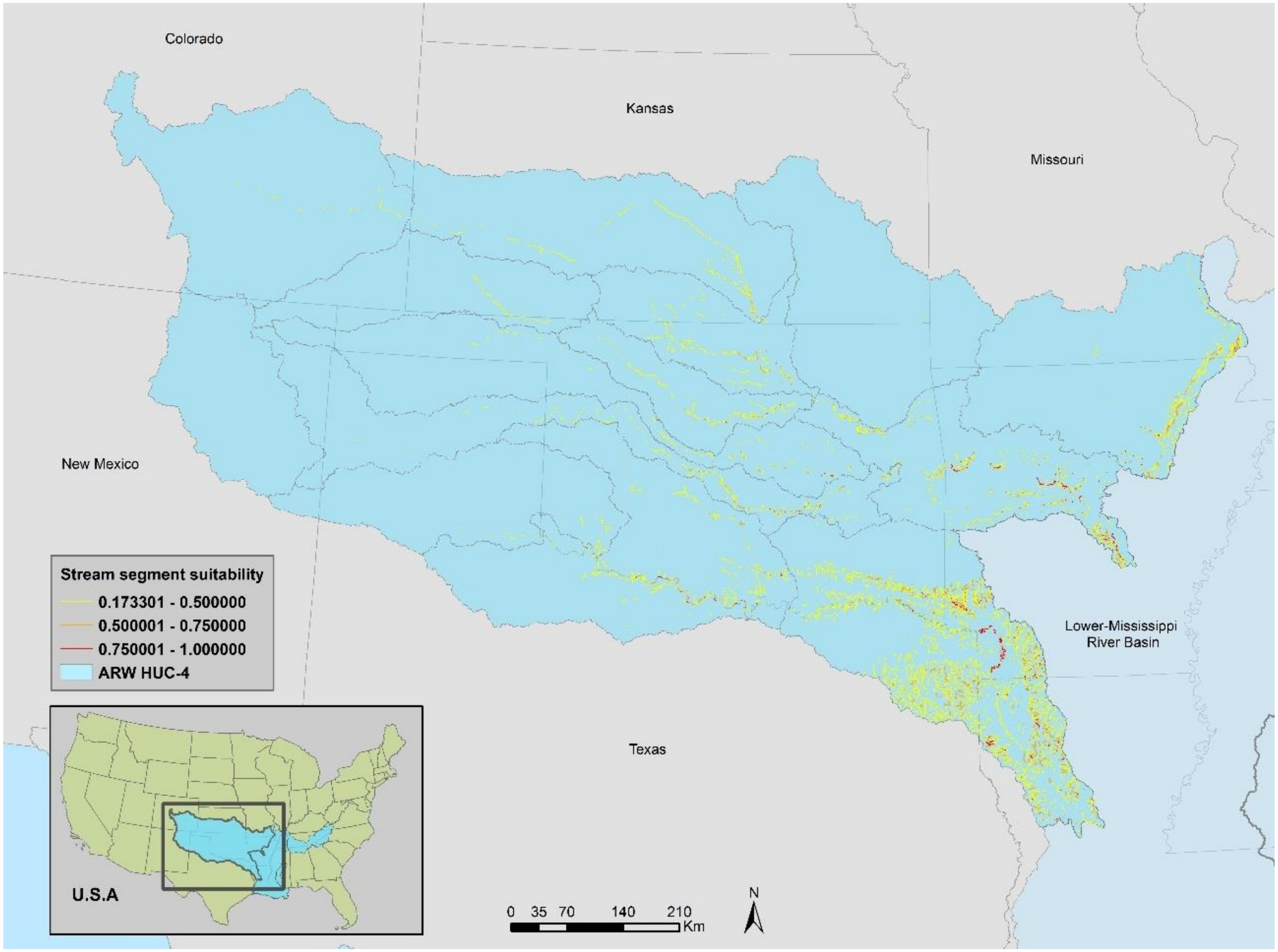

Figure 5. LMT model estimated stream segment suitability projected onto the Arkansas-Red-White basin.

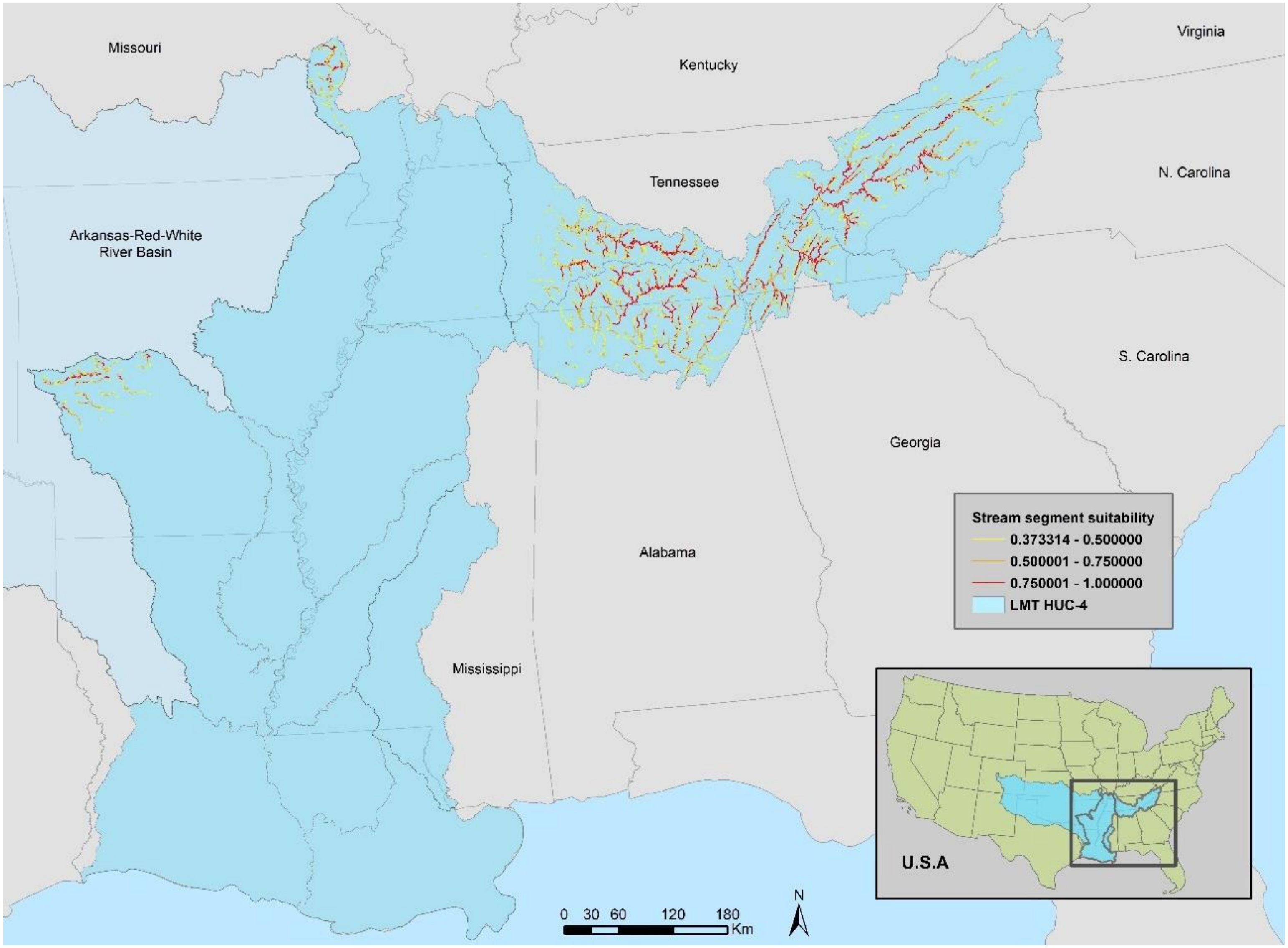

Figure 6. ARW model estimated stream segment suitability projected onto the Lower-Mississippi and Tennessee river basin.

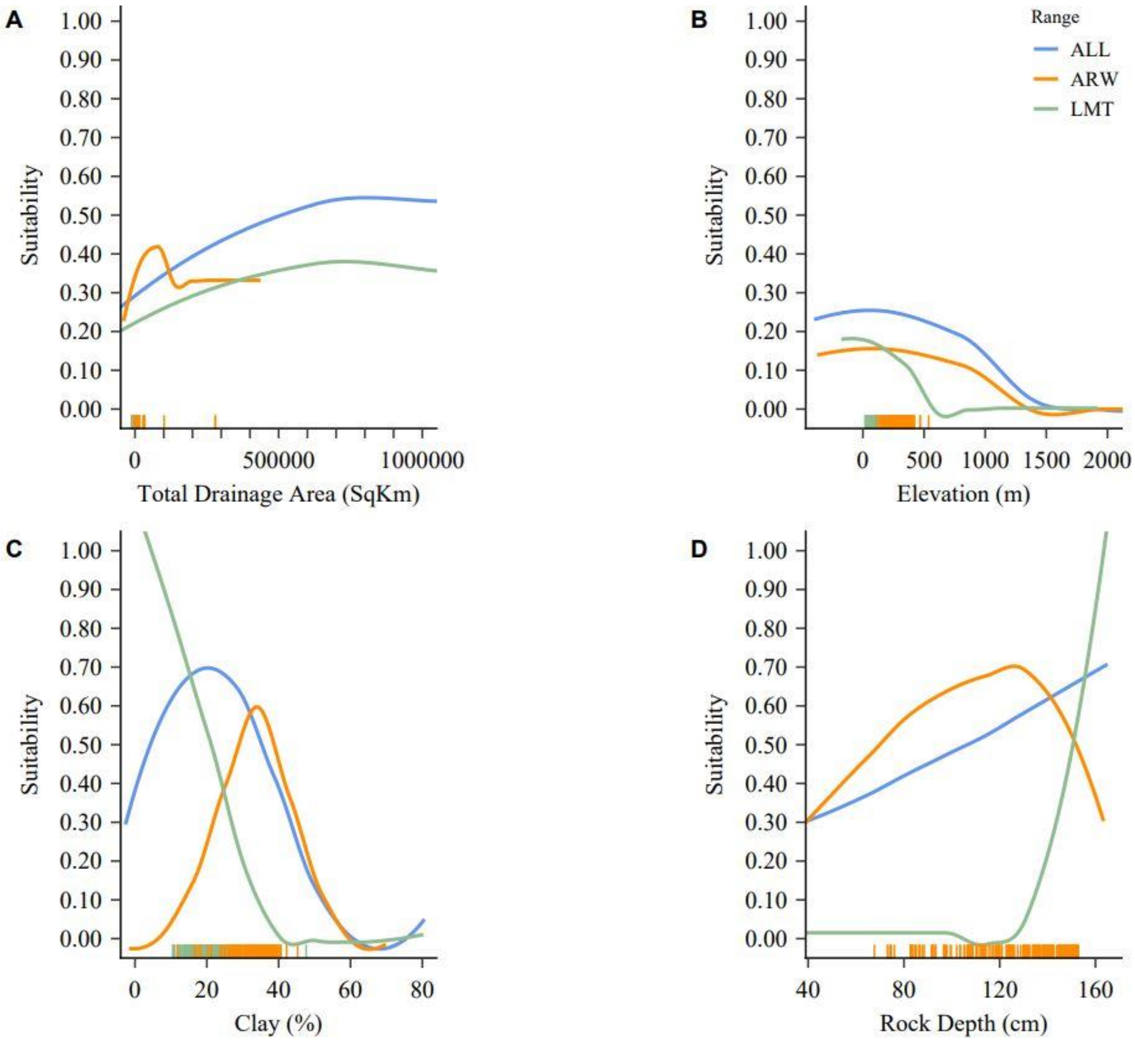

Figure 7. Response curves depicting the highest contributing environmental variables plotted against predicted suitability values of Bluntface Shiner for the ALL, ARW, and LMT ranges. Occurrence records corresponding with the curves are represented along the x-axis (tick marks).

Identifying environmental factors that affect detection and occupancy of Bluntface Shiner (*Cyprinella camura*) in wadeable streams of Oklahoma

Abstract

In Oklahoma, Bluntface Shiner (*Cyprinella camura*; BFS*)* population status is considered "decreasing" or "unknown," and the species is designated as a Tier II species of greatest conservation need. Yet, current information on the distribution and habitat associations of BFS is lacking. To address these knowledge gaps, we used a single season occupancy modeling framework to account for imperfect detection to estimate contemporary detection and occupancy probabilities for BFS. Our aims were to identify the best gear (seine net versus backpack electrofisher) for detecting BFS, the environmental conditions that influence gear effectiveness, and landscape scale factors that influence occupancy of the species in wadable streams within their historic range in Oklahoma. We conducted field surveys during the summers (May – August) of 2021 and 2022. We surveyed 61 sites and captured BFS at a total of 18 sites across the Chikaskia, Caney, Verdigris, Spring, Elk, and Lake O' the Cherokees watersheds for a naïve occupancy rate of 29.5%. Our best detection models revealed gear efficiency was dependent on environmental conditions at the site; however, the seine net was generally the most effective gear at capturing BFS. The best supported occupancy model estimated larger total drainage area size was positively related to occupancy probabilities for BFS. When accounting for imperfect detection, occupancy rate of our sites was 29.5%. Our surveys throughout the historic distribution within Oklahoma suggest BFS are rare and have undergone declines across large portions of their range. Our findings call for focused BFS conservation and management efforts in Oklahoma.

Introduction

Anthropogenic activities pose a major threat to freshwater fish diversity (Jelks et al., 2008; Malmqvist and Rundle, 2002; Dudgeon et al., 2006). Alteration of natural flow regimes can decrease dissolved oxygen levels, increase turbidity and eutrophication, and lead to homogenization of habitat, which may contribute to shift fish assemblages toward tolerant species (Paul and Meyer, 2001; Walsh et al., 2005; Lookingbill et al., 2009; Elmore and Kaushal, 2008; Perkin et al., 2017; Gido et al., 2010; Allan, 2004). Though stream fishes are adapted to natural disturbances (Matthews, 1988; Dodds et al., 2014), anthropogenic disturbances exacerbate the severity of disturbance effects on fish (Perkin et al., 2015). However, for many riverine fishes there is little knowledge about their life history and habitat requirements, which is crucial information for assessing the effects of anthropogenic disturbances on species. Indeed, lack of this information poses a major hinderance to conservation and management of threatened species (Cooke et al., 2012). In Great Plains streams of central North America, many imperiled or threatened Leuciscid species are in need of greater research and monitoring efforts to fill knowledge gaps of their life history, habitat requirements, and distribution (e.g., Worthington et al., 2017). One such species in need of increased study is the Bluntface Shiner (*Cyprinella camura*; BFS).

BFS is native to the Arkansas, Lower-Mississippi, and Tennessee river drainages. BFS occurs within two disjunct areas on either side of the Mississippi River valley. To the west, BFS occur within the Arkansas River basin which includes the Middle-Arkansas, Verdigris, Neosho, and the Lower-Arkansas drainages. To the east, BFS occur in the Lower-Mississippi River basin which includes the Hatchie, Yazoo, Coldwater, and Big Black drainages. Also to the east, BFS exists in some western tributaries within the Tennessee River Basin. On both sides of the Mississippi River valley, BFS is associated with medium-to-large sized streams with clear, flowing waters over riffle and run habitats (Wilkinson and Edds, 2001; Cross, 1954; Etnier and Starnes, 1993; Farr, 1996). In the west, BFS is less abundant in lowland streams that generally have lower average base flow, higher turbidities, and sand, silt, and mud substrates (Cross and Calvin, 1971; Metcalf et al., 2010; Wilkinson and Edds, 2001). Similarly, BFS occurring east of the Mississippi River are mostly associated with upland headwaters of tributaries with moderate- to-swift flow and sand or gravel substrates (Johnston, 1999; Mayden, 1989; Farr, 1996; Ross and Brenneman, 2001). Spawning generally occurs during the spring and summer months from late April to August in the western range (Distler, 2014; Robinson and Buchanan, 2020; Miller and Robinson, 2004), and from March to August in the eastern range (Ross, 2001; Ross and Brenneman, 2001; Etner and Starnes, 1993). As crevice spawners, BFS requires access to rocky substrates for egg deposition and sufficient flow for egg aeration (Mayden 1989; Johnston, 1999). Thus, despite some variation across their range, BFS has specific habitat requirements for survival and reproduction.

Like many riverine fishes, BFS is experiencing apparent declines across its historic range. Currently, the International Union for Conservation of Nature considers this species to be of Least Concern with an unknown population trajectory; however, state agency listings suggest that BFS is experiencing local declines. Within Kansas, Mississippi, and Tennessee, BFS is secure (Mounts, 2019; Ross and Brenneman, 2001; Etner and Starnes, 1993); however, in parts of the Arkansas River basin populations are vulnerable and possibly imperiled in Oklahoma, Missouri, and Arkansas (Oklahoma Department of Wildlife Conservation, 2016; Missouri Department of Conservation, 2021; Robinson and Buchanan, 2020). Likewise, populations in Louisiana are imperiled and at high risk of extirpation (LWF, 2022).

Within the western range of BFS, there is evidence that BFS may have been experiencing declines well before the 2000's. Cross and Braasch (1968) compared fish communities of the upper Neosho River from 1952 to 1967 to characterize their response to increased agricultural activity in the area. At sites where investigators commonly detected the species in years past, BFS were no longer found during the return surveys of 1967 (Cross and Braasch, 1968). In the Arkansas River drainage of Arkansas, no BFS have been documented since the 1960's, suggesting the species has been extirpated (Robinson and Buchanan, 2020). More recently, the Oklahoma Department of Wildlife Conservation (ODWC) Stream Team's community surveys only captured BFS at 12 of 99 sites within the species' historic range (Anthony Rodger, ODWC Biologist, Pers. Comm.).

Unfortunately, there is a general lack of recent, in-depth investigations on BFS habitat needs and distribution to inform conservation measures. BFS is considered sensitive to anthropogenic degradation of water quality and in-stream habitat quality degradation (Jester et. Al., 1992). For example, Cross and Calvin (1971) found that the once abundant BFS had been mostly extirpated after increased cattle ranching activity in the upper Neosho River. They attributed this decline to low oxygen-stress tolerance and optimal habitat being limited during summer months which, in conjunction with increased pollution, prevented BFS from reestablishing after fish-kills. Where BFS cannot recover from disturbance, it is also possible that they are replaced by more tolerant species such as Red Shiner (*Cyprinella lutrensis*), which is tolerant to a range of water conditions including low dissolved oxygen, high turbidities, and thermal shock (Cross and Calvin, 1971; Jester et al., 1992; Matthews and Hill, 1997). This may explain past observations of Red Shiner having higher abundances in stream segments where BFS abundance was low and vice versa (Cross, 1954). However, more work is needed to understand basic habitat associations of BFS and to identify drivers of range loss.

Within this investigation, we conducted field surveys across much of the known historic range of BFS in Oklahoma (see Chapter 1) and used occupancy modeling (MacKenzie et al., 2002) to examine physical habitat factors that affect detection and occupancy of BFS in wadable streams. Occupancy modeling is often used by ichthyologists to identify important habitat factors that drive species occurrence and distribution (e.g., Albanese et al., 2014; Dextrase et al., 2014; Shea et al., 2015; Kuehne and Olden, 2016; Taylor, 2016; Potts et al., 2021) and to assess how effective different sampling gears are for detecting species (e.g., Pregler et al., 2014; Wedderburn, 2018; Reid and Haxton, 2017). In natural settings, a target species may be elusive to capture and difficult to detect despite being present at a site (Bayley and Peterson, 2001; Mackenzie et al., 2002). If ignored, imperfect detection may create misleading inferences regarding species-habitat associations and species distributions (Lahoz-Monfort et al., 2014). The potential for imperfect detection to bias inferences is especially relevant in Great Plains streams, which are characterized by highly variable environmental conditions (Dodds et al., 2014) that can differentially impact the effectiveness of sampling gear (e.g., Schlosser et al., 2012). Therefore, a secondary objective of our study was to examine the effectiveness of seining versus backpack electrofishing for detecting BFS.

Methods

Study Area

The study area spanned across north-central and eastern Oklahoma and included streams from watersheds along the Arkansas River that represent the historic distribution of BFS (Figure 1; also see Chapter 1). The climate of the area is semi-humid with long summers that can reach extreme heat and a rainy season during late spring and early summer. Thus, stream hydrology is highly variable and largely driven by precipitation (United States Army Corps of Engineers, 1987). Typically, the mean maximum streamflow occurs during the spring when rainstorms are common, while the mean minimum stream flow occurs during the summer when temperature and evapotranspiration is highest (Adamski et al., 1995). Especially in the late summer months, many streams within the study area face annual desiccation and become intermittent or completely dry (Matthews, 1988). In the western parts of our study area, streams have relatively low gradients with riffles separated by long stretches of slow-moving pools. Riffles here were often underlaid by gravel, boulder, or bedrock, while pools had substrates of silt, sand, or clay. Eastern portions of the study area transition to upland Ozark Highlands and Boston Mountain streams with higher gradients and well-defined riffles and runs underlaid by gravel, boulder, and bedrock with some sand and clay outcrops.

Field Surveys

We performed targeted field surveys within Oklahoma from late May to early August of 2021 and 2022, coinciding with the BFS spawning season. The first field season focused on watersheds in the north-central part of the state, from the Chikaskia to the Middle Verdigris watershed. The second field season searched the northeast corner moving southward from the Spring to the Dirty-Greenleaf watershed. To select survey sites, we performed an aerial-visual survey using Google Earth Pro to identify up to 40 potential sites per field season that appeared to have safe access to the stream channel and were ostensibly wadable (<1.5 meters deep). Sites were selected at exact and proximal locations in relation to existing BFS occurrence records, as well as exploratory sites within streams where the minnow had not been previously recorded (Fig.1, 2). As such, our field sites were focused in areas we considered to have a higher likelihood of being occupied by BFS compared to a randomized selection of sites across the landscape. We aimed for four visits to each site for replication. During a revisit to the site, we surveyed the same reach. With each visit, we alternated between the backpack electrofisher and seine net to compare effectiveness in capturing BFS.

Backpack electrofishing was conducted with an ETS ABP-4-MR and two dipnets, generally following standard sampling methods (Rabeni et al. 2009). We aimed for an average output of 2.5 amps for each channel unit and standardized duty cycles between 10-15% and rates between 45-60 Hz. Voltage, duty cycle, and rate were adjusted to elicit desirable level of electrotaxis. In high conductivity (>700 µS/cm), we adjusted settings outside of these parameters with a focus on lower voltage but increased current (Meyer et al., 2021). Initial shock settings were tested outside the study reach before sampling began. Seining used a 5 x 1.2 m seine with 3/16" mesh. For either gear, we performed the necessary effort to cover the entire channel.

Upon the first visit to a site, we recorded location descriptions and GPS coordinates and established a sampling reach consisting of available channel unit types (i.e., riffle, run, and pool). Each channel unit within the reach was treated as independent for fish and habitat data collection. Sampling began at the most downstream channel unit and progressed in the upstream direction, and we recorded sampling effort for each channel unit. Fishes that were collected within a unit were kept in containers until sampling of the entire channel unit was completed. After sampling the channel unit, all captured fish were identified, enumerated, and released back into the midpoint of the channel unit. All captured BFS were given a small fin-clip to allow for identification of recaptures during revisits.

Water quality measurements of pH, conductivity, salinity, and turbidity were recorded at the midpoint of the reach prior to sampling. We also visually assessed aquatic vegetation (percentage of unit area covered), large woody debris (count within the unit), and substrate embeddedness was scaled between 1 (least embedded) to 5 (most embedded) for each channel unit. Along the reach, the length of the channel unit was recorded, and we placed three transects at equal distances within each channel unit perpendicular to stream flow. We recorded wetted channel width at each transect, and at a minimum of five evenly spaced points along the transects (exceptionally large transects called for more points), we measured the substrate type (similar to Bain et al., 1992), water depth, and water velocity. Additionally, we measured water temperature and dissolved oxygen at the mid-point of the middle transect at each unit. For additional environmental variables that may explain occupancy across the landscape, we used a spatial join in ArcMap to link our sample site locations to stream segment data in NHDplusV2 (McKay et al., 2012) and StreamCat (Hill et al., 2016).

Occupancy and Detection Modeling

Detection and occupancy models were constructed with R coding language using the package 'unmarked' (Fiske and Chandler, 2011; R Core Team, 2020) following a single-season framework that accounts for imperfect detection (i.e., detection rates < 1; MacKenzie et al., 2002; 2006). Models were created based on *a priori* hypotheses of environmental factors that influence the detectability and occurrence of BFS (Table 1; Table 3). Pearson's correlation tests were run to remove variables with correlation coefficients of $r > 0.7$ (Dormann et al. 2013). All continuous variables were standardized with a natural long transformation followed by calculation of a z-score, whereas percentage variables were arcsine square root transformed.

Selection of the best approximating model was based on Akaike Information Criterion (AIC) and Akaike weights (W_t). Models were ranked using AIC values (Burnham and Anderson, 2001). The model with the lowest AIC value was considered the best model, but we also considered models with delta AIC (∆AIC) scores less than 2.00 to have support. Detection probabilities were estimated with logistic regression (MacKenzie et al., 2006). We used the logit link function to back transform estimates into interpretable results. We first considered a candidate set of detection models using survey-level detection covariates with occupancy held constant ($\Psi(.)\rho(Cov)$). The detection model with the most support was chosen for use in testing hypothesized relationships among site covariates and occupancy . In this group of models, occupancy was allowed to vary as a function of site-level covariates while detection was modeled as a constant ($\psi(Cov)p(.)$ or as a function of a covariate $\psi(Cov)p(Cov)$ We assessed model fit on the global model (all covariates) with Pearson's chi-squared statistic and a measure of overdispersion ($\hat{c}$) estimated with parametric bootstrapping (n = 10,000), wherein $\hat{c} > 1.2$ would be indicative of lack of fit (MacKenzie and Bailey, 2004). Occupancy probability estimates were calculated using empirical Bayes methods (Fiske and Chandler 2011; Fiske and Chandler 2015).

Results

Field surveys

The study area spanned a wide area within northeastern Oklahoma, and sample sites were diverse in physical and chemical conditions (Table 3). The goal of 40 sites per summer with four replicates each was not achieved due to high stream flows early in the field season, coupled with a few sites going dry later in the field season. A total of 61 sites were surveyed with 191 total visits. We performed 104 surveys at 31 sites in 2022 and 87 surveys at 30 sites in 2022. Of the 61 sites, 10 were exact locations of a historic BFS record, 33 were proximal to a historic record, and 18 were exploratory locations (Fig. 2). During the first field season, we captured BFS at 14 of the 31 sites (45%) while in the second field season, we captured BFS at 4 of the 30 sites (13%). Therefore, without accounting for imperfect detection, we had a naïve occupancy estimate of 29.5% (Figure 3). Of the sites where BFS were captured, 2 were exact historical locations, 13 were proximal to a historic record, and 1 was an exploratory location.

Detection Models

The candidate model set of 19 models resulted in six well supported models (ΔAIC ≤ 2.00; Table 4). The highest ranked detection model contained turbidity with a gear interaction (Turb*Gear; $W_t = 0.20$). Other models with support included a gear interaction, including: wetted channel width with a gear interaction (WCW*Gear; $W_t = 0.16$); an additive relation between conductivity and turbidity with a gear interaction (Cond + Turb*Gear; $W_t = 0.15$); depth with a gear interaction (Dep*Gear; $W_t = 0.09$); velocity with a gear interaction (Vel*Gear; $W_t = 0.07$); and conductivity with a gear interaction (Cond*Gear; $W_t = 0.07$).

Further analysis parsed detection probability estimates between sampling gears. The models revealed differences in detection probability of BFS with backpack electrofishing (BPEF) and seine (SE) as a function of the mean of the survey-level covariates measured at the sites (Table 2). Model ρ(Turb*Gear) estimated a detection probability of 61% with SE and 42% with BPEF. There was an estimated a positive relationship between turbidity and SE wherein detection probability with SE increased as turbidity increased, and a negative relationship between turbidity and BPEF wherein detection probability with BPEF decreased as turbidity increased (Fig. 6A). Model ρ(Cond + Turb*Gear) estimated a detection probability of 71% with SE and 55% with BPEF. This model demonstrated the same trend, though overall detection probabilities were higher for both gears than when estimated with turbidity alone (Fig. 6C). Estimated detection probabilities for ρ(Cond*Gear) were similar to the aforementioned models; however, detection probability remained constant at 72% with SE and 56% with BPEF (Fig. 6F). Model ρ(Vel*Gear) estimated detection probabilities of 83% with SE and 56% with BPEF. In contrast to the other models with gradual trends, detection probability under this model sharply decreased for both gears as water velocity increased (Fig. 6E). The remaining models estimated opposing trends between gears with overlap. Model ρ(WCW*Gear) estimated a detection probability of 25% with SE and 43% with BPEF. Detection probability with SE increased as WCW increased, while detection probability with BPEF decreased as WCW increased (Fig. 6B). Lastly, model ρ(Dep*Gear) estimated detection probabilities of 59% with SE and 52% with BPEF. Detection probability with BPEF decreased as depth increased, while detection probability with SE increased as depth increased (Fig. 6D).

Occupancy Models

The Turb*Gear detection model was the most supported model; therefore, we adopted it for the combined model set and occupancy models. The goodness of fit test showed the global model fit the data ($\hat{c}$ = 0.92). Only one of the eight occupancy models was well supported (ΔAIC $\leq$ 2.00; Table 4). The highest ranked model included total drainage area as the occupancy covariate ((ψ(zmeanTotDA)ρ(Turb*Gear); W_t = 0.88). The estimated occupancy probability based on the mean total drainage area of our sites was 32.6%. Plotting occupancy probability against total drainage area revealed a positive relationship in which occupancy probability increased as total drainage area size increased (Fig. 7). Empirical bayes estimated the proportion of area occupied to be 29.5% (95% CI = 29.5, 68.9).

Discussion

Under a single season framework for detection and occupancy modeling, we identified environmental factors that influenced the detection of Bluntface Shiner (BFS) and their occurrence in wadable streams of Oklahoma. Our investigation contributes to the growing body of literature that demonstrates the utility of detection and occupancy models for elucidating habitat associations of fishes (e.g., Anderson et al., 2012; Dextrase et al., 2014; Potoka et al., 2016; Jensen and Vokoun, 2013; Shea et al., 2014) and the best gears for capturing them (e.g., Reid and Haxon, 2017; Pregler et al., 2015; Smith et al., 2105; Gibson-Reinemer et al., 2016; Schlosser et al., 2012; Price and Peterson, 2010). We found that BFS can be elusive in scenarios where certain abiotic factors interact with gear type. We also estimated that BFS currently occupy less than a third of our study sites despite focusing our sampling efforts within the historic range of BFS. Our results provide evidence for a restricted contemporary distribution of BFS within Oklahoma compared to their estimated historic distribution (see Chapter 1).

Detection models

Our models identified several habitat covariates that influenced detection probability of our gears. In general, SE was estimated to have higher average detection probabilities than BPEF under a broad range – but not all – of the environmental conditions encountered in our study. Our results contrast with other investigators who found that electrofishing was overall superior to SE for sampling stream-dwelling fish in their respective streams (Mercado-Silva and Escandon-Sandoval, 2008; Poos et al. 2007). However, no single sampling gear may produce perfect detection rates in all conditions (Price and Peterson, 2010; Pregler et al., 2015; Schlosser et al., 2012; Gibson-Reinemer et al., 2016), and our findings reiterate that sampling gear effectiveness is largely influenced by in situ habitat conditions (Rabeni et al., 2009; Temple and Pearsons, 2007; Bayley and Herendeen, 2000). To illustrate this idea, consider the most well supported detection model involving turbidity that estimated an increase in detection probability with SE as turbidity increased while negatively effecting BPEF as turbidity increased. A general rule for electroshocking is that the anode should be visible for dip-netting fish affected by the electric shock-field (Beaumont, 2016; Barbour et al., 1999). In circumstances with low water clarity, line of sight is obscured, and accurate dip-netting becomes increasingly difficult with higher turbidities. As such, detection probability with electrofishing decreases with increased turbidity (Lyon et al., 2013) and net-gears are the preferred method for sampling in turbid conditions (Utrup and Fisher, 2006). In contrast, sampling with SE involves indiscriminate area-coverage, regardless of visibility. Further, we observed BFS in clear waters (identified by a solid white band at the base of the tail and confirmed by captures) to be extremely wary of our presence. In this regard, turbidity may have been advantageous in limiting BFS avoidance response to incoming SE hauls (Poos et al., 2007).

We attributed SE generally outperforming BPEF in detecting BFS to the versatility of SE for surveying streams across a large, geographically diverse area wherein environmental conditions can vary drastically from stream to stream. For this reason, if only one gear could be adopted due to time or logistical constraints, we recommend using SE for the best chances of detecting BFS in future surveys conducted in a similar manner (i.e., wadeable streams during summer months). Beyond this, it would be worthwhile to incorporate detection probabilities in future BFS monitoring programs to determine threshold conditions for effective gear use, or to inform gear efficacy prior to implementation (Schlosser et al., 2012). We acknowledge a limitation of our sampling design in that SE and BPEF were alternated between visits to a site regardless of in-stream conditions. Other investigations may employ a more informed sampling approach by selecting the appropriate gear based on conditions present at the site (e.g., turbidity, conductivity, depth, etc.) which could yield different detection results. Additionally, historic records and our occupancy modeling results suggests that BFS occupy larger streams, which can be difficult to sample with SE and BPEF. Future investigations could employ a similar detection modeling framework to explore detection of BFS with gears such as tow-barge electrofishing in larger, wadable streams or boat-trawl for deep mainstreams of large rivers to further refine sampling protocol for BFS across habitat types.

Occupancy models

Our occupancy model informed by detection estimated 29.5% of our sites were occupied by BFS, which was not different than our naïve occupancy of 29.5%. This occupancy estimate covers less than a third of our study sites and was quite low considering that our survey area was entirely within the historic distribution of BFS, and we biased our site locations based on proximity to historic records. However, we know that true occupancy was higher because single individuals were captured by the Oklahoma Department of Wildlife Conservation (ODWC) at or nearby sites where we did not capture BFS during the same summer (Anthony Rodger, Pers. Comm.). Had we captured BFS at these sites, our occupancy results would have likely differed.

Total drainage area size (TotDA) was the most important occupancy variable, and there was a positive relationship between TotDA size and BFS occupancy probability wherein larger TotDA sizes had higher occupancy estimates. This result matched a separate investigation that identified TotDA as the highest contributing variable that explained the historic distribution of BFS (see Chapter 1). Similarly, literature and historic records suggest that BFS are most associated with relatively larger streams. Drainage area is correlated with many factors including increasing water flow, depth, stream size and length, and various physicochemical properties that change with increasing size and are closely linked to fish assemblage structure and richness (Allen et al., 2021; Mattews, 2012). It would be difficult to make any direct inferences from TotDA, but if we consider the harsh and dynamic conditions that characterize prairie streams in the historic range of BFS, there is a case for larger drainage area sizes being important in metapopulation dynamics involving source and sink populations of BFS. Source habitats would include areas suitable for spawning, rearing, and refugia, while sink habitats are those with unfavorable conditions where local extinction would occur without rescue from populations elsewhere (Falke and Fausch, 2010). Across a watershed, the availability of these habitats varies unpredictably across space and time (Falke et al., 2012), and access to and connectivity between them are critical for population persistence, reproduction and dispersal (Dodds et al., 2004; Sedell et al., 1990; Labbe and Fausch, 2000; Falke et al., 2010). Based on these ideas, we hypothesize that larger TotDA sizes may be important for BFS populations by providing environmental stability, greater access to important microhabitats, and increased chances of successful colonization (Hoagstrom and Berry, 2006).

We found no additional site-level covariates to influence the occupancy of BFS, though literature suggests that several other environmental covariates may affect the occurrence of BFS. For example, substrate composition and dissolved oxygen are cited as habitat requirements for BFS; however, their effects on BFS occupancy may be obscured in our models by the highly variable environments we surveyed. For landscape variables such as agriculture, occupancy effects may have been concealed by the relatively narrow size of our study area and lack of reference conditions. Had we surveyed sites outside of the historic range, we may have encountered a wider range of covariate values which would better refine the importance of these variables. Despite our results, variables such as these are worth considering in investigations of BFS because this species is sensitive to poor water quality and habitat quality (Jester et al., 1992; Cross and Calvin, 1997). Alternatively, the wide variety of habitat conditions at sites where we detected BFS could suggest that the disjunct population in the Arkansas River Basin are habitat generalists.

Sites with BFS detections were clustered within the Chikaskia, Caney, Verdigris, Spring, Lake O' the Cherokees, and Elk watersheds which coincided with the more recent historic records of the minnow. The Caney and Verdigris watersheds may represent strongholds for BFS because we captured approximately 200 individuals from each watershed during our surveys. In contrast, our surveys yielded no detections in the Dirty – Greenleaf, Illinois, and Lower Neosho watersheds, which held a high number of historic records and our species distribution models estimated highly suitable conditions for BFS (see Chapter 1). We suspected fair chances of detecting BFS in the latter watersheds, though historic records showed that the BFS were last documented in the area during the

1990s. Populations still persist to some capacity in parts of these watershed as evident by captures of single individuals from Dirty Creek (no historic records) and Bayou Maynard (contained several historic records) during surveys by the ODWC Stream Team in the summer of 2022. These relatively large tributaries were un-impounded with unrestricted flow into the Lower Arkansas, which differentiated them from other tributaries in the area (Greenleaf Creek).

The distribution of BFS appears to be naturally patchy across the landscape (see Chapter 1), and our occupancy results coupled with the lack of recent records indicates that dams and large impoundments increasingly disconnect BFS populations. These physical barriers raise concerns about the true, contemporary range of the BFS. Take the Illinois River Basin as example. Historic records in the area mostly predated the completion of Lake Tenkiller, and one may suspect a chance of capturing BFS in the watershed given the number of historic records in the area. Yet, our sites, nor those of the ODWC Stream Team who also surveyed the watershed during this time, yielded detections of BFS (Anthony Rodger, Pers. Comm). This was alarming because even in the farther upstream reaches of the Illinois River in the state of Arkansas, no BFS have been documented since the 1960s. Thus, it is presumed to be extirpated from the watershed (Robinson and Buchanan Robison, 2020). The creation of Lake Tenkiller is likely not the single cause of the suspected extirpation (given that other extant populations exist above impoundments), but the existence of historic records submerged within the reservoir indicates that a large area of suitable BFS habitat was lost, and the dam precludes natural recolonization from populations in adjacent watersheds. Other studies documented declines in fluvial fish species associated with the creation of impoundments, which the authors attributed to changes in hydrology, invasive species, and fragmentation of home range that disrupts metapopulation dynamics (Luttrell et al., 1999; Schrank et al., 2001; Winston et al., 1991; Wilde and Ostrand, 1999; Hubbs and Pigg, 1976), but we cannot make definitive assertions on the effects of impoundments on BFS beyond the relation between these structures and the historic records.

Management and Conservation Implications

The BFS is a species of greatest conservation need in Oklahoma, but information on life history and habitat requirements is sparse, and BFS distribution and population status within the state is considered declining or unknown. Basic natural history information is fundamental to conservation efforts and development of effective management strategies for at-risk species (Matthews, 2015; Cooke et al., 2012; Mace and Kunin, 1994). Consequently, we expanded the understanding of this species by using occupancy models to inform sampling protocols for detecting BFS and elucidating environmental variables that influence their detection and occupancy in wadable streams of Oklahoma.

Our data indicate that BFS have declined from its historic range and exist as disjunct populations across a limited number of watersheds within Oklahoma. Even within its historic range, this species is rare in the sense that populations are “clumped” within small areas across their distribution (McDonald and Thompson, 2004). Habitat protection and monitoring of extant populations, as well as identifying factors driving their decline, is warranted given their limited range and demographic isolation, but there still exists knowledge gaps that would impede such efforts. Mainly, we still do not have a complete picture of what functional or natural habitats are required for BFS to complete their life cycle; such information is fundamental to effective management and conservation planning (Roni et al., 2008). The Caney and Verdigris are model watersheds in these respects given the high abundance of BFS. These populations provide opportunities for investigating reproduction, growth, macro- and microhabitat requirements, and movement patterns of healthy populations, which can be leveraged in investigations of BFS populations elsewhere. However, the highly heterogenous geography and climate across the historic distribution may mean that suitable conditions in one watershed will not necessarily apply to a separate watershed (e.g., Ozarks vs. lowland prairie watersheds). These issues can be addressed by continuing to quantify and refine the known distribution of BFS by searching other habitats (i.e., large mainstream rivers) or watersheds where records are sparse or nonexistent (e.g., Salt Fork, Black Bear – Red Rock, Polecat – Snake, Lower Verdigris). For example, there is substantial evidence of BFS inhabiting large mainstreams of rivers, but biological reasons are never provided. Hill et al., (1981) captured BFS exclusively from their sites within the mainstem of the Grand River and not in tributary creeks nearby, whereas in the Chikaskia, we found juveniles in the mainstream river, but adults in tributaries. Future surveys should consider mainstream river habitats as being of equal importance as smaller wadeable streams in surveys for BFS to capture drastically different abiotic conditions that are potentially suitable for BFS. Likewise, it would also be worthwhile to expand searches beyond the watersheds we investigated into areas estimated to have highly suitable habitats (see chapter 1). Detections of BFS in these watersheds would further refine our understanding of where BFS persist and habitat conditions that are most suitable. Taken together, these measures would lend additional clues toward the drivers of BFS range loss in Oklahoma.

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Table 1. Survey- and site-level covariates used for detection and occupancy modeling

Environmental covariates	Abbreviation	Scale	Description
Sampling gear (BPEF or Seine)	Gear	Survey-level	Gear type used during the survey
Conductivity (μS/cm)	Cond	Survey-level	Ease at which electricity passes through water
Average depth of the reach (m)	Dep	Survey-level	Average water depth at transects
Maximum depth (m) of the reach	MaxD	Survey-level	Deepest water depth within the reach (maximum capped at 1.5 m)
Proportion of coarse substrates	PropCoarse	Survey-level	Proportion of boulder and bedrock substrates in transects
Turbidity (NTU)	Turb	Survey-level	Point estimate of water clarity
Average surface velocity of the reach (m/s)	Vel	Survey-level	Average surface velocity at transects
Average wetted channel width (m)	WCW	Survey-level	Average width of the wetted channel at transects
Average substate type	AvgSub	Site-level	Average score (from 1 [most fine] to 5 [most coarse]) for substrate type at transects
Standard deviation of substrate type	SubSD	Site-level	Describes which site had the most variable substrate composition
Base flow index	BFI	Site-level	Base flow index of the catchment
Dissolved Oxygen (ppt)	DO	Site-level	Point estimate of dissolved oxygen
Elevation (m)	Elev	Site-level	Elevation at the site
Percent hay or crop land use (%)	AgLand	Site-level	Percentage of hay or crop land use in the catchment
Total drainage area (km^2)	TotDA	Site-level	Total drainage area upstream of the reach

Table 2. Descriptions and hypotheses of Bluntface Shiner detection and occupancy models for wadable streams in Oklahoma during the summer months of 2021-2022.

Candidate model set		
Hypotheses (Detection)	Model	Model structure
There is no covariate effect on detection probability	ρ(.)	β_0
Gear type affects detection	ρ(Gear)	$\beta_0 + \beta_1$(Gear)
Depth affects detection	ρ(Dep)	$\beta_0 + \beta_1$(Dep)
Max depth affects detection	ρ(MaxD)	$\beta_0 + \beta_1$(MaxD)
Proportion of coarse substrate affects detection	ρ(PropCoarse)	$\beta_0 + \beta_1$(PropCoarse)
Turbidity affects detection	ρ(Turb)	$\beta_0 + \beta_1$(Turb)
Water velocity affects detection	ρ(Vel)	$\beta_0 + \beta_1$(Vel)
Wetted channel width affects detection	ρ(WCW)	$\beta_0 + \beta_1$(WCW)
Depth with a gear interaction affects detection	ρ(Dep*Gear)	$\beta_0 + \beta_1$(Dep) x β_2(Gear)
Max depth with a gear interaction affects detection	ρ(MaxD*Gear)	$\beta_0 + \beta_1$(MaxD) x β_2(Gear)
Turbidity with a gear interaction affects detection	ρ(Turb*Gear)	$\beta_0 + \beta_1$(Turb) x β_2(Gear)
Proportion of coarse substrate with a gear interaction affects detection	ρ(PropCoarse*Gear)	$\beta_0 + \beta_1$(PropCoarse) x β_2(Gear)
Water velocity with a gear interaction affects detection	ρ(Vel*Gear)	$\beta_0 + \beta_1$(Vel) x β_2(Gear)
Wetted channel width with a gear interaction affects detection	ρ(WCW*Gear)	$\beta_0 + \beta_1$(WCW) x β_2(Gear)
Conductivity and turbidity with a gear interaction affects detection	ρ(Cond + Turb*Gear)	$\beta_0 + \beta_1$(Cond) + β_2(Turb) x β_3(Gear)
Depth and velocity with a gear interaction affects detection	ρ(Dep + Vel*Gear)	$\beta_0 + \beta_1$(Dep) + β_2(Vel) x β_3(Gear)
Depth, velocity, and wetted channel width with a gear interaction affects detection	ρ(Dep + Vel + WCW*Gear)	$\beta_0 + \beta_1$(Dep) + β_2(Vel) + β_3 (WCW) x β_4(Gear)
Depth, velocity, and proportion of coarse substrate with a gear interaction affects detection	ρ(Dep + Vel + PropCoarse*Gear)	$\beta_0 + \beta_1$(Dep) + β_2(Vel) + β_3(PropCoarse) x β_4(Gear)
Depth, velocity, wetted channel width, and proportion of coarse substrate with a gear interaction affects detection	ρ(Dep + Vel + WCW + PropCoarse*Gear)	$\beta_0 + \beta_1$(Dep) + β_2(Vel) + β_3(WCW) + β_4(PropCoarse) x β_5(Gear)
Combined model set		
Hypothesis (Occupancy)	Model	Model structure
No covariate affects occurrence	Ψ(.)	β_0
Average substrate type affects occupancy	Ψ(zmeanAvgSub)	$\beta_0 + \beta_1$(zmeanAvgSub)
Substrate variability affects occupancy	Ψ(zmeanSubSD)	$\beta_0 + \beta_1$(zmeanSubSD)
percent hay or crop land use affects occupancy	Ψ(arcsqAgLand)	$\beta_0 + \beta_1$(arcsqAgLand)
Base flow index affects occupancy	Ψ(zmeanBFI)	$\beta_0 + \beta_1$(zmeanBFI)
Dissolved oxygen affects occupancy	Ψ(zmeanDO)	$\beta_0 + \beta_1$(zmeanDO)
Elevation affects occupancy	Ψ(zmeanElev)	$\beta_0 + \beta_1$(zmeanElev)
Total drainage area affects occupancy	Ψ(zmeanTotDA)	$\beta_0 + \beta_1$(zmeanTotDA)

Table 3. Survey-level and site-level covariates used in detection and occupancy models. Survey-level covariates measured in the field were water depth (m; Dep); water velocity (m/s; Vel); wetted channel width (m, WCW); proportion of coarse substrate (%; PropCoarse); maximum depth (m; MaxD); conductivity (µS/cm; Cond); and turbidity (NTU; Turb). Site-level covariates collected in the field or using GIS were average substrate type (from 1 [most fine] to 5 [most coarse]; AvgSub); standard deviation of the substrate type (representative of substrate variability; SubSD); and dissolved oxygen (ppt; DO), total drainage area (km^2; TotDA); base flow index (BFI); elevation (m; Elev); and percentage hay or crop land use (%; AgLand).

Survey-level covariates							
	Dep	Vel	WCW	PropCoarse	MaxD	Cond	Turb
Min.	0.11	0	4.15	0	0.28	126.1	0.23
Median	0.28	0.23	9.41	36.67	0.86	466	6.52
Mean	0.3112	0.3081	10.63	40.08	1.227	641.6	12.223
Max.	0.83	1.98	72.6	293.75	64	3910	150
Site-level covariates							
	AvgSub	SubSD	DO	TotDA	BFI	Elev	AgLand
Min.	0.0531	0.0302	0.045	3.97	10	158.6	0
Median	0.95	1.22	1.11	86.32	26.2	260.5	21.98
Mean	1.57	1.49	1.55	231.2	25.67	261.7	24.22
Max.	6.31	5.58	9.97	4417	47.96	348.9	71.83

Table 4. Model ranking based on AIC scores.

Model	K	AIC	ΔAIC	AICWt	Cum.Wt	LL
Candidate model set						
Model	K	AIC	ΔAIC	AICWt	Cum.Wt	LL
Ψ(.)ρ(Turb*Gear)	7	149.9	0	0.2	0.2	-67.95
Ψ(.)ρ(WCW)	7	150.35	0.45	0.16	0.35	-68.17
Ψ(.)ρ(Cond + Turb*Gear)	8	151.42	0.52	0.15	0.5	-67.21
Ψ(.)ρ(Dep*Gear)	7	151.4	1.5	0.09	0.59	-68.7
Ψ(.)ρ(Vel*Gear)	7	151.64	1.74	0.08	0.68	-69.82
Ψ(.)ρ(Cond*Gear)	7	151.85	1.95	0.07	0.75	-68.92
Ψ(.)ρ(PropCoarse*Gear)	7	152.1	2.19	0.07	0.82	-69.05
Ψ(.)ρ(Dep + Vel*Gear)	8	153.63	3.73	0.03	0.85	-68.82
Ψ(.)ρ(Dep + Vel + WCW*Gear)	9	153.91	4.01	0.03	0.87	-67.96
Ψ(.)ρ(.)	2	153.99	4.09	0.03	0.9	-75
Combined model set						
Model	K	AIC	ΔAIC	AICWt	Cum.Wt	LL
(ψ(zmeanTotDA)ρ(Turb*Gear))	8	141.97	0	0.88	0.88	-62.99
(ψ(zmeanDO + zmeanAvgSub + zmeanSubSD + zmeanTotDA + zmeanBFI + zmeanElev + arcsqAgLand)ρ(Turb*Gear)	14	148.31	6.33	0.04	0.92	-60.15
(ψ(zmeanBFI)ρ(Turb*Gear))	8	149.56	7.59	0.02	0.94	-66.78
(ψ(.)ρ(.))	8	149.73	7.76	0.02	0.96	-66.87

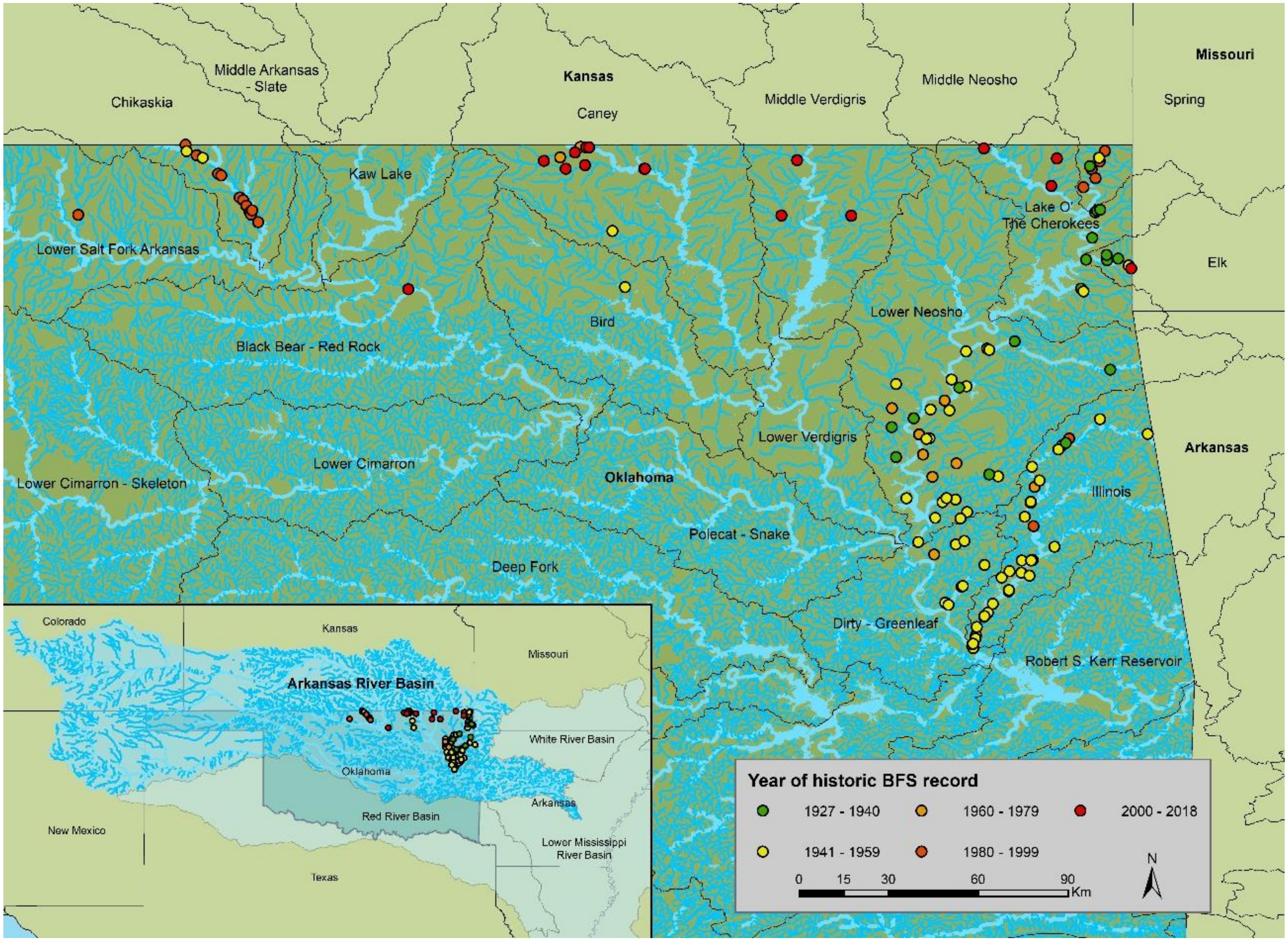

Figure 1. Map of historic Bluntface Shiner occurrence records within Oklahoma colored by year of collection.

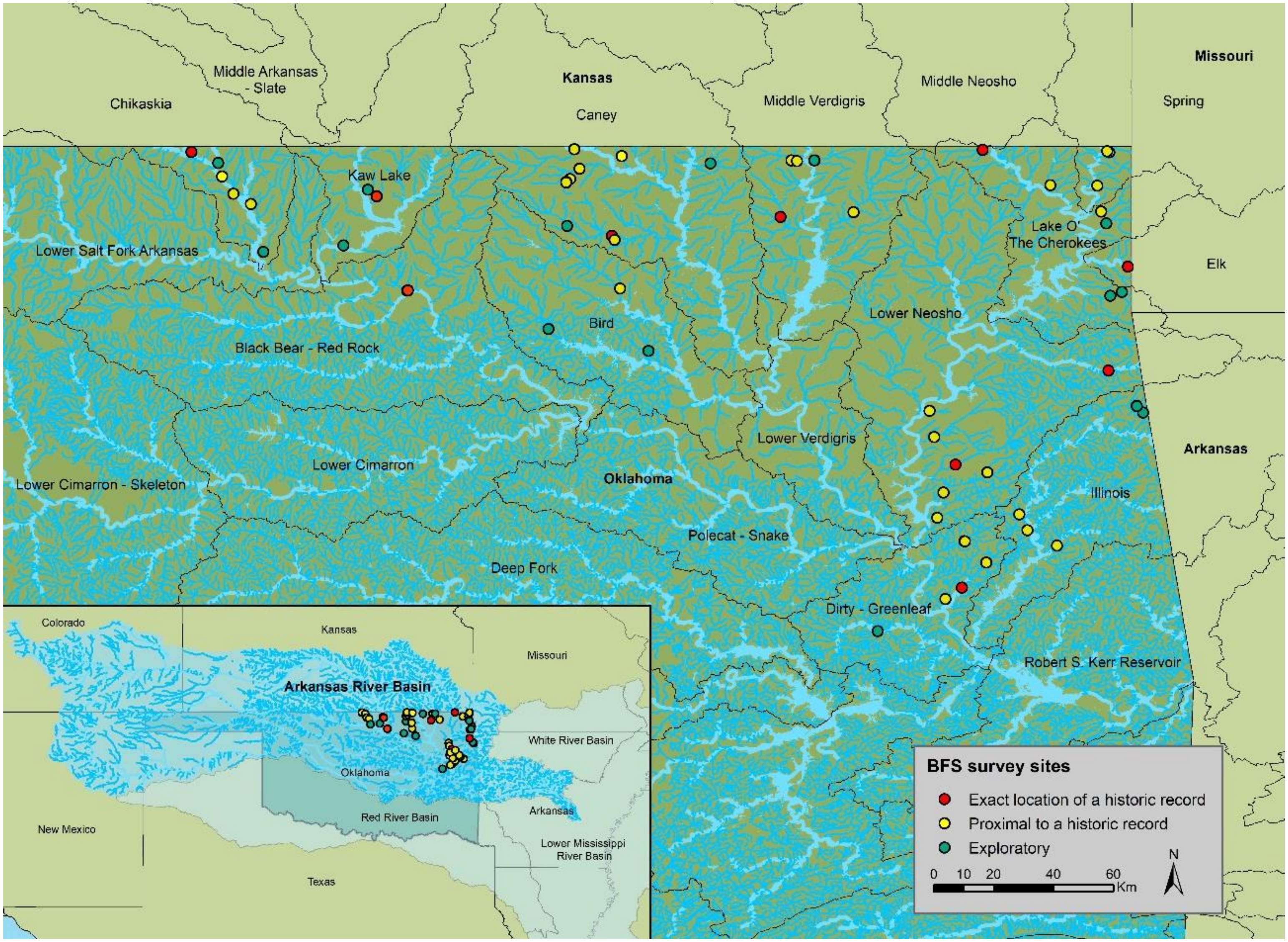

Figure 2. Map of Bluntface Shiner survey locations categorized by proximity to historical records.

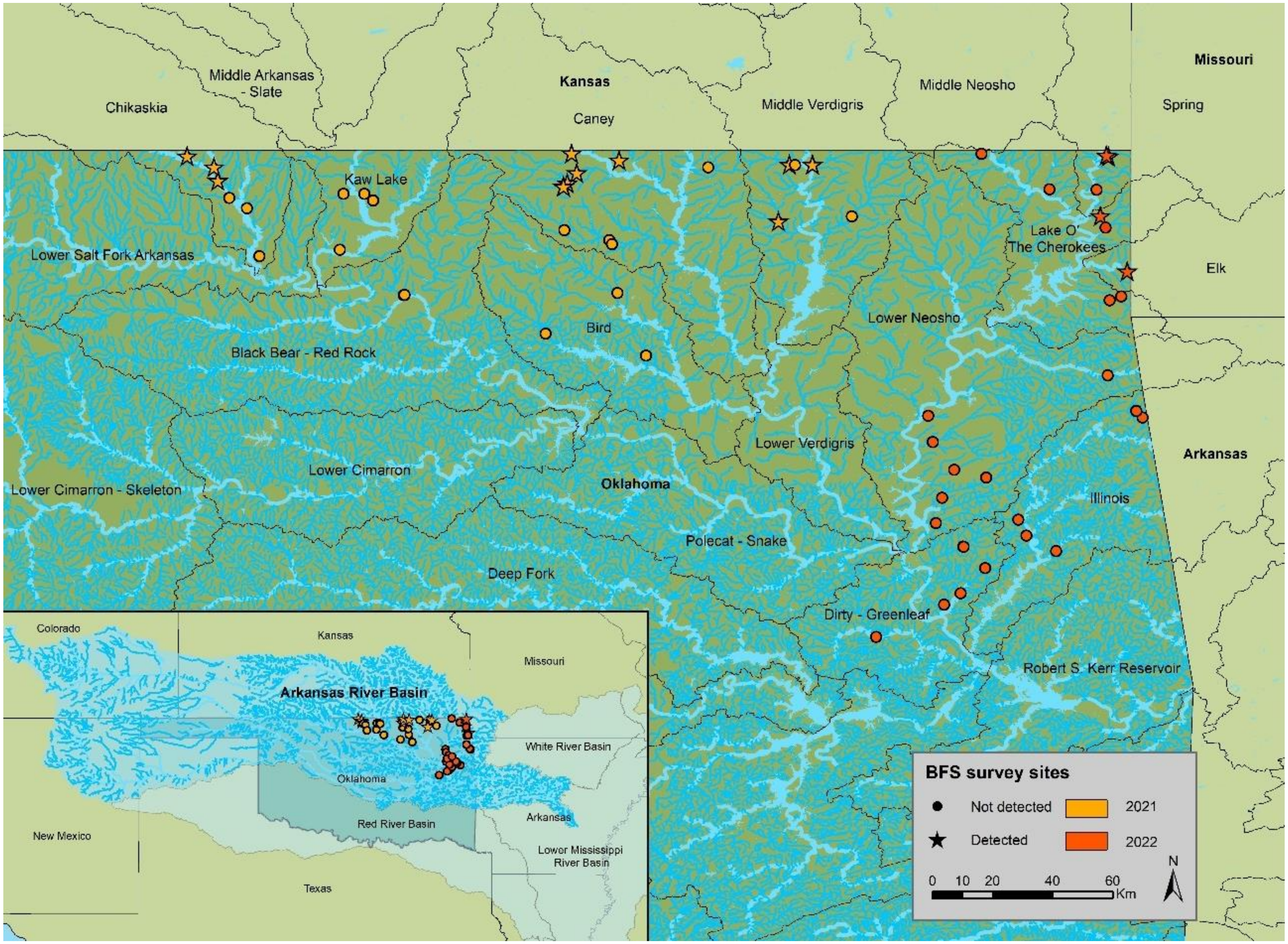

Figure 3. Map of Bluntface Shiner detections in 2021 and 2022.

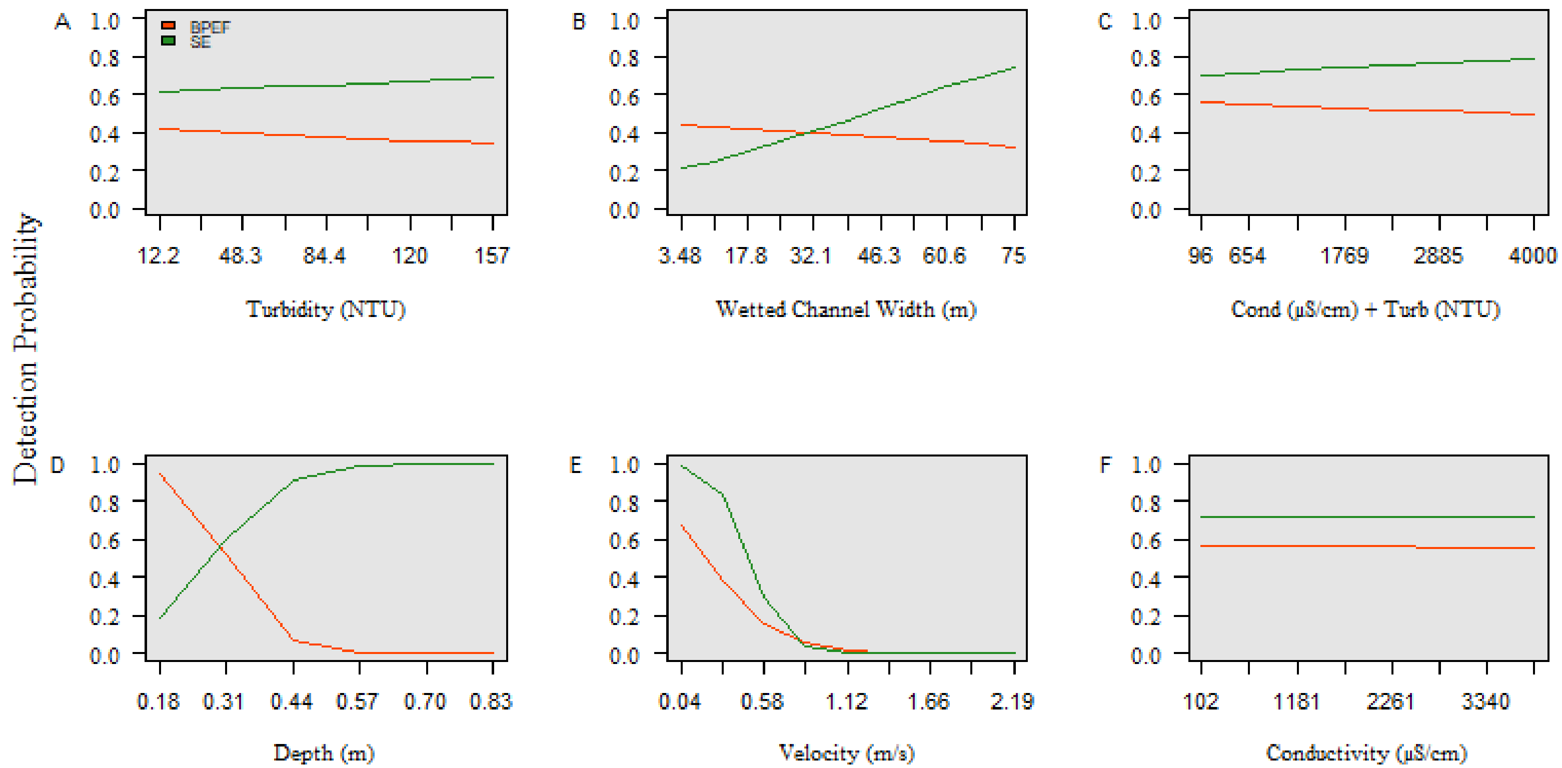

Figure 4. Comparison of relationships between detection probability of Bluntface Shiner and survey-level covariates across gear types (SE = seine net; BPEF = backpack electrofisher).

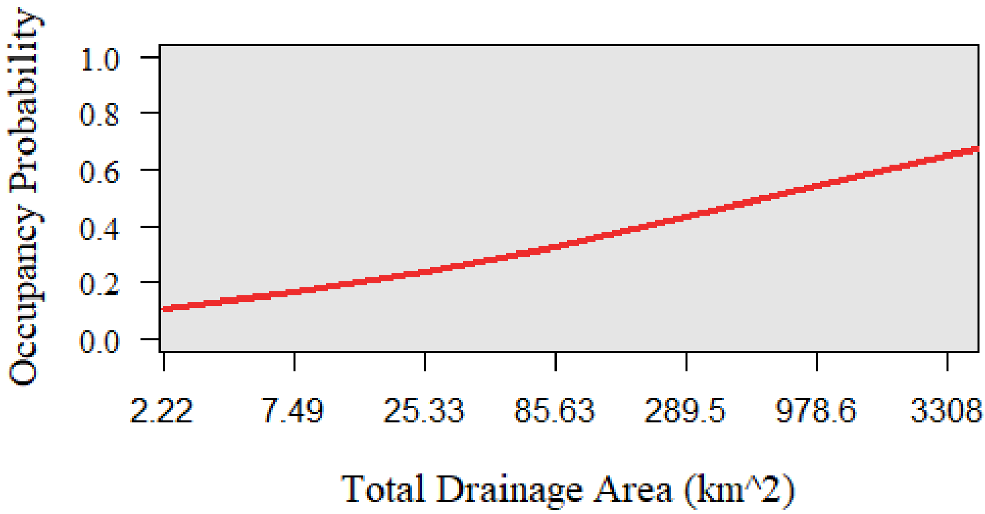

Figure 5. Estimated relationship between occupancy probability of Bluntface Shiner and total drainage area.