Species Status Assessment Report
for
The Rabbitsfoot
(*Quadrula cylindrica cylindrica*, Say 1817)

Version 1.0

Photo: Dr. Chris Barnhart, Missouri State University

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EXECUTIVE SUMMARY

U.S. Fish and Wildlife Service (Service) developed this species status assessment (SSA) report for the Rabbitsfoot (*Quadrula cylindrica cylindrica*), which was listed as threatened under the Endangered Species Act in 2013. The SSA framework uses the conservation biology principles of resiliency, representation, and redundancy to assess a species' viability at specific points in time. The purpose of this report is to support development of a recovery plan. The contents of the report are as follows: (1) summary of the subspecies' biology at the individual, population, and subspecies levels; (2) description of the influences on resource needs and viability within the framework of the three factors that contributed to listing; (3) discussion of conservation actions that biologists have implemented to benefit this mussel and its habitat; (4) description of the subspecies' current condition in terms of resiliency, representation, and redundancy; (5) calculation of projected future risk of extirpation or low condition; and (6) identification of a portfolio of watersheds that maximize viability of the Rabbitsfoot.

At the individual level, freshwater mussels need suitable physical habitat, water quality conditions, food, and host fish species to survive and reproduce. Like most freshwater mussels, the Rabbitsfoot occurs in lotic waters, which provide riverine mussel species with dissolved oxygen for respiration, remove wastes, transport food items to the juvenile and adult life history stages, and provide a diverse assemblage of suitable host fish for glochidia to encyst upon and metamorphose into a transformed juvenile. At the population level, resource needs at the individual level must be met at a broader scale, both spatially and temporally, so populations are resilient to stochastic events and the effects of anthropogenic activities. Resilient populations occur in stream reaches with a sufficient spatial extent to support an abundance of individuals of multiple age classes and with evidence of recruitment of juveniles. For a species to maintain viability, resilient populations must occur across its historical range to the maximum extent possible with hydrological connectivity maintained among them to increase the ability of the species to withstand or recover from catastrophic events.

We selected the 10-digit hydrologic unit code watershed (hereafter referred to as watersheds) as the analysis unit and aggregated records of occurrence and demographic data accordingly. Because genetic structure of populations is lacking, we grouped watersheds that occupy geographically and ecologically comparable areas into nine representation units. We assigned an ordinal classification of resilience of low, medium, or high condition within watersheds based on demographic and distributional criteria and estimated range reduction by subtracting the area of watersheds where the Rabbitsfoot is considered extirpated from the total area of watersheds historically occupied. To estimate representation and redundancy, we tallied the number of watersheds classified in each condition across the nine representation units. We modeled ordinal classification of resilience for each watershed as a function of threats to mussel persistence associated with hydrological alteration, erosion and sediment, climate change, and nutrient and chemical pollution using a Bayesian approach to ordinal regression.

The Rabbitsfoot historically occurred within at least 434 watersheds located throughout the nine representation units. Twenty-three watersheds are classified as unknown, because they contained historical occurrences, but there are no records of surveys in the watersheds in the last 30 years. Two hundred eighty-eight watersheds are classified as extirpated, a reduction of between 63%

and 70% of its historical range, depending on if unknown watersheds are classified as either extant or extirpated, respectively. Of the 123 watersheds that are extant, 83 (67.5%) are classified as low condition, 12 (9.8%) as medium, and 28 (22.8%) as high. High condition watersheds are generally isolated from other high and medium condition watersheds. Patch density of developed land use had the greatest negative effect on current condition, with a mean parameter estimate of –0.24 (95% credible interval [CI] = –0.40, –0.09). The percentage of developed land use had a similar negative effect (–0.18), although there was less certainty in the estimate.

Approximately 70% of extant watersheds are classified as low condition meaning the populations located within or among them have low resilience. Because of the substantial reduction in historical range, number of watersheds classified as low condition, and isolation of watersheds classified as high and medium condition from each other, representation for the Rabbitsfoot across its historical range is also low. For wide-ranging species like the Rabbitsfoot, assessing redundancy is challenging given it's distribution over such a large geographic extent relative to e.g., a narrow endemic species. However, the substantial reduction in historical range and lack hydrological connectivity among watersheds classified as high and medium condition decrease the ability of the Rabbitsfoot to withstand or recover from catastrophic events.

We found that the threats affecting current condition were consistent with threats affecting other freshwater mussel species. The negative effect of patch density and percentage of developed land use on current condition may reflect decreasing habitat quality from increased erosion and sedimentation and poor water quality associated with increased urban development. All watersheds currently classified as medium and high condition are isolated from major metropolitan regions, suggesting that even modest increases in developed land use could quickly decrease resiliency. Our model identified range-wide threats affecting current condition. It did not include proxy variables for some local scale threats known to affect freshwater mussels because their inclusion decreased model performance or data were not available. Natural resource professionals should consider the effect of local scale threats on the current condition of populations of the Rabbitsfoot in conservation and management planning decisions.

Using the model parameters estimated in the current condition analysis, i.e., the relative effects of each threat, we modeled the probability that a watershed would be classified as each ordinal category based on historical land use and climate patterns. These probabilities represent the sub-species present risk profile, or baseline projections, accounting for past climate and land use change. We used 2005 as the baseline for historical land use patterns because it roughly corresponds to the midpoint of contemporary occurrences and has an available land cover dataset. We calculated the historical baseline for climate patterns from 1950–2005, which represents the full historical period available from the climate dataset.

To assess future conditions, we projected the probability that a watershed would be classified as each ordinal category out to 2050, approximately three generations, to incorporate uncertainty under future scenarios of global change. We calculated future probabilities of extirpation using model parameters estimated from present risk of extirpation or low (baseline) conditions, land use projections under four emission scenarios (A1B, A2, B1, and B2), and climate projections under 12 climate scenarios derived from six global climate models (bcc-csm1-1-m, BNU-ESM, CanESM2, GFDL-ESM2G, GFDL-ESM2G, HadGEM2-ES365, and IPSL-CM5A-MR) and two representative concentration pathways (RCP 4.5 and 8.5).

Patterns for both the present risk and projected future risk of watersheds being classified as either extirpated or low condition were clear; based on land use within them, all 123 currently extant watersheds are more likely than not (>0.5) to be classified as either extirpated or low condition in the future. While the Rabbitsfoot has a high risk of extirpation range wide, spatial patterns in the amount and direction of uncertainty demonstrate useful information for decision making. For example, the present risk of extirpation is generally lower and more uncertain in the lower portions of the Little, Rolling Fork, and Cossotot rivers in the Red River representation unit, suggesting that watersheds within these rivers may represent key strongholds in the future for populations located within or among them. The modeling approach we used focuses on range-wide drivers of conditions within watersheds and does not adequately capture some site-specific threats with high consequences. For example, the negative effects of mining on mussels in parts of the range of the Rabbitsfoot have been well documented, and the risk posed by this threat as well as others not included in the model, e.g., invasive, nonnative aquatic species, must be considered in conservation planning in addition to the model estimates presented here.

We used a three-part optimization approach to identify watersheds that maximize viability of the Rabbitsfoot. The first part focuses on maintaining or increasing resilience in all watersheds that are currently classified as high condition. The second part focuses on increasing resilience in watersheds currently classified as low and medium condition with the lowest projected risk of extirpation at 2050 to increase redundancy at the sub-species level. The third part focuses on increasing representation and redundancy through reintroduction at historical sites. We applied our approach to each representation unit. We maintained all watersheds currently classified in high condition and selected ≤5 watersheds per representation unit for both the second and third parts of our approach. We constrainted the optimization to select watersheds containing critical habitat firstwhen available.

The results of our optimization approach provide a portfolio of 113 watersheds that maximize probability of persistence (i.e., resilience) and spatial spread across the landscape (i.e., representation) as an example of how the results of SSA analyses can be applied to conservation planning. The number of watersheds selected can be altered to achieve a certain level of resiliency and/or redundancy in each representation unit. By incorporating this optimization approach with local knowledge on the feasibility of and response to implementation of conservation actions and anticipated costs across the three parts of the approach, natural resource professionals can achieve an efficient, cost-effective, and transparent recovery strategy for the Rabbitsfoot.

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CHAPTER 1 – INTRODUCTION

1.1 Listing Status and Recovery Planning

The Service identified the Rabbitsfoot (*Quadrula cylindrica cylindrica*) as a candidate for protection under the Endangered Species Act (Act) on November 15, 1994, (59 FR 58982) and assigned it a status Category 2 designation. The status Category 2 list was eliminated in 1996 (61 FR 74596). On November 9, 2009, the Service added the Rabbitsfoot to its candidate list (74 FR 57804). The Service published a proposed rule to list the Rabbitsfoot as a threatened species and designate critical habitat on October 16, 2012 (77 FR 63440). The Service issued a final rule listing the Rabbitsfoot as threatened on September 17, 2013 (78 FR 57076).

At listing, the Service assigned the Rabbitsfoot a species' recovery priority number of 12, which signifies it is a subspecies with a moderate degree of threat, low recovery potential. The recovery potential for the Rabbitsfoot is considered low because of the extent of its decline, its apparent sensitivity to common pollutants, and historical and continued effects upon its riverine habitat. The factors affecting its habitat are difficult to remove or mitigate as threats are numerous, complex, not fully understood, and encompass many landowners. Additionally, issues such as nonpoint source pollution and urban industrial waste management are complex and require considerable time and effort to resolve.

The Service published a final rule designating critical habitat for the Rabbitsfoot on October 13, 2015 (70 FR 59808). In total, the Service designated 2,312 river kilometers (1,437 river miles [RMs]) for a total of 31 units in Alabama, Arkansas, Illinois, Indiana, Kansas, Kentucky, Mississippi, Missouri, Ohio, Oklahoma, Pennsylvania, and Tennessee (Figure 1.1). The Service did not designate any unoccupied stream reaches, as defined in the proposed critical habitat rule (77 FR 63440, October 16, 2012), as critical habitat. The Service defined occupied habitat as those stream reaches that contain sizeable and small populations as defined by Butler (2005, pp. 88–89).

The Service published initiation of a 5-year status review for the Rabbitsfoot in 2019 (84 FR 14669) and completed the review in 2020 (Service 2020). The status of the Rabbitsfoot detailed in the review was similar to its status at time of listing with widely scattered individuals that occur in mostly isolated areas of at least 63 rivers or creeks. Threats associated with the factors the Service used to determine the Rabbitsfoot warranted listing continue at levels similar to those detailed at that time. The Service recommended the status of the Rabbitsfoot remain as threatened.

1.2 SSA Purpose and Framework

The SSA report provides an assessment of a species current and future status to support policy-guided decisions (Service 2016, p. 4). The SSA report does not result in a decision directly. Any decisions that the Service makes regarding the legal classification of a species will be made after reviewing the SSA report and all relevant laws, regulations, and policies. The results of a proposed decision will be announced in the *Federal Register*, with appropriate opportunities for

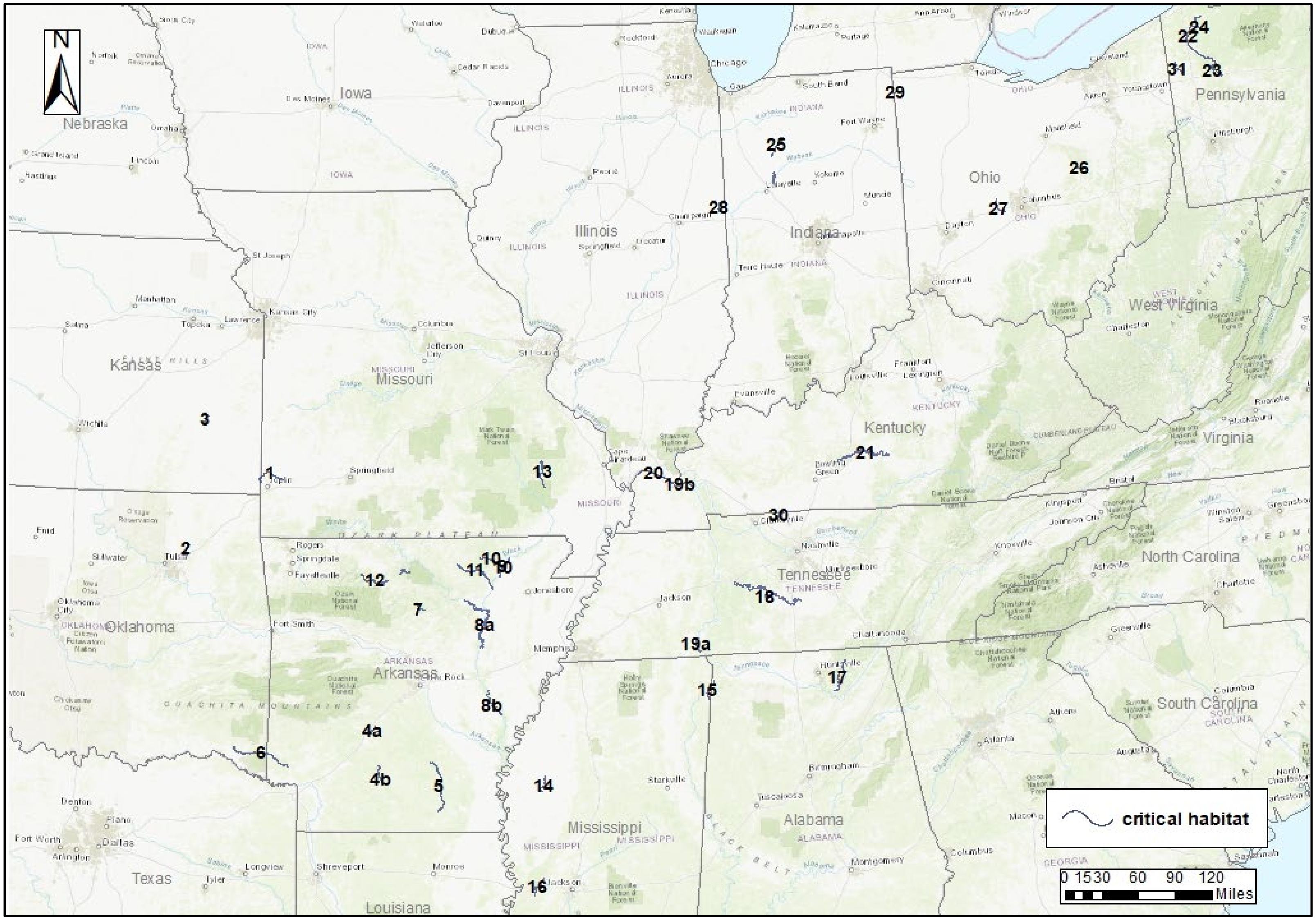

FIGURE 1.1. Index map illustrating the 2,312 river kilometers (1,437 river miles) of designated critical habitat across 31 units in Alabama, Arkansas, Illinois, Indiana, Kansas, Kentucky, Mississippi, Missouri, Ohio, Oklahoma, Pennsylvania, and Tennessee.

public input. The Service's intent is to update SSA reports with new information as it becomes available and to use them to support all functions of the Endangered Species Program from listing, to consultations, and recovery. The purpose of this SSA report is to support development of a recovery plan.

The SSA framework consists of three successive stages: (1) species' needs relative to supporting self-sustaining populations; (2) species' current condition in terms of those needs; and (3) species' forecasted future condition under plausible future scenarios (Figure 1.2). The framework guides the development of an SSA report, which is an in-depth review of the species' biology and threats, an evaluation of its biological status, and an analysis of the resources and conditions needed to maintain long-term viability (Service 2016, entire; Smith et al. 2018, entire). In the SSA analysis, we considered the likely changes—past, present, and future—that are occurring within the range of the Rabbitsfoot to help us understand what factors drive its viability. For the purpose of this assessment, we define viability as the ability of the Rabbitsfoot to sustain healthy populations in natural river systems within a biologically meaningful timeframe. We used the best scientific and commercial information available to characterize its viability.

The SSA framework uses the principles of resiliency, redundancy, and representation (i.e., "the three Rs"; Wolf et al. 2015, entire; Service 2016, entire) to assess a species' viability at specific points in time. A species with a high degree of resiliency, representation, and redundancy is better able to adapt to novel changes and to tolerate environmental stochasticity and catastrophes. In general, species viability will increase with increases in resiliency, redundancy, and representation (Smith et al. 2018, p. 306).

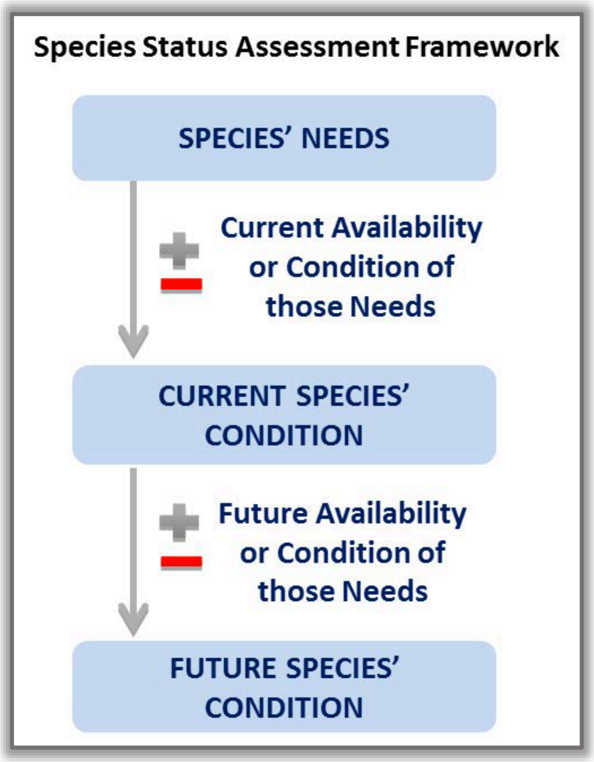

FIGURE 1.2. Species Status Assessment Framework and its three successive stages.

- **Resiliency** is the ability of a population to withstand environmental stochasticity (normal, year-to-year variations in environmental conditions such as temperature, rainfall), periodic disturbances within the normal range of variation (fire, floods, storms), and demographic stochasticity (normal variation in demographic rates such as mortality and fecundity) (Redford et al. 2011, p. 40). Simply stated, resiliency is the ability to sustain populations through the natural range of favorable and unfavorable conditions.

We can gauge resiliency by evaluating population level metrics such as: (1) demography (abundance and the components of population growth rate -- survival, reproduction, and migration); (2) genetic health (effective population size and heterozygosity); (3) connectivity (gene flow and population rescue); and (4) habitat quantity, quality, configuration, and heterogeneity. Also, for species prone to spatial synchrony (regionally correlated fluctuations among populations), distance between populations and degree of spatial heterogeneity (diversity of habitat types or microclimates) are also important considerations. Highly resilient populations are better able to withstand demographic and environmental stochasticity as well as the effects of anthropogenic activities.

- **Representation** is the ability of a species to adapt to both near-term and long-term changes in its physical (climate conditions, habitat conditions, habitat structure, etc.) and biological (pathogens, competitors, predators, etc.) environments. The ability to adapt to new environments, i.e., adaptive capacity, is essential for viability, as species need to continually adapt to their continuously changing environments (Nicotra et al. 2015, p. 1269). Species adapt to novel changes in their environment by either moving to new, suitable environments or by altering their physical or behavioral traits (phenotypes) to match the new environmental conditions through either plasticity or genetic change (Beever et al. 2016, p. 132; Nicotra et al. 2015, p. 1270). The latter (evolution) occurs via the evolutionary processes of natural selection, gene flow, mutations, and genetic drift (Crandall et al. 2000, p. 290-291; Sgro et al. 2011, p. 327; Zackay 2007, p. 1).

We can gauge representation by examining the breadth of genetic, phenotypic, and ecological diversity found within a species and its ability to disperse and colonize new areas. In assessing breadth of variation, it is important to consider both larger-scale variation such as morphological, behavioral, or life history differences, which might exist across the range and environmental or ecological variation across the range and smaller-scale variation, which might include measures of interpopulation genetic diversity. In assessing dispersal ability, it is important to evaluate the ability and likelihood of the species to track suitable habitat and climate over time. Lastly, to evaluate the evolutionary processes that contribute to and maintain adaptive capacity, it is important to assess natural levels and patterns of gene flow, degree of ecological diversity occupied, and effective population size. A species that has populations that exhibit geographic, genetic, or life history variation has greater ability to adapt to changing conditions. The more representation, or diversity, a species has, the more it is capable of adapting to changes in its environment, whether natural or anthropogenic. In the absence of species-specific genetic and ecological diversity information, we evaluate representation based on the extent and variability of habitat characteristics across the geographical range.

- **Redundancy** is the ability of a species to withstand catastrophes. Catastrophes are stochastic events that are expected to lead to population collapse regardless of population health and for which adaptation is unlikely (Mangal and Tier 1993, p. 1083). Redundancy is about spreading the risk among multiple populations to minimize the potential extinction of the species from a catastrophic event.

We can gauge redundancy by analyzing the number and distribution of populations relative to the scale of anticipated species-relevant catastrophic events. The analysis entails assessing the cumulative risk of catastrophes occurring over time. Redundancy can be analyzed at a population or regional scale, or for narrow-ranged species, at the species level. Redundancy gauges the probability that the species has a margin of safety to withstand or return from catastrophic events and thus, the viability of a species increases as its redundancy increases.

CHAPTER 2 – SPECIES BIOLOGY

2.1 Taxonomy

Say (1817, no pagination but p. 13 of publication) originally described the Rabbitsfoot as *Unio cylindrica.* The type locality is the Wabash River (Parmalee and Bogan 1998, p. 210), probably near New Harmony, Posey County, Indiana, and adjacent Illinois. Taxonomists have assigned the Rabbitsfoot to the genera *Unio*, *Mya*, *Margarita*, *Margaron*, and *Orthonymus* at various times in history. Lewis (1870, p. 218) was the first taxonomist to assign the Rabbitsfoot to the genus *Quadrula*.

Turgeon et al. (1998, p. 37) recognized *Quadrula cylindrica* as 2 subspecies, the Rabbitsfoot (*Quadrula cylindrica cylindrica*; Say 1817) and Rough Rabbitsfoot (*Quadrula cylindrica strigillata*; Wright 1898, p. 6). Serb et al. (2003, entire) used mitochondrial NADH dehydrogenase subunit 1(*nadh*1) gene marker to construct a molecular phylogeny of the genus *Quadrula* and continued to recognize the Rabbitsfoot as a member of the monophyletic *Quadrula metanevra* species group (Simpson 1900, p. 773). They did not examine enough samples to test the taxonomic validity of the Rough Rabbitsfoot as a subspecies. Sproules et al. (2006, entire) also sequenced *nadh*1 to examine variation among populations of the Rabbitsfoot from the Duck River, Tennessee, Illinois and Ouachita rivers, Arkansas, and Green River, Kentucky, and a population of the Rough Rabbitsfoot from the Clinch River, Tennessee. Their results indicated that populations in the Ouachita and Green rivers represented unique populations that warrant special conservation measures for protection. However, because of the small sample sizes and and use of only *nadh*1 they were not able to resolve the relationships between populations in the other three rivers in this pilot study. They recommended a comprehensive study using multiple genetic markers, increased sample sizes, and life-histories to reliably assess the phylogenetic relationships among the three other populations. Williams et al. (2017, pp. 52–53) used Turgeon et al. (1998, p. 37), Serb et al. (2003, entire), Sproules et al. (2006, entire), and discussions with experts on mussel systematics to develop a revised taxonomic classification and comprehensive list of the freshwater mussels of the United States and Canada that reflected recent refinement of mussel systematics. Williams et al. (2017, p. 53) reassigned the Rabbitsfoot as *Theliderma cylindrica* and no longer recognized the Rough Rabbitsfoot as a subspecies based on the results of molecular analyses from Serb et al. (2003, entire) and Sproules et al. (2006, entire). Since reassignment, Lopes-Lima et al. (2019, entire) used *nadh*1 and mitochondrial cytochrome c oxidase subunit 1 (*cox*1) gene markers and morphometric data to conduct a phylogenetic analysis of species within Tribe Quadrulini (*Cyclonaias, Quadrula, Theliderma, Trittogonia*). This study did not include new *nadh*1 sequences or additional markers for the Rabbitsfoot, so it did not provide new information on the validity of the Rough Rabbitsfoot as a subspecies. However, by including *cox*1, they identified two distinct clades for the closely related *Theliderma metanevra*, one corresponding to specimens from the Tennessee River Basin and the other to specimens from the Mobile River Basin. This result demonstrates the importance of examining multiple genetic markers, as *nadh*1 analyses alone did not detect this separation. It also provides further support of the need to evaluate the taxonomic validity of reassignment of the Rabbitsfoot and Rough Rabbitsfoot as *Theliderma cylindrica* given that it did not include new *nadh*1 sequences or additional markers for the Rabbitsfoot and that there may be a separation among these populations similar to that found for *Theliderma metanevra* using *cox*1. The Service has neither recognized the change in nomenclature nor reassignment of both subspecies as *Theliderma cylindrica* through the rule-making process and currently recognizes the following taxonomic classification for the Rabbitsfoot:

Phylum	Mollusca
Class	Bivalvia
Order	Unionoida
Family	Unionidae
Genus	Quadrula
Species	*Quadrula cylindrica*
Subspecies	*Quadrula cylindrica cylindrica*

2.2 Morphological Description

The Rabbitsfoot is a highly distinctive medium to large freshwater mussel. It has two elongate, rectangular shells that reach 12 cm (5 in) in length (Oesch 1984, p. 91-93). The periostracum is generally smooth and yellowish, greenish, or olive in color becoming darker and yellowish-brown with age and usually covered with dark green or nearly black chevrons and/or triangular markings pointed ventrally (Say 1817, p. 13; see cover photo). These patterns are absent in some individuals. Soft parts generally have an orange coloration (Davis and Fuller 1981, pp. 228–233 and 241; Oesch 1984, p. 91; Parmalee and Bogan 1998, pp. 211–212). However, Vidrine (1993, pp. 55) noted that in the Ouachita River system in Louisiana the Rabbitsfoot had black soft parts.

Bogan and Parmalee (1983, pp. 66–67) provided the following description of the shell:

> Beaks are moderately elevated; sculpture consists of a few irregular, strong ridges or coarse folds ending or continuing on the umbonal area as small tubercles. The posterior ridge is full and rounded, extending diagonally from the umbo to the posterior ventral margin while above there is a wide radial impression that may end in a slight sinus behind.
>
> The left valve has two low triangular, radially split pseudocardinal teeth and two long straight lateral teeth. The right valve has a single low triangular, deeply striated pseudocardinal tooth, often with a smaller elongated tubercular tooth on either side; the single lateral tooth is long and straight. The interdentum is either narrow or absent and the beak cavity is moderately deep. Nacre white and iridescent. The shell is much thinner posteriorly.

Oesch (1984, p. 91) described the Rabbitsfoot as follows:

> Branchial opening moderately large with brownish-yellow tentacles; anal finely papillose; supra-anal briefly connected to anal by mantle edge; gills very long and narrow, outer more narrow anteriorly; inner laminae free from visceral mass; palpi long, narrow; connected one-third of their length antero-dorsad; color of soft parts peculiar, foot with orange background striped in black; visceral mass uniorange, mantles with black pigment especially along the margins at siphonal openings.

2.3 Genetics

Sproules et al. (2006, entire) sequenced mitochondrial DNA region (*nadh*1) to characterize genetic structure of populations of the Rabbitsfoot in the Duck, Green, Illinois, and Ouachita rivers in Tennessee, Kentucky, and Arkansas. They found that individuals from the Green and Ouachita rivers separated into their own unique groups, suggesting that these populations have little if any genetic exchange with the Illinois and Duck rivers and recommended that if augmentation occur for populations in the Green and Ouachita rivers natural resource professionals should carefully consider the source population. Fobian (2007, p. 47) presented results of a genetic analysis that compared DNA sequences of 600 base pairs of *nadh*1 among individuals from the Spring, Black, Little, Verdigris, and St. Francis rivers. The samples showed no discernible genetic differences for female Rabbitsfoot among these rivers and recommended further studies be conducted to test for genetic differences among populations. These results demonstrate the need to conduct more studies that elucidate genetic structure among populations of the Rabbitsfoot across its historical range. Results of such studies will provide essential information for future propagation and reintroduction efforts.

2.4 Life Cycle

Like other freshwater mussels in the order Unionida, the Rabbitsfoot has a complex life cycle that includes an obligate ectoparasitic stage, which requires a host fish for successful reproduction (Figure 2.1). Generally, freshwater mussels pass through four life stages: reproductive, glochidia, juvenile, and adult. These stages are detailed below.

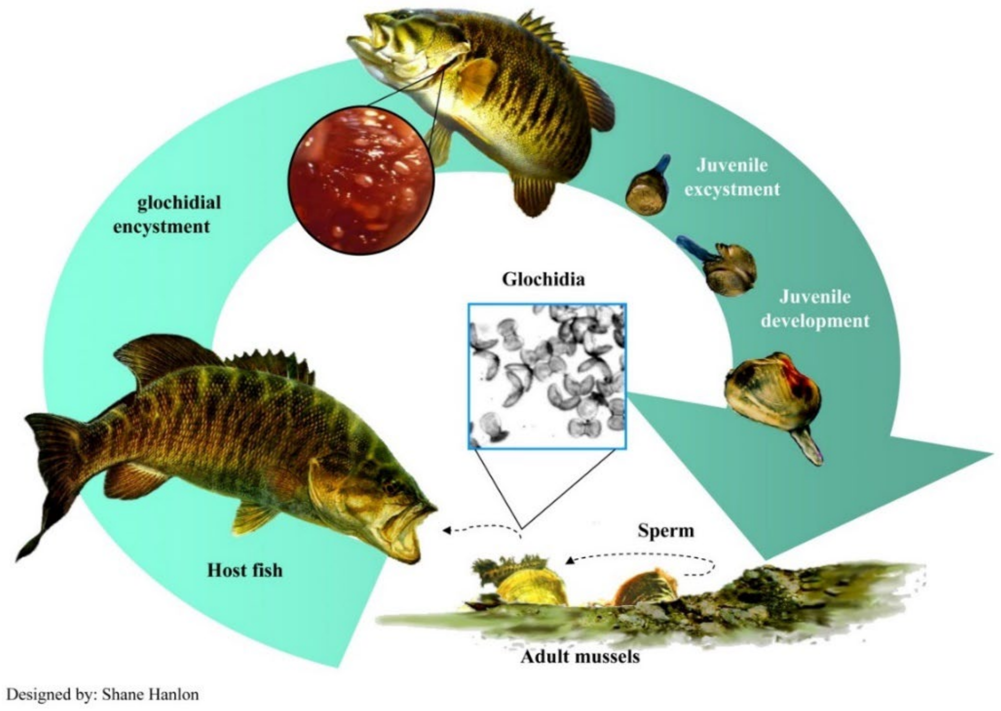

FIGURE 2.1. General mussel life cycle. Designed by Shane Hanlon, U.S. Fish and Wildlife Service..

2.4. 1 Reproductive

Males release sperm into the water column. Females deposit eggs in the interlamellar spaces, or water tubes, of the gills. The eggs are fertilized by sperm as females filter water through their incurrent siphon during feeding and respiration. Galbraith and Vaughn (2011, p. 197) examined reproductive traits of the Rabbitsfoot at three sites on the Little River in southeastern Oklahoma and found that incidence of hermaphroditism ranged from 1–17% and significantly more females at all sites combined but did not observe this difference at individual sites.

2.4.2 Glochidia

The modified larvae, called glochidia, begin maturation in the female's gills until they reach an obligate ectoparasitic stage that requires a host fish. Fobian (2007, p. 65) calculated a mean length and height of glochidia as 0.20 mm (0.008 in) and 0.19 mm (0.007 in), respectively. Like other members of the genus *Quadrula,* females of this mussel species use all four gills as a brooding pouch, called the marsupium, for the glochidia (LeFevre and Curtis 1912, pp. 115–118; Howard 1914, p. 21; Ortmann 1919, p. 54; Fobian 2007, p. 26). The Rabbitsfoot is tachytictic meaning a short-term brooder, with females brooding between May and late August in the Spring, Black, and Little rivers in Arkansas, Kansas, and Missouri (Barnhart and Baird 2000, p. 6; Fobian 2007, p. 64).

Studies have reported estimates of fecundity for freshwater mussels ranging from 75,000–3,000,000 (Coker et al. 1921, p. 144) and 9,647–325,709 (Haag and Staton 2003, pp. 2125) glochidia per female depending on species, age, and size of the individual. Fobian (2007, pp. 18–19) quantified fecundity as total number of glochidia per female in captivity. His counts excluded unfertilized eggs and any portions of the brood released before capture, and therefore are a conservative estimate of fecundity. Mean fecundity for females from the Little and Spring rivers was 50,525 ± SD 17,326 (n=4) and 46,016 ± SD 26,400 (n=3), respectively. Yeager and Neves (1986; p. 333) found mean fecundity of females from the Black River (169,484 ± SD 51,716, n=8) was greater than mean fecundity for the Little and Spring rivers.

Female Rabbitsfoot release glochidia within conglutinates, which are defined as aggregates of eggs, formed as molds in the water tubes of the marsupial demibranch of the female (Lefevre and Curtis 1912, p. 136; Figure 2.2). The females typically retain glochida and fragmentary conglutinates in their mantle and attract sight-feeding minnows by using it as a visual lure (Barnhart et al. 2008, p. 379–380; Figure 2.3) at which time they reflexively release small quantities of conglutinate fragments and free glochidia into the water column. Some glochidia attach and encyst on the gills of a suitable host fish (see 3.1.4) which provide nourishment for the glochidia to continue development.

2.4.3 Juvenile

When metamorphosis is complete, the juvenile mussels excyst from the host fish and drop onto the stream floor. If habitat and water quality conditions are favorable, juveniles develop into adults. Fobian (2007, p. 13) observed that individuals from the rivers he studied were often highly aggregated with many even-aged individuals suggesting juveniles excyst simultaneously from host fish and remain in the same general location throughout their entire life.

FIGURE 2.2. Glochidia within conglutinates. Photo: Dr. Chris Barnhart, Missouri State University

FIGURE 2.3. A brooding female Rabbitsfoot displaying mantle lure, which consists of orange excurrent aperture encircled by a white mantle. Photo: Dr. Chris Barnhart, Missouri State University

2.4.4 Adult

Rabbitsfoot mussels reach adulthood when they become sexually mature. Fobian (2007, p. 50) examined shell annuli of reproductive individuals and inferred age at sexual maturity of the Rabbitsfoot as between four to six years.

2.5 Survival and Growth

Freshwater mussels in the order Unionida are generally described as long-lived and slow-growing in large part based on data from the Freshwater Pearl Mussel (*Margaritifera margaritifera)*, which grows slowly and reaches an age of 100–200 years (Bauer, 1992. entire; Ziuganov et al., 2000, entire). Studies of other mussel species reported shorter life spans. Michaelson and Neves (1995, pp. 331–333) used age-length keys to estimate average age for the Dwarf Wedgemussel (*Alasmidonta heterodon)* as 10 years. Haag and Rypel (2011, pp. 234–236) estimated median ages for mussels in Tribes Ambemini, Lampsilini, Pleuobemini, and Quadrulini as 25, 15, 32, and 36 years, respectively. Most studies of age and growth have been based on the rings laid down in the shell, which biologists interpreted as reflecting annual pauses in mussel growth during the winter. This inference is supported by analyses of shell microchemistry (Veinott and Cornett 1996, entire). However, recent studies of growth rates based on direct measurements of marked individuals in the field suggest that growth rings are not annual and that earlier estimates based on growth rings may underestimate longevity by a factor of 3–10 (Anthony et al. 2001, pp. 1352–1354).

Fobian (2007, p. 32) reported that shell length of 32 brooding female Rabbitsfoot ranged from 82–122 mm and from that estimated ages as between 6–17 years. Yeager and Neves (1986, p. 332) counted external growth rings of 39 gravid females belonging to the closely related Rough Rabbitsfoot and from that estimated ages as between 10–22 years. As part of the same project, Yeager and Saylor (1995, p. 4) estimated the ages of gravid females of the Cumberland Monkeyface (*Quadrula intermedia*) as between 14–22 years. Christian et al. (2000, p. 48) fitted von Bertalanffy growth curves to estimate longevity for the Mapleleaf (*Quadrula quadrula)* as between 15–30 years. Sansom et al. (2016, p. 22) applied dendrochronology techniques and Ford-Walford analyses to estimate longevity for the Pimpleback (*Cyclonaias pustulosa)* and Pistolgrip (*Trigonia verrucosa)* as between 25–34 years for both species. Because of their long life spans, freshwater mussel species could persist decades under conditions of negative population growth, which would make it challenging to understand declines in abundance (Strayer et al. 2004, p. 433).

Growth rates for freshwater mussels vary among species, rivers and streams, but generally they grow relatively rapidly for the first few years (Chamberlain 1931, Scruggs 1960, pp. 28–30; Negus 1966, pp. 517–518) then slow appreciably (Bruenderman and Neves 1993, p. 88; Hove and Neves 1994, pp. 34–36). This reduction in growth rate with years is correlated with sexual maturity and hypothesized to be a result of the diversion of energy from growth to gamete production (Baird 2000, pp. 63–71). Although we do not have sufficient data to assess growth rate for the Rabbitsfoot, it is likely similar to other freshwater mussels.

2.6 Movement

In general, freshwater mussels are sedentary animals (Fuller 1974, p. 241). Fobian (2007, p. 13) considered the Rabbitsfoot highly mobile relative to other mussel species. He observed tracks created by the Rabbitsfoot of 1–2 m in length while walking along shallow banks in the Black River, Arkansas, and one individual move approximately 1.5 m in just a few hours after disturbance (Fobian 2007, p. 13–14). He also reported observing large numbers of adults traveling up and down the bank toward shallower water during the summer months but not in the winter. He hypothesized that this lateral migration towards shallower water during the summer months may indicate seasonal movement to facilitate reproduction by increasing exposure to the opposite sex during spawning or to host fish (Fobian 2007, p. 25). Amyot and Downing (1998, pp. 355–357) found that aggregation varied over the season for a population of *Elliptio complanata* bringing mussels closer together during spawning and suggested a functional reproductive role for this horizontal movement. Further studies are needed to assess movement of adult Rabbitsfoot relative to other species and as a reproductive strategy.

CHAPTER 3 – RESOURCE NEEDS

3.1 Individual-level Resource Needs

Resource needs at the individual level are those life history characteristics that influence the successful completion of each life history stage. In other words, these are the survival and reproduction needs that make the species resilient to particular natural or anthropogenic influences. At the individual level, North American freshwater mussels need suitable physical habitat, water quality conditions, food, and host fish species to survive and reproduce (Table 3.1). Like most freshwater mussels, the Rabbitsfoot occurs in lotic waters, i.e., continuous flowing water such as rivers, streams, and creeks. Lotic waters provide riverine mussel species with dissolved oxygen for respiration, remove wastes, transport food items to the juvenile and adult life history stages, and provide a diverse assemblage of suitable host fish for glochidia to encyst upon and metamorphose into a transformed juvenile.

TABLE 3.1. Life history stages and individual resource needs of the Rabbitsfoot.

Life History Stage	Resource Needs[1]	Resource Function[2]
All	▪ Small- to medium-sized streams, and some larger rivers with appropriate flows for each life history stage ▪ Appropriate water quality conditions	B, F, S, D
Reproductive: March through May	▪ Sexually mature males in close proximity to sexually mature females ▪ Appropriate spawning temperatures	B, S
Glochidia: May through August	▪ Presence of brooding females ▪ Sufficient abundance of suitable host fish species for attachment	B, D
Juveniles	▪ Host fish dispersal ▪ Suitable substrate for settlement ▪ Adequate food availability	F, S, D
Adults	▪ Suitable substrate ▪ Adequate food availability	F, S

[1] These resource needs are common among North American freshwater mussels; however, due to lack of species-specific research, parameters specific to the Rabbitsfoot are unavailable.

[2] B=breeding; F=feeding; S=sheltering; D=dispersal

3.1.1 Habitat

As is the case for other freshwater mussels, biologists classify habitat of the Rabbitsfoot using generalized physical characteristics from anecdotal observations. The Rabbitsfoot occurs in small- to medium-sized streams and some larger rivers (Ortmann 1919, p. 56; Gordon and Layzer 1989, p. 32; Cummings et al. 1992, p. 32). Substrates include sand, sediment, gravel to medium-sized cobble, and a mixture of sand and gravel (Scammon 1906, p. 348; Parmalee 1967, p. 37; Gordon and Layzer 1989, p. 32; Fobian 2007, p. 13). Biologists have reported the Rabbitsfoot from shallow water areas near banks and adjacent runs, riffles, and shoals, with current ranging from moderate to swift (Scammon 1906, p. 348; Ortmann 1919, p. 56; Murray and Leonard 1962, p. 65; Gordon and Layzer 1989, p. 32), but also from depths of 3.0 m (Parmalee 1967, p. 37). Watters (1988, p. 13) found specimens along the shore adjacent to a good current and laying in a pile of decaying branches. Fobian (2007, p. 13) found specimens in shallow water near flow refuges along the bank between 0.1–2.0 m deep, adjacent to good current and often near Water Willow (*Justicia* spp.) beds and woody debris. Flow refuges are patches of stable riverine habitat characterized by a combination of low shear hydraulic stress (the force exerted by moving water near the bottom of the river) and high sediment stability during high flow periods and some minimum level of current velocity during low flow periods (Layzer and Madison 1995, pp. 335–337; Strayer 1999a, pp. 471–475; Hardison and Layzer 2001, pp. 79–82; Morales et al. 2006, pp. 668–671). Researchers hypothesize that flow refuges allow freshwater mussels to maintain their position in suitable habitat during floods (low shear stress) and also to access food, oxygen, and removal of waste product during low water periods (Strayer et. al 2004, pp. 433–434). Some biologists observed adult Rabbitsfoot laying horizontally on the surface of the sediment rather than burrowing into it (Ortmann 1919, p. 56; Watters 1988, p. 13; Fobian 2007, p. 13) and hypothesized flow refuges may play an important role in decreasing the likelihood of displacement into unsuitable habitat. However, in higher energy habitats where flow refuges are not available, e.g., along the lower Buffalo River, Tennessee, biologists observed adult Rabbitsfoot in relatively high concentrations burrowing completely into banks composed of sandy silt (T. Amacker, Tennessee Valley Authority [TVA], pers. comm., 2016).

Habitat requirements for the juvenile life history stage of freshwater mussels are largely unknown. The initial distribution of juveniles is dependent upon the location of their host fish when they excyst from it and flow conditions within the water column at that time. Neves and Widlak (1987, pp. 3–5) quantitatively sampled juvenile mussels in three habitat types, pools, runs, and riffles and two microhabitat types, the downstream side of boulders in the stream bed and along stream banks. They found that adults occurred most frequently in riffles and density of juvenile mussels was greatest behind boulders; however, because such habitat was limited at their study site, the abundance of juveniles was greatest in riffles and runs. Yeager et al. (1994, p. 219) found that juveniles of the Rainbow Mussel (*Villosa iris*) burrowed into the sediment or into interstitial spaces in substrate and fed on algae, bacteria, and detritus found in the interstitial water. Hastie et al. (2000, pp. 65–66) found a strong positive relationship between density of adult and juvenile Freshwater Pearl Mussels, demonstrating that both life stages were usually found in the same parts of the river bed and thus, had broadly similar habitat requirements. Both adults and juveniles had a ‘preference’ for boulder-dominated substrata, but density of adults was strongly assocated with surface substratum diversity, whereas density of juveniles was strongly associated with proportion of fine interstitial material i.e., areas dominated with boulders that also contained enough sand for burrowing. Geist and Auerswald (2007, pp. 2305–2309) observed that adult and juvenile Freshwater Pearl Mussels occurred simultaneously in the same habitat patches, but used different microhabitas. During summer, adults were usually partly buried and filtered free-flowing water, whereas juveniles were completely buried and thus exposed to interstitial water. Additional experiments are needed to determine whether juveniles have habitat preferences or their distributions are determined more by chance differential survival rates among habitats.

3.1.2 Water Quality Conditions

Biologists have not conducted investigations to identify the appropriate range of water quality parameters that define suitable habitat conditions for the Rabbitsfoot. However, as a group freshwater mussels are particulary sensitive to changes in the following water quality parameters: (1) dissolved oxygen < approximately 2–3 parts per million (ppm) (Chen et al. 2001, p. 214); (2) ammonia >0.5 ppm total ammonia-nitrogen (TAN) (Augspurger et al. 2003, p. 2572); (3) potassium <4 mg/L (Imlay 1973, p. 97); (4) elevated temperature > approximately 30° C (Pandolfo et al. 2010, p. 960; Ganser et al. 2013, p. 1172); (5) heavy metals (Valenti et al. 2005, pp. 1242–1245; Wang et al. 2010, p. 2062; Wang et al. 2016, pp. 5–10); and (6) excessive total suspended solids (Gascho Landis and Stoeckel 2016, p. 232–234; Hansen et al. 2016, entire). Fobian (2007, p. 20) found successfully metamorphosed juveniles dropped from host fish beginning at 23° C.

3.1.3 Food and Feeding Behavior

Although the diet and feeding behavior of the Rabbitsfoot are unknown, they are likely similar to other freshwater mussels. Freshwater mussels are omnivores that feed across trophic levels on detritus, bacteria, phytoplankton, zooplankton, and dissolved organic matter (Churchill and Lewis 1924 pp. 460–461; Silverman et al. 1997, p. 1859; Nichols and Garling, 2000, pp. 873–874; Roditi et al 2000, entire; Christian et al. 2004, pp. 105–108). Adult freshwater mussels primarily feed by filtering microscopic particulate matter suspended in the water column but also are capable of deposit feeding by accessing food items from interstitial spaces in substrate through cilia-generated water currents that pull material in through the anterior portion of the shell while the foot is extended and of pedal feeding directly from the sediment (Allen 1914, p. 128; Yeager et al. 1994, p. 219–221; Raikow and Hamilton 2001, pp 516–517). Studies of juvenile freshwater mussels have found that they primarily use pedal feeding behavior to deposit particulate matter into the mantle cavity through the pedal gape until they fully develop structures for filter feeding (Reid et al 1992, entire; Yeager et al. 1994, p. 219–221; Gatenby et al. 1996; pp. 601–602).

3.1.4 Host Fish

Barnhart and Baird (2000, p. 16) collected glochidia of the Rabbitsfoot from the Black River, Arkansas, and tested five species of fish as potential hosts. They observed transformation of glochidia on Blacktail Shiner (*Cyprinella venusta*) but not on Fathead Minnow (*Pimephales promelas*), Mimic Shiner (*Notropis volucellus*), Emerald Shiner (*Notropis atherinoides*) or Bluegill (*Lepomis macrochirus)*. Fobian (2007, pp. 36–42) tested 27 species of fish within four families as potential hosts for the Rabbitsfoot from the Spring, Black, and Little rivers in Arkansas, Kansas, and Missouri. Blacktail Shiner, Red Shiner (*Cyprinella lutrensis*), Bluntface Shiner (*Cyprinella camura*), Spotfin Shiner (*Cyprinella spiloptera*), and Cardinal Shiner (*Luxilus cardinalis*) were suitable hosts for glochidia of individuals from all three rivers. He observed variable metamorphosis on the Rosyface Shiner (*Notropis rubellus*). Glochidia of individuals from the Black River metamorphosed at a much higher percentage than sympatric glochidia of individuals from the Spring River, but he was unable to conclude if metamorphosis differences between glochidia of individuals from these two rivers were significant due to small sample sizes

(Fobian 2007 pp. 41–42). He tested one Striped Shiner (*Luxilus chrysocephalus*) from the Little River with glochidia of individuals from the Little River, but this fish had a metamorphosis success of 70% (Fobian 2007 pp. 40–42). Metamorphosis success of glochida of individuals from the Black River and Little River was high on Emerald Shiner (*Notropis atherinoides*) from the Black River. However Emerald Shiners from the Little River were poor hosts for glochidia of individuals from the Little and Spring rivers (Fobian 2007 p. 42). Watters et al. (2009, p. 19) confirmed transformation of glochidia collected from gravid female Rabbitsfoot on Rainbow Darter (*Etheostoma caeruleum*) and Striped Shiner. More recently, host fish trials identified Scarlet Shiner (*Lythrurus fasciolaris*) and Striped Shiner as suitable hosts for glochidia collected from females in the Paint Rock River, Alabama (T. Fobian, Alabama Department of Conservation and Natural Resources [ADCNR], pers. comm., 2020). However, more studies are needed both in the laboratory to determine suitable host fish species and if they vary among populations and in the field to verify that these relationships occur in nature, especially for the eastern part of the range of the Rabbitsfoot. Furthermore, demographic and movement studies of host fish species are also needed since the success of conservation actions such as augmentation or reintroduction through captive propagation is contigent upon the distribution and abundance of viable populations of suitable host fish species.

3.2 Population-level Resource Needs

Resource needs at the population level are components of the life history of the Rabbitsfoot that most influence the ability of populations to be resilient to stochastic events such as droughts and floods. For populations of the Rabbitsfoot to be resilient to stochastic events, their resource needs at the individual level must be met at a broader scale, both spatially and temporally. Populations need to occur in stream reaches with a sufficient spatial extent to support an abundance of individuals of multiple age classes and with evidence of recruitment such that populations are able to recover from stochastic events (Figure 3.1). Populations that are distributed across tributaries in addition to the mainstem of a river provide the spatial complexity necessary to facilitate recovery of populations from stochastic events if there are no barriers to movement of host fish (Figure 3.1). Because males release sperm into the water column, where it drifts until females filter water containing sperm through their incurrent siphon, successful individual reproduction and population viability require sufficient numbers of females downstream of sufficient numbers of males.

3.3 Species-level Resource Needs

For a species to maintain viability, it must have sufficient representation and redundancy across its range. Representation describes the ability of a species to adapt to changing environmental conditions over time. It can be measured by the breadth of genetic and environmental diversity within and among populations and gauges the probability that a species is capable of adapting to environmental changes. For the Rabbitsfoot to have sufficient representation, resilient populations should occur across a wide geographic extent and in several river systems. Natural levels of connectivity should be maintained among populations to allow for the exchange of novel and benficial adaptations where connectivity is high or is the mechanism for localized adaptation and variation where connectivity is lower or the species is more naturally isolated (Figure 3.1). Redundancy describes the ability of a species to withstand catastrophic events. It can be measured by the number of populations, their resilience, distribution, and connectivity. For the Rabbitsfoot, redundancy is characterized by having multiple, resilient and representative populations distributed across a wide geographic extent while maximizing connectivity. Connectivity allows for immigration and emigration among populations and increases the likelihood of recolonization should a population become extirpated (Figure 3.1).

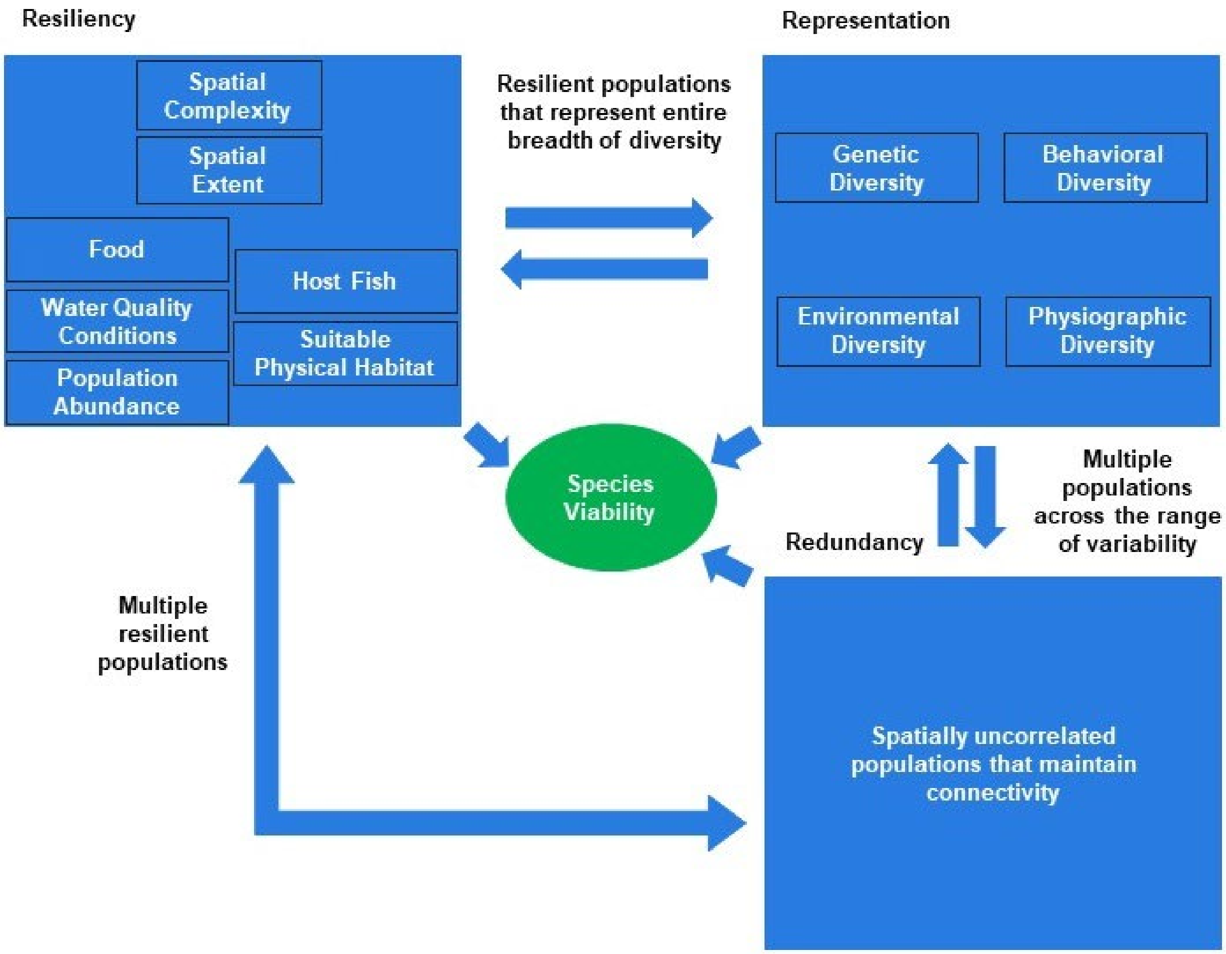

FIGURE 3.1. How resiliency, representation, and redundancy are related to species viability.

CHAPTER 4 – INFLUENCES ON VIABILITY

The Service's final rule for the Rabbitsfoot (78 FR 57076) listed it as threatened due to the immediacy, severity, and scope of threats from a combination of three out of the five factors described in section 4(a)(1) of the Act: (A) The present or threatened destruction, modification, or curtailment of its habitat or range; (D) the inadequacy of existing regulatory mechanisms; and (E) other natural or manmade factors affecting its continued existence. The Service did not include Factor B, Overutilization for Commercial, Recreational, Scientific, or Educational Purposes, or Factor C, Disease or Predation, in the final rule because there was no evidence that threats from these factors were affecting the status of the Rabbitsfoot. The Service also listed these three factors as the primary threats to the species in the 2020 5-year review (Service 2020, pp. 46–59). In this chapter, we describe the influences on resource needs and viability of the Rabbitsfoot (Figure 4.1) within the framework of the three factors that contributed to its listing as threatened under the Act and discuss conservation actions that biologists have implemented to benefit this mussel and its habitat.

4.1 Factor A: Destruction, Modification, or Curtailments of Its Habitat or Range

Limited mobility, long lifespans, filter-feeding habits, and a parasitic larval stage make freshwater mussels in the family Unionidae highly vulnerable to human alteration of their habitats. Losses of many populations of unionid mussels prior to the 1950s are directly associated with the acute effects of pollution from chemical spills or industrial effluents or habitat destruction from construction of dams or channelization (Strayer et al. 2004, p. 435). More recently, biologists identify chronic effects of these threats e.g., fragmentation of habitat and isolation of populations as well as interactions among them as the cause for the continued decline in populations of freshwater mussels (Strayer et al. 2004, p. 435; Galbraith et al. 2010, entire).). Yet, the decline in freshwater mussel species remains enigmatic as the chronic effects of these threats do not fully explain declines in their populations (Haag et al. 2019, entire).

4.1.1 Impoundments

Declines in populations of unionid mussels are most frequently attributed to the destruction and modification of riverine habitat by dams and resultant impoundments constructed for navigation, flood protection, and generation of hydroelectric power (Neves et al. 1997, pp. 63–64; Strayer 2006, p. 278; Downing et al. 2010, pp. 156–159). Within impounded areas upstream of dams, decreased survival and reproductive success of riverine mussels are associated with the direct loss of free-flowing reaches and associated flow refuges, increase in water depth and sedimentation, decrease in dissolved oxygen levels, and alteration or loss of resident host fish species (Neves et al. 1997, pp. 63–64; Pringle et al. 2000, pp. 810–815; Watters 2000, 265–266). Downstream of dams, decreased survival and reproductive success of riverine mussels are associated with changes in flow and temperature regimes, channel scouring and bank erosion, decreased dissolved oxygen levels, and changes in fish assemblages (Williams et al. 1992, p. 7; Layzer et al. 1993, p. 69; Neves et al. 1997, pp. 63–64; Pringle et al. 2000, pp. 810–815; Watters 2000, 265–266). The remaining populations of freshwater mussels are isolated from each other making them more likely to become extirpated from demographic stochastic events, such as random fluctuations in reproductive success as well as environmental stochastic events, such as droughts, floods, and effects of anthropogenic activities e.g., chemical spills or unauthorized discharges (Layzer et al. 1993, pp. 68–69; Neves et al. 1997, pp. 63–75; Watters 2000, pp. 264–265, 268; Pringle et al. 2000, pp. 810–815).

Researchers have directly attributed the decline of freshwater mussels within the Arkansas, Red, White, Tennessee, Cumberland, Mississippi, and Ohio River basins to construction of numerous dams and resultant impoundments (Miller et al. 1984, p. 109; Williams and Schuster 1989, pp. 7–10; Layzer et al. 1993, pp. 68–69; Neves et al. 1997, pp. 63–64; Obermeyer et al. 1997, pp. 113–115; Watters 2000, pp. 262–263; Sickel et al. 2007, pp. 71–78; Hanlon et al. 2009, pp. 11–12; Vaughn and Taylor 1999, pp. 915–917; Watters and Flaute 2010, pp. 3–7). Changes in the diversity and abundance of freshwater mussel species as a result of impoundment are especially well documented for the Tennessee River Basin. This river basin once possessed the most diverse and abundant mussel fauna on any river in North America, but now consists of a series of reservoirs formed by the construction of 36 multi-purpose dams that impound 2,287 RMs of the mainstem of the Tennessee River and its largest tributaries (Holston, Little Tennessee, Clinch, Elk, Flint, and Sequatchie rivers, and Bear Creek) (Neves et al. 1997, pp. 63–64). Five dams on the mainstem of the Cumberland River downstream of Cumberland Falls either impounded or impacted riverine habitat through hypolimnetic releases along approximately 90% of its 526 RMs. Dams on six of its major tributaries (Caney Fork, Obey, Laurel, and Stones rivers and Martin’s Fork of the Cumberland River) impounded >100 RMs and impacted approximately 200 RMs of additional habitat for freshwater mussels through hypolimnetic releases of (Layzer et al. 1993, p. 65). The U.S. Army Corps of Engineers (USACE) began navigational improvements on the Ohio River in 1830, which now include 21 lock and dam structures stretching from Pittsburgh, Pennsylvania, to Olmsted, Illinois, near its confluence with the Mississippi River. Additionally, the USACE constructed lock and dam structures on numerous tributaries of the Ohio River. For example, the USACE constructed nine locks and dams that alter a 72-RM stretch of the Allegheny River in Pennsylvania from Armstrong County to Pittsburgh and a series of six locks and dams on the lower half of the Green River that extend upstream to the western boundary of Mammoth Cave National Park, Kentucky.

Construction of dams and resultant impoundments likely contributed more to range reduction and declines in populations of the Rabbitsfoot throughout its range than any other single threat. Vaughn and Taylor (1999, p. 915) found that populations of freshwater mussels including the Rabbitsfoot did not recover from the effects of hypolimnetic releases on the Little River in Oklahoma until 12 RMs downstream of the impoundment, with a peak of species richness and abundance of mussel species at 33 RMs downstream. Within the Ohio River Basin, biologists have attributed declines in populations of the Rabbitsfoot to navigational locks and dams on the Ohio, Allegheny, Monongahela, Muskingum, Kentucky, Green, Barren, and White rivers. Because these dams and resultant impoundments are essential for navigation, flood protection, and generation of hydroelectric power removal of them is not reasonable, but conservation actions can be developed to address how to modify operation to support ecologically suitable conditions for imperiled aquatic species.

Even low-head dams, low water crossings, and perched culverts constructed across a river or creek channel can have some of these same adverse effects on freshwater mussels and their host

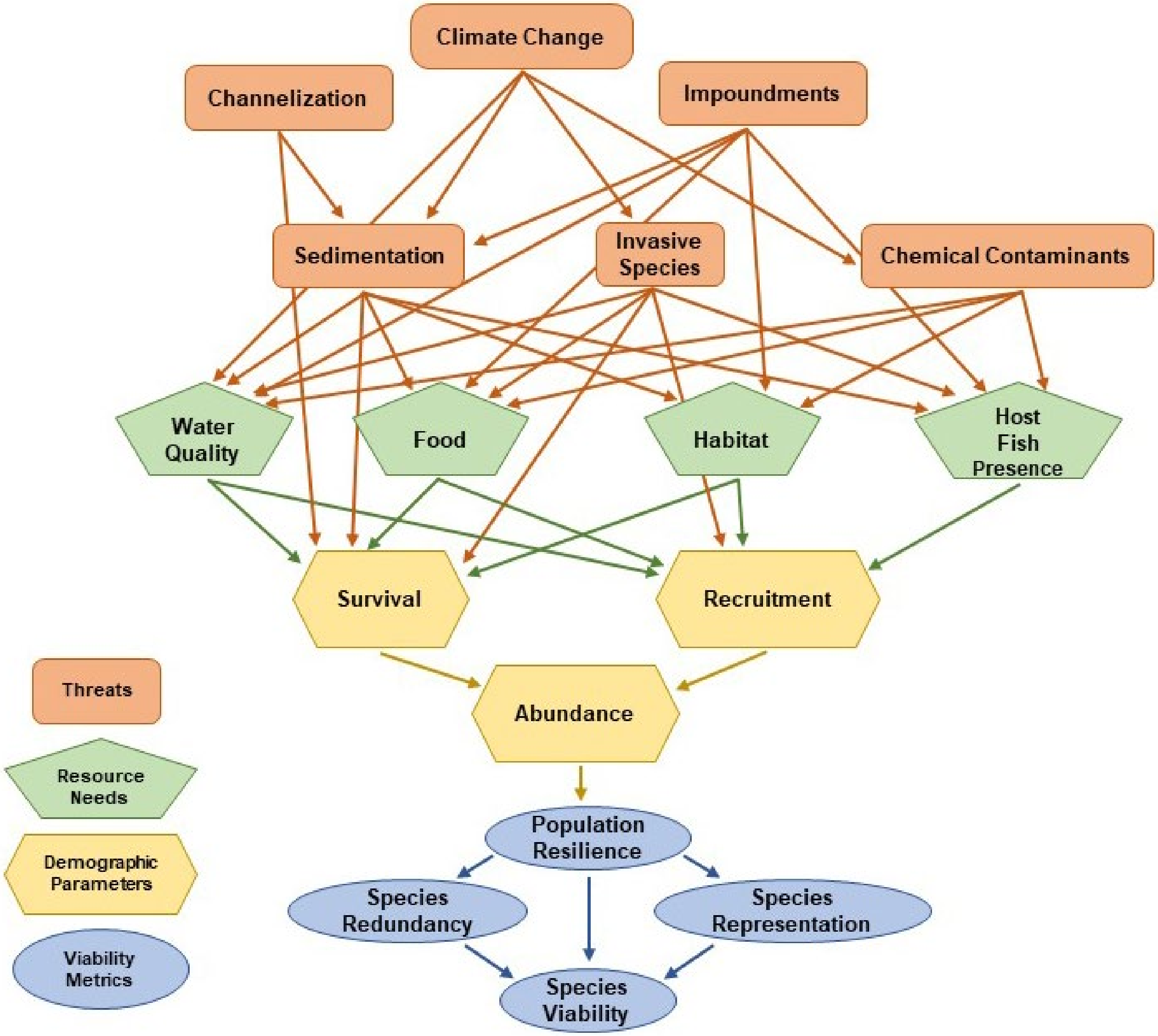

FIGURE 4.1. Influence diagram illustrating relationships between threats associated with the three factors, resource needs, demographic parameters, and viability of the Rabbitsfoot.

fish, particularly reducing species richness and evenness of mussel taxa and impeding movement of host fish species. Watters (1996, pp. 80–81) found that upstream distribution of the Fragile Papershell (*Leptodea fragilis)* and Pink Heelsplitter (*Potamilus alatus*) was restricted to the portion of the river downstream of the low-head dams suggesting that the host fish of these mussel species were unable to move upstream of these obstacles. Dean et al. (2002, pp. 235–236) surveyed freshwater mussels upstream and downstream of two low-head dams on the Neosho River and found a significant difference in mean species richness and eveness, but not abundance suggesting low-head dams affect freshwater mussel assemblages on this river. However, results of other studies demonstrate that low-head dams may have unanticipated positive effects on downstream habitat and biota (reviewed in Gangloff 2013, pp.476–479). In rivers and creeks affected by anthropogenic activities, low-head dams and their impoundments may perform some key ecological functions including filtrating and detoxifying anthropogenically-elevated nutrient loads, oxygenating low-gradient streams during low-water periods, stabilizing portions of the stream beds that are needed for the persistence of fish and mussels, and barriers to movement of non-native, invasive species. Low-head dams may be indirectly or directly related to declines or extirpation of the Rabbitsfoot as noted by biologists in the Mahoning, Walhonding, Tuscarawas, Scioto, Olentangy, Eel, Vermilion, Duck, Neosho, and Spring (Arkansas River Basin) rivers and Alum, Big Walnut, Big Darby, and Sugar (Wabash River tributary) creeks (Butler 2005, p. 97).

4.1.2 Channelization

The objectives of most channelization activities are to simplify drainage for flood control, enhance navigation, and decrease the extent of inundation of the natural floodplain (Hubbard et al. 1993, p. 136). Belt (1975, p. 684) reported that straightening, a channelization activity implemented to enhance navigation, increased flood heights due, in part, to a decrease in stream length and an increase in gradient. As a result, flood events may be exacerbated, conveying large quantities of sediment, potentially with adsorbed contaminants, into streams. Snagging is another channelization activity, and it is used to remove obstructions such as fallen trees and debris from the channel to prevent a river from inundating the natural floodplain. The obstructions are dragged from the river bed, which decreases habitat heterogeneity important for aquatic diversity, creates unstable substrates, and increases bank erosion (Marzlof 1978, pp. 14–22). Dredging is another channelization activity that adversely affects freshwater mussels, most notably by crushing and/or removing individuals caught in the dredge path. However, the adverse effects of dredging are not limited to the excavated site as they can extend upstream and downstream of it. For example, headcutting, which is the upstream progression of river bed destabilization and accelerated bank erosion as the channel reestablishes a stable base level throughout the watershed after dredging, is especially destructive in areas without bedrock or other controls on base level (Hartfield 1993, pp. 131–139). Accelerated erosion increases sedimentation, which may smother mussels or expose them to resuspended contaminants (Stansbery 1970, p. 10; Engler 1979, p. 328–344).

Channelization activities that occurred decades ago have adversely affected habitat in many rivers across the range of the Rabbitsfoot. Snagging and dredging to maintain channels for navigation continue to adversely affect habitat in the lower portions of the mainstem of the Ohio, Tennessee, and White rivers (Butler, 2005, p. 98). The USACE (2007) dredged the lowest reaches of the Verdigris River to create a navigation channel to the Arkansas River to decrease distances traversed by barge traffic. Various entities implemented channelization activities to drain areas for agriculture and/or to decrease the probability and frequency of flood events along hundreds of RMs of the Eel, North Fork Vermilion, and Embarras rivers in the Ohio River Basin, Paint Rock River and Bear Creek in the Tennessee River Basin, and St. Francis River in the White River Basin (Haag 2012, p. 330). The scope of channelization activities over extensive areas alters physical habitat and degrades water quality, which affects the species at the population level.

4.1.3 Sedimentation

Excessive sedimentation is a frequently invoked cause of declines in populations of freshwater mussels (Brim Box and Mossa 1999, p. 99). Researchers have cited the adverse effects of excessive sedimentation from human activities on populations of freshwater mussels since the late 1800s (Kunz 1898, p. 328). However, sediment loads, whether natural or human-induced, are difficult to identify and quantify, so much of the information available to biologists about the effects of excessive sedimentation on freshwater mussels is largely descriptive or anecdotal; strong evidene supporting sedimentation as a direct cause of declines in populations of mussels is lacking (Brim Box and Mossa 1999, pp. 104–106) and biologists have not conducted any studies to examine the effects of sedimentation on the Rabbitsfoot.

Biologists have documented several adverse biological effects of excessive sediment, especially fine sediments, on freshwater mussels. Fine sediments such as silt and clay particles may directly adversely affect freshwater mussels by decreasing feeding and respiratory efficiency from clogged gills, disrupting metabolic processes, decreasing growth and survival rates, limiting burrowing activity, suffocating individuals, and decreasing effectiveness of visual lures used by some brooding females to attract sight-feeding host fish (Ellis 1936, pp. 39–40; Marking and Bills 1979, p. 210; Vannote and Minshall 1982, pp. 4105–4106; Hartfield and Hartfield 1996, p. 373). Fine sediments may indirectly adversely affect freshwater mussels if turbidity levels significantly reduce the amount of light penetrating through the water for photosynthesis, and thus, the production of certain food items (Kanehl and Lyons 1992, p. 7). Excessive sedimentation potentially affects freshwater mussels physically in several ways. They are potentially affected by changes in suspended and bed material load, bed sediment composition associated with increased sediment production and runoff in the watershed, changes in channel form, position, and degree of stability, changes in depth or the width/depth ratio, which affects light penetration and flow regime, actively aggrading (filling) or degrading (scouring) channels, and changes in channel position (Vannote and Minshall 1982, p. 4106; Kanehl and Lyons 1992, pp. 4–5; Brim Box and Mossa 1999, pp. 109–112). These changes may dislodge, transport downstream, or leave mussels stranded.

Results of experimental studies evaluating the responses of freshwater mussels to burial by fine sediments and sand found that covering adult mussels under as little as 14 cm (0.55 in) of sediment significantly increased mortality rates of some of the species, suggesting that the primary effects of excessive sediment on mussels are often sublethal, and detrimental effects may not be immediately apparent (Ellis 1936, 39–40; Vannote and Minshall 1982, pp. 4015–4016; Brim Box 1999, pp. 71–78). However, these experimental studies lacked controls and the rate and depth of accumulation of sediment used in them is not likely to occur in free-flowing streams (Haag 2012, p. 360). More recently, Gascho Landis and Stoeckel (2016, p. 232–234) conducted a laboratory experiment to examine the effects of suspended solids on mussel reproduction in both a short-and long-term brooder (*Reginaia ebenus* and *Ligumia subrostrata*, respectively) and found that excessive inputs of sediment disrupted early reproductive stages of both species and thus, may be an important driver of declines in populations of freshwater mussels. Hansen et al. (2016, entire) developed a dyanamic, process-based interaction model to test the hypothesis that chronic exposure to increased suspended sediment and food limitation are the primary factors controlling native freshwater mussel population density in the Minnesota and St. Croix river basins and found that suspended sediment and food limation were good predictors of long-term changes in freshwater mussel population density in these river basins. Lummer et al. (2018, p. 1344) evaluated effects of three particle size classes and increasing sediment concentration on the behavior of the Painter's mussel (*Unio pictorum*). They found that mussels did not avoid the uptake of suspended sediments by decreasing their Hall activity (clearance of suspended particles out of the water column), neither for different particle size classes nor for increasing concentrations of the smallest particle size class, which is contradictory to the findings by Ellis (1936, 39–40). These results demonstrate the need to conduct more studies that evaluate the effects of particle size classes and increasing sediment concentration on the behavior of freshwater mussels.

Because of the interest in rearing juveniles in captivity for use in conservation, other studies have focused on the ability of fine sediments to block interstitial water flow, which could reduce levels of dissolved oxygen and interfere with feeding of juvenile mussels. Geist and Auerswald (2007, pp. 2308–2309) found that sites with good recruitment of juvenile Freshwater Pearl Mussels had coarser and better sorted substrata with significantly lower quantities of fine sediments whereas sites lacking recruitment of juveniles had a more variable and greater penetration resistance indicating clogging of the interstitial macropore system by the deposition of mud and compaction of the stream bed. Osterling et al. (2010, pp.763–764) examined possible causes of population declines of the Freshwater Pearl Mussel by relating age distribution, density and growth with turbidity, sedimentation, and density of host fish. They found that turbidity and sedimentation were 3-4 times lower in streams with recent recruitment compared with those without recent recruitment suggesting turbidity and sedimentation do affect the juvenile life history stage of freshwater mussels. Conversely, Strayer and Malcom (2010, pp. 1783–1786) sampled recruiting and non-recruiting populations of the Common Pearly Mussel (*Elliptio complanata*) in southeastern New York to test whether status of the populations was associated with interstitial un-ionized ammonia, fine sediments, interstitial dissolved oxygen, disappearance of primary host fish, or crayfish densites and found no association between recruitment failure and fine sediments, interstitial dissolved oxygen, disappearance of primary host fish, or crayfish, but did find a strong relationship between recruitment failure and high concentrations (>0.2 jug N/L) of un-ionized ammonia. Denic et al. (2014, pp. 115–117) investigated the effects of fine sediment and physicochemical variables on recruitment of juveniles in a population of the Thick-shelled River Mussel (*Unio crassus)* in the Sallingback River, northeast of Munich, Germany, and also did not find a relationship between fine sediment deposition and juvenile recruitment. Tuttle-Raycraft et al. (2017, pp. 1164) examined the effect of total suspended solid concentration on the clearance rate, the volume of water a mussel clears of particles within a certain time frame, of newly transformed juvenile and adult Wavy-rayed Lampmussel (*Lampsilis fasciola*), Fatmucket (*Lampsilis siliquoidea*), Eastern Pondmussel (*Ligumia nasuta*), and Rainbow mussel

(*Villosa iris*) and found a decrease in clearance rate as total suspended solid concentration increased at both juvenile and adult stages, but the decrease in feeding was five times greater in juvenile compared to adult bivalves.

4.1.4 Chemical Contaminants

Chemical contaminants are frequently cited as contributing to declines in densities, ranges, and diversity of populations of freshwater mussels (Richter et al. 1997, p. 1081; Strayer et al. 2004, p. 436; Haag and Williams 2014, p. 51). Chemical contaminants enter rivers through point and non-point sources including spills, industrial and municipal effluents, and urban, silvicultural, and agricultural runoff. These sources contribute chemical contaminants in the form of heavy metals, nutrients, pesticides, organic compounds, and pharmaceuticals to the aquatic environment. Although chemical spills and other point sources of contaminants may directly cause localized mortality of freshwater mussels, widespread decreases in density and diversity may result in part from the subtle, pervasive effects of chronic, low-level contamination (Naimo 1995, p. 342).

Cope et al. (2008, entire) evaluated the pathways of exposure to chemical contaminants for all four life history stages of freshwater mussels and found that each one has both common and unique characteristics that contribute to observed differences in exposure and sensitivity. Free glochidia are exposed to waterborne contaminants briefly e.g., a duration of seconds to days through surface water (Cope et al. 2008, p. 454–457; Ingersoll et al. 2007, pp. 101–104). The obligate parasitic stage also might be exposed to waterborne contaminants through surface water while partially encysted or to contaminants in host fish tissue while fully encysted for a duration of weeks to months, which could affect transformation success of glochicia into juveniles (Cope et al. 2008, p. 457; Ingersoll et al. 2007, pp. 101–104). Because juveniles live largely burrowed in the sediment for the first 0 to 4-years of life, sediment, interstitial water, and diet are the primary pathways of exposure for this life history stage, but surface water also might contribute to exposure during certain periods and environmental conditions (Cope et al. 2008, pp. 457–458). The primary pathways of exposure to contaminants for adult freshwater mussels are surface water, sediment, interstitial water, and diet, and because of their longevity the duration of exposure for adults may be years to decades (Cope et al. 2008, pp. 452–454).

Heavy metals including cadmium, chromium, copper, lead, mercury, nickel, and zinc occur in point-source discharges such as industrial and wastewater effluents. Heavy metals can cause mortality and affect biological processes including disrupting enzyme efficiency, altering filtration rates, reducing feeding and growth, and changing behavior of freshwater mussels (Salanki and Balogh 1989, pp. 446–447; Jacobson et al. 1997, p. 2390; Valenti et al. 2005, p. 1244; Wang et al. 2007b, pp. 2052–2055; 2010, p. 2053). The early life history stages of freshwater mussels are among the most sensitive aquatic organisms tested for heavy metals (Keller and Zam 1991, pp. 543–544). Low but chronic heavy metal and other toxicant inputs may reduce mussel recruitment (Yeager et al. 1994, p. 217; Ahlstedt and Tuberville 1997, p. 75). Wang et al. (2010, p. 2062) assessed the acute and chronic toxicity of glochidia and juveniles of the Fatmucket (*Lampsilis siliquoidea*) and Neosho Mucket (*Lampsilis rafinesqueana*) and found that in comparison with other tested freshwater organisms, newly transformed juveniles (5 days old) were relatively sensitive to the acute toxicity of lead, cadmium, or zinc, and 2- to 4-month-old juveniles were relatively sensitive to the chronic toxicity of Pb, but were moderately sensitive to the chronic toxicity of cadmium or zinc. Valenti et al. (2005, pp. 1242–1245) conducted chronic toxicity tests with the glochidia and juvenile life history stages of the Rainbow Mussel (*Villosa iris*) and found that glochidia were more sensitive than juveniles to mercury with a median lethal concentration value of 14 µg/L and 114 µg/L, respectively and that juveniles exposed to mercury ≥8 µg/L exhibited decreased growth.

Runoff from agricultural, silvicultural, and urban land use practices such as animal feedlots, fertilized row crops, construction sites, and landscapes are major non-point sources of phosphorus and nitrogen to aquatic ecosystems (U.S. Environmental Protection Agency [EPA] 2020). Bauer (1988, p. 244; 1992, p. 425) demonstrated that excessive nitrogen concentrations can be lethal to the adult Freshwater Pearl Mussel and can decrease the life span and size of other mussel species. Eutrophication caused by excessive inputs of nitrogen and phosphorus may cause algal blooms in surface waters and contribute to a wide range of water-related problems including summer fish kills(Carpenter et al. 1998, pp. 560–561). Eutrophication may be particularly detrimental to juvenile mussels that inhabit the interstitial spaces in the substrate where lower dissolved oxygen concentrations are more likely than on the sediment surface where adults tend to live (Sparks and Strayer 1998, pp. 132–133).

Pesticides are acutely toxic to freshwater mussels, especially early life history stages, and freshwater mussels are more sensitive to pesticides than other aquatic organisms (Conners and Black 2004, pp. 364–365; Bringolf et al., 2007b, p. 2103–2105). Bringolf et al. (2007a, pp. 2095–2099) tested the toxicity of glyphosate, formulations (i.e., Roundup® and Aqua Star), and the surfactant used in Roundup, (i.e., MON 0818) to the glochidia and juvenile life history stages of the Fatmucket and found that MON 0818 was the most toxic constituent and that glochidia were the most sensitive life stage tested to date. Roundup®, technical grade glyphosate isopropylamine salt, and isopropylamine were also acutely toxic to juveniles and glochidia (Bringolf et al. 2007a, p. 2097). Results of Bringolf et al. (2007b, p. 2103–2105) demonstrated that other pesticides, including atrazine, chlorpyrifos, and permethrin are also toxic to the glochidia and juvenile life history stages of the Fatmucket. Wu et al. (2010, entire) reported atrazine from 75% of stream surface water and 40% of the groundwater in the agricultural region of the United States.

Direct and indirect effects of water quality degradation and chemical contamination as a result of active or past mining activities also may affect populations of freshwater mussels. From 1850–1970, commercial extraction of lead- and zinc-bearing ores occurred in the Tri–State Mining District, which encompasses approximately 6,500 km^2 in southeastern Kansas and adjacent portions of southwestern Missouri, and northeastern Oklahoma (Angelo et al. 2007, p. 469). Angelo et al. (2007, pp. 477–485) surveyed freshwater mussels throughout the Spring River Basin to evaluate the effect of this mining activity on freshwater mussel communities and found a strong negative correlation between the distribution and abundance of native freshwater mussels, including the Rabbitsfoot, and sediment concentrations of lead, zinc, and cadmium. Sediment and water quality samples exceeded the EPA 2006 threshold effect concentrations for cadmium, lead, and zinc at numerous sampling locations within the Tri–State Mining District (Angelo et al. 2007, p. 484). In 2016 and 2017, EcoAnalysts, Inc. (2018, pp. 44–56) quantitatively sampled freshwater mussel community metrics (density, richness, recruitment, mortality, community composition) at 22 sites distributed over a range of sediment metal contaminant levels in the Spring River Basin. They found a negative correlation between the metrics and sediment toxicity, suggesting sediment metal concentrations adversely affected the freshwater mussel community.

The early life history stages of freshwater mussels are among the most sensitive aquatic organisms tested for effects of ammonia (Goudreau et al. 1993, pp. 219–220; Augspurger et al. 2003, p. 2574; Newton et al. 2003, p. 2557). Augspurger et al. (2003, p. 2574) demonstrated that ammonia is lethal to juveniles at concentrations as low as 0.7 parts per million (ppm) TAN, normalized to pH 8 (range = 0.7–19.7 ppm) and lethal to glochidia at concentrations as low as 2.4 ppm TAN, normalized to pH 8 (range = 2.4–10.4 ppm). They considered ammonia a limiting factor for survival and recovery of freshwater mussels because it is toxic at relatively low concentrations, and it is a common contaminant in lotic waters not only through anthropogenic inputs like agricultural runoff and municipal wastewater treatment plants, but also through natural processes like nitrogen fixation, ammonification, and dissimilatory reduction of nitrate. Toxic effects of ammonia are more pronounced at higher pH and water temperatures because the level of the un-ionized form increases as a percentage of total ammonia (Mummert et al. 2003, p. 2545). Documented toxic effects of ammonia include reduced time valves are open for respiration and feeding (Naimo 1995, p. 355), impaired secretion of the byssal thread (Reddy and Menon 1979, pp. 317–320), reduced ciliary action, depleted lipid, glycogen, and other carbohydrate stores (Chetty and Indira 1994, p. 693), altered metabolism (Chetty and Indira 1995, pp. 84–88), significant reduction in growth (Newton et al. 2003, p. 2558), and acute toxicity (Goudreau et al. 1993, pp. 219–220). Because of the results of studies testing the sensitivity of aquatic organisms to ammonia, freshwater mussels are recognized as the most sensitive aquatic organisms ever tested to the effects of ammonia (Wang et al. 2007b, pp. 2039–2045; 2007*c*, pp. 2052–2055). These results prompted the EPA to revise the ammonia water quality criteria to protect freshwater mussels (78 FR 52192, August 22, 2013; p. 2).

Biologists have not conducted acute toxicity tests to evaluate the sensitivity of the Rabbitsfoot to any chemicals. However, Wang et al. (2016, pp. 2–5) conducted acute 96-hour toxicity tests to evaluate the sensitivity of five phylogenetically diverse species of juvenile mussels (multi-species study) to 10 chemicals (alachor, metolachlor, ammonia, potassium, chloride, sulfate, chromium VI, copper, nickel, and zinc). They also compared the sensitivity of the Fatmucket in acute exposures to the 10 chemicals to the sensitivity of the other mussel species (single species study). The results of the multi-species study indicated that freshwater mussels representing different families or tribes had similar sensitivity to most of the tested chemicals regardless of modes of toxic action (Wang et al. 2016, pp. 6). The results of the single species study indicated sensitivity of the Fatmucket was similar to the other freshwater mussel species tested and thus, it can be a good surrogate for protecting other freshwater mussels in acute exposures (Wang et al. 2016, pp. 6–7).

Endocrine disrupters, e.g., contraceptives, antidepressents, and livestock growth hormones, are common in riverine ecosystems and originate mainly from municipal sewage effluents and livestock production operations (Gagné et al. 2001, p. 266; Fent et al. 2006, p. 125). In a nationwide study, Kolpin et al. (2002, pp. 1208–1210) sampled 139 stream sites in 30 states and detected the presence of numerous pharmaceutical drugs, hormones, and other organic waste products downstream from urban development and livestock production operations. In 2004, Galloway et al. (2005, pp. 4–22) surveyed seven streams in northwestern Arkansas and found pharmaceuticals or other organic wastewater constituents at 16 of 17 sites. Endrocrine disrupters in municipal effluents affect mussels in a number of complex ways, including by increasing serotonin and dopamine levels and inducing spawning in gonadal tissue in vitro (Gagné and Blaise 2003, pp. 120–123; Gagné et al. 2004, pp. 36–39). Gagné et al. (2001, pp. 263–267) found that chronic exposure to estrogenic compounds in effluents caused feminization of male freshwater mussels, but these individuals did not produce eggs, suggesting a major disruption of reproductive function. The long-term effects of these compounds on freshwater mussels and other aquatic organisms in general are unknown and probably limited to areas near their source (Gagné et al. 2004, pp. 42–43). Biologists have not conducted studies examining the effects of endocrine disrupters specifically on the Rabbitsfoot, but populations are likely exposed to numerous sources, such as sewage effluents, livestock operations, and other agriculture practices that exist within their range.

4.2 Factor D: Inadequacy of Existing Regulation

4.2.1 Clean Water Act

The objective of the Federal Water Pollution Control Act, commonly referred to as the Clean Water Act (CWA) (33 U.S.C. 1251 *et seq.*), is to restore and maintain the chemical, physical, and biological integrity of the nation's waters by preventing point and non-point sources of contaminants. States are responsible for setting and implementing water quality standards that align with the requirements of the CWA. Non-point source pollution within the watersheds occupied by the Rabbitsfoot include timber clear-cutting, clearing of riparian vegetation, urbanization, road construction, and other practices that allow contaminants to enter streams. For example, streams labeled as impaired in the Duck River watershed (300 RMs) are losing 5–55% more soil per year than streams not labeled as impaired (USACE 2011a). There is no information concerning the implementation of the CWA regarding non-point source pollution specific to protection of the Rabbitsfoot and very little information on other freshwater mussels, which makes it difficult to determine whether the CWA is adequately addressing threats to freshwater mussels.

Even though the glochidia and juvenile life history stages of freshwater mussels are reported as more sensitive to some chemicals when compared to commonly tested aquatic organisms (Keller and Zam 1991, pp. 543–544; Augspurger et al. 2003, p. 2574), the EPA has not routinely used the toxicity data generated from freshwater mussels in the derivation of water quality criteria, mainly due to a lack of a standardized method for conducting toxicity tests with freshwater mussels (Ingersoll et al. 2007, pp. 101–104). In 2006, the American Society for Testing and Materials (ASTM) published standard methods for conducting acute and chronic toxicity tests with glochidia and juvenile life history stages of freshwater mussels (ASTM 2006, pp. 1393–1444). Wang et al. (2007a, entire) demonstrated that these methods can be used to consistently generate toxicity data with acceptable precision and accuracy. Wang et al. (2007b, entire) used the ASTM standard methods to determine acute toxicity of copper, ammonia, and chlorine to glochidia and juvenile freshwater mussels and found that these early life history stages generally were more sensitive to copper and ammonia than other aquatic organisms. These results prompted the EPA to revise the ammonia water quality criteria to protect freshwater mussels (78 FR 52192, August 22, 2013; p. 2). However, not all states within the range of the Rabbitsfoot have adopted the new ammonia criteria. Results of other acute toxicity tests using ASTM standard methods and the juvenile life history stage of a variety of freshwater mussel species demonstrated that the sensitivity of freshwater mussels to ammonia, chlorine, nickel, and zinc is lower than the current EPA water quality criteria for those chemical contaminants excluding potassium because currently there is no water quality criteria for it (March et al. 2007, pp. 2068–2070; Gibson 2015, pp. 47–49 and 72–73; Wang et al. 2016, pp. 5–10). Use of these results by the EPA to establish water quality critieria for these contaminants and adoption of those criteria at the state level would improve protection of freshwater mussels from exposure to these chemical contaminants.

4.3 Factor E: Other Natural or Man-made Factors

4.3.1 Population Fragmentation and Isolation

Anthropogenic changes to the landscape such as the construction of dams and resultant impoundments, have fragmented riverine habitat and isolated populations of freshwater mussels from each other and their host fish. Fragmentation and isolatioin make extirpation of the remaining populations from demographic stochastic events, such as variation in fertilization success among individuals, and from environmental stochastic events, such as droughts that severely decrease abundance more likely (Haag 2012, pp. 336–338). Fragmentation of riverine habitat also may affect the genetic structure of populations. It reduces the natural interchange of genetic material between populations by impeding host fish movement and the genetic diversity within populations by decreasing population size, both of which can lead to inbreeding depression (Avise and Hambrick 1996, p. 461). Inbreeding depression can result in early mortality, decreased fertility, smaller body size, loss of vigor, reduced fitness, and various chromosome abnormalities (Smith 1990, pp. 311–321). Genetic inbreeding and loss of heterozygosity, low fecundity, decreased reproductive success and recruitment are all factors that could result in the Allee effect, a decline in individual fitness at low population size or density, that may result in a minimum population size below which it is not able to recover (Courchamp et el. 1999, p. 409).

We do not have data to examine how population size and isolation affect genetic diversity of the Rabbitsfoot. However, given the reduction in range and number of areas represented by a few individuals (Service 2020), it is likely that some populations of the Rabbitsfoot are below effective population size (number of breeding individuals who add offspring to the next generation). Achieving effective population size is necessary for a population to adapt to environmental change and maintain long-term viability (Soulé 1980, pp. 162–164). Biologists reported evidence of recruitment for 13 of the 63 (21%) rivers or creeks where the Rabbitsfoot currently occurs, indicating likely recruitment reduction or failure (Service 2020). Even in the absence of existing or new anthropogenic threats, low effective population size may reduce population viability and presents conservation challenges.

4.3.2 Invasive Nonnative Species

Since its introduction in the mid-1980s through release of ballast water into the Great Lakes, colonization of the Zebra Mussel (*Dreissena polymorpha)* has resulted in the decline and regional extirpation of some populations of freshwater mussels in North America (Schloesser et al. 1996, p. 303). The best known mechanism by which the Zebra Mussel affects native freshwater mussels is direct fouling, where it attaches to the shells of native mussels (reviewed in Strayer 1999b, pp. 77–80). Heavy infestions of the Zebra Mussel on the shell of a native host mussel may injure it by impeding both lateral and vertical locomotion, interfering with normal valve movements, deforming valve margins, and locally depleteing food resources and increasing waste products. Heavy infestations of the Zebra Mussel also may stress native host mussels by reducing energy stores. The Zebra Mussel can adversely affect native mussels by mechanisms other than fouling, for example by reducing food concentrations to levels too low to support reproduction and survival. Large deposits of pseudofeces of the Zebra Mussel may degrade habitat for native mussels (Vaughn 1997, p. 11).

In the mid-1990s, expansion of the Zebra Mussel in North America slowed, and for the most part, it was not successful in colonizing rivers in the southern United States (Haag 2019, p. 52). Strayer et al. (2011, entire) conducted a 20-year study of the Hudson River population of the Zebra Mussel and its effects on native freshwater mussels. They found that annual survivorship of the Zebra Mussel decreased >100-fold during this time. After approximately eight years, populations of native freshwater mussels approached pre-invasion densities indicating that some native freshwater mussels are able to recover after an intial period of severe impact. The Zebra Mussel co-occurs with the Rabbitsfoot in some rivers including the Ohio, Tennessee, White, Allegheny, lower Verdigris, and Neosho rivers as well as French and Bear creeks. Populations of the Zebra Mussel occur primarily in streams with barge navigation (Stoeckel et al. 2003, p. 334). Biologists have reported that native freshwater mussel populations are able to survive when abundance of the Zebra Mussel is low (Butler 2005, B. E. Fisher, Indiana Department of Natural Resources [IN DNR], pers. comm., 2009), which tends to be the case for rivers with no barge traffic and warmer water temperatures.

Since its introduction in the early twentieth century, biologists have documented occurrence of Asian Clams (*Corbicula* spp.) throughout the range of the Rabbitsfoot (Benson and Williams 2021, pp. 17–39). While biologists have not conducted studies to evaluate the effects of Asian Clams on populations of the Rabbitsfoot specifically, they have proposed several mechanisms by which Asian Clams can adversely affect native freshwater mussels (reviewed in Strayer 1999b, p. 82). Dense populations of Asian Clams may compete with native mussels, particularly juveniles, for resources such as food, nutrients, and space, may ingest sperm, glochidia, and newly metamorphosed juveniles, may reduce habitat for juvenile mussels while actively disturbing sediments. Periodic die–offs of Asian Clams may produce enough ammonia and consume enough dissolved oxygen to kill native mussels (Strayer 1999b p. 82; Cherry et al. 2005, p. 377). Yeager et al. (2000, pp. 254–256) examined the interactions between newly metamorphosed juveniles of the Rainbow Mussel and high densities of adult Asian Clams. They found that consumption of glochidia and displacement of juveniles by adult Asian Clams adversely affected the survival and growth of the Rainbow Mussel, which could limit recruitment. Ferreira-Rodríguez et al. (2018, p. 89) conducted field and laboratory experiments to assess the effects of the Asian Clam (*Corbicula fluminea,* Müller 1774) on the growth, physiological condition, and locomotor activity of *Unio delphinus*, a native freshwater mussel species from Europe. They found *Unio delphinus* exhibited lower growth and physiological condition and greater locomotor activity at greater densities of *Corbicula fluminea*. Although they were not able to establish the main mechanism(s) responsible for these results, they hypothesized competition for food resources, and/or abiotic changes in microhabitat features as a result of bioturbation activities and production of feces and pseudofeces by *Corbicula fluminea*. Haag et al. (2021, pp. 451–454) examined relationships between survival and growth of captively-reared juveniles of the Cumberland Bean (*Venustaconcha troostensis)*, Painted Creekshell (*Villosa taeniata)*, Plain Pocketbook (*Lampsilis cardium)*, and Fluted Kidneyshell (*Ptychobranchus subtentus)* and water temperature, water chemistry, and abundance of *Corbicula fluminea* at 17 sites in the Rockcastle River system, Kentucky. For all four mussel species, they found a positive relationship between water temperature and mussel growth and a negative relationship between mussel growth and abundance of *Corbicula fluminea*. These results are similar to the laboratory results of Yeager et al (2000, pp. 254–256) in that they support a measurable, negative relationship between mussel growth and a relatively low abundance of *Corbicula fluminea*, which suggests that it may be a major factor in widespread mussel declines.

The Black Carp (*Mylopharyngodon piceus*) is a large, fish species native to China. Biologists have documented its occurrence in Arkansas, Illinois, Indiana, Mississippi, and Missouri (Nico et al. 2005, p. 155; B. E. Fisher, IN DNR, pers. comm., 2021), Louisiana, Tennessee, and Kentucky (Nico and Neilson 2018, U.S. Geological Survey Nonindigenous Aquatic Species Database). Nico et al. (2005, entire) studied the diet of the Black Carp within its native range in east Asia. They concluded that because of its large size (>1.0-m total length and >50 kg) and long life span (>15 years) the Black Carp has the potential to cause significant harm to native freshwater mussels as a predator of multiple age classes (Nico et al. 2005, p. 77). Poulton et al. (2019, p. 182) inventoried diet items consumed by wild-caught Black Carp obtained from the Mississippi River drainage. They identified 59 animal taxa, including 21 mollusks, 27 insects, 11 other noninsect invertebrates, and various plant material including nuts and seeds. Because of the breadth of taxa consumed by the wild-caught Black Carp, ingestion of high abundances of a single diet taxon, and multiple species of unionid mussels that inhabit both lentic and lotic habitats, they concluded that their results confirmed that the invasion of the Black Carp in the U. S. poses a risk to native aquatic fauna.

The Round Goby (*Neogobius melanostomus*) is another invasive, nonnative fish species released in the 1980s that is well established and likely to spread through the Mississippi River system (Strayer 1999b). This species is an aggressive competitor of similar-sized benthic fishes (sculpins and darters), as well as a voracious carnivore, despite its size (<25.4 cm in length), preying on a variety of foods, including small mussels and fishes that could serve as glochidial hosts for freshwater mussel species (Strayer 1999b, Janssen and Jude 2001). In 2014, biologists with Pennsylvania Fish and Boat Commission discovered the Round Goby in the French Creek Watershed (Lake LeBoeuf). From 2016 to 2018, Stauffer and Wilson (2018, p. 5) documented them in the main stem of French Creek and expansion of their range downstream every year since 2014. They also documented consumption of glochidia of unionid mussels from multiple size classes of the Round Goby and a shift in habitat occupied by host fish species in response to the presence of the Round Goby (Stauffer and Wilson, 2018, pp. 6–10).

4.3.3 Climate Change

Climate change has the potential to increase the vulnerability of freshwater mussels to random catastrophic events (McLaughlin et al. 2002, entire). We used NatureServe Climate Change Vulnerability Index 3.02 (CCVI; Young et al. 2011, entire) to conduct a coarse-scale assessment of the relative vulnerability of the Rabbitsfoot to climate change and Climate Wizard (Girvetz et al. 2009, entire) to examine the magnitude of predicted temperature and moisture change for each of the 15 states within the historical range. For all 15 states, air temperature is predicted to increase over the next 30 years and in nine of the states by as much as 5.6–6.0° F (approximately 15° C), which corresponds with predictions from the U.S. Global Climate Change Research Program (2017, p. 207). Although it is not a 1:1 ratio, warmer air temperatures increase water temperatures and reduce dissolved oxygen, which is likely to adversely affect both the Rabbitsfoot and host fish species. Annual moisture is predicted to decrease over the next 30 years by as much as -0.097 – -0.119% in three of the states and as little as -0.028 – -0.050% in others. Altered stream flows resulting from more severe drought events decrease water flow and dissolved oxygen levels and increase water temperature in stream and rivers, which are likely to adversely affect both the Rabbitsfoot and host fish species.

We used the best available scientific and commercial data about the ecology and life history of the Rabbitsfoot to score its indirect exposure and species-specific sensitivity as extremely vulnerable; abundance and/or range extent within geographical area assessed is extremely likely to substantially decrease or disappear by 2050. The indirect exposure and sensitivity factors that contributed to this score include the following: (1) presence of anthropogenic barriers like impoundments that impede movement of host fish among suitable habitats, limiting dispersal and gene flow among populations of freshwater mussels; (2) predicted effects of land use changes resulting from human responses to climate change e.g., reduction in stream flows caused by diversion of water resources for urban and agricultural use in response to drought conditions; and (3) dependence on host fish for propagule dispersal. Global status of the Rabbitsfoot is ranked as vulnerable, and state status varies among vulnerable, critically imperiled, or presumed extirpated. Global and state status do not account for vulnerability to climate change. Thus, these results indicate that climate change itself not only is likely to further declines in populations, but also to exacerbate the effects of anthropogenic changes that natural resource professionals already identified as primary threats jeopardizing persistence of populations. For example, predicted decreases in precipitation and increases in air temperature are likely to further reduce stream flow beyond the affect impoundments have had on the viability of the Rabbitsfoot and thus, escalate declines in populations although the locations and magnitude of effects of changing climate conditions are difficult to predict.

4.4 Conservation Actions

The most fundamental goal of freshwater mussel recovery and conservation is to increase the amount of occupied habitat and connectivity so species can sustain localized catastrophic events and adapt to more subtle but long-term environmental changes. As such, conservation actions to protect, improve, and restore habitat and to reintroduce and augment populations are integral to recovery and conservation of the Rabbitsfoot. Removal of navigation dams essential to commerce (e.g., Ohio and Tennessee rivers) is not practical, but conservation measures to avoid, minimize, and mitigate the loss and degradation of habitat resulting from effects associated with their operation can be implemented to provide more ecologically suitable conditions for imperiled aquatic species downstream of them. In contrast, non-functional, aging and unsound low-head dams may be candidates for removal and subsequent restoration of habitat. Similarly, river reaches where populations are declining but otherwise have intact habitat should be prioritized for improvement, especially if they have the potential to increase connectivity among populations in unaffected tributaries. Finally, because of the level of geographic isolation and number of remaining populations represented by a few individuals, reintroduction and augmentation through captive propagation are necessary actions to achieve the recovery goal of delisting.

4.4.1 Habitat Protection, Improvement, and Restoration

Numerous parcels of public and private lands occur along historical and extant rivers or creeks with records of occurrence for the Rabbitsfoot. Biologists have reported records of occurrence from the Erie, Green River, Little River, Pond Creek, and Tennessee National Wildlife refuges (NWR) or from the watersheds within which those refuges are located. The Peoria Tribe of Indians of Oklahoma manages resources in approximately a 12- and 5-mile reach of the Neosho and Spring rivers, respectively, in Ottawa County, Oklahoma. Policy makers designated the Allegheny and Buffalo (Arkansas) rivers and Big and Little Darby creeks as National Wild and Scenic rivers and Duck and Illinois rivers as state scenic rivers. In Indiana, approximately 16 miles of the Tippecanoe River and >50 miles of Sugar Creek, a tributary of the Wabash River, qualify for designation as state natural and scenic rivers under Indiana Code IC-14-29-6 (https://www.in.gov/dnr/outdoor/5355.htm). The location of Mammoth Cave National Park, which is also a World Heritage Site and Biosphere Reserve, in the upper Green River provides a significant level of localized watershed protection for the Rabbitsfoot in that system. The Nature Conservancy (TNC) partnered with governmental and non-governmental agencies to establish bioreserves along the Green, Tippecanoe, Paint Rock, Duck, and Strawberry rivers and Fish, French, Big Darby, and Little Darby creeks, which currently all have extant watersheds in which the Rabbitsfoot occurs. Although TNC has few riparian inholdings in these watersheds, they developed and implemented innovative community-based projects that address the conservation of aquatic species and habitat restoration on multiple scales e.g., development of a forest bank, which is a market-based approach to sustainable forest management for the Green and Clinch rivers and Big Brushy Creek watersheds (Master et al. 1998, pp. 28–29).

The Service’s Partners for Fish and Wildlife program has funded millions of dollars in projects to private landowners to enhance riparian habitat in streams with populations of the Rabbitsfoot. For example, watershed level projects funded by Partners for Fish and Wildlife in the Green and Duck rivers enhanced and protected habitat for the Rabbitsfoot. Other funding sources for habitat restoration and conservation include CWA Section 319, U.S. Department of Agriculture, Natural Resource Conservation Service(NRCS) programs (e.g., Environmental Quality Incentives Program, Wildlife Habitat Improvement Program, Conservation Reserve Enhancement Program [CREP]), Landowner Incentive Program, Private Stewardship Grant, National Fish and Wildlife

Foundation, and numerous other Federal and State programs are potential sources of funding for various projects that benefit freshwater mussels. For example, Kentucky secured a CREP grant to remove ≤100,000 acres of riparian lands from agricultural production in the upper Green River watershed. Efforts will focus on areas that should be of direct benefit to populations of freshwater mussels in the Green River. Rivers designated as TNC bioreserves, such as the Strawberry River, which is on the CWA Section 303(d) list for sedimentation impairment (TNC 2004), secured Section 319 grants for sediment remediation activities. Various non-governmental organizations work with riparian landowners to help them restore and protect streambanks and riparian zones along several rivers and creeks where the Rabbitsfoot occurs, and these organizations partner with various other stakeholders in conserving aquatic resources.

Since 2000, the Service's State Wildlife Grants program has provided federal funds to develop and implement programs that benefit wildlife and their habitats, especially non-game species. Priority is placed on projects that benefit species of greatest conservation need, and many states identified mollusks as a focal group. These funds must be used to address conservation needs identified within a state's Wildlife Action Plan such as research, surveys, species and habitat management, and monitoring. In turn, the states administer funds to undertake work on their own or support cooperator projects. The Service also provides grants to states and territories for species and habitat conservation on private lands. This program is known as Section 6 of the Act and offers four categories of grants to support species conservation actions: (1) Conservation Grants; (2) Habitat Conservation Plan Land Acquisition; (3) Habitat Conservation Planning Assistance; and (4) Recovery Land Acquisition Grants. The state of Pennsylvania and USACE are cooperating to develop flow management plans for dams that occur on the Shenango and Allegheny rivers to enhance habitat conditions for the plants and animals that depend on downstream river flows (Pennsylvania State Wildlife Action Plan 2015–2025 Appendix 1.4F-Mussels p. 36).

In 1997, the Kansas legislature amended K.S.A 32-962 to create conservation and recovery plan agreements with landowners. Recovery plan agreements must meet the following criteria: (1) participant must carry out management activities specified in the recovery plan for the Rabbitsfoot and three other state-listed mussel species (Obermeyer 2000); (2) property must pass critical habitat designation guidelines for the targeted threatened and endangered species; (3) duration of agreement shall be five years; and (4) Kansas Department of Wildlife, Parks, and Tourism and other essential personnel will have access privileges to the property for the duration of the agreement for monitoring purposes. A landowner who meets the recovery criteria is eligible for a habitat management income tax credit equal to the amount of property taxes paid on enrolled property during each year of the agreement. A landowner also may be eligible for state income tax credit equal to the cost incurred for compliance with the recovery plan. This cost may include expenses from maintaining easement roads, planting and maintaining riparian habitat, building fences for excluding livestock from accessing streams, and constructing alternative watering sources for livestock.

Service biologists have initiated Safe Harbor Agreements (SHA) for listed species and Candidate Conservation Agreements with Assurances (CCAA) for candidate species with private landowners to conserve populations of aquatic organisms. The Rabbitsfoot will benefit from conservation measures associated with these agreements when populations occur with other targeted listed and candidate species. For example, the Service, TNC, Arkansas Game and Fish Commission, and NRCS entered into SHA and CCAA agreements to benefit aquatic species in the Upper Little Red River watershed, Arkansas (Service 2015). The partners will utilize the agreements to provide technical and financial assistance to interested non-federal landowners who are willing to conduct voluntary management activities focused on improving water quality and habitat on their properties. Three federally listed aquatic species, including the Rabbitsfoot, occur in this watershed.

The U.S. Fish and Wildlife Service's Alabama Ecological Services Field Office developed a conservation model, and it is being implemented in a major restoration project that involves TNC, Natural Resource Conservation Service, Geological Survey of Alabama, and Alabama Department of Conservation and Natural Resources. The cooperators designated the Paint Rock River, Big Canoe Creek, and Locust Fork watersheds as Strategic Habitat Units based on water quality data, habitat conditions, and biological assessments from the Alabama Rivers and Streams Network. Over the next five years, TNC will complete several streambank restoration projects in these watersheds to improve water quality and quantity, preserve biotic integrity, and promote restoration efforts for Alabama's critical waterways. Streambank restoration and planting riparian buffers will help to reconnect the rivers to floodplains, slow surface water runoff and pollution, improve aesthetic property value, and reduce further erosion (alh2o.org).

In 1988, the TVA initiated the Reservoir Release Improvement (RRI) Program in the Tennessee River Basin, which resulted in the installation of equipment to increase dissolved oxygen concentrations below 16 dams and initiation of operational changes and installation of equipment to ensure minimum water flows through its dams (tva.com). Results from implementation of the RRI program include improved dissolved oxygen concentrations, stable water temperatures, decreased bank erosion, and stabilization of habitat in rivers within the Tennessee River Basin including at Normandy Dam in the Duck, Holston and French Broad rivers (Scott et al. 1996, p. 5; Ahlstedt and Johnson 2005, pp. 3–4). In 2008, the TVA altered tailwater releases from Tims Ford Dam in the Elk River to reduce severity of flow fluctuations and increase water temperatures to provide more ecologically suitable conditions for imperiled aquatic species downstream of the dam (tva.com). Biologists have observed evidence of recruitment for the Rabbitsfoot and determined the status of this population as stable (T. Amacker, TVA, pers. comm., 2020).

The Sustainable Rivers Project (SRP) is a national partnership between the USACE and TNC. It focuses on modifying operations at USACE dams to enhance habitat conditions for the plants and animals that depend on downstream river flows. The SRP formally began in 2002 when USACE implemented an interim plan designed with TNC to create more natural regimes of flow and stream temperature by modifying operations at the Green River Dam, Kentucky (Hickey and Warner 2006, p. 11). In 2006, the USACE approved the plan and officially integrated it into the water management policies of the Louisville District. Biologists conducted studies of invertebrate diversity and production and fish diversity along the Green River Bioreserve from the dam to the confluence with the Nolin River in Mammoth Cave National Park (reviewed in Konrad et al. 2012, pp. 782–784). Moles and Layzer (2008, pp. 215–217) examined gravidity, fecundity, and fertilization success of the Mucket (*Actinonaias ligamentina)* at four sites along a 63-mi reach of the Green River immediately below the dam and found rates of these demographic parameters increased downstream with distance from the dam and generally were less than those in other rivers. The USACE and TNC are also working on the White River system in Arkansas and Missouri to balance the need to provide hydroelectric power and a water supply for the public with the need to improve habitat conditions for the plants and animals that depend on downstream river flows (USACE 2011b).

Some states have removed non-functional, aging, and unsound low-head dams to improve hydrological conditions for aquatic species downstream and to restore upstream fish passage. In 2012, partners from the National Fish Habitat Program, Ohio River Basin Fish Habitat Partnership, and Manchester University, removed two low-head dams located on the Eel River in Wabash County, Indiana. The removal of these dams resulted in the reconnection of over 190 stream miles (fws.gov/fieldnotes/regmap.cfm?arskey=35135). Since 2017, partners removed two more low-head dams located on the Eel River in Miami an Whitley counties, Indiana, and are working to remove another located in Logansport, Cass County, Indiana (agrinews-pubs.com/2020/12/05/ecologist-leads-projects-to-protect-preserve-the-eel-river/ahfzw8y/). In 2014, the Tennessee Wildlife Resources Agency (TWRA), along with the help of partners removed Browns Mill Dam, a low-head dam located near Murfreesboro, Rutherford County, from the East Fork Stones River (D. W. Hubbs, TWRA, pers. comm. 2019). In 2017, governmental and non-governmental partners collaborated to remove Green River Lock and Dam 6 in the Ohio River Basin, Kentucky (fws.gov). Lock and Dam 6 was the first of four dams the partners scheduled for removal along the Green River. Lock and Dam 5, about 14 miles downstream from Lock and Dam 6, is scheduled for removal in 2021. Once all four dams are removed, free-flowing hydrological conditions will extend the length of the Green River to its confluence with the Ohio River where the Green River NWR, established in 2019, is located. The Illinois Department of Natural Resources removed the Danville Dam from the mainstem of the Vermilion River and Ellsworth Dam from the North Fork Vermilion River, both located in Danville, Vermilion County, Illinois, in 2018 and 2019, respectively (Tiemann et al. 2018, entire). The Ohio Department of Natural Resources (ODNR) began removal of the Six Mile Dam from the Walhonding River, in October, 2020. They will complete removal of the dam in spring, 2021, at which time the Walhonding River will flow freely from Mohawk Dam to its confluence with the Tuscarawas River for the first time in 180 years (Ashland Times-Gazette, October 9, 2020). Once the natural flow of the river is re-established it will restore upstream fish passage, redistribute coarse sediment supply to starved reaches downstream of the dam, and address impairments that constrain aquatic communities within the dam pool (Environmental Solutions and Innovations, Inc., 2019). Biologists with ODNR, the Service, and volunteers salvaged >600 Rabbitsfoot mussels, some of them juveniles, during removal of the dam and reallocated them upstream (A. Boyer, Service, pers. comm. 2020).

4.4.2 Reintroduction and Augmentation through Captive Propagation

The Cumberlandian Region Mollusk Restoration Committee (CRMRC) consists of biologists from state, federal, and non-governmental natural resource organizations. In 2010, the committee finalized a plan for the population restoration and conservation of imperiled freshwater mollusks of the Cumberlandian Region (CRMRC 2010). It identifies extant populations, potential augmentation and relocation localities, and recommended priority actions for 57 freshwater mussels and 25 snails. The committee categorized the Rabbitsfoot as a Tier 3 species: taxa that have experienced a significant decline in range and abundance or are extirpated from the region (CRMRC 2010, p. 85). The committee identified several locations in the region as potential opportunities for reintroduction of the Rabbitsfoot (CRMRC 2010, p. 85). However, the committee categorized the reintroduction potential for the Rabbitsfoot as low and propagation difficulty as high (CRMRC 2010, p. 85). The committee did not identify any rivers or creeks as opportunities for augmentation. They determined the Paint Rock and Duck river populations represented opportunities to translocate adults for reintroduction and augmentation activities (CRMRC 2010, p. 85).

In 2012, Dr. Paul Johnson and his staff at the Alabama Aquatic Biodiversity Center reintroduced 12 mussel and three snail species back into areas where they no longer occurred (alh2o.org). This effort included reintroduction of the Rabbitsfoot through propagation in Limestone Creek in the Tennessee River Basin. Staff at the Center for Mollusk Conservation, Frankfort, Kentucky, are attempting to propagate the Rabbitsfoot (M. McGregor, Kentucky Department of Fish and Wildlife Resources, 2020, pers. comm.). Rearing them has proven difficult. If successful, staff and partners anticipate reintroducing the Rabbitsfoot in the Licking River.

Natural resource professionals from Missouri State University (MSU), Kansas Department of Wildlife, Parks, and Tourism, U.S. Fish and Wildlife Service, Oklahoma Department of Wildlife Conservation, The Environmental Program of the Peoria Tribe of Indians of Oklahoma, and the Kansas City Zoo are working cooperatively to augment and restore Kansas populations of the Rabbitsfoot through captive propagation (Barnhart 2017). Using information about historical occurrence of the Rabbitsfoot, occurrence of host fish species, and habitat restoration efforts, they identified suitable sites for reintroduction in the upper Verdigris and Neosho river systems, particularly the Cottonwood River in Kansas. They obtained glochidia from gravid females collected from the lower Verdigris River, below Oologah Reservoir in Oklahoma, as the source population for the upper Verdigris River restoration site and from a site along the lower Spring River upstream of the YY Bridge crossing in Cherokee County, Kansas as the source population for the upper Neosho River system. The Kansas City Zoo and MSU cultured juveniles until they were large enough to tag by laser engraving (>2 cm shell length), approximately 2.5 years. They also engraved brooding females for identification and returned them to the collection site within one month of collection.

Because of high water, they were unable to access brooding females from the lower Verdigris River in 2014 and 2015. In 2016 and 2017, natural resource professionals collected brooding females from the lower Verdigris River and propagated juveniles on host fish and in-vitro. However, as a result of equipment failures, only a very small number of these juveniles survived. In 2018, natural resource professional collected 13 brooding females from the lower Verdigris River and are culturing juveniles on host fish at MSU and the Peoria Tribe's Aquatic Facility. In 2014, they stocked 191 juveniles propagated from glochida collected from the Spring River site to a site on the Cottonwood River. They recovered a recently dead shell from this site in 2015, indicating good growth after stocking. In 2015, they stocked 24 adults tagged with passive integrated transponders in the Spring River, Cherokee County, the same site from which they originated as glochidia, as part of a study to monitor movements. On successive visits over the winter, the mussels had moved very little and had repositioned their orientation in the substrate. The biologists returned in May, 2016, and relocated 18 of the 24 mussels. After more flooding throughout the summer, they visited the site in late July, 2016, relocated five of the 24 mussels and presumed the rest of the mussels washed downstream. The biologists suspended subsequent efforts to relocate these mussels because of habitat instability and plan to monitor reintroduction sites every two years.

The Service developed develop a decision-support tool, specifically a Bayesian Network (BN), to identify and prioritize conservation propagation for threatened, endangered, and declining native aquatic species for the National Fish Hatchery system in the Mountain-Prairie Region (Service 2021, entire). The goal of this project is to align regionalproduction hatcheries in the Mountain-Prairie Region with national priorities establishing conservation propagation facilities that assist with landscape conservation and state partnership needs. The Rabbitsfoot is on the list of priority species.

CHAPTER 5 – CURRENT CONDITION

Using the SSA framework, we describe the subspecies' current condition in terms of resiliency, representation, and redundancy. After delineating analysis and representation units, we used demographic and distributional criteria to assign each extant watershed an ordinal classification of resilience of low, medium, or high condition. For quantitative interpretation, we defined each qualitative condition as a probability of persistence over a 30-year time period. We summarized resilience by representation units to describe representation and redundancy for the Rabbitsfoot range-wide. We used a Bayesian approach to ordinal regression to model current condition of each watershed as a function of threats to mussel persistence.

5.1 Methods

5.1.1 Summarizing Data

We compiled known localities of the Rabbitsfoot (n = 7,453) within a Geographic Information System to facilitate spatial analyses. Data sources for known localities included results from quantitative and qualitative surveys, peer-reviewed literature, agency reports, personal communications, and museum records (Service 2020, pp. 8–47). For each locality, we included as much of the following information as available: (1) year of survey; (2) number of individuals; (3) evidence of recruitment; and (4) status of the population at the 2020 5-year status review. We used information reported by biologists from the last 30 years (1990 to present), which is approximately three generations, to represent current condition of the Rabbitsfoot.

We considered a watershed extant if biologists reported a live individual or fresh dead specimen since 1990. If biologists last surveyed a watershed before 1990, then we assigned its condition as unknown. We considered the Rabbitsfoot extirpated from a watershed if biologists conducted a survey since 1990 but did not locate live individuals or fresh dead specimens or if suitable habitat was no longer available (e.g., impounded stream reaches). Confirming extirpation is difficult, and this category also may include watersheds where the Rabbitsfoot exists at such low densities that biologists fail to detect individuals. (i.e., likely functionally extirpated) We did not specifically define a minimum level of effort because not all surveys consistently reported effort in terms of person-hours or other measures. However, surveys generally spanned multiple sites covering the length of the mainstem within a particular watershed.

5.1.2 Delineating Analysis Units

Few data on the genetic structure of populations of the Rabbitsfoot are available to inform the spatial scale of analysis (Serb et al. 2003, entire; Sproules et al. 2006; Fobian 2007, p. 47), so we selected the 10-digit hydrologic unit code (HUC 10) watershed as the analysis unit. We identified the HUC 10 watershed (hereafter, watershed) as the most appropriate unit because it reflects relevant differences in hydrologic conditions e.g., separation by major impoundments and represents the greatest spatial resolution supported by available data. Our approach does not define populations and therefore differs from how biologists previously delineated analysis units for the Rabbitsfoot. Butler (2005 pp. 20–90) and the Service (77 FR 63446–63448; 2020 pp. 8–47) aggregated records of occurrence and demographic data for the entire length of a river or creek rather than a watershed and referenced the result as a population. For comparison with the information provided in these documents, we summarized watersheds by river or creek in Appendix A and in the results and discussion sections of chapters 5 and 6.

5.1.3 Delineating Representation Units

The historical range of the Rabbitsfoot includes the lower Great Lakes and Mississippi river sub-basins and Ohio, Cumberland, Tennessee, White, Arkansas, and Red river systems located across 15 states (Butler 2005 pp. 15–18; 77 FR 63446–63448; Service, 2020 pp. 43–45; Figure 5.1). Biologists consider the Rabbitsfoot extirpated in West Virginia and Georgia and 86 of the 149 (57.7%) rivers or creeks across the range. Because genetic structure of populations of the Rabbitsfoot is lacking, we used the two sub-basins and six river systems as the foundation for grouping watersheds that occupy geographically and ecologically comparable areas into representation units. However, the Ohio River system is much larger in size and spans a wider range of biotic and abiotic conditions relative to the others, so we used EPA level III ecoregions to divide it into two units for a total of nine representation units (Figure 5.1). The upper and lower Ohio River representation units are approximately equal in size and denote similar patterns and composition in geology, landforms, soils, vegetation, climate, land use, wildlife, and hydrology, all of which affect differences in ecosystem quality and integrity (Omernik and Griffith 2014, entire). The upper Ohio River representation unit includes the Central and North Central Appalachains and Western Allegheny Plateau ecoregions. The lower Ohio River representation unit includes the Interior River Valleys and Hills, Interior Plateau, and Central and Eastern Cornbelt Plains ecoregions.

5.1.4 Assessing Current Condition

We assigned an ordinal classification of resilience of low, medium, or high condition within watersheds based on demographic and distributional criteria (Table 5.1). Data are not available with which to estimate demographic criteria directly, so we used number of individuals and evidence of reproduction or status of a river or creek at the 2020 5-year status review (Service 2020, pp. 8–47) as indirect measures of abundance and reproduction, respectively (Table 5.1). We used evidence of reproduction provided in the data sources. As our distributional criterion, we used stream km occupied within a watershed, calculated as the total length of stream reaches between all mussel localities along U.S. National Hydrography Dataset Plus flowlines using the packages sf and stplanr in R version 3.6.1 (Lovelace and Ellison 2018, entire; Pebesma 2018, entire; R Core Team 2019, entire). We developed these criteria *a posteriori* to provide transparent and reproducible thresholds that reflect available data and expert judgement on subspecies' current condition within watersheds. These criteria do not reflect population estimates and are dependent on the subspecies and spatial resolution of the present analysis.

We estimated range reduction by subtracting the area of watersheds where the Rabbitsfoot is considered extirpated from the total area of watersheds historically occupied. We represented uncertainty in range reduction by assuming watersheds with unknown condition are currently extant (lower value) and currently extirpated (upper value). We calculated area from data referenced to the Albers Equal Area North American Datum 1983 Coordinate Reference System (EPSG = 42303) using the sf package in R.

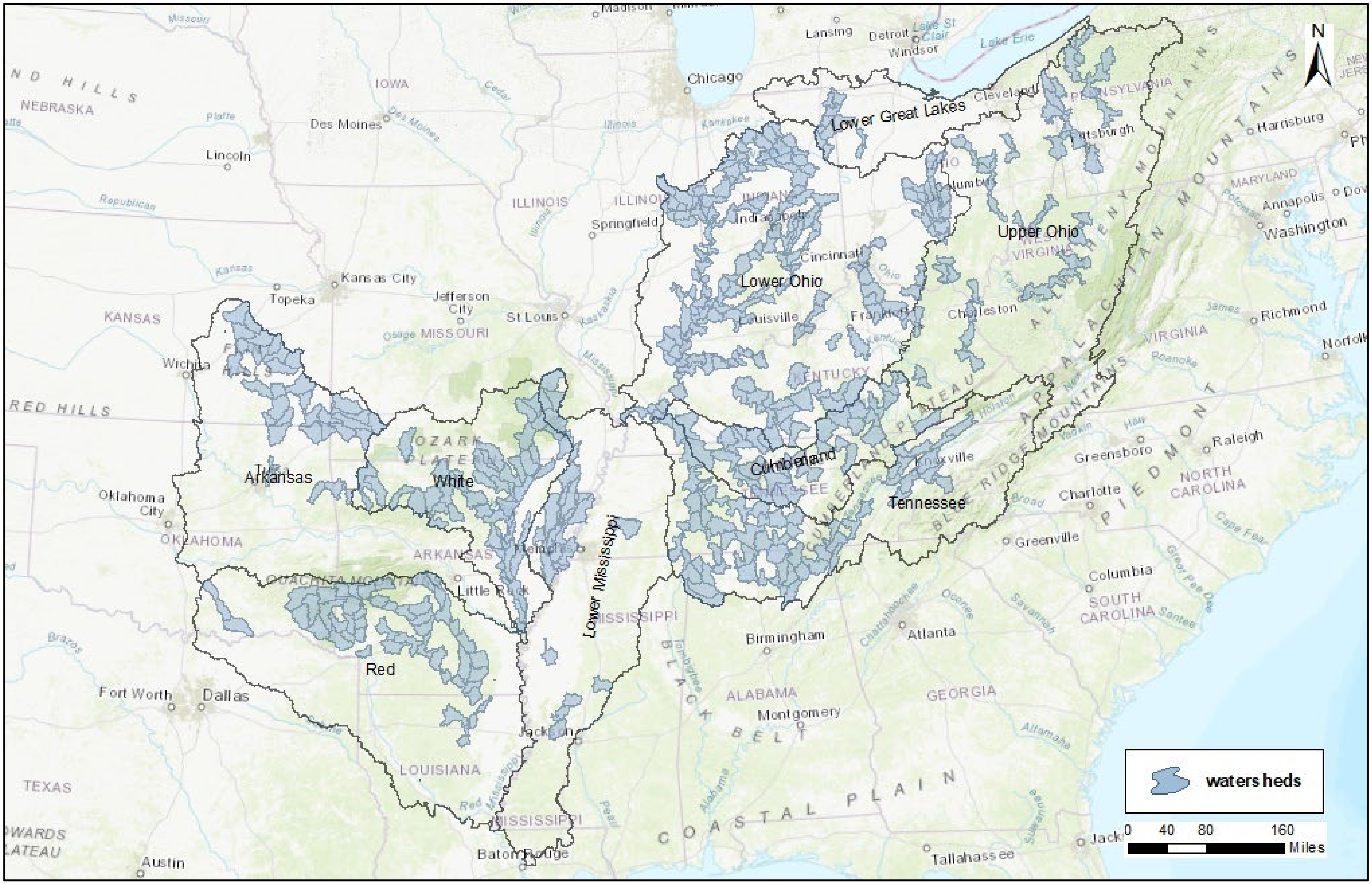

FIGURE 5.1. Historical range (HUC 10 watersheds) of the Rabbitsfoot distributed across nine representation units.

We ranked both criteria from 1 (low) to 3 (high) and used a 2 demographic:1 distributional weighting to reflect the importance of demography on resilience. We summed the results into a final score and used it to assign each extant watershed an ordinal classification of resilience of low (3–5), medium (6–7), or high (8–9) condition (Appendix A). Ordinal classification of resilience is qualitative, so we provided a probability of persistence (>0.75, 0.25–0.75, <0.25) for each classification as a quantitative interpretation of it to facilitate ecologically meaningful and interpretable classification e.g., a watershed classified as high condition has a <25% chance of being extirpated in the next 30 years or >75% chance of persisting in the next 30 years (Table 5.1). We estimated representation and redundancy by tallying the number of watersheds classified in each condition across the nine representation units.

TABLE 5.1. Demographic and distributional criteria used to assign resilience within watersheds and to assess their current condition.

Condition	Abundance	Reproduction	Distributional Criterion	Probability of Persistence[†]
High	100s of individuals	Evidence of reproduction or stable/increasing at 5-yr review	Occurs in more than 50 river km	> 0.75
	10s of individuals	Evidence of reproduction or increasing status at 5-yr review		
Medium	100s of individuals	Decreasing trend at 5-yr review	Occurs in 10–50 river km	0.25–0.75
	10s of individuals	Unknown or stable at 5-yr review		
	Fewer than 10 individuals	Evidence of reproduction or increasing status at 5-yr review		
Low	10s of individuals	Decreasing trend at 5-yr review	Occurs in fewer than 10 river km	< 0.25
	Fewer than 10 individuals	Unknown or stable at 5-yr review		
	Presence-absence data	Unknown or stable at 5-yr review		
Unknown		Historical records of occurrence in watershed with no surveys in past 30 years	Not Applicable	Not Applicable
Extirpated		No individuals collected in surveys within the past 30 years	No areas occupied	Not Applicable

[†] Probability of persistence represents expected risk of extirpation over 30 years. It reflects authors' judgments and is provided only to reduce linguistic uncertainty in verbal category descriptions, following best practices for communicating risk.

5.1.5 Modeling Framework and Relative Effects of Threats

We modeled ordinal classification of resilience for each watershed as a function of threats to mussel persistence using a Bayesian approach to ordinal regression (Kruschke 2015, pp. 671–702). This model is similar to basic linear regression, except that the response variable is composed of discrete, ordered categories (i.e., extirpated, low, medium, high, or unknown) rather than a continuous variable (i.e., probability of persistence). We used the model to estimate the relative effects of a set of candidate predictor variables on current conditions. Complete details of the model are provided in Appendix B.

Currently, biologists identify the chronic effects of hydrologic alteration, habitat loss, decreased water quality, and changes in precipitation as primary threats to mussel persistence (Strayer et al. 2004, p. 435; Galbraith et al. 2010, entire). We considered 17 proxies for these threats as predictor variables and selected 11 of them for use in our model (Table 5.2). In addition, we included a standardized measure of edge density, landscape shape index, as a measure of heterogeneity. We calculated mean values of the predictor variables for each watershed using publicly available land use and downscaled climate datasets (Abatzoglou 2013, entire; Hill et al. 2016, entire; Sohl et al. 2018, entire) and functions provided in the packages sf, raster, and landscape metrics.

TABLE 5.2. Summary of threats identified, including proxy variables considered as predictor variables in the final model based on Deviance Information Criterion.

Threats	Proxy variables considered	Final model parameters
Hydrologic alteration	Reservoir surface area (% open water)	Included
	Percentage developed land use (% developed)	Included
	Patch density of developed land use (PD developed)	Included
	Number of dams	Included
Erosion and sedimentation	Mean monthly precipitation	Included
	Percentage developed land use (% developed)	Included
	Patch density of developed land use (PD developed)	Included
	Patch density of forested land (PD forest)	Included
	Mean runoff	Included
	Mean runoff from agriculture (agricultural runoff)	Included
	Patch density of cropland (PD cropland)	Included
	Percentage pasture land (% pasture)	Included
	Soil erodibility index	Not included
Climate change	Coefficient of variation mean monthly precipitation	Not included
	Mean monthly precipitation	Included
	Mean monthly air temperature	Not included
Nutrient and chemical pollution	Percentage developed land use (% developed)	Included
	Patch density of developed land use (PD developed)	Included
	Patch density of cropland (PD cropland)	Included
	Percentage pasture land (% pasture)	Included
	Percentage of mining land use & number of mines	Not included
	Stream km impaired (EPA 303(d) and TMDL lists)	Not included
Nonnative competitors	No suitable proxy for competitive effects identified	Not included
Lack of existing protection	Percentage of protected area	Included

5.2 Results

The Rabbitsfoot historically occurred within 434 watersheds located throughout the nine representation units (Table 5.3; Figure 5.2). Two hundred eighty-eight watersheds are classified as extirpated, a reduction of between 63% (watersheds with unknown condition assumed extant) and 70% (watersheds with unknown condition assumed extirpated) of its historical range (Table 5.3; Figure 5.2). Of the 123 watersheds that are extant, 83 (67.5%) are classified as low condition, 12 (9.8%) as medium, and 28 (22.7%) as high (Table 5.3; Figure 5.2). Twenty-three watersheds are classified as unknown (Table 5.3; Figure 5.2). High condition watersheds are generally isolated from other high and medium condition watersheds (Figure 5.2).

TABLE 5.3. Number of HUC 10 watersheds by representation unit and current condition.

Representation Unit	Extirpated	Unknown	Low	Medium	High	Total Watersheds
Arkansas	21	0	6	3	1	31
Cumberland	25	2	3	0	0	30
Lower Great Lakes	7	0	2	0	0	9
Lower Mississippi	14	5	4	1	0	24
Lower Ohio	118	6	21	1	9	155
Red	25	0	14	4	4	47
Tennessee	31	1	10	2	7	51
Upper Ohio	21	6	11	0	2	40
White	26	3	12	1	5	47
Total Watersheds	**288**	**23**	**83**	**12**	**28**	**434**

The lower Ohio River representation unit contains a total of 10 watersheds in either medium or high condition, five of which are concentrated in the Tippecanoe River (Table 5.3; Appendices A and C). The remaining watersheds in this unit are located in the lower mainstem of the Ohio River below the confluence with the Tennessee River, mainstem of the Ohio River, and the Green River. The Tennessee River representation unit contains a total of nine watersheds in either medium or high condition, concentrated within the Duck and Paint Rock rivers, and the mainstem of the Tennessee River at the confluence with the Ohio River (Table 5.3; Appendices A and C). The remaining watersheds in this unit are isolated within the mainstem of the Tennessee, Elk, and Buffalo rivers, and Bear Creek. The Red River representation unit contains a total of eight watersheds in either medium or high condition, six of which are concentrated within the Little, Cossatot, and Saline (Ouachita River) rivers with the remaining watersheds isolated within the Ouachita River and Bayou Bartholomew (Table 5.3; Appendices A and C). The White River representation unit contains a total of six watersheds in either medium or high condition, four of which are concentrated within the Strawberry and Spring rivers, and the others are isolated within the Middle Fork Little Red River and War Eagle Creek (Table 5.3; Appendices A and C). The upper Ohio River representation unit contains two isolated high condition watersheds; one is located within French Creek and the other within the Walhonding River (Table 5.3; Appendices A and C). The Arkansas River representation unit contains a total four watersheds in either medium or high condition, two of which are located within the Spring River and the others are isolated within the Neosho and Verdigris rivers. The lower Mississippi River representation unit contains one watershed in medium condition. No high or medium condition watersheds occur within the lower Great Lakes or Cumberland River representation units (Table 5.3; Appendices A and C).

We found that patch density and percentage of developed land use, agricultural runoff, and patch density of forested land use affected current condition (Figure 5.4). Patch density of developed land use had the greatest negative effect on current condition, with a mean parameter estimate of –0.24 (95% credible interval [CI] = –0.40, –0.09). The percentage of developed land use had a similar negative effect (–0.16), although there was less certainty in the estimate (95% CI = –0.40, 0.00). Several predictor variables including agricultural runoff, mean precipitation, and number

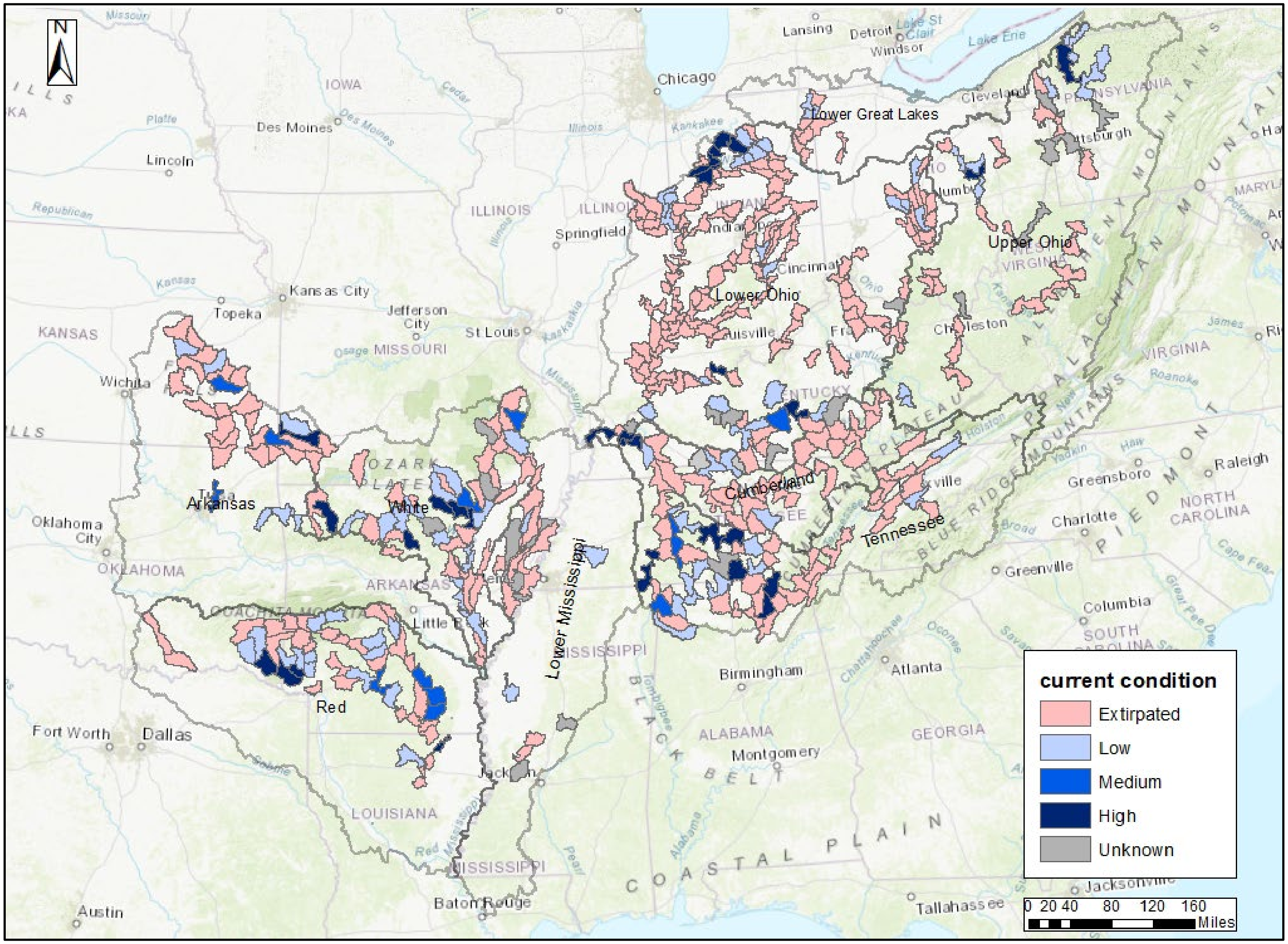

FIGURE 5.2. Current condition for the Rabbitsfoot by HUC 10 watershed. See Appendix C for detailed maps by representation unit.

of dams had a mean negative effect (50% CI < 0), but the 95% CI overlapped zero (Figure 5.4). Patch density of forested land use had a positive effect on current condition of the Rabbitsfoot (mean = 0.08; 95% CI = 0.07, –0.24). Mean runoff also had a positive effect on current condition of the Rabbitsfoot (95% CI = 0.04, 0.34).

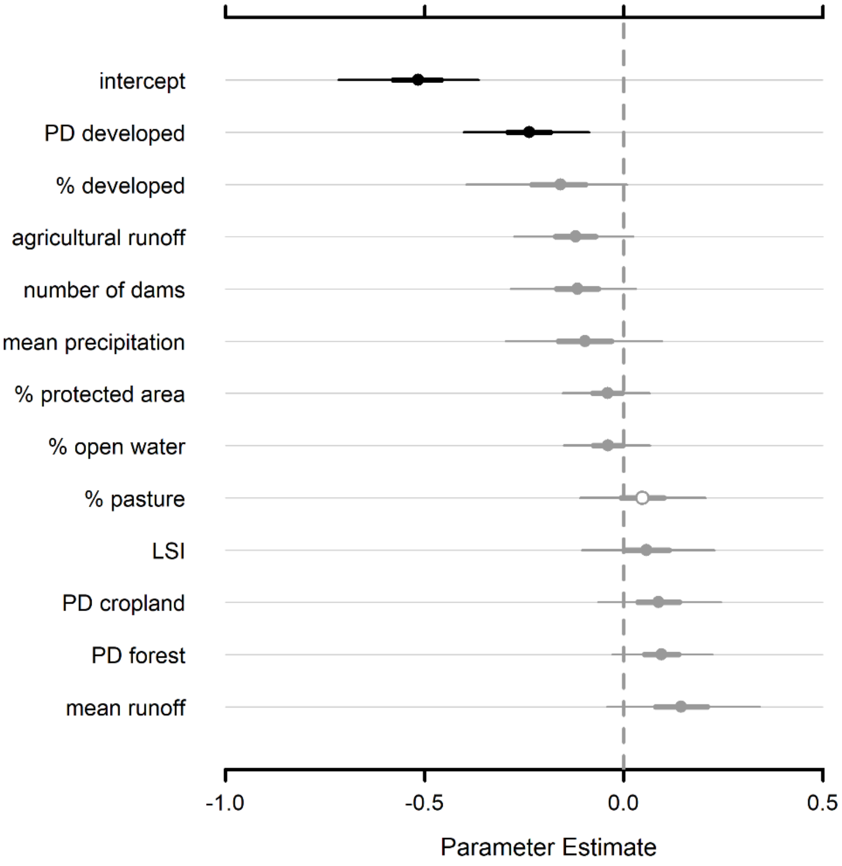

FIGURE 5.4. Posterior parameter estimates for effects of predictor variables on the condition of watersheds. Symbols represent medians, thick lines represent 50% credible intervals (CI), and thin lines represent 95% CIs. Symbols are color coded black if the 95% CI did not overlap zero, grey if the 50% CI did not overlap zero, and white if the 50% CI contains zero. PD stands for patch density and LSI for landscape shape index.

5.3 Discussion

The Rabbitsfoot historically occurred throughout a large portion of the Mississippi River and Great Lakes basins but has experienced a reduction between 63 and 70% of its historical range. While estimated reductions in range are dependent on the spatial scale of analysis, values from the present study are consistent with estimates from the literature suggesting a 60% reduction (Butler 2005, pp. 89–92; Service 2020, pp. 43–44). Our approach represents an underestimate of reduction, as we only considered watersheds with confirmed records as historically occupied. Because early surveys did not always record exact localities and many watersheds experienced hydrologic alterations prior to comprehensive sampling, the actual historical range was likely greater than our estimation. In addition, sampling has not occurred in a standardized manner and many watersheds with an unknown current condition may reflect the opinions of experts that these areas are unlikely to support individuals, i.e., sample selection bias. This information suggests that the upper value of our estimate of range reduction (70%) may be more representative of the actual reduction from historical range.

Our approach to assessing current condition is dependent upon the criteria we used to assign an ordinal classification of resilience within watersheds. Some agencies have established long-term monitoring programs to assess trends in mussel abundance and their results are consistent with observed classifications. For example, two watersheds within the Duck River, Tennessee, are classified as high condition and results from sampling at four sites within these watersheds demonstrate increasing or stable trends in mean density for the Rabbitsfoot between 2010 and 2020 (Wisniewski 2020, pp. 6–13). Results of surveys conducted at the same sites over decades documented declines in relative abundance in 2 watersheds classified as low condition within the mainstem of the lower White River, Arkansas, (Harris and Christian 2000, p. 12) and in three watersheds classified as low condition within the Buffalo River, Arkansas (Matthews 2009, p. 122).

For populations of the Rabbitsfoot to be resilient to stochastic events and the effects of anthropogenic activities, they need to occur in stream reaches with a sufficient spatial extent to support an abundance of individuals of multiple age classes and with evidence of reproduction and recruitment of juveniles into the population. Approximately 70% of extant watersheds are classified as low condition meaning the populations located within or among them have low resilience. In fact, whether or not individuals located in watersheds classified as low condition are contributing to population resilience at all is uncertain. It is possible that many of the populations located within or among low condition watersheds have so few breeding individuals that they are below effective population size (Soulé 1980, pp. 162–164).

To have sufficient representation, resilient populations should be distributed across a wide geographic extent in several river systems the historical range or be abundant enough to have a breadth of genetic, phenotypic, and ecological diversity of a species to the maximum extent possible. Because we have no information about genetic, phenotypic, or ecological variation across the range of the Rabbitsfoot, we used representation units as a proxy measure for adaptive capacity because they represent a range of abiotic and biotic factors that likely have influenced evolution of a species, and thus the ability of a species to adapt to changes in its environment. Despite the substantial reduction in the historical range, extant watersheds remain distributed throughout all nine representation units. However, the lower Great Lakes and lower Missssissippi and Cumberland River representation units only contain extant watersheds classified as low condition suggesting a decrease in representation of the Rabbitsfoot as their contribution to population resilience is uncertain. Furthermore, 35% of the watersheds classified as high and medium condition are isolated from each other, which limits the exchange of novel and beneficial adaptations among populations that increases the ability of a species to adapt to changing environmental conditions over time. Because of the substantial reduction in historical range, number of watersheds classified as low condition, and isolation of watersheds classified as high and medium condition from each other, representation for the Rabbitsfoot is low.

Redundancy is characterized by a species having multiple, resilient populations distributed across its range and representation units relative to the spatial occurrence of catastrophic events as well as connectivity among populations to increase the ability of a species to withstand or recover from catastrophic events. For wide-ranging species like the Rabbitsfoot, assessing redundancy is challenging given it's distribution over such a large geographic area relative to e.g., a narrow endemic species. However, the substantial reduction in historical range of the Rabbitsfoot decreases its' ability to withstand or recover from catastrophic events. Although watersheds classified as extant are distributed across the range of the Rabbitsfoot and all nine representation units, most are classified as low condition and may contain populations that are below effective population size. Most watersheds classified as high and medium condition lack hydrological connectivity, so immigration and emigration among those watersheds are not likely, which further decreases the ability of the Rabbitsfoot to withstand or recover from catastrophic events.

We found that the threats affecting current condition of the Rabbitsfoot were consistent with threats affecting other freshwater mussel species (reviewed in Haag 2012, pp. 316–389). Of the proxies we considered as predictor variables for threats associated with hydrological alteration, erosion and sediment, and nutrient and chemical pollution, percentage and patch density of developed land use had the greatest negative effect on current condition. The negative effect of these two variables may reflect decreasing habitat quality from increased erosion and sedimentation and poor water quality associated with increased urban development. Hopkins (2009, pp. 948) also found percentage and density of developed land negatively impacted occurrence of the Rabbitsfoot in the upper Green River system in south-centeral Kentucky. Studies evaluating stream health using a variety of criteria (pollutant loads, habitat quality, aquatic species diversity and abundance, etc.) demonstrated that stream health decreases with increasing impervious surface cover (ISC) of a watershed starting at levels of imperviousness as low as 10–20% (reviewed in Schueler et al. 2009, entire). Hilderbrand et al. (2010, p. 1010) used response thresholds of benthic macroinvertebrate taxa to derive taxon loss curves across the ISC gradient for each of the 15 major watersheds in Piedmont and Coastal Plain physiographic province of Maryland. Their results forecasted that 50% of benthic macroinvertebrate taxa will be lost in Coastal Plains watersheds by the time ISC reaches 20%; 20% and 30–40% of the taxa will be lost in Piedmont watersheds by the time ISC reaches 5% and 10%, respectively (Hilderbrand et al. 2010, p. 1012). All watersheds currently classified as medium and high condition are isolated from major metropolitan regions, suggesting that even modest increases in developed land use and corresponding increases in impervious surface cover could quickly decrease resiliency.

The mean effect of other predictor variables on current condition of the Rabbitsfoot was consistent with what we predicted, but their estimates were less precise. Agricultural runoff and mean precipitation had a mean negative effect on current condition, which may reflect decreasing habitat quality from increased erosion and sedimentation and poor water quality associated with increased agricultural development (Bauer 1988, p. 244; 1992, p. 425; Bringolf et al. (2007b, p. 2103–2105). Number of dams also had a mean negative effect on current condition. Biologists directly associate losses of many populations of freshwater mussels prior to the 1950s with habitat destruction from construction of dams (Strayer et al. 2004, p. 435). Dams continue to isolate populations from each other making them more likely to become extirpated from demographic stochastic events, such as random fluctuations in reproductive success as well as environmental stochastic events, such as droughts, floods, and effects of anthropogenic activities e.g., chemical spills or unauthorized discharges (Layzer et al. 1993, pp. 68–69; Neves et al. 1997, pp. 63–75; Watters 2000, pp. 264–265, 268; Pringle et al. 2000, pp. 810–815). Patch density of forested land use had a mean positive effect on current condition. Jones et al. (1999, p. 1460) found that contiguity of riparian forest plays a major role in regulating aquatic conditions and noted that even relatively small deforested riparian strips were associated with markedly increased loading of fine sediments and homogenization of stream habitat in a southern Appalachian stream. Mean runoff also had a mean positive effect on current condition, which is not what we predicted. This result may reflect regional differences in status and precipitation patterns given the large range of the Rabbitsfoot, rather than a true positive effect of runoff.

While our model identified range-wide threats affecting current condition of the Rabbitsfoot, we did not include proxy variables for some local scale threats known to affect freshwater mussels in our model because inclusion of these variables decreased model performance or data were not available. For example, results of studies evaluating the effect of mining activity on freshwater mussel communities throughout the Spring River Basin in southeastern Kansas and adjacent portions of southwestern Missouri, and northeastern Oklahoma, found a strong negative correlation between the distribution and abundance of native freshwater mussels, including the Rabbitsfoot, and sediment concentrations of lead, zinc, and cadmium (Angelo et al. 2007, pp. 477–485; EcoAnalysts, Inc. 2018, pp. 44–56). In Pennsylvania, Stauffer and Wilson (2018, pp. 6–10) documented expansion of the range of the Round Goby from the main stem of French Creek downstream during a 3-year period, their consumption of glochidia of unionid mussels, and a shift in habitat occupied by host fish species in response to the presence of this invasive, nonnative fish species. Natural resource professionals should consider the effect of local scale threats on the current condition of populations of the Rabbitsfoot in conservation and management planning decisions.

CHAPTER 6 – FUTURE CONDITION

We used parameter estimates, i.e., relative effects of threats on mussel persistence, from the current condition analysis and historical land use and climate patterns to predict the present, or baseline, risk of extirpation or low condition for the Rabbitsfoot. Then, we used the present risk estimates and multiple future scenarios of land use and climate to calculate projected future risk of extirpation or low condition for the Rabbitsfoot by 2050. To represent uncertainty in model predictions for present risk of extirpation, we used the 95% CI for each watershed. For projected future risk of extirpation, we represented uncertainty as the maximum and minimum 95% CI for the probability of extirpation across all combinations of land use and climate scenarios.

6.1 Methods

6.1.1 Present Risk of Extirpation or Low Condition

Using the model parameters estimated in the current condition analysis, i.e., the relative effects of each threat, we modeled the probability that a watershed would be classified as each ordinal category based on historical land use and climate patterns. These probabilities represent the sub-species present risk profile, or baseline projections, accounting for past climate and land use change. We used 2005 as the baseline for historical land use patterns because it roughly corresponds to the midpoint of contemporary occurrences and is the most recent year available in the historical land cover dataset (Sohl et al. 2018, entire). We calculated the historical baseline for climate patterns from 1950–2005, which represents the full historical period available from the climate dataset (Abatzoglou 2013, entire). The analysis focused on the probability that a watershed would be classified as either extirpated or low condition. Because watersheds classified as low in the current condition analysis contain few individuals and no evidence of reproduction or a decreasing or unknown status at the 2020 five-year review (Table 5.1), they have an inherently high risk of extirpation within several generations, i.e., low resilience. We included watersheds classified as unknown in the model and estimated their condition during model fitting. For each watershed, we used 95% CI to represent uncertainty in model predictions. Credible intervals represent the Bayesian equivalent of 95% confidence intervals, and were specifically calculated as highest posterior density intervals. Importantly, these baseline estimates are not impacted by uncertainty in future climate or land use scenarios because they are derived from currently observed patterns across the landscape.

6.1.2 Projected Future Risk of Extirpation or Low Condition

We projected the probability that a watershed would be classified as each ordinal category out to 2050, approximately three generations, to incorporate uncertainty under future scenarios of global change. We calculated future probabilities of extirpation using model parameters estimated from present relative risk of extirpation or low (baseline) conditions, land use projections under four emission scenarios (A1B, A2, B1, and B2), and climate projections under 12 climate scenarios derived from six global climate models (bcc-csm1-1-m, BNU-ESM, CanESM2, GFDL-ESM2G, GFDL-ESM2G, HadGEM2-ES365, and IPSL-CM5A-MR) and 2 representative concentration pathways (RCP 4.5 and 8.5). We averaged climate projections over 30-year periods centered on 2050, i.e., 2035–2065. We selected global climate models from 20 models available in the climate dataset to represent the full range of potential change in precipitation across the study region. Future projections for mean runoff, number of dams, and percentage of protected area were not available, so we held them constant. We represented uncertainty in the future risk of extirpation as the maximum and minimum 95% CI for the probability of extirpation across all combinations of land use and climate scenarios. This analysis approach separates model uncertainty, (i.e., structural uncertainty in the relationship between condition of watershed and various threats), from uncertainty in future climate projections, (i.e., a range of equally plausible futures), in a way that may facilitate decision-making processes associated with the Act.

6.2 Results

6.2.1 Present Risk of Extirpation or Low Condition

The present risk probility of a watershed being classified as either extirpated or low condition revealed consistent patterns across the range of the Rabbitsfoot with these conditions being more likely than not for all watersheds (95% CI >0.5; Figure 6.1). While there is uncertainty in the model estimates, this range of values suggests that the Rabbitsfoot has a high risk of being in low condition or extirpated rangewide and is consistent with the current status of this mussel sub-species. The mean present risk is high for several watersheds classified as high condition, including watersheds within the Tippecanoe (95% CI >0.80), Duck (95% CI > 0.80), Strawberry (95% CI >0.90), and lower mainstem of the Ohio (95% CI >0.90) rivers.

6.2.2 Projected Future Risk of Extirpation or Low Condition

The projected probability of a watershed being classified as extirpated or low condition at 2050 displayed small changes across future scenarios when compared to present risk probabilities (Figure 6.2). The mean probability of future risk increased or decreased by <0.10 in comparison to present risk for most watersheds (Figure 6.2, panel B) and 95% CI for many watersheds increased due to the added uncertainty across the future scenarios (Figure 6.2, panels A and C). For example, the 95% CI for the Lower Rolling Fork watershed in the Red River representation unit changed from 0.48–0.85 under present risk conditions to 0.34–0.86 under projected conditions at 2050, while the mean probability changed only slightly from 0.69 to 0.65. The 95% CI remained >0.50 for all but six watersheds. Future risk (mean probability) of a watershed being classified as extirpated or low condition at 2050 is illustrated for each representation unit in Appendix C.

6.3 Discussion

Patterns for both the present risk and projected future risk of watersheds being classified as either extirpated or low condition were clear; based on land use within them, all watersheds are more likely than not (95% CI >0.5) to be classified as either extirpated or low condition in the future. In other words, of the 123 watersheds that are currently extant, 83 are currently classified as low condition, and the 40 currently classified as either medium or high condition have a high risk of being classified as either extirpated or low condition in the future. While the Rabbitsfoot has a high risk of extirpation range wide, spatial patterns in the amount and direction of uncertainty

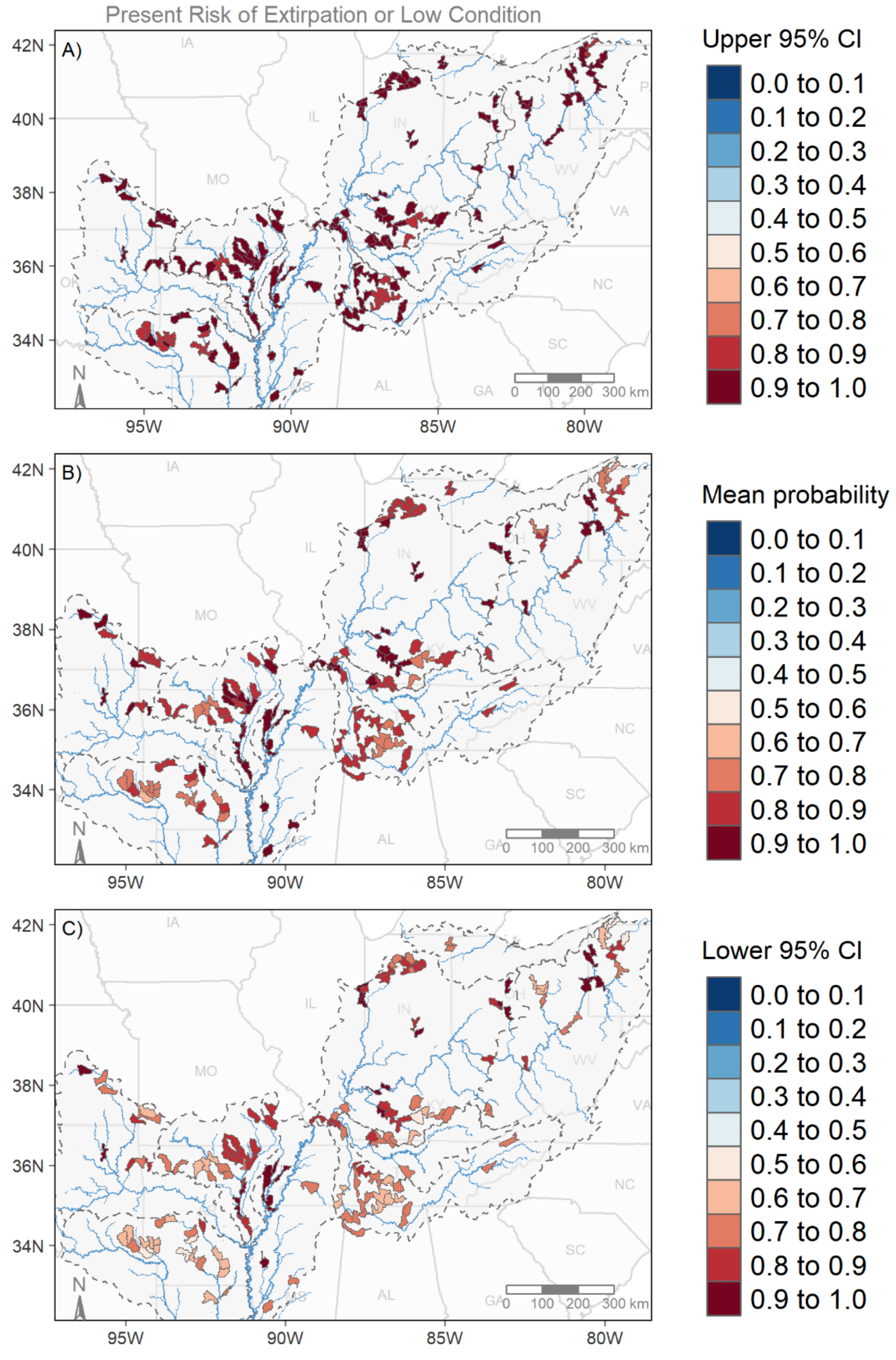

FIGURE 6.1. Present risk of extirpation or low condition for the Rabbitsfoot. Panels show the mean probability (B) and upper and lower 95% CI (A and C, respectively) of being classified as extirpated or low condition for each watershed. Warm colors (reds) represent probabilities that are more likely than not (>0.50) and cool colors (blues) represent probabilities that are less likely than not (<0.50). Dashed lines are representation units. Blue lines are major rivers.

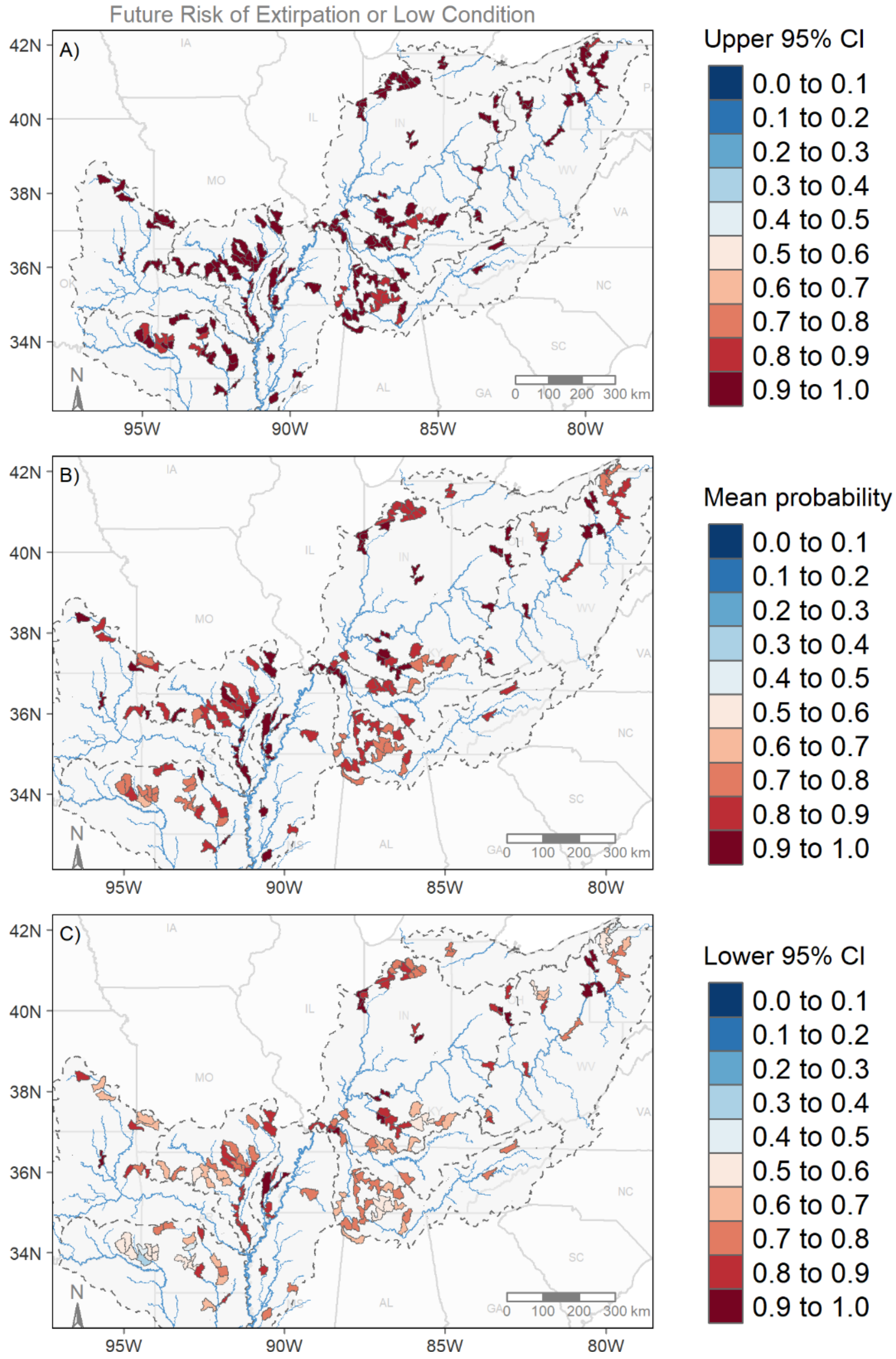

FIGURE 6.2. Projected future risk of extirpation or low condition for the Rabbitsfoot under four landuse and 12 climate scenarios. Panels show the mean probability of being classified as extirpated or low condition for each watershed (B) and maximum upper and minimum lower 95% CI (A and C, respectively) across all future scenarios. Warm colors (reds) represent probabilities that are more likely than not (>0.5) and cool colors (blues) represent probabilities that are less likely than not (<0.50). Dashed lines are representation units. Blue lines are major rivers.

demonstrate useful information for decision making. For example, the present risk of extirpation is generally lower and more uncertain in the lower portions of the Little, Rolling Fork, and Cossotot rivers in the Red River representation unit, suggesting that watersheds within these rivers may represent key strongholds in the future for populations located within or among them. Watersheds within the Duck River in the Tennessee River representation unit also represent current strongholds, yet the estimated future risk tended to be greater for these watersheds. While past efforts to halt the construction of Columbia Dam and improve discharge from the upstream Normandy Dam may have contributed to the current classification of watersheds as high condition in the mainstem of the Tennessee River (Ahlstedt et al. 2017, pp. 4, 101), proximity to the rapidly developing Nashville metropolitan area leaves the future of populations located within or among these watersheds uncertain. Even modest increases in developed land use within these watersheds e.g., surrounding Columbia or Shelbyville, Tennessee, could lead to rapid increases in the future risk of extirpation.

Several important factors impact model uncertainty and the scope of inference provided by our analysis. In order to best match the scale of available survey data, the analysis unit is relatively large and certain watersheds may contain dramatically different conditions throughout. For example, the upper and lower third of watersheds located in the lower Tippecanoe River (0512010612, Honey Creek-Tippecanoe River and 0512010613, Tippecanoe River) are currently classified as high condition, while the areas in between are heavily impacted by Lake Shafer and Lake Freeman. In addition, many surveys target mainstem reaches within each watershed and may miss important differences in how larger rivers and smaller tributaries respond to landscape level effects. For example, while populations of several mussel species have shown signs of recovery in the middle Duck River, species richness within its tributaries has declined dramatically (Ahlstedt et al. 2017, entire). This spatial heterogeneity decreases the precision of model estimates, and in both examples, overestimates the area presently supporting stable populations.

The modeling approach we used focuses on range-wide drivers of conditions within watersheds and does not adequately capture some site-specific threats with high consequences, e.g., mining, chemical spills, and emerging enigmatic die offs. While we considered multiple predictor variables related to mining during model selection, mining does not explain range-wide patterns of watershed condition and was not included in the final model. The negative effects of mining on mussels in parts of the range of the Rabbitsfoot have been well documented (Angelo et al. 2007, pp. 477–485; EcoAnalysts, Inc. 2018, pp. 44–56), and the risk posed by this threat as well as others not included in the model, e.g., invasive, nonnative aquatic species, must be considered in conservation planning in addition to the model estimates presented here.

CHAPTER 7 – INFORMING CONSERVATION PLANNING

The Service developed the SSA framework in part to separate biological risk assessments from conservation policy decisions (Doremus 1997, entire; Smith et al. 2018, p. 303). Studies have called for increased integration of quantitative modeling in conservation planning (Morris et al. 2002, entire). This chapter outlines an optimization approach to identify watersheds that maximize viability of the Rabbitsfoot. The approach consists of three parts: 1) maintaining all watersheds classified as high condition; 2) improving habitat conditions within watersheds classified as low and medium condition and, when appropriate, augmenting populations located within or among them; and 3) reintroducing individuals into watersheds currently classified as extirpated or unknown condition once threats are eliminated or abated. We provide a portfolio of watersheds as a starting point for the implementation of conservation actions across the historical range of the Rabbitsfoot (Appendix D).

7.1 Methods

7.1.1 Optimization Concept

Because it is not practical to try to improve habitat conditions in all watersheds across the historical range of the Rabbitsfoot, natural resource professionals need an approach that facilitates identification of an optimal portfolio of watersheds within which to implement conservation actions in order to achieve the recovery vision. A common method for solving such conservation planning problems is mathematical optimization using integer linear programming to maximize or minimize an objective function that describes the relationship between conservation actions and outcomes (Beyer et al. 2016, entire). The fundamental objective in developing a recovery strategy for a species is to maximize its viability. For the Rabbitsfoot, we framed our approach in terms of identifying a portfolio of watersheds that maximizes the probability of persistence (i.e., resilience) and geographic extent (i.e., representation) for a specific number of watersheds (i.e., redundancy). Specifically, the problem we seek to solve is as follows:

$$maximize \sum_{i=1}^{N} p_i x_i + w \sum_{i=1}^{N} \sum_{j=1;i<j}^{N} d_{ij} y_{ij}$$

$$subject\ to \sum_{i=1}^{N} x_i \leq T$$

$$y_{ij} - x_i \leq 0$$

$$y_{ij} - x_j \leq 0$$

$$y_{ij} - x_i - x_j \geq -1$$

$$(x_i, y_{ij}) \in \{0, 1\}$$

where x_i and y_{ij} are binary decision variables determining whether the i^th^ and j^th^ watersheds are selected, p_i is the probability of being classified as medium or high condition at 2050 for the i^th^ watershed, d_{ij} is a distance matrix of the watersheds, T is the number of watersheds, and w is a constant that controls the relative weighting of the spatial objective. The remaining constraints are required in order to consider the spatial arrangement among watersheds while maintaining a linear objective function (Beyer et al. 2016, pp. 15–16).

7.1.2 Optimization Approach

The potential conservation actions are based on our three-part approach. The first part focuses on maintaining or increasing resilience in all watersheds that are currently classified as high condition. Natural resource professionals can achieve this part by eliminating or abating threats to the persistence of the Rabbitsfoot through activities that protect and/or improve habitat such as developing conservation easements, implementing best management practices to improve management of reservoir releases, removing non-functional, aging, and unsound low-head dams, planting riparian vegetation to stabilize stream banks and decrease water temperature, and replacing culverts and bridges that restrict fish passage with new ones that allow passage and accommodate increased flows.

The second part of the approach focuses on increasing resilience in watersheds currently classified as low and medium condition with the lowest projected risk of extirpation at 2050 to increase redundancy at the sub-species level. Because this part focuses on increasing resilience within watersheds currently classified as low and medium condition, natural resource professionals should first eliminate or abate threats by implementing activities that improve habitat such as those described in the first part of the approach until populations demonstrate increasing numbers and successful recruitment over a specified number of years. After known threats are eliminated or abated, natural resource professionals may utilize augmentation to increase genetic diversity and the ability of the species to adapt to environmental changes or to increase population numbers above depensation thresholds related to Allee effects and environmental stochasticity (Strayer et al. 2019, p. 3).

The third part of the approach focuses on increasing representation and redundancy through reintroduction at historical sites. Again, this part of the approach will first require implementation of activities that improve habitat to eliminate or abate the historical threats that caused extirpation. It also will require development of successful techniques for propagation of the Rabbitsfoot as part of controlled propagation plans per Service guidelines (FR 56916–56922).

7.1.3 Applying the Optimization Approach to Representation Units

We applied our approach to each representation unit. We maintained all watersheds currently classified in high condition and selected ≤5 watersheds per representation unit for both the second and third parts of our approach. We selected ≤5 watersheds because it provided the level of redundancy needed per representation unit to achieve the recovery vision and hence goal of delisting. Maintenance of all high condition watersheds and ≤5 watersheds for both the second and third parts of the approach resulted in a total of 7–19 watersheds per representation unit.

Because all watersheds had a low probability of being classified as medium or high condition at 2050, we incorporated uncertainty into our approach by using the upper 95% CI as the probability of persistence estimate, which natural resource professionals may use to prioritize implementation of conservation activities in specific watersheds. We standardized Euclidean distances between the centroids of each watershed to 0–1 and weighted the distance objective as half important as the probability of persistence objective (w = 0.5). If watersheds did not contain critical habitat for the Rabbitsfoot, we multiplied expected utility by 0.1, which effectively constrained the optimization to select critical habitat if available. We used the optimization modeling package in R to solve the models (Schumacher 2020, entire).

7.2 Results

The need to maintain watersheds currently classified as high condition for the Rabbitsfoot is evident. Although all high condition watersheds were part of the optimal portfolio, ranking these watersheds based on projected levels of risk of extirpation at 2050 may help natural resource professional prioritize allocation of resources for conservation activities in the short-term (Figure 7.1; Appendix D). For example, watersheds in the Ohio, Tennessee, Tippecanoe, Strawberry, and Duck rivers are projected to have greater risk compared with other watersheds currently classified as high condition. Natural resource professionals may focus on limiting the effects of expanding urban development in some watersheds (e.g., Duck River) or improving ecological flows for populations downstream of dams (e.g., Ohio, Tennessee, Duck, and Tippecanoe rivers). For example, TVA's initiation of the RRI program to improve dissolved oxygen concentrations and increase minimum flows from Normandy Dam on the Duck River (Ahlstedt et al. 2017, p. 101) and efforts by the ODNR to remove Six Mile Dam from the Walhonding River (Ashland Times-Gazette, October 9, 2020) are consistent with the first part of the approach and may contribute to maintaining the high condition of these watersheds.

The optimal portfolio also identified ≤5 watersheds classified as low or medium condition for each representation unit where habitat improvement may increase resilience (Figure 7.2, Appendix D). Although natural resource professionals may have additional reasons to prioritize specific watersheds, such as cost, land ownership or the presence of local conservation groups or public support, the factors considered in the future conditions model and the optimization selected watersheds consistent with ongoing conservation actions. For example, three of the watersheds selected in the Lower Ohio representation unit are along the Green River, where public-private partnerships are planning the removal of four dams (fws.gov). Similar efforts by the TWRA to remove Browns Mill Dam in the East Fork Stones River (D. W. Hubbs, TWRA, pers. comm. 2019) are consistent with the watersheds selected for the Cumberland representation unit. Because the Cumberland representation unit only contained three low condition watersheds, the optimal portfolio included all available watersheds. The optimization also selected watersheds identified in several other conservation plans, including strategic habitat and river reaches units for aquatic species of conservation concern in Alabama and adjoining states (Bear Creek and Elk River in Tennessee representation unit) (alh2o.org) and priority areas for agriculture best management practices in Pennsylvania's Wildlife Action Plan (French Creek in Upper Ohio representation unit) (Pennsylvania State Wildlife Action Plan 2015–2025 Appendix 1.4F-Mussels p. 36).

Habitat protection and improvement may not be equally feasible in all watersheds identified in the optimal portfolio. For example, the optimization selected Owl Creek and Wolf Creek watersheds in the Neosho River of the Arkansas River representation unit despite them being listed as impaired on the EPA 303(d) list from effects of mining. The optimization included the Owl Creek-Neosho River watershed primarily because it contains critical habitat. Although habitat improvement may be possible in this watershed, it may be more cost-effective to improve habitat in nearby watersheds e.g., within the Spring River. This result highlights the need to consider additional local-scale effects and the costs of conservation actions when evaluating optimization results. Future application of the optimization approach could be improved by incorporating estimates for the relative costs across activities and watersheds into the objective function. Natural resource professionals also may utilize augmentation through captive propagation in these watersheds if numbers are critically low; however, the environmental or biological factors limiting mussel abundance must be understood before efforts are likely to succeed (Strayer et al. 2019, p. 3). For example, if host fish species are a limiting factor, then augmentation may only lead to short-term gains in mussel abundance and resources may be better directed elsewhere (Strayer et al. 2019, p. 3).

Reintroduction through captive propagation will be an integral component of a recovery strategy for the Rabbitsfoot because adult mussels have low dispersal ability and fragmentation of riverine habitat has reduced opportunities for larval dispersal by host fish species throughout much of the historical range. The optimal portfolio selected ≤5 watersheds for reintroductions in each representation unit, many of which are consistent with other conservation prioritization projects (Figure 7.3, Appendix D). For example, natural resource professionals identified the Widows Creek-Tennessee River (downstream of Nickjack Dam), Upper Flint River, and Richland Creek watersheds as strategic habitat units and river reaches for aquatic species of conservation concern in Alabama and adjoining states (alh2o.org). The optimization selected other watersheds for reintroduction of the Rabbitsfoot that natural resource professionals specifically identified for potential reintroduction including the Roaring Paunch Creek-Big South Fork Cumberland River and Sinking Creek-Rockcastle River in the Cumberland representation unit (CRMRC 2010, p. 85) and Johnson Creek-Licking River the Lower Ohio River representation unit and Fox-Creek Licking River in the Upper Ohio River representation unit (alh2o.org). The optimization did not select some watersheds identified by natural resource professionals primarily due to the added objective to maximize distance between watersheds. For example, CRMRC (2010, p 85) identified the tailwaters of Guntersville Dam on the Tennessee River as a potential reintroduction site, but this watershed was not included in the optimal portfolio due its proximity to two high condition watersheds in the Paint Rock River.

Reintroductions are only likely to be successful if the threat that caused extirpation is eliminated (Strayer et al. 2019, pp. 2–3). Because mussel declines can sometimes be caused by unknown factors (Haag 2012, entire) and the optimization only assessed watersheds relative to others, it will be necessary for natural resource professionals to confirm that potential threats are eliminated or abated before attempting reintroductions. For example, seven of the nine watersheds in the Lower Great Lakes representation unit are classified as extirpated, which results in the Saint Mary’s River being included in the optimal portfolio despite containing heavily developed areas around Fort Wayne, Indiana. Although inclusion of this watershed in the portfolio maximizes the probability of persistence relative to the remaining watersheds, it may

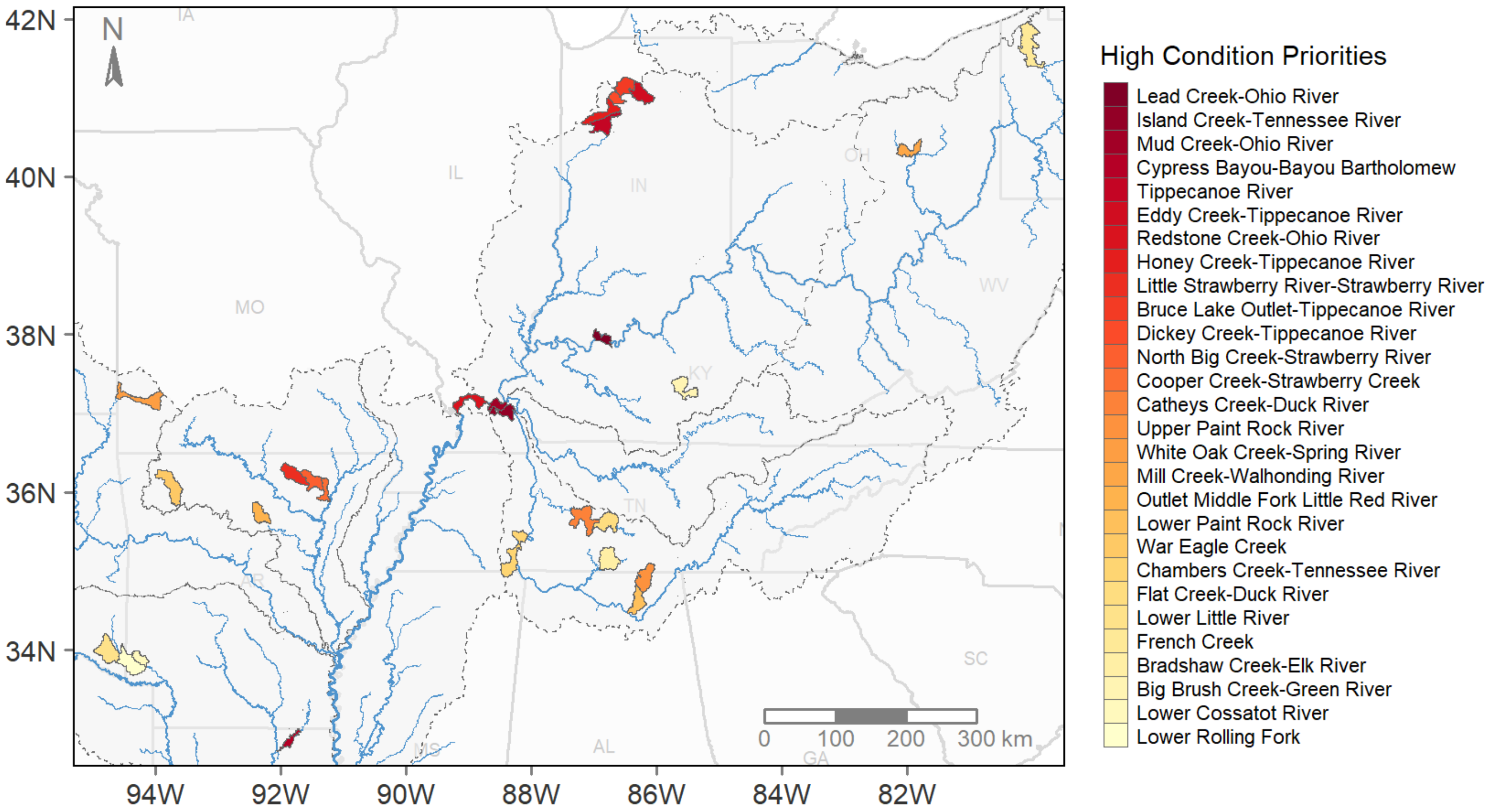

FIGURE 7.1. Watersheds in the optimal portfolio that are currently classified in high condition. Colors represent the relative risk at 2050, where reds have a greater projected probability of being classified as low or extirpated. Dotted lines mark boundaries of representation units. The full portfolio is provided in Appendix D.

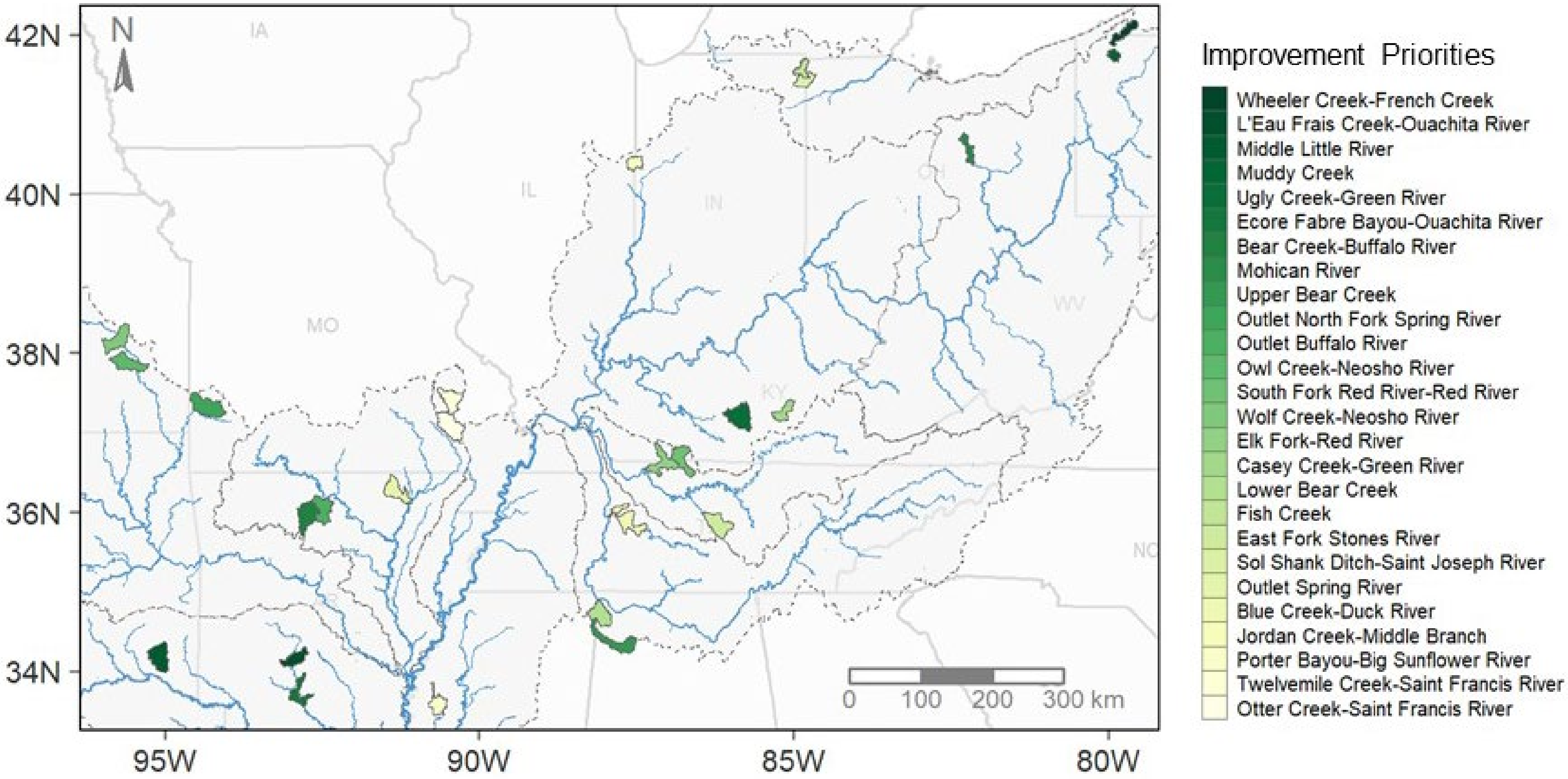

FIGURE 7.2. Watersheds prioritized for habitat improvement in the optimal portfolio that are currently classified as low or medium condition. Only three watersheds are displayed per representation unit for clarity of labels. Darker colors represent watersheds with a higher probability of being classified in medium or high condition at 2050. Dotted lines mark boundaries of representation units. The full portfolio is provided in Appendix D.

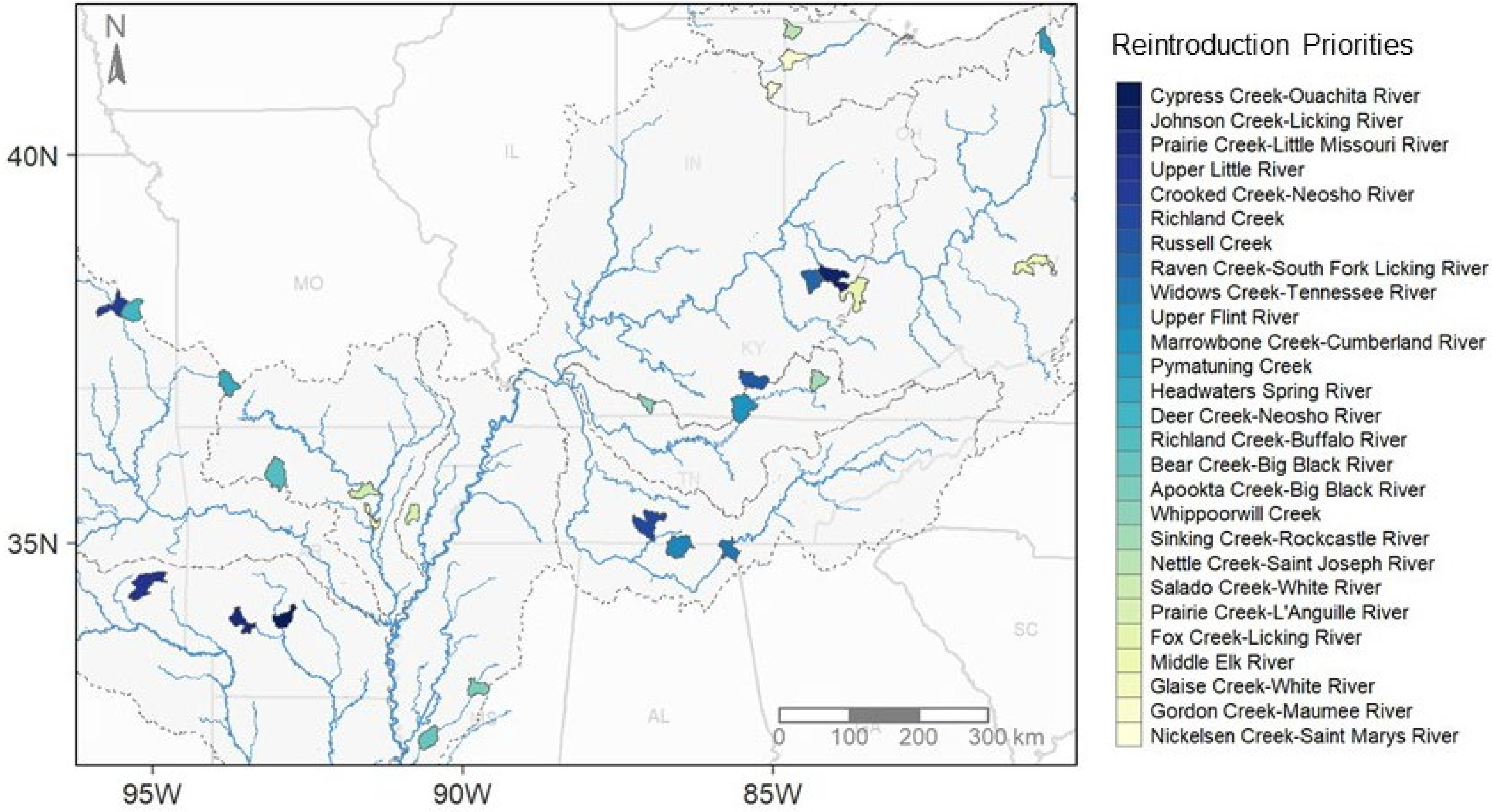

FIGURE 7.3. Watersheds prioritized for reintroduction in the optimal portfolio that are currently classified as extirpated or unknown condition. Only three watersheds are displayed per representation unit for clarity of labels. Darker colors represent watersheds with a higher probability of being classified in medium or high condition at 2050. Dotted lines mark boundaries of representation units. The full portfolio is provided in Appendix D.

not represent a feasible opportunity for reintroduction. The optimization could be further constrained to only include watersheds above a specified probability of persistence; however, a limited number of historically occupied watersheds will still affect opportunities for reintroduction in certain representation units. Reintroduction efforts may benefit from an experimental approach, where individuals are first released on a smaller scale to identify the likelihood of success (Haag and Williams 2014, pp. 9–10).

We constrainted the optimization to first select watersheds containing critical habitat by weighting them 10 times greater in the objective function. This weighting had a large effect on the optimal portfolio of some representation units. For example, in the Arkansas River representation unit the optimization selected the Owl Creek-Neosho River watershed as a possibility for habitat improvement and Deer Creek-Neosho River watershed as a possibility for reintroduction even though seemingly less impaired watersheds exist within the Spring River and Shoal Creek. The optimal portfolio for the White River representation unit was especially impacted by this constraint, with 12 of the 15 watersheds selected containing critical habitat. If we decrease the weight to 5 times greater instead of 10, for example, the optimal portfolio for the White River representation unit contains 10 watersheds with critical habitat out of 15 instead of 12. This suggests that constraining the optimization to select watersheds containing critical habitat within which to implement conservation actions should be carefully considered as other watersheds not containing critical habitat may provide larger increases in resilience or representation. We included the critical habitat constraint because the Service determined these specific areas within the range were occupied by the Rabbitsfoot at the time it was listed, contained features essential to the conservation of this subspecies (e.g., adequate water quality and quantity, floodplain connectivity, stream channel stability, water flow, and presence of host fish species), and detailed actions needed to eliminate or abate threats to these features (70 FR 59808, pp. 24710–24714). However, natural resource professionals should carefully consider the relative weight of critical habitat if the strategy for recovery is to maximize probability of persistence and distance between occupied watersheds.

7.3 Discussion

These results provide an optimal portfolio of 113 watersheds that maximize the probability of persistence (i.e., resilience) and geographic extent (i.e., representation) for the Rabbitsfoot as an example of how the results of SSA analyses can be applied to conservation planning. The number of watersheds selected can be altered to meet specific recovery criteria, such as the number of redundant watersheds in each representation unit. Recovery criteria for achieving a certain level of representation could be incorporated by constraining the optimization to include a specified percentage of the historical range based on the areas of each watershed (Wolf et al. 2015, p. 205). By incorporating this optimization approach with local knowledge on the feasibility of and response to implementation of conservation actions and anticipated costs across the three-part approach, natural resource professionals can achieve an efficient, cost-effective, and transparent recovery strategy for the Rabbitsfoot.

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Appendix A. Current and future condition of HUC 10 watersheds summarized by rivers and creeks used as analysis units in the listing rule and 2020 5-year status review. Mean probabilities of future condition (Extirpated, Low, Medium, High) refer to baseline probabilities referenced in the SSA report.

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
Arkansas	Center Creek	1107020706	Center Creek	Extirpated	0.67	0.21	0.03	0.09
Arkansas	Cottonwood River	1107020304	Outlet Cottonwood River	Low	0.85	0.11	0.01	0.03
Arkansas	Fall River	1107010201	Headwaters Fall River	Extirpated	0.84	0.11	0.01	0.03
Arkansas	Fall River	1107010203	Indian Creek-Fall River	Extirpated	0.66	0.21	0.03	0.10
Arkansas	Illinois River	1111010304	Headwaters Illinois River	Low	0.74	0.17	0.02	0.06
Arkansas	Illinois River	1111010306	Upper Illinois River	Low	0.68	0.20	0.03	0.09
Arkansas	Illinois River	1111010308	Lower Illinois River	Low	0.71	0.19	0.03	0.07
Arkansas	Neosho River	1107020102	Rock Creek-Neosho River	Extirpated	0.71	0.18	0.03	0.08
Arkansas	Neosho River	1107020103	Dow Creek-Neosho River	Extirpated	0.83	0.12	0.01	0.03
Arkansas	Neosho River	1107020104	Neosho River-John Redmond Reservoir	Extirpated	0.73	0.18	0.03	0.07
Arkansas	Neosho River	1107020401	Wolf Creek-Neosho River	Low	0.73	0.17	0.02	0.07
Arkansas	Neosho River	1107020402	Crooked Creek-Neosho River	Extirpated	0.59	0.24	0.04	0.13
Arkansas	Neosho River	1107020403	Deer Creek-Neosho River	Extirpated	0.70	0.19	0.03	0.08
Arkansas	Neosho River	1107020404	Owl Creek-Neosho River	Medium	0.67	0.21	0.03	0.09
Arkansas	Neosho River	1107020501	Canville Creek-Neosho River	Extirpated	0.71	0.18	0.03	0.08
Arkansas	Neosho River	1107020502	Hickory Creek-Neosho River	Extirpated	0.74	0.17	0.02	0.07
Arkansas	Neosho River	1107020506	Cherry Creek-Neosho River	Extirpated	0.64	0.22	0.04	0.10
Arkansas	Neosho River	1107020601	Tar Creek-Neosho River	Extirpated	0.64	0.22	0.04	0.11
Arkansas	North Fork Spring River	1107020703	Outlet North Fork Spring River	Low	0.57	0.24	0.04	0.14
Arkansas	Shoal Creek	1107020707	Headwaters Shoal Creek	Extirpated	0.65	0.21	0.03	0.11
Arkansas	Shoal Creek	1107020708	Shoal Creek	Extirpated	0.78	0.15	0.02	0.05
Arkansas	Spring River	1107020701	Headwaters Spring River	Extirpated	0.62	0.22	0.04	0.12
Arkansas	Spring River	1107020705	White Oak Creek-Spring River	High	0.58	0.24	0.04	0.14
Arkansas	Spring River	1107020709	Shawnee Creek-Spring River	Medium	0.76	0.16	0.02	0.05

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
Arkansas	Spring River	1107020710	Outlet of Spring River	Extirpated	0.68	0.20	0.03	0.09
Arkansas	Verdigris River	1107010101	Headwaters Verdigris River	Extirpated	0.78	0.15	0.02	0.05
Arkansas	Verdigris River	1107010105	Chetopa Creek-Verdigris River	Extirpated	0.72	0.18	0.03	0.07
Arkansas	Verdigris River	1107010301	Big Hill Creek-Verdigris River	Extirpated	0.78	0.15	0.02	0.05
Arkansas	Verdigris River	1107010302	Pumpkin Creek-Verdigris River	Extirpated	0.71	0.19	0.03	0.07
Arkansas	Verdigris River	1107010303	California Creek-Verdigris River	Extirpated	0.62	0.23	0.04	0.12
Arkansas	Verdigris River	1107010502	Sweetwater Creek-Verdigris River	Medium	0.86	0.10	0.01	0.03
Cumberland	Beaver Creek	513010305	Beaver Creek	Extirpated	0.75	0.17	0.02	0.06
Cumberland	Big South Fork Cumberland River	513010405	Roaring Paunch Creek-Big South Fork Cumberland River	Extirpated	0.60	0.23	0.04	0.13
Cumberland	Big South Fork Cumberland River	513010407	Sinking Creek-Big South Fork Cumberland River	Extirpated	0.70	0.19	0.03	0.08
Cumberland	Caney Fork	513010809	Lower Caney Fork	Extirpated	0.73	0.18	0.02	0.06
Cumberland	Cumberland River	513010102	Yellow Creek-Cumberland River	Extirpated	0.77	0.16	0.02	0.05
Cumberland	Cumberland River	513010302	Pitman Creek-Cumberland River	Extirpated	0.74	0.18	0.02	0.06
Cumberland	Cumberland River	513010304	Wolf Creek-Cumberland River	Extirpated	0.71	0.19	0.03	0.08
Cumberland	Cumberland River	513010306	Crocus Creek-Cumberland River	Extirpated	0.68	0.20	0.03	0.09
Cumberland	Cumberland River	513010307	Marrowbone Creek-Cumberland River	Extirpated	0.58	0.24	0.04	0.14
Cumberland	Cumberland River	513010601	Sugar Creek-Cumberland River	Extirpated	0.75	0.17	0.02	0.06
Cumberland	Cumberland River	513020103	Wilburn Creek-Cumberland River	Extirpated	0.67	0.21	0.03	0.09
Cumberland	Cumberland River	513020203	Whites Creek-Cumberland River	Extirpated	0.94	0.04	0.00	0.01
Cumberland	Cumberland River	513020503	Half Pone Creek-Cumberland River	Extirpated	0.79	0.15	0.02	0.05
Cumberland	Cumberland River	513020504	Saline Creek-Cumberland River	Extirpated	0.70	0.20	0.03	0.08
Cumberland	Cumberland River	513020507	Eddy Creek-Cumberland River	Extirpated	0.75	0.17	0.02	0.06
Cumberland	East Fork Obey River	513010501	East Fork Obey River	Extirpated	0.72	0.18	0.03	0.07
Cumberland	East Fork Stones River	513020301	East Fork Stones River	Low	0.66	0.21	0.03	0.09
Cumberland	Harpeth River	513020401	Upper Harpeth River	Extirpated	0.71	0.19	0.03	0.07
Cumberland	Harpeth River	513020406	Lower Harpeth River	Extirpated	0.93	0.05	0.01	0.01
Cumberland	Obey River	513010503	Big Eagle Creek-Obey River	Extirpated	0.69	0.20	0.03	0.08

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
Cumberland	Red River	513020601	Summers Branch-Red River	Unknown	0.67	0.21	0.03	0.09
Cumberland	Red River	513020602	South Fork Red River-Red River	Low	0.60	0.24	0.04	0.13
Cumberland	Red River	513020606	West Fork Red River	Unknown	0.73	0.18	0.03	0.07
Cumberland	Red River	513020607	Elk Fork-Red River	Low	0.64	0.22	0.03	0.10
Cumberland	Rockcastle River	513010201	Middle Fork Rockcastle River	Extirpated	0.66	0.21	0.03	0.10
Cumberland	Rockcastle River	513010204	Little Rockcastle River-Rockcastle River	Extirpated	0.70	0.20	0.03	0.08
Cumberland	Rockcastle River	513010205	Sinking Creek-Rockcastle River	Extirpated	0.59	0.24	0.04	0.13
Cumberland	Stones River	513020303	Stones River	Extirpated	0.81	0.14	0.02	0.04
Cumberland	West Fork Stones River	513020302	West Fork Stones River	Extirpated	0.74	0.17	0.02	0.06
Cumberland	Whippoorwill Creek	513020603	Whippoorwill Creek	Extirpated	0.62	0.23	0.04	0.12
Lower Great Lakes	Auglaize River	410000709	Jennings Creek-Auglaize River	Extirpated	0.61	0.23	0.04	0.12
Lower Great Lakes	Fish Creek	410000304	Fish Creek	Low	0.88	0.09	0.01	0.02
Lower Great Lakes	Maumee River	410000501	Headwaters Maumee River	Extirpated	0.71	0.19	0.03	0.07
Lower Great Lakes	Maumee River	410000502	Gordon Creek-Maumee River	Extirpated	0.63	0.23	0.04	0.11
Lower Great Lakes	Saint Joseph River	410000303	Nettle Creek-Saint Joseph River	Extirpated	0.66	0.21	0.03	0.09
Lower Great Lakes	Saint Joseph River	410000305	Sol Shank Ditch-Saint Joseph River	Low	0.80	0.14	0.02	0.04
Lower Great Lakes	Saint Joseph River	410000308	Saint Joseph River	Extirpated	0.76	0.16	0.02	0.06
Lower Great Lakes	Saint Mary's River	410000405	Nickelsen Creek-Saint Marys River	Extirpated	0.88	0.09	0.01	0.02
Lower Great Lakes	Saint Mary's River	410000406	Saint Marys River	Extirpated	0.61	0.23	0.04	0.12
Lower Mississippi	Big Black River	806020107	Apookta Creek-Big Black River	Unknown	0.73	0.17	0.03	0.07
Lower Mississippi	Big Black River	806020207	Bear Creek-Big Black River	Unknown	0.83	0.10	0.01	0.05
Lower Mississippi	Big Creek	802020203	Big Creek	Extirpated	0.79	0.15	0.02	0.05
Lower Mississippi	Big Sunflower River	803020705	Porter Bayou-Big Sunflower River	Low	0.85	0.11	0.01	0.03
Lower Mississippi	Hatchie River	801020805	Cypress Creek-Hatchie River	Low	0.62	0.23	0.04	0.11
Lower Mississippi	Saint Francis River	802020202	Stouts Creek-Saint Francis River	Extirpated	0.76	0.16	0.02	0.05
Lower Mississippi	Saint Francis River	802020204	Twelvemile Creek-Saint Francis River	Medium	0.68	0.20	0.03	0.09
Lower Mississippi	Saint Francis River	802020205	Otter Creek-Saint Francis River	Low	0.74	0.17	0.02	0.06

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
Lower Mississippi	Saint Francis River	802020304	Mingo Ditch-Saint Francis River	Extirpated	0.76	0.17	0.02	0.06
Lower Mississippi	Saint Francis River	802020305	Wilhelmina Cutoff-Saint Francis River	Extirpated	0.78	0.15	0.02	0.05
Lower Mississippi	Saint Francis River	802020306	Big Slough Ditch-Saint Francis River	Extirpated	0.79	0.15	0.02	0.05
Lower Mississippi	Saint Francis River	802020308	Little Slough Ditch	Low	0.84	0.11	0.01	0.03
Lower Mississippi	Saint Francis River	802020309	Saint Francis River Floodway-Saint Francis River	Extirpated	0.81	0.13	0.02	0.04
Lower Mississippi	Saint Francis River	802020310	Saint Francis Bay	Unknown	0.93	0.05	0.01	0.01
Lower Mississippi	Saint Francis River	802020312	Tyronza River	Extirpated	0.89	0.08	0.01	0.02
Lower Mississippi	Saint Francis River	802020313	Copeland Slough-Saint Francis River	Extirpated	0.78	0.15	0.02	0.05
Lower Mississippi	Saint Francis River	802020314	Fifteenmile Bayou	Extirpated	0.89	0.09	0.01	0.02
Lower Mississippi	Saint Francis River	802020315	Blackfish Bayou	Unknown	0.83	0.12	0.01	0.03
Lower Mississippi	Saint Francis River	802020316	L'Anguille River-Saint Francis River	Extirpated	0.84	0.11	0.01	0.03
Lower Mississippi	Saint Francis River	802020408	Buffalo Creek Ditch	Extirpated	0.82	0.13	0.02	0.03
Lower Mississippi	Saint Francis River	802020409	Pemiscot Bayou	Unknown	0.92	0.06	0.01	0.01
Lower Mississippi	Saint Francis River	802020502	Prairie Creek-L'Anguille River	Extirpated	0.77	0.16	0.02	0.05
Lower Mississippi	Yazoo River	803020607	Piney Creek-Yazoo River	Extirpated	0.84	0.11	0.01	0.04
Lower Mississippi	Yazoo River	803020608	O'Neil Creek-Yazoo River	Extirpated	0.73	0.18	0.03	0.07
Lower Ohio	Alum Creek	506000116	Alum Creek-Big Walnut Creek	Extirpated	0.95	0.04	0.00	0.01
Lower Ohio	Barren River	511000202	Peter Creek-Barren River	Unknown	0.52	0.26	0.05	0.17
Lower Ohio	Barren River	511000204	Bays Fork-Barren River	Extirpated	0.55	0.25	0.05	0.16
Lower Ohio	Barren River	511000207	Park City-Barren River	Extirpated	0.60	0.23	0.04	0.12
Lower Ohio	Barren River	511000209	Little Muddy Creek-Barren River	Low	0.79	0.15	0.02	0.05
Lower Ohio	Big Blue River	512020401	Duck Creek-Big Blue River	Extirpated	0.87	0.10	0.01	0.02
Lower Ohio	Big Blue River	512020408	Big Blue River	Extirpated	0.79	0.14	0.02	0.04
Lower Ohio	Big Darby Creek	506000119	Headwaters Big Darby Creek	Extirpated	0.80	0.14	0.02	0.04
Lower Ohio	Big Darby Creek	506000121	Worthington Ditch-Big Darby Creek	Low	0.83	0.12	0.01	0.03
Lower Ohio	Big Darby Creek	506000122	Hellbranch Run-Big Darby Creek	Extirpated	0.90	0.08	0.01	0.02
Lower Ohio	Big Monon Creek	512010611	Big Monon Creek	Extirpated	0.68	0.20	0.03	0.09

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
Lower Ohio	Big Walnut Creek	506000115	Blacklick Creek-Big Walnut Creek	Extirpated	0.94	0.05	0.00	0.01
Lower Ohio	Black Creek	512020206	Black Creek	Extirpated	0.77	0.16	0.02	0.05
Lower Ohio	Brandywine Creek	512020403	Brandywine Creek	Extirpated	0.83	0.12	0.01	0.03
Lower Ohio	Deer Creek	506000202	Sugar Run-Deer Creek	Extirpated	0.78	0.15	0.02	0.05
Lower Ohio	Deer Creek	506000203	Hay Run-Deer Creek	Extirpated	0.69	0.20	0.03	0.08
Lower Ohio	Drakes Creek[1]	511000206	Drakes Creek	Extirpated	0.65	0.21	0.03	0.10
Lower Ohio	Driftwood River	512020409	Driftwood River	Extirpated	0.86	0.11	0.01	0.03
Lower Ohio	East Fork White River	512020602	Little Sand Creek-East Fork White River	Extirpated	0.88	0.09	0.01	0.02
Lower Ohio	East Fork White River	512020605	Thompson Slough-East Fork White River	Extirpated	0.80	0.14	0.02	0.04
Lower Ohio	East Fork White River	512020606	Hough Creek-East Fork White River	Extirpated	0.74	0.18	0.02	0.06
Lower Ohio	East Fork White River	512020803	Rock Lick Branch-East Fork White River	Extirpated	0.52	0.26	0.05	0.17
Lower Ohio	East Fork White River	512020810	Leatherwood Creek-East Fork White River	Extirpated	0.67	0.21	0.03	0.09
Lower Ohio	East Fork White River	512020814	Barn Run-East Fork White River	Extirpated	0.58	0.24	0.04	0.14
Lower Ohio	East Fork White River	512020815	East Fork White River	Extirpated	0.60	0.24	0.04	0.13
Lower Ohio	Eel River	512010404	Clear Creek-Eel River	Low	0.67	0.21	0.03	0.09
Lower Ohio	Eel River	512010405	Paw Paw Creek-Eel River	Low	0.67	0.21	0.03	0.09
Lower Ohio	Eel River	512010406	Weesau Creek-Eel River	Low	0.65	0.21	0.03	0.10
Lower Ohio	Eel River	512010407	Eel River	Low	0.68	0.20	0.03	0.09
Lower Ohio	Embarras River	512011212	Honey Creek-Embarras River	Extirpated	0.66	0.21	0.03	0.09
Lower Ohio	Embarras River	512011215	Indian Creek-Embarras River	Extirpated	0.79	0.14	0.02	0.04
Lower Ohio	Flatrock River	512020501	Shankatank Creek-Flatrock River	Extirpated	0.82	0.13	0.02	0.04
Lower Ohio	Flatrock River	512020504	Mill Creek-Flatrock River	Extirpated	0.74	0.17	0.02	0.06
Lower Ohio	Flatrock River	512020506	Flatrock River	Low	0.85	0.11	0.01	0.03
Lower Ohio	Green River	511000101	South Fork Green River-Green River	Unknown	0.61	0.23	0.04	0.12
Lower Ohio	Green River	511000103	Casey Creek-Green River	Low	0.60	0.24	0.04	0.12
Lower Ohio	Green River	511000107	Big Brush Creek-Green River	High	0.46	0.27	0.05	0.21

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
Lower Ohio	Green River	511000108	Ugly Creek-Green River	Medium	0.42	0.28	0.06	0.24
Lower Ohio	Green River	511000113	Beaverdam Creek-Green River	Low	0.72	0.18	0.03	0.07
Lower Ohio	Green River	511000114	Big Reedy Creek-Green River	Extirpated	0.61	0.23	0.04	0.12
Lower Ohio	Green River	511000303	Indian Camp Creek-Green River	Unknown	0.73	0.18	0.03	0.07
Lower Ohio	Green River	511000305	Lewis Creek-Green River	Unknown	0.85	0.11	0.01	0.03
Lower Ohio	Green River	511000505	Race Creek-Green River	Extirpated	0.84	0.12	0.01	0.03
Lower Ohio	Kentucky River	510020507	Clear Creek-Kentucky River	Extirpated	0.63	0.22	0.04	0.11
Lower Ohio	Licking River	510010111	Johnson Creek-Licking River	Extirpated	0.49	0.26	0.05	0.19
Lower Ohio	Licking River	510010112	Grassy Creek-Licking River	Extirpated	0.73	0.18	0.03	0.07
Lower Ohio	Licking River	510010113	Banklick Creek-Licking River	Extirpated	0.97	0.03	0.00	0.00
Lower Ohio	Little Darby Creek	506000120	Little Darby Creek	Low	0.80	0.14	0.02	0.04
Lower Ohio	Little Miami River	509020208	Turtle Creek-Little Miami River	Extirpated	0.82	0.13	0.02	0.03
Lower Ohio	Middle Branch North Fork Vermilion River	512010907	Jordan Creek-Middle Branch	Low	0.78	0.15	0.02	0.05
Lower Ohio	Middle Fork Vermilion River	512010901	Big Four Ditch	Extirpated	0.78	0.15	0.02	0.05
Lower Ohio	Middle Fork Vermilion River	512010905	Middle Fork Vermillion River	Extirpated	0.85	0.11	0.01	0.03
Lower Ohio	Mississinewa River	512010305	Massey Creek-Mississinewa River	Extirpated	0.83	0.12	0.01	0.03
Lower Ohio	Mississinewa River	512010306	Mississinewa River	Extirpated	0.76	0.16	0.02	0.05
Lower Ohio	Nolin River	511000110	Middle Nolin River	Low	0.60	0.23	0.04	0.12
Lower Ohio	North Fork Vermilion River	512010908	North Fork Vermilion River	Low	0.86	0.10	0.01	0.02
Lower Ohio	Ohio Brush Creek	509020105	Lick Fork-Ohio Brush Creek	Extirpated	0.52	0.26	0.05	0.18
Lower Ohio	Ohio River	509020106	Cabin Creek-Ohio River	Unknown	0.72	0.19	0.03	0.07
Lower Ohio	Ohio River	509020301	Mill Creek	Extirpated	0.95	0.04	0.00	0.01
Lower Ohio	Ohio River	509020302	Muddy Creek-Ohio River	Extirpated	0.98	0.02	0.00	0.00
Lower Ohio	Ohio River	514010103	Corn Creek-Ohio River	Extirpated	0.78	0.15	0.02	0.05
Lower Ohio	Ohio River	514010106	Goose Creek-Ohio River	Extirpated	0.84	0.12	0.01	0.03
Lower Ohio	Ohio River	514010109	Beargrass Creek-Ohio River	Extirpated	0.95	0.04	0.00	0.01
Lower Ohio	Ohio River	514020102	Clover Creek-Ohio River	Extirpated	0.74	0.17	0.02	0.06

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
Lower Ohio	Ohio River	514020107	Lead Creek-Ohio River	High	0.83	0.12	0.01	0.03
Lower Ohio	Ohio River	514020108	Pup Creek-Ohio River	Extirpated	0.78	0.15	0.02	0.05
Lower Ohio	Ohio River	514020112	Caney Creek-Ohio River	Low	0.88	0.09	0.01	0.02
Lower Ohio	Ohio River	514020204	Canoe Creek-Ohio River	Extirpated	0.94	0.05	0.00	0.01
Lower Ohio	Ohio River	514020206	Bayou Creek-Ohio River	Extirpated	0.89	0.08	0.01	0.02
Lower Ohio	Ohio River	514020301	Goose Pond Ditch-Ohio River	Extirpated	0.70	0.19	0.03	0.08
Lower Ohio	Ohio River	514020303	Hurricane Creek-Ohio River	Low	0.58	0.24	0.04	0.14
Lower Ohio	Ohio River	514020309	Barren Creek-Ohio River	Unknown	0.63	0.22	0.04	0.11
Lower Ohio	Ohio River	514020601	Mud Creek-Ohio River	High	0.75	0.17	0.02	0.06
Lower Ohio	Ohio River	514020602	Massac Creek-Ohio River	Extirpated	0.66	0.21	0.03	0.10
Lower Ohio	Ohio River	514020607	Redstone Creek-Ohio River	High	0.76	0.16	0.02	0.06
Lower Ohio	Olentangy River	506000110	Grave Creek-Olentangy River	Low	0.84	0.12	0.01	0.03
Lower Ohio	Mississinewa River	512010306	Mississinewa River	Extirpated	0.76	0.16	0.02	0.05
Lower Ohio	Olentangy River	506000111	Rush Run-Olentangy River	Extirpated	0.96	0.03	0.00	0.00
Lower Ohio	Patoka River	512020904	Altar Ceek-Patoka River	Extirpated	0.60	0.23	0.04	0.12
Lower Ohio	Pipe Creek	512010116	Little Pipe Creek-Wabash River	Extirpated	0.81	0.14	0.02	0.04
Lower Ohio	Pipe Creek	512020104	Pipe Creek	Extirpated	0.80	0.14	0.02	0.04
Lower Ohio	Rough River	511000405	Muddy Creek-Rough River	Low	0.80	0.14	0.02	0.04
Lower Ohio	Russell Creek	511000104	Russell Creek	Extirpated	0.52	0.26	0.05	0.18
Lower Ohio	Salamonie River	512010203	Black Creek-Salamonie River	Extirpated	0.75	0.17	0.02	0.06
Lower Ohio	Salamonie River	512010204	Salamonie River	Extirpated	0.75	0.17	0.02	0.06
Lower Ohio	Salt Fork Vermilion River	512010902	Saline Branch Drainage Ditch	Extirpated	0.93	0.06	0.01	0.01
Lower Ohio	Salt Fork Vermilion River	512010903	Upper Salt Fork Drainage Ditch-Salt Fork	Extirpated	0.85	0.11	0.01	0.03
Lower Ohio	Salt Fork Vermilion River	512010904	Stony Creek	Extirpated	0.87	0.10	0.01	0.02
Lower Ohio	Salt Fork Vermilion River	512010906	Salt Fork	Extirpated	0.83	0.12	0.01	0.03
Lower Ohio	Salt River	514010203	Beech Creek-Salt River	Extirpated	0.62	0.23	0.04	0.11
Lower Ohio	Scioto River	506000112	Indian Run-Scioto River	Extirpated	0.91	0.07	0.01	0.02
Lower Ohio	Scioto River	506000123	Scioto Big Run-Scioto River	Low	0.91	0.07	0.01	0.01

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
Lower Ohio	Scioto River	506000204	Scippo Creek-Scioto River	Extirpated	0.75	0.17	0.02	0.06
Lower Ohio	Scioto River	506000205	Kinnikinnick Creek-Scioto River	Extirpated	0.75	0.17	0.02	0.06
Lower Ohio	South Fork Licking River	510010204	Raven Creek-South Fork Licking River	Extirpated	0.49	0.27	0.05	0.19
Lower Ohio	South Fork Licking River	510010205	Fork Lick Creek-South Fork Licking River	Extirpated	0.61	0.23	0.04	0.12
Lower Ohio	Sugar Creek (Wabash)	512011001	Browns Wonder Creek-Sugar Creek	Extirpated	0.82	0.13	0.02	0.04
Lower Ohio	Sugar Creek (Wabash)	512011004	Prarie Creek-Sugar Creek	Extirpated	0.83	0.12	0.01	0.03
Lower Ohio	Sugar Creek (Wabash)	512011006	Sugar Creek	Extirpated	0.76	0.16	0.02	0.05
Lower Ohio	Sugar Creek (White)	512020404	Little Sugar Creek-Sugar Creek	Extirpated	0.88	0.09	0.01	0.02
Lower Ohio	Sugar Creek (White)	512020407	Sugar Creek	Low	0.79	0.15	0.02	0.04
Lower Ohio	Tippecanoe River	512010603	Trimble Creek-Tippecanoe River	Extirpated	0.70	0.19	0.03	0.08
Lower Ohio	Tippecanoe River	512010604	Chippewanuck Creek-Tippecanoe River	Low	0.61	0.23	0.04	0.12
Lower Ohio	Tippecanoe River	512010605	Eddy Creek-Tippecanoe River	High	0.71	0.19	0.03	0.07
Lower Ohio	Tippecanoe River	512010606	Bruce Lake Outlet-Tippecanoe River	High	0.61	0.23	0.04	0.12
Lower Ohio	Tippecanoe River	512010609	Dickey Creek-Tippecanoe River	High	0.62	0.23	0.04	0.12
Lower Ohio	Tippecanoe River	512010612	Honey Creek-Tippecanoe River	High	0.69	0.20	0.03	0.08
Lower Ohio	Tippecanoe River	512010613	Tippecanoe River	High	0.75	0.17	0.02	0.06
Lower Ohio	Vermilion River	512010810	Swank Creek-Little Vermilion River	Extirpated	0.80	0.14	0.02	0.04
Lower Ohio	Vermilion River	512010909	Vermilion River	Low	0.85	0.11	0.01	0.03
Lower Ohio	Wabash River	512010113	Loon Creek-Wabash River	Extirpated	0.72	0.18	0.03	0.07
Lower Ohio	Wabash River	512010114	Treaty Creek-Wabash River	Extirpated	0.75	0.17	0.02	0.06
Lower Ohio	Wabash River	512010502	Rock Creek-Wabash River	Extirpated	0.75	0.17	0.02	0.06
Lower Ohio	Wabash River	512010503	Rattlesnake Creek-Wabash River	Extirpated	0.75	0.17	0.02	0.06
Lower Ohio	Wabash River	512010506	Sugar Creek-Wabash River	Extirpated	0.76	0.16	0.02	0.05
Lower Ohio	Wabash River	512010802	Burnett Creek-Wabash River	Extirpated	0.86	0.10	0.01	0.02
Lower Ohio	Wabash River	512010805	Kickapoo Creek-Wabash River	Extirpated	0.67	0.21	0.03	0.09
Lower Ohio	Wabash River	512010806	Big Shawnee Creek-Wabash River	Extirpated	0.73	0.18	0.03	0.07
Lower Ohio	Wabash River	512010808	Jordan Creek-Wabash River	Extirpated	0.75	0.17	0.02	0.06

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
Lower Ohio	Wabash River	512010809	Coal Creek	Extirpated	0.64	0.22	0.03	0.10
Lower Ohio	Wabash River	512010816	Mill Creek-Wabash River	Extirpated	0.75	0.17	0.02	0.06
Lower Ohio	Wabash River	512011106	Coal Creek-Wabash River	Extirpated	0.94	0.05	0.00	0.01
Lower Ohio	Wabash River	512011109	Clear Creek-Wabash River	Extirpated	0.75	0.17	0.02	0.06
Lower Ohio	Wabash River	512011111	Prairie Creek-Wabash River	Extirpated	0.64	0.22	0.04	0.10
Lower Ohio	Wabash River	512011113	Raccoon River-Wabash River	Extirpated	0.59	0.24	0.04	0.13
Lower Ohio	Wabash River	512011116	Turtle Creek-Wabash River	Extirpated	0.64	0.22	0.03	0.10
Lower Ohio	Wabash River	512011117	No Business Creek-Wabash River	Extirpated	0.70	0.20	0.03	0.08
Lower Ohio	Wabash River	512011119	Kelso Creek-Wabash River	Extirpated	0.85	0.11	0.01	0.03
Lower Ohio	Wabash River	512011301	Raccoon Creek	Extirpated	0.59	0.24	0.04	0.13
Lower Ohio	Wabash River	512011302	River Deshee-Wabash River	Extirpated	0.73	0.18	0.03	0.07
Lower Ohio	Wabash River	512011303	Coffee Bayou-Wabash River	Extirpated	0.78	0.15	0.02	0.05
Lower Ohio	Wabash River	512011304	Bonpas Creek	Extirpated	0.64	0.22	0.03	0.10
Lower Ohio	Wabash River	512011306	French Creek-Wabash River	Extirpated	0.74	0.18	0.02	0.06
Lower Ohio	Wabash River	512011308	Fox River-Wabash River	Extirpated	0.65	0.22	0.03	0.10
Lower Ohio	Wabash River	512011309	Levy Slough-Wabash River	Extirpated	0.67	0.21	0.03	0.09
Lower Ohio	Wabash River	512011409	Pond Creek-Little Wabash River	Extirpated	0.67	0.21	0.03	0.09
Lower Ohio	Wabash River	512011410	Lick Creek-Little Wabash River	Extirpated	0.66	0.21	0.03	0.10
Lower Ohio	Wabash River	512011505	Lost Creek-Skillet Fork	Extirpated	0.64	0.22	0.03	0.10
Lower Ohio	Wabash River	514020207	Lower Highland Creek-Ohio River	Extirpated	0.79	0.15	0.02	0.05
Lower Ohio	Wabash River	514020405	Cane Creek	Extirpated	0.62	0.23	0.04	0.12
Lower Ohio	Walnut Creek	506000118	Little Walnut Creek-Walnut Creek	Extirpated	0.88	0.09	0.01	0.02
Lower Ohio	West Fork White River	512020101	Muncie Creek-White River	Extirpated	0.76	0.16	0.02	0.06
Lower Ohio	West Fork White River	512020103	Killbuck Creek-White River	Extirpated	0.90	0.07	0.01	0.01
Lower Ohio	West Fork White River	512020107	Stony Creek-White River	Extirpated	0.84	0.12	0.01	0.03
Lower Ohio	West Fork White River	512020110	Crooked Creek-White River	Extirpated	0.92	0.06	0.01	0.01
Lower Ohio	West Fork White River	512020112	Pleasant Run-White River	Extirpated	0.94	0.04	0.00	0.01
Lower Ohio	West Fork White River	512020114	Clear Creek-White River	Extirpated	0.90	0.08	0.01	0.01

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
Lower Ohio	West Fork White River	512020115	Lambs Creek-White River	Extirpated	0.89	0.08	0.01	0.02
Lower Ohio	West Fork White River	512020117	Butler Creek-White River	Extirpated	0.68	0.20	0.03	0.09
Lower Ohio	West Fork White River	512020202	Fish Creek-White River	Extirpated	0.57	0.25	0.04	0.14
Lower Ohio	West Fork White River	512020204	Lattas Creek-White River	Extirpated	0.76	0.17	0.02	0.05
Lower Ohio	West Fork White River	512020205	First Creek-White River	Extirpated	0.70	0.20	0.03	0.08
Lower Ohio	Wabash River	512011303	Coffee Bayou-Wabash River	Extirpated	0.78	0.15	0.02	0.05
Lower Ohio	West Fork White River	512020208	Indian Creek-White River	Extirpated	0.74	0.17	0.02	0.06
Lower Ohio	West Fork White River	512020209	Kessinger Ditch-White River	Extirpated	0.75	0.17	0.02	0.06
Lower Ohio	Whetstone Creek	506000109	Whetstone Creek	Extirpated	0.68	0.20	0.03	0.09
Lower Ohio	White River	512020210	White River	Extirpated	0.66	0.21	0.03	0.09
Red	Bayou Bartholomew	804020510	Cypress Bayou-Bayou Bartholomew	High	0.68	0.20	0.03	0.09
Red	Bayou D'Arbonne	804020609	Lower Bayou D'Arbonne	Low	0.60	0.23	0.04	0.13
Red	Blue River	1114010201	Upper Blue River	Extirpated	0.52	0.26	0.05	0.17
Red	Blue River	1114010202	Lower Blue River	Extirpated	0.66	0.21	0.03	0.10
Red	Caddo River	804010202	Upper Caddo River	Extirpated	0.72	0.18	0.03	0.08
Red	Caddo River	804010203	Middle Caddo River	Extirpated	0.66	0.21	0.03	0.10
Red	Caddo River	804010204	Lower Caddo River	Extirpated	0.58	0.24	0.04	0.14
Red	Cossatot River	1114010904	Headwaters Cossatot River	Extirpated	0.65	0.21	0.03	0.11
Red	Cossatot River	1114010905	Upper Cossatot River	Low	0.53	0.26	0.05	0.16
Red	Cossatot River	1114010906	Lower Cossatot River	High	0.49	0.26	0.05	0.20
Red	Glover River	1114010702	Glover River	Low	0.51	0.26	0.05	0.18
Red	Little Missouri River	804010305	Prairie Creek-Little Missouri River	Extirpated	0.46	0.27	0.05	0.21
Red	Little Missouri River	804010306	Terre Rouge Creek	Extirpated	0.57	0.24	0.04	0.14
Red	Little Missouri River	804010309	Outlet Little Missouri River	Low	0.46	0.27	0.05	0.21
Red	Little River	1114010701	Upper Little River	Extirpated	0.54	0.25	0.05	0.17
Red	Little River	1114010703	Middle Little River	Low	0.53	0.26	0.05	0.16
Red	Little River	1114010704	Lower Little River	High	0.58	0.24	0.04	0.14
Red	Little River	1114010903	Lower Rolling Fork	High	0.42	0.28	0.06	0.25

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
Red	Little River	1114010912	Outlet Little River	Extirpated	0.54	0.25	0.05	0.16
Red	Mountain Fork Little River	1114010801	Upper Mountain Fork	Extirpated	0.71	0.19	0.03	0.08
Red	Mountain Fork Little River	1114010802	Middle Mountain Fork	Extirpated	0.52	0.26	0.05	0.17
Red	Mountain Fork Little River	1114010803	Lower Mountain Fork	Extirpated	0.53	0.26	0.05	0.17
Red	North Fork Saline River	804020301	North Fork Saline River	Extirpated	0.74	0.17	0.02	0.06
Red	Ouachita River	804010101	Headwaters Ouachita River	Extirpated	0.74	0.17	0.02	0.07
Red	Ouachita River	804010102	Big Brushy Creek-Ouachita River	Low	0.70	0.19	0.03	0.08
Red	Ouachita River	804010103	Ouachita River-Lake Ouachita	Low	0.68	0.19	0.03	0.10
Red	Ouachita River	804010201	De Roche Creek-Ouachita River	Low	0.59	0.24	0.04	0.13
Red	Ouachita River	804010205	L'Eau Frais Creek-Ouachita River	Low	0.43	0.28	0.06	0.24
Red	Ouachita River	804010206	Cypress Creek-Ouachita River	Extirpated	0.35	0.28	0.06	0.31
Red	Ouachita River	804010209	Ecore Fabre Bayou-Ouachita River	Medium	0.47	0.27	0.05	0.21
Red	Ouachita River	804020107	Locust Bayou-Ouachita River	Low	0.67	0.21	0.03	0.09
Red	Ouachita River	804020108	Mill Creek-Ouachita River	Extirpated	0.66	0.21	0.03	0.10
Red	Ouachita River	804020202	Big Brushy Creek-Ouachita River	Extirpated	0.69	0.20	0.03	0.08
Red	Ouachita River	804020204	Shiloh Creek-Ouachita River	Extirpated	0.72	0.18	0.03	0.07
Red	Ouachita River	804020702	Bayou De Siard-Ouachita River	Extirpated	0.82	0.13	0.02	0.04
Red	Rolling Fork Little River	1114010902	Upper Rolling Fork	Low	0.48	0.27	0.05	0.20
Red	Saline River (Little River)	1114010907	Headwaters Saline River	Extirpated	0.58	0.24	0.04	0.14
Red	Saline River (Little River)	1114010908	Messer Creek-Saline River	Low	0.47	0.27	0.05	0.21
Red	Saline River (Little River)	1114010909	Saline River-Millwood Lake	Low	0.48	0.27	0.05	0.20
Red	Saline River (Ouachita River)	804020303	Middle Fork Saline River-Alum Fork Saline River	Extirpated	0.70	0.19	0.03	0.08
Red	Saline River (Ouachita River)	804020307	Trace Creek-Saline River	Low	0.75	0.17	0.02	0.06
Red	Saline River (Ouachita River)	804020308	Lost Creek-Saline River	Extirpated	0.56	0.25	0.04	0.15
Red	Saline River (Ouachita River)	804020309	Grisly Creek-Saline River	Extirpated	0.56	0.25	0.04	0.15
Red	Saline River (Ouachita River)	804020403	Seay Creek-Saline River	Medium	0.54	0.25	0.05	0.17
Red	Saline River (Ouachita River)	804020404	Brown Creek-Saline River	Medium	0.58	0.24	0.04	0.14
Red	Saline River (Ouachita River)	804020405	Snake Creek-Saline River	Medium	0.53	0.25	0.05	0.17

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
Red	Saline River (Ouachita River)	804020406	L'Aigle Creek-Saline River	Extirpated	0.56	0.25	0.04	0.15
Tennessee	Bear Creek	603000601	Upper Bear Creek	Low	0.57	0.25	0.04	0.15
Tennessee	Bear Creek	603000602	Cedar Creek	Extirpated	0.58	0.24	0.04	0.14
Tennessee	Bear Creek	603000603	Lower Bear Creek	Medium	0.58	0.24	0.04	0.13
Tennessee	Big Rock Creek	604000205	Big Rock Creek	Extirpated	0.56	0.25	0.04	0.15
Tennessee	Buffalo River	604000401	Upper Buffalo River	Extirpated	0.69	0.20	0.03	0.08
Tennessee	Buffalo River	604000403	Lower Buffalo River	Medium	0.65	0.21	0.03	0.10
Tennessee	Clinch River	601020508	North Fork Clinch River-Clinch River	Low	0.63	0.22	0.04	0.11
Tennessee	Clinch River	601020510	Indian Creek-Clinch River	Extirpated	0.62	0.23	0.04	0.11
Tennessee	Clinch River	601020511	Norris Lake-Clinch River	Extirpated	0.78	0.15	0.02	0.05
Tennessee	Clinch River	601020703	Poplar Creek	Extirpated	0.90	0.08	0.01	0.02
Tennessee	Clinch River	601020704	Clinch River	Extirpated	0.89	0.08	0.01	0.02
Tennessee	Duck River	604000201	Normandy Lake-Duck River	Extirpated	0.81	0.13	0.02	0.04
Tennessee	Duck River	604000203	Fall Creek-Duck River	Low	0.60	0.23	0.04	0.13
Tennessee	Duck River	604000207	Flat Creek-Duck River	High	0.47	0.27	0.05	0.20
Tennessee	Duck River	604000305	Catheys Creek-Duck River	High	0.65	0.21	0.03	0.10
Tennessee	Duck River	604000309	Blue Creek-Duck River	Low	0.70	0.19	0.03	0.08
Tennessee	Elk River	603000304	Tims Ford Lake-Elk River	Extirpated	0.82	0.13	0.02	0.04
Tennessee	Elk River	603000307	Norris Creek-Elk River	Low	0.52	0.26	0.05	0.17
Tennessee	Elk River	603000309	Bradshaw Creek-Elk River	High	0.50	0.27	0.05	0.19
Tennessee	Elk River	603000402	Richland Creek	Unknown	0.52	0.26	0.05	0.17
Tennessee	Elk River	603000404	Anderson Creek-Elk River	Low	0.51	0.26	0.05	0.18
Tennessee	Flint River	603000203	Upper Flint River	Extirpated	0.63	0.22	0.04	0.11
Tennessee	Flint River	603000204	Lower Flint River	Extirpated	0.70	0.19	0.03	0.08
Tennessee	French Broad River	601010704	French Broad River	Low	0.65	0.21	0.03	0.10
Tennessee	Holston River	601010403	Holston River	Extirpated	0.72	0.18	0.03	0.07
Tennessee	Holston River	601020102	Sinking Creek-Tennessee River	Extirpated	0.89	0.08	0.01	0.02
Tennessee	Little Pigeon River	601010703	Little Pigeon River	Extirpated	0.57	0.24	0.04	0.14

Representation Unit	**Listing Rule Population**	**Number**	**Name**	**Current Condition**	**Extirpated**	**Low**	**Medium**	**High**
Tennessee	Little Tennessee River	601020405	Lower Tellico Lake	Extirpated	0.66	0.21	0.03	0.10
Tennessee	Lookout Creek	602000111	Lookout Creek	Extirpated	0.80	0.14	0.02	0.04
Tennessee	Paint Rock River[2]	603000201	Upper Paint Rock River	High	0.63	0.22	0.04	0.11
Tennessee	Paint Rock River	603000202	Lower Paint Rock River	High	0.57	0.25	0.04	0.14
Tennessee	Sequatchie River	602000403	Lower Sequatchie River	Extirpated	0.69	0.20	0.03	0.08
Tennessee	Shoal Creek	603000505	Wilson Lake-Shoal Creek	Low	0.65	0.21	0.03	0.10
Tennessee	Tennessee River	602000106	Dallas Lake-Tennessee River	Extirpated	0.76	0.16	0.02	0.06
Tennessee	Tennessee River	602000112	Nickajack Lake-Tennessee River	Extirpated	0.87	0.10	0.01	0.02
Tennessee	Tennessee River	603000102	Widows Creek-Tennessee River	Extirpated	0.52	0.26	0.05	0.17
Tennessee	Tennessee River	603000104	Mud Creek-Tennessee River	Extirpated	0.46	0.27	0.05	0.22
Tennessee	Tennessee River	603000106	Upper Guntersville Lake	Extirpated	0.65	0.21	0.03	0.10
Tennessee	Tennessee River	603000109	Lower Guntersville Lake	Extirpated	0.63	0.21	0.04	0.12
Tennessee	Tennessee River	603000209	Tennessee River-Wheeler Lake	Extirpated	0.66	0.21	0.03	0.09
Tennessee	Tennessee River	603000211	Swan Creek-Wheeler Lake	Extirpated	0.72	0.18	0.03	0.07
Tennessee	Tennessee River	603000508	Tennessee River-Pickwick Lake	Low	0.66	0.21	0.03	0.09
Tennessee	Tennessee River	603000510	Bluff Creek-Pickwick Lake	Extirpated	0.63	0.22	0.04	0.11
Tennessee	Tennessee River	603000512	Indian Creek-Pickwick Lake	Extirpated	0.71	0.18	0.03	0.08
Tennessee	Tennessee River	604000105	Chambers Creek-Tennessee River	High	0.55	0.25	0.04	0.15
Tennessee	Tennessee River	604000107	Beech Creek-Tennessee River	Extirpated	0.62	0.23	0.04	0.11
Tennessee	Tennessee River	604000110	Lick Creek-Tennessee River	Extirpated	0.65	0.21	0.03	0.10
Tennessee	Tennessee River	604000504	Tennessee River-Kentucky Lake	Extirpated	0.83	0.12	0.02	0.03
Tennessee	Tennessee River	604000509	Standing Rock Creek-Kentucky Lake	Extirpated	0.69	0.19	0.03	0.09
Tennessee	Tennessee River	604000510	Jonathan Creek-Kentucky Lake	Low	0.73	0.17	0.03	0.08
Tennessee	Tennessee River	604000605	Island Creek-Tennessee River	High	0.77	0.15	0.02	0.05
Upper Ohio	Allegheny River	501000301	Upper Allegheny River	Low	0.49	0.26	0.05	0.20
Upper Ohio	Allegheny River	501000309	Lower Allegheny River	Low	0.71	0.18	0.03	0.07
Upper Ohio	Allegheny River	501000611	Allegheny River	Unknown	0.67	0.21	0.03	0.09
Upper Ohio	Beaver River	503010401	Beaver River	Extirpated	0.94	0.04	0.00	0.01

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
Upper Ohio	Big Sandy River	507020404	Whites Creek-Big Sandy River	Extirpated	0.85	0.11	0.01	0.03
Upper Ohio	Black Fork Mohican River	504000202	Rocky Fork-Black Fork Mohican River	Extirpated	0.83	0.12	0.01	0.03
Upper Ohio	Elk River	505000706	Middle Elk River	Extirpated	0.65	0.21	0.03	0.10
Upper Ohio	Elk River	505000709	Lower Elk River	Extirpated	0.71	0.19	0.03	0.08
Upper Ohio	French Creek	501000403	Wheeler Creek-French Creek	Low	0.41	0.28	0.06	0.26
Upper Ohio	French Creek[3]	501000409	French Creek	High	0.52	0.25	0.05	0.18
Upper Ohio	Killbuck Creek	504000308	Doughty Creek-Killbuck Creek	Low	0.55	0.25	0.04	0.16
Upper Ohio	LeBoeuf Creek	501000404	LeBoeuf Creek	Low	0.47	0.26	0.05	0.21
Upper Ohio	Levisa Fork	507020302	Mud Creek-Levisa Fork	Extirpated	0.84	0.12	0.01	0.03
Upper Ohio	Levisa Fork	507020305	Georges Creek-Levisa Fork	Extirpated	0.84	0.11	0.01	0.03
Upper Ohio	Licking River	510010108	Fox Creek-Licking River	Extirpated	0.66	0.21	0.03	0.09
Upper Ohio	Little Kanawha River	503020303	Upper Little Kanawha River	Extirpated	0.66	0.21	0.03	0.10
Upper Ohio	Mahoning River	503010308	Mill Creek-Mahoning River	Extirpated	0.96	0.03	0.00	0.00
Upper Ohio	Mohican River	504000208	Mohican River	Low	0.52	0.26	0.05	0.18
Upper Ohio	Monongahela River	502000508	Pigeon Creek-Monongahela River	Extirpated	0.97	0.03	0.00	0.00
Upper Ohio	Muddy Creek	501000405	Muddy Creek	Low	0.44	0.27	0.06	0.23
Upper Ohio	Muskingum River	504000403	Symmes Creek-Muskingum River	Low	0.70	0.19	0.03	0.08
Upper Ohio	Muskingum River	504000408	Brush Creek-Muskingum River	Extirpated	0.65	0.21	0.03	0.10
Upper Ohio	Muskingum River	504000412	Rainbow Creek-Muskingum River	Extirpated	0.68	0.20	0.03	0.09
Upper Ohio	Not Applicable	503010103	Montour Run-Ohio River	Unknown	0.98	0.01	0.00	0.00
Upper Ohio	Ohio River	503010111	Kings Creek-Ohio River	Unknown	0.96	0.03	0.00	0.00
Upper Ohio	Ohio River	503020110	French Creek-Ohio River	Unknown	0.65	0.22	0.03	0.10
Upper Ohio	Ohio River	509010110	Symmes Creek-Ohio River	Unknown	0.76	0.16	0.02	0.05
Upper Ohio	Pymatuning Creek	503010203	Pymatuning Creek	Extirpated	0.54	0.25	0.05	0.16
Upper Ohio	Scioto River	506000213	Big Beaver Creek-Scioto River	Extirpated	0.72	0.19	0.03	0.07
Upper Ohio	Scioto River	506000216	Camp Creek-Scioto River	Extirpated	0.78	0.15	0.02	0.05
Upper Ohio	Scioto River	509010306	Little Scioto River-Ohio River	Extirpated	0.80	0.14	0.02	0.04
Upper Ohio	Shenango River	503010204	Big Run-Shenango River	Low	0.94	0.05	0.00	0.01

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
Upper Ohio	Shenango River	503010206	Yankee Run-Shenango River	Unknown	0.92	0.06	0.01	0.01
Upper Ohio	South Fork Kentucky River	510020304	Buffalo Creek-South Fork Kentucky River	Low	0.63	0.22	0.04	0.11
Upper Ohio	South Fork Kentucky River	510020306	Meadow Creek-South Fork Kentucky River	Low	0.66	0.21	0.03	0.10
Upper Ohio	Tuscarawas River	504000112	Pigeon Run-Tuscarawas River	Extirpated	0.82	0.13	0.02	0.04
Upper Ohio	Tuscarawas River	504000117	Stone Creek-Tuscarawas River	Extirpated	0.77	0.16	0.02	0.05
Upper Ohio	Walhonding River	504000309	Mill Creek-Walhonding River	High	0.56	0.24	0.04	0.15
Upper Ohio	West Fork River	502000203	Middle West Fork River	Extirpated	0.67	0.21	0.03	0.09
Upper Ohio	West Fork River	502000206	Lower West Fork River	Extirpated	0.81	0.13	0.02	0.04
White	Black River	1101000704	Logan Creek	Unknown	0.68	0.20	0.03	0.09
White	Black River	1101000705	Clearwater Lake-Black River	Extirpated	0.69	0.20	0.03	0.08
White	Black River	1101000706	Big Brushy Creek-Black River	Extirpated	0.69	0.20	0.03	0.08
White	Black River	1101000707	Indian Creek-Black River	Extirpated	0.76	0.16	0.02	0.05
White	Black River	1101000708	Big Hunting Slough-Black River	Extirpated	0.84	0.12	0.01	0.03
White	Black River	1101000710	Murray Creek-Black River	Extirpated	0.84	0.12	0.01	0.03
White	Black River	1101000901	Fourche River	Unknown	0.69	0.20	0.03	0.08
White	Black River	1101000902	Big Running Water Creek-Black River	Low	0.80	0.14	0.02	0.04
White	Black River	1101000903	Curia Creek-Black River	Extirpated	0.62	0.23	0.04	0.11
White	Buffalo River	1101000502	Headwaters Buffalo River	Low	0.62	0.23	0.04	0.12
White	Buffalo River	1101000503	Richland Creek-Buffalo River	Extirpated	0.58	0.24	0.04	0.14
White	Buffalo River	1101000504	Bear Creek-Buffalo River	Low	0.52	0.26	0.05	0.17
White	Buffalo River	1101000505	Outlet Buffalo River	Low	0.50	0.26	0.05	0.19
White	Current River	1101000806	Big Barren Creek-Current River	Extirpated	0.60	0.24	0.04	0.13
White	Current River	1101000809	Current River	Extirpated	0.63	0.22	0.04	0.11
White	Little Red River	1101001402	South Fork Little Red River	Extirpated	0.57	0.25	0.04	0.14
White	Little Red River	1101001409	Tenmile Creek-Little Red River	Extirpated	0.75	0.17	0.02	0.06
White	Middle Fork Little Red River	1101001403	Headwaters Middle Fork Little Red River	Extirpated	0.60	0.24	0.04	0.13
White	Middle Fork Little Red River	1101001404	Outlet Middle Fork Little Red River	High	0.59	0.24	0.04	0.13

Representation Unit	Listing Rule Population	Number	Name	Current Condition	Extirpated	Low	Medium	High
White	North Fork White River	1101000610	Norfork Lake-North Fork River	Extirpated	0.80	0.14	0.02	0.04
White	Reeses Fork Cache River	802030208	Cache Bayou-Cache River	Extirpated	0.82	0.13	0.02	0.04
White	South Fork Spring River	1101001001	Myatt Creek	Extirpated	0.65	0.21	0.03	0.10
White	South Fork Spring River	1101001003	South Fork Spring River	Low	0.81	0.13	0.02	0.04
White	Spring River	1101001002	Headwaters Spring River	Low	0.71	0.19	0.03	0.08
White	Spring River	1101001005	Outlet Spring River	Medium	0.72	0.18	0.03	0.07
White	Spring River	1101001104	Lower Eleven Point River	Extirpated	0.65	0.22	0.03	0.10
White	Strawberry River	1101001202	Little Strawberry River-Strawberry River	High	0.73	0.18	0.02	0.06
White	Strawberry River	1101001203	North Big Creek-Strawberry River	High	0.71	0.19	0.03	0.08
White	Strawberry River	1101001205	Cooper Creek-Strawberry Creek	High	0.66	0.21	0.03	0.10
White	War Eagle Creek	1101000106	War Eagle Creek	High	0.60	0.23	0.04	0.13
White	White River	802030105	Raft Creek-White River	Low	0.84	0.12	0.01	0.03
White	White River	802030206	Cow Lake Ditch-Bayou De View	Extirpated	0.84	0.12	0.01	0.03
White	White River	802030207	Buffalo Creek-Bayou De View	Extirpated	0.82	0.13	0.02	0.03
White	White River	802030305	Indian Bayou-White River	Low	0.82	0.13	0.02	0.04
White	White River	802030306	La Grue Bayou-White River	Extirpated	0.84	0.11	0.01	0.03
White	White River	802030307	Honey Locust Bayou-White River	Extirpated	0.79	0.14	0.02	0.05
White	White River	1101000103	Lake Sequoyah-White River	Extirpated	0.69	0.20	0.03	0.08
White	White River	1101000107	Beaver Lake-White River	Extirpated	0.80	0.14	0.02	0.05
White	White River	1101000301	Lake Taneycomo-White River	Extirpated	0.82	0.12	0.02	0.04
White	White River	1101000311	Bull Shoals Lake-White River	Extirpated	0.77	0.16	0.02	0.05
White	White River	1101000402	Hicks Creek-White River	Low	0.53	0.26	0.05	0.16
White	White River	1101000404	Lafferty Creek-White River	Unknown	0.59	0.24	0.04	0.13
White	White River	1101000406	Wolf Bayou-White River	Low	0.63	0.23	0.04	0.11
White	White River	1101000407	Salado Creek-White River	Extirpated	0.66	0.21	0.03	0.09
White	White River	1101001303	Willow Slough-White River	Low	0.77	0.16	0.02	0.05
White	White River	1101001305	Taylor Bay-White River	Low	0.84	0.12	0.01	0.03
White	White River	1101001306	Glaise Creek-White River	Extirpated	0.70	0.19	0.03	0.08

[1] HUC 511000206 Drakes Creek includes West Fork Drakes Creek from the listing rule populations

[2] HUC 603000201 Upper Paint Rock River includes Estill and Larkin Forks Paint Rock rivers and Hurricane Creek from the listing rule populations

[3] HUC 501000409 French Creek includes Conneautee Creek from the listing rule populations

Appendix B – Model Details

We modeled ordinal classifications for each watershed as a function of major threats to mussel persistence using a Bayesian approach to ordered probit regression, with a thresholded cumulative normal inverse-link function (Kruschke 2015, pp. 671-702). This model assumes the observed categories (i.e., extirpated, low, medium, high) reflect a latent continuous variable (i.e., probability of persistence) and calculates the probability of each ordinal outcome as the area under the normal curve between thresholds for that outcome. Specifically, the model can be expressed as follows:

$$[\boldsymbol{\theta}, \boldsymbol{\beta}, \sigma^2 \,|\boldsymbol{y}] \propto \prod_{i=1}^{n} [y_i \,|\boldsymbol{\beta}, \boldsymbol{x}_i, \sigma^2] \prod_{k=1}^{K} [\theta_k]\, [\sigma^2]\, [\boldsymbol{\beta}]$$

$$y_i \sim categorical(p_i)$$

$$p_i = \int_{-\infty}^{k=1} [z_i \mid \mu_i, \sigma^2]\, dz_i, \int_{k=1}^{k=2} [z_i \mid \mu_i, \sigma^2]\, dz_i, \ldots, \int_{k=K}^{k=\infty} [z_i \mid \mu_i, \sigma^2]\, dz_i$$

$$\mu_i = \beta_0 + \beta_1 x_1 + \ldots + \beta_n x_n$$

$$\boldsymbol{\beta} \sim normal(0, 50)$$

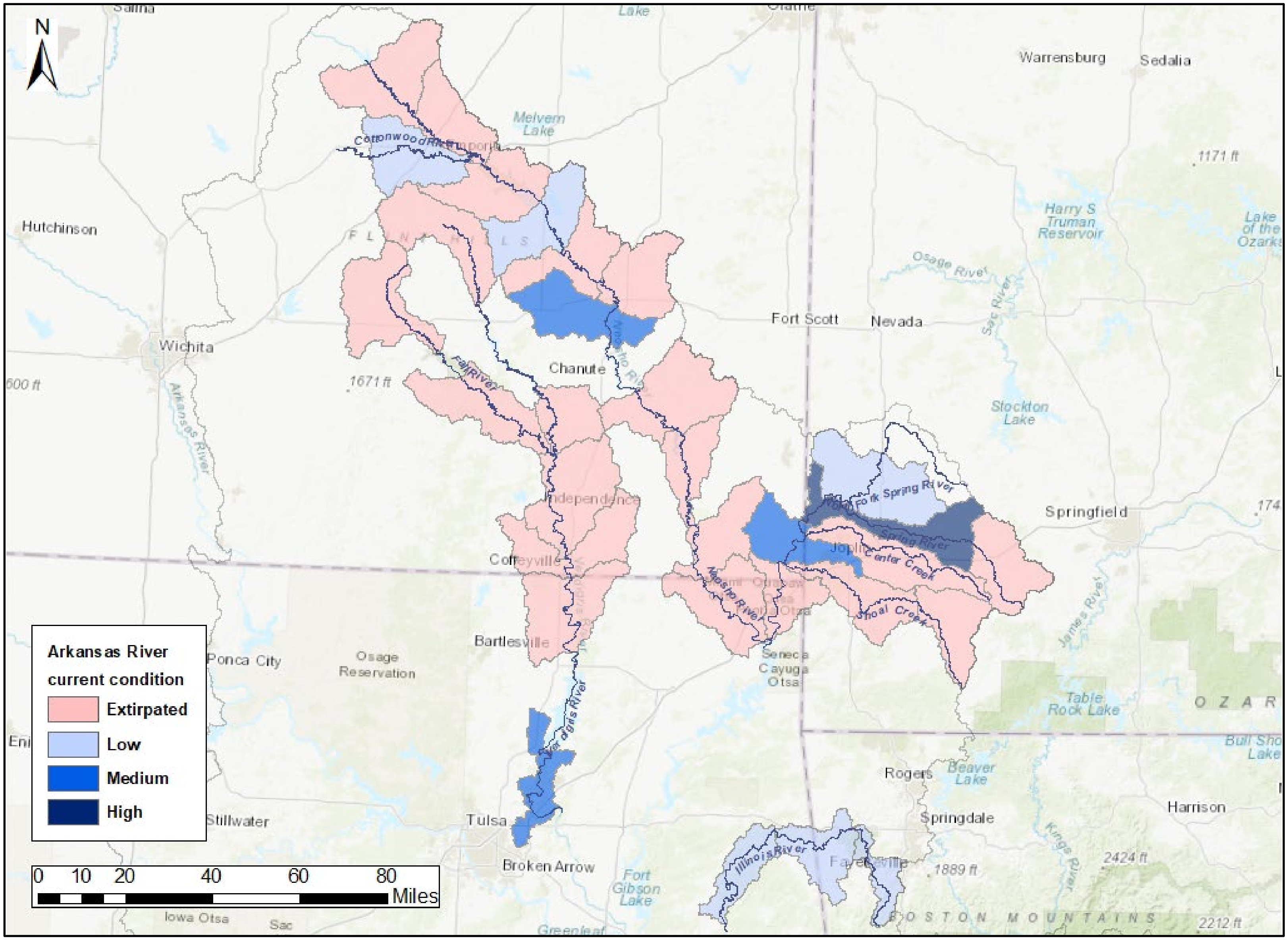
$$\sigma^2 \sim uniform(0, 100)$$

$$\theta_2 \sim uniform(0, 1)$$

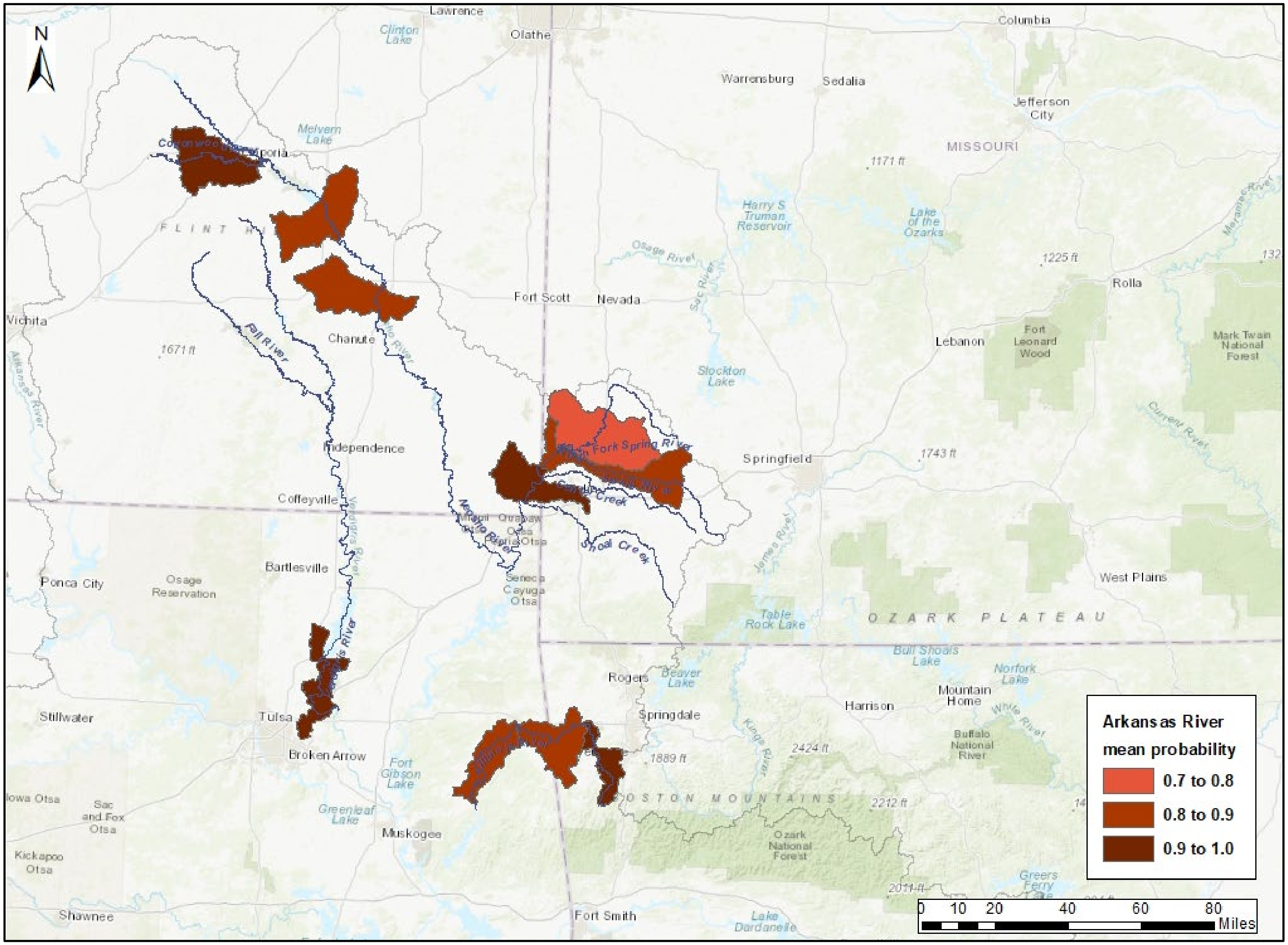
where y_i is the observed ordinal category, p_i is a vector of probabilities of the ordinal categories, and z_i represents the latent metric variable for the ith watershed. θ_k represents the kth ordered category, and θ_1 and θ_3 are fixed at 0 and 0.75, respectively, to handle issues of identifiability of model parameters. We modeled the mean of the underlying metric distribution (μ_i) as a linear function of land-use and climatic variables for each watershed and the variance was assumed constant across watersheds. We used parameter estimates from this linear function (**β**) to identify the relative effect of threats to mussel persistence.

We based model selection on the Deviance Information Criterion and posterior predictive checks using Bayesian p-values for the percentage of watersheds classified in each category (Hobbs and Hooten 2015, pp. 181-194, 216-217). We modeled the relative effects of a set of predictor variables with the program JAGS version 4.3.0 using the package rjags (Plummer 2019, entire). Markov Chain Monte Carlo samplers adapted over 20,000 iterations and posterior parameter estimates were based on three chains of length 30,000 with a burn-in of 20,000. Convergence was assessed using the Gelman-Rubin diagnostic and visual inspection of trace plots. Predictor variables were standardized to mean zero and unit variance to improve convergence and all priors were diffuse on the scale of the data. Watersheds with unknown current condition were treated as random variables and estimated during model fitting.

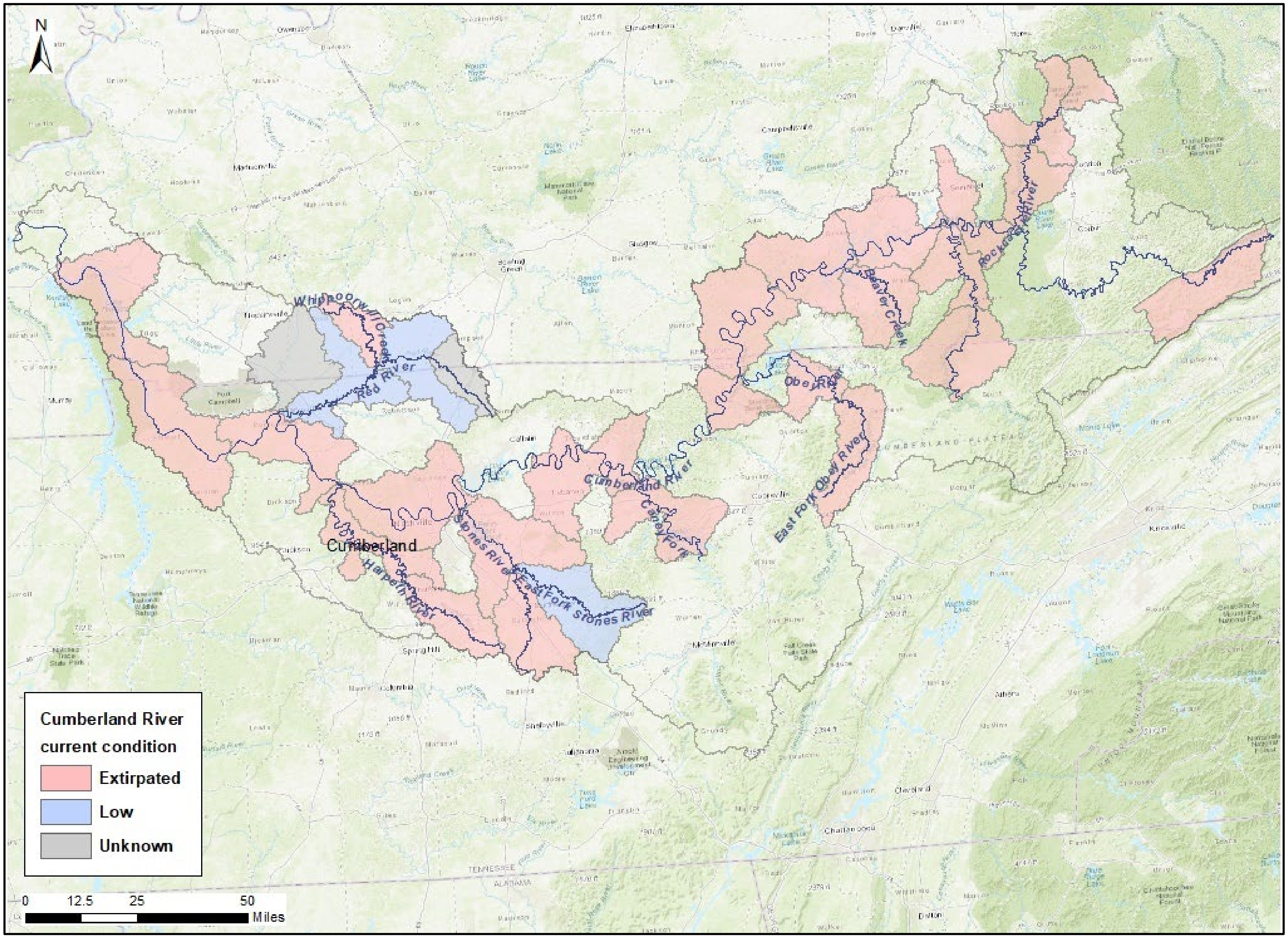
Appendix C. Current Condition for the Rabbitsfoot within each watershed by representation unit followed by mean probability of future risk of extirpation or low condition at 2050 for the Rabbitsfoot under four landuse and 12 climate scenarios for the same representation unit.



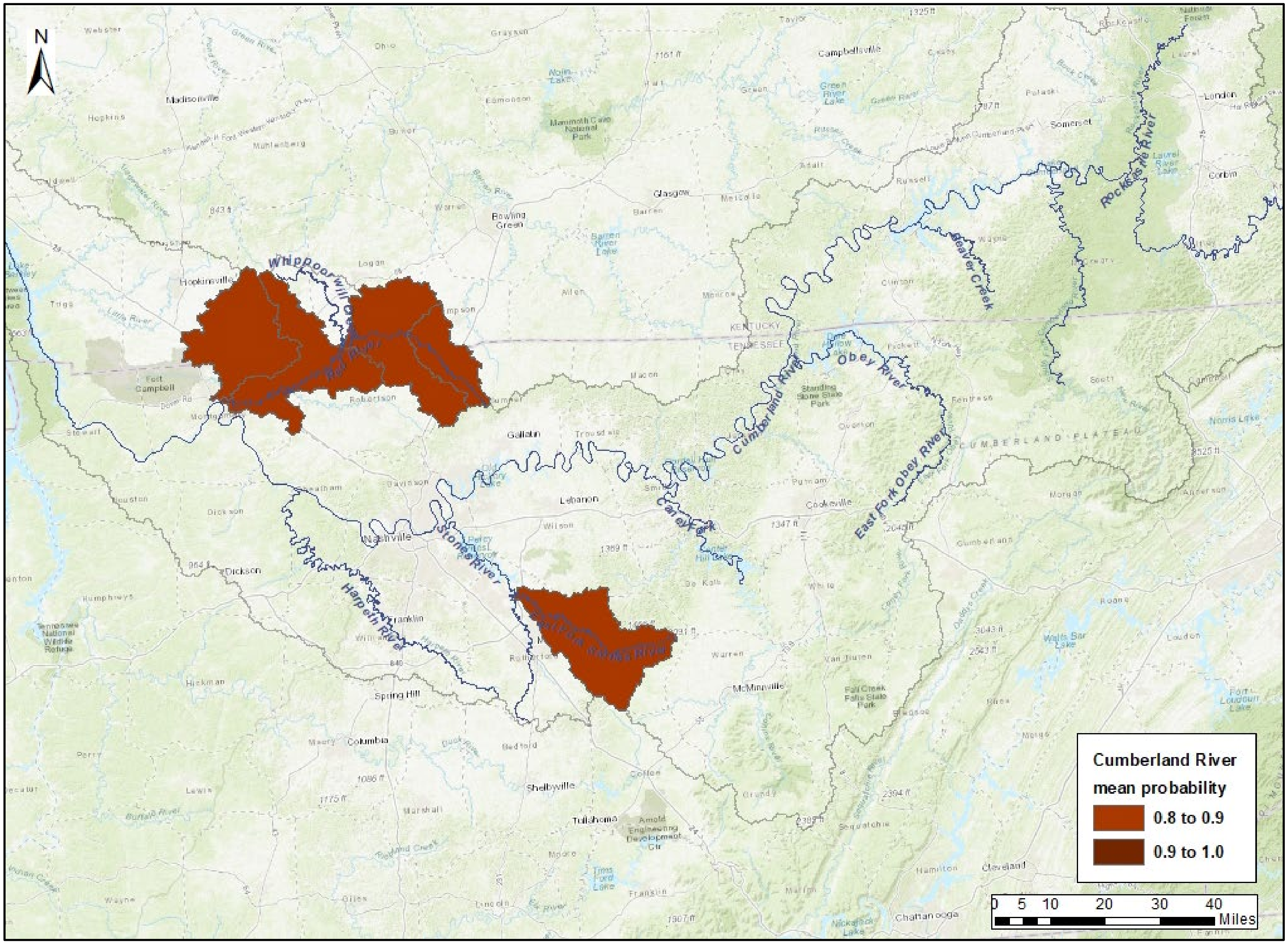
Current Condition for the Rabbitsfoot within each watershed of the Arkansas River representation unit.



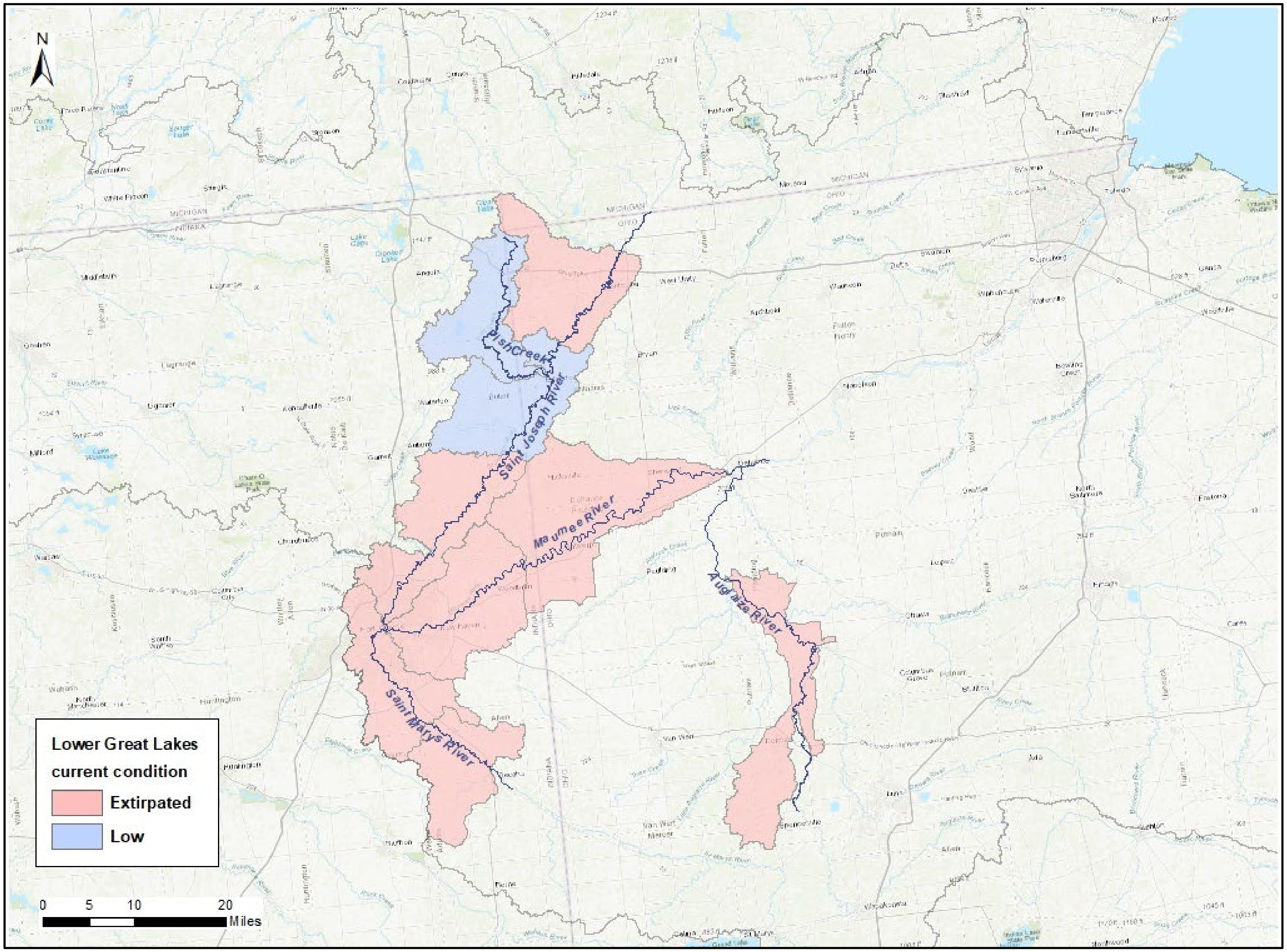
Mean probability of future risk of extirpation or low condition for watersheds within the Arkansas River representation unit.



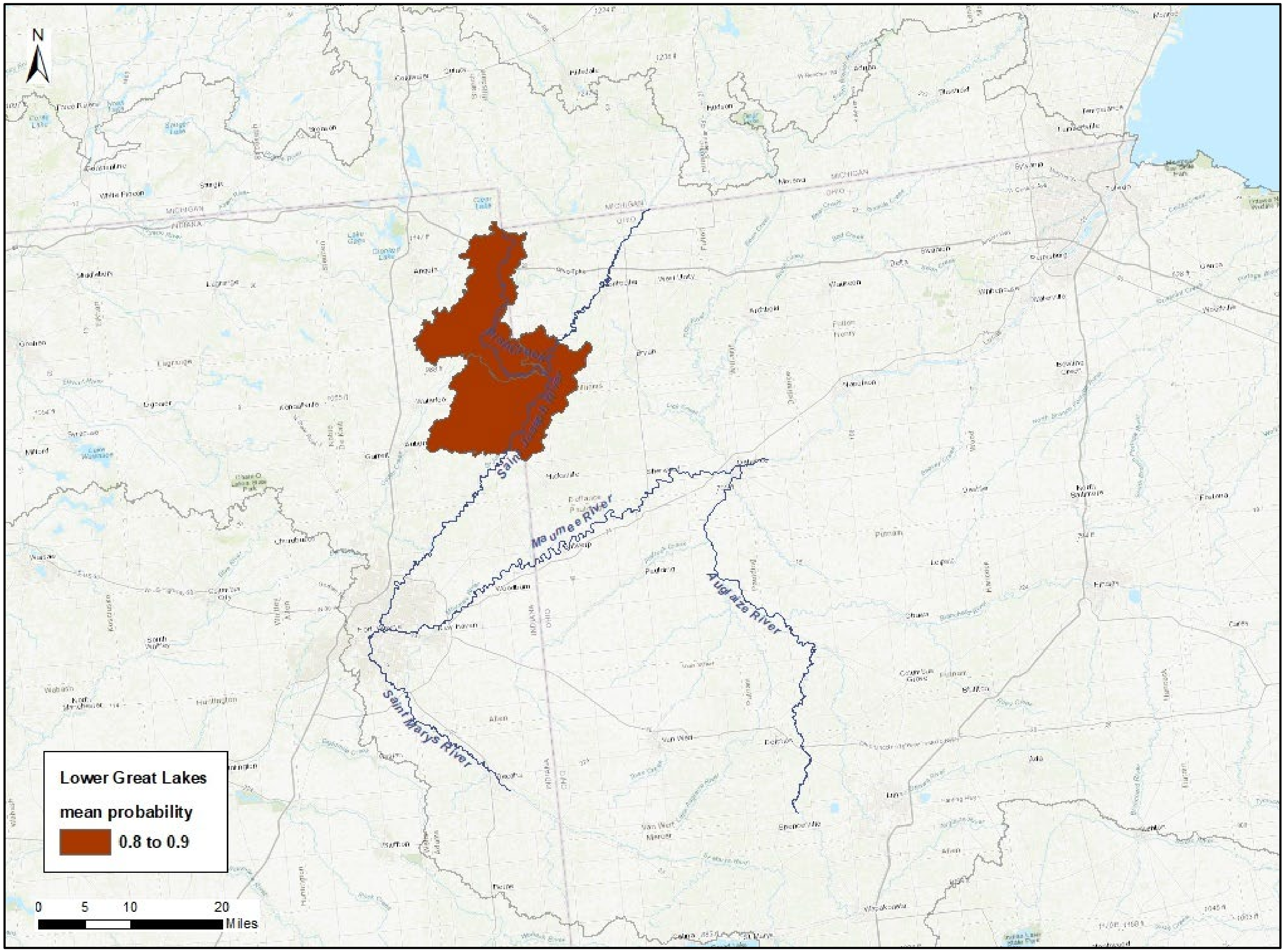
Current Condition for the Rabbitsfoot within each watershed of the Cumberland River representation unit.



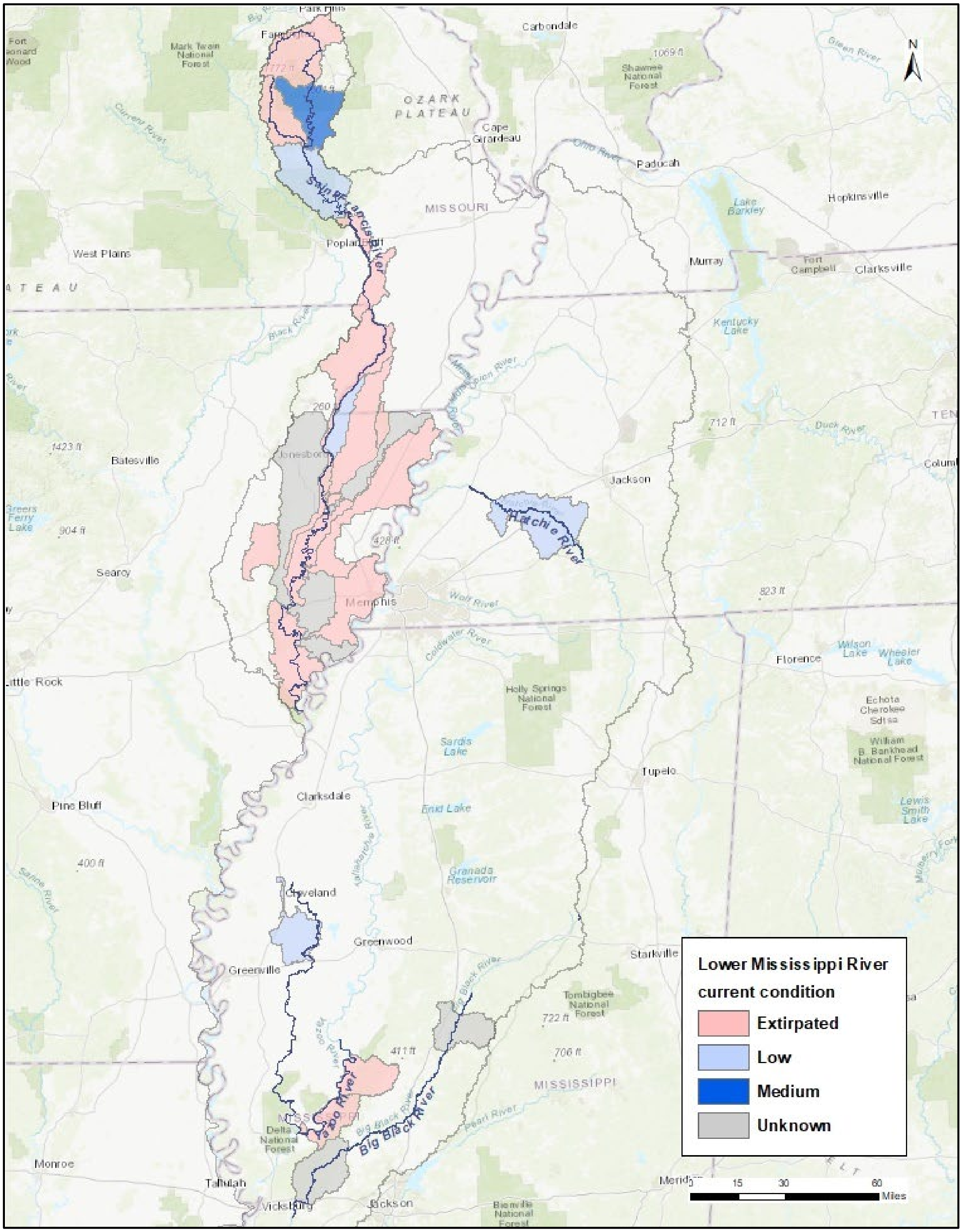
Mean probability of future risk of extirpation or low condition for watersheds within the Cumberland River representation unit.



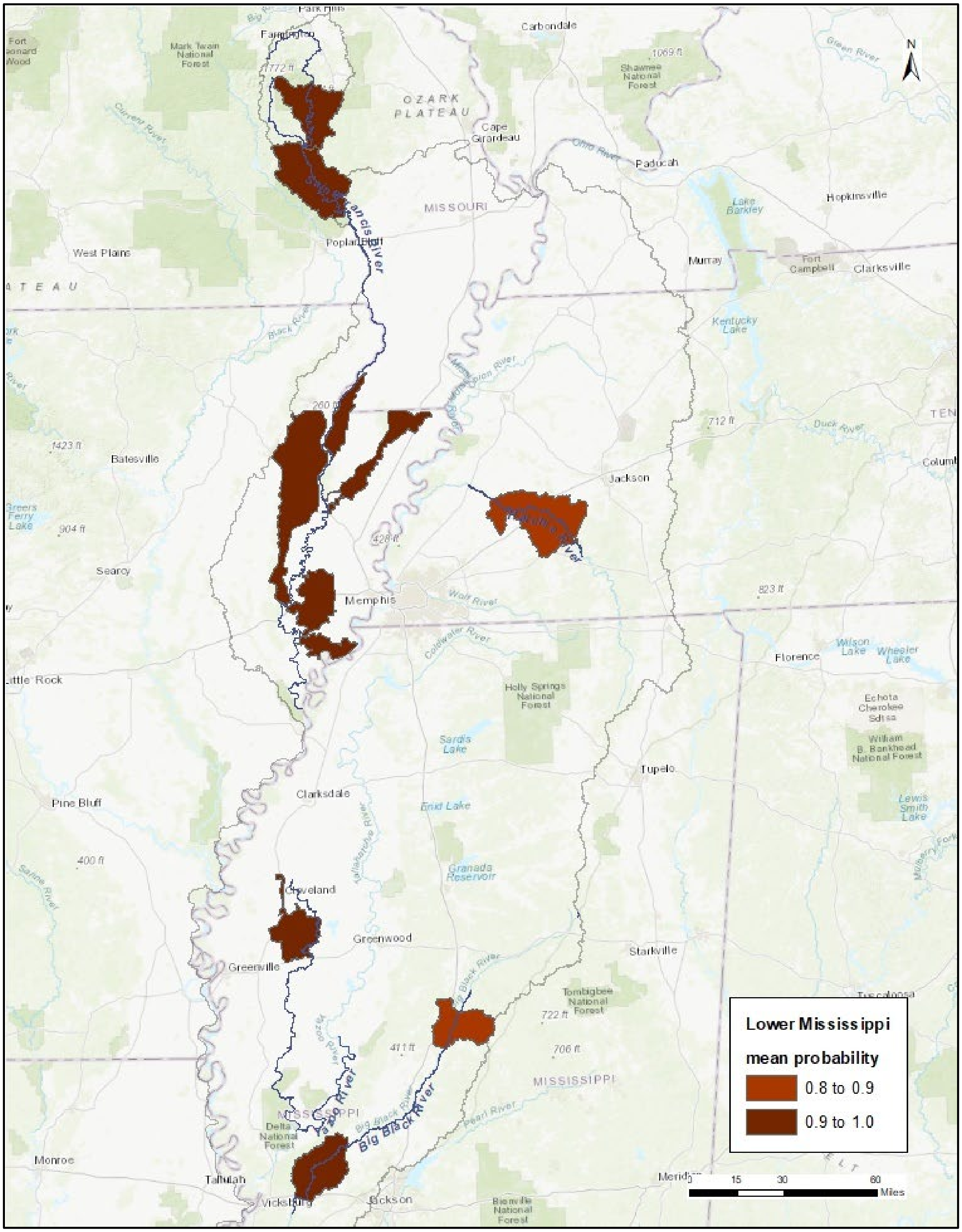
Current Condition for the Rabbitsfoot within each watershed of the Lower Great Lakes representation unit.



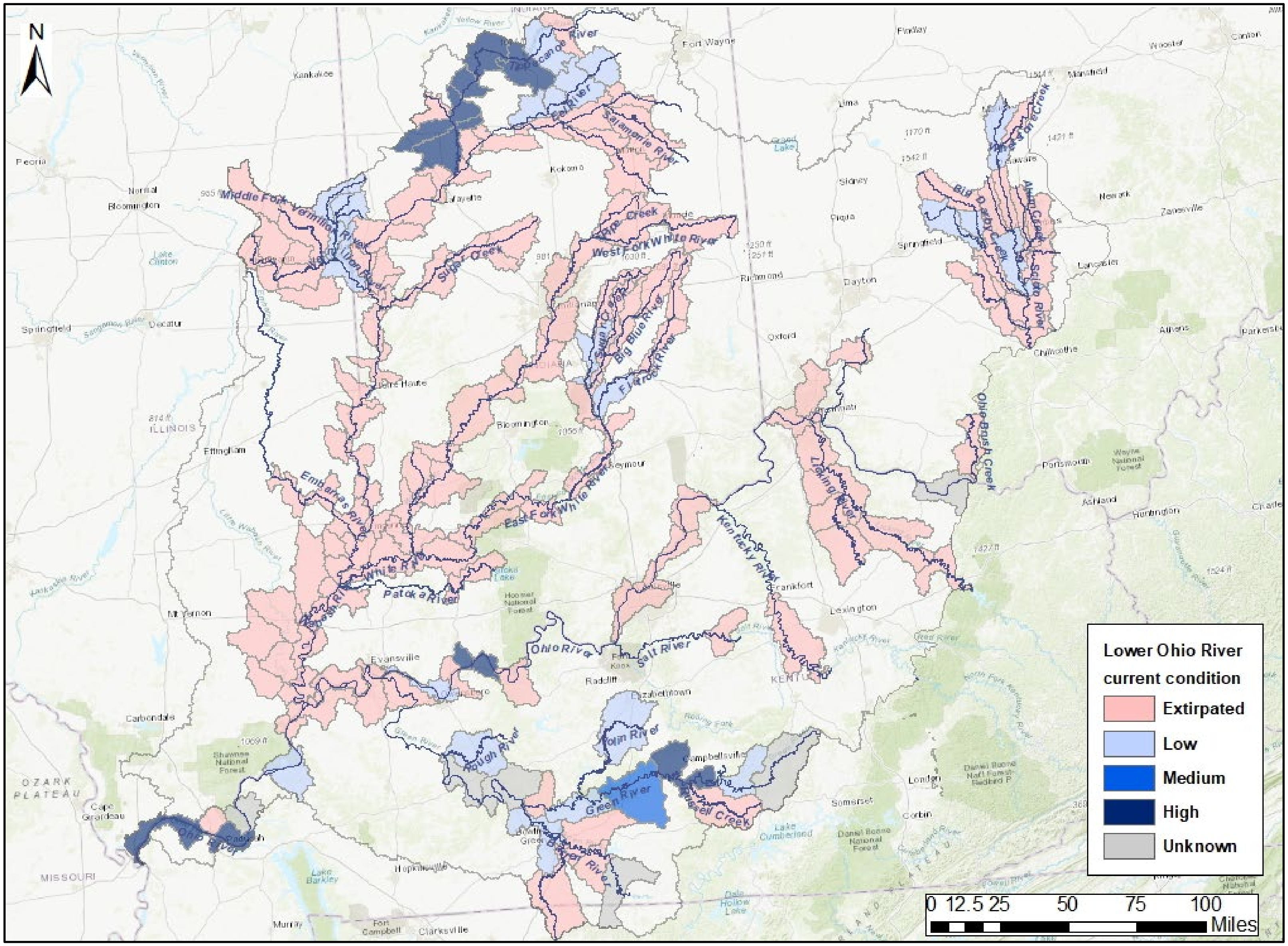
Mean probability of future risk of extirpation or low condition for watersheds within the Lower Great Lakes representation unit.



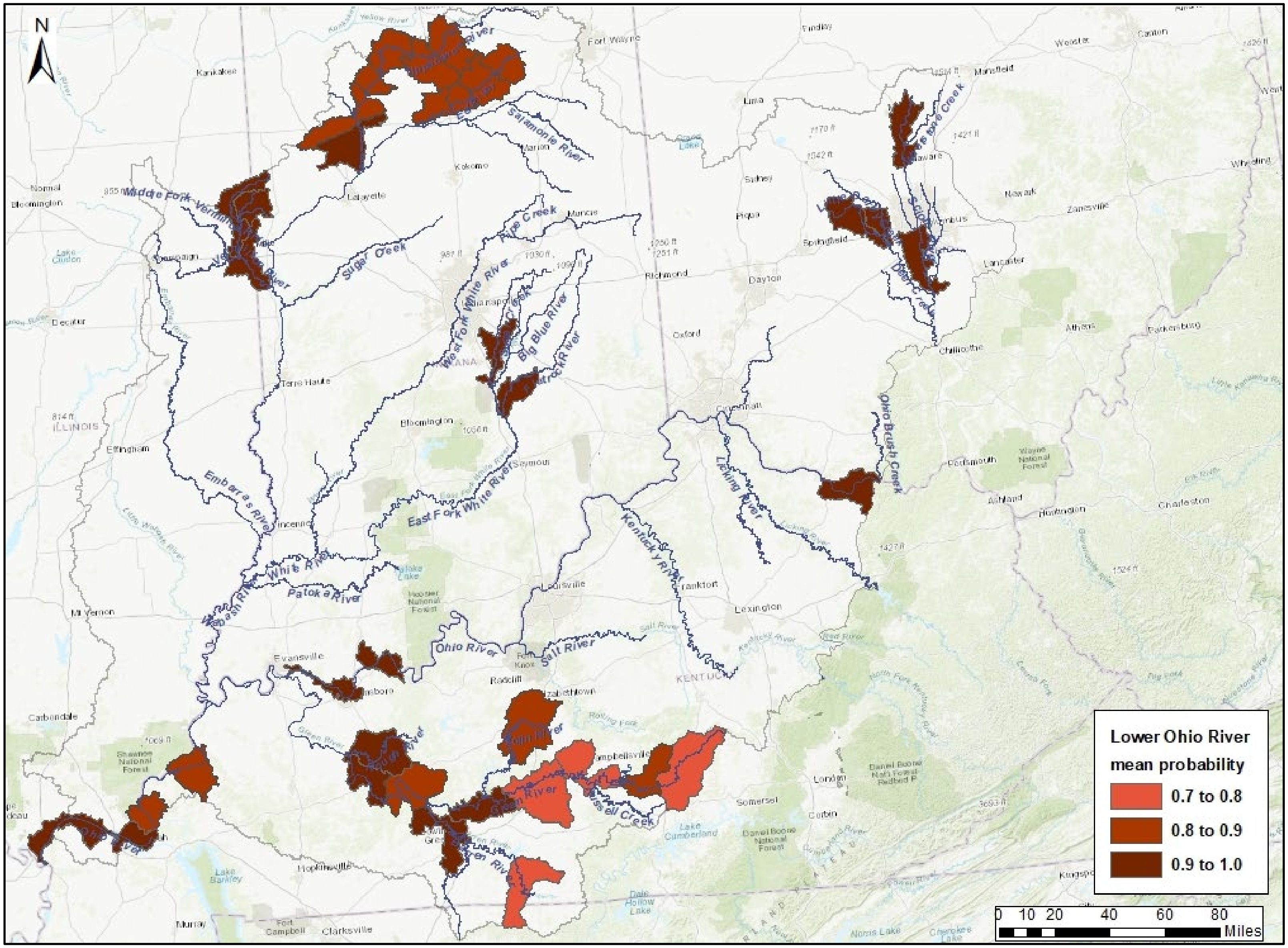
Current Condition for the Rabbitsfoot within each watershed of the Lower Mississippi River representation unit.



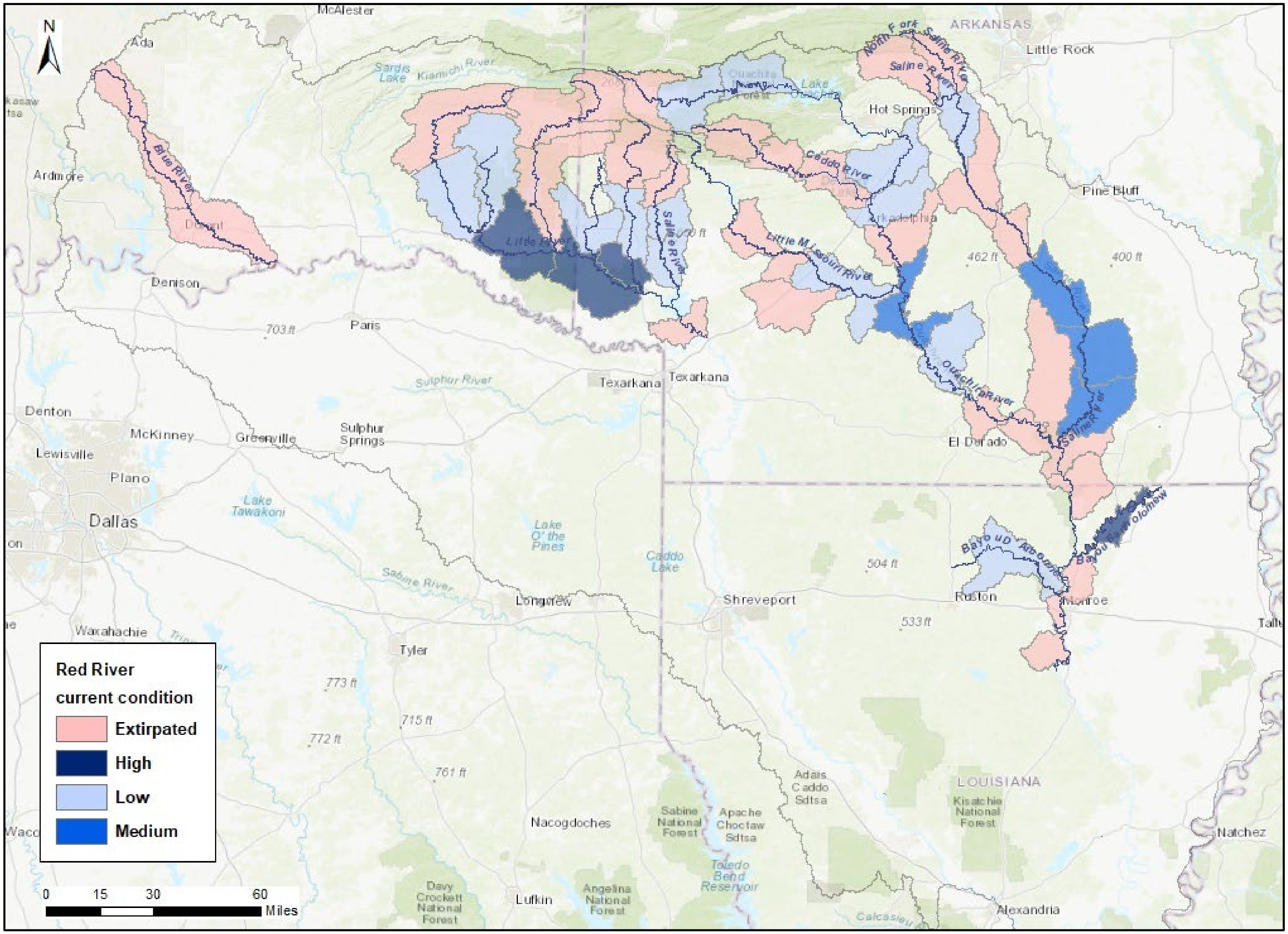
Mean probability of future risk of extirpation or low condition for watersheds within the Lower Mississippi River representation unit.



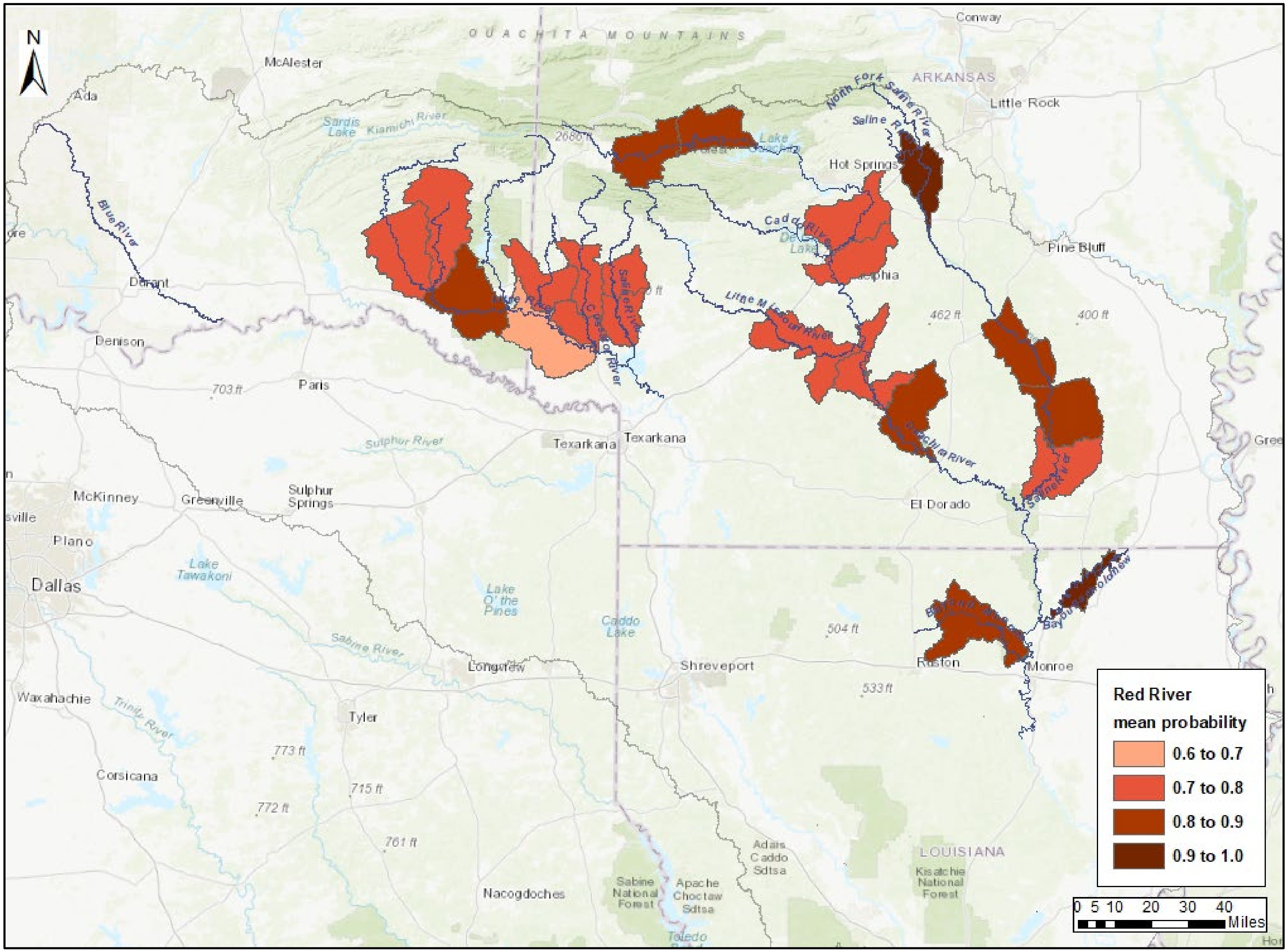
Current Condition for the Rabbitsfoot within each watershed of the Lower Ohio River representation unit.



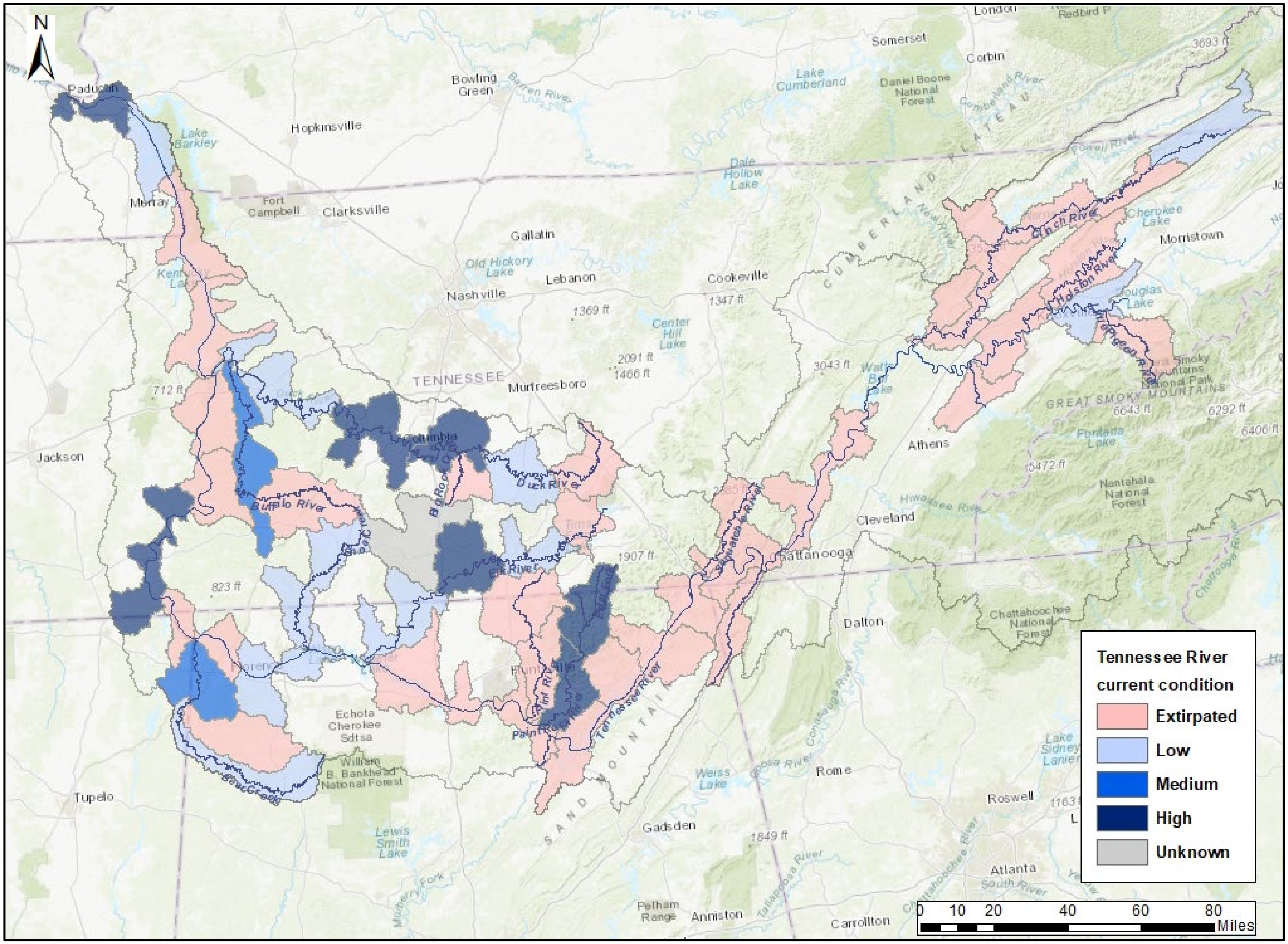
Mean probability of future risk of extirpation or low condition for watersheds within the Lower Ohio River representation unit.



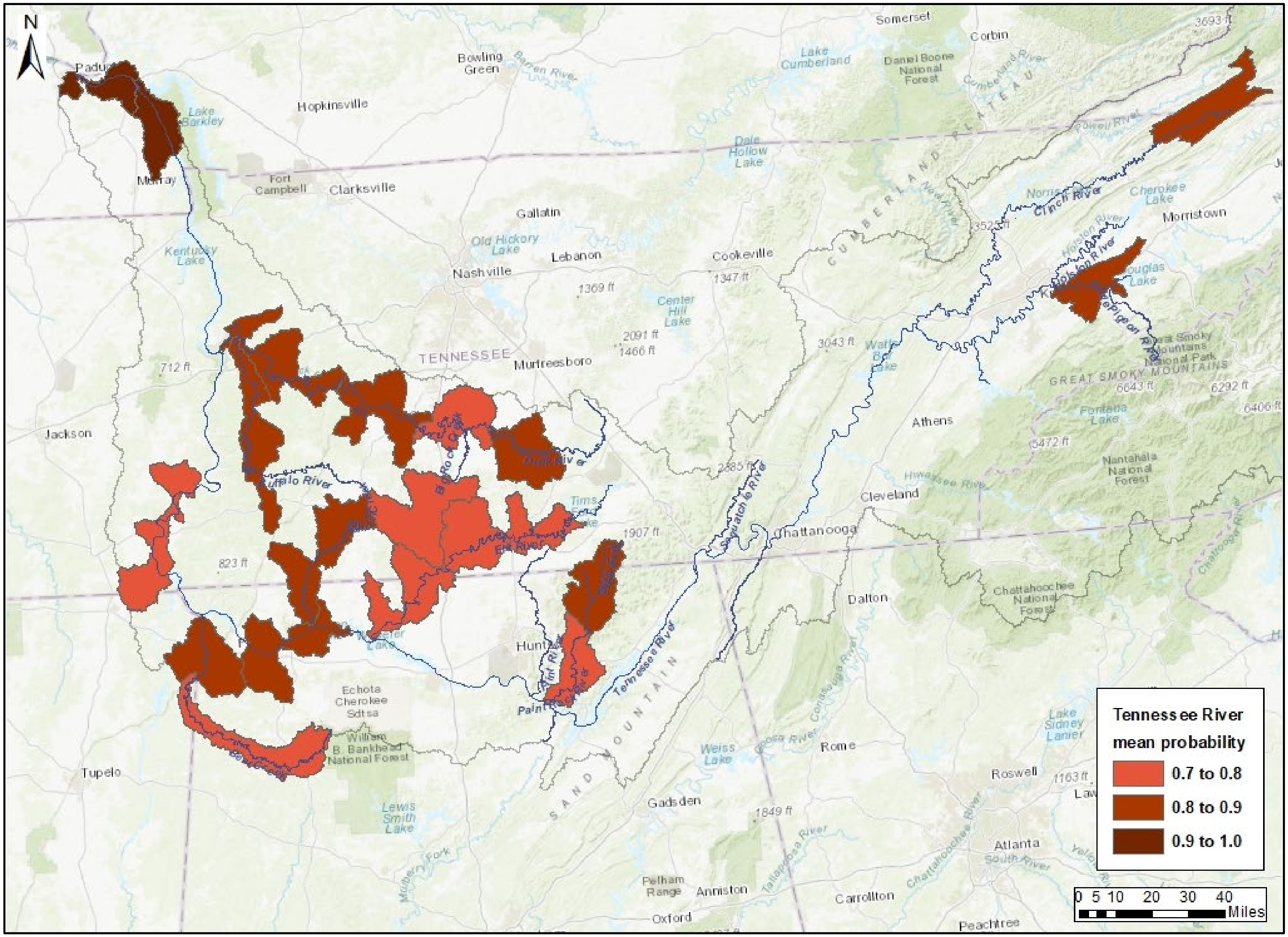
Current Condition for the Rabbitsfoot within each watershed of the Red River representation unit.



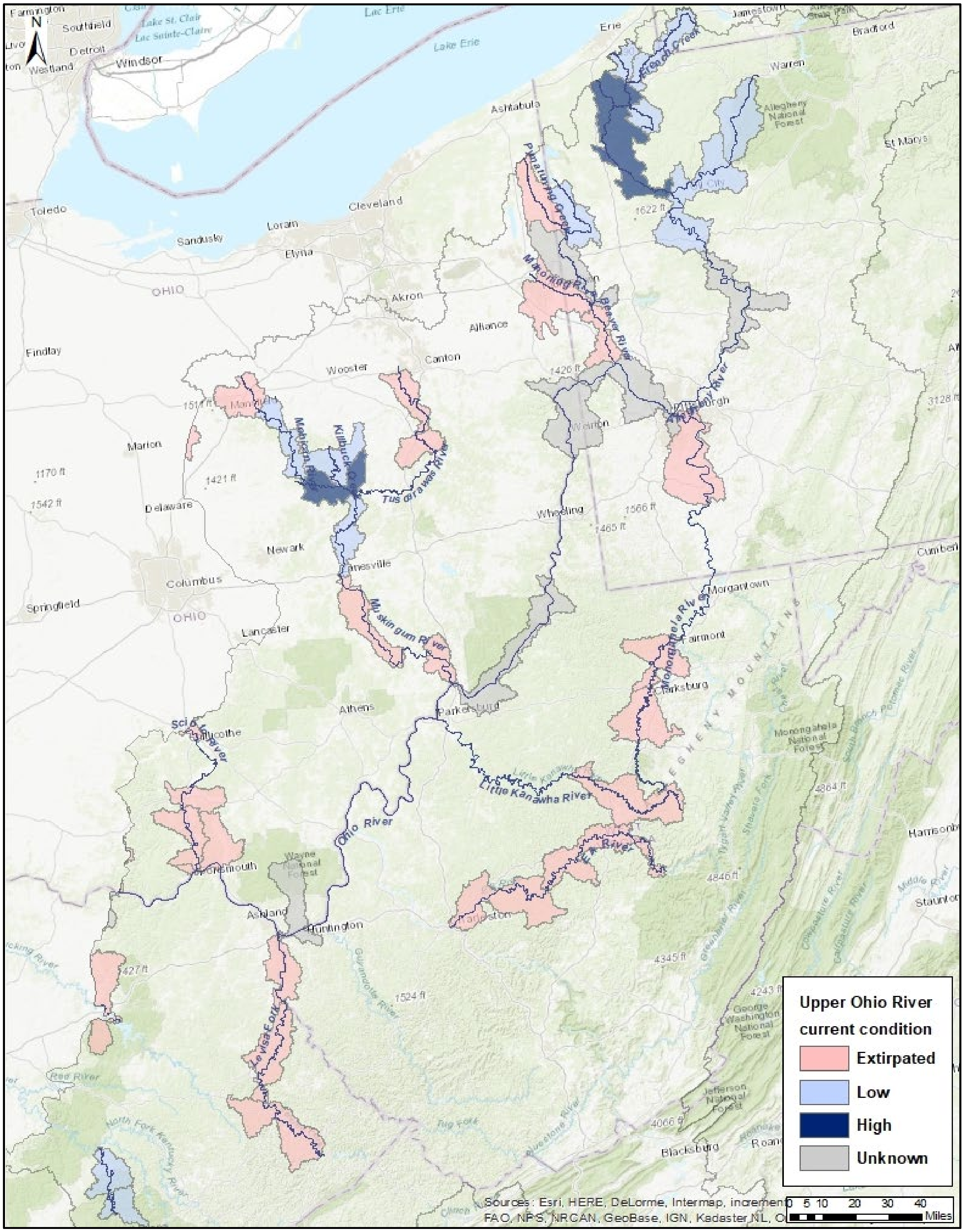
Mean probability of future risk of extirpation or low condition for watersheds within the Red River representation unit.



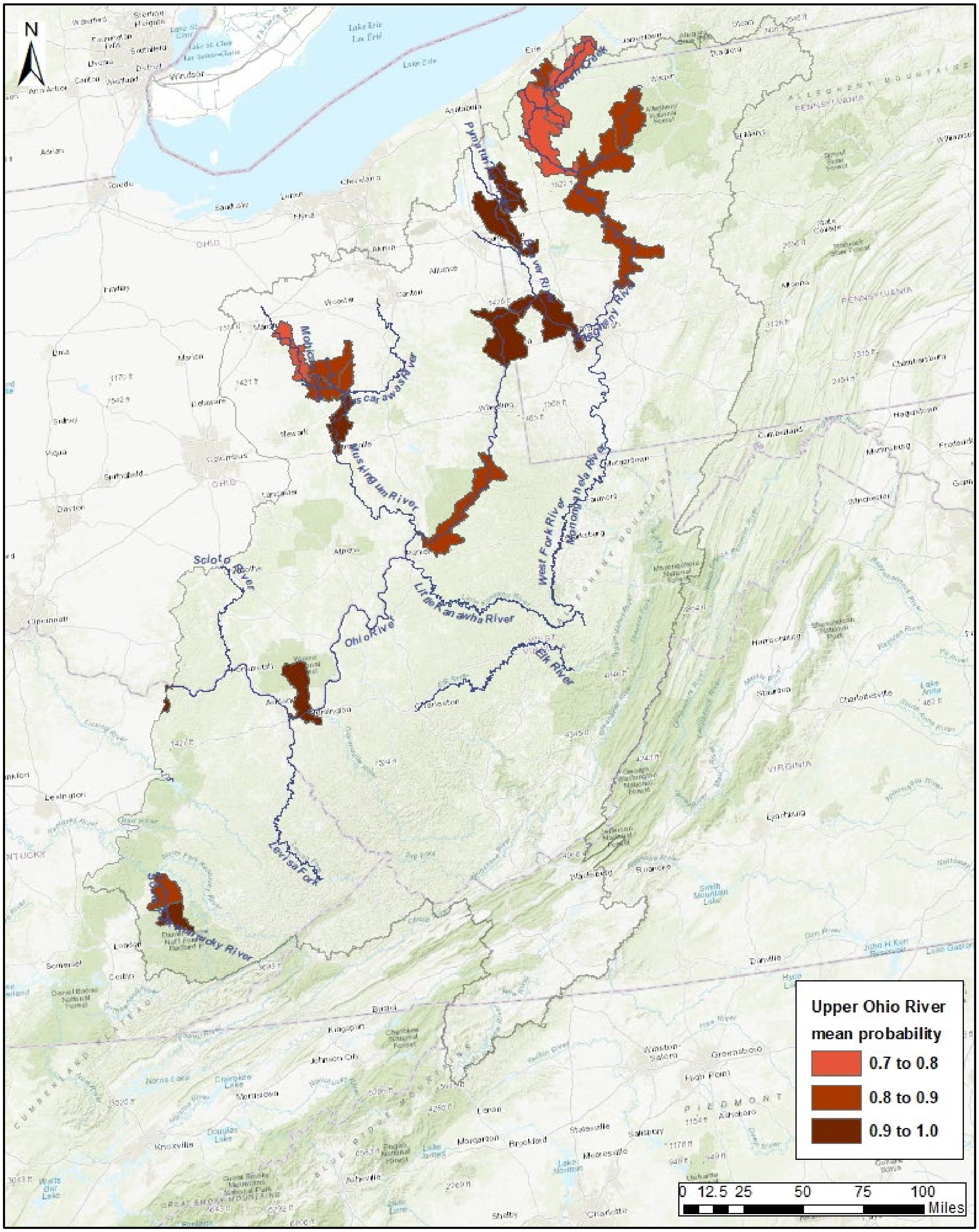
Current Condition for the Rabbitsfoot within each watershed of the Tennessee River representation unit.



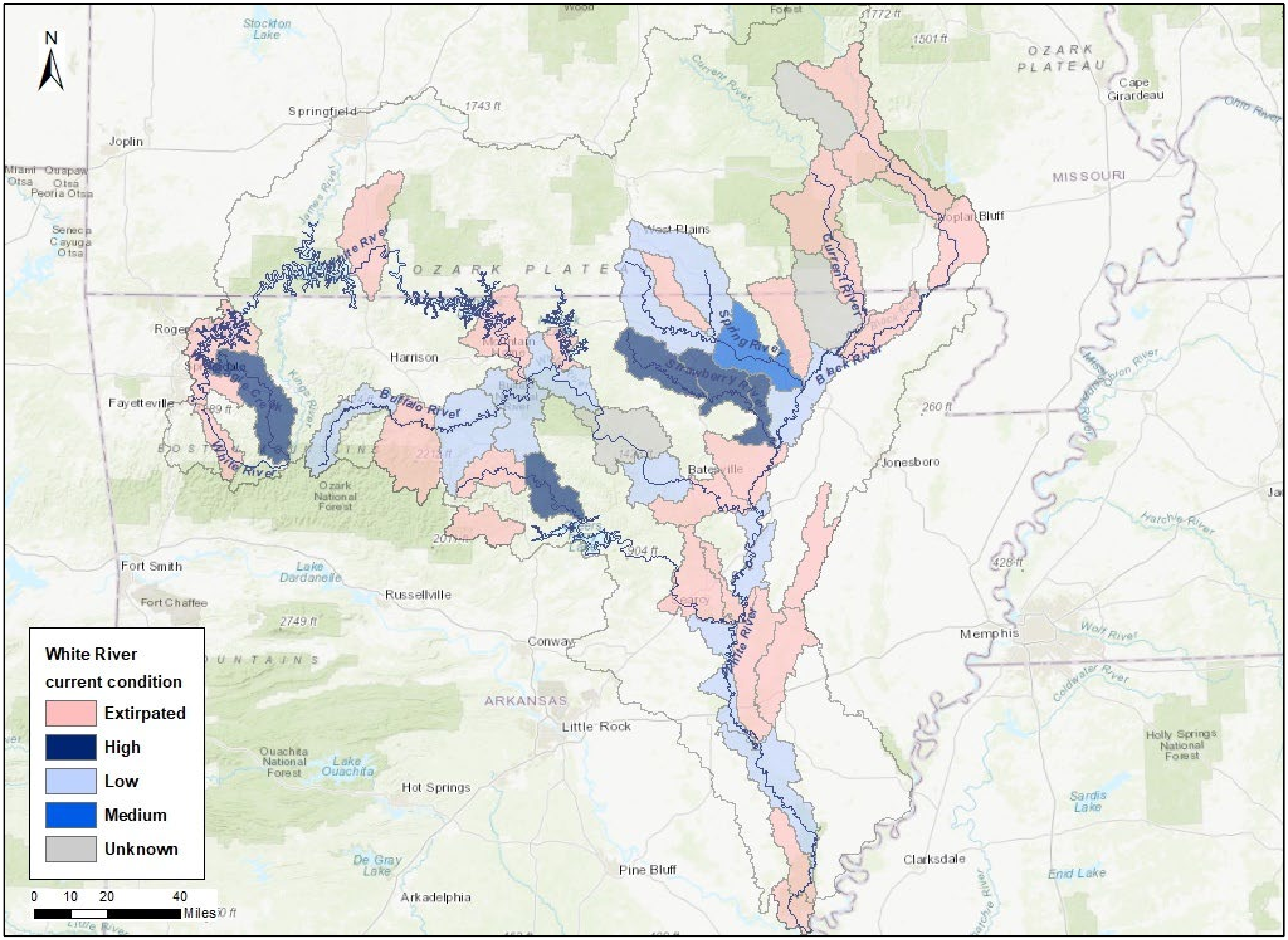
Mean probability of future risk of extirpation or low condition for watersheds within the Tennessee River representation unit.



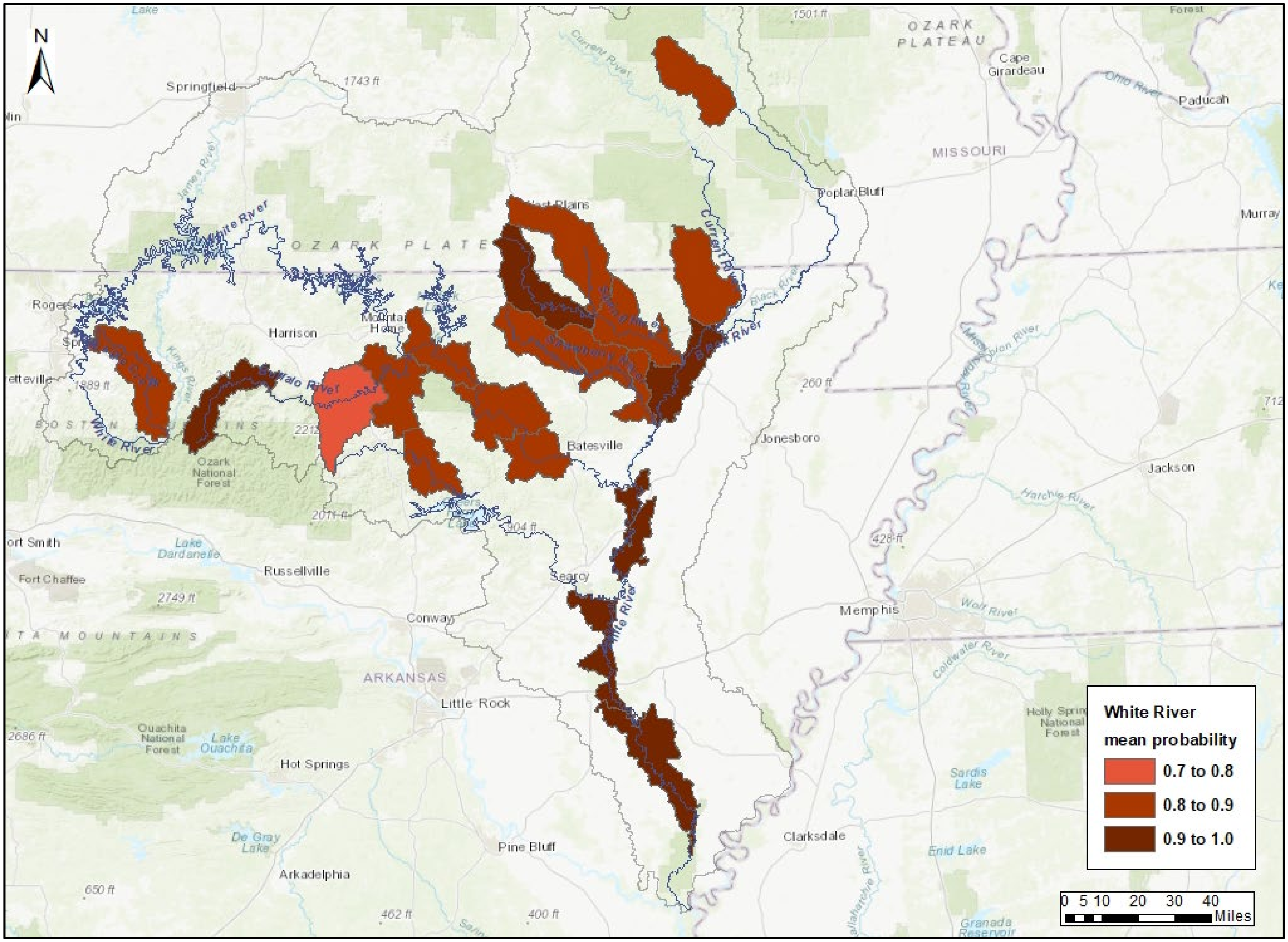
Current Condition for the Rabbitsfoot within each watershed of the Upper Ohio River representation unit.



Mean probability of future risk of extirpation or low condition for watersheds within the Upper Ohio River representation unit.



Current Condition for the Rabbitsfoot within each watershed of the White River representation unit.



Mean probability of future risk of extirpation or low condition for watersheds within the White River representation unit.

Appendix D. An example of an optimal portfolio of watersheds in which to target conservation actions based on the objectives of maximizing the probability of being classified in medium or high condition at 2050 and the distance between occupied watersheds. Watersheds are prioritized according to the three-part approach: (1) high condition; (2) habitat improvement and (3) reintroduction.

Priority	Representation Unit	Listing Rule Population	Number	Name	Current Condition	Critical Habitat
High Condition	Arkansas	Spring River	1107020705	White Oak Creek-Spring River	High	yes
High Condition	Lower Ohio	Green River	511000107	Big Brush Creek-Green River	High	yes
High Condition	Lower Ohio	Tippecanoe River	512010605	Eddy Creek-Tippecanoe River	High	no
High Condition	Lower Ohio	Tippecanoe River	512010606	Bruce Lake Outlet-Tippecanoe River	High	yes
High Condition	Lower Ohio	Tippecanoe River	512010609	Dickey Creek-Tippecanoe River	High	yes
High Condition	Lower Ohio	Tippecanoe River	512010612	Honey Creek-Tippecanoe River	High	yes
High Condition	Lower Ohio	Tippecanoe River	512010613	Tippecanoe River	High	yes
High Condition	Lower Ohio	Ohio River	514020107	Lead Creek-Ohio River	High	no
High Condition	Lower Ohio	Ohio River	514020601	Mud Creek-Ohio River	High	yes
High Condition	Lower Ohio	Ohio River	514020607	Redstone Creek-Ohio River	High	yes
High Condition	Red	Bayou Bartholomew	804020510	Cypress Bayou-Bayou Bartholomew	High	no
High Condition	Red	Little River	1114010704	Lower Little River	High	yes
High Condition	Red	Little River	1114010903	Lower Rolling Fork	High	yes
High Condition	Red	Cossatot River	1114010906	Lower Cossatot River	High	yes
High Condition	Tennessee	Paint Rock River	603000201	Upper Paint Rock River	High	yes
High Condition	Tennessee	Paint Rock River	603000202	Lower Paint Rock River	High	yes
High Condition	Tennessee	Elk River	603000309	Bradshaw Creek-Elk River	High	no
High Condition	Tennessee	Tennessee River	604000105	Chambers Creek-Tennessee River	High	yes
High Condition	Tennessee	Duck River	604000207	Flat Creek-Duck River	High	yes
High Condition	Tennessee	Duck River	604000305	Catheys Creek-Duck River	High	yes
High Condition	Tennessee	Tennessee River	604000605	Island Creek-Tennessee River	High	yes
High Condition	Upper Ohio	French Creek	501000409	French Creek	High	yes
High Condition	Upper Ohio	Walhonding River	504000309	Mill Creek-Walhonding River	High	yes
High Condition	White	War Eagle Creek	1101000106	War Eagle Creek	High	no
High Condition	White	Strawberry River	1101001202	Little Strawberry River-Strawberry River	High	yes
High Condition	White	Strawberry River	1101001203	North Big Creek-Strawberry River	High	yes
High Condition	White	Strawberry River	1101001205	Cooper Creek-Strawberry Creek	High	yes
Habitat Improvement	Arkansas	Neosho River	1107020401	Wolf Creek-Neosho River	Low	no
Habitat Improvement	Arkansas	Neosho River	1107020404	Owl Creek-Neosho River	Medium	yes
Habitat Improvement	Arkansas	North Fork Spring River	1107020703	Outlet North Fork Spring River	Low	no

Priority	**Representation Unit**	**Listing Rule Population**	**Number**	**Name**	**Current Condition**	**Critical Habitat**
Habitat Improvement	Arkansas	Spring River	1107020709	Shawnee Creek-Spring River	Medium	yes
Habitat Improvement	Arkansas	Illinois River	1111010306	Upper Illinois River	Low	no
Habitat Improvement	Cumberland	East Fork Stones River	513020301	East Fork Stones River	Low	no
Habitat Improvement	Cumberland	Red River	513020602	South Fork Red River-Red River	Low	yes
Habitat Improvement	Cumberland	Red River	513020607	Elk Fork-Red River	Low	yes
Habitat Improvement	Lower Great Lakes	Fish Creek	410000304	Fish Creek	Low	yes
Habitat Improvement	Lower Great Lakes	St. Joseph River	410000305	Sol Shank Ditch-Saint Joseph River	Low	yes
Habitat Improvement	Lower Mississippi	Hatchie River	801020805	Cypress Creek-Hatchie River	Low	no
Habitat Improvement	Lower Mississippi	St. Francis River	802020204	Twelvemile Creek-Saint Francis River	Medium	yes
Habitat Improvement	Lower Mississippi	St. Francis River	802020205	Otter Creek-Saint Francis River	Low	yes
Habitat Improvement	Lower Mississippi	St. Francis River	802020308	Little Slough Ditch	Low	no
Habitat Improvement	Lower Mississippi	Big Sunflower River	803020705	Porter Bayou-Big Sunflower River	Low	yes
Habitat Improvement	Lower Ohio	Little Darby Creek	506000120	Little Darby Creek	Low	yes
Habitat Improvement	Lower Ohio	Green River	511000103	Casey Creek-Green River	Low	yes
Habitat Improvement	Lower Ohio	Green River	511000108	Ugly Creek-Green River	Medium	yes
Habitat Improvement	Lower Ohio	Green River	511000113	Beaverdam Creek-Green River	Low	yes
Habitat Improvement	Lower Ohio	Middle Branch North Fork Vermilion River	512010907	Jordan Creek-Middle Branch	Low	yes
Habitat Improvement	Red	Ouachita River	804010201	De Roche Creek-Ouachita River	Low	yes
Habitat Improvement	Red	Ouachita River	804010205	L'Eau Frais Creek-Ouachita River	Low	yes
Habitat Improvement	Red	Ouachita River	804010209	Ecore Fabre Bayou-Ouachita River	Medium	yes
Habitat Improvement	Red	Saline River (Ouachita River)	804020405	Snake Creek-Saline River	Medium	yes
Habitat Improvement	Red	Little River	1114010703	Middle Little River	Low	yes
Habitat Improvement	Tennessee	Elk River	603000307	Norris Creek-Elk River	Low	no
Habitat Improvement	Tennessee	Elk River	603000404	Anderson Creek-Elk River	Low	no
Habitat Improvement	Tennessee	Bear Creek	603000601	Upper Bear Creek	Low	yes
Habitat Improvement	Tennessee	Bear Creek	603000603	Lower Bear Creek	Medium	yes
Habitat Improvement	Tennessee	Duck River	604000309	Blue Creek-Duck River	Low	yes

Priority	Representation Unit	Listing Rule Population	Number	Name	Current Condition	Critical Habitat
Habitat Improvement	Upper Ohio	Allegheny River	501000309	Lower Allegheny River	Low	yes
Habitat Improvement	Upper Ohio	French Creek	501000403	Wheeler Creek-French Creek	Low	yes
Habitat Improvement	Upper Ohio	LeBoeuf Creek	501000404	LeBoeuf Creek	Low	yes
Habitat Improvement	Upper Ohio	Muddy Creek	501000405	Muddy Creek	Low	yes
Habitat Improvement	Upper Ohio	Mohican River	504000208	Mohican River	Low	yes
Habitat Improvement	White	Buffalo River	1101000502	Headwaters Buffalo River	Low	yes
Habitat Improvement	White	Buffalo River	1101000504	Bear Creek-Buffalo River	Low	yes
Habitat Improvement	White	Buffalo River	1101000505	Outlet Buffalo River	Low	yes
Habitat Improvement	White	Spring River	1101001005	Outlet Spring River	Medium	yes
Habitat Improvement	White	White River	1101001303	Willow Slough-White River	Low	yes
Reintroduction	Arkansas	Verdigris River	1107010303	California Creek-Verdigris River	Extirpated	no
Reintroduction	Arkansas	Neosho River	1107020402	Crooked Creek-Neosho River	Extirpated	no
Reintroduction	Arkansas	Neosho River	1107020403	Deer Creek-Neosho River	Extirpated	yes
Reintroduction	Arkansas	Neosho River	1107020501	Canville Creek-Neosho River	Extirpated	no
Reintroduction	Arkansas	Spring River	1107020701	Headwaters Spring River	Extirpated	no
Reintroduction	Cumberland	Rockcastle River	513010205	Sinking Creek-Rockcastle River	Extirpated	no
Reintroduction	Cumberland	Cumberland River	513010307	Marrowbone Creek-Cumberland River	Extirpated	no
Reintroduction	Cumberland	Big South Fork Cumberland River	513010405	Roaring Paunch Creek-Big South Fork Cumberland River	Extirpated	no
Reintroduction	Cumberland	Red River	513020601	Summers Branch-Red River	Unknown	no
Reintroduction	Cumberland	Whippoorwill Creek	513020603	Whippoorwill Creek	Extirpated	yes
Reintroduction	Lower Great Lakes	St. Joseph River	410000303	Nettle Creek-Saint Joseph River	Extirpated	no
Reintroduction	Lower Great Lakes	St. Mary's River	410000405	Nickelsen Creek-Saint Marys River	Extirpated	no
Reintroduction	Lower Great Lakes	St. Mary's River	410000406	Saint Marys River	Extirpated	no
Reintroduction	Lower Great Lakes	Maumee River	410000502	Gordon Creek-Maumee River	Extirpated	no
Reintroduction	Lower Great Lakes	Auglaize River	410000709	Jennings Creek-Auglaize River	Extirpated	no
Reintroduction	Lower Mississippi	St. Francis River	802020305	Wilhelmina Cutoff-Saint Francis River	Extirpated	no
Reintroduction	Lower Mississippi	St. Francis River	802020502	Prairie Creek-L'Anguille River	Extirpated	no

Priority	Representation Unit	Listing Rule Population	Number	Name	Current Condition	Critical Habitat
Reintroduction	Lower Mississippi	Yazoo River	803020607	Piney Creek-Yazoo River	Extirpated	no
Reintroduction	Lower Mississippi	Big Black River	806020107	Apookta Creek-Big Black River	Unknown	no
Reintroduction	Lower Ohio	Licking River	510010111	Johnson Creek-Licking River	Extirpated	no
Reintroduction	Lower Ohio	Green River	511000101	South Fork Green River-Green River	Unknown	no
Reintroduction	Lower Ohio	Russell Creek	511000104	Russell Creek	Extirpated	no
Reintroduction	Lower Ohio	Barren River	511000204	Bays Fork-Barren River	Extirpated	no
Reintroduction	Red	Ouachita River	804010206	Cypress Creek-Ouachita River	Extirpated	no
Reintroduction	Red	Little Missouri River	804010305	Prairie Creek-Little Missouri River	Extirpated	no
Reintroduction	Red	Little Missouri River	804010306	Terre Rouge Creek	Extirpated	no
Reintroduction	Red	Saline River (Ouachita River)	804010308	Lost Creek-Saline River	Extirpated	no
Reintroduction	Red	Saline River (Ouachita River)	804010309	Grisly Creek-Saline River	Extirpated	no
Reintroduction	Red	Cossatot River	1114010904	Headwater Cossatot River	Extirpated	no
Reintroduction	Red	Saline River (Little River)	1114010907	Headwaters Saline River	Extirpated	no
Reintroduction	Tennessee	Little Pigeon River	601010703	Little Pigeon River	Extirpated	no
Reintroduction	Tennessee	Tennessee River	603000102	Widows Creek-Tennessee River	Extirpated	no
Reintroduction	Tennessee	Flint River	603000203	Upper Flint River	Extirpated	no
Reintroduction	Tennessee	Elk River	603000402	Richland Creek	Unknown	no
Reintroduction	Tennessee	Bear Creek	603000602	Cedar Creek	Extirpated	no
Reintroduction	Upper Ohio	Pymatuning Creek	503010203	Pymatuning Creek	Extirpated	no
Reintroduction	Upper Ohio	Little Kanawha River	503020303	Upper Little Kanawha River	Extirpated	no
Reintroduction	Upper Ohio	Elk River	505000706	Middle Elk River	Extirpated	no
Reintroduction	Upper Ohio	Elk River	505000709	Lower Elk River	Extirpated	no
Reintroduction	Upper Ohio	Licking River	510010108	Fox Creek-Licking River	Extirpated	no
Reintroduction	White	White River	1101000407	Salado Creek-White River	Extirpated	yes
Reintroduction	White	Buffalo River	1101000503	Richland Creek-Buffalo River	Extirpated	yes
Reintroduction	White	Black River	1101000903	Curia Creek-Black River	Extirpated	no
Reintroduction	White	Spring River	1101001104	Lower Eleven Point River	Extirpated	no
Reintroduction	White	White River	1101001306	Glaise Creek-White River	Extirpated	yes