AN ASSESSMENT OF STREAM FISH VULNERABILITY AND AN EVALUATION OF CONSERVATION NETWORKS IN MISSOURI

A Thesis

Presented to

the Faculty of the Graduate School

at the University of Missouri

In Partial Fulfillment

Of the Requirements for the Degree

Master of Science

---

by

NICHOLAS A. SIEVERT

DR. CRAIG P. PAUKERT, THESIS SUPERVISOR

DECEMBER 2014

The undersigned, appointed by the dean of the Graduate School, have examined the thesis entitled:

AN ASSESSMENT OF STREAM FISH VULNERABILITY AND AN EVALUATION OF CONSERVATION NETWORKS IN MISSOURI

Presented by Nicholas A. Sievert

A candidate for the degree of Master of Science

And hereby certify that, in their opinion, it is worthy of acceptance.

____

Dr. Craig Paukert

____

Dr. Joanna Whittier

____

Dr. Timothy Matisziw

____

Dr. Michelle Staudinger

ACKNOWLEDGMENTS

I would first like to thank the United States Geological Service National Climate Change and Wildlife Science Center for funding this project. I would also like to thank the Missouri Department of Conservation (MDC) for providing the fish community data which served as the foundation upon which this project was completed. Specifically, I would like to thank Matt Combes and Dr. Doug Novinger, who not only provided me with tremendous sources of data for Missouri's stream fish communities, but also shared with me their expertise and knowledge by reviewing my work and offering invaluable insights. Dorothy Butler of MDC also generously provided fish records from the Missouri Natural Heritage Database. I would also like to thank Gust Annis and the Missouri Resource Assessment Partnership for providing me with GIS data without which this project would not have been possible. Dennis Figg of MDC also provided me with both insight and data related to Missouri's conservation networks. Bob Hrabik and Amy Davis, both of MDC, provided valuable feedback and insight throughout the development of this thesis. I would also like to thank those who attended the Missouri Conservation Network Evaluation Meeting we organized at the MDC Central Regional Office in April of 2014. We were honored to have 29 participants from a variety of groups including MDC, US Fish and Wildlife Service, US National Parks Service, and the Nature Conservancy attend and provide valuable feedback. I would also like to thank Dr. Yin-Phan Tsang and Dr. Dana Infante from Michigan State University for providing information on species responses to climate change as well as feedback on my work.

There have been many people throughout this process who have contributed greatly to both my personal and professional growth, while giving me great memories and lots of laughs. I'd like to thank my fellow graduate students and post docs in the Department of Fisheries and Wildlife Sciences, specifically Landon Pierce, Jason Harris, Jeff Fore, Pablo Oliero, Dr. Jacob Westhoff, Emily Pherigo, and Andy Dinges, for their mentorship, thoughtfulness, friendship, and for keeping me sane during long hours in the basement. I would like to thank my parents Randy and Sandy Sievert for always encouraging me to learn and to follow my passions, and my fiancé Veronica Conaway for her support, encouragement, and patience throughout this process.

I would also like to thank Dr. Tim Matisziw and Dr. Michelle Staudinger for providing me with unique perspectives and helpful feedback throughout the development of my thesis. Dr. Jodi Whittier was instrumental to this project. She was always available to share her knowledge and experience with me, without which I would have a lot less hair on top of my head and this project would not have been as successful. Most of all I would like to thank my advisor, Dr. Craig Paukert, for doing everything he could to give me an opportunity to be successful. Craig's guidance helped me grow as a thinker, professional, researcher, writer, and person. I could not have asked for a more supportive, accessible, and good-natured advisor. I owe so much gratitude to so many people; this thesis is truly a product of the efforts and encouragement of all who have supported me.

TABLE OF CONTENTS

LIST OF ILLUSTRATIONS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDICES

AN ASSESSMENT OF STREAM FISH VULNERABILITY AND AN EVALUATION OF CONSERVATION NETWORKS IN MISSOURI

Nicholas A. Sievert

Dr. Craig P. Paukert, Thesis Supervisor

ABSTRACT

The conservation of aquatic biodiversity is largely dependent on the ability of researchers and managers to identify vulnerable species and prioritize conservation actions. Stream fish are facing tremendous threats due to habitat degradation, stream network fragmentation, and climate change. For this project two vulnerability indices were developed to quantify the relative vulnerability of stream fish in Missouri. These indices allowed us to identify the most vulnerable stream fish in Missouri, and also to compare how the use of traits differed from the use of measured species responses in quantifying vulnerability. The most valuable areas for stream fish conservation within and complementary to Missouri's conservation networks were identified through the use of a systematic conservation planning approach. Species representation, weighting based on vulnerability and listing status, and species-specific responses to upstream habitat degradation were utilized to identify priority areas with the conservation planning tool Zonation. This information can assist managers in identifying the species most in need of conservation and where the best opportunities exist for taking management actions. The frameworks for assessing both vulnerability and conservation value can be used or adapted for addressing similar concerns in other regions.

DESCRIPTION OF CHAPTERS

The two chapters of this thesis were written as independent manuscripts which will be submitted to peer-reviewed journals. For this thesis we included a general introduction and conclusion to tie the elements of the two chapters together into a more complete narrative. Because each chapter was written as an independent manuscript there is some overlap in the information conveyed in the introductions, as well as in the study area and data sections of the materials and methods, and references, tables, and figures that were included separately for each chapter. A single set of appendices were generated for the entire thesis. Intended co-authors of the manuscripts were listed after each chapter title and the thesis was written using plural nouns to include co-authors.

GENERAL INTRODUCTION

Lotic ecosystems are among the most imperiled on the planet (Allan and Flecker 1993, Abell 2002, Dudgeon *et al.* 2006). Freshwater biodiversity is declining at a faster rate than in any terrestrial biome (Ricciardi and Rasmussen 1999), and the threats posed by anthropogenic land-use, climate change, flow modification, and invasive species are likely to exacerbate the situation (Ricciardi and Rasmussen 1999, Dudgeon *et al.* 2006, Staudt *et al.* 2013). A combined extinction rate for unionid mussels, crayfishes, fish, and amphibians is near 4% per decade, which is significantly higher than that of other taxa (Ricciardi and Rasmussen 1999). Throughout most of the world, information on the status of biodiversity is limited, biased, or absent (Dudgeon *et al.* 2006). However, in areas with robust data, researchers are finding alarming trends in the number of imperiled species. In North America approximately 39% of aquatic species are considered imperiled (Jelks *et al.* 2008) and in regions with less data, such as Asia, the pressures of dense human populations, rapid growth, and high levels of biodiversity and endemics likely present an even bleaker scenario (Dudgeon *et al.* 2006). This evidence suggests the conservation of aquatic biodiversity likely requires conservation and management actions, or substantial biodiversity losses are likely. In Missouri, the status of aquatic biodiversity is following similar trends. Of the 210 native fish species, 66 (31%) have been designated as species of conservation concern (Sowa *et al.* 2007, Missouri Natural Heritage Program 2012). The long-term conservation of aquatic biodiversity depends on our ability to identify threats, determine species vulnerability, and develop effective conservation strategies.

Lotic biodiversity is expected to be influenced by a number of threats including warming stream temperatures, alterations to flow regimes, and habitat degradation across the globe (Allan and Flecker 1993, Eaton and Scheller 1996, Poff *et al.* 2002, Allan 2004). Stream temperatures are expected to increase by approximately 3.6 degrees Celsius by 2095 based on an intermediate (A1B) estimate of average annual air temperature and air to stream temperature conversions (Eaton and Scheller 1996, Girvetz *et al.* 2009), which has been predicted to result in species extirpations and range contractions (Eaton and Scheller 1996, Mohseni *et al.* 2003, Lyons *et al.* 2010). Even if temperature shifts do not result in range loss, they have the potential to affect the performance of populations by influencing energetics, growth, and feeding (Pease and Paukert 2013, Westhoff and Paukert 2014). Alterations to stream flow regimes from changes in the timing, amount, and type (snow vs rain) of precipitation are expected to impact stream fish (Poff *et al.* 2002). Some stream fish species depend upon predictable seasonal flow patterns for successful reproduction, and changes to these patterns could result in a decline of reproductive success (Poff *et al.* 2002). Increases in the intensity of rainfall on fewer rain days are predicted, leading to an increased magnitude and frequency of flooding, which has been linked to shifts in species composition and local species extirpations (Poff *et al.* 1997, 2002). Summer and fall stream flow may be reduced in many regions where increased frequency and magnitudes of droughts have been predicted (Poff *et al.* 2002). Species which require consistent or high amounts of discharge may be lost from stream segments impacted by drought conditions (Larimore *et al.* 1959, Stanley *et al.* 1997). Habitat alteration and degradation resulting from anthropogenic land use impacts is another major threat to stream fish (Malmqvist and Rundle 2002). Continued agricultural and urban development and the modification of streams and rivers through channelization, dredging, and damming, are expected to continue to have negative impacts on sensitive stream fish species (Malmqvist and Rundle 2002, Allan 2004).

The state of Missouri is heavily impacted by both urban and agricultural development. Approximately 37% of Missouri is cropland (which includes both harvested and pastured lands), and while urban land comprises only 3.7% of Missouri's area, it is highly concentrated in the St. Louis and Kansas City areas (Blodgett and Lea 2005). Land use varies substantially by region, with agriculture and grassland dominating in the north, forests in the south and urban around St. Louis and Kansas City (Blodgett and Lea 2005). Identifying areas for protection of aquatic biodiversity is important as urban and agricultural land use encompass about 40% of the state (Blodgett and Lea 2005).

In order to assess the status of aquatic biodiversity conservation in Missouri, we addressed two primary questions:

1: How vulnerable are Missouri species and communities to current and future threats?

2: How well is aquatic biodiversity represented within Missouri's conservation networks and which areas are the most valuable for stream fish conservation?

We can prioritize conservation efforts to benefit the species and communities in greatest need by developing an understanding of which species are vulnerable to climate change and habitat degradation and identifying areas which are important to the conservation of those species. We evaluated aquatic species representation within Missouri's established conservation networks using species distribution models and identified priority areas for stream fish conservation using systematic conservation planning techniques. These networks include the existing conservation network (ECN) composed of areas established for the primary purpose of wildlife conservation, Conservation Opportunity Areas (COAs) which are areas established in the Missouri Comprehensive Wildlife Strategy (Hoskins 2005) as target areas for conservation action, and Priority Watersheds (PWs) which were established as a part of the Missouri Department of Conservation Fisheries Program as target areas for conservation and management actions (Corson *et al.* 2010). This allowed for the evaluation of how well aquatic biodiversity is represented within these networks, and a ranking of stream segments within and complementary to each established network based on their value for stream fish conservation.

REFERENCES

Abell, R. 2002. Conservation Biology for the Biodiversity Crisis: a Freshwater Follow-up. Conservation Biology **16**(5), 1435–1437.

Allan, J. 2004. Landscapes and Riverscapes: The Influence of Land Use on Stream Ecosystems. Annual Review of Ecology, Evolution, and Systematics **35**, 257–284.

Allan, J., and A. Flecker. 1993. Biodiversity Conservation in Running Waters. BioScience **43**(1), 32–43.

Blodgett, C., and R. Lea. 2005. Missouri Landcover Project. Missouri Resource Assessment Partnership. Columbia, Missouri.

Corson, A., P. Blanchard, and P. Calvert. 2010. Strategies for Watershed Management. Missouri Department of Conservation, Jefferson City, MO.

Dudgeon, D., A. H. Arthington, M. O. Gessner, Z. I. Kawabata, D. J. Knowler, C. Lévêque, R. J. Naiman, A. H. Prieur-Richard, D. Soto, and M. L. J. Stiassny. 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. Biological Reviews **81**(2), 163–182.

Eaton, J., and R. Scheller. 1996. Effects of Climate Warming on Fish Thermal Habitat in Streams of the United States. Limnology and Oceanography **41**(5), 1109–1115.

Girvetz, E., C. Zganjar, G. Raber, E. Maurer, P. Kareiva, and J. Lawler. 2009. Applied Climate-Change Analysis: The Climate Wizard Tool. PLoS ONE **4**(12) e8320.

Hoskins, J. 2005. Missouri's Comprehensive Wildlife Strategy. Missouri Department of Conservation.

Jelks, H., S. Walsh, N. Burkhead, S. Contreras-Balderas, E. Diaz-Pardo, D. Hendrickson, J. Lyons, N. Mandrak, F. McCormick, and J. Nelson. 2008. Conservation status of imperiled North American freshwater and diadromous fishes. Fisheries **33**(8), 372–407.

Larimore, R. W., W. F. Childers, and C. Heckrotte. 1959. Destruction and Re-establishment of Stream Fish and Invertebrates Affected by Drought. Transactions of the American Fisheries Society **88**(4), 261–285.

Lyons, J., J. Stewart, and M. Mitro. 2010. Predicted effects of climate warming on the distribution of 50 stream fishes in Wisconsin, U.S.A. Journal of Fish Biology **77**(8), 1867–1898.

Malmqvist, B., and S. Rundle. 2002. Threats to the running water ecosystems of the world. Environmental Conservation **29**(2), 134–153.

Missouri Natural Heritage Program. 2012. Missouri Species and Communities of Conservation Concern Checklist. Missouri Department of Conservation.

Mohseni, O., H. Stefan, and J. Eaton. 2003. Global Warming and Potential Changes in Fish Habitat in U.S. Streams. Climatic Change **59**(3), 389–409.

Pease, A. A., and C. P. Paukert. 2013. Potential impacts of climate change on growth and prey consumption of stream-dwelling smallmouth bass in the central United States. Ecology of Freshwater Fish **23**(3), 336-346.

Poff, N., M. Brinson, and J. Day. 2002. Aquatic ecosystems and global climate change. Pew Center on Global Climate Change, Arlington, VA:44.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The Natural Flow Regime. BioScience **47**(11), 769–784.

Ricciardi, A., and J. Rasmussen. 1999. Extinction rates of North American freshwater fauna. Conservation Biology **13**(5), 1220–1222.

Sowa, S., G. Annis, M. Morey, and D. Diamond. 2007. A gap analysis and comprehensive conservation strategy for riverine ecosystems of Missouri. Ecological Monographs **77**(3), 301–334.

Stanley, E. H., S. G. Fisher, and N. B. Grimm. 1997. Ecosystem Expansion and Contraction in Streams. BioScience **47**(7), 427–435.

Staudt, A., A. K. Leidner, J. Howard, K. A. Brauman, J. S. Dukes, L. J. Hansen, C. Paukert, J. Sabo, and L. A. Solórzano. 2013. The added complications of climate change: understanding and managing biodiversity and ecosystems. Frontiers in Ecology and the Environment **11**(9), 494–501.

Westhoff, J. T., and C. P. Paukert. 2014. Climate Change Simulations Predict Altered Biotic Response in a Thermally Heterogeneous Stream System. PloS one **9**(10), e111438.

CHAPTER 1

VULNERABILITY ASSESSMENT FOR MISSOURI STREAM FISH TO CLIMATE CHANGE AND HABITAT DEGRADATION

Nicholas Sievert and Dr. Craig P. Paukert

ABSTRACT

Understanding the future impacts and threats of climate and land use change are critical for long-term biodiversity conservation. We developed and compared two indices to assess the vulnerability of stream fish in Missouri. Both indices assessed vulnerability based on species environmental tolerances, rarity, range size, dispersal ability, and the average connectivity of the streams occupied by each species. These indices differed in how the environmental tolerance components were classified, specifically vulnerability to habitat degradation, warming stream temperatures, and alterations to flow regimes. In one index environmental tolerance components were classified based on measured species responses, while in the second index environmental tolerance was classified based on species traits. We also determined 1) if vulnerability scores were consistent between indices, 2) the relationship between species conservation status (species listed as Missouri species of conservation concern) and vulnerability score, and 3) how species vulnerable to these three threats were spatially distributed. Vulnerability scores were calculated for all 133 species with the trait association index, while only 99 species were able to be evaluated using the species response index, because a number of species lacked response data. The range and

mean for scores from the trait association index were greater than those from the species response index, largely due to the species response index's inability to evaluate many rare species, which generally have high vulnerability scores for the trait association index. The indices were consistent in classifying vulnerability to habitat degradation, but showed substantial variation for vulnerability to increases in stream temperature and alterations to flow regimes. This is likely because the current climate change impacts have had a minimal measurable influence on stream fish communities, limiting the opportunities to observe species-specific responses to these threats, while traits can be more generally linked to vulnerability. Both indices showed higher mean vulnerability scores for listed species than unlisted species, which provided a coarse measure of validation. The distribution of vulnerable species in Missouri showed consistent patterns between indices, with the Ozark subregion generally having higher numbers and proportions of vulnerable species per site than in other subregions, which suggests that if conservation actions target habitats rather than species, both indices will perform similarly. This vulnerability assessment provides valuable information regarding the status of Missouri stream fish species and we believe this framework can be used, and updated, for the development of vulnerability assessments in other regions.

INTRODUCTION

Stream ecosystems have some of the most imperiled communities on Earth and freshwater biodiversity is declining at a higher rate than most other taxa groups (Allan and Flecker 1993; Abell 2002; Dudgeon *et al.* 2006). Continuing habitat degradation, warming stream temperatures, and alterations to flow regimes will likely cause continuing declines (Ricciardi and Rasmussen 1999; Dudgeon *et al.* 2006). Conservation actions aimed at protecting aquatic biodiversity are critical for preventing future biodiversity losses (Master *et al.* 1998). In order to plan for long-term biodiversity conservation we must gain a better understanding of how impacts and threats, such as climate change and habitat degradation, affect aquatic species. Conservation and management of stream fish species will require researchers and managers to identify which species are vulnerable, or likely to experience harm, under future conditions (Turner *et al.* 2003; Glick *et al.* 2011; Poff *et al.* 2012). This study developed a new vulnerability assessment framework that identifies species vulnerable to warming stream temperatures, alterations in flow regimes, and habitat degradation. These are among the primary threats identified as drivers of future stream fish declines (Malmqvist and Rundle 2002; Poff *et al.* 2002).

Climate change, which is expected to increase stream temperatures and alter flow regimes, is one of the most significant threats facing stream fish (Eaton and Scheller 1996; Poff *et al.* 2002). In Missouri, the average annual air temperature is expected to rise approximately 4 degrees Celsius by 2095 based on an intermediate (A1B) climate scenario (Girvetz *et al.* 2009). This change in air temperature is predicted to increase Missouri stream temperatures by approximately 3.6 degrees Celsius (Eaton and Scheller 1996), although this relationship is complicated locally by factors such as groundwater input, riparian shading, and stream size (Mohseni and Stefan 1999). In Wisconsin, a 3 degrees Celsius air temperature increase is predicted to result in the loss of 343,034 km of stream habitat for cool- and cold-water fishes, which includes a species extirpation (Lyons *et al.* 2010). Eaton and Scheller (1996) found that the amount of habitat available for cold- and cool-water fish species would be reduced nationwide by approximately 50% with a doubling of atmospheric CO_2, resulting in species extirpations in many streams and dramatic range contractions for many species. In another study of streams across the United States cold-water fish habitat was predicted to decrease by 36% and cool-water fish habitat by 15% with a doubling of atmospheric CO_2 concentrations (Mohseni *et al.* 2003). Warm-water streams in the Great Plains sometimes achieve maximum temperatures which are at or near the physiological limits of some fish species, and an increase in stream temperature of just a few degrees is predicted to result in local extirpation and even extinction of some species (Matthews and Zimmerman 1990). Even in cases where stream fish are not extirpated from a stream segment, research has shown that some species are likely to be subject to decreases in fitness based on changes in energetics and growth (Pease and Paukert 2013; Westhoff and Paukert 2014). These studies provide strong evidence which suggests some species of stream fish will decline or face local extirpations as stream temperatures warm.

Changes to flow regimes, related to the seasonal timing and amount of precipitation and snowmelt, are expected to have substantial impacts on stream fish (Poff *et al.* 2002). Increases in the intensity of rainfall on fewer rain days are predicted, which may lead to an increased magnitude and frequency of flooding; this has been linked to shifts in species composition and local species extirpations (Poff *et al.* 1997, 2002). Summer and fall stream flow may be reduced in many regions because of increased evapotranspiration and decreased precipitation frequency, earlier snowmelt could reduce spring flows, and increased frequency and magnitudes of droughts and floods are expected to occur (Poff *et al.* 2002; Wuebbles and Hayhoe 2004). Stream fish vary in their ability to cope with prolonged periods of low flow, and species which require high levels of discharge are likely to be eliminated from many sites (Larimore *et al.* 1959; Stanley *et al.* 1997). The variability of precipitation patterns is likely to increase, which may cause declines in species which exhibit equilibrium or periodic life history strategies (Poff *et al.* 2002; Olden and Kennard 2010; Mims and Olden 2012, 2013).

Another major threat to lotic biodiversity is habitat alteration and degradation (Malmqvist and Rundle 2002). Anthropogenic modifications including the conversion of land to agricultural or urban uses or the modification of streams and rivers from channelization, dredging, and damming often result in altered and degraded stream conditions (Malmqvist and Rundle 2002; Allan 2004). Degraded streams have been associated with losses of aquatic biodiversity (Allan 2004). Indices of biological integrity (IBIs) have identified many stream fish species as intolerant of habitat degradation

(Esselman *et al.* 2011). Additionally, life history traits such as lithophilic spawning, and benthic invertivory have been used as criteria for IBIs measuring habitat degradation, because species with those traits are sensitive to degradation (Berkman and Rabeni 1987; Barbour *et al.* 1999; Simon 1999).

With this knowledge of threats facing stream fish species the vulnerability of species can be assessed. For the purposes of this thesis, vulnerability is defined as the extent to which a species is likely to be negatively impacted by the cumulative effects of climate change and habitat degradation (Turner *et al.* 2003; Schnieder *et al.* 2007; Glick *et al.* 2011). In order to conduct a vulnerability assessment for stream fish species, a framework which accounts for the impacts of multiple threats is needed. Vulnerability is often determined using a framework that assesses a species sensitivity, exposure, and adaptive capacity to threats (Turner *et al.* 2003; IPCC 2007; Glick *et al.* 2011a; Poff *et al.* 2012; Staudinger *et al.* 2013). A number of organizations, government agencies, and researchers have developed vulnerability assessments using this framework (Bagne *et al.* 2011; Glick *et al.* 2011; Young *et al.* 2011). Two prominent vulnerability assessment tools, the System for Assessing Vulnerability of Species (SAVS) developed by the United States Forest Service, and the NatureServe Climate Change Vulnerability Index (NSCCVI) developed for NatureServe, were applied to Missouri stream fish (Bagne *et al.* 2011; Young *et al.* 2011). The applicability of these vulnerability assessments to stream fish species in Missouri to be limited. In Missouri accurate models of current and future stream temperature and flow have not yet been developed. The absence of these models limits our ability to assess the exposure component of these assessments. Both assessment techniques use air temperature models to predict changes in thermal habitat, however in stream systems the relationship between air and water temperatures is confounded by riparian shading, groundwater contribution, and stream size (Mohseni and Stefan 1999; Allan 2004; Caissie 2006; Whitledge *et al.* 2006; Webb *et al.* 2008; Westhoff and Paukert 2014). Because this study is only able to express exposure in the most general terms (warming is expected to occur, flow regimes are expected to become less predictable and more extreme events are expected to occur) it could not be used to differentiate species and therefore cannot be included as a scoring component, at least not until adequate models have been developed. Additionally, these tools were designed for use with a wide array of taxa over large spatial scales, which limits their ability to account for some of the nuances of stream fish vulnerability. Another issue with the application of these tools to stream fish is that they depend on information which is largely unknown for stream fish species; examples include knowledge of a species reliance on interspecific interactions, measures of genetic variation, occurrence of bottlenecks in recent evolutionary history, and phenological response to changing seasonal temperature and precipitation dynamics (Bagne *et al.* 2011; Young *et al.* 2011). Currently, there simply is not enough information available to broadly apply these assessment tools to stream fish in a meaningful way. The inability to reasonably apply available vulnerability assessment tools to determine the vulnerability of Missouri’s stream fish necessitates the development of a new methodology.

Poff *et al.* (2012) developed a framework for assessing the threat posed by climate change to freshwater diversity. This framework is a function of three components: exposure to the threats, sensitivity of species to impacts of threats, and freshwater resilience or ability to adapt to or cope with the threats (Table 1). Although this framework deviates from the definitions and structure presented by Glick *et al.* (2011) which is widely used for developing vulnerability assessments, we believe that it provides a better mechanism for assessing stream fish vulnerability based on the information that is currently available, and therefore this thesis conformed to structure and definitions found in Poff *et al.* (2012). Species environmental tolerances, which are often expressed as sensitivity in the vulnerability assessment literature, specifically to habitat degradation, stream temperature warming, and alterations to flow regimes, as well as factors such as a species range, rarity, dispersal ability, and the hydrological connectivity of a species habitat can be incorporated into this framework to create a method for assessing the vulnerability of stream fish species. This analysis of species vulnerability will focus on the sensitivity and freshwater resilience components of the Poff *et al.* (2012) framework as adequate information is not yet available to assess exposure in Missouri.

Species tolerances of habitat degradation, altered flow regimes, and increasing stream temperatures have been assessed using two different approaches; species trait associations (Angermeier 1995; Parent and Schriml 1995; Poff 1997; Olden *et al.* 2007, 2008; Culp *et al.* 2011; Mims and Olden 2013) and measured species responses (Hering *et al.* 2006; Lyons *et al.* 2010, Tsang and Infante, Michigan State University, Unpublished data). For this thesis two separate indices were developed, one which scores environmental tolerance based on traits and the other which scores based on species responses. The same scoring framework was used for both indices. The trait association approach to classifying environmental tolerance is based on traits which have been linked to vulnerability to habitat degradation, altered flow regimes, and increases in stream temperature in peer-reviewed literature. The species response approach is based on species-specific observations of sensitivity to each of the environmental tolerance components. Species environmental tolerances provide the foundation for the sensitivity component of this vulnerability assessment framework.

Another aspect of the sensitivity component of the vulnerability assessment framework is species dispersal ability (Poff *et al.* 2012). The ability of a fish species to disperse throughout a stream network is critical to the persistence of species faced with threats (Albanese *et al.* 2009; Poff *et al.* 2012). Dispersal, a one-way movement from a site (Lidicker Jr. and Stenseth 1992), may benefit species facing threats by allowing 1) for gene flow between subpopulations (Hanski 1998; Heggenes *et al.* 2006), 2) increased colonization of newly available habitats (Detenbeck *et al.* 1992; Albanese *et al.* 2009), and 3) species to shift their ranges as environmental changes occur at the landscape scale (Matthews and Zimmerman 1990; Tonn 1990).

Both range size and population size can influence the vulnerability of a species (Poff *et al.* 2012). Species with smaller ranges are limited in their ability to withstand stochastic environmental and demographic fluctuations and are therefore more vulnerable (Angermeier 1995). Species constrained to a small range are more likely to lack access to refugia or an adequate gradient of conditions necessary to cope with the impacts of threats. Rare species, even those which are widely distributed across the landscape, may have increased vulnerability to both deterministic threats such as habitat loss, as well as stochastic events such as invasions and epidemics (Mace *et al.* 2008). Species with narrowly restricted distributions and rare species generally have high levels of vulnerability.

Freshwater resilience, defined as the hydrological connectivity of the landscape, is another component of the vulnerability assessment framework developed by Poff *et al.* (2012). A stream system with high levels of connectivity provides species the opportunity to emigrate to suitable habitats as conditions change (Poff *et al.* 2012). Not only is dispersal difficult because of the restrictive, branching nature of stream networks, but humans have increased the level of fragmentation through the creation of barriers such as dams and road crossings (Jackson and Marmulla 2001; Fagan *et al.* 2002). Species with highly fragmented populations may have an elevated risk of extinction (Fagan *et al.* 2002).

Scientists and decision makers benefit from the ability to identify vulnerable species to make effective decisions regarding the conservation of stream fish. The objectives of this chapter are to 1) classify species' vulnerability based on environmental tolerance to habitat degradation, stream temperature warming, and alterations to flow regimes, dispersal ability, restricted distribution, rarity, and freshwater resilience 2) compare the use of traits and species responses to identify species environmental tolerance 3) develop indices to measure stream fish vulnerability using our species classifications, 4) determine whether our measures of vulnerability correlate with listing status, and 5) analyze distribution patterns of vulnerable species in Missouri. Our aim is to develop a framework for assessing stream fish vulnerability and to make recommendations for assessing stream fish vulnerability based on an application of our methodology to Missouri's stream fish species.

MATERIALS AND METHODS

Study area

The state of Missouri can be classified into three ecologically unique subregions: the Central Plains, Ozarks, and Mississippi Alluvial Basin (MAB) (Pflieger 1970; Pflieger and Missouri Department of Conservation 1989; Sowa *et al.* 2007) (Figure 1). These subregions feature major differences in geology, landform, soils, land cover, and groundwater influence which create unique habitats and fish communities in the streams of each area (Table 2). The Central Plains, in northern Missouri, is dominated by open grassland and agriculture (Figure 1) with streams associated with wide, gently sloping valleys which have a relatively low gradient, high turbidity, and fine silt and sand substrates (Sowa *et al.* 2007). The Ozarks, in southern Missouri (except the southeastern corner), is a mix of forested areas and open pastureland and agriculture (Sowa *et al.* 2007) (Figure 1). Most of the region has rugged terrain with high relief, which creates higher average stream gradients. Many streams in the Ozarks have substantial groundwater inputs. The substrate of these streams is highly variable but often consists of gravel, cobble, or bedrock with sand and silt in slower moving areas and pools. The MAB is in southeastern Missouri, and is a broad flat plain with low stream gradients, most of which have been channelized for agriculture, which is the predominant land cover in the region (Figure 1). The substrate varies throughout the subregion but generally the larger and slower moving streams are characterized by fine silt while streams with faster flow are comprised of sand and small gravel substrates (Sowa *et al.* 2007).

Fish data

Data detailing fish occurrence records were provided by the Missouri Department of Conservation (MDC) from the Resource Assessment and Monitoring Program (RAM) database. This study used 1499 fish samples which were collected using a statewide stratified random sampling design (Fischer and Combes 2003) of unique stream segments collected between April and September, 2000 to 2011 from permanent wadeable streams (2^{nd}-5^{th} Strahler order) (Figure 2). All samples within the RAM database were collected using standardized methods with block nets enclosing a sample area which was then thoroughly sampled via electrofishing and seining (Fischer and Combes 2003), with all individual specimens identified to species and enumerated.

Scoring system and classification of species vulnerability to threats

Species vulnerability scores were calculated using two different approaches: one based on species responses and the other linked to species traits (Figure 3). Species vulnerability was classified and scored for both indices according to species environmental tolerance, dispersal ability, restricted distribution, rarity, and freshwater resilience. Species environmental tolerance scores were calculated by summing the scores for habitat, temperatures, and flow vulnerability separately for each of the two indices, while all other criteria were scored the same way for both indices (Figure 3). Species sensitivity scores were calculated by multiplying rarity scores by the sum of environmental tolerance, dispersal ability, and range size scores, while the final species score is the sum of the sensitivity and freshwater resilience scores (Figure 3). Scores were assigned to each species based on the classification of species vulnerability to each threat category. Rarity was incorporated as a multiplicative factor rather than an additive component because rarity is expected to compound the impacts of the other components. Scores for each threat category were assigned as either discrete (0 or 1) for vulnerable or not classifications, continuous-additive (0 to 1) for vulnerability which was quantified along a gradient and incorporated into the vulnerability index as an additive component, or continuous-multiplicative (1 to 2) for vulnerability which was quantified along a gradient and incorporated into the vulnerability index as a multiplicative component. Multiplicative components were scaled from 1 to 2, rather than 0 to 1, because these components are only intended to have no effect or increase a species score, not reduce it.

Environmental tolerance was scored using the discrete value system by summing the vulnerable (1) or not (0) classifications for vulnerability to habitat degradation, warming stream temperatures, and alterations to flow regimes (Figure 3). Two separate environmental tolerance scores were calculated; one using a traits-based approach and another using a measured species response approach. Classifications for the traits-based approach were made after a thorough review of the literature identified traits linked to vulnerability to each of the threats. Lithophilic spawning and benthic invertivory were used to classify species as vulnerable to habitat degradation as both have been linked to sensitivity to sedimentation, a frequent product of habitat degradation (Berkman and Rabeni 1987; Mims *et al.* 2010). Species which exhibit either of these traits were classified as vulnerable to habitat degradation. Species classified as cool- or cold-water adapted have been identified as vulnerable to warming stream temperatures and were therefore classified as vulnerable to warming stream temperatures in this assessment (Matthews and Zimmerman 1990; Eaton and Scheller 1996; Mohseni *et al.* 2003; Lyons *et al.* 2010; Mims *et al.* 2010). As stream flows become less predictable, and incidences of both extreme droughts and floods increase (Poff *et al.* 2002), species exhibiting the periodic life history strategy (prefer predictable flow patterns), and the equilibrium life history strategy (prefer low variability in flow patterns) are expected to be more vulnerable than species exhibiting the opportunistic life history strategy (thrive in streams with unpredictable and variable flow patterns) (Winemiller and Rose 1992; Olden and Kennard 2010; Mims and Olden 2012, 2013).

The second index used species responses to identify the environmental tolerance of species to habitat degradation, stream temperature warming, and alterations to flow regimes (Figure 3). A substantial amount of research has been conducted to identify species which are intolerant of habitat degradation, particularly due to the proliferation of IBIs (Karr 1981; Karr *et al.* 1986; Lyons *et al.* 1996; Karr and Chu 1997; Whittier *et al.* 2007; Pont *et al.* 2009). Esselman *et al.* (2011) conducted a literature review of fish based IBIs from across North America and compiled a list of intolerant species. Species which have been classified as intolerant within the World Wildlife Fund ecoregions (Abell *et al.* 2008) which intersect Missouri (ecoregions numbered 143, 146, 147, 148, 149) were classified as vulnerable to habitat degradation for our species response index. Tsang *et al.* (Michigan State University, Unpublished data) used stream gauge station temperature and flow data to identify species which decreased in abundance with temperature and flow metrics corresponding to future climate change predictions. Species demonstrating decreased abundance in streams with flow metrics associated with climate change predictions were classified as vulnerable to predicted alterations to flow regimes. Species showing decreased abundance in streams with temperature metrics associated with climate change predictions were classified as vulnerable to warming stream temperatures.

Species ability to disperse throughout the stream network is critical to recolonization after disturbance, establishment in new habitat made available via changes to the environment, and maintenance of healthy metapopulation dynamics (Albanese *et al.* 2009; Campbell Grant 2011; Radinger and Wolter 2013). Species with limited dispersal abilities are expected to be vulnerable (Poff *et al.* 2012). Radinger *et al.* (2013) quantified the average dispersal distance for the mobile component (dispersers) for nine fish families. Species from three families (Fundulidae, Cottidae, and Percidae), which had an average dispersal distance of less than 250 meters (Radinger and Wolter 2013), were classified as vulnerable based on their limited dispersal abilities and received scores of 1 (Figure 3). Species from all other families received scores of 0 for the dispersal ability component of these vulnerability indices.

Species range size and rarity have been linked to vulnerability due to environmental change (Poff *et al.* 2012). Range size was scored using the continuous-additive approach (Figure 3). In order to quantify vulnerability based on range size the number of ecological drainage unites (EDUs) in which a species has a known occurrence were counted. These values were then scaled from 0 for species which occur in all 17 EDUs within Missouri to 1 for species which occur in a single EDU. Intermediate values were calculated along this scale; for instance if a species was found in 25% of EDUs it would receive a score of 0.75, if a species was found in 88% of drainages it would receive a score of 0.12. Similarly, species rarity was calculated by using scaled values from 1 to 2. First, a cutoff for which species would be considered common was needed. This was achieved by examining a histogram of the percent of sites occupied by each species. A threshold was identified at less than 10% of sites (87 species; Figure 4). Using the continuous-multiplicative approach, species which were found at <10% of sites received scores scaled from 1 (10% of sites) to 2 (A single site, 0.01% of sites). Through this scaling system a species found at seven percent of sites would receive a score of 1.3 while a species found at two percent of sites would receive a score of 1.8.

Freshwater resilience scores were calculated as the ratio of the number of connected stream kilometers per occurrence scaled using the continuous additive approach from 0 for the maximum value (1,787 km/occurrence) to 1 for the minimum value (123 km/occurrence) (Figure 3). The amount of connected stream kilometers was calculated by partitioning out 2nd-6th order stream segments in the Missouri Resource Assessment Partnership's stream network GIS layer (Blodgett and Lea 2005). Stream segments classified as stream or stream/river (i.e., not reservoir or impoundment) were isolated via a definition query, which clipped each stream network at any location which flowed into an impoundment. The number of connected stream kilometers for each intact stream segment was calculated with the Rivex tool (Hornby 2014). This information allowed us to quantify the number of connected stream kilometers for each species occurrence. This information was used to determine the average number of connected kilometers for the occurrences of each species. Species with high average connected stream kilometers per occurrence inhabit areas with relatively high stream network connectivity, while species with relatively low values occur in fragmented habitats with low connectivity.

Index comparisons

Several analyses were conducted to determine whether the species response and trait association approaches for classifying environmental tolerance achieved similar results, whether state-listed species received higher vulnerability scores than unlisted species, and whether there were patterns in the distribution of vulnerable species between aquatic subregions. The percent agreement between the two approaches for classifications of vulnerability to habitat degradation, warming stream temperatures, and alterations to flow regimes was used to determine the consistency of the approaches to classifying environmental tolerance. A Fischer's exact test was used to determine if the indices showed a positive odd's ratio ($\alpha<0.01$) for vulnerable classifications of the environmental tolerance components of the two approaches. A two-way analysis of variance was used to determine if mean vulnerability score differed by listing status, Missouri species of conservation concern including all federally threatened and endangered species (Missouri Natural Heritage Program 2012), and environmental tolerance classification approach. Pearson's correlation analysis was used to determine if the two indices were related based on the number and proportion of vulnerable species at each of the study sites for each of the three components of environmental tolerance. An analysis of variance was used to determine whether the mean number and mean proportion of vulnerable species to each of the three environmental tolerance components differed by Missouri's three aquatic subregions. All statistical calculations were performed using R statistical software (R Development Core Team 2011).

RESULTS

Index development

Vulnerability scores for all 133 species were calculated using a trait association approach to classifying environmental tolerance (Appendix 1). The mean vulnerability score using the trait association approach was 3.84 (+/- 0.35 95% CI). Using the species response approach for classifying environmental tolerance 99 of the 133 species could be evaluated and the mean score for this approach was 2.38 (+/- 0.28 95% CI) (Appendix

1). The remaining 34 species were not evaluated because they lacked the information necessary for classifying environmental tolerances. The trait association approach classified 71, 55, and 54 species vulnerable to habitat degradation, warming stream temperatures, and alterations to flow regimes respectively (Appendix 2). The species response approach classified 43, 3, and 18 species vulnerable to habitat degradation, warming stream temperatures, and alterations to flow regimes respectively (Appendix 3). For each of the 133 species rarity, dispersal, range, and freshwater resilience scores were calculated (Appendix 4). The mean rarity score was 1.48 with 65.4% of species classified as being rare based on occurrence rates of <10% of sites. Thirty four species (25.6%) were classified as vulnerable based on limited dispersal ability. The mean number of EDU's in which a species was present was 8 (of 17 total), which equates to a score of 0.53. The mean value for freshwater resilience was 0.53 which equates to an average of 743 connected kilometers per occurrence.

Index comparisons

The species response and trait association approaches to classifying environmental tolerance showed similarities and differences in classification. The most consistent classification between methods was for vulnerability to habitat degradation (65.7%), followed by warming temperatures (60.1%) and lastly flow regime alteration (46.5%) (Table 3). There was a high degree of positive association between the trait and response based approaches to classifying vulnerability based on habitat degradation (Odd's ratio of 3.68, p=0.002), while no significant association was found between the trait and response based approaches to classifying vulnerability based on warming temperatures (Odd's ratio of 0.80, $p=0.99$) or changes in flow regimes (Odd's ratio of 0.49; $p= 0.2$) (Table 3).

Mean vulnerability scores differed by both listing status ($p<0.001$), and index ($p<0.001$), with no interaction between listing status and index ($p=0.794$). The mean species response score for listed species was 3.35, while unlisted species averaged scores of 2.29; for the trait association approach, listed species averaged scores of 4.84, while unlisted species averaged scores of 3.57 (Figure 5). Scores for rare species (species occurring in less than 10% of sites) were compared to common species and it was found that rare species had significantly higher vulnerability scores for both approaches (Response: Rare=2.8, Not =1.9, $P<0.001$; Trait: Rare=4.6, Not=2.5, $P<0.001$).

The number of vulnerable species at each site had the highest correlation between the trait and response-based approaches for habitat degradation ($r=0.94$), followed by temperature ($r=0.37$), and flow ($r=0.23$) (Figure 6). The proportion of vulnerable species at each site had the highest correlation between both indices for habitat degradation ($r=0.87$), followed by temperature ($r=0.15$), and finally flow ($r=-0.25$) (Figure 6). The mean number of vulnerable species per site was greatest in the Ozark subregion for both habitat degradation and temperature using both approaches of classifying environmental tolerances (Table 4). The greatest numbers of species vulnerable per site for flow occurred in the Mississippi Alluvial Basin according to the trait association approach, and the Plains according to the species response approach (Table 4). The mean proportion of vulnerable species per site was highest in the Ozarks for habitat and temperature for the trait association approach and habitat for the species response approach (Table 4). The greatest proportion of flow vulnerable species per site was highest in the Mississippi Alluvial Basin for the trait association, and in the plains for the species response approach (Table 4).

DISCUSSION

The results of this assessment provide a framework for assessing stream fish vulnerability to climate change and habitat degradation impacts. Stream communities in regions across the globe will be impacted by climate change and habitat degradation (Eaton and Scheller 1996; Poff *et al.* 2002; Heino *et al.* 2009), and evaluating species vulnerability will be critical for conservation planning efforts (Poff *et al.* 2012). This assessment provides a new approach for stream fish vulnerability assessment which can be applied and modified in other regions and improved with further evaluation and refinement. The trait association approach to classifying environmental tolerance was applied to all species while the species response approach could only able be applied to 74 percent of species in this analysis. The trait association approach produced higher vulnerability scores (3.84 versus 2.38), and exhibited a greater range of scores (0-8.75 versus 0-7.82). Both of these discrepancies between the indices largely stems from the inability of the species response index to classify the environmental tolerances of rare species. Of the 34 species for which environmental tolerance could be classified based on measured species responses, 32 were classified as rare (occurring at less than 10% of the study sites). Rare species had significantly higher vulnerability scores than common species using both approaches which likely resulted in higher average and maximum scores for the trait association index compared to the species response index. Although some regions, such as California, appear to have sufficient information regarding the vulnerability of species to specific metrics, many regions lack the data necessary to elicit the responses of species to environmental threats necessary for a broadly applied vulnerability assessment (Moyle *et al.* 2013). To further assess and quantify the impact of changes in the environment, quantitative assessments of stream fish relationships with potential stressors is needed (Schlosser 1991). Broad scale vulnerability assessments which rely on measured species responses to environmental threats will depend on additional research to determine the impacts of threats on all species. Many of these rare species are likely the most vulnerable, so a failure to include them in a vulnerability assessment would likely leave an important data gap and present a biased assessment.

A traits based approach allows for a more complete assessment of species vulnerability as traits have been described for the vast majority of North American stream fish (Frimpong and Angermeier 2009; Mims *et al.* 2010). The use of traits has flourished for the assessment of stream fish communities, and has proven to be a useful tool in stream fish ecology and conservation research (Parent and Schriml 1995; Poff 1997; Olden and Kennard 2010; Culp *et al.* 2011; McManamay *et al.* 2014). The results presented here suggest a traits based approach allows for a more complete representation of species; while more research linking specific traits to environmental threats would help decrease the uncertainty of these assessments and validate measures of vulnerability.

Substantial discrepancies existed in the number of species classified as vulnerable to environmental tolerance components using the trait and response based approaches. Considering only species evaluated using both the trait association and species response approaches, it was determined that both approaches classified a similar number of species as vulnerable to habitat degradation (47 versus 43 respectively). However, the number of species classified as vulnerable to both warming stream temperatures (38 versus 3) and alterations to flow regimes (47 versus 18) was considerably higher for the trait association approach. This is likely because results derived using traits to classify vulnerability could be extrapolated based on expectations of how species will respond, however when using measured species responses classifications are restricted to observations of responses which have already occurred. Because habitats have already been widely degraded there have been many opportunities to measure species responses in both degraded and intact habitats, however currently observed changes in temperature and flow regimes have only accounted for a small proportion of long-term predictions (Heino *et al.* 2009), and thus responses to these changes may not yet have taken place. This lack of observations of more extreme conditions predicted in the future has likely led to an inability to classify those species which may show negative responses under future climactic conditions.

Differences were observed in which species classified as vulnerable by the two indices. Vulnerability to habitat degradation was classified relatively consistently by both approaches (65.7% agreement), suggesting that both indices were likely successful in capturing and representing species vulnerability to land use changes. In cases where the trait association index classified species as vulnerable while the species response index did not, species are often those which have been identified as some of the most tolerant of degradation (e.g. Johnny Darter *Etheostoma nigrum*, Orangethroat Darter *Etheostoma spectabile*, and Golden Redhorse *Moxostoma erythrurum* (Karr 1981)), and other common, tolerant species (e.g. Orangespotted Sunfish *Lepomis humilis* (Meador and Carlisle 2007), Creek Chub *Semotilus atromaculatus* (Miller *et al.* 1988), and Black Bullhead *Ameiurus melas* (Novomeská and Kováč 2009)). Species classified as vulnerable to habitat degradation by the species response index, which were not by the trait association index, included a number of Centrarchids, (e.g. Warmouth *Lepomis gulosus*, Longear Sunfish *Lepomis megalotis*, Smallmouth Bass *Micropterus dolomeiu*, Shadow Bass *Ambloplites ariommus*, Ozark Bass *Ambloplites constellatus*, and Rock Bass *Ambloplites rupestris*), and some species of the genus *Cyprinella* (e.g. Whitetail Shiner *Cyprinella galactura*, Spotfin Shiner *Cyprinella spiloptera*, and Steelcolor Shiner *Cyprinella whipplei*). This suggests that although these indices are relatively consistent in classifying vulnerability to habitat degradation, there are still some vulnerable species which were excluded from one approach or the other. The relative consistency between the approaches of classifying habitat vulnerable species suggests that the current understanding of vulnerability to habitat degradation is sufficient for the purposes of a broad-scale assessment using either approach.

In contrast, vulnerability to stream warming and alterations to flow regimes showed major discrepancies between the two approaches. Although only three Missouri stream fish species (White Sucker *Catostomus commersonii*, Smallmouth Bass, and Central Stoneroller *Campostoma anomalum*) have shown measured negative responses to warming stream temperatures, under future conditions many additional species could be vulnerable. There are examples of predicted range contraction for many species under future climate projections, even if that range contraction has yet to be observed (Mohseni *et al.* 2003; Heino *et al.* 2009; Lyons *et al.* 2010). Based on the trait association approach, which allows negative responses to be extrapolated even though they may not yet have been observed, a large number of Cyprinids (14 of 33) and Percids (8 of 15) will likely be vulnerable to future warming.

Discrepancies between the two indices were greatest for flow which suggests that there is a need to resolve the conflicting results between vulnerability predictions based on life history strategy (Mims *et al.* 2010) versus vulnerability predictions based on measured species responses (Tsang and Infante, Michigan State University, Unpublished data). Discrepancies between these indices found a number of Catostomids (12 species), Centrarchids (12 species), and Ictalurids (6 species) had traits linked to vulnerability but showed no negative response, while no species within these families showed a negative response without traits linked to vulnerability. The lack of a measured response for these equilibrium and periodic life history strategists suggests that other factors may complicate the predicted relationship between these traits and decreases in abundance and occupancy associated with changes in stream flow. In contrast, several families had much higher numbers of species showing only negative species responses than only traits linked to vulnerability; including Cyprinids (8 versus 4) and Percids (4 versus 1). Based on the trait research literature it was expected that these opportunistic species would tolerate or even benefit from projected changes in stream flow regimes (Mims and Olden 2013; McManamay *et al.* 2014), however observations of fish responses to predicted flow metrics show decreases in species abundance (Tsang and Infante, Michigan State University, Unpublished data). These discrepancies suggest that a thoughtful comparison between the response and trait association approaches should be used to select the most appropriate approach, and future research should be conducted to rectify these differences. When selecting an approach for classifying environmental tolerances researchers and managers may need to consider if sufficient information is available for target species, and which approach to classifying environmental tolerance better fits the most current understanding of species vulnerability, particularly regarding the alteration of flow regimes.

Our species vulnerability scores were positively correlated with listing status, which suggests that both of the indices achieve results that align with those of more traditional methods of identifying vulnerable species. In Missouri, species of conservation concern are determined through expert knowledge while also considering rarity, population trends, and threats following the NatureServe Conservation Status Assessment Criteria (Missouri Natural Heritage Program 2012). Using this information as a partial validation it appears that the quantitative assessment of stream fish vulnerability presented here is consistent with expert opinion. All 23 species of conservation concern were able to be evaluated using the trait association index, while only 5 species of conservation concern were with the species response index. Based on the correlations the indices have with species of conservation concern it appears that species which are currently considered imperiled are likely to be vulnerable to future threats.

Our analysis of the distribution of vulnerable species showed strong spatial trends. Both approaches to classifying environmental tolerance identified the Ozarks as generally having both the highest numbers and proportions of vulnerable species per site than in the other subregions. The exception to this was for flow vulnerability, for which the number and proportion of vulnerable species per site was highest in the MAB according to the trait association approach and the Plains according to the species response approach. Streams in the Ozarks have steeper stream gradients, more diverse substrates, cooler stream temperatures, and less human development than Plains streams and are able to support more sensitive species (Sowa *et al.* 2007). The spatial consistency of the counts and proportions of species classified as vulnerable to habitat degradation and stream warming suggests that management for those threats can be focused on the Ozarks regardless of which approach to classifying environmental tolerance is used. The inconsistency regarding flow vulnerability corroborates the previous comparison of environmental tolerance classification by confirming that further research to determine species vulnerability to predicted alterations to flow regimes is needed. Streams are predicted to experience greater variation in flows (IPCC 2002; Poff *et al.* 2002), however the impact of increased variation in flow on stream fish is unclear. Decision makers may need to consider which approach to classifying environmental tolerance best fits their understanding of flow vulnerable species when making decisions regarding the identification and selection of areas for management of these species.

Understanding the impact of uncertainty is crucial for any vulnerability assessment (Patt *et al.* 2005; Füssel and Klein 2006) and there are many sources of uncertainty that affect our understanding of stream fish vulnerability in Missouri. One of the primary sources of uncertainty in any climate change work is how much the climate will change (Stainforth *et al.* 2007). Many scenarios have been developed which predict a range of future temperature and precipitation conditions (IPCC 2013). In stream systems this uncertainty is compounded by the fact that there is also uncertainty in how these changes in climate will impact stream conditions (Whitledge *et al.* 2006; Westhoff and Paukert 2014). In Missouri models do not yet exist which allow us to predict stream temperature and flow conditions now or under predicted future climate scenarios. Although the general trends are known for changes in stream temperature and flow regimes, currently suitable data is not available to differentiate levels of exposure across the Missouri landscape. Because of this exposure, which is often used in vulnerability assessments (Glick *et al.* 2011; Staudinger *et al.* 2013), cannot be used as a factor of discrimination in species vulnerability analysis. This assessment is also bound to the uncertainty inherent in predicting the negative effects of environmental threats on species based on both traits and measured responses. Understanding the impact of uncertainty on this assessment is crucial for the interpretation of these results, the applicability of the framework for others, and for providing opportunities to refine the assessment in order to reduce uncertainty when new information becomes available.

There are clear tradeoffs between the two approaches of assessing environmental tolerance. Benefits of a trait association approach include broad applicability to many species, consistency between studies/regions, and ability to link to a causal mechanism (Parent and Schriml 1995; Poff 1997; Frimpong and Angermeier 2009; Culp *et al.* 2011). Benefits of a measured species response approach include direct evidence of species intolerance of threats, and the ability to account for complex interactions between species characteristics and their responses (Williams *et al.* 2008; Lyons *et al.* 2010, Tsang and Infante, Michigan State University, Unpublished Data). Disadvantages of a trait association approach include difficulty representing variability in environmental tolerances, limited inference to known and quantified traits, and difficulty accounting for complex species/environmental relationships. Disadvantages of a measured species response approach include restriction to species which have data available (analysis of many rare species are limited by lack of data), limitation of inference to threat impacts which have already occurred in the environment or dependence on laboratory studies to simulate threat impacts, and difficulty isolating the impact of a threat from other factors which may affect a species response. A trait association approach may currently be better suited to assessing stream fish vulnerability because this approach allows us to analyze more species of interest using extrapolations rather than direct observations. Research which investigates species responses to threats could add to the feasibility of a response based assessment of vulnerability. A more complete set of information regarding species thermal and flow tolerances, bioenergetics, and population viability coupled with landscape models of stream temperatures and flow regimes could lend insight for the development of response based models which could predict species specific vulnerability.

Many stream fish species are already threatened with local extirpation, and increases in habitat degradation, stream warming, and alterations to flow regimes of streams will likely result in increases in the number of species impacted as well as the magnitude of this imperilment (Poff *et al.* 2012). To more effectively conserve stream fish, decision makers need a better understanding of which species are most vulnerable to each of the threats. This study created a quantifiable index of stream fish vulnerability based on the outline by Poff *et al.* (2012) that fills this need for the state of Missouri. As more information is gathered for stream fishes the application of more traditional vulnerability assessments following the criteria and framework established by Glick *et al.* (2011) may be possible, however based on currently available information we believe this approach provides a suitable alternative. The assessment of vulnerability presented here allows for identification of species vulnerable to specific threats and also gives us a cumulative measure of a species vulnerability to a suite of threats. This study developed a framework which can serve as a foundation for future vulnerability assessments and can be adapted to include new information regarding species environmental tolerances, additional threats, and the weighting or scale of threats. Our hope is that others can apply and modify this framework for assessment of vulnerable stream fish species in other regions.

REFERENCES

Abell, R. (2002) Conservation Biology for the Biodiversity Crisis: a Freshwater Follow-up. *Conservation Biology* **16**, 1435–1437.

Abell, R., Thieme, M.L., Revenga, C., *et al.* (2008) Freshwater ecoregions of the world: a new map of biogeographic units for freshwater biodiversity conservation. *BioScience* **58**, 403–414.

Albanese, B., Angermeier, P.L. and Peterson, J.T. (2009) Does mobility explain variation in colonisation and population recovery among stream fishes? *Freshwater Biology* **54**, 1444–1460.

Allan, J. (2004) Landscapes and Riverscapes: The Influence of Land Use on Stream Ecosystems. *Annual Review of Ecology, Evolution, and Systematics* **35**, 257–284.

Allan, J. and Flecker, A. (1993) Biodiversity Conservation in Running Waters. *BioScience* **43**, 32–43.

Angermeier, P. (1995) Ecological Attributes of Extinction-Prone Species: Loss of Freshwater Fishes of Virginia. *Conservation Biology* **9**, 143–158.

Bagne, K., Friggens, M. and Finch, D. (2011) *A System for Assessing Vulnerability of Species (SAVS) to Climate Change*. USDA Forest Service. Albuquerque, New Mexico.

Barbour, M.T., Gerritsen, J., Snyder, B.D. and Stribling, J.B. (1999) Rapid bioassessment protocols for use in streams and wadeable rivers. *USEPA, Washington* D.C.

Berkman, H. and Rabeni, C. (1987) Effect of siltation on stream fish communities. *Environmental Biology of Fishes* **18**, 285–294.

Blodgett, C. and Lea, R. (2005) *Missouri Landcover Project*. Missouri Resource Assessment Partnership. Columbia, Missouri.

Caissie, D. (2006) The thermal regime of rivers: a review. *Freshwater Biology* **51**, 1389–1406.

Campbell Grant, E. (2011) Structural complexity, movement bias, and metapopulation extinction risk in dendritic ecological networks. *JNABS Journal* **30**, 252–258.

Culp, J.M., Armanini, D.G., Dunbar, M.J., *et al.* (2011) Incorporating traits in aquatic biomonitoring to enhance causal diagnosis and prediction. *Integrated Environmental Assessment and Management* **7**, 187–197.

Detenbeck, N.E., DeVore, P.W., Niemi, G.J. and Lima, A. (1992) Recovery of temperate-stream fish communities from disturbance: a review of case studies and synthesis of theory. *Environmental Management* **16**, 33–53.

Dudgeon, D., Arthington, A.H., Gessner, M.O., *et al.* (2006) Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews* **81**, 163–182.

Eaton, J. and Scheller, R. (1996) Effects of Climate Warming on Fish Thermal Habitat in Streams of the United States. *Limnology and Oceanography* **41**, 1109–1115.

Esselman, P.C., Infante, D.M., Wang, L., Wu, D., Cooper, A.R. and Taylor, W.W. (2011) An index of cumulative disturbance to river fish habitats of the conterminous United States from landscape anthropogenic activities. *Ecological Restoration* **29**, 133–151.

Fagan, W., Unmack, P., Burgess, C. and Minckley, W. (2002) Rarity, fragmentation, and extinction risk in desert fishes. *Ecology* **83**, 3250–3256.

Fischer, S. and Combes, M. (2003) *Resource Assessment and Monitoring Program: Standard Operation Procedures - Fish Sampling*. Missouri Department of Conservation.

Frimpong, E. and Angermeier, P. (2009) Fish traits: A database of ecological and life-history traits of freshwater fishes of the United States. *Fisheries* **34**, 487–495.

Füssel, H.-M. and Klein, R.J.T. (2006) Climate Change Vulnerability Assessments: An Evolution of Conceptual Thinking. *Climatic Change* **75**, 301–329.

Girvetz, E., Zganjar, C., Raber, G., Maurer, E., Kareiva, P. and Lawler, J. (2009) Applied Climate-Change Analysis: The Climate Wizard Tool. *PLoS ONE* **4**, e8320.

Glick, P., Stein, B. and Edelson, N. (2011) *Scanning the conservation horizon: a guide to climate change vulnerability assessment*. National Wildlife Federation Washington, DC, USA.

Hanski, I. (1998) Metapopulation dynamics. *Nature* **396**, 41–49.

Heggenes, J., Qvenild, T., Stamford, M.D. and Taylor, E.B. (2006) Genetic structure in relation to movements in wild European grayling (Thymallus thymallus) in three Norwegian rivers. *Canadian Journal of Fisheries and Aquatic Sciences* **63**, 1309–1319.

Heino, J., Virkkala, R. and Toivonen, H. (2009) Climate change and freshwater biodiversity: detected patterns, future trends and adaptations in northern regions. *Biological Reviews* **84**, 39–54.

Hering, D., Johnson, R.K., Kramm, S., Schmutz, S., Szoszkiewicz, K. and Verdonschot, P.F.M. (2006) Assessment of European streams with diatoms, macrophytes, macroinvertebrates and fish: a comparative metric-based analysis of organism response to stress. *Freshwater Biology* **51**, 1757–1785.

Hornby, D. (2014) *RivEX (Version 10.15) [Software]. http://rivex.co.uk/index.html*.

IPCC (2013) Climate Change 2013: The Physical Science Basis. Working Group 1 Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Chagne. 1535 pp. Intergovernmental Panel on Climate Change, Cambridge, United Kingdom and New York, NY.

IPCC (2007) *Climate Change 2007: Impacts, Adaptation and Vulnerability: Working Group II Contribution to the Fourth Assessment Report of the IPCC Intergovernmental Panel on Climate Change*, (Vol. 4). Cambridge University Press.

IPCC (2002) Climate Change and Biodiversity. IPCC Technical Paper V. Intergovernmental Panel on Climate Change: Working Group II.

Jackson, D.C. and Marmulla, G. (2001) The influence of dams on river fisheries. *FAO Fisheries technical paper*, 1–44.

Karr, J. (1981) Assessment of biotic integrity using fish communities. *Fisheries* **6**, 21–27.

Karr, J.R. and Chu, E.W. (1997) Biological monitoring: essential foundation for ecological risk assessment.

Karr, J.R., Fausch, K.D., Angermeier, P.L., Yant, P.R. and Schlosser, I.J. (1986) Assessing biological integrity in running waters. *A method and its rationale. Illinois Natural History Survey, Champaign, Special Publication* **5**.

Larimore, R.W., Childers, W.F. and Heckrotte, C. (1959) Destruction and Re-establishment of Stream Fish and Invertebrates Affected by Drought. *Transactions of the American Fisheries Society* **88**, 261–285.

Lidicker Jr., W. and Stenseth, N.C. (1992) To disperse or not to disperse: who does it and why?. In: *Animal Dispersal*. (eds N.C. Stenseth and W. Lidicker Jr.). Springer Netherlands, pp 21–36.

Lyons, J., Stewart, J. and Mitro, M. (2010) Predicted effects of climate warming on the distribution of 50 stream fishes in Wisconsin, U.S.A. *Journal of Fish Biology* **77**, 1867–1898.

Lyons, J., Wang, L. and Simonson, T.D. (1996) Development and validation of an index of biotic integrity for coldwater streams in Wisconsin. *North American Journal of Fisheries Management* **16**, 241–256.

Mace, G.M., Collar, N.J., Gaston, K.J., *et al.* (2008) Quantification of extinction risk: IUCN's system for classifying threatened species. *Conservation Biology* **22**, 1424–1442.

Malmqvist, B. and Rundle, S. (2002) Threats to the running water ecosystems of the world. *Environmental conservation* **29**, 134–153.

Master, L.L., Flack, S.R., Stein, B.A. and Conservancy, N. (1998) *Rivers of life: critical watersheds for protecting freshwater biodiversity*. Nature Conservancy Arlington, Virginia.

Matthews, W.J. and Zimmerman, E.G. (1990) Potential effects of global warming on native fishes of the southern Great Plains and the Southwest. *Fisheries* **15**, 26–32.

McManamay, R.A., Bevelhimer, M.S. and Frimpong, E.A. (2014) Associations among hydrologic classifications and fish traits to support environmental flow standards. *Ecohydrology*, n/a–n/a.

Meador, M.R. and Carlisle, D.M. (2007) Quantifying tolerance indicator values for common stream fish species of the United States. *Ecological Indicators* **7**, 329–338.

Miller, D.L., Hughes, R.M., Karr, J.R., *et al.* (1988) Regional Applications of an Index of Biotic Integrity for Use in Water Resource Management. *Fisheries* **13**, 12–20.

Mims, M.C. and Olden, J.D. (2013) Fish assemblages respond to altered flow regimes via ecological filtering of life history strategies. *Freshwater Biology* **58**, 50–62.

Mims, M.C. and Olden, J.D. (2012) Life history theory predicts fish assemblage response to hydrologic regimes. *Ecology* **93**, 35–45.

Mims, M.C., Olden, J.D., Shattuck, Z.R. and Poff, N.L. (2010) Life history trait diversity of native freshwater fishes in North America. *Ecology of Freshwater Fish* **19**, 390–400.

Missouri Natural Heritage Program (2012) *Missouri Species and Communities of Conservation Concern Checklist*. Missouri Department of Conservation.

Mohseni, O. and Stefan, H. (1999) Stream temperature/air temperature relationship: a physical interpretation. *Journal of Hydrology* **218**, 128–141.

Mohseni, O., Stefan, H. and Eaton, J. (2003) Global Warming and Potential Changes in Fish Habitat in U.S. Streams. *Climatic Change* **59**, 389–409.

Moyle, P.B., Kiernan, J.D., Crain, P.K. and Quinones, R.M. (2013) Climate change vulnerability of native and alien freshwater fishes of California: a systematic assessment approach. *PloS one* **8**, e63883.

Novomeská, A. and Kováč, V. (2009) Life-history traits of non-native black bullhead Ameiurus melas with comments on its invasive potential. *Journal of Applied Ichthyology* **25**, 79–84.

Olden, J.D. and Kennard, M.J. (2010) Intercontinental comparison of fish life history strategies along a gradient of hydrologic variability. In: *American Fisheries Society Symposium*, Vol. 73. pp 83–107.

Olden, J., Hogan, Z. and Zanden, M. (2007) Small fish, big fish, red fish, blue fish: size-biased extinction risk of the world's freshwater and marine fishes. *Global Ecology and Biogeography* **16**, 694–701.

Olden, J., Poff, N. and Bestgen, K. (2008) Trait synergisms and the rarity, extirpation, and extinction risk of desert fishes. *Ecology* **89**, 847–856.

Parent, S. and Schriml, L. (1995) A model for the determination of fish species at risk based upon life-history traits and ecological data. *Canadian Journal of Fisheries and Aquatic Sciences* **52**, 1768–1781.

Patt, A., Klein, R.J.T. and de la Vega-Leinert, A. (2005) Taking the uncertainty in climate-change vulnerability assessment seriously. *Comptes Rendus Geoscience* **337**, 411–424.

Pease, A.A. and Paukert, C.P. (2013) Potential impacts of climate change on growth and prey consumption of stream-dwelling smallmouth bass in the central United States. *Ecology of Freshwater Fish* **23**, 336-346.

Pflieger, W. (1970) A distributional study of Missouri fishes. University of Kansas Publications, Museum of Natural History. **20**, 225-570.

Pflieger, W.L. and Missouri Deptartment of Conservation (1989) *Aquatic community classification system for Missouri*. Missouri Dept. of Conservation.

Poff, N. (1997) Landscape filters and species traits: towards mechanistic understanding and prediction in stream ecology. *Journal of the North American Benthological Society*, 391–409.

Poff, N., Brinson, M. and Day, J. (2002) Aquatic ecosystems and global climate change. *Pew Center on Global Climate Change, Arlington, VA*, 44.

Poff, N.L., Allan, J.D., Bain, M.B., *et al.* (1997) The Natural Flow Regime. *BioScience* **47**, 769–784.

Poff, N.L., Olden, J.D. and Strayer, D.S. (2012) Climate change and freshwater extinction risk. In: *Saving a Million Species: Extinction Risk from Climate Change*. (ed L. Hannah). Island Press, pp 309–336.

Pont, D., Hughes, R.M., Whittier, T.R. and Schmutz, S. (2009) A predictive index of biotic integrity model for aquatic-vertebrate assemblages of western US streams. *Transactions of the American Fisheries Society* **138**, 292–305.

Radinger, J. and Wolter, C. (2013) Patterns and predictors of fish dispersal in rivers. *Fish and Fisheries* **15**, 456-473.

R Development Core Team (2011) *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria.

Ricciardi, A. and Rasmussen, J. (1999) Extinction rates of North American freshwater fauna. *Conservation Biology* **13**, 1220–1222.

Schlosser, I.J. (1991) Stream Fish Ecology: A Landscape Perspective. *BioScience* **41**, 704–712.

Schnieder, S., Semenov, S., Patwardhan, A., *et al.* (2007) Assessing key vulnerabilities and the risk from climate change. *Climate change 2007: Impacts, adaptation and vulnerability: Contribution of working group II to the fourth assessment report of the intergovernmental panel on climate change*, 779–810.

Simon, T.P. (1999) *Assessing the sustainability and biological integrity of water resources using fish communities*. CRC Press.

Sowa, S.P., Annis, G., Morey, M.E. and Diamond, D.D. (2007) A Gap Analysis and Comprehensive Conservation Strategy for Riverine Ecosystems of Missouri. *Ecological Monographs* **77**, 301–334.

Stainforth, D.A., Allen, M.R., Tredger, E.R. and Smith, L.A. (2007) Confidence, uncertainty and decision-support relevance in climate predictions. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences* **365**, 2145–2161.

Stanley, E.H., Fisher, S.G. and Grimm, N.B. (1997) Ecosystem Expansion and Contraction in Streams. *BioScience* **47**, 427–435.

Staudinger, M.D., Carter, S.L., Cross, M.S., *et al.* (2013) Biodiversity in a changing climate: a synthesis of current and projected trends in the US. *Frontiers in Ecology and the Environment* **11**, 465–473.

Tonn, W. (1990) Climate change and fish communities: a conceptual framework. *Transactions of the American Fisheries Society* **119**, 337–352.

Turner, B.L., Kasperson, R.E., Matson, P.A., *et al.* (2003a) A framework for vulnerability analysis in sustainability science. *Proceedings of the national academy of sciences* **100**, 8074–8079.

Webb, B., Hannah, D., Moore, R., Brown, L. and Nobilis, F. (2008) Recent advances in stream and river temperature research. *Hydrological Processes* **22**, 902–918.

Westhoff, J.T. and Paukert, C.P. (2014) Climate Change Simulations Predict Altered Biotic Response in a Thermally Heterogeneous Stream System. *PloS one* **9**, e111438.

Whitledge, G., Rabeni, C., Annis, G. and Sowa, S. (2006) Riparian Shading and Groundwater Enhance Growth Potential for Smallmouth Bass in Ozark Streams. *Ecological Applications* **16**, 1461–1473.

Whittier, T.R., Hughes, R.M., Stoddard, J.L., Lomnicky, G.A., Peck, D.V. and Herlihy, A.T. (2007) A structured approach for developing indices of biotic integrity: three examples from streams and rivers in the western USA. *Transactions of the American Fisheries Society* **136**, 718–735.

Williams, S.E., Shoo, L.P., Isaac, J.L., Hoffmann, A.A. and Langham, G. (2008) Towards an Integrated Framework for Assessing the Vulnerability of Species to Climate Change. *PLoS Biol* **6**, e325.

Winemiller, K.O. and Rose, K.A. (1992) Patterns of life-history diversification in North American fishes: implications for population regulation. *Canadian Journal of Fisheries and Aquatic Sciences* **49**, 2196–2218.

Wuebbles, D. and Hayhoe, K. (2004) Climate Change Projections for the United States Midwest. *Mitigation and Adaptation Strategies for Global Change* **9**, 335–363.

Young, B., Byers, E., Gravuer, K., Hall, K., Hammerson, G. and Redder, A. (2011) Guidelines for Using the NatureServe Climate Change Vulnerability Index. NatureServe.

TABLES

Table 1: Vulnerability assessment components used to assess Missouri stream fish vulnerability based on Poff *et al.* 2012.

Component	**Definitions from Poff *et al.* 2012**
Exposure	Deviation in physico-chemical conditions relative to regional baselines.
Sensitivity	Intrinsic factors related to a species environmental tolerance, dispersal ability, genetic adaptation, range, and population size.
Freshwater Resilience	Connectivity of aquatic habitats which provides opportunities for species dispersal.

Table 2: Land use type percentages for the three Missouri aquatic subregions and statewide (Blodgett and Lea 2005).

	Missouri Land Use (%)			
	Statewide	**Plains**	**Ozarks**	**Mississippi Alluvial Basin**
Urban	3.7	4.1	3.5	1.8
Cropland	23.9	37.1	7.0	83.0
Grassland	33.3	35.9	34.4	3.8
Forest	34.5	17.3	51.4	5.0
Wetland	2.1	3.5	1.0	2.5
Water	2.2	2.0	2.1	3.9
Other	0.3	0.1	0.6	0.1

Table 3: Comparison of trait association and species response approaches for classifying species environmental tolerances based on percent agreement of species classifications, odd’s ratio and p-value calculated using Fischer’s exact test.

Threat	**Percent Agreement**	**Odd’s Ratio**	**P-Value**
Habitat Degradation	65.7	3.68	0.002
Temperature	60.1	0.80	1.000
Flow	46.5	0.49	0.200

Table 4: Mean number and proportion of vulnerable species at sites within each subregion by index and threat; highest value in bold. P-values tested whether the mean number or proportion of species differed by subregion using a one-way ANOVA. "MAB"=Mississippi Alluvial Basin; "Num."=Number of species; "Prop."=Proportion of species

Index		**Plains**		**Ozarks**		**MAB**		**P-Value**	
	Threat	**Num.**	**Prop.**	**Num.**	**Prop.**	**Num.**	**Prop.**	**Num.**	**Prop.**
Trait Association									
	Habitat	4.6	0.30	**10.6**	**0.55**	3.3	0.18	<0.001	<0.001
	Flow	5.9	0.41	8.0	**0.42**	**10.1**	0.28	<0.001	<0.001
	Thermal	3.5	0.24	**8.0**	**0.42**	5.1	0.28	<0.001	<0.001
Species Response									
	Habitat	3.0	0.19	**9.2**	**0.51**	2.4	0.16	<0.001	<0.001
	Flow	**3.5**	**0.25**	**2.3**	0.13	2.1	0.14	<0.001	<0.001
	Thermal	1.1	**0.01**	**1.5**	**0.01**	0.0	0.0	0.003	0.027

FIGURES

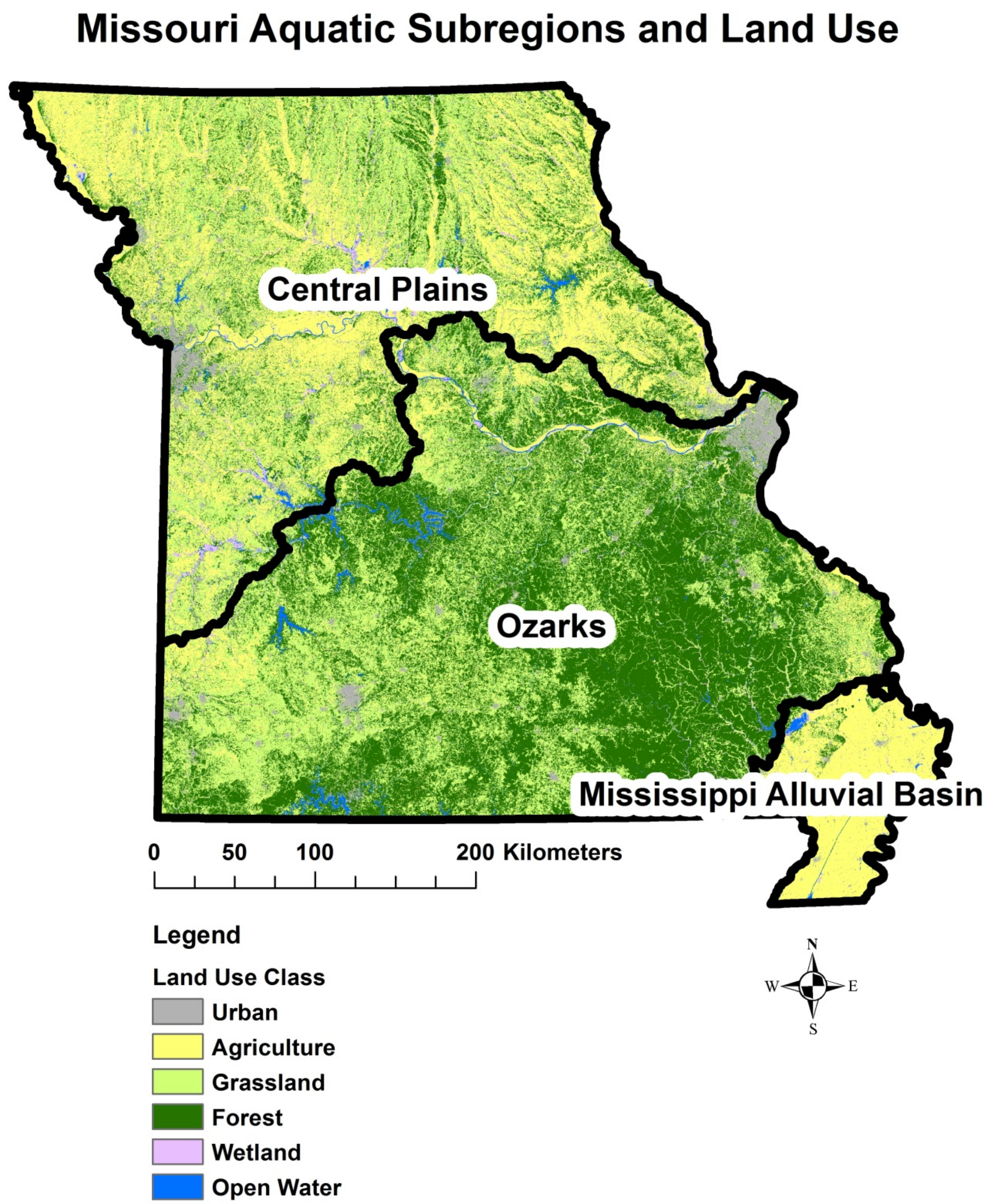

Figure 1: Map showing Missouri's aquatic subregions and land use.

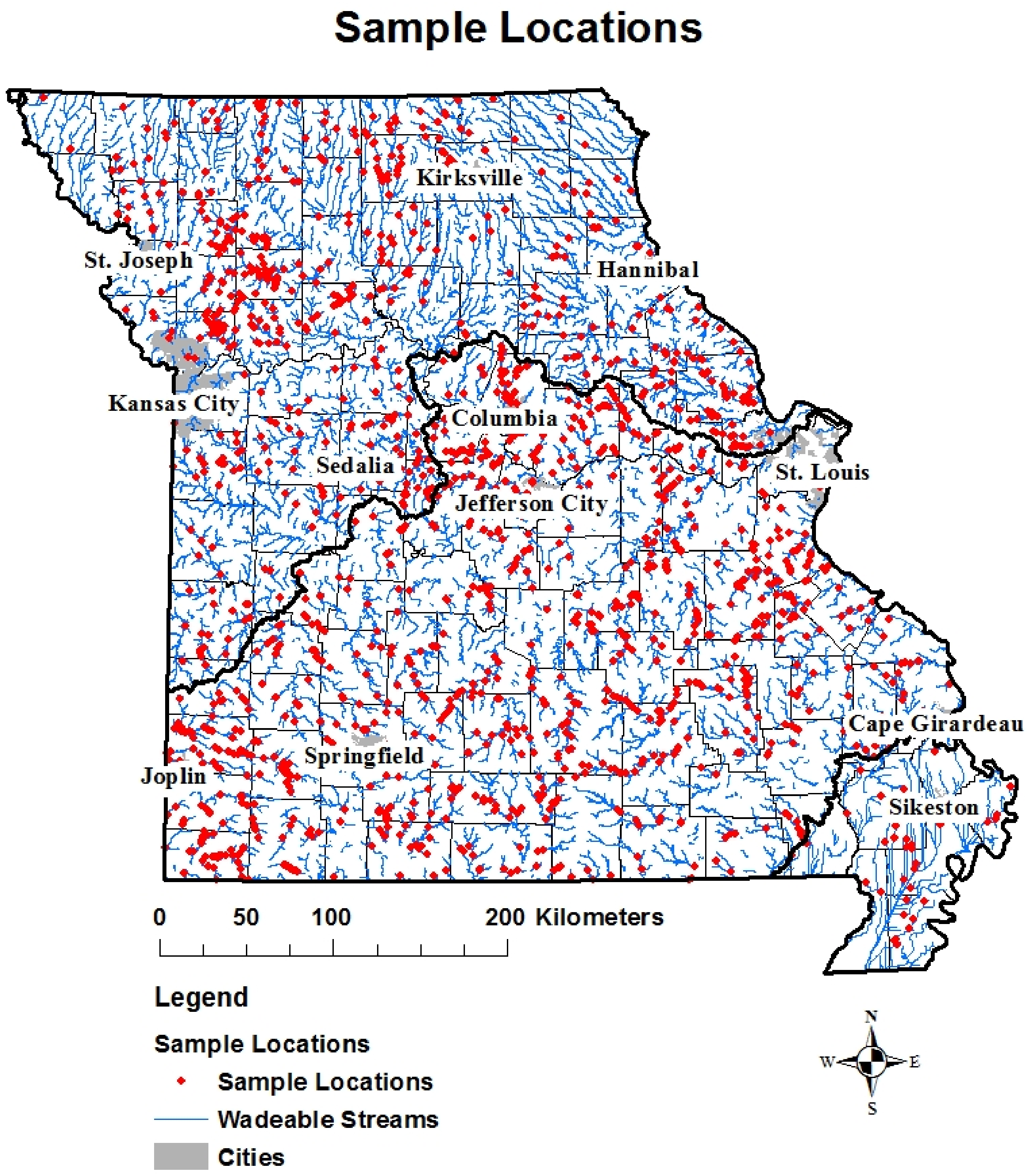

Figure 2: Map of sample locations used in this study for fishes collected from 2000 to 2011 in wadeable streams in Missouri.

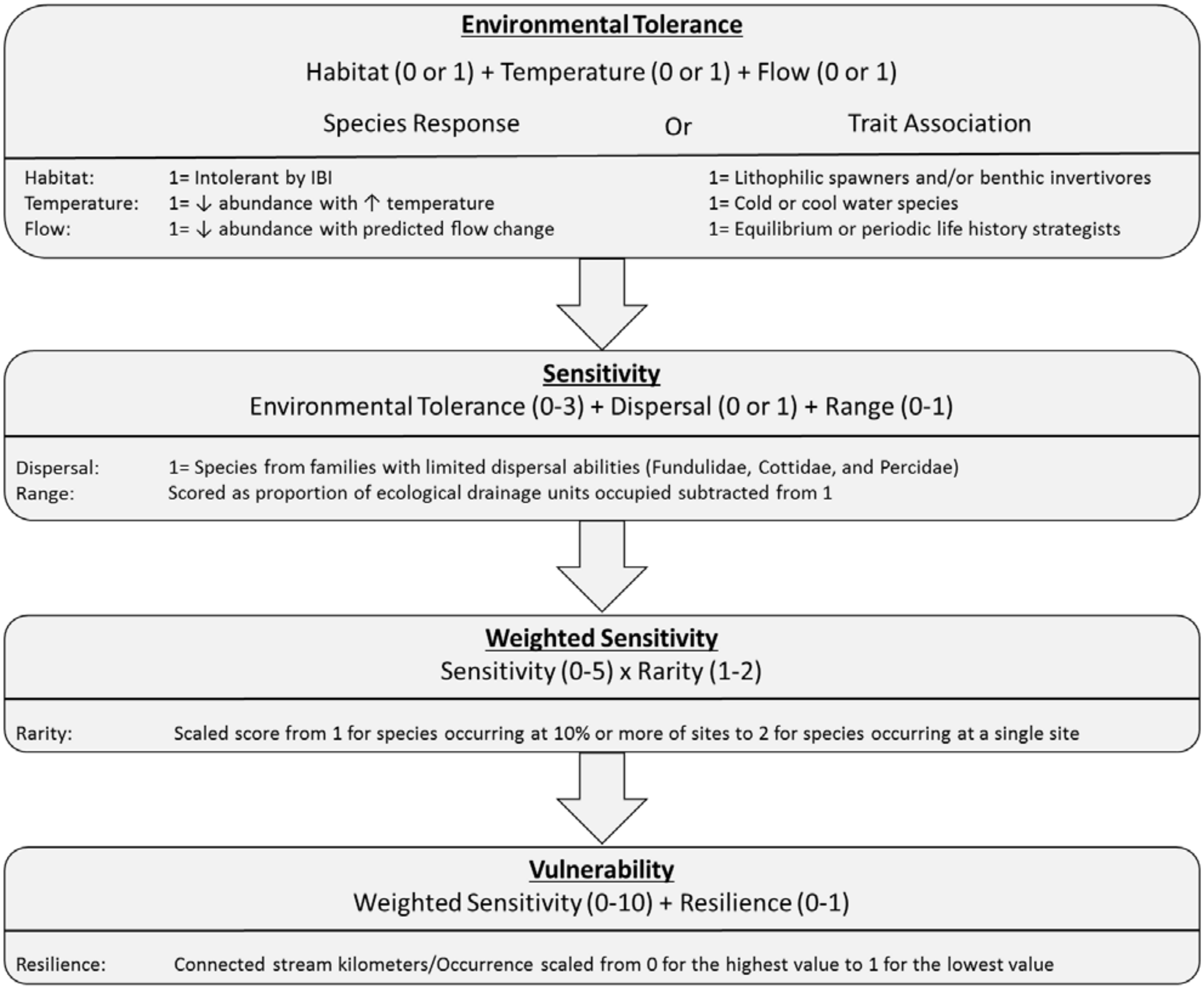

Figure 3: Conceptual diagram of stepwise procedure for calculating species vulnerability scores using the Species Response Index and Trait Association Index. Inputs for dispersal, range, rarity and freshwater resilience were the same for both indices; differences between index calculations occurred only for the environmental tolerance scoring criteria. Numbers in parenthesis indicate potential scoring range. For all "0 or 1" scoring, species meeting listed criteria receive a score of 1, while all others receive a score of 0.

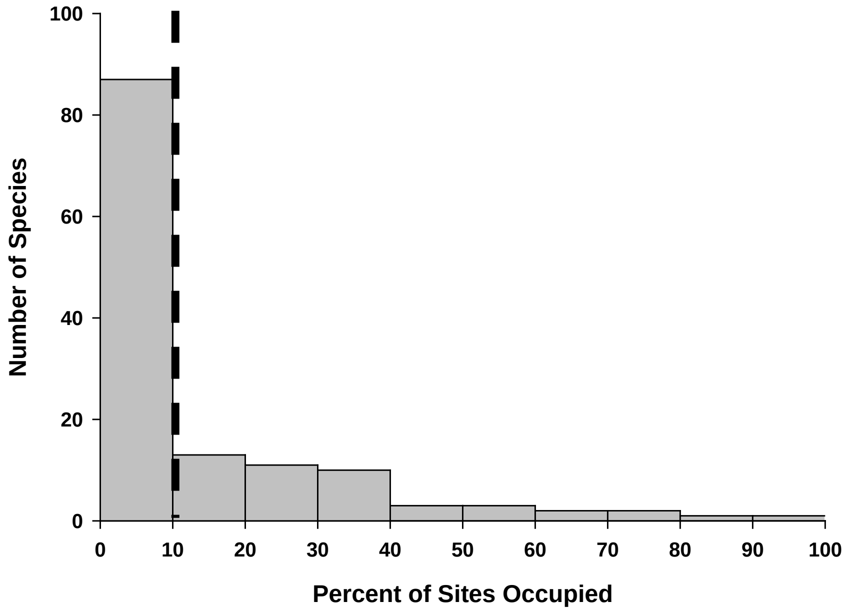

Figure 4: Histogram of species rarity for fish sampled from 1,499 wadeable stream segments in Missouri from 2000 to 2011. The dashed line represents threshold used to classify rare species.

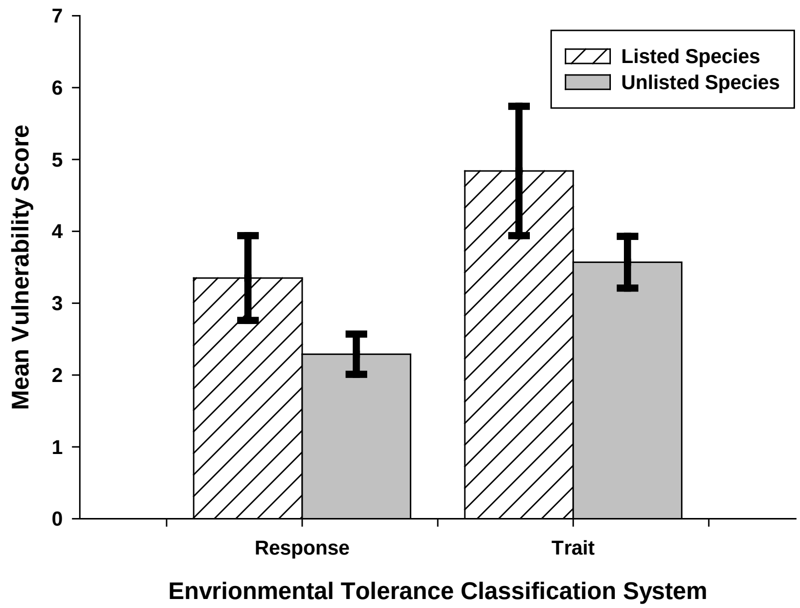

Figure 5: Mean score (+/- 95% Confidence Interval) for listed and unlisted species using the species response and traits association approaches for classifying environmental tolerance.

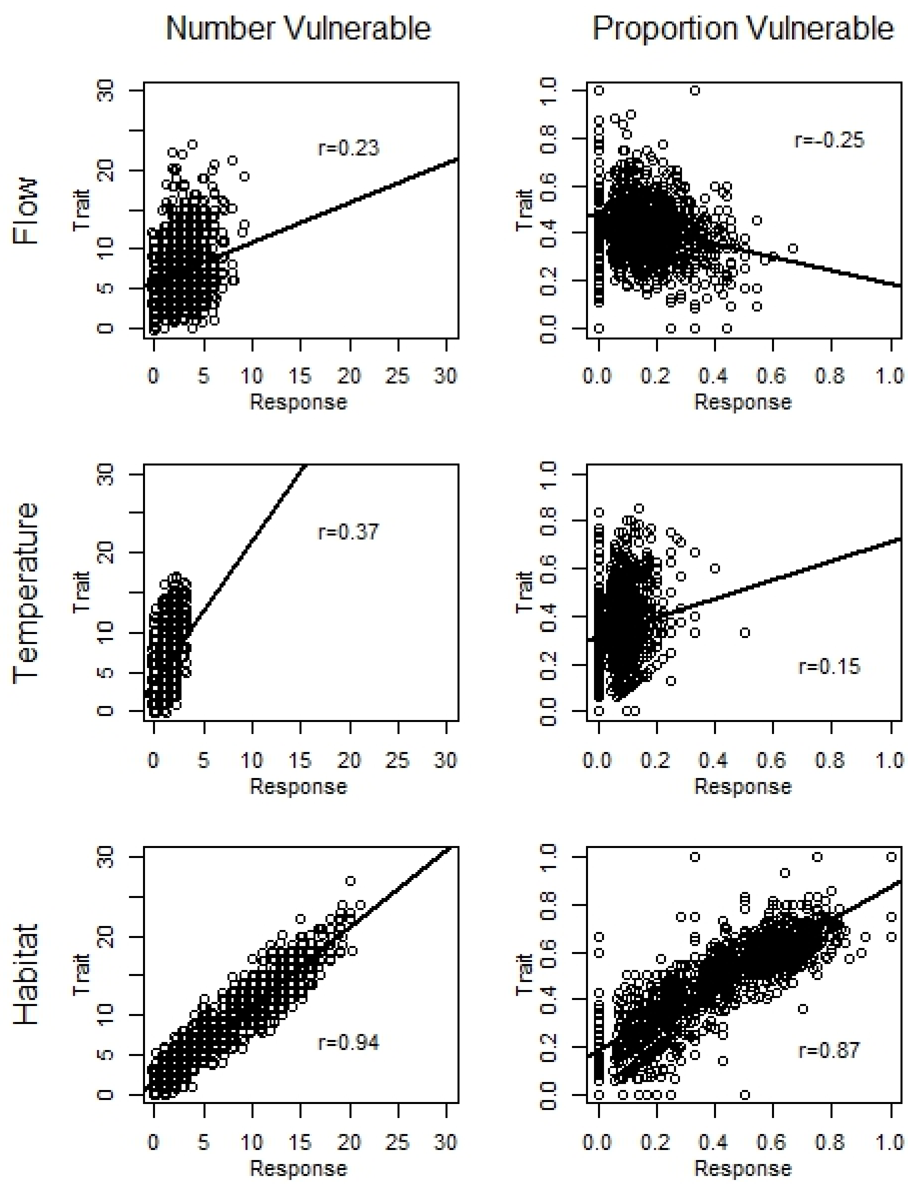

Figure 6: Correlation between the trait and response based approaches for classifying environmental tolerance based on the number and proportion of vulnerable species at each of the study sites, r = Pearson's correlation coefficient.

CHAPTER 2

EVALUATION OF MISSOURI'S CONSERVATION NETWORKS FOR STREAM FISH CONSERVATION

Nicholas Sievert, Dr. Craig P. Paukert, and Dr. Joanna Whittier

ABSTRACT

Successful conservation of stream fishes will become increasingly dependent on the use of conservation networks as the effects habitat degradation and climate change increase. We developed a framework for identifying both the most valuable stream segments for fish conservation within established conservation networks, as well as areas outside of conservation networks which best complement what is already being protected. This methodology was applied to three established conservation networks in Missouri, USA: Priority Watersheds, Conservation Opportunity Areas, and the existing conservation network. We also examined patterns in conservation values based on the landscape and habitat characteristics of sites, the impact of constraining prioritization based on the established conservation network compared to using a blank slate approach, how species representation varied in sites selected based on measure of conservation value compared to sites selected based solely on habitat integrity, and how decisions to include other variables (e.g., weighting of species, metric of upstream watershed integrity, and the use of different conservation planning algorithms) may inform conservation planning decisions. There were a number of sites with low scores based on landscape-level habitat integrity, but had conservation values in the top 10%

based on the conservation value approach. Constraining prioritization to established conservation networks generally resulted in lower species representation, however the established networks did include representation of almost all species and provided a more feasible mechanism for implementing large-scale conservation than starting from a blank slate. Species weighting had very little impact on the conservation value of stream segments, likely because highly weighted species generally had smaller distributions and were therefore retained to maintain representation regardless of weight. The inclusion of upstream habitat integrity and the prioritization technique both resulted in differences in the conservation value assigned to sites suggesting that decision makers may need to consider the implications of choices regarding these options on results. We believe this framework can be used in other regions to identify priority sites for the conservation of stream species within and complementary to established conservation networks.

INTRODUCTION

Conservation networks are important for the protection of increasingly imperiled stream communities (Saunders *et al.* 2002; Abell *et al.* 2007; Nel *et al.* 2009a). Habitat degradation, warming stream temperatures, and alterations to the flow regime are expected to contribute to future declines of stream fish (Dudgeon *et al.* 2006). Conservation plans are important for the protection of aquatic biodiversity from these threats, and have been developed for stakeholders around the globe (Abell *et al.* 2007). The primary goal of most aquatic biodiversity conservation plans is to protect the current suite of species or communities, often through reserves or protected areas (Saunders *et al.* 2002; Abell *et al.* 2007). Resources for the establishment and management of reserves and protected areas are often limited; therefore it is important to make scientifically informed decisions to achieve the greatest conservation outcomes.

Effective conservation planning depends on the consideration of both biodiversity and the amount of resources available for protection, management and research to meet goals (Linke *et al.* 2011). Typically, conservation plans for freshwater ecosystems have been centered on the development of a spatial network which supports a targeted set of species or features of interest. Because resources for conservation are often limited, these plans seek to accomplish this while minimizing cost or area by focusing on the complementarity of selected sites (Nel *et al.* 2009a). Recently, there has been a proliferation of research aimed at developing quantitative methods for creating conservation plans with an emphasis on complementarity (Moilanen *et al.* 2008, 2009a; Hermoso *et al.* 2011; Linke *et al.* 2011). Generally, these tools have been used to design conservation networks or set priorities while treating the project area as a blank slate, that is, areas are prioritized without consideration of any already established conservation networks (Sowa *et al.* 2007; Wenger *et al.* 2009; Strecker *et al.* 2011; Esselman *et al.* 2013; Pool *et al.* 2013). However, in many areas the slate is not blank, as conservation networks have already been established (e.g., state and federal wildlife management areas, refuges, and parks). For many managers and decision makers the flexibility to start from scratch and create new conservation networks or priority areas is not feasible given the costs associated with the creation of new networks, as well as the existing obligations of agencies to its stakeholders (Naughton-Treves *et al.* 2005). In areas where the slate is not blank, evaluation and prioritization techniques which consider established conservation networks constitute a more realistic, practical approach.

This study aims to provide a framework for freshwater conservation planning which incorporates established conservation networks into the planning process, with an application to the state of Missouri. The objectives of this project were to rank wadeable stream segments (based on predicted stream fish representation, species vulnerability to land use and climate change, and the upstream integrity of sites) within and complementary to three established conservation networks in Missouri: the existing conservation network (ECN), Conservation Opportunity Areas (COAs), and Priority Watersheds (PWs). This study also evaluated how decisions related to the inclusion of species weighting, species specific connectivity requirements, and the method used for prioritizing stream segments affected the results.

The ECN is comprised of public (Missouri Natural Areas, Missouri Department of Conservation (MDC) lands, US Fish and Wildlife Service Refuges, Department of Natural Resources lands, U.S. Forest Service lands, National Park Service lands, and Army Corps of Engineers lands) and private areas (Wetlands Reserve Program lands, The Nature Conservancy Preserves, Missouri Prairie Foundation lands, the Pioneer Forest, and Ozark Regional Land Trust lands), which are managed with a primary purpose of wildlife conservation (Hoskins 2005). In Missouri there are 3,943 ECN units encompassing 13,183 km^2 and 2,590 km of wadeable streams (Abbitt *et al.* 2004; Figg 2011). COAs consist of 35 units located throughout Missouri that have been identified by MDC as the best locations to achieve all wildlife conservation. These units are a mixture of public and private lands where MDC and 18 partner organizations representing national and state agencies, and national, regional, and local non-profit conservation organizations work together to utilize "technology, expertise and resources for all wildlife conservation" (Missouri Department of Conservation 2012). These COAs encompass 22,855 km^2 and 4,801 km of wadeable streams (Abbitt *et al.* 2004; Figg 2011). Each COA has a stakeholder team which develops goals and coordinates management and conservation actions for the area and attempts to facilitate conservation actions on both the public and private lands within the areas. PWs were created by the MDC Fisheries Division in 2010 by identifying focal areas across the state for conserving aquatic biodiversity and providing quality areas and opportunities for outdoor recreation. These areas were selected based on a ranking system which included Aquatic COAs, the presence of streams and impoundments managed by MDC for recreation, local buy-in from landowners, feasibility, and multiple priority achievement in overlapping areas (Corson *et al.* 2010). In total there are 58 Priority Watersheds encompassing 38,931 km^2 and 8,826 km of wadeable streams (Abbitt *et al.* 2004; Figg 2011).

MATERIALS AND METHODS

Study area

Missouri is comprised of three ecologically unique aquatic subregions: the Central Plains, Ozarks, and Mississippi Alluvial Basin (MAB) (Figure 1; Pflieger 1970; Pflieger and Missouri Deptartment of Conservation 1989; Sowa *et al.* 2007). Differences in the geology, landform, soils, land cover, and the influence of groundwater between these regions have given rise to unique habitats and fish communities in the streams of each area. The Central Plains encompasses all of northern Missouri and much of the western portion of the state, and characterized by open grassland and agricultural land use (Figure 1; Sowa *et al.* 2007). The topography is predominately composed of wide, gently sloping valleys (Sowa *et al.* 2007). Streams are generally characterized as having a relatively low gradient, high turbidity, and fine silt and sand substrates (Sowa *et al.* 2007). Most of the southern portion of Missouri is considered the Ozarks, which is predominately composed of a mix of forested areas and open pastureland (Figure 1; Sowa *et al.* 2007). The topography has high relief and rugged terrain. Streams in the Ozarks tend to have higher average stream gradients with gravel, cobble, or bedrock as dominant channel substrates with sand and silt substrates in slower moving areas and pools (Sowa *et al.* 2007). The far southeastern portion of the state is the Mississippi Alluvial Basin (MAB), which has low relief and low gradient streams (Sowa *et al.* 2007). Most of this region has been channelized and ditched for agricultural production (Figure 1). The substrate varies throughout the subregion but generally the larger and slower moving streams are characterized by fine silt while streams with faster flow are comprised of sand and small gravel substrates (Sowa *et al.* 2007).

Data

Fish community data was provided by MDC and included records from 1990 to 2011 (Figure 2). Data collected between 1990 and 1999 was contained in the MDC Fish Community Database, while data collected between 2000 and 2011 was contained in the Resource Assessment and Monitoring Program (RAM) database. Data used for this study was limited to fish community collections from wadeable streams (2^{nd}-5^{th} order and classified as permanent), and had at least 0.5 hours of effort using seines and/or electrofishing. This allowed for a scope of inference which included all wadeable, permanent streams in Missouri (Figure 2). All 769 samples within the MDC Fish Community Database were representative of community sampling efforts; while all 1,107 samples within the RAM database were sampled using a standard procedures which used block nets to enclose the sample area which was then thoroughly sampled via electrofishing and seining (Fischer and Combes 2003). Sites were selected for RAM sampling based on a stratified random approach which selected wadeable stream segments throughout Missouri (Fischer and Combes 2003). In this study both the RAM and Fish Community Databases were used to account for stream fish occurrence in a presence/absence format. In addition to the above databases that contained information on community samples, point data obtained from the Missouri Natural Heritage Program database, which was collected between 1990 and 2011, was used for species for which models could not be developed.

Environmental data was provided by the Missouri Resource Assessment Partnership (MoRAP). This geospatial data was linked to each confluence to confluence stream segment and included 27 variables related to biogeography, stream features, local landscape, upstream landscape, and anthropogenic impacts (Appendix 5; Abbitt *et al.* 2004). These variables were selected because they have been linked to fish species distributions in Missouri and elsewhere (Sowa *et al.* 2007; Strecker *et al.* 2011).

Species distribution models

Species distribution models were developed to determine representation of each species in each wadeable stream segment in Missouri. Models were developed using an ensemble approach that averaged the results of four component models which met minimum evaluation standards. The four component models included multivariate adaptive regression splines (MARs), generalized additive models (GAMs), boosted regression trees (BRTs), and random forest models. Species models were developed separately for each of Missouri's aquatic subregions because species exhibit different relationships with landscape features between subregions (Sowa *et al.* 2007). Species presence or absence served as the response variable while predictor variables were drawn from both continuous and categorical environmental data (Appendix 5). Models were developed for species which had at least 40 occurrences within a subregion. All models were developed using R statistical software using the 'earth' package (MARs; Milborrow 2014), 'gam' package (GAM; Hastie 2014), 'gbm' package (BRT; Ridgeway 2013), and the 'randomForest' package (random forest; Liaw *et al.* 2014). Each model was built from a random subset of 70% of the species occurrence data, with the remaining 30% being used for model evaluation.

Model accuracy was evaluated using the area under the receiver operator characteristic curve (AUC), calculated using the 'ROCR' package in r (Sing *et al.* 2013), model bias (number of occurrences predicted compared to observed expressed as a percentage), and model fit as the mean absolute error (MAE) of a calibration curve with 10% probability of occurrence bins. Ensemble models, which predicted probability of occurrence for all species in each wadeable stream segment in Missouri, were created by averaging the predictions of all component models which met each of the minimum evaluation standards: AUC ≥0.6, bias of +/- 25%, and MAE of ≤0.125.

Species representation

Species representation within Missouri's established conservation networks was assessed by determining the number of stream segments a species was predicted to occupy within each conservation network. Species occupancy was accounted for in two ways. For species which met the modeling requirements predicted probabilities of occurrence were drawn from the species distribution models. For species, for which acceptable models were not created, observed occurrence records were used to assign stream segments a probability of occurrence of 1 where the species had been collected, and the species frequency of occurrence for each ecological drainage unit (EDU) was then used to assign probabilities of occurrence to all unsampled stream segments. This was accomplished for each of the 17 ecological drainage units (EDUs) in Missouri by dividing the number of occurrences from the number of samples in each EDU. Predicted number of occupied stream segments within each of the established conservation networks was calculated for each species by summing the probabilities of occurrence for all stream segments which intersected a conservation network.

Conservation value

Stream segments were prioritized within and complementary to Missouri's conservation networks based on a measure of conservation value. Conservation value rankings were calculated based on weighted species representation while accounting for upstream watershed integrity using Zonation conservation planning software version 3.1 (Moilanen *et al.* 2012). Conservation values were calculated via two zonation algorithms which calculated the proportional loss of species representation based on the removal of each stream segment. The stream segment with the lowest conservation value was removed, and conservation values for all remaining cells were recalculated. This process is done iteratively until only a single stream segment remains producing a hierarchy of conservation values from the least to most valuable stream segment. This analysis was completed for a number of scenarios which incorporated different methods of accounting for representation, weighting species, and upstream integrity for each of the three conservation networks as well as the blank slate approach (i.e., assumes no established conservation networks).

Species representation was accounted for using two distinct methods: core-area Zonation and with the additive benefit function. Core-area Zonation by removing the cell which minimizes biological loss by selecting the stream segment with the lowest value for the most valuable species occurrence in that cell; this approach emphasizes unique species occurrences and generally results in better representations of uncommon species (Moilanen *et al.* 2012). The additive benefit function minimizes biological loss by selecting the stream segment that results in the lowest marginal loss of all species in a given cell for removal (Moilanen *et al.* 2012). This method emphasizes species richness and generally results in higher average species representation but lower levels of representation for uncommon species (Moilanen *et al.* 2012).

Species representation was weighted in three different ways to capture a number of options by which decision makers may value the representation of species. These weights were based on species vulnerability, species listing status, or all species weighted equally. Species were weighted based on vulnerability and listing status to emphasize the selection of areas for species in greatest need of conservation, and was compared to the analysis where species had equal weights. Vulnerability weight was based on a species vulnerability to habitat degradation, warming stream temperatures, and alterations to the flow regime while also considering a species dispersal ability, range size, rarity, and range-wide fragmentation (Chapter 1). The trait association index developed in Chapter 1 was used to calculate our vulnerability scores because it could assess all species of interest (Chapter 1). Vulnerability weights were scaled from 1 (lowest vulnerability score) to 2 (highest vulnerability score). Conservation prioritization was also completed using state listing status as a weight, where species of conservation concern received weights of 2, and unlisted species received weights of 1, as well as with all species weighted equally.

Species-specific penalty curves were developed, adapting the procedure developed by Moilanen *et al.* (2008), to account for the potential effect of habitat degradation of upstream sites on the value of a local stream segments fish community (Figure 3). Species-specific penalty curves were developed using the RAM fish community data by dividing observed species occurrences into three classes based on upstream habitat degradation (Percent of upstream watershed area classified as urban or agricultural land use). These classes were low (0-33%) which served as a baseline, medium (>33-66%), and high (>66-100%). Frequency of occurrence within each class was calculated for each species which had a minimum of ten occurrences, were then divided by the baseline occurrence rate, and values were then rounded to the nearest value of 1, 0.66, or 0.33 to represent the multiplicative change in biological value of a stream segment based on the remaining proportion of its upstream watershed. Values for each class were restricted to being no higher than the previous, more intact class, when this exception occurred the more intact class was assigned the value of the less intact class. This was done to provide a more conservative estimate of the loss of biological value and to eliminate situations in which a loss of upstream sites would result in an increase in biological value. This exception occurred for 20 of the 118 species which were evaluated. Fifteen species did not have a sufficient number of occurrences (10) for analysis so were analyzed without accounting for upstream habitat integrity. Integrity curves were then created with x-axis values of 0 (low class), 0.5 (medium class), and 1 (high class) with the associated y-axis values representing multiplicative changes in value (1, 0.66, 0.33).

Stream segments were ranked both within and complementary to Missouri's established conservation networks, which forced stream segments within each conservation network to be retained until all areas outside of the network were removed (Moilanen *et al.* 2012). This process created a ranking of stream segments outside of the established conservation network from least to most valuable respective of what is represented within the established conservation networks. Stream segments outside of the conservation network with predicted occurrences of species not well represented within the conservation network received high conservation values, particularly for species with high weights, whereas, sites with species which are already well represented within the conservation network received lower values. Stream segments within the conservation network were then removed, creating a ranking within each established conservation network. This analysis was conducted separately for Priority Watersheds, Conservation Opportunity Areas, and the existing conservation network in Missouri. In addition, this analysis was completed without accounting for any established conservation networks to develop a blank slate solution for the purpose of comparison. This analysis was conducted for each of the established conservation networks, and the blank slate, using all possible combinations of accounting for species representation (core-area and additive benefit), species weighting (vulnerability, listing status, all equal), and upstream watershed integrity (with and without).

This study analyzed whether areas of high conservation value differed by landscape-level habitat features including aquatic subregion, ecological drainage unit, Strahler stream order, segment gradient, and percent of watershed in forest, agriculture, grassland, and urban land cover for each of the three aquatic subregions, as land use was substantially different among subregions (Figure 1). Continuous features were broken down into two classes; top 10 percent of sites and remaining 90 percent of sites, while categorical features consisted of classes based on their discrete features. A one way analysis of variance (ANOVA) was performed for each feature to determine whether mean conservation values differed by segment characteristics.

The representation of species in areas selected through the use of the most basic prioritization analysis (core area, equal weights, no upstream integrity consideration) was compared to the selection of the least degraded stream segments to determine the potential benefits of applying a systematic, representation based approach. The level of degradation of a stream segment was classified based on a habitat threat index (HTI) for Missouri and surrounding states (Annis *et al.* 2009). Within each of the three established conservation networks the top ten percent of stream segments based on greatest conservation value and the top ten percent based on lowest HTI score (i.e., lowest landscape-level threats) were selected for each of the three subregions. The number of species predicted to be lacking representation in each solution, the percent of species predicted to occupy the greater number of stream segments in the solution derived using conservation value versus HTI scores, and the mean, maximum, and minimum percent difference between the conservation value and HTI derived solutions were all calculated. The top 2.5% of complementary areas selected based on the two approaches were analyzed for each established network in each of the aquatic subregions by calculating the percent of species with more predicted occupied stream segments in the solution derived using conservation value, and the average, maximum, and minimum percent differences from the HTI derived solution to the conservation value derived solution.

The level of species representation achieved by the established conservation networks was compared to an optimal network generated using the blank slate approach. The number of species expected to be represented within each established conservation network and the corresponding optimal networks were calculated. This study also determined the proportion of each species predicted occurrences within each conservation network as well as within an optimal conservation network of the same size, which was used to identify the species with the lowest proportion of its occurrences protected within each of the established conservation networks, as well as the optimal networks of the same size. The mean proportion of species occurrences protected within a conservation network was also calculated for each of the established networks and the corresponding optimal networks. These results were grouped by network and conservation prioritization technique (Cell removal algorithm: core area or additive benefit) and were analyzed using a three way ANOVA to determine if there was a difference in both the minimum and mean proportions of species occurrences between established networks and the optimal network while including connectivity and species weighting scheme as factors.

A sensitivity analysis was conducted to determine the effects of the prioritization technique, inclusion of upstream connectivity, and species weighting, on conservation value scores. Percent congruence for sites ranked in the top 1%, 5%, as well as each 10% increment between 0 and 100% of the prioritization results were calculated within and complementary to each of the conservation networks . These comparisons were made at a pairwise level in which all settings and inputs were the same except the input or setting analyzed, which allowed us to determine how sensitive the prioritization analysis is to the use of different prioritization algorithms, inclusion of connectivity, and the weighting of species.

RESULTS

Species distribution models were developed for 40 species in the Plains subregion, and 68 in the Ozarks subregion. All species for which attempts were made to create models, except Black Bullhead *Ameiurus melas* and Green Sunfish *Lepomis cyanellus* (both Plains models), had at least one of the four ensemble models which met minimum evaluation standards and were therefore represented via a distribution model (Appendix 6). All remaining species representations (44, 64, and 50 species for the Plains, Ozarks, and Mississippi Alluvial Basin respectively) had less than 40 occurrences within a subregion and were therefore classified based on occurrence locations and frequency of occurrence data. Based on the three model evaluation criteria (AUC, BIAS, and MAE) Random Forest models generated the most acceptable component models (95), followed by BRTs (89), GAMs (81), and finally MARS (76) (Table 1). The ensemble approach allowed 108 species distribution models to be created, which was a 42 percent increase over MARS alone, 33 percent increase over GAMs alone, 21 percent increase over BRTs alone, and 14 percent increase over Random Forest models alone. More detailed information regarding the species distribution models can be found in the appendices including: regional species model list (Appendix 6), ensemble components (Appendix 7), component model evaluation statistics (Appendix 8), and maps and shapefiles of results (Appendix 9).

All species were predicted to occur in at least 1 stream segment within each of Missouri's established conservation networks (Appendix 10). In stream segments within the established conservation networks 6, 8, and 7 species were predicted to be found in less than 10 stream segments for PWs, the ECN, and COAs respectively. There were 97, 90, and 90 species which were predicted to occupy greater than 100 stream segments in PWs, the ECN, and COAs respectively.

Conservation value was variable but showed patterns based on subregion and EDU (Ps<0.01; Appendix 11). All three subregions contain stream segments with both high and low conservation values (Figure 4; Appendix 11). The MAB had the highest mean conservation value (0.79) followed by the Ozarks (0.64) and Plains (0.30)

($P<0.001$). In the Plains average conservation value scores ranged from 0.42 in the Nishnabotna/Platte EDU to 0.16 in the Osage/South Grand EDU ($P<0.001$). In the Ozarks average conservation values ranged from 0.89 EDU in the Neosho EDU to 0.41 in the Moreau/Loutre EDU ($P<0.001$). Finally, in the Mississippi Alluvial Basin average conservation values ranged from 0.96 in the Black/Cache EDU to 0.78 in the St. Francis/Little EDU ($P<0.001$). Average conservation value tended to increase with increasing Strahler order for the Plains (0.23 for 2nd order to 0.44 for 5th order; $P<0.001$) and the Ozarks (0.56 2nd order to 0.76 for 5th order; $P<0.001$), and the MAB (0.78 for 5th order to 0.80 for 2nd order; $P<0.001$), but the differences in the MAB were likely not ecologically significant. The top 10 percent of highest conservation value sites in the Plains had higher forest cover in the watershed (15.6% to 12.9%; $P<0.001$), lower agriculture (39.2% to 43.7%; $P<0.001$), and lower grassland (39.2% to 43.7%; $P<0.001$) than sites in the lower 90% of conservation values. In the more forested and less agriculturally-dominated Ozarks the top 10 percent of highest conservation value sites had lower levels of agriculture in the watershed (2.1% to 4.9%; $P<0.001$), and lower stream segment gradients (2.3 m/km to 3.1m/km; $P<0.001$), but forest cover was similar (48.3% to 49.7%; $P<0.001$). In the Mississippi Alluvial Basin high value stream segments had more forest in their watersheds (13.3% to 6.8%) and less agriculture (72.7% to 80.4%) in this agriculturally-dominated subregion. These results suggest that land use and land cover was related to conservation value, but was region-specific.

Stream segments selected based on conservation value resulted in greater species representation, particularly for uncommon species when compared to sites selected based on habitat integrity alone (HTI scores) (Table 2). The number of species lacking representation in the top ten percent of sites within established conservation networks based on conservation value were one species in Priority Watersheds, two in the existing conservation network, and five in Conservation Opportunity Areas, while the number of species lacking representation in a corresponding number of sites selected based on HTI were 10 in Priority Watersheds, 13 in the existing conservation network, and 17 in Conservation Opportunity Areas. Between 70 and 77 percent of species were predicted to have higher levels of representation in the top ten percent of segments within conservation networks selected using the conservation value framework than those selected based on habitat integrity alone (Table 2). Additionally, the average the use of conservation value retained 166% more occurrences per species than using HTI for Priority Watersheds, 311% more for Conservation Opportunity Areas, and 190% more for the existing conservation network. Uncommon species had the largest increases in number of occurrences based on the selection of sites using conservation value as opposed to using habitat integrity with increases of 6,145% (0.2 predicted occurrences for sites selected based on HTI vs 15 predicted occurrences for sites selected based on conservation value) for the Redspot Chub *Nocomis asper* in the existing conservation network, and 7,099% (0.4 vs 28.9 occurrences) and 16,122% (0.1 vs 21.8 occurrences) for the Niangua Darter *Etheostoma nianguae* in Priority Watersheds and Conservation Opportunity Areas respectively. Several common species were less well represented in sites selected based on conservation value rather than HTI score including the Mottled Sculpin *Cottus bairdii* in Priority Watersheds (65% less predicted occurrences; 120 vs 47 sites), the Creek Chubsucker *Erimyzon oblongus* in Conservation Opportunity Areas (61%; 55 vs 21) and the existing conservation network (65%; 55 vs 19). The amount of species which were had more occurrences when selections were made using conservation value ranged from 70% in Conservation Opportunity Areas to 73% in both Priority Watersheds and the existing conservation network. The average percent difference ranged from 11% in Priority Watersheds to 14% in Conservation Opportunity Areas, with maximum differences ranging from 127% in Conservation Opportunity Areas to 321% in the existing conservation network and minimum differences ranging from -8% for the existing conservation network to -16% for Conservation Opportunity Areas.

Established networks retained 4.2 to 10.0% higher proportion of representation than the blank slate alternative depending on network for the species which lost the greatest proportion of their original representation in each solution developed using the additive benefit function (Table 3). When core-area Zonation was used to create blank slate alternatives there was no significant difference for Priority Watersheds (-1.0%; $P=0.195$), but both Conservation Opportunity Areas (-1.2%), and the established conservation network (-3.3%) had decreased representation when compared to the blank slate alternatives (Table 3; $Ps<0.001$). The average proportion of representation lost across all species was less in the optimal network alternatives than in the established networks using both additive benefit (Table 3; -27.4%, -11.7%, and -23.5% change for Priority Watersheds, Conservation Opportunity Areas, and the existing conservation network respectively), and core-area Zonation (Table 3; -15.9%, -5.7%, and

-12.6% change for Priority Watersheds, Conservation Opportunity Areas, and the existing conservation network respectively). Therefore, developing a network using a blank slate approach typically resulted in a lower reduction of species distribution compared to established conservation networks.

Our assessment of the influence of connectivity, prioritization technique, and species weighting on the conservation ranking of stream segments revealed that species weighting and connectivity had less influence on the results than prioritization technique. Conservation scores of stream segments determined with and without connectivity revealed high levels of congruence even among the most valuable sites (Figure 5A). Congruence analysis of the conservation scores of stream segments using core-area Zonation compared to the additive benefit function revealed lower levels of congruence suggesting the choice of prioritization technique can have substantial effects on the outcome (Figure 5B). Finally, congruence of conservation rankings was relatively high between all pairwise combinations of species weighting (none, vulnerability, listing status) suggesting that the weighting of species had only a moderate effect on the conservation solutions (Figure 5C, D, E).

DISCUSSION

This study provides a framework that may be a more realistic and effective approach for prioritizing stream fish conservation efforts over a broad spatial scale when conservation networks have already been established. This framework, may serve as a more feasible option for conservation planning than previous work in the area which has largely used a blank slate approach (Sowa *et al.* 2007; Wenger *et al.* 2009; Strecker *et al.* 2011; Esselman *et al.* 2013; Pool *et al.* 2013). Budgets for agencies tasked with managing and creating conservation areas and protecting stream fish are often limited, necessitating the use of efficient prioritization schemes for achieving the greatest conservation outcomes (Naughton-Treves *et al.* 2005). Therefore, the ability to prioritize stream fish conservation within established networks may be critical, as agencies typically do not have the resources necessary to implement stream fish conservation actions for all stream segments within conservation networks. By incorporating established conservation networks in the planning process, stream segments which complement what is already being protected can also be identified. This method emphasizes the selection of areas which have occurrences of species which may not already be well represented within an established network resulting in an expanded network with a more comprehensive and diverse assemblage of species. This framework allows managers and decision makers to make more informed decisions regarding the spatial prioritization of stream fish conservation.

Analysis revealed that Missouri's established conservation networks typically do contain the vast majority of fish species in the state. All 133 species which were evaluated were predicted to be represented within each of Missouri's established conservation networks except Black Buffalo, *Ictiobus niger*, a species which is more commonly found in rivers than the wadeable streams that were sampled for this study (Pflieger 1997). Protecting multiple areas to allow for redundancy in case of catastrophic declines and changes in habitat due to climate change, invasive species, or anthropogenic disturbances may be needed to ensure species persistency (Stein *et al.* 2000), and our study revealed that the majority (68-72%, depending on conservation network) of species were predicted to occur in over 100 stream segments in each of the established conservation networks suggesting they are relatively secure. However 5 to 7 species (depending on conservation network) were predicted to occur within less than 10 stream segments, suggesting their long-term protection is less secure. These less secure species often had restricted distributions such as the Channel Darter *Percina copelandi*, Arkansas Saddled Darter *Etheostoma euzonum*, Flier *Centrarchus macropterus* and Bluntface Shiner *Cyprinella Camura*, were rare or difficult to detect such as the Southern Brook Lamprey *Icthyomyzon gagei*, and Northern Brook Lamprey *Icthyomyzon fossor*, or were primarily found in riverine habitats rather than wadeable streams such as the Black Buffalo *Ictiobus niger*, Western Silvery Minnow *Hybognathus argyritis*, Plains Minnow *Hybognathus placitus*, and Highfin Carpsucker *Carpiodes velifer* (Pflieger 1997; Galat *et al.* 2005). When possibilities for network expansion arise targeting areas with known or predicted occurrences of underrepresented species may improve the long-term outlook for those species, and bolster the comprehensiveness of the established network.

The use of surrogates, such as landscape and habitat characteristics, is widely used in conservation planning, particularly when access to biological data is limited (Nel *et al.* 2009b). The findings of this study suggest that results from habitat-based assessment may not produce the same results as assessments based on species representation. High value areas which are excluded when sites are selected based on landscape and habitat surrogates are likely to include species which are restricted to non-surrogate habitat classes and rare species. This study found that stream segments of both high and low conservation value were distributed throughout the state and in a variety of stream sizes, ecological subregions, drainage basins, and land use classes. Selecting sites based only on high value habitat surrogates would have meant that some stream segments in the top 10 percent of conservation values, belonging to groups associated with the lowest average conservation values (Plains aquatic subregion, the Osage/South Grand, Grand Chariton, and Blackwater/Lamine drainages, 2^{nd} order streams, and agricultural watersheds) would have been excluded. Species restricted to these areas, and rare species would have much lower representations if a prioritization system using only landscape and habitat surrogates was used. Although the use of landscape and habitat surrogates are a useful alternative when sufficient data on species representation is not available (Trakhtenbrot and Kadmon 2005), a systematic conservation planning approach which includes species representation may be important for comprehensive species conservation (Margules and Pressey 2000; Linke *et al.* 2011). In cases where species representation data is not available the results of this study suggest that landscape and habitat based surrogates do a reasonable job of identifying sites with high conservation values. However, when sufficient biological data is available, representation based, systematic conservation planning is likely to result in solutions which better represent rare species and species which occupy habitats not associated with high value surrogates.

One commonly used method for creating protected areas is the selection of areas which are pristine or have minimal anthropogenic impacts, often because these areas are rugged, isolated, or have limited economic value (Margules and Pressey 2000). A number of studies have developed tools for assessing the ecological integrity of catchments (Mattson and Angermeier 2007; Annis *et al.* 2009; Paukert *et al.* 2011), and although we believe this information is a valuable component of conservation planning, our results suggest that relying on threat data alone, without consideration of species representation, may result in limited or no protection for many species. Using this studies measure of conservation value for prioritizing sites yielded much higher predicted levels of species representation (9 to 12 more species were represented in the top 10% of sites within each conservation network) than if stream segments would have been selected based solely on a measure of habitat integrity (Annis *et al.* 2009). This suggests that systematic, representation based conservation planning is likely to maintain higher levels of representation for the majority of species (species averaged between 166-311% greater representation in the top 10% of sites within each of conservation network) and particularly so for uncommon species both within and complementary to established conservation networks. In order to maintain comprehensive species representation prioritization of sites based on threat data alone is likely insufficient.

Systematic conservation planning often considers a variety of variables, in addition to species representation, to account for factors which influence conservation success (Moilanen *et al.* 2008; Leathwick *et al.* 2010; Strecker *et al.* 2011; Pool *et al.*

2013). The weighting of species is commonly used in conservation planning which allows emphasis to be placed on sites which are important for priority species (Moilanen *et al.* 2005; Early and Thomas 2007; Gordon *et al.* 2009; Carroll *et al.* 2010; Summers *et al.* 2012). However, the results showed that weighting species had a relatively minor effect on the conservation values of stream segments. Species which were weighted highly also tended to be the least common, and because this study's methodology placed emphasis on maintaining representations of all species, stream segments with occurrences of uncommon species were given high conservation priority, regardless of weighting. The lack of sensitivity to species weighting suggests that uncertainty or differences in weighting schemes are unlikely to have substantial impacts on conservation priorities for others employing this framework.

Upstream habitat integrity appeared to have an intermediate impact on the prioritization of sites both within and complementary to established conservation networks. Stream fish communities are heavily influenced by conditions in the upstream watershed (Weaver and Garman 1994; Wang *et al.* 1997, 2001; Wenger *et al.* 2008). Many conservation plans incorporate upstream habitat integrity requirements (Moilanen *et al.* 2008; Hermoso *et al.* 2011; Strecker *et al.* 2011; Esselman *et al.* 2013), but there has been little evaluation of how the inclusion of connectivity influences results. The inclusion of connectivity in these analyses tended to increase the value of upstream segments and showed some additional clustering when compared to results where connectivity was not included.

The results of this study suggest that the choice of prioritization technique had a substantial impact on the results of the analysis. A variety of prioritization techniques have been developed which assign conservation values in a number of unique ways (Moilanen 2007; Moilanen *et al.* 2009b; Watts *et al.* 2009). The use of two prioritization techniques, which 1) emphasizes unique species representation (core area), and 2) emphasizes species richness (additive benefit; Moilanen *et al.* 2012) showed substantial differences (<60% congruence for top 10 percent of sites), and the proportion of representation lost by the average species and the species which lost the greatest proportion of its predicted occurrences. Prioritizations performed using the core-area implementation had higher levels of representation for the worst off species, while those performed using the additive benefit function maintained higher levels of representation for the average species, as was expected based on the literature (Moilanen *et al.* 2012). The sensitivity of conservation value rankings to the choice of prioritization technique suggests that researchers and decision makers should carefully consider their objectives when determining which approach to use. If the primary objective of the prioritization is to maintain representation of all species the core-area implementation is likely the most appropriate approach, while if the primary objective is to select stream segments with higher species richness and retain higher average levels of species representation implementation the additive benefit function may be a better approach.

Our study developed tools which can be used by managers and decision makers to prioritize management, land acquisitions, and guide the formation of partnerships for stream fish conservation in Missouri, and an approach for conservation prioritization which can be applied to other regions. Stream fish face a number of threats including habitat degradation, climate change, invasive species, and habitat fragmentation (Strayer and Dudgeon 2010). In order to conserve aquatic biodiversity in the face of these threats managers undertake a variety of management actions such as riparian restoration, watershed management, and barrier removal (Roni *et al.* 2002). Conservation planning techniques can help identify where these actions can be taken in order to achieve the greatest returns. The conservation value of complementary sites to established conservation networks may be used to identify opportunities for land acquisition or partnerships which offer the greatest value.

We believe this framework for incorporating established conservation networks into the conservation prioritization process can be used for aquatic biodiversity conservation for regional or national prioritization projects around the globe. In the United States data on protected areas is widely available through the National Gap Analysis Program Protected Area Database (DellaSala *et al.* 2001), which contains 635,500 geospatial records from 49 agency types including state and federal agencies, municipal governments, and non-profit organizations for all 50 states (Duarte 2012). Protected and priority areas, such as South Africa's National Freshwater Ecosystem Priority Areas (Nel *et al.* 2011), Europe's Natura 2000 sites (Commission of the European Communities 2002), and State Wildlife Action Plans in the United States (Pugh and Hall 2006) among many others represent a vast set of established networks upon which this framework for prioritizing stream fish conservation could be applied. The ability to consider established conservation networks during the prioritization process adds realism to the systematic conservation planning process. In regions where sufficient data exists to estimate the distributions of aquatic biodiversity, this framework can be utilized to inform management decisions within established networks, and can guide land acquisition and partnerships in selecting areas which best complement what is already being protected.

REFERENCES

Abbitt, R., Annis, G. and Sowa, S. (2004) Missouri Streams with Landscape Variables. Missouri Resource Assessment Partnership, Columbia, Missouri.

Abell, R., Allan, J. and Lehner, B. (2007) Unlocking the potential of protected areas for freshwaters. *Biological Conservation* **134**, 48–63.

Annis, G., Sowa, S. and Diamond, D. (2009) Developing Synoptic Human Threat Indices for Assessing the Ecological Integrity of Freshwater Ecosystems in EPA Region 7. 192 pp. Missouri Resource Assessment Program, Columbia, Missouri.

Blodgett, C. and Lea, R. (2005) Missouri Landcover Project. Available at: http://morap.missouri.edu/index.php/land-cover/ [Accessed September 15, 2012].

Carroll, C., Dunk, J.R. and Moilanen, A. (2010) Optimizing resiliency of reserve networks to climate change: multispecies conservation planning in the Pacific Northwest, USA. *Global Change Biology* **16**, 891–904.

Commission of the European Communities (2002) Commission Working Document on Natura 2000. European Commission on the Environment, Brussels.

Corson, A., Blanchard, P. and Calvert, P. (2010) Strategies for Watershed Management. Missouri Department of Conservation, Jefferson City, MO.

DellaSala, D.A., Staus, N.L., Strittholt, J.R., Hackman, A. and Iacobelli, A. (2001) An updated protected areas database for the United States and Canada. *Natural Areas Journal* **21**, 124–135.

Duarte, L. (2012) Protected Areas Database of the United States (PADUS) version 1.3.

Dudgeon, D., Arthington, A., Gessner, M., *et al.* (2006) Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews* **81**, 163–182.

Early, R. and Thomas, C.D. (2007) Multispecies conservation planning: identifying landscapes for the conservation of viable populations using local and continental species priorities. *Journal of Applied Ecology* **44**, 253–262.

Esselman, P.C., Edgar, M., Breck, J., Hay-Chmielewski, E.M. and Wang, L. (2013) Riverine connectivity, upstream influences, and multi-taxa representation in a conservation area network for the fishes of Michigan, USA. *Aquatic Conservation: Marine and Freshwater Ecosystems* **23**, 7–22.

Figg, D. (2011) Conservation Priorities Decision Support Tool. Missouri Department of Conservation, Jefferson City, Missouri.

Fischer, S. and Combes, M. (2003) *Resource Assessment and Monitoring Program: Standard Operation Procedures - Fish Sampling*. Missouri Department of Conservation.

Galat, D.L., Berry, C.R., Gardner, W.M., *et al.* (2005) Spatiotemporal patterns and changes in Missouri River fishes. American Fisheries Society Symposium. **45**, 249-291.

Gordon, A., Simondson, D., White, M., Moilanen, A. and Bekessy, S.A. (2009) Integrating conservation planning and landuse planning in urban landscapes. *Landscape and Urban Planning* **91**, 183–194.

Hastie, T. (2014) *gam: Generalized Additive Models*. R Package. V. 1.09.1.

Hermoso, V., Linke, S., Prenda, J. and Possingham, H. (2011) Addressing longitudinal connectivity in the systematic conservation planning of fresh waters. *Freshwater Biology* **56**, 57–70.

Hoskins, J. (2005) Missouri’s Comprehensive Wildlife Strategy. Missouri Department of Conservation.

Leathwick, J., Moilanen, A., Ferrier, S. and Julian, K. (2010) Complementarity-based conservation prioritization using a community classification, and its application to riverine ecosystems. *Biological Conservation* **143**, 984–991.

A. Liaw and M. Wiener (2002). Classfication and Regression by randomForest. R News **2**(3), 18-22.

Linke, S., Turak, E. and Nel, J. (2011) Freshwater conservation planning: the case for systematic approaches. *Freshwater Biology* **56**, 6–20.

Margules, C. and Pressey, R. (2000) Systematic conservation planning. *Nature* **405**, 243–253.

Mattson, K.M. and Angermeier, P.L. (2007) Integrating human impacts and ecological integrity into a risk-based protocol for conservation planning. *Environmental Management* **39**, 125–138.

Milborrow, S. (2014) *earth: Multivariate Adaptive Regression Spline Models*. R Package. V. 3.2-7.

Missouri Department of Conservation (2012) Conservation Opportunity Areas. Available at: mdc.mo.gov/node/3330 [Accessed June 27, 2012].

Moilanen, A. (2007) Landscape Zonation, benefit functions and target-based planning: Unifying reserve selection strategies. *Biological Conservation* **134**, 571–579.

Moilanen, A., Franco, A.M., Early, R.I., Fox, R., Wintle, B. and Thomas, C.D. (2005) Prioritizing multiple-use landscapes for conservation: methods for large multi-species planning problems. *Proceedings of the Royal Society B: Biological Sciences* **272**, 1885–1891.

Moilanen, A., Kujala, H. and Leathwick, J.R. (2009a) The Zonation framework and software for conservation prioritization. In A. Moilanen, K. Wilson, and H. Possingham (eds.), *Spatial Conservation Prioritization*, 196–210.

Moilanen, A., Leathwick, J. and Elith, J. (2008) A method for spatial freshwater conservation prioritization. *Freshwater Biology* **53**, 577–592.

Moilanen, A., Meller, L., Leppanen, J., Pouzols, F.M., Arponen, A. and Kujala, H. (2012) *Zonation Conservation Planning Software*. Biodiversity Conservation Informatics Group, University of Helsinki, Finland.

Moilanen, A., Wilson, K.A. and Possingham, H.P. (2009b) *Spatial conservation prioritization: quantitative methods and computational tools*. Oxford University Press Oxford, UK.

Naughton-Treves, L., Holland, M.B. and Brandon, K. (2005) The Role of Protected Areas in Conserving Biodiversity and Sustaining Local Livelihoods. *Annual Review of Environment and Resources* **30**, 219–252.

Nel, J.L., Murray, K., Maherry, A., *et al.* (2011) Technical Report for the National freshwater Ecosystem Priority Areas Project. WRC Report No. 1801/2/11.

Nel, J., Reyers, B., Roux, D. and Cowling, R. (2009a) Expanding protected areas beyond their terrestrial comfort zone: Identifying spatial options for river conservation. *Biological Conservation* **142**, 1605–1616.

Nel, J., Roux, D., Abell, R., *et al.* (2009b) Progress and challenges in freshwater conservation planning. *Aquatic Conservation: Marine and Freshwater Ecosystems* **19**, 474–485.

Paukert, C., Pitts, K., Whittier, J. and Olden, J. (2011) Development and assessment of a landscape-scale ecological threat index for the Lower Colorado River Basin. *Ecological Indicators* **11**, 304–310.

Pflieger, W. (1970) A distributional study of Missouri fishes. University of Kansas Publications, Museum of Natural History. **20** (3), 225-570.

Pflieger, W. (1997) *The Fishes of Missouri*, Second Edition. Conservation Commission of the State of Missouri, Jefferson City, MO.

Pflieger, W.L. and Missouri Deptartment of Conservation (1989) *Aquatic community classification system for Missouri*. Missouri Dept. of Conservation.

Pool, T.K., Strecker, A.L. and Olden, J.D. (2013) Identifying Preservation and Restoration Priority Areas for Desert Fishes in an Increasingly Invaded World. *Environmental Management* **51**, 631–641.

Pugh, C. and Hall, D. (2006) State Wildlife Action Plans, From Vision to On-the-Ground Action. 116 pp. Association of Fish & Wildlife Agencies.

Ridgeway, G. (2013) *gbm: Generalized Boosted Regression Models*. R Package. V. 2.1.

Roni, P., Beechie, T.J., Bilby, R.E., Leonetti, F.E., Pollock, M.M. and Pess, G.R. (2002) A Review of Stream Restoration Techniques and a Hierarchical Strategy for Prioritizing Restoration in Pacific Northwest Watersheds. *North American Journal of Fisheries Management* **22**, 1–20.

Saunders, D., Meeuwig, J. and Vincent, A. (2002) Freshwater Protected Areas: Strategies for Conservation. *Conservation Biology* **16**, 30–41.

Sing, T., Sander, O., Beerenwinkel, N. and Lengauer, T. (2013) *ROCR: Visualizing the performance of scoring classifiers*. R Package. V. 1.0-5.

Sowa, S.P., Annis, G., Morey, M.E. and Diamond, D.D. (2007) A Gap Analysis and Comprehensive Conservation Strategy for Riverine Ecosystems of Missouri. *Ecological Monographs* **77**, 301–334.

Stein, B.A., Kutner, L.S. and others (2000) *Precious heritage: The status of biodiversity in the United States*. Oxford University Press.

Strayer, D.L. and Dudgeon, D. (2010) Freshwater biodiversity conservation: recent progress and future challenges. *Journal of the North American Benthological Society* **29**, 344–358.

Strecker, A., Olden, J., Whittier, J. and Paukert, C. (2011) Defining conservation priorities for freshwater fishes according to taxonomic, functional, and phylogenetic diversity. *Ecological Applications* **21**, 3002–3013.

Summers, D.M., Bryan, B.A., Crossman, N.D. and Meyer, W.S. (2012) Species vulnerability to climate change: impacts on spatial conservation priorities and species representation. *Global Change Biology* **18**, 2335–2348.

Trakhtenbrot, A. and Kadmon, R. (2005) Environmental cluster analysis as a tool for selecting complementary networks of conservation sites. *Ecological Applications* **15**, 335–345.

Wang, L., Lyons, J., Kanehl, P. and Bannerman, R. (2001) Impacts of urbanization on stream habitat and fish across multiple spatial scales. *Environmental Management* **28**, 255–266.

Wang, L., Lyons, J., Kanehl, P. and Gatti, R. (1997) Influences of Watershed Land Use on Habitat Quality and Biotic Integrity in Wisconsin Streams. *Fisheries* **22**, 6–12.

Watts, M.E., Ball, I.R., Stewart, R.S., *et al.* (2009) Marxan with Zones: software for optimal conservation based land-and sea-use zoning. *Environmental Modelling & Software* **24**, 1513–1521.

Weaver, L.A. and Garman, G.C. (1994) Urbanization of a Watershed and Historical Changes in a Stream Fish Assemblage. *Transactions of the American Fisheries Society* **123**, 162–172.

Wenger, S.J., Hagler, M.M. and Freeman, B.J. (2009) Prioritizing areas of the Conasauga River sub-basin in Georgia and Tennessee for preservation and restoration. *Proceedings of the Southeastern Fishes Council* **51**, 31–38.

Wenger, S.J., Peterson, J.T., Freeman, M.C., Freeman, B.J. and Homans, D.D. (2008) Stream fish occurrence in response to impervious cover, historic land use, and hydrogeomorphic factors. *Canadian Journal of Fisheries and Aquatic Sciences* **65**, 1250–1264.

TABLES

Table 5: Summary statistics for the evaluation metrics for each of the four component model types. Accept= Percent of models which met minimum evaluation criteria; AUC= Area under the receiver operating characteristic curve; MAE= Mean absolute error; MARS= Multivariate adaptive regression splines; GAM= Generalized additive model; BRT= Boosted regression tree; RndmFrst= Random forest

	AUC		Bias		MAE	
Model	Mean (Range)	Accept	Mean (Range)	Accept	Mean (Range)	Accept
MARS	0.78 (0.44-1.00)	92.6 %	1.00 (0.52-1.73)	83.3 %	0.08 (0.03-0.20)	88.0%
GAM	0.80 (0.52-0.97)	94.4 %	0.97 (0.57-1.85)	84.3 %	0.08 (0.02-0.21)	91.7 %
BRT	0.83 (0.48-1.00)	99.1 %	1.05 (0.74-2.49)	87.9 %	0.07 (0.03-0.17)	95.3 %
RndmFrst	0.83 (0.53-1.00)	97.2 %	0.99 (0.61-1.59)	89.8 %	0.08 (0.04-0.24)	94.4 %

Table 6: Comparison of sites selected based on habitat threat index (HTI; Annis *et al.* 2009) versus conservation value scores. Percent difference was calculated based on the number of stream segments predicted to be occupied for a species for segments selected based on conservation value versus segments selected based on HTI, where positive % differences represent increased representation when using conservation value and negative % differences represent decreased representation when using conservation value. The number of species unrepresented is the number of species which are not predicted to be represented using either HTI or conservation value for stream segment selection. PW=Priority Watersheds; COA=Conservation Opportunity Areas; ECN=existing conservation network

	PW	COA	ECN
Top 10% of Conservation Network			
% Species Better Represented by Conservation Value	77	73	70
Mean % Difference	166	311	190
Max % Difference	7,099	16,122	6,145
Min % Difference	-61	-61	-65
# Species Unrepresented (HTI)	10	17	13
# Species Unrepresented (Conservation Value)	1	5	2
Entire Conservation Network and Top 2.5% Complementary Areas			
% Species Better Represented by Conservation Value	73	70	73
Mean % Difference	11	14	11
Max % Difference	199	127	321
Min % Difference	-14	-16	-8

Table 7: Comparison of the mean proportion of distribution lost for 1) the species which has lost the highest proportion of distribution and 2) the average proportion of distribution lost when restricted to each of the conservation networks compared to proportional distribution lost when restricted to an optimal network of equivalent size. P values calculated using a three way anova based on weighting method, connectivity, and current/optimal network. PW=Priority Watershed, COA=Conservation Opportunity Area, ECN=Existing Conservation Network. Method acronyms: CAZ=Core Area Zonation, AdBen=Additive Benefit Function.

		Species with Highest Proportion Lost				Average Proportion Lost			
Network	**Method**	**Network Mean**	**Optimal Mean**	**Percent Change**	**P**	**Network Mean**	**Optimal Mean**	**Percent Change**	**P**
PW	CAZ	0.899	0.890	-1.0%	0.195	0.730	0.614	-15.9%	<0.001
	AdBen	0.899	0.991	10.2%	<0.001	0.730	0.530	-27.4%	<0.001
COA	CAZ	0.955	0.944	-1.2%	0.013	0.804	0.758	-5.7%	<0.001
	AdBen	0.955	0.995	4.2%	<0.001	0.804	0.710	-11.7%	<0.001
ECN	CAZ	0.932	0.901	-3.3%	0.006	0.744	0.650	-12.6%	<0.001
	AdBen	0.932	0.992	6.4%	<0.001	0.744	0.569	-23.5%	<0.001

FIGURES

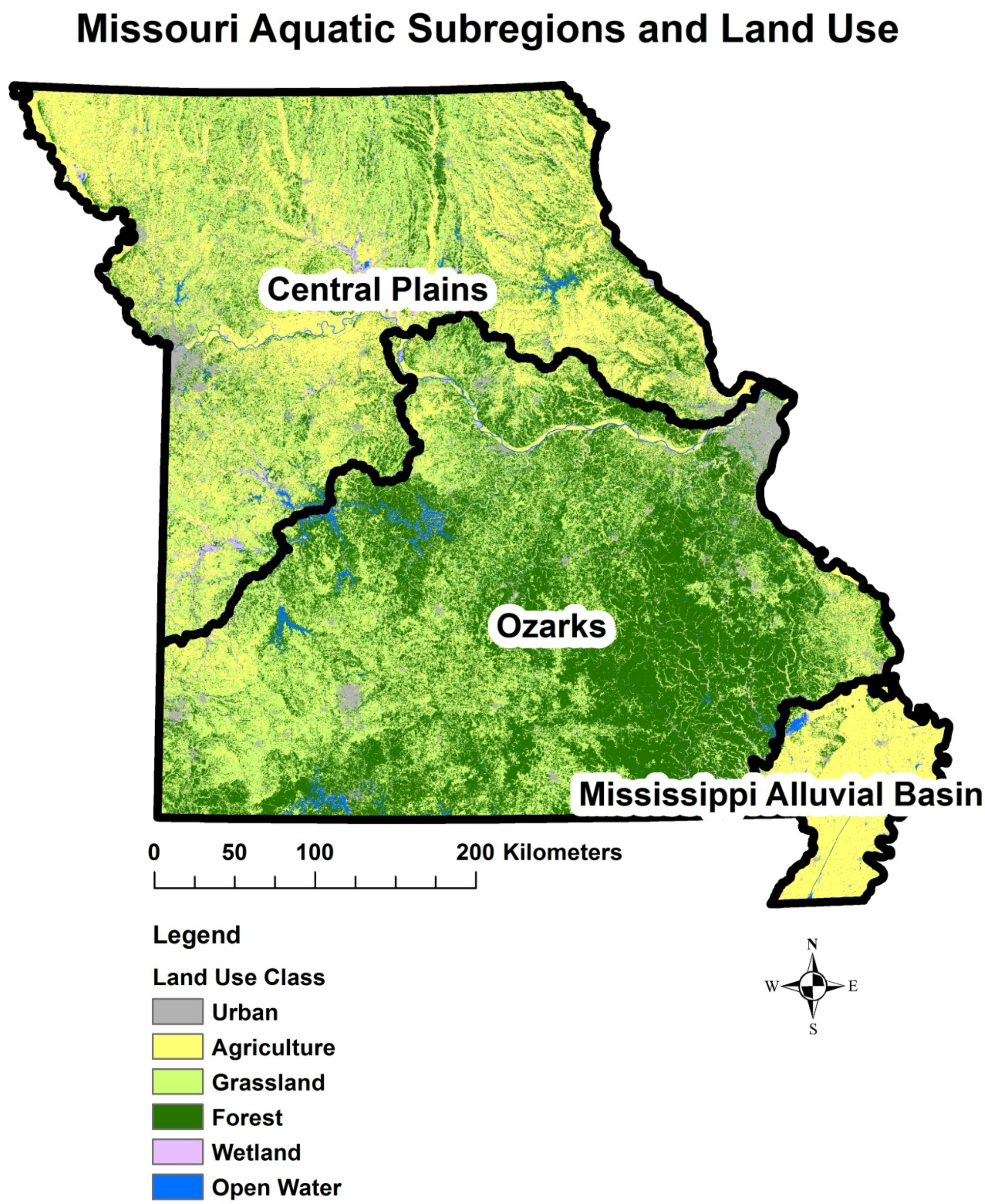

Figure 7: Map showing Missouri's aquatic subregions and land use classifications.

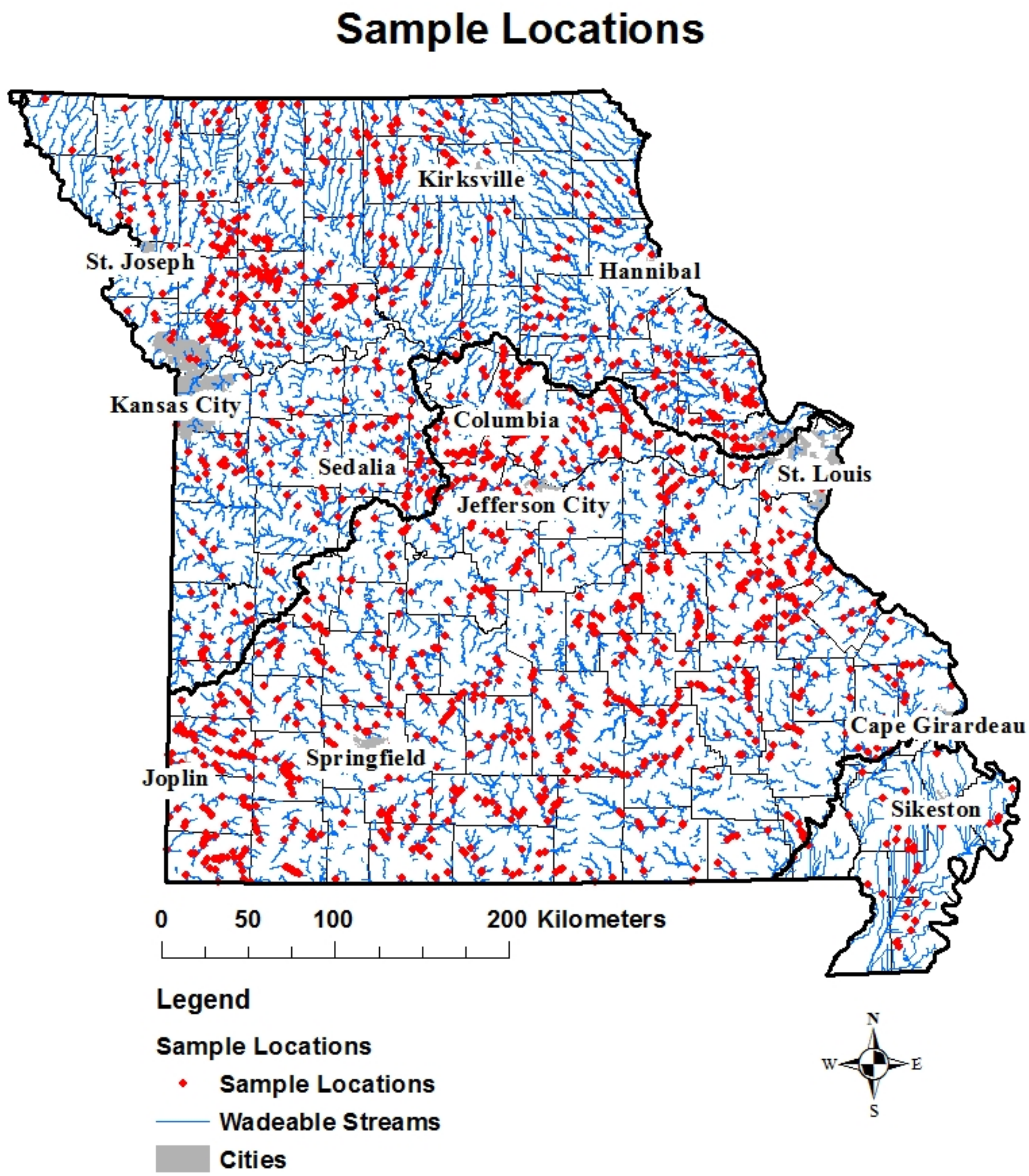

Figure 8: Map of fish community sample locations, collected in Missouri from 1990 to 2010, which were used in this study.

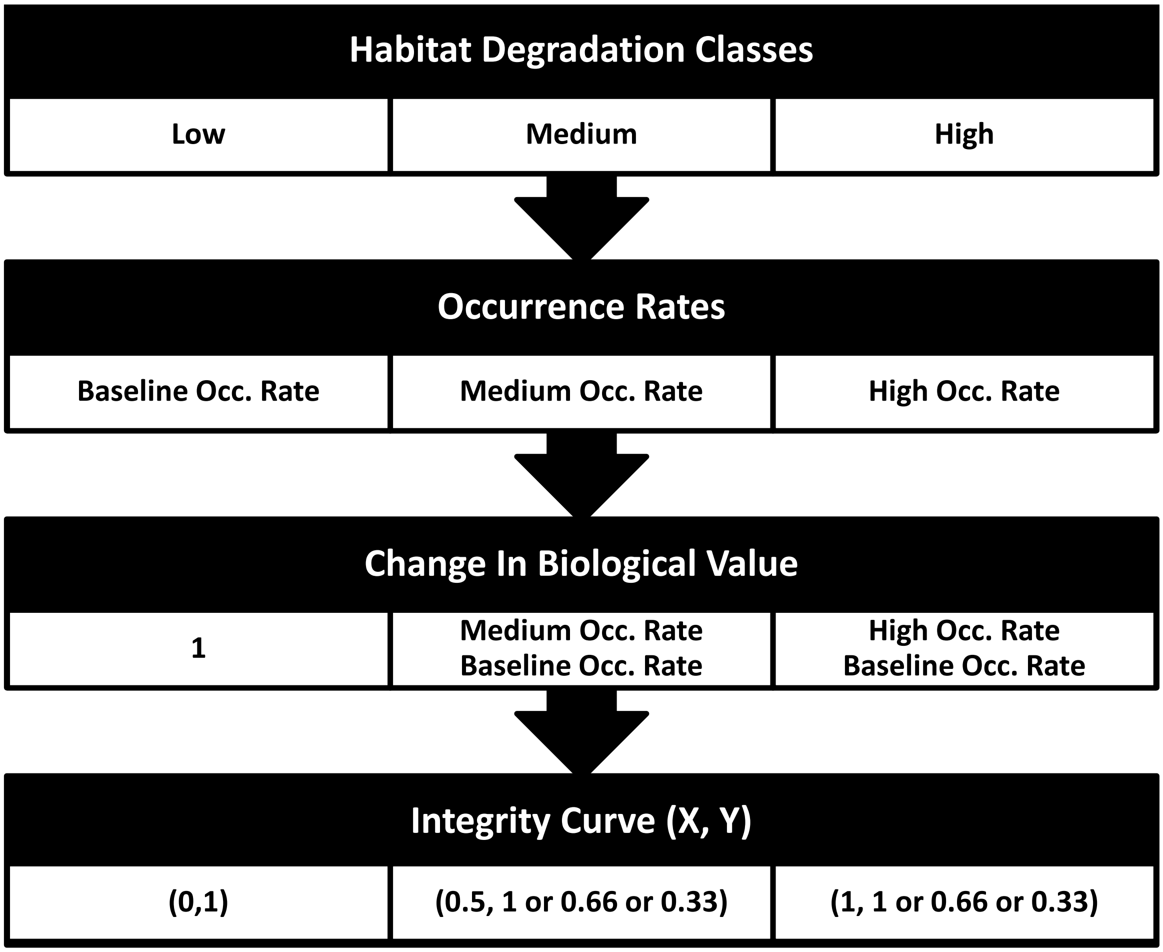

Figure 9: Framework used for developing upstream habitat integrity penalty curves.

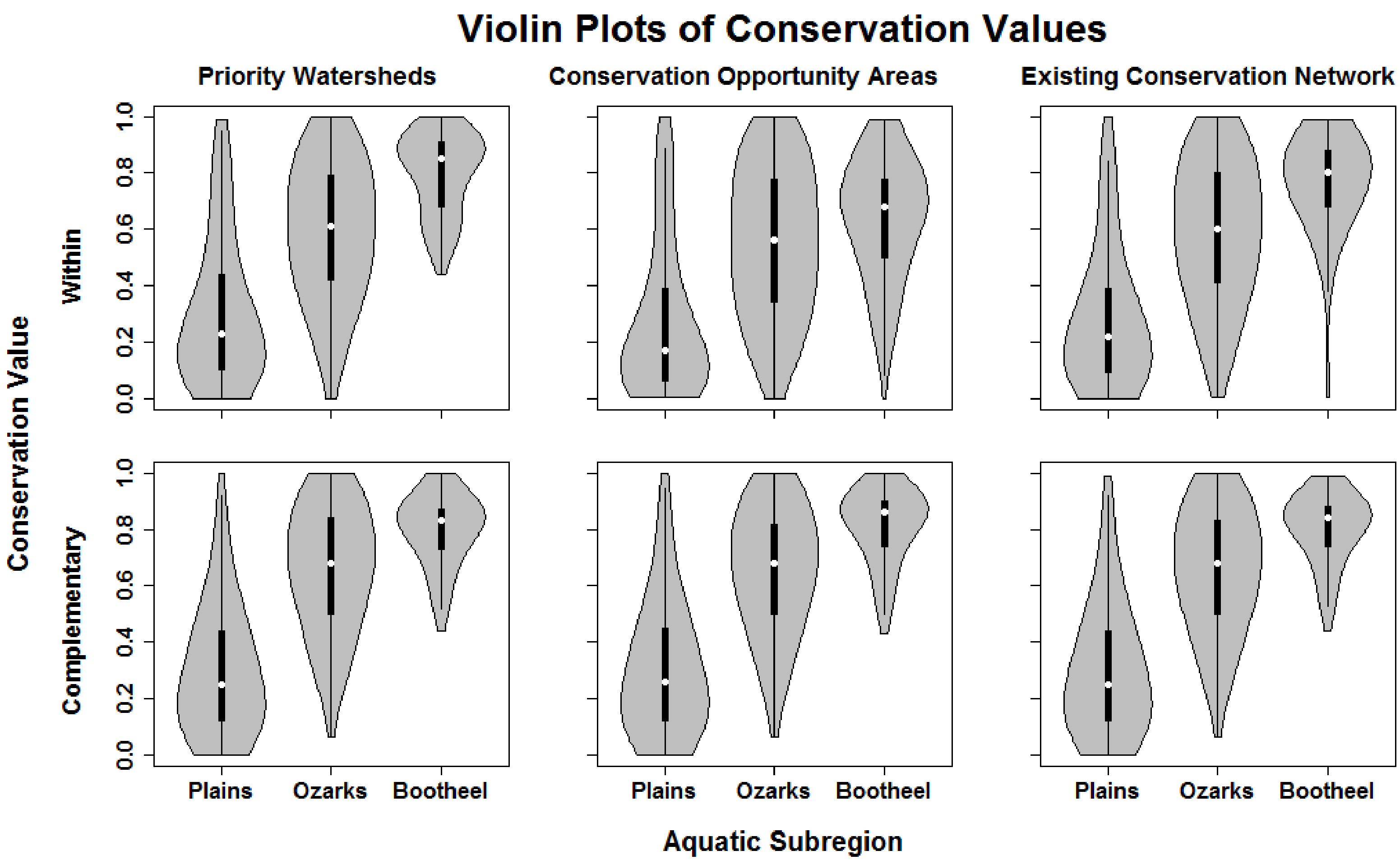

Figure 10: Violin plots of conservation value scores by subregion, separated by conservation network and within/complementary. The white point represents the median value, black bar represents the interquartile range, black line represents the overall range, and gray shaded area is a kernel density plot.

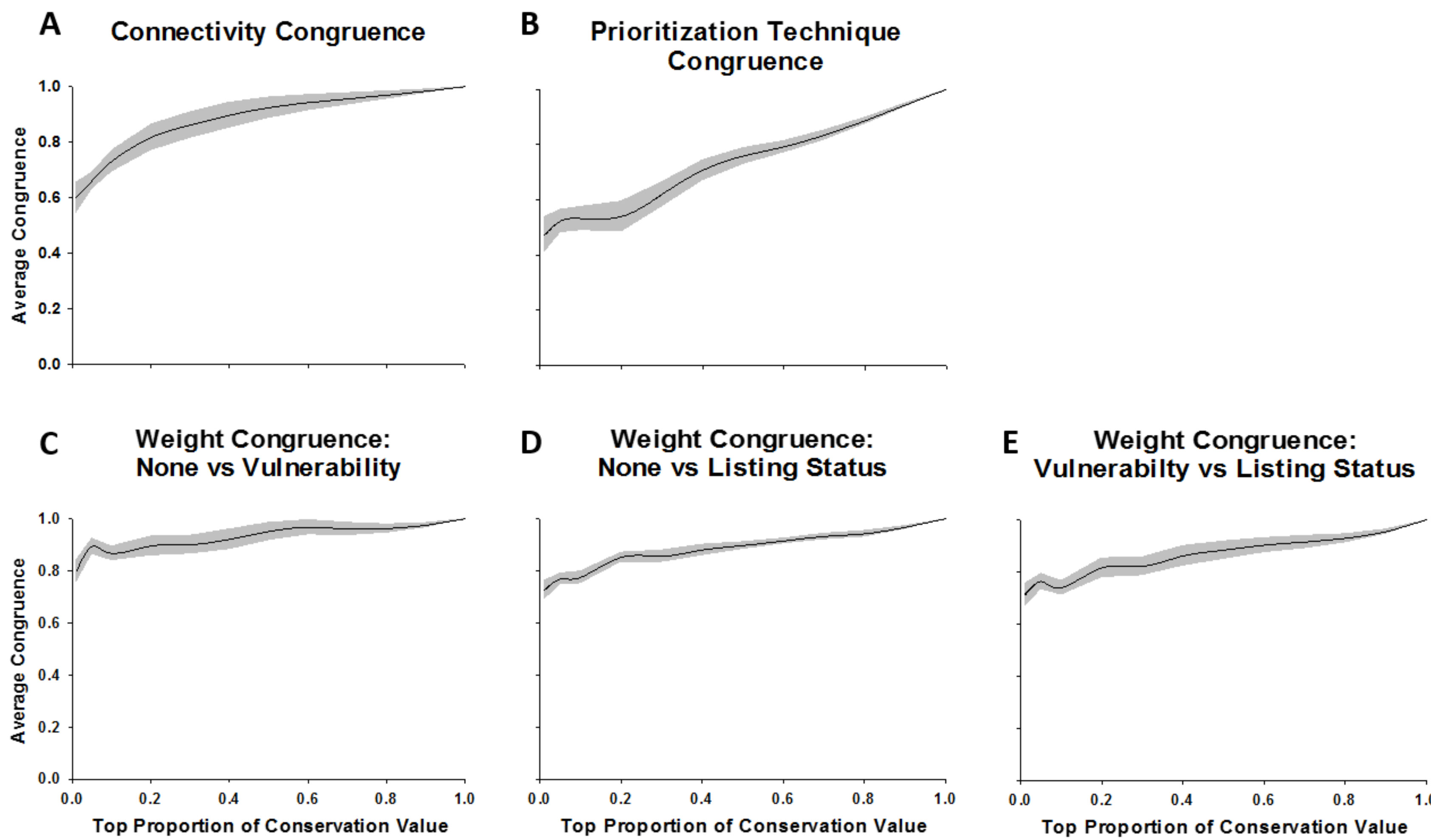

Figure 11: Plot of average percent congruence for the top proportion of conservation values with and without upstream connectivity requirements (A), each prioritization technique (B), and pairwise comparisons of each of the vulnerability approaches (C, D, E). Shaded areas represent the 95% confidence interval.

CONCLUSION

Our results may be used to support the management and conservation of stream fish in Missouri and can provide a framework for assessments in other regions. Our research suggested that the degree to which stream fish are vulnerable to climate change is likely to vary among species, and that different approaches to classifying vulnerability are likely to achieve different results (Chapter 1). We also found that high priority areas for stream fish conservation could be identified both within and complementary to established conservation networks (Chapter 2). We believe the ability to identify vulnerable fish species and areas which are important for their conservation is critical to the protection of stream fish species.

Our analysis of stream fish vulnerability revealed that although we have a relatively strong understanding of the role habitat degradation plays in species vulnerability, there is a great need for additional research to examine the effects of stream warming and alterations of flow regimes on stream fish. Our comparison of trait versus measured response-based approaches to assigning environmental tolerances yielded substantial discrepancies. Currently, the lack of data available for measuring species responses to climate related stressors limits the broad applicability and increases the uncertainty of a response based approach. The ability to extrapolate trait based sensitivities to all species, for which traits are known, makes the applicability of a trait association approach much more feasible. In addition, the mechanistic nature of a trait association approach provides logical support for assigning environmental tolerances. However, additional factors not accounted for by traits may affect the environmental tolerances of some species. Based on currently available information for stream fish, we believe that the trait association index provides the most complete and reliable assessment of stream fish vulnerability. However, future research linking traits with environmental tolerances or measuring species responses to stressors is critical to improving the accuracy and decreasing the uncertainty of vulnerability assessments for stream fish.

The ability to assess the representation of stream fish within conservation networks is critical to developing effective conservation strategies. We found that by targeting areas within and complementary to established conservation networks, locations which provided comprehensive and redundant species representation could be identified. The comparison of results from our approach to selecting priority areas versus the use of surrogates, such as habitat factors or threat scores, revealed substantial increases in the predicted representation of species, particularly species which are rare or have restricted distributions, when our approach was used. The use of different prioritization techniques, species weighting, and inclusion of connectivity requirements had variable influences on the results of prioritization, suggesting careful consideration of inputs and settings is strongly advisable. Our comparison of species representation within the each of the conservation networks compared to an optimal network of the same size, generated using a blank slate approach, revealed that the proportion of representation lost for the species which lost the greatest proportion of its original distribution was the same or slightly higher when using core-area Zonation, but lower when using the additive benefit function. The same comparison for the average proportion of distribution lost for all species revealed that for both core-area and additive benefit implementations species lost more representation when restricted to established networks than optimal networks of the same size. This suggests that predicted representations of species could be improved if conservation networks were developed from scratch although the relatively small differences suggest that these gains would be marginal and the benefits of working within established infrastructure likely outweigh the gains of starting from a blank slate. We believe that conservation value rankings of stream segments within conservation networks can be used to help prioritize the selection of sites for management actions. The conservation value rankings of stream segments complementary to established conservation networks can be used to aid in decisions related to land acquisition and the establishment of partnerships. Our framework for assessing stream segment scale conservation value while incorporating conservation networks which have already been established could be applied around the globe to assist in meeting the prioritization needs of managers and decision makers.

Our work also revealed a number of areas which would benefit greatly from additional research. One of the greatest needs is the development of stream temperature and flow models which represent both current and future conditions. Without this information the options for assessing the exposure of stream fish to climate change are extremely limited and riddled with uncertainty. Stream fish vulnerability assessments would also benefit greatly from research which quantifies the relationships between species or traits with stream temperature and flow regimes. Data on these topics for stream fish is currently very limited, making it difficult to draw conclusions as to the impact stressors will have on stream fish communities in the future. In order to more effectively plan for the conservation of stream fish species it would be beneficial to gain a greater understanding of how the landscape affects stream fish. It would be particularly useful to gain a better understanding of species-specific habitat requirements, specifically developing better measures of how upstream habitat degradation impacts a species, the area of suitable habitat which needs to be maintained for the persistence of a population (i.e. How big do protected areas need to be?) and how do factors such as connectivity dispersal ability, and metapopulation dynamics impact the viability of populations (i.e. Can a patchwork of sites protect a population and if so how close together do these sites need to be?).

APPENDICES

Appendix 1: Species vulnerability scores for the response and trait based approaches to classifying environmental tolerance

Species	Score (Response)	Score (Traits)
AMBLOPLITES ARIOMMUS	3.12	4.65
AMBLOPLITES CONSTELLATUS	3.96	5.56
AMBLOPLITES RUPESTRIS	3.2	2.2
AMEIURUS MELAS	1.69	2.69
AMEIURUS NATALIS	0.6	1.6
AMEIURUS NEBULOSUS	2.87	4.86
APHREDODERUS SAYANUS	1.92	1.92
APLODINOTUS GRUNNIENS	0.94	2.48
CAMPOSTOMA ANOMALUM PULLUM	1.71	0.71
CAMPOSTOMA OLIGOLEPIS	1.01	2.01
CARPIODES CARPIO	1.05	2.49
CARPIODES CYPRINUS	1.18	2.83
CARPIODES VELIFER	4.61	4.61
CATOSTOMUS COMMERSONII	3.79	3.79
CENTRARCHUS MACROPTERUS		4.86
COTTUS BAIRDII	3	5
COTTUS CAROLINAE	3.04	4.04
COTTUS HYPSELURUS		6.22
CYPRINELLA CAMURA	2.47	2.47
CYPRINELLA GALACTURA	3.98	2.17
CYPRINELLA LUTRENSIS	0.86	0.86
CYPRINELLA SPILOPTERA	5.82	1.98
CYPRINELLA VENUSTA	1.67	3.42
CYPRINELLA WHIPPLEI	3.51	1.89
DOROSOMA CEPEDIANUM	0.67	2.67
ERIMYSTAX HARRYI		5.63
ERIMYSTAX X-PUNCTATUS		5.63
ERIMYZON OBLONGUS	2.85	5.63
ESOX AMERICANUS VERMICULATUS		4.06
ESOX NIGER	3.94	3.94
ETHEOSTOMA BLENNIOIDES	3.87	2.87
ETHEOSTOMA BURRI		7.87
ETHEOSTOMA CAERULEUM	2.99	3.99
ETHEOSTOMA CRAGINI		7.82
ETHEOSTOMA EUZONUM		8.11
ETHEOSTOMA FLABELLARE	2.97	3.97
ETHEOSTOMA GRACILE		7.15
ETHEOSTOMA JULIAE		8.48

Species	Score (Response)	Score (Traits)
ETHEOSTOMA MICROPERCA		7.71
ETHEOSTOMA NIANGUAE		8.75
ETHEOSTOMA NIGRUM	2.94	2.94
ETHEOSTOMA PROELIARE		7.34
ETHEOSTOMA PUNCTULATUM	3.34	4.34
ETHEOSTOMA SPECTABILE	1.88	3.88
ETHEOSTOMA STIGMAEUM	7.82	5.84
ETHEOSTOMA TETRAZONUM		6.87
ETHEOSTOMA UNIPORUM		7.72
ETHEOSTOMA ZONALE	3.74	3.74
FUNDULUS CATENATUS	2.88	1.88
FUNDULUS DISPAR		6.71
FUNDULUS NOTATUS	1.88	2.88
FUNDULUS OLIVACEUS	1.93	3.93
FUNDULUS SCIADICUS		3.17
GAMBUSIA AFFINIS		0.62
HYBOGNATHUS ARGYRITIS		1.89
HYBOGNATHUS HANKINSONI	3.47	3.47
HYBOGNATHUS PLACITUS		1.91
HYBOPSIS AMBLOPS	2.56	2.56
HYPENTELIUM NIGRICANS	2.02	4.02
ICHTHYOMYZON FOSSOR		6.44
ICHTHYOMYZON GAGEI		4.66
ICTALURUS PUNCTATUS	0.63	1.63
ICTIOBUS BUBALUS	1.15	2.94
ICTIOBUS CYPRINELLUS	1.51	3.32
LAMPETRA AEPYPTERA	2.07	5.91
LEPISOSTEUS OCULATUS		3.66
LEPISOSTEUS OSSEUS	0.82	2.22
LEPISOSTEUS PLATOSTOMUS	0.96	2.46
LEPOMIS CYANELLUS	0.61	1.61
LEPOMIS GULOSUS	1.93	1.93
LEPOMIS HUMILIS	0.7	1.7
LEPOMIS MACROCHIRUS	0.62	1.62
LEPOMIS MEGALOTIS	1.68	2.68
LEPOMIS MICROLOPHUS	0.91	2.21
LEPOMIS MINIATUS		3.29
LUXILUS CARDINALIS	3.45	4.9
LUXILUS CHRYSOCEPHALUS	1.89	1.89
LUXILUS CORNUTUS	2.83	2.83
LUXILUS PILSBRYI	3.54	4.92
LUXILUS ZONATUS		2.05
LYTHRURUS UMBRATILIS	0.66	0.66

Species	Score (Response)	Score (Traits)
MACRHYBOPSIS STORERIANA	1.71	5.69
MICROPTERUS DOLOMIEU	2.99	1.99
MICROPTERUS PUNCTULATUS	0.86	2.16
MICROPTERUS SALMOIDES	0.62	1.62
MINYTREMA MELANOPS	2.56	4.15
MORONE CHRYSOPS	1.77	3.69
MOXOSTOMA ANISURUM	3.83	7.55
MOXOSTOMA CARINATUM	4.19	6.12
MOXOSTOMA DUQUESNEI	2.04	3.04
MOXOSTOMA ERYTHRURUM	0.96	2.96
MOXOSTOMA MACROLEPIDOTUM	0.82	5.26
NOCOMIS ASPER	3.8	6.94
NOCOMIS BIGUTTATUS	2	4
NOTEMIGONUS CRYSOLEUCAS	0.7	1.7
NOTROPIS ATHERINOIDES	2.79	2.79
NOTROPIS BOOPS	2.03	3.03
NOTROPIS BUCCATUS	2.34	4.1
NOTROPIS BUCHANANI		3.9
NOTROPIS DORSALIS	2.05	2.05
NOTROPIS GREENEI		3.2
NOTROPIS HETEROLEPIS		2.33
NOTROPIS NUBILUS	2	3
NOTROPIS OZARCANUS		5.83
NOTROPIS STRAMINEUS	1.99	0.99
NOTROPIS TELESCOPUS	2.5	3.61
NOTROPIS TEXANUS		1.87
NOTROPIS VOLUCELLUS	3.1	1.46
NOTURUS ALBATER		4.68
NOTURUS EXILIS	1.93	3.93
NOTURUS FLAVATER		6.16
NOTURUS FLAVUS	4.61	6.33
NOTURUS GYRINUS	1.57	3.3
NOTURUS MIURUS	2.2	4.11
NOTURUS NOCTURNUS	1.4	7.34
OPSOPOEODUS EMILIAE		1.34
PERCINA CAPRODES	2.79	2.79
PERCINA COPELANDI	4.43	6.4
PERCINA CYMATOTAENIA		7.88
PERCINA EVIDES	5.98	7.92
PERCINA MACULATA	6.17	6.17
PERCINA PHOXOCEPHALA	4.76	6.39
PERCINA SCIERA	3.9	5.89
PHENACOBIUS MIRABILIS	1.84	1.84

Species	Score (Response)	Score (Traits)
PHOXINUS ERYTHROGASTER	1.95	1.95
PIMEPHALES NOTATUS	1.61	0.61
PIMEPHALES PROMELAS	1.64	0.64
PIMEPHALES VIGILAX	1.66	1.66
POMOXIS ANNULARIS	0.62	1.97
POMOXIS NIGROMACULATUS	1.17	3.04
PYLODICTIS OLIVARIS	0.96	2.6
SANDER VITREUS	4.21	8.19
SEMOTILUS ATROMACULATUS	0.65	2.65

Appendix 2: Environmental tolerance classifications for the trait association index (1 = vulnerable, 0 = not vulnerable)

Species	Habitat	Thermal	Flow
AMBLOPLITES ARIOMMUS	0	1	1
AMBLOPLITES CONSTELLATUS	0	1	1
AMBLOPLITES RUPESTRIS	0	0	1
AMEIURUS MELAS	1	0	1
AMEIURUS NATALIS	0	0	1
AMEIURUS NEBULOSUS	0	0	1
APHREDODERUS SAYANUS	0	0	0
APLODINOTUS GRUNNIENS	0	0	1
CAMPOSTOMA ANOMALUM PULLUM	0	0	0
CAMPOSTOMA OLIGOLEPIS	1	0	0
CARPIODES CARPIO	0	0	1
CARPIODES CYPRINUS	0	0	1
CARPIODES VELIFER	0	0	1
CATOSTOMUS COMMERSONII	1	1	1
CENTRARCHUS MACROPTERUS	0	0	1
COTTUS BAIRDII	1	1	1
COTTUS CAROLINAE	1	0	1
COTTUS HYPSELURUS	1	1	1
CYPRINELLA CAMURA	0	0	0
CYPRINELLA GALACTURA	0	0	0
CYPRINELLA LUTRENSIS	0	0	0
CYPRINELLA SPILOPTERA	0	0	0
CYPRINELLA VENUSTA	0	1	0
CYPRINELLA WHIPPLEI	0	0	0
DOROSOMA CEPEDIANUM	0	1	1
ERIMYSTAX HARRYI	1	1	0
ERIMYSTAX X-PUNCTATUS	1	1	0
ERIMYZON OBLONGUS	1	1	1
ESOX AMERICANUS VERMICULATUS	0	0	1
ESOX NIGER	0	0	1
ETHEOSTOMA BLENNIOIDES	1	0	0
ETHEOSTOMA BURRI	1	1	0
ETHEOSTOMA CAERULEUM	1	1	0
ETHEOSTOMA CRAGINI	1	1	0
ETHEOSTOMA EUZONUM	1	1	0
ETHEOSTOMA FLABELLARE	1	1	0
ETHEOSTOMA GRACILE	1	1	0
ETHEOSTOMA JULIAE	1	1	0
ETHEOSTOMA MICROPERCA	1	1	0

Species	Habitat	Thermal	Flow
ETHEOSTOMA NIANGUAE	1	1	0
ETHEOSTOMA NIGRUM	1	0	0
ETHEOSTOMA PROELIARE	1	1	0
ETHEOSTOMA PUNCTULATUM	1	1	0
ETHEOSTOMA SPECTABILE	1	1	0
ETHEOSTOMA STIGMAEUM	1	0	0
ETHEOSTOMA TETRAZONUM	1	1	0
ETHEOSTOMA UNIPORUM	1	1	0
ETHEOSTOMA ZONALE	1	0	0
FUNDULUS CATENATUS	0	0	0
FUNDULUS DISPAR	0	1	0
FUNDULUS NOTATUS	0	1	0
FUNDULUS OLIVACEUS	1	1	0
FUNDULUS SCIADICUS	0	0	0
GAMBUSIA AFFINIS	0	0	0
HYBOGNATHUS ARGYRITIS	0	0	0
HYBOGNATHUS HANKINSONI	0	1	0
HYBOGNATHUS PLACITUS	0	0	0
HYBOPSIS AMBLOPS	1	0	0
HYPENTELIUM NIGRICANS	1	1	1
ICHTHYOMYZON FOSSOR	1	1	0
ICHTHYOMYZON GAGEI	1	0	0
ICTALURUS PUNCTATUS	0	0	1
ICTIOBUS BUBALUS	0	0	1
ICTIOBUS CYPRINELLUS	0	0	1
LAMPETRA AEPYPTERA	1	1	0
LEPISOSTEUS OCULATUS	0	0	1
LEPISOSTEUS OSSEUS	0	0	1
LEPISOSTEUS PLATOSTOMUS	0	0	1
LEPOMIS CYANELLUS	0	0	1
LEPOMIS GULOSUS	0	0	1
LEPOMIS HUMILIS	1	0	0
LEPOMIS MACROCHIRUS	0	0	1
LEPOMIS MEGALOTIS	0	1	1
LEPOMIS MICROLOPHUS	0	0	1
LEPOMIS MINIATUS	0	0	1
LUXILUS CARDINALIS	1	1	0
LUXILUS CHRYSOCEPHALUS	1	0	0
LUXILUS CORNUTUS	1	0	0
LUXILUS PILSBRYI	1	1	0
LUXILUS ZONATUS	1	0	0
LYTHRURUS UMBRATILIS	0	0	0

Species	Habitat	Thermal	Flow
MACRHYBOPSIS STORERIANA	1	1	0
MICROPTERUS DOLOMIEU	0	0	1
MICROPTERUS PUNCTULATUS	0	0	1
MICROPTERUS SALMOIDES	0	0	1
MINYTREMA MELANOPS	1	0	1
MORONE CHRYSOPS	0	0	1
MOXOSTOMA ANISURUM	1	1	1
MOXOSTOMA CARINATUM	1	0	1
MOXOSTOMA DUQUESNEI	1	0	1
MOXOSTOMA ERYTHRURUM	1	0	1
MOXOSTOMA MACROLEPIDOTUM	1	1	1
NOCOMIS ASPER	1	1	1
NOCOMIS BIGUTTATUS	1	1	1
NOTEMIGONUS CRYSOLEUCAS	0	0	1
NOTROPIS ATHERINOIDES	0	1	0
NOTROPIS BOOPS	1	1	0
NOTROPIS BUCCATUS	1	0	0
NOTROPIS BUCHANANI	1	0	0
NOTROPIS DORSALIS	0	1	0
NOTROPIS GREENEI	1	0	0
NOTROPIS HETEROLEPIS	0	0	0
NOTROPIS NUBILUS	1	1	0
NOTROPIS OZARCANUS	1	1	0
NOTROPIS STRAMINEUS	0	0	0
NOTROPIS TELESCOPUS	1	1	0
NOTROPIS TEXANUS	0	0	0
NOTROPIS VOLUCELLUS	0	0	0
NOTURUS ALBATER	1	0	1
NOTURUS EXILIS	1	1	1
NOTURUS FLAVATER	1	0	1
NOTURUS FLAVUS	1	1	1
NOTURUS GYRINUS	1	0	0
NOTURUS MIURUS	1	0	0
NOTURUS NOCTURNUS	1	1	1
OPSOPOEODUS EMILIAE	0	0	0
PERCINA CAPRODES	1	0	0
PERCINA COPELANDI	1	0	0
PERCINA CYMATOTAENIA	1	1	0
PERCINA EVIDES	1	1	0
PERCINA MACULATA	1	1	0
PERCINA PHOXOCEPHALA	1	1	0
PERCINA SCIERA	1	0	0

Species	Habitat	Thermal	Flow
PHENACOBIUS MIRABILIS	1	0	0
PHOXINUS ERYTHROGASTER	0	1	0
PIMEPHALES NOTATUS	0	0	0
PIMEPHALES PROMELAS	0	0	0
PIMEPHALES VIGILAX	0	0	0
POMOXIS ANNULARIS	0	0	1
POMOXIS NIGROMACULATUS	0	0	1
PYLODICTIS OLIVARIS	0	0	1
SANDER VITREUS	0	1	1
SEMOTILUS ATROMACULATUS	1	1	0

Appendix 3: Environmental tolerance classifications for species response index (1 = vulnerable, 0 = not vulnerable)

Species	Habitat	Thermal	Flow
AMBLOPLITES ARIOMMUS	1	0	0
AMBLOPLITES CONSTELLATUS	1	0	0
AMBLOPLITES RUPESTRIS	1	0	1
AMEIURUS MELAS	0	0	1
AMEIURUS NATALIS	0	0	0
AMEIURUS NEBULOSUS	0	0	0
APHREDODERUS SAYANUS	0	0	0
APLODINOTUS GRUNNIENS	0	0	0
CAMPOSTOMA ANOMALUM PULLUM	0	1	0
CAMPOSTOMA OLIGOLEPIS	0	0	0
CARPIODES CARPIO	0	0	0
CARPIODES CYPRINUS	0	0	0
CARPIODES VELIFER	1	0	0
CATOSTOMUS COMMERSONII	1	1	1
CENTRARCHUS MACROPTERUS			
COTTUS BAIRDII	0	0	1
COTTUS CAROLINAE	1	0	0
COTTUS HYPSELURUS			
CYPRINELLA CAMURA	0	0	0
CYPRINELLA GALACTURA	1	0	0
CYPRINELLA LUTRENSIS	0	0	0
CYPRINELLA SPILOPTERA	1	0	1
CYPRINELLA VENUSTA	0	0	0
CYPRINELLA WHIPPLEI	1	0	0
DOROSOMA CEPEDIANUM	0	0	0
ERIMYSTAX HARRYI			
ERIMYSTAX X-PUNCTATUS			
ERIMYZON OBLONGUS	1	0	0
ESOX AMERICANUS VERMICULATUS			
ESOX NIGER	1	0	0
ETHEOSTOMA BLENNIOIDES	1	0	1
ETHEOSTOMA BURRI			
ETHEOSTOMA CAERULEUM	1	0	0
ETHEOSTOMA CRAGINI			
ETHEOSTOMA EUZONUM			
ETHEOSTOMA FLABELLARE	1	0	0
ETHEOSTOMA GRACILE			
ETHEOSTOMA JULIAE			
ETHEOSTOMA MICROPERCA			

Species	Habitat	Thermal	Flow
ETHEOSTOMA NIANGUAE			
ETHEOSTOMA NIGRUM	0	0	1
ETHEOSTOMA PROELIARE			
ETHEOSTOMA PUNCTULATUM	1	0	0
ETHEOSTOMA SPECTABILE	0	0	0
ETHEOSTOMA STIGMAEUM	1	0	1
ETHEOSTOMA TETRAZONUM			
ETHEOSTOMA UNIPORUM			
ETHEOSTOMA ZONALE	1	0	0
FUNDULUS CATENATUS	1	0	0
FUNDULUS DISPAR			
FUNDULUS NOTATUS	0	0	0
FUNDULUS OLIVACEUS	0	0	0
FUNDULUS SCIADICUS			
GAMBUSIA AFFINIS			
HYBOGNATHUS ARGYRITIS			
HYBOGNATHUS HANKINSONI	0	0	1
HYBOGNATHUS PLACITUS			
HYBOPSIS AMBLOPS	1	0	0
HYPENTELIUM NIGRICANS	1	0	0
ICHTHYOMYZON FOSSOR			
ICHTHYOMYZON GAGEI			
ICTALURUS PUNCTATUS	0	0	0
ICTIOBUS BUBALUS	0	0	0
ICTIOBUS CYPRINELLUS	0	0	0
LAMPETRA AEPYPTERA	0	0	0
LEPISOSTEUS OCULATUS			
LEPISOSTEUS OSSEUS	0	0	0
LEPISOSTEUS PLATOSTOMUS	0	0	0
LEPOMIS CYANELLUS	0	0	0
LEPOMIS GULOSUS	1	0	0
LEPOMIS HUMILIS	0	0	0
LEPOMIS MACROCHIRUS	0	0	0
LEPOMIS MEGALOTIS	1	0	0
LEPOMIS MICROLOPHUS	0	0	0
LEPOMIS MINIATUS			
LUXILUS CARDINALIS	1	0	0
LUXILUS CHRYSOCEPHALUS	1	0	0
LUXILUS CORNUTUS	0	0	1
LUXILUS PILSBRYI	1	0	0
LUXILUS ZONATUS			
LYTHRURUS UMBRATILIS	0	0	0

Species	Habitat	Thermal	Flow
MACRHYBOPSIS STORERIANA	0	0	0
MICROPTERUS DOLOMIEU	1	1	0
MICROPTERUS PUNCTULATUS	0	0	0
MICROPTERUS SALMOIDES	0	0	0
MINYTREMA MELANOPS	1	0	0
MORONE CHRYSOPS	0	0	0
MOXOSTOMA ANISURUM	0	0	1
MOXOSTOMA CARINATUM	1	0	0
MOXOSTOMA DUQUESNEI	1	0	0
MOXOSTOMA ERYTHRURUM	0	0	0
MOXOSTOMA MACROLEPIDOTUM	0	0	0
NOCOMIS ASPER	1	0	0
NOCOMIS BIGUTTATUS	1	0	0
NOTEMIGONUS CRYSOLEUCAS	0	0	0
NOTROPIS ATHERINOIDES	0	0	1
NOTROPIS BOOPS	1	0	0
NOTROPIS BUCCATUS	0	0	0
NOTROPIS BUCHANANI			
NOTROPIS DORSALIS	0	0	1
NOTROPIS GREENEI			
NOTROPIS HETEROLEPIS			
NOTROPIS NUBILUS	1	0	0
NOTROPIS OZARCANUS			
NOTROPIS STRAMINEUS	1	0	0
NOTROPIS TELESCOPUS	1	0	0
NOTROPIS TEXANUS			
NOTROPIS VOLUCELLUS	0	0	1
NOTURUS ALBATER			
NOTURUS EXILIS	1	0	0
NOTURUS FLAVATER			
NOTURUS FLAVUS	1	0	1
NOTURUS GYRINUS	0	0	0
NOTURUS MIURUS	0	0	0
NOTURUS NOCTURNUS	0	0	0
OPSOPOEODUS EMILIAE			
PERCINA CAPRODES	1	0	0
PERCINA COPELANDI	0	0	0
PERCINA CYMATOTAENIA			
PERCINA EVIDES	1	0	0
PERCINA MACULATA	1	0	1
PERCINA PHOXOCEPHALA	1	0	0
PERCINA SCIERA	0	0	0

Species	Habitat	Thermal	Flow
PHENACOBIUS MIRABILIS	1	0	0
PHOXINUS ERYTHROGASTER	1	0	0
PIMEPHALES NOTATUS	0	0	1
PIMEPHALES PROMELAS	0	0	1
PIMEPHALES VIGILAX	0	0	0
POMOXIS ANNULARIS	0	0	0
POMOXIS NIGROMACULATUS	0	0	0
PYLODICTIS OLIVARIS	0	0	0
SANDER VITREUS	0	0	0
SEMOTILUS ATROMACULATUS	0	0	0

Appendix 4: Species scores for rarity, dispersal, range and freshwater resilience

Species	Rarity	Dispersal	Range	Resilience
AMBLOPLITES ARIOMMUS	1.53	0	0.765	0.42
AMBLOPLITES CONSTELLATUS	1.6	0	0.942	0.85
AMBLOPLITES RUPESTRIS	1	0	0.589	0.61
AMEIURUS MELAS	1	0	0.177	0.51
AMEIURUS NATALIS	1	0	0	0.6
AMEIURUS NEBULOSUS	1.99	0	0.942	1
APHREDODERUS SAYANUS	1.82	0	0.765	0.53
APLODINOTUS GRUNNIENS	1.54	0	0.236	0.58
CAMPOSTOMA ANOMALUM PULLUM	1	0	0.118	0.59
CAMPOSTOMA OLIGOLEPIS	1	0	0.412	0.6
CARPIODES CARPIO	1.44	0	0.471	0.37
CARPIODES CYPRINUS	1.65	0	0.412	0.5
CARPIODES VELIFER	1.98	0	0.883	0.88
CATOSTOMUS COMMERSONII	1	0	0.177	0.61
CENTRARCHUS MACROPTERUS	1.99	0	0.942	1
COTTUS BAIRDII	1	1	0.471	0.53
COTTUS CAROLINAE	1	1	0.412	0.63
COTTUS HYPSELURUS	1.22	1	0.648	0.55
CYPRINELLA CAMURA	1.99	0	0.942	0.6
CYPRINELLA GALACTURA	1.81	0	0.824	0.68
CYPRINELLA LUTRENSIS	1	0	0.295	0.56
CYPRINELLA SPILOPTERA	1.92	0	0.706	0.62
CYPRINELLA VENUSTA	1.75	0	0.648	0.54
CYPRINELLA WHIPPLEI	1.62	0	0.765	0.65
DOROSOMA CEPEDIANUM	1	0	0.059	0.61
ERIMYSTAX HARRYI	1.87	0	0.824	0.35
ERIMYSTAX X-PUNCTATUS	1.9	0	0.589	0.71
ERIMYZON OBLONGUS	1.39	0	0.648	0.56
ESOX AMERICANUS VERMICULATUS	1.85	0	0.765	0.79
ESOX NIGER	1.86	0	0.765	0.66
ETHEOSTOMA BLENNIOIDES	1	1	0.236	0.63
ETHEOSTOMA BURRI	1.76	1	0.942	0.93
ETHEOSTOMA CAERULEUM	1	1	0.353	0.64
ETHEOSTOMA CRAGINI	1.82	1	0.942	0.65
ETHEOSTOMA EUZONUM	1.99	1	0.942	0.27
ETHEOSTOMA FLABELLARE	1	1	0.295	0.67
ETHEOSTOMA GRACILE	1.94	1	0.589	0.19
ETHEOSTOMA JULIAE	1.93	1	0.942	0.87
ETHEOSTOMA MICROPERCA	1.97	1	0.824	0.18
ETHEOSTOMA NIANGUAE	1.99	1	0.942	0.91

Species	Rarity	Dispersal	Range	Resilience
ETHEOSTOMA NIGRUM	1	1	0.353	0.59
ETHEOSTOMA PROELIARE	1.95	1	0.765	0
ETHEOSTOMA PUNCTULATUM	1	1	0.648	0.69
ETHEOSTOMA SPECTABILE	1	1	0.236	0.64
ETHEOSTOMA STIGMAEUM	1.98	1	0.706	0.48
ETHEOSTOMA TETRAZONUM	1.68	1	0.706	0.64
ETHEOSTOMA UNIPORUM	1.95	1	0.942	0.03
ETHEOSTOMA ZONALE	1.26	1	0.471	0.63
FUNDULUS CATENATUS	1	1	0.236	0.64
FUNDULUS DISPAR	1.98	1	0.883	1
FUNDULUS NOTATUS	1	1	0.177	0.7
FUNDULUS OLIVACEUS	1	1	0.236	0.69
FUNDULUS SCIADICUS	1.72	1	0.648	0.34
GAMBUSIA AFFINIS	1	0	0	0.62
HYBOGNATHUS ARGYRITIS	1.97	0	0.883	0.15
HYBOGNATHUS HANKINSONI	1.78	0	0.883	0.12
HYBOGNATHUS PLACITUS	1.98	0	0.824	0.28
HYBOPSIS AMBLOPS	1.09	0	0.648	0.76
HYPENTELIUM NIGRICANS	1	0	0.353	0.67
ICHTHYOMYZON FOSSOR	1.99	0	0.883	0.7
ICHTHYOMYZON GAGEI	1.97	0	0.942	0.83
ICTALURUS PUNCTATUS	1	0	0.118	0.51
ICTIOBUS BUBALUS	1.79	0	0.353	0.52
ICTIOBUS CYPRINELLUS	1.81	0	0.412	0.76
LAMPETRA AEPYPTERA	1.92	0	0.883	0.37
LEPISOSTEUS OCULATUS	1.79	0	0.765	0.5
LEPISOSTEUS OSSEUS	1.4	0	0.177	0.57
LEPISOSTEUS PLATOSTOMUS	1.5	0	0.295	0.52
LEPOMIS CYANELLUS	1	0	0	0.61
LEPOMIS GULOSUS	1.19	0	0.177	0.53
LEPOMIS HUMILIS	1	0	0.118	0.58
LEPOMIS MACROCHIRUS	1	0	0	0.62
LEPOMIS MEGALOTIS	1	0	0	0.68
LEPOMIS MICROLOPHUS	1.3	0	0.177	0.68
LEPOMIS MINIATUS	1.63	0	0.706	0.51
LUXILUS CARDINALIS	1.45	0	0.883	0.72
LUXILUS CHRYSOCEPHALUS	1	0	0.295	0.59
LUXILUS CORNUTUS	1.27	0	0.706	0.66
LUXILUS PILSBRYI	1.38	0	0.942	0.86
LUXILUS ZONATUS	1	0	0.471	0.58
LYTHRURUS UMBRATILIS	1	0	0.177	0.48
MACRHYBOPSIS STORERIANA	1.99	0	0.706	0.31

Species	Rarity	Dispersal	Range	Resilience
MICROPTERUS DOLOMIEU	1	0	0.353	0.64
MICROPTERUS PUNCTULATUS	1.3	0	0.118	0.71
MICROPTERUS SALMOIDES	1	0	0	0.62
MINYTREMA MELANOPS	1.59	0	0.295	0.5
MORONE CHRYSOPS	1.92	0	0.53	0.75
MOXOSTOMA ANISURUM	1.86	0	0.706	0.66
MOXOSTOMA CARINATUM	1.93	0	0.824	0.67
MOXOSTOMA DUQUESNEI	1	0	0.353	0.69
MOXOSTOMA ERYTHRURUM	1	0	0.236	0.72
MOXOSTOMA MACROLEPIDOTUM	1.48	0	0.118	0.65
NOCOMIS ASPER	1.57	0	0.942	0.75
NOCOMIS BIGUTTATUS	1	0	0.412	0.59
NOTEMIGONUS CRYSOLEUCAS	1	0	0.059	0.64
NOTROPIS ATHERINOIDES	1.65	0	0.295	0.65
NOTROPIS BOOPS	1	0	0.412	0.62
NOTROPIS BUCCATUS	1.76	0	0.824	0.89
NOTROPIS BUCHANANI	1.96	0	0.706	0.56
NOTROPIS DORSALIS	1	0	0.589	0.46
NOTROPIS GREENEI	1.59	0	0.648	0.58
NOTROPIS HETEROLEPIS	1.95	0	0.765	0.84
NOTROPIS NUBILUS	1	0	0.353	0.65
NOTROPIS OZARCANUS	1.93	0	0.824	0.38
NOTROPIS STRAMINEUS	1	0	0.412	0.58
NOTROPIS TELESCOPUS	1.11	0	0.765	0.54
NOTROPIS TEXANUS	1.86	0	0.824	0.34
NOTROPIS VOLUCELLUS	1.64	0	0.471	0.69
NOTURUS ALBATER	1.43	0	0.824	0.64
NOTURUS EXILIS	1	0	0.236	0.69
NOTURUS FLAVATER	1.98	0	0.883	0.45
NOTURUS FLAVUS	1.72	0	0.353	0.56
NOTURUS GYRINUS	1.73	0	0.589	0.55
NOTURUS MIURUS	1.91	0	0.824	0.63
NOTURUS NOCTURNUS	1.98	0	0.589	0.23
OPSOPOEODUS EMILIAE	1.89	0	0.706	0.01
PERCINA CAPRODES	1	1	0.118	0.67
PERCINA COPELANDI	1.97	1	0.942	0.6
PERCINA CYMATOTAENIA	1.98	1	0.883	0.19
PERCINA EVIDES	1.94	1	0.706	0.73
PERCINA MACULATA	1.59	1	0.53	0.56
PERCINA PHOXOCEPHALA	1.63	1	0.53	0.64
PERCINA SCIERA	1.99	1	0.765	0.39
PHENACOBIUS MIRABILIS	1	0	0.353	0.49

Species	Rarity	Dispersal	Range	Resilience
PHOXINUS ERYTHROGASTER	1	0	0.353	0.6
PIMEPHALES NOTATUS	1	0	0	0.61
PIMEPHALES PROMELAS	1	0	0.177	0.46
PIMEPHALES VIGILAX	1.72	0	0.589	0.65
POMOXIS ANNULARIS	1.35	0	0.059	0.54
POMOXIS NIGROMACULATUS	1.87	0	0.236	0.73
PYLODICTIS OLIVARIS	1.64	0	0.295	0.48
SANDER VITREUS	1.99	1	0.765	0.7
SEMOTILUS ATROMACULATUS	1	0	0.059	0.59

Appendix 5: Environmental variables and associated units which were used for the creation of species distribution models

Variable	Units
Ecological Drainage Unit	Categorical
Aquatic Ecological System Type	Categorical
Strahler Stream Order	Continuous Categorical
Downstream Size	Continuous Categorical
Reach Gradient	m/km
Segment Gradient	m/km
Local Geological Classification	Categorical
Urban Landcover- Local	% of Local Watershed Area
Agricultural Landcover- Local	% of Local Watershed Area
Grassland Landcover- Local	% of Local Watershed Area
Forest Landcover- Local	% of Local Watershed Area
Wetland Landcover- Local	% of Local Watershed Area
Urban Landcover- Upstream Watershed	% of Upstream Watershed Area
Agricultural Landcover- Upstream Watershed	% of Upstream Watershed Area
Grassland Landcover- Upstream Watershed	% of Upstream Watershed Area
Forest Landcover- Upstream Watershed	% of Upstream Watershed Area
Wetland Landcover- Upstream Watershed	% of Upstream Watershed Area
Number of Springs- Local	Count
Number of Springs- Watershed	Count
Number of Dams- Local	Count
Number of Dams- Upstream Watershed	Count
Number of Headwater Impoundments- Local	Count
Number of Headwater Impoundments- Upstream Watershed	Count
Number of Road/Stream Crossings per KM^2- Local	Ratio
Number of Road/Stream Crossings per KM^2- Upstream Watershed	Ratio
Local Habitat Threat Index Score	Value
Upstream Habitat Threat Index Score	Value
Percent of the Local Riparian Intact	% of Local Riparian Not Classified as Agriculture or Urban

Appendix 6: A list of species codes, scientific names, and common names for all study species, as well as whether a model was constructed for each species for both the Plains and Ozarks subregions. Blank indicates no model constructed, "Yes" indicates a satisfactory model was created, "Yes (Failed)" indicates a model was created but did not meet the model evaluation criteria.

Species Code	Scientific Name	Plains Model	Ozark Model
A_ARIOMMUS	AMBLOPLITES ARIOMMUS		Yes
A_CONSTELLATUS	AMBLOPLITES CONSTELLATUS		Yes
A_GRUNNIENS	APLODINOTUS GRUNNIENS	Yes	
A_MELAS	AMEIURUS MELAS	Yes (Failed)	Yes
A_NATALIS	AMEIURUS NATALIS	Yes	Yes
A_NEBULOSUS	AMEIURUS NEBULOSUS		
A_RUPESTRIS	AMBLOPLITES RUPESTRIS		Yes
A_SAYANUS	APHREDODERUS SAYANUS		
C_BAIRDII	COTTUS BAIRDII		Yes
C_CAMURA	CYPRINELLA CAMURA		
C_CAROLINAE	COTTUS CAROLINAE		Yes
C_CARPIO	CARPIODES CARPIO	Yes	
C_COMMERSONII	CATOSTOMUS COMMERSONII	Yes	Yes
C_CYPRINUS	CARPIODES CYPRINUS	Yes	
C_GALACTURA	CYPRINELLA GALACTURA		Yes
C_HYPSELURUS	COTTUS HYPSELURUS		Yes
C_LUTRENSIS	CYPRINELLA LUTRENSIS	Yes	Yes
C_MACROPTERUS	CENTRARCHUS MACROPTERUS		
C_OLIGOLEPIS	CAMPOSTOMA OLIGOLEPIS		Yes
C_PULLUM	CAMPOSTOMA ANOMALUM PULLUM	Yes	Yes
C_SPILOPTERA	CYPRINELLA SPILOPTERA		
C_VELIFER	CARPIODES VELIFER		
C_VENUSTA	CYPRINELLA VENUSTA		
C_WHIPPLEI	CYPRINELLA WHIPPLEI		Yes
D_CEPEDIANUM	DOROSOMA CEPEDIANUM	Yes	Yes
E_BLENNIOIDES	ETHEOSTOMA BLENNIOIDES		Yes
E_BURRI	ETHEOSTOMA BURRI		
E_CAERULEUM	ETHEOSTOMA CAERULEUM		Yes
E_CRAGINI	ETHEOSTOMA CRAGINI		
E_EUZONUM	ETHEOSTOMA EUZONUM		
E_FLABELLARE	ETHEOSTOMA FLABELLARE	Yes	Yes
E_GRACILE	ETHEOSTOMA GRACILE		
E_HARRYI	ERIMYSTAX HARRYI		
E_JULIAE	ETHEOSTOMA JULIAE		
E_MICROPERCA	ETHEOSTOMA MICROPERCA		

Species Code	Scientific Name	Plains Model	Ozark Model
E_NIANGUAE	ETHEOSTOMA NIANGUAE		
E_NIGER	ESOX NIGER		
E_NIGRUM	ETHEOSTOMA NIGRUM	Yes	Yes
E_OBLONGUS	ERIMYZON OBLONGUS		Yes
E_PROELIARE	ETHEOSTOMA PROELIARE		
E_PUNCTULATUM	ETHEOSTOMA PUNCTULATUM		Yes
E_SPECTABILE	ETHEOSTOMA SPECTABILE	Yes	Yes
E_STIGMAEUM	ETHEOSTOMA STIGMAEUM		
E_TETRAZONUM	ETHEOSTOMA TETRAZONUM		Yes
E_UNIPORUM	ETHEOSTOMA UNIPORUM		
E_VERMICULATUS	ESOX AMERICANUS VERMICULATUS		
E_X-PUNCTATUS	ERIMYSTAX X-PUNCTATUS		
E_ZONALE	ETHEOSTOMA ZONALE		Yes
F_CATENATUS	FUNDULUS CATENATUS		Yes
F_DISPAR	FUNDULUS DISPAR		
F_NOTATUS	FUNDULUS NOTATUS	Yes	Yes
F_OLIVACEUS	FUNDULUS OLIVACEUS		Yes
F_SCIADICUS	FUNDULUS SCIADICUS		Yes
G_AFFINIS	GAMBUSIA AFFINIS	Yes	Yes
H_AMBLOPS	HYBOPSIS AMBLOPS		Yes
H_ARGYRITIS	HYBOGNATHUS ARGYRITIS		
H_HANKINSONI	HYBOGNATHUS HANKINSONI		
H_NIGRICANS	HYPENTELIUM NIGRICANS		Yes
H_PLACITUS	HYBOGNATHUS PLACITUS		
I_BUBALUS	ICTIOBUS BUBALUS		
I_CYPRINELLUS	ICTIOBUS CYPRINELLUS		
I_FOSSOR	ICHTHYOMYZON FOSSOR		
I_GAGEI	ICHTHYOMYZON GAGEI		
I_NIGER	ICTIOBUS NIGER		
I_PUNCTATUS	ICTALURUS PUNCTATUS	Yes	Yes
L_AEPYPTERA	LAMPETRA AEPYPTERA		
L_CARDINALIS	LUXILUS CARDINALIS		Yes
L_CHRYSOCEPHALUS	LUXILUS CHRYSOCEPHALUS	Yes	Yes
L_CORNUTUS	LUXILUS CORNUTUS	Yes	Yes
L_CYANELLUS	LEPOMIS CYANELLUS	Yes (Failed)	Yes
L_GULOSUS	LEPOMIS GULOSUS		Yes
L_HUMILIS	LEPOMIS HUMILIS	Yes	Yes
L_MACROCHIRUS	LEPOMIS MACROCHIRUS	Yes	Yes
L_MEGALOTIS	LEPOMIS MEGALOTIS	Yes	Yes
L_MICROLOPHUS	LEPOMIS MICROLOPHUS		Yes

Species Code	Scientific Name	Plains Model	Ozark Model
L_MINIATUS	LEPOMIS MINIATUS		
L_OCULATUS	LEPISOSTEUS OCULATUS		
L_OSSEUS	LEPISOSTEUS OSSEUS		Yes
L_PILSBRYI	LUXILUS PILSBRYI		Yes
L_PLATOSTOMUS	LEPISOSTEUS PLATOSTOMUS	Yes	
L_UMBRATILIS	LYTHRURUS UMBRATILIS	Yes	Yes
L_ZONATUS	LUXILUS ZONATUS		Yes
M_ANISURUM	MOXOSTOMA ANISURUM		
M_CARINATUM	MOXOSTOMA CARINATUM		
M_CHRYSOPS	MORONE CHRYSOPS		
M_DOLOMIEU	MICROPTERUS DOLOMIEU		Yes
M_DUQUESNEI	MOXOSTOMA DUQUESNEI		Yes
M_ERYTHRURUM	MOXOSTOMA ERYTHRURUM	Yes	Yes
M_MACROLEPIDOTUM	MOXOSTOMA MACROLEPIDOTUM		Yes
M_MELANOPS	MINYTREMA MELANOPS		Yes
M_PUNCTULATUS	MICROPTERUS PUNCTULATUS		Yes
M_SALMOIDES	MICROPTERUS SALMOIDES	Yes	Yes
M_STORERIANA	MACRHYBOPSIS STORERIANA		
N_ALBATER	NOTURUS ALBATER		Yes
N_ASPER	NOCOMIS ASPER		Yes
N_ATHERINOIDES	NOTROPIS ATHERINOIDES		
N_BIGUTTATUS	NOCOMIS BIGUTTATUS		Yes
N_BOOPS	NOTROPIS BOOPS	Yes	Yes
N_BUCCATUS	NOTROPIS BUCCATUS		
N_BUCHANANI	NOTROPIS BUCHANANI		
N_CRYSOLEUCAS	NOTEMIGONUS CRYSOLEUCAS	Yes	Yes
N_DORSALIS	NOTROPIS DORSALIS	Yes	
N_EXILIS	NOTURUS EXILIS	Yes	Yes
N_FLAVATER	NOTURUS FLAVATER		
N_FLAVUS	NOTURUS FLAVUS	Yes	
N_GREENEI	NOTROPIS GREENEI		Yes
N_GYRINUS	NOTURUS GYRINUS		
N_HETEROLEPIS	NOTROPIS HETEROLEPIS		
N_MIURUS	NOTURUS MIURUS		
N_NOCTURNUS	NOTURUS NOCTURNUS		
N_NUBILUS	NOTROPIS NUBILUS		Yes
N_OZARCANUS	NOTROPIS OZARCANUS		
N_STRAMINEUS	NOTROPIS STRAMINEUS	Yes	Yes
N_TELESCOPUS	NOTROPIS TELESCOPUS		Yes
N_TEXANUS	NOTROPIS TEXANUS		
N_VOLUCELLUS	NOTROPIS VOLUCELLUS		

Species Code	Scientific Name	Plains Model	Ozark Model
O_EMILIAE	OPSOPOEODUS EMILIAE		
P_ANNULARIS	POMOXIS ANNULARIS	Yes	
P_CAPRODES	PERCINA CAPRODES	Yes	Yes
P_COPELANDI	PERCINA COPELANDI		
P_CYMATOTAENIA	PERCINA CYMATOTAENIA		
P_ERYTHROGASTER	PHOXINUS ERYTHROGASTER		Yes
P_EVIDES	PERCINA EVIDES		
P_FLAVESCENS	PERCA FLAVESCENS		
P_MACULATA	PERCINA MACULATA	Yes	
P_MIRABILIS	PHENACOBIUS MIRABILIS	Yes	Yes
P_NIGROMACULATUS	POMOXIS NIGROMACULATUS		
P_NOTATUS	PIMEPHALES NOTATUS	Yes	Yes
P_OLIVARIS	PYLODICTIS OLIVARIS	Yes	
P_PHOXOCEPHALA	PERCINA PHOXOCEPHALA	Yes	
P_PROMELAS	PIMEPHALES PROMELAS	Yes	Yes
P_SCIERA	PERCINA SCIERA		
P_VIGILAX	PIMEPHALES VIGILAX		
S_ATROMACULATUS	SEMOTILUS ATROMACULATUS	Yes	Yes
S_VITREUS	SANDER VITREUS		

Appendix 7: A list of component models which went into the ensemble for each species. Component models assigned a score of 0 did not meet minimum requirements for at least one evaluation statistic and therefore were not included in the ensemble, while component models assigned a score of 1 met minimum requirements for all three model evaluation statistics and were included in the final ensemble.

Plains Subregion

Species	MARS	GAM	BRT	RndmFrst	Sum
AMEIURUS MELAS	0	0	0	0	0
AMEIURUS NATALIS	0	0	1	1	2
APLODINOTUS GRUNNIENS	0	0	0	1	1
CAMPOSTOMA ANOMALUM PULLUM	0	1	1	1	3
CARPIODES CARPIO	1	1	1	1	4
CARPIODES CYPRINUS	1	1	0	1	3
CATOSTOMUS COMMERSONII	0	1	0	1	2
CYPRINELLA LUTRENSIS	1	1	1	1	4
DOROSOMA CEPEDIANUM	0	1	0	1	2
ETHEOSTOMA FLABELLARE	1	1	1	1	4
ETHEOSTOMA NIGRUM	1	1	1	1	4
ETHEOSTOMA SPECTABILE	1	1	1	1	4
FUNDULUS NOTATUS	1	1	1	1	4
GAMBUSIA AFFINIS	1	1	1	1	4
ICTALURUS PUNCTATUS	0	0	1	1	2
LEPISOSTEUS PLATOSTOMUS	0	0	1	0	1
LEPOMIS CYANELLUS	0	0	0	0	0
LEPOMIS HUMILIS	0	1	1	1	3
LEPOMIS MACROCHIRUS	0	0	1	0	1
LEPOMIS MEGALOTIS	1	0	0	1	2
LUXILUS CHRYSOCEPHALUS	1	1	1	1	4
LUXILUS CORNUTUS	1	1	1	1	4
LYTHRURUS UMBRATILIS	1	1	0	1	3
MICROPTERUS SALMOIDES	0	0	1	1	2
MOXOSTOMA ERYTHRURUM	1	1	1	1	4
NOTEMIGONUS CRYSOLEUCAS	1	1	0	0	2
NOTROPIS BOOPS	1	1	1	0	3
NOTROPIS DORSALIS	1	1	1	1	4
NOTROPIS STRAMINEUS	1	1	1	1	4
NOTURUS EXILIS	1	1	1	1	4
NOTURUS FLAVUS	1	1	1	1	4
PERCINA CAPRODES	0	0	1	1	2
PERCINA MACULATA	1	0	0	0	1
PERCINA PHOXOCEPHALA	1	1	1	1	4

Species	MARS	GAM	BRT	RndmFrst	Sum
PHENACOBIUS MIRABILIS	1	0	1	0	2
PIMEPHALES NOTATUS	1	1	1	1	4
PIMEPHALES PROMELAS	1	1	1	1	4
POMOXIS ANNULARIS	0	1	1	1	3
PYLODICTIS OLIVARIS	1	0	1	0	2
SEMOTILUS ATROMACULATUS	1	1	1	1	4

Ozarks Subregion

Species	MARS	BRT	GAM	RndmFrst	Sum
AMBLOPLITES ARIOMMUS	0	1	0	1	2
AMBLOPLITES CONSTELLATUS	0	1	0	1	2
AMBLOPLITES RUPESTRIS	1	1	1	1	4
AMEIURUS MELAS	0	1	1	1	3
AMEIURUS NATALIS	0	0	0	1	1
CAMPOSTOMA ANOMALUM PULLUM	1	1	1	1	4
CAMPOSTOMA OLIGOLEPIS	1	1	1	1	4
CATOSTOMUS COMMERSONII	1	1	0	1	3
COTTUS BAIRDII	1	1	1	1	4
COTTUS CAROLINAE	1	1	1	1	4
COTTUS HYPSELURUS	1	1	1	1	4
CYPRINELLA GALACTURA	0	0	1	1	2
CYPRINELLA LUTRENSIS	1	1	1	1	4
CYPRINELLA WHIPPLEI	1	0	1	1	3
DOROSOMA CEPEDIANUM	0	1	0	0	1
ERIMYZON OBLONGUS	1	1	1	1	4
ETHEOSTOMA BLENNIOIDES	1	1	1	1	4
ETHEOSTOMA CAERULEUM	1	1	1	1	4
ETHEOSTOMA FLABELLARE	1	1	0	1	3
ETHEOSTOMA NIGRUM	1	1	1	1	4
ETHEOSTOMA PUNCTULATUM	1	1	1	1	4
ETHEOSTOMA SPECTABILE	1	1	1	1	4
ETHEOSTOMA TETRAZONUM	1	0	0	1	2
ETHEOSTOMA ZONALE	1	1	1	1	4
FUNDULUS CATENATUS	1	1	1	1	4
FUNDULUS NOTATUS	1	1	1	1	4
FUNDULUS OLIVACEUS	1	1	1	1	4
FUNDULUS SCIADICUS	1	0	1	1	3
GAMBUSIA AFFINIS	1	1	1	1	4
HYBOPSIS AMBLOPS	1	1	1	1	4
HYPENTELIUM NIGRICANS	0	1	1	1	3
ICTALURUS PUNCTATUS	0	1	0	1	2

Species	MARS	BRT	GAM	RndmFrst	Sum
LEPISOSTEUS OSSEUS	1	0	0	1	2
LEPOMIS CYANELLUS	0	1	1	1	3
LEPOMIS GULOSUS	1	1	1	0	3
LEPOMIS HUMILIS	1	1	1	1	4
LEPOMIS MACROCHIRUS	0	1	1	1	3
LEPOMIS MEGALOTIS	1	1	0	1	3
LEPOMIS MICROLOPHUS	0	0	1	1	2
LUXILUS CARDINALIS	1	1	1	1	4
LUXILUS CHRYSOCEPHALUS	1	1	1	1	4
LUXILUS CORNUTUS	1	1	1	1	4
LUXILUS PILSBRYI	1	1	1	1	4
LUXILUS ZONATUS	1	1	1	1	4
LYTHRURUS UMBRATILIS	0	1	1	1	3
MICROPTERUS DOLOMIEU	1	1	0	1	3
MICROPTERUS PUNCTULATUS	1	1	1	1	4
MICROPTERUS SALMOIDES	0	1	1	0	2
MINYTREMA MELANOPS	1	1	0	1	3
MOXOSTOMA DUQUESNEI	1	1	1	1	4
MOXOSTOMA ERYTHRURUM	1	1	1	1	4
MOXOSTOMA MACROLEPIDOTUM	1	0	1	1	3
NOCOMIS ASPER	0	1	1	1	3
NOCOMIS BIGUTTATUS	1	1	1	1	4
NOTEMIGONUS CRYSOLEUCAS	1	1	0	1	3
NOTROPIS BOOPS	1	1	1	1	4
NOTROPIS GREENEI	1	1	1	1	4
NOTROPIS NUBILUS	0	1	1	1	3
NOTROPIS STRAMINEUS	1	1	1	1	4
NOTROPIS TELESCOPUS	1	1	1	1	4
NOTURUS ALBATER	0	0	1	1	2
NOTURUS EXILIS	0	1	1	1	3
PERCINA CAPRODES	1	1	1	1	4
PHENACOBIUS MIRABILIS	1	1	1	1	4
PHOXINUS ERYTHROGASTER	1	1	1	1	4
PIMEPHALES NOTATUS	1	1	1	1	4
PIMEPHALES PROMELAS	0	1	0	0	1
SEMOTILUS ATROMACULATUS	1	1	1	1	4

Appendix 8: Component model evaluation statistics

Generalized Additive Models: Plains Subregion

Species	AUC	Bias	MAE
AMEIURUS MELAS	0.570	0.860	0.208
AMEIURUS NATALIS	0.597	1.055	0.139
APLODINOTUS GRUNNIENS	0.645	0.571	0.130
CAMPOSTOMA ANOMALUM PULLUM	0.766	0.972	0.087
CARPIODES CARPIO	0.884	0.785	0.077
CARPIODES CYPRINUS	0.726	1.231	0.042
CATOSTOMUS COMMERSONII	0.667	1.120	0.084
CYPRINELLA LUTRENSIS	0.736	0.925	0.087
DOROSOMA CEPEDIANUM	0.846	1.092	0.051
ETHEOSTOMA FLABELLARE	0.908	0.906	0.064
ETHEOSTOMA NIGRUM	0.813	0.936	0.109
ETHEOSTOMA SPECTABILE	0.858	0.934	0.093
FUNDULUS NOTATUS	0.900	1.142	0.040
GAMBUSIA AFFINIS	0.759	1.024	0.083
ICTALURUS PUNCTATUS	0.816	0.795	0.128
LEPISOSTEUS PLATOSTOMUS	0.887	0.743	0.063
LEPOMIS CYANELLUS	0.517	0.951	0.146
LEPOMIS HUMILIS	0.699	0.810	0.113
LEPOMIS MACROCHIRUS	0.584	1.062	0.093
LEPOMIS MEGALOTIS	0.816	0.565	0.115
LUXILUS CHRYSOCEPHALUS	0.898	0.944	0.045
LUXILUS CORNUTUS	0.899	0.841	0.063
LYTHRURUS UMBRATILIS	0.711	0.941	0.099
MICROPTERUS SALMOIDES	0.575	0.967	0.116
MOXOSTOMA ERYTHRURUM	0.795	1.105	0.059
NOTEMIGONUS CRYSOLEUCAS	0.637	1.000	0.103
NOTROPIS BOOPS	0.671	1.004	0.082
NOTROPIS DORSALIS	0.877	0.963	0.061
NOTROPIS STRAMINEUS	0.722	0.947	0.091
NOTURUS EXILIS	0.930	0.795	0.076
NOTURUS FLAVUS	0.696	1.074	0.041
PERCINA CAPRODES	0.831	0.675	0.088
PERCINA MACULATA	0.918	0.700	0.091
PERCINA PHOXOCEPHALA	0.953	1.065	0.076
PHENACOBIUS MIRABILIS	0.672	0.950	0.143
PIMEPHALES NOTATUS	0.788	1.014	0.087
PIMEPHALES PROMELAS	0.755	0.961	0.112
POMOXIS ANNULARIS	0.791	0.833	0.099

Species	AUC	Bias	MAE
PYLODICTIS OLIVARIS	0.740	0.747	0.076
SEMOTILUS ATROMACULATUS	0.701	0.960	0.091

Generalized Additive Models: Ozark Subregion

Species	AUC	Bias	MAE
AMBLOPLITES ARIOMMUS	0.918	0.631	0.092
AMBLOPLITES CONSTELLATUS	0.682	1.778	0.021
AMBLOPLITES RUPESTRIS	0.892	1.012	0.056
AMEIURUS MELAS	0.714	0.990	0.046
AMEIURUS NATALIS	0.651	0.977	0.126
CAMPOSTOMA ANOMALUM PULLUM	0.751	1.010	0.078
CAMPOSTOMA OLIGOLEPIS	0.836	0.949	0.077
CATOSTOMUS COMMERSONII	0.841	0.745	0.081
COTTUS BAIRDII	0.910	0.925	0.072
COTTUS CAROLINAE	0.829	0.986	0.073
COTTUS HYPSELURUS	0.884	0.816	0.074
CYPRINELLA GALACTURA	0.875	1.011	0.047
CYPRINELLA LUTRENSIS	0.940	0.914	0.073
CYPRINELLA WHIPPLEI	0.955	0.770	0.065
DOROSOMA CEPEDIANUM	0.628	1.536	0.063
ERIMYZON OBLONGUS	0.915	0.813	0.057
ETHEOSTOMA BLENNIOIDES	0.825	0.888	0.071
ETHEOSTOMA CAERULEUM	0.927	0.963	0.062
ETHEOSTOMA FLABELLARE	0.737	0.746	0.115
ETHEOSTOMA NIGRUM	0.872	0.840	0.063
ETHEOSTOMA PUNCTULATUM	0.874	0.962	0.057
ETHEOSTOMA SPECTABILE	0.813	0.933	0.060
ETHEOSTOMA TETRAZONUM	0.911	0.666	0.064
ETHEOSTOMA ZONALE	0.818	0.955	0.054
FUNDULUS CATENATUS	0.843	1.061	0.095
FUNDULUS NOTATUS	0.826	1.122	0.079
FUNDULUS OLIVACEUS	0.857	0.967	0.058
FUNDULUS SCIADICUS	0.955	0.935	0.057
GAMBUSIA AFFINIS	0.738	1.026	0.114
HYBOPSIS AMBLOPS	0.860	0.905	0.095
HYPENTELIUM NIGRICANS	0.721	1.043	0.105
ICTALURUS PUNCTATUS	0.750	1.324	0.048
LEPISOSTEUS OSSEUS	0.658	0.711	0.054
LEPOMIS CYANELLUS	0.646	0.974	0.095
LEPOMIS GULOSUS	0.761	0.895	0.057

Species	AUC	Bias	MAE
LEPOMIS HUMILIS	0.823	1.214	0.033
LEPOMIS MACROCHIRUS	0.673	1.036	0.088
LEPOMIS MEGALOTIS	0.645	0.985	0.126
LEPOMIS MICROLOPHUS	0.818	1.008	0.046
LUXILUS CARDINALIS	0.901	1.157	0.021
LUXILUS CHRYSOCEPHALUS	0.828	1.122	0.061
LUXILUS CORNUTUS	0.967	1.179	0.046
LUXILUS PILSBRYI	0.963	1.032	0.039
LUXILUS ZONATUS	0.921	0.984	0.037
LYTHRURUS UMBRATILIS	0.788	0.905	0.077
MICROPTERUS DOLOMIEU	0.766	1.001	0.128
MICROPTERUS PUNCTULATUS	0.818	0.993	0.060
MICROPTERUS SALMOIDES	0.696	1.003	0.101
MINYTREMA MELANOPS	0.875	1.849	0.031
MOXOSTOMA DUQUESNEI	0.752	1.002	0.089
MOXOSTOMA ERYTHRURUM	0.720	1.078	0.114
MOXOSTOMA MACROLEPIDOTUM	0.689	0.950	0.044
NOCOMIS ASPER	0.911	0.833	0.056
NOCOMIS BIGUTTATUS	0.851	0.947	0.109
NOTEMIGONUS CRYSOLEUCAS	0.671	0.750	0.070
NOTROPIS BOOPS	0.835	1.197	0.096
NOTROPIS GREENEI	0.920	0.997	0.078
NOTROPIS NUBILUS	0.762	0.977	0.076
NOTROPIS STRAMINEUS	0.874	1.175	0.058
NOTROPIS TELESCOPUS	0.944	0.938	0.081
NOTURUS ALBATER	0.939	0.964	0.102
NOTURUS EXILIS	0.673	0.922	0.113
PERCINA CAPRODES	0.788	0.953	0.052
PHENACOBIUS MIRABILIS	0.808	1.043	0.044
PHOXINUS ERYTHROGASTER	0.810	1.010	0.075
PIMEPHALES NOTATUS	0.878	0.964	0.067
PIMEPHALES PROMELAS	0.595	0.602	0.067
SEMOTILUS ATROMACULATUS	0.767	1.011	0.088

Multivariate Adaptive Regression Splines: Plains Subregion

Species	AUC	Bias	MAE
AMEIURUS MELAS	0.527	0.865	0.143
AMEIURUS NATALIS	0.598	1.002	0.164
APLODINOTUS GRUNNIENS	0.688	1.733	0.080
CAMPOSTOMA ANOMALUM PULLUM	0.680	1.020	0.161

Species	AUC	Bias	MAE
CARPIODES CARPIO	0.725	0.834	0.093
CARPIODES CYPRINUS	0.815	0.817	0.065
CATOSTOMUS COMMERSONII	0.601	1.111	0.159
CYPRINELLA LUTRENSIS	0.874	1.052	0.076
DOROSOMA CEPEDIANUM	0.763	0.737	0.067
ETHEOSTOMA FLABELLARE	0.866	1.006	0.056
ETHEOSTOMA NIGRUM	0.785	1.041	0.107
ETHEOSTOMA SPECTABILE	0.904	0.943	0.080
FUNDULUS NOTATUS	0.914	0.960	0.064
GAMBUSIA AFFINIS	0.786	1.130	0.090
ICTALURUS PUNCTATUS	0.716	0.995	0.130
LEPISOSTEUS PLATOSTOMUS	0.713	1.267	0.040
LEPOMIS CYANELLUS	0.582	1.001	0.044
LEPOMIS HUMILIS	0.683	0.710	0.107
LEPOMIS MACROCHIRUS	0.565	0.980	0.120
LEPOMIS MEGALOTIS	0.829	0.946	0.063
LUXILUS CHRYSOCEPHALUS	0.844	0.847	0.054
LUXILUS CORNUTUS	0.865	1.221	0.054
LYTHRURUS UMBRATILIS	0.735	1.080	0.091
MICROPTERUS SALMOIDES	0.520	0.974	0.186
MOXOSTOMA ERYTHRURUM	0.879	0.751	0.064
NOTEMIGONUS CRYSOLEUCAS	0.653	1.194	0.123
NOTROPIS BOOPS	0.942	0.858	0.059
NOTROPIS DORSALIS	0.862	1.091	0.098
NOTROPIS STRAMINEUS	0.737	1.046	0.125
NOTURUS EXILIS	0.923	0.841	0.067
NOTURUS FLAVUS	0.650	0.896	0.047
PERCINA CAPRODES	0.859	1.439	0.046
PERCINA MACULATA	0.847	0.778	0.066
PERCINA PHOXOCEPHALA	0.931	0.823	0.077
PHENACOBIUS MIRABILIS	0.742	1.043	0.091
PIMEPHALES NOTATUS	0.756	0.942	0.097
PIMEPHALES PROMELAS	0.767	0.992	0.109
POMOXIS ANNULARIS	0.655	1.256	0.076
PYLODICTIS OLIVARIS	0.865	0.847	0.057
SEMOTILUS ATROMACULATUS	0.756	0.997	0.065

Multivariate Adaptive Regression Splines: Ozark Subregion

Species	AUC	Bias	MAE
AMBLOPLITES ARIOMMUS	0.895	1.254	0.040
AMBLOPLITES CONSTELLATUS	0.968	0.712	0.082

Species	AUC	Bias	MAE
AMBLOPLITES RUPESTRIS	0.870	0.886	0.109
AMEIURUS MELAS	0.442	0.555	0.064
AMEIURUS NATALIS	0.560	1.039	0.199
CAMPOSTOMA ANOMALUM PULLUM	0.739	1.050	0.055
CAMPOSTOMA OLIGOLEPIS	0.847	1.067	0.088
CATOSTOMUS COMMERSONII	0.829	0.892	0.072
COTTUS BAIRDII	0.908	0.966	0.065
COTTUS CAROLINAE	0.827	0.989	0.087
COTTUS HYPSELURUS	0.748	0.969	0.050
CYPRINELLA GALACTURA	0.898	0.515	0.072
CYPRINELLA LUTRENSIS	0.967	0.902	0.055
CYPRINELLA WHIPPLEI	0.961	0.960	0.064
DOROSOMA CEPEDIANUM	0.651	1.429	0.046
ERIMYZON OBLONGUS	0.820	1.175	0.037
ETHEOSTOMA BLENNIOIDES	0.756	0.931	0.111
ETHEOSTOMA CAERULEUM	0.882	0.996	0.079
ETHEOSTOMA FLABELLARE	0.710	0.912	0.077
ETHEOSTOMA NIGRUM	0.847	1.088	0.082
ETHEOSTOMA PUNCTULATUM	0.817	1.145	0.088
ETHEOSTOMA SPECTABILE	0.747	1.111	0.105
ETHEOSTOMA TETRAZONUM	0.800	0.815	0.053
ETHEOSTOMA ZONALE	0.795	1.184	0.076
FUNDULUS CATENATUS	0.860	1.061	0.101
FUNDULUS NOTATUS	0.809	1.114	0.063
FUNDULUS OLIVACEUS	0.773	1.040	0.102
FUNDULUS SCIADICUS	0.804	0.771	0.056
GAMBUSIA AFFINIS	0.734	0.997	0.084
HYBOPSIS AMBLOPS	0.855	0.961	0.055
HYPENTELIUM NIGRICANS	0.660	0.961	0.139
ICTALURUS PUNCTATUS	0.872	1.412	0.033
LEPISOSTEUS OSSEUS	0.678	0.852	0.050
LEPOMIS CYANELLUS	0.630	1.160	0.161
LEPOMIS GULOSUS	0.657	1.070	0.026
LEPOMIS HUMILIS	0.801	1.044	0.036
LEPOMIS MACROCHIRUS	0.628	0.935	0.138
LEPOMIS MEGALOTIS	0.639	0.978	0.087
LEPOMIS MICROLOPHUS	0.600	0.588	0.077
LUXILUS CARDINALIS	0.993	0.873	0.053
LUXILUS CHRYSOCEPHALUS	0.852	1.066	0.067
LUXILUS CORNUTUS	0.926	1.007	0.053
LUXILUS PILSBRYI	0.996	0.961	0.046

Species	AUC	Bias	MAE
LUXILUS ZONATUS	0.913	1.015	0.049
LYTHRURUS UMBRATILIS	0.728	1.310	0.043
MICROPTERUS DOLOMIEU	0.749	1.051	0.108
MICROPTERUS PUNCTULATUS	0.826	0.930	0.049
MICROPTERUS SALMOIDES	0.572	1.084	0.179
MINYTREMA MELANOPS	0.769	1.083	0.038
MOXOSTOMA DUQUESNEI	0.717	0.931	0.109
MOXOSTOMA ERYTHRURUM	0.691	1.068	0.115
MOXOSTOMA MACROLEPIDOTUM	0.815	0.790	0.054
NOCOMIS ASPER	0.971	0.666	0.087
NOCOMIS BIGUTTATUS	0.900	1.033	0.066
NOTEMIGONUS CRYSOLEUCAS	0.654	0.965	0.033
NOTROPIS BOOPS	0.836	0.875	0.107
NOTROPIS GREENEI	0.905	1.248	0.042
NOTROPIS NUBILUS	0.758	1.084	0.126
NOTROPIS STRAMINEUS	0.912	1.069	0.056
NOTROPIS TELESCOPUS	0.950	0.919	0.094
NOTURUS ALBATER	0.894	1.328	0.061
NOTURUS EXILIS	0.665	1.264	0.143
PERCINA CAPRODES	0.661	1.209	0.084
PHENACOBIUS MIRABILIS	0.691	1.066	0.039
PHOXINUS ERYTHROGASTER	0.812	1.090	0.065
PIMEPHALES NOTATUS	0.845	0.935	0.088
PIMEPHALES PROMELAS	0.625	0.611	0.054
SEMOTILUS ATROMACULATUS	0.779	1.052	0.097

Boosted Regression Trees: Plains Subregion

Species	AUC	Bias	MAE
AMEIURUS MELAS	0.478	0.899	0.174
AMEIURUS NATALIS	0.684	0.858	0.116
APLODINOTUS GRUNNIENS	0.910	1.363	0.055
CAMPOSTOMA ANOMALUM PULLUM	0.796	1.085	0.090
CARPIODES CARPIO	0.824	0.887	0.064
CARPIODES CYPRINUS	0.814	1.916	0.049
CATOSTOMUS COMMERSONII	0.724	1.036	0.126
CYPRINELLA LUTRENSIS	0.840	0.977	0.059
DOROSOMA CEPEDIANUM	0.803	1.450	0.072
ETHEOSTOMA FLABELLARE	0.943	1.125	0.055
ETHEOSTOMA NIGRUM	0.796	0.926	0.125
ETHEOSTOMA SPECTABILE	0.882	0.843	0.110

Species	AUC	Bias	MAE
FUNDULUS NOTATUS	0.969	1.166	0.082
GAMBUSIA AFFINIS	0.817	0.928	0.082
ICTALURUS PUNCTATUS	0.849	0.884	0.078
LEPISOSTEUS PLATOSTOMUS	0.845	0.835	0.053
LEPOMIS HUMILIS	0.711	1.042	0.064
LEPOMIS MACROCHIRUS	0.643	0.947	0.091
LEPOMIS MEGALOTIS	0.966	0.742	0.077
LUXILUS CHRYSOCEPHALUS	0.988	0.980	0.068
LUXILUS CORNUTUS	0.888	1.036	0.048
LYTHRURUS UMBRATILIS	0.715	0.892	0.136
MICROPTERUS SALMOIDES	0.647	1.017	0.095
MOXOSTOMA ERYTHRURUM	0.870	1.217	0.041
NOTEMIGONUS CRYSOLEUCAS	0.643	0.817	0.135
NOTROPIS BOOPS	0.964	0.760	0.086
NOTROPIS DORSALIS	0.897	1.038	0.072
NOTROPIS STRAMINEUS	0.792	0.870	0.119
NOTURUS EXILIS	0.952	1.031	0.073
NOTURUS FLAVUS	0.702	1.054	0.045
PERCINA CAPRODES	0.865	1.034	0.075
PERCINA MACULATA	0.864	2.491	0.046
PERCINA PHOXOCEPHALA	0.947	1.150	0.061
PHENACOBIUS MIRABILIS	0.666	1.104	0.112
PIMEPHALES NOTATUS	0.823	1.021	0.033
PIMEPHALES PROMELAS	0.847	0.880	0.082
POMOXIS ANNULARIS	0.676	1.071	0.092
PYLODICTIS OLIVARIS	0.917	0.880	0.063
SEMOTILUS ATROMACULATUS	0.811	0.999	0.057

Boosted Regression Trees: Ozark Subregion

Species	AUC	Bias	MAE
AMBLOPLITES ARIOMMUS	0.916	0.924	0.080
AMBLOPLITES CONSTELLATUS	0.979	0.779	0.068
AMBLOPLITES RUPESTRIS	0.909	0.911	0.057
AMEIURUS MELAS	0.759	0.818	0.041
AMEIURUS NATALIS	0.693	1.203	0.128
CAMPOSTOMA ANOMALUM PULLUM	0.827	1.011	0.030
CAMPOSTOMA OLIGOLEPIS	0.891	1.010	0.094
CATOSTOMUS COMMERSONII	0.826	0.925	0.053
COTTUS BAIRDII	0.924	1.146	0.047
COTTUS CAROLINAE	0.824	0.929	0.088

Species	AUC	Bias	MAE
COTTUS HYPSELURUS	0.852	0.877	0.072
CYPRINELLA GALACTURA	0.912	2.094	0.030
CYPRINELLA LUTRENSIS	0.942	0.902	0.070
CYPRINELLA WHIPPLEI	0.977	1.412	0.066
DOROSOMA CEPEDIANUM	0.739	0.908	0.033
ERIMYZON OBLONGUS	0.784	0.889	0.073
ETHEOSTOMA BLENNIOIDES	0.819	0.991	0.066
ETHEOSTOMA CAERULEUM	0.920	0.985	0.063
ETHEOSTOMA FLABELLARE	0.743	0.907	0.095
ETHEOSTOMA NIGRUM	0.854	0.981	0.052
ETHEOSTOMA PUNCTULATUM	0.871	0.928	0.067
ETHEOSTOMA SPECTABILE	0.827	1.040	0.059
ETHEOSTOMA TETRAZONUM	0.927	1.528	0.056
ETHEOSTOMA ZONALE	0.859	1.095	0.061
FUNDULUS CATENATUS	0.887	1.034	0.061
FUNDULUS NOTATUS	0.835	0.957	0.074
FUNDULUS OLIVACEUS	0.831	1.030	0.067
FUNDULUS SCIADICUS	0.922	1.396	0.044
GAMBUSIA AFFINIS	0.786	0.927	0.059
HYBOPSIS AMBLOPS	0.907	0.806	0.060
HYPENTELIUM NIGRICANS	0.733	0.926	0.087
ICTALURUS PUNCTATUS	0.872	1.129	0.046
LEPISOSTEUS OSSEUS	0.624	1.406	0.027
LEPOMIS CYANELLUS	0.687	0.969	0.080
LEPOMIS GULOSUS	0.697	1.020	0.031
LEPOMIS HUMILIS	0.907	1.144	0.026
LEPOMIS MACROCHIRUS	0.698	1.032	0.092
LEPOMIS MEGALOTIS	0.691	0.987	0.115
LEPOMIS MICROLOPHUS	0.838	1.367	0.072
LUXILUS CARDINALIS	0.999	1.117	0.046
LUXILUS CHRYSOCEPHALUS	0.872	1.013	0.038
LUXILUS CORNUTUS	0.979	1.038	0.057
LUXILUS PILSBRYI	1.000	1.033	0.051
LUXILUS ZONATUS	0.961	0.981	0.059
LYTHRURUS UMBRATILIS	0.830	1.061	0.038
MICROPTERUS DOLOMIEU	0.833	0.946	0.080
MICROPTERUS PUNCTULATUS	0.767	1.225	0.059
MICROPTERUS SALMOIDES	0.732	0.975	0.079
MINYTREMA MELANOPS	0.797	0.841	0.046
MOXOSTOMA DUQUESNEI	0.784	1.049	0.046
MOXOSTOMA ERYTHRURUM	0.793	1.052	0.046
MOXOSTOMA MACROLEPIDOTUM	0.733	1.480	0.037

Species	AUC	Bias	MAE
NOCOMIS ASPER	0.988	0.933	0.070
NOCOMIS BIGUTTATUS	0.878	0.944	0.085
NOTEMIGONUS CRYSOLEUCAS	0.637	1.076	0.072
NOTROPIS BOOPS	0.846	0.982	0.052
NOTROPIS GREENEI	0.934	1.132	0.064
NOTROPIS NUBILUS	0.827	1.000	0.085
NOTROPIS STRAMINEUS	0.882	0.974	0.091
NOTROPIS TELESCOPUS	0.944	0.820	0.082
NOTURUS ALBATER	0.963	1.489	0.067
NOTURUS EXILIS	0.703	0.870	0.105
PERCINA CAPRODES	0.804	0.924	0.038
PHENACOBIUS MIRABILIS	0.858	1.018	0.060
PHOXINUS ERYTHROGASTER	0.838	0.933	0.065
PIMEPHALES NOTATUS	0.882	0.984	0.058
PIMEPHALES PROMELAS	0.607	0.920	0.083
SEMOTILUS ATROMACULATUS	0.807	0.944	0.082

Random Forest: Plains Subregion

Species	AUC	Bias	MAE
AMEIURUS MELAS	0.697	0.746	0.240
AMEIURUS NATALIS	0.606	0.950	0.117
APLODINOTUS GRUNNIENS	0.949	1.137	0.073
CAMPOSTOMA ANOMALUM PULLUM	0.842	0.925	0.078
CARPIODES CARPIO	0.779	1.088	0.054
CARPIODES CYPRINUS	0.897	0.829	0.112
CATOSTOMUS COMMERSONII	0.692	1.059	0.097
CYPRINELLA LUTRENSIS	0.860	1.015	0.072
DOROSOMA CEPEDIANUM	0.796	1.397	0.042
ETHEOSTOMA FLABELLARE	0.905	0.817	0.069
ETHEOSTOMA NIGRUM	0.855	1.027	0.077
ETHEOSTOMA SPECTABILE	0.928	0.818	0.124
FUNDULUS NOTATUS	0.940	1.029	0.059
GAMBUSIA AFFINIS	0.761	0.868	0.106
ICTALURUS PUNCTATUS	0.891	1.042	0.078
LEPISOSTEUS PLATOSTOMUS	0.784	0.738	0.115
LEPOMIS CYANELLUS	0.542	0.959	0.134
LEPOMIS HUMILIS	0.770	1.313	0.091
LEPOMIS MACROCHIRUS	0.527	1.110	0.189
LEPOMIS MEGALOTIS	0.943	1.052	0.048
LUXILUS CHRYSOCEPHALUS	0.953	1.182	0.090

Species	AUC	Bias	MAE
LUXILUS CORNUTUS	0.964	1.048	0.077
LYTHRURUS UMBRATILIS	0.766	0.942	0.088
MICROPTERUS SALMOIDES	0.666	0.916	0.115
MOXOSTOMA ERYTHRURUM	0.910	1.167	0.049
NOTEMIGONUS CRYSOLEUCAS	0.545	0.971	0.181
NOTROPIS BOOPS	0.970	0.665	0.083
NOTROPIS DORSALIS	0.933	1.038	0.067
NOTROPIS STRAMINEUS	0.847	0.964	0.052
NOTURUS EXILIS	0.921	0.912	0.060
NOTURUS FLAVUS	0.728	0.763	0.079
PERCINA CAPRODES	0.774	1.588	0.078
PERCINA MACULATA	0.915	0.605	0.092
PERCINA PHOXOCEPHALA	0.909	1.138	0.043
PHENACOBIUS MIRABILIS	0.623	0.986	0.184
PIMEPHALES NOTATUS	0.822	1.121	0.111
PIMEPHALES PROMELAS	0.791	1.082	0.074
POMOXIS ANNULARIS	0.702	0.777	0.103
PYLODICTIS OLIVARIS	0.937	0.685	0.107
SEMOTILUS ATROMACULATUS	0.770	1.021	0.082

Random Forest: Ozark Subregion

Species	AUC	Bias	MAE
AMBLOPLITES ARIOMMUS	0.931	1.070	0.065
AMBLOPLITES CONSTELLATUS	0.982	1.075	0.064
AMBLOPLITES RUPESTRIS	0.931	0.931	0.057
AMEIURUS MELAS	0.673	1.237	0.070
AMEIURUS NATALIS	0.719	0.928	0.090
CAMPOSTOMA ANOMALUM PULLUM	0.783	0.982	0.073
CAMPOSTOMA OLIGOLEPIS	0.882	1.083	0.061
CATOSTOMUS COMMERSONII	0.847	0.986	0.051
COTTUS BAIRDII	0.917	1.068	0.039
COTTUS CAROLINAE	0.853	0.984	0.074
COTTUS HYPSELURUS	0.850	0.764	0.112
CYPRINELLA GALACTURA	0.909	0.789	0.062
CYPRINELLA LUTRENSIS	0.962	1.056	0.061
CYPRINELLA WHIPPLEI	0.973	0.788	0.063
DOROSOMA CEPEDIANUM	0.818	0.695	0.067
ERIMYZON OBLONGUS	0.824	0.791	0.098
ETHEOSTOMA BLENNIOIDES	0.874	0.980	0.076
ETHEOSTOMA CAERULEUM	0.924	1.011	0.051

Species	AUC	Bias	MAE
ETHEOSTOMA FLABELLARE	0.813	1.042	0.065
ETHEOSTOMA NIGRUM	0.877	1.111	0.065
ETHEOSTOMA PUNCTULATUM	0.907	1.030	0.054
ETHEOSTOMA SPECTABILE	0.807	1.000	0.056
ETHEOSTOMA TETRAZONUM	0.964	0.999	0.067
ETHEOSTOMA ZONALE	0.864	0.874	0.063
FUNDULUS CATENATUS	0.860	0.996	0.060
FUNDULUS NOTATUS	0.868	1.136	0.036
FUNDULUS OLIVACEUS	0.880	1.033	0.071
FUNDULUS SCIADICUS	0.952	0.988	0.064
GAMBUSIA AFFINIS	0.755	0.932	0.080
HYBOPSIS AMBLOPS	0.890	1.240	0.054
HYPENTELIUM NIGRICANS	0.711	1.043	0.085
ICTALURUS PUNCTATUS	0.884	0.765	0.059
LEPISOSTEUS OSSEUS	0.753	0.905	0.063
LEPOMIS CYANELLUS	0.654	1.002	0.096
LEPOMIS GULOSUS	0.721	1.305	0.059
LEPOMIS HUMILIS	0.839	1.202	0.042
LEPOMIS MACROCHIRUS	0.720	1.046	0.123
LEPOMIS MEGALOTIS	0.720	0.983	0.108
LEPOMIS MICROLOPHUS	0.772	0.900	0.061
LUXILUS CARDINALIS	0.997	0.929	0.054
LUXILUS CHRYSOCEPHALUS	0.883	0.991	0.061
LUXILUS CORNUTUS	0.971	0.823	0.065
LUXILUS PILSBRYI	1.000	1.033	0.056
LUXILUS ZONATUS	0.950	0.929	0.071
LYTHRURUS UMBRATILIS	0.896	1.052	0.052
MICROPTERUS DOLOMIEU	0.848	1.005	0.078
MICROPTERUS PUNCTULATUS	0.786	1.209	0.035
MICROPTERUS SALMOIDES	0.638	0.945	0.138
MINYTREMA MELANOPS	0.845	0.814	0.061
MOXOSTOMA DUQUESNEI	0.762	0.984	0.076
MOXOSTOMA ERYTHRURUM	0.811	1.084	0.051
MOXOSTOMA MACROLEPIDOTUM	0.837	1.094	0.099
NOCOMIS ASPER	0.996	0.988	0.059
NOCOMIS BIGUTTATUS	0.916	0.881	0.083
NOTEMIGONUS CRYSOLEUCAS	0.654	0.817	0.102
NOTROPIS BOOPS	0.896	0.836	0.074
NOTROPIS GREENEI	0.914	0.917	0.057
NOTROPIS NUBILUS	0.827	1.041	0.065
NOTROPIS STRAMINEUS	0.937	0.990	0.068
NOTROPIS TELESCOPUS	0.965	0.858	0.065

Species	AUC	Bias	MAE
NOTURUS ALBATER	0.918	1.146	0.053
NOTURUS EXILIS	0.700	0.916	0.106
PERCINA CAPRODES	0.775	0.980	0.043
PHENACOBIUS MIRABILIS	0.773	1.213	0.044
PHOXINUS ERYTHROGASTER	0.853	0.979	0.096
PIMEPHALES NOTATUS	0.873	1.055	0.065
PIMEPHALES PROMELAS	0.675	1.356	0.040
SEMOTILUS ATROMACULATUS	0.794	1.134	0.088

Appendix 9: Electronic data with PDF of species distribution models and a shapefile of species distribution model results.

Appendix 10: Number of stream segments within each established conservation network each species is predicted to occupy

Species_Code	COA Segments	ECN Segments	PW Segments
A_ARIOMMUS	333.224	312.602	190.134
A_CONSTELLATUS	191.073	202.543	370.077
A_GRUNNIENS	312.024	346.505	276.972
A_MELAS	356.84	425.111	680.19
A_NATALIS	1266.47	1371.009	1945.802
A_NEBULOSUS	12.516	10.38	17.072
A_RUPESTRIS	324.638	407.552	608.175
A_SAYANUS	77.158	74.228	125.509
C_BAIRDII	439.734	492.169	625.104
C_CAMURA	27.348	6.978	13.322
C_CAROLINAE	832.542	930.886	1009.521
C_CARPIO	242.071	384.908	360.436
C_COMMERSONII	548.134	742.002	1282.065
C_CYPRINUS	198.531	254.096	329.692
C_GALACTURA	222.156	190.053	229.144
C_HYPSELURUS	412.178	464.614	300.444
C_LUTRENSIS	912.407	1326.801	1810.257
C_MACROPTERUS	33.6	15.8	6.2
C_OLIGOLEPIS	1456.787	1507.307	1630.861
C_PULLUM	2125.527	2330.205	3049.566
C_SPILOPTERA	84.163	80.43	186.09
C_VELIFER	6.104	8.204	11.708
C_VENUSTA	170.177	146.401	208.932
C_WHIPPLEI	158.337	187.606	491.829
D_CEPEDIANUM	567.991	560.338	535.041
E_BLENNIOIDES	1138.156	1146.546	1648.462
E_BURRI	92.391	88.268	27.531
E_CAERULEUM	1721.378	1720.852	2072.02
E_CRAGINI	50.004	19.332	30.192
E_EUZONUM	22.25	20.93	8.06
E_FLABELLARE	918.181	926.81	1358.272
E_GRACILE	58.168	54.488	55.796
E_HARRYI	139.302	130.406	81.804
E_JULIAE	95.721	107.23	173.854
E_MICROPERCA	57.04	75.769	37.193
E_NIANGUAE	33.792	37.879	54.502
E_NIGER	106.802	83.763	57.2
E_NIGRUM	564.169	790.43	1528.231
E_OBLONGUS	299.748	264.634	398.976

E_PROELIARE	55.124	58.456	58.104
E_PUNCTULATUM	427.052	493.718	725.927
E_SPECTABILE	1176.104	1206.013	1931.835
E_STIGMAEUM	86.117	73.076	58.28
E_TETRAZONUM	183.452	236.027	260.582
E_UNIPORUM	180.541	168.859	54.685
E_VERMICULATUS	125.286	115.482	110.629
E_XPUNCTATUS	57.672	63.74	132.266
E_ZONALE	476.071	481.235	570.76
F_CATENATUS	1557.236	1549.964	1737.935
F_DISPAR	31.354	12.418	10.51
F_NOTATUS	600.436	576.15	1139.801
F_OLIVACEUS	1449.142	1433.267	2005.976
F_SCIADICUS	105.019	146.294	111.938
G_AFFINIS	988.319	1123.657	1391.275
H_AMBLOPS	489.888	398.22	607.802
H_ARGYRITIS	3.842	7.751	11.333
H_HANKINSONI	34.222	54.862	106.378
H_NIGRICANS	1317.83	1316.261	1632.585
H_PLACITUS	7.312	13.952	21.573
I_BUBALUS	166.85	125.569	156.312
I_CYPRINELLUS	143.521	115.563	129.897
I_FOSSOR	5.642	5.385	16.537
I_GAGEI	5.48	5.152	5.376
I_NIGER	2.96	5.288	7.272
I_PUNCTATUS	577.307	785.867	845.382
L_AEPYPTERA	102.171	99.879	59.139
L_CARDINALIS	96.268	87.251	147.858
L_CHRYSOCEPHALUS	1167.124	1173.394	1922.021
L_CORNUTUS	179.518	225.801	337.47
L_CYANELLUS	2397.419	2709.253	3684.798
L_GULOSUS	345.131	289.807	271.16
L_HUMILIS	469.552	482.951	739.338
L_MACROCHIRUS	2418.909	2615.015	3471.637
L_MEGALOTIS	2064.565	1970.876	2438.593
L_MICROLOPHUS	185.973	175.574	282.198
L_MINIATUS	196.628	194.65	163.016
L_OCULATUS	114.438	74.358	149.09
L_OSSEUS	219.758	185.294	236.645
L_PILSBRYI	354.921	383.906	607.76
L_PLATOSTOMUS	337.486	345.116	234.484
L_UMBRATILIS	489.611	538.704	1079.536
L_ZONATUS	1566.112	1579.114	1625.662

M_ANISURUM	48.501	59.233	122.735
M_CARINATUM	14.184	20.616	41.304
M_CHRYSOPS	53.952	59.586	51.379
M_DOLOMIEU	1197.13	1232.507	1469.864
M_DUQUESNEI	567.748	614.134	857.632
M_ERYTHRURUM	658.646	649.264	1059.4
M_MACROLEPIDOTUM	190.842	193.195	237.92
M_MELANOPS	225.481	145.166	177.67
M_PUNCTULATUS	400.249	383.568	472.881
M_SALMOIDES	1664.5	1857.451	2758.534
M_STORERIANA	15.78	19.042	24.607
N_ALBATER	413.477	358.548	260.499
N_ASPER	42.447	50.022	94.257
N_ATHERINOIDES	219.081	185.473	259.305
N_BIGUTTATUS	1388.826	1380.565	1458.036
N_BOOPS	751.893	681.436	1267.109
N_BUCCATUS	57.893	57.814	119.696
N_BUCHANANI	22.862	40.274	51.512
N_CRYSOLEUCAS	477.946	574.028	778.319
N_DORSALIS	313.691	638.961	1139.174
N_EXILIS	1230.81	1282.544	1899.379
N_FLAVATER	68.018	59.236	41.846
N_FLAVUS	119.489	182.883	238.85
N_GREENEI	494.088	476.583	463.179
N_GYRINUS	80.633	109.355	206.751
N_HETEROLEPIS	33.93	42.708	26.84
N_MIURUS	79.458	69.785	29.361
N_NOCTURNUS	49.718	47.006	36.682
N_NUBILUS	1536.215	1514.813	1737.241
N_OZARCANUS	119.198	110.086	83.054
N_STRAMINEUS	670.381	1117.532	1693.882
N_TELESCOPUS	683.748	609.556	516.686
N_TEXANUS	57.732	59.286	104.15
N_VOLUCELLUS	221.653	152.981	286.023
O_EMILIAE	87.132	76.933	84.607
P_ANNULARIS	351.843	452.291	441.372
P_CAPRODES	491.314	453.792	744.809
P_COPELANDI	20.935	4.969	8.262
P_CYMATOTAENIA	49.108	76.411	30.437
P_ERYTHROGASTER	798.875	849.866	908.488
P_EVIDES	87.196	76.751	132.765
P_FLAVESCENS	3.222	3.573	5.724
P_MACULATA	85.65	160.374	284.123

P_MIRABILIS	422.573	685.439	989.015
P_NIGROMACULATUS	117.195	123.195	167.549
P_NOTATUS	1653.102	1948.989	2984.05
P_OLIVARIS	209.695	264.393	209.786
P_PHOXOCEPHALA	164.651	228.351	198.93
P_PROMELAS	417.229	694.736	1116.33
P_SCIERA	48.423	47.53	37.859
P_VIGILAX	106.931	131.265	197.611
S_ATROMACULATUS	1648.951	2005.182	2772.665
S_VITREUS	14.106	16.863	26.287

Appendix 11: Electronic data with PDF of maps and a shapefile which contain the results for all conservation planning results