



Development and assessment of indices to determine stream fish vulnerability to climate change and habitat alteration



Nicholas A. Sievert^{a,*}, Craig P. Paukert^b, Yin-Phan Tsang^c, Dana Infante^d

^a Missouri Cooperative Fish and Wildlife Research Unit, Department of Fisheries and Wildlife Sciences, University of Missouri, 302 Anheuser-Busch Natural Resources Building, Columbia, MO 65211, United States

^b U.S. Geological Survey, Missouri Cooperative Fish and Wildlife Research Unit, Department of Fisheries and Wildlife Sciences, University of Missouri, 302 Anheuser-Busch Natural Resources Building, Columbia, MO 65211, United States

^c Department of Natural Resources and Environmental Management, University of Hawaii, Sherman 243, 1910 East-West Road, Honolulu, HI 96822, United States

^d Department of Fisheries and Wildlife, Michigan State University, Manly Miles Building, Suite 318, 1405 South Harrison Road, East Lansing, MI 48823, United States

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ABSTRACT

Understanding the future impacts 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, USA based on species environmental tolerances, rarity, range size, dispersal ability and on the average connectivity of the streams occupied by each species. These two indices differed in how environmental tolerance was classified (i.e., vulnerability to habitat alteration, changes in stream temperature, and changes to flow regimes). Environmental tolerance was classified based on measured species responses to habitat alteration, and extremes in stream temperatures and flow conditions for one index, while environmental tolerance for the second index was based on species' traits. The indices were compared to determine if vulnerability scores differed by index or state listing status. We also evaluated the spatial distribution of species classified as vulnerable to habitat alteration, changes in stream temperature, and change in flow regimes. Vulnerability scores were calculated for all 133 species with the trait association index, while only 101 species were evaluated using the species response index, because 32 species lacked data to analyze for a response. Scores from the trait association index were greater than the species response index. This is likely due to the species response index's inability to evaluate many rare species, which generally had high vulnerability scores for the trait association index. The indices were consistent in classifying vulnerability to habitat alteration, but varied in their classification of vulnerability due to increases in stream temperature and alterations to flow regimes, likely because extremes in current climate may not fully capture future conditions and their influence on stream fish communities. Both indices showed higher mean vulnerability scores for listed species than unlisted species, which provided a coarse measure of validation. Our indices classified species identified as being in need of conservation by the state of Missouri as highly vulnerable. The distribution of vulnerable species in Missouri showed consistent patterns between indices, with the more forest-dominated, groundwater fed streams in the Ozark subregion generally having higher numbers and proportions of vulnerable species per site than subregions that were agriculturally dominated with more overland flow. These results suggest that both indices will identify similar habitats as conservation action targets despite discrepancies in the classification of vulnerable species. Our vulnerability assessment provides a framework that can be refined and used in other regions.

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1. 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). Anthropogenic habitat alteration, changes in stream temperatures, and change in flow regimes

* Corresponding author. Tel.: +1 9204952645.

E-mail addresses: NAS4tf@mail.missouri.edu (N.A. Sievert), paukertc@missouri.edu (C.P. Paukert), tsangy@hawaii.edu (Y.-P. Tsang), infanted@anr.msu.edu (D. Infante).

will likely cause continuing declines in aquatic biota (Ricciardi and Rasmussen, 1999; Dudgeon et al., 2006). Conservation actions to protect aquatic biodiversity are critical for preventing future biodiversity losses (Master et al., 1998). In order to plan for long-term biodiversity conservation, a better understanding of how impacts and threats, such as climate change and habitat alteration, affect aquatic species is needed. 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). An assessment of stream fish vulnerability requires the identification of species which are threatened by primary drivers of future stream fish declines such as habitat alteration, predicted changes in stream temperatures, and predicted change in flow regimes (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, USA, the average annual air temperature is expected to rise approximately 4 °C by 2095 based on an intermediate (A1B) climate scenario (Girvetz et al., 2009). This is expected to increase stream temperature by an average of 3.6 °C (Eaton and Scheller, 1996), which may have a substantial effect on stream fishes. In Wisconsin, USA, a 3 °C air temperature increase is predicted to result in loss of 343,034 km of stream habitat for cool- and cold-water fishes, which includes a species extirpation (Lyons et al., 2010). Studies have predicted substantial losses of cool- and cold-water stream habitats (15–50%) with a doubling of atmospheric CO₂ (Eaton and Scheller, 1996; Mohseni et al., 2003). Warm-water streams in the Great Plains, USA, achieve maximum temperatures at or near the physiological limits of some resident fish species, and an increase in stream temperature of just a few degrees is predicted to result in local extirpation and extinction (Matthews and Zimmerman, 1990). In addition, decreases in fitness based on changes in energetics and growth may be a non-lethal consequence of warming temperatures (Pease and Paukert, 2014; Westhoff and Paukert, 2014). These studies provide strong evidence that some stream fish species may decline or face local extirpations as stream temperatures warm.

Changes to flow regimes are also expected to have substantial impacts on stream fish (Poff et al., 2002). Predicted increased frequency and extremes of flood and droughts may lead to shifts in species composition and local species extirpations (Poff et al., 1997, 2002). In Missouri, mean annual precipitation is expected to increase only slightly (~10 mm) between historic (1965–2015) and future (2080–2100) time periods, however this precipitation is expected to come in the form of heavier precipitation (~10 mm increase on wet days) on fewer days (12 less wet days) (Girvetz et al., 2009). 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).

Habitat alteration caused by anthropogenic modifications including the conversion of land to agricultural or urban uses or the direct modification of streams and rivers from channelization, dredging, and damming often result in altered and degraded stream conditions and losses of aquatic biodiversity (Malmqvist and Rundle, 2002; Allan, 2004). Indices of biological integrity (IBIs) have identified many stream fish species as intolerant of habitat alteration (Esselman et al., 2011), and life history traits sensitive to the effects of habitat alteration such as lithophilic spawning, and benthic invertivory have been used as criteria for IBIs measuring habitat alteration (Berkman and Rabeni, 1987; Barbour et al., 1999; Simon, 1999).

A framework for the assessment of stream fish vulnerability (i.e., the extent to which a species is likely to be impacted by the cumulative effects of climate change and habitat alteration [Turner et al.,

2003; Schnieder et al., 2007; Glick et al., 2011]) can be developed by associating the effects of climate change and habitat alteration with species or trait-specific impacts. 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., 2011; Poff et al., 2012; Staudinger et al., 2013). A number of vulnerability assessments have been developed 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); Bagne et al., 2011), and the NatureServe Climate Change Vulnerability Index (NSCCVI; Young et al., 2011) could not be applied to Missouri stream fish due to a lack necessary data for evaluating many of the criteria these assessment techniques use. These assessments depend on models of current and future climate conditions which have not yet been developed for stream temperature and flow in Missouri. Additionally, these tools were designed for a wide array of taxa over large spatial scales, and 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 available information is too limited to broadly apply these assessment tools to stream fishes in Missouri. The inability to apply currently available vulnerability assessment tools to determine stream fish vulnerability 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 flow and temperature conditions which deviate from regional baselines; sensitivity of species based on intrinsic factors related to a species environmental tolerance, dispersal ability, genetic adaptation, range, and population size; and habitat resilience or the level of connectivity of habitat which provides opportunities for adaptation via dispersal. Although this framework deviates from the definitions and structure presented by Glick et al. (2011), we believe that it provides a mechanism for assessing stream fish vulnerability based on the information that is currently available, so we adapted it for our use.

Species environmental tolerances, which are often expressed as sensitivity in the vulnerability assessment literature, specifically to habitat alteration, changes in stream temperature, and change 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. Our analysis of species vulnerability will focus on the sensitivity and habitat resilience components of the Poff et al. (2012) framework as adequate information is not yet available to assess exposure in Missouri.

Species tolerance of habitat alteration, changes in 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; Poff and Zimmerman, 2012; Lyons et al., 2010). We developed two separate indices, 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, however the indices used different methods for classifying environmental tolerance. The trait association approach to classifying environmental tolerance is based on traits which have been linked to vulnerability to habitat alteration, changes to flow regimes, and increases in stream temperature in peer-reviewed literature, and the species response approach is based on

species-specific observations of sensitivity to each of the environmental tolerance components.

Another aspect of the sensitivity component 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 by threats (Albanese et al., 2009; Poff et al., 2012). Dispersal, a one-way movement from a site (Lidicker 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 a 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). Therefore, species with narrowly restricted distributions and rare species generally have high levels of vulnerability.

Habitat 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), while a system with high levels of fragmentation, often caused by the creation of barriers such as dams and road crossings (Jackson and Marmulla, 2001; Fagan et al., 2002), can limit the ability of species to cope with threats causing populations to face an elevated risk of extinction (Fagan et al., 2002). Fragmentation caused by dams is particularly problematic for lotic obligate species as the lentic habitat created upstream of dams is not accessible for these species (Gillette et al., 2005).

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 study are to (1) classify species' vulnerability based on environmental tolerance of habitat alteration, changing stream temperatures, and changes to flow regimes, dispersal ability, restricted distribution, rarity, and habitat 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 a large area (the ecologically diverse state of Missouri, USA). Our aim is to develop a framework for assessing stream fish vulnerability at coarse spatial scales and demonstrate its applicability to Missouri's stream fish species.

2. Materials and methods

2.1. Study area

The state of Missouri is over 180,500 km² and is classified into three ecologically unique subregions: the Central Plains, Ozarks, and Mississippi Alluvial Basin (MAB) (Fig. 1; Pflieger, 1970; Pflieger and Missouri Department of Conservation, 1989; Sowa et al., 2007). These subregions are characterized by major differences in geology, landform, soils, land cover, and groundwater influence which create unique habitats and fish communities in the streams of each

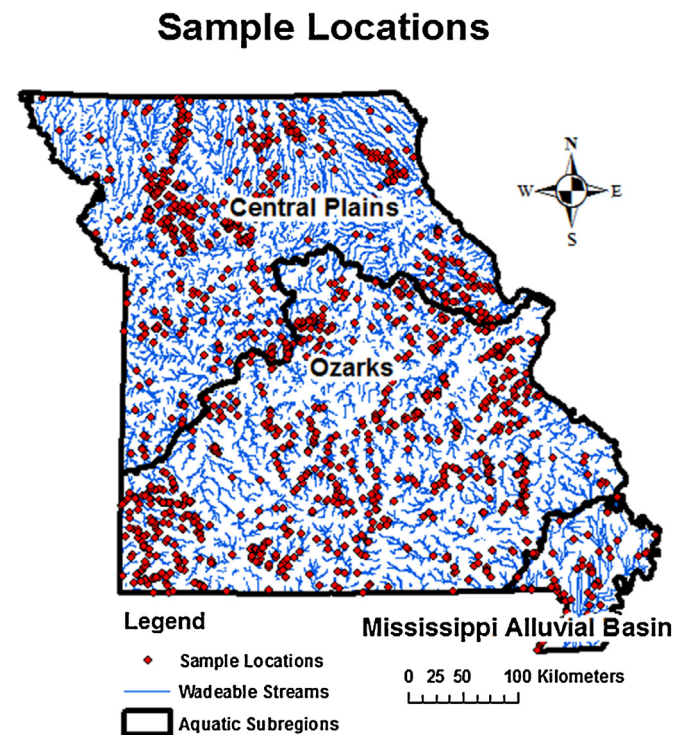


Fig. 1. Map of sample locations used in this study for fishes collected from 2000 to 2011 in wadeable streams in Missouri.

area. The Central Plains, in northern Missouri, is dominated by open grassland and agriculture (78%; Blodgett and Lea, 2005) with streams associated with wide, gently sloping valleys which have a relatively low gradient, high turbidity, and fine silt and sand substrates (Blodgett and Lea, 2005; Sowa et al., 2007). The Ozarks, in southern Missouri (except the southeastern corner), is a mix of forested areas (51.4%) and agriculture (7%; Blodgett and Lea, 2005; Sowa et al., 2007). Most of the region has rugged terrain with high relief and higher stream gradients. Many streams in this region 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, in southeastern Missouri, is a broad flat plain with low stream gradients, most of which have been channelized for agriculture, the predominant land cover in the region (83%; Blodgett and Lea, 2005; Sowa et al., 2007). The substrate varies throughout the subregion but larger and slower moving streams generally have fine silt while streams with faster flow are comprised of sand and small gravel substrates (Sowa et al., 2007). Among subregions, agricultural and forest land cover vary substantially, however the percent urban land use ranges only from 1.8% to 4.1% (Blodgett and Lea, 2005).

2.2. Fish data

Our study used 1499 fish samples which were collected using a statewide stratified random sampling design (Fischer and Combes, 2003) of unique confluence to confluence stream segments based on the 1:100,000 stream network file developed by Sowa et al. (2007). Samples were collected between April and September, 2000–2011 from permanent wadeable streams (2nd–5th Strahler order) as a part of a wadeable stream monitoring program (Resource Assessment and Monitoring Program; RAM) through the Missouri Department of Conservation (Fig. 1). All samples within the RAM database were collected using standardized methods; block nets enclosed a sample reach (length 40 times the mean wetted width, minimum 150 m) which was then thoroughly sampled

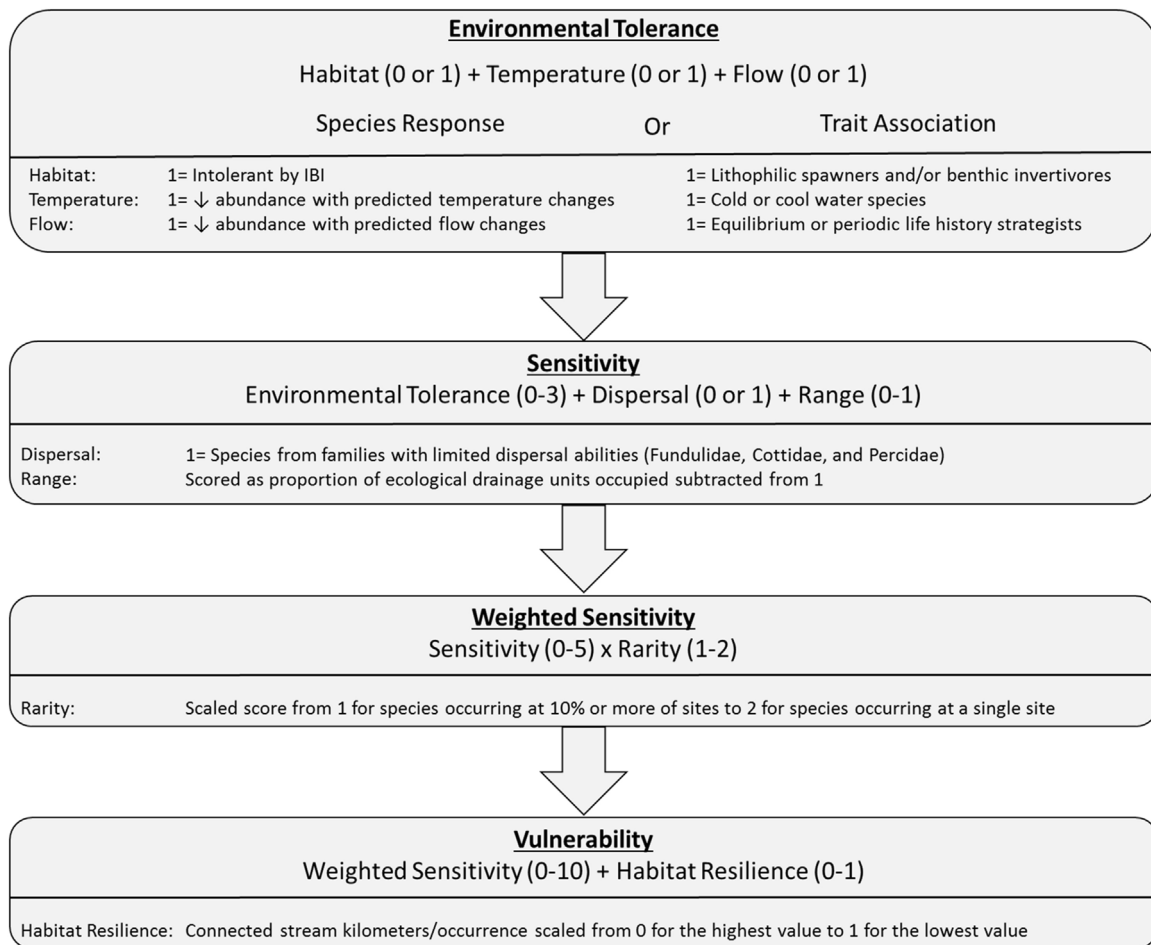


Fig. 2. Conceptual diagram of stepwise procedure for calculating species vulnerability scores using the Species Response Index and Trait Association Index. Inputs for dispersal, range, rarity and habitat resilience were the same for both indices; differences between index calculations occurred only for 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.

via electrofishing and seining (Fischer and Combes, 2003), with all individuals identified to species and enumerated.

2.3. Scoring system and classification of species vulnerability to threats

Species vulnerability scores were calculated using two different approaches: one based on species responses to measured threats and the other based on species traits (Fig. 2). Both indices were classified and scored for species environmental tolerance, dispersal ability, restricted distribution, and rarity as well as the habitat resilience of their habitats. Species environmental tolerance scores were calculated by summing the scores for habitat condition, temperature, and flow vulnerability separately for each of the two indices, while all other criteria were scored the same way for both indices (Fig. 2). 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 habitat resilience scores (Fig. 2). 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 factor 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 (1) or not (0) classifications, continuous-additive (0–1) for vulnerability which was quantified

along a gradient and incorporated into the vulnerability index as an additive component, or continuous-multiplicative (1–2) 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 alteration, change in stream temperatures, and change to flow regimes (Fig. 2). 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 from a review of literature that identified traits linked to vulnerability to each of the threats. Species which exhibit either lithophilic spawning or benthic invertivory were classified as vulnerable to habitat alteration as both have been linked to sensitivity to sedimentation (Berkman and Rabeni, 1987; Mims et al., 2010). Cool- or cold-water adapted species have been identified as vulnerable to warming stream temperatures and were classified as such for our indices (Matthews and Zimmerman, 1990; Eaton and Scheller, 1996; Mohseni et al., 2003; Lyons et al., 2010; Mims et al., 2010). Species with 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 species response index used species specific criteria to identify the environmental tolerance of species to habitat alteration, changes in stream temperature, and change to flow regimes (Fig. 2). Esselman et al. (2011) conducted a literature review of fish based IBIs from across North America and compiled a list of intolerant species. Species classified as intolerant by Esselman et al. (2011) within the World Wildlife Fund ecoregions (Abell et al., 2008) which intersect Missouri were classified as vulnerable to habitat alteration for our species response index. We determined species' associations with extremes in water temperature and stream flow metrics (i.e., representative of changes that could occur with changes in climate) using Indicator Species Analysis (ISA, Dufrière and Legendre, 1997). Fish data used for this determination were acquired from state and federal programs targeting entire stream assemblages in the Temperate Plains (TPL) and Southern Appalachians (SAP) aggregated ecoregions comprising Missouri (Daniel et al., 2015). To characterize water temperature and flow condition, daily stream flow and temperature records were acquired from the USGS Water Data for the Nation (U.S. Geological Survey, 2010) and summarized into 171 flow metrics (using the National Hydrologic Assessment Tool, <https://www.sciencebase.gov/catalog/item/5387735ee4b0aa26cd7b5461>) and 169 temperature metrics (Tsang et al., submitted for publication). Locations of fish samples were associated with stream reaches of the NHD-PlusV1 where temperature and flow data were collected, and thus were presumed to characterize habitat conditions affecting fish species in those stream reaches. To test for associations, ISA calculates indicator values for groupings of sites having high species abundances with extreme values for a particular metric. High indicator values determined for a grouping of sites with high species abundances at extremes in values for a particular metric (and corresponding low indicator values for another grouping for more moderate metric values) support the idea that a species may be associated with extremes in conditions. For this study, we used Threshold Indicator Taxa Analysis (TITAN; Baker and King, 2010) to perform ISA iteratively, testing values of metrics that split species into two groupings across gradients represented by each flow and temperature metric. TITAN calculates indicator value scores for all possible groupings, and then identifies the point along the metric gradient that yields the group of sites with the highest indicator value of all groupings tested (i.e., the change point). Significant indicator values for groups on either side of a change point were used to suggest positive or negative association between species and flow metrics. We identified significant associations between species and extremes in metrics using a p -value of 0.01, purity ≥ 0.95 , and reliability ≥ 0.9 , following Baker and King (2010). Relationships between temperature and flow metrics and current climate metrics (Hostetler et al., 2011) were assessed using Pearson product-moment correlations. Any correlation coefficient with an absolute value greater than 0.5 was considered to show strong associations. We used data from GENMOM, a global climate model which was dynamically downscaled to North America, to determine how climate would change between current and future conditions (five timesteps; 2020–2024, 2030–2034, 2040–2044, 2065–2069, and 2085–2089). The GENMOM GCM was selected because it was the most conservative for late century predictions of temperature and precipitation of the downscaled climate models available for our study region (GENMOM, ECHAM5, and GFDL) and was demonstrated to perform similarly to other climatologies based on PMIP2 and PMIP3 simulations, paleoclimate data-model analyses, and proxy reconstructions (Hostetler et al., 2011; Herb et al., 2014; Alder and Hostetler, 2015). The directionality of the change in climate metrics between current and future

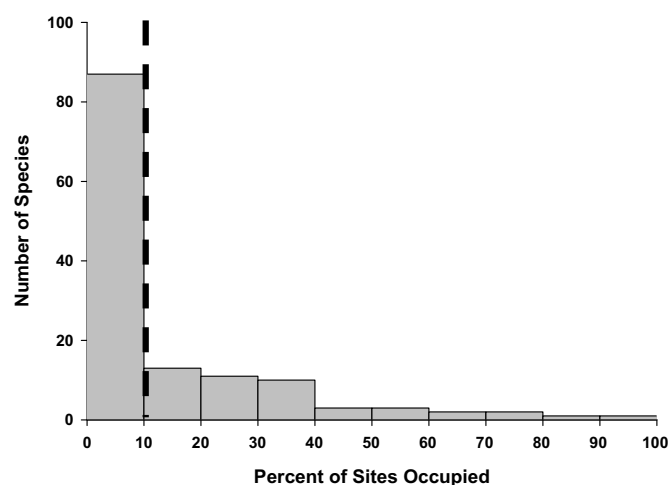


Fig. 3. Histogram of species rarity for fish sampled from 1499 Wadeable stream segments in Missouri from 2000 to 2011. The dashed line represents threshold used to classify rare species.

conditions was used to determine whether a species group (positive or negative association with a flow or temperature metric) would shift toward increased or decreased abundance based on predicted changes in climate. Species which were linked to a decrease in abundance by any temperature or flow metric were classified as vulnerable to predicted changes in temperature or stream flow, respectively.

We classified species with low dispersal ability as vulnerable. Radinger and Wolter (2013) quantified the average dispersal distance for the mobile component (dispersers) for nine fish families. Species from three families, Fundulidae, Cottidae, and Percidae, had limited dispersal abilities, averaging less than 250 m. Species in these families were classified as vulnerable and received scores of 1 (Fig. 2), with the exception of *Sander vitreus*, a Percid which is known to undertake annual spawning migrations of greater than 100 km (Dupont et al., 2006). Species from all other families received scores of 0 for the dispersal ability component. 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 (Fig. 2) and was quantified based on the number of ecological drainage units (EDUs) in which a species has a known occurrence. Ecological drainage units generally follow the boundaries of the major river basins in the state and were developed by identifying contiguous units made up of multiple USGS 8-digit hydrologic units with relatively similar fish assemblages (Sowa et al., 2007). EDUs are nested within subregion boundaries with a total of 5 EDUs in the Plains, 9 in the Ozarks, and 3 in the MAB (Sowa et al., 2007). 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 (e.g. a species found in 25% of EDUs would receive a score of 0.75, a species found in 88% of drainages would receive a score of 0.12). Similarly, species rarity was calculated by using scaled values from 1 to 2. Species were considered common if they occurred at greater than 10% of sites (87 species; Fig. 3). 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). Therefore 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.

Habitat 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 (1787 km/occurrence) to 1 for the minimum value

(123 km/occurrence) (Fig. 2). The amount of connected stream kilometers was calculated by partitioning out 2nd–6th order stream segments of the stream network GIS layer (Blodgett and Lea, 2005). Stream segments classified as anything other than stream or stream/river (i.e., reservoir or impoundment) were removed, which clipped the 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), from which we quantified the average number of connected stream kilometers per occurrence for 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.

2.4. Index comparisons

Analyses were conducted to determine whether the species response and trait association approaches for classifying environmental tolerance achieved similar results, whether listed species received higher vulnerability scores than unlisted species, and whether there were patterns in the distribution of vulnerable species between the ecologically diverse aquatic subregions of Missouri. The percent agreement between the two approaches for classifications of vulnerability to habitat alteration, changes in stream temperatures, and change 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 Odds 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 index. 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 aquatic subregion. All statistical calculations were performed using R statistical software (R Development Core Team, 2011).

3. Results

3.1. Index development

The trait association approach to classifying environmental tolerance was used to calculate vulnerability scores for all 133 species with a mean vulnerability score of 3.83 ± 0.36 95% CI (Appendix A). Environmental tolerance was classified for 101 of the 133 species using the species response approach. This index produced a mean vulnerability score of 2.60 ± 0.30 95% CI (Appendix A). The remaining 32 species were not evaluated because we did not have a sufficient number of occurrences which could be associated with temperature and flow metrics. A total of three species declined in abundance with predicted changes in four temperature metrics (Appendix A), whereas a total of 38 species were shown to have declines in abundance with predicted changes in five flow metrics (Appendix B) and 43 species were identified as being vulnerable to habitat alteration (Appendix C). The trait association approach classified 71, 55, and 54 species vulnerable to habitat alteration, predicted changes in stream temperature, and predicted changes in flow regimes, respectively (Appendix C). For each of the

Table 1

Comparison of trait association and species response approaches for classifying species environmental tolerances based on percent agreement of species classifications, Odds ratio and *p*-value calculated using Fischer's exact test.

| Threat | Percent agreement | Odds ratio | <i>p</i> -Value |
|--------------------|-------------------|------------|-----------------|
| Habitat alteration | 65.3 | 3.60 | 0.003 |
| Temperature | 60.4 | 0.79 | 0.999 |
| Flow | 51.5 | 1.05 | 0.999 |

133 species rarity, dispersal, range, and habitat resilience scores were calculated (Appendix C). The mean rarity score was 1.48 with 65.4% of species classified as rare (i.e., occurred in <10% of sites). Thirty three species (24.8%) 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 habitat resilience was 0.53 which equates to an average of 743 connected kilometers per occurrence.

3.2. 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 alteration (65.3%), followed by changing temperatures (60.4%) and changes to flow regimes (51.5%) (Table 1). If a species was classified as vulnerable to habitat alteration for one index, it was 3.60 times more likely to be classified as vulnerable to habitat alteration with the other index (Odds ratio of 3.60, $p = 0.002$). In contrast, no association was found between the trait and response based approaches to classifying vulnerability based on changing temperatures (Odds ratio of 0.79, $p = 0.999$) or changes to flow regimes (Odds ratio of 1.05; $p = 0.99$) (Table 1).

Mean vulnerability scores differed by both listing status ($p < 0.001$), and index ($p < 0.001$), with no interaction between listing status and index ($p = 0.693$). For each of the indices listing status was significantly related to score ($p < 0.001$) with mean scores for listed species of 1.0 (response) to 1.4 (trait) units greater than unlisted species (Fig. 4). In addition, rare species had higher vulnerability scores for both approaches (Response: Rare = 3.05, Not = 2.10, $p < 0.001$; Trait: Rare = 4.58, Not = 2.47, $p < 0.001$).

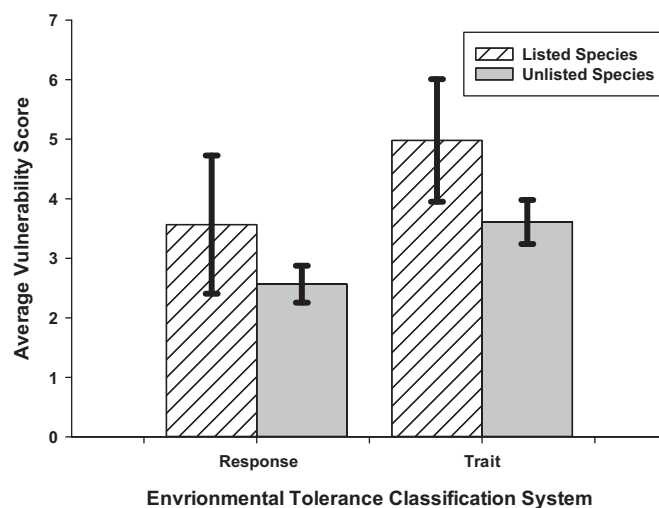


Fig. 4. Mean score ($\pm 95\%$ Confidence Interval) for listed and unlisted species using the species response and traits association approaches for classifying environmental tolerance.

Table 2

Mean number and proportion of vulnerable species at sites within each subregion by index and threat. *p*-Values tested whether the mean number or proportion of species differed by subregion using a one-way ANOVA. “MAB” = Mississippi Alluvial Basin.

| Index | Plains | | Ozarks | | MAB | | <i>p</i> -Value | |
|-------------------|--------|------------|--------|------------|--------|------------|-----------------|------------|
| | Number | Proportion | Number | Proportion | Number | Proportion | Number | Proportion |
| 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 | 0.57 | <0.001 | <0.001 |
| Temperature | 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 |
| Temperature | 1.1 | 0.01 | 1.5 | 0.01 | 0.0 | 0.0 | 0.003 | 0.027 |

The number of vulnerable species at each site was highly correlated between indices for habitat alteration ($r=0.94$), but much less for temperature ($r=0.37$), and flow alteration ($r=0.23$) (Fig. 5). Similarly, the proportion of vulnerable species at each site had the highest correlation between indices for habitat alteration ($r=0.87$), but less so for temperature ($r=0.15$), and flow alteration ($r=-0.25$) (Fig. 5). The mean number of vulnerable species per site was greatest in the forest-dominated Ozark subregion for both habitat alteration and temperature using both approaches of classifying environmental tolerances (Table 2). The greatest numbers of species vulnerable per site for flow alteration occurred in the MAB with the trait association approach, and the Plains with the species response approach (Table 2). 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 2). The greatest proportion of flow vulnerable species per site was highest in the MAB for the trait association, and in the plains for the species response approach (Table 2).

4. Discussion

The results of this assessment provide a framework for assessing stream fish vulnerability to climate change and habitat alteration. Stream communities worldwide are likely to be impacted by climate change and habitat alteration (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). Our assessment provides a new approach for stream fish vulnerability assessment which can be applied and modified for use in other regions. The classification of species as vulnerable based on their environmental tolerances can be refined as new data becomes available. Models of stream temperature and flow will allow exposure to change to be evaluated. Additionally, these indices can be modified to fit the needs of specific management interests.

The trait association approach produced higher vulnerability scores, and exhibited a greater range of scores than the species response approach. Both of these discrepancies between the indices largely stem from the lack of data available to classify the environmental tolerances of rare species using the response-based approach. Of the 32 species for which environmental tolerance could not be classified based on measured species responses, 31 were classified as rare. 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. Although some regions appear to have sufficient information regarding the vulnerability of all species to specific metrics (Moyle et al., 2013), many regions lack the stream temperature and flow data necessary to elicit the responses of species to environmental threats necessary for a broadly applied vulnerability assessment (Deweber et al., 2014). Quantitative assessments of stream fish relationships with potential stressors are needed (Schlosser, 1991) and 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 allowed for a more complete assessment of species vulnerability as traits have been described for most North

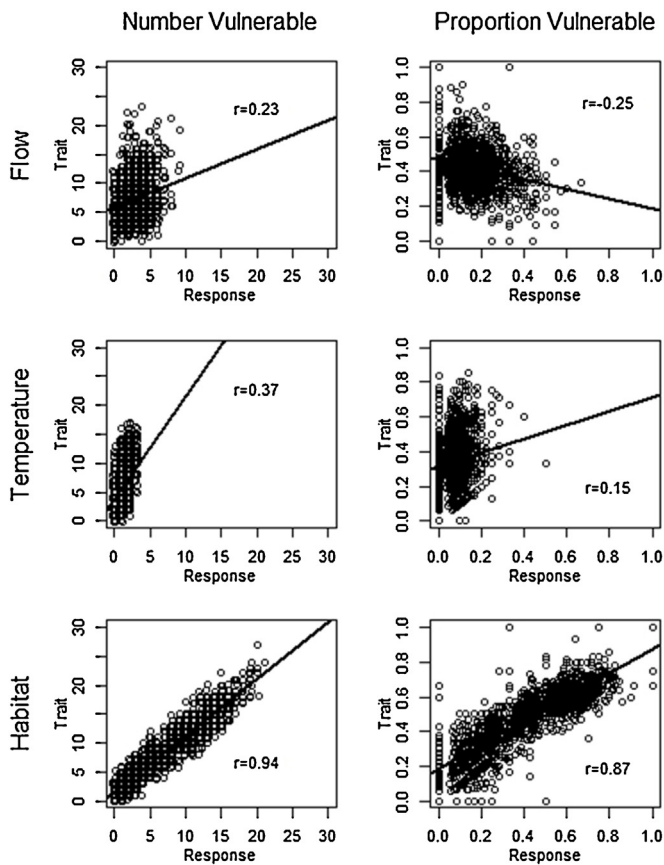


Fig. 5. Correlation between the trait and response based approaches for classifying environmental tolerance based on the number and proportion of vulnerable species at each of the study sites, *r* = Pearson’s correlation coefficient.

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 is a useful tool in stream fish ecology and conservation (Parent and Schriml, 1995; Poff, 1997; Olden and Kennard, 2010; Culp et al., 2011; McManamay et al., 2014). Our results suggest a traits based approach allows for a more complete representation of species; however uncertainty still exists regarding our ability to link traits to species vulnerability. Species have complex relationships with the environment and stressors, and the use of single or even a suite of traits may not provide enough information to fully account for this complexity. More research linking specific traits to environmental threats would help decrease the uncertainty of these assessments and validate trait based 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. For species which were assessed using both indices a similar number of species were classified as vulnerable to habitat alteration (50 versus 43, respectively) and changes to flow regime (47 versus 38), however the number of species classified as vulnerable to changing stream temperatures (39 versus 3) was considerably higher for the trait association approach. The lack of consensus between the use of traits versus measured responses for temperature and flow classifications may stem from the lack of available temperature and flow data available to link species responses to projected changes in climate. As additional stream temperature and flow data become available the relationships between species and these variables will be able to be evaluated more fully. As more data become available we will be able to evaluate a broader range of conditions and also have more replicates across this range. This information will aid in identifying causal, mechanistic explanations for the relationship between stream fish and stream temperature and flow.

Vulnerability to habitat alteration 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. Species classified as vulnerable to habitat alteration with the trait association index but not the measured response index were often those identified as tolerant of alteration but belong to families often viewed as intolerant (e.g. johnny darter (*Etheostoma nigrum*), orangethroat darter (*Etheostoma spectabile*), and golden redbreast (*Moxostoma erythrum*); 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 alteration by the species response index but not by the trait association index included a number of Centrarchids (e.g. warmouth (*Lepomis gulosus*), longear sunfish (*Lepomis megalotis*), smallmouth bass (*Micropterus dolomieu*), 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*)). Although these indices are relatively consistent in classifying vulnerability to habitat alteration, there are still some vulnerable species which were excluded by each approach. The relative consistency between the approaches of classifying habitat vulnerable species suggests that the current understanding of vulnerability to habitat alteration may be sufficient for the purposes of a broad-scale assessment using either approach.

In contrast, vulnerability to changes in stream temperature and changes 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 (*Camptostoma anomalum*)) have shown decreased abundances with future predicted climatic conditions according to our response based index, many additional species may be vulnerable. In studies completed in areas where future stream temperature models exist, species which have yet to exhibit decreased abundance or range contractions are predicted to in the future, and the same is likely for some Missouri species (Mohseni et al., 2003; Heino et al., 2009; Lyons et al., 2010). This could be due to our inability to assess a number of species using the response based approach due to limited data availability and because the current range of observable conditions does not fully reflect the range of future conditions (we are unable to assess future extremes which have not yet occurred). The trait association approach allows decreased abundances and range contractions to be extrapolated even though they may not yet have been observed, and a large number of Cyprinids (14 of 33) and Percids (8 of 15) may be vulnerable to future warming based on this approach, however uncertainty exists as to whether this approach is able to account for the complexities of the impacts of temperature on fish. Species may appear vulnerable based on a certain trait, but may also have other characteristics which allow them to be more or less resilient to increasing temperatures than we can infer based on the simplistic, trait-based approach.

Substantial discrepancies also existed between the two indices 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 our measured species responses. The majority of Ictalurids (10/12), and Centrarchids (15/16) were classified as vulnerable for the trait based approach, however very few of the species in these families were classified as vulnerable based on measured responses (3/10), (3/14), respectively (two species from each of these families were not evaluated based on the species response approach). This 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. 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 may consider rectifying 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 predicted changes in flow regimes and stream temperatures. The use of a traits based approach allows for a broader species assessment, however it is limited in its ability to account for some of the complexity of species/environment interactions. The response based approach provides more direct evidence of species vulnerability, however it is limited in the number of species it can be used to assess based on data availability and limited to the current range of observable environmental conditions, which likely does not fully encompass the range of future conditions.

Species vulnerability scores for both indices were greater for species listed as threatened, endangered, or of conservation concern in Missouri suggesting 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 were determined through expert knowledge while also considering rarity, population trends, and threats following the NatureServe Conservation Status Assessment Criteria (Missouri Natural Heritage Program, 2012). All 23 species of conservation concern were able to be evaluated using the trait

association index, while only five species of conservation concern were with the species response index. Our quantitative assessment of stream fish vulnerability presented here is consistent with expert opinion and provides a general validation of our method. It also suggests that species of conservation concern are likely to become increasingly vulnerable when faced with 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 alteration 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 alteration and changes in stream temperature suggest 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 many sources of uncertainty affect our understanding of stream fish vulnerability in Missouri. Projected changes in climate have substantial uncertainty (Stainforth et al., 2007) so multiple scenarios have been developed which predict a range of future temperature and precipitation conditions (IPCC, 2013). Additional uncertainty surrounds how these predicted changes in climate will impact stream temperature and flow, and currently Missouri does not have any models that make these predictions over a broad spatial scale (Whitledge et al., 2006; Pease and Paukert, 2014; Westhoff and Paukert, 2014). Because of these limitations, 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 our species vulnerability analysis. Additionally, accounting for additional threats which were not considered for in our assessment such as invasive species and pollution may shift the relative vulnerability of some species.

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 responses to threats, and the ability to account for complex interactions between species characteristics, their responses, and the environment (Williams et al., 2008; Lyons et al., 2010). 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, limitation of inference to threat impacts which are observable 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. Additional 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 temperature and flow tolerances, bioenergetics, and population viability coupled with landscape models of stream temperatures and flow regimes would assist in 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 alteration, changes in stream temperature, and changes to the 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 conserve stream fish more effectively, decision makers may need a better understanding of which species are most vulnerable to each of the threats and overall. Our study created a quantifiable index of stream fish vulnerability based on the outline by Poff et al. (2012) for 133 species in a Midwest US state. 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. Our assessment of vulnerability 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 on 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.

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Appendix A. Species response categorization based on temperature metrics for species which showed decreases in abundance with predicted future conditions (species which did not show a decrease in abundance for any of the metrics were not included in the table). Signs indicate direction of future change based on correlation between temperature metrics and climate data. 'X' indicates the metric to which a given species is predicted to show declines in abundance with predicted future changes to temperature metrics.

| Scientific name | Temperate Plains ecoregion | | Southern Appalachian ecoregion | | |
|-------------------------------|---|--|--------------------------------|-------------------------------|----------------------------------|
| | Maximum of 7 day moving average of daily mean (+) | | Spring average daily mean (+) | Summer average daily mean (+) | Average daily mean in winter (+) |
| <i>Campostoma pullum</i> | | | | | X |
| <i>Catostomus commersonii</i> | | | X | | |
| <i>Micropterus dolomieu</i> | | | | | X |

Appendix B. Species response categorization based on flow metrics for species which showed decreases in abundance with predicted future conditions (species which did not show a decrease in abundance for any of the metrics were not included in the table). Signs indicate direction of future change based on correlation between flow metrics and climate data. 'X' indicates species which will show declines in abundance with predicted future changes to flow metrics.

| Scientific name | Temperate Plains ecoregion | | | Southern Appalachian ecoregion | |
|---------------------------------|--|---|--|--|---|
| | Variability in annual flows divided by median annual flows, where variability is calculated as interquartile (+) | High flow duration above 25th median flow (-) | Variability in Julian date of annual minimum (-) | Coefficient of variation in mean monthly flows (+) | Variability in annual maxima of 90-day means of daily discharge (+) |
| <i>Ambloplites rupestris</i> | X | | | X | X |
| <i>Ameiurus melas</i> | | | X | | |
| <i>Aplodinotus grunniens</i> | | X | | | |
| <i>Campostoma pullum</i> | X | | | | |
| <i>Carpiodes carpio</i> | | X | | | |
| <i>Carpiodes cyprinus</i> | | X | | | X |
| <i>Catostomus commersonii</i> | | | X | X | X |
| <i>Cottus bairdii</i> | X | | | | X |
| <i>Cyprinella spiloptera</i> | | | | | X |
| <i>Cyprinella whipplei</i> | X | | | | |
| <i>Etheostoma blennioides</i> | X | | | | X |
| <i>Etheostoma caeruleum</i> | X | | | | |
| <i>Etheostoma nigrum</i> | | X | X | | |
| <i>Etheostoma stigmaeum</i> | | | | | X |
| <i>Etheostoma zonale</i> | | | | | X |
| <i>Hybognathus hankinsoni</i> | | X | X | | |
| <i>Hybopsis amblops</i> | X | | | | |
| <i>Hypentelium nigricans</i> | X | | | | X |
| <i>Ictalurus punctatus</i> | | X | | | |
| <i>Lepomis megalotis</i> | X | | | | |
| <i>Luxilus chrysocephalus</i> | X | | | | X |
| <i>Luxilus cornutus</i> | | X | X | | X |
| <i>Micropterus dolomieu</i> | | | | | X |
| <i>Moxostoma anisurum</i> | | X | | | |
| <i>Moxostoma carinatum</i> | X | | | | |
| <i>Moxostoma duquesnei</i> | X | | | | |
| <i>Moxostoma erythrum</i> | | X | | | X |
| <i>Moxostoma macrolepidotum</i> | | X | | | |
| <i>Notropis atherinoides</i> | | | | | X |
| <i>Notropis dorsalis</i> | | X | X | | |
| <i>Notropis stramineus</i> | | X | | | |
| <i>Notropis volucellus</i> | | | | | X |
| <i>Noturus flavus</i> | | X | | | |
| <i>Percina caprodes</i> | X | | | | X |
| <i>Percina maculata</i> | | X | X | | |
| <i>Pimephales notatus</i> | | | | | X |
| <i>Pimephales promelas</i> | | X | X | | |
| <i>Sander vitreus</i> | | X | | | |

Appendix C. Species scores and classification criteria.

| Scientific name | Response score | Trait score | Response | | | Trait | | | Rarity | Range | Dispersal | Resilience |
|-------------------------------------|----------------|-------------|----------|------|---------|-------|------|---------|--------|-------|-----------|------------|
| | | | Flow | Temp | Habitat | Flow | Temp | Habitat | | | | |
| <i>Ambloplites ariommus</i> | 3.12 | 4.65 | 0 | 0 | 1 | 1 | 1 | 0 | 1.53 | 0.77 | 0 | 0.42 |
| <i>Ambloplites constellatus</i> | 3.96 | 5.56 | 0 | 0 | 1 | 1 | 1 | 0 | 1.6 | 0.94 | 0 | 0.85 |
| <i>Ambloplites rupestris</i> | 3.20 | 2.20 | 1 | 0 | 1 | 1 | 0 | 0 | 1 | 0.59 | 0 | 0.61 |
| <i>Ameiurus melas</i> | 1.69 | 2.69 | 1 | 0 | 0 | 1 | 0 | 1 | 1 | 0.18 | 0 | 0.51 |
| <i>Ameiurus natalis</i> | 0.60 | 1.60 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 0 | 0 | 0.6 |
| <i>Ameiurus nebulosus</i> | 2.87 | 4.86 | 0 | 0 | 0 | 1 | 0 | 0 | 1.99 | 0.94 | 0 | 1 |
| <i>Aphredoderus sayanus</i> | 1.92 | 1.92 | 0 | 0 | 0 | 0 | 0 | 0 | 1.82 | 0.77 | 0 | 0.53 |
| <i>Aplodinotus grunniens</i> | 2.48 | 2.48 | 1 | 0 | 0 | 1 | 0 | 0 | 1.54 | 0.24 | 0 | 0.58 |
| <i>Campostoma oligolepis</i> | 1.01 | 2.01 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0.41 | 0 | 0.6 |
| <i>Campostoma pullum</i> | 2.71 | 0.71 | 1 | 1 | 0 | 0 | 0 | 0 | 1 | 0.12 | 0 | 0.59 |
| <i>Carpiodes carpio</i> | 2.49 | 2.49 | 1 | 0 | 0 | 1 | 0 | 0 | 1.44 | 0.47 | 0 | 0.37 |
| <i>Carpiodes cyprinus</i> | 2.83 | 2.83 | 1 | 0 | 0 | 1 | 0 | 0 | 1.65 | 0.41 | 0 | 0.5 |
| <i>Carpiodes velifer</i> | 4.61 | 4.61 | 0 | 0 | 1 | 1 | 0 | 0 | 1.98 | 0.88 | 0 | 0.88 |
| <i>Catostomus commersonii</i> | 3.79 | 3.79 | 1 | 1 | 1 | 1 | 1 | 1 | 1 | 0.18 | 0 | 0.61 |
| <i>Centrarchus macropterus</i> | | 4.86 | | | | 1 | 0 | 0 | 1.99 | 0.94 | 0 | 1 |
| <i>Cottus bairdii</i> | 3.00 | 5.00 | 1 | 0 | 0 | 1 | 1 | 1 | 1 | 0.47 | 1 | 0.53 |
| <i>Cottus carolinae</i> | 3.04 | 4.04 | 0 | 0 | 1 | 1 | 0 | 1 | 1 | 0.41 | 1 | 0.63 |
| <i>Cottus hypselurus</i> | | 6.22 | | | | 1 | 1 | 1 | 1.22 | 0.65 | 1 | 0.55 |
| <i>Cyprinella camura</i> | 2.47 | 2.47 | 0 | 0 | 0 | 0 | 0 | 0 | 1.99 | 0.94 | 0 | 0.6 |
| <i>Cyprinella galactura</i> | 3.98 | 2.17 | 0 | 0 | 1 | 0 | 0 | 0 | 1.81 | 0.82 | 0 | 0.68 |
| <i>Cyprinella lutrensis</i> | 0.86 | 0.86 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.3 | 0 | 0.56 |
| <i>Cyprinella spiloptera</i> | 5.82 | 1.98 | 1 | 0 | 1 | 0 | 0 | 0 | 1.92 | 0.71 | 0 | 0.62 |
| <i>Cyprinella venusta</i> | 1.67 | 3.42 | 0 | 0 | 0 | 0 | 1 | 0 | 1.75 | 0.65 | 0 | 0.54 |
| <i>Cyprinella whipplei</i> | 5.13 | 1.89 | 1 | 0 | 1 | 0 | 0 | 0 | 1.62 | 0.77 | 0 | 0.65 |
| <i>Dorosoma cepedianum</i> | 0.67 | 2.67 | 0 | 0 | 0 | 1 | 1 | 0 | 1 | 0.06 | 0 | 0.61 |
| <i>Erimystax harryi</i> | | 5.63 | | | | 0 | 1 | 1 | 1.87 | 0.82 | 0 | 0.35 |
| <i>Erimystax x-punctatus</i> | 1.83 | 5.63 | 0 | 0 | 0 | 0 | 1 | 1 | 1.9 | 0.59 | 0 | 0.71 |
| <i>Erimyzon oblongus</i> | 2.85 | 5.63 | 0 | 0 | 1 | 1 | 1 | 1 | 1.39 | 0.65 | 0 | 0.56 |
| <i>Esox americanus vermiculatus</i> | | 4.06 | | | | 1 | 0 | 0 | 1.85 | 0.77 | 0 | 0.79 |
| <i>Esox niger</i> | 3.94 | 3.94 | 0 | 0 | 1 | 1 | 0 | 0 | 1.86 | 0.77 | 0 | 0.66 |
| <i>Etheostoma blennioides</i> | 3.87 | 2.87 | 1 | 0 | 1 | 0 | 0 | 1 | 1 | 0.24 | 1 | 0.63 |
| <i>Etheostoma burri</i> | | 7.87 | | | | 0 | 1 | 1 | 1.76 | 0.94 | 1 | 0.93 |
| <i>Etheostoma caeruleum</i> | 3.99 | 3.99 | 1 | 0 | 1 | 0 | 1 | 1 | 1 | 0.35 | 1 | 0.64 |
| <i>Etheostoma cragini</i> | | 7.82 | | | | 0 | 1 | 1 | 1.82 | 0.94 | 1 | 0.65 |
| <i>Etheostoma euzonum</i> | | 8.11 | | | | 0 | 1 | 1 | 1.99 | 0.94 | 1 | 0.27 |
| <i>Etheostoma flabellare</i> | 2.97 | 3.97 | 0 | 0 | 1 | 0 | 1 | 1 | 1 | 0.3 | 1 | 0.67 |
| <i>Etheostoma gracile</i> | | 7.15 | | | | 0 | 1 | 1 | 1.94 | 0.59 | 1 | 0.19 |
| <i>Etheostoma juliae</i> | | 8.48 | | | | 0 | 1 | 1 | 1.93 | 0.94 | 1 | 0.87 |
| <i>Etheostoma microperca</i> | | 7.71 | | | | 0 | 1 | 1 | 1.97 | 0.82 | 1 | 0.18 |
| <i>Etheostoma nianguae</i> | | 8.75 | | | | 0 | 1 | 1 | 1.99 | 0.94 | 1 | 0.91 |
| <i>Etheostoma nigrum</i> | 2.94 | 2.94 | 1 | 0 | 0 | 0 | 0 | 1 | 1 | 0.35 | 1 | 0.59 |
| <i>Etheostoma proeliare</i> | | 7.34 | | | | 0 | 1 | 1 | 1.95 | 0.77 | 1 | 0 |
| <i>Etheostoma punctulatum</i> | 3.34 | 4.34 | 0 | 0 | 1 | 0 | 1 | 1 | 1 | 0.65 | 1 | 0.69 |
| <i>Etheostoma spectabile</i> | 1.88 | 3.88 | 0 | 0 | 0 | 0 | 1 | 1 | 1 | 0.24 | 1 | 0.64 |
| <i>Etheostoma stigmaeum</i> | 7.82 | 5.84 | 1 | 0 | 1 | 0 | 0 | 1 | 1.98 | 0.71 | 1 | 0.48 |
| <i>Etheostoma tetrazonum</i> | | 6.87 | | | | 0 | 1 | 1 | 1.68 | 0.71 | 1 | 0.64 |
| <i>Etheostoma uniporum</i> | | 7.72 | | | | 0 | 1 | 1 | 1.95 | 0.94 | 1 | 0.03 |
| <i>Etheostoma zonale</i> | 5.00 | 3.74 | 1 | 0 | 1 | 0 | 0 | 1 | 1.26 | 0.47 | 1 | 0.63 |
| <i>Fundulus catenatus</i> | 2.88 | 1.88 | 0 | 0 | 1 | 0 | 0 | 0 | 1 | 0.24 | 1 | 0.64 |
| <i>Fundulus dispar</i> | | 6.71 | | | | 0 | 1 | 0 | 1.98 | 0.88 | 1 | 1 |
| <i>Fundulus notatus</i> | 1.88 | 2.88 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 0.18 | 1 | 0.7 |
| <i>Fundulus olivaceus</i> | 1.93 | 3.93 | 0 | 0 | 0 | 0 | 1 | 1 | 1 | 0.24 | 1 | 0.69 |
| <i>Fundulus sciadicus</i> | | 3.17 | | | | 0 | 0 | 0 | 1.72 | 0.65 | 1 | 0.34 |
| <i>Gambusia affinis</i> | 0.62 | 0.62 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0.62 |
| <i>Hybognathus argyritis</i> | | 1.89 | | | | 0 | 0 | 0 | 1.97 | 0.88 | 0 | 0.15 |
| <i>Hybognathus hankinsoni</i> | 3.47 | 3.47 | 1 | 0 | 0 | 0 | 1 | 0 | 1.78 | 0.88 | 0 | 0.12 |
| <i>Hybognathus placitus</i> | | 1.91 | | | | 0 | 0 | 0 | 1.98 | 0.82 | 0 | 0.28 |
| <i>Hybopsis amblops</i> | 3.65 | 2.56 | 1 | 0 | 1 | 0 | 0 | 1 | 1.09 | 0.65 | 0 | 0.76 |
| <i>Hypentelium nigricans</i> | 3.02 | 4.02 | 1 | 0 | 1 | 1 | 1 | 1 | 1 | 0.35 | 0 | 0.67 |
| <i>Ichthyomyzon fossor</i> | | 6.44 | | | | 0 | 1 | 1 | 1.99 | 0.88 | 0 | 0.7 |
| <i>Ichthyomyzon gagei</i> | | 4.66 | | | | 0 | 0 | 1 | 1.97 | 0.94 | 0 | 0.83 |
| <i>Ictalurus punctatus</i> | 1.63 | 1.63 | 1 | 0 | 0 | 1 | 0 | 0 | 1 | 0.12 | 0 | 0.51 |
| <i>Ictiobus bubalus</i> | 1.15 | 2.94 | 0 | 0 | 0 | 1 | 0 | 0 | 1.79 | 0.35 | 0 | 0.52 |
| <i>Ictiobus cyprinellus</i> | 1.51 | 3.32 | 0 | 0 | 0 | 1 | 0 | 0 | 1.81 | 0.41 | 0 | 0.76 |
| <i>Lampetra aepyptera</i> | 2.07 | 5.91 | 0 | 0 | 0 | 0 | 1 | 1 | 1.92 | 0.88 | 0 | 0.37 |
| <i>Lepisosteus oculatus</i> | | 3.66 | | | | 1 | 0 | 0 | 1.79 | 0.77 | 0 | 0.5 |
| <i>Lepisosteus osseus</i> | 0.82 | 2.22 | 0 | 0 | 0 | 1 | 0 | 0 | 1.4 | 0.18 | 0 | 0.57 |
| <i>Lepisosteus platostomus</i> | 0.96 | 2.46 | 0 | 0 | 0 | 1 | 0 | 0 | 1.5 | 0.3 | 0 | 0.52 |
| <i>Lepomis cyanellus</i> | 0.61 | 1.61 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 0 | 0 | 0.61 |
| <i>Lepomis gulosus</i> | 1.93 | 1.93 | 0 | 0 | 1 | 1 | 0 | 0 | 1.19 | 0.18 | 0 | 0.53 |
| <i>Lepomis humilis</i> | 0.70 | 1.70 | 0 | 0 | 0 | 0 | 0 | 1 | 1 | 0.12 | 0 | 0.58 |
| <i>Lepomis macrochirus</i> | 0.62 | 1.62 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 0 | 0 | 0.62 |
| <i>Lepomis megalotis</i> | 2.68 | 2.68 | 1 | 0 | 1 | 1 | 1 | 0 | 1 | 0 | 0 | 0.68 |
| <i>Lepomis microlophus</i> | 0.91 | 2.21 | 0 | 0 | 0 | 1 | 0 | 0 | 1.3 | 0.18 | 0 | 0.68 |

Appendix C (Continued)

| Scientific name | Response score | Trait score | Response | | | Trait | | | Rarity | Range | Dispersal | Resilience |
|---------------------------------|----------------|-------------|----------|------|---------|-------|------|---------|--------|-------|-----------|------------|
| | | | Flow | Temp | Habitat | Flow | Temp | Habitat | | | | |
| <i>Lepomis miniatus</i> | | 3.29 | | | | 1 | 0 | 0 | 1.63 | 0.71 | 0 | 0.51 |
| <i>Luxilus cardinalis</i> | 3.45 | 4.90 | 0 | 0 | 1 | 0 | 1 | 1 | 1.45 | 0.88 | 0 | 0.72 |
| <i>Luxilus chrysocephalus</i> | 2.89 | 1.89 | 1 | 0 | 1 | 0 | 0 | 1 | 1 | 0.3 | 0 | 0.59 |
| <i>Luxilus cornutus</i> | 2.83 | 2.83 | 1 | 0 | 0 | 0 | 0 | 1 | 1.27 | 0.71 | 0 | 0.66 |
| <i>Luxilus pilsbryi</i> | 3.54 | 4.92 | 0 | 0 | 1 | 0 | 1 | 1 | 1.38 | 0.94 | 0 | 0.86 |
| <i>Luxilus zonatus</i> | | 2.05 | | | | 0 | 0 | 1 | 1 | 0.47 | 0 | 0.58 |
| <i>Lythrurus umbratilis</i> | 0.66 | 0.66 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.18 | 0 | 0.48 |
| <i>Macrhybopsis storeriana</i> | 1.71 | 5.69 | 0 | 0 | 0 | 0 | 1 | 1 | 1.99 | 0.71 | 0 | 0.31 |
| <i>Micropterus dolomieu</i> | 3.99 | 1.99 | 1 | 1 | 1 | 1 | 0 | 0 | 1 | 0.35 | 0 | 0.64 |
| <i>Micropterus punctulatus</i> | 0.86 | 2.16 | 0 | 0 | 0 | 1 | 0 | 0 | 1.3 | 0.12 | 0 | 0.71 |
| <i>Micropterus salmoides</i> | 0.62 | 1.62 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 0 | 0 | 0.62 |
| <i>Minytrema melanops</i> | 2.56 | 4.15 | 0 | 0 | 1 | 1 | 0 | 1 | 1.59 | 0.3 | 0 | 0.5 |
| <i>Morone chrysops</i> | 1.77 | 3.69 | 0 | 0 | 0 | 1 | 0 | 0 | 1.92 | 0.53 | 0 | 0.75 |
| <i>Moxostoma anisurum</i> | 3.83 | 7.55 | 1 | 0 | 0 | 1 | 1 | 1 | 1.86 | 0.71 | 0 | 0.66 |
| <i>Moxostoma carinatum</i> | 6.12 | 6.12 | 1 | 0 | 1 | 1 | 0 | 1 | 1.93 | 0.82 | 0 | 0.67 |
| <i>Moxostoma duquesnei</i> | 3.04 | 3.04 | 1 | 0 | 1 | 1 | 0 | 1 | 1 | 0.35 | 0 | 0.69 |
| <i>Moxostoma erythrurum</i> | 1.96 | 2.96 | 1 | 0 | 0 | 1 | 0 | 1 | 1 | 0.24 | 0 | 0.72 |
| <i>Moxostoma macrolepidotum</i> | 2.30 | 5.26 | 1 | 0 | 0 | 1 | 1 | 1 | 1.48 | 0.12 | 0 | 0.65 |
| <i>Nocomis asper</i> | 3.80 | 6.94 | 0 | 0 | 1 | 1 | 1 | 1 | 1.57 | 0.94 | 0 | 0.75 |
| <i>Nocomis biguttatus</i> | 2.00 | 4.00 | 0 | 0 | 1 | 1 | 1 | 1 | 1 | 0.41 | 0 | 0.59 |
| <i>Notemigonus crysoleucas</i> | 0.70 | 1.70 | 0 | 0 | 0 | 1 | 0 | 0 | 1 | 0.06 | 0 | 0.64 |
| <i>Notropis atherinoides</i> | 2.79 | 2.79 | 1 | 0 | 0 | 0 | 1 | 0 | 1.65 | 0.3 | 0 | 0.65 |
| <i>Notropis boops</i> | 2.03 | 3.03 | 0 | 0 | 1 | 0 | 1 | 1 | 1 | 0.41 | 0 | 0.62 |
| <i>Notropis buccatus</i> | 2.34 | 4.10 | 0 | 0 | 0 | 0 | 0 | 1 | 1.76 | 0.82 | 0 | 0.89 |
| <i>Notropis buechanani</i> | | 3.90 | | | | 0 | 0 | 1 | 1.96 | 0.71 | 0 | 0.56 |
| <i>Notropis dorsalis</i> | 2.05 | 2.05 | 1 | 0 | 0 | 0 | 1 | 0 | 1 | 0.59 | 0 | 0.46 |
| <i>Notropis greeniei</i> | | 3.20 | | | | 0 | 0 | 1 | 1.59 | 0.65 | 0 | 0.58 |
| <i>Notropis heterolepis</i> | | 2.33 | | | | 0 | 0 | 0 | 1.95 | 0.77 | 0 | 0.84 |
| <i>Notropis nubilus</i> | 2.00 | 3.00 | 0 | 0 | 1 | 0 | 1 | 1 | 1 | 0.35 | 0 | 0.65 |
| <i>Notropis ozarcanus</i> | | 5.83 | | | | 0 | 1 | 1 | 1.93 | 0.82 | 0 | 0.38 |
| <i>Notropis stramineus</i> | 2.99 | 0.99 | 1 | 0 | 1 | 0 | 0 | 0 | 1 | 0.41 | 0 | 0.58 |
| <i>Notropis telescopus</i> | 2.50 | 3.61 | 0 | 0 | 1 | 0 | 1 | 1 | 1.11 | 0.77 | 0 | 0.54 |
| <i>Notropis texanus</i> | | 1.87 | | | | 0 | 0 | 0 | 1.86 | 0.82 | 0 | 0.34 |
| <i>Notropis volucellus</i> | 3.10 | 1.46 | 1 | 0 | 0 | 0 | 0 | 0 | 1.64 | 0.47 | 0 | 0.69 |
| <i>Noturus albatris</i> | | 4.68 | | | | 1 | 0 | 1 | 1.43 | 0.82 | 0 | 0.64 |
| <i>Noturus exilis</i> | 1.93 | 3.93 | 0 | 0 | 1 | 1 | 1 | 1 | 1 | 0.24 | 0 | 0.69 |
| <i>Noturus flavater</i> | | 6.16 | | | | 1 | 0 | 1 | 1.98 | 0.88 | 0 | 0.45 |
| <i>Noturus flavus</i> | 4.61 | 6.33 | 1 | 0 | 1 | 1 | 1 | 1 | 1.72 | 0.35 | 0 | 0.56 |
| <i>Noturus gyrinus</i> | 1.57 | 3.30 | 0 | 0 | 0 | 0 | 0 | 1 | 1.73 | 0.59 | 0 | 0.55 |
| <i>Noturus miurus</i> | 2.20 | 4.11 | 0 | 0 | 0 | 0 | 0 | 1 | 1.91 | 0.82 | 0 | 0.63 |
| <i>Noturus nocturnus</i> | 1.40 | 7.34 | 0 | 0 | 0 | 1 | 1 | 1 | 1.98 | 0.59 | 0 | 0.23 |
| <i>Opsopoeodus emiliae</i> | | 1.34 | | | | 0 | 0 | 0 | 1.89 | 0.71 | 0 | 0.01 |
| <i>Percina caprodes</i> | 3.79 | 2.79 | 1 | 0 | 1 | 0 | 0 | 1 | 1 | 0.12 | 1 | 0.67 |
| <i>Percina copelandi</i> | 4.43 | 6.40 | 0 | 0 | 0 | 0 | 0 | 1 | 1.97 | 0.94 | 1 | 0.6 |
| <i>Percina cymatotaenia</i> | | 7.88 | | | | 0 | 1 | 1 | 1.98 | 0.88 | 1 | 0.19 |
| <i>Percina evides</i> | 5.98 | 7.92 | 0 | 0 | 1 | 0 | 1 | 1 | 1.94 | 0.71 | 1 | 0.73 |
| <i>Percina maculata</i> | 6.17 | 6.17 | 1 | 0 | 1 | 0 | 1 | 1 | 1.59 | 0.53 | 1 | 0.56 |
| <i>Percina phoxocephala</i> | 4.76 | 6.39 | 0 | 0 | 1 | 0 | 1 | 1 | 1.63 | 0.53 | 1 | 0.64 |
| <i>Percina sciera</i> | 3.90 | 5.89 | 0 | 0 | 0 | 0 | 0 | 1 | 1.99 | 0.77 | 1 | 0.39 |
| <i>Phenacobius mirabilis</i> | 1.84 | 1.84 | 0 | 0 | 1 | 0 | 0 | 1 | 1 | 0.35 | 0 | 0.49 |
| <i>Phoxinus erythrogaster</i> | 1.95 | 1.95 | 0 | 0 | 1 | 0 | 1 | 0 | 1 | 0.35 | 0 | 0.6 |
| <i>Pimephales notatus</i> | 1.61 | 0.61 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 0 | 0.61 |
| <i>Pimephales promelas</i> | 1.64 | 0.64 | 1 | 0 | 0 | 0 | 0 | 0 | 1 | 0.18 | 0 | 0.46 |
| <i>Pimephales vigilax</i> | 1.66 | 1.66 | 0 | 0 | 0 | 0 | 0 | 0 | 1.72 | 0.59 | 0 | 0.65 |
| <i>Pomoxis annularis</i> | 0.62 | 1.97 | 0 | 0 | 0 | 1 | 0 | 0 | 1.35 | 0.06 | 0 | 0.54 |
| <i>Pomoxis nigromaculatus</i> | 1.17 | 3.04 | 0 | 0 | 0 | 1 | 0 | 0 | 1.87 | 0.24 | 0 | 0.73 |
| <i>Pylodictis olivaris</i> | 0.96 | 2.60 | 0 | 0 | 0 | 1 | 0 | 0 | 1.64 | 0.3 | 0 | 0.48 |
| <i>Sander vitreus</i> | 4.22 | 6.28 | 1 | 0 | 0 | 1 | 1 | 0 | 1.99 | 0.77 | 0 | 0.7 |
| <i>Semotilus atromaculatus</i> | 0.65 | 2.65 | 0 | 0 | 0 | 0 | 1 | 1 | 1 | 0.06 | 0 | 0.59 |

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