ASSESSING THE VULNERABILITY OF STREAM COMMUNITIES AND THE CONSISTENCY

AND USE OF BIOTIC INDICES IN LEAST-DISTURBED STREAMS

_____

A Thesis

presented to

the Faculty of the Graduate School

at the University of Missouri-Columbia

_____

In Partial Fulfillment

of the Requirements for the Degree

Master of Science

_____

by

JACOB M. SCHWOERER

DR. CRAIG PAUKERT, THESIS SUPERVISOR

FEBRUARY 2016

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

thesis entitled:

ASSESSING THE VULNERABILITY OF STREAM COMMUNITIES AND THE CONSISTENCY

AND USE OF BIOTIC INDICES IN LEAST-DISTURBED STREAMS

Presented by Jacob Schwoerer,

A candidate for the Masters of Science,

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

_____

Dr. Craig Paukert (Advisor)

_____

Dr. Jodi Whittier

_____

Dr. Chris Wikle

Acknowledgments

Table of Contents

List of Tables

Chapter One

Chapter Two

List of Figures

Chapter One

Chapter Two

**Assessing the Vulnerability of Stream Communities and the Consistency and Use of
Biotic Indices in Least-Disturbed Streams**

Jacob M. Schwoerer

Craig Paukert, Thesis Advisor

Abstract

The need in freshwater conservation is to understand the current status of aquatic
biota so that we can recognize when degradation or changes occur. Because stream
habitats and communities are dynamic, it is important to understand the natural
variability through time and space so that departures may be used to make inferences on
stability or instability. Additionally, attempting to predict aquatic communities that are
more likely to experience a change in diversity, abundance, and function from
anthropogenic impacts may help to prioritize locations for management action. Finally,
assessing the consistency of various biotic indices (quantitative tools used to convey lotic
ecosystem health) will aid in conveying a more holistic depiction of stream condition and
to prioritize locations and biota for management action. To address each of the
aforementioned data gaps, we used fish and aquatic invertebrate community data
collected from 1988 to 2013 from 88 sites within seven National Park Service (NPS)
units represented within the Heartland Inventory and Monitoring Network. The fish
community (Index of Biotic Integrity) at each of the seven NPS units was less temporally
variable than spatially variable. This relationship was not found with aquatic invertebrate

community (Hilsenhoff Biotic Index) in that only three of the seven NPS units were less temporally variable than spatially variable. Aquatic invertebrate communities at each NPS unit were most vulnerable to an altered flow regime (mean among parks: 81% ± 6% of the community vulnerable) while the fish community was most vulnerable to in-stream physical habitat alteration (mean among parks: 53% ± 15% of the community vulnerable). Generally, relationships among biotic indices were highly variable (ρ = -0.02 to 0.87) and uncorrelated (12 of 15 pairwise comparisons) and biotic indices that were correlated differed by river system. Within faunal group indices were more related within the Buffalo National River (ρ = -0.28 to 0.87) while richness indices were more related within the Ozark National Scenic Riverways (ρ = 0.60 to 0.81). Implementing these components into monitoring programs may lead to a more thorough understanding of lotic ecosystems, their aquatic biota, and their integrity status now and into the future.

General Introduction

Lotic ecosystems are dynamic features within the landscape and are variable in water chemistry, water temperature, discharge, physical habitat, and biota through time and space (Matthews 1988; Poff et al. 1997; Jackson et al. 2001; Dodds et al. 2004). The variability of biotic communities found within a lotic ecosystem is a product of variability in stream physio-chemical habitat (Horwitz 1978; Schlosser 1982; Ross et al. 1985; Poff and Allan 1995) and species have adapted to thrive in such areas by having life histories that mimic this natural variability (Poff and Allan 1995). However, even locations with stable stream habitats can experience species disappearance during brief unfavorable conditions, but which later recolonize when conditions improve (Matthews 1988; Dodds et al. 2004). The degree of natural variability in stream habitat (hydrology, temperature, habitat volume, dissolved oxygen, etc.) often determines the fish community structure at a site, with highly variable stream habitats consisting of resource generalist species, while stable and less variable stream habitats being characterized by a higher proportion of species with specialized life histories (Poff and Allan 1995). Understanding the natural variability in stream habitats and assemblages is important for developing a baseline measure used to assess long term stability of stream biota.

Lotic communities have the ability to overcome a shifting habitat mosaic and natural disturbances, however, long term or permanent shifts in regional climates (Eaton and Scheller 1996; Lyons et al. 2010) and human expansion (Schlosser 1991; Allan 2004; Dudgeon et al. 2005; Scott 2006) beyond the adaptive capacity of stream organisms have created large losses in biodiversity and an altered community structure. Streams recently

influenced by changes in water temperature, precipitation patterns, and the conversion of natural lands that cause broad habitat degradation have altered streams from historic conditions. Large-scale biodiversity loss has resulted from these disruptions in the equilibrium of lotic ecosystems and is the reason freshwater biomes are among the most imperiled on the planet (Ricciardi and Rasmussen 1999; Abell 2002; Dudgeon et al. 2005).

Habitat degradation is among the greatest threat to native biota in lotic ecosystems (Fausch et al. 2002; Scott 2006) and thus areas with minimal degradation are useful to identify the potential biological integrity in a region (Angermeier and Karr 1986; Hughes et al. 1998; Lawrence et al 2011). Removal or alteration of physical habitat alters a streams hydrograph, channel morphology, reduces water quality, and increases the likelihood of flooding (Prestegaard 1988; Scott 2006). For example, Removal of riparian vegetation causes bank destabilization, a change in channel morphology, and an increased sedimentation and nutrient loads which alter physical and chemical habitat. Barriers to movement, such as dams, impoundments, and road crossings negatively affect many stream organisms attempting to immigrate or migrate to fulfill life history requirements. In addition, these barriers create changes in the natural flow regime within a system; harming organisms adapted to survive in such streams (Poff et al. 1997). Biodiversity decreases when anthropogenic stream modifications occur because degradation often reduces habitat heterogeneity that native organisms need to persist through time. These habitat degradations impact the biota by altering trophic interactions which create a cascading effect throughout the ecosystem (Matthews 1988; Meyer et al. 1999). Although the streams used in this research are protected within National Park

Service (NPS) boundaries, each parks ability to protect the stream varies depending on the amount of upstream watershed area that lies outside park boundaries.

Urban land use plays a disproportionate role in affecting aquatic resources (Allan 2004). One of the greatest alterations in urban settings is the amount of impermeable surfaces, creating a lack of water infiltration into subsurface areas and a decrease base flow (Poff et al. 1997; Allan 2004; Stanfield and Kilgour 2006). This also lessens the time it takes for precipitation to enter a stream channel, thus increasing the magnitude in the hydrograph, which alters channel morphology, increases water temperature, and reduces water quality (Scott 2006; Standfield and Kilgour 2006). Areas with dense human populations contribute large amounts of pollution to lotic ecosystems (Allan 2004) and have the potential to remove sensitive species and alter the biotic integrity (Kolpin et al. 2002; Allan 2004). With the world population continuing to grow, these effects will likely be exacerbated in the future. Understanding what regions or streams are most vulnerable to urbanization is important for prioritization of management action.

With higher human population growth, there is a greater dependency on agricultural production to meet consumptive needs. Streams with a high proportion of agriculture in its watershed experience increases in overland flow, sedimentation, and agricultural fertilizers and pesticides which cause eutrophication and an array of physiological responses by aquatic biota (Schlosser 1991; Poff et al. 1997; Dodds et al. 2004; Allan 2004). Hydrologic disruptions in agricultural watersheds are then exacerbated with the installation of tile drains, which increase the speed of precipitation towards the channel and create flashy systems that are characterized by a shorter lag time and a higher magnitude in the hydrograph (Matthews 1988; Allan et al. 1997). This tends

3

to entrench stream channels and alter the biotic community (Brookes 1988). Irrigation disrupts natural stream hydrology by drawing down the water table, causing streams to become lower or even dry during parts of the year, which fragments the channel (Weeks and Stangland 1971; Perkin et al. 2015). Whether natural landscapes get converted to an urban or agricultural setting, changing the natural dynamics of a stream creates changes in habitat and biodiversity, and thus identifying where these losses may be greatest aid resource management agencies to protect and restore threatened resources

Climate change is among the leading threats facing ecosystems worldwide, and is of particular concern for lotic ecosystems (Eaton and Scheller 1996; Mohseni et al. 2003; Xenopoulos et al. 2005; Lyons et al. 2010). Air temperatures are projected to increase 1-7°C depending on the global circulation model (Ficke et al. 2007), which will correspond to increases in stream water temperatures (Erickson and Stefan 1996; Mohseni and Stefan 1999) and redefine thermal habitats for aquatic organisms (Eaton and Scheller 1996; Rahel 2002; Mohseni et al. 2003; Lyons 2010). While approximately 50 percent of available cold- and cool-water fish habitats in the US are expected to disappear under temperature trends associated with doubling of atmospheric CO_2 (Eaton and Scheller 1996), warm-water fish thermal habitat is expected to increase by 31 percent (Mohseni et al. 2003). Yearly precipitation totals are predicted to increase, decrease and/or experience a change in timing, severity or duration of events, depending on geographic location (Meyer et al. 1999; Ficke et al. 2006). Alterations in the timing of key temperatures and discharges have the potential to send mistimed cues to aquatic organisms such as mistimed spawning, hatching, migration or other life history characteristics (Welcomme 1979; Nesler et al. 1988; Fausch and Bestgen 1997; Meyer et al. 1999). Ultimately, these

changes in thermal habitat and flow regime due to climate change will be an important factor associated with major changes in species biogeography and persistence. Therefore, resource managers may need to consider how climate change may affect aquatic biota in their systems, and what systems may be more at risk to the impacts of climate change.

Biotic communities are a reflection of the environment in which they live in, where high quality, functioning, and diverse communities are a reflection of high quality and functioning ecosystems (Karr 1981). Biotic indices are quantitative tools that allow managers to assess the condition of the stream ecosystem using biota found within them (Karr et al. 1986; Hilsenhoff 1987; Karr 1991). Indices can be single- or multi-metric with common metrics including species richness, species abundances, trophic composition, sensitivity or tolerance values, and individual condition, amongst others. They are developed to be taxa specific, with fish (Karr 1981; Lyons 1992; Lyons et al. 1996; Dauwalter 2003) and aquatic invertebrates (Hilsenhoff 1987; Lenat 1988; 1993) being the most common aquatic faunal groups used. Through the use of various fish and aquatic invertebrate community biotic indices, an overall status of ecosystem health is often inferred; however, these indices rely heavily on interpretations because they are often used in various systems over a wide geographic area. Because these indices are used to describe broad lotic ecosystem health, consistency of interpretations between any two indices should be similar; even between fish and aquatic invertebrate biotic indices. However, there has been limited research looking into the consistency of multiple aquatic biotic indices to portray ecosystem health (Lammert and Allan 1999; Ogren and Huckins 2014; 2015). Because decisions about management are often based on biotic index interpretations (USEPA 2011), it places emphasis on understanding the relationship

among health interpretations produced by various indices using the same and different faunal groups.

Being able to determine where a stream lies on the continuum of stability allows managers to better prioritize areas of conservation need. Therefore, I will develop baseline stream community (fish and aquatic invertebrate) variability across time and space so that departures from this can be used to assert whether a community is within the bounds of stability in future settings. Understanding the degree in which a stream community naturally varies allows increased variability to infer degrees of instability. Because it is also important to understand how much a community is likely to change with various types of degradation, I will assess stream community (fish and aquatic invertebrate) vulnerability to anthropogenic impacts and attributes linked to stream community vulnerability. This information can then be used to prioritize not only what streams are more or less vulnerable, but to determine what type of disturbance will impact the community the greatest. Since both fish and aquatic invertebrates are commonly used to assess stream health, I will determine how consistent stream health interpretations produced from multiple biotic indices are within and across faunal groups. In addition, I will determine how these biotic indices are related to ecological gradients at various scales in a least-disturbed setting which will allow considerations to be taken into account when interpreting index scores. This research will give managers a utility to understand the current condition of stream environments and to more holistically determine aquatic resource condition by understanding what communities are more vulnerable to degradation and by knowing what set of biotic indices to use depending on the objectives of the monitoring effort.

Literature Cited

Allan, J. D., D. L. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. Freshwater Biology 37: 149-161.

Allan, J. D., 2004. Landscapes to riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology, Evolution, and Systematics 35: 257-284.

Angermeier, P. L., and J. R. Karr. 1986. Applying and index of biotic integrity based on stream-fish communities: considerations in sampling and interpretation. North American Journal of Fisheries Management 6: 418-429.

Brooks, A., 1988. Channelized rivers, perspectives for environmental management. New York: John Wiley and Sons.

Dauwalter, D. C., E. J. Pert, and W. E. Keith. 2003. An index of biotic integrity for fish assemblages in Ozark Highland Streams of Arkansas. Southeastern Naturalist 2: 447-468.

Dodds, W. K., K. Gido, M. R. Whiles, K. M. Fritz, and W. J. Matthews. 2004. Life on the edge: the ecology of Great Plains prairie streams. Bioscience 54: 205-216.

Dudgeon, D., A. H. Arthington, M. O. Gessner, Z. Kawabata, D. J. Knowler, C. Leveque, R. J. Naiman, A. Prieur-Richard, D. Soto, M. L. J. Stiassny, and C. A. Sullivan. 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. Freshwater Biodiversity 81: 163-182.

Eaton, J. G., and R. M. Scheller. 1996. Effects of climate warming on fish thermal habitat in streams of the United States. Limnology and Oceanography 41: 1109-1115.

Erickson, T. R., and H. G. Stefan. 1996. Correlations of Oklahoma stream temperatures with air temperatures. University of Minnesota St. Anthony Falls Laboratory Project Report No. 398.

Fausch, K. D. and K. R. Bestgen. 1997. Ecology of fishes indigenous to the central and southwestern Great Plains. Ecology and Conservation of Great Plains Vertebrates 125: 131-166.

Fausch, K. D., C. E. Torgersen, C. V. Baxter, and H. W. Li. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. Bioscience 52: 483-498.

Ficke, A., C. Myrick, and L. Hansen. 2007. Potential impacts of global climate change on freshwater fisheries. Reviews in Fish Biology and Fisheries 17: 581-613.

Hilsenhoff, W. L., 1987. An improved biotic index of organic stream pollution. Great Lakes Entomologist 20: 31-39.

Horwitz, R. J., 1978. Temporal variability patterns and the distributional patterns of stream fishes. Ecological Monographs 48: 307-321.

Hughes, R. M., P. R. Kaufmann, A. T. Herlihy, T. M. Kincaid, L. Reynolds, and D. P. Larsen. 1998. A process for developing and evaluating indices of fish assemblage integrity. Canadian Journal of Fisheries and Aquatic Sciences 55: 1618-1631.

Jackson, D. A., P. R. Peres-Neto, and J. D. Olden. 2001. What controls who is where in freshwater fish communities – the roles of biotic, abiotic, and spatial factors. Canadian Journal of Fisheries and Aquatic Sciences 58: 157-170.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6: 21-27.

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

Karr, J. R. 1991. Biological integrity: a long-neglected aspect of water resource management. Ecological Applications 1: 66-84.

Kolpin, D. W., E. T. Furlong, M. T. Meyer, E. M. Thurman, S. D. Zaugg, L. B. Barber, H. T. Buxton. 2002. Pharmaceuticals, hormones, and other organic wastewater contaminants in U.S. streams, 1999-2000: a national reconnaissance. Environmental Science and Technology 36: 1201-1211.

Lammert, M., and J. D. Allan. 1999. Assessing biotic integrity of streams: effects of scale in measuring the influence of land use/cover and habitat structure on fish and macroinvertebrates. Environmental Management 23: 257-270.

Lawrence, D. J., E. R. Larson, C. A. R. Liermann, M. C. Mims, T. K. Pool, and J. D. Olden. 2011. National parks as protected areas for U.S. freshwater fish diversity. Conservation Letters 4: 364-371.

Lenat, D. R. 1988. Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. Journal of the North American Benthological Society 7: 222-233.

Lenat, D. R. 1993. A biotic index for the southeastern United States: derivation and list of tolerance values, with criteria for assigning water-quality ratings. Journal of the North American Benthological Society. 12: 279-290.

Lyons, John 1992. Using the index of biotic integrity (IBI) to measure environmental quality in warmwater streams of Wisconsin. General Technical Report NC-149. St. Paul, MN: U.S. Dept. of Agriculture, Forest Service, North Central Forest Experiment Station

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

Lyons, J., J. S. 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: 1867-1898.

Matthews, W. J., 1988. North American prairie streams as systems for ecological study. Journal of the North American Benthological Society 7: 387-409.

Meyer, J. L., M. J. Sale, P. J. Mulholland, and N. L. Poff. 1999. Impacts of climate change on aquatic ecosystem functioning and health. Journal of the American Water Resources Association 35: 1373-1386.

Mohseni, O., and H. G. Stefan. 1999. Stream temperature/air temperature relationship: a physical interpretation. Journal of Hydrology 218: 128-141.

Mohseni, O., H. G. Stefan, and J. G. Eaton. 2003. Global warming and potential changes in fish habitat in U.S. streams. Climate Change 59: 389-409.

Nesler, T. P., R. T. Muth, and A. F. Wasowicz. 1988. Evidence for baseline flow spikes as spawning cues for Colorado squawfish in the Yampa River, Colorado. Transactions of the American Fisheries Society Symposium 5: 68-79.

Ogren, S. A., and C. J. Huckins. 2014. Evaluation of suitability and comparability of stream assessment indices using macroinvertebrate data sets from the Northern Lakes and Forests Ecoregion. Ecological Indicators 40: 117-126.

Ogren, S. A., and C. J. Huckins. 2015. Applicability and interpretation of fish indices of biotic integrity (ibi) for bioassessment in the upper Midwest. North American Journal of Fisheries Management 35: 281-295.

Perkin, J. S., K. B. Gido, A. R. Cooper, T. F. Turner, M. J. Osborne, E. R. Johnson, and K. B Mayes. 2015. Fragmentation and dewatering transform Great Plains stream fish communities. Ecological Monographs 85: 73-92.

Poff, N. L. and J. D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology 76: 606-627.

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: 769-784.

Prestegaard, K. L., 1988. Morphological controls on sediment delivery pathways. Pages 533-540 in Walling DE, ed. Sediment budgets. Wallingford (UK):IAHS Press. International Association of Hydrological Sciences Publication nr 174.

Rahel, F. J., 2002. Using current biogeographic limits to predict fish distributions following climate change. American Fisheries Society Symposium 32: 99-110.

Ricciardi, A., and J. B. Rasmussen. 1999. Extinction rates of North American freshwater fauna. Conservation Biology 13: 1220-1222.

Ross, S. T., W. J. Matthews, and A.A. Echelle. 1985. Persistence of stream fish assemblages: effects of environmental change. The American Naturalist 126: 24-40.

Schlosser, I. J., 1982. Fish community structure and function along two habitat gradients in a headwater stream. Ecological Monographs 52: 395-414.

Schlosser, I. J., 1991. Stream ecology: a landscape perspective. Bioscience 41: 704-712.

Scott, M. C., 2006.Winners and losers among stream fishes in relation to land use legacies and urban development in southeastern US. Biological Conservation 127: 301-309.

Stanfield, L. W., and B. W. Kilgour. 2006. Effects of percent impervious cover on fish and benthos assemblages and in-stream habitats in Lake Ontario tributaries. American Fisheries Society Symposium 48: 577-599.

USEPA (US Environmental Protection Agency). 2011. A Primer on Using Biological Assessments to Support Water Quality Management. EPA-810-R-11-01. Office of Science and Technology, Office of Water, Washington, DC.

Weeks, E. P., and H. G. Stangland. 1971. Effects of irrigation on streamflow in the central sand plain of Wisconsin. Wisconsin Department of Natural Resources and the Wisconsin Geological and Natural history Survey.

Welcomme, R. L., 1979. Fisheries ecology of floodplain rivers. Longman Inc., New York. 317p.

Xenopoulos, M. A., D. M. Lodge, J. Alcamo, M. Marker, K. Schulze, and D. P. Van Vuuren. 2005. Scenarios of freshwater fish extinctions from climate change and water withdrawal. Global Climate Biology 11: 1557-1564.

<div align="center">**Chapter One**</div>

<div align="center">**Assessing the variability of stream communities and their vulnerability to land use and climate change**</div>

<div align="center">Jacob M. Schwoerer, Craig P. Paukert, and Hope R. Dodd</div>

Abstract

Stream habitats and communities are dynamic; therefore it is important to understand the natural variability through time, so that departures may be used to make inferences of instability. In addition, climate and land use change are major contributors to stream degradation, but the extent that these impacts have on various regions and streams likely differ. Determining a stream's susceptibility to change from anthropogenic impacts may be needed to determine the stream communities most likely to experience a decrease in diversity, abundance, and function in the future. Our goal was to determine the natural spatial and temporal variability and vulnerability of fish and aquatic invertebrate communities using data collected from 1988 to 2013 at 88 sites throughout seven National Park Service (NPS) units in the central United States. We used a trait-based vulnerability assessment for fish and aquatic invertebrates that allowed us to determine overall community vulnerability, and factors driving a stream's vulnerability (temperature, flow, habitat degradation, dispersal ability, or species persistence through time). The fish community at each of the seven NPS units was less temporally variable than spatially variable, but there was no consistent relationship found with aquatic invertebrates across park units. Aquatic invertebrate communities at each NPS unit were most vulnerable to an altered flow regime (mean among parks: 81% ± 6% of the community vulnerable) while the fish community was most vulnerable to in-stream physical habitat alteration (mean among parks: 53% ± 15% of the community

vulnerable). The most and least vulnerable park was consistent between fish and aquatic invertebrate assessments. Homestead National Monument, a small, low species-rich, prairie stream with fine substrate was the least vulnerable, and George Washington Carver National Monument, a small, low species-rich, coarse substrate Ozark stream was most vulnerable. Our results provide a framework that can be used to determine aquatic biota vulnerability throughout Midwestern streams.

Introduction

Streams exhibit natural variability in habitat and biota through time and space even in a natural functioning and relatively stable community (Horwitz 1978; Schlosser 1982; Ross et al. 1985; Poff and Ward 1989; Poff and Allan 1995). However, additive degrees of variability (instability) are often linked to anthropogenic and landscape-level disturbances (Schlosser 1991; Poff et al.1997). Changes in land use and climate can cause aquatic ecosystem degradation, with the level of degradation depending on the extent of change and the vulnerability of the stream itself to such changes (Allan et al. 1997; Meyer et al. 1999). For aquatic resource managers, understanding a streams vulnerability or risk to such anthropogenic changes would allow them to better prioritize, manage, and conserve threatened resources.

Identifying the current health of stream communities is vital to more effectively plan for future changes. Ideally, identifying a least-disturbed stream may be needed to make relative conclusions about the health of other streams, and allows for the investigation of factors attributed to its change (Harding et al. 1998; Foster et al. 2003; Gido et al 2013). However, due to the natural variability of stream biota from seasonal or climatic variability, it is difficult to determine the health status of a stream without using

12

multiple biotic assessments through time (Poff and Allan 1995; Gido et al 2013). Likewise, due to spatial heterogeneity in habitat, stream community structure is variable across space and therefore characterizing stream condition by only sampling a single stream reach may be problematic. Previous studies have concluded that even in stable communities, there are fluctuations in abundance and presence of fish species and aquatic invertebrate taxa among sites, seasons, and years (Matthews 1986; Freeman et al. 1988; McElravy et al. 1989; Boulton et al. 1992; Hansen and Ramm 1994). Therefore, knowing whether a streams biotic community is within the natural bounds of variability is crucial to understanding its current health and to assess the impacts from disturbance or management actions (McElravy et al. 1989).

A streams vulnerability to land use and climate change is unique to its catchment and is dependent in part by the biotic community that occupies the area under a natural disturbance regime. Although increases in agricultural or urban land use are often directly attributed to changes in stream health (Allan 2004; Diana et al. 2006; Moerke and Lamberti 2006), the extent of stream degradation depends on the extent of land use change and catchment characteristics. Areas with increased slopes, impermeable soils, or erodible soils may increase the degree of stream degradation, which may make a stream community more vulnerable to climactic or landscape level changes (Nerbonne and Vondracek 2001). Resource managers often use changes in taxa to assess the impacts that land use, climate change, and other anthropogenic impacts have on an ecosystem and to gauge vulnerability (Nerbonne and Vondracek 2001; Schweizer and Matlack 2005; Gido et al. 2013). However, the magnitude of system vulnerability to land use or climate change in the future depends on the baseline conditions of the stream. A community

comprised of tolerant species that persist under a range of physical and chemical habitat conditions may not be vulnerable when high degrees of habitat degradation occur, while a community comprised of sensitive or intolerant species that need specific physical and chemical habitat conditions may be vulnerable when limited habitat degradation occurs (Poff and Allan 1995). Therefore, identifying the baseline biotic community present in a system may help identify how vulnerable the system is to change.

Vulnerability assessments are tools used to depict a representative risk value for a faunal group, species, or habitat. To be effective, these biotic vulnerability assessments link specific life history traits to increased risk of extirpation or extinction (O'Grady et al. 2004; Olden et al. 2006; Olden et al. 2007; Lyons et al. 2010) so an accurate portrayal of risk may be calculated and provide managers with a starting point from which conservation efforts may be directed. The risk of extirpation or extinction may be linked to anthropogenic alterations to the in-stream habitat, water temperature, stream flows, and landscape-level stream fragmentation. Anthropogenic habitat degradation may lead to changes in substrate composition, channel shape, water depth, and current velocity which may limit the ability of a species to occupy a given stream reach (Poff and Allan 1995; Poff et al. 1997). Increases in water temperature often redefine thermal habitats and displace cold and cool water species (Eaton and Scheller 1996; Rahel 2002; Mohseni et al. 2003; Lyons 2010) whereas changes in the timing, severity, and amount of annual precipitation (Meyer et al. 1999; Ficke et al. 2007) may disrupt natural flow regimes and alter a streams water quality, energy sources, physical habitat, and biotic interactions (Poff et al. 1997). In particular, changes in the timing of key temperatures and discharges may send mistimed cues to aquatic organisms, triggering spawning, hatching, migration,

or other life history characteristics to become out of place (Welcomme 1979; Nesler et al. 1988; Fausch and Bestgen 1997; Meyer et al. 1999). The 75,000 large dams and countless small dams in the continental Unites States have created a fragmented riverscape that alter flow and thermal regimes and act as barriers to movement, which may affect many stream organisms attempting to immigrate or migrate to fulfill life history requirements (Graf 1999). While some fish and aquatic invertebrates may persist under each of these severe changes, others may become impacted if any number of these changes were to occur. Therefore, resource managers need to know what threats would have the greatest impacts on the community in order to prioritize protection.

The objectives of this study were to determine the spatial and temporal variability of fish and aquatic invertebrate communities within National Park Service (NPS) units in the central United States so that a baseline measure of variability may be used to assess community stability into the future. We were not intending to evaluate biotic indices across a range of disturbances gradients, but rather to determine the natural variability in space and time. In addition, we sought to determine climactic, landscape, and in-stream habitat metrics associated with variability to predict how future conditions may influence community stability. We then developed a community based vulnerability assessment for fish and aquatic invertebrates and determined whether geographic location, in-stream habitat, or climate variables were linked to species vulnerability to inform managers as to streams and biota that may be most sensitive to environmental change. We hypothesized that fish will be more variable across sites due to habitat heterogeneity (fish were sampled along a length of stream while aquatic invertebrates were sampled in only riffle habitats) while aquatic invertebrates will be more variable over time (Freeman et al.

1988; McElravy et al. 1989; Boulton et al. 1992). In addition, aquatic communities in smaller streams were hypothesized to be more temporally variable since smaller or headwater streams are associated with temporal shifts in channel morphology and resource availability (Schlosser 1982) while larger streams will be more spatially variable since there is likely a different arrangement of physical habitat among sites leading to a difference in community structure. We also hypothesize that streams within the Great Plains ecoregion will receive the lowest community vulnerability scores since these systems are commonly considered harsh, and thus comprised primarily of generalist species that can tolerate a wide range of environmental conditions (Poff and Ward 1989; Bramblett and Fausch 1991; Franssen et al. 2006).

Methods

Study Area

We used data from the NPS Heartland Inventory and Monitoring (I&M) Network, which encompasses the Midwest region from Kansas to Ohio and from Minnesota to Arkansas, USA. The purpose of these NPS I&M networks is to facilitate baseline inventories and to measure resource condition over time within the National Parks and to convey this information to stakeholders (Fancy and Bennetts 2012). The streams and sites used in this study were within seven parks in five states within the Heartland I&M Network: Pipestone National Monument in Minnesota (PIPE), Homestead National Monument of America in Nebraska (HOME), Tallgrass Prairie National Preserve in Kansas (TAPR), George Washington Carver National Monument in Missouri (GWCA), Wilsons Creek National Battlefield in Missouri (WICR), Ozark National Scenic Riverways in Missouri (OZAR), and Buffalo National River in Arkansas (BUFF) (Figure

1). Sampling reaches at PIPE, TAPR, and HOME were selected based on a subset of historically sampled sites to retain comparability with those samples or based on stream locations that contained consistent water availability for sampling aquatic biota. Reaches at WICR and GWCA were selected to obtain a representative and unbiased sample through inclusion of comprehensive channel unit types (run, riffle, pool, etc.), and reaches at OZAR and BUFF were selected using a generalized random tessellation stratified method that produces an independent random site selection that is spatially balanced (Stevens and Olsen 2004; Dodd et al. 2008; Petersen et al. 2008). Parks were located within the Great Plains (PIPE, TAPR, and HOME) and Ozarks ecoregions (GWCA, WICR, BUFF, OZAR), with streams in the Great Plains having lower gradients, finer substrates, lower biotic diversity, less annual precipitation, and are more hydraulically and thermally variable than Ozark region streams (Matthews 1988; Dodds et al. 2004). The streams within PIPE, HOME, and TAPR consisted of wadeable prairie streams (4.1 m mean wetted width) with fine substrates and few species (average 27 fish and 103 aquatic invertebrate taxa), while GWCA and WICR were wadeable Ozarks streams (7.7 m mean wetted width) with medium fine/coarse substrate and relatively few species (average 28 fish and 141 aquatic invertebrate taxa; Table 1). At OZAR and BUFF, both the mainstem river and tributaries were sampled. The mainstem river of OZAR and BUFF consisted of wadeable and non-wadeable sites (33 meter mean wetted channel width OZAR; 40 meter mean wetted channel width BUFF) with relatively more species (average 69 fish and 162 aquatic invertebrate taxa; Table 1). The tributaries at OZAR and BUFF were wadeable Ozark streams with relatively higher species richness than WICR and GWCA.

The data used was collected by the Heartland I&M staff during their standard fish and aquatic invertebrate sampling cycle for long-term monitoring (Table 2) at each of the seven parks. Standard sampling protocols for fish, aquatic invertebrates, water chemistry, and habitat were developed by the network to ensure consistency for comparison between sites and years (Bowles et al. 2007; Bowles et al. 2008; Dodd et al. 2008; Petersen et al. 2008). Fish sampling at PIPE, TAPR, and HOME was conducted using seines while GWCA and WICR use backpack or tow barge electrofishing. Depending on the size of the site, BUFF and OZAR mainstem and tributaries were sampled using a combination of backpack, tow barge, and boat electrofishing as well as seining. Reach length at WICR, GWCA, OZAR, and BUFF was 20 times the mean wetted stream width, while reach length at PIPE, TAPR, and HOME was the length required to encompass all channel units present at a site (riffles, runs, pools, glides) and averaged from 46 to 94 meters at these three park units. Fish community composition, abundance, and effort data was collected from fish sampling. Water chemistry measurements collected at each site included water temperature (°C), dissolved oxygen (mg/l), specific conductance (μS/cm), pH, and turbidity (NTU) and were recorded in conjunction with fish sampling at each site. For reaches sampled using seines, each habitat unit (run, riffle, pool, etc.) seined within the reach had habitat data (current velocity, wetted width, depth, and substrate size) measured at the upstream, downstream, and midpoint within the habitat unit. For reaches sampled using electrofishing, these same habitat measurements were taken at three equidistant points across the stream channel within 11 equally spaced transects through the reach. Since the categorical substrate data collected differed by NPS unit and

faunal group (dominant substrate type in Plains versus Wentworth particle size class in Ozarks) we created a hybrid classification method as follows: 1 = muck, detritus, and silt, < 0.06mm, and 1 on the Wentworth scale; 2 = sand, 0.06-2mm, and 2-6 on the Wentworth scale; 3 = pea gravel, 2-16mm, and 7-11 on the Wentworth scale; 4 = coarse gravel, 16-64mm, and 12-15 on the Wentworth scale; 5 = cobble, 64-256mm, and 16-19 on the Wentworth scale; 6 = boulder, 256-4000mm, and on the 20-23 Wentworth scale.

Aquatic invertebrate sampling was conducted at each of the aforementioned NPS units with exception to TAPR (Table 2). Each time a site was sampled for aquatic invertebrates, the same habitat and water chemistry data as for the fish sites was collected but only at the location where aquatic invertebrate samples were collected (Typically riffle channel units; Table 2). The parks PIPE, GWCA, and WICR were sampled using a surber sampler, BUFF and OZAR were sampled with a slack surber, and HOME was sampled with hester-dendy samplers due to sandy substrate. For PIPE, GWCA, WICR, BUFF, and OZAR, samples were taken at the first three riffles from the downstream boundary of each reach. Each riffle had three samples taken equidistant across it. At HOME, hester-dendy samplers were placed at the upper and lower sample reach in grids of five and deployed for a minimum of 30 days for colonization. Comparability of these methods were justified by previous works that found no relationship between sampling method and *Ephemeroptera*, *Plecoptera*, and *Trichoptera* richness (Letovsky et al. 2012) and low variability between Hilsenhoff Biotic Index (HBI; Hilsenhoff 1987) and sampling method (Guild et al. 2014). Water temperature (°C), dissolved oxygen (mg/l), specific conductance (μS/cm), pH, and turbidity (NTU) were also collected at each sampling location. Habitat measurements were collected at each sampling location at

each riffle (except HOME) and include riffle length and width, depth, current velocity, and dominant substrate. Substrate data again was converted to the hybrid classification system described above.

Assessment of Biotic Variability

Fish community structure was assessed at each site during each year it was sampled using one of two Indices of Biotic Integrity (IBI) depending on whether the park was in the Great Plains or Ozark ecoregion (Plains; Fausch et al. 1984; Ozarks; Dauwalter et al. 2003). Aquatic invertebrate community structure was assessed at each site during each year it was sampled using the HBI (Hilsenhoff 1987). These metrics allowed us to measure the variability in community condition and functional organization which may be lost by solely testing species abundance across time. The coefficient of variation (CV) of the IBI and HBI was used to assess the spatial and temporal variability of stream communities at each NPS unit. Temporal variability for each site (within each NPS unit) was measured by calculating the CV (of the IBI or HBI) for each site over all the years it was sampled and spatial variability was measured by calculating the CV (of the IBI or HBI) of all the sites sampled in any one year (separately by NPS unit). Variability scores were then averaged (for spatial variability and temporal variability) within each NPS unit so that each unit has a single spatial and temporal variability score (Figure 2). A repeated measures ANOVA (NPS units among a region) was then used to determine if the mean variability differed by faunal group, variability type (spatial vs. temporal), or an interaction of the two. To focus on variability of species rather than community condition, we also evaluated the persistence of fish species and aquatic invertebrate families across time within each NPS unit by investigating the proportion of

20

sampling seasons the species/family was present at each NPS unit. A t-test was then used to determine if there was a difference in the mean persistence (proportion of years the species/family was present) by region (Plains or the Ozarks).

Fish Community Vulnerability

A trait based approach incorporating a species feeding ecology, life history, habitat preference, and tolerance traits was used to assess fish community vulnerability to anthropogenic disturbance. Species traits were obtained from the Fishtraits database (Frimpong and Angermeier 2009) and from a freshwater fish life history trait database (Mims et al. 2010). Measuring community vulnerability was accomplished by calculating a vulnerability score for each species found within a park (Table 3). Scoring accounted for susceptibility to disturbances (habitat degradation, warming temperature, and altered flow regime) as well as dispersal ability and persistence at a park through time. Scoring for habitat, temperature, flow, and dispersal ability was defined as by Sievert (2014), where species vulnerable to each metric received a 1 and all others received a 0. Lithophilic spawners or benthic invertivores received a 1 in habitat degradation, cool- and cold-water species received a 1 in warming temperatures, equilibrium or periodic life history strategists received a 1 in altered flow regime, and members in the *Fundulidae*, *Cottidae*, and *Percidae* family received a 1 in dispersal ability. The species persistence score was defined as one subtracted by the ratio of the number of years the species was present at a park by the number of years the park was sampled. A species vulnerability score was the sum of scores received from habitat degradation, flow modification, temperature increase, dispersal ability, and species persistence. Translating the species vulnerability score to community vulnerability score was done by calculating the mean

21

species vulnerability score within each park. The proportion of the community at each park that was vulnerable to each of the four main disturbance types (habitat degradation, flow modification, temperature increase, and dispersal ability) was then summarized to better quantify the impacts that specific disturbance types may have on fishes. A t-test was then used to determine if there was a difference in the mean proportion of the fish community vulnerable to habitat degradation, flow alteration, temperature increase, and dispersal ability between NPS units located in the Plains or the Ozarks.

Aquatic Invertebrate Community Vulnerability

Species traits were also used to calculate the vulnerability scores for aquatic invertebrates. The United States Environmental Protection Agency (EPA) Freshwater Biological Traits Database was used to obtain aquatic invertebrate feeding group, species life history, habitat preference, and tolerance traits (EPA 2012). Due to the lack of comprehensive trait data for aquatic invertebrate taxa, we selected only *Ephemeroptera*, *Plecoptera*, and *Trichoptera* taxa since they were the best represented taxa within our assemblages that had trait data available, tend to be sensitive indicators of stream condition (Lenat 1988), and there is minimal differences in EPT richness using a hester-dendy and surber sampler (Letovsky et al. 2012). The aquatic invertebrate disturbance vulnerability assessment incorporated habitat degradation, altered flow regime, and warming temperature (Table 3). Scoring used the 0 or 1 values as used in the fish vulnerability index where taxa with a sensitivity value less than 4.5 (Hilsenhoff 1987) received a 1 in habitat degradation, taxa represented in a single flow classification of moderate flow velocity or higher received a score of 1 in vulnerable to an altered flow regime (Rader and Belish 1999), and received a 1 in temperature increase if classified as

a cold stenotherm (<5°C) or cold-cool eurytherm (0-15°C; Poff et al. 2010) taxa. Community vulnerability was calculated using the same methods for fish vulnerability. Similar to fish, the proportion of taxa vulnerable to each of the three disturbance types (habitat degradation, flow modification, and temperature increase) were compared to determine what disturbance would have the greatest impact on the community. A t-test was then used to determine if there was a difference in the mean proportion of the aquatic invertebrate community vulnerable to habitat degradation, flow alteration, and temperature increase between NPS units located in the Plains or the Ozarks.

Areas of Heightened Variability/Vulnerability

We used fish and aquatic invertebrate variability and vulnerability scores to determine if location, in-stream habitat, or climate variables were related to heightened variability or vulnerability (Table 4). Location variables included region (Plains or Ozarks), degrees latitude, and land cover (percent agriculture or urban in the upstream watershed). In-stream habitat variables included the means and CV of stream size (wetted channel width), substrate, current velocity, and water depth. Climate variables included average annual maximum and minimum air temperatures, average annual mean air temperature, average annual total precipitation, and average annual maximum precipitation event (24 hour period). Collinearity was then assessed individually among variables related to location, habitat, and climate, and those with $r^2 > 0.70$ were removed from analysis. We used regression analysis (for continuous variables) and ANOVA (for categorical variables) to determine if greater fish and aquatic invertebrate vulnerability and variability was associated with the retained geographic location, in-stream habitat, or climate metrics. Finally, geospatial data layers of predicted air temperature and

23

precipitation in the year 2080 (mean of 16 General Circulation Models) under the medium A1B emission scenario (Girvetz et al. 2009) were used if a climactic variable was able to best describe variability or vulnerability and to determine how these influences may impact parks or regions in the future.

Results

Across all sites, years, and NPS units, a total of 177,291 individuals within 99 species of fish were sampled and a total of 1,143,620 aquatic invertebrate individuals were sampled, 73.4% were identified to genus. Since 24 tributaries within the BUFF and 14 tributaries within the ONSR were only sampled once, we were unable to calculate temporal variability for those sites and were discarded from the spatial and temporal variability analysis. Although we were able to obtain trait data to discern all 99 species of fish as vulnerable to habitat degradation, flow modification, temperature increase, and dispersal ability, only 55 of the 98 unique EPT taxa had trait data to discern if it was vulnerable to habitat degradation, flow modification, and temperature increase. Therefore at each NPS unit, 58 to 68 percent of the EPT taxa (17-31% of total community taxa) had trait data available in all three vulnerability categories. The temporal range of our study was long enough (7-17 years) so that there was likely greater than one complete turnover of all individuals, which is the minimum time period to assess the stability or persistence of a community (Connell and Sousa 1983; Hansen and Ramm 1994). The spatial scale of our study varied based on stream size and the area encompassed in each NPS unit but was sufficient to encounter both complementation and supplementation of habitats at each unit (Connell and Sousa 1983; Dunning et al. 1992). Mean annual minimum air temperature was removed from analysis because it was related to mean annual air

temperature ($r^2 > 0.98$), mean annual maximum precipitation event in a 24 hour period was removed from analysis because it was related to total annual precipitation ($r^2 > 0.85$), and CV of wetted channel width were removed from analysis because it was related to wetted channel width ($r^2 > 0.85$).

The mean IBI scores ranged from fair (37; Fausch et al. 1984) to reference (80; Dauwalter et al. 2003) and the mean HBI score ranged from fairly poor to very good (6.33 to 3.76; Hilsenhoff 1987). Variability was different between faunal group and variability type (interaction between faunal group and variability type; P = 0.02) and thus we used a single factor ANOVA for each faunal group. The mean CV of fish IBI across sites in a year (spatial variability) was higher (CV ranging from 0.12 to 0.25) compared to temporal variability (CV ranging from 0.04 to 0.17; P = 0.03; Table 5). By contrast, the mean CV in aquatic invertebrate HBI across sites in a year (0.06 to 0.15; spatial variability) did not differ from the mean CV in a site across years (0.10 to 0.16; temporal variability) (Table 5). In general, we found that the fish IBI varies more from site to site within a park than year to year, but in contrast, the aquatic invertebrate HBI did not vary differently from year to year within a park than site to site. The proportion of fish species at each NPS unit that were present during each year the park was sampled ranged from 0.17 at PIPE to 0.57 at BUFF and the proportion of aquatic invertebrate taxa (family level) at each NPS unit that were present during each year the park was sampled ranged from 0.02 at HOME to 0.46 at OZAR (Table 6). Ozark NPS units had a greater proportion of fish species present each year the park was sampled than Plains sites (P = 0.04) while this was not the case for aquatic invertebrates (P = 0.13).

Temporal variability in the fish IBI was not related to any of the location, habitat, or climate variables investigated, but spatial variability was linked to land use. More urban development in the upstream watershed was linked to greater spatial variability in the fish IBI across all parks ($r^2 = 0.70$, $P = 0.04$, Figure 3). However, this relationship may have been influenced by the spatial arrangement of sites within parks. For example, PIPE has sites above and below a waterfall which have different community compositions while WICR has its three sites distributed among three different branches of a river which again have different community compositions regardless of upstream land use. When investigating the effects of removing the outlier (WICR) from the analysis, the relationship was still evident despite not being significant ($r^2 = 0.72$, $P = 0.07$) due to the limited sample size. Mean temporal variability in aquatic invertebrate HBI was greater in the Plains rather than the Ozarks Ecoregion (*ANOVA*; $P = 0.03$), northern latitudes ($r^2 = 0.87$, $P = 0.007$), and cooler average annual mean air temperatures ($r^2 = 0.82$, $P = 0.01$). In general, locations farther north, which tended to be Plains NPS units with cooler air temperatures had greater temporal variability in HBI scores (Figure 4). Spatial variability in aquatic invertebrates was greater in streams with a larger wetted channel width ($r^2 = 0.76$, $P = 0.02$), greater annual total precipitation ($r^2 = 0.73$, $P = 0.03$), and in locations with less agriculture in the upstream watershed ($r^2 = 0.76$, $P = 0.02$; Figure 5). Under the medium A1B emission scenario for the year 2080, mean annual air temperatures are projected to increase 3.55 to 4.71°C and total annual precipitation is projected to change -0.9 to 33.4cm depending on NPS unit. These changes would correspond to a 1-3% decrease in the temporal variability and a -0.5-3% change in the spatial variability in the HBI.

Of the 99 species of fish that were found in the seven NPS units, 55 were vulnerable to habitat degradation, 41 were vulnerable to warming temperatures, 43 were vulnerable to an altered flow regime, and 22 had poor dispersal ability (Appendix 1). Eleven species were not vulnerable to any of these categories while 88 species were vulnerable to at least one of these categories. Only one species, the Ozark Sculpin *Cottus hypselurus*, was vulnerable to each of these four categories and was the most vulnerable species at BUFF, OZAR, and WICR. The Stonecat *Noturus flavus* was the most vulnerable species at GWCA, HOME, and TAPR and the Iowa Darter *Etheostoma exile* and Yellow Perch *Perca flavescens* were the most vulnerable species at PIPE. The proportion of the community vulnerable to habitat alteration was greater in the Ozarks region (P = 0.01) and was lowest at HOME (0.27) and highest at GWCA (0.69). Although NPS units in the Ozarks region tended to have a higher proportion of the community vulnerable to water temperature increase compared to Plains region parks (0.19 at TAPR to 0.63 at GWCA), this difference was not significant (P = 0.12). The proportion of the community vulnerable to a change in flow regime ranged from 0.33 (PIPE) to 0.53 (HOME) with no difference between regions (P = 0.51). NPS units in the Ozarks region also had a greater proportion of the community with limited dispersal ability than NPS units in the Plains (P = 0.03) with proportion of the community as poor dispersers ranging from 0 (HOME) to 0.38 (GWCA; Table 7). The mean species scores at each NPS unit (park vulnerability score) ranged from 1.45 to 2.31 out of a possible ~5 (Table 7). HOME was found to be the least vulnerable whereas GWCA was found to be the most vulnerable NPS unit with the general trend being that NPS units in the Ozarks were more vulnerable than NPS units in the Plains (P = 0.08). We also found that fish

vulnerability was linked to substrate size ($r^2 = 0.70$, $P = 0.02$) where streams with coarser substrate tended to be more vulnerable (Figure 6). Because NPS units within the Ozarks tend to have coarser substrate than in the Plains ecoregion, this result is also likely due to a regional difference.

Of the 55 EPT taxa used for vulnerability analysis, 11 were vulnerable to water temperature increase, 38 were vulnerable to habitat degradation, and 42 were vulnerable to flow regime shift (Appendix 2). Six taxa were not vulnerable to any of the three categories while 49 taxa were vulnerable to at least one of the three categories. Nine taxa were vulnerable to all three categories (score of 3). Similar to fish, Ozark NPS units had a greater proportion of the aquatic invertebrate community vulnerable to habitat alteration ($P = 0.01$) with proportions at each of the seven NPS units from 0.48 at HOME to 0.74 at OZAR. Proportion of the aquatic invertebrate community vulnerable to temperature increase ranged from 0.14 at HOME to 0.25 at GWCA, with no difference between regions ($P = 0.29$). As with fish, HOME, with 0.9 of the community vulnerable to a change in flow regime, was found to have the greatest proportion of the community vulnerable to this (Table 7). The mean species scores at each NPS unit (park vulnerability score) ranged from 1.52 (HOME) to 1.75 (GWCA) out of a possible 3 (Table 7). We found that aquatic invertebrate vulnerability was greater in the Ozarks than the Plains ecoregion ($P = 0.04$).

Discussion

Our study builds upon the knowledge regarding the spatial and temporal variability of fish and aquatic invertebrate communities and their vulnerability to anthropogenic degradation. In addition, we re-conceptualize the methods in which

community stability may be inferred. The conventional definition of stability developed by Connell and Sousa (1983) is the consistency in the numbers of individuals within a species over time, and has been regularly used (Matthews 1986; McElravy et al. 1989; Ross et al. 1985; Hansen and Ramm 1994; Schaefer et al. 2012). However, this definition requires accounting for frequency and severity of disturbance and their impacts on habitat and biota, and also accounting for natural variability through time and space (Horwitz 1978; Schlosser 1982; Poff and Ward 1989; Poff and Allan 1995). Therefore, the Connell and Sousa (1983) definition better serves as a baseline measure of variability. Instability, then, should be characterized by increases in this natural variability. Since we were unable to discern the difference between natural and additive variability due to anthropogenic impacts, we measured current variability (Connell and Sousa's 1983 definition of stability) so that future monitoring can use departures from this baseline variability to assess decreases in stability. By using biotic index scores, we were still able to consider abundances of individuals (typically broken into functional groups instead of species) as well as community integrity to determine the health status of the system. This minimized the influence of natural fluctuations in individual species abundances because species within a community often have inverse or compensatory relationships over time (McNaughton 1977; Naeem and Li 1997; Bai et al. 2004). In addition, using biotic index scores allows stability to be compared among streams with differing species compositions (Karr 1981).

Our results suggest that spatial variability in fish communities was similar or greater than temporal variability at each of the NPS units. Matthews (1986) found that the fish community in an Ozark stream recovered from a 90 year flood in eight months which

supports our findings of low temporal variability in fish communities, even after large-scale disturbances. Similarly, Dodds et al. (2004) observed rapid recovery in fish biodiversity after floods or desiccation events within Great Plains streams. While fish communities in upstream locations tended to vary over time due to shifting habitats, downstream habitats remain relatively more stable which leads to a less temporally variable community (Horwitz 1978; Karr et al. 1987). Despite greater habitat heterogeneity being linked to greater spatial variability in fish communities, Freeman et al. (1988), Karr et al. (1987) and Brown (2003) found that greater habitat heterogeneity reduces temporal variability in stream communities. Therefore, we believe that an artifact of promoting a heterogeneous and diverse stream environment with low temporal variability may be greater community spatial variability (Franssen et al. 2011; Shaefer et al. 2012).

Whether the aquatic invertebrate community was more spatially or temporally variable was contingent upon which NPS unit was being investigated. The parks with lower temporal variability in HBI scores averaged very good or good quality according to the HBI classification categories (Hilsenhoff 1987), which supports our previous claim with fish, that increased habitat heterogeneity is linked to greater community spatial variability, and in turn reduces temporal variability (Karr et al. 1987; Brown 2003). Community temporal variability is partially dependent upon intensity and length of time since previous disturbances (Connell and Sousa 1983). Timing of disturbances influences the degree in which a community is affected (John 1964; Matthews 1986; McElravy 1989); therefore, community stability likely differs between fish and aquatic invertebrates due to differences in life history, and modes and rates of recovery post disturbance

(Matthews 1986; McElravy et al. 1989; Fausch and Bramblett 1991; Boulton et al. 1992).

Fish may be more persistent over time because they are more mobile throughout

ontogeny (Dunning et al. 1992) and efficient at seeking refugia to avoid periods of stress

(Karr 1981). In contrast, aquatic invertebrates are more sedentary within the stream, are

prone to displacement from disturbance events, and are less efficient at recolonization

post-disturbance because they rely on aerial sources (during brief hatch periods) and drift

(Williams and Hynes 1976), which may explain why they are less persistent over time.

Streams that received greater total annual precipitation were positively related to

the spatial variability in aquatic invertebrates, which supports previous findings on the

relationship between hydrology and community variability (Horwitz 1978; Matthews

1986; McElravy 1989; Poff and Ward 1989; Boulton et al. 1992; Poff and Allan 1995). In

addition, streams with more agriculture within their watershed were associated with a

decrease in spatial variability in the aquatic invertebrate community, which is consistent

with Scott and Helfman (2001), Rahel (2002), and Walters et al. (2003) who showed

increased agriculture and urban in upstream watersheds tends to homogenize in-stream

habitats occupied by a more tolerant generalist community that are less spatial variable.

However, we also found an increase in spatial variability in the fish community was

associated with an increase urban development which refutes this argument. Our results

likely were driven by three sites at WICR that had different branches with urban ranging

from 9 to 65 percent in the watershed (average 38). Therefore, it is likely that the degree

of habitat homogenization differs among each of the branches and may explain why there

were consistently different communities among these three sites.

Although temporal variability in aquatic invertebrates was associated with mean air temperature and northern latitudes, the relationship appears to be driven by NPS units in the Plains being located in northern latitudes with colder climates. Prairie streams often exist on a balance of flood and desiccation events which play important roles in structuring aquatic organism abundance and distribution (Matthews 1988; Ostrand and Wilde 2002, 2004; Scheurer et al. 2003; Fritz and Dodds 2005). This may explain why there was increased spatial variability and the low persistence of fish and aquatic invertebrates found in the Prairie streams versus the Ozark streams throughout our study period.

Both of the fish and aquatic invertebrate vulnerability assessments showed agreement among the most and least vulnerable NPS units. Because prairie streams typically experience high flood and drought frequency, extremes in high and low temperatures, and dissolved oxygen and salinity, biota have adapted to be tolerant generalist species that are reliant on rapid recolonization and high fecundity to persist (Matthews 1988; Ostrand and Wilde 2002; Dodds et al. 2004). Despite this stochastic environment, HOME was found to have a large proportion of *Centrarchids* and *Ictalurids*, which tend to be equilibrium life strategists (Mims et al. 2010) and explains why HOME was the most vulnerable to an altered flow regime, despite being the least vulnerable NPS unit overall. GWCA had the most vulnerable fish and aquatic invertebrate communities. This NPS unit has cool-water influence from springs (Dodd et al. 2011) and may explain why it had the highest proportion of fish and aquatic invertebrates that were vulnerable to warming temperatures. Streams that are spring influenced tend to have more cool- or cold-water species/taxa, and therefore these taxa

are more vulnerable to warming temperatures as was found in the Ozark NPS units.

However, since the thermal regime from many Ozark rivers/streams is driven by stable

discharges of cool groundwater and is thermally buffered by the impacts of air

temperature (Eaton and Scheller 1996; Mohseni and Stephan 1999; Mohseni et al. 2003;

Lyons et al. 2010; Westhoff and Paukert 2014), it is unlikely that these streams will

experience substantial increases in water temperature. Therefore, it appears that although

the communities are sensitive to warming temperatures, the streams themselves may not

necessarily be vulnerable to warming temperatures due to the thermal buffering capacity

of groundwater inputs.

The aquatic invertebrate community in the Ozarks region was determined to be

more vulnerable than the Plains region. However, aquatic invertebrate vulnerability was

not related to any of the other location, land use, and climate metrics measured. Prairie

streams are harsh environments, and because of this are made up of tolerant generalist

species (Matthews 1988; Ostrand and Wilde 2002; Dodds et al. 2004). Ozark streams in

our study were stable environments driven by steady and predictable discharges and

temperatures (Converse 1994; Kennen et al. 2009) due to the presence of springs and due

to their lower network position, which have produced stable environmental conditions

(Karr et al. 1987; Lohr and Fausch 1997; Ostrand and Wilde 2002). Since stable

environments allow species to become specialized to efficiently exploit resources and

persist through time, they then become more vulnerable to changes in environmental

condition (Munday 2004), which supports our findings that aquatic invertebrate

communities in streams in the Ozarks are more vulnerable than in streams in the Plains.

Fish community vulnerability was greater in Ozark streams with larger substrates than Plains streams with find substrate. Although it is likely that fish vulnerability is greater in the Ozarks for similar reasons as with aquatic invertebrates above, there is further evidence to suggest why streams with larger substrates have heightened vulnerability scores. Goldstein and Meador (2004) found that streams similar in size to our Ozark streams had larger substrate dominated by gravel and cobble and the greatest percent of benthic species and invertivore species. Our study used benthic invertivores as a trait to define vulnerability and thus may help to further explain this result. Habitat degradation is often associated with substrate embeddedness (Schlosser 1991; Poff et al. 1997; Allan 2004), and the community at sites that are already degraded (with finer substrates) will likely be comprised of the tolerant generalist species in the community with low vulnerability.

In response to a rapidly growing human population leading to changes in land use, climate, and environmental conditions, it is becoming ever more important to understand current condition of stream systems so that measures of degradation or change can be determined. Since the early signs that biota are under environmental stress are changes in "normal" abundances and persistence (Ross et al. 1985), determining the current fluctuations in abundance and persistence is important to define, such that departures from this can be used as signs of environmental stress and possible focuses for protection. This study has provided evidence that streams are inherently spatially and temporally variable even within systems that generally have limited disturbance, and that low temporal variability may be linked with increase in ecological integrity (Karr et al. 1987; Brown 2003). Although our results suggest that changes in climate through the year 2080

may only affect the spatial and temporal variability of aquatic communities a maximum of 3%, they provide elements to monitor as land becomes developed and air temperature and precipitation patterns shift. Therefore, it places importance on understanding current fluctuations in the community under current conditions so that departures from this baseline can be used to infer community instability due to environmental change in future settings.

Literature Cited

Allan, J. D., D. L. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. Freshwater Biology 37: 149-161.

Allan, J. D., 2004. Landscapes to riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology, Evolution, and Systematics 35: 257-284.

Bai, Y., X. Han, J. Wu, Z. Chen, and L. Li. 2004. Ecosystem stability and compensatory effects in the inner Mongolian grassland. Nature 431: 181-184.

Boulton, A. J., C. G. Peterson, N. B. Grimm, and S. G. Fisher. 1992. Stability of an aquatic macroinvertebrate community in a multiyear hydrologic disturbance regime. Ecology 73: 2192-2207.

Bowles, D. E., J. A. Luraas, L. W. Morrison, H. R. Dodd, M. H. Williams, G. A. Rowell, M. D. DeBacker, J. A. Hinsey, F. D. Usrey, and J. L. Haack. 2007. Protocol for monitoring aquatic invertebrates at Ozark National Scenic Riverways, Missouri, and Buffalo National River, Arkansas. Natural Resource Report NPS/HTLN/NRR – 2007/009. National Park Service, Fort Collins, Colorado.

Bowles, D. E., M. H. Williams, H. R. Dodd, L. W. Morrison, J. A. Hinsey, C. E. Ciak. G. A. Rowell, M. D. DeBacker, and J. L. Haack. 2008. Protocol for monitoring aquatic invertebrates of small streams in the heartland network inventory & monitoring network. Natural Resource Report NPS/HTLN/NRR – 2008/042. National Park Service, Fort Collins, Colorado.

Bramblett, R. G., and K. D. Fausch. 1991. Variable fish communities and the index of biotic integrity in a western Great Plains river. Transactions of the American Fisheries Society 120: 752-769.

Brown, B. L. 2003. Spatial heterogeneity reduces temporal variability in stream insect communities. Ecology Letters 6: 316-325.

Connell, J. H. and W. P. Sousa. 1983. On the evidence needed to judge ecological stability and persistence. The American Naturalist 121: 789-824.

Converse, J. W. 1994. Water chemistry, nutrient dynamics, and macrophyte production of a large cold-water spring. MS thesis. University of Missouri, Columbia.

Dauwalter, D. C., E. J. Pert, and W. E. Keith. 2003. An index of biotic integrity for fish assemblages in Ozark Highland Streams of Arkansas. Southeastern Naturalist 2: 447-468.

Diana, M., J. D. Allan, and D. Infante. 2006. The influence of physical habitat and land use on stream fish assemblages in southeastern Michigan. American Fisheries Society Symposium 48: 359-374.

Dodd, H. R., D. G. Peitz, G. A. Rowell, D. E. Bowles, and L. W. Morrison. 2008. Protocol for monitoring fish communities in small streams in the heartland inventory and monitoring network. Natural Resource Report NPS/HTLN/NRR – 2008/052. National Park Service, Fort Collins, Colorado.

Dodd, H.R. 2011. Fish community monitoring at George Washington Carver National Monument: 2006, 2007 and 2010 status report. Natural Resource Technical Report NPS/HTLN/NRTR—2011/138. National Park Service, Fort Collins, Colorado.

Dodds, W. K., K. Gido, M. R. Whiles, K. M. Fritz, and W. J. Mattews. 2004. Life on the edge: the ecology of Great Plains prairie streams. Bioscience 54: 205-216.

Dunning, J. B., B. J. Danielson, and H. R. Pulliam. 1992. Ecological processes that affect populations in complex landscapes. Oikos 65: 169-175.

Eaton, J. G., and R. M. Scheller. 1996. Effects of climate warming on fish thermal habitat in streams of the United States. Limnology and Oceanography 41: 1109-1115.

Fancy, S. G. and R. E. Bennetts. 2012. Institutionalizing and effective long-term monitoring program in the US National Park Service. Design and Analysis of Long-term Ecological Monitoring Studies. Pages 481-497. Cambride University Press. New York, NY.

Fausch, K. D. and K. R. Bestgen. 1997. Ecology of fishes indigenous to the central and southwestern Great Plains. Ecology and Conservation of Great Plains Vertebrates 125: 131-166.

Fausch, K. D., J. R. Karr, and P. R. Yant. 1984. Regional applications of an index of biotic integrity based on steam fish communities. Transactions of the American Fisheries Society 113: 39-55.

Ficke, A., C. Myrick, and L. Hansen. 2007. Potential impacts of global climate change on freshwater fisheries. Reviews in Fish Biology and Fisheries 17: 581-613.

Foster, D., F. Swanson, J. Aber, I. Burke, N. Brokaw, D. Tilman, and A. Knapp. 2003. The importance of land-use legacies to ecology and conservation. Bioscience 53: 77-88.

Franssen, N. R., K. B. Gido, C. S. Guy, J. A. Tripe, S. J. Shrank, T. R. Strakosh, K. N. Bertrand, C. M. Franssen, K. L. Pitts, and C. P. Paukert. Effects of floods on fish assemblages in an intermittent prairie stream. Freshwater Biology 51: 2072-2086.

Freeman, M. C., M. K. Crawford, J. C. Barrett, D. E. Facey, M. G. Flood, D. J. Stouder, and G. D. Grossman. 1988. Fish assemblage stability in a southern Appalachian stream. Canadian Journal of Fisheries and Aquatic Sciences 45: 1949-1958.

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

Fritz, K. M., and W. K. Dodds. 2005. Harshness: characterization of intermittent stream habitat of space and time. Marine and Freshwater Research 56: 13-23.

Gido, K. B., D. L. Propst, J. D. Olden, and K. R. Bestgen. 2013. Multidecadal responses of native and introduced fishes to natural and altered flow regimes in the American Southwest. Canadian Journal of Fisheries and Aquatic Sciences 70: 554-564.

Girvetz, E. H., C. Zganjar, G. T. Raber, E. P. Maurer, P. Kareiva, and J. J. Lawler. 2009. Applied climate-change analysis: the climate wizard tool. PlosOne DOI: 10.1371/journal.pone.0008320

Goldstein, R. M. and M. R. Meador. 2004. Comparisons of fish species traits from small streams to large rivers. Transactions of the American Fisheries Society 133: 971-983.

Graf, W. L. 1999. Dam nation: a geographic census of American dams and their large-scale hydrologic impacts. Water Resource Research 35: 1305-1311.

Guild, K., A. Anthony, M. Bilger, and J. Holt. 2014. Assessment of passive and active macroinvertebrate collection methods in adjacent reaches on the upper main stem of the Susquehanna River. Journal of the Pennsylvania Academy of Science 88: 47-56.

Hansen, M. J., and C. W. Ramm. 1994. Persistence and stability of fish community structure in a southwest New York stream. The American Midland Naturalist 132: 52-67.

Harding, J. S., E. F. Benfield, P. V. Bolstad, G. S. Helfman, and E. B. D. Jones III. 1998. Stream biodiversity: the ghost of land use past. Proceedings of the National Academy of Sciences. 95: 14843-14847.

Hilsenhoff, W. L., 1987. An improved biotic index of organic stream pollution. Great Lakes Entomologist 20: 31-39.

Horwitz, R. J., 1978. Temporal variability patterns and the distributional patterns of stream fishes. Ecological Monographs 48: 307-321.

John, K. R. 1964. Survival of fish in intermittent streams of the Chiricahua Mountains, Arizona. Ecology 45:112-119.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6: 21-27.

Karr, J. R., P. R. Yant, K. D. Fausch, and I. J. Schlosser. 1987. Spatial and temporal variability of the index of biotic integrity in three Midwestern streams. Transactions of the American Fisheries Society 116: 1-11.

Kennen, J.G., J. A. Henriksen, J. Heasley, B. S. Cade, and J. W. Terrell. 2009. Application of the Hydroecological Integrity Assessment Process for Missouri Streams: U.S. Geological Survey Open-File Report 2009–1138, 57 p.

Lenat, D. R. 1988. Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. Journal of the North American Benthological Society 7: 222-233.

Letovsky, E., I. E. Myers, A. Canepa, and D. J. McCabe. 2012. Differences between kick sampling techniques and short-term Hester-Dendy samples for stream macroinvertebrates. BIOS 83: 47-55.

Lohr, S. C. and K. D. Fausch. 1997. Multiscale analysis of natural variability in stream fish assemblages of a western Great Plains watershed. Copia 1997: 706-724.

Lyons, J., J. S. 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: 1867-1898.

Matthews, W. J. 1986. Fish faunal structure in an Ozark stream: stability, persistence and a catastrophic flood. Copeia 1986: 388-397.

Matthews, W. J., 1988. North American prairie streams as systems for ecological study. Journal of the North American Benthological Society 7: 387-409.

McElravy, E. P., G. A. Lamberti, and V. H. Resh. 1989. Year-to-year variation in the aquatic macroinvertebrate fauna of a northern California stream. Journal of North American Benthological Society 8: 51-63.

McNaughton, S. J. 1977. Diversity and stability of ecological communities: a comment on the role of empiricism in ecology. The American Naturalist 111: 515-525.

Meyer, J. L., M. J. Sale, P. J. Mulholland, and N. L. Poff. 1999. Impacts of climate change on aquatic ecosystem functioning and health. Journal of the American Water Resources Association 35: 1373-1386.

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

Moerke, A. H. and G. A. Lamberti. 2006. Scale-dependent influences of water quality, habitat, and fish communities in streams of the Kalamazoo River Basin, Michigan (USA). Aquatic Sciences 68: 193-205.

Mohseni, O., and H. G. Stefan. 1999. Stream temperature/air temperature relationship: a physical interpretation. Journal of Hydrology 218: 128-141.

Mohseni, O., H. G. Stefan, and J. G. Eaton. 2003. Global warming and potential changes in fish habitat in U.S. streams. Climate Change 59: 389-409.

Munday, P. L. 2004. Habitat loss, resource specialization, and extinction on coral reefs. Global Change Biology 10: 1642-1647.

Naeem, S., and S. Li. 1997. Biodiversity enhances ecosystem reliability. Nature 390: 507-509.

Nerbonne, B. A., and B. Vondracek. 2001. Effects of local land use on physical habitat, benthic macroinvertebrates, and fish in the Whitewater River, Minnesota, USA. Environmental Management. 28: 87-99.

Nesler, T. P., R. T. Muth, and A. F. Wasowicz. 1988. Evidence for baseline flow spikes as spawning cues for Colorado squawfish in the Yampa River, Colorado. Transactions of the American Fisheries Society Symposium 5: 68-79.

O'Grady, J. J., D. H. Reed, B. W. Brook, and R. Frankham. 2004. What are the best correlates of predicted extinction risk?. Biological Conservation 118: 513-520.

Olden, J. D., N. L. Poff, and K. R. Bestgen. 2006. Life-history strategies predict fish invasions and extirpations in the Colorado River basin. Ecological Monographs 76: 25-40.

Olden, J. D., Z. S. Hogan, and M. J. Vander Zanden. 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.

Ostrand, K. G., and G. R. Wilde. 2004. Changes in prairie stream fish assemblages restricted to isolated streambed pools. Transactions of the American Fisheries Society 133: 1329-1338.

Ostrand, K. G., and G. R. Wilde. 2002. Seasonal and spatial variation in a prairie stream-fish assemblage. Ecology of Freshwater Fish 11: 137-149.

Petersen, J. C., B. G. Justus, H. R. Dodd, D. E. Bowles, L. W. Morrison, M. H. Williams, and G. A. Rowell. 2008. Methods for monitoring fish communities of Buffalo National River and Ozark national Scenic Riverways in the Ozark Plateaus of Arkasas and Missouri: Version 1. U. S. Geological Survey Open-File Report 2007-1302, 94p.

Poff, N. L. and J. V. Ward. 1989. Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patters. Canadian Journal of Fisheries and Aquatic Sciences 46: 1805-1818.

Poff, N. L. and J. D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology 76: 606-627.

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: 769-784.

Poff, N. L., M. I. Pyne, B. P. Bledsoe, C. C. Cuhaciyan, and D. M. Carlisle. 2010. Developing linkages between species traits and multiscaled environmental variation to explore vulnerability of stream benthic communities to climate change. Journal of North American Benthological Society 29: 1441-1458.

Rader, R. B., and T. A. Belish. 1999. Influence of mild to severe flow alterations on invertebrates in three mountain streams. Regulated Rivers: Research and Management 15: 353-363.

Rahel, F. J., 2002. Using current biogeographic limits to predict fish distributions following climate change. American Fisheries Society Symposium 32: 99-110.

Ross, S. T., W. J. Matthews, and A.A. Echelle. 1985. Persistence of stream fish assemblages: effects of environmental change. The American Naturalist 126: 24-40.

Schaefer, J. F., S. R. Clark, and M. L. Warren. 2012. Diversity and stability in Mississippi stream fish assemblages. Freshwater Science 31: 882-894.

Scheurer, J. A., K. D. Fausch, and K. R. Bestgen. 2003. Multiscale processes regulate brassy minnow persistence in a Great Plains river. Transactions of the American Fisheries Society 132: 840-855.

Schlosser, I. J., 1982. Fish community structure and function along two habitat gradients in a headwater stream. Ecological Monographs 52: 395-414.

Schlosser, I. J., 1991. Stream ecology: a landscape perspective. Bioscience 41: 704-712.

Schweizer, P. E., and G. R. Matlack. 2005. Annual variation in fish assemblages of watersheds with stable and changing land use. The American Midland Naturalist 153: 293-308.

Scott, M. C., and G. S. Helfman. 2001. Native invasions, homogenization, and the mismeasure of integrity of fish assemblages. Fisheries 26: 6-15.

Sievert, N. A. 2014. Missouri stream fish vulnerability and representation within conservation networks. MS thesis. University of Missouri, Columbia.

Stevens, D. L., and A. R. Olson. 2004. Spatially balanced sampling of natural resources. Journal of the American Statistical Association 99: 262-278.

U. S. EPA. Freshwater Biological Traits Database (Final Report). U. S. Environmental Protection Agency, Washington, DC, EPA/600/R-11/038F, 2012.

Walters, D. M., D. S. Leigh, and A. B. Bearden. 2003. Urbanization, sedimentation, and the homogenization of fish assemblages in the Etowah River Basin, USA. Hydrobiologia 494: 5-10.

Welcomme, R. L., 1979. Fisheries ecology of floodplain rivers. Longman Inc., New York. 317p.

Westhoff, J. T., and C. P. Paukert. 2014. Climate change simulations predict altered biotic response in a thermally heterogeneous stream system. PloS One 9: e111438. doi:10.1371/journal.pone.0111438.

Williams, D. D., and H. B. N. Hynes. 1979. The recolonization mechanisms of stream benthos. Oikos 27: 265-272.

Tables

Table 1. Number of sites sampled, fish and aquatic invertebrate richness, and mean wetted width for individual streams sampled from 1988 to 2013 within each of the National Park Service units.

Park[a]	Stream	Number of Sites		Species Richness		Mean Wetted
		Fish	Invertebrate	Fish	Invertebrate	Width (m)
PIPE	Pipestone Creek	*2 or 4	*1 or 2	28	93	4.0
TAPR	11 Small Streams	*13 or 18	0	38	Not Sampled	3.2
HOME	Cub Creek	1	2	16	113	5.0
GWCA	Entire Park	3	3	19	142	4.0
	Carver Creek	1	1	14	112	4.7
	Williams Branch	1	1	11	95	3.6
	Harkin's Branch	1	1	19	87	3.7
WICR	Entire Park	3	3	37	140	11.1
	Wilson's Creek	1	1	35	92	16.5
	Skegg's Branch	1	1	11	87	6.0
	Terrell Creek	1	1	18	94	10.7
OZAR	Entire Park	25	25	76	153	27.7
	Current R.	6	6	75	120	39.4
	Jacks Fork	3	3	46	105	21.5
	Tribs of Current R.	13	13	41	118	7.6
	Tribs of Jacks Fork	3	3	31	79	6.8
BUFF	Entire Park	36	36	62	171	23.4
	Buffalo R.	6	6	56	134	36.8
	Tribs of Buffalo R.	30	30	52	143	9.8

[a] Pipestone National Monument (PIPE), Tallgrass Prairie National Preserve (TAPR), Homestead National Monument of America (HOME), George Washington Carver National Monument (GWCA), Wilsons Creek National Battlefield (WICR), Ozark National Scenic Riverways (OZAR), and Buffalo National River (BUFF)
* Number of sites sampled differed by year.

Table 2. Years in which fish and/or aquatic invertebrates have been sampled at each of seven National Park Service units. Pipestone National Monument (PIPE), Tallgrass Prairie National Preserve (TAPR), Homestead National Monument of America (HOME), George Washington Carver National Monument (GWCA), Wilsons Creek National Battlefield (WICR), Ozark National Scenic Riverways (OZAR), and Buffalo National River (BUFF).

Year	PIPE Fish	PIPE Invert	TAPR Fish	TAPR Invert	HOME Fish	HOME Invert	GWCA Fish	GWCA Invert	WICR Fish	WICR Invert	OZAR Fish	OZAR Invert	BUFF Fish	BUFF Invert
1988-1990										X				
1996		X				X		X		X				
1997		X				X		X		X				
1998		X				X				X				
1999		X				X				X				
2000		X				X				X				
2001	X	X	X			X				X				
2002	X	X	X			X				X				
2003	X	X	X			X				X				
2004	X	X	X		X	X				X				
2005	X	X	X			X		X		X	X	X		X
2006	X	X	X		X	X	X	X	X	X	X	X	X	X
2007	X	X	X			X	X	X	X	X	X	X	X	X
2008	X		X		X	X					X	X	X	X
2009	X		X			X					X	X	X	X
2010	X	X			X	X	X	X	X	X	X	X	X	X
2011	X					X								X
2012						X					X	X		
2013	X						X		X				X	

Table 3. Framework for developing species vulnerability scores (Sievert 2014) that was used to calculate the community vulnerability score. Fish vulnerable to habitat degradation (lithophilic spawners or benthic invertivores), warming temperatures (cool and cold-water species), altered flow regime (equilibrium or periodic life history strategists), or being a poor disperser (members in the *Fundulidae*, *Cottidae*, and *Percidae* family) received a score of 1 in that category and all others received a 0. Aquatic Invertebrates vulnerable to habitat degradation (taxa with < 4.5 HBI value), warming temperatures (stenotherm or cold-cool eurytherm), altered flow regime (species found in only one flow velocity classification of moderate or higher) received a 1 and all others received a 0. Aquatic invertebrate vulnerability did not take into account dispersal ability or persistence.

Species Vulnerability Template		
Scoring	Habitat Degradation	0 or 1
	Warming Temperature	0 or 1
	Altered Flow Regime	0 or 1
	Dispersal Ability	0 or 1
	Species Persistence	1 - (# Years Present/Total # Years Sampled)
Calculating	Environmental Tolerance	Habitat + Temperature + Flow
	Sensitivity	Dispersal Ability + Species Persistence
	Vulnerability	Environmental Tolerance + Sensitivity

Table 4. Location, habitat, and climate variables used to investigate relationships with spatial and temporal variability scores and vulnerability scores for both fish and aquatic invertebrates in seven national park service units distributed across the central United States that were sampled from 1988 to 2013. CV of wetted channel width, mean annual minimum air temperature, and maximum precipitation in a 24-hour period were removed due to collinearity with other variables.

Category	Variable
Location	Region
	Degrees Latitude
	Percent Agriculture in Upstream Watershed
	Percent Urban in Upstream Watershed
Habitat	Wetted Channel Width (m)
	Substrate Size Class (modified Wentworth scale)
	CV of Substrate
	Depth (m)
	CV of Depth
	Velocity (m/s)
	CV of Velocity
Climate	Mean Annual Maximum Air Temperature (°C)
	Mean Annual Air Temperature (°C)
	Total Annual Precipitation (cm)

Table 5. The mean, range, and standard deviation of the Index of Biotic Integrity (Fausch et al. 1984; Dauwalter et al. 2003) and the Hilsenhoff Biotic Index (Hilsenhoff 1987) at seven national park service units distributed across the central United States. Temporal variability is the coefficient of variation (IBI or HBI) of a single site over all the years then averaged across all sites at a park. Spatial variability is the coefficient of variation (IBI or HBI) of all the sites within a single year, and then averaged among years. Pipestone National Monument (PIPE), Tallgrass Prairie National Preserve (TAPR), Homestead National Monument of America (HOME), George Washington Carver National Monument (GWCA), Wilsons Creek National Battlefield (WICR), Ozark National Scenic Riverways (OZAR), and Buffalo National River (BUFF). Since HOME only has one site that is sampled for fish, spatial variability could not be computed.

Park	Index of Biotic Integrity (IBI)				Hilsenhoff Biotic Index (HBI)			
	Mean (SD)	Score Range	Temporal (CV)	Spatial (CV)	Mean (SD)	Score Range	Temporal (CV)	Spatial (CV)
BUFF	80 (7)	58-95	0.07	0.13	4.81 (0.60)	3.47-5.94	0.11	0.13
OZAR	75 (10)	55-95	0.12	0.12	3.76 (0.54)	2.63-4.74	0.10	0.15
TAPR	42 (5)	28-50	0.12	0.16		Not Sampled		
HOME	37 (2)	35-38	0.04		6.33 (0.85)	2.26-7.12	0.13	0.06
PIPE	37 (7)	24-50	0.12	0.20	5.32 (0.87)	4.48-6.79	0.16	0.09
GWCA	55 (11)	46-81	0.17	0.19	4.85 (0.59)	4.02-6.12	0.10	0.10
WICR	62 (14)	45-83	0.10	0.25	5.82 (0.80)	2.45-6.78	0.12	0.10

Table 6. Proportion of fish species and aquatic invertebrate taxa (family level) at each National Park Service unit that were present each year the park was sampled (100%) and more than half the time the park was sampled (> 50%) from 1988 to 2013.

Park	Fish		Invertebrate	
	100%	> 50%	100%	> 50%
Buffalo National River	0.75	0.87	0.43	0.69
George Washington Carver National Monument	0.56	0.88	0.34	0.59
Wilsons Creek National Battlefield	0.49	0.88	0.10	0.41
Ozark National Scenic Riverways	0.43	0.82	0.46	0.74
Tallgrass Prairie National Preserve	0.39	0.64	N/A	N/A
Homestead National Monument of America	0.27	0.73	0.02	0.42
Pipestone National Monument	0.17	0.46	0.16	0.44

Table 7. Proportion of fish and aquatic invertebrate (invert) species/taxa at seven national park service units within the central United States vulnerable to habitat degradation (fish: lithophilic spawners or benthic invertivores; aquatic invertebrates: taxa with < 4.5 HBI value), warming temperatures (fish: cool and cold-water species; aquatic invertebrates: stenotherm or cold-cool eurytherm), altered flow regime (fish: equilibrium or periodic life history strategists; aquatic invertebrates: taxa found in only one flow velocity classification of moderate or higher), or being a poor disperser (fish only: members in the *Fundulidae*, *Cottidae*, and *Percidae* family) as well as overall park vulnerability score. Pipestone National Monument (PIPE), Tallgrass Prairie National Preserve (TAPR), Homestead National Monument of America (HOME), George Washington Carver National Monument (GWCA), Wilsons Creek National Battlefield (WICR), Ozark National Scenic Riverways (OZAR), and Buffalo National River (BUFF).

	Habitat		Temperature		Flow		Dispersal	Vulnerability	
Park	Fish	Invert	Fish	Invert	Fish	Invert	Fish	Fish	Invert
HOME	0.27	0.48	0.27	0.14	0.53	0.90	0.00	1.45	1.52
TAPR	0.42		0.19		0.42		0.11	1.47	
OZAR	0.56	0.74	0.39	0.19	0.50	0.74	0.19	1.85	1.68
BUFF	0.64	0.72	0.47	0.19	0.45	0.74	0.25	1.95	1.65
PIPE	0.50	0.53	0.50	0.18	0.33	0.82	0.17	2.00	1.53
WICR	0.66	0.64	0.51	0.16	0.46	0.80	0.29	2.17	1.60
GWCA	0.69	0.67	0.63	0.25	0.44	0.83	0.38	2.31	1.75
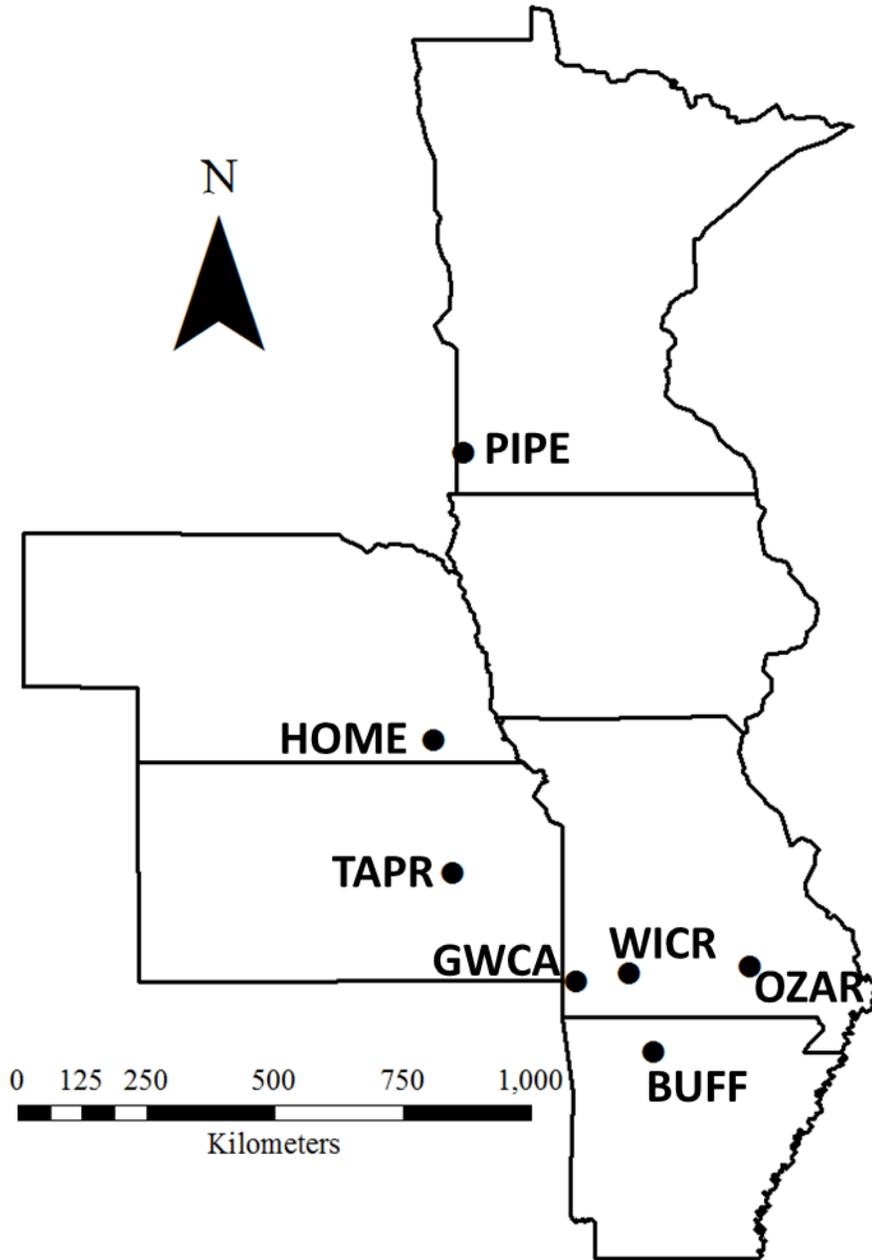
Figure 1. National Park Service units used to investigate the variability and vulnerability of fish and aquatic invertebrate communities from 1988 to 2013 Pipestone National Monument (PIPE), Tallgrass Prairie National Preserve (TAPR), Homestead National Monument of America (HOME), George Washington Carver National Monument (GWCA), Wilsons Creek National Battlefield (WICR), Ozark National Scenic Riverways (OZAR), and Buffalo National River (BUFF).
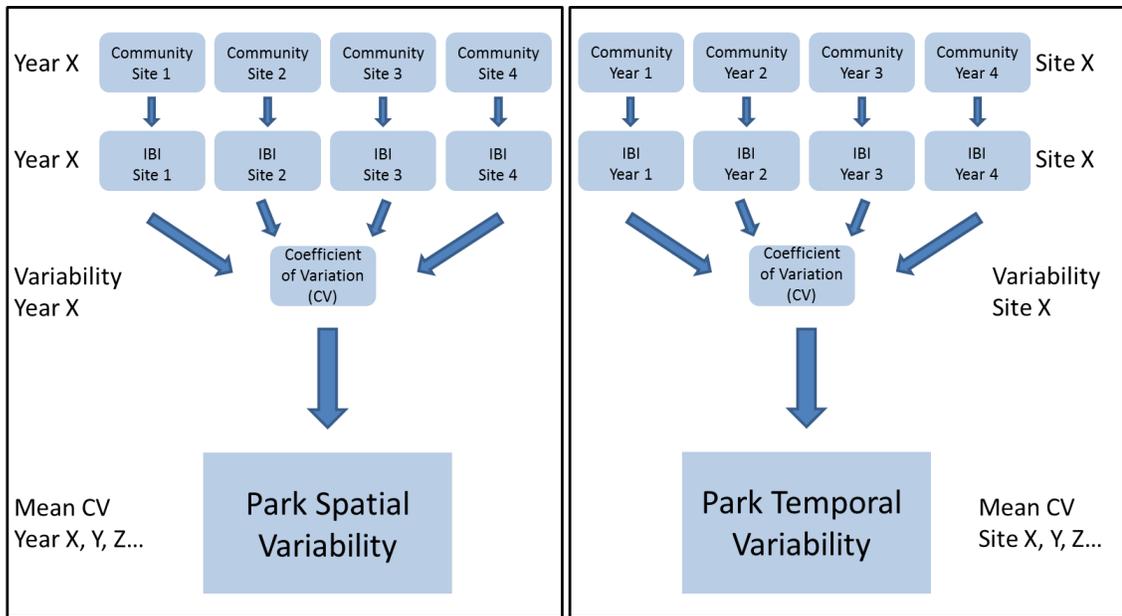
Figure 2. Conceptual model of the process to convert of stream community data (fish in this example) to a Park Spatial Variability score and a Park Temporal Variability score.
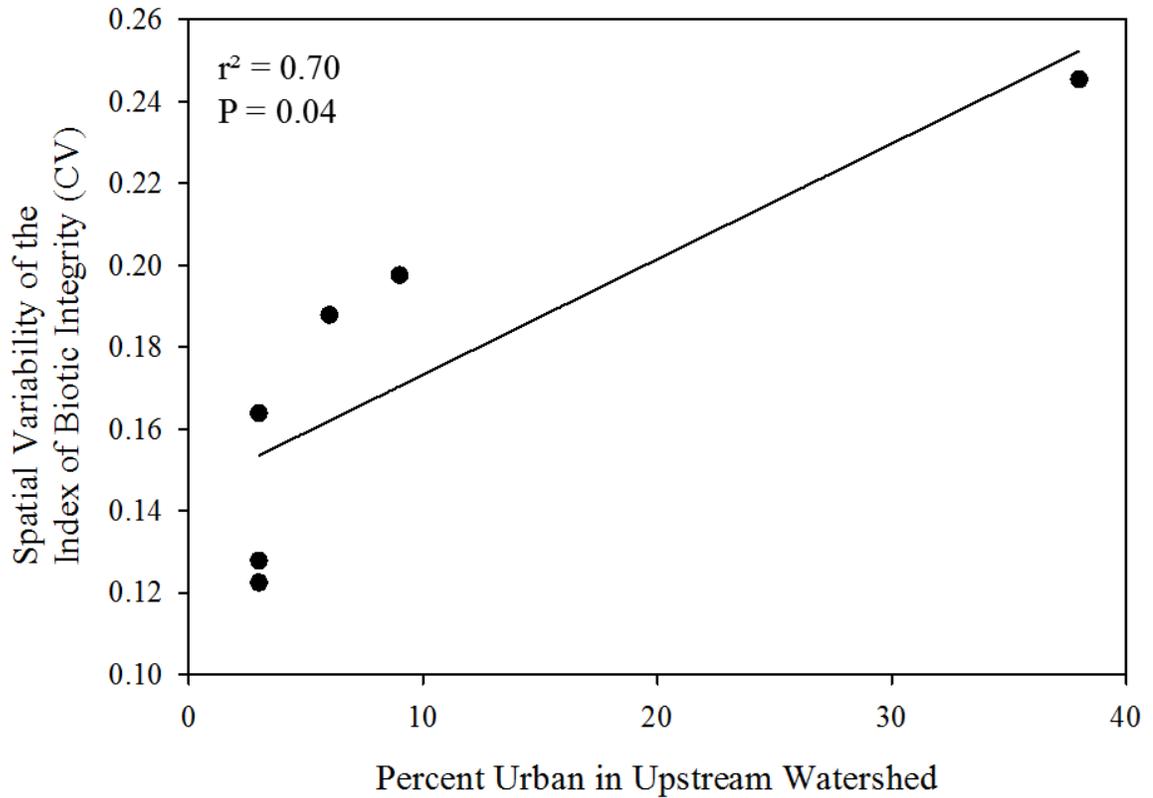
Figure 3. The relationship between percent urban in the upstream watershed and spatial variability in fish communities in Pipestone National Monument, Tallgrass Prairie National Preserve , Homestead National Monument of America, George Washington Carver National Monument, Wilsons Creek National Battlefield, Ozark National Scenic Riverways, and Buffalo National River.
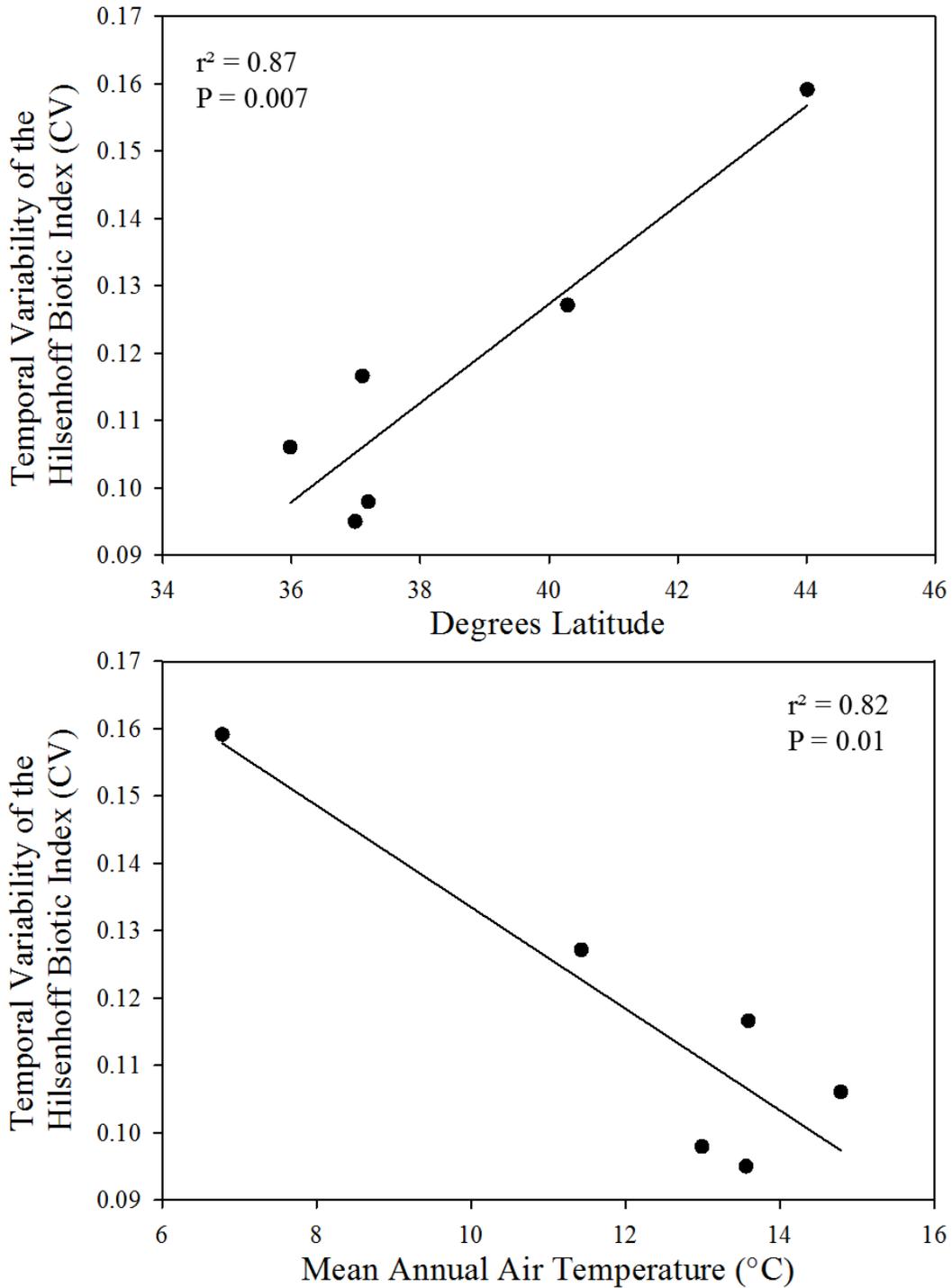
Figure 4. Relationships between latitude or average annual air temperature and temporal variability in aquatic invertebrate communities in Pipestone National Monument, Tallgrass Prairie National Preserve, Homestead National Monument of America, George Washington Carver National Monument, Wilsons Creek National Battlefield, Ozark National Scenic Riverways, and Buffalo National River.
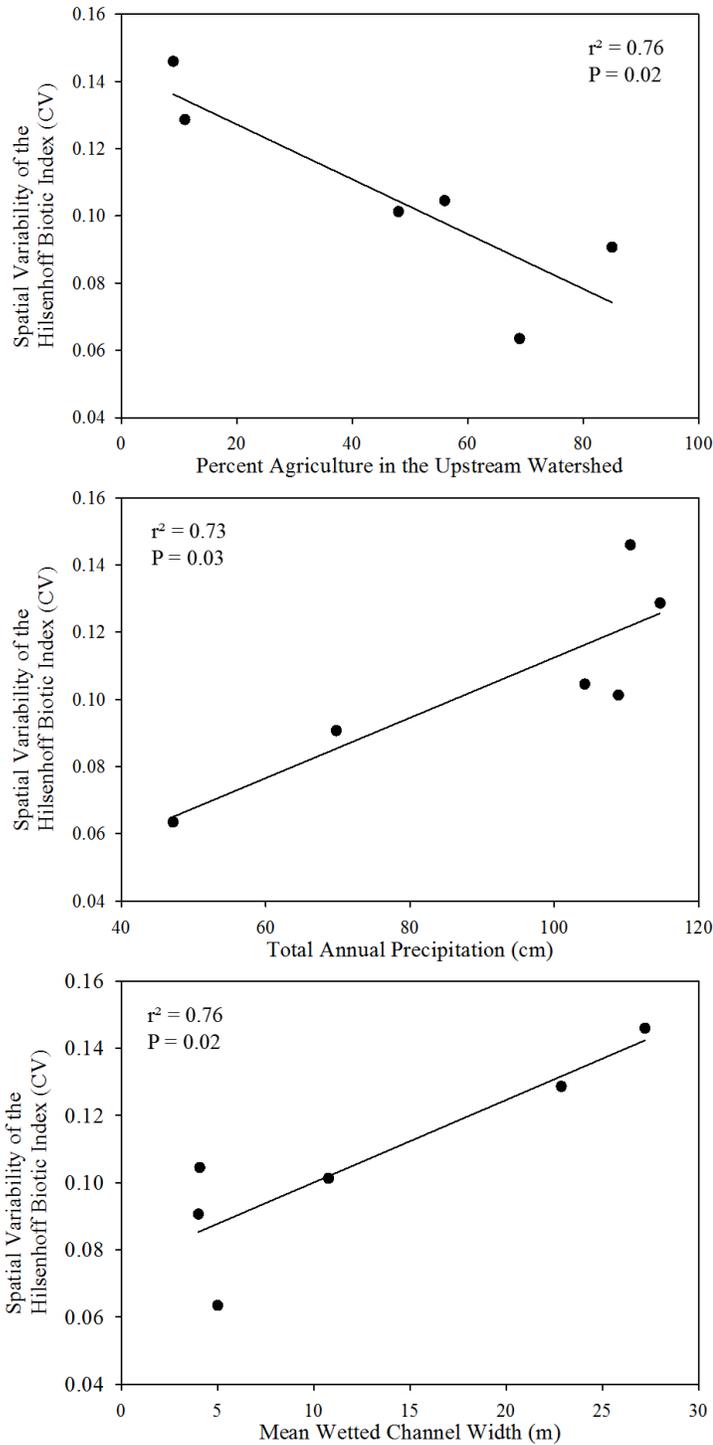
Figure 5. Relationships between percent agriculture in the upstream watershed, wetted channel width, total annual precipitation, and maximum precipitation within 24 hours and spatial variability in aquatic invertebrate communities in Pipestone National Monument, Tallgrass Prairie National Preserve, Homestead National Monument of America, George Washington Carver National Monument, Wilsons Creek National Battlefield, Ozark National Scenic Riverways, and Buffalo National River.
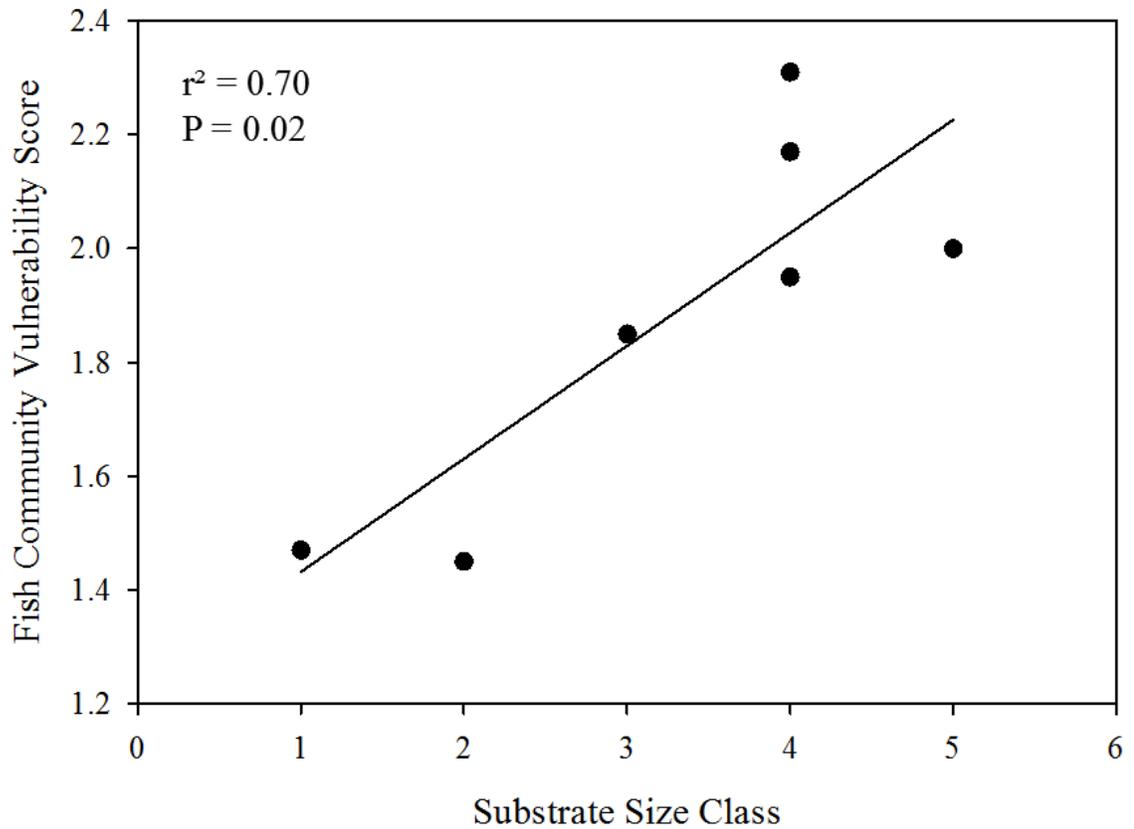
Figure 6. The relationship between dominant substrate size and fish community vulnerability. Substrate classification scheme a follows: 1 = muck, detritus, and silt (< 0.06mm), 2 = sand (0.006-2mm), 3 = pea gravel (2-16mm), 4 = coarse gravel (16-64mm), and 5 = cobble (64-256mm).

Appendix 1. Fish species vulnerable (1) to habitat degradation, temperature increase, altered flow, and poor dispersal ability for species sampled in Pipestone National Monument, Tallgrass Prairie National Preserve, Homestead National Monument of America, George Washington Carver National Monument, Wilsons Creek National Battlefield, Ozark National Scenic Riverways, and Buffalo National River from 2001 to 2013.

Scientific Name	Common Name	Habitat	Temperature	Flow	Dispersal
Ambloplites constellatus	Ozark bass		1	1	
Ambloplites ariommus	Shadow bass			1	
Ameiurus melas	Black bullhead	1		1	
Ameiurus natalis	Yellow bullhead			1	
Anguilla rostrata	American eel			1	
Aphredoderus sayanus	Pirate perch				
Campostoma anomalum	Central stoneroller	1			
Carpiodes carpio	River carpsucker			1	
Catostomus commersoni	White sucker		1	1	
Cottus carolinae	Banded sculpin	1		1	1
Cottus hypselurus	Ozark sculpin	1	1	1	1
Culaea inconstans	Brook stickleback		1		
Cyprinella lutrensis	Red shiner				
Cyprinella galactura	Whitetail shiner				
Cyprinus carpio	Common carp			1	
Dorosoma cepedianum	Gizzard shad		1	1	
Erimystax harryi	Ozark chub	1			
Erimyzon oblongus	Creek chubsucker	1		1	
Esox niger	Chain pickerel			1	
Esox lucius	Northern pike	1	1	1	
Etheostoma cragini	Arkansas darter	1	1		1
Etheostoma euzonum	Arkansas saddled darter	1	1		1
Etheostoma zonale	Banded darter	1			1
Etheostoma uniporum	Current River darter	1			1
Etheostoma euzonum	Current River saddled darter	1	1		1
Etheostoma flabellare	Fantail darter	1	1		1
Etheostoma blennioides	Greenside darter	1			1
Etheostoma exile	Iowa darter	1	1		1
Etheostoma nigrum	Johnny darter	1			1
Etheostoma spectabile	Orangethroat darter	1	1		1
Etheostoma caeruleum	Rainbow darter	1	1		1
Etheostoma punctulatum	Stippled darter	1	1		1
Etheostoma juliae	Yoke darter	1	1		1
Fundulus olivaceus	Blackspotted topminnow	1	1		1

Fundulus notatus	Blackstripe topminnow				1
Fundulus catenatus	Northern studfish				1
Gambusia affinis	Mosquitofish				
Hybognathus hankinsoni	Brassy minnow		1		
Hypentelium nigricans	Northern hog sucker	1	1	1	
Ichthyomyzon castaneus	Chestnut lamprey	1	1	1	
Ictalurus punctatus	Channel catfish			1	
Ictiobus cyprinellus	Bigmouth buffalo			1	
Labidesthes sicculus	Brook silverside				
Lampetra appendix	American Brook lamprey	1	1		
Lampetra aepyptera	Least Brook lamprey	1	1		
Lepisosteus osseus	Longnose gar			1	
Lepomis macrochirus	Bluegill			1	
Lepomis cyanellus	Green sunfish			1	
Lepomis megalotis	Longear sunfish		1	1	
Lepomis humilis	Orangespotted sunfish	1			
Lepomis microlophus	Redear sunfish			1	
Lepomis miniatus	Redspotted sunfish			1	
Lepomis gulosus	Warmouth			1	
Luxilus zonatus	Bleeding shiner	1			
Luxilus cardinalis	Cardinal shiner	1			
Luxilus cornutus	Common shiner	1			
Luxilus pilsbryi	Duskystripe shiner	1	1		
Luxilus chrysocephalus	Striped shiner	1			
Lythrurus umbratilis	Redfin shiner				
Micropterus salmoides	Largemouth bass			1	
Micropterus dolomieu	Smallmouth bass			1	
Micropterus punctulatus	Spotted bass			1	
Minytrema melanops	Spotted sucker	1		1	
Moxostoma duquesnei	Black redhorse	1		1	
Moxostoma erythrurum	Golden redhorse	1		1	
Moxostoma carinatum	River redhorse	1		1	
Moxostoma macrolepidotum	Shorthead redhorse		1	1	
Nocomis biguttatus	Hornyhead chub	1	1	1	
Notemigonus crysoleucas	Golden shiner			1	
Notropis amblops	Bigeye chub	1	1		
Notropis boops	Bigeye shiner	1	1		
Notropis dorsalis	Bigmouth shiner				
Notropis percobromus	Carmine shiner				
Notropis atherinoides	Emerald shiner		1		
Notropis volucellus	Mimic shiner				

57

Species	Common name				
Notropis nubilus	Ozark minnow	1	1		
Notropis ozarcanus	Ozark shiner	1			
Notropis stramineus	Sand shiner				
Notropis telescopus	Telescope shiner	1	1		
Notropis topeka	Topeka shiner	1	1		
Notropis greenei	Wedgespot shiner	1			
Noturus maydeni	Black River madtom			1	
Noturus flavater	Checkered madtom	1		1	
Noturus albater	Ozark madtom	1		1	
Noturus exilis	Slender madtom	1	1	1	
Noturus flavus	Stonecat	1	1	1	
Oncorhynchus mykiss	Rainbow trout	1	1	1	
Perca flavescens	Yellow perch		1	1	1
Percina maculata	Blackside darter	1	1		1
Percina evides	Gilt darter	1	1		1
Percina caprodes	Logperch	1			1
Phenacobius mirabilis	Suckermouth minnow	1			
Phoxinus erythrogaster	Southern redbelly dace		1		
Pimephales notatus	Bluntnose minnow		1		
Pimephales promelas	Fathead minnow				
Pimephales tenellus	Slim minnow	1			
Pylodictis olivaris	Flathead catfish			1	
Rhinichthys atratulus	Blacknose dace	1	1		
Semotilus atromaculatus	Creek chub	1	1		

Appendix 2. Ephemeroptera, Plecoptera, and Trichoptera taxa vulnerable (1) to habitat degradation, temperature increase, and altered flow for taxa sampled in Pipestone National Monument, Homestead National Monument of America, George Washington Carver National Monument, Wilsons Creek National Battlefield, Ozark National Scenic Riverways, and Buffalo National River from 1988 to 2012.

Order	Family	Genus	Habitat	Temperature	Flow
Ephemeroptera	*Ameletidae*	*Ameletus*			1
Ephemeroptera	*Baetidae*	*Baetis*		1	1
Ephemeroptera	*Baetidae*	*Paracloeodes*			1
Ephemeroptera	*Baetidae*	*Procloeon*			1
Ephemeroptera	*Baetidae*	*Pseudocloeon*	1		1
Ephemeroptera	*Baetiscidae*	*Baetisca*	1		1
Ephemeroptera	*Caenidae*	*Caenis*			1
Ephemeroptera	*Ephemeridae*	*Ephemera*	1		1
Ephemeroptera	*Ephemeridae*	*Hexagenia*			
Ephemeroptera	*Heptageniidae*	*Epeorus*	1	1	1
Ephemeroptera	*Heptageniidae*	*Heptagenia*	1	1	1
Ephemeroptera	*Heptageniidae*	*Rhithrogena*	1	1	1
Ephemeroptera	*Heptageniidae*	*Stenacron*			1
Ephemeroptera	*Heptageniidae*	*Stenonema*	1		1
Ephemeroptera	*Isonychiidae*	*Isonychia*	1		1
Ephemeroptera	*Leptophlebiidae*	*Habrophlebiodes*			
Ephemeroptera	*Leptophlebiidae*	*Leptophlebia*			
Ephemeroptera	*Leptophlebiidae*	*Paraleptophlebia*	1	1	1
Ephemeroptera	*Siphlonuridae*	*Siphlonurus*	1	1	1
Ephemeroptera	*Tricorythidae*	*Tricorythodes*			
Plecoptera	*Capniidae*	*Allocapnia*	1	1	1
Plecoptera	*Capniidae*	*Paracapnia*	1		1
Plecoptera	*Chloroperlidae*	*Alloperla*	1	1	1
Plecoptera	*Leuctridae*	*Leuctra*	1		1
Plecoptera	*Leuctridae*	*Zealeuctra*	1		1
Plecoptera	*Nemouridae*	*Amphinemura*	1	1	1
Plecoptera	*Nemouridae*	*Prostoia*			
Plecoptera	*Perlidae*	*Acroneuria*	1		
Plecoptera	*Perlidae*	*Neoperla*	1		
Plecoptera	*Perlidae*	*Paragnetina*	1		
Plecoptera	*Perlidae*	*Perlesta*	1		
Plecoptera	*Perlidae*	*Perlinella*	1		
Plecoptera	*Perlodidae*	*Helopicus*			
Plecoptera	*Perlodidae*	*Hydroperla*	1		1
Plecoptera	*Perlodidae*	*Isoperla*	1	1	1
Plecoptera	*Pteronarcyidae*	*Pteronarcys*	1	1	
Plecoptera	*Taeniopterygidae*	*Strophopteryx*	1		
Plecoptera	*Taeniopterygidae*	*Taeniopteryx*	1		1
Trichoptera	*Brachycentridae*	*Brachycentrus*	1		1
Trichoptera	*Brachycentridae*	*Micrasema*	1		1
Trichoptera	*Glossosomatidae*	*Agapetus*	1		1
Trichoptera	*Glossosomatidae*	*Glossosoma*	1		1
Trichoptera	*Glossosomatidae*	*Protoptila*	1		1
Trichoptera	*Helicopsychidae*	*Helicopsyche*	1		1

Trichoptera	*Hydropsychidae*	*Ceratopsyche*	1	1
Trichoptera	*Hydropsychidae*	*Cheumatopsyche*		1
Trichoptera	*Hydropsychidae*	*Diplectrona*	1	1
Trichoptera	*Hydropsychidae*	*Hydropsyche*	1	1
Trichoptera	*Hydropsychidae*	*Potamyia*		1
Trichoptera	*Hydroptilidae*	*Hydroptila*		1
Trichoptera	*Hydroptilidae*	*Mayatrichia*		1
Trichoptera	*Hydroptilidae*	*Ochrotrichia*		1
Trichoptera	*Lepidostomatidae*	*Lepidostoma*	1	1
Trichoptera	*Leptoceridae*	*Ceraclea*	1	1
Trichoptera	*Leptoceridae*	*Nectopsyche*	1	1

**Relationships among biotic indices and their responses to ecological gradients
in least-disturbed Ozark streams**

Jacob M. Schwoerer, Craig P. Paukert, and Hope R. Dodd

Abstract

Biotic indices are commonly used to convey lotic ecosystem health by using a
single quantitative score that can be grouped into a single health category. Many indices
have been developed for fish and aquatic invertebrates and thus allows for multiple
measures of stream condition. We assessed the consistency of three fish biotic indices
(Index of Biotic Integrity, percent intolerant individuals to habitat alteration, and species
richness), and three aquatic invertebrate indices (Hilsenhoff Biotic Index,
Ephemeroptera, *Plecoptera*, *Trichoptera* richness, and taxa richness) to determine if
indices using the same or different faunal group were related. We also determined
whether each of the biotic indices were related to water quality, in-stream habitat, and/or
land cover metrics. We used data from repeat sampling from 2005 to 2013 in two
relatively least-disturbed river systems within two National Park Service units in the
Ozarks region of Missouri (Ozark National Scenic Riverways (ONSR); N = 24) and
Arkansas (Buffalo National River (BNR); N = 31). Spearman correlations (ρ) revealed
highly variable relationships among indices (ρ = -0.02 to 0.87). Indices at both locations
were generally uncorrelated (12 of 15 pairwise comparisons) and correlated indices
differed by river system. Within faunal group indices were more related in the BNR (ρ =
-0.28 to 0.87) while richness indices were more related in the ONSR (ρ = 0.60 to 0.81).
Generally, we found that water temperature was positively related to fish index scores but
negatively related to aquatic invertebrate index scores in the BNR. Indices in the ONSR

were positively associated with increased upstream watershed area, possibly due to the stronger influence of spring in this watershed, which buffers the high and low water temperatures. Streams with smaller watershed areas appeared to have inherently lower biotic index scores, indicating the need to interpret integrity differently along stream size gradients. In conclusion, we found inconsistencies in which indices were related within two similar and relatively least-disturbed river systems. Stream size and thermal regimes influence these indices in relatively undisturbed systems suggesting decision makers may need to consider these factors before comparing sites or rivers with different water temperatures or sizes.

Introduction

Community composition of stream fish and aquatic invertebrates is a function of channel characteristics, chemical variables, flow conditions, energy sources, and biotic interactions (Karr 1981; Karr et al. 1986; Karr 1991), which makes them ideal candidates for measuring the overall condition and health of lotic systems. Biotic indices measure stream health using community traits, such as faunal diversity, species dominance and abundance, trophic structure, tolerance to degraded or polluted conditions, and individual condition (e.g. parasites, lesions, eroded fins) to score the integrity of a community (Karr et al. 1981; Hughes et al. 1998). Biotic indices calculated from newly sampled sites can be compared to reference or least-disturbed sites within an ecoregion (Hughes et al. 1998; Lammert and Allan 1999), allowing community health to be characterized along a gradient of impairment and relative conclusions to be formed. However, since sites may vary in community composition over time and space even within least-disturbed streams (Fausch et al. 1984; Fisher and Paukert 2008), it can lead to fuzzy interpretations of

ecological integrity. To increase consistency of interpretation, indices have been tailored for specific stream types (e.g. stream size, hydrologic regime, thermal regime, and region), which has resulted in the creation of many different indices, including ones specifically for fish or aquatic invertebrates (Hughes et al. 1998). However, limited information exists on the consistency among multiple indices to assess biotic integrity, even in areas with low levels of degradation.

The use of fish to assess stream health is a common practice among fisheries scientists. Some benefits of using fish as bio-indicators is that the general public recognizes their economic and recreational importance, responses in multiple trophic levels to be deduced since fish species incorporate multiple trophic levels (e.g. herbivore, insectivore, piscivore), and they can be used in a variety of regions due to their applicability and availability of reference data (Karr 1981; Berkman et al. 1986; Lammert and Allan 1999). The shortcomings of using fish for bio-assessments are that fish are mobile, which can enable avoidance of environmental stressors (Karr 1981). Additionally, species diversity may differ between drainages or sites within a stream for reasons other than stream health, which could confound interpretation of indices (Berkman et al. 1986). Multiple metrics of stream fish health have been developed and range from fairly straightforward metrics such as total species richness (Fitzpatrick et al. 2001) to the multimetric Index of Biotic Integrity (IBI; Table 1). Various IBIs have been developed for fish based on a number of factors including thermal class, stream size, and geographic ecoregions (Bramblett and Fausch 1991; Lyons 1992; Lyons et al. 1996; Hughes et al. 1998; Dauwalter et al 2003). However, these indices have a similar framework and incorporate metrics such as species richness, dominance, abundance,

trophic structure, tolerance to degraded and polluted conditions, individual health, and reproductive and functional feeding guilds (Karr et al. 1981; Berkman and Rabeni 1987; Hughes et al. 1998).

Aquatic invertebrate biotic indices are commonly used to determine health of lotic environments because aquatic invertebrates are relatively sedentary, allowing for the detection of localized disturbance, are shorter lived than fish which allows for the effects of pollution to be examined, and are typically found in complex communities with a wide variety of trophic and taxonomic groups (Berkman et al. 1986; Lammert and Allan 1999). However, inconsistencies can arise since similar microhabitats and substrates need to be available at each site for suitable comparisons, drift species may be found in non-characteristic locations, and the collection and identification of specimens is often time, labor, and cost intensive (Berkman et al. 1986; Lammert and Allan 1999). Although multiple aquatic invertebrate biotic indices have been established, three have become standards for habitat condition assessments: total taxa richness, *Ephemeroptera*, *Plecoptera*, and *Trichoptera* taxa richness (EPT), and the Hilsenhoff Biotic Index (HBI; Hilsenhoff 1987; Table 1). Aquatic invertebrate taxa richness is an indicator of water quality, habitat diversity, and habitat suitability (Lenat 1988; McElravy et al. 1989; Sponseller et al. 2001). The EPT index uses *Ephemeroptera*, *Plecoptera*, and *Trichoptera* taxa as an indicator of water quality conditions because these taxa are considered sensitive to pollution and human disturbance (Lenat 1988; Sponseller et al. 2001; Nerbonne and Vondracek 2001). The HBI uses pollution tolerance values and relative abundance of taxa to calculate a community tolerance score used to measure community impairment to organic pollution (Hilsenhoff 1987). In theory, index interpretations in

least-disturbed stream systems should score marks of high integrity. Although there are differences among biotic indices, they collectively describe some aspect of stream health which allows managers to better understand and protect threatened areas as well as identify and prioritize degraded areas for restoration. However, few studies have determined the relatedness of multiple indices in least-disturbed systems and their ability to portray similar stream health interpretations.

Although both fish and aquatic invertebrate indices may be used within the same stream reach to define overall stream health, they may reflect different environmental characteristics at various spatial scales. Aquatic invertebrates are more affected by stream segment and reach scale habitat and substrate (Richards and Host 1993, 1994; Carter et al. 1996; Lammert and Allan 1999; Fitzpatrick et al 2001), while fish respond more to land cover at the watershed and stream segment scales (Carter et al 1996; Richards et al. 1996; Roth et al 1996; Lyons et al. 2000). Lammert and Allan (1999) found that three aquatic invertebrate indices were more correlated with each other than with the fish IBI. Therefore, indices based on aquatic invertebrates and fish may provide different indications of stream condition (Lammert and Allan 1999) and decision makers may need to determine which index best meets their objectives. However, managers often make decisions based on limited data due to fiscal and logistical constraints (Berkman et al. 1986) and therefore need to understand what environmental characteristics or spatial scales influence index interpretations and whether other indices portray similar interpretations so that redundant measures of integrity may be reduced. Because decisions about management and prioritization are often based on index scores (USEPA 2011), understanding relationships among indices, and their strengths and limitations for

describing stream and watershed conditions will aid interpretation. Investigating the differences between index interpretations in a least-disturbed system could shed light on the strengths and limitations of individual indices and help managers more accurately interpret scores by understanding what indices under what conditions are comparable.

We investigated the consistency of stream health interpretations based on six biotic indices in two federally protected and managed Ozark river systems: the Ozark National Scenic Riverways (ONSR) in southeastern Missouri, USA, and the Buffalo National River (BNR) in northcentral Arkansas, USA. Three fish index scores (IBI; Dauwalter et al. 2003, species richness, and percent intolerant individuals to habitat alteration) and three aquatic invertebrate index scores (taxa richness, EPT taxa richness, and the HBI; Hilsenhoff 1987) were compared within and across faunal groups (Table 1). Since fish and aquatic invertebrates occupy different spatial and temporal scales and respond and recover from environmental perturbations at different rates and by different modes (Berkman et al. 1986; Matthews 1986; McElravy et al. 1989; Fausch and Bramblett 1991; Boulton et al. 1992; Lammert and Allan 1999), we hypothesized that index interpretations within faunal groups would be more consistent than across faunal groups. We hypothesized that in these least-disturbed stream systems, index scores would be influenced by environmental gradients found naturally within a heterogeneous stream environment. Since aquatic invertebrates occupy smaller spatial and temporal scales, we hypothesized that in-stream physical habitat characteristics would influence these index scores, whereas because fish occupy larger spatial and temporal scales to fulfill life history requirements, broad watershed land cover characteristics will influence these index scores (Plafkin et al. 1989). Because these environmental gradients exist in

federally protected and least-disturbed river systems, it is important to assess how index scores vary along these gradients in order to guide more accurate stream health interpretations.

Methods

Study Area

The ONSR in southeastern Missouri and the BNR in northwestern Arkansas are two National Park Service (NPS) units located in the Ozark Highlands ecoregion and are among the few free flowing rivers remaining in the conterminous United States (Figure 1). Both watersheds are located in the Springfield and Salem Plateaus that are characterized by limestone and dolomite geology, which produce a karst topography that includes sinkholes, caves, springs, and gaining/losing reaches on the river (Dodd 2009; 2013). The upper portion of BNR is located in the Boston Mountains which consists of sandstone and shale geology. The ONSR is a 327 km^2 NPS Unit that encompasses 217 km of the Jacks Fork and Current rivers in Missouri. The Current River, which originates at Montauk Spring, enters the ONSR as a 4th order stream and reaches 6th order before exiting, while the Jacks Fork is a 5th order stream that joins the Current River within NPS boundaries. The BNR is a 382 km^2 NPS Unit that encompasses 135 km of the Buffalo River in Arkansas, which enters the NPS boundary as a 4th order stream and reaches 5th order before exiting the park. There are approximately 122 and 74 fish species identified as likely present in the OSNR and BNR, respectively (NPSpecies; https://irma.nps.gov/App/ Species/Search). Site selection for sampling fish and aquatic invertebrate communities at each park was accomplished using a Generalized Random

Tessellation Stratified method that generates a spatially-balanced and independent random sample of sites (Stevens and Olsen 2004). Nine mainstem river sites within ONSR (six on the Current River and three on the Jacks Fork), and 15 tributary streams (one site in each tributary) were sampled (Figure 1). At BNR, six mainstem river and 25 tributaries sites were sampled. The sampling reach length was determined using 20 times the mean wetted channel width for each site with a minimum length of 150 m and a maximum of 1000 m.

Data Collection

At ONSR, the nine mainstem sites were sampled annually from 2005-2009 and in 2012 for both fish and aquatic invertebrates. The 15 tributaries were sampled on a rotating panel with 13 tributaries sampled once during 2006-2010 and two additional tributaries were re-sampled in 2012. Sites on the ONSR were sampled for fish in late September to late October in all years sampled except for 2005 when sites were sampled from November to early December. Aquatic invertebrate data was collected between November and January each year sampling was conducted at ONSR. The six mainstem sites at BNR were sampled annually for aquatic invertebrates from 2005-2009 and in 2011 and sampled for fish community data from 2006-2010 and in 2013. The 25 tributaries within BNR were sampled for both fish and aquatic invertebrates on a rotating panel with 24 tributaries sampled once during 2006-2010. One additional tributary was sampled in 2011 for aquatic invertebrates and in 2013 for fish. In the BNR, fish community sampling occurred in May and June while aquatic invertebrate community sampling occurred from November to February for each of the years they were sampled.

Fish, aquatic invertebrate, and corresponding habitat data from the ONSR and BNR were collected by the NPS Heartland Inventory and Monitoring Network (HTLN) staff using standardized protocols (Bowles 2007; Petersen et al. 2008). Fish were sampled at mainstem sites using pulsed DC boat electrofishing supplemented by towed barge or backpack electrofishing in shallower habitats. Most tributaries were too shallow for a boat and were sampled with towed barge or backpack electrofishing gear. Fish were identified to species and enumerated in the field. At each site and sample year, water quality (dissolved oxygen (mg/l), specific conductance (μS/cm), pH, and water temperature (°C)) were measured before and after fish community data was collected. In-stream habitat (water depth, velocity, dominant substrate size, substrate embeddedness, and wetted channel width) was collected at three equally spaced points (left, center, right) along 11 equally spaced cross-sectional transects. At each site, segment sinuosity, upstream drainage density, upstream watershed area, and land cover was calculated using Arcmap 10.2, the National Hydrography Dataset, and the 2006 National Land Cover Database (Fry et al. 2011). Land cover (percent agriculture and percent urban/developed) was calculated for the upstream watershed beginning at the downstream boundary at each site (watershed scale) and for a 100 meter buffer surrounding each of the sampled reaches (local or reach scale). Segment sinuosity measurement was the ratio of stream distance to linear distance between the nearest upstream and downstream confluences for each site, and drainage density was the ratio of river kilometers to upstream watershed area (Rosgen 1994).

Aquatic invertebrates were collected using a Slack Surber sampler (Bowles et al. 2007). Collections were taken at three replicate locations equidistant across a riffle within

three consecutive riffles starting from the downstream boundary of each site (nine surber

samples per site). Samples were brought back to the lab for genus level identification

(when possible) and enumeration. Water quality parameters (dissolved oxygen (mg/l),

specific conductance (µS/cm), pH, water temperature (°C)) were collected at each riffle.

Physical habitat, including dominant substrate size, substrate embeddedness, water

velocity, water depth, and wetted channel width were collected at each sample location

within each riffle (nine per site).

Biotic Index Score Calculations

Each of the six biotic indices (three fish and three aquatic invertebrate; Table 1)

were calculated for each site and for each year sampled. The fish IBI used had been

developed for wadeable streams in the Ozark Highland region of Arkansas and consisted

of seven metrics (Dauwalter et al. 2003; Table 2). This index incorporated single-pass

electrofishing samples collected during base flow conditions; similar to the sampling

methods used by HTLN staff. Percent intolerant fish individuals were calculated as the

percent of the individuals sampled that were classified as species intolerant to habitat

alterations (Barbour et al. 1999; Dauwalter et al. 2003), and species richness was the total

number of fish species collected. Likewise, aquatic invertebrate taxa richness was the

total number of unique taxa collected in each sample and EPT is the total number of

genera within the orders *Ephemeroptera*, *Plecoptera*, and *Trichoptera*. We used the

genus level HBI with taxa tolerance values developed by Hilsenhoff (1987) and Lenat

(1993) to calculate a weighted community tolerance value based on proportion of taxa in

each group. A mean annual site score was calculated for each aquatic invertebrate index

from the nine samples (three replicates within each of three riffles). Since mainstem sites

and some tributaries were sampled multiple years, the overall site score for each of the six indices was calculated as the mean over the multiple years, which eliminated the temporal dependence among samples.

Data Analysis

To investigate consistency among biotic indices (Table 1), Spearman's Rank Correlation was used to make pairwise comparisons between each of the index scores. Correlations were performed separately between the ONSR and the BNR to minimize spatial dependence from geographic location of the parks. The significance level was adjusted using a Bonferroni correction ($\alpha = 0.003$) due to the 15 pairwise comparisons for each park. This analysis allowed us to determine how closely each index was correlated with another and to determine if indices within a faunal group were more related to themselves than across faunal groups.

An information-theoretic approach, using Akaike's Information Criterion adjusted for small sample size (AICc), in a linear model framework, was used to determine which environmental factors were most related to each of the six indices (Table 1). Each of the six biotic indices were ran against a suite of *a priori* models encompassing in-stream physical habitat, water quality, and land cover (both local and watershed scale; Table 3). Models included a watershed scale and local scale model that included variables represented within each respective scale (land cover and stream habitat characteristics). Four models were developed to represent the function of these biotic indices (ecosystem health, habitat complexity, water quality, and organic pollution; Table 1). Since in-stream habitat data was collected specifically for fish and aquatic invertebrates, we used each faunal groups respective habitat values in the analysis. In addition, analysis was

performed separately between the ONSR and the BNR to reduce spatial dependence related to geographic location and to investigate potential differences in index interpretation categories. For each analysis, the model with the lowest AICc value and those within two AICc values from the lowest were considered top predictive models (Burnham and Anderson 2002).

Results

A total of 60 fish species and 241 aquatic invertebrate taxa were collected at the BNR and 70 fish species and 218 aquatic invertebrate taxa were collected at the ONSR. The range and mean for each biotic index score were similar between NPS units (Table 4). In the BNR, 87% of sites were classified as good or reference condition using the IBI (IBI > 60) and 51% of the sites were classified as good, very good, or excellent using the HBI (HBI < 5.00). In the ONSR, 100% of sites were classified as good or reference using the IBI and 96% of the sites were classified as good, very good, or excellent using the HBI.

The average upstream percent agriculture was 11% for the BNR (range 0 to 30%) and 10% for the ONSR (range 0 to 22%), whereas the average percent urban in the upstream watershed was 3% (range 1 to 5%) for both the BNR and ONSR (Table 5). Based on an assessment of stream habitat condition for all streams in Missouri (Sowa et al. 2007), these percentages are low enough to classify every sample site in these two NPS units as least-impacted from agriculture or urban development, with exception to two sites in the BNR which were within the slightly-impacted category. Mainstem river sites within the ONSR had greater percent agriculture within their upstream watersheds than tributary sites ($P < 0.001$) but this result was not found in the BNR ($P = 0.73$).

Watershed area in the BNR ranged from 4 to 2988 km^2 while watershed area in the ONSR ranged from 7 to 4664 km^2 (Table 5). The large differences between minimum and maximum watershed area in each of the river systems were due to differences in tributary sites (ONSR: n=15, BNR: n=25) and mainstem river sites (ONSR: n=9, BNR: n=6). Water temperature in the BNR ranged from 11.4°C to 27.6°C during fish collections and 2.9°C to 13.9 °C during aquatic invertebrate collections. Water temperature in the ONSR ranged from 6.0°C to 20.3°C during fish collections and 7.2°C to 14.0°C while collecting aquatic invertebrates (Table 5).

Few of the biotic metrics were significantly correlated with each other, regardless of faunal group. Only three of the 15 pairwise comparisons were significantly related (P < 0.003 after Bonferroni correction) within each of the NPS units (Table 6). All significant relationships within the BNR were between indices within the same faunal group with fish IBI and species richness, aquatic invertebrate EPT and HBI, and aquatic invertebrate taxa richness and EPT having strong correlations (Table 6; Figure 2). Two of the three significant relationships within the ONSR were between a fish and an aquatic invertebrate index (fish species richness and EPT richness; fish species richness and aquatic invertebrate taxa richness). For both NPS units, aquatic invertebrate taxa richness and EPT were significantly correlated. The only index that was not significantly related to any other indices at either of the NPS units was percent intolerant fish individuals.

The watershed scale and organic pollution model were found to be most predictive for fish and aquatic invertebrate biotic indices within the ONSR, whereas in the BNR, multiple models, which included water quality, watershed scale, organic pollution, and habitat complexity were most predictive (Table 7). Only two indices were

related to models mimicking what they were created to describe; EPT describing water quality and fish species richness describing habitat complexity (Table 1; Table 7). For all three aquatic invertebrate indices in ONSR (EPT, HBI, and taxa richness), the watershed model was most predictive with larger watershed area and more agriculture in the upstream watershed (for only EPT and taxa richness) linked to higher biotic integrity (Figure 3; 4). In contrast, the aquatic invertebrate indices in the BNR were more closely tied to the water quality model, with lower water temperatures linked to higher biotic integrity using EPT and taxa richness indices (Figure 4) and greater drainage density was significantly linked to lower integrity using the HBI.

For the IBI and fish species richness indices in the ONSR, the watershed model was the best predictive model (Table 7). Organic pollution model was also a good predictive model for IBI. However, the best model for percent intolerant fish species was organic pollution but explained no variation after being adjusted for multiple parameters in the model. Fish species richness increased with watershed area and agriculture in the upstream watershed, however percent agriculture averaged only 10% among sites and was at most 22% (Table 5; Figure 3). In the BNR, the IBI was best predicted by the water quality model, with warmer water temperature linked to higher biotic integrity (Table 7; Figure 4). The percent intolerant fish individuals was best predicted by the organic pollution as well as the water quality model, with the only variable significantly describing increased percent of intolerant fish being lower water temperature. Fish species richness was best predicted by the habitat complexity model, with increasing depth and width to depth ratio describing greater species richness (Table 7).

Within the ONSR, the EPT, aquatic invertebrate taxa richness, and fish species richness indices, which were all significantly correlated to one another (Table 6), also responded to the same watershed model (Table 7), suggesting that the percent agriculture in the watershed and watershed area were associated to each index (Figure 3; 4). In the BNR, EPT and aquatic invertebrate taxa richness indices were both highly correlated (Table 6), and both indices responded to the water quality model by having lower biological integrity with warmer water temperatures (Table 7; Figure 4). The strong relationship between EPT and HBI (Table 6) within BNR were both predicted by the water quality model (Table 7), with higher HBI (lower integrity) relating to lower pH and higher EPT related to lower temperature water. The only indices that were significantly correlated but were best described by different models or variables were IBI and fish species richness (Table 6), which were described by the water quality and habitat complexity models, respectively (Table 7).

Discussion

Our study is the first to our knowledge that compares multiple fish indices with multiple aquatic invertebrate indices within and across spatially distinct yet very similar least-disturbed watersheds. Our results demonstrate that biotic indices were relatively unrelated in undisturbed streams and rivers. Previous studies have found that same faunal group indices tend to portray more similar interpretations of stream integrity than when comparing indices across faunal groups (Lammert and Allan 1999). This partially supports our findings from the BNR, where although only three of the 15 index comparisons were related, these three comparisons were each within the same faunal group. The life histories of these faunal groups may influence how each react and recover

to environmental perturbations, natural or anthropogenic. Fish are more mobile and can often recover quicker than aquatic invertebrates (Matthews 1986; McElravy et al. 1989; Fausch and Bramblett 1991; Boulton et al. 1992). The rate at which in-stream habitat, fish communities, and aquatic invertebrate communities return to a natural state after a disturbance differs (Matthew 1986; McElravy et al. 1989; Fausch and Bramblett 1991; Boulton et al. 1992). Thus, there may be opposing integrity classifications established depending on the faunal group analyzed and the amount of time post-disturbance that integrity was evaluated.

A total of 80% (12 of 15) of the relationships in each park were not significant, suggesting most biotic integrity metrics do not reflect the same measure of stream health even at the same site. Similarly, Ogren and Huckins (2014) found that two aquatic invertebrate health metrics, the HBI and the biological condition gradient, were unrelated to each other but suggested this result was likely due to the limited range in scores, which also may have occurred in our study as our index scores were relatively high given these were least disturbed systems. Herbst and Silldorff (2006) and Hawkins et al. (2010) found stronger relationships among aquatic invertebrate indices within the low diversity river systems of the Sierra Nevada's (Spearmans's r = 0.70-0.86) and the Columbia River basin (Pearson's r = 0.63-0.92), respectively. Because our study took place in diverse streams with more complex species interactions than in the western US, it may explain why relationships among indices were harder to distinguish. Our results from the ONSR revealed that only simplistic stand-alone richness indices are comparable to one another regardless of faunal group used. However, stand-alone richness indices do not account for composition, abundances, or evenness within the community and may include a large

proportion of tolerant generalist species (Scott and Helfman 2001; Fleishman et al. 2006), which does not capture a full depiction of stream integrity. In addition, richness is known to change based on stream size and disturbance frequency (Vannote et al. 1980; Resh et al. 1988; Fausch and Bramblett 1991; Schaefer et al. 2012), which are independent of degradation and may make interpreting integrity of sites along this gradient problematic.

Despite the BNR and the ONSR being relatively similar in terms of stream sizes, underlying geology, physical habitat, water chemistry, and land cover, environmental variables that influenced biotic index scores differed between NPS units. This places emphasis on understanding biotic index relationships along environmental gradients at a fine spatial scale so that accurate interpretations can be made to guide sound management (Lyons 1996; Hawkins et al. 2010; Ogren and Huckins 2015). Our results supported our hypothesis and previous work that related indices will be sensitive to the similar environmental characteristics, in that each of the three related indices in the ONSR and two of the three related indices comparisons in the BNR were sensitive to the watershed and water quality models, respectively (Lammert and Allan 1999; Fitzpatrick et al. 2001). We found similar models or variables described both fish and aquatic invertebrate faunal groups, although models and variables differed by NPS unit (BNR: water quality model; ONSR: watershed model). In contrast, other studies have suggested that fish were influenced by upstream watershed area, land cover at both the upstream watershed and stream buffer scales, and flow variability whereas aquatic invertebrates were influenced by in-stream habitat, substrate, and substrate embeddedness (Berkman et al. 1986; Carter et al. 1996; Lammert and Allan 1999; Fitzpatrick et al. 2001). Agriculture within a watershed may lower the overall biotic integrity (Lammert 1995; Roth et al. 1996; Allan

et al. 1997; Fitzpatrick et al. 2001), however, the range and maximum percent agriculture in these previous studies were much greater (~1 to 90%) than our study (0 to 30%). We found that EPT, aquatic invertebrate taxa richness, and fish species richness indices in the ONSR were all positively related to percent agriculture in the upstream watershed, which was similar to other studies linking fish species richness and IBI to agriculture in the watershed (Fitzpatrick et al. 2001; Wang et al. 2001). However, the positive relationship we found dissolved when analyzing the mainstem river sites from the tributary sites separately and thus was likely due to larger watershed areas possessing larger percent agriculture within them. When evaluating only mainstem sites (watershed areas of 383 to 4664 km^2), fish species richness had a significant negative relationship with percent agriculture ($P = 0.003$). Therefore, there may be evidence that even a limited amount of agriculture in the watershed negatively influences biotic metrics when accounting for watershed area.

Both river systems studied may be considered least-disturbed (relative to many other watersheds in the region) due to being partially protected by NPS boundaries (Saunders et al. 2002; Lawrence et al. 2011) and thus have limited impacted land cover in their upstream watersheds (Sowa et al. 2007), and in turn had high ecological integrity based on our index interpretations. Therefore, habitat and landscape-level variables that were related to these indices may be useful for delineating biotic integrity classifications breaks in these least-disturbed systems. In the BNR, the percent intolerant fish individuals, aquatic invertebrate taxa richness, and EPT indices all increased and the IBI and fish species richness decreased with decreased water temperature. This relationship was likely not present in the ONSR since it receives stronger spring influence, with

spring discharge creating warm water refugia in cold months and cold water refugia in warm months, thus buffering high and low temperatures (Converse 1994; Westhoff and Paukert 2014). The lower levels of groundwater influence in the BNR may also explain why water temperatures were higher in warmer months when fish sampling (19.0°C versus 15.1°C), but lower in cold months for aquatic invertebrate sampling (8.7°C versus 10.8°C) compared to the ONSR. Therefore, even within these warm-water systems in the Ozarks, stream temperature and groundwater influence may need to be considered before comparing sites or stream systems across divergent temperatures. Thus our results agree with Ogren and Huckins (2015) and Lyons (1996) that by categorizing a stream's thermal regime and selecting the best corresponding index leads to a more accurate and consistent portrayal of stream integrity. Other research in Wisconsin has recognized the differences in community assemblages across thermal regimes and has created separate IBI's specifically for warm- and cold-water streams (Lyons 1992; Lyons et al. 1996), and our results suggest a similar categorization may be useful within warm water streams of the Ozarks that are and are not thermally buffered by spring influence.

We found that in the ONSR, streams with smaller watershed size appear to inherently receive lower integrity scores despite being protected and theoretically least-disturbed. This result places emphasis on the importance of using watershed areas as a delineating factor when interpreting integrity scores within the ONSR. Network position and stream size make intermediate sized stream reaches inherently more species rich (Vannote et al. 1980; Ward 1998; Fisher and Paukert 2009; Schaefer et al. 2012). Therefore, it is important to account for both of these factors comparing streams distributed along a gradient of network position. Some indices have recognized this and

have accounted for it in their scoring criteria or limited application to only a narrow window of stream sizes (Fausch et al. 1984; Lyons 1992; Lyons et al. 1996; Dauwalter et al. 2003). Although the IBI used in this study (Dauwalter et al. 2003) warned of its limited application to only wadeable streams (Strahler order 1-3), it has previously been applied to fourth through sixth order streams (Dauwalter et al. 2007; Dodd 2009; 2013). We found that in the BNR, larger watersheds had higher scores using this IBI, regardless of whether fourth and fifth order streams were included in analysis (order 2-3: P = 0.006; orders 2-5: P < 0.001). This result partially validates the application of this IBI to larger streams within this region, while still placing emphasis on interpreting stream health along the gradient of stream size.

Although the use of biotic indices is common in aquatic science because they convey the health of lotic systems to managers, stakeholders, and the public through a single quantitative score that can be grouped into a single health category (Karr 1981; 1986; Hilsenhoff 1987; Lenat 1988), we found that without proper considerations, the interpretation of these scores may at times be misconstrued, even in relatively similar systems. We found that many indices reflect attributes unrelated to degradation (e.g. network position, thermal regime). Despite finding that warmer temperatures and streams with larger watershed areas appear to have inherently higher integrity using IBI scores, the IBI used in this study (Dauwalter et al. 2003) provided a single interpretation classification system (reference: 80-100; good: 60-80; fair: 40-60; poor: 20-40; very poor: 0-20) for all wadeable streams. Similarly, overall integrity ratings by using the HBI, which also has a single interpretation classification system (excellent: 0-3.75; very good: 3.76-4.25; good: 4.26-5; fair: 5.01-5.75; fairly poor: 5.76-6.5; poor: 6.51-7.25; very poor:

7.26-10), increased with increased watershed area in the ONSR. Despite downstream reaches typically being warmer, having lower gradient, lower water velocity, finer substrates, and additive level of organic pollution from upstream reaches (Vannote et al. 1980), we still found these reaches to be higher integrity than upstream reaches. Because this contradicts expectations, it further suggests the need to account for network position or watershed area while interpreting biotic integrity scores. By accounting for watershed area and water temperature, a more accurate depiction of stream condition may be conveyed. In conclusion, we found that to capture a holistic depiction of stream integrity and evaluate how a lotic system responds to the environment, multiple indices incorporating multiple faunal groups is needed (Fitzpatrick et al. 2001). By understanding the variables or scale that each index was related to in a least-disturbed system, a more rooted interpretation of integrity can be realized, ultimately leading to better stream prioritization for management and a more accurate monitoring effort.

Literature Cited

Allan, J. D., D. L. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. Freshwater Biology 37: 149-161.

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water; Washington, D.C.

Berkman, H. E., C. R. Rabeni, and T. P. Boyle. 1986. Biomonitors of stream quality in agricultural areas: fish versus invertebrates. Environmental Management 10: 413-419.

Berkman, H. E., and C. R. Rabeni. 1987. Effects of siltation on stream fish communities. Environmental Biology of Fishes 18: 285-294.

Boulton, A. J., C. G. Peterson, N. B. Grimm, and S. G. Fisher. 1992. Stability of an aquatic macroinvertebrate community in a multiyear hydrologic disturbance regime. Ecology 73: 2192-2207.

Bowles, D. E., J. A. Luraas, L. W. Morrison, H. R. Dodd, M. H. Williams, G. A. Rowell, M. D. DeBacker, J. A. Hinsey, F. D. Usrey, and J. L. Haack. 2007. Protocol for monitoring aquatic invertebrates at Ozark National Scenic Riverways, Missouri, and Buffalo National River, Arkansas. Natural Resource Report NPS/HTLN/NRR – 2007/009. National Park Service, Fort Collins, Colorado.

Bramblett, R. G., and K. D. Fausch. 1991. Variable fish communities and the index of biotic integrity in a western Great Plains river. Transactions of the American Fisheries Society 120: 752-769.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Springer.

Carter, J. L., S. V. Fend, and S. S. Kennelly. 1996. The relationships among three habitat scales and stream benthic invertebrates community structure. Freshwater Biology 35: 109-124.

Converse, J. W. 1994. Water chemistry, nutrient dynamics, and macrophyte production of a large cold-water spring. MS thesis. University of Missouri, Columbia.

Dauwalter, D. C., E. J. Pert, and W. E. Keith. 2003. An index of biotic integrity for fish assemblages in Ozark Highland Streams of Arkansas. Southeastern Naturalist 2: 447-468.

Dodd, H.R. 2009. Fish community monitoring at Buffalo National River: 2006-2007 status report. Natural Resource Technical Report NPS/HTLN/NRTR—2009/185. National Park Service, Fort Collins, Colorado.

Dodd, H. R. 2013. Fish community monitoring at Ozark National Scenic Riverways: 2005-2010 summary report. Natural Resource Data Series NPS/HTLN/NRDS—2013/599. National Park Service, Fort Collins, Colorado.

Fausch, K. D., J. R. Karr, and P. R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. Transactions of the American Fisheries Society 113: 39-55.

Fausch, K. D., and R. G. Bramblett. 1991. Disturbance and fish communities in intermittent tributarites of a western Great Plains river. Copeia 1991: 659-674.

Fisher, J. R., and C. P. Paukert. 2008. Habitat relationships with fish assemblages in minimally disturbed Great Plains regions. Ecology of Freshwater Fish 17: 597-609.

Fisher, J. R., and C. P. Paukert. 2009. Effects of sampling effort, assemblage similarity, and habitat heterogeneity on estimates of species richness and relative abundance of stream fishes. Canadian Journal of Fisheries and Aquatic Sciences 66: 277-290.

Fitzpatrick, F. A., B. C. Scudder, B. N. Lenz, and D. J. Sullivan. 2001. Effects of multi-scale environmental characteristics on agricultural stream biota in eastern Wisconsin. Journal of the American Water Resources Association 37: 1489-1507.

Fleishman, E., R. F. Noss, and B. R. Noon. 2006. Utility and limitations of species richness metrics for conservation planning. Ecological Indicators 6: 543-553.

Fry, J. A., G. Xian, S. Jin, J. A. Dewitz, C. G. Homer, C. A. Barnes, N. D. Herold, and J. D. Wickham. 2011. Completion of the 2006 national land cover database for the conterminous United States. Photogrammetric Engineering and Remote Sensing 77: 858-864.

Hawkins, C. P., Y. Cao, and R. Roper. 2010. Method of predicting reference conditions affects the performance and interpretations of ecological indices. Freshwater Biology 55: 1066-1085.

Herbst, D. B., and E. L. Silldorff. 2006. Comparison of the performance of different bioassessment methods: similar evaluations of biotic integrity from separate programs and procedures. Journal of the North American Benthological Society 25: 513-530.

Hilsenhoff, W. L., 1987. An improved biotic index of organic stream pollution. Great Lakes Entomologist 20: 31-39.

Hughes, R. M., P. R. Kaufmann, A. T. Herlihy, T. M. Kincaid, L. Reynolds, and D. P. Larsen. 1998. A process for developing and evaluating indices of fish assemblage integrity. Canadian Journal of Fisheries and Aquatic Sciences 55: 1618-1631.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6: 21-27.

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

Karr, J. R. 1991. Biological integrity: a long-neglected aspect of water resource management. Ecological Applications 1: 66-84.

Lammert, M., and J. D. Allan. 1999. Assessing biotic integrity of streams: effects of scale in measuring the influence of land use/cover and habitat structure on fish and macroinvertebrates. Environmental Management 23: 257-270.

Lawrence, D. J., E. R. Larson, C. A. R. Liermann, M. C. Mims, T. K. Pool, and J. D. Olden. 2011. National parks as protected areas for U.S. freshwater fish diversity. Conservation Letters 4: 364-371.

Lenat, D. R. 1988. Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. Journal of the North American Benthological Society 7: 222-233.

Lenat, D. R. 1993. A biotic index for the southeastern United States: derivation and list of tolerance values, with criteria for assigning water-quality ratings. Journal of the North American Benthological Society. 12: 279-290.

Lyons, John 1992. Using the index of biotic integrity (IBI) to measure environmental quality in warmwater streams of Wisconsin. General Technical Report NC-149. St. Paul, MN: U.S. Dept. of Agriculture, Forest Service, North Central Forest Experiment Station

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

Lyons, J. B. M. Weigel, L. K. Paine, and D. J. Undersander. 2000. Influence of intensive rotational grazing on bank erosion, fish habitat quality, and fish communities in

southwestern Wisconsin trout streams. Journal of Soil and Water Conservation 55: 271-276.

Matthews, W. J. 1986. Fish faunal structure in an Ozark stream: stability, persistence and a catastrophic flood. Copeia 1986: 388-397.

McElravy, E. P., G. A. Lamberti, and V. H. Resh. 1989. Year-to-year variation in the aquatic macroinvertebrate fauna of a northern California stream. Journal of North American Benthological Society 8: 51-63.

Nerbonne, B. A., and B. Vondracek. 2001. Effects of local land use on physical habitat, benthic macroinvertebrates, and fish in the Whitewater River, Minnesota, USA. Environmental Management. 28: 87-99.

NPSpecies - The National Park Service Biodiversity Database. IRMA Portal version. https://irma.nps.gov/NPSpecies/ (certified species list - park status view; accessed February 1, 2016).

Ogren, S. A., and C. J. Huckins. 2014. Evaluation of suitability and comparability of stream assessment indices using macroinvertebrate data sets from the Northern Lakes and Forests Ecoregion. Ecological Indicators 40: 117-126.

Ogren, S. A., and C. J. Huckins. 2015. Applicability and interpretation of fish indices of biotic integrity (ibi) for bioassessment in the upper Midwest. North American Journal of Fisheries Management 35: 281-295.

Petersen, J. C., B. G. Justus, H. R. Dodd, D. E. Bowles, L. W. Morrison, M. H. Williams, and G. A. Rowell. 2008. Methods for monitoring fish communities of Buffalo National River and Ozark national Scenic Riverways in the Ozark Plateaus of Arkasas and Missouri: Version 1. U. S. Geological Survey Open-File Report 2007-1302, 94p.

Plafkin, J.L., M.T. Barbour, K.D. Porter, S.K. Gross, and R.M. Hughes. 1989. Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish. U.S. Environmental Protection Agency, Office of Water Regulations and Standards, Washington, D.C. EPA 440-4-89-001.

Resh, V. H., A. V. Brown. A. P. Covich, M. e. Gurtz, H. W. Li. G. W. Minshall, S. R. Reice, A. L. Sheldon, J. B. Wallace, and R. C. Wissmar. 1988. The role of disturbance in stream ecology. Journal of the North American Benthological Society 7: 433-455.

Richards, C., and G. E. Host. 1993. Identification of predominant environmental factors structuring stream macroinvertebrate communities within a large agricultural catchment. Freshwater Biology 29: 285-294.

Richards, C., L. B. Johnson, and G. E. Host. 1996. Landscape-scale influences on stream habitats and biota. Canadian Journal of Fisheries and Aquatic Science 53: 295-311.

Richards, C., and G. E. Host. 1994. Examining land use influences on stream habitats and macroinvertebrates – a GIS approach. Water Resources Bulletin 30: 729-738.

Rosgen, D. L. 1994. A classification of natural rivers. Catena 22: 169-199.

Roth, N. E., J. D. Allan, and D. L. Erickson. 1996. Landscape influences on stream biotic integrity assessed at multiple spatial scales. Landscape Ecology 11: 141-156.

Saunders, D. L., J. J. Meeuwig, and A. C. J. Vincent. 2002. Freshwater protected areas: strategies for conservation. Conservation Biology 16: 30-41.

Schaefer, J. F., S. R. Clark, and M. L. Warren. 2012. Diversity and stability in Mississippi stream fish assemblages. Freshwater Science 31: 882-894.

Scott, M. C., and G. S. Helfman. 2001. Native invasions, homogenization, and the mismeasure of integrity of fish assemblages. Fisheries 26: 6-15.

Sowa, S. P., G. Annis, M. E. Morey, and D. D. Diamond. 2007. A gap analysis and comprehensive conservation strategy for riverine ecosystems of Missouri. Ecological Monographs 77: 301-334.

Sponseller, R. A., E. F. Benfield, and H. M. Valett. 2001. Relationships between land use, spatial scale and stream macroinvertebrate communities. Freshwater Biology 46: 1409-1425.

Stevens, D. L., and A. R. Olson. 2004. Spatially balanced sampling of natural resources. Journal of the American Statistical Association 99: 262-278.

USEPA (US Environmental Protection Agency). 2011. A Primer on Using Biological Assessments to Support Water Quality Management. EPA-810-R-11-01. Office of Science and Technology, Office of Water, Washington, DC.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37: 130-137.

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

Ward, J. V. 1998. Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. Biological Conservation 83: 269-278.

Westhoff, J. T., and C. P. Paukert. 2014. Climate change simulations predict altered biotic response in a thermally heterogeneous stream system. PloS One 9: e111438. doi:10.1371/journal.pone.0111438.

Tables

Table 1. Biotic indices and fauna commonly used and the aspect of integrity the index was intended to describe within lotic ecosystems.

Index	Fauna	Utility	Citation
Index of Biotic Integrity (IBI)	Fish	Ecosystem Health	Dauwalter et al. 2003
Species Richness	Fish	Habitat Complexity	Fitzpatrick et al. 2001
Percentage of Intolerant Individuals	Fish	Ecosystem Health	Barbour et al. 2000
Taxa Richness	Invertebrate	Ecosystem Health	Lenat 1988
EPT Richness	Invertebrate	Water Quality	Lenat 1988
Hilsenhoff Biotic Index (HBI)	Invertebrate	Organic Pollution	Hilsenhoff 1987

Table 2. Metrics comprised within the multimetric Index of Biotic Integrity developed for the Ozark Highland region (Dauwalter et al. 2003).

Index of Biotic Integrity Metrics
Percent individuals algivorous/herbivorous, invertivorous, and piscivorous
Percent individuals with black spot/anomaly
Percent individuals as green sunfish, bluegill, yellow bullhead, and channel catfish
Percent individuals as invertivorous
Percent individuals as top carnivore
Number of darter, sculpin, and madtom species
Number of lithophilic spawning species

Table 3. A priori models at watershed and local scales used for determining the scale that each index is sensitive to. Local landcover refers to a 100 meter buffer surrounding the stream reach. Models three to six are designed to imitate what various biotic indices are intended to measure.

	Model Name	Model Components
1	Watershed	Percent Agriculture in Watershed + Percent Urban Watershed + Watershed Area + Drainage Density + Sinuosity
2	Local	Percent Agriculture in Reach + Percent Urban Reach + Substrate + Velocity + Depth + Embeddedness + Width : Depth
3	Water Quality	Dissolved Oxygen. + Sp. Conductance + Water Temperature + pH
4	Habitat Complexity	Drainage Density + Sinuosity + Depth + Substrate + Velocity + Width : Depth
5	Organic Pollution	Percent Agriculture (Watershed) + Percent Urban (Watershed) + Embeddedness + Dissolved Oxygen
6	Ecosystem Health	Sinuosity + Embeddedness + Width : Depth + Dissolved Oxygen + Specific Conductance + Water Temperature + pH
7	**Global**	

Table 4. The mean and range of biotic index scores in the Ozark National Scenic Riverways (ONSR; 24 sites) and Buffalo National River (BNR; 31 sites) from 2005 to 2013.

Index	ONSR		BNR	
	Mean	Range	Mean	Range
Index of Biotic Integrity (Dauwalter et al. 2003)	78	65 - 92	73	50 - 85
Percent Intolerant Fish Individuals	50	17 - 85	57	9 - 96
Fish Species Richness	23	4 - 36	20	7 - 37
Aquatic Invertebrate Taxa Richness	25	19 - 31	23	7 - 35
Ephemeroptera, *Plecoptera*, *Trichoptera* Richness	12	6 - 17	12	2 - 21
Hilsenhoff Biotic Index (Hilsenhoff 1987)	4.05	3.09 - 5.09	4.89	3.02 - 7.14

Table 5. The mean and range of habitat, location, and climate covariates used when investigating attributes linked to biotic index scores in the Buffalo National River in Arkansas and Ozark National Scenic Riverways in Missouri from 2005 to 2013.

| | Buffalo National River | | Ozark National Scenic Riverways | |
| | Fish | Aquatic Invertebrate | Fish | Aquatic Invertebrate |
Covariate	Mean (Range)	Mean (Range)	Mean (Range)	Mean (Range)
Percent Agriculture Watershed	11 (0-30)	11 (0-30)	10 (0-22)	10 (0-22)
Percent Urban Watershed	3 (1-5)	3 (1-5)	3 (1-5)	3 (1-5)
Percent Agriculture Reach	2 (0-3)	2 (0-3)	1 (0-16)	1 (0-16)
Percent Urban Reach	4 (0-28)	4 (0-28)	5 (0-25)	5 (0-25)
Watershed Area (km²)	368 (4-2988)	368 (4-2988)	759 (7-4664)	759 (7-4664)
Drainage Density	1.41 (0.13-2.01)	1.41 (0.13-2.01)	1.63 (1.39-1.98)	1.63 (1.39-1.98)
Sinuosity	1.24 (1.02-2.13)	1.24 (1.02-2.13)	1.29 (1.00-2.34)	1.29 (1.00-2.34)
Substrate (Wentworth	12 (11-24)	14 (12-15)	12 (1-15)	14 (12-15)
Velocity (m/sec)	0.24 (0.03-0.68)	0.49 (0.14-0.82)	0.19 (0.02-0.58)	0.52 (0.09-0.88)
Depth (m)	0.5 (0.15-1.33)	0.27 (0.07-0.72)	0.63 (0.15-1.47)	0.30 (0.09-0.62)
Width : Depth	29.5 (13.9-54.7)	55.8 (19.9-121.0)	24.5 (14.8-47.3)	55.5 (20.3-156.2)
Embeddedness	2 (0-3)	2 (2)	2 (2-3)	2 (2-3)
Water Temperature (°C)	19.1 (11.4-27.6)	8.68 (2.9-13.9)	15.2 (6.0-20.3)	10.8 (7.2-14.0)
pH	7.63 (6.91-8.23)	7.87 (6.64-8.62)	7.90 (6.96-8.25)	7.96 (7.36-8.49)
Specific Conductance (μS/cm)	246 (85-435)	237 (86-419)	339 (278-460)	295 (152-492)
Dissolved Oxygen (mg/l	9.24 (6.85-11.71)	11.78 (7.48-15.08)	8.97 (6.59-11.89)	10.59 (8.27-12.07)

Table 6. Spearman's rank correlations (ρ) between three fish and three aquatic invertebrate indices in the Buffalo National River, Arkansas and the Ozark National Scenic Riverways, Missouri from 2005 to 2013. Correlations in gray were statistically significant after a Bonferroni correction (α = 0.003). IBI = Index of Biotic Integrity (Dauwalter et al. 2003). % Intolerant = percent of fish individuals intolerant to habitat alteration. Spp. Richness = Total number of fish species. EPT = Number of unique taxa in the orders *Ephemeroptera*, *Plecoptera*, and *Trichoptera*. HBI = Family level Hilsenhoff Biotic Index (Hilsenhoff 1987). Taxa Richness = Number of distinct aquatic invertebrate taxa.

	Buffalo National River		Ozark National Scenic Riverways	
Comparison	ρ	P	ρ	P
Fish				
IBI - % Intolerant	-0.28	0.124	-0.11	0.597
IBI - Fish Richness	0.77	< 0.001	-0.16	0.441
Fish Richness - % Intolerant	-0.49	0.006	-0.12	0.561
Aquatic Invertebrate				
EPT - HBI	-0.54	0.002	-0.47	0.021
Invert Richness - EPT	0.87	< 0.001	0.81	< 0.001
Invert Richness - HBI	-0.43	0.016	-0.34	0.102
Fish-Aquatic Invertebrate				
% Intolerant - EPT	-0.14	0.459	-0.09	0.670
% Intolerant - HBI	0.22	0.237	0.07	0.756
% Intolerant - Invert Richness	-0.03	0.879	-0.04	0.837
IBI - EPT	0.08	0.648	-0.27	0.197
IBI - HBI	-0.03	0.872	0.20	0.345
IBI - Invert Richness	-0.02	0.928	-0.05	0.799
Fish Richness - EPT	0.00	0.982	0.69	< 0.001
Fish Richness - HBI	-0.03	0.870	-0.55	0.006
Fish Richness - Invert Richness	-0.15	0.407	0.60	0.002

Table 7. Top models to predict biotic index scores in the Ozark National Scenic Riverways (ONSR) and the Buffalo National River (BNR). EPT = Number of taxa in orders *Ephemeroptera, Plecoptera,* and *Trichoptera.* Taxa Richness = Number of distinct aquatic invertebrate taxa. HBI = Family level Hilsenhoff Biotic Index (Hilsenhoff 1987). IBI = Index of Biotic Integrity (Dauwalter et al. 2003). Percent Intolerant = Percent of fish individuals intolerant to habitat alteration. Species Richness = Total number of fish species. Arrows represent coefficient relationship within the model (↑: Positive, ↓: Negative). Bolded variables refer to significant relationships.

| Park	Index	Model Name	Model Components	ΔAICc	w_i
ONSR	EPT	Watershed	**Percent Ag.** ↑ + Percent Urban + Sinuosity + Drainage Density + **Area** ↑	0	0.67
		Organic Pollution	**Percent Ag. Watershed** ↑ + Percent Urban Watershed + D.O. + **Embeddedness** ↓	1.46	0.32
	Taxa Richness	Watershed	**Percent Ag.** ↑ + Percent Urban + Sinuosity + Drainage Density + **Area** ↑	0	0.91
	HBI	Watershed	Percent Ag. + Percent Urban + Sinuosity + Drainage Density + **Area** ↓	0	0.86
	IBI	Watershed	Percent Ag. + Percent Urban + Sinuosity + Drainage Density + Area	0	0.68
		Organic Pollution	Percent Ag. Watershed + Percent Urban Watershed + D.O. + Embeddedness	1.55	0.32
	Percent Intolerant	Organic Pollution	Percent Ag. Watershed + Percent Urban Watershed + D.O. + Embeddedness	0	0.94
	Species Richness	Watershed	**Percent Ag.** ↑ + Percent Urban + Sinuosity + Drainage Density + **Area** ↑	0	0.99
BNR	EPT	Water Quality	D. O. + Sp. Cond. + **Water Temperature** ↓ + pH	0	0.93
	Taxa Richness	Water Quality	D. O. + Sp. Cond. + **Water Temperature** ↓ + pH	0	0.78
	HBI	Watershed	Percent Ag. + Percent Urban + Sinuosity + **Drainage Density** ↑ + Area	0	0.44
		Water Quality	D. O. + Sp. Cond. + Water Temperature + **pH** ↑	0.43	0.35
	IBI	Water Quality	D. O. + Sp. Cond. + **Water Temperature** ↑ + pH	0	0.81
	Percent Intolerant	Organic Pollution	Percent Ag. Watershed + Percent Urban Watershed + D.O. + Embeddedness	0	0.52
		Water Quality	D. O. + Sp. Cond. + **Water Temperature** ↓ + pH	0.44	0.42
	Species Richness	Habitat Complexity	Drainage Density + Sinuosity + **Depth** ↑ + Substrate + Velocity + **Width to Depth Ratio** ↑	0	0.95

Figures
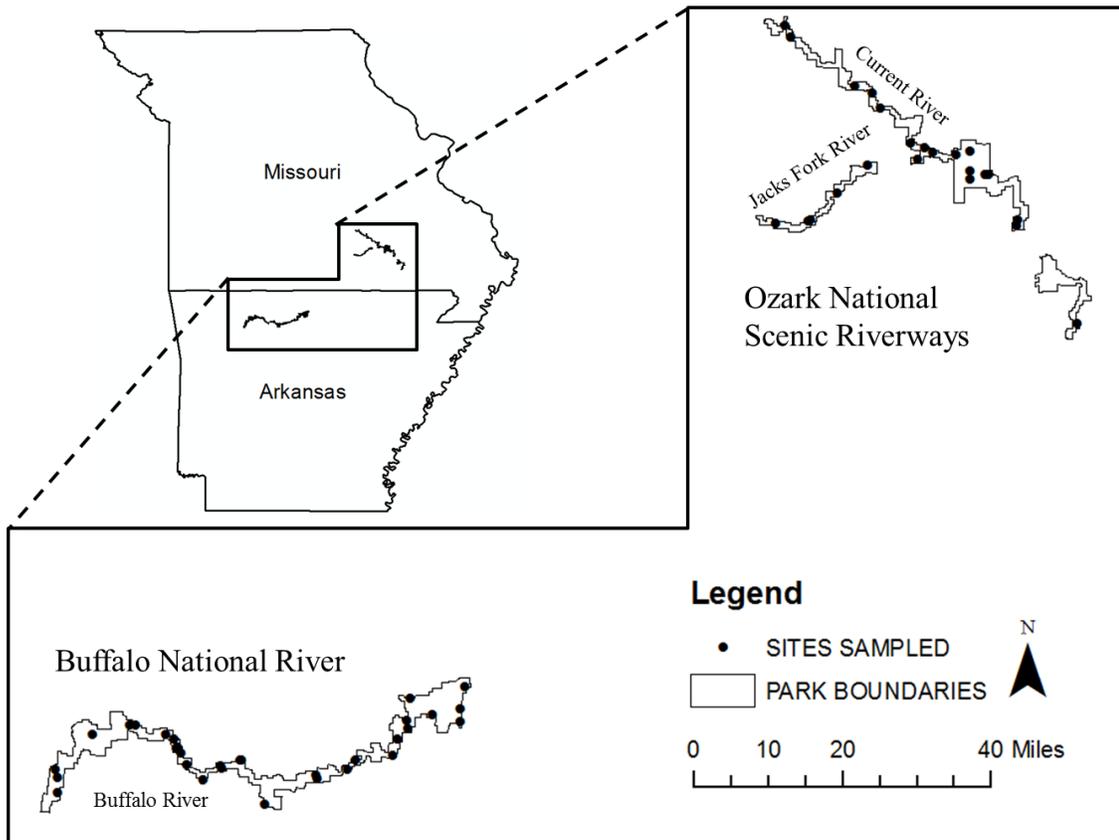
Figure 1. Map of National Park Service boundaries and fish and aquatic invertebrate sampling locations in the Ozark National Scenic Riverways, Missouri (N=24) and the Buffalo National River, Arkansas (N=31) from 2005 to 2013.
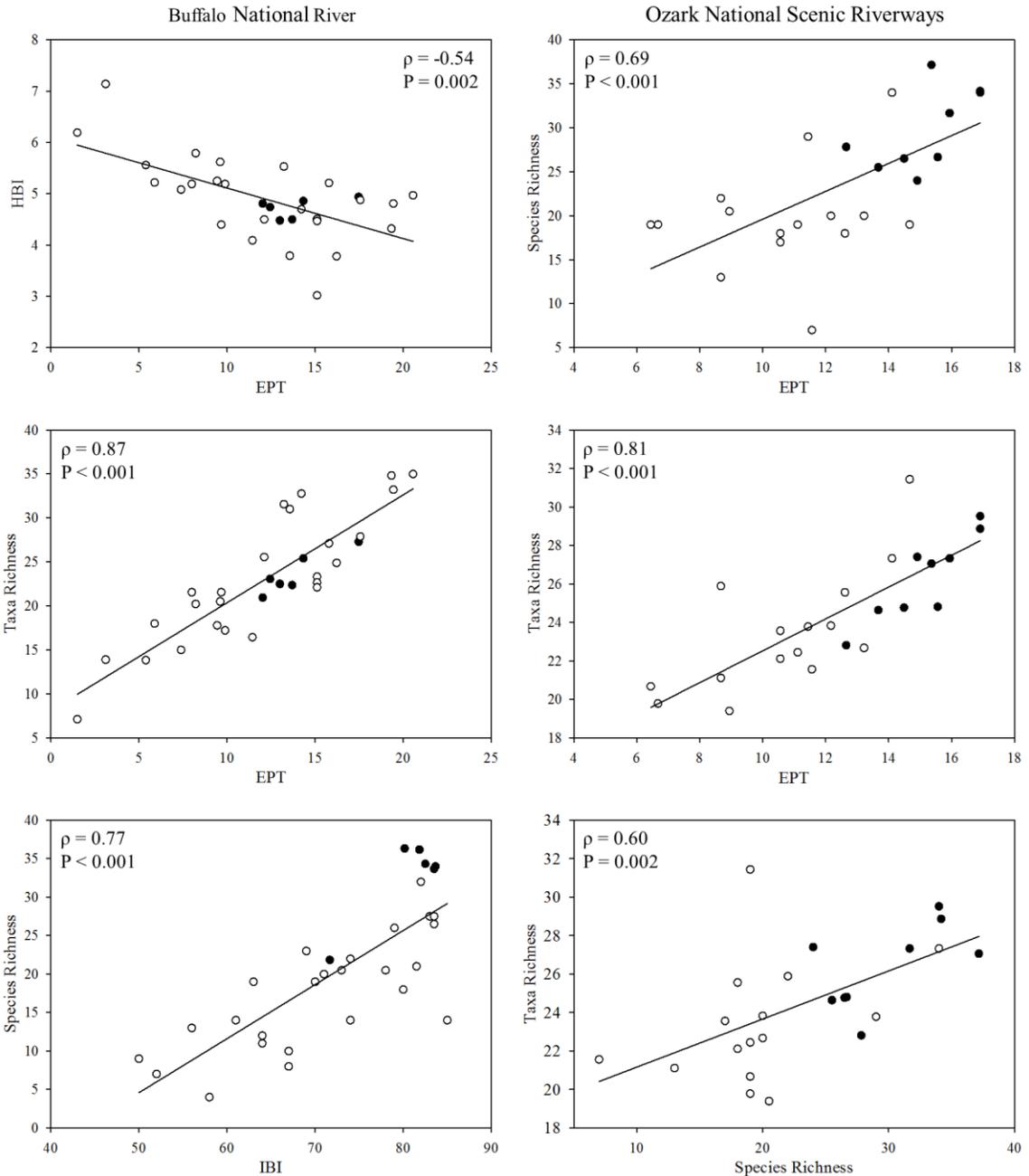
Figure 2. Significant spearman's rank correlations (ρ) between biotic index scores from the Buffalo National River (left) and the Ozark National Scenic Riverways (right). The Buffalo National River was analyzed using data from n=31 sites and the Ozark National Scenic Riverways was analyzed using data from n=24 sites collected from 2005 to 2013. IBI = Index of Biotic Integrity (Dauwalter et al. 2003), EPT = *Ephemeroptera*, *Plecoptera*, and *Trichoptera* richness, HBI = Hilsenhoff Biotic Index (Hilsenhoff 1987). Species Richness refers to fish while taxa richness refers to aquatic invertebrates. Solid circles are mainstem river sites and hollow circles are tributary sites.
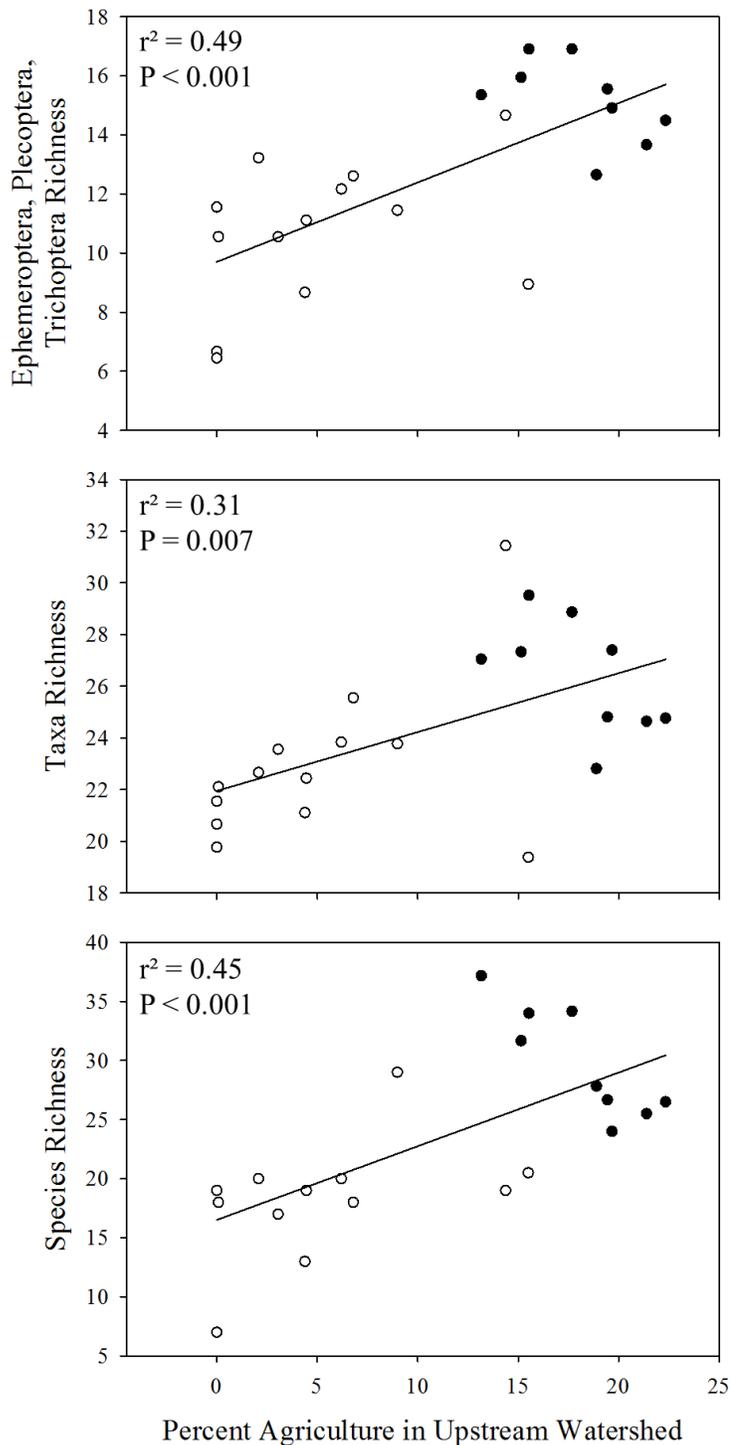
Figure 3. Relationship between percent agriculture in the upstream watershed and fish species richness, aquatic invertebrate taxa richness, and *Ephemeroptera*, *Plecoptera*, and *Trichoptera* richness in the Ozark National Scenic Riverways, Missouri from 2006 to 2013. Mainstem river sites are solid circles while tributary sites are hollow circles. All significant positive relationships dissolve when analyzing the mainstem and tributary sites separately.
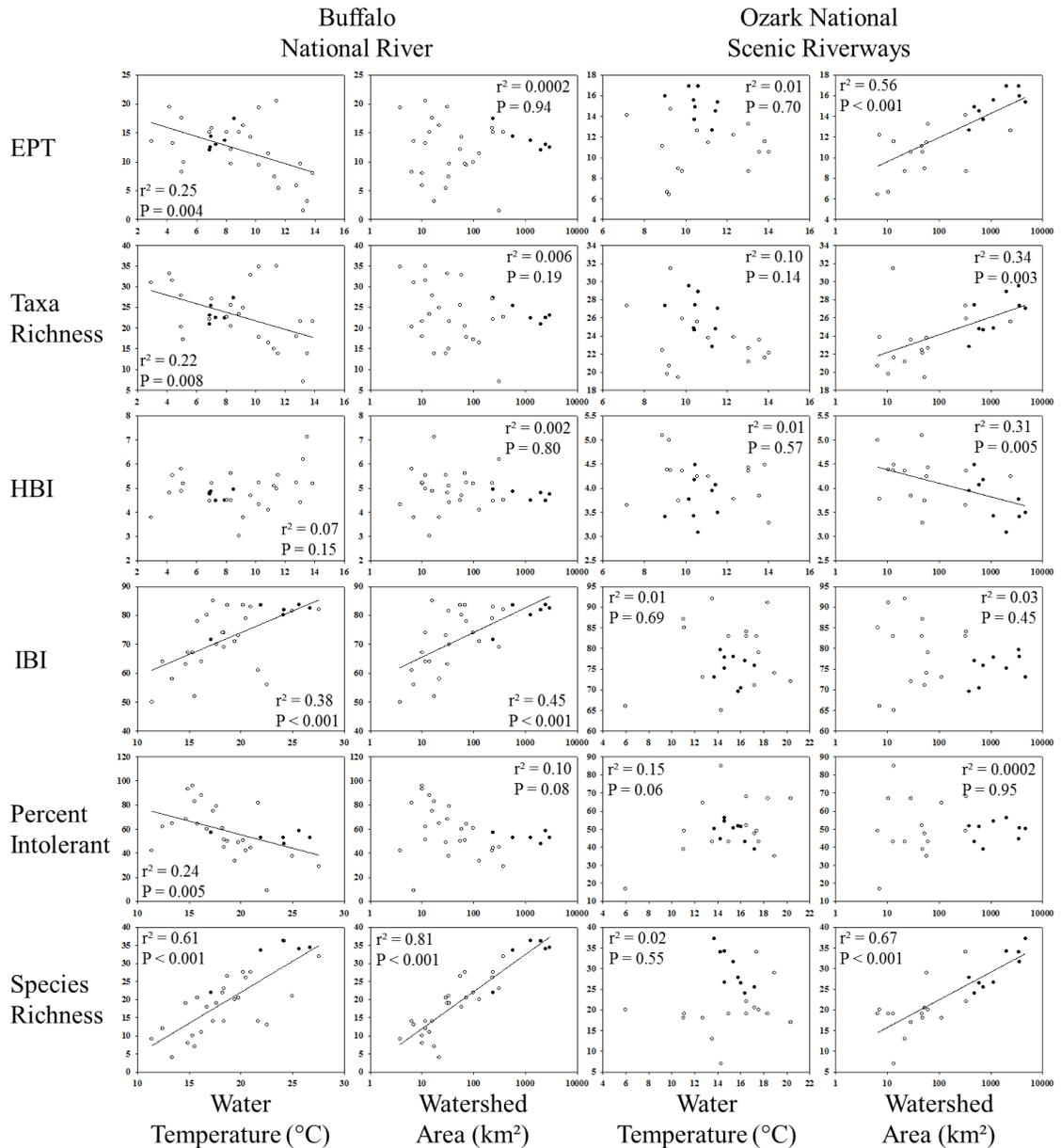
Figure 4. Relationships with *Ephemeroptera*, *Plecoptera*, and *Trichoptera* species richness (EPT), aquatic invertebrate taxa richness, Hilsenhoff Biotic Index (HBI; Hilsenhoff 1987), Index of Biotic Integrity (IBI; Dauwalter et al. 2003), percent intolerant fish individuals, and fish species richness in the Buffalo National River, Arkansas and the Ozark National Scenic Riverways, Missouri versus water temperature and watershed area (log10 transformed). Solid circles are mainstem river sites and hollow circles are tributary sites. Significant relationships have a regression line. Sampling occurred from 2005 to 2013.

Conclusion

As our world continues to change, so do the ways in which we approach conserving biological resources. Our understanding of how aquatic biota may respond to changes in habitat caused by anthropogenic disturbance is constantly evolving and provides resource managers with a more efficient means of conservation by using the best available science. My study assessed the current condition of stream ecosystems so that we may recognize when changes occur, predicted attributes that will alter aquatic communities in the future in order to prioritize locations for protection, and assessed the limitations of current biotic health indices to aid in the accuracy in depicting current conditions and locations for future management.

Stream ecosystems are dynamic features in the landscape (Horwitz 1978; Schlosser 1982; Ross et al. 1985; Poff and Ward 1989; Poff and Allan 1995), thus to determine where a stream lies on a continuum of stability, managers need to understand how a community naturally fluctuates over time and space. Departures from this baseline community variability can then be used in future settings to make inferences as to the stability of a community. I found that fish communities are consistently less temporally variable than spatially variable, likely do to their mobility and ability to avoid periods of stress and efficient means of recolonization post-disturbance. This relationship did not hold true with aquatic invertebrates, in that some parks were less temporally variable while others were less spatially variable. Because aquatic invertebrates are relatively sedentary and are less efficient at recolonizing (Williams and Hynes 1976), they are more impacted by disturbances for a longer period of time. Therefore, this result may illustrate how disturbances have a more legacy affect in influencing aquatic invertebrate

community structure over time. Although habitat, climate, and region was not strongly associated with fish community variability, aquatic invertebrate spatial variability was found to be related to degrees of habitat homogenization and proximity of sites to one another while temporal variability was found to be greater in Plains streams rather than Ozark streams.

In the United States alone, there are over 250,000 rivers containing approximately 3.5 million river miles (Krammerer 1990). Therefore, prioritizing rivers and reaches for management is an important objective. We can prioritize streams based on locations likely to experience more severe anthropogenic alterations and changes in climate as well as the communities within them that are more prone to change with a changing environment. My vulnerability assessment took the latter approach and I was able to assess stream communities that are more or less likely to become impacted with a changing environment. I found that Plains streams were less vulnerable than Ozark streams, likely because Plains streams are harsh environments composed of tolerant generalist species (Matthews 1988; Ostrand and Wilde 2002; Dodds et al. 2004). Because Ozark streams are more hydraulically and thermally stable, the community was more specialized and thus more vulnerable to change (Munday 2004). By overlaying my results of where aquatic biota are most vulnerable with areas likely to experience the largest change in climate and anthropogenic influence will give an even more refined criterion for prioritizing management locations.

Assessing the consistency, strengths, and limitations of various biotic health indices will aid in the accuracy in depicting current conditions and prioritizing locations for management (Ogren and Huckins 2014; 2015). In two similar river systems in the

Ozarks, I found few biotic indices had consistent integrity scores and that indices that were related differed between systems. However, in both systems, results emphasized that to portray a more holistic depiction of stream integrity, multiple indices incorporating both fish and aquatic invertebrate taxa may be needed. However, our results suggested that biotic integrity scores were not linked to habitat degradation, likely because the rivers had limited urban and agriculture in the watershed (Sowa et al. 2007) and generally protected within NPS boundaries (Saunders et al. 2002; Lawrence et al. 2011). However, the index scores were related to natural environmental gradients. For example, because I found that streams with smaller watershed size seemed to inherently receive lower biotic integrity scores using four out of the six indices with the Ozark National Scenic Riverways, it places emphasis on interpreting streams along the gradient of watershed area separately. Similarly, I found that water temperature influenced five out of the six indices with the Buffalo National River, and also places emphasis on interpreting stream integrity depending on where it lies along the gradient water temperature. By understanding that multiple indices incorporating multiple faunal groups are needed to get a broader depiction of stream health and that index scores may need to take into account environmental gradients unrelated to degradation for accurate interpretations, it may lead to more accurate depiction of current conditions and areas to prioritize for management.

My research provides a platform to more accurately assess the status of lotic ecosystems, now and into the future. Departures from the baseline community variability over time and space may lead to assertions of stability and provide managers with locations to manage. Stream vulnerability using both fish and aquatic invertebrate

communities can be used to assess streams that are more or less vulnerable as well as forms of degradation that would impact the community the greatest and allow for more focused management. However, multiple indices and multiple faunal groups are needed to reveal a comprehensive depiction of stream condition. By implementing these strategies into monitoring programs, a more rooted understanding of lotic ecosystems, the biota within them, and their status with regards to health or condition may be made now and into the future.

Literature Cited

Dodds, W. K., K. Gido, M. R. Whiles, K. M. Fritz, and W. J. Matthews. 2004. Life on the edge: the ecology of Great Plains prairie streams. Bioscience 54: 205-216.

Horwitz, R. J., 1978. Temporal variability patterns and the distributional patterns of stream fishes. Ecological Monographs 48: 307-321.

Kammerer, J.C., May, 1990, Largest Rivers in the United States, US Geological Survey Fact Sheet, Open File Report 87-242.

Lawrence, D. J., E. R. Larson, C. A. R. Liermann, M. C. Mims, T. K. Pool, and J. D. Olden. 2011. National parks as protected areas for U.S. freshwater fish diversity. Conservation Letters 4: 364-371.

Matthews, W. J., 1988. North American prairie streams as systems for ecological study. Journal of the North American Benthological Society 7: 387-409.

Munday, P. L. 2004. Habitat loss, resource specialization, and extinction on coral reefs. Global Change Biology 10: 1642-1647.

Ogren, S. A., and C. J. Huckins. 2014. Evaluation of suitability and comparability of stream assessment indices using macroinvertebrate data sets from the Northern Lakes and Forests Ecoregion. Ecological Indicators 40: 117-126.

Ogren, S. A., and C. J. Huckins. 2015. Applicability and interpretation of fish indices of biotic integrity (ibi) for bioassessment in the upper Midwest. North American Journal of Fisheries Management 35: 281-295.

Ostrand, K. G., and G. R. Wilde. 2002. Seasonal and spatial variation in a prairie stream-fish assemblage. Ecology of Freshwater Fish 11: 137-149.

Poff, N. L. and J. V. Ward. 1989. Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patters. Canadian Journal of Fisheries and Aquatic Sciences 46: 1805-1818.

Poff, N. L. and J. D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology 76: 606-627.

Ross, S. T., W. J. Matthews, and A.A. Echelle. 1985. Persistence of stream fish assemblages: effects of environmental change. The American Naturalist 126: 24-40.

Saunders, D. L., J. J. Meeuwig, and A. C. J. Vincent. 2002. Freshwater protected areas: strategies for conservation. Conservation Biology 16: 30-41.

Schlosser, I. J., 1982. Fish community structure and function along two habitat gradients in a headwater stream. Ecological Monographs 52: 395-414.

Sowa, S. P., G. Annis, M. E. Morey, and D. D. Diamond. 2007. A gap analysis and comprehensive conservation strategy for riverine ecosystems of Missouri. Ecological Monographs 77: 301-334.

Williams, D. D., and H. B. N. Hynes. 1979. The recolonization mechanisms of stream benthos. Oikos 27: 265-272.