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## Cover Page Footnote

We thank several people who assisted us with fieldwork or sample processing, including Hope Dodd, Jessica Luraas, Shawn Hodges, Jared James, Angela Bandy, Ryan Green, Cary Nabors, and Kyle Ellis. Views, statements, findings, conclusions, recommendations, and data in this paper are solely those of the authors and do not necessarily reflect views and policies of the U.S. Department of Interior, National Park Service. Mention of trade names or commercial products does not constitute endorsement or recommendation for use by the National Park Service.

# Long-term Aquatic Invertebrate Monitoring at Buffalo National River, Arkansas

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Running title: Aquatic invertebrate monitoring at the Buffalo River

## Abstract

Aquatic invertebrate community structure was used to assess long-term water quality integrity in the mainstem of the Buffalo National River, Arkansas from 2005 to 2013. Nine benthic invertebrate samples were collected from each of six sampling sites using a Slack-Surber sampler. The Stream Condition Index (SCI) developed for Ozark streams was used to assess integrity of the invertebrate communities. This index is calculated using taxa richness, EPT (Ephemeroptera, Plecoptera, Trichoptera) Richness, Shannon's Diversity Index, and Hilsenhoff Biotic Index (HBI). Sørensen's similarity index was used to assess community similarity among sites, and scores were then analyzed using ascendant hierarchical cluster analysis. The benthic invertebrate fauna was diverse with 167 distinct taxa identified from all sites, with similarities ranging from 70% to 83%. Cluster analysis showed that sites were clustered in a longitudinal progression, with those sites closest to one another in linear distance generally being the most closely related. Overall, the invertebrate taxa of the Buffalo River are largely intolerant (mean tolerance value= 4.38). Taxa richness was typically greater than 20 among samples, and EPT richness values consistently were greater than 12 for all sites in most years. Shannon's diversity index values generally ranged from 2.0 to 2.5 among sites and years. Metric values tended to decrease in a downstream direction to Site 4, and then increase to levels observed upstream. The exception was for HBI, which did not show this response and values for this metric generally were below 5. SCI scores among sampling sites were variable but not generally impaired and were fully biologically-supporting. Water quality (temperature, dissolved oxygen, specific conductance, pH, turbidity) met state standards in all instances. Habitat data were summarized, but found to be poorly correlated with invertebrate metrics (<30% significant). Although the condition of invertebrate communities and water quality in the Buffalo River are largely sound and have high integrity, numerous ongoing and projected threats to these resources remain, and those threats largely originate outside of the park's jurisdictional boundaries. Inherent variability of invertebrate community diversity and density across sites and years highlights the importance of using multi-metric assessment and multiyear monitoring to support management decisions.

## Introduction

The Buffalo National River (BUFF) was established in 1972 to protect the corridor of the Buffalo River and its tributaries. However, the NPS jurisdictional boundary of the Buffalo River is generally a narrow corridor that encompasses only about 11% of the watershed, while over 60% of the watershed is in private ownership (Mott and Luraas 2004). This leaves much of the watershed unprotected from human activities such as timber management, landfills, grazing, livestock operations, urbanization, gravel mining, stream channelization, and removal of riparian vegetation. Wadeable streams of the Ozarkian region, including those at BUFF, generally are in relatively good condition, but the previously noted stressors threaten their integrity (Petersen and Femmer 2002; Petersen 2004; Huggins *et al*. 2005; United States Environmental Protection Agency 2006). Since the establishment of BUFF, more of the watershed has been deforested than is protected within the boundaries of the National River (Scott and Hofer 1995; Scott and Udouj 1999; Mott 2000). This is problematic because land use practices at the watershed level tend to overwhelm localized protection of stream corridors (Roth *et al*. 1996; Heino *et al*. 2003; ZumBerge *et al*. 2003). For example, increases in bank erosion rates and changes in channel morphology through time have been correlated with increased land clearing of steep uplands within a stream basin (Stephenson and Mott 1992; Jacobson and Primm 1997), as well as historical riparian land clearing (Panfil and Jacobson 2001). Moreover, the Buffalo River basin is located in an area of extensive karst topography, making its streams vulnerable to contaminated groundwater recharge and interbasin transfer of groundwater from adjacent watersheds (Brahana *et al.* 2016; Watershed Conservation Resource Center 2017). Although all new discharges to the catchments of the Buffalo River are prohibited as part of an anti-degradation strategy (United States Code of Federal Regulations 2012), historical and ongoing pollutant discharges remain (Hovis 2014; Usrey 2013; Brahana *et al.* 2016; Watershed Conservation Resource Center 2017). Protecting and maintaining the integrity of the natural resources of the Buffalo River is a high priority because this river also serves as a major economic contributor to the region largely through tourism and park visitation (Cui *et al*. 2013; Cullinane *et al*. 2014).

Aquatic invertebrates are an important tool for understanding and detecting changes in ecosystem integrity, and they can be used to reflect cumulative impacts that cannot otherwise be detected through traditional water quality monitoring (Barbour *et al.* 1999; Moulton *et al*. 2000, 2002). Benthic community structure can be quantified to reflect stream integrity in several ways, including the occurrence of pollution sensitive taxa, dominance by a particular taxon combined with low overall taxa richness, or appreciable shifts in community composition relative to a reference condition (Lazorchak *et al*. 1998; Barbour *et al*. 1999; Bonada *et al.* 2006).

Stream assessments using aquatic invertebrates are typically short-term, single events aimed at assessing stream integrity for a given section of stream in relation to stressors such as bacterial or chemical pollution, and habitat disturbance. By comparison, long-term monitoring at fixed, permanent sites is much less common. Such long-term monitoring is particularly important because the variability over time of metrics used in bioassessments has been shown to be high in other studies (Bruce 2002; Jackson and Füreder 2006; Mazor *et al*. 2009; Vaughan and Ormerod 2012; Bowles *et al*. 2013a, 2013b). Evaluation of long-term variability helps researchers and managers better understand alterations in stream condition relative to climatic variability and change, as well as other anthropogenic disturbances (Jackson and Füreder 2006; Vaughan and Ormerod 2012).

There have been several previous studies conducted on stream invertebrate communities at BUFF for the purpose of assessing water quality impacts and ecological integrity (see Bowles *et al*. 2007 for review). They include Kittle (1975), Geltz and Kenny (1982), Bryant 1997, Mathis (1990, 2001), Mott (1997), Radwell (2000), and Usrey (2001). All of these works exist as gray literature and have not been published. Additionally, these studies were based on either single season events, or multiple season events within the same year. Other aquatic invertebrate studies at BUFF have attempted to take a more comprehensive and long-term approach to assessing invertebrate community dynamics and stream integrity. For example, Mathis (2001) developed an Index of Community Integrity (ICI) for the Buffalo River based on multiple metrics from seasonal collections within the river basin.

The National Park Service's Heartland Inventory and Monitoring Network (HTLN) began monitoring at BUFF in 2005. Bowles *et al*. (2007) included the ICI in the original monitoring protocol to assess long-term aquatic invertebrate community structure at fixed, randomly selected sites at BUFF and directed towards maintaining the ecological integrity of the river and its tributaries. Subsequently, the ICI was not selected for further use because it was judged inferior to the simpler Stream Condition Index (SCI) developed for neighboring Missouri (see DeBacker *et al*. 2012). Bowles *et al*. (2013c) presented a summary of the first few years of this monitoring program. A previous study addressed aquatic invertebrate communities in BUFF tributaries (Mixon-Hinsey 2008).

Here, the results of monitoring aquatic invertebrate community structure and habitat at permanent mainstem Buffalo River sites conducted from 2005 to 2013 are summarized.

## Methods and Materials

### *Site Selection*

Sampling was conducted at 6 permanent mainstem river sites on the Buffalo River annually from 2005 to 2009, and again in 2011 and 2013 (Fig. 1). See Bowles *et al.* (2007) for a description of site selection and supporting data. All samples were collected from riffles during a November through February index period with most samples being collected during December and January. Site 1 was dry during the index period in 2005 and could not be sampled, and in 2006 Site 6 was flooded during most of the index period and also could not be sampled.

### *Aquatic Invertebrates*

Three benthic invertebrate samples were collected from each of three successive riffles at each sampling site using a Slack-Surber sampler (500 µm mesh, 0.25 $m^2$, n=9). The sample area was agitated for 2 minutes with a garden cultivation tool, and large pieces of substrate were scrubbed with a brush as necessary to


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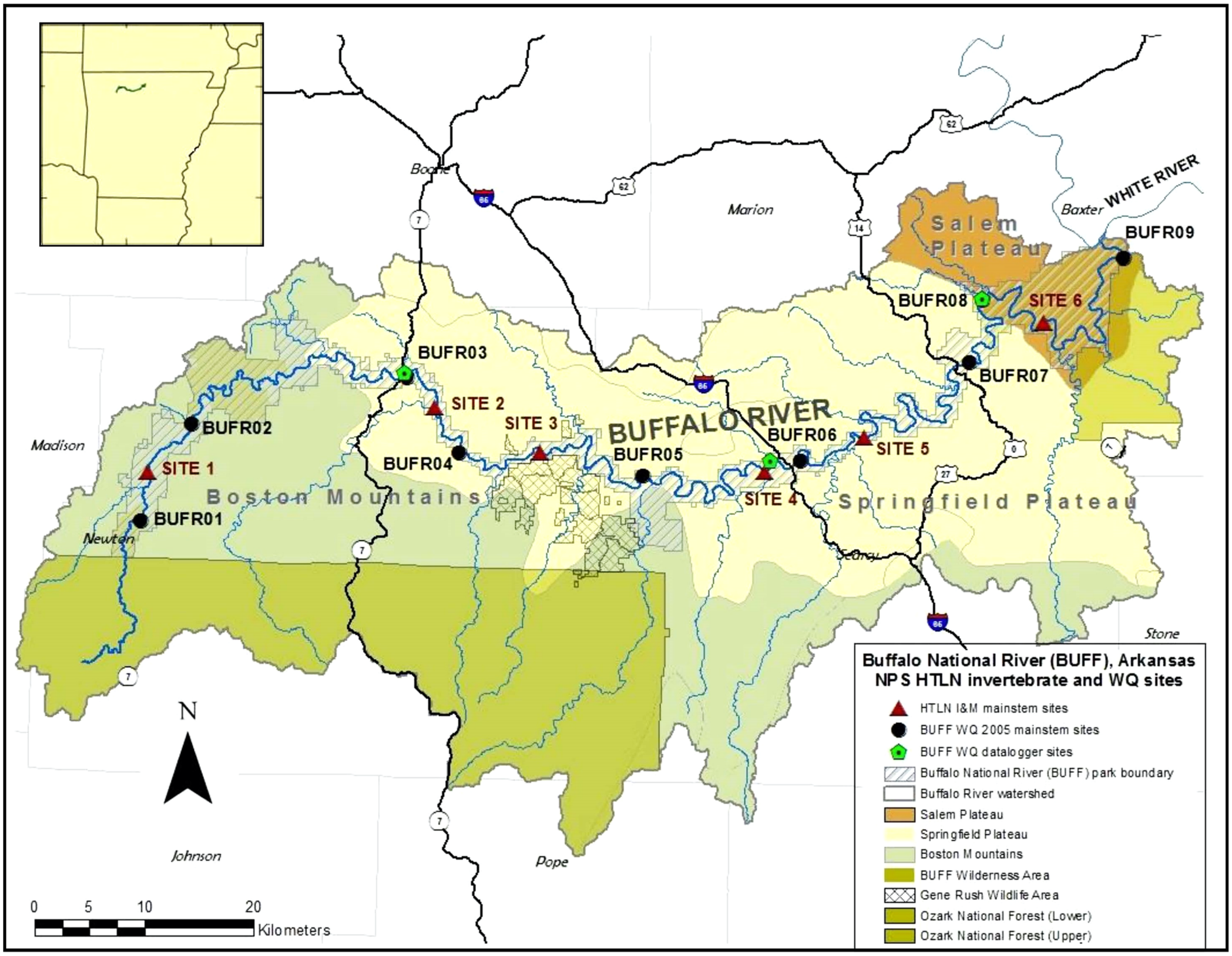


Figure 1. Location of water quality and benthic invertebrate sampling sites on the Buffalo River. BUFF water quality sampling locations are black circles, HTLN monitoring sites are red triangles, and data logger sites are green pentagons.

remove attached invertebrates. Samples were placed in plastic jars and preserved with either 99% isopropyl or 95% ethyl alcohol. Samples were sorted in the laboratory following a subsampling routine described in Bowles *et al*. (2007), and taxa were identified to the lowest practical taxonomic level (usually genus) and counted.

In addition to sampling conducted by the HTLN, BUFF natural resources staff collected invertebrate samples from nine mainstem Buffalo River water quality sites during a BUFF water quality bioassessment study in 2005 (Fig. 1). The data from that study are maintained in the HTLN database (HTLN 2016). Collection methods used by BUFF staff were analogous to those reported here and the data can therefore be directly compared. Data from that study are analyzed in this report for the purpose of comparison to our monitoring sites and data to provide a broader picture of invertebrate community structure and integrity.

### *Multi-metric Index*

The Stream Condition Index (SCI), a multi-metric index developed by Rabeni *et al*. (1997) for the state of Missouri, was used to assess integrity of invertebrate community data. The SCI is a multi-metric index founded on data collected from 26 reference streams in the Ozarks region (Rabeni *et al*. 1997). This index is calculated using four metrics as measures of community structure and balance, including taxa richness, EPT (Ephemeroptera, Plecoptera, Trichoptera) richness, Shannon's diversity index, and Hilsenhoff Biotic Index (HBI; Hilsenhoff 1982, 1987, 1988). High values are preferred for all metrics, except for HBI, where smaller values are the desired response. An increase in HBI values over time is undesired, because that would reflect the community's increasing tolerance to disturbance. See Bowles *et al*. (2007) for sources of assigned tolerance values. The chosen metrics are sound measures of community structure and balance and are generally considered sufficiently sensitive to detect a variety of potential pollution problems in Ozark streams (Rabeni *et al.* 1997) (Table 1). All metric values used are normalized so that they become unitless and can be compared, and have equal influence on the SCI results. The lower or upper quartile of the distribution for each metric is used as the minimum value representative of reference conditions (Table 1). Mean metric values were established by averaging the values for each of three samples per riffle and then averaging the means for the three riffles to establish a site mean (n=3). Procedures for calculating and scoring these four metrics and the SCI can be found in Bowles *et al*. (2007) and Sarver *et al*. (2002). The SCI produces three possible levels of stream condition: 1) fully biologically supporting (unimpaired), 2) partially biologically supporting (impaired), and 3) non-biologically supporting (very impaired). Unimpaired or reference sites score ≥16 and have the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region. Both partially biologically supporting (SCI 10-14) and non-biologically supporting (SCI 4-8) categories indicate impaired streams that do not meet the beneficial use of protection of aquatic life.

### *Habitat and Water Quality Assessment*

Dominant substrate was visually estimated from three randomly selected pieces within the sampling net frame using the Wentworth scale (Wentworth 1922). Depth (cm) and current velocity (m/sec) were measured immediately in front of the sampling net frame using a top-setting wading rod fitted with a Marsh-McBirney Flow-Mate 2000 flow meter. Qualitative habitat variables (percent substrate embeddedness, periphyton, filamentous green algae, and aquatic vegetation) were estimated within the sampling net frame as percentage categories (0, <10, 10-40, 40-75, >75). Habitat data were analyzed as midpoints of each category across years for each site.

Static readings of water quality parameters (temperature, dissolved oxygen, specific conductance, pH) were recorded at each riffle sampled with calibrated, hand-held instruments (YSI models 55, 63, ProPlus). In addition, hourly readings of water quality parameters (temperature, dissolved oxygen, specific conductance, pH, turbidity) were recorded continuously at 1 hour intervals at least 1 week prior to sampling using calibrated data loggers (YSI models 6600, 6920) at three fixed sites on the Buffalo River located near Site 2, Site 4, and between Site 5 and Site 6 (Fig. 1). The water quality data collected for this study are only intended to describe the prevailing conditions that may influence the structure of invertebrate communities, and they represent only a small snapshot of the broader range of possible conditions over longer periods. Due to the limitations of using water quality data obtained with data loggers, the invertebrate community is used here as a surrogate of long-term water quality conditions. Water quality data are summarized across years and presented as single means to represent each site.

### *Statistical Analysis*

Sørensen's Similarity Index (presence/absence) was used to analyze similarity of taxa occurrences among the different sampling sites (Southwood and Henderson 2000; Hammer *et al*. 2001). Similarity index scores among sites were analyzed using ascendant hierarchical cluster analysis (Ward 1963) following the recommendation of Magurran (2004).

Pairwise correlation coefficients for each pair of metrics and habitat variables were calculated using nonparametric Kendall's tau (Daniel 1990) because examination of histograms revealed lack of normality for many of the habitat variables. This analysis evaluated correlations between the four biological metrics calculated from aquatic invertebrate samples and 11 habitat variables. The habitat variables included: embeddedness, vegetation, periphyton, algae, depth, velocity, substrate size, dissolved oxygen, temperature, specific conductance, and pH. Data were grouped separately and analyzed by year and by site. When grouped by year, all riffles from all sites were included in the same analysis, and the analysis was repeated for each year (N = 7 years; n = 18 observations for each correlation: 3 riffles x 6 sites) (4 metrics x 11 habitat variables x 7 years = 308 total correlations). This approach provided the strongest level of independence among observations. When grouped by site, all years of data for all riffles of each site were included, and the analysis was repeated for each site (N = 6 sites; n = 21 observations for each correlation: 3 riffles x 7 years) (4 metrics x 11 habitat variables x 6 sites = 264 total correlations). Such analyses produced many correlation coefficients and P-values, with an unknown actual Type I error rate. Thus, a meta-analytic approach was applied in interpreting the results. The number of significant ($P<0.05$) correlations was summarized for each pair of


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Table 1. Descriptive statistics, quartiles and scores for aquatic invertebrate metrics calculated using single habitat coarse substrate (riffle) data during a fall index period (from Rabeni *et al*. 1997). Summary statistics are from riffle habitat of reference streams (n=5) in the Ozark ecoregion during the fall index period.

| Metric | Statistics | | | | Quartiles | | | Scores | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | Mean | Standard Error | Minimum | Maximum | 25% | 50% | 75% | 5 | 3 | 1 |
| Taxa Richness | 28.3 | 3.3 | 23.5 | 41.0 | 21 | 26 | 29 | >=21 | 20-11 | <11 |
| EPT Richness | 13.1 | 0.7 | 11.5 | 15.0 | 9 | 11 | 12 | >=9 | 8-5 | <5 |
| HBI | 4.3 | 0.3 | 3.3 | 5.0 | 3.6 | 4.9 | 5.3 | <=5.3 | 5.4-7.7 | >7.7 |
| Shannon's Diversity Index | 2.4 | 0.1 | 2.1 | 2.7 | 2.29 | 2.44 | 2.61 | >=2.29 | 2.28-1.15 | <1.15 |

SCI Scoring: ≥16 not impaired, 10-14 impaired, 4-8 very impaired.

metrics and habitat variables. Then percentage of significant correlations for each pair of metrics and habitat variables, summarized over all metrics, was determined. Although it is unknown which correlations may be spurious, habitat variables with a greater overall percentage of significant correlations are likely to have, in general, greater potential to explain variability in these metrics. SPSS version 20.0 was used to calculate correlation coefficients (IBM Corp. 2011).

## Results and Discussion

### *Aquatic invertebrates*

The aquatic invertebrate fauna of the Buffalo River is diverse and many taxa are shared across sampling sites. Among all sites, 167 distinct taxa were identified with similarities ranging from 70% to 83% (Table 2). Because Chironomidae were not identified beyond family level, the number of distinct taxa is likely much higher. Considering the Chironomidae at the family level does not appreciably change the metrics used in this paper (Rabeni and Wang 2001). A complete list of invertebrate taxa at each site, their abundances and associated environmental data are too voluminous to present here, but can be obtained from the senior author. Cluster analysis showed that sites are clustered in a longitudinal progression (Fig. 2). Generally, those sites closest to one another in linear distance were most closely related (Fig. 1, Fig. 2). The exception was site 2, which formed a cluster with site 1 rather than with site 3 as expected, and this cluster was distinct from the remaining sites. This may be partially due to the physical conditions at those sites and stressors acting on the invertebrate communities. Site 1 typically has lower specific conductance (Fig. 8C) and larger substrate size (Fig. 6A) compared to the other sites, and it often has intermittent flows, especially during late summer. Such environmental and habitat conditions are likely reflected in the invertebrate community structure observed at this location. Site 2 is located about 3.5 km downstream of Mill Creek, which has had ongoing high loadings of human coliform bacteria (Usrey 2013). Manner and Mott (1991) found that 96% of the nitrogen load being carried by the Buffalo River below the confluence with Mill Creek was supplied by this stream, and the contamination likely came from the interbasin transfer of groundwater within a nearby watershed.

The metric values recorded clearly exceeded the minimum reference stream values (maximum for HBI) in some years, but not in others (Table 1, Figs. 3A-D). With the exception of HBI, values tended to decrease in a downstream direction to Site 4, and then increase to levels observed upstream. Such variation may not be biologically significant and may be due to the stretch of river upstream of this site experiencing seasonal drying and intermittent flows during most summers. Taxa richness was typically greater than 20 among samples. It is noteworthy that representatives of the intolerant EPT orders were abundant across all sites, and EPT richness values consistently were greater than 12 for all sites in most years. EPT values generally were high relative to Ozark reference streams (Table 1), although not as high as for other regional streams (Bowles *et al*. 2016).

Overall, the invertebrate taxa of the Buffalo River are largely intolerant (mean tolerance value=4.38), and HBI values generally were below 5. Tolerant taxa (tolerance values ≥5) were present in most samples, but they were generally not as well represented in the benthos as intolerant taxa. Individual metrics were highly variable among years and sites, although such among the invertebrate communities shows the importance of using a multi-metric index for stream assessment and multi-year sampling so that too much variability is not unexpected (Mazor *et al*. 2009). HBI values of 5.5 or less are generally considered good, although some organic pollution may be possible (Hilsenhoff 1982, 1988). Mean HBI across years for all

Table 2. Sørensen similarity index for aquatic invertebrate taxa among collecting sites at the Buffalo National River, Arkansas.

| | Site 2 | Site 3 | Site 4 | Site 5 | Site 6 |
|---|---|---|---|---|---|
| Site 1 | 0.70 | 0.79 | 0.76 | 0.76 | 0.72 |
| Site 2 | | 0.77 | 0.75 | 0.73 | 0.73 |
| Site 3 | | | 0.83 | 0.82 | 0.80 |
| Site 4 | | | | 0.81 | 0.74 |
| Site 5 | | | | | 0.81 |

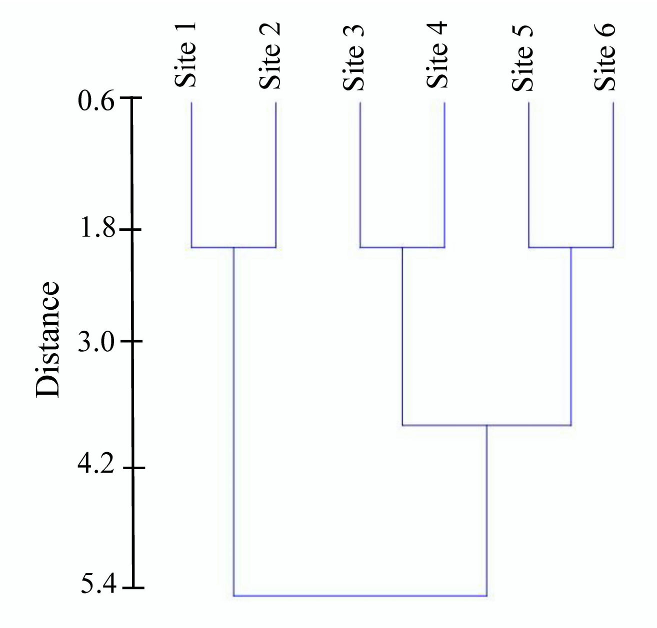


Figure 2. Dendogram showing results for ascendant hierarchical cluster analysis and relative distance of Sørensen's similarity index scores of the aquatic invertebrate communities at sampling sites along the Buffalo River, Arkansas, 2005-2013.

sites ranged from 4.42 to 4.78, which reflects good conditions. Shannon's diversity index values generally ranged from 2.0 to 2.5 among sites and years. Values for Site 4 were generally less than 2, however. For biological data, Shannon's diversity index ranges generally from 1.5 (low species richness and evenness) to 3.5 (high species evenness and richness) (McDonald 2003), but the actual value is contingent on the number of species in the community. The variability observed weight is not placed on the value of a single metric or year. Environmental stressors, such as extended drought and flooding, may impact invertebrate communities and influence assessment results in any given year (Bunn and Arthington 2002; Lake 2003).

SCI scores among sites and years were variable, but they showed that sampling sites are generally not impaired and are fully biologically-supporting (Figs. 4A-F). The lower scores observed in some years are likely due to interannual variability of invertebrate communities coupled with flow dynamics (flood, drought) that occur at those sites rather than anthropogenic disturbances. These data also show the importance of collecting data during multiple years and at multiple sites so that low scores in any given year do not unduly influence management decisions for corrective actions (Mazor *et al*. 2009). SCI scores calculated from data collected during an earlier study conducted by BUFF staff (HTLN 2016) showed a similar response to data collected during this study (site means for all years) with scores being lowest in the mid reaches of the river but then increasing to values similar to those observed upstream (Fig. 5). This finding lends further support to the idea that the losing reaches upstream of site 4 are influencing downstream invertebrate community structure. The higher SCI value for BUFR05 is based on a single sampling event and therefore may not be entirely representative of the range of variation that occurs at that site.

Although the Buffalo River may be classified as relatively high quality, some anthropogenic impacts have occurred there and other threats are ongoing. Previous water-quality and invertebrate community monitoring at BUFF (Mathis 1990; Bryant 1997; Mott 1997; Usrey 2001; Mott and Luraas 2004) showed strong negative correlations between nonpoint source pollution (fecal coliform bacteria, nitrates, phosphorus), stream water quality, and invertebrate community structure along the river's course. In some instances, non-point source pollution has substantial inputs to the river. For instance, Usrey (2001) reported that nitrogen levels of four mid-reach tributaries of the Buffalo River (Mill Creek, Little Buffalo River, Big Creek, and Davis Creek) represented approximately 40% of the total nitrogen loading to the river and average nitrate values were 2 to 4 times higher in these tributaries than in the adjacent river. Usrey (2001) also found that the decreasing abundance of pollution intolerant EPT taxa was associated with higher nitrate concentrations, and increasing orthophosphate concentrations were positively correlated with increasing densities of pollution tolerant dipterans. Inadequately treated wastewater discharges in the Mill Creek watershed continues largely unabated (Watershed Conservation Resource Center 2017). Thus, nutrient loading of the Buffalo River may be among the most significant threats


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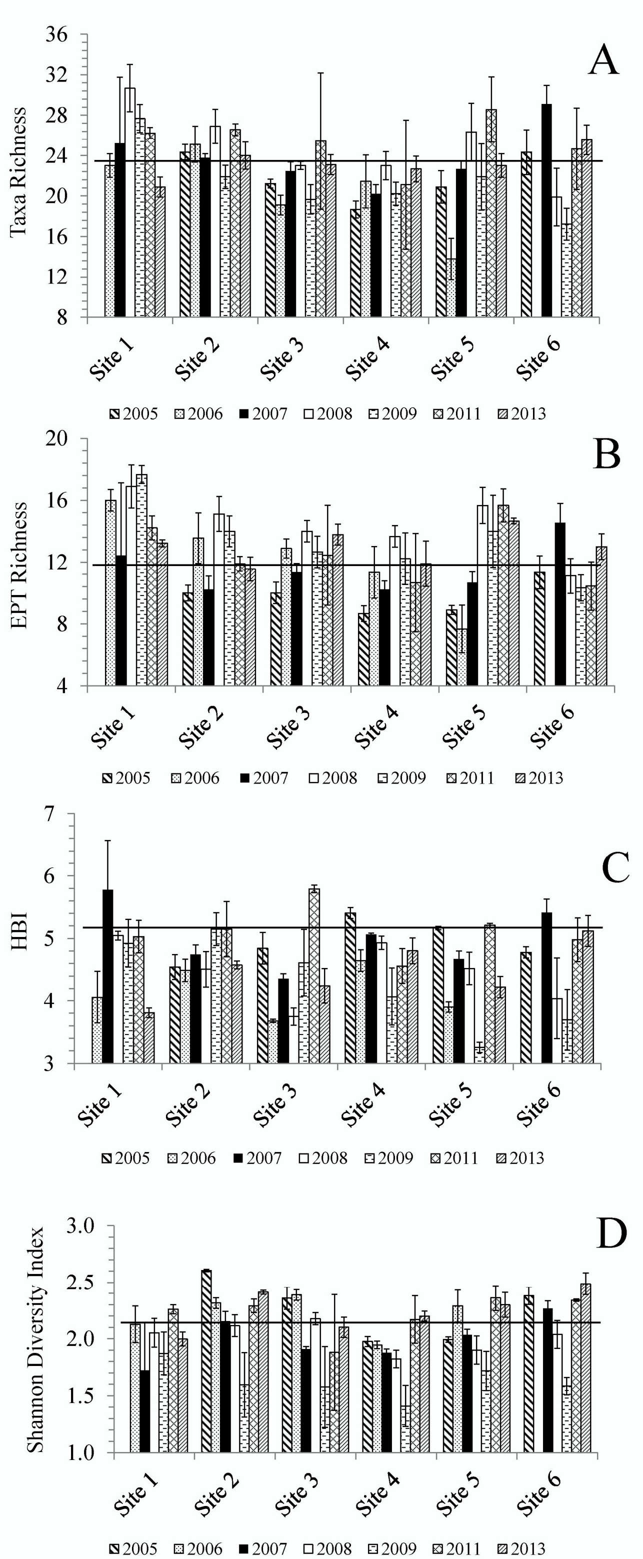



Figs. 3A-D. Aquatic invertebrate community metrics for the Buffalo River, Arkansas, 2005-2013. Values are means and error bars represent one standard error. The horizontal line conforms to the minimum reported value for Ozark reference streams, except for HBI, which is the maximum reported value (from Rabeni *et al.* 1997).

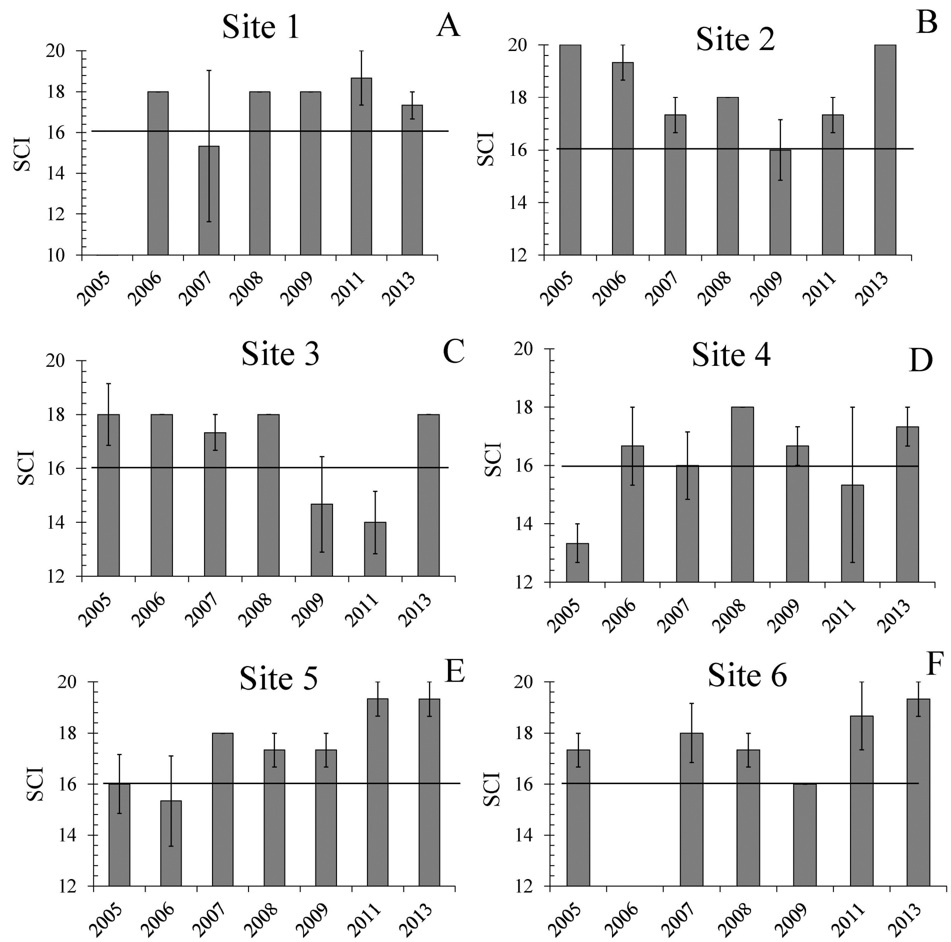
to the integrity of its resident biological communities. The present and previously reported data collectively show the utility of using aquatic invertebrates for assessing water quality integrity. The data also show that mainstem river water quality can be degraded from impairments to tributaries, which in turn degrades biological communities.

### *Habitat and Water Quality*

Mean riffle depth where samples were collected ranged from around 20 to 35 cm, and mean current velocities ranged from about 0.40 to 0.95 m/sec. Substrate was larger at Site 1 (Wentworth Scale=15-16, 45-90 mm, large cobble) compared to the other sites, all of which had similarly sized substrate (Wentworth Scale=13-15, 32-64 mm, large pebble) (Fig. 6A). Substrate embeddedness was similar at most sites generally, ranging from 25 to 30%, but was least at the upstream most site (~20%) and slightly higher at the downstream most site (~40%) (Fig. 6B).

Aquatic vegetation (mostly mosses) and filamentous green algae were poorly represented at all sampling sites (<20%) and those data are not presented here. Periphyton densities growing on the rock substrates were generally consistent at the upper 3 sampling sites and at site 6 (~25%), but were frequently higher at sites 4 and 5 (~35% and 30%, respectively) (Fig. 7). Sites 4 and 5 are downstream of the Woolum Access of the Buffalo River and this stretch of river has two prominent losing reaches where surface flows are periodically diverted completely to subsurface flow, especially during summer (Moix and Galloway 2004). These losing reaches are approximately 5 km and 4.5 km long, respectively, and are separated by a 4 km long gaining reach. The latter losing reach ends less than 1 km upstream of site 4. It is possible that this losing reach located above the sampling site may stimulate increased growth of periphyton at those sites due to increased temperatures and nutrient loading associated with the resulting pooling of the river (Petersen and Femmer 2002). Upstream nutrient loading from tributaries could also play a role in stimulating growth. Shorter losing reaches (~2 km) are located in the upper Buffalo River including one located immediately upstream and partially overlapping with site 1, but that site has been dewatered only once during our sampling window (2005).

Habitat conditions were generally consistent among sites and years. Overall, no habitat variables exhibited persistently strong correlations with any of the metrics, and the percentage of "significant" correlations was relatively low (<30%) in all cases (Table 4). In addition,




Figs. 4A-F. Mean SCI values and standard errors for collecting sites on the Buffalo River, 2005-2013. The horizontal line represents an SCI of 16, the lower limit for rating a site unimpaired.

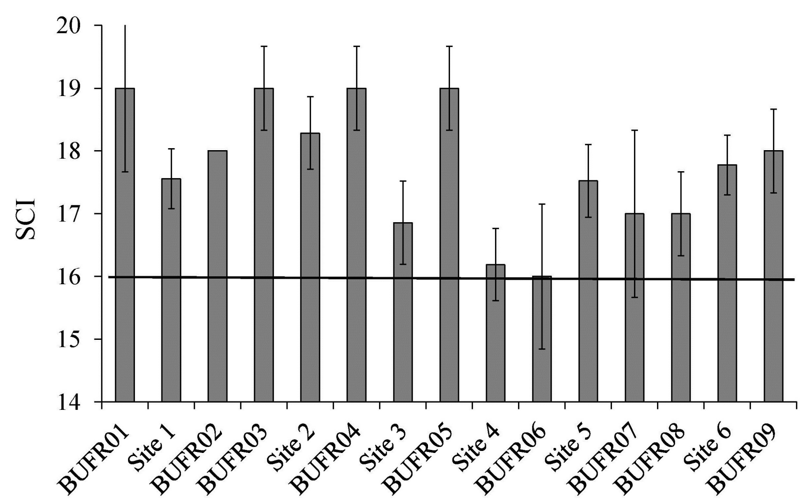



Figure 5. Mean SCI scores and standard errors for Buffalo River water quality bioassessment sampling sites collected in 2005 and Heartland Inventory and Monitoring Network sampling sites (2005-2013). See Figure 1 for site locations. The horizontal line represents an SCI of 16, the lower limit for rating a site unimpaired.



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Table 3. Summary of BUFF pairwise correlations (Kendall's tau) organized by year (i.e., correlations conducted among all sites in each year) and by site (i.e., correlations conducted among all years at each site). Values are number of significant correlations/percentage of significant correlations of total.

| Variables | HBI | Taxa Richness | EPT Richness | Shannon's Diversity | **Total** |
|---|---|---|---|---|---|
| | | | By Year | | |
| Filamentous algae | 2/0.29 | 2/0.29 | 1/0.14 | 1/0.14 | 6/0.21 |
| Current velocity | 1/0.14 | 2/0.29 | 1/0.14 | 1/0.14 | 5/0.18 |
| Dissolved oxygen | 2/0.29 | 1/0.14 | 1/0.14 | 1/0.14 | 5/0.18 |
| Temperature | 2/0.29 | 1/0.14 | 0/0 | 1/0.14 | 4/0.14 |
| Substrate size | 1/0.14 | 1/0.14 | 2/0.29 | 0/0 | 4/0.14 |
| Specific conductance | 2/0.29 | 0/0 | 1/0.14 | 1/0.14 | 4/0.14 |
| Substrate embeddedness | 0/0 | 1/0.14 | 2/0.29 | 0/0 | 3/0.11 |
| pH | 0/0 | 1/0.14 | 1/0.14 | 1/0.14 | 3/0.11 |
| Periphyton | 1/0.14 | 1/0.14 | 1/0.14 | 0/0 | 3/0.11 |
| Depth | 0/0 | 1/0.14 | 1/0.14 | 0/0 | 2/0.07 |
| Vegetation | 1/0.14 | 0/0 | 0/0 | 0/0 | 1/0.04 |
| Total | 12/0.16 | 11/0.14 | 11/0.14 | 6/0.08 | 40/0.13 |
| | | Expected number of spurious correlations = 15 | | | |
| | | | By Site | | |
| Filamentous algae | 0/0 | 3/0.50 | 4/0.67 | 0/0 | 7/0.29 |
| Current velocity | 1/0.17 | 2/0.33 | 2/0.33 | 1/0.17 | 6/0.25 |
| Dissolved oxygen | 2/0.33 | 1/0.17 | 1/0.17 | 2/0.33 | 6/0.25 |
| Temperature | 0/0 | 2/0.33 | 2/0.33 | 1/0.17 | 5/0.21 |
| Substrate size | 2/0.33 | 1/0.17 | 0/0 | 1/0.17 | 4/0.17 |
| Specific conductance | 1/0.17 | 1/0.17 | 1/0.17 | 1/0.17 | 4/0.17 |
| Substrate embeddedness | 0/0 | 1/0.17 | 2/0.33 | 1/0.17 | 4/0.17 |
| pH | 1/0.17 | 0/0 | 1/0.17 | 1/0.17 | 3/0.13 |
| Periphyton | 1/0.17 | 0/0 | 1/0.17 | 1/0.17 | 3/0.13 |
| Depth | 0/0 | 0/0 | 0/0 | 1/0.17 | 0/04 |
| Vegetation | 0/0 | 0/0 | 0/0 | 0/0 | 0/0 |
| Total | 8/0.12 | 11/0.17 | 14/0.21 | 10/0.15 | 43/0.16 |
| | | Expected number of spurious correlations = 13 | | | |

a certain number of spurious correlations are expected (1 in 20 for alpha = 0.05) in analyses such as those conducted here. The number of expected spurious correlations ranged from 22 to 38% of the observed "significant" correlations (Table 3). Algae, current velocity, dissolved oxygen, temperature, substrate, and specific conductance usually had a greater percentage of "significant" correlations than the other variables, across all analyses, but some of these variables are autocorrelated, hence their biological significance may not be relevant (Martínez-Abraín 2008). The low number of significant correlations for some habitat variables is likely due to the categorical scale used to assess some habitat data (see Methods), and the lack of variability in the values observed for these variables. This analysis shows that the habitat data collected in relation to benthic invertebrate samples presently has limited value for correlating with community and diversity metrics, but that finding does not rule out further analyses with individual invertebrate taxa or groups of

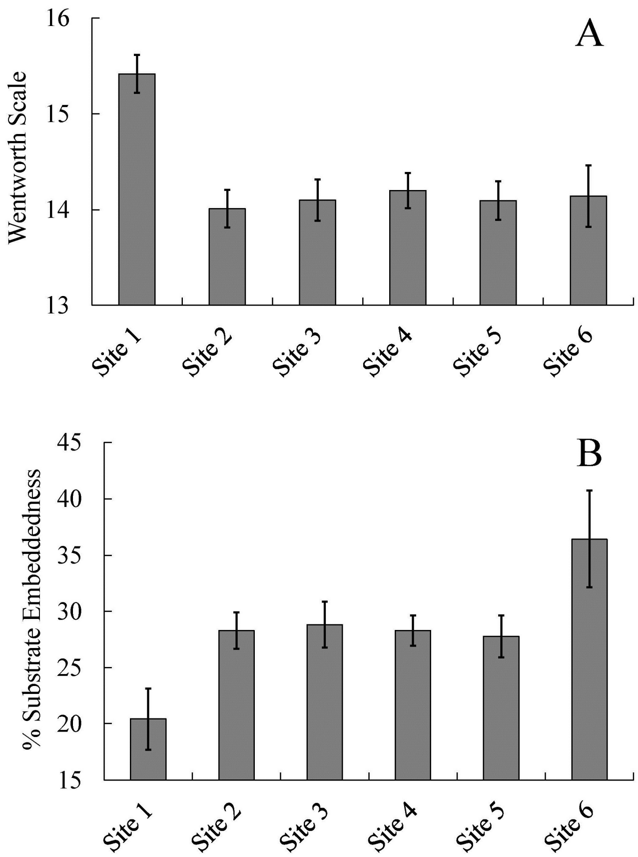


Figure 6A-B. Mean substrate size (Wentworth scale) and percent substrate embeddedness associated with benthic invertebrate samples from the Buffalo River, Arkansas, 2005-2013. Error bars represent one standard error.

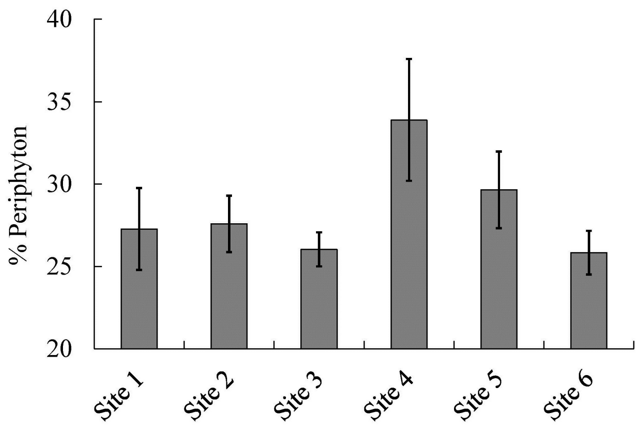


Figure 7. Percent periphyton associated with benthic invertebrate samples from the Buffalo River, Arkansas, 2005-1013. Values are means; error bars represent one standard error.

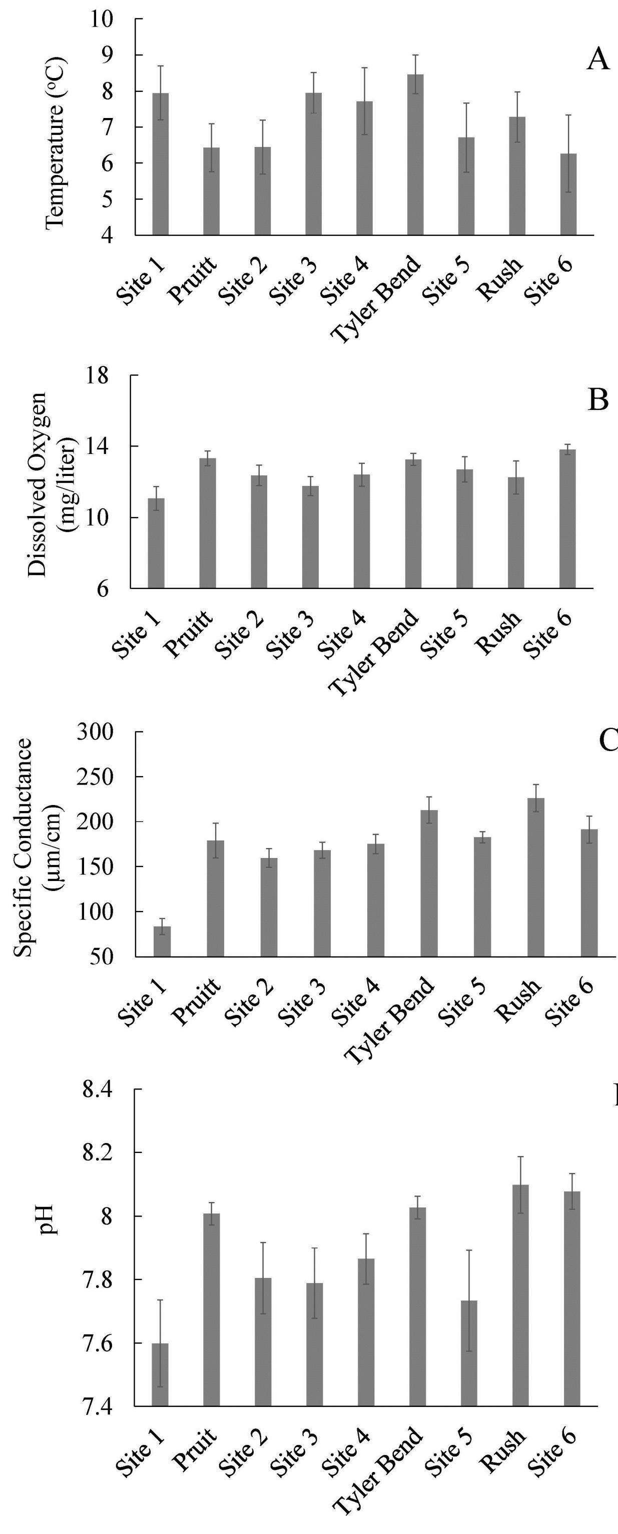


Figure 8A-D. Water physical-chemical data for sampling sites on the Buffalo River, Arkansas, 2005-2013. Values are means with standard errors. Data were collected as static readings using hand-held meters at sampling sites 1-6, while data were collected continuously using dataloggers at Pruitt, Tyler Bend and Rush locations. See methods for other details.



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taxa (e.g., EPT). Accordingly, only summary data are presented here to generally characterize the conditions in which samples were collected, and a further analysis of this data is beyond the scope of this paper.

Water quality met state standards in all instances (Arkansas Pollution Control and Ecology Commission 2017) (Fig. 8A-D). Temperature was variable among sampling sites and years, which was expected due to climatic variations among years sampled as well as location of sampling sites along the length of the river. Dissolved oxygen levels were high in all instances and were at or above saturation across years and sites (means=11.1-13.9, range 8.4-15.3 mg/liter). Specific conductance was lowest at the upstream most sampling site across years (mean=83.5 µm/cm, range 48.5-126.7 µm/cm) while mean values increased in a downstream direction for the other sites (means=154, 170, 175, 184 and 192, respectively µm/cm). pH was consistent and similar among all sampling sites and years sampled (means=7.6-8.1), and values are reflective of the karst topography of the Buffalo River basin. Turbidity, not shown here, was nearly always below 10 NTU. The water quality values we report are consistent with those recorded by other studies (Moix and Galloway 2004, Huggins *et al*. 2005, Watershed Conservation Resource Center 2017) with the exception of temperature because their data were recorded during different seasons.

## Conclusions

This paper provides baseline invertebrate, habitat and water quality data for selected sites on the Buffalo River, Arkansas. Invertebrate community structure in the Buffalo River generally is diverse and reflects above average water quality. Inherent variability of invertebrate community diversity and density across sites and years highlights the importance of multiyear assessment and monitoring to support management decisions. Although the condition of invertebrate communities and water quality in the Buffalo River exceeded water quality standards and have high integrity, numerous ongoing and projected threats to these resources remain, and those threats largely originate outside of the park's jurisdictional boundaries. Aquatic invertebrate monitoring at BUFF provides a sound tool to recognize both deterioration and chronic decline of water quality.

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