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Running Title: Water Quality Assessment of Sager Creek

Abstract

An annual rapid bioassessment and physiochemical survey of Sager Creek in Northwest Arkansas was conducted. Sager Creek is a first to second order stream that flows through the city of Siloam Springs, AR. Invertebrate collections and water samples were collected at three different reaches, with the most downstream reach being below the effluent of the Siloam Springs Wastewater Treatment Plant. Benthic arthropods were collected, identified, and counted to produce a family-level biotic index and a family-level index of diversity. Statistical analysis revealed that these indices were significantly different for the effluent- influenced reach. However, this difference could not be correlated to any measured physiochemical parameter.

Key words:--- Aquatic insects, macroinvertebrates, rapid bioassessment, water quality

Introduction

Benthic macroinvertebrates possess several characteristics that make them especially useful in assessing water quality. First, they occupy several trophic levels and are principle components in stream food webs. They typically have life cycles which extend over multiple seasons and experience varying environmental conditions. They tend to display low motility within the stream benthos and also show varying levels of tolerance to environmental conditions including stream pollution (Kuep et al. 1966).

In 1988, in recognition of a need for a rapid field-based assessment tool, Hilsenhoff published a biotic index based on the pollution tolerance levels of families of benthic arthropods. Although the family-level biotic index (FBI) tended to overestimate the pollution level of clean streams and underestimate the pollution level of polluted streams, compared to a species-level biotic index (BI), it still provided valuable information for assessing water quality in lotic environments (Hilsenhoff 1987, 1988).

Diversity indices are also used to evaluate the structure of macroinvertebrate communities. Simpson's Index of Diversity (SID) calculates the probability that two sampled organisms will belong to different taxonomic groups (Simpson 1949). In other words, as diversity increases, the probability that the two individuals sampled will belong to different taxonomic groups also increases. Although SID is most commonly applied at the species level, taxonomic sufficiency (Ellis 1985) has been demonstrated at higher taxonomic levels in both marine and freshwater systems (Warwick 1988, Marchal 2005, Marshall et al. 2006, Jones 2008). Significant correlations between reduced taxonomic diversity and polluted water have also been indicated (Wright et al. 1993, Nedeau et al. 2003).

The practice of using benthic macroinvertebrate surveys as an assessment of water quality has been applied to many streams and creeks in Arkansas (Shackleford 1988, Brown et al. 1997, Burns 2001, Williams et al. 2002, Grippo and McCord 2006, McCord et al. 2007, Brueggen-Boman and Bouldin 2012). However, no macroinvertebrate stream assessment study has been published on Sager Creek, a small stream in Northwest Arkansas.

The objectives of this study were to: (1) provide baseline water quality conditions of the understudied Sager Creek (2) determine if rapid bioassessments using benthic macroinvertebrates is sensitive enough to assess the health of Sager Creek.

Materials and Methods

The Sager Creek watershed, which is located in the Ozark Highlands Ecoregion of Northwest Arkansas, (Omernick 1987) encompasses approximately 44 km^2. Wet weather tributaries of the stream extend east of the city of Siloam Springs by as much as 6.7 km. However, the principle flow of Sager Creek begins at

Box Springs, an underground aquifer that opens to the surface on the Siloam Springs Municipal Golf Course (GBM[c] & Associates 2005). This first to second order stream (Vannote et al. 1980) flows west through the city of Siloam Springs, through the campus of John Brown University and into Oklahoma. Approximately 300 m from the state line, effluent from the Siloam Springs Wastewater Treatment Plant is discharged into the stream.

Sampling of Sager Creek occurred during September of 2009 and continued through July of 2010. A total of three test reaches are located along Sager Creek. The Honeycutt reach (H) is the highest upstream site and is located near Box Springs. The JBU reach (JBU) is located to the north and northeast of the John Brown University campus and is downstream from the business district of Siloam Springs. The Wastewater reach (W) is the farthest downstream and is below the Wastewater Treatment Plant effluent. Each of the three reaches was further broken down into eight individual riffle sites labeled A-H.

A 500-micron D-net was used to collect benthic macroinvertebrate samples. The D-net was placed randomly in the riffle, downstream of the water-flow, and an approximate 0.30 meter by 0.30 meter area was scrubbed for thirty seconds in front of the D-net to dislodge the organisms. This process was performed at two different locations within each of the eight sites contained in a reach. The sample was then transferred from the D-net through a 5-mm rock screen into a bucket. The screen and D-net were inspected and all clinging organisms were removed. Organisms that would contribute to the FBI (i.e. insect larvae) were transferred to the bucket, while noncontributing invertebrates (i.e. Decapodans, Oligochaetes and Hirudineans) were discarded. The sample was then filtered through a 500-micron screen to remove excess water. The final sample was emptied into a collection container and preserved with 95% ethyl alcohol.

In the laboratory, each sample was dispensed into a gridded counting tray and a grid was chosen at random to begin the organism count. A one hundred organism subsample was separated, identified to the family level (Needham and Needham 1962, Voshell 2002), and recorded. A FBI and a family-level SID (FSID) were calculated according to Hilsenhoff (1988) and Simpson (1949) for each subsample. Tolerance values for the FBI were assigned according to the Missouri Department of Natural Resources database (Sarver 2005). A mean family-level biotic index (MFBI) was calculated for each reach utilizing all eight of the individual site FBI. MFBI during the months of June and July were calculated utilizing only four individual site's FBI.

Calculations for stream water flow were performed according to EPA standards (USEPA 2004). Physiochemical data were collected using various means. A Milwaukee portable pH meter (model MW100) was used to record stream pH. A handheld thermometer was used to record stream temperature. These tests were performed at three randomly selected sites at each reach and a mean value for each parameter was recorded. At these same sites, approximately 120 ml of water was collected, according to EPA standards (USEPA 2004), for additional physiochemical tests. The 120 ml unfiltered water samples, were tested for dissolved oxygen (HRDO method 8166), nitrogen (cadmium reduction method 8039), and phosphorous concentrations (USEPA method 365.2) using a Hach™ colorimeter (model DR/850). A mean value for each concentration was calculated and recorded.

Physiochemical data, MFBI and FSID were compared using paired t-tests, with an application of the Bonferroni Multiple Comparison Test (Triola and Triola 2006), with an alpha value set at 0.05.

Results

Arthropods from six different insect orders, representing 17 different families were collected along with two groups of crustaceans; isopods and amphipods (Table 1). Mayflies and isopods were the most commonly counted organisms collected in the Honeycutt reach, averaging 59% and 20% of all organisms counted respectively. The most commonly counted organisms from the JBU reach were again the mayflies (42%) with the true flies (Dipterans) a close second (35%). However the Wastewater reach was almost completely dominated by the true flies (83%) with mayflies the next largest group at only 13%.

According to the FBI established by Hilsenhoff (1988), an increasing value represents an increasing level of organic pollution. The Honeycutt reach's MFBI (5.057) would place it in the "fair" ranking, indicating that a "fairly substantial amount of organic pollution was likely". The JBU reach's MFBI (4.935) would place it in the "good" ranking, indicating "some organic pollution was probable". However, as can be seen in Fig. 1, the Honeycutt reach is only slightly higher than the JBU reach and there was no significant difference between these two values (P = 0.821). The Wastewater reach's MFBI (5.736) was ranked at the very upper end of the "fair" ranking, just below the

Table 1. List of organisms collected, identified, counted and utilized in the production of the mean Family Biotic Index (MFBI) and Family-level Index of Diversity (FSID). All numbers are the average number of individuals identified per reach per sample day.

	Honeycutt						JBU						Wastewater					
	10/7 '09	11/6 '09	2/3 '10	3/10 '10	6/18 '10	7/14 '10	9/30 '09	11/2 '09	1/27 '10	3/3 '10	6/19 '10	7/15 '10	10/12 '09	11/11 '09	2/10 '10	3/29 '10	6/21 '10	7/20 '10
Ephemerotera																		
Baetidae	46.00	73.50	41.88	43.88	53.00	37.25	9.25	8.25	3.50	2.38	43.75	31.50	3.50	2.63	0.00	0.63	2.50	18.00
Caenidae	4.88	4.38	0.50	1.00	0.00	3.00	7.00	17.00	1.38	1.00	0.50	4.00	1.50	1.63	0.00	1.13	0.00	2.25
Heptageniidae	9.50	11.25	6.63	5.13	1.50	1.50	25.25	30.63	25.00	3.63	15.75	11.00	4.75	0.88	0.25	0.38	0.00	1.25
Leptophlebiidae	0.00	0.50	3.13	1.50	3.50	0.00	0.00	0.00	1.88	0.38	0.00	0.00	0.00	0.00	0.13	0.13	0.00	0.00
Isonychiidae	0.00	0.00	0.00	0.38	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Leptohyphidae	0.00	0.00	0.00	0.25	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Odonata																		
Coenagrionidae	2.88	0.88	0.88	0.38	0.00	0.00	2.63	1.13	2.88	0.00	1.00	1.50	0.38	0.25	0.00	0.00	0.00	0.00
Calopterygidae	0.00	0.25	0.00	0.00	0.00	0.00	1.13	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Libellulidae	0.00	0.00	0.00	0.00	0.00	0.00	0.38	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Trichoptera																		
Hydropsychidae	3.25	2.13	2.88	0.13	3.00	6.00	6.88	5.13	4.50	1.50	9.25	9.25	0.50	0.50	0.25	0.25	0.00	3.50
Philopotamidae	3.00	1.75	1.25	0.00	3.50	7.00	2.25	2.50	3.38	0.38	10.00	6.75	0.13	0.13	0.00	0.00	0.00	0.00
Lepidoptera																		
Pyralidae	0.00	0.00	5.75	0.00	0.00	0.00	0.13	0.25	0.00	0.00	0.25	0.50	0.00	0.13	0.00	0.00	0.00	0.00
Coleoptera																		
Elmidae	6.38	9.13	0.00	3.00	5.00	9.50	7.00	1.13	0.50	1.38	3.00	8.00	0.50	0.38	0.00	0.50	0.00	0.00
Psephenidae	0.25	0.00	0.00	0.00	0.00	0.00	16.38	7.13	1.88	0.38	3.75	5.50	1.88	1.25	0.00	0.00	0.00	1.75
Diptera																		
Ceratopogonidae	0.88	0.00	0.50	0.25	0.00	0.00	1.13	0.00	1.38	0.25	0.00	0.00	1.38	0.13	0.00	0.00	0.00	0.00
Chironomidae	1.13	3.50	11.38	26.50	0.00	2.00	15.50	6.75	43.50	83.00	28.00	23.50	28.00	11.63	8.00	22.88	120.00	81.75
Simuliidae	0.00	0.00	0.00	0.13	0.00	0.50	0.63	0.13	0.00	0.00	1.00	0.50	0.25	0.00	0.00	0.25	0.00	0.00
Amphipoda																		
Gammaridae	0.25	0.88	0.13	0.25	0.00	0.50	0.00	0.13	0.50	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Isopoda																		
Asellidae	7.38	18.88	18.75	20.50	41.00	12.50	0.13	0.25	0.13	0.00	0.00	0.50	0.38	0.00	0.25	0.50	0.00	0.00

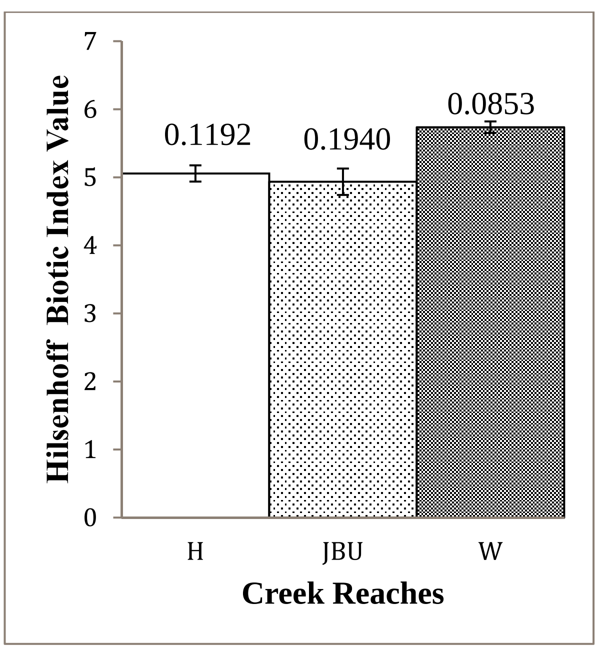

Fig. 1. Mean Family-level biotic index value. Values range from 0 to 10 indicating increasing pollution levels. Standard error bars and values are also indicated.

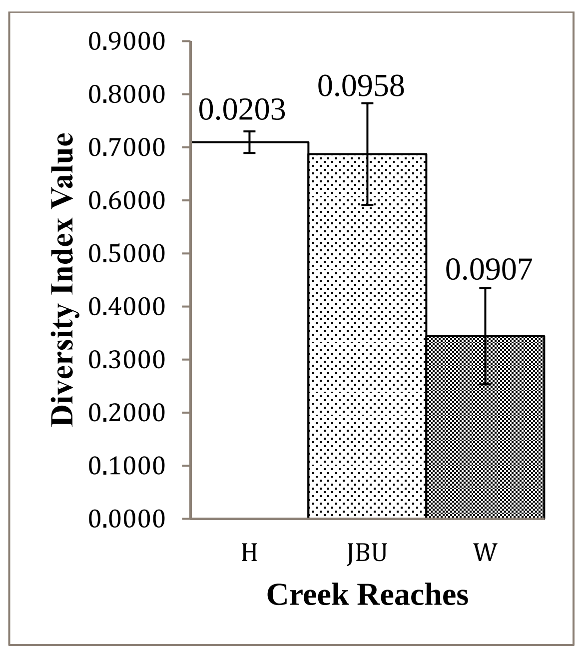

Fig. 2. Mean Family-level Simpson Index of Diversity value. Values range from 0 to 1 indicating increasing diversity. Standard error bars and values are also indicated.

"fairly poor" ranking, indicating that "substantial pollution is likely", and was significantly different than both the JBU reach (P = 0.009) and the Honeycutt reach (P = 0.0005).

Similarly, the FSID of both the Honeycutt and JBU reaches (0.7097 and 0.6874) were notably high, indicating a fairly high level of diversity within the population of benthic macroinvertebrates (Fig. 2). When compared to each other, there was no significant difference between these two reaches (P = 1.245). However, the FSID of the Wastewater reach (0.3441) was significantly lower than both the JBU reach (P = 0.038) and the Honeycutt reach (P = 0.009) indicating a much lower level of diversity.

Physiochemical calculations (Table 2) revealed no significant difference between temperature, dissolved oxygen, nitrate and phosphate levels within any of the reaches. There was a significant difference between the level of water-flow between the Honeycutt and Wastewater reaches (P = 0.015) and a significant difference in the pH level of the Honeycutt reach compared to the Wastewater reach (P = 0.017).

Discussion

Utilizing stream macroinvertebrates as indicators of potential stream pollution has been a practice in the U.S. for approximately 100 years (Weston and Turner 1917). During that time period the number of different macroinvertebrate indices used to assess water quality has grown significantly (Perkins 1983, Resh 1994). However, there seems to be substantial debate concerning the accuracy of these indices in predicting water quality, particularly when those indices utilize "rapid" bioassessments (Hannaford and Resh 1995, Taylor 1997). Rapid bioassessments utilize techniques that are designed to fulfill two primary objectives: reduce the costs and efforts of assessments relative to more labor intensive, highly-specialized traditional approaches; and make the results of the assessments meaningful to a more generalized audience (Resh and Jackson 1993). The first of these objectives was of paramount importance in this current study as both financial and man-power resources were both extremely limited. In a final analysis, the use of a rapid bioassessment protocol seems warranted considering that the USEPA sanctions this approach (Barbour et al. 1999).

The presence of such a large number of Dipterans, particularly the Chironomids in the Wastewater reach heavily influence both the FSID and MFBI resulting in this reach's significant difference from both of the upstream reaches. Some Chironomidae genera have a very rapid life cycle and may produce several generations of individuals within a season, particularly during the warm summer months (Pinder 1986). Thus, it might be concluded that the high Chironomid numbers in the Wastewater reach, during the months of

Table 2. Mean and standard error values for water-flow, temperature, pH, dissolved nitrate, phosphate and oxygen tests performed on water samples from Sager Creek. Significant P-values of compared reaches are in bold type.

	Honeycutt	JBU	Wastewater	Comparison	P-value
Water-flow (m^3/s)	0.296 ± 0.058	0.556 ± 0.086	1.147 ± 0.206	H vs. JBU	0.118
				H vs. W	**0.015**
				JBU vs. W	0.103
Temp. °C	16.74 ± 1.783	18.24 ± 3.105	19.50 ± 3.036	H vs. JBU	0.581
				H vs. W	0.246
				JBU vs. W	0.333
pH	7.022 ± 0.109	7.755 ± 0.296	7.650 ± 0.138	H vs. JBU	0.090
				H vs. W	**0.017**
				JBU vs. W	1.134
Nitrate (ppm)	3.793 ± 0.647	2.500 ± 0.431	2.367 ± 0.649	H vs. JBU	0.181
				H vs. W	0.352
				JBU vs. W	1.304
Phosphate (ppm)	0.143 ± 0.061	0.158 ± 0.068	0.866 ± 0.398	H vs. JBU	0.897
				H vs. W	0.133
				JBU vs. W	0.131
Oxygen (ppm)	8.712 ± 0.393	9.200 ± 0.791	7.978 ± 0.607	H vs. JBU	0.738
				H vs. W	0.425
				JBU vs. W	0.279

June and July, distorted the overall FSID when in fact this is only a seasonal effect. However an analysis of family percentages per reach sample reveals that the percentage of Chironomids in the Wastewater reach remained relatively high throughout the entire year ranging from 60% to 98%, with the second highest percentage (90%) occurring during the month of Feburary. By comparison, the Honeycutt reach's Chironomid percentage ranged from 0% to 26%. The JBU reach had one sample with a high Chironomid percentage (88%) while all others ranged from 8% to 48%. Thus, the high percentage of Chironomids in the Wastewater reach appears to be reach-specific rather than seasonal.

The family-level pollution tolerance value (6) for the Chironomids is fairly high (Sarver 2005). When this is compared to the tolerance values of the dominating families of mayflies from the two upstream reaches, (Baetidae 4 and Heptageniidae 4), the reason for the MFBI differences becomes clear. However, it should be noted that the Chironomid family is highly diverse with many genera, some of which have widely varying levels of pollution tolerance. Thus it is possible that the MFBI for this reach is overestimated. This would be consistent with Hilsenhoff's initial study in which the family-level biotic index tended to overestimate the pollution level of clean streams (Hilsenhoff 1988).

Hilsenhoff (1988) identified eight different insect orders from which individual larvae could be utilized in producing the FBI. The absence of two orders from our Sager Creek data was of particular interest. In all of the samples collected and surveyed, no Plecopterans (stoneflies) or Megalopterans (dobsonflies and alderflies) were identified. This is somewhat surprising as early exploration of a nearby stream (Flint Creek) revealed the presence of both of these orders in some abundance (Wakefield, unpublished data). The absence of the Plecopterans could be explained by their pollution sensitivity. All of the families within this order have tolerance values that range from 0-3 (Sarver 2005). Thus, it is possible that even the "fair" to "good" rankings of the two upstream sites indicate water quality that is not suitable for this sensitive order of insects.

The absence of Megalopterans is more difficult to explain. Megaloptera consists of families that act as predators within the stream benthos. Predators, by their trophic position in a stream food web, should be relatively low in number. This was seen in the low number of Odonatans (damselflies and dragonflies) found in the subsamples. The dominant Odonatan (Coenagrionidae or narrow-winged damselflies) in samples has one of the highest pollution tolerance values (9) of any insect larvae collected (Sarver 2005). The dobsonflies (Corydalidae) have a tolerance value of 4, while the alderflies (Sialidae) have a tolerance value of 7.5 (Sarver 2005). Although both of these tolerance values are below the tolerance of the narrow-winged damselflies, they are well within the range of other organisms that were identified in the subsamples. Thus pollution intolerance does not seem to be the reason for the absence of this order. It's possible that patchy distribution and preferred habitat of the organisms resulted in noncapture.

Differences in MFBI and FSID are also not easily correlated to physiochemical analysis. The level of water-flow at the Wastewater reach was significantly higher than the level at the Honeycutt reach. The effluent of the wastewater treatment plant adds approximately 11.4 x 10^7 liters/day to the stream, (water-flow measures of 81.0 -116.0 x 10^7 liters/day include this effluent.) Even without this, the level of water-flow should be expected to increase downstream as many small springs and wet-weather tributaries feed into the creek as it grows from a first to second order stream. Vannote et al. (1980) suggested that taxonomic diversity should actually increase as stream size increases reaching a maximum level of diversity in mid-order streams. However, this suggested diversity increase is not due to increased water flow but increased instability in the physical parameters of the growing stream system including diel temperature changes, riparian shading, and shifts in food resources (Vannote 1980). The fact that this study found a decreased diversity in the higher-ordered portion of the stream suggests a negative impact from some physiochemical parameter, but does not suggest that increased water-flow causes a decrease in taxonomic diversity.

Also, although the pH levels of the Honeycutt and Wastewater reaches were significantly different from each other (7.022 vs 7.650), both are still clearly within the suitable pH range (6.5-9.0) as previously established by the U.S. Environmental Protection Agency (USEPA 1986). Thus, it is doubtful factors impacting pH alone is directly responsible for the differences seen in the MFBI or FSID. Since no significant differences were found in any of the other tested parameters, the reasons for the differences in arthropod populations is still not discernible.

A comparison of the results of this study to previous studies in Arkansas (Shackleford 1988, Brown et al. 1997, Burns 2001, Williams et al. 2002, Grippo and McCord 2006, McCord et al. 2007) is difficult due to a variety of factors. These factors include, but are not limited to, the fact that these surveys were done on streams in different ecoregions of the state and these studies used a variety of different indices to determine stream quality. However, Brueggen-Boman and Bouldin's (2012) study of the Strawberry River watershed was in the same ecoregion as Sager Creek. Additionally this study utilized the Hilsenhoff Biotic Index (HBI) and some family-level tolerance values for calculating a biotic index score as one indicator of stream health (Hilsenhoff 1987, Brueggen-Boman and Bouldin 2012). Although this study was primarily focused on agricultural/grazing impacts on stream health, some results were similar to what was found in Sager Creek. In particular, stream reaches that had a high percentage of Dipterans (over 80%) also scored in the high range of the "fair" ranking on the HBI and indicated a low level of diversity within the ephemeroptera, plecoptera, trichoptera orders. Water chemistry data was not collected in this study, instead habitat assessments of the riparian zone and changes in the watershed land usage over time were evaluated in conjunction with macroinvertebrate indices. The authors concluded that water quality was being affected by changes in land usage, most notably loss of forested land and increasing urbanization (Brueggen-Boman and Bouldin 2012).

Although the precision of family-level bioassessments remains in question, it seems clear that these tools do provide valuable information for assessing stream health. The results of these assessments support the conclusion that the effluent from the Wastewater Treatment Plant is having a negative effect on the health of Sager Creek. However, the absence of plecopterans and megalopterans from the upstream sites and the conclusion drawn from the Strawberry River study may indicate a larger problem that encompasses the entire watershed. A more comprehensive study of the Sager Creek watershed including land usage data may be valuable in elucidating the causes of the declining stream health.

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Literature Cited

Barbour MT, **J Gerritsen**, **BD Snyder** and **JB 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 (DC): USEPA337 p. <http://water.epa.gov/scitech/monitoring/rsl/bioassessment/download.cfm> Accessed 18 May 2012.

Brown AV, **Y Aguila**, **KB Brown** and **WP Fowler**. 1997. Responses of benthic macroinvertebrates in small intermittent streams to silvicultural practices. Hydrobiologia 347:119-125.

Brueggen-Boman TR and **JL Bouldin**. 2012. Characterization of temporal and spatial variation in subwatersheds of the Strawberry River, AR, prior to implementation of agricultural best management practices. Journal of the Arkansas Academy of Sciences 66:41-49.

Burns JJM. 2001. Changes in watershed land use, geomorphology, and macroinvertebrate assemblages in Clear Creek, Northwest Arkansas, from 1948-1999. [MS thesis]. Fayetteville (AR): University of Arkansas 194 p. UMI Microform EP 15387. p 97.

Ellis D. 1985. Taxonomic sufficiency in pollution assessment. Marine Pollution Bulletin 16:459.

GBM[c] & Associates. 2005. Sager Creek Watershed Assessment: Completed for the City of Siloam Springs. p 73.

Grippo RS and **SM McCord**. 2006. Bioassessment of silviculture best management practices in Arkansas. Arkansas State University Program in Environmental Sciences and Department of Biological Sciences. Jonesboro (AR) p 206.

Hannaford MJ and **VH Resh**. 1995. Variability in macroinvertebrate rapid-bioassessment surveys and habitat assessments in a northern California stream. Journal of the North American Benthological Society 14:430-439.

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

Hilsenhoff WL. 1988. Rapid field assessment of organic pollution with a family-level biotic index. Journal of the North American Benthological Society 7:65-68.

Jones FC. 2008. Taxonomic sufficiency: The influence of taxonomic resolution on freshwater bioassessments using benthic macroinvertebrates. Environmental Review 16:45-69.

Keup LE, **WM Ingram** and **KM MacKenthum.** 1966. The role of bottom-dwelling macrofauna in water pollution investigations. US Dept. of Health, Education and Welfare. Cincinnati, (OH) p 23.

Marchal J. 2005. An evaluation of the accuracy of order-level biotic indices for southern Appalachian streams. Bios 76:61-67.

Marshall JC, **AL Steward** and **BD Harch**. 2006. Taxonomic resolution and quantification of freshwater macroinvertebrate samples from an Australian dryland river: the benefits and costs of using species abundance data. Hydrobiologia 572:171-194.

McCord SB, **RA Grippo** and **DM Eagle**. 2007. Effects of silviculture using best management practices on stream macroinvertebrate communities in three ecoregions of Arkansas, USA. Water Air & Soil Pollution 184:299–311.

Nedeau EJ, **RW Merritt** and **MG Kaufman**. 2003. The effect of an industrial effluent on an urban stream benthic community: water quality vs. habitat quality. Environmental Pollution 123:1–13.

Needham JG and **PR Needham**. 1962. A guide to the study of freshwater biology. 5th ed. San Francisco (CA): Holden-Day, Inc. 108 p.

Omernik JM. 1987. Ecoregions of the conterminous United States. Annals of the Association of American Geographers 77:118-125.

Perkins JL. 1983. Bioassay evaluation of diversity and community comparison indexes. Journal (Water Pollution Control Federation) 55:522-530

Pinder LCV. 1986. Biology of freshwater Chironomidae. Annual Review of Entomology 31:1-23.

Resh VH. 1994. Variability, accuracy, and taxonomic costs of rapid assessment approaches in benthic macroinvertebrate biomonitoring. Bolletino di zoologia 61:375-383.

Resh VH and **JK Jackson**. 1993. Rapid assessment approaches to biomonitoring using benthic macroinvertebrates. *In;* Rosenberg DM and VH Resh, editors. Freshwater biomonitoring and benthic macroinvertebrates. New York: Chapman and Hall p 192-228.

Sarver R. 2005. Taxonomic Levels for Macroinvertebrate Identifications. Missouri Department of Natural Resources Air and Land Protection Division Environmental Services Program Standard Operating Procedures. p 30.

Shackleford B. 1988. Rapid bioassessments of lotic macroinvertebrate communities: biocriteria development Arkansas Department of Environmental Quality. Water Division. p 52.

Simpson EH. 1949. Measurement of diversity. Nature 163:688.

Taylor BR. 1997. Rapid assessment procedures: radical re-invention or just sloppy science? Human and Ecological Risk Assessment: An International Journal 3:1005-1016.

Triola MM and **MF Triola**. 2006. Biostatistics for the Biological and Health Sciences. Boston (MA):Pearson. 699 p.

U.S. Environmental Protection Agency (USEPA). 1986. Quality criteria for water 1986. Washington (DC):USEPA 447 p. <http://yosemite.epa.gov/water/owrccatalog.nsf/9da204a4b4406ef885256ae0007a79c7/18888fcb7d1b9dc285256b0600724b5f!OpenDocument> Accessed on 16 May 2012.

U.S. Environmental Protection Agency (USEPA) Office of Water. 2004. Wadeable Streams Assessment: Field Operations Manual. Washington (DC): USEPA. 119 p. <www.epa.gov/owow/monitoring/wsa/wsa_fulldocument.pdf> Accessed on 5 June 2009.

Vannote RL, **GW Minshall**, **KW Cummins**, **JR Sedell** and **CE Cushing**. 1980. The river continuum concept. The Canadian Journal of Fisheries and Aquatic Sciences 37:130-137.

Voshell JR. 2002. A guide to common freshwater invertebrates of North America. Blacksburg, (VA): The McDonald & Woodward Publishing Company. 442 p.

Warwick RM. 1988. The level of taxonomic discrimination required to detect pollution effects on marine benthic communities. Marine Pollution Bulletin 19:259-268.

Weston RS and **CE Turner**. 1917. Studies on the digestion of a sewage-filter effluent by a small and otherwise unpolluted stream. *In* Sedgewick WT, editor. Contributions from the Sanitary Research Laboratory and Sewage Experiment Station Cambridge (MA): Massachusetts Institute of Technology 96p. Available at: http://books.google.com/books?id=Q7EhAQAAMAAJ&pg=PA5&lpg=PA5&dq=Weston,+Turner,+Coweeset&source=bl&ots=jQfmHD7W7f&sig=CIAMdn2qadfh6FyQMqP9wax5AU&hl=en&sa=X&ei=VFC2T4n1I4aA2wX4idisCQ&ved=0CEkQ6AEwAw#v=onepage&q=Weston%2C%20Turner%2C%20Coweeset&f=false. Accessed 2012 May 18.

Williams LR, **CM Taylor**, **ML Warren**, **Jr**. and **JA Clingenpeel**. 2002. Large-scale effects of timber harvesting on stream systems in the Ouachita Mountains, Arkansas, USA. Environmental Management 29:76-87.

Wright JF, **MT Furse**, **PD Armitage** and **D Moss.** 1993. New procedures for identifying running-water sites subject to environmental stress and for evaluating sites for conservation based on the macroinvertebrate fauna. Archiv fur Hydrobiologie 127:319–326.