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T. R. Brueggen-Boman
*Arkansas State University*, teresa.brueggen@smail.astate.edu

J. L. Bouldin
*Arkansas State University*

# Characterization of temporal and spatial variation in subwatersheds of the Strawberry River, AR, prior to implementation of agricultural best management practices

T.R. Brueggen-Boman[1] and J.L. Bouldin[1, 2]

[1] *Environmental Sciences Graduate Program, Arkansas State University, State University, AR 72467*
[2]*Department of Biological Sciences, Arkansas State University, State University, AR 72467*

[1]Correspondence: teresa.brueggen@smail.astate.edu

## Abstract

Benthic macroinvertebrate (BMI) assessments indicate alterations in physical and/or chemical factors making them valuable tools when attempting to assess agricultural best management practices (BMPs). The purpose of this study was to obtain pre-BMP land cover and macroinvertebrate community data in order to characterize temporal and spatial variation in three catchments: Little Strawberry (LS), Greasy Creek (GC), and Sandy Creek (SC) of the Strawberry River Watershed, located in north central Arkansas, in order to establish baseline conditions. BMIs were collected for the three subwatersheds in 1995, 1999, 2002, 2003 and 2009. Taxa richness, Hilsenhoff Biotic Index (HBI) and Ephemeroptera, Plecoptera and Trichoptera (EPT) score, % dominant taxa, and % Diptera were used as indicators of water quality and habitat changes. All subwatersheds experienced decreased forest land cover indicating land use transition. All resulting BMI measures indicate overall stable habitat and water conditions in the LS subwatershed. Results for GC subwatershed vary with taxa richness, EPT and HBI indicating a stable habitat while increasing % dominant taxa and Diptera indicate a decreasing trend in overall habitat quality. All BMI measures in the SC subwatershed indicate decreasing trend in habitat quality. The most sensitive measure to temporal changes was % Diptera. Trends were not consistent among subwatersheds indicating the importance of continued data collection to establish baseline data and truly monitor changes in aquatic systems over time.

## Introduction

Benthic macroinvertebrate assessments are important tools in evaluating alterations in water and/or habitat quality. Their fast life cycles, specialized respiratory appendages and feeding structures make them ideal for indicating alterations or stress to a system. Continual use of these assessments can be valuable in detecting changes in a system. One focal point of research incorporating such assessments is stream reaches affected by no exclusion cattle grazing, as this practice has been shown to degrade quality of water ways in the vicinity.

Catchments with streams that have open access to grazing cattle have been shown to have higher sediment loads and increased total suspended solids (Wohl and Carline 1996). Excessive sediment loads can alter biological assemblages, especially macroinvertebrates, due to deterioration of benthic habitat (Wood and Armitage 1997). The presence of cattle in streams can not only cause negative impacts to water quality and aquatic habitat, but also alter macroinvertebrate communities within streams (Strand and Merrit 1999). Berry et al. (2003) describes the direct and indirect biological consequences of siltation or suspended sediment in aquatic systems. Direct effects to aquatic invertebrates include abrasion, clogging of filtration mechanisms that interfere with ingestion and respiration, and habitat burial (Wilber and Clarke 2001). Indirect effects include decreased light attenuation and changes in stream bed morphology resulting in decreased suitable habitat (Berry et al. 2003). Waters (1995) cites the deposition of suspended sediment on benthic invertebrates as one of the most important concerns of sediment pollution. Sediment has recently been listed as one of the top contributors to impairment of rivers and streams (USEPA 2004)

An assessment of multiple measures of benthic macroinvertebrates in a system over time can indicate environmental changes. Before alterations take place in an agricultural region (i.e. land use changes, implementation of BMPs) it is vital to obtain baseline macroinvertebrate data for a system to truly assess the impact of the alterations.

Studies of benthic populations to the effectiveness of BMPs to improve water quality parameters in streams impacted by cattle have provided varying results. It has been determined that improved macroinvertebrate richness and increases in pollution intolerant taxa are possible in streams when cattle are excluded (Rinne 1988, Galeone, 2000). Herbst and Kane (2009) determined significant increases in Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa and proportion and diversity of sensitive taxa in post management monitoring of a stream bank restoration project following suspended cattle grazing; they additionally determined a decrease in tolerant taxa and an increase in shredders. Scrimgeour and Kendall (2003), while not determining increases in overall benthic abundances, noted an increase of shredders in cattle excluded stream reaches compared to cattle-grazed reaches. Carline and Walsh (2007) suggested that the increase in density determined in their study was a response to decreased fine sediments from stabilized banks in pasture reaches. Other studies have determined that population responses may depend on the riparian buffer width and length (Parkyn et al. 2003) or the type of vegetation used in riparian restoration (Sovell et al. 2000).

The Arkansas Department of Environmental Quality (ADEQ) (2008) 303d list of Impaired Waterbodies cited seven reaches of the Strawberry River Watershed as not supporting aquatic life due to excess turbidity from surface erosion. Agricultural activities within the watershed are thought to be the major source of the siltation (ADEQ 2003). The implementation of BMPs took place December 2008 to June 2011 in the upper Strawberry River Watershed. Landowner participation was encouraged through an Environmental Protection Agency (EPA) 319 grant issued to the Fulton County Conservation District. The BMPs implemented in the subwatersheds included fencing to exclude livestock from streams, pasture planting, grassland maintenance, and livestock watering facilities with heavy use protection areas.

The objective of this study was to characterize land cover data and benthic macroinvertebrate data for the three upper subwatersheds of the Strawberry River collected prior to implementation of all BMPs incorporated through the EPA 319 grant. This will provide a broad picture of spatial and temporal changes and allow for a more in-depth comparison of pre- and post-BMP results in the assessment of the effectiveness of the implemented practices.

## Materials and Methods

### *Study location and watershed characteristics*

The Strawberry River Watershed, located in north central Arkansas, encompasses approximately 2,023 $km^2$ (ADEQ 2003) (Fig. 1). Confined animal feeding operations, dairies, cattle production, and row cropping are agricultural activities that occur within this watershed (ADEQ 2003). The Arkansas Pollution Control and Ecology Commission (APCEC) (2010) designated the uses of the Strawberry River as: 1) an Extraordinary Resource Water; 2) an Ecologically Sensitive Waterbody; 3) a Natural and Scenic Waterway; 4) Primary Contact (full body, e.g. swimming); 5) Secondary Contact (partial body, e.g. wading); 6) Domestic, Industrial and Agricultural Water Supply; and 7) Fisheries. This watershed supports a diverse community of aquatic macroinvertebrates, with 313 known species, including several federally endangered freshwater mussel and regionally endemic species (Harp and Robison 2006).

The three uppermost subwatersheds of the Strawberry are: Little Strawberry (LS), Sandy Creek (SC) and Greasy Creek (GC). The largest of the three is LS, followed by SC and GC, respectively (Table 1). Primary land use in the subwatersheds is dominated by forest and pasture (Table 2). There are four dairy operations (<100 cows/dairy) located within the subwatersheds (Figure 2). Spring benthic macroinvertebrate collections have been collected nine times since 1995 in the LS subwatershed, seven times in the SC subwatershed and four times in the GC subwatershed (Table 1). At the time of the spring 2009 benthic collection, 4% of fencing (none excluding cattle directly from streams), 19% of brush management and 17% of pasture/hay planting had been implemented. Additional BMPs put in after this time included pasture establishment, watering facilities, ponds, heavy use impact areas (HUAs) and wells.

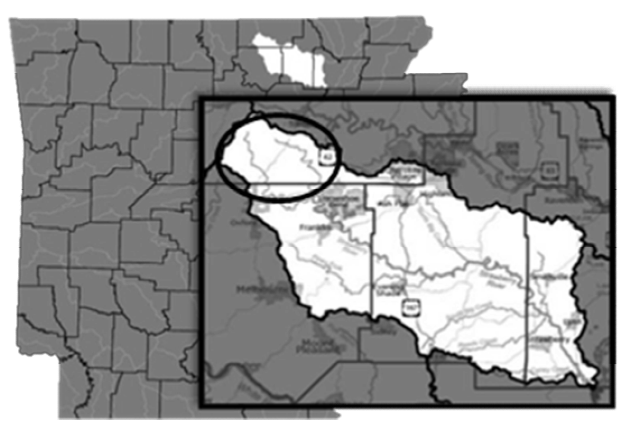

Figure 1: The Strawberry River Watershed, located in north central AR. The circled portion in the inset indicates the portion of the watershed where benthic macroinvertebrate collections occurred, Fulton and Izard Counties (Arkansaswater.org, 2012).

***Methodology***

All 1995 collections, taken by the Arkansas Department of Pollution Control and Ecology (ADPC&E) and 1999, 2002 and 2003 collections, taken by the ADEQ, were located in riffles (ADPC&E 1996, ADEQ 2012). In 2009 collections, taken by Arkansas State University Ecotoxicology Research Facility (ASU-ERF), four riffle (LS09-1, LS09-2, SC09-1, and GC09-1) and two shallow (<1.5m) run locations (SC09-2 and GC09-2) were sampled. Collection locations for 2009 where chosen based upon proximity to grab sample locations for chemical parameters for another aspect of a larger study. ADPC&E collections took place late spring to early summer. All other collections were considered spring collections as defined by the AQEQ (2003). There was some variation in sampling methods among the studies. In 1995, a Portable Invertebrate Box Sampler (PIBS) was used in locations that had sufficient depth and flow stream flow. In locations with insufficient depth a kick seine was used. Two to three sample areas were composited. The 1999, 2002, 2003 and 2009 samples were obtained using a traveling kick method over a 100-m stretch and a D-frame dip net (500 μm mesh). In all collections, samples were preserved in 70% ethanol and returned to a laboratory for analysis. All ADPC&E and ADEQ samples were subsampled until at least 95 organisms were obtained. The 2009 collections were whole samples. All organisms were identified to the lowest taxonomic level possible, usually genus, with the exception of Chironomidae. The 2009 collections were sent to an outside laboratory for quality assurance. Riffle habitat assessments were performed for collections from 2002, 2003 and 2009.

***Data Analysis***

Published land use data was compared to determine any transitions within the subwatersheds throughout the time of the benthic macroinvertebrate collections. This included data from 1999, 2004, and 2006 (Arkansas Watershed Information System 2012). Percent transition was calculated by subtracting 1999 data from 2006 data.

Riffle habitat assessments were compiled to determine comparability of sampling sites in 2002, 2003 and 2009 collections.

Several measures that had been targeted in previous studies were used to determine if change had occurred temporally within each watershed. The measures calculated included: taxa richness, Hilsenhoff Biotic Index (HBI), Ephemeroptera, Plecoptera and Trichoptera (EPT) score, percent dominant taxa and percent Diptera. Taxa richness was calculated by adding the total number of genus found at each location. Each taxon was assigned a tolerance based upon values provided by ADEQ personnel (based from Barbour et al. 1999, and Merritt and Cummins 2008). Family level values were used in the situation that no genus tolerance value was available, e.g. *Maccaffertium* (Ephemeroptera: Heptageniidae) and *Pseudocentroptiloides* (Ephemeroptera: Baetidae) or organism could not be keyed beyond family, e.g. Chloroperlidae. Tolerance values were multiplied by number of organism and divided by total number of organisms (Hilsenhoff, 1987). Water quality at time of collection was then inferred, ranging from very poor to excellent, according to Hilsenhoff (1987). The EPT index was calculated by adding all taxa of those orders present. Percentage of dominant taxa was calculated

Table 1: Subwatershed size and BMI collection sites (Arkansas Watershed Information System, 2006).

| | | Collection Years and Site Locations[1] (Figure 2) | | | | |
|---|---|---|---|---|---|---|
| Subwatershed | Size ($km^2$) | 1995 | 1999 | 2002 | 2003 | 2009 |
| LS | 107.70 | LS95-1* | | LS02-1 | LS03-1 | LS09-1 |
| | | LS95-2 | | LS02-2 | LS03-2 | LS09-2* |
| | | | | LS02-I-1 | LS03-I-1 | |
| | | | | LS02-I-2 | LS03-I-2 | |
| SC | 88.27 | SC95 | | SC02-1 | SC03-1 | SC09-1 |
| | | | | SC01-2 | SC03-2 | SC09-2 |
| GC | 72.73 | GC95 | GC99* | | | GC09-1* |
| | | | | | | GC09-1 |

[1]Location names are identified by abbreviated subwatershed followed by year of collection. If collections took place in Izard County, AR, year is followed by "I". All other collections sites are located in Fulton County, AR. If more than one location took place in the year, the year is followed by a number.
*Same location

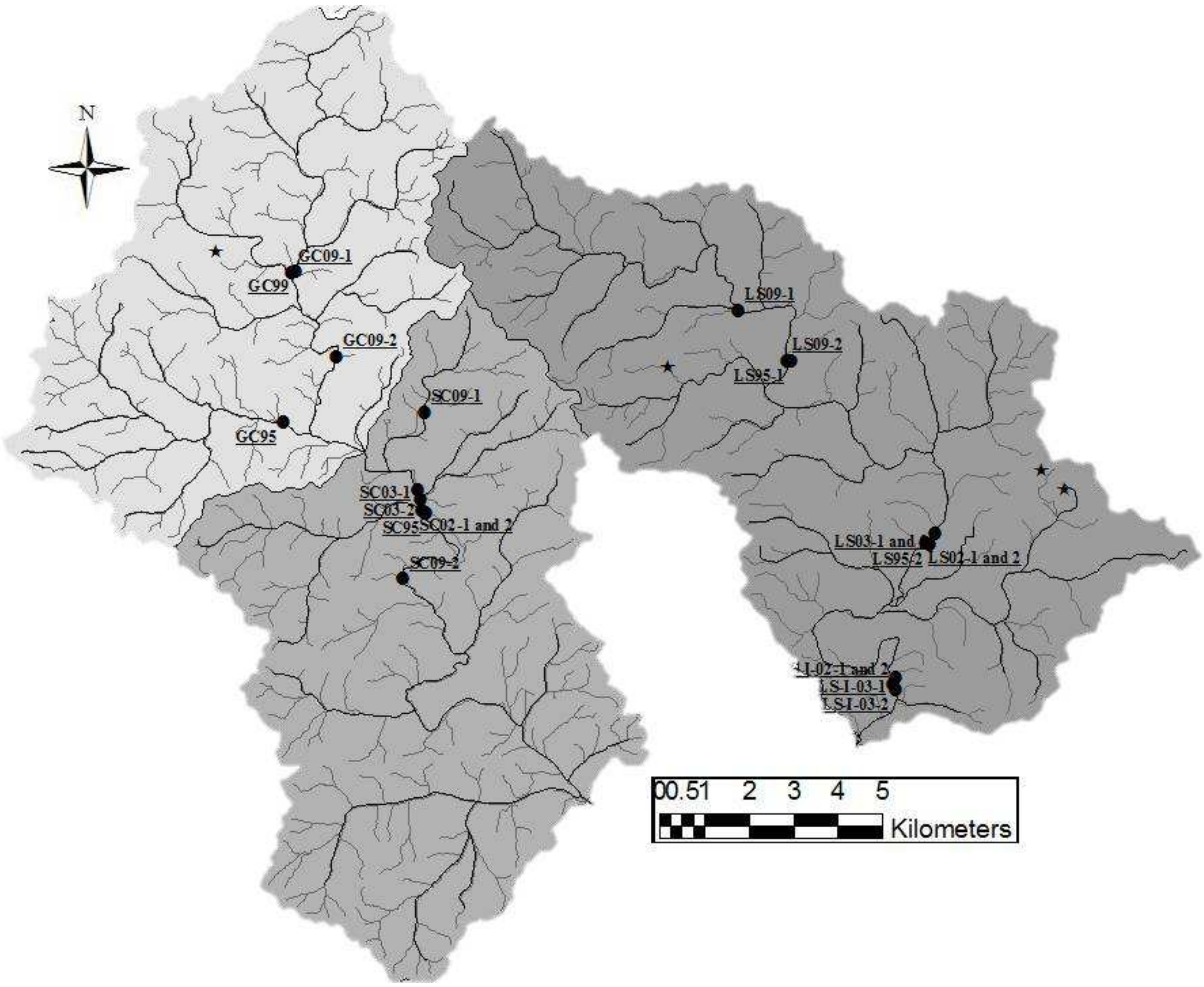


Figure 2: Macroinvertebrate sampling locations from 1995, 1999, 2002, 2003, and 2009 the subwatersheds of the Strawberry River. LS-Little Strawberry, SC=Sandy Creek, GC=Greasy Creek. Stars indicate locations of dairy farms (<100 cows). Darker streams indicate larger tributaries.

Table 2: Percent land cover in three subwatersheds: LS, SC, and GC for 1999, 2004, 2006 with percent change from 1999 to 2006 (Arkansas Watershed Information System, 2006).

| | LS | | | | SC | | | | GC | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Land Cover* | 99 (%) | 04 (%) | 06 (%) | % change | 99 (%) | 04 (%) | 06 (%) | % change | 99 (%) | 04 (%) | 06 (%) | % change |
| Forest | 53.2 | 46.2 | 45.1 | **-8.1** | 54.1 | 46.0 | 45.9 | **-8.2** | 58.0 | 51.2 | 50.4 | **-7.6** |
| Pasture | 44.5 | 47.9 | 45.7 | **+1.3** | 44.4 | 46.8 | 43.3 | **-1.1** | 41.4 | 47.3 | 45.1 | **+3.8** |
| Urban | 1.8 | 4.2 | 4.7 | **+2.9** | 1.2 | 6.0 | 6.5 | **+5.3** | 0.3 | 0.5 | 0.8 | **+0.5** |
| Herbaceous | 0.2 | 1.3 | 3.0 | **+2.8** | 0.1 | 1.0 | 3.0 | **+2.9** | 0.1 | 0.8 | 2.7 | **+2.6** |
| Crops | 0.0 | 0.1 | 0.8 | **+0.8** | 0.0 | 0.1 | 0.8 | **+0.8** | 0.0 | 0.0 | 0.4 | **+0.4** |
| Water | 0.4 | 0.4 | 0.6 | **+0.2** | 0.2 | 0.2 | 0.4 | **+0.2** | 0.2 | 0.3 | 0.5 | **+0.3** |
| Bare | 0.0 | 0.0 | 0.1 | **+0.1** | 0.0 | 0.0 | 0.1 | **+0.1** | 0.0 | 0.0 | 0.1 | **+0.1** |

*Definitions based on a modified Anderson Level II-III classification schema (CAST, 2006).

by dividing the total number of taxa with the highest abundance by the total number of organisms in that collection. Percent Diptera was calculated by dividing the total number of dipteran organisms present by the total number of organisms.

An increasing trend in taxa richness would indicate improved water quality, habitat diversity and suitability. An increasing trend in EPT index or decreasing trend in HBI score would indicate improved water quality. An increasing trend in % dominant taxa and % Diptera would be indicative of increasing environmental stress. No trend over time would indicate stable conditions or that the chosen metrics were not sensitive to occurring changes. Assessing individual data indicates the condition of the habitat at time of collection.

The original HBI incorporated a 0-5 scale for water quality assessment (Hilsenhoff 1977). This range was used in the determination of the 1995 HBI scores. Therefore, all previously published values for all measures were recalculated to confirm consistency of values reported.

Analysis of Variance (ANOVA) ($\alpha$=0.05) was run using Minitab 13.30 to determine if measures were significantly different between sampling years for the subwatersheds as a whole. If data was not normal, a Kruskal-Wallis ANOVA ($\alpha$=0.05) was performed. Sample from 1999 was excluded from analysis.

## Results

All three subwatersheds had a decrease of approximately 8% forest land cover from 1999 to 2006 (Table 2). Sandy Creek had a slight decrease in pasture land cover over this time period, while LS and GC experienced growth. All had an increase in urbanization with the greatest percentage in SC. All had slight increases in land cover of crops, herbaceous, water, and bare land.

Habitat scores revealed similar overall scores

Table 3: Comparison of riffle habitat assessment scores* from 2002, 2003 and 2009 collections (ADEQ, 2012).

| | | Bank Stability | | | | Channel | | Riparian | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Sites | ALT | Left | Right | SD | EMB | Flow Status | RIF | Left | Right | ES/AC | V/D | **Total Score**** |
| LS02-1 | 16 | 10 | 13 | 10 | 11 | 13 | 11 | 16 | 13 | 11 | 10 | **134** |
| LS02-2 | 16 | 15 | 11 | 13 | 13 | 11 | 13 | 18 | 8 | 13 | 13 | **144** |
| LS-I-02-1 | 16 | 13 | 16 | 13 | 13 | 15 | 13 | 10 | 15 | 13 | 13 | **150** |
| LS-I-02-2 | 16 | 13 | 15 | 11 | 13 | 15 | 13 | 8 | 10 | 11 | 13 | **138** |
| LS03-1 | 16 | 11 | 11 | 13 | 13 | 11 | 11 | 16 | 10 | 13 | 10 | **135** |
| LS03-2 | 15 | 16 | 10 | 13 | 13 | 11 | 11 | 16 | 6 | 13 | 13 | **137** |
| LS-I-03-1 | 16 | 11 | 18 | 11 | 11 | 10 | 13 | 8 | 15 | 13 | 10 | **136** |
| LS-I-03-2 | 16 | 11 | 16 | 11 | 8 | 10 | 13 | 8 | 13 | 11 | 10 | **127** |
| LS09-1 | 19 | 16 | 11 | 13 | 13 | 15 | 13 | 10 | 5 | 18 | 13 | **146** |
| LS09-2 | 16 | 5 | 15 | 11 | 9 | 11 | 11 | 5 | 15 | 11 | 13 | **122** |
| | | | | | | | | | | | | |
| SC02-1 | 16 | 6 | 6 | 8 | 10 | 13 | 11 | 3 | 5 | 10 | 13 | **101** |
| SC02-2 | 16 | 13 | 11 | 6 | 6 | 13 | 10 | 16 | 3 | 8 | 13 | **115** |
| SC03-1 | 15 | 11 | 10 | 13 | 8 | 13 | 11 | 5 | 5 | 11 | 13 | **115** |
| SC03-2 | 18 | 15 | 8 | 10 | 8 | 13 | 11 | 18 | 5 | 11 | 10 | **127** |
| SC09-1 | 15 | 19 | 15 | 18 | 10 | 19 | 9 | 15 | 7 | 19 | 8 | **154** |
| SC09-2 | 19 | 17 | 15 | 13 | 16 | 18 | 9 | 20 | 3 | 11 | 7 | **148** |
| | | | | | | | | | | | | |
| GC09-1 | 19 | 17 | 4 | 13 | 8 | 11 | 15 | 19 | 6 | 15 | 17 | **144** |
| GC09-2 | 15 | 19 | 19 | 16 | 17 | 20 | 10 | 19 | 19 | 17 | 2 | **173** |

ALT: channel alteration; SD: sediment deposition; EMB: embeddedness; RIF: frequency of riffles; ES/AC: epifaunal substrate/available cover; V/D: velocity/depth;
*Poor: 0-5; marginal: 6-10; sub-optimal: 11-15; optimal: 16-20; **Total score out of 220 possible points.

Table 4: Comparison of taxa richness, EPT index, % dominant taxa, HBI score and interpretation of water quality based on HBI score for all collections (ADPC&E 1996, ADEQ 2012).

| Sites | Taxa richness | EPT | % Dominant taxa*[KW] | % Diptera*[KW] | HBI* | Interpretation of water quality based on HBI score |
|---|---|---|---|---|---|---|
| **LS95-1** | 11 | 8 | 47.2% | 14.8% | 5.14 | Good† |
| **LS95-2** | 13 | 9 | 49.1% | 8.0% | 5.88 | Fair |
| **LS02-1** | 17 | 8 | 34.3% | 5.8% | 3.45 | Excellent |
| **LS02-2** | 14 | 8 | 27.2% | 24.4% | 4.31 | Very Good |
| **LS-I-02-1** | 15 | 7 | 22.4% | 26.2% | 4.44 | Very Good |
| **LS-I-02-2** | 16 | 9 | 35.9% | 43.6% | 4.72 | Good |
| **LS03-1** | 17 | 11 | 35.6% | 3.4% | 2.57 | Excellent |
| **LS03-2** | 21 | 12 | 30.5% | 7.3% | 3.30 | Excellent |
| **LS-I-03-1** | 32 | 16 | 29.8% | 5.6% | 4.61 | Good |
| **LS-I-03-2** | 27 | 14 | 32.0% | 6.8% | 3.80 | Very Good |
| **LS09-1** | 18 | 7 | 31.5% | 27.1% | 5.55 | Fair |
| **LS09-2** | 28 | 18 | 41.5% | 41.5% | 5.01 | Good |
| Trend | **I** | **I** | **S** | **I** | **S** | |
| | | | | | | |
| **SC95** | 10 | 5 | 46.7% | 52.4% | 4.60 | Good† |
| **SC02-1** | 14 | 6 | 31.6% | 25.1% | 4.37 | Very Good |
| **SC02-2** | 13 | 8 | 51.6% | 5.6% | 4.20 | Very Good |
| **SC03-1** | 20 | 10 | 41.5% | 7.6% | 4.96 | Good |
| **SC03-2** | 15 | 7 | 33.0% | 32.2% | 5.51 | Fair |
| **SC09-1** | 11 | 4 | 95.9% | 95.9% | 5.96 | Fair |
| **SC09-2** | 8 | 3 | 47.6% | 52.4% | 4.58 | Good |
| Trend | **D** | **D** | **I** | **I** | **S** | |
| | | | | | | |
| **GC95** | 13 | 7 | 57.4% | 18.3% | 5.49 | Good |
| **GC99** | 19 | 7 | 29.3% | 25.2% | 5.89 | Fair |
| **GC09-1** | 25 | 12 | 60.2% | 62.6% | 5.06 | Good |
| **GC09-2** | 15 | 4 | 84.4% | 84.6% | 5.66 | Fair |
| Trend | **S** | **S** | **I** | **I** | **S** | |

I=increasing, D=decreasing, S=stable; †Originally published "fair", "good" based on recalculation of HBI. *Metric significantly different year to year, one-way ANOVA ($P \leq 0.02$). [KW]Kruskal-Wallis one-way ANOVA performed.

among sampling locations within the same subwatershed (Table 3). Scores for LS ranged from 122-150 with that largest individual variation occurring in riparian habitat scores. Results for SC were broader ranging from 101-154 with the largest variation in individual scores occurring in bank stability. Scores for GC ranged from 144-173, with the largest variations in bank stability and riparian zone scores.

Comparisons of BMI measures in LS subwatershed indicate stable or improving habitat and/or water quality between 1995 and 2009 collections, with the exception of increasing % Diptera (Table 4). There is an increasing trend in taxa richness and EPT score which are both indicative of improving conditions. Percent dominant taxa and HBI score have remained fairly stable across the collections. The HBI scores in 1995 are comparable to 2009, both of which are slightly higher than 2002 and 2003 collections. Interpretation of HBI scores indicate water quality ranged from fair to excellent. Percent Diptera is extremely variable with a range of 3.4 to 43.6, with low values detected in all 1995 and 2003 collections. The lowest taxa richness was detected in LS95-1 (11). The lowest EPT score (7) was detected in LS-I-02-1 and LS09-1. The highest % dominant taxa (49.1) and HBI score were detected in LS95-2. The highest %

Diptera (43.6) was detected in LS-I-02-2.

All measures in the SC subwatershed indicate decreasing aquatic integrity over the time of the study; with the exception of a fairly stable HBI score (Table 4). The stable HBI scores indicate water quality conditions ranging from fair to very good. Negative trends in taxa number and EPT score, combined with increasing % dominant taxa and % Diptera are all signs of temporal disturbance occurring in this system. The lowest taxa richness (8) and EPT score (3) were detected in SC09-2. The highest % dominant taxa (95.9), % Diptera (95.9), and HBI score (5.96) were detected in SC09-1.

The GC subwatershed measures indicate varying trends (Table 4). The taxa richness, EPT score and HBI score indicate fairly stable conditions. Interpretation of HBI scores indicate water quality ranging from fair to good. The lowest taxa richness (13) was detected in GC95. GC09-2 had the lowest EPT score (4). The highest HBI score (5.89) was detected in GC99. Increasing trends in % dominant taxa and % Diptera are indicators of decreasing quality of aquatic conditions. The highest results for all these measures were detected in GC09-2 (84.4 and 84.6, respectively).

Comparison among years indicated significant ($P \leq 0.02$) differences in % dominant taxa, % Diptera and HBI scores. Median % dominant taxa and mean HBI scores were higher in 1995 and 2009 compared to 2002 and 2003. Median % Diptera was 33% higher in 2009, than the next closest year 2002 (24.5%).

## Discussion

Individual benthic macroinvertebrate collections may only indicate quality of biological communities at the specific time frame prior to the collection or be skewed due to a one-time disturbance. Thus, it becomes extremely valuable to have collections over time to assess temporal changes. These evaluations, combined long-term, can provide a true characterization of the health of a waterway. Having continued land cover data to assess spatial changes over time is additionally pertinent as basic BMI metrics are more indicative of an effect and not a cause. With this information combined with BMI metrics, one can make an inference to an indirect cause (i.e. deforestation leads to increased sedimentation causing increased % Diptera). In this study, the decrease in forest cover indicates changing land use patterns within these subwatersheds. Two of the three subwatersheds had increases of 2 to 4 percent in pasture land, while all had increases of approximately half a percent in crop land, three percent in herbaceous cover and up to five percent urbanization, pointing to the potential for anthropogenic stressors for these waterways. The transition from forest cover to other land cover types support the need for BMPs in these subwatersheds to potentially improve current waterway conditions, or at the least maintain stable conditions as development continues.

Consistency of sampling methods is important when attempting to compare measures over time. While the collection method for the majority of collections was consistent, there is some concern as to the validity of comparing subsample results to whole sample results. According to Baker and Huggins (2005) whole samples provide a more accurate assessment of taxa richness. The best way to avoid assessment errors of a stream reach is to whole sample (Doberstein et al. 2000). Dobertein et al. (2000) supported previous findings by Resh (1979), Courtemanch (1996) and Vinson and Hawkins (1996) that there is risk for loss of substantial information when subsampling occurs. This argument indicates that measures from 1995, 1999, 2002 and 2003 may not be as representative of actual conditions as 2009. When assessing the presented data, even a slight increase or decrease in any individual collection does not lead to substantial variation of the overall trends.

Available habitat scores indicate fairly comparable collection sites. When comparing riffle to run habitat, it is clear that GC09-2, a fairly shallow run location (<1.5m), had greatest indicators for poor quality conditions in three of the five measures. The taxa richness and HBI for this site were comparable to the other three locations. The second shallow run location (<1.5m), SC09-2, additionally yielded the lowest EPT score, but this value along with the other measures did not seem to indicate a large difference between run and riffle locations. Lenat (1988) notes that many water quality assessments focus only on riffles, while large portions of invertebrate species can be found in other habitats. The run locations were selected for accessibility and proximity to water collection sites for a larger study assessing the effectiveness of BMP implementation. They are valuable in the characterization of these subwatersheds. If these sites were eliminated from comparison, overall concluding trends for the subwatersheds would be maintained.

Comparison of measures among years indicates that some measures may be more sensitive in detecting differences temporally than others.

## Conclusion

The 1995 assessment concluded that water quality was being affected by activity within the watershed and that results were not what should be expected of an Extraordinary Resource Stream (ADPC&E 1996). The report suggested BMP implementation to minimize impacts of current land use noted at that time, cattle production and clearing of timber.

It is clear that land use transition is occurring in all three subwatersheds and each system is responding independently. The SC subwatershed, which has the highest percentage of forest loss and increase in urbanization, indicates the highest impact to quality of aquatic habitat. The other two subwatersheds appear to have maintained fairly stable conditions, but maintaining a stable condition from the 1995 results is not ideal for these watersheds. This study provides invaluable knowledge of aquatic conditions within the three subwatersheds prior to BMP implementation. The ability to monitor system's integrity temporally and spatially allows the effects of potentially indirect and long-term sources of pollution to be truly assessed. This makes it an invaluable tool in assessing cattle/agricultural BMP effectiveness.

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