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In-Stream Monitoring of Sediments and Water in the Lower Ouachita River for Site Impact to Aquatic Biota

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Abstract

Reported reduced sportfish densities in the main channel of the Ouachita River prompted an investigation, beginning in 1990, into potential causes of ongoing impairment to aquatic biota. In-stream monitoring that incorporated toxicity testing of sediments and water was conducted to discern potential sources of contaminants that might be related to the suboptimal fishery populations. Organisms selected to evaluate chronic impairment included larval fish, clams, midges and water fleas. The fathead minnow (*Pimephales promelas*) and cladoceran (*Ceriodaphnia dubia*) were used to estimate patterns of toxicity associated with water from seven designated reaches and selected tributaries of the Ouachita River. Larval survival and growth tests were conducted using the fathead minnow, while survival and reproduction were assessed for the cladoceran. An enzyme assay using the Asian clam (*Corbicula fluminea*), and growth and survival tests with *Chironomus tentans* were used to evaluate ambient sediment toxicity within these same reaches and tributaries. Ambient toxicity was rarely observed in the mainstem of the River and, moreover, represented intermittent events. However, impaired growth in larval fish, poor reproduction in cladocera, and reduced enzyme activity in clams were evident for several tributaries. Results of 10-day whole sediment tests showed significant growth reductions in *C. tentans* exposed to sediments collected from West and East Two bayous, Smackover and Coffee creeks. These results suggest there is intermittent impairment in tributaries of the Ouachita River due to ambient water and sediment conditions that are aside from current concerns for mercury contamination.

Introduction

Although chemical monitoring of specific contaminants may validate the use of water quality criteria, concern over bioavailability and effect requires an integrated approach to insure human health and maintain biological communities. Information gathered from a series of stream fishery surveys conducted by Arkansas Game and Fish Commission fisheries biologists from 1987 to 1990 supported the contention that traditional fisheries management efforts would not be sufficient to improve the river's fisheries with such a diverse array of known anthropogenic impacts (Wise et al., 1993). In 1990, concern over reported reduced sportfish densities in the main channel of the Ouachita River prompted the formation of the Lower Ouachita River Work Group (LORWG) to evaluate chemical and biological monitoring of the basin. The lower Ouachita River is defined as that reach which extends from downstream of Remmel Dam, which impounds Lake Catherine, to the Arkansas Louisiana State Line, downstream of the Felsenthal Lock and Dam. This lower section of the Ouachita River has historically been impacted by the release of water from upstream reservoirs, which has had an effect on flow patterns, temperature and water chemistry (Nix et al., 1996). Additionally, several agricultural, municipal and industrial sources are known to contribute to both point and nonpoint discharges. Downstream from Camden, brine and petroleum products have had a significant impact in the past. Before the river exits the state, Felsenthal Lock and Dam furnishes a large green tree reservoir managed for fish and wildlife.

During a 1991 review of available river data from various state and federal agencies, LORWG determined that additional studies were needed to evaluate whether contamination sources or effects were the cause for the reduced fishery resources of the river. These additional assessments included evaluations of water and sediment quality (Nix et al., 1996), fish community structure (Price and Rodgers, 1996), fish bioaccumulation, mussel community analysis (Posey et al., 1996), flow regimes and river flood history (Lee, pers. comm.), and ambient water and sediment toxicity presented in this study.

This study presents the summary of measured in-stream and laboratory impairment of water and sediment-dwelling organisms exposed to ambient conditions in the lower Ouachita River. Beginning in August 1992, ambient toxicity screening of seven tributaries and the main stem of the Ouachita River was initiated with standard test organisms. Toxicity tests were performed on ambient water from the

Table 1. Location limits for each identified segment of the Ouachita River.

Segment Number	Location (km downstream of Remmel Dam)	
1	Remmel Dam to Rockport	(3.7)
2	Rockport to confluence with Caddo River	(16.1)
3	Confluence with Caddo River to confluence with Little Missouri River	(107.8)
4	Confluence with Little Missouri River to Camden, AR	(115.9)
5	Camden, AR to confluence with Smackover Creek	(215.7)
6	Confluence with Smackover Creek to upper end of Felsenthal Pool	(267.2)
7	Felsenthal Pool	(326.7)
8	Felsenthal Dam to Louisiana state boundary	(337.9)

main stem of the Ouachita River on seven occasions between July 1992 and July 1993. Tributary samples were collected and tested from one to six times through 1994. Sediment characterization, sediment toxicity testing using an invertebrate species, and 30-day in situ monitoring using Asian clams were utilized to aid in the evaluation of sediment contaminant data.

Materials and Methods

Description of Sites.--The Ouachita River headwaters arise in the Ouachita Mountains Ecoregion near Mena, Arkansas and drain a portion of the Ouachita Mountains composed mostly of sandstone and shale. The river flows almost due east through three impoundments (Lakes Ouachita, Hamilton, and Catherine) before the gradient changes significantly as it enters the Gulf Coastal Plain near Malvern and changes again downstream of Camden (ADPC&E, 1987). Here, the river is more characteristic of a lowland stream and is affected by land use and drainage of the lower Ouachita River watershed. The river was divided into eight segments for the LORWG investigation (Nix et al., 1996) on the basis of tributary confluences, hydrologic structures or other stream features (Table 1). Specific river sites or stations in toxicity evaluations were selected to delineate areas of suspected impact either from tributaries or river sections known to have reduced fish densities or displaying ambient toxicity revealed in a 1992 screening study. A total of 23 stations contained within eight segments was sampled from July 1992 through December 1994 (Table 2). It should be noted that Reach #7 covers the pool formed by Felsenthal Lock and Dam. This section of the river is affected by processes different from those governing a strictly riverine environment and was therefore not sampled and evaluated for ambient toxicity screening.

Ambient Water Toxicity Testing.--Beginning in 1992, samples were collected from major tributaries of the Ouachita River. Tributaries were screened for toxicity and impairment on a convenience basis prior to 1994, however, exclusive screening of major tributaries was conducted during 1994. Water samples were collected at midstream of each location, and samples from tributaries were usually collected from the bank or a bridge. Efforts were made to collect samples away from pools and primary tributaries. An eight-liter sample provided ample water for use in toxicity tests and routine chemical analysis. Water samples were placed in polyethylene bottles, iced, and returned to Ouachita Baptist University (OBU) laboratories for analysis. If toxicity tests were initiated on the day the samples were collected, then samples were not cooled upon collection. Temperature, dissolved oxygen, specific conductance, and pH were measured in situ using a Hydrolab Surveyor II® system calibrated according to manufacturer's specifications

Table 2. Description of sampling locations for sediment and water in the lower Ouachita River used for toxicity assessments conducted from July 22, 1992 through December 1, 1994.

Station	Site Description	Station	Site Description
1A	0.4 km downstream of Remmel Dam	6A	West Two Bayou
1B	1.2 km downstream of confluence with Cove Creek	E2b	100 m upstream of West Two Bayou
2A	1.5 km upstream of confluence with Chatman Creek	E2B	East Two Bayou
2B	4.0 km downstream of confluence with Chatman Creek	6A	Upstream of Champagnolle Creek, 1.0 km downstream of confluence with Smackover Creek
3A	1.0 km upstream of confluence with Reynold's Ditch	6B	0.5 km downstream of confluence with Champagnolle Creek
4A	Reynold's Ditch	6C	0.7 km downstream of Cation Lake
5A	1.0 km upstream of confluence with Little Missouri River	6D	0.5 km upstream of confluence with Moro Bayou
5B	1.0 km downstream of confluence with Little Missouri River	6E	1.0 km downstream of confluence with Moro Bayou
5C	0.5 km upstream of Highway 7 bridge at Camden	7A	Inflow to Felsenthal Pool
E2A	3.0 km downstream of Highway bridge at Camden	8A	1.0 km upstream of confluence with Coffee Creek
E2b	1.5 km upstream of confluence with Smackover Creek	C	Mouth of Coffee Creek
		8B	0.5 km downstream of confluence with Coffee Creek

each day prior to use. Alkalinity, hardness, and pH were determined in the laboratory according to methods described by the American Public Health Association (1992) or the U.S. Environmental Protection Agency (U.S. EPA, 1970).

Chronic Toxicity Screening.--Chronic toxicity tests were conducted on water samples collected from each of the seven designated reaches and selected tributaries of the Ouachita River. These ambient toxicity tests were conducted utilizing the microcrustacean, *Ceriodaphnia dubia*, and the fathead minnow, *Pimephales promelas*. Methods follow those outlined by U.S. EPA (1989) with the following modifications cited by Knight and Waller (1987):

1) Daily sample water renewals for both *C. dubia* and fathead minnow tests were made from a single sample stored at 4°C between renewals;
2) *C. dubia* were fed a combination of green algae, *Selenastrum capriconutum* and an extract of cereal leaves, Cerophyl;
3) Statistical design for *C. dubia* tests utilized a randomized block design, while the design for fathead minnow tests utilized a completely random design.

Test Endpoints.--The endpoints of *C. dubia* tests were acute survival and chronic productivity. Productivity is defined as the number of young produced per female over the course of seven days. *C. dubia* will produce their third brood within the seven-day test duration. The endpoints for fathead minnow tests consisted of survival and growth, with growth measured as mean dry weight of test organisms after seven-day exposures. Due to the nature of sampling on the

Ouachita River (by reach), ambient toxicity tests were used to identify areas of the river that were consistently impacted. However, identifying the source for measured in-stream impairment was limited since ambient testing differs from effluent evaluations. Ambient tests can provide information on toxicity patterns associated with a water body that enables investigators to pose hypotheses about the sources of toxicity.

Statistical Methods.--All chronic data using *C. dubia* and *P. promelas* were analyzed using TOXSTAT software (Gulley et al., 1991). Data were evaluated for normality and homogeneity of variances prior to subjection to analysis of variance (ANOVA). If the data were found to be normally distributed and the variances homogeneous, Tukey's Multiple Comparison Procedure was utilized to determine statistically significant differences. If the data did not follow a normal distribution, or if the variances were heterogeneous, then the nonparametric Kruskal-Wallis procedure was used. Performance controls (OBU laboratory water, reconstituted moderately hard) were analyzed with the ambient water toxicity data.

In-stream Monitoring of Clams.--Adult Asian clams, Corbicula fluminea. used for monitoring were collected from the Middle Fork of the Saline River near the town of Benton, Saline County, Arkansas in July 1993. Prior knowledge of the condition of this collection site was provided by John Harris (pers. comm.). Clams were sorted according to shell length and selected to range in size from 14 to 16 mm, and allowed to acclimate in the Ouachita River downstream of Remmel Dam. Clams were kept in chilled, aerated river water between stations from the time of collection until arrival at monitoring placements. Upon arrival at each monitoring location, thirty clams were sorted into each of three plastic crates measuring 180 mm x 180 mm x 180 mm. Crates were lined with number 20 fiberglass screen mesh (850 micrometer opening) and filled with cobble to provide substrate for the clams and a settling surface for collected fine silt. Crates were then secured into a group of three and lowered at each sampling location to a depth of 2-3 m and secured to a tethered line extending from a marked shore object (either tagged trees or major snags). At the time of placement, sediment samples were collected from each site location using a petite ponar dredge (150 mm x 150 opening) preserved in plastic bags, chilled and transferred to the laboratory for characterization.

After one month of exposure, clams were retrieved from crates, dissected aboard the boat, transferred to cryovials, and frozen in liquid nitrogen until enzyme analysis could be performed at Arkansas State University's (ASU) Ecotoxicology Research Facility. Sediments were again collected by petite ponar from each location in September 1993 and preserved. Sediment accumulated in each crate was transferred to bags immediately after retrieval, chilled, and transported to the laboratory for characterization and use in *Chironomus* testing. Overlying water was retained in all bags until laboratory use, insuring that sediments remained hydrated until analysis. Clams collected from baskets were used for evaluation of cellulolytic activity and tissue analysis.

Cellulolytic Enzyme Assay.--Six clams from each site were randomly chosen and dissected for cellulolytic enzyme analysis according to methods used by Farris et al., 1989. Since degradation of substrate cellulose takes place through the combined action of endocellulase and exocellulase, the product of two specific assays was used to determine the activity level of the enzyme group of the monitored organism. Endocellulase, with a synthetic cellulose medium [1% carboxymethylcellulose, (CMC) solution] as a substrate, was assayed for viscosity changes based upon the rate and amount of breakdown of 1 mg of reduced sugar by the available cellulases within the organism at the time of dissection. Exocellulase uses a combination of CMC as the substrate and dinitrosalicylic acid (DNS) as a color indicator to bind with any accessible glucose liberated during the degradation of available cellulose (Farris et al., 1989). The product of the two measurements was used to determine accessible enzyme availability extracted from the clams.

Enzyme extracts from individual clams were prepared from whole-body homogenates. Samples were homogenized in phosphate buffer at pH 6.1 at a wet mass to buffer ratio of 0.2 g/ml. Samples were centrifuged for 15 minutes at 15,000 rpm, supernatants decanted for endo/exocellulase analysis, and the pellets recovered for dry mass measurements.

Statistical Analysis.--One-way analysis of variance (ANOVA) was used to evaluate the effects of site water on *C. fluminea* cellulolytic activity (Statistical Analysis Systems, 1985). Significance was inferred $\alpha = 0.05$, and Duncan's multiple range test was used to determine means that were significantly different from control activities.

Sediment Toxicity Tests.--*Chironomus tentans* were used in ten-day, solidphase tests to determine relative toxicity among collection sites (U.S. EPA, 1994). Sediment grab and trap samples were collected from 11 sites in the Ouachita River basin during August and September 1993. Samples were transferred to 500-ml plastic bags and cold packed in coolers. Coolers were then shipped overnight to the Arkansas State University (ASU) Ecotoxicology Research Facility. Upon arrival, samples were checked for chain of custody records, and then transferred to a refrigerator for storage. Prior to all toxicity tests, sediments were homogenized by stirring with a spoon.

Approximately 400 ml of the homogenized sample was then sieved through a #40 (425 µm), #270 (53 µm), and a collecting pan sieve series using one liter of moderately hard synthetic water used as the overlying water. The fraction collected in the #270 sieve was used as the test sediment. The water, including the silt fraction in the collecting pan, was

Table 3. Measured impairment and toxicity in *Ceriodaphnia dubia* and *Pimepales promelas* exposed to ambient water from sites throughout the lower Ouachita River, July 22, 1992 through December 1, 1994. The number of toxicity tests conducted per site are shown with dates for the collections that elicited chronic impairment.

Site	*Cerodaphnia dubia*	*Pimephales promelas*	Site	*Cerodaphnia dubia*	*Pimephales promelas*
1A	9	9 (7-22-94)‡	6E	8	7
1B	9 (12-1-94)‡	9	7A		
2A	7		8A	10(1-14-93)‡	10(1-14-93)‡
2B	7		C	2†‡	3
3A	7	7	8B	10(1-14-93)†‡	10(7-22-94)‡
3B	7	7	E2A		
3C	7		E2b		
4A	7		E2B	2(10-23-92)†‡	
5A	10(1-14-93)†‡	10(9-16-93)‡	Champagnolle Creek	1(8-21-92)†‡	
5B	10	10(9-16-93)‡	Smackover Creek	1	
5C	12	12(8-5-94)‡	Cove 1	3(4-16-94)‡	2(12-1-94)‡
6A	15(1-14-93)†	13(8-5-94)†	Cove 2	3(4-16-94)‡	2(12-1-94)‡
6B	11	10	Coffee Creek 1	2(7-22-94)‡	3
6C	7	7	Coffee Creek 2	2(12-1-94)‡	3
6D	8	7	Coffee Creek 3	2(7-22-94 / 12-1-94)‡	3

†Statistically significant difference in survival
‡Statistically significant difference in productivity (growth or reproduction)

used as overlying water. The fine particles suspended in water were allowed to settle for 24 hours. Approximately 30 ml of sediment from the #270 sieve was placed in a 250ml beaker. The dilution water from the collecting pan was placed in a beaker, resuspended by stirring, and 150 ml was poured gently into the beaker. Test chambers were covered and placed in a temperature-controlled room or incubator at 23±2 °C, and sediment was allowed to settle for 24 hours. All test chambers were continuously aerated and oxygen levels maintained above 5 mg/L. After settling for 24 hours, ten 3rd-instar larvae cultured in house at ASU were added from a newly-hatched culture unit from each replicated test vessel. All larvae were released beneath the water surface using a widebore, fired pipette to avoid entrapment of air bubbles which may cause floatation. Individuals found floating were removed and replaced with larvae from the same culture unit. Each beaker received a 0.5 ml suspension of Tetramin® flake food, and renewed overlying water, daily. Moderately hard synthetic water was used as overlying test water and renewed by siphoning approximately 2/3 of the volume from each chamber.

The test was terminated after ten days of exposure by pouring the contents of each beaker through a #40 (425 µm) sieve. The contents in the sieve were rinsed into a sorting tray, and larvae were collected and preserved in ethanol. The number of individuals recovered in each beaker was recorded. The larvae were placed in labeled, tared aluminum boats and placed in a drying oven for at least two hours at 100°C. The boats were transferred to a desiccator for at least 1 hour and weighed. Mean dry weights of each replicate were taken to compare net weight gain among exposed individuals for each of the 26 sites tested. Significant differences in growth and survival among sites were determined using Dunnett's analysis. Analysis of variance procedure was used to compare *C. tentans* survival and weight gain among all the site sediments (U.S. EPA, 1989).

Sediment Characterization.--Sediment characteristics routinely studied in association with toxicity testing were included for analysis based on their anticipated effects on contaminant sorption to bottom sediments. Particle size composition, cation exchange capacity (CEC), percent total solids and pH were determined as in Moore et al. (1996) on sediments from 11 study sites.

Results and Discussion

Ambient Water Toxicity Screening.--The main stem of the Ouachita River was sampled for toxicity screening seven times between July 1992 and July 1993. Samples were collected on twelve occasions from seven tributaries between August 1992 and December 1994. Toxicity evaluations on ambient water using *C. dubia* and *P. promelas* for all sites are summarized in Table 3. Testing in August 1992 established that toxic responses were measurable in *C. dubia* exposed to water collected from the mouth of Champagnolle Creek.

Table 4. Cellulolytic enzyme activity measured from *Corbicula fluminea* in 30-day in situ exposures, at 10 river locations in the lower Ouachita River.

River Sites	Product	Index	River Sites	Product	Index
Reynolds Ditch (3B)	1171.9 (±205.6)	18 (±3)	Champanolle (6B)	1130.8 (±245.5)	17 (±4)
Little MO (4A)	374.2 (±138.4)	6 (±2)	Felsenthal (7A)	50167.7 (±681.7)	76 (±10)
E2A	1356.1 (±183.5)	20 (±3)	8A	6638.7 (±1198.7)	100 (±18)
E2b	4660.3 (±1494)	70 (±23)	Coffee Creek (C)	1234.1 (±529.0)	49 (±8)
E2B	2365.2 (±475.9)	36 (±7)	8B	3729.7 (±2127.3)	56 (±32)
Smackover (6A)	1417.7 (±165.9)	21 (±3)			

The mean productivity of 4.5 young per female, was statistically different from productivity measured in water exposures from Station 5C, Smackover Creek and Station 6A. Large variance measures were associated with mean production estimates for stations and often resulted in failure of the homogeneity test on additional test dates. Description of test means for reproduction and survival can be found in greater detail in Knight et al. (1995).

Impairment of both reproduction and growth, as well as toxicity, was rarely observed in either *C. dubia* and *P. promelas* when exposed to water from the mainstem of the Ouachita River. However, toxicity to *C. dubia* was measured in at least one section of Champagnolle Creek four of the five times this site was tested. Repeated test impairment and mortality was also measured in *C. dubia* when exposed to water from East Two Bayou (four of five times tested) and from West Two Bayou (three of five times tested). Toxicity in *P. promelas* was observed in the testing of water from East Two Bayou during September 1993. This corresponded with significant impairment of reproduction measured in C. dubia when exposed to the same water. Impairment in reproduction and growth for *C. dubia* and *P. promelas*, respectively, was measured in water collected during July 1994 at Cove Creek. This was the greatest significant reduction in measured parameters obtained during the toxicity assessment.

In-stream Clam Monitoring.--The use of *Corbicula* as a suitable surrogate for native freshwater bivalves has been successfully validated in several monitoring studies (Farris et al., 1988; Farris et al., 1996; Milam and Farris, 1998). Clams have been shown to establish significant biomarker responses to in-stream exposures of various metal discharges and to recover to background physiological conditions following removal of the exposure. Cellulolytic activity measured in *Corbicula* from the Ouachita River varied from 6,638 x 10^5 to 1,130 x 10^5 units/g dry weight for stations sampled. The highest enzyme activity was found in clams collected approximately 6.5 km downstream of Felsenthal Dam (Station 8A) and at the inflow to Felsenthal Reservoir (Station 7A) (Table 4). The remaining clams from all other in-river stations located downstream of tributaries showed reduced enzyme activity. Most notable activity reductions were found in clams held at stations within influence of East and West Two bayous (reductions to 36 and 20% of upstream activities, respectively) and clams downstream of both Smackover (21% of upstream) and Champagnolle (17% of upstream) creeks. Clams did not display significant mortality in any of the replicate baskets for all sites during the month-long exposures. The observed low digestive activity in clams from the Little Missouri River was attributed to low production in the river as gut contents were noted to contain less mass and color than in clams from the Ouachita River. This observed difference in enzyme activity between clams from tributaries and mainstem rivers has been measured and validated in comparisons of the Guest and Clinch rivers in Virginia (Farris et al., 1996). However, the reduction to 18% of upstream activities in clams from Reynold's Ditch was not attributed to absence of sufficient organics or suspended matter for feeding. Indeed, the observed amount of gut contents would suggest that the clams were actively processing food but failed to utilize useable foodstuffs due to measured enzyme reduction. Clam enzyme assays measured reduced glucose equivalents from areas that have been suggested as possibly impacted with regards to native mussel densities (Harris, 1991). The same sites noted by Harris as having poor mussel communities from the East Two Bayou confluence to downstream of Champagnolle Creek are comparable to sites having reduced cellulolytic

Table 5. Results of 10-day sediment toxicity tests using C. tentans exposed to sediments collected from the lower Ouachita River using a ponar grab sampler and in-river sediment traps. Values represent mean survival and growth following acute tests.

	Survival (%)		Growth (mg)			Survival (%)		Growth (mg)	
	Ponar	Trap	Ponar	Trap		Ponar	Trap	Ponar	Trap
Reynolds Ditch (3B)	47	–	0.07	—	Smackover (6A)	63	–	0.02	—
Little MO (7A)	93	83	0.13	0.21	Champagnolle (6B)	100	100	0.14	0.20
E2A	90	93	0.21	0.05*	Felsenthal (7A)	83	93	0.11	0.63
E2b	60	97	0.06	0.04*	Coffee Creek (C)	83	87	0.09	0.27
E2B	46	90	0.06	0.03*	8B	87	100	0.18	0.13

*Values were statistically significant between tested groups.

activity from these studies. Overall, the mussel community of the lower Ouachita River is similar in species composition, distribution, and density when compared to other rivers of the state (Posey et al., 1996). However, Posey (1997) stated that nearly all collected from the Ouachita River were "dwarfed" when compared to those of other rivers. This phenomenon of reduced growth is unexplained to date and doesn't seem limited to site specific locations along the river.

Cellulolytic activity of clams in the Ouachita River appeared altered in a predictable fashion in relation to suspected impacted areas of the river. However, variability in enzyme activity from the upstream clams not experiencing effluent or chemical treatment was large at times. The sensitivity of detecting statistically significant changes across stations was reduced by this variability. The importance of the cellulolytic enzyme complex for providing usable foodstuffs for clams, mussels, and snails appears intuitively correct in that previous studies (Farris et al., 1988; 1989) suggested declines in cellulase activity cascaded into growth reductions and preceded mortality and changes in glycogen content (Haag et al., 1993). Unionids and clams feed by siphoning small particles from the water column or water-sediment interface. Feeding by these bivalves may be impaired by loss of suspended particles, behavioral avoidance of contaminants, or actual inhibition of digestive enzymes by associated bioaccumulation.

Chironomus tentans *Growth on Sediments.* –Results of 10-day tests with *C. tentans* on sampled sediments of the Ouachita River showed statistically significant differences in reduced growth from stations around West and East Two Bayous (E2A, E2B,E2b), Smackover (6A), and Coffee (6B) creeks (Table 5). Tests utilizing ponarcollected sediment from Station 6A downstream of Smackover Creek resulted in the lowest measured *C. tentans* growth in six series of tests from all stations (individual dry weight = 0.02 mg). Growth of *C. tentans* on control sediments must produce 0.06 mg for individual dry weight. Two separate tests upon the same sediment (6A) measured growth per individual below 0.03 mg. Mortality in these tests was not a significant factor in reducing growth estimates per individual by reducing the sample size comparisons among stations. Rather, *C. tentans* only had reduced survival to 50% upon sediments from Reynolds Ditch and downstream of Station E2B. Collection method was not a factor for sediment from Station E2B as tests with both trapped and grabbed sediments resulted in growth impairment. However, measured impairment in recently deposited sediments was evident from other stations comparing ponar and trap samples.

Any correlation between growth impairment and sediment characterization was not apparent from these tests. However, there is a strong indication that the observed impairment from sediment tests is in agreement with areas where the clam enzyme reductions were most apparent. Clam responses during in-stream monitoring efforts, growth tests from collected sediments, and Harris' (1991) mussel surveys form a collective indication that reductions associated with siphoning, sessile animals or with macroinvertebrates that ingest affected sediments are able to delineate impact zones within the Ouachita River. Macroinvertebrate surveys conducted by Price and Rodgers (1996) also reported impacted communities downstream of West Two Bayou near Camden to just upstream of Smackover Creek and downstream of Felsenthal Dam. While habitat variations were reported to account for linear declines in the upper Ouachita comparisons, they could not account for all of the differences noted in sites downstream of Camden and

Felsenthal Dam. Those results also support the observed differences in fish communities downstream of Reach #4 reported by Wise et al. (1993). Causes for measured spatial fluctuations in the fish community have also indicated point source impacts related to areas affected by West Two Bayou and Smackover Creek.

Sediment Characterization.--Sediment characteristics varied among sites, but were similar among replicated samples (Tables 6 and 7). Reynolds Ditch had the highest percent of sand and clay particles with calculated values of 75.3 and 6.5%, respectively. Trap samples from site E2B had the greatest percent silt, 96.3%. Percent total solids varied from 34.7- 78.7% among all sites collected in September 1993. Measured values for pH were similar among all sites, ranging from 5.24 at E2b in August 1993 to 7.48 downstream of Little Missouri River (LM) in September of the same year. Cation exchange capacity was less than 3.0 for sediments collected from all sites, with the exception of trapped sediments collected from Site 7A (4.3).

Sediment characterization indicated that sites were variable in relation to those parameters measured in this study. However, one important parameter that should be analyzed in future work, is total organic carbon (TOC). Laboratory and field data support the contention that TOC concentrations in sediment control the availability of organic chemicals to interact with organisms (Giesy et al., 1990). This analysis, combined with analytical evaluation of available water soluble fractions of contaminants, would allow better delineation of impacted areas having reduced growth of *Chironomus* and clam enzyme activity.

Measured impairment responses were coincident in sampled sediment and water from East Two Bayou, Smackover and Coffee creeks, and to a lesser degree in Champagnolle Creek and downstream of Reynolds Ditch (Table 8). The potential cause of the observed impairments may be explained in part by water and sediment conditions measured during September 1992 through September 1993 (Nix et al., 1996). Measured water quality parameters that exceeded the Arkansas Water Quality Standards (WQS) for pH, turbidity, and certain trace metals were sufficient to impact biota as measured and represented by both acute and chronic aquatic life criteria. While most measured concentrations of pesticides were very low, dieldrin concentrations exceeded the proposed federal criteria at a number of sites. If not for sediment TOC concentrations exceeding 5% in the lower Ouachita River, dieldrin would be expected to exert a greater influence at all sites. Significant increases in measured water quality parameters such as specific conductance, alkalinity, and turbidity were often correlated with high concentrations of sulfate, chloride, and trace metals for the three impacted tributary sites and East Two Bayou. Elevated zinc, copper, and arsenic concentrations in Coffee Creek, as high as 10 times the acute Arkansas WQS were measured in January 1994. High concentrations of cadmium, copper, nickel, and zinc (as high as 35 times the zinc WQS, specifically) were again evident in July 1994. Elevated concentrations of nickel, chromium, vanadium, and zinc were measured in Champagnolle Creek in October

Table 6. Characterization of Ouachita River sediments collected by ponar dredge and silt trap on September 14, 1993.

Site	pH	Temp (°C)	Sand (%)	Silt (%)	Clay (%)	CEC	Total Solids (%)
Reynolds Ditch	6.45	10.0	75.3	18.2	6.5	3.0	41.4
8B-trap	6.07	10.6	16.0	82.4	1.6	1.7	78.0
8B-ponar	6.06	11.9	59.5	10.5	0.0	0.7	72.2
7A-trap	5.54	11.8	7.4	92.6	0.0	4.3	34.7
7A-ponar	6.83	12.8	21.5	78.8	0.0	2.4	50.2
6B-trap	5.95	15.1	12.4	87.6	0.0	2.7	52.8
6B-ponar	6.35	15.9	50.5	49.5	0.0	1.3	69.9
6A-trap	6.29	13.4	25.2	73.3	1.5	1.8	55.5
6A-ponar	6.58	8.1	44.1	55.9	0.0	1.4	76.6
Coffee Creek-trap	6.97	9.4	13.4	86.7	0.0	2.8	52.8
Coffee Creek-ponar	6.64	9.4	60.4	39.6	0.0	1.1	72.0
8A-trap	7.30	12.5	28.9	71.1	0.0	1.0	50.0
8A-ponar	6.54	7.6	60.2	38.2	1.6	0.9	73.6
Little MO-ponar	7.48	9.8	55.5	39.1	5.4	1.2	79.6
E2B-trap	6.70	10.8	7.9	96.3	0.0	2.0	45.2
E2B-ponar	5.88	11.5	54.2	45.8	0.0	1.8	78.7
E2A-trap	6.81	10.5	6.4	93.6	0.0	1.6	44.5
E2A-ponar	6.01	10.5	44.0	61.6	0.0	–	64.9

Table 7. Characterization of Ouachita River sediments collected by ponar dredge on August 5, 1993.

Site	pH	Temp (°C)	Sand (%)	Silt (%)	Clay (%)	CEC	Total Solids (%)
Reynolds Ditch	–	–	–	–	–	–	82.7
E2A	6.49	19.1	68.2	31.8	0.0	0.5	71.6
E2b	5.24	12.5	42.8	57.2	0.0	0.8	67.9
E2B	6.40	13.1	43.0	57.0	0.0	1.0	64.7
6A	6.12	11.8	48.5	51.5	0.0	0.4	72.4
6B	6.12	12.6	58.9	41.1	0.0	0.7	77.0

Table 8. Coincident impairment measured from nine sites in the lower Ouachita River, downstream of Camden, AR on October 5, 1993. Checks represent measured parameters that were significantly reduced ($\alpha = 0.05$).

	C. fluminea Enzyme Activity	*C. tentans* Survival & Growth	*C. dubia* Survival & Reproduction	*P. promelas* Survival & Growth
River Sites				
Reynolds Ditch (3B)	✓	✓		
E2A	✓	✓		
E2b		✓		
E2B		✓	✓	
6A	✓	✓	✓	✓
6B	✓		✓	✓
8A			✓	✓
Coffee Creek (C)		✓	✓	✓
8B			✓	✓

1994. Elevated water quality parameters on separate dates and at the sites delineated by these toxicity and impairment assessments suggest that the study did serve to screen stations for suspected impact on the lower Ouachita River.

Monitoring efforts designed to address the link between contaminants and adverse impacts on aquatic communities in the ambient environment have been recently previewed extensively in such large watersheds as the Chesapeake Bay (Hall and Alden, 1997). As in this study of the Ouachita River, ambient toxicity tests have been utilized on a much broader scale than traditional effluent toxicity tests. These ambient toxicity tests are now being used in a tiered approach to identify areas where future assessment efforts are warranted. The two-year database showed that responses from both water column and sediment tests, measured at specific stations, were generally similar. The ambient toxicity data collected during this study demonstrate the need for integrated, multispecies water and sediment evaluations to strengthen impact assessments.

What began as routine sampling of fish populations in the lower Ouachita River has now not only indicated suboptimal fish population levels, but has drawn attention to the presence of toxic effects on other aquatic communities in contaminated areas. This assessment of biologically significant environmental contamination eventually widened to include an advisory against the consumption of fish in Louisiana sections of the Ouachita River because of mercury contamination. Fish in the portion of the Ouachita River downstream of Smackover Creek to the Louisiana border were found to contain mercury concentrations that exceeded the U.S. Federal Food and Drug Administration's (FDA) recommended advisory limits of 1.0 ppm. A health advisory on the consumption of predatory fish was subsequently issued by the Arkansas Department of Health in 1992.

Now that ambient toxicity screening of water and sediments has been field validated with a suite of sensitive lethal and sublethal bioassays for resident aquatic organisms, risk ranking is needed to substantiate cause and effect relationships that are aside from current concerns for mer-

cury contamination. It is clear from this study that some sites are contaminated, and others demonstrate localized ambient toxicity. Application of risk ranking methods, such as those used by Hartwell (1997), can be used to assess (1) whether localized toxic contaminant effects are influencing populations in the lower Ouachita River as a whole, (2) if low-level but wide spread contamination is a greater problem, or (3) if a combination of the two is at work. Such approaches assume that biologically significant environmental contamination is not necessarily predictable based solely on chemical analysis. This is confirmed with the current bioaccumulative properties of mercury in areas of the river where water and sediment mercury values are typically below detection (Nix et al., 1996). Confirmed high levels of trace metals and one pesticide in at least three areas of the lower Ouachita River suggest that future adverse effects on aquatic communities may be linked to bioavailable contaminants shown to exert effects in bioassays.

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