Transactions of the American Fisheries Society 133:705–717, 2004

Effects of Lowhead Dams on Riffle-Dwelling Fishes and Macroinvertebrates in a Midwestern River

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Abstract.—Many studies have assessed the effects of large dams on fishes and macroinvertebrates, but few have examined the effects of lowhead dams. We sampled fishes, macroinvertebrates, habitat, and physicochemistry monthly from November 2000 to October 2001 at eight gravel bar sites centered around two lowhead dams on the Neosho River, Kansas. Sites included a reference site and a treatment site both upstream and downstream from each dam. Multivariate analysis of variance indicated that habitat, but not physicochemistry, varied immediately upstream and downstream from the dams, with resultant effects on macroinvertebrate and fish assemblages. Compared with reference sites, upstream treatment sites were deeper and had lower velocities and downstream treatment sites were shallower and had higher velocities; both upstream and downstream treatment sites had greater substrate compaction than reference sites. Macroinvertebrate richness did not differ among site types, but abundance was lowest at downstream treatment sites and evenness was lowest at upstream treatment sites. Fish species richness did not differ among site types, but abundance was highest at downstream reference sites and evenness was highest at upstream sites. The abundance of some benthic fishes was influenced by the dams, including that of the Neosho madtom *Noturus placidus*, which was lowest immediately upstream and downstream from dams, and those of the suckermouth minnow *Phenacobius mirabilis*, orangethroat darter *Etheostoma spectabile*, and slenderhead darter *Percina phoxocephala*, which were highest in downstream treatment sites. Although limited to one system during a 1-year period, this study suggests that the effects of lowhead dams on fishes, macroinvertebrates, and habitat are similar to those reported for larger dams, providing important considerations for riverine ecosystem conservation efforts.

About 2 million dams exist in the United States, including 75,000 dams over 2 m in height; Kansas ranks second in dam number among all states, with 5,699 dams (Shuman 1995). Dams fragment rivers, reducing connectivity and resulting in negative effects on stream biota upstream and downstream from the impoundment (Doeg and Koehn 1994; Rabeni 1996; Kanehl et al. 1997). Dams block movement of fishes and affect habitat and physicochemical conditions of streams by converting lotic habitats to lentic, changing streamflow, altering water quality, and modifying channel morphology and bed structure by increasing siltation upstream and erosion downstream (Watters 1996; Helfrich et al. 1999; Porto et al. 1999). These alterations cause changes in assemblage structure of fishes and macroinvertebrates via shifts in composition, abundance, and diversity both upstream and downstream from the impoundment. Although many studies have addressed effects of large dams on fishes (e.g., Martinez et al. 1994; Clarkson and Childs 2000; Wildhaber et al. 2000b), few have examined effects of lowhead dams (<4 m in height) (e.g., Benstead et al. 1999; Helfrich et al. 1999; Beasley and Hightower 2000), and none has done so in a midwestern U.S. stream.

Our objectives were to investigate possible effects of two lowhead dams on the fish and macroinvertebrate assemblages, habitat, and physicochemistry of the Neosho River, Kansas. We pre-

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Received March 10, 2003; accepted November 19, 2003

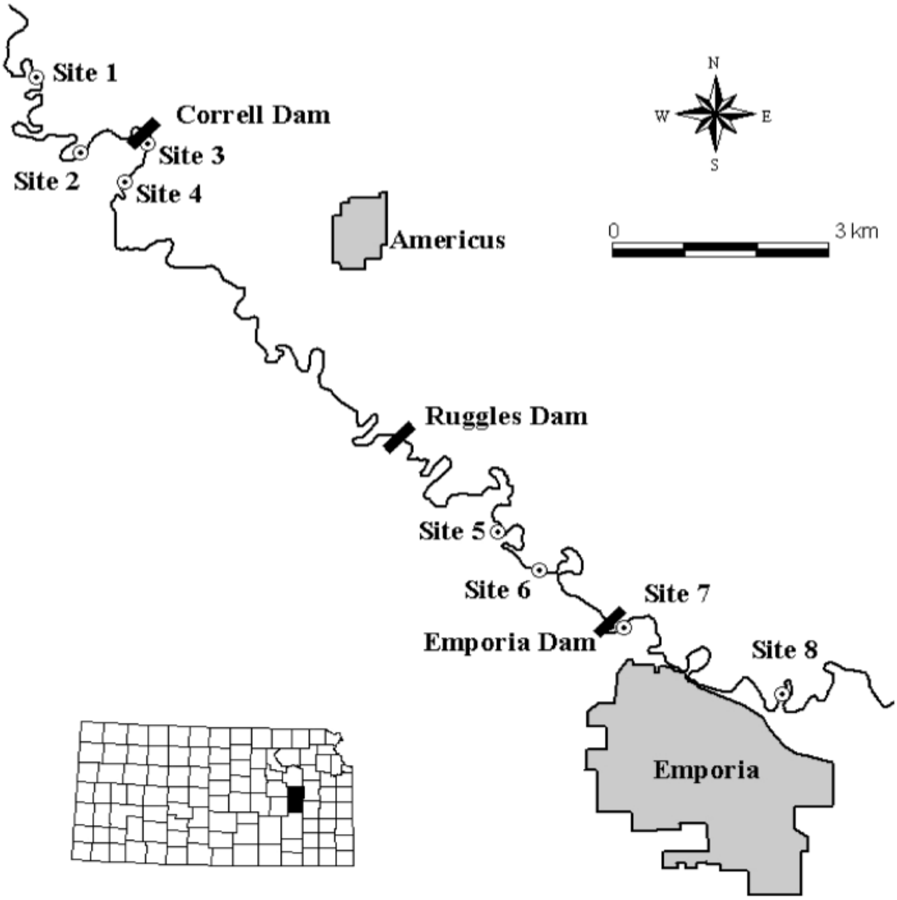

FIGURE 1.—Study area along the Neosho River in Lyon County, Kansas, where fish and macroinvertebrate assemblages, habitat variables, and physicochemical variables were sampled in relation to two lowhead dams (Correll and Emporia).

dicted that, because of differences in habitat and physicochemistry, fish and macroinvertebrate assemblages would differ among treatment sites (areas immediately upstream and downstream from lowhead dams) and reference sites (areas outside the direct zone of influence of dams). We had two a priori hypotheses. First, we hypothesized that because of inundation, upstream treatment sites would be deeper and have lower velocities, greater siltation, and greater substrate compaction than reference sites, resulting in gravel bars with fewer lotic-type fishes (e.g., madtoms *Noturus* spp. and darters *Etheostoma* and *Percina* spp.) and macroinvertebrates (e.g., mayflies [Ephemeroptera], stoneflies [Plecoptera], and caddisflies [Trichoptera]) but more lentic-type fishes (e.g., sunfishes *Lepomis* spp.) and macroinvertebrates (e.g., dragonflies [Odonata]). Upstream treatment sites would also have lower fish and macroinvertebrate abundance, richness, and evenness than reference sites due to habitat homogeneity. Second, we hypothesized that because of scouring, downstream treatment sites would be shallower and have higher velocities and a higher proportion of large substrate than reference sites, resulting in gravel bars with lower fish and macroinvertebrate abundance, richness, and evenness due to habitat homogeneity. We compared fish and macroinvertebrate abundance, richness, and evenness against 10 habitat variables and 7 physicochemical variables at upstream and downstream treatment and reference sites to test for localized effects of lowhead dams.

Methods

Study area and sampling methodology.—Our study sites were eight gravel bars situated along a 34-km stretch of the Neosho River in Lyon County, Kansas (Figure 1), within the Prairie Parkland

Province Ecoregion (Chapman et al. 2001). This portion of the Neosho River is a fifth-order stream impounded by three lowhead dams (Correll, Ruggles, and Emporia) designed for water supply. The Neosho River basin is primarily agricultural, with principal crops of mixed grasses, corn, wheat, and soybeans, and small riparian zones lie adjacent to crop fields. We sampled a segment of river with a mean gradient of 0.54 m/km and mean widths ranging from 14 to 35 m. Council Grove Reservoir is located near the headwaters of the Neosho River, 39 km upstream from site 1 (Figure 1). Designed for flood control, the reservoir impounds 1,310 ha at conservation pool and has a 5-m-diameter, epilimnetic outlet that regulates the flow of the Neosho River.

We sampled the sites in random order during daylight hours monthly from November 2000 to October 2001 (Tiemann 2002). Our eight sites were comprised of four site types (upstream reference, upstream treatment, downstream treatment, and downstream reference) positioned near two lowhead dams, Correll and Emporia (Figure 1). Given that the proportion of habitat made up of gravel bars is relatively constant along this length of the Neosho River, we selected sites based on presence of a gravel bar composed mainly of gravel smaller than 64 mm, proximity to the dams, and landowner permission. We focused on gravel bars because fish assemblages in these habitats are most influenced by disturbances from impoundments (Wildhaber et al. 2000b). The Correll Dam (38°31′19″N, 96°19′05″W) is situated in the upper part of the study area, is 2.3 m high and 45 m long, and impounds approximately 20 ha; this dam is no longer used for water supply. The Emporia Dam (38°26′11″N, 96°12′28″W), located downstream of the Correll Dam, is 3.7 m high and 22 m long, impounds approximately 25 ha, and is used as a water supply station. Because no pre-impoundment data on fish and macroinvertebrate assemblage structure were available, we chose treatment sites as the first samplable gravel bars directly upstream and downstream from each dam, and reference sites as the first samplable gravel bars outside the zone of direct dam influence on flow. Site 1 (upstream reference) and site 2 (upstream treatment, or inundated) were located 7.0 and 1.9 km upstream from the Correll Dam; site 3 (downstream treatment) and site 4 (downstream reference) were located 0.1 and 1.1 km downstream from the Correll Dam. Site 5 (upstream reference) and site 6 (upstream treatment, or inundated) were located 4.1 and 2.7 km upstream from the Emporia Dam; site 7 (downstream treatment) and site 8 (downstream reference) were located 0.1 and 7.0 km downstream from the Emporia Dam. We judged these sites to be appropriate and valid standards for presently free-flowing portions of the Neosho River.

We chose to examine dam effects on these relatively homogeneous mesohabitats because their inhabitants can be sampled more easily and efficiently and with more standardized methodology than bigger fishes of deep pools, for which several different types of gear would be necessary. Also, we chose gravel bars over pools or runs, not only because gravel bars are more samplable, but also because we expected gravel bars to be more affected (e.g., change of lentic to lotic) than pools or runs, and because gravel bar fauna generally is more sensitive than those of pools or runs. We used the sampling methods of Wildhaber et al. (2000a), which are appropriate for gravel bars. At each site, we evenly spaced five transects at least 5 m apart perpendicular to the river channel along the length of the gravel bar, and sampled up to five points on each transect, maintaining a minimum of 0.5 m between points. To minimize disturbance, we sampled transects from downstream to upstream, sampled points from near shore to far shore, and sampled variables in the following order at each site: fishes, water depth and stream velocity, substrate compaction and composition, macroinvertebrates, and physicochemistry. We could not sample sites 5 and 6 in December, January, or February, or site 2 in January or February, due to ice cover.

Macroinvertebrates and fishes.—To collect fishes, we kick-seined a 4.5-m^2 area at each point by disturbing the substrate 3 m upstream from a stationary, 1.5-m, 3-mm-mesh seine and proceeding downstream to the seine. We identified, counted, and released all fishes upon completion of sampling at a site.

We sampled macroinvertebrates at three random points per site in undisturbed substrate at the head of the gravel bar, in accordance with the strongly upstream-biased distribution of macroinvertebrates within gravel bars (Brown and Basinger-Brown 1984). We used a D-net to dredge a 0.09-m^2 area of substrate, and placed the sample into a bucket partially filled with water. We stirred the substrate for 2 min, strained the water through a 1-mm-mesh net, and preserved the contents in 45% isopropyl alcohol. In the laboratory, we sorted samples to family, except nematodes, which were identified to order.

Habitat quality and physicochemistry.—We assessed water depth, stream velocity, substrate compaction, and substrate composition at each point. We measured water depth with a meter stick and measured stream velocity with a Global Flow Probe FP101 current meter (Global Water, Gold River, California) positioned at 60% depth from the surface. We estimated substrate compaction by touch, and coded loose substrate as 1, medium as 2, firm as 3, and bedrock as 4 (Fuselier and Edds 1995). We sampled substrate with a shovel (Grost et al. 1991; Bain 1999) and estimated composition visually (Mullner et al. 2000) on a modified Wentworth scale to determine percentages of clay and silt, sand, gravel, pebble, cobble, boulder, and bedrock (Cummins 1962). We verified field estimates in the laboratory based on reference samples taken at the beginning and end of the study.

At the head of each gravel bar, upstream from the area sampled for fishes and macroinvertebrates, we measured temperature with a thermometer and dissolved oxygen with a Hach Model AL-36B kit (Hach Chemical Company, Loveland, Colorado). We then collected a water sample for laboratory analyses of alkalinity (Hach Model AL-36B kit), ammonia (Hach surface waters kit), and turbidity (Hach 2100P turbidimeter). By use of a vacuum pump and Pall type-A/C, glass-fiber filters, we filtered 100 mL of water through two filters and stored them at −10°C for subsequent chlorophyll-*a* and particulate organic carbon (POC) analyses at the Columbia Environmental Research Center (CERC) in Columbia, Missouri. At the CERC, we used a model 10-AU-005 Field Fluorometer (Turner Designs, Sunnyvale, California) to measure chlorophyll *a* and a Coulometrics Model 5014 carbon analyzer (UIC, Inc., Joliet, Illinois) to measure POC in the filtered samples.

Statistical analysis.—We averaged scores for all points to obtain a mean site value for each month and pooled these data for analysis at the treatment level (upstream reference, upstream treatment, downstream treatment, and downstream reference; Bain 1999; Wildhaber et al. 2000a). We used the Shapiro–Wilk test (Zar 1999) to evaluate distributions of means for normality, and we used Levene's test (Milliken and Johnson 1984) to evaluate homogeneity of variance. We $\log_{10}$ transformed nonnormal variables and arcsine-square-root-transformed proportional variables (Zar 1999). Transformation normalized the data, and we accepted the premise that *F*-statistics used to compare means of normally distributed variables are effective whether or not variances are equal, especially when sample sizes are equal or nearly so (Milliken and Johnson 1984). We used Tukey's studentized range test for pairwise comparisons among treatments. We eliminated fish species and macroinvertebrate taxa occurring in less than 5% of all samples (<5 of the 88 samples) from abundance analyses following Gauch (1982). We conducted all statistical tests in the Statistical Analysis System (SAS Institute, Inc., Cary, North Carolina) and considered them significant at *P*-values less than 0.05. Because of multiple tests, we applied a sequential Bonferroni correction ($\alpha = 0.05$), where appropriate, to help control overall experimental type I error rate (Rice 1989).

We performed separate three-way (site type, dam, and month) multivariate analyses of variance (MANOVA) to test for effects of lowhead dams on habitat and physicochemical variables and on fish and macroinvertebrate abundance (number per m^2). We used Wilk's lambda (λ; Zar 1999) to test for significance, the error term being the three-way interaction. We followed significant MANOVAs with a step-down analysis of covariance (ANCOVA) (Tabachnick and Fidell 1983) to examine the contributions of individual variables.

As a measure of assemblage response in terms of species diversity, we calculated richness (number of species) (Menhinick 1964) and evenness (equitability) (Williams 1964) of fishes and macroinvertebrates for each sample. Because richness values depend upon area sampled, we used rarefaction for fishes (unequal area sampled per site) but not for macroinvertebrates (equal area sampled per site) (Glowacki and Penczak 2000; Wildhaber et al. 2000a). We chose an evenness index that is independent of richness (Smith and Wilson 1996). We performed three-way MANOVAs on individual habitat and physicochemical variables, in addition to individual fish and macroinvertebrate taxa abundances, richness, and evenness, to further test for effects of lowhead dams. We also calculated Pearson's correlation coefficient to examine potential relationships of statistically significant habitat and physicochemical variables with fish and macroinvertebrate abundance, richness, and evenness.

Results

Habitat Quality and Physicochemistry

Habitat characteristics varied significantly among site types (MANOVA: $\lambda = 0.0003$; $n = 88$; $P < 0.0001$). Bedrock (step-down ANCOVA: $F = 425.23$; df = 60, 27; $P < 0.0001$) and substrate compaction (step-down ANCOVA: $F = 16.29$; df

TABLE 1.—Means (SDs in parentheses) and analysis of variance results (*F*-values, with *P*-values in parentheses) for habitat and physicochemical variables by site type in the Neosho River, Kansas, from November 2000 to October 2001; *N* is the number of samples per site type, lowercase letters within rows indicate significant Tukey's groupings, and asterisks indicate significant sequential Bonferroni-adjusted *P*-values.

Habitat or physicochemical variable	Upstream reference ($N = 21$)	Upstream treatment ($N = 19$)	Downstream treatment ($N = 24$)	Downstream reference ($N = 24$)	Site type *F*-value (df = 3, 27)
Water depth (cm)	48.4 (13.1) z	57.7 (4.4) y	24.0 (12.4) x	35.5 (15.1) z	57.16 (<0.0001)*
Stream velocity (m/s)	0.24 (0.15) z	0.05 (0.08) y	0.42 (0.27) x	0.32 (0.15) z	29.64 (<0.0001)*
Substrate compaction	1.9 (0.2) z	2.3 (0.3) y	2.8 (0.4) y	1.7 (0.3) z	99.77 (<0.0001)*
Clay/silt (<0.06 mm; %)	20.0 (61) z	11.5 (6.7) z	4.1 (5.4) y	14.8 (7.2) z	22.86 (<0.0001)*
Sand (0.06–1 mm; %)	5.0 (2.4) z	3.3 (1.4) z	2.8 (2.3) y	5.4 (3.2) z	5.83 (0.003)*
Gravel (2–15 mm; %)	41.7 (3.4) z	38.4 (6.9) z	24.5 (6.4) y	42.6 (6.9) z	85.85 (<0.0001)*
Pebble (16–63 mm; %)	33.0 (6.7) z	37.1 (5.7) z	22.7 (7.6) y	37.1 (6.6) z	26.09 (<0.0001)*
Cobble (64–256 mm; %)	0.3 (0.8) z	7.2 (1.9) y	4.3 (2.7) y	0.1 (0.2) z	54.08 (<0.0001)*
Boulder (>256 mm; %)	0.0 (0.0) z	2.5 (2.6) y	1.0 (2.0) z	0.0 (0.0) z	8.14 (0.0005)*
Bedrock (solid bottom)	0.0 (0.0) z	0.0 (0.0) z	40.7 (6.1) y	0.0 (0.0) z	425.25 (<0.0001)*
Temperature (°C)	14.9 (9.9)	15.0 (9.5)	15.5 (10.8)	15.3 (10.8)	0.45 (0.72)
Dissolved oxygen (mg/L)	8.9 (2.3)	8.3 (2.3)	10.0 (2.5)	9.4 (2.2)	1.70 (0.11)
Alkalinity (mg/L)	171.5 (49.2)	176.0 (58.8)	176.0 (54.4)	179.6 (65.3)	0.35 (0.79)
Ammonia (mg/L)	0.01 (0.03)	0.02 (0.04)	0.02 (0.04)	0.03 (0.05)	0.42 (0.74)
Chlorophyll *a* (μg/L)	678.4 (708.8)	680.0 (521.6)	535.1 (574.4)	421.0 (393.3)	1.38 (0.27)
POC[a] (mg/L)	170.1 (97.0)	164.6 (81.3)	179.7 (91.3)	166.3 (74.7)	0.68 (0.57)
Turbidity[b]	35.9 (35.8)	31.1 (21.8)	40.6 (47.6)	40.5 (40.9)	1.90 (0.15)

[a] Particulate organic carbon.

[b] Nephelometric turbidity units.

= 3, 30; $P < 0.0001$) contributed significantly to the variation among site types. Multivariate ANOVA indicated that all habitat variables differed significantly among site types (Table 1). Tukey's test indicated that upstream treatment sites were deeper and had slower velocities than reference sites, whereas downstream treatment sites were shallower and had faster velocities than reference sites (Figure 2); both treatment site types had higher substrate compaction than reference sites (Figure 2). Downstream treatment sites had a different particle size distribution compared to reference sites and upstream treatment sites. Tukey's test indicated that downstream treatment sites had lower percentages of clay/silt, sand, gravel, and pebble substrates, and a higher percentage of bedrock than reference sites and upstream treatment sites, whereas upstream treatment sites had a higher percentage of boulder than reference sites; both treatment site types had higher percentages of cobble than reference sites had (Figure 3).

Physicochemistry did not vary significantly among site types (MANOVA: $\lambda = 0.47$; $n = 82$; $P = 0.64$). None of the seven variables differed significantly among site types (Table 1).

Macroinvertebrates and Fishes

We collected 11,594 macroinvertebrates representing 26 identified taxa (12 orders encompassing 25 families, plus the nematode order Rhabditida), of which 23 were sufficiently common to be retained for abundance analysis (Table 2). Aquatic insects comprised 94.9% of the macroinvertebrates sampled. Chironomidae (order Diptera) was the most abundant family collected (64.0%), followed by Hydropsychidae (order Trichoptera; 10.3%) and Heptageniidae (order Ephemeroptera; 6.5%).

Macroinvertebrate abundance varied significantly among site types (MANOVA: $\lambda = 0.000006$; $n = 88$; $P < 0.0001$). Mean (±SD) macroinvertebrate abundance varied from 46.4 ± 6.2 individuals/m^2 in upstream reference sites and 48.1 ± 12.4 individuals/m^2 in upstream treatment sites to 25.9 ± 6.6 individuals/m^2 in downstream treatment sites and 55.4 ± 1.8 individuals/m^2 in downstream reference sites. Abundances of 12 of the 23 taxa were significantly different among site types (Table 2). Abundances of Culicidae (Diptera) (step-down ANCOVA: $F = 40.84$; df = 60, 27; $P < 0.0001$), Lestidae (order Odonata) (step-down ANCOVA: $F = 7.48$; df = 3, 33; $P = 0.0002$), Chironomidae (step-down ANCOVA: $F = 9.62$; df = 3, 32; $P < 0.0001$), and Heptageniidae (step-down ANCOVA: $F = 9.86$; df = 3, 1; $P < 0.0001$) contributed significantly to variation in abundance among site types. Tukey's test indicated that Culicidae, Chironomidae, and Lestidae had higher abundances at upstream treatment sites compared with other site types, Chironomidae had lower abundance at downstream treatment sites com-

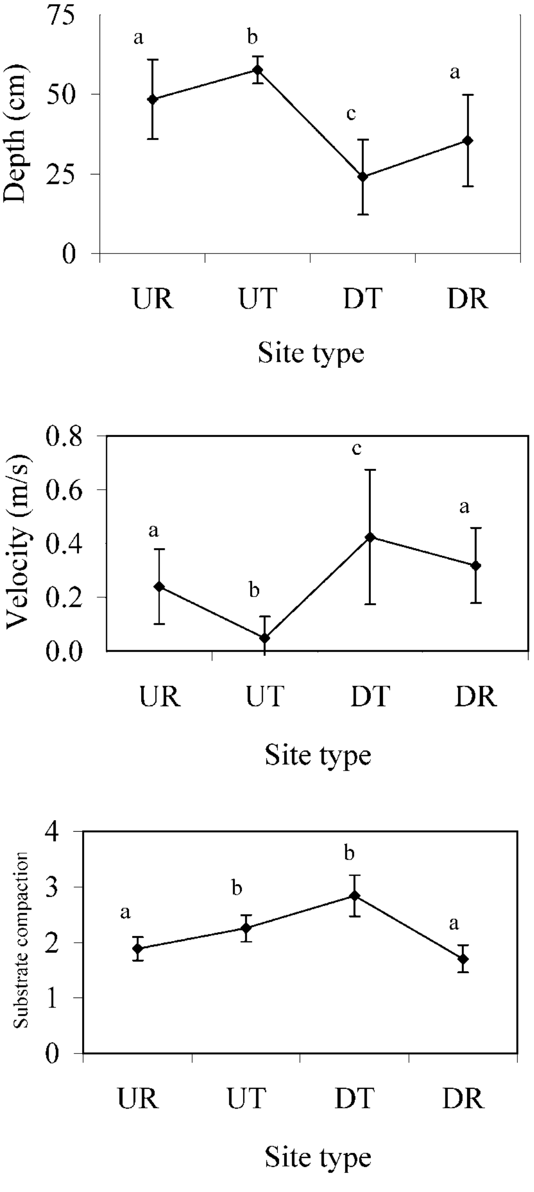

FIGURE 2.—Mean water depth, stream velocity, and substrate compaction (±SD) per site type (UR = upstream reference; UT = upstream treatment; DT = downstream treatment; DR = downstream reference) in the Neosho River, Kansas, November 2000–October 2001. The lowercase letters in the lowest panel indicate significant groupings according to Tukey's test.

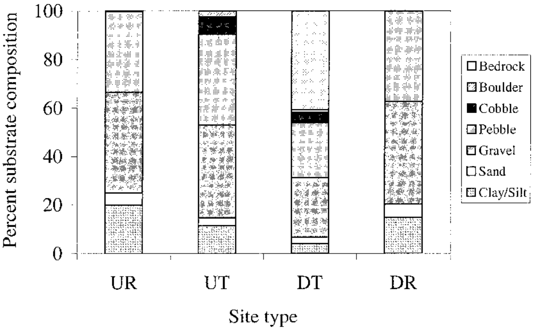

FIGURE 3.—Mean substrate composition percentages per site type (UR = upstream reference; UT = upstream treatment; DT = downstream treatment; DR = downstream reference) in the Neosho River, Kansas, November 2000–October 2001.

pared to other site types, and Heptageniidae had higher abundances in reference sites than treatment sites. Macroinvertebrate abundance was positively correlated with percent pebble substrate (Pearson's correlation: $r = 0.46$; $P < 0.0001$) and negatively correlated with percent bedrock substrate (Pearson's correlation: $r = -0.32$; $P = 0.0003$).

Macroinvertebrate taxa richness did not significantly differ among site types (ANOVA: $F = 0.74$; df = 3, 27; $P = 0.54$), but evenness did (ANOVA: $F = 8.37$; df = 3, 27; $P = 0.0004$). Mean macroinvertebrate evenness varied from 0.48 ± 0.01 in upstream reference sites and 0.39 ± 0.18 in upstream treatment sites to 0.41 ± 0.01 in downstream treatment sites and 0.47 ± 0.03 in downstream reference sites. Tukey's test indicated that upstream treatment sites had lower evenness than reference sites and downstream treatment sites. Neither macroinvertebrate taxa richness nor evenness was significantly correlated with fish abundance or any habitat or physicochemical variable.

In 88 samples, we caught 15,222 fish representing 10 families, 19 genera, and 31 species, of which 21 species were sufficiently common to be retained for abundance analysis (Table 3). Eleven species occurred at all eight sites (central stoneroller, red shiner, ghost shiner, bluntnose minnow, bullhead minnow, channel catfish, orangespotted sunfish, bluegill, orangethroat darter, logperch, and slenderhead darter). Red shiner was the most abundant species collected (47.8%), followed by ghost shiner (10.6%) and bluntnose minnow (10.0%).

Fish abundance varied significantly among site types (MANOVA: $\lambda = 0.0002$; $n = 88$; $P < 0.0001$). Mean fish abundance varied from 1.37 ± 0.29 fish/m^2 in upstream reference sites and 1.85 ± 0.52 fish/m^2 in upstream treatment sites to 2.68 ± 0.42 fish/m^2 in downstream treatment sites and 3.09 ± 0.44 fish/m^2 in downstream reference sites. Abundances of 4 of the 21 species were signifi-

TABLE 2.—Mean macroinvertebrate taxa abundance per square meter (SDs in parentheses) and analysis of variance results (*F*-values, with *P*-values in parentheses) by site type in the Neosho River, Kansas, from November 2000 to October 2001; *N* is the number of samples per site type, lowercase letters within rows indicate significant Tukey's groupings, and asterisks indicate significant sequential Bonferroni-adjusted *P*-values.

Benthic invertebrates	Upstream reference (N = 21)	Upstream treatment (N = 19)	Downstream treatment (N = 24)	Downstream reference (N = 24)	Site type *F*-value (df = 3, 27)
Order Ephemeroptera					
Potamanthidae	0.23 (0.03)	0.22 (0.00)	0.06 (0.02)	0.40 (0.03)	3.35 (0.03)
Baetidae	1.38 (0.42) z	0.40 (0.21) y	0.24 (0.13) y	1.94 (0.29) z	10.75 (<0.0001)*
Heptageniidae	4.43 (0.85) z	1.75 (1.14) y	0.96 (0.19) y	4.18 (0.13) z	21.67 (<0.0001)*
Order Plecoptera					
Perlidae	0.64 (0.11)	1.39 (0.68)	0.14 (0.04)	0.72 (0.04)	7.14 (0.001)
Order Trichoptera					
Limnephilidae	0.01 (0.00)	0.04 (0.00)	0.03 (0.00)	0.10 (0.03)	0.79 (0.65)
Hydropsychidae	6.52 (0.50) z	1.66 (1.05) y	2.71 (0.91) y	6.75 (0.10) z	18.30 (<0.0001)*
Order Odonata					
Gomphidae	0.00 (0.00)	0.12 (0.00)	0.00 (0.00)	0.00 (0.00)	7.17 (0.001)
Lestidae	0.00 (0.00) z	0.95 (0.01) y	0.00 (0.00) z	0.00 (0.00) z	26.00 (<0.0001)*
Order Coleoptera					
Carabidae	0.75 (0.06) z	0.15 (0.08) y	0.31 (0.02) y	1.11 (0.00) y	19.13 (<0.0001)*
Dytiscidae	1.98 (0.56)	0.26 (0.16)	0.64 (0.31)	1.86 (0.12)	5.36 (0.005)
Gyrinidae	0.73 (0.22) z	0.13 (0.07) y	0.15 (0.01) y	0.74 (0.05) z	12.31 (<0.0001)*
Order Hemiptera					
Corixidae	0.00 (0.00) z	0.12 (0.03) y	0.00 (0.00) z	0.01 (0.00) z	11.71 (<0.0001)*
Belostomatidae	0.00 (0.00)	0.18 (0.01)	0.01 (0.00)	0.04 (0.00)	5.56 (0.004)
Order Diptera					
Chironomidae	26.78 (2.91) z	34.57 (6.45) y	17.29 (3.96) x	33.71 (0.46) z	25.30 (<0.0001)*
Chaoboridae	0.73 (0.06)	0.48 (0.05)	0.31 (0.08)	0.65 (0.07)	1.04 (0.39)
Culicidae	0.00 (0.00) z	1.15 (0.34) y	0.03 (0.00) z	0.00 (0.00) z	40.84 (<0.0001)*
Simuliidae	0.87 (0.35)	0.65 (0.27)	1.38 (0.66)	1.04 (0.05)	0.31 (0.82)
Order Oligochaeta					
Tubificidae	0.77 (0.11)	1.20 (0.40)	0.35 (0.11)	0.76 (0.09)	3.60 (0.03)
Order Rhynchobdellida					
Glossiphoniidae	0.00 (0.00) z	0.29 (0.10) y	0.03 (0.00) z	0.00 (0.00) z	19.70 (<0.0001)*
Order Heterodonta					
Corbiculidae	0.18 (0.00) z	0.00 (0.00) y	0.03 (0.00) y	0.90 (0.23) z	10.43 (<0.0001)*
Order Gastropoda					
Lymnaeidae	0.00 (0.00)	0.34 (0.09)	0.32 (0.13)	0.00 (0.00)	6.30 (0.002)
Order Decapoda					
Cambaridae	0.29 (0.01) z	1.69 (0.84) y	0.86 (0.00) y	0.22 (0.02) z	8.57 (0.0004)*
Order Rhabditida	0.07 (0.03)	0.24 (0.09)	0.08 (0.04)	0.21 (0.13)	0.52 (0.67)

cantly different among site types (Table 3). Abundances of orangethroat darter (step-down ANCOVA: $F = 14.86$; df = 60, 27; $P < 0.0001$) and suckermouth minnow (step-down ANCOVA: $F = 5.96$; df = 3, 72; $P = 0.001$) contributed significantly to variation in abundance among site types. Tukey's test indicated that the orangethroat darter, suckermouth minnow, and slenderhead darter were more abundant in downstream treatment sites compared to other sites, and that the Neosho madtom was less abundant in treatment sites compared to reference sites (Figure 4). Fish abundance was not significantly correlated with macroinvertebrate abundance or any habitat or physicochemical variable.

Fish species richness did not significantly differ among site types (ANOVA: $F = 2.83$; df = 3, 27; $P = 0.06$), but evenness did (ANOVA: $F = 4.83$; df = 3, 27; $P = 0.008$). Mean evenness varied from 0.55 ± 0.02 in upstream reference sites and 0.52 ± 0.05 in upstream treatment sites to 0.44 ± 0.07 in downstream treatment sites and 0.45 ± 0.04 in downstream reference sites. Tukey's test indicated that both upstream site types had higher

TABLE 3.—Mean fish species abundance per square meter (SDs in parentheses) and analysis of variance results (*F*-values, with *P*-values in parentheses) by site type in the Neosho River, Kansas, from November 2000 to October 2001; *N* is the number of samples per site type, lowercase letters within rows indicate significant Tukey's groupings, and asterisks indicate significant sequential Bonferroni-adjusted *P*-values.

Fishes	Upstream reference (*N* = 21)	Upstream treatment (*N* = 19)	Downstream treatment (*N* = 24)	Downstream reference (*N* = 24)	Site type *F*-value (df = 3, 27)
Cyprinidae					
Central stoneroller					
Campostoma anomalum	0.013 (0.008)	0.037 (0.023)	0.028 (0.011)	0.025 (0.000)	1.10 (0.37)
Red shiner					
Cyprinella lutrensis	0.386 (0.002)	0.454 (0.069)	1.421 (0.031)	1.773 (0.095)	4.53 (0.01)
Ghost shiner					
Notropis buchanani	0.310 (0.033)	0.333 (0.167)	0.193 (0.005)	0.156 (0.034)	0.86 (0.47)
Sand shiner					
N. stramineus	0.015 (0.009)	0.006 (0.000)	0.006 (0.002)	0.049 (0.031)	4.00 (0.02)
Suckermouth minnow					
Phenacobius mirabilis	0.019 (0.011)^{a}	0.001 (0.000)^{a}	0.064 (0.009)^{b}	0.027 (0.008)^{a}	14.38 (<0.0001)*
Bluntnose minnow					
Pimephales notatus	0.200 (0.055)	0.321 (0.138)	0.215 (0.061)	0.341 (0.047)	1.24 (0.31)
Slim minnow					
Pimephales tenellus	0.048 (0.032)	0.091 (0.000)	0.020 (0.001)	0.091 (0.035)	6.42 (0.002)
Bullhead minnow					
Pimephales vigilax	0.091 (0.035)	0.057 (0.019)	0.041 (0.008)	0.233 (0.067)	4.73 (0.009)
Catostomidae					
Golden redhorse					
Moxostoma erythrurum	0.001 (0.000)	0.003 (0.000)	0.001 (0.000)	0.001 (0.000)	0.56 (0.65)
Ictaluridae					
Channel catfish					
Ictalurus punctatus	0.022 (0.003)	0.008 (0.005)	0.014 (0.000)	0.019 (0.008)	2.24 (0.74)
Stonecat					
Noturus flavus	0.003 (0.000)	0.000 (0.000)	0.006 (0.001)	0.005 (0.000)	1.12 (0.38)
Neosho madtom					
N. placidus	0.014 (0.007)^{a}	0.003 (0.001)^{b}	0.001 (0.000)^{b}	0.021 (0.003)^{a}	9.66 (0.0002)*
Poeciliidae					
Western mosquitofish					
Gambusia affinis	0.006 (0.000)	0.004 (0.001)	0.028 (0.019)	0.000 (0.000)	0.93 (0.44)
Centrarchidae					
Green sunfish					
Lepomis cyanellus	0.000 (0.000)	0.009 (0.001)	0.010 (0.003)	0.012 (0.002)	0.95 (0.42)
Orange spotted sunfish					
L. humilis	0.097 (0.045)	0.331 (0.041)	0.083 (0.027)	0.178 (0.039)	3.73 (0.03)
Bluegill					
L. macrochirus	0.005 (0.002)	0.008 (0.001)	0.005 (0.003)	0.003 (0.001)	0.65 (0.59)
Percidae					
Orangethroat darter					
Etheostoma spectabile	0.041 (0.028)^{a}	0.048 (0.013)^{a}	0.133 (0.010)^{b}	0.037 (0.021)^{a}	14.86 (<0.0001)*
Logperch					
Percina caprodes	0.009 (0.002)	0.014 (0.001)	0.023 (0.013)	0.006 (0.002)	2.87 (0.05)
Channel darter					
P. copelandi	0.009 (0.000)	0.013 (0.000)	0.000 (0.000)	0.015 (0.008)	2.65 (0.07)
Slenderhead darter					
P. phoxocephala	0.007 (0.017)^{a}	0.102 (0.046)^{a}	0.377 (0.216)^{b}	0.095 (0.044)^{a}	9.85 (0.0001)*
Sciaenidae					
Freshwater drum					
Aplodinotus grunniens	0.000 (0.000)	0.002 (0.000)	0.002 (0.000)	0.002 (0.000)	0.24 (0.87)

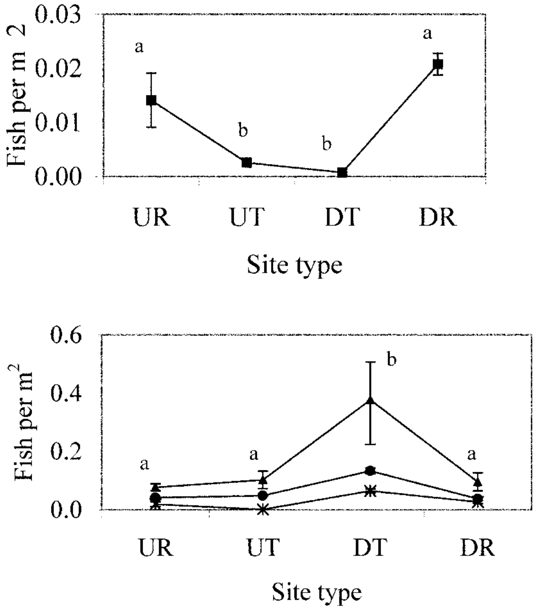

FIGURE 4.—Mean abundances (±SD) of Neosho madtom (top panel) and suckermouth minnow (asterisks), orangethroat darters (circles), and slenderhead darters (triangles) (bottom panel) abundance per for each site type (UR = upstream reference; UT = upstream treatment; DT = downstream treatment; DR = downstream reference) in the Neosho River, Kansas, November 2000–October 2001. The lowercase letters indicate significant groupings according to Tukey's test.

evenness than downstream treatment sites, but only upstream reference sites differed from downstream reference sites. Neither fish species richness nor evenness was significantly correlated with macroinvertebrate abundance or any habitat or physicochemical variable.

Discussion

A strong correlation exists between habitat variables and fish and macroinvertebrate assemblages, which implies that fragmentation and modification (e.g., inundation, scouring, and channelization) of riverine habitat can have profound effects on biotic integrity, including declines in abundance and diversity of macroinvertebrates and fishes (Neves and Angermeier 1990; Dynesius and Nilsson 1994; Luttrell et al. 1999). Although our study was conducted only during a period of 1 year and on a single river, our data demonstrate influences of lowhead dams on habitat quality, macroinvertebrates, and fishes in this midwestern stream. Correll and Emporia dams affected water depth, stream velocity, and substrate characteristics immediately upstream and downstream, with resultant effects on macroinvertebrate and fish assemblage structure. Results were similar to those for lowhead dams in other parts of North America (e.g., Helfrich et al. 1999; Porto et al. 1999; Beasley and Hightower 2000), and to a lesser extent resembled those for large dams (e.g., Martinez et al. 1994; Camargo and Voelz 1998; Wildhaber et al. 2000b).

Habitat Quality and Physicochemistry

As a stream is deepened, water velocity is decreased and its ability to carry sediment in the water column is reduced, generally resulting in increased sedimentation of the substrate (Kondolf 1997; Wood and Armitage 1997). Our upstream treatment sites were deeper and had lower velocity and higher substrate compaction than the other site types, but percentages of fine substrates were not significantly different from those of reference sites. Upstream treatment sites had a higher proportion of larger particles (cobble and boulder), perhaps as a result of the parent material existing prior to inundation or the relationship between mean current velocity and the size of particles that can be transported after inundation. Our study could not adequately address this issue. Downstream treatment sites were shallower and had higher velocities than other site types, and differed from other site types in 8 of 10 substrate characteristics. Treatment sites immediately downstream from the dams had greater substrate compaction and larger mean substrate size, which was reflected in more bedrock and lower percentages of clay/silt, sand, gravel, and pebble compared to reference sites. Over time, water flowing over these dams appears to have scoured finer substrates and taken the gravel bar down to bedrock, which accounted for the differences in bedrock among sites. A coarsening of substrate can result from streambed erosion by ''sediment-hungry'' release waters with increased velocity, and this process typically reduces habitat diversity (Kondolf 1997; Camargo and Voelz 1998). In rivers with large dams, effects on substrate size composition typically are greatest immediately downstream from the dam, causing scouring of organisms that sometimes leaves streambeds devoid of much of their fauna (Camargo and Voelz 1998).

Mean daily extraction from the Emporia water supply station was about 30 million liters (~0.34 m^3/s) (City of Emporia 2001), whereas mean daily discharge from Council Grove Reservoir during our study was approximately 1.73 m^3/s (USACE

2001). During August, site 7 (immediately downstream from Emporia Dam) had periods of no flow due to water extraction levels that exceeded Council Grove Reservoir discharge. We found no previous reports of the effects of lowhead dams and water extraction. Water extraction behind lowhead dams could indirectly degrade downstream substrate and affect benthic organisms. Reduced discharge, whether natural or artificial, can expose portions of gravel bars and cause compaction by the drying of organic material in interstitial spaces. If the substrate remains compacted following return to normal water levels, benthic organisms could be forced into less suitable areas, resulting in decreased survival (Wildhaber et al. 2000a; Bulger and Edds 2001).

Physicochemistry values were within the range reported by Wildhaber et al. (2000a) and Bulger and Edds (2001) for undammed portions of the Neosho River, and there were no significant differences among site types for any of the seven physicochemical variables. Unlike large dams (Wildhaber et al. 2000b), the lowhead dams we studied did not seem to affect physicochemistry of the Neosho River, perhaps because of lower water retention time. Although Hach kits do not provide sufficient accuracy or precision to be defensible, our results are comparable among our sites because we used the same kits throughout the study. In addition, our water quality data were comparable to those reported in other studies in the Neosho River.

Effects of Adjacent Dams

Ruggles Dam is another lowhead structure that impounds the Neosho River between the Correll and Emporia dams (Figure 1); we were unable to obtain landowner permission to sample around this dam. As with most North American rivers (Benke 1990), the Neosho River is highly regulated, having 2 reservoir dams and 15 lowhead dams in Kansas. We could not remove the potential effects of these other dams. Rather, in our analysis of the localized impacts of the Correll and Emporia dams, we chose reference sites outside the zone of direct dam influence on flow; these reference sites represented the normal condition for presently undammed portions of the Neosho River and therefore acted as appropriate and valid standards. Downstream effects of lowhead dams depend on dam size, hydrology, geology, faunal composition, and other factors (Baxter 1977).

Council Grove Reservoir Dam, 39 km upstream from site 1, also affects the river. During our study, discharge of the Neosho River at Americus (Figure 1) (USGS 2001) mirrored releases from Council Grove Reservoir (USACE 2001). However, because Council Grove is not a hydroelectric facility and because it has epilimnetic release, it produces no regular pulses of discharge and no alteration of the thermal regime downstream. Given these factors, the facility’s relatively small size, and the considerable downstream distance of our study area from the reservoir, we felt confident in the assumption that our sites were outside the direct influence of this dam.

Macroinvertebrates and Fishes

Macroinvertebrate abundance was lowest at downstream treatment sites, perhaps as a result of substrate coarsening and reduced habitat diversity (Baxter 1977; Kondolf 1997; Camargo and Voelz 1998). Macroinvertebrate abundance depends upon presence of a mixture of heterogeneous gravel, pebble, and cobble substrates, and moderate, consistent flow (Waters 1995), which were not characteristics of our downstream treatment sites. Macroinvertebrates inhabiting degraded streambed substrates are subjected to scouring, which could make them more susceptible to predation through dislodgment (Newcombe and MacDonald 1991). No macroinvertebrate taxon had higher abundance immediately downstream from the dams.

Macroinvertebrate taxa richness was not significantly different among site types, but evenness was lowest at upstream treatment sites, as the fauna was dominated by some resilient lentic taxa (Merritt and Cummins 1996), including Culicidae, Chironomidae, and Lestidae, that reached their highest abundances or occurred only at these inundated sites. More-sensitive, lotic taxa (Merritt and Cummins 1996), including Baetidae (Ephemeroptera), Heptageniidae, and Hydropsychidae, were less abundant at upstream and downstream treatment sites than at reference sites. Given that these organisms are good environmental indicators (Brown and Basinger-Brown 1984; Brown and Brussock 1991; Merritt and Cummins 1996), our results suggest that the lowhead dams we studied have negatively impacted habitat quality of the Neosho River.

Fish species richness did not differ significantly among site types. Evenness was lower at downstream reference sites, mainly due to a February sample in which 606 of 607 fish were red shiners. Fish abundance was highest at downstream reference sites and lowest at upstream reference sites.

Helfrich et al. (1999) suggested that a series of lowhead dams might present a serious cumulative challenge to fish passage, leading to gradual alteration of fish assemblage structure in a river. Given the presence of 17 dams on the Kansas portion of the Neosho River, such extensive modification could have a collective impact on fish populations. Although our study did not specifically address this issue, we did not see a significant longitudinal effect on fish assemblages (Gillette 2002).

Differences in fish assemblage structure were reflected mainly in abundances of benthic species immediately upstream and downstream from the dams. For example, abundance of the federally listed Neosho madtom was lower at upstream treatment sites, which had more cobble and boulder, deeper, slower water, and higher substrate compaction than reference sites. The Neosho madtom and many other substrate-oriented fishes are habitat specialists whose abundances vary according to stream velocity and substrate composition (Cross and Collins 1995; Pflieger 1997). Downstream treatment sites had shallower water depths, greater stream velocities, and lower percentages of clay/silt, sand, gravel, and pebble substrates, favoring the orangethroat darter, slenderhead darter, and suckermouth minnow. These species prefer ample stream velocities and sites free of silt (Pflieger 1997), and they dominated the assemblage in downstream treatment areas. However, as with macroinvertebrates, scoured downstream treatment sites lacked the loosely compacted substrate required by many substrate-oriented fishes. For example, the Neosho madtom prefers loose, clean gravel/pebble substrate in moderate water depths and stream velocities (Fuselier and Edds 1994; Bulger and Edds 2001). Compared with the situation at reference sites, the abundance of this fish was significantly lower immediately downstream from the dams, where water was shallower and faster and where gravel and pebble substrates were less prevalent. Compaction of substrate in downstream treatment areas might force substrate-oriented fishes into less suitable areas, where they could experience lower survival rates (Bulger and Edds 2001). It is also possible that the larger interstitial spaces in the cobble, which was more abundant immediately upstream and downstream from the dams, might not offer as many macroinvertebrates to feed on or as much protection from predators for substrate-oriented fishes as do gravel and pebble (Wildhaber et al. 2000a). It should be noted that our upstream treatment sites were not located directly behind the dams because seining there was not possible. By standardizing our sampling to kick-seining of gravel bars, our collections were efficient and comparable among site types. However, this sampling was most effective for small, lotic fishes, and probably underrepresented some Neosho River fishes, including the larger lentic taxa like centrarchids and catostomids.

Conclusions

Our findings suggest that lowhead dams cause differences in habitat immediately upstream and downstream, producing effects on fish and macroinvertebrate assemblages that are similar to, but less extensive than, the effects of large dams. The dams in our study were associated with significant differences in water depth, stream velocity, substrate compaction, and substrate composition that appear to affect macroinvertebrate and fish abundance and evenness, especially for habitat specialists. Our study contributes insights into the effects of lowhead dams on riverine habitat and fish and macroinvertebrate assemblages in the Midwest. Additional studies in other drainages and regions, with differing faunas and hydrologic regimes, should be conducted to gain a better understanding of how lowhead dams affect the biology and hydrology of stream ecosystems. Knowledge of the effects of these barriers can be used in the conservation and protection of riverine biotic integrity.

Acknowledgments

This research was supported by the U.S. Geological Survey (USGS) under USGS Cooperative Agreement Number 00CRAG0025, with additional funding from an Emporia State University (ESU) Faculty Research and Creativity Grant and an ESU Graduate Student Research Grant. We thank the landowners (G. Gulde, W. Leffler and M. Leffler, P. Matile and D. Matile, L. Schlesener, Girl Scout Council of the Flint Hills, the City of Emporia, and ESU Natural Areas) for allowing access to the river. Thanks to D. Moore, L. Scott, and D. Zelmer of ESU for statistical advice and helpful discussions. B. Johnson of Trout Unlimited and H. Dodd of the Illinois Natural History Survey (INHS) provided background information. D. Mulhern and V. Tabor of the U.S. Fish and Wildlife Service loaned a Hach kit. R. Ferguson, B. Flock, B. Harkins, G. Head, G. Sievert, R. Sleezer, L. Sneed, and L. Westerman of ESU and J. Anderson of the Kansas Biological Survey provided technical assistance. B. Chance, J. Dean, L. Freeman, J. Howard, S. Sherraden, and I. Singh of ESU assisted with fieldwork, and J. Albers, A. Allert, J. Fairchild, B. Lakish, S. Olson, and C. Witte of the USGS assisted with water chemistry analysis. G. Levin, T. Rice, C. Taylor, and D. Thomas of INHS reviewed the manuscript and provided constructive criticism.

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