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Barry C. Poulton
U.S. Geological Survey, Columbia Environmental Research Center, Columbia, Missouri

Mark L. Wildhaber
U.S. Geological Survey, Columbia Environmental Research Center, Columbia, Missouri

Collette S. Charbonneau
U.S. Fish and Wildlife Service, Region 3 Field Office, Fort Snelling, Minnesota

James F. Fairchild
U.S. Geological Survey, Columbia Environmental Research Center, Columbia, Missouri

Brad G. Mueller
U.S. Geological Survey, Jackson Field Research Station, Jackson, Wyoming

See next page for additional authors

Authors
Barry C. Poulton, Mark L. Wildhaber, Collette S. Charbonneau, James F. Fairchild, Brad G. Mueller, and Christopher J. Schmitt

A LONGITUDINAL ASSESSMENT OF THE AQUATIC MACROINVERTEBRATE COMMUNITY IN THE CHANNELIZED LOWER MISSOURI RIVER*

BARRY C. POULTON[1]*, MARK L. WILDHABER[1],
COLLETTE S. CHARBONNEAU[2], JAMES F. FAIRCHILD[1], BRAD G. MUELLER[3]
and CHRISTOPHER J. SCHMITT[1]

[1] *U.S. Geological Survey, Columbia Environmental Research Center, Columbia, Missouri, U.S.A.;* [2] *U.S. Fish and Wildlife Service, Region 3 Field Office, Fort Snelling, Minnesota, U.S.A.;* [3] *U.S. Geological Survey, Jackson Field Research Station, Jackson, Wyoming, U.S.A.*
(author for correspondence, e-mail: barry_poulton@usgs.gov)*

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Abstract. We conducted an aquatic macroinvertebrate assessment in the channelized reach of the lower Missouri River, and used statistical analysis of individual metrics and multimetric scores to identify community response patterns and evaluate relative biological condition. We examined longitudinal site differences that are potentially associated with water quality related factors originating from the Kansas City metropolitan area, using data from coarse rock substrate in flowing water habitats (outside river bends), and depositional mud substrate in slack water habitats (dike fields). Three sites above river mile (RM) 369 in Kansas City (Nebraska City, RM = 560; St. Joseph, RM = 530; Parkville, RM = 377) and three below (Lexington, RM = 319; Glasgow, RM = 228; Hermann, RM = 94) were sampled with rock basket artificial substrates, a qualitative kicknet method, and the Petite Ponar. We also compared the performance of the methods used. A total of 132 aquatic macroinvertebrate taxa were collected from the lower Missouri River; one third of these taxa belonged to the sensitive EPOT insect orders (Ephemeroptera, Plecoptera, Odonata, and Trichoptera). Rock baskets had the highest mean efficiency (34.1%) of the methods, and the largest number of taxa was collected by Ponar (n = 69) and kicknet (n = 69) methods. Seven of the 15 metrics calculated from rock basket data, and five of the nine metrics calculated from Ponar data showed highly significant differences (ANOVA, $P < 0.001$) at one or more sites below Kansas City. We observed a substantial reduction in net-spinning Trichoptera in rock habitats below Kansas City (Lexington), an increase in relative dominance of Oligochaeta in depositional habitats at the next site downstream (Glasgow), and lower relative condition scores in rock habitat at Lexington and depositional habitat at Glasgow. Collectively, these data indicate that some urban-related impacts on the aquatic macroinvertebrate community are occurring. Our results suggest that the methods and assessment framework we used in this study could be successfully applied on a larger scale with concurrent water and sediment chemistry to validate metrics, establish impairment levels, and develop a specific macroinvertebrate community index for the lower Missouri River. We recommend accomplishing this with longitudinal multi-habitat sampling at a larger number of sites related to all potential sources of impairment, including major tributaries, urban areas, and point sources.

Keywords: assessment, biological condition, habitat, Kansas City, macroinvertebrates, Missouri River, petite ponar, rock substrate

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1. Introduction

Aquatic macroinvertebrates are an important component in bioassessment studies designed to evaluate overall water resource quality (Shackleford, 1988; Fausch *et al.*, 1990; Karr and Kerans, 1991), food habits of benthic fishes (Todd and Stewart, 1985), ecological function (Benke *et al.*, 1979; Newswanger *et al.*, 1982; Benke *et al.*, 1986), and effects of specific anthropogenic disturbances (Hellawell, 1986; House *et al.*, 1993; Wildhaber and Schmitt, 1998) in flowing waters. Aquatic macroinvertebrates are valuable for determining biological condition because they are limited in mobility and complete the majority of their life cycle in water. They are effective indicators of historical conditions and integrate the combined effects of all impacts acting on a water body (Friedrich *et al.*, 1992), and are therefore especially valuable in cases where chemical-specific analysis cannot separate the cumulative effects of multiple stressors. Of the nation's 'great' rivers that have recently been defined as those with watersheds exceeding 3226 sq mi. (Simon and Lyons, 1995), few macroinvertebrate community studies have been conducted. Large river macroinvertebrate community studies are greatly outnumbered by those in wadeable streams, partially due to greater ecological complexity, severely altered habitat conditions, and a resident aquatic fauna that is usually more poorly known and difficult to sample. Most community-level biological assessments in larger rivers have occurred in Europe (Battagazzore *et al.*, 1992; Camargo, 1992; Depauw *et al.*, 1994), although some have recently been conducted in the U.S. based on surveys of fishes (Gammon, 1991; Simon and Emery, 1995). However, no community-level longitudinal evaluations of biological condition have been completed for the Missouri River. Most lower Missouri River studies were conducted during or shortly after reservoir construction in North and South Dakota and concentrated on comparisons between channelized and non-channelized portions along the Nebraska, South Dakota, and Iowa borders (McMahon *et al.*, 1972; Wolf *et al.*, 1972; Nord and Schmulbach, 1973; Hesse and Mestl, 1985). Others were designed to evaluate effects of power plant discharges (Camp Dresser and McKee, 1981; Carter *et al.*, 1982) or to describe biological assemblages that utilize river training structures such as rip-rap, revetments, and dike fields (Burress *et al.*, 1982; Atchison *et al.*, 1986). Early literature suggested that turbidity and high current velocities in the lower river were unfavorable for optimum macroinvertebrate productivity, and the channelized portion was thought to have low species richness (Berner, 1951; Morris *et al.*, 1968; Hansen and Dillon, 1973). Other studies on channelized segments have documented differences in benthic density (Beckett and Pennington, 1986), productivity (Dixon, 1986; Mestl and Hesse, 1993), and species richness (Barnum and Bachmann, 1988) among habitats or have characterized communities inhabiting specific habitats (Jennings, 1979; Brunsing, 1993). The macroinvertebrate community monitoring programs that exist on other great rivers such as the Ohio (Ohio River Valley Water Sanitation Commission, ORSANCO, unpublished) are in the early stages of development (E. Emery, pers. comm.), or in the case of the Mississippi River system, have only relied on specific indicator taxa (e.g. Long Term Resource Monitoring Program; Thiel and Sauer, 1995).

There is a critical need to identify relationships between macroinvertebrate community structure and the combined effects of water quality and habitat degradation in all interjurisdictional rivers, because aquatic resources are still declining in many systems. There is an even greater need for developing and implementing bioassessments in large rivers to fulfill legislative mandates regarding ecological integrity, use attainment status for aquatic life, and the identification of causes and sources of impairment (Yoder, 1991; Karr, 1993; Barbour, 1997). In the 1970s, Munger *et al*. (1974) documented benthic macroinvertebrate community responses related to urban and industrial sources of water pollutants in the lower Missouri River below Kansas City and St. Joseph, MO. During the same period, the U.S. Army Corps of Engineers (USACE) also conducted several studies that documented the success of macroinvertebrate sampling methods (Mathis *et al*., 1982; Beckett and Pennington, 1986) and included initial characterization of communities inhabiting different habitats and substrates (Atchison *et al*., 1986; Barnum and Bachmann, 1988). Since that time, improvements in the treatment of municipal and industrial wastewater and other regulatory changes have been implemented under the Clean Water Act. In recent years, ecologists have begun to incorporate the concept of biological integrity into the assessment frameworks currently used to evaluate water resource quality (Karr, 1991, 1993; Friedrich *et al*., 1992). It is now recognized that implementation of chemical criteria alone is not enough to substantially improve ecological integrity in all cases, since physical habitat degradation, effects of persistent contaminant mixtures, and exotic species are still impeding biological recovery in many systems (U.S. EPA, 1990). The U.S. EPA now requires state pollution control agencies to include more comprehensive, systematic evaluations of biological condition and impairment in lotic systems. The currently accepted process of biological evaluation for flowing waters includes site comparisons based on multimetric approaches and interpretation of community attributes (i.e. metrics, Karr, 1993; Barbour *et al*., 1995). Application of this process to both current and historical macroinvertebrate data is considered a valid and sensitive approach as long as methods are comparable (Kerans *et al*., 1992; Barbour *et al*., 1996). However, none of the previous macroinvertebrate studies on the lower Missouri River have used this approach to determine biological integrity or relative condition, even though the entire lower segment in the state of Missouri is listed as an impaired waterway under Section 303(d) of the Clean Water Act and is slated for mandatory development of TMDL's (total maximum daily loads, Missouri Dept. of Natural Resources, unpublished). This is largely due to difficulties in data collection, lack of standard methods, differences in criteria and aquatic life use designations among states, and the unknown pollution tolerances for large river aquatic species.

The goal of our study was to utilize aquatic macroinvertebrate communities as a screening tool for evaluating relative biological condition in relation to effects of water quality factors originating from the Kansas City metropolitan area. To achieve this goal, we utilized methods that would characterize the aquatic macroinvertebrate community in selected habitats on the lower, channelized portion of the Missouri River. We selected methods and assessment procedures for interpretation of community attributes that are widely used for evaluating wadeable streams (Karr and Kerans, 1991; Barbour *et al.*, 1995, 1999). Kansas City was selected as a potential source of cumulative stressors in the lower river based on previous literature that has identified combined sewer outflows, industrial pollutants, and other historical water quality problems directly downstream (Walter, 1971; Ford, 1980; Schmulbach *et al.*, 1992; Welsh, 1992). Our specific objectives were: (1) to evaluate longitudinal differences in macroinvertebrate community structure at six locations (three above Kansas City, and three below) in the mainstem of the lower Missouri River using multimetric community data from two important habitats, (2) to evaluate performance of specific gear types, (3) to compare our data with that of previous lower Missouri River studies conducted in the same habitats (Munger *et al.*, 1974; Carter *et al.*, 1982; Atchison *et al.*, 1986), and (4) to make recommendations for future large river macroinvertebrate assessments in this system.

2. Study Area

The Missouri River basin drains approximately 22 million ha of the U.S. and contains a population of over 25 million people. Lower portions of the river have been highly modified by reservoir discharges, channelization, and flood control; only 10% of the predevelopment floodplain of the lower Missouri River remains (Hesse, 1987). Modifications to facilitate agriculture and navigation between Sioux City, IA and St. Louis, MO have isolated the lower Missouri River from its historically productive floodplain (Hesse *et al.*, 1988; Junk *et al.*, 1989; Schmulbach *et al.*, 1992). Wing deflectors (dikes), rip-rap, and revetments composed of rock were added to reduce bank erosion as part of the quarry-stone construction method of channel modification approved in 1949 (Ferrell, 1996). By 1981, the USACE reported that channelization and bank stabilization projects had been completed for the lower 1187 km of the Missouri River. The outside bend of nearly all meanders has been reveted with rock or mattress blocks in the channelized section of the river (Ferrell, 1996), and rock is still added for maintenance and repair. In 1993 alone, 194,000 tons of waterway improvement material were carried by barges to maintain these river training structures (USACE, 1993). Channel modifications coincided with observed declines in the abundance of several Missouri River native fishes, many of which are considered benthic and feed on aquatic macroinvertebrates (reviewed by Hesse *et al.*, 1989; Hesse *et al.*, 1993). Water quality problems have also been documented in the lower Missouri River since 1910. The Kansas City metropolitan area contains municipal and industrial point source discharges, including petroleum refineries, industrial chemical manufacturing plants, and processing wastes from meatpacking and livestock yards (Ford, 1980; Schmulbach *et al.*, 1992). Oxygen depletion and associated fish kills caused by municipal sewage pollution occurred below Kansas City during the 1960s and 1970s, and the presence of oil, floating debris, and objectionable tastes and odors in drinking water and fish were considered problems during this period. More recently, elevated concentrations of polychlorinated biphenyls (PCB's) and organochlorine pesticides (OC's) such as chlordane have been reported in several Missouri River fishes, resulting in the issuance of public health advisories during the 1980s for the river reach between Kansas City and St. Louis (Missouri Dept. of Conservation, unpublished). Contaminant sampling with Semi-Permeable Membrane Devices (SPMD'S) has documented elevated levels of toxaphene, chlordane, dieldren, PCB's, and polycyclic aromatic hydrocarbons (PAHs) associated with urban runoff below Kansas City (Petty *et al.*, 1995, 1998). Aquatic community structure of fish and macroinvertebrates in the lower portions of the Missouri River system had been largely ignored until the recent 500 yr flood events in the 1990s, which created improvements in floodplain habitats and partial restoration of river connectivity.

3. Methods

To provide longitudinal site comparisons in the lower Missouri River, six sites were selected (three above and three below Kansas City) based on available access, and included one site directly below the metropolitan area where cumulative impacts, if present, would be the least difficult to detect (Figure 1): (1) Nebraska City, NE (RM = 560), (2) St. Joseph, MO (RM = 530), (3) Parkville, MO immediately above Kansas City (RM = 377), (4) Lexington, MO about 25 km below Kansas City (RM = 319), (5) Glasgow, MO (RM = 228), and (6) Hermann, MO (RM = 94). The sampling methods, season of collection, and habitats were chosen to yield repeatable, representative samples under the harsh conditions inherent with the Missouri River system–water level fluctuations, high current velocities, and a predominance of unstable substrates. The difficulty in obtaining macroinvertebrate samples under these conditions, and large rivers in general, has been identified previously (Anderson and Mason, 1968; Rosenberg and Resh, 1982). Artificial substrates such as baskets filled with rock or gravel (Jacobi, 1971) are among the most successful options for deep riverine habitats where water level fluctuations are common (Dickson *et al.*, 1971; Mason *et al.*, 1973; Slack *et al.*, 1986), and are consistent with other large river macroinvertebrate studies (Anderson and Mason, 1968; Depauw *et al.*, 1994). We selected late autumn as an index period for sampling because during this season, water levels in the Missouri River are relatively stable and provide the best opportunity for successful retrieval of artificial substrates. Stable water levels also allowed the use of qualitative kicknet sampling along the shoreline, with a method similar to that used in wadable streams (Lenat,

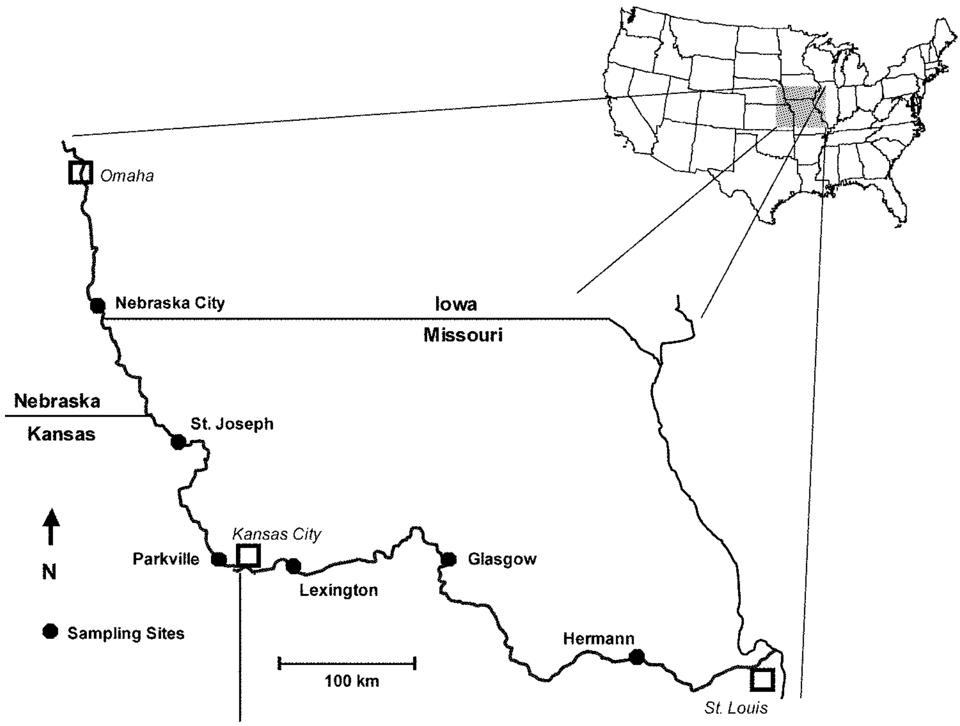

Figure 1. Map of aquatic macroinvertebrate sampling sites evaluated during a longitudinal assessment of biological condition in the channelized lower Missouri River.

1988; Shackleford, 1988; Barbour *et al.*, 1999). Two habitat types were selected for study: (1) main channel revetments or rock wing dikes with visible flow and coarse rock substrate, and (2) slack water areas behind wing dikes with no apparent flow containing soft-bottom mud substrate with visible organic matter.

3.1. Rock habitats

Artificial substrates were constructed of 3.5 cm mesh plastic aquaculture netting folded into a 15 cm cubical design and fastened together with plastic zip-ties. Baskets were tied onto an 8 m section of nylon rope fastened to the bank with steel rod about 1–2 m above the waterline. The baskets were filled with 4 L of 5 cm crushed limestone, tossed upstream into the current, and allowed to settle to the bottom in about 2–3 m of water with the rope positioned perpendicular to the bank. Limestone substrate was chosen because it closely simulated the construction materials used for wing dikes and revetments on the river. At each site, the baskets were deployed for six weeks of macroinvertebrate colonization from late October through early December 1996. To select segments of each revetment that conformed to a current velocity range of 0.4–0.7 m sec^{-1}, velocity measurements were taken at mid-depth with a Marsh McBurney 2000 digital current meter during rock basket deployment and immediately before basket retrieval in December. Substrates were retrieved from a boat by detaching the bank end of the rope, and gently raising the basket vertically through the water column in a hand-over-hand fashion, to reduce the possibility of organism loss due to vibrations or abrupt movements. Baskets were immediately placed into buckets at the waterline. Substrate was removed from the baskets and the entire mesh, rocks, and associated organic matter were rinsed in the buckets to isolate organisms. The sample was concentrated into a 530 μm sieve and placed into labeled 1 L sample jars with 90% ethanol preservative. Concurrently with rock basket sampling, a D-frame kicknet with 500 μm mesh was used to collect two, 100 organism field-sorted samples from rock habitats at each site – one at substrate deployment in October and one at basket retrieval in December. We expected similar community composition between kicknet and rock basket methods; kicknet samples were taken from the same locations as the rock basket deployment to provide method comparisons and identify any excessive loss of organisms or taxa during basket retrieval. Kicknet samples were taken by repeatedly disturbing rock and gravel substrate upstream of the net in about 0.5–1.0 m water depths along the river margin. Samples were placed in a large white tray, and alternating large and small organisms were picked from the tray and net in an attempt to reduce possible bias towards the larger sized individuals. One hundred organisms were selected over a 45–60 min period to acquire the maximum diversity, and were preserved in labeled sample bottles with 80% ethanol.

3.2. Depositional habitats

Slack water depositional areas behind wing dikes were sampled concurrently with rock basket retrieval, with a Petite Ponar as recommended for sediment-dwelling organisms in large rivers (Slack *et al.*, 1986; Burt *et al.*, 1991). We selected relatively stable locations corresponding with dike fields at inside river bends with no visible flow, and where dark-colored sediments containing fine organic matter had not been covered by sand from previous high water events. Five separate Ponar samples were taken and rinsed with a 500 μm sieve bucket. Organic matter and organisms were concentrated in the bucket, removed with a plastic spoon and forceps, and preserved with 80% ethanol in labeled 1 L jars.

3.3. Laboratory processing

Aquatic macroinvertebrates in Ponar samples were separated from debris in their entirety under a dissecting stereomicroscope using fine forceps and 10$\times$ magnification. Due to the large amounts of organic debris present in rock basket samples, a 0.75 m $\times$ 0.5 m subsampling tray marked with 54 numbered 5 cm square grids was used to randomly split these samples similar to the method described by Moulton *et al.* (2000). Debris and organisms from 11 of the 54 grids (20% of the sample) were removed and sorted in the same manner as the Ponar samples. Organisms from all methods and samples were identified to the lowest possible taxonomic level, usually genus or species. Midges (Diptera: Chironomidae) were mounted on glass slides with CMCP-10 mounting media (Masters Chemical Co.) and identified to genus level with a compound microscope. The Oligochaeta were identified to family; Tubificidae and Naididae were lumped together as one taxon due to the low numbers of Naididae and numerous tubificid fragments.

3.4. Statistical analysis

We evaluated the efficiency of different sampling methodologies (Elliot and Drake, 1981) using the total taxa archive that included every distinct taxon collected in the lower Missouri River by all methods (including light-trap data and other qualitative collections) from ongoing studies on the lower Missouri River since 1992. This reference taxa list was used as the basis for comparisons of community composition between the two habitats and among the three methods used, in addition to identifying taxa that were unique to one habitat or collected with only one method. The degree of resemblance in catch composition between methods was expressed as % taxa similarity, calculated by determining the percentage of overlap in taxa collected between the two methods being compared (Sanders, 1960; Boesch, 1977). Mean % efficiency was calculated separately for each method by dividing the mean number of taxa collected across all samples, by the total number of taxa collected by that method. The % of total taxa in each of the two habitats was also calculated for each method.

Aquatic macroinvertebrate community metrics used for this assessment were selected from those in current literature, and those that have potential value as indicators of biological condition for large rivers or specific habitats. We selected metrics 'a priori' from each of the categories outlined by Barbour *et al.* (1992, 1995) and from the Rapid Bioassessment Protocols (Plafkin *et al.*, 1989; Barbour *et al.*, 1999) which included commonly used indicators of biological condition in wadeable streams. The selected metrics and their abbreviations used throughout this paper include: (1) structure metrics (total taxa richness = TTR; percent Ephemeroptera, Plecoptera and Trichoptera = % EPT; EPT taxa richness = EPT; percent Chironomidae = % CH; Chironomidae taxa richness = CHR); (2) community balance metrics (Hilsenhoff Biotic Index, Hilsenhoff, 1982, 1987 = HBI; ratio of EPT and Chironomidae abundances = EPT/C; percent dominant taxon = % DT; percent model affinity, Novak and Bode 1992 = % MA); and (3) functional feeding group metrics (ratio of scrapers to filtering collectors = SFR). Assignment of functional groups for individual taxa followed the designations given in Merritt and Cummins (1996). Other commonly used metrics we included were the Shannon-Wiener Diversity Index (SDI), and percent Ephemeroptera (% EPH). Additional metrics were added to the analysis based on their potential use in describing large river community characteristics, and included richness of EPT and Odonata taxa (EPOT), percent EPOT (% EPOT), and percent large river restricted taxa (% LRRT). The selection of % LRRT as a metric was based on preliminary sampling that indicated several large river invertebrate species were present in the lower Missouri system, and is parallel to the choice of metrics for the large river fish guild used by Simon and Emery (1995). Density (DEN) was the only quantitative metric calculated in this study, and is expressed as number per m^2 for ponar samples and number per L interstitial volume for rock baskets. Interstitial volume was calculated by subtracting water displacement of the 5 cm limestone used as substrate in the rock baskets.

We performed two different statistical analyses on individual metrics: (1) comparisons among all six sites, and (2) comparing above versus below Kansas City. Since the five samples taken at each site were essentially independent of each other within a site, we used all samples in analyses for among site differences (n = 29, one sample missing from both Ponar and rock basket methods). Because the six sites were not independent, we averaged samples at each site to evaluate differences above versus below Kansas City (n = 6). Before any statistical analyses could be conducted, the data were analyzed for assumptions of normality using SAS/LAB software (SAS, 1992), and we tested variance constancy with Lavene's Test (Milliken and Johnson, 1984). After applying the transformations suggested by the software, the data for each metric were analyzed using one-way analysis of variance (ANOVA). The experiment-wise error rate was controlled using the Tukey method for multiple comparisons (SAS, 1992), and significant differences were indicated at the $P < 0.05$ level.

The individual metrics used for relative scoring of sites for both rock and depositional habitats were selected based on a combination of the following criteria: (1) discrimination among sites identified with statistical significance, (2) widespread historical use in wadeable stream studies, and (3) where a longitudinal pattern was observed even where there was no statistical significance among sites. The 4-metric score (TTR, EPT, HBI, SDI) for rock basket and kicknet sampling methods included the four standard attributes presently used for site scoring by the Missouri Dept. of Natural Resources for coarse substrate samples in wadeable streams as part of the development of state biocriteria (MDNR, unpublished and R. Sarver, pers. comm.). A 10 metric score was calculated for rock basket data and included the above four standard metrics, plus six additional metrics which included % EPOT, % CH, DEN, % LRRT, SFR, and EPT/C. The 5 metric score calculated from ponar data included the most appropriate metrics for depositional habitats based on the above criteria (TTR, % CH, SDI, DEN, and % EPH). We scored sites using a conservative percentile of 50% for each metric across all sites and split the lower 50% in half to provide reasonable cut-off ranges for progressively lower scores, similar to that suggested by Barbour *et al.* (1995) for cases where no true reference conditions exist. This approach was also used by Simon and Emery (1995) for fish community data from the Ohio River. For each site, individual metrics were scored as 5 (above 50th percentile), 3 (between 25th and 50th percentile), and 1 (lower quartile). Scores for each combination of metrics were obtained for each site by adding individual scores for each of the metrics used.

4. Results

4.1. Method and habitat comparisons

A total of 118 distinct macroinvertebrate taxa were collected from the Missouri River during this study, and an additional 14 species were collected with supplementary qualitative and blacklight collection methods. Of these 132 known taxa, approximately one third belong to the EPOT insect orders. In rock habitats, rock baskets had the highest mean efficiency (34.1%) but collected lower numbers of unique taxa (15) and lower percentage of the total number of taxa (Table I). Kicknet sampling collected a larger mean number of taxa (19.2) and the highest percentage of the total known taxa within the habitat (88.4%). Depositional habitats sampled with the Ponar yielded the largest number of unique taxa (22). Rock basket and kicknet methods were the most similar to each other, with a 75.3% overlap in species composition. Rock basket and Ponar methods were nearly as similar (73.1%); however, this high similarity is partially due to incidental taxa (N = 11) normally found in flowing water habitats that were collected frequently in depositional zones with Ponar sampling. Net-spinning caddisflies (Trichoptera: Hydropsychidae), several species of mayflies in the family Heptageniidae, and stoneflies (Plecoptera) were the dominant organisms colonizing the rock basket samples (Figure 2). The kicknet samples contained higher overall taxa richness in the EPOT orders. In particular, the number of stonefly and mayfly species found in kicknet samples was more than twice that colonizing rock basket samples (Figure 2). Oligochaeta and Chironomidae were the dominant organisms in the depositional habitat samples (Figure 2). Chironomidae taxa richness made up from 10% in the kicknet samples and up to 40% in the Ponar samples.

4.2. Site comparisons

Rock basket and ponar data indicated statistically significant differences in metric values, both among sites and above versus below Kansas City. Analysis of rock basket data for sites upstream vs. downstream of Kansas City indicated weakly significant differences in means ($P < 0.10$) for three of the metrics (ANOVA, Table II); % DT and EPT/C were lower and HBI was higher downstream. For among-site analysis of rock basket samples, 12 of 15 metrics showed highly significant differences in means at one or more sites (ANOVA, $P < 0.001$). Of these metrics, seven indicated that one or two of the sites below Kansas City were significantly different than the other sites. At Lexington, the metrics EPT, % EPT, % EPOT, % DT, DEN, % LRRT, and EPT/C were all lower as compared to the other sites. The lower percentage of EPOT individuals colonizing rock baskets at Lexington is

TABLE I

Similarity and % efficiency matrix for macroinvertebrate species collected by 3 sampling methods used to evaluate biological condition in the lower Missouri River. The total of 132 species known to occur in this river reach include an additional 14 species collected by qualitative methods (including light-trapped specimens) from 1992–1998. Unique taxa are those collected with only one method. Percent efficiency was calculated by dividing the mean number of taxa collected with the method, by the total number of taxa collected with the method, across all sites. Percent Similarity was calculated by dividing the total number of taxa common to both methods by the combined total collected by both methods

Similarity comparison	Macroinvertebrate sampling method		
	Rock basket (n = 29)	Kicknet (n = 12)	Petite ponar (n = 29)
Total taxa collected by method	52	69	69
Unique taxa collected by method	15	19	22
Mean # of taxa collected by method	17.7	19.2	14.2
(standard deviation in parentheses)	(7.1)	(3.4)	(3.9)
Mean % efficiency of method	34.1	27.8	20.6
(range in parentheses)	(15.3–69.2)	(15.9–34.7)	(8.6–36.2)
% Of total known taxa (N = 132)	39.4	52.2	52.2
% Of taxa known within habitat	66.6 (rock)	88.4 (rock)	97.1 (depositional)
% Taxa similarity with Rock Basket	–	75.3	73.1
% Taxa similarity with Kicknet	75.3	–	39.1
% Taxa similarity with Ponar	73.1	39.1	–

partially due to higher numbers (and relative abundance) of oligochaeta at this site (Figure 3). Only four stoneflies were found among the five rock baskets deployed at Lexington, yet stoneflies were among the numerically dominant organisms at the other sites. Lexington and Glasgow also had significantly higher HBI values (Table II), suggesting a higher degree of organic enrichment. Analysis of ponar data upstream vs. downstream of Kansas City yielded four metrics with significantly different ($P < 0.05$) means: % MA, % EPH, and SDI were all lower and % DT was higher below Kansas City (Table III). Among-site analysis of ponar samples indicated that the first three of these metrics were significantly lower at Glasgow. Glasgow also had the lowest % CH in ponar samples (3.8%), whereas Nebraska City had the highest (30.9%, Table III).

Relative biological condition scores for all sites using different combinations of metrics are given in Table IV. For rock basket data, the 4 metric score was slightly lower directly below Kansas City at Lexington, and at Nebraska City. The

TABLE II

List of arithmetic means (range in parentheses) and statistical significance (P values from the two analyses with F values in brackets) for macroinvertebrate metrics determined from colonization of rock basket artificial substrates in the lower Missouri River. Means or groups of means with the same superscript letter(s) across sites are not significantly different as identified using Tukey's method of pairwise comparisons (controls for experiment-wise error using a $P < 0.05$ significance level). + = used in scoring (Table IV)

Metric	Sampling site							
	Among sites (N = 29)	Above vs. below KC (N = 6)	Nebraska City	St. Joseph	Parkville	Lexington	Glasgow	Hermann
+ Taxa richness	0.0039	0.689	10.6^{a}	25.4^{bc}	20.4^{bc}	14.6^{abc}	16.4^{abc}	19.5^{abc}
	[4.77]	[0.185]	(9–14)	(19–36)	(13–32)	(8–19)	(12–24)	(13–26)
+ EPT richness	0.0005	0.490	8.0^{ab}	14.6^{bc}	12.4^{bc}	6.4^{a}	8.6^{abc}	13.5^{bc}
	[6.89]	[0.575]	(7–9)	(11–18)	(9–18)	(4–8)	(5–12)	(9–20)
Percent (%) EPT	<0.0001	0.2287	96.0^{b}	91.4^{b}	95.2^{b}	69.1^{a}	91.4^{b}	90.5^{b}
	[39.96]	[2.016]	(92.5–98.1)	(87.7–96.2)	(93.0–97.4)	(57.3–74.6)	(89.0–94.3)	(86.7–93.4)
EPOT richness	0.002	0.542	8.2^{ab}	15.0^{b}	13.0^{b}	6.8^{a}	9.6^{ab}	14.0^{b}
	[5.37]	[0.442]	(8–9)	(12–19)	(9–19)	(4–8)	(5–15)	(9–21)
+ Percent (%)	<0.0001	0.231	96.1^{b}	91.5^{b}	95.5^{b}	69.6^{a}	91.7^{b}	90.8^{b}
EPOT	[42.95]	[1.991]	(92.5–98.1)	(87.7–96.2)	(93.1–97.4)	(59.8–74.6)	(89.0–94.7)	(86.9–93.4)
Chironomidae	<0.0001	0.830	0.6^{a}	6.2^{b}	5.8^{b}	5.2^{b}	4.4^{b}	4.2^{b}
Richness	[13.03]	[0.052]	(0–2)	(3–11)	(4–10)	(3–8)	(4–6)	(3–6)
+ Percent (%)	<0.0001	0.136	0.7^{a}	5.16^{bc}	3.8^{bc}	18.6^{cd}	5.5^{bc}	7.3^{bcd}
Chironomidae	[23.09]	[3.45]	(0–2.8)	(2.3–6.9)	(2.1–6.1)	(11.1–23.3)	(3.7–8.3)	(6.0–12.2)

TABLE II
(continued)

Metric	Sampling site							
	Among sites (N = 29)	Above vs. below KC (N = 6)	Nebraska City	St. Joseph	Parkville	Lexington	Glasgow	Hermann
% Dominant	<0.0001	0.079	60.3^{c}	59.2^{c}	69.1^{c}	28.2^{a}	56.3^{bc}	43.1^{b}
Taxon	[21.02]	[5.44]	(47.4–68.2)	(45.6–73.3)	(64.9–71.3)	(26.4–32.4)	(51.4–61.4)	(33.3–60.4)
+ Hilsenhoff	<0.0001	0.075	4.90^{a}	4.96^{a}	4.96^{a}	5.20^{b}	5.19^{b}	4.98^{a}
(HBI)	[10.75]	[5.68]	(4.78–4.97)	(4.89–5.03)	(4.94–4.99)	(5.07–5.34)	(5.02–5.35	(4.82–5.13)
+ Shannon-	0.0003	0.169	1.35^{ab}	1.62^{ab}	1.29^{ab}	2.03^{b}	1.46^{ab}	1.82^{ab}
Wiener Index	[7.49]	[2.80]	(1.13–1.55)	(1.19–2.10)	(1.24–1.42)	(1.78–2.36)	(1.26–1.66)	(1.46–2.09)
+ Density	0.0001	0.373	269^{a}	1802^{b}	1713^{c}	237^{a}	942^{bc}	790^{abc}
	[8.99]	[1.004]	(116–404)	(784–3425)	(392–4320)	(110–437)	(601–979)	(497–1347)
% Model affinity	0.0004	0.563	48.2^{a}	55.7^{ab}	47.0^{a}	65.5^{b}	46.6^{a}	50.8^{a}
	[7.11]	[0.395]	(39.7–59.6)	(48.8–64.9)	(43.3–54.2)	(59.9–70.0)	(42.2–52.7)	(40.5–62.2)
+ % Large river	<0.0001	0.154	74.1^{c}	69.5^{bc}	76.6^{c}	48.6^{a}	72.7^{bc}	60.8^{ab}
restricted taxa	[14.24]	[3.07]	(66.7–86.0)	(61.1–78.6)	(70.2–79.2)	(44.1–56.8)	(66.4–76.3)	(51.2–68.5)
+ Scraper/filterer	0.072	0.921	0.12^{ab}	0.10^{ab}	0.08^{ab}	0.18^{b}	0.05^{a}	0.07^{ab}
ratio	[2.36]	[0.011]	(0.02–0.23)	(0.07–0.13)	(0.03–0.12)	(0.10–0.27)	(0.02–0.08)	(0.01–0.19)
+ EPT/chironomid	<0.0001	0.079	82.0^{c}	22.1^{bc}	29.9^{bc}	4.0^{a}	18.0^{b}	19.5^{b}
ratio	[14.39]	[5.49]	(33–131)	(11.9–42.0)	(15.3–45.9)	(2.6–6.1)	(10.7–25.0)	(7.1–15.5)

TABLE III

List of arithmetic means (range in parentheses) and statistical significance (P values from the two analyses with F values in brackets) for benthic macroinvertebrate metrics determined from Petite Ponar sampling in depositional habitats of the Lower Missouri River. Means or groups of means with the same superscript letter(s) across sites are not significantly different as identified using Tukey's method of pairwise comparisons (controls for experiment-wise error using a $P < 0.05$ significance level). + = used in scoring (Table IV)

Metric	Sampling site							
	Among sites (N = 29)	Above vs. below KC (N = 6)	Nebraska City	St. Joseph	Parkville	Lexington	Glasgow	Hermann
+ Taxa richness	0.154 [1.791]	0.373 [1.002]	18.2^{a} (12–25)	14.8^{a} (13–16)	12.2^{a} (10–15)	14.4^{a} (12–15)	12.6^{a} (6–20)	12.8^{a} (8–13)
Chironomidae richness	0.036 [2.876]	0.325 [1.253]	8.8^{b} (6–12)	5.0^{ab} (2–7)	5.5^{ab} (3–10)	3.4^{a} (2–6)	5.6^{ab} (2–9)	5.6^{ab} (4–8)
+ Percent (%) Chironomidae	<0.0001 [19.86]	0.1314 [3.58]	30.9^{d} (17-4-41.4)	8.0^{b} (3.8–13.0)	17.1^{cd} (13.6–24.4)	9.4^{bc} (8.6–10.6)	3.8^{a} (2.7–7.0)	7.8^{ab} (4.0–10.7)
% Dominant Taxon	<0.0001 [13.31]	0.0455 [8.23]	52.04^{a} (39.0–62.9)	69.6^{ab} (57.6–81.7)	58.3^{ab} (23.3–75.9)	72.6^{ab} (59.6–83.3)	91.1^{c} (86.6–94.5)	80.3^{bc} (73.2–88.6)
+ Shannon-Wiener Index	<0.0001 [13.75]	0.0535 [7.347]	1.69^{c} (1.38–2.27)	1.28^{bc} (0.93–1.61)	1.51^{bc} (1.05–2.21)	1.18^{bc} (0.85–1.48)	0.49^{a} (0.29–0.68)	0.91^{b} (0.55–1.28)
+ Density	0.0002 [7.89]	0.1842 [2.57]	6070^{bc} (3013–9515)	4830^{ab} (3272–7663)	2421^{a} (732–3358)	5287^{bc} (4047–6544)	10393^{c} (6329–17178)	6621^{bc} (5855–7448)
% Model affinity	0.0022 [5.314]	0.0644 [6.418]	51.9^{bc} (48–57)	41.8^{b} (31.0–50.4)	51.2^{bc} (39.1–70.0)	41.1^{b} (31.7–53.8)	23.8^{a} (20.5–28.4)	34.4^{ab} (26.4–40.8)
% Large river Restricted taxa	0.0073 [4.205]	0.3869 [0.941]	3.86^{ab} (0.9–7.0)	4.98^{ab} (0.9–5.6)	11.0^{bc} (1.7–33.3)	8.08^{bc} (2.8–17.0)	0.84^{a} (0.0–2.0)	1.64^{ab} (0.7–3.0)
+ Percent (%) Ephemeroptera	<0.0001 [13.53]	0.0951 [4.738]	2.22^{b} (1.5–2.9)	2.96^{b} (1.8–4.7)	5.52^{bc} (3.6–6.8)	1.24^{b} (0.7–2.6)	0.34^{a} (0.0–0.8)	5.86^{bc} (4.7–7.4)

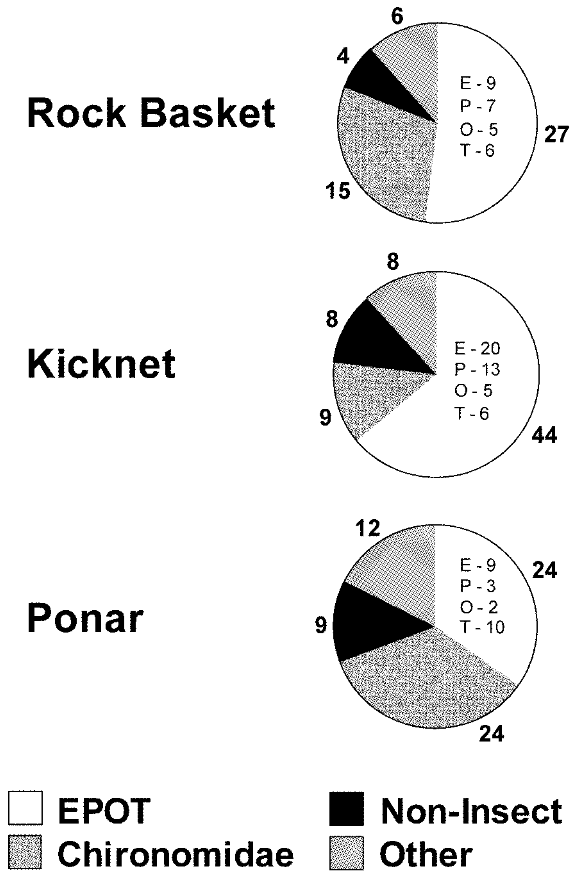

Figure 2. Number of aquatic macroinvertebrate taxa and relative contribution of taxonomic groups collected by 3 sampling methods in the lower Missouri River. Basket artificial substrates and kicknet methods were used in rock habitats, while the Petite Ponar was used in depositional habitats behind wing dikes. E = Ephemeroptera, P = Plecoptera, O = Odonata, T = Trichoptera.

10 metric score, which includes other potentially valuable large river metrics such as % LRRT and % CH, indicated substantially lower scores at Lexington. The 5 metric score for depositional habitats indicated a slightly lower score for Glasgow (Table IV). The kicknet data did not show any discernable longitudinal pattern between sites for individual metrics, but the 4 metric score was substantially lower at Lexington. The four commonly used metrics calculated for both kicknet and rock basket data showed consistent differences; for kicknet samples, HBI values were lower and SDI values were higher than for rock baskets. However, both October and December kicknet samples were comparable to each other (Table IV), and individual methods were consistent within a site. Of the seven metrics for which Lexington differed significantly from the others, % EPT and % EPOT were signi-

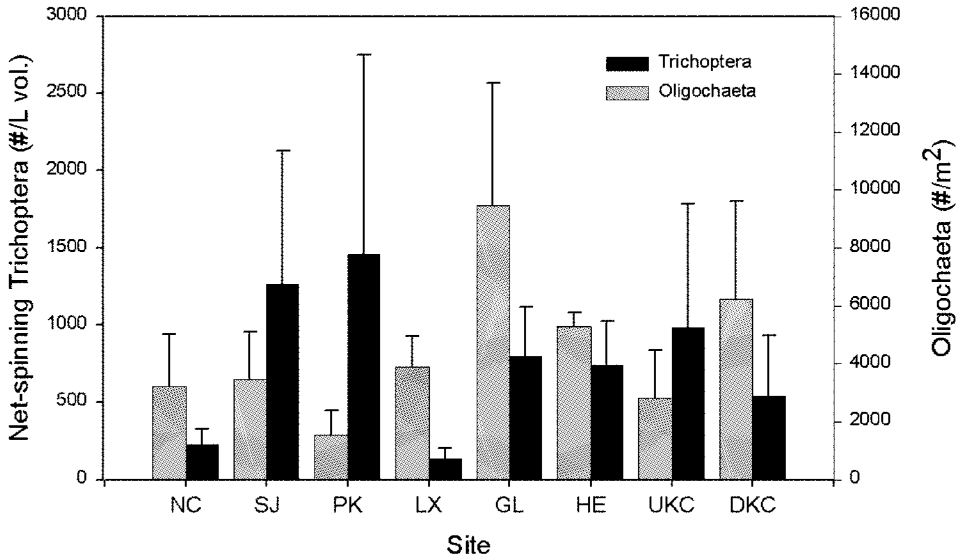

Figure 3. Mean density of net-spinning Trichoptera in rock baskets deployed on rock revetments and mean density of Oligochaeta in depositional habitats behind wing dikes sampled with a Petite Ponar (plus one standard deviation), for 6 sites in the channelized lower Missouri River. NC = Nebraska City, SJ = St. Joseph, PK = Parkville, LX = Lexington, GL = Glasgow, HE = Hermann, UKC = sites above Kansas City, and BKC = sites below Kansas City.

ficantly lower because of the relatively higher numbers of oligochaeta and lower numbers of net-spinning Trichoptera at this site (Table II, Figure 3).

5. Discussion

Our research is the first attempt to determine relative biological condition in the lower Missouri River using a multimetric community-level assessment and providing support for the interpretations with statistical analysis of the individual metrics. Assessments that use additive combinations of community attributes to measure and describe whether resident biota of aquatic systems within a region, stream reach or watershed are similar to one another, have become the basis for determination of biological status in flowing waters (Gerritsen, 1995). Multimetric indices such as the Index of Biotic Integrity (IBI, Karr *et al.*, 1986) and the Invertebrate Community Index (ICI, Karr and Kerans, 1991) are currently well-established as measures of biological condition (Lenat, 1993; Kerans and Karr, 1994), and have succeeded because they use sound ecological theory to integrate the cumulative effects of all human activities, including chemical contamination (Fore *et al.*, 1994). We utilized multiple-attributes in our assessment because this method of interpretation has received widespread usage in wadeable streams, and recent literature has identified the potential of modifying these indices (Simon and Emery, 1995; Lyons

TABLE IV

Total combined scores (mean and range) of relative biological condition for 6 sites and 2 habitats on the lower Missouri River using different combinations of benthic macroinvertebrate metrics. Individual values for the 4 standard metrics (TRR, EPT, SDI, and HBI) determined from the kicknet method are also given. Combined scores were determined by adding individual scores for each of the metrics, using the following criteria: above 50th percentile = score of 5, between 25th and 50th percentile = score of 3, and lower quartile = score of 1. Metrics included in scoring are marked in Tables II—III and explained in the methods section of the text

Combination of metrics		Sites above Kansas City			Sites below Kansas City		
		Nebraska City	St. Joseph	Parkville	Lexington	Glasgow	Hermann
10 Metrics (Rock) Rock Basket		36.4 (32–42)	38.8 (36–42)	44.4 (40–48)	18.8 (14–22)	40.8 (34–48)	37.5 (34–42)
5 Metrics (Depositional) Petite Ponar		21.4 (19–23)	19.4 (17–23)	18.5 (15–21)	18.6 (15–23)	11.4 (9–13)	19.4 (13–25)
4 Standard Metrics (Rock) Rock Basket		12.8 (10–16)	18.0 (16–20)	16.0 (14–18)	12.0 (8–14)	14.8 (8–20)	18.0 (14–20)
4 Standard Metrics (Rock) Kicknet		23.6 (20–28)	20.8 (16–26)	28.0 (24–30)	6.8 (6–8)	26.0 (24–28)	20.8 (14–26)
Kicknet Taxa richness	10/96	11	20	24	16	18	23
	12/96	20	18	19	20	20	22
Kicknet EPT Taxa richness	10/96	10	13	15	10	13	14
	12/96	17	12	15	13	13	17
Kicknet Shannon Diversity	10/96	1.81	2.42	2.45	2.33	2.37	2.59
	12/96	2.44	2.37	2.43	2.40	2.66	2.47
Kicknet HBI	10/96	4.36	3.88	4.73	3.94	5.02	4.33
	12/96	4.32	3.53	3.63	4.02	4.11	3.58

et al., 1996) or developing new ones (Kerans and Karr, 1994; Barbour *et al.*, 1996; Barbour, 1997) for large rivers. Although reference or best-attainable conditions have never been defined for the Missouri River, alternative interpretation methods have been suggested for comparing sites in cases where no reference condition can be determined. We used the cumulative data set for each attribute to define metric expectations and provide percentiles for relative scoring of sites as done previously in both larger rivers (Simon and Emery, 1995) and smaller streams (Lenat, 1993; Barbour *et al*., 1995).

We selected both commonly used metrics and those that represent modifications for potential application in large rivers. The information provided here is only an initial step towards full-scale evaluation and validation of metrics that has been done in other studies. We observed patterns and statistically significant differences among sites for many metrics that have been selected by others as core attributes for biotic assessments and state monitoring programs (TTR, EPT, CHR, and % DT, Barbour *et al*., 1996). These metrics are typically among the least variable (HBI, TTR, % EPT, % DT, and EPT/C, Hannaford and Resh, 1995; Poulton *et al*., 1995), are reliable predictors of water quality (TTR, EPT, Lenat, 1988; Barbour *et al*., 1992; Yoder and Rankin, 1995), or both. Even though density of Oligochaeta (# m^{-2}) in depositional habitats and net-spinning Trichoptera (# L^{-1} interstitial volume) in rock habitats were not analyzed statistically, these 'a posteriori' attributes were depicted graphically in this paper (Figure 3) because they helped interpret patterns we observed in some of the other metrics. The selection of metrics to be used for relative scoring of sites was based primarily on the criteria listed earlier, although some were not used because they had a high similarity with other metrics (% EPT and EPOT) or the validity of their use for large rivers is uncertain (% DT and % MA). Chironomid richness (CHR) was not used because it has been known to provide a somewhat bimodal response to perturbations (Lenat, 1983). Even though % DT and % MA showed among-site differences, they were not used for scoring because a reduction in net-spinning Trichoptera (primarily the dominant taxa *Hydropsyche orris*) directly below Kansas City at Lexington (Figure 3) resulted in a lower % DT and a higher % MA as compared to other sites, both of which are opposite of the response that might be expected due to perturbation (Novak and Bode, 1992). Other metrics such as SFR suggested that the reduction in net-spinning Trichoptera at Lexington is possibly due to other pollution-related factors such as bound toxicants (see Plafkin *et al*,. 1989; Camargo, 1992). Similarly, SDI was higher in rock habitats at Lexington, but this metric was retained for site scoring because of its widespread usage in the literature and to enhance data comparisons with wadeable stream monitoring programs. Although many of the attributes we selected should theoretically be useful for large rivers, their suitability for larger systems will remain uncertain until they can be validated. Tolerances to different pollutants for some large river taxa are unknown, and for metrics such as HBI, tolerance values for large river taxa had to be assigned based on the most closely related species. Taxa such as *Attaneuria ruralis* (Plecoptera: Perlidae) and *Raptoheptagenia cruenata* (Ephemeroptera: Heptageniidae) are examples of lower Missouri River species that are restricted to large rivers in the central U.S. and for which tolerance values are not listed by Lenat (1988) or Hilsenhoff (1987). Fore *et al*. (1994) indicated that impaired sites should have had higher variability due to loss of ability in the community to buffer changes. Nevertheless, taxa not listed in these publications did not make up a significant portion of the samples, and HBI variability was low as has been found in other studies (Poulton *et al.*, 1995).

5.1. Rock habitats

The heterogeneous nature of the crushed limestone substrate that makes up the dikes and revetments on the Missouri River includes a large range of particle sizes from finer gravel to large boulders (5–50 cm and occasionally up to 1.0 m, see Atchison *et al.*, 1986; Sandheinrich and Atchison, 1986) and probably provides turbulent conditions with a much larger range of velocities and microhabitats than that of unstable fine material. Rock is the most stable and heterogeneous substrate in flowing waters of the main channel, and due to the removal and reduction of large woody debris during channelization, rock habitats may represent the largest in total area along the river margins. In our study, rock baskets were meant to simulate the community composition in these habitats. The dominant taxa in rock basket samples at all sites were the net-spinning caddisflies *Hydropsyche orris, H. scalaris*, and *Potamyia flava*, and the mayfly *Stenonema integrum* and many small instars of the stonefly species *Isoperla bilineata*. We added the Odonata to the EPT index (EPOT) because of their large size, potential importance as fish food items, taxonomic richness in the river (N = eight species), and the frequency of collection in both depositional and rock substrates. Artificial rock substrates have been shown to provide representative macroinvertebrate samples from rock habitats (Slack *et al.*, 1986), even though Mason *et al.* (1973), and Carter *et al.* (1982), both reported that dominance by a few taxa is a common occurrence in artificial substrates deployed in large rivers. In our study, these samplers yielded a mean efficiency estimate of 34.1% (Table III) and collected a larger number of taxa than that reported by other studies (Slack *et al.*, 1986).

The density estimates (#/m or #/L) we generated from rock basket samples could not be directly compared with some previous studies because others either did not sample in areas with current flow (Carter *et al.*, 1982), the volume of rock used in the baskets was not measured (Munger *et al.*, 1974), or a rock removal method that did not sample interstitial invertebrates was used (Atchison *et al.*, 1986). Mathis *et al.* (1982) implanted rock baskets on top of revetments in the lower Mississippi River and reported densities of over 100 000 m^{-2}, which falls in the range of the mean estimates in our study (range = 16,116–122,536 m^{-2} or 237–1802 L^{-1} interstitial volume). Carter *et al.* (1982) did not deploy rock baskets in flowing water habitats but rather behind the dikes in areas with lower velocity. This factor, along with a summer sampling season, may explain why their samples had lower numbers of taxa, higher dominance of oligochaeta, and lacked a significant stonefly (Plecoptera) fauna. We lost only one artificial substrate sampler during this study; however, it is not known whether our method of deploying rock baskets would allow successful retrieval during spring or early summer when flood pulses are more common. Our results suggest that rock baskets deployed during late fall seasons can be easily retrieved and yield representative samples from rock habitats adjacent to the main channel without being covered with silt, a problem noted in other large river studies (Munger *et al*., 1974; Mathis *et al*., 1982).

We used a kicknet concurrent with deployment of artificial substrates to provide a more complete list of taxa inhabiting rock habitats, identify any taxa lost during retrieval of rock baskets, and generate data that could be compared with other methods and studies. This method was included because of the accessibility of rock rubble substrate along revetments during stable or declining water levels, and because several variations of this method have received widespread use in wadeable stream studies. Fluctuating water levels in depths of 1 m or less which occur often in regulated rivers, may prevent use of this method for long-term monitoring studies that require a well-defined index period each year. Kicknet samples had 75.3% taxonomic similarity with rock baskets, but these two methods did not reveal the same longitudinal patterns. Even though we attempted to reduce any possible bias towards larger individuals during field picking, only four metrics were calculated from these samples because of their qualitative nature. The 4 metric combination indicated a lower score at Lexington, and based on our scoring criteria this equates to a higher probability that a sample taken from rock habitats at this site would yield values for these four metrics that would fall below the 50th percentile. These four standard metrics are used in rapid assessment techniques (Plafkin *et al*., 1989) and in stream monitoring programs by many state agencies, including Missouri (MDNR, unpublished). Both combined scores and several individual metrics derived from rock basket data indicated that the Lexington site may have a more impaired benthic fauna than the other sites, but kicknet data revealed this pattern only with the combined relative score of the four metrics. Because we simultaneously attempted to obtain the highest taxa richness in a 45–60 min period with the kicknet method, our estimates for the other three metrics may be biased.

Both SDI and HBI were consistently different between the two methods used in rock habitats, and the kicknet method yielded taxa richness estimates that were nearly always higher than that of rock baskets. However, no distinct pattern in taxa richness among sites was evident with either method. Kicknet samples were dominated by the mayflies *Isonychia* sp., *Baetis* sp., and *S. integrum*, the caddisfly *P. flava*, and the stoneflies *Acroneuria abnormis* and *Isoperla bilineata*. Kicknet samples had higher mean and total numbers of taxa and higher numbers of unique taxa than rock basket samples deployed in the same habitat (Table I). Highly mobile swimming mayflies *Isonychia* sp. and *Baetis* sp. that were common in the kicknet samples, were nearly absent from rock baskets; it is possible that a significant number of individuals belonging to these taxa escaped from rock baskets upon retrieval. Although these two methods were used within the same habitat, the differences in relative abundance and taxa richness we observed may be due to spatial differences in communities between near-shore areas sampled with the kicknet (1 m depths) and that of deeper water a further distance from shore where the rock baskets were deployed (2–4 m).

5.2. Depositional habitats

In this study, ponar sampling from depositional habitats yielded a larger number of taxa (69) and efficiency estimate (20.6%) than in some lentic studies (Tsui and Breedlove, 1978) and in other large river macroinvertebrate research (Carter *et al.*, 1982; Slack *et al.*, 1986; Hornbach *et al.*, 1989). Chironomids and Oligochaeta were numerically dominant and overall composition was similar to that reported for mud substrates in abandoned channels of the lower Missouri River (Atchison *et al.*, 1986; Beckett and Pennington, 1986) and backwater lakes on the upper Mississippi (Hornbach *et al.*, 1989). However, previous studies have not reported burrowing mayflies (Ephemeroptera: Ephemeridae, *Hexagenia* spp., *Pentagenia vittigera*) as a significant portion of the fauna in the lower Missouri system, even though a recent study documented large densities of stranded individuals behind dikes after drops in water levels (Braaten and Guy, 1997). These mayflies require specific sediment texture with sufficient oxygen and organic matter content for survival (Ericksen, 1968; Wright and Mattice, 1981; Elstad, 1986). We suggest that they have not regularly been reported as a significant portion of the fauna in other Missouri River studies because their distribution behind dikes is patchy. Barnum and Bachmann (1988) documented that coarse-textured sediments in flowing areas behind dikes were nearly void of benthic organisms. At all sites, we commonly collected these mayflies from the stable zone of dark colored mud substrate with visible organic matter and no current flow. This substrate may have been more suitable for mayflies than that sampled in other Missouri River studies (Carter *et al.*, 1982; Atchison *et al.*, 1986; Beckett and Pennington, 1986). Therefore, % EPH may be an appropriate metric for large river assessments if this zone can be specifically targeted during ponar sampling because burrowing mayflies can make up a significant portion of the biomass in depositional areas of large rivers (Hornbach *et al.*, 1989) and are known to be positively correlated with water and sediment quality (Fremling, 1964; Schloesser *et al.*, 1991).

The range of our total density estimates (#/m) from backwater areas (2421–10,393) was comparable to that reported from abandoned channels (12,624, Atchison *et al.*, 1986; 7243, Beckett and Pennington, 1986) and the depositional areas behind dikes (maximum of 4410, Carter *et al.*, 1982) in the middle Missouri River, but were much higher than those reported from silt areas in the same river reaches (79.5, Johnson *et al.*, 1974). We also documented the presence of incidental taxa that were unexpected in backwaters. Even though no visible flow was present in this habitat, the ponar regularly collected taxa that are normally associated with flow and stony substrate, including net-spinning caddisflies, stoneflies, and heptageniid mayflies. It is possible that the large numbers of individuals of these taxa that are in the drift (Modde and Schmulbach, 1973; Carter *et al.*, 1982) are inadvertently transported to slack water habitats behind wing dikes and may not be able to readily move back into adjacent flowing areas after they settle out of the water column. This phenomenon was observed at every site, and may be related to channel flow patterns and formation of depositional zones behind dikes.

5.3. Assessment of Biological Condition

Our study design was based on the assumption that upstream/downstream site comparisons relative to a metropolitan area would offer the best possibility for detecting cumulative effects of all potential anthropogenic influences on large river macroinvertebrate communities. We also relied on extant water quality information from previous studies suggesting that the Kansas City metropolitan area represents a significant cumulative source of contaminants and urban-related impacts in the lower Missouri River (Ballentine *et al.*, 1970; Ford, 1980; Schmulbach *et al.*, 1992; Petty *et al.*, 1995, 1998). This is further supported by unpublished information on contaminants and unregulated chemical constituents originating from tributaries in the Kansas City metro area (Dale Blevins and Don Wilkison, USGS, pers. comm.), and by recent documentation of elevated levels of OC's, urban-related PAH's, and xenoestrogenic compounds in SPMD's deployed below Kansas City at the Napoleon-Lexington reach (Petty *et al.*, 1995, 1998).

Yoder and Rankin (1995) recognized nine types of pollution and disturbance sources that result in discernable response patterns (i.e. biological response signatures) in fish and macroinvertebrate communities, all of which may affect these communities when urban areas are geographically situated on large, regulated rivers. Several of the urban-related source types they listed have been suspected as causes of aquatic community changes in other large river macroinvertebrate studies, and include combined sewer outflows (Seager and Abrams, 1990), industrial discharges (Camargo, 1992; Battegazzore *et al.*, 1992), toxics and complex mixtures (Hellawell, 1986; Ramade, 1989), and oxygen-demanding substances associated with urban stormwater runoff (Pratt and Coler, 1979; Willemsen *et al.*, 1990). These studies have also demonstrated specific responses in macroinvertebrate community attributes, many of which were observed to some degree in the results of our study. Rock habitats at Lexington, directly below Kansas City, had lower mean richness and percent of clean-water EPT organisms (EPT and % EPT), EPT/C, DEN, and % LRRT. Only four stoneflies were found in the five replicate samples from this site. Depositional habitats further downstream at Glasgow had significantly lower SDI and % EPH, and had lower % CH, % LRRT, and % MA. Only five burrowing mayflies were found in the replicate samples taken from depositional habitats at this site. Rock habitats at both Lexington and Glasgow had significantly higher HBI values, indicating a higher level of organic enrichment than the other sites. A higher dominance of Oligochaeta in rock habitats at Lexington (Figure 3) and a community comprising over 90% Oligochaeta in the depositional habitats at Glasgow, also indicate organic pollution (Burt *et al.*, 1991; Lenat, 1993). Oligochaete worms as a group are also tolerant of toxics (Yoder and Rankin, 1995). A noticeably lower number of filtering Trichoptera that we observed at Lexington (Figure 3), has been documented in large rivers below urban areas in other studies, and the presence of toxicants from industrial effluents has been the suspected cause of this response (Plafkin *et al.*, 1989; Camargo, 1992; Yoder and Rankin, 1995). All of these indicators, including the lower relative condition scores at Lexington and in the depositional habitats at Glasgow, suggest that some impacts resulting from the Kansas City area were detected. In sampling with artificial substrates in the River Po in Italy, Battegazzore *et al.* (1992) found that effects of pollutants from industrial point sources were relatively short-lived from a longitudinal standpoint. Our study also suggests this pattern because relative condition scores and values for most of the metrics at Glasgow and Hermann were more similar to sites upstream of Kansas City than those at Lexington.

Our data do not suggest whether Chironomidae taxa richness and relative abundance would be important indicators in large rivers or for particular habitats, because Nebraska City had the highest CHR and % CH in depositional zones, and the lowest values for these parameters in rock habitats. Lenat (1983) reported that reduced chironomid taxa richness can be associated with either severe water pollution or clean water and relatively undisturbed habitats, and that often the highest chironomid taxa richness can be found where moderate levels of pollution prevail. Dominant chironomid taxa such as *Cricotopus, Dicrotendipes, Ablabesmyia* and certain species of *Polypedilum* are known to be associated with agricultural and sewage inputs (Rae, 1989). These taxa were among the dominant chironomid genera at all of our sites, in both depositional and rock habitats. The lower amount of fine interstitial substrate material we observed in the rock habitats at Nebraska City suggested that this revetment had been more recently constructed than those at the other sites. Low numbers of chironomids and the second lowest density estimates at this site suggest that rock revetments, as colonization habitat for macroinvertebrates, may improve with age when they become more embedded with finer material. It is also possible that the Omaha, NE metropolitan area, about 89 km upstream, may be influencing the community composition at this site. However, our condition scores for Nebraska City were relatively similar among all other sites except for rock habitat at Lexington and the depositional habitat at Glasgow.

6. Conclusions and Recommendations

Perhaps the most important conclusion reached in this research is that a multimetric bioassessment approach utilizing upstream-downstream comparisons can be used as an initial screening tool for determining relative biological condition and measuring cumulative effects of urban-related anthropogenic stressors in large rivers. Our study is the first to apply this approach to the lower Missouri River. Previous macroinvertebrate studies on the channelized reaches have attributed low estimates of productivity, diversity, and density with high current velocities and substrate instability (Berner, 1951; Morris *et al.*, 1968; Mestl and Hesse, 1993). Our density and taxa richness estimates are comparable to that reported for other physically altered large rivers including the lower Mississippi (Mathis *et al.*, 1982), the Illinois (Richardson, 1921), and the Ohio (Mason *et al.*, 1971). Sandheinrich and Atchison (1986) noted the importance of rock structures in providing habitat for macroinvertebrates that may not otherwise be available; our study confirms that rock revetments and wing dikes represent a diverse and productive habitat in the lower Missouri river even though most of this substrate has been artificially added. Future large river monitoring programs should take advantage of the availability of this stable substrate for acquiring representative, quantitative macroinvertebrate data in situations where logistical problems in sampling are much greater than in wadeable streams. Selection of this habitat for pollution assessment in large rivers is parallel to the current usage of heterogeneous substrate in riffle areas of smaller streams. In addition to flowing water areas adjacent to the main channel, we recommend a multi-habitat approach with inclusion of soft-bottom mud substrates in depositional zones, because this habitat contributes unique taxa and additional contaminant-related information that may not be evident from sampling of rock substrates alone.

State agencies are responsible for evaluating the biological condition of flowing waters and the ability of these waters to support aquatic life. The development of biocriteria (Southerland and Stribling, 1995) and determination of use attainment status for large rivers will require further understanding of cause-effect relationships and application of an upstream/downstream study design related to all influences from urban areas, major tributaries, point sources, and biogeographic factors. This is a formidable task in large, interjurisdictional rivers when there is often a shortage of resources and lack of agreement on approaches. Alternative assessment methods and site comparisons that have been suggested in cases where reference conditions cannot be determined (Simon and Emery, 1995; Barbour *et al.*, 1995), such as those used in this study, may be the best available framework for measuring relative biological condition in these systems. Large river modifications of aquatic community indices such as the ICI (Karr and Kerans, 1991), and subsequent identification of least-impaired or best-attainable community characteristics for specific reaches, are in its early stages for the Ohio River (ORSANCO, E. Emery, pers. comm.). To further evaluate metrics and to aid in the development of large river biological criteria, we recommend a similar approach for the Missouri River system. The index period and methodologies we used in this study appear replicable and our initial results could be fine-tuned by applying this framework over extended spatial and temporal scales at a larger number of sites. We also propose additional metrics that have potential as sensitive community indicators for evaluating relative condition attributable to anthropogenic influences in large rivers, such as those associated with species restricted to large systems (% LRRT) and those directly associated with potential contaminant loading in sediments (% EPH).

Since our study did not include a temporal component or comprehensive sampling of water and sediment quality, we did not attempt to validate individual metrics, identify reference conditions and reaches, establish impairment thresholds, or develop a specific macroinvertebrate community index for the lower Missouri River. However, this could be accomplished by acquiring data from a large number of sites and simultaneous analysis of water chemistry parameters, sediment contaminants, and body burdens in taxa such as burrowing mayflies. This information would have strengthened our interpretations, although including these measurements at a scale that would have been required to demonstrate cause-effect relationships was beyond the scope of this study. The numerous categories of urban impacts simultaneously affecting biota in large rivers and the intermittent nature of water quality-related stressors, often prevents the separation of individual effects on biota as has been attempted previously in environments where complex contaminant mixtures are a problem (Wildhaber and Schmitt, 1998). For this reason, the value of macroinvertebrate communities as integrators of cumulative effects may be especially important for determining biological condition in large rivers such as the Missouri system, where native aquatic species are still in decline (Karr and Chu, 1999) and biota may still be adjusting to the broad scale physical alterations that have occurred (Hesse and Sheets, 1993).

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