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Abstract Fish and benthic macroinvertebrate assemblages often provide insight on ecological conditions for guiding management actions. Unfortunately, land use and management legacies can constrain the structure of biotic communities such that they fail to reflect habitat quality. The purpose of this study was to describe patterns in fish and benthic macroinvertebrate assemblage structure, and evaluate relationships between biota and habitat characteristics in the Chariton River system of south-central Iowa, a system likely influenced by various potential management legacies (e.g., dams, chemical removal of fishes). We sampled fishes, benthic macroinvertebrates, and physical habitat from a total of 38 stream reaches in the Chariton River watershed during 2002–2005. Fish and benthic macroinvertebrate assemblages were dominated by generalist species tolerant of poor habitat quality; assemblages failed to show any apparent patterns with regard to stream size or longitudinal location within the watershed. Metrics used to summarize fish assemblages and populations [e.g., presence–absence, relative abundance, Index of Biotic Integrity for fish (IBI_F)] were not related to habitat characteristics, except that catch rates of piscivores were positively related to the depth and the amount of large wood. In contrast, family richness of benthic macroinvertebrates, richness of Ephemeroptera, Trichoptera, and Plecoptera taxa, and IBI values for benthic macroinvertebrates

(IBI_{BM}) were positively correlated with the amount of overhanging vegetation and inversely related to the percentage of fine substrate. A long history of habitat alteration by row-crop agriculture and management legacies associated with reservoir construction has likely resulted in a fish assemblage dominated by tolerant species. Intolerant and sensitive fish species have not recolonized streams due to downstream movement barriers (i.e., dams). In contrast, aquatic insect assemblages reflected aquatic habitat, particularly the amount of overhanging vegetation and fine sediment. This research illustrates the importance of using multiple taxa for biological assessments and the need to consider management legacies when investigating responses to management and conservation actions.

Keywords Biological assessment · Management legacy · Assemblage structure · Iowa · Agriculture

Introduction

Reductions in freshwater biodiversity have been well documented (Jelks et al. 2008; Strayer and Dudgeon 2010; Burkhead 2012), as have the mechanisms responsible for the declines (Ricciardi and Rasmussen 1999; Dudgeon et al. 2006). Threats to native biota include overexploitation and interactions with nonnative species, but alteration of habitat is generally recognized as the most widespread cause of reduced biodiversity and the greatest hindrance to conservation efforts (Richter et al. 1997; Dudgeon et al. 2006). Because stream systems and their biota are so closely integrated with their watersheds, understanding how biotic communities change in response to habitat degradation or restoration remains an important focus of basic and applied ecology.

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Anthropogenic effects on aquatic ecosystems are widespread and although many types of land use alteration [e.g., urbanization (Wang et al. 2001; Walters et al. 2009), timber harvest (Campbell and Doeg 1989), and mining (Mol and Ouboter 2004; Freund and Petty 2007)] have negatively affected aquatic systems, the effects of agriculture can be particularly deleterious to the structure and function of aquatic ecosystems (Zimmerman et al. 2003; Wagenhoff et al. 2012). Agriculture is often defined broadly, but the type and extent of agriculture is important with regard to effects on aquatic habitats and their biota. For instance, low-intensity pasture agriculture often has minimal effects on biotic communities in streams (Strayer et al. 2003), whereas row-crop agriculture usually results in dramatic alterations to habitat quality and stream biota (Jacquemin and Pyron 2011).

Few areas have been so highly influenced by agriculture as the Midwestern USA, particularly in Iowa where row-crop agriculture, pasture, and urban development dominate watershed land uses (Opsomer et al. 2003; Secchi et al. 2008). Prior to the European settlement, Iowa was primarily covered with tallgrass prairie, forests, and wetlands (Bultena et al. 1996). By the turn of the 21st century, nearly 80 % of Iowa's land area had been converted to cropland or pasture with consequent losses in terrestrial and aquatic biodiversity (Bultena et al. 1996; Gallant et al. 2011). Negative effects on biota have been especially dramatic in aquatic systems. For instance, the Clean Water Act Section 303(d) list of impaired waters contained 52 % of Iowa's waterbodies in 2012 (Iowa DNR 2013). The most frequent cause of impairment of rivers and streams was indicator bacteria (i.e., *E. coli*), closely followed by biological impairment (i.e., based on fish or benthic macroinvertebrate assemblages). The poor ecological condition of aquatic systems is further reflected in the status of aquatic organisms. Of the 297 species classified by the Iowa Department of Natural Resources (IDNR) as species of greatest conservation need (SGCN), 135 (48.4 %) are aquatic species (Zohrer 2005). Approximately 44 % of Iowa's native fishes are identified as SGCN, a proportion exceeded only by freshwater mussels (53 % of all species are SGCN).

The linkage between land use and the integrity of aquatic habitats and their biota is clear. This linkage serves as the foundation for biological assessment (bioassessment) where biotic communities are used to evaluate the ecological condition of an ecosystem (Davis and Simon 1995; Wilson and Xenopoulos 2008). Biological assessments have been conducted using a variety of techniques, such as ratios of observed to expected assemblages (Clarke et al. 1996; Carlisle et al. 2008) and multimetric indices like the Index of Biotic Integrity (IBI; Karr et al. 1986; Davis and Simon 1995; Roset et al. 2007). Bioassessments are widely used to determine current biotic conditions and their

response to environmental stressors, monitor aquatic ecosystem dynamics through space or time, and assist in the regulatory process. In aquatic systems, fishes and macroinvertebrate assemblages are commonly used in bioassessments (Davis and Simon 1995; Simon 1999). While concordance between fish and invertebrate responses to physical and chemical conditions has been observed (Kilgour and Barton 1999; Freund and Petty 2007), others have shown differential relationships with habitat (Lammert and Allan 1999; Carlisle et al. 2008; Walters et al. 2009). The use of at least two assemblages has been suggested as a means to provide a comprehensive assessment of ecological conditions (Yoder and Rankin 1995). Thus, there is a need to better understand how multiple taxa respond to ecological conditions.

The use of aquatic organisms in assessments and evaluations is based on the premise that their occurrence and (or) abundance reflects ecological conditions. Unfortunately, the ability of biota to provide information on habitat quality is often hindered by legacies created by prior land use or management actions. The concept of management legacies, including the "ghost of land use past," suggests that prior disturbances fundamentally alter ecosystems such that recovering systems are structurally and functionally different than similar undisturbed systems (Harding et al. 1998; Foster et al. 2003). Importantly, management legacies confound evaluations of management actions because expected responses may be unattainable. Streams and rivers in Iowa have experienced a long history of intensive agricultural land use, water development (i.e., dams), and fishery management activities (e.g., stocking, chemical removal of fishes); however, the importance of management legacies on the structure of aquatic communities remains unknown. Given the status of freshwater systems, understanding the response of aquatic organisms to ecological stressors is critical for guiding management and conservation efforts. The purpose of this study was to evaluate relations between fish and benthic macroinvertebrate assemblage structure and habitat characteristics in Iowa streams and to better understand the potential influence of management legacies on stream biota. We hypothesized that fish and benthic macroinvertebrate assemblages would exhibit concordant structure and that species intolerant of poor habitat quality would be absent from habitats with degraded habitat (e.g., high fine sediment, little instream cover).

Methods

Study Area

This study was conducted on streams in the Chariton River basin in south-central Iowa, upstream of Rathbun Reservoir

Fig. 1 Sampling reaches in the Chariton River basin of south-central Iowa, 2002–2005. Only the upper portion of Rathbun Reservoir is shown

(Fig. 1). Streams in the basin typically have well-developed pool-riffle-run sequences, have wetted widths of 2–5 m and average thalweg depths of 0.5–1 m during summer baseflow, a diverse mixture of substrates varying from fine silt to large boulders, and abundant instream cover in the form of wood and overhanging vegetation. The watershed has an area of about 1,480 km², and is characterized by rolling uplands and a diversity of riparian habitats varying from broad alluvial plains to narrow, forested riparian corridors (Mayhew 1977; Paragamian 1977). Prior to the European settlement, the landscape contained a diverse mixture of prairie, forest, and savanna (Burras and McLaughlin 2002). Intensive farming has occurred in the basin since the 1860s; approximately 50 % of the land area in the watershed is currently used for row-crop agriculture (primarily corn *Zea mays* and soybeans *Glycine max*) and another 10–20 % is used as pasture for livestock (Schilling and Libra 2000; Burras and McLaughlin 2002). The remainder of the basin has steep slopes with highly erodible clay soils that alternate between being either too wet or too dry for row crops (Opsomer et al. 2003). The watershed provides an average of 15 million L of water daily to over 50,000 residents in Iowa and Missouri; therefore, maintaining high water quality is a top priority for resource managers in the region. Over the last 40 years, a variety of best management practices (BMPs) have been implemented to reduce the effects of agriculture on aquatic systems, including rest-rotation systems for livestock, no-

till planting systems, terracing, grassed waterways, managed riparian buffers, and removing lands from production through the Conservation Reserve Program.

Fish and Benthic Macroinvertebrate Sampling

Fish and benthic macroinvertebrates were sampled from a total of 38 randomly selected sites during 2002–2005 in the Chariton River basin (Fig. 1) using standard IDNR protocols (Wilton 2004). Sampling reaches were 35 times the mean stream width, except that a minimum reach length of 175 m and maximum length of 700 m was established for all reaches. One upstream electrofishing pass was conducted during the summer using a backpack electrofishing unit (Smith-Root, Inc., Vancouver, WA). Voltage output was adjusted to maximize efficiency and reduce incidental mortality of fishes. At least three personnel used dip nets (6 mm delta mesh) to collect fishes. Captured fish were identified to species in the field and then released at the point of capture. Unidentified specimens were fixed in 10 % formalin and transferred to the laboratory for identification.

Benthic macroinvertebrates were sampled during the late summer to early fall using two sampling protocols: (1) standard-habitat (SH) and (2) multi-habitat (MH) samples (Wilton 2004). Standard-habitat samples were collected by deploying an array of Hester-Dendy artificial habitat samplers (8 plates, each plate was 102 mm long × 102 mm

wide $\times$ 6 mm thick separated with 25.4 mm spacers, 1,832.3-cm² surface area). Samplers were wired to metal posts driven into the streambed at transitions between riffle and pool habitats. If riffle habitats were not present, samplers were placed at the transition between run and pool habitats. All samplers were placed just off the stream bottom. Four replicate SH samples were collected at each site. After 30 days, samplers were transported back to the laboratory where they were disassembled and washed. The water and debris was poured across a 500- μ m sieve, and the material was preserved in 10 % formalin. In the laboratory, benthic macroinvertebrates were poured across a sieve that had a grid printed on the surface. When large numbers of organisms were encountered, grid cells were randomly chosen and all organisms were removed from those grids to obtain a minimum subsample of 100 organisms. Each sample was processed separately, but data were later aggregated to characterize the benthic macroinvertebrate assemblage (Wilton 2004).

The objective of the MH sample was to maximize the number of taxa collected. The MH sample was collected from the entire reach by handpicking benthic macroinvertebrates from all habitat types. Common types of substrates sampled as part of the MH sample included silt, sand, rock, detritus, woody debris, and aquatic vegetation. In addition to directly removing organisms by hand, D-frame nets were used to sample invertebrates by vigorously sweeping nets through the substrate (e.g., aquatic vegetation). A combined sampling time of 90 min was divided among three collectors who covered the entire sampling reach. All of the organisms were combined into one sample for the reach. Benthic macroinvertebrates were preserved in 10 % formalin and transported back to the laboratory where they were identified and counted. All invertebrates (i.e., SH and MH samples) were identified to the family level.

Habitat Sampling

Stream habitat assessment was conducted following the methods of Simonson et al. (1994a, b) and Wang et al. (1998). Wetted stream width, thalweg depth, substrate composition, instream cover for fish, bank conditions, and riparian vegetation characteristics were measured along twenty transects that were equally spaced within the reach. Substrate was classified using a modified-Wentworth scale and included silt, sand, gravel, cobble, boulder, and bedrock (Armantrout 1998). Substrate was later defined as bedrock, fine substrate (i.e., silt and sand), or rocky substrate (i.e., gravel, cobble, boulder). The dominant substrate was determined at 0.20, 0.40, 0.50, 0.60, and 0.80 of the transect width. Substrate softness at each point was estimated by scoring the substrate from 0 (hard) to 5 (soft). The surface area of instream cover that occurred in water at

least 0.30 m in depth was measured. Instream cover categories included boulders, overhanging vegetation, submerged macrophytes, emergent macrophytes, undercut banks, and woody debris (i.e., logs, root wads, log or branch complexes). Riparian vegetation was measured as the percentage of land covered by terrestrial vegetation within 10 m of the stream margin. Substrate composition and instream cover were summarized as the percentage of the reach (based on surface area) with each substrate or cover type. Mean width, depth, substrate softness, and percentage of land covered by riparian vegetation were also estimated for the reach. The coefficient of variation of thalweg depth was estimated to provide an indication of depth heterogeneity. Bank conditions were characterized as either vegetated or actively eroding and summarized as a percentage of the reach length. In addition to transect-based measurements of habitat characteristics, we also estimated the percentage of the reach that was channelized. Channelization age was estimated as old (channelized > 20 years prior to the sampling), intermediate (10–20 years prior to the sampling), or recent (within 10 years of sampling). Age was estimated by examining existing vegetation (e.g., presence and size of trees), discussions with landowners, and governmental agency reports. Sinuosity of each reach was calculated using a Geographic Information System as the ratio of a 1,000-m segment of stream (centered on the reach) divided by the straight line distance between the beginning and end of the segment.

Data Summarization and Analysis

Frequency of occurrence across reaches and percent composition of the sample (i.e., by reach) were calculated by species (fish) and family (invertebrates). Both the SH and MH samples were used to estimate frequency of occurrence for invertebrates, but only the SH samples were used to estimate percent composition of the sample at a reach. Catch-per-unit-effort (CPUE) was estimated as the number of fish per minute of electrofishing. Density of benthic macroinvertebrates was estimated from the SH sample as the number of organisms per square centimeter. A fish-based (IBI_F) and benthic macroinvertebrate-based Index of Biotic Integrity (IBI_{BM}), developed by Wilton (2004), was calculated for each reach to provide insight on ecological conditions. Each index was composed of twelve metrics that broadly reflect assemblage structure and function (Table 1). Organisms were placed into functional and tolerance guilds (e.g., trophic and spawning guilds, sensitivity to habitat degradation) following Wilton (2004). Metrics were scored and combined to obtain IBI_F and IBI_{BM} scores, both of which had a possible score varying from 0 (poor) to 100 (best). Additional details on index derivation and

Table 1 Metrics used to calculate an Index of Biotic Integrity using data on benthic macroinvertebrate (IBI_{BM}) and fish (IBI_F) assemblages in the Chariton River basin, Iowa, 2002–2005

IBI _{BM}	IBI _F
Taxa richness (MH sample)	Native fish species richness
Taxa richness (SH sample)	Catostomidae species richness
Ephemeroptera-Plecoptera-Trichoptera richness (MH sample)	Sensitive species richness
Ephemeroptera-Plecoptera-Trichoptera richness (SH sample)	Benthic-invertivore species richness
Sensitive taxa richness (MH sample)	Percentage of the three dominant species
Percentage of the three dominant taxa (SH sample)	Percentage of benthic-invertivores
Biotic index (SH sample)	Percentage of omnivores
Percentage of EPT taxa (SH sample)	Percentage of top carnivores
Percentage of Chironomidae (SH sample)	Percentage of simple lithophilic spawners
Percentage of Ephemeroptera (SH sample)	Fish assemblage tolerance index
Percentage of scrapers (SH sample)	Adjusted catch-per-unit-effort
Percentage of the dominant functional feeding group (SH sample)	Percentage of fish with deformities, erosions, lesions, or tumors

The IBI_{BM} was calculated using data from a standard-habitat (SH) and multi-habitat (MH) macroinvertebrate samples

calculation can be found in Wilton (2004). The IBI_F and IBI_{BM} have been shown to provide an excellent characterization of ecological conditions in Iowa streams (Wilton 2004; Heitke et al. 2006; Litvan et al. 2008a).

A number of multivariate and univariate statistical techniques were also used to evaluate patterns in fish and benthic macroinvertebrate assemblage structure and relations with environmental features in the watershed. Similarities in assemblages among reaches were evaluated using Jaccard's index of assemblage similarity (Jongman et al. 1995). The matrix of similarity indices was then clustered using the unweighted pair-group method to produce a dendrogram depicting clusters of reaches with similar assemblages. Separate analyses were conducted using fish and invertebrate data. Calculation of similarity indices and cluster analysis were conducted using NTSYSsp version 2.1 (Exeter Software, Setauket, New York).

Nonmetric multidimensional scaling (NMDS) was used to identify patterns in fish and benthic macroinvertebrate assemblage structure (McCune and Grace 2002). For fish, two NMDS analyses were conducted; one analysis used presence-absence data and the other used CPUE data.

Similar analyses were conducted for benthic macroinvertebrates using presence-absence (SH and MH) and density data (SH only). Analyses were based on Bray-Curtis distance matrices (McCune and Grace 2002). The strength of correlation between physical habitat and fish assemblage structure was evaluated by calculating Pearson's correlation coefficients of mean physical habitat variables with sample scores from the NMDS ordinations. Because many habitat variables were redundant (e.g., multiple variables describing substrate composition), Pearson's correlations were calculated for all pairs of habitat variables to ensure that no highly correlated (Pearson's correlation coefficient ≥ 0.70) variables were retained in the analysis. When two variables were correlated, the most ecologically significant variable or the variable with the highest relevance for management was retained for further analysis. Based on this analysis, ten variables were retained: sinuosity; mean width; mean depth; percent of the reach that was channelized; percent of the riparian corridor with vegetation; percentage of the bank actively eroding; coverage of overhanging vegetation, undercut bank, or large woody debris; and percentage of fine substrate. All NMDS analyses were conducted using PC-ORD version 5.0 (MjM Software Design, Gleneden Beach, Oregon; McCune and Grace 2002).

Stepwise multiple-regression analysis was used to further evaluate the relationship between fish and invertebrate assemblages and habitat conditions in sampling reaches. Dependent variables used in the analysis of the fish assemblage included species richness; richness of native species, catostomids, and sensitive species; percent composition and CPUE of lithophilic spawners, piscivores, and omnivores; and IBI_F. Dependent variables associated with the benthic macroinvertebrate assemblage included family richness, richness of Ephemeroptera, Plecoptera, and Trichoptera (EPT) families, percentage of individuals as EPT taxa, and IBI_{BM}. Independent variables included the ten habitat variables used in the NMDS analysis. The contribution of individual variables was tested using *F*-tests, and regression equations were limited to variables that contributed significantly (i.e., $P \leq 0.05$) to the model (Zar 2010). Regressions were conducted using transformed data (e.g., $\log_{10}[\text{CPUE} + 1]$), as well as untransformed data. Results were similar regardless of whether or not data were transformed; therefore, all results reflect those using untransformed data. In addition, nonlinear relationships were evaluated by examining bivariate plots and by including quadratic terms in the models. Nonlinear patterns were not evident. Regression analyses were conducted using SAS version 9.2 (SAS Institute 2006).

Historical data were used to provide insight on whether observed patterns in the data were consistent with observations prior to the construction of Rathbun Dam on the

Chariton River. We acquired historical data on fish sampling efforts in the watershed using the Iowa Rivers Information System (IRIS). The IRIS was developed as a tool for professionals to access fish assemblage survey data on fishes in Iowa's streams and rivers (Loan-Wilsey et al. 2005). Fish assemblage data were obtained from the published literature, agency reports, museum collections, IDNR reports and field notes, statewide biological survey databases, graduate theses and dissertations, and unpublished field notes (Loan-Wilsey et al. 2005). The database was completed in 2005 and contains 10,993 fish assemblage samples collected from 2,969 unique U.S. Geological Survey National Hydrography Dataset stream segments between 1884 and 2002. We queried the data set for all records within the study area from 1884–1964 (construction of Rathbun Dam began in September 1964). Data for 24 unique sampling sites were obtained. The difference in frequency of occurrence (i.e., percentage of sites occupied) of individual species between historical (i.e., 1884–1964) and recent surveys (i.e., 2002–2005) was calculated to provide a coarse measure of fish assemblage changes.

Results

Twenty-six species of fish and 26 families of invertebrates were sampled (Tables 2, 3). Although several fish species were relatively common across the watershed, most species were sampled in less than half of the reaches (Table 2). Percent composition of the fish assemblage (by number) varied from 0.02 to 73.9 % across species. On average across sites, the four most common fishes also dominated the total number of individuals (61.8 % of all individuals). The most common and abundant fishes were those typically considered generalist species (e.g., green sunfish, red shiner, sand shiner, fathead minnow; scientific names are provided in Table 2). Only two of the species sampled, suckermouth minnow and tadpole madtom, might be considered species sensitive to environmental stressors and only one species, tadpole madtom, is a SGCN. Similarly, a small number of invertebrate families were common across reaches (Table 3). Percent composition of benthic macroinvertebrates varied from 0.02 to 100 % among taxa and reaches. Midges (Diptera: Chironomidae) and flathead mayflies (Ephemeroptera: Heptageniidae) were the most common and, on average, the most abundant taxa. Although we anticipated observing similarities in fish assemblages based on whether a reach was a main river or tributary site, or in the northern or southern portion of the watershed, spatial structure in the fish assemblage was not apparent (Fig. 2). Similar results were observed for benthic macroinvertebrates (Fig. 3).

Nonmetric multidimensional scaling ordinations were a poor fit (i.e., high stress) to the presence–absence and

CPUE data for fish (Fig. 4) and none of the habitat data were correlated with NMDS axis scores (Table 4). The two-dimensional NMDS ordination for the occurrence of benthic macroinvertebrate families (combined SH and MH samples) was a reasonable fit to the data (stress = 12.07; Fig. 5a). The two-dimensional ordination for benthic macroinvertebrate abundance (SH samples only) was an excellent fit to the data (stress = 7.94). In particular, most of the EPT taxa (e.g., Heptageniidae, Helicopsychidae, Perlidae) were related to low levels of fine substrate, abundant large wood, and well-vegetated riparian areas (Fig. 5b; Table 4). High densities of chironomids were present in reaches with a high proportion of fine substrate.

Only one regression model explained a significant amount of variation in the fish data (Table 5). Specifically, CPUE of piscivores increased with the amount of large wood and depth ($R^2 = 0.52$; $P = 0.0001$). Family richness of benthic macroinvertebrates was positively related to the amount of riparian vegetation, but the model explained little variation in taxa richness ($R^2 = 0.22$; $P = 0.01$). Richness of EPT taxa ($R^2 = 0.29$; $P = 0.009$), the percentage of the invertebrate assemblage composed of EPT taxa ($R^2 = 0.72$; $P = 0.0001$), and the IBI_{BM} ($R^2 = 0.66$; $P = 0.0001$) were all positively related to the amount of overhanging vegetation and inversely related to the amount of fine substrate.

Despite the higher number of samples in the recent survey (38 sites) compared to the historical data (24 sites), there were 17 more species collected in the historical samples (Fig. 6). Six fish SGCN were present in the historical samples, compared to only one in the recent surveys. Some species that have declined or been lost from the assemblage appear to be those that rely on hard or rocky substrate for all or a portion of their life history (e.g., stonecat *Noturus flavus*, blackside darter *Percina maculata*); other species were common across a variety of habitats (e.g., quillback *Carpinus cyprinus*; silver chub *Macrhybopsis storeriana*). When the frequency of occurrence (i.e., percentage of sampling sites occupied) of a species in the historical samples was compared to the frequency of occurrence in recent surveys, 30 species “declined” in occurrence. Species that appear to have “increased” in occurrence are highly tolerant species (e.g., bluntnose minnow, fathead minnow) or those common in lentic habitats (e.g., largemouth bass, walleye). No species was unique to recent samples.

Discussion

Understanding patterns of community structure has long been a major focus of aquatic ecologists, and a primary goal has been identifying patterns and determining the mechanisms regulating community structure across the

Table 2 Frequency of occurrence and mean composition (numbers in parentheses are minimum and maximum values) of fishes sampled in the Chariton River basin, Iowa, 2002–2005

Species			Frequency of occurrence (%)	Mean composition (%)
Common name	Scientific name	Abbreviation		
Green sunfish	<i>Lepomis cyanellus</i>	GSF	97.3	17.54 (2.79, 73.98)
Red shiner	<i>Notropis lutrensis</i>	RDS	94.7	14.17 (2.05, 68.89)
Creek chub	<i>Semotilus atromaculatus</i>	CKC	86.8	12.10 (2.11, 59.09)
Sand shiner	<i>Notropis stramineus</i>	SAS	73.7	18.03 (3.60, 73.38)
Bluegill	<i>Lepomis macrochirus</i>	BLG	68.4	5.75 (1.17, 29.79)
Largemouth bass	<i>Micropterus salmoides</i>	LMB	68.4	4.28 (0.86, 18.52)
Bigmouth shiner	<i>Notropis dorsalis</i>	BMS	60.5	5.25 (0.94, 17.14)
Bluntnose minnow	<i>Pimephales notatus</i>	BNM	60.5	6.03 (1.37, 35.80)
Fathead minnow	<i>Pimephales promelas</i>	FHM	55.3	6.61 (2.37, 64.89)
Black bullhead	<i>Ameiurus melas</i>	BLB	47.4	2.59 (0.90, 28.57)
Johnny darter	<i>Etheostoma nigrum</i>	JDT	42.1	1.46 (0.53, 17.39)
Yellow bass	<i>Morone mississippiensis</i>	YEB	26.3	0.54 (0.23, 6.52)
Common carp	<i>Cyprinus carpio</i>	CRP	23.7	0.69 (0.32, 9.09)
White crappie	<i>Pomoxis annularis</i>	WHC	21.1	0.60 (0.31, 11.11)
White sucker	<i>Catostomus commersonii</i>	WHS	21.1	0.71 (0.30, 8.88)
Orangespotted sunfish	<i>Lepomis humilis</i>	OSS	18.4	0.85 (0.46, 16.28)
Suckermouth minnow	<i>Phenacobius mirabilis</i>	SMM	15.8	0.67 (0.39, 11.36)
Channel catfish	<i>Ictalurus punctatus</i>	CCF	10.5	0.11 (0.07, 2.22)
Freshwater drum	<i>Aplodinotus grunniens</i>	FWD	10.5	0.33 (0.24, 9.09)
Golden shiner	<i>Notemigonus crysoleucas</i>	GOS	10.5	0.12 (0.06, 1.85)
River carpsucker	<i>Carpionodes carpio</i>	RVC	10.5	0.22 (0.14, 5.00)
Brassy minnow	<i>Hybognathus hankinsoni</i>	BRM	7.9	0.81 (0.54, 18.42)
Walleye	<i>Sander vitreus</i>	WAE	7.9	0.27 (0.17, 5.71)
Bigmouth buffalo	<i>Ictiobus cyprinellus</i>	BMB	5.3	0.17 (0.12, 3.92)
Gizzard shad	<i>Dorosoma cepedianum</i>	GZS	2.3	0.09 (0.06, 2.22)
Tadpole madtom	<i>Noturus gyrinus</i>	TPM	2.6	0.02 (0.02, 0.77)

landscape (Schlosser 1982; Marsh-Matthews and Matthews 2000; Jackson et al. 2001). Fish and invertebrates often exhibit clear spatial patterns in assemblage structure (Vannote et al. 1980). The concepts of longitudinal zonation and species addition are based on predictable patterns of fish assemblage structure (Rahel and Hubert 1991; Matthews 1998; Jackson et al. 2001). Longitudinal addition is the continual addition of species to downstream reaches (Sheldon 1968; Quist et al. 2004a); whereas, faunal zonation results from the segregation of ecologically similar species into discrete faunal zones (e.g., coldwater versus warmwater species) in response to changes in stream geomorphology, thermal characteristics, or other large-scale factors (Rahel and Hubert 1991; Matthews 1998). Despite the spatial extent of our study and differences in physical habitat among stream reaches, fish and invertebrate assemblages in the Chariton River basin were not spatially structured (i.e., upstream versus downstream, river versus streams).

Fish assemblages were dominated by species tolerant of disturbance and (or) those species characterized as habitat

generalists. The only fish species sampled that might be considered sensitive were suckermouth minnow and tadpole madtom; however, neither species was common or abundant and even suckermouth minnows are frequent in degraded habitats (Quist et al. 2003). Comparisons with pre-impoundment data suggest that the fish assemblage in the Chariton River was likely more diverse and contained a greater number of habitat specialists and sensitive species than the current fish assemblage. Only 24 reaches were sampled in the watershed prior to 1965, yet 44 species were sampled, six of which were SGCN. In the current study, 38 reaches were sampled, with arguably more efficient sampling gears, but only 26 species were sampled (one was SGCN). Since 1965, additional sampling in the watershed not associated with the current project has failed to capture species other than those reported in the current study. Similar reductions in fish diversity have been observed throughout the Midwestern USA (Karr et al. 1985; Patton et al. 1998; Jacquemin and Pyron 2011), including Iowa (Zohrer 2005; Parks 2013).

The construction and subsequent management of the Rathbun Dam may explain many of the changes to the fish

Table 3 Frequency of occurrence and mean composition (numbers in parentheses are minimum and maximum values) of benthic macroinvertebrates sampled in the Chariton River basin, Iowa, 2002–2005

Family	Abbreviation	Frequency of occurrence (%)	Mean composition (%)
Chironomidae	CHIR	84.2	36.14 (5.47, 98.67)
Heptageniidae	HEPT	81.6	44.97 (5.57, 96.39)
Caenidae	CAEN	58.9	7.86 (1.83, 57.14)
Coenagrionidae	COEN	34.2	2.05 (1.04, 36.67)
Gyrinidae	GYRN	18.4	0.89 (0.50, 13.89)
Hydropsychidae	HYDR	13.2	0.71 (0.42, 14.29)
Baetidae	BAET	10.5	0.32 (0.19, 6.25)
Sialidae	SIAL	10.5	0.59 (0.31, 8.75)
Helicopsychidae	HELI	5.3	0.06 (0.05, 1.63)
Aeshnidae	AESH	5.3	0.07 (0.05, 1.85)
Calopterygidae	CALO	5.3	0.04 (0.03, 0.97)
Libellulidae	LIBL	5.3	2.66 (2.63, 100)
Gomphidae	GOMP	5.3	0.04 (0.03, 0.80)
Corydalidae	CORY	5.3	0.36 (0.28, 10.00)
Dryopidae	DRYP	5.3	0.11 (0.07, 2.35)
Dytiscidae	DYTS	5.3	0.05 (0.04, 1.37)
Isonychiidae	ISON	2.6	0.11 (0.11, 4.10)
Perlidae	PERL	2.6	0.03 (0.03, 1.00)
Leptoceridae	LEPT	2.6	0.06 (0.06, 2.46)
Limnephilidae	LIMN	2.6	0.02 (0.02, 0.88)
Polycentropodidae	POLY	2.6	0.13 (0.13, 4.92)
Curculionidae	CURC	2.6	0.02 (0.02, 0.80)
Elmidae	ELMI	2.6	0.06 (0.06, 2.35)
Corixidae	CORX	2.6	0.02 (0.02, 0.88)

assemblage in the Chariton River basin. Rathbun Dam was constructed on the Chariton River in 1965 to provide flood control, serve as a source for municipal water, and provide recreational opportunities. Prior to the completion of the dam, the entire watershed was chemically treated with rotenone (a piscicide) to remove “non-desirable” fishes. Once construction of the dam was complete, the reservoir was stocked with sport fishes to provide a recreational fishery (Paragamian 1977). Large-scale fish removal efforts were a common management practice across North America several decades ago and still remain an important tool for many management objectives (Cailteux et al. 2001; Wiley 2008). As witnessed by the species sampled in the current study, removal efforts in the Chariton River basin failed to remove every fish from the system (i.e., few of the species sampled were reintroduced). Nevertheless, treatments undoubtedly had a significant effect on fish assemblage structure. In addition to direct removal via chemical treatment, impounding the Chariton River likely also had indirect effects on fish assemblages. The effects of impoundment on downstream aquatic habitats have been studied extensively and include alterations to flow and thermal regimes, sediment and nutrient dynamics, and barriers to movement (Patton and Hubert 1993; Dynesius

and Nilsson 1994; Rosenberg et al. 2000; Miranda and Bettoli 2010). Impoundments can also influence upstream fishes by facilitating biotic interactions (e.g., predation) with introduced fishes, preventing movement, and serving as a population sink (Winston et al. 1991; Luttrell et al. 1999; Gido et al. 2002; Quist et al. 2004b, 2005). The direct and indirect effects of Rathbun Reservoir are unknown, but have likely played a role in structuring fish assemblages in upstream habitats.

The other major stressor on the Chariton River basin is intensive agriculture. Row-crop agriculture has been implicated in reduced abundance and diversity of aquatic organisms across much of North America (Wenger et al. 2008; Jacquemin and Pyron 2011; Maloney and Weller 2011). Row-crop agriculture typically results in altered flow regimes, eutrophication due to high nutrient inputs, reduced instream cover associated with diminished riparian zones (e.g., large woody debris recruitment), and high fine sediment inputs (Waters 1995; Jones et al. 2001; Yates and Bailey 2010). The Chariton River watershed has been inhabited by European settlers since the mid-1800s and agriculture has long been a dominant feature on the landscape. Like most of Iowa, the amount of land dedicated to production of row crops increased in the 1900 s following

Fig. 2 Dendrogram depicting fish assemblage similarity among reaches sampled in the Chariton River basin, Iowa, 2002–2005. Reaches with the same letters are from the same stream and the

numbers ascend from downstream to upstream. Reach abbreviations are provided in Fig. 1 and species abbreviations are provided in Table 2

increased demand and various technological advances (Burras and McLaughlin 2002; Opsomer et al. 2003). Unfortunately, BMPs were not widely practiced in Iowa until the mid-1980s even though such practices are now common in the watershed (Bultena et al. 1996; Schilling and Libra 2000; Burras and McLaughlin 2002). One would expect that widespread use of BMPs over the last several decades would improve habitat quality. Despite several years of effort, we were unable to secure data on BMPs in the watershed. Although we could not directly quantify the effects of BMPs on instream habitat, personal observations and accounts by residents indicate that habitat has improved dramatically with regard to riparian conditions, water clarity, flow characteristics, and substrate composition. As such, we expected that fish and invertebrate assemblages would reflect habitat characteristics of stream systems.

Fish assemblage structure and relative abundance of individual species in the Chariton River watershed were not related to instream habitat or riparian habitat characteristics, likely because fish assemblages were dominated by generalist species. The only exception was catch-per-

unit-effort of piscivorous fishes which was positively related to depth and the abundance of large wood. The primary piscivores in the study reaches were green sunfish, large creek chubs, and largemouth bass. Tilma et al. (1998) found that large wood explained most of the variation in abundance and biomass of spotted bass *Micropterus punctulatus* in Kansas streams. Also in Kansas streams, Quist and Guy (2001) found that growth of green sunfish and creek chub was positively related to the amount of wood. Although the specific mechanisms are unknown, deep habitats and large wood likely concentrate prey fishes and serve as cover (i.e., ambush) for piscivores.

In contrast to fish assemblages, benthic macroinvertebrate assemblages were closely related to instream habitat; the observed patterns conformed to what would be expected based on the literature. Specifically, reaches with high-quality substrates (i.e., low fines) and intact riparian vegetation tended to have higher IBI_{BM} values and family richness, particularly those taxa (e.g., EPT) sensitive to habitat degradation. Using a combination of field experiments and observational surveys in West Virginia streams, Angradi (1999) found that benthic macroinvertebrate

Fig. 3 Dendrogram depicting benthic macroinvertebrate assemblage similarity among reaches sampled in the Chariton River basin, Iowa, 2002–2005. Reaches with the same letters are from the same stream

and the numbers ascend from downstream to upstream. Reach abbreviations are provided in Fig. 1 and family abbreviations are provided in Table 3

density, biomass, and EPT taxa richness declined with increasing fine sediment. In Missouri streams, macroinvertebrate taxa richness, density, EPT richness, and EPT density were significantly correlated (negatively) with fine sediment (Zweig and Rabeni 2001). Similar findings are ubiquitous in the literature (Freeman and Schorr 2004; Kaller and Hartman 2004; Wagenhoff et al. 2012), including research focused on Iowa streams (Litvan et al. 2008b; Herringshaw et al. 2011).

Our study suggests that management legacies are influencing the structure of fish assemblages in the Chariton River system. Management legacies have been well documented over the last 15–20 years in terrestrial ecosystems, but comparatively few studies have been conducted in aquatic systems (Foster et al. 2003; Maloney et al. 2008). Nevertheless, a number of recent studies illustrate the importance and complexity of management legacies in aquatic systems. For instance, Harding et al. (1998) found that historical land use (agriculture) was a better predictor of fish and benthic invertebrate community structure than current conditions in small streams of western North

Carolina. Utz et al. (2009) found that few benthic macroinvertebrate taxa in Maryland streams were sensitive to agricultural land use in areas historically dominated by farming; whereas, several taxa were sensitive to agriculture in a region where agriculture was not historically widespread. Zimmerman and Covich (2003) found that past logging practices altered stream substrates and riparian forests in headwater streams of Puerto Rico to the point where freshwater crab *Epilobocera sinuifrons* densities continued to be affected several decades following cessation of disturbance. Burcher and Benfield (2006) evaluated the influence of urbanization on fish and benthic macroinvertebrates in North Carolina streams. Although urbanization was related to altered community structure, previous agricultural land use overwhelmed the influence of urbanization in many of the study watersheds. In small streams at Fort Benning, Georgia, stream habitat and fish and benthic macroinvertebrate assemblages were closely related to contemporary land use (Maloney et al. 2008). However, residual variation in several variables (e.g., diatom density, benthic particulate organic matter) was significantly related

Fig. 4 Nonmetric multidimensional scaling (NMDS) for fish assemblages in the Chariton River basin, Iowa, 2002–2005. Ordinations were conducted using presence–absence and catch-per-unit-effort (CPUE; fish per minute of electrofishing). Species abbreviations are provided in Table 2

to historical land use. Cuffney et al. (2010) evaluated the response of benthic macroinvertebrates to urbanization in nine metropolitan areas across the USA. The authors found that the response and rate of response of macroinvertebrate assemblages varied depending on prior land use. Although a direct evaluation of previous land use change is not possible with our data, interactions between historical land use and current disturbances are undoubtedly working to structure fish and benthic macroinvertebrate assemblages in the Chariton River system.

Across Iowa and other regions of the Midwestern USA, intensive row-crop agriculture has degraded riparian and stream habitats with consequent changes in aquatic organisms (Karr et al. 1985; Fischer et al. 2010; Jacquemin and Pyron 2011). In response, BMPs have been widely implemented to reduce the effects of agriculture (e.g., reduce nutrient and sediment input; Osborne and Kovacic 1993; Vondracek et al. 2005). Traditional habitat restoration practices generally focus on enhancing or creating habitat and then allowing organisms to recolonize (Roni

et al. 2008; Miller et al. 2010). Such restorations often fail to reach management goals for a number of reasons, many of which can be attributed to management legacies. For example, the residual effects of a land use or management practice may be so severe that restoration activities fail to enhance habitat quality (Roni et al. 2002; Lake et al. 2007; Sundermann et al. 2011), at least at temporal and spatial scales relevant to management. Another major factor that prevents expectations from being met is lack of connectivity and (or) dispersal constraints that limit recolonization (Lake et al. 2007). Reid et al. (2008) examined factors influencing the distribution of several species of redhorse *Moxostoma* spp. in the Grand River watershed, Ontario. Historical discharges of domestic sewage and industrial waste were such that virtually all fishes, including redhorse, were extirpated from the system. Despite major improvements in water quality, redhorse remained absent. The authors hypothesized that dams were preventing upstream recolonization by downstream redhorse populations. Fischer et al. (2010) examined relations between fish assemblage structure and BMPs (i.e., riparian buffer strips) in small streams of central Iowa. The study streams had been exposed to intensive row-crop agriculture (>80 % of the watershed area) for over a century. In addition, fish kills were a common occurrence, primarily due to low dissolved oxygen from eutrophication or acute runoff events from confined animal feeding operations. Over the last 20 years, the system has been the focus of extensive habitat restoration efforts (Isenhardt et al. 1997). In fact, much of the study was conducted within the Bear Creek National Restoration Demonstration Watershed (U.S. Department of Agriculture—Natural Resources Conservation Service). Despite improved habitat quality (i.e., reduced nutrient and sediment input; increased depth, coarse substrate, and large woody debris; reduced fine substrate) in reaches with riparian buffers, fish assemblages were dominated by tolerant, generalist species and were not related to measured habitat characteristics. All of the study streams were tributaries to the South Skunk River which contains a highly diverse fish assemblage. However, downstream barriers to movement in the streams prevented recolonization of fish to upstream habitats. We argue that similar factors explain patterns observed in the current study where the additive effects of chemical removal and habitat degradation from intensive row-crop agriculture resulted in a fish assemblage composed almost entirely of tolerant, generalist species; recolonization from downstream source populations has been prevented by Rathbun Dam. In contrast, dams are not a barrier to dispersal for most aquatic macroinvertebrates and explains why benthic macroinvertebrate assemblage structure reflected habitat quality.

Results of this study have obvious application to the management and assessment of stream systems. Biological

Table 4 Pearson's correlation coefficients (numbers in parentheses are associated *P*-values) between stream habitat characteristics and nonmetric multidimensional scaling (NMDS) axis scores for fishes and benthic macroinvertebrates sampled from the Chariton River basin, Iowa, 2002–2005

Variable	Fish P-A		Fish CPUE		Macroinvertebrate P-A		Macroinvertebrate density	
	NMDS1	NMDS2	NMDS1	NMDS2	NMDS1	NMDS2	NMDS1	NMDS2
Sinuosity	0.16 (0.33)	0.29 (0.08)	-0.12 (0.48)	-0.29 (0.08)	0.08 (0.65)	0.26 (0.13)	-0.15 (0.36)	-0.06 (0.74)
Mean width	-0.24 (0.15)	-0.19 (0.23)	-0.26 (0.12)	0.03 (0.86)	0.01 (0.97)	-0.05 (0.76)	0.21 (0.21)	0.17 (0.30)
Channelized	-0.26 (0.11)	-0.24 (0.15)	0.01 (0.99)	0.25 (0.13)	0.02 (0.89)	-0.24 (0.15)	0.16 (0.34)	-0.14 (0.40)
Riparian buffer	-0.20 (0.23)	-0.11 (0.49)	0.13 (0.44)	0.27 (0.09)	0.15 (0.35)	-0.25 (0.13)	0.04 (0.72)	0.41 (0.01)
Bank erosion	0.13 (0.43)	0.14 (0.39)	-0.12 (0.47)	-0.25 (0.13)	-0.16 (0.34)	-0.06 (0.72)	0.24 (0.15)	0.18 (0.28)
Mean depth	0.24 (0.14)	-0.02 (0.90)	-0.10 (0.92)	0.28 (0.08)	-0.04 (0.79)	-0.03 (0.82)	0.24 (0.14)	-0.02 (0.90)
Overhanging vegetation	-0.13 (0.43)	-0.27 (0.10)	0.03 (0.87)	0.17 (0.32)	-0.02 (0.87)	-0.01 (0.99)	-0.14 (0.41)	-0.29 (0.07)
Undercut bank	0.09 (0.58)	0.08 (0.65)	0.02 (0.89)	0.06 (0.72)	-0.51 (0.001)	0.55 (0.0004)	-0.28 (0.08)	-0.04 (0.83)
Large woody debris	-0.16 (0.34)	-0.14 (0.41)	0.04 (0.82)	0.06 (0.72)	0.08 (0.61)	0.05 (0.79)	-0.05 (0.78)	0.51 (0.001)
Fine substrate	0.02 (0.89)	-0.06 (0.70)	-0.01 (0.95)	0.18 (0.28)	-0.39 (0.02)	-0.50 (0.001)	0.40 (0.01)	-0.70 (0.0001)

The NMDS analyses were conducted using fish and benthic macroinvertebrate presence-absence (P-A), fish catch-per-unit-effort (CPUE), fish per minute of electrofishing, and benthic macroinvertebrate density (individuals per m²). Values in bold were significant at $P \leq 0.05$

Fig. 5 Nonmetric multidimensional scaling (NMDS) for benthic macroinvertebrate assemblages in the Chariton River basin, Iowa, 2002–2005. Ordinations were conducted using presence-absence and density (individuals per m²). The proportion of the reach (based on surface area) with undercut bank (UCB), fine substrate (Fine), large woody debris (LWD), and riparian vegetation (RipVeg) were significantly correlated ($P \leq 0.05$) with NMDS axis scores and are shown as vectors. Family abbreviations are provided in Table 3

assessments are widely conducted to evaluate ecosystem integrity and are based on the assumption that biological communities are an accurate reflection of ecological conditions (Karr et al. 1986; Simon 1999). Other work has focused on appropriate taxa for inclusion in bioassessments (Heino 2010). For instance, benthic macroinvertebrates are excellent indicators of local conditions because they are typically less mobile (i.e., at the larval stage) than vertebrates and display a wide range of tolerance to physical and chemical stressors (Blocksom and Johnson 2009). Invertebrates may also respond quickly to perturbations due to their short life spans and generation times. Fishes are also used in biological assessments and have been shown to be excellent indicators of ecological conditions (Simon 1999; Emery et al. 2003). Most fish are mobile and tend to be longer lived than invertebrates. Because many fishes are reliant on linear connectivity of

Table 5 Statistically significant ($P \leq 0.05$) multiple-regression models for fish and benthic macroinvertebrates sampled from the Chariton River, Iowa, 2002–2005

Model	R^2	P
Fish		
$CPUE_{pisc} = -2.215 + 0.187 (LWD) + 4.329 (DEPTH)$	0.52	0.0001
Benthic macroinvertebrates		
$S_{BM} = 0.376 + 0.072 (BUFFER)$	0.22	0.01
$S_{EPT} = 3.691 + 0.053 (OHV) - 0.019 (FINE)$	0.29	0.009
$EPT = 145.145 + 1.005 (OHV) - 1.337 (FINE)$	0.72	0.0001
$IBI_{BM} = 2.835 + 0.046 (OHV) - 0.035 (FINE)$	0.66	0.0001

Dependent variables included catch-per-unit-effort of piscivorous fishes ($CPUE_{pisc}$; fish per minute of electrofishing); taxa richness of benthic macroinvertebrates (S_{BM}); richness of Ephemeroptera, Plecoptera, and Trichoptera taxa (S_{EPT}); percentage of the sample composed of EPT taxa, and Index of Biotic Integrity based on benthic macroinvertebrates (IBI_{BM}). Independent variables included the amount of large woody debris (LWD; percentage of surface area), mean depth (DEPTH; m), riparian buffer (BUFFER; percentage of land cover with vegetation with 10 m of the stream margin), overhanging vegetation (OHV; percentage of surface area), and fine substrate (FINE; percentage of surface area with clay, silt, or sand substrate)

habitats, they also provide important information on habitat fragmentation that may not be evident with other taxa. Using multiple taxa is ideal (Yoder and Rankin 1995; Lammert and Allan 1999), but more often than not, the choice of taxa used in a biological assessment is based on financial resources. Solely focusing on financial or logistical constraints is problematic because management legacies can introduce substantial uncertainty and confusion into biological assessments. Based on the fish data (i.e., IBI_F), 97 % of the reaches in the current study would have been characterized as being in poor to fair ecological condition. Only one site was characterized as being in good condition and no sites were considered in excellent condition using data on the fish assemblage. If benthic macroinvertebrates were used to evaluate ecological integrity (i.e., IBI_{BM}), only 60 % would be characterized as fair to poor and the remaining 40 % of sites would be considered to be in good to excellent ecological condition. Discordant results between taxa in the Chariton River basin suggest that the mechanisms associated with response to prior and current disturbances are assemblage specific and that

Fig. 6 Difference in the frequency of occurrence (i.e., percentage of sampling sites occupied) of fishes in Chariton River basin, Iowa, sampled during historical (1884–1964; $n = 24$ reaches) sampling events and the current study (2002–2005; $n = 38$ reaches). Negative

values indicate a “decline” in occurrence in recent surveys. Species with an asterisk were only observed in the historical samples. Species of greatest conservation need (SGCN) are also identified

conducting bioassessments without considering management legacies can lead to improper conclusions and hinder evaluations of management and conservation actions. On a global scale, anthropogenic disturbances to terrestrial and aquatic systems will undoubtedly continue to increase in extent and intensity, as will monitoring of ecological conditions and efforts to restore ecosystem structure and function. As such, the need to consider management legacies when making inferences and management decisions is critical.

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