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[Article]

Fish Assemblages in a Western Iowa Stream Modified by Grade Control Structures

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Abstract.—Over 400 riprap grade control structures (GCSs) have been built in streams of western Iowa to reduce erosion and protect bridges, roads, and farmland. In conjunction with a companion study evaluating fish passage over GCSs in Turkey Creek, we evaluated the differences in fish assemblage and habitat characteristics in reaches immediately downstream from GCSs (GCS sites) and reaches at least 1 km from any GCS (non-GCS sites). The GCS sites were characterized by greater proportions of pool habitat, maximum depths, fish biomass, and abundance of juvenile largemouth bass *Micropterus salmoides* than were non-GCS sites. Index of biotic integrity (IBI) scores were poor or fair (<43 on a 0–100 scale) and not significantly different between the GCS and non-GCS sites. Additionally, we investigated both the longitudinal changes in fish assemblages in this GCS-fragmented stream and the changes in fish assemblages after slope modifications of three GCSs to facilitate fish passage. Thirteen fish species were present throughout the study area, whereas another 15 species exhibited truncated distributions not extending to the most upstream sampling location. After modification of the GCSs, IBI scores increased at seven of nine sites (mean increase = 4.6 points). Also, channel catfish *Ictalurus punctatus* were detected 7.3 km upstream at sites where, 2 years before GCS modification, they had been absent from collections. Given the number and distribution of GCSs in western Iowa streams, understanding the effects of these structures is vital to the conservation and management of fish assemblages in this and other regions where GCSs or similar structures are used.

In western Iowa, grade control structures (GCSs) are commonly built in streams to halt the upstream progression of channel headcuts and to stabilize streambanks (Figure 1). Streams in this ecoregion (Loess Hills and Rolling Prairies) are characterized by actively incising channels, sparse riparian vegetation, high sediment and nutrient loads, and low diversity of fish and macroinvertebrates (Wilton 2004; Heitke et al. 2006; Litvan et al. 2008b). Heitke et al. (2006) found that the streams in this ecoregion had the lowest average index of biotic integrity (IBI) scores, percent sensitive species, and species richness of all Iowa ecoregions. Moreover, the streams in this ecoregion had the lowest average width-to-depth ratio, the highest average bank slope, and a relatively high percentage of streambanks without canopy cover—all indicating a past history of stream channelization and removal of riparian vegetation (Heitke et al. 2006). In an attempt to slow channel headcutting and to protect bridges, roads, and farmland from damage caused by severe bank erosion, over 400 GCSs have been built in streams across western Iowa since the early 1990s and many more are planned. The majority of these structures consist of a 1.2-m vertical steel sheet piling and a downstream apron of rock riprap. Grade control structures are placed immediately downstream from bridges, forming large backwater pools that promote sediment deposition and bank stability around bridge pilings (Figure 1).

In addition to impeding fish passage (Litvan et al. 2008a, this issue), GCSs alter fish habitat by affecting local flow, depth, and substrate characteristics (Cooper and Knight 1987; Shields and Hoover 1991; Shields et

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FIGURE 1.—A grade control structure on Turkey Creek at low-flow conditions.

al. 1995; Shields et al. 1998). Downstream of GCSs, deep scour pools form. These deepwater areas may provide critical habitat to pool-dwelling species and refuges for all species during drought or freezing conditions (Cooper and Knight 1987; Shields and Hoover 1991; Voegele 1997). In addition, aeration of water flowing over GCSs delivers well-oxygenated water into downstream scour pools. The riprap used to construct GCSs provides coarse substrate in streams, such as those in western Iowa, that otherwise would be dominated by sand and silt. Riprap supports greater biomass, density, and diversity of macroinvertebrates than do the naturally occurring fine substrates and thus provides an enhanced local food resource for fish and other aquatic vertebrates (Litvan et al. 2008b). Some studies have shown that fish communities respond positively to increased depth and substrate heterogeneity found near GCSs (Cooper and Knight 1987; Shields and Hoover 1991; Shields et al. 1995). For example, the construction of stone weirs in streams in Mississippi that had been experiencing erosion problems similar to those in western Iowa resulted in increased pool habitat and substrate heterogeneity, which in turn led to more diverse assemblages (Shields and Hoover 1991) and larger body size (Shields et al. 1995). Poulet's (2007) study of weir effects in a French stream found that total fish species richness was higher immediately downstream of weirs than at sites distant from weirs. However, in another investigation of GCSs in Mississippi, Raborn and Schramm (2003) found that richness and assemblage structure of fish species did not differ significantly between stream segments altered by GCSs and unaltered segments. Other studies have documented deleterious effects of riprap used as bank stabilization in streams, including loss of streambank vegetation and undercut banks, reduction in lateral migration of the stream channel, and decreased salmonid biomass (Knudsen and Dilly 1987; Schmetterling et al. 2001). Furthermore, reduced fish passage, well documented for both large and small dams (Pringle et al. 2000; Santucci et al. 2005), is also a potential consequence of small, riprap structures such as GCSs (e.g., Litvan et al. 2008a).

The goal of this study was to evaluate fish assemblage structure in Turkey Creek, a western Iowa stream modified by multiple GCSs. The specific objectives were to (1) evaluate differences in habitat characteristics and fish assemblage structure in reaches immediately downstream from GCSs and reaches at least 1 km from any GCS and (2) examine the changes in fish assemblages after the slope of three GCSs was modified to facilitate better fish passage. A companion study (Litvan et al. 2008a) evaluated fish passage over GCSs in Turkey Creek. Because of the severity of erosion and resulting widespread use of GCSs in western Iowa streams, understanding the effects of these structures on habitat and fish assemblages is a necessary first step toward improving the health of these streams.

Study Area

Turkey Creek, located in the Loess Hills and Rolling Prairies ecoregion of western Iowa, is a tributary of the East Nishnabotna River and part of the Missouri River

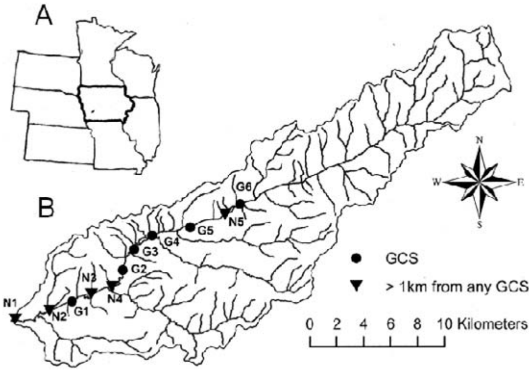

FIGURE 2.—Locations of **(A)** Turkey Creek and **(B)** the sampling sites on Turkey Creek at or 1 km from a grade control structure (GCS). Sites G1–G6 and N4 were sampled by means of passive gear and electrofishing, sites N1–N3 and N5 by electrofishing only. At least six additional stream stabilization structures (GCSs or riprap piles) are located upstream from site G6 in the main channel of Turkey Creek and its headwater tributaries.

drainage network (Omernik et al. 1993; Figure 2). Originating in northwestern Adair County, Iowa, Turkey Creek flows 49 km southwest through Cass County and drains a watershed of 331 km^2 (Iowa Department of Natural Resources Watershed Initiative 2002). Land use in the Turkey Creek watershed is dominated by intensive agriculture, 54% of the landscape being devoted to row crops and an additional 16% to livestock grazing (Iowa Department of Natural Resources Watershed Initiative 2002). Precipitation in the watershed is approximately 80 cm/year. Turkey Creek is gently sloping with a main channel gradient of 1.3 m/km (Iowa DNR Watershed Initiative 2002).

Turkey Creek has been significantly altered by anthropogenic activities, including channelization, removal of riparian vegetation, and placement of GCSs (Bulkley 1975; Larson et al. 2004). Channelization during the late 1800s and more recent projects in 1929 and 1958 left 85% of the channel nonmeandering, with a main channel sinuosity ratio less than 1.5 (Bulkley 1975). Turkey Creek has been described as resembling a ditch rather than a natural stream, with nearly vertical banks up to 6 m high (Harlan et al. 1987). Streambank erosion is prevalent and benthic substrates are dominated by silt and sand. Beginning in 1996, nine GCSs have been built downstream of bridges in Turkey Creek to stabilize the stream channel and halt the upstream progression of knickpoints (e.g., Figure 1).

The study area consisted of 11 sampling sites (Figure 2). Site names beginning with the letter *G* were located immediately downstream from GCSs and are hereafter referred to as GCS sites. Site names beginning with the letter *N* were located at least 1 km from any GCS and are hereafter referred to as non-GCS sites. The most downstream site (N1) was located 0.3 km upstream from the creek's confluence with the East Nishnabotna River, draining a watershed of 331 km^2, whereas the site most upstream (G6), 23.9 km upstream, drained a watershed of 133 km^2. Within the study area, Turkey Creek ranges from a third- to fourth-order stream approximately 3–15 m in wetted width. All sampling sites were accessed at bridge crossings; six sites (G1–G6) were reaches immediately downstream from GCSs and five sites (N1–N5) were reaches at least 1 km from any GCS and accessed by bridges without GCSs. None of the bridges in this study were low-water crossings or contained structural elements that could have restricted fish passage. Stretches of stream that appeared to be affected by bridge presence were excluded from sampling reaches. A companion study (Litvan et al. 2008a) investigated fish passage over five of the GCSs (G1–G5) included in this study. During the winter of 2004–2005, three of these structures (G1, G3, and G4) were modified to more gradual slopes to better facilitate fish passage (Litvan et al. 2008a).

Methods

Fish sampling.—Fish data collection consisted of a combination of electrofishing surveys and passive gear sampling. Five electrofishing surveys were conducted between October 2004 and May 2006 to recapture marked fish for a companion study (Litvan et al. 2008) and collect fish assemblage data. Before the slope of three GCSs (G1, G3, and G4) within the study area was modified, an electrofishing survey was conducted in October 2004. After the modifications were complete, electrofishing surveys were conducted in four separate seasons: April–May 2005, July–August 2005, October 2005, and May 2006. The October 2004 and April–May 2005 surveys were conducted at 10 sites (all except N2), and the remaining three surveys were conducted at all 11 sites (see Table 2 in Litvan et al. 2008a). The length of all stream reaches sampled with electrofishing was 280 m, 40 times the mean summer wetted width of all sites (approximately 7.0 m; Lyons 1992). Although stream wetted width varied between sites, we chose a consistent sampling length of 280 m to provide consistent recapture effort at all mark–recapture stations for the fish passage companion study (Litvan et al. 2008a). At all non-GCS sites and GCS sites with deep (>1.5-m) scour pools (G1, G2, and G3), a block net was placed at the upstream boundary of the sampling reach. The upstream endpoints of sampling reaches at all non-GCS sites were located at least 20 m downstream from bridges at the point at which bridge-related habitat effects were no longer

TABLE 1.—Ranges, means, and standard errors of habitat variables measured at sites at least 1 km from a grade control structure (non-GCS sites; $n = 5$) and at sites with such structures (GCS sites; $n = 6$) in Turkey Creek. Habitat surveys were conducted in July and August 2005; Wilcoxon rank-sum P-values indicating significant differences ($P < 0.05$) between site types appear in bold italics.

	Non-GCS sites			GCS sites			
Variables	Range	Mean	SE	Range	Mean	SE	*P*-value
Distance upstream (km)	0.3–22.4	8.9	3.7	6.3–23.9	15.4	2.5	0.1775
Mean wetted width (m)	5.0–10.5	7.6	0.9	4.5–8.7	6.7	0.6	0.4286
Mean bankfull width (m)	7.7–14.4	10.6	1.1	7.5–10.6	9.1	0.4	0.2468
Maximum depth (m)	0.47–0.72	0.61	.05	0.64–1.80	1.22	0.17	***0.0260***
Mean thalweg depth (m)	0.24–0.36	0.27	0.02	0.27–0.49	0.37	0.04	***0.0303***
Width-to-depth ratio	0.20–0.39	0.28	0.04	0.14–0.21	0.18	0.01	***0.0173***
Mean bankfull height (m)	0.76–0.98	0.86	0.04	0.79–1.07	0.94	0.04	0.1775
Mean incised height (m)	2.65–5.10	3.78	0.41	3.83–4.79	4.36	0.15	0.2468
Mean left bank angle (°)	23–53	37	6	22–73	37	7	0.7922
Mean right bank angle (°)	16–50	30	6	23–65	41	6	0.3290
Woody debris in channel (m^3)	4.6–50.6	20.2	8.0	3.8–35.3	16.0	4.6	0.7922
% Canopy cover	38–60	47	4	17–75	41	9	0.4286
% Pool	1–18	7	3	8–35	23	4	***0.0108***
% Run	70–95	88	5	47–89	67	6	***0.0281***
% Riffle	0–12	5	2	0–19	10	4	0.4199
% Hardpan	0–7	3	1	1–29	10	5	0.4026
% Fines	29–100	50	13	43–74	57	5	0.2468
% Sand	0–58	33	11	3–46	20	7	0.4286
% Gravel	0–22	12	4	3–17	7	2	0.3290
% Cobble	0–4	1	1	0–5	2	1	0.4697
% Boulder	0–5	1	1	0–8	3	1	0.1342
% Concrete	0	0	0	0–4	1	1	0.4545

evident. At GCS sites with nonwadeable scour pools (G1, G2, and G3), the upstream boundary of the sampling reach was the point at which depth became too great for wading. At GCS sites with wadeable scour pools (G4, G5, and G6), the upstream boundary of the sampling reach was located at the base of the GCS apron. Beginning 280 m downstream from the block net or GCS base, two backpack electrofishing units were used to collect fish in a single upstream pass of the sampling reach (Simonson and Lyons 1995).

Passive gears were set at sites G1, N4, G2, G3, G4, G5, and G6 for 24-h periods throughout four summer field seasons (2002–2005; see Table 2 in Litvan et al. 2008a). To decrease the bias of the mesh size on the size of fish captured, we used two types of passive gear: hoop nets (total net length, 2.6 m; hoop diameter, 61 cm; front throat diameter, 15.2 cm; back throat diameter, 10.2 cm; and mesh size, 1.9 cm) and minnow traps (throat diameter, 2.54 cm; and mesh size, 0.64 cm). At each site, one hoop net and one minnow trap baited with soy cake were set on each side of the stream channel, for a total of two hoop nets and two minnow traps at each site. To avoid influence from bridges and to locate sufficient water depth for hoop nets, passive gears were set 50–100 m downstream from GCSs or bridges. Hook-and-line sampling was also utilized during summer field seasons to supplement mark–recapture data for a companion study (Litvan et al. 2008a).

All collected fish were identified and counted. For all electrofishing surveys except the October 2004 survey, we measured batch weights of all species to determine total fish biomass. Because of heavy siltation and hazardous conditions for wading at site N1, this site was shifted upstream 200 m in July 2005, before the habitat and electrofishing surveys in summer 2005. Throughout this study our fish sampling efforts at GCS sites focused on reaches immediately downstream from structures; we did not sample fish directly upstream from GCSs. Upstream from GCSs, the water is impounded, forming moderately deep pools that promote siltation and result in hazardous wading conditions. Because of extreme channel incision and lack of boat access points to the stream, we were not able to utilize boat electrofishing gear for these impoundments but were limited to backpack electrofishing in wadeable reaches directly downstream from GCSs.

Surveys of physical habitat.—We conducted habitat surveys of the 11 fish sampling sites in July–August 2005, which were scheduled to coincide with the July–August 2005 electrofishing survey. All reaches surveyed were 280 m in length and were the same reaches sampled during the July–August 2005 electrofishing survey. Using the wadeable streams habitat assessment protocol of the U.S. Environmental Protection Agency, we measured habitat characteristics at equally spaced transects within each stream reach (Kaufmann et al.

TABLE 2.—Fish species collected in Turkey Creek. The sites at which they were present and the number collected were determined from all of the electrofishing, passive gear, and hook-and-line data collected over the period 2002–2006. Mean CPUE (number of fish collected per 100 m of stream) was calculated by combining data from five electrofishing collections from 2004–2006. Species are listed in descending order of mean CPUE.

Species	Sites where present[a]	Number collected	CPUE	
			Mean	% of total
Sand shiner *Notropis stramineus*	All	26,530	166	44
Red shiner *Cyprinella lutrensis*	All	44,486	135	36
Creek chub *Semotilus atromaculatus*	All	3,596	18	5
Bigmouth shiner *Notropis dorsalis*	All	2,143	14	4
Fathead minnow *Pimephales promelas*	All	2,257	14	4
Suckermouth minnow *Phenacobius mirabilis*	All	1,403	9	3
Green sunfish *Lepomis cyanellus*	All	1,025	5	1
Bluegill *Lepomis macrochirus*	All	960	4	1
Largemouth bass *Micropterus salmoides*	N1–G5, G6	660	4	1
White sucker *Catostomus commersonii*	G1, N4–G6	270	1	<0.5
Channel catfish *Ictalurus punctatus*	N1–G5, G6	1,581	1	<0.5
Yellow bullhead *Ameiurus natalis*	All	720	1	<0.5
River carpsucker *Carpiodes carpio*	N1, G1, N4–G5	206	1	<0.5
Brassy minnow *Hybognathus hankinsoni*	N2–G3, G5	76	<0.5	<0.5
Common carp *Cyprinus carpio*	N1, G1–G5	92	<0.5	<0.5
Stonecat *Noturus flavus*	N1, G1–G6	86	<0.5	<0.5
Gizzard shad *Dorosoma cepedianum*	N1, G1, G3, G5	21	<0.5	<0.5
Black bullhead *Ameiurus melas*	G1, N4–G5, G6	888	<0.5	<0.5
Flathead catfish *Pylodictus olivaris*	N1, G1–G4	30	<0.5	<0.5
White crappie *Pomoxis annularis*	N1, G1, N4–G3	467	<0.5	<0.5
Goldeye *Hiodon alosoides*	N1–G1, N4, G3	15	<0.5	<0.5
Shortnose gar *Lepisosteus platostomus*	N1, G1–G4	31	<0.5	<0.5
Black crappie *Pomoxis nigromaculatus*	N1, G1, N4–G4	394	<0.5	<0.5
Orangespotted sunfish *Lepomis humilis*	N1, G1, G3	5	<0.5	<0.5
Quillback *Carpiodes cyprinus*	N1, G1, N4, G3	7	<0.5	<0.5
Shorthead redhorse *Moxostoma macrolepidotum*	G1, G3, G4	9	<0.5	<0.5
Freshwater drum *Aplodinotus grunniens*	N1, N4, G3, G4	5	<0.5	<0.5
Longnose gar *Lepisosteus osseus*	G1, G3	2	<0.5	<0.5
Golden shiner *Notemigonus crysoleucas*	G5	1	<0.5	<0.5
Total fish		87,963	377	100

[a] Follows order of sites in Figure 2. For example, a species listed as present at sites G1–G2 was found at sites G1, N3, N4, and G2.

1999). Except for maximum depth, total woody debris volume, percent canopy cover, substrate composition proportion, and channel unit (i.e., run, riffle, or pool) proportion, we calculated a mean value for each habitat parameter (e.g., average left bank angle and average thalweg depth) for each stream reach surveyed.

Data analysis.—To evaluate significant differences in habitat variables between GCS and non-GCS sites, we used the Wilcoxon rank-sum test. To examine fish species distributional patterns, we tabulated species presence/absence for each sampling site and plotted the results in relation to distance upstream. All data available from electrofishing, passive gear, and hook-and-line sampling from 2002 to 2006 were used to tabulate species presence/absence at each site. For visual analysis of longitudinal species distributions, we plotted presence/absence of 15 species with truncated distributions (i.e., distributions that did not extend to the most upstream sampling location) in relation to distance upstream.

Data collected from three electrofishing surveys conducted during base flow conditions (October 2004; July–August 2005; and October 2005) were used to assess differences in fish assemblage structure between GCS and non-GCS sites via Wilcoxon rank-sum tests. Data from spring electrofishing surveys (April–May 2005 and May 2006) were excluded from this analysis because of turbid and high flow conditions that impaired sampling efficiency. Data from passive gear sampling were excluded from this analysis because passive gear did not provide a representative sample of the entire fish community and were used to sample only one non-GCS site. The variables included in the rank-sum analyses were species richness, total fish abundance and biomass, individual species abundance and biomass, and index of biotic integrity (IBI) score. Index of biotic integrity is a tool used to assess overall stream health (Karr et al. 1986). The IBI used in this study was developed by the Iowa Department of Natural Resources (IDNR) and contains 12 metrics: number of native species, number of sucker species, number of sensitive species, number of benthic invertivore species, percent abundance of the top three most abundant species, percent of fish present that are benthic invertivores, percent of fish present that are omnivores, percent of fish present that are top carnivores, percent of fish that are simple lithophilous spawners, fish assemblage tolerance index, adjusted catch per unit effort (CPUE), and adjustment for high percentages of deformities, erosions, lesions, and tumors, as discussed by Wilton (2004). This IBI defines adjusted CPUE as the number of fish per 30.5 m of stream length, excluding fish that are classified as tolerant or exotic or introduced (Wilton 2004). Using this tool, we calculated IBI scores and individual metrics for each site during three seasons (October 2004; July–August 2005; and October 2005). Rank-sum tests were conducted by using SAS (version 9.1; Statistical Analysis System, Cary, North Carolina). Results were considered significant at $P < 0.05$.

To determine whether habitat variables other than site type were related to fish community health, we conducted regression analyses of habitat variables and IBI scores. We also used regression to investigate the relationship between IBI score and distance upstream from the stream's confluence with the East Nishnabotna River. Regression analyses were performed in SYSTAT (version 9; SPSS, Chicago, Illinois). Results were considered significant at $P < 0.05$.

Nonmetric multidimensional scaling (nMDS) ordination was used to illustrate patterns in fish assemblage composition among sites. For this analysis, data from the July–August 2005 electrofishing survey were used because of the presence of migratory fish species (i.e., Ictaluridae) and stable base flow conditions. After square-root transformation of fish abundance data, Bray–Curtis similarities between sites were computed and nMDS was used to ordinate sites based on similarities in fish assemblages (Clarke and Gorley 2001). In the resulting ordination plot, distances between sites are proportional to the overall similarity of their fish assemblages (Clarke and Warwick 1994). The nMDS ordination was performed in Primer version 5 (Clarke and Gorley 2001).

Wilcoxon sign-rank tests were used to examine changes in fish assemblages after modification of GCSs. To reduce the variation due to season, only data from the premodification October 2004 and post-modification October 2005 electrofishing surveys of nine sites (G1, N3, N4, G2, G3, G4, G5, N5, and G6) were included in the sign-rank analysis. Site N2 was excluded from the analysis because it was not sampled in the 2004 survey. Site N1 was also excluded from the analysis because the sampling reach was shifted upstream 200 m after the October 2004 electrofishing survey; any differences in fish assemblages at site N1 after modification could have resulted from the different habitat characteristics present in this new reach. The variables included in the sign-rank analyses were IBI score, individual IBI metrics, species richness, total fish abundance, individual species abundance, and total biomass of channel catfish *Ictalurus punctatus*, black bullhead *Ameiurus melas*, yellow bullhead *A. natalis*, and creek chub *Semotilus atromaculatus* (for which biomass data were gathered as part of the companion study). Sign-rank tests were conducted by using SAS (version 9.1). Results were considered significant at $P < 0.05$.

Finally, to examine longitudinal changes in the abundance of four target species (i.e., those species marked as part of the companion study evaluating fish passage) before and after GCS modification, we plotted abundance in relation to distance upstream. For this analysis, the passive gear CPUE from the 2004 and 2005 summer field seasons was used to quantify fish abundance in the sampling reaches before and after GCS modification. For this part of the study, we defined CPUE as the number of fish captured per passive gear set (two minnow traps and two hoop nets at each site) per 24-h period.

Results

Physical Habitat

The GCS sites had significantly greater proportions of pool habitat ($W = 16.5$, $P = 0.011$), higher maximum depths ($W = 18.0$, $P = 0.026$), and higher average thalweg depth ($W = 18.0$, $P = 0.030$) than did non-GCS sites (Table 1). Non-GCS sites had significantly greater run habitat ($W = 42$, $P = 0.028$) and width : depth ratios ($W = 43$, $P = 0.017$; Table 1) than GCS sites. All other variables measured (i.e., substrate composition, canopy cover, total woody debris volume, and others; Table 1) were not significantly different between site type ($P > 0.134$).

Fish Assemblages

A total of 29 species and nearly 88,000 fish were sampled in Turkey Creek from 2002 to 2006 (Table 2). Thirteen species, including black and yellow bullheads, stonecats, largemouth bass, green sunfish, bluegills, creek chub, sand shiners, red shiners, fathead minnow, suckermouth minnow, bigmouth shiners, and white suckers were present at sites extending from downstream of the lowermost GCS (G1) to the sampling location farthest upstream (G6), a distance of 17.6 km, both before and after slope modification of the three GCSs in the study area. Fifteen species sampled at more than one sampling site during at least two sampling seasons exhibited truncated distributions within Turkey Creek (i.e., their distributions did not

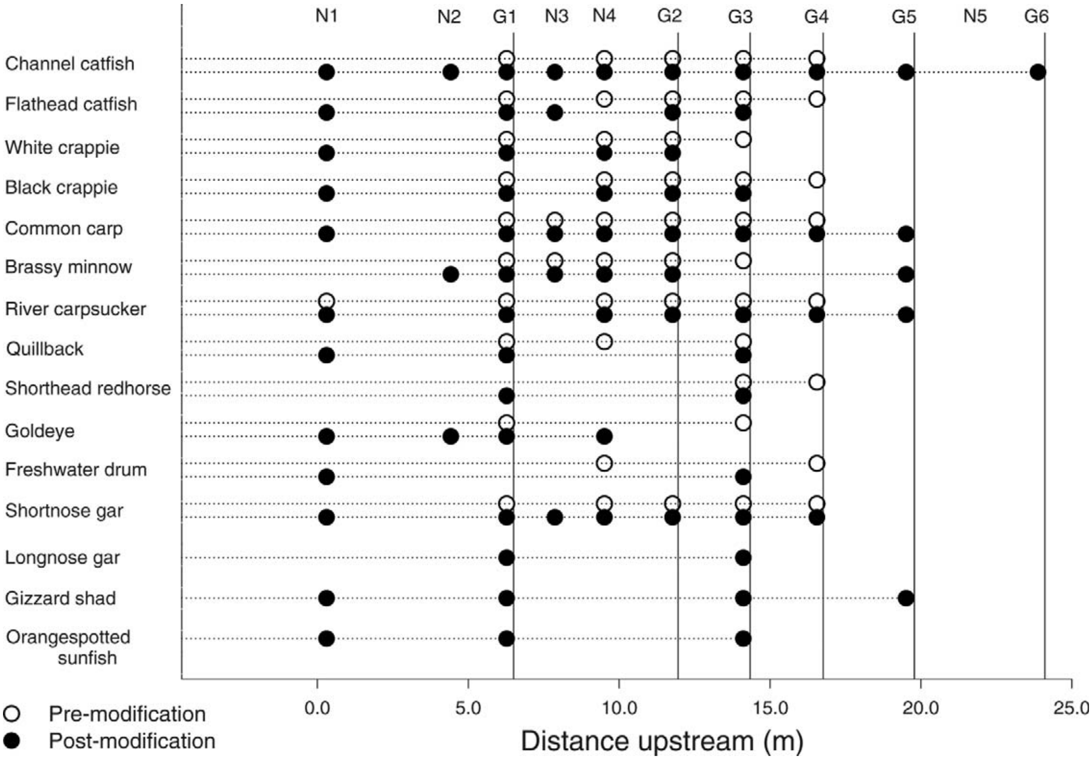

FIGURE 3.—Species with truncated distributions (i.e., ones not extending from the most downstream site to the most upstream site) in Turkey Creek. Species presence at each site is indicated for sampling conducted before and after modification of the slopes at the grade control structures at sites G1, G3, and G4. See Figure 2 for site locations.

extend to the most upstream sampling site; Figure 3). From 2002 to 2006, no white crappies, orangespotted sunfish, quillback, goldeyes, or longnose gars were found in samples from upstream of site G3 and no flathead catfish, shorthead redhorses, freshwater drum, shortnose gars, or black crappies were found in samples taken upstream of site G4 (Figure 3). Additionally, no common carp, brassy minnow, river carpsuckers, or gizzard shad were collected upstream of site G5 from 2002 to 2006 (Figure 3). In October 2005, a single golden shiner was sampled at site G5. In general, there was a 50% reduction in species richness from downstream of the most downstream GCS site to the most upstream site.

Electrofishing surveys conducted at base flow conditions indicated that GCS sites are generally characterized by greater abundance and biomass of centrarchids (mostly largemouth bass and bluegills) than non-GCS sites. In the October 2004 survey, no fish community variables (i.e., species richness, total fish abundance and biomass, and individual species abundance and biomass) were significantly different between GCS and non-GCS sites ($P > 0.114$). In the July–August 2005 survey, abundance and biomass of largemouth bass were significantly greater at GCS sites than at non-GCS sites ($P < 0.017$; Figure 4A, D). In the October 2005 survey, the abundance and biomass of largemouth bass and bluegill were significantly greater at GCS sites than at non-GCS sites ($P < 0.052$, Figure 4B, C, E, F). The majority of largemouth bass collected at GCS sites were juvenile fish (approximately 80–100 mm total length) found in scour pool habitat at the downstream base of structures. In addition, total fish biomass was significantly greater at GCS sites than at non-GCS sites during the October 2005 survey ($P = 0.017$, Figure 4G). No other fish community variables (i.e., total abundance and the remaining individual species abundance and biomass) were significantly different between site types in any electrofishing survey ($P > 0.05$).

Index of biotic integrity scores ranged from 15 to 43 (Table 3), the biotic integrity of sites being classified as "poor" (IBI score, 0–25) or "fair" (26–50; Wilton 2004). No sensitive species were collected (Table 3). Index of biotic integrity scores and IBI metrics did not differ significantly between GCS and non-GCS sites (*P*

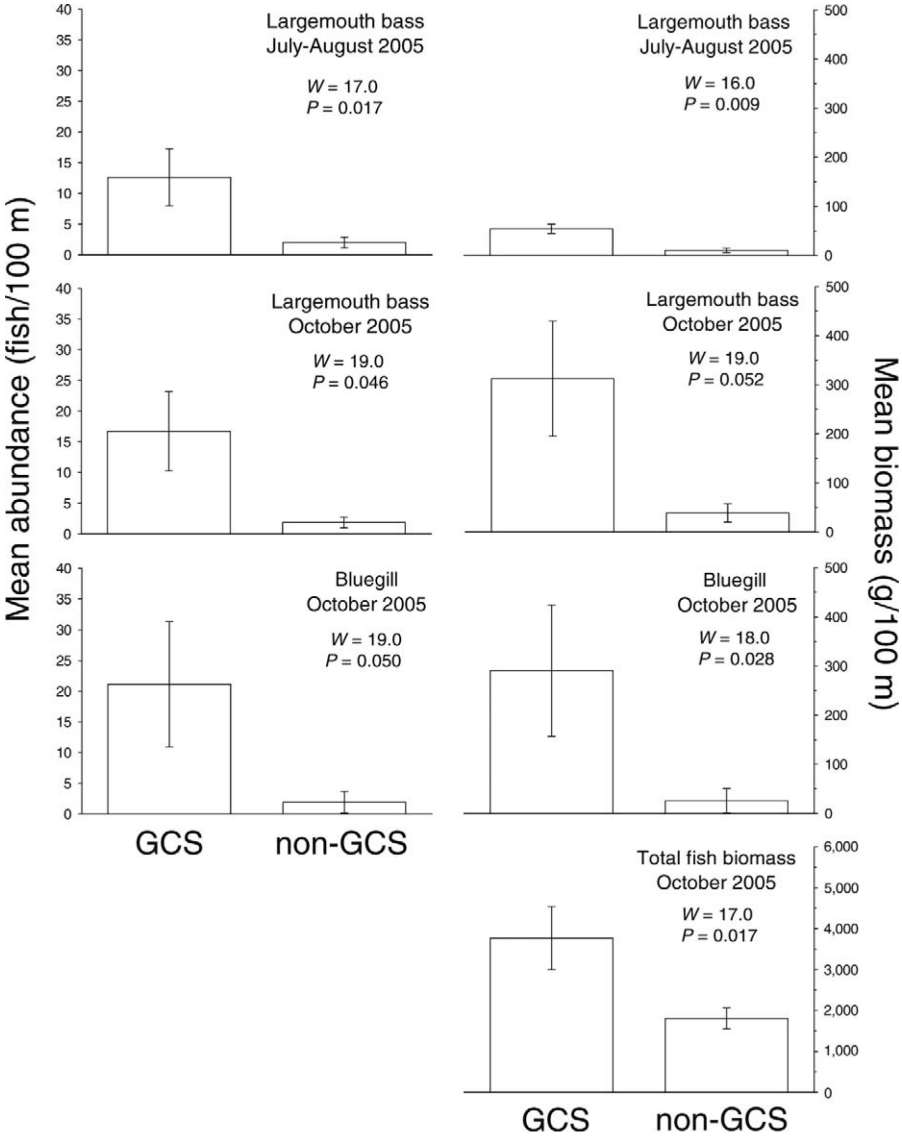

FIGURE 4.—Mean abundance of largemouth bass and bluegills (left panels) and mean biomass of largemouth bass, bluegills, and total fish (right panels) collected during July–August 2005 and October 2005 electrofishing surveys in Turkey Creek. Data are grouped by site type (GCS = sites with a grade control structure, non-GCS = sites at least 1 km from such a structure). Error bars = SEs; W = the Wilcoxon rank-sum statistic.

> 0.05), although mean IBI scores for GCS sites were greater than for non-GCS sites in all three electrofishing surveys tested. The IBI score was positively correlated with the proportion of riffle habitat (r = 0.71, P = 0.014) and negatively correlated with the proportion of run habitat ($r = -0.72$, $P = 0.013$; Figure 5). There was no significant relationship between IBI score and distance upstream ($r^2 \leq 0.18$, $P > 0.108$), indicating that fish community integrity as measured by IBI did not change significantly from downstream to upstream in this GCS-fragmented system.

Ordination by the nMDS procedure illustrated that longitudinal position along the stream, rather than site type, defined the pattern of fish assemblage similarity in Turkey Creek (Figure 6). Sites downstream from the most downstream GCS (sites N1, G1, and N2) differed from sites located upstream from the most downstream GCS (sites G2–G6). Site G1, directly downstream of

TABLE 3.—Ranges, means, and standard errors of IBI scores and metrics for sites at least 1 km from a grade control structure (non-GCS sites) and at sites with such structures (GCS sites) in Turkey Creek sampled by backpack electrofishing. Site N2 was not sampled in October 2004. See text for descriptions of metrics. The *P*-values are from Wilcoxon rank-sum tests comparing non-GCS and GCS sites; none of the metrics were statistically significant ($P \geq 0.05$).

IBI or metric	Non-GCS sites Range	Mean	SE	GCS sites Range	Mean	SE	*P*-value
Oct 2004							
IBI	15–35	27.3	4.7	28–41	32.0	1.9	0.7190
Native species	8–13	9.8	1.1	9–15	11.7	1.0	0.2000
Sucker species	0–2	1.0	0.4	0–3	1.5	0.4	0.5238
Sensitive species	0–0	0	0	0–0	0	0	1.000
Benthic invertivore species	1–2	2.0	0.4	2–3	2.2	0.2	1.000
% Top three species	77.6–95.4	88.4	4.0	75.1–94.7	84.9	3.5	0.7619
% Benthic invertivore species	0.8–6.3	2.6	1.3	0.8–13.0	5.4	2.0	1.000
% Omnivores	14.8–57.8	36.1	8.8	25.3–55.0	38.5	4.2	0.7619
% Top carnivores	0.0–0.2	0.1	0.1	0.0–0.6	0.1	0.1	0.6190
% Lithophilous spawners	0.8–6.2	2.5	1.3	0.8–12.8	4.9	2.1	0.3524
Tolerance index	7.0–8.0	7.5	0.2	7.0–8.0	7.5	0.2	0.9143
Adjusted CPUE	22–238	97	48	30–210	89	30	0.9143
DELT[a]	0–0	0	0	0–0	0	0	1.000
Jul–Aug 2005							
IBI	20–30	25.2	1.9	26–40	30.0	2.1	0.1883
Native species	8–11	10.0	0.5	10–14	11.2	0.7	0.4156
Sucker species	0–2	0.8	0.4	0–3	1.2	0.4	0.4935
Sensitive species	0–0	0	0	0–0	0	0	1.000
Benthic invertivore species	1–3	2.2	0.5	2–3	2.5	0.2	0.7078
% Top three species	65.4–87.6	77.1	4.2	70.7–81.5	77.5	1.6	0.9307
% Benthic invertivore species	1.3–7.8	4.3	1.1	1.2–11.3	5.5	1.6	0.7078
% Omnivores	31.3–56.9	41.5	4.7	26.9–48.1	39.0	3.1	0.9307
% Top carnivores	0.0–11.8	4.1	2.3	0.0–2.1	0.8	0.4	0.5736
% Lithophilous spawners	0.0–7.8	3.6	1.3	0.8–9.0	4.4	1.3	0.6623
Tolerance index	6.9–8.6	7.6	0.3	6.9–8.3	7.5	0.2	0.6623
Adjusted CPUE	5.1–28.1	15.8	3.9	16.0–38.3	22.8	3.5	0.2468
DELT[a]	0.0–1.1	0.6	0.2	0.0–1.8	0.9	0.3	0.5736
Oct 2005							
IBI	19–38	31.2	3.2	27–43	37.0	2.3	0.1169
Native species	9–12	10.4	0.6	10–16	12.3	1.1	0.2965
Sucker species	0–1	0.4	0.2	0–3	1.7	0.5	0.0887
Sensitive species	0–0	0	0	0–0	0	0	1.000
Benthic invertivore species	2–3	2.6	0.2	2–3	2.8	0.2	0.5455
% Top three species	75.0–97.2	86.4	4.0	78.6–94.5	86.0	2.7	0.9307
% Benthic invertivore species	0.3–4.8	2.8	0.8	0.8–7.8	3.6	1.0	0.5455
% Omnivores	15.1–67.8	40.2	9.4	14.2–64.8	35.0	7.2	0.6623
% Top carnivores	0.0–20.2	4.6	3.9	0.0–0.9	0.3	0.2	0.3853
% Lithophilous spawners	0.2–4.0	2.3	0.7	0.6–7.7	3.2	1.0	0.6623
Tolerance index	6.1–8.6	7.3	0.4	6.8–8.3	7.3	0.2	0.7922
Adjusted CPUE	14–360	152	57	93–217	134	18	1.000
DELT[a]	0–0.1	0.02	0.02	0–0	0	0	0.4545

[a] Percent with deformities, erosions, lesions, and tumors.

the most downstream GCS, was also dissimilar from sites N2 and N3, non-GCS sites located approximately 1,800 m downstream and 1,600 m upstream, respectively, from site G1. Species collected frequently from sites downstream of the most downstream GCS but rare or absent at upstream sampling locations include short and longnose gars, gizzard shad, goldeyes, freshwater drum, and other species characteristic of larger rivers. In contrast, the sites furthest upstream, including sites G5, G6, and N5, were similar to one another in species diversity and characterized by abundance of creek chub and other cyprinids. Sites located in the middle of the stream (N3, N4, G2, G3, and G4) also were similar to one another, having greater abundances of yellow bullheads and fathead minnow than those in either the downstream or upstream sites (Figure 6).

After modification of the three GCSs, IBI scores increased by a mean of 4.6 points in October ($S = 15.5$, $P = 0.031$; Figure 7). After modification of the GCS at site G4, the October IBI score increased by 10 points at each of the GCS sites upstream from this modification (G5 and G6). Examining individual IBI metrics indicates that this overall increase in IBI after modification was primarily due to a combination of

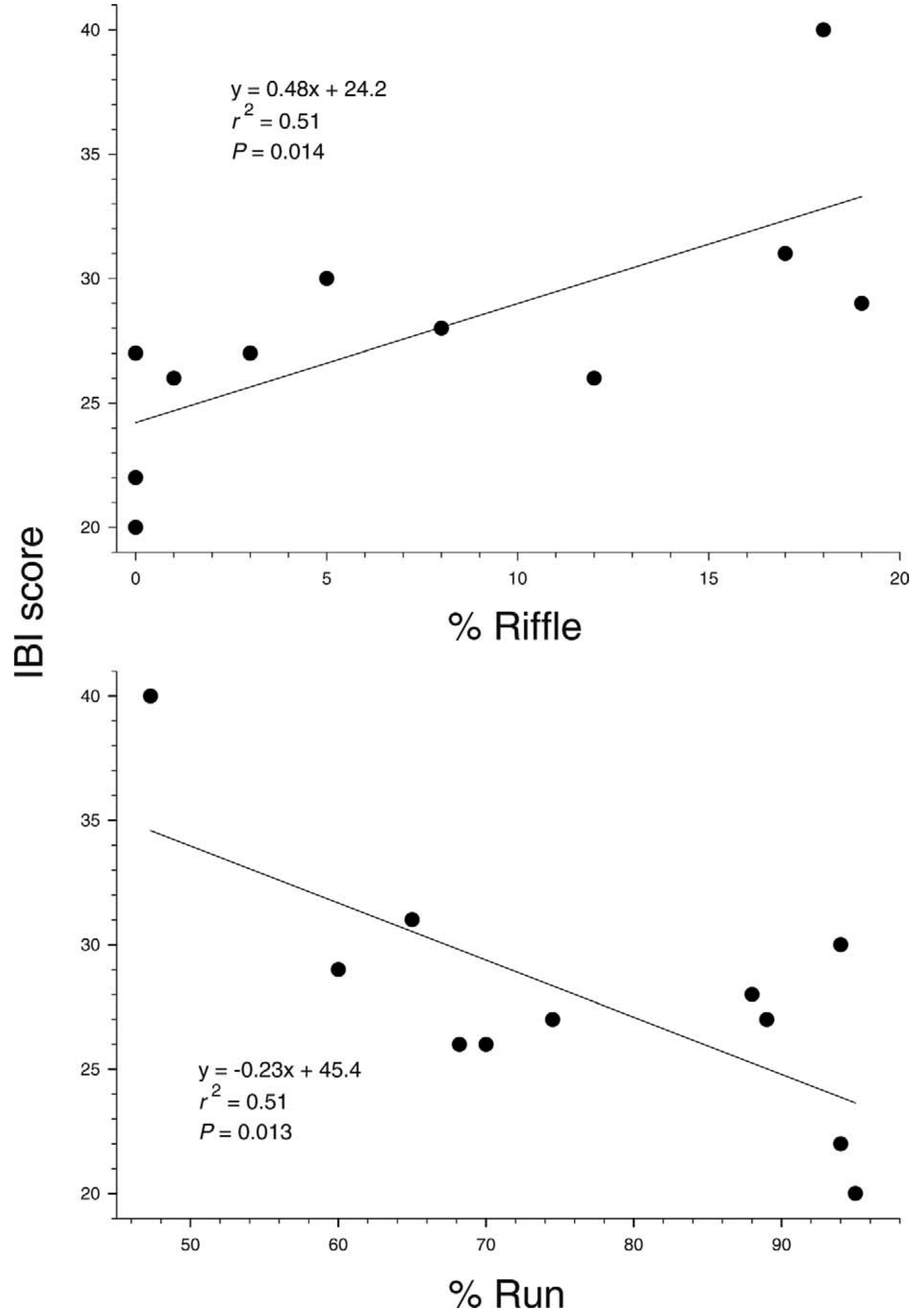

FIGURE 5.—Correlations between index of biotic integrity (IBI) scores and the proportions of riffle and run habitat measured at 11 sites on Turkey Creek during July–August 2005.

increased benthic invertivore species ($S = 7.5$; one-sided test: $P = 0.031$) and increased percentage of fish that were top carnivores ($S = 7.5$, $P = 0.078$). In addition, the abundances of total ictalurids and total centrarchids collected during October electrofishing surveys were significantly greater in 2005, after the GCS modifications ($S > 18.0$, $P < 0.008$). In particular, abundances of yellow bullheads, largemouth bass, green sunfish, and creek chub were significantly greater in the October 2005 survey ($S > 13.0$, $P < 0.031$).

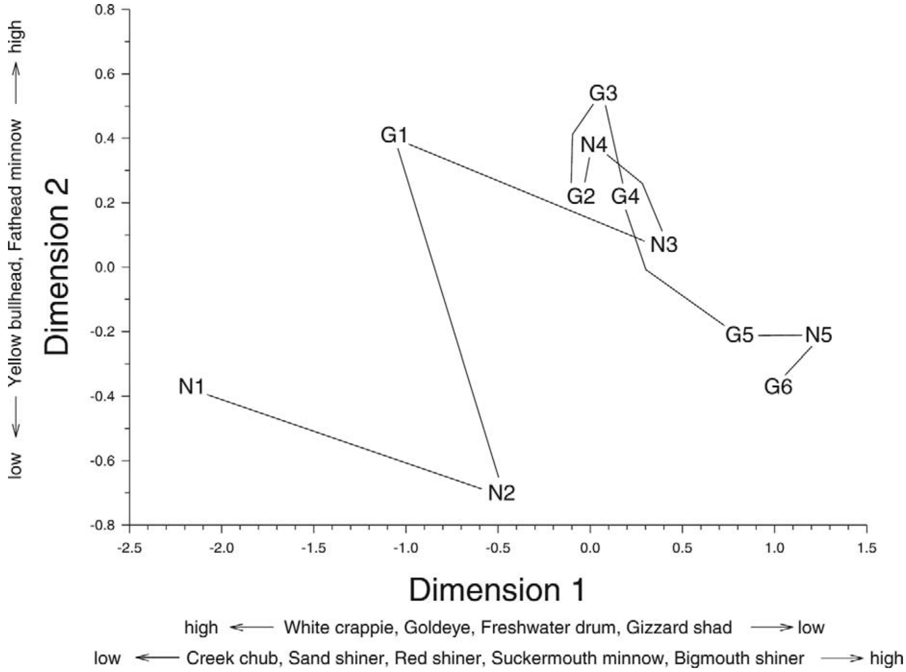

FIGURE 6.—Nonmetric multidimensional scaling ordination of sites on Turkey Creek, calculated from a Bray–Curtis similarity matrix of fish species abundance data from the July–August 2005 electrofishing survey. The distances between sites are proportional to the similarity of their fish assemblages. Lines connect the sites in longitudinal sequence, beginning at site N1 (the furthest downstream) and ending at site G6 (the furthest upstream). The species listed along the axes were significantly correlated with dimension scores ($P < 0.05$) and are included to facilitate interpretation. The ordination stress value was 0.04.

In both 2004 and 2005, the passive gear CPUE of channel catfish was greatest downstream of the most downstream GCS (site G1) and decreased with distance upstream (Figure 8). No channel catfish were present in samples taken upstream of site G4 from 2002 to 2004, before modification of the GCSs. After modification of the GCS at site G4, channel catfish were caught at sites G5 and G6, indicating that the modification of this structure may have allowed channel catfish to move upstream. In both 2004 and 2005, CPUE of yellow bullheads and creek chub increased with distance upstream and was highest at sites G5 and G6 (Figure 8). In 2005, CPUE of black bullheads was greatest at downstream GCS sites and lower at the non-GCS site and upstream GCS sites (Figure 8).

Discussion

Previous studies have shown that alteration of stream habitat by impoundments, specifically the formation of deep pools and backwaters, may lead to a local shift from assemblages comprised primarily of lotic species to assemblages dominated by lentic species preferring slow-water habitat (Pringle et al. 2000; Taylor et al. 2001; Gehrke et al. 2002). In a study of a warmwater channelized stream in Ohio, Edwards et al. (1984) found that relative abundances of centrarchids, including largemouth bass, green sunfish, smallmouth bass *Micropterus dolomieu*, bluegills, longear sunfish *Lepomis megalotis*, and rock bass *Ambloplites rupestris*, were significantly greater in a channelized reach rehabilitated by artificial riffles than in a naturally meandering reach and an unmitigated channelized reach. Shields and Hoover (1991) also reported a high abundance of centrarchids, particularly longear sunfish, in pools created by GCSs, and Shields et al. (1995) reported an increase in the relative abundance of centrarchids from 11% to 55% after restoration of a channelized stream with stone weirs. In a study of the effects of impounding a warmwater stream in southern Illinois, Taylor et al. (2001) found that the fish assemblage shifted from a preimpoundment community dominated by cyprinids to a postimpoundment community dominated by centrarchids, which accounted for 54% of the total stream fish community and 78% of fish collected from impounded lentic habitat.

Our study in a typical western Iowa stream indicated that reaches immediately downstream from GCSs are deeper, have greater proportions of pool habitat, and

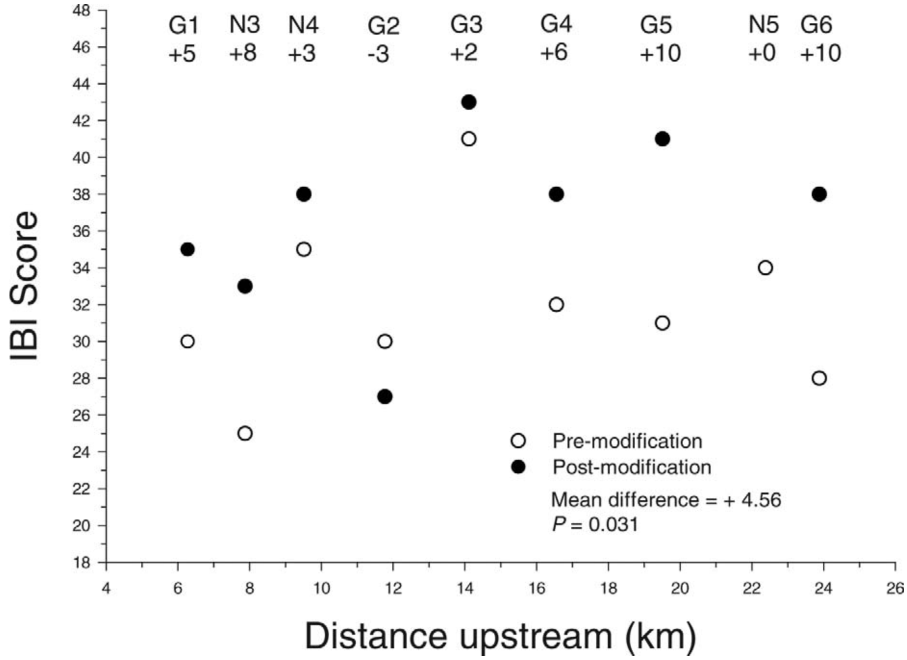

FIGURE 7.—Index of biotic integrity (IBI) scores in Turkey Creek before and after modification of the slopes at the grade control structures at sites G1, G3, and G4. The premodification scores are from October 2004, the postmodification scores from October 2005. The sampling sites and changes in IBI score are indicated at the top.

support greater centrarchid abundance and biomass, particularly largemouth bass and bluegills, than reaches not associated with GCSs. In addition, in the post-modification (October 2005) electrofishing survey we found that GCS sites supported greater total fish biomass than non-GCS sites. The greater biomass at GCS sites may have resulted from the greater proportion of deepwater habitat that support larger-bodied fish; non-GCS sites, by contrast, are shallower and dominated by small cyprinids (i.e., red shiners and sand shiners). The largemouth bass and bluegills in Turkey Creek may have washed in from nearby farm ponds or been colonists from riverine populations. In the Missouri River, largemouth bass are generally found near shore but are more abundant in relatively deep nearshore areas with structure than in shallow areas of comparable position (Harlan et al. 1987; Berry et al. 2004). In Turkey Creek, pool habitat is scarce other than near GCSs; thus pool-dwelling species such as largemouth bass and bluegills concentrate in scour pools downstream of GCSs, which are the largest and deepest pools available. Future research should seek to determine whether instream structures such as artificial riffles and GCSs stimulate increased fish production or are merely acting as fish attractors, drawing fish away from less preferred habitats nearby.

Overall, the IBI scores at GCS sites were not significantly different from those at non-GCS sites, although they averaged slightly higher at GCS sites in all three surveys analyzed. Similarly, no significant differences between GCS and non-GCS sites were seen in total fish abundance, species richness, or any other IBI metrics. Previous studies have also failed to demonstrate significant fish assemblage effects of GCSs and artificial riffles compared with that in reference reaches (Pretty et al. 2003; Raborn and Schramm 2003). The apparent lack of response to instream structures that enhance habitat may be due to poor water quality, altered hydrologic regime, barriers to fish passage, or pervasive degradation in the surrounding watershed, any of which could neutralize the potential benefit of localized habitat improvements (Pretty et al. 2003). A more holistic approach, including the improvement of water quality and restoration of a more natural hydrologic regime as well as the mitigation of barriers to fish passage may be necessary to fully realize the potential of localized instream physical habitat restoration to enhance fish assemblages (Pretty et al. 2003).

The longitudinal changes in fish assemblage structure that we documented in Turkey Creek are probably a reflection of both natural longitudinal patterns and fragmentation from multiple GCSs. Several species (e.g., red shiners, green sunfish, creek chub, and 10

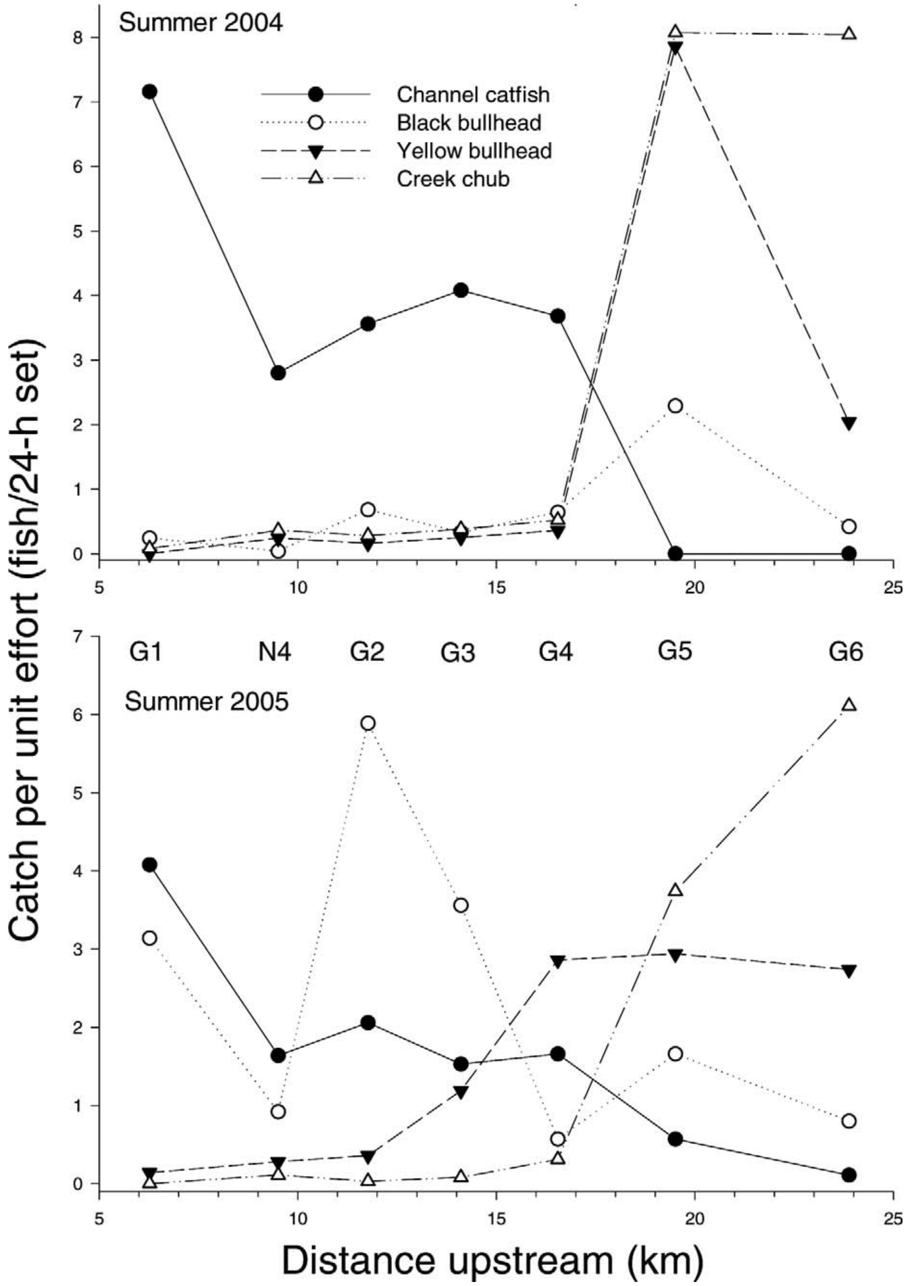

FIGURE 8.—Catch per unit effort with passive gear of channel catfish, black bullheads, yellow bullheads, and cheek chub in summer 2004 and summer 2005 in Turkey Creek. Sampling sites are indicated at the top of the bottom panel.

others) were present in all portions of the stream we sampled, whereas 15 other species had truncated distributions that did not extend to the most upstream sampling location. Accordingly, the pattern of similarity among sites illustrated by our nMDS ordination was primarily driven by longitudinal position rather than by site type. Longitudinal patterns in biotic assemblages, and the related pattern of assemblage changes with changing stream size, are well documented (Sheldon 1968; Evans and Noble 1979; Vannote et al. 1980; Rahel and Hubert 1991; Allan 1995). Other studies have shown that in streams with barriers species richness declines more in locations upstream from the barriers than would naturally be expected because of elevation or distance upstream (Reyes-Gavilán et al. 1996; Porto et al. 1999; Joy and Death 2001; Dodd et al. 2003). In southwestern Iowa there are no streams of comparable size to Turkey Creek without GCSs (Larson et al. 2004). This lack of a suitable reference stream, coupled with the intense effort required for the mark–recapture component of a companion study (Litvan et al. 2008a), limited our study to one stream system. Therefore, we were unable to rigorously separate the natural longitudinal effects from the effects of fragmentation resulting from GCSs. However, because of the pervasiveness of GCSs in this region, their demonstrated effects on fish passage (Litvan et al. 2008a) and habitat, and their likely effects on fish assemblages, our results should provide important guidance for managers and benchmarks for future evaluations.

After modification of the GCSs, IBI scores increased from 2 to 10 points in seven of the nine sites. Although encouraging, these increases should be interpreted with caution. Because we were able to compare IBI scores from only one premodification and one postmodification sampling period and had no reference stream, we cannot rule out the possibility that the differences were due to some uncontrollable interannual effect. With appropriate caution, we believe our results suggest that modification of barriers to facilitate fish passage may have an overall positive effect on the fish assemblages in Turkey Creek and similar streams in western Iowa and elsewhere; furthermore, we believe that IBIs are a useful tool with which to evaluate future changes in the health of streams impacted by GCSs.

In addition to improving overall fish assemblage health, GCS modification is intended to increase the number of sport fish, particularly channel catfish. Our results indicated that channel catfish abundance declined with distance upstream in this GCS-fragmented stream and was greatest downstream of the most downstream GCS, both before and after GCS modification. In a study of an unfragmented Missouri River tributary, Dames et al. (1989) found that the majority of channel catfish (72%) were located within the lower 8 km of the tributary and that 95% of channel catfish were located in the lower 20 km. The decline of channel catfish abundance from downstream to upstream in Turkey Creek may reflect a natural tendency of this species to remain closer to the larger East Nishnabotna River. However, because channel catfish are known to move between larger rivers and their tributaries, often using smaller tributaries as spawning and nursery areas (Hubert 1999), reduced passage because of GCSs will probably be deleterious to this species in Turkey Creek over time. After modification of three GCSs in Turkey Creek to facilitate fish passage, we found a significant increase in ictalurids collected upstream of the modified structures. Many of the ictalurids collected after GCS modification were young-of-year fish, indicating that channel catfish were successfully spawning in this GCS-modified stream.

The inverted longitudinal distributions of channel catfish versus yellow bullheads and creek chub suggest another possible role of GCSs: providing refuge from predation or competition by blocking the passage of predator or competitor species. Before modification, channel catfish were not collected upstream of the GCS at site G4, where the downstream apron of riprap had washed away from the sheet piling, creating in essence a vertical low-head dam. In contrast, yellow bullheads and creek chub were not only present upstream of site G4 but much more abundant upstream than downstream of site G4. Although our study was not designed to test for biotic interactions, these distributions suggest that release from predation by or competition with channel catfish explains the greater abundance of yellow bullheads and creek chub upstream of site G4. Previous studies have observed abrupt changes in species abundance upstream of barriers that may have been the result of release from predation or competition (Fraser et al. 1995; Joy and Death 2001). After modification of the GCS at site G4 to improve fish passage, channel catfish were detected upstream from this structure but at relatively low abundance compared with downstream sites. Yellow bullheads and creek chub continued to persist at relatively high abundances upstream of site G4 after GCS modification; it is unclear whether this means that (1) channel catfish had no influence on their distributions in the first place, (2) channel catfish were not yet abundant enough upstream of site G4 to reduce the abundances of those species as they had downstream of site G4, or (3) our study was not long enough to detect interactions. Additional research will be required to resolve this question, but clearly the potential role of biotic interactions should be considered when assessing the effects of barriers to passage and the consequences for fish assemblages.

Management Implications

The GCSs in western Iowa streams were constructed solely for the purposes of bank stabilization and the protection of bridges, roads, and farmland; enhancement of aquatic habitat was not a goal in the original design of these structures. Beneficial consequences of these structures—such as increased pool depth, increased substrate and flow heterogeneity, and increased macroinvertebrate abundance and diversity (Litvan et al. 2008b)—are unintentional and should be weighed against their negative effects as barriers to fish passage.

In fact, the creation of scour pools by GCSs is an unintentional effect of structure design and may eventually undermine the stability of the structure, causing it to fail. Thus, future GCSs should be designed to prevent the formation of scour pools; allowing fish passage as well as stabilizing streambanks and preventing headcutting should be high priorities. To improve stream habitat and fish assemblage health in this region, resource managers should modify existing GCSs to allow fish passage under a wide spectrum of flow conditions and should construct new GCSs judiciously, keeping overall stream health as an important design goal.

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