EFFECTS OF GRADE CONTROL STRUCTURES ON THE MACROINVERTEBRATE ASSEMBLAGE OF AN AGRICULTURALLY IMPACTED STREAM

M. E. LITVAN,[a]* T. W. STEWART,[a] C. L. PIERCE[b] and C. J. LARSON[c]

[a] *Department of Natural Resource Ecology and Management, Iowa State University, Ames, Iowa 50011, USA*
[b] *U.S. Geological Survey, Iowa Cooperative Fish and Wildlife Research Unit, Iowa State University, Ames, Iowa 50011, USA*
[c] *Iowa Department of Natural Resources, 57744 Lewis Road, Lewis, Iowa 51544, USA*

ABSTRACT

Nearly 400 rock rip-rap grade control structures (hereafter GCS) were recently placed in streams of western Iowa, USA to reduce streambank erosion and protect bridge infrastructure and farmland. In this region, streams are characterized by channelized reaches, highly incised banks and silt and sand substrates that normally support low macroinvertebrate abundance and diversity. Therefore, GCS composed of rip-rap provide the majority of coarse substrate habitat for benthic macroinvertebrates in these streams. We sampled 20 sites on Walnut Creek, Montgomery County, Iowa to quantify macroinvertebrate assemblage characteristics (1) on GCS rip-rap and at sites located (2) 5–50 m upstream of GCS, (3) 5–50 m downstream of GCS and (4) at least 1 km from any GCS (five sites each). Macroinvertebrate biomass, numerical densities and diversity were greatest at sites with coarse substrates, including GCS sites and one natural riffle site and relatively low at remaining sites with soft substrates. Densities of macroinvertebrates in the orders Ephemeroptera, Trichoptera, Diptera, Coleoptera and Acariformes were abundant on GCS rip-rap. Increases in macroinvertebrate biomass, density and diversity at GCS may improve local efficiency of breakdown of organic matter and nutrient and energy flow, and provide enhanced food resources for aquatic vertebrates. However, lack of positive macroinvertebrate responses immediately upstream and downstream of GCS suggest that positive effects might be restricted to the small areas of streambed covered by GCS. Improved understanding of GCS effects at both local and ecosystem scales is essential for stream management when these structures are present. Copyright © 2007 John Wiley & Sons, Ltd.

KEY WORDS: grade control structure; artificial riffle; benthic macroinvertebrates; bank erosion; stream bank stabilization

INTRODUCTION

Throughout much of the world, streams have been degraded by anthropogenic stresses including channelization, removal of riparian vegetation, agricultural and industrial pollution, hydrologic alterations and other deleterious land-use practices (Karr *et al*., 1985; Brookes, 1988; Harper *et al*., 1998). In particular, channelization of streams is a long established method of river regulation that has been practiced worldwide and is prevalent in agricultural regions (Brookes, 1988). Channelization decreases habitat for aquatic organisms by increasing peak stream flows and reducing benthic substrate heterogeneity (Brookes, 1988). The lateral area over which stream flow can dissipate is reduced when a straightened channel is disconnected from its floodplain (Simon and Darby, 2002). Consequently, stream energy is forced vertically down and peak discharges increase, resulting in channel incision and stream bank erosion (Simon and Darby, 2002). Sediment from overland runoff and within-channel erosion buries natural rocky substrates and storm flow events wash woody debris and leaf packs downstream, destroying habitat for aquatic organisms. Furthermore, as incised channels erode, infrastructure and farmland are lost by widening of stream banks (Baumel *et al*., 1994). In response to deteriorating conditions of streams, many rehabilitation projects use in-stream structures such as boulder weirs, artificial riffles and flow deflectors to increase habitat heterogeneity and

---

*Correspondence to: Mary E. Litvan, Missouri Department of Conservation, 1110 S. College Avenue Columbia, Missouri 65201, USA.
E-mail: Mary.Litvan@mdc.mo.gov

stabilize the stream channel (Shields and Hoover, 1991; Ebrahimnezhad and Harper, 1997; Laasonen *et al.*, 1998; Harrison *et al.*, 2004).

In western Iowa, USA, streams have been highly impacted by channelization and agricultural land use and are characterized by actively incising channels, sparse riparian vegetation, high sediment and nutrient loads and low fish and macroinvertebrate diversity (Wilton, 2004; Heitke *et al.*, 2006). In this region, stream bank erosion is exacerbated by wind deposited silt (loess) that originates from the Missouri River alluvial plain and forms rolling hills that are highly prone to erosion (Prior, 1991). Drastic down cutting occurs in streams of this region, resulting in nearly vertical banks over 5 m deep (Figure 1A). Channels have also widened dramatically, leading to an estimated US$1.1 billion loss of farmland and bridge infrastructure (Baumel *et al.*, 1994). To protect bridge stability and decrease loss of farmland by erosion, grade control structures (GCS) that consist of a vertical metal dam and a downstream piling of rock rip-rap (Figure 1B) have been placed in streams to slow upstream progression of knickpoints and reduce bank erosion upstream from structures (Gu *et al.*, 1999). Nearly 400 GCS of this design have been constructed in Missouri River tributary streams in western Iowa since the early 1990s, and many more structures have been proposed or are currently under construction (Larson *et al.*, 2004).

Use of rock rip-rap for bank stabilization is a common practice of river engineers (Schmetterling *et al.*, 2001; Linhart *et al.*, 2002), and many studies have suggested both beneficial and deleterious ecological effects of rip-rap
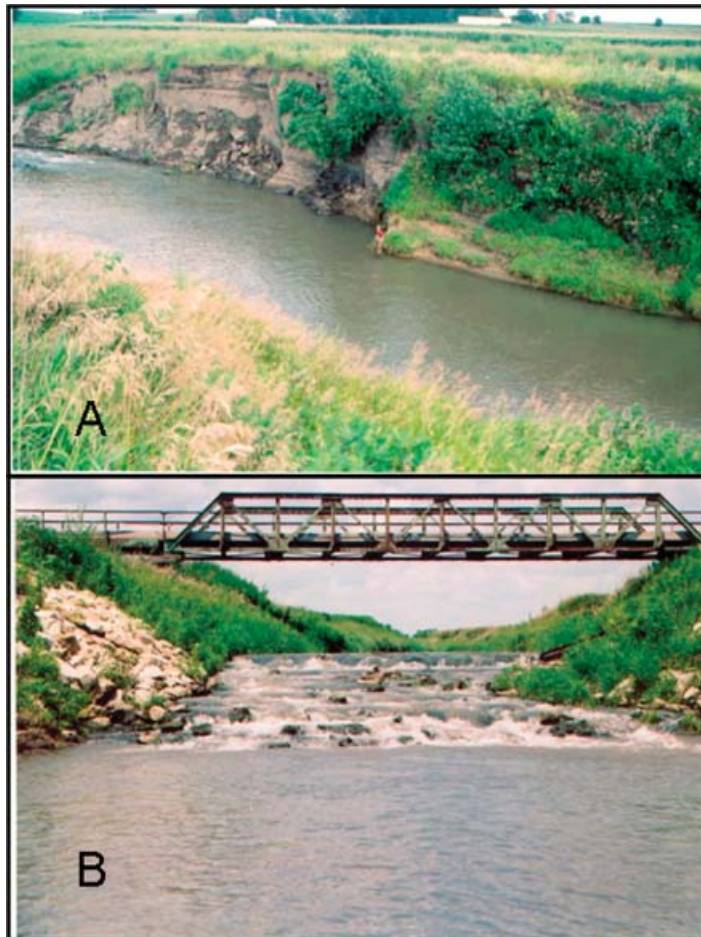

Figure 1. Walnut Creek, Iowa is characterized by an actively incising channel and severe bank erosion. For scale, note the researcher standing below the cut bank (A). Grade control structures (GCS), consisting of a metal dam and a downstream apron of rock rip-rap, have been constructed downstream of bridges to stabilize banks and reduce erosion (B)

and GCS in aquatic ecosystems (Shields and Hoover, 1991; Schmetterling *et al.*, 2001; Linhart *et al.*, 2002). Steeply sloped structures in western Iowa may restrict fish movement, and efforts are currently underway to design GCS that will facilitate easier fish passage (Larson *et al.*, 2004). Rip-rap used in bank stabilization projects may be deleterious to stream ecosystems by restricting natural fluvial processes such as lateral channel migration, thereby causing riparian vegetation loss, bank undercutting and homogenization of near-bank habitat (Schmetterling *et al.*, 2001). In addition, Knudsen and Dilley (1987) found that rip-rapped banks negatively impacted juvenile salmonid biomass and Peters *et al.* (1998) found lower salmonid abundance near rip-rapped banks than at natural banks. However, Linhart *et al.* (2002) found aquatic mosses on rip-rap support a diverse and abundant invertebrate meiofauna community. By increasing local substrate size and heterogeneity, artificial riffles produced by rip-rap could benefit aquatic organisms, including benthic macroinvertebrates, by providing additional habitat in degraded streams dominated by soft substrates. Macroinvertebrates are a critical link in aquatic food webs and play a vital role in stream ecosystem function (Wallace and Webster, 1996). By consuming primary producers and decomposing organic matter, and in turn serving as prey for vertebrates, aquatic macroinvertebrates function as conduits of energy and nutrients to organisms at higher trophic levels (Wallace and Webster, 1996).

Environmental features, including flow velocity, depth, canopy cover, water chemistry and food supply, have been shown to affect the composition and distribution of macroinvertebrate communities (Merritt and Cummins, 1996). In particular, substrate is a primary determinant of community structure, with taxa richness and total macroinvertebrate abundance often increasing with increased habitat heterogeneity (Beisel *et al.*, 1998; Downes *et al.*, 2000; Negishi and Richardson, 2003). Previous studies demonstrated that macroinvertebrates readily colonize rip-rap and boulders placed in lakes and streams, and achieve relatively high levels of abundance and diversity because of increased habitat surface area and complexity offered by these substrates (Schmude *et al.*, 1998; Negishi and Richardson, 2003). In addition to changes in substrate, GCS and lowhead dams affect flow characteristics within a stream, often resulting in an area of slow flow dominated by fine substrate particles upstream of the structure, and a high-energy scour pool downstream of the structure (Shields and Hoover, 1991; Harper *et al.*, 1998). By altering substrate, depth and flow characteristics, GCS in western Iowa streams may significantly affect local macroinvertebrate community structure.

The overarching goal of this study was to identify effects of GCS on the macroinvertebrate assemblage structure in Walnut Creek, Iowa, an agriculturally impacted stream modified by GCS. Specific objectives were to: (1) quantify and compare total macroinvertebrate biomass, numerical density and taxa richness at GCS sites, nearby upstream and downstream sites that were likely greatly influenced by GCS, and sites far from (at least 1 km) any GCS, (2) identify macroinvertebrate taxa that responded positively and negatively to conditions at GCS and nearby sites and (3) quantify and compare physicochemical features at GCS, nearby sites and sites far from GCS, and relate these to macroinvertebrate community features. Modification of streams with GCS is pervasive in western Iowa, with nearly every low order stream containing at least one in-stream structure. It is therefore imperative that we understand GCS effects on aquatic communities in these altered stream ecosystems.

METHODS

Study area

Walnut Creek, located in the steeply rolling loess prairies ecoregion of western Iowa, is a tributary of the West Nishnabotna River and part of the Missouri River drainage network (Omernik *et al.*, 1993; Figure 2). Originating in southeastern Shelby County, Iowa, Walnut Creek flows 104 km south through Pottawattamie and Montgomery Counties to north central Fremont County and drains a watershed of 582 km^2 (Iowa DNR Watershed Initiative, 2002). During the first three decades of the 20th century, Walnut Creek was extensively channelized, resulting in a loss of 48 km of stream channel or a 40% reduction in total stream length (Bulkley, 1975). Post-channelization, main channel gradient is 1.02 m km^{-1}, 98% of the channel is non-meandering with a main channel sinuosity ratio less than 1.5, streambanks are highly incised and headcutting and erosion of the stream channel is prevalent (Bulkley, 1975; Iowa DNR Watershed Initiative, 2002). Land use in the Walnut Creek watershed is dominated by intensive agriculture with about 72% of the landscape devoted to row crops and an additional 6% to livestock grazing (Iowa DNR Watershed Initiative, 2002). Although the majority of native riparian vegetation has been
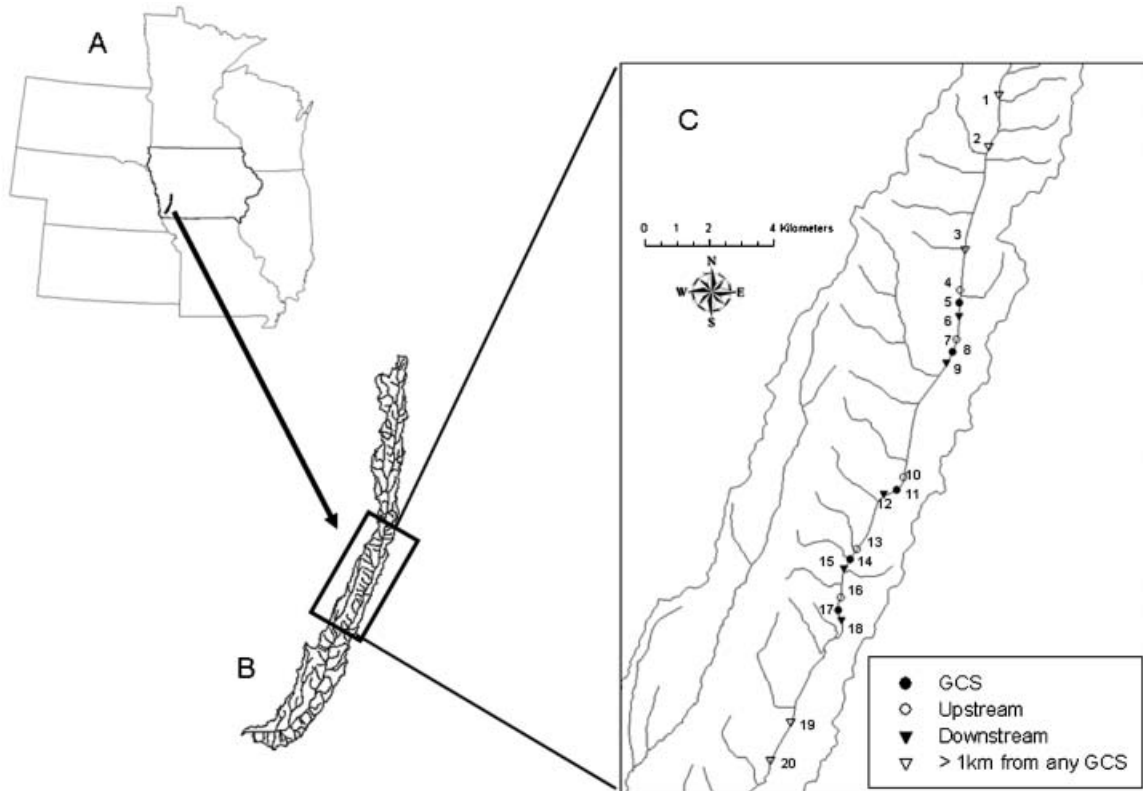
Figure 2. The midwestern United States, with Iowa outlined in bold and the Walnut Creek watershed in black (A). The Walnut Creek watershed with the study area is also outlined (B). Sites 1–20 within the study area are indicated (C). Descriptions of GCS sites (black circles), upstream (open circles) and downstream (black inverted triangles) sites near GCS, and sites >1 km from any GCS (open inverted triangles) are provided in the text

replaced with row crop agriculture, a small amount of remnant riparian forest exists. Annual precipitation in the watershed is approximately $80 \, \text{cm} \, \text{year}^{-1}$.

There are currently 16 GCS in Walnut Creek. The five GCS sampled in this study were constructed in March 2001 and have 20:1 back slopes of approximately 20–30 m in length (Larson *et al*., 2004). Tributaries of Walnut Creek contain an additional 19 stream stabilization structures. At sites sampled in this study, Walnut Creek is a 3rd order stream ranging from about 5 to 20 m wide. The study area was in the middle reaches of the creek (41° 2.59′ N, 95° 17.32′ W to 41° 12.94′ N, 95° 13.09′ W), with the uppermost site draining a watershed of $193 \, \text{km}^2$ and the most downstream site draining a watershed of $283 \, \text{km}^2$.

Sampling occurred at 20 sites on Walnut Creek that were distributed between four basic site types ($n = 5$ sites for each site type; Figure 2). The four site types included: (1) GCS, specifically the rock rip-rap forming the downstream apron, (2) immediately upstream of GCS in an area characterized by impounded water and silt deposition, (3) immediately downstream of GCS where water flowing over GCS is delivered and streambed scouring occurs and (4) stream reaches located at considerable distance from GCS. At each of the five separate bridge crossings with an associated GCS, sampling sites were located directly on top of the GCS substrate ($n = 5$), 5–50 m upstream from GCS ($n = 5$) and 5–50 m downstream from GCS ($n = 5$; Figure 2). The fourth site type sampled included sites at least 1 km away from any GCS structure where local habitat conditions did not appear to be heavily influenced by GCS; these sites ($n = 5$) were accessed by bridges not associated with a GCS (Figure 2).

Macroinvertebrate assemblages

Benthic macroinvertebrates were sampled from 20 sites on Walnut Creek from 25 to 27 July 2004 (Figure 2). At each of five separate bridge crossings with associated GCS, a sample was taken directly from substrate composing

each GCS. Additionally, a sample was taken 5–50 m upstream of each GCS, and also 5–50 m downstream of each GCS. An additional five samples were taken at sites at least 1 km away from any GCS.

At each sampling site, a 10-m long section of stream was designated as the sampling transect. Within this transect, two locations were randomly selected from a grid of 100 equally spaced points that covered the entire transect length and width. A 30.5-cm wide D-frame kick net was used to collect macroinvertebrates using methods modified from the USEPA's wadeable streams bioassessment protocol (Barbour *et al.*, 1999). After placing the net frame firmly against the substrate, a $0.09\,m^2$ plot was visually determined. At GCS and other sites with coarse substrates and strong currents, all rocks lying at least halfway within this plot were held in front of the net and scrubbed by hand so that dislodged organisms drifted into the net. Rocks with less than 50% of their surface area enclosed in the plot were removed from the plot and therefore excluded from the sample. Scrubbed rocks were placed outside of the sampling plot and remaining substrate was disturbed by shuffling feet for 30 s. In areas of low flow velocity, the net was swept through the sampling plot while kicking substrate into the net for 30 s. All sampling was performed by the same investigator. Both samples from locations within a single transect were combined to form a composite sample of organisms representative of the entire site (the total area sampled at each site was $0.18\,m^2$). Each composite sample (hereafter referred to as sample) was filtered through a 500-μm sieve and retained material was preserved in 10% formalin containing rose bengal dye. Formalin was replaced by 70% alcohol within 24 h of sample collection.

In the laboratory, all components of each sample were examined under $10\times$ magnification and macroinvertebrates were separated from other material. Insects and mollusks were identified to family and other taxa were identified to order or class (Merritt and Cummins, 1996; Thorp and Covich, 2001). Numbers of individuals in each taxon were counted and densities were expressed as number of individuals m^{-2} horizontal benthic surface area. Taxa richness of each sample was expressed as number of taxa $(0.18\,m)^{-2}$. Total macroinvertebrate biomass at each site was quantified by drying invertebrates at $60^\circ C$ for 24 h and ashing at $550^\circ C$ for 3 h to determine ash-free dry mass (g AFDM m^{-2}; APHA 2005).

Physicochemical variables

Substrate composition, dissolved oxygen, pH, canopy coverage, water temperature, depth, channel wetted width, flow velocity and distance upstream from the most downstream sampling site were measured at each sampling location. Similar to macroinvertebrate data, both values for each variable from within the same transect were averaged to yield a single value representative of the sample site. Substrate composition in each sampling plot was visually estimated as the proportion of each of the following substrate particle size classes: silt/clay/mud (<0.06 mm diameter), sand (0.06 mm–2 mm diameter), gravel (>2 mm–64 mm diameter) and cobble/boulder (>64 mm diameter; Gordon *et al.*, 1992). We quantified substrate composition by assigning codes to substrate size classes in the following manner: 1 = silt, 2 = sand, 3 = gravel and 4 = cobble/boulder (e.g. Bain *et al.*, 1985; Inoue and Nunokawa, 2002; Negishi and Richardson, 2003). A substrate composition score ranging from 1.0 to 4.0 was calculated based on the proportion of each substrate type at each sampling location. For example, a sample collected from a plot containing 60% cobble and 40% gravel would have a substrate score of 3.6 $((0.6 \times 4) + (0.4 \times 3) = 3.6)$. Sites with high coverage by cobble/boulder, including GCS sites and one natural riffle, had high substrate composition scores, whereas sites dominated by silt and sand had low substrate composition scores.

From the middle of each sampling location, dissolved oxygen and pH were measured with electronic probes (Milwaukee SM600 Smart DO meter, Oakton pH Tester 1), canopy coverage was determined with a spherical densiometer, temperature was measured with a standard lab thermometer, depth was measured to the nearest centimetre and channel wetted width was measured to the nearest 0.1 m. Due to malfunction of our electronic flow meter while in the field, stream velocity was measured at each sample location by recording time required for a tennis ball to travel the 10 m length of the sampling transect and pass over the sampling location (Gordon *et al.*, 1992). To obtain the most accurate velocity measurement possible, the object was placed in the water upstream of the sampling location, so that it would pass over the sampling location while travelling downstream. Two velocity trials were conducted for each sampling location and time values were averaged to determine stream surface

velocity for that sampling location. GPS coordinates of sampling sites were recorded and the distance upstream from the most downstream sample site was measured using ArcGIS (version 9.0, ESRI, Redlands, California).

Finally, in July 2005 we estimated the proportion of naturally occurring coarse benthic substrate (gravel/cobble/boulder) within the entire study area of Walnut Creek, using a canoe to travel the stream from Site 1 to Site 20, a distance of 22.9 km. Length of stream with any observed natural coarse substrate was measured and divided by the total distance travelled to obtain a conservative estimate of the per cent of natural coarse substrate within the main channel of the study area. Of the 22.9 km of main stream channel within this study area, 1.3 km (6%) contained any natural coarse benthic substrates, illustrating the scarcity of natural coarse substrates in this stream.

Data analysis

Analysis of variance (ANOVA) and Tukey's pair-wise comparisons were used to determine differences in macroinvertebrate and physicochemical variables between site type. Macroinvertebrate variables analysed with ANOVA included total macroinvertebrate density, individual taxon density and total macroinvertebrate biomass. We also gained insight into differences in biological integrity across site types using four macroinvertebrate-based Index of Biotic Integrity (IBI) metrics employed by the Iowa Department of Natural Resources (Wilton, 2004). Metrics included total taxa richness, EPT taxa richness (number of taxa in the orders Ephemeroptera, Plecoptera and Trichoptera), proportion EPT taxa contributing to total macroinvertebrate density and proportion of the three most numerically dominant taxa at each site (Wilton, 2004). We also tested for site differences in the following physicochemical variables: substrate composition score, depth, flow velocity, dissolved oxygen, pH, temperature, per cent canopy cover, stream wetted width and distance upstream from the most downstream sampling location.

Of the five sites located at least 1 km from any GCS, four sites were characterized by primarily silt and sand substrate, while one site was a natural riffle with large gravel and cobble substrate. Graphical analysis of the natural riffle site indicated that macroinvertebrate community and physicochemical features for this site were outliers relative to the four sites with silt and sand substrate. Therefore, data from this natural riffle site were excluded from ANOVAs, but data values for this natural riffle site are included in tables and graphs for comparison purposes.

Principal components analysis (PCA), including taxa whose numbers composed at least 1% of any sample, was used to illustrate invertebrate taxon associations at sites. In addition, correlation analyses of PCA factor scores with physicochemical variables at each site were used to determine physicochemical features related to variation in macroinvertebrate taxa densities.

Prior to conducting statistical analyses, data for macroinvertebrate density, biomass and taxa richness were examined to determine if they deviated from assumptions of parametric statistical techniques (i.e. normality, homoskedasticity) and were transformed when necessary (Zar, 1999). Macroinvertebrate abundance and biomass data were highly heteroskedastic, and therefore $\log_{10} (x + 1)$ transformed (Zar, 1999). Additionally, canopy cover measurements (percentages) were arcsine-square-root-transformed (Zar, 1999). All data analyses were performed in SYSTAT (version 9, SPSS, Chicago, Illinois) and relationships were considered statistically significant at $p \leq 0.05$. When multiple tests were conducted on the same data set simultaneously, we report unadjusted p values in addition to Bonferroni corrected critical p values ($\alpha = 0.05$/number of dependent variables) that are used to maintain the overall probability of committing Type I error at $p \leq 0.05$ (Scheiner, 2001).

RESULTS

Physicochemical variables

Sites located directly on GCS substrate resembled artificial riffles and were characterized by coarse substrate and high substrate composition scores, shallow depths and swift flow velocities; a natural riffle site located >1 km from any GCS exhibited conditions similar to GCS sites (Table I). Sites located 5–50 m upstream from GCS were relatively deep, due to impoundment of water by GCS (Table I). Relative to GCS, these sites had reduced water velocity and low substrate composition scores due to deposition of fine sediment (Table I). Due to scouring effect of swiftly flowing water leaving the base of each GCS riffle, sites located 5–50 m downstream from GCS resembled

Table I. Mean values ± SE for physicochemical variables in four categories of sites in Walnut Creek, Iowa

Physicochemical variable	Site category					ANOVA p-value
	5–50 m upstream from GCS (n = 5)	Grade control structure (GCS) (n = 5)	5–50 m downstream from GCS (n = 5)	>1 km from any GCS sand/silt (n = 4)	>1 km from any GCS natural riffle (n = 1)	
Substrate score	1.1 ± 0.0 (a)	3.8 ± 0.1 (b)	1.7 ± 0.2 (c)	1.7 ± 0.3 (a,c)	3.2	<0.001
Velocity (m/s)	0.09 ± 0.03 (a)	0.73 ± 0.11 (b)	0.27 ± 0.08 (c)	0.46 ± 0.05 (b,c)	0.56	<0.001
Depth (m)	0.64 ± 0.05 (a)	0.28 ± 0.02 (b)	0.67 ± 0.06 (a)	0.45 ± 0.01 (b)	0.16	<0.001
% Canopy cover	12 ± 7	0 ± 0	2 ± 1	20 ± 13	15	0.115
Dissolved oxygen (mg/L)	8.1 ± 0.1	8.2 ± 0.2	8.1 ± 0.1	8.2 ± 0.1	9.3	0.946
pH	8.3 ± 0.0	8.3 ± 0.0	8.3 ± 0.0	8.3 ± 0.0	8.4	0.327
Temperature (°C)	20.2 ± 0.9	20.0 ± 0.5	20.0 ± 0.7	22.8 ± 0.6	23.0	0.053
Wetted width (m)	11.6 ± 0.4	11.7 ± 0.8	9.4 ± 1.1	8.3 ± 0.7	18.4	0.030*
Distance upstream (km)	10.83 ± 2.06	10.80 ± 2.06	10.75 ± 2.06	15.54 ± 5.28	1.36	0.624

Physicochemical variables measured at one natural riffle are also included for comparison. Distance upstream is the distance from the most downstream site of the study area (site number 20). The natural riffle site was excluded from ANOVAs, but values for this site are provided for comparative purposes. Letters in parentheses represent site types that are significantly different based on Tukey's pairwise comparisons. Sites with the same letter are not significantly different at $p \leq 0.05$.
*Tukey's multiple comparisons test showed no differences between specific sites ($p \geq 0.058$).

pools with relatively deep, slow moving water and were dominated by sand and silt substrate that also resulted in low substrate composition scores (Table I). Excluding the natural riffle site, the remaining four sites located >1 km from any GCS resembled run habitat typical of channelized streams and were characterized by moderate depth and flow velocity values, slightly greater per cent canopy cover than sites located near GCS and silt and sand substrate which resulted in low substrate composition scores (Table I).

Exclusive of the Bonferroni adjusted critical p-value ($\alpha = 0.05$), ANOVA results indicated significant differences in depth, flow velocity, substrate composition score and stream-wetted width between site types ($p \leq 0.030$; Table I). Depths at GCS sites and sand/silt sites >1 km from GCS were similar and shallower than sites 5–50 m upstream and downstream from GCS (Table I). Flow velocity was lower at sites immediately upstream of GCS relative to all other site types, and also lower at sites immediately downstream from GCS relative to GCS sites (Table I). Substrate composition score (i.e. mean particle size) was greater at GCS sites than all other sites, and lower immediately upstream of GCS than at sites immediately downstream of GCS (Table I). Although ANOVA indicated an overall difference in wetted width among site types ($p = 0.03$), multiple comparisons tests revealed that differences among specific site types were weak (Turkey's test; $p \geq 0.058$). With a Bonferroni adjusted critical p-value ($\alpha = 0.05/9$ environmental factors $= 0.006$), ANOVA results indicated significant differences in depth, flow velocity and substrate composition score between site types ($p < 0.001$; Table I). Canopy cover, dissolved oxygen, pH, temperature and distance upstream did not differ between site types ($p \geq 0.053$; Table I).

Macroinvertebrate density and biomass

ANOVA indicated a significant difference in macroinvertebrate density ($p < 0.001$) and biomass ($p = 0.003$) between site types (Figure 3; Table II). Sites located on GCS had greater total macroinvertebrate densities than all other site types (Turkey's test; $p < 0.001$; Figure 3A; Table II). Total macroinvertebrate densities on GCS rip-rap ranged from 1756 to 15 215 individuals m^{-2} and were 1.9–124 times greater at GCS sites than at all other sites, excluding one natural riffle sample which had a density of 13 792 individuals m^{-2}. Total macroinvertebrate densities at sites immediately upstream from GCS (range $= 262$–939 individuals m^{-2}), immediately downstream from GCS (123–498 individuals m^{-2}) and at fine-particle substrate sites at least 1 km from any GCS (226–708 individuals m^{-2}) did not differ (Turkey's test; $p \geq 0.29$; Figure 3A; Table II).

GCS sites also had significantly greater macroinvertebrate biomass than all other site types (Turkey's test; $p \leq 0.013$; Figure 3B; Table II). Total macroinvertebrate biomass on GCS rip-rap ranged from 0.22 to 4.78 g AFDM m^{-2} and was 1.4–239 times greater than at 13 of the remaining sites with fine-substrate streambeds. Biomass at one GCS site (site 17) exceeded the biomass of the natural riffle site (4.25 g AFDM m^{-2}; Figure 3B; Table II). Total macroinvertebrate biomass at sites immediately upstream from GCS (0.03–0.11 g AFDM m^{-2}), immediately downstream from GCS (0.02–0.16 g AFDM m^{-2}), and at soft substrate sites at least 1 km from any GCS (0.04–0.33 g AFDM m^{-2}) were not different (Turkey's test; $p \geq 0.997$; Figure 3B; Table II). One site >1 km from any GCS that was dominated by fine substrates had a relatively high total macroinvertebrate biomass due to abundant Sphaeriidae clams (Figure 3B; Table II).

Taxa richness and other biotic integrity metrics

ANOVAs indicated significant differences in total taxa richness, EPT taxa richness, proportion EPT taxa contributing to total macroinvertebrate density and proportion of the three most numerically dominant taxa between site types ($p \leq 0.027$; Figure 4). Taxa richness at GCS sites was greater than at the other site types (Turkey's test; $p \leq 0.003$; Figure 4A). Taxa richness was 1.2–3.2 times greater at GCS (13–16 taxa (0.18 m)$^{-2}$) and the natural riffle site (15 taxa (0.18 m)$^{-2}$) than at all other sites (5–11 taxa (0.18 m)$^{-2}$; Figure 4A). Taxa richness at sites located upstream from GCS, downstream from GCS and at least 1 km from any GCS did not differ (Turkey's test; $p \geq 0.965$; Figure 4A).

EPT taxa richness and proportion of EPT taxa contributing to total macroinvertebrate density were significantly greater at GCS sites than at other site types (Turkey's test; $p \leq 0.051$; Figure 4B–C). Proportion of the three most numerically dominant taxa was greater at sites immediately upstream of GCS than at GCS sites (Turkey's test; $p = 0.022$; Figure 4D). However, values for this metric at GCS sites and sites immediately upstream from GCS did
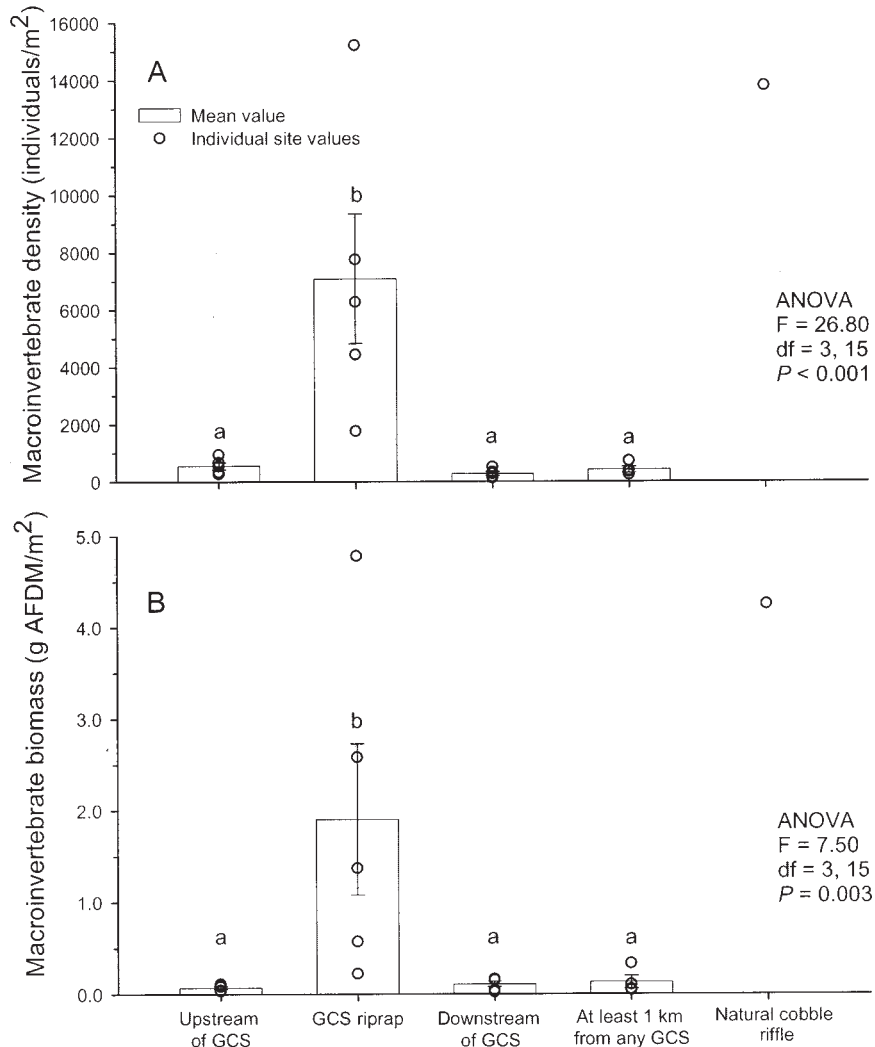
Figure 3. Mean (±SE) macroinvertebrate density (A) and biomass (B) at GCS sites, sites immediately upstream and downstream of GCS, and sites at least 1 km from any GCS. Individual site values are indicated with open circles. Values for density and biomass at one natural cobble riffle site are provided for comparison purposes

not differ from sites located immediately downstream from GCS and sites at least 1 km from any GCS (Turkey's test; $p \geq 0.152$; Figure 4D).

Individual taxon responses

Twenty-eight taxa were identified in this study (Table II). ANOVAs indicated a significant difference in numerical densities of 13 macroinvertebrate taxa between site types ($p \leq 0.034$; Table II). In general, macroinvertebrate communities on GCS rip-rap were dominated by ephemeropteran, trichopteran and dipteran taxa (Table II). Nine taxa, including the ephemeropteran taxa Baetidae, Heptageniidae, Isonychiidae and Tricorythidae, the megalopteran taxon Corydalidae, the trichopteran taxa Hydropsychidae and Hydroptilidae and the dipteran taxa Chironomidae and Simuliidae were most abundant at GCS sites and significantly greater at GCS sites than all other site types (Tukey's test; $p \leq 0.045$); these taxa were also numerous at the natural riffle site (Table II). Two of these taxa, including the megalopteran taxon Corydalidae and the caddisfly taxon Hydroptilidae were present only at GCS sites and the natural riffle (Table II). Water mites (order Acariformes) were more abundant at GCS sites than

Table II. Taxa densities (mean $\pm$ SE individuals m^{-2}), total densities and total biomass (g AFDM/m^2) at four site types and one natural riffle in Walnut Creek, Montgomery County, Iowa

Taxon densities (individuals m^{-2})	Site category					ANOVA p-value
	5–50 m upstream from GCS (n = 5)	Grade control structure (GCS) (n = 5)	5–50 m downstream from GCS (n = 5)	>1 km from any GCS sand/silt (n = 4)	Natural riffle (n = 1)	
Turbellaria	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	11	No value
Hirudinea	0.0 ± 0.0	1.1 ± 1.1	1.1 ± 1.1	0.0 ± 0.0	0	0.565
Oligochaeta	383.8 ± 128.5 (a)	72.0 ± 46.0 (b)	96.7 ± 31.0 (a,b)	123.6 ± 69.5 (a,b)	623	0.059
Mollusca						
Lymnaeidae	1.1 ± 1.1 (a)	1.1 ± 1.1 (a)	2.1 ± 2.1 (a,b)	26.9 ± 13.7 (b)	0	0.035
Physidae	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0	0.454
Sphaeriidae	1.1 ± 1.1	0.0 ± 0.0	2.1 ± 2.1 (a)	0.0 ± 0.0	0	0.454
Acariformes	0.0 ± 0.0 (a)	8.6 ± 3.2 (b)	0.0 ± 0.0 (a)	26.9 ± 13.7 (b)	0	0.005
Amphipoda	2.1 ± 1.3 (a,b)	0.0 ± 0.0	0.0 ± 0.0	1.3 ± 1.3	0	0.007
Ephemeroptera	54.8 ± 30.1 (a)	4,465.5 ± 1,396.4 (b)	89.2 ± 27.2 (a)	82.0 ± 33.6 (a)	8115	<0.001
Baetidae	23.6 ± 15.6 (a)	1,005.1 ± 335.4 (b)	2.1 ± 1.3 (a)	5.4 ± 3.1 (a)	2816	<0.001
Caenidae	1.1 ± 1.1	4.3 ± 3.1	22.6 ± 8.4	21.5 ± 14.1	5	0.138
Ephemeridae	12.9 ± 3.6 (a)	0.0 ± 0.0 (b)	0.0 ± 0.0 (b)	0.0 ± 0.0 (b)	0	<0.001
Heptageniidae	5.4 ± 5.4 (a)	489.1 ± 85.8 (b)	9.7 ± 4.6 (a)	16.1 ± 9.3 (a)	376	<0.001
Isonychiidae	7.5 ± 6.3 (a)	2,181.1 ± 1,026.2 (b)	11.8 ± 7.7 (a)	6.7 ± 4.0 (a)	3273	<0.001
Tricorythidae	4.3 ± 3.1 (a)	785.8 ± 132.3 (b)	43.0 ± 23.7 (a)	32.2 ± 17.8 (a)	1645	<0.001
Odonata						
Gomphidae	4.9 ± 2.2	3.2 ± 2.2	9.7 ± 3.6	1.3 ± 1.3	0	0.318
Hemiptera						
Corixidae	1.1 ± 1.1	1.1 ± 1.1	0.0 ± 0.0	0.0 ± 0.0	0	0.630
Megaloptera						
Corydalidae	0.0 ± 0.0 (a)	8.6 ± 5.0 (b)	0.0 ± 0.0 (a)	0.0 ± 0.0 (a)	16	0.016
Trichoptera	19.3 ± 12.3 (a)	1,771.6 ± 726.0 (b)	21.5 ± 13.7 (a)	20.2 ± 13.9 (a)	3510	0.001
Hydropsychidae	19.3 ± 12.3 (a)	1,450.1 ± 617.6 (b)	21.5 ± 13.7 (a)	20.2 ± 13.9 (a)	2962	0.001
Hydroptilidae	0.0 ± 0.0 (a)	319.3 ± 125.5 (b)	0.0 ± 0.0 (a)	0.0 ± 0.0 (a)	548	<0.001
Leptoceridae	0.0 ± 0.0	2.1 ± 2.1	0.0 ± 0.0	0.0 ± 0.0	0	0.454
Lepidoptera						
Crambidae	1.1 ± 1.1	0.0 ± 0.0	0.0 ± 0.0	0.0 ± 0.0	0	0.454
Coleoptera	3.2 ± 2.2 (a,b)	27.9 ± 11.9 (a)	8.6 ± 4.0 (a,b)	0.0 ± 0.0 (b)	151	0.043
Dryopidae	0.0 ± 0.0	4.3 ± 3.1	0.0 ± 0.0	0.0 ± 0.0	11	0.123
Dytiscidae	0.0 ± 0.0	2.1 ± 2.1	0.0 ± 0.0	0.0 ± 0.0	0	0.454
Elmidae	3.2 ± 2.2 (a,b)	21.5 ± 9.2 (b)	8.6 ± 4.0 (a,b)	0.0 ± 0.0 (a)	140	0.051
Diptera	77.4 ± 17.5 (a,b)	722.4 ± 155.2 (b)	44.1 ± 16.3 (a)	153.2 ± 79.9 (a,b)	1366	0.007
Ceratopogonidae	2.1 ± 1.3	0.0 ± 0.0	3.2 ± 1.3	64.5 ± 44.4	0	0.147
Chironomidae	75.2 ± 16.5 (a)	664.3 ± 133.1 (b)	39.8 ± 16.0 (a)	87.3 ± 41.5 (a)	1328	0.003
Empididae	0.0 ± 0.0 (a)	7.5 ± 3.2 (b)	0.0 ± 0.0 (a)	1.3 ± 1.3 (a,b)	27	0.034
Simuliidae	0.0 ± 0.0 (a)	50.5 ± 32.1 (b)	1.1 ± 1.1 (a)	0.0 ± 0.0 (a)	11	<0.001
Total density	545.4 ± 122.2 (a)	7,084.4 ± 2,265.4 (b)	271.8 ± 66.5 (a)	407.8 ± 104.5 (a)	13 792	<0.001
Total biomass	0.07 ± 0.02 (a)	1.90 ± 0.83 (b)	0.11 ± 0.03 (a)	0.13 ± 0.07 (a)	4.25	0.003

The natural riffle site was excluded from ANOVAs, but values for this site are provided for comparative purposes. Letters in parentheses represent site types that are significantly different based on Tukey's pairwise comparisons. Sites with the same letter are not significantly different at $p \leq 0.05$.
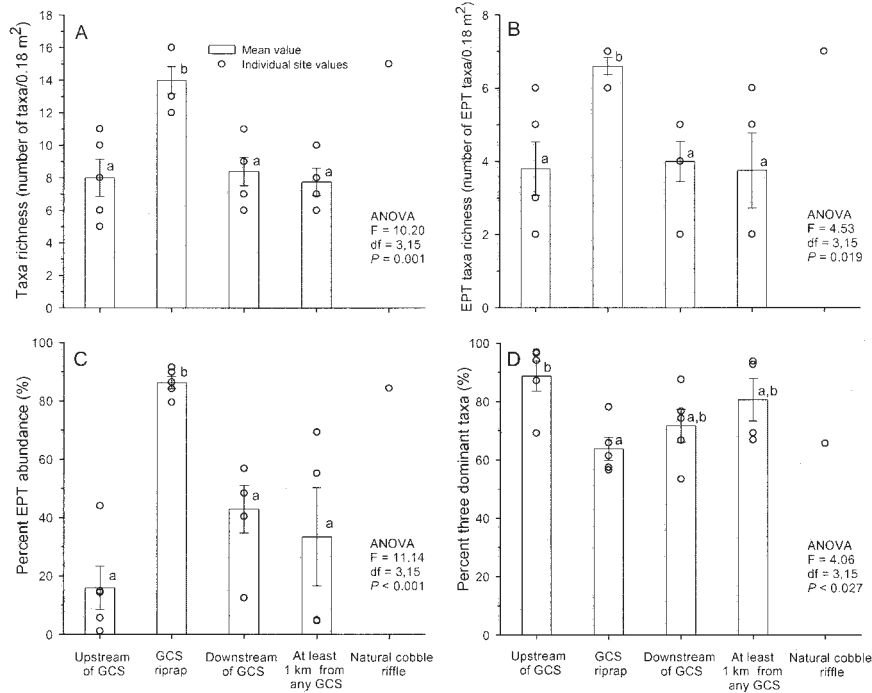
Figure 4. Mean (±SE) total taxa richness (A), EPT taxa richness (B), per cent EPT abundance (C) and per cent three most dominant taxa (D) immediately upstream from GCS, on GCS rip-rap, immediately downstream from GCS, and at sites >1 km from GCS with fine substrates. Values for macroinvertebrate variables at one natural cobble riffle site are provided for comparison purposes. Individual site values are indicated with open circles

sites downstream and >1 km from GCS (Tukey's test; $p \leq 0.015$; Table II). The dipteran Empididae was absent from sites upstream and downstream from GCS and was greater at GCS sites (Tukey's test; $p = 0.048$; Table II). The mayfly Ephemeridae responded negatively to coarse substrates, and was only recorded from sites 5 to 50 m upstream from GCS (Tukey's test; $p < 0.001$; Table II). Sphaeriidae clams were greater at sites at least 1 km from any GCS than at all other sites (Tukey's test; $p < 0.030$) and were absent at GCS riffles, the natural riffle and sites 5–50 m upstream from GCS (Table II). No Plecoptera taxa were collected from any site.

Because 28 separate ANOVA tests were conducted on this data set, the Bonferroni-adjusted critical p-value for each analysis was 0.0018 (i.e. $p = 0.05/28$). If this adjusted critical p-value was used, eight taxa were significantly different between site type, including four Ephemeroptera families, two Trichoptera families and one Diptera family that were significantly greater at GCS sites and one Ephemeroptera family that was significantly greater at sites 5–50 m upstream from GCS ($p \leq 0.001$; Table II).

Ordination of macroinvertebrate taxa densities with PCA indicated that two principal components (PC) explained 62% of variation in taxa densities across all sites (Figure 5). PC 1 was responsible for 49% of this variation. Taxa loading positively on PC 1 were most abundant at GCS and natural riffle sites while taxa loading negatively on PC 2 were characteristic of soft substrate sites upstream, downstream and at least 1 km from GCS (Table II). PCA Factor 1 scores were positively correlated with substrate score ($r = 0.70$, $p = 0.001$), flow ($r = 0.56$, $p = 0.011$), pH ($r = 0.52$, $p = 0.018$) and stream width ($r = 0.53$, $p = 0.016$), but were negatively correlated with depth ($r = -0.63$, $p = 0.003$). PCA Factor 2 scores were negatively correlated with substrate score ($r = -0.77$, $p < 0.001$) and flow ($r = -0.59$, $p = 0.007$) but were positively correlated with depth ($r = 0.61$, $p = 0.004$).

DISCUSSION

Our results indicate that GCS in Walnut Creek are significantly altering habitat conditions and associated benthic macroinvertebrate assemblages at GCS riffles and at sites directly upstream and downstream from GCS. In our

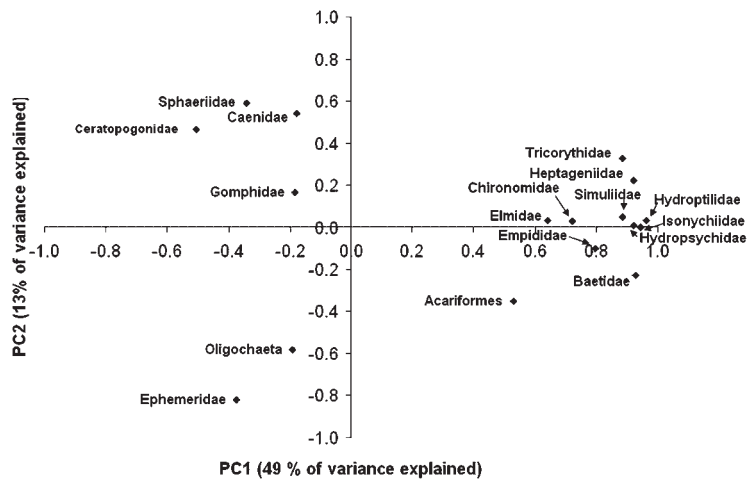## PCA of macroinvertebrate taxa densities

Figure 5. PCA plot based on densities of invertebrate taxa composing at least 1% of total macroinvertebrate density at any site. Taxa loading positively on principal component (PC) 1 were abundant at GCS rip-rap sites, whereas taxa loading negatively on PC 1 were more common at soft-substrate sites upstream or downstream of GCS or at least 1 km from any GCS

study, analyses of physicochemical variable data with ANOVA and correlations with PCA factor scores revealed that substrate particle size (i.e. substrate composition score), flow velocity and depth were heavily influenced by GCS and were likely driving variation in macroinvertebrate assemblages at sites on and near these structures. Although total macroinvertebrate density and biomass, taxa richness and many individual taxa densities were greatest on GCS rip-rap, low levels of benthic macroinvertebrate abundance and diversity occurred at sites immediately upstream and downstream of these areas. Although many taxa responded positively to conditions on GCS rip-rap (e.g. multiple mayfly and caddisfly families) and one taxa responded positively to conditions upstream from GCS (i.e. the burrowing mayfly Ephemeridae), other taxa avoided sites located on or nearby GCS. For example, Sphaeriidae clams were greatest at sights >1 km from any GCS and Empididae flies avoided sites upstream and downstream from GCS. Impoundment of water upstream from GCS and scouring of the streambed immediately downstream might have produced poor-quality habitat for these and other macroinvertebrates.

GCS rip-rap sampled in this study supported substantially greater abundance and diversity of benthic macroinvertebrates than sites dominated by fine substrate. In both lentic and lotic ecosystems, rocks, woody debris and a variety of other structurally complex substrates usually support higher macroinvertebrate abundance and diversity than finer substrates (Schmude *et al.*, 1998; Downes *et al.*, 2000; Voelz and McArthur, 2000). Relative to fine, soft substrates, both natural and artificial coarse substrates provide greater surface area for colonization and foraging and reduce likelihood of localized extinctions due to chance (Downes *et al.*, 2000). In streams, heterogeneous habitat created by coarse substrates also support taxa with diverse requirements, and provide refuges from predation and shear forces caused by floods (Hart and Finelli, 1999; Voelz and McArthur, 2000). Additionally, coarse substrates trap drifting coarse particulate organic matter (CPOM), increasing availability of food and habitat for macroinvertebrates (Bilby and Likens, 1980; Stewart *et al.*, 2003). GCS in western Iowa streams have been observed functioning as organic debris dams, trapping corn stalks, grasses, leaves, twigs and other CPOM at the upstream face of the metal dam, and between adjacent rip-rap particles. Debris dams increase food availability and refuge from floods for many invertebrate taxa, including Chironomidae (Palmer *et al.*, 1995; Palmer *et al.*, 1996). By retaining CPOM and increasing its residence time in streams, coarse substrates enable macroinvertebrates to colonize and process this material (Bilby and Likens, 1980; Palmer *et al.*, 1996; Stewart *et al.*, 2003). CPOM degraded into fine-particulate organic matter (FPOM) by shredding macroinvertebrates can be assimilated locally by filter-feeding and collector-gathering macroinvertebrates (Bilby and Likens, 1980).

Our results are consistent with previous studies finding overall positive effects of coarse substrate on macroinvertebrate assemblages (Beisel *et al.*, 1998; Downes *et al.*, 2000; Negishi and Richardson, 2003).

Specifically, macroinvertebrate taxa richness, total biomass and density and densities of several taxa were greater at GCS and a natural riffle site than at other sites dominated by fine substrates. Additionally, macroinvertebrate assemblage characteristics at our natural riffle site and GCS sites were very similar. These results support those from other studies demonstrating similarities in macroinvertebrate assemblages within artificial and natural coarse substrates (Ebrahimnezhad and Harper, 1997). Consequently, GCS and restored or created riffles have potential for providing coarse substrate habitat for benthic macroinvertebrates that has been reduced or eliminated by human land-use practices in agricultural and urbanized landscapes.

Responses of total macroinvertebrate density, biomass, taxa richness, EPT taxa richness and proportion of EPT abundance to substrate composition and related physicochemical factors were driven by individual taxon requirements and population responses. Isonychiidae mayflies and Hydropsychidae caddisflies were the most abundant taxa at coarse-substrate sites, and contributed greatly to positive effects of GCS riffles on total macroinvertebrate biomass and density. Potential mechanisms for responses of these filter-feeding organisms to GCS include abundant hard substrate for clinging and net construction, and high flow velocities and food concentrations (Merritt and Cummins, 1996; Stewart et al., 2003). Other mayflies (Baetidae, Heptageniidae, Tricorythidae) and caddisflies (Hydroptilidae) that responded favourably to GCS are classified as collector-scrapers and/or collector-gatherers and likely benefit from abundant food in the form of periphyton, bryophytes and other particulate matter that accumulate on and between rocks (Merritt and Cummins, 1996; Downes et al., 2000; Linhart et al., 2002). Similar to other studies, riffle beetles (Elmidae) responded positively to substrate size and abundance of rocky habitat (Bowles et al., 2003; Stewart et al., 2003). In a study of the biological effectiveness of artificial riffles in a channelised stream in the UK, Ebrahimnezhad and Harper (1997) found that Hydropsychidae, Simuliidae, Baetidae, Elmidae and Acariformes responded positively to habitat conditions provided by coarse substrates. Likewise, Harrison et al. (2004) found Hydropsychidae, Elmidae, and Baetidae positively associated with artificial riffles in lowland rivers of the United Kingdom. In our study, all of these taxa were present in GCS samples, and densities of these taxa responded positively to habitat conditions on GCS riffles. Greater proportion and greater taxa richness of Ephemeroptera and Trichoptera organisms at GCS sites than at remaining sites indicate an increased potential for breakdown of organic matter and transfer of nutrients and energy through aquatic food webs at GCS rip-rap. However, although filter-feeding, collector-scraping and collector-grazing organisms were abundant at GCS, shredding organisms (i.e. Plecoptera, Cambaridae, Crambidae, Leptoceridae) were absent or rare at all sites, possibly indicating a lack of allochthonous material in this stream with sparse riparian vegetation. Upstream of a GCS, water flow rate is reduced as it backs up against the vertical sheetpile dam that constitutes the facing of the structure. By creating a silt-dominated depositional zone, GCS indirectly creates habitat for burrowing organisms. However, depositional habitat is unfavourable for most lotic taxa, resulting in reduced taxa richness and a higher proportion of total density composed of a few numerically dominant taxa in silt substrate.

GCS, boulder weirs and artificial riffles are widely used in stream management and restoration projects to reduce streambank erosion and create habitat for organisms (Shields and Hoover, 1991; Gore et al., 1998; Harper et al., 1998). By stimulating increases in abundance and diversity of macroinvertebrates and other benthic organisms, these structures may improve ecosystem health in streams and rivers where coarse substrate and riffles are otherwise scarce. Macroinvertebrates increase nutrient cycling- and energy-transfer efficiencies through aquatic food webs by consuming and physically breaking down periphyton and detritus, and subsequently excreting nutrients and serving as prey for vertebrates (Wallace and Webster, 1996). Additionally, macroinvertebrates and their predators sequester and process nutrients and other pollutants derived from human land-use practices (Olson, 2004). Quantitative indicators of the capacity for addition of coarse substrates to improve stream ecosystem health were provided by Negishi and Richardson (2003), who found that total macroinvertebrate abundance and particulate organic matter storage increased 280% and 550%, respectively, after placement of boulder clusters in a British Columbia stream. Likewise, Laasonen et al. (1998) found that boulder dams used in restoration of channelized streams increased litter retention and invertebrate abundance.

In addition to enhanced organic matter processing, macroinvertebrates colonizing GCS are likely important prey for aquatic vertebrates such as fish, reptiles, amphibians and waterfowl (Ellis and Gowing, 1957; Waters, 1988; Wallace and Webster, 1996). Assuming a conversion of 1 g AFDM of invertebrates to 5.55 kcal of energy (Waters, 1988), our data show that GCS sites (with a maximum biomass of 4.78 g of macroinvertebrate $AFDM\,m^{-2}$) represent up to $26.5\,kcal\,m^{-2}$ potentially available to predators of macroinvertebrates during summer. Therefore,

compared to sites with fine substrates (having a mean biomass of 0.1 g AFDM m^{-2}), a 200-m^2 GCS (approximately 20 m long $\times$ 10 m wide) can potentially support over 900 g AFDM of macroinvertebrates more than an equivalent area of streambed covered by soft substrates. This additional macroinvertebrate biomass could potentially supply nearly 5000 kilocalories (900 g AFDW $\times$ 5.55 kcal g AFDM^{-1}) of additional food energy to invertivores. Because the streambed in our study area was dominated by soft substrate, with no more than 5–6% of non-GCS streambed containing gravel or larger particles, GCS are likely critical sources of food for local populations of fish and other vertebrates. If fish can exploit macroinvertebrates through foraging at GCS sites, or by consuming drifting invertebrates that were dislodged from these structures, individual fish body condition and total fish biomass and production may be enhanced by GCS (Ellis and Gowing, 1957; Waters, 1988).

Most GCS in western Iowa streams were originally constructed with downstream 4:1 back slopes. Although this design is economically practical due to requirement of relatively little rip-rap material, there is increasing concern and supporting preliminary evidence that fish movements are impeded by these steep structures (Larson *et al.*, 2004). Consequently, GCS constructed in the future will have more gradual slopes (10:1, 15:1 or 20:1), to increase likelihood of successful fish passage over these structures (Larson *et al.*, 2004). GCS sampled in our study had 20:1 downstream slopes. Computer simulation of artificial riffles in a western Tennessee, USA stream predicted that structures with a 20:1 slope would increase available macroinvertebrate habitat under all flow regimes, and this prediction correlated well to direct observation of macroinvertebrate response to constructed riffles (Gore *et al.*, 1998). Our results demonstrate that 20:1 sloped GCS in a predominantly soft-bottomed, agriculturally impacted stream not only support distinct macroinvertebrate assemblages, but also greater benthic macroinvertebrate abundance and diversity than can be found at almost any other location.

Despite positive responses of macroinvertebrates on GCS rip-rap and the potential for benefits to stream ecosystems, possible deleterious effects of in-stream structures are numerous and well documented (Pringle *et al.*, 2000; Santucci *et al.*, 2005). In western Iowa, potential unintended but harmful effects of GCS include degradation of habitat near structures, removal of riparian vegetation during construction, restriction of fish passage and reduction in lateral channel migration and natural fluvial processes. Boyken (1998) observed impoundment of water upstream from GCS in western Iowa extending up to a maximum of 1.07 km from structures. Although impounded water is an intentional goal of structure design in order to reduce streambank erosion around bridges located upstream from GCS, the impoundment of stream habitat may reduce significant portions of lotic habitat and lead to shifts in aquatic fauna assemblages in streams with multiple structures. These effects must be considered during construction of such structures, so that environmental damage is minimized, and ecological benefits maximized, while still enabling the structure to function as intended. Future designs for grade control and bank stabilization in western Iowa should consider use of soft-engineering techniques, including trees and large woody debris, as an alternative to the placement of rip-rap and metal dams in streams (Schmetterling *et al.*, 2001.) In addition, in-channel movement of rip-rap and structural failure of GCS is widespread in western Iowa (Voegele, 1997). Therefore, expected lifetime of in-stream structures such as GCS should be determined so that proper maintenance and structural repair can be planned for prior to initial construction (Thompson, 2002). In western Iowa, the placement of GCS in streams arose out of a necessity to reduce erosion and channel headcutting that originated due to widespread stream channelization and poor land-use practices. Although GCS may provide bank stability near bridges and reduce loss of farmland by erosion, improved watershed and riparian zone management is needed for overall effective stream rehabilitation. Because GCS are placed in streams of western Iowa solely for purposes of bank stability and erosion control, enhanced macroinvertebrate abundance and diversity on GCS rip-rap is an unintended effect of these structures. Several hundred GCS now occur in western Iowa streams, and more are being planned for construction. Understanding effects of GCS at both local and watershed scales is necessary for effective stream management and rehabilitation.

REFERENCES

APHA. 2005. *Standard Methods for the Examination of Water and Wastewater*, 21st edition. APHA: Washington, DC.

Bain MB, Finn JT, Booke HE. 1985. Quantifying stream substrate for habitat analysis studies. *North American Journal of Fisheries Management* **5**: 499–500. DOI: 10.1577/1548-8659(1985)<499:QSSFHA>2.0.CO;2.

Barbour MT, Gerritsen J, Snyder BD, Stribling JB. 1999. *Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates, and Fish,* 2nd edition. EPA/841-B-99-002. U.S. Environmental Protection Agency, Office of Water, Assessment and Watershed Protection Division: Washington, D.C.

Baumel CP, Morris LL, McVey MJ, Yang X. 1994. *Impacts of Degrading Western Iowa Streams on Private and Public Infrastructure Costs.* Agriculture and Home Economic Experiment Station, Iowa State University: Ames, Iowa.

Beisel JN, Usseglio-Polatera P, Thomas S, Moreteau JC. 1998. Stream community structure in relation to spatial variation: the influence of mesohabitat characteristics. *Hydrobiologia* **389**: 73–88. DOI: 10.1023/A:1003519429979.

Bilby RE, Likens GE. 1980. Importance of organic debris dams in the structure and function of stream ecosystems. *Ecology* **61**: 1107–1113.

Bowles DE, Barr CB, Stanford R. 2003. Habitat and phenology of the endangered riffle beetle Heterelmis comalensis and a coexisting species, Microcylloepus pusillus, (Coleoptera: Elmidae) at Comal Springs, Texas, USA. *Archiv Fur Hydrobiologie* **156**: 361–383.

Boyken MS. 1998. Hydrologic and hydraulic analyses of stream stabilization and grade control structures in western Iowa. Masters thesis, Iowa State University: Ames, Iowa.

Brookes A. 1988. *Channelized Rivers*. John Wiley & Sons: Chichester.

Bulkley RV. 1975. *Inventory of major stream alterations in Iowa, Completion Report: A study of the effects of stream channelization and bank stabilization on warm water sport fish in Iowa*. Subproject No. 1. U.S. Fish and Wildlife Service Contract No. 14-16-0008-745. Iowa Cooperative Fisheries Research Unit: Ames, Iowa.

Downes BJ, Lake PS, Schreiber ESG, Glaister A. 2000. Habitat structure, resources and diversity: the separate effects of surface roughness and macroalgae on stream invertebrates. *Oecologia* **123**: 569–581. DOI: 10.1007/PL00008862.

Ebrahimnezhad M, Harper DM. 1997. The biological effectiveness of artificial riffles in river rehabilitation. *Aquatic Conservation: Marine and Freshwater Ecosystems* **7**: 187–197. DOI: 10.1002/(SICI)1099-0755(199709)7:3<187::AID AQC219>3.0.CO;2.-B.

Ellis RJ, Gowing H. 1957. Relationship between food supply and condition of wild brown trout, Salmo trutta Linnaeus, in a Michigan stream. *Limnology and Oceanography* **2**: 299–308.

Gordon ND, McMahon TA, Finlayson BL. 1992. *Stream Hydrology: An Introduction for Ecologists*. John Wiley & Sons: Chichester.

Gore JA, Crawford DJ, Addison DS. 1998. An analysis of artificial riffles and enhancement of benthic community diversity by physical habitat simulation (PHABSIM) and direct observation. *Regulated Rivers: Research and Management* **14**: 69–77. DOI: 10.1002/(SICI)1099-1646(199801/02)14:1<69::AID.-RRR477>3.0.CO;2-D.

Gu R, Voegele DM, Lohnes RA. 1999. Evaluation of stream grade control structures in loess soil region. *Journal of Hydraulic Engineering* **125**: 882–885. DOI: 10.1061/(ASCE)0733-9429(1999)125:8(882).

Harper D, Ebrahimnezhad M, Climent I, Cot F. 1998. Artificial riffles in river rehabilitation: setting the goals and measuring the successes. *Aquatic Conservation: Marine and Freshwater Ecosystems* **8**: 5–16. DOI: 10.1002/(SICI)1099-0755(199801/02)8:1<5::AID.-AQC266>3.0.CO;2-G.

Harrison SSC, Pretty JL, Shepherd D, Hildrew AG, Smith C, Hey RD. 2004. The effect of instream rehabilitation structures on macro-invertebrates in lowland rivers. *Journal of Applied Ecology* **41**: 1140–1154. DOI:10.1111/j.0021-8901.2004.00958.x.

Hart DD, Finelli CM. 1999. Physical-biological coupling in streams: the pervasive effects of flow on benthic organisms. *Annual Review of Ecology and Systematics* **30**: 363–395. DOI:10.1146/annurev.ecolsys.30.1.363.

Heitke JD, Pierce CL, Gelwicks GT, Simmons GA, Siegwarth GL. 2006. Habitat, land use, and fish assemblage relationships in Iowa streams: preliminary assessment in an agricultural landscape. In *Influences of Landscape on Stream Habitat and Biological Communities*, Wang L, Hughes R (eds). American Fisheries Society, Symposium 48: 287–303. American Fisheries Society: Bethesda, Maryland.

Inoue M, Nunokawa M. 2002. Effects of longitudinal variations in stream habitat structure on fish abundance: an analysis based on subunit-scale habitat classification. *Freshwater Biology* **47**: 1594–1607. DOI:10.1046/j.1365-2427.2002.00898.x.

Iowa DNR Watershed Initiative. 2002. http://www.igsb.uiowa.edu/nrgislibx/watershed/watersheds.htm Retrieved 19 Aug 2007

Karr JR, Toth LA, Dudley DR. 1985. Fish communities of midwestern rivers: a history of degradation. *BioScience* **35**: 90–95.

Knudsen EE, Dilley SJ. 1987. Effects of riprap bank reinforcement on juvenile salmonids in four western Washington streams. *North American Journal of Fisheries Management* **7**: 351–356. DOI: 10.1577/15488659(1987)<351:EORBRO>2.0.CO;2.

Laasonen P, Muotka T, Kivijärvi I. 1998. Recovery of macroinvertebrate communities from stream habitat restoration. *Aquatic Conservation: Marine and Freshwater Ecosystems* **8**: 101–113. DOI: 10.1002/(SICI)1099-0755(199801/02)8:1<101::AID.-AQC251>3.0.CO;2-4.

Larson CJ, Atchison GJ, Menzel BW. 2004. *Annual progress report to the Iowa. Department of Natural Resources Fisheries Bureau for an evaluation of effects of weirs in Walnut Creek on fish abundance and movement*. Contract No: 03-8250-01. Iowa Department of Natural Resources: Des Moines, Iowa.

Linhart J, Vlčková S, Uvíra V. 2002. Bryophytes as a special mesohabitat for meiofauna in a rip-rapped channel. *River Research and Applications* **18**: 321–330. DOI: 10.1002/rra.671.

Merritt RW, Cummins KW. 1996. *An introduction to the aquatic insects of North America*, 3rd edition. Kendall/Hunt: Dubuque, Iowa.

Negishi JN, Richardson JS. 2003. Responses of organic matter and macroinvertebrates to placements of boulder clusters in a small stream of southwestern British Columbia, Canada. *Canadian Journal of Fisheries and Aquatic Sciences* **60**: 247–258. DOI: 0.1139/F03-013.

Olson RA. 2004. Components, processes, and design of created palustrine wetlands. In *Wetland and Riparian Areas of the Intermountain West: Ecology and Management*, McKinstry MC, Hubert WA, Anderson SH (eds). University of Texas Press: Austin, Texas; 185–215.

Omernik JM, Griffith GE, Pierson SM. 1993. *Ecoregions and Western Corn Belt Plains Subregions of Iowa*. U.S. Environmental Protection Agency, Environmental Research Laboratory: Corvallis, Oregon.

Palmer MA, Arensburger P, Botts PS, Hakenkamp CC, Reid JW. 1995. Disturbance and the community structure of stream invertebrates–patch-specific effects and the role of refugia. *Freshwater Biology* **34**: 343–356. DOI:10.1111/j.1365-2427.1995.tb00893.x.

Palmer MA, Arensburger P, Martin AP, Denman DW. 1996. Disturbance and patch-specific responses: the interactive effects of woody debris and floods on lotic invertebrates. *Oecologia* **105**: 247–257. DOI: 10.1007/BF00328554.

Peters RJ, Missildine BR, Low DL. 1998. Seasonal fish densities near river banks stabilized with various stabilization methods. First year report of the Flood Technical Assistance Project. U.S. Fish & Wildlife Service, North Pacific Coast Ecoregion, Western Washington Office, Aquatic Resources Division. Lacey, Washington.

Pringle CM, Freeman MC, Freeman BJ. 2000. Regional effects of hydrologic alterations on riverine macrobiota in the new world: tropical-temperate comparisons. *BioScience* **50**: 807–823. DOI: 10.1641/0006-3568(2000)050[0807:REOHAO]2.0.CO;2.

Prior JC. 1991. *Landforms of Iowa*. University of Iowa Press: Iowa City, Iowa.

Santucci VJ Jr, Gephard SR, Pescitelli SM. 2005. Effects of multiple low-head dams on fish, macroinvertebrates, habitat, and water quality in the Fox River, Illinois. *North American Journal of Fisheries Management* **25**: 975–992. DOI: 10.1577/M03-216.1.

Scheiner SM. 2001. MANOVA. In *Design and Analysis of Ecological Experiments*, 2nd edition, Scheiner SM, Gurevitch J (eds). Oxford University Press: 99–115.

Schmetterling DA, Clancy CG, Brandt TM. 2001. Effects of riprap bank reinforcement on stream salmonids in the western United States. *Fisheries* **26**(7): 6–13. DOI: 10.1577/1548-8446(2001)026,0006:EORBRO.2.0.CO;2.

Schmude KL, Jennings MJ, Otis KJ, Piette RR. 1998. Effects of habitat complexity on macroinvertebrate colonization of artificial substrates in north temperate lakes. *Journal of the North American Benthological Society* **17**: 73–80.

Shields FD, Hoover JJ. 1991. Effects of channel restabilization on habitat diversity, Twentymile Creek, Mississippi. *Regulated Rivers: Research and Management* **6**: 163–181.

Simon A, Darby SE. 2002. Effectiveness of grade-control structures in reducing erosion along incised river channels: the case of Hotophia Creek, Mississippi. *Geomorphology* **42**: 229–254. DOI:10.1016/SO169-555X(01)00088-5.

Stewart TW, Shumaker TL, Radzio TA. 2003. Linear and nonlinear effects of habitat structure on composition and abundance in the macroinvertebrate community of a large river. *American Midland Naturalist* **149**: 293–305. DOI: 10.1674/0003-0031(2003)149[0293:LANEOH]2.0.CO;2.

Thompson DM. 2002. Long-term effects of instream habitat-improvement structures on channel morphology along the Blackledge and Salmon Rivers, Connecticut, USA. *Environmental Management* **29**(1): 250–265. DOI: 10.1007/s00267-001-0069-0.

Thorp JH, Covich AP. 2001. *Ecology and Classification of North American Freshwater Invertebrates*, 2nd edition. Academic Press: San Diego, California.

Voegele DM. 1997. The evaluation of riprap and low-cost grade control structures in western Iowa. Master's thesis, Iowa State University, Ames, Iowa.

Voelz NJ, McArthur JV. 2000. An exploration of factors influencing lotic insect species richness. *Biodiversity and Conservation* **9**: 1543–1570. DOI: 10.1023/A:1008984802844.

Wallace JB, Webster JR. 1996. The role of macroinvertebrates in stream ecosystem function. *Annual Review of Entomology* **41**: 115–139. DOI:10.1146/annurev.en.41.010196.000555.

Waters TF. 1988. Fish production-benthos production relationships in trout streams. *Polskie Archiwum Hydrobiologii* **35**: 545–561.

Wilton T. 2004. *Biological assessment of Iowa's wadeable streams*. Final Report. Iowa Department of Natural Resources: Des Moines, Iowa.

Zar JH. 1999. *Biostatistical Analysis*, 4th edition. Prentice-Hall: Upper Saddle River, New Jersey.