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RESEARCH

Can Warmwater Streams Be Rehabilitated Using Watershed-Scale Standard Erosion Control Measures Alone?

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Abstract Degradation of warmwater streams in agricultural landscapes is a pervasive problem, and reports of restoration effectiveness based on monitoring data are rare. Described is the outcome of rehabilitation of two deeply incised, unstable sand-and-gravel-bed streams. Channel networks of both watersheds were treated using standard erosion control measures, and aquatic habitats within 1-km-long reaches of each stream were further treated by addition of instream structures and planting woody vegetation on banks (''habitat rehabilitation''). Fish and their habitats were sampled semiannually during 1–2 years before rehabilitation, 3–4 years after rehabilitation, and 10–11 years after rehabilitation. Reaches with only erosion control measures located upstream from the habitat measure reaches and in similar streams in adjacent watersheds were sampled concurrently. Sediment concentrations declined steeply throughout both watersheds, with means ≥40% lower during the post-rehabilitation period than before. Physical effects of habitat rehabilitation were persistent through time, with pool habitat availability much higher in rehabilitated reaches than elsewhere. Fish community structure responded with major shifts in relative species abundance: as pool habitats increased after rehabilitation, small-bodied generalists and opportunists declined as certain piscivores and larger-bodied species such as centrarchids and catostomids increased. Reaches without habitat rehabilitation were significantly shallower, and fish populations there were similar to the rehabilitated reaches prior to treatment. These findings are applicable to incised, warmwater streams draining agricultural watersheds similar to those we studied. Rehabilitation of warmwater stream ecosystems is possible with current knowledge, but a major shift in stream corridor management strategies will be needed to reverse ongoing degradation trends. Apparently, conventional channel erosion controls without instream habitat measures are ineffective tools for ecosystem restoration in incised, warmwater streams of the Southeastern U.S., even if applied at the watershed scale and accompanied by significant reductions in suspended sediment concentration.

Keywords River restoration · Fish · Monitoring · Restoration assessment · Ecosystem rehabilitation · Instream structures · Channel incision

Introduction

Streams in the southeastern United States support remarkably high levels of biodiversity, but many species are imperiled because of habitat and water quality degradation (Karr and others 2000, Warren and others 2000). Physical habitat degradation in these streams often triggers symptoms of ecosystem distress syndrome (Rapport and others 1985): reductions in the stability and diversity of aquatic ecosystems, elimination of the longer-lived, larger species, and a tendency to favor small, short-lived opportunistic species (Berkman and Rabeni 1987, Ebert and Filipek 1988,

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Meffe and Sheldon 1988, Menzel and others 1984, Paller 1994, Shields and others 1994, 1995a, Lamberti and Berg 1995, Rabeni and Smale 1995, Oscoz and others 2005). A conceptual framework for fish communities in small warmwater (*sensu* Winger 1981) streams proposed by Schlosser (1987a) and supported by subsequent research (e.g., Taylor and Warren 2001) offers insight into the mechanisms involved in these changes and possible pathways for rehabilitation (Shields and others 1998). Schlosser argued that fish community structure in warmwater streams is the expression of a complex interplay between abiotic and biotic factors, generally trending from highly variable communities in shallow headwaters to more stable communities in larger downstream reaches. Species richness and fish size increase while population density decreases along a physical gradient of hydrologic variability, habitat heterogeneity, and pool development. Reaches with flashy hydrology, spatially uniform conditions, and shallow depths support populations dominated by large numbers of small-bodied, rapidly reproducing cyprinids and are devoid of larger fishes. These assemblages are termed ‘‘colonizing’’ communities. Temporally unstable physical conditions in these reaches result in considerable variation in fish species richness, population density, and age structure (Adams and others 2004). As pool volume, temporal stability, and habitat heterogeneity increase, communities feature fewer, larger piscivores such as centrarchids, more pool-dwelling catostomids, and fewer small invertivores and omnivores because of increased predation and competition for refugia. This type of structure was referred to as ‘‘stable’’ by Schlosser (1987a).

Stream ecosystem integrity is governed by landscape-level controls on stream hydrology such as geology and land use that are reflected in stream depth, width, and water quality (Hynes 1975, Roth and others 1996, Allan and others 1997, Fitzpatrick and Giddings 1997, Schlosser 2002, Ward 1998, Fausch and others 2002, Allan 2004). Rehabilitation of incised streams (or restoration, *sensu* Society 2004) ideally would include restoring watershed hydrology. In the case of incised streams perturbed by shifts in the amount and rate of runoff, restoring watershed hydrology would imply shifting from a regime dominated by surface inflows and characterized by flashy hydrographs, high peaks, and extremely low base flows to one characterized by more moderate fluctuations, groundwater inflows, and higher base flow (Shields and Cooper 1994, Shields and others 1994, Doyle and Shields 1998). Although many factors control sediment transport, the nonlinear relationship between water and sediment discharge implies that lower flow peaks would result in lower sediment loads. Presumably, such hydrologic restoration would produce shifts in physical aquatic habitat that would lead to ecological recovery. Accordingly, some workers suggest that ecosystem rehabilitation through reach-scale manipulations (i.e., building instream structures) is ineffective and that an approach consisting of halting anthropogenic practices and press disturbances that prevent natural recovery, revegetating riparian zones, and restoring lateral and off-channel habitats is preferred (Beschta and others 1994, Thompson and Stull 2002, Pretty and others 2003, Thompson 2005, Harrison and others 2004, Thompson 2006). Others propose preservation of entire watersheds in more or less pristine condition (Harding and others 1998). Still others admit the joint importance of reach- and watershed-scale influences, but they emphasize the importance of processes acting at larger spatial scales (Roth and others 1996, Rabeni and Sowa 1996, Wissmar and Beschta 1998, Lepori and others 2005), and favor the restoration of functional processes (e.g., conditions leading to island formation) rather than forms (e.g., constructing islands) (Ward and others 2002). In fact, political, technical, and economic realities often preclude large-scale efforts to restore hydrologic or sediment regimes, and restoration workers must limit themselves to local or watershed-scale erosion control or habitat structures.

Additional information is needed to provide a scientific basis for restoration strategies (Kondolf 1995). However, even though stream restoration projects are numerous, with total funding estimated to exceed several billion dollars annually, monitoring efforts are sparse (Bernhardt and others 2005). Reports of long-term effects of restoration of lowland, warmwater streams are rare relative to evaluations of restorations in salmonid streams (Hill and Platts 1998, Schmetterling and Pierce 1999, Lyons and Courtney 1990). Projects that are monitored often reveal structural failures over the long- or short term (e.g., Shields and others 2003, Thompson and Stull 2002, Thompson 2005, but see Schmetterling and Pierce 1999), unexpected physical function (Thompson 2002a, Sear and Newson 2004), or unanticipated or undesired biological responses (Moerke and Lamberti 2003, Pretty and others 2003).

We developed and implemented a stream habitat rehabilitation project in 1992 and another in 1993 that consisted of modifying existing stone erosion control structures and planting willow cuttings in 1-km reaches near the mouths of two watersheds (Shields and others 1995c, 1995d). We sampled the rehabilitated reaches and reaches affected only by standard erosion controls before and after habitat rehabilitation using a ‘‘pulsed

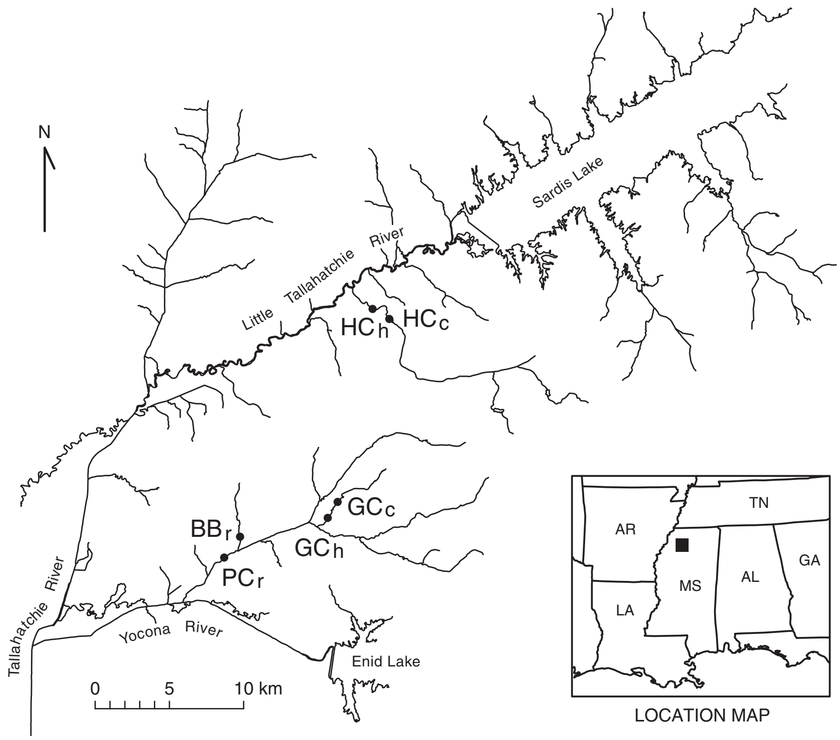

Fig. 1 Location of study sites

monitoring strategy'' (Bryant 1995). This study allowed comparison of the relative effects of two stream restoration approaches: watershed-wide erosion controls and watershed-wide controls plus instream habitat rehabilitation. Our hypothesis was that a shift in fish community structure from colonizing to stable could be triggered by the latter but not the former.

Sites

Study reaches were located within the Hotophia Creek (HC) and Peters Creek (PC) watersheds in northwestern Mississippi within the hilly region of the upper Yazoo River watershed (Fig. 1). Watershed centroids are separated by only about 15 km, and thus the areas experience nearly identical weather. Topography, soils, land use, and general stream characteristics are also similar (Table 1), producing similar hydrologic regimes. Relatively flat cultivated floodplains flank the channels, with steep forested or grassed hillslopes rising to the watershed divides. Soils (primarily loess and loess-derived alluvium) are highly erodible, and channels are extremely unstable, producing average annual sediment yield about twice the national average (~1000 t km^{-2}) for watersheds of this size (Shields and others 1995a). Because of their inclusion in federally funded erosion control programs (e.g., Hudson 1997), the geomorphology of both watersheds has been described in several publications (e.g., Whitten and Patrick 1981, Little and others 1982, Simon and Darby 2002). European settlement (1835–1850) was followed by deforestation, cultivation of hillsides, rapid erosion and gully development, and up to 2 m of valley sedimentation. In order to drain valley bottoms for agriculture, all perennial channels within both watersheds were channelized at least once between approximately 1880 and 1965, and fluvial response between about 1965 and the present is consistent with conceptual models of incised channel evolution (Schumm and others 1984, Simon 1989a). Extensive erosion control works were constructed throughout both watersheds, most between 1986 and 1996 (Tables 1 and 2).

Habitat rehabilitation activities were performed on 1-km reaches of HC and Goodwin Creek (GC), a PC tributary, in early 1992 and early 1993, respectively (Shields and others 1995c and 1995d). These reaches are referred to as ''habitat reaches'' below and shown on Figures 1 and 2 as HC_h and GC_h. The conceptual foundation for the rehabilitation project was linked to prevailing geomorphic processes that were actively transforming channel morphology during the course of the study (Shields and others 1992 and 1998). Prior to rehabilitation, aquatic habitats were typical of incised channels in the region and somewhat similar to degraded streams associated with agricultural watersheds

Table 1 Description of study watersheds and sampled reaches

	Hotophia Creek HC	Goodwin Creek GC	Bobo Bayou BB	Peters Creek PC
Drainage area, km^2	91	21	16	205
Land use (row crops/idle or pasture/forest), %	8/40/52	9/61/30	13/54/33	11/53/36
Slope	0.0011	0.0016	0.0025	0.0009
Sinuosity	1.4	1.3	1.12	1.1
Bed material	Sand	Sand and gravel	Sand and gravel	Sand and gravel
Bank height, m	3–7	4–5	4–5	2–6
Channel width,[a] m	40–60	20–70	25–30	55–85
Instream structures in treated reaches	Stone toe protection,[b] stone spurs	Grade control structure immediately downstream, stone toe, stone weirs	n/a	n/a
Instream structures in untreated reach	Stone toe protection along one bank	Stone toe protection along one bank	n/a	Stone toe protection along one bank
Watershed control measures	Grade controls, bank protection, drop pipes,[c] small reservoirs, land treatment	Grade controls, bank protection, drop pipes, small reservoirs, land treatment	Drop pipes	Grade controls, bank protection, drop pipes, small reservoirs, land treatment
Sampling dates (Spring and Fall)				
Physical habitat and fish (electrofishing)	1991–95, 2002	1991–95, 2003	1991–95, 2003	1991–95, 2002
Fish (concussive detonation cord)	2003	2004	2004	2003

[a] Average distance between top banks

[b] Windrow of quarry stone placed along and parallel to the toe of steep, eroding banks. See Shields and others (2000)

[c] Earthen embankments fitted with L-shaped corrugated metal pipes to pass runoff and control riparian gully erosion. See Shields and others (2002)

Table 2 Partial list of federally funded erosion control measures placed within study watersheds after Shields and Knight (2003)

Type of structure	Hotophia Creek (HC)	Peters Creek (PC)
Bank stabilization (km)	17.9	15.2
Riser (drop inlet) pipes	148	77
Low drop grade control	10	17
High drop grade control	3	0
Small reservoirs	5	4
Debris basins	123	100
Estimated total cost ($\$10^6$)	9.8	11.3

in the northern Midwest (Talmage and others 2002): hydrology was flashy, pool habitats were rare and temporally unstable, substrate was dominated by shifting sands, and woody debris was scarce. Macroscale bedforms (point bars, midchannel bars and riffles) tended to be temporally unstable and dominated by medium sands; deposits of gravel without sand-fill interstices were absent. Eroding banks were common, and riparian vegetation was sparse (Shields and others 1994). Grade control structures were constructed at the lower ends of GC_h and HC_h in the early 1980s, and stone bank protection structures were constructed in 1989–1991 and 1986–1990 in GC_h and HC_h, respectively. Baseflow channels were extremely shallow within much wider high flow channels enlarged by erosion, and the bank protection structures did not protrude far enough into the base flow channels to provide cover, scour pools, or overall habitat heterogeneity.

All study streams except for Bobo Bayou (BB) (Fig. 1) were monitored for water quality as described by Lizotte and others (2001, 2002, 2003a, 2003b). Observations were summarized by others (Cooper and Knight 1991, Slack 1992, Rebich 1993, Maul and others 2004): water quality conditions were generally acceptable in light of criteria for aquatic life with the exception of transient events that produced elevated suspended solids concentrations (~10^3 mg L^{-1}) and fecal coliform counts. Specific conductance levels tend to be rather low (generally <100 μs/cm), whereas pH values were near neutral, and dissolved oxygen (DO) values were near saturation. Fish populations were typical of those found in many streams damaged by incision (Shields and others 1994).

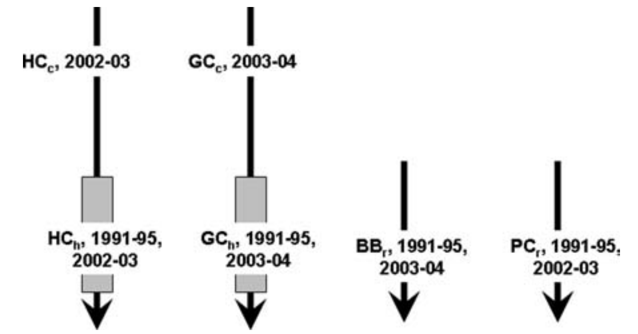

Fig. 2 Schematic of study plan. Habitat reaches are shown in gray. Years are sampling dates. Arrows show direction of streamflow. HC_c, Hotophia Creek control reaches; GC_c, Goodwin Creek control reaches; HC_h, Hotophia Creek habitat reach; GC_h, Goodwin Creek habitat reach; BB_r, Bobo Bayou reference reaches; PC_r Peters Creek reference reaches

The primary objective of rehabilitation was to increase the availability of stable pool habitat at base flow, thus providing refugia during periods of extreme hydrologic variation (Taylor and Warren 2001) and shifting the fish community toward the ‘‘stable’’ mode (Schlosser 1987a, Shields and others 1998). For rehabilitation, the existing stone spur dikes (deflectors) were extended riverward to provide stony substrate and trigger formation of stable pool habitats similar to those found associated with large wood and beaver dams in the narrower, sinuous, lightly impacted streams in the region that have not undergone incision (Shields and others 1994), and, we infer from aerial photographs, in these streams prior to about 1950. Spur dike extensions resulted in longer spurs in HC_h, and low weirs in GC_h (Fig. 3). Rehabilitation work required addition of less than 18% more stone than was previously placed in the same reaches for standard bank stabilization. Willow posts were planted in sandbars and cohesive banks near the structures in order to accelerate channel evolution and increase shade, riparian cover, and contributions of wood and other detritus to the stream.

For comparison with habitat reaches HC_h and GC_h (watershed scale erosion control measures + instream habitat rehabilitation), 1-km ‘‘reference’’ reaches (PC_r and BB_r, Fig. 1 and 2), which were influenced by watershed scale measures only, were sampled concurrently with habitat reaches. The reference reaches were selected because of their similarity to the pre-rehabilitation state of HC_h and GC_h in terms of geomorphology, physical habitat, and watershed-scale erosion controls. For additional confirmation of our findings regarding the effects of reach-scale habitat rehabilitation, 1-km ‘‘control’’ reaches immediately upstream from the habitat reaches (HC_c and GC_c, Fig. 1 and 2) were sampled concurrently with the habitat reaches 10 and 11 years after rehabilitation. Control reaches obviously experienced nearly identical water quality and hydrology as the habitat reaches downstream, and they were influenced by standard types of erosion controls (Table 2), but not instream habitat rehabilitation measures.

Methods

Stage and discharge were recorded by the U.S. Geological Survey at HC (station 07273100) and PC (station 07275530) during water years 1987–2001 and at GC by our laboratory during water years 1988–2002 (http://www.ars.usda.gov/Business/docs.htm?docid=5120, accessed February 16, 2007). Daily mean suspended-sediment concentration and load were recorded during water years 1988–1997 for HC and PC and during water years 1987–2002 for GC. The record of daily mean suspended sediment concentration for HC and PC was extended through water year 2001 using regression formulas and sediment rating curves as described by Shields and Knight (2003).

In each sampled reach, four 100-m long zones spaced ~200 m apart were sampled. Water depth was measured with a wading rod at five regularly spaced grid points along five transects placed at uniform intervals within each zone. Bed material type was classified visually at each point where depth was measured. Thus, water depth and bed type were sampled at 100 points (5 points $\times$ 5 transects $\times$ 4 zones) within each reach. Flow width was measured at each transect, and visual estimates were made regarding the size of each woody debris formation (Barbour and others 1999), the dominant type and size of bank vegetation (classified as bare soil or stone, weeds and herbaceous vegetation, woody vegetation <5 m high, or woody vegetation [trees] >5 m high) visible from the channel for 5 m on either side of an imaginary line that coincided with each transect, and percent canopy (portion of sky intercepted by vegetation when viewed by an observer standing in the middle of the base flow channel). Discharge was measured on each physical habitat sampling date at the downstream end of each study reach using a wading rod, tape, and an electromagnetic current meter. Current velocity was measured at 0.6 times depth at a number of verticals across a uniform cross section, and integrated with cross-sectional area to compute stream discharge. Flow dependency of water depth and wetted width was explored using scatter plots and by computing correlation coefficients. Mean values of width and depth for each sampling zone were

Fig. 3 Habitat rehabilitation measures in (a) Goodwin Creek and (b) Hotophia Creek. (**a**) Goodwin Creek reach GC_h. Existing stone erosion control spurs have been extended across the baseflow channel to create a v-shaped weir. Dormant willow posts have been planted in sandy deposits downstream from weir (Shields and others 1995d). (**b**) Hotophia Creek reach HC_h. Existing stone spurs are being extended about one third of the way across the base flow channel. A stone toe has been placed on the opposite bank, and dormant willow posts were planted shortly thereafter (Shields and others 1995c)

plotted against discharge. Separate plots were prepared for the periods before and after habitat rehabilitation for zones within the habitat reaches.

Fish were collected using a backpack-mounted electroshocker. Within each 1-km reach, the same four 100-m-long zones sampled for physical variables were fished for ~10 minutes of electric field application concurrently with physical habitat data collection in 1991–1995 and in 2002 or 2003 (Table 1). In 1993 and 2002, deeper regions of GC_h and HC_h, respectively, were sampled by placing the electroshocker in a small boat operated by two people assisted by others who waded in shallower regions nearby. After collection, fishes were identified, measured for total length, and weighed. For analysis, each collection was characterized by the number of fish, catch per unit effort (fish/min of electrical field application), the number of species, the mean mass of fish, percent of catch biomass, and percent of catch numbers as centrarchids, and as cyprinids. Larval fish were not included.

Electrofishing sometimes produces biased results in deeper habitats (Shields and others 2000). To augment long-term electrofishing collections, 25-m segments of each study reach were re-sampled 11 years after rehabilitation (2003 and 2004) using concussive sampling (Metzger and Shafland 1986). All six study reaches (Figures 1 and 2) were sampled at least twice with concussive sampling, and a total of 13, 25-m segments of the 100-m zones sampled by electrofishing were sampled. Explosive charges were composed of commercially available detonation cord of 10.63 PETN/m and Number 8 electric blasting caps secured by appropriately placed weights. Prior to concussive sampling, block nets were placed simultaneously above and below the sampling reach.

Comparison of pooled spring and fall electrofishing collections with concussive samples taken from the same zones the year before concussive sampling verified that the latter was more effective in sampling sites with mean depths >50 cm for larger individuals. However, depths this great were limited to only two sampling zones. Therefore, a one-way repeated-measures randomized analysis of variance (ANOVA) of the fish data based on sampling method (electrofishing or explosive) was conducted. F-values generated by one-way ANOVA of the observed data were compared to the distributions of F obtained by resampling the data for 1000 trials using a Monte Carlo routine. F-values based on observed data that were large enough to produce $p < 0.05$ and that were exceeded by random assignment only 5% of the time were accepted as statistically significant. There were no significant differences in numbers, biomass, or number of species per sample; mean fish size or percent biomass as centrarchids or cyprinids taken with the two methods. We therefore pooled these descriptors of the electrofishing and concussive samples for further analysis. However, there were differences in relative abundance of cyprinids and centrarchids, with electrofishing yielding higher values for percent catch as cyprinids by number and lower values for percent centrarchids than concussive sampling.

Physical habitat conditions in each of the 100-m sampling zones on each electrofishing sampling date were described by tabulating the average water depth and width, the decimal fraction of sampled points with bed covered by gravel, the large wood density, canopy coverage, and fraction of the bankline classified as supporting woody vegetation. Fish collections from each sampling zone and date were represented in the data set by total numbers, number of species, catch biomass, percent of numerical and biomass catch as cyprinids and as centrarchids, percent of biomass catch as catostomids, catch per unit of effort (electrofishing samples only), and mean fish size. Therefore, our basic data set was a matrix with rows representing sampling events at 100-m (25-m for concussive samples) zones on specific dates. Columns represented physical (mean depth, etc.) or biological (catch biomass, etc.) attributes measured on that date in that zone. No physical data were collected concurrently with concussive samples.

For analysis of rehabilitation effects, semiannual fish and physical habitat data from each stream were assigned to three groups: pre-rehabilitation (1 to 2 years), short term (up to 4 years after rehabilitation), and long term (10–11 years after rehabilitation). Two sets of data analyses were conducted. The first consisted of ANOVA of *within-reach temporal changes* across the three periods mentioned above using data from habitat and reference reaches. F-values generated by one-way ANOVA of the observed data were compared to the distributions of F obtained by randomly assigning data to the pre-rehabilitation, short-term, or long-term periods for 1000 trials using a Monte Carlo routine. F-values for which $p < 0.05$ and that were exceeded by random assignment only 5% of the time were accepted as statistically significant. The second set of data analyses consisted of one-way randomized ANOVA of *spatial differences between the reference and control reaches* 10–11 years after rehabilitation. Procedures and significance levels were the same as used for the first set of analyses.

Faunal similarity of fish collections and associated habitat characteristics were explored using nonmetric multidimensional scaling (NMS) of species abundances. NMS analysis was based on a matrix representing the abundance of 41 species in 219 collections (a collection was a group of fish captured from a single 100-m [electrofishing] or 25-m [concussive] sample site on a single date). Species represented by a single capture were omitted from the matrix. A distance matrix was derived from the matrix of species abundances containing the Sorensen (Bray-Curtis) distances between collections (McCune and others 2002). A matrix of key physical habitat variables was correlated with the axes derived from a second NMS analysis using concurrent fish and physical habitat collections (41 species in 184 collections) to explore relationships between habitat and faunal response. The matrix of species abundances was smaller in the second case because physical data were missing or incomplete for some of the fish collections. In both cases, multiple runs (stepping down in dimensionality from six to one axis) with real (40 runs) and randomized data (50 runs) were used to determine the dimensionality of the best solution (that which produced the lowest final stress from a real run) (McCune and Mefford 1999).

Results

Physical

In both HC and PC, suspended-sediment discharge fell sharply in the years after rehabilitation, due at least in part to lower flows (Table 3). Suspended sediment loads during water years 1993–2001 were only 14% to 40% of pre-rehabilitation (1988–1991) loads. However, measured discharges for habitat and fish sampling dates varied little for each stream across the study. Because samples were always collected at baseflow within a narrow window of dates in Spring and Fall, instantaneous measured discharges varied little from sample to sample. Coefficients of variation for measured discharges were 0.99, 0.28, 0.59, and 0.75 for BB, PC, GC, and HC, respectively. Mean values for wetted width and depth for each sampling zone and each date were weakly correlated (or uncorrelated) with discharge, with discharge typically explaining less than 30% of the variation in the width and depth. Therefore, mean values of width and depth were not corrected for flow variations prior to analysis.

Significant changes in physical habitats occurred within the habitat reaches (Fig. 4A), with average depth increasing in both streams, and increasing width associated with the weirs in GC_h. Consistent temporal changes in reference reaches were limited to slight increases in mean water depth and width in BB_r because of migration of a headcut into the lower end of the reach. During the 4 years immediately after rehabilitation, HC_h mean flow depth decreased slightly, although the depth and size of scour holes associated with the stone spurs more than doubled (Shields and others 1995b). However, after 10 years, HC_h mean depth had almost tripled, and was about three times larger than mean depth in reference and upstream control reaches (Fig. 4A and B). GC_h was also significantly ($p < 0.0005$) deeper than the upstream control GC_c. Substrate composition remained stable at all sites except HC_h, where gravel became more common ($p < 0.0001$), increasing from nearly absent prior to rehabilitation to about 11% coverage of the bed. HC_h also displayed an order of magnitude greater ($p < 0.001$) amount of instream large wood over the long term, reflecting revegetation of banks and associated improved wood retention. Even though ~70% of the

Table 3 Average of mean annual precipitation and mean-daily water and sediment discharge before (water years 1988–1991) and after (water years 1993–2001) habitat rehabilitation

Stream	Variable	Means		
		Before	After	Change(%)
GC	Mean annual precipitation (mm)	1378	836	–39%
BB		nd	nd	
GC	Mean-daily water discharge (m^3/s)	0.44	0.26	–41%
PC		4.6	2.8	–39%
HC		2.1	1.2	–43%
BB		nd	nd	
GC	Mean-daily sediment load (metric ton)	49	20	–60%
PC		823	235	–71%
HC		304	44	–86%

GC Goodwin Creek, BB Bobo Bayou, PC Peters Creek, HC Hotophia Creek, nd not determined

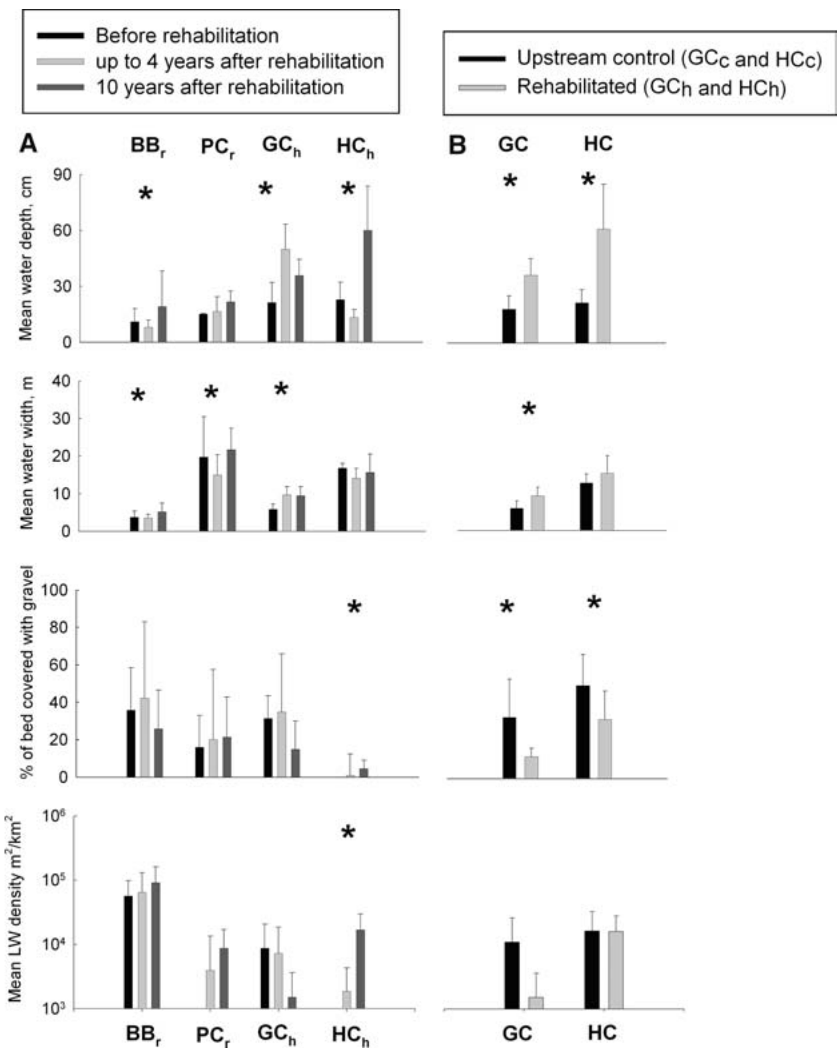

Fig. 4 Key physical aquatic habitat variables in (**A**) reference and rehabilitated habitat reaches before, shortly after, and a decade after rehabilitation, and (**B**) in rehabilitated habitat reaches and control reaches immediately upstream a decade after rehabilitation. LW = large wood. Error bars are +1 standard deviation of reach means. Asterisks indicate statistically significant differences ($p < 0.05$, see text for analysis details)

planted willow posts died (Shields and others 1995b), vegetation invaded stone revetments and sandy berms, consistent with the conceptual incised channel evolution model (Simon 1989a).

Ten years after rehabilitation, the habitat reaches were two to three times as deep and about half as gravelly as the control reaches immediately upstream (Fig. 4B). GC_h (stone weirs and toe protection) was 56% wider ($p < 0.01$) and had about half as much ($p = 0.03$) woody riparian cover than GC_c, which had only stone toe protection along concave banks. Instream large wood levels were similar in habitat and upstream reaches. Woody species failed to colonize banks of GC_h, likely because of infertile soils and competition from the exotic vine, *Pueraria lobata* (kudzu). Accordingly, large wood (LW) loadings remained depressed in GC_h relative to upstream, but differences were not statistically significant ($p = 0.28$).

Fish

Fish sampling collected 35,530 individuals of 46 species (Table A1). Although they were dominated by small cyprinids, reference reaches BB_r and PC_r yielded 32 and 37 species, respectively, over the course of the study. These values compare with 33 and 22 species captured from HC_h and GC_h, respectively, after rehabilitation. HC_h and GC_h fish populations responded differently to habitat rehabilitation (Fig. 5A). Fish numbers fell at GC_h after installation of weirs and attendant formation of deep pool habitats, whereas mean fish size increased slightly ($p = 0.07$). Addition of structure and pool habitats in HC_h was followed by a tripling of the mean number of fish per sample ($p < 0.05$) and a doubling of the mean number species per sample ($p < 0.001$) over the short term, but probably not over the long term. Mean size of fish captured at HC_h increased dramatically, with mean lengths for the pre-rehabilitation, short-term, and long-term periods at 7.0, 49, and 51 g, respectively, but because of large variances in the data, these changes were not significant ($p = 0.25$) (Fig. 5A).

On the other hand, HC_h and GC_h experienced similar, significant shifts in fish species composition. Assemblages in both reaches shifted away from small cyprinids typical of shallow, incised streams toward larger-bodied fishes, and these shifts were even more pronounced over the long term (Fig. 5). For example, pre-rehabilitation samples from habitat reaches comprised an average of 62% cyprinids by number and 21% by biomass, but samples collected a decade after restoration were comprised of only 25% cyprinids by number and 3% by biomass. These shifts made the habitat reaches less similar to the shallow, unstable BB_r, where cyprinids comprised 82% of numbers and 67% of sample biomass over the course of the study (Fig. 5A). As cyprinids declined in GC_h centrarchids became more dominant, increasing from a pre-rehabilitation average of 44% up to 86% of biomass 10 years afterward. At HC_h, catostomids comprised 11%, 62%, and 73% of pre-rehabilitation, short-term, and long-term total fish catch biomass, respectively. However, the collection means for catostomid biomass were 8%, 27%, and 16%, for pre-rehabilitation, short-term, and long-term periods, respectively, and these were not significantly different ($p = 0.23$). Trends in the habitat reaches are especially clear when catch biomass of major family groups was plotted against time, with cyprinid biomass declining in both streams while centrarchids increase and catostomids fluctuate (Fig. 6). The effects of pool development associated with habitat rehabilitation on family composition are also evident when collections from the habitat reaches are compared with those from control reaches just upstream (Fig. 5B and Table 4). Relative to the control reaches, cyprinids were far less dominant in the habitat reaches and centrarchids were more dominant. Concussive sampling results suggest that communities in habitat reaches were made up of fewer but larger fishes of slightly fewer species relative to control reaches.

Fish and Habitat

A matrix representing the abundance of 41 fish species (species represented by a single capture were omitted) in 219 collections produced two axes in NMS that together explained 70% of the variance in the original distance matrix. The abundances of several small-bodied species tolerant of extremely shallow flows over shifting sand beds, *Semotilus atromaculatus, Etheostoma artesiae, Pimephalus notatus*, and *Luxilus chrysocephalus*, were negatively correlated ($r < -0.34$) with both axes. Abundances of the larger-bodied centrarchids, *Lepomis megalotis* and *L. cyanellus*, were negatively correlated with Axis I and positively correlated with Axis II. Long-term post-rehabilitation collections from habitat reaches GC_h and HC_h received NMS Axis II scores that were significantly higher than for pre-rehabilitation collections (randomized one-way ANOVA, $p < 0.008$). HC_h also received significantly lower scores on Axis I ($p = 0.001$) after rehabilitation.

The second NMS (using only the 184 fish collections for which reliable values for sampling zone habitat variables were available) produced three axes that together explained 78% of the variance in the original distance matrix, and the first two axes explained 61% of the variance. The point cloud was rotated around its centroid to maximize the correlation of mean water depth with Axis I (McCune and Mefford 1999) (Table 5). After this rotation, the abundance of the pool-dwelling centrarchid *L. megalotis* was positively correlated ($r > 0.39$) with both axes (Table 6 and Fig. 7). Abundances of the small opportunists,

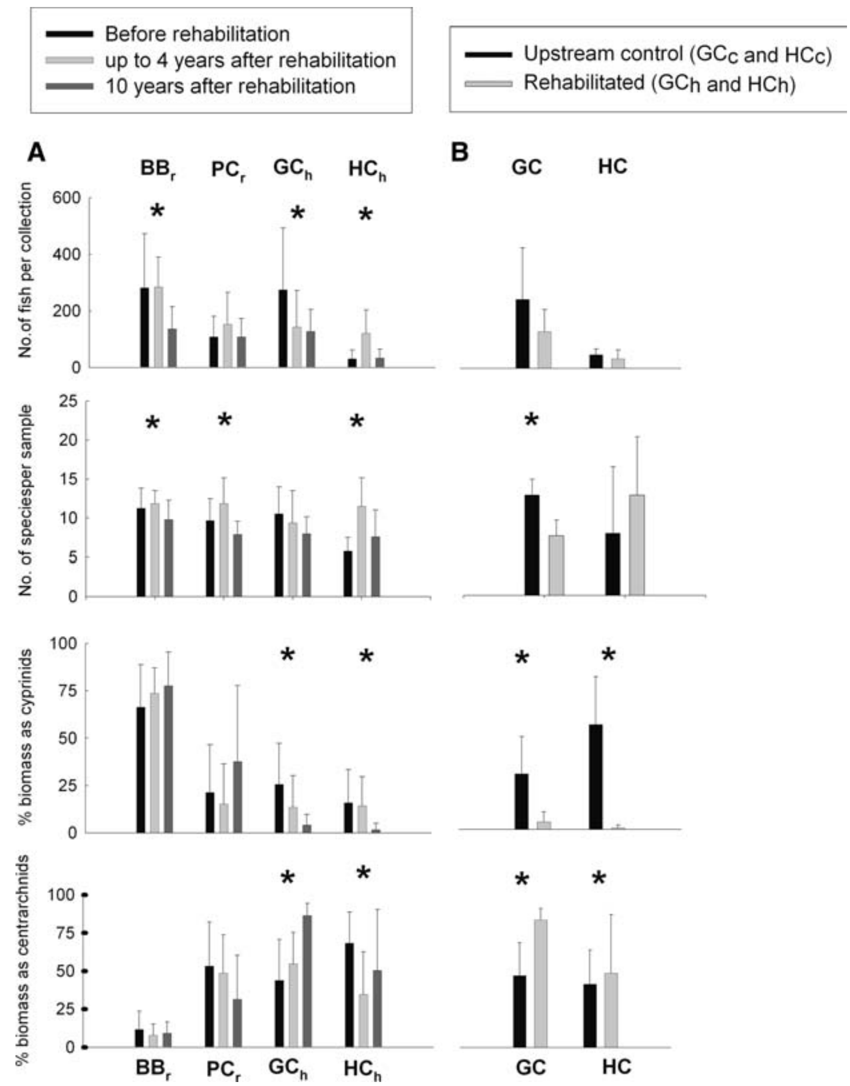

Fig. 5 Mean characteristics of fish populations in (**A**) reference and rehabilitated habitat reaches before, shortly after, and a decade after rehabilitation; and (**B**) in rehabilitated habitat reaches and control reaches immediately upstream a decade after rehabilitation. Error bars are + 1 standard deviation of reach means. Asterisks indicate statistically significant differences ($p < 0.05$, see text for analysis details)

Notropis rafinesquei and *Etheostoma artesiae* were negatively correlated with Axis I ($r < -0.52$). Mean water depth and width were positively correlated with Axis I (Table 5), while the density of large wood and gravel coverage of the bed were negatively related to Axis I. Abundances of the topminnow, *Fundulus olivaceus*, and the small cyprinids, *L. chrysocephalus* and *P. notatus* were positively correlated ($r > 0.44$) with Axis II. Gravel coverage was positively correlated with Axis II, whereas water width was negatively correlated ($r < -0.31$). When points representing each collection are plotted in the plane formed by Axis I and II, collections from habitat reaches clearly are displaced toward the lower right region, which is associated with deeper, wider habitats (Fig. 7). Accordingly, the pool-dwelling perch, *Aphredoderus sayanus*, the piscivorous gars, *Lepisosteus oculatus* and *Lepisosteus osseus*, and the catostomid, *Moxostoma poecilurum* also plot in this portion of the plane.

Discussion

Stream and river restoration projects are becoming increasingly common, with annual expenditures on the order of $1 billion for small and midsize projects in the United States (Bernhardt and others 2005). Stream restoration projects, particularly those targeted at ecological rehabilitation, are economically valuable (Collins and others 2005). However, major gaps exist in the science base needed to plan and design new restorations (Wohl and others 2005). Less than 10% of stream restoration projects examined in a large national survey included assessment or evaluation (Bernhardt and others 2005). The work described here represents a substantive effort in monitoring and assessment of restoration practice. Although our findings are strictly applicable to streams and watersheds similar to those we studied, they partially address the need identified by Bernhardt and others (2005).

Table 4 Comparison of means for key descriptors of fish collections 10 yr after rehabilitation[a]

Characteristic	Concussive sampling		Electrofishing	
	Habitat rehabilitation	Reference and control	Habitat rehabilitation	Reference and control
No. of fish per sample	85	212	83	118
Biomass per sample (kg)	3.8	0.89	0.9	0.9
No. of species per sample	9.0	9.1	7.6	10.3
Mean size of fish (g)	63.5	4.3	16.8	13.9
% of numbers as cyprinids	43	83	12	55
% of biomass as cyprinids	24	70	38	69
% of numbers as centrarchids	41	13	70	29
% of biomass as centrarchids	46	23	50	22

[a] Electrofishing samples were collected from the 100-m reaches in Spring and Fall (n = 26), whereas concussive sampling occurred in 25-m segments of the same reaches the following spring or summer (n = 13)

Table 5 Pearson and Kendall correlations between ordination axes and physical habitat variables

Habitat variable	DCA I			DCA II		
	r	r^2	τ	r	r^2	τ
Mean water depth	0.42	0.17	0.41	–0.02	0.00	–0.01
Large wood density	–0.51	0.26	–0.37	0.09	0.01	0.11
Fraction of bed covered with gravel	–0.39	0.15	–0.24	0.31	0.10	0.24
Mean water width	0.56	0.32	0.44	–0.30	0.09	–0.27
Fraction of bank covered with woody vegetation	–0.08	0.01	–0.01	0.10	0.01	0.02

DCA detrended correspondence analysis

Table 6 Pearson and Kendall correlations between ordination axes and fish species for which $r^2 > 0.10$

Fish species	NMS I			NMS II		
	r	r^2	τ	r	r^2	τ
Etheostomea artesiae	–0.53	0.28	–0.53	0.33	0.11	0.35
Luxilus chrysocephalus	–0.46	0.21	–0.48	0.41	0.17	0.42
Lepomis megalotis	0.42	0.17	0.37	0.39	0.15	0.28
Pimephatus notatus	–0.43	0.18	–0.48	0.49	0.24	0.42

NMS nonmetric multidimensional scaling

We found that both study watersheds experienced lower flows and sediment concentrations in the years after habitat rehabilitation than in the years just prior to rehabilitation (Table 3), and these observations agree with work by others that considered slightly different spatial or temporal domains (Runner and Rebich 1997). Although determination of causal links is beyond the scope of this study, these trends were likely due to the combined influence of lower precipitation, conversion of cultivated lands to forest or pasture (Kuhnle and others 1996), and channel stabilization (Simon and Darby 2002). At any rate, it appears that habitat and fish responses observed in this study reflect the effect of a major improvement in water quality in the form of lower suspended sediment concentrations. Furthermore, it appears that reducing sediment concentrations and installing standard erosion controls at the watershed scale without consideration of other habitat issues were not adequate to rectify critical issues that limited recovery of fish community structure in the control and reference reaches.

Major shifts in physical habitat characteristics followed habitat rehabilitation, and these changes persisted for at least a decade. Both habitat reaches became deeper and one of them became wider, whereas upstream control reaches remained relatively shallow. Current velocity decreased in the habitat reaches over the short term (Shields and others 1998), and although current velocity was not measured over the long term, it must have remained lower because of increased flow cross-sectional area. Habitat changes in the rehabilitated reaches produced fish assemblages more similar to a nonincised reference stream about

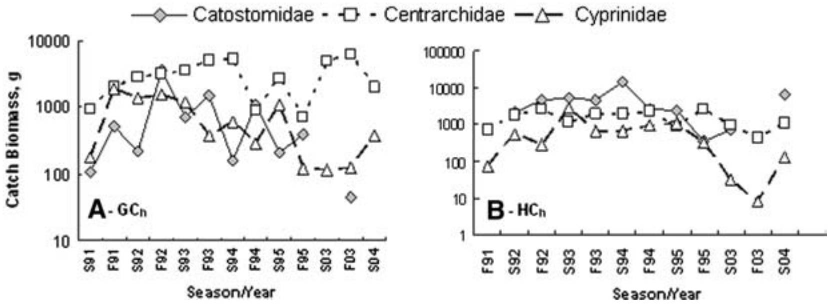

Fig. 6 Total catch biomass comprising catostomids, centrarchids, and cyprinids from (**A**) Hotophia Creek habitat reach (HC_h) and (**B**) Goodwin Creek habitat reach (GC_h). X-axis labels indicate season (S = Spring, F = Fall) and year of sampling. Points for 1991–1995 are the total biomass collected by electrofishing four 100-m-long sampling zones of each stream. Points for GC for 2003 are also electrofishing totals, whereas 2004 represents catch from concussive sampling two 25-m-long sampling zones. Points for HC for 2002 are electrofishing totals, whereas 2003 represents catch from concussive sampling two 25-m-long sampling zones.

25 km away, where small cyprinids comprised 29% of numbers and <1% biomass (Shields and others 1998), and persisted for >10 years, consistent with previous observations (Shields and others 1994, 1997, 1998). Others have suggested that stable pool habitats in warmwater streams enhance food web development and predator fish populations (Schlosser 1987b, Schlosser and Angermeier 1990). Multivariate analyses (NMS) confirmed the association of larger-bodied centrarchids, catostomids, and piscivores with the kinds of physical habitat changes produced by rehabilitation.

Stream fish communities depend on multiple habitat types in natural spatial and temporal arrangements that allow fish to complete various life cycle stages and survive extreme events (Schlosser 1995, 2002). Thus, restoration projects need to be planned with all aspects of system integrity in mind, but efficient, incremental rehabilitation may often be achieved by attacking the ‘‘critical elements’’ or limiting physical habitat factors, at least in watersheds similar to those studied here. In some settings, identification of critical elements may be difficult or impossible, but were addressed here by adding stable pool habitats and woody riparian vegetation. Clearly, pool habitats and riparian vegetation alone do not remedy all types of ecological degradation, and too much pool habitat can be worse than too little. The lower gravel substrate availability in our habitat reaches relative to upstream control reaches (Fig. 5B) may be viewed as a negative effect of the habitat rehabilitation work, because many native species depend on gravel substrate for various functions. Furthermore, the low rate of survival for willow post plantings and the low levels of LW density in the habitat reaches even 10 years after rehabilitation were

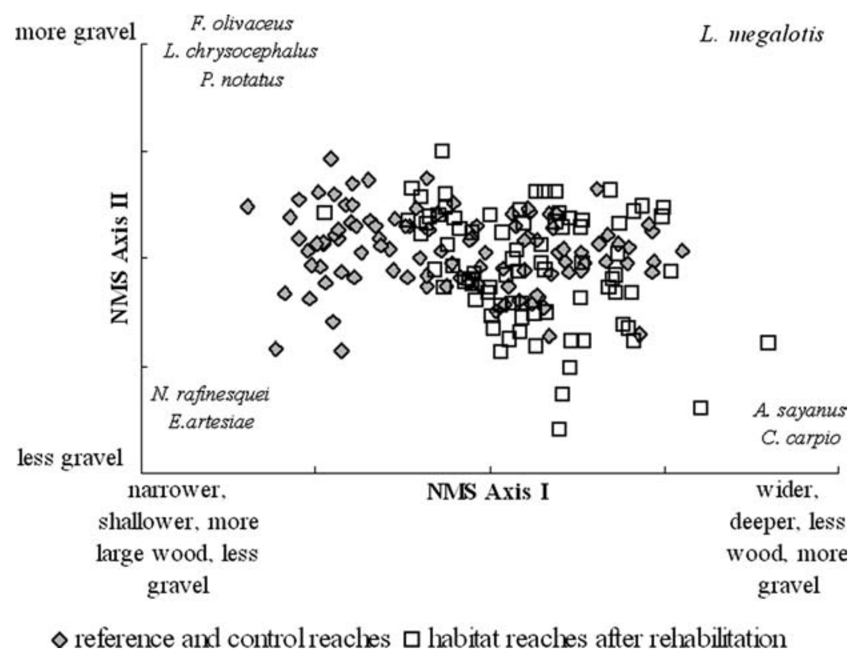

Fig. 7 Ordination of fish collections from sites with and without habitat rehabilitation. Ordination is based on first two axes of nonmetric multidimensional scaling (NMS) as described in the text

disappointing, especially at GC_h. The importance of wood to many native species is well established (Warren and others 2002). The negative correlation between large wood density and our NMS Axis I reflects the failure of our rehabilitation projects to trigger higher levels of large wood input and retention. Meanwhile, large wood densities remained high in the incised, degraded reference stream BB_r because of wood input from rapid mass wasting of channel banks.

Work by others suggests that stream ecosystems may not be restored by simply modifying physical habitat at the reach scale (Beschta and others 1994, Thompson and Stull 2002, Pretty and others 2003, Thompson 2005, Thompson 2006, Lepori and others 2005), and that existing stream conditions may reflect long-term legacy effects that are almost irreversible over the near term (Harding and others 1998, Jackson and others 2005). However, much evidence attests to the association between warmwater stream fish community structure and instream habitat characteristics even in degraded systems with less than optimal water quality (Meffe and Sheldon 1988, Shields and others 1995b, Warren and others 2002, Shields and others 2003, Smiley and others 2005). It is important to note that the positive changes in fish community composition reported here were produced after implementation of both watershed-scale erosion controls and instream habitat measures. The fundamental finding of this effort seems to underscore the principle that stream ecosystems are constrained by phenomena acting at different scales in a hierarchical fashion. Where both large- and small-scale constraints occur, effective restoration requires a coordinated attack on both (Palmer and others 2005). Such a coordinated attack could produce large ecological returns for small incremental investments over and above standard erosion controls.

The persistence of restoration effects over the longer term is related to the performance of the instream structures. Other projects have experienced initial improvements in habitat quality and attendant shifts in fish community structure that were later reversed by failing structures (e.g., Shields and others 2003, 2006a). Extensive reviews of stream habitat rehabilitation structure performance report high rates of failure (e.g., Frissell and Nawa 1992, Thompson 2005, but see Roper and others 1998 and Schmetterling and Pierce 1999). The basic designs used in this study were not substantially different from those used for generations (Shields 1983, Thompson 2006), and recent advances were not available at the time of design (e.g., Schwartz and others 2002, Kuhnle and others 2002, Thompson 2002b). However, the habitat projects were planned and sited with full consideration of the geomorphic context and currently dominant processes (Shields and others 1992): both habitat reaches were located in aggrading reaches (channel evolution model Type V, Simon 1989a, 1989b) upstream from grade control structures that provided bed stability.

Human activities have tended to impact warmwater stream ecosystems by degrading physical habitat characteristics as well as water quality. Although considerable attention has been devoted to improving instream water quality and to controlling watershed sediment yield, much evidence suggests that the most injurious impacts to stream ecosystems are driven by processes associated with channelization, incision, and riparian zone deforestation. The watersheds described in this study have experienced considerable instability since European settlement in the mid-nineteenth century, and re-stabilization of channel beds and banks required expenditure of more than $20 million between 1985 and 2001. Despite these efforts, stream habitats and associated fish communities remain in a degraded condition except for two, 1-km-long reaches that were modified for this research at a very small incremental cost. These two projects meet the five criteria proposed by Palmer and others (2005) for ecologically successful river restoration, but the practices they demonstrate have not been widely used across the rural landscape in this region. Current environmental quality initiatives are more focused on the effects of management practices that are applied outside the stream corridor on water quality rather than instream measures and physical habitat (Shields and others 2006b). We find that these practices must be paired with in-channel restoration measures to be successful, and so wide recovery of stream ecological resources is likely to be difficult, expensive, and slow. Perhaps most disheartening is the fact that after nearly 40 years of federally funded research on the deleterious environmental effects of traditional stream channelization, this type of stream management continues to be practiced by federal water resources agencies in northern Mississippi.

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Table A1 Fishes collected before and after habitat rehabilitation projects in two reaches of incised Northern Mississsippi streams, GC_h and HC_h as well as from unrehabilitated reaches upstream, HC_c and GC_c, and degraded reference streams BB_r and PC_r, see Figure 2 for definition of short-term and long-term periods

	BB_r			GC_h			GC_c	HC_h			HC_c	PC_r		
	Before	Short term	Long term	Before	Short term	Long term	Long term	Before	Short term	Long term	Long term	Before	Short term	Long term
Aphredoderidae														
Aphredoderus sayanus									2		2			
Atherinidae														
Labidesthes sicculus									6	4	2			
Catostomidae														
Carpiodes carpio	1			7	2				58	8	2	29	7	1
Erimyzon oblongus	97	187	1	231	190	1	10		6					
Ictiobus bubalus				3					49	1			22	
Ictiobus niger									1					
Moxostoma poecilurum								1	2	3	3		5	1
Centrarchidae														
Lepomis cyanellus	30	10	2	130	245	158	79	43	192	23	12	68	166	31
Lepomis gulosus		3							1				1	
Lepomis humilis											4			
Lepomis macrochirus	55	32	10	127	74	163	114	28	585	51	29	110	235	32
Lepomis marginatus		43			62				45		5		71	
Lepomis megalotis	83	82	21	396	646	516	475	13	277	93	113	268	797	217
Lepomis microlophus						1							1	
Micropterus punctulatus	22	9	4	7	12	8	21	3	120	8	17	37	119	14
Micropterus salmoides	1	1		7	9	2	3		9				8	
Pomoxis annularis									1				1	
Clupeidae														
Dorosoma cepedianum								1	40	6		9	1	
Cyprinidae														
Campostoma anomalum		41												
Cyprinella venusta	88	163	39	107	41			92	625	24	22	148	234	239
Cyprinella camura	858	884	149	1529	687	165	573	17	738	19	123	585	923	379
Cyprinella lutrensis	67	20	99	172	42	93	260			2	34	11	16	36
Cyprinus carpio									6				2	
Hybognathus nuchalis			1									23		
Luxilus chrysocephalus	194	897	70	114	275	4	110						31	
Lythrurus umbratilis		13	2	4			5							
Notemigonus crysoleucas	10						1						1	
Notropis atherinoides		4			7			3	95				29	1
Notropis rafinesquei	1738	1064	684	606	208	33	273	4	348		11	113	62	33
Pimephales notatus	147	1059	122	426	378	13	99	9	76			148	392	11
Pimephales vigilax		81							34	44	7	3		7
Semotilus atromaculatus	507	1342	50	78	64	2	75						7	

Table A1 Continued

	BB_r			GC_h			GC_c	HC_h			HC_c	PC_r		
	Before	Short term	Long term	Before	Short term	Long term	Long term	Before	Short term	Long term	Long term	Before	Short term	Long term
Fundulidae														
Fundulus notatus		4	8		2					9	15		14	16
Fundulus olivaceus	260	352	39	188	262	72	90	15	348	26	41	23	295	11
Ictaluridae														
Ameiurus natalis	44	94	5	145	86	18	75	2	11	1	4	27	36	4
Ictalurus punctatus		2							64	8	12	93	49	19
Noturus phaeus		1									4		1	
Pylodictis olivaris														1
Lepisosteidae														
Lepisosteus oculatus									25	3	3	9	24	7
Lepisosteus osseus									13	1		1	10	
Percidae														
Etheostoma artesiae	278	381	37	42	112	22	85		3			4	22	2
Etheostoma nigrum		2											1	
Etheostoma parvipinne	6													
Percina sciera	7			8					26		8	19	28	12
Poeciliidae														
Gambusia affinis	9	67	27	51	40		4	15	66	1			30	4
Sciaenidae														
Aplodinotus grunniens						1								
Grand total	4502	6838	1370	4378	3444	1272	2352	246	3872	335	473	1728	3641	1078

BB_r Bobo Bayou reference reaches, GC_h Goodwin Creek habitat reach, GC_c Godwin Creek control reaches, HC_h Hotophia Creek habitat reach, HC_c Hotophia Creek control reaches, PC_r Peters Creek reference reaches

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