Rehabilitation of aquatic habitats in warmwater streams damaged by channel incision in Mississippi *

F. D. Shields, Jr., S. S. Knight & C. M. Cooper
U.S. Dept. of Agriculture, Agricultural Research Service, National Sedimentation Laboratory, P.O. Box 1157, Oxford, MS 38655-1157, U.S.A.; Tel. +1 601-232-2919 (voice); Fax: +1 601-232-2915; E-mail address: shields@sedlab.olemiss.edu

Abstract

Channel incision has major impacts on stream corridor ecosystems, leading to reduced spatial habitat heterogeneity, greater temporal instability, less stream-floodplain interaction, and shifts in fish community structure. Most literature dealing with channel incision examines physical processes and erosion control. A study of incised warmwater stream rehabilitation was conducted to develop and demonstrate techniques that would be economically feasible for integration with more orthodox, extensively employed watershed stabilization techniques (e.g., structural bank protection, grade control structures, small reservoirs, and land treatment). One-km reaches of each of five northwest Mississippi streams with contributing drainage areas between 16 and 205 km^2 were selected for a 5-year study. During the study two reaches were modified by adding woody vegetation and stone structure to rehabilitate habitats degraded by erosion and channelization. The other three reaches provided reference data, as two of them were degraded but not rehabilitated, and the third was only lightly degraded. Rehabilitation approaches were guided by conceptual models of incised channel evolution and fish community structure in small warmwater streams. These models indicated that rehabilitation efforts should focus on aggradational reaches in the downstream portions of incising watersheds, and that ecological status could be improved by inducing formation and maintenance of stable pool habitats.

Fish and physical habitat attributes were sampled from each stream during the Spring and Fall for 5 years, and thalweg and cross-section surveys were performed twice during the same period. Rehabilitation increased pool habitat availability, and made the treated sites physically more similar to the lightly degraded reference site. Fish communities generally responded as suggested by the aforementioned conceptual model of fish community structure. Species composition shifted away from small colonists (principally cyprinids and small centrarchids) toward larger centrarchids, catostomids, and ictalurids. Fish density and species richness increased at one rehabilitated site but remained stable at the other, suggesting that the sites occupied different initial states and endpoints within the conceptual model, and differed in their accessibility to sources of colonizing organisms. These experiments suggest that major gains in stream ecosystem rehabilitation can be made through relatively modest but well-designed efforts to modify degraded physical habitats.

Introduction

Accelerated erosion and sedimentation associated with channel incision is the driving force behind some of the most intensive and extensive destruction and degradation of stream corridor habitats. Channel incision may be triggered by a variety of causes, e.g., reservoir construction, land use change, channelization, and natural events (Galay, 1983), and has been observed to result in rapid erosion, increasing channel width by as much as an order of magnitude within

a few months or years. Case studies of channel incision and its physical effects are available for Missouri (Emerson, 1971), Iowa (Daniels, 1960; Lohnes, 1997), New Hampshire (Yearke, 1971), Oklahoma (Barclay, 1980; Schoof et al., 1986), Mississippi (Kesel & Yodis, 1992; Whitten & Patrick, 1981), Tennessee (Simon & Hupp, 1986), the Midwest (Simon et al., 1996) and Indiana in the USA (Barnard, 1977), Alberta, Canada (Parker & Andres, 1976), and central Italy (Rinaldi & Simon, in press), and effects on terrestrial riparian vegetation in Western Tennessee have been studied by Hupp (1992). Channel incision is a worldwide problem, but is particularly prevalent in the warmwater streams (*sensu* Winger, 1981) of the Southeastern U.S. due to human disturbances, abundant precipitation, highly erodible soils, and the absence of bedrock channel controls. The degradation of these streams is of special concern because of their relatively high levels of biodiversity. For example, 123 fish species have been reported for the $32\,600\,km^2$ Yazoo River basin which contains the sites studied herein (Ross & Brenneman, 1991). Previous studies of selected sites in watersheds located in the northwestern part of the state of Mississippi (hereafter simply northwest Mississippi) ranging in size from 91 to $550\,km^2$ between 1985 and 1991 yielded 31–52 fish species, and two collections from 27 sites in 1992–1993 yielded 65 species (Shields et al., 1995a).

Incision and associated processes radically transform stream corridors. Changes in channel geometry in incising systems have been generally described by similar conceptual models proposed by Schumm et al. (1984) and Simon (1989a) (Figure 1). These incised channel evolution models (ICEM) were developed based on observations of fluvial systems undergoing rapid headward incision following baselevel lowering. Channel deepening occurs as knickpoints or knickzones progress upstream. When deepening renders banks higher and steeper than a threshold based on geotechnical properties of the bank materials, widening ensues, elevating sediment loads. Suspended sediment yields reported for actively incising watersheds include 704–1673 t km^{-2} yr^{-1} (Rebich, 1993) and 157–876 t km^{-2} yr^{-1}, while average from a nonincised watershed in the same region was 57 t km^{-2} yr^{-1} (Simon, 1989b). Reaches downstream of knickpoints aggrade by trapping part of the sediment derived from upstream erosion, eventually developing a two-tiered cross-section within the enlarged channel (Figure 1). Thorne (1997) has suggested existing ICEMs be modified to account for late-stage adjustments wherein ultimate stability is re-established as the two-tiered channel develops a sinuous path and heavily vegetated bar tops.

Channel incision can have major impacts on aquatic habitat quality. Case-study evidence suggests that straightened, incised channels have sharper ('flashier') hydrographs (Campbell et al., 1972; Shankman & Pugh, 1992; Shields & Cooper, 1994; Woltemade & Potter, 1994; Wyzga, 1996), because channel deepening and enlargement eliminate stream-floodplain interactions and accentuate flow peaks. Unlike non-incised streams which overflow onto floodplains and afford low-velocity refugia to fish during catastrophic events (Jungwirth et al., 1995; Matheney & Rabeni, 1995), deeply incised channels contain all but the most infrequent floods within banks, subjecting biota to extremely harsh conditions. Shields & Cooper (1994) compared stage hydrographs for two streams in northwest Mississippi during 11 concurrent storms. One site was deeply incised and contained all flows within channel, while the other was not incised and experienced overbank flow for about half of the events. Mean time to peak and event duration were greater for the non-incised stream despite the fact that the incised stream's watershed was 2.4 times larger, 1.7 times longer, and contained less urban area.

Pool habitats of incising northwest Mississippi channels are extremely transitory, scouring or filling with nearly every event (Cooper & Knight, 1987), and pool habitat of any type is in short supply. Beds are dominated by shifting sand. Shields et al. (1994) found that three incised channels in Mississippi had mean base flow water depths ranging from 17 to 26 cm and only 9–31% of water area with depth >30 cm. Similar measures for a non-incised reference channel were 35 cm and 42%, respectively. A broader assessment of incised channels in northwest Mississippi showed that in contrast to less-disturbed warmwater streams, 77% of 25 streams sampled in the summer of 1993 had mean baseflow water depths less than 25 cm, and 53% had sand covering more than 75% of their beds (Shields et al., 1995a). Streams were often dwarfed within oversized channels enlarged by erosion. At baseflow, fewer than 10% of the sampled streams occupied more than 5% of their channel cross-sections. In contrast, a non-incised reference stream occupied an average of about 20% of its channel cross-section.

Physical habitat degradation associated with channel incision triggers many of the symptoms of ecosystem distress syndrome (Rapport et al., 1985): reductions in the stability and diversity of aquatic ecosys-
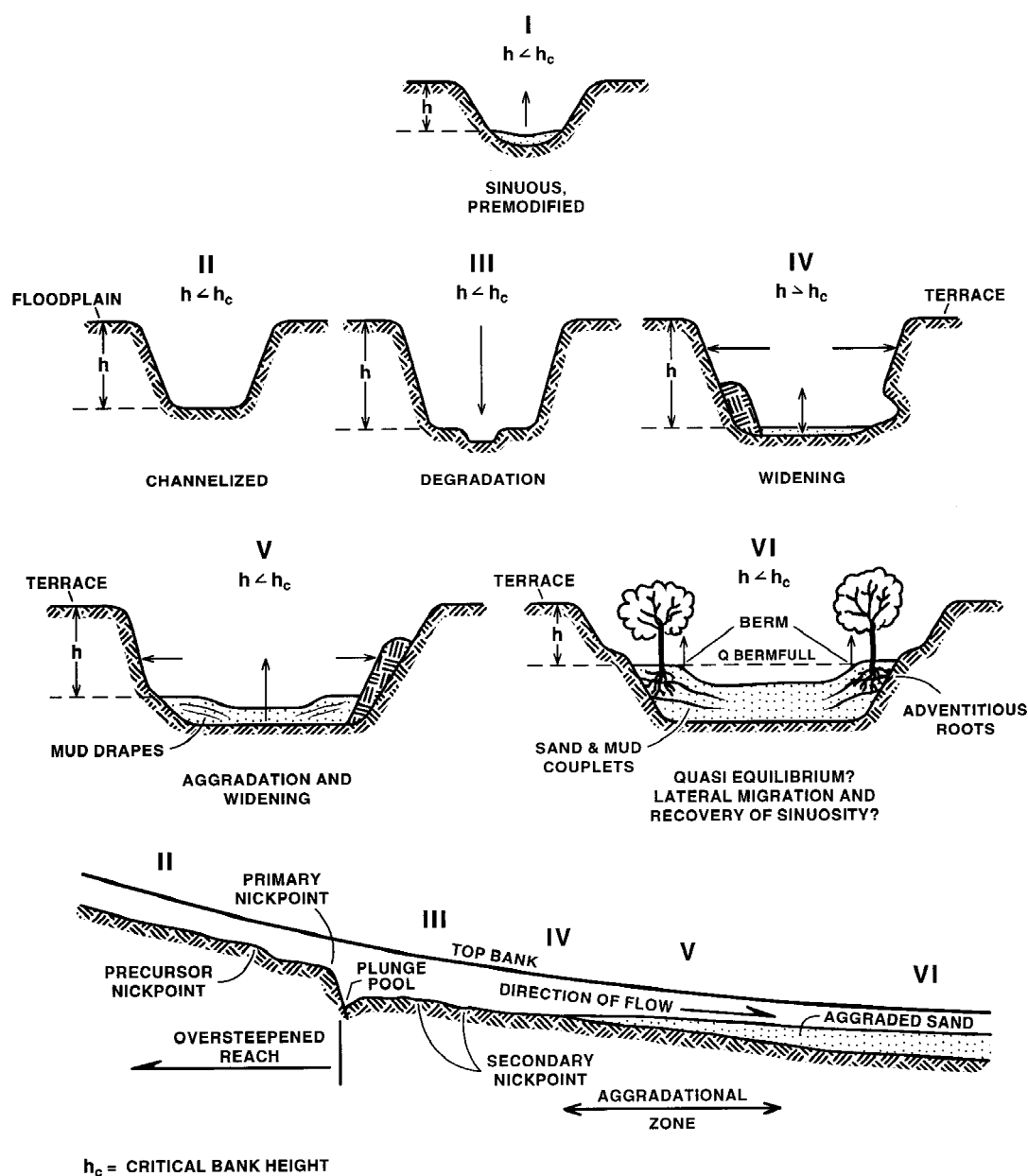
I
$h < h_c$

SINUOUS,
PREMODIFIED

II
$h < h_c$
FLOODPLAIN

III
$h < h_c$

IV
$h > h_c$
TERRACE

h

h

h

CHANNELIZED

DEGRADATION

WIDENING

V
$h < h_c$
TERRACE

VI
$h < h_c$
TERRACE

h

h

BERM
Q BERMFULL

ADVENTITIOUS
ROOTS

MUD DRAPES

SAND & MUD
COUPLETS

AGGRADATION AND
WIDENING

QUASI EQUILIBRIUM?
LATERAL MIGRATION AND
RECOVERY OF SINUOSITY?

II
PRIMARY
NICKPOINT

III

IV

V

TOP BANK

VI

PRECURSOR
NICKPOINT

PLUNGE
POOL

DIRECTION OF FLOW

AGGRADED SAND

OVERSTEEPENED
REACH

SECONDARY
NICKPOINT

AGGRADATIONAL
ZONE

h_c = CRITICAL BANK HEIGHT

Figure 1. The incised channel evolution model after Schumm et al. (1984) and Simon (1989).

tems, elimination of the longer-lived, larger species, and a tendency to favor small, short-lived opportunistic species (Menzel et al., 1984; Shields et al., 1994, 1995a). A conceptual framework for fish communities in small warmwater streams ('Schlosser's framework') proposed by Schlosser (1987) offers insight into effects of channel incision on fish and possible pathways for rehabilitation and ecological recovery, particularly when viewed concurrently with the chan-

nel evolution model and results of habitat assessments in incising channels (Figure 2). Schlosser argued that fish community structure in warmwater streams is the expression of a complex interplay between abiotic and biotic factors. He described the framework in terms of aquatic habitat heterogeneity and volume, fish species composition, fish density, fish population age structure, fish species richness and species density.
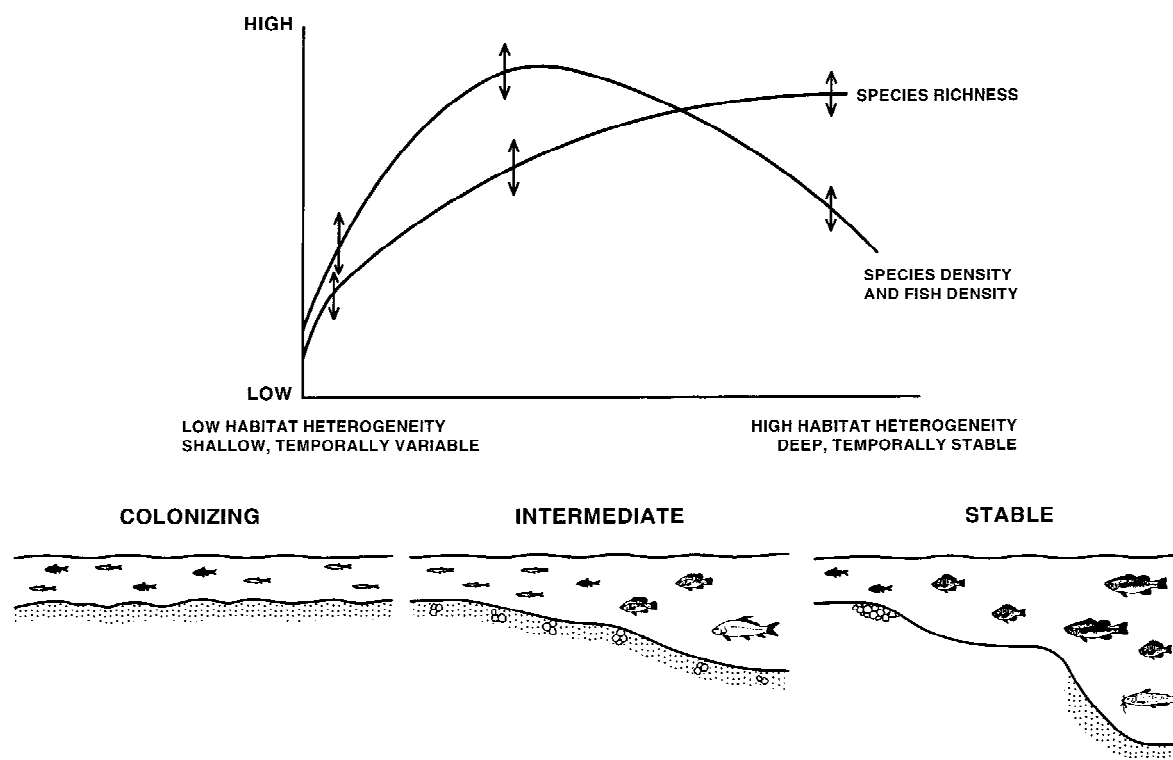
Figure 2. A conceptual framework for fish communities in small warmwater streams (after Schlosser, 1987). The adjectives colonizing, intermediate, and stable apply to the fish community, not the channel. We hypothesize that disturbance due to channel incision typically results in habitat changes and transformation of fish communities toward the left of the figure while natural recovery results in opposite trends. Rehabilitation should assist natural recovery processes.

The framework (Schlosser, 1987) includes a pattern of increasing species richness and fish size and decreasing population density along a physical gradient of habitat heterogeneity and pool development (Figure 2). Pool habitat development is a key factor in spatial and temporal heterogeneity because of its effects on habitat volume and temporal stability. Reaches with uniform conditions and shallow depths support populations dominated by juvenile cyprinids and devoid of larger fishes. Schlosser termed these communities, 'colonizing,' and noted that they were dominated by species with rapid maturity, prolonged breeding seasons, high reproductive rates, and strong dispersal capability of young (e.g., *Pimephales notatus*). According to Schlosser's framework, unstable physical conditions (e.g., wide fluctuations in streamflow), result in considerable variation in fish species richness, population density, and age structure. As pool volume, temporal stability, and habitat heterogeneity increase, species richness and population density increase due to the addition of older cyprinids and younger centrarchids and castostomids. Finally, as pool depth and volume increase further, major shifts occur in fish age/size structure, species composition, and trophic structure. Communities feature fewer, larger piscivores (e.g., large centrarchids) and fewer small invertivores and omnivores due to increased predation and competition for refugia. Schlosser termed this type of community as 'stable.' Temporal stability, both in physical and biological attributes, is higher in reaches with large, well-developed pools relative to shallow, uniform areas. However, droughts and floods still create major but temporary shifts in community pattern (Matthews 1986; Schlosser, 1990). Multiple years of observation are required to gage true differences in fish communities that are driven by physical habitat differences. Most of Schlosser's concepts were derived from studies conducted in agricultural watersheds in central Illinois. Investigations of warmwater streams in other regions have yielded similar observations (Berkman & Rabeni, 1987; Ebert and Filipek, 1988; Lamberti & Berg, 1995; Meffe & Sheldon,

Table 1. Implications of channel incision for fish communities based on the incised channel evolution models proposed by Schumm et al. (1984) and Simon (1989b) and the conceptual framework proposed by Schlosser (1987).

ICEM stage (Simon 1989)	Typical habitat conditions	Warmwater fish community	Remarks
I	Abundant deep, stable pools. High woody debris densities. Relatively narrow, deep channels that overflow frequently. Relatively high levels of heterogeneity.	Stable	Initial pre-disturbance condition. Sediment loads are low.
II	Reduced heterogeneity.	Varies depending on site-specific factors.	Post-channelization, but pre-incision.
III	High levels of temporal instability, erosion, sediment transport. However, temporary pools may form just downstream of knickpoints.	Colonizing to intermediate depending on site-specific factors	Physical habitat heterogeneity may be high due to disruption of uniform stage II conditions. Sediment loads increase by an order of magnitude.
IV	Temporal instability. Wide, shallow baseflow channels dominated by runs with shifting sand beds. Relatively little pool habitat. If banks are well-vegetated, debris densities will be high due to debris input from failing banks.	Colonizing	Sediment loads remain high due to continued bank failure
V	Similar to IV, but aggradation in the form of berms or alternate bars creates a meandering base flow channel within the enlarged cross-section. Pool development hindered by the absence of debris and well-vegetated, steep baseflow channel boundaries. Relatively low debris densities.	Colonizing	Restablishment of vegetation reduces sediment load somewhat, but degrading reaches upstream continue to provide elevated supply
VI	Low debris densities. Pool development hindered by the absence of well-vegetated steep baseflow channel boundaries, but as berms become vegetated, pools slowly recover.	Colonizing—intermediate	Sediment load lower than Stages III and IV, but higher than I

1988; Menzel et al., 1984; Paller, 1994; Rabeni & Smale 1995; Shields et al., 1995a).

The ICEM (Figure 1) and the Schlosser (1987) framework (Figure 2),when applied together, are useful for predicting responses of stream fish communities to channel incision. Although the ICEM focuses on the enlarged channel cross section, the base flow channel is more important when considering aquatic habitat, and a slightly different perspective is required. Habitat conditions (Shields & Smith, 1992; Shields *et al.*, 1994; Wallerstein et al., 1997) and fish community characteristics likely to develop during each ICEM stage are listed in Table 1. In general, Schlosser's framework suggests that incision transforms fish habitat in higher-order streams to the harsher, more temporally variable conditions typical of undisturbed headwater streams (Schlosser, 1990).

The key role that pool habitat availability and quality plays in incising channel ecosystems is illustrated by findings of other studies. Scour holes adjacent to stabilization structures in unstable, incised channels have been found to support more species of fish and larger fish than surrounding channel habitats without structures (Knight & Cooper, 1991; Shields & Hoover, 1991; Winger et al., 1976), and fish populations in some incised channels have been shown to respond quickly (∼ 1 year) to restoration of pool habitats (Swales & O'Hara, 1983). TerHaar & Herricks (1989) conducted an analysis of channelized streams in Illinois based on habitat suitability indices and con-
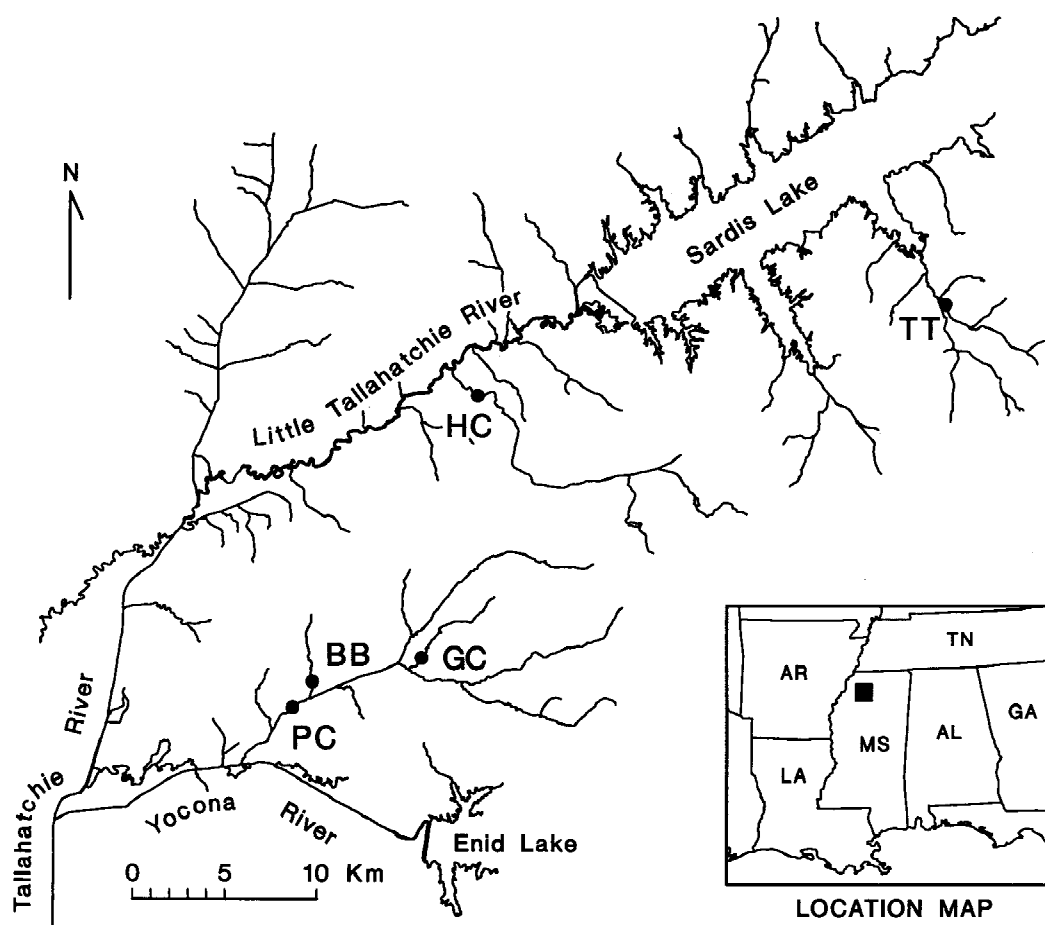
Figure 3. Study site locations.

cluded that the most critical need for the selected fish species was for creation of high quality pool habitats.

The objective of this study was to develop and demonstrate rehabilitation approaches for stream corridor ecosytems disrupted by channel incision. In particular, methods are needed that are cost-effective and compatible with more orthodox, large-scale watershed stabilization techniques. Selection of sites and treatments was guided by the ICEM and framework described above.

Study sites

One-km reaches in each of five northwest Mississippi watersheds were selected for study (Figure 3) based on their locations, disturbance histories, and the availability of hydrologic and geomorphologic data. Watersheds were located in the East Gulf Coastal Plain

Physiographic Province bordering the bluffline of the Mississippi River Valley (Fenneman, 1938). Soils, topography, and land use were typical of many streams along the eastern side of the lower Mississippi River floodplain. Ridges are capped with loess deposits, and valleys are filled with alluvium derived from post-European settlement erosion overlying a complex of six or more erodible stratigraphic units (Grissinger et al., 1982). Mean annual rainfall was about 1400 mm.

European settlement of the area began about 1830 and was followed by deforestation, cultivation, rapid erosion of hillsides, and accelerated valley sedimentation (Happ et al., 1940). Valley bottoms were covered by up to several meters of sediments eroded from hillslopes (Grissinger & Murphey, 1982; Happ et al., 1940), and swampy conditions developed as channels filled with sediment. Landowners, acting as individuals and through drainage districts, attempted to reclaim valley lands by channelizing streams and construct-

Table 2. Geometric characteristics of study reaches

Stream	Drainage area (km^{-2})	Bank height (m)	Channel width (m)	Slope (m km^{-1})	Thalweg sinuosity
PC (degraded reference)	205	2–6	55–85	0.9	1.08
HC (rehabilitated)	91	3–7	37–58	1.1	1.40
BB (degraded reference)	16	4–5	25–29	2.5	1.12
GC (rehabilitated)	21	4–5	20–70	1.6	1.12
TT (lightly degraded reference)	39	1–2	7–9	2.1	1.25

Table 3. Vegetation and structure in study reaches

Stream	Percent of bank dominated by woody vegetation	Percent canopy	Large woody debris density (m^2 km^{-2})	Stabilization works in place prior to rehabilitation
PC (degraded reference)	23	3	3940	Stone toe protection
HC (rehabilitated)	28	4	1860	Grade control weir downstream, stone toe, short stone spurs
BB (degraded reference)	44	50	62 700	None
GC (rehabilitated)	32	14	7 080	Grade control weir downstream, stone toe, short stone spurs
TT (lightly degraded reference)	71	62	103 000	None

ing drainage ditches between about 1840 and 1930. A second round of channelization and construction of major flood-control reservoirs by federal agencies occurred between about 1930 and 1960. Channels responded to channelization and reservoirs by incising headward as much as 5 m between about 1960 and the present (Grissinger et al., 1982; Simon & Darby 1997; Whitten & Patrick 1981).

All study reaches were fourth-order (Strahler) except for PC, which was fifth order (Figure 3). Sites were paired based on watershed characteristics, channel geometry (Table 2) and pre-rehabilitation habitat quality (Table 3). Peters Creek (PC) was selected to serve as a degraded and untreated reference for the incised and rehabilitated Hotophia Creek (HC), while Bobo Bayou (BB) served as a degraded and untreated reference for the incised and rehabilitated Goodwin Creek (GC). Toby Tubby Creek (TT), a non-incised, lightly degraded stream, served as a point of reference for all other sites. All of the study reaches except for TT were straightened, deeply incised and widened channels flanked by cultivated fields. Fields (pasture,

cotton, and soybeans) also flanked TT, but a ∼600-m wide band of forested wetlands laced with beaver (*Castor canadensis*) ponds separated the edge of fields and the channel. Woody debris formations and beaver dams were common within the TT channel but scarce elsewhere (Table 3). Periphyton often blanketed bed sediments of degraded sites during prolonged low flow periods, but was generally absent at TT. Conversely, aquatic macrophytes were found in certain reaches of TT where canopy openings occurred, but were rare elsewhere. Available water quality data indicated adequate conditions for most endemic and native aquatic species (Cooper & Knight, 1991; Shields et al., 1994; Slack et al., 1992). However, visual evidence suggested sediment loads were elevated at all sites, which was confirmed by measurements. Mean annual suspended sediment yields at PC and HC were 1460 and 1110 metric t km^{-2} year^{-1}, respectively (Rebich, 1993). Mean annual total load of sand size and finer sediment at GC was 1220 metric t km^{-2} year^{-1} (Grissinger et al., 1991). All sites except for BB and TT were stabilized prior to this study with

stone toe protection or very short spur dikes, and HC and GC also had grade controls at the downstream ends of the study reaches. HC and TT were sand bed streams, while the other sites had mixed beds of sand and gravel.

Rehabilitation

Since channel incision results in long-term morphological transformation at the landscape scale, its potential for degrading biotic integrity is greater than point or nonpoint source pollution (Karr, 1991). Unassisted recovery, therefore, can be expected to be slow (Yount & Niemi, 1990). Workers attempting to rehabilitate incised streams are faced with major problems in deciding where to begin and what methods to adopt. The information in Table 1 suggests that opportunities are greatest in reaches that have evolved to ICEM Stages V and VI and are experiencing some natural ecological recovery. Furthermore, attempts at rehabilitation in reaches in earlier stages of evolution or in Stage V or VI reaches that regress to earlier stages due to changes in boundary conditions, are likely to fail due to the effects of erosion on structures and vegetation. Therefore rehabilitation strategies should focus on accelerating the development of narrower, deeper baseflow channels; large, stable pools; and revegetation of bars and berms in relatively stable or depositional Stage V or VI reaches. We hypothesized that this would shift 'colonizing' type fish communities toward 'stable' communities. The overall objective of rehabilitation was to demonstrate techniques that would be economically feasible for integration with larger scale watershed stabilization activities elsewhere, and therefore redevelopment of meandering alignments was not considered. This project was a component of a federally-funded demonstration program (Shields et al., 1995c) primarily targeted at channel erosion control in rural areas. Within this program, projected total costs average \$88 900 km^{-2} of watershed area, while bank stabilization costs are typically on the order of \$150 per m of treated bankline. Only habitat rehabilitation features costing less than 10–15% of total bankline stabilization cost were considered to be economically feasible.

After 1 and 2 years of pre-rehabilitation data collection, respectively, HC and GC were modified by extending the existing stone bank protection structures into the base flow channel and by planting dormant willow posts. Modified structures were located at 50 m intervals along the channel. At HC the extensions were 12-m long sloping crest angled tips for the existing spurs (Figure 4a), while at GC the extensions spanned the base flow channel to create low weirs which were upstream pointing vees in plan (Figure 4b). At the upstream end of the GC reach, eleven stone spurs were placed perpendicular to 170 m of the existing stone toe. Spurs were ∼3 m long, had level crests and were spaced ∼7 m apart. Stone requirements for rehabilitation structures at HC and GC were less than 18% by volume of the quantity of stone required for the unmodified stabilization structures.

Willow posts 1.5 m long by 8–30 cm diameter were planted 1 m deep in toe the of eroding banks by making holes using a metal ram mounted on a hydraulic hoe. Smaller plant materials were not used because previous experiments with stakes in high-energy incising channels had been unsuccessful (Shields et al., 1995b). Dormant posts were also planted in sandbars in an effort to restore riparian zones and canopy destroyed by incision-induced channel widening. Posts were planted in a series of 1.2-m deep pits excavated in the sandbar just parallel to the base flow channel. Square pits (about 1.2 m×1.2 m) were dug within 5 m of the water's edge with an hydraulic hoe, and 10–12 posts were placed vertically around the perimeter of each pit before refilling.

Methods

Data collection

Discharge and stage were measured continuously at sites PC, HC, and GC throughout the study, and stage was measured at TT during the last 3 years of the study. BB was ungaged, but was confluent with PC. Channel cross-sections and thalweg profiles were surveyed twice at each site during the course of the study. Samples of surficial bed material (∼4 kg) were collected from the center and quarterpoints of the low flow channel at 12 cross-sections at each site before and after rehabilitation for sieve analysis.

Fish and physical habitat variables were sampled concurrently at baseflow during the Spring (May–June) and Fall (Sept.–Oct.) for 5 years (1991–95). For PC and HC, 1991 represents pre-rehabilitation conditions, while 1992–95 were post-rehabilitation. At the other three sites, 1991–92 was pre-rehabilitation, and 1993–95 was post-rehabilitation. TT was not sampled during the Spring of the first year. Since pool-riffle
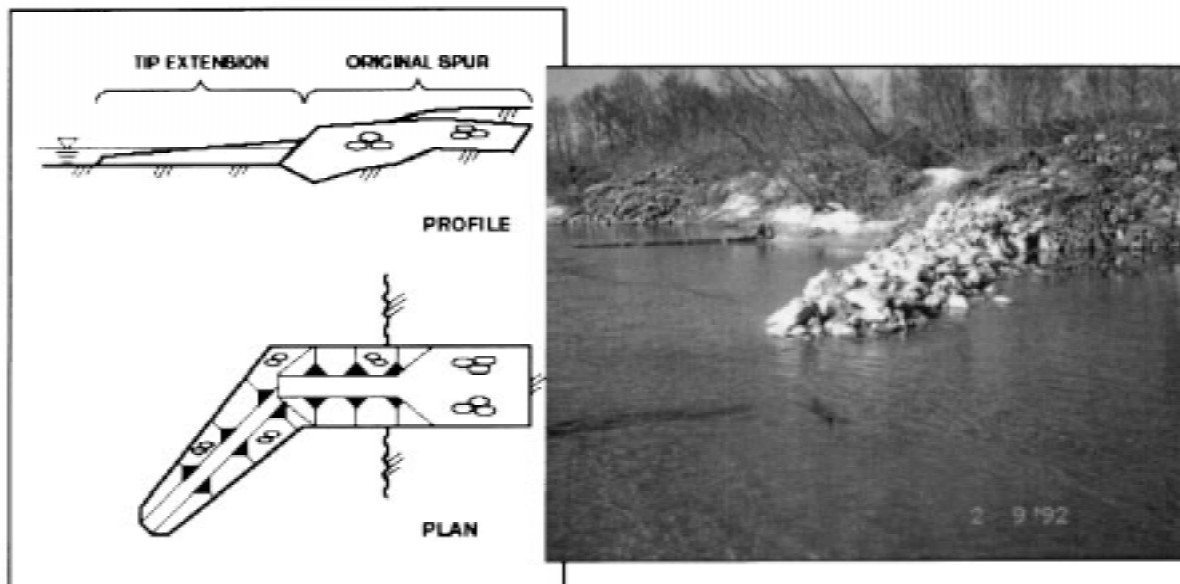
(a)
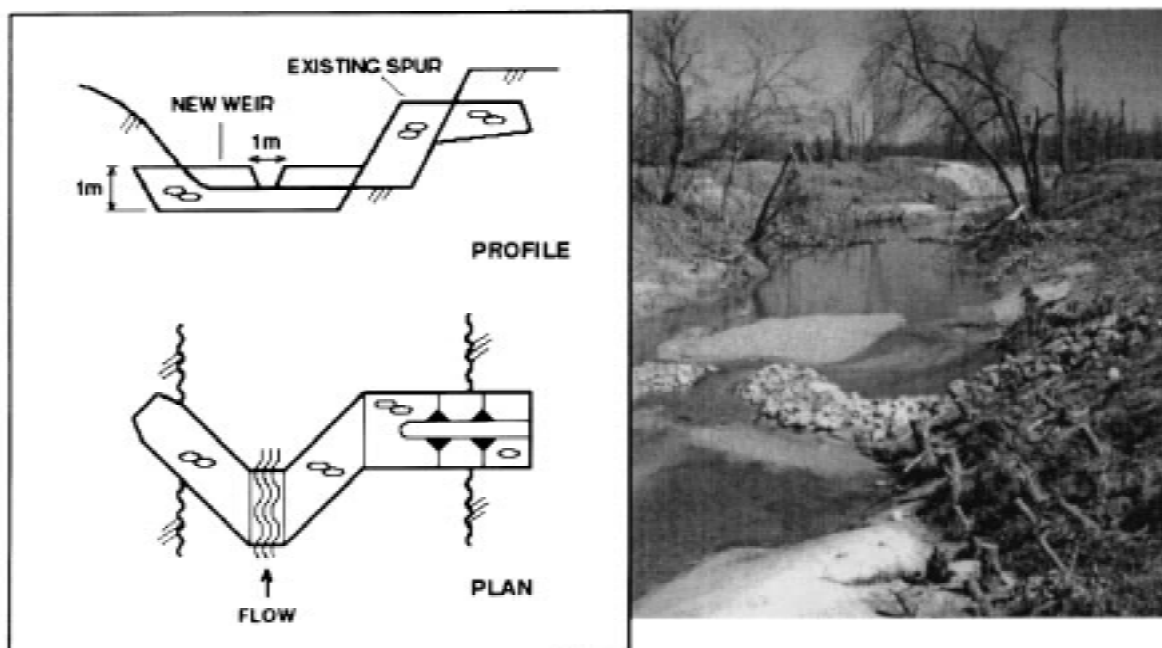
(b)

Figure 4. Rehabilitation structures (*a*) at HC and (*b*) GC.

Table 4. Categories used for calculation of physical habitat diversity (Gorman & Karr, 1978)

Variable	1	2	3	4	5	6
Depth (cm)	0–8	8–20	20–50	>50	–	–
Velocity (cm s^{-1})	0–5	5–10	10–25	>25	–	–
Bed type	Clay	Sand	Gravel	Riprap	Debris	Other

sequences were absent or poorly developed, sampling reaches could not be located based on habitat units. Therefore we sampled four 100-m long zones distributed along each of the 1-km long study reaches. Research findings published subsequent to initiation of our study indicate that sampling reach lengths should be at least a certain multiple of baseflow stream water surface widths. Additional discussion of this issue is provided below; however, the lengths of the zones we sampled were approached or exceeded the published criteria.

Physical data collection protocols were designed to measure the most important governing variables (Aadland, 1993; Meffe & Sheldon, 1988) over the entire area sampled for fishes with a spatial sampling frequency high enough to provide suitable accuracy and precision (Simonson et al., 1994). Flow depth and velocity at 0.6 times depth (to obtain the mean velocity for the vertical) were measured with wading rod and current meter at regularly spaced grid points along 20–25 transects in each stream. A minimum of 100 grid points were sampled from each stream on each sampling date. Flow width was measured at each transect, and visual estimates were made regarding the number and size of woody debris formations, dominant type and size of bank vegetation, and percent canopy. Instantaneous discharge was measured using the wading rod and velocity meter. The maximum depth of each scour hole at the modified stone structures at HC and GC was measured with a wading rod annually at midsummer low flow.

Fish were collected using a backpack-mounted electroshocker. At each stream, the same four 100-m stream zones sampled for physical variables were fished for several minutes (mean = 8.4; S.D. = 3.3) of electric field application. Fishes longer than about 15 cm were identified, measured for total length, and released. Smaller fish, and fish that could not be identified in the field were preserved in 10% formalin solution and transported to the laboratory for identification and measurement. At GC, it appeared that

electroshocking efficiency declined after rehabilitation due to increased depth. Therefore fish were sampled using rotenone following Spring electroshocking during the final year of the study. Rotenone data was excluded from statistical analysis. Water temperature, pH, dissolved oxygen, and conductivity were measured at midday (between 10:00 and 15:00 hours) on dates when fish were sampled.

Data analysis

Habitat volume was computed by numerically integrating measured depths with width to obtain cross-sectional area, and multiplying average area by reach length. Habitat heterogeneity (H') was measured in three dimensions: depth, current, and substrate. Calculation of the Shannon index of habitat diversity followed Gorman & Karr (1978) (Table 4). To facilitate comparison of depth–velocity data sets (a 'data set' is all depth–velocity pairs collected from one stream on one date), the depth-velocity domain was divided into four broad categories or 'habitat types'. These types were defined arbitrarily as *shallows* (depth < 8 cm, velocity < 25 cm s^{-1}), *riffles* (depth < 8 cm, velocity > 25 cm s^{-1}), *pools* (depth > 20 cm, velocity < 10 cm s^{-1}), and *runs* (all other). The percentage of aquatic habitat in each category was plotted against time for each study site.

Electrofishing results were used to determine species composition, fish density (number and biomass per 100-m of stream sampled and per minute of electrofishing), length distribution for selected species, and species density (number of species captured per stream on a given sampling date and per 100 individuals). The number of species per 100 individuals was computed by dividing the number of species captured from each 100-m sampling zone on a given date by the number of fish captured from the zone on that date and multiplying the resulting quantity by 100.

Catch per unit stream length and per unit of effort were used as indicators of fish density rather than catch

Table 5. Faunal groupings based on trophic status, life history, and adult size for most common fish species captured in this study

Group	Family	Species	Common name
Minnows	*Cyprinidae*	*Semotilus atromaculatus*	Creek chub
		Luxilus chrysocephalus	Striped shiner
		Cyprinella camura	Bluntface shiner
		Pimephales notatus	Bluntnose minnow
		Notropis rafinesquei	Yazoo shiner
		Cyprinella venusta	Blacktail shiner
		Notropis atherinoides	Emerald shiner
	Fundulidae	*Fundulus olivaceus*	Blackspotted topminnow
Suckers	*Catostomidae*	*Erimyzon oblongus*	Creek chubsucker
		Carpiodes carpio	River carpsucker
		Moxostoma poecilurum	Blacktail redhorse
		Minytrema melanops	Spotted sucker
		Ictiobus bubalus	Smallmouth buffalo
Sunfishes	*Centrarchidae*	*Lepomis megalotis*	Longear sunfish
		Lepomis macrochirus	Bluegill
		Lepomis cyanellus	Green sunfish
		Micropterus punctulatus	Spotted bass
		Micropterus salmoides	Largemouth bass
		Lepomis marginatus	Dollar sunfish
		Lepomis gulosus	Warmouth
		Lepomis punctatus	Spotted sunfish

per unit water surface area (Schlosser, 1987), because the water surface width varied widely. We examined community composition at the species level, and at the level of groups (catfish, minnows, suckers, and other) we defined based on trophic status, life history, and adult size following Schlosser (1987) (Table 5). Species selected for length distribution analysis were present in numbers > 11 in pre- and post-rehabilitation samples and were pool-dwellers that normally grow to lengths > 15 cm when conditions are favorable.

Physical and biological data collected before and after rehabilitation were compared using *t*-tests if data were normally distributed and possessed homogeneous variances. Data which failed to meet these criteria were compared using a Mann–Whitney rank sum test. Species distributions were compared by computing Spearman rank order correlation coefficients pre- and post-rehabilitation distributions for each site. All statistical analyses were performed using SigmaStat 2.0 software (Jandel Scientific, 1995).

Results

Hydrology

The first year of this study was the wettest recorded during the 10-year period of record (Figure 5a). Peak discharges in 1990 and 1991 were the highest for the period of record (Figure 5b). According to Schlosser's framework, modified channels tend to experience higher levels of temporal variability in habitat heterogeneity and volume. Schlosser illustrated this aspect by comparing coefficients of variation calculated for various physical attributes that were measured monthly. Mean daily stage records for three of our sites highlight the effect of incision on habitat temporal variability (Figure 5c and Table 6). The incised streams PC and HC had stage frequency distributions strongly skewed to the right, while the non-incised TT stages were less strongly skewed. Hydrological conditions in the incised channels were typified by long periods of extremely low base flow punctuated by short, sharp high flows. Conditions at TT were mod-
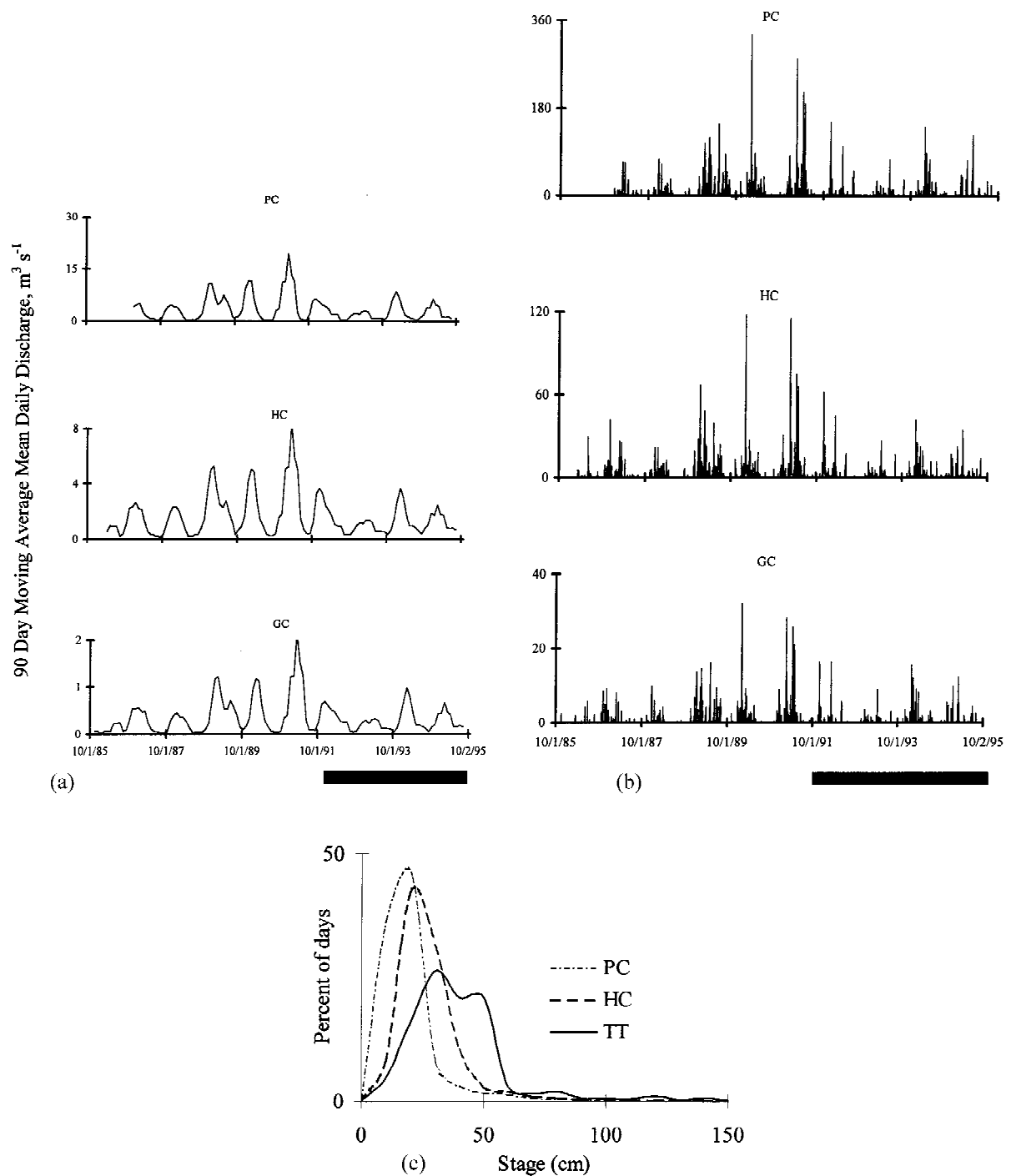
Figure 5. (*a*) Three-month moving average of mean monthly discharge (m^3 s^{-1}) for sites PC, HC, and GC. Black bar at bottom indicates duration of this study. (*b*) Mean daily discharge (m^3 s^{-1}) for sites PC, HC, and GC. Black bar at bottom indicates duration of this study. (*c*) Frequency distribution of mean daily stage at sites PC, HC, and TT. See Table 6 for descriptive statistics.

Table 6. Descriptive statistics for mean daily stage: Frequency distributions are shown in Figure 5c

Stream	Number of daily means (n)	Coefficient of variation (%)	Skewness	Kurtosis
PC (degraded reference)	966	10	4.5	27.6
HC (rehabilitated)	1003	12	3.2	15.7
TT (lightly degraded reference)	853	25	2.5	9.6

erate: base flows were relatively higher, while storm hydrographs were lower and lasted longer. The differences depicted in Table 6 and Figure 5c would likely be greater if instantaneous data collected at hourly or shorter intervals were used in the analysis in place of daily means (Watson & Biedenharn, in press).

Instantaneous baseflow discharges measured when habitat data were collected were far less variable at the nonincised site than the incised sites. The coefficient of variation for instantaneous discharge (Fall 1991–Fall 1995) ranged from 33 to 62% at all incised channel sites but was only 13% at the nonincised TT.

Habitat volume, bed type, and heterogeneity

Visual observations indicated that HC and GC stream channels responded quickly to the rehabilitation structures, with deep scour holes forming adjacent to structures following the high flow events which occurred in the first 3–4 weeks following construction. Conversely, the willow posts had little effect on physical aquatic habitat during this study. Only about 30 and 50% of posts survived two growing seasons at HC and GC, respectively. Post mortality was related to competition by the fast-growing exotic vine, *Pueraria lobata* (kudzu) and the infertility of bank soils exposed by incision (Grissinger & Bowie, 1984). Surviving posts grew slowly. However, woody vegetation cover on one side (convex point bar) of the HC site increased from 38% before planting to 78% after two growing seasons.

At HC, mean water depths were smaller following rehabilitation due to lower discharges on the post-rehabilitation sampling dates (Table 7). Pool habitat increased from 2% before rehabilitation to 14% a year later but then declined to values comparable to the degraded site PC (2%–7%) (Figure 6). Scour holes that formed at the tips of the rehabilitation structures were more persistent: the average of the maximum depth at extended spurs (corrected for stage variation) increased from 32 cm before rehabilitation to 84 cm a
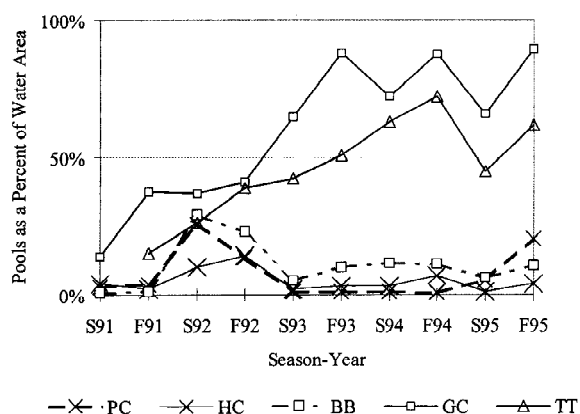

Figure 6. Pool habitat versus time for rehabilitated and reference sites.

few months afterward, and means measured during the following 3 years were 72, 60 and 79 cm ($n = 15$). Laboratory data (Kuhnle et al., 1997) from sand-bed flume experiments indicate that the volume of a spur dike scour hole pool is proportional to the cube of maximum depth. Thus we suspect that the volume of scour hole pools declined as maximum scour depth declined, but was still about 14 times greater than before rehabilitation. Longer spurs, or structures which entirely spanned the channel may have produced larger scour holes, but would have been more prone to failure. At one location within the study reach, spurs were placed on opposite sides of the stream, creating a constriction. Both spurs were destroyed when they fell into the resultant scour hole during the first high flow after construction. Some individual scour holes experienced cycles of almost complete filling and re-scour. Waves of bed material from upstream reaches contributed to scour hole instability. In addition, as sand eroded from beneath the dike extensions they sank into the bed, and 1–2 m of the riverward tip of each structure subsided beneath the bed surface.

At GC, where stone structures spanned the entire base flow width and gravel was present in the

Table 7. Aquatic habitat before/after rehabilitation

Stream	Mean instantaneous discharge ($L\,s^{-1}$)	Mean width (m)	Mean depth (cm)	Mean velocity ($cm\,s^{-1}$)	Mean habitat volume (m^3/100 m of stream)	Mean pool habitat (% of water area)	Median size of surficial bed material (mm)	Percent of bed covered with sand (Gravel)	Mean habitat heterogenity (H')
PC (degraded reference)	478/496	17.3/18.0	15/17	**17/21**	244/296	3/9	0.49/0.49	77/71(16/17)	2.49/2.36
HC (rehabilitated)	**1165/684**	**16.7/13.8**	**22/12**	**24/19**	**382/194**	3/6	0.40/0.35	92/90 (0/1)	1.87/1.98
BB (degraded reference)	7/8	**2.8/3.0**	**5/8**	7/5	**26/29**	14/9	0.52/2.0	**62/29**(25/44)	2.21/2.32
GC (rehabilitated)	46/45	**6.7/8.7**	**20/51**	**5/3**	**125/483**	**32/78**	0.55/0.75	44/39(34/31)	**2.83/2.32**
TT (lightly degraded reference)	348/408	**6.2/6.7**	**35/50**	**19/10**	**250/332**	**27/56**	0.35/0.40	63/60(0/0)	2.39/2.49

Boldface means indicate significant differences between before and after values ($p < 0.05$)

Table 8. Fish species richness and density before/after rehabilitation.

Stream	Total number of species, entire study	Mean number of species per stream on a given date	Mean number of species per 100 individuals
PC (degraded reference)	37	**15/19**	12/12
HC (rehabilitated)	31	**11/19**	26/16
BB (degraded reference)	32	16/17	**6/4**
GC (rehabilitated)	24	16/15/18	**6/11**
TT (lightly degraded reference)	48	17/17	25/22

The third value for GC represents results of rotenone collections during the last year of the study. Boldface values indicate significant differences between before and after means ($p < 0.04$)

bed, physical changes were greater and more persistent through time. Pool habitat availability doubled following rehabilitation as a result of the scour holes created by the structures and because beavers used many of the structures as foundations for dams. Mean depth increased 155%, and velocity decreased 40%. The rehabilitation structures were more stable than at HC, and average maximum scour hole depths for the three summers following rehabilitation were 129, 141 and 129 cm. GC habitat quality became less similar to the degraded reference site BB and more similar to the lightly degraded reference, TT. Beaver activity was limited to periods of prolonged low flow, and beaver dams were removed by higher flows each winter. Gains in GC pool habitat were at the expense of runs and shallows, but there was little temporal change in riffles.

Aside from increasing mean depth at TT, reference sites showed little change. Pool habitat increased at TT over the period of observation, perhaps due to beaver activity and increasing levels of woody debris density in the stream.

Bed material became coarser at BB as waves of gravel were transported into the study reach from degrading reaches upstream and replaced sand (Table 7). Bed-type distribution at the other sites changed little (Pearson correlation coefficient between pre- and post-bed type distributions, $r > 0.97$, $P < 0.002$). Periphyton blooms occurred on the beds of all four of the degraded sites during the low flow periods late 1993 and 1994, covering an average of 21% of the bed surface. During periods of prolonged low flow, periphyton blanketed much of the bed of degraded reaches except for HC, covering an average of 14, 18 and 27% of the beds in Fall 1992, Spring 1993 and Fall 1993. In contrast, periphyton was virtually absent from the bed of TT, perhaps because of its 62% canopy cover (Table 3). Despite the increased pool area at GC, TT and HC, overall habitat heterogeneity changed little following rehabilitation. Mean H' values were actually slightly lower at GC following rehabilitation, perhaps due to the large areas impounded by beaver dams constructed on the stone weirs during the summer.

Species richness and density

Following rehabilitation, mean species richness increased by 72% at HC and by 27% at its degraded reference, PC, but remained unchanged at the other sites (Table 8). The relatively low level of average species richness at TT (17) relative to the large number of species found there during the entire study (48) is due to the large number of rare species. Eleven species found at TT were represented by only one individual. Conversely, the incised sites PC, HC, BB, and GC produced seven, four, two, and no species, respectively, represented by only one individual. Electroshocking efficiency declined at GC following rehabilitation as depth, width, and water volume increased. Despite lower electrofishing efficiency, sampling with rotenone produced only three more species than electrofishing (Table 8), indicating that electroshocking samples produced reasonable estimates of species richness.

The number of species found per 100 individuals captured fluctuated widely at all sites, and significant differences were found between pre- and post-rehabilitation periods only at GC and BB (Table 8). While the number of species per 100 individuals fell from six to four at BB, it nearly doubled at GC, increasing from six to 11. Schlosser's framework (Figure 2) indicates that species and population density both trend downward as habitat recovery progresses. However, if fish density falls faster, the number of
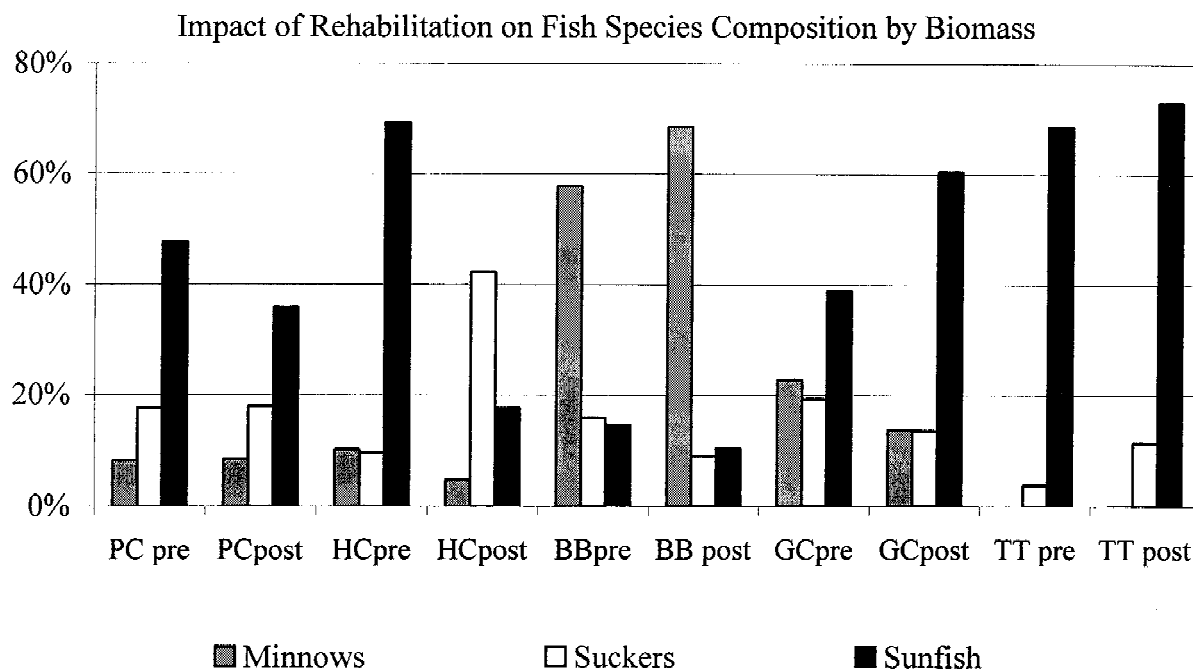
Impact of Rehabilitation on Fish Species Composition by Biomass

Figure 7. Fish species composition before and after rehabilitation.

Table 9. Similarity of ranks of species based on (*a*) numbers of 14 most dominant species (*b*) biomass distribution of 13 most dominant species, before/after rehabilitation.

Stream	Degraded Reference (PC for HC, BB for GC)	Lightly degraded reference (TT)
(*a*) Numbers[a]		
HC	**0.801/0.738**	0.296/0.372
GC	**0.753**/−0.033	−0.338/−0.300
(*b*) Biomass[b]		
HC	0.390/**0.845**	−0.077/0.367
GC	0.465/0.280	−0.501/0.072

[a]Spearman's rank order correlation coefficient; bold values are significant with $P < 0.003$.
[b]Spearman's rank order correlation coefficient; bold values are significant with $P < 0.001$.

species per 100 individuals will rise, as at GC. Relatively low numbers of species per 100 individuals at the degraded reference sites PC and BB (six to 12) are indicative of colonizing and intermediate communities dominated by a few opportunists.

Species composition

A complete list of fish species and total number of captures obtained during this study is provided as Appendix A. Spearman rank-order correlation coeffi-

cients based on numerical frequency of the 14 most common species (species which comprised > 1% of the catch) indicated that the rehabilitated site communities became slightly more similar to the lightly degraded reference and less similar to the degraded references following rehabilitation (Table 9a). In general, these shifts were not statistically significant. Similar results were obtained for distributions of biomass among the 13 most dominant species (Table 9b). However, definite shifts in community structure did occur, and these changes were consistent with the Schlosser framework. Before rehabilitation, small, opportunistic species were dominant, and those we classified as 'minnows' comprised an average of 44% of the numbers and 25% of biomass of fish samples from all sites except the lightly degraded reference, TT, where minnows were 29% of numbers and less than 1% of sample biomass (Figure 7). After rehabilitation, minnows comprised 36% of fish numbers at the rehabilitated sites but only 9% of biomass. In accordance with Schlosser's framework, increasing pool size and stability led to communities composed of larger individuals, principally sunfishes. Prior to rehabilitation, sunfishes comprised 15% of the numbers and 39% of the biomass in GC samples, but 28% of numbers and 60% of biomass afterward in electrofishing samples

Table 10. Fish density before/after rehabilitation.

Stream	Mean number of fish per 100 m	Mean number of fish per minute of sampling	Mean fish biomass (kg per 100 m)	Mean fish biomass (g/minute of sampling)
PC (degraded reference)	112/142	15/21	1.54/2.11	238/303
HC (rehabilitated)	**40/123**	10/16	**0.18/2.70**	**43/320**
BB (degraded reference)	289/285	44/40	0.43/0.78	**72/108**
GC (rehabilitated)	274/144/4,096	**37/12**	1.33/1.20/10.0	164/105
TT (lightly degraded reference)	73/83	10/8	**2.65/0.72**	**418/71**

The third value for GC represents results of rotenone collections during the last year of the study. Boldface values indicate significant differences between before and after values. ($P < 0.011$)

and 26% of numbers and 55% of biomass in rotenone samples.

At HC, the primary change in species composition was the relative decline of small sunfishes and minnows and the appearance of several large species that were not found prior to rehabilitation. Six of these (*Carpiodes carpio, Cyprinus carpio, Ictalurus punctatus, Ictiobus bubalus, Lepisosteus oculatus, Lepisosteus osseus*), comprised 6% of the post-rehabilitation samples by number and 76% of sample biomass. In addition, the piscivorous centrarchid *M. punctulatus* became more abundant: only three individuals ranging in length from 5 to 10 cm were captured before rehabilitation, but 120 ranging in length from 2 to 44 cm were found following rehabilitation. Evidently these relatively large fishes were able to migrate upstream from a larger river about 1.5 km downstream of the HC study reach and exploited the new pool habitats created as a result of scour adjacent to the stone spurs.

The lightly degraded reference site TT harbored fewer minnows and more sunfish than the other sites. Sunfish populations fluctuated widely at TT, but comprised 60% of the fish by number and 70% of the biomass collected from this site during the five-year study. Peaks in sunfish abundance followed overbank flooding in early 1991 and 1994.

Fish density

Following rehabilitation, numerical and biomass catch increased sharply at HC and decreased at GC (Table 10). Biomass at HC exhibited strongest response in the first 2 years following rehabilitation and declined to levels similar to those at PC in the last 2 years. At GC, several measures of density declined after rehabilitation, but with the exception of numerical catch per unit effort, these changes were not statistically significant. Rotenone collections from GC also

indicated little change in density following rehabilitation. Rotenone collections from rehabilitated reaches yielded an average of 13.4 fish with a biomass of 110 g m of stream length (Table 10), which compares with 16.0 fish and 118 g m of stream reported for rotenone collections for an untreated reach upstream (Shields et al., in press).

Fish collections from the reference sites changed relatively little during the period of observation except for a six-fold drop in biomass at TT (Table 10). Initially high biomass levels ($\sim$ 3 to 9 kg/100 m) may have been related to prolonged regional flooding in the months just prior to this study (Figure 5b). Collections from TT tended to contain fewer, larger individuals than from the degraded sites, consistent with Schlosser's framework.

Fish size (length)

According to Schlosser's framework, habitat rehabilitation should lead to development of communities with older, larger individuals. Captured adults of selected pool-dwelling species were slightly larger at GC following rehabilitation, though in only one case (*L. megalotis*) were differences in length statistically significant (Table 11). More pronounced differences appeared for some species collected with rotenone. Post-rehabilitation rotenone collections produced an average of 54 ictalurids representing the species *A. natalis* per 100 m of stream that were longer than 8 cm. Rotenone collections from a 100-m long untreated reach upstream reported by Shields et al. (in press) yielded only three individuals of this species and size. Similar changes were not observed at HC, but as noted above, post-rehabilitation collections contained relatively large individuals representing six species not present before. Selected species at the reference sites

Table 11. Comparison of median lengths (in cm) of selected fish species before/after rehabilitation.

Stream	Centrarchidae				Ictaluridae
	L. macrochirus	*L. megalotis*	*L. cyanellus*	*M. punctulatus*	*A. natalis*
PC (degraded reference)	9/10	**9/8**	8/8	**16/11**	14/11
HC (rehabilitated)	7/7	**9/8**	**8/7**	9/9	15/11
BB (degraded reference)	**8/5**	6/7	6/9	8/10	**9/7**
GC (rehabilitated)	6/7	**7/8**	8/8	12/5	11/13
TT (lightly degraded reference)	**9/6**			15/13	

Boldface medians are different ($P < 0.03$).

were generally smaller during the post-rehabilitation period.

Discussion

Methods

Research published after our study was underway indicated that the length of stream reaches sampled for fish and their habitats should be a multiple of baseflow stream water surface width. For example, sample-reach lengths of 86–584 m were found necessary to capture 95% of the fish species in each of nine Wisconsin streams (Lyons, 1992), and based on this work, Simonson et al. (1994) recommended sampling a stream reach 35 stream widths long. Angermeier & Smogor (1995) reported a sampled reach length of 22–67 stream widths was adequate to find 90% of the fish species present, while less effort was required to obtain species relative abundance, and more effort to find available microhabitat types. Greater effort was required in more homogeneous streams. Accuracy and precision of results increased exponentially with the number of stream widths sampled: in the most homogeneous stream, sampling 20 stream widths yielded species relative abundance that had 90% similarity with the 'true' value, while sampling 30 stream widths increased similarity only up to 97%. It should be noted that these findings were based on study of relatively lightly degraded streams that had more clearly structured pool-riffle habitats than our study sites. Although the approaches used to sample physical habitat variables were similar to ours, fish sampling was conducted with electric seines rather than backpack electroshockers. Most importantly, these studies sought to determine the species richness, species composition, and the number of available microhabitats in a given stream reach based on a one-time sample. However, the general shape of the exponential relationship between the length of stream sampled and the precision of the resulting data (Angermeier & Smogor, 1995) would still hold in our case. Our protocol therefore yielded slightly more reliable results for the smaller site GC (46–60 mean stream widths sampled) and its untreated reference BB (140 widths) than for the larger HC (24–29 widths) and its untreated reference PC (22–23 widths). We sampled about 60 mean stream widths in TT, the lightly degraded reference site. Repetitive sampling of each site (two to four times before rehabilitation and six to eight times afterward) elevated confidence in study findings.

The approach used in this study for measuring stream habitat quality has strengths and weaknesses. Values of water depth and velocity used to define the boundaries between habitat types were subjective (e.g., pools were all areas with $d > 20$ cm, and $v < 10$ cm s^{-1}), and were designed to be sensitive to changes in habitat produced by rehabilitation. Use of depths and velocities measured at grid points to define habitat type (pool, run, etc.) rather than visual

classification of each stream segment eliminated the problems associated with the subjectivity of the latter approach. On the other hand, an isolated point with relatively deep water and low velocity is not a pool in the sense that a large area of deep, slow water is, and does not provide the cover and structural attributes that a natural pool in a well-developed pool-riffle sequence does.

Hydrology and rehabilitation monitoring

Impacts of rehabilitation on fish communities and their habitats must be viewed in light of the influences of other physical factors imposed on the systems. Elevated flows in the year prior to this study and during the pre-rehabilitation period may partially account for the quality of habitat and fish community structure observed before rehabilitation and the magnitude of response following rehabilitation. However, the reference sites were subject to the same climatic and hydrological influences, and provide a good basis for estimating the relative influences of natural recovery from high flows and rehabilitation-assisted recovery. Based on a study of Oregon streams, Pearsons et al. (1992) suggested that fish assemblages in hydraulically complex streams (less disturbed) were more resistant to floods than those in hydraulically simple ones.

Physical habitat rehabilitation

Rapid physical response to spurs and weirs, including significant increases in pool area and volume, was consistent with observations by others (House et al., 1991; Jungwirth et al., 1995). Our efforts to move the fish community structures of these extremely degraded systems along the x-axis of Schlosser's diagram (Figure 2) by imposing physical changes were only partially effective. Pool habitat availability increased slightly at HC and greatly at GC, but overall habitat diversity as measured by a Shannon index did not increase. Three factors may have contributed to the static Shannon index values:
(1) the HC structures were not large or numerous enough to produce greater change;
(2) GC structures and the beaver dams they supported produced so much pool development that overall habitat complexity declined; and
(3) other than addition of some riprap, the rehabilitation projects did little to diversify the monotonic sandy substrates, which reflected the high sediment loads prevalent in incising channels.

Structures likely added temporal stability to pools as well as increasing their size, but willow plantings had minimal effects on habitat during the period of observation. Throughout the study, woody debris densities in the degraded channels were an order of magnitude or more lower than for the lightly degraded reference, TT, and canopy and woody riparian vegetation were also low. In addition, our projects were not designed to address one of the major perturbations within these systems—their flashy hydrology. Spring floods in warmwater streams have been associated with large reductions in juvenile centrarchids and cyprinids (Schlosser & Angermeier, 1990).

Fish community rehabilitation

The main response of the fish communities to rehabilitation was in their species composition, and this response mirrored the pattern predicted by the framework (Schlosser, 1987). Numbers and biomass of fishes representing pool-dwelling species increased significantly at the treated sites following rehabilitation, evidently in response to the physical transformations produced by the structures and by beaver. Concurrent shifts in invertebrate populations, perhaps due to increased predation by fish have been documented elsewhere (Cooper et al., 1997).

Physical response to rehabilitation was greater at GC than HC, but fish density followed a different trend: fish numbers and biomass increased at HC, but showed little change at GC. There are four factors that may be account for the differential response. First, electrofishing efficiency declined with increasing water depth and width. However, we note that rotenone collections did not produce different findings. Second, the sites may have represented different final conditions within Schlosser's framework. The framework indicates a pattern of initially increasing and then decreasing fish density with increasing depth and temporal stability. This suggests that post-rehabilitation conditions produced an intermediate community at HC and a stable community at GC. Third, in streams with strong hydrological environmental variation, fish abundance and distribution are less closely associated with the physical habitat features affected by these rehabilitation projects than in streams subject to less environmental variability, but may reflect extinction/recolonization processes (Angermeier & Schlosser, 1989). Finally, the differential response of the fish communities at the two sites also may be related to their accessibility for colonists.

There were grade control weirs at the downstream end of both sites, but the HC weir was often submerged by high flows, while the GC weir normally presented a 2-m vertical obstacle to upstream migrants. HC was 1.7 km upstream from a confluence with a large, non-incised river, while GC was 11.4 km upstream from a similar confluence. The sudden appearance of large adult catostomids at HC less than 4 months after rehabilitation was likely due to the interaction of colonization and habitat quality. A source of colonists was in suitable proximity to the rehabilitated habitat that did not exist prior to the project. Furthermore, during high flows, the path from the source to the study reach was free of obstacles. Similar colonization did not occur at GC. Schlosser (1995) noted the importance of the availability of refugia to fish population dynamics, and the interaction between colonization processes and environmental heterogeneity is a fundamental attribute of his framework. Reaches with high levels of overall physical heterogeneity and stable pools will not achieve their biotic potential if they are isolated from other aquatic habitats which serve as sources of colonists and furnish refugia to stream biota during extreme events.

The framework indicated that temporal fluctuations in density should be less severe in reaches with higher habitat quality, but fluctuations in fish biomass density at TT were quite large relative to other sites and may reflect stream–floodplain interactions not possible for the deeply incised channels. In particular, we suspect that the large peaks in centrarchid abundance and biomass observed at TT may be related to the movement of fish between floodplain beaver ponds (Schlosser, 1995) or the downstream reservoir and the stream during periods of prolonged higher flows.

Schlosser (1987) hypothesized that large fish would avoid shallower reaches because of space limitations or increased vulnerability to terrestrial predators. Accordingly, we anticipated addition of stable pools would foster development of populations with larger fish. However, the size of only one of five selected pool-dwelling species increased in GC electrofishing collections following rehabilitation. Although two of four selected species at HC were slightly smaller following rehabilitation, numerous individuals representing larger species not found before rehabilitation appeared, increasing the mean length of all fish from 4.2 to 7.7 cm. Our observations of species richness and density were consistent with the notion that changes at HC represented movement from a colonizing to an intermediate state, while GC was transformed from intermediate to stable. Species richness and density both increased at HC, while at GC richness was stable and density fell.

Conclusions

Findings reported herein should be applicable to sand and mixed sand and gravel bed streams with bed slopes on the order of 0.001 or less and draining humid, temperate watersheds smaller than about 200 km^2. Channel incision and attendant erosion and sedimentation represent one of the most powerful destructive influences acting on stream corridor ecosystems, particularly in streams lacking bedrock controls. Limited rehabilitation of incised stream corridors is possible for minimal incremental cost; perhaps as little as 10% additional to expenditures required for orthodox erosion controls without habitat considerations. Physical response to instream habitat structures in warmwater streams may be quite rapid, and response is greater for structures spanning the channel than for those that simply project into it. Minimal stone structures placed in sand bed channels are vulnerable to failure if not placed on filter layers or fabric. Aquatic habitat response to shoreline revegetation is more gradual than for instream structure, and revegetation is hindered by the presence of infertile soils and exotics.

Pool habitat availability is a critical factor in many incised streams. Addition of relatively deep, stable pools in a way that preserves and enhances overall physical heterogeneity can produce a shift in fish species composition and abundance as indicated by the framework for small warmwater stream fish communities proposed by Schlosser (1987). Although it may be hindered by the lack of floodplain interaction and by the flashy hydrology typical of incised streams, this shift makes incised stream communities more similar to those found in less degraded streams. More complete rehabilitation requires activities that treat long reaches and impact processes (e.g., sediment transport, organic matter production and transport) at the watershed scale. Only limited success can be expected for reach-scale habitat rehabilitation if the treated area is isolated with extreme degradation up- and downstream.

Acknowledgements

References

Aadland, L. P., 1993. Stream habitat types: Their fish assemblages and relationship to flow. North Am. J. Fisheries Management 13: 790–806.

Angermeier, P. L. & I. J. Schlosser, 1989. Species–area relationships for stream fishes. Ecology 70: 1450–1462.

Angermeier, P. L. & R. A. Smogor, 1995. Estimating number of species and relative abundances in stream-fish communities: Effects of sampling effort and discontinuous spatial distributions. Can. J. Fish. Aquat. Sci. 52: 936–949.

Barclay, J. S., 1980. Impact of stream alterations on riparian communities in Southcentral Oklahoma. FWS/OBS-80-17 (Fish and Wildlife Service Contract 14-16-0008-2039), Fish and Wildlife Service, Washington.

Barnard, R. S., 1977. Morphology and morphometry of a channelized stream: The case history of Big Pine Creek Ditch, Benton County, Indiana. In Studies in Fluvial Geomorphology No. 4. Technical Report Number 92. Purdue University, Water Resources Research Center, West Lafayette.

Berkman, H. E. & C. F. Rabeni, 1987. Effect of siltation on stream fish communities. Envir. Biol. Fishes 18: 285–294.

Campbell, K. L., S. Kumar & H. P. Johnson, 1972. Stream straightening effects on flood-runoff characteristics. Trans. ASAE 15: 94–98.

Cooper, C. M. & S. S. Knight, 1987. Fisheries in man-made pools below grade-control structures and in naturally occurring scour holes of unstable streams. J. Soil Wat. Conservation 42: 370–373.

Cooper, C. M. & S. S. Knight, 1991. Water quality cycles in two hill land streams subjected to natural, municipal, and non-point agricultural stresses in the Yazoo Basin of Mississippi, USA (1985–1987). Verh. Int. Verein. Limnol. 24: 1654–1663.

Cooper, C. M., S. Testa III, & F. D. Shields, Jr. 1997. Invertebrate response to physical habitat changes resulting from rehabilitation efforts in an incised unstable stream. In Wang, S.Y., E. Langendoen & F.D. Shields, Jr. (eds), Management of Landscapes Disturbed by Channel Incision, Stabilization, Rehabilitation, and Restoration, Center for Computational Hydroscience and Engineering, University of Mississippi, University, Mississippi: 887–892.

Daniels, R. B., 1960. Entrenchment of the Willow Drainage Ditch, Harrison County, Iowa. Am. J. Sci. 258: 161–76.

Emerson, J. W., 1971. Channelization: A case study. Science 173: 325–326.

Ebert, D. J. & S.P. Filipek, 1988. Response of fish communities to habitat alternation in a small Ozark stream. Proc. Arkansas Acad. Sci. 42: 28–32.

Fenneman, N. M., 1938. Physiography of the Eastern United States. McGraw-Hill, New York.

Galay, V. J. 1983. Causes of river bed degradation. Wat. Resources Res. 19: 1057–1090.

Gorman, O. T. & J. R. Karr, 1978. Habitat structure and stream fish communities. Ecology 59: 507–515.

Grissinger, E. H. & A. J. Bowie, 1984. Material and site controls of stream bank vegetation. Trans. Am. Soc. Agric. Eng. 27 (6): 1829–1835.

Grissinger, E. H. & J. B. Murphey, 1982. Present 'problem' of stream channel instability in the bluff area of Northern Mississippi. J. Mississippi Acad. Sci. 27: 117–128.

Grissinger, E. H., J. B. Murphey & R. L. Frederking, 1982. Geomorphology of Upper Peters Creek Catchment, Panola County, Mississippi: Part II, Within channel characteristics. In Proceedings of the International Symposium on Rainfall–Runoff Modeling, Water Resources Publications: 267–282.

Grissinger, E. H., A. J. Bowie & J. B. Murphey. 1991. Goodwin Creek bank instability and sediment yield. In: Proceedings of the Fifth Federal Interagency Sedimentation Conference (FIASC). Federal Energy Regulatory Commission, Las Vegas, Nevada: PS32 – PS39.

Happ, S. C., G. Rittenhouse & G. C. Dobson, 1940. Some principles of accelerated stream and valley sedimentation. Technical Bulletin No. 695, U.S. Dept. of Agriculture, Washington.

House, R., V. Crispen & J. M. Suther, 1991. Habitat and channel changes after rehabilitation of two coastal streams in Oregon. Am. Fisheries Soc. Symp. 10: 150–159.

Hupp, C. R., 1992. Riparian vegetarian recovery patterns following stream channelization: A geomorphic perspective. Ecology 73: 1209–1226.

Jandel Scientific, 1995. SigmaStat Statistical Software User's Manual. Jandel Scientific, San Rafael.

Jungwirth, M., S. Muhar & S. Schmutz, 1995. The effects of recreated instream and ecotone structures on the fish fauna of an epipotamal river. Hydrobiologia 303: 195–206.

Karr, J. R., 1991. Biological integrity: A long-neglected aspect of water resource management. Ecol. Appl. 1: 66–84.

Kesel, R. H. & E. G. Yodis, 1992. Some effects of human modifications on sand-bed channels in southwestern Mississippi, USA. Environ. Geol. Wat. Sci. 20: 93–104.

84

Knight, S. S. & C. M. Cooper, 1991. Effects of bank protection on stream fishes. In Proceedings of the Fifth Federal Interagency Sedimentation Conference (FIASC USDA, ARS), National Sedimentation Laboratory, Oxford: 13-34 – 13-39.

Kuhnle, R. A., C. V Alonso & F. D. Shields Jr., 1997. Geometry of scour holes around spur dikes, an experimental study. . In Wang, S.Y., E. Langendoen & F. D. Shields, Jr. (eds), Management of Landscapes Disturbed by Channel Incision, Stabilization, Rehabilitation, and Restoration, University of Mississippi, University, Mississippi: 283–287.

Lamberti, G. A. & M. B. Berg, 1995. Invertebrates and other benthic features as indicators of environmental change in Juday Creek, Indiana. Natural Areas J. 15: 249–258.

Lohnes, R. A., 1997. Stream degradation and stabilization: the Iowa experience. In Wang, S.Y., E. Langendoen & F. D. Shields, Jr. (eds), Management of Landscapes Disturbed by Channel Incision, Stabilization, Rehabilitation, and Restoration, University of Mississippi, University, Mississippi: 35–44.

Lyons, J., 1992. The length of stream to sample with a towed electrofishing unit when fish species richness is estimated. North Am. J. Fish. Management 12: 198–203.

Matheney & C. F. Rabeni. 1995. Patterns of movement and habitat use by northern hog suckers in an Ozark stream. Trans. Am. Fish. Soc. 124: 886–897.

Matthews, W. J., 1986. Fish faunal structure in an Ozark Stream: Stability, persistence and catastrophic flood. Copeia 388–397.

Meffe, G. K. & A. L. Sheldon. 1988. The Influence of habitat structure on fish assemblage composition in southeastern blackwater streams. Am. Midland Naturalist 120 (2): 225–240.

Menzel, B. W., J. B. Barnum & L. M. Antosch, 1984. Ecological alterations of Iowa prairie-agricultural streams. Iowa State J. Res. 59: 5–30.

Paller, M. H., 1994. Relationships between fish assemblage structure and stream order in South Carolina Coastal Plain streams. Trans. Am. Fish. Soc. 123: 150–161.

Parker, G. & D. Andres, 1976. Detrimental effects of river channelization. In Proceedings of the Symposium on Inland Waters for Navigation, Flood Control & Water Diversions: 1248–1266.

Pearsons, T. N., H. W. Li, & G. A. Lamberti. 1992. Influence of habitat complexity on resistance to flooding and resilience of stream fish assemblages. Trans. Am. Fish. Soc. 121: 427–436.

Rabeni, C. F. & M. A. Smale, 1995. Effects of siltation on stream fishes and the potential mitigating role of the buffering riparian zone. Hydrobiologia 303: 211–219.

Rapport, D. J., A. Reiger & T. C. Hutchison, 1985. Ecosystem behavior under stress. Am. Nat. 125: 617–640.

Rebich, R. A., 1993. Preliminary trend analyses of stream discharge and sediment data for the Demonstration Erosion Control Project, North-Central Mississippi, July 1985 through September 1991. In Daniel, B.J. (ed.), Proceedings of the Twenty-third Mississippi Water Resources Conference, Mississippi State University, Mississippi State: 98–109.

Rinaldi, M., & A. Simon, In press. Bed-level adjustments in the Arno River, Italy. Geomorphology.

Ross, S. T. & W. M. Brenneman, 1991. Distribution of freshwater fishes in Mississippi. Completion report, Dingell-Johnson project F-69, Mississippi Dept. Wildlife, Fisheries, and Parks, Bureau of Fisheries and Wildlife, Jackson, 548 pp.

Schoof, R. R., P. B. Allen & G. A. Gander, 1986. Evolution of two dredged channels in Oklahoma. In Proceedings of the Fourth Federal Interagency Sedimentation Conference (FIASC),Washington, DC: 5-93–5-102.

Schlosser, I. J., 1987. A conceptual framework for fish communities in small warmwater streams. In Matthews W. J. & D. C. Heins (eds), Community and Evolutionary Ecology of North American Stream Fishes. University of Oklahoma, Norman.

Schlosser, I. J., 1995. Critical landscape attributes that influence fish population dynamics in headwater streams. Hydrobiologia 303: 71–81.

Schlosser, I. J., 1990. Environmental variation, life history attributes, and community structure in stream fishes: Implications for environmental management and assessment. Envir. Management 14: 621–628.

Schlosser, I. J. & P. L. Angermeier, 1990. The influence of environmental variability, resource abundance, and predation on juvenile cyprinid and centrarchid fishes. Polish Archives of Hydrobiology 37: 265–284.

Schumm, S.A., M. D. Harvey & C. C. Watson, 1984. Incised channels: morphology, dynamics and control. Water Resources Publications, Littleton, Colorado.

Shankman, D. & T. B. Pugh, 1992. Discharge response to channelization of a coastal plain stream. Wetlands 12: 157–162.

Shields, F. D. Jr., & Cooper, C. M. 1994. Riparian Wetlands and Flood Stages. In Controneo G. V. & R. R. Rumer (eds), Hydraulic Engineering '94, Proceedings of the 1994 Conference, American Society of Civil Engineers, New York: 351–355.

Shields, F. D. Jr. & Hoover, J. J. 1991. Effects of channel restabilization on habitat diversity, Twentymile Creek, Mississippi. Regulated Rivers: Res. Management 6(3): 163–181.

Shields, F. D. Jr. & R. H. Smith, 1992. Effects of large woody debris removal on physical characteristics of a sand-bed river. Aquat. Conservation: Mar. Freshwater Ecosystems 2: 145–163.

Shields, F. D. Jr., S. S. Knight & C. M. Cooper, 1994. Effects of channel incision on base flow stream habitats and fishes. Environ. Management 18: 43–57.

Shields, F. D. Jr., S. S. Knight & C. M. Cooper, 1995a. Use of the index of biotic integrity to assess physical habitat degradation in warmwater streams. Hydrobiologia 312: 191–208.

Shields, F. D. Jr., A. J. Bowie & C. M. Cooper, 1995b. Control of streambank erosion due to bed degradation with vegetation and structure. Wat. Resources Bullet. 31: 475–489.

Shields, F. D. Jr., Knight, S. S., & Cooper, C. M., 1995c. Rehabilitation of watersheds with incising channels in Mississippi. Wat. Resources Bullet. 31(6): 971–982.

Simon, A. 1989a. A model of channel response in disturbed alluvial channels. Earth Surface Processes Landforms 14, 11–26.

Simon, A., 1989b. The discharge of sediment in channelized alluvial streams. Wat. Resources Bull. 25: 1177–1188.

Simon, A. & S. Darby, 1997. Disturbance, channel evolution, and erosion rates: Hotophia Creek, Mississippi. In Wang, S.Y., E. Langendoen & F. D. Shields, Jr. (eds), Management of Landscapes Disturbed by Channel Incision, Stabilization, Rehabilitation, and Restoration, University of Mississippi, University, Mississippi: 476–481.

Simon, A., & C. R. Hupp, 1986. Channel evolution in modified Tennessee channels. In Proceedings of the Fourth Federal Interagency Sedimentation Conference (FIASC), Washington, DC: 5-71–5-82.

Simon, A., M. Rinaldi, and G. Hadish, 1996. Channel evolution in the loess area of the Midwestern United States. In Proceedings of the Sixth Federal Interagency Sedimentation Conference, Interagency Advisory Committee on Water Data, Washington, DC: III-86–III-93.

Simonson, T. D., J. Lyons & P. D. Kanehl, 1994. Guidelines for evaluating fish habitat in Wisconsin streams. Forest Service General Technical Report NC-164. Forest Service-United States Department of Agriculture, St. Paul.

Slack, L. J., 1992. Water-quality and bottom-material-chemistry data for the Yazoo River Basin Demonstration Erosion Control Project, North-Central Mississippi, February 1988–September 1991. U.S. Geological Survey Open-File Report 92-469. U.S. Geological Survey, Jackson: 197 pp.

Swales, S. & K. O'Hara, 1983. A short-term study of the effects of a habitat improvement programme on the distribution and abundance of fish stocks in a small lowland river in Shropshire. Fish. Management 14: 135–144.

TerHaar, J. & E. Herricks, 1989. Management and development of aquatic habitat in agricultural drainage systems. WRC Research Report No. 212, Project No. G1560-05, PB90-173790.

Thorne, C. R., 1997. Fluvial processes and channel evolution in the incised rivers of north-central Mississippi. In Wang, S.Y., E. Langendoen & F. D. Shields, Jr. (eds), Management of Landscapes Disturbed by Channel Incision, Stabilization, Rehabilitation, and Restoration, University of Mississippi, University, Mississippi: 23–34.

Wallerstein, N., C. R. Thorne & M. W. Doyle, 1997. Spatial distribution and impact of large woody debris in northern Mississippi. In Wang, S.Y., E. Langendoen & F. D. Shields, Jr. (eds), Management of Landscapes Disturbed by Channel Incision, Stabilization, Rehabilitation, and Restoration, University of Mississippi, University, Mississippi: 145–150.

Watson, C. & D. Biedenharn, In Press. Hydrologic response in incised channels. In Proceedings of the Wetlands Engineering and River Restoration Conference 1998, American Society of Civil Engineers, New York.

Whitten, C. B. & D. M. Patrick, 1981. Engineering geology and geomorphology of streambank erosion (Report 2: Yazoo River Basin Uplands, Mississippi). Technical Report GL-79-7, Report 2 of a Series.

Winger, P. V., 1981. Physical and chemical characteristics of warmwater streams: A review. In Krumholz, L. A. (ed.), The Warmwater Streams Symposium, Southern Division, American Fisheries Society, Lawrence, Kansas: 32–44.

Winger, P. V., C. M. Bishop, R. S. Glesne & R. M. Todd, 1976. Evaluation study of channelization and mitigation structures in Crow Creek, Franklin County, Tennessee, and Jackson County, Alabama. Final Report for Contract No. AG47 SCS-00141, U. S. Soil Conservation Service, Nashville.

Woltemade, C. J. & K. W. Potter, 1994. A watershed modeling analysis of fluvial geomorphologic influences on flood peak attenuation. Wat. Resources Res. 30: 1933–1942.

Wyzga, B., 1996. Changes in the magnitude and transformation of flood waves subsequent to the channelization of the Raba River, Polish Carpathians. Earth Surface Processes Landforms 21: 749–763.

Yearke, L. W., 1971. River erosion due to channel relocation. Civil Eng. ASCE 41: 39–41.

Yount, J. D. & G. J. Niemi, 1990. Recovery of lotic communities and ecosystems from disturbance: A narrative review of case studies. Envir. Management 14: 547–569.

Appendix A. Fishes captured from five streams in northwestern Mississippi 1991–1995

Family	Species	PC pre	PC post	HC pre	HC post	BB pre	BB post	GC pre	GC post	TT pre	TT post	Sum
Aphredoderus	*Aphredoderus sayanus*				2					1	1	4
Atherinidae	*Labidesthes sicculus*				6					4	5	15
	Carpiodes carpio	28	8		58	1		7	2			104
Catostomidae	*Erimyzon oblongus*				6	97	187	231	190	1	12	724
	Ictiobus bubalus		22		49			3				74
	Ictiobus niger				1							1
Centrarchidae	*Lepomis macrochirus*	86	259	28	585	54	32	127	74	417	1149	2811
	Minytrema melanops									7	14	21
	Moxostoma poecilurum		5	1	2						1	9
	Lepomis megalotis	137	928	13	277	83	82	396	646	0	8	2570
	Lepomis cyanellus	38	196	43	192	30	10	130	245		1	885
	Lepomis marginatus		71		45		43		62		7	228
	Lepomis gulosus		1		1		3			7	24	36
	Lepomis punctatus									9	13	22
	Lepomis microlophus		1							2	4	7
	Lepomis humilis										5	5
	Micropterus punctulatus	22	134	3	120	22	9	7	12	88	34	451
	Micropterus salmoides		8		9	1	1	7	9	26	14	75
	Pomoxis nigromaculatus									7		7
	Pomoxis annularis		1		1							2
Clupeidae	*Dorosoma cepedianum*	7	3	1	40					2	1	54
	Campostoma anomalum						41			1		42
Cyprinidae	*Cyprinella camura*	252	1256	17	738	858	884	1529	687			6221
	Cyprinella lutrensis		27			67	20	172	42			328
	Cyprinella venusta	90	292	92	625	88	163	107	41	107	5	1610
	Cyprinus carpio		2		6					3	1	12
	Hybognathus nuchalis	23									1	24
	Luxilus chrysocephalus		31			194	859	114	275		4	1477
	Lythrurus umbratilis						13	4			5	22
	Notemigonus crysoleucas		1			10				1	2	14
	Notropis rafinesquei	60	115	4	348	1738	1064	606	208			4143
	Notropis atherinoides		29	3	95	0	4		7	72	603	813
	Notropis texanus										5	5
	Opsopoeodus emiliae										1	1
	Pimephales notatus	31	509	9	76	147	1059	426	378	9	1	2645
	Pimephales vigilax		3		34		81					118
	Semotilus atromaculatus		7			507	1342	78	64			1998
Esocidae	*Esox niger*										1	1
Fundulidae	*Fundulus olivaceus*	8	310	15	348	260	352	188	262	16	80	1839
	Fundulus notatus		14				4		2			20
Ictaluridae	*Ameiurus natalis*	20	43	2	11	44	94	145	86		3	448
	Ictalurus punctatus	90	52		64		2			3	1	212
	Noturus phaeus		1				1					2
Lepisosteidae	Lepisosteus oculatus	3	30		25					5		63
	Lepisosteus osseus		11		13							24
Percidae	*Etheostoma whipplei*		26		3	278	368	42	112			829
	Etheostoma parvipinne					6						6
	Etheostoma nigrum		1				2				3	6
	Etheostoma sp		2									2
	Etheostoma chlorosomum									1		1
	Percina sciera	1	46		26	7		8				88
Petromyzontidae	*Icthyomyzon gagei*										5	5
Poeciliidae	*Gambusia affinis*		30	15	66	9	67	51	40		3	281
Sciaenidae	*Aplodinotus grunniens*									2	5	7
	Total number	896	4475	246	3872	4501	6787	4378	3444	791	2022	31 412
	Total species	16	36	14	31	22	27	21	21	24	34	54

All streams except TT were severely degraded by channel incision and attendant erosion. HC and GC were rehabilitated using vegetation and stone structure in February 1992 and March 1993, respectively. For PC and HC, 1991 represents pre-rehabilitation conditions, while 1992–95 were post-rehabilitation. At the other three sites, 1991–92 was pre-rehabilitation, and 1993–95 was post-rehabilitation.