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INCISED STREAM PHYSICAL HABITAT RESTORATION WITH STONE WEIRS

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ABSTRACT

The initial results are described of the restoration of 1 km long reach of Goodwin Creek, a channelized sand and gravel bed warm-water stream draining a 21 km^2 lowland catchment in north-west Mississippi. Although a series of grade control weirs and bank protection works had been constructed before restoration, sediment production from channel erosion remained high ($>1200\,t\,km^{-2}\,y^{-1}$) and aquatic habitats were of poor value. At base flow, only 5–20% of the water area was classified as pool habitat (depth >30 cm and velocity $<10\,cm\,s^{-1}$). Restoration works were designed to be compatible with existing channel stabilization works and economic critera. Stone was added to extend the existing groynes across the base flow channel to create 18 small weirs. The effects of restoration were quantified by collecting fish and physical habitat data semi-annually for two years before and during the first year after restoration from the restored reach and from two reference streams. Restoration increased pool habitat availability, overall physical heterogeneity, riparian vegetation, shade and woody debris density. After restoration, mean width, depth and velocity exhibited changes of +56, +150 and −56%, respectively, despite discharge levels that averaged 43% lower during data collection periods. The pool area increased to 72% of the water area. Bed types became more heterogeneous, with larger fractions of clay, debris and riprap, and less sand and gravel. The fish response to restoration measures was modest, but distinct. Before restoration cyprinids and centrarchids comprised 74 and 11%, respectively, of the numerical catch, but 32 and 55% after restoration. Fish species composition and relative abundance after restoration were slightly more similar to that of the non-incised reference site than before restoration. The median lengths of five selected fish species were greater after restoration, but were unchanged at reference sites.

KEY WORDS: physical habitat restoration; incised streams; stone weirs

INTRODUCTION

Channel incision is a common and very destructive mode of accelerated channel erosion (Galay, 1983), particularly in watersheds smaller than about 500 km^2 drained by channels with little or no geological control on stream base level. Increases in channel width due to incision-related erosion triggered by channelization of as high as 100 to 1000% have been observed in Missouri (Emerson, 1971), Iowa (Daniels, 1960), New Hampshire (Yearke, 1971), Oklahoma (Barclay, 1980; Schoof *et al.*, 1986), Mississippi (Kesel and Yodis, 1992), Alberta (Parker and Andres, 1976), Tennessee (Simon and Hupp, 1986) and Indiana (Barnard, 1977). Suspended sediment yields reported for actively incising watersheds include 704–1673 $t\,km^{-2}\,y^{-1}$ (Rebich, 1993) and 157–876 $t\,km^{-2}\,y^{-1}$ (Simon, 1989).

The ecosystems of incised channel corridors suffer adverse impacts due to elevated sediment loads, hydrological modifications associated with the isolation of the stream from its flood plain and the loss of stream habitat structures (pools, riffles, woody debris, undercut banks, etc.). 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 to 31% of the water area had a depth >30 cm. Similar measures for a non-incised reference channel were 35 cm and 42%, respectively. Channel deepening and enlargement eliminate physical and biological stream–flood plain

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interactions and accentuate flow peaks. Shields and Cooper (1994) analysed 11 concurrent storm stage hydrographs from a non-incised stream flanked by forested wetlands and an adjacent incised channel. Average times to peak and event durations were 31 and 42% longer for the non-incised stream, even though its watershed was only 42% as large as that of the incised channel. In the absence of erosion-resistant outcrops, the effects of channel lowering propagate upstream and affect the entire watershed.

As incision results in long-term morphological transformation at the landscape scale, its potential for degrading biotic integrity is greater than point or non-point source pollution (Karr, 1991) and unassisted recovery can be expected to be slow (Yount and Niemi, 1990). Rehabilitation of aquatic habitats in incised streams, particularly warm water, sand-bed streams, has received little attention relative to other types of stream habitat manipulation (Swales and O'Hara, 1983; Shields *et al.*, 1992; Osborne *et al.*, 1993), and careful scientific evaluation of fish habitat restoration projects in all stream types is often absent or inadequate (Frissell and Nawa, 1992). Despite the high levels of biodiversity found in warm-water streams of the south-eastern USA and the disproportionate number of channelized streams in that region (Brookes, 1988), few stream restoration investigations have been conducted. In a comprehensive review by Lyons and Courtney (1990), only three of 25 studies dealing with habitat improvement in warm-water streams were sited in the south-eastern USA.

Some authorities counsel against attempting habitat restoration in flashy, unstable, sand-bed streams with high sediment loads and eroding banks (Van Haveren and Jackson, 1986; Brookes, 1989; Heede and Rinne, 1990; Frissell and Nawa, 1992). However, scour holes adjacent to stabilization structures in unstable, incised channels have been found to support more species of fish and larger fish than the surrounding channel habitats without structures (Winger *et al.*, 1976; Shields and Hoover, 1991; Knight and Cooper; 1991), and fish populations in incised channels have been shown to respond quickly (about one year) to the restoration of pool habitats (Swales and O'Hara, 1983; Shields *et al.*, 1993). On the basis of these findings, we hypothesized that the fish community in an incised, warm-water stream could be significantly improved by increasing the availability and locational stability of pool habitats. Herein we present the short-term (less than one year) results of a restoration project in an incised channelized stream in north-western Mississippi. Restoration planning and design were based on geomorphic and ecological concepts and were intended to produce a low cost approach suitable for routine, widespread incorporation into watershed stabilization projects.

Table I. Description of study sites

	Site IR	Site R	Site NR
Meaning of designation	Incised, reference	Incised, restored	Non-incised reference
Watershed area (km^{-2})	16	21	38
Chanelization after watershed aforestation and valley sedimentation	Before 1957	Before 1937	Headwater tributaries before early 1930s
Response to perturbations	Head-cutting and incision	Head-cutting and incision: channel width doubled 1968–1977; cross-sectional area increased 25% 1977–1983 (Murphey and Grissinger, 1985)	
Conditions at the time of this study			
Land use (%)			
Urban: crops: pasture: forest	0: 13: 54: 33	0: 13: 60: 28	12: 15: 22: 48
Channel width (m)	25–29	20–70	10–14
Channel depth (m)	4–5	4–5	2
Sinuosity	1·13	1·12	1·25
Thalweg slope ($m\,km^{-1}$)	2·5	1·6	2·1
Bed material D_{50}(mm)	0·42–1·5	0·24–8·5	0·06–0·55
Observed base flow ($m^3\,s^{-1}$)	0·01	0·06	0·4

IR = Incised reference stream; R = incised, restored stream; and NR = non-incised reference stream.

STUDY SITES

One kilometre reaches in each of three fourth-order north-west Mississippi watersheds were selected for study (Figures 1 and 2, and Table I) based on their locations, disturbance histories and the availability of hydrological and geomorphological data. Two of the watersheds [Goodwin Creek (R) and Bobo Bayou (IR)] had experienced widespread incision; the third [Toby Tubby Creek (NR)] had not. R was selected for restoration, whereas IR and NR were studied to provide points of reference.

Watersheds were located in the East Gulf Coastal Plain Physiographic Province along the bluffline of the Mississippi River Valley. In all three watersheds, the soils, topography and land use are typical of many streams along the eastern side of the lower Mississippi River flood plain. 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 stratigraphic units, all of which are erodible (Grissinger *et al.*, 1982). The mean annual rainfall was about 1400 mm. The available data indicated that water quality was adequate for the maintenance of healthy communities of aquatic organisms (Cooper and Knight, 1991; Shields *et al.*, 1994).

European settlement of the area containing the study watersheds, which began about 1830, 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 metres of sediments eroded from hillslopes (Happ *et al.*, 1940; Grissinger and Murphey, 1986) and swampy conditions developed due to impaired drainage. Landowners, acting as individuals and through drainage districts, attempted to reclaim valley lands by channelizing streams and constructing drainage ditches between about 1840 and 1930. Most of these efforts were poorly planned and ineffective. A second round of channelization and construction of major flood-control reservoirs on receiving streams by federal agencies occurred between about 1930 and 1960.

Sites R and IR responded to channelization and the reduction of flood stages on receiving streams by rapid incision. Incision often occurred by upstream progression of knick-points ('head-cutting'), as described by Whitten and Patrick (1981) and Smith and Patrick (1991). In contrast, the non-incised site NR experienced valley sedimentation and channel aggradation subsequent to European settlement, but channelization was

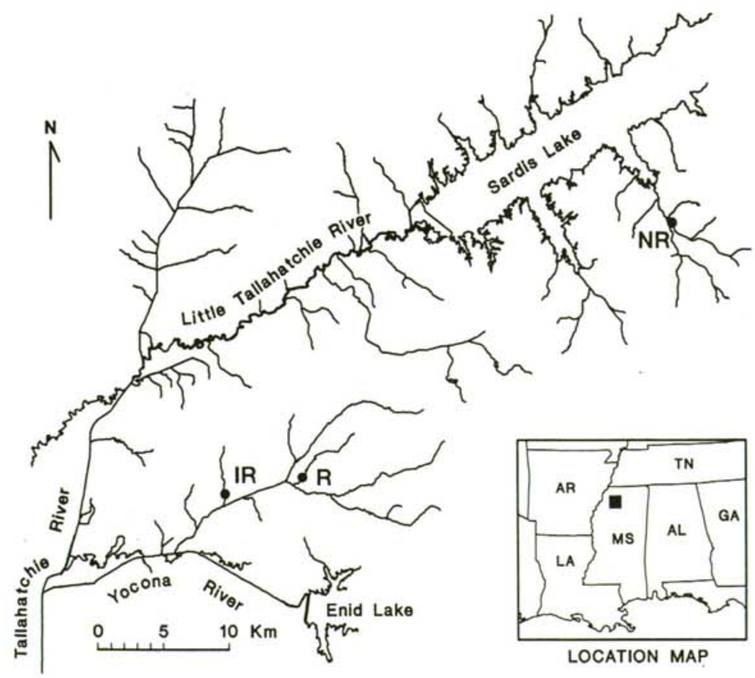

Figure 1. Location of study sites

Figure 2. Photographs of study reaches (a) Site IR; (b) site R before restoration; (c) site R after restoration; and (d) site NR

confined to reaches upstream and incision did not occur. Furthermore, the base level for site NR was elevated rather than lowered because a dam was placed on its receiving stream downstream of the confluence rather than upstream as for the other three study reaches. During this study (1991–1993), all three study sites had relatively stable banklines and bed profiles. However, all continued to receive large sediment loads from uplands areas within their watersheds that were still actively incising.

Before restoration, R channel morphology and aquatic habitats were similar to IR (Table I). At base flow, less than about 10% of the water area was classified as pool habitat (depth >30 cm and velocity <10 cm s^{-1}). Local bed and bank erosion and deposition were common, but the overall channel alignment and thalweg profile were stable. Sediment yield averaged about 1200 tkm^{-2} y^{-1} (Grissinger *et al.*, 1991). Sites R and IR differed in two aspects: shade canopy and woody debris were more common at site IR, and structures were common at site R and within its watershed. Stone riprap groynes and longitudinal toe protection were placed at selected locations within and just upstream of the R study reach in 1989–1991, and a series of grade control weirs was constructed throughout the R watershed in about 1980 (Bowie and Sansom, 1986). There were no stabilization structures such as weirs, revetments or groynes in the IR study reach or nearby.

Although the two incised channels were flanked by cultivated fields, site NR was bordered by a 600 m wide band of forested wetlands (Figure 2d). During summer much of the channel was almost entirely covered by tree canopy, and large woody debris formations were common year-around. Beaver dams were common on adjacent flood plains and were occasionally found in the NR channel itself.

RESTORATION

Restoration schemes for unstable channels should be planned with consideration of the factors that induced the instabilities (Kondolf, 1990). In incising watersheds, the restoration potential is greatest in the lower, relatively stable reaches where the general trend is towards deposition (Van Haveren and Jackson, 1984; Shields *et al.*, 1992) and vegetated bars form along the channel (Hupp and Simon, 1991; Hupp, 1992). Reaches upstream of grade-control structures are less likely to undergo general destabilization leading to the failure of habitat structures (Heede and Rinne, 1990). The overall goal of restoration in these systems should be to accelerate natural processes of recovery of channel equilibrium, riparian vegetation and stream–flood plain interaction (Van Haveren and Jackson, 1986; Shields *et al.*, 1992). Specific objectives for the site R restoration project were to increase pool habitat availability and riparian cover. The project was intended to demonstrate and test a restoration approach that would be feasible for integration with ongoing channel and watershed stabilization activitivies. Furthermore, habitat structures and plant materials had to be maintenance-free and durable enough to withstand the high energy, high sediment transport conditions typical of incised channels.

For restoration, stone was added to extend existing groynes across the base flow channel to create 18 small weirs at intervals roughly equal to six times the average base flow channel width in a 1 km long reach immediately upstream from a grade-control structure (Heede and Rinne, 1990) (Figure 3). Stone weirs were selected because they have been successfully used to create and maintain pools or to function as artificial riffles in a variety of channelized streams (Swales, 1982; Shields, 1983; Geiger and Schroter, 1983; Lewis and Williams, 1984; Edwards *et al.*, 1984; Takahashi and Higashi, 1987). Although weirs are more likely to suffer structural failure than spurs (current deflectors) (Babcock, 1982; Frissell and Nawa, 1992), they also are more likely to significantly improve physical habitat (Swales and O'Hara, 1983). In plan, the weirs were v-shaped with vertices upstream to focus the overtopping flow and resultant scour in the centre of the channel. The crests were about 2 m wide and the side slopes were equal to the angle of repose of the stone. Crest elevations were 0·6 m higher than the existing stream bed, except for central gaps 1 m wide, for which crest elevations were set equal to stream bed elevation. One end of each weir was tied into an existing groyne, and the opposite end was keyed into the low flow channel bank a distance of 3 m. Stone size, ranged from 0·2 to 450 kg, with 50 to 85% of the stones weighing less than 36 kg. The total volume of stone required for restoration was equal to 17·5% of the stone already emplaced for bank stabilization.

Eighty to 90 dormant native black willow (*Salix nigra* spp.) posts 1·5 m long by 8–30 cm diameter were planted immediately downstream from each weir using a ram mounted on an hydraulic hoe. Posts were

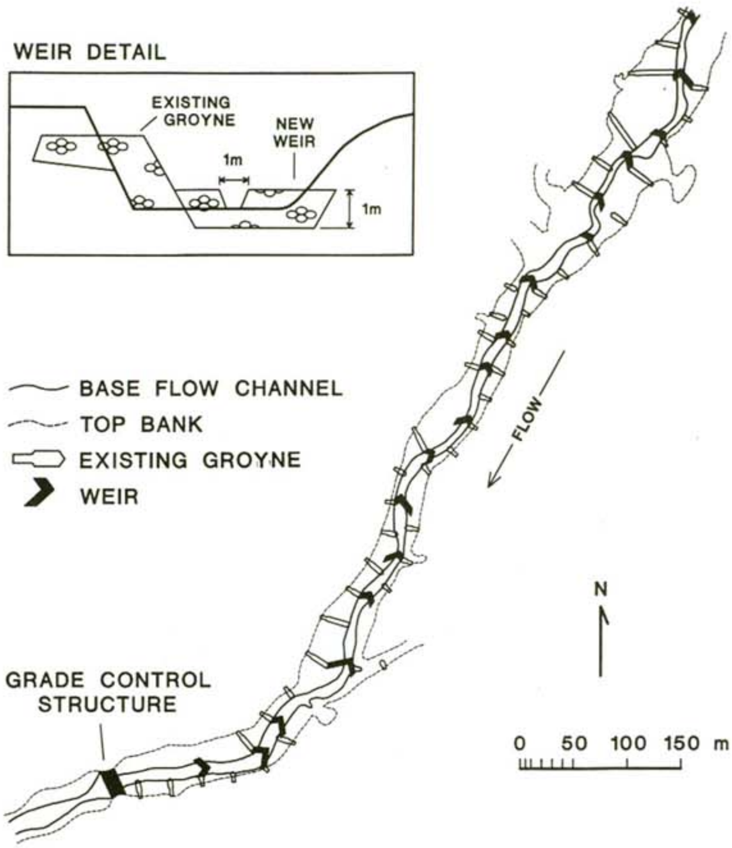

Figure 3. Plan of restoration details for site R study reach

planted 0·8–1·2 m deep on approximately 1·2 m centres in three staggered rows that ran roughly parallel to the channel. Every effort was made to preserve the small amount of existing woody vegetation along the banks. Posts were harvested from an area within 1·5 km of the planted sites and were planted butt down (with the same vertical orientation they had before harvest) within 24 hours of cutting.

METHODS

The effects of restoration were quantified by collecting fish and physical habitat data semi-annually (spring and autumn) for two years before (1991–1992) and during the first year after restoration (1993). The non-incised reference site (NR) was sampled only during the autumn of the first year, and because of strong seasonal influences in fish collections, the NR autumn 1991 data were not included in the analysis. Physical aquatic habitat variables were sampled from each study reach. The design of physical data collection and analysis procedures was partially based on work by others (e.g. Gorman and Karr, 1978; Berkman *et al.*, 1986; Schlosser, 1987; Plafkin *et al.*, 1989; Petersen, 1992). Data were collected at 75–140 points at each site arrayed at roughly equal intervals along 15–24 transects. Side channels were sampled only if they were hydraulically connected to the main channel at the up- and downstream ends. Depth was measured with a wading rod, velocity was measured at 0·6 depth using a Marsh–McBirney current meter (brand names provided for information only) and bed type was visually classified as clay, sand, gravel, riprap, vegetation, debris (leaves, twigs or larger woody debris) or other (e.g. periphyton or man-made items). At each point, the current meter probe was oriented to measure maximum velocity regardless of its direction.

Velocities were visually estimated when the depth was less than 4–5 cm. Where two or more bed types were closely mingled, the classification was based on the type that dominated the area covered by the 'foot' of the wading rod (an 8 cm diameter disc). Water surface width was measured with a tape at each transect. The presence and number of beaver dams and man-made structures (e.g. weirs, revetments, jacks, etc.) within or immediately downstream from each reach were noted, and the area of each large woody debris formation in the plane of the water surface was visually estimated. Discharges were measured using the wading rod and current meter at the downstream end of each study reach. Basic water quality parameters (pH, conductivity, dissolved oxygen, temperature) were measured using a Martek water quality meter. Physical habitat data were used to compute summary statistics for pre- and post-restoration periods. Depth, bed type and velocity data were used to compute a Shannon-type index of heterogeneity (Gorman and Karr, 1978).

The dimensions of scour holes below each weir were measured during midsummer base flow (i.e. not during the passage of a storm hydrograph) using a tape and a wading rod. A thalweg profile was obtained using surveying instruments three months after construction and was compared with a profile obtained in 1985 by the US Army Corps of Engineers. Finally, detailed maps of a 100 m long reach near the middle of the restoration project were obtained by measuring depths at 1 m intervals along cross-sections spaced 5 m apart two years before and three months after construction.

Fish were collected in spring and autumn concurrently with physical data using a Coffelt BP-4 backpack-mounted electroshocker. One of the electroshockers was mounted on a small two-person fishing raft and was used to sample the scour holes at site R that were too deep to wade during the autumn post-restoration collection. Three 100 m stream subreaches within each 1 km study reach were fished for approximately 900 seconds of electric field application. Fish longer than about 15 cm were identified, measured for total length and released. Smaller fish and larger fish that could not be identified in the field were preserved in 10% formalin solution and transported to the laboratory for identification and measurement. Owing to uncertainties in identification, larval forms and individuals shorter than 2 cm were not included in the data for analysis. Fish data were divided into sets representing collections made before and after restoration of site R.

Species abundance data for each site before and after restoration were used to compute Sorenson quantitative similarity indices (Magurran, 1988) and Spearman rank correlation coefficients (Glantz, 1992). Similarity indices were computed for each pair of sites using the formula: similarity $= 2jN/(aN + bN)$, where $jN =$ the sum of the lower of the two abundances recorded for species found in both sites, $aN =$ total number of individuals at site A and $bN =$ total number of individuals at site B. Perfect similarity $= 1{\cdot}0$, whereas sites with no species in common would have a similarity of zero. Effects of habitat restoration on fish size were examined by comparing non-parametric statistics for the length of five selected species that were well represented in collections made before and after restoration.

RESULTS

During the year after construction, the restored reach experienced peak discharges of 65·2 and 44·8 $m^3 s^{-1}$.

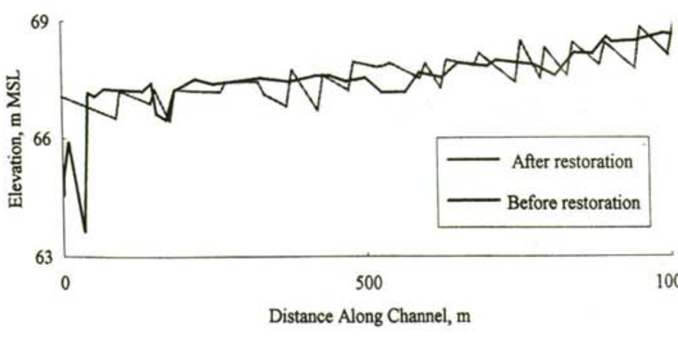

Figure 4. Site R thalweg profile before and after restoration

These values were equalled or exceeded 14 and 30 times, respectively, during the 10 years before our study. The physical response to weir construction was rapid. Scour holes formed downstream of each structure, with average (SD) dimensions of 11 (2·6) m wide by 14 (6·9) m long by 130 (42) cm deep. In many cases, gravel deposits formed immediately upstream of the weirs, creating riffle-like habitats. The thalweg profile of the entire restoration project showed that the series of 18 weirs and associated scour holes formed a saw-tooth of stair-step profile typical of streams with naturally well-developed pool–riffle series (Figure 4). In contrast, the pre-restoration profile contained only four pools at irregular intervals, one of which appeared to be a knick-point. Examination of detailed maps of the 100 m reach showed local increases in water width of over 100% and local increases in depth of 0–1000%.

Restoration promoted physical changes in site R habitats that made them less similar to site IR and more similar to NR (Figure 5). Mean width, depth and velocity exhibited changes of +56, +150 and −56%, respectively, after restoration (Table II), in spite of the fact that discharge during data collection periods

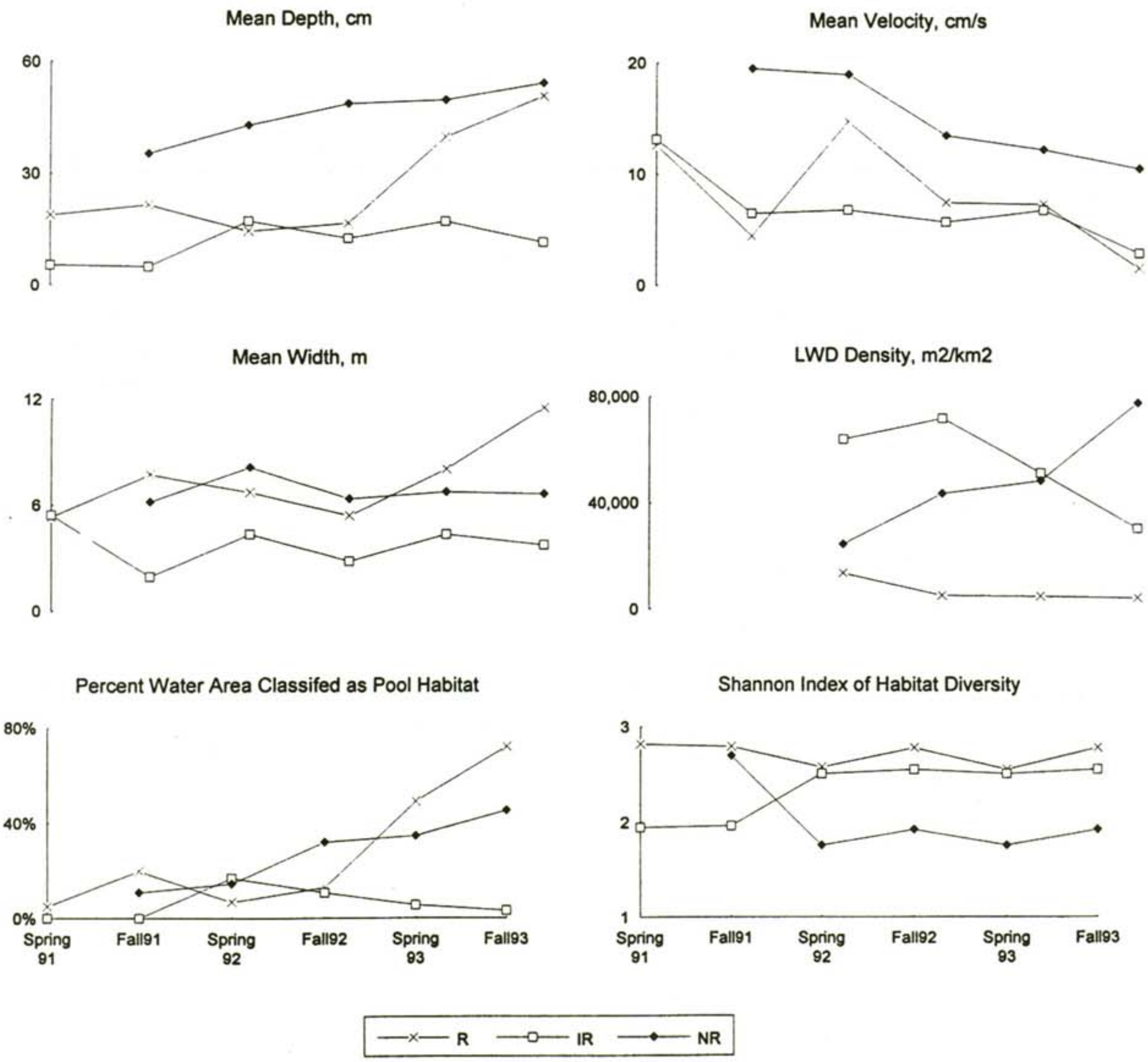

Figure 5. Selected descriptors of base flow physical habitat conditions measured concurrent with fish sampling

Table II. Summary statistics describing physical habitat before and after restoration. Statistics in bold are significantly different ($p < 0{\cdot}01$) before and after restoration (within site comparison)

	Site IR	Site R	Site NR
Mean discharge ($L\,s^{-1}$)			
Before	14 (37, 6, 10, 3)	63 (120, 40, 53, 38)	423 (468, 378)
After	8 (13, 4)	48 (52, 44)	354 (374, 334)
Mean depth (cm)			
Before	**8 ± 10**	**18 ± 15**	45 ± 22
After	**11 ± 9**	**45 ± 34**	52 ± 24
Mean velocity ($cm\,s^{-1}$)			
Before	8 ± 10	**9 ± 12**	**16 ± 10**
After	6 ± 9	**4 ± 10**	**11 ± 7**
Mean width (m)			
Before	3.0 ± 1·9	**6·2 ± 2·7**	6·3 ± 0·6
After	4·0 ± 2·3	**9·7 ± 3·8**	6·7 ± 0·7
Mean LWD density ($m^2\,km^{-2}$)			
Before	67 808	9222	33 913
After	40 690	4354	62 959
Percentage of water area classified as pool habitat*			
Before	7	11	23
After	4	61	40

IR = Incised reference stream; R = incised, restored stream; and NR = non-incised reference stream.
* Arbitrarily defined as all areas with depth >30 cm and velocity <10 $cm\,s^{-1}$.

averaged 43% lower. For quantification, the pool area was arbitrarily defined as the water area with depth >30 cm and velocity <10 $cm\,s^{-1}$. The year before restoration, physical habitat data collected concurrently with fish sampling showed that pool area comprised only 7 and 13% of the site R water area on spring and autumn sample dates, respectively, which was typical of the incised reference site, IR (Figure 5). Three months after restoration, the pool area increased to 49% of the water surface area. Seven months after construction, habitats had been further modified by beaver (*Castor canadensis*) dams constructed on top of seven of the stone weirs and at two locations between weirs. The pool area increased to 72%, the mean water surface width was nearly double the pre-restoration value, and the mean water depth approached the mean depth found in the non-incised reference stream, NR. The mean velocity declined sharply in response to beaver dams.

Habitat complexity increased due to the addition of deep pools below the weirs and swift, shallow divided flow reaches around gravel bars just upstream of weirs (Figure 2c). Sandy substrates in the less shady, shallow incised channels were often covered by a veneer of periphyton after prolonged periods of low flow, but periphyton was not observed on the bed of the more heavily shaded NR, which was also deeper and swifter. Discounting the influence of periphyton, the bed composition at the two reference sites remained roughly the same in pre- and post restoration periods, whereas bed types at R became more heterogeneous, with

Table III. Distribution of bed types at base flow in per cent

Bed type	IR pre	IR post	R pre	R post	NR pre	NR post
Sand	66·3	38·8	46·4	36·2	65·3	78·0
Sand covered by living or dead periphyton	0·5	23·3	7·0	16·3		
Gravel	22·7	15·5	35·8	24·8		
Clay	3·1	2·6	6·0	9·9	22·7	6·0
Debris	7·2	19·8	1·3	8·5	12·0	16·0
Riprap	0·5	0·0	3·5	4·3		

IR = incised reference stream; R = incised, restored stream; and NR = non-incised reference stream.

Table IV. Number of fish species (individuals) captured from an incised stream and two reference streams in Mississippi before and after restoration. Restoration works were completed two months before the spring 1993 samples were collected.

Season	IR	R	NR	All sites
Spring 1991	12 (192)	14 (174)		366
Autumn 1991	19 (1820)	16 (1388)		3208
Spring 1992	16 (318)	14 (892)	14 (55)	1265
Autumn 1992	13 (775)	16 (1320)	15 (101)	2196
Spring 1993	16 (1060)	16 (800)	12 (48)	1908
Autumn 1993	19 (768)	11 (313)	11 (266)	1347
All seasons	27 (4933)	20 (4887)	24 (470)	40 (10 290)

IR = incised reference stream; R = incised, restored stream; and NR = non-incised reference stream.

larger fractions of riprap, debris and clay (Table III). Physical habitat heterogeneity, as measured by a Shannon-type index, remained unchanged after restoration ($p = 0{\cdot}53$, two-tailed t-test), despite the larger fraction of area influenced by the scour pools and backwater effects of beaver dams (Figure 5). Physical habitat Shannon indices for NR were significantly lower than for R and IR (ANOVA, $p = 0{\cdot}01$), possibly because of the absence of gravel at NR.

Throughout the study, large woody debris density at site R was about an order of magnitude lower than for the other two sites, reflecting the relative scarcity of woody vegetation through the R study reach (Figure 5). Effects of planting willow posts on habitats were limited by high rates of post mortality (~80%). Mortality was caused by competition by kudzu and other herbaceous species, impermeable soils (Grissinger and Bowie, 1984), droughty soils and beaver activity.

Fish collections included 10 290 individuals representing 40 species. There were 48 to 1820 individuals and 11 to 19 species per collection (collection = all fishes from a site on a given date) (Table IV). Catch per unit of effort ranged 1·5 to 75·8 fish per minute, with the lowest figure occurring at the deeper, swifter non-incised reference site (NR, Table V). Expansion of pool habitat reduced the sampling efficiency at site R because deep pools are more difficult to electrofish than shallow runs. However, the average catch rates for centrarchids before and after restoration were equal (4·1 fish per minute).

The non-incised reference site, NR, yielded 13 species that were not found at the other two sites, whereas the incised reference site, IR, produced five species not found elsewhere. The cyprinid *Notropis atherinoides* comprised 53% of the catch from NR, but was not captured elsewhere. No unique species was captured from site R. To facilitate comparison of conditions before and after restoration, fish data were bisected into subsets corresponding to periods before and after restoration (Table VI). Subset data for each site contained

Table V. Catch per unit of effort (fish per minute) for an incised stream and two reference streams in Mississippi before and after restoration. Restoration works were completed two months before the spring 1993 samples were collected

Season	IR	R	NR	Mean
Spring 1991	10·1	5·7		7·9
Autumn 1991	75·8	68·8		72·3
Spring 1992	18·5	34·3	3·4	18·7
Autumn 1992	33·2	51·1	4·4	29·6
Spring 1993	39·8	25·2	1·5	22·2
Autumn 1993	30·4	5·6	9·3	15·1
Mean	36·4	25·7	4·7	24·2

IR = Incised reference stream; R = incised, restored stream; NR = non-incised reference stream.

equal effort across seasons (spring and autumn). Collections from IR and from R were numerically dominated by two cyprinids: the bluntface shiner (*Cyprinella camura*) and the Yazoo shiner (*Notropis rafinesquei*). These two species, which comprised more than 50% of the catch by number from IR and R before restoration and more than 38% afterwards, specialize in shallow, sandy stream habitats (Suttkus, 1991) and are ubiquitous in the unstable, damaged channels in this region (Shields *et al.*, in press). Six species comprised over 75% of the pre-restoration catch from IR and R, but the same six species comprised less than 10% of the catch by number from NR. Centrarchids comprised an average of 26% of catches from

Table VI. Fish species captured from an incised stream and two reference streams in Mississippi before and after restoration

Family	Genus	Species	IR-pre	IR-post	R-pre	R-post	NR-pre	NR-post	Totals
Aphredoderidae	*Aphredoderus*	*sayanus*					1		1
Atherinidae	*Labidesthes*	*sicculus*					3	1	4
Catostomidae	*Carpiodes*	*carpio*	1		7				8
	Erimyzon	*oblongus*	75	43	224	51		1	394
	Minytrema	*melanops*					1	1	2
Centrarchidae	*Lepomis*	*cyanellus*	24	5	98	51			178
		gulosus		1			2	6	9
		macrochirus	30	8	95	20	34	47	234
		marginatus		30		12			42
		megalotis	74	40	225	269			608
		punctatus					1		1
	Micropterus	*punctulatus*	13	4	4		11	8	40
		salmoides	1	1	3	4	8	5	22
Cyprinidae	*Cyprinella*	*camura*	559	442	1324	299	3	4	2631
		lutrensis	55		172				227
		venusta	39	89	95	30	5	1	259
	Cyprinus	*carpio*					3		3
	Luxilus	*chrysocephalus*	141	267	112	44			564
	Lythrurus	*fumens*					14		14
		umbratilis		13				4	17
	Notemigonus	*crysoleucas*	10						10
	Notropis	*atherinoides*					39	210	249
		rafinesquei	1215	244	584	181			2224
	Pimephales	*notatus*	125	193	423	46	5		792
		vigilax		81					81
	Semotilus	*atromaculatus*	311	180	78	17			586
Esocidae	*Esox*	*niger*						1	1
Fundulidae	*Fundulus*	*olivaceus*	165	47	154	37	5	11	419
Ictaluridae	*Ameiurus*	*natalis*	29	26	83	28			166
	Ictalurus	*punctatus*					2		2
	Noturus	*phaeus*					6		6
Moronidae	*Morone*	*saxatilis*		38					38
Percidae	*Etheostoma*	*nigrum*		1					1
		parvipinne	6						6
		proeliare					5		5
		sp						1	1
		whipplei	220	34	41	16			311
	Percina	*sciera*	3		8		7	10	28
Poeciliidae	*Gambusia*	*affinis*	9	41	44	8			102
Sciaenidae	*Aplodinotus*	*grunniens*					1	3	4
Grand total			3115	1828	3774	1113	156	314	10 290
Catch per unit effort (fish/min)			37	35	37	13	2	5	23
Number of species			22	22	19	16	18	15	40

IR = Incised reference stream; R = incised restored stream; and NR = non-incised reference stream.

Table VII. Sorenson quantitative similarity indices for fish collections before (1991–1992) and after (1993) restoration at R. Within-site comparisons are in bold

	IR pre	IR post	R pre	R post	NR pre	NR post
IR pre	1·00					
IR post	**0·563**	1·00				
R pre	0·581	0·502	1·00			
R post	0·415	0·553	**0·432**	1·00		
NR pre	0·039	0·032	0·034	0·066	1·00	
NR post	0·035	0·033	0·039	0·057	**0·455**	1·00

IR = Incised reference stream; R = incised, restored stream; and NR = non-incised reference stream.

NR, but only 5 and 11% of catches from IR and from R before restoration, respectively. After restoration centrarchids comprised 32% of the catch from R. In contrast, cyprinids declined from 74 to 55% of the numerical catch.

Effects of restoration on species composition and relative abundance were examined by computing Sorenson quantitative similarity indices (Magurran, 1988) and Spearman rank correlation coefficients (Glantz, 1992) using the data presented in Table VI. A matrix of similarity indices is presented in Table VII, which shows that R pre- and post-restoration collections were 43% similar, whereas the reference site collections across the same time periods were only slightly more similar. Incised channel (IR and R) collections were similar to one another before (58%) and after (55%) restoration, whereas NR remained distinct from the other two sites. Similarity between sites R and NR increased from 3% before restoration to 6% afterwards. These results were confirmed by the Spearman correlation analysis (Table VIII). Only the 22 most abundant species were used in the correlation analysis (species with grand total numbers for all sites and dates >20). Collections from incised channels were significantly correlated with one another before and after restoration, but NR collections were distinctive.

Pool habitats created by restoration provided habitat for larger fish. Length distributions for three centrarchids (*Lepomis cyanellus*, *L. macrochirus* and *L. megalotis*), the catostomid *Erimyzon oblongus*, and an ictalurid (*Ameiurus natalis*) were compared using a non-parametric test (Mann–Whitney rank sum). As shown in Table IX, all five species tended to be larger at R after restoration, with median differences significant at the $p \leqslant 0{\cdot}005$ level in four of the five cases. Similar changes in fish size were not observed at the two reference sites.

Water quality data indicated the conditions were within acceptable criteria for aquatic life before and after restoration at all three sites.

DISCUSSION

Erosion and deposition patterns after weir construction were typical of patterns observed for low weirs in

Table VIII. Spearman rank order correlation coefficients for fish collections before and after restoration at R. Coefficients in bold are significant at $p = 0{\cdot}005$

	IR pre	IR post	R pre	R post	NR pre	NR post
IR pre	1·000					
IR post	**0·624**	1·000				
R pre	**0·810**	0·512	1·000			
R post	**0·708**	**0·629**	**0·829**	1·000		
NR pre	−0·198	−0·333	−0·153	−0·233	1·000	
NR post	−0·142	−0·338	−0·116	−0·167	**0·880**	1·000

IR = Incised reference stream; R = incised, restored stream; and NR = non-incised reference stream.

Table IX. Comparison of length distribution of five selected fish species before and after restoration. IR = incised reference stream, R = incised, restored stream, NR = non-incised reference stream. Statistics in bold indicate median values were different at $p \leqslant 0.005$

Family	Genus	Species	Statistic	IR-pre	IR-post	R-pre	R-post	NR-pre	NR-post
Catostomidae	*Erimyzon*	*oblongus*	Median	10·1	8·8	**9·0**	**13·0**		
			75th centile	11·9	10·8	**10·3**	**16·0**		
			p	0·467		**<0·0001**			
			No.	75	43	224	51	0	1
Centrarchidae	*Lepomis*	*cyanellus*	Median	5·8	5·4	7·5	8·1		
			75th centile	8·1	7·3	9·3	9·7		
			p	0·931		0·332			
			No.	24	5	98	51	0	0
Centrarchidae	*Lepomis*	*macrochirus*	Median	7·8	5·5	**6·0**	**8·0**	7·0	6·0
			75th centile	8·7	5·8	**7·7**	**10·0**	13·0	11·4
			p	0·111		**0·002**		0·500	
			No.	30	8	95	20	34	47
Centrarchidae	*Lepomis*	*megalotis*	Median	5·5	6·7	**7·2**	**8·0**		
			75th centile	7·8	8·5	**9·5**	**9·2**		
			p	0·04		**0·004**			
			No.	74	40	225	269	0	0
Ictaluridae	*Ameiurus*	*natalis*	Median	9·0	9·6	**11·1**	**14·0**		
			75th centile	10·6	14·0	**13·0**	**17·5**		
			p	0·349		**0·005**			
			No.	29	26	83	28	0	0

steeper streams (Klassen and Northcote, 1986), larger rivers (Walker *et al.*, 1992) and in physical models (Barton and Winger, 1973; Lin and Chen, 1992). A rapid physical response to low weirs is consistent with observations by others. House *et al.* (1991) reported two- to five-fold increases in pool area along 10·4 km of two Oregon coastal streams within one year of the installation of 289 habitat structures, including 114 that fully spanned the channel. Carline and Klosiewski (1981) found that rock weirs placed in a channelized Ohio stream failed at high flow and therefore created habitats similar to shallow riffles, but did not form downstream scour holes.

The stone weirs facilitated beaver activity. Beaver dams constructed on top of the weirs amplified the effects on physical habitats. Pool habitats were greatly expanded and riffle habitats were reduced. Winger *et al.* (1976) noted that restoration weirs along Crow Creek, Alabama were less effective when placed so close together that the riffle areas were submerged. Although the loss of riffle areas is not desirable, creating conditions that facilitate beaver-induced stream alterations may represent a return to the more natural conditions that prevailed before European settlement (Naiman *et al.*, 1986; 1988). Beaver translocation has been used to restore riparian conditions along eroding streams in south-western Wyoming. The channel at site R is so deeply incised that the beaver activity had no effect on drainage or flooding of adjacent fields, and during the following winter all traces of beaver dams were removed by high flows. As noted earlier, beaver ponds are common riparian features along the non-incised reference stream (NR).

Weirs or low dams are often used to restore stream habitats without evaluation of their ecological effects. The generalization of the available published data points towards shifts in species composition after weir placement and increases in fish biomass due to reductions in current velocity and increases in water depth and habitat volume (Swales, 1989). Our findings emphasize these ideas. However, much of the published data reviewed by Swales (1989) is based on studies of cold-water streams, principally in Norway. Studies of warm-water stream fish populations have revealed the importance of pool availability (TerHaar and Herricks, 1989; Ebert *et al.*, 1991). The presence of larger individuals of some species (Schlosser, 1982; Meffe and Sheldon 1988; Lobb and Orth, 1991), species richness (Schlosser, 1987), and total fish numbers (Foltz, 1982) are positively related to pool habitat.

Our findings generally correlate well with work by others on the impacts of weirs on fish populations of channelized, warm-water streams. Cooper and Knight (1987) compared fish populations of natural pools and weir-stilling basins in rapidly eroding, north-west Mississippi streams. Weir-stilling basins supported fewer species than natural scour holes, but larger individuals, particularly centrarchids and ictalurids. Shields and Hoover (1991) reported reaches of severely eroded, channelized stream in north-east Mississippi containing grade-control weirs which supported fish populations with higher levels of species richness, species diversity and faunistically distinct from populations of adjacent reaches without weirs. Carline and Klosiewski (1985) found that rock weirs were ineffective in creating pools in a channelized, warm-water stream in Ohio, and accordingly had insignificant effects on fish numbers, biomass and species richness. They attributed the low centrarchid density to the absence of pool habitats. Conversely, Edwards *et al.* (1984) reported positive ecological responses to weir-like mitigation structures placed in another channelized stream in Ohio. Structures created riffle-like habitats and pools with maximum depths of 2·5 m. Four to six years after channelization and mitigation, game fish (centrarchids and ictalurids) were more abundant in the reach with mitigation structures than in a channelized reach without structures, which was dominated by non-game fish (catostomids and cyprinids). Hortle and Lake (1983) found that a small weir and associated cover and pool habitats partially mitigated the effects of channelization in a river in southern Victoria. The reach just upstream of the weir had significantly higher species richness, total fish biomass, numerical density and standing crop than two other channelized reaches without weirs. Swales and O'Hara (1983) studied the effects of the addition of restoration structures to a channelized river in England. During the first year after construction, two pool-dwelling fishes (*Leuciscus leuciscus* L. and *Leuciscus cephalus* L.) exhibited increases in population density and standing cross of 276 and 150%, respectively, in reaches upstream of two small weirs.

The biological results of restoration at site R were less pronounced than those observed in an adjacent watershed, where the extension of existing stone spurs doubled the availability of pool habitats. There, during the year after restoration, the number of fish species and mean fish length doubled, and fish biomass increased by an order of magnitude (Shields *et al.*, 1993). However, this adjacent site is only 1·7 km upstream from the confluence with a larger river that evidently serves as a source of organisms which rapidly move into new habitats created by restoration. The only obstacle to the influx of fish is a low grade control weir which is submerged during high stages. There is a grade control structure just downstream of R that presents an approximately 2 m vertical obstacle to upstream bound fish, and this location is about 11·4 km upstream from the confluence with a major river.

Based on the observed changes in mean depth and width, the volume of aquatic habitat in the study reach increased by a factor of four after restoration. Sport and commercial fishing were not significant factors. We therefore hypothesize that resident populations were unable to expand fast enough to fully exploit the new habitat, and migration from downstream reaches was prevented by the grade control weir. Reduced population density in addition to the inherent difficulty of electrofishing deeper pools led to a reduced catch per unit of effort after restoration. The fact that the catch per unit of effort for centrarchids was constant after restoration is noteworthy, and a positive indication of the effectiveness of restoration.

Additional monitoring of these sites continues (weekly water quality sampling, semi-annual sampling for physical habitat, benthic macroinvertebrates, fish; biannual channel and bed-size surveys and continuous hydrological monitoring) and the real value of the restoration concepts and designs described here will be quantified after the system has had more time to respond (Swales and O'Hara, 1983). The goal of this effort is to produce '... a channel in dynamic equilibrium that supports a self-sustaining and functionally diverse community assemblage' (Osborne *et al.*, 1993). Clearly, more time is required to evaluate progress towards this goal. Two distinct possibilities exist for project failure: sedimentation and erosion. Deposition of sediments could eliminate the new pool habitats and local scour could undermine the weirs. The first possibility seems unlikely in spite of the plentiful supply of sediments from upstream because the pools are created by scouring action from weirs rather than backwater effects. Pools created by backwater from beaver dams are eliminated during the events which carry most of the sediments because the beaver dams are washed out. As for scour, many of the stone weirs may fail if time series of water and sediment discharge produce certain conditions. However, the probability of failure is unknown. Hydraulic design tools which would allow economic analysis of structural performance in response to a complex combination of time varying three-

Table X. Screening criteria for low stone weirs for warm-water stream aquatic habitat restoration

Stream characteristic	Criteria	Range for seven successful applications reported by Shields *et al.* (1992)
Water quality	Acceptable for aquatic life	
Habitat	Pool-deficient	
Hydrology	Unsteady flow	$18\,m^3\,s^{-1} < Q_{bankfull} < 283\,m^3\,s^{-1}$
Slope	Moderate	0·0007–0·0470
Bed material	Must be erodible. However, highly erodible materials will require more careful design to prevent weir undermining or flanking	Clay to cobbles
Channel width	Small to moderate	8·2–38·1 m (bottom width)
Channel banks	Stable and well-defined enough to anchor both ends of the structure 1–2 m into banks	
Channel bed	Not actively degrading (lowering). Moderate aggradation tolerable	

dimensional flow phenomena in a small channel with a wide range of sediment sizes are presently beyond the state of the art.

The preliminary data presented here indicate that low stone weirs hold much promise as cost-effective tools for warm-water stream restoration. These structures should be designed and constructed to increase pool habitat availability by creating and maintaining downstream scour holes. Rudimentary screening criteria for use of low weirs based on this study and work by others (Weshe, 1985; Brookes, 1989; Shields *et al.*, 1992) are presented in Table X.

Restoration of 1 km of stream at site R may possibly benefit adjacent reaches. Expanded pools may offer refugia to larger fish from upstream reaches during prolonged droughts, and higher levels of primary and secondary production in large pools created by stone weirs and beaver dams may contribute to biomass exported downstream. Existing data are not adequate to test these hypotheses, however.

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