LONG-TERM MONITORING

Stream Ecosystem Restoration: Is Watershed-Scale Treatment Effective Without Instream Habitat Rehabilitation?

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With erosion reduced, the fish community responded positively to restoration of a degraded stream reach.

Many researchers have written that the integrity of stream ecosystems is governed by watershed-level landscape variables, such as geology and land use (Hynes 1975, Roth and others 1996, Allan and others 1997, Schlosser 2002), while others have coined the term "riverscape" to emphasize the role of landscape-scale variables in stream ecosystems (Fausch and others 2002). Accordingly, some researchers and practitioners prefer a "passive approach" to stream restoration. This method consists of halting anthropogenic practices and process disturbances that prevent recovery, while suggesting that reach-scale manipulations are typically ineffective (Beschta and others 1994). Others admit the dual importance of reach- and watershed-scale influences, but emphasize the importance of processes acting at larger spatial scales (Roth and others 1996, Wissmar and Beschta 1998). They favor the restoration of functional processes rather than forms (for example, restoring conditions leading to island formation as opposed to constructing islands with equipment) (Ward and others 2002).

Clearly, efforts toward ecosystem restoration should be informed by an understanding of the main drivers involved in degradation and attendant ecological responses. In warmwater streams, physical habitat degradation associated with channel erosion triggers many of the symptoms of ecosystem distress syndrome (Rapport and others 1985) including reductions in the stability and diversity of aquatic ecosystems; elimination of the longer-lived, larger species; and a tendency to favor small, short-lived opportunistic species (Menzel and others 1984, Berkman and Rabeni 1987, Ebert and Filipek 1988, Meffe and Sheldon 1988, Paller 1994, Shields and others 1994 and 1995b, Lamberti and Berg 1995, Rabeni and Smale 1995).

A conceptual framework for fish communities in small warmwater streams proposed by Isaac Schlosser (1987) offers insight into the effects of such erosion on fish. Schlosser argued that fish community structure in warmwater streams is the expression of a complex interplay between abiotic and biotic factors, generally trending from highly variable communities in shallow headwaters to more stable communities in larger downstream reaches. Species richness and fish size increase and population density decreases along a physical gradient of habitat heterogeneity and pool development. Pool habitat development is a key factor in spatial and temporal heterogeneity because of its effects on habitat

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volume and temporal stability. Pools offer larger fishes, including top predators, refugia from terrestrial predators, high velocities, and high temperatures. Accordingly, reaches with uniform conditions and shallow depths support populations dominated by juvenile minnows and are devoid of larger fishes. Unstable physical conditions (for example, 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 minnows and younger sunfish (*Alectis ciliaris*) and suckers (*Catostomus* spp.). Finally, as pool depth and volume increase further, major shifts occur in fish age and size structure, species composition, and trophic structure. Communities feature fewer, larger predatory fish, like bass (*Micropterus* spp.), more pool-dwelling suckers, and fewer small, insect and invertebrate-eating fish (invertivores) and omnivores due to increased predation and competition for refugia. Schlosser (1987) 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. The importance of physical factors in governing fish community structure has been further established by studies across biogeographically distinct regions (Lamouroux and others 2002). Shields and his colleagues (1998a) have adapted Schlosser's framework to identify possible pathways for rehabilitation.

These concepts, along with the results of studies in degraded streams in the southeastern United States, suggest that simply increasing the availability of stable pool habitats might trigger limited ecological recovery (Cooper and Knight 1987, Knight and Cooper 1991, Shields and Hoover 1991, Shields and others 1997). Accordingly, we developed and implemented two stream habitat rehabilitation projects in 1992 and 1993 that consisted of modifying existing stone erosion control structures and planting willow cuttings (Shields and others 1995b and 1995c). The conceptual foundation for both projects was linked to prevailing geomorphic processes that were actively transforming channel morphology during the course of the study (Shields and others 1992 and 1998a). Treated and untreated streams were monitored before and after restoration. Effects on fish and their habitats generally were positive during the first three to four years following the restoration effort (Shields and others 1998a).

Many researchers have called for emphasis on post-project monitoring for stream ecosystem rehabilitation (for example, Kondolf 1995), but reports of long-term effects on warmwater, sand-bed streams are rare. Evaluations of passive (Hill and Platts 1998) and active (Schmetterling and Pierce 1999) stream restorations in salmonid streams in the western United States are more common. Bryant (1995) proposed a "pulsed monitoring strategy" consisting of a series of short-term (3-5 years), high-intensity studies separated by longer periods (10-15 years) of low-density data collection. In this article, we describe the results of sampling one of the two previously mentioned rehabilitated streams 10 and 11 years after restoration.

Sites and Methods

Hotophia and Peters Creeks are located in northwestern Mississippi within the hilly region of the upper Yazoo River watershed. Due to their inclusion in federally funded erosion control programs (Hudson 1997), both creeks and their watersheds have been described in several publications (Whitten and Patrick 1981, Little and others 1982, Knight and Cooper 1990, Simon and Darby 2002). The spatial centers of the watersheds are separated by only about 9 miles (15 km), and thus experience nearly identical weather. Topography, soils, and land use are also similar (Table 1), producing similar hydrologic regimes.

All perennial channels within both watersheds were channelized at least once between about 1880 and 1965, and the fluvial response from roughly 1965 to the present is consistent with conceptual models of incised channel evolution (Schumm and others 1984, Simon 1989). Extensive erosion control works were installed throughout both watersheds between 1986 and 1996.

Habitat rehabilitation activities were performed on a 0.6-mile (1-km) reach near the mouth of Hotophia Creek (Reach 1) from January through February 1992 (Shields and others 1995b). Prior to the project, aquatic habitat was typical of many incised channels in the region (Shields and others 1994): 1) base flow conditions were extremely shallow, 2) pool habitats were rare and temporally unstable, 3) substrate was dominated by shifting sands, and 4) woody debris was scarce (Figure 1a). Eroding banks were common

Table 1. Description of 1-km long study reaches.

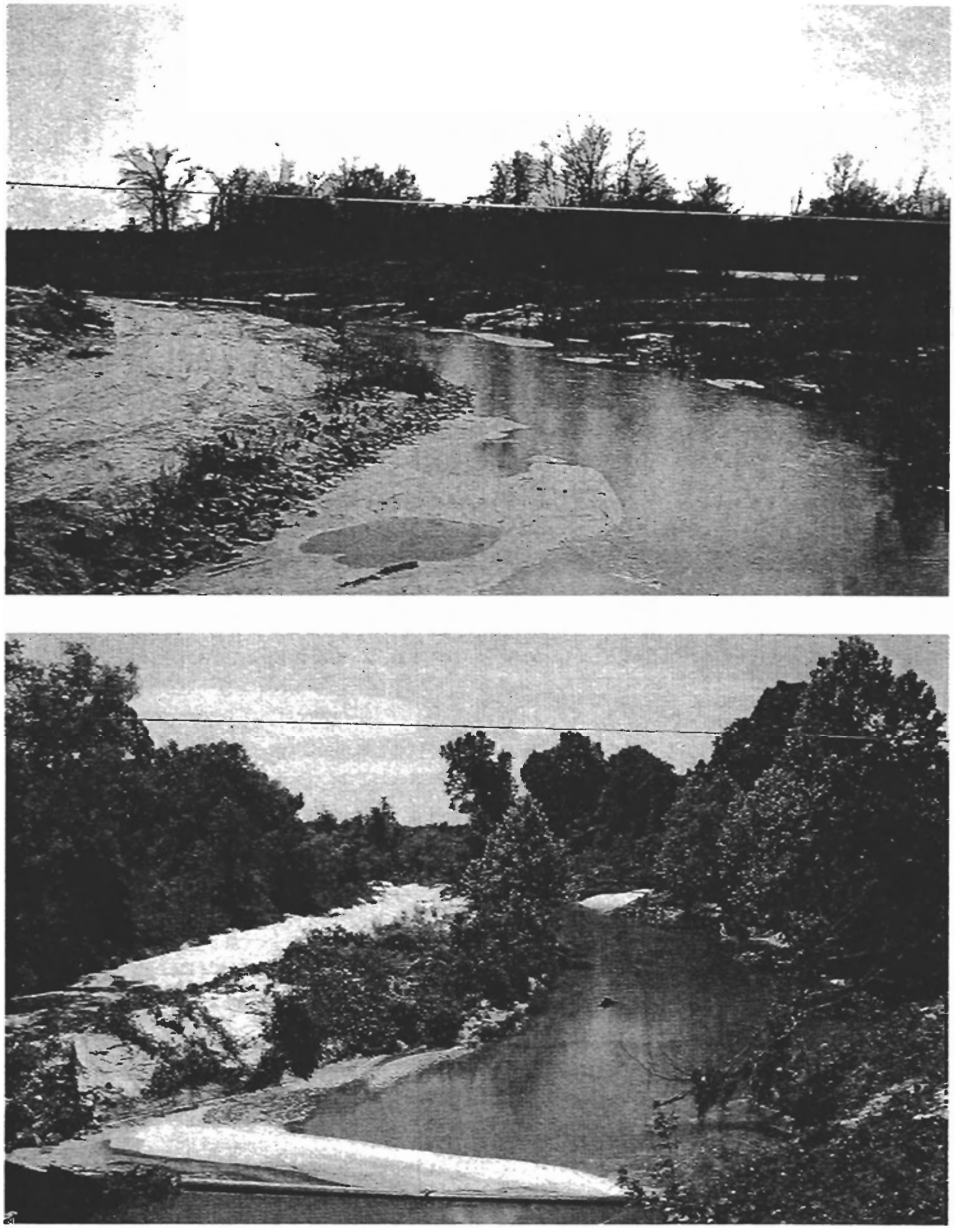
	Hotophia Creek Reach 1 (restored)	Hotophia Creek Reach 2 (untreated)	Peters Creek (untreated)
Drainage area, km²	91	91	205
Land use	8% row crop, 40% idle or pasture, 52% forest		11% row crop, 53% idle or pasture, 36% forest
Slope	0.001	0.001	0.0009
Sinuosity	1.4	1.3	1.1
Bed material	Sand	Sand	Sand and gravel
Bank height, m	3-7	3-7	2-6
Channel width, m	40-60	40-60	55-85
Structures	Low-drop grade control structure immediately downstream, stone toe, stone spurs	Stone toe protection along one bank	Stone toe protection along one bank

and riparian vegetation was sparse (Shields and others 1998a). Although stone structures had been placed for bank protection, they provided very little habitat and, as a result, fish populations were typical of those found in many incised streams (Shields and others 1994).

Habitat restoration consisted of extending existing stone spur dikes riverward to provide stony substrate and trigger formation of stable pool habitats. Workers planted willow posts in the sandbar opposite the stone spurs, and placed a low windrow of stone along the planted bars (Figure 1b). The primary objective of the project was to increase the availability of stable pool habitat at base flow. Dike extensions required addition of only 10 percent more stone than was previously placed in the same reach for bank stabilization.

Three 0.6-mile-long reaches near the mouths of Hotophia and Peters Creek watersheds were sampled for this study (Table 1). Reach 2 of Hotophia Creek was located immediately upstream from Reach 1 and thus experienced virtually identical hydrology and water quality. Reach 1 of Hotophia Creek represented a restoration strategy that included watershed-scale erosion controls plus instream habitat treatments, while Reach 2 and Peters Creek represented a "watershed-scale treatment only" strategy. In this article, we refer to Reach 1 of Hotophia Creek as "treated," while Reach 2 and Peters Creek are termed "untreated."

We sampled the physical habitat conditions in Reach 1 of Hotophia Creek and Peters Creek at baseflow during May-June and September-October in 1991-1995 and 2002, and in Reach 2 during the same months as Reach 1 in 2002. We used similar procedures to sample the study reaches in all years. Since pool-riffle sequences were absent or poorly developed, we could not locate sampling reaches in relation to habitat units. Therefore, we sampled four 100-m-long zones distributed along each of the 1-km-long study reaches. We measured water depth with a wading rod at five, regularly spaced grid points along five transects placed at uniform intervals within each zone. We classified bed material type visually at each point where depth was measured. We measured flow width at each transect, and made visual estimates regarding the number and size of woody debris formations, dominant type and size of bank vegetation, and percent canopy. Instantaneous discharge was measured at one transect within each reach using a wading rod and velocity meter.



Figures 1a and 1b. View of the Reach 1 section of Hotophia Creek in northwestern Mississippi immediately after construction (1a) and in 2002 (1b). This reach included watershed-scale erosion controls and instream habitat treatments. Note toe and willow post plantings on left and the spur dike extension on right side of top photograph. *Photos courtesy of F. Douglas Shields, Jr.*

While collecting physical habitat data in 1991-1995 and 2002, three members of our team used dip nets and a Coffelt BP-1 backpack-mounted electroshocker to collect fish within each reach. We "fished" the same four 100-m-long zones for about ten minutes each. However, because water depth in Reach 1 had increased significantly by 2002, and since electrofishing efficiency is inversely related to water depth (Shields and others 2000), we felt the 2002 results were questionable.

In 2003, we resampled 25-m segments of each reach using explosive detonation

Table 2. Average of mean-daily water and sediment discharge before (water years 1988-1991) and after (water years 1993-2001) habitat rehabilitation.

	Hotophia Creek			Peters Creek		
	Before	After	Change (%)	Before	After	Change (%)
Mean annual precipitation, mm	1591	1283	-19	1591	1283	-19
Mean-daily water discharge, m^3/second	2.1	1.2	-71	4.6	2.8	-65
Mean-daily sediment load, tons	304	44	-593	823	235	-251

cord. Detonation cord is a rope-like, linear explosive that is commonly used to trigger groups of blasting charges. It has been used successfully for concussion sampling of fish populations where other methods fail because of water depth or strong current (Metzger and Shafland 1986). Explosive charges were composed of commercially available explosive detonation cord of 10.63 PETN/m and Number 8 electric blasting caps. Preparation for collecting was simple. A sampling reach (25 m) was measured and flagged 24 to 36 hours before sampling. Prior to sampling, we placed block nets simultaneously above and below the sampling reach. Detonation cord was secured by appropriately placed weights in a longitudinal fashion so that no length of detonation cord was more than 20 ft (6 m) from another length of cord or the streambank. The actual length of cord varied from 410 ft (125 m) to more than 820 ft (250 m) per site. We positioned the cord so that irregular stream areas were adequately sampled. Since depth was almost entirely 3 ft (1 m) or less and the concussive distance was 10 ft (3 m) laterally, the kill was virtually complete.

After sampling, we made visual observations to determine if there were any fish not swept by current into the downstream net. These observations revealed no live fish. Both nets were then pulled and the fish collected. Following collection, we identified fish that were more than 6 inches (15 cm) long, measured them for total length, and then disposed of the fish on-site. Smaller fish and fish that we could not identify were preserved in 10-percent formalin solution and transported to the laboratory for identification and measurement.

During water years 1987–2001, the U.S. Geological Survey recorded stage and discharge data within Reach 1 (station 07273100) and within the sampled reach of Peters Creek (station 07275530). [Editor's note: The term "water year" is used in USGS reports that discuss surface-water supply, and is defined as the 12-month period from October 1, for any given year through September 30, of the following year. The water year is designated by the calendar year in which it ends and which includes nine of the 12 months. Thus, the year ending September 30, 1999 is called the "1999" water year.] Daily mean suspended-sediment concentration and load were also recorded by the USGS during water years 1988 through 1997. In addition, biweekly grab samples (samples dipped from the surface at the channel centerline) from both sites were obtained and analyzed for suspended sediment concentration (total suspended solids, Clesceri and others 1998) by the National Sedimentation Laboratory (NSL) during the periods April 1985 to October 2001 and December 1991 to October 2001 for Peters and Hotophia Creeks, respectively. The record of daily mean suspended sediment concentration was extended through water year 2001 using suspended sediment ratings that were developed using these grab sample concentrations, daily mean water discharge records, and a regression formula relating mean-daily suspended sediment load reported by the USGS to the grab sample suspended sediment concentration reported by the NSL (Shields and Knight 2003).

Results

Suspended-sediment discharge fell sharply following restoration in both streams, due at least in part to drier conditions (Table 2). Hotophia Creek suspended sediment load dropped by a factor of six. Significant morphologic changes occurred within Reach 1 (Table 3). During the four years immediately following restoration, mean water depth increased only slightly, although the depth and size of scour holes associated with the stone spurs more than doubled (Shields and others 1995a). However, by 2002, the mean water depth in Reach 1 was 2.5 to 3 times greater than

Table 3. Effects of restoration on physical aquatic habitat at comparable discharges.

	Hotophia Creek (Reach 1—restored)			Hotophia Creek (Reach 2—untreated)	Peters Creek (untreated)	
Mean + standard deviation	Before (Fall 1991)	Short-term (Spring 1992)[a]	Long-term (Spring 2002)	Long-term (Spring 2002)	Short-term (Spring 1994)	Long-term (Spring 2002)
Instantaneous water discharge (m^3/s)	0.80	0.66	0.63	Not measured, but similar to reach 1	0.64	0.65
Mean water width (m)	18 + 4	16 + 3	16 + 6	13.4 + 3.1	22 + 8	16 + 6
Mean water depth (cm)	16 + 8	19 + 17	54 + 34	17 + 12	13 + 6	22 + 13
Fraction of bed covered with sand (%)	100	91	76	70	86	60
Average woody debris density, m^2/km^2	—	1,520	14,600	8,820	1,120	9,950
Bank line covered with trees or brush, %	—	19	55	56	15	36

[a]Treatments were placed in Winter 1992. The bed responded quickly to the stone structures, so widths and depths represent post-rehabilitation. However, woody debris and riparian vegetation responded more slowly, so Spring 1992 values of these variables represent pre-rehabilitation conditions.

Table 4. Effects of restoration on electrofishing collections. Means followed by different subscripts indicate significantly different distributions for the same reach but different time periods ($p < 0.05$, One-way ANOVA using log-transformed data or Kruksal-Wallis ANOVA on ranks). Before = 1991, Short term = 1992-1995, Long term = 2002.

Mean + standard deviation	Hotophia Creek (Reach 1—restored) Before (Fall 1991)	Short-term (Spring 1992)[a]	Long-term (Spring 2002)	Hotophia Creek (Reach 2—untreated) Long-term (Spring 2002)	Peters Creek (untreated) Short-term (Spring 1994)	Long-term (Spring 2002)
Number of species per 100 m sampling zone	$6 + 2^{a}$	$12 + 3^{b}$	$6 + 2^{a}$	9 + 2	$11 + 3^{a}$	$8 + 2^{b}$
Number of individuals per 100 m	$40 + 42^{a}$	$123 + 82^{b}$	$21 + 7^{a}$	44 + 15	$136 + 100^{a*}$	$82 + 39^{a*}$
Number of individuals captured per min of electrical field application	$9 + 10^{a}$	$15 + 12^{a}$	$2 + 1^{b}$	4 + 1	$19 + 13^{a}$	$9 + 6^{b*}$
Fish biomass (kg/100m)	$0.2 + 0.2^{a\$}$	$2.7 + 2.5^{b\$}$	$0.8 + 0.8^{a\$}$	0.8 + 0.8	$2.0 + 2.7^{a*}$	$1.1 + 0.9^{a*}$
Percentage of numerical catch comprised of minnows	$56 + 20^{a}$	$44 + 21^{a}$	$12 + 16^{b}$	31 + 12	$39 + 30^{a*}$	$37 + 33^{a*}$
Percentage of numerical catch comprised of sunfishes	$36 + 17^{a}$	$35 + 15^{a}$	$60 + 17^{b}$	44 + 16	$45 + 27^{a*}$	$48 + 29^{a*}$
Percentage of numerical catch comprised of suckers	$1 + 3^{a*}$	$6 + 12^{a*}$	$2 + 3^{a*}$	2 + 6	$1 + 3^{a*}$	$0 + 1^{a*}$
Length of fish, cm	$5 + 3^{a\$}$	$7 + 6^{b\$}$	$9 + 10^{c\$}$	8 + 6	$8 + 5^{a\$}$	$8 + 6^{a\$}$

*ß< 0.80

$Even log-transformed data were non-normally distributed, so a Kruksal-Wallis ANOVA on ranks was used.

[a]Treatments were placed in Winter 1992. The bed responded quickly to the stone structures, so widths and depths represent post-rehabilitation. However, woody debris and riparian vegetation responded more slowly, so Spring 1992 values of these variables represent pre-rehabilitation conditions.

the same reach before restoration, the untreated reach located upstream, and Peters Creek (Table 3). Both streams became narrower (referring to water width, not channel width), and bed material became more heterogeneous.

About 70 percent of the willow posts planted in Reach 1 died during the first two years after planting (Shields and others 1995b), likely due to poor soil conditions (Shields and others 1998b) or competition from exotic species (Shields and others 1995c). Other vegetation, however, invaded stone revetments and sandy berms, consistent with the conceptual incised channel evolution model (Simon 1989), roughly doubling woody cover on banks in both treated and untreated reaches. Woody debris density increased by an order of magnitude in both streams by 2002 (Table 3).

Fish communities in Hotophia Creek responded dramatically to habitat restoration over the short term (1992-1995, Table 4) with species richness, numbers, and biomass increasing by factors of 1.7, 3.0, and 14.0, respectively (Shields and others 1998a). Fish species composition shifted away from small minnows typical of shallow, incised streams toward larger-bodied, longer-lived sunfishes and suckers typical of less degraded streams in this region (Shields and others 1994, 1997, 1998a). Data collected in 1991 and prior to restoration in 1992 revealed that relatively large-bodied, pool-dwelling sunfishes comprised an average of 36 percent of the fish collected and smaller, highly tolerant minnows 67 percent of fish collections. In 2002, sunfishes comprised an average of 60 percent of the individuals captured from the treated reach, while the highly tolerant minnows comprised only 31 percent. The remaining catch was comprised of catfish (*Ictalurus* spp.), gar (*Lepisosteus osseus* or *Atractosteus spatula*), and suckers. Moreover, fish captured from Reach 1 in 2002 were larger than fish collected from the untreated sites.

Electrofishing in 2002 was hampered by an equipment-based bias that produced lower species richness and overall fish numbers in the deeper waters of Reach 1 when compared to the shallower depths of the untreated reaches. The data from the 2003 detonation cord fish collections revealed patterns similar to the short-term (1992-1995) trends.

The restored Reach 1 supported larger individuals, about 60 percent more species, and an order of magnitude more fish biomass than the untreated reaches. We also found fewer, but larger, fish in the restored reach, and the relative abundance of sunfishes was 43 percent—a level more closely emulating collections from a non-incised reference stream about 40 miles (25 km) away, which were 60-70 percent sunfish (Shields and others 1998a). By 2003, the distributions of fish biomass among family groups presented a particularly stark contrast among the three reaches: biomass of collections from untreated reaches of Hotophia and Peters creeks were 63 percent and 95 percent minnows, respectively. Similar collections from the treated reach of Hotophia were comprised of 75 percent suckers, 13 percent sunfishes and only 2 percent minnows as biomass.

Discussion and Conclusions

The expenditure of more than $3,900 per square mile ($100,000 per km²) for erosion control structures and a period of drier weather reduced suspended sediment yield from Hotophia Creek watershed by a factor of about six during the last decade. Determination of the relative importance of erosion controls in this decline is, however, beyond the scope of this paper. Nevertheless, we did observe that baseflow channels of all three stream reaches became narrower and deeper, consistent with changes expected for systems with declining sediment load (Werrity 1997) and increasing bank vegetation (Hey and Thorne 1986).

The 2002 electrofishing data from Peters Creek and Reach 2 of Hotophia Creek is strikingly similar to the 1991 data from Hotophia Creek and to the 1991-1995 data from Peters Creek (Table 4), which strongly suggests that baseflow channel narrowing and deepening and the significant improvements in water quality outlined above were insufficient to produce the recovery of fish populations in these two reaches. Conversely, habitat restoration structures (stone spurs) and vegetation placed along Reach 1 of Hotophia Creek in 1992 were followed by strong and progressive responses in fish populations consistent with the conceptual framework proposed by Shields and others (1998a). We suggest that these responses were triggered by providing deep pool habitats and vegetated banklines that were not produced by watershed-level treatments.

Clearly, shoreline vegetation and pool habitats alone do not remedy all types of stream corridor ecological degradation, and too much pool habitat can be worse than too little. Stream fish communities depend on multiple habitat types in natural spatial and temporal arrangements that allow fish to complete various life cycle stages and survive extreme events (Schlosser 1995, 2002). Thus, while riverine restoration projects need to be planned with all aspects of system integrity in mind, attacking the "critical elements" or limiting physical habitat factors may be efficient first steps in an incremental restoration process.

As previously stated, opinions differ on the relative importance of watershed-scale compared to local, instream measures for restoring stream ecosystems and reducing elevated sediment loads (Kuhnle and others 1996, Simon and Darby 2002). While we concede that restoring watershed hydrology or halting anthropogenic influences may be a preferred approach to stream restoration, we like to point out that time, cost, and legal constraints often prevent such comprehensive methodologies. Moreover, the research reported here further supports our contention that restorationists can trigger ecological recovery by simply increasing the availability of certain limited stream habitat features. A systematic examination of links among channel form, instream habitat, and aquatic communities within another southeastern watershed revealed that modification of instream habitat and channel form are necessary for biological recovery (Smiley 2002). In Hotophia Creek, we found that exploiting the prevailing geomorphic processes in order to assist reach-based restoration techniques was cost-effective in accelerating ecological recovery. However, the difficulty of anticipating future changes in watershed water and sediment yield and the response of physical habitat factors to these changes should not be minimized.

Stream ecosystem restoration may be produced by a combination of active and passive approaches that consider physical habitat as well as water quality if habitat factors are limiting. Major gains in ecological recovery are available for relatively minimal incremental costs by the inclusion of habitat features in erosion control projects placed in severely degraded, incised, warmwater streams. Failure to include these features represents poor stewardship and loss of important opportunities.

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