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LARGE WOOD ADDITION FOR AQUATIC HABITAT REHABILITATION IN AN INCISED, SAND-BED STREAM, LITTLE TOPASHAW CREEK, MISSISSIPPI[†]

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ABSTRACT

Large wood (LW) is a key component of stream habitats, and degraded streams often contain little wood relative to less-impacted ones. Habitat rehabilitation and erosion control techniques that emphasize addition of natural wood in the form of individual elements or structures are increasingly popular. However, the efficacy of wood addition, especially in physically unstable, warmwater systems is not well established. The effects of habitat rehabilitation of Little Topashaw Creek, a sinuous, sand-bed stream draining 37 km^2 in northwest Mississippi are described herein. The rehabilitation project consisted of placing 72 LW structures along eroding concave banks of a 2-km reach and planting 4000 willow cuttings in sandbars opposite or adjacent to the LW structures. Response was measured by monitoring flow, channel geometry, physical aquatic habitat and fish populations in treated and untreated reaches for 2 years before and 4 years after rehabilitation. Initially, LW structures reduced high flow velocities at concave bank toes. Progressive failure of the LW structures and renewed erosion began during the second year after rehabilitation, with only 64% of the structures and about 10% of the willow plantings surviving for 3 years. Accordingly, long-term changes in physical habitat attributable to rehabilitation were limited to an increase in LW density. Fish biomass increased in the treated reach, and species richness approximately doubled in all reaches after rehabilitation, suggesting the occurrence of some sort of stressful event prior to our study. Fish community composition shifted toward one typical of a lightly degraded reference site, but similar shifts occurred in the untreated reaches downstream, which had relatively high levels of naturally occurring LW. Large wood is a key component of sand-bed stream ecosystems, but LW addition for rehabilitation should be limited to sites with more stable beds and conditions that foster rapid woody plant colonization of sediment deposits. Published in 2006 by John Wiley & Sons, Ltd.

KEY WORDS: stream restoration; large woody debris; fish; erosion; sediment; physical habitat

INTRODUCTION

Warmwater streams in the southeastern United States have remarkably high levels of biodiversity and are thus important ecological resources, but resident fauna are apparently experiencing accelerated extinction rates (Ricciardi and Rasmussen, 1999; Warren *et al.*, 2000). Many species are imperiled due to habitat and water quality degradation associated with erosion and sedimentation caused by channelization, watershed development and other human activities (Karr *et al.*, 2000; Warren *et al.*, 2000). Watershed development and channel modifications have triggered headward-progressing channel incision in the upper parts of watersheds and attendant downstream sedimentation throughout much of the central United States (Simon *et al.*, 1996), and in many urbanizing watersheds elsewhere. In north-central Mississippi, annual sediment yield in incising watersheds is $\sim$1000 t km^{-2}, or about an order of magnitude more than the national average (Shields *et al.*, 1995). Physical aquatic habitat quality is poor in incised reaches, usually exhibiting a surplus of shallow water depths and shifting, sandy substrate and a deficit of woody debris, pool habitats and stable substrates (Shields *et al.*, 1994). Stage and flow fluctuations

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tend to be more rapid and erratic than for nonincised streams (Doyle and Shields, 1998). Large wood (LW) is input to channels by bank failure processes, and in-channel debris accumulations are associated with sediment retention (Downs and Simon, 2001), in some cases reversing incision (Shields *et al.*, 2000). However, as banks fail and channels widen, incised reaches become less retentive of LW. Bank failure often destroys remnants of floodplain forest, and thus the fully incised channels lose the 'mediating influence' (Brooks and Brierly, 2002) of riparian vegetation and LW on geomorphic processes.

LW is an important component of aquatic habitat in warmwater streams, retaining particulate organic matter such as leaf litter (Bilby and Likens, 1980; Angermeier and Karr, 1984; Brookshire and Dwire, 2003), providing substrate for biomass production by benthic macroinvertebrates (Benke *et al.*, 1985), and fostering higher levels of invertebrate species richness and abundance (Cooper and Testa, 1999). Large wood creates zones of flow acceleration and deceleration that provide higher levels of physical diversity (Shields and Smith, 1992; Dolloff and Warren, 2003), which are important to fish (Thevenet and Statzner, 1999; Warren *et al.*, 2002). Native species are likely adapted to high LW densities typical of North American streams prior to European settlement when LW was abundant due to beaver (*Castor canadensis*) activity and the absence of human actions to remove LW and old-growth forests. Review of field and laboratory experimental studies indicates that fish use submerged LW for overhead cover from predators, as a velocity shelter, as a visual barrier from other fish and possibly for orientation as well as a source of prey (Crook and Robertson, 1999). Laboratory experiments with salmonids indicate that the most complex types of LW structures that provide cover, shade and velocity shelter are more valuable than formations that provide only one attribute (McMahon and Hartman 1989).

Large wood addition is increasingly common as a river habitat rehabilitation measure, but most reports are for coarse-bed, coldwater montane streams (White, 1996; Crook and Robertson, 1999; Reich *et al.*, 2003, Abbe *et al.*, 2003). Of 29 LW addition projects reviewed by Reich *et al.* (2003), none were located in sand-bed streams draining cultivated lands. Only seven of the 29 were in channels wider than 9.9 m, and only four had stated goals of channel stabilization. North American reports were limited to the montane West (14), the Appalachians (2), and the Montreal River of Quebec (1), a trout stream. In contrast, moderate success was reported by Bond and Lake (2005) for habitat rehabilitation using LW in a sand bed stream in southeastern Australia. The stream studied by Bond and Lake had undergone extreme aggradation due to sand deposition ('sand slug'), and insertion of LW triggered formation of pools that provided cover and refugia during flow extremes. Incising sand bed streams such as the one studied here feature extreme channel widths, absence of coarse sediments for ballast, rapid and frequent flow fluctuations, highly unstable channel boundaries and rapid wood decay relative to cooler climates (Roni *et al.*, 2002). Features contributing to natural LW retention (Gurnell *et al.*, 2000) are absent or suppressed. In comparison to coarse-bed montane streams, available wood tends to be smaller, cobble and boulders for ballast are unavailable, and channel erosion rates (relative to channel width) are higher. Channel width-depth ratios are an order of magnitude smaller than cobble- or gravel-bed rivers, so storm flows tend to be deep, and structures are more frequently submerged but for brief periods.

LW addition has improved aquatic habitat diversity with corresponding positive responses in benthic macroinvertebrates (Gerhard and Reich, 2000) and trout (Zika and Peter, 2002) in small channelized streams in Western Europe. Large wood structures have also been applied to relatively large gravel and cobble-bed streams in the USA. Northwest and Australia (Brooks *et al.*, 2001) for channel control and habitat rehabilitation. We hypothesized that recovery of physical aquatic habitat and fish community structure could be accelerated by placing LW structures to stabilize banks and in-channel sand deposits in an incised, warmwater stream. In order to isolate effects of LW addition on fish and their habitats, we monitored reaches with and without LW addition in a small, sand-bed stream in northern Mississippi. Our study period spanned six years, including more than a year prior to LW addition.

Study site

A site was selected along Little Topashaw Creek, a fourth-order stream (1:24,000 topographic map) in north central Mississippi draining about 37 km^2 (Figure 1). Criteria used in site selection included rapid bank erosion, an abundant supply of sandy bed material from upstream, nearby sources of native plant and animal colonists, and an

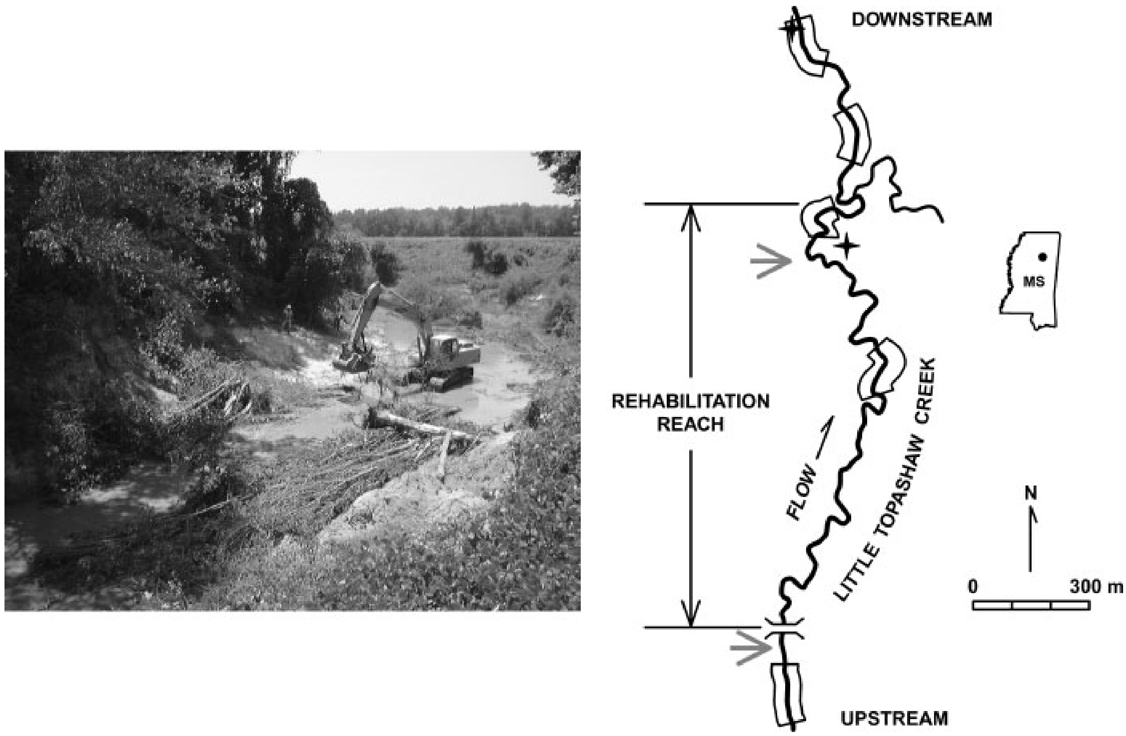

Figure 1. Location of Little Topashaw Creek rehabilitation project and reaches sampled for physical habitat and fish. Boxed regions indicate subreaches sampled for habitat and for fish. Inset photograph shows construction of LW structure. Horizontal arrows show tracer dye injection points, and stars indicate points where dye concentrations were measured

advanced stage of incised channel evolution (Simon and Darby, 1999). Watershed land use was dominated by pine and mixed hardwood forest, with only about 12% of the area in cultivation or used as pasture. Cultivated lands were concentrated within the floodplain immediately adjacent to the study reach. The single-thread, meandering channel had an average sinuosity of 2.1, an average slope of 0.0025, an average width of 33 m, and an average depth of 3.6 m. Channel bed materials were primarily 0.2 to 0.3 mm sand. Channel morphology was extremely dynamic, typical of incising channels in the region. Historical aerial photos suggest mean channel width increased by a factor of 4 to 5 between 1955 and 1997. A 2-km reach was selected for rehabilitation, with adjacent untreated reaches immediately up- and downstream monitored for comparison purposes (Figure 1).

A geomorphic evaluation performed immediately prior to rehabilitation indicated that the downstream end of the study area was in the aggradational stage V of the Simon (1989) conceptual model of incised channel evolution, while the middle part was stage IV (degradation and bank failure), and the upstream segments were still degrading (stage III). A knickpoint was located between zones classified as stage IV and stage III, with thalweg slopes $\sim$0.003 upstream of the knickpoint and $\sim$0.002 downstream. In general, concave banks on the outside of meander bends were failing by mass wasting, and sand was accreting on large point bars opposite failing banks. Outside of bends, eroding banks were invading adjacent cultivated fields, while inside bends and abandoned sloughs were vegetated with a diverse mixture of hardwood trees and associated species. Surveys of 13 cross sections before and after a flow of 55 $m^3 s^{-1}$ that occurred 3 months prior to rehabilitation indicated an average increase in cross-sectional area of 10% with bank retreat as great as 7.6 m. This event, in which peak stages reached mid-bank elevation, triggered 60 m of upstream migration of a 0.6-m high headcut and produced two chute cutoffs across point bars. The estimated magnitude of the 2-year event computed using a regional regression equation is 74 $m^3 s^{-1}$ with a standard error of 35% (Ries and Crouse, 2002). High flow events tended to be extremely brief (<30 hours) and frequent, with base flows generally <0.10 $m^3 s^{-1}$.

Large wood structure design and construction

Design details are presented by Shields *et al.* (2004) and will only be summarized here. Key aspects of the design problem include (1) use of buoyant materials, (2) use of materials that gradually decay, and (3) dual objectives of channel stabilization and habitat rehabilitation. The long-term purpose of LW placed in an incised sand-bed stream is to amplify dominant geomorphic processes leading to development of a sinuous two-stage channel with wooded berms (Stage VI, Simon, 1989). Initial success of LW structures depends upon their ability to resist flotation, while their long-term success is contingent upon their creation of suitable habitat for plants by inducing sediment deposition. Dense colonies of woody plants can secure and stabilize the channel margins over the longer term as the LW decays (Jacobson *et al.*, 1999).

LW structure geometry was specified by crest angle, length, elevation and spacing (Figure 2). Additionally, the maximum and minimum dimensions for individual logs and the spacing of logs within structures may also be specified. The crest angle (angle between a line normal to the approach flow vector and the weir crest) was set at 15° upstream to promote deflection of overtopping flow away from eroding banks (Derrick, 1997), although others have suggested angles between the bank and the weir crest of 25° to 30° based on straight channel flume tests (Johnson *et al.*, 2001). Crest length was based on a target mean bottom width for the design cross-section based on regional curves for incised channels in advanced stages of evolution. Crest elevations were selected so that the sediment berms that formed over the structures transformed adjacent vertical banks into stable geometries as indicated by geotechnical analyses (Darby and Simon, 1999). LW structures were spaced 1.5 to 2.0 times the crest length apart based on criteria for river training groins (Petersen, 1986).

For design, forces acting on the LW structures were partitioned into buoyancy and fluid drag (D'Aoust and Millar, 2000). Forces due to ice and impact by floating wood were neglected. The buoyant force, F_b (in newtons), is equal to the product of the difference between the specific weight of water, γ_w, and wood, γ_d, (both in $N\,m^{-3}$) and

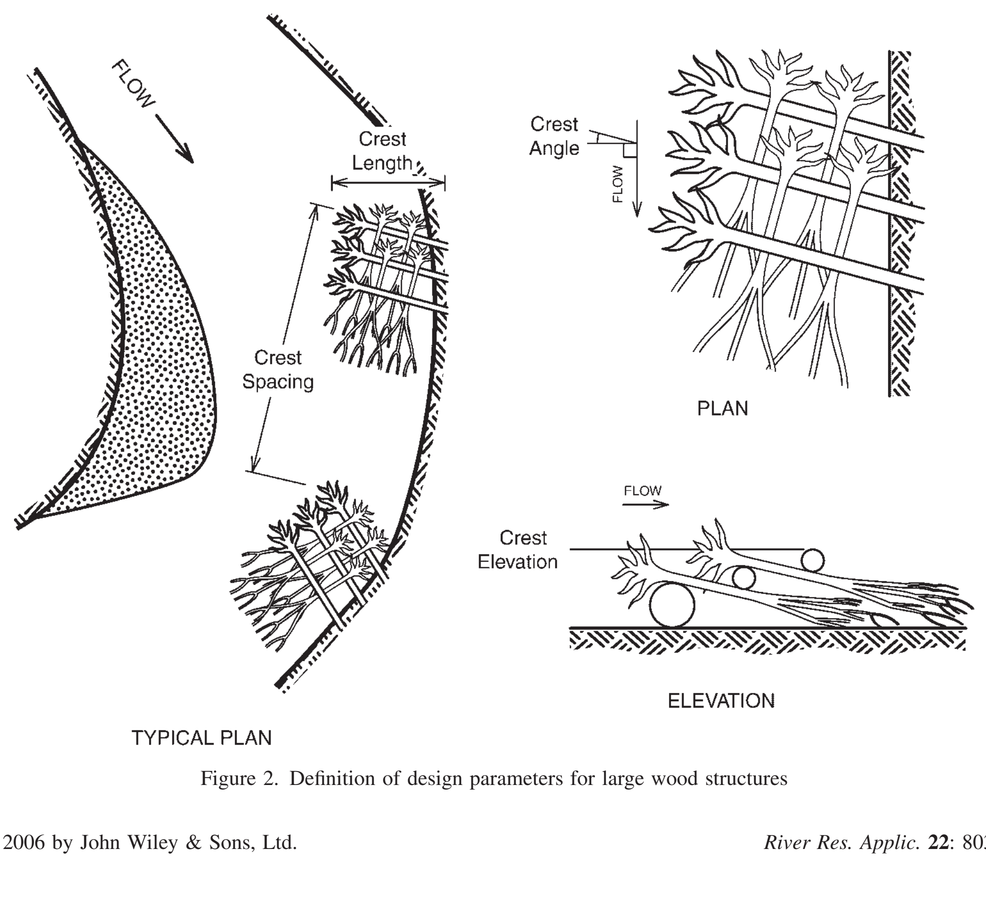

Figure 2. Definition of design parameters for large wood structures

the submerged volume of the logs (Braudrick and Grant, 2000):

$$F_b = (\gamma_w - \gamma_d) \sum_{i=1}^{i=n} V_i \tag{1}$$

where V_i is the submerged volume in m^3 for the ith log in a structure comprised of n logs. It was assumed that all of the members had volumes approximated by cylinders. The assumption of cylindrical volumes overestimated LW volume because it neglected stem tapering, but this factor was balanced by the volume of branches. Since wood submergence along incised, sand bed streams is limited to periods shorter than a few days, a specific weight (γ_d/γ_w) of 0.45 was used for buoyant force computations based on published and measured values (Shields *et al.*, 2004).

The drag force on the structures was computed by:

$$F_d = \frac{\gamma_w V^2 A C_D}{2g} \tag{2}$$

where F_d = drag force in N, V = approach flow velocity in m s^{-1}, A = area in m^2 of LW structure projected in the plane perpendicular to flow, and C_D = drag coefficient. Here the structure was treated as a single body, rather than a collection of individual cylinders (Gippel *et al.*, 1996). For design, the cross-section mean velocity was increased by a factor of 1.5 to allow for higher velocities on the outside of bends (U.S. Army Corps of Engineers, 1991). Drag coefficients were computed using an empirical formula (Shields and Gippel, 1995), and typically ranged from ~0.7 to 0.9. However, drag coefficients reach values as high as 1.6 for cylinders that are barely submerged because of the forces associated with the formation of standing waves (Wallerstein *et al.*, 2001; Alonso *et al.*, 2005), and in retrospect, higher values may have been more appropriate. Nevertheless, the buoyant and drag forces computed using the flow velocities and depths observed in this project showed that the net buoyant force was an order of magnitude greater than the drag force. Net applied forces acting on structures were resisted by sediment deposition induced during lower flows prior to structure submergence and by earth anchors. Four earth anchors were pressed ~1.7 m into the bed at the corners of each of 80% of the structures, load tested to 4.5 kN, and attached to each other with steel cables running over the top of the structure.

A total of 72 LW structures was constructed on concave, eroding banks using either woody debris (~10%) or living trees (~90%). Living trees were ≥0.20 m diameter at breast height, an average of 6.7 ± 3.2 m long, and were harvested with root balls and crowns intact using heavy equipment. A total of 1,168 trees was obtained by clearing 3.4 ha of fencerows and ditchlines > 10 m from the top bank of the channel. Most trees were harvested more than 200 m from the channel, and no impacts of tree harvest on channel morphology or habitat were observed. Most trees were oaks (*Quercus* sp.), but ash (*Fraxinus* sp.), cherry (*Prunus* sp.), hickory (*Carya* sp.), elm (*Ulmus* sp.), sweetgum (*Liquidambar styraciflua)* and sycamore (*Platanus occidentalis*) were also used. About 4,000 willow (*Salix nigra*) cuttings were planted on point bars and in sediment deposits adjacent to selected LW structures using a water-jetting technique. Including willow planting, costs for rehabilitation were approximately US\$88 m^{-1} channel treated, roughly 20% to 50% of costs for recent construction of traditional stone stabilization projects in the region.

METHODS

Effects of LW addition were observed by monitoring the treated reach and adjacent reaches up- and downstream before and after construction of our project to produce a before-and-after-control-impact study scheme. Large wood structures were constructed during July-August 2000, and willows were planted during January-February 2001, but the October 2000 data were influenced only minimally by the LW structures because a prolonged drought prevented the structures from exerting any effects on channel morphology until November 2000. Indeed, many of the structures were not in contact with the stream during the extremely low flow of October 2000. Accordingly, the 1999–2000 data were classified as 'before rehabilitation,' and the 2001–2004 data, 'after rehabilitation.'

Selection of an appropriate reference condition is an important step in stream restoration planning, and restoration success should be measured against a reference. Previous studies (Shields *et al.*, 1994 and 1998) of Toby

Tubby Creek, a lightly-degraded nonincised reference stream with bed material and contributing drainage area almost identical to Little Topashaw Creek were used to provide a reference data set.

Little Topashaw Creek precipitation was monitored using two tipping bucket rain gages located adjacent to the channel at opposite ends of the treated reach. Stream flow was computed using a rating based on data collected at 15-min intervals at the upstream end of the treated reach using acoustic Doppler/pressure transducer instruments that logged water depth and depth-averaged velocity. Effects of LW addition on physical habitat quality and fish were quantified by semiannual (June and late September or early October) sampling at base flow during 1999–2004 inclusive, except habitat was not sampled in 2004.

Effects of LW addition on flow patterns that contribute to hydraulic retention were quantified using a tracer dye experiment 9 months after construction. Discharges during the dye experiment ($\sim$0.3 $m^3 s^{-1}$) were above base flow, but well below high flow levels. Slug-injections of Rhodamine WT dye were made immediately upstream and downstream from the treated reach (Figure 1), and passage of the resulting dye cloud was documented by periodically collecting grab samples for several hours at points downstream. The reach traversed by the downstream dye cloud had similar flow resistance characteristics (bed slope, channel cross section, bed material size, and sinuosity) to the treated reach except for the absence of the LW structures and the attendant features created in the channel bed by scour and deposition adjacent to the structures. The downstream study was completed first to avoid interference from the upstream injection. Time-concentration curves were normalized by dividing the time values by reach length, producing a frequency distribution of flow-through velocities for each of the two reaches.

A more direct measure of the longer-term effects of the structures on organic matter retention was made by sampling carbon content in stream bed sediments of the base flow channel. Samples of the top 10 cm of hyporheic zone sediments, including organic matter lying on the bed surface, were taken from channel transects located at $\sim$50 m intervals along the entire treated reach. At each transect, $\sim$250 g samples were taken from the base flow channel centreline and from the region between water's edge and the quarterpoint of the base flow channel (three samples per transect). All samples were analyzed for organic C via dry combustion using a LECO CN2000 (brand name for information purposes only) at a temperature of 1300–1350°C as described by Stofleth *et al*. (2004).

Effects of LW on aquatic habitats during high flows were observed using acoustic-Doppler depth-velocity loggers (Shields *et al.*, 2001). Two loggers were secured above the stream bed along each of two transects in each of two meander bends: one where LW structures had been placed on the concave bank and one untreated bend. Depth and velocity measurements were recorded every 5 min during major runoff events. LW effects on erosion and deposition were quantified using cross-section and thalweg surveys conducted before and during the first year after construction.

Fish and physical habitat variables were sampled semiannually during base flow within five, 150-m-long (20-30 water widths) subreaches (Figure 1): two were downstream of the modified region in a reach geomorphically similar to the treated reach (Simon stage IV/V), two were within the treated reach, and one was in a straight, relatively narrow channel immediately upstream (Simon Stage III). Within each reach, physical habitat variables were measured along 10 transects placed at 15-m intervals. Along each transect, water depth and substrate were recorded at a point 25 cm from the left water's edge and at four to six additional points spaced at equal intervals (e.g., spacing for four points would be local water width divided by 5). Substrate was visually classified as clay, sand, gravel, organic debris or other (usually man made objects or vegetation). Water surface width was measured with a tape. Discharge was computed using depth and velocity data collected using a wading rod and an electromagnetic current meter. Fish were sampled concurrently with physical habitat using a single pass with a backpack-mounted electroshocker. Each subreach was fished for several minutes (electric field application time mean = 12.4 min, std dev = 5.0 min). Shocking crews, which consisted of one person carrying the electroshocker and two people with dipnets, worked from downstream to upstream, sampling all habitats, with greater concentration on those yielding fishes. Fishes longer than about 150 mm were identified, measured for total length, and released. Smaller fish, and fish that could not be identified in the field were preserved in 10 percent Formalin and identified and measured in the laboratory. Fish biomasses were determined by weighing or using species-specific regression formulas based on total length derived from previously obtained specimens.

Fish collections were compared across reaches (upstream, treated, and downstream) and sampling periods (before and after rehabilitation) using two-way ANOVA. Fish community composition was compared among the three reaches by computing Spearman Rank Order correlation coefficients using abundances of the 14 most abundant species. In addition, species lists for each sample were used to compute five indicators of ecological integrity (Wichert and Rapport, 1998; Shields *et al.*, 2000). These indicators, termed species association characteristic scores (SACS), were based only on the presence or absence of a species. SACS were computed by assigning integer scores to each fish species based upon habitat orientation (e.g., surface, benthic, littoral or vegetation, etc.) and feeding group (e.g., omnivore, herbivore, general invertebrates, etc.) in such a way that higher scores were associated with greater sensitivity to ecosystem stress. SACS values were computed for each sample as follows:

$$SACS_j = \frac{\sum_{i=1}^{N} SCS_{ij}}{N}$$

where $\mathrm{SACS_j}$ = species association characteristic score based on indicator j, $\mathrm{SCS_{ij}}$ = value of indicator j for species i provided by Shields *et al.* (2000), and N = number of species. Mean SACS were compared across sites and dates using two-way ANOVA.

RESULTS

Annual precipitation during the period of observation averaged 1,480 mm. A severe drought occurred in late 2000 (72 mm total July-October) coincident with LW structure construction and a very wet period followed during the winter of 2000–2001 (955 mm November-March). Annual peak discharges ranged from 40–60 $\mathrm{m^3\,s^{-1}}$, with maximum depths of about 3 m and occasional velocities as great as 3 $\mathrm{m\,s^{-1}}$. Three events with peaks $>$20 $\mathrm{m^3\,s^{-1}}$ occurred during the first few months after construction, and 11 events of this magnitude occurred over the second and third years following construction. High flow events were of short duration (a few hours), with brief rise times and sharp peaks. Stream bank erosion was initially slowed by placement of the LW structures, and deposition of sand berms adjacent to steep, concave banks increased slope stability during the first year following rehabilitation. Initial performance of willow cuttings was strong, with more than half of monitored plants surviving the first growing season (Martin *et al.*, 2005). However, longer term willow survival was less than 10% (Pezeshki *et al.*, submitted), and scour of sediment deposits and attendant bed degradation resulted in progressive failure (loss of woody materials) of the LW structures (Table I) and renewed erosion of banks.

Table I. Fate of large wood structures, Little Topashaw Creek, Mississippi. Percentages refer to totals in first column

	As built (2000)	After one year (2001)	After two years (2002)	After three years (2003)
No. of remaining structures	72 (100%)	68 (94%)	50 (69%)	46 (64%)
No. of remaining structures located in bends	39 (100%)	38 (97%)	30 (77%)	27 (69%)
No. of remaining structures located in straight reaches	33 (100%)	30 (91%)	20 (61%)	19 (58%)
Mean crest elevation above bed, m	2.1 ± 0.5	2.5 ± 0.6	2.4 ± 0.5	—
Length, m	13.9 ± 3.9	10.2 ± 4.2	—	—
Width, m	5.3 ± 1.9	6.0 ± 2.3	—	—
No. of remaining structures without anchors	14 (100%)	13 (93%)	11 (79%)	9 (64%)
No. of remaining structures with anchors	58 (100%)	55 (95%)	39 (67%)	37 (64%)

Table II. Summary of base flow physical habitat conditions before and after habitat rehabilitation, Little Topashaw Creek, Mississippi. Mean values are given ± standard deviation. Values with the same superscripts are not significantly different ($P = 0.05$, Two-way ANOVA)

Quantity	Upstream reach		Reach modified by LW addition and willow planting		Downstream reach	
	Before	After	Before	After	Before	After
Mean water depth, cm	35 ± 34^{a}	24 ± 28^{c}	6 ± 7^{b}	11 ± 11^{d}	6 ± 8^{b}	12 ± 13^{d}
Mean water width, m	6.7 ± 3.3^{a}	6.2 ± 3.8^{a}	4.9 ± 2.7^{b}	4.8 ± 1.7^{b}	4.5 ± 2.1^{b}	6.8 ± 2.4^{c}
Large wood density, m^2/ 100 m^{2*}	0.56 ± 0.25^{a}	0.19 ± 0.22^{a}	9.40 ± 7.12^{b}	33.75 ± 12.57^{c}	4.77 ± 2.48^{b}	6.63 ± 3.59^{b}

*Units refer to gross area of large wood formations in the plane of the water surface divided by water surface area.

The treated reach instream LW density was tripled by the rehabilitation project, and base flow became deeper but not wider in the treated reach (Table II and Figure 3). Large wood structures performed well during initial high flow events. However, during the next two years, progressive failure of the structures was observed (Table I), particularly due to failure of earth anchors, wood decay and breakage, and scouring of sediments deposited within the structures (Shields *et al.*, 2004). Structural failure was accompanied by a reversion to shallow habitat conditions. The untreated reach downstream also became deeper after rehabilitation, while the morphologically dissimilar upstream reach became shallower (Table II). Water depths were greatest in the upstream reach throughout the study due to the presence of several upstream-migrating nickpoints and associated scour pools. The upstream reach was typical of a transitional phase that is a precursor of the inferior conditions downstream (Shields *et al.*, 1998).

The dye test, conducted during the first year after rehabilitation, showed that flow resistance and hydraulic retention was greater in the treated reach than downstream. Mean velocity for the treated reach was $0.17\ m\ s^{-1}$, but $0.29\ m\ s^{-1}$ for the downstream reach. Although the mean velocity was less in the reach treated with LW, the shape of the frequency distribution of velocity was similar in the two reaches. The 90 percentile velocity was 1.25 times

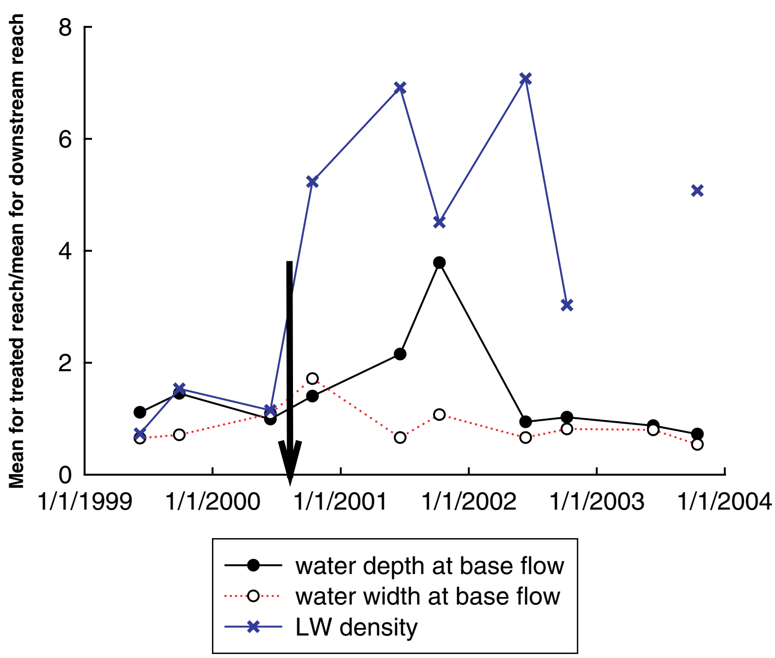

Figure 3. Means of base flow water depth, width, and large wood loading in the treated reach divided by respective means for the untreated reach downstream. Approximate date for LW addition is shown by the vertical arrow

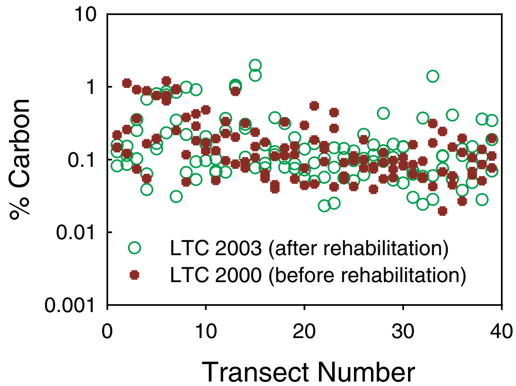

Figure 4. Carbon content of surficial bed sediments of treated reach of Little Topashaw Creek, Mississippi, immediately prior to and 2 years after rehabilitation with LW structures

the median velocity (0.19 m s^{-1}) within the rehabilitated reach, while the same ratio for the downstream reach was 1.36 (median = 0.30 m s^{-1}).

Bed sediment organic carbon levels were confined primarily to a range between 0.05% and 0.5% with the exception of occasional spikes (~1%) associated with samples crumbled from consolidated cohesive scarps and outcrops in the upstream portions of the study reach (Figure 4). Since this C was not associated with POM, these values were excluded from statistical analyses. Mean bed C concentrations were intermediate to values for highly and lightly degraded reaches elsewhere in the region (Stofleth *et al*., 2004). One-way ANOVA based on ranks indicated that bed sediment C concentrations were not significantly different before and three years after rehabilitation ($P < 0.05$) (Table III).

Acoustic-Doppler loggers recorded velocity magnitudes within LW structures that were only about one-third of those measured in the channel adjacent to the structure (Figure 5). Preferred habitat for centrarchids and ictalurids generally lies within the 0.10 to 0.50 m s^{-1} range (e.g., McMahon and Terrell, 1982; Stuber *et al.*, 1982). Prior to placement of LW structures, highest velocities within bends occurred at the bank toe, but the LW created a velocity shelter in this region and confined higher velocities to the channel centre. Velocities within the LW structure were generally less than 0.30 m s^{-1}, and usually below 0.10 m s^{-1}, even during events that were large enough to produce flow depths >3 m.

A total of 9,129 fish representing 32 species were collected during the course of the study. Fish numbers, biomass, and size increased in all reaches following rehabilitation, but the only statistically significant changes were increasing biomass and size in the treated reach ($P < 0.05$, Table IV). In the treated reach, at least 18 of the captured species and 9 of the 12 species that exhibited greater mean abundance following LW addition have documented associations with LW (Table V). Similar ratios for the upstream and downstream reaches were 6/8 and 9/11.

All three reaches became more speciose following rehabilitation, and the average number of species per sample (a single collection from a single 150-m reach) approximately doubled following LW addition (Table IV). There was no significant difference in the number of species per sample collected from the treated reach and the other

Table III. Total carbon concentrations in bed sediments of treated reach of Little Topashaw Creek, Mississippi before and after rehabilitation

Sampling date	Mean ± Standard deviation (%)	Skewness	Kurtosis
May–June 2000	0.13 ± 0.08	1.18	0.82
July–Aug 2003	0.14 ± 0.15	5.56	42.55

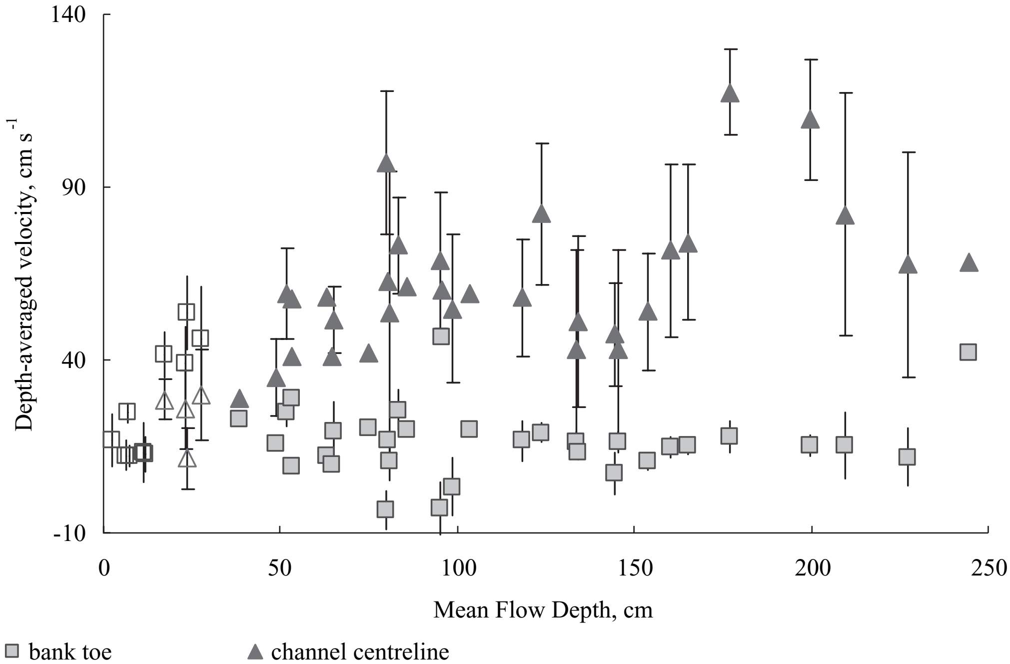

Figure 5. Comparison of runoff event-mean water velocities and depths within LW structures ('bank toe') and along the adjacent channel centreline. Open symbols show conditions during three months prior to placement of LW structures, while filled symbols represent events during the first 4 months after placement. Error bars are ± one standard deviation

reaches either before or after rehabilitation ($P > 0.27$). Species composition was influenced more strongly by channel morphology and associated fluvial processes than by LW addition. Fish species abundance in the treated reach was similar to the downstream reach (both Simon stage IV/V) throughout the experiment, with Spearman Rank Order correlation coefficients (Glantz, 1992) for the 14 most abundant species (Table V) of 0.85 and 0.88 before and after rehabilitation ($P < 0.001$ in both cases). In contrast, the Spearman coefficients between the abundance lists for the upstream reach (Simon Stage III) and the other two reaches were <0.36 ($P > 0.20$) before rehabilitation and <0.64 ($P > 0.013$) afterward, highlighting the differences in physical characteristics. SACS indicators were generally unchanged following rehabilitation. Only the SACS indicator based on maximum body size displayed a significant difference between pre- and post-rehabilitation periods, and that difference was limited to the downstream reach.

Table IV. Summary of electrofishing catch before and after rehabilitation, Little Topashaw Creek, Mississippi. Mean values are given ± one standard deviation. Values with the same superscripts are not significantly different ($P = 0.05$, Two-way ANOVA)

Quantity	Upstream reach		Treated reach		Downstream reach	
	Before	After	Before	After	Before	After
Mean no. of fish per sample*	74 ± 79^{a}	143 ± 149^{a}	129 ± 75^{a}	177 ± 164^{a}	141 ± 76^{a}	186 ± 139^{a}
Mean biomass, g per sample	262 ± 155^{a}	$337 \pm 192^{a,b}$	150 ± 78^{a}	407 ± 429^{b}	168 ± 97^{a}	$397 \pm 192^{a,b}$
Total no. of species	13	22	19	26	17	27
Mean no. of species per sample	6.8 ± 0.5^{a}	11.4 ± 4.4^{b}	6.8 ± 3.0^{a}	12.8 ± 2.9^{b}	6.3 ± 2.9^{a}	13.1 ± 1.4^{b}
Length of largest individual in each sample (length, cm)	13 ± 4^{a}	$16 \pm 3^{a,b}$	9 ± 3^{a}	14 ± 6^{b}	$10 \pm 5^{*}$	$12 \pm 2^{a,b}$

*The expression 'sample' here refers to a collection from a 150-m long sampling reach.

Table V. List of fish species taken from Little Topashaw Creek, Mississippi, before and after construction of habitat rehabilitation project in order of descending abundance. Abundance values are not corrected for differences in levels of effort

Species	Preconstruction			Postconstruction			Total	Use of Large Wood (Dolloff and Warren 2003)
	downstream	restored	upstream	downstream	restored	upstream		
Cyprinella venusta	410	368	64	753	778	230	2603	egg attachment
Notropis rafinesquei	315	279	3	928	751	136	2412	
Pimephales notatus	83	124	92	397	430	167	1293	nesting cover, egg attachment
Lepomis cyanellus	48	25	33	176	163	40	485	cover, feeding
Luxilus chrysocephalus		8		177	150	4	339	
Lepomis megalotis			9	109	110	110	338	
Gambusia affinis	50	102		57	26	25	260	
Pimephales vigilax	19	2		50	98	67	236	nesting cover, egg attachment
Fundulus olivaceus	3	3	2	81	66	36	191	cover
Lepomis macrochirus	3	1	8	41	53	37	143	cover, feeding
Semotilus atromaculatus	17	4	15	53	27	1	117	feeding (juvenile), diurnal cover
Erimyzon oblongus	2	2	6	45	32	24	111	cover
Ameiurus natalis	13	19	1	45	23	7	108	nesting cover
Lythrurus umbratilis			1	17	11	47	76	
Notropis atherinoides			57				57	
Noturus miurus	1	10		10	20	13	54	
Noturus phaeus				10	27	10	47	diurnal cover, probably nesting cover
Etheostoma nigrum				3	12	25	40	nesting cover, egg attachment
Aphredoderus sayanus	1	4	4	5	12	5	31	cover, feeding
Micropterus punctulatus		1		10	11	6	28	
Fundulus notatus				1	25		26	
Micropterus salmoides	3			3	9	7	22	
Notropis volucellus		12					12	
Noturus nocturnus	1	2		2	1	1	7	cover
Ictalurus punctatus	1	1			1		3	
Notemigonus crysoleucas					2		3	
Etheostoma chlorosomum	1			1	1		2	cover, egg attachment
Ameiurus nebulosus						1	1	nesting cover
Etheostoma parvipinne		1					1	cover, egg attachment
Etheostoma whipplei				1			1	cover[1]
Percina sciera				1			1	cover
Pomoxis nigromaculatus				1			1	

[1]Dolloff and Warren *E. artesiae* as name for this species.

Collections were dominated by cyprinids (81% of total number, 51% of biomass) and centrarchids (10% of total number, 32% of biomass), and the relative dominance of cyprinids expressed as a fraction of total fish biomass was inversely related to the mean water depth ($r = -0.36$, $P = 0.013$). Opposite trends were indicated for the centrarchids with $r = 0.46$ ($P < 0.001$), for the association between their fraction of catch biomass and mean depth. In the treated reach, centrarchid numbers and biomass increased from 3% to 26% of the catch and from 11% to 38% of the catch, respectively (Table VI). Significant, but smaller increases in centrarchid dominance occurred in the downstream reach. Conversely, the dominance of the cyprinids as a fraction of catch biomass declined in all three

Table VI. Relative dominance of centrarchids and cyprinids before and after rehabilitation, Little Topashaw Creek, Mississippi. Mean values are given ± standard deviation. Values with the same superscripts are not significantly different ($P = 0.05$, Two-way ANOVA)

Quantity	Upstream reach		Treated reach		Downstream reach	
	Before	After	Before	After	Before	After
Mean % of numbers as cyprinids	65 ± 20^{a}	50 ± 28^{a}	78 ± 27^{a}	75 ± 15^{a}	79 ± 24^{a}	75 ± 14^{a}
Mean % of biomass as cyprinids	28 ± 19^{a}	15 ± 15^{a}	68 ± 10^{b}	49 ± 16^{c}	66 ± 5^{b}	$49 \pm 18^{b,c}$
Mean % of numbers as centrarchids	29 ± 21^{a}	$27 \pm 24^{a,c}$	3 ± 2^{b}	26 ± 16^{c}	5 ± 4^{b}	16 ± 10^{c}
Mean % of biomass as centrarchids	57 ± 21^{a}	$46 \pm 27^{a,c}$	11 ± 5^{b}	38 ± 13^{c}	18 ± 13^{b}	30 ± 9^{c}

reaches in the four years following rehabilitation, but only the treated reach experienced significant changes ($P = 0.037$). Two centrarchid species typically associated with deeper habitats were captured in the treated reach following LW addition but not before (*Micropterus salmoides*, *Lepomis megalotis*). The abundance of *Lepomis megalotis* was found to be positively associated with natural LW in an experiment by Angermeier and Karr (1984). *Micropterus punctulatus*, a species dependent upon channel margin habitat with LW and reduced velocities (Scott and Angermeier, 1998), was represented by only one 4.8 cm-long specimen prior to rehabilitation, and this one was found in the treated reach. Following rehabilitation, 6, 11 and 10 individuals were captured in the upstream, treated and downstream reaches, respectively, with maximum lengths of 7, 18, and 8 cm, respectively. The mean length of centrarchids captured increased following rehabilitation in the treated and downstream reaches ($P < 0.001$), but decreased upstream ($P = 0.043$).

The lightly-degraded reference stream, Toby Tubby Creek, was deeper, contained more LW, and was more retentive of organic matter than Little Topashaw Creek (Table VII). These conditions supported a speciose fish community that was dominated by relatively large centrarchids, with small cyprinid opportunists only lightly represented.

Table VII. Comparison of study stream, Little Topashaw Creek, with reference stream, Toby Tubby Creek (data from Shields *et al.*, 1998)

	Toby Tubby Creek, 1991–1995	Little Topashaw Creek before/after rehabilitation		
		Upstream reach	Treated reach	Downstream reach
Large wood density, $m^2/100\ m^2$	10.3	0.6/0.2	9.4/33.8	4.8/6.6
Fraction of bed covered with organic debris, %	20	1/4	5/5	4/2
Mean C concentration in bed sediment, %	0.26	nd	0.13	nd
Mean baseflow water depth, cm	45	35/24	6/11	6/12
Total number of fish species	48	13/22	19/25	17/27
Fish catch biomass, g/150 m	1080*	262/337	150/407	168/397
Average length of largest individual in each sample, cm	32	13/16	9/14	10/12
% of catch comprised by cyprinids (biomass)	<1	24/29	75/53	71/52
% of catch comprised by centrarchids (biomass)	70	59/47	13/38	25/28

*Based on 1993–1995. Mean for 1991–1992 was about 4 times higher, perhaps due to prolonged regional flooding in early 1991 (Shields *et al.*, 1998).

DISCUSSION

Addition of LW in the form of engineered structures produced temporary improvements in physical habitat quality in a rapidly incising sand bed stream. However, only 64% of the LW structures and ~10% of the planted willow cuttings survived more than three years. Use of LW and vegetation for stream habitat rehabilitation requires sites with more compatible climatic conditions (for less rapid wood decay) and geomorphic processes that contribute to the stability of sediment deposits within LW structures. Stability of the structures used in this project would have been increased with more robust anchoring systems; however, the long-term success of LW structures in this type of geomorphic setting is dependent on rapid plant colonization of sediment deposition induced by the structures. Without vegetation to hold the sediments in place, they will be eroded as the LW gradually decays and disintegrates.

Evidently the improved retention measured by the dye study after year 1 was dissipated by the gradual failure of the LW structures during years 2 and 3, and thus the bed organic carbon levels in 2003 were not significantly different from those in 2000. Despite the failure of rehabilitation to produce significantly deeper, more retentive aquatic habitat, fish populations shifted toward larger species more typical of the reference stream. Fish community responses were muted, but were consistent with findings by Angermeier and Karr (1984), who observed that the adaptive associations between fish and wood during base flow in a small Illinois stream appeared to be more closely related to cover rather than food availability or current shelter. They found that all three of the centrarchid species captured in their study exhibited preference for control (unaltered) stream reaches over those from which all wood had been removed. Johnson *et al.* (1988) reported that complex cover with small interstices is important for small centrarchids such as *Lepomis macrochirus* experiencing predation pressures. The appearance of larger individuals within our study reach following LW placement is particularly significant, as fish age/size structure is diagnostic of conditions in less degraded streams, where high levels of habitat heterogeneity, and deep, temporally stable pools allow fishes to withstand hydrologic extremes and elude predators during vulnerable juvenile phases (Webb and de Buffrenil, 1990) as they grow larger and older. Fish food webs grow longer and more complex, leading to a decline in the dominance of generalists and the appearance of predators.

The reference stream (Toby Tubby Creek) fish community was typical of the ''stable'' warmwater stream type (Schlosser, 1987). Incised channels, such as Little Topashaw Creek, tend to support ''colonizing'' assemblages (Schlosser, 1987) dominated by small-bodied opportunists such as juvenile cyprinids (Shields *et al.*, 1998; Adams *et al.*, 2004). Our efforts to rehabilitate stream habitats within Little Topashaw Creek were partially successful in moving key attributes of the treated reach closer to reference conditions (Table VII). However, similar, although slightly less pronounced physical and biological shifts also occurred in the untreated reach downstream. Even the untreated upstream reach experienced a significant increase in the number of fish species (Table VI), suggesting that some watershed-wide event (e.g. flood, drought, or pollution) depressed fish populations in the period prior to our study (Matthews, 1987; Matthews *et al.*, 1988). Temporal instability in fish communities, probably because of hydrologic perturbations, is typical of incised streams in this region (Shields *et al.*, 1998; Adams *et al.*, 2004). Restoration of a self-perpetuating, functional stream ecosystem on a trajectory toward the reference condition did not occur at Little Topashaw Creek.

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