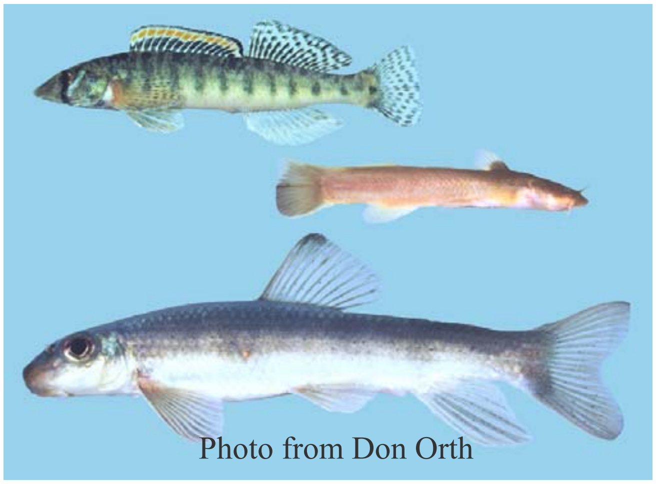
Warmwater Streams Symposium II



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Warmwater Streams Fisheries Management Session

Frank Fiss, Tennessee Wildlife Resources Agency
Moderator and Session Chair

Then and Now: Twenty-Four Years of Warmwater Streams Management

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An old proverb says: "The more things change, the more they stay the same." The same could be said about warmwater streams management. Each generation of scientists builds on the body of knowledge given to them by the previous generation of scientists. Each generation answers questions using the technology they have at hand. A previous generation of warmwater stream managers convened a national symposium on warmwater streams in Knoxville, Tennessee in March 1980 (Krumholz, 1981). The 1980 symposium emphasized the importance of warmwater streams, summarized current ecological knowledge about warmwater streams, discussed warmwater streams resources, problems, and management needs, examined methods for protecting their integrity, and stimulated fresh ideas and new approaches in warmwater streams management. While the past 24 years have brought new innovations and approaches, some management techniques remain the same.

Our symposium is designed to provide an updated report on issues, principles, and practices in warmwater streams management. Our objectives are three-fold: (1) to present topics promoting better monitoring, management, and conservation of warmwater stream fisheries; (2) to present practical solutions for and associated fish community response to problems involving instream habitat, fish movement, riparian disturbance and streambank erosion control; and, (3) to feature multiparty, cooperative efforts restoring individual species or aquatic communities in areas of extirpation or altered habitats. We organized contributed papers into sessions on Warmwater Streams Fisheries Management, Stream Habitat Improvement, and Species Restoration and Recovery Successes to meet these objectives.

We will compare and contrast changes in warmwater stream management over the 24 years since the previous symposium was held. In many ways, we will provide guidance to a new generation of warmwater stream managers, drawing on the visionaries of the last generation for guidance. The need for quality management of warmwater streams is more critical today than it was in 1980. In 1980, based on the 1975 National Survey of Hunting and Fishing, 19% of all freshwater fishing occurred in streams and rivers in the U.S. (Stroud, 1981). Today, based on the 2001 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation, 25% of all freshwater fishing occurs in streams and rivers in the United States.

Shifts in the focus of stream managers are evident when we compare "then" to "now" in at least seven dimensions:

1. Dominant Goals
2. Biological Dimension
3. Scientific Approach
4. Spatial Scale
5. Management Paradigm
6. Reservoir Connections, and
7. Flow

Then, dominant goals were recognizing and quantifying environmental degradation, future potentials, small-scale bank stabilization, and identifying environmental injustice. Now, we focus more effort on environmental restoration and management options and promote multiple values plus environmental justice. Large-scale restoration projects, such as the Kissimmee River restoration, are now being initiated (Dahm et al. 1995). Then, biologists examined game fish and rough fish and calculated diversity indices. Now, we examine mussels, threatened and endangered species, multispecies complexes, and important subsistence fisheries and we are seeking ecological integrity. Then, we used normal scientific approaches, such as upstream-downstream studies to detect effects, erring on the side of reducing type I error. Now, we are beginning to adopt post normal scientific approaches by reducing type II error, adopting Bayesian statistics and establishing normative conditions. Then, biologists focused on sites in the stream channel and limiting point source pollutants. Now, biologists examine fish and flood plain connections, watershed-scale processes of fragmentation and connectivity (Dynesius and Nilsson 1994), identify hot spots of biodiversity and centers of population density and are involved in total maximum daily loading and best management practices implementation.

Then, the primary management paradigm was command-and-control, which respected the hierarchy and power differential among agencies. Today, we promote and encourage a collaborative and adaptive management paradigm and require public involvement in the process.

Then, the majority of state agency resources were devoted to the challenges of managing novel species complexes in reservoirs while fisheries in flowing waters were often ignored. Now, we look downstream from reservoirs and examine, through the Federal Energy Regulatory Commission relicensing process, alternative management strategies appropriate for riverine fisheries.

Then, the concept of minimum instream flow was a western concept, which was seldom a priority issue in warmwater streams. Freshwater appropriations are now a global concern, including warmwater streams (Postel et al. 1996). Now, we have adopted a natural flow paradigm, examining multiple elements of the flow regime that influence warmwater streams.

Management options available to the last generation of warmwater stream managers are still in use today. Regulations, stocking, and habitat improvement were all major topics discussed by Fajen (1980). Due to changing angler demographics, common regulations have changed. In 1980, many bass populations were managed using minimum size limits and fishery closures to protect spawning stocks. Today, anglers are more interested in catching quality bass than taking some home for the frying pan. As a result, regulation management has shifted to the use of high-end slot limits and catch-and-release regulations designed to provide trophy bass angling. Fajen (1980) mentioned supplemental stocking as a management option in cases where streams lack adequate spawning conditions or are overexploited, but he cautioned against this management technique. In this symposium, one of the smallmouth bass presenters suggests that supplemental stocking may be a viable tool to provide consistent recruitment in warmwater streams.

In 1980, game species like smallmouth bass garnered most of the management interest. Today, other species are of interest as well. In this symposium, nongame fish species and mussels are part of the management focus in warmwater streams. Another symposium at this meeting is focusing exclusively on ictalurid fish management in warmwater streams. A large portion of Fajen's (1980) discussion focused on instream habitat improvement as a supplemental tactic within a more comprehensive strategy of watershed management. The stream habitat improvement session in this symposium also has that orientation. What has likely changed most about instream habitat improvement is the technology that is used to assess channel disturbance and guide habitat placement, as talks in our Stream Habitat Improvement Session will demonstrate.

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Smallmouth Bass Recruitment Based on the Goldilocks Principle: Sometimes It's Too Wet, Sometimes It's Too Dry, and Sometimes It's Just Right

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Stream-dwelling populations of smallmouth bass Micropterus dolomieu commonly experience highly variable recruitment, frequently associated with abiotic factors (Paragamian 1984; Slipke et al. 1998; Buynak and Mitchell 2002). This variability can affect these populations and their associated fisheries (e.g., Paragamian 1984; Buynak and Mitchell 2002). Lotic populations of smallmouth bass in Virginia follow this pattern of irregular recruitment.

Our study contained three primary objectives. The first objective was to determine an index of recruitment success in smallmouth bass populations. The second objective was to identify predictable relationships between smallmouth bass recruitment and stream discharge parameters. The final objective was to examine a conceptual model utilizing supplemental stocking to enhance or stabilize recruitment.

This study was conducted on the James, Rappahannock, and Shenanodah Rivers, Virginia. These are all moderately large Atlantic slope rivers. Smallmouth bass were collected annually with electrofishing boats during the fall season. Sampling on the James River took place from 1991-2001, while sampling on the Rappahannock and Shenandoah Rivers took place from 1996-2002.

Catch-per-unit-effort (CPUE; fish/hr) described the relative abundance of smallmouth bass. Age 0 CPUE was compared with the abundance of older fish in succeeding years. Age 0 CPUE was also compared with selected stream discharge variables before, during, and after the primary spawning months. Finally, a Beverton-Holt dynamic pool model of the James River smallmouth bass population was constructed using FAST 2.0 (Slipke and Maceina 2000). This model simulated the effects of supplemental stocking on the population. Two supplemental stocking strategies were compared. One stocking strategy was to supplement weak year classes, while the other was to supplement average year classes. For all statistical tests, α was set at 0.10.

The CPUE of age 0 smallmouth bass displayed high annual variability in the study (range = 0.9-48.0 fish/hr). The CPUE of age 0 fish was positively related to the CPUE of age 1 fish the following year in the James (r^2 = 0.83; P = 0.01) and Rappahannock Rivers (r^2 = 0.82; P = 0.01). These data were unavailable for the Shenandoah River. Additionally, age 0 CPUE was positively related to catch curve residuals in the James River (r^2 = 0.84; P < 0.01), indicating that the CPUE of age 0 fish identified year-class strength. These relationships in the Rappahannock and Shenandoah Rivers were positive; but not significant, possibly due to low sample sizes ($N \leq 6$). Thus, the CPUE of age 0 smallmouth bass in the fall was a valid indicator of year-class strength.

Monthly stream discharge variables (e.g., mean, minimum, maximum, standard deviation, etc.) were compared with age 0 CPUE for the months of March-July. Of the variables examined, mean June discharge had the strongest relationship with smallmouth bass recruitment. This relationship was nonlinear in all cases (Figure 1). The models were statistically significant in the James and Shenandoah Rivers ($\underline{r}^2 = 0.83$; $\underline{P} < 0.01$ and $\underline{r}^2 = 0.81$; $\underline{P} = 0.08$, respectively). Similar, but non-significant, results were found in the Rappahannock River ($\underline{r}^2 = 0.66$; $\underline{P} = 0.11$). Strong year classes were associated with near normal flows during June, while weaker year classes were observed in years with lower or higher than normal June flows.

The James River smallmouth bass population was modeled using empirical data with FAST 2.0 (Slipke and Maceina 2000), resulting in a total population of age 1 and older fish ranging from 77,000-195,000 fish annually (mean = 131,293; $\underline{s} = 249.5$). The first supplemental stocking strategy (SS 1) stocked 50,000 fingerling smallmouth when natural recruitment was at or below the 20th percentile (67,492 fish) based on 100 years of modeling (Figure 2). This changed a weak year class into an average one. The second supplemental stocking strategy (SS 2) stocked 50,000 fingerlings when natural recruitment fell between the 40th and 60th percentiles (91,800-114,700 fish), creating strong year classes from average ones. Both stocking strategies increased the number of fish in the population, assuming survival rates of stocked and naturally reproduced fish were equal. Supplementing weak year classes (SS 1) eliminated weak year classes from a frequency of 14/100 years to 0/100 years. Supplementing average year classes nearly doubled the frequency of strong year classes from 18/100 years to 33/100 years. In either stocking strategy, fish were stocked 20 times over the course of 100 years, so the total number stocked was identical.

Variable recruitment has affected smallmouth bass populations in this study and elsewhere (Paragamian 1984; Buynak and Mitchell 2002). We found strong and weak year classes were persistent, similar to results from others (e.g., Cleary 1956), and that the CPUE of fall-collected age 0 fish accurately assessed year-class strength.

While other studies noted associations between stream discharge and smallmouth bass recruitment success, these described negative linear relationships (Cleary 1956; Slipke et al. 1998; Buynak and Mitchell 2002). Our findings indicated a parabolic relationship, with reduced recruitment success at both high and low flows. Similarly, Paragamian and Wiley (1987) found a parabolic relationship between stream discharge and growth of age 1 smallmouth bass. We did not identify the actual mechanisms controlling smallmouth bass recruitment as related to stream discharge, but they are likely different at high and low flows.

Variable recruitment of smallmouth bass affected the quality of the smallmouth bass fisheries in our rivers. As flows in these rivers were essentially unregulated, altering stream discharge to control smallmouth bass recruitment was not an option. Supplemental stocking may offer one option for managing recruitment. Stocking fingerling smallmouth bass to eliminate weak year classes (SS 1) could stabilize these fisheries. This management strategy might maintain angler catch rates at acceptable levels. Stocking fingerlings to create strong year classes (SS 2) would not increase stability in these fisheries. However, it could increase the frequency of years with high angler catch rates. Additionally, as these smallmouth bass populations experience relatively slow growth and moderate-high mortality, the abundance of trophy-sized fish may be positively linked to strong year classes, as suggested by preliminary evidence and modeling. The SS 2 management option may increase the number of trophy-sized fish available to anglers.

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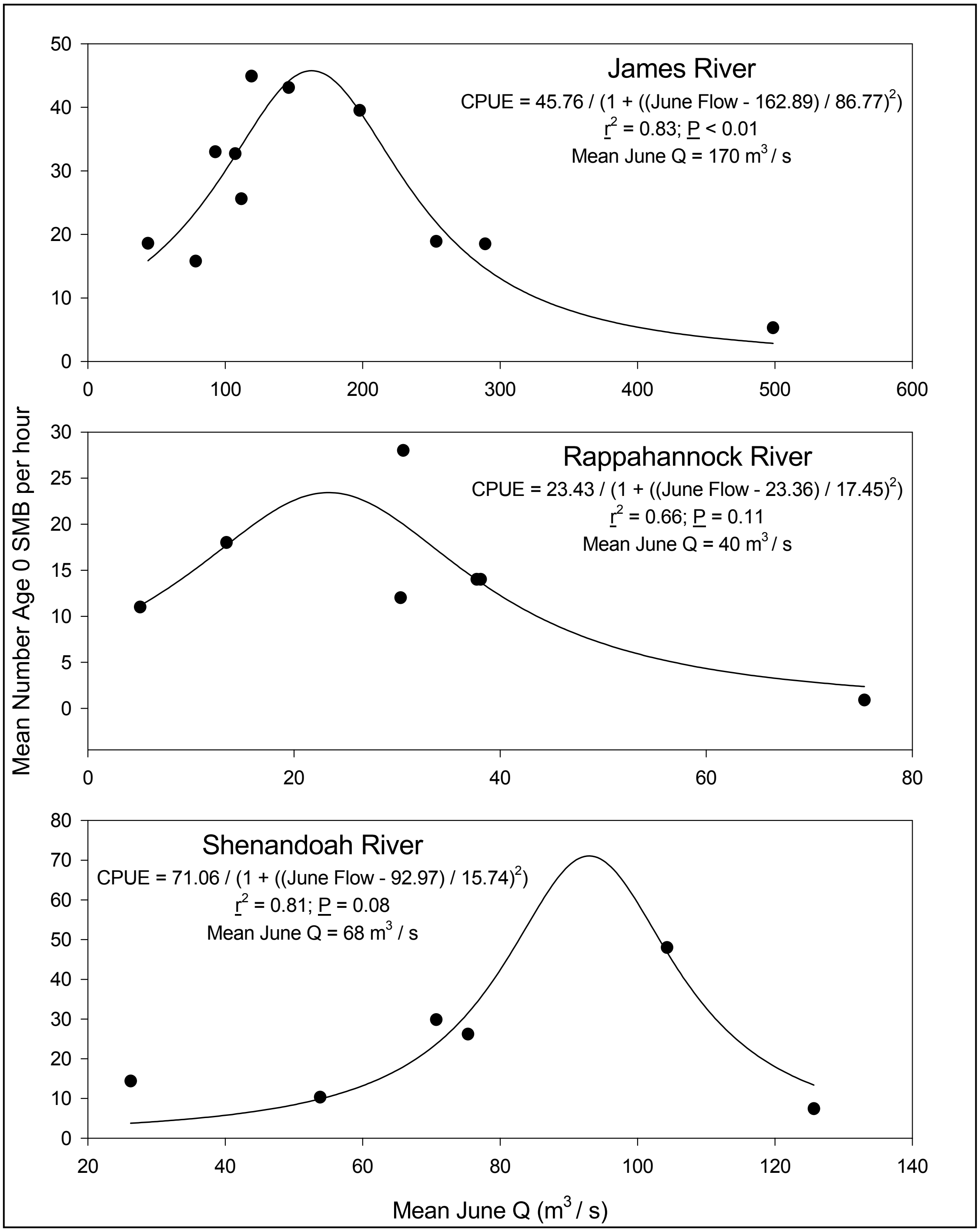

Figure 1. Mean CPUE (n/h) of age 0 smallmouth bass and mean June stream discharge (m^3/s) at Cartersville (James River), Fredericksburg (Rappahannock River), and Millville, WV (Shenandoah River), 1991-2002.

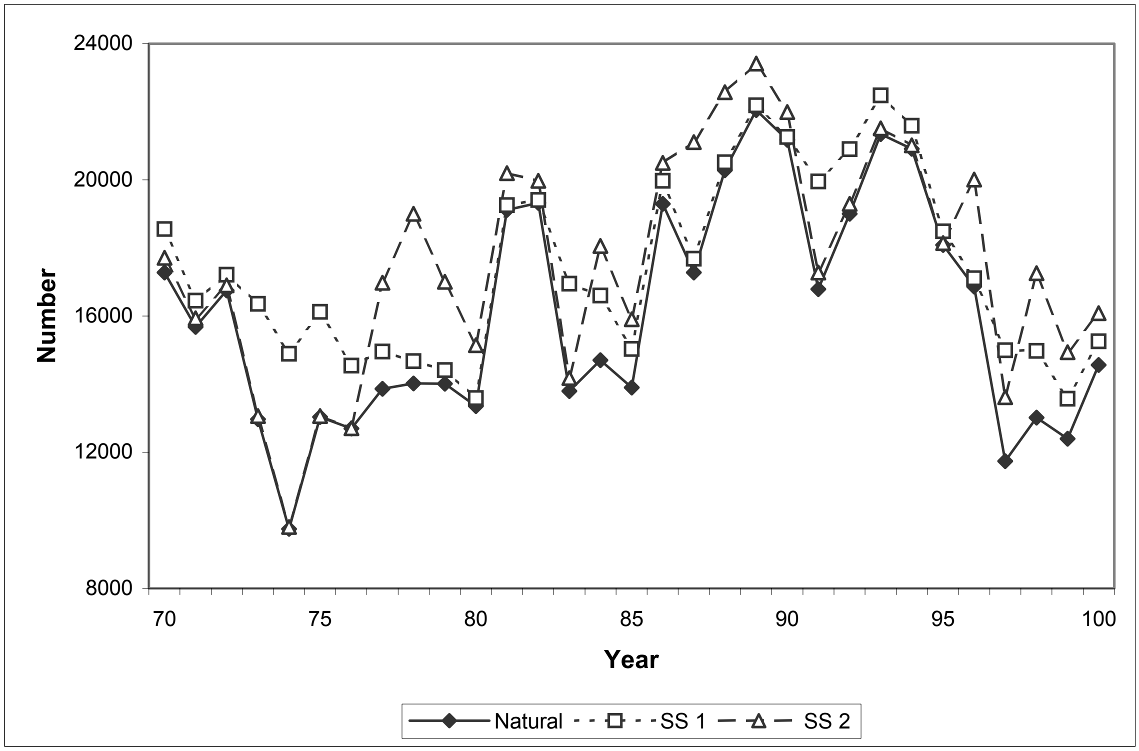

Figure 2. Number of smallmouth bass ≥ age 5 in the James River for years 70-100 as estimated by FAST modeling for 100 years. The solid line (diamonds) represents numbers from natural recruitment alone. The dotted line (squares) represents numbers with supplemental stocking (SS 1) of 50,000 fish in years when natural recruitment < 67,492 fish. The dashed line (triangles) represents numbers with supplemental stocking (SS 2) of 50,000 fish in years when natural recruitment ranges from 97,861-114,707 fish.

Estimating Smallmouth Bass Population Size and Biomass in Virginia Rivers Using Multiple-Pass Boat Electrofishing

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Biologists with the Virginia Department of Game and Inland Fisheries routinely use single-pass electrofishing surveys to evaluate smallmouth bass relative abundance and population size structure. Age determinations are made from otoliths, and estimates of year class strength, growth and mortality are determined. Single-pass electrofishing samples are conducted because multiple-pass runs are labor and time intensive. However, single-pass electrofishing may provide erroneous estimates of size structure and population parameters due to numerous biases. Although also bound by important assumptions, more intensive depletion electrofishing utilizing multiple vessels may provide better estimation of true population size structure. Additionally, depletion electrofishing can obtain population and biomass estimates.

We conducted a study to determine if smallmouth bass could be successfully depleted in Virginia rivers. Additionally, we wanted to estimate smallmouth bass biomass and determine the efficacy of annual single-pass electrofishing surveys relative to more intensive depletion sampling.

Depletion electrofishing was conducted at four sites on the Rappahannock River (4134 km^2 watershed and average annual discharge of 47 m^3/s) and five sites on the James River (17,503 km^2 watershed and average annual discharge of 211 m^3/s). Rappahannock River sites averaged 283 m in length, and average widths ranged from 46 to 162 m. James River sites averaged 734 m in length, and average widths ranged from 50 to 165 m. Mean sample areas on the Rappahannock and James Rivers were 2.9 and 8.6 ha.

Sampling was conducted with four to nine electrofishing boats. Sampling began at downstream gradient barriers and progressed upstream to upper gradient barriers. After three runs were completed, counts of adult smallmouth bass collected in each run were regressed against run numbers. Runs continued until statistics were deemed significant (either $P<0.10$ or $r^2>0.85$) or six runs were completed.

Depletion data were analyzed using Microfish 3.0 (Kulp and Moore 2000). Microfish software generated population estimates from removal data based on the Burnham maximum-likelihood estimation theory. Estimates were generated for young-of-year (YOY) and adult smallmouth bass. Maximum YOY total length (120 and 140 mm on the Rappahannock and James Rivers, respectively) was based on previous research and length frequency distributions. Smallmouth bass that were not YOY were considered adults. Biomass estimates were determined using weights recorded during sampling. Population estimates from Microfish were then compared to those derived from the Leslie method – least squares linear regressions of catch per minute against cumulative catch at each site (Maceina et al. 1993).

Comparisons were made between depletion data and single-pass data collected in 2000 and 2001 to determine if single-pass sampling provided biased or incomplete representation of smallmouth bass size structure compared to depletion sampling. Mean total lengths of stock-size fish from depletion sampling were compared with single-pass sampling using ANOVA, and size structures from these datasets were compared using PSD and RSD-P (Miranda 1993).

Adult smallmouth bass were successfully depleted in three to five runs at most sites. At two sites on the Rappahannock River, highly significant regression statistics were derived after only three electrofishing passes, while other sites on both rivers required four or five passes to produce acceptable statistics. Sites producing the best regression statistics with the least amount of effort were either higher in the watershed or characterized by less habitat complexity.

Population estimates for adult smallmouth bass on the Rappahannock River varied from 167/km to 740/km (mean=386/km, SE=132) and 13/ha to 93/ha (mean=47/ha, SE=17) (Table 1). Estimates for adult smallmouth bass on the James River varied from 50/km to 525/km (mean=265/km, SE=90) and 3/ha to 115/ha (mean=38/ha, SE=20). Population estimates for young-of-year smallmouth bass on the Rappahannock River varied from 49/km to 327/km (mean=221/km, SE=60) and 10/ha to 70/ha (mean=28/ha, SE=14). Estimates for young-of-year smallmouth bass on the James River varied from 129/km to 596/km (mean= 248/km, SE=88) and 8/ha to 82/ha (mean=31/ha, SE=14). Capture probability (P) was highest (mean=0.40) for adult smallmouth bass in the Rappahannock River and lowest (mean=0.17) for young-of-year smallmouth bass in the James River.

Smallmouth bass biomass estimates on the Rappahannock River averaged 69 kg/km and 8.6 kg/ha (SE=21, 3.4), while estimates on the James River averaged 66 kg/km and 9.3 kg/ha (SE=20, 4.8; Table 2).

Leslie depletions of catch vs. cumulative catch usually resulted in negative linear relationships. Population estimates based on Leslie depletions were greater than those generated by Microfish, but differences were consistent, and most sites differed by 28% or less.

Comparisons between mean total length of stock-size smallmouth bass from depletion electrofishing and single-pass electrofishing on the Rappahannock River suggested that single-pass runs resulted in size selectivity bias at 50% of sample sites (mean lengths were greater for depletion samples). Conversely, comparisons of James River smallmouth bass suggested little size selectivity bias existed between depletion and single-pass samples, as only one site had significantly different mean total lengths.
Stock index evaluation supported mean total length data. Size structures varied in one of two Rappahannock River comparisons, but James River comparisons had minimal bias. This study suggests that adult smallmouth bass can be successfully depleted from sample reaches within Virginia rivers. Although the upper tier of sampling effort included five electrofishing passes with nine boats, desirable outcomes were achieved with less effort.

Estimates derived from this study will be used to begin planning potential smallmouth bass supplemental stocking regimes. Additionally, this study provides cautious optimism that single pass fall electrofishing surveys adequately describe smallmouth bass population size structure in some Virginia rivers. However, further study is required to elucidate this relationship, which will become more important as efforts increase to evaluate recently enacted slot length limits. Future study should incorporate additional population estimation techniques such as mark-recapture, telemetry, or underwater observation.

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Table 1. - Population estimates of adult and young-of-year (YOY) smallmouth bass calculated with Microfish 3.0 and expanded for number of fish per kilometer and hectare. River codes: RAP=Rappahannock River, JAM=James River. Means given with standard errors in parentheses. P=Capture Probability.

Adults

River	Site	No. Runs	Catch	Estimate	95% C.I.	P	No/km	No/ha
RAP	Embrey	3	37	50	37-75	0.36	204	13
RAP	I95	5	184	225	194-256	0.29	740	49
RAP	Elys	4	105	111	105-119	0.51	432	93
RAP	Phelps	3	44	54	44-71	0.45	167	34
Mean			93 (34)	110 (41)		0.40	386(132)	47(17)
JAM	Columbia	6	67	129	67-235	0.11	108	8
JAM	Bremo	3	37	43	37-55	0.47	50	3
JAM	Lynchburg	5	199	330	216-443	0.17	411	30
JAM	Buchanan	3	217	413	222-604	0.22	231	32
JAM	Lick Run	3	259	281	265-297	0.57	525	115
Mean			156 (44)	239(67)		0.31	265(90)	38(20)

YOY

River	Site	No. Runs	Catch	Estimate	95% C.I.	P	No/km	No/ha
RAP	Embrey	3	12	60	12-579	0.07	245	15
RAP	I95	5	54	80	54-121	0.20	263	17
RAP	Elys	4	26	84	26-351	0.09	327	70
RAP	Phelps	3	16	16	16-18	0.67	49	10
Mean			27 (9)	60 (16)		0.26	221(60)	28(14)
JAM	Columbia	6	40	150	40-609	0.05	158	12
JAM	Bremo	3	19	95	19-757	0.07	129	8
JAM	Lynchburg	5	75	97	75-124	0.25	155	11
JAM	Buchanan	3	138	384	138-812	0.14	596	82
JAM	Lick Run	3	61	88	61-129	0.32	200	44
Mean			67 (20)	163 (56)		0.17	248(88)	31(14)

Table 2. – Biomass estimates of smallmouth bass (adults and young-of-year) from two Virginia rivers. River codes: RAP=Rappahannock River, JAM=James River. Means given with standard errors in parentheses.

River	Site	Kg/km	Kg/ha
RAP	Embrey	40	2.5
RAP	I95	121	8.0
RAP	Elys	85	18.3
RAP	Phelps	28	5.6
Mean		69 (21)	8.6 (3.4)
JAM	Columbia	41	3.0
JAM	Bremo	25	1.5
JAM	Lynchburg	46	3.3
JAM	Buchanan	82	11.3
JAM	Lick Run	137	27.4
Mean		66 (20)	9.3 (4.8)

A Process-Driven Approach for Assessing Warmwater Streams in Wyoming

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Fisheries management in the western United States has been dominated largely by sport fisheries interests with most of the focus on coldwater salmonids. However, the decline of native fishes species throughout the western United States has created a need to understand the effects of anthropogenic activities on warmwater streams and the consequent effects on native fish assemblages. Adequate assessment techniques are difficult to develop for warmwater stream systems due to high fish species richness, diverse life history strategies, and complex interactions among abiotic and biotic processes. Although a variety of methods have been proposed for sampling fish and habitat, the only methods that provide some level of assessment use indices (e.g., Index of Biotic Integrity). Assessment techniques that rely on an index are appealing due to their simplicity and ability to summarize large, complex data sets. However, index-based approaches do not provide assessments that focus on the ecology of individual species. Similarly, indices of habitat conditions may obscure patterns in the data because habitat that is important for one species may not be important for other species. Moreover, most index-based techniques have neither provided an integration nor interpretation of fish and habitat information. Consequently, we developed a method, termed the Warmwater Stream Assessment (WSA), for evaluating habitat characteristics and fish assemblages in warmwater streams that avoids the use of indices, focuses on the ecology of fishes, and integrates information on fish and habitat characteristics.

Sampling for the WSA is conducted at the reach scale. Reaches are selected to represent characteristics of larger stream segments. Prior to sampling, information on anthropogenic disturbances in the watershed is obtained and a list of possible species is compiled based on distribution maps for each species and the large-scale characteristics of the reach (e.g., elevation). Fish are sampled by electrofishing and seining, and fish are identified to species and counted. Habitat is categorized as either reach-scale or channel-unit (e.g., pool, riffle) habitat. Estimates of reach-scale habitat include channel morphology (e.g., percentage of the reach as pool habitat), elevation, turbidity and intermittence characteristics, mean width, and maximum depth. Measurement of channel-unit habitat focuses on depth, substrate composition, and instream cover. Because warmwater fishes are generally dependent on combination of habitat, channel-unit habitat is measured using a data matrix approach where combinations of depth strata, substrate type, and instream cover are recorded for individual channel units.

The fish and habitat data are organized and evaluated using decision trees and a summary table. Decision trees are hierarchical and provide a systematic account of the zoogeography, reach-scale and channel-unit habitat requirements, and sensitivity to biotic interactions for each species. The summary table summarizes and assimilates information on abiotic and biotic characteristics of the reach and requirements of fishes. Decision trees focus on individual species, while the summary table focuses on patterns in the fish assemblage. The last step in the WSA process is to determine the abiotic and biotic features that are likely to have been changed since settlement by Europeans, and how and why conditions have been altered. These insights are obtained by assimilating information regarding anthropogenic activities in the watershed and fish assemblage and habitat characteristics of the reach.

Data collection, analysis, and evaluation should be considered as a synergistic process, not simply the activity of acquiring data. However, few sampling programs progress to the level of assessment. The WSA provides information on the expected native fish assemblage in a stream reach, facilitates comparison to the current assemblage of native and introduced species, and provides insight as to major factors influencing the presence of individual and groups of species. The WSA is a simple process that enables biologists to assimilate habitat and fish assemblage data and integrate the data with information on the ecology of each species. The WSA provides insight on causal mechanisms and identifies factors that are likely to have been altered from their historic condition. It can provide a foundation for more intensive study because knowledge of probable influential factors in a reach ensures that more intensive studies are focused, time and cost efficient, and answer meaningful questions related to fish and habitat management.

Influence of Long-Term Streamflow Variation on Recruitment of Riverine Fish Populations

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River regulation and water development are the foremost problems threatening stream fishes and other lotic biota in the southern United States. The influence of these factors, however, occurs in systems with high natural variability over time that must be assessed and quantified before relationships can be attributed to management actions. Thus, we used a long-term (13 year) dataset to assess the inter-annual variability in the recruitment of 26 warmwater fish species in a large Midwestern stream and related it to variation in river discharge. Our analyses included economically and recreationally important species, such as largemouth bass and smallmouth bass, as well as non-game catostomid, clupeid, cyprinid, and atherinid species.

We estimated young-of-year (YOY) fish density from fish assemblage survey data collected annually from a 2.5-km reach of the Kankakee River in Will County, Illinois, in late summer from 1977 to 1990 (excluding 1980). The Kankakee River is a large, low-gradient, unregulated prairie river typical of the region. For each species, we identified YOY fishes using length-frequency histograms, and estimated species-specific YOY density by adjusting catch data with gear-specific capture efficiency models (Bayley and Dowling 1990). We used hierarchical linear models with a random intercept to evaluate guild and discharge variables as predictors of YOY fish density. We assessed relative importance of the predictor variables using Akaike's Information Criteria with small sample bias adjustment (AIC_c; Burnham and Anderson 2002).

We evaluated eight guilds as potential predictors of YOY recruitment. Guilds were created by evaluating similarity among species based on one or more life history characteristics via hierarchical cluster analysis and the Jaccard similarity index. We initially constructed a life history matrix composed of 31 variables among eight general groupings: general habitat use, general flow requirements, month when spawning initiated, spawning flow requirements, spawning duration, spawning guilds (Balon 1975), lifespan, and age at maturity. We created one guild using all 31 characteristics, one guild using subsets of spawning characteristics, and four guilds based on individual spawning characteristics. We also evaluated trophic guilds as defined by Poff and Allan (1995) and taxonomic groups to family. We evaluated influence of five discharge (m^3/s) variables during spawning/rearing (April-July): mean, minimum, maximum, coefficient of variation (CV), and percent mean daily change in discharge (delta).

The most plausible model explaining YOY fish density contained a guild defined by the timing of spawning initiation (April, May, or June) and mean discharge. The spawn initiation guild was 9.3 times more likely the best guild predictor of YOY density than the next best predictor, the guild containing all life history variables. The species group that began spawning in April was composed of six sucker species, carp, white and black crappie, and rock bass. Six cyprinid species, five centrarchid species (including largemouth and smallmouth bass), and blackstripe topminnow began spawning in May. Only two species, brook silverside and mimic shiner, began spawning in June. Considering these groups within the spawning initiation guild, the most plausible models of YOY fish density contained all groups, mean discharge, and interactions between mean discharge and groups that began spawning in April and June (Figure 1).

Among measures of discharge, mean discharge was 4.4 times more likely than the next best predictor, maximum discharge (Table 1). However, mean discharge was positively correlated with maximum discharge ($r = 0.89$, $P < 0.0001$), minimum discharge ($r = 0.89$, $P < 0.0001$), and percent mean daily change ($r = 0.55$, $P < 0.0001$). In a regulated river system with artificially high and low flows, patterns in relationships among these variables would differ, and mean discharge would likely have less predictive value.

Recruitment of all species decreased as mean discharge during spawning/rearing increased (Figure 1). Mean discharge during spawning/rearing in the Kankakee ranged from 95 - 313 m^3/s. Recruitment failure by any group did not occur under these conditions, but years at the upper end of this range experienced low recruitment relative to drier years. Species that began spawning in May had the highest overall density in the Kankakee River for all discharge conditions, however, this group was more sensitive to increasing discharge than groups that began spawning in April or June. It appears that river characteristics under high discharge conditions (high velocities, turbidity) more negatively affected the cyprinid and centrarchid fishes in this group than other species.

Our results indicate that mean flow during the spawning/rearing period was more important than flow stability or habitat availability (minimum flow) in predicting YOY density in the Kankakee River. However, we recommend testing these hypotheses in river systems with differing geomorphologic and fish assemblage characteristics to better evaluate relationships between YOY recruitment and discharge variables across a range of environmental conditions. We will integrate patterns observed in this study as we continue to develop predictive models to assess the impacts of river regulation on fish assemblages to aid in management of fisheries in regulated rivers.

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Table 1. Importance (sum of weights of all models containing each variable) of the discharge variables we evaluated as predictors of YOY density on the Kankakee River: mean, maximum (max), minimum (min), no flow variable in model (.), coefficient of variation (cv), and percent mean daily change in discharge (delta).

Discharge	Importance
Mean	0.58
Max	0.13
Min	0.09
.	0.07
CV	0.06
Delta	0.06

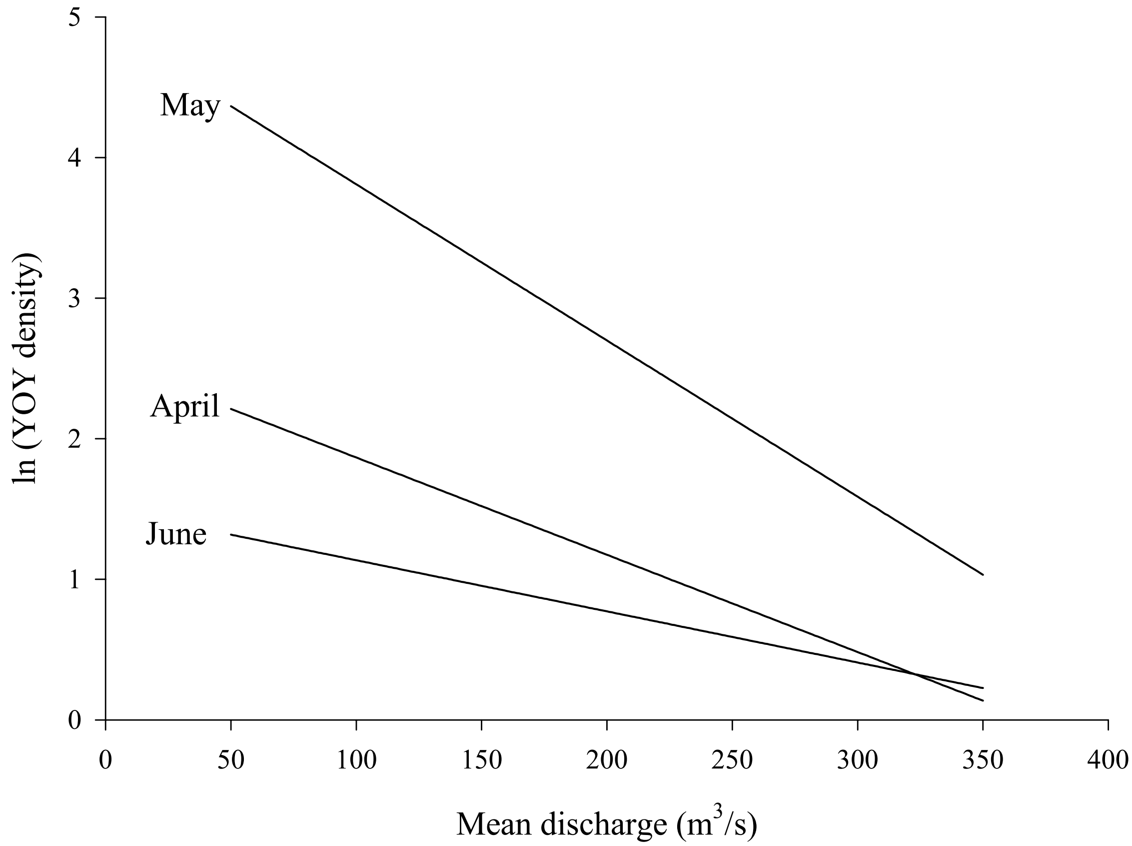

Figure 1. Natural log of YOY density versus mean discharge for all months within the spawning initiation guild.

Sampling Considerations for Assessing Prairie River Fish Assemblages

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Success at incorporating biotic integrity into water resource management depends on an appropriate and cost-effective procedure for sampling aquatic organisms, including fish (Karr 1991). At present, there is no formal protocol to sample and assess the overall health of fish assemblages in large prairie rivers. Prairie rivers are generally characterized as having a wide, relatively shallow, braided channel with a sandy substrate and little in-stream structure or vegetation. Prairie rivers also are flashy (i.e. highly variable discharge) and have relatively high conductivities (>1000 µS/cm). The purpose of this study was to develop a cost effective, yet scientifically sound, protocol to sample fish in large prairie rivers in Oklahoma.

Development of the protocol has proceeded through four phases: 1.) classification of in-stream habitat, 2.) delineation of site reach, 3.) evaluation of gear efficiency, and 4.) determination of sampling effort.

Six habitat types were initially identified based on water depth and flow measured in the field (Gorman and Karr 1978). Fish were collected from all 6 habitat types at three different locations on the Cimarron River near the towns of Dover, Guthrie, and Coyle, Oklahoma. Cluster analysis and analysis of similarities (ANOSIM) of these fish collections subsequently reduced these six habitat types to three general types (shallow water, deep water, and backwater; Table 1).

Habitat was delineated for a 43 km section of the Cimarron River, between Coyle and Ripley, Oklahoma, from Digital Orthophoto Quads using ArcView GIS software. This section was divided into 50-m transects, and the frequency of habitat types was calculated for each transect. Average proportion of each habitat type was calculated over the entire reach, and the minimum river distance needed to achieve this proportion was calculated for each habitat type. Shallow water averaged 58.8% of the area over the entire section and reached this proportion after 1250 m. Deep water averaged 37.4% and reached this proportion after 1400 m. Backwater averaged 3.8% and reached this proportion after 1150 m. From these results we concluded that, at minimum, a 1400 m section needs to be sampled in this reach of the Cimarron River to encounter habitats in proportion to their occurrence.

Because of the high conductivity in prairie rivers we were unable to use electrofishing. Therefore, we used a seine as the primary gear type for shallow water and backwater habitats and a hoop net as the primary gear type for deep water habitats. Seine and hoop net efficiency were evaluated based on the number of species detected in each of the three habitat types. Seine efficiency was evaluated and averaged for 3 shallow water habitats and 4 backwater habitats. Hoop net efficiency was evaluated and averaged over a combined 36 net nights (or 24 h sets).

In shallow water and backwater habitats, fish were sampled with one pass by a 6.1 m x 1.2 m (4.8 mm mesh) seine. The remaining fish were sampled with a 5.0% rotenone solution at 3 ppm for 15 min. Efficiency was calculated as the ratio of the number of species detected by the seine and the total number of species detected from both methods. On average, the seine detected 54.2% of the species in shallow water habitats and 57.7% of the species in backwater habitats.

In deep-water habitats, hoop nets were used to calculate the mean number of species detected per net night. Two types of hoop nets were used; a large net (0.9 m x 3.7 m x 50.8 mm mesh) and a small net (0.6 m x 2.4 m x 25.4 mm mesh). We found no significant difference between the two net sizes in terms of the number of species caught. Regardless of which net was set out, 1 to 5 species were detected with an average of 1.5 species per net night.

A species accumulation curve was created based on habitat type using fish data collected from the Cimarron River near the towns of Dover, Guthrie, and Coyle, Oklahoma (Figure 1). Maximum sampling effort was defined as the point on the species accumulation curve where no new species were encountered in each habitat type. Maximum species richness was attained after 7 samples in shallow water habitats, 2 samples in deep-water habitats, and 4 samples in backwater habitats. These results suggest a disproportionate sampling design is needed to maximize efficiency. For example, more “rare” habitat types (e.g. backwater) and fewer “common” habitats (e.g. shallow water) would need to be sampled relative to their areal coverage.

With this protocol, we found that a representative sample of the fish assemblage could be attained in a two-day period in the Cimarron River. The first day would consist of in-stream habitat classification and mapping followed by sample allocation and setting of hoop nets (in deep-water habitats based on the disproportionate sampling technique). On the second day, hoop nets would be pulled (after 24 h) and seining would be executed in the shallow and backwater habitats based on the disproportionate sampling allocation from the previous day. We plan to test the protocol in other prairie rivers to evaluate its efficiency.

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Table 1. ANOSIM of fish collections from sites on the Cimarron River near Dover, Guthrie, and Coyle, Oklahoma. An R-statistic less than 0.5 indicates the two habitat types have a similar species composition, and a significance level greater than 5% means the two habitat types are significantly similar in terms of species composition. Initial habitat types were: shallow fast (SF), shallow slow (SS), deep fast (DF), deep slow (DS), non-wadeable (NW), and backwater (BW). An asterisk indicates significantly similar habitat types. Final habitat types, reduced from the initial six types were: shallow water (SW), deep water (DW), and backwater (BW). All three final types had a significantly different species composition.

Habitat Types	R Statistic	Significance Level (%)
Initial comparisons		
DS, NW*	-0.018	60.0
SS, SF*	-0.010	50.9
DS, DF*	-0.071	46.7
DF, NW*	0.344	8.6
BW, DS	0.841	1.8
SS, DS	0.933	0.7
SF, DS	0.947	0.7
SS, NW	0.886	0.3
SS, BW	0.548	0.1
SS, DF	0.964	0.1
SF, BW	0.776	0.1
SF, DF	0.928	0.1
SF, NW	0.927	0.1
BW, DF	0.905	0.1
BW, NW	0.726	0.1
Final comparisons		
SW, BW	0.699	0.1
SW, DW	0.896	0.1
BW, DW	0.469	0.1

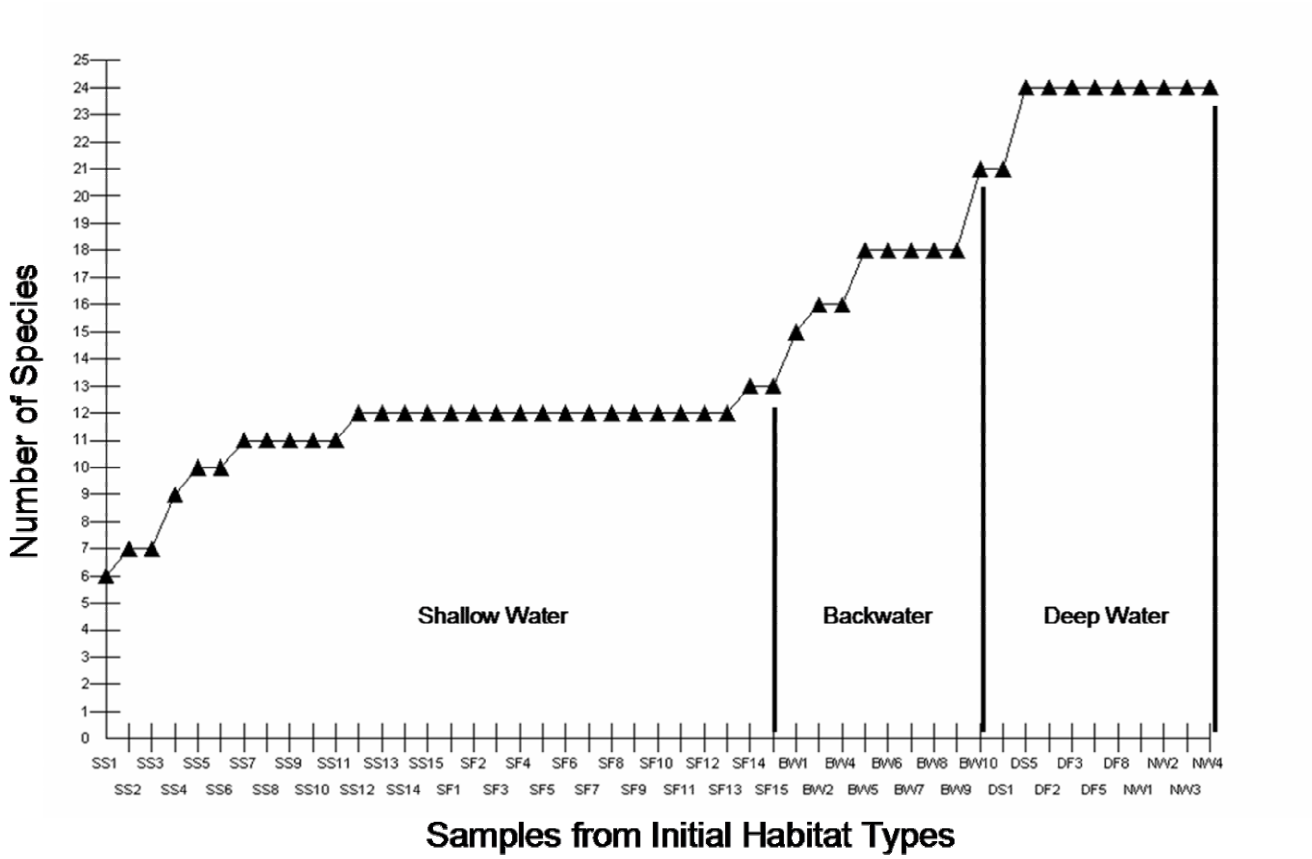

Figure 1. Species accumulation curve for fish collected from the Cimarron River near the towns of Dover, Guthrie, and Coyle, Oklahoma. Fish were sampled from the six initial habitat types: shallow fast (SF), shallow slow (SS), deep fast (DF), deep slow (DS), non-wadeable (NW), and backwater (BW). Samples are grouped by final habitat type.

Random Selection of Stream Sites: An Important Step in Fluvial Geomorphic and Fishery Surveys

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Researchers and managers survey streams to assess the status and trends of their physical and biological components. However, the selection of sampling locations has historically been dependent on handpicked stream sites or sites with easy access and has rarely incorporated a random element.

Sampling streams at sites that have not been randomly chosen can lead to erroneous conclusions regarding stream conditions and population trends of lotic organisms. Jacobs and Cooney (1997) reported that overestimates of coho salmon *Oncorhynchus kisutch* escapement in Oregon coastal streams occurred when using traditional standard survey sites when compared to estimates made at sites selected by a stratified random sampling survey. Use of the traditional survey sites lead to an overestimate of the naturally spawning coho salmon population. Balkenbush and Fisher (1997) found that public areas on the Glover River, Oklahoma contained higher densities of centrarchids, more instream cover, and were also deeper than remote areas. They concluded that employing an accessibility sampling strategy may yield biased estimates of population sizes and habitat availability. Employing a non-random or accessibility sampling strategy could have further consequences if erroneous conclusions are drawn regarding the status and trends of fish populations and habitat availability, which may lead to mismanagement or less-beneficial allocation of resources.

Stevens (1994) discussed the inherent characteristics of a probability sampling design. First, the population has to be described explicitly. Second, every element of the population has some opportunity of actually being sampled. And third, the selection is carried out by a process that includes an explicit random element. Employing a probability sampling design allows inferences to be made regarding the defined population.

Several examples of probability sampling designs for streams exist. The U.S. Environmental Protection Agency's Environmental Monitoring and Assessment program used a spatially balanced, randomized procedure to select 100 stream sites where chemical and biological information were collected. From those stream locations, they were able to make valid inferences about the chemical and biological integrity of 184,600 km of streams in the Mid-Atlantic Highland region (Herlihy et al. 2000). At a state level, Firman and Jacobs (2001) discussed the Oregon Department of Fish and Wildlife (ODFW) design for monitoring the status and trends of stream habitat and salmonids in Oregon. The ODFW design integrates three ongoing projects: spawning surveys, juvenile salmonids, and aquatic habitat inventory. It also incorporates the random element needed to make inferences about unsampled stream habitats and fish populations.

We used a geographic information system to develop a stratified random sampling design for a fluvial geomorphic and black bass survey of eastern Oklahoma streams. Our target population was the length of all Strahler stream orders 1-4 in the Boston Mountains, Central Irregular Plains, Ouachita Mountains, and Ozark Highlands ecoregions within Oklahoma. We randomly selected 160 sites from that population. To select stream sites, we first generated a stream network by using a 30-m digital elevation model from the National Elevation Dataset. The model constituted elevation data for all drainages for 1-5 order streams in eastern Oklahoma. A stream network was then created by identifying all grid cells that had a watershed greater than 1.35 km^2, and Strahler orders were assigned to all stream segments. Ten random points were selected per stream order for each ecoregion, i.e., ten sites per strata combination. At each stream site a reach of 20 times the mean channel width was delineated and channel units (e.g., riffles, runs, pools) were mapped and snorkeled for black bass *Micropterus spp.* Stream channels were classified by using cross-sectional data and maps. Measurement of geomorphic and fish variables at the selected stream sites facilitates valid statistical comparisons at multiple spatial scales and across different ecoregions.

About half of U.S. state fish and wildlife agencies survey stream fish populations every year, and over half allocate ≤5% of agency budgets to stream fisheries management and research (Fisher and Burroughs 2003). It is unknown to what degree most state agencies use probability-based sampling to monitor the status and trends of their stream resources. Employing a large-scale probability-based sample may not be feasible due to limited resources. However, using a sampling program without an explicit random element incorporated may limit the usefulness of collected data, or lead to a misunderstanding of the status and trends of aquatic resources. Monetary or personnel limitations may require that the defined population of interest be reduced in order to maintain the statistical validity of inferences made from samples. Moreover, state agencies may need to collaborate and combine resources with other agencies to meet the needs of a large-scale sampling program.

The importance of applying probability sampling designs to stream surveys has been known for sometime now. However, application of such designs has mostly been implemented by agencies with large jurisdictional boundaries for large-scale environmental and biological monitoring programs (Stevens 1994; Herlihy et al. 2000), with some exceptions (Firman and Jacobs 2001). We reiterate the importance of using probability sampling during surveys that are intended to infer states and conditions from explicitly defined stream populations, such as a single stream, a stream network in a watershed, or all streams within a large geographic area.

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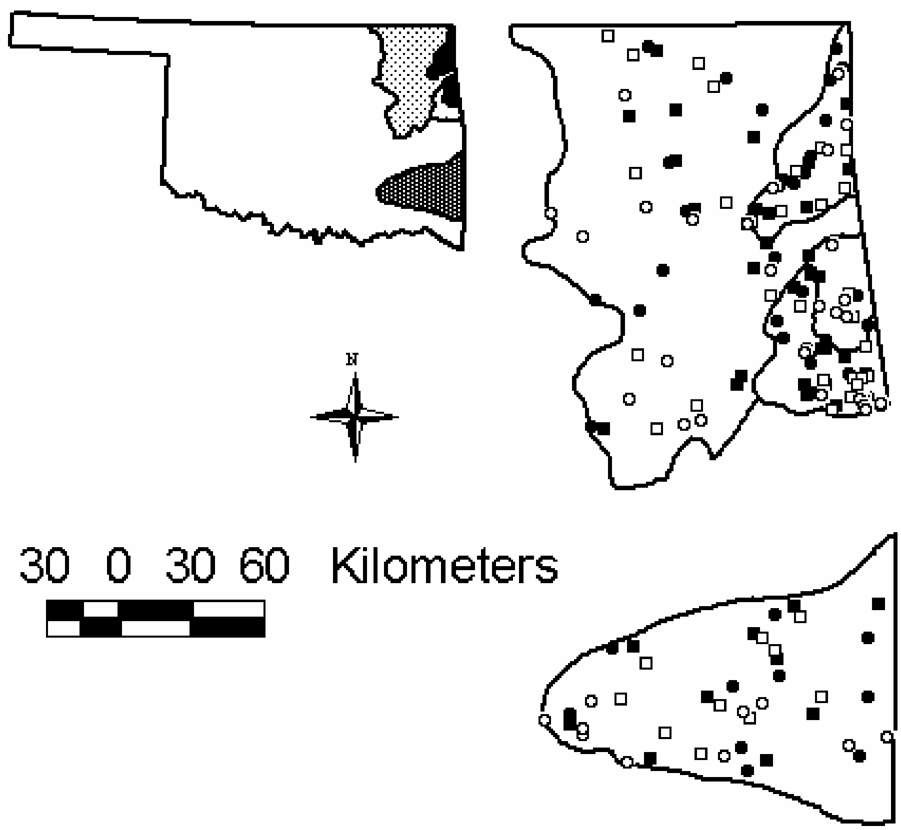

Figure 1. One-hundred sixty stream sites to be surveyed for fluvial geomorphic and black bass population variables. Ten sites were selected within stream orders 1 (●), 2 (■), 3 (□), and 4 (○) in the Boston Mountains (white), Central Irregular Plains (light grey), Ouachita Mountains (dark grey), and Ozark Highlands (black) ecoregions.

Stream Fisheries Management in the Southeastern United States: Trends from a Survey of State Agencies

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Warmwater stream and river fisheries in the southeastern United States have traditionally provided a variety of recreational opportunities for people in the region. However, they have received varying degrees of management, in part because agencies have been slow to acknowledge that warmwater stream fisheries deserve or require the type of management given to other aquatic habitats (Rabeni and Jacobson1999). Fisher et al. (1998) compared survey findings on stream and river fishing activity in the southeastern United States with those about stream management programs in the region and found a direct relationship between stream fishing activity and stream management program development. They concluded that, in general, stream and river fisheries in the Southeast are not being managed in proportion to their values. In a subsequent survey of stream fisheries management programs administered by state agencies in the United States, Fisher and Burroughs (2003) found regional differences in management activities and recommended a model program with clearly-defined goals, sufficient financial and human resources, up-to-date information, and cooperation with other agencies and groups to effectively manage stream fisheries resources. Herein we focus on the characteristics of stream fisheries management programs in the Southeast, based on findings from Fisher and Burroughs (2003) and the relationship described by Fisher et al. (1998).

We surveyed 15 state fish and wildlife agencies in the southeastern United States in 2000 to evaluate characteristics of their stream fisheries management programs. State agencies surveyed were in Alabama (AL), Arkansas (AR), Florida (FL), Georgia (GA), Kentucky (KY), Louisiana (LA), Maryland (MD), Mississippi (MI), North Carolina (NC), Oklahoma (OK), South Carolina (SC), Tennessee (TN), Texas (TX), Virginia (VA), and West Virginia (WV). The survey questionnaire contained 26 multi-part questions about program management, agency resources, fisheries resources, cooperative efforts, and management issues.

Almost half (47%) of the respondents indicated their state had a comprehensive stream fisheries management program, including four states (GA, TN, VA, WV) that Fisher et al. (1998) identified as having extensive stream management programs and three states (AR, OK, TX) that had limited programs. Sixty-two percent of the respondents listed maintaining and improving ecosystem integrity, and increasing angling quality and opportunities as their primary goals for managing streams fisheries. The average number of full-and part-time employees managing streams fisheries was 28 (range: 4 for MS to 96 for GA), which accounted for just over half (53%) of all freshwater fisheries agency employees. Agencies expended the greatest proportion of their budget on management, hatcheries, and law enforcement and the least on habitat improvement, land acquisition, and consultative services. Respondents said employees allocated a third of their time to management activities and another third to research and hatchery activities combined.

Agencies in the Southeast manage all components of stream fisheries, including populations, habitats and anglers. Nearly all agencies (96%) survey sportfish populations statewide annually or at some other interval by electrofishing to estimate population abundance for the establishment of regulations. Most agencies (60%) survey instream channel habitat units and riparian features as needed using channel unit identification and microhabitat measurements primarily for stream bank stabilization and riparian restoration projects. Only a minority (40%) of the respondents indicated they use geographic information and global positioning system technology to survey habitat. A clear majority of agencies (87%) conduct creel surveys of anglers using a variety of methods (mail and telephone surveys and direct observation) mainly to measure angler satisfaction, identify fishing trends, and determine resource utilization.

Cooperation with federal and state agencies and non-governmental agencies enable state fish and wildlife agencies in the Southeast to better manage stream resources. Federal agencies that most (>60%) state agencies cooperated with were the U.S. Fish and Wildlife Service, Army Corps of Engineers, U.S. Geological Survey, and U.S.D.A. Natural Resources Conservation Service. All southeastern state fish and wildlife agencies cooperated with their state water quality agency, and most (>60%) worked with state water resources and environment agencies. Non-governmental organizations that most respondents indicated their agency cooperated with included Trout Unlimited, B.A.S.S, The Nature Conservancy, and a variety of other organizations. Citizen groups, such as local fishing and hunting clubs and watershed organizations, were primary cooperators for stream management agencies.

Many of the papers presented at the inaugural Warmwater Streams Symposium in 1980 dealt with water pollution and impoundment impacts (Krumholz 1981). In our survey, all respondents rated water quality and contaminants, harvest regulations, and fishing pressure as either moderately or very important issue in the Southeast (Figure 1). Many respondents also rated impacts associated with impoundments (e.g., instream flow assessment, impoundment releases, hydropower relicensing) as important. In fact, of the 15 issues presented, only interbasin water transfer was rated as not important by a majority (54%) of the respondents. This is somewhat surprising given some of the more contentious water transfer projects (e.g., Santee-Cooper diversion project, Tri-State [AL, FL, GA] study) in the Southeast (Meador 1996).

Results from this survey indicate that several southeastern fish and wildlife agencies have increased their stream fisheries management activities compared with a similar survey conducted in the mid-1990s. This is particularly important in the

Southeast where increasing impacts from urban development, associated land transformations, and water demands, such as in the southern Piedmont (Conroy et al. 2003) and in the Dallas, Texas metroplex, threaten the sustainability of fisheries resources and stream ecosystems. Careful and comprehensive planning and new attitudes about sustainable natural resources management will be needed to successfully maintain and restore stream fisheries resources in the Southeast.

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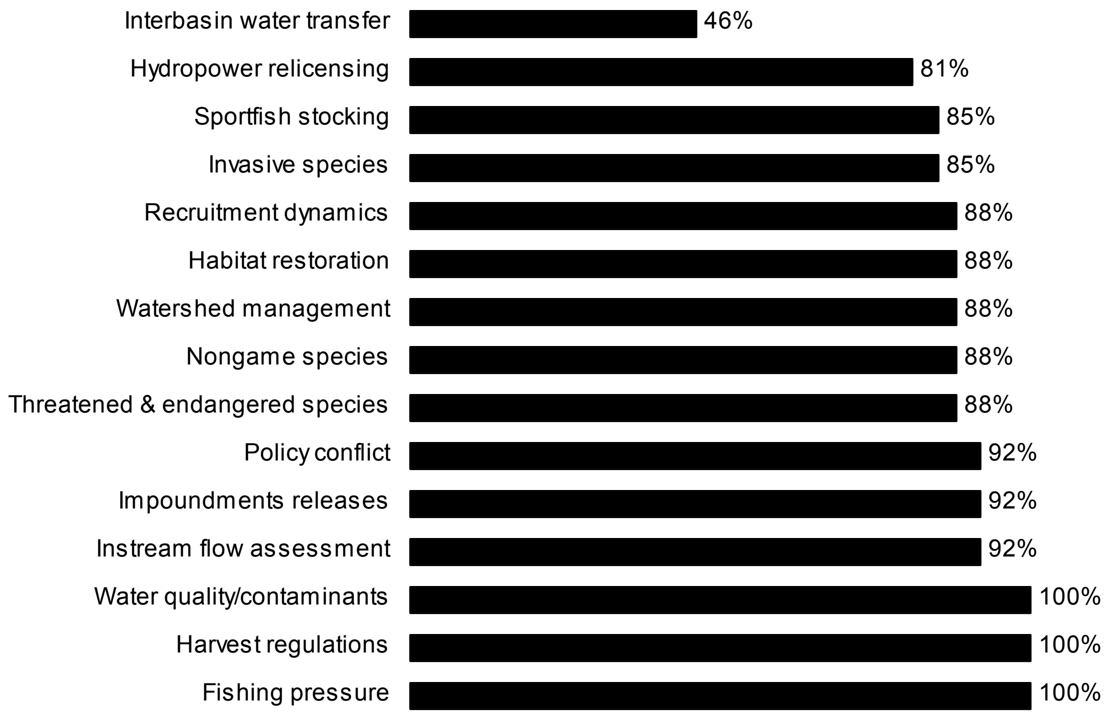

Figure 1. Percent of survey respondents from agencies in the southeastern United States that rated 15 stream management issues as "moderately important" or "very important."

Stream Habitat Improvement Session

Paul Balkenbush, Oklahoma Department of Wildlife Conservation
Moderator and Session Chair

Physical Evaluation of Stream Crossings using Applied Fluvial Geomorphology Techniques

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Streams in the Ouachita uplift of southeastern Oklahoma are characterized by bedrock, boulder and cobble substrates, moderate width to depth ratios, relatively steep gradients and narrow valleys. While sensitivity of these channels to anthropogenic impacts is relatively low, the aquatic communities which inhabit them are susceptible to disturbance.

The Ouachita National Forest includes 352,000-acres in Oklahoma. Remaining land ownership in the Ouachita uplift is dominated by the silviculture industry including Weyerhaeuser, John Hancock and Georgia Pacific companies. Significant silviculture activity in the region is facilitated by a large road network. Prescribed limits of one road mile per square land mile are presently exceeded by three orders of magnitude in many areas. Thousands of bridge crossings associated with these roads have been constructed to facilitate timber resource access. Historically, structure durability and economics dictated bridge construction protocols and little consideration was given to fish movement. Many of these structures now act as fish movement barriers either directly by construction design or indirectly by subsequent changes in stream bed morphology.
Literature reviews indicate significant responses in biotic communities to stream crossings. Barriers to fish movement may compromise the ability of fish communities to maintain their integrity (Gagen and Rajput 2002) and increase species extinction risks (Bestgen and Platania 1991). For example, gravel substrates are the preferred spawning habitat of the federally listed leopard darter *Percina pantherina* (Jones et al., 1984). Inability of these fish to access this habitat along the stream continuum may threaten year class success, long term survival and fitness of the species (Schaefer, et al. 2000). Much research has indicated significant impacts to fish communities caused by disruptions of the stream continuum.

The objectives of this study were to: 1) develop regional curves for the bankfull metrics of cross-sectional width, depth and area; 2) assess changes in streambed morphology associated with road crossings; and 3) interpolate impacts of these changes on life cycles of stream fish.

Regional curves facilitate the accurate identification of bankfull stage, defined as the incipient point of flooding (Rosgen 1996), by graphically depicting drainage area plotted versus channel cross-sectional metrics for a given stream location. Historical flood frequency, stage elevation and discharge data were collected for nine southeast

Oklahoma gaging stations and analyzed to create regional curves. Cross-sectional area, mean depth and width were collected at the 1.25-year flood stage at the individual gaging stations. Each of these metrics were then digitally plotted versus known drainage areas for each site. Finally, a regression line was fitted to these plots to establish a curve for each parameter.

Stream geomorphic assessment methods described by Rosgen (1996) were used to assess impacts of road crossings at each study site. Study site selection was based on: 1) schedule for bridge renovation/removal projects; 2) magnitude of fish migration barrier; and 3) stream type and order. A reference reach as described by Rosgen (1996) was used as the study control. A Level I survey was completed to determine steam classification. A Level II assessment was then performed within the reference reach and study site to thoroughly describe channel metrics. The Level II assessment includes a site description, longitudinal profile, cross-sectional surveys, pebble counts and subsequent calculations of dimensionless ratios. All metrics were relative to bankfull flow which was identified using physical characteristics then validated with regional curves. Slope of key channel features, a principal indicator of the stream's morphological behavior, was determined with a longitudinal profile. Monumented cross-sections were then used both upstream and downstream from the crossings to capture significant bed elevation changes. Randomized pebble counts were used to characterize sediment stratification and included 100 samples taken from 10 transects occurring in both riffles and pools.

Entrenchment ratios, width/depth ratios, sinuosity, slope and dominant substrate sizes were calculated from morphometric data. Statistical analyses were used to assess differences in these metrics between sites. Sediment competency (maximum particle size transportable) and capacity (particle amount transportable) were relatively compared between the study and control sites. Velocity estimates derived from collected metrics were correlated with critical swimming speeds to evaluate fish movement potential.

Preliminary results show significant changes in sediment distribution, energy slope, cross-sectional dimension, mesohabitat composition and velocity distribution above and below crossing and compared to control sites. Sediment capacity and competency were severely diminished by the presence of the crossing, leading to headward aggradation and downstream scouring of substrate. A bimodal sediment distribution was also observed at the study site. Finally, the critical swimming speeds of many fish were rapidly exceeded during flood events within the crossing structure.

Results from this study will be used to streamline road crossing removal, improvement and future design protocols. Additional research is planned to increase the scope and magnitude of this study.

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The Influence of Road Crossings on Fish Movement and Fish Communities in Ouachita Mountain Streams, Ouachita National Forest

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Studies of fish passage at road crossings began in the early 1990's on the Ouachita National Forest. The earliest study (Standage et al. 1993) found that a 70-inch (1.8 meter) diameter corrugated metal pipe with a 14-inch (53.6 centimeter) drop to the stream surface was a barrier to green sunfish (*Lepomis cyanellus*), orangebelly darters (*Etheostoma radiosum*), central stonerollers (*Campostoma anomalum*) and longear sunfish (*Lepomis megalotis*) that were found below the structure but not above. Nine weeks of monitoring after culvert reconstruction showed over a hundred darters moved through the baffled pipe and grouted rip rap ramp. By mid-April the stream was drying and no fish other than the orangebelly darter were found upstream of the ramped pipe. Subsequent visits to the site have not detected any fish above the crossing other than orangebelly darters.

Warren and Pardew (1998) examined fish movement at nine crossings including natural-bottomed fords, box culverts, vented low-water fords (low-water crossing with pipe or box culverts), and an unvented low-water ford. The natural ford and box culverts were found to have movements of marked fish through the crossings comparable with or higher than marked fish movements detected for natural reaches. The vented and unvented low-water crossings had reduced or no movements detected through the crossings.

For six low-water vented fords studied by Gagen and Landrum (2000), species richness upstream of the crossings was only half that found downstream (6.3 versus 12.5 below). Recovery of marked fish that had moved across reaches with low-water fords was less than half that of marked fish found to have moved across similar reaches without low-water fords. Marked fish found to have moved were twice as likely to move downstream rather than upstream. Plunge pools below the aprons were eliminated for three of the low-water vented fords in the study by back-filling with riprap. Upstream fish passage was only detected at two of the modified crossings.

Schaefer et al. (2003) studied leopard darter (*Percina pantherina*) movements at a natural ford and a vented low-water crossing with one corrugated pipe and several box culverts in 1999 and 2000. Downstream movement of one marked leopard darter was detected at the low-water crossing on two of eight occasions. Downstream movement at the natural crossing (a shallow riffle) was observed on two of the eight re-surveys of first one and then two marked leopard darters. No upstream movement of marked leopard darters across either crossing was ever detected. Laboratory trials conducted during the same study found that box and round culverts reduce leopard darter movement and wider culverts may allow more movement than narrow ones. A significant finding of this study was that leopard darters seek out thermal refugia in deeper waters when their shallower habitats exceed 29°C. It was concluded that impassable road crossings blocking access to thermal refugia can have a profound effect on population size and habitat suitability.

Gagen and Rajput (2002) examined 21 low-water vented fords and found that mean species richness was 7.1 upstream versus 9.3 downstream. Of movements detected, creek chubs, green sunfish and longear sunfish had the greatest propensity to move (in that order). Total abundance (total number of all individuals of all species) was significantly lower in the combined upstream reaches versus the combined downstream reaches. Darters, with the exception of the more common and endemic orangebelly darters, were consistently missing from upstream communities while present in downstream communities. Bluegill (*Lepomis macrochirus*), northern hogsucker (*Hypentelium* nigricans) and pirate perch (*Aphredoderus sayanus*) were also present in downstream communities, but were absent upstream of low-water crossings. Fewer species were found upstream of the crossings in 67% of the 21 study streams with spring baseflow culvert velocities ranging from 16 cm/s to 85 cm/s. At 60 cm/s and above, only species losses occurred upstream of the crossings (Rajput 2003). Plunge pools were found at five of the 21 stream crossings, all of which were associated with species loss upstream of the low water crossing. Four stream crossings with aprons above the downstream water level were associated with fewer species upstream of the crossing.

Stream crossings, if not designed and functioning properly, have a high probability of being fish passage barriers which impact fish species diversity, population size and fish community structure. Recolonization needs following natural and man-caused perturbations and the need to access thermal refugia must be factors considered in stream crossing construction, reconstruction and maintenance.

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Design and Installation of Two Streambank Stabilization Projects in Northeast Oklahoma

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Sedimentation from streambank erosion is a chief pollutant of streams. Eroding streambanks increase the amount of sediment entering streams which impacts sport fisheries by reducing fish habitat diversity, macroinvertebrate production and subsequent sport fish growth rates. Many technologies using applied fluvial geomorphology concepts are available to streams managers to locally minimize this problem.

The Oklahoma Department of Wildlife Conservation has successfully completed two streambank stabilization projects to demonstrate for landowners the application, design, cost and effectiveness of these structures. The USFWS partners for fish and wildlife grants funded both projects which were completed on Spring Creek in northeast Oklahoma. Spring Creek is a third order gravel dominated stream that supports a quality sport fishery which includes the genetically distinct Neosho smallmouth bass.

The first project used a modified cedar tree revetment to stabilize approximately 107-m of rapidly eroding streambank. The first step in this process was to reshape the current vertical streambank to a 1:1 slope to reduce the shear-stress at the toe of the bank. The second step was to install five large root-wads equally spaced along the toe of the streambank for increased bank protection and fish habitat. The trunks of the root-wads were buried at least 30-ft into the bank to prevent them from washing out. A 0.76-cm biodegradable erosion control mat was then placed over the exposed soil and anchored in place with wooden stakes. Cedar trees were then anchored along the entire lower portions of the streambank extending from the base flow elevation up to the bankfull elevation. Each tree was anchored to the bank facing upstream with stainless steel cable and duckbill anchors. The bottom of each tree was then connected to the top of the tree immediately below it with stainless steel cable. The remainder of the stream bank was then vegetated with native grasses and trees and fenced off.

Unfortunately, Spring Creek experienced three flood events before the vegetation could become well established. During the first two high flow events the structure experienced flows equal to a bankfull discharge and the structure performed excellent but during the third event discharges were commensurate with a five-year flood event and 75% of the structure failed. The project was completed a second time but we increased the thickness of the erosion control matting to 1.52-cm, reshaped the stream bank to a 1:2 ratio, armored the stream bank above the bankfull stage and installed more root-wads (Figure 1). To date the stream has not experienced flows of a magnitude that would cause concern and the vegetation has become well established so we feel confident that it will be able to withstand future high flow events. The total cost of this project was $7,354 which includes $5,000 dollars spent on the original project and an additional $2,354 to repair the structure.

The second project used five J-hook rock vanes to stabilize 180-m of rapidly eroding streambank. Each rock vane was installed pointing upstream at a 20-30^o angle from the streambank with a slope of 2-7^o (Figure 2). The arm of the vane extended to 1/3 of the bankfull width (Rosgen 1998). Footer rocks were placed about 4-ft below the structure on top of bedrock to prevent the stream from undermining the structure and causing it to collapse. These structures were constructed of 81-91-cm diameter rocks that were put in place with a track-hoe. Construction began at the downstream end of the streambank and vane spacing was determined by projecting a 90^0 angle from the tip of one vane back to the upstream streambank. Eight hundred tons of rocks were used to construct the five rock vanes at a cost of $12,400. To date no visible signs of streambank erosion has occurred since the rock vanes were installed.

A shortcoming of this project was that the amount of rock needed to complete the work was underestimated. This precluded use of the J-hook portion of the rock vanes which would have provided additional fish habitat by creating large scour holes below each structure (Rosgen 1996). We also learned since this project was completed that each rock vane would protect three times its length in stream bank (Rosgen, personal communications). Had we known this prior to construction we would have spaced the rock vanes approximately 150-ft apart. This would have reduced the cost by about 50% and the J-hook portions could have been installed.

We anticipate morphological changes in the stream channel after flows increase and the thalweg becomes more defined in the center of the stream channel. This, coupled with added in-stream habitat created by the rock vanes, should benefit the fishery. We are currently cooperating with the Oklahoma Cooperative Fish and Wildlife Research Unit at Oklahoma State University in a project designed to monitor fish community responses to the installation of these structures.

Both of these structures appear to be viable options for landowners who want to reduce property loss caused by streambank erosion or want to improve the in-stream habitat. Each structure has advantages and disadvantages to landowners or streams managers interested in using these methods. The cost, risks, labor and machinery needed to complete the projects must be evaluated by land owners along with their budgets and available resources. For example, the cedar tree revetments cost much less than the J-hook rock vanes to install but they are more likely to fail if a flood event occurs before the vegetation can become well established. The cedar tree revetments require the use of a back-hoe or bulldozer to reshape the stream bank while a track-hoe with an active thumb is needed to install J-hook rock vanes. While the cost of constructing a cedar tree revetment is much lower than that of the J-hook rock vanes the process is much more labor intensive and the rock vanes provide better in-stream habitat.

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Figure 1. Picture of completed cedar tree revetment at Spring Creek in northeast Oklahoma.

Figure 2. Picture of J-hook rock vanes installed at Spring Creek in northeast Oklahoma.

Responses of Fish Populations to the Installation of Rock Vanes in Spring Creek, Oklahoma

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Human activities in watersheds and along riparian corridors can accelerate stream bank erosion. Use of rock vanes in streams is one option for controlling stream bank erosion (Johnson et al. 2001). Rock vanes also enhance fish habitat (Rosgen 1998). In September 2002, the Oklahoma Department of Wildlife Conservation installed five rock vanes in a 180-m section of Spring Creek, Oklahoma to help control stream bank erosion and improve fish habitat quality. The rock vanes were constructed with boulders, and ranged from 13.9 to 20.4-m in length. Each vane was anchored in the left stream bank, extended across the left 1/3 of the channel, and was angled 20 to 30 degrees upstream and 2 to 7 degrees downward into the stream (Rosgen 1998).

We used a before-after, control-impacted design to assess changes in fish habitat and fish populations in response to the installation of rock vanes. The control and project sites were 160 and 180-m long pools, respectively. Fish population, fish assemblage, and habitat data were collected in August and October 2002, and August 2003 at both sites. Centrarchids were also sampled in October 2001 at the project site. Fishes were collected using a Smith-Root 2.5 GPP boat electrofisher, and fish habitat was measured along transects spaced 20-m apart throughout each study site. The same transects were resampled during each sampling period.

Installation of rock vanes changed stream habitat. Substrate distributions did not change at the control site among dates (Fisher exact test; df = 6; $P = 0.313$), but included bedrock, boulders, and more silts at the project site after the rock vanes were installed (df = 10; $P = 0.001$). Abundance of submergent vegetation increased more at the project site when compared to the control site. Water depth heterogeneity among transects did not change at the control (Levene's test; df = 2, 18; $P = 0.971$) or project (df = 2, 24; $P = 0.412$) site among sample dates. Heterogeneity of water velocities also did not change at either site (Control; df = 2, 18; $P = 0.283$; Project; df = 2, 24; $P = 0.540$).

The project also resulted in changes of some fish populations. Relative to the control site, smallmouth bass *Micropterus dolomieu* catch-per-effort (CPE) did not change, but shadow bass *Ambloplites ariommus* CPE responded differently (Figure 1). Smallmouth bass proportional stock density (PSD) continuously decreased at the control site, whereas shadow bass PSD was always zero. Conversely, smallmouth and shadow bass PSDs initially decreased at the project site post installation, but subsequently increased. Smallmouth bass relative weights did not change at the control site among dates (ANOVA; df = 2, 22; $P = 0.092$), whereas shadow bass relative weights did (df = 2, 29; $P = 0.034$; Figure 2). Relative weights of both smallmouth (df = 3, 60; $P = 0.001$) and shadow bass (df = 3, 72; $P = 0.002$) decreased among sample dates after rock vanes were installed (Figure 2).

Fish assemblage stability did not differ between sites. Mean coefficients of variation for CPE at the project site (mean = 64; SD = 28) among sample dates was not different from those at the control site (mean = 57; SD = 35) (t-test; df = 30; $P = 0.499$). Species with CPE trends that differed between sites were banded sculpins *Cottus carolinae*, cardinal shiners *Luxilus cardinalis*, northern hogsuckers *Hypentelium nigricans*, Ozark minnows *Notropis nubilus*, and redspot chubs *Nocomis asper*. A difference in trends suggested that species abundance was affected.

The installation of five rock vanes in Spring Creek resulted in changes in fish habitat, some fish population characteristics, and the abundance of some fish species. Substrate was altered with the addition of boulders used to build rock vanes, which created fish velocity shelters where silt accumulated. An increase in submergent vegetation was observed; a majority of velocity shelters containing silt substrates had high densities of coontail *Ceratophylum demursum*. Conversely, increased habitat diversity, in terms of water depths and velocities, was not observed, probably because a dominant discharge (i.e., bankfull) event has yet to scour streambed materials since the vanes were installed. Nevertheless, changes were observed in some fish populations. Shadow bass abundance appeared to respond negatively at first to the project, but then showed an increase, whereas abundance at the control site decreased. Smallmouth bass abundance did not appear to change at the project site. Changes in fish abundance at the project site could have occurred through a loss in habitat from large woody debris containing rootwads, which were removed to install the vanes, or a gain in boulders and vegetation. Smallmouth bass associate with woody debris and boulders, but shadow bass associate mostly with rootwads (Probst et al. 1984). However, it is unknown whether changes in abundance resulted from differences in survival, or immigration and emigration (Gowan and Fausch 1996). A decline in smallmouth bass PSD at the control site remains unexplained. However, changing smallmouth bass PSDs at the project site may have resulted from a change in habitat use within a species; Probst et al. (1984) found that habitat selection changed with smallmouth bass size. Smallmouth and shadow bass relative weights decreased in response to the project, suggesting some adverse impact on habitat quality. In addition to responses of specific populations, fish assemblage stability between sites was not different. Each assemblage fluctuated moderately, although less so than most assemblages analyzed by Grossman et al. (1990). However, the CPE of some numerically dominant species changed relative to the control, which suggests that the observed habitat change may have affected some species because of their habitat preferences (Gorman 1988).

We anticipate further changes in fish habitat and fish populations. Future large discharge events may change fish habitat further by scouring the streambed below the rock vanes. Additionally, changes we observed in fish population characteristics may not be permanent due to slow response times. Although the primary goal of curtailing stream bank erosion appears to have been met, subsidiary goals of improving fish habitat will continue to be evaluated after further monitoring.

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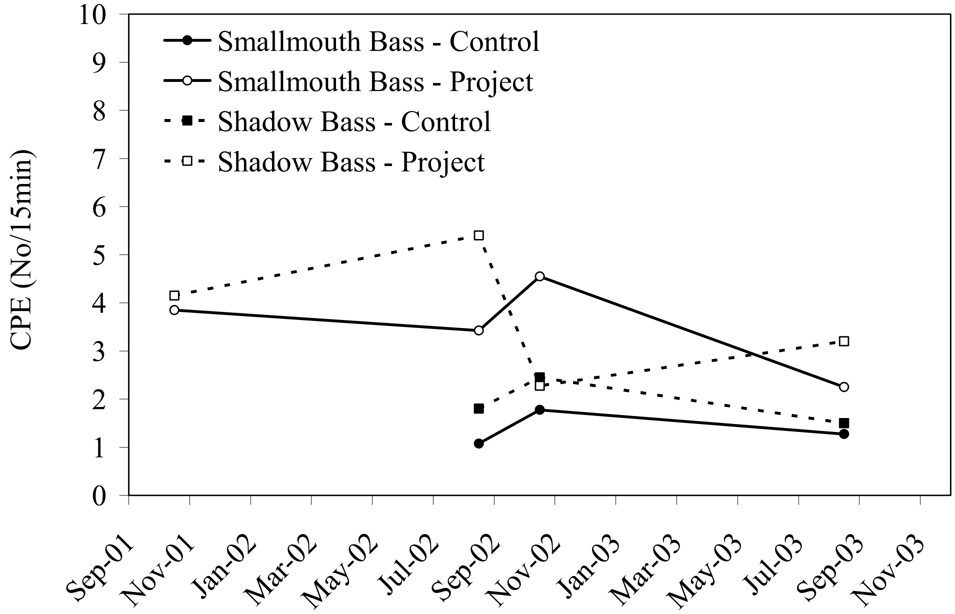

Figure 1. Catch-per-effort (CPE; number per 15 minutes of electrofishing) of shadow and smallmouth bass among sample dates at the control and project sites. Rock vanes were installed in September 2002.

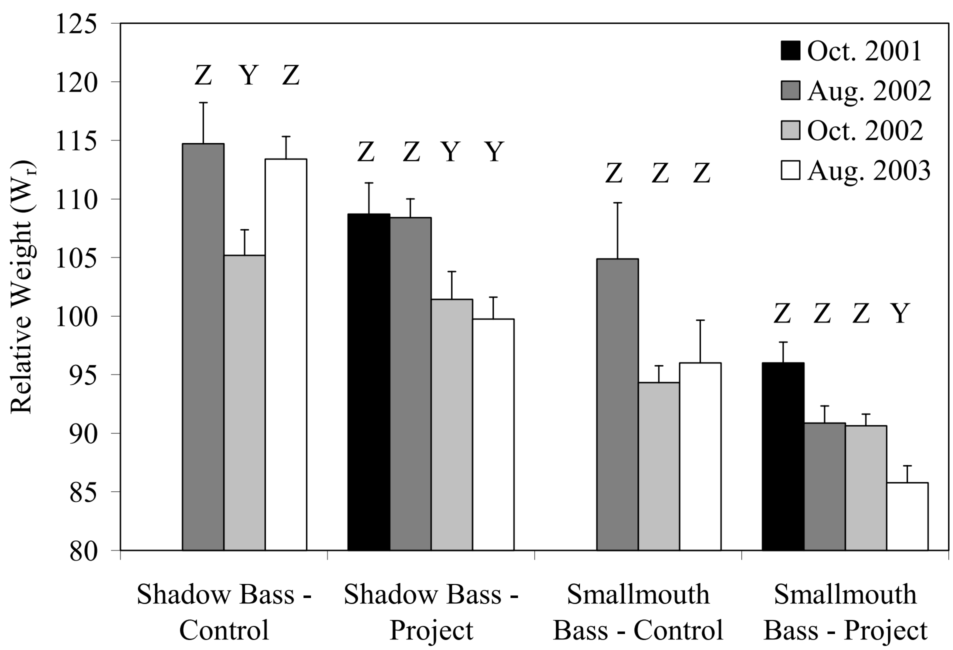

Figure 2. Relative weights (± SE) of shadow and smallmouth bass among sample dates at the control and project site. A multiple contrast was used to detect significant differences ($P \leq 0.05$) between the two dates before rock vanes were installed and each date thereafter for each site and species. Significant differences are indicated by different letters. Rock vanes were installed in September 2002.

Species Restoration and Recovery Successes Session

Christopher S. Thomason, South Carolina Department of Natural Resources
Session Chair

Chris O'Bara, West Virginia Department of Natural Resources
Moderator

Restoring Biodiversity in an East Tennessee Tailwater

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The construction and operation of dams have greatly altered the physical and chemical properties, and the composition of aquatic fauna in tailwaters. Without dam removal, some of these changes are irrevocable; however, changing the quality and pattern of discharge may allow restoration of much of the original fauna. Douglas Dam, located on the French Broad River in Tennessee is operated primarily for flood control and hydroelectric peaking power. The historic fauna of the lower French Broad River is poorly known. Based on information in Etnier and Starnes (1993), Parmalee and Bogan (1998), data from an aboriginal site (unpublished data, P. Parmalee), unpublished reports, a collection of relic shells from a muskrat midden (unpublished data, J.R. Shute and P. Rakes), and our own collection records, we believe that the historic fauna likely consisted of about 98 species of fish and 54 species of mussels. Historically, there was no minimum flow release from Douglas Dam, and during periods of lake stratification, water released from the hypolimnion contained low dissolved oxygen (DO) concentrations. By 1977, the original fauna was reduced to 48 species of fish and 12 species of mussels.

In 1987, the Tennessee Valley Authority (TVA) began testing methods to increase DO in discharges from 16 hydroelectric projects in the upper Tennessee River system. A combination of three aeration methods was developed at Douglas Dam, and the ability to restore consistent DO levels to 4 mg/L was achieved in 1993. A minimum flow release was also initiated in 1987. Biological monitoring of warmwater fish communities in the French Broad River below Douglas Dam showed dramatic improvements at three fish sampling stations by 1996. Recognizing these positive biological responses, the US Fish and Wildlife Services (FWS), US Geological Survey (USGS), Tennessee Technological University (TTU), Tennessee Wildlife Resources Agency, Tennessee Aquarium, Southeastern Aquatic Research Institute, World Wildlife Fund, and Conservation Fisheries, Inc., and TVA teamed up under the leadership of R. Biggins (FWS) to restore the lost biological diversity. Source populations for many (but not all) of the fishes extirpated and their connectivity with the Douglas Dam tailwater exist. In contrast to the fish fauna, most mollusk species extirpated have little opportunity to recolonize the tailwater. Therefore, about 20,000 adult freshwater mussels of 18 species were collected from the lower Tennessee River and translocated to the tailwater. Additionally, several hundred spiny riversnails (*Io fluvialis*), and >20,000 lake sturgeon (*Acipenser fulvescens*) have been reintroduced.

Since 1994, we collected 78 native fish species (80 % of the historic fauna), and 9 introduced species in the tailwater (Table 1). Following improvements in discharge, many relatively intolerant species including lampreys, catostomids, and percids colonized the tailwater. Many of the cyprinids historically occurring in the French Broad River were also collected; however, only a few individuals represented several of these species.

Species that have not recolonized the tailwater include mainly those that have been extirpated from the entire Tennessee River system, are isolated by other dams or inhabit the mid-water column. We did collect some mid-water cyprinid species; however, they were uncommon. Although the minimum flow release provides a high diversity of habitat, few velocity refuges exist during full generation. The lack of low-velocity habitat is likely limiting the recolonization of mid-water column species.

Radio tracking and monitoring indicated stocked lake sturgeon have dispersed throughout the upper Tennessee River system. Growth of recaptured fish has been good. Because of their age at maturity (>10 yrs), it will be many years before we can determine whether a self-reproducing population of lake sturgeon has been established.

At each of eight permanently marked plots, we determined mussel survival at 1-year intervals following translocation with a stratified random sampling design. Qualitative searches were also conducted to determine the presence of extant species. One year after translocation, estimated densities of live mussels varied from 52 to 108% of the densities translocated within each plot. The decrease in density between years was due to a combination of mortality and movement. During a qualitative search immediately downstream of 2 plots, we found 76 live and 62 dead translocated mussels. Retention of translocated mussels within a plot was positively correlated with the density of nontranslocated mussels ($r= 0.84$; $p < 0.05$). The size and eroded condition of most extant species found suggest they were living when Douglas Dam was constructed; however, recent recruitment was evident for five species.

Results of translocating mussels indicate that many extirpated endangered species can be reestablished in the lower French Broad River; however, continued operation of the Douglas Dam as a peaking hydroelectric facility will constrain the number of species. Of the 54 mussel species historically occurring in the lower French Broad River, 5 are now extinct, and we believe it is unlikely that 16 species can be reestablished because of the unavailability of a donor population, or the absence of their hosts. The larvae (glochidia) of all extirpated mussels are obligate parasites on fish. Moreover, many mussel species are host specific and can metamorphose on only one or a few species of fish. Although 80% of the historic fish fauna now occurs in the Douglas Dam tailwater and additional species may colonize it, the general lack of mid-water column species will prevent the restoration of some mussel species. Furthermore, the lack of flow refuges during full generation makes it unlikely that these mid-water column species can be reestablished. Despite these constraints, we believe that reproducing populations of at least 31 species, including 7 endangered, can be reestablished in the Douglas Dam tailwater. These federally listed species may soon be transplanted to this river stretch, pending their designation by the FWS as experimental, non-essential populations.

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Table 1. Numbers of species per family for the historic and recent fish faunas in the Douglas Dam tailwater.

	Number of Species		
Family	Historic	1977	>1987
Petromyzontidae	3	0	3
Acipenseridae	2	0	1[a]
Polyondontidae	1	0	0
Lepisosteidae	2	2	2
Clupeidae	3	2	3
Hiodontidae	1	1	1
Cyprinidae	24	12[b]	23[c]
Catostomidae	16	6	13
Ictaluridae	8	3	6
Esocidae	1	0	0
Salmonidae	0	0	2[d]
Fundulidae	2	1	2
Poeciliidae[d]	1	1	1
Atherinidae	1	1	1
Cottidae	1	1	1
Moronidae	2	1	3[b]
Centrarchidae	11	7[b]	12[b]
Percidae	18	6	15[b]
Sciaenidae	1	1	1

[a]Recaptures of stocked lake sturgeon.
[b]Includes one introduced species.
[c]Includes four introduced species.
[d]Introduced

A Survey of Pigeon River Re-introduction Efforts

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For most of the twentieth century, the Pigeon River has suffered environmental degradation in the form of industrial and municipal waste, and hydrological alteration. The primary polluter has been a paper mill that began operations in 1908 located in Canton, NC. This Appalachian stream was once so polluted that North Carolina classified the best use of her waters to be for waste disposal (Messer 1964).

Historically, it is estimated that 40 species of native mussels and 95 species of native fishes thrived in this river (Etnier and Saylor 2001). In the last 12-15 years, water quality has steadily improved as the quality of the mill effluent increased, and has led to the return of many native fish species. However, not all aquatic species can return because surviving populations are isolated from or within the Pigeon River drainage. The improved health of the river has led state, federal, and private agencies to re-introduce several native aquatic species into the river.

As of 2003, eight species of fish, six genera of snails and eight genera of mussels have been re-introduced into the Pigeon River (Table 1). The first re-introductions were common snails (*Leptoxis sp*., *Pleurocera sp*., *Eliminia* sp.) in 1996 by the U.S. Geological Survey (USGS) and Tennessee Wildlife Resources Agency (TWRA). Snails were re-introduced to determine their chances of survival, which if successful, would set the stage for future re-introductions of mussels and fish (S. Ahlstedt, USGS, personal communication).The survival of these snails led to additional re-introductions in 1999 (*Io* sp., *Campeloma sp*., *Lithasia sp*.), and the transplanting of nine species of common mussels (*Alasmidonta marginata, Amblema plicata, Cyclonaias tuberculata, Elliptio dilatata, Lampsilis fasciola, L. ovata/cardium, Ptychobranchus fasciolaris, Quadrula pustulosa*, *Strophitus undulates)* in 2000.

Subsequent recruitment of *Leptoxis* (an indicator species) and *Pleurocera* encouraged the formation of the Pigeon River Recovery Project in 2001 with the goal of re-introducing native fish into the river. The initial candidates were small, non-game fish collected from area streams with similar habitat within the French Broad

River drainage. These fish were typically collected with seine crews; on occasion, electroshocking was used. The first natives were the blueside darter (*Etheostoma jessiae*), bluebreast darter (*E. camurum*), and gilt darter (*Percina evides*).

The primary release site was at Tannery Island, characterized as riffle habitat with predominately cobble, boulder, and some sand substrate.

To assess survival of relocated species, visible implant fluorescent elastomer (VIE) tags were employed. VIE is a bio-compatible, medical grade silicone that, when injected as a liquid, cures to a pliable solid. The tag was placed just under the skin and above the muscle layer at the base of the dorsal fin. The five VIE color variations used in each batch tagging effort represented the river source, the season (spring or fall), and the year of collection. Tag colors were easily observed in typical daylight conditions; in reduced light conditions, a blue LED light enhanced tag visibility.

Direct underwater observations have proven to be the most efficient method for the collection of life history data, microhabitat use, and census collection of benthic non-schooling species (Dinkins and Shute 1996). During preliminary surveys at the Tannery Island release site, it was determined that the identity and tag of a target species could not be confirmed at a distance greater than 60 cm (2 feet) in full sunlight and less than 45 cm (1.5 feet) in shade. Therefore, in the subsequent quantitative snorkel survey, transect lanes were 1.2 cm in width. Surveys were conducted downstream, do to the flighty response of gilt darters when approached from behind (upstream). Each snorkeler was tethered with a rope around one ankle and was guided by the tether person (Figure 2). The rope was attached to a reel, which was operated by the tether person. The tetherer's job was to keep each snorkeler in a straight line with the others. Snorkelers communicated with hand signals. The snorkeler kept his/her body centered over a lead-core rope, which marked the center of each lane or transect. Fish observed along transect lane boundaries or fish passing between transects were noted by adjacent snorkelers to reduce duplicate counts. When one of the target species was sighted, the line of snorkelers was halted and the species and color of the VIE tag (or no tag) was recorded on the PVC slate of the individual observer.

Snorkeling surveys during the summer of 2002 revealed healthy gilt and bluebreast darters, but no blueside darters; the occasional observation of untagged gilts suggested reproduction. Subsequent re-introductions have included stargazing minnows (*Phenacobius uranops*), mountain madtoms (*Noturus eleutherus*), stripetail darters (*E. kennicotti*), American brook lampreys (*Lampetra appendix*), and mountain brook lampreys (*Ichthyomyzon greelyi*).

Snorkeling surveys conducted during the summer and fall of 2003 made random transects of 21 sites covering a 22.4 km (13.5mile) reach of the Pigeon. No bluebreast or blueside darters were observed; above average precipitation resulted in high water levels which may have moved these darters outside of the survey area. Interspecific competition may have been a factor as well. Gilt darters were located at seven of these sites: the re-introduction site, one site directly upstream and the five sites directly downstream of the re-introduction site. A total of 102 gilts were observed, including 18 tagged and 84 untagged juveniles and young-of the-year; the presence of untagged fish indicated successful reproduction. Three genera of river snails (*Io*, *Leptoxi*s, *Pleurocera*) were located at five sites. Significant reproduction in *Leptoxis* and *Pleurocera* has been documented, and *Io* has been observed laying egg masses for the past three years.

Heavy rains and flooding conditions this year may be a factor in the apparent movement of both fish and snails. Snorkel surveys provided an efficient, non-invasive sampling method to monitor re-introduced species. Accuracy and precision of visual census may vary according to riverine topography, time of the day or season, and the observer skill level. Re-introductions of all species in the Tennessee reach will continue, and re-introductions will begin in the North Carolina reach in 2004.

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Table 1. Pigeon River re-introductions, 1996-2003.

Year	Organism	Genera/Species	#Individuals
1996-2003	Snails	6	60k-80k
2000-2003	Mussels	9	145
2001-2003	Darters	4 sp.	2608
2002-2003	Madtom	1 sp.	381
2002-2003	Minnow	1 sp.	270
2003	Lampreys	2 sp.	716

Figure 1. Snorkel survey in progress.

Propagation and Culture of Endangered Juvenile Mussels (Unionidae) at the Freshwater Mollusk Conservation Center, Virginia Tech

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Although North America contains the greatest diversity of freshwater mussels in the world, roughly 300 species, this family of mollusks is the most imperiled taxon in the United States. Already, 35 species are presumed extinct and 70 species are listed as endangered or threatened. Without immediate efforts to recover federally protected species in watersheds throughout the country, the extinction of additional species is likely. The Tennessee Wildlife Resources Agency, U.S. Fish and Wildlife Service, Virginia Department of Game and Inland Fisheries, and Virginia Tech have entered into a cooperative program to fund the production, culture, and release of large numbers of endangered juvenile mussels into rivers in Tennessee and Virginia. The goal of this project is to augment natural reproduction at sites with these species and to release juvenile mussels at historic sites within those rivers to expand population ranges.

Biologists at the Freshwater Mollusk Conservation Center have developed methods to produce and culture juvenile mussels to help recover these populations. Freshwater mussels have a unique life history, requiring the use of particular species of fish in the life cycle. The process of producing juvenile mussels begins by collecting suitable host fish from the river and holding them in captivity until gravid female mussels can be found. In the laboratory, the larvae (glochidia) in the gills of the female mussel are flushed out using a hypodermic needle filled with water. This non-lethal method allows us to return females to the river once her progeny have been removed. We have collected and transported female mussels of various species to our laboratory, removed their glochidia, and released them back to the site of capture. The following year we have then recaptured some of these female mussels and found them gravid. The larvae can number more than 200,000 per female. These larvae are then introduced into a bucket holding the host fish, and aeration is used to keep the water agitated to allow larvae to attach to the gills of the fish. After 1 hour of exposure, the fish are moved to large aquaria where the attached larvae begin the transformation process, which requires 2-3 weeks. Aquarium systems are adapted to the flow and cover requirements of the specific host fishes. Glochidia are transformed at cool temperatures (19-22°), which increases survival of host fish and allows glochidia to transform unharmed to the juvenile stage. Once these young juveniles drop from their host fish, they are collected by siphoning the tank bottoms. Newly metamorphosed juveniles are held in small containers with cultured algae and sediments for 1-2 weeks before release to the wild, or cultured long-term in recirculating aquaculture stream systems.

Long-term (2-6 months) culture of juvenile mussels in recirculating aquaculture troughs is a feasible, cost-effective method to produce juveniles for population augmentation of endangered species, toxicity testing, or other research needs. The process begins by placing newly metamorphosed juveniles in individual containers in the raceway of the recirculating aquaculture trough. The juveniles are cultured in dishes containing fine sediments. The culture unit is a 3 m long, 225 L plastic livestock feed trough. A 50:50 mixture of conditioned (dechlorinated) municipal water and well water is used in the culture system, with hardness ranging from 250 to 350 mg/L $CaCO_3$. A 50 L square, plastic container serves as a sump reservoir, and PVC piping is used for delivery and return lines. Water is pumped through the raceway using a centrifuge or magnetic drive pump, and gravity-fed back to the sump reservoir through a standpipe. The juveniles are fed small (5-10 μm) green algae, e.g., *Neochloris oleoabundans* or *Nannochloropsis oculata*, at a daily concentration of 20,000-30,000 cells/mL. For the best results, juveniles are cultured at temperatures ranging from 21-24°C. Sustained temperatures > 27°C seem to be detrimental to survival and growth of young juveniles in our recirculating aquaculture systems. Generally, survival of juvenile mussels is influenced by seasonal viability of newly metamorphosed juveniles, species differences, substrate composition, water quality, and predators. For example, the common rainbow mussel *Villosa iris* is much easier to culture than the endangered oyster mussel *Epioblasma capsaeformis* ($p<0.05$); additionally, the survival rate of transformed juveniles of both species is greater in the spring ($p<0.05$). Long-term (60-90 d) survival of endangered juveniles has ranged from 0-50%; however, techniques are now greatly improved and survival is expected to increase.

Between 1998 and 2003, nearly 400,000 juvenile mussels of 9 endangered mussel species were released into the Big South Fork Cumberland, Clinch, Powell and Hiwassee rivers. These juveniles are typically between 700-1200 μm long at the time of their release into the wild. Monitoring efforts at release sites have documented variable survival of juveniles. For example, survival of released juveniles of *E. capsaeformis* in the Clinch River has been documented, and augmentation efforts in the river appear successful. In contrast, released juveniles of the same species in the Powell River have shown no signs of survival. Propagation is now a viable tool to implement recovery of all federally listed mussel species.

Candidate Conservation Agreement with Assurances for the Robust Redhorse, *Moxostoma robustum*, Ocmulgee River, Georgia

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The robust redhorse *Moxostoma robustum* is a large, rare sucker that was originally described from the Yadkin River, North Carolina in 1869 by Edward Cope. Few specimens were collected and the species status was uncertain until 1991 when a single population of robust redhorse was discovered by Georgia Department of Natural Resources (GADNR) biologists along a 70-mile reach of the Oconee River in central Georgia, downstream of Georgia Power's Sinclair Dam. The robust redhorse is currently listed as endangered by the State of Georgia and is a species of management concern to the U. S. Fish and Wildlife Service (USFWS). This original population is believed to consist of 600 individuals ± 180 (1 SE) in a defined reach of the Oconee River (Jennings et al, 2000).

The Robust Redhorse Conservation Committee (RRCC) is implementing management efforts for the robust redhorse. The RRCC is a cooperative effort established in 1995 through a Memorandum of Understanding among State and Federal agencies, private interests, research scientists, industry, and conservation organizations. The RRCC works to determine the status of known robust redhorse populations, establish additional populations, and implement necessary research and other actions to maintain or enhance the survival of this species within its historic range.

The RRCC developed a Conservation Strategy for the robust redhorse that includes short- and long-term goals for the conservation and recovery of the species (RRCC, 2003). The short-term goals of the Conservation Strategy include:

- Establishing refugial populations to reduce the impact of potential catastrophic events on the species' survival;
- Determining habitat and life history requirements of the robust redhorse; and
- Establishing reintroduction plans or agreements to facilitate conservation actions for specific sites.

The long-term goal of the Conservation Strategy is to establish or maintain at least six self-sustaining populations of robust redhorse distributed throughout the historic range.

One of the proposed reintroduction sites was a reach of the Ocmulgee River below another Georgia Power facility, Lloyd Shoals Dam. This area is physically similar to the Oconee River and is in an adjacent watershed (Figure 1). This proposal by university researchers and natural resource agencies raised a number of concerns regarding the consequences of such actions to Georgia Power if Georgia DNR introduced a species, which may be a candidate for future listing under the ESA, below a federally licensed project.

Numerous discussions about the basis for this recommendation and possible ways to address Georgia Power's concerns regarding introduction of a non-federally listed species led to the development of a Candidate Conservation Agreement with Assurances (CCAA). The CCAA policy is designed to encourage landowners, including those who operate facilities that may affect adjacent habitat, to take conservation actions to enhance the survival of rare and potentially imperiled species. The landowner is provided assurances under the CCAA policy by the USFWS that additional conservation actions will not be required nor will additional land, water or resource restrictions be imposed beyond those in the agreement should the species be listed under the ESA at some point in the future. The assurance in this specific situation precludes the USFWS from requiring alteration of the operation of Lloyd Shoals Dam for the benefit of the robust redhorse, including alteration of the flow regime specified in the FERC license, for the remaining period of the license.

In order to accomplish the objectives of this CCAA, the parties agreed to undertake the following conservation actions.

1. Stock the Project Site

As a result of the CCAA, Georgia DNR will stock the project site with approximately 4,000 hatchery reared robust redhorse fingerlings each year, for five years (not necessarily consecutive), subject to availability. The goal of this action is to establish a refugial population from the Oconee River parental stock that consists of a minimum of five year classes. As of the end of 2002, an estimated 6,692 individuals have been released into the Ocmulgee River from six year classes spanning 1997 through 2002.

2. Study the movement of introduced juvenile robust redhorse

Georgia Power is responsible for studies of the movement of introduced juvenile robust redhorse using radio or sonic transmitters attached to a subset of the stocked fish. The first set of studies used both technologies and found that approximately 60% of the telemetered fish stayed within the 19 mile reach of the project site.

3. Monitor abundance and distribution of introduced robust redhorse

Georgia Power will monitor abundance and distribution of introduced robust redhorse through periodic surveys. Initial electrofishing surveys collected six individuals from the introduced population distributed from Lloyd Shoals downstream to Warner Robbins, Georgia.

4. Estimate population size
Georgia Power will fund an assessment of population size based on sampling at intervals not to exceed three years

This CCAA represents a significant and important milestone in the cooperative conservation efforts for the species. It is a logical extension of the cooperative conservation efforts by the RRCC and is consistent with section 2(a)(5) of the ESA, which encourages creative partnerships among public, private, and government sectors to conserve imperiled species and their habitats. It recognizes that no single participant has both the authority and resources to fully implement all actions necessary for protecting this species. Finally, this agreement is a positive step by natural resource agencies in addressing the risks and concerns that naturally result when landowners contemplate conservation actions which encourage potentially imperiled species to occupy or increase on property they are responsible for managing.

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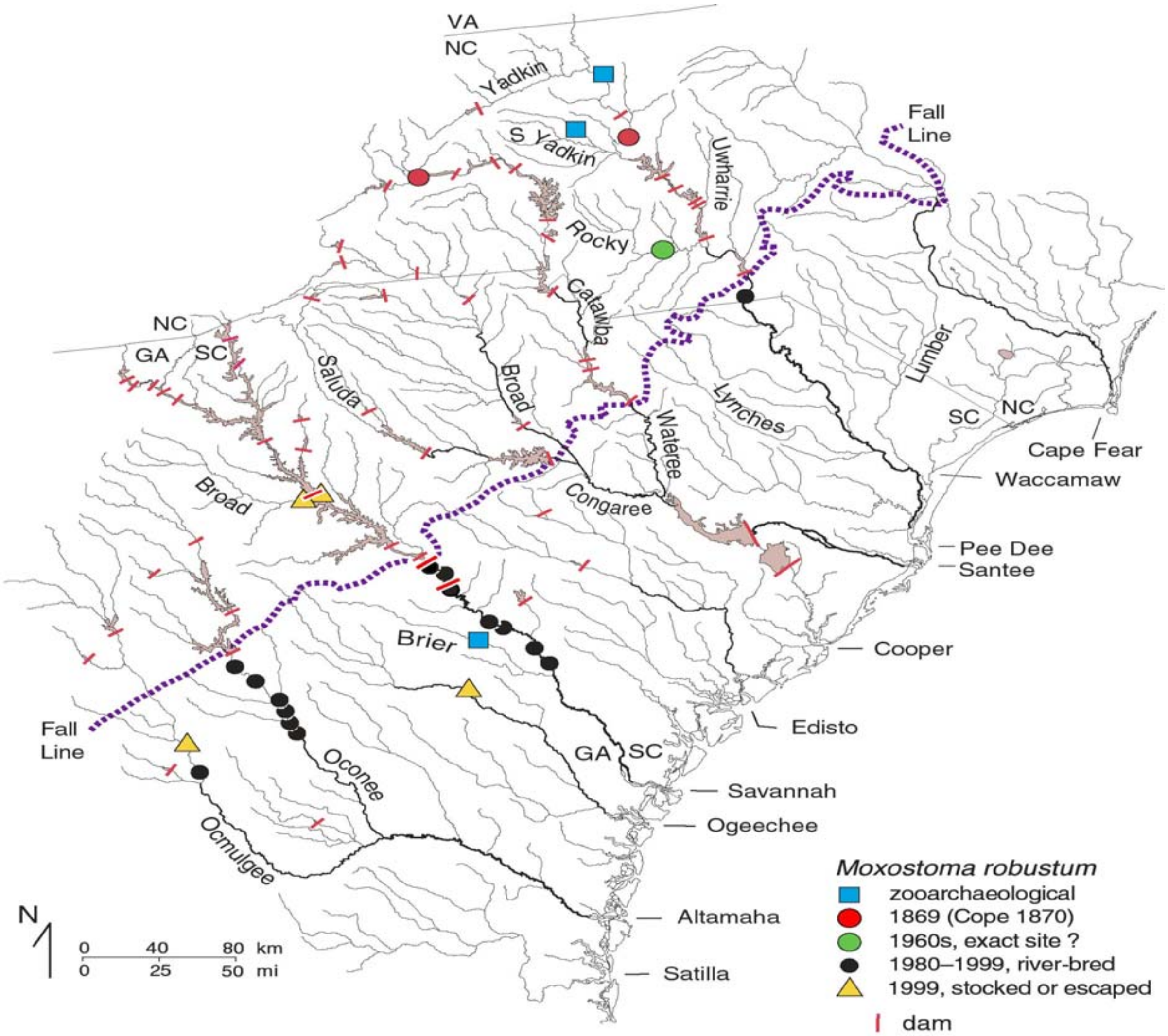
VA
NC
Yadkin
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Uwharrie
Fall Line
Rocky
Catawba
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GA
SC
Saluda
Broad
Lynches
Wateree
Lumber
SC
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Cape Fear
Waccamaw
Broad
Congaree
Pee Dee
Santee
Brier
Cooper
Fall Line
Edisto
GA
SC
Oconee
Savannah
Ogeechee
Ocmulgee
Moxostoma robustum
zooarchaeological
1869 (Cope 1870)
1960s, exact site ?
1980–1999, river-bred
1999, stocked or escaped
dam
Altamaha
Satilla
N
0 40 80 km
0 25 50 mi

Ohio River Paddlefish: Management Strategies for a Multi-jurisdictional Species

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Paddlefish are native to the Ohio River sub-basin of the Mississippi River drainage. The Ohio River sub-basin represents a linear distance of 20,856 km or 22% of the endemic paddlefish range. Historically, habitat degradation, exploitation by sport and commercial fishers, and illegal harvest significantly impacted paddlefish population throughout their range. Recently, increased commercial exploitation of paddlefish, because eggs are a high-grade substitute for sturgeon caviar, has placed an additional stressor on paddlefish populations.

Paddlefish within the Ohio River sub-basin migrate among multi-jurisdictional boundaries. Consequently, to effectively manage this great river species, a coordinated strategy must be employed. The Ohio River Fisheries Management Team (ORFMT) provided a framework for such an approach. Five state agencies actively participate in ORFMT tasks with two additional states contributing when appropriate. In 2001, the ORFMT Technical Group developed an Ohio River Sub-basin Paddlefish Strategic Plan that clearly outlined both policies and tasks. The plan recognizes the multi-jurisdictional framework of managing great river species, and considers individual state's management strategies. Policies on genetic concerns, commercial and sport harvest regulations, and data sharing are addressed and implemented in this strategic plan. Coordinated field tasks related to tagging and movement, standardized population characterizations, and stocking are also addressed and are currently ongoing. Through this approach, the management of paddlefish within the Ohio River sub-basin is truly multi-jurisdictional, providing for a more realistic strategy for this great river species.