STATEWIDE RESEARCH – FRESHWATER FISHERIES

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Job Title: An evaluation of multiple families of striped bass stocked in Lake Wateree in 2008

Period Covered July 1, 2010 – June 30, 2011

Summary

In May and June of 2008 striped bass fingerlings from 6 genetic families were stocked in Lake Wateree. To assess recruitment by family to age 1+, fish were collected during the Winter and Spring of 2009-2010. All fish collected from the 2008 year class were identified to family using microsatellite data previously generated from parents of the 2008 year class. Contribution by family was calculated. Because families were stocked on two different dates, contributions were evaluated by family for each stocking date using the G test. Families returned at rates different than those expected based on stocking rates. One family that comprised 14.2% of total fingerlings stocked in 2008 accounted for 54.2% of returns at age 1+. Conversely, two families combined provided 75.9% of stocked fingerlings, but accounted for 29.2% of total recaptures at age 1+. Family differences in returns were still evident when evaluated separately by stock date. Two families stocked June 13 returned in similar proportions to those at stocking, but four families stocked May 29 returned in proportions significantly different to those expected. Three May 29 families returned at rates lower than those at which they were stocked, while one family's return rate of 76.5% represents a 27% increase over its stock rate. These results confirm the need to consider familial differences in stocking evaluations that seek to compare time of stocking, stocking location etc. Data is currently being generated for condition of fish from each of these families at stocking, and will be included in further evaluation.

Introduction

Multiple factors in the production and stocking of hatchery reared striped bass can contribute to a batch's potential for survival and eventual recruitment to a fishery. The need exist for a better understanding of how, and which, factors contribute significantly to the ultimate success of stocked fish. Ideally study designs will allow for a homogenized gene pool across treatments. The development of microsatellite markers for striped bass provides an excellent tool in that it allows the evaluation of multiple treatment batches of fish. Elimination of genetic effects on treatment groups is not possible however when treatments are identified by their genetic mark. Wang et al. (2006) found that dam and sire effects on juvenile growth and growth rate were significant in hybrid striped bass (*M. chrysops* female x *M. saxatilis* male). Results for measurement at two time intervals also suggested that selection for growth rate at an early life stage could affect growth rate at a later life stage. Thus, genetic effects on growth, and on other aspects of performance, are important to consider when evaluating effects such as time or location of stocking. In 2008, striped bass from 6 genetic families were stocked in Lake Wateree, with plans to assess recruitment by family to age 1+. In the last year work has focused on identification of collected age 1+ fish to family, and evaluation of returns by family.

Materials and Methods

Finclip tissues from selected fish collected in 2009/2010 were evaluated at Marine Resources Research Institute using 12 microsatellite markers (Fountain et al. 2009). All fish were identified to year class, and then to parental cross and family based on striped bass broodstock evaluations. Because fish from two families, X and Y, were grown out as fingerlings in the same pond and their individual stocking rates are not known, family XY stocking and return numbers were combined and evaluated as one family. Deviations in actual rates of return from expected rates were evaluated by family for each of two distinct stock dates, using the G test.

Results

Of N=210 fin clip tissue samples evaluated, all were of hatchery origin and were identified to their broodstock parents. N=168 were from the 2008 year class. All other fish were from the 2007 (N=34) and 2009 (N=2) year classes. These year classes are not being followed as part of this evaluation and are not included in any further analysis.

Fish were collected from each of 6 genetic families stocked in Lake Wateree in 2008. Catch rates varied from stocking rates. Family A comprised 14.2% of total fingerlings stocked in 2008, but accounted for 54.2% of returns at age 1+. Conversely, families XY and Z combined provided 75.9% of stocked fingerlings, but accounted for just 29.2% of total recaptures at age 1+. When statistically evaluated by date, catch rates were similar to stocking rates for two families stocked on June 13. Four families stocked May 29 returned in proportions different to those expected. Three May 29 families returned at rates lower than those at which they were stocked, while one family's return rate of 76.5% represents a 27% increase over its stock rate (Table 1.).

Table 1. Stock and return data for striped bass fingerlings stocked in Lake Wateree in 2008. Data is presented by stock date and genetic family. G test statistics and P-values are presented by stock date, and evaluate the difference in actual and expected (based on stocking proportions) rates of return.

Stock Date	Family	N Stocked	N Returned	Stock Proportion	Return Proportion	G	P-value
5/29/2008	A	38,517	91	60.21	76.47	17.369	0.0006
	B	17,108	23	26.74	19.33		
	C	1,015	1	1.59	0.84		
	D	7,332	4	11.46	3.36		
6/13/2008	XY	71,312	21	36.50	42.86	0.591	0.442
	Z	124,064	28	63.50	57.14		

Discussion

Genetic marks have become an important tool in the evaluation of stocking strategies for striped bass. They enable us to evaluate returns based on a wide range of factors. These may include but are not limited to timing of stocking, stocking location or zone, stocking method, and source of fingerlings. An important factor to consider in the use of genetic marks is that they preclude the homogenization of genetic families across treatment groups prior to stocking. This introduces the possibility of a family effect inherent to those treatments we wish to evaluate.

A family effect on return data is evidenced in returns of striped bass stocked in Lake Wateree in 2008. For four families stocked May 29, rates of return at age 1+ differed considerably from expected rates. From the point of fingerling harvest to stocking these families were all treated equally, including being spread equally among the hauling compartments on the transport truck used for stocking. So, any hauling or stocking stress on these fingerlings would be spread evenly across all four families. The increase in return rate of family A over stocking rate is highly significant and underscores the importance of design in these types of experiments. When experimental design allows it, families should be genetically homogenized across stocking treatments to eliminate any effects inherent to family. When design specifics preclude mixing of families across treatments, investigators should do what they can to maximize the number of families within each treatment. Statistical evaluation of data should include individual family as a variable, along with any other treatment variables being evaluated.

Recommendations

Based on current evaluation, ensure that any study design that incorporates genetic marks considers family as a recruitment variable in data analysis. Complete data collection for condition at stocking of each family stocked in Lake Wateree in 2008. Repeat data evaluation in consult with statistician, including condition, family, stock date, method and time of capture as variables. Prepare final report.

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Wang Xiaoxue, Kirstin E. Ross, Eric Saillant, Delbert M. Gatlin III, John R. Gold. 2006. Quantitative genetics and heritability of growth-related traits in hybrid striped bass (*Morone chrysops x Morone saxatilis*). Aquaculture 261: 535-545

Prepared By: Jean Leitner Title: Biologist

Job Title: Assessment of condition, growth, contribution to fish community, and diet of striped bass, white perch, and American shad young-of-the-year in the Santee-Cooper lakes, South Carolina

Period Covered July 1, 2010 – June 30, 2011

Summary

Boat electrofishing was conducted each month at two Lake Marion sites during summer of FY11 and FY12 to evaluate relative abundance, growth, and condition of key juvenile fish species. During summer 2011 white perch and American shad were the most abundant species accounting for 42% and 27% of all young-of-year fish collected. Gizzard shad juveniles, which have been very rare in previous years, were common representing 17% of all fish collected. Striped bass relative abundance decreased each study year from a high of 20% in 2009 to only 6% during 2011. Catch rates of striped bass during 2011 were also significantly lower than those observed during 2009. Growth of American shad, threadfin shad and white perch during 2011 was similar to the growth observed during 2010 but slower than that observed during 2009. Neither clupeid species grew much, if at all, between August and October 2011. Mean TL of American shad was highly variable throughout the summer. Striped bass growth during 2011, although highly variable, appeared to be similar to that observed during 2009 and 2010. Condition (Kn) of American shad and striped bass varied each with the 2011 values intermediate of those observed during 2009 and 2010. Twenty-six different taxa were identified in the diet of young-of-year white perch and striped bass. The most frequently encountered items were microc-rustaceans (e.g., cladocerans, copepods and ostracods) and larval insects.

Introduction

'Fingerling mortality' of striped bass is a key issue for the Santee-Cooper striped bass stakeholders and it has been a key issue of the DNR for many years. Many hypotheses have been generated to define the causes of either good or poor recruitment in a given year. These hypotheses include, but are not limited to, reduction in the adult spawning stock, competition with resident and anadromous species, and reduced nutrient inflow due to drought. The Santee-Cooper Comprehensive study group of the DNR defined investigation of the 'competition for resources' hypotheses as its primary short-term goal. A strategy was needed to obtain key monitoring data on the species of interest. The objectives of this study are to, 1) Define growth and condition of key juvenile species, 2) describe the diet of each species and 3) define the relative abundance of each key species.

Materials and Methods

Growth, condition and relative abundance

Young-of-year (YOY) American shad, blueback herring, threadfin shad, white perch and striped bass were collected monthly from two Lake Marion sites with boat electrofishing gear. At each site night-time electrofishing was conducted for roughly 10 minutes at each of three transects. We attempted to collect all young-of-year (YOY) of the targeted species. Specimens were preserved on ice and measured (TL, mm) and weighed (mg) within 24 hours of collection.

Diet

Up to 15 of each key species per site were preserved in 10% formalin on every sample date during 2009. The stomach contents of the preserved striped bass and white perch specimens were examined under a dissecting microscope and identified the lowest practical taxon. Frequency of occurrence was calculated as the proportion of fish stomachs that contained one or more individuals of a given food type.

Results

Growth, condition and relative abundance

Young-of-year morones and clupeids were collected at night from two Lake Marion sites with boat electrofishing during June – September 2011. The “Big Water” site was located near I-95 on the Clarendon County side (34.5178, -80.4349) and the “Indian Bluff” site was located midway down the reservoir on the Orangeburg County side (33.4319, -80.3621). Three transects were sampled at each site on 5 different dates. Each site received approximately 0.5 h of electrofishing effort on each sample date. During 2011 thirty transects were sampled with a total electrofishing effort of 5.1 h (Table 1).

Table 1. Number of transects sampled on each date and electrofishing effort (h) during nighttime electrofishing at two sites on Lake Marion, SC during 2011.

	Big Water		Indian Bluff		Total	
Date	Transects	Effort (h)	Transects	Effort (h)	Transects	Effort (h)
6/7/2011	3	0.47	3	0.48	3	0.95
6/22/2011	3	0.50	3	0.50	3	1.00
7/20/2011	3	0.50	3	0.50	3	1.00
8/10/2011	3	0.50	3	0.55	3	1.05
9/7/2011	3	0.50	3	0.59	3	1.09
Total	15	2.48	15	2.62	15	5.10

Overall white perch and American shad dominated the community representing 42% and 27% of all YOY fish collected during 2011, respectively (Figure 1). Gizzard shad YOY, which were very rare in 2009 and 2010, were abundant during 2011 representing 17% of all fish collected. Striped bass and threadfin shad were common, accounting for 6% and 7%, respectively of the fish collected during 2011. Blueback herring were rare accounting for < 1% of all fish collected during all years. Relative abundance of the target species varied by site and year. American shad were a larger component of the sample at the Big Water site during 2011, where they accounted for 45% of all fish collected, than the Indian Bluff site where they represented only19% of all fish collected (Figure 1). In all years American shad have been a larger component of the fish community at the Big Water site than the Indian Bluff site. American shad relative abundance in 2011 was less than that observed during 2010 but similar to the levels observed during 2009. Gizzard shad accounted for 23% of juvenile fish collected during 2011 at Indian Bluff and represented only 5% of all fish collected at Big Water. During 2011 striped bass relative abundance was higher at Big Water (11%) than Indian Bluff (4%). Over the last three years striped bass relative abundance has decreased each year from roughly 20% of the total relative abundance during 2009 to only 6% of the total relative abundance during 2011. Relative abundance of threadfin shad at the Big Water and Indian Mound sites was similar during 2011. Overall relative abundance of threadfin shad during 2011 was similar to that observed during 2010, but less than that observed during 2009.

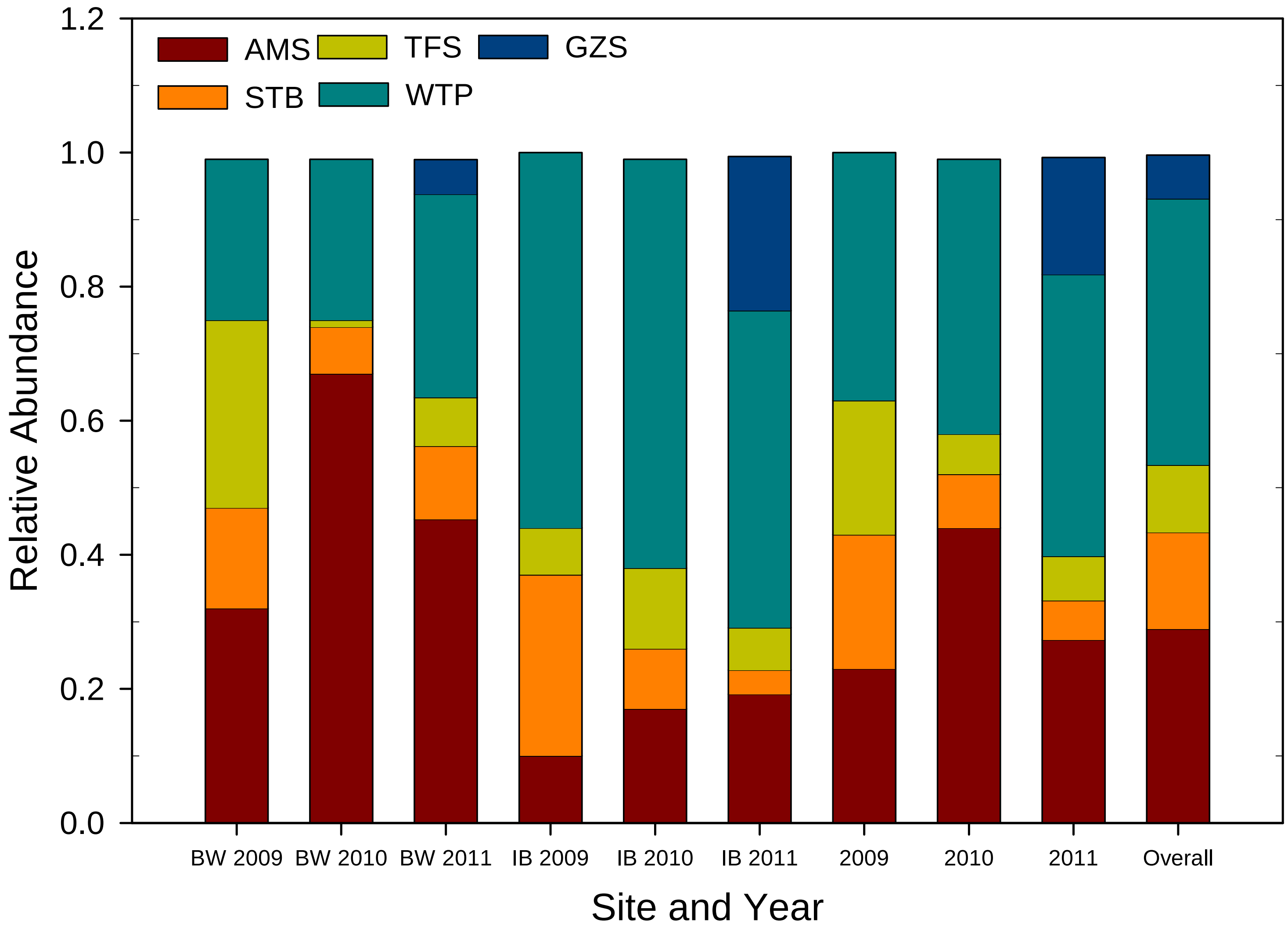

Figure 1. Relative abundance of American shad (AMS), threadfin shad (TFS), striped bass (STB), gizzard shad (GZS) and white perch (WTP) collected from the Big Water (BW) and Indian Bluff (IB) sites on Lake Marion, South Carolina, during 2009-2011. Overall relative abundance is given for each year.

Catch per unit effort (CPUE) varied among species and dates. During 2011 American shad CPUE (No/h) ranged from 29 to 615 and was similar between sites (Table 2). Striped bass CPUE ranged from 11 to 76 and was similar between sites. White perch CPUE ranged from 10 to 711 and was significantly higher at the Indian Bluff site than the Big Water site (ANOVA; $P > 0.05$) (Table 2).

Table 2. Mean catch per unit effort (no/h), standard error in parentheses, for young-of-year American shad, striped bass, and white perch at each of two Lake Marion sites sampled with boat electrofishing during 2011.

	American shad		Striped bass		White perch	
Date	Big Water	Indian Bluff	Big Water	Indian Bluff	Big Water	Indian Bluff
6/7/2011	29 (14)	152 (57)	22 (22)	11 (11)	10 (10)	21 (8)
6/22/2011	138 (35)	70 (39)	40 (16)	66 (60)	110 (95)	196 (94)
7/20/2011	106 (119)	414 (92)	76 (16)	64 (14)	116 (23)	580 (360)
8/10/2011	615 (512)	49 (19)	22 (2)	5 (3)	175 (91)	711 (216)
9/7/2011	74 (52)	184 (47)	48 (9)	15 (12)	233 (80)	496 (92)
Mean 2011	192 (77)	173 (41)	41(8)	32 (13)	129 (33)	400 (101)

During 2009 -2011 overall CPUE of American shad ranged from 101 to 208 and CPUE of white perch ranged from 156 to 265; CPUE of both species was similar among years (Table 3). CPUE of striped bass ranged from 34 to 96 and was significantly higher during 2009 than 2010 and 2011 (ANOVA; $P < 0.05$).

Table 3. Mean catch per unit effort (no/h), standard error in parentheses, for young-of-year American shad, striped bass, and white perch collected from Lake Marion with boat electrofishing during 2009-2011.

Year	American Shad	Striped Bass	White Perch
2009	101 (34)	96 (21)	156 (28)
2010	208 (58)	34 (9)	185 (34)
2011	183 (43)	36 (7)	265 (58)

On 7 June 2011 American shad mean total length (TL) was 63 mm (SE = 2.0), American shad grew slowly throughout the summer and reached a mean TL of 68 mm (SE = 0.5) by early October (Figure 2). Growth of American shad was highly variable during 2011 and appeared to be much slower in 2011 than 2009, but similar to the growth observed during 2010. Growth of threadfin shad during 2011 was similar to 2010, but slower than the growth observed during 2009. During early September 2009 threadfin shad mean TL was 82 mm TL (SE = 0.47), but during September 2011 mean TL of threadfin shad was only 71 mm TL (SE = 0.6) (Figure 1).

In early June white perch mean total length was 49 mm (SE = 1.4), white perch grew steadily throughout the summer and attained a mean TL of 73 mm (SE = 0.97) by early October (Figure 3). White perch growth during 2011 was similar to the growth observed during 2010, but slower than that observed during 2009. Striped bass mean TL in early June was 74 mm (SE = 4.3) (Figure 2). With the exception of an anomalous observation in August striped bass grew steadily through the summer reaching a mean TL of 140 mm (SE = 9.8) by October. Striped bass growth during 2011 was comparable to the growth observed in 2009 and 2010, but much slower than that observed in 2008.

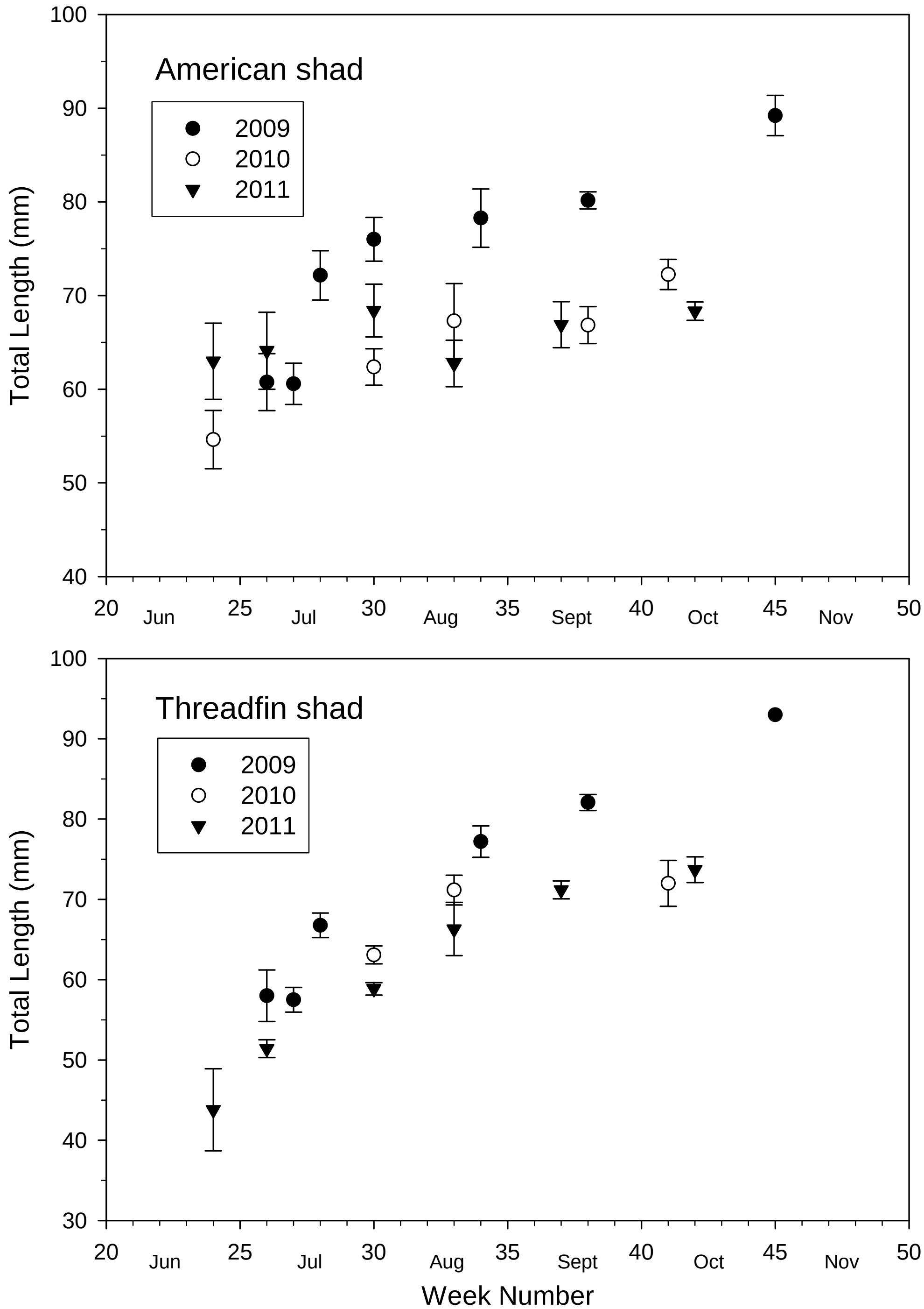

Figure 2. Mean total length (± 2 SE) of American shad and threadfin shad collected from Lake Marion, South Carolina during 2009 - 2011.

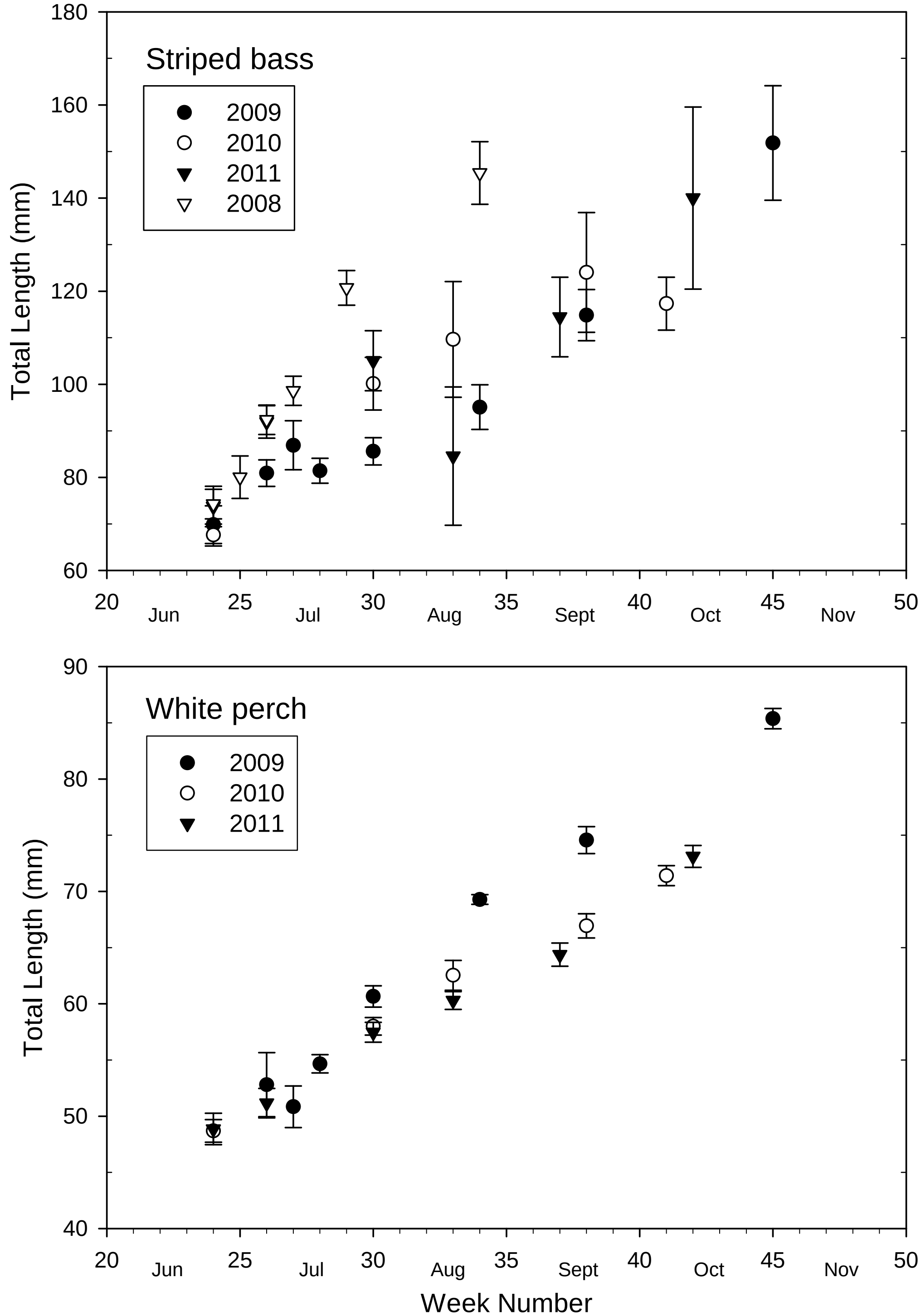

Figure 3. Mean total length (± 2 SE) of striped bass and white perch collected from Lake Marion, South Carolina during 2009 and 2010.

Condition (Kn) of juvenile striped bass and American shad was calculated for fish collected during 2009 -2011. American shad Kn was significantly different each year with the highest values observed during 2009 and the lowest value observed during 2010 (ANOVA $P < 0.05$) (Table 4). Striped bass Kn was also different each year with the highest value observed during 2010 and the lowest value observed during 2009. Although condition varied statistically among years, mean condition was not “poor” in any year and difference may not be of biological signifigance.

Table 4. Mean condition (Kn), number of observations (N) and standard error (SE) for American shad and striped bass collected during nighttime electrofishing at Lake Marion, SC during 2009 - 2011.

	American Shad			Striped Bass		
Year	N	Kn	SE	N	Kn	SE
2009	317	104	0.77	405	99	0.38
2010	466	97	0.88	178	103	0.57
2011	517	102	0.93	244	101	0.47

Diet

During 2009 a sample of each of the key species was retained on every date for diet analysis. During FY11 109 striped bass (mean TL = 92 mm; range 53 -165 mm TL) and 120 white perch (mean TL = 65 mm; range 45 -82 mm TL) stomachs collected during 2009 were excised and examined for contents. Six of the striped bass stomachs and 9 white perch stomachs were empty, but all remaining stomachs contained at least some items. A total of 26 different taxa were identified in the dissected stomachs (Table 5). The most numerous taxa encountered were cladocera in the family Sididae, cyclopoid copepods and chironomids. Taxa from six different cladocera were found including the invasive *D. lumholtzi*. The only fish identified in the stomachs of striped bass and white perch were tessellated darter.

Table 5. Taxa identified in the stomachs of striped bass and white perch collected from Lake Marion, SC during summer 2009. The number of stomachs (N) containing each taxa, and the total number of each taxa observed.

				Striped Bass		White Perch	
				N	Total	N	Total
Nematoda				14	55	1	1
Mollusca							
	Bivalve			1	1	1	1
	Gastropoda			3	8	0	0
Annelida							
	Oligochaeta			6	8	2	2
Bryozoa							
	Bryozoan			1	1	0	0
Chelicerata							
	Hydracarina			4	27	3	4
Crustacea							
	Amphipoda			4	13	14	43
	Cladocera						
		Bosminidae		1	4	0	0
		Daphniidae	*Daphnia lumholtzi*	1	3	0	0
		Sididae	*Diaphanosoma sp.*	19	1384	37	220
			Latona sp.	7	111	38	393

		Ilyocryptidae	*Ilyocryptus sp.*	0		1	2
		Leptodoridae	*Leptodora kindti*	2	33	0	0
		Chydoridae		3	12	18	57
	Copepoda						
		Cyclopoida		30	545	85	1819
	Ostracoda			2	2	33	146
Hexopoda							
	Coleoptera		*Gyrinidae dineutus*	1	1	0	0
	Diptera						
		Ceratopogonidae		0	0	7	7
		Chaoboridae		15	40	22	89
		Chironomidae		47	299	94	1294
		Ephemeroptera					
			Caenis sp.	15	50	42	109
			Hexagenia limbata.	9	14	1	2
	Odonata		*Anisoptera*	1	1	0	0
	Trichoptera			2	15	6	12
Fish							
			Etheostoma olmstedi	10	15	0	0

Aquatic dipterians (pupae and larvae) and fish were the most commonly encountered items in striped bass stomachs occurring in 59% and 39% of stomachs, respectively (Table 6). Copepods, ephemeroptera (larvae and adults), and cladocerans were common occurring in at least 24% of striped bass stomachs. Nematodes occurred in 14% of striped bass stomachs and all other prey items were rarely (<5% of fish) encountered. Despite fish occurring in 39% of striped bass stomachs the only fish we were able to identify was tessellated darter.

Nearly every (98%) white perch stomach contained aquatic diptera (pupae and larvae). Copepods, cladocerans, and ephemeroptera larvae occurred in at least 38% of all white perch stomachs. Although ostracods and amphipods were rarely seen in striped bass they were commonly encountered in white perch, occurring in 29% and 13% of stomachs, respectively.

White perch and striped bass young-of-year consumed similar prey items during summer 2009. In prey taxa groups where more than one individual striped bass or white perch consumed a prey taxa there was very little difference in the items consumed. Striped bass consumed tessellated darters, gastropods and *Leptodora kindti* which were not consumed by white perch. The only prey taxa consumed by white perch, but not striped bass was ceratopogonids. There were size related differences in prey taxa consumed. Striped bass larger than 100 mm TL rarely consumed mircocrustaceans and instead fed on larger prey. Fish were consumed by 72% of striped bass larger than 100 mm TL.

Table 6. Frequency of occurrence of prey taxa in examined stomachs and the total number of each taxa found in the stomachs of striped bass and white perch collected from Lake Marion, South Carolina during 2009.

		Striped bass		White perch	
Phyla	Prey taxa	Percent	Total	Percent	Total
Nematoda	Nematoda	14%	55	1%	1
Mollusca	Bivalve	1%	1	1%	1
	Gastropoda	3%	8	0%	0
Annelida	Oligochaeta	6%	8	2%	2
Bryozoa	Bryozoa	1%	1		
Chelicerata	Hydracarina	4%	27	3%	4
Crustacea	Amphipod	4%	13	13%	43
	Cladoceran	24%	1851	59%	777
	Copepod	29%	545	77%	1819
	Ostracod	2%	2	30%	146
Hexopoda	Coleoptera	1%	1	0%	
	Diptera	59%	511	98%	1735
	Ephemeroptera	28%	70	39%	116
	Odonata	1%	1	0%	
	Trichoptera	2%	15	5%	12
Fish	Fish	39%	57	5%	5

Discussion

Size ranges of American shad and striped bass caught during summer 2010 and 2011 were highly variable. For example, striped bass TL on 12 October 2011 ranged from 76 mm to 237 mm and on 20 July 2011 American shad TL ranged from 32 mm to 100 mm. It is not clear whether the disparity in sizes is due to different growth rates, earlier or later spawned cohorts, or potentially the difference between wild and hatchery reared fish. The huge size range of American shad caught during June lead us to review 18 otoliths to ensure that the larger fish (> 70 mm TL) were from the 2011 year class. Each of the otoliths reviewed appeared to be from age-0 fish, and based on cursory examination appeared to show some fish were hatched perhaps a month earlier in the year than others.

Preliminary analysis of striped bass and white perch stomach contents show considerable diet overlap until striped bass reach 100 mm TL. Once larger than 100 mm TL striped bass feed less on microcrustaceans and relied more heavily on fish. Whether or not the diet overlap of young-of-year white perch and striped bass results in resource competition would largely depend on the consumption rates of each species and the availability of the prey resources. Surprisingly the only fish we found in the stomachs of striped bass were tessellated darter, there was no evidence of YOY striped bass feeding on clupeids during the summer. Based on first summer growth rates of striped bass and clupeids in Lake Marion it appears that YOY clupeids are largely unavailable to striped bass as a prey resource during their first summer.

Recommendations

During FY12 we will combine juvenile fish data collected from Lake Marion with similar data collected from Lake Moultrie. Once a database has been constructed the data will be used to describe relative abundance, growth and condition of each species and evaluate spatial and temporal differences within the lakes. American shad diet samples collected from Lake Marion during 2009 will be processed and the potential for resource competition assessed among the key species.

Genetic samples collected from 176 YOY striped bass during summer 2011 will be processed to examine the contribution of stocked fish at our sample locations.

Literature Cited

None

Prepared By: Jason Bettinger Title: Wildlife Biologist III

Job Title: Twelvemile Creek Dam Removal Monitoring

Period Covered January 1, 2011 through December 31, 2011

Summary

This report details fish collections for two 2011 Twelvemile Creek samples; one immediately after the removal of Woodside I Dam (April 2011), and the other immediately after the removal of Woodside II Dam (October 2011). We used two methods to examine initial changes in biological composition due to dam removal. We plotted fish metrics by site and year to evaluate temporal trends, and used non-parametric multidimensional scaling (NMDS) to examine, visualize, and interpret changes in community composition. We found that prior to dam removal, the biological composition of impoundments were distinct from their immediate downstream free-flowing counterparts. Impoundments were characterized by greater densities of sunfish and bass, and free-flowing sections were characterized by greater densities of darters, shiners, and madtom species. After dam removal, the upper impoundment (Woodside I Above) has become more similar in composition to its immediate downstream free-flowing counterpart (Woodside I Below) and the upstream reference reach. Species richness, darter density, and cyprinid density have increased at the former upper impoundment. In contrast, the lower impoundment (Woodside II Above) and its downstream free-flowing counterpart (Woodside II Below) have both become more similar in composition to the alluvial downstream reference reach. We observed a decrease in darters and cyprinids, and an increase in invasive species at these sites immediately following the removal of the lower dam. This contrast may be partially due to cumulative downstream habitat disturbances resulting from the dam removal process (increased deposited sediment, increased turbidity, decreased depths and velocities).

Introduction

Dams alter riverine environments by converting lotic habitats to lentic ones, thereby altering physical habitat, flow-regimes, temperature-regimes, sediment transport, dendritic connectivity, and nutrient cycling (Bednarek 2001). As a consequence, dams change the composition, structure, and function of native fish communities (Martinez et al. 1994, Taylor et al. 2001, Santucci et al. 2005). Few evaluations of the ecological effects of dam removal have been conducted in North America due to the lack of opportunity, particularly in the Southeast. A rare opportunity has presented itself with the removal of two mainstem dams on Twelvemile Creek, Pickens County, South Carolina (Figure 1).

Twelvemile Creek was extensively polluted with PCBs originating from a capacitor manufacturing plant from 1955 – 1975; the waste site and its receiving waters were listed with the EPA Superfund program in 1990. Under the CERCLA statute (Superfund law), a natural resources board of trustees is authorized to act as trustees of natural resources on behalf of the public, and within that role they may assess and recover damages for injuries and losses to natural resources caused by a hazardous waste site. As part of the settlement for damages caused by PCB contamination, a natural resources board of trustees facilitated the removal of two mainstem dams on Twelvemile Creek in order to 1) remove any remaining contaminated sediments that have accumulated behind the dams, and 2) to promote sediment transport to further 'cap' contaminated sediments downstream and in Lake Hartwell. Dam removal began in August 2009 with the initial dredging behind the upper dam (Woodside I); this dam was completely removed by April 2011. Dredging and removal preparations began on the lower dam (Woodside II) in April 2011, and removal was completed in September 2011.

The objective of this investigation is to document changes in the aquatic ecosystem before and after the removal of the two dams (Woodside I and Woodside II). The project should provide information on a series of questions:

1) How do biological communities and environmental factors in the impounded reaches differ from those found in free-flowing sections of Twelvemile Creek?
2) What are the effects of dam removal on downstream biological communities, habitat, water quality, and channel dimensions?
3) How long does it take for the biological community, habitat, water quality and geomorphology in the impounded reaches to recover to a typical stream ecosystem?

This report details fish collections for two 2011 samples, one immediately after the removal of Woodside I Dam (April 2011), and the other immediately after the removal of Woodside II Dam (October 2011). We used two methods to examine how community composition has changed through the initial process of dam removal. We plotted fish metrics by site and year to evaluate temporal trends, and used non-parametric multidimensional scaling (NMDS) to examine, visualize, and interpret changes in community composition.

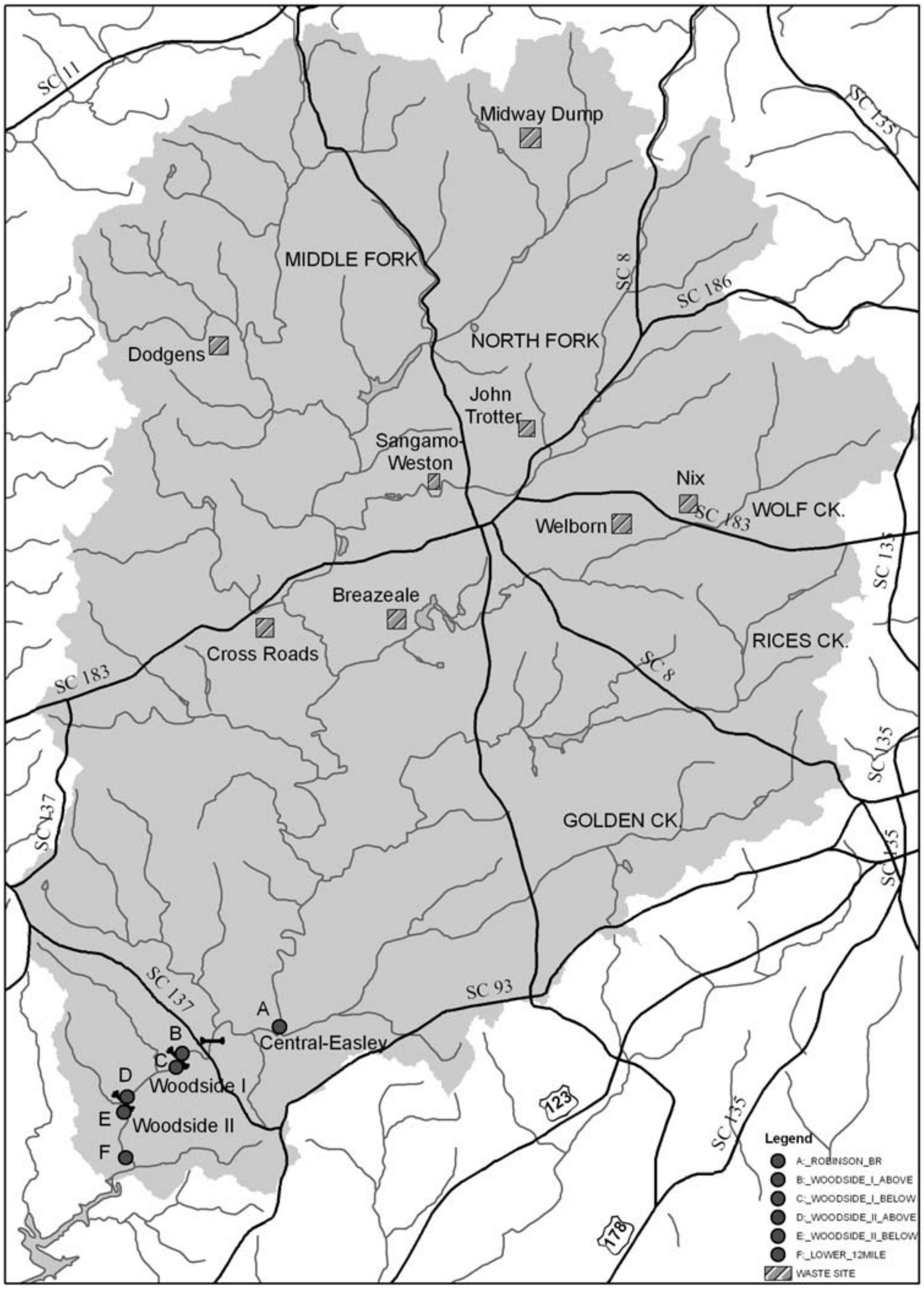

Figure 1. Map of Twelvemile Creek drainage (shaded) showing the two former.

Woodside Dams, the remaining Easley-Central Dam, eight sample site locations (A-D), Sangamo-Weston (primary waste site), and six satellite waste sites.

Materials and Methods

Eight sampling stations were established for collecting biological, habitat, and water quality data, and conducting geomorphological measurements (Figure 1). Six stations are located on Twelvemile Creek, distributed as follows: 1) the alluvial stream section downstream of Woodside II dam (Twelvemile Lower), 2) the bedrock-constrained stream section downstream of Woodside II Dam (Woodside II Below), 3) the impounded area above Woodside II Dam (Woodside II Above), 4) the bed-rock constrained flowing section downstream of Woodside I Dam (Woodside I Below), 5) the impounded area above Woodside I Dam (Woodside I Above), and 6) a reference station in the flowing section upstream of the Easley-Central Water District Dam and Reservoir (Robinson Bridge). Two stations are located in nearby Three and Twenty Creek, a stream system that is similar in physiography and drainage area but lacking major mainstem dams. The two Three and Twenty Creek reference stations (LaFrance and Burns Bridge) are located a similar longitudinal distance apart as the extreme upstream and downstream Twelvemile stations, and will be monitored concurrently with the Twelvemile Creek stations to document variation in aquatic variables longitudinally and over time in a system not undergoing dam removal.

Two fish samples were conducted in April and October of 2011 (Table 1). The April sample occurred approximately 1 month after Woodside I was removed. The October sample occurred approximately 1 month after Woodside II was removed, and ~7 months post Woodside I removal. Fishes were collected within 300m segments at each station with a standardized effort using electrofishing gear and seines. Backpack electrofishers and seines were used in wadeable stream segments to sample a standard area of 15 m^2. All fishes encountered were collected, field identified to species level, recorded, and released.

Table 1. The eight stations locations and sample dates for April and October of 2011. April samples were taken approximately 1 month after the removal of Woodside I, and the October samples were taken approximately 1 month after the removal of Woodside II.

Site	Sample Date April 2011	Sample Date October 2011	Lattitude (°N)	Longitude (°W)
Robinson Bridge	4-Apr-2011	18-Oct-2011	34.78079	-82.75465
Woodside I Above	4-Apr-2011	18-Oct-2011	34.77456	-82.77877
Woodside I Below	4-Apr-2011	18-Oct-2011	34.7717	-82.77998
Woodside II Above	8-Apr-2011	17-Oct-2011	34.76583	-82.79163
Woodside II Below	8-Apr-2011	17-Oct-2011	34.76262	-82.79202
Twelvemile Lower	7-Apr-2011	17-Oct-2011	34.75367	-82.79219
3&20 LaFrance	7-Apr-2011	2-Nov-2011	34.60878	-82.76286
3&20 Burns Bridge	7-Apr-2011	2-Nov-2011	34.58987	-82.78222

Fish assemblage metric scores were plotted by site and year to evaluate temporal trends before and after dam removal. Metrics evaluated included: species richness, Simpson diversity, darter density, cyprinid density, sunfish density, and invasive species density (Table 2). Additionally, I used non-metric multidimensional scaling (NMDS) to examine changes in community composition before and after dam removal. NMDS is a non-parametric ordination technique that translates the n-dimensional (n= # taxa) community in relatively few dimensions (usually 2 or 3) so that differences between sites are readily interpreted visually. In a robust NMDS plot, distances between points on the plot are directly representative of the differences in species composition of communities.

Table 2. Fish metric definitions.

Fish Metric	Definition
Species Richness	number of non-zero elements in a row
Simpson Diversity	1-sum (Pi*Pi), where Pi = importance probability in element I (element I reletivized by row total)
Darter Density	# darter individuals / total number of seine sets (darters = BBD, TQD)
Cyprinid Density	# cyprinid individuals / total numberof seine sets (cyprinids = STS, WFS, YFS)
Sunfish Density	# sunfish individuals / total number of seine sets (sunfish = BLG, GSF, RBS, RES, WAR)
Invasive Density	# invasive individuals / total number of seine sets (invasives = GSF, FCF, SPB)

Results

Fish sampling in the April 2011 sample of Twelvemile Creek sites resulted in the collection of 1079 individuals representing 23 species (Table 3). The catch among Twelvemile Creek sites was numerically dominated by two sunfishes (*Lepomis macrochirus*: n=183, *Lepomis microlophus*: n=173,), and one cyprinid *(Nocomis leptocephalus*: n=170). Conservation priority species were represented by *Ameiurus brunneus, Ameiurus platycephalus, Hybopsis rubrifrons, Micropterus coosae, Moxostoma collapsum,* and *Etheostoma inscriptum*, and comprised 17.97% of total collections in Twelvemile Creek samples (Kohlsaat 2005). One nonnative species, *Lepomis cyanellus*, was collected in three Twelvemile sample locations. We collected 204 individuals representing 18 species in the Three and Twenty Creek sites (Table 3). The catch among Three and Twenty Creek sites was numerically dominated by one sunfish (*Lepomis macrochirus*: n=72), and one cyprinid (*Nocomis leptocephalus*: n=40). Four conservation priority species were collected in the Three and Twenty Creek samples (*Ameiurus platycephalus*, *Micropterus coosae*, *Hybopsis rubrifrons*, *Ameiurus natalis*), and comprised 3.9% of total collections.

Fish sampling in the October 2011 sample of Twelvemile Creek sites resulted in the collection of 790 individuals representing 24 species (Table 4). The catch among Twelvemile Creek sites was numerically dominated by one sunfish (*Lepomis macrochirus*: n=127), and four cyprinids (*Notropis hudsonius*: n=112, *Notropis lutipinnis*: n=94, *Nocomis leptocephalus*: n=93, and *Cyprinella nivea*: n=89). Conservation priority species were represented by *Ameiurus brunneus, Ameiurus platycephalus, Hybopsis rubrifrons, Micropterus coosae, Moxostoma collapsum,* and *Etheostoma inscriptum*, and comprised 11.8% of total collections in Twelvemile Creek samples (Kohlsaat 2005). We collected 116 individuals representing 13 species in the Three and Twenty Creek sites (Table 4). The catch among Three and Twenty Creek sites was numerically dominated by one sunfish (*Lepomis macrochirus*: n=28), one darter (*Percina nigrofasciata*: 25), and one cyprinid (*Notropis lutipinnis*: n=18). No conservation priority species were collected in the October 2011 Three and Twenty Creek samples.

Habitat variables differed dramatically after dam removal in Twelvemile Creek; of note, median particle size increased substantially at impounded sites. Median particle size in the impoundment above Woodside I was 111.5 mm in April (~ 1 month post dam removal), and 52.5 mm in October 2011 (~ 7 months post dam removal), where prior to dam removal the impoundment was dominated by fine particles (~ 0.5 mm). Median particle size at the free-flowing site immediately below Woodside I was 23 mm in April and 780.5 mm in October, indicating that finer sediments present immediately after dam removal were largely washed out by 7 months post dam removal. We observed a similar initial pattern of change in inorganic substrate composition at the sites flanking Woodside II. Median particle size in the impoundment above Woodside II was 1.5 mm in April (prior to dam removal), and was 12 mm in October (~ 1 month post removal). Median particle size at the free-flowing site immediately below Woodside II was 415 mm in April (prior to dam removal), and was 0.75 mm in October (~ 1 month post removal), indicating that a large amount of fine sediment was transported downstream during and after the removal of the Woodside II Dam. Although not directly quantified, we observed a greater amount of fine sediments downstream of Woodside II immediately after dam removal than we observed downstream of Woodside I immediately after dam removal.

Water quality measurements taken at Twelvemile Creek stations were relatively consistent between April and October 2011 samples. Water temperatures ranged from (10.9 - 13.9 °C) in April, and (9.65– 13.64 °C) in October. Dissolved oxygen levels were near 11 mg/L in April, and near 12 mg/L in October. Conductivities among all Twelvemile sites in April (~ 1 month past

Woodside I dam removal) were between 35-36 μS/cm. In October, conductivities were slightly higher in sites downstream that were affected by the removal of Woodside II (47-51 μS/cm: ~ 1 month past Woodside II dam removal) than found in sites above Woodside II (43-45 μS/cm). pH remained between 7 and 8 in both April and October samples. Turbidities ranged between (8.76 – 15.05 NTU) in April 2011, and (4.15 – 6.92 NTU) in October 2011.

Fish assemblage metric scores were plotted by site and year (Figure 2). Species richness was greater at all of the non-impounded sites (Robinson Br., Woodside I Below, Woodside II Below, and Twelvemile Lower) than the two impounded sites (Woodside I Above, Woodside II Above) prior to dam removal. However, after dam removal species richness increased at impounded sites. Of note, species richness in the impoundment above Woodside I was 9 or less prior to dam removal, and increased to 13 seven months post dam removal. Simpson's diversity was higher and more consistent through time at the two sites below all dams prior to dam removal (connected to Lake Hartwell: Twelvemile Lower, Woodside I Below). Diversity has not increased in sites after dam removal, however it is highly likely that future samples will show increased diversity in sites above former dams as new species migrate to these reaches. We observed a decrease in darter density in the free-slowing sections below Woodside I and Woodside II during and after the process of dam removal (August 2009 through September 2011); reduced densities may be the result of general habitat disturbances (increased fine sediment, increased turbidity) resulting from dam removal activities. Interestingly, we observed a substantial increase in darter density in the impoundment above Woodside I at approximately seven months post dam removal, accompanying the change from lentic to lotic conditions and a general increase in median particle size. We observed high seasonal variation in cyprinid density at the two sites below previous dams (Woodside II Below, Twelvemile Lower), with spikes occurring in the fall. We observed greater cyprinid density in the impoundment above the upper dam (Woodside I Above) at ~ 7 months post dam removal than was observed prior to dam removal. Sunfish density also showed high seasonal variability the two sites below previous dams, with spikes occurring in the spring. We observed a decrease in sunfish density in the impoundment above Woodside II after dam removal, as the section reverted from lentic to lotic conditions. The density of invasive species was variable through time at all sites. However, the density of invasive species was greater at both sites below previous dams (Twelvemile Lower, Woodside II Below) immediately after the removal of Woodside II, the lower dam. Of particular concern, we captured *Micropterus punctulatus* for the first time in Twelvemile Creek in our October 2011 samples. We captured *M. punctulatus* in the most downstream site (Twelvemile Lower), as well as in both sites above and below the former lower dam (Woodside II Below and Above). No *M. punctulatus* were captured at any of the Twelvemile sample locations in five sampling rounds prior to dam removal, even in the two sites below former dams (fully connected to Lake Hartwell). However, we regularly captured *Micropterus coosae* at all sites prior to dam removal. This presents an ecological concern given that *M. punctulatus* hybridizes with *M. coosae*. Prior to their removal, the Woodside Dams acted as barriers to invasion of *M. punctulatus* in Twelvemile Creek. Fortunately, a third dam (Easley-Central Dam) remains on Twelvemile Creek that effectively blocks the upper reaches from invasion. If *M. coosae* is indeed a separate species, as genetics studies indicate, we may need to weigh the benefits of retaining the Easley-Central Dam as a protective barrier against the invasion of *M. punctulatus* from below. The introduction of *M. punctulatus* above the Easley-Central dam may also impact other native species as well.

NMDS showed that differences in community composition among sites were strongly related to changes in habitat conditions before and after dam removal, and the assemblage likely varied longitudinally. Three general site groupings emerged in the ordination 1) both impoundments before dam removal, 2) downstream (Twelvemile Lower) samples, Woodside II free-flowing after dam removal, Woodside II impoundment after dam removal, and 3) upstream (Robinson Br.) samples, Woodside I free-flowing sites before and after dam removal, and Woodside I impoundment after dam removal (Figure 3). Group 1 was characterized by low velocities, relatively shallow, and sand substrates. Common fishes found in impoundments included sunfishes and bass. Group 2 was characterized by intermediate velocities, small median particle size, and relatively shallow depths. Common fishes include sunfish and bass, as well as shiners, darters, and madtoms. Group 3 was characterized by high velocities, large substrates, and greater depths. Species associated with group 3 sites were dominantly darters and cyprinids.

The ordination showed that both impoundments (Woodside I and II Above) had similar species compositions prior to dam removal. After dam removal, the composition of the upper impoundment (Woodside I Above) became more similar in composition to both the free-flowing section immediately downstream (Woodside I Below) and the upstream reference reach (Robinson Br.). In contrast, after dam removal the composition of the lower impoundment (Woodside II Above) has initially become more similar to the downstream reference site, Twelvemile Lower. The ordination showed that both free-flowing sites (Woodside I and II Below) had similar species compositions prior to dam removal. After dam removal, the composition of the free-flowing site immediately below Woodside I remained similar in composition as observed prior to dam removal. In contrast, the composition of the free-flowing site immediately below the lower dam, Woodside II Below, became more similar to the composition of the downstream reference site. We observed a greater amount of fine sediments downstream of Woodside II immediately after dam removal than we observed downstream of Woodside I immediately after dam removal. While the communities of Woodside II Above and Below were initially more similar to the lower reference immediately after dam removal, we think this may be partially due to habitat disturbances due to the dam removal process (increased deposited sediment, increased turbidity, decreased depths and velocities), since the underlying habitat pallet is more similar to that of the upstream sites (high gradient reaches). However, continued monitoring may highlight longitudinal differences in community structure due to proximity to Lake Hartwell.

Table 3. Summary of fish species and numbers collected in Twelvemile and Three and Twenty Creeks in April 2011.

Species\Site	Robinson Bridge	Woodside I Above	Woodside I Below	Woodside II Above	Woodside II Below	Twelvemile Lower	3&20 LaFrance	3&20 Burns Bridge	Total
BBD	6	0	28	0	13	7	12	2	68
BHC	11	0	25	110	24	3	21	19	213
BLC	0	0	0	0	0	0	0	2	2
BLG	4	1	3	14	128	33	9	63	255
CCF	0	0	0	4	0	3	0	0	7
CRC	0	0	0	5	0	0	0	0	5
FBH	1	0	5	1	0	0	1	0	8
GLS	0	0	0	8	0	0	0	0	8
GSF	1	0	3	5	0	0	3	5	17
LMB	0	0	2	0	2	1	0	0	5
MGM	1	0	3	1	1	0	5	9	20
NHS	4	2	5	10	3	1	5	4	34
NLR	0	0	6	3	0	2	0	0	11
RBS	4	2	10	16	0	2	2	6	42
REB	0	0	1	0	0	0	1	1	3
RES	0	0	0	1	150	22	1	2	176
RFC	71	4	11	7	0	0	1	0	94
SBH	1	0	17	1	0	1	2	2	24
STS	0	0	30	0	3	6	2	5	46
TQD	0	0	43	0	19	0	0	0	62
WAR	0	0	0	0	0	0	0	1	1
WFS	2	5	12	8	1	48	0	1	77
WTB	0	0	0	0	1	0	0	0	1
YFS	5	0	30	15	22	4	9	5	90
YLP	0	0	0	0	11	0	3	0	14
Total	111	14	234	209	378	133	77	127	1283
Richness	12	5	17	16	13	13	15	15	25

Table 4. Summary of fish species and numbers collected in Twelvemile and Three and Twenty Creeks in October 2011.

Species\Site	Robinson Bridge	Woodside I Below	Woodside II Above	Woodside II Below	Twelvemile Lower	3&20 LaFrance	3&20 Burns Bridge	Total
BBD	20	23	3	0	12	19	6	83
BHC	22	66	0	1	4	9	6	108
BLC	0	0	0	0	1	0	0	1
BLG	10	9	0	1	107	12	16	155
CCF	0	0	0	3	6	0	0	9
FBH	2	1	0	1	1	0	0	5
FCF	0	0	0	5	0	0	0	5
GSF	1	1	0	2	4	1	2	11
LMB	0	0	1	0	2	0	0	3
MGM	3	1	0	0	0	2	5	11
MSQ	0	0	0	0	5	0	0	5
NHS	13	42	3	0	2	2	3	65
NLR	3	1	0	0	1	0	0	5
RBS	2	6	0	1	6	4	2	21
REB	0	5	1	0	2	0	0	8
RES	0	0	0	0	8	2	1	11
RFC	39	2	0	0	0	0	0	41
SBH	2	11	3	1	0	1	2	20
SPB	0	0	1	2	1	0	0	4
STJ	0	0	0	0	0	2	0	2
STS	0	0	6	12	94	0	3	115
TQD	0	17	0	0	0	0	0	17
WAR	0	0	0	1	1	0	0	2
WFS	13	10	4	0	62	0	2	91
YFS	14	72	3	5	0	10	4	108
Total	144	267	25	35	319	64	52	906
Richness	13	15	9	12	18	10	12	25

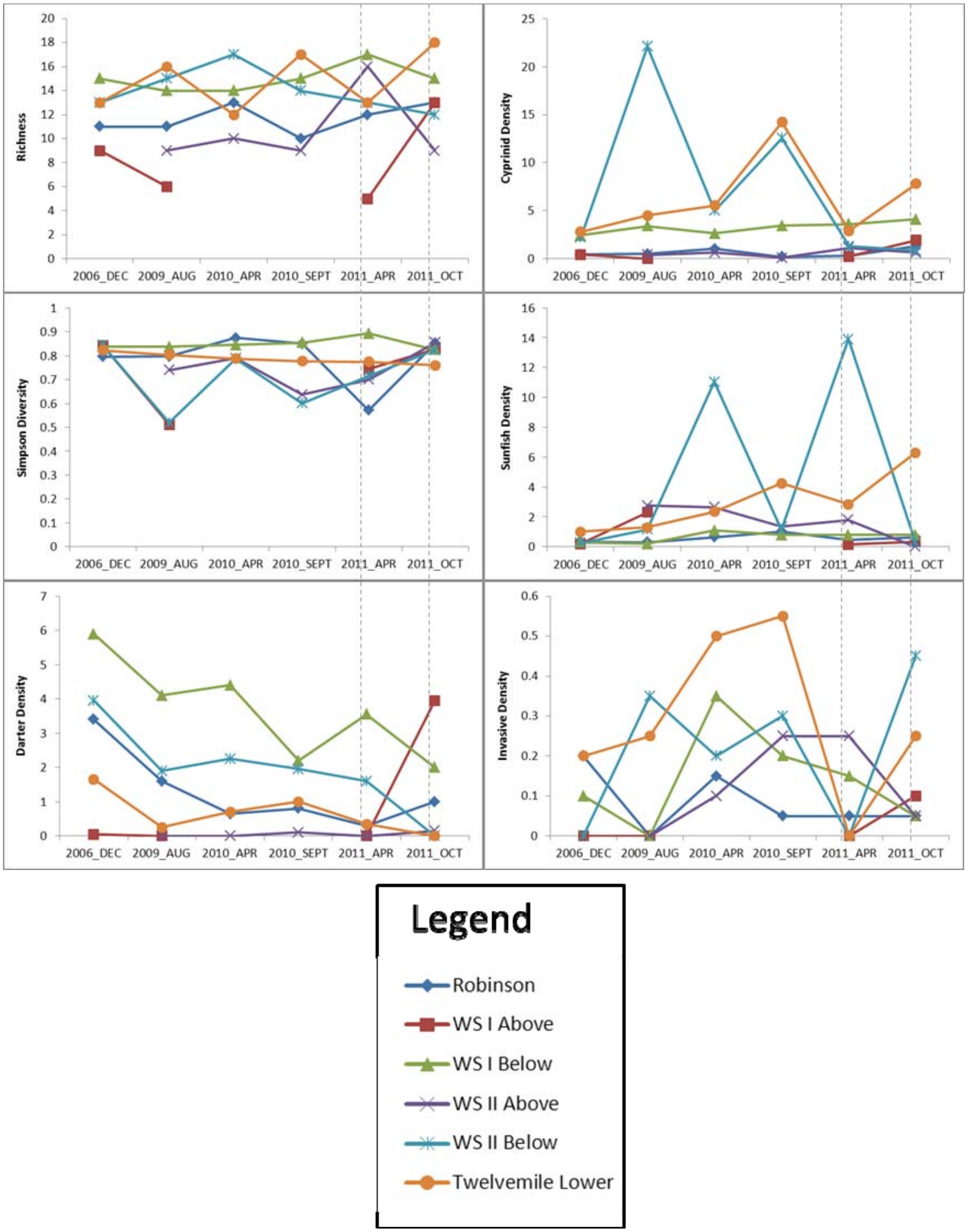

Figure 2. Fish species richness, Simpson diversity, darter density, cyprinid density, sunfish density, and invasive species density through time at Twelvemile Creek sample sites. The left-most vertical dashed line represents the complete removal of Woodside I dam, and the right-most vertical dashed line represents the complete removal of Woodside II dam.

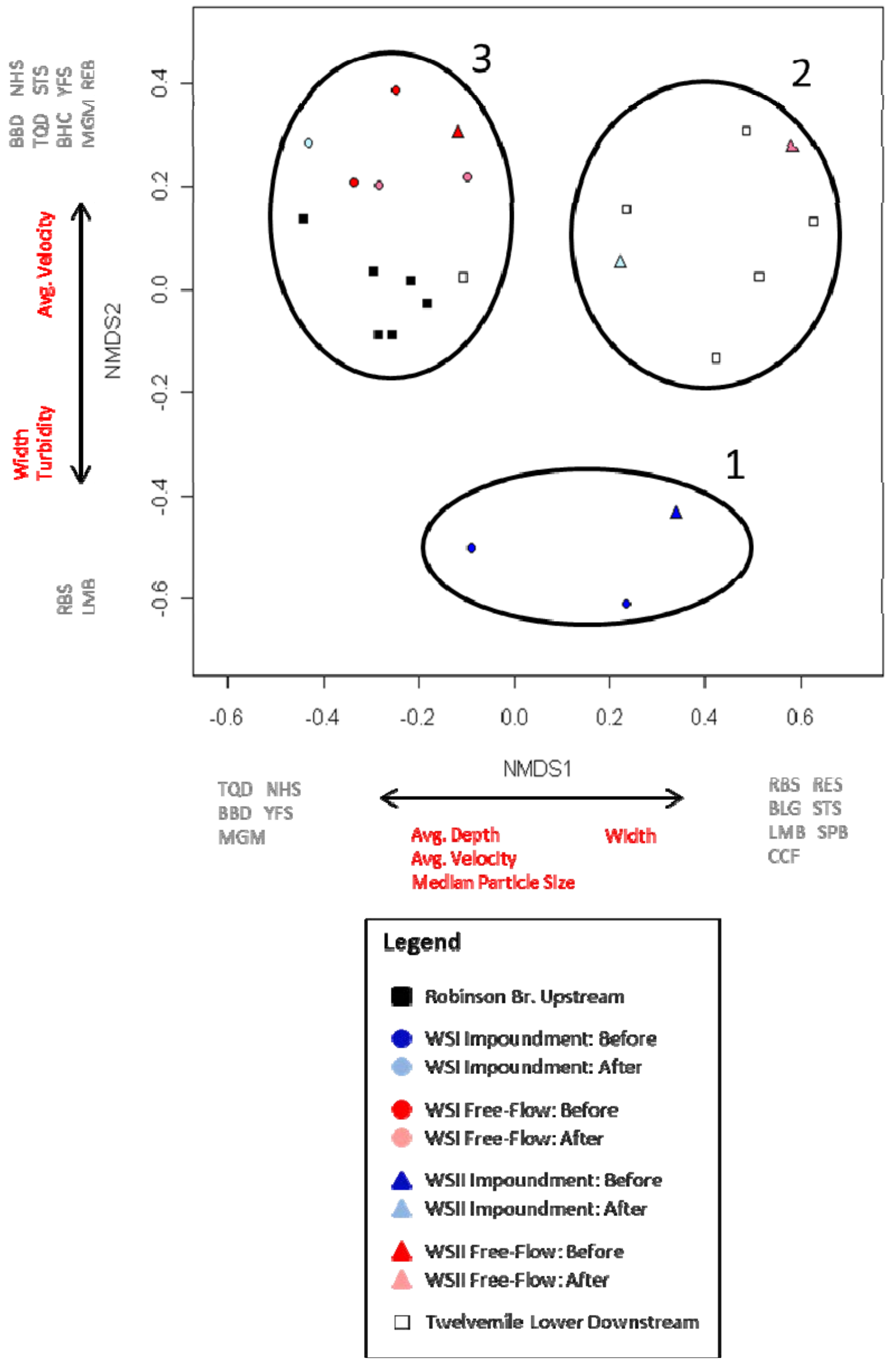

Figure 3. Non-metric multidimensional scaling of fish species by site. Points closer to one another in the ordination are more similar in species composition. Species names in gray refer to species that correlate strongly on each axis, whether positively or negatively. Environmental parameters in red are parameters that were strongly correlated with each axis. Species correlations are inherent weights (i.e. the ordination is based on the species) whereas environmental correlations are post-hoc. Inferred groups are indicated with black circles.

Discussion

Prior to dam removal, the biological composition of impoundments were distinct from their immediate downstream free-flowing counterparts. Impoundments were characterized by greater densities of sunfish and bass, and free-flowing sections were characterized by greater densities of darters, shiners, and madtom species. After dam removal, the upper impoundment (Woodside I Above) has become more similar in composition to its immediate downstream free-flowing counterpart (Woodside I Below) and the upstream reference reach. Species richness, darter density, and cyprinid density have increased at the former upper impoundment. In contrast, the lower impoundment (Woodside II Above) and its downstream free-flowing counterpart (Woodside II Below) have both become more similar in composition to the alluvial downstream reference reach. We observed a decrease in darters and cyprinids, and an increase in invasive species at these sites immediately following the removal of the lower dam. This contrast may be partially due to cumulative downstream habitat disturbances resulting from the dam removal process (increased deposited sediment, increased turbidity, decreased depths and velocities).

Recommendations

The sampling events from 2011 will serve as benchmarks for the initial ecological impacts due to dam removal activities. We will continue standardized sampling according to schedule at Twelvemile Creek and Three and Twenty Creek to provide a multi-year record of post dam-removal ecological conditions.

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Job Title: Recovery of the Main Stem Reedy River Fish Community from a Major Oil Spill

Period Covered January 1, 2011 through December 31, 2011

Summary

Previous research (Kubach et al. 2011) indicated that fish assemblage structure of Reedy River sites impacted by the 1996 oil spill largely recovered by 2000 (4.3 years post-disturbance). The incorporation of data from two additional sampling periods (2008, 2011) reiterated this long-term pattern of recovery, however also showed compositional differences from previous sample years (1996-2005). A NMDS ordination indicated that the 2008 and 2011 sample periods were characterized by a decreased abundance of Ictalurid species, and increased abundances of several shiner and darter species. The observed compositional differences may be due to changes in stream habitat resulting from prolonged drought, resultant low flows, and the loss of deep water habitats preferred by riverine Ictalurid species.

Introduction

In one of the largest inland oil spills in United States history, a petroleum pipeline ruptured and released 22,800 barrels (957,000 gallons) of diesel fuel into the Reedy River near Fork Shoals, South Carolina, on June 26, 1998. Early July 1996 surveys confirmed near-complete fish and macroinvertebrate extirpation for 37 km downstream to the headwaters of Boyd Mill Pond, a 74-ha impoundment on the Reedy River (Rankin et al. 1996, Glover 1996). Approximately 94% of the oil was recovered within 12 days of the incident with the remainder primarily infiltrating the ground water near the spill site. Therefore, the oil spill represented an acute impact of a relatively short-lived stressor. The overall objective of this research has been to document the recovery of fish assemblage structure within the portion of the river impacted by the oil spill. Kubach et al. (2011) found that impacted Reedy River sample locations showed initial recovery in 1997, species re-colonization in 1998, and that overall recovery was largely achieved by 2000 (4.3 years post-disturbance). Data collected in 2005 further recapitulated this finding. The focus of this report is on the incorporation of two subsequent fish samples (2008, 2011) to the recovery analysis, as well as provide a general characterization of the most recent and final Reedy River mainstem fish sample, September 2011.

Materials and Methods

A longitudinal sampling framework was implemented to monitor the recovery of the affected river section. Five fixed sites—an undisturbed reference site approximately 5 km upstream of the oil spill origin, and four sites ranging from 2-30 km downstream within the disturbed section—were each sampled once in August 1996 (1.5 months post-disturbance), October 1996 (4 months post-disturbance), October 1997 (16 months post-disturbance), October 1998 (28 months post-disturbance), October 2000 (52 months post-disturbance), September-October 2005 (112 months post-disturbance), September-October 2008 (148 months post-disturbance), and September 2011 (182 months post-disturbance) (Table 1). Site A was not sampled in August 1996; therefore, a total of 39 samples have been conducted throughout the course of this research.

Table 1. Reedy River mainstem site recovery monitoring study sites relative to the oil spill origin.

Site	Type	Position Relative to Spill Site (river km)
833-REF	Reference	5.4 km upstream
845-A	Disturbed	1.8 km downstream
835-B	Disturbed	14.2 km downstream
778-C	Disturbed	20.6 km downstream
070-D	Disturbed	29.5 km downstream

Fish sampling consisted of three-pass depletion electrofishing by 12-15-person crews. Due to extremely low flows observed in the fall of 2011 (avg. 40-50 cfs), backpack electrofishing gear proved sufficient to sample sites 833-REF, 845-A, 835-B, and 778-C. A combination of backpack and barge-mounted electrofishing gear was used in site 070-D, which was characterized by greater depth than the upstream sites. The entire wetted channel was sampled in a downstream to upstream direction over a reach length of 150m at each sample location. All fish were collected, identified to species, recorded, and released. Abundance of each species was summed across electrofishing passes for each sample prior to analysis.

Fish assemblage recovery was evaluated as a function of relative compositional similarity among sites through time with the hypothesis that the disturbed sites would be initially dissimilar to the reference site and become increasingly similar to the reference site through time. I used non-metric multidimensional scaling (NMDS) to extract spatiotemporal patterns in fish assemblage structure. NMDS is an ordination technique that translates the n-dimensional (n= # of taxa) community in relatively few dimensions (usually 2 or 3) so that differences between sites are readily interpreted visually or using simple statistical tests. In a robust NMDS plot, distances between points (e.g. samples, years) on the plot are directly representative of the differences in species composition of communities. I preformed NMDS using R (vegan: Oksanen 2010) on a 39 (sample) by 32 (species) ranked similarity matrix using the Bray-Curtis similarity coefficient, following a 4th root transformation. *Gambusia* was excluded from the analysis based on its extreme variability in abundance (Kubach et al. 2011). I assessed the appropriate dimensionality of the NMDS solution using a scree plot, and guidelines proposed by McCune and Grace (2002). The final stress value obtained from the original data was compared to stress values produced by a Monte-Carlo test with 20 iterations and 10 randomized runs. I examined the stability of the solution to further assess how well the selected model fit the data. I plotted the final solution, and grouped samples within years (sample periods) together in 95% confidence ellipsoids, and displayed reference site sample through years in a separate 95% confidence ellipsoid.

Results

Fish sampling in September 2011 resulted in a collection of 1470 individuals representing 21 species. The catch among all sites was numerically dominated by *Notropis lutipinnis*: n=298, *Nocomis leptocephalus*: n=213, *Lepomis auritus*: n=185, and *Cyprinella chlorista:* n=172. Conservation priority species were represented by *Etheostoma thalassinum* and *Cyprinella chlorista*, and compromised 12.93% of total collections (Kohlsaat 2005).

The following species displayed reduced abundances in 2011 samples compared to previous sampling years: *Micropterus salmoides*, *Lepomis macrochirus*, *Ameiurus catus*, *Ameiurus platycephalus*, and *Ictalurus punctatus*. Additionally, the following species were generally found in greater abundances than in previous sampling years: *Notropis lutipinnis*, *Cyprinella chlorista*, and *Etheostoma thalassinum*. The differences in community composition observed in 2011, as also found in 2008, may reflect changes in habitat conditions due to prolonged drought and resultantly low sustained flow conditions.

A distinct spatiotemporal pattern within the ordination reiterated the recovery patterns found by Kubach et al. (2011). An examination of the scree plot indicated that a 3-dimensional solution provided far greater reductions in stress than later axes. The final stress for the 3-dimensional solution was 9.19, the final instability was 0.0087, and the Monte-Carlo test was significant with $p < 0.0001$ indicating that the solution derived produced stronger axes than expected by chance.

Recovery in the fish assemblage structure was clearly illustrated by the position of impacted sites (grouped by year) in relation to the reference site through time (Figure 1). In both ordination plots (axes 1 v. 2, and axes 1 v.3), samples (grouped by year) are increasingly similar to reference through time, and display less variability through time, with recovery largely achieved by the year 2000 (Kubach et al. 2011). Samples not included in the Kubach et al. (2011) analysis (years 2008, 2011) reiterated this pattern of recovery. However, the ordination plot of axis 1. vs. 2 does show a separation of the 2008 and 2011 samples along the second axis. A correlation of the original species data with axis 2 indicated that these two years differed in composition from other years by having decreased Ictalurid species (*Ameiurus natalis, Ameiurus platycephalus, Ameiurus catus*), and increased cyprinid (shiner) and percid (darter) species (*Cyprinella chlorista, Notropis scepticus, Etheostoma thalassinum*). This shift in species composition may be due to changes in stream habitat resulting from the prolonged drought during this time-period, which resulted in sustained low flows and the loss of deep water habitats generally preferred by riverine Ictalurid species.

<u>Discussion</u>

Previous research (Kubach et al. 2011) indicated that fish assemblage structure of Reedy River sites impacted by the 1996 oil spill largely recovered by 2000 (4.3 years post-disturbance). The incorporation of two additional sampling periods (2008, 2011) reiterated this long-term pattern of recovery, however also showed compositional differences from previous sample years (1996-2005). A NMDS ordination indicated that the 2008 and 2011 sample periods were characterized by the decreased abundance of Ictalurid species, and increased abundances of several shiner and darter species. The observed compositional changes may be due to changes in stream habitat resulting from prolonged drought, resultant low flows, and the loss of deep water habitats preferred by riverine Ictalurid species.

Recommendations

The Reedy River oil spill represents a valuable empirical context from which to address disturbance in aquatic community ecology. Ensuing efforts will be aimed at the completion of a final report to synthesize the body of research produced from studies of both the Reedy River mainstem and a selection of its larger tributary streams.

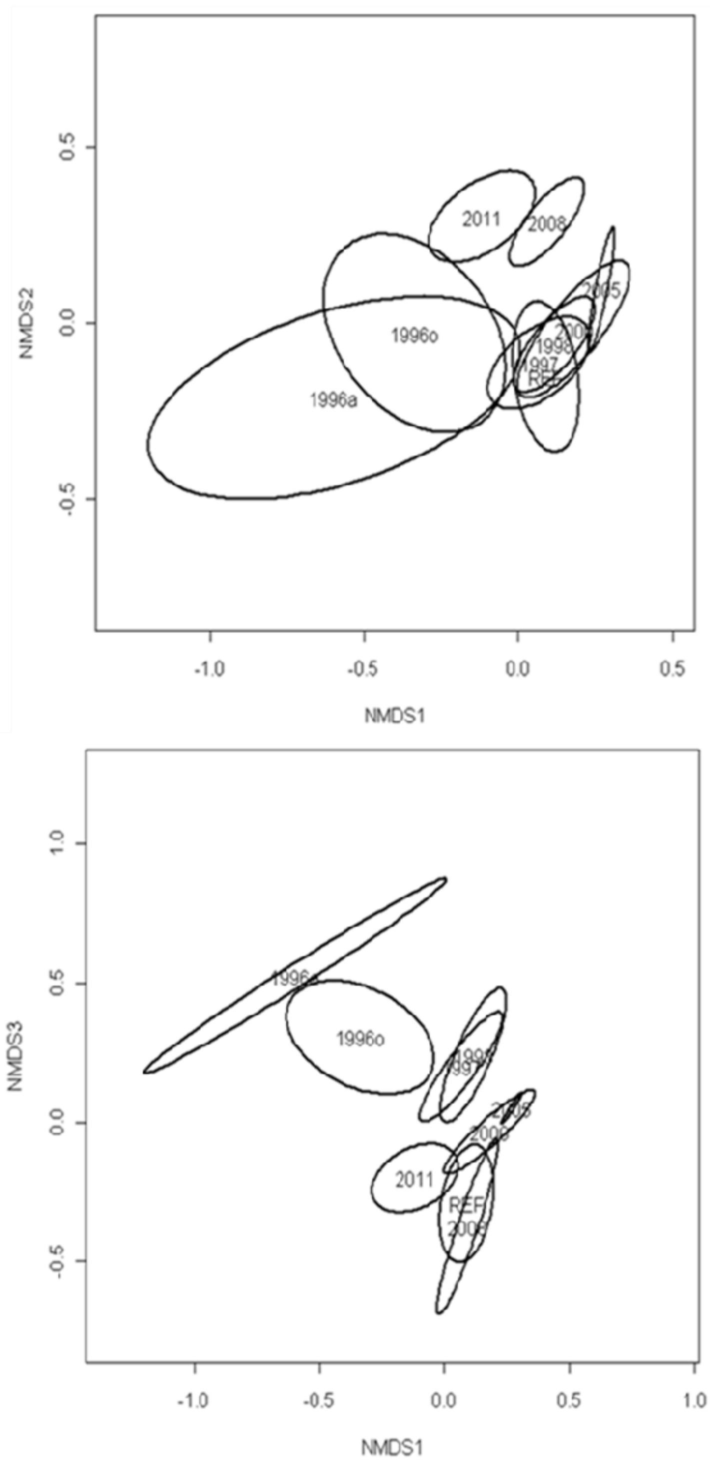

Figure 1. Non-metric multidemsional scaling of fish species by site. Sites are grouped as either reference or sample year within 95% confidence ellipsoids. Ellipsoids that are closer to one another in the ordination are more similar in species composition; overlapping ellipsoids have strongly similar species compositions.

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Prepared By: Cathy Marion Title: Fisheries Research Biologist

Job Title: South Carolina Stream Assessment

Period Covered October 1, 2010 – September 30, 2011

Summary

Sixty-nine randomly selected sites were sampled from 01 October 2010 – 30 September 2011 according to South Carolina Stream Assessment (SCSA) Standard Operating Procedures. Sites were sampled in four river basins as defined in the SCSA: Ashepoo-Combahee-Edisto (ACE; Sand Hills, Atlantic Southern Loam Plains ecoregions), Broad (Outer Piedmont ecoregion), Congaree/Lower Santee (Sand Hills, Atlantic Southern Loam Plains, Carolina Flatwoods ecoregions), and Pee Dee (Slate Belt, Sand Hills, Atlantic Southern Loam Plains, Carolina Flatwoods ecoregions). Data have been added to the SCSA dataset for stream resource analyses and development of decision-support conservation models.

Introduction

The degradation of aquatic ecosystems and subsequent imperilment of native aquatic faunas observed in the southeastern United States underscore the demand for proactive, watershed-based conservation. The South Carolina Stream Assessment (SCSA), a multi-organization effort, was implemented in 2006 to address the need for science-based resource management. The goals are to characterize the biological, physical, and chemical condition of wadeable freshwater streams statewide, and relate these stream indicators to conditions in their watersheds.

Watersheds are distributed according to "ecobasins," spatial strata representing unique combinations of South Carolina's four major river basins and seven level-IV ecoregions, with sample size proportional to ecobasin area. Fixed, annually-sampled reference sites are established within each ecobasin to reflect least-disturbed watersheds and capture temporal dynamics in measured parameters. In addition, 75-100 randomly selected sites are sampled annually for spatial representation of watershed conditions, with statewide coverage scheduled by 2011.

Stream reach-scale biological variables include fish and macroinvertebrate assemblage structure as well as crayfish, mussel, and herpetofaunal distribution. Physical stream habitat is assessed in addition to channel geomorphology and water chemistry. Watershed-scale and riparian indicators are derived from land cover and pollution discharge data, facilitating the development of quantitative models describing the effects of watershed management scenarios on aquatic habitats and biological communities. Ultimately, we hope to provide land planners and managers with an empirically-derived, spatially-explicit decision support framework for watershed and riparian management.

Materials and Methods

Sixty-nine randomly selected sites were sampled from 01 October 2010 – 30 September 2011 according to South Carolina Stream Assessment (SCSA) Standard Operating Procedures (SCDNR 2009; Table 1). Sites were sampled in four river basins as defined in the SCSA: Ashepoo-Combahee-Edisto (ACE; Sand Hills, Atlantic Southern Loam Plains ecoregions), Broad (Outer Piedmont ecoregion), Congaree/Lower Santee (Sand Hills, Atlantic Southern Loam Plains, Carolina Flatwoods ecoregions), and Pee Dee (Slate Belt, Sand Hills, Atlantic Southern Loam Plains, Carolina Flatwoods ecoregions).

Table 1. South Carolina Stream Assessment randomly selected sample sites, 01 October 2010 – 30 September 2011.

River Basin	Ecoregion	Site Number	Sample Date	Stream
ACE	Atlantic S. Loam Plains	251812	19-Apr-2011	Beech Creek
ACE	Atlantic S. Loam Plains	269910	28-Jun-2011	Bull Swamp Creek
ACE	Atlantic S. Loam Plains	277289	21-Jun-2011	Murph Mill Creek
ACE	Atlantic S. Loam Plains	283834	14-Apr-2011	Big Beaver Creek
ACE	Atlantic S. Loam Plains	284551	20-Apr-2011	Big Beaver Creek
ACE	Atlantic S. Loam Plains	285081	22-Jun-2011	Caw Caw Swamp
ACE	Atlantic S. Loam Plains	285099	22-Jun-2011	Saddler Swamp
ACE	Atlantic S. Loam Plains	289921	26-May-2011	Flea Bite Creek
ACE	Atlantic S. Loam Plains	298149	17-Aug-2011	Pond Branch
ACE	Atlantic S. Loam Plains	299102	21-Apr-2011	Little Bull Creek
ACE	Atlantic S. Loam Plains	300478	25-May-2011	Rocky Swamp Creek
ACE	Atlantic S. Loam Plains	304912	28-Jun-2011	Goodland Creek
ACE	Atlantic S. Loam Plains	314722	13-Apr-2011	Whaley Creek
ACE	Atlantic S. Loam Plains	317466	19-Apr-2011	Windy Hill Creek
ACE	Atlantic S. Loam Plains	320715	23-Jun-2011	Roberts Swamp
ACE	Atlantic S. Loam Plains	324816	13-Apr-2011	Little Salkehatchie River
ACE	Atlantic S. Loam Plains	326516	25-May-2011	Little Salkehatchie River
ACE	Atlantic S. Loam Plains	329877	13-Apr-2011	Trib. to Toby Creek
ACE	Atlantic S. Loam Plains	342938	9-Aug-2011	Toby Creek
ACE	Atlantic S. Loam Plains	348611	12-Apr-2011	Parker Branch
ACE	Atlantic S. Loam Plains	351886	24-May-2011	Birds Branch
ACE	Atlantic S. Loam Plains	362227	29-Jun-2011	Miller Swamp
ACE	Atlantic S. Loam Plains	363045	12-Apr-2011	Trib. to Jackson Branch
ACE	Atlantic S. Loam Plains	365040	24-May-2011	Jackson Branch
ACE	Sand Hills	250149	16-Aug-2011	Black Creek
Broad	Outer Piedmont	4877	3-Nov-2010	Buffalo Creek
Broad	Outer Piedmont	5685	18-Aug-2011	South Pacolet River
Broad	Outer Piedmont	118022	12-Oct-2010	Beaver Creek
Broad	Outer Piedmont	148342	12-Oct-2010	Cannons Creek
Broad	Outer Piedmont	155535	12-Oct-2010	Crims Creek
Congaree/Santee	Atlantic S. Loam Plains	239142	5-Oct-2010	Cedar Creek
Congaree/Santee	Atlantic S. Loam Plains	245198	6-Oct-2010	Sandy Run Creek
Congaree/Santee	Atlantic S. Loam Plains	256365	4-Oct-2010	Buckhead Creek
Congaree/Santee	Carolina Flatwoods	349446	21-Jul-2011	Wambaw Creek
Congaree/Santee	Sand Hills	206029	5-Oct-2010	Gills Creek
Congaree/Santee	Sand Hills	255769	4-Oct-2010	Big Beaver Creek
Pee Dee	Atlantic S. Loam Plains	124431	14-Jul-2011	Horse Creek
Pee Dee	Atlantic S. Loam Plains	157374	20-Jul-2011	High Hill Creek
Pee Dee	Atlantic S. Loam Plains	159918	10-Aug-2011	Catfish Canal
Pee Dee	Atlantic S. Loam Plains	161334	31-Aug-2011	Jordan Creek
Pee Dee	Atlantic S. Loam Plains	162435	20-Jul-2011	Boggy Gully Swamp
Pee Dee	Atlantic S. Loam Plains	186538	10-Aug-2011	Willow Creek
Pee Dee	Atlantic S. Loam Plains	198278	11-Aug-2011	Rocky Bluff Swamp
Pee Dee	Carolina Flatwoods	203569	31-Aug-2011	Trib. to Big Swamp

River Basin	Ecoregion	Site Number	Sample Date	Stream
Pee Dee	Carolina Flatwoods	216403	30-Aug-2011	Hope Swamp
Pee Dee	Carolina Flatwoods	242601	30-Aug-2011	Long Branch
Pee Dee	Carolina Flatwoods	314075	19-Jul-2011	Boggy Swamp
Pee Dee	Sand Hills	46339	13-Jul-2011	Whites Creek
Pee Dee	Sand Hills	60805	16-Mar-2011	Little Fork Creek
Pee Dee	Sand Hills	62904	8-Jun-2011	Huckleberry Creek
Pee Dee	Sand Hills	69005	4-May-2011	Big Bear Creek
Pee Dee	Sand Hills	72138	4-May-2011	Fork Creek
Pee Dee	Sand Hills	76767	17-Mar-2011	Bay Branch
Pee Dee	Sand Hills	77498	8-Jun-2011	Black Creek
Pee Dee	Sand Hills	88961	18-May-2011	Rocky Creek
Pee Dee	Sand Hills	89665	16-Mar-2011	Trib. to Juniper Creek
Pee Dee	Sand Hills	91052	12-Jul-2011	Buffalo Creek
Pee Dee	Sand Hills	96049	3-May-2011	Big Sandy Creek
Pee Dee	Sand Hills	99509	16-Mar-2011	South Buffalo Creek
Pee Dee	Sand Hills	101902	7-Jun-2011	Little Lynches River
Pee Dee	Sand Hills	117754	15-Mar-2011	Mills Creek
Pee Dee	Sand Hills	122370	19-May-2011	Beaverdam Creek
Pee Dee	Sand Hills	128722	9-Jun-2011	Beaverdam Creek
Pee Dee	Sand Hills	155952	15-Mar-2011	Trib. to Nancy Branch
Pee Dee	Sand Hills	228241	3-May-2011	Trib. to Hatchet Camp Branch
Pee Dee	Slate Belt	55326	5-May-2011	North Branch Wildcat Creek
Pee Dee	Slate Belt	86893	24-Mar-2011	Little Lynches River
Pee Dee	Slate Belt	94450	24-Mar-2011	Little Lynches River
Pee Dee	Slate Belt	97604	18-May-2011	Hanging Rock Creek

Results

Ashepoo-Combahee-Edisto (ACE) River Basin (Atlantic Southern Loam Plains ecoregion)

Forty-seven fish species including eight Priority species (Kohlsaat et al. 2005) were collected altogether from 24 randomly selected sites in the ACE basin / Atlantic Southern Loam Plains ecoregion (Table 2). Fish species richness among sites averaged 15.9 (range 7 – 25). Sites exhibiting the highest fish species richness were Birds Branch (25 species; Site No. 351886), Caw Caw Swamp (22 species; Site No. 285081; also a regional reference site), Goodland Creek (22 species; Site No. 304912) and Little Salkehatchie River (20 species; Site No. 326516). The most frequently encountered species by number of sites occupied were pirate perch, redfin pickerel and tessellated darter (Table 2). By total abundance, the top three species collected were dusky shiner, lowland shiner and pirate perch. Overall relative abundance of Priority species was 16.8%.

The turquoise darter (*Etheostoma inscriptum*) was collected at two sites in the ecobasin. This ecobasin represents the extreme southeastern limit of the known range of *E. inscriptum*.

Table 2. Fish species collected from SCSA random sample sites in the Ashepoo-Combahee-Edisto basin / Atlantic Southern Loam Plains (01 October 2010 – 30 September 2011) and Conservation Priority according to Kohlsaat et al. (2005). Site occupancy values are out of a possible 24 sites sampled. Continued on following page.

Scientific Name	Common Name	Conservation Priority	n Sites Occupied	% Sites Occupied	Relative Abundance
Chologaster cornuta	Swampfish		1	4.2%	0.04%
Anguilla rostrata	**American eel**	**Highest**	18	75.0%	1.52%
Aphredoderus sayanus	Pirate perch		24	100.0%	7.98%
Labidesthes sicculus	Brook silverside		1	4.2%	0.03%
Erimyzon oblongus	Creek chubsucker		17	70.8%	1.89%
Erimyzon sucetta	Lake chubsucker		2	8.3%	0.17%
Minytrema melanops	Spotted sucker		6	25.0%	0.49%
Acantharchus pomotis	**Mud sunfish**	**Moderate**	13	54.2%	0.39%
Centrarchus macropterus	Flier		2	8.3%	0.11%
Enneacanthus gloriosus	Bluespotted sunfish		4	16.7%	0.10%
Lepomis auritus	Redbreast sunfish		19	79.2%	7.05%
Lepomis gibbosus	Pumpkinseed		2	8.3%	0.11%
Lepomis gulosus	Warmouth		10	41.7%	0.70%
Lepomis macrochirus	Bluegill		12	50.0%	3.13%
Lepomis marginatus	Dollar sunfish		16	66.7%	4.84%
Lepomis microlophus	Redear sunfish		2	8.3%	0.06%
Lepomis punctatus	Spotted sunfish		18	75.0%	3.96%
Micropterus salmoides	Largemouth bass		9	37.5%	0.33%
Nocomis leptocephalus	Bluehead chub		4	16.7%	2.42%
Notemigonus crysoleucas	Golden shiner		2	8.3%	0.14%
Notropis chalybaeus	Ironcolor shiner		3	12.5%	1.32%
Notropis cummingsae	Dusky shiner		14	58.3%	21.95%
Notropis lutipinnis	Yellowfin shiner		6	25.0%	3.17%
Notropis maculatus	Taillight shiner		1	4.2%	0.01%
Notropis petersoni	Coastal shiner		6	25.0%	0.36%
Opsopoeodus emiliae	**Pugnose minnow**	**Moderate**	3	12.5%	0.13%
Pteronotropis stonei	**Lowland shiner**	**Moderate**	16	66.7%	13.83%
Semotilus atromaculatus	Creek chub		1	4.2%	0.04%
Elassoma zonatum	Banded pygmy sunfish		5	20.8%	0.36%
Esox americanus	Redfin pickerel		22	91.7%	4.74%
Esox niger	Chain pickerel		9	37.5%	0.25%
Fundulus lineolatus	Lined topminnow		2	8.3%	0.08%

Scientific Name	Common Name	Conservation Priority	n Sites Occupied	% Sites Occupied	Relative Abundance
Ameiurus brunneus	**Snail bullhead**	**Moderate**	7	29.2%	0.14%
Ameiurus natalis	Yellow bullhead		14	58.3%	0.89%
Ameiurus nebulosus	Brown bullhead		1	4.2%	0.01%
Ameiurus platycephalus	**Flat bullhead**	**Moderate**	1	4.2%	0.06%
Noturus gyrinus	Tadpole madtom		4	16.7%	0.42%
Noturus insignis	Margined madtom		6	25.0%	0.44%
Noturus leptacanthus	Speckled madtom		11	45.8%	1.65%
Etheostoma fricksium	**Savannah darter**	**Highest**	9	37.5%	0.71%
Etheostoma fusiforme	Swamp darter		1	4.2%	0.01%
Etheostoma inscriptum	**Turquoise darter**	**High**	2	8.3%	0.07%
Etheostoma olmstedi	Tessellated darter		20	83.3%	4.63%
Etheostoma serrifer	Sawcheek darter		3	12.5%	0.11%
Percina nigrofasciata	Blackbanded darter		14	58.3%	2.14%
Gambusia holbrooki	Eastern mosquitofish		18	75.0%	7.01%
Umbra pygmaea	Eastern mudminnow		1	4.2%	0.01%

Pee Dee River Basin (Sand Hills)

Altogether, 46 fish species including 11 Priority species (Kohlsaat et al. 2005) were collected from 18 randomly selected sites in the Pee Dee basin / Sand Hills ecoregion (Table 3). Mean fish species richness was 11.2 (range 1 – 25). The most species-rich sites were Little Lynches River (25 species; Site No. 101902), Rocky Creek (21 species; Site No. 88961) and Huckleberry Creek (20 species; Site No. 62904). Species occurring at the most sites were pirate perch, redfin pickerel, dusky shiner, margined madtom and yellow bullhead (Table 3). The most abundant species were dusky shiner, bluehead chub and tessellated darter. Priority species represented 18.4% by abundance of all fishes collected.

The Sandhills chub (*Semotilus lumbee*), a narrowly distributed species of Highest Priority, was collected at 22% of sites in this ecobasin, which includes the majority of its limited range. Outside of the Pee Dee / Sand Hills ecobasin, *S. lumbee* was collected at only one other randomly selected site in the SCSA, in the adjacent portion of the Catawba/Wateree (Santee) basin in 2009.

Sites supporting *S. lumbee* were primarily in small-medium headwater stream reaches, although fish species richness ranged from 8 – 15 at these sites.

Non-native green sunfish (*Lepomis cyanellus*) were collected at two sites (11%) in this ecobasin, increasing the known range of this potentially invasive species in the Pee Dee River basin. *L. cyanellus* has recently been documented from sites in all ecoregions of the Pee Dee basin in South Carolina (K. Kubach annual reports, 2007 and 2009). Potential impacts of *L. cyanellus* on native species include competition, predation and hybridization (Rohde et al. 2009); introduced populations warrant monitoring.

Table 3. Fish species collected from SCSA random sample sites in the Pee Dee basin / Sand Hills (01 October 2010 – 30 September 2011) and Conservation Priority according to Kohlsaat et al. (2005). Site occupancy values are out of a possible 18 sites sampled. Continued on following page.

Scientific Name	Common Name	Conservation Priority	n Sites Occupied	% Sites Occupied	Relative Abundance
Chologaster cornuta	Swampfish		1	5.6%	0.04%
Anguilla rostrata	**American eel**	**Highest**	3	16.7%	0.60%
Aphredoderus sayanus	Pirate perch		13	72.2%	5.49%
Erimyzon oblongus	Creek chubsucker		9	50.0%	2.00%
Erimyzon sucetta	Lake chubsucker		2	11.1%	0.34%
Moxostoma collapsum	**Notchlip redhorse**	**Moderate**	1	5.6%	0.09%
Scartomyzon sp.	Brassy jumprock		1	5.6%	0.04%
Acantharchus pomotis	**Mud sunfish**	**Moderate**	7	38.9%	0.85%
Centrarchus macropterus	Flier		4	22.2%	0.72%
Enneacanthus gloriosus	Bluespotted sunfish		7	38.9%	0.85%
Lepomis auritus	Redbreast sunfish		8	44.4%	7.07%
Lepomis cyanellus	Green sunfish		2	11.1%	0.17%
Lepomis gibbosus	Pumpkinseed		2	11.1%	0.34%
Lepomis gulosus	Warmouth		9	50.0%	0.77%
Lepomis macrochirus	Bluegill		7	38.9%	4.51%
Lepomis marginatus	Dollar sunfish		8	44.4%	2.30%
Lepomis microlophus	Redear sunfish		2	11.1%	0.17%
Micropterus salmoides	Largemouth bass		3	16.7%	0.26%
Pomoxis nigromaculatus	Black crappie		1	5.6%	0.09%
Cyprinella chloristia	**Greenfin shiner**	**Moderate**	1	5.6%	4.56%
Cyprinella pyrrhomelas	**Fieryblack shiner**	**Moderate**	1	5.6%	0.94%
Hybognathus regius	Eastern silvery minnow		1	5.6%	1.36%
Nocomis leptocephalus	Bluehead chub		4	22.2%	7.96%

Scientific Name	Common Name	Conservation Priority	n Sites Occupied	% Sites Occupied	Relative Abundance
Notemigonus crysoleucas	Golden shiner		4	22.2%	0.94%
Notropis altipinnis	Highfin shiner		2	11.1%	3.36%
Notropis chlorocephalus	**Greenhead shiner**	**High**	1	5.6%	3.41%
Notropis cummingsae	Dusky shiner		12	66.7%	11.37%
Notropis petersoni	Coastal shiner		1	5.6%	0.60%
Notropis procne	Swallowtail shiner		1	5.6%	2.77%
Notropis scepticus	Sandbar shiner		1	5.6%	0.68%
Pteronotropis stonei	**Lowland shiner**	**Moderate**	3	16.7%	2.98%
Semotilus lumbee	**Sandhills chub**	**Highest**	4	22.2%	1.79%
Esox americanus	Redfin pickerel		13	72.2%	3.41%
Esox niger	Chain pickerel		8	44.4%	0.77%
Ameiurus brunneus	**Snail bullhead**	**Moderate**	2	11.1%	0.09%
Ameiurus melas	Black bullhead		1	5.6%	0.09%
Ameiurus natalis	Yellow bullhead		10	55.6%	1.62%
Ameiurus platycephalus	**Flat bullhead**	**Moderate**	2	11.1%	0.26%
Noturus gyrinus	Tadpole madtom		3	16.7%	0.21%
Noturus insignis	Margined madtom		12	66.7%	7.11%
Etheostoma fusiforme	Swamp darter		1	5.6%	0.13%
Etheostoma olmstedi	Tessellated darter		8	44.4%	7.83%
Etheostoma serrifer	Sawcheek darter		4	22.2%	0.21%
Percina crassa	**Piedmont darter**	**High**	3	16.7%	2.85%
Gambusia holbrooki	Eastern mosquitofish		7	38.9%	0.98%
Umbra pygmaea	Eastern mudminnow		2	11.1%	5.07%

Pee Dee River Basin (Slate Belt)

Thirty-three fish species including five Priority species (Kohlsaat et al. 2005) were collected altogether from four randomly selected sites in the Pee Dee basin / Slate Belt ecoregion (Table 4). On average, 17 fish species were collected per site in this ecobasin (range 10 – 25). Sites producing highest species richness were Little Lynches River (25 species at Site No. 94450 and 14 species at Site No. 86893) and North Branch Wildcat Creek (19 species; Site No. 55326). Four species occurred at all four sample sites: bluehead chub, creek chub, redbreast sunfish and tessellated darter (Table 4). By total abundance, the top three species collected were tessellated darter, highfin shiner and bluehead chub. Overall relative abundance of Priority species was 3.3%.

Striped jumprock (*Scartomyzon rupiscartes*) were collected at two sites in the Lynches River drainage, likely representing the first confirmed records for this species in the Pee Dee basin in South Carolina based on Rohde et al. (2009). A voucher specimen from one of these sites exhibited the following characteristics, confirming *S. rupiscartes* (Rohde et al. 2009): dorsal rays = 12; gill rakers on first arch = 23 – 25; dark stripes on body wider than light stripes. Prior reports of *S. rupiscartes* from the Lynches River drainage were determined to be misidentifications (Rohde et al. 2009). The similar brassy jumprock (*Scartomyzon sp.*) was also collected at two sites, at one of which it was sympatric with *S. rupiscartes*.

The greenhead shiner (*Notropis chlorocephalus*) was collected at a relatively low proportion of sites in the Pee Dee / Slate Belt (25%, or one site) and Pee Dee / Sand Hills (5.6%), which mark the approximate eastern limit of its range in South Carolina. It appears to be less common and abundant, even in similar habitats, than the closely related *N. lutipinnis* of other river basins to the west. The apparently low occurrence and abundance of *N. chlorocephalus* in these ecobasins, however, may in part reflect their location at the periphery of the range of this species.

Non-native green sunfish (*Lepomis cyanellus*) were present at three of four sites in this ecobasin, suggesting establishment in this portion of the Pee Dee basin (Lynches River drainage).

Table 4. Fish species collected from SCSA random sample sites in the Pee Dee basin / Slate Belt (01 October 2010 – 30 September 2011) and Conservation Priority according to Kohlsaat et al. (2005). Site occupancy values are out of a possible 4 sites sampled.

Scientific Name	Common Name	Conservation Priority	n Sites Occupied	% Sites Occupied	Relative Abundance
Aphredoderus sayanus	Pirate perch		3	75.0%	3.17%
Erimyzon oblongus	Creek chubsucker		3	75.0%	2.13%
Scartomyzon rupiscartes	Striped jumprock		2	50.0%	0.16%
Scartomyzon sp.	Brassy jumprock		2	50.0%	0.60%
Lepomis auritus	Redbreast sunfish		4	100.0%	3.55%
Lepomis cyanellus	Green sunfish		3	75.0%	2.08%
Lepomis gibbosus	Pumpkinseed		2	50.0%	0.55%
Lepomis gulosus	Warmouth		2	50.0%	0.33%
Lepomis macrochirus	Bluegill		3	75.0%	1.64%
Lepomis microlophus	Redear sunfish		1	25.0%	0.11%
Micropterus salmoides	Largemouth bass		1	25.0%	0.05%
Clinostomus funduloides	Rosyside dace		3	75.0%	2.46%
Cyprinella chloristia	**Greenfin shiner**	**Moderate**	1	25.0%	0.16%
Nocomis leptocephalus	Bluehead chub		4	100.0%	14.43%
Notemigonus crysoleucas	Golden shiner		1	25.0%	0.11%
Notropis altipinnis	Highfin shiner		3	75.0%	20.23%
Notropis chlorocephalus	**Greenhead shiner**	**High**	1	25.0%	1.42%
Notropis cummingsae	Dusky shiner		1	25.0%	0.05%
Notropis petersoni	Coastal shiner		1	25.0%	0.27%
Notropis procne	Swallowtail shiner		2	50.0%	4.37%
Notropis scepticus	Sandbar shiner		1	25.0%	0.22%
Semotilus atromaculatus	Creek chub		4	100.0%	2.02%
Esox americanus	Redfin pickerel		2	50.0%	0.55%
Esox niger	Chain pickerel		1	25.0%	0.05%
Ameiurus brunneus	**Snail bullhead**	**Moderate**	2	50.0%	0.33%
Ameiurus natalis	Yellow bullhead		1	25.0%	0.16%
Ameiurus platycephalus	**Flat bullhead**	**Moderate**	2	50.0%	0.98%
Noturus gyrinus	Tadpole madtom		2	50.0%	0.16%
Noturus insignis	Margined madtom		3	75.0%	2.84%
Etheostoma olmstedi	Tessellated darter		4	100.0%	34.17%
Percina crassa	**Piedmont darter**	**High**	1	25.0%	0.44%
Gambusia holbrooki	Eastern mosquitofish		1	25.0%	0.11%
Umbra pygmaea	Eastern mudminnow		1	25.0%	0.05%

Other Ecobasins

In addition to scheduled ecobasins, sites in seven other ecobasins were sampled during this reporting period: ACE basin: Sand Hills (1 site); Broad basin: Outer Piedmont (5); Congaree/Lower Santee basin: Sand Hills (2), Atl. S. Loam Plains (3) and Carolina Flatwoods (1); Pee Dee basin: Atl. S. Loam Plains (7) and Carolina Flatwoods (4; Table 1). These sites could not be sampled in previous years due to logistical or environmental constraints. The majority of sites from these ecobasins are presented in previous annual reports (2006 – 2010) and all data will be included in final reports.

Discussion

Blackbanded Sunfish (Enneacanthus chaetodon) Population Status

Efforts are currently underway to assess the population status of the blackbanded sunfish (*Enneacanthus chaetodon*) throughout its range, in portions of which it is known to be declining or imperiled. The South Carolina Stream Assessment (SCSA) employs random sampling of wadeable streams, providing a means of quantifying species abundances at several spatial scales and measuring rarity. SCSA sampling in 2011 included 55 sites within the known range of *E. chaetodon* (Sand Hills, Atlantic Southern Loam Plains, and Carolina Flatwoods ecoregions); however, *E. chaetodon* was not collected at any of these sites. Through 2011, the total number of *E. chaetodon* collected at SCSA randomly selected sites remains low, at 17 individuals from 5 of 230 sites sampled (2.2%) within its potential range in the coastal plain. However, the apparently low presence and abundance of *E. chaetodon* at SCSA sites may in part reflect sampling selectivity towards wadeable, channel-constrained streams (i.e. those effectively sampled using backpack electrofishing). Historic data suggest *E. chaetodon* may be more abundant in wider and deeper habitats (e.g. swamps) that are not currently sampled as part of the SCSA and thus further evaluation of these habitats is necessary to fully assess the population status of this and other species with similar habitat requirements.

Recommendations

This report summarizes SCSA sampling of randomly selected sites from 01 October 2010 – 30 September 2011. Forthcoming final reports will focus on standardized estimation of stream resources (summarized by river basin and ecoregion strata), including development of conservation criteria for South Carolina stream fishes based on standardized abundance estimates and other measures. These criteria will assist biologists and resource managers in assigning conservation status in future efforts such as revisions of the Comprehensive Wildlife Conservation Strategy.

Additional analyses aim to develop watershed-scale models of land use change on stream resources (physical, chemical and biological) for conservation and decision-support applications.

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Prepared By: Kevin Kubach Title: Fisheries Biologist

Job Title: Trophic resources for larval fish in Lake Marion

Period Covered July 1, 2010 – June 30, 2011

Results and Discussion

The work reported here is part of an ongoing program of studies directed toward developing process-based models of food resources and other factors that may limit recruitment of key resident and anadromous fish species in the Santee-Cooper system,

In 2008, the South Carolina Department of Natural Resources (SCDNR) re-convened the Santee-Cooper Comprehensive Study Group to provide an update and overview of current conditions in the system and to guide and promote development of a scientific basis for management decisions about aquatic resources within the Santee-Cooper basin. The most critical short-term goal identified by the Study Group was to evaluate whether zooplankton abundance may limit the recruitment of key fish species, including striped bass, American shad, blueback herring, threadfin shad, and white perch.

These key species have overlapping spawning seasons (April to June), shared nursery areas in Upper Lake Marion, and similar preferences for zooplankton during early life stages. During recent years, blueback herring and striped bass recruitment dropped to historically low levels (Lamprecht, S., SCDNR, personal communication). The causes for these declines, and the implications for success of the Santee-Cooper anadromous fish passage and restoration efforts, are presently unknown. To date, striped bass has received more attention than the other key species in Lake Marion. However, because all of the key species share habitat and resources during early development, reduced recruitment of striped bass probably indicates changing conditions for the other species.

Investigations of factors influencing successful striped bass recruitment were conducted in Santee-Cooper in the 1980s and early 1990s. Successful recruitment depends on the abundance and timing of zooplankton production. Striped bass appear to require zooplankton densities on the order of 100 animals/liter or more (Bulak et al., 1997).

For Lake Marion, the most important controls on zooplankton abundance in spring are probably intensity of predation, adequacy of phytoplankton, and losses resulting from the high flushing rate of water through the system. Because spring temperatures are fairly consistent between years, they are unlikely to produce great differences in zooplankton abundances. Feeding by the larval fish could suppress zooplankton abundances, and larval fish may compete for this resource (for example, the hypothesized interaction between anadromous American shad and salmon in the Columbia River; Fresh, 1996). The benthos may also affect the plankton in Lake Marion. *Corbicula fluminea*, the invasive Asiatic clam, is abundant. *Corbicula* can be highly productive (Sousa et al., 2008). *Corbicula* spp. have greatly suppressed phytoplankton and phytoplankton in other shallow systems (for example, Hwang et al., 2004; Lopez et al., 2006), causing major changes in trophic structure.

Our work during this reporting period included: 1) processing and analyzing benthic samples from Middle and Lower Lake Marion; 2) estimating abundances and birth rates of zooplankton in Upper Lake Marion; and 3) constructing and using a model to estimate the impact of flushing rate on spring plankton populations in Upper Lake Marion.

Benthic samples from Middle and Lower Lake Marion

We collected 25 benthic samples on 5 transects from Middle Lake Marion in June 2010 and 17 benthic samples on 5 transects from Lower Lake Marion in June-July 2010. Methods were similar to previously reported sampling in Upper Lake Marion in June-July 2009 and April 2010.

Stations were accessed by airboat. Water depth was determined using Sonar, and geographic coordinates were recorded with a hand-held GPS unit. As with our previous benthic sampling, collaboration with Santee Cooper greatly facilitated the process.

Two samples were taken at each station with a Petite Ponar Bottom Grab sampler (152 mm by 152 mm). One sample was placed on a 0.5-mm stainless steel screen and gently rinsed with water pumped from the lake. Material retained on the screen was preserved in 70% alcohol. Sediment was collected from the other sample for analysis of texture and organic carbon content.

Sediment texture and organic carbon content samples were processed by Santee Cooper Analytical and Biological Services. Particles <500 microns were analyzed on a Cilas 930 Laser Particle Size Analyzer; samples were sieved to remove particles >500 microns before processing. Particles >500 microns were analyzed by screening dry sediment with a 500-micron sieve, then weighing both fractions. Two replicates were run for each of the textural analyses. Samples for organic carbon content were dried and ground with a mortar and pestle before analysis.

In the laboratory, benthos samples was rinsed on a 0.5-mm Nitex sieve to remove sediment and preservative, then placed in a metal tray for sorting. All visible invertebrates were removed. Most were grouped in 5-mm size classes for counting. Prior simulations with plausible length distributions indicated that this grouping would have little impact on the accuracy of the biomass estimates (<5% difference from estimates based on 1-mm size classes).

We identified common benthic invertebrates to species, rarer taxa to family or higher level using standard keys and additional reference materials. Dr. Robert Dillon of College of Charleston consulted on gastropod identifications, and Mr. David Eargle of South Carolina Department of Health and Environmental Control consulted on bivalve and mayfly identifications.

Biomasses (dry weight) were estimated using regressions from Benke et al. (1999) for *Corbicula fluminea* (Lauritzen and Mozley's summer equation for a population in North Carolina) and *Hexagenia limbata* (Smock's equation for *H. munda* in North Carolina) and a function fit to data for *Viviparus subpurpureus* from Richardson and Brown (1989). The equation for *Corbicula fluminea* was also used for the sphaeriids, which are similar in form to small *Corbicula*. An average biomass of 0.1 mg was used for *Chaoborus punctipennis* (Taylor, unpublished data for mainly 4th instar larvae of *Chaoborus punctipennis* from Pond 4 on the Savannah River Site in South Carolina). The same value was also used for the chironomids, which were similar in size.

The mean depth of benthic samples increased from 3.4 m in Upper Lake Marion to 7 m in Lower Marion (Table 1). Silts (means of 75% to 77%) and clays (means of 9% to 15%) dominated sediment composition in all three regions. Mean organic carbon content of the sediment was near 2.5% for all three regions.

The abundance and biomass of the benthos decreased from downlake from Upper Lake Marion to Lower Lake Marion (Table 2, Figures 1 and 2). As in Upper Lake Marion, the benthos of Middle and Lower Lake Marion was dominated by the Asiatic clam *Corbicula fluminea* and the olive mystery snail *Viviparus subpurpureus*. The mayfly *Hexagenia limbata* ranked distant third in biomass in Middle Lake Marion, as in Upper Lake Marion, but was absent from Lower Lake Marion. Only dipteran larvae (Chironomidae and Chaoboridae) had similar abundances across regions. Overall, biomass decreased by about half from Upper to Middle Lake Marion, by about two-thirds from Middle to Lower Lake Marion.

Zooplankton samples from Upper Lake Marion

We processed quantitative zooplankton samples collected at six stations on five dates in April-June 2009. Most of the samples were dominated by rotifers; copepods and cladocerans were sparse (Table 3). The most common rotifers were *Synchaeta* (most dates), *Polyarthra*, *Conochilus*, *Keratella*, and *Filinia*. The most common cladocerans were *Bosmina* (all dates) and *Ceriodaphnia*. Cyclopoids, including *Mesocyclops edax*, predominated among the copepods. Other taxa included dipteran larvae, water mites, and worms. Abundances in April and early May were well below the concentration (100 animals/liter) required by striped bass larvae (Bulak et al., 1997).

We compared these results with five previous studies in Upper Lake Marion and its tributaries in 1980s and 1990s. Abundances of major taxa fell generally within the range of variation reported in the four previous studies in the 1980s and 1990s. To the extent that reliable comparisons could be made, composition showed little change. The invasive cladoceran *Daphnia lumholtzi*, which is morphologically conspicuous, was not reported in the 1980s and 1990s. It is present at low abundance in some 2011 samples (see below).

Because counts of egg-carrying zooplankton were low, the 2009 data proved to be only marginally suitable for birth rate and production estimates. The estimates, generally appropriate for cladocerans and rotifers, are based on a measure of population structure--the ratio of eggs to animals--and a taxon-specific estimate of egg development time based on temperature. The few birth rate estimates we were able to make with these data showed high variation among samples, which may have been due either to real differences in conditions among the stations or to sampling error (see "Field estimates" in Figure 3). Birth rates were not estimated in any previous studies at Lake Marion.

In June 2011, we began to collect additional zooplankton samples in conjunction with night sampling of juvenile fish in mid- and Upper Lake Marion. The new samples will provide better estimates of species composition and birth rates. The new samples also suggest that diel vertical migration of larger zooplankton may be substantial, raising the possibility that samples taken during daytime, including the spring 2009 samples and nearly all of the samples from the 1980s and 1990s, have underestimated abundances of larger zooplankton, such as late copepodid and adult stages of the cycloid and calanoid copepods.

Model to estimate the impact of flushing rate on spring zooplankton in Upper Lake Marion

We built a model to evaluate the potential impact of loss rates due to flushing on zooplankton populations in Upper Lake Marion using a water temperature function, a hydrologic model, the ranges of egg ratios observed in the spring 2009 zooplankton samples, and published functions for the temperature-dependencies of zooplankton egg development times.

The water temperature function was fitted to surface water temperature data from Upper Lake Marion stations SC-010 and SC-015 for 1983-2009 from the EPA STORET database.

The main nursery area for striped bass larvae in Upper Lake Marion extends roughly from mile marker 150 to I-95. We estimated daily inflow rates for this segment for 1983-2010, using daily mean discharge data from USGS stations in the Wateree and Congaree Rivers. We estimated the volume for the section using GPS data obtained in conjunction with the benthic sampling. We then computed retention times and flushing rates for this segment, adjusting its volume according to daily water level data for Lake Marion from USGS.

We illustrate the comparison between zooplankton birth rates and loss rates due to flushing for an annual cycle based on data for 2009 (Figure 3). The overlapping ranges for birth rates and loss rates during spring suggest that flushing may limit growth of zooplankton populations in spring, with more severe impacts on cladocerans than on rotifers.

Recommendations

Continue to develop a process-oriented, modeling framework to allow continued refinement of a system-based ecological model, as more data are obtained and lake processes continue to change. Specific management applications resulting from this effort may include predicting optimal levels and times for striped bass stocking.

Table 1. Sample depth and characteristics of benthic sediments from Upper, Middle, and Lower Lake Marion.

	Upper (n=50)		***Middle (n=25)***		***Lower (n=17)***	
	Mean	***Range***	***Mean***	***Range***	***Mean***	***Range***
Depth (m)	3.4	(1.2, 5.1)	5.2	(1.2, 8.5)	7.0	(1.2, 12.8)
Clay	9%	(1%, 15%)	15%	(7%, 56%)	11%	(3%, 16%
Silt	75%	(6%, 90%)	77%	(36%, 89%)	77%	(22%, 92%)
Very fine sand	8%	(0%, 36%)	3%	(0%, 19%)	4%	(0%, 19%)
Fine sand	6%	(0%, 38%)	2%	(0%, 19%)	3%	(0%, 22%)
Medium sand	2%	(0%, 46%)	1%	0%, 12%)	2%	(0%, 18%)
Coarse sand	1%	(0%, 19%)	1%	(0%, 17%)	4%	(0%, 44%)
Total organic carbon	2.4%	(0.0%, 6.0%)	2.5%	(0.1%, 4.2%)	2.5%	(0.1%, 4.4%)

Table 2. Benthic invertebrates in Upper, Middle, and Lower Lake Marion. Biomasses were not estimated for some sparsely abundant or small taxa. Upper Lake Marion results, which have been reported previously, are included for comparison.

		Upper Lake Marion June-July 2009 (n=50)		***Middle Lake Marion June 2010 (n=25)***		***Lower Lake Marion July 2010 (n=17)***	
Taxon	***Size range (mm)***	***Abundance (#/m^2)***	***Biomass (g/m^2)***	***Abundance (#/m^2)***	***Biomass (g/m^2)***	***Abundance (#/m^2)***	***Biomass (g/m^2)***
ALL BIVALVES		**551**	**48.3**	**248**	**25.0**	**196**	**8.4**
Corbiculidae	5-40	421	48.1	232	25.0	117	8.3
Sphaeriidae[1]	<5-15	120	0.2	14	0.0	76	0.1
Unionidae: *Elliptio* spp[2]	5-110	9		2		3	
Lampsilis splendida/radiata	50-60	2		0		0	
ALL GASTROPODS		**328**	**17.7**	**241**	**8.0**	**84**	**2.3**
Physidae	<5-10	3		0		10	
Planorbidae	<5	1		0		0	
Valvatidae: *Valvata bicarinata*	<5	5		0		0	
Viviparidae: *Viviparus subpurpureus*	<5-30	320	17.7	241	8.0	74	2.3
ALL INSECTS		**569**	**2.8**	**520**	**0.7**	**326**	**0.0**
Coleoptera: Dytiscidae	10-15	1		0		0	
Elmidae	5-10	1		0		0	
Diptera: Chaoboridae: *Chaoborus punctipennis*		120	0.0	213	0.0	132	0.0
Chironomidae		275	0.0	177	0.0	186	0.0
Ephemeroptera: Caenidae: *Caenis*		7		0		0	
Ephemeridae: *Hexagenia limbata*	5-30	158	2.8	130	0.6	0	0.0
Odonata: Gomphidae		3		0		0	
Trichoptera	5-10	3		0		8	
ALL CRUSTACEANS		**12**		**2**		**64**	
Amphipoda: Gammaridae: *Gammarus*		10		2		3	
Talitridae: *Hyalella*		0		0		59	
Copepoda: Cyclopoida		1		0		0	
Isopoda		1		0		0	
Ostracoda		0		0		3	
TOTAL		**1,460**	**68.8**	**1,010**	**33.7**	**670**	**10.7**

[1]Includes sphaeriids *Eupera cubensis*, *Pisidium* sp., *Sphaerium/Musculium* sp., and, in Upper and Middle Lake Marion samples only, *Corbicula* <5 mm

[2]Includes forms resembling *E. producta*, *E. fisheriana*, and *E. folliculata/angustata*

Table 3. Abundances of zooplankton in Upper Lake Marion. Values for six stations are summarized for each sampling date.

		Mean and range (animals liter^{-1})									
Group		***10-Apr-09***		***23-Apr-09***		***07-May-09***		***21-May-09***		***04-Jun-09***	
Rotifera		3.8	(1.0, 14.9)	23.2	(2.4, 73.3)	30.6	(10.5, 58.9)	67.5	(5.9, 151.6)	167.2	(11.8, 497.1)
Copepoda	**nauplii**	3.0	(0.7, 9.5)	5.5	(2.0, 20.1)	11.2	(3.5, 25.8)	10.3	(1.3, 45.2)	4.5	(1.1, 11.4)
	copepodids	0.5	(0.0, 2.6)	0.4	(0.0, 1.7)	1.4	(0.2, 3.9)	1.1	(0.3, 3.2)	1.0	(0.4, 2.2)
Cladocera		0.6	(0.0, 2.8)	0.6	(0.0, 1.3)	2.6	(1.2, 4.6)	1.7	(0.8, 4.6)	1.2	(0.3, 2.8)
Other		0.2	(0.0, 0.4)	0.2	(0.0, 0.5)	0.0	(0.0, 0.2)	0.2	(0.0, 0.2)	0.1	(0.0, 0.2)

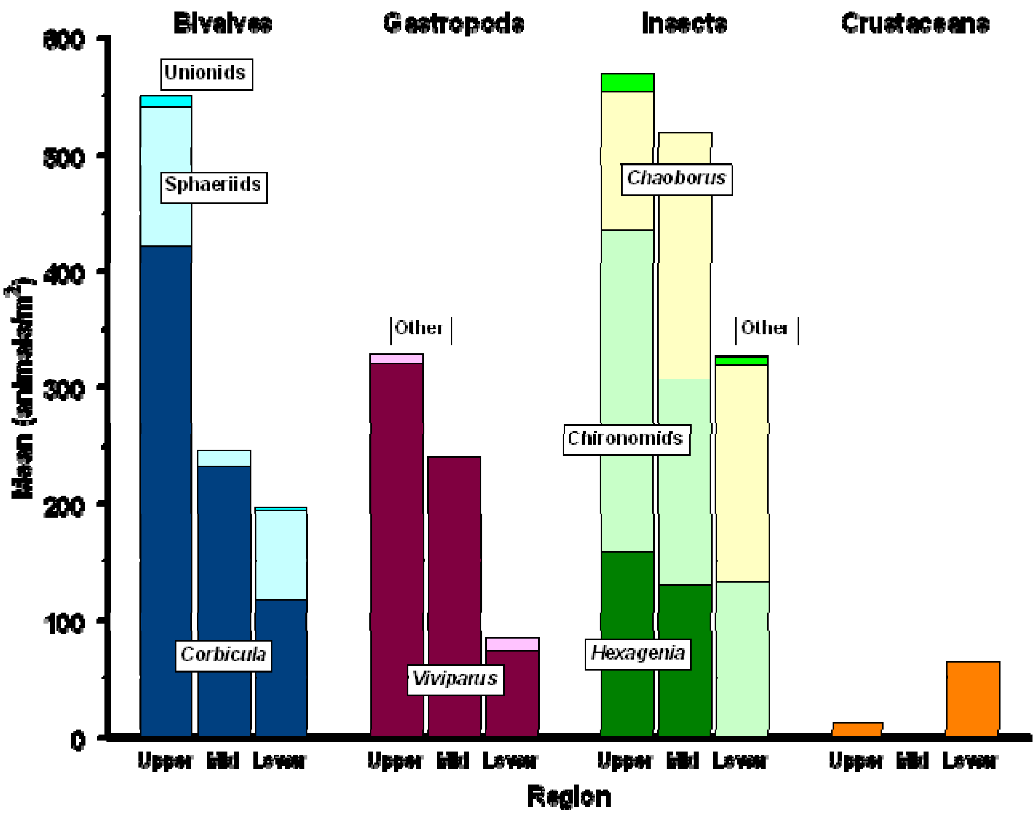

Figure 1. Mean abundances of benthic invertebrates by taxonomic group in Upper, Middle, and Lower Lake Marion. Upper Lake Marion results, which have been reported previously, are included for comparison.

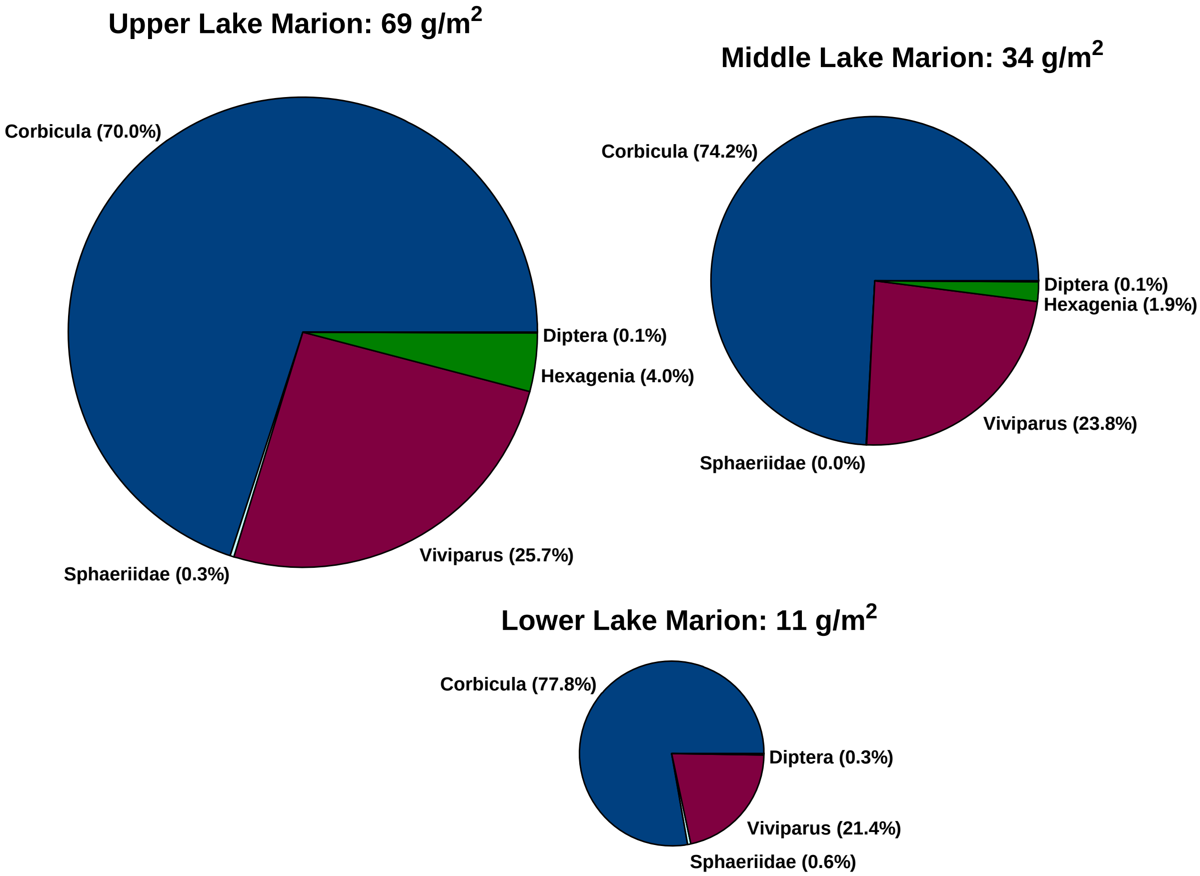

Figure 2. Biomasses of benthic invertebrates by taxonomic group in Upper, Middle, and Lower Lake Marion. Percentages are derived from values in Table 2. Upper Lake Marion results, which have been reported previously, are included for comparison.

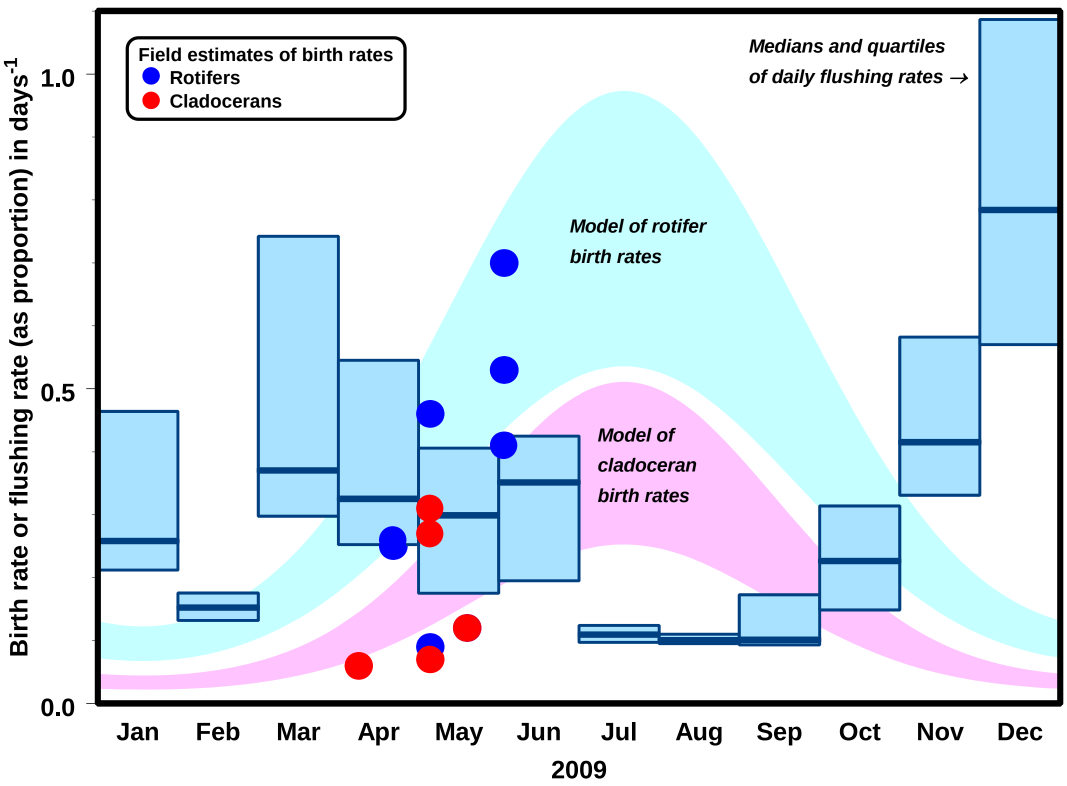

Figure 3. Comparison between zooplankton birth rates and loss rates due to flushing in Upper Lake Marion.

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Job Title: Crayfishes, shrimps, mussels, and snails from the Statewide Stream Assessment

Period Covered October 1, 2010 – September 30, 2011

Summary

Between October 1, 2010 – September 30, 2011 stream surveys were done at 69 sites in 10 ecobasins as part of the South Carolina Stream Assessment. Data were gathered on physical habitat, water chemistry, fish community, herpetofauna, and selected invertebrates. Five sites in the Broad River Outer Piedmont ecobasin were surveyed. In the lower Santee River drainage, the Atlantic Southern Loam Plains (ASLP; 3 sites), Sandhills (2 sites), and Carolina Flatwoods (1 site) ecobasins were surveyed and in the Ashepoo-Combahee-Edisto rivers (ACE) the ASLP (24 sites) and Sandhills (1 site) were sampled. In the Pee Dee River drainage, four ecobasins were sampled: ASLP (7 sites), Sandhills (18 sites), Slate Belt (4 sites) and Carolina Flatwoods (4 sites). Overall, at least 12 species of crayfishes, 1 species of shrimp, 5 species of snails (and limpets), several species of mussels, and *Corbicula* sp. (cf. *fluminea*) were collected.

Introduction

Invertebrate organisms constitute about 95% of the species diversity on Earth. Knowledge of many invertebrate groups continues to lag behind that of the more well known vertebrates and a few well-studied groups of invertebrates. The South Carolina Stream Assessment (SCSA) was intended to gather information on distribution and abundance of the aquatic fauna (fishes and select invertebrates), physical habitat, and certain water chemistry attributes at randomly-selected wadeable streams sites across the State. These data may be useful for relating environmental conditions to the presence of particular species or assemblages and for defining what factors are necessary to maintain or restore relatively natural, undisturbed faunal assemblages. The data also will be useful for prioritizing protection of high-quality aquatic communities and habitats.

Materials and Methods

Between October 1, 2010 – September 30, 2011 stream surveys were done at 69 sites in 10 ecobasins. Ecobasins are defined as specific river drainage subsets of the larger level IV ecoregions developed for South Carolina by Griffith et al. (2002); some of the smaller ecoregions are not included. Five sites in the Broad River Outer Piedmont ecobasin were surveyed in 2010. In the Lower Santee River drainage, the Atlantic Southern Loam Plains (ASLP; 3 sites; 2010), Sandhills (2 sites; 2010), and Carolina Flatwoods (1 site; 2011) ecobasins were surveyed and in the Ashepoo-Combahee-Edisto rivers (ACE) the ASLP (24 sites; 2011) and Sandhills (1 site; 2011) were sampled. In the Pee Dee River drainage, four ecobasins were sampled in 2011: ASLP (7 sites), Sandhills (18 sites), Slate Belt (4 sites) and Carolina Flatwoods (4 sites) (Table 1).

Table 1. Streams sampled during the period October 1, 2010 – September 30, 2011 (arranged by ecobasin). ACE = Ashepoo-Combahee-Edisto rivers, ASLP = Atlantic Southern Loam Plains, FW = Carolina Flatwoods.

Ecobasin	**Stream**	**Site Number**	**Date Sampled**
Broad Outer Piedmont	Buffalo Creek	4877	3 Nov 2010
Broad Outer Piedmont	Crims Creek	155535	12 Oct 2010
Broad Outer Piedmont	Beaver Creek	118022	12 Oct 2010
Broad Outer Piedmont	Cannons Creek	148342	12 Oct 2010
Broad Outer Piedmont	South Pacolet River	5685	18 Aug 2011
ACE Sandhills	Black Creek	250149	16 Aug 2011
ACE ASLP	Tributary to Toby Creek	329877	13 Apr 2011
ACE ASLP	Parker Branch	348611	12 Apr 2011
ACE ASLP	Little Salkehatchie River	324816	13 Apr 2011
ACE ASLP	Little Bull Creek	299102	21 Apr 2011
ACE ASLP	Windy Hill Creek	317466	19 Apr 2011
ACE ASLP	Whaley Branch/Whaley Creek	314722	13 Apr 2011
ACE ASLP	Big Beaver Creek	283834	14 Apr 2011

Ecobasin	**Stream**	**Site Number**	**Date Sampled**
ACE ASLP	Tributary to Jackson Branch	363045	12 Apr 2011
ACE ASLP	Rocky Swamp Creek	300478	25 May 2011
ACE ASLP	Birds Branch	351886	24 May 2011
ACE ASLP	Saddler Swamp	285099	22 Jun 2011
ACE ASLP	Murph Mill Creek	277289	21 Jun 2011
ACE ASLP	Beech Creek	251812	19 Apr 2011
ACE ASLP	Jackson Branch	365040	24 May 2011
ACE ASLP	Miller Swamp	362227	29 Jun 2011
ACE ASLP	Flea Bite Creek	289921	26 May 2011
ACE ASLP	Big Beaver Creek	284551	20 Apr 2011
ACE ASLP	Little Salkehatchie River	326516	25 May 2011
ACE ASLP	Roberts Swamp	320715	23 Jun 2011
ACE ASLP	Bull Swamp Creek	269910	28 Jun 2011
ACE ASLP	Caw Caw Swamp	285081	22 Jun 2011
ACE ASLP	Pond Branch	298149	17 Aug 2011
ACE ASLP	Toby Creek	342938	9 Aug 2011
ACE ASLP	Goodland Creek	304912	28 Jun 2011
Lower Santee ASLP	Cedar Creek	239142	5 Oct 2010
Lower Santee ASLP	Buckhead Creek	256365	4 Oct 2010
Lower Santee ASLP	Sandy Run Creek	245198	6 Oct 2010
Lower Santee Sandhills	Gills Creek	206029	5 Oct 2010
Lower Santee Sandhills	Big Beaver Creek	255769	4 Oct 2010
Lower Santee FW	Wambaw Creek	349446	21 Jul 2011
Pee Dee ASLP	High Hill Creek	157374	20 Jul 2011
Pee Dee ASLP	Jordan Creek	161334	31 Aug 2011
Pee Dee ASLP	Willow Creek	186538	10 Aug 2011
Pee Dee ASLP	Horse Creek	124431	14 Jul 2011
Pee Dee ASLP	Rocky Bluff Swamp	198278	11 Aug 2011
Pee Dee ASLP	Catfish Canal	159918	10 Aug 2011
Pee Dee ASLP	Boggy Gully Swamp	162435	20 Jul 2011
Pee Dee Sandhills	Trib. to Hatchet Camp Branch	228241	3 May 2011
Pee Dee Sandhills	Tributary to Nancy Branch	155952	15 Mar 2011
Pee Dee Sandhills	Tributary to Juniper Creek	89665	16 Mar 2011
Pee Dee Sandhills	Buffalo Creek	91052	12 Jul 2011
Pee Dee Sandhills	Bay Branch	76767	17 Mar 2011
Pee Dee Sandhills	Little Fork Creek	60805	16 Mar 2011
Pee Dee Sandhills	Mills Creek	117754	15 Mar 2011
Pee Dee Sandhills	South Buffalo Creek	99509	16 Mar 2011
Pee Dee Sandhills	Huckleberry Branch	62904	8 Jun 2011
Pee Dee Sandhills	Big Bear Creek/N Prong Creek	69005	4 May 2011
Pee Dee Sandhills	Beaverdam Creek	128722	9 Jun 2011
Pee Dee Sandhills	Beaverdam Creek	122370	19 May 2011
Pee Dee Sandhills	Rocky Creek	88961	18 May 2011
Pee Dee Sandhills	Big Sandy Creek	96049	3 May 2011
Pee Dee Sandhills	Fork Creek	72138	4 May 2011
Pee Dee Sandhills	Black Creek	77498	8 Jun 2011
Pee Dee Sandhills	Little Lynches River	101902	7 Jun 2011

Ecobasin	Stream	Site Number	Date Sampled
Pee Dee Sandhills	Whites Creek	46339	13 Jul 2011
Pee Dee Slate Belt	Hanging Rock Creek	97604	18 May 2011
Pee Dee Slate Belt	North Branch Wildcat Creek	55326	5 May 2011
Pee Dee Slate Belt	Little Lynches River	86893	24 Mar 2011
Pee Dee Slate Belt	Little Lynches River	94450	24 Mar 2011
Pee Dee FW	Tributary to Big Swamp	203569	31 Aug 2011
Pee Dee FW	Hope Swamp/Boykin Creek	216403	30 Aug 2011
Pee Dee FW	Boggy Swamp	314075	19 Jul 2011
Pee Dee FW	Long Branch	242601	30 Aug 2011

Results and Discussion

Between October 1, 2010 – September 30, 2011 stream surveys were done at 69 sites in 10 ecobasins as part of the SCSA. Five sites in the Broad River Outer Piedmont ecobasin were surveyed. In the lower Santee River drainage, the ASLP (3 sites), Sandhills (2 sites), and Carolina Flatwoods (1 site) ecobasins were surveyed and in the ACE the ASLP (24 sites) and Sandhills (1 site) were sampled. In the Pee Dee River drainage, four ecobasins were sampled: Atlantic Southern Loam Plains (7 sites), Sandhills (18 sites), Slate Belt (4 sites) and Carolina Flatwoods (4 sites).

Collections of crayfishes and shrimps were made at all sites in the ACE (25), Lower Santee (6), and Broad (5) river basins and at all except 1 site in the Pee Dee River basin (32 of 33 sites) sampled in late 2010 through 2011 and included a total of at least 4 species of *Cambarus*, 8 species of *Procambarus* (7 native, 1 introduced), and 1 species of shrimp (*Palaemonetes* sp.). A total of 9 species of crayfishes and 1 species of shrimp were identified from localities in the Pee Dee basin , 4 species of crayfishes and 1 species of shrimp from the ACE basin, 2 species of crayfishes and 1 species of shrimp from the Lower Santee basin, and 2 species of crayfishes (no shrimp collected) were identified from sites in the Broad basin. Species richness ranged from 0–4 species of crayfishes and shrimp, with an average of 2 species per site, and abundances of species at sites were 1–97 individuals. Some collections containing only juveniles/ subadults or adult females of *Cambarus* spp. could not be identified to species. Only 14% of collections of *Cambarus* spp. contained adult form I males compared with 29% of collections of *Procambarus* spp.

During the 2010–2011 surveys, six crayfish species of conservation concern (Kohlsaat et al., 2005) were collected from 53 sites in 9 ecobasins. *Procambarus hirsutus* ("Moderate" conservation concern) was collected most frequently (21 of 69 sites), and all sites were within the ACE ASLP and Sandhills ecobasins. A species of "Highest" conservation concern, *Procambarus echinatus,* was collected at 3 sites in the ACE ASLP ecobasin. *Procambarus chacei*, a species of "Moderate" conservation concern, was collected at 9 of 69 sites in the Lower Santee ASLP, Sandhills, and Carolina Flatwoods ecobasins and in the Pee Dee ASLP. *Procambarus lepidodactylus*, a species of "High" conservation concern, was collected at 7 of 69 sites within the Pee Dee ASLP and Sandhills ecobasins. *Procambarus acutus* and/or *P. blandingii*, the latter being a species of "Moderate" conservation concern, were collected at 16 of 69 sites. During 2009–2011, more species of conservation concern were collected, and from more sites, compared with the 2006–2008 sampling (Poly, 2009, 2010, this report). Much of the discrepancy among sampling periods is likely due to a combination of differences in number of sites and ecobasins sampled within the periods and the effects of drought conditions experienced in 2007 and 2008. A single specimen of *Cambarus reduncus* (no conservation status) was captured in the Pee Dee Slate Belt ecobasin. The non-native species, *Procambarus clarkii*, was collected at only two sites in the Pee Dee and Broad river drainages.

Mussels and snails were kept from sites where they were observed, but many of the mussel collections have not been identified yet, whereas snail identifications have been completed with the assistance of Robert T. Dillon, Jr. (College of Charleston). Of the 69 sites sampled in 2010 and 2011 mussels were recorded from 21 sites (1–29 individuals per site; including live animals and shells), snails (and limpets) were caught at 28 sites (1–38 individuals per site), and the non-native, *Corbicula* sp., was found at 19 sites (3–21 individuals per site, but at some sites was noted as present only). Most of the mollusks were collected incidentally while netting fish and crayfish; therefore, the abundances likely do not reflect actual densities of mollusks accurately.

Mussels were found in five ecobasins, including ACE ASLP, Pee Dee ASLP and FW, and Lower Santee FW and ASLP. Many of the mussels (specimens and photos) have not been identified yet. Two shells of *Pyganodon cataracta* were found at one site in the Pee Dee ASLP. In Buckhead Creek (Lower Santee ASLP), a small, lanceolate species of *Elliptio* was collected. *Elliptio* cf. *producta* was collected at one site in the Pee Dee ASLP and *Elliptio* cf. *complanata* was collected at several sites. Most of the *Elliptio* species in South Carolina belong to species complexes and require further study in order to identify them accurately. Therefore, live mussels from several sites were preserved in 95% ethyl alcohol for future use in genetic studies that should help resolve some of the current taxonomic uncertainty with South Carolina's freshwater mussels. Unfortunately, the needed taxonomic and biological research on mussels and other invertebrates in South Carolina has been and continues to be underfunded. *Corbicula* sp. (cf. *fluminea*) was collected in 7 ecobasins and occurs statewide.

Campeloma decisum was the most common snail, being found at 23 sites in 5 ecobasins, and abundance was 1–38 individuals per site. This species has been the most common and abundant snail throughout the Coastal Plain during the SCSA project. Other snails and limpets included *Helisoma anceps* (1 site), *Laevapex fuscus* (1 site), *Physa acuta* (2 sites), and *Pleurocera catenaria dislocata* (1 site). None of the 4 snail species of conservation concern were collected; these consist of *Gillia altilis*, *Lioplax subcarinata*, *Physa carolinae*, and *Somatogyrus* spp. (*virginicus*). *Physa carolinae* was included on the list of conservation species, in part, because it was an undescribed species at the time; however, it is known from 6 counties on the outer Coastal Plain of South Carolina and can be abundant at times (Wethington et al., 2009). Three fingernail clams were collected at a single site in the ACE ASLP. At least some species of snails and the fingernail clams are probably more widespread and abundant than the limited data from the SCSA indicate.

Recommendations

Perform biological, distribution, taxonomic, and habitat studies on specific species of conservation concern.

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Prepared By: William Poly Title: Aquatic Biologist

Job Title: American eel abundance, and distribution along the spillways of the Lake Wateree Dam on the Wateree River and Columbia Dam on the Broad River

Period Covered July 1, 2010 – June 30, 2011

Summary

During 2011 we continued our efforts to monitor American eel abundance and distribution along the spillways of the Columbia and Wateree dams. Eel traps were fished at four locations along each dam during 2011 for a total effort of 890 ramp days at Wateree Dam and 380 ramp days at Columbia Dam. Backpack electrofishing was conducted on 15 dates between March and November with a total electrofishing effort of 305 minutes at Columbia Dam and 189 minutes at Wateree Dam. A total of 7 American eels were captured; 3 at Wateree Dam and 4 at Columbia Dam. Catch rates of American eels at both dams during 2011 were very low and comparable to catch rates observed during 2010. Based on ramp trap collections and backpack electrofishing along the spillways of the dams there appeared to be very few eels in the vicinity of the two dams during 2011.

Introduction

Since the 1980's a decrease in American eel *Anguilla rostrata* catch rates has heightened concerns over the status of the population. The cause of this decline is unknown, but several factors (e.g. migration barriers, habitat loss and degradation, overfishing, etc.) have been identified that could affect population size and distribution. American eel were historically abundant along the Atlantic slope where their range extended into the Wateree and Broad rivers and their tributaries. Dams constructed along those rivers and tributaries have impeded the inland migration of juvenile eels as well as the seaward migration of adults and altered their distribution within the Santee River Basin. Facilitating passage of American eel around migration barriers should benefit American eel populations and augment restoration efforts. Juvenile eels may exhibit specific habitat preferences that could influence where along the dam they attempt upstream passage. Maximizing eel passage will require effective placement of passage facilities.

Materials and Methods

Eel ramp traps and backpack electrofishing were used to identify when and where eel passage and collection devices should be placed to maximize passage of American eels. Eel ramp traps were installed at Wateree Dam during March 2010 and at Columbia Dam during May and June 2010. The design of the ramps is similar to those that worked very well at Roanoke Rapids, NC. The ramp traps were constructed from ¾ inch plywood and range from roughly 7 ft to 13 ft in length and are 12 inches wide. The ramp deck is covered with 1-in polyethylene Akwadrain material and terminates at a covered collection bucket. Water is supplied to each ramp and collection bucket through gravity fed supply lines. Three ramp traps were installed at Wateree Dam during spring 2010 and a fourth trap added during 2011 (Figure 1). At Columbia Dam 3 ramp traps were installed during spring 2010 and a fourth trap, a box-style, trap was added during 2011. During 2011 all the ramp traps, except for trap 1 at Columbia Dam, were replaced with aluminum ramps with similar dimensions and water supply features as the original wooden ramps. In addition to ramp traps Fukui traps or minnow traps covered with nylon stocking material were baited with cut gizzard shad or cat food and fished at 4 and 6 locations at Columbia Dam and Wateree Dam, respectively.

Figure 1. Eel ramp trap locations at Columbia Dam (upper panel) and Wateree Dam (lower panel) during 2011.

Eel ramp traps, as well as baited traps, were monitored at least monthly until April, and then every Monday, Wednesday, and Friday through June. After June eel traps were monitored at least biweekly for the remainder of the year. The base of each dam was visually surveyed each sampling day to identify congregations of eels in areas not sampled with traps. The presence of eels in the vicinity of the Wateree Dam and their abundance were evaluated during 2011 by backpack electrofishing at least monthly March –June and at least quarterly the remainder of the year. All eels collected were enumerated, measured and released. Water temperature at each trap location was recorded continuously with temperature loggers, dissolved oxygen, and conductivity were recorded during each sampling visit.

Results

The minimum number of days eel ramp traps were in operation varied by site and trap location from 24 to 291 days during 2010 and 14 to 249 days during 2011 (Table 1). Ramp traps at Wateree Dam were in operation fairly consistently while those at Columbia dam often lost prime due to the small elevation change from the pond to the ramp traps. Ramp traps at Columbia Dam were also dislodged during high water events while those at Wateree Dam have remained in place since installation. Ramp traps 2 and 3 at Columbia Dam were frequently dislodged during spring and summer 2010 and during 2011 often lost prime due to low water levels.

Table 1. Installation date of each eel ramp trap at each site and the minimum number of days each ramp trap was running each year through 15 November 2011.

			Min Trap Days	
Site	Trap	Installation Date	2010	2011
Wateree	0	3/17/2011	*	222
	1	3/10/2010	291	211
	2	3/10/2010	224	249
	3	3/10/2010	236	208
Columbia	1	5/20/2010	161	191
	2	5/20/2010	24	53
	3	6/8/2010	63	82
	4	5/11/2011	*	14
	5	4/27/2011	*	40

Backpack electrofishing was used to supplement ramp trap effort and was conducted during spring through fall at each site during 2010 and 2011 (Table 2). An effort was made to sample for 10 minutes at each ramp trap location; however, occasionally discharge was too high to effectively sample some of the locations at each site. During July and August 2011 we increased our backpack electrofishing sampling at Columbia Dam to account for poor ramp trap performance due to low water levels. Backpack electrofishing effort during 2011 ranged from 10 to 97 minutes per month at Columbia Dam and 0 to 52 minutes per month at Wateree Dam. During the first two years of the study we have expended over 13 hours of effort backpack electrofishing in the vicinity of ramp locations at the two dams.

Table 2. Backpack electrofishing effort in minutes at each site by month during 2010 and 2011.

		Site		
Year	Month	Columbia	Wateree	Total effort
2010				
	April	0	46	46
	May	0	36	36
	June	35	18	53
	July	29	27	57
	August	10	27	38
	October	24	27	51
	November	0	21	21
2011				
	March	10	28	38
	April	10	30	40
	May	50	28	78
	June	31	52	82
	July	79	0	79
	August	97	23	120
	November	28	28	57
Total Effort		404	391	795

During 2011 we collected 7 American eels (Mean Total Length [TL] = 233 mm; range 203 – 251 mm TL) from the two sites (Table 3). Four eels were collected from Columbia Dam; three eels were collected while electrofishing near trap location 1, and the fourth eel was captured in a Fukui trap (similar to minnow trap) near trap location 1. During 2011 three eels were collected from Wateree Dam; one eel was captured in ramp traps at each location 1 and 2, and the third eel was collected while backpack electrofishing near ramp location 1. Backpack electrofishing catch rates (number/hour) of American eel at Columbia Dam were 0.6/hour during both 2010 and 2011. At Wateree Dam backpack electrofishing catch rates were 0.0/hour during 2010 and 0.3/hour during 2011. During both 2010 and 2011 ramp trap catch rates (eels/trap day) at Wateree Dam have been 0.002/day. No eels have been collected in ramp traps at Columbia Dam.

Table 3. Total length of American eel collected by date from each site and ramp location, and the method of capture during 2010-2011.

Date	Site	Location	TL (mm)	Method
4/21/210	Wateree	1	108	Ramp
8/10/2010	Wateree	2	394	Ramp
8/25/2010	Columbia	1	314	EF
5/2/2011	Wateree	1	235	Ramp
5/25/2011	Columbia	1	203	EF
6/17/2011	Wateree	1	249	EF
6/17/2011	Wateree	2	272	Ramp
6/17/2011	Columbia	1	203	EF
6/17/2011	Columbia	1	217	EF
6/29/2011	Columbia	1	251	Fukui trap

Discussion

Catch of American eels was very low at both dams during 2010 and 2011. It does not appear that many eels utilized the bypassed area below Wateree Dam, nor were eels abundant below Columbia Dam during 2010 or 2011. The low catch rates of American eel below Columbia and Wateree dams are consistent with backpack electrofishing catch rates of American eel in wadeable streams within the Congaree and Wateree drainages (Figure 2). Lower in the Santee Drainage, below the Santee-Cooper lakes, catch rates of American eel in wadeable streams are much higher ranging from 9 to 14 eels per hour (Figure 2). It is clear, based on backpack electrofishing catch rates of wadeable streams, that eels are much more abundant lower in the system, below Pinopolis Dam on the Cooper River and Wilson Dam on the Santee River. Future efforts should focus on getting American eel past those migration barriers lower in the system so that passage higher in the system at Columbia and Wateree dams can be evaluated.

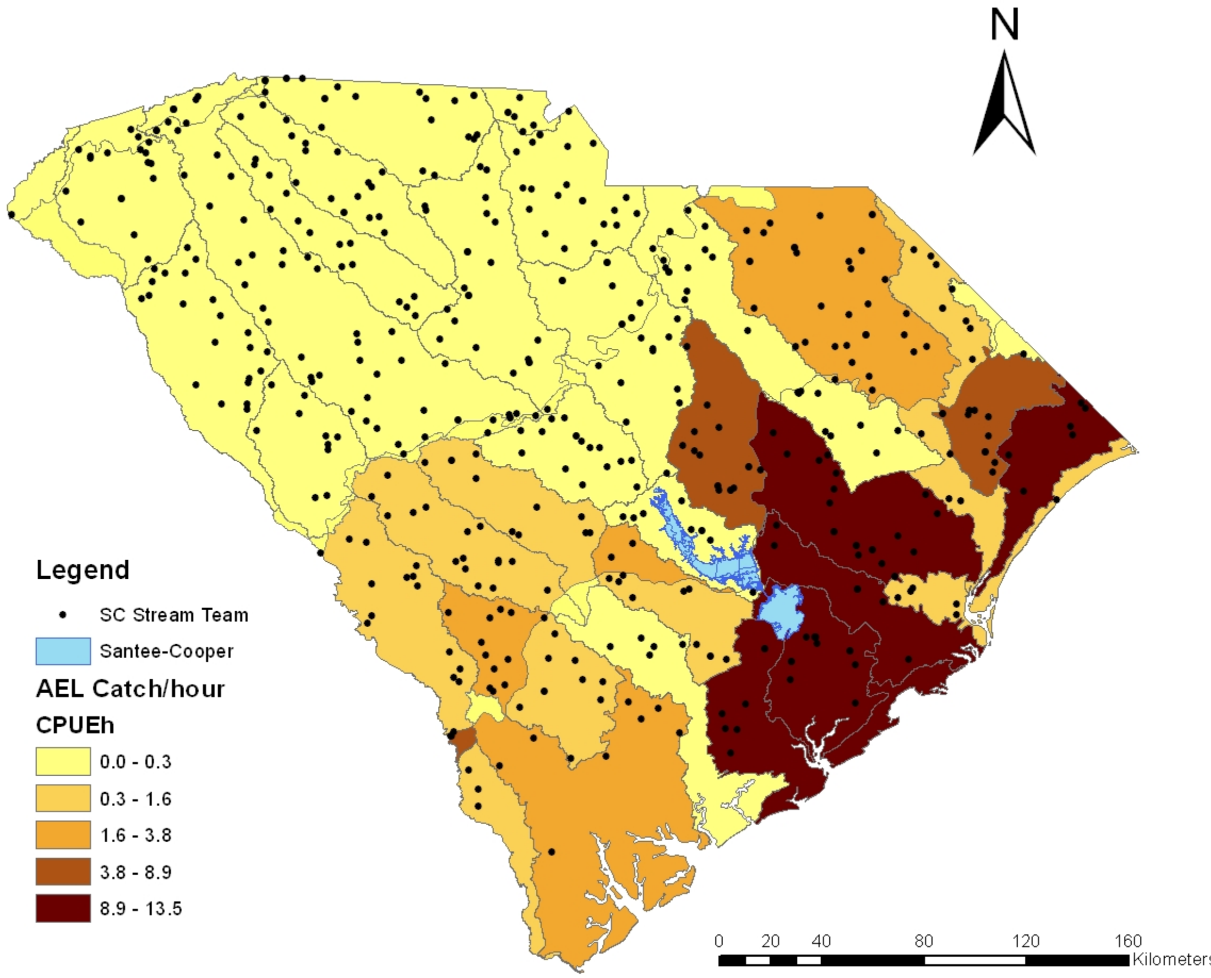

Figure 2. Mean backpack electrofishing catch rates (number/hour) of American eel in South Carolina wadeable streams by ecobasins. Mean catch rates calculated from data collected by the SCDNR stream team during 2005-2011.

The low American eel catch rates below both dams has limited our ability to recommend a suitable location for future passage facilities. All the eels collected at Wateree Dam have been collected on the west side of the dam nearest the powerhouse. We have expended over 2.3 hours of backpack electrofishing effort along the East side of the dam, and have had and eel ramp trap fishing for nearly two years in that location, but no eels have been collected. At Broad River dam all the eels have been collected on the east side of the dam, near the fish passage facility; however, our collection effort has been much greater in that area. The west side of the dam is not accessible when water is spilling which has decreased our opportunities for electrofishing and water flowing over the dam in that area has frequently dislodged our traps. At this time, based on our limited data, we cannot recommend a passage facility site for either dam.

Recommendations

Continue the study as planned, maintaining ramp traps as needed and backpack electrofishing as scheduled through fall 2012.

Prepared By: Jason Bettinger   Title: Wildlife Biologist III

Job Title: Smallmouth bass stocking assessment – Broad River Lake Jocassee, and Lake Robinson

Period Covered July 1, 2009 – June 30, 2010

Summary

We continued our study evaluating the SCDNR smallmouth bass stocking program. Fish stocked as fry and fingerlings into the Broad River during 2009 made a significant contribution to the year class, representing 46% of age-1 smallmouth bass collected during fall of 2010. Marking efficacy continues to be good at the Cheraw State Fish Hatchery where smallmouth bass marking efficacy was 100% during 2009. In 3 of 5 study years fingerling stockings were more economical than fry stockings.

Introduction

Smallmouth bass have been stocked intermittently into the Broad River and Lake Jocassee since 1984 and 1980, respectively. Each of those systems has developed small, but unique fisheries that have demonstrated the ability to grow trophy-size smallmouth bass. Numbers and sizes of fish stocked have varied greatly depending on availability. Routinely fry and fingerling smallmouth bass are stocked each year; however, it is not known which of these stockings has the higher survival and ultimately contributes to the fishery. Identifying which stocking size has the greater relative survival and adjusting that value for production costs will allow hatchery managers to focus production on the most economically beneficial size group.

Materials and Methods

OTC Marking and Stocking

Smallmouth bass fry (mean TL = 42 mm; range 26 - 63 mm TL) and fingerlings (mean TL = 150 mm; range 89 – 234 mm TL) were reared and marked with OTC at the Cheraw State Fish Hatchery in accordance with the SCDNR protocol for immersion marking juvenile fish. Fish stocked as fry received a single OTC mark and were stocked during spring and those stocked during fall as fingerlings received a second OTC mark to facilitate differentiation of the two size groups. OTC Marking efficacy was determined for each marking (immersion) event. Up to 30 fish from each marking event were retained and held separately in raceways or aquariums at the Cheraw State Fish Hatchery for at least 14 (preferably 21 d) days post immersion. Sagittal otoliths were removed from each fish and mark detection conducted at the Eastover Lab.

Stocking of smallmouth bass fry and fingerlings occurred each year from 2005 through 2010. During late May smallmouth bass fry were stocked into the Broad River and Lake Jocassee. Approximately 8,000 smallmouth bass fry were equally divided and stocked into three reaches (upper, middle, and lower) of the Broad River. Roughly 9,000 smallmouth bass fry were divided equally and stocked into Lake Jocassee at two locations. During October approximately 2,800 fingerling fish were stocked in equal proportions into the Broad River and Lake Jocassee, respectively, at the fry stocking locations.

Field Data Collection

Boat electrofishing during late summer and early fall, prior to fall stocking of fingerlings, was used to collect smallmouth bass from the Broad River during 2005-2011. Angling was also be used to collect fish when sufficient numbers were not collected with boat electrofishing gear.

Up to 80 age-1 fish from each of the three river sections were collected each year for evaluation, but all smallmouth bass collected were retained for ageing.

Boat electrofishing and littoral gill netting was used to collect smallmouth bass from Lake Jocassee. Electrofishing was conducted in March. Smallmouth bass were also collected with littoral gill net sets. Gill nets were experimental multi-filament nylon nets, 150 feet x 6 feet, containing three 10-foot panels each of five mesh sizes (1, 1.5, 2, 2.5, and 3 inch, bar measure). Nets were set horizontal on the bottom (littoral sets) at depths ranging from 10-50 feet for two consecutive days at five standardized locations during the months of January, March, May, and November, for a total of forty net-nights each year. This is an on-going standardized sampling program on Lake Jocassee, and was utilized to collect fish for this study.

Total length and weight was recorded for each smallmouth bass collected. Sagittal otoliths were removed from each fish to estimate age. Otoliths of fish from the 2005 – 2010 year class were examined for OTC marks.

Analytical Methods

The contribution of fingerling and sub-adult stocked fish as well as naturally reproduced fish to each year-class was calculated by dividing the number of otoliths with single mark (fingerling stocked) and double mark (sub-adult stocked) or no mark (naturally produced) by the total number of otoliths examined. Relative survival (RS) by year-class between fry and fingerling stockings to account for unequal stocking rates was calculated as:

$$RS = (nf /Nf)/(ne /Ne),$$

Where nf = number of smallmouth stocked at size f and recaptured, Nf = number of smallmouth stocked at size f, ne = number of smallmouth stocked at size e and recaptured, Ne = number of smallmouth stocked at size e. Because production costs increase significantly with fish size due to a variety of factors (e.g., extended feeding, maintenance, mortality) RS was used in conjunction with production costs to determine the cost benefit of each stocking size. Based on current national production costs of $0.69 for a two inch “fry” and $2.49 for a 6-inch “fingerling” smallmouth bass the RS ratio would need to be at least 3.7:1 in favor of fingerling stockings to warrant their stocking in lieu of fry.

Results

OTC Marking and Stocking

On 25 May 2010 an estimated 9,000 smallmouth bass fry were divided equally and stocked at four locations into the Broad River. On 26 October 2010 an estimated 2,100 smallmouth bass fingerlings were divided equally and stocked at three locations into the Broad River All fish received a single OTC mark in one immersion event and fall stocked fingerlings received a second OTC mark in a second immersion marking event. All OTC immersion marking occurred at the Cheraw State Fish Hatchery. Overall marking efficacy of spring and fall-stocked smallmouth bass was evaluated by reviewing the otoliths of at least 20 fall-stocked fingerlings from each of the four stocking locations. Marking efficacy was 100% with all 82 otoliths reviewed containing two clearly readable marks.

Broad River

During October 2010 smallmouth bass were collected with angling gear from three river sections on 4 sampling days (Table 1). Two days of electrofishing were conducted on two river sections during November to augment the limited number of fish collected with angling gear (Table 2). In all, 181 smallmouth bass were collected during 2010 and their otoliths were read to estimate their age (Table 3).

Table 1. River section sampled, number of anglers, angling effort, and CPUE (No/h) of smallmouth bass (SMB) collected from the Broad River with angling gear during October 2010.

Date	River Section	No Anglers	Time Fished (h)	Total Effort (h)	SMB Collected	CPUE (no./h)
10/18/2010	Below Neal Shoals	5	7.9	39.5	45	1.14
10/20/2010	Below Gaston Shoals	4	7.5	30.0	19	0.63
10/21/2010	Below Gaston Shoals	2	8.5	17.0	28	1.65
10/27/2010	Below 99-islands	4	5.0	20.0	6	0.30
Total					98	0.92

Table 2. River section sampled, electrofishing effort, number of smallmouth bass collected and catch per unit effort (CPUE) of smallmouth bass collected from the Broad River during Fall 2010.

Date	River Section	Effort (h)	SMB Collected	CPUE (no./h)
11/9/2010	Below Neal Shoals	4.24	59	13.9
11/15/2010	Below 99-islands	1.47	24	16.3
Total			83	14.5

Table 3. Age, number of smallmouth collected, mean total length (TL) mm, and standard error (SE) of smallmouth bass collected during fall 2010.

Age	Number	Mean TL	SE
0	47	148	3.8
1	39	233	4.4
2	65	284	4.6
3	25	295	9.3
4	3	420	16.3

Otoliths from179 smallmouth bass collected from the Broad River during 2010 were successfully reviewed for OTC marks to determine whether they were wild fish or hatchery stocked fish. Of the 47 age-0 fish collected and successfully reviewed for OTC marks 14 were marked, eight otoliths had a single mark indicating they were stocked in spring 2010 as fry, and 6 were double marked indicating they were stocked during fall 2010 as fingerlings, the other 33 age-0 fish were presumably wild (Table 4). Otoliths from 39 age-1 fish were successfully reviewed for OTC marks, 21 of those fish were unmarked (wild), 6 were single marked (fry-stocked during spring) and 12 were double marked (fingerling-stocked during fall) (Table 4). The contribution of stocked fish to the 2010 year class one year post-stocking was 46% (Table 5).

Table 4. Collection year, year class (YC) and the number of wild spawned, spring-stocked and fall-stocked smallmouth bass, based on differential OTC marks, collected from the Broad River, South Carolina.

Year	YC	Wild Fish	Spring Stocked	Fall Stocked	Number Reviewed
2006					
	2005	29	2	24	55
	2006	92	3		95
2007					
	2005	5			5
	2006	154	4	2	160
	2007	70	3		73
2008					
	2005	5			5
	2006	57	2	1	60
	2007	188	12	6	206
	2008	71	5		76
2009					
	2005	1			1
	2006	22			22
	2007	67	4		71
	2008	92	1	4	97
	2009	4	2	3	9
2010					
	2006	3			3
	2007	25			25
	2008	64	1		65
	2009	21	6	12	39
	2010	33	8	6	47

Between 2005 and 2009 the contribution of fry-stocked fish at age-1 has ranged from 1% to 15% while that of fingerling-stocked fish has ranged from 1% to 44% (Table 5). The total contribution of both sized stockings has ranged from 4% to 47% and averaged 22%. The RS of fingerling stocked fish ranged from 2.03 to 35.14.

Table 5. The number of fry and fingerling smallmouth bass stocked each year, the number of otoliths collected from age-1 smallmouth bass and reviewed for OTC marks, percent contribution of each size stocking at age-1, and the relative survival (RS) of fingerling-stocked fish when compared to fry-stocked fish in the Broad River, South Carolina.

	Number stocked			Percent Contribution			
Year	Fry	Fingerling	N	Fry	Fingerling	Total	RS-Fingerling
2005	8200	2800	55	0.04	0.44	0.47	35.14
2006	11340	2000	160	0.03	0.01	0.04	2.84
2007	12000	3226	194	0.06	0.03	0.09	2.03
2008	8500	3500	97	0.01	0.04	0.05	9.71
2009	10000	3500	39	0.15	0.31	0.46	5.71

Lake Jocassee and Lake Robinson

Otoliths were not collected from Lake Jocassee or Lake Robinson during 2010.

Discussion

In the Broad River the contribution of stocked fish to the 2005 and 2009 year class was 47 % and 46%, respectively, but the contribution of stocked fish to the 2006 - 2008 year classes averaged only 6% (range; 4% - 9%). Based on five years of data collection it appears that there is large annual variation in the recruitment of wild and stocked fish to age-1 in the Broad River. That variation could be due, in part, to river discharge. High or low river discharges can influence success of natural recruitment and survival of young-of-the-year wild and stocked smallmouth bass. During 2005 and 2009 the Broad River experienced average spring water levels with a wet summer during 2005 and a very wet fall during 2009 (Figure 1). In each of those years stocked fish made a significant contribution to their respective year classes. However, during 2006-2008 river water levels were well below average for most of the year and contribution of stocked fish was poor. A final collection of smallmouth bass was made during 2011 that will allow us to evaluate the success of stockings during 2010, a year when water levels after March were well below average and consistent with those observed during 2006-2008.

In three of the five study years fingerling stockings were more economical than fry stockings. In those three years, two of which were years when stocked fish made a significant contribution to the year class, fingerling stocked smallmouth bass had a survival rate at least 5.7 times greater than fry stocked smallmouth bass. If SCDNR smallmouth bass production costs are similar to the national average ($0.69/fry and $2.49/fingerling) then fingerling smallmouth bass should be stocked in lieu of fry during years that smallmouth bass are stocked in the Broad River.

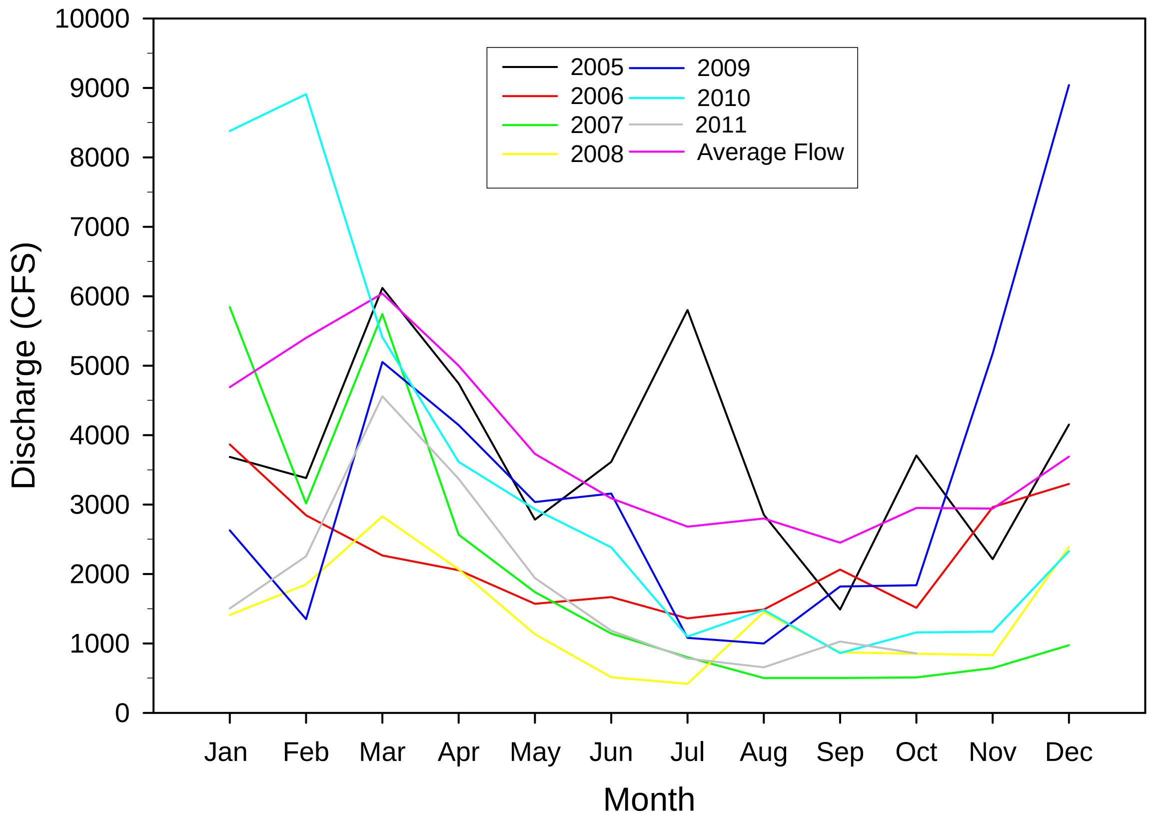

Figure 1. Average monthly discharge (cfs) of the Broad River at Carlisle, South Carolina, mid-point of the river, during 2005 – 2011.

Recommendations

During FY12 we need to determine whether or not a smallmouth bass fishery has developed in Lake Robinson and whether or not that fishery warrants future stocking. Smallmouth bass collected from the Broad River during fall 2011 should be processed for OTC marks to evaluate the contribution of stocked fish during a year when water levels after March were well below average and consistent with those observed during 2006-2008. Based on five years of data collection it appears that fingerling stockings are more economical than fry stockings in the Broad River.

Prepared By: Jason Bettinger Title: Wildlife Biologist III

Job Title: Distribution of striped bass in J. Strom Thurmond Reservoir, South Carolina-Georgia, in relation to pump storage operation and hypolimnetic oxygenation

Period Covered July 1, 2010 – June 30, 2011

Summary

During FY11 fifty-seven striped bass collected from the Lake Russell Tailrace and tributaries to Thurmond Reservoir were implanted with acoustic transmitters. Sixty-five fish have been successfully implanted since the study began in FY10. Implanted striped bass were detected by 52 different receivers stationed throughout the reservoir and were manually tracked on 37 dates. Sixty-nine percent of implanted fish appeared to be alive at the end of FY11, 9% have been harvested, and the remaining fish have either died (6%) or are missing (15%). The Russell Tailrace was an important habitat for striped bass during 2010 and 2011 with the majority (> 77%) of fish occupying the tailrace at some point during each summer.

Introduction

J. Strom Thurmond (Thurmond) Reservoir supports a popular recreational striped bass fishery. Striped bass production at Thurmond is largely due to suitable habitat provided by artificially oxygenated, hypolimnetic releases from Richard B. Russell (Russell) Dam, that provide cool well oxygenated water in the tailrace and upper portions of Thurmond Reservoir.

During 2011 Russell Dam commenced expanded pump-storage operations which could result in warmer tailrace temperatures below Russell Dam, possibly reducing suitable habitat for some species of fish. Given the unsuitable striped bass habitat throughout most of the reservoir during the summer the loss of the refuge in the Russell tailrace and upper Thurmond could have a negative impact on the striped bass fishery. To mitigate for the potential loss of striped bass habitat in the Russell tailrace and upper Thurmond, the USACE installed an oxygen injection system in the lower portion of Thurmond to provide striped bass habitat.

It is unknown how striped bass will utilize the expected reduction of habitat in the Russell tailrace and upper Thurmond or the new artificially oxygenated area in the lower reservoir. Considerable expense has been expended in the development and installation of the new oxygen injection system and it is important to document the extent of striped bass use of the newly-created habitat. Information on the seasonal distribution of striped bass after project implementation will be important for successful management of the striped bass fishery in Thurmond Reservoir

Materials and Methods

The study will monitor the seasonal movement of adult striped bass in Thurmond Reservoir. Specifically monitoring their seasonal use of the current refuge area in the upper reaches of Thurmond as well as the enhanced area below Modoc, SC. In spring of 2010 and 2011 striped bass were collected from the Russell tailrace and at least two major tributaries (e.g., Little River, GA and Little River, SC) and surgically implanted with individually coded temperature sensing acoustic transmitters. Two different transmitters manufactured by Sonotronics were used based on fish length. A high powered long-range transmitter (Model CHP-87-L) expected to last 18 months was implanted in striped bass > 575 mm TL and a less powerful transmitter (Model CTT-83-3) expected to last 36 months was implanted in striped bass > 480 mm TL.

An array of remote acoustic receivers (SUR-3BT, Sonotronics Inc.) was used to collect movement data from transmitter implanted fish. Receivers were positioned throughout the mainstem reservoir with expanded arrays in the tailrace and oxygen injected area to achieve continuous coverage of the Savannah River channel in those areas. Additional location data was collected with a hand held ultrasonic receiver (USR-08, Sonotronics Inc.) to identify other potential refuges and locate missing fish. Temperature and oxygen profiles at 1-m depth intervals were determined biweekly during the summer study period at a series of fixed stations throughout the monitored area.

Results

Fifty-seven striped bass (mean TL = 698 mm; range 480 – 1300 mm TL) collected from the Lake Russell Tailrace and four tributaries were implanted with acoustic transmitters during FY11 (Table 1). During August 2010 twenty-one striped bass (mean TL = 647 mm; range 480 - 1040 mm TL) collected from the Russell Tailrace were implanted with acoustic transmitters. During spring 2011 thirty-six striped bass (Mean TL = 728 mm; range 550 – 1300 mm TL) were captured from the Russell Tailrace and four tributaries and implanted with acoustic transmitters. Since FY10 a total of 74 striped bass have been implanted with transmitters.

Between 20 August and 17 September 2010 submersible acoustic receivers (SUR-3BT) were deployed at 34 locations (Figure 1). During November 2010 eleven additional receivers were deployed to provide increased coverage of the Savannah River Channel and during February 2011 seven receivers were deployed in the Little River, SC. As of June 2011 there were 52 receivers in the Thurmond receiver array. Striped bass implanted with transmitters were manually tracked on 37 dates during FY11.

There have been nearly 1.7 million detections at 52 receiver locations and all fish known to have survived transmitter implantation have been detected at least once. During manual tracking events 33 different fish were located at least once (Table 1). At the conclusion of FY 11, 45 implanted striped bass were assumed to alive, 4 fish had died, 6 fish were harvested, 10 fish were missing, and 9 fish either expired from transmitter implantation or their transmitters have malfunctioned. During FY11 an Access database was constructed to store all receiver and manually collected location data.

Table 1. Date of implantation, transmitter ID, Total Length (TL), location of implantation, fate, number of detections with receivers and while manual tracking, and the number of days tracked post implantation for transmitter implanted striped bass in Lake Thurmond, SC-GA through June 2011. Fate codes are; Alive (A), Dead (D), Harvested (H), Missing (M), Tagging Mortality or Faulty Transmitter (TM/FT).

					Detections		Days
Date	ID	TL	Location	Fate	Receiver	Manual	Tracked
4/20/2010	2	1200	Broad River, GA	H	43694	4	304
4/16/2010	3	665	Little River, SC	TM/FT	14		
5/4/2010	4	1400	Broad River, GA	M	47459	4	400
5/4/2010	5	800	Broad River, GA	M	20	3	50
4/16/2010	6	650	Long Cane Cr., SC	D	12249	3	
5/4/2010	7	1200	Broad River, GA	TM/FT	13	2	
5/4/2010	8	930	Broad River, GA	A	78943	9	422
4/20/2010	9	693	Thurmond	D	11		
4/16/2010	10	730	Long Cane Cr., SC	TM/FT	45	2	
5/4/2010	11	863	Broad River, GA	A	58668	4	422
5/4/2010	17	950	Broad River, GA	TM/FT		3	
4/28/2010	18	690	Little River, GA	TM/FT		2	
4/16/2010	19	655	Little River, SC	A	32128	3	440
4/16/2010	20	820	Little River, SC	M	74269	1	252
4/28/2010	21	632	Little River, GA	A	43376	5	428
4/28/2010	22	565	Little River, GA	A	21122	4	428
5/4/2010	23	722	Broad River, GA	TM/FT			
8/24/2010	24	1040	JST Tailrace	A	30014		310
8/24/2010	25	582	JST Tailrace	H	29560	3	98
8/25/2010	32	613	JST Tailrace	D			309
8/24/2010	33	604	JST Tailrace	M	7872		59
8/25/2010	34	970	JST Tailrace	D	32658		47
8/24/2010	35	635	JST Tailrace	A	37280	1	310
8/25/2010	36	588	JST Tailrace	M	8451		28
8/24/2010	37	573	JST Tailrace	A	106356	3	310
8/18/2010	38	650	JST Tailrace	A	35386	1	316
8/24/2010	39	708	JST Tailrace	A	83418	2	310

Table 1. Continued.

Date	ID	TL	Location	Fate	Detections Receiver	Manual	Days Tracked
8/25/2010	40	645	JST Tailrace	A	40135	7	309
8/25/2010	41	934	JST Tailrace	A	133157	1	309
8/24/2010	47	616	JST Tailrace	A	106654	3	310
8/25/2010	48	593	JST Tailrace	A	30874	4	309
8/18/2010	49	549	JST Tailrace	A	33914	1	316
8/24/2010	50	530	JST Tailrace	A	42704	1	310
8/24/2010	51	480	JST Tailrace	A	110464	3	310
8/24/2010	52	510	JST Tailrace	TM/FT	2876	2	310
8/18/2010	53	622	JST Tailrace	A	28937	3	301
8/18/2010	54	547	JST Tailrace	A	35542	3	316
8/18/2010	56	605	JST Tailrace	H	28099	2	306
3/24/2011	67	682	Long Cane Cr., SC	A	6503		98
3/24/2011	96	620	Long Cane Cr., SC	H	1902		78
3/24/2011	65	670	Long Cane Cr., SC	A	33576	1	98
3/24/2011	100	630	Long Cane Cr., SC	H	3577		46
3/24/2011	113	810	Long Cane Cr., SC	A	5116		98
3/24/2011	62	680	Long Cane Cr., SC	A	13427	1	98
3/24/2011	71	705	Long Cane Cr., SC	A	5346		98
3/24/2011	63	702	Long Cane Cr., SC	A	16498		98
3/24/2011	68	723	Long Cane Cr., SC	H	4186		28
3/24/2011	86	925	Little River, SC	A	8973		98
3/24/2011	64	600	Little River, SC	M	21395		66
3/24/2011	82	862	Little River, SC	A	9422		98
4/5/2011	80	690	Little River, GA	A	8895		86
4/5/2011	112	620	Little River, GA	TM/FT	2		
4/5/2011	84	785	Little River, GA	A	5576		86
4/5/2011	70	680	Little River, GA	M	1331		85
4/5/2011	78	690	Broad River, GA	A	4308		86
4/5/2011	69	780	Broad River, GA	A	19210		86
4/5/2011	66	652	Broad River, GA	A	17809	1	86
4/5/2011	79	735	Broad River, GA	A	8705		86
4/5/2011	77	590	Broad River, GA	A	6511		86
4/5/2011	83	550	Broad River, GA	A	3333		86
4/5/2011	81	765	Broad River, GA	M	3966		44
4/8/2011	101	650	Little River, GA	A	2050		83
4/8/2011	93	670	Little River, GA	A	6157		83
4/18/2011	98	1200	Little River, GA	M	3302		73
4/18/2011	94	1300	Little River, GA	A	6782		73

Table 1. Continued.

Date	ID	TL	Location	Fate	Detections Receiver	Manual	Days Tracked
4/18/2011	97	705	Little River, GA	A	16821		73
5/25/2011	115	675	Russell Tailrace	TM	1418		4
5/25/2011	95	638	Russell Tailrace	A	5017		36
5/25/2011	100	643	Russell Tailrace	A	26898		36
5/25/2011	108	622	Russell Tailrace	A	3754		36
5/25/2011	114	574	Russell Tailrace	M	984		4
5/25/2011	85	702	Russell Tailrace	A	2281		11
5/25/2011	68	990	Russell Tailrace	A	2799		13
5/25/2011	92	695	Russell Tailrace	A	21652		36

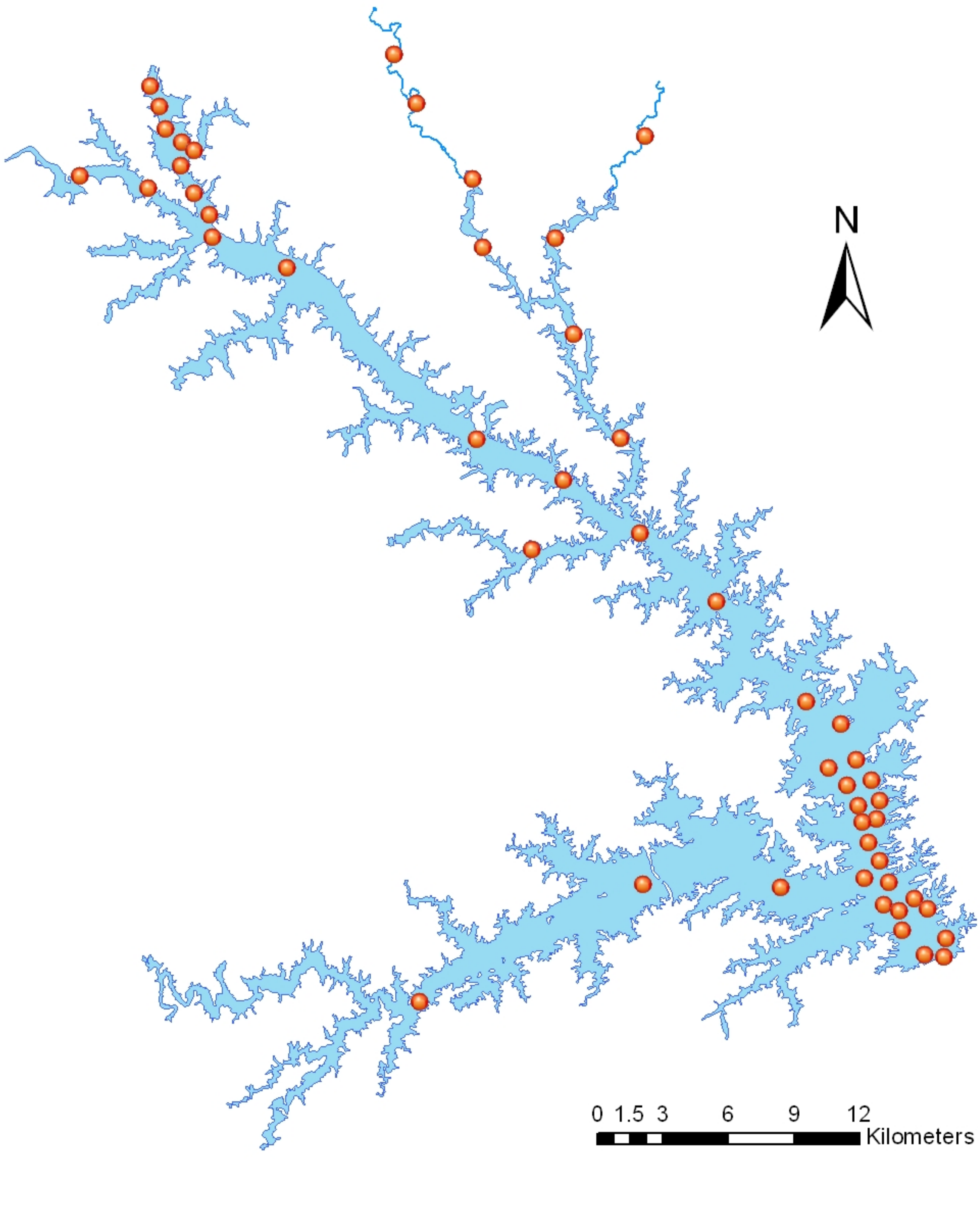

Figure 1. Acoustic receiver locations in J. Strom Thurmond Reservoir SC-GA, during 2011.

Discussion

Location data downloaded from receivers during FY11 has been incorporated into an Access database; however, rigorous analysis of those data has not been completed. Cursory examination of the data does show the importance of the Russell Tailrace as a summer habitat for striped bass. During summer 2010 and 2011 the majority of striped bass inhabited the tailrace at some point during the summer. Four of 5 fish implanted during spring 2010 and monitored through the 2010 summer utilized the Russell Tailrace, while only one fish stayed in the lower lake during summer. Thirty-five of 45 fish alive during summer 2011 spent some portion of the summer in the Russell Tailrace while the remaining 10 fish spent the summer in the lower and mid lake regions. It also appears that fish that inhabit Little River, GA during spring are less likely to utilize the Russell Tailrace during summer. Of 10 fish implanted in Little River, GA during spring 2011 only 3 fish utilized the tailrace during summer, the remaining 7 fish restricted their summer movements to the lower portion of the reservoir.

Recommendations

We will continue the study as planned. During spring of 2012 we will attempt to implant another 35 - 40 striped bass with acoustic transmitters. Striped bass movements will be monitored with our receiver array and by manually tracking fish throughout the year.

Prepared By: Jason Bettinger Title: Wildlife Biologist III

Job Title: Functional Relationships of Macroinvertebrate Community Metrics with Instream and Landscape Factors from the South Carolina Stream Assessment

Period Covered October 1, 2010 – September 30, 2011

Introduction

Under current understanding of watershed processes, stream hydrology, chemistry, geomorphology and biotic assemblage structure are tightly linked with a suite of regional terrestrial factors including climate, vegetation, slope, elevation, geology and land cover/land use (Johnson and Gage 1997). Because stream ecosystems are so linked with their terrestrial watersheds, disturbances of land surfaces cumulatively lead to alterations of water quality, instream habitat, and biological communities within their corresponding aquatic ecosystems (Allan et al. 1997, Scott et al. 2002). The identification and prediction of land use impacts on stream habitats, channel form and biotic assemblages constitutes an essential component of aquatic ecosystem management (Richards et al. 1996).

Aquatic macroinvertebrate communities can be useful indicators of water quality because they respond to integrated stresses over time, and reflect fluctuating environmental conditions. Community response to various pollutants (e.g. organic, toxicants, fine sediments) may be assessed through interpretation of diversity, known organism tolerances and, in some cases, relative abundance and feeding behavior types. In this case, our intent was to use macroinvertebrate community indicators of ecological integrity from randomly selected watersheds across the target river basins to determine the levels at which disturbance in the watershed leads to biological degradation. Knowledge of these functional responses, especially the existence of thresholds, is critical to designing effective BMPs and mitigation strategies. Reductions in nonpoint source pollution can be achieved through use of GIS-based tools that quantify these responses on the landscape, leading to efficient controls by recommendation and enactment of greenspace ratios, riparian buffer ordinances, or sustainable development zoning.

The necessary components of such an endeavor include 1) information on ecological response variables across a gradient of land use intensities using a statistically-valid sampling design; 2) statistical techniques to estimate complex ecological cause-effect relationships and interactions, along with an assessment of uncertainty; and 3) a decision-support framework in which information can be utilized effectively and communicated to policy makers, planners, and managers. Here, we utilize a subset of data collected under the the probabilistic design of the South Carolina Stream Assessment (Scott 2008) to derive functional relationships of macroinverbebrate community metrics to stream and watershed factors.

Materials and Methods

Stream Data

Data from ninety-three randomly-selected stream sites in the Savannah and Saluda River basins were included in the analysis; however, not all sites had a full complement of variables associated with them. We wanted to assess the predictive capability of the wide range of stream and watershed factors so eight sites missing data were dropped from further analysis, leaving thirty-two sites from the Saluda River and fifty-three sites from the Savannah basin. Data were collected in 2008 according to South Carolina Stream Assessment (SCSA) Standard Operating Procedures (SCDNR 2009). Channel form measurements were accomplished by establishing surveyed cross sections at each sample site. Morphological calculations such as ratios of width to depth, bank height, and cross sectional area were determined. Availability of stable habitat is a critical factor in the development and persistence of benthic macroinvertebrate communities; we used the EPA Rapid

Bioassessment Protocol (RBP) to evaluate in-stream habitat. Benthic macroinvertebrate sampling at each site will consist of 15 man minutes sampling in all available habitat types. In addition, there will be a 15 man minute visual inspection and collection from rocks, woody debris, aquatic macrophytes and other stable substrates. Benthic macroinvertebrates and associated detritus will be collected in 500 micron mesh nets and sampled material from different habitat types will be pooled as a single sample. All sampled material will be preserved in ethanol and returned to the laboratory where benthic macroinvertebrates will be picked from the detritus and identified to the lowest possible taxonomic level. The benthic macroinvertebrate sampling protocol and methodology was accomplished through our collaboration with Dr. Rockie English's laboratory at Clemson University, which is certified by South Carolina Department of Health and Environmental Control (SCDHEC). Benthic macroinvertebrate data was organized and analyzed to provide taxa richness, Ephemeroptera, Plecoptera and Trichoptera (EPT) richness, SCDHEC values for bioassesment and calculated water quality tolerance values.

Watershed Data

We utilized ESRI's ArcGIS v. 9.3 to generate catchments associated with each sample location; catchments included the entire drainage area upstream of stream sample sites. We delineated catchments using highest resolution available digital elevation models, which were produced by combining SCDNR lidar data (3m) and U.S. Geological Survey (http://seamless.usgs.gov) seamless digital elevation models (10m, 30m). Land cover categories for 2006 produced by the Environmental Protection Agency's (EPA) Multi-Resolution Land Characteristics Consortium (MRLC) were extracted at the catchment scale (land use 'x' (km^2)/total catchment area (km^2). Land use categories were defined and categorized according to the USGS NLCD 1992-2001 Land Cover Change Retrofit Product (Fry et al. 2009). All map layers were projected in the Universal Transverse Mercator (UTM) projection system (zone 17 N), using the North American 1983 datum.

Data Analysis

Traditional statistical approaches to the prediction of biological indicators are challenged by datasets that have large numbers of predictor variables with complex covariance structures and high-order interactions, and are constrained by linear assumptions (many environmental variables have naturally skewed distributions; King et al. 2005, Carlisle et al. 2009). Machine learning techniques provide an alternative modeling paradigm to traditional statistics, where no a priori model is defined, and complex data structures (non-normal distributions, interactions) are permitted. Machine learning techniques use an algorithm to learn the relationship between the response and its predictors by identifying dominant patterns in the dataset (Breiman 2001, Elith et al. 2008).

We applied Random Forests which represent an advance in machine learning techniques that have increased the accuracy and prediction power of single classification and regression trees by the creation of an ensemble of trees (Breiman 2001). Random forests are non-parametric, can handle both categorical and continuous data as either predictor and/or response variables, can handle high-order interactions, and are insensitive to outliers (Breiman 2001, De'ath and Fabricius 2000, Urban 2002). Random Forests fit an ensemble of trees to a dataset, where each individual tree in the forest is built using a randomly selected bootstrap sample of the training dataset. In addition, only a random subset of predictor variables is considered for node and splitpoint selection (Amit and German 1997). In this way, two elements of randomness are injected into the procedure. Observations not included in the bootstrap samples are passed down their respective trees, and each tree's terminal nodes contain mean responses to different combinations of observed values among predictor variable pathways. Each tree has a 'vote' in the most important predictive variables to split on, and on the mean responses of different values of input combinations; and the majority of votes among the ensemble of trees 'wins'. Therefore, we can a) predict and rank variables that most strongly influence the outcome, b) predict the mean outcome based on different values among variable combination pathways, and c) isolate and examine the behavior of individual predictors on the outcome, while holding the effect of all other predictive variables constant.

We applied four regression RF models to our invertebrate dataset for the Saluda and Savannah basins; one for each of our four biological indicator outcome variables. We used the pseudo-R^2 and the root-mean-squared error (RMSE) as our model performance measures. Top-ranked variables for predicting biological indicators as measured by Random Forests were visualized with variable importance plots. The importance measure shows how much mean square error is increased when a given variable is randomly permuted. Important variables will change the regression prediction more than insignificant variables when randomly permuted, therefore larger increases in mean square error indicate more important variables. We plotted relationships between the response and individual predictors using partial dependence plots, which allowed us to visualize the individual contribution of a predictive variable while holding the effect of all other predictive variables constant. Although we can visualize the relationships between individual predictor variables and the response, it remains important to note that variables work in synchrony to predict the actual mean outcome.

Results

Taxa Richness

Macroinvertebrate taxa richness ranged from 15 to 102 at the stream sites. A random forest regression model explained nearly 24% of variation in taxa richness among sites. Figure 1 shows variable importance plots for the random forest model (regression model is on the left, disregard plot on right as this pertains to classification strength). This plot indicates that scoring on EPA Rapid Bioassessment Protocol (RBP) habitat parameter number 5. Channel Flow Status was the most important predictor, all others held constant, indicating that at least some flow in the channel is important for taxa richness. Figure 2 depicts partial dependence plots showing relationships between the response and individual predictors, which allows visualization of the individual contribution of a predictive variable while holding the effect of all other predictive variables constant. From Figure 2 it is apparent that taxa richness decline steadily as RBP parameter 5 Flow scores drop below about 7, a score below which reflects water flow in less than 25% of the stream channel (taken from RBP field assessment form description). Urban land cover in the watershed was also an important predictor (Figure 1), with greatest taxa richness at the lowest levels of urbanization and a threshold effect at about 20% urban land cover (Figure 2). RBP parameter 10 Riparian zone width scores and stream wetted width were also important predictors.

EPT

Taxa richness of Ephemeroptera, Plecoptera and Trichoptera (EPT) ranged from 0 to 40 at the stream sites. A random forest regression model explained 59.4% of variation in EPT richness among sites. Figure 3 shows variable importance plots for the model, indicating that water

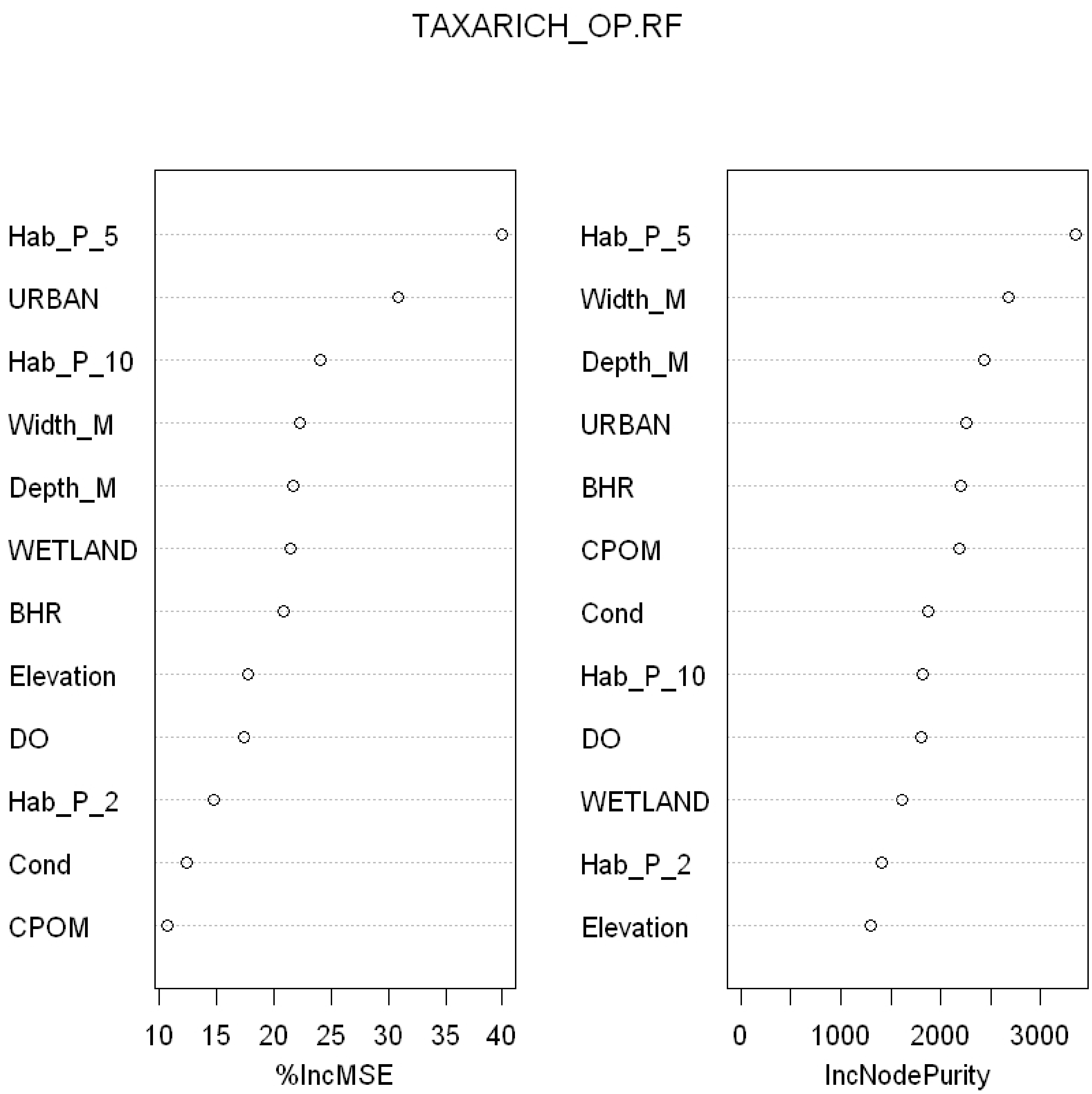

Figure 1. Variable importance plot for taxa richness random forest regression model (left) shows how much mean square error is increased when a given variable is randomly permuted. Important variables will change the regression prediction more than insignificant variables when randomly permuted, therefore larger increases in mean square error indicate more important variables. The random forest regression model explained 23.9% of variation in taxa richness among sites.

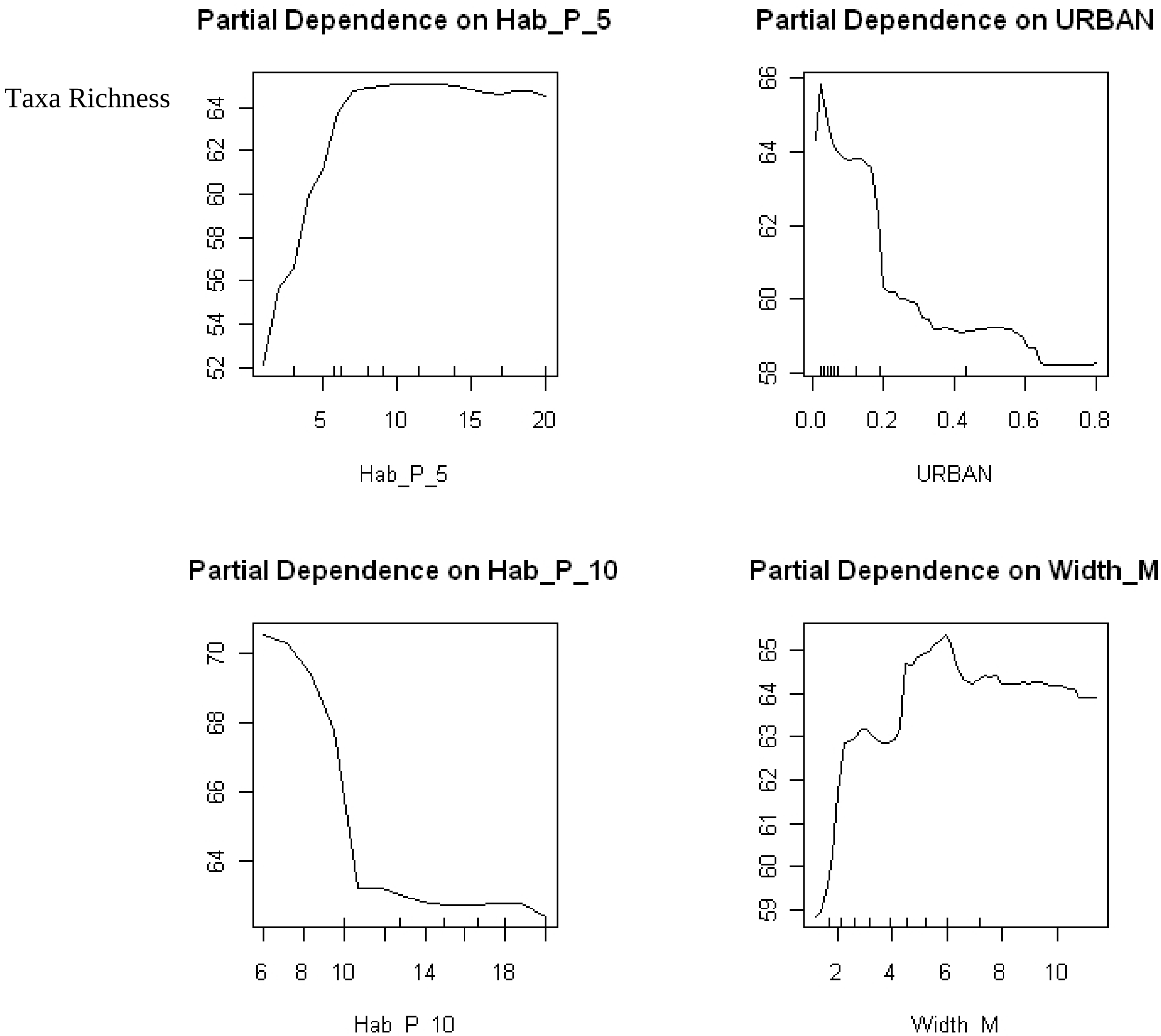

Figure 2. Partial dependence plots showing response of mean taxa richness to individual predictors, which allows visualization of the effect of a predictive variable while holding the effect of all other predictive variables constant.

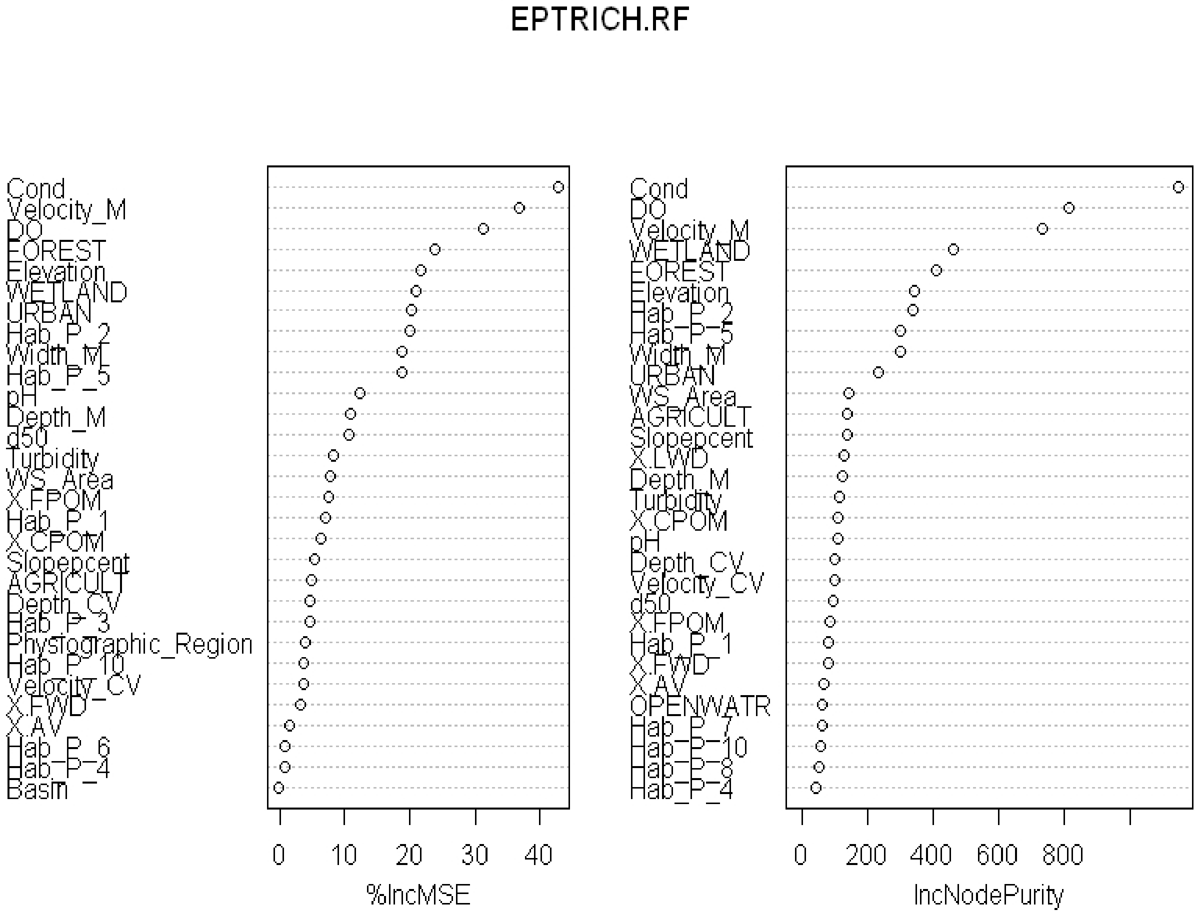

Figure 3. Variable importance plot for EPT richness random forest regression model (left) shows how much mean square error is increased when a given variable is randomly permuted. Important variables will change the regression prediction more than insignificant variables when randomly permuted, therefore larger increases in mean square error indicate more important variables. The random forest regression model explained 59.4% of variation in EPT richness among sites.

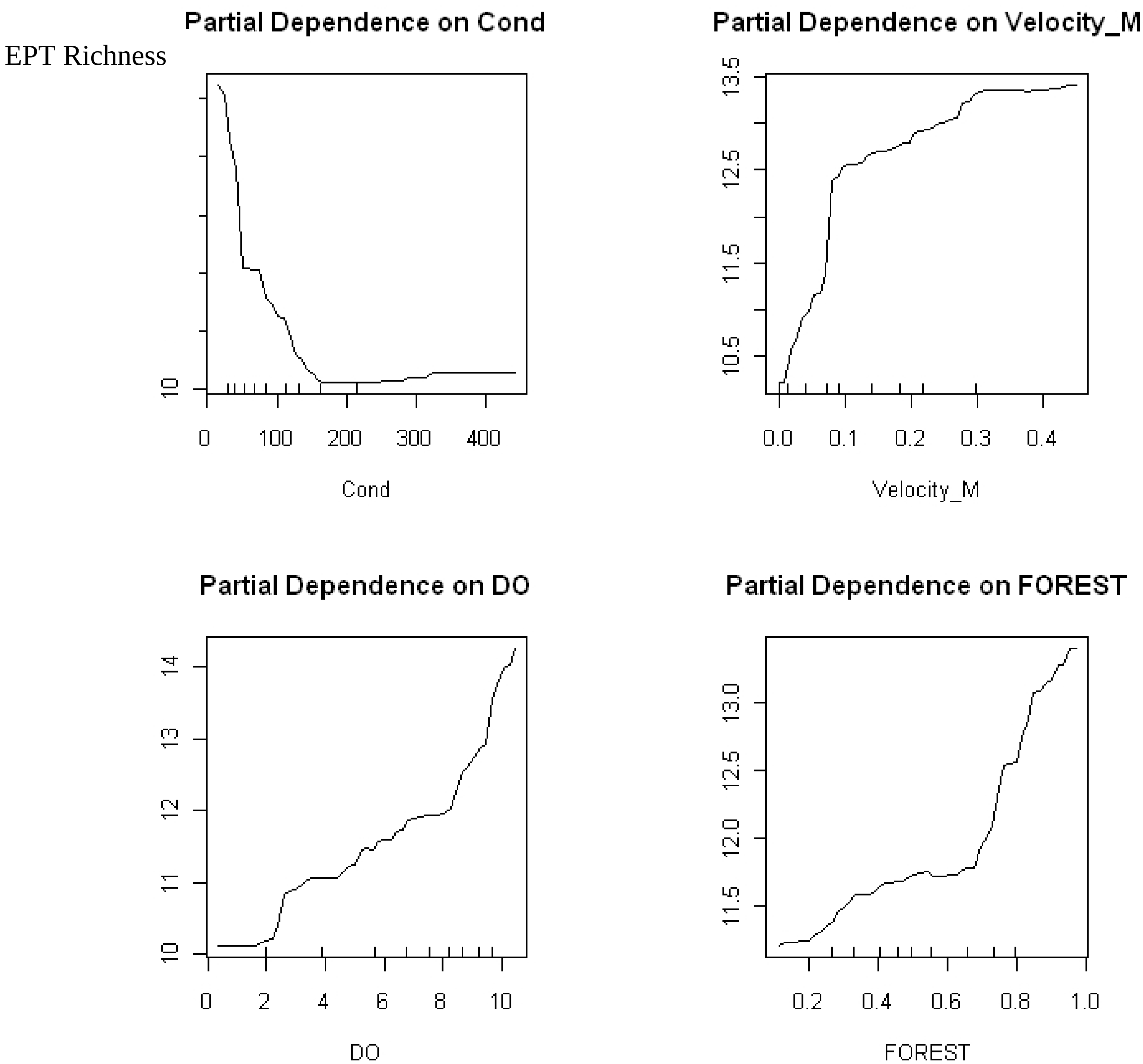

Figure 4. Partial dependence plots showing response of mean EPT richness to individual predictors, which allows visualization of the effect of a predictive variable while holding the effect of all other predictive variables constant.

conductivity, mean water flow velocity, dissolved oxygen concentration, and forest land cover in the watershed were the most important predictors. Partial dependence plots (Figure 4) show the functional relationships of the four most important predictors, showing that EPT richness is higher on average at lower conductivity levels and at greater water velocities. The similarity of dissolved oxygen and forest cover relationships is striking, suggesting some link in these predictors. In terms of cause and effect, it seems logical to surmise that high levels of forest cover creates conditions of higher dissolved oxygen and lower conductivity in stream water, to which the sensitive EPT taxa respond directly.

Tolerance Values

Tolerance values for macroinvertebrate taxa ranged from 2.15 to 8.93 at the stream sites. Under these calculations, high tolerance values correspond to greater dominance by taxa tolerant of poor water quality; low values correspond to a more sensitive community. A random forest regression model explained 73.4% of variation in tolerance values among sites. Figure 5 shows variable importance plots for the model, indicating that water conductivity, dissolved oxygen concentration, mean water flow velocity, and RBP habitat metric 5. Channel flow status were the most important predictors. Partial dependence plots (Figure 6) show similar but inverse patterns as the plots for EPT (Figure 4). As Figure 5 shows, forest cover was also an important variable for tolerance value prediction and further suggests its importance as a driver of biological integrity.

Bioassessment Values

Bioassessment values for macroinvertebrate taxa are calculated by DHEC to reflect whether a community meets 303(d) criteria; they ranged from 1 to 5 at the stream sites. A random forest regression model explained 62.3% of variation in bioassessment values among sites. Figure 7 shows

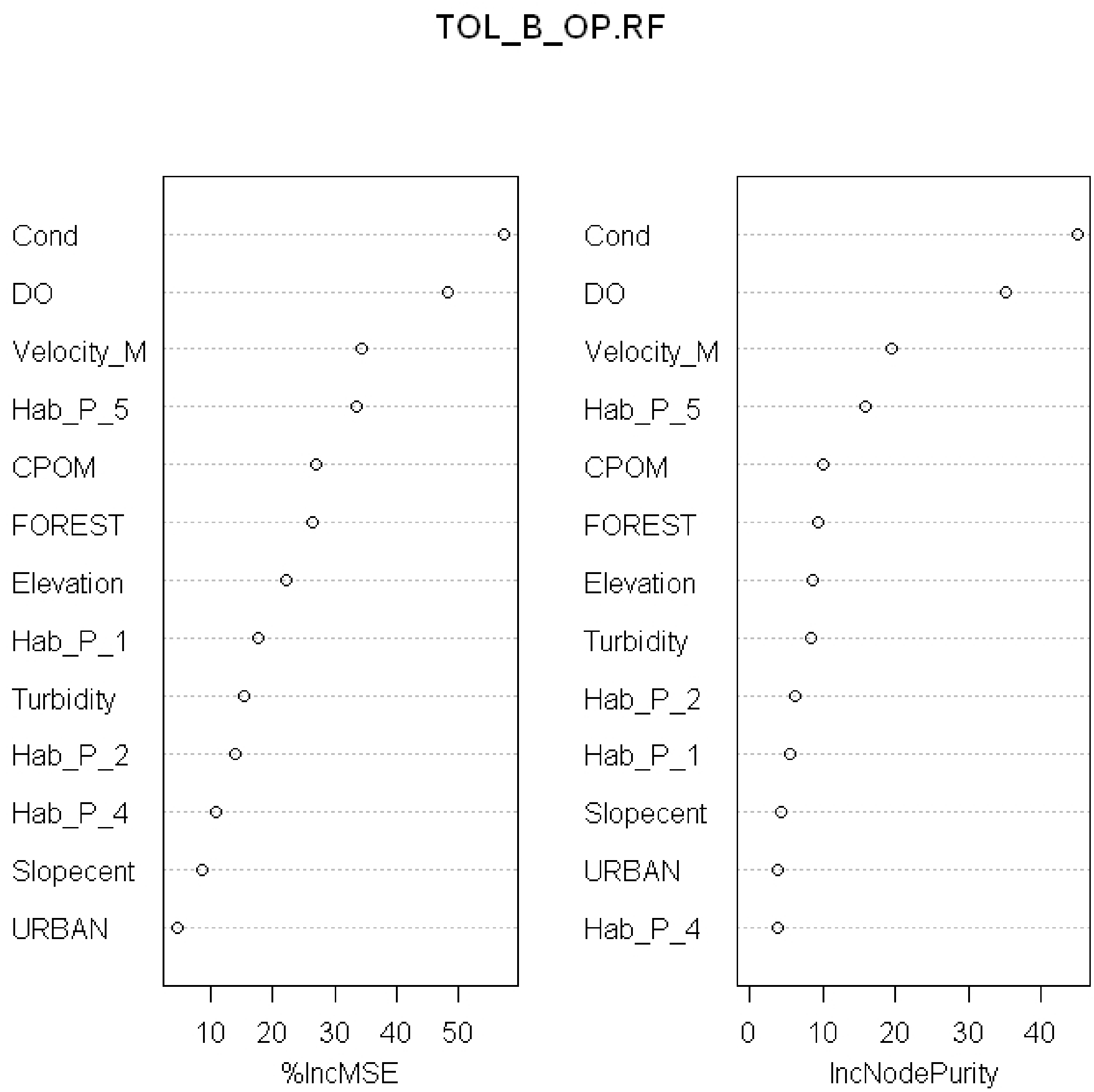

Figure 5. Variable importance plot for the random forest regression model (left) to predict water quality tolerance values of macroinvertebrate taxa shows how much mean square error is increased when a given variable is randomly permuted. Important variables will change the regression prediction more than insignificant variables when randomly permuted, therefore larger increases in mean square error indicate more important variables. The random forest regression model explained 73.4% of variation in tolerance values among sites.

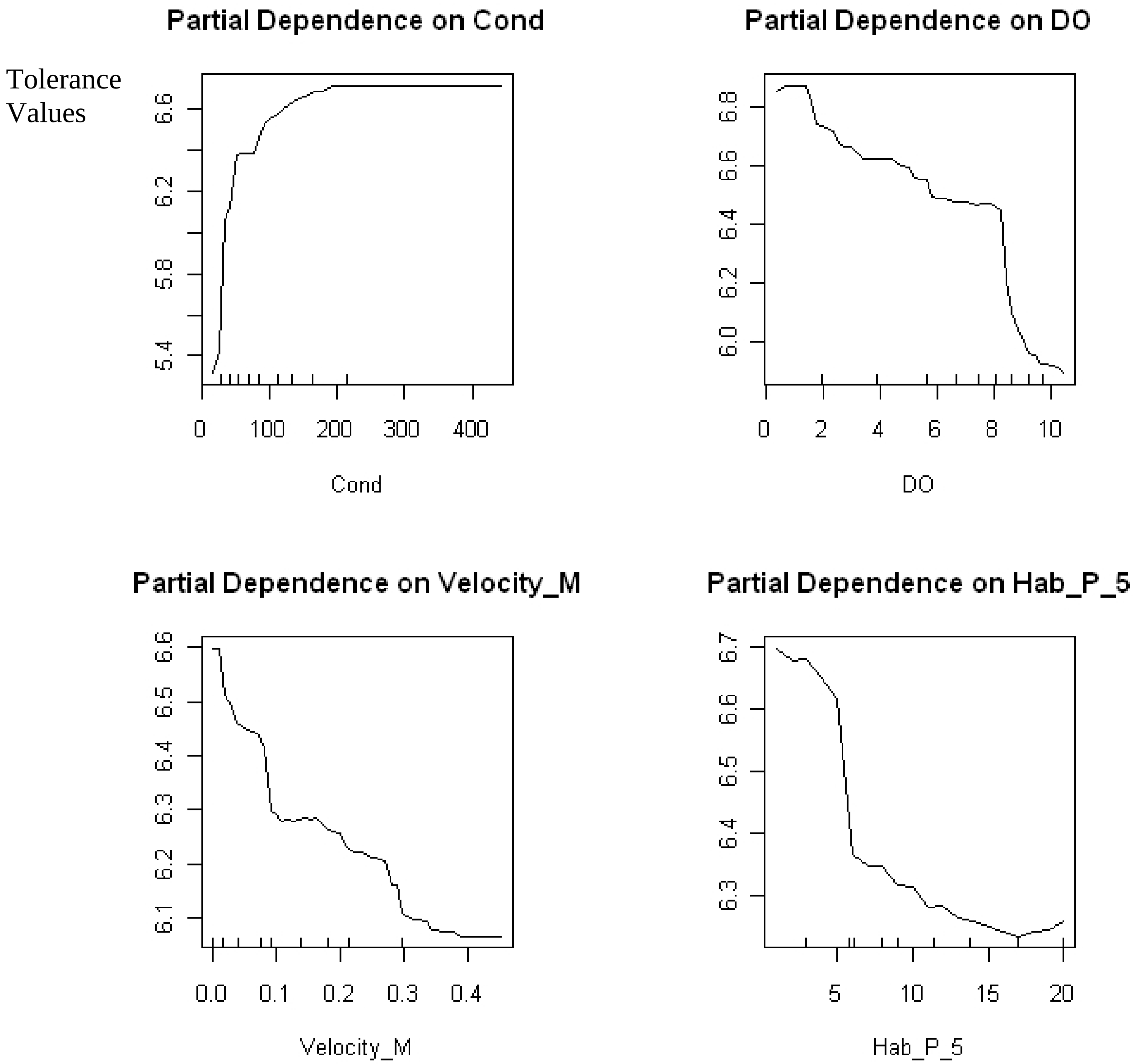

Figure 6. Partial dependence plots showing response of mean tolerance values to individual predictors, which allows visualization of the effect of a predictive variable while holding the effect of all other predictive variables constant.

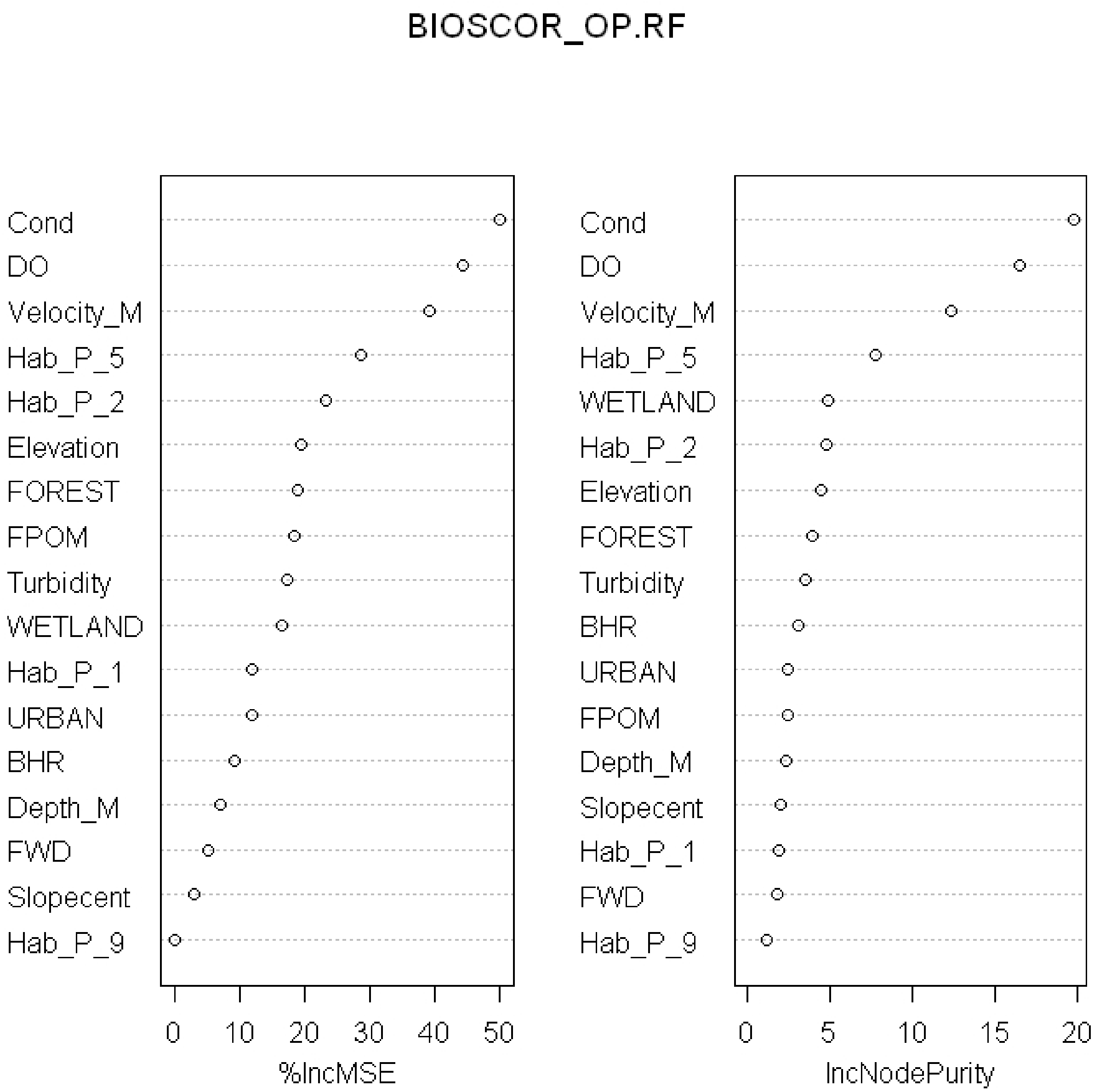

Figure 7. Variable importance plot for bioassessment value random forest regression model (left) shows how much mean square error is increased when a given variable is randomly permuted. Important variables will change the regression prediction more than insignificant variables when randomly permuted, therefore larger increases in mean square error indicate more important variables. The random forest regression model explained 62.3% of variation in bioassessment values among sites.

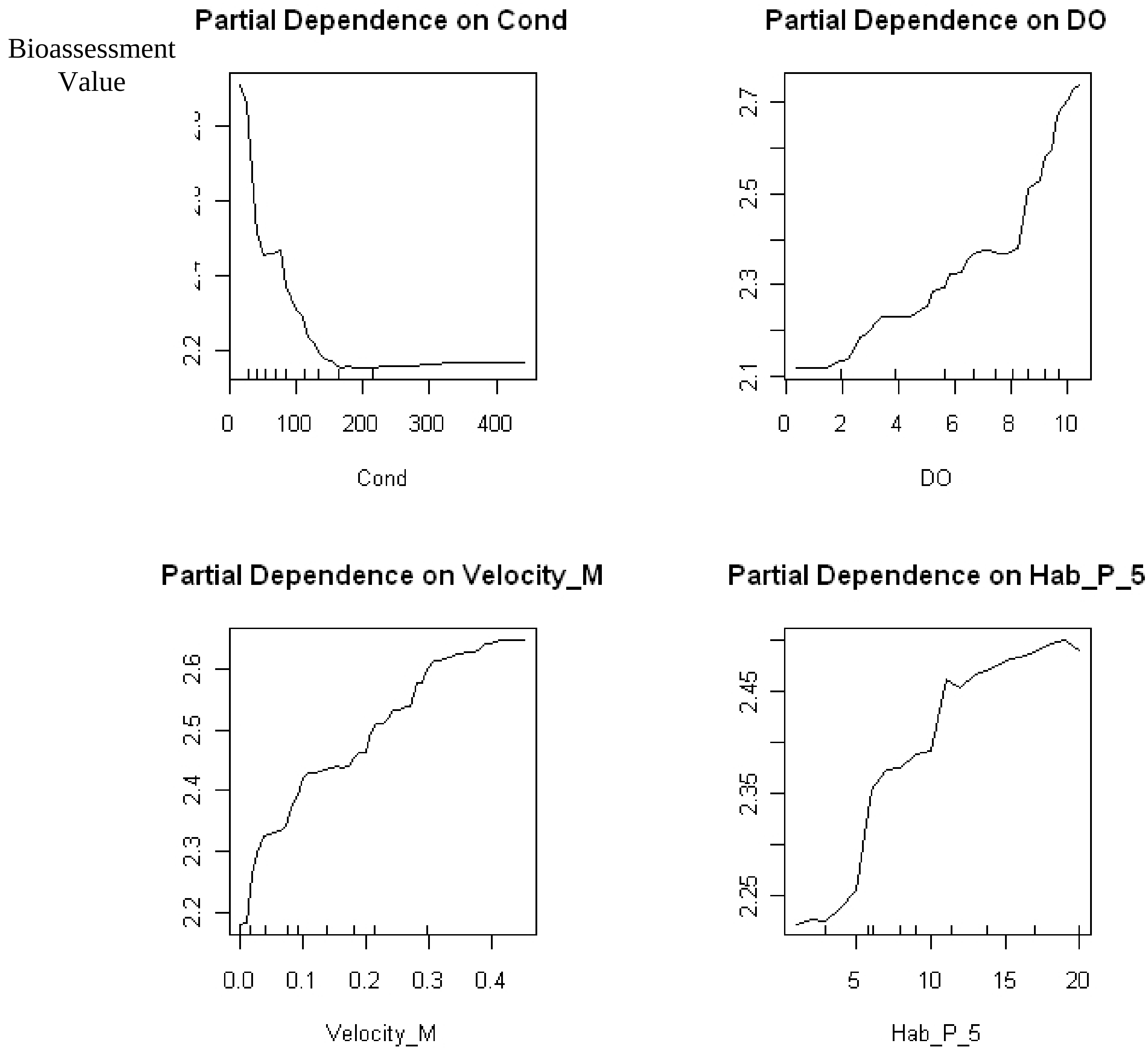

Figure 8. Partial dependence plots showing response of mean bioassessment value to individual predictors, which allows visualization of the effect of a predictive variable while holding the effect of all other predictive variables constant.

variable importance plots for the model that reflect very similar results as for tolerance values. Partial dependence plots (Figure 8) show very similar responses as the EPT model (Figure 4), which paints an overall picture of the macroinvertebrate community relationship to stream and watershed factors in Savannah and Saluda basin streams that is quite robust across metrics.

Discussion

The models derived here demonstrate that biological integrity, as indicated by macroinvertebrate assemblage metrics currently in use by DHEC, can be predicted from stream and watershed factors. Significantly, functional relationships were shown to be nonlinear in most cases, with breaks and thresholds in responses that can be identified and serve as management targets. The importance of water quality (e.g., dissolved oxygen, conductivity), habitat condition (channel flow status, riparian zone), and watershed land cover (urbanization, forest cover) to determining biotic metrics was shown. Controlling variables can deduced from the results here coupled with broad evidence from other studies: loss of forest from 100% down to about 70% of a watershed is associated with a steep decline in sensitive taxa, and increased urban land cover beyond 20% of a watershed is associated with steep taxa losses, other things being equal. The relationships detailed here suggest that watershed protection and aquatic resource management can move from a reactive mode into a proactive one, where these relationships are employed in decision-support tools that can make spatially-explicit forecasts of effects under different development or management scenarios.

Recommendations

This report summarizes recent results from analysis of SCSA sample data, focusing on macroinvertebrate assemblage metrics. Forthcoming analyses will center on stream fishes, both species of conservation priority as well as metrics of taxonomic and functional integrity. Of interest will be the extent of correspondence in the macroinvertebrate and fish data to stream and watershed factors, and to set overall criteria to protect stream systems. These criteria will assist biologists and resource managers in assigning conservation status in future efforts such as revisions of the Comprehensive Wildlife Conservation Strategy. Furthermore, we aim to deploy watershed-scale models of stream condition within decision-support applications that can be utilized by resource agencies, land planning bodies, and grassroots conservation organizations alike.

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Prepared By: Mark Scott with Cathy Marion Title: Fisheries Biologist

Job Title: Assessing introgressive hybridization within and habitat requirements of native South Carolina redeye bass

Period Covered October 1, 2010 – September 30, 2011

Summary

In 2003-2004 black bass were collected from stream and reservoir populations in the Savannah drainage, and from one stream population in Santee drainage, to assess presence of non-native species and hybridization with native redeye bass *Micropterus coosae*. To assess change in these populations over time, collections were repeated in 2009-2010. In the last year, genetic analysis of N= 907 black bass was completed. Comparisons to 2003-2004 data indicate a continued decline in redeye bass in reservoir populations, and an increase in both hybrids Alabama spotted bass *M. henshalli*. New data also confirms presence of hybrids in some stream populations where they were not collected previously, indicating increased spread of the Alabama spotted bass impact on native redeye populations. Work to determine the origin of redeye bass populations in the neighboring Santee drainage and their status with respect to hybridization has continued with collections from new sites on the Enoree river. Genetically confirmed redeye bass as well as hybrids with Alabama spotted bass were collected from all sites sampled.

Introduction

The redeye bass *Micropterus coosae* (Hubbs and Bailey 1940) is one of two black bass native to South Carolina, and has been identified by South Carolina's Comprehensive Wildlife Conservation Strategy as a Species of Highest Priority due its restricted range and threats from introduced species (Kohlsaat et al. 2005). The species' native range is restricted compared to others of its genus and includes the Savannah, Altamaha and Ogeechee River drainages on the Atlantic slope, and the Mobile Bay and Apalachicola drainages on the Gulf slope. Redeye bass occupy habitats above the Fall Line in fast moving, cool-water streams (Rhode et al. 2009). In addition to native headwater streams and tributaries, *M. coosae* has thrived within four of the Savannah River basin's man-made reservoirs; Jocassee, Keowee, Hartwell and Russell (Koppelman and Garret 2002).

Recent studies have examined the relationship among populations of redeye bass across the range of the species. Mobile Bay drainage redeye bass are morphologically distinct from Atlantic Slope populations, with the common name Bartram's bass assigned to the latter (Bud Freeman, unpublished data). DNA sequence data supports this distinction, and further suggests species-level divergence between Savannah River redeye bass and those of other Atlantic Slope drainages. Savannah River redeye bass represent a highly divergent and distinct evolutionary lineage (Oswald 2007).

Introductions of the non-native Alabama spotted bass (*Micropterus punctulatus henshalli*) into lakes Keowee and Russell have put Savannah River redeye bass at risk due to introgessive hybridization (Barwick et al. 2006). A 2004 genetic survey showed that Alabama spotted bass have expanded within the drainage, as have their hybrids with redeye bass (Oswald 2007). Both are present in all four lakes surveyed. While the survey of tributaries of the drainage showed that those redeye populations were for the most part still unimpacted by hybridization, spotted bass are known to take advantage of stream habitats, and the continued spread of Alabama spotted bass and their hybrids throughout the drainage is a possibility.

Objectives of this study include repeat sampling of redeye bass populations surveyed in 2004 and an assessment of genetic change over time, and a genetic evaluation of redeye bass and other co-distributed species in Santee drainage to further evaluate the redeye's status in Santee drainage as introduced. Work in the last year has focused on completion of sequencing of collected Savannah drainage individuals, compilation of genetic data, comparison to 2004 survey data, and new Santee drainage collections.

Materials and Methods

Genetic sequences were generated for black bass collected from Savannah drainage reservoirs and tributary populations in 2009 and 2010. For all fish collected, sequences were generated for one mitochondrial and three nuclear dna loci following the procedures outlined by Oswald (2007). Fish were placed in one of six classifications based on their genetic profile; redeye bass, largemouth bass, Alabama spotted bass, smallmouth bass, redeye x Alabama spotted bass hybrid, or other hybrid. Hybrid individuals were further classified as F1 or backcross hybrids.

For reservoir collections that duplicated the 2004 survey, proportions of fish in each classification were calculated for individual sampling sites, and the average proportion of each classification across sites was calculated for each reservoir. Genotype scores were generated by Max Bangs, Master's student at University of South Carolina, for individual redeye bass, Alabama spotted bass, and hybrids between these two species, by counting the number of Alabama spotted bass alleles at each of the three nDNA loci examined. Scores ranged from 0 for 'pure' redeye bass to 6 for 'pure' Alabama spotted bass. A histogram of genotype scores was generated, with color coded bars to designate proportions of mitochondrial DNA from each species. To assess change in species and genome composition over time, results were compared to 2004 survey data. For tributary collections, genetic data was examined for any new incidences of Alabama spotted bass or hybrids.

Fish were collected from new Santee drainage sites. Following a first recorded collection of a redeye bass from Enoree River, a desk inventory of shoal habitats in Enoree River was conducted using Google Earth. Accessible areas were scouted and selected for sampling. Fish were collected by backpack electrofishing. For all black bass encountered and collected, field identification, total length and weight were recorded. Fish were photographed, and fin clips were taken and stored in 100% non-denatured ethanol for genetic analysis. Fish were then stored frozen for future reference. Sequencing was completed and fish were identified to species or hybrid status based on sequencing results as described above.

Results

Sequencing was completed for N=907 black bass collected in 2009 and 2010, including collections from Lakes Jocassee, Keowee, Hartwell and Russell, and from 18 tributary populations from the Savannah and Santee drainages. Fish were analyzed from all reservoir sites sampled. Data for two Savannah tributary populations surveyed was not completed due to a machine malfunction during sequencing.

Data from our reservoir samples show a precipitous decline in redeye bass collected from two reservoirs. Our 2004 survey indicated redeye bass had been virtually eliminated from Lakes Keowee and Russell, where they comprised 0% and 2% of black bass collected, respectively. Collections in 2010 show little change in redeye bass proportions from these two lakes, but a decline is evident in Lakes Hartwell and Jocassee. For Lake Hartwell redeye bass comprised 26% of fish collected in 2004 and 8% in 2010. There is a corresponding increase in hybrids between redeye and Alabama spotted bass, from 26% to 43%. For Lake Jocassee genetically pure redeye bass comprised 39% percent of black bass collected in 2004, and only 14% in 2010. Hybrids between redeye and Alabama spotted bass increased, from 29% in 2004 to 54% in 2010. Also of interest on Lake Jocasee is new collections of hybrids between redeye bass and smallmouth bass, which comprised 5% of fish analyzed from the lake in 2010 (Table 1).

Table 1. Percent composition of black bass species classifications collected from Lakes Jocassee, Keowee, Hartwell and Russell in 2004 and 2010; redeye bass (REB), Alabama spotted bass (ASB), largemouth bass (LMB), smallmouth bass (SMB). Percent values are reported as proportions for each lake, and are average proportions over all sites sampled.

Species	Jocassee		Keowee		Hartwell		Russell	
	2004 N=127	2010 N=140	2004 N=161	2010 N=137	2004 N=171	2010 N=183	2004 N=144	2010 N=172
REB	.39	.14	-	<.01	.26	.08	.02	-
ASB	-	.01	.25	.26	<.01	.05	.17	.27
REBxASB	.29	.54	.38	.42	.26	.43	.37	.27
LMB	.20	.15	.37	.31	.47	.42	.44	.46
SMB	.12	.11	-	-	-	-	-	-
Other hybrids	-	.05	-	<.01	-	.01	-	<.01

In examining the proportions of redeye bass, Alabama spotted bass and their hybrids alone, an increase in pure Alabama spotted bass is evident for Lakes Hartwell and Russell. The proportion of fish in this classification from Lake Hartwell increased from 1% in 2004 to 10% in 2010. From Lake Russell proportions increased from 30% to 50% (Table 2).

Table 2. Average proportions of redeye bass (REB), Alabama spotted bass (ASB) and hybrids between the two species collected from Lakes Jocassee, Keowee, Hartwell and Russell in 2004 and 2010. Proportions reported for each lake are average proportions over all sites sampled.

Species	Jocassee		Keowee		Hartwell		Russell	
	2004 N=86	2010 N=97	2004 N=101	2010 N=90	2004 N=89	2010 N=102	2004 N=81	2010 N=93
REB	.57	.20	-	.01	.49	.14	.04	-
ASB	-	.02	.39	.37	.01	.10	.30	.50
REBxASB	.43	.78	.61	.62	.49	.77	.66	.50

Genotype scoring results, similarly to species proportions, reflect little change for Lake Keowee. All other reservoir populations show a decrease in 'pure' redeye bass (score of 0) and a corresponding increase in 'pure' Alabama spotted bass (score of 6). Over all scores increased over time for each lake. This is most notable for Lake Hartwell which scored 1.3 in 2004 and 3.2 in 2010. Increased genotype scoring reflects the increase in fish classified as pure Alabama spotted bass, as well as an increase in the segment of Alabama spotted bass alleles among hybrid individuals From each population in 2010, a portion of the individuals scoring as pure Alabama spotted bass possessed redeye bass haplotypes at the mitochondrial locus, indicating a portion of fish classified as pure Alabama spotted bass are actually the result of high order backcrossing (Figure 3, Bangs 2011).

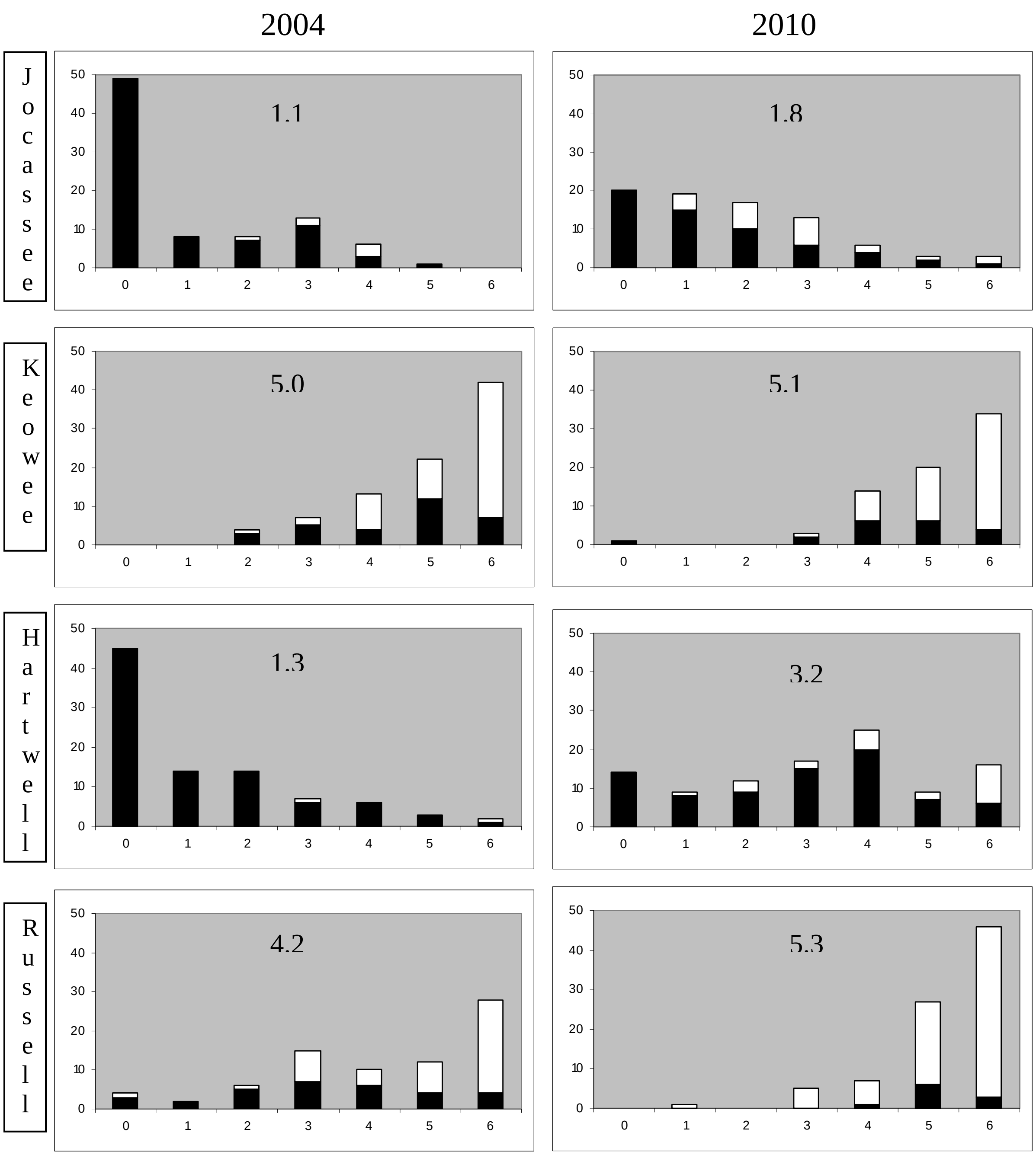

Figure 3*: Genotype scoring across three nDNA loci. Numbers on the x-axis represent the total number of ASB alleles across three nDNA loci, so 0 represents “pure REB” and 6 represents “pure ASB”. White bars represent ASB mtDNA and black bars represent REB mtDNA. Y-axis is number of individuals. The number at the top of each graph is the average genotype score for that lake at that time period.

*From Bangs 2011.

Genetic anlaysis confirmed non-natives and/or hybrids from 5 tributary sites in 2010, an increase since the 2004 survey. Hybrids were collected from Little River, Little Coldwater Creek, Savannah River at Augusta Shoals, and from the lower sampling site on Chauga River. One fish collected from Big Generostee Creek was homozygous for an Alabama spotted bass allele at one locus, but incomplete data prevents classifying it as pure or hybrid. In 2004 Little River was the only tributary population of these from which we collected non-natives or hybrids. All other 2010 collections, including at an upper Chauga River site, yielded only pure redeye or largemouth bass (Table 4).

Table 4. Genetic identifications of black bass collected from Savannah and Santee (Saluda River at Pelzer) Drainage streams in 2009 and 2010; redeye bass (REB), largemouth bass (LMB), Alabama spotted bass (ASB), smallmouth bass (SMB), hybrid (HYB).

		Species (N)					
Stream	Date	REB	LMB	ASB	ASBxREB	SMB	SMBxREB
Steven’s Ck.*	7/29/09	-	-	-	-	-	-
Big Generostee Ck.*	7/30/09	-	-	-	-	-	-
Saluda River	9/9/09	16	9	0	0	0	0
Eastatoee Ck.	9/24/09	8	0	0	0	0	0
Chauga River lower	9/14/09, 9/29/09	9	2	0	8	0	0
Chauga River - upper	9/29/09	15	0	0	0	0	0
Little River Lower**	9/30/09	-	-	-	-	-	-
Little River upper	9/30/09	2	0	0	2	0	0
Chatooga River	8/4/10	18	0	0	0	0	0
Little Coldwater Ck.	9/1/10	18	3	0	3	0	0
Savannah River	9/16/10	15	4	0	0	6	2

*Genetic data from Steven’s Creek and Big Generostee are inclomplete.
**No fish were collected from the Little River lower site.

Seven shoal habitat sites were identified on the Enoree River for black bass collections. Sites were included from areas both above and below the first collection site at Riverdale Mills. The lower most site was at Musgrove Mill near Clinton. The upper most site was above Gibb Shoals, near the town of Greer. Redeye bass were field identified from all sites sampled (n=57). Genetic analysis confirmed species status for n = 50, while n = 7 were found to be redeye bass x Alabama spotted bass hybrids. No pure Alabama spotted bass were collected. Redeye bass were the most commonly encountered black bass (Table 5). Largemouth bass (n=3) were also collected. No smallmouth bass, or redeye x smallmouth bass hybrids were collected, though smallmouth are common in the Broad River, which the Enoree River is a tributary of.

Table 5. Species identifications for black bass collected from the Enoree River in 2010 and 2011; redeye bass (REB), redeye bass x Alabama spotted bass (REB x ASB), largemouth bass (LMB).

			Species		
Site	Site Coordinates	Date	REB	REB x ASB	LMB
Musgrove Mill	34.594096, -81.854859	12 May 2011	6	1	0
Hwy 49	34.603519, -81.910633	12 May 2011	4	1	0
Riverdale below dam	34.649847, -81.960254	11 Nov 2010	8	0	1
Riverdale above dam	“ “	11 Nov 2010	5	0	0
Hwy 418	34.80691, -82.165668	23 May 2011	25	4	0
Pelham Falls	34.853782, -82.221352	19 May 2011	1	0	0
Gibb Shoals	34.871623, -82.238625	9 Jun 2011	1	0	0
Above Gibb Shoals	34.876518, -82.245847	9 Jun 2011	0	1	2

Discussion

The decline in proportions of redeye bass in Lakes Hartwell and Jocassee indicates these populations are moving in the same direction as Keowee and Russell genetically. Considering this, the outlook for the continued presence of redeye bass in Savannah drainage reservoirs is certainly grim.

It is interesting to note that while the proportions of black bass from the reservoirs classified as pure Alabama spotted bass has increased, we are not able to confirm what of these are actually pure. Genotype scoring results shows that these lake populations are producing fish homozygous for Alabama spotted bass alleles at all nuclear loci, which are still clear backcrosses due to their redeye bass mtDNA. The increase over time in Alabama spotted bass mitochondrial haplotypes among hybrids, as redeye haplotypes become more rare, would indicate that a number of fish classified as pure Alabama spotted bass are actually the result of high order backcrossing. At any rate, the increase in Alabama spotted bass alleles among hybrids in the reservoirs, and the increased occurrence of 'pure' Alabama spotted bass indicate that over time the redeye bass genome in these reservoirs will be completely replaced by that of Alabama spotted bass.

Any new collection of non-native bass in redeye bass streams is disturbing in that it represents the potential for loss of a pure population through introgression. It also documents further spread of these species within the Savannah drainage, and highlights the need for public education on the ramifications of such species introductions. The increased incidence of Alabama spotted bass alleles in tributary populations is important, and warrants further study. While hybrids were collected previously from Little River (Leitner 2009), 2010 collections indicate they have since moved past an area hoped to be a barrier to upstream movement. Chauga River, hybrids were collected below and within a major shoal, but not at the collection site further upstream. In general, upper Savannah tributary sampling sites where non-native or hybrid bass were collected are more closely associated geographically with the reservoirs than those where only native bass were collected. Our current data represents snapshots of the populations in these tributaries, but gives us limited information toward the extent of hybridization throughout these streams. Further study is needed to establish genetic baselines longitudinally within these systems.

Smallmouth bass and there hybrids with redeye bass have been collected previously from the Savannah River at Augusta Shoals (Leitner 2009), and were confirmed again with 2010 collections. As with our upper Savannah tributary sites, our data confirms the presence of a non-native bass and it's hybrids with native redeye bass at one site in the Savannah River. To determine the true impact of this on redeye bass, and to track change over time a more extensive baseline is needed. The shoals at Augusta are difficult to sample and obtaining a representative collection of fish from there is problematic. Still, a more rigorous baseline of the proportion of bass species within the shoals, and the extent of their spread above and below this area would be of value.

Work is continuing toward determining the origin of redeye bass in the Santee drainage. Genetic analysis of fish collected from the Saluda River, near Pelzer, S.C., indicate they were introduced from a Savannah drainage source (Oswald 2007). However, historical collections suggest redeye bass may be native to the Santee drainage (Gilbert 2009). Collections of the species from Enoree River represent new records, though work on the Enoree has been limited, and it is unclear for how long the fish has been present there. Redeye bass have been collected from the Broad River as well. These fish were taken from areas that have been extensively sampled in recent years, and would seem to indicate recent introductions and movement of the species in that portion of the drainage. Whether native or introduced, genetically pure populations of redeye bass in Santee drainage may serve as refuge points for the Savannah genome of the species. Stream team collections in the Santee in 2008 recorded redeye bass in new locations in the Saluda portion of the drainage (Kubach 2008), and these collections are in the process of being analyzed. Collections from Enoree river and the genetic results reported here will also be analyzed, together with a suite of co-distributed species. Divergence will be compared to that found in redeye bass.from the Savannah and Santee drainages.

Recommendations

Complete sequencing of all collected fish. Characterize all fish to species and/or hybrid status. Develop GIS database that incorporates all genetic and spatial data. Examine parameters contributing to presence/absence of Alabama bass or hybrids. Complete collections and genetic analysis of Santee populations of redeye bass and four other species. Examine divergence between the two drainages for each species to assess status of Santee drainage redeye bass as native or introduced. Identify tributary systems for more extensive baseline generation. Identify conservation measures to preserve redeye bass tributary populations in Savannah drainage. Launch education/media campaign that targets movement of fish, and impacts on native species, black bass in particular. Develop partnerships for funding of future work. Write final reports. Continue work to publish earlier and current results.

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Prepared By: Jean Leitner Title: Fisheries Biologist

Job Title: Redbreast Stocking Evaluation – Edisto River

Period Covered July 1, 2010 – June 30, 2011

Summary

A study to evaluate the contribution of stocked redbreast sunfish into the Edisto River fishery was initiated. Broodstocks were established from redbreast sunfish captured by electrofishing from the Edisto River. Produced fingerlings were harvested and immersion marked with oxytetracycline (OTC) in October 2010. Fingerlings (n= 276,300) were stocked into an 11.12 mile section of the main stem of the Edisto River, bounded by SC Hwy 61 and US Hwy 17A. Subsamples of the OTC marked redbreast were grown out until April 2011 for mark evaluation. All field work was performed by Region III staff and is included in that region's progress report. OTC mark evaluations were conducted on a blind set of otoliths including known marked and unmarked redbreast. Marks were confirmed, with an estimated proportion of marked fish of 63% versus an actual proportion of 67%.

Introduction

Redbreast sunfish *Lepomis auritus* is a much sought after sport fish on the Edisto River. Collections made in 2004 spanned a very high water event. Those collections suggest that once hydrologic conditions normalized, allowing for greater river access and angling, larger fish were quickly exploited and removed (Bulak 2005). The annual stocking of redbreast sunfish began in Edisto River in 1995. This was in response to public concerns that introduced flathead catfish were negatively impacting the popular fishery. Records show approximately 13.7 million redbreast stocked in the river since 1995, with annual stocking ranging from 0.45-2.2 million.

The supplemental stocking of redbreast sunfish in Edisto River has never been evaluated. Collections of microtagged redbreast sunfish that were stocked in Little Pee Dee River from 1990 – 1992 suggested minimal contribution, though further sampling was recommended before drawing conclusions from the available data (Crochet and Sample 1993). Genetic survey of redbreast sunfish populations across five South Carolina drainages indicated Edisto river redbreast were markedly less diverse than redbreast populations from other drainages (Leitner 2006). This could be a result of lost diversity in the former hatchery population and its impact on the receiving population in the river, or could be an indication of some bottleneck events occurring in the wild. To best manage this resource, we need a basic understanding of whether supplemental stocking is contributing to the redbreast sunfish population and fishery of the Edisto River.

Materials and Methods

Known marked redbreast sunfish fingerlings from two mark events were provided to this lab by Region III staff. Fingerlings had been grown out for approximately 6 months. Otoliths were processed according to standard procedures for OTC mark evaluation. Additional otoliths from known unmarked redbreast were obtained and used to produce a blind set for evaluation. Mark evaluations were conducted by two independent readers. Otoliths that the two readers did not agree on were excluded from further analysis. The estimated proportion of marked otoliths was calculated.

Results

Ninety percent of otoliths were correctly classified by both readers. The estimated proportion of marked fish in the blind set was 67%, while the actual proportion of marked fish was 63%. It was recommended that the study go forward with collections of the marked year class from the wild, and with marking and stocking of an additional year class.

Discussion

The successful marking of redbreast sunfish has been demonstrated, and is a vital step toward full implementation of this study. Evaluation of wild caught fish should continue forward. As with any study involving the OTC marking of fish, great care should be taken to adhere to section protocols during marking and stocking of subsequent year classes. A sufficient grow out period is essential to evaluation of known marked fish. For sunfish marked in the Fall, this period should span at least 6 months. Ideally a set of known unmarked fish from the same year class will also be grown out, to ensure availability of suitable size and age fish of the same species for development of blind OTC evaluation sets.

Recommendations

Continue study. Collect 2010 year class from Editsto River for evaluation of contribution of stocked fish. Repeat marking and stocking for additional year classes. Ensure an extended growout is allowed for all known marked fish evaluated.

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Prepared By: Jean Leitner Title: Fisheries Biologist

Job Title: First record of the leech, *Macrobdella sestertia* (Annelida: Hirudinida), in South Carolina

Period Covered October 1, 2010 – September 30, 2011

Summary

Two species of the leech genus *Macrobdella*, *M. decora* and *M. ditetra*, have been reported from South Carolina. Recent collecting revealed the presence of a third species, *Macrobdella sestertia* Whitman, 1886. This is the first report of *M. sestertia* in South Carolina, and it is a significant southward extension of its distribution, which formerly included only Massachusetts and Maine.

Introduction

Currently, there are four valid species of North American medicinal leeches recognized in the genus *Macrobdella*: *M. decora*, *M. sestertia*, *M. ditetra*, and *M. diplotertia*. All of the species are sanguivorous, feeding on vertebrates (mainly frogs); however, only one, *M. decora*, is known to feed on the blood of humans. Sawyer (1973) discussed cases of leech attacks on swimmers in several South Carolina lakes and indicated that *M. decora* was the species involved in the attacks at Lake Jemiki (Oconee Co., SC) in the late 1960s and early 1970s.

Macrobdella decora has a wide distribution in eastern and central North America from northern Mexico to southern Canada (Klemm, 1982); however, only a single locality has been reported in South Carolina (Sawyer and Pass, 1972). *Macrobdella ditetra* occurs in coastal states from Virginia to Louisiana and inland to Arkansas (Sawyer and Shelley, 1976; Klemm, 1982), and has been collected at seven locations on the outer coastal plain of South Carolina (Sawyer and Shelley 1976). *Macrobdella diplotertia* has been found in Missouri, Kansas, and Arkansas

(Turbeville and Briggler, 2003). *Macrobdella sestertia* occurs in Massachusetts and Maine (Whitman, 1886; Smith, 1977; Smith and Hanlon, 1997), with reports from Louisiana being erroneous (Smith, 1977). In 2008, specimens of *M. sestertia* were collected from several streams in South Carolina, which is a significant extension of the known range of this species.

Materials and Methods

Leeches were collected by hand and with dipnets during stream electrofishing surveys in South Carolina in 2008 and 2011; collection locations are given in the Appendix. Some leeches were narcotized by slow addition of 70% ethanol to water in their containers, then were preserved in 70% ethanol. Others were fixed in 70% ethanol in the field without narcotization; however, these had been killed by summer heat and were relaxed prior to preservation. Leech identifications were made using morphological characteristics given in original descriptions and subsequent works (Whitman, 1886; Moore, 1953; Klemm, 1982; Davies, 1991).

Results and Discussion

A total of 10 specimens of *Macrobdella* spp. were collected from four locations in 2008 and 2011. Eight leeches had the following characteristics that identify them as *Macrobdella sestertia*: 1) male and female gonopores separated by two and one-half annuli, 2) a total of 24 copulatory gland pores arranged in four rows of six, 3) median longitudinal row of pale orange spots with marginal rows of quadrangular black blotches, and 4) body pigmentation olive green dorsally, orange ventrally. The copulatory gland pores are inconspicuous and lying hidden between annuli in smaller leeches, becoming more exposed as the glands develop. The three largest leeches had noticeable gland development that appeared white, contrasting with the orange coloration on the ventral side of the body. One additional specimen from Turkey Creek was considered to be *M. sestertia*; however, it escaped after capture. In addition to collections of *M. sestertia*, one specimen of *M. ditetra* was collected from Willow Creek, Florence Co., SC in 2011 and appears to be a new county record for this species. This specimen has male and female gonopores separated by two annuli and possesses a total of 8 copulatory gland pores arranged in two rows of four. Its pigmentation was gray/brown dorsally with two narrow, longitudinal stripes and rusty brown ventrally (lacking any black blotches) and was consistent with what has been reported for *M. ditetra*.

All of the streams containing *M. sestertia* are in the upper portion of the Stevens/Turkey creek basin (Savannah River drainage) in northeastern Edgefield County. The streams had rocky bottoms with clear, shallow water, and the streams were under drought conditions when the collections were made. Leeches were found by moving or disturbing cobble-sized rocks. The coastal plain stream where *M. ditetra* was collected had a clay substrate with coarse organic and woody debris.

Recommendations

A manuscript will be prepared to publish these significant distribution records.

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Appendix

***Macrobdella ditetra*:** n = 1, Willow Creek (site # 186538), approx. 11 km SE of Florence, downstream of Flowers Road (SSR 726), Florence Co., 34.11604° N / -79.67809° W, 10 August 2011, W.J. Poly, K.M. Kubach, M.T. Cribb, A.R. Gelder, J. Johnston, S. Mycko

***Macrobdella sestertia*:** n = 1, Turkey Creek (site # 232326), approx. 10 km NNW of Edgefield, upstream of Elmwood Road (SSR 100), Edgefield Co., 33.88178° N / 81.96879° W, 29 July 2008, W.J. Poly, K.M. Kubach, C.A. Marion, M.T. Cribb, A.R. Gelder, A. Sayer, G. Satterfield, C. Guinn; n = 3, Little Stevens Creek (site # 225891), approx. 13 km N of Edgefield, Edgefield Co., 33.90094° N / 81.96870° W, 29 July 2008, W.J. Poly, K.M. Kubach, C.A. Marion, M.T. Cribb, A.R. Gelder, A. Sayer, G. Satterfield, C. Guinn; n = 1, Sleepy Creek (site # 222764), upstream of Sleepy Creek Road (SSR 62) and downstream of US Route 378, Edgefield Co., 33.92844° N / 81.97770° W, 29 July 2008, W.J. Poly, K.M. Kubach, C.A. Marion, M.T. Cribb, A.R. Gelder, A. Sayer, G. Satterfield, C. Guinn; n = 4, Sleepy Creek (same location), 31 July 2008, W.J. Poly

Prepared By: William Poly　　　　Title: Aquatic Biologist

Job Title: Stocking augmentation of redear sunfish in Stevens Creek Reservoir

Period Covered July 1, 2010 – June 30, 2012

Summary

The objectives of this effort were to evaluate the effectiveness of stocking and learn about the redear sunfish *Lepomis microlophus* population in Stevens Creek reservoir. A three month angler survey showed that largemouth bass *Micropterus salmoides* and sunfish are the primary species sought by anglers; the reservoir is almost exclusively a 'local' fishery. Stocking was successfully performed in the fall of 2006 and 2007. Electrofishing evaluations the following year showed that stocked fish were making a substantial contribution to the stocked cohort. Growth data showed that hatchery fish were larger than wild fish. In the future, stocking appears to be a good management tool for this reservoir, however, continued evaluation of possible effects on the wild spawn is warranted.

Introduction

The redear sunfish *Lepomis microlophus* is a popular sport species in South Carolina. Compared to other sunfish commonly encountered in South Carolina, bluegill *L. macrochirus* and redbreast sunfish *L. auritus*, the redear sunfish has the largest size potential, commonly exceeding 1 kg. In fact, the current International Game Fish Association all-tackle world record for redear sunfish (2.48 kg) was caught in South Carolina.

Water level fluctuations in reservoirs can negatively affect spawning success of sunfish by exposing nests or destabilizing the shallow water environment. Like other sunfishes, the redear sunfish spawn their eggs into a nest, generally in shallow (i.e. < 2 m in depth) waters (Rohde et al 2009). Spawning generally occurs from late spring to early summer. Clark et al.

(2008) showed that the predicted egg-to-dispersal survival of white crappie *Pomoxis annularis* and smallmouth bass *Micropterus dolomieu*, also nest builders from the sunfish family, declined with increasing amplitude of water level fluctuation; highest survival was always predicted for the no-water-level-fluctuation condition.

Stevens Creek reservoir is a 970 hectare impoundment located on the Savannah River, just upstream of Savannah Georgia. The reservoir was first impounded in 1912 for the sole purpose of generating electricity. The Stevens Creek facility now operates as a re-regulating facility to mitigate the effects of highly variable discharges from the upstream, J. Strom Thurmond dam. Normal daily water level fluctuation in Stevens Creek reservoir is from 0.7 to 1.4 meters. Concern existed that fluctuating water levels may negatively affect the spawning success of nest-building fishes, such as redear sunfish.

Stocking of sunfish in small ponds is an often-used fishery management tool. However, stocking sunfish to augment population size in a relatively large impoundment, such as Stevens Creek Reservoir, has received surprisingly little evaluation. The overall of objective of this work was to evaluate the potential for augmenting redear sunfish in Stevens Creek.

The objectives of this study were:

1. Stock substantial numbers of oxytetracycline-marked redear sunfish into Stevens Creek reservoir in 2006 and 2007
2. Evaluate the contribution of hatchery-reared redear sunfish to the age-1 cohort in boat electrofishing samples collected during fall of 2007 and 2008.
3. Describe the population structure and growth of redear sunfish.
4. Conduct a 12-week creel survey in April-June, 2009, to assess angler use of Stevens Creek reservoir.

Note: Stevens Creek reservoir was defined as the waters downstream from the Highway 28 bridge downstream to Stevens Creek dam. Except for an initial survey in 2006, all stocking, sampling, and census efforts occurred in this area.

Materials and Methods

Stocking

In 2006 and 2007, redear sunfish were spawned at South Carolina Department of Natural Resources (SCDNR) hatcheries, grown-out in ponds, and stocked in the fall, approximately 6 months after spawning. Two stocking sites were used – the Fury Ferry landing just downstream of highway 28 on the South Carolina side and the South Carolina Electric and Gas landing just upstream of the dam on the Georgia side of the reservoir. Fish were transported to one of the two stocking sites in oxygenated hauling tanks. At the landing, fish were tempered until water temperature in the hauling tank was within 1° C of the Stevens Creek reservoir water temperature. Fish were then transferred to an oxygenated hauling tank in a boat and were transported to and stocked at various beds of vegetation within the reservoir. Stocking sites were recorded and an attempt was made to evenly spread the stocked fish throughout the reservoir. A sample of the stocked fish were obtained from each hauling truck prior to stocking, placed on ice, and measured (total length (TL) and weight) within 24 hours

Marking with oxytetracycline

Redear sunfish were marked with oxytetracycline (OTC) prior to stocking. Fish were immersed in OTC at concentrations ranging from 500 to 700 mg/L. The marking solution was buffered with Tris to reduce acidity associated with OTC and provide an effective pH for marking.

In 2006, samples of fish from the various marking treatments were held for grow-out at either Eastover Research Lab or Styx Fish Hatchery. At a later time, these fish were sacrificed and the otoliths were removed. The otoliths were inspected under a fluorescent compound scope by two separate readers to confirm or reject the presence of a good OTC mark. In 2007, after

OTC immersion and prior to stocking, a sub-sample of each marking batch was coded wire tagged (CWT) and re-stocked into a grow-out pond. At a later time, these fish were harvested and the coded wire tags were inspected under a microscope to determine the location, date, and batch of the OTC marking. Then, a random stratified sub-sample of 33 otoliths from the CWT, OTC-marked fish was independently inspected under a microscope by two readers for the presence of an OTC mark. These samples were mixed into the otolith samples from electrofishing samples of redear sunfish from Stevens Creek so that the readers had no prior knowledge that these fish were marked with OTC.

Electrofishing recapture sampling

During the fall of 2006, 2007, and 2008, boat electrofishing was conducted to assess the redear sunfish population in Stevens Creek Reservoir. In 2006, sampling was conducted from Steven's Creek dam upstream to Lake Thurmond dam by SCDNR and the U.S. Army Corps of Engineers. The objective of this initial sampling was to characterize general population characteristics prior to stocking.

In 2007 and 2008, electrofishing was conducted from Stevens Creek dam upstream to the highway 28 bridge crossing. Each year, sampling was conducted at locations throughout this area, attempting to cover sites representative of the entire sampling zone. The sampling zone was divided into an upper and lower sampling zone; the dividing line between zones was located at 33.58144 N, -82.09484 W (on the southern bank of the reservoir) to a point due north on the northern shoreline; an attempt was made to collect a representative sample of fish from each zone during 2007 and 2008.

Collected redear sunfish were measured (total length (TL), mm) and weighed (g). Otoliths were removed for later determination of age. Number of annuli was determined by two, trained, independent readers. Percent agreement between readers was determined. Only those fish for which age was agreed upon by both readers were used in further analysis of growth and length at age. Mean length at age was determined for the composite 2006-2008 sample. A T-test with equal variances was used to evaluate differences in average total length of hatchery and wild redear sunfish with one annulus from the fall 2007 electrofishing collection. In 2008, the same procedure was used to compare average total length of hatchery and 'all other' redear sunfish with one annulus from the fall 2008 electrofishing collection.

A length weight regression was determined for all redear sunfish collected in 2006-2008. A length frequency distribution was calculated separately for fish collected in 2006, 2007, and 2008; each fish length was put into the nearest 2.54 cm grouping, or nearest inch group, to create usable categories.

Angler survey

From April 3, 2009 to June 25, 2009, an angler survey was conducted. This time period was broken up into three, 28 day periods, each containing 20 weekdays and 8 weekend days. Each day was divided into two, six hour survey periods, 7 AM to 1 PM and 1 PM to 7 PM. During each 28 day period, 6 weekend and 7 weekday half-day sampling periods were randomly selected. Within the survey period, an instantaneous count of bank and boat angler use was made at a randomly selected time. During the remainder of the survey period, creel clerks interviewed anglers, determining hours fished, angler catch by species, and asking questions about the angler's assessment of the fishery and where they resided .Survey results were analyzed and expanded by the University of South Carolina statistics lab according to the methods of Malvestuto (1996).

Results

A total of 148,111 and 99,491 redear sunfish were stocked during the fall of 2006 and 2007, respectively (Table 1). Weighted mean size was 76.1 mm and 6.7 g in 2006 and 100.9 mm and 18.2 g in 2007. In 2007, fish produced at the Heath Springs hatchery had a mean size of 125.9 mm and 38.7 g demonstrating the potential for raising near catchable size redear sunfish in one growing season.

Table 1. Number and average size of redear sunfish stocked into Stevens Creek Reservoir in 2006 and 2007.

Date	Stocking site	Number stocked	Mean total length, mm, (standard error)	Mean weight, grams, (standard error)	Sample size
10/12/2006	Fury's Ferry	8,300	-	-	-
10/24/2006	Fury's Ferry	41,500	73.8 (0.9)	6.2 (0.2)	60
11/7/2006	SCE&G	64,800	75.1 (0.9)	6.4 (0.2)	61
11/14/2006	Fury's Ferry	34,211	80.6 (1.5)	7.8 (0.5)	47
10/24/2007	Fury's Ferry	22,710	97.9 (0.6)	13.8 (0.3)	153
10/31/2007	SCE&G	21,835	93.7 (0.9)	12.8 (0.6)	107
11/6/2007	SCE&G	5,744	142.5 (2.9)	61.6 (3.9)	21
11/7/2007	Fury's Ferry	13,682	118.9 (1.4)	29.1 (1.3)	84
11/12/2007	SCE&G	17,520	97.4 (0.8)	15.0 (0.4)	111
11/16/2007	Fury's Ferry	18,000	89.8 (1.2)	11.3 (1.0)	53

In 2006, redear sunfish were successfully marked with OTC. Difficulty was experienced growing out the fish to a size where the mark was distinct from the edge of the otolith. On April 23, 2007, 20 otoliths from fish that were grown-out at Eastover from the October 12, 2006 stocking were inspected for marks; total length of the fish at that time ranged from 79 to 113 mm. OTC marks were clearly present near the edge of the otolith on the five largest fish (103-113 mm, TL); marks were not clearly visible on the other fish (79 to 99 mm TL). It was hypothesized that growth was not adequate in the smaller fish to observe the OTC mark.

Subsequently, in May, 2007, otoliths from fish > 100 mm TL were inspected for an OTC mark; fish came from the 10/24/07 (N=5), 11/7/07 (N=3), and 11/14 (N=1) stocking. Both readers judged that all otoliths were clearly marked with OTC, though the marks were still very close to the edge. A third sample of fish were grown out from the 11/14/06 stocking to 6/4/07 and ranged in size from 81 to 121 mm TL. Both readers saw clear marks in all inspected fish (N=9).

In 2007, marking success was inconsistent. Of 32 known-marked fish that were inspected only 10 (31%) were identified as 'marked' by both readers. Some marking dates produced higher percentages of successfully marked fish than other dates (Table 2). Weighting percentage marked for a stocking date by the total number stocked on a stocking date, produced an estimate that 29 % of redear sunfish stocked in 2007 were clearly marked.

Table 2. Oxytetracycline mark identification by two independent reads of marked redear sunfish.

			Number	
Date (2007)	Hatchery	Inspected	Both = 'marked'	One = 'marked'
10/23	Cheraw	7	6	0
10/30	Cheraw	4	0	0
11/5	Spring Stevens	5	0	1
11/6	Spring Stevens	7	1	0
11/15	Cheraw	2	0	1
11/15	Cohen Campbell	7	3	1
Total		32	10	3

A total of 760 redear sunfish were collected by electrofishing during the 2006-2008. The relation between total length and weight was defined as:

Log10 weight (g) = -4.94 + 3.11 * log10 total length (mm); R^2 = 0.99; N = 760.

A total of 723 fish had otoliths that could be aged by both independent readers; age agreement between both readers occurred for 612 (85%) fish. Mean length was calculated for these fall-collected fish (Table 3) and, for 2006-2008, was 130 , 171, and 203 mm for redear sunfish with

1, 2, and 3 annuli, respectively; length at age was similar among years. Of note, a fish with 18 annuli was collected in 2006.

Table 3. Mean length at age for redear sunfish from Stevens Creek Reservoir, 2006-2008. Fish were collected in fall, so there was considerable growth beyond the last annulus.

Year	Annuli	N	Mean Total Length (mm)	Standard Deviation
	1	10	124	28
	2	49	171	25
	3	11	192	28
2006	4	8	235	40
	5	3	263	5
	6	2	248	59
	8	1	305	NA
	18	1	318	NA
	1	153	128	24
	2	58	169	27
	3	16	202	17
2007	4	6	210	33
	5	3	244	6
	6	2	285	7
	8	1	313	NA
	0	13	85	13
	1	147	133	29
	2	90	172	28
2008	3	27	208	16
	4	8	216	25
	5	3	210	45

Approximately one year from stocking, results strongly suggested that hatchery fish were significantly larger than wild fish. In fall of 2007 collections of redear sunfish with one annulus, hatchery fish were significantly ($P < 0.001$) longer than wild fish; hatchery and wild fish averaged 137 (N=72) and 117 (N=70) mm TL, respectively. Since we could not clearly differentiate hatchery from wild fish from the fall 2007 stocking, we compared hatchery fish (i.e. both readers saw an oxytetracycline mark) with ‘all other’ fish. In fall of 2008 collections of redear sunfish with one annulus, hatchery fish were significantly (P < 0.001) longer than 'all other' fish; hatchery and 'all other' averaged 167 (N=13) and 129 (N=134) mm TL, respectively. In 2006, prior to stocking, redear sunfish with one annulus averaged 124 mm TL (N=10; 95% confidence interval = ± 17 mm).

Length-frequency histograms were prepared for the three study years (Figure 1) and modes generally agreed with average length at age. However, 2008 length-frequency showed a modal size for fish with one annulus that was lower than the average size.

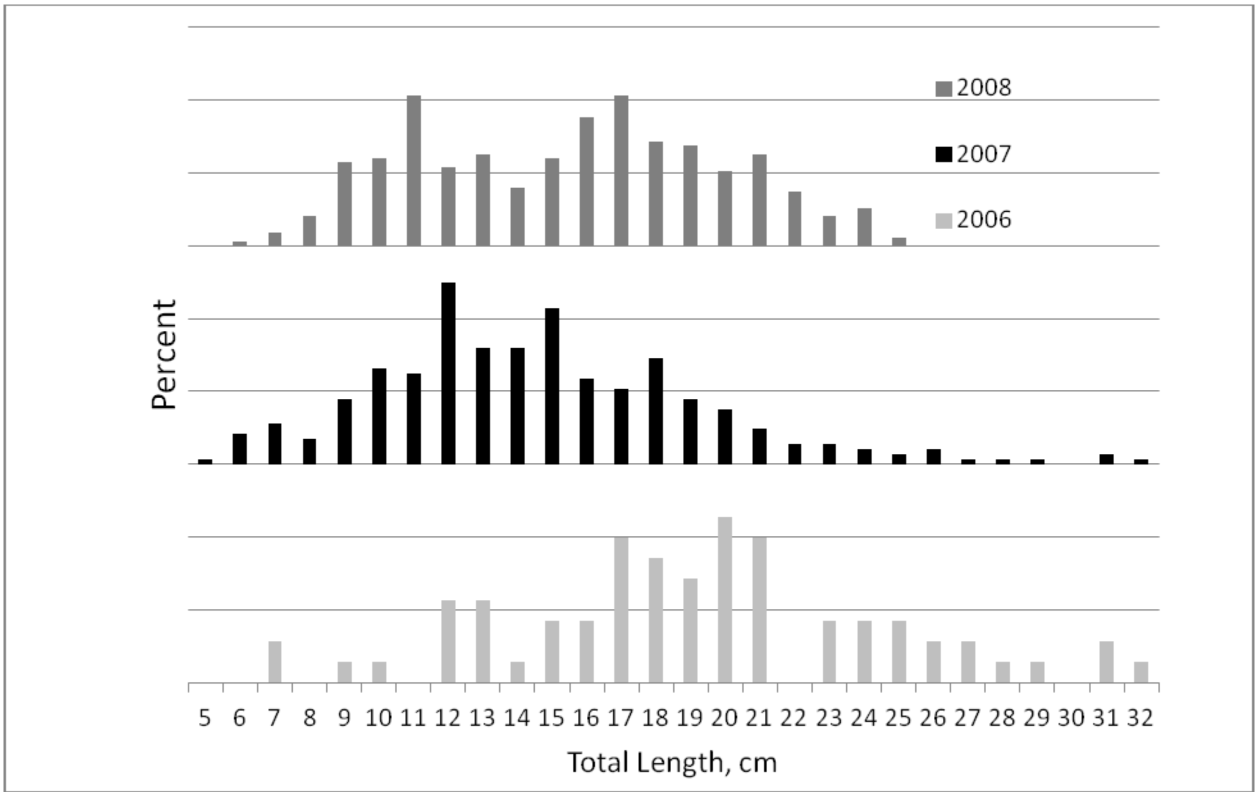

Figure 1. Length frequency histogram of redear sunfish collected from Stevens Creek Reservoir in 2006 (N=70), 2007 (N=289), and 2008 (N=351). Each horizontal gridline represents 5% of the total catch for each year.

In 2007, 147 redear sunfish otoliths with one annulus were inspected for the oxytetracycline mark associated with the fall 2006 stocking. In four instances, the two readers disagreed upon whether a mark was visible. For the remaining otoliths, 72 (50%) were marked and 71 were not marked. One otolith reader blindly read a subsample of otoliths a second time; the reader agreed with his initial interpretation of 72 of 80 (90%) otoliths.

In 2008, 70 redear sunfish otoliths with one annulus were inspected for the oxytetracycline mark associated with the fall 2007 stocking. In eight instances, the two readers disagreed upon whether a mark was visible. For the remaining otoliths, 13 (21%) were marked and 49 were not marked.

Angler Survey

During the 84-day survey period, anglers fished for 3,242 (relative standard error (RSE) = 14.9) hours, which, on average, breaks down to approximately 39 hours of angler effort per day. Boat angling accounted for 93% of effort. Anglers harvested an estimated 2,834 (RSE = 30.2) fish; anglers released 2,378 fish (RSE 30.9).

The estimated catch rate (harvested and released fish) was 1.6 fish/hour (RSE = 21.2). Expansion of surveys revealed that the catch was dominated by bluegill (N=1,772; RSE = 214), largemouth bass (N=873; RSE=246), redear sunfish (N=644; RSE=363), and redbreast sunfish (N=563; RSE=627), though variance associated with these estimates too high to be deemed reliable estimates.

Bluegill dominated the fish directly observed as harvested by the clerks. Of those observed fish, bluegill was 54%, redbreast sunfish 18%, largemouth bass 14%, and redear sunfish comprised 11%.

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Intended fishing effort, based on the target species identified by the angler, indicated that largemouth bass and sunfish were the most sought after species. Interviews indicated that 36% sought largemouth bass, 25% sought “bream” (i.e. any of the sunfish species), 15% sought redear sunfish, and 15% sought ‘anything;’ all other categories were less than 5% of intended effort.

Anglers were local (within 50 miles) in origin. Most came from GA (63%) and the remainder from SC. These anglers (N=27) classified the fishery as excellent (11%), good (59%), fair (26%), and poor (4%).

Discussion

A 3-month angler survey showed that Stevens Creek was a local fishery where anglers were primarily targeting largemouth bass and sunfish. Angling pressure was light to moderate while the majority of anglers classified the fishery as ‘good.’ This study was not designed to show whether the fishery was enhanced, in terms of effort and catch rate, due to the stocking. However, the stocking did appear to positively augment the redear sunfish fishery, which would likely result in increased effort and greater success by anglers.

Hatchery stocking made a substantial contribution to the redear sunfish population of Stevens Creek reservoir. The 2006 stocking accounted for 50% of redear sunfish with one annulus collected during the fall of 2007. The 2007 stocking could not be evaluated as effectively as only 31% of the known-marked fish were successfully identified. However, even with this limitation, 21% of the fish with one annulus collected in 2008 were identified as hatchery fish by both otolith readers. If we assume that only 31% of the hatchery fish were successfully identified, then the actual percentage of hatchery fish would have been much greater.

Length comparisons of redear sunfish with one annulus showed that hatchery-stocked fish were larger than wild fish. In 2006, prior to stocking, wild fish averaged 124 mm TL. Hatchery fish from the 2006 stocking were, on average, 20 mm longer than wild fish in the fall of 2007 collections. Identifiable hatchery fish from the 2007 stocking were, on average, 38 mm longer in the fall 2008 collections than ‘all other’ fish, which was an unknown mix of wild and unidentified hatchery fish. The larger size of hatchery fish from the 2007 stocking appears to relate to size at stocking; in 2006 and 2007, fish averaged 76 and 101 mm TL at stocking, respectively.

It appears that stocked fish had a size advantage on wild fish at stocking, which was maintained approximately one year after stocking. Unfortunately, this study did not produce a reliable estimate of size of age-0 wild fish at stocking, as the electrofishing gear was not an effective sampling gear for these fish. This size advantage may allow stocked fish to escape a food bottleneck earlier than wild fish, allowing them to recruit faster to the sport fishery. An earlier preliminary study with a high school intern showed that Stevens Creek redear sunfish switched from plankton/insects to snails at approximately 150 mm TL. Thus, from a management perspective, stocking of redear sunfish appears to augment a year class and produce fish that will recruit more quickly to the sport fishery. However, this study did not have adequate information to discern whether hatchery fish were limiting the abundance or growth of wild-spawned fish. Future studies must obtain additional measures of the abundance and size at age-0 to evaluate this.

Recommendations

If the economics of the fishery warrant, develop and implement a long-term redear sunfish stocking and monitoring plan on Stevens Creek reservoir. Based on the information obtained in this study, the stocking/monitoring plan should have the following characteristics:

- Mark all stocked fish
- Use the stocking rate, method of stocking, and size of stocked fish that was used in 2008, as larger stocked fish seem to have advantages for the fishery and for the redear sunfish population
- Develop a sampling method of capable of estimating the average size of age-0 redear sunfish during late September or early October, prior to stocking; this is needed to compare size of hatchery fish to size of wild fish.
- Implement a stocking plan where stocking occurs for 3 consecutive years followed by 3 consecutive years of non-stocking to better evaluate the effects of stocking.
- Implement a recapture sampling method in the fall that has fixed stations and effort so that catch per unit of effort can be used an index of abundance; take into account reservoir stage fluctuations, which can affect sampling efficiency.
- Consider enlisting local anglers to help monitor the fishery.

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