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Development of a multi-scale management perspective for wadeable stream fisheries in Mississippi

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DEVELOPMENT OF A MULTI-SCALE MANAGEMENT PERSPECTIVE FOR WADEABLE STREAM FISHERIES IN MISSISSIPPI

By

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I used multivariate, hierarchical analyses to examine the relative influence of watershed-, riparian- and channel-scale environmental characteristics on catch per unit effort (CPUE: fish/angler-hour) and species composition of sport fisheries in Mississippi wadeable streams. Partial canonical correspondence analyses indicated that riparian-scale variables (31.1%) explained more variation in sport fish relative abundances compared to watershed-scale (24.4%) and channel-scale variables (18.9%). Largemouth bass *M. salmoides* and longear sunfish *Lepomis megalotis* were more abundant in smaller-watershed streams with dense forest cover and greater woody debris, alkalinity and diverse substrates. Spotted bass *M. punctulatus* and bluegill *L. macrochirus* were more abundant in larger-watershed streams with moderate to dense forest cover yet more open riparian canopies.

Regional-scale characteristics also influenced relative abundances of these fisheries. Total CPUE, total bass CPUE and largemouth bass CPUE were greatest in watersheds

draining the Blackland Prairie-Flatwoods compared to other level III ecoregions. This ecoregion contains fertile soils that influence stream productivity, because alkalinity tends to be large in forested streams draining this ecoregion.

I developed and validated watershed-scale models and found that percentage forest cover, stream density, total road density and primary highway density predicted mean total CPUE, mean total sunfish CPUE and mean total bass CPUE accurately (Sign tests comparing observed versus predicted mean CPUE, $P > 0.05$). The models were precise ($R^2 > 0.71$), explaining 83%, 71% and 80%, respectively, of variation in mean total CPUE, mean total sunfish CPUE and mean total bass CPUE from independent data. Species-specific models performed poorly, suggesting biotic relationships may hinder development of meaningful habitat models for species.

My study supports forest conservation to sustain sport fisheries in Mississippi's wadeable streams. Forests mediate sediment and nutrient loading to stream channels, influence hydrology and channel morphology and provide woody habitat for sport fish and their forage base (benthic macroinvertebrates). My small sample size was small; (N = 13 reaches), thus caution is advised before engaging in comprehensive management of wadeable streams based on my results. Nevertheless, my watershed models can be applied at very low cost using a GIS or topographic maps to identify reaches state-wide that support wadeable stream sport fisheries.

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CHAPTER I

INTRODUCTION

Management of wadeable stream fisheries in Mississippi

Wadeable streams in Mississippi support black bass *Micropterus* spp., sunfish *Lepomis* spp. and channel catfish *Ictalurus punctatus* fisheries (Robinson and Rich 1980; Robinson and Rich 1981; Robinson and Rich 1984). However, state-wide stock assessments of these fisheries have not been conducted since the 1980's (Robinson and Rich 1980; Robinson and Rich 1981; Robinson and Rich 1984). Currently, Mississippi's wadeable streams are not managed for recreational fisheries. Instead, they are managed almost exclusively for water quality and flood control (MDEQ 2003; Bressler et al. 2006).

Historic anecdotal evidence suggests that prior to 1970, Mississippi anglers valued wadeable streams for their sport fisheries. Interviews conducted by the Mississippi Department of Wildlife, Fisheries and Parks (MDWFP) reported that largemouth bass *M. salmoides*, spotted bass *M. punctulatus*, longear sunfish *L. megalotis* and bluegill *L. macrochirus* were targeted by anglers in small southern Mississippi streams (Rich 1968; Rich 1970; Rich 1972). On average, these anglers caught 1.4 kg of bass and 0.3 kg of sunfish during four- to six-hour fishing trips. After 1970, however, the anglers abandoned wadeable streams as a source of recreation and subsistence. They claimed that poor agricultural and timber practices in the streams' respective watersheds and riparian zones caused sediment erosion into the stream channels, which filled in deep fishing holes. In addition, they considered the fish meat unpalatable because of oil and gas well pollution (Rich 1970; Rich 1972).

A more recent assessment of angler characteristics shows that Mississippi anglers spend more time and money fishing in impoundments than in streams (Schramm et al. 1996). Subsequently, the bulk of fisheries management activities in the state are devoted to farm ponds, state-operated lakes and flood-control reservoirs (Fisher and Burroughs 2003). Although anglers still exploit Mississippi's wadeable streams (albeit to some unknown extent), these resources continue to be relegated to fairly low management priority in relation to other fisheries in the state.

The importance of habitat assessments as fisheries management tools

In other regions of the U.S. (e.g., Mid-western, Western, Pacific Northwest, Northeast, Mid-Atlantic and Ozark highlands), habitat assessments are very important to fisheries management in wadeable streams (Scarnecchia and Bergerson 1987; Lyons 1991; Sowa and Rabeni 1995; Zorn and Seelbach 1995; White 1996; Wang et al. 1998). Water temperature, in-stream flows, substrate composition, large woody debris (LWD), overhanging vegetation, riffle-pool frequency and residual pools are examples of environmental characteristics in wadeable streams that are associated with fish stock characteristics (Angermeier and Karr 1984; Angermeier 1985; Neves and Angermeier 1990; Rabeni and Jacobson 1993; Clarkson and Wilson 1995; Zorn and Seelbach 1995; Maddock 1999; Talmage et al. 2002). In the Western U.S., trout biomass has been monitored indirectly using habitat quality indices (HQI), which are regression models that incorporate flow and substrate characteristics of stream channels to predict biomass of trout in these systems (Binns and Eisermann 1979; Raleigh et al. 1986; Jacobs et al.1987). Furthermore, habitat restoration and enhancement influence fisheries management in wadeable streams because the fish and their prey (e.g., benthic invertebrates), as well as production by aquatic algae, macrophytes, fungi and bacteria, are often limited by the physical and chemical environment in stream channels (Pomeroy and Wiebe 1988; Poff and Ward 1990; Wootton and Power 1993; Riley and Fausch 1995; Zorn and Seelbach 1995; Nisbet et al. 1997; Hieber and Gessner 2002).

Although habitat management activities can increase or maintain abundance of catchable-size fish (TL $\geq$ 180 mm) in particular habitats or reaches in the short-term (i.e., < 10 years), there is little evidence to suggest that habitat management enhances the long-term survival and production of fish in wadeable streams (Thompson 2006; Budy and Schaller 2007; Cooperman et al. 2007). Studies conducted in Northwest Mississippi streams draining predominately agricultural landscapes, like the Mississippi "Delta" region, have shown that artificial structures (e.g., stone spurs, large wood) used to rehabilitate stream channels often fail over time or require periodic maintenance (Shields et al. 2000; Shields et al. 2003). In addition, increases in catchable-size bass and catfish relative abundances over a ten-year period have been attributed to their immigration from large rivers rather than resulting from habitat enhancement in the affected reaches (Shields et al. 2003). In several Pacific Coastal U.S. streams, artificial structures used to restore trout habitat failed due to cumulative effects of watershed erosion originating from upstream tributaries (Frissell and Nawa 1992). These studies illustrate that human land use in landscapes throughout stream watersheds limit the success of channel-scale habitat and fisheries restoration in specific reaches.

Influence of scale on fisheries management in wadeable streams

Streams function hierarchically through space and time (Frissell et al. 1986; Imhof et al. 1996; Smiley and Dibble 2005). Subsequently, physical, chemical and biological attributes of streams at smaller scales (e.g., the fish, their forage and their habitats) are constrained by climatic and geological processes (e.g., precipitation, evapo-transpiration, hillslope erosion) occurring at larger spatial and temporal scales (Frissell and Nawa 1992; Roth et al. 1996; Allan 2004). Ecological studies conducted at multiple spatial scales often find that biota and habitats within stream channels exhibit strong correlations with geomorphic, climatic and land use/land cover characteristics in the streams' respective watersheds and riparian zones (Lanka et al. 1987; Clarkson and Wilson 1995; Kauffman et al. 1997; Herlihy et al. 1998; Strayer et al. 2003). Still, some fisheries models rely on environmental variables that reflect one spatial scale, and they typically address only the channel scale characteristics (e.g., habitat suitability indices or HQI models for trout in Western U.S. streams).

Channel characteristics of wadeable streams are tightly linked to larger-scale riparian and watershed characteristics. For example, riparian vegetation buffers sediment and chemical pollution to stream channels (Gregory et al. 1991; Naiman and Decamps 1997). Overhanging riparian foliage shades stream channels, mediating water temperature (Opperman and Merelender 2004). Riparian vegetation also provides allochthonous inputs of organic matter to stream channels, which ultimately influences energy dynamics of stream biota (Hynes et al. 1974; Vannote et al. 1980). Furthermore, the cumulative impacts of geology, climate and land use on watersheds ultimately determine hydrological and geomorphic characteristics of riparian zones and stream channels (Lanka et al. 1987; Bencala 1993; Fausch et al. 2002; Wiens 2002). For example, annual precipitation, evapotranspiration, soil type and structure, topography, vegetation, and human land use are characteristics of watersheds that ultimately define the intensity, frequency and duration of sediment, water and nutrient inputs to stream channels (Allan and Johnson 1997; Poole 2002; Allan 2004; Durance et al. 2006). As Hynes (1970) so eloquently stated regarding the structure and function of stream ecosystems, "…in every respect, the valley rules the stream".

Objectives

I used stream hierarchy theory as a working hypothesis (Allen and Starr 1982) to examine relationships between catchable-size sport fish relative abundances and environmental characteristics of wadeable streams in Mississippi at different spatial scales. Frissell et al. (1986) outlined this theoretical approach for studying stream ecosystems, proposing that stream management activities should consider hierarchical controls on system functions (e.g., biogeochemical cycling, primary and secondary production, survival and abundance of stream biota). A hierarchical perspective is needed because stream processes operate at a wide range of scales (10^{-7} to 10^{8} m spatially and 10^{-8} to 10^{7} yr temporally), and the relative importance of environmental factors controlling stream biota and their habitats changes with the spatial and temporal scale of resolution. This approach is an appropriate management perspective for my study, enabling me to address the proportional significance of different spatial scales on wadeable stream fisheries in Mississippi.

The objectives of my research were to (1) identify important environmental variables at watershed, riparian and channel scales of resolution in Mississippi wadeable streams (Figure 1), (2) determine the appropriate hierarchical scale for modeling catchable sport fish relative abundances, (3) develop and evaluate a set of candidate models that can be used by fisheries managers to locate wadeable stream reaches that potentially support sport fisheries and (4) determine if catchable-size sport fish relative abundances differ among even larger-scale, hierarchical ecoregions. Finally, I develop a multi-scale management framework towards the initiation of a wadeable stream fisheries program in Mississippi.

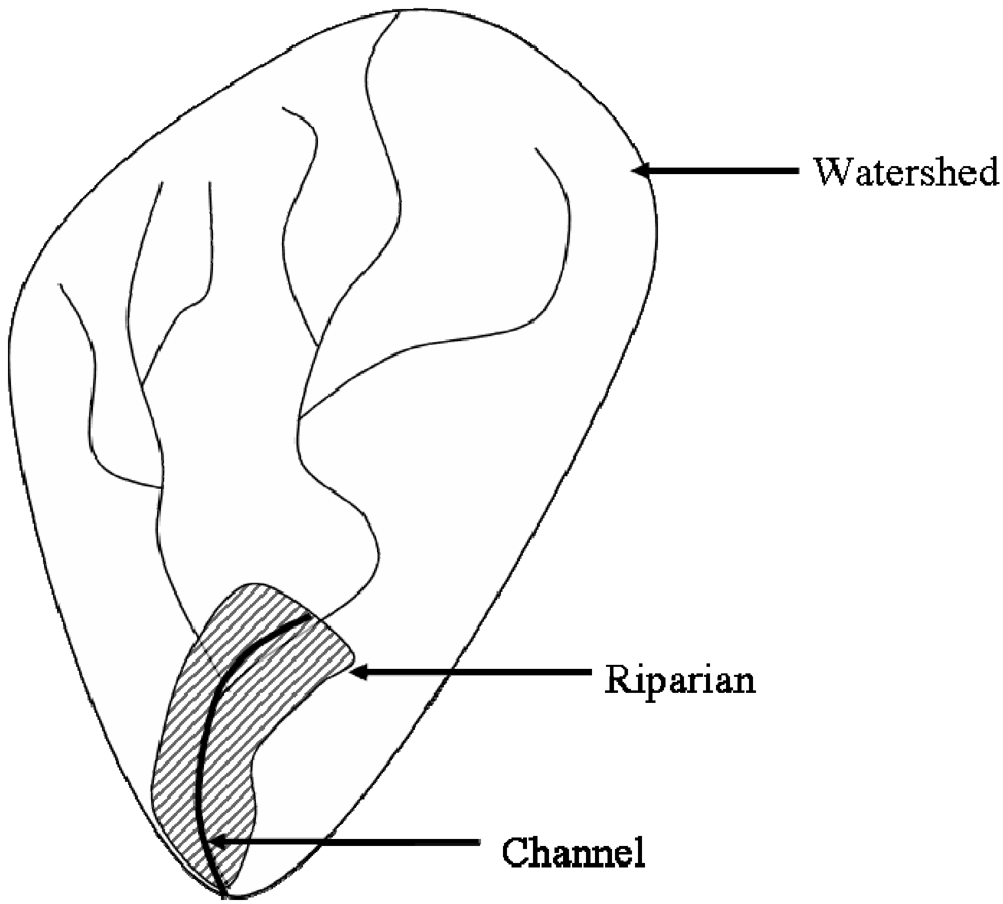


Figure 1. An illustration of multiple spatial scales at which environmental data were collected in my study. The watershed scale is the largest scale, containing the land area and upstream tributaries that drain to the bottom of the sample reach. The riparian scale included a 300 m perpendicular distance from the right and left channel margins. The channel scale included all in-stream physical habitat, water chemistry and benthic macroinvertebrates. The boundaries of the channel were the top of the first terrace on the right and left banks.

CHAPTER II

METHODS

Study areas

In 2004 the U.S. Environmental Protection Agency (USEPA) conducted the National Wadeable Streams Assessment (WSA) (USEPA 2006). Environmental and benthic macroinvertebrate sampling for the WSA followed the protocols in the USEPA Environmental Monitoring and Assessment Program (EMAP). I used the same sample units from Mississippi that were included in this WSA. These sample units consisted of one reach within each of 13 wadeable streams (Figure 2). The reaches were chosen randomly by USEPA personnel using a generalized random-tessellation stratified (GRTS) design (Stevens and Olsen 2004). The population of interest in my study was all wadeable-stream reaches in Mississippi from the U.S. Geological Survey's (USGS) National Hydrography Dataset (www.nhd.usgs.gov) mapped at the 1:100,000 scale.

Wadeable streams were identified as stream orders 1 through 5 (Strahler 1964) and were generally < 1 m deep throughout most of the reach. Reaches that were accessible only by boat were not considered wadeable. Sampling occurred at low flow conditions during summer (July-October). The sample reach length was 40 times the mean wetted width at the X-site, where the X-site is the mid-point of the reach (Figure 3). Research from the EMAP program suggests that this reach length typically encompasses

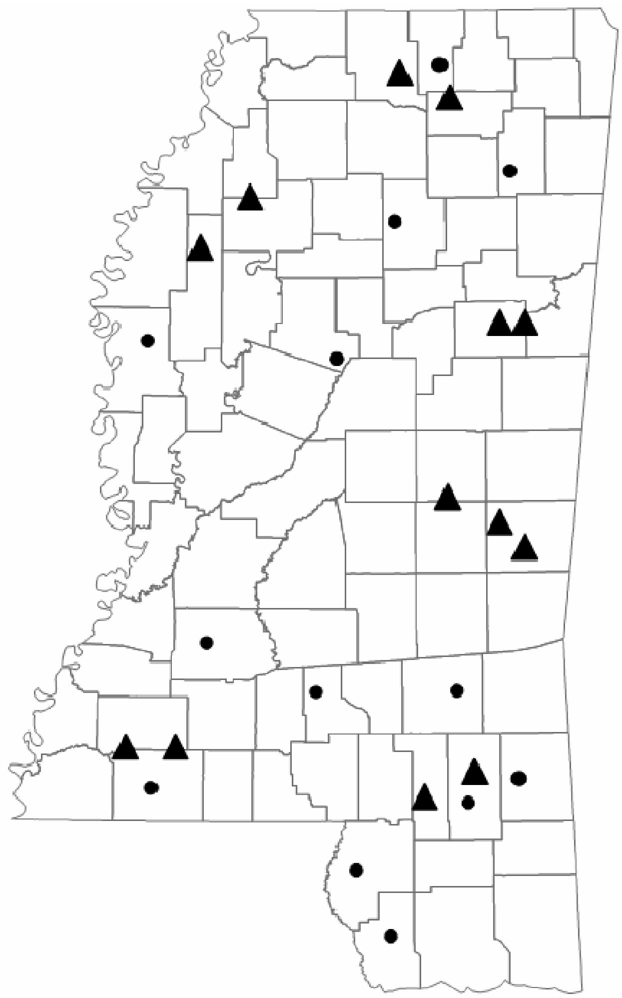


Figure 2. Locations of wadeable stream reaches sampled during summers 2003-2005. Circles are reaches sampled during 2004 and 2005 for the U. S. EPA National Wadeable Streams Assessment (USEPA 2006). Data from these reaches were used to develop models of catchable-size sport fish relative abundances. Triangles are independent reaches sampled during summers 2003 and 2004. Data from these reaches were used to evaluate model performance.

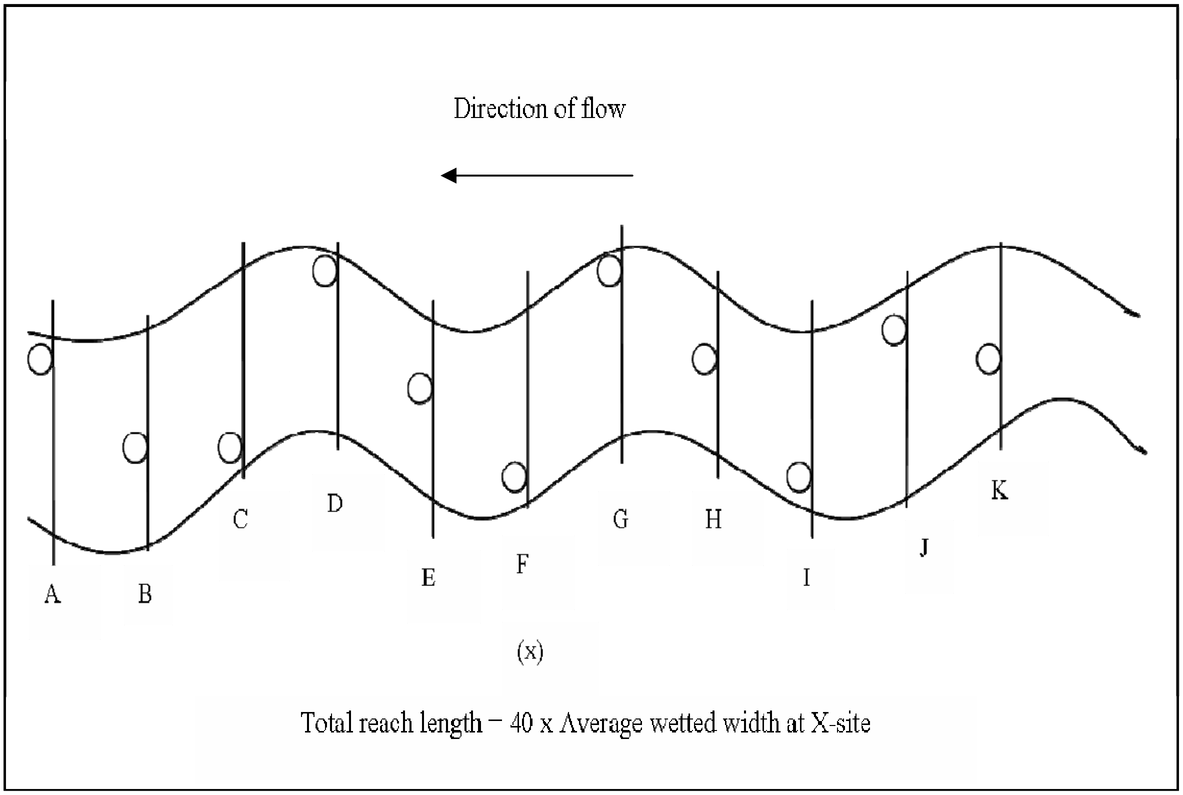


Figure 3. An illustration of the sample reach layout used to collect channel-scale data, benthic macroinvertebrates and catchable sport fishes in Mississippi wadeable streams during summers 2004 and 2005. This layout followed the U.S. EPA environmental monitoring and assessment protocols (EMAP) used in the 2004 National Wadeable Streams Assessment (USEPA 2006). Letters correspond to the eleven equidistant transects at which data and benthic macroinvertebrates were collected.

two meanders in wadeable streams, and it is generally the most appropriate reach length for characterizing physical habitat, benthic macroinvertebrate and fish assemblages from wadeable streams (Kaufman et al. 1999).

I used data from 13 additional stream reaches in Mississippi as an independent dataset to test models generated from the WSA dataset (see Figure 2). Fish catch data for 11 of these streams were taken from Shewmake and Jackson (2004), who sampled reaches within the streams during summers 2003-2004. Angling methods and equipment in their study were similar to that used in my research. Two other streams located in the Mississippi "Delta" region were selected by the GRTS design but were discarded from the WSA. One reach was a human-constructed canal and was not considered representative according to the WSA sampling protocol (USEPA 2004), and the other was considered unsafe due to a recent precipitation event. However, these two reaches were fished on later occasions during summer 2004 under safer, low-flow conditions.

Assessment of watershed characteristics

Watershed data were collected from geographic information systems (GIS). These data were gathered by remote sensing technology and compiled into a dataset managed by personnel at the USEPA in Corvalis, OR. The watershed boundary was defined as the upland geographic area that drained to the downstream end of a reach, specifically to the final downstream transect of the reach. Total watershed area (km^2) was delineated by digitizing a polygon along the boundary of a watershed onto a map layer in ArcMap (Environmental Systems Research Institute, Inc., 2005). The watershed area was then calculated as the area of the polygon.

Land use/land cover (LULC) data and human population data were collected by the USGS in 2001 and obtained from their Seamless Data Distribution System dataset (www.seamless.usgs.gov). The LULC elements characterized by the U.S.G.S. data set included the percentage of the watershed covered by natural vegetation, unnatural vegetation, forest, wetlands, total agriculture (i.e., row-crop, pasture/hay, silviculture) and urban/recreational areas. Forest connectivity data were collected as well and included average forest connectivity, percentage perforated forest and percentage forest edge. Road densities were calculated as the length of road (km)/watershed area (km^2). Roads were classified as Class 1 (primary highways like interstate and state highways), Class 2 (secondary highways and county roads), and Class 3 (rural roads, off-road trails, neighborhood roads, logging trails). To assess extent of human development in watersheds, percentage impervious land use was measured to reflect percentage watershed with connected pavement (e.g., parking lots, city blocks). In addition, population change was measured as percentage difference in human population density from the 1990 census to the 2000 census.

In the field, ground-truthing was conducted for remotely-sensed watershed characteristics while traveling to and from the site, and while standing in the channel. Evidence of human impacts was estimated visually. Categories of impacts assessed were residential, recreational, agricultural, industrial, and stream management (USEPA 2004). Each impact within a category was scored 0-3, corresponding to none, low, medium or high, respectively, degree of human impact. Residential impacts included residences, maintained lawns, construction, pipes/drains, dumping, roads, bridges/culverts, and sewage treatment. Recreational impacts included hiking trails, parks/campgrounds, primitive parks/camping, trash/litter, and surface films. Categories describing agricultural impacts were cropland, pasture, livestock use, orchards, poultry facilities, irrigation equipment, and water withdrawal. Industrial impacts consisted of industrial plants, mines/quarries, oil/gas wells, power plants, evidence of fire, odors, and commercial activities. Stream management impacts included liming, chemical treatment, angling effort, dredging, channelization, water level fluctuations, fish stocking and dams (USEPA 2004). Finally, the dominant land use of the watershed was visually classified by the field team as forest, agriculture, pasture, urban or suburban/town. If the dominant land use was forest, then the observer classified the dominant age-class (estimated visually by tree size) as 0-25 years, 25-75 years, or >75 years old (USEPA 2004).

Assessment of riparian characteristics

In the field, riparian zone characteristics were estimated from a 10 m x 10 m plot at each transect within reaches, with transect lines dissecting the middle of riparian plots (Figure 4). Canopy and understory vegetation were classified as deciduous, coniferous, broadleaf evergreen, mixed or none (USEPA 2004).

The size structure of the riparian zone was estimated visually, and three size categories of riparian vegetation were used: canopy (> 5 m high), understory (0.5-5 m high), and ground cover (< 0.5 m high). Diameter-at-breast height (dbh) of riparian trees was estimated visually. Canopy vegetation was divided into large trees (> 0.3 m dbh) or small trees (< 0.3 m dbh). Understory and ground cover vegetation were divided into woody shrubs/saplings or non-woody herbs/grasses/forbs. Density of each division was scored (0-4) as percentage cover of the 10 m x 10 m plot. Categories that

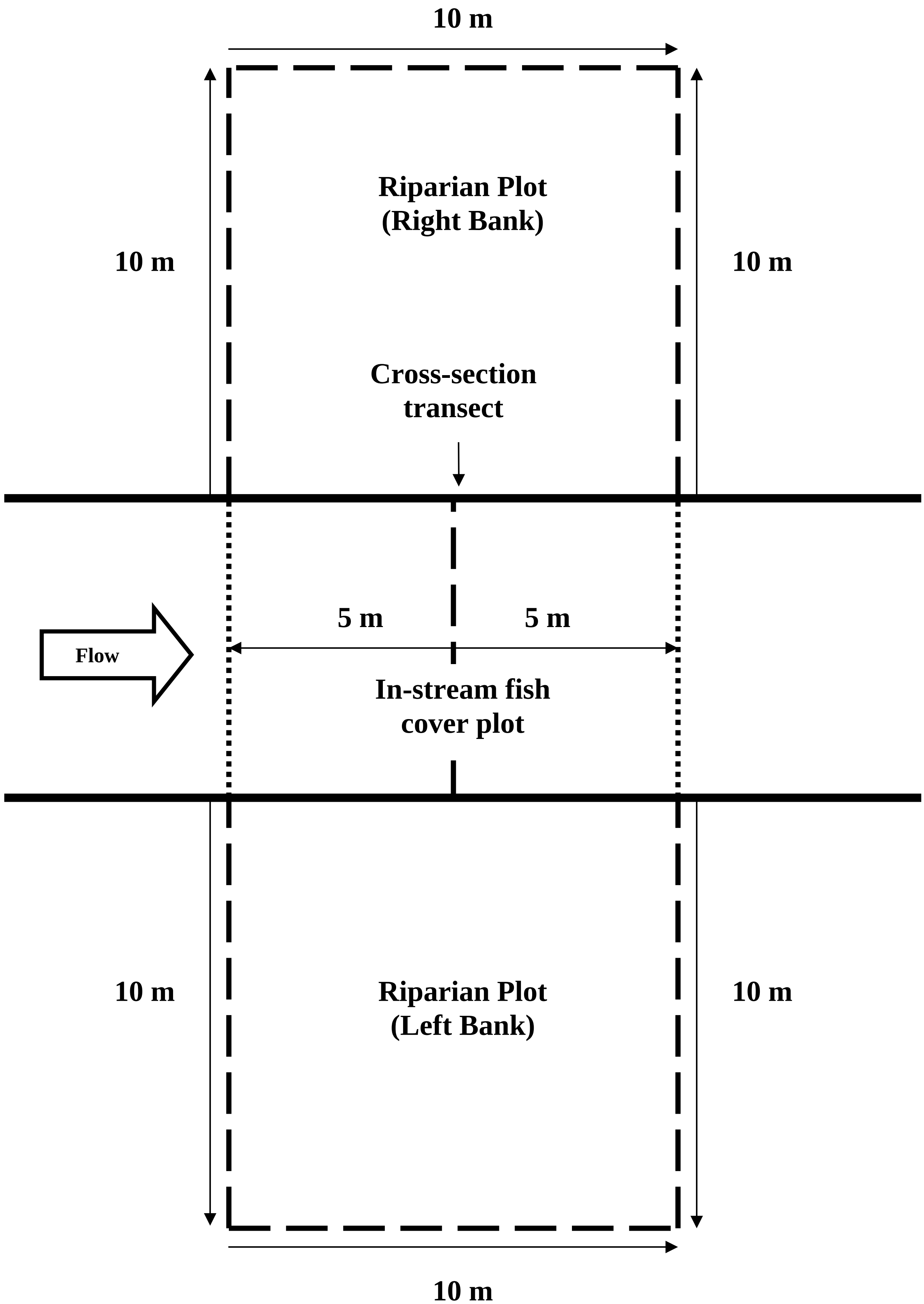

10 m
Riparian Plot
(Right Bank)
10 m
10 m
Cross-section
transect
Flow
5 m
5 m
In-stream fish
cover plot
10 m
Riparian Plot
(Left Bank)
10 m
10 m

Figure 4. Riparian boundaries at individual transects (figure from USEPA 2006). These plots were used in the National Wadeable Streams Assessment to visually estimate size-structure and taxonomic composition of riparian vegetation, intensity of human disturbances (e.g., trash, pipes, logging, residences) and percentage composition of various channel-scale fish cover habitats.

described riparian size structure included "absent" (0%), "sparse" (<10%), "moderate" (10-40%), "dense" (40-75%), and "very dense" (>75%) (USEPA 2004).

Human influences to the riparian zone were quantified with a total human disturbance score (USEPA 2004). Total human disturbance was measured at transects, and it was the sum of scores from the following impact categories: rip-rap/dam, buildings, pavement/cleared lot, road/railroad, pipes, landfill/trash, park/lawn, row-crops, pasture/hay field, logging operations, and mining activity. Each of the preceding categories of human disturbance was measured as "not present" = 0, "occurs > 10 m from bank" = 1, "occurs within 10 m of bank" = 2, or "on the bank" = 3.

Legacy tree characteristics were surrogate measures of riparian zone disturbance. A legacy tree was identified as the largest (dbh) and tallest tree that could be seen from the channel along the transect line (USEPA 2004). Their characteristics included taxonomic category (e.g., oak *Quercus* spp., sycamore *Platonus* spp., pine *Pinus* spp.), dbh (m), height (m), and distance (m) from the wetted margin (DWM). Legacy tree dbh scores were grouped into five categories: (1) 0-0.1 m dbh, (2) 0.11-0.3 m dbh, (3) 0.31-0.75 m dbh, (4) 0.76-2 m dbh and (5) >2 m dbh. Legacy tree heights were grouped into four categories: (1) <5 m, (2) 5-15 m, (3) 16-30 m and (4) >30 m. The perpendicular distance of the legacy tree to the wetted margin was estimated directly by the observer from the channel.

Assessment of channel characteristics

Water chemistry

In 2004, samples of water for chemical analysis were taken immediately upon entering the stream at the X-site before transects were demarcated (see Figure 3). These 2004 samples, as well as chemistry samples collected in other states, were specifically analyzed for the WSA project administered by the U.S. EPA. A 4-liter plastic cubicontainer and two 60-cc disposable syringes were used to sample stream water. The 2004 samples were placed in the stream until all physical data were recorded and benthic macroinvertebrate samples had been collected. At the end of the sampling period, the water samples were stored at 4°C, and they were shipped overnight to the U. S. EPA laboratory in Corvalis, Oregon for processing (USEPA 2004). All water samples were processed within 72 hours of collection to minimize effects of organic material decomposition on pH, dissolved carbon and nutrient concentrations in the samples. Dissolved oxygen (DO) was not estimated for the WSA because DO measurements and equipment could not be standardized and/or calibrated in a practical manner among state-specific field crews participating in the WSA.

The 2005 grab samples were collected the same as the 2004 samples, but they were processed at Mississippi State University on the same day they were collected. These samples were not used by the U.S. EPA for the WSA. The 2005 water samples were analyzed with a LaMotte Freshwater Aquaculture test kit (Model AQ-2). Additional water chemistry measurements were taken *in situ.* These measurements included specific conductivity (μS), water temperature (°C) and DO (mg O_2/L). These data were collected with a YSI model 85 meter (YSI Incorporated, Yellow Springs, Ohio) and recorded at the downstream end of the sample reach before angling started. The 2004 and 2005 chemical data were not combined for statistical analyses because they were analyzed using different methodologies.

Physical characteristics of the stream channel

Physical habitat characteristics were measured across transects and encompassed substrate composition, geomorphology, fish cover, and canopy cover (see Figure 4) (USEPA 2004). Cross-sectional measurements of water depth (cm) and substrate composition were taken at five equidistant points from the left bank to the right bank. At each point along the transect, substrate was grabbed by hand and classified according to the following size-class codes: bedrock, concrete/asphalt, large boulder (1,000-4,000 mm), small boulder (250-999 mm), cobble (64-249 mm), coarse gravel (16-63 mm), fine gravel (2.1-16 mm), sand (0.06-2 mm), fine silt/mud, hardpan clay, or wood. In addition, percentage embeddedness was recorded as the percentage of the substrate that was buried in the bottom of the channel (e.g., bedrock or hardpan clay was 0% embedded, whereas sand or silt was 100% embedded). The bank angle (0-360°) was measured with a clinometer at the right and left banks of transects. The clinometer rested on a survey pole, and the measurement was taken when ≥50% of the survey pole was flat against the bank. If undercut banks were present, then the width (m) of the undercut bank was measured. From the left bank, the wetted width, bar width and bankfull widths (m) were measured across transects with a survey pole.

At the left or right bank of transects, the bankfull height and channel incision height were measured. The bankfull height was the vertical distance from the wetted margin to the top of the bankfull or "active" channel. The bankfull channel is defined as the channel that is filled by moderate-sized flood events that typically occur every two or three years (USEPA 2004). These flows do not generally inundate the floodplain, but they shape channel morphology in most streams. The channel incision height was the vertical distance from the wetted margin to the level of the first valley terrace (USEPA 2004). Channel incision height measures the degree of stream down-cutting below the general level of its valley. Stream channels typically respond to incision by aggrading (depositing sediment) downstream of incised "nick-points" to compensate for new channel widths and depths created by the incision process (Montgomery and Buffington 1998; Kauffman et al. 1999; Simon and Rinaldi 2006).

Canopy cover measurements were recorded to quantify the canopy density that potentially shaded transects. Six measurements were taken with a densiometer at both banks facing the middle of the channel. Measurements were taken at the center of the channel while facing upstream, left, downstream, and the right sides of the channel. At each of the six measurement locations, vegetation counts were made from zero to 17, where zero represented no canopy cover, and 17 represented the greatest canopy cover density.

Thalweg profiles were conducted longitudinally between transects. Thalweg measurements occurred at ten equidistant points between transects. The thalweg was defined as the deepest point in the channel with the fastest current flow. Characteristics of the thalweg profile included thalweg depth (cm), presence or absence of a bar, soft sediment, a side channel, and backwater. At each of the ten thalweg points, the channel was classified as a plunge pool, trench pool, lateral scour pool, backwater pool, glide, riffle, rapid, cascade, falls, or dry channel (USEPA 2004). Pools were classified according to the structure that might have formed them, such as large woody debris, rootwad, boulder or bedrock, or an unknown fluvial event. Large woody debris units (LWD) were counted and summed between transects. A LWD unit (diameter at the smallest end >10 cm and length ≥1.5 m) was defined as any woody debris touching the water or hanging over the water from the bank.

Other geomorphologic characteristics included water surface slope, channel bearing, discharge, and the extent of beaver *Castor canadensis* modifications to current flow. Percentage water surface slope was measured at successive transects with a clinometer. Channel bearing (0-360°) was a surrogate measure of channel sinuosity and was measured at the mid-point of transects. An estimate of discharge (m^3/s) was made by using the neutrally-buoyant object method. Specifically, a plastic sphere with holes was floated three separate times at a measured distance (usually 5 m). To obtain the cross-sectional area, the wetted width was recorded at three equidistant points along the distance of the float path, and the distances were averaged. Five depth measurements were taken at equidistant points across each of the three wetted widths, and these distances were averaged. Flow modifications from beaver activity were classified as “none”, “minor”, or “major”. Beaver dams that created pools but allowed a detectible flow across the dam were considered “minor” modifications. Beaver dams that created pools with little or no visible flow across the dam were considered "major" modifications.

A rapid habitat assessment (RHA) was conducted for the entire reach. The procedures for the assessment followed the EPA's Rapid Bioassessment Protocols (Barbour et al. 1999) and incorporated refined procedures that were developed specifically for low-gradient streams. The RHA scores were based on visual estimates of "habitat quality" for resident aquatic organisms. Reach-wide habitat diversity (presence of pools, multiple substrate-types, riffles, etc.) and stability (e.g., degree of bank erosion, substrate dominated by fine particles) were scored as "poor", "marginal", "sub-optimal" or "optimal" for biotic diversity and production. Scores ranged from zero to 120, where larger RHA scores reflected better "habitat quality".

Fish cover characteristics (Figure 4) were measured at each of the ten transects and included the percentage cover of filamentous algae, macrophytes, large woody debris (diameter at largest end $\geq$ 0.3 m), small woody debris (diameter at largest end < 0.3 m), rootwads or live trees, overhanging vegetation, undercut banks, boulders and artificial structures. Depending on their coverage within the transect area, fish cover characteristics were scored (0-4) if they were "absent" (0% = 0), "sparse" (< 10% = 1), "moderate" (10-40% = 2), "heavy" (41-75% = 3) or "very heavy" (>75% = 4).

Sampling protocol for benthic macroinvertebrates

Collecting, sorting, identification and quality control/quality assurance procedures for benthic macroinvertebrate (BMI) sampling followed standardized protocol for the national WSA (USEPA 2006). Benthic macroinvertebrates were collected with a 500 µm-mesh D-frame kicknet at left, center or right portions of transects for a period of 30 seconds (n=11 transects/reach). The starting point at transect "A" (left, center, or right side of channel) was chosen randomly. All other transects (B through K) were sampled systematically in the order left, center then right sides of the channel (see Figure 3 for systematic sampling design). Sampling points were 1 m downstream of the transect line and 1 m away from banks. Samples from each transect were combined into one jar and preserved in 70-80% ethanol at the site. This reach-wide sample was brought back to the fish research laboratory at Mississippi State University where invertebrates were sorted and identified.

Benthic macroinvertebrates were sorted with 500 µm-mesh gridded screens. Individual grids were subsampled randomly, and all individuals from a selected grid were counted and placed in a 20 ml scintillation vial with 70-80% ethanol. The sample sorting process continued until 500 (± 20%) organisms were counted (USEPA 2004). All organisms were identified to genus using dichotomous keys. Individuals were assigned to functional feeding groups (collectors, shredders, predators, scrapers) and habit-type groups (burrowers, sprawlers, clingers, climbers, swimmers) (Merritt and Cummins 1996). For statistical analyses, I used percentage composition of these behavioral groups (USEPA 2004) as an indicator of a group's abundance relative to other behavioral groups. The identified samples were sent to Rithron Associates, Inc. (Billings, Montana) for taxonomic verification (a quality control procedure for the WSA protocol).

Sampling protocol for catchable-size sport fishes

During summers 2003-2005, fishes were sampled by angling with ultra-light fishing gear consisting of a spinning reel with 1.8 kg test line and a rod 1.7 m long (Figure 5). This sampling technique was chosen because it focused the sampling effort on species of interest in Mississippi's wadeable streams (i.e., catchable centrarchids) (Robinson and Rich 1980; Schramm et al. 1996; Shewmake and Jackson 2004). Angling is an important sampling technique for fisheries resources in streams because it applies directly to fisher experience (potential or realized) in these systems. Angling has been a valuable sampling technique for small stream fish stocks in Alaska (Hetrick and Bromaghin 2006) and catfish stocks in South Dakota streams (Arterburn and Berry2002); especially when other gear types were ineffective at sampling certain sizes or species of fish. Hetrick and Bromaghin (2006) and Arterburn and Berry (2002) reported that stock characteristics from their studies were estimated as well or better with angling compared to other traditional sampling techniques (e.g., electrofishing). Recently, Tsuboi and Endou (2008) found that observed angling CPUE (fly-rod with caddisfly larvae as bait) and expected abundances (observed by snorkeling and electrofishing pools) of white-spotted char *Salvelinus leucomaenis* were not different in a Japanese stream (Chi-square; $P = 0.94$).

In my study, "beetle-spin" lures were used to fish all of the reaches (Figure 5). These lures consisted of chartreuse grub bodies (5.1 cm long), 3.5 g chartreuse jig heads and #0 nickel spinner blades. I chose this lure-type because it effectively captures the target sport fishes, specifically catchable-size bass and sunfish (TL $\geq$ 180 mm and 80 mm, respectively). Two anglers entered the respective reach at its downstream end and fished in an upstream direction until the entire reach was sampled. All fish were processed in

Figure 5. Equipment and methods used to sample sport fishes in 26 Mississippi wadeable streams during the summers of 2003-2005. The top photo is a "beetle-spin" lure used to capture fish. The middle photo illustrates the angling procedure using an ultra-light rod and spinning reel. The bottom photo shows the process of collecting fish data such as total length.

the stream and released. For every fish that was caught, the species was recorded, and the total length (mm) was measured. Scales were removed from the left side and above the lateral line for age analysis. Stream reaches were fished on at least three occasions during June-September 2004 and 2005. In my study, catch per unit of effort (CPUE: fish/angler-hour), as well as percentage composition, were used as indices of relative abundance. These measures were used to characterize fish relative abundances because fishing effort (thus catchability) was standardized, and no fish were removed from the streams (Tsuboi and Endou 2008).

Experimental design and statistical analyses

Objective 1: Identify important environmental gradients

Multivariate gradient analysis at different scales

To reduce size and collinearity of data, and identify important environmental gradients, I used principal components analyses (PCA) to create new, more comprehensive environmental and BMI variables. The primary aim of the PCA was to produce a small number of orthogonal composite variables (i.e., components, gradients or axes) that described much of the variation in the original datasets (McCune and Grace 1999). A separate PCA was performed for watershed-scale data, riparian-scale data, channel-scale data and BMI data as well as the combined watershed, riparian and channel data. I used PC-Ord software (McCune and Grace 1999) to conduct all PCA.

A principle component (i.e., axis or gradient) consists of a set of linear sample scores. In my study, a sample score was calculated for each reach. The sample score reflected the reach's similarity to the other reaches regarding a particular environmental or BMI gradient. The principle components then were used as independent variables in subsequent correlational analyses (e.g., canonical correspondence analysis) to describe sport fish relative abundances. In theory, the principle components were not correlated (McCune and Grace 1999). Therefore, subsequent correlational analyses using PCA sample scores as descriptor variables were less likely to suffer from effects of multicollinearity, compared to analyses that use the actual variables as descriptors (e.g., multiple regression).

I used Pearson correlation coefficients to form the cross-products matrix in the PCA. I interpreted only those principle components with eigenvalues > 1.0. I chose this cut-off value because components with eigenvalues > 1.0 tend to contain more information than expected by chance (McCune and Grace 1999). Variables that loaded heaviest on a component were those with the largest eigenvectors for that component. I used these variables to interpret the meaning of a principle component. The eigenvectors calculated by a PCA are conceptually similar to Pearson's r-statistic (McCune and Grace 1999). Relative to other variables in the dataset, the variables with larger eigenvectors were those most strongly associated with a component. Prior to all PCA, proportion data were arcsine-square root transformed, and all other data were ln-transformed to minimize bias with regard to a variable's unit of measure and to improve univariate normality.

Objective 2: Determine an appropriate scale for modeling catchable-size sport fishes

I used three separate analyses to determine the relative importance of scale on modeling associations among catchable sport fish relative abundances and environmental gradients. First, I conducted a PCA on the combined-scales environmental dataset to determine if environmental characteristics of the streams were hierarchically arranged. If the combined-scales PCA created components with spatially explicit variables, this would suggest that Mississippi wadeable streams were not arranged in a spatial hierarchy (i.e., each component would consist of only watershed-scale data, only riparian data, or only channel-scale data). However, if variables from multiple spatial scales loaded heavily on a particular component, then this would suggest that Mississippi wadeable streams were hierarchically organized. If the streams exhibited a spatial hierarchy, then models would need to take into account this hierarchical structure. Thus, smaller-scale models (e.g., the channel) could potentially be influenced by larger-scale characteristics (e.g., the watershed).

Second, I used Mantel tests (Mantel 1967) to determine if environmental dissimilarity matrices from watershed, riparian and channel scales were correlated. The Mantel test works much like a regression, except that the data are arranged in multivariate matrices. The standardized Mantel statistic (r) functions like Pearson's correlation coefficient (Sokal and Rohlf 1995). It ranges from -1 to 1, and the test produces a calculated P-value by conducting a randomization procedure. In this case, I chose Mantel's asymptotic approximation method to construct a P-value for each test. Each matrix consisted of sample scores from the first six principle components from the scale-dependent PCA in objective 1. I conducted pair-wise comparisons between PCA scores from each scale and scale combination (e.g., watershed scale vs. riparian scale or watershed + riparian scales vs. channel scale). If a significant relationship was found (at α=0.05), then I concluded the environmental data from the scale comparisons were correlated. Consequently, I would consider the streams to be hierarchically structured, and the largest scale in the comparison(s) would be considered the most appropriate scale for modeling sport fish relative abundances in Mississippi wadeable streams. If no significant relationship was observed, then the scale that explained more variation in fish relative abundances in the scale-dependent regression models would be considered the most appropriate scale for modeling purposes.

Finally, I conducted partial canonical correspondence analyses (pCCA) to assess the relative importance of spatial structure and scale on fish relative abundances (ter Braak 1986; Borcard et al. 1992; Cushman and McGarigal 2002). A CCA is a constrained, multivariate ordination that uses multiple linear regressions to ordinate samples in multidimensional space. In our study, the main matrix (i.e., the response variables) consisted of mean total CPUE (all fish species combined), mean total bass CPUE (largemouth bass + spotted bass), mean total sunfish CPUE (all *Lepomis* spp.), mean largemouth bass CPUE, mean spotted bass CPUE, mean longear sunfish CPUE, mean bluegill CPUE and percentage relative abundances of the species/species groups. The second matrices (i.e., the independent variables) consisted of principle components from the scale-dependent PCA.

I used PCA sample scores from watershed-scale data only for the secondary matrix in one analysis, followed by riparian-scale PCA scores and channel-scale PCA scores. Then, I used watershed + riparian-scale PCA scores, followed by watershed + channel-scale PCA scores, riparian + channel-scale PCA scores and finally watershed + riparian + channel-scale PCA scores. I assessed the relative importance of each of these spatial scales and spatial structure (i.e., spatial autocorrelation) by partitioning the percentage variation explained by environmental variables relative to a correspondence analysis conducted on the fish data without the constraint of the environmental variables (Borcard et al. 1992; Grand and Cushman 2003; Gido et al 2006; Wang et al. 2006). I used the geographical coordinates of each stream reach (latitude and longitude at the X-sites) as covariables in each pCCA to assess effect of spatial structure among fish taxa on the environmental correlations at each scale. Similar to partial regression analysis, I isolated the important environmental variables for each scale-dependent CCA that were correlated with fish relative abundances after accounting for potential effects of spatial structure among fish taxa and environmental variables at the other scales. These analyses followed the methods described by Anderson and Gribble (1998) and Grand and Cushman (2003).

I used CANOCO 4.5 software (ter Braak and Smilauer 1998) to conduct all CCA. For all ordinations, proportion data were arcsine-square root-transformed and all other data were log(x) + 1-transformed to minimize bias regarding a variable's unit of measure and to improve normality. I focused on inter-species distances and biplot scaling (ter Braak and Smilauer 1998). At $\alpha = 0.05$, I used the Monte Carlo test with 999 unrestricted permutations under the reduced model to test the statistical significance of canonical axes calculated by each CCA (ter Braak and Smilauer 1998).

Objective 3: Develop candidate models and evaluate model performance

Multiple linear regression analysis

I used multiple linear regression analysis (MLR) to construct testable models and identify important environmental variables associated with catchable sport fish CPUE in Mississippi wadeable streams (Draper and Smith 1998). I used best-subsets regression at $\alpha = 0.05$ to create a set of candidate models. I observed Mallow's C*p*-statistics, adjusted R^2 values and Akaike's Information Criterion (AIC) values to evaluate performance of each candidate model. To evaluate relative strength of independent variables in a candidate model, I calculated partial correlation coefficients (PCC) and standardized regression coefficients (SRC) for each variable in the model. Both of these parameters account for influences of other variables in the models. Variance inflation factors (VIF) were calculated for each independent variable in the candidate models. Variables with VIF < 4.0 do not share a collinear relationship with other independent variables in the model, i.e., they do not artificially increase parameter estimates and their standard errors (Draper and Smith 1998). Therefore, the best candidate models were retained only if they had variables with VIF < 4.0.

Assumptions regarding regression analyses were addressed using studentized residual plots for homogeneous variances and mean error of zero, normality plots of residuals and the Durbin-Watson test for uncorrelated errors (Draper and Smith 1998). To minimize effects of very large or very small numeric values on regression results, all proportion data were arcsine-square root-transformed, whereas all other data were ln + 0.01-

transformed. All regressions were performed with SAS version 9.1 (SAS Institute, Cary, North Carolina, USA, 2004).

Once an appropriate scale was determined, I tested accuracy and precision of MLR models with independent data from 13 different wadeable streams in Mississippi (see Figure 2). I followed model validation procedures by Kocovsky and Carline (2006) and Rashleigh et al. (2005), who used these methods to evaluate regression models that predicted trout abundances Pennsylvania and Mid-Atlantic Highland streams, respectively.

Given the appropriate environmental data from the independent dataset, CPUE responses were predicted for each independent reach. To assess model accuracy, I used the non-parametric Sign test to compare predicted median response from the independent data to their observed median response. This test is similar to a paired *t*-test. Models were considered accurate if no significant differences ($P > 0.05$) were detected between median predicted responses and median observed responses. To evaluate model precision, I regressed mean CPUE of fish stock variables from the independent data using the watershed-scale variables from the candidate models as descriptor variables (MLR; $\alpha = 0.05$). The candidate models were considered precise if coefficients of determination (R^2 values) for the independent regression models were ≥ 0.60.

Objective 4: Compare sport fish CPUE among larger-scale ecoregions and stream size categories

Stream ecosystems tend to be hierarchically organized (Frissell et al. 1986; Allan 1997; Johnson 1997; Poole 2002; Smiley and Dibble 2005; Durance et al. 2006). Thus, biological assemblages and habitats within stream channels are constrained by geological and climatic histories at the watershed or basin scale. One of the limitations of using EMAP data from the WSA is that most of the watershed-scale data reflect human land use instead of more natural characteristics such as soil type or climate. In addition, stream ecosystems naturally change from upstream to downstream. They exhibit geomorphic gradients from headwaters to mouth as drainage area and channel width increases (Vannote et al. 1980). Using the tenets of stream hierarchy as a framework, I compared catchable sport fish relative abundances among two different stream size categories nested within broad, landscape-scale ecoregions.

I tested the hypothesis that angler catch rates of principal sport fishes in Mississippi wadeable streams do not differ with respect to stream size (Gomi et al. 2002) or Level III ecoregion (Omernik 1987; Omernik and Bailey 1997; USEPA 2003). Ecoregions are broad, geographic units containing particular hydrologies, soil types, vegetation, and other physical and chemical characteristics (Omernik 1987). Therefore, a stream within a particular ecoregion is theoretically more similar to other streams within that ecoregion, whereas streams in different ecoregions are not as similar (Omernik and Bailey 1997). I chose Level III ecoregions (USEPA 2003) (Figure 6) as landscape-scale physiographic units because they are more representative of natural geomorphology and climate than

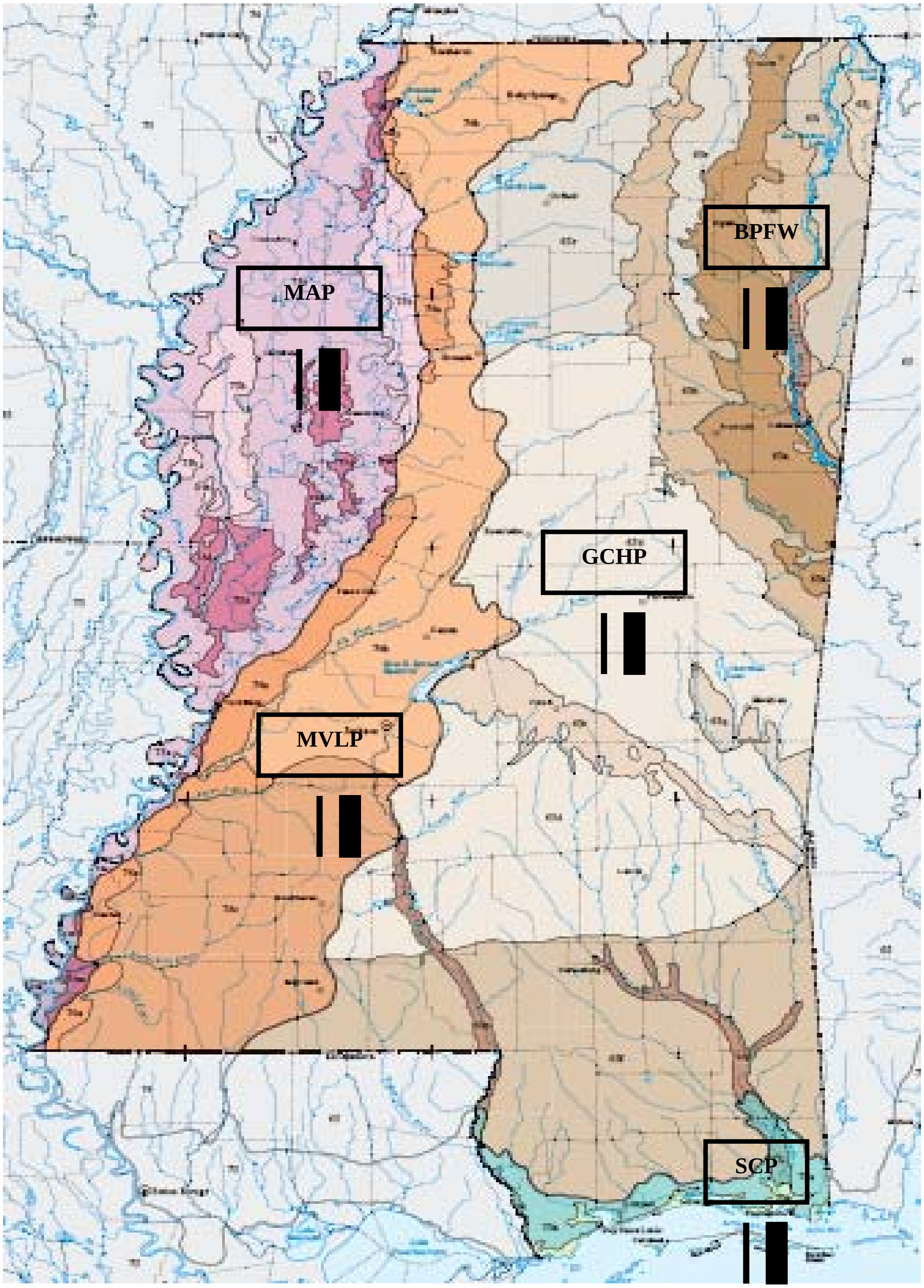

MAP
BPFW
GCHP
MVLP
SCP

Figure 6. An experimental layout used for multiple experimental designs comparing sport fish catch rates (fish/angler-hour) in 26 wadeable streams from different physiographic units. Boxes represent level III ecoregions that occur in Mississippi (USEPA 2003). MAP is Mississippi River Alluvial Plain (purple shades), BPFW is Blackland Prairie-Flatwoods (brown shades), GCHP is Gulf Coast Hills and Plains (beige shades), MVLP is Mississippi Valley Loess Plains (orange shades) and SCP is Southern Coastal Plain (teal shades). Streams were categorized by size as headwater streams (Strahler orders 1-2; thin bars) and network streams (Strahler orders 3-4; thick bars) which reflect their hydrological utility in line with the process domains concept.

watershed-scale data from the WSA. In addition, using ecoregions as experimental treatments is advantageous because watersheds nested within ecoregions can be considered independent, non-overlapping experimental units in the experimental design. I also compared catch rates between two categories of stream size: headwater streams versus network streams (Gomi et al. 2002).

In the context of the process domains concept (Montgomery 1999), headwater streams are sources of sediment erosion and are impacted directly by surface and subsurface runoff from the surrounding landscape. Headwater streams are those classified as stream orders 1-2, and network streams are those classified as stream orders 3-4. I chose these two categories because, from a hydrological standpoint, headwater streams tend to function differently from network streams (Figure 6) (Montgomery 1999; Gomi et al. 2002). Network streams are transport corridors of sediment, and they receive water from upland headwater streams. They continually aggrade and degrade as sediment moves through the channel network from headwaters to large rivers. Because they are physically dissimilar from a hydrological and geomorphological perspective, headwater streams and network streams should differ, theoretically, with respect to their ecological characteristics, which include sport fish populations.

Instead of relying on only one statistical test, I used three experimental designs to test my hypotheses. First, I ran mixed-model analyses of variance (ANOVA) with a nested, repeated-measures design (i.e., split-plot design; *sensu* Maceina et al. 1994 and Keith et al. 1998). Ecoregion was the main plot factor, and stream size was the nested sub-plot factor (Figure 6). These variables were fixed effects in the model, whereas sample date was the random effect (N = 76 samples). I used five Level III ecoregions that occur in Mississippi as the ecoregion treatment: (1) Blackland Prairie-Flatwoods (BPFW), (2) Gulf Coast Hills and Plains (GCHP), (3) Mississippi Valley Loess Plains, (4) Mississippi River Alluvial Plain (MAP) and (5) Southern Coastal Plain (SCP). I incorporated two levels of stream size: (1) headwaters and (2) network. All 26 reaches from the WSA dataset and the independent dataset were used for analyses. The CPUE for each sample date and species/species group were the response variables. Interaction effects (ecoregion*sample date and ecoregion*stream size) and main effects were tested at α = 0.05.

In addition to the nested ANOVA, I used a randomized complete block design (RCBD) to test for differences in mean CPUE among ecoregions and stream size (N = 26). Ecoregion was the treatment, and stream size was the blocking factor. Interaction effects (ecoregion*stream size) and main effects were tested at $\alpha = 0.05$. For these parametric analyses, CPUE data were ln-transformed to meet assumptions regarding normality and homogeneous variances of residuals, and they were conducted in SPSS version 10.0 (SPSS Incorporated, Chicago, Illinois, 2000).

I also used a non-parametric test in support of the latter parametric analyses. In this regard, I compared mean CPUE (N = 26) among ecoregions and stream size categories using a multi-response permutation procedure (MRPP) in PC-Ord statistical software (McCune and Grace 1999). An MRPP is a multivariate analysis that uses a matrix of values as response variables. It uses a randomized, resampling technique to generate a test statistic (A) and a *P*-value for the statistic. Treatment effects (ecoregion and stream size) were considered significantly different at $\alpha = 0.05$. For this test, the main matrix (i.e. response variable) consisted of the mean CPUE of each species/species group. I did not use a blocked MRPP because I did not have a balanced design (McCune and Grace 1999).

CHAPTER III

RESULTS

Summary of environmental, BMI and fish catch characteristics

Watershed-scale characteristics

Geomorphic, hydrologic and land use characteristics varied widely across watersheds. Watershed areas were 0.5-345.8 km^2, and elevation at the X-sites decreased steadily from 137 m above sea level in the northern region of Mississippi to 3 m above sea level near the Gulf Coast. Total stream length was 1.0-783.4 km. Stream density was 0.7-2.0 km/km^2. Total road density was 0-3.4 km/km^2. Percentage population change during 1990-2000 declined slightly in some watersheds (e.g., 1.8% at Middleton Creek), but increases in percentage population change during this period were dramatic in some watersheds, especially those nearer the Gulf Coast (e.g., 48.8% for Jourdan River and 26.5% for West Hobolochitto Creek).

Except for Bogue Phalia, the watersheds were covered primarily by natural vegetation (11.6% for Bogue Phalia, and 62.3-100% for the remaining watersheds). Likewise, percentage forest cover dominated the vegetation structure in the watersheds; forest cover varied from 61.1% to 100% at most sites, whereas it was only 1.3% at Bogue Phalia. Wetland cover was small in most watersheds, varying only from 0 to 17.7%. Agriculture (row crops, silviculture and pastures/hayfields combined) comprised most of the land use activity in the watersheds (0-87.6%), followed by percentage of urban areas (0-11.8%).

Riparian-scale characteristics

Hardwoods and mixed hardwood-conifer vegetation dominated the riparian assemblage structure in the 10 m wide riparian zones. Relatively young, secondary growth (percentage of trees with dbh < 0.3 m) dominated the riparian canopy cover. The understory riparian structure (vegetation 0.5-5 m tall) consisted of relatively even densities of woody shrubs/saplings and grasses/herbs/forbes. Riparian ground-cover (vegetation < 0.5 m tall) was primarily dominated by grasses/herbs/forbes with minimal amounts of barren dirt.

Remotely sensed land cover/land use characteristics within 0, 30 m and 300 m wide riparian buffers reflected the land use/land cover characteristics at the watershed scale. Percentage forest cover dominated land cover at most sites, except for Bogue Phalia, which was dominated by row-crop agriculture.

Channel-scale characteristics

Dissolved organic carbon varied broadly from 1.3 mg/L to 8.3 mg/L, total phosphorus varied 6.9-918.9 µg/L, ammonia-nitrogen varied 0.6-6.1 µeq/L and nitrate-nitrogen varied 0-31.0 µeq/L. Specific conductivities in some streams were low (<40 µS/cm), primarily the Southern Coastal Plain streams. However, two of these streams, Little Bogue Homo and Magee tributary, had relatively high specific conductivities (136.3 and 142.6 µS/cm, respectively). The "Delta" stream, Bogue Phalia, had the greatest specific conductivity (503.3 μS/cm). Calculated also alkalinities varied widely (e.g., 7.3-4,400.1μeq/L).

In summer 2005, mean water temperatures was 22.0-33.0 ˚C. Mean dissolved oxygen concentration was 2.9-7.0 mg/L. Mean specific conductivity was 39.7-1,019.0 μS/cm. Mean pH was 5.4-7.3 (calculated as the average measured pH using the LaMotte Freshwater Aquaculture test kit).

The streams exhibited geomorphic characteristics typical of wadeable streams that drain coastal plain landscapes. Because the channels were unconstrained in relatively broad valleys, they were generally much wider relative to their depth (mean width:depth ratio 28.7). Based on average wetted widths, reach lengths were 150-792 m. Mean wetted widths were 1.2-19.3 m. Mean thalweg depths were 8.0-148.3 cm.

Channel morphologies were typical of glide-pool streams that contain shifting, small substrates. Glides and pools comprised 96.3% of the 4.4 total kilometers of stream length (all reach lengths combined), whereas riffle units made up only 3.7%. The average maximum residual pool depth was 75.0 cm. In general, the streams had low discharges (< 3.1 m^3/s), except for West Hobolochitto Creek (discharge = 8.3 m^3/s). The streambeds were relatively unstable (mean RBS = 59.7), because they were primarily composed of fine-grained substrates (mean substrate diameter = 5.3 mm) such as clay, silt, sand and fine gravel.

Overhanging vegetation dominated in-stream fish cover followed by SWD (< 10 cm dbh) and LWD, with total mean percentage cover scores (PCS) of 0.56, 0.28 and 0.12, respectively. Percentage canopy cover reflected the dominance of overhanging vegetation in these small streams, averaging 78.5% at the banks and 71.3% at mid-channel. Attached algae, macrophyte, undercut bank and rock/boulder fish cover were very sparse (total mean PCS 0.01).

Benthic macroinvertebrates

Total numbers of benthic macroinvertebrates (N = 3,079) collected from these streams varied from 96 to 429 individuals per reach-wide sample. Immature dipterans were the most numerically abundant aquatic insect group in the samples, especially chironomids (51.5-93.4% relative abundance), followed by the Ephemeroptera (0-20.0% relative abundance), Trichoptera (0-27% relative abundance) and Plecoptera (0-18.8% relative abundance). Behavioral guilds were primarily dominated by clingers (26.2-82.3%), followed by burrowers (5.5-49.4%), climbers (8.3-33.0%), sprawlers (3.4-32.3%) and swimmers (0-9.6%). Feeding guilds were dominated by collector-filterers (3.9-52.0%) and collector-gatherers (11.8-56.6%), followed by predators (2.3-31.7%), scrapers (0.8-30.4%) and shredders (6.1-34.4%).

Fish catch characteristics

Largemouth bass, spotted bass, longear sunfish and bluegill dominated the total catch, comprising 81% of all sampled individuals (298 fish total). Largemouth bass comprised 38% of the total catch, followed by longear sunfish (20%), spotted bass (12%) and bluegill (11%). Overall, centrarchids made up 95% of the total catch, which in addition to the principle species listed above included white crappie *Pomoxis annularis*, black crappie *Pomoxis nigromaculatus*, shadow bass *Ambloplites arriommus* and redear sunfish *Lepomis microlophus*. Non-centrarchid fishes that were caught included channel catfish, shortnose gar *Lepisosteus osseus*, spotted gar *Lepisosteus occuleatus*, freshwater drum *Aplodinotes grunniens*, pickerel *Esox* spp. and creek chub *Semotilus atromaculatus*.

Mean total CPUE averaged 0.85 fish/angler-hour (S.E. = 0.14) and varied from zero to 2.5 fish/angler-hour among reaches. Mean total bass CPUE and mean total sunfish CPUE averaged 0.57 (S.E. = 0.11) and 0.35 (S.E. = 0.07) fish/angler-hour, respectively. Mean CPUE of largemouth bass, spotted bass, longear sunfish and bluegill averaged 0.22 (S.E. = 0.05), 0.18 (S.E. = 0.04), 0.19 (S.E. = 0.06) and 0.11 (S.E. = 0.04) fish/angler-hour, respectively. On average, mean total lengths of largemouth bass and spotted bass were 219 mm (S.E. = 12.3) and 194 mm (S.E. = 6.9), respectively. Longear sunfish and bluegill mean total lengths were 126 mm (S.E. = 2.0) and 157 mm (S.E. = 4.4), respectively.

Objective 1: Identification of important environmental gradients

Watershed-scale PCA

I retained the first six principal components for interpretation of the PCA on watershed-scale data (Table 1). These six axes represented 91.4% of the data in the original watershed matrix, which consisted of 26 variables. I interpreted axis one as a land use gradient. Percentage unnatural land use and percentage total agriculture were correlated negatively with this axis and represented 40.3% of the total variation in the watershed-scale data. Axis two was a forest cover gradient (19% of the total variation).

Table 1. Scale-dependent principle components analyses (PCA) on environmental and benthic macroinvertebrate data collected from 13 wadeable streams in Mississippi during summer 2004. Signs (+ and -) represent the direction of the correlation between the variables and the axis.

Interpretation of first six axes	Eigen-value	Percentage variance represented in original matrix	Cumulative percentage variance represented
Watershed scale			
Land use (-)	22.2	40.3	40.3
Forest cover (+)	10.5	19.0	59.3
Road density (-)	9.4	17.1	76.4
Upland agriculture-population density (-)	3.5	6.3	82.7
Latitude (+)	3.2	5.8	88.5
Elevation (+)	1.6	2.9	91.4
Riparian Scale			
Agriculture (+)	22.4	39.5	39.5
Forest cover (-) and pipes (+)	10.1	17.4	56.9
Structural complexity (+)	5.7	9.8	66.7
Wetlands (-)	5.0	8.6	75.3
Urban land use and open canopy (+)	3.5	6.1	81.4
Canopy cover (-)	2.7	4.7	86.1
Channel scale			
Alkalinity-conductivity-pH (-)	20.5	25.6	25.6
Channel size (-) and fish cover density (+)	16.4	20.5	46.1
Residual pool depth and density (+)	9.8	12.3	58.4
NH_4-small woody debris-fines (+)	7.8	9.7	68.1
Substrate complexity and turbidity (+)	6.9	8.6	76.7
NO_3-rocky substrates-glides (+)	4.5	5.7	82.4
Benthic macroinvertebrates			
Chironomidae (-) and EPT (+)	20.9	22.7	22.7
Assemblage diversity (+)	19.9	21.6	44.3
Mollusca (+)	14.2	15.5	59.8
Ephemeroptera (-)	9.4	10.2	70.0
Predator richness (+)	7.5	8.1	78.1
Plecoptera (+)	5.1	5.5	83.6

Variables such as percentage forest cover, percentage perforated forest, percentage forest edge and average forest connectivity were positively correlated with the axis. Axis three was a road density gradient. The variables percentage impervious cover from roads, primary highway density and total road crossings were correlated negatively with axis three (17.1% of the total variation). Axis four was an upland agriculture/population density gradient (6.3% of total variation). The variables percentage total agriculture, percentage row crop agriculture and percentage pasture land use on valley slopes > 9%, as well as human population density in 1990 and 2000, were correlated negatively with this axis. Axes five and six represented latitude (5.8% of total variation) and elevation (2.9% of total variation) gradients, respectively. These two variables loaded positively onto their respective axes.

Riparian-scale PCA

I interpreted the first six principal components from the riparian scale data (see Table 1). These axes represent 86.1% of the riparian data, consisting of 39 variables. Similar to the watershed-scale PCA results, axis one was an agriculture gradient (22.4% of the total variation in riparian-scale data). Percentage total agriculture, percentage pasture land use, percentage row crop within 0, 30 m and 300 m wide riparian buffers as well as percentage total agriculture on valley slopes >3% within 0 and 30 m wide riparian buffers were correlated negatively with this axis. Axis two was a forest cover gradient (10.1% of the total variation). The variables percentage natural land use and percentage forest in 0, 30 m and 300 m wide corridors were correlated positively with axis two, whereas proximity of pipes to the stream was negatively correlated with the axis. Axis three was a structural complexity gradient (5.7% of the total variation). The variables canopy + mid-layer woody density and canopy + mid-layer + ground layer woody density were correlated positively with axis three. Axis four was a wetlands gradient (5% of the total variation). Percentage wetland cover in the 0, 30 m and 300 m wide corridors were correlated negatively with this axis. Axis five was an urban land use/open canopy gradient (3.5% of the total variation). Percentage urban land use in the 0, 30 m and 300 m wide buffers and percentage of reach with mid- and ground-layers vegetation were correlated positively with axis five. Axis six was a closed canopy gradient (2.7% of the total variation). Mean canopy density at bank and mean canopy density at mid-channel were correlated negatively with this axis.

Channel-scale PCA

I interpreted the first six principal components derived from the channel-scale data, and they represented 82.4% of the data in the original matrix (see Table 1), consisting of 76 variables. Axis one was an alkalinity gradient (20.5% of the total variation in the channel-scale data). The water chemistry variables pH, conductivity, alkalinity, concentrations of Ca^{+2}, OH^{-}, HCO_3^{-}, CO_3^{-2}, sum of cations, sum of anions and sum of base cations were correlated negatively with axis one, whereas H^{+} concentration was positively correlated with the axis. Axis two was a channel size/fish cover gradient (16.4% of the total variation). Bankfull width, wetted width and discharge were correlated negatively with this axis, whereas percentage overhanging vegetation was associated positively with the axis. Axis three was residual pool gradient (9.8% of the total variation). Maximum residual pool depth, number of residual pools > 1m deep and mean vertical profile area of residual pools were correlated positively with axis three. Axis four was a gradient of ammonium, small woody debris and detritus (i.e., coarse particulate organic matter or CPOM) and fine substrates (i.e., silt and clay) (7.8% of the total variation). Axis five was a substrate complexity gradient (6.9% of total variation). Axis six was a gradient of nitrate, rocky substrates and reaches primarily composed of glides, because percentage of reach with pools was correlated negatively with the axis (4.5% of the total variation).

Benthic macroinvertebrate PCA

The first six principal components derived from the BMI matrix represented 83.6% of the total variation in the original data (see Table 1), which were comprised of 28 metrics. Percentage Chironomidae individuals was correlated negatively to axis one, whereas EPT richness was correlated positively to axis one (20.9% of the total variation in the BMI data). Axis two was an assemblage diversity gradient (19.9% of the total variation). Total richness and Shannon-Wiener uncertainty index (Brower et al. 1997) were correlated positively to the axis, whereas Simpson's Dominance index (Brower et al. 1997) and sprawler species richness were correlated negatively with axis two. Percentage Mollusca individuals and percentage Ephemeroptera individuals represented axes three (14.2% of the total variation) and four (9.4% of the total variation), respectively. Axis five (7.5% of the variation) was a predator species richness gradient, and axis six (5.1% of the total variation) was a Plecoptera relative abundance gradient.

Objective 2: Determine an appropriate scale for modeling catchable sport fish

Combined-scales PCA of environmental data

The comprehensive PCA on the combined-scales environmental dataset supported the results of the scale-dependent PCA. Most of the variation in the environmental data consisted of a large number variables from different spatial scales (i.e., watershed + riparian + channel). Sixty-five percentage of the total variation was represented by axes that were defined by interrelated variables from multiple spatial scales (Table 2). I interpreted the first three principal components, because sample scores from this PCA were not used in subsequent correlational analyses (i.e., CCA).

Many of the variables that loaded heaviest on the combined-scale PCA were the same variables that defined the gradients in the scale-dependent PCA from objective 1 (Table 3). Axis one was a land use gradient (31.8% of the total variation in the environmental data). A combination of watershed- and riparian-scale variables defined this axis, including percentage unnatural land use, percentage total agriculture, percentage pasture, percentage row crop, percentage pasture and row crop on valley slopes > 3%, and residential/rural road density. Additionally, the watershed variables total road density and primary highway crossings loaded heaviest on this land use gradient.

A combination of watershed, riparian and channel scale variables helped define axis two, representing 20.5% of the total variation in the dataset. This axis was defined as a forest cover/pipes/conductivity gradient. Percentage natural land cover, percentage forest cover, average forest connectivity index and percentage forest interior at the watershed

Table 2. Environmental variables used to interpret the meaning of axes from a principle components analysis (PCA) on combined-scales environmental data collected from 13 wadeable streams in Mississippi during summer 2004. These variables loaded the heaviest on their respective axis with eigenvectors 0.14-0.20. The superscripts "W", "R" and "C" indicate watershed, riparian and channel scale variables, respectively.

Axis name	Variables that defined the axis
Land use	Percentage unnatural land use $^{(W, R)}$
	Percentage total agriculture $^{(W, R)}$
	Percentage pastoral agriculture $^{(W, R)}$
	Percentage crop agriculture $^{(W, R)}$
	Percentage pasture on slopes ≥3% $^{(W, R)}$
	Percentage row-crop on slopes ≥3% $^{(W, R)}$
	Residential road density $^{(W, R)}$
	Total road density $^{(W)}$
	Primary highway stream crossings $^{(W)}$
Forest cover / pipes / conductivity	Percentage natural land cover $^{(W)}$
	Percentage forest cover $^{(W)}$
	Average forest connectivity $^{(W)}$
	Percentage forest interior $^{(W)}$
	Proximity of pipes to stream $^{(R)}$
	Specific conductance $^{(C)}$
	Magnesium ion concentration $^{(C)}$
	Sum of cations $^{(C)}$
	Sum of anions $^{(C)}$
	Sum of base cations $^{(C)}$
	Ionic strength $^{(C)}$
Channel morphology / canopy cover	Mean bank angle $^{(C)}$
	Percentage coniferous riparian canopy $^{(R)}$
	Mean bank canopy density $^{(R)}$
	Mean mid-channel canopy density $^{(R)}$
	Water surface slope $^{(C)}$
	Residual pool maximum depth $^{(C)}$
	Residual pool mean depth $^{(C)}$
	Channel sinuosity $^{(C)}$

Table 3. Results of a combined-scales principle components analysis (PCA) on environmental data collected from 13 wadeable streams in Mississippi during summer 2004.

Names of first six axes	Eigen-value	Percentage variance represented in original matrix	Cumulative percentage variance represented
Watershed and riparian land use	49.3	31.8	31.8
Watershed and riparian forest cover, pipes, conductivity	31.7	20.5	52.3
Channel morphology, riparian canopy cover	19.6	12.7	65.0

scale were correlated negatively with axis two. In contrast, proximity of pipes to the stream (riparian scale), specific conductance, Mg^{+2}, ionic strength and the sums of cations, base cations and anions (channel scale) were correlated positively with axis two.

Axis three was a channel morphology and canopy cover gradient (19.6% of the total variation). A combination of riparian- and channel-scale variables defined this axis. The riparian-scale variables percentage coniferous canopy, mean bank canopy density and mean mid-channel canopy density were correlated negatively with axis three. The channel-scale variables mean bank angle, water surface slope, maximum and mean residual pool depths and channel sinuosity were correlated positively to axis three. The results of this comprehensive PCA suggested that environmental characteristics of Mississippi wadeable streams at larger scales (e.g. watershed) were correlated with environmental characteristics at smaller spatial scales (e.g., channel).

Pair-wise correlation analysis of environmental data from multiple spatial scales

The pair-wise correlation analysis of PCA scores (i.e., environmental gradients) from different spatial scales supported the results of the PCA on the combined-scales data. All Mantel tests suggested that environmental data from different spatial scales were significantly and strongly associated (all r-statistics > 0.46 and all P-values < 0.05; Table 4). For example, PCA scores from watershed-scale data were correlated significantly with PCA scores from riparian-scale data (r-statistic = 0.87; $P < 0.001$) and channel-scale data (r-statistic = 0.46; $P = 0.04$). This analysis also suggests that Mississippi wadeable streams are structured hierarchically.

Table 4. Pair-wise multivariate correlations (Mantel tests) of PCA scores representing environmental characteristics of 13 Mississippi wadeable streams collected at different spatial scales.

Spatial scale comparison	Mantels' *r*-statistic	*P*-value
Watershed vs. Riparian	0.87	< 0.01
Watershed vs. Channel	0.46	0.04
Riparian vs. Channel	0.53	0.01
Watershed + Riparian vs. Channel	0.51	0.02
Watershed + Channel vs. Riparian	0.82	< 0.01
Riparian + Channel vs. Watershed	0.74	0.01

Partial CCA of multi-scale environmental gradients

Canonical correspondence analyses revealed that total amount of variation in sport fish relative abundance data was 0.254, which is the sum of all unconstrained canonical eigenvalues (i.e., the total inertia). Partial CCA results indicated that the percentage of this variation explained by watershed + riparian + channel-scale environmental variables independent of spatial structure was 79.8% (Figure 7), albeit this was not significant statistically (Monte Carlo test; $P = 0.06$). Spatial factors alone (i.e., geographic position of streams with environmental effects removed) explained only 1.3% of the total variation in fish relative abundances (Monte Carlo test; $P = 0.05$). Thus, spatial autocorrelation did not appear to confound the partitioning of the environmental associations with fish relative abundances.

When spatial structure and scaling effects were accounted for (Figure 8), riparian variables explained the most amount of variation in sport fish relative abundances (31.1%), followed by watershed (24.4%) and channel variables (18.9%). However, environmental variables from these three scales are interrelated strongly. Indeed, most of the variation in the sport fish relative abundances (43.7%) is due to confounding interactions among watershed, riparian and channel variables. Watershed variables exhibited stronger confounding effects with channel variables (33.0%) than riparian variables (10.6%). Watershed and riparian variables exhibited very low confounding effects (4.7%).

Plots of the CCA at different spatial scales showed that sport fish relative abundances at the species level differed with respect to their correlations with environmental variables (Figure 9). At the watershed scale, largemouth bass and longear sunfish

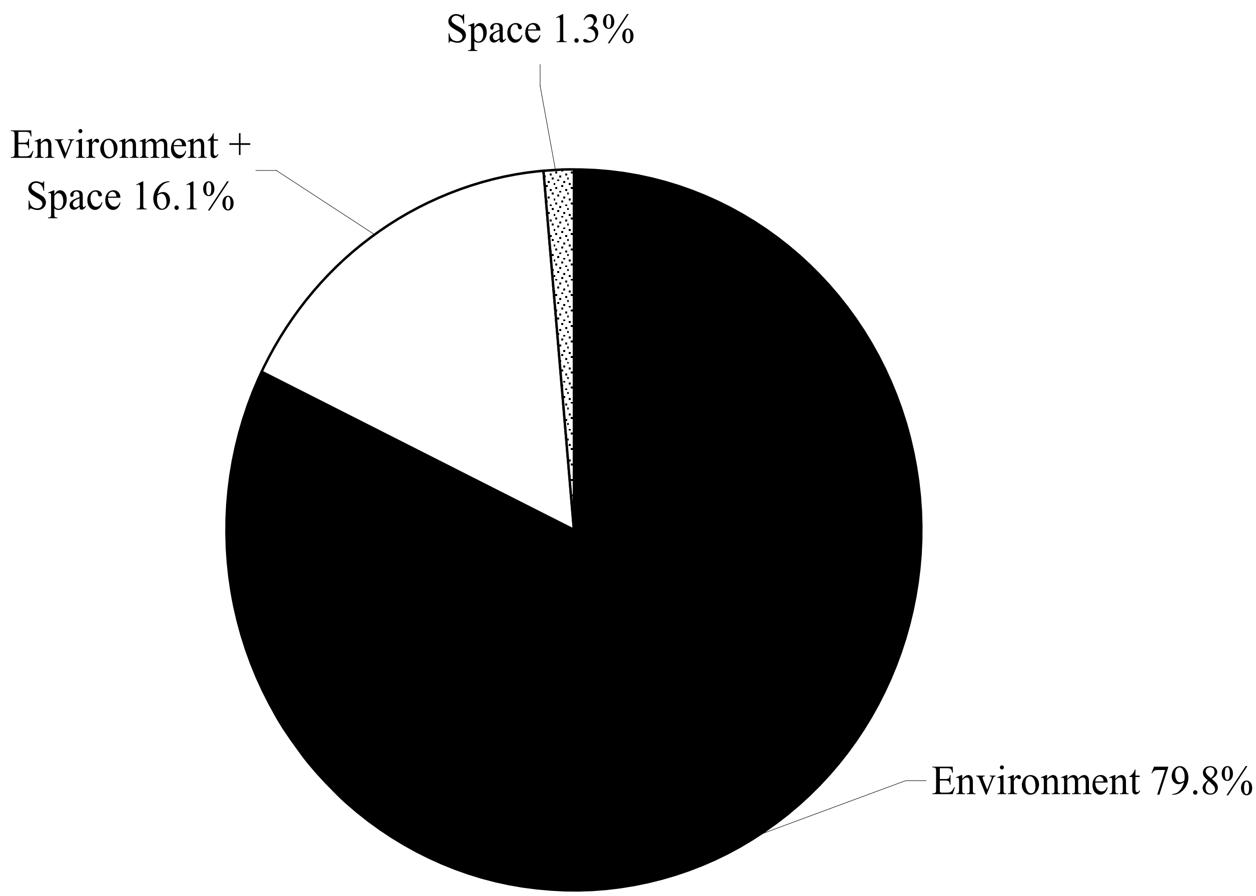


Figure 7. Summary of partial canonical correspondence analyses showing the percentage total variation in sport fish relative abundances explained by environmental variables at watershed, riparian and channel scales, as well as spatial variables (geographic coordinates) and the combination of environmental and spatial variables.

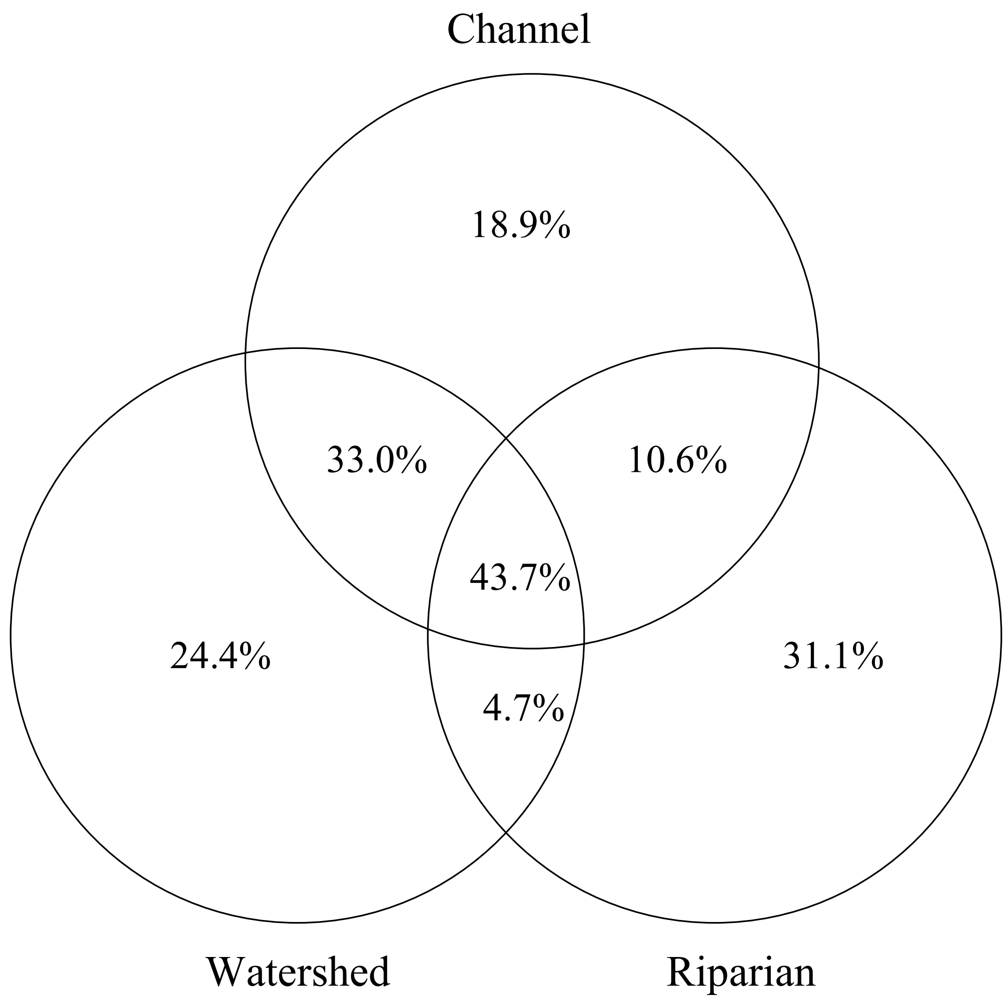
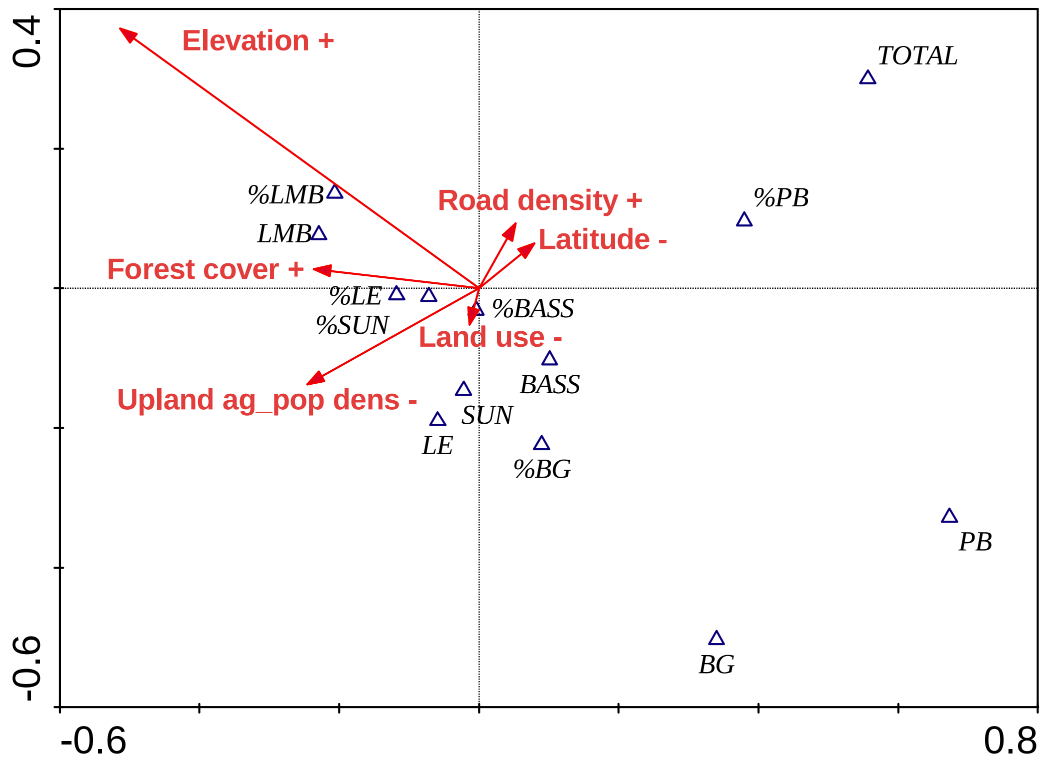


Figure 8. Results of partial canonical correspondence analyses showing the percentage variation in sport fish relative abundances explained by environmental variables from watershed, riparian and channel scales, as well as variation explained by combinations of the three scales.

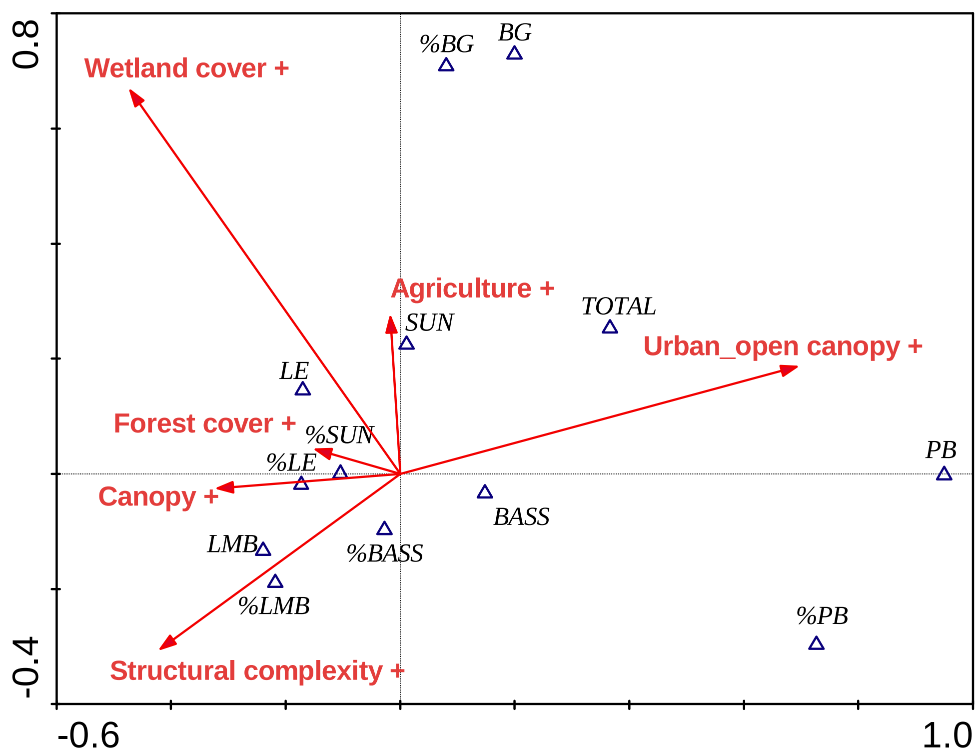
relative abundances were associated negatively with the upland agriculture-human population density and land use gradients, and associated positively with the forest cover, latitude and elevation gradients. In contrast, spotted bass relative abundances were associated positively with the upland agriculture-human population density gradient and associated negatively with latitude and elevation. Bluegill relative abundances and total CPUE were associated positively with the road density gradient. No bass or sunfish were caught in streams draining landscapes dominated by agriculture (>90% land cover), such as those found in the "Delta" region of Northwestern Mississippi.

At the riparian scale (Figure 9), largemouth bass and longear sunfish relative abundances were associated negatively agriculture and urban land use-open canopy gradients but associated positively with the structural complexity (i.e., more vertical layers in the riparian zones) and closed canopy cover gradients. Longear sunfish CPUE was also associated positively with the wetlands gradient. Spotted bass relative abundances and total CPUE were associated positively with the urban land use-open canopy gradient and associated negatively with the wetlands gradient. Bluegill relative abundances were associated negatively with forest cover and closed canopies and associated positively with presence of pipes and agriculture.

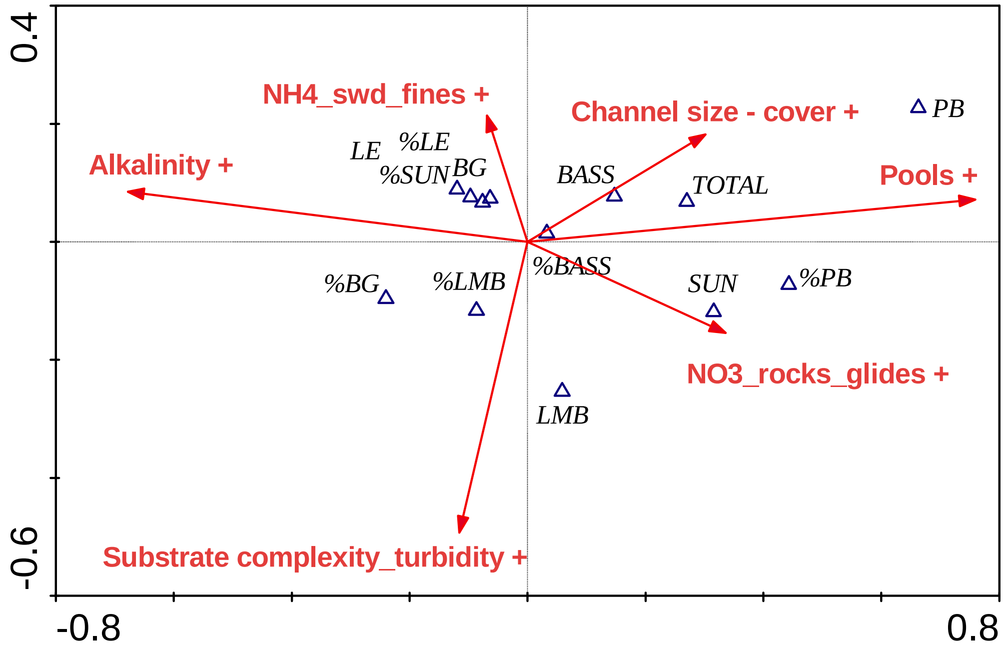
At the channel scale (Figure 9), largemouth bass relative abundances were associated negatively with smaller channel sizes and greater fish cover density, primarily overhanging vegetation and woody debris. In addition, they were associated positively



0.4
-0.6
-0.6
0.8
Elevation +
TOTAL
%LMB
Road density +
%PB
LMB
Latitude -
Forest cover +
%LE
%BASS
%SUN
Land use -
BASS
Upland ag_pop dens -
SUN
LE
%BG
PB
BG



0.8
-0.4
-0.6
1.0
%BG
BG
Wetland cover +
Agriculture +
TOTAL
SUN
Urban_open canopy +
LE
Forest cover +
%SUN
PB
%LE
Canopy +
BASS
LMB
%BASS
%LMB
%PB
Structural complexity +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-0.6
-0.8
0.8
NH4_swd_fines +
Channel size - cover +
PB
Alkalinity +
LE
%LE
%SUN
BG
BASS
TOTAL
Pools +
%BASS
%BG
%LMB
SUN
%PB
NO3_rocks_glides +
LMB
Substrate complexity_turbidity +

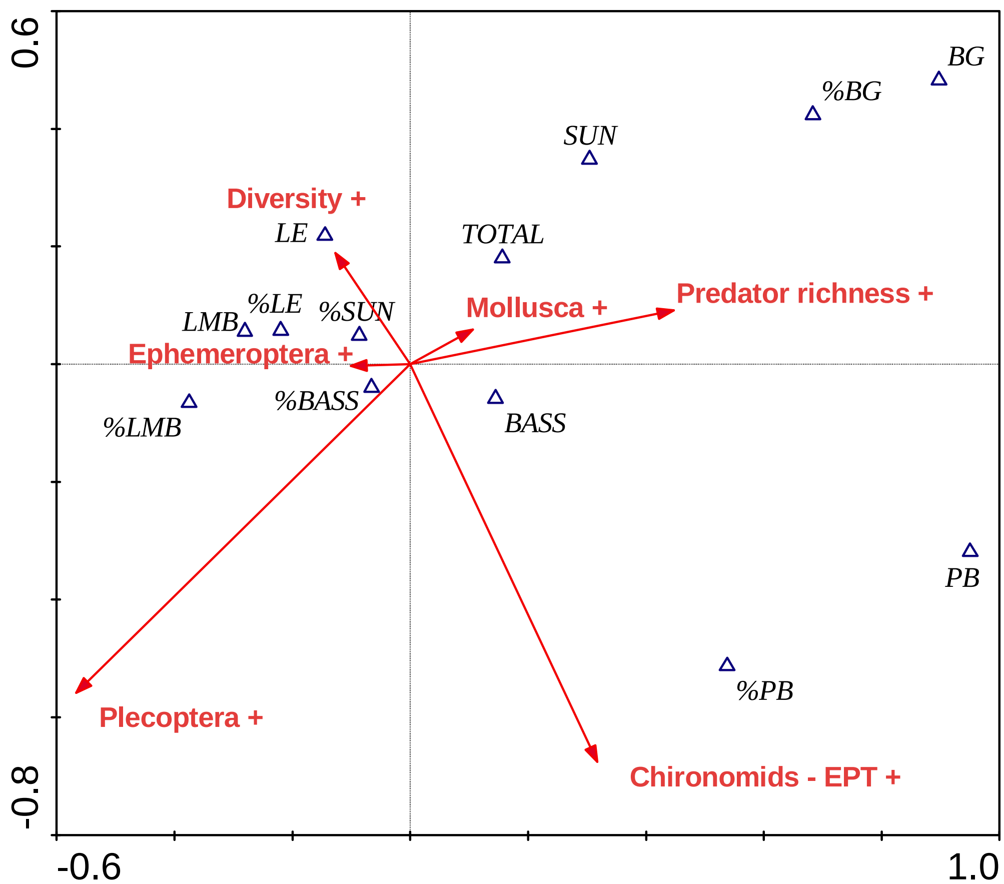

0.6
-0.8
-0.6
1.0
BG
%BG
SUN
Diversity +
LE
TOTAL
%LE
%SUN
LMB
Mollusca +
Predator richness +
Ephemeroptera +
%BASS
%LMB
BASS
PB
%PB
Plecoptera +
Chironomids - EPT +

Figure 9. Joint plots of partial CCA between sport fish relative abundances and environmental variables at watershed, riparian and channel scales in Mississippi wadeable streams during summers 2004 and 2005. Axes 1 and 2 are shown (Monte Carlo tests, 999 permutations, $P < 0.05$). Independent variables (arrows) consist of sample scores from PCA on scale-specific environmental data (see Table 1). The plots are in order from watershed, riparian to channel scales. Plots for each scale were constructed after partialling out confounding effects of variables from other scales and spatial structure among fish taxa. Signs are the correlation direction between descriptor and fish variables. TOTAL = total CPUE, SUN = total sunfish CPUE, BASS = total bass CPUE, LMB = largemouth bass CPUE, PB = spotted bass CPUE, LE = longear sunfish CPUE, BG = bluegill CPUE. %BASS = percentage composition of bass, %SUN = percentage composition of sunfish, %LMB = percentage composition of largemouth bass, %PB = percentage composition of spotted bass, %LE = percentage composition of longear sunfish, %BG = percentage composition of bluegill. Axes 1 and 2 are shown for all plots (Total inertia = 0.254, Monte Carlo test, 999 permutations, $P < 0.05$ for all axes).

with greater substrate complexity and turbidity. Bluegill and longear sunfish relative abundances, total CPUE and total sunfish CPUE were associated positively with NH_4-nitrogen-swd-fine substrates and alkalinity gradients. Spotted bass relative abundances were associated positively with NO_3-nitrogen-rocky substrates-glide habitat and residual pool gradients, reflecting an indirect relationship with watershed size. When sport fish relative abundances were analyzed using environmental variables from all three scales, the results were similar to the preceding scale-dependent CCA (Figure 10).

Taxonomic groups and functional guilds of BMI also explained some variation in catchable fish abundances (see Figure 9). Longear sunfish relative abundances were associated positively with relative abundance of Ephemeroptera individuals and assemblage diversity. Spotted bass relative abundances were associated negatively with abundance of Chironomidae individuals and associated positively with relative abundances of EPT individuals. Largemouth bass relative abundances were associated negatively with predator richness, whereas bluegill relative abundances were associated positively with predator richness. Overall, these results suggest that catchable-size sport fish relative abundances were associated with BMI that tend to be pollution-intolerant.

Objective 3: Develop candidate models and evaluate model performance

The best watershed-scale regression models showed that relative abundances of catchable-size sport fishes were influenced by a combination of human land use and natural geomorphic characteristics. Mean total CPUE, mean total bass CPUE, and mean

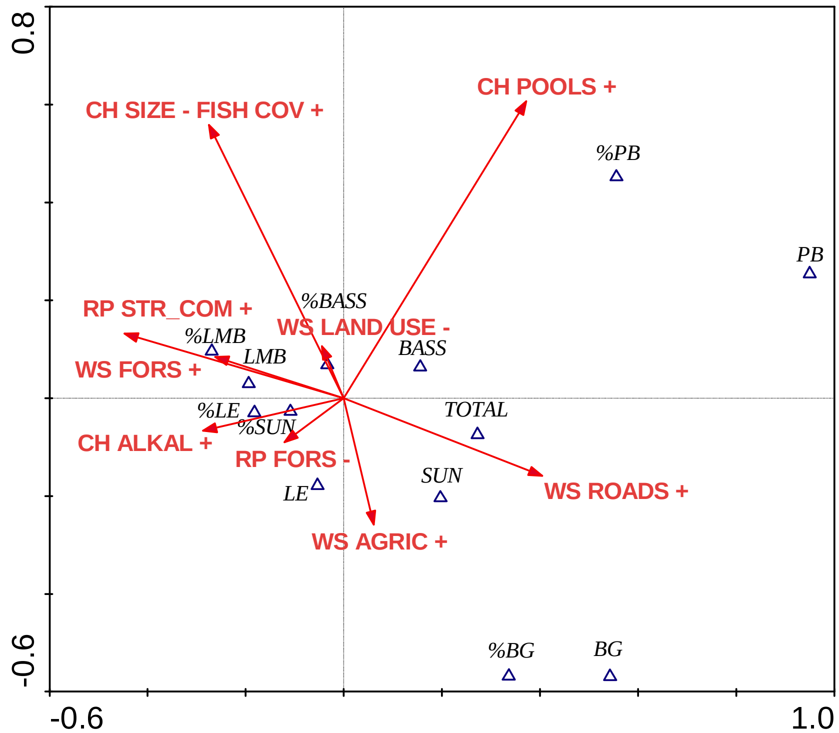


Figure 10. CCA joint plots describing associations between sport fish relative abundances and multi-scale environmental variables independent of spatial structure among fish taxa. Descriptor variables consist of sample scores from PCA on scale-specific environmental data (see Table 1). The first plot is from watershed-scale data only, followed by riparian-scale and channel-scale data. Signs are the direction of correlation between descriptor variables and fish variables. TOTAL = total CPUE, SUN = total sunfish CPUE, BASS = total bass CPUE, LMB = largemouth bass CPUE, PB = spotted bass CPUE, LE = longear sunfish CPUE, BG = bluegill CPUE. %BASS = percentage composition of bass, %SUN = percentage composition of sunfish, %LMB = percentage composition of largemouth bass, %PB = percentage composition of spotted bass, %LE = percentage composition of longear sunfish, %BG = percentage composition of bluegill.

WS = watershed scale, RP = riparian scale and CH = channel scale. Axes 1 and 2 are shown for all plots (Total inertia = 0.254, Monte Carlo test, 999 permutations, $P < 0.05$ for all axes).

total sunfish CPUE were associated positively with percentage forest cover, stream density, total road density and primary highway density (Table 6). Partial coefficients revealed that stream density (PCC > 0.50) and total road density (PCC > 0.60) were stronger descriptors of total CPUE, total bass CPUE and total sunfish CPUE than percentage forest (PCC > 0.47) or rural road density (PCC > 0.24). Mean largemouth bass CPUE was associated negatively with percentage total agriculture but associated positively with rural road density. Percentage total agriculture (PCC = 0.67) had a slightly stronger influence on mean largemouth bass CPUE than did rural road density (PCC = 0.64). Mean spotted bass CPUE was associated negatively with elevation and number of rural road crossings/stream km but associated positively with rural road density. Elevation (PCC = -0.75) had a stronger influence on mean spotted bass CPUE than rural road crossings (PCC = -0.27) and rural road density (PCC = 0.29). Percentage total agriculture (PCC = -0.42) was associated negatively with mean longear sunfish CPUE, whereas rural road density (PCC = 0.51) was associated positively with mean longear sunfish CPUE. Mean bluegill CPUE was not associated significantly with any particular watershed-scale variable.

Candidate models developed for mean total CPUE, mean total bass CPUE and mean total sunfish CPUE were accurate when tested with data from independent reaches. Predicted response values for these catch descriptors did not differ significantly from

Table 5. Linear regression models and their associated statistics describing associations between catchable sport fish CPUE (fish/angler-hour) and watershed-scale characteristics from 13 wadeable streams in Mississippi.

Estimated regression model *	Model statistics
Mean total CPUE = -3.63 + 1.30(PFOR) + 18.14(STRMDENS) + 2.59(RDDENS) + 0.52(RDDENSC1)	$F_{(4,12)}$=18.27 P = 0.0004 R^2 = 0.90
Mean total bass CPUE = -8.03 + 2.25(PFOR) + 17.95(STRMDENS) + 2.69(RDDENS) + 0.53(RDDENSC1)	$F_{(4,12)}$=13.25 P = 0.001 R^2 = 0.87
Mean total sunfish CPUE = -7.46 + 2.07(PFOR) + 16.19(STRMDENS) + 2.36(RDDENS) + 0.62(RDDENSC1)	$F_{(4,12)}$=7.80 P =0.01 R^2 = 0.80
Mean largemouth bass CPUE = 3.17 – 2.20(PAGT) + 3.72(RDDENSC3)	$F_{(2,12)}$=13.54 P = 0.001 R^2 = 0.73
Mean spotted bass CPUE = -4.13 – 1.42(Elevation) + 3.02(RDDENSC3) – 3.44(STRXDC3)	$F_{(3,12)}$=4.84 P =0.03 R^2 = 0.62
Mean longear sunfish CPUE = 1.58 – 1.54(PAGT) + 2.85(RDDENSC3)	$F_{(2,12)}$=6.43 P = 0.02 R^2 = 0.56

* PFOR = percentage of watershed forested; STRMDENS = stream density (stream km/km^2 of watershed); RDDENS = total road density (road km/km^2 of watershed); RDDENSC1 = primary highway density; PAGT = percentage of watershed with agriculture; RDDENSC3 = rural road density; ELEVATION = elevation above sea level (m); STRXDC3 = no. of rural road crossings/stream km

their observed response values (Sign tests; *P*-values = 0.27, 0.58 and 0.58, respectively). The watershed variables from these models also exhibited relatively good precision when tested with the independent data. These variables explained 83%, 71% and 80% of the variation in mean total CPUE, mean total bass CPUE and mean total sunfish CPUE, respectively, in the independent dataset (Table 6).

Models developed for mean largemouth bass CPUE, mean spotted bass CPUE and mean longear sunfish CPUE were not accurate (Sign tests; $P < 0.003$). In addition, the watershed variables were relatively imprecise ($P > 0.15$ for all models) at explaining variation in the independent mean CPUE data, representing only 19%, 32% and 33% of the variation in mean largemouth bass CPUE, mean longear sunfish CPUE and mean spotted bass CPUE, respectively.

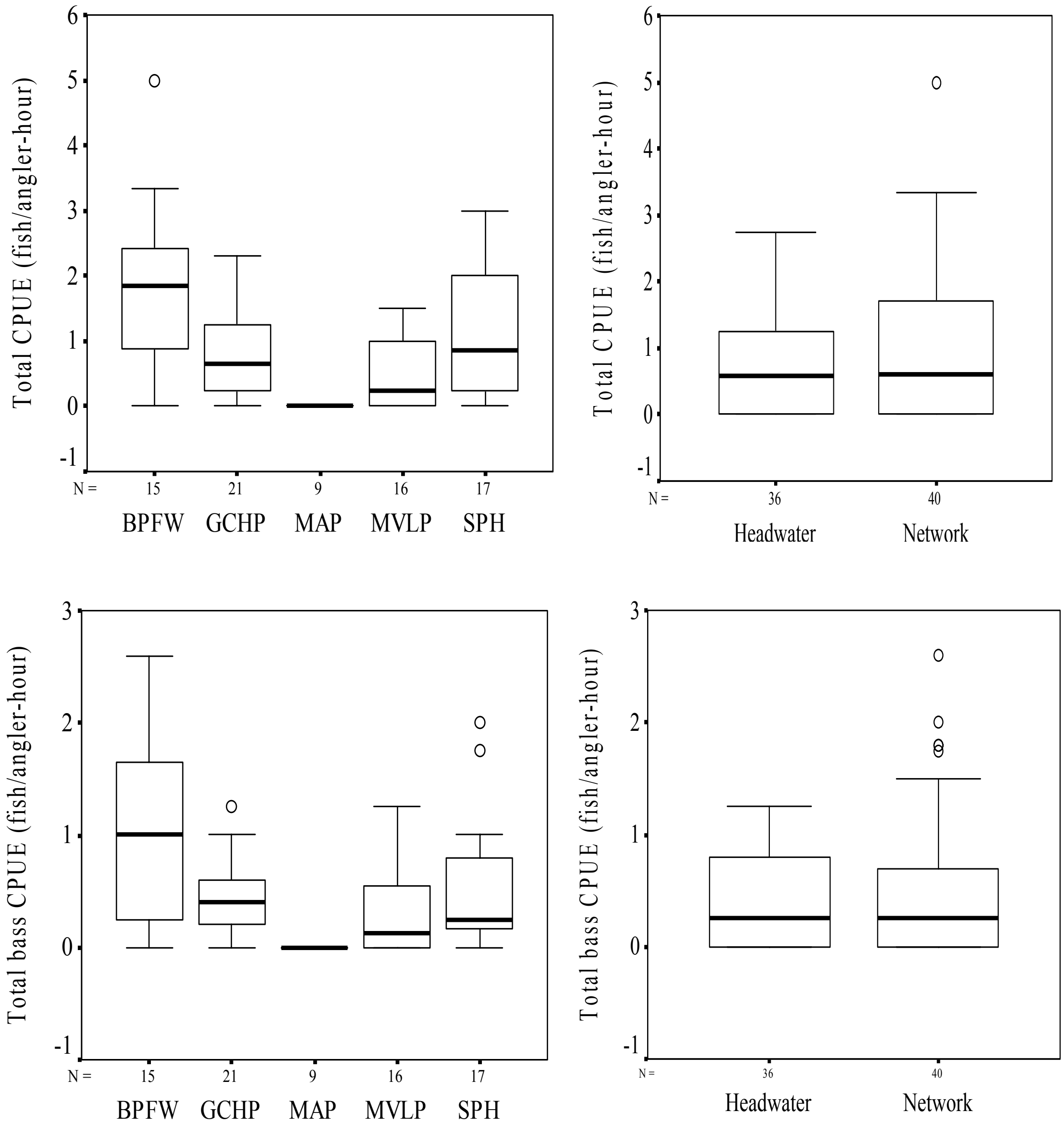
Objective 4: Ecoregion and stream size effects on catchable sport fish CPUE

The repeated measures nested ANOVA with sample dates as replicates (N=76 samples from 26 streams), stream size as the subplot factor and ecoregion as the main plot factor revealed no significant ecoregion*stream size interaction or sample date*ecoregion interaction ($P > 0.05$). Analyses of main effects indicated that total CPUE, total bass CPUE, total sunfish CPUE, largemouth bass CPUE and longer sunfish CPUE did not differ significantly between headwater streams (stream orders 1-2) and network streams (stream orders 3-4), but that they differed with respect to Level III ecoregion ($P < 0.05$; Figure 8). Spotted bass CPUE and bluegill CPUE did not differ with respect to stream size or ecoregion.

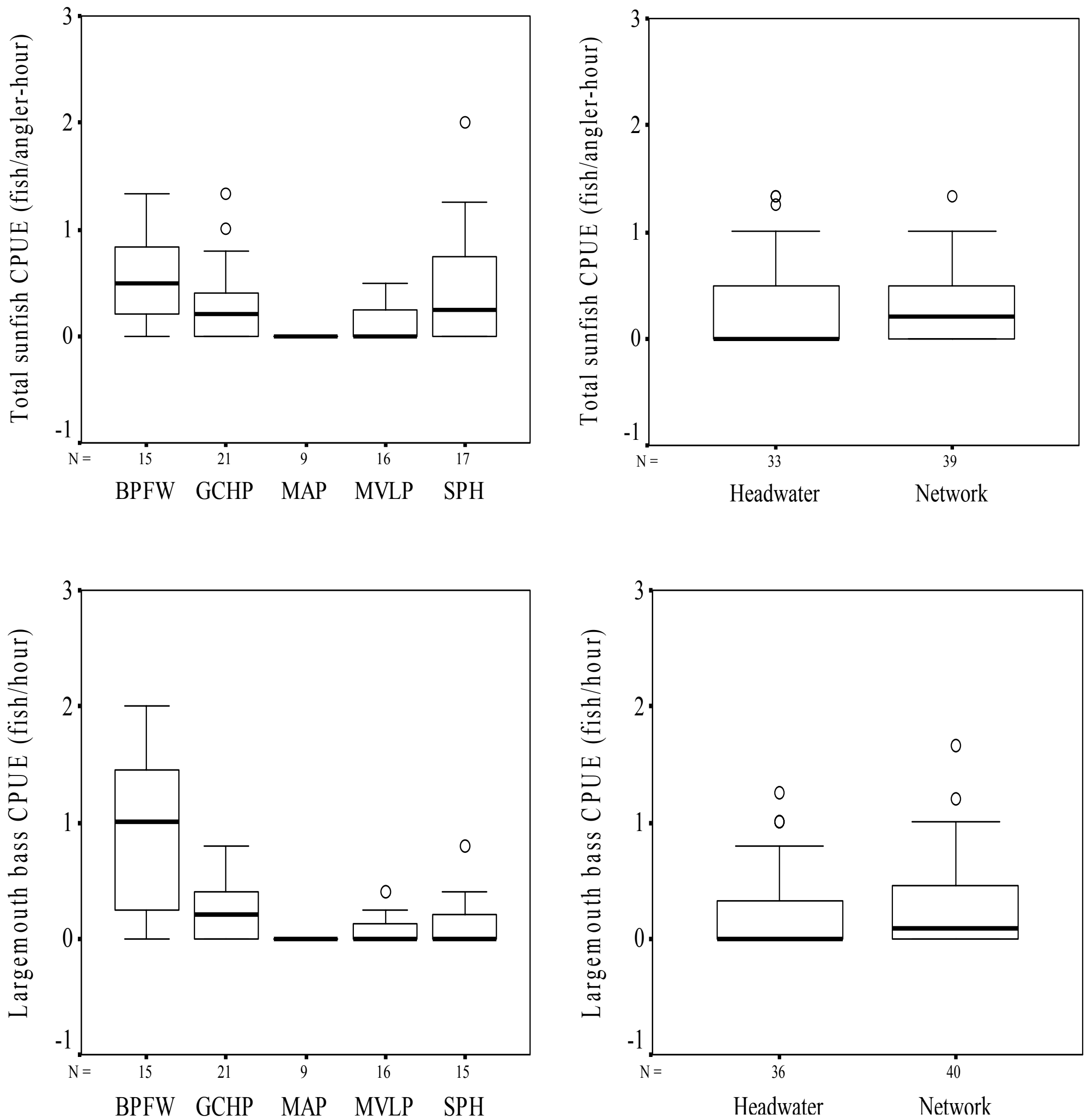
Table 6. Independent regressions evaluating the predictive abilities of watershed-scale variables used to describe mean CPUE (fish/angler-hour) of catchable sport fishes from Mississippi wadeable streams.

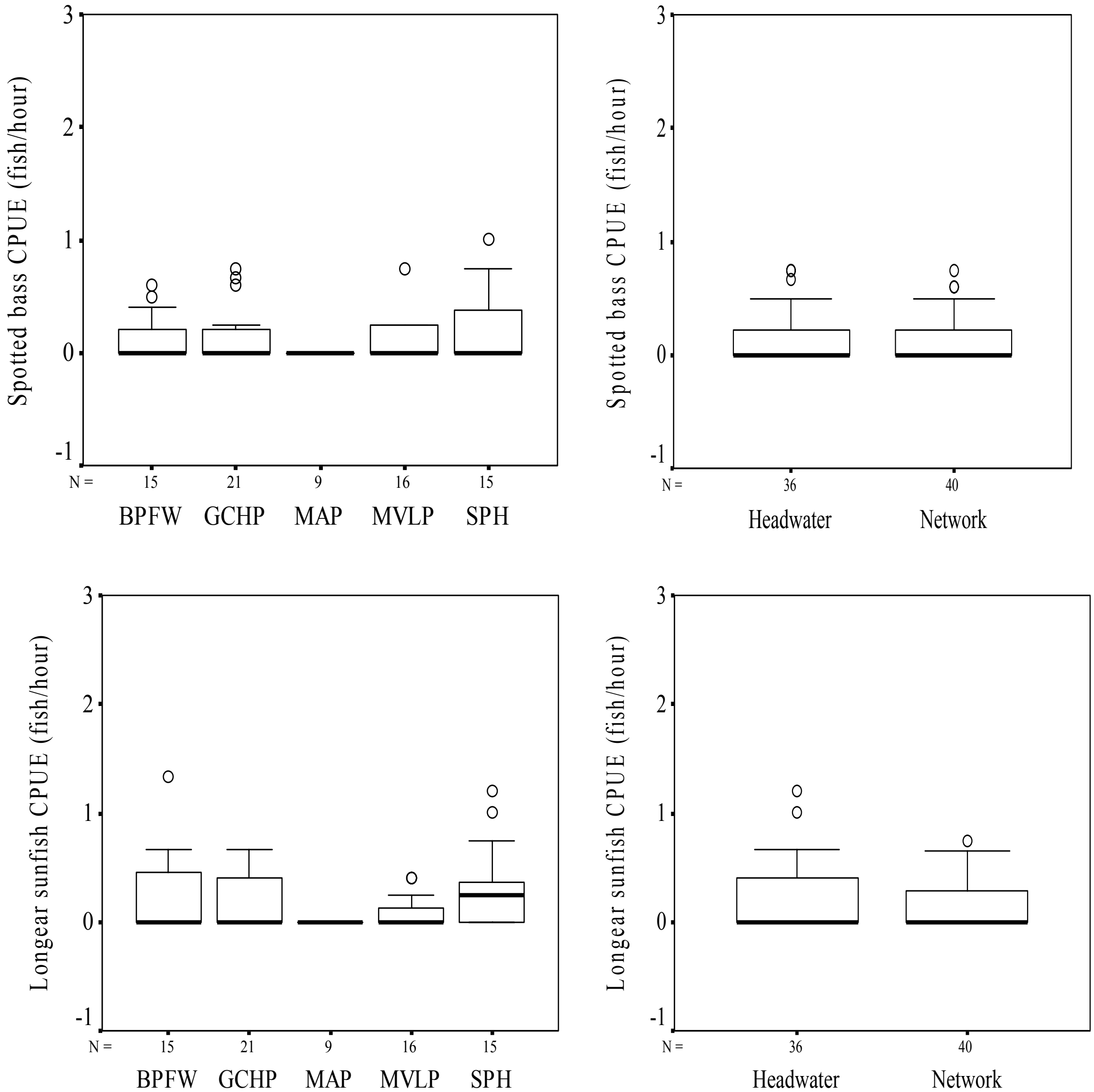
Mean CPUE (fish/angler-hour)	Descriptor variables *	F-statistic	*P*-value	R^2-value
Total catch	PFOR RDDENS STRMDENS RDDENSC1	9.88	0.003	0.83
Total bass	PFOR RDDENS STRMDENS RDDENSC1	4.93	0.03	0.71
Total sunfish	PFOR RDDENS STRMDENS RDDENSC1	7.97	0.01	0.80
Largemouth bass	PAGT RDDENSC3	1.15	0.36	0.19
Spotted bass	ELEVATION RDDENSC3 STRXDC3	1.44	0.39	0.33
Longear sunfish	PAGT RDDENSC3	2.30	0.15	0.32

* PFOR = percentage of watershed area forested; STRMDENS = stream density (stream km/km^2 of watershed); RDDENS = total road density (road km/km^2 of watershed); RDDENSC1 = primary highway density; PAGT = percentage of watershed area with agriculture; RDDENSC3 = rural road density; ELEVATION = elevation above sea level (m); STRXDC3 = number of rural road crossings/stream km



Total CPUE (fish/angler-hour)
N = 15 21 9 16 17
BPFW GCHP MAP MVLP SPH
Total CPUE (fish/angler-hour)
N = 36 40
Headwater Network
Total bass CPUE (fish/angler-hour)
N = 15 21 9 16 17
BPFW GCHP MAP MVLP SPH
Total bass CPUE (fish/angler-hour)
N = 36 40
Headwater Network


Total sunfish CPUE (fish/angler-hour)
3
2
1
0
-1
N =
15
21
9
16
17
BPFW
GCHP
MAP
MVLP
SPH
Total sunfish CPUE (fish/angler-hour)
33
39
Headwater
Network
Largemouth bass CPUE (fish/hour)
15
21
9
16
15
BPFW
GCHP
MAP
MVLP
SPH
Largemouth bass CPUE (fish/hour)
36
40
Headwater
Network


Spotted bass CPUE (fish/hour)
3
2
1
0
-1
N =
15
21
9
16
15
BPFW
GCHP
MAP
MVLP
SPH
Spotted bass CPUE (fish/hour)
36
40
Headwater
Network
Longear sunfish CPUE (fish/hour)
15
21
9
16
15
BPFW
GCHP
MAP
MVLP
SPH
Longear sunfish CPUE (fish/hour)
36
40
Headwater
Network

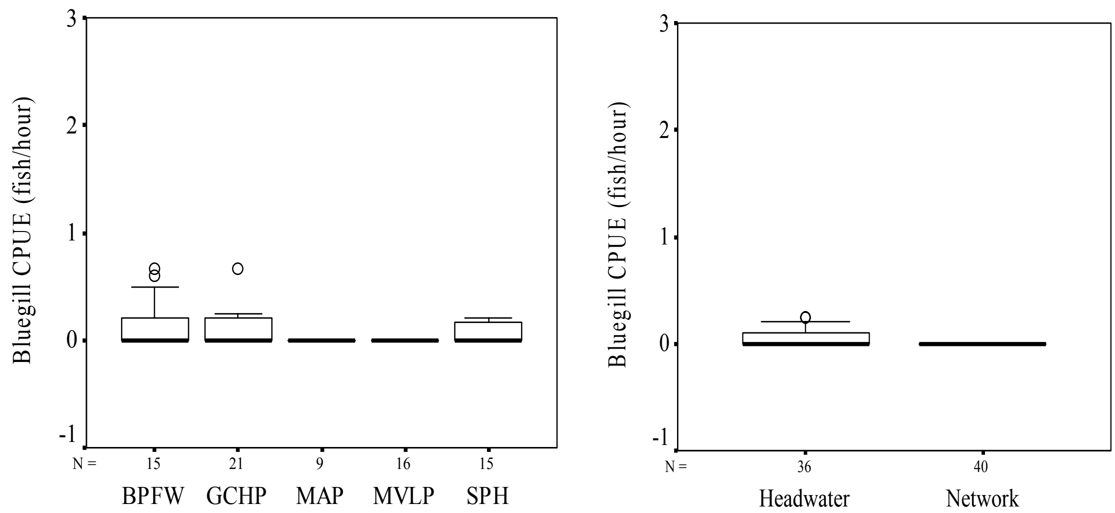


Figure 11. Box and whisker plots comparing CPUE (fish/angler-hour) of Mississippi wadeable-stream fisheries from different stream sizes and ecoregions. Stream size is delineated hydrologically by headwater reaches (stream orders 1-2), and network reaches (stream orders 3-4). Ecoregions are Blackland prairie-flatwoods (BPFW), Gulf coast hills and plains (GCHP), Mississippi River alluvial plain (MAP), Mississippi valley loess plains and hills (MVLP) and Southeastern coastal plains (SCP).

On average, largemouth bass CPUE was greatest in the Blackland Prairie-Flatwoods ecoregion (BPFW) (LSD multiple comparisons; $P < 0.05$). In addition, largemouth bass CPUE from the Gulf Coast Hills and Plains (GCHP), Mississippi Valley Loess Plains (MVLP) and Southern Coastal Plain (SCP) were similar to each other but significantly greater than catch rates from the Mississippi River Alluvial Plain (MAP). Longear sunfish CPUE was similar among BPFW, GCHP and MVLP, whereas longear sunfish CPUE from these ecoregions was significantly greater than that of the MAP. Total CPUE, total bass CPUE and total sunfish CPUE were similar between BPFW, GCHP, and SCP but greater than those of the MVLP and MAP.

Differences in ln-transformed CPUE also were detected using a randomized complete block design with ecoregion as the treatment factor and stream order (1-4 instead of headwater/network) as the blocking factor. There was no significant interaction between stream order and ecoregion ($P > 0.05$) for any of the response variables. Catch rates did not differ significantly among stream orders. However, total CPUE, total sunfish CPUE and largemouth bass CPUE did not differ significantly with respect to ecoregion. No ecoregion effect was detected for total bass CPUE, spotted bass CPUE, longear sunfish CPUE or bluegill CPUE. It must be noted, however, that the assumption regarding homogeneity of residual variances was violated for all tests using both experimental designs (Levene's test; $P < 0.05$). Therefore, caution must be used regarding conclusions obtained from these parametric analyses. Results from multivariate, non-parametric analysis (MRPP) of catch rates (CPUE) among ecoregions and stream size categories were similar to that for the univariate, parametric analyses. The MRPP showed that multivariate CPUE did not differ significantly among ecoregions (T = -11.39; A = 0.15; $P < 0.0001$), whereas multivariate CPUE did not differ significantly with respect to stream size (headwater versus network streams) (A = 0.003; T = -0.55; $P = 0.22$).

CHAPTER IV

DISCUSSION

Multi-scale influences on wadeable stream sport fisheries in Mississippi

A conceptual framework for Mississippi wadeable streams can be viewed holistically, as a functionally integrated system composed of interrelated compartments operating at different spatial and temporal scales (Figure 12). In any particular stream reach, relative abundances of sport fisheries resources are constrained by "trickle-down" effect of hierarchical processes initiated at larger scales (Frissell et al. 1986; Schlosser 1995; Imhoff et al. 1996; Smiley and Dibble 2005; Durance et al. 2006). When a forested watershed is converted to a predominately agricultural, pastoral or urban landscape, the geologic and hydrologic templates shift (Zimmerman et al. 2003). Stream channels in these impacted landscapes respond by altering their flow, sediment and nutrient regimes to a new dynamic equilibrium (e.g., flashier hydrographs with much lesser base flows and greater maximum flows). Consequently, the stream community either resists the physical and chemical changes to habitat conditions, or it shifts to a community dominated by species that will persist in the new environment (Schlosser 1982; Poff and Ward 1990; Harding et al. 1998; Jones et al. 1999; Sawyer et al. 2004).

On a geologic time scale, anthropogenic changes at the watershed or regional scale impart rather abrupt physical disturbances to ecosystems within effected stream channels,

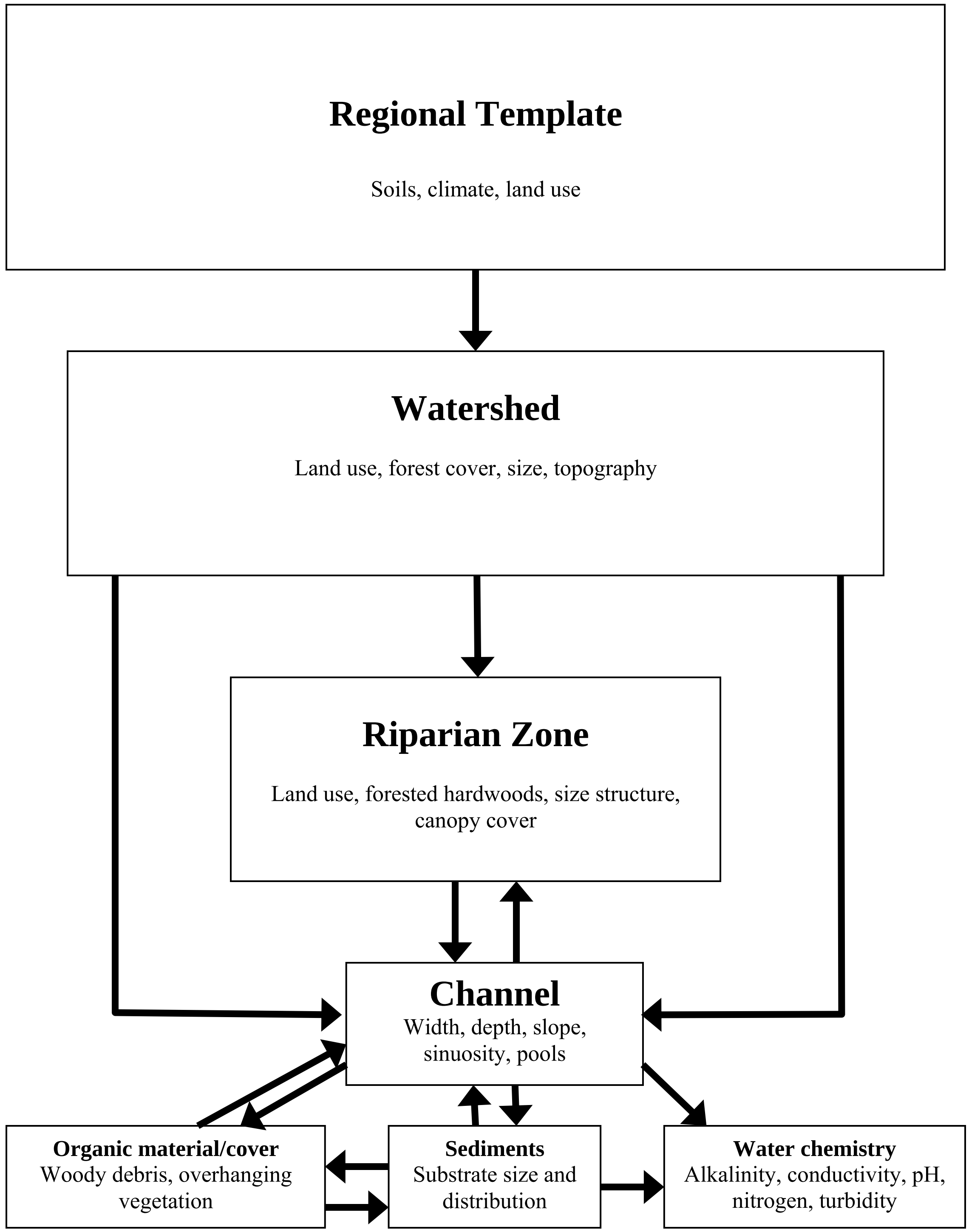


Figure 12. A conceptual model illustrating hierarchical relationships among environmental characteristics that are associated with wadeable stream fisheries in Mississippi.

resulting in altered physical and chemical states over a brief period of days, weeks or years (Poff and Ward 1990; Peterson and Kwak 1999; Chadwick et al. 2006). At any particular point in a stream channel, inorganic chemicals, sediment and organic matter are processed or moved downstream in a span of hours, days, weeks and/or several years. However, the magnitude and variation of available habitat (i.e., the disturbance regime, *sensu* Poff and Ward 1990) for organisms at this scale are constrained ultimately by regional, watershed and riparian templates operating at larger spatial scales.

In Mississippi, conversion of forested landscapes to agriculture yields more open canopies, eroded banks, fewer and shallower pool habitats, greater turbidity and softer sediments in stream channels (Shields et al. 2000; Shields et al. 2003). Consequently, the system shifts from a bass and sunfish fishery to a fishery composed primarily of catfishes and suckers (Catostomidae) (Jackson and Ye 2000; Jackson 2004). Forested streams in the Upper Yazoo River Basin that were cleared to support row-crop agriculture (60-80% of watershed area) support catfish and sucker fisheries in stream channels, whereas centrarchid fisheries (crappies, black bass and sunfishes) predominate in more backwater habitats (e.g., sloughs, oxbow lakes) (Jackson 2004). In addition, channel catfish tend to grow faster in rivers draining proportionately less bottomland hardwood forest relative to agriculture, in addition to greater soil fertility and lower altitude (Shephard and Jackson 2006).

Environmental gradients from different spatial scales are interrelated, and terrestrial components of wadeable stream ecosystems, including riparian zones, watersheds and ecoregions, are intimately connected to bass and sunfish sport fisheries at reach-specific scales. From a multi-scale perspective, substrate composition, water chemistry, woody debris, channel size and residual pools at the channel scale are associated with riparian size structure, canopy cover and land use. Moreover, riparian and channel characteristics are linked tightly to watershed characteristics, particularly land use, forest cover, road density, elevation and latitude. In turn, relative abundances of bass and sunfish fisheries in these systems are associated with environmental characteristics at all of these spatial scales, yet riparian-scale variables appear to have the strongest influence followed by watershed- then channel-scale variables.

Regional-scale characteristics provide the template, and thus constrain physical, chemical and biological dynamics of stream ecosystems at the watershed scale (Omernik and bailey 1997; Zorn et al. 2002; Wang et al. 2003). Ecoregions, for example, have been sculpted by geologic, hydrologic and human forces over centuries and millennia. Likewise, watersheds within ecoregions have been shaped by the forces of precipitation, erosion, sedimentation, vegetation, and human land use (i.e., pre-Columbian peoples) at a millennial pace. Subsequently, stream communities, which include sport fisheries resources, have adapted relatively slowly to physical and chemical fluctuations at the regional scale (Maret et al. 1997; Matthews and Robison 1998).

Ecoregion characteristics influence relative abundances of sport fishes in Mississippi wadeable streams. Angling catch rates of sport fisheries (total CPUE, total bass CPUE and largemouth bass CPUE) were greatest in watersheds draining the BPFW ecoregion in Northeast Mississippi compared to other level III ecoregions in the state. The BPFW ecoregion consists of chalk, marl and calcareous clay soils with smectite or carbonate minerals (USEPA 2003). The soils of this ecoregion are fertile, and its productive capacity potentially influences productivity in aquatic environments. Water moving through watersheds in this ecoregion and into stream channels is typically very alkaline, which is a characteristic that tends to influence primary and secondary productivity in aquatic environments (Kwak and Waters 1997; Townsend et al. 1997; McClurg et al. 2007). Alford and Jackson (2006) found that a densely forested BPFW stream with total alkalinity concentrations up to 155 mg/L $CaCO_3$ during summer base flow had large total angling catch rates (on average, 2.1fish/angler-hour) and relatively large black bass (up to 356 mm TL) compared to other densely forested streams in Mississippi having much lesser total alkalinities (e.g., < 50 mg/L $CaCO_3$).

In contrast to the BPFW, and other level III ecoregions in the state, wadeable streams in the MAP ecoregion will not support bass and sunfish sport fisheries without dramatic reforestation of riparian zones and watersheds. No bass or sunfish were caught using my angling methods in the agricultural MAP ecoregion; only freshwater drum, shortnose gar and spotted gar were caught in these systems. These systems are characterized by chronic sedimentation, chemical inputs (e.g., insecticides, mercury), stagnant base flows, oxygen depletion during warm summer months and removal or downstream transport of woody debris (MDEQ 2003; Shields et al. 2003; Simon and Rinaldi 2006). Even if watersheds in the MAP were converted once again to predominately forested landscapes, (e.g., via Conservation Reserve Programs), decades of land use may still impede the full recovery of bass and sunfish fisheries in this region (*sensu* effects of "the ghost of land use past" in Harding et al. [1998]).

After accounting for collinear effects of spatial structure among samples and environmental variables at other scales, I was able to isolate important scale-dependent environmental gradients to bass and sunfish sport fisheries. In must be noted that my small sample size was small; (N = 13 reaches), thus caution is advised before engaging in a comprehensive state-wide program to manage wadeable stream fisheries based on my results. Nevertheless, at the watershed scale, forest cover, land use and topography are the most important environmental gradients. Largemouth bass and longear sunfish relative abundances increase in smaller watersheds containing dense levels of forest cover in higher elevation latitudes (e.g., > 60% forest cover, watershed area < 100 km^2). In contrast, spotted bass and bluegill relative abundances are greater in larger, lower elevation watersheds with moderate to dense levels of forest cover (e.g., 40-60% forest cover, watershed area 100-500 km^2) and increasing road densities.

Compared to agricultural, pastoral or urban watersheds, forested watersheds minimize surface runoff to streams, keeping them relatively free of chemical pollutants, excessive fine sediments and warmer water temperatures (Bencala 1993; Ziemer and Lisle 1998; Malard et al. 2002; Moerke and Lamberti 2006). In addition, flow regimes (e.g., magnitude and annual variation in discharge) are more stable in forested watersheds than agricultural or urban watersheds (Ziemer and Lisle 1998). Because forests influence water and sediment yields, they ultimately regulate availability of habitats within channels for stream biota at lesser yet very connected trophic levels, including periphyton, diatoms and BMI (Nisbet et al. 1997; Naymik et al. 2005; Kiffney and Roni 2007). Subsequently, forested watersheds are critical to the sustainability of bass and sunfish fisheries in wadeable stream ecosystems (Ziemer and Lisle 1998; Poole 2002; Zorn and Wiley 2006).

Nested within watersheds, I found that largemouth bass and longear sunfish were more abundant in streams with closed riparian canopies, structurally complex riparian vegetation and decreasing or no human land use within 30-m and 300-m buffers. In contrast, spotted bass and bluegill were more abundant in streams with more open canopies and greater human land use in 30-m and 300-m buffers. However, for spotted bass this is likely a function of increased watershed or stream size, based on additional environmental and BMI relationships to spotted bass at the channel scale. Forested riparian zones are the primary source of energy in wadeable stream ecosystems, especially in headwater reaches (Vannote et al. 1980; Kiffney et al. 2003; England and Rosemond 2004). Ultimately, availability of forage for sport fishes depends on the supply of organic matter from the riparian zone (Sponseller and Benfield 2001; Nislowe and Lowe 2006; Dineen et al. 2007), especially from deciduous leaves and woody debris (Gregory et al. 1991; Naiman and Decamps 1997).

At the channel scale, channel size and morphology, fish cover, substrate composition, turbidity, NO_3-nitrogen, NH_4-nitrogen and residual pools were the important environmental characteristics associated with bass and sunfish relative abundances. In contrast to more highland streams, low-gradient streams in Mississippi tend to lack cobble and boulder substrates that typically serve as fish cover for salmonids and smallmouth bass (Binns and Eisermann 1979; Raleigh et al. 1986; Nelson et al. 1992; Sowa and Rabeni 1995). Instead, I found that small woody debris and overhanging vegetation were the dominant sources of fish cover for catchable-size largemouth bass and longear sunfish in Mississippi wadeable streams. Similarly, Insaurralde (1992) found that size-structure (proportional stock density) of flathead catfish *Pylodictis olivaris* was associated positively ($R^2 = 0.61$) with LWD density (number of snags/100 m) in four medium-size Mississippi rivers (wetted width 10-60 m). In addition, he found that relative abundance of flathead catfish was associated positively with percentage of riparian vegetation classified as old growth ($R^2 = 0.77$).

In the Southeastern Coastal Plain ecoregion, woody debris is the primary substrate for attachment by many species of BMI (Benke et al. 1985), because stream beds in this ecoregion consist primarily of shifting sand, clay, silt and small gravel substrates. Woody debris tends to resist flow and remain in a stream reach longer than fine substrates, providing opportunities for periphyton and invertebrates to colonize this structurally stable organic material (Anderson 1978; Wallace and Webster 1996). Stream reaches that have been snagged (i.e., removed of wood for navigation purposes) tend to support less biomass of fisheries resources and less diverse fish assemblages (Angermeier and Karr 1984; Jackson 2000) relative to reaches that are not snagged. Consequently, woody debris functions as an important forage base and habitat for many fishes in this region, especially for centrarchid and catfish fisheries.

Residual pools are important channel-scale characteristics associated with spotted bass relative abundances. Pool habitats, especially during low-flow periods during warm summer months, are important refugia for wadeable stream fisheries. For example, pool depth and frequency are often associated with production or presence/absence of smallmouth bass and salmonid fisheries in more highland streams (Bowlby and Rolf 1986; Rankin 1986; Lyons 1991, Clarkson and Wilson 1995). In fact, habitat restoration and enhancement programs have been oriented specifically towards the creation of pool habitats for sport fisheries using artificial structures (e.g., log weirs, boulders, rock vanes) (Shields et al. 2000; Binns 2004).

The CCA conducted on sport fish relative abundances and BMI gradients also indicated that diverse BMI assemblages (as evidenced by EPT and diversity metrics) and increased trophic complexity (e.g., predator species richness) were associated with relative abundances of the principle species. Most state and federal agencies in the U.S. have developed region-specific indices of biotic integrity (IBI) to assess water quality in their wadeable streams (Lyons et al. 1996b; Roth et al. 1996; Hughes et al. 2004; Bramblett et al. 2005). Regardless of ecoregion, these studies find that streams in good condition (i.e., those with greater IBI scores) tend to be directly related to relative abundances of EPT taxa, diversity measures (e.g., Shannon-Wiener index, taxa richness) and relative abundances of certain guilds such as shredders, predators, climbers and clingers (Bressler et al. 2006). These results suggest that bass and sunfish fisheries in Mississippi wadeable streams are associated with BMI metrics that tend to be fair to good indicators of biotic integrity in these systems (Karr 1991; USEPA 2006; Bressler et al. 2006; Rohasliney and Jackson 2008).

Development and evaluation of model performance

Because Mississippi wadeable streams are structured hierarchically, I chose the watershed scale as an appropriate scale to develop and test habitat models of catchable-size sport fish CPUE in Mississippi wadeable streams. When I tested my watershed-scale regression models with independent data collected from additional Mississippi streams, the models predicted mean total CPUE, mean total bass CPUE and mean total sunfish CPUE accurately, indicating the models for these stock descriptors performed reasonably well. The independent variables from the watershed-scale models explained a large amount of variation in mean total CPUE ($R^2 = 0.83$), mean total bass CPUE ($R^2 = 0.71$) and mean total sunfish CPUE ($R^2 = 0.80$) in the independent dataset. Therefore, the watershed variables percentage forest cover, stream density, total road density and primary highway density can be used by fisheries managers to identify stream reaches in Mississippi that potentially support bass and sunfish fisheries. These watershed models are better than riparian or channel models because the environmental data can be obtained from topographic maps or a GIS at much less cost compared to field work necessary for most riparian and channel data.

However, my watershed models did not predict mean CPUE accurately for the individual species. One of the advantages of using ordination as an exploratory tool is that it can uncover relationships among species that go undetected with more traditional univariate or bivariate analyses such as MLR. The CCA results from my study suggest that biotic relationships, in addition to abiotic associations, influence the relative abundances of principle sport fishes in Mississippi wadeable streams (Stoneman and Jones 2000). For example, resource partitioning (e.g., habitat segregation) may help to explain the gradient in relative abundances between largemouth bass, longear sunfish and spotted bass in the watershed-, riparian- and channel-scale CCA (*sensu* Schlosser 1982; Scott and Angermeier 1998; Sammons and Bettoli 1999).

Predation (or predation risk) may help to explain the dichotomy between largemouth bass and bluegill in the CCA conducted with BMI gradients (Schlosser 1987). Bluegill relative abundances were associated positively with BMI predator species richness, whereas largemouth bass was associated negatively with predator species richness. I attribute this BMI gradient to variation in the number of odonate species in streams where bluegill were captured because abundances of Plecoptera, consisting of primarily predacious species (Merritt and Cummins 1996), were associated negatively with bluegill relative abundances. Immature odonates typically inhabit streams with soft substrates (Merritt and Cummins 1996). Bluegill may be relegated to reaches with proportionately more soft substrates and lesser riparian canopy cover to avoid predation by largemouth bass (Schlosser 1987).

Management perspectives for wadeable stream fisheries in Mississippi

My study supports conservation of forested landscapes to provide sustainable bass and sunfish sport fisheries in Mississippi wadeable streams. There is little evidence that habitat restoration alone improves survival or recruitment of fish to the harvestable size (Riley and Fausch 1995; Roni et al. 2002; Thompson 2006; Budy and Shaller 2007; Cooperman et al. 2007). Over time, artificial habitat structures fail or simply degrade (e.g. wood structures will decompose, rocks will be transported or eroded). These degradations or failures are generally a result of cumulative effects from upstream tributaries and hydrological processes that remain unaddressed such as uncontrolled sediment inputs and inappropriate flow regimes in predominately agricultural or urban watersheds (Frissell and Nawa 1992; Rabeni and Jacobson 1993; Shields et al. 2003). In Mississippi, an appropriate management orientation for wadeable stream bass and sunfish sport fisheries will focus on preservation and reforestation of riparian zones and watersheds. Habitat-specific management activities should be conducted in impacted reaches already containing densley forested riparian zones and watersheds (e.g., streams impacted by channelization or road construction) or reaches in landscapes that will be converted to forest (e.g., abandoned pastures).

The watershed scale is an appropriate scale for managing wadeable stream fisheries. Results from my pCCA analyses show that when environmental variables and spatial structure among fish taxa are accounted for, one spatial scale explains nearly as much information in sport fish relative abundances (12.6%) as to two or three combined spatial scales. This is significant from a management perspective, because collecting additional, but redundant, data from different spatial scales does not substantially increase one's ability to develop meaningful habitat models of these sport fisheries. Moreover, collecting data on stream characteristics at two or three spatial scales is time consuming and more costly than collecting less information at only one scale of resolution.

Fisheries managers must come to terms with this trade-off between collecting a lot of data on a large number of variables that leads to greater model precision, and collecting fewer, more parsimonious datasets that lead to lesser model precision. However, it is important that the models have practical application for fisheries managers. Models should (1) be able to predict response variables (e.g. CPUE) accurately, (2) contain relatively few independent variables (e.g., 1-5), and (3) the independent variables should lack redundancy (i.e., collinearity). My watershed-scale models for mean total CPUE, mean total bass CPUE and mean total sunfish CPUE are accurate (Sign tests; $P > 0.05$). They incorporate only four independent variables that can be measured from topographic maps or a GIS, allowing managers to locate a large number of candidate reaches supporting bass and sunfish fisheries (Johnson and Gage 1997; Creque et al. 2005; Kocovsky and Carline 2006). In addition, these variables exhibited virtually no collinearity (VIF < 1.4 for all variables in models). Once suitable stream reaches are located using the watershed models, the template will be set for more traditional management activities at reach-specific scales of resolution (e.g., stock assessments, habitat management).

Stream management conducted at inappropriate scales creates much uncertainty and leads to unattainable habitat restoration goals for wadeable stream fisheries. When habitat restoration is not the prime management focus, wadeable stream fisheries management has taken a deterministic, stable-state approach to managing fish populations. For example, size or creel limits, which are based upon equilibrium concepts of population dynamics, have been used to manage smallmouth bass and trout fisheries in wadeable streams (Lyons et al. 1996a; Slipke et al. 1998; Fiss and Churchill 2003). However, when viewed in the context of different spatial and temporal scales, stream ecosystems exhibit a great deal of variability with respect to flow, habitat availability, disturbance regime (e.g., flood magnitude and frequency) and productivity. Therefore, fisheries managers should embrace this stochastic dynamic and consider a watershed-scale perspective rather than more a more localized, channel-scale perspective.

Active management of terrestrial landscapes for stream resources can be challenging. Watersheds typically course through property controlled by several landowners who have diverse value systems, and they encompass land areas so large that watersheds may be impractical for active management (e.g., lime application, reforestation). Therefore, public outreach programs (e.g., extension workshops) can be used to educate landowners on the value of conserving forested watersheds, especially regarding their influence on wadeable stream fisheries. Suitable reaches around the state can be promoted as “largemouth bass and longear sunfish streams” or “spotted bass and bluegill streams”, similar to the way other states promote their wadeable streams as “trout streams” or “smallmouth bass streams”. Mississippi’s scenic streams program (Whitehurst 2003) and the Conservation Reserve Program (U. S. Department of Agriculture, Natural Resources Conservation Service) would be appropriate stewardship venues for advancing forest preservation and reforestation of agrarian landscapes. Finally, access to wadeable stream sport fisheries in national forests, national wildlife refuges, state parks, state wildlife management areas and other public lands could be promoted to increase awareness of these resources.

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APPENDIX A

WATERSHED CHARACTERISTICS OF 13 WADEABLE STREAM REACHES IN MISSISSIPPI. FIELD DATA WERE COLLECTED DURING SUMMER 2004-2005. REMOTELY SENSED DATA WERE COLLECTED IN 2001. CC = COLES CREEK, GR = GREEN CREEK, JR = JOURDAN RIVER, BP = BOGUE PHALIA, BC = BRUSHY CREEK, HC = HORSE CREEK, MT = MAGEE TRIBUTARY, CYP =CYPRESS CREEK, LBH = LITTLE BOGUE HOMO, WHC = WEST HOBOLOCHITTO CREEK, MC =MIDDLETON CREEK, CR = COLDWATER RIVER, QT = QUITMAN TRIBUTARY

	CC	GC	JR	BP	BC	HC	MT	CYP	LBH	WHC	MC	CR	QT
Latitude	33.85	34.17	30.41	32.23	31.93	33.33	31.88	31.02	31.74	30.76	31.35	34.85	32.02
Longitude	-89.47	-88.40	-89.54	-90.78	-90.21	-89.66	-89.74	-89.01	-89.00	-89.65	-90.84	-89.39	-89.68
Area (km^2)	13.5	14.7	345.8	148.3	60.9	18.8	7.5	13.8	9.9	29.3	11.6	44.7	0.5
Elevation at (m)	76	76	3	26	70	86	126	33	70	35	83	137	87
% natural land use	83.8	91.3	64.9	11.6	66.7	96.0	44.5	92.8	81.6	70.4	92.8	62.3	100
% unnatural land use	16.2	8.7	35.1	88.4	33.3	4.0	55.4	7.2	18.4	29.6	7.2	37.7	0
% forest cover	83.8	90.5	50.4	1.3	66.0	94.6	44.5	75.1	73.2	61.1	92.8	61.7	100
% wetland cover	0	0.8	14.4	10.3	0.7	1.4	0	17.7	8.4	9.3	0.02	0.6	0
% urban cover	0	0.1	1.0	0.8	3.2	0	11.8	0.2	0.2	0.2	0	0.1	0
% man-made barren	6.7	0.9	2.2	0.002	4.6	1.4	1.6	3.8	6.4	3.4	2.2	0.1	0
% agriculture	9.5	7.7	32.9	87.6	25.6	2.7	42.1	3.3	11.8	26.0	5.0	37.6	0
% pasture	6.2	7.1	27.2	5.4	18.2	1.5	34.7	2.3	7.1	21.7	3.7	20.5	0
% row crop	3.3	0.6	5.7	81.8	7.2	1.2	7.2	1.0	4.7	4.3	1.3	17.0	0

Appendix A continued.

	CC	GC	JR	BP	BC	HC	MT	CYP	LBH	WHC	MC	CR	QT
Average forest connectivity	91.2	93.9	80.4	18.4	85.7	96.4	70.9	89.6	90.2	83.5	95.7	77.6	100
% patch forest	1.3	0.3	5.2	1.2	2.5	0.2	5.0	1.3	1.6	4.4	0.2	3.8	0
% transitional forest	4.0	1.9	7.3	0.03	5.9	0.3	9.3	5.0	4.3	7.1	1.0	8.6	0
% edge forest	12.9	13.4	15.2	0	17.2	2.9	11.6	26.5	18.0	17.2	13.2	12.1	0
% perforated forest	32.1	38.1	13.0	0	26.1	38.2	17.2	19.6	24.8	16.8	25.2	32.9	0
% interior forest	33.4	36.8	9.8	0	14.2	53.0	1.4	22.6	24.4	15.6	53.3	4.4	100
Total stream length (km)	15.6	16.0	383.8	1,783.4	68.8	20.5	5.3	141.1	100.9	295.8	9.6	36.3	1.0
Stream density (km/km^2)	1.2	1.1	1.1	1.2	1.1	1.1	0.7	1.0	1.0	1.0	0.8	0.8	2.0
%impervious cover	0	0	0	0	0	0	10.2	0	0	0	0	0	0
Total road length (km)	14.2	12.9	358.9	1,552.1	99.8	12.0	25.5	149.4	117.3	285.7	5.3	37.6	0
Total road density (km of roads/km^2)	1.0	0.9	1.0	1.0	1.6	0.6	3.4	1.1	1.2	1.0	0.5	0.8	0
Number road crossings/km of stream	0.3	0.1	0.3	0.4	0.3	0.2	1.3	0.2	0.3	0.3	0.1	0.2	0
% human pop. change (1990-2000)	-1.6	-1.3	48.8	-2.6	2.1	15.4	-1.8	7.8	10.0	26.5	1.3	10.1	-1.6

APPENDIX B

RIPARIAN CHARACTERISTICS OF 13 WADEABLE STREAMS IN MISSISSIPPI.

CC = COLES CREEK, GR = GREEN CREEK, JR = JOURDAN RIVER, BP = BOGUE PHALIA, BC = BRUSHY CREEK, HC = HORSE CREEK, MT = MAGEE TRIBUTARY, CYP = CYPRESS CREEK, LBH = LITTLE BOGUE HOMO, WHC = WEST HOBOLOCHITTO CREEK, MC = MIDDLETON CREEK, CR = COLDWATER RIVER, QT = QUITMAN TRIBUTARY

	CC	GC	JR	BP	BC	HC	MT	CYP	LBH	WHC	MC	CR	QT
% canopy and understory layers	100	100	100	90.9	100	100	95.5	100	86.4	100	90.5	95.5	95.5
% understory and ground cover	100	100	100	95.5	100	100	100	100	100	100	95.2	100	100
% canopy, understory, and ground cover	100	100	100	90.9	100	100	95.5	100	86.4	100	90.5	95.5	95.5
% understory-ground cover herbaceous	90.9	13.6	100	59.1	59.1	54.5	47.6	13.6	100	40.9	71.4	86.4	100
% understory-ground cover woody	100	100	100	95.5	100	95.5	100	100	100	100	90.9	86.4	90.9
Mean large trees (>0.3 m dbh) (scored 0-4)	0.33	0.34	0.16	0.03	0.36	0.19	0.09	0.26	0.22	0.14	0.33	0.13	0.30
Mean canopy cover density (scored 0-4)	0.58	0.94	0.71	0.49	0.88	0.41	0.31	0.89	0.63	0.31	0.89	0.55	0.76
Mean ground cover (scored 0-4)	0.67	1.07	0.88	1.00	1.01	0.93	0.77	0.95	0.81	0.73	0.85	0.93	0.56
Mean ground layer as barren (score 0-4)	0.28	0.05	0.07	0.14	0.05	0.05	0.18	0.05	0.07	0.05	0.05	0.05	0.07
Mean vegetation as woody cover (scored 0-4)	1.10	1.45	1.28	1.01	1.45	1.12	1.11	1.63	1.11	0.65	1.32	0.95	1.08
Mean bank canopy density (%)	83.2	100	37.7	47.9	84.2	80.7	88.2	89.0	75.4	*	79.4	86.4	90.1
Mean mid-channel canopy density (%)	81.4	99.2	24.1	5.5	86.8	79.7	90.2	66.0	73.3	*	78.6	83.6	89.4

Appendix B continued.

	CC	GC	JR	BP	BC	HC	MT	CYP	LBH	WHC	MC	CR	QT
%e reach adjacent to natural land use	75.9	89.9	90.8	30.3	79.3	95.2	53.9	98.5	92.4	90.9	94.0	67.8	100
% reach adjacent to forest	75.9	85.6	52.4	2.5	76.1	92.2	53.9	33.9	68.0	52.8	94.0	63.7	100
% reach adjacent to wetlands	0	4.3	38.2	27.8	3.3	2.9	0	64.6	24.4	38.1	0	4.1	0
% reach adjacent to human land use	24.1	10.1	9.2	69.7	20.6	4.8	46.1	1.5	7.6	9.1	6.0	32.2	0
% reach adjacent to urban land use	0	0.3	0.1	0.4	2.52	0	7.4	0.2	0.1	0	0	0	0
% reach adjacent to barren land	5.3	0	0.2	0	1.1	0	0.5	0.8	2.7	2.1	5.0	0	0
% reach adjacent to agriculture	18.8	9.8	8.8	69.4	17.1	4.8	38.2	0.7	4.8	7.1	1.0	32.2	0
% 30-m wide buffer with natural land use	74.3	91.7	88.2	26.4	77.4	94.7	52.4	97.9	91.6	88.8	92.8	63.4	100
% 30-m wide buffer forest	74.3	87.8	52.6	2.0	74.4	91.9	52.4	40.1	68.2	54.4	92.8	60.8	100
% 30-m wide buffer wetlands	0	3.8	35.5	24.4	3.0	2.8	0	57.8	23.4	34.5	0	2.6	0
% 30-m wide buffer human land use	25.7	8.3	11.8	73.4	22.6	5.3	47.6	2.1	8.4	11.2	7.2	36.6	0
% 30-m wide buffer urban	0	0.3	0	0.4	2.0	0	7.6	0	0	0	0	0	0
% 30-m wide buffer barren	5.2	0	0.4	0	1.5	0	1.0	1.2	3.0	2.4	4.5	0	0
% 30-m wide buffer agriculture	20.5	8.0	11.4	73.2	19.2	5.3	38.8	0.9	5.4	8.8	2.7	36.6	0

Appendix B continued.

	CC	GC	JR	BP	BC	HC	MT	CYP	LBH	WHC	MC	CR	QT
% 30-m wide buffer as pasture	14.1	8.0	9.9	5.9	13.2	1.0	33.3	0.7	3.4	7.2	2.6	22.8	0
% 30-m wide buffer row crop	6.4	0	1.5	67.2	6.0	4.3	5.5	0.1	2.0	1.6	0.2	13.9	0
% 30-m wide buffer natural land use	80.6	91.5	70.0	12.5	67.8	95.9	51.1	93.6	82.5	74.9	93.9	59.0	100
% 30-m wide buffer forest	80.6	90.3	52.1	1.3	66.8	93.9	51.1	70.0	70.5	61.8	93.9	58.4	100
% 30-m wide buffer wetlands	0	1.1	17.7	11.2	1.0	2.0	0	23.5	12.0	13.0	0	0.7	0
% 300-m wide buffer with human land use	19.4	8.5	30.1	87.5	32.2	4.1	48.9	6.4	17.5	25.1	6.1	40.9	0
% 300-m wide buffer urban	0	0.1	0.1	0.5	2.5	0	11.4	0.1	0.3	0.1	0	0	0
% 300-m wide buffer barren	7.2	0.7	1.7	0	4.2	1.0	2.2	4.0	5.5	3.8	2.5	0	0
% 300-m wide buffer agriculture	12.2	7.8	28.3	87.0	25.5	3.2	35.3	2.4	11.7	21.3	3.6	40.9	0
% 300-m wide buffer pasture	8.2	7.2	23.4	5.6	18.3	1.6	30.4	1.6	7.4	17.6	3.2	23.2	0
% 300-m wide buffer row crop	4.0	0.5	4.9	81.3	7.1	1.5	4.6	0.7	4.3	3.6	0.5	17.8	0

APPENDIX C

CHANNEL CHARACTERISTICS OF 13 WADEABLE STREAM REACHES IN MISSISSIPPI. FIELD DATA WERE COLLECTED DURING SUMMER 2004-2005. ASTERISKS ARE MEAN *IN SITU* MEASUREMENTS TAKEN IN 2005. CC = COLES CREEK, GR = GREEN CREEK, JR = JOURDAN RIVER, BP = BOGUE PHALIA, BC = BRUSHY CREEK, HC = HORSE CREEK, MT = MAGEE TRIBUTARY, CYP = CYPRESS CREEK, LBH = LITTLE BOGUE HOMO, WHC = WEST HOBOLOCHITTO CREEK, MC = MIDDLETON CREEK, CR = COLDWATER RIVER, QT = QUITMAN TRIBUTARY

	CC	GC	JR	BP	BC	HC	MT	CYP	LBH	WHC	MC	CR	QT
pH	7.1	6.7	6.4	8.2	7.0	6.7	6.3	5.9	7.1	5.8	6.5	6.8	7.8
Conductivity (μS)	89.4	31.0	38.7	503.3	47.8	41.1	39.4	27.5	36.3	31.0	44.5	97.6	142.6
Turbidity (NTU)	13.3	7.4	5.0	23.2	11.2	9.6	20.3	1.8	5.8	11.7	2.3	1.6	23.5
Total suspended solids (mg/L)	39.5	2.0	10.4	40.5	2.8	4.9	18.5	1.0	1.4	28.2	12.5	3.8	55.4
Color (PCU)	15	15	20	15	10	10	0	35	25	30	20	10	15
Dissolved organic carbon (mg/L)	2.3	2.2	5.9	3.9	2.2	2.9	0.7	5.8	7.6	8.3	3.4	1.3	2.4
Dissolved inorganic carbon (mg/L)	9.7	2.6	1.2	53.1	2.7	3.8	5.4	0.4	10.2	2.2	3.8	10.7	18.6
Total P (μg/L)	52.6	23.6	31.0	119.0	35.0	32.4	62.0	6.9	918.9	59.1	26.3	11.7	34.3
Ca (μeq/L)	276.1	115.1	70.3	2,798.6	107.1	125.8	84.7	30.0	829.2	74.6	124.9	438.3	1,035.1
Mg (μeq/L)	230.6	79.9	73.5	1,682.9	90.3	100.1	56.7	41.2	148.6	58.1	107.8	217.4	137.4
Na (μeq/L)	280.1	51.3	131.3	880.0	196.4	127.9	120.4	131.1	239.1	72.8	131.7	109.5	145.7
K (μeq/L)	31.2	25.6	44.5	110.4	41.2	21.6	25.8	11.4	26.0	41.5	37.0	54.5	27.7
NH_4 (μeq/L)	0.8	1.4	0.6	0.6	0.6	0.5	6.1	0.6	2.6	0.9	0.6	3.0	1.6
SO_4 (μeq/L)	61.2	13.9	66.7	648.7	73.6	54.0	6.2	17.8	63.7	59.2	70.7	186.9	93.8
NO_3 (μeq/L)	3.6	2.6	5.9	0	6.1	0	31.0	3.1	8.3	12.2	0	13.1	4.5
Cl (μeq/L)	97.0	43.7	163.0	412.2	134.6	72.5	99.1	149.6	371.5	95.2	126.0	82.4	123.1
Total N (μg/L)	213	149	458	508	161	101	534	265	506	681	225	366	221

Appendix C continued.

	CC	GC	JR	BP	BC	HC	MT	CYP	LBH	WHC	MC	CR	QT
H (µeq/L)	0.1	0.2	0.4	0.01	0.1	0.2	0.5	1.4	0.1	1.5	0.3	0.2	0.02
OH (µeq/L)	0.1	0.1	0.02	1.7	0.1	0.1	0.02	0.01	0.1	0.01	0.03	0.1	0.6
CO_3 (µeq/L)	681.6	152.8	54.6	4,329.7	184.3	221.8	211.8	8.6	713.3	42.6	191.2	642.8	1,489.0
CO_2 (µeq/L)	0.8	0.1	0.01	68.9	0.2	0.1	0.04	0.001	0.8	0.003	0.1	0.3	8.8
Alkalinity (µeq/L)	682.3	152.8	54.2	4,400.1	184.4	221.8	211.4	7.3	714.1	41.2	191.0	643.0	1,498.4
Sum cations (µeq/L)	818.9	273.4	320.5	5,480.5	429.6	376.1	294.2	215.6	1,245.5	249.4	402.3	822.8	1,347.5
Sum anions (µeq/L)	844.3	213.3	290.2	5,461.3	398.9	348.5	348.2	179.2	1,157.6	209.2	388.1	925.7	1,719.9
Sum base cations (µeq/L)	818.0	271.9	319.5	5,479.9	429.0	375.4	287.5	213.6	1,242.9	247.1	401.4	819.7	1,345.9
Organic anions (µeq/L)	22.6	21.5	55.8	38.6	21.3	28.1	6.5	52.5	74.2	75.0	32.5	12.2	23.6

Appendix C continued.

	CC	GC	JR	BP	BC	HC	MT	CYP	LBH	WHC	MC	CR	QT
Mean bank angle (°)	36.4	54.5	33.4	28.6	49.5	37.9	51.2	49.6	40.5	*	38.1	27.0	20.0
Mean undercut bank distance (m)	0.02	0.03	0.03	0	0.01	0.14	0.01	0.02	0	0.14	0.01	0	0.005
Mean bankfull width (m)	6.4	5.0	37.6	21.9	12.4	20.3	2.2	12.6	7.2	20.3	2.2	10.4	3.0
Mean bankfull height (m)	0.86	0.37	1.42	0.93	0.25	0.53	0.73	0.45	0.78	*	0.34	0.37	0.53
Mean channel incision height (m)	3.73	1.09	2.28	3.14	2.39	3.00	1.30	3.00	1.39	*	1.73	1.74	2.93
Relative bed stability	68.9	25.4	184.4	67.7	20.1	44.4	73.8	60.4	79.3	*	23.3	29.5	97.6
Reach length (m)	238	158	673	713	396	198	150	436	198	792	150	150	150
Mean thalweg depth (cm)	33.57	51.16	66.10	69.23	41.88	24.94	19.39	44.74	71.19	148.34	8.00	8.99	11.40
St. dev. of thalweg depth	19.6	27.6	32.0	27.0	19.5	11.9	14.4	20.4	38.0	47.5	7.1	5.8	8.2
Mean wetted width (m)	5.4	4.1	19.3	19.1	9.5	5.0	2.0	10.1	7.1	18.7	1.2	6.9	1.4
Mean width:depth ratio	22.4	11.5	33.2	32.2	28.1	23.1	13.0	29.6	11.6	14.3	37.4	98.7	18.5
Discharge (m^3/s)	0.056	0.142	3.031	0.904	0.815	0.241	0.008	0.349	0.395	8.303	$4*10^{-5}$	0.610	0.0001
Sinuosity (m/m)	1.04	3.60	1.11	1.62	1.49	1.06	1.67	1.07	1.24	*	1.28	1.01	1.92
Mean water surface slope (%)	1.00	0.66	1.40	0.70	0.60	1.20	1.41	1.25	1.05	*	1.02	1.00	2.40

Appendix C continued.

	CC	GC	JR	BP	BC	HC	MT	CYP	LBH	WHC	MC	CR	QT
Mean % embeddedness	70.7	92.2	95.3	95.1	49.4	88.0	24.2	76.6	87.5	68.8	68.4	100	36.5
Mean algae fish cover (scored 0-4)	0	0	0.13	0	0	0	0	0.005	0	0	0	0.03	0
Mean macrophyte fish cover (scored 0-4)	0	0	0	0	0	0	0	0	0.16	0.009	0	0	0
Mean LWD fish cover (scored 0-4)	0.03	0.24	0.14	0.01	0.03	0.36	0.14	0.28	0.11	0.07	0.08	0	0
Mean SWD fish cover (scored 0-4)	0.16	0.74	0.08	0.31	0.04	0.23	0.48	0.34	0.76	0.11	0.34	0.05	0.02
Mean overhanging vegetation fish cover (scored 0-4)	0.71	0.88	0.10	0.07	0.80	0.45	0.82	0.32	0.61	0.07	0.88	0.77	0.80
Mean undercut bank fish cover (scored 0-4)	0.02	0.03	0.01	0	0.01	0.004	0.02	0.01	0	0.02	0.03	0	0.005
Mean rock/boulder fish cover (scored 0-4)	0	0	0	0	0.08	0	0	0	0	0	0	0	0
Mean artificial fish cover (scored 0-4)	0	0	0	0.01	0	0	0	0	0	0	0	0	0.29
Mean fish cover density-all types (scored 0-4)	0.92	1.88	0.33	0.40	0.96	1.06	1.33	0.96	1.48	0.27	1.33	0.82	1.11
Mean total human disturbance (scored 0-4)	1.49	0	0.45	2.83	0.38	0.23	0.17	0.19	0.67	0.29	0	1.52	0.97
Mean substrate diameter (mm)	0.05	0.01	0.35	0.01	17.8	0.48	0.01	2.54	0.05	3.11	1.72	0.35	39.03

Appendix C continued.

	CC	GC	JR	BP	BC	HC	MT	CYP	LBH	WHC	MC	CR	QT
No. residual pools >75 cm deep	1	2	3	1	0	0	0	1	1	2	0	0	0
No. residual pools >100 cm deep	0	1	1	0	0	0	0	0	1	2	0	0	0
Maximum residual pool depth (cm)	77.7	100.1	122.7	79.1	68.7	59.2	67.9	79.6	133.1		50.3	22.5	39.3
Mean residual pool depth (m^2/100 m)	17.41	29.06	16.25	20.50	15.92	9.11	12.47	15.16	34.81		5.00	3.23	4.64
St. dev. of substrate diameter	74.6	1.0	3.4	3.8	28.9	5.9	380.9	24.1	6.8	54.0	13.3	1.0	103.9
% fine substrates (silt/clay)	1.0	73.3	2.9	96.2	0	0	23.8	0	38.1	5.7	0	0	1.0
% concrete substrate	0	0	0	0	0	0	0	0	0	0	0	0	2.9
% sand substrate	62.5	0	90.5	0	33.3	81.0	4.8	63.8	38.1	57.1	62.9	100	41.0
%e hardpan clay substrate	24.0	0	1.0	0	1.0	2.9	68.6	9.5	0	17.1	3.8	0	41.9
% fine, sand, and gravel substrate (<16 mm diameter)	65.4	73.3	94.3	96.2	35.2	81.0	28.6	64.8	76.2	66.7	65.7	100	53.3
% coarse gravel/rock substrate (>16 mm diameter)	1.9	0	0	1.0	63.8	0	0	21.0	0	12.4	22.9	0	4.8
%boulder/bedrock substrate (>250 mm diameter)	0	0	0	1.0	16.2	0	0	0	0	0	0	0	0
% wood/detrital substrate	8.7	26.7	4.8	2.9	0	16.2	2.9	3.8	23.8	3.8	7.6	0	0
LWD volume (no./100 m)	6.4	7.7	6.0	2.7	10.1	64.2	3.4	28.2	15.3	5.9	4.0	0.1	0.3

APPENDIX D

BENTHIC MACROINVERTEBRATE METRICS FROM 13 WADEABLE STREAM REACHES IN MISSISSIPPI. SAMPLES COLLECTED DURING SUMMER 2004. CC = COLES CREEK, GR = GREEN CREEK, JR = JOURDAN RIVER, BP = BOGUE PHALIA, BC = BRUSHY CREEK, HC = HORSE CREEK, MT = MAGEE TRIBUTARY, CYP = CYPRESS CREEK, LBH = LITTLE BOGUE HOMO, WHC = WEST HOBOLOCHITTO CREEK, MC = MIDDLETONCREEK, CR = COLDWATERRIVER, QT =QUITMAN TRIBUTARY

Metric	CC	GC	JR	BP	BC	HC	MT	CYP	LBH	MC	CR	QT
Total individuals	395	412	379	271	96	281	235	344	203	268	429	396
Richness	31	56	44	35	26	49	32	48	25	39	31	37
Simpson dominance	0.17	0.06	0.09	0.09	0.15	0.06	0.14	0.09	0.18	0.09	0.28	0.16
Shannon-Weiner (H`)	2.31	3.26	2.96	2.77	2.50	3.31	2.59	2.97	2.20	2.89	1.87	2.41
Hilsenhoff's index	4.92	4.57	3.93	6.50	3.71	5.17	5.52	4.60	3.86	3.53	3.96	4.97
% Chironomidae	93.4	45.9	54.4	78.2	53.1	49.5	82.6	61.9	20.7	51.5	92.3	85.1
% Ephemeroptera	2.0	1.9	12.1	7.4	12.5	20.0	0	2.0	5.4	3.4	0.9	5.6
% Plecoptera	0	0.2	0	0	18.8	0.4	0	0.3	0	0	0	0
%Trichoptera	0.5	8.5	13.7	5.9	7.3	2.8	3.8	4.4	27.1	0	4.4	1.5
EPT richness	5	10	13	6	9	12	1	12	6	4	3	5
% Non-insects	2.5	13.1	0.79	3.3	1.0	17.8	5.1	6.7	10.8	1.9	0	1.3
%Mollusca	0.3	12.4	0	1.1	0	0	2.6	0.3	9.4	0	0	0
% Odonata	0.8	3.2	0.8	1.1	2.1	2.1	1.7	1.5	1.0	0.4	0.9	0.5
%Coleoptera	0.5	17.0	16.4	3.3	5.2	2.8	5.5	17.4	33.5	9.7	0.5	5.1
%Oligochaeta	0	0	0.3	1.5	0	1.8	1.3	0.3	0	0	0	0.8

Appendix D continued.

Metric	CC	GC	JR	BP	BC	HC	MT	CYP	LBH	MC	CR	QT
% Pollution intolerant	21.5	28.6	39.6	12.5	27.1	16.4	11.5	25.0	33.5	43.3	36.6	31.1
% Facultative-tolerant	64.8	52.9	49.6	37.6	70.8	53.0	65.6	67.2	57.6	45.9	60.6	41.9
% Pollution tolerant	10.9	17.5	10.3	48.3	2.1	28.5	22.1	7.6	8.9	3.7	2.3	26.0
%Clingers	27.1	44.4	57.3	26.2	72.3	29.9	56.6	26.5	82.3	38.1	66.2	36.9
% Burrowers	34.4	35.4	19.5	49.4	11.5	32.4	5.5	38.7	14.3	38.8	22.1	41.2
% Climbers	32.7	8.3	14.8	17.7	33.3	10.0	10.2	13.1	14.3	16.0	21.0	16.2
%Swimmers	0.5	0.7	3.7	0	3.1	7.1	1.3	9.6	0	7.5	0.7	0.8
%Sprawlers	9.1	18.7	14.8	22.5	8.3	18.5	32.3	20.1	3.4	7.5	10.5	8.3
%Collector-filterers	47.6	25.5	27.4	24.4	17.7	3.9	42.1	9.0	36.0	19.4	52.0	38.4
% Collector-gatherers	65.1	34.2	29.0	39.8	17.7	44.8	12.8	45.6	11.8	25.4	18.9	56.6
% Predators	5.3	30.8	12.4	15.1	27.1	21.4	31.5	31.7	4.9	13.4	8.9	2.3
%Scrapers	0.8	4.4	20.3	3.3	9.4	11.7	5.5	9.6	38.4	9.7	1.6	4.8
%Shredders	6.1	12.6	16.9	31.7	34.4	11.7	8.1	27.6	14.3	31.7	21.0	7.8

APPENDIX E

CENTRARCHID CATCH CHARACTERISTICS FROM 13 WADEABLE STREAM REACHES IN MISSISSIPPI DURING SUMMER 2004-2005. COEFFICIENTS OF VARIATION (CV) ARE IN PARENTHESES

Catch characteristics	Coles	Green	Jourdan	Bogue Phalia	Brushy	Horse
CPUE (no. fish/ angler-hour):						
Mean total CPUE	1.67 (59.2)	1.06 (32.7)	2.50 (20.3)	0.17 (173.2)	1.12 (0.56)	1.09 (100.3)
Mean LMB CPUE	0.72 (67.2)	0.25 (173.2)	0	0	0.23 (0.19)	0.46 (69.7)
Mean PB CPUE	0	0.06 (173.2)	1.50 (8.8)	0	0.62 (0.70)	0
Mean LE CPUE	0.56 (114.9)	0.16 (173.2)	0.25 (35.9)	0	0.22 (0.57)	0.51 (141.4)
Mean BG CPUE	0.28 (124.4)	0.17 (173.2)	0.33 (51.5)	0	0	0
Mean total bass CPUE (LMB + PB)	0.72 (67.2)	0.46 (173.2)	1.50 (8.8)	0	1.08 (0.72)	1.09 (100.3)
Mean total sunfish CPUE (all *Lepomis*)	0.95 (118.5)	0.60 (58.1)	0.75 (22.1)	0	0.05 (0.57)	0.12 (0.50)
Percentage composition:						
percentage largemouth bass	63.6	21.4	0	0	36.4	50.0
percentage spotted bass	0	7.1	63.6	0	36.4	0
percentage longear sunfish	18.2	14.3	9.1	0	18.2	37.5
percentage bluegill sunfish	18.2	21.4	27.3	0	0	0

LMB = largemouth bass, PB = spotted bass, LE = longear sunfish, BG = bluegill.

Appendix E continued.

Catch characteristics	Magee trib.	Cypress	L. Bogue Homo	W. Hobolochitto	Middleton	Coldwater	Quitman trib.
<u>CPUE (no. fish/ angler-hour):</u>							
Mean total CPUE	0.27 (86.6)	0.61 (35.1)	1.94 (22.7)	0.72 (57.4)	0	0	0
Mean LMB CPUE	0.10 (173.2)	0.47 (131.5)	0.44 (99.0)	0.22 (86.6)	0	0	0
Mean PB CPUE	0	0.11 (173.2)	0.17 (173.2)	0	0	0	0
Mean LE CPUE	0.17 (173.2)	0.11 (173.2)	0.17 (173.2)	0.33 (113.8)	0	0	0
Mean BG CPUE	0	0	0.08 (173.2)	0.11 (86.6)	0	0	0
Mean total bass CPUE (LMB+ PB)	0.27 (173.2)	0.69 (137.2)	0.78 (0.54)	0.55 (86.6)	0	0	0
Mean total sunfish CPUE (all *Lepomis* spp.)	0.17 (173.2)	0.22 (99.0)	1.17 (1.20)	0.17 (153.4)	0	0	0
<u>Catch composition:</u>							
percentage largemouth bass	33.3	60.0	20.8	22.2	0	0	0
percentage spotted bass	0	10.0	4.2	0	0	0	0
percentage longear sunfish	66.7	10.0	16.7	44.4	0	0	0
percentage bluegill sunfish	0	0	5.8	22.2	0	0	0

LMB = largemouth bass, PB = spotted bass, LE = longear sunfish, BG = bluegill.