POPULATION AND CONSERVATION BIOLOGY OF THE THREATENED LEOPARD DARTER

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CHAPTER I

INTRODUCTION

The leopard darter Percina pantherina is a threatened percid endemic to five streams in the Little River drainage in southeastern Oklahoma and southwestern Arkansas. Before 1977, only 165 specimens had been collected from four streams in the drainage (Robison 1978). Between 1977 and 1988, a total of 1210 leopard darters were collected during studies conducted in the entire drainage (Jones and Maughan 1984, James et al. 1991, Zale et al. 1994). It was apparent that abundances were higher than previously believed, but much of the increase may have reflected improvements in sampling and a better understanding of leopard darter habitat use. However, the overall abundance, survival, and distribution of leopard darters in the drainages was still poorly understood prior to this study (see chapters II and III).

After obtaining data on overall abundances and distribution of leopard darters, we can begin to examine how life history and other (e.g., abiotic and biotic) factors interact to influence patterns in abundance. Previous work with the species described habitat use (Jones 1981, James 1989), spawning behavior and habitat (James et al. 1989), growth (Jones et al. 1983, James et al. 1991), and diet (James et al. 1991). This study adds to previous population biology information and extends it by examining how human and environmental impacts affect the abundance, survival, and habitats of the species.

The first challenge in determining overall abundance of leopard darters was to develop a method of extrapolating population abundance samples to unsampled areas. A confounding factor was that not all habitat in the streams is capable of supporting leopard darters (James and Collins 1993). We developed a method for using maps of mesohabitat and previous data on leopard darter habitat preferences (James 1989) to classify the quality of mesohabitat in two streams (Chapter II). We found that mesohabitat structure differed among streams and apparently influenced overall abundance (Chapter III). We also looked at how road crossings (Chapter IV) and land use (Chapter V) could influence leopard darter populations.

The objectives of the study were to: (1) determine abundance of leopard darters in Big Eagle Creek and West Fork Glover River, Oklahoma, (2) examine factors that influence variation in abundance of leopard darters among streams and from year to year, (3) examine swimming performance of the leopard darter in relation to culverts in road crossings in the Glover River, and (4) examine how land use influences sedimentation in the streams and affects the quality of leopard darter habitat.

CHAPTER II

USING GEOGRAPHIC INFORMATION SYSTEMS TO DETERMINE ABUNDANCE OF A STREAM-DWELLING FISH

Abstract: We present a method for using geographic information systems (GIS) to map and classify mesohabitat and estimate the abundance of stream fishes. Stream mesohabitat types are classified in terms of quality for a given species, and these quality categories are used to stratify population sampling. Population samples and mesohabitat quality then are used with the GIS to predict abundances in areas that were not sampled and calculate an abundance estimate for an entire stream. We present the basic method and illustrate its use with a case study from Big Eagle Creek in southeastern Oklahoma. Our method is flexible and accounts for mesohabitat variation in quality and its influence on fish abundance. We consider our method a first step in the utilization of GIS technology for population estimation and in quantitatively sampling at scales that have been difficult to examine in the past.

Introduction

Fisheries biologists are increasingly required to work at larger spatial scales as many natural resource and land management agencies move toward an ecosystem approach to managing fisheries resources (NRC 1993; U.S.D.A. Forest Service 1993). In the past, emphasis has been placed on sampling at the microhabitat to stream-reach level, and a variety of methods have been developed to characterize streams at these scales (Hudson et al. 1992; Meador et al. 1993; O'Neill and Abrahams 1984; Orth and Maughan 1982). Techniques for sampling at larger scales are under-developed (Lewis et al. 1996). Such techniques are necessary to begin to understand processes at the landscape level.

When sampling at larger scales, adequate sampling designs are needed to obtain precise estimates of stream characteristics or fish population parameters and to relate these estimates to unsampled areas. Fishery surveys often are conducted at fixed sites (Johnson and Nielsen 1983; King et al. 1981; Wilde and Fisher 1996) that are chosen subjectively to be representative of the entire system or to maximize the catch of a targeted species. The major drawback of sampling at fixed sites is that they may not be representative of conditions in the system (Wilde and Fisher 1996). Data collected from such sites, therefore, are applicable only to those sites and cannot reliably be extrapolated to describe the rest of the system. Improvements in estimates can be made with a two-stage sampling design. In two-stage sampling, a stream is first divided into sections, and then fish abundance is estimated within randomly selected sections (Hankin and Reeves 1988). Population estimates from the subsample of stream sections are then extrapolated to the entire stream. With this method the variance associated with extrapolating to unsampled areas is much greater than the variance from estimating fish abundances at sampling locations (Hankin and Reeves 1988). Although random selection of sampling locations in the two-stage sampling method reduces problems that occur with sampling at fixed sites, the method does not account for variation in habitat quality within sections or its effect on fish abundances. Methods that identify strata on the basis of individual units of mesohabitat (i.e., pools and riffles) can be used to further reduce variance due to extrapolation.

Implementing a probability sampling design (e.g., random, stratified, systematic, or cluster) can be difficult in stream environments because of factors such as cost, time, and access. Balancing the difficulty of sampling large areas and the need for an appropriate sampling design has been problematic for many projects. An approach using fish-habitat relationships to define habitat classes that serve as strata in fish abundance sampling (i.e., Hankin 1986; Peterson and Rabeni 1995; Wilde and Fisher 1996) can provide this balance. There is a clear need for methods that enable assessment of available habitat, stratify habitat on the basis of predicted usage or quality, and employ a probability sampling design. This is essentially a spatial problem that lends itself well to Geographic Information Systems (GIS) technology.

During our study of the abundance and population dynamics of a federally threatened species, the leopard darter *Percina pantherina*, we needed a method to estimate abundances at unsampled areas, suspecting that not all available habitat was capable of supporting similar abundances of leopard darters. Using GIS, we developed a stratified random sampling design that accounted for both logistical concerns (access) and spatial concerns (amount of preferred habitat). In this paper, we present a general method for using GIS to estimate abundance of a fish in an entire stream. We illustrate this method with a case study of the leopard darter in Big Eagle Creek in southeastern Oklahoma. Finally, we offer suggestions for modifications and applications of this method for other fisheries and aquatic scientists.

The Method

Habitat Classification and Mapping

Our method consists of classifying and mapping stream mesohabitats, quantifying the suitability of these mesohabitats to stream fishes, and sampling fish populations and estimating their abundance over an entire stream or stream reach (Figure 1). A map of mesohabitat types is generated, and transect samples of habitat characteristics are used to create mesohabitat strata. The strata are used in selecting fish sampling sites, generating density functions, and developing an overall abundance estimate for the stream. In our description of the method, mesohabitat "type" refers to the classification of a section of stream into categories of pools, riffles, runs, etc., whereas mesohabitat "unit" refers to a single instance of a particular mesohabitat type.

During the initial mesohabitat mapping and quantification stage of the method (Figure 1), a base map depicting stream boundaries is created for the entire stream or stream reach, and field surveys are conducted to locate, map, identify, and quantify mesohabitat types in the stream. Stream boundaries can be digitized from aerial photographs, although spatial distortions caused by relief displacement, position within the photograph, camera tilt (Lillesand and Kiefer 1994), and shadowing from riparian vegetation will need to be corrected during later ground-truthing surveys. Spatial distortions can be partly corrected during the spatial registration process using ground control points identified on the photographs and on 1:24,000 topographic quadrangle maps (Lillesand and Kiefer 1994). Alternatively, 1:24,000 topographic quadrangle maps may be used as source material for digitizing boundaries if the river is large. As more orthophoto quads become available, the problem of spatial distortion will be negated since orthophotos do not contain scale, tilt and relief distortions (Lillesand and Kiefer 1994). Increases in satellite imagery resolution also may reduce future dependence on photographs.

Once stream boundary maps are prepared, mesohabitat types in the stream are identified and classified using a habitat evaluation system. Several systems can be used to identify different types of mesohabitat during field surveys. For example, Bisson et al. (1982) and McCain et al. (1990) developed similar mesohabitat classification systems to describe fundamental pool/riffle forms based on differences in streambed topography, low water-surface slope, degree of surface turbulence, position of scouring, and location relative to the main channel. Hawkins et al. (1993) proposed a similar hierarchical system with fewer categories. Both Bisson et al. (1982) and McCain et al. (1990), however, provide diagrams and text descriptions for their mesohabitat types.

Habitat Quantification and Suitability Classification

In the second stage of the method (Figure 1), mesohabitat types are classified according to suitability for the fish species of interest. Mesohabitat classification systems rely on visual estimates; thus, classification differences are likely to arise if there are multiple observers or if observers have little training and are required to classify mesohabitat into a large number of similar categories (Roper and Scarnecchia 1995). In addition, descriptions of mesohabitat types developed for some stream systems may not transfer easily to other systems (e.g., boulder-bedrock versus alluvial streams). Because of the difficulty in transferring stream-classification systems and disagreements over classification of certain mesohabitat types (e.g., runs/glides), quantification of mesohabitat types (O'Neill and Abrahams 1984; Rabeni and Jacobson 1993) may be necessary. Quantitative measurements of microhabitat characteristics within mesohabitat types can be used to verify visual designations of mesohabitat types and also can be compared with known fish preferences to define quality of mesohabitat types for a given species.

Transect sampling in mesohabitat units is used to generate values for habitat variables for which a species' preference is known. We recommend pre-determining the total number of mesohabitat units to be sampled to facilitate the logistics of sampling. After estimating this total number, all rare mesohabitats (i.e., < 5-7 examples in the entire stream) should be sampled, and remaining effort should be divided proportionally among the rest of the mesohabitat units. Depending on length of the study area and accessibility, the area may need to be divided into segments and sampled proportionally within segments.

To determine transect spacing, we recommend the method developed by Simonson et al. (1994). In streams with a mean stream width (MSW) <5 m, transects are placed three MSWs apart. Transects are placed two MSWs apart in streams that have a MSW ≥5 m (Simonson et al. 1994). The authors determined that habitat characteristics measured at 2-3 MSW intervals were within 5% of true values 95% of the time (Simonson et al. 1994). Although the method was developed for sampling stations that may include several different types of mesohabitat, we feel that it is appropriate for determining transect placement within single units of mesohabitat.

To categorize quality of mesohabitat types, we recommend use of a method of classification developed by

Thomas and Bovee (1993). Frequency distributions for each microhabitat variable are created from original data of habitats used by a species or by using previously obtained data. Habitat conditions in the central 50% of the frequency distribution are classified as optimal. Conditions in the central 95% are classified as suitable with the differences between optimal and suitable ranges representing habitat that is usable but of a lower quality than the optimum range. Remaining habitat conditions are classified as unsuitable (Thomas and Bovee 1993). A composite suitability is created with all individual habitat suitabilities. If all habitat conditions are optimal, the location is classified as optimal. If any of the individual habitat suitabilities are classified as usable, the composite suitability at that location is usable. Habitat suitability is classified as unsuitable if any individual habitat conditions are unsuitable (Thomas and Bovee 1993).

After classification of habitat quality at transect sampling points, a subjective scale is developed in our method to classify the suitability of each mesohabitat type based on quantity and quality of all points sampled within each type. For example, if ≥50% of all points sampled within a mesohabitat type are classified as optimal or usable, we designate that mesohabitat type as optimal. If 30-49% of the points in a mesohabitat type are optimal or usable, the type is classified as suitable. Mesohabitat types with poorer quality transect points (<30% optimal or usable) are classified as unsuitable. Using GIS, each mesohabitat unit in the database is reclassed into one of the three suitability classes and a resultant map is generated (Figure 2).

Fish Population Sampling and Estimation

The final stage is to use mesohabitat suitability classes as strata for fish population abundance sampling (Figure 1). Fish sampling is stratified by mesohabitat suitability and, if previously defined, by stream segment. Samples should be randomly selected from all mesohabitat units in each strata.

Any method to estimate fish abundance (e.g., mark-recapture, depletion, transect) may be used. The entire area of a sampling location should be measured following the population sampling in order to convert abundance to density. Although the area of each mesohabitat unit is available in the GIS, logistical concerns may limit fish sampling to a subsection of an individual mesohabitat unit.

After estimates of population abundance are generated, an optional step to account for longitudinal trends in fish abundance may be necessary. Longitudinal trends in abundance may arise from differences in habitat complexity (Schlosser 1982, 1987), geomorphological barriers (Gilliam et al. 1993), biotic interactions (Baltz et al. 1982; Gilliam et al. 1993), and physicochemical characteristics (Baltz et al. 1987). If there is a longitudinal trend, a mathematical function can be generated to account for changes in abundance within mesohabitat suitability classes with respect to longitudinal position.

Finally, the classified maps of mesohabitat are used in conjunction with population estimates to predict abundances of fish within unsampled units of mesohabitat. If there is no longitudinal trend in fish abundance, we recommend that mean estimates for each mesohabitat suitability class be applied to the unsampled units. In situations where an abundance function is created, mesohabitat data generated by the GIS can be input into the predictive model.

Densities predicted for each mesohabitat unit are then used with area of each unit of mesohabitat in the GIS to calculate the predicted abundance of fish within each unit. The GIS can be used to calculate an abundance estimate for the entire stream, or the data can be exported to a statistical software package to calculate abundance and a confidence interval.

Case Study: Leopard Darter Abundance in Big Eagle Creek, Oklahoma

Study Area and Species

Big Eagle Creek originates in the Ouachita Mountains in southern LeFlore County, Oklahoma, and flows SSE for 31 km to its confluence with the Mountain Fork River. The drainage basin, covering an area of approximately 240 km^2, is composed largely of sandstone and shale sedimentary rocks. Most of the land surrounding Big Eagle Creek is heavily forested, and silviculture and poultry farming are the major landuse activities. The upper reaches of the creek consist of shallow, short scour pools and riffles. Farther downstream, habitat shifts to deeper, longer midchannel pools and riffles. Substrata are primarily cobble, boulder, and bedrock, although smaller-sized substrata are present in isolated locations. Because of our inability to access the upper 10 km of the stream and the likelihood that leopard darters are absent in this area (Zale et al. 1994), we sampled the lower 21 km of continuous mainstem channel.

The leopard darter is a percid endemic to five streams of the Little River drainage in southeastern Oklahoma and southwestern Arkansas. Its apparent rarity (Cloutman and Olmsted 1974, Robison et al. 1974) led to its designation as a threatened species (U.S. Fish and Wildlife Service 1978), although several recent studies (James et al. 1991, Zale et al. 1994, Toepfer et al. 1996) have found much greater abundances than previously estimated (Jones et al. 1984). Because of impoundments and unsuitable habitat in some areas,leopard darters are confined to the middle and upper reaches of the larger streams in the drainage (James and Collins 1993).

Habitat Classification and Mapping

Initial base maps depicting the edges of Big Eagle Creek were digitized from tracings of 1:7920 aerial photographs obtained from the Natural Resources Conservation Service. During float trips, we observed 10 mesohabitat types (Table 1) using the classification scheme by McCain et al. (1990). Our initial selection of transect sampling sites was based primarily on logistics; we pre-determined the amount of habitat that could be sampled within 4-5 days. Big Eagle Creek was divided into four segments based on access, and mesohabitat types were quantified by segment. We attempted to sample all occurences of rare mesohabitat types (occurring <5 times in the entire stream). For uncommon mesohabitats (occurring <5 times in each segment), one randomly chosen example was sampled from each segment. Remaining effort was divided proportionally by the frequency of common mesohabitat types and by stream segment.

The location of the first transect in each unit of mesohabitat was subjectively placed within 10 m of the upstream end of the unit, and in most cases a minimum of three transects was sampled. When a mesohabitat unit was too short for three transects we used two transects and two additional randomly located sampling points. A total of ten subjectively located points were used to classify habitat at each of two narrow cascades created by concrete dams. Water depth, current velocity (at 0.6 depth), and substrata were measured at four equally-spaced points along each transect or at each extra sampling point. A Modified Wentworth particle size scale (Bovee and Cochnauer 1977) was used to characterize substrate in a 1-m^2 area around each sampling point by assigning a number for each 25% coverage of the area. A weighted mean was constructed for each sampling point by multiplying the dominant substratum score by four, the secondary substratum by three, tertiary by two, and quaternary by one, and dividing the quantity by 10. We needed four days to sample 23% of all available mesohabitat units in Big Eagle Creek.

Habitat Quantification and Suitability Classification

The next step in classifying habitat suitability was to translate the data collected during transect sampling into suitability classes for all mesohabitat types (Table 2, Fig. 2). Habitat suitability criteria were derived from frequency distributions of depth, velocity, and substrate (James 1989) at points where individual leopard darters were first observed. We had difficulties using the three suitability classes of Thomas and Bovee (1993) because there was an apparent longitudinal gradient in quality within mesohabitat types. For example, although midchannel pools in the headwaters had more high-quality habitat than riffles in the same area, they had considerably less high-quality habitat than midchannel pools farther downstream. Because of these differences, none of the mesohabitat types were classified as optimal after using our initial suitability classification methods. Even after using samples from 23% of the mesohabitat units we were unable to factor out the longitudinal effects on habitat quality. Thus, we reduced the three levels of suitability classes by Thomas and Bovee (1993) to two, preferred and non-preferred. In our application of the method, preferred habitat corresponded to the suitable category (central 95%) of Thomas and Bovee (1993), and non-preferred habitat corresponded to their unsuitable category. Preferred water depth was 25 to 90 cm, current velocity was 0 to 28 cm/s, and substrata were gravel, cobble, and boulder. We applied these criteria to all sampling points, and, using Boolean logic, derived a preference rating for each point. That is, if all three habitat variables (depth, velocity and substrate) at a sampling point were preferred, the habitat at that point was classified as preferred. If any of the variables were determined to be not preferred, the habitat was classified as non-preferred (after Thomas and Bovee 1993). After deriving a suitability rating for each sampling point, we re-classified mesohabitat types based on the frequency of preferred and non-preferred points taken in that type (Table 2).

Fish Population Sampling and Estimation

The suitability classification of mesohabitat types was used to stratify mark-recapture sampling locations for leopard darters. Based on previous sampling, abundance estimates of this rare fish were difficult to obtain, so we biased our sampling toward optimal mesohabitats to ensure that at least a few estimates of abundance would be determined. We were able to access the two upstream-most segments in the stream, but low summer flow and poor accessibility limited our sampling to the upper and lower third of the two downstream-most stream segments. During each mark-recapture trip, we sampled three mesohabitat units. If we were unable to sample a site because of low water or poor visibility, the nearest location from the same suitability category was chosen. Larger mesohabitat units (> 750 m^2) could not be completely sampled so we chose an area that could be effectively sampled by two people.

We used mark-recapture to estimate population size. In the marking run, two snorkelers with small, hand-held dip nets each searched the entire area at least 3-4 times and captured leopard darters. In areas that contained leopard darters, we searched until no additional fish were sighted for a period of thirty minutes. All captured leopard darters were anesthetized with MS-222, marked with an injectable elastomer material (Northwest Marine Technology, Inc.) and released after they recovered equilibrium. Recapture runs were made for two days following the initial tagging. During the recapture runs all unmarked darters were marked and released. Schnabel abundance estimates (Krebs 1989) were determined for each sampling location, and abundances were converted to densities based on the area sampled.

The abundance estimates for leopard darters showed a longitudinal trend with extremely low abundance in optimal and suitable mesohabitat in the headwaters. Abundance increased in an exponential manner in downstream samples with a greater rate of increase in optimal compared to suitable mesohabitat (Figure 3). We used nonlinear regression to create models to predict density based on longitudinal position of mesohabitat units in the stream. The distance of the center of each mesohabitat unit from the top of the stream was determined using GIS and was used in the regression models with the dependent variable, density. Our data showed an exponential trend, but we assumed that densities would reach an asymptote, resulting in a logistic curve. Initial values for slope and intercept of the logistic curve were obtained by solving the full equation in terms of those values and generating a linear regression. These values were then used as starting values to fit a model with nonlinear least squares regression (SAS Institute 1985). A total of 100 iterations with the multivariant secant method was used to generate the final models. Separate models were generated for optimal and suitable mesohabitat in Big Eagle Creek (Figure 3).

We obtained the area of each unit from the GIS and used these values with the predicted density values to generate a total abundance estimate for the entire stream. Since we were lacking an abundance estimate from suitable mesohabitat in the lower 5 km of Big Eagle Creek we constrained the predicted density to avoid extrapolating the regression line beyond our sample data. Seven of the 25 units of suitable mesohabitat were downstream from our last sampling location. Although we may have underestimated densities by constraining the density downstream, we feel that the error was negligible. Densities in optimal mesohabitat were about one order of magnitude higher than densities in suitable mesohabitat, and the total area of optimal mesohabitat was four times higher. Approximately 98.5% of the overall abundance estimate was composed of predicted abundances in optimal mesohabitat.

We used the statistical differential method (Kempthorne and Folks 1971) in which the first term of a Taylor expansion series is retained to derive a variance term for predicted density in each unit of mesohabitat. The variances were weighted by mesohabitat area and summed to calculate a 95% confidence-interval for the overall abundance estimate. The abundance estimate and 95%

confidence interval for Big Eagle Creek was 98,441 ± 3,293. In contrast, without consideration of mesohabitat differences an extrapolation of mean density would have resulted in an abundance estimate and 95% confidence-interval of 72,197 ± 75,231.

Conclusions

We believe our method is an improvement over previous methods used to estimate population abundance of stream fishes. Applying densities per unit stream length to an entire stream, or sampling in randomly selected segments (Hankin and Reeves 1988) does not allow for spatial variation in habitat quality or its effects on fish density. By collecting quantitative habitat data within individual mesohabitat units one can assess variability in habitat quality. If mesohabitat types then are reclassified into suitability strata based on habitat preferences of a species, the result is sampling units that are more likely to have a direct relationship with fish abundance. Also, by coupling longitudinal abundance functions with the spatial mesohabitat data in a GIS, the variance in abundance due to extrapolation can be reduced.

We acknowledge that problems associated with data collection, design, and analysis still exist in our assessment of habitat in Big Eagle Creek, Oklahoma. In our case study, we feel that both the baseline spatial data and our sampling methods were not adequate to fully characterize the habitat and habitat suitability in the stream. Our major difficulty was trying to make classification decisions at a mesohabitat scale based on quantitative data collected at a microhabitat scale. Preliminary fine-scale (1 m^2) sampling at six locations showed that high-quality habitat tended to occur in small patches that could easily be missed by transect points, resulting in an underestimate of the amount of high-quality habitat. Classifying mesohabitat within segments (i.e., upstream, midstream, downstream) might alleviate this problem, but our sample sizes precluded that type of data subsetting.

Our case study was also limited by small sample sizes in our population estimates. We were hindered by the life cycle and activity levels of the leopard darter. There is about a 3-4 month period from summer to early fall in which leopard darter recruits are large enough to safely handle and active enough for us to capture large numbers. Afterwards, they appear to hide in interstices of cobble and boulders and our capture efforts were largely unsuccessful. In addition, we are limited to capturing individuals with hand-held dip nets and require high visibility while snorkeling. The logistical constraints we faced may not occur in other studies. However, the method was quite useful for estimating fish abundance in Big Eagle Creek. It also was effective in reducing variation during abundance extrapolation and gave us a much clearer understanding of longitudinal abundance distribution of the leopard darter.

In our application, GIS allowed us to classify a large area of stream and develop a more accurate assessment of abundance patterns of leopard darters. Without mapping habitat we would have had to make assumptions about unsampled and possibly unseen areas. An additional advantage, and one of the most important benefits of GIS, is that once the data are collected they can be manipulated or queried for a variety of questions. For example, our existing database of habitat for Big Eagle Creek can now be used with species-specific habitat requirements to generate maps of habitat quality for other species of fish. The data also provide a good representation of the meso-scale of habitat and can be linked to other spatial scales. For instance, we are currently modeling impacts of land use in the watershed on leopard darter habitat quality.

We expect the use of GIS by fisheries and aquatic scientists to be more common in the future (Fisher and Toepfer, unpublished data). The technology and techniques used by geographers have advanced far beyond the applications that aquatic researchers have developed. The challenge is for researchers and managers to apply this geographic tool, coupled with spatial statistics and models, to improve our ability to conserve fisheries and aquatic resources at all spatial scales.

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Table 1. Mesohabitat category descriptions for Big Eagle Creek, Oklahoma (modified from McCain et al. 1990).

Category	Description
Low Gradient Riffle	Shallow, swift current; cobble and boulder substrate
High Gradient Riffle	Moderately deep, swift current; cobble and boulder substrate
Cascade	Extremely steep and swift; one or more waterfalls;
Secondary Channel Pool	Small pool at confluence of a tributary and the main channel;
Backwater Pool	Moderately deep; little to no current; gravel, sand or detritus
Lateral Scour Pool	One side deep; opposite side is shallower; cobble or boulder
Midchannel Pool	Deep in center; current typically slow; substrate varies from gravel to bedrock
Glide	Shallow pool with constant depth; low current with little turbulence; substrate varies
Run	Shallow reaches; varying depths; swift current with turbulence; cobble, boulder or bedrock
Step Run	Runs separated by short riffles; swift current; boulder or bedrock

Table 2. Classification of suitability of mesohabitat types based on percentage of transect points classified as preferred and non-preferred in Big Eagle Creek, Oklahoma.

Mesohabitat type	Total Number	Number Sampled	Percent Preferred	Percent Non-preferred	Suitability Class
Low Gradient Riffle	68	15	25	75	Unsuitable
High Gradient Riffle	26	5	28	72	Unsuitable
Cascade	7	3	0	100	Unsuitable
Secondary Channel Pool	1	1	58	42	Optimal
Backwater Pool	3	2	59	41	Optimal
Lateral Scour Pool	12	4	45	55	Suitable
Midchannel Pool	45	9	57	43	Optimal
Glide	13	4	73	27	Optimal
Run	32	5	30	70	Suitable
Step Run	1	0	0	0	NA

List of Figures

HABITAT CLASSIFICATION AND MAPPING

Prepare base map of stream from source materials
(e.g., aerial photographs, topographic maps, satellite imagery)

↓

Visually identify and classify mesohabitat types
in stream and digitize onto base map

↓

HABITAT QUANTIFICATION AND SUITABILITY CLASSIFICATION

Select mesohabitat units to quantify microhabitat characteristics
(use transect sampling and base transect spacing on MSW)

↓

Derive habitat suitability criteria for fish species and
separate into suitability classes (optimal, suitable, unsuitable)

↓

Apply suitability rating to microhabitat sample points
in mesohabitat units using Boolean logic

↓

Reclassify mesohabitat units into suitability classes
based on frequency of microhabitat suitability ratings

↓

FISH POPULATION SAMPLING AND ESTIMATION

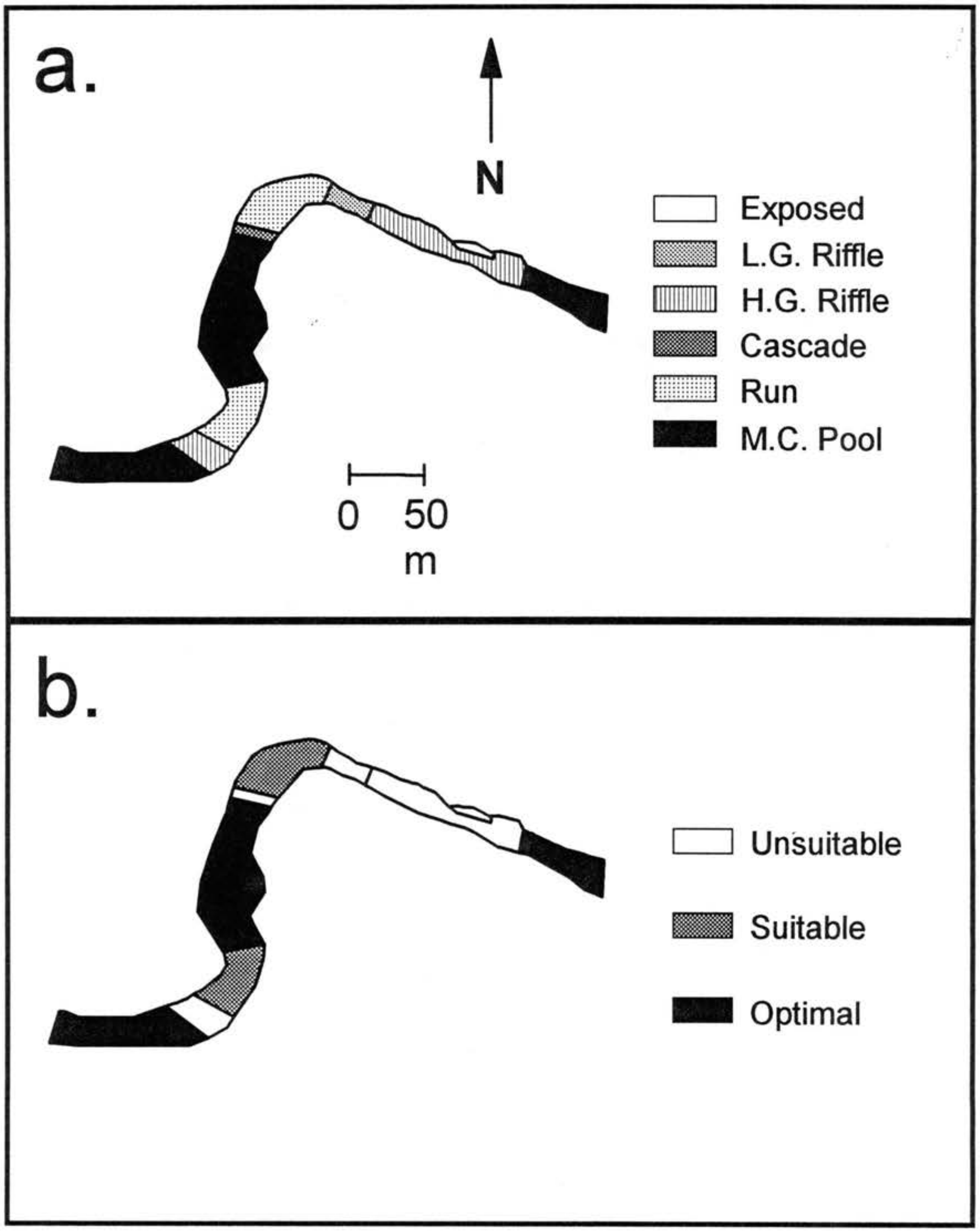
Stratify fish population sampling sites using mesohabitat suitability
ratings, and randomly select mesohabitat units to be sampled

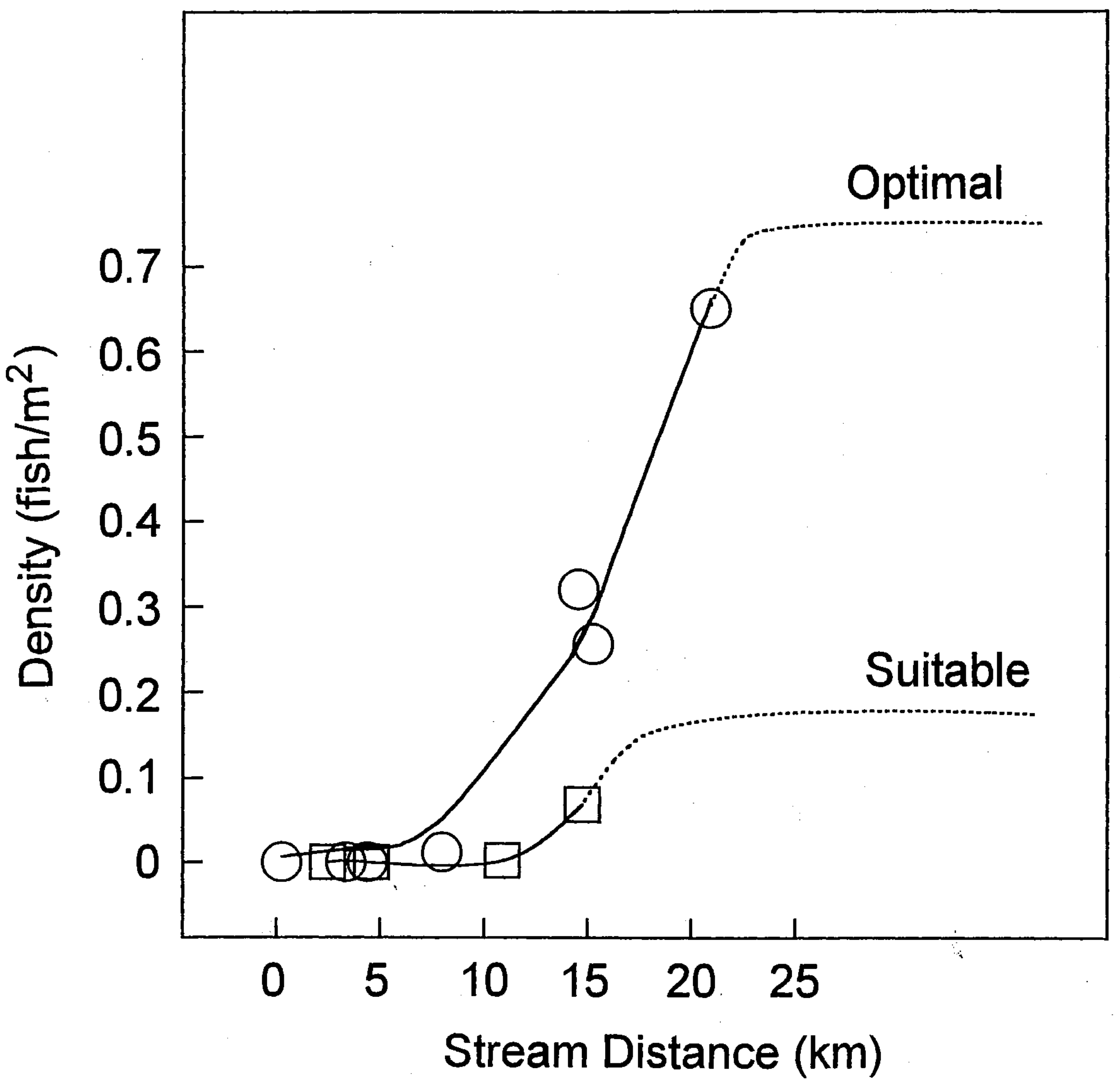
↓

Sample fish population (e.g., mark-recapture, depletion,
transect) and estimate population size

↓

Expand population estimate to stream using mesohabitat suitability
and, if present, longitudinal density function


a.
N
Exposed
L.G. Riffle
H.G. Riffle
Cascade
Run
M.C. Pool
0 50
m
b.
Unsuitable
Suitable
Optimal


Optimal
Suitable
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
Density (fish/m²)
0
5
10
15
20
25
Stream Distance (km)

CHAPTER III

FACTORS INFLUENCING ABUNDANCE OF THE THREATENED LEOPARD DARTER

The leopard darter *Percina pantherina* is a threatened species endemic to the Little River drainage in southeastern Oklahoma and southwestern Arkansas. Collections from 1977-1988 indicated that the species was more common than previously thought, but only two attempts were made to determine abundance of the species in an entire stream. We used maps of mesohabitat in a geographic information system to determine abundances of leopard darters in Big Eagle Creek and West Fork Glover River, Oklahoma, and Robinson Fork River, Arkansas. Among-stream differences in darter abundance reflect differences in habitat structure. Reduced abundances in 1996 apparently resulted from a drought and an associated delay in the spawning season.

The leopard darter *Percina pantherina* is a percid endemic to five streams of the Little River drainage in southeastern Oklahoma and southwestern Arkansas. Its apparent rarity (Cloutman and Olmsted, 1974; Robison et al., 1974) led to its designation as a threatened species (U.S. Fish and Wildlife Service, 1978). Although only 165 specimens had been collected prior to 1977, more recent studies found greater abundances. From 1977 to 1988, 1210 leopard darters were collected during surveys and population sampling at over 100 sites along the Saline, Cossatot, Rolling Fork, Mountain Fork, Robinson Fork, and upper Little rivers (Jones et al., 1984; James et al., 1991; Zale et al., 1994). Although the number of darters captured by Jones et al. (1984) and James et al. (1991) may have included multiple captures of several individuals, abundance of leopard darters appears greater than previously believed.

It became apparent during our recent studies, however, that abundances of leopard darters among three streams (Big Eagle Creek, West Fork Glover River, and Robinson Fork River) were considerably different. Geomorphological structure of the three streams differed, and we hypothesized that the interaction between geomorphology and darter distribution patterns likely influenced observed differences in abundance. Jones et al. (1984) showed that pools are preferred habitat, and subsequent studies (James et al., 1991; Zale et al., 1994) have confirmed this. Although most leopard darters occur in pools during late winter and early spring, they move into the lower part of riffles in mid-spring in association with spawning (James and Maughan, 1989). However, the size and distribution of pools and riffles differs among the three streams. Since the life cycle of leopard darters depends on both pools and riffles, a factor potentially influencing population abundances is the complementation, or spatial proximity, of the two habitat types to each other (Schlosser, 1995).

In addition to variation in abundance across drainages, densities of leopard darters at several of our sampling sites decreased dramatically from 1995 to 1996. The short lifespan of leopard darters likely influences inter-annual variations in abundance. Jones et al. (1984) indicated that leopard darters lived for a period of approximately three years. However, James et al. (1991) created monthly length-frequency histograms indicating a rapid growth rate and a high mortality rate for the species and concluded that maximum longevity was only about 18 months. Because of the short lifespan, individual leopard darters usually have only one reproductive season. Thus, reduced breeding success in any one year could have a marked effect on population size.

Our objective was to relate variation in leopard darter abundances to geomorphological characteristics and hydrologic conditions using life-history information. We examined both inter-stream and inter-annual variation in abundance.

MATERIALS AND METHODS

Study area.--Big Eagle Creek originates in the Ouachita Mountains in southern LeFlore County, Oklahoma, and flows SSE for 31 km to its confluence with the Mountain Fork River (Figure 1). West Fork Glover River originates in the Beaver's Bend Hills of the Ouachita Mountains in northern McCurtain County, Oklahoma and flows south for 33 km to its confluence with the mainstem of the Glover River (Figure 1). Robinson Fork River arises in McCurtain County, Oklahoma and flows SSE for approximately 32 km to its confluence with the Rolling Fork River at DeQueen Reservoir in Sevier County, Arkansas. During 1994 to 1996 we sampled mesohabitat and leopard darter abundances in approximately 21 km of Big Eagle Creek and West Fork Glover River and about 13 km of Robinson Fork River.

Rivers in the Ouachita mountains are composed largely of sandstone, shale, and novaculite sedimentary rocks (Thornbury, 1965). Most of the land surrounding the streams is heavily forested, and silviculture and poultry farming are the major landuse activities. Upper reaches of Big Eagle Creek consist of shallow, short, scour-pools and riffles. Farther downstream, mesohabitat shifts to deeper, longer, midchannel pools and riffles. West Fork Glover River consists mostly of long, deep, midchannel pools and narrow riffles. The portion of Robinson Fork River habitable by leopard darters is approximately half the size of the other two streams. Habitat in this river ranges from short pools and high gradient riffles in the headwaters to long midchannel pools and runs in the tailwaters. Periodic flash flooding scours the channels of all three of these streams, resulting in primarily cobble, boulder, and bedrock substrata, although smaller substrata are present in isolated locations.

Fish abundance and survival.--Preliminary surveys of Big Eagle Creek and West Fork Glover River indicated that a large amount of the available mesohabitat was potentially usable by leopard darters. We developed a sampling technique using Geographic Information Systems (GIS) to map and classify mesohabitat in terms of its suitability for leopard darters.

Stream boundaries were digitized from 1:7920 aerial photographs. Type and location of mesohabitat units (e.g., midchannel pool, low gradient riffle) were classified and indicated on plots of the stream boundaries during float trips. A stratified random design was used to choose sampling locations for measuring water depth, current velocity, and substrata at four points along each of several transects. Spacing of transects was determined by mean stream width. In units of mesohabitat with a mean stream width (MSW) <5 m, transects were placed three MSWs apart. Transects were placed two MSWs apart in units of mesohabitat with a MSW of ≥5 m (Simonson et al., 1994). We compared data from the transect samples with habitat preferences of leopard darters (James, 1989) and classified mesohabitat types in terms of quality for the species (see Chapter II). We then used the mesohabitat quality classes in a stratified random design for selection of abundance sampling sites. We conducted three-day mark-recapture runs and generated Schnabel abundance estimates (Krebs, 1989) at each sampling location. A longitudinal pattern in leopard darter abundance was apparent in Big Eagle Creek, so nonlinear functions were created to predict densities at unsampled areas (see Chapter II). No longitudinal pattern was detected in West Fork Glover River, so a single Schnabel estimate was generated by pooling mark-recapture data from all population sampling sites. Predicted density estimates in unsampled areas were used with mesohabitat areas in the GIS to calculate abundance estimates for each entire stream. A confidence interval for abundance in Big Eagle Creek was constructed using the statistical differential method (Kempthorne and Folks, 1971) in which the first term of a Taylor expansion series is retained to derive a variance term. Because there was no function describing a longitudinal abundance trend in West Fork Glover, we used the confidence interval for the density calculated from the pooled Schnabel estimate to generate a confidence interval for overall abundance.

Unlike Big Eagle Creek and West Fork Glover River, much of the available mesohabitat in Robinson Fork River was unsuitable for leopard darters. In addition, we were unable to float this river because of its smaller size and lower flow. Therefore, we modified our sampling method to map and classify mesohabitat quality. A preliminary survey for presence of leopard darters was conducted to delimit the distribution of the species within Robinson Fork River. Stream boundary maps also were digitized from 1:7920 aerial photographs. We hiked the entire portion of the river occupied by leopard darters and indicated mesohabitat types on maps of the stream boundary. While mapping mesohabitat we used a systematic sampling design at 500-m increments to quantify water depth, current velocity, and substrata in specific mesohabitat types. Similar to our methods in the other two streams, we compared habitat data with known leopard darter preferences to classify quality of mesohabitat types. We randomly selected population sampling sites from historical locations and additional areas that were accessible. Density estimates obtained from pooled Schnabel estimates were used with the mesohabitat maps to predict densities in unsampled areas and generate an abundance estimate for the entire stream. A confidence interval for overall abundance was computed by using the confidence interval for the pooled Schnabel estimate.

Annual survival rates in Big Eagle Creek and West Fork Glover River were estimated with length-frequency histograms. Based on breaks in the histograms and previous work (James, 1989) we assumed two age classes: age 0 (<60 mm standard length) and age 1+ (≥60 mm SL). Given the limited lifespan of the leopard darter (James et al., 1991), we considered survival to spring spawning to be especially critical. We used data from Big Eagle Creek to examine survival from September 1994 (post-recruitment) to February 1995 (pre-spawning).

Stream habitat, geomorphology, and hydrology.--To describe geomorphological structure in all three streams, we measured mesohabitat diversity and dominance (Turner, 1989) and quantified the degree of habitat complementation (Schlosser, 1995). Mesohabitat maps created for the population sampling were used to determine the diversity and dominance indices. We excluded the headwaters of all streams because of inaccessibility and a lack of suitable leopard darter habitat in these areas. We calculated the mesohabitat indices for each entire stream and also for four equally-sized segments in each stream.

Habitat complementation, or the proximity of two required habitat types (Schlosser, 1995), was calculated using GIS. The mesohabitat maps were converted using GRASS4.0 software into 3-m resolution cells and were reclassified into two additional maps, one consisting of all areas of mesohabitat capable of supporting leopard darters, and one consisting of all riffles. We created concentric buffer zones in 3-m increments outward from each riffle to generate a new map indicating the distance of every non-riffle cell to the nearest riffle cell. We then generated a table indicating the areas within each 3-m distance class. These areas were determined for entire streams and separately for individual units of usable mesohabitat within each stream. We summarized the resulting distance-gradient table by combining the 3-m distance classes into 75-m classes.

Mean daily discharge data were collected from a USGS gaging station on the Mountain Fork River near Smithville, Oklahoma. Although the gaging station was not located on any of our three study streams, it was within 2 km of lower Big Eagle Creek. We used these data as a measure of flow trends over the three-year study period.

RESULTS

Fish abundance and survival.--Using the mesohabitat maps and the nonlinear density functions, we calculated an abundance estimate of 98,441 leopard darters (95% CI; ±3,293) for Big Eagle Creek in 1995; this river was not sampled in 1996. For West Fork Glover River, we were able to sample only five optimal mesohabitat units in 1995, and our abundance estimate for that year was 56,530 fish (±40,223). In 1996, we sampled 14 mesohabitat units in West Fork Glover River and generated an abundance estimate of 32,614 fish (±44,572). For Robinson Fork River, we generated an estimate of 4,848 (±1,478) for 1995; in 1996, we were unable to find enough individuals to make an estimate of population abundance. Reduced abundance in 1996 was reflected in reduced density at five of the seven locations sampled in both 1995 and 1996 in West Fork Glover and Robinson Fork Rivers (Figure 2). Although there was not a significant decrease in density between years (Wilcoxon signed ranks test, _P_ = 0.128), there was a decrease of 60-100% at five of the seven sites.

Analyses of length-frequency histograms from 1994 and 1995 in Big Eagle Creek (Figure 3)indicated an annual survival rate of 7%. The annual survival rate in West Fork Glover River from 1995 to 1996 was slightly higher at 12%. The survival rate from early fall 1994 (post-recruitment) to February 1995 (pre-spawning) in Big Eagle Creek was approximately 55% (Figure 4). We were unable to capture enough individuals in Robinson Fork River to generate the length-frequency histograms required for estimates of survival rates.

Stream habitat,geomorphology, and hydrology.--We determined from our mesohabitat maps that Big Eagle Creek contained about 606,700 m^2 of mesohabitat that was usable by leopard darters. West Fork Glover River contained about 622,978 m^2 of usable mesohabitat, and Robinson Fork River contained only about 221,780 m^2. Big Eagle Creek had ten different mesohabitat types, resulting in the highest mesohabitat diversity (H = 1.512) of the three streams (Table 1). Robinson Fork River, with nine different types of mesohabitat, had the second highest diversity, while West Fork Glover River, with eight mesohabitat types, had the lowest mesohabitat diversity. Dominance was highest in West Fork Glover River, intermediate in Robinson Fork, and lowest in Big Eagle Creek. All three streams showed some dominance by midchannel pools, a mesohabitat that encompassed 53-72% of the total area of the streams. There also were longitudinal changes in diversity among the streams (Table 1). In Big Eagle Creek, diversity of mesohabitat decreased from 2.157 in the upstream-most quarter (Segment 1) of the stream to 0.872 in the downstream-most quarter (Segment 4). There was also a general decrease in diversity in West Fork Glover River. Robinson Fork River showed the reverse trend with mesohabitat diversity increasing from 1.105 to 1.393 from the upstream end to the downstream end. As diversity decreased downstream in Big Eagle Creek, dominance of midchannel pools increased. However, dominance decreased downstream in Robinson Fork River. There was less of a longitudinal gradient in mesohabitat dominance in West Fork Glover River, which had two extremely long pools (about 1 and 1.5 km long) that led to higher dominance in the mid-reaches of the stream.

The distribution of predicted fish abundances also reflected the patterns of mesohabitat diversity and dominance in each stream. The highest densities of leopard darters were found in the downstream portion of Big Eagle Creek (see Chapter II), corresponding with increased mesohabitat quality in downstream areas. These factors resulted in a strong longitudinal gradient in leopard darter abundance with few fish upstream and high abundance downstream. Abundances of leopard darters within segments of the West Fork Glover failed to show any longitudinal trend. The lack of strong trends in mesohabitat diversity and dominance in this stream suggested that there were few differences in habitat along the entire length of the stream. In contrast to the previous two streams, the population of leopard darters in Robinson Fork appeared highly fragmented. A small group of darters inhabited a single pool in the top quarter of the stream while the remaining individuals inhabited two pools in the lower quarter of the stream.

In quantifying mesohabitat complementation, about 80% of the high-quality mesohabitat in Big Eagle Creek was ≤150 m from a riffle (Figure 5). However, only about 60% of high-quality mesohabitat in the other two streams was ≤150 m from a riffle.

Mean daily flow indicated that streams in the area tend to be flashy with frequent large changes in discharge (Figure 6). In 1994 and 1995, there were several peaks in discharge during most of the year with slightly lower flow during periods of 1994. In fall 1995, however, a drought occurred in the area and discharge was generally < 1 m^3/s. In early spring 1996, there were larger discharge events, but none approached discharges observed in previous years.

DISCUSSION

Our estimates of leopard darter abundance in Big Eagle Creek and West Fork Glover River were much higher than previous estimates. Estimates for the entire Glover River ranged from 3,000 to 10,000 (James, 1996) compared to our estimates of 32,000-56,000 for a single tributary. No other attempts have been made for estimating abundances for entire streams, but densities have been estimated at selected study sites in past studies. From fall 1978 to summer 1980, Jones et al. (1984) searched for leopard darters at fourteen locations in the Glover River and estimated densities ranging from 0 to 0.017 darters/m^2. James et al. (1991) sampled leopard darters at six sites in the Glover River from summer 1985 to fall 1988 and found densities ranging from 0.001 to 0.065 darters/m^2. Our densities ranged from 0 to 0.65 darters/m^2 in Big Eagle Creek and 0 to 0.21 darters/m^2 in West Fork Glover River. It had been suggested that the most abundant populations of leopard darters were in the Glover River (Zale et al., 1994). We, however, found higher densities and overall abundance (Table 1) in Big Eagle Creek than in West Fork Glover River. In contrast to Big Eagle Creek and West Fork Glover River, our density estimates in Robinson Fork River tended to be much lower, ranging from 0 to 0.058 darters/m^2.

We believe that our larger estimates of abundance in Big Eagle Creek or West Fork Glover River than those of previous workers were the result of improved sampling methods. Jones et al. (1984) captured fish with electroshocking and supplementary seining, both of which may be inefficient in the deep pools with cobble boulder substrata that are common in the Little River system. The underwater capture techniques used by James et al. (1991) and in our study appeared to be a more efficient means of capture. In addition, the abundance estimates by James et al. (1991) were one-day depletion estimates. Based on our work at a long-term movement/migration study site, we found that the daily activity levels of leopard darters vary widely. The three-day mark-recapture method we used may have provided a more accurate estimate of population abundances. However, observed differences in abundance may also have been a function of inter-annual variation and spatial variation between studies completed over a fifteen year period.

Amount of usable mesohabitat was similar between Big Eagle Creek and West Fork Glover, but in 1995, the latter stream had 33% fewer leopard darters. In that same year, Robinson Fork River with about one-third the amount of usable mesohabitat as the other two streams had only 5-8% as many leopard darters. These differences in abundance among drainages likely reflect differences in habitat structure. For example, although Big Eagle Creek and West Fork Glover had similar quantities of usable mesohabitat, the characteristics and distribution of individual units of mesohabitat differed.

Because the maximum values for mesohabitat diversity and dominance (Turner, 1989) were a function of the total number of mesohabitat types present, it was difficult to directly compare the three streams or segments within streams. In Big Eagle Creek, which had the highest densities of leopard darters, midchannel pools were the dominant mesohabitat in all four segments and showed a marked downstream trend in increased importance. Pools in the upper part of the drainage were smaller with swifter currents, and had lower densities of leopard darters. West Fork Glover River showed a similar but less pronounced trend toward increased dominance of midchannel pools. Pools in West Fork Glover tended to be considerably deeper (Appendices 4 and 5) and longer than those in Big Eagle Creek. Robinson Fork River, which had the lowest densities of leopard darters, was also dominated by midchannel pools, but there was no downstream trend toward increased dominance by this mesohabitat type. Quality of mesohabitat types in Robinson Fork varied more than in the other two streams, however. Unlike most areas in the other two streams, the majority of pool habitat in Robinson Fork had substrata composed of angular slabs of bedrock with little or no cobble or smaller rock material. Such areas are unsuitable for leopard darters (James, 1989). In comparison, most pools in the other two streams contained large areas of cobble and boulder substrata.

As noted by Dunning et al. (1992) and Schlosser (1995), the degree of mesohabitat complementation, which was highest in Big Eagle Creek, can affect population abundance. Leopard darters require pools for year-round habitat and riffles for spawning (James and Maughan, 1989). Areas of high-quality mesohabitat (pools and glides) in Big Eagle Creek were relatively small and closely associated with riffles (high complementation), a factor that might explain the high densities of leopard darters in this stream. In West Fork Glover River, where densities were somewhat lower, areas of high-quality mesohabitat often were very long pools. Darters in the center of these pools would be able to traverse high-quality areas the entire distance to the riffle. In Robinson Fork River, where densities were lowest, areas of high-quality mesohabitat often were separated from riffles by hundreds of meters of unsuitable habitat (slabs of bedrock). Since leopard darters migrate from pools to riffles (James, 1989), the reduction of mesohabitat complementation in West Fork Glover and Robinson Fork rivers may account for their lower population abundances.

We observed inter-annual fluctuations in abundance in West Fork Glover and Robinson Fork rivers. From 1995 to 1996, our abundance estimate for West Fork Glover River declined 42% (Table 1) and densities at three of four sampling sites declined 60-80%. In Robinson Fork River, we were unable to find enough fish to even estimate overall abundance in 1996, and density at two sampling sites declined 100%. The low survival rate of leopard darters and the limitation of a single spawning season makes the species susceptible to large fluctuations in abundance. We attribute the decline in abundance in 1996 to the patterns of discharge in the preceding months (Figure 6). Although high flow events, such as those observed during the spawning season (late February to early May) in 1994 and 1995 (Figure 6), could possibly disrupt leopard darter reproduction, this likely does not occur because high flow events often affect summer spawners more than spring spawners (Schlosser, 1985). In contrast, during the fall and winter of 1995 flow was considerably reduced, and a high-flow event did not occur until six weeks after the start of the normal spawning season. Correspondingly, standard lengths of young-of-year individuals in July 1996 were about 10 mm smaller than in previous years.

A combination of stressed adults and delayed spawning might have affected recruitment in 1996. In late May, we found that 14 of 22 adult leopard darters in a sampling site in West Fork Glover River had obvious fungal infections. Most individuals had fungal growth at the base of the anal and dorsal fins, while the worst infections included dense growths on their gills. The fungal infections may have resulted from stress associated with poor water quality.

The leopard darter occurs in only five streams in the Little River drainage (Miller and Robison, 1973). Although the streams appear to be similar, there are subtle differences in habitat structure that likely influence population abundances in each stream. Because leopard darters have essentially only one reproductive season, populations can fluctuate markedly in response to disturbances such as the drought that occurred in 1995-1996. In addition, more information is needed on how land use activities affect stream habitat and water quality. Although the species is much more abundant than previously believed, elements of its population biology may make it especially susceptible to natural and anthropogenic impacts.

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Table 1. Mesohabitat diversity and dominance and leopard darter abundances for each stream and for four equally-sized segments within each stream.

Stream	Mesohabitat Diversity	Mesohabitat Dominance	Darter Abundance
Big Eagle Creek	1.512	0.791	98,441
Segment 1	2.157	0.040	870
Segment 2	1.629	0.450	3,493
Segment 3	1.188	0.758	17,055
Segment 4	0.872	0.791	77,014
Robinson Fork	1.370	0.827	4,848
Segment 1	1.105	0.841	342
Segment 2	1.310	0.769	0
Segment 3	1.483	0.714	0
Segment 4	1.393	0.553	4,506
West Fork Glover	0.918	1.160	47,321
Segment 1	1.214	0.578	9,125
Segment 2	1.014	0.932	12,531
Segment 3	0.670	0.716	13,413
Segment 4	0.814	0.572	12,252

LIST OF FIGURES

Figure 1. Little River drainage in southeastern Oklahoma and southwestern Arkansas. WF indicates the West Fork Glover River, BE indicates Big Eagle Creek, and RF indicates Robinson Fork River.

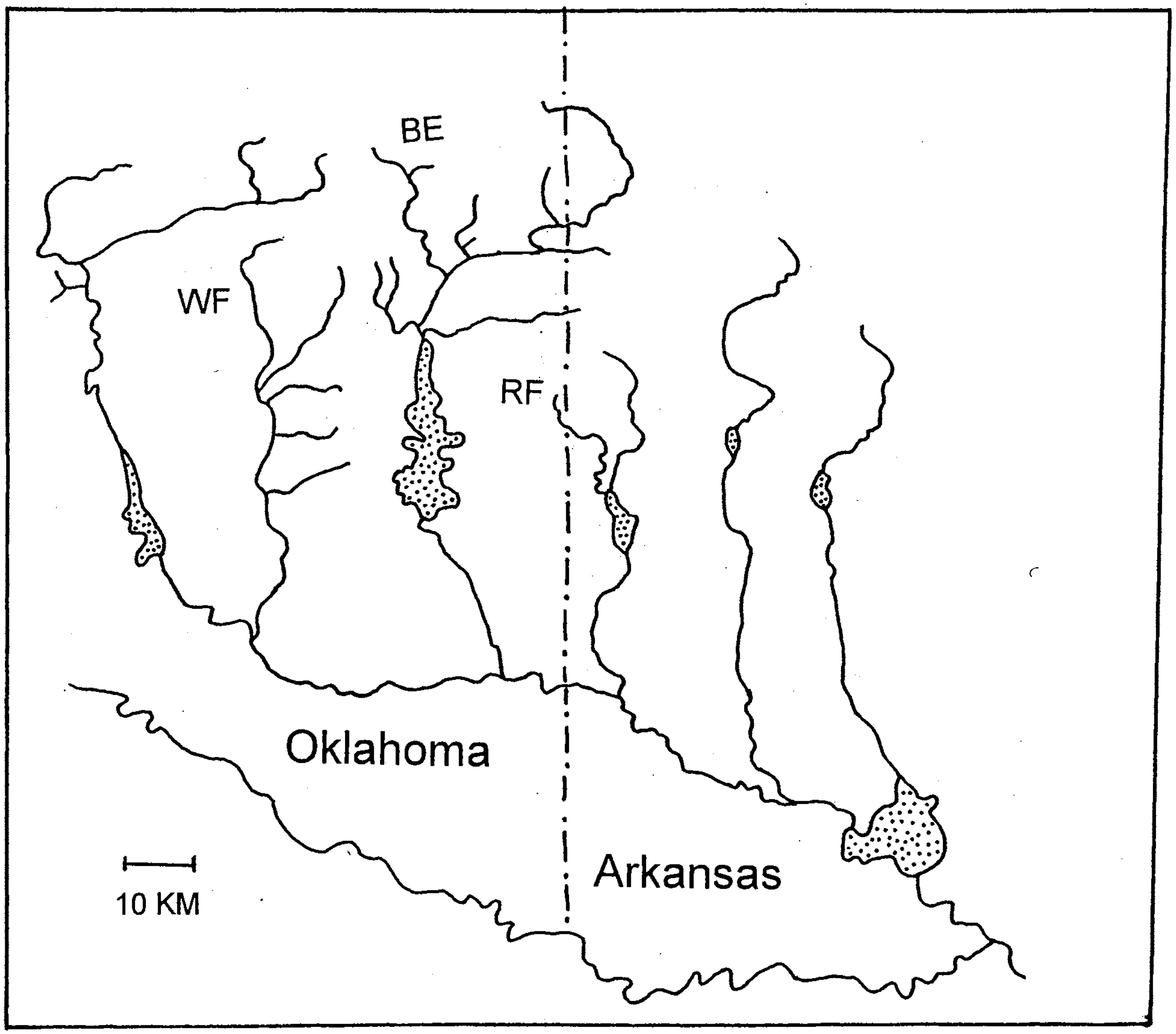
Figure 2. Inter-annual abundance in density of leopard darters. WF indicates sites in West Fork Glover River and RF indicates sites in Robinson Fork River.

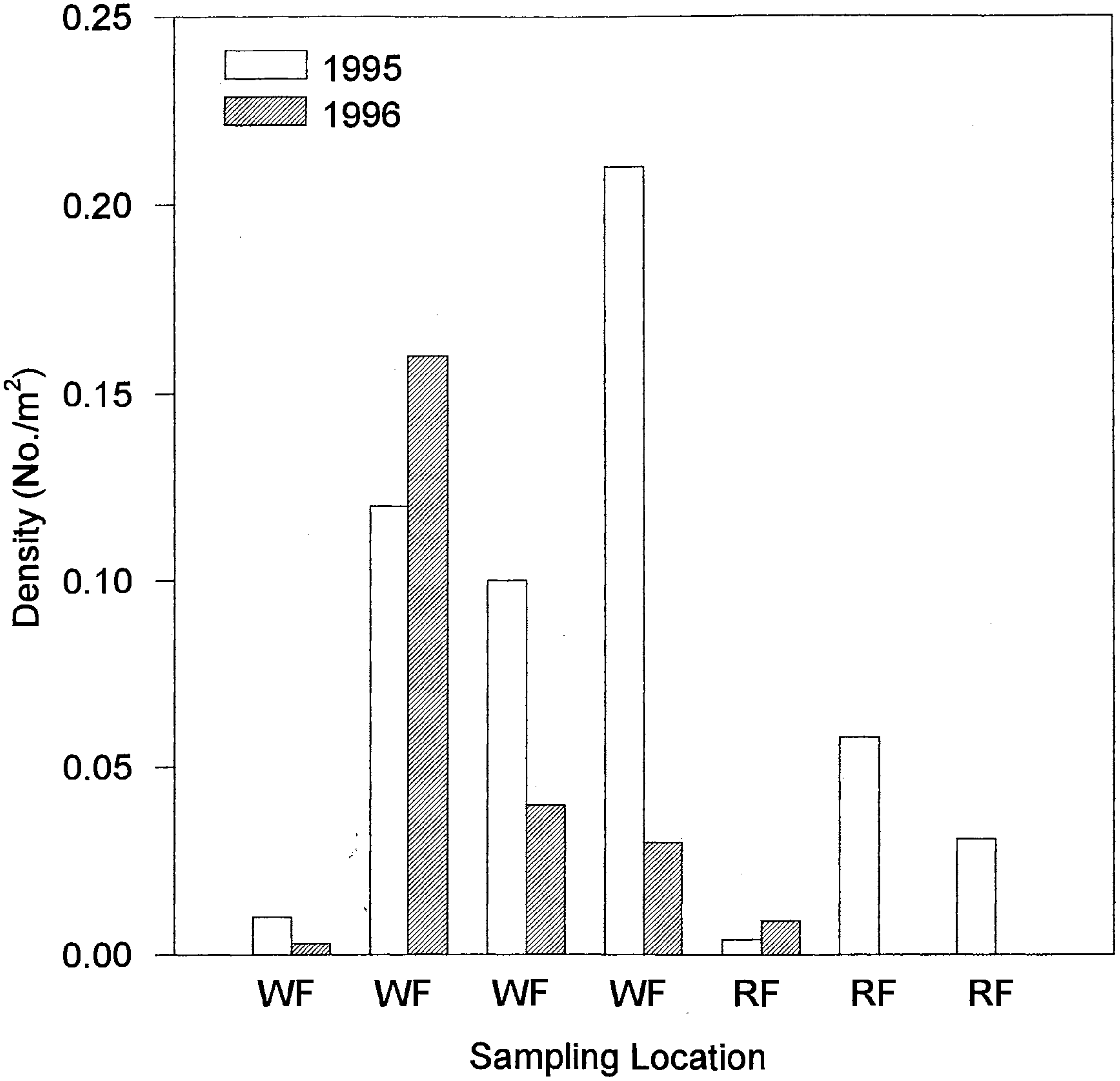
Figure 3. Length-frequency histograms for annual survival of leopard darters in Big Eagle Creek and West Fork Glover River.

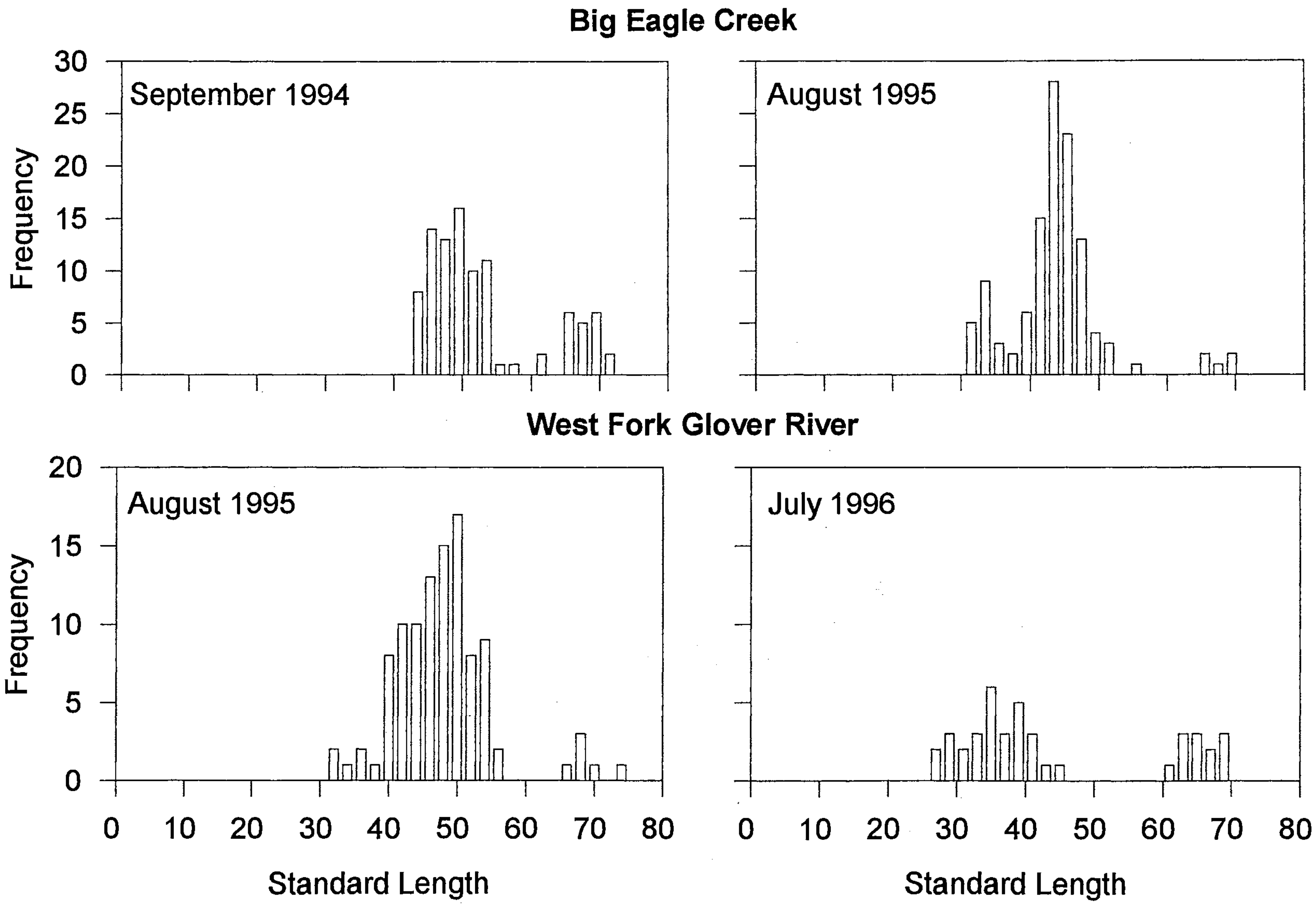
Figure 4. Length-frequency histograms for post-recruitment to pre-spawning survival.

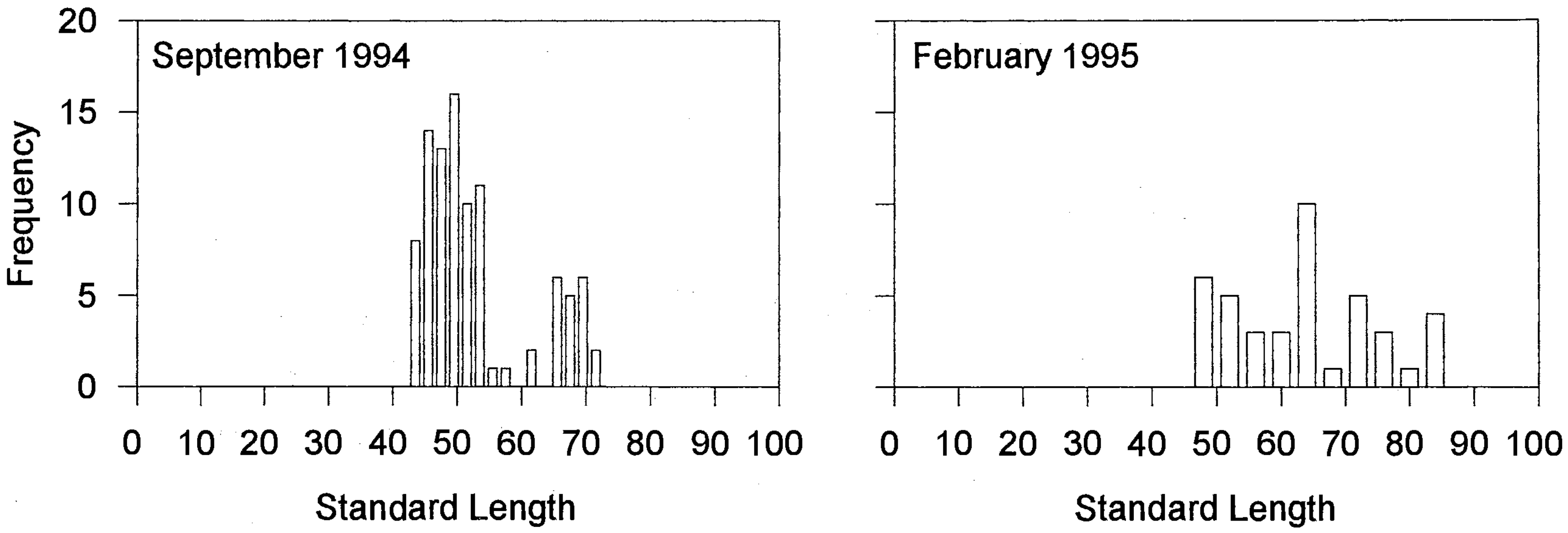
Figure 5. Habitat complementation measured as percent of usable habitat within 75 m distance classes from the nearest riffle.

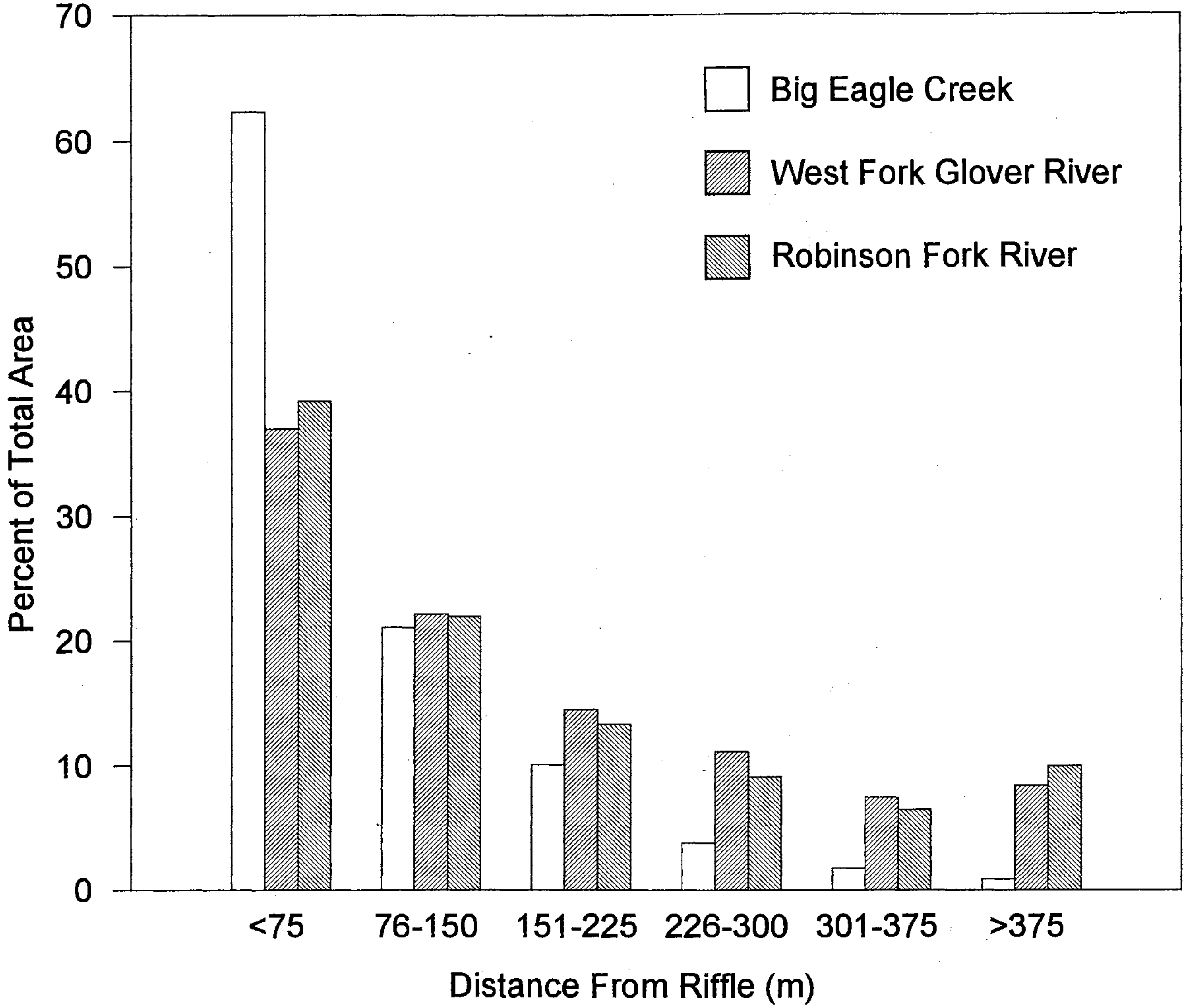
Figure 6. Mean daily discharge for a three-year period in the Mountain Fork River near Smithville, OK.

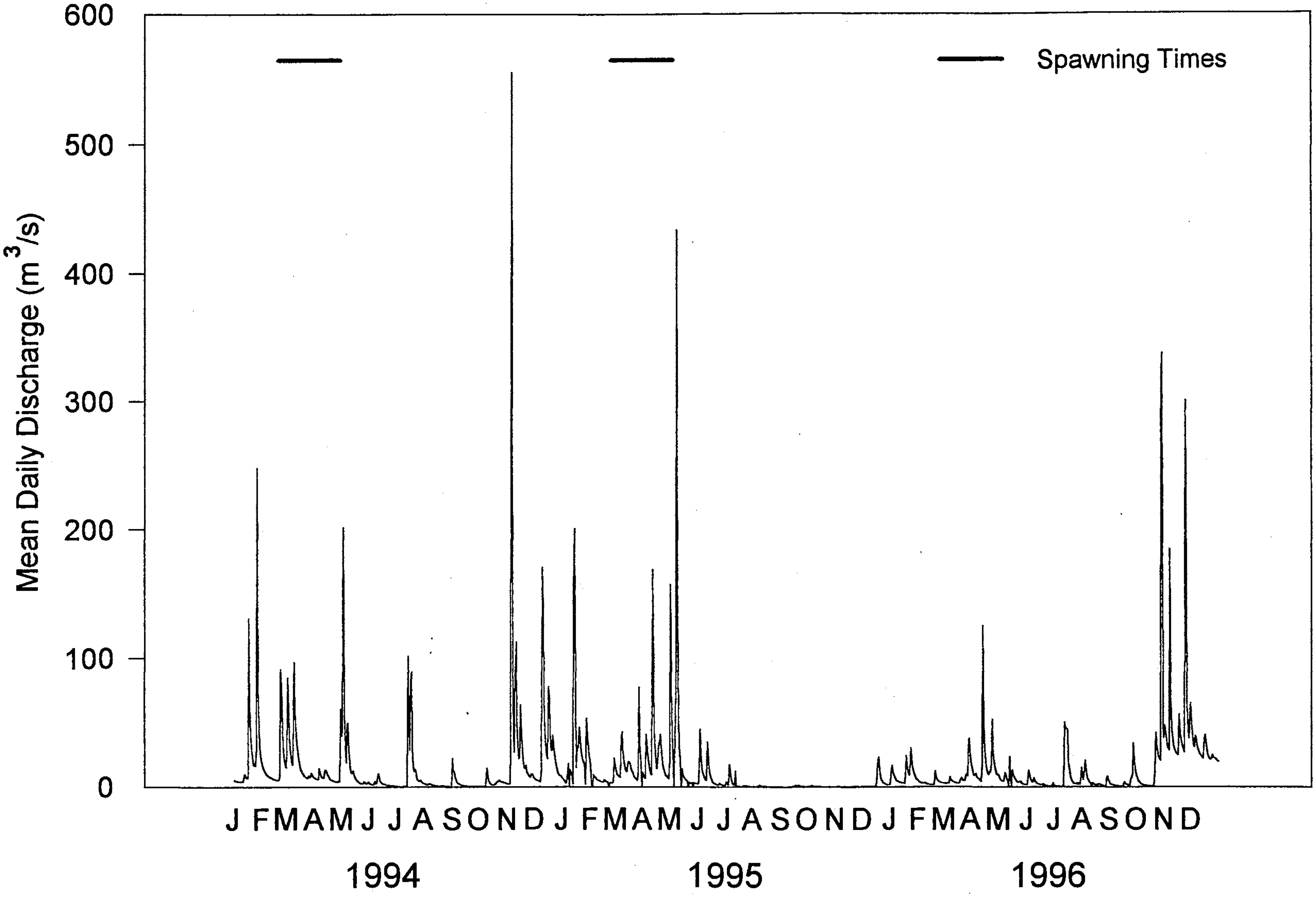

BE
WF
RF
Oklahoma
Arkansas
10 KM


1995
1996
Density (No./m^2)
0.25
0.20
0.15
0.10
0.05
0.00
WF
WF
WF
WF
RF
RF
RF
Sampling Location


Big Eagle Creek
September 1994
August 1995
Frequency
30
25
20
15
10
5
0
West Fork Glover River
August 1995
July 1996
Frequency
20
15
10
5
0
0
10
20
30
40
50
60
70
80
Standard Length
Standard Length


September 1994
February 1995
Frequency
20
15
10
5
0
0
10
20
30
40
50
60
70
80
90
100
Standard Length
Standard Length


Big Eagle Creek
West Fork Glover River
Robinson Fork River
Percent of Total Area
0
10
20
30
40
50
60
70
<75
76-150
151-225
226-300
301-375
>375
Distance From Riffle (m)


Mean Daily Discharge (m^3/s)
600
500
400
300
200
100
0
Spawning Times
J F M A M J J A S O N D J F M A M J J A S O N D J F M A M J J A S O N D
1994
1995
1996

Appendix 1. Mark-recapture results for Big Eagle Creek in 1995.

Mesohabitat Unit	Legal Description	Starting Date	Day 1			Day 2			Day 3		
			C	M	R	C	M	R	C	M	R
Unsuitable Mesohabitat											
HGR 3-2	T1N R25E Sec. 31	6/27/95	0	0	0	0	0	0	0	0	0
LGR 3-15	T1S R25E Sec. 4	8/23/95	0	0	0	0	0	0	0	0	0
LGR 4-3	T1S R25E Sec. 9	7/18/95	0	0	0	0	0	0	0	0	0
Suitable Mesohabitat											
RUN 1-4	T1N R25E Sec. 17	7/18/95	0	0	0	0	0	0	0	0	0
LSP 2-2	T1N R25E Sec. 27	7/11/95	0	0	0	0	0	0	0	0	0
RUN 3-7	T1S R25E Sec. 5	6/27/95	1	1	0	1	1	0	0	0	0
LSP 4-1	T1S R25E Sec. 9	8/23/95	3	2	0	4	4	0	10	0	2
Optimal Mesohabitat											
MCP 1-1	T1N R25E Sec. 17	8/23/95	0	0	0	0	0	0	0	0	0
MCP 2-2	T1N R25E Sec. 20	7/11/95	0	0	0	0	0	0	0	0	0
MCP 2-4	T1N R25E Sec. 28	7/11/95	0	0	0	0	0	0	0	0	0
MCP 3-1	T1N R25E Sec. 30	6/27/95	6	0	0	3	0	0	2	0	0
MCP 3-16	T1S R25E Sec. 4	8/23/95	11	10	0	23	9	1	17	0	4
MCP 4-2	T1S R25E Sec. 9	7/18/95	22	22	0	19	17	2	25	0	11
MCP 4-16	T1S R25E Sec. 22	8/15/95	17	17	0	28	22	6	35	0	7

Appendix 2. Mark-recapture results for West Fork Glover River in 1995.

Mesohabitat Unit	Legal Description	Starting Date	Day 1			Day 2			Day 3		
			C	M	R	C	M	R	C	M	R
MCP 1-1	T2S R23E Sec. 6	8/08/95	2	2	0	3	1	0	2	0	1
MCP 1-2	T2S R23E Sec. 6	8/08/95	5	5	0	12	9	3	7	0	1
MCP 1-6	T2S R23E Sec. 7	8/09/95	3	3	0	5	2	2	4	0	1
MCP 2-7	T2S R23E Sec. 20	8/29/95	10	9	0	17	11	5	18	0	6
MCP 3-20	T3S R23E Sec. 7	8/29/95	11	11	0	15	14	1	16	0	4

Appendix 3. Mark-recapture results for West Fork Glover River in 1996.

Mesohabitat Unit	Legal Description	Starting Date	Day 1			Day 2			Day 3		
			C	M	R	C	M	R	C	M	R
		Unsuitable Mesohabitat									
HGR 1-1	T2S R23E Sec. 6	7/01/96	0	0	0	0	0	0	0	0	0
HGR 3-1	T2S R23E Sec. 20	7/01/96	0	0	0	0	0	0	0	0	0
		Suitable Mesohabitat									
LGR 1-4	T2S R23E Sec. 6	7/01/96	0	0	0	0	0	0	0	0	0
LGR 2-2	T2S R23E Sec. 7	7/09/96	0	0	0	0	0	0	0	0	0
LGR 2-5	T2S R23E Sec. 18	7/09/96	0	0	0	0	0	0	0	0	0
LGR 3-1	T2S R23E Sec. 20	7/09/96	0	0	0	0	0	0	0	0	0
LGR 3-13	T2S R23E Sec. 32	7/09/96	0	0	0	0	0	0	0	0	0
		Optimal Mesohabitat									
MCP 1-2	T2S R23E Sec. 6	7/22/96	8	8	0	9	7	2	5	0	2
RUN 2-2	T2S R23E Sec. 18	7/09/96	2	2	0	0	0	0	0	0	0
MCP 2-1	T2S R23E Sec. 18	8/09/96	9	9	0	7	6	1	3	0	2
MCP 2-6	T2S R23E Sec. 20	7/22/96	3	3	0	5	0	0	3	0	0
MCP 3-10	T2S R23E Sec. 32	9/09/96	0	0	0	0	0	0	0	0	0
MCP 3-11	T2S R23E Sec. 32	9/09/96	1	1	0	2	2	0	0	0	0
MCP 3-20	T3S R23E Sec. 7	7/22/96	6	6	0	1	1	0	0	0	0

Appendix 4. Habitat characteristics in Big Eagle Creek. Values are means ± 1 SD.

Mesohabitat Type	Depth (cm)	Velocity (cm/s)	Substrate
	Unsuitable Mesohabitat		
Cascade	32.14 ± 26.70	67.96 ± 44.57	7.91 ± 0.30
Low Gradient Riffle	28.43 ± 12.95	35.32 ± 29.11	6.22 ± 0.32
High Gradient Riffle	29.98 ± 15.32	34.55 ± 27.90	6.40 ± 0.52
	Suitable Mesohabitat		
Lateral Scour Pool	66.09 ± 36.33	8.33 ± 10.57	6.69 ± 0.93
Run	48.07 ± 31.14	18.27 ± 19.99	7.06 ± 0.89
	Optimal Mesohabitat		
Backwater Pool	81.68 ± 34.99	0.95 ± 1.81	5.91 ± 0.12
Glide	53.88 ± 24.20	10.81 ± 9.48	6.41 ± 0.62
Midchannel Pool	69.58 ± 35.46	6.85 ± 8.15	6.51 ± 0.73
Secondary Channel Pool	38.50 ± 17.46	0.67 ± 1.23	5.49 ± 0.56

Appendix 5. Habitat characteristics in West Fork Glover River. Values are means ± 1 SD.

Mesohabitat Type	Depth (cm)	Velocity (cm/s)	Substrate
	Unsuitable Mesohabitat		
Backwater Pool	25.13 ± 8.29	0.00 ± 0.00	7.89 ± 0.27
High Gradient Riffle	30.00 ± 15.64	57.46 ± 40.89	6.55 ± 0.57
	Suitable Mesohabitat		
Low Gradient Riffle	34.79 ± 14.25	29.94 ± 26.66	6.46 ± 0.50
	Optimal Mesohabitat		
Glide	42.07 ± 15.51	15.84 ± 23.30	6.09 ± 0.36
Midchannel Pool	99.26 ± 49.33	2.56 ± 3.52	6.33 ± 0.97
Run	64.58 ± 32.89	10.21 ± 11.11	6.71 ± 0.52
Step Run	28.33 ± 14.45	30.25 ± 27.59	6.86 ± 0.63

CHAPTER IV

SWIMMING PERFORMANCE OF THE THREATENED LEOPARD DARTER IN RELATION TO ROAD CULVERTS

Abstract--We investigated the relationship between leopard darter swimming performance evaluated in the laboratory and current velocities measured at the ends of corrugated pipe and open-box culverts through road crossings. We tested eight darters at each of four current velocities, 0, 5, 12, and 25 cm/s and measured burst frequency, duration, and distance. We used ANOVA to analyze burst frequency and total distance covered during a ten-minute period and found that at a current velocity of 25 cm/s fish swam more frequently and for a greater distance than at lower velocities. When nested ANOVAs were used to remove individual variation, we found that mean burst distance, mean time swimming, and mean swimming speed also differed significantly, with fish in the highest velocity bursting greater distances at higher speeds for longer periods of time. Current velocities in box and pipe culverts tended

to be higher than 25 cm/s, and several crossings had structural barriers in addition to high current velocities. Although there is no evidence that culverts act as long-term barriers to migration or dispersal of leopard darters, they may prevent migratory activity during certain years, thereby negatively affecting localized populations.

Current velocity through poorly-designed culverts may act as a barrier to migration and dispersal of fishes (Baker and Votapka 1990; Clay 1995). For example, Derksen (1980) concluded from a mark-recapture study that water velocities in five culverts acted as a near-total barrier to migration of spring-spawning Arctic grayling *Thymallus arcticus*, northern pike *Esox lucius*, and longnose suckers *Catostomus catostomus*.

Although estimates of critical swimming speed or swimming performance of fishes (e.g., Brett 1967; Dorn et al. 1979; Berry and Pimentel 1985) have used experimental flumes or tunnels that resemble culverts, few studies have directly related swimming performance to culvert passage by fish. Jones et al. (1974) evaluated swimming speeds of 17 fish species in the Mackenzie River and generated curves to show the smallest size of each species that could be expected to traverse a 100-m culvert at a range of current velocities. In addition to potential current-velocity barriers, poorly designed culverts also may present barriers such as shallow water depths within the culvert, absence of refuge pools at either end, an hydraulic jump at the inlet, or a large drop from the outlet to the stream surface (Baker and Votapka 1990). All of these are considered barriers for large fishes but may have even greater affects on smaller fishes.

The leopard darter Percina pantherina is a small federally threatened percid (USFWS 1978) endemic to the Little River drainage of southeastern Oklahoma and southwestern Arkansas (Miller and Robison 1973). Abundance and distribution of the species may be limited by anthropogenic factors such as silviculture and associated road development, gravel removal, runoff from agriculture and poultry industries, and reservoirs (James and Collins 1993). For example, leopard darters historically inhabited the lower Mountain Fork and Cossatot rivers (Eley et al. 1975), but these populations were extirpated by construction of Broken Bow and Gillham reservoirs (James and Collins 1993). In addition, spawning habitat appears limited because not all riffles in the drainage contain gravel suitable for spawning (James 1989). These factors make the leopard darter potentially vulnerable to localized extirpation, and the presence of physical barriers such as culverts may have adverse effects on recolonization or spawning migrations.

Leopard darters inhabit pools during most of the year and spawn in riffle tailwaters from early March to mid-May (James and Maughan 1989). Individuals are usually found in water depths of 25-75 cm, over cobble and boulder substrata, and in areas with little to no current (James et al. 1991). Individuals typically are seen swimming 5-10 cm above the substrate and are rarely seen resting on the bottom (James et al. 1991; Toepfer, personal observation). Leopard darters spawn in areas with current velocity as high as 50 cm/s (James and Maughan 1989), but in such situations they usually are observed resting on gravel and cobble substrates and appear to have difficulty swimming in swift currents (James et al. 1991; Toepfer, personal observation). Swimming activity in swifter currents involves short movements directly on the surface of the substrate, and individuals that enter the water column are generally swept downstream (Toepfer, personal observation).

Recent concerns regarding fish passage (R. Standage, Ouachita National Forest and T. Melchiors, Weyerhaeuser Company) in the range of the leopard darter require an understanding of whether culverts at road crossings act as potential barriers. Our objective was to relate swimming abilities of leopard darters at different current velocities to those observed during the spawning season in culverts within the Glover River drainage of Oklahoma.

Methods

Laboratory study.--Thirty-two leopard darters (59.19 ± 6.12 mm standard length) were captured from Big Eagle Creek in southeastern Oklahoma during September 1996 and transported to Oklahoma State University in northcentral Oklahoma. The swimming performance apparatus (Figure 1) was a flow-through system modified from a design by Layher (1993) and was connected to a 3/4-hp pump with a 3.81-cm intake and outlet. Water for the experiment was recirculated through a Living Stream (Frigid Units, Toledo, Ohio) with a capacity of approximately 760 l. Two diverter valves were placed at the outlet of the pump to allow control of flow through the apparatus. An expanding joint was used to direct flow from the pump into a test chamber consisting of a 91.5-cm long clear PVC pipe with a 7.62-cm diameter. The flow continued through two 90-degree turns to return to the living stream reservoir. We marked the diverter valve nearest the test chamber with five equally-spaced marks from completely closed to completely open. The second diverter valve which diverted flow away from the test chamber was left completely open when leopard darters in preliminary tests were unable to survive high current velocities (approximately 40 cm/s). A petcock valve at the upstream end of the test chamber allowed excess air to be bled from the system, and an access plug at the downstream end was used to introduce and remove fish from the apparatus. Plastic mesh was placed at both ends of the test chamber to restrict fish to the chamber and minimize turbulent flow.

We measured the swimming performance of leopard darters at four treatments (current velocities), and randomly selected the order in which they would be tested. We first placed individual leopard darters into the test chamber and allowed them to acclimate for five minutes. After acclimation, they were immediately exposed to the test current velocity.

We observed the fish and used an audio tape-recorder to record the burst activity of the fish during a period of ten minutes. Eight randomly chosen darters were used in each treatment, and each individual was used once. Data recorded on the tapes included burst frequency (bursts/10 min), duration (s), distance (cm) and speed (body lengths/ sec), and total distance of bursts (cm). Swimming bursts in the 5-25 cm/s treatments consisted of an individual orienting into the current and making a rapid burst upstream. All fish swam along the bottom of the test chamber. After active swimming ceased, the fish drifted backward to the mesh at the back of the test chamber. The point at which active forward movement stopped was considered the end of a burst.

For presentation, mean values were calculated from all burst events without consideration to variation by individual fish. Data for burst frequency and total distance of bursts were analyzed with ANOVA. Since individual fish showed varying numbers of bursts, the remaining variables (burst duration, distance, and speed) were analyzed with a nested ANOVA to separate out the variance due to individuals. A Bonferroni joint estimation procedure (Neter et al. 1990) was used for multiple comparisons.

Immediately following the last treatment, we used a dye injected into the petcock valve to measure current velocity at each mark on the diverter valve. Velocity was determined by timing movement of the dye over 50 cm. Five replicates were taken for each valve setting and the mean was used as the current velocity for that treatment. The four velocities were significantly different (ANOVA F = 333.21, $P < 0.0001$) and were approximately equal to 0, 5, 12, and 25 cm/s.

Field measurements--We measured current velocity at the inlet and outlet of open-box and corrugated pipe culverts at road crossings in the Glover River drainage. Because spawning typically begins in early March (James and Maughan 1989), measurements were taken once in mid-February and once in early March 1996. One current velocity measurement was taken near the bottom surface of the inlet and outlet of small pipe culverts with a Marsh-McBirney meter (Model 201). In larger pipe culverts, we measured velocities at two points along a perpendicular transect at both ends, and we measured four to five points in a perpendicular transect across both ends of box culverts. We pooled the velocity measurements from all culverts and both sampling dates for each road crossing. Data from crossings with both pipe and box culverts were pooled by culvert type.

Results

Behavior of leopard darters in the swimming apparatus was similar to that observed at high current velocities in the field. After being placed in the apparatus, each darter typically rested on or near the plastic mesh at the back of the test chamber. In velocities of 0 and 5 cm/s, most individuals remained in the same position for the entire ten minutes. Individuals were more active at velocities of 12 and 25 cm/s, and at 25 cm/s they occasionally began a new burst before drifting completely back to the mesh. Only forward movement was considered part of the burst. No fish were observed to swim actively downstream.

Leopard darter swimming performance at the highest velocity (25 cm/s) was significantly different from that at the three lower velocities (Table 1). Another notable difference was that only one fish showed any swimming activity at 0 and 5 cm/s but all eight fish were active at 25 cm/s. At velocities of 25 cm/s, fish swam for longer periods at higher speeds and for greater distances than at the lower velocities. Darters in the highest velocity also swam more frequently than those in velocities of 0 and 5 cm/s. Burst duration exhibited the only significant nested effect.

Box plots for water velocities measured at both ends of culverts through several road crossings indicated that values for the majority of culverts were greater than our highest laboratory velocity (Figure 2). At five of the seven road crossings with pipe culverts, 13-31% of the points were below 25 cm/s, indicating that each crossing may have had one or more culverts with low velocities. One crossing (EF2) had only one point with a velocity higher than 25 cm/s while another crossing (MG1) had extremely high velocities(median > 1.4 m/s; only one observation below 60 cm/s). Water velocities within box culverts tended to be lower and less variable than those in pipe culverts. Velocities at two of the crossings were near 25 cm/s, whereas almost 100% of the velocities at three crossings were higher than 25 cm/s.

Water velocity is only one of the potential barriers at road crossings, and six of ten crossings had culverts with multiple barriers. These barriers included scour-created cascades immediately downstream from a culvert, absence of refuge pools at the downstream or upstream end of culverts, and a high hydraulic head at the inlet of culverts. Also, all of the culverts were considerably longer (mean box culvert length = 4.6 m, mean pipe culvert length = 5.4 m) than the total distance traversed by leopard darters in the laboratory apparatus (Table 1).

Discussion

Many culverts in the Glover River drainage may pose a problem for passage of leopard darters during certain discharge levels. Water velocities at most culverts were well above the highest velocity tested in the lab (25 cm/s), although it appeared that at least one culvert at each crossing had velocities lower than 25 cm/s. However, single culverts with low current velocities are effective only if they are in the pathway of fish migration (Baker and Votapka 1990), and leopard darters may not locate that culvert. In addition, current velocities in pipe culverts at one crossing (MG1; Figure 2) and box culverts at two crossings (EF4 and MG3) were nearly always greater than 25 cm/s.

Swimming activity increased at the highest water velocity tested in the lab (25 cm/s). We were unable to conclude that higher velocities were barriers to movement and thus determine an endpoint velocity at which leopard darters would not be able to swim. Our permit did not allow collection of more individuals that could be used to test higher velocities. However, Jones et al. (1974) concluded that water velocities in culverts should be below 30-40 cm/s to allow passage of most migratory species.

In addition to the difficulty of swimming against high current velocities, leopard darters are exposed to a variety of other barriers at some road crossings. All of the culverts were longer than the total distance traversed in our experiment (Table 1). The greatest total distance covered by an individual leopard darter during a ten minute trial was 1.55 m compared to the 4-6 m length of culverts. Mean burst lengths also were considerably shorter. In the 25 cm/s water velocity, three fish had single bursts of 90 cm, nearly the entire length of the swimming chamber. All of the remaining bursts in that velocity ($\underline{N} = 44$), however, were <20 cm. Except for one box culvert with heavy algal growth, none of the culverts appeared to have water-velocity refuges at distances corresponding to the mean burst distance. Leopard darters in the laboratory apparatus did not maintain position after a swimming burst and drifted back to the end of the test chamber. Without current velocity refuges in the culverts, leopard darters would likely have difficulty holding position long enough to traverse a culvert through multiple swimming bursts. In addition, six crossings had additional barriers such as an hydraulic jump at the upstream end, a cascade at the downstream end, or an absence of a refuge pool at one or both ends. The crossing with the highest current velocities (MG1; Figure 2) had all of the additional barriers, including a drop of approximately 0.75 m just downstream from the culverts.

An additional factor that may influence the ability of leopard darters to traverse culverts is the presence of pathogens. In 1995, we exposed leopard darters to a higher velocity (about 40 cm/s) and had nearly 100% mortality by the next day. That experiment, however, was confounded by the presence of an unknown pathogen in the lab (M. Ewing, Oklahoma State University), but pathogens are occasionally evident in the field. For example, in spring 1996, 61% of darters captured at sites in the West Fork Glover River were infected by fungal growths on the fins and gills (Toepfer, personal observation).

It is unclear whether leopard darters make long-distance movements or even need to move long distances. Migration of fishes often is associated with reproduction, and some darters migrate during spring spawning (Winn 1958; Ingersoll et al. 1984). Several _Percina_ spp. also are presumed to migrate in association with spawning (Trautman 1981). There is some evidence that leopard darters show migration from pools into riffles associated with spawning (Jones et al. 1984; James and Maughan 1989) although one systematic effort to examine migration of marked individuals was inconclusive (Toepfer et al. 1996). In addition, spawning does not occur in all riffles even though adjacent pools contain leopard darters during the year (James 1989), suggesting that individuals in those locations are required to migrate to other riffles.

Because leopard darters have essentially one reproductive season during their estimated lifespan of 18 months (James et al. 1991), any barrier to migratory movement might be critical to maintaining local abundances.

During higher flow, when water covers the road, there may be opportunities for darters to move along the edges of the stream, although some darters appear to move infrequently during high flow events (Freeman 1995). In years such as 1996, however, rainfall during the entire spawning season was low and all flow at road crossings was through culverts.

Finally, the combination of anthropogenic factors and the leopard darter's limited lifespan and reproductive opportunities (James et al. 1991) make the species especially vulnerable to localized extirpation. In such a species, recolonization from other areas may be particularly important for the persistence of local populations. For example, in November 1976 a chemical spill extirpated leopard darters from a 16-km reach of the upper Mountain Fork River (Robison 1978), and by 1987 the species had recolonized the area (Zale et al. 1994). Anthropogenic impacts from silviculture, pesticides, fertilizers, and poultry and swine farming could cause periodic extirpations of local populations of leopard darters. Although culverts probably are not long-term barriers to migration or dispersal of leopard darters, their ability to act as barriers during some years may have dramatic effects on populations in areas near road crossings.

Acknowledgments

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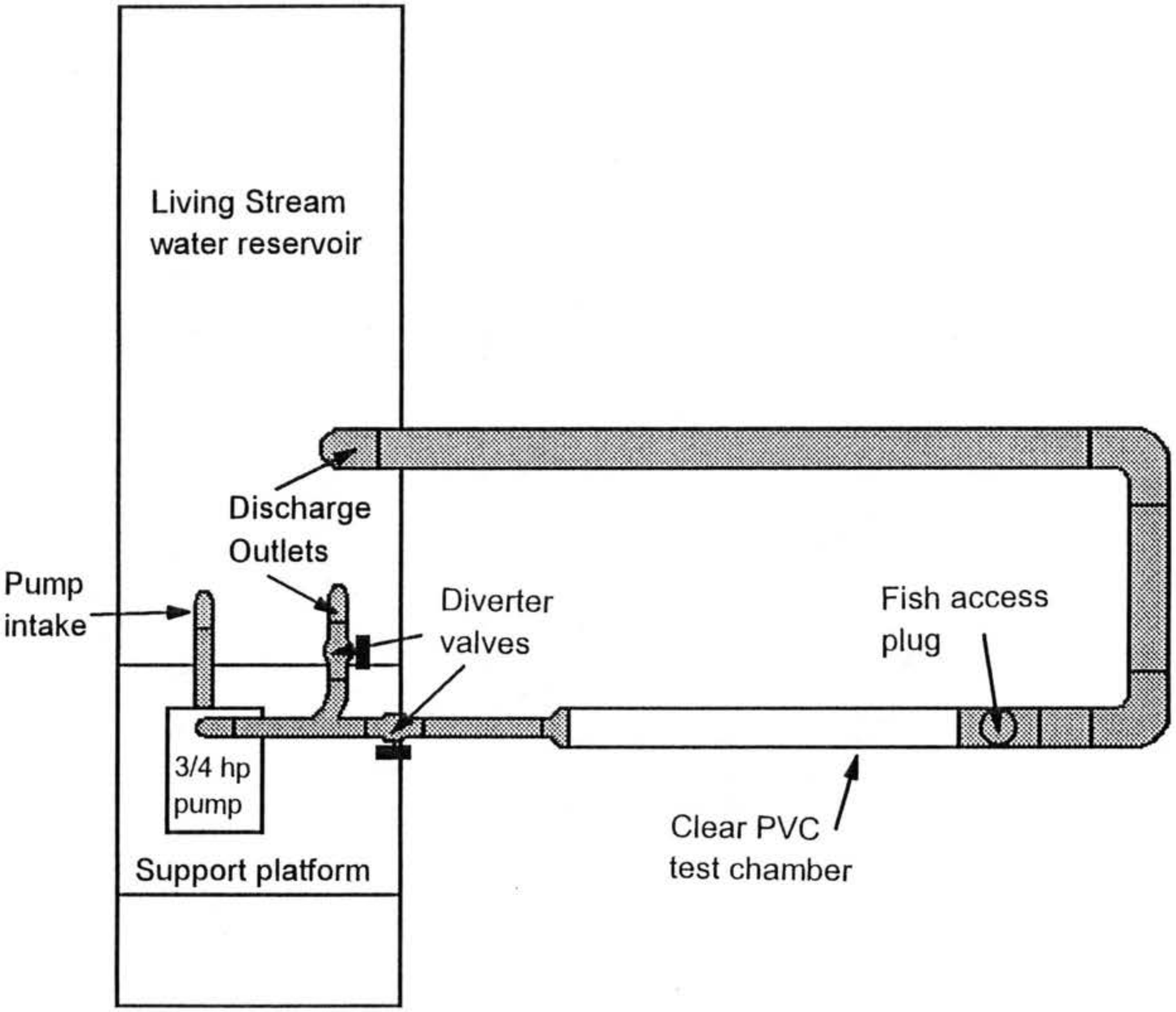
Table 1. Mean values (± 1 SD) for swimming performance variables. Results for the treatment and nested effects in the ANOVAs are at the bottom. Values with different letters were significantly different in the Bonferroni multiple comparisons test.

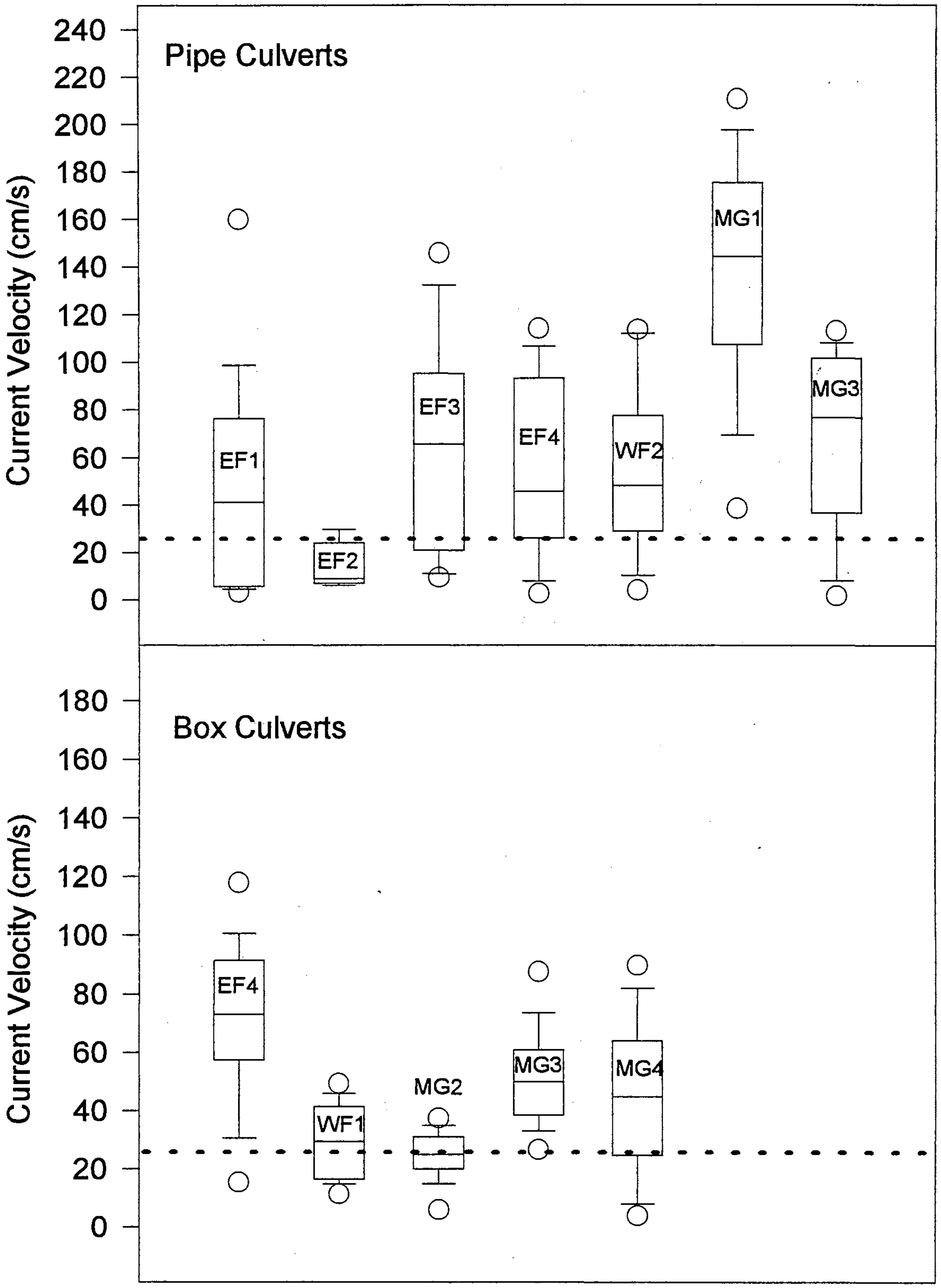
Current Velocity (cm/s)	Number Active	Burst Frequency	Duration of Burst (s)	Burst Distance (cm)	Burst Speed (BL/s)	Total Distance (cm)
0	1	0.25 ± 0.71x	0.44 ± 0.90x	0.44 ± 0.88x	0.04 ± 0.07x	0.50 ± 1.41x
5	1	1.00 ± 2.83x	0.55 ± 0.55x	1.13 ± 1.24x	0.17 ± 0.19x	2.12 ± 6.01x
12	5	3.12 ± 3.60xy	0.98 ± 0.47x	2.28 ± 1.38x	0.38 ± 0.25x	8.00 ± 10.28x
25	8	5.88 ± 2.90y	1.53 ± 1.61y	14.23 ± 20.01y	1.40 ± 1.04y	90.75 ± 57.79y
ANOVA *F*						
Vel		7.26	81.52	12.98	10.85	17.56
Fish(Vel)		NA	4.41	1.63	0.73	NA
ANOVA *P*						
Vel		<0.001	<0.0001	<0.025	<0.025	<0.025
Fish(Vel)		NA	<0.0001	>0.050	>0.050	NA

Figure Captions

Figure 1. Schematic of the flow-through swimming performance apparatus, top view.

Figure 2. Box and whisker plots of current velocity in culverts at each road crossing in the Glover River drainage. The boxes cover the central 50 percent of the observations, and the whiskers extend to the 5th and 95th percentiles. The dashed line across each section of the figure indicates 25 cm/s, the highest velocity tested in the laboratory swimming performance experiment. EF indicates crossing in East Fork Glover River, WF indicates those in West Fork Glover River, and MG indicates those in mainstem Glover River.


Living Stream water reservoir
Discharge Outlets
Pump intake
Diverter valves
Fish access plug
3/4 hp pump
Support platform
Clear PVC test chamber


Pipe Culverts
Current Velocity (cm/s)
240
220
200
180
160
140
120
100
80
60
40
20
0
EF1
EF2
EF3
EF4
WF2
MG1
MG3
Box Culverts
Current Velocity (cm/s)
180
160
140
120
100
80
60
40
20
0
EF4
WF1
MG2
MG3
MG4

Appendix 1. Swimming activity of leopard darters.

Current Velocity	Fish	Standard Length (mm)	Burst Number	Burst Duration (s)	Burst Distance (cm)	Burst Speed (BL/s)
0	1	54	0	0	0	0
0	2	63	1	2.4	2	0.13
			2	1.6	2	0.20
0	3	55	0	0	0	0
0	4	66	0	0	0	0
0	5	60	0	0	0	0
0	6	62	0	0	0	0
0	7	55	0	0	0	0
0	8	61	0	0	0	0
5	1	56	0	0	0	0
5	2	64	1	1.1	2	0.28
			2	0.9	3	0.52
			3	1.1	3	0.43
			4	1.4	2	0.22
			5	0.8	1	0.19
			6	0.8	2	0.39
			7	1.0	1	0.16
			8	1.2	3	0.39
5	3	53	0	0	0	0
5	4	54	0	0	0	0
5	5	55	0	0	0	0

Appendix 1. Continued.

Current Velocity	Fish	Standard Length (mm)	Burst Number	Burst Duration (s)	Burst Distance (cm)	Burst Speed (BL/s)
5	6	61	0	0	0	0
5	7	55	0	0	0	0
5	8	56	0	0	0	0
12	1	71	1	1.4	6	0.60
			2	1.2	2	0.23
			3	1.2	3	0.35
			4	1.6	3	0.26
			5	1.0	2	0.28
			6	1.8	3	0.23
			7	1.8	4	0.31
			8	1.6	2	0.18
12	2	50	1	0.9	2	0.44
			2	0.8	1	0.25
			3	0.8	2	0.50
			4	0.8	3	0.75
12	3	54	0	0	0	0
12	4	53	0	0	0	0
12	5	53	1	1.4	2	0.27
			2	0.7	1	0.27

Appendix 1. Continued.

Current Velocity	Fish	Standard Length (mm)	Burst Number	Burst Duration (s)	Burst Distance (cm)	Burst Speed (BL/s)
12	6	60	0	0	0	0
12	7	55	1	1.2	3	0.45
12			2	1.2	2	0.30
12	8	55	1	1.1	2	0.33
12			2	0.8	5	1.14
			3	0.7	1	0.26
			4	0.7	2	0.52
			5	1.0	2	0.36
			6	1.1	3	0.50
			7	1.1	2	0.33
			8	0.7	2	0.52
			9	0.8	4	0.91
25	1	71	1	0.9	2	0.31
			2	0.8	1	0.18
			3	0.7	1	0.20
			4	0.8	1	0.18
25	2	60	1	11.6	90	1.29
25	3	59	1	0.9	3	0.56
			2	0.9	6	1.13
			3	1.0	10	1.69
			4	1.8	6	0.56
			5	1.0	4	0.68

Appendix 1. Continued.

Current Velocity	Fish	Standard Length (mm)	Burst Number	Burst Duration (s)	Burst Distance (cm)	Burst Speed (BL/s)
25	4	74	1	3.0	12	0.54
			2	1.2	8	0.90
			3	1.9	16	1.14
			4	0.8	6	1.01
			5	1.2	9	1.01
25	5	61	1	0.9	9	1.64
			2	4.6	90	3.21
			3	0.7	8	1.87
			4	1.4	15	1.76
			5	1.1	10	1.49
			6	1.4	11	1.29
			7	1.2	12	1.64
25	6	61	1	1.2	6	0.82
			2	0.9	11	2.00
			3	1.7	7	0.68
			4	2.9	18	1.02
			5	0.9	3	0.55
			6	2.9	7	0.39
			7	2.5	90	5.90
25	7	71	1	1.4	25	2.51
			2	1.1	25	3.20
			3	1.1	30	3.84
			4	0.8	16	2.82

Appendix 1. Continued.

Current Velocity	Fish	Standard Length (mm)	Burst Number	Burst Duration (s)	Burst Distance (cm)	Burst Speed (BL/s)
			5	1.4	6	0.60
			6	0.7	4	0.80
			7	0.9	15	2.35
			8	1.7	9	0.75
			9	1.1	8	1.02
			10	1.1	6	0.77
			11	1.0	8	1.13
25	8	56	1	1.1	9	1.46
			2	1.0	8	1.43
			3	1.2	6	0.89
			4	1.1	12	1.95
			5	1.7	8	0.84
			6	0.9	11	2.18
			7	1.2	11	1.64
			8	1.0	13	2.32
			9	1.1	4	0.65
			10	1.4	12	1.53
			11	1.1	8	1.30

Appendix 2. Current velocities in culverts at road crossings in Glover River drainage. NM indicates points that were not measurable.

Legal Description	Sample Date	Culvert Type	Culvert Number	Upstream/ Downstream	Point	Current Velocity (cm/s)
T2S R24E Sec. 5	2/16/96	Pipe	1	Up	1	9
Weyerhauser 63000				Down	1	31
Crossing			2	Up	1	6
				Down	1	NM
		Box	1	Up	1	24
					2	18
					3	17
					4	15
					5	15
				Down	1	43
					2	39
					3	52
					4	45
					5	44
	3/03/96	Pipe	1	Up	1	7
				Down	1	24
			2	Up	1	7
				Down	1	24
		Box	1	Up	1	24
					2	18

Appendix 2. Continued.

Legal Description	Sample Date	Culvert Type	Culvert Number	Upstream/ Downstream	Point	Current Velocity (cm/s)
					3	16
					4	16
					5	8
				Down	1	40
					2	35
					3	38
					4	47
					5	39
T2S R24E Sec. 8	2/16/96	Pipe	1	Up	1	3
Weyerhauser 60000				Down	1	30
Road Crossing			2	Up	1	174
				Down	1	76
			3	Up	1	5
				Down	1	5
	3/03/96	Pipe	1	Up	1	4
				Down	1	41
			2	Up	1	77
				Down	1	15
			3	Up	1	6
				Down	1	5

Appendix 2. Continued.

Legal Description	Sample Date	Culvert Type	Culvert Number	Upstream/ Downstream	Point	Current Velocity (cm/s)
T2S R24E Sec. 7 Y-road off Weyerhauser 60000	2/16/96	Pipe	1	Up	1	1
				Down	1	16
			2	Up	1	24
				Down	1	97
			3	Up	1	56
				Down	1	117
			4	Up	1	28
				Down	1	83
	3/03/96	Pipe	1	Up	1	7
				Down	1	31
			2	Up	1	54
				Down	1	107
			3	Up	1	37
				Down	1	102
				Up	1	28
				Down	1	89
T2S R23E Sec. 26 Weyerhauser 56500 Road Crossing	2/16/96	Pipe	1	Up	1	21
				Down	1	95
			2	Up	1	65
				Down	1	131
			3	Up	1	19
				Down	1	95

Appendix 2. Continued.

Legal Description	Sample Date	Culvert Type	Culvert Number	Upstream/ Downstream	Point	Current Velocity (cm/s)
			4	Up	1	6
				Down	1	21
			5	Up	1	72
				Down	1	142
			6	Up	1	11
				Down	1	50
	3/04/96	Pipe	1	Up	1	43
				Down	1	105
			2	Up	1	79
				Down	1	120
			3	Up	1	16
				Down	1	82
			4	Up	1	11
				Down	1	39
			5	Up	1	82
				Down	1	154
			6	Up	1	35
				Down	1	66
T2S R23E Sec. 18	2/16/96	Pipe	1	Up	1	6
				Down	1	70
			2	Up	1	51
				Down	1	117

Appendix 2. Continued.

Legal Description	Sample Date	Culvert Type	Culvert Number	Upstream/ Downstream	Point	Current Velocity (cm/s)
			3	Up	1	42
				Down	1	103
			4	Up	1	28
				Down	1	112
			5	Up	1	34
				Down	1	105
			6	Up	1	45
				Down	1	112
	3/04/96	Pipe	1	Up	1	0
				Down	1	19
			2	Up	1	43
				Down	1	74
			3	Up	1	30
				Down	1	77
			4	Up	1	32
				Down	1	78
			5	Up	1	24
				Down	1	68
			6	Up	1	11
				Down	1	56
T3S R23E Sec. 7	2/16/97	Box	1	Up	1	54
					2	54

Appendix 2. Continued.

Legal Description	Sample Date	Culvert Type	Culvert Number	Upstream/ Downstream	Point	Current Velocity (cm/s)
Weyerhauser 56000 Road Crossing					3	53
					4	53
					5	62
				Down	1	69
					2	47
					3	40
					4	60
					5	64
			2	Up	1	91
					2	84
					3	77
					4	79
					5	73
				Down	1	38
					2	9
					3	21
					4	28
					5	75
			3	Up	1	42
					2	29
					3	37
					4	24
					5	43

Appendix 2. Continued.

Legal Description	Sample Date	Culvert Type	Culvert Number	Upstream/ Downstream	Point	Current Velocity (cm/s)
				Down	1	23
					2	27
					3	64
					4	44
					5	2
			4	Up	1	12
					2	12
					3	12
					4	8
					5	4
				Down	1	96
					2	23
					3	20
					4	58
					5	50
	3/04/96	Box	1	Up	1	44
					2	44
					3	45
					4	47
					5	50
				Down	1	62
					2	51

Appendix 2. Continued.

Legal Description	Sample Date	Culvert Type	Culvert Number	Upstream/ Downstream	Point	Current Velocity (cm/s)
					3	50
					4	43
					5	52
			2	Up	1	77
					2	80
					3	73
					4	75
					5	80
				Down	1	82
					2	47
					3	103
					4	106
					5	86
			3	Up	1	37
					2	41
					3	41
					4	11
					5	34
				Down	1	42
					2	86
					3	60
					4	37
					5	15

Appendix 2. Continued.

Legal Description	Sample Date	Culvert Type	Culvert Number	Upstream/ Downstream	Point	Current Velocity (cm/s)
			4	Up	1	3
						2 4
					3	4
					4	0
					5	5
				Down		NM
T3S R23E Sec. 32 Weyerhauser 72000 Road Crossing	2/16/96	Pipe	1	Up	1	106
					2	70
				Down	1	99
					2	104
			2	Up	1	71
					2	6
				Down	1	46
					2	35
		Box	1	Up	1	49
					2	42
					3	58
					4	62
				Down	1	62
					2	60
					3	75
					4	93

Appendix 2. Continued.

Legal Description	Sample Date	Culvert Type	Culvert Number	Upstream/ Downstream	Point	Current Velocity (cm/s)
	3/05/96	Pipe	1	Up	1	82
					2	89
				Down	1	91
					2	108
			2	Up	1	0
					2	115
				Down	1	28
					2	38
		Box	1	Up	1	43
					2	24
					3	33
					4	40
				Down	1	37
					2	34
					3	54
					4	51
T4S R23E Sec. 32 Weyerhauser 71400 Road Crossing	2/16/96	Pipe	1	Up	1	174
				Down	1	125
					2	130
			2	Up	1	211
				Down	1	192
					2	195

Appendix 2. Continued.

Legal Description	Sample Date	Culvert Type	Culvert Number	Upstream/ Downstream	Point	Current Velocity (cm/s)
			3	Up	1	114
				Down	1	186
			4	Up	1	75
				Down	1	213
					2	171
			5	Up	1	47
				Down	1	188
					2	37
			6	Up	1	153
				Down	1	174
					2	179
	3/05/96	Pipe	1	Up	1	159
				Down	1	144
					2	126
			2	Up	1	NM
				Down	1	21
			3	Up	1	101
				Down	1	170
					2	161
			4	Up	1	94
				Down	1	207
					2	168
			5	Up	1	136

Appendix 2. Continued.

Legal Description	Sample Date	Culvert Type	Culvert Number	Upstream/ Downstream	Point	Current Velocity (cm/s)
				Down	1	105
					2	127
			6	Up	1	108
				Down	1	110
					2	96
T5S R23E Sec. 9 Weyerhauser 71000 Road Crossing	2/16/96	Box	1	Up	1	32
					2	36
					3	34
					4	34
					5	24
				Down	1	30
					2	30
					3	31
					4	20
					5	0
			2	Up	1	25
					2	32
					3	29
					4	31
					5	38
				Down	1	19
					2	31

Appendix 2. Continued.

Legal Description	Sample Date	Culvert Type	Culvert Number	Upstream/ Downstream	Point	Current Velocity (cm/s)
					3	24
					4	24
					5	18
			3	Up	1	41
					2	38
					3	37
					4	34
					5	30
				Down	1	24
					2	30
					3	36
					4	30
					5	21
	3/05/96	Box	1	Up	1	18
					2	29
					3	30
					4	25
					5	14
				Down	1	6
					2	16
					3	29
					4	23
					5	23

Appendix 2. Continued.

Legal Description	Sample Date	Culvert Type	Culvert Number	Upstream/ Downstream	Point	Current Velocity (cm/s)
			2	Up	1	17
					2	31
					3	5
					4	20
					5	16
				Down	1	20
					2	23
					3	26
					4	26
					5	19
			3	Up	1	25
					2	31
					3	32
					4	30
					5	18
				Down	1	14
					2	24
					3	25
					4	22
					5	6

CHAPTER V

IMPACTS OF LAND USE ON STREAM HABITAT FOR THE LEOPARD DARTER

Abstract. We examined the effects of land use in two watersheds in southeastern Oklahoma on quality of habitat for an endemic, threatened fish, the leopard darter *Percina pantherina*. Results from a nonpoint source model indicated that sediment yields were concentrated in the lower portion of mainstems in each watershed, and the yields in one watershed were 2-3 times higher than in the other. Within the mainstems there were nearly equal amounts of habitat that were potentially usable by leopard darters, although the distribution of usable habitat differed longitudinally. There were significant differences in fine sediments and turbidity between streams, indicating a potential reduction in habitat quality because of sedimentation in one stream.

INTRODUCTION

Ecologists and natural resource managers have long recognized the relationships between land and water ecosystems. Management activities in these systems, however, have often been conducted with little regard to their effects on each other. With many natural resource and land management agencies moving toward using an ecosystem approach for managing and conserving natural resources (USDA Forest Service 1993, National Research Council 1993), information about land-water interactions and their effects on stream biota is needed. Recent developments in remote sensing and Geographic Information Systems (GIS) permit examination of ecological patterns and land use at large spatial scales, although such analyses require complementary measurements at finer scales to determine meaningful associations (King 1993). Incorporating multiple spatial scales and establishing their linkages will not only provide a more integrated approach to natural resource management, but will also promote more efficient and effective conservation of biological diversity.

Threatened or endangered species often serve as indicators of ecosystem integrity (King 1993) and their ability to persist depends upon interactions between life history traits (e.g., population dynamics, habitat preferences, and seasonal movements) and prevailing environmental conditions (Lubchenco et al. 1991). Efforts directed toward recovery of rare and endangered aquatic species must focus on activities in water and land ecosystems and their influence on critical habitats. For example, intensive land use activities, such as clear-cutting timber harvest and associated road construction, may have an impact on the quality of stream habitat for rare or imperiled species.

The leopard darter *Percina pantherina* is a percid endemic to five streams of the Little River drainage in southeastern Oklahoma and southwestern Arkansas. Its apparent rarity (Cloutman and Olmsted 1974, Robison et al. 1974) led to its designation as a threatened species (U.S. Fish and Wildlife Service 1978). Leopard darters are confined to the middle and upper reaches of many of the streams in the drainage because of impoundments and unsuitable habitat in some areas (Zale et al. 1994).

An important factor regulating populations of threatened species, such as the leopard darter, is the amount and quality of habitat available to support the population. Although fine-scale habitat preferences have been documented for leopard darters (Jones et al. 1984), little has been done to determine quality of stream reaches for concerted protection or reintroduction efforts. Critical habitat has been designated (U.S. Fish and Wildlife Service 1978), but not all areas of the critical habitat are actually capable of supporting leopard darters (James and Collins 1993). To facilitate leopard darter management, it is necessary to identify the river localities with suitable habitat that support existing populations or could serve as potential sites for future reintroduction of the species.

Our objective was to use a multi-scale approach to examine potential land use effects on quality of usable habitat for leopard darters. Specifically, we used watershed models to predict sedimentation generated by land use activities. We then identified areas within two streams that had habitat usable by leopard darters. Finally, we compared fine sediments and turbidity in the streams to see if land use effects had reduced the quality of usable habitat in each stream.

METHODS

Study Area

Big Eagle Creek watershed is located in the Ouachita Mountains in southern LeFlore and northern McCurtain counties, Oklahoma (Figure 1). West Fork Glover River watershed is at the southern edge of the Ouachita Mountains and is located almost entirely in northern McCurtain County, Oklahoma. Big Eagle Creek watershed covers an area of approximately 240 km^2, and West Fork Glover River watershed is approximately 271 km^2.

Soils in the Ouachita mountains are derived largely from sandstone, shale, and novaculite sedimentary rocks (Thornbury, 1965). Most of the land in both watersheds is heavily forested, and silviculture and poultry farming are the major land use activities.

Watershed Model

We used the Agricultural Nonpoint Source (AGNPS) model (Young et al. 1987) to predict sediment yield in the two watersheds. A single storm producing 4.3" of rain was simulated in each model. AGNPS is a distributed parameter model in which a watershed is divided into square grid units. Runoff characteristics and sediment transport are calculated for each cell and routed through cells to the outlet of the watershed. A modified version of the Universal Soil Loss Equation (USLE) is used by AGNPS to determine sediment yields (Young et al. 1987). The model requires input of 22 parameters that describe the topography, soil types, land use, and channel morphologies within the watershed (Table 1). We used the WATER, Soil, Hydro-Environmental Decision Support System (WATERSHEDSS) from the North Carolina State University Water Quality Group to generate data layers for the AGNPS model. WATERSHEDSS uses GRASS4.1 to generate data for the AGNPS model, runs the model, and exports AGNPS output. Four basic data layers were obtained and these layers were either modified within GRASS4.1 or by WATERSHEDSS to generate the remaining parameters for the model.

The watershed boundaries were created by digitizing boundaries from 1:24000 topographic quad maps. The vector boundaries were then converted into 200-m resolution cells for use in AGNPS. We created 600 m buffers around the watersheds to use in "clipping" the soil and topography layers to reduce data processing in later steps.

Topography information was obtained from a 1:250,000 digital elevation model (DEM) through the U.S. Geological Survey. The data were processed to 200-m resolution for the AGNPS model, and the model interface automatically generated slope and aspect from the initial topography layer. We edited the aspect layer by hand, however, since many of the cells were not properly routed downstream.

Soil data were obtained from the Map Information Assembly Display (MIADS) database provided by the Natural Resources Conservation Service (NRCS). We copied the soil data to several new maps and edited the categories in each to obtain new data layers describing specific soil features. The USLE K-factor and soil hydrologic groups were determined from Soil Conservation Survey soil surveys for Leflore, McCurtain, and Pushmataha counties in Oklahoma (Table 2). A weighted K-factor was determined for soil complexes and associations by summing the products of the K-factor for each soil and its percent composition in the complex. When available in the soil surveys, percent sand and percent clay were also determined for each soil type.

In some cases a range for percent clay was given for a soil type, and we calculated the mean of the upper and lower values for use in the model. Percent sand was often unavailable in the soil surveys. In those cases we used data for percent of soil passing through P200 (75 μm) and P10 (2 mm) sieves to calculate a value (C. Sample, NRCS, personal communication): % sand = 100 - (P200/P10 * 100). There was also a range for the P200 and P10 sieve fractions; we again used the mean of the upper and lower values in our calculations.

Land use from 1985 also was obtained from the MIADS database (Table 3). We copied the land use map to new maps and edited the category labels to generate additional layers required by AGNPS. The C-factor is a measure of the effects of different cover and management regimes compared to an identical area in tilled, continuous fallow (Dissmeyer and Foster 1984). C-factor values for most land use types (Table 4) were taken directly from tables (Dissmeyer and Foster 1984, Young et al. 1987). We calculated the C-factor for clearcuts using subfactors provided by Dissmeyer and Foster (1984). Other layers derived from the land use layer were fertilizer/nutrient application, type of machine used in tillage, and management practice. Although we did input values for each land use category, we assumed that these factors would have little influence on sedimentation since agriculture is rare in both watersheds (Table 3).

Stream Habitat Determination

Initial base maps depicting the edges of Big Eagle Creek and West Fork Glover River were digitized from tracings of 1:7920 aerial photographs obtained from NRCS. Because leopard darters do not occur in headwater areas, we mapped the lower 21-km of continuous mainstem in each watershed. We then used a classification scheme by McCain et al. (1990) to identify and map the location of mesohabitat types during float trips down each stream.

We used transect sampling to measure habitat characteristics within mesohabitat types. Our initial selection of transect sampling sites was based primarily on logistics; we pre-determined the amount of habitat that could be sampled within 4-5 days. We attempted to sample all available locations of rare mesohabitat types (occurring <5 times in the entire stream). If a mesohabitat type was uncommon (occurred <5 times in each segment), one randomly-chosen example of that type was sampled from each segment. Remaining effort was divided proportionally by the frequency of common mesohabitat types and by stream segment.

We subjectively placed the first transect within 10 m of the upstream end of each unit of mesohabitat. In most cases a minimum of three transects was sampled. When mesohabitat units were too short for three transects we used two transects and two additional randomly located sampling points. A total of ten subjectively located points were used to classify habitat at two narrow cascades created by concrete dams in Big Eagle Creek. Water depth, current velocity (at 0.6 depth), and substrata were measured at four equally-spaced points along each transect or at each extra sampling point. A Modified Wentworth particle size scale (Bovee and Cochnauer 1977) was used to characterize substrata in a 1 m^2 area around each sampling point by assigning a number for each 25% coverage of the area. A weighted mean was constructed for substrata at each sampling point by multiplying the dominant substratum score by four, the secondary substratum by three, tertiary by two, and quaternary by one, and dividing the quantity by 10.

Habitat suitability criteria for each mesohabitat type were derived from frequency distributions of depth, velocity, and substrate (James 1989) at points where individual leopard darters were first observed. We reduced the three levels of suitability classes by Thomas and Bovee (1993) into two, preferred and non-preferred. Preferred habitat corresponded to the suitable category (central 95%) of Thomas and Bovee (1993), and non-preferred habitat corresponded to their unsuitable category (remaining 5%). Preferred water depth was 25 to 90 cm, current velocity was 0 to 28 cm/s, and substrate was gravel, cobble, and boulder. We applied these criteria to all sampling points and derived a preference rating for each point. If all three habitat variables (depth, velocity and substrate) at a sampling point were preferred, the habitat at that point was classified as preferred. If any of the variables were not preferred, the habitat was classified as non-preferred (after Thomas and Bovee 1993). After deriving a suitability rating for each sampling point, we re-classified mesohabitat types based on the frequency of preferred and non-preferred points taken in that type (Appendices 1 and 2).

Sediment Effects

AGNPS generated a map indicating sediment yield for every cell in each watershed. We overlayed the vector image of the mesohabitat map and queried the sediment map to determine the sediment yield within individual units of mesohabitat. Many mesohabitat units were entirely within a single 4000-m^2 cell and had only one sediment yield value. Other mesohabitat units, however, extended across multiple cells. In those instances we determined the sediment yield at each end of the unit and calculated a mean yield. We also determined the linear distance from the upstream end of the mapped stream channel to the center of each mesohabitat unit.

Midchannel pools were not the highest quality mesohabitat in the mesohabitat analysis (Appendices 1 and 2), but our observations during mark-recapture population sampling (Chapter III) indicated that midchannel pools contained the highest densities of leopard darters. In addition, midchannel pools were the dominant mesohabitat type in both streams and accounted for 54% of the total mesohabitat area in Big Eagle Creek and 72% in West Fork Glover River. Therefore, we examined the substrata in midchannel pools in both streams to determine if potential sedimentation had impacted the drainages. We used the BASS pebble count procedure (Clingenpeel 1994) to quantify the substrate in midchannel pools in both streams. In the BASS method, ten substrata particles are sampled along a single midpoint, perpendicular transect in the pool. An observer waded or swam a transect in each pool and measured substrata at ten, equidistant points. At each point, the observer averted his eyes, touched the streambed with a single finger, and selected the first particle touched. The intermediate axis of each particle was measured and classified into a modified Wentworth scale that differed from the original Wentworth scale by a factor of square root of 2 (Schaub 1996). Although the BASS method is much less labor intensive than other pebble count methods, Schaub (1996) found that it generated data in two other Ouachita Mountain streams that were not significantly different from data obtained in more labor-intensive methods. Because leopard darters spawn near or in riffles at the head of pools (James and Maughan 1989), we placed an additional transect at the head of each pool. We also took a turbidity sample as an indirect measure of impacts from suspended sediments. Data for percent of fine sediments (<2 mm) and turbidity were pooled by stream and analyzed with t-tests. Percentages for fine sediments were arcsine transformed before analysis.

RESULTS

The maps of sediment yield in each watershed (Figures 2 and 3) indicated that erosion is fairly constant across most of both watersheds. There were areas (clearcuts) in the southwest corner of Big Eagle Creek watershed that had higher sediment yields than most of the upper watershed. Maximum sediment yield in Big Eagle Creek watershed was 2280 tons in a glide 1.2 km upstream from the outlet. At the outlet, yield declined slightly to 2224 tons. Sediment yield in most of the West Fork Glover watershed also was fairly homogenous but was about 2-3 times higher than in Big Eagle Creek. The maximum sediment yield in West Fork Glover watershed (6147 tons) was located about 13.4 km upstream from the outlet, and yield at the outlet was 5084 tons. Both mainstems and major tributaries concentrated sediment from the surrounding areas.

At the mesohabitat scale, the two streams were similar in amount of available high-quality habitat. Big Eagle Creek had a total of 40.07 hectares of optimal mesohabitat compared to 54.35 hectares in West Fork Glover River (Appendices 1 and 2). The distribution of mesohabitat differed between streams, however (Chapter III). In Big Eagle Creek, midchannel pools increased in frequency and size in the lower part of the drainage. There was less of a longitudinal trend in West Fork Glover River, and the largest pools were located in middle sections of the stream reach.

Sediment yield also varied longitudinally in both streams. There were two large increases in sediment yield within the first 10 km of the West Fork Glover mainstem (Figure 4), and clearcuts were near the stream in both instances. It appeared that much of the sediment was deposited within a few hundred meters after the second increase in sediment yield. There was a smaller increase in sediment yield about halfway down Big Eagle Creek but this area of the stream was upstream from most clearcutting activity.

Mean percent fine sediments along transects taken at the head of pools in Big Eagle Creek (3.66 ± 6.62) was not significantly different ($\underline{T}$ = 1.93, $\underline{P}$ = 0.06) from that at the head of pools in West Fork Glover River (9.47 ± 16.82). Percent fine sediments, however, did differ ($\underline{T}$ = 2.64, $\underline{P}$ = 0.01) in the middle of midchannel pools (Big Eagle, 4.63 ± 8.39; West Fork, 12.63 ± 14.85). Mean turbidity (NTU) in Big Eagle Creek (4.03 ± 1.29) also was significantly lower ($\underline{T}$ = 3.58, $\underline{P}$ = 0.001) than turbidity in West Fork Glover River (5.67 ± 1.60).

DISCUSSION

Sediment yields for a single-storm event were 2-3 times higher in West Fork Glover than in Big Eagle Creek. These results, however, were a measure of the total amount of sediment leaving each cell rather than sediment deposition. Because the majority of sediment yield is accounted for by storage and periodic flushing of alluvium (Schumm 1977), we interpreted the patterns of sediment yield as an indirect measure of sediment deposition.

The difference in sediment yields between watersheds was difficult to attribute to one cause such as land use. Erosion is affected by the combination of soil properties, land use, and topography within any one cell of the AGNPS model. There were some general differences between the two watersheds, however. Half of the soils in Big Eagle Creek watershed were in the B hydrologic group (Table 2) with the remaining soils divided between the C and D hydrologic groups. In contrast, 73.6% of the soils in West Fork Glover were in the C and D hydrologic groups. Soils in the C and D hydrologic groups have slow to very slow infiltration rates when thoroughly wet and have a higher rate of runoff. Therefore, erosive runoff may have occurred earlier and in greater quantities in West Fork Glover River than in Big Eagle Creek. Higher runoff may have had little effect, though, because the lower soil K-factors (Table 2) indicated that the soils generally were less erodible than those in Big Eagle Creek.

Although soils were similar between the watersheds, land use was considerably different. Over 91% of the total land area in Big Eagle Creek watershed was covered by forests (Table 3) which have low erosion rates (Dissmeyer and Foster 1984). Reforested clearcuts accounted only for

4.9% of the total land area. West Fork Glover watershed in comparison was only 49.3% forests, and over 40% of the watershed was reforested clearcuts. Clearcut areas in the southern area of Big Eagle Creek watershed produced more sediment relative to the upper watershed (Figure 2), but the largest increase in sediment yield (Figure 4) was upstream from those areas. In West Fork Glover, however, there were two large increases in sediment yield (Figure 4) that were near extensive clearcuts.

Differences in sediment yield were reflected by differences in measures of sediment in the two streams. Both percentage of fine sediments along mid-pool transects and turbidity were significantly higher in West Fork Glover River. Although there were differences in sediment at the center of pools, there did not appear to be an impact in potential spawning beds at the head of pools.

The modeling process had limitations that may have influenced the sediment yield results. The major limitation was that our available land use information was from 1985. Land use patterns likely have changed between that period and our mapping of mesohabitat and collection of sediment data. In addition, there was only one category in the land-use map for clearcuts. It is unlikely that all clearcuts in each watershed were the same age. We assumed during calculation of the C-factor that all clearcuts were 2 years old. Erosion from older clearcuts would be lower after pine replanting.

Another limitation of our models was that we were unable to account for erosion from logging roads. Average sediment yield from roads in a watershed in the Ouachita Mountains in Oklahoma range from 0.038 to 0.048 tons/acre/year (Miller et al. 1985, Scoles et al. 1995). Because AGNPS modeled a single storm, we were unable to account for the input of sediment from roads. Erosion from roads decrease rapidly as traffic levels drop after logging activities are reduced (Beschta 1978, Reid and Dunne 1984). In addition, culverts can redirect water flow from roads through vegetation and reduce the amount of sediment delivered to streams (Trimble and Sartz 1957, Waters 1995). Sediment delivery is highest when road culverts empty directly into stream channels. Miller et al. (1985) examined roads in an Arkansas watershed managed for silivculture and found that culverts emptying directly into stream channels accounted for only 5.5% of the drainage structures per kilometer of road. As a result, sediment delivery from roads to streams in the watershed was only about 1% of the total amount eroded from roads (Miller et al. 1985). Miller et al. (1985) suggested that the number of crossings may be more important than the actual area of roads. While tributaries in Big Eagle Creek and West Fork Glover watersheds had numerous road crossings, areas of the mainstems inhabited by leopard darters had few crossings. The mainstem of Big Eagle Creek had four crossings, including two paved roads, and West Fork Glover River had five crossings, one of which was paved.

We were not able to show a direct link between patterns of potential sedimentation and leopard darter populations. In Big Eagle Creek watershed, the highest sediment yield was at the lower end of the mainstem (Figure 2), but the highest densities of leopard darters also occurred in these areas (Chapter III). There was a similar trend in potential sedimentation in West Fork Glover, but densities of leopard darters did not show a distinct distributional pattern (Chapter III). In addition, sedimentation between watersheds and leopard darter densities showed an inverse relationship. Densities were about three times lower in West Fork Glover, which also had sediment yields that were 2-3 times higher than Big Eagle Creek.

Most of the work concerning effects of sediment on stream fishes has focused on salmonids in the Pacific Northwest, and the effects in other areas are still poorly understood (Waters 1995). Boschung and O'Neil (1981) found few differences in water quality or fish communities in reference and clearcut watersheds in Alabama. In contrast, Berkman and Rabeni (1987) found that siltation from agriculture activities had negative effects on fishes in northeast Missouri. Fishes that were benthic insectivores and those that required clean gravel for spawning, both characteristics of leopard darters (James 1989), showed decreases in abundance. Forested land tends to be associated with better water quality, lower sedimentation, and higher measures of biotic integrity than other land uses such as agriculture (Richards et al. 1996, Allan et al. 1997, Wang et al. 1997). Although these factors may have higher variation when land use includes nonforested land (i.e., Wang et al. 1997), biotic communities tend to show negative relationships with nonforested land use patterns.

Land use activities in the Ouachita Mountains do not appear to have dramatic negative impacts on the quality of stream habitat or water quality. Clingenpeel (1994) examined three pairs of reference and managed watersheds in Arkansas for a three-year period. He found that percent fine sediments and embeddedness did not differ between streams in each pair, which suggests that sedimentation may not be a factor in reducing quality of stream habitat. However, cumulative effects over a long time period may have more of a negative impact. Maughan et al. (1984) studied the impact of timber harvest on aquatic organisms in southeastern Oklahoma and concluded that there were no major changes in community composition over a 30-year period, but they did indicate that rare species had declined. In addition, Rutherford et al. (1987) indicated that intensive clear-cutting and associated activities (road building) in the 1960s caused a decline in species restricted to the eastern part of Oklahoma. Rutherford et al. (1992) also suggested that r-selected species (small, short-lived) may respond more quickly to perturbations from clear-cutting activity. The leopard darter is restricted to five streams in the Little River drainage and has a lifespan of only about 18 months (James et al. 1991). Its limited distribution and lifespan potentially make it more vulnerable to effects of land-use activities.

We consider the results of this study to be the first step in determining the effects of land use on populations of the leopard darter. Multi-scale approaches are particularly powerful in determining how activities at the watershed scale affect quality of habitat at larger scales (Allan et al. 1997). The quality of habitat for leopard darters appeared to be negatively affected by land use patterns at the watershed level. Future studies should focus on directly relating these effects to population parameters (e.g., growth, mortality, recruitment) of this species.

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Table 1. AGNPS model parameters, input data layer, and source (User-supplied or derived by AGNPS/GRASS interface). Asterisks indicate optional parameters.

Parameter	Data Layer	Source
Cell number	Topography	Interface
Receiving cell	Topography	Interface
Aspect	Topography	Interface/User
Land slope	Topography	Interface
Slope shape factor	Topography	Interface
Field slope length	Topography	Interface
Channel indicator	Topography	Interface/User
SCS curve number	Land use	Interface
Manning roughness coefficient	Land use	Interface
USLE management (C) factor	Land use	User
USLE support (P) factor	Land use	Interface
Soil condition constant	Land use	Interface
Soil hydrologic group	Soils	User
USLE erodibility (K) factor	Soils	User
Soil textural class	Soils	User
Fertilization level	Land use	User
Fertilizer availability factor	Land use	User
Point source indicator*	Topography	User
Gully source indicator*	Topography	User
Impoundment factor*	Topography	User
Chemical oxygen demand*	Land use	User

Table 2. Areas (hectares) of soil in Big Eagle Creek and West Fork Glover River watersheds by hydrologic group and soil erodibility factor. Percent of total area in each watershed is indicated in parentheses.

	Hydrologic Group			K (Erodibility) Factor		
Stream	B	C	D	≤0.20	0.21-0.30	>0.30
Big Eagle	12,000 (50.0)	11,132 (46.4)	852 (3.6)	3,100 (12.9)	18,772 (78.3)	2,112 (8.8)
West Fork	7,152 (26.4)	18,364 (67.8)	1,586 (5.8)	15,848 (58.6)	10,596 (39.2)	608 (2.2)

Table 3. Land use in Big Eagle Creek and West Fork Glover River watersheds.

	Big Eagle Creek		West Fork Glover River	
Land-use	Hectares	% Total Area	Hectares	% Total Area
Row Crops	NA	NA	4	0.01
Pastureland	728	3.03	2488	9.20
Pastureland-Brushy, Canopy >20%	404	1.68	68	0.25
Clearcut/Reforested	1176	4.90	10,928	40.40
Shortleaf Pine, >70% Pine	2488	10.37	2956	10.93
Shortleaf Pine/Oak, Mixed Forest	19,160	79.88	10,140	37.48
Oak/Hickory, >70% Deciduous	12	0.05	68	0.25
Bottomland Hardwoods	12	0.05	168	0.62
Postoak/Blackjack Oak	NA	NA	4	0.01
Native Pasture	NA	NA	160	0.59
Confined Feeding Area	4	0.02	20	0.07
Urban/Built-Up Land	NA	NA	48	0.18
Totals	23,984		27,052	

Table 4. C-factors for each type of land use. Most values were taken directly from tables provided by Dissmeyer and Foster (1984).

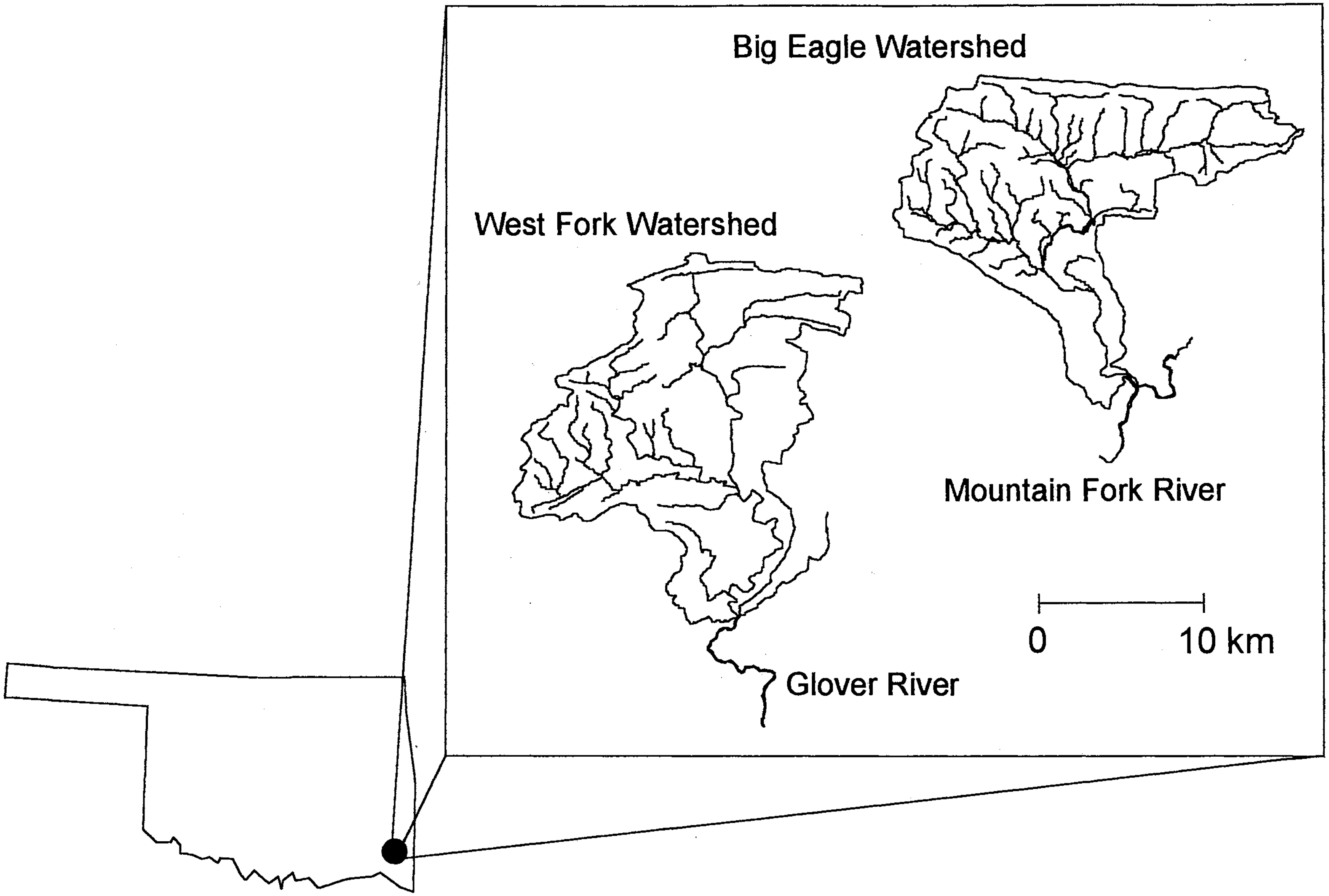
Land-use	C-factor
Row Crops	0.200
Pastureland	0.040
Pastureland-Brushy, Canopy >20%	0.039
Clearcut/Reforested[1]	0.120
Shortleaf Pine, >70% Pine	0.003
Shortleaf Pine/Oak, Mixed Forest	0.003
Oak/Hickory, >70% Deciduous	0.003
Bottomland Hardwoods	0.003
Postoak/Blackjack Oak	0.003
Native Pasture	0.015
Confined Feeding Area	0.070
Urban/Built-Up Land	0.100

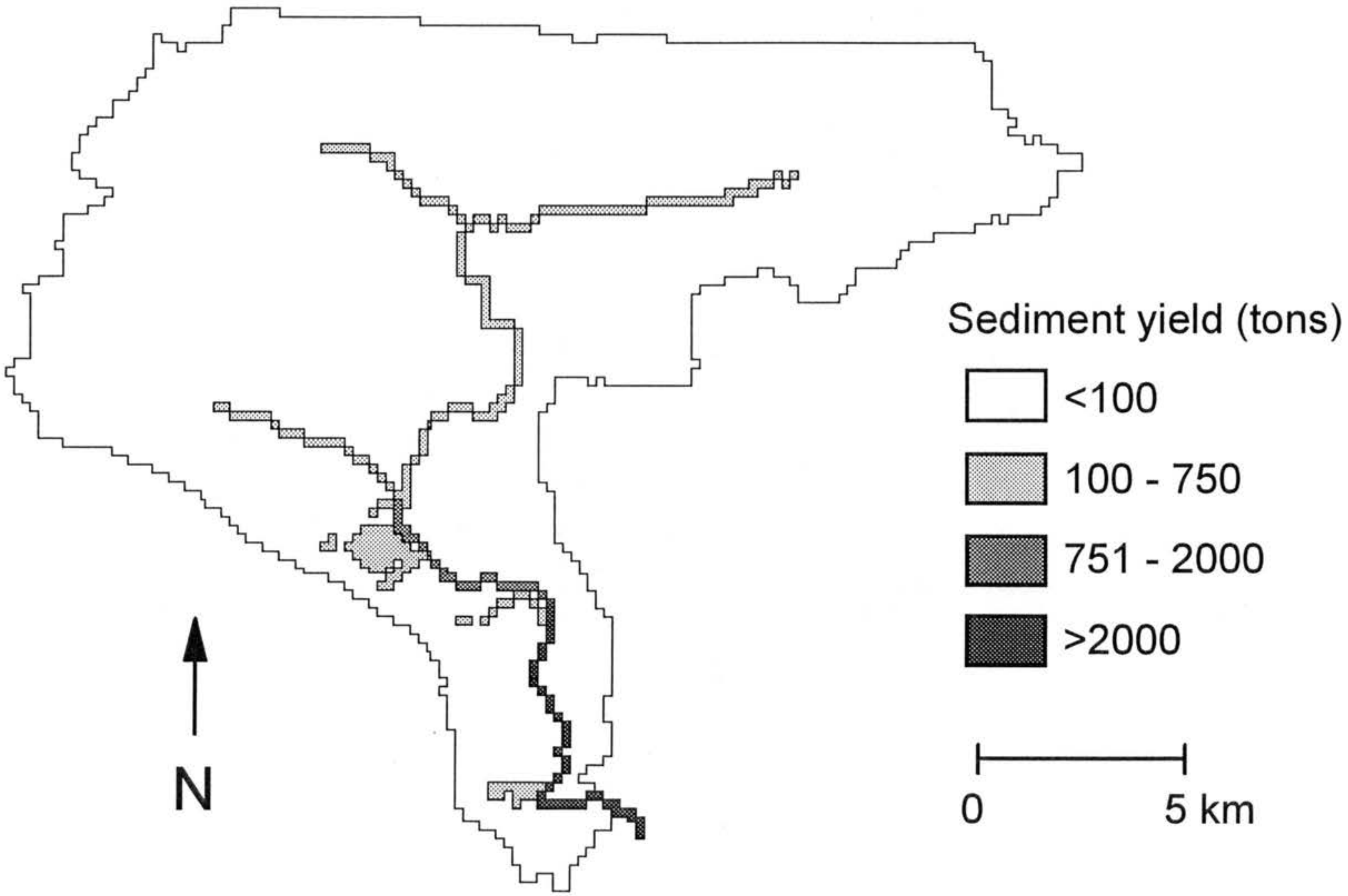
[1] Clearcut/Reforested C-factor calculated with the following subfactors:

70% bare soil/12-36 months since tillage	0.296
Canopy height 8 m/10% canopy	0.990
50% soil with fine roots	0.470
Tillage 8-12% slope/on contour	0.900

C-factor = (0.296)(0.990)(0.470)(0.900) = 0.120

List of Figures


Big Eagle Watershed
West Fork Watershed
Mountain Fork River
Glover River
0
10 km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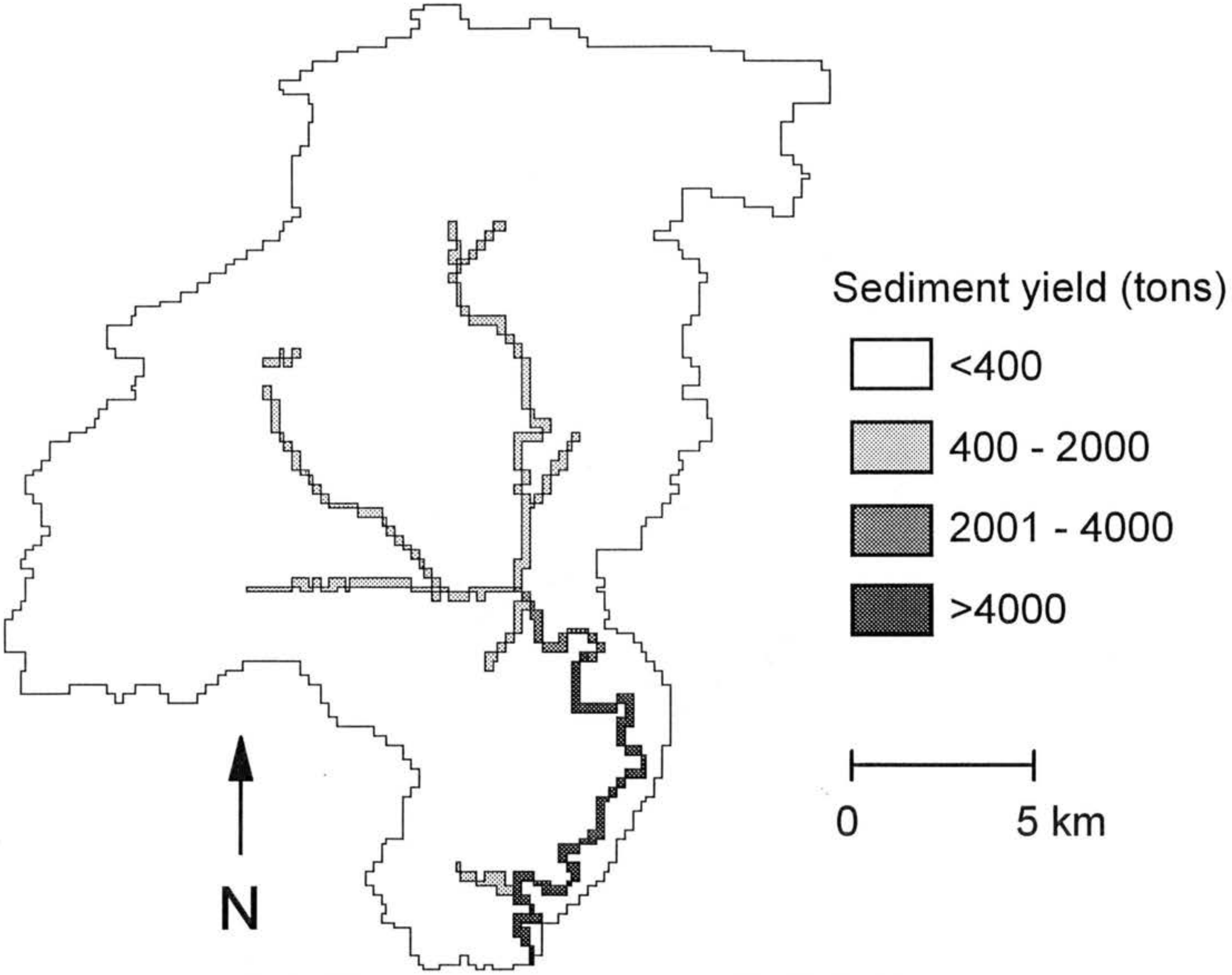
<100
100 - 750
751 - 2000
>2000
N
0
5 km


Sediment yield (tons)
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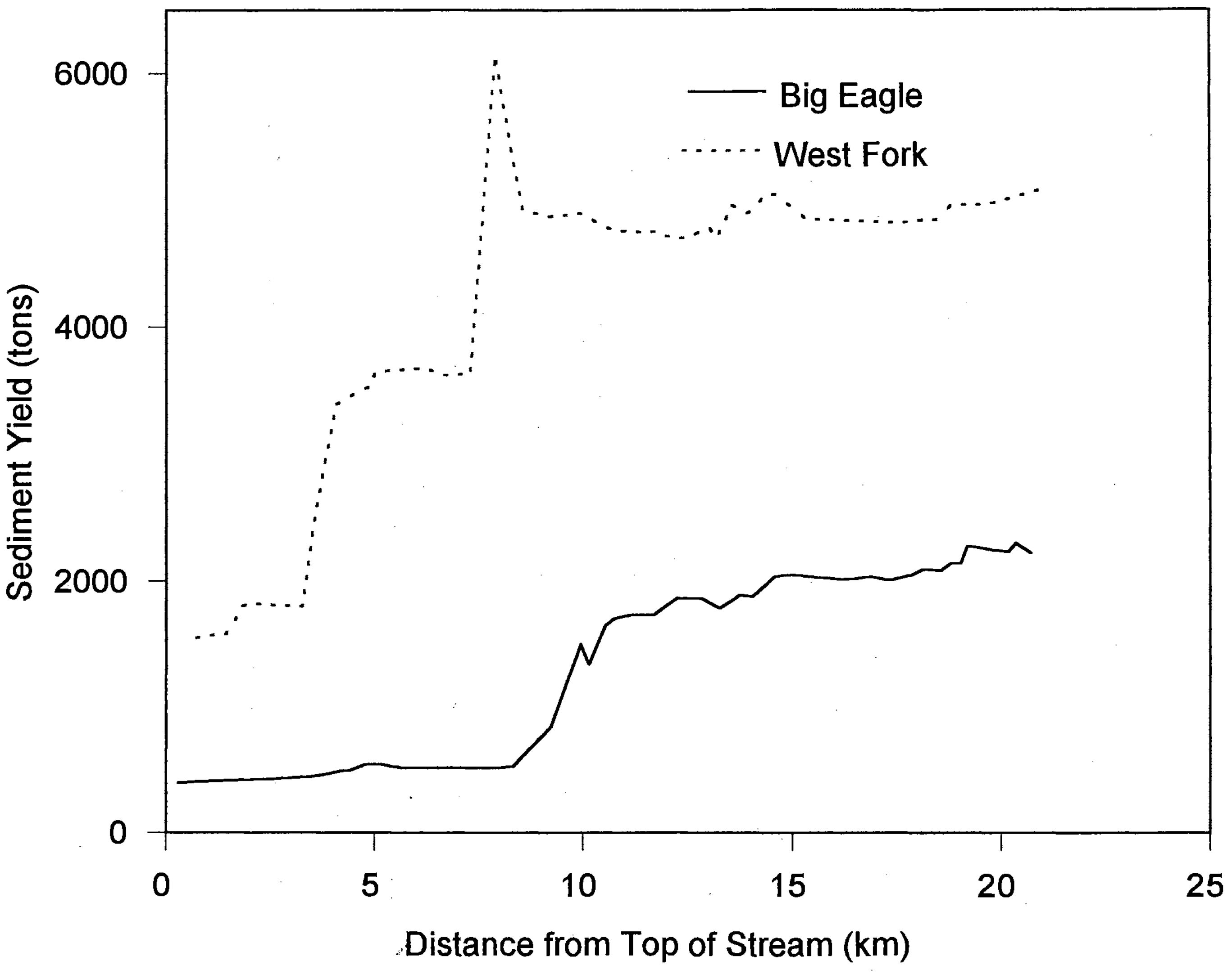
400 - 2000
2001 - 4000
>4000
0
5 km
N


Big Eagle
West Fork
Sediment Yield (tons)
6000
4000
2000
0
0
5
10
15
20
25
Distance from Top of Stream (km)

Appendix 1. Classification of mesohabitats in Big Eagle Creek, Oklahoma. Ratings are based on percentage of transect points classified as suitable or unsuitable. The total area for each mesohabitat type is for each entire stream.

Mesohabitat Type	Total Number	Number Sampled	Percent Suitable	Percent Unsuitable	Suitability	Area (ha)
Low Gradient Riffle	68	15	25	75	Unsuitable	7.67
High Gradient Riffle	26	5	28	72	Unsuitable	4.24
Cascade	7	3	0	100	Unsuitable	0.44
Secondary Channel Pool	1	1	58	42	Optimal	1.02
Backwater Pool	3	2	59	41	Optimal	1.02
Lateral Scour Pool	12	4	45	55	Suitable	2.67
Midchannel Pool	45	9	57	43	Optimal	34.47
Glide	13	4	73	27	Optimal	3.56
Run	32	5	30	70	Suitable	8.95
Step Run	1	0	0	0	NA	0.16
Total	208	48				64.20

Appendix 2. Classification of mesohabitats in West Fork Glover River, Oklahoma. Ratings are based on percentage of transect points classified as suitable or unsuitable. The total area for each mesohabitat type is for each entire stream.

Mesohabitat Type	Total Number	Number Sampled	Percent Suitable	Percent Unsuitable	Suitability	Area (ha)
Low Gradient Riffle	50	10	47	53	Suitable	7.74
High Gradient Riffle	2	2	29	71	Unsuitable	0.13
Backwater Pool	1	1	0	100	Unsuitable	0.11
Midchannel Pool	37	9	45	55	Optimal	44.90
Glide	12	5	64	36	Optimal	3.51
Run	19	6	77	23	Optimal	5.88
Step Run	1	1	50	50	Optimal	0.06
Total	118	34				62.33

CHAPTER VI.

CONCLUSIONS

Previous estimates of leopard darter abundance focused on individual sampling sites. Attempts to extrapolate densities to an entire stream did not account for differences in fish distribution or habitat quality along the entire length of the stream. In addition, it appeared that one-day depletion estimates during previous sampling were not indicative of actual densities at sampling locations. The method developed here accounted for variability in leopard darter density and habitat quality as a function of position within stream channels. Therefore, abundances of leopard darters appear to be about an order of magnitude higher than previous estimates.

An additional advantage of our sampling method is that we now have a better understanding of the distribution of leopard darters in two streams and should be able to more properly manage the species. For example, densities are low in upper Big Eagle Creek even though it has ostensibly high-quality habitat. Any activities in the upper watershed will likely have less of a direct effect on leopard darters than would activities in lower Big Eagle Creek where leopard darters are 10 times more dense than estimates in previous studies.

As we begin to understand more about the basic population biology of leopard darters we can start to expand to look at factors that can alter darter abundances or distributions. Although we currently do not understand movement patterns of leopard darters, it appeared that culverts at road crossings could serve as barriers to migration or dispersal under some flow conditions. In addition, clearcutting activity has led to an increase of sedimentation within West Fork Glover River. Although we do not have evidence of a direct effect, density and overall abundance of leopard darters are lower in West Fork Glover River compared to Big Eagle Creek.

One factor of leopard darter biology that is still sorely lacking is their general utilization of space in streams. We do not know if leopard darters migrate or how they utilize available areas of large units of habitat. An understanding of their spatial patterns would improve many of the analyses in this and future studies. For example, we assumed that the middle of large midchannel pools had identical densities as each end during our abundance extrapolations. If the pattern is not valid, our estimates of abundance are too high.

Much of the value of this study has been the development of new techniques. With further refinement, our abundance extrapolation method should be useful for determining abundances in other areas of the Little River drainage for which we have little to no data. The extrapolation method and the multi-scale modeling also may prove to be helpful in future management of the species. With an understanding of darter and habitat distribution, we will better be able to identify and manage the most critical areas of the drainage and may be able to identify portions of the historical range that are suitable for reintroduction efforts.

VITA

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