

ASSESSING PERSISTENCE OF TWO RARE DARTER SPECIES
USING POPULATION VIABILITY ANALYSIS MODELS

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VITA

Wendi Winter Hartup, the daughter of Richard Lee and Miriam (Werner) Winter, was born July 27, 1974 in Emporia, Kansas. She graduated from Emporia High School in 1992. She attended Cottey College, Nevada, Missouri, for two years and graduated with an Associates of Art degree in Marine Biology in August 1994. She then attended Troy State University, Troy, Alabama in 1994 and graduated with her Bachelor of Science degree in Marine Biology and a Minor in Mathematics in November 1996. In 1997, she began working for the Alabama Water Watch program, based in the Department of Fisheries and Allied Aquacultures of Auburn University. She married Chad Allen Hartup, son of Larry and Verlene (Howell) Hartup, on April 22, 2000. While working full-time, she began graduate school in the Department of Fisheries and Allied Aquacultures of Auburn University in January, 2001.

THESIS ABSTRACT
ASSESSING PERSISTENCE OF TWO RARE DARTER SPECIES
USING POPULATION VIABILITY ANALYSIS MODELS

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The objective of this study was to assess whether results predicted from population viability analysis (PVA) models resemble historical and current presence/absence data for *Etheostoma boschungi* and *Etheostoma brevirostrum*. *Etheostoma boschungi* is a migratory species and has two distinct, but adjacent habitats: non-breeding and breeding. *Etheostoma boschungi* spawn from late February to late March and are known from 30 sites in tributaries of the Tennessee River drainage of Alabama and Tennessee. *Etheostoma brevirostrum* prefers rocky runs and riffles with a fast current and spawn from early April to late May. *Etheostoma brevirostrum* is known to occur in disjunct populations located in tributaries of the Coosa River drainage and are believed to live three years. We used information on fecundity, life span, and population size to determine fertility and survival vital rates for a three-stage, pre-breeding, PVA model for

each species. The age-structured model used Leslie projection matrices to calculate deterministic population growth rates (λ) and relative elasticities of the vital rates.

The population size for *E. boschungii* decreases when adult fertility rates are below 0.896. The population size for *E. boschungii* falls below one individual after five years when $\lambda = 0.309$ for a one-batch fecundity and after five years when $\lambda = 0.322$ for a two-batch fecundity. The *E. boschungii* population size increases to more than 1,000 individuals after ten years when $\lambda = 1.158$ and after nine years when $\lambda = 1.184$ for one-batch and two-batch fecundities, respectively. The population size for *E. brevirostrum* decreases when adult fertility rates are below 0.737, 0.929, and 0.818 for the upstream, downstream, and combined populations, respectively. *Etheostoma brevirostrum* population size projected for upstream, downstream, and combined stream segments (using *E. coosae* and *E. pyrrhogaster* survivals) resulted in more than 50 times the initial population size after one year. As long as $\lambda < 1$, then the population will decrease but when $\lambda > 1$, then the population increases. The elasticity analysis of the matrix indicated the fertilities made the largest relative contribution to λ , while the survivals were smaller for all sites and for both species.

Despite some of the limitations in gathering data for rare species, we believe PVA models are useful for studying fishes. State agencies should focus on improving the habitat within each of the watersheds to improve survival of the juveniles to year one, in addition to monitoring the watershed more closely for possible land use or pollution impacts.

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INTRODUCTION

Many populations of fishes appear to be declining at rates warranting their protection. These declines could be due to factors that occur from natural disasters or anthropogenic activities. Natural factors, such as floods, drought and unseasonable water temperatures (either too cold or too warm), can displace fishes, increasing the chance of local extirpation at multiple scales (Kuehne and Barbour, 1983).

Anthropogenic activities may contribute to population declines through habitat alteration, water table recession, and surface/groundwater contamination. Habitats are often altered directly during construction of roads, ditches, and/or dams, and affected indirectly by sedimentation from urban development or logging practices (Boschung and Nieland, 1986; Allan, 1995; McGregor and Shepard, 1995; Beissinger and McCullough, 2002; Primack, 2002). Habitat alteration is usually permanent and is considered a significant cause of population decline (Boyce, 1992; Primack, 2002). Drawing from the water table for drinking purposes or agriculture practices can affect stream levels, which can mirror drought conditions in some small streams. Runoff of animal wastes from feedlots and pesticides from cropland can overload surface water with nutrients, which can cause algal blooms and low dissolved oxygen levels (Allan, 1995).

Most darters, particularly those of the genus *Etheostoma*, have short life spans (2-4 years), have limited distributions, and frequently live in restricted habitats. Fishes with limited distributions may be confined to a single stream system and/or have very few

populations. Many darters that live in restricted habitats also have water quality requirements which include high dissolved oxygen levels (> 8 ppm) with inversely low temperatures (< 10 °C). Streams with these values are indicative of near-pristine waters, as most aquatic organisms need dissolved oxygen levels over 5 ppm to thrive (Ultsch et al., 1978; Mettee et al., 1996). The vulnerability to water chemistry changes make the presence of species of *Etheostoma* an indicator of good water quality but also make them vulnerable to changes in environmental quality.

Any number of the previous factors alone or combined can be limiting to a species' survival. In order to protect a species from drastic declines in population size or eventual extinction, conservationists must determine the stability of populations under a variety of circumstances so that conservation efforts can be targeted appropriately. To do this, conservationists try to conduct comprehensive studies of a species by examining available life history information (growth, survival, and reproduction), collecting additional life history information (population size, life span, and fecundity), and monitoring existing population levels. Often policy decisions are based on an estimation of how long a population can persist and the rate at which the size of a population is changing (Boyce, 1992; Heppell et al., 2000b; Primack, 2002). These data are frequently lacking, especially for endangered species (Beissinger, 2002). Meanwhile, resource managers may have to make judgments that will affect numerous species and recommend alternatives for imperiled species, sometimes using incomplete data sets. By utilizing short-term predictions, early feedback on a management action can be evaluated so that future decisions are easier to make (Burgman et al., 1993; Fagan et al., 2001; Hanski, 2002; Primack, 2002).

One tool utilized by resource managers to determine a population's persistence is population viability analysis (PVA). A variety of computer software packages are now available that construct simple and complex models with uses ranging from determination of the risk of endangerment for a population to providing guidance on land use planning strategies (Akçakaya, 2000; Fagan et al., 2001). Detailed, complex models are not necessary if the primary variable causing population decline is loss or altered habitat, which is the situation with most species under the Endangered Species Act (Boyce, 2002). Rarely are there enough data available for endangered species to allow estimation of parameters required for complex models, however, workable models (for shorter-lived species) can still be built even when there are gaps in data for a population (Burgman et. al, 1993; Beissinger, 2002). Simple PVA models can predict dynamics of imperiled species, which may be adequate for conservation planning (Belovsky et al., 2002).

There are numerous types of PVAs with some based on deterministic and others on stochastic projections. Many PVAs utilize matrix models to estimate the likelihood that a population will persist for a certain period of time; these models can be used to provide resource management with insights into what parameters influence survival or extinction rates (Boyce, 1992; Williams et al., 1999; Ferrara, 2001; Beissinger and McCullough, 2002). In most matrix-based PVA models, males are ignored because including both sexes underestimates the risk of extinction (Brook et. al, 2000). Additionally, females are usually the limiting sex during breeding so abundances are expressed in terms of females and productivity (Burgman et. al, 1993; Brook et. al, 2000). Unless males are the

limiting sex, then vital rates of males will not influence population growth rates as significantly.

Models are constructed by first classifying individuals into age classes or life history stages (i) at time (t) to determine the size of the matrix. Leslie models are the simplest matrices and integrate age-specific survival and fertility rates (Leslie, 1945). Leslie matrices can be used to estimate the probability of reaching a critical population level and to calculate average extinction times. Population size and fecundity are used to derive estimates of fertility (F_i) and survival (P_i) rates, which are needed for each age class to be modeled. Generally, age classes are determined by annual growth (Caswell, 2001). The fertility rate of a population is the number of live births produced over some time period and is often expressed as young females produced per female in the population (Krebs, 1985). The fecundity of a population is the potential level of reproductive performance and is usually greater than fertility (Krebs, 1985). Survival rates represent the fraction of those individuals of age i that survive to be of age $i+1$ (Krebs, 1985; Burgman et al., 1993; Heppell et al., 2000a; Caswell, 2001).

These vital rates are incorporated into a matrix model to determine population growth trends. An example Leslie Matrix for four ages looks like the following 4x4 projection matrix.

$$\text{Projection Matrix } (A) = \begin{bmatrix} F_1 & F_2 & F_3 & F_4 \\ P_1 & 0 & 0 & 0 \\ 0 & P_2 & 0 & 0 \\ 0 & 0 & P_3 & 0 \end{bmatrix}$$

The projection matrix is multiplied by a population vector (the number of individuals n in each stage i at time t) to project the population through time. The vector must have the same number of rows as the matrix (see example below).

$$\text{Population Vector} = \begin{bmatrix} n_{it} \\ n_{it} \\ n_{it} \\ n_{it} \end{bmatrix}$$

If all the parameters in the model never varied, it would be deterministic. The expected population growth rate (λ) is the largest real positive (dominant) eigenvalue of the projection matrix. The left eigenvector (reproductive values) corresponds with the dominant eigenvalue and measures the value of an individual in each of the age classes based on the anticipated total number of offspring (Burgman et al., 1993). The unit sum of the right eigenvector (stable age distribution) estimates the proportion of individuals in each age class even though the total population density grows or declines (Caswell, 2001; Morris and Doak, 2002).

Sensitivity analysis has become an essential component of demographic analyses. Each element in a sensitivity matrix measures how small a change in the corresponding element of the projection matrix A would change the dominant eigenvalue, while keeping all other elements in A constant (Caswell, 1994; Morris and Doak, 2002; Wilson, 2002).

The sensitivity (s_{ij}) of λ to changes in the elements of A is defined by:

$$s_{ij} = \frac{\partial \lambda}{\partial a_{ij}} = \frac{v_i w_j}{\langle \mathbf{w}, \mathbf{v} \rangle}$$

where i refers to the row and j refers to the column of the matrix element a , w and v are the left and right eigenvectors, respectively of A corresponding to λ , and the $\langle \mathbf{w}, \mathbf{v} \rangle$

denotes the scalar product of w and v (Caswell, 1994, 2001). The most common form of sensitivity analysis is elasticity analysis or proportional sensitivities (Caswell, 1994; de Kroon et al., 2000). Elasticities are rescaled sensitivities to account for the magnitude of λ and the matrix element (Caswell, 2001):

$$e_{ij} = s_{ij} \frac{a_{ij}}{\lambda} = \frac{\partial \log \lambda}{\partial \log a_{ij}}$$

Simply put, the elasticity analysis of A compares the relative influence of each vital rate on λ . The largest elasticity values represent the life history stage with the strongest influence on population growth rates (Burgman et al., 1993; Mills and Lindberg, 2002). This information can be used to infer how management that affects a particular vital rate will affect the trend of λ (Heppell et al., 2000a).

Various government agencies now use PVA models to evaluate species proposed for listing under the Endangered Species Act (Boyce, 1992). For the past two decades, over 50 PVAs have aided in the conservation efforts of species of plants, tortoises, bears, and birds (McCarthy et al., 2001). For example, several age-structured models have been constructed for long-lived seabirds and resulted in adult survival rates as the most influential parameter on population growth (Cuthbert et al., 2001). Desert tortoise and sea turtle models indicated diminishing the mortality of large individuals would have the greatest influence on population growth (Doak et al., 1994; Heppell et al., 2000a).

Many species commonly analyzed with PVA models are long-lived, have decades of life history information available and can easily be studied for population size, behavior and dynamics. For example, determination of the number of offspring for vertebrates is observed in the field. Few fishes have been analyzed with PVA models other than large,

more long-lived fish, like sturgeon, or more abundant fish, such as salmon and trout (Boyce, 1992; Williams et al., 1999).

Short-lived species, such as fishes, are more difficult to observe in the field. For example, to determine offspring numbers in fishes, especially *Etheostoma* species, fecundity must be determined by dissecting ovaries, staging ova into size classes, and counting the number of ova. Prior to the early 1990s, fecundity was determined from counts of all eggs produced by individual females or from guesses as to the number of ripe ova produced by females (Heins and Baker, 1993a; 1993b). Heins and Baker (1988; 1989) provided science with a standard way to examine ova and classify them by developmental stage and clutch size. Problems can arise when researchers try to compare historical published fecundity with fecundity determined by the current standard. Additionally, whether or not a fish spawns multiple times during a breeding season or just once can affect values for fecundity. There are many conflicting methods to determine if a fish spawns multiple times (Weddle and Burr, 1991). The age of fishes can be determined empirically by direct observation, statistically with length frequencies and anatomically from certain fish structures (Jearld, 1983). However, for many darters it is difficult to gather dependable estimates of age, especially if age is determined by size or growth controlled by environmental variation (Burgman et al., 1993).

Objectives

The objectives of this study were to: a) assess population levels at historic sites for *Etheostoma boschungii*, the Slackwater Darter and *Etheostoma brevirostrum*, the Holiday Darter, b) assess which parameters have the greatest relative influence on the population growth rate, c) predict possible extinction time periods and d) determine the population

trend. The results from this study will address the utility of PVA models for rare fishes and provide information that can be utilized for conservation of these fishes.

METHODS

Study species

The study species are *Etheostoma boschungi*, the Slackwater Darter, and *Etheostoma brevirostrum*, the Holiday Darter. Although these two darter species are in the same family (Percidae), their life histories differ. The following provides information on the breeding habits, historical locations, length, life span, and published population size for each species.

Etheostoma boschungi is a migratory species and has two distinct, but adjacent habitats: non-breeding and breeding. For most of the year, they live in gravel-bottomed pools of creeks where they burrow under piles of old leaf litter or detritus that accumulate where water flow is slow (Wall and Williams, 1974; Boschung, 1976a; 1976b; 1979; McGregor and Shepard, 1995). Streams inhabited during most of the year are usually 0.6-12 m wide and 0.15-2 m deep with high dissolved oxygen levels (8.1-12.4 ppm). In November, *E. boschungi* migrates approximately 3-6 km to the breeding habitat. The breeding habitat is shallow water (5-10 cm deep), which originates in seeps, boils or flooded fields that slowly runs off into adjacent streams. Once winter rains increase water levels in the streams, the darters have access to these shallow waters. *Etheostoma boschungi* spawn from late February to late March (Boschung, 1976a; 1976b; 1979). McGregor (pers. comm.) stated *E. boschungi* is opportunistic and will spawn in vegetation found in stream channels as long as there is enough flowing water to keep

eggs oxygenated, even in landscaped yards, lawns, or similar areas. In April to early May, juveniles migrate to the non-breeding habitat (Boschung, 1976a).

Historically, *E. boschungi* has been collected from 30 sites in tributaries of the Tennessee River drainage of Alabama and Tennessee (Fig. 1) (Boschung, 1976a; 1976b; McGregor and Shepard, 1995). The total population size of the species was not estimated prior to my study but a 1976 study estimated the largest population (found within the Cypress Creek watershed) to contain 3,600 individuals (Boschung, 1976a). Boschung determined that the life span of *E. boschungi* is no more than four years and they reach a maximum size of 65 mm standard length (SL) (Boschung, 1976a; Etnier and Starnes, 1993; Page, 1983). Standard length is a typical measurement for fishes and is the distance from the tip of the snout to the hypural plate. Juvenile *E. boschungi* range in size from 10-12 mm SL by early April and by the end of their first year are 30 mm \pm 2 mm SL (Boschung, 1976a). Boschung estimated fecundity to be an average of 320 ripe eggs from three specimens (Boschung, 1976a; Boschung and Nieland, 1986).

Etheostoma brevirostrum, unlike *E. boschungi*, does not migrate in order to spawn and prefers rocky runs and riffles with a fast current (Suttkus and Etnier, 1991; Johnston and Phillips, 2001). Streams in which they are found are 3-12 m wide and 7-44 cm deep. *Etheostoma brevirostrum* spawn from early April to late May.

Etheostoma brevirostrum is known to occur in disjunct populations located in tributaries of the Coosa River drainage (Suttkus and Etnier, 1991; Mettee et al., 1996; Johnston and Phillips, 2001). These populations may represent two or more taxa and the Shoal Creek population is a new species (B. R. Kuhadja, pers. comm.). This study concentrates on the population that is restricted to Shoal Creek between Sweetwater and

Whiteside Mill Lakes in the Talladega National Forest (Fig. 2). Little is known about the life history of *E. brevirostrum* but they are believed to live three years and reach a maximum size of 53 mm SL (Suttkus and Etnier, 1991). Fecundity analysis has not been conducted for *E. brevirostrum*.

Threats to each species

Etheostoma boschungi is federally listed as threatened (U.S. Fish and Wildlife Service, 1984; 1990). Their habitat and complex cycles are ultimately, what limit their abundance and distribution (U.S. Fish and Wildlife Service, 1984). Potential threats include: ditching to drain areas with shallow groundwater, urban development, surface and groundwater contamination from point and nonpoint source pollution, and seepage areas diked to form ponds (U.S. Fish and Wildlife Service, 1984). Additionally, reproduction of *E. boschungi* is dependent upon rainfall and water temperature. Rainfall must be heavy enough to cause the main channel to flood into spawning grounds and temperatures must be more than 14 °C for spawning to occur (U.S. Fish and Wildlife Service, 1984). If weather conditions lower the water temperature, then the spawning season could be shortened (Boschung, 1979).

Etheostoma brevirostrum is not federally protected; however, living within the boundaries of the Talladega National Forest does provide some protection. Every national forest has a forest plan that includes streamside management zones, watershed assessment and aquatic viability from a habitat standpoint (USDA Forest Service, 2004). In Alabama, the focus of the forest plan is to use a watershed approach for management decisions and to ensure activities conducted within the National Forest do not have negative effects on water quality, and are suitable to maintain native aquatic communities

(D. Thurmond, pers. comm). The forest plan implements State best management practices, such as streamside management zones (SMZs), to protect water quality from upslope land use practices (USDA Forest Service, 2004). Streamside management zones are areas adjacent to waterbodies and typically contain sediment filter strips to restrict ground disturbance and protect stream banks (USDA Forest Service, 2004). Since the forest plan was implemented, policies concerning clear cutting and ecosystem management have caused the numbers of acres of regeneration harvesting to decline (USDA Forest Service, 2000).

The city of Heflin, Alabama could impact the quantity of water available for aquatic species because Heflin is growing rapidly compared to the State average and there is a potential to need additional water resources from the impoundments of Shoal Creek (USDA Forest Service, 2000). It is also possible that SMZs are not always properly implemented or maintained and could introduce sediment loads and runoff to the streams (D. Thurmond, pers. comm.). Currently, the main threat to the *E. brevirostrum* population is division into two subpopulations by a small impoundment that blocks movement (Johnston and Phillips, 2001). Habitat fragmentation, reduced water quality (due to pollution and/or poor land use), drought, and unseasonable temperatures could all have an effect on spawning capabilities of either population, ultimately affecting population levels.

Field data collection

Two darter species were assessed in relation to life history traits (fecundity, number of age classes, age specific population sizes, and age specific survival rates) during the

breeding season. Data for *E. boschungi* were collected from one site in Cypress Creek and data for *E. brevirostrum* data were collected from four sites in Shoal Creek.

In November 2000, January 2001, March 2001, and March 2002 a survey for the presence of sizable populations of *E. boschungi* was conducted at 30 historical sites located throughout the Tennessee Valley, representing both breeding and non-breeding habitats (Johnston and Hartup, 2001; 2002) (Fig. 1). Two to three collectors used dip nets and 3.3 m seines to sample one to two hours for *E. boschungi* at each site, however, *E. boschungi* was present at only two sites. Darters present at the Speedway site (McGregor and Shepard, 1995) near the headwaters of Shoal Creek, Tennessee, consisted of less than ten individuals. We concluded that sampling this population further could be detrimental to their persistence in the future.

The other population, found near the headwaters of Middle Cypress Creek, appeared to be large enough to study. This particular location is a historical breeding site of *E. boschungi*, known in many publications as the Dodd site (Boschung, 1976a; 1976b; McGregor and Shepard, 1995). The darters migrate to and breed in a seepage stream created by boils or springs that gather enough water to create two small streams (each < 0.5 m wide). These tiny streams combine (1 m wide) and flow 75 m across a rural farm to join Middle Cypress Creek (Fig. 3).

A mark-recapture study was conducted during the breeding seasons of 2001 and 2002 for *E. boschungi* at the Dodd site. Fish were collected for 45 min with dip nets and placed immediately into ice chests with aerated stream water. Each individual was sexed (male, female, or undeterminable/juvenile; Wall and Williams, 1974), measured with a

caliper to the nearest 0.01 mm SL, clipped on the top portion of the caudal fin (Murphy and Willis, 1996), and then released. Attempts at recapture occurred within a week.

Etheostoma brevirostrum were collected during the 2001 and 2002 breeding seasons in Shoal Creek, Alabama, from two sites above and two sites below Highrock Lake (Fig. 4). Transect data (measurements of riffle length, three widths and three depths per site) were collected from each site for use in determining stream volume (length x mean width x mean depth). Two to three collectors used 3.3 m seines for up to two hours at each site. Each site was sampled with at least six passes of seining or until no more *E. brevirostrum* were found. Not enough specimens were available for mark-recapture at all sites, so *E. brevirostrum* were measured with a caliper to the nearest 0.01 mm SL, sexed (male, female, or undeterminable/juvenile; Suttkus and Etnier, 1991) and released. I assumed all available fish were caught.

Laboratory procedures and data collection

To reduce population impacts, I attempted to use museum specimens for fecundity analysis (this requires sacrifice of females). However, existing *E. boschungii* specimens from the University of Alabama were not useable. No large series of museum species collected during the breeding season was available for *E. brevirostrum*. Therefore, I generated original data for fecundity of both species.

Specimens were fixed in 5% buffered formalin and then transferred to water. The SL of each specimen was measured to the nearest 0.1 mm with a digital caliper. Each specimen was weighed to the nearest 0.01 g. The right ovary was then removed from the specimen, air dried, and weighed to the nearest 0.01 g.

Fecundity was determined by dissecting the right ovary and classifying ova into one of six developmental stages described by Heins and Baker (1988): ripe (RE), ripening (MR), mature (MA), late maturing (LM), early maturing (EM), and latent (LA). All ova were classified by developmental stage under a dissecting microscope and counted. Ten ova were measured (digital caliper under a dissecting microscope) for each of the three largest size classes (RE, MR, and MA). Although the RE developmental stage is sometimes present, it is usually excluded from fecundity analysis because it is difficult to determine if any eggs have been oviposited or not (Heins and Baker, 1989; 1993a). The most important developmental stages of ova for determining fecundity are the MR and MA stages. Specimens were vouchered in the Auburn University museum collection.

The number of ova per batch (developmental stage) or batch fecundity spawned during the breeding season remains unpublished for both of these species. Some species are multiple spawners, meaning they spawn more than one batch of ova per breeding season. The average number of ova in the MR developmental stage is called one-batch fecundity. Two-batch fecundity is the average number of MR ova plus the average number of MA ova. For *E. boschungii*, we compare models using one-batch and two-batch fecundities. The spawning period for this species is short, and it is not known for any species of *Ozarka* how many clutches are spawned. Like other members of the subgenus *Ulocentra*, *E. brevirostrum* is likely a multiple spawner (O'Neil, 1981; Page and Madden, 1981; Erickson and Mahan, 1982; Carney and Burr, 1989; Weddle and Burr, 1991; Johnston and Haag, 1996). I constructed all models for *E. brevirostrum* with two-batch fecundities.

To determine age classes for each species, we used histograms of length frequencies. For *E. boschungii*, I used all historical length data in addition to what we collected to determine size ranges for each age class. Historical length data are not available for *E. brevirostrum*, so only data collected during this study were used for length-frequency analysis.

To estimate the population size for each age class (n_{it}) of *E. boschungii*, we used mark-recapture data from females captured during the study. The adjusted Petersen estimate (Chapman, 1951) was used to calculate population size, where m is the number of females captured and marked in the initial sample; c is the total number of females captured in a subsequent sample; and r is the number of marked females recaptured in c .

$$n_{it} = \frac{(m+1)(c+1)}{(r+1)} \quad (1)$$

Equation (1) represents the number of females in the population at time t , given that no changes have occurred in the population size and that adequate mixing has occurred between sampling periods. This formula provides an unbiased estimate for smaller populations (Ricker, 1975). Ricker (1975) recommends replacing r in equation 1 with variables from a Poisson distribution chart to calculate confidence limits. The estimated population size is then multiplied by the distribution of each age class.

Transect data from *E. brevirostrum* sampling were separated into three data sets to determine the volume of the riffles because we wanted to compare population levels from two sites above and two sites below Highrock Lake as well as with a hypothetical population that combines data from all sites. The density of fish/ m³ was estimated by

dividing the number of females per age class (n_i) by riffle volume (equation 2) (Gordon et. al., 1992; U. S. Environmental Protection Agency, 1997).

$$\text{Density of Fish/ m}^3 = n_i / \text{riffle volume} \quad (2)$$

Three segments of Shoal Creek were measured in meters with computer software (MAPINFO PROFESSIONAL, version 6.0, MapInfo Professional Corporation, Troy, NY, 2000, unpubl.). Tributaries were not included in the measurements because results from a 2001 movement study in Shoal Creek showed *E. brevirostrum* had access to many of the tributaries but did not utilize them (Johnston and Phillips, 2001). Therefore, only the mainstem of Shoal Creek (from Sweetwater Lake to Whiteside Mill Lake) was used for these measurements. Measurements were separated into the following segments to correspond with each of the models: upstream of Highrock Lake to Sweetwater Lake; downstream of Highrock Lake to Whiteside Mill Lake; and from Sweetwater Lake to Whiteside Mill Lake. The last measurement included the length of Highrock Lake.

To estimate the population size for each age class of *E. brevirostrum*, we multiplied density of fish/ m³ in equation 2 by the estimated stream segment lengths measured with MapInfo (equation 3).

$$\text{Total population for each age class} = (\text{equation 2})(\text{stream segment length}) \quad (3)$$

Analysis of data

The mean size, standard deviation (SD), size range and population 95% confidence limits (95% CL) were calculated for all field data. For laboratory data, the mean size and number of ova, SD, and range of size and number of ova per female per developmental stage were calculated for both species.

The age-based matrix model

First we constructed a life history table using the following basic ecology formulas (Krebs, 1985):

$$n_{i+1} = n_i - d_i \quad (4)$$

$$q_i = d_i / n_i \quad (5)$$

$$P_i = n_{i+1} / n_i \quad (6)$$

$$L_i = (n_{i+1} + n_i) / 2 \quad (7)$$

$$T_i = \sum_i^{\infty} L_i \quad (8)$$

$$e_i = T_i / n_i \quad (9)$$

$$F_i = (P_0)(f_i) / 2 \quad (10)$$

where i = age interval, n_i = number of females at start of age interval i , P_i = proportion of organisms surviving to start of age interval i , d_i = number dying during the age interval i to $i+1$, q_i = rate of mortality during the age interval i to $i+1$, L_i = number of individuals alive on average during the age interval i to $i+1$, T_i = cumulative sum of L_i in units of individuals times time units, e_i = mean expectation of life for organisms alive at start of age i , F_i = fertility rate or number of female offspring per female aged i per year, P_0 = proportion of organisms surviving to start of age one and f_i = batch fecundity (number of eggs produced per batch). Fertilities of all mature ages classes are assumed to be equal.

Since this was a breeding census for both species, we did not collect enough juvenile to one-year-old data to calculate survival. Although some of the historical data showed population levels for *E. boschungii* (3,600 individuals; Boschung, 1976a), these were not

distributed by age. Individuals up to one-year-old could not be determined. We collected survival data on closely related species to *E. boschungii* and *E. brevirostrum* and substituted the P_0 of sister species to determine F_i . We also estimated instantaneous annual mortality rates (Z) from the fishery analysis software package, FAST (SLIPKE, J. W. AND M. J. MACEINA. 2000. Fishery Analysis and Simulation Tools, version 2.1. Auburn University, AL.). Total annual mortality rates are derived by adding fishing mortality (F^*) to natural mortality (M). Fishing mortality is the result of fish harvesting and would be zero for both our species because darters are not usually harvested. Natural mortality is the result of old age, predation, parasites, diseases, and abiotic causes. For our purposes, Z is equal to M . FAST calculates M from a selection of formulas (Pauly, 1980; Hoenig, 1983; Peterson and Wroblewski, 1984; Chen and Watanabe, 1989; Jensen, 1996; Quinn and Deriso, 1999), which is then converted to annual survival rates with the following formula:

$$P = e^{-Z} = e^{-M} \quad (11)$$

PopTools was used to construct the models used in this study (POPTOOLS: software for the analysis of ecological models, version 2.5.9., G. M. Hood, <http://www.cse.csiro.au/poptools/>, 2003, unpubl.). PopTools is an add-in to the computer program, Microsoft Excel, which has tools for analyzing a Leslie projection matrix, an elasticity analysis of the projection matrix, and a population projection. We constructed a three-stage, pre-breeding, birth-pulsed population model (Fig. 5). The three age classes used in our model for each species include one-year-old adults (n_1), two-year-old adults (n_2), and three-year-old adults (n_3). We assumed for the model that these darters at birth are zero years old until one year later, at which time they have the capability to

reproduce. The PVA matrix model assumes the fish live three years, no migration occurs into or out of a population, sampling was conducted just before breeding season, and n_3 is the last reproductive stage (individuals die after they breed at this age).

The fertility and survival rates are put into a 3x3, Leslie projection matrix, which is deterministic. I used the basic analysis function in PopTools (Hood 2003) to calculate λ (expected population growth rate), the stable age distribution (right eigenvector) and reproductive value (left eigenvector) for the deterministic projection matrix.

I could not estimate the fertility rates for the matrix directly from the data we collected because we did not sample juvenile survival. Sister species of *E. boschungii* in the subgenus *Ozarka* lacked juvenile survival as well. Some of the sister species for *E. brevirostrum* in the subgenus *Ulocentra* had published juvenile survival rates, which I substituted into equation 10 for *E. brevirostrum* fertility rates.

In order to estimate fertility rates for *E. brevirostrum* and *E. brevirostrum*, I used instantaneous survival rates for each age class from my study as well as estimated survival rates from FAST in the Leslie projection matrix. I then set the SOLVER tool in Windows Excel to $\lambda = 1$. The SOLVER tool calculates what the fertility rates should be with the provided survival rates when $\lambda = 1$. I then used the elasticity function in PopTools to conduct an analysis of the matrix to assess which parameters have the greatest relative influence on the population growth rate. I also projected the population over time to determine when and if the population approaches one and to determine the population trend.

RESULTS

Field Data

Etheostoma boschungii --In 2001 a total of 274 *E. boschungii* (152 males: 26-50 mm SL; \bar{x} SL = 34 mm; SD = 4.45; and 122 females: 27-50 mm SL; \bar{x} SL = 34 mm; SD = 4.70) were collected from the Dodd site on Middle Cypress Creek, Tennessee. The overall sex ratio, 0.8 females:1 male, was not significantly different ($\chi^2 = 3.28$, $p > 0.05$). In 2002 a total of 79 *E. boschungii* (21 males: 38-49 mm SL; \bar{x} SL = 43 mm; SD = 2.80; and 58 females: 32-52 mm SL; \bar{x} SL = 42 mm; SD = 4.86) were collected from the Dodd site on Middle Cypress Creek, Tennessee. The overall sex ratio, 2.76:1, was significantly female biased ($\chi^2 = 17.33$, $p < 0.001$).

Length frequencies were used to determine age classes for *E. boschungii* (Fig. 6). Data for Fig. 5A were from University of Alabama museum specimens collected in the 1970s (unpubl.). Data for Fig. 5B were from a Geological Survey of Alabama study in the 1990s (McGregor and Shepard, 1995) and data for Fig. 5C were collected in 2001 and 2002 (Johnston and Hartup, 2001; 2002). Historical collections depicted in Fig. 5A and Fig. 5B was sampled for other purposes than population abundance. The color scheme for the graphs depicts the pre-breeding time period (blue, red and yellow), where only adults are present and the post-breeding period (green), where mostly juveniles are present with very few adults. Based on these graphs, the data indicates the following four

age classes: juveniles (n_0) = <26 mm SL, one-year-old adults (n_1) = 27-39 mm SL, two-year-old adults (n_2) = 40-48 mm SL and three-year-old adults (n_3) = >49 mm SL.

Mark-recapture data from 2001 resulted in an estimated female population size of 234 (95% CL = 145-398), a male population size of 331 (95% CL = 209-552) and a total population size for the Dodd Site at 581 individuals (95% CL = 413-846) (Table 1). Mark-recapture data from 2002 resulted in a significantly larger female population size of 450 (95% CL = 136-818; $\chi^2 = 68.21$, $p < 0.001$), a significantly smaller male population size of 63 (95% CL = 19-115; $\chi^2 = 182.3$, $p < 0.001$) and a total population size for the Dodd Site at 545 individuals (95% CL = 199-1362) that was not statistically different.

The 2001 mark-recapture census resulted in the following mean number of individuals in each age class: $n_1 = 48$, $n_2 = 4$ and $n_3 = 2$. The proportion of individuals per age class resulted in a decreasing population size, as the fish become older (Table 2). The 2002 mark-recapture census resulted in the following mean number of individuals in each age class: $n_1 = 9$, $n_2 = 17$ and $n_3 = 3$, which resulted in a population size that does not decrease as the fish grow older.

Etheostoma brevirostrum -- In 2001 a total of 143 *E. brevirostrum* were collected from four sites on Shoal Creek. Exactly 45 males (35-49 mm SL; \bar{x} SL = 42 mm; SD = 3.51) and 41 females (30-48 mm SL; \bar{x} SL = 39 mm; SD = 3.66) were collected from the two sites upstream of Highrock Lake. A total of 57 *E. brevirostrum* (29 males: 34-44 mm SL; \bar{x} SL = 39 mm; SD = 2.46; and 28 females: 33-44 mm SL; \bar{x} SL = 37 mm; SD = 2.60) were collected from the two sites downstream of Highrock Lake.

In 2002 a total of 74 *E. brevirostrum* were collected from the same four sites on Shoal Creek. Eight males (37-49 mm SL; \bar{x} SL = 43 mm; SD = 3.96) and 9 females (24-45 mm SL; \bar{x} SL = 37 mm; SD = 7.42) were collected from the two sites upstream of Highrock Lake. A total of 57 *E. brevirostrum* (21 males: 36-46 mm SL; \bar{x} SL = 40 mm; SD = 2.99; and 36 females: 22-50 mm SL; \bar{x} SL = 38 mm; SD = 4.49) were collected from the two sites downstream of Highrock Lake. None of the sex ratios for 2001 was significantly different from a 1:1 relationship, which would be expected for most darters. In 2002, the sex ratio, 1.7:1, was significantly female biased ($\chi^2 = 3.95$, $p < 0.05$) for the downstream sites of Highrock Lake but was not statistically different for the upstream sites of Highrock Lake or for Shoal Creek.

Length frequencies were used to determine age classes for *E. brevirostrum* (Fig. 7). Collection data were from a 2001 movement study (Johnston and Phillips, 2001) and our 2002 collection in Shoal Creek. The graph indicates four age classes: juveniles (n_0) = <30 mm SL, one-year-old adults (n_1) = 31-40 mm SL, two-year-old adults (n_2) = 41-45 mm SL, and three-year-old adults (n_3) = >46 mm SL.

Riffle volume determined from transect data resulted in 198 m³ for the sites upstream of Highrock Lake, 174 m³ for the sites downstream of Highrock Lake and 372 m³ for all sites combined (Table 3). The lengths for segments of Shoal Creek mainstem were determined as follows: upstream of Highrock Lake to Sweetwater Lake= 3.8 km, downstream of Highrock Lake to Whiteside Mill Lake= 2.5 km, and from Sweetwater Lake to Whiteside Mill Lake = 6.5 km.

Although the number of fish collected per sampling season ranged from 2-54 individuals per age class in 2001 and 1-32 individuals per age class in 2002, the density of fish per stream section resulted in < 0.2 fish/m³ for all sites in 2001 and 2002 (Table 4). We did not use the juvenile data in the extrapolated populations because the models are set up as pre-breeding censuses. The distribution of population size for each age class in 2001 showed a decrease in number as the fish became older for both populations upstream and downstream of Highrock Lake. In 2002, the extrapolated population of the upstream segment did not decrease, as the fish grew older. The upstream population size with 732 *E. brevirostrum* in 2001 was almost double that of the downstream population size with 400 *E. brevirostrum*. In 2002, the upstream population of 135 *E. brevirostrum* was much less abundant than those collected in 2001 and was smaller than the 2002 downstream population. The downstream population size increased slightly from 2001 to 2002 with 400 *E. brevirostrum* to 498 *E. brevirostrum*.

Laboratory Data

Etheostoma boschungii --Ten females (40.6 – 48.8 mm SL; \bar{x} SL = 44.2 mm; SD = 2.53) were sampled during the breeding season for fecundity analysis. The mean number and diameter of ova in the three largest developmental size classes were, from largest to smallest: RE (\bar{x} = 8, SD = 4.56 and \bar{x} = 1.23 mm, SD = 0.09); MR (\bar{x} = 92, SD = 54.53 and \bar{x} = 0.95 mm, SD = 0.09); and MA (\bar{x} = 105, SD = 52.46 and \bar{x} = 0.73 mm, SD = 0.07). The largest egg class (RE) was present in 50% of the fish but was not used in determining fecundity. The one-batch fecundity was the mean number of ova from the

MR stage ($\bar{x} = 92$) and the two-batch fecundity was the mean number of ova from the MR stage added to the MA stage (197).

Etheostoma brevirostrum --Five females (33.1-42.8 mm SL; \bar{x} SL = 36.4 mm; SD = 3.87) were sampled for fecundity analysis. The mean number and diameter of ova in the three largest developmental size classes were, from largest to smallest: RE ($\bar{x} = 3$, SD = 1.15 and $\bar{x} = 1.34$ mm, SD = 0.19); MR ($\bar{x} = 66$, SD = 26.83 and $\bar{x} = 0.98$ mm, SD = 0.09); and MA ($\bar{x} = 109$, SD = 28.59 and $\bar{x} = 0.61$ mm, SD = 0.16). The largest egg class (RE) was present in 60% of the fish but was not used in determining fecundity. The two-batch fecundity was the mean number of ova from the MR stage added to the MA stage (175).

Vital rates

Etheostoma boschungii --We used the population size determined from 2001 data in Table 2 for each age class because the information fit in the best with the life history table (Table 5). Fertility rates could not be calculated for the basic life history table from the existing data we collected because we did not collect any information on juvenile survival. We were able to calculate survival for the PVA model for each age class of adults ($P_1 = 0.074$, CL = 0.073-0.076; $P_2 = 0.571$, CL = 0.5-0.58; $P_3 = 0$). The rate of mortality during the interval from one age to the next was $q_1 = 0.926$; $q_2 = 0.429$; $q_3 = 1.0$ and the average expectation of life for each age class was $e_1 = 0.62$ yrs; $e_2 = 1.07$ yrs; $e_3 = 0.5$ yrs.

Using the *E. boschungii* survival from Table 5, the adult fertility rates would have to be 0.896 for the population to be stable (stable age distribution: $n_1 = 0.896$, $n_2 = 0.066$, n_3

= 0.038 and reproductive values: $n_1 = 0.303$, $n_2 = 0.426$, $n_3 = 0.271$). By back-calculating with equation 10, the juvenile survival rate would be 0.019 for a one-batch fecundity of 92 ova and 0.009 for a two-batch fecundity of 197 ova. The elasticity analysis of the matrix using our calculated survival rates and the SOLVER fertility indicated the fertilities made the largest relative contribution to λ with elasticities of 0.79, 0.06, and 0.03 while survivals were smaller at 0.09 and 0.03.

None of the species closely related to *E. boschungii* had estimates for juvenile survival to year one (Table 6). Of the species closely related to *E. boschungii* that had available information, survival rates ranged from 0.396 to 0.549 ($\bar{x} = 0.38 \pm 0.23$ SD) for one-year-old adults and 0.429 to 0.855 ($\bar{x} = 0.64 \pm 0.3$ SD) for two-year-old adults. For each of these species with published survival rates, the authors assume that each age class was collected in proportion to its relative number in the population, the population is stationary, and the number of juveniles entering the population is constant. The species with the closest percentage of individuals in each age class to our data from Table 2 was *E. cragini*. Five of the closely related species die sometime before their third or fourth year, like *E. boschungii*, while *E. punctulatum* lives up to five years.

Of the closely related species from Table 6, *E. cragini* was the only species with population data compatible with our data. Using *E. cragini* survivals and setting $\lambda = 1$, results in $F_i = 0.937$ with the SOLVER tool (stable age distribution: $n_1 = 0.937$, $n_2 = 0.044$, $n_3 = 0.019$ and reproductive values: $n_1 = 0.305$, $n_2 = 0.409$, $n_3 = 0.286$). Using equation 10, results in juvenile survival rates of 0.020 and 0.010 when using one-batch and two-batch fecundities, respectively.

Annual survival rates estimated with the program FAST ranged from 0.153 to 0.368 ($\bar{x} = 0.23 \pm 0.07$ SD) (Table 7). By using the FAST annual survival rates for both P_1 and P_2 and setting $\lambda = 1$, the SOLVER tool estimated fertility rates ranging from 0.665 to 0.850 ($\bar{x} = 0.781 \pm 0.06$ SD). One-batch fecundities resulted in juvenile survival rates ranging from 0.014 to 0.018 ($\bar{x} = 0.017 \pm 0.001$ SD) and two-batch fecundities resulted in juvenile survival rates ranging from 0.007 to 0.009 ($\bar{x} = 0.008 \pm 0.001$ SD). Juvenile survival rates derived from two-batch fecundities were almost half the juvenile survival rates derived from one-batch fecundities.

The population size decreases when adult fertility rates are below 0.896 and juvenile survival rates are below 0.019 and 0.009 for one- and two-batch fecundities, respectively (Table 8). The population size falls below one individual after five years when $P_0 = 0.004$, adult fertility rates are 0.184 and $\lambda = 0.309$ for a one-batch fecundity of 92 ova (Table 8, Fig. 8). For a two-batch fecundity of 197, the population size falls below one individual after five years when $P_0 = 0.002$, adult fertility rates are 0.197 and $\lambda = 0.322$. The population size increases to 1,000 individuals after ten years when $\lambda = 1.158$ (one-batch fecundity used) and after nine years when $\lambda = 1.184$ (two-batch fecundity used). As long as $\lambda < 1$, then the population will decrease but when $\lambda > 1$, then the population increases.

Etheostoma brevirostrum -- We used the population size for each age class determined from 2001 data in Table 4 because the proportion of individuals fit in the best with the life history table (Table 9). Fertility rates could not be calculated for the basic life history table from the existing data we collected because we did not collect any

information on juvenile survival. Survival rates were highest among the population upstream of Highrock Lake and lowest from the population downstream of Highrock Lake. The average expectation of life for each age class was highest for the upstream population as well.

The SOLVER tool along with *E. brevirostrum* survivals and two-batch fecundities from Table 9 and setting $\lambda = 1$, resulted in estimated adult fertility rates of 0.737 (stable age distribution: $n_1 = 0.737$, $n_2 = 0.211$, $n_3 = 0.053$ and reproductive values: $n_1 = 0.376$, $n_2 = 0.347$, $n_3 = 0.277$) for the upstream population and a back-calculated juvenile survival rate of 0.008. A stable downstream population results in adult fertility rates of 0.929 (stable age distribution: $n_1 = 0.929$, $n_2 = 0.071$, $n_3 = 0$ and reproductive values: $n_1 = 0.350$, $n_2 = 0.325$, $n_3 = 0.325$) and a back-calculated juvenile survival rate of 0.011. For the entire Shoal Creek population, adult fertility rates would have to be 0.818 (stable age distribution: $n_1 = 0.818$, $n_2 = 0.152$, $n_3 = 0.30$ and reproductive values: $n_1 = 0.357$, $n_2 = 0.351$, $n_3 = 0.292$) to be stable with a back-calculated juvenile survival rate of 0.009. The elasticity analysis of the matrix using our calculated survival rates and the SOLVER fertility rates indicated the fertilities made the largest relative contribution to λ , while the survivals were smaller for all sites (Table 10). The highest elasticity value for fertility rates came from the downstream matrix.

Many of the annual survival rates estimated with FAST were the same for *E. brevirostrum* as for the *E. boschungii* because many of the formulas are based on size and maximum age (Table 11). Annual survivals ranged from 0.177 to 0.425 ($\bar{x} = 0.308 \pm 0.10$ SD). By using the FAST annual survival rates for both P_1 and P_2 and setting $\lambda = 1$,

the SOLVER tool estimated fertility rates ranging from 0.623 to 0.827 ($\bar{x} = 0.716 \pm 0.08$ SD). Two-batch fecundities resulted in juvenile survival rates ranging from 0.007 to 0.009 ($\bar{x} = 0.008 \pm 0.001$ SD).

Four closely related species to *E. brevirostrum* had estimates for juvenile survival to year one and die sometime before their third or fourth year, like *E. brevirostrum*, while *E. simoterum* lives up to two years and *E. zonale* lives up to five years (Table 12). Of the species that had survival information available, survivals ranged from 0.123-0.761 for P_0 ($\bar{x} = 0.54 \pm 0.29$ SD), 0-0.286 for P_1 ($\bar{x} = 0.16 \pm 0.10$ SD) and 0-0.182 for P_2 ($\bar{x} = 0.06 \pm 0.08$ SD). For each of the species with published survival rates, the authors assume that each age class was collected in proportion to its relative number in the population, the population is stationary, and the number of juveniles entering the population is constant.

Of the closely related species from Table 12, *E. raneyi* and *E. zonale* were the only species with population data for the same number of age classes but neither of them had juvenile survival. I used the published juvenile survivals from the other species to determine results for the upstream, downstream, and combined stream segments (Table 13). The juvenile survival rates used for *E. brevirostrum* from *E. coosae* and *E. pyrrhogaster* cause the fertility rates to be more than five times higher than the *E. simoterum* fertility rates. Population size projected for upstream, downstream, and combined stream segments (using *E. coosae* and *E. pyrrhogaster* survivals) resulted in more than 50 times the initial population size after one year. The calculated right eigenvectors were similar among species as were the left eigenvectors.

The downstream population size falls below one individual after six years when $P_0 = 0.004$, adult fertility rates are 0.350 and $\lambda = 0.415$ and the upstream population falls below one individual after six years when $P_0 = 0.001$, adult fertility rates are 0.079 and $\lambda = 0.254$ (Table 14). The population size for *E. brevirostrum* falls below one individual after four years when $P_0 = 0.001$, adult fertility rates are 0.088 and $\lambda = 0.224$ for the combined population. The population size increases to 1,000 individuals after eight years for the downstream population when $\lambda = 1.122$, and after three years for the upstream population when $\lambda = 1.142$. The combined population is already above 1,000 individuals when $\lambda = 1$.

DISCUSSION

Currently both species appear to be present in very low population levels within their respective watersheds. Thirty years ago, *E. boschungi* was scattered throughout the Tennessee Valley in small, disjunct populations (Wall and Williams, 1974; Boschung, 1976a; 1976b; 1979). A study in 1995 discovered new breeding/non-breeding sites, indicating a larger distribution (McGregor and Shepard, 1995). However, after assessment of population levels at these historic sites, only two populations of *E. boschungi* were found to currently exist (Johnston and Hartup, 2002) indicating within the last ten years population levels have decreased drastically. The last mark-recapture in 2002 showed the population consisted primarily of two- to three-year-old adults at the Dodd site, which has been a fairly, stable stronghold over the years. Historical data collected for *E. boschungi* was for presence/absence rather than to collect population size data. Based on these data as well as our population data, the entire population is declining. In many studies with limited data, count-based information is best viewed as a tool that provides relative measures of the health of two or more populations (Morris and Doak, 2002).

Few data exist for *E. brevirostrum* and historical population levels were not assessed. The total range of the species has been reduced to the mainstem of Shoal Creek. Johnston and Phillips (2001) suggest the two populations found in the upstream and downstream sites of Highrock Lake are at levels less than 1,000 individuals. For darters

when the number of individuals is more than 1,000, it is considered an abundant and viable population (Primack, 2002; Shaffer et al., 2002)

To assure resiliency in a population, the population needs to be large enough to avoid genetic problems, persist for extended periods of time and have the ability to respond to favorable conditions in the environment when they occur (Primack, 2002; Shaffer et al., 2002). Low numbers for both these species could be attributed to annual variation in population sizes, but sampling or data compilation error could also be factors. Both species were difficult to sample. *Etheostoma boschungi* appeared to disappear when they left the breeding habitat and migrated to the non-breeding habitat. They were difficult to find or sample outside of the breeding season. This was consistent with published monitoring of the species (Boschung, 1976a, 1976b; McGregor and Shepard, 1995). Populations could have gone into deeper pools than our equipment or collectors could handle or they could have migrated further downstream than presumed. Many of the streams containing historical breeding sites have also been altered (Johnston and Hartup, 2002). While *E. boschungi* is more difficult to find and sample outside of the breeding season, determining a way to monitor population levels year-round would provide better estimates for survival rates and population size.

Etheostoma brevirostrum live in wide streams in the strong currents of riffle mesohabitat. Large, slippery boulders make sampling a challenge. Populations could easily move into crevices of rocks too heavy for collectors to move so it is possible they exist in larger numbers than we estimated for this study. Additionally the rainy, wet season of the year is from February to June, which is when both species breed (Kuehne and Barbour, 1983). Rainfall during this time can limit chances of collecting either

species by flooding sampling sites. Collectors have to sample larger areas that are less confined since stream banks and flood plains often overflow. This situation can allow fish easier escape from collectors, transport them downstream to breed in new locations or allow the fish to become prey.

Estimates of population size were possibly underestimated for *E. boschungi*, because mark-recaptures were only conducted once per season. Multiple mark-recapture events can help provide a better picture of the population. For *E. brevirostrum*, densities from defined areas were extrapolated to the upper, lower, and combined reach of Shoal Creek occupied by the species. In doing this, we assumed each segment was one large riffle, which could make our numbers higher than in reality. To solve this, we could estimate the volume for all riffles per segment, sum the totals, and calculate density.

Estimates of fecundity for both species were based on egg counts for a one-batch spawn and two-batch spawn per female. We determined *E. boschungi* had a one-batch fecundity of 92 ova, which is comparable to the mature ova (102) of a closely related species and another member of the *Ozarka* subgenus, *E. trisella* (Ryon, 1985). However, two-batch fecundity for *E. trisella* (256) is higher than our estimate. Boschung's (1976) fecundity estimates were more comparable to the two-batch fecundity of *E. trisella* than to any of our results. We suspect Boschung's estimates represent counts of several developmental stages, resulting in larger counts than ours. We also compared a closely related species, *E. coosae*, which occurs sympatrically with *E. brevirostrum* in the Coosa River. *Etheostoma coosae* has a lower fecundity (126; O'Neil, 1981) than what we determined for *E. brevirostrum*. Historical *E. boschungi*, *E. trisella* and *E. coosae* fecundities were likely not as comparable to our results because the fecundities were

evaluated with varying ova classification methods prior to the current standard by Heins and Baker (1988).

There are many conflicting methods to determine if a fish spawns multiple times during a breeding season, which also affects the fecundity of fishes. Multiple spawning may represent a bet-hedging strategy in warm water streams (Weddle and Burr, 1991). Bet-hedging species are those that produce large numbers of eggs and invest very little parental care (Promislow and Harvey, 1990). Although the spawning period for *E. boschungi* is approximately one month, it is unlikely there is enough time to spawn multiple batches (Boschung, 1976).

Although results from the elasticity analyses of both species show fertilities make the largest relative contribution to λ , the component of equation 10 that most changes is the survival of the juveniles. This illustrates that fecundity is not the influential parameter on population growth. Therefore, if the timing of spawning was not long enough and the fecundity was not the major determinant of population persistence, it is not important whether *E. boschungi* is a single- or multiple-batch spawner. For a multiple-batch spawner, like *E. brevirostrum*, fecundity also does not contribute as much to persistence of the population.

Etheostoma boschungi has persisted as a species at some level for the last 30 years even though many of the populations have declined. It is possible the population at the Dodd site could persist for another 30 years and we could rely on the juvenile survival rate of 0.016 from Table 8. The loss of so many locations for the distribution of *E. boschungi* would appear to make it the more imperiled species. *Etheostoma brevirostrum* appears to be persisting but in small numbers. It is possible the juvenile survival rates

could be different for upstream vs. the downstream populations. If we were to rely on published juvenile survival rates of sister species, the number of individuals observed in the field ought to be much higher. It is possible this species has persisted for at least 10 years (since it was first described) and we could rely on a juvenile survival rate of 0.003 for the upstream population and 0.005 for the downstream population from Table 14. It is also doubtful the juvenile survival rate is as high as those published for other species in Table 12. Both *E. boschungii* and *E. brevirostrum* are egg attachers, which mean there is no parental care for larvae and the larvae are not protected from predation or disease (Page, 1983).

These models are fairly straightforward and with enough information about a fish species, they could be useful to resource managers in predicting possible extinction time periods and the population trend. The essential information needed for PVA models is the survival rates and population levels broken down by age classes. Journals are publishing less life history papers because many of the species appear to be the same in all aspects, however, each species is unique when analyzing survival and fertility rates. If researchers were to continue studying the life history of a species before it became rare, it might be possible to prevent population declines. The more available information about a species makes constructing models easier.

State agencies should focus on improving the habitat within each of the watersheds to improve survival of the juveniles to their first year, while monitoring the watershed more closely for possible land use or pollution impacts. This will correspond with improved population growth rates and larger population levels. More comprehensive status surveys should be conducted for each species and historical sites should be revisited on a regular

basis. Once more information is gathered, a stochastic model would be more realistic to project future population growth even though deterministic models are simpler to design and run. Stochasticity can incorporate chaos and randomness into the model, which can mimic drastic changes in the environment as well as predict the effect resource management has on the population growth rate.

Resource managers should explore new methods or techniques (captive breeding; if a method can be found that preserves genetic diversity) to introduce more native species to the population or reintroduce species back to the population. Further studies can provide more information for future modeling, which can be incorporated into management plans. More than ever the importance of public education for imperiled species should be implemented by state agencies. The public rarely has knowledge of such information as to why a species is important to the ecosystem and even to human survival. We all need water to exist and if aquatic species are indicators of the trend of our water quality, resource managers should strive to educate and provide more information about imperiled species as well as how to assist recovery. The best case scenario would be to inform the public before the species has a high probability of extinction. If enough interest is generated within a community, the public will volunteer to be involved in the planning process to help a population recover. Any one of these efforts would start the process for allowing the persistence of these species to improve but immediate conservation efforts are needed.

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TABLE 1. MARK-RECAPTURE SAMPLING RESULTS FROM THE 2001 AND 2002 BREEDING SEASONS OF *Etheostoma boschungii*. Confidence limits in parentheses are calculated with Poisson distribution.

	2001			2002		
	Females	Males	Total	Females	Males	Total
Marked (m)	67	63	130	29	8	37
Caught (c)	54	87	141	29	13	42
Recapture (r)	15	16	31	1	1	2
	(8.4-24.8)	(9.2-26)	(21-44)	(0.1-5.6)	(0.1-5.6)	(0.2-7.2)
Population (\hat{N})	234	331	581	450	63	545
	(145-397.9)	(208.6-552.2)	(413.4-845.5)	(136.4-818.2)	(19.1-114.5)	(199.3-1361.7)

TABLE 2. MARK-RECAPTURE RESULTS FOR FEMALE *Etheostoma boschungii* DATA IN 2001 AND 2002. n_0 = juveniles, n_1 = one-year-old adults, n_2 = two-year-old adults, n_3 = three-year-old adults, \hat{N} = estimated population size per age class with 95 percent confidence limits in parentheses, mean size in mm SL is followed by ± 1 standard deviation (SD) with range in parentheses.

	2001 Females			2002 Females		
	Proportion	\hat{N}	Mean Size \pm SD	Proportion	\hat{N}	Mean Size \pm SD
n_0	-	-		-	-	
n_1	89.6%	209 (129.9-356.5)	33.2 \pm 2.77 (27-39)	31.0%	140 (42.3-253.9)	36.2 \pm 1.90 (32-39)
n_2	6.6%	15 (9.6-26.3)	42.9 \pm 2.65 (40-48)	58.6%	264 (79.9-479.6)	43.7 \pm 2.41 (40-48)
n_3	3.8%	9 (5-15)	50 \pm 0 (49-54)	10.3%	47 (14.1-84.6)	50.3 \pm 1.35 (49-54)
Total		234			450	

TABLE 3. SUMMARY OF 2002 TRANSECT DATA, RIFFLE VOLUME, AND STREAM LENGTH ON SHOAL CREEK. Length, width and depth from transects were measured in meters and stream segments were measured with mapping software. US = upstream of Highrock Lake, DS = downstream of Highrock Lake, Combo = upstream and downstream of Highrock Lake combined, SL = standard length, SD = standard deviation.

	Transect Summary			Riffle Volume (m ³)	Length of Stream Segment (m)
	Length ± SD	Width ± SD	Depth ± SD		
US	159.96 ± 3.60	4.29 ± 2.24	0.29 ± 0.27	198.28	3818.5
DS	124.43 ± 3.87	6.19 ± 3.34	0.23 ± 0.20	173.64	2475.3
Combo	284.39			371.92	6531.7

TABLE 4. NUMBER OF *Etheostoma brevirostrum* COLLECTED IN THE FIELD, THE ESTIMATED NUMBER OF FISH PER M³ AND THE EXTRAPOLATED POPULATION SIZE FOR EACH AGE CLASS. Values in parentheses are 95 percent confidence limits. US = upstream of Highrock Lake, DS = downstream of Highrock Lake, Combo = upstream and downstream of Highrock Lake combined, n_1 = one-year-old adults, n_2 = two-year-old adults, n_3 = three-year-old adults, \hat{N} = extrapolated population size per age class.

		2001			2002		
		Collected	Fish/m ³	\hat{N}	Collected	Fish/m ³	\hat{N}
US	n_1	28 (27.06-28.94)	0.141 (0.136-0.146)	539 (521-557)	3 (0.39-5.61)	0.015 (0.002-0.028)	58 (7-108)
	n_2	8 (7.36-8.64)	0.040 (0.037-0.044)	154 (142-166)	4 (2.53-5.47)	0.020 (0.013-0.028)	77 (49-105)
	n_3	2 (0.14-3.86)	0.010 (0.001-0.019)	39 (3-74)	-		
DS	n_1	26 (25.31-26.69)	0.150 (0.146-0.154)	371 (361-381)	29 (28.16-29.84)	0.167 (0.162-0.172)	413 (401-425)
	n_2	2 (0.73-3.27)	0.012 (0.004-0.019)	29 (10-47)	5 (3.70-6.30)	0.029 (0.021-0.036)	71 (53-90)
	n_3	-			1	0.006	14
Combo	n_1	54 (53.40-54.60)	0.145 (0.144-0.147)	948 (938-959)	32 (31.21-32.79)	0.086 (0.084-0.088)	562 (548-576)
	n_2	10 (9.40-10.60)	0.027 (0.025-0.029)	176 (165-186)	9 (8.09-9.91)	0.024 (0.022-0.027)	158 (142-174)
	n_3	2 (0.14-3.86)	0.005 (0.000-0.010)	35 (2-68)	1	0.003	18

TABLE 5. LIFE HISTORY TABLE FOR *Etheostoma boschungii*. This table uses one-batch fecundity. All parameters are the same with the two-batch fecundity. See text for sources and definitions.

Age (yr) (i)	2001 (n_i)	95% CL		P_i	L_i	d_i	q_i	T_i	e_i (yr)	f_i	F_i
0	-										
1	209	130	357	0.074	112	194	0.926	129.00	0.62	92	
2	15	10	26	0.571	12	7	0.429	16.54	1.07	92	
3	9	5	15	0	4	9	1.000	4.41	0.50	92	
\dot{N}	234	145	398								

TABLE 6. SPECIES CLOSELY RELATED TO *Etheostoma boschungii*. P_i = survival rates for each age class, f_i = mature ova or fecundity, % = assumption that each age class was collected in proportion to its relative number in the population, that the population was neither increasing nor decreasing, and that the number of juveniles entering the population each year was constant.

Subgenus	Species	f_i	Survival and Population Proportion					References
			P_0	P_1	P_2	P_3	P_4	
<i>Ozarka</i>	<i>Etheostoma cragini</i>	380	-	0.047	0.429	0		a, e, f, g, i
				94%	4%	2%		
<i>Ozarka</i>	<i>Etheostoma pallididorsum</i>	78	-	0.539	0			a, c, e, g
				65%	35%			
<i>Ozarka</i>	<i>Etheostoma punctulatum</i>	524		0.549	0.855	0.264	0	a, d, e, g, j
				47%	26%	22%	6%	
<i>Ozarka</i>	<i>Etheostoma trisella</i>	102	-	0.396	0			a, e, g, h
<i>Fuscatelum</i>	<i>Etheostoma parvipinne</i>				0			b, e, g
<i>Psycromaster</i>	<i>Etheostoma tuscumbia</i>	78			0			a, e, g

^a Bart, Jr. and Page, 1992

^b Etnier and Starnes, 1993

^c Hambrick and Robison, 1979

^d Hotalling and Taber, 1986

^e Kuehne and Barbour, 1983

^f Labbe and Fausch, 2000

^g Page, 1983

^h Ryon, 1985

ⁱ Taber, Taber, and Topping, 1986

^j Vives, 1987

TABLE 7. ESTIMATED ANNUAL SURVIVAL RATES FROM FAST WITH *Etheostoma boschungii* DATA. F_i = adult fertility rates and P_0 = juvenile survival rate (derived from one-batch and two-batch fecundities).

Method	Estimated Survival	F_i	P_0	
			one-batch	two-batch
Chen/Watanabe, 1989	0.226	0.783	0.017	0.008
Hoenig, 1983	0.242	0.769	0.017	0.008
Jensen, 1996	0.153	0.850	0.018	0.009
Pauly, 1980	0.199	0.808	0.018	0.008
Peterson/Wroblewski, 1984	0.207	0.800	0.017	0.008
Quinn/Deriso, 1999 (used 1%)	0.215	0.792	0.017	0.008
Quinn/Deriso, 1999 (used 5%)	0.368	0.665	0.014	0.007

TABLE 8. ESTIMATED POPULATION SIZE PROJECTIONS FOR *Etheostoma boschungii*.
 F_i = adult fertility rates, f_i = one-batch (92 ova) and two-batch (197 ova) fecundity, P_i = survival rates, λ = population growth rate, and \dot{N} = total population size.

P_1	P_2	P_3	P_0	f_i	F_i	λ	$\dot{N} < 1$ after i yrs	$\dot{N} > 1000$ in i yrs
0.074	0.571	0	0.004	92	0.184	0.309	5	
0.074	0.571	0	0.002	197	0.197	0.322	5	
0.074	0.571	0	0.005	92	0.230	0.355	6	
0.074	0.571	0	0.006	92	0.276	0.400	6	
0.074	0.571	0	0.007	92	0.322	0.444	7	
0.074	0.571	0	0.003	197	0.296	0.419	7	
0.074	0.571	0	0.008	92	0.368	0.488	8	
0.074	0.571	0	0.009	92	0.414	0.533	9	
0.074	0.571	0	0.010	92	0.460	0.577	9	
0.074	0.571	0	0.004	197	0.394	0.513	9	
0.074	0.571	0	0.005	197	0.493	0.608	10	
0.074	0.571	0	0.011	92	0.506	0.621	11	
0.074	0.571	0	0.012	92	0.552	0.666	13	
0.074	0.571	0	0.013	92	0.598	0.710	15	
0.074	0.571	0	0.006	197	0.591	0.703	15	
0.074	0.571	0	0.014	92	0.644	0.755	19	
0.074	0.571	0	0.015	92	0.690	0.799	24	
0.074	0.571	0	0.007	197	0.690	0.799	24	
0.074	0.571	0	0.016	92	0.736	0.844	32	
0.074	0.571	0	0.017	92	0.782	0.889	46	
0.074	0.571	0	0.008	197	0.788	0.894	48	
0.074	0.571	0	0.018	92	0.828	0.933	79	
0.074	0.571	0	0.019	92	0.874	0.978	>100	
0.074	0.571	0	0.009	197	0.887	0.990	>100	
0.074	0.571	0	0.020	92	0.920	1.023		64
0.074	0.571	0	0.021	92	0.966	1.068		22
0.074	0.571	0	0.010	197	0.985	1.087		18
0.074	0.571	0	0.022	92	1.012	1.113		14
0.074	0.571	0	0.023	92	1.058	1.158		10
0.074	0.571	0	0.011	197	1.084	1.184		9
0.074	0.571	0	0.024	92	1.104	1.204		8
0.074	0.571	0	0.025	92	1.150	1.249		7
0.074	0.571	0	0.012	197	1.182	1.280		6
0.074	0.571	0	0.013	197	1.281	1.377		5

TABLE 9. LIFE HISTORY TABLE FOR *Etheostoma brevirostrum*. This table uses two-batch fecundity. US = upstream of Highrock Lake, DS = downstream of Highrock Lake, Combo = upstream and downstream of Highrock Lake combined. See text for sources and definitions.

US	Age (yr)	2001	95% CL		P_i	L_i	d_i	q_i	T_i	e_i	f_i	F_i
	(i)	(n_i)								(yr)		
	0	-										
	1	539	521	557	0.286	347	385	0.714	462.20	0.86	175	
	2	154	142	166	0.250	96	116	0.750	115.55	0.75	175	
	3	39	3	74	0	19	39	1.000	19.26	0.50	175	
	\dot{N}	732	666	798								

DS	Age (yr)	2001	95% CL		P_i	L_i	d_i	q_i	T_i	e_i	f_i	F_i
	(i)	(n_i)								(yr)		
	0	-										
	1	371	361	381	0.077	200	342	0.923	213.83	0.58	175	
	2	29	10	47	0	14	29	1.000	14.26	0.50	175	
	3					0	0		0		175	
	\dot{N}	399	371	427								

Combo	Age (yr)	2001	95% CL		P_i	L_i	d_i	q_i	T_i	e_i	f_i	F_i
	(i)	(n_i)								(yr)		
	0	-										
	1	948	938	959	0.186	562	772	0.814	684.68	0.72	175	
	2	176	165	186	0.197	105	141	0.803	122.68	0.70	175	
	3	35	2	68	0	17	35	1.000	17.34	0.50	175	
	\dot{N}	1159	1105	1213								

TABLE 10. ELASTICITY VALUES FOR *Etheostoma brevirostrum*. See text for sources and definitions.

	n_i	P_i	F_i	Elasticity	
				F_i	P_i
US	1	0.286	0.737	0.560	0.200
	2	0.250	0.737	0.160	0.040
	3	0	0.737	0.040	0
DS	1	0.077	0.929	0.867	0.067
	2	0	0.929	0.067	0
	3		0.929	0	0
Combo	1	0.186	0.818	0.675	0.150
	2	0.197	0.818	0.125	0.025
	3	0	0.818	0.025	0

TABLE 11. ESTIMATED ANNUAL SURVIVAL RATES FROM FAST WITH *Etheostoma brevirostrum* DATA. F_i = adult fertility rates and P_0 = juvenile survival rate (derived from two-batch fecundities).

Method	Estimated Survival	F_i	P_0
Chen/Watanabe, 1989	0.378	0.658	0.008
Hoening, 1983	0.242	0.769	0.009
Jensen, 1996	0.348	0.681	0.008
Pauly, 1980	0.425	0.623	0.007
Peterson/Wroblewski, 1984	0.177	0.827	0.009
Quinn/Deriso, 1999 (used 1%)	0.215	0.792	0.009
Quinn/Deriso, 1999 (used 5%)	0.368	0.665	0.008

TABLE 12. SPECIES CLOSELY RELATED TO *Etheostoma brevirostrum*. P_i = survival rates for each age class, f_i = mature ova or fecundity, % = assumption that each age class was collected in proportion to its relative number in the population, that the population was neither increasing nor decreasing, and that the number of juveniles entering the population each year was constant.

Subgenus	Species	f_i	Survival and Population Proportion					References
			P_0	P_1	P_2	P_3	P_4	
<i>Ulocentra</i>	<i>Etheostoma coosae</i>	88	0.556 60%	0.186 33%	0 6%			a, g, h, i, k
<i>Ulocentra</i>	<i>Etheostoma pyrrhogaster</i>	1=28.4, 2,3=144	0.761 51%	0.286 38%	0 11%			a, b, e, g, k
<i>Ulocentra</i>	<i>Etheostoma raneyi</i>	52	-	0.104 47%	0.104 6%	0		f, k
<i>Ulocentra</i>	<i>Etheostoma simoterum</i>	152	0.123 89%	0 11%				a, g, i, j, k
<i>Ulocentra</i>	<i>Etheostoma zonale</i>	40	-	0.180 82%	0.182 15%	- 3%	0	a, c, e, g, i, k
<i>Ulocentra</i>	<i>Etheostoma zonistium</i>	1=77, 2,3=128	0.728 54%	0.181 39%	0 7%			a, b, e, k
<i>Ulocentra</i>	<i>Etheostoma baileyi</i>	35				0		e, g, k
<i>Ulocentra</i>	<i>Etheostoma barrenense</i>	800				0		e, g, k
<i>Ulocentra</i>	<i>Etheostoma duryi</i>					0		e, g, i, k
<i>Ulocentra</i>	<i>Etheostoma etnieri</i>					0		e, g, i, k
<i>Ulocentra</i>	<i>Etheostoma flavum</i>					0		d, e, g, k

^a Bart, Jr. and Page, 1992

^b Carney and Burr, 1989

^c Erickson and Mahan, 1982

^d Etnier and Bailey, 1989

^e Etnier and Starnes, 1993

^f Johnston and Haag, 1996

^g Kuehne and Barbour, 1983

^h O'Neil, 1981

ⁱ Page, 1983

^j Page and Mayden, 1981

^k Porter, Cavender and Fuerst, 2002

TABLE 13. ESTIMATED FERTILITY RATES AND GROWTH RATE FROM CLOSELY RELATED SPECIES WITH *Etheostoma brevirostrum*. P_0 = juvenile survival, F_i = estimated adult fertility rate, λ = growth rate, SAD = stable age distribution and Reprod = reproductive values.

		<i>Etheostoma coosae</i>			<i>Etheostoma pyrrhogaster</i>			<i>Etheostoma simoterum</i>		
		US	DS	COMBO	US	DS	COMBO	US	DS	COMBO
	P_0	0.56	0.56	0.56	0.76	0.76	0.76	0.12	0.12	0.12
	F_i	48.61	48.61	48.61	66.58	66.58	66.58	10.76	10.76	10.76
	λ	48.90	48.69	48.80	66.86	66.65	66.76	11.04	10.83	10.94
	\dot{N} after 1 yr	35,767	19,432	56,536	48,914	26,602	77,351	8,066	4,323	12,676
SAD	1	0.994	0.994	0.996	0.996	0.999	0.997	0.974	0.993	0.983
	2	0.006	0.006	0.004	0.004	0.001	0.003	0.025	0.007	0.017
	3	0.000	0.000	0.000	0.000	0.000	0.000	0.001	0.000	0.000
Reprod	1	0.334	0.334	0.334	0.334	0.334	0.334	0.337	0.335	0.34
	2	0.334	0.334	0.334	0.334	0.333	0.334	0.335	0.333	0.34
	3	0.332	0.332	0.332	0.332	0.333	0.333	0.328	0.333	0.33

TABLE 14. ESTIMATED POPULATION SIZE PROJECTIONS FOR *Etheostoma brevirostrum*. F_i = adult fertility rates, f_i = two-batch fecundity, P_i = survival rates, λ = population growth rate, and \dot{N} = total population size.

	P_1	P_2	P_3	P_0	f_i	F_i	λ	$\dot{N} < 1$ after i yrs	$\dot{N} > 1000$ in i yrs
Combo	0.186	0.197	0	0.001	175	0.088	0.224	4	
DS	0.077	0		0.004	175	0.350	0.415	6	
US	0.286	0.250	0	0.001	175	0.079	0.254	6	
US	0.286	0.250	0	0.001	175	0.088	0.268	7	
Combo	0.186	0.197	0	0.003	175	0.263	0.429	8	
US	0.286	0.250	0	0.002	175	0.175	0.387	8	
DS	0.077	0		0.005	175	0.438	0.504	9	
US	0.286	0.250	0	0.003	175	0.263	0.492	9	
DS	0.077	0		0.006	175	0.525	0.593	11	
US	0.286	0.250	0	0.004	175	0.350	0.591	12	
Combo	0.186	0.197	0	0.005	175	0.438	0.613	14	
DS	0.077	0		0.007	175	0.613	0.682	15	
US	0.286	0.250	0	0.005	175	0.438	0.686	17	
DS	0.077	0		0.008	175	0.700	0.770	22	
US	0.286	0.250	0	0.006	175	0.525	0.779	26	
Combo	0.186	0.197	0	0.007	175	0.613	0.792	30	
DS	0.077	0		0.009	175	0.788	0.858	39	
US	0.286	0.250	0	0.007	175	0.613	0.871	47	
Combo	0.186	0.197	0	0.008	175	0.700	0.881	>50	
DS	0.077	0		0.010	175	0.875	0.946	>50	
DS	0.077	0		0.012	175	1.050	1.122		8
US	0.286	0.250	0	0.010	175	0.875	1.142		3
Combo	0.186	0.197	0	0.010	175	0.875	1.057		0

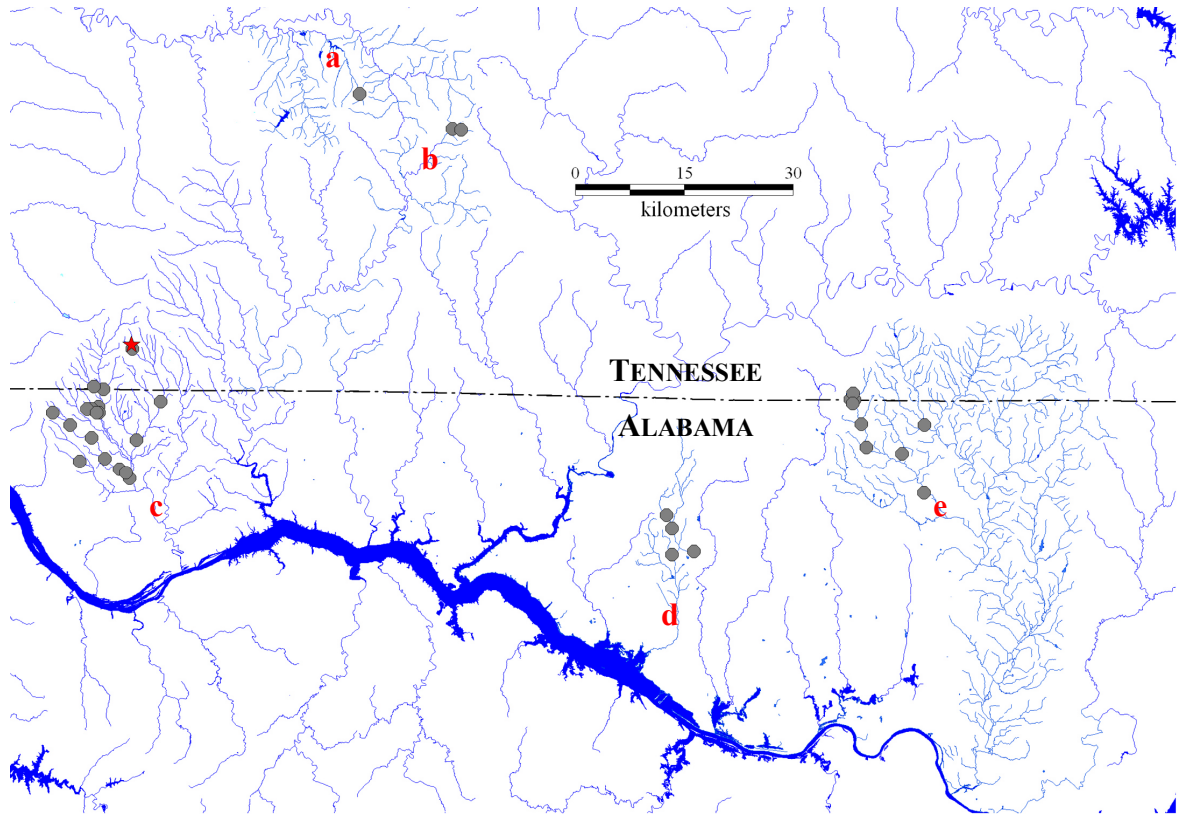


Fig. 1. Historical locations (circles) and sampling site (star) of *Etheostoma boschungii* within tributaries of the Tennessee River drainage, USA. Letters on the map correspond to the following: a) South Fork Buffalo River and Chief Creek of Buffalo River, b) Shoal Creek, c) Cypress Creek, d) Swan Creek, and e) Flint River.

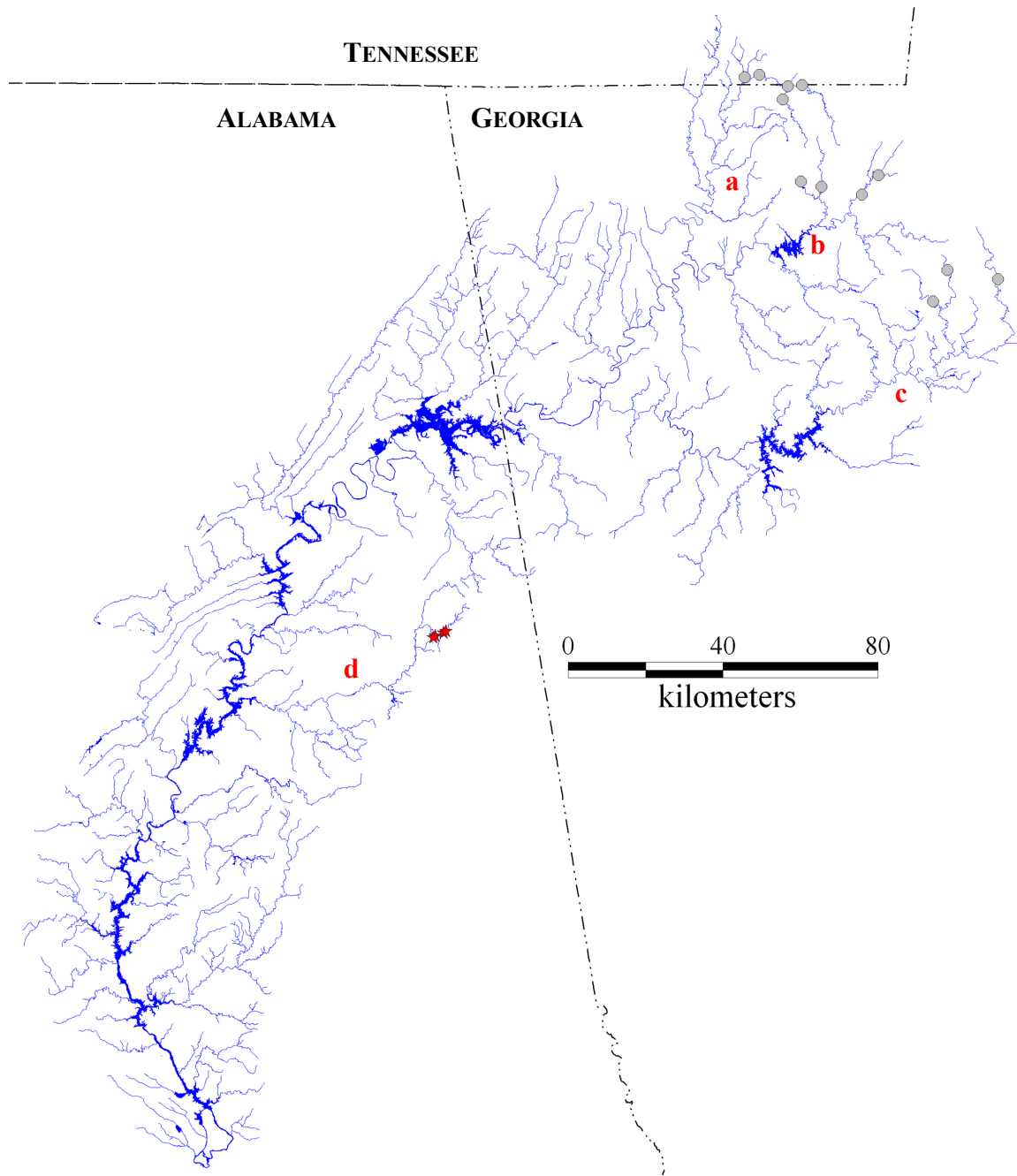


Fig. 2. Historical locations (circles) and sampling sites (star) of *Etheostoma brevirostrum* within tributaries of the Coosa River drainage, USA. Letters on the map correspond to the following: a) Conasauga River, b) Coosawattee River, c) Etowah River, and d) Choccolocco Creek.

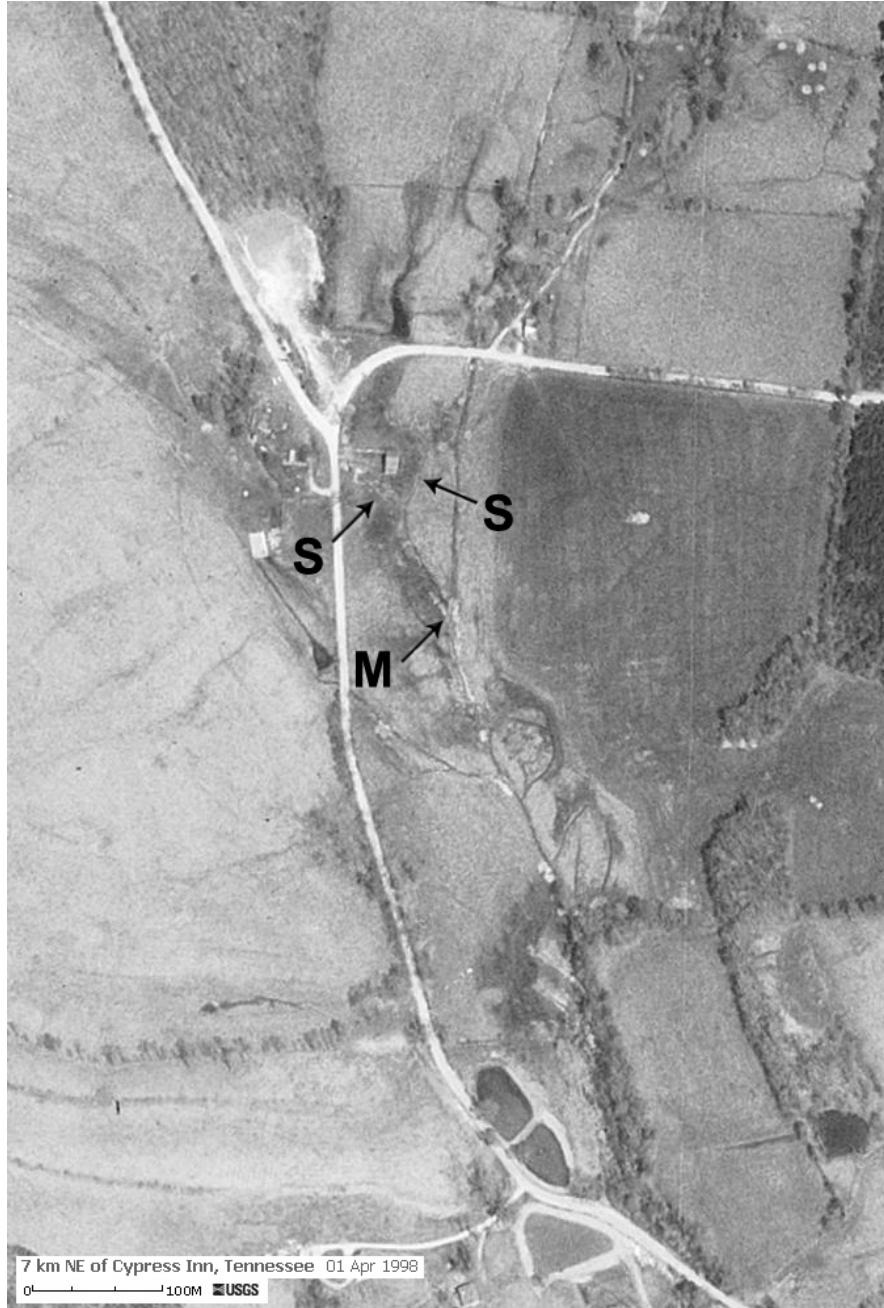


Fig. 3. Map of the Dodd Site, Middle Cypress Creek on Dodd Road, Wayne County, Tennessee (S = small seepage stream, M = area where the larger seepage stream joins with Middle Cypress Creek) (TERRASERVER-USA: An online provider of USGS digital maps [web application], <http://terraservice.net/>, 2004, unpubl.).

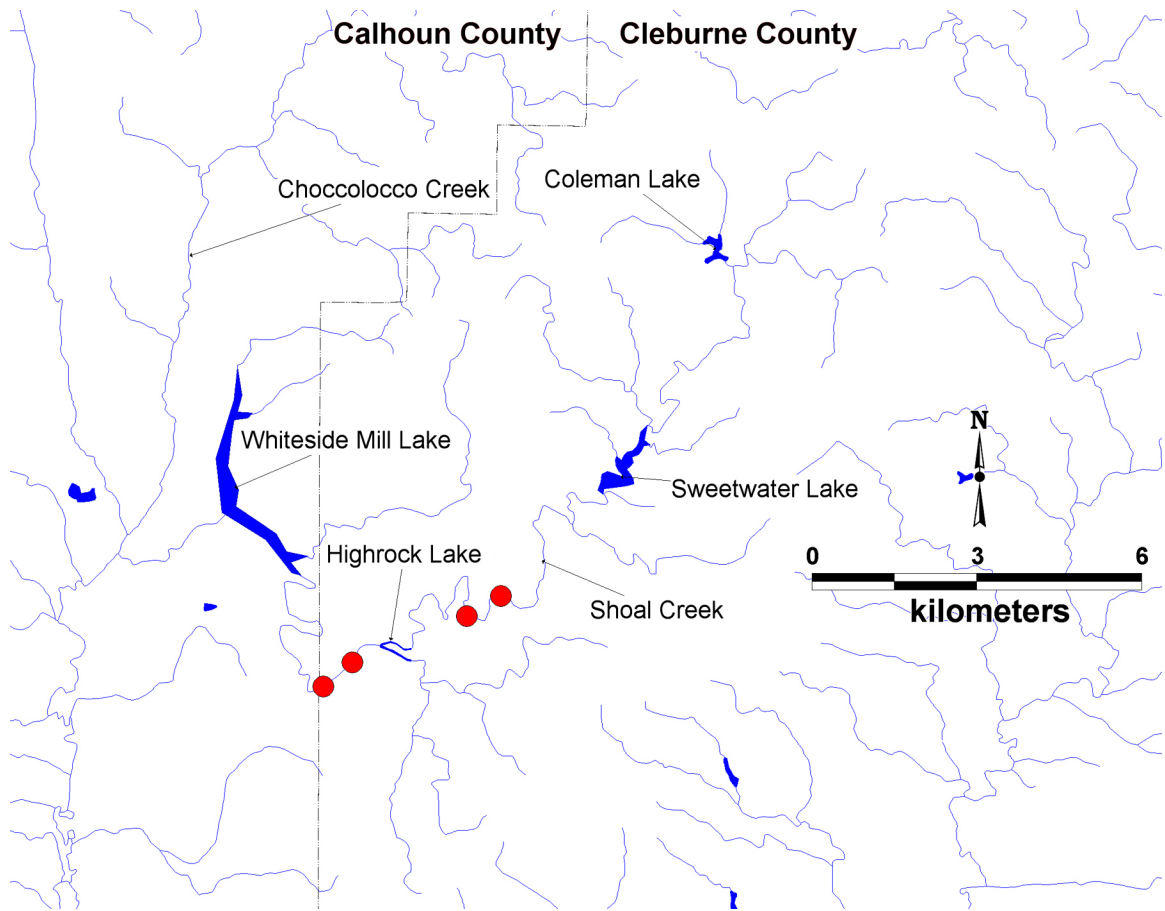


Fig. 4. Sampling locations (circles) of *E. brevirostrum* within Shoal Creek of the Coosa River drainage in Alabama, USA.

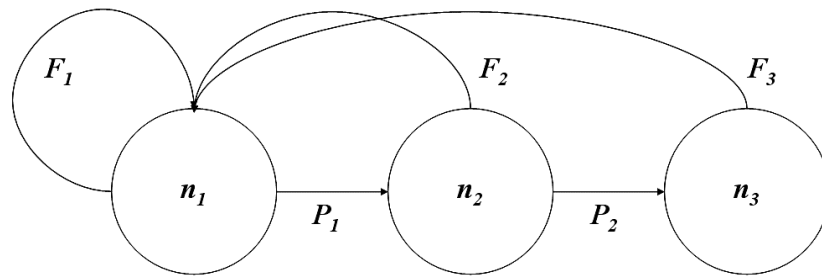


Fig. 5. A three-stage, birth-pulsed, pre-breeding model for adult populations of *E. boschungii* and *E. brevisrostrum*. Circles denote stages in the age-structured model and arrows indicate transition probabilities.

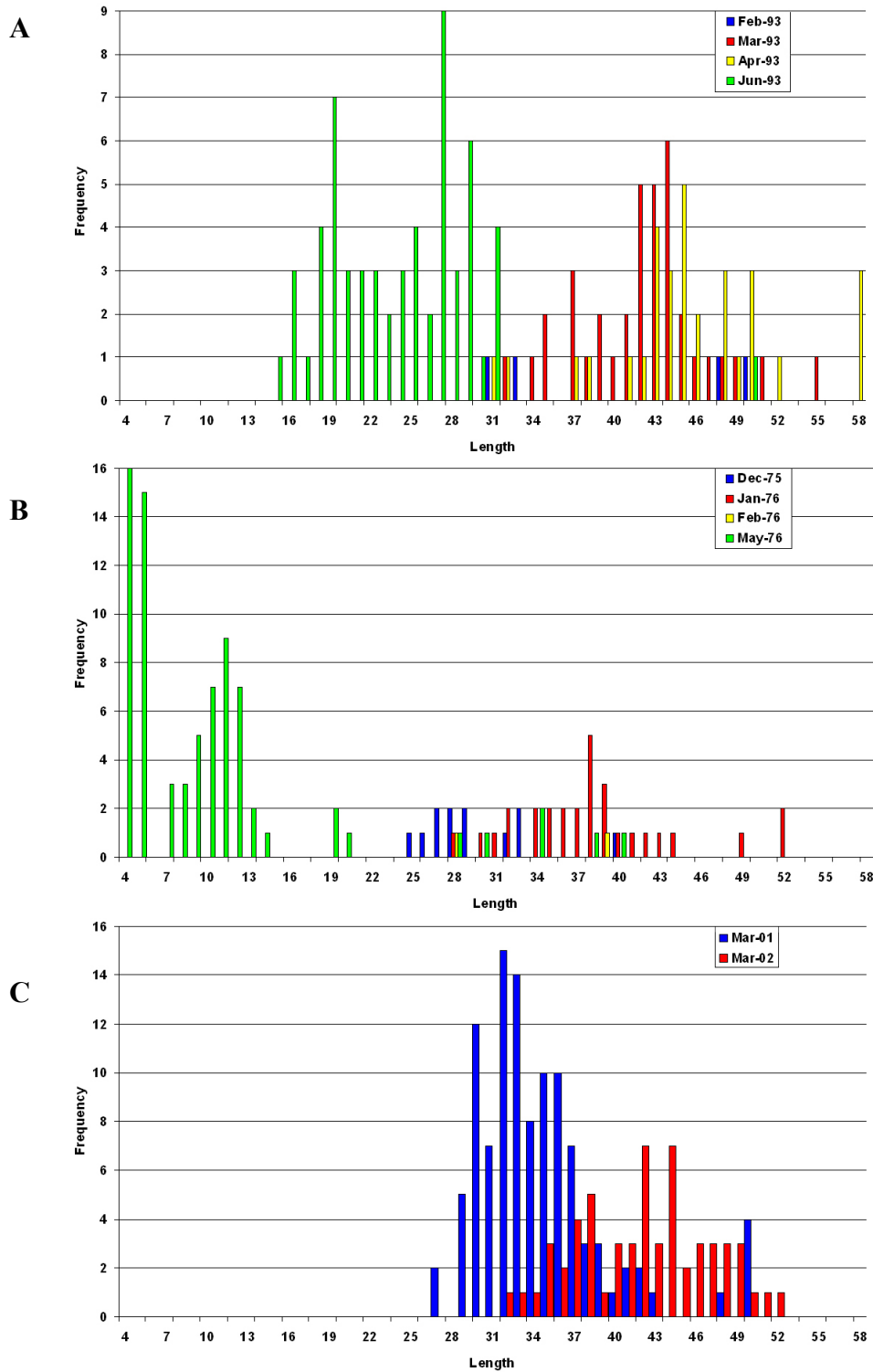


Fig. 6. Length-frequency distribution of collected *Etheostoma boschungii* from selected years. A = University of Alabama museum specimens, B = (McGregor and Shepard, 1995) C = 2001-2002 census data.

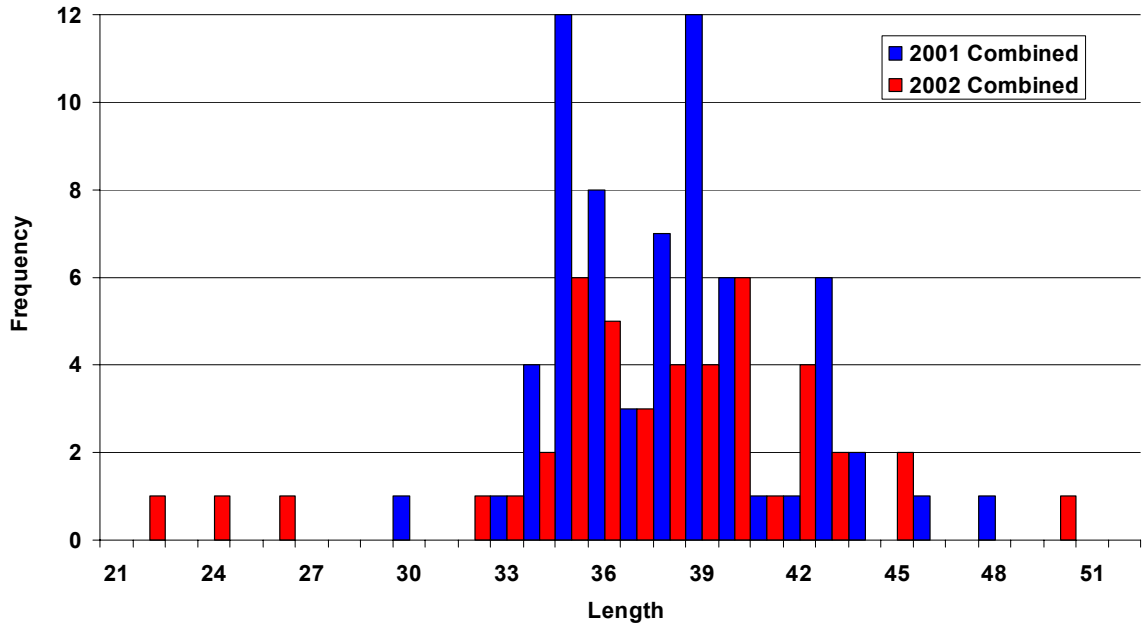


Fig. 7. Length-frequency distribution of collected *Etheostoma brevirostrum* from 2001 and 2002 in Shoal Creek.

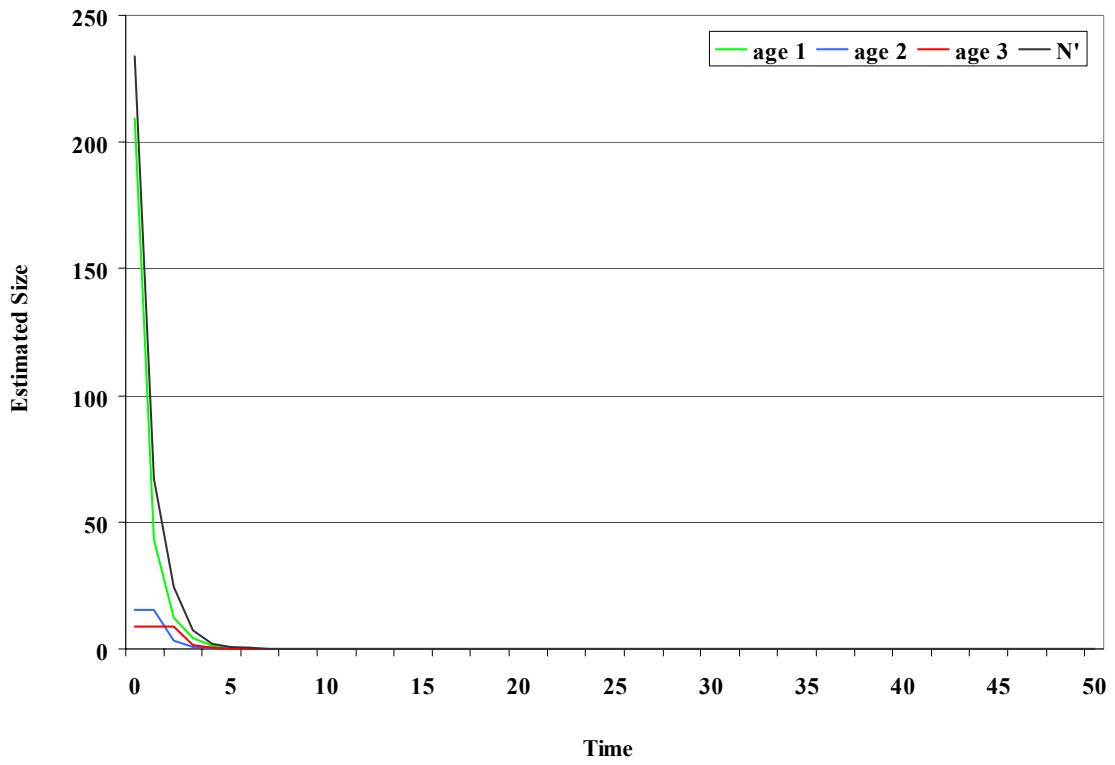
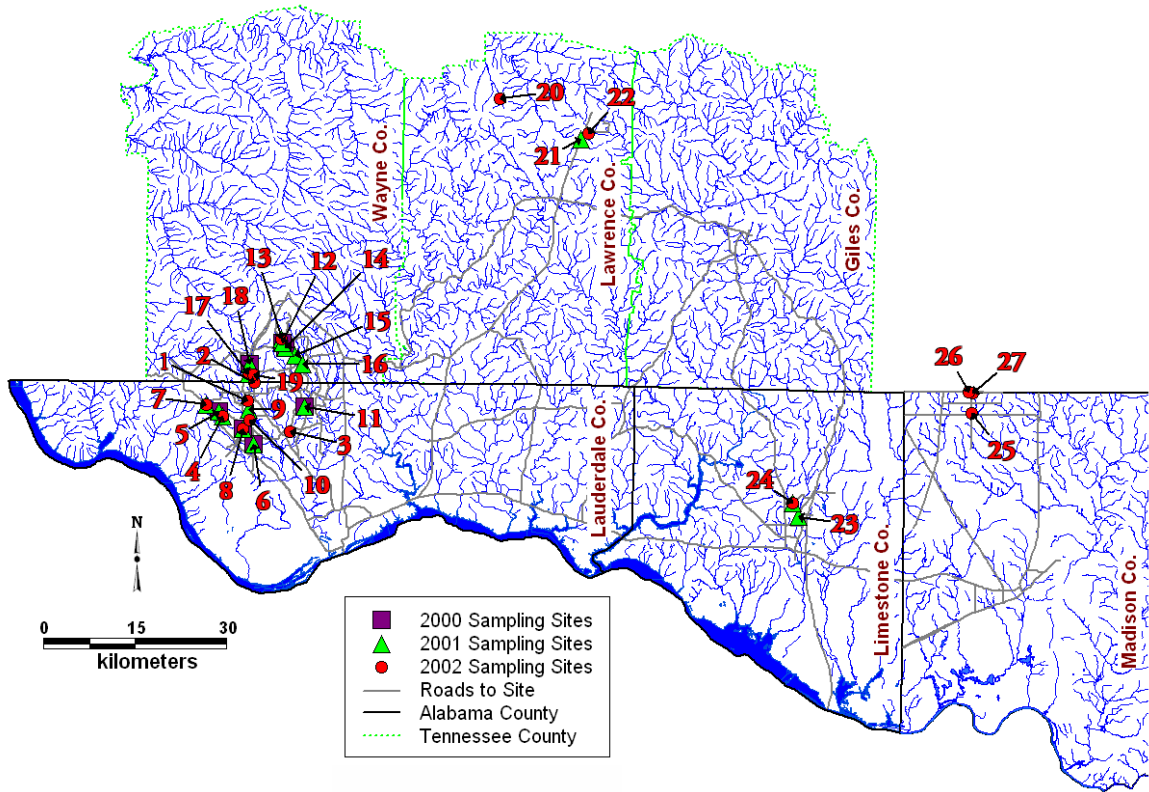


Fig. 8. Example of projected population size per age class of *Etheostoma boschungii* when $P_0 = 0.004$, $P_1 = 0.074$, $P_2 = 0.571$, $P_3 = 0$, and $F_i = 0.184$, using a one-batch fecundity.

APPENDIX I: Description and map of sampling sites for each species

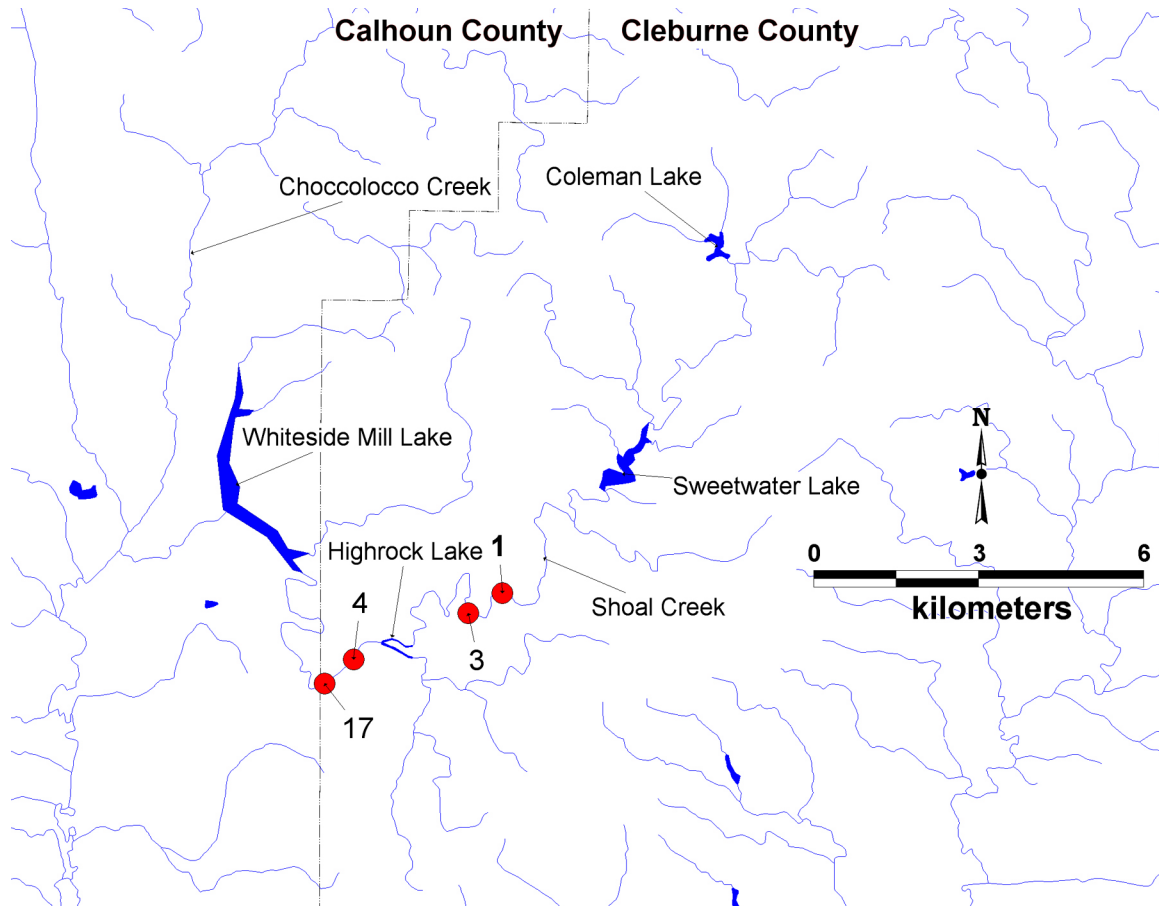
Study sites of *E. boschungi* (circles).



Summary of *E. boschungii* sampling sites. Latitude and longitude for each site is shown in decimal degrees.

Code	Location	Longitude/Latitude (Decimal degree)	System	County	State
1	Cemetary Branch @ Hwy 5	-87.824980/ 34.981177	Cypress	Lauderdale	AL
2	Dulin Branch @ Hwy 157	-87.813611/ 35.005278	Cypress	Lauderdale	AL
3	Greenbriar Branch @ CR 8/CR 139	-87.757863/ 34.941673	Cypress	Lauderdale	AL
4	Lindsey Ck @ CR152/Wallace Ridge Rd - Z site	-87.864201/ 34.961195	Cypress	Lauderdale	AL
5	Lindsey Ck @ CR8/CR10	-87.870651/ 34.967825	Cypress	Lauderdale	AL
6	Lindsey Ck @ CR81/154	-87.815310/ 34.925086	Cypress	Lauderdale	AL
7	Lindsey Ck @ Murphy's Chapel/CR 8	-87.890841/ 34.976175	Cypress	Lauderdale	AL
8	Lindsey Ck @ Natchez Trace/CR5	-87.831700/ 34.945085	Cypress	Lauderdale	AL
9	North Fork Cypress Ck, Cemetery Br @ Natchez Trace Pkwy - D Austin site	-87.822409/ 34.969915	Cypress	Lauderdale	AL
10	Threet Creek @ Natchez Trace	-87.821567/ 34.956233	Cypress	Lauderdale	AL
11	Middle Cypress Ck @ CR139	-87.734198/ 34.974254	Cypress	Lauderdale	AL
12	Middle Cypress Ck @ Dodd Rd (stream crosses under road)	-87.770027/ 35.055643	Cypress	Wayne	TN
13	Middle Cypress Ck @ Dodd Rd -on farm	-87.771527/ 35.061713	Cypress	Wayne	TN
14	Middle Cypress Ck @ Gilchrist Rd	-87.764747/ 35.049803	Cypress	Wayne	TN
15	Middle Cypress Ck @ Middle Cypress Ck Rd	-87.750307/ 35.039523	Cypress	Wayne	TN
16	Middle Cypress Ck @ Pumping Station Rd/HWY227	-87.736967/ 35.029253	Cypress	Wayne	TN
17	Cooper's Br trib @ Natchez Trace Pkwy-seep, 1mi from picnic area	-87.823150/ 35.015767	Cypress	Wayne	TN
18	Cypress Ck @ Trace Pkwy (by park)	-87.820859/ 35.029254	Cypress	Wayne	TN
19	Dulin Branch Trib @ N of Hwy 227	-87.815556/ 35.014444	Cypress	Wayne	TN
20	Chief Ck @ Hwy 240	-87.425400/ 35.372783	Buffalo	Lawrence	TN
21	Little Shoal Ck @ Lawrenceburg, Hwy 43	-87.296125/ 35.320359	Shoal	Lawrence	TN
22	Little Shoal Creek @ Speedway, Dooley Rd off Hwy 43	-87.285074/ 35.327870	Shoal	Lawrence	TN
23	Swan Ck @ Elkton Rd/ CR86	-86.951677/ 34.830918	Swan	Limestone	AL
24	Swan Ck @ Piney Chapel Rd/CR81 - L&N site	-86.960567/ 34.848417	Swan	Limestone	AL
25	Briar Fork trib @ Scott Road	-86.677500/ 34.964722	Flint	Madison	AL
26	Briar Fork trib @ Scott Rd/State Line Rd	-86.681667/ 34.992778	Flint	Madison	AL
27	Briar Fork @ Scott Rd-Scott Orchard	-86.675536/ 34.991691	Flint	Madison	AL

Study sites of *E. brevirostrum* located above and below Highrock Lake (circles).



Summary of *E. brevirostrum* sampling sites for 2001 and 2002 movement studies (US = Upstream site of Highrock Lake Dam, DS = Downstream site of Highrock Lake Dam). Latitude and longitude for each site is shown in decimal degrees.

Site	US/DS of Highrock	Site Description
1	US	Shoal Creek, Pine Glenn Recreation Area off Forest Road 531, 8.1 mi. N Hwy 78, Cleburne Co., AL (N33.725470/W-85.602710)
3	US	Shoal Creek, 5 mi. N Heflin, 1 mi. downstream from Pine Glenn Recreation Area, Cleburne Co., AL (N33.722222/W-85.609370)
4	DS	Shoal Creek, off Forest Road 531, below Highrock Lake, near Cleburne-Calhoun County Line, Cleburne Co., AL (N33.714630/W-85.631850)
17	DS	Shoal Creek, 0.5 mi. from end of FS rd. 530, Cleburne Co., AL (N33.710730/W-85.637580)