

Final Report:

**Ecology of the Western Chicken Turtle (*Deirochelys reticularia miaria*) in the
Arkansas Valley: Development of Survey and Monitoring Protocols for a Rare
and Secretive Species**



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Introduction:

Because of its scarcity and poorly understood biology and ecology, the Western Chicken Turtle (*Deirochelys reticularia miaria*) has been designated as an S3 species (i.e., rare to uncommon) with fewer than 25 locality records in Arkansas. At this time, development of a conservation and management plan is inconceivable due to a lack of baseline ecological data. This information is critical for the development of suitable conservation strategies and to define appropriate distribution and abundance survey protocols for this rare species.

Virtually nothing is known about the Western Chicken Turtle (*D. r. miaria*) in Arkansas, except that they occur in the state. In fact, little is known about the ecology of the three Chicken Turtle subspecies. Few studies have examined the ecology of Chicken Turtles since 1969, and those were conducted on the eastern subspecies (*D. r. reticularia*) in Virginia and South Carolina. Most accounts of Chicken Turtles that appear in general herpetological surveys and reference books lack or provide insufficient ecological data.

The limited ecological and biological data that are available for Chicken Turtles suggest they use a variety of habitats, including terrestrial habitats, during their annual cycle. Chicken Turtles may have bimodal nesting seasons, with the first season occurring in early spring and the second in late summer/early fall. Preliminary data from a local population suggests that female Western Chicken Turtles are gravid in mid-June, potentially using terrestrial habitats for most of the year, and occupying seasonal wetlands during late spring and early summer. Survey protocols for Western Chicken Turtles do not exist. In addition, a potentially limited activity season, low population densities, and specific habitat requirements may make the detection and determination of abundance very difficult for this species. In other words, Arkansas currently has no indication of the population status, distribution, or abundance of this species within the state.

Because Chicken Turtles potentially occupy small, seasonal wetlands, there is the possibility that Chicken Turtles exist in metapopulations. In other words, individuals may move between small wetlands (e.g., small populations) on a seasonal or potentially yearly basis. Because these seasonal wetlands are subject to environmental fluctuations, as well as anthropogenic impacts, a Chicken Turtle population may be dependant upon the ability to colonize new and re-colonize old wetlands.

The principal goal of this study is to determine and document population structure and size, reproductive ecology, and seasonal habitat selection of Western Chicken Turtles in central Arkansas. We proposed to elucidate these variables and develop appropriate survey protocols including appropriate survey seasons and techniques, which will allow for distribution and

abundance surveys to be conducted accurately across the state and throughout the species range. Although official funding for this project began in August 2007, data were collected beginning in 2005.

Materials and Methods:

Study Site. The study was conducted at a floodplain wetland on the East Fork Cadron Creek located near Holland, Arkansas. The wetland consisted mostly of shallow water, less than 1.5 m in depth, dominated by emergent vegetation including Smartweed (*Polygonum* spp.), Water-primrose (*Ludwigia* spp.), and Buttonbush (*Cephalanthus occidentalis*). The surrounding land consisted of a mixed hardwood forest on the eastern side of the wetland and a pine plantation on the western side with Bald-cypress (*Taxodium distichum*) located along the water fringes. The wetland was inundated from fall and spring rains with maximum levels obtained during spring when the East Fork Cadron Creek overflowed its banks. After spring rains ceased, the wetland would often dry completely by midsummer until rains began again in the fall (Figures 1-3).

General Methodology. This study was conducted under the Arkansas Game and Fish Commission Scientific Collection Permit 022520083 and in accordance with Institutional Animal Care and Use Committee animal use protocol 2005-001. Turtles were collected each year (2006 - 2008) from late March through early July using unbaited fyke nets (Vogt, 1980a). A fyke net consists of 2 turtle traps (3 3-ft rings, 1 inch mesh, Arkansas throat) connected by a 25- or 50-ft lead net (6 ft deep, 1 inch mesh). The traps were submerged at least past the throat, but shallow enough to not be completely submerged (allows turtles to breathe). The concept of the trap is that turtles would swim into the lead net and not be able to swim above or below it. They would then swim or crawl along the net and end up in either trap. Nets were positioned throughout the wetland and checked every 1 - 2 days for turtles. All *D. r. miaria* were taken to the University of Central Arkansas for data collection. Each turtle was given an individual identification mark by notching a unique pattern into their marginal scutes. Straight-line carapace length (CL, ± 0.1 mm) was measured for each turtle and sex was determined by the preanal length of a turtle's tail (Gibbons, 1969). Age was estimated by counting the number of growth rings on the abdominal plastral scutes. However, larger individuals tended to have smooth shells preventing an estimation of their age. Females were examined for eggs through palpation and ultrasound. Two clutches were collected in 2006 by injecting two different gravid females with oxytocin and a third was obtained when a female prematurely expelled her clutch (she flipped herself in a holding pen and was trapped inverted while held in captivity in 2008). Mean clutch size was determined for the three clutches; however, only the two intentionally collected clutches were used to determine mean egg length (± 0.1 mm), width (± 0.1 mm), and mass (± 0.1 g). Eggs were subsequently incubated at 29 °C. Once the eggs hatched, mass (± 0.1

g), CL (± 0.1 mm), and plastron length (PL) (± 0.1 mm) were recorded for each hatchling. The hatchlings were released into the population in spring 2007.

Growth models were constructed by using the von Bertalanffy growth model (Fabens, 1965):

$$L_t = a (1 - be^{-kt}), \quad (1)$$

where L_t is CL at age t , a is the maximum CL, b is a variable related to hatchling size, e is the base of the natural logarithm, and k is the intrinsic growth factor. The growth interval method (Fabens, 1965), using the rearranged form of the von Bertalanffy growth model:

$$L_r = a - (a - L_i)e^{-kd}, \quad (2)$$

where L_r is the CL at recapture, L_i is the length at initial capture, and d is the length of time between captures, was used to calculate k for males and females. Carapace length from individuals that were captured at least twice (11 males and 5 females) were used to construct the growth model. The estimated values of k for males and females were then used to solve Equation 1 for b for each sex. Carapace lengths for the growth curves were estimated for males and females (ranging in age from 0 - 25 years) using the estimated values of k and b . Growth rates were compared between the sexes using the estimated CL via an ANCOVA with age as the covariate.

Population Modeling. Encounter histories were constructed for each individual, using the method of Cooch and White (2000). During a trapping season (i.e., 2006, 2007, 2008), a turtle was recorded as “captured” if it was caught or as “not captured” if it was never caught during that year. Once a turtle was captured in a season, subsequent recaptures were not considered in the analysis. Four adult turtles were recovered dead during the study, two presumably from predation and two from disease or injury, though the exact causes are uncertain. Capture histories for these four turtles were constructed to represent the dead recoveries as specified by Cooch and White (2000). Turtles were grouped as male, female, or juvenile (i.e., those too small to determine sex with certainty).

Population parameters were estimated based on encounter histories using Program MARK (White and Burnham, 1999). Models for apparent survival (Φ) and probability of capture (p) were constructed using the Cormack-Jolly-Seber model for open populations (Cormack, 1964; Jolly, 1965; Seber, 1965). A saturated model, which held Φ and p to be dependent on group (i.e., male, female, or juvenile) and time, was fit to the Cormack-Jolly-Seber model followed by constructing models with reduced parameters. The fitness of the saturated model was

determined by comparing its model deviance to the mean deviance of 1000 bootstrap replicated models. The saturated model deviance approached a significant difference ($P = 0.10$) from the mean bootstrap deviance sufficiently enough to be taken into consideration during model selection. In order to determine which model best represented the data, model selection was conducted using maximum-likelihood methods (Burnham et al., 1994). The corrected Akaike Information Criterion (AICc) was used to determine the model that was the most parsimonious (Anderson and Burnham, 1994; Burnham et al., 1995). Due to the degree of overdispersion in the data, as indicated by the bootstrap analysis result approaching a significant P-value, the model deviance of the AICc ($\hat{c} = 1$) was adjusted to the deviance of the saturated model ($\hat{c} = 2.148$). The adjusted \hat{c} required that model selection be conducted using the quasi-likelihood estimator, QAICc (Burnham and Anderson, 1998). Apparent survival and probability of capture were estimated using weighted averages from the QAICc model weights. Program MARK had difficulty in estimating parameters for some models, resulting in unrealistically high estimates of parameters and standard errors, which affected the weighted averages. All problematic models had little support from the QAICc. In order to deal with the error, all models with less than one percent support were dropped from consideration. Weighted parameter estimates were then obtained from the reduced model list.

Population size estimates were obtained using the POPAN formulation of the Jolly-Seber model (Schwarz and Arnason, 1996). This formulation varies from the original Jolly-Seber model in that it estimates population size in addition to Φ and p . However, this model was not used to determine Φ and p for the reason that Program MARK is unable to calculate weighted averages for the POPAN formulation models where it will for Cormack-Jolly-Seber models (Cooch and White, 2000). The POPAN formulation model also includes the additional variable of probability of entrance of an individual into the population (PENT). Models were constructed using the three most parsimonious models from the Cormack-Jolly-Seber model (i.e., the models with over ten percent support from the QAICc). PENT was allowed to vary with time or group. Program MARK was unable to fit any models with PENT held constant or the model with Φ constant, p varying by time, and PENT varying by group. Constructed models were compared using AICc to determine the most parsimonious model, which was then used to estimate population size for each turtle group.

Population growth (λ) was estimated using the Pradel- λ formulation of the Jolly-Seber model, which includes an estimate of population growth (λ) in addition to Φ and p (Pradel, 1996). Similar to the POPAN formulation model, the Pradel- λ model does not allow calculation of weighted averages for Φ and p and was not used to estimate values for them. Models were constructed using the three most parsimonious models from the Cormack-Jolly-Seber models. Lambda was allowed to vary with time or was held constant. Model selection was achieved

using AICc to determine the most parsimonious model from which a population growth estimate was obtained. Juvenile captures were problematic in estimating population growth because the majority of individuals captured (eleven of the thirteen total captures) were ones that had been hatched in captivity and subsequently released into the population. This headstarting is thought to increase hatching success and survival of hatchlings (Heppell et al., 1996), which would likely result in artificially increasing population growth. Due to possible bias in the growth rate from the captive hatching and release of juveniles, the same models were constructed and compared using only the encounter histories from the adult turtles. A population growth estimate was obtained from the most parsimonious model selected through AICc.

Microhabitat Selection. Radio transmitters (R1235 Fish Body Implants, Advanced Telemetry Systems, Inc., Isanti, MN) were epoxied to the carapaces of 8 turtles in 2007 and 10 turtles in 2008. The transmitter and epoxy weighed approximately 40 g. In order to cause the least affect on a turtle's mobility from the added weight of the transmitter, only turtles over 400 g were used in the study. Turtles were located once every 3-4 days during the active period. Once a turtle became inactive (i.e., they remained in one place for at least three consecutive locations), the intervals were extended to one week. Upon location, the microhabitat characteristics of the turtle's position were recorded. The characteristics recorded were dependant upon whether the turtle was aquatic or terrestrial. Additional, the same microhabitat characteristics were recorded from 5 random points. These points were established by travelling a random distance and direction from the turtle's location. The distances were limited to between 7 and 26 m because they represented the interquartile range of daily distances moved in a pilot study.

Microhabitat characteristics measured during aquatic activity periods included water temperature at turtle depth (°C), distance to nearest structure (m), and canopy cover (%). Also, we used a 1 m² Daubenmire frame to assess additional variables including maximum water depth (cm), minimum water depth (cm), submerged vegetation (%), emergent vegetation (%), woody debris (%). Microhabitat characteristics measured during terrestrial activity periods included surface temperature (°C), temperature at turtle (°C), canopy cover (%), vegetative cover (%), detritus depth (cm), and distance from previous location.

Results:

General Results. In Arkansas, *Deirochelys* appears to be active primarily during the spring months (Figure 4) and aestivates during the summer, fall, and winter. During the course of the study, 42 individual turtles were captured and marked. Although not significant, the population approached a male bias (1.9:1, $\chi^2 = 2.79$, $P = 0.095$, Figure 5). However, the population was

significantly adult biased (2.2:1 adult to juvenile ratio, $\chi^2 = 6.09$, $P = 0.014$). The majority of the juveniles captured (11 of 13) were individuals that had been headstarted. Females were significantly longer (CL) than males (Student's t-test, $F_{11,8} = 3.43$, $P = 0.005$) and grew at a significantly faster rate (ANCOVA, $F_{1,50} = 68.15$, $P < 0.0001$; Figure 6). Estimated values of k were 0.31 ± 0.18 (SE) and 0.10 ± 0.05 (SE) and values of b were 0.78 and 0.86 for males and females, respectively.

Reproduction. Sixty percent (3 out of 5) of the females captured in 2006 were gravid while 40% (2 out of 5) of the females caught in both 2007 and 2008 were gravid. Mean gravid female CL was 198.1 ± 7.5 mm (SE; range = 169.5 – 210.8 mm) and mean PL was 174.9 ± 6.9 mm (SE; range = 150.5 - 186.9 mm). Mean clutch size was 11 ± 1 (SE; range = 9 - 12, $n = 3$). For the two clutches obtained through inducing ovipositing, egg length was 36.4 ± 0.3 mm (SE; range 32.0 - 39.1 mm), egg width was 22.3 ± 0.1 mm (SE; range = 21.1 – 23.5 mm), and egg mass was 11.0 ± 0.2 g (SE; range = 9.9 - 12.2 g). Eighteen of the twenty-one eggs (86%) hatched after a 68 day incubation period. Hatchling turtles had a carapace length of 34.4 ± 0.2 mm (SE; range = 31.7 - 35.4 mm), a plastron length of 31.5 ± 0.1 mm (SE; range = 30.5 - 32.3 mm), and a mass of 8.2 ± 0.1 g (SE; range = 6.8 - 9.2 g). Clutch size, egg size and mass, and hatchling size and mass were all very similar to the values obtained for the eastern populations (Table 1).

Unfortunately, we were unable to locate nest sites due to transmitter failure.

Population Parameter Estimates. QAICc model selection for the Cormack-Jolly-Seber models indicated that the most parsimonious models were those with constant Φ and p , or those in where only one variable changed with time (Table 2). Elimination of models with less than one percent support had no effect on the ordering of models and only minimal effect on model weights. Weighted averages of Φ and p showed little variation among group or between sampling periods (Table 3), further indicating that the most likely model is one where Φ and p did not vary over the study. Furthermore, model selection of the POPAN and Pradel- λ formulations of the Jolly-Seber model determined models with constant Φ and p to be the most parsimonious. Population size estimates from the POPAN formulation model agreed with the observed male bias sex ratio ($N = 21 \pm 2$ SE males, $N = 11 \pm 1$ SE females). However, estimates indicate that there is possibly a greater abundance of juveniles than observed ($N = 32 \pm 12$ SE). The population growth estimate that includes juvenile encounter histories indicates that the population is growing ($\lambda = 1.25 \pm 0.16$ SE). However, the estimate for the model excluding juvenile data indicates that the population is stable ($\lambda = 1.07 \pm 0.14$ SE).

Microhabitat Selection. All transmitters either failed, the turtles left the study site, were depredated by raccoons (possibly otters), or shed their scutes along with the transmitters. As a

result, we have very limited habitat selection data (n = 6 turtles in 2007, n = 7 turtles in 2008). In fact, we only have data for April, May and June. It is my opinion that telemetry is futile with this species unless the transmitter is bolted to the shell. However, the shell of *Deirochelys* seems to be quite brittle. Indeed, many of the marks that were originally square deteriorated into broad ovals within the time frame of the study.

We initially planned to use non-metric multidimensional scaling (NMS) to determine (dis)similarities for our sample units (turtle and random locations). NMS is an ordination technique that summarizes complex relationships among sample units in a graphical manner and reports the variation, or differences, among those sample units with the least amount of variables needed (axis or dimensions). However, the initial analyses indicated very little difference between the groups and NMS failed to stabilize in some cases (April, June terrestrial data) due to heterogeneity issues with the data which is probably due to extremely low sample sizes. Indeed, follow-up analyses with MRPP (multi-response permutation procedure) failed to detect any significant differences between turtle microhabitat use and microhabitat availability in May or June (P= 0.7025 for May, P= 0.3942 for June); the two months with the most robust sample sizes. Indeed, the means and standard errors for each variable are very similar for each month (Tables 4 and 5).

Discussion:

Turtles are characterized as having low juvenile survival with a gradual increase as they age (Wilbur and Morin, 1988). In many turtle populations, adult survival is often around 0.90 (Frazer et al., 1990; Congdon et al., 1993; Converse et al., 2005). Therefore, the estimate of Φ for this population (0.70) is somewhat alarming. High adult survival is necessary to counter low hatching success and juvenile survival and to maintain a viable population (Congdon et al., 1993). Without high survival, the reproductive potential of adults is drastically reduced, which further reduces recruitment and would likely result in population decline. The cause of the low survival in this population is unclear because the cause of death in the four observed occurrences could not be fully determined. Alternatively, *D. reticularia* may naturally exhibit a lower survival. For instance, survival estimates from a population in South Carolina was as low as 0.40 and did not exceed 0.70 during the three years the population was studied (Buhlmann and Gibbons, 2001). Furthermore, the species is thought to be short-lived after reaching sexual maturity (Gibbons and Greene, 1978; Gibbons, 1987). A naturally low survival rate may be part of the reason for the presumably reduced longevity in the species. Further study of this population, as well as other populations, is needed to determine if the estimated survival is typical.

Juveniles turtles are expected to have low survival rates that often results from high predation (Congdon et al., 1983). However, the Cormack-Jolly-Seber model estimated juvenile survival to be equivalent to that of adults. Some caution is warranted when examining this estimation because of a possible bias. The majority of juveniles in the study came from the two clutches hatched in captivity, meaning that most of the juveniles possibly had their early survival artificially increased. Although possible bias exists in the estimates, juvenile *D. r. miaria* appear to have a high survivorship because 61% of the headstarted juveniles were encountered the year after they were released.

The similarity in adult and juvenile survival indicates that survival rates are independent of age or size, and that *D. reticularia* demonstrates a Type II survivorship life history (Pearl, 1927) instead of the Type III often associated with chelonians (Wilbur and Morin, 1988). However, a Type III survivorship has been shown in turtles that have markedly different life history traits. For instance, *Emydoidea blandingii* is a long-lived species with delayed sexual maturity (Cogdon et al., 1993) while *D. reticularia* is relatively short-lived (Gibbons and Greene, 1978; Gibbons, 1987). In addition, *D. reticularia* mature at a younger age (2 - 6 years; Gibbons and Greene, 1978). Similar life history characteristics have been observed in *Chrysemys picta*, which also shows a Type II survivorship (Gibbons, 1968; Wilbur, 1975).

Chicken Turtles are believed to be able to exist in small populations (Trauth et al., 2004; Buhlmann et al., 2008). If this is true, the low population estimate should not be as much concern for the population's survival as the estimates for population growth. In order for a population to survive, it must have a growth value (λ) equal to or greater than 1.0 (Williams et al., 2002). The growth estimate obtained from the Pradel- λ formulation including all individuals indicates that the population is actually growing. However, inclusion of the released juveniles in the estimate possibly biased the estimate. When the model was limited to only individuals older than two years, a more reliable estimate was obtained that indicates the population was stable rather than growing. Stable population growth estimates have been determined for other turtle populations; *Chrysemys picta* (Mitchell, 1988), *Emydoidea blandingii* (Congdon et al., 1993), *Kinosternon flavescens* (Iverson, 1991), and *Trachemys scripta* (Frazer et al., 1990). Caution should be taken when interpreting these results because turtle population growth has been shown to vary greatly year to year (Converse et al., 2005). This estimate may only be the result of sampling during a stable period in a stochastic population. For that reason, these estimates best represent a snapshot in the populations' history; further study is necessary in order to determine if any trends are present.

Gibbons (1969) recognized a sex bias in the sampling methods he used to obtain *D. reticularia*. He attributed the male-biased ratio obtained from aquatic traps to the propensity of males to

move around more. A study on a population of *D. reticularia* in Virginia indicated that males may have larger home ranges and greater movements (Buhlmann, 1995). Aquatic traps were used in the current study and a male-biased sex ratio was obtained. However, it is arguable that the bias is not the result of a trapping bias as assumed by Gibbons and Greene (1978). Estimates on the probability of capture obtained from the Cormack-Jolly-Seber model indicate that no sex bias was present in the sampling method because the most parsimonious model held probability of capture constant among groups, as well as among sampling occasions. Therefore, it is unlikely that the population sex bias was the result of capture bias; it more likely represents a sex bias in the population. Furthermore, estimates based on the POPAN formulation model of population size indicate that approximately all males and females within the population had been captured with the possible exception of only a few individuals.

Turtle populations vary greatly in their sex ratios (Gibbons, 1990). For instance, Emydid turtle populations vary from skewed female biases (*Emydoidea blandingii*, Ruane et al., 2008; *Graptemys pseudogeographica*, Vogt, 1980b), to populations with a male bias (*G. geographica*, Vogt, 1980b; *Malaclemys terrapin*, Gibbons 1990). In addition, some species, such as *Chrysemys picta* (Moll, 1973, Raney and Lachner, 1942) and *Trachemys scripta* (Parker, 1984; Cagle, 1942), have different populations with each bias. Therefore, it is not surprising to find a male-skewed bias in this study's population, especially considering that similar and even exceedingly male-biased populations have been observed in *D. reticularia* (Gibbons, 1990).

Growth rates have previously been reported for a *D. reticularia* population in South Carolina based on PL (Gibbons, 1969; Gibbons and Greene, 1978). These studies are limited mainly to younger individuals and do not compare growth rates over an extended range of age classes. Gibbons (1969) reported some difficulties in aging turtles resulting from the smoothing of the shell as the turtles aged. In this study, only two males were obtained that could be aged to more than six years. Using the growth interval method of the von Bertalanffy growth model allowed the construction of models without knowing the age of the turtles. Generally, turtles show their greatest rate of growth as juveniles with a gradual decrease in the rate after reaching maturity (Auffenberg and Iverson, 1979). Males in this study showed such a decrease around the age of four. However, females appear to have an only slightly reduced growth rate by the time they reach the tenth year, which is likely the result of females reaching maturity at a later age and larger size.

The reproductive ecology of *D. reticularia* has been studied from populations in South Carolina and Florida (Gibbons, 1969; Gibbons and Greene, 1978; Congdon et al., 1983; Jackson, 1988). Because reproductive characteristics have been shown to vary across the distribution of some turtle species (Ruane et al., 2008), it is necessary to examine the reproductive ecology of *D. r.*

miaria in order to determine whether they are similar to the eastern subspecies. The similarity in reproductive traits observed in this study to those of previous studies indicates that these variables are similar throughout the distribution of *D. reticularia*. One major difference in reproductive ecology observed in this study was the lack of the necessity to diapause eggs for successful incubation. Previous studies have reported very low hatching success in clutches that were not maintained at around 20 °C before being warmed to initiate development (Buhlmann, 1998). It is uncertain if this represents a true difference between western and eastern populations or if their low success rates were the result of additional factors.

Size at maturity could not be adequately determined for this population. Previous studies used sacrificed individuals to determine maturity (Gibbons, 1969; Gibbons and Greene, 1978; Jackson, 1988). In the present study, maturity could only be determined for females that were gravid, whereas males had no conclusive visual evidence of their maturity. The smallest female recovered with palpable eggs was 169.5 mm CL and 150.5 mm PL. Previous studies indicate that maturity is likely reached around 140 mm PL (Gibbons, 1969; Gibbons and Greene, 1978; Jackson, 1988), 10 mm lower than obtained in this study. The discrepancy in values is most likely the result of the low numbers of females obtained and a general lack of females within the 130 - 150 mm PL range captured during the study. However, considering that other aspects of their reproductive ecology are consistent between *D. r. miaria* and the eastern populations, the size of maturity is likely the same, though it needs to be verified.

Management of chelonian populations involves both in-situ (e.g., habitat conservation) and ex-situ methods (e.g., captive breeding and headstarting) (Moll and Moll, 2004). Although most habitat conservation programs are generally considered beneficial to species, the effectiveness of some programs, such as headstarting, are somewhat controversial. Headstarting involves captive hatching of clutches in an attempt to increase recruitment into a population. However, in populations with low juvenile survival and high adult survival (Type III survivorship) headstarting is likely to have minimal impact on the population (Heppell et al. 1996). In contrast, *D. r. miaria* appears to demonstrate more of a Type II survivorship and as such would depend more on high recruitment than on adult survival to maintain populations. Therefore, headstarting is likely a much more viable management practice in chelonian species that demonstrate a Type II survival, such as *D. r. miaria*. For instance, headstarting showed some promise in this study by high percentage of released juveniles that were encountered the following year.

The demographic and reproductive characteristics of *D. r. miaria* in Arkansas appear to be similar to those of the eastern populations (Gibbons, 1969; Bennett et al., 1970; Gibbons, 1970; Gibbons et al., 1983; Jackson, 1988; Buhlmann, 1995; Burke and Gibbons, 1995; Buhlmann and

Gibbons, 2001). However, the true variability in the demographic and reproductive traits studied is still unknown because of the short duration of this study. In addition, some features require further research in order to better expand the data. For instance, additional study of the species' movements would be beneficial to better interpret its effects on biased sex ratios. Survival also warrants more investigation both from this population and additional ones throughout the species' distribution to determine if their life history does indeed vary from the normal life history associated with chelonians or if these results are an artifact of a very stochastic population. Nevertheless, these uncertainties should not overshadow the contribution this data makes in expanding the knowledge on this understudied species and its usefulness toward managing it.

Unfortunately, the habitat selection study did not produce any conclusive results. The turtles we followed used all parts of the wetland and did not seem to select any area or characteristic over another (Table 4). We can say that at least some Chicken Turtles will seek terrestrial refugia during the warmest months. However, these turtles also seem to remain active and move considerably (Table 5). Other turtles may seek refuge in the nearby creek, as evidenced by one animal appearing in a hoop net in June 2004 and one of our transmittered animals moving to the creek in June 2008. Our suggestion for future research is to look at macrohabitat selection, rather than microhabitat selection. This would be most easily accomplished using trapping surveys for distribution and abundance, coupled with GIS analysis and field-based data collection. From these data, a spatially explicit occupancy-estimation model could be generated to predict where larger populations of Chicken Turtles could be found. The model could then be refined and validated by visiting predicted sites and surveying for populations.

Survey and Monitoring Protocols. During the study, we primarily used fyke nets to capture turtles. We briefly used drift fences with box traps, but had little success. At this site, pitfall traps would have been useless because the water table was at or just below the surface. In other words, the pitfall traps would have filled with water. In addition, the deployment and expense of a drift fence and box trap array for survey purposes would be extremely cumbersome, time consuming, and inefficient. Rather, we suggest that future surveys use fyke nets (Figures 7A, 7B). Our nets used 2 hoop nets (turtle traps) connected by a lead net. We did not use wing nets. Our hoop traps ranged in diameter from 20 in to 3 ft, and lead nets ranged from 25 ft to 50 ft long (all were 6 ft deep). The lead nets were tied into the throat of the hoop nets to encourage a turtle to enter the throat. Bait is not necessary, but can certainly be used. At our study site, we typically deployed between 9 and 18 fyke nets. Our attempt was to saturate the area with nets in order to catch turtles quickly.

According to our trapping data, surveys should begin in March and end in June. Trapping success will undoubtedly be impacted by spring temperatures and water levels. We currently do not have accurate data for fall activity, which still remains a possibility but is unlikely given the water levels observed over 3 years. In order to generate population demographics for a monitoring program, at least two years of trapping should be attempted assuming the population can be justified as “closed” (3 years would be the minimum for an “open” population). We suggest that traps run for at least one week at each site.

Using this survey methodology, we trapped an additional 2 sites along the Cadron Creek. All sites were located on private lands. One site was trapped for two weeks (maximum time allotted by landowner) whereas the remaining site was trapped for approximately 2 months. We captured a total of 5 Chicken Turtles at these sites, which suggests the sites did not support large populations. Interestingly, 3 of the turtles were captured within the first week of trapping. Whether these sites represented sink habitats (main study site would probably be the source) is unknown. However, Red-Eared Sliders (*Trachemys scripta*) that were marked at our main study site were captured at one of these sites. Genetic analyses would have revealed patterns of relatedness and possible gene flow between sites, but this part of the proposal was removed and not funded.

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Table 1. Comparison of the reproductive traits of *Deirochelys reticularia* from various populations. Values listed are means \pm standard error. Values of variances from Congdon and Gibbons (1985) and Jackson (1988) are adjusted to standard error for comparison.

Location	Clutch size	Egg			Hatchling		
		Length (mm)	Width (mm)	Mass (g)	CL (mm)	PL (mm)	Mass (g)
Arkansas ^a	11 \pm 1	36.4 \pm 0.3	22.3 \pm 0.1	11.0 \pm 0.2	34.4 \pm 0.2	31.5 \pm 0.1	8.2 \pm 0.1
South Carolina ^b	8 \pm 2	36.0 \pm 0.8	22.0 \pm 0.4	10.7 \pm 0.6	-	24.3 \pm 1.0	6.7 \pm 0.9
South Carolina ^c	8 \pm 1	34.8 \pm 0.2	20.8 \pm 0.2	9.1 \pm 0.2	-	-	-
Florida ^d	10 \pm 1	36.5 \pm 0.2	22.4 \pm 0.1	10.7 \pm 0.1	-	30.1 \pm 0.5	8.5 \pm 0.1

^a Current study

^b Congdon et al. 1983

^c Congdon and Gibbons 1985

^d Jackson 1988

Table 2. Reduced Cormack-Jolly-Seber model selection based on the corrected Quasi-Akaike Information Criterion (QAICc) with an adjusted $\hat{c} = 2.148$. Model names indicate if survival (Φ) or probability of capture (p) were held constant (.) or varied with group (g) or time (t). Δ QAICc is the difference in QAICc value between a model and the most parsimonious model. Model weight indicates the support for a model. Number of parameters specifies the number of variables estimated for a model. QDeviance is the deviance of a model from the fully saturated model.

Model	QAICc	Δ QAICc	Model Weight	Model Likelihood	Number of Parameters	QDeviance
$\Phi(.)p(.)$	31.72	0.00	0.465	1.000	2	4.87
$\Phi(t)p(.)$	33.88	2.16	0.158	0.339	3	4.68
$\Phi(.)p(t)$	33.88	2.16	0.158	0.339	3	4.68
$\Phi(.)p(g)$	35.08	3.36	0.087	0.186	4	3.39
$\Phi(g)p(.)$	36.18	4.46	0.050	0.108	4	4.48
$\Phi(t)p(t)$	36.37	4.65	0.045	0.098	4	4.68
$\Phi(t)p(g)$	37.60	5.88	0.025	0.053	5	3.26
$\Phi(g)p(t)$	38.77	7.05	0.014	0.029	5	4.44

Table 3. Apparent survival between 2006 and 2007 (Φ_{1-2}) and between 2007 and 2008 (Φ_{2-3}) and probability of capture during 2007 (p_2) and 2008 (p_3) for each sex grouping classification. Values listed are the weighted averages from compared models \pm the weighted standard error from the models.

	Apparent survival		Probability of capture	
	Φ_{1-2}	Φ_{2-3}	p_2	p_3
Male	0.71 \pm 0.19	0.69 \pm 0.18	0.85 \pm 0.19	0.82 \pm 0.20
Female	0.71 \pm 0.19	0.69 \pm 0.18	0.81 \pm 0.23	0.78 \pm 0.24
Juvenile	0.70 \pm 0.20	0.68 \pm 0.19	0.81 \pm 0.24	0.78 \pm 0.24

Table 4. Aquatic microhabitat availability and use by Western Chicken Turtles (*Deirochelys reticularia miaria*).

Month	Water Temp. (°C)	Canopy Cover (%)	Water Depth (cm)			Vegetative Cover Class (%)		
			Max.	Min.	Difference	Submerged	Emergent	Woody Debris
April								
Available	18 ± 1	27 ± 19	41 ± 11	44	12	75-100	50-75	25-50
Turtle	18 ± 1	40 ± 18	56 ± 15	42 ± 5	22 ± 19	75-100	75-100	25-50
May								
Available	23 ± 1	21 ± 7	41 ± 4	32 ± 4	9 ± 2	75-100	25-50	50-75
Turtle	23 ± 1	27 ± 10	43 ± 3	37 ± 3	6 ± 1	75-100	75-100	50-75
June								
Available	26 ± 1	34 ± 7	33 ± 2	23 ± 2	9 ± 2	25-50	75-100	25-50
Turtle	26 ± 0	35 ± 8	41 ± 5	28 ± 2	14 ± 5	50-75	75-100	75-100

Table 5. Terrestrial microhabitat availability and use by Western Chicken Turtles (*Deirochelys reticularia miaria*).

Month	Surface Temp. (°C)	Turtle Temp. (°C)	Turtle Depth (cm)	Canopy Cover (%)	Vegetative Cover (%)	Detritus Depth (cm)	Distance from Previous Location (m)
June							
Available	29 ± 1	26 ± 2	.	91 ± 6	75-100	3 ± 2	.
Turtle	30 ± 1	27 ± 2	4 ± 2	61 ± 30	75-100	2 ± 1	43 ± 13

Figure 1: This is the entrance to the study site during spring (April 2007). However, the Cadron Creek has crested its banks and site access is impossible.



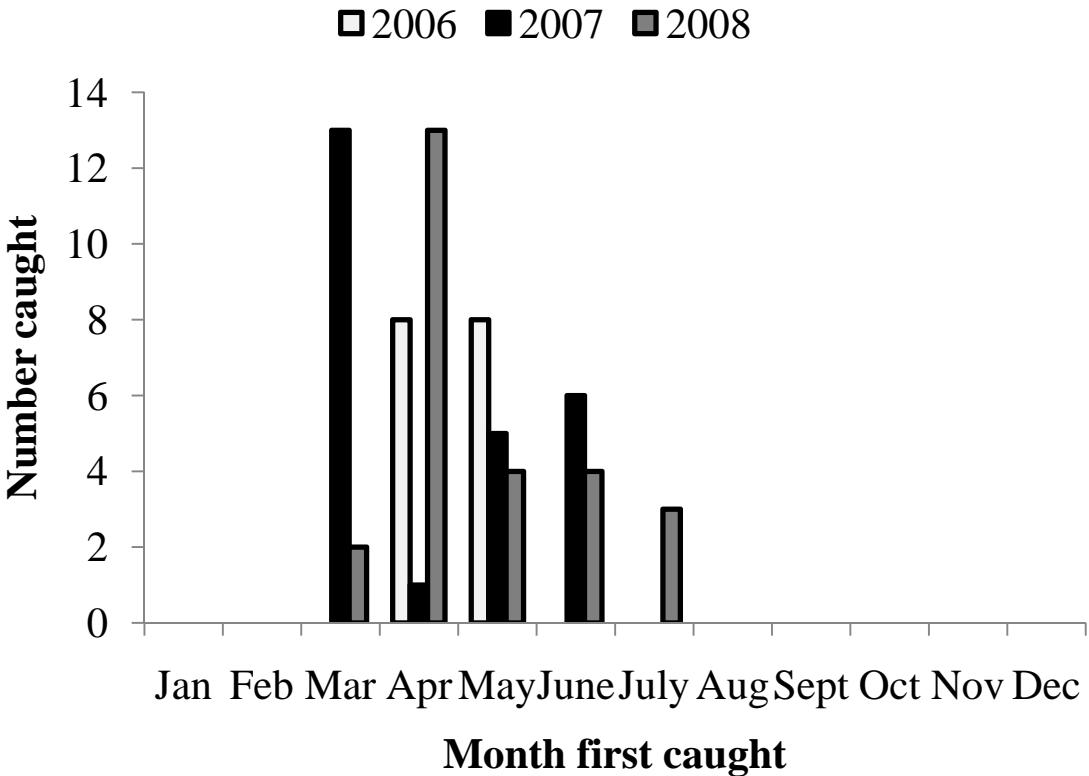
Figure 2: This is the dried pond section of the main study site during September 2006. The section of trees in the background is wooded area shown in Figure 1.



Figure 3. Typical water levels at the main study site during May 2006. This is the entrance and the pond is to the left. Turtles were found throughout the main site.



Figure 4. Month of first capture for individual *Deirochelys reticularia* separated by sampling year. The dates of the first captured turtle each year were 21 April 2006, 24 March 2007, and 13 March 2008.



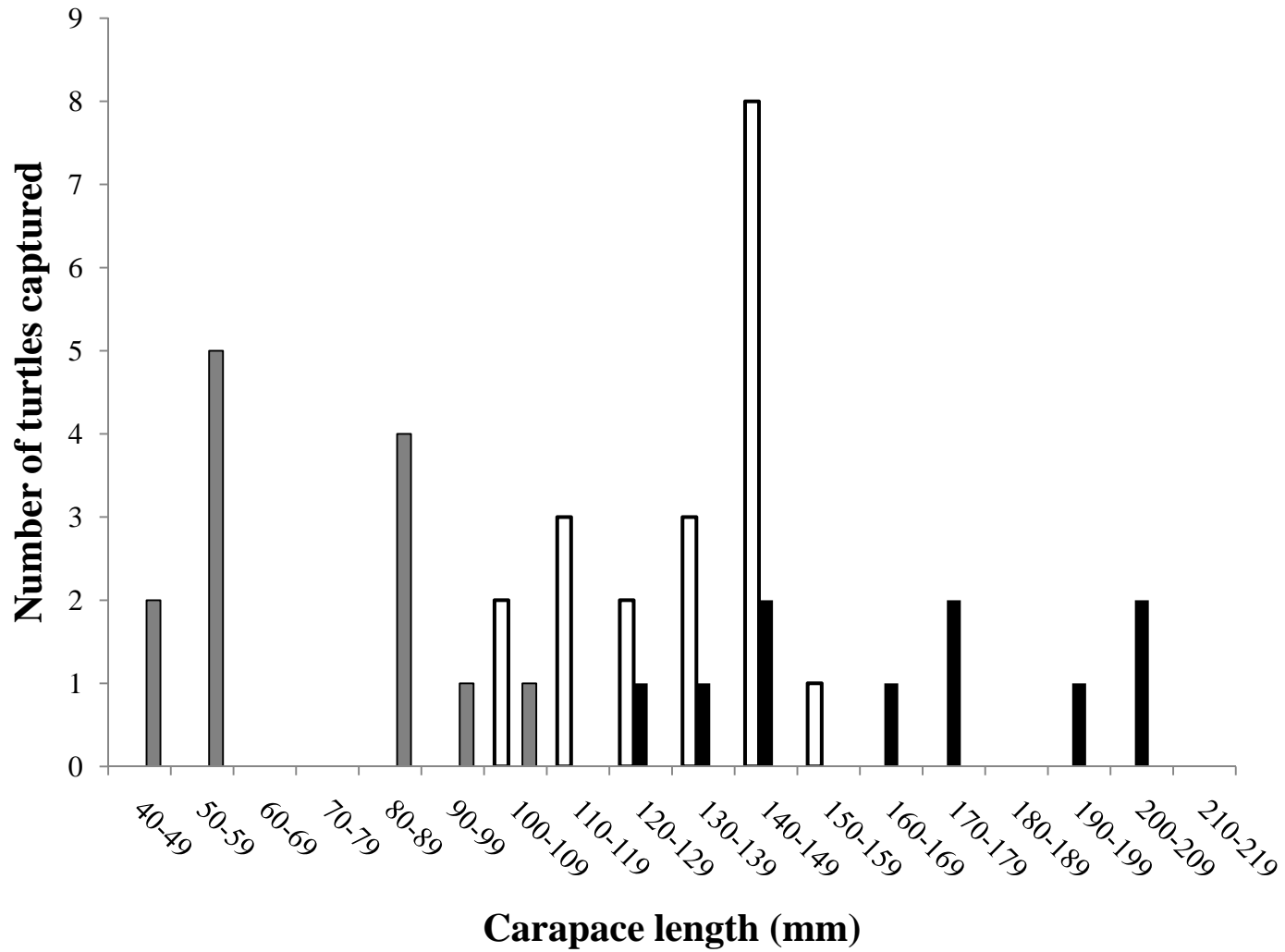


Figure 5. Size distribution of *Deirochelys reticularia miaria* captured at the study pond. Males are designated by white boxes, females by black boxes, and juveniles by grey boxes.

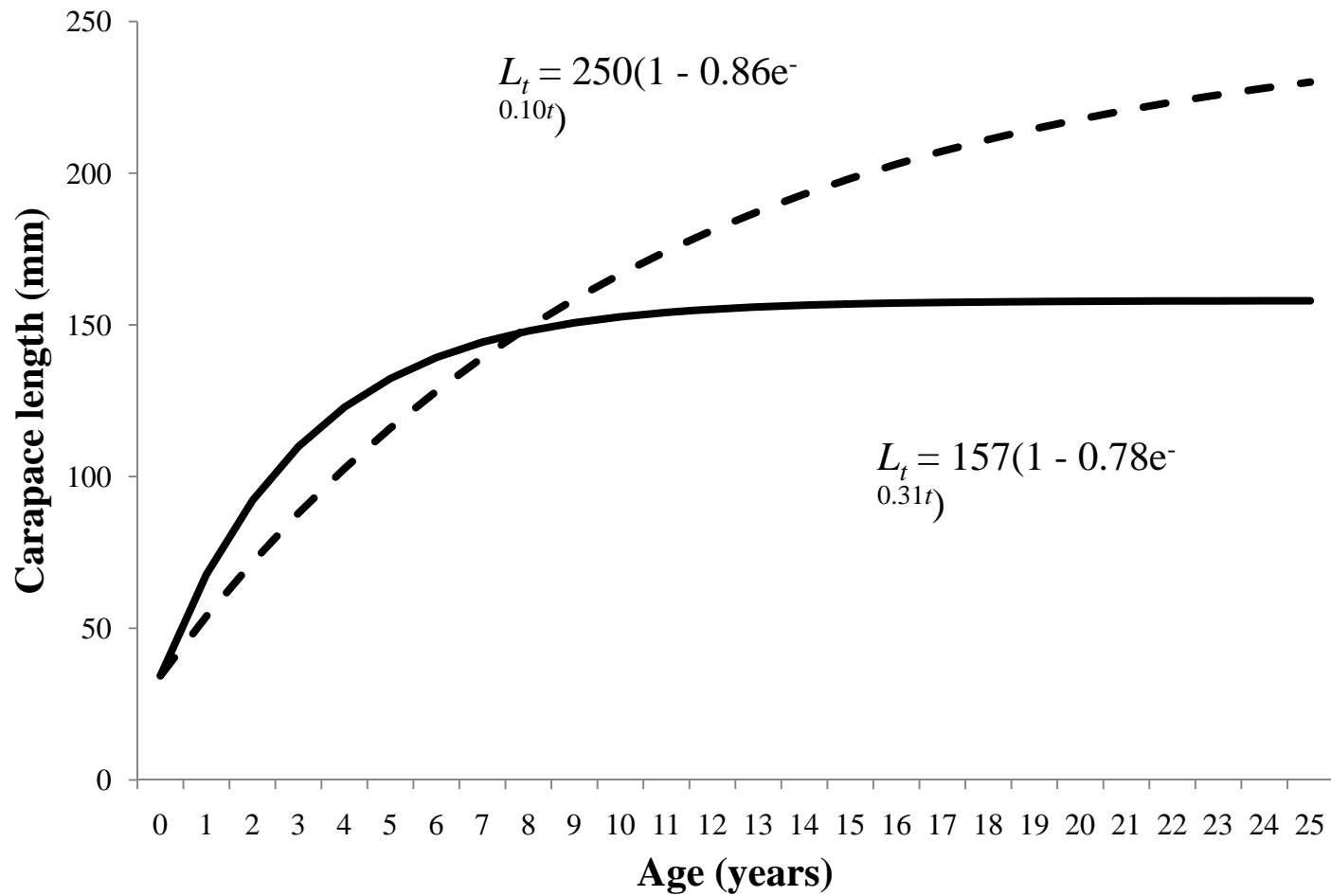


Figure 6. Von Bertalanffy growth curves for male (solid line) and female (dashed line) *Deirochelys reticularia miaria* based on straight-line carapace length. Equations are included below the curve for males and above the curve for females. L_t is the estimated carapace length at age t .

Figure 7A- 7B. Fyke nets.

