FINAL PERFORMANCE REPORT

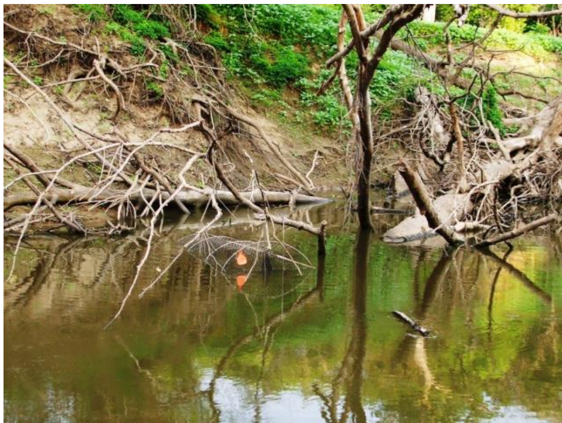
Federal Aid Grant No. F10AF00168 (T-52-1)

A Survey of Alligator Snapping Turtles and Other Turtle Species in Three Northeastern Oklahoma Rivers

Oklahoma Department of Wildlife Conservation

July 1, 2010 through June 30, 2014

A SURVEY OF ALLIGATOR SNAPPING TURTLES AND OTHER TURTLE SPECIES IN THREE NORTHEASTERN OKLAHOMA RIVERS: 1 JULY 2010–30 JUNE 2014



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OBJECTIVES

1. To measure growth and survivorship rates of reintroduced alligator snapping turtles in the Caney River. The locations, number, size, and growth rates of individuals will be provided in Performance reports.

2. To measure turtle community structure on the Spring River upstream of Grand Lake and, if deemed suitable, reintroduce alligator snapping turtles to the system. The locations, species, size distribution and number of individuals caught will be provided in Performance reports.

3. To measure turtle community structure on the Verdigris River upstream of Oologah Lake and, if deemed suitable, reintroduce alligator snapping turtles to the system. The locations, species, size distribution and number of individuals caught will be provided in Performance reports.

OVERVIEW

We report on two interrelated issues. The first section reports the growth, body condition, and survival of released alligator snapping turtle juveniles in the Caney River in northeastern Oklahoma, a site where an otherwise robust turtle community persists but where alligator snapping turtles were extirpated. The second section compares and contrasts the aquatic turtle communities in the Caney River, Verdigris River, and Spring River, and analyzes several environmental gradients and how they are influencing these communities. Globally, freshwater turtle populations are declining at an alarming rate, the causes of which include overharvest, habitat modification, pollution, and collection for the pet trade (Gibbons and Stangel 1999, Gibbons et al. 2000). While primarily focused in Asia, freshwater turtle conservation is now a worldwide dilemma. The southeastern United States boasts a rich diversity of freshwater turtles, but these turtles are often illegally taken and shipped overseas, decimating local populations (Moll and Moll, 2004).

Alligator snapping turtles (*Macrochelys temminckii*) have experienced significant population declines. This species has been particularly affected by a combination of habitat alteration, commercial harvest and an iteroparous reproductive strategy (Pritchard, 2006). The species is currently afforded some level of protection in all states that it occurs. In Oklahoma, alligator snapping turtles occur in the eastern one-third of the state and are listed as a Species of Special Concern (Riedle et al., 2005). Surveys conducted over three years at 67 sites in 15 counties in eastern Oklahoma resulted in only 63 captures at four sites (Riedle et al., 2005). Previously, *M. temminckii* had been reported at 26 sites around the state (Riedle et al., 2005). In response to the apparent disappearance of this top-level predator, a reintroduction program was started at Tishomingo National Fish Hatchery in Oklahoma, in which individuals were hatched and raised at the hatchery and spent a year in hatchery ponds before being released into suitable habitat (Riedle et al., 2008).

1. CONSERVATION AND REINTRODUCTION OF ALLIGATOR SNAPPING TURTLES: GROWTH, BODY CONDITION, AND SURVIVAL

1.1 Introduction

Reintroduction of imperiled species is an increasingly important conservation management tool for species that have experienced population declines, but for which suitable habitat persists (Snyder et al., 1996; Seddon et al., 2007; Seddon et al., 2012). Reintroductions may be conducted to satisfy a variety of objectives (Seddon, 2010), but most frequently aim to either repopulate areas where a species has been extirpated or supplement depleted populations that lack sufficient numbers to recover without intervention (Seddon et al., 2007). Such efforts can have profound effects on an ecosystem, especially when focused on keystone species or top-level predators (Mittelbach et al., 1995; Ripple and Beschta, 2003; Ritchie et al., 2012).

A variety of potential drawbacks to reintroductions have been either documented or postulated, including aberrant behavior resulting from captive rearing (Crane and Mathis, 2010), low genetic diversity among released stock (Groombridge et al., 2012), and high mortality rates of released animals due to inexperience finding local resources or identifying and evading predators (Snyder et al., 1996; Reinert and Rupert, 1999; Roe et al. 2010). Many of these drawbacks can be addressed in a well-designed conservation program. For instance, training has shown promise for conditioning captive-bred animals to recognize and avoid predators in a diverse range of taxa (Berejikian et al., 1999; Alberts, 2007; Crane and Mathis, 2010; Olson et al., 2012). Issues related to low genetic diversity can be addressed with well-designed breeding programs that maintain captive populations with adequate effective population sizes, maximize interbreeding among available subpopulations of captive stock, and minimize the number of generations produced in captivity (Frankham, 2007).

Finally, the negative impacts of animals' inexperience finding local resources can often be minimized using a "soft-release" approach where individuals slated for translocation or reintroduction are exposed to natural environmental conditions within the confines of a protected site (Van Leuven et al., 2004; Tuberville et al., 2006). For instance, exposure to natural foraging conditions with the absence of predation pressure offers animals an opportunity to become proficient at locating and handling prey (Brown et al., 2003; Escobar et al., 2010). Additionally, restricting movements to a large but enclosed area can limit the 'wandering' behavior that has been described in several reintroduction studies (Tuberville et al., 2006; Rittenhouse et al., 2007; Roe, 2010). Wandering likely increases exposure to predation and limits individuals' familiarity with locally available resources—such as food patches and shelter—that are necessary for survival. Characteristics of animals conducive to a soft-realease approach include species with instinctive behavior (i.e. lack of parental fostering), species that are at the top of the local food chain, or species reintroduced into an environment free of potential predators (Snyder et al., 1996).

Regardless of the pre-release measures that are taken to ensure the success of a reintroduction initiative, post-release monitoring to evaluate actual success is critical for informing long-term conservation (Nichols and Armstrong, 2012). Effective post-release monitoring is expensive, time-consuming, and may require years or decades to determine the ultimate success or failure of a reintroduction project. As a result, post-release monitoring efforts were not always incorporated into early reintroduction efforts (Sarrazin and Barbault, 1996; Snyder et al., 1996; Seddon et al., 2007).

The alligator snapping turtle (*Macrochelys temminckii*) possesses a suite of characteristics that make it an attractive candidate for a captive propagation-reintroduction conservation approach.: *Macrochelys temminckii* is long-lived (Ernst and Lovich, 2009), inhabits a variety of river, lake, swamp, and slough habitats (Pritchard, 1989), has a catholic diet (Sloan et al., 1996; Elsey, 2006; East, 2012), produces large clutches, typically ranging from 9 to 61 eggs per clutch with a mean of 27.8 eggs per clutch (Ernst and Lovich, 2009), and is relatively easy to propagate (B. Fillmore, unpublished data). State laws throughout the specie's range protect alligator snapping turtles. In Oklahoma, the species is listed as a Species of Special Concern and both harvest and possession are prohibited. The species historically occurred across much of the eastern one-third of the state, but today is restricted to just a few river systems in the east-central and southeastern portions of the state (Riedle et al., 2005; 2006). Surveys conducted over three years at 67 sites in 15 counties in eastern Oklahoma resulted in only 63 captures at four sites (Riedle et al., 2005).

Because of this decline, it was determined that reintroduction efforts to re-establish viable populations in suitable habitat were warranted (Riedle et al., 2008). Using head-started juvenile *M. temminckii*, 90 turtles were released into the Caney River system in 2008, and an additional 60 and 96 juvenile *M. temminckii* were released in 2009 and 2010, respectively. All turtles were 3–7 years old at the time of release. To assess the impact of these reintroductions, mark-recapture surveys were conducted in 2008–2013 to measure growth rates, changes in body condition, and annual survival rates. Although none of these metrics are definitive measures of success, all are informative indicators of the progress of a reintroduction effort.

1.2Methods

Study Sites—Trapping surveys were conducted in northeastern Oklahoma on the Caney River, which has its headwaters in the tallgrass prairie ecoregion in Kansas and is dammed to form Hulah Lake in Osage County, Oklahoma, approximately 20 river kilometers south of the Kansas border. The river has a narrow riparian buffer that is surrounded by agricultural fields and prairie. Because of its isolation from metropolitan areas, human activity on the river is low in comparison to many rivers in the state, but includes low levels of fishing, camping, swimming, and boating.

The extent of the Caney River that we sampled was restricted by limitations imposed by the navigability of the river and the availability of public access points. We sampled a combined16.4 km of the river and one of its tributaries, Pond Creek. We identified 80 locations that were suitable for setting a hoop net. During each day of sampling, nets were placed at a randomly selected subset of 6–15 locations.

Trapping—We used four-hoop and three-hoop hoop traps consisting of 76-cm diameter hoops and 2.5-cm square mesh. The traps were stretched by attaching notched PVC to the outermost hoops. Traps were baited with either canned sardines or fresh fish that were either by-catch in the hoop traps or caught in gill or trammel nets. Traps were set between 13:00–18:00 and checked the following morning. Trapping effort was alternated each day between the main channel of the Caney River and Pond Creek. The trapping surveys in 2008 and 2009 consisted of four days each of trapping during the month of July. Fifteen traps were set daily, for a total effort of 60 trap nights in each of those two years. In 2010 we sampled in June and July for 11 days and a total of 189 trap nights. In 2011 we sampled May–August for 21 days with a total of 169 trap nights. In 2012 we sampled May–August for 23 days with a total of 171 trap nights. In 2013 we sampled May–July for 20 days, and in 2014 in June for 9 days. All individuals trapped had a unique passive integrated transponder (PIT) tag number for identification. This study was approved by the Missouri State University Institutional Animal Care and Use Committee (protocol number 10015) and the Oklahoma Department of Wildlife Conservation (permit #5376).

Data Analysis—Growth of recaptured individuals was assessed in three ways. First, changes in size were assessed by comparing individual turtles' midline straight carapace length (MCL) at the time of release to MCL at their first and second recapture and analyzed with a repeated-measures ANOVA. Second, because animals of different sizes are expected to grow at different rates, we calculated size-corrected growth as $((MCL_i – MCL_r)/MCL_r)$/(years between captures), where MCL_r was MCL at the time of release and MCL_i was the MCL at the i-th recapture. Finally, changes in body condition were calculated by regressing $\log_{10}$(mass) on $\log_{10}$(MCL) and using the resulting residuals to generate a body condition index (Jakob et al., 1996). For purposes of comparison, head-started turtles that remained at the hatchery were included in measures of body condition. The hatchery turtles were divided into two groups: 1) individuals that were maintained indoors where they were fed dead fish and fish-based pellets *ad libitum*; and 2) individuals that were maintained for a year in an outdoor pond at the hatchery where they were exposed to natural cycles and foraged much as released or wild turtles might. Finally, body condition of released turtles was regressed against time of year (Julian date) to assess seasonal changes in body condition.

Survival and capture probability were analyzed using Cormack-Jolly-Seber models (CJS) (Nichols 1992) in Program MARK (White and Burnham, 1999). Analyses were conducted using mark-recapture data from the 2010–2012 sampling efforts.

1.3 Results

Recaptured individuals consistently exhibited measurable increases in length, both in comparison to their size at release and to their size at previous recaptures (F = 82.05, df = 27, P = 0.0005) (Figure 1-1). Following release, turtles grew 5–41% in MCL per year (mean = 17 ± 1%) and 18–442% in mass per year (mean = 82 ± 11%). There was a positive correlation between mass and carapace length among recaptures and turtles that were retained in captivity (slope = 0.63, R^2 = 0.97, P = 0.0005; Figure 1-2). Body condition did not vary among animals that were maintained indoors or outdoors in a hatchery pond, or at the time of initial release, or after first or second recaptures (F = 0.24, df = 3, 301, P = 0.87; Figure 1-3). Successive body condition measurements made at the time of release and at each subsequent recapture did not differ significantly (P = 0.162–0.729). Similarly, body condition did not correlate with time of year (R^2 = 0.068, P = 0.101, slope = 0.04). Annual growth rates for individuals were higher for recaptured releases than at the hatchery, except in year 2011 (Figure 1-4). Low sample sizes for turtles recaptured in consecutive years did not allow statistical analysis to compare annual growth rates to animals at the hatchery and released animals.

The survival estimate for all trapping periods in 2010–2012 was 0.64 ± 0.08 (95% CI = 0.51–0.78). The capture probability for this survey period was 0.30 ± 0.10 (95% CI = 0.15–0.51).

1.4 Discussion

All recaptured individuals exhibited substantial and consistent growth, and body condition remained comparable to that of turtles that remained in captivity and were fed *ad libitum*. This suggests that reintroduced turtles successfully and quickly located the resources needed to survive and flourish. One turtle exhibited exceptional growth (Fig. 1-1, identified with an asterisk). The individual was a year class 2004 male; its MCL at release in 2008 was 187.6 mm, and MCL at second recapture in 2010 was 296.6 mm. This individual was larger than average in comparison to others in its cohort (mean = 172.97 mm) at the time of release, and his MCL increased 28% per year while mass increased 89% per year (mean increase in MCL = 8.74% and in mass = 74.50%). The trajectory of its MCL growth followed a slope of 54.51 compared to the average slope of growth for all other turtles (14.82).

Even discounting this exceptional outlier, growth of released turtles was robust. One-year growth of released turtles was higher than that of captive turtles kept outdoors in a well-stocked pond or even from those maintained indoors year-round and fed *ad libitum* (Figure 1-4). However, sample sizes of recaptured releases were too low to assess significance. In particular, there was only one individual represented by the year 2011 release group, which is a likely explanation for the apparent decline in growth observed in 2011. Two other studies of translocated and/or reintroduced *M. temminckii* also reported good growth and body condition a year after release. In Louisiana, a subadult *M. temminckii* that was translocated to a site presumed to be outside of its natural home range exhibited modest growth one year after its release (Bogosian, 2010). Although the small number of translocated subadults included in that study prohibited statistical analysis of growth, the single translocated animal grew approximately twice as much as resident subadults that were of comparable size and monitored during the same period. In a study conducted in southern Oklahoma, captive-reared *M. temminckii* were released into an oxbow and monitored for more than a year after release (Moore, 2010). Not only did the turtles grow in that study, but actually exhibited greater body condition after a year than did animals from the same cohort that remained in captivity. These studies, in combination with our results, suggest that *M. temminckii* can thrive in a novel environment, and may be much better suited to a reintroduction conservation approach than some other chelonians that apparently only perform well after acquiring information about the spatial distribution of patchy resources (Tuberville et al., 2006; Rittenhouse et al., 2007).

Locating resources and maintaining good body condition do not ensure the long-term success of a reintroduction effort—individuals must also survive to adulthood and reproduce. Although insufficient time has elapsed to measure these more decisive endpoints, this is the first study to assess survival rates of reintroduced *M. temminckii*. The survival estimate was low in the analysis of all survey periods in 2010–2012, and if this estimate was an accurate reflection of annual survival rates then the population is unlikely to persist long-term. However, factors other than mortality seem likely to have contributed to this survival estimate. First, the model indicated low capture probability, and this affected the magnitude of the confidence intervals around the survival estimate. Such low capture probabilities appear consistent with previous mark–recapture efforts for this species. For instance, in a captive population of 30 adult turtles being maintained as brood stock in two ponds totaling 0.63 ha surface area at a national fish hatchery, 400 trap nights of effort using hoop traps baited with fresh fish managed to capture only 19 animals (D. Thompson, pers. comm.).

Second, survival estimates in Program MARK are based on the presence or absence of individuals over multiple survey periods. Because the model was derived from data divided into nine discrete sampling periods, the survival estimate would have decreased when turtles that were alive and present in the sampled reach of river were not recaptured in later sampling periods, which we expect may have occurred frequently because capture probability was low. While mortality would certainly account for animals not being recaptured late in the study, so too would emigration out of the sampling area. In fact, calculations based on the Cormack-Jolly-Seber model in Program MARK should be expected to underestimate survival; the model assumes that both emigration and immigration will occur, but in a reintroduction scenario no animals originate outside of the sampling area, so immigration will not occur and therefore will not balance emigration from the point or region that releases occurred (White and Burnham, 1999). Unfortunately, the degree to which survival is underestimated because of the discrepancy between model assumptions and reality are not known. Therefore, we can only conclude that the estimated annual survival rate represents a minimum threshold. East (2012) reported a survival rate of 0.46 of resident individuals at a national wildlife refuge. Survivorship values for eastern snapping turtles (*Chelydra serpentina*) have been reported between 0.74-0.76 (Steyermark et al., 2008). Therefore, survivorship of released turtles at the Caney River falls between a population in decline and normal values for a closely-related species.

The degree to which turtles emigrated from the study area is unknown, but there was ample opportunity for them to do so. The total available aquatic habitat that turtles could have dispersed to covers approximately 114 ha of the Caney River and Pond Creek and 10,765 ha of Hulah Lake. The total area of river in which turtles were released and subsequently trapped was equal to approximately 56 ha, or about 49% of the total riparian areas and 0.52% of the total area when the lake is included. Emigration of the released population from this larger area is limited by two factors. One is that, except during flooding events, both the main channel and its tributary enter shallow riffles upstream of the area sampled. Secondly, the dam forming Hulah Lake prevents further emigration downstream. River modifications such as these currently contribute to limitations of emigration and result in genetic isolation of populations throughout the species' range (Roman et al., 1999).

Continued close monitoring of this reintroduced *M. temminckii* population will be necessary to ascertain the ultimate success of the conservation endeavor. In the future, released turtles should be tracked via radio telemetry to get a better estimate of survival, to measure movement and emigration patterns, and to determine the extent to which animals utilize the nearby reservoir. The population structure will also need to be monitored, as some individuals are likely on the cusp of attaining sexual maturity. The onset of maturity will necessitate monitoring of nesting activity and nest depredation, and will mark the beginning of a substantively new phase in the progression to a viable, self-sustaining population.

Table 1-1. Age distribution, number, and size of Alligator Snapping Turtles released in the Caney River. Values reported are mean ± SE.

Release year	Year Class	n	Mass	Carapace length
2008	2004	46	1352±84.34	169.32±3.48
	2005	44	328.67±9.26	109.69±1.18
2009	2005	60	661.8±19.11	137.55±1.28
2010	2003	2	2315±280	208.98±8.01
	2005	23	834.57±40.56	138.74±4.78
	2006	17	322.65±24.34	107.44±2.94
	2007	54	174.17±6.58	86.6±1.07

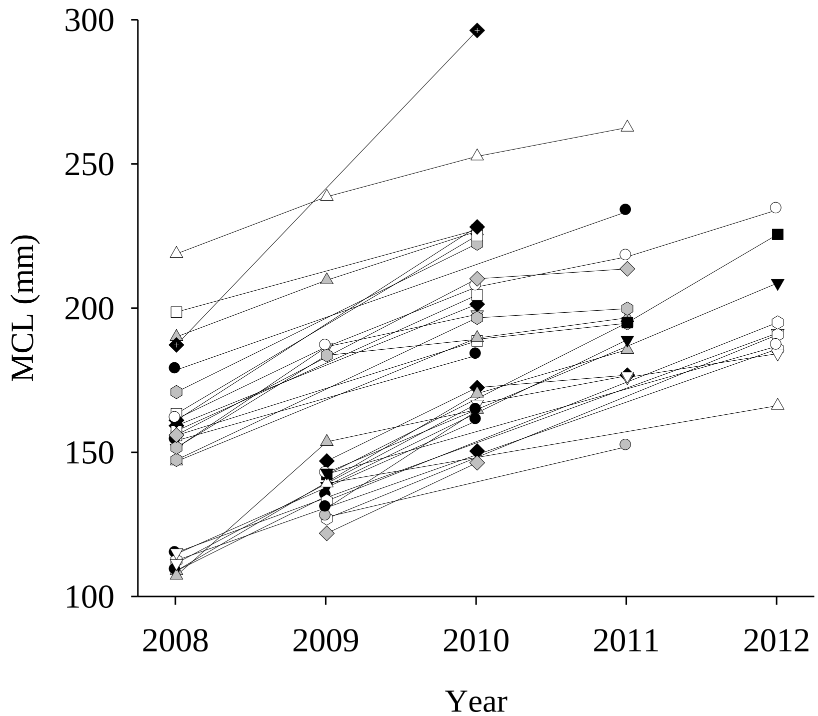

Figure 1-1. Midline carapace length growth of 40 recaptured individuals. Lines connect points representing a single turtle. The first point in each set indicates size at the time of release, and each subsequent point represents a recapture.

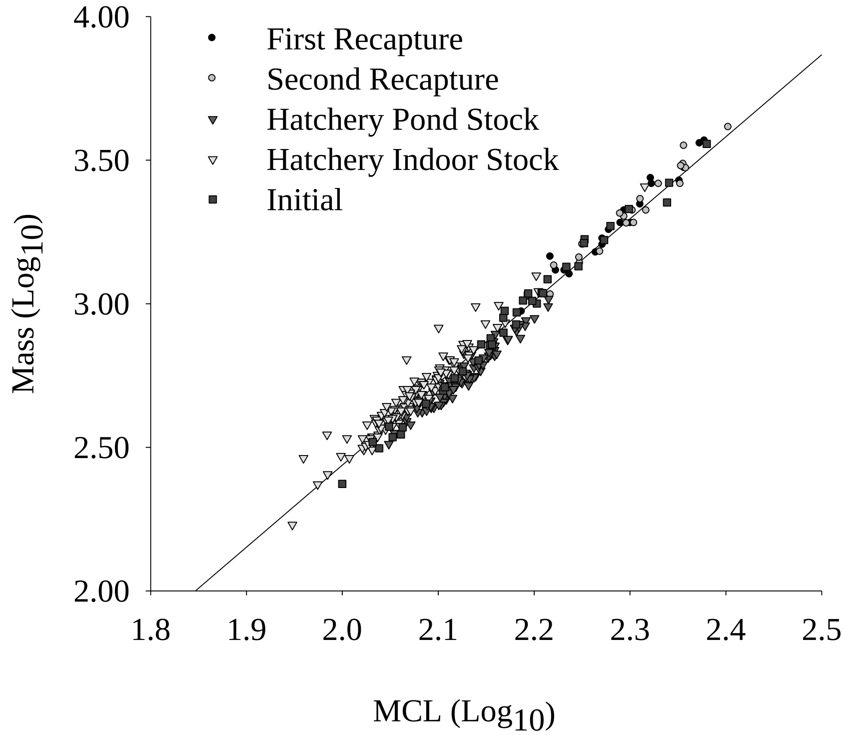

Figure 1-2. Relationship of mass to midline carapace length of alligator snapping turtles recaptured from the Caney River, as well as turtles that remained indoors or in an outdoor pond at Tishomingo NFH (R^2 = 0.97, P = 0.0005, slope = 0.63).

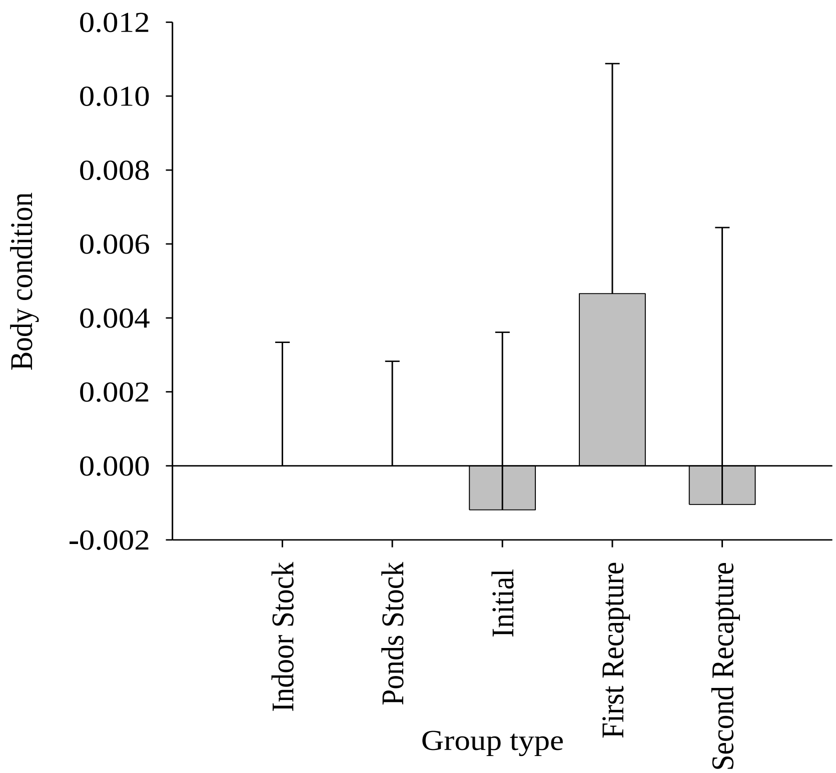

Figure 1-3. Average body condition of alligator snapping turtles measured under different conditions. Error bars are ± 1 SE. Average values for hatchery stock were too close to zero to generate visible bars.

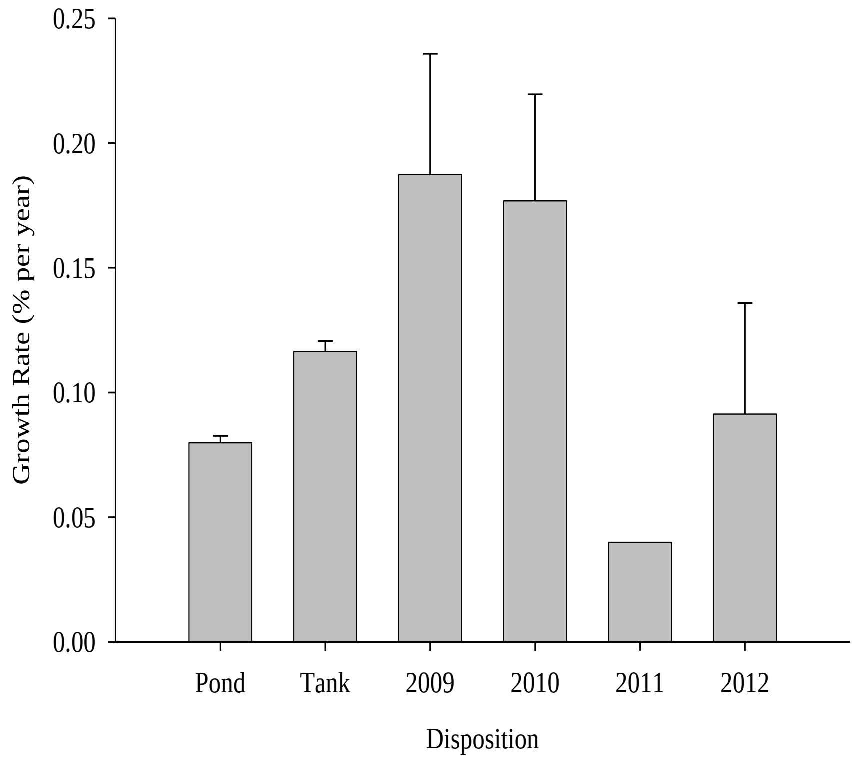

Figure 1-4. Average growth rate of MCL of alligator snapping turtles measured under different conditions and by year of recapture after release. The sample size of each group is listed above each bar. Error bars are ± 1 SE.

2. FRESHWATER TURTLE COMMUNITY STRUCTURE AND HABITAT SELECTION IN OKLAHOMA'S NORTHEASTERN RIVERS

2.1 Introduction

Several consistent patterns have been observed through the study of community data. Lawton (1999) described several common patterns including the presence of more smaller species than larger species within assemblages, larger areas will contain more species than smaller areas, species with larger ranges tend to be higher in abundance, species diversity decreases as latitude increases; and systems with higher energy inputs tend to have higher species diversity. Vellend (2010) described four major processes that explain patterns of community assemblages including natural selection that causes differential reproduction, genetic drift that can reduce heterozygosity of local gene pools, speciation that causes an increase in species richness, and dispersal. Despite these patterns, a criticism of community ecology is that it is a "soft science" that is full of many unique patterns with few encompassing laws. Universal laws are difficult to come by because there are many different environments and each has different organisms that are adapted for these environments, and thus the rules and laws tend to be contingent upon any given situation (Lawton, 1999). Ricklefs (2008) went so far as to dismiss the idea of describing local communities altogether in favor of only describing regional classes of communities. A rebuttal to Ricklefs (2008) by Brooker et al. (2009) pointed out that communities are where organisms interact and place selective pressures on one another within the ecosystem, but analyses at larger scales such as at the regional level may mask these patterns. Knowledge of the community structure and interactive effects of environmental variables are essential, especially to describe differences among habitats and to monitor temporal changes in habitat that can lead to changes in community structure. For instance, declines of many freshwater turtles all over the world have been documented, and the causes of these declines vary, including overharvest, habitat loss, competition with nonnative species, and climate change (Moll and Moll, 2004). The implications of many of these population changes are impossible to assess because of the paucity of data related to community assemblages or interactions.

While freshwater turtles are certainly not the only operating unit in their respective habitats, a group of turtles often make up a significant fraction of the total biomass in the habitats in which they occur (Iverson, 1982; Congdon et al., 1986). In a review of turtle biomass in a variety of habitats, Iverson (1982) found turtles constituted a standing biomass comparable to or exceeding that of fish, and at least an order of magnitude greater than that typical of endotherm biomass, and only rivaled by large herbivores in terrestrial systems. Being long-lived and constituting a significant amount of biomass, freshwater turtles also play a vital role in energy and nutrient flow in freshwater ecosystems (Moll and Moll, 2004). In addition, freshwater turtles play a vital role in food web dynamics (Aresco, 2005). Therefore, freshwater turtles serve as a major operating unit of the entire freshwater ecosystem.

Several turtle ecology studies have described the community structure and habitat associations at a single location or of select species in a variety of locations, including kinosternids (Mahmoud, 1969), *Apalone* species (Bury, 1979; Fuselier and Edds, 1994; Barko and Briggler, 2006), *Graptemys* species (Vogt, 1981; Lindeman, 1999; Aresco, 2005), and select species in a tropical stream (Moll, 1990). However, only a few studies have assessed the entire aquatic turtle community assemblages in a variety of locations (Cagle, 1942; Vandewalle and Christiansen, 1996; DonnerWright et al., 1999; Bodie and Semlitsch, 2000; Dreslik et al., 2005;

Atkinson; 2009) and more such studies are needed to address a general lack of freshwater turtle community assemblage data.

Freshwater turtles are of conservation concern in Oklahoma. Sampling efforts were conducted in the 1990s and 2000s to document the extent of population declines in the eastern one-third of the state. Studies have described the aquatic turtle communities with baseline population data in several locations across the state while also measuring environmental data to aid in explaining community patterns (Riedle et al., 2009; Johansen, 2011).

One particular species of conservation concern, *Macrochelys temminckii* (alligator snapping turtle), has been extirpated from much of its native range in the southeastern United States (Pritchard, 2006) and surveys confirmed that viable populations persisted in Oklahoma in just one or two locations (Riedle et al. 2005; 2006). In response to declines elsewhere in the state, a reintroduction effort was initiated in the Caney River in northeastern Oklahoma. Analysis of other suitable release sites was recommended to assess the suitability of other potential release sites (Riedle et al., 2008).

This species exhibits several characteristics that make it a favorable candidate for reintroduction. *Macrochelys temminckii* is long-lived (Ernst and Lovich, 2009), inhabits a variety of river, lake, swamp, and slough habitats (Pritchard, 2006), has a catholic diet (Sloan et al., 1996; Elsey, 2006; East, 2012), produces large clutches, typically ranging from 9 to 61 eggs per clutch with a mean of 27.8 eggs per clutch (Ernst and Lovich, 2009), and is relatively easy to propagate (B. Fillmore, unpublished data). Further, prior studies on translocation and/or reintroduction of *M. temminckii* have reported good growth and body condition a year after release. In Louisiana, a subadult *M. temminckii* that was translocated to a site outside of its natural home range and exhibited modest growth one year after release (Bogosian, 2010). In southern Oklahoma, captive-reared *M. temminckii* released into an oxbow exhibited greater body condition after a year of monitoring than animals from the same cohort that remained in captivity (Moore, 2010).

In addition to assessing habitat suitability, it is important to understand community-level effects of reintroducing an extirpated species, especially when dealing with a large omnivorous species like *M. temminckii*, which has been absent from a community for at least several years. Antagonistic interactions between *M. temminckii* and its closest relative, *Chelydra serpentina* (eastern snapping turtle) have been observed in the field. The two species also have a tendency to prefer sites with a higher amount of submerged woody debris, and therefore competition for these sites is plausible. Resource partitioning in terms of diets between released *M. temminckii* and sympatric wild *Graptemys* species has also been explored (East, 2012). The larger *M. temminckii* can grow to be much larger than any other sympatric turtle species, and predation on other turtle species is plausible. The consequences of reintroduction of a top-level predator can be far-reaching and sometimes unintended, as has been exemplified with the reintroduction of other species. For example, the reintroduction of gray wolves at Yellowstone National Park resulted in a top-down effect of the food chain, where the predation of overrun elk as well as behavior changes of the elk caused riparian plant life to return (Ripple and Beschta, 2003).

The objectives of this project were to sample the turtle community of three rivers in northeastern Oklahoma. In two rivers, our primary objectives were to describe the habitat and aquatic turtle community assemblages and assess the suitability of each river for possible future reintroduction of *M. temminckii*. *Macrochelys temminckii* were already reintroduced in the third river sampled, and we assessed the turtle community structure there in order to make comparisons with those rivers where the species has remained absent.

2.2 Methods

Study Sites—We conducted trapping surveys in northeastern Oklahoma on the Caney, Verdigris, and Spring rivers in 2011 and 2012. The headwaters of the Caney and Verdigris rivers originate in the tallgrass prairie ecoregion in Kansas. The Caney River is impounded at Hulah Lake in Oklahoma approximately 20 river kilometers south of the Kansas border. Once a vivacious area for locals to go for a variety of recreational activities, Hulah Lake has become severely silted in. There are two impoundments on the Verdigris River before it joins the Arkansas River in eastern Oklahoma; the first forms Toronto Reservoir in central Kansas and the second forms Oologah Lake in northeastern Oklahoma. In contrast, the Spring River has headwaters in the Ozarks ecoregion in Missouri, flows through the southeast corner of Kansas, and is impounded at its confluence with the Neosho River to form Grand Lake O' the Cherokees in Oklahoma. The major characteristics of each river system were as follows: the Caney River, Verdigris River, Pond Creek, and Big Creek were characterized by slow-moving current, a substrate that was mostly sand-silt, and turbid water. In contrast, the two reaches of the Spring River that were sampled included two distinct areas. The upstream reach had faster current, a gravelly substrate, and clearer water. All of the three rivers surveyed in this study include nearby development, and agriculture, and support recreational uses such as fishing, camping, swimming, and boating. All three rivers are prone to significant flooding when heavy rains occur upstream, especially near the reservoirs. Two tributaries were also sampled, including Pond Creek (Caney River) and Big Creek (Verdigris River). The extent of the study areas on each river varied due to limitations imposed by the navigability of each river and tributary as well as the location of public access points. At each location we excluded at least 100 m of river adjacent to boat ramps to limit trap theft and possible trapping bias stemming from anthropogenic activities.

Trapping—Traps consisted of four-hoop and three-hoop hoop traps consisting of 76-cm diameter hoops and 2.5-cm square mesh. The traps were stretched by attaching notched PVC to the outermost hoops. Traps were baited with either canned sardines or fresh fish caught in the hoop traps, caught in gill or trammel nets, or provided by local fishermen. Within each study area 67 to 94 sites suitable for setting a trap were identified, and then a random subset of 6–15 were selected and used each day. Traps were set between 13:00–18:00 and checked the following morning. Trapping efforts were alternated daily between the main river channels and tributaries on the Caney and Verdigris rivers.

Data Collection—Each emydid and kinosternid turtle was given a unique combination of scute notches using a rotary tool (adapted from Cagle, 1939). Trionychids and chelydrids were marked with passive integrated transponder (PIT) tags injected into the left femoral region.

The habitat variables measured at each net site included the number of basking sites and submerged structure, water temperature (near the surface and up to 3 m below the surface), dissolved oxygen, conductivity, water clarity, canopy cover, and mid-channel water depth. The number of available basking sights within a 3-m radius of each net was scored on a 0–3 qualitative scale, with 0 indicating no basking sites and 3 representing very high basking site density. The same qualitative scale measured the underwater structure at each site. A depth finder unit (386ci, Humminbird, Eufaula, AL) aided in determining submerged structure in 2012. Canopy cover was measured with a concave densiometer (Lemmon, 1957). In 2011, water depth was measured with a weighted line with demarcations spaced at 10 cm intervals. These data were obtained from a depth finder in 2012.

Analyses—Species diversity was assessed using both the Shannon diversity index and species evenness. Site species composition similarity was assessed using the Sorenson similarity index (Magurran, 1983). The higher the result of the Shannon diversity index, the higher the species diversity is of that location. Species evenness ranges from 0 – 1; results closer to 1 indicate a higher degree of evenness or species abundances. The Sorenson index ranges from 0–1; results closer to 1 indicate a more similar community assemblage. Pair-wise t-tests were run to test for differences between sample sites for results of the Shannon diversity index (Magurran, 1983). The locations of captures were observed using a detrended correspondence analysis (DCA). The DCA designates weighted species scores based on the location of where captures occurred in relation to one another without the inclusion of environmental variables (Hill and Gauch, Jr., 1980). Interaction effects between species found and environmental variables measured were analyzed using canonical correspondence analysis (CCA). The CCA designates weighted species scores as the dependent variable and the environmental factor scores as the independent variables. Performing a CCA relies on prior knowledge of habitat associations of species in order to maximize explanatory power of measured variables (Ter Braak, 1986; Palmer, 1993). Ordination analyses were done using CANOCO software. In order to compare possible species interactions in relation to environmental variables across rivers we only utilized species captured in all three stream systems within our ordination analyses. Additionally, to reduce the influence of rarely captured species, only those species with greater than 10 captures were used. The one exception was the inclusion of *M. temminckii* to test for the influence of the introduction of this species into the Caney River. A Monte-Carlo statistical test was performed for each CCA analysis to see how well the measured environmental variables explained species distribution. Also, each river system was compared in terms of several environmental variables measured to elucidate significant differences in habitat availability using a one-way ANOVA (Minitab version 6). All conclusions were based on a Type I error rate of 0.05.

2.3 Results

A total of 533 net-nights were conducted (1 net-night = one net set for one night) of sampling in 2011, 586 net-nights in 2012, 200 net-nights in 2013, and 115 net-nights in 2014. The amount of effort that we exerted in 2013 decreased because of problems that arose with sustained flooding at our study sites. The catch per unit effort (CPUE) calculated across both years was highest in the Caney River, followed by the Spring River, Big Creek, Pond Creek, and the Verdigris River (Table 2-5). We were only able to sample the northern reach of the Spring River in June 2011 due to low water levels during the remainder of the study. The remaining sampling periods at the Spring River occurred 6.7 km downstream. This reach was much closer to the reservoir and was characterized by deeper, more turbid water and silt substrate.

Nine species were captured (Table 2-4). *Trachemys scripta* was consistently the most abundant species in all of the rivers sampled, followed by *Graptemys ouachitensis* and *Apalone spinifera* (Table 2-4). *Graptemys pseudogeographica* and *Chelydra serpentina* were also captured at major locations (Table 2-4). *Macrochelys temminckii* was only captured at sites where they have been reintroduced, including the Caney River and Pond Creek (Table 2-4). The one individual captured in the Verdigris River was part of a release that occurred in late July of 2012, and so was not used in our analyses. We captured five *Apalone mutica* in our two seasons of sampling, and all were caught at the upstream-most site that we sampled on the Spring River. We captured six *Sternotherus odoratus* (stinkpot turtles) and 13 *Pseudemys concinna* (river cooters) (Table 2-4).

Species diversity did not differ much between the Caney River and its main tributary, Pond Creek (Table 2-2), and data for the two were pooled in 2014 due to high incidence of movement between the two channels. Species diversity between the Verdigris River and its main tributary Big Creek was significantly different ($P = 0.05$) (Table 2-2). All other comparisons of species diversity were significantly different. Unsurprisingly given the similarity in species diversity and their connectedness, the Caney River and Pond Creek exhibited the most similarity in species composition (Table 2-3). Species evenness was greatest at the Caney River and lowest at the Spring River (Table 2-5).

The Caney and Verdigris River DCAs (Figure 2-1) showed that both species of *Graptemys* were closely associated, but this was not the case on the Spring River. At the Spring River, *G. pseudogeographica* was most closely associated with *A. spinifera*. *Macrochelys temminckii* was captured at different sites than *C. serpentina*. The Caney River CCA (Figure 2-1) revealed two axes based on submerged structure and water depth (Axis 1) and canopy cover and basking structure (Axis 2). *Graptemys pseudogeographica* was associated closely with water depth and fell out close to *M. temminckii*, while *C. serpentina* was tied closely to sites with basking structure and increasing depth. Percent variance for the first axis was 69% and the addition of the second axis explained an additional 19%. Results of a Monte-Carlo test confirmed that the environmental data adequately explained species locations (*trace* = 0.045, $F = 1.762$, $P = 0.0320$).

Axis 1 for the Verdigris River CCA (Figure 2-1) was defined by water depth and basking structure, while Axis 2 represented canopy cover and submerged structure. Both species of *Graptemys* were captured at sites with increasing water depth. *Apalone spinifera* and *C. serpentina* were both tied more closely to submerged structure than the other species, and *T. scripta* was located near the middle of the graph. Percent variance for the first axis was 88% and the addition of the second axis explained an additional 10%. Results of a Monte-Carlo test confirmed that the environmental data adequately explained species locations (*trace* = 0.091, $F = 5.236$, $P = 0.0020$).

The Spring River CCA (Figure 2-1) revealed axes based on water depth (Axis 1) and basking structure, submerged structure, and canopy cover (Axis 2). Both species of *Graptemys* and *A. spinifera* were captured at sites with decreasing water depth. *Chelydra serpentina* was closely tied to basking structure, and *T. scripta* was located near the middle of the graph, indicating there was not one environmental factor that explained presence of this species. Percent variance for the first axis was 75% and the addition of the second axis explained an additional 14%. Results of a Monte-Carlo test confirmed that the environmental data adequately explained species locations (*trace* = 0.062, $F = 3.087$, $P = 0.0020$).

Similarities and differences were observed when comparing species locations between river systems. For instance, the CCA graphs indicated that *T. scripta* was a generalist by its central location on each graph. On the other hand, both species of *Graptemys* were found at similar locations as observed on the CCA graphs of the Verdigris River and Spring River, but a separation of both species occurred at the Caney River, indicating possible competitive exclusion either between each species or due to the presence of another species. *Apalone spinifera* was observed with no particular association with other species on both the DCA and CCA graphs for the Caney River and the Verdigris River, but a closer association with both *Graptemys* species was observed in the Spring River. Also, *C. serpentina* was associated with basking structure on the CCA graphs of the Caney and Spring rivers, but at the Verdigris River the species was associated more with submerged structure.

A number of differences were observed when environmental variables were compared among the river systems. Water clarity was significantly greater at the Spring River than at the

Caney or Verdigris rivers (Table 2-1). Canopy cover was significantly greater at the Spring River than at the Caney or Verdigris rivers (Table 2-1). Water depth was significantly greater at the Verdigris River than at the Caney or Spring rivers (Table 2-1). The number of basking sites was significantly greater at the Caney River than at the Verdigris or Spring rivers (Table 2-1). Dissolved oxygen in the water was significantly higher at the Spring River than at the Caney or Verdigris rivers (Table 2-1).

2.4 Discussion

Trachemys scripta consistently dominated the catch at all locations. Other turtle community studies have reported high densities of this generalist species (Dreslik and Phillips, 2005; Stone et al., 2005; Riedle, 2009; Glorioso et al., 2010). Overall, a greater number of *G. ouachitensis* than *G. pseudogeographica* were captured at each site. *Chelydra serpentina* was captured occasionally at every location except the Verdigris River; however, the species was captured regularly in its tributary, Big Creek. A higher amount of submerged woody debris and lentic water characterized this tributary, and both are habitat characteristics commonly associated with this species (Ernst and Lovich, 2009). The fact that *Apalone mutica* was only detected at the upstream-most reach of the Spring River may reflect habitat limitations for the species stemming from the environmental conditions found at that particular reach. *Apalone spinifera* appeared to be more of a habitat generalist than *A. mutica*, as well as captured or observed in all of the other rivers. Similar observations of *A. mutica* as more of a habitat specialist have been made (Williams and Christiansen, 1981; Barko and Briggler, 2006). The capture of *Pseudemys concinna* was curious given that all individuals were adults and that adults of this species are primarily herbivorous (Ernst and Lovich, 2009). The species has historically been a difficult one to capture using baited traps due to its "wary nature and herbivorous diet" (Dreslik, 1997). Cahn (1937) stated that *P. concinna* populations should be sampled over long periods due to its rarity in being captured.

The Shannon diversity index takes into account the abundance of each species captured, and thus the high number of *T. scripta* captured likely contributed to lower scores at each river. The effect is particularly noticeable at the Spring River, where *T. scripta* dominated captures and species diversity and evenness were lowest. The Caney River and Pond Creek, locations where juvenile *M. temminckii* have been released periodically since 2007 (Chapter 1), had the highest overall diversity scores as well as the most similar turtle communities. Furthermore, species richness was similar between the Spring River and the Caney River, but evenness was not similar between these two rivers due to the abundance of *T. scripta*.

Several notable patterns were observed from the ordination analyses. Both species of *Graptemys* were clustered along a gradient on the DCA graph of the Caney River, but were separated on the CCA graph of the Caney River. This observation could be explained a couple of ways. First, it does appear that *G. ouachitensis* was found at sites with similar attributes as the sites that *M. temminckii* was captured, which included moderate water depth and both submerged structure and basking structure. Telemetry studies have indicated that *M. temminckii* prefer habitat sites with submerged structure as well as sites with abundant canopy cover (Harrel et al., 1996; Riedle et al., 2006). *Macrochelys temminckii*, a larger and more aggressive species, could be pushing *G. pseudogeographica* out of preferred habitat. The second explanation for this segregation is that when the measured variable water depth was included, *G. ouachitensis* appeared to be trapped at sites that contained deeper water than at sites where *G. pseudogeographica* was was primarily found. *Graptemys pseudogeographica* reportedly associates with habitat characteristics such as abundant aquatic vegetation, basking sites, and slow currents, and (Fuselier and Edds, 1994). Also, *G. ouachitensis* was captured more often at all locations, and it appears that from the DCA that *G. pseudogeographica* was usually captured with *G. ouachitensis*. The association of both *M. temminckii* and *C. serpentina* with both submerged structure and basking structure is explained by the fact that at many sites, submerged structure also constituted the basking structure that was counted. The Caney River DCA also shows a segregation of both chelydrid species, but the Caney River CCA shows that both were found at sites with similar attributes as commented on above. This indicates that both species prefer the same measured variables but they tend to avoid one another. A common observation at all three rivers was the location of *T. scripta* near the middle of each CCA plot. This result indicates that the species was found across a variety of locations and did not tend to associate with any particular environmental variable included.

The alpha and beta diversity results at each location add vital information to the growing number of aquatic turtle community studies in Oklahoma. The capture information can be used as baseline data to determine locations that need more observation and management. The results of the DCA and CCA analyses for observing species community patterns are helpful, but they are not a means to an end. Rather, the ordination method is a good starting point when looking for possible community patterns that exist at a location. A number of environmental variables were measured for the analysis that could also be used as baseline data to monitor changes that will continue to affect the aquatic turtle communities at the locations sampled. Future studies at these locations should also sample for alpha and beta diversity as well as environmental variables and look for possible temporal changes. Specific species interactions should also be explored for possible competitive exclusion, particularly at locations before and after the reintroduction of *M. temminckii*.

Table 2-1. Mean (± SD) environmental variables collected at each site, by location, pooled over 2011 and 2012. Sampling was restricted to the Caney River in 2013 and 2014. Within rows, means followed by the same letter are not different α = 0.05.

Variable	Caney River	Verdigris River	Spring River
Water Clarity (cm)	45.11 ± 17.74 a	47.85 ± 21.79 a	53.80 ± 15.51 b
Water Temperature	29.50 ± 3.29 a	29.17 ± 4.29 a	29.66 ± 3.31 a
Submerged Structure	0.91 ± 0.81 a	0.81 ± 0.77 a	0.89 ± 0.76 a
% Canopy Cover	35.53 ± 34.60 a	53.33 ± 32.82 b	66.69 ± 34.28 c
Water Depth	388.13 ± 235.52 a	462.90 ± 170.07 b	422.86 ± 209.98 c
Basking Sites	1.06 ± 0.84 a	0.94 ± 0.87 b	0.91 ± 0.83 b
Dissolved Oxygen (mg/L)	5.15 ± 3.37 a	5.49 ± 3.93 a	8.68 ± 3.41 b

Table 2-2. Pair-wise t–tests of the Shannon diversity indices as compared between locations. A result of P = 0.05 or less indicates species diversity between the two sites was significantly different.

Site	Caney River	Pond Creek	Verdigris River	Big Creek
Spring River	t = 270.61 P = 0.0005	t = 11.48 P = 0.0005	t = 8.84 P = 0.0005	t = 6.75 P = 0.005
Big Creek	t = -5.08 P = 0.005	t = -5.14 P = 0.005	t = 1.7 P = 0.05	
Verdigris River	t = 3.51 P = 0.005	t = -3.65 P = 0.005		
Pond Creek	t = -0.29 P = 0.25			

Table 2-3. Sorenson similarity index results. Sites with more similar turtle communities are indicated by values approaching 1.00.

Site	Caney River	Pond Creek	Verdigris River	Big Creek
Spring River	0.80	0.88	0.86	0.77
Big Creek	0.83	0.77	0.73	
Verdigris River	0.77	0.86		
Pond Creek	0.93			

Table 2-4. Total captures of each species at each study location. APSP = *Apalone spinifera*, CHSE = *Chelydra serpentina*, GROU = *Graptemys ouachitensis*, GRPS = *Graptemys pseudogeographica*, MATE = *Macrochelys temminckii*, and TRSC = *Trachemys scripta*

Site	APMU	APSP	CHSE	GROU	GRPS	MATE	PSCO	STOD	TRSC
Caney R.	0	98	10	89	21	41	0	1	420
Pond Cr.	0	121	5	42	37	29	1	2	356
Verdigris R.	0	45	0	202	13	1	3	0	340
Big Cr.	0	91	16	62	8	0	0	0	442
Spring R.	5	48	26	113	7	0	10	3	1085

Table 2-5. Total net nights, catch per unit effort (CPUE), species evenness, and species diversity index value (H') for each location.

Site	Net Nights	CPUE	Species Evenness	H'
Caney	189	3.60	0.61	1.19
Pond Creek	189	3.14	0.58	1.21
Verdigris	203	2.98	0.56	1.00
Big Creek	185	3.35	0.56	0.92
Spring	364	3.56	0.32	0.66

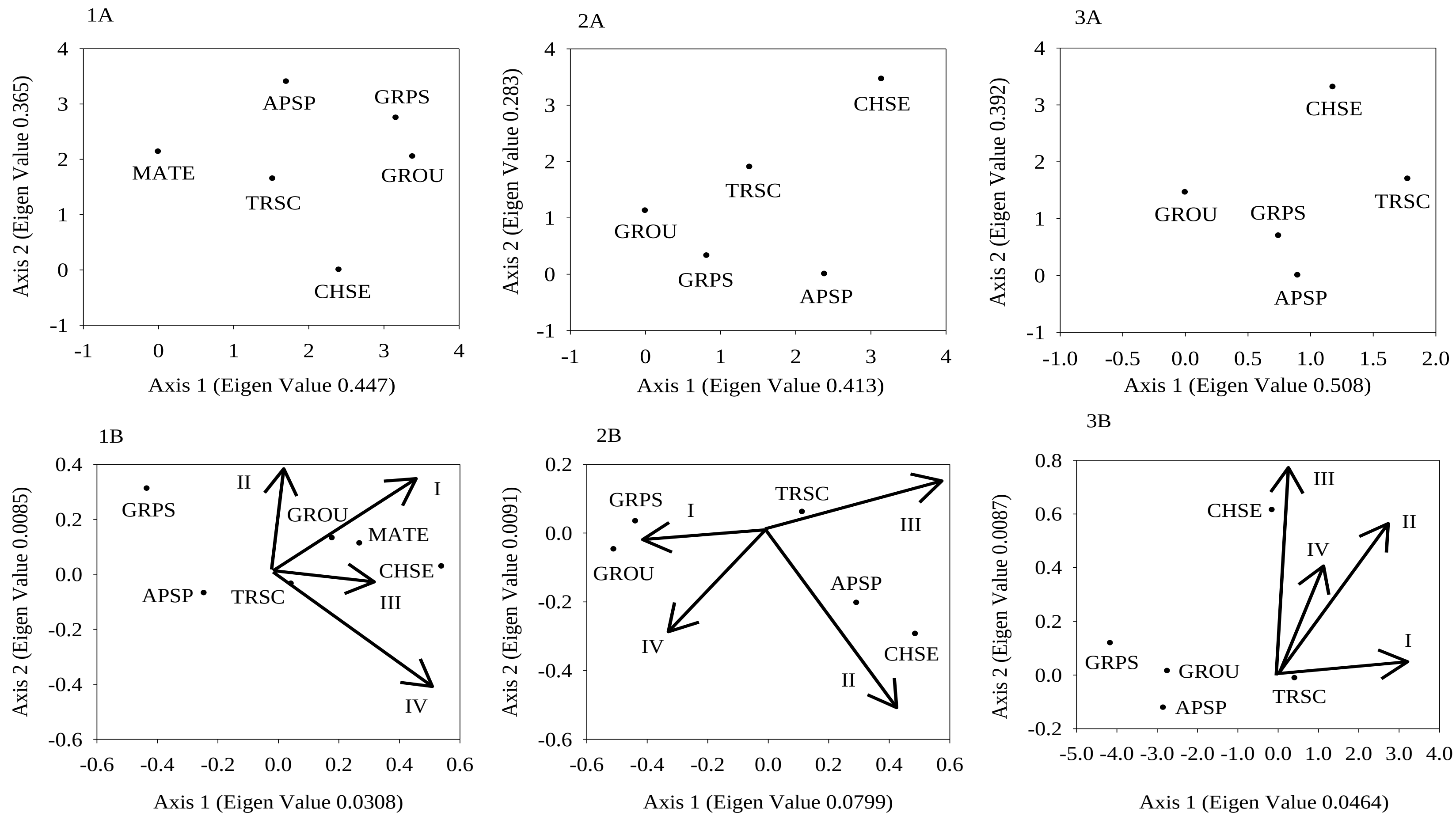

Figure 2-1. DCA (1A–3A) and CCA (1B–3B) plots of the (1) Caney River, (2) Verdigris River, and (3) Spring River. I = water depth, II = submerged structure, III = basking structure, IV = canopy cover. APSP = *Apalone spinifera*, CHSE = *Chelydra serpentina*, GROU = *Graptemys ouachitensis*, GRPS = *Graptemys pseudogeographica*, MATE = *Macrochelys temminckii*, and TRSC = *Trachemys scripta*.

3. SUMMARY

Macrochelys temminckii has experienced declines across its native range. Current reintroduction efforts underway in northeastern Oklahoma aim to establish viable populations in rivers and creeks where the species occurred historically. A necessary step in any reintroduction program is to monitor the fate of released individuals; otherwise these efforts might be futile if the growth and survival rates are low. In relation to reintroduction monitoring, it is important to also monitor the local aquatic turtle communities to assess the effects of reintroducing a species that has been absent for an extended period.

The present study shows that released alligator snapping turtles that were captive-raised at a hatchery using a soft release strategy of reintroduction are surviving and growing at the Caney River. The results from this study along with other studies exploring reintroduction or translocation indicate that *M. temminckii* can survive and thrive in a novel environment. In terms of the species' conservation in the region, this should be very promising news for the future of this population as well as other releases that may take place in the future.

The turtle communities in each of the rivers sampled in this study commonly had a very large population of *T. scripta*, a generalist species that is able to reproduce and thrive in a variety of environmental conditions. On the other hand, a species with a narrower range of habitat requirements, *A. mutica*, was only captured in low numbers in one particular area sampled of the Spring River. The CCA analysis was helpful in observing relationships between species and environmental variables. The graphs indicated a strong relationship between measured environmental variables and presence. Of particular interest to this study were any examples of possible competitive exclusion between released *M. temminckii* and resident turtle communities. The Caney River CCA indicated possible exclusion between *M. temminckii* and *G. pseudogeographica*; the separation between the two species of *Graptemys* was not observed in the Verdigris or Spring rivers. No reintroduction events have taken place at the Spring River, while a late summer 2012 reintroduction in the Verdigris River only included one trapping interval a short time later. Competitive exclusion might be occurring between the two species, and this observation will need to be monitored and investigated further.

In the future, more information can be obtained concerning the survival and movements of released *M. temminckii*. Projects are planned involving following released individuals via radio telemetry with the goal to fill in missing gaps of information. The species is notoriously cryptic and difficult to detect using standard methods of baited funnel net trapping. The captures that were recorded of the entire aquatic turtle community at each location can be used as baseline reference data for future ecological evaluations. In addition, the data collected from the suite of environmental variables measured throughout this study can also be used as baseline data to monitor changes that may occur at these locations in the future.

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Appendix 1. Total captures of aquatic turtles on the Caney River in northern Oklahoma in 2011. MCL = mid-line straight carapace length. Metrics are expressed as $\bar{x} \pm 1$ SE.

Site	Species	Sex class	n	MCL (mm)	Mass (g)
Caney River	*A. spinifera*	M	20	184.64 ± 4.22	595.00 ± 25.63
		F	19	277.04 ± 13.50	2011.22 ± 260.24
		U	5	156.8 ± 7.80	382.00 ± 50.36
	C. serpentina	M	4	296.25 ± 14.36	10337.50 ± 3064.81
		F	3	292.43 ± 9.08	6370.00 ± 614.19
		U	0	-	-
	G. ouachitensis	M	24	98.76 ± 1.64	123.67 ± 8.35
		F	18	162.73 ± 6.15	618.33 ± 57.80
		U	2	80.90 ± 16.10	80.00 ± 35.00
	G. pseudogeographica	M	10	103.33 ± 3.86	110.80 ± 8.35
		F	5	149.44 ± 9.74	441.60 ± 65.86
		U	0	-	-
	M. temminckii	M	0	-	-
		F	0	-	-
		U	17	175.59 ± 4.90	1458.82 ± 152.66
	T. scripta	M	122	177.66 ± 2.35	780.09 ± 26.29
		F	98	189 ± 3.74	1060.59 ± 50.07
		U	4	84.33 ± 7.69	83.00 ± 9.19
Pond Creek	*A. spinifera*	M	20	173.89 ± 3.55	517.60 ± 35.75
		F	21	291.88 ± 14.58	2377.14 ± 322.84
		U	4	194.18 ± 44.17	785.00 ± 462.05
	C. serpentina	M	0	-	-
		F	2	276.15 ± 6.85	3078.50 ± 1721.50
		U	0	-	-
	G. ouachitensis	M	15	101.65 ± 2.03	124.47 ± 7.15
		F	11	146.95 ± 14.41	552.46 ± 107.52
		U	1	76.15 ± n/a	68.00 ± n/a
	G. pseudogeographica	M	8	108.43 ± 2.35	149.25 ± 11.46
		F	10	155.14 ± 9.22	489.20 ± 76.14
		U	1	65.80 ± n/a	45.00 ± n/a
	M. temminckii	M	0	-	-
		F	0	-	-
		U	16	201.92 ± 5.32	2202.00 ± 197.40
	P. concinna	M	0	-	-
		F	1	127.20 ± n/a	270.00 ± n/a
		U	0		
	S. odoratus	M	0	-	-
		F	1	106.00 ± n/a	210.00 ± n/a
		U	0	-	-
	T. scripta	M	91	167.21 ± 3.50	700.32 ± 30.74
		F	97	185.62 ± 3.77	1009.03 ± 46.76
		U	3	90.33 ± 6.84	131.33 ± 34.34

Appendix 2. Total captures of aquatic turtles on the Verdigris River in northern Oklahoma in 2011. MCL = mid-line straight carapace length. Metrics are expressed as x̄ ± 1 SE.

Site	Species	Sex class	n	MCL (mm)	Mass (g)
Verdigris River	*A. spinifera*	M	0	-	-
		F	1	183.50 ± n/a	670.00 ± n/a
		U	0	-	-
	G. ouachitensis	M	20	99.21 ± 5.13	126.30 ± 6.88
		F	39	169.54 ± 3.4	702.95 ± 34.33
		U	0	-	-
	G. pseudogeographica	M	3	102.30 ± 1.33	128.33 ± 15.90
		F	4	175.75 ± 21.16	850.00 ± 320.31
		U	0	-	-
	T. scripta	M	79	187.90 ± 2.91	952.29 ± 36.50
		F	31	198.96 ± 5.78	1181.00 ± 77.09
		U	0	-	-
Big Creek	*A. spinifera*	M	14	163.53 ± 7.02	460.69 ± 53.53
		F	26	290.55 ± 13.01	2447.40 ± 299.81
		U	0	-	-
	C. serpentina	M	1	314.00 ± n/a	9800.00 ± n/a
		F	1	292.00 ± n/a	6200.00 ± n/a
		U	1	128.00 ± n/a	500.00 ± n/a
	G. ouachitensis	M	23	96.09 ± 1.77	111.48 ± 4.77
		F	19	175.57 ± 6.92	772.63 ± 62.58
		U	4	66.30 ± 3.78	60.75 ± 8.93
	G. pseudogeographica	M	1	104.50 ± n/a	150.00 ± n/a
		F	7	163.69 ± 15.57	648.75 ± 123.67
		U	0	-	-
	T. scripta	M	182	185.45 ± 2.22	918.41 ± 25.91
		F	47	166.46 ± 6.18	794.64 ± 81.33
		U	1	94.65 ± n/a	142.00 ± n/a

Appendix 3. Total captures of aquatic turtles on the Spring River in northern Oklahoma in 2011. MCL = mid-line straight carapace length. Metrics are expressed as $\bar{x} \pm 1$ SE.

Site	Species	Sex class	n	MCL (mm)	Mass (g)
Spring River	*A. mutica*	M	4	202.93 ± 10.44	764.50 ± 143.94
		F	2	252.6 ± 13.30	1320.00 ± 250.00
		U	0	-	-
	A. spinifera	M	6	165.77 ± 7.62	465.50 ± 60.47
		F	22	264.8863 ± 15.71	2117.82 ± 327.23
		U	2	187.00 ± 8.00	564.00 ± 114.00
	C. serpentina	M	2	285.00 ± 37.00	5725.00 ± 1175.00
		F	5	268.66 ± 22.19	4718 ± 832.84
		U	0	-	-
	G. ouachitensis	M	31	107.39 ± 3.48	168.84 ± 28.96
		F	26	171.11 ± 4.41	688.92 ± 51.96
		U	1	72.00 ± n/a	58.00 ± n/a
	G. pseudogeographica	M	1	117.10 ± n/a	213.00 ± n/a
		F	2	207.00 ± 15.00	1187.00 ± 303.00
		U	0	-	-
	P. concinna	M	1	224.00 ± n/a	1400.00 ± n/a
		F	6	238.42 ± 12.42	1731.67 ± 275.25
		U	0	-	
	S. odoratus	M	2	94.35 ± 4.25	110.00 ± 20.00
		F	0	-	-
		U	0	-	-
	T. scripta	M	328	178.76 ± 1.29	796.19 ± 15.30
		F	128	199.19 ± 2.33	1142.64 ± 30.92
		U	3	99.17 ± 2.74	163.33 ± 23.33

Appendix 4. Total captures of aquatic turtles on the Caney River in northern Oklahoma in 2012. MCL = mid-line straight carapace length. Metrics are expressed as $\bar{x} \pm 1$ SE.

Site	Species	Sex class	n	MCL (mm)	Mass (g)
Caney River	*A. spinifera*	M	11	179.28 ± 5.35	606.36 ± 47.36
		F	45	274.13 ± 11.17	2177.84 ± 177.16
		U	0	-	-
	C. serpentina	M	1	293.00 ± n/a	7500.00 ± n/a
		F	2	269.75 ± 4.25	4925.00 ± 75.00
		U	1	300.30 ± n/a	6000.00 ± n/a
	G. ouachitensis	M	16	103.34 ± 4.81	160.63 ± 36.33
		F	23	162.60 ± 5.48	619.30 ± 55.30
		U	1	56.90 ± n/a	40.00 ± n/a
	G. pseudogeographica	M	4	95.25 ± 4.14	101.25 ± 12.64
		F	7	173.54 ± 6.59	718.57 ± 63.07
		U	0	-	-
	M. temminckii	M	0	-	-
		F	0	-	-
		U	24	189.66 ± 5.31	1898.75 ± 168.14
	S. odoratus	M	0	-	-
		F	1	107.30 ± n/a	130.00 ± n/a
		U	0	-	-
	T. scripta	M	116	162.92 ± 2.77	657.19 ± 30.02
		F	80	178.96 ± 4.14	941.06 ± 57.32
		U	0	-	-
Pond Creek	*A. spinifera*	M	18	181.83 ± 2.95	635.00 ± 25.30
		F	61	287.01 ± 7.05	2216.48 ± 134.51
		U	1	120.00 ± n/a	245.00 ± n/a
	C. serpentina	M	2	304.90 ± 15.60	9250.00 ± 3250.00
		F	1	286.00 ± n/a	5000.00 ± n/a
		U	0	-	-
	G. ouachitensis	M	4	99.00 ± 2.56	130.00 ± 9.13
		F	8	176.39 ± 5.93	768.75 ± 67.11
		U	3	57.07 ± 3.60	31.00 ± 9.54
	G. pseudogeographica	M	3	112.13 ± 6.51	181.67 ± 29.49
		F	16	155.3 ± 10.25	609.38 ± 95.10
		U	0	-	-
	M. temminckii	M	0	-	-
		F	0	-	-
		U	13	203.87 ± 6.78	2225.00 ± 250.38
	S. odoratus	M	0	-	-
		F	1	95.80 ± n/a	155.00 ± n/a
		U	0	-	-
	T. scripta	M	96	174.11 ± 2.63	770.63 ± 30.87
		F	69	192.63 ± 4.23	1074.64 ± 53.52
		U	0	-	-

Appendix 5. Total captures of aquatic turtles on the Verdigris River in northern Oklahoma in 2012. MCL = mid-line straight carapace length. Metrics are expressed as $\bar{x} \pm 1$ SE.

Site	Species	Sex class	n	MCL (mm)	Mass (g)
Verdigris River	*A. spinifera*	M	12	167.97 ± 7.09	572.50 ± 55.16
		F	29	221.38 ± 11.35	1251.38 ± 206.36
		U	0	-	-
	G. ouachitensis	M	55	99.83 ± 0.88	128.32 ± 3.05
		F	88	143.27± 4.59	485.63 ± 35.07
		U	1	59.50 ± n/a	25.00 ± n/a
	G. pseudogeographica	M	2	102.25 ± 8.25	130.00 ± 0
		F	5	174.56 ± 9.73	666.00 ± 94.21
		U	0	-	-
	M. temminckii	M	0	-	-
		F	0	-	-
		U	1	126.00 ± n/a	580.00 ± n/a
	P. concinna	M	2	230.00 ± 22.00	1362.50 ± 287.50
		F	0	-	-
		U	0	-	-
	T. scripta	M	158	181.95 ± 2.26	890.66 ± 25.86
		F	77	192.51 ± 3.82	1085.71 ± 53.67
		U	1	180.00 ± n/a	860.00 ± n/a
Big Creek	*A. spinifera*	M	22	163.31 ± 4.85	508.18 ± 41.85
		F	34	225.51 ± 12.20	1369.71 ± 227.08
		U	0	-	-
	C. serpentina	M	6	261.35 ± 22.09	3650.00 ± 610.80
		F	7	252.51 ± 9.09	4514.29 ± 160.99
		U	1	165.50 ± n/a	1150.00 ± n/a
	G. ouachitensis	M	13	103.66 ± 1.39	143.08 ± 3.86
		F	13	172.65 ± 9.05	730.77 ± 81.77
		U	0	-	-
		U	0	-	-
	T. scripta	M	182	183.43 ± 1.99	884.87 ± 23.46
		F	35	166.85 ± 7.12	782.71 ± 84.62
		U	1	82.10 ± n/a	75.00 ± n/a

Appendix 6. Total captures of aquatic turtles on the Spring River in northern Oklahoma in 2012. MCL = mid-line straight carapace length. Metrics are expressed as x̄ ± 1 SE.

Site	Species	Sex class	n	MCL (mm)	Mass (g)
Spring	*A. spinifera*	M	12	153.73 ±	482.14 ± 95.37
River		F	7	13.61	769.09 ± 313.36
		U	0	173.07 ± 16.78	-
	C. serpentina	M	15	283.52 ± 12.65	6253.33 ± 809.42
		F	7	246.17 ± 19.69	4750 ± 608.51
		U	0	-	-
	G. ouachitensis	M	19	104.49 ± 1.51	147.79 ± 6.67
		F	35	174.49 ± 5.19	758.44 ± 53.39
		U	2	61.15 ± 52.50	52.50 ± 27.50
	G. pseudogeographica	M	4	105.00 ± 3.89	121.25 ± 17.84
		F	1	215.70 ± n/a	1250 ± n/a
		U	0	-	-
	P. concinna	M	2	195.00 ± 8.00	905.00 ± 85.00
		F	1	171.00 ± n/a	650.00 ± n/a
		U	0	-	-
	S. odoratus	M	1	105.00 ± n/a	170.00 ± n/a
		F	0	-	-
		U	0	-	-
	T. scripta	M	447	183.76 ± 1.09	875.08 ± 12.88
		F	172	193.14 ± 2.43	1089.07 ± 31.38
		U	0	-	-

Appendix 7. Total captures of aquatic turtles on the Caney River and Pond Creek in northern Oklahoma in 2013. MCL = mid-line straight carapace length. Metrics are expressed as x̄ ± 1 SE.

Site	Species	Sex class	n	MCL (mm)	Mass (g)
Caney River & Pond Creek (combined)	*A. spinifera*	M	29	178.14 ± 5.77	540.40 ± 53.11
		F	39	256.79 ± 9.25	1803.97 ± 209.92
		U	-	-	-
	C. serpentina	M	1	251.5 ± NA	9050.00 ± NA
		F	3	290.33 ± 9.51	8666.67 ± 1440.29
		U	4	294.75 ± 15.07	6516.67 ± 1303.95
	G. ouachitensis	M	37	99.61 ± 1.78	123.64 ± 8.22
		F	47	153.54 ± 5.67	545.02 ± 44.71
		U	2	35.50 ± 0.50	8.00 ± 0.00
	G. pseudogeographica	M	13	99.56 ± 1.96	122.38 ± 6.53
		F	17	160.78 ± 12.52	659.38 ±105.81
		U	0	-	-
	M. temminckii	M	-	-	-
		F	-	-	-
		U	49	203.11 ± 4.52	2477.33 ± 273.02
	T. scripta	M	157	169.22 ± 2.05	697.69 ± 23.37
		F	96	191.20 ± 2.95	1061.36 ± 42.39
		U	4	121.88 ± 20.56	273.00 ± 75.99
	P. concinna	M	1	225 ± NA	1225 ± NA
		F	1	266 ± NA	2320 ± NA
		U	0	-	-

Appendix 8. Total captures of aquatic turtles on the Caney River and Pond Creek in northern Oklahoma in 2013. MCL = mid-line straight carapace length. Metrics are expressed as x̄ ± 1 SE.

Site	Species	Sex class	n	MCL (mm)	Mass (g)
Caney River & Pond Creek (combined)	*A. spinifera*	M	23	180.70±19.66	541.09±181.10
		F	80	286.44±56.08	2166.33±1183.93
		U	0	-	-
	C. serpentina	M	12	277.79±58.69	5770.0±2425.22
		F	7	262.14±64.28	5162.86±2244.15
		U	0	-	-
	G. ouachitensis	M	7	94.86±19.10	110.71±71.09
		F	21	156.79±43.89	607.38±345.68
		U	5	86.88±7.28	73.33±37.86
	G. pseudogeographica	M	21	102.21±12.35	126.00±47.03
		F	16	157.57±46.62	626.25±414.42
		U	4	81.00±3.58	58.75±28.10
	M. temminckii	M	0	-	-
		F	0	-	-
		U	67	223.04±34.05	2869.00±1313.62
	T. scripta	M	283	171.82±30.23	745.67±360.37
		F	240	187.31±36.32	995.06±456.24
		U	2	126.50±12.02	127.00±131.52
	P. concinna	M	0	-	-
		F	0	-	-
		U	1	102.00±NA	160.00±NA
	S. odoratus	M	1	109.00±NA	200.00±NA
		F	1	99.00±NA	150.00±NA
		U	0	-	-