LOWER RIO GRANDE VALLEY FRESHWATER TURTLE POPULATIONS:

THREE DECADES OF CHANGE

THESIS

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ABSTRACT

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Substantial commercial harvesting of wild freshwater turtles has occurred in the lower Rio Grande Valley of Texas since the 1990s. State regulations were created in 2007 to eliminate turtle harvesting in public waters, while common turtle species have no harvest protection in private waters. In addition to harvest, road mortality may be increasing due to extensive human population growth since the 1970s. I repeated a study conducted in 1976 to determine if demographic changes have occurred in freshwater turtle populations over the last three decades. Original trapping locations were re-located and when possible re-trapped with similar trapping effort using baited hoop nets. Original locations rendered unsuitable by anthropogenic or natural changes were replaced with proximal or similar locations. Species, sex, carapace length and width, plastron length and width, body depth, and weight were recorded for individual turtles. Data were analyzed for red-eared slider (*Trachemys scripta elegans*) and Texas spiny softshell (*Apalone spinifera emoryi*) captures. Capture-rates and carapace lengths were compared using unequal variance *t*-tests or randomization tests, adult sex-ratios were compared using Chi-square goodness-of-fit tests, and correlations between red-eared slider carapace lengths and roads were tested using Spearman Rank Correlation Coefficient tests. The 1976 data were analyzed by season to determine if the 2008 results were potentially biased. The mean red-eared slider capture-rate was significantly lower in 2008, but only when all counties were included in the analysis. The mean carapace length for male red-eared sliders was significantly shorter in 2008 for Cameron County. Mean carapace lengths for male and female red-eared sliders were significantly longer in 2008 for Hidalgo County and all counties combined. Sex-ratios for red-eared slider adults were typically more male-biased in 2008. The mean carapace length for female Texas spiny softshells was significantly longer in 2008, and the adult sex-ratio was significantly more male-biased. A significant positive weak correlation was detected for carapace lengths and road density within 1 km and 5 km of trapping locations for female red-eared sliders. Capture-rates and carapace lengths were significantly different between May to July and August to November 1976. Sex-ratios were significantly different between May to July and August to November 1976. The changes detected cannot be attributed solely to harvest. They are likely the result of several factors including harvest, differential mortality, changes in habitat availability, and natural fluctuations.

CHAPTER I

INTRODUCTION

Texas Parks and Wildlife Department (TPWD) instituted regulations in October 2007 (R. Macdonald, Texas Parks and Wildlife Department, personal communication) designed to protect non-game animals from over-harvesting. The regulations were largely a response to substantial commercial harvesting of wild freshwater turtles for domestic and international food markets, traditional Chinese medicine, turtle farms, the pet trade, reptile expositions, zoos and aquariums (Warwick et al. 1990, Asian Turtle Trade Working Group 1999, Fisher 2000, Telecky 2000, Ceballos 2001, Ceballos and Fitzgerald 2004, Fitzgerald et al. 2004, Moll and Moll 2004). Turtles are common food items in many parts of the world and are heavily exploited in the Asian area (Rhodin 2000, Moll and Moll 2004, Georges et al. 2006). The collapse of native Asian turtle populations has created a worldwide turtle market (Williams 1999, Moll and Moll 2004, Guynup 2005), which includes the state of Texas as a major supplier (Close and Seigel 1997, Ceballos 2001, Ceballos and Fitzgerald 2004).

Under the new TPWD regulations, all freshwater turtles are protected from harvesting in public waters (R. Macdonald, Texas Parks and Wildlife Department, personal communication). However, several turtle species remain unprotected in private waters. These species include red-eared sliders (*Trachemys scripta elegans*), commonsnapping turtles (*Chelydra serpentina serpentina*) and softshell turtles (*Apalone spp.*). At least 377,534 freshwater turtles were exported from Texas between 1995 and 2000, with the number of exports increasing annually (Ceballos and Fitzgerald 2004). Of the 16,110 wild-caught turtles reported in 1999, spiny softshells (*Apalone spinifera*) and red-eared sliders (*Trachemys scripta elegans*) accounted for 87.9% of the take. Further, 87.1% of the take came from four counties: Hidalgo (38.5%), Cameron (27.6%), Lamar (17.3%), and Willacy (3.8%) (Ceballos 2001). Concentrated harvest of this magnitude is likely unsustainable, but there are currently no data suggesting proper harvest levels.

Little research has been done addressing harvesting impacts on turtle populations. A recent study in Minnesota resulted in a higher catch-per-unit-effort of painted turtles in non-harvested versus harvested lakes (Gamble and Simons 2004). Close and Seigel (1997) found harvested ponds in Louisiana had significantly smaller male and female red-eared sliders. However, substantial research done in fisheries has shown selective harvesting has been responsible for population declines and alteration of population structures for many fish species. Populations of all five species of Pacific salmon decreased in size throughout the 1900s, presumably as a result of substantial harvest (Ricker 1981). Julliard et al. (2001) demonstrated that fishing was responsible for the majority of mortality in cod (*Gadus morhua*) greater than one-year old and suggested high fishing pressure caused a decrease in the Norwegian Skagerrak cod populations. Selective gill nets caused a reduction in the mean length of European whitefish (*Coregonus lavaretus*) in the Gulf of Finland (Heikinheimo and Mikkola 2004). Long-term selective harvesting resulted in a decrease in size at maturity for North Sea plaice (*Pleuronecetes platessa*) (Rijnsdorf 1993) and caused accelerated growth rates, a reduction in age of maturity, and a shorter life-span in Atlantic (Arctic-Norwegian) cod (*Gadus morhua morhua*) (Borisov 1978).

Freshwater turtle populations in south Texas have experienced a long-term decline in suitable habitat due to changes in land-use, water availability, and recent substantial harvesting. Following the introduction of the railroad system in the early 1900s, agricultural production increased greatly in south Texas (Levine 2007). Over 90% of native woodlands in Cameron County were lost between 1930 and 1983, primarily to agricultural expansion (Tremblay et al. 2005). Substantial human population growth has occurred in the lower Rio Grande Valley over the last three decades (U.S. Census Bureau 1982, 2007), resulting in heavy urbanization. The growing population has resulted in high road densities and traffic. Centerline distances for Cameron, Hidalgo, and Willacy counties in 2006 were 1,032.6, 1,277.4, and 355.6 km, respectively (Texas Department of Transportation 2008). The daily distances traveled for Cameron, Hidalgo, and Willacy counties in 2006 were 9,240,750, 17,395,779, and 716,176 km, respectively. High road densities and usage are now commonplace throughout the United States, such that an estimated 22% of the contiguous United States is ecologically altered by roads (Forman 2000).

Overall, rainfall has remained steady since the 1960s, with a mean monthly precipitation of 5.5 cm in Cameron County (National Oceanic and Atmospheric Administration (NOAA) 2008) (Fig. 1). However, precipitation can vary substantially from year-to-year, resulting in unreliable sustainability of turtle habitats. The mean monthly precipitation from 1973 through November 1976 was 6.3 cm; whereas, the mean monthly precipitation from 2005 through June 2008 was only 4.2 cm (Fig. 2-3) (NOAA 2008). Furthermore, as urban populations grow, water storage rights are being transferred from agricultural to urban sectors (Levine 2007), reducing habitat availability in agricultural areas.

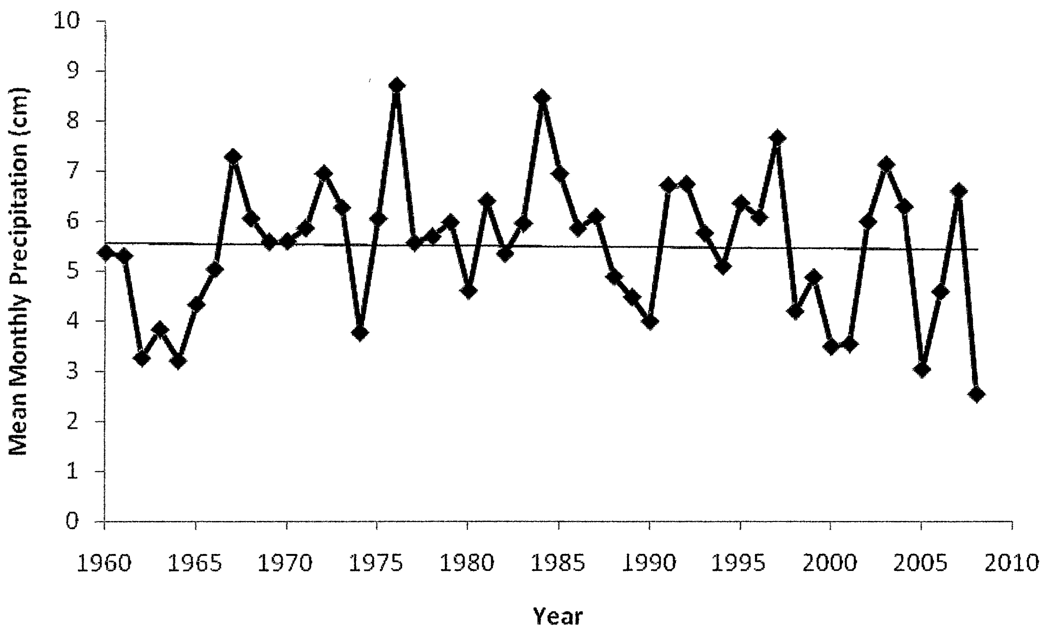


Figure 1. Mean monthly precipitation in Cameron County from 1960 through June 2008. Long-term precipitation has remained nearly constant with an average of 5.5 cm per month, despite substantial variation between years.

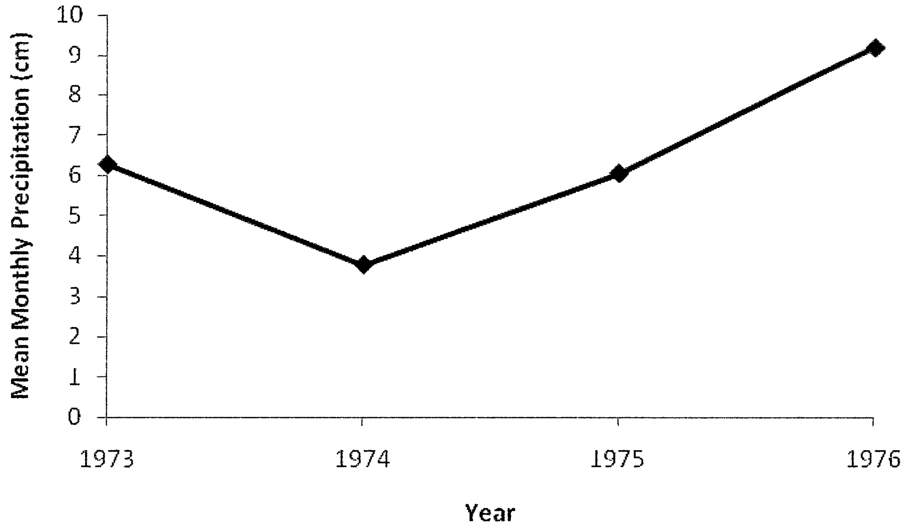


Figure 2. Mean monthly precipitation in Cameron County from 1973 through November 1976. The mean monthly precipitation for this time period was 6.3 cm.

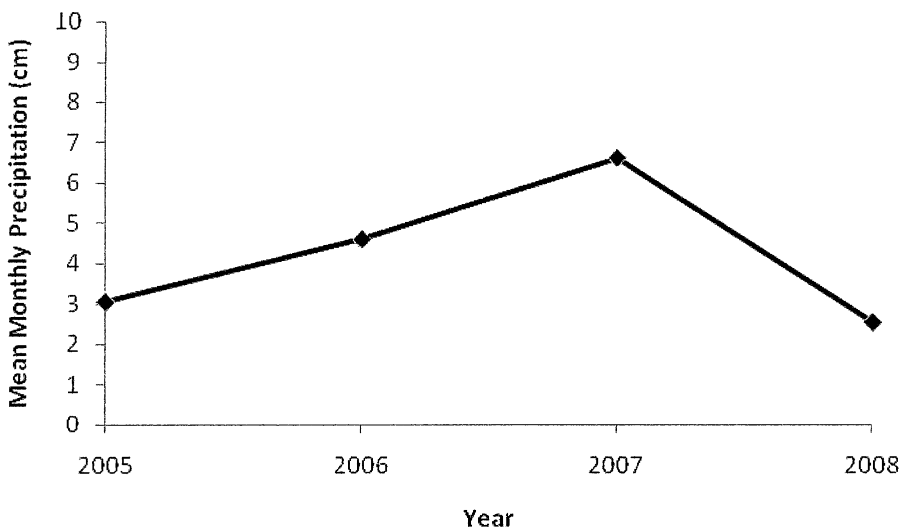


Figure 3. Mean monthly precipitation in Cameron County from 2005 through June 2008. The mean monthly precipitation for this time period was 4.2 cm.

Individual turtles are often killed by vehicles on roads (Ashley and Robinson 1996). Differential mortality by sex could result in a change in sex-ratios over time. Adult slider population sex-ratios are controlled by four factors: 1) hatchling sex-ratio; 2) differential mortality; 3) differential immigration and emigration; and 4) differential age of maturity (Gibbons 1990). Incubation temperatures < 29.4°C produce a male-biased sex ratio, incubation temperatures at about 29°C produce an equal sex-ratio, and incubation temperatures > 31°C produce all females (Willingham 2005). Spiny softshells are one of the few turtle species that do not have temperature-dependent sex determination (Bull and Vogt 1979). Sex-ratios for this species are typically 1:1 at birth (Ernst et al. 1994). In sliders, overland movements by males peak during the breeding season, and overland movement by females peak during the nesting season (Morreale et al. 1984). There is evidence that older, melanistic males exhibit a greater tendency to migrate to new breeding locations than younger, non-melanistic males (Thomas and Parker 2000). This life-history characteristic would create site-specific probabilities for male road mortality based on the age structure of a population at a given location. Females move onto land to nest and migrate to more suitable habitats (Ernst et al. 1994). Schubauer et al. (1990) found the mean home range size was 731 m and 401 m, respectively, for male and female red-eared sliders, and home range size was positively correlated with body size for females.

There is evidence that female turtles are disproportionately vulnerable to road mortality. Steen and Gibbs (2004) found both painted turtles (*Chrysemys picta*) and common snapping turtles (*Chelydra serpentina*) had substantially higher male-biased sex-ratios in wetlands surrounded by high road densities compared to wetlands surrounded by low road densities. Haxton (2000) reported higher road mortality for female common snapping turtles in central Ontario, with the number of road mortalities increasing greatly during the nesting season. Gibbs and Steen (2005) performed a meta-analysis on freshwater turtle sex-ratio data and found the proportion of males had increased linearly since 1930, with more male-bias in states with higher road densities. Steen et al. (2006) performed a meta-analysis on reports pertaining to freshwater turtle surveys conducted on and off-roads and found that for aquatic turtles, off-road surveys were male-biased and on-road surveys were female-biased. Therefore, it is likely that road systems jeopardize the persistence of turtle populations (Gibbs and Shriver 2002).

In 1976, Eric Grosmaire surveyed freshwater turtle populations in the lower Rio Grande Valley of Texas for his Master of Science thesis titled '*Aspects of the natural history of freshwater turtles within the lower Rio Grande Valley of Texas*' (Grosmaire 1977). Trapping occurred at public and private localities in Cameron, Hidalgo, and Willacy counties. Pertinent information obtained included sex-ratios, size-classes, and population sizes of freshwater turtles in the lower Rio Grande Valley.

The purpose of my study was to compare current freshwater turtle population demographics to those of turtle populations in the lower Rio Grande Valley of three decades ago. This was accomplished by repeating the study conducted by Eric Grosmaire (1977). The objectives were to gain useful information on current population demographics, determine if demographic changes have occurred since 1976, and infer the causes of detected changes.

CHAPTER II

STUDY AREA

This study was conducted in Cameron, Hidalgo, and Willacy counties (Fig. 4). Trapping occurred at 11 discrete locations within these counties. Human populations in the lower Rio Grande Valley have grown considerably since 1976 (U.S. Census Bureau 1982, 2007). The population in Cameron County has increased from 189,400 (~57 people/km^2) in 1976 to 387,717 (~117 people/km^2) in 2006. Hidalgo County has increased from 249,000 (~61 people/km^2) to 700,634 people (~171 people/km^2). Willacy County has experienced little population growth in the last three decades, growing from 17,400 (~9 people/km^2) to 20,645 people (~10 people/km^2). Cameron, Hidalgo, and Willacy counties, respectively, are projected to continue growing over the next three decades to 614,396 (a 36.9% increase), 1,256,080 (a 44.2% increase) and 28,280 (a 27.0% increase) in 2040 (Rio Grande Regional Water Planning Group 2001).

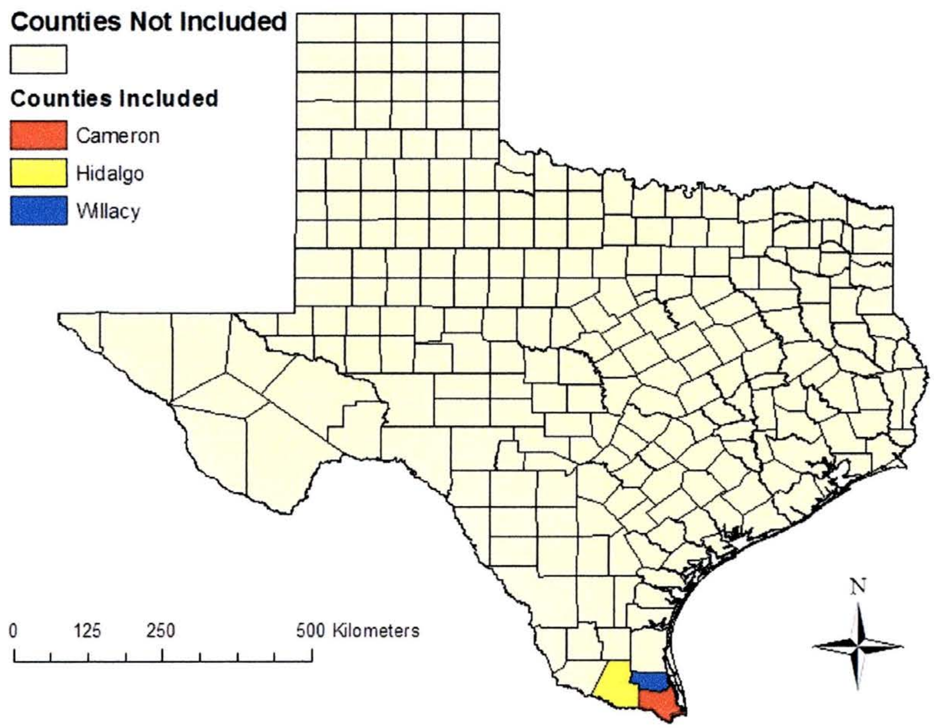


Figure 4. Counties included in this study on changes in freshwater turtle population demographics. Trapping sites within these counties were chosen based on a study conducted in 1976. Cameron and Hidalgo were two of the three counties comprising 78% of wild freshwater turtle harvest in 1999 (Ceballos and Fitzgerald 2004).

CHAPTER III

MATERIALS AND METHODS

The purpose of my study was to obtain data comparable to that gathered by Eric Grosmaire for his thesis in 1976. I re-located and, when possible, re-trapped the same locations with similar trapping effort. Eric Grosmaire trapped sporadically between 21 May 1976 and 15 November 1976. Due to time constraints I did not temporally repeat his trapping effort. The total trapping effort was repeated between 10 May 2008 and 14 June 2008.

Trapping Effort and Locations

Original effort was delineated using Eric Grosmaire's thesis, the original data sheets, and the assistance of James R. Dixon, Eric Grosmaire's thesis advisor. All trapping localities were re-located and photographed (Appendix A). Land-use adjacent to sites was noted. Several trapping locality errors were reported in Eric Grosmaire's thesis and corrected (Appendix B). Accessible locations with currently suitable conditions were re-trapped with similar effort. Locations without currently suitable conditions were replaced with proximal or similar locations and trapped (Table 1, Fig. 5-7).

Table 1. Locations, type of water body, current water status and land-use type surrounding 1976 and 2008 trap sites. Sites are listed from north to south by county. Characters correspond to locations in Figure 1.

1976	2008	County	Coordinates*	Water Body	Status	Title	Land-use
1		Cameron	N26.25166, W097.61370	Pond	Wet	Public	Agricultural
2		Cameron	N26.22903, W097.34863	Pond	Dry	Laguna Atascosa	NWR**
	A	Cameron	N26.22029, W097.60605	Canal	Wet	Abbott Reservoir	Agricultural
3		Cameron	N26.19527, W097.60181	River	Wet	Arroyo Colorado	Industrial
	B	Cameron	N25.85420, W097.39588	Resaca	Wet	Southmost Preserve	Preserve
	C	Cameron	N25.85227, W097.39743	Resaca	Wet	Southmost Preserve	Preserve
	D	Cameron	N25.85032, W097.39867	River	Wet	Rio Grande	Preserve
	E	Cameron	N25.84070, W097.38863	River	Wet	Rio Grande	Preserve
	F	Hidalgo	N26.29286, W098.13398	Pond	Wet	Edinburg Wetlands	City Park
4		Hidalgo	N26.27031, W097.96054	Canal	Wet	Public	Agricultural
5	G	Hidalgo	N26.17882, W098.38714	Resaca	Wet	Bentsen-Rio Grande	State Park
	H	Hidalgo	N26.16792, W098.37992	Resaca	Wet	Bentsen-Rio Grande	State Park
	I	Hidalgo	N26.18517, W098.37948	Canal	Wet	Bentsen-Rio Grande	State Park
	J	Hidalgo	N26.14711, W097.98901	Pond	Wet	Frontera Audubon	Preserve
6		Hidalgo	N26.07981, W098.14074	Pond	Dry	Santa Ana	NWR**
	K	Hidalgo	N26.07840, W098.13047	Pond	Wet	Santa Ana	NWR**
7	L	Hidalgo	N26.07390, W098.15336	Pond	Wet	Santa Ana	NWR**
	M	Hidalgo	N26.06833, W098.13741	River	Wet	Santa Ana	NWR**
8		Willacy	N26.50363, W097.48840	Canal	Wet	Public	Undeveloped
9		Willacy	N26.48814, W097.65675	Pond	Dry	Private	Agricultural
10		Willacy	N26.48223, W097.74853	Pond	Wet	Private	Agricultural
11		Willacy	N26.46308, W097.70819	Pond	Dry***	Private	Agricultural
12	N	Willacy	N26.45587, W097.76271	Pond	Wet	Frank Quintero	Cattle pasture
13		Willacy	N26.45451, W097.75909	Pond	Dry	Private	Agricultural
14		Willacy	N26.45341, W097.93415	Canal	Wet	Public	Agricultural
15		Willacy	N26.45316, W097.61968	Pond	Dry	Private	Agricultural
16	O	Willacy	N26.45218, W097.77656	Pond	Wet	Public	Agricultural
	P	Willacy	N26.43910, W097.61353	Canal	Wet	Public	Pasture
17		Willacy	N26.41776, W097.59951	Resaca	Dry	Private	Agricultural
18		Willacy	N26.40231, W097.61563	Canal	Wet	Public	Agricultural
19		Willacy	N26.40108, W097.70241	Resaca	Dry	Private	Agricultural
20		Willacy	N26.40091, W097.71147	Canal	Dry	Public	Agricultural
21		Willacy	N26.39356, W097.71952	Canal	Wet	Public	Agricultural
22		Willacy	N26.38800, W097.71957	Pond	Dry	Private	Agricultural
	Q	Willacy	N26.35752, W097.58618	Canal	Wet	Public	Agricultural
23		Willacy	N26.35751, W097.73042	Pond	Dry	Private	Agricultural
24		Willacy	N26.34240, W097.78249	Pond	Wet	Private	Residential
25		Willacy	N26.33024, W097.47494	Creek	Wet	Arroyo Colorado	Residential

*Coordinates are in decimal degrees using the WGS 84 datum
**National Wildlife Refuge
***Pond appeared dry from a distance

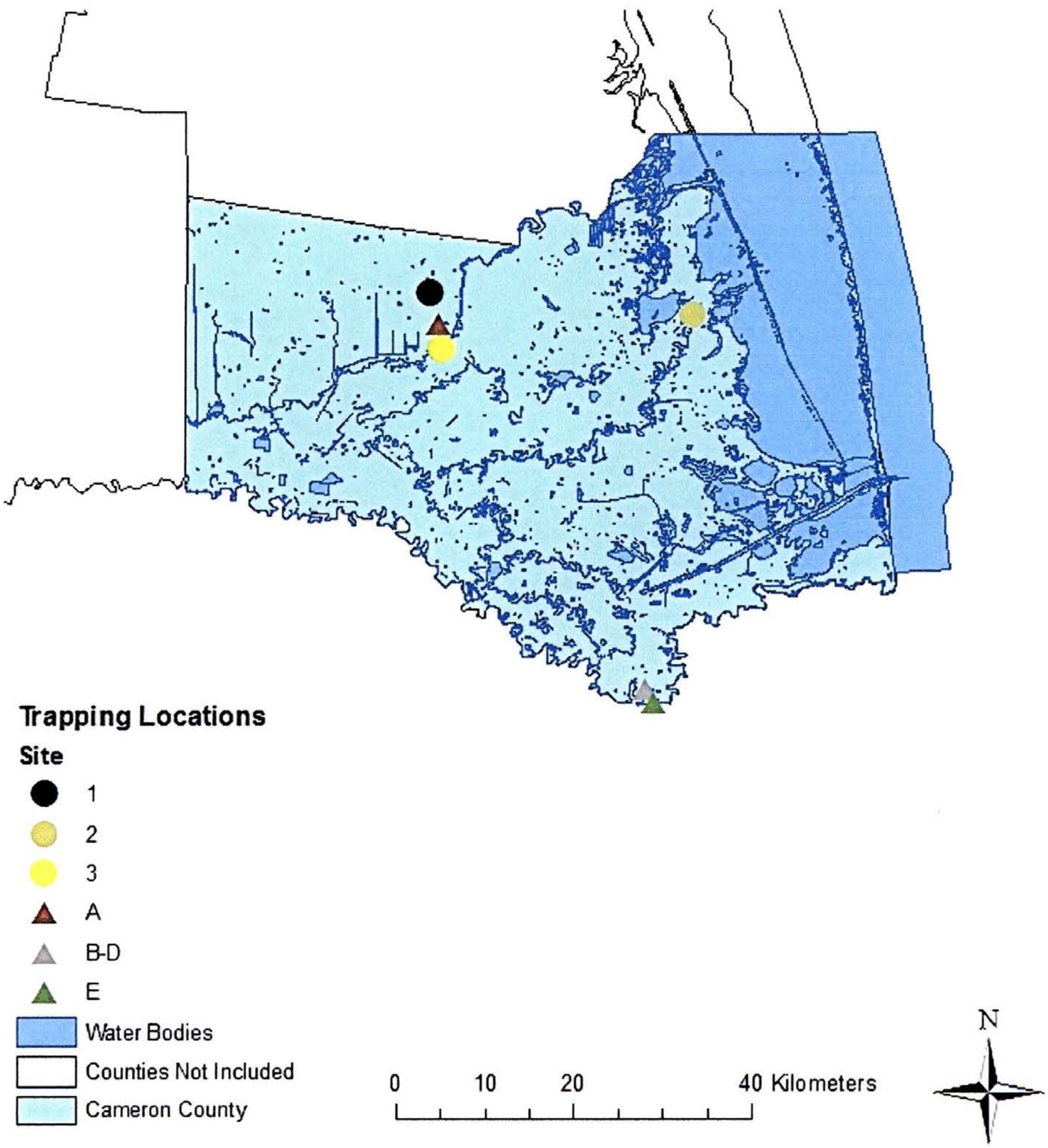


Figure 5. Turtle trapping localities in Cameron County. Due to the comparable size and protected status, Southmost Preserve (B-E) was chosen as a replacement for the currently dry Headquarters Pond (2) at Laguna Atascosa National Wildlife Refuge, which accounted for 80% of the 1976 trapping effort in this county.

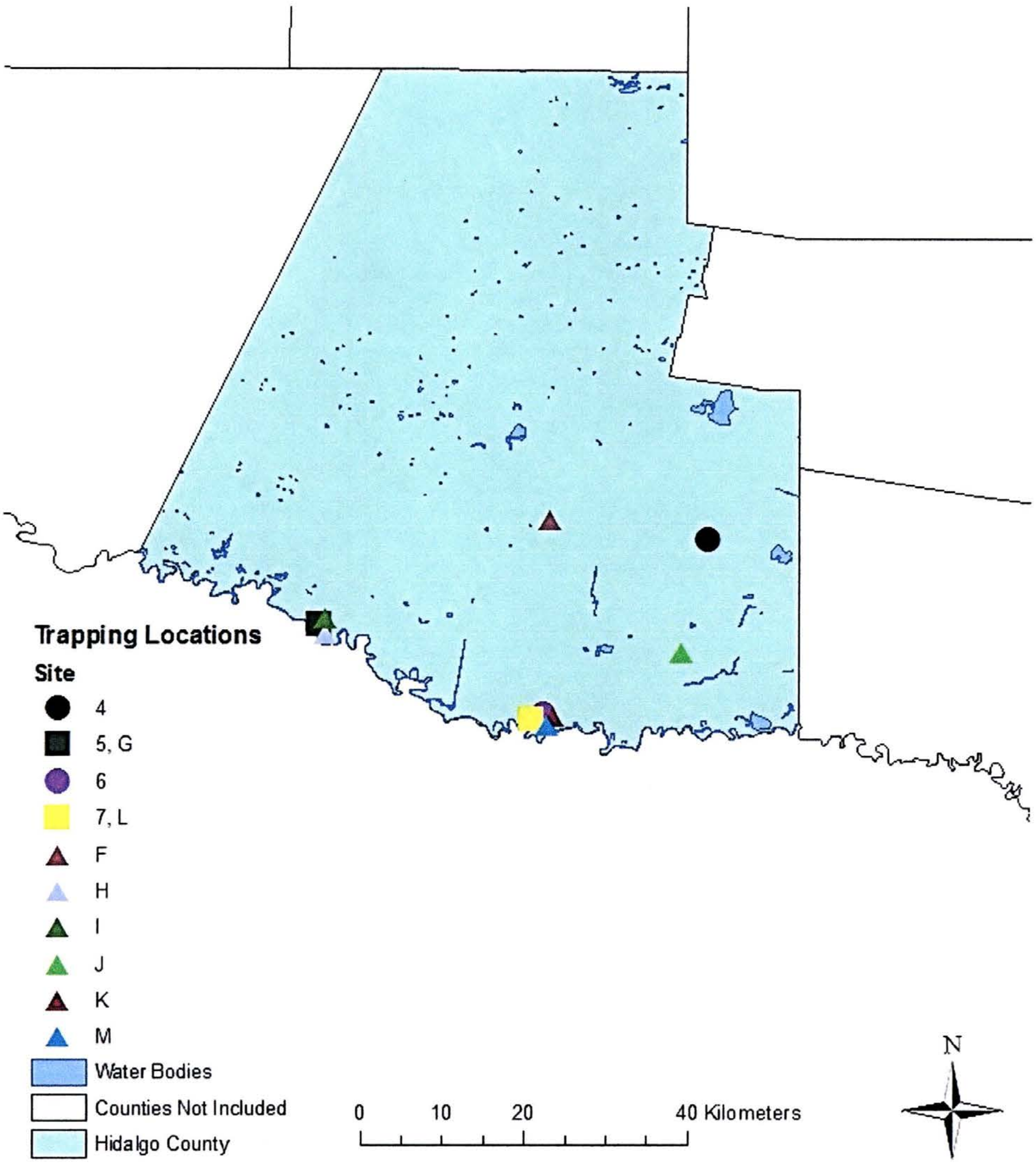


Figure 6. Turtle trapping localities in Hidalgo County. Santa Ana National Wildlife Refuge (6, 7, K, L, M) accounted for 85% of the 1976 trapping effort in this county. I expanded the trapping locations to obtain more representative results for Hidalgo County due to low capture success at Santa Ana.

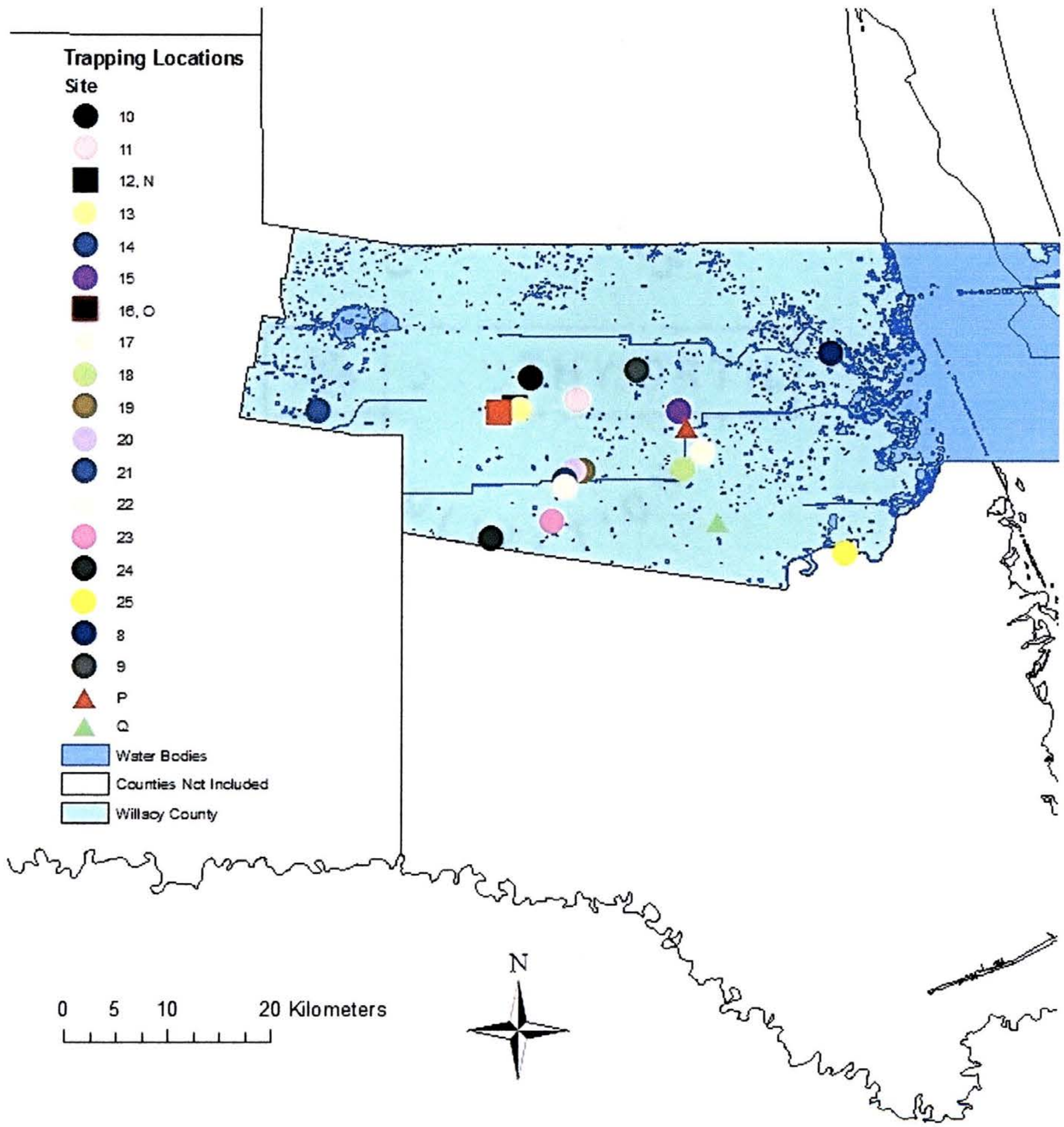


Figure 7. Turtle trapping localities in Willacy County. In 1976, many locations were trapped in this county for short amounts of time. I trapped fewer sites for longer amounts of time, primarily due to currently unsuitable habitat conditions at the majority of the original sites.

Trapping Methods

Eric Grosmaire used twenty 76.2 cm diameter double-throated steel hoop nets for turtle-trapping. I obtained 11 of the original hoop nets, and supplemented the set with twenty-seven 76.2 diameter fiberglass single-throated hoop nets. Therefore, I reduced trapping duration by increasing trap effort at each location. The bait used in the 1976 trapping included canned fish, fresh fish, and beef scraps. I used canned fish, fresh fish, and shrimp. The method of trap placement in 1976 was unknown, but almost certainly followed the general guidelines provided by James R. Dixon, who assisted with trapping in both 1976 and 2008. I placed traps along canal, river, and pond borders; securing traps to reeds or other vegetation. Traps were placed equidistant to one another when possible, with distances between traps ranging from 2 m to 6 m. To avoid capture bias in locations subjected to long-term trapping (i.e., Southmost Preserve and Edinburg Scenic Wetlands), a portion of the traps were moved to new locations within the site every two days.

Data collected in both studies included species, sex, carapace length, carapace width, plastron length, plastron width, body depth, and weight. Length measurements accurate to 1.0 mm were taken in 1976 using homemade vernier calipers laid on a steel rule (Grosmaire 1977). Length measurements accurate to 1.0 cm were taken in 2008 using Haglof® tree calipers (Haglof, Madison, MS). A device using English measurements accurate to 1 oz was used to weigh individuals in 1976 (Grosmaire 1977). Weight measurements accurate to 20 g were taken in 2008 using Pesola® precision scales (Pesola, Baar, Switzerland). For individuals weighing more than 2500 g, a portable scale accurate to 100 g was used.

In both studies, sex was determined using secondary sexual characteristics. Adult male red-eared sliders have elongated foreclaws and the pre-cloacal portion of the tail lies beyond the edge of the carapace (Gibbons and Lovich 1990). The pre-cloacal portion of the tail of male Texas spiny softshells is also substantially longer (Conant and Collins 1998). Small juvenile red-eared sliders (individuals with a carapace length < 90 mm) were not sexed unless obvious male characteristics were expressed. Red-eared sliders and yellow mud turtles (*Kinosternon flavescens flavescens*) were individually marked by notching the carapace (Cagle 1939) in 1976 using a file and in 2008 using a Dremel® (Dremel, Racine, Wisconsin) (Fig. 8). Texas spiny softshells (*Apalone spinifera emoryi*) were individually marked in 1976 using metal fish tags attached to the carapace and in 2008 by imprinting individual numbers into the posterior edge of the carapace using a Dremel.

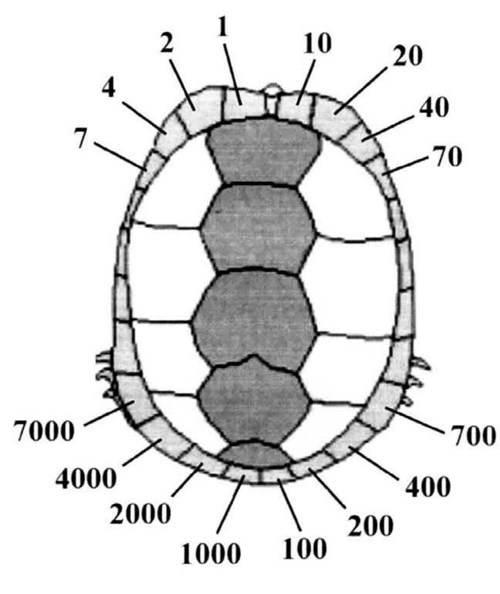


Figure 8. Diagram showing the numbering system used to mark the carapace of turtles in 2008. For example, a turtle to be marked 286 would be notched on the marginals for 200, 70, 10, 4 and 2.

Statistical Analyses

Data sets for red-eared sliders were pooled by county and analyzed as a whole for comparisons of capture-rates and demographic characteristics. I only analyzed the total capture data for Texas spiny softshells due to a low number of captures per county in both studies. Data for captured yellow mud turtles were not analyzed because of a low number of total captures in both studies. Red-eared sliders were classified as juveniles if the plastron length was < 100 mm for males and < 160 mm for females, respectively (Gibbons and Greene 1990). Texas spiny softshells were classified as juveniles if the plastron length was < 88 mm and < 160 mm respectively for males and females (Webb 1962).

I used the carapace rather than the plastron to test for length differences. These two measurements have been shown to be highly correlated (Gibbons and Lovich 1990). Only adults were used in sex-ratio and carapace length comparisons. For sex-ratio comparisons, captures outside of traps (i.e., roadside captures) were excluded from the data sets to reduce capture bias. For comparisons of total captures, all individuals captured in traps were included in analyses. Recaptures were not included in any analyses.

I compared relative abundances by converting the number of captures to a capture-rate per trap day and then compared capture-rates. I used *t*-tests to compare all sample means that were approximately normally distributed (Fowler et al. 1998, Quinn and Keough 2002). The assumption of equal variances was tested using an *F*-ratio test prior to running *t*-tests (Quinn and Keough 2002). When variances were approximately equal ($P \geq 0.05$), I compared means using an unequal variance *t*-test (i.e., unpooled variance *t*-test) (Ruxton 2006). This test has been found to perform as well as a Student's *t*-test when variances are equal (Moser and Stevens 1992), but controls Type 1 errors better when variances are unequal (Ruxton 2006). When variances were found to be unequal ($P < 0.05$), I normalized the data through transformation (Fowler et al. 1998). When the data could not be normalized, I compared means using an unequal variance *t*-test, if both the 1976 and 2008 sample distributions were approximately normal (based on graphical interpretation) (Fowler et al. 1998, Ruxton 2006). If either distribution was not approximately normal, I compared means using a randomization test with 10,000 permutations. This test computes the probability of obtaining a test statistic as great as or greater than the one obtained, which in this case was the difference between two means (Ramsey and Schafer 2002). Means are reported with standard deviations.

I compared sex-ratios using a Chi-square goodness-of-fit test with Yates correction factor (Fowler et al. 1998), using expected frequencies generated with the 1976 sex-ratio data. To determine if season may be a confounding factor for the results, I compared red-eared slider capture-rates, carapace lengths, and sex-ratios for data collected from May to July and August to November of 1976. I could not test this for Texas spiny softshells due to the low number of summer captures in 1976.

I investigated the potential effect of road mortality on red-eared slider populations by performing correlation analyses between male and female carapace lengths and distance to road, road density within 1 km, and road density within 5 km (Table 2). Road data were gathered using ArcGIS® 9.2 (ESRI, Redlands, California) and publicly available data layers from the Texas Natural Resources Information System (TNRIS 2008). Distance to road was measured using the distance measurement tool. Road density was found by buffering trap sites, joining the attribute tables for buffered trap sites and road layers, and calculating distances for road segments within the buffers. Road segments that were located within the buffer, but extended beyond the buffer, were included in the GIS road distance calculation. The excess distances were removed manually using the distance tool. The road data could not be normalized and correlations were analyzed using the non-parametric Spearman rank correlation coefficient test (Fowler et al. 1998). Though not statistically analyzed for significance due to a low number of samples, I also graphed the mean carapace length and sex-ratio for the three locations with at least 30 red-eared slider captures (i.e., Edinburg Scenic Wetlands-Hidalgo County, Frank Quintero Laguna- Willacy County, and Southmost Preserve-Cameron County) with respect to distance to road, road density within 1 km, and road density within 5 km. A regression line was included to determine if a linear relationship existed between mean carapace lengths or sex-ratios and roads for red-eared sliders. I used 95% confidence intervals for all analyses (i.e., $P \leq 0.05$) (Fowler et al. 1998). Statistical analyses were performed using Excel® 2007 (Microsoft, Redmond, WA), JMP® 7 (SAS Institute, Cary, NC), and R 2.7.2 (The R Foundation for Statistical Computing, Vienna, Austria).

Table 2. Distance to road and road density within 1 km and 5 km for 2008 trap sites. Only locations with red-eared slider captures are included.

Trap Site	Distance to Road (m)	1 km Road Density (km)	5 km Road Density (km)
A	420	1.85	155.86
B-C	373	3.65	91.07
D	573	1.48	81.53
E	1122	0	47.48
F	167	16.13	398.53
H	128	5.02	62.47
J	137	33.27	468.62
K	85	6.73	107.73
M	1443	0	82.70
12, N	82	10.84	236.81
16, O	20	11.85	262.78
P	0	4.1	95.01
Q	0	7.6	127.07

CHAPTER IV

RESULTS

Trapping Locations

Eric Grosmaire trapped turtles at 25 locations in Cameron, Hidalgo, and Willacy counties. Only four locations contained suitable conditions and were re-trapped for this study. Eleven of the original trapping locations were either dry or had been converted to other forms of land-use; typically agricultural fields or housing. The other 10 locations were unsuitable for trapping because they contained water levels too low to support turtle populations or use of traps, contained a lack of border vegetation, were inaccessible, or presented a high risk of trap theft. Thirteen sites were chosen from available proximal or similar sites to be used as replacements for the original trap sites.

Turtle Population Comparisons

Eric Grosmaire captured 458 individuals in 1380 trap days. I captured 313 individuals in 1400 trap days (Table 3). Red-eared sliders accounted for 86.7% and 81.8% of captures in 1976 and 2008, respectively.

Table 3. Trapping effort and number of freshwater turtles captured in Cameron, Hidalgo, and Willacy counties in 1976 and 2008.

County	Trap Days 1976	Trap Days 2008	*Red-eared sliders* 1976	*Red-eared sliders* 2008	*Texas spiny softshells* 1976	*Texas spiny softshells* 2008	*Yellow mud turtles* 1976	*Yellow mud turtles* 2008
Cameron	400	412	56	97	11	13	6	1
Hidalgo	820	819	296	109	18	37	1	0
Willacy	160	169	45	50	23	4	2	2
Total	*1380*	*1400*	*397*	*256*	*52*	*54*	*9*	*3*

Red-eared sliders in Cameron County.— A total of 56 red-eared sliders were captured in 1976 in 400 trap days. I captured 97 red-eared sliders in 412 trap days. The capture data were normalized with an arcsin transformation ($F_{11,12}$ = 1.41, P = 0.28). The capture-rate was not different between years (t_{22} = -0.51, P = 0.61). The adult sex-ratio for Cameron County was 1:1.88 (male:female) in 1976 and 1:1.41 in 2008. The sex-ratio was not different between the two years (χ^2_1 = 1.52, 0.10 < P < 0.90 , N = 77).

The mean carapace length for males was 176.4 ± 17.5 mm in 1976 and 162.7 ± 28.7 mm in 2008. The mean carapace length for females was 214.5 ± 16.8 mm in 1976 and 216.0 ± 18.4 mm in 2008. I could not normalize the data for males ($F_{31,17}$ = 2.70, P = 0.02). Using a randomization test, the mean carapace length for males was significantly different (P < 0.01) (Fig. 9). The mean female carapace length was not different (t_{85} = 0.39, P = 0.70) (Fig. 10).

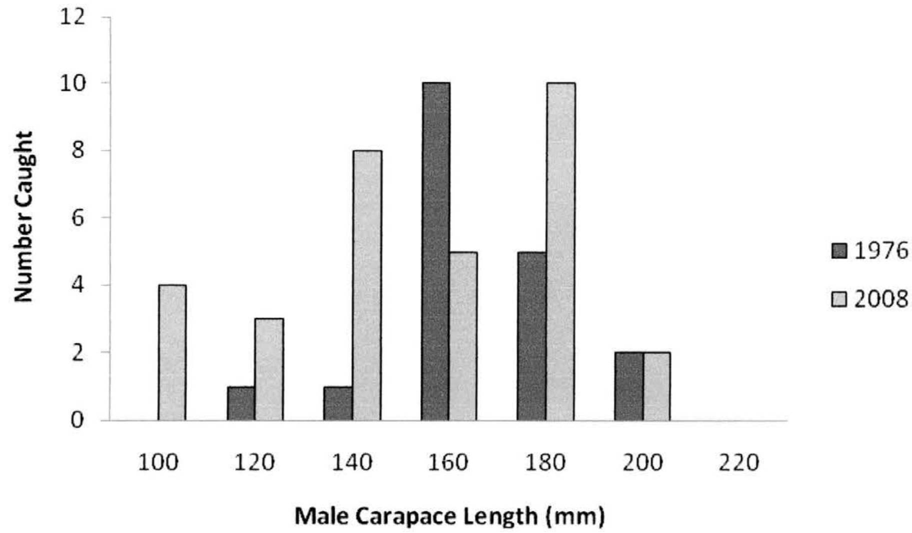


Figure 9. Carapace lengths of adult male red-eared sliders in Cameron County. The mean carapace length in 1976 (176.4 ± 17.5 mm) was significantly longer than the mean carapace length in 2008 (162.7 ± 28.7 mm).

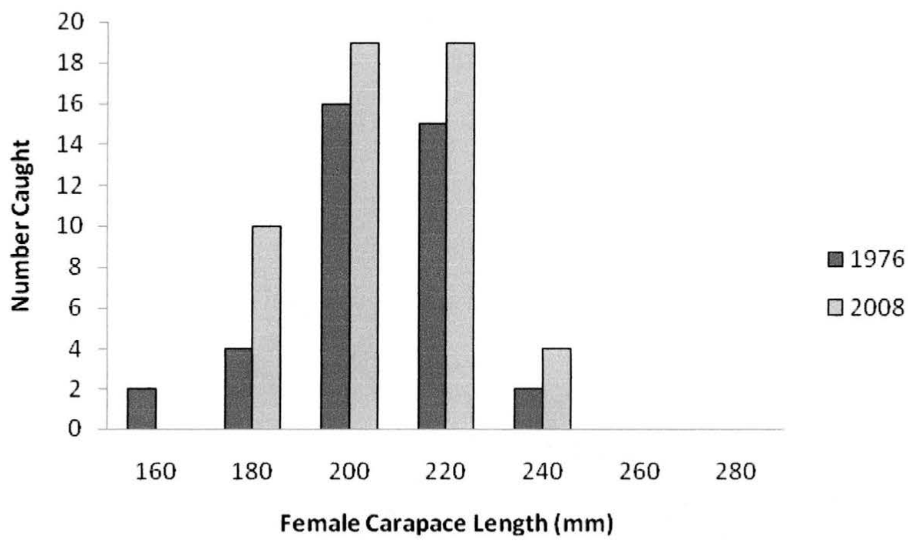


Figure 10. Carapace lengths of adult female red-eared sliders in Cameron County. The mean carapace length was not significantly different between years.

Red-eared sliders in Hidalgo County.— A total of 296 red-eared sliders were captured in 1976 in 820 trap days. I captured 109 red-eared sliders in 819 trap days. I was unable to normalize the data ($F_{28,19} = 47.06$, $P < 0.01$). Using a randomization test, the capture-rate was not different between years ($P = 0.08$). The adult sex-ratio was 1:0.91 (male:female) in 1976 and 1:1.10 in 2008. The sex-ratio was not different between the two years ($\chi^2_1 = 0.89$, $0.10 < P < 0.90$, $N = 84$).

The mean carapace length for males was 155.4 ± 23.4 mm in 1976 and 169.5 ± 29.1 mm in 2008. The mean carapace length for females was 200.5 ± 17.3 mm in 1976 and 227.8 ± 24.9 mm in 2008. I normalized the data for males ($F_{39,136} = 1.46$, $P = 0.06$). The mean carapace length for both males ($t_{56} = 2.61$, $P = 0.01$) (Fig. 11) and females ($t_{60} = 6.81$, $P < 0.01$) (Fig. 12) was significantly different.

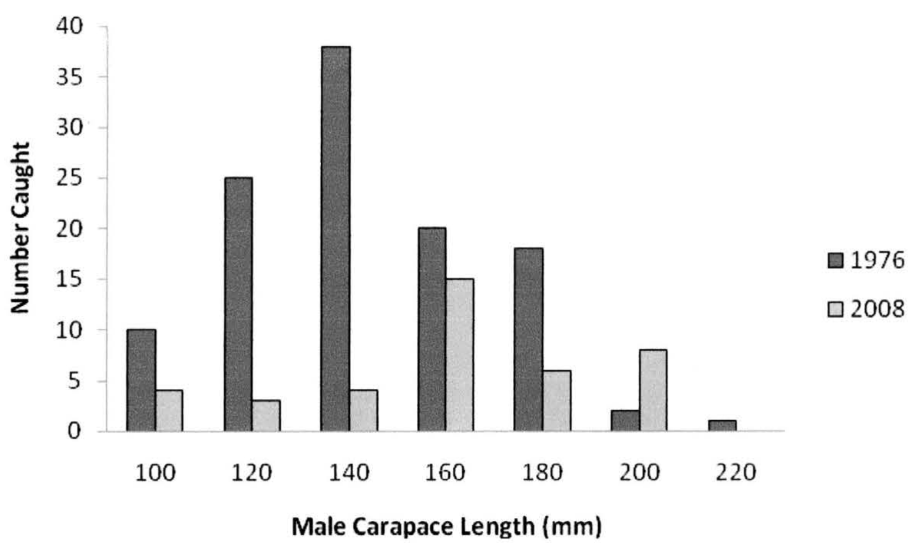


Figure 11. Carapace lengths of adult male red-eared sliders in Hidalgo County. The mean carapace length in 1976 (155.4 ± 23.4 mm) was significantly shorter than the mean carapace length in 2008 (169.5 ± 29.1 mm).

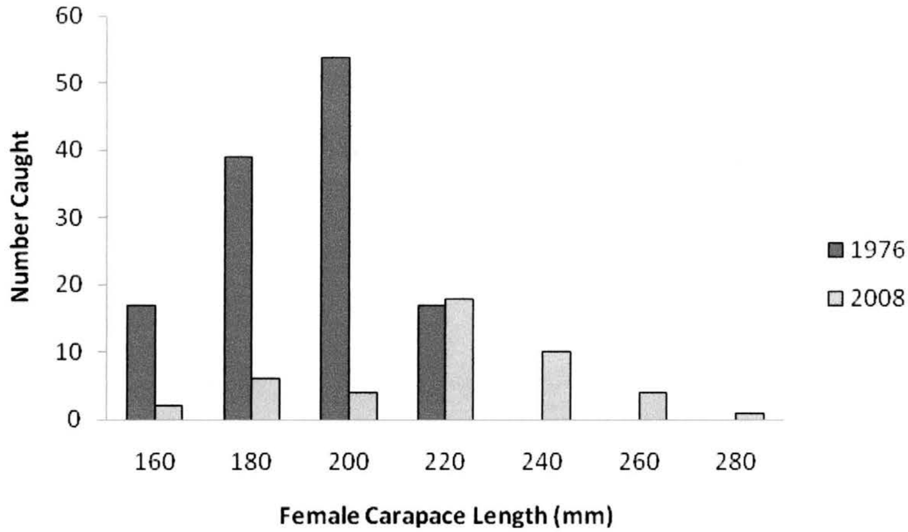


Figure 12. Carapace lengths of adult female red-eared sliders in Hidalgo County. The mean carapace length in 1976 (200.5 ± 17.3 mm) was significantly shorter than the mean carapace length in 2008 (227.8 ± 24.9 mm).

Red-eared sliders in Willacy County.— A total of 45 red-eared sliders were captured in 1976 in 160 trap days. I captured 50 red-eared sliders in 169 trap days. The capture-rate was not different (t_{12} = 0.005, P = 1.00). The adult sex-ratio was 1:1.56 (male:female) in 1976 and 1:0.95 in 2008. The sex-ratio was not different between the two years (χ^2_1 = 2.58, 0.10 < P < 0.90, N = 43).

The mean carapace length for males was 166.6 ± 25.1 mm in 1976 and 161.5 ± 25.9 mm in 2008. The mean carapace length for females was 211.3 ± 25.3 mm in 1976 and 217.2 ± 24.0 mm in 2008. The mean carapace length was not different between years for males (P = 0.55) (Fig. 13) or females (t_{43} = -0.81, P = 0.42) (Fig. 14).

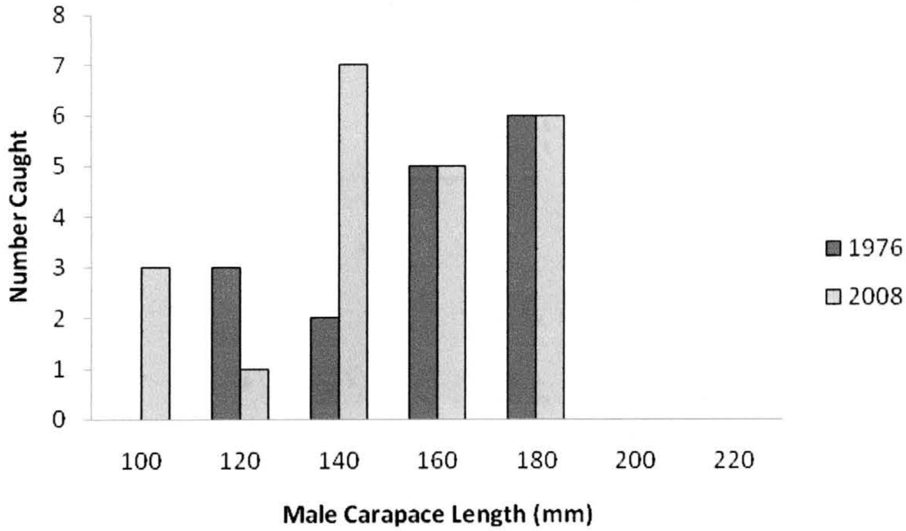


Figure 13. Carapace lengths of adult male red-eared sliders in Willacy County. The mean carapace length was not significantly different between years.

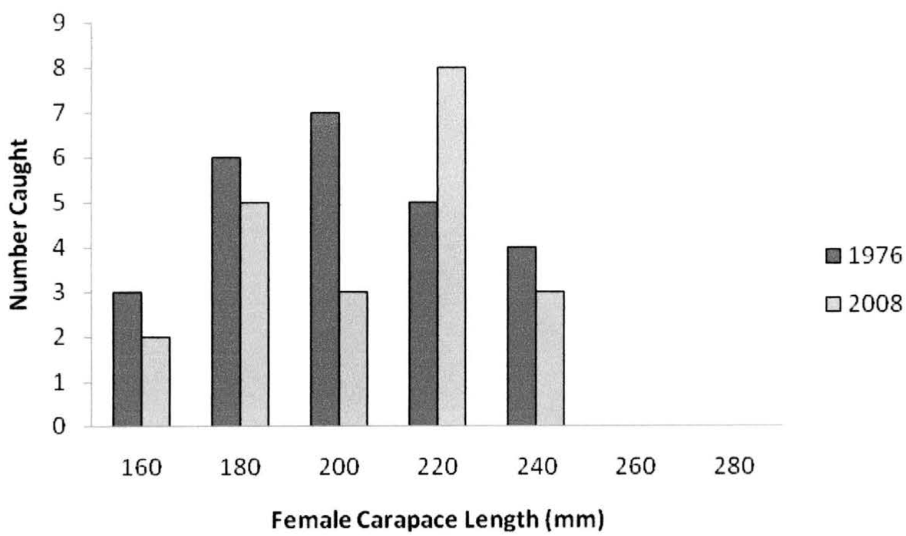


Figure 14. Carapace lengths of adult female red-eared sliders in Willacy County. The mean carapace length was not significantly different between years.

Total captured red-eared sliders.— A total of 397 red-eared sliders were captured in 1976 in 1380 trap days. I captured 256 red-eared sliders in 1400 trap days. I was unable to normalize the data ($F_{46,40} = 8.44$, $P < 0.01$). Using a randomization test, the total capture-rate was significantly different ($P < 0.01$). A total of 40 juvenile red-eared sliders were captured in 1976 and 48 juveniles were captured in 2008. The juvenile capture-rate was not different between years ($t_{85} = -0.04$, $P = 0.97$). The adult sex-ratio was 1:1.07 (male:female) in 1976 and 1:1.16 in 2008. The sex-ratio was not different between the two years ($\chi^2_1 = 0.32$, $0.10 < P < 0.90$, $N = 205$).

The mean carapace length for males was 158.6 ± 23.9 mm in 1976 and 165.3 ± 28.2 mm in 2008. The mean carapace length for females was 204.8 ± 19.3 mm in 1976 and 220.8 ± 22.6 mm in 2008. I normalized the data for females ($F_{116,190} = 1.17$, $P = 0.17$). The mean carapace length for both males ($t_{167} = 1.94$, $P = 0.05$) (Fig. 15) and females ($t_{231} = 6.29$, $P < 0.01$) (Fig. 16) was significantly different between years.

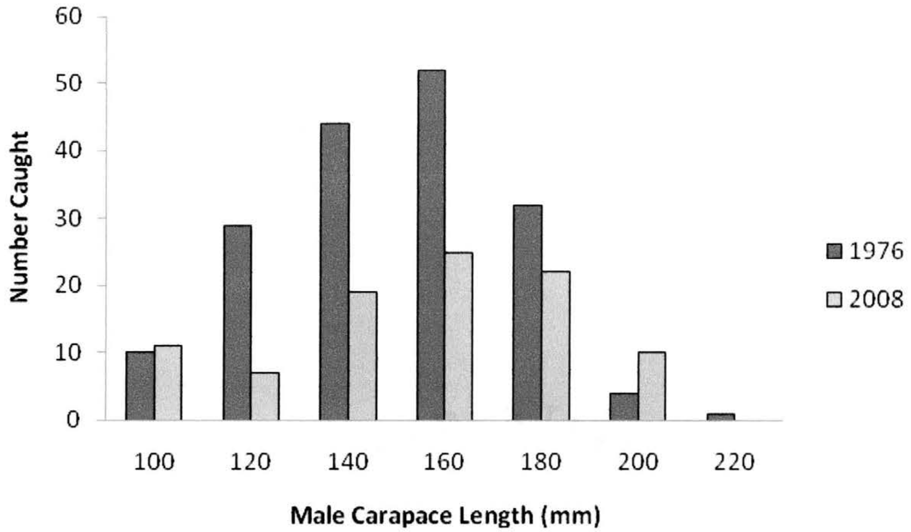


Figure 15. Carapace lengths of adult male red-eared sliders in all counties. The mean carapace length of 158.6 ± 23.9 mm in 1976 was significantly shorter than the mean carapace length of 165.3 ± 28.2 mm in 2008.

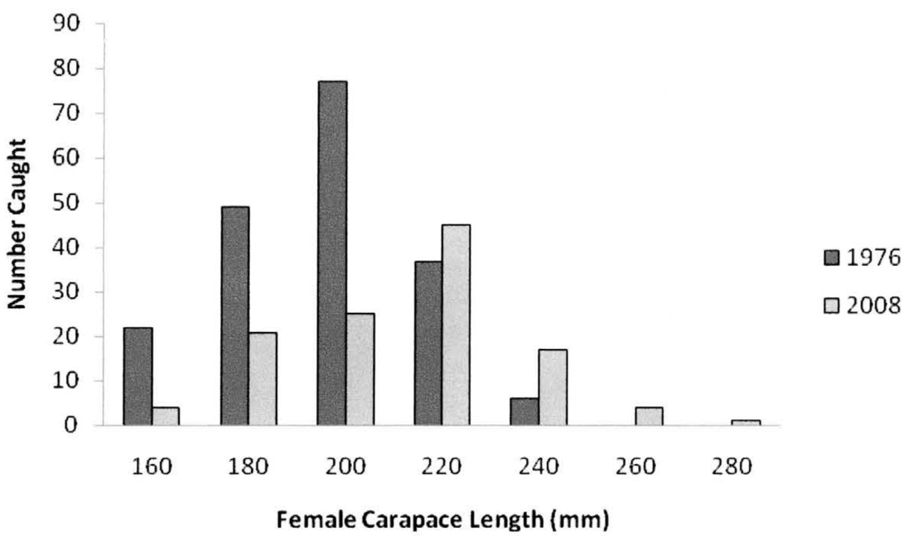


Figure 16. Carapace lengths of adult female red-eared sliders in all counties. The mean carapace length in 1976 (204.8 ± 19.3 mm) was significantly shorter than the mean carapace length in 2008 (220.8 ± 22.6 mm).

Total captured Texas spiny softshells.— A total of 52 Texas spiny softshells were captured in 1976 in 1380 trap days. I captured 54 in 1400 trap days. I was unable to normalize the data ($F_{50,40} = 3.35$, $P < 0.01$). Using a randomization test, the total capture-rate was not different ($P = 0.77$).The sex-ratio was 1:1.00 (male:female) in 1976 and 1:0.38 in 2008. The sex-ratio was significantly different between years ($\chi^2_1 = 9.11$, $P < 0.01$, $N = 44$).

The mean carapace length for males was 163.3 ± 15.3 mm in 1976 and 170.6 ± 19.1 mm in 2008. The mean carapace length for females was 295.9 ± 37.3 mm in 1976 and 364.9 ± 51.2 mm in 2008. The mean carapace length was not different for males ($t_{51} = 1.57$, $P = 0.12$) (Fig. 17). The mean carapace length for females was significantly different ($P < 0.01$) (Fig. 18).

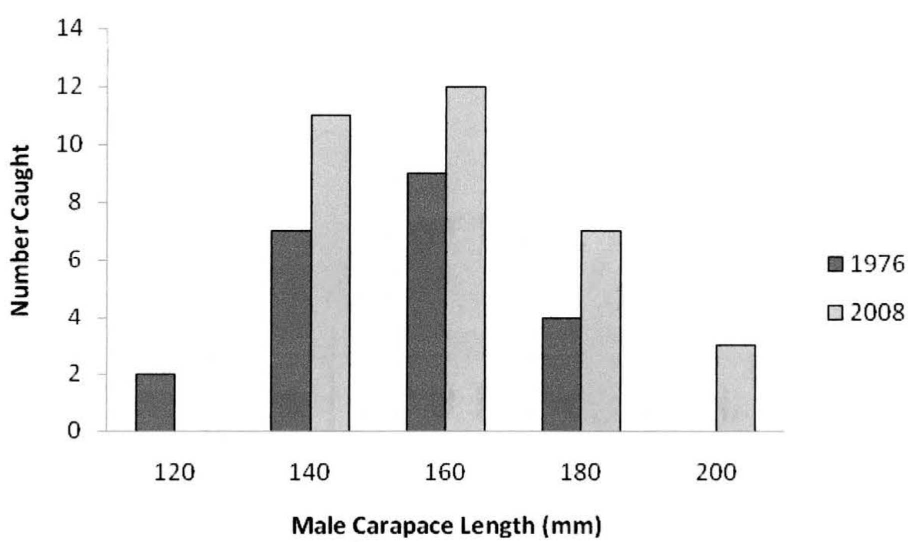


Figure 17. Carapace lengths of adult male Texas spiny softshells in all counties. The mean carapace length was not significantly different between years.

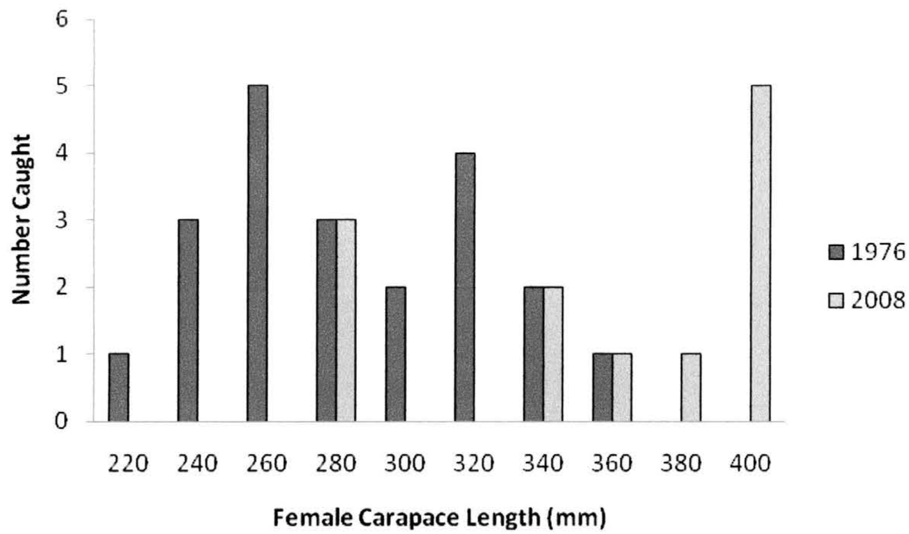


Figure 18. Carapace lengths of adult female Texas spiny softshells in all counties. The mean carapace length in 1976 (295.9 ± 37.3 mm) was significantly shorter than the mean carapace length in 2008 (364.9 ± 51.2 mm).

Season as a Potentially Confounding Factor

The red-eared slider capture-rate (per trap/day) in 1976 was 0.72 turtles between May and July, and 0.25 turtles between August and November. I was unable to normalize the data ($F_{16,29} = 5.72$, $P < 0.01$). Using a randomization test, the capture-rate was significantly different between time periods ($P = 0.03$) (Figure 19). The sex-ratio was 1:0.87 (male:female) between May and July and 1:1.49 between August and November. Using the August to November sex-ratio to generate expected frequencies, the sex-ratios were significantly different ($\chi^2_1 = 15.82$, $P < 0.01$, $N = 215$).

The mean carapace length for males was 152.5 ± 23.3 between May and July and 171.6 ± 19.9 between August and November. The mean carapace length for females was 200.2 ± 17.1 between May and July and 209.0 ± 20.6 between August and November. I normalized the data for females ($F_{82,99} = 1.39$, $P = 0.06$). The mean carapace length for both males (t_{123} = -5.54, $P < 0.01$) and females (t_{161} = -2.93, df = 161, $P < 0.01$) was significantly different.

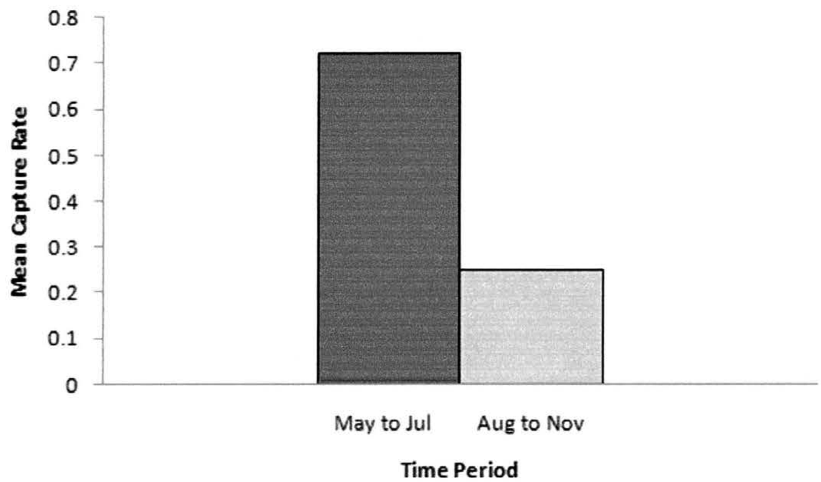


Figure 19. Red-eared slider capture-rates in 1976 from May to July and August to November. The mean capture-rate from May to July (0.72 turtles per trap day) was significantly different from the mean capture-rate from August to November (0.25 turtles per trap day) ($P = 0.03$).

Correlations between Red-eared Slider Data and Roads

No correlation was detected between male carapace length and distance to road (r_s = -0.003, $P = 0.98$, $n = 95$), road density within 1 km of trap site ($r_s = 0.079$, $P = 0.45$, $n = 95$), or road density within 5 km of trap site ($r_s = 0.139$, $P = 0.18$, $n = 95$). No correlation was detected between female carapace length and distance to road ($r_s = -0.100$, $P = 0.28$, $n = 116$). However, there was a significant positive weak correlation between female carapace length and road density within 1 km ($r_s = 0.198$, $P = 0.03$, $n = 116$) and 5 km (r_s = 0.197, $P = 0.03$, $n = 116$) of trap site. There was a significant positive strong correlation between the two road densities ($r_s = 0.729$, $P < 0.01$, $n = 11$).

Regression analyses using the three locations with at least 30 captures showed no apparent linear relationship between mean male carapace length and distance to road (r^2 = 0.31). However, there was a strong positive relationship between mean male carapace length and road density within 1 km (r^2 = 0.94) and 5 km (r^2 = 0.98) of trap site (Fig. 20). Likewise, no apparent relationship was found between mean female carapace length and distance to road (r^2 = 0.56), but there was a strong positive relationship between mean female carapace length and road density within 1 km (r^2 = 1) and 5 km (r^2 = 0.99) (Fig. 21). No correlation was found between sex-ratio and road density within 1 km (r^2 = 0.20) or 5 km (r^2 = 0.12). A potential relationship was found between sex-ratio and distance to road (r^2 = 0.86), with male-biased sex-ratios decreasing with distance to road.

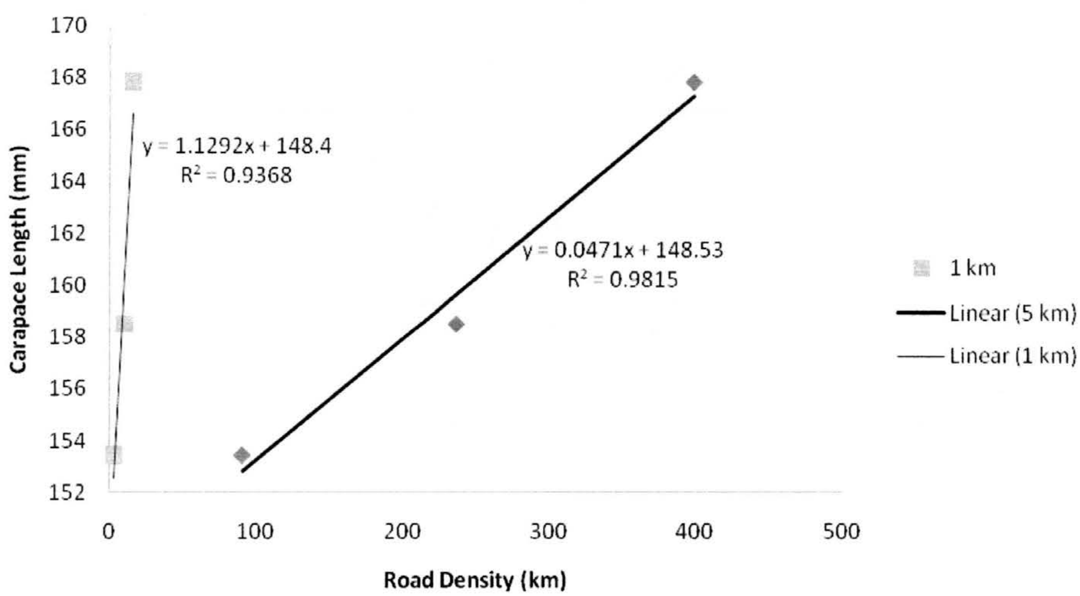


Figure 20. Regression analysis between mean male red-eared slider carapace length and road density. The three locations with at least 30 captures were used for the analysis. Both the 1 km (r^2 = 0.94) and 5 km (r^2 = 0.98) road densities displayed a strong positive linear relationship between carapace length and road density.

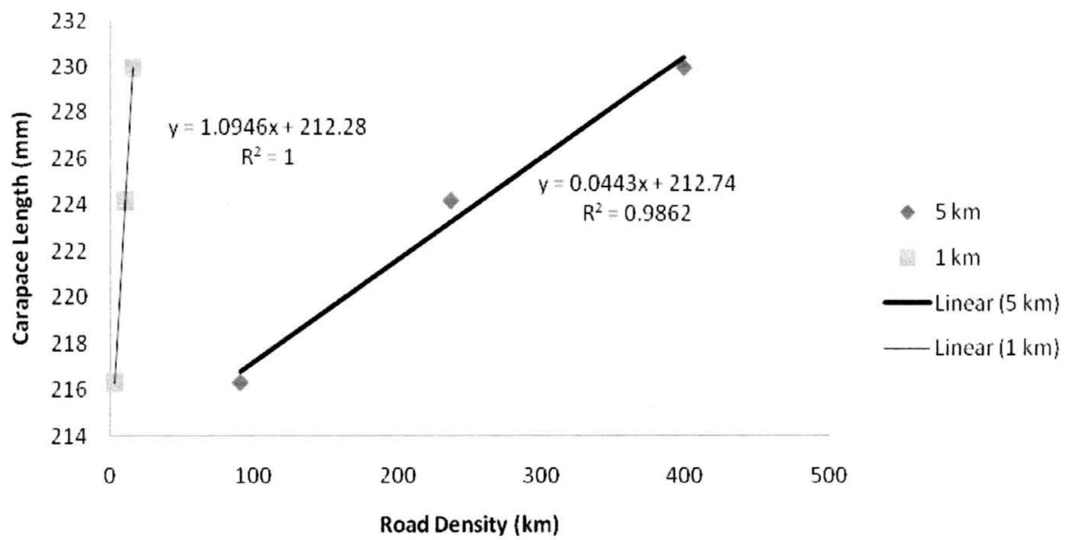


Figure 21. Regression analysis between mean female red-eared slider carapace length and road density. The three locations with at least 30 captures were used for the analysis. Both the 1 km ($r^2 = 1$) and 5 km ($r^2 = 0.99$) road densities displayed a strong positive linear relationship between carapace length and road density.

CHAPTER V

DISCUSSION

None of the three counties included in this study showed a significant difference in mean red-eared slider capture-rate. However, there was a significant difference when all counties were analyzed together. The lower mean capture-rate in 2008 could be a consequence of turtle harvest, the long-term effects of road mortality, or other seemingly less likely factors. I captured more red-eared sliders in Cameron County than were captured in 1976. This is somewhat surprising given the substantial harvest that has occurred in this county (Ceballos 2001). However, Eric Grosmaire performed 80% of his trapping effort for this county at Laguna Atascosa NWR, with low capture success (0.09 turtles per trap day). Southmost Preserve likely contains more suitable conditions for freshwater turtles (i.e., no saltwater and no American alligators (*Alligator mississippiensis*)).

I captured fewer red-eared sliders in Hidalgo County than were captured in 1976. This county accounted for the majority of freshwater turtle harvest in 1999 (Ceballos 2001). However, I chose to trap locations in this county that were not trapped in 1976.

Eric Grosmaire's 820 trap days in this county were primarily limited to two locations: Santa Ana National Wildlife Refuge (NWR) (680 trap days), and Bentsen-Rio Grande State Park (120 trap days). I replicated the 120 trap days at Bentsen-Rio Grande State Park, but limited the trap days at Santa Ana NWR to 187. I chose to trap other locations in Hidalgo County because utilizing the remaining trap days at Santa Ana NWR could have led to severely biased results. Eric Grosmaire captured 0.45 turtles per trap day at Santa Ana NWR, while I captured only 0.02 turtles per trap day. This is due to the current waterfowl management regime for the refuge leading to unsuitable turtle habitat for the locations Grosmaire (1977) used. The ponds at Santa Ana NWR are drained periodically to allow for growth and planting of vegetation, and water levels are kept low in most ponds to support waterfowl foraging activity.

The mean red-eared slider capture-rate was approximately the same for Willacy County. This is not surprising given the low amount of reported take in this county and the lack of extensive population growth. However, Grosmaire (1977) trapped at 18 sites, whereas I only trapped at 4 sites due to the majority of original sites being dry. Therefore, comparing capture-rates likely underestimated changes in turtle abundances.

Although the total number of turtles captured was much lower in 2008, the design of this project makes it difficult to infer changes in abundance. This is due to both the substantial number of replacement sites that were necessary and the lack of a random sampling design in both studies. Statistical inference issues aside, far less suitable turtle habitat was available in 2008. This is illustrated by the number of formerly suitable sites no longer available, the recorded drought history for the area (Stahle and Cleaveland 1988), and increased and re-directed water usage caused by the substantial increase in the human population (U.S. Census Bureau 1982, 2007). Because there was less available habitat, turtle populations would either have been present in higher concentrations in the remaining suitable locations (assuming populations were not operating at carrying capacity in 1976), or at similar densities in the remaining sites with the overall populations decreased by habitat loss. The first scenario should have translated into higher capture-rates at a given location. Because that generally wasn't the case, the inferred reduction in turtle abundances is probably much greater than what was detected. This conclusion is further supported by the significantly higher capture-rates in 1976 from May to July (all of the 2008 trapping took place during this time period).

The red-eared slider adult sex-ratio was not significantly different in any of the three counties included in this study. However, it was substantially more male-biased in Cameron and Willacy counties in 2008. The sex-ratio in Hidalgo County was slightly higher in 2008. The approximately equal sex-ratio in Hidalgo County was an expected result given that by 1976 Hidalgo County was already well developed and supporting a relatively large number of people. Thus, the potential for road mortality was already high. Willacy County was and remains a rural county, although some population growth has occurred. The increased male-biased sex-ratio may be due to an increase in agricultural vehicular traffic. The human population in Cameron County has increased drastically since 1976 and, although not significant, the difference in the red-eared slider sex-ratio may represent the effects of increased female road mortality. Rose and Manning (1996) found a red-eared slider sex-ratio of 1:1.75 (male:female) in two west Texas ponds, a ratio similar to that obtained in south Texas in 1976. However, I only used hoop nets to capture turtles in this study. Rose and Manning (1996) used both hoop nets and basking traps. Hoop nets tend to be a male-biased capture method (Ream and Ream 1966, Thomas et al. 1999). Still, sampling methods were the same between this study and the 1976 study. Due to the use of only one type of trap, the sex-ratios obtained in both studies may not be representative of true population sex-ratios. Further, the sex-ratio was more male-biased from May to July in 1976. Because all of the 2008 trapping occurred during this time period, detected sex-ratios were likely more male-biased than true population sex-ratios.

The mean carapace length for male red-eared sliders was significantly longer in Hidalgo County in 2008, significantly shorter in Cameron County in 2008, and not significantly different in Willacy County. The mean carapace length for female red-eared sliders was also significantly longer in Hidalgo County in 2008 and not significantly different in Cameron or Willacy counties. Combining the data from all three counties showed a significantly longer mean carapace length in 2008 for both males and females. The adult mean carapace length for both males and females was significantly longer during the August to November trapping period in 1976. Therefore, carapace length differences were probably even greater than what was detected.

It is unclear why mean carapace lengths for both male and female red-eared sliders were typically longer in 2008. Theoretically, harvest should result in smaller turtles because the younger individuals are not given sufficient time to grow into larger size-classes. However, harvest and mortality affect populations by reducing abundance and consequently reproductive potential. Therefore, larger individuals may be a product of lower reproduction due to loss of females through harvest or road mortality. Larger, and thus older, individuals remain in the population while recruitment into the adult age-class is lower.

The lack of evidence supporting the effects of harvesting on red-eared sliders may also be due to sampling locations. The majority of trap days in both studies were performed on properties where turtles were protected from harvest during the preceding decade or longer. Harvesting effects on protected populations would be the result of immigration and emigration, and thus take longer to be detectable. Exact localities from which turtles were harvested in the counties I sampled were unknown and, thus, not trapped.

The number of Texas spiny softshells captured was nearly identical between years and the capture-rate was not significantly different. The adult sex-ratio was significantly male-biased in the 2008 captures (1:0.38, male:female). Given their highly aquatic nature (Ernst et al. 1994), softshells may be more vulnerable to altered sex-ratios caused by differential road mortality than sliders. Although they typically nest near the water, female spiny softshells have been found to travel up to 100 m inland to nest (Vogt 1981, Ernst et al. 1994). Eric Grosmaire found a sex-ratio of 1:1 in 1976, which agrees with other spiny softshell studies (Cagle 1942, Breckenridge 1955, Vogt and Bull 1982). The disparity between the two results may be cause for concern. The mean carapace length for Texas spiny softshells was longer in males (not significant) and females (significant) in 2008. This may be indicative of lower recruitment, and thus a lack of younger, smaller adults.

Although roads may be a factor responsible for the increase in mean red-eared slider carapace lengths, I was unable to test the road data against Texas spiny softshell carapace lengths due to the lack of captures at most trapping locations. However, the significant positive weak correlation found between female red-eared slider carapace lengths and major road density within 1km and 5 km, coupled with the linear relationships found using mean carapace lengths, show that roads may indeed be influencing freshwater turtle population demographics in the lower Rio Grande Valley. This study was not designed to test the effects of roads on turtle populations. Further research on this subject should include an experimental design suitable for obtaining such information.

Management Implications

Based on the results of this study, I cannot conclude from these data that harvest is responsible for changes in freshwater turtle population demographics in the lower Rio Grande Valley. However, given the similar results between species, it seems likely that the same factors are influencing red-eared slider and Texas spiny softshell population demographics. Despite the possibility of reduced reproduction, I consider the presence of larger individuals a positive result. Differential size-based harvest not only affects current size and age-classes by removing larger individuals, over time it can act as a selective pressure towards smaller size-classes by altering the allele frequencies of the reproductive class so that individuals genetically pre-disposed to being smaller or having slower growth rates will dominate reproductive cycles (Ratner and Lande 2001). Turtles are often sold by size-class (Close and Seigel 1997), thus incentives exist to harvest larger turtles.

The changes detected are likely the result of several influential factors, including harvest, differential mortality, changes in habitat availability, and natural fluctuation. Conserving adult freshwater turtles is crucial to long-term population viability due to low fecundity, low hatching success, and delayed maturity (Congdon et al. 1993, 1994, Heppell 1998). Due to their life-history characteristics, turtle populations have a very low tolerance for additive mortality. There is evidence that additive mortality as low as 1% to 5% to adult age classes may be the most turtles can tolerate before incurring negative population growth rates (Doroff and Keith 1990, Congdon et al. 1993, 1994). Furthermore, other factors like road mortality and changes in water levels can be as important a regulatory factor for turtle populations as harvest.

Given that south Texas is historically drought-prone (Stahle and Cleaveland 1988), and that approximately 94% of land in Texas is privately owned (Texas Center for Policy Studies 2000), it is unlikely that public water protection will be adequate for long-term turtle conservation. Sliders frequently migrate to new locations, with migration distances that can exceed 5 km (Ernst et al. 1994). These migrations are often a response to unsuitable habitat conditions driven by changes in water levels (which can occur on both public and private property) (Cagle 1950). Turtles are currently being managed under the unrealistic assumption that little movement occurs between public and private waters.

A potential alternative management regime for freshwater turtles is spatial harvest control. Spatial harvest management theory is based on the concept that harvest can be successfully regulated through the designation of protected and unprotected areas (McCullough 1996). The number, size, and placement of protected areas change in response to harvest trends, theoretically resulting in protected areas that serve as robust source populations. It is attractive because the only required population data are estimated numbers of individuals harvested per location per unit time, a current requirement for turtle harvesters.

However, because successful spatial harvest management depends on dispersal from protected areas, it assumes that protected areas continually house robust populations. There is no evidence supporting the ability of protected areas in Texas to maintain robust long-term source populations for turtles. Furthermore, harvest response may be slow due to the life-history characteristics of sliders, which entail delayed maturity, low fecundity, and long life-spans (Gibbons and Lovich 1990, Ernst et al. 1994). Although age at maturity varies, females have been found to be sexually mature at five to eight years of age, and males at two to five years of age (Gibbons et al. 1981, Gibbons and Greene 1990). If dispersal rates are high, there may be a substantial time-lag before overexploitation is detected, possibly resulting in depletion of turtles populations. Therefore, although it would likely be more effective than current management, slow response times and the inability to ensure that protected habitats remain suitable make spatial control a risky harvest management tool for turtles. In addition, this management regime would require that private waters be included in the harvest control.

Understanding the effects of both harvest and road mortality are critical for freshwater turtle conservation in Texas. Slow population response times make it difficult to test the long-term effects of harvest on populations within realistic research time limits. Therefore, to understand the effects of harvest known harvested locations should be compared to protected locations containing similar turtle habitat. The level of take and length of time the location has been harvested should also be known. This study has revealed that roads may be negatively affecting turtle populations in the lower Rio Grande Valley. Further research in this area should include a sampling design that allows a representative number of captures at locations surrounded by varying road densities.

Given the lack of conclusive evidence concerning the influence of both harvest and road mortality on freshwater turtle populations in south Texas, I recommend that a more conservative approach be taken for harvest management. In addition to spatial control, harvest regulations should be modified to prevent turtle harvest during breeding and nesting seasons. Furthermore, bag limits should be enforced for female turtles due to their substantially greater influence on population viability.

APPENDIX A: TRAP-SITE LOCATION PHOTOGRAPHS

Photographs are presented as north, east, south and west, clockwise from top-left. General site descriptions are given above each photograph set. Sites and order of listing correspond to Table 1.

Site 1: Public pond. The site was unsuitable for trapping in 2008 due to a lack of security and proper border vegetation. The surrounding landscape consisted of agricultural fields.

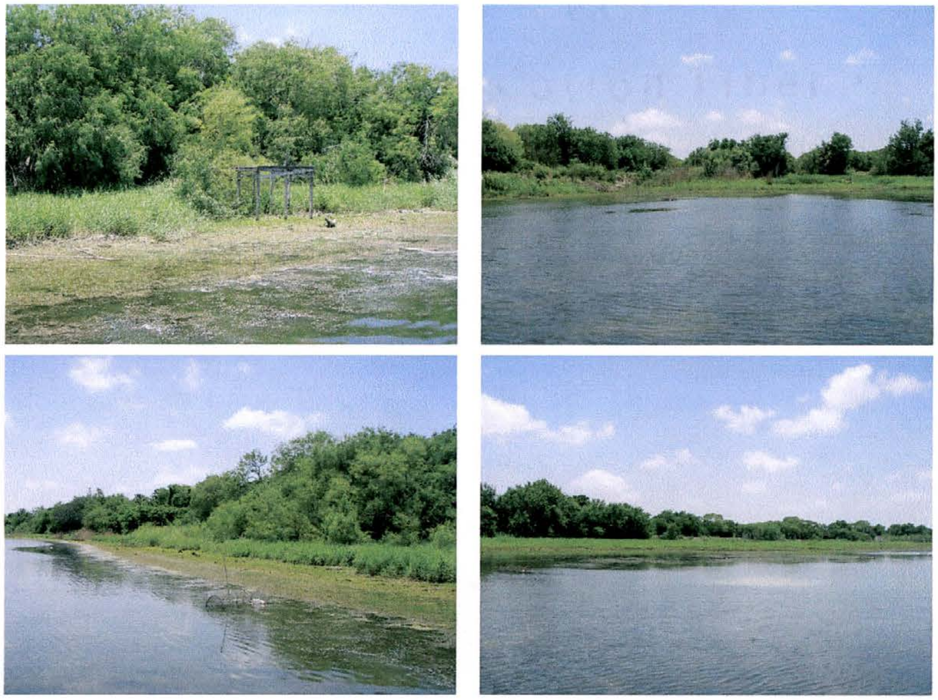
Site 2: Headquarters Pond at Laguna Atascosa National Wildlife Refuge. The pond, a major trapping site in 1976, was dry in 2008.

Site A: Abbott Reservoir. The site, surrounded by an agricultural matrix, provided suitable turtle habitat.

Site 3: Arroyo Colorado. The site was unsuitable for trapping in 2008 due to a high risk of trap theft. This portion of the river was imbedded in an industrial and residential landscape with patches of presumably unmanaged land.

Site B: The permanent resaca in the Nature Conservancy of Texas' Southmost Preserve. The site provided suitable turtle habitat for all life stages.

Site C: The ephemeral resaca in the Nature of Conservancy of Texas' Southmost Preserve. The site, adjacent to the permanent resaca, provided temporary turtle habitat.



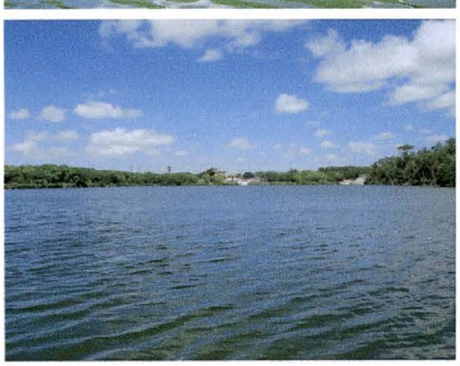
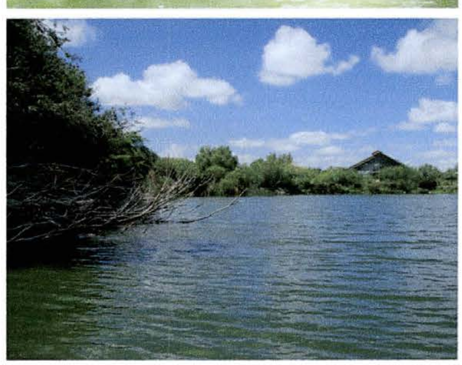
Site D: One of two Rio Grande sites trapped on the Nature Conservancy of Texas' Southmost Preserve.

Site E: One of two Rio Grande sites trapped on the Nature Conservancy of Texas' Southmost Preserve. This site was located upstream from Site D.




Site F: Edinburg Scenic Wetlands. This wetland supported large turtle populations.



Site 4: Public canal. The site contained water, but was unsuitable for trapping in 2008 due to a lack of border vegetation. The surrounding landscape consisted of a matrix of agricultural fields and low density housing.

Site 5, G: West resaca at Bentsen-Rio Grande State Park. This resaca has held water since it was trapped in 1976.

Site H: East resaca at Bentsen-Rio Grande State Park. This resaca is periodically filled and serves as temporary turtle habitat.

Site I: Canal at Bentsen-Rio Grande State Park. The canal runs along the northern border of the park.

Site J: Frontera Audubon. This patch of suitable habitat is surrounded by urban development.




Site 6: Willow Lake at Santa Ana National Wildlife Refuge. This was one of Eric Grosmaire's primary trapping sites and contained only shallow standing water in 2008.




Site K: Pintail Lake at Santa Ana National Wildlife Refuge. This site was used as a replacement for Willow Lake.

Site 7, L: Cattail Lake at Santa Ana National Wildlife Refuge. This site contained new water and was an unsuccessful trapping location. The site was nearly dry at the time of the photo, approximately a month after trapping.

Site M: Rio Grande at Santa Ana National Wildlife Refuge. This site contained suitable turtle habitat.

Site 8: Public canal. The site was located in an accessible undeveloped area, but contained shallow water.

Site 9: Private pond. The pond was dry in 2008 and surrounded by agricultural fields.




Site 10: Private pond. The site contained water of an unknown depth and was surrounded by agricultural fields. The pond was formerly much larger.




Site 11: Private pond. I was unable to obtain access to the pond and am uncertain of its water status. It was assumed to be dry given the surrounding agricultural matrix.

Site 12, N: Private pond owned by Frank Quintero. This pond, located in a cattle pasture, served as a point habitat for turtles within a surrounding agricultural matrix.

Site 13: Private pond. The site was dry in 2008 and surrounding by agricultural fields.

Site 14: Public canal. The site contained water, but appeared shallow. The surrounding landscape included agricultural fields and pastures.

Site 15: Private pond. The site was dry in 2008 and no former pond could be located. The surrounding landscape consisted of an agricultural matrix.

Site 16, O: Public pond. This was a relatively large runoff pond with an island in the center. The surrounding landscape consisted of agricultural fields.

Site P: Public canal. The site contained relatively deep water and enough suitable border vegetation to serve as a replacement for the currently unsuitable canals.

Site 17: Private resaca. The site was dry when visited, but looked potentially ephemeral. The surrounding landscape included agricultural fields and pastures.

Site 18: Public canal. The site appeared suitable for trapping, but contained very shallow water. Agricultural fields dominated the surrounding landscape.

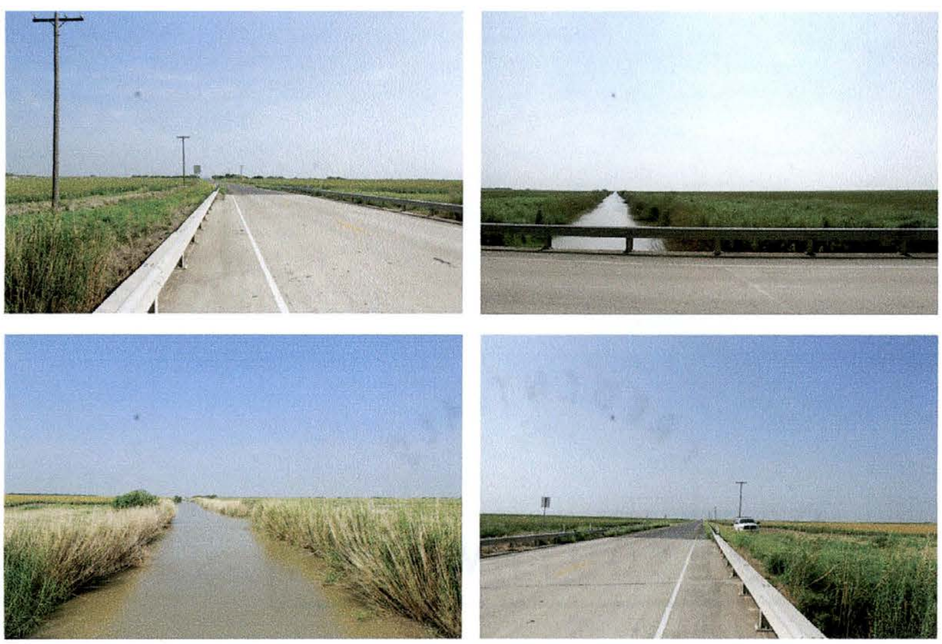
Site 19: Private resaca. The site was dry in 2008, but was clearly a former resaca. Agricultural fields surrounded the site.

Site 20: Public canal. The site was dry in 2008 and surrounded by agricultural fields.

Site 21: Public canal. The site contained water, but was not trappable due to a lack of suitable border vegetation. This was the same canal as Site L. The surrounding landscape consisted of an agricultural matrix.

Site 22: Private pond. The site was dry in 2008 but was formerly a relatively large pond. Agricultural fields dominated the surrounding landscape.

Site Q: Public canal. The site contained relatively deep water and sufficient border vegetation for trapping. The site was embedded in an agricultural matrix.



Site 23: Private pond. The site was dry in 2008 but appeared to serve as a small ephemeral pond. The surrounding landscape contained agricultural fields and pastures.

Site 24: Private pond. The site contained water but appeared to be very shallow. The surrounding landscape consisted of pastureland and housing subdivisions.

Site 25: Arroyo Colorado. The site contained water and appeared potentially suitable for trapping, but was avoided due to a high risk of trap theft. The surrounding landscape consisted of residential housing and pasturelands.

APPENDIX B: ERRORS REPORTED IN GROSMAIRE (1977) AND CORRECT ORIGINAL TRAPPING LOCALITIES

Incorrect Locality	Correct Locality
Arroyo Colorado at FM 506	Arroyo Colorado at FM 106
Intersection of Hwy 77 and FM 490, SE corner	Intersection of Hwy 77 and FM 498, SE corner
On FM 490, 1 61 km east of Hwy 77	On FM 498, 1.61 km east of Hwy 77
On FM 498, 1.77 km west of FM 1520	On FM 498, 1.77 km west of FM 1420
Reservoir north of FM 408, between Rio Hondo and FM 507	Reservoir north of FM 508, between Rio Hondo and FM 507

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