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The Effects of Commercial Harvest on the Density and Demography of Aquatic Turtles in Arkansas

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ABSTRACT

The United States is home to the second highest concentration of turtle species in the world, after Asia. As of 2018, there are 57 turtle species recognized within the US, 40% of which are listed as threatened or endangered, with the primary threats to population persistence identified as over-consumption and/or habitat loss. Within the US, the Mississippi Alluvial Valley (MAV) region represents the second highest turtle species richness, after the Mobile River Basin. The MAV region of Arkansas is one of the least regulated in terms of commercial aquatic turtle harvest and has undergone large-scale habitat conversion from bottomland hardwood forest wetlands to agriculture, yet little is known about freshwater turtle populations within this region. As awareness of the plight of turtles worldwide increases and studies continue to find current levels of commercial harvest unsustainable, biologists and conservation organizations have begun petitioning states to close or strictly regulate commercial turtle harvest. Baseline data on turtle populations in the MAV of Arkansas is lacking. We conducted a three-year capture-mark-recapture (CMR) study of turtle community composition, density, and demography in agricultural ditches and aquaculture ponds of eastern Arkansas, two abundant wetland habitats which are often targeted by commercial turtle harvesters. We captured and marked over 4,000 individual turtles of nine species including red-eared sliders (*Trachemys scripta;* N = 2695), spiny softshell turtles (*Apalone spinifera;* N = 640), common musk turtles (*Sternotherus odoratus;* N = 508), eastern mud turtles (*Kinosternon subrubrum;* N = 81), common snapping turtles *(Chelydra serpentina;* N = 56), river cooters (*Pseudemys concinna;* N = 11), southern painted turtles (*Chrysemys dorsalis;* N = 7), Mississippi map turtles (*Graptemys pseudogeographica kohnii*; N = 7*)*, and alligator snapping turtles (*Macrochelys temminckii;* N = 2). We found that harvest severely reduces density of red-eared sliders and spiny softshell turtles in both pond and ditch systems for up to two years post-harvest and potentially causes shifts in community composition detectable for years after the initial removal event. We found no differences in turtle species richness or diversity between harvest status in ponds, but harvested ditches had higher mean Simpson's diversity and species richness. There were relatively few consistent differences in density or demography within ditches, likely because the dynamic hydrology of ditches results in frequent immigration and emigration. Recently harvested aquaculture ponds had lower densities of red-eared sliders and spiny softshell turtles than unharvested ponds and were missing size cohorts, persisting for at least 5 years after harvest. Using supervised classification in a GIS, we delineated 22,317 ha of aquaculture ponds and more than 18,350 linear km of agricultural ditches occurring in the MAV. Based on our density calculations, we estimate that more than 2 million red-eared sliders and 427,000 spiny softshell turtles occur in ditches and aquaculture ponds of the region. Overall density of sliders was greater in ditches, with approximately 65% of our extrapolated abundance existing in this habitat type, while spiny softshell turtles are far more common in ponds, with only about 17% of our extrapolated abundance occurring in ditches across the MAV. Our density estimates were moderate compared to other reports in the literature. These turtles are clearly utilizing these habitats, sometimes occurring at high densities, yet they are not limitless. Harvest can reduce their populations and managers must take this into account.

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encouragement, and support in any way I needed it. Life is so much better with y’all by my side.

Cheers to many more years tackling whatever life throws our way as a team!

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CHAPTER I: INTRODUCTION

When the Asian turtle market collapsed in the 1990s due to over-exploitation of native species, the United States became a major exporter of wild caught turtles (Mali et al. 2014, Reed and Gibbons 2003). In just one decade (2002-2012), a total of 126,600,529 individual turtles of 14 genera were exported from the US (Mali et al. 2014). In light of the current known declines in turtle populations worldwide, conservationists and state agencies have begun to assess the impact this large-scale removal and exportation may be having on local turtle populations. As many studies find current harvest levels unsustainable, states have begun to close (e.g., Texas, Missouri, Alabama) or strictly regulate (e.g., Connecticut, New Jersey, Iowa) the practice (Brown et al. 2012, Zimmer-Shaffer et al. 2014). However, there is concern that harvest pressure will shift to states that have yet to implement strict regulations (e.g., Arkansas), especially in the southeastern region where turtle diversity is believed to be highest (Buhlmann et al. 2009). Landscapes of this region have shifted since the 1940s from vast natural wetlands to agriculture (Oswalt 2013). Eastern Arkansas alone accounts for half the rice production in the United States and ranks second in aquaculture production (Engle et al. 2020, Laws 2020). As a result of these land use practices, eastern Arkansas is a landscape dominated by anthropogenic land use that has converted natural wetlands into highly human-altered wetlands. These anthropogenic ecosystems appear to provide extensive habitat for freshwater turtles yet limited previous research has examined turtle density or species composition in these altered aquatic habitats and impacts of commercial harvest on overall turtle populations within these systems is unknown.

Additionally, in 2018 the Center for Biological Diversity, the IUCN SSC Tortoise and Freshwater Turtle Specialist Group, and dozens of leading turtle biologists urged Arkansas to ban or strictly regulate commercial harvest of its freshwater turtle species, yet baseline data required to make informed commercial turtle harvest management and regulation decisions is lacking and little is known about the effects the current level of harvest has on local turtle populations. In response to these research needs, this study was conducted to provide basic turtle assemblage data and population estimates appropriate for harvest sustainability modelling and management decisions.

In the first portion of this thesis, we focus on overall abundances and differences in turtle assemblages between our two anthropogenic wetland types – aquaculture ponds and agricultural ditches of eastern Arkansas. We used a Capture-Mark-Recapture approach over three summers (2019-2021) to estimate abundances in 41 aquaculture ponds and 21 agricultural ditches. We used supervised classification within GIS to delineate these wetland types across the region. We then used a Monte Carlo approach to extrapolate our estimates across the entire region.

In the second portion of this thesis, we examine differences in estimated densities and turtle assemblages within aquaculture ponds and agricultural ditches that have and have not experienced commercial turtle harvest.

Our objectives were to, 1) evaluate turtle assemblages inhabiting agricultural ditches and aquaculture ponds across the MAV of Arkansas, 2) quantify the total amount of available agricultural ditch and aquaculture pond habitat in the region in order to extrapolate how many turtles of each species may occupy these habitat types across the MAV in Arkansas, and 3) evaluate differences in the turtle community composition between harvested and unharvested sites.

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CHAPTER II: AQUATIC FRESHWATER TURTLE ASSEMBLAGES AND DENSITIES IN TWO COMMON ANTHROPOGENIC AQUATIC HABITATS OF EASTERN ARKANSAS

ANDRHEA D. MASSEY, BRETT DEGREGORIO, AND J.D. WILLSON

Abstract

The Mississippi Alluvial Valley (MAV) of Arkansas is a drastically human-altered landscape dominated by anthropogenic land use, namely agriculture and aquaculture. As a result of these land use practices, many of the available wetlands within this region are aquaculture ponds or irrigation ditches for agriculture. Using supervised classification in a GIS, we determined that in this region there is 22,317 ha of aquaculture ponds and more than 18,350 linear km of agricultural ditches. These anthropogenic ecosystems provide extensive habitat for freshwater turtles yet limited previous research has examined turtle density or species composition in these altered aquatic habitats. We used a Capture-Mark-Recapture approach over three summers (2019-2021) to evaluate population demography and community composition of freshwater turtles in 32 agricultural ditch and 51 aquaculture pond sites across the MAV. We captured over 4,000 individual turtles of nine species of turtle red-eared sliders (*Trachemys scripta;* N = 2695), spiny softshell turtles (*Apalone spinifera;* N = 640), common musk turtles (*Sternotherus odoratus;* N = 508), eastern mud turtles (*Kinosternon subrubrum;* N = 81), common snapping turtles *(Chelydra serpentina;* N = 56), river cooters (*Pseudemys concinna;* N = 11), southern painted turtles (*Chrysemys dorsalis;* N = 7), Mississippi map turtles (*Graptemys pseudogeographica kohnii*; N = 7), and alligator snapping turtles (*Macrochelys temminckii;* N = 2). Diversity and richness did not differ between the two wetland types. One species, the red-eared slider, dominated the turtle community in both wetland types, comprising 66% (±22% SD) of all captures in agricultural ditches and 63% (± 32% SD) in aquaculture ponds. We estimated densities of only the three most commonly captured species (*T. scripta*, *A. spinifera*, and *S. odoratus*) due to lack of appropriate recapture rates in all other species. Population density of red-eared sliders ranged from 0 turtles/ha or linear km to 500.08 turtles/km with a median of

37.05 turtles/unit area. The spiny softshell turtle was more frequently captured in ponds than ditches and attained average densities of 25.08 (± 18.68) and 6.76 (± 3.48) respectively. By extrapolating our density estimates to each habitat type we estimated that upwards of two million red-eared sliders occur in aquaculture ponds and agricultural ditches across Eastern Arkansas. Our results suggest that although much of the landscape is human dominated, these habitats provide abundant habitat for a few generalist turtle species.

Introduction

Turtles are one of the most threatened major vertebrate groups, with their global decline attributed largely to anthropogenic causes, namely overharvesting (for the pet trade and consumption) and habitat loss or alteration (Elston et al. 2016, Stanford et al 2020). Of the ~360 living species, about 60% are threatened, endangered, or recently extinct (Lovich et al 2018, Stanford et al 2020). Freshwater turtle diversity in the United States is believed to be highest in the southeastern region, where landscapes have shifted since the 1940s from vast natural wetlands to agriculture (Buhlmann et al. 2009, Oswalt 2013). Several studies have researched turtle populations in human dominated landscapes (Budischak et al. 2006, Gibbons et al. 2000, Lovich et al. 2018) but few regions have experienced as much habitat conversion as the Mississippi Alluvial Valley (MAV) of Arkansas.

Historically, the MAV was the largest tract of bottomland hardwood wetlands in North America and today covers approximately 26.7 million acres (King et al. 2010, Oswalt 2013). The ecoregion is a mostly flat alluvial floodplain running south along the Mississippi River from Illinois and Missouri through Arkansas, Tennessee, Mississippi and Louisiana. Historically, the MAV was comprised mainly of bottomland hardwood forest, but the largescale deforestation that occurred in the 1940s-1970s led to the land being converted to agriculture such as rice and soybeans. In fact, in 2020 alone, Arkansas accounted for 47.5% of the United States' total rice production, with the state harvesting approximately 583,152 ha of rice crops from the MAV (www.uaex.edu/farm-ranch/crops-commercial-horticulture/rice/, accessed 16 September 2021). Because rice is a semi-aquatic plant, flood-irrigation must be implemented, which has led to the development of agricultural ditches used to direct water both to and from flooded fields. Many of these ditches are partially or fully drained on a seasonal schedule, along with the rice fields which are drained for harvest.

In addition to ranking third in irrigated acres in 2018, with 4.25 million acres irrigated, Arkansas ranks second among states in aquaculture production and is home to 181 fish hatcheries and over 100,000 farm ponds. In 2018, Arkansas produced approximately 68% of the total baitfish and 35% of the total sportfish sold in the United States, with the majority of this aquaculture production occurring in the MAV region (Arkansas Aquaculture 2020, USDA 2019). Within the MAV region of Arkansas, there are 138 fish hatcheries with ponds varying greatly in size (~5 ha to 50 ha; Clements et al. 2021, Engle et al. 2020).

Historically, the MAV was home to a diverse suite of freshwater turtle species that inhabited the river systems, bayous, and oxbow lakes (Nickerson et al. 2019). Yet, community composition and density of turtles in the anthropogenic habitats of the MAV is unclear. Numerous studies have shown that some turtles can colonize and attain high densities in anthropogenic habitats such as golf course ponds, farm ponds, and reservoirs (Congdon et al. 1986, Major 1975, DeGregorio et al. 2012, Galbraith et al. 1988, Rose and Manning 1996). However, few studies have evaluated turtle populations in agricultural ditches (Elston et al. 2016, Homyack et al. 2016) or aquaculture ponds (Failey et al. 2007, Mahmoud 1969). Understanding turtle population densities and community composition in the anthropogenic habitats of Arkansas is especially important because these are currently the predominant wetlands in the MAV and are legally targeted by commercial turtle harvesters (Irwin 2007). In order to understand the effects of harvest on turtle populations, conservationists and managers need to understand the possible range of densities and demographics of turtles inhabiting these wetlands.

Our objectives were to, 1) evaluate turtle assemblages inhabiting agricultural ditches and aquaculture ponds across the MAV of Arkansas, 2) compare the density of the species in these wetlands to reported densities from other wetland types, and 3) quantify the total amount of available agricultural ditch and aquaculture pond habitat in the region and extrapolate how many turtles of each species may occupy these habitat types across the MAV in Arkansas.

Materials and Methods

Study sites - We conducted our study widely across 27 counties in Eastern Arkansas (Figure 1). We chose our study region to overlap with the MAV region of Arkansas (21 counties) but included six adjacent counties where commercial aquatic turtle harvest is allowed. We conducted short (5 d), intense trapping sessions at 62 sites across the MAV consisting of two wetland types – agricultural ditches and aquaculture ponds. This approach allowed us to rigorously generate point estimates of density and species composition as well as understand variation in density across the region using Closed Capture-Mark-Recapture (CMR) models. We sampled freshwater turtle populations in 21 ditches varying in size from 0.06 – 3.14 km and 41 aquaculture ponds, varying in size from less than 0.5 ha to more than 10 ha (Fig 1). Ten aquaculture ponds and seven ditch sites were re-sampled in different years in order to produce site-year estimates, as has been done in other wildlife studies, including those with freshwater turtles (Bailey et al. 2004, Dreslik et al 2005). Given high inter-annual variation in habitat and environmental condition, and turtle densities, we treated the site-year as the replicate in our analyses, resulting in a total sample size of 51 ponds and 32 ditch segments. Ditch sites were selected without prior knowledge of turtle densities according to the following set of criteria: 1) were accessible with a 4wd vehicle, 2) must have had enough water at time of sampling for traps to be properly partially submerged, 3) were a minimum of 450 m long or connected to a series of ditch sections, creating a combined minimum of 450 m of ditch, and 4) were not a channelized stream or stream section. Aquaculture farms were selected if they were accessible (with landowner permission) and had either no harvest (n = 21) or a prior harvest history (n = 20). We trapped at five different aquaculture farms and treated individual aquaculture ponds within the farms as separate replicates. Beyond these criteria, we selected sites that were representative of available habitat across our study region with the intention of selecting a broad range of habitats within the constraints of accessibility.

Turtle Sampling – Sampling occurred from late April to early August for three consecutive summers (2019 – 2021). Turtles were captured using baited hoop net traps (Memphis Net and Twine, Memphis, Tenn.) set approximately 15 - 30 m apart and left in place for five days (i.e., set Monday, pulled Friday). Number of traps set per site varied from 5 to 35 traps depending on water levels, accessibility, and pond size or ditch length. We used custom hoop nets, replicating the trap most often used by local commercial turtle harvesters such that traps had two ‘Arkansas style’ funnel throats and mesh size of 3.175 cm and 50.8 cm diameter. The front of the trap was oriented to face downstream in ditches, held open with a wooden stake while the back of the trap was held up with PVC over rebar, so that a minimum of approximately 40% of the trap remained above water, allowing captured turtles to breathe. To avoid traps becoming submerged in the case of unexpected rapid water level rise, a common occurrence in agricultural ditches, each trap was also fitted with a flotation device in the back section of the trap. In the case of rising water levels, the PVC allowed the trap to slide up, lifted by the float, avoiding the drowning of captured turtles. We baited traps with raw chicken, or on rare occasions, fish. To avoid spreading disease in fish farms we used exclusively chicken as bait and decontaminated traps with a 1% Virkon® Aquatic solution between each trapping session (Stockton-Fiti 2017).

We checked traps daily to remove all captured turtles and to replace bait. We recorded the mass (g), straight-line carapace length (SCL [mm]), and straight-line plastron length (SPL [mm]) of each captured turtle. We recorded the sex of each turtle by examining secondary sexual characteristics specific to each species (Ernst and Lovich 1994). We individually marked all captured turtles. We marked hard-shelled turtles using a Dremel or hand file to notch the marginal scutes with a unique code, a modified version of the Nagle et al. (2017) marking schematic. Soft-shelled turtles were marked with a 2.5 cm metal clip tag displaying a unique number attached to the posterior edge of the carapace (Ostovar et al. 2021; National Band and Tag Company, Newport, KY, USA). We released all turtles near the trap they had been captured in immediately after processing.

Diversity and Community Composition – In order to describe the turtle assemblage within each habitat type we calculated several indices of diversity. First, we calculated the raw species richness (number of species captured) for each site. We also calculated the percent composition of each species at each site. Finally, we calculated the Simpson’s Diversity Index for each site. Simpson’s Diversity Index is a measure of diversity which takes into account both the number of species present (richness) and the relative abundance of each species (evenness) through the following equation:

$$D = 1 - (\Sigma n(n-1)/N(N-1)$$

where n is the number of individuals of each species and N is the total number of individuals of all species (Simpson 1949). Values range from 0 (no diversity) to 1 (infinite diversity). We compared Simpson's Diversity and species richness between habitat types using two sample *t*-tests.

Density Estimation – We used closed capture-mark-recapture (CMR) analyses in Program Mark (Cooch and White 2012) to estimate the density of the three most frequently captured turtle species (*Apalone spinifera, Trachemys scripta,* and *Sternotherus odoratus*) at each site. Data (i.e., weekly encounter histories) assumed populations were closed for each of the 5-day trapping sessions. We ran three models (M_0, M_t, M_b) for the 5-day encounter histories for each site-year using the Closed Populations – Full Likelihood p and c data type. This data type assumes 1) the population is closed to births, deaths, immigration, and emigration during the (5-day) sampling period, 2) no tags are lost, 3) there is no human error in recording data, and 4) probability of capture during each sampling occasion is constant and equal. Model M_0, the "null" model, assumes capture probability is constant with respect to all factors; model M_t assumes capture probabilities vary over time or trapping occasion (day), and model M_b assumes capture probabilities vary due to behavioral responses (i.e., trap happy or trap shy response), with initial capture probability (p) allowed to vary from recapture probability (c) (Otis et al. 1978). Once each model was run, models that failed to converge or yielded nonsensical population estimates were removed and model averaging was conducted within Program MARK to estimate abundance of each species and the associated 95% CI for each site-year. To convert abundance to density, we divided each abundance estimate by the area that was trapped, which was calculated manually in ArcGIS v10.7.1 (ESRI Inc. Redlands, CA, USA) using measurement units of linear kilometers for agricultural ditches and hectares for ponds.

Some of our trapping sites had few captures or recaptures of some species, precluding CMR abundance estimation for some species at some sites. To estimate density at these sites, we extrapolated density based on that site's capture per unit effort (CPUE; total number of individuals captured per species divided by effort [trap-days]). We first used linear regression analyses to explore the relationship between CPUE and density estimated using CMR for each species within each habitat type (Figure 2). We then used the resulting species/habitat-specific regression equations to extrapolate density based on CPUE at sites where that species was captured, but CMR failed. We were unable to estimate density of *S. odoratus* at any pond sites due to low captures and recaptures (see below). Thus, we extrapolated density of *S. odoratus* for pond sites using the CPUE-density regression from ditches, essentially assuming 1 ha of aquatic habitat per km of ditch, or an average ditch width of 10 m. We assumed zero densities for species at sites where no individuals were captured.

Quantifying available habitat – To quantify the total available aquaculture pond and irrigation ditch habitat available within our 27-county focal region and to estimate how many turtles may occur across the region in these habitats, we digitized all aquaculture ponds in ArcGIS using the USGS National Hydrology Dataset Plus High Resolution (NHDPlus HR) layers and the most recent 30-m resolution LandSat-8 imagery, obtained from the USGS Earth Explorer. We delineated irrigation ditches using the NHDPlus HR layers. We examined, through ground-truthing, a large number (N ~ 50) of known ditches and aquaculture ponds to ensure they were selected using our methods, as well as ensuring that all trapping locations were properly identified. We cleaned our final available habitat database by removing mis-labelled sites (e.g., a slough or large reservoir identified as an exceptionally large aquaculture pond).

Extrapolation of Turtle Density Across the MAV – We used a Monte-Carlo approach to extrapolate density of the three most common turtle species (*T. scripta*, *A. spinifera*, and *S. odoratus*) across the estimated total available habitat in the MAV of Arkansas. We composed simulations in Program R (v.3.5.3) (R Development Core Team, 2019) that randomly assigned turtle densities to each pond (N = 4,723) and ditch segment (N = 24,916) identified using GIS (see above), based on a random draw from density estimates from our ditch (N = 32) and pond (N = 51) sites. Specifically, in each simulation, each pond and ditch was assigned a density based on a draw from the 95% Confidence Interval (CI) of one of our pond or ditch sites, respectively. Because density estimates extrapolated from CPUE (see above) lacked an associated 95% CI, we generated a CI for each of these sites based on the average width of the CI (% of mean) across our CMR estimates for that species/habitat. In a few cases, model averaging resulted in negative lower 95% CI values; in these cases, we adjusted the lower CI to the known minimum density (# unique individuals captured/area) for that species/site. For sites with no captures, we assigned a zero density with no variation. We conducted 1,000 simulations of our Monte-Carlo algorithm for each species and tabulated the total number of turtles in each habitat type and total across the region for each simulation.

Results

Turtle sampling - We trapped at 62 sites across the Mississippi Alluvial Valley of Arkansas, including 41 aquaculture ponds and 21 agricultural ditch segments (Figure 1). We totaled 2434 trap days in agricultural ditches and 1772 trap days in aquaculture ponds. In total, we captured 4,007 individuals of nine species: spiny softshell turtle, red-eared slider, common snapping turtle, eastern mud turtle, common musk turtle. southern painted turtle, Mississippi map turtle, river cooter, and alligator snapping turtle (Table 1).

Diversity and Community Composition – While most species occurred in both habitat types, *M. temminckii* was only encountered in one agricultural ditch and all but one of the eleven *P. concinna* were also captured in agricultural ditches. Four of the five most commonly captured species (*T. scripta*, *S. odoratus*, *K. subrubrum*, *C. serpentina*) comprised a greater percentage of captures in agricultural ditches than in aquaculture ponds, with the exception being *A. spinifera* which comprised a greater percentage of captures in aquaculture ponds than in agricultural ditches (Table 2). The most commonly captured species were *Trachemys scripta* (n = 2695), *Apalone spinifera* (n = 640), *Sternotherus oderatus* (n =508), *Kinosternon subrubrum (n = 81)* and *Chelydra serpentina (n = 56)*. *T. scripta* accounted for a mean (± 1 SD) of 66% (±22%) of all captures in agricultural ditches and 63% (±32%) of all captures in aquaculture ponds. Percent composition of *A. spinifera* was greater in ponds (mean = 26%, ± 29%) than ditches (6% ± 6%; t = -4.26, P > 0.001). (Table 2). *Sternotherus odoratus* was the second most frequently captured turtle in agricultural ditches and accounted for a larger proportion of the turtle community in ditches (mean = 16%, ± 15%) than in ponds (5% ± 7%; t =2.67, P = 0.015) (Table 2). *K. subrubrum* accounted for an average 6% (± 9%) of captures in agricultural ditches and 2% (±

9%) of captures in aquaculture ponds. *C. serpentina* accounted for an average of 6% (± 17%) of captures in agricultural ditches and 3% (± 11%) of captures in aquaculture ponds. Both *C. serpentina* and *K. subrubrum* were caught in similar proportions in each habitat (P > 0.176; Table 2). All other species *(Chrysemys dorsalis, Graptemys pseudogeographica kohnii, Pseudemys concinna,* and *Macrochelys temminckii*) collectively accounted for less than 1% of all unique captures in aquaculture ponds and less than 1% of all unique captures in agricultural ditches.

Species richness of agricultural ditches ranged from 0 to 8, with an average of 3.76 and richness for aquaculture ponds ranged from 0 to 7, with an average of 3.19. Richness was not significantly different (t = 0.81, P = 0.42) between wetland types. Simpson's Diversity values for agricultural ditches ranged from 0 to 0.679, with an average of 0.35 and aquaculture ponds ranged from 0 to 0.714 with an average of 0.347. Simpson's Diversity Index (t = -0.32, P = 0.75) also did not vary between wetland types.

Density Estimation – Through CMR and extrapolation, we were able to calculate 390 density estimates. Of the 62 sites we trapped at, we had sufficient captures and recaptures to generate 86 species-specific abundance estimates through Program MARK. We were most frequently able to estimate the density of *T. s. elegans* (N = 53). For sites sampled multiple times within a year, the CMR estimate with the tightest CI was selected as our site-year estimate. After selecting one estimate per site-year for *T. scripta, A. spinifera,* and *S. odoratus,* we had 249 estimates to use for our analysis. When corrected for the area of wetlands trapped, density estimates for *T. scripta* ranged from 0 – 127.77 (median = 15.64) turtles/ha in aquaculture ponds and 0 – 500.08 (median = 59.58) turtles/km in agricultural ditches. *A. spinifera* estimates ranged from 0 – 67.19 (median = 11.09) turtles/ha in aquaculture ponds and 0 -11.75 *(*median = 4.22) turtles/km in agricultural ditches. *S. odoratus* densities ranged from 0 – 302.36 (median = 8.53) turtles/km in agricultural ditches (Table 3). Due to low unique captures and recaptures, we were unable to estimate densities of *S. odoratus* in aquaculture ponds via CMR.

Habitat Quantification and Total Number of Turtles - Within our study area, we delineated a total of 24,916 agricultural ditches ranging from 0.002 – 10.59 (mean = 0.74) linear kilometers and 4,723 aquaculture ponds ranging from 0.01 – 52.86 (mean = 4.73) hectares (Figure 3). We used the previously presented density estimates for each species in each habitat (Table 4) to extrapolate the total number of each of the three most common species occurring in these habitats across the entirety of the MAV region. Mean total abundance of *T. scripta* across the MAV was 2,168,803 (95% CI = 2,167,081 – 2,170,524) based on 1,000 iterations of our Monte-Carlo simulation (Figure 4). Approximately 65% of that total (mean = 1,408,241) occurred within ditches, whereas 35% (mean = 760,562) was in aquaculture ponds. Mean overall abundance of *A. spinifera* was 427,747 (95%CI = 426,976 – 428,517), the majority (83%; mean = 355,233) of which were in aquaculture ponds, with relatively few (17%; mean = 72,514) in ditches (Figure 4). Mean total abundance of *S. odoratus* was 816,090 (95%CI = 815,210 – 816,970), with 72% occurring in ditches, and 28% in aquaculture ponds (Figure 4).

Discussion

This study presents valuable data collected in two anthropogenic aquatic habitats – agricultural ditches and aquaculture ponds – across a wide study area that was historically part of the largest tract of bottomland hardwood wetlands in North America and has experienced widespread habitat conversion. We provide the first region-wide freshwater turtle abundance estimates for eastern Arkansas, which can be used by management officials to monitor population trends over time and determine conservation regulations region-wide.

Although we captured a total of 9 turtle species in agricultural ditches and aquaculture ponds, the community was dominated by a small number of generalist species. In both habitats, *T. scripta* accounted for >60% of the turtles captured. *T. scripta* are habitat generalists with the ability to disperse overland and colonize new wetlands rapidly (Abigayle 2009, Salzberg 2000). Because of these traits, *T. scripta* have been shown to dominate in other wetland habitats as well, including both natural (Nickerson et al. 2019) and artificial (Dreslik et al. 2005, Elston et al. 2016, Glorioso et al. 2010) wetlands. Clearly, these anthropogenic habitats provide the resource needs for this species. The same is likely true for *K. subrubrum* and *S. odoratus*, which are generalists and have been found in numerous natural and artificial wetland types (Cagle 1942, Konvalina et al. 2016, Sutton and Christiansen 1999). It appears that *A. spinifera* do not constitute a large proportion of the turtle assemblage in ditches but do in aquaculture ponds. *A. spinifera* is an economically and ecologically important species that is in decline in some regions and listed as an Endangered Species in Canada (Hughes 1999, Mahoney and Lindeman 2016, Mali et al. 2014, Zimmer-Shaffer et al. 2014) which makes the ability of *A. spinifera* to attain high densities in aquaculture ponds notable.

Although 14 species of freshwater turtle occur in Arkansas (Trauth et al. 2004), some species known to occur within our study area were not captured in agricultural ditches or aquaculture ponds, suggesting that these habitats are not suitable or used by all species in Arkansas. The chicken turtle (*Deirochelys reticularia miaria*), the smooth softshell turtle (*Apalone mutica*), the razor-backed musk turtle (*Sternotherus carinatus*) and two species of map turtle (*Graptemys ouachitensis, Graptemys geographica*) were not captured as part of this study.

Additionally, some species were captured only rarely (*Graptemys pseudogeographica kohnii*, *Pseudemys concinna*, and *Macrochelys temminckii)*. For most of these species, their absence is likely due to a lack of habitat compatibility, as *A. mutica* and *Graptemys spp*. prefer riverine habitats (Anderson et al. 2002, Barko and Briggler 2006, Moll and Moll 2004). While *M. temminckii* has been occasionally found to use agricultural ditches, the species generally prefers cypress swamps (Harrel et al. 1996). *P. concinna* is broadly distributed, but typically prefers river and lake wetlands (Dreslik et al. 2003). *S. carinatus* is most often found in creeks or rivers with gravel, sand or cobble substrate and their geographic range only overlapped with the southern portion of our study region (Lindeman 2008). *D. r. miaria* range overlaps with our study area and they are known to prefer shallow, slow to still waters with abundant vegetation, which describes many of our agricultural ditch sites, but have been found to avoid bodies of water occupied by large numbers of other turtle species (Buhlmann et al. 2009). Additionally, *D. r. miaria* is rare and declining across this region (Trauth et al. 2004) so the absence of any captures in this study is unsurprising. While the riverine species may occasionally use agricultural ditches because they are often connected to stream and river habitats, they do not form a substantial portion of the turtle assemblage in these anthropogenically altered habitats.

T. scripta represent a large proportion of the turtle assemblage in both ditch systems and aquaculture ponds. Nickerson et al. (2019) found *T. scripta* at high densities in a man-made lake located in the Missouri portion of the MAV, with approximately 252.6 turtles per ha. *T. scripta* was the most abundant species at their study sites in the Mississippi portion of the MAV as well, making up more than 75% of all captures. Similar percentages of *T. scripta* have been found in man-made lakes of Arkansas (77%; Konvalina et al. 2016), urban ditches of Arkansas (85%; Elston et al. 2016), wetlands in the Missouri portion of the MAV (78%; Bodie et al. 2000), and

Illinois wetlands (67%; Dreslik et al. 2005). Based on these findings, it is not uncommon for turtle assemblages in anthropogenic habitats to be comprised heavily of one dominant species, specifically *T. scripta.* Although *T. scripta* were the dominant species the densities we report here (mean = 31.75 turtles/ha; Table 4) in aquaculture ponds was lower than has been reported in many farm, golf course, and city ponds which is surprising given the food supplementation that occurs in fish farm ponds.

There have been fewer investigations of *T. scripta* density in linear habitats, such as the agricultural ditches studied here. However, several examples do exist, and our reported density (85.89 turtles/km) appears to fall in the middle of the reported range of density estimates. Both Elston et al. (2016) and Moll and Legler (1971) reported higher densities of *T. scripta* in ditches and rivers than our study. However, Munscher et al. (2020) reported only 43 turtles/ha of a river and lake system in Texas, lower than our estimated mean. *T. scripta* benefit from habitats that are anthropogenically altered (Mota et al. 2021) and have been expanding their range (Nickerson and Pitt 2012), suggesting that high densities of this species in anthropogenically altered habitats is to be expected.

In contrast to *T. scripta*, *A. spinifera* numbers were estimated to be greatest in aquaculture ponds, with our density estimates ranging between 0 – 67.19 (median = 11.09) turtles/ha, a higher density than in the reported literature, with values ranging from 0.39 turtles/ha in an Illinois lake (Dreslik et al. 2005) to 13.6 turtles/ha in a Missouri lake (Nickerson et al. 2019). These higher densities may be due to supplemental feeding, as aquaculture managers distribute food intended for their fish on a daily basis. *A. spinifera* are also likely attracted to aquaculture ponds because they are often clear, deep pools that are managed to be free of vegetation, habitat attributes often preferred by this species (Barko and Briggler 2006, Ernst and Lovich 2009). However, our density estimates for *A. spinifera* in agricultural ditches (mean = 5.00 ± 2.32 turtles/km) were lower than reported literature for canals, streams and rivers. Shaffer et al. (2017) reported numbers similar to ours in a section of the Missouri river experiencing harvest pressure, but all other reported numbers, including additional estimates from Shaffer et al. (2017), are higher than our estimated densities. This suggests *A. spinifera* populations in agricultural ditches of Arkansas are not as robust as populations in other linear habitats. In addition, the agricultural ditches available in our region likely do not provide the habitat traits that other linear habitats occupied by *A. spinifera* offer, such as clear and cool water.

There was substantial variation in our density estimates both among sites and between site years, with few or no captures one year and hundreds the next. Unlike most studies of turtle abundance, our study used many short-term sampling sessions across a large number of sites to estimate the range of densities that turtles occur in our two focal habitat types. This approach was important for our system because density can be influenced by the habitat but also by legacy effects. For example, some of our sites had experienced commercial turtle harvest or systematic removal which has been shown to reduce turtle density for many years due to the slow life histories of most turtles (Brown et al. 2012, Congdon et al. 1994, Mali et al. 2016). For agricultural ditch sites, and to some extent aquaculture ponds, unpredictable and dynamic hydrology likely causes constantly shifting densities of turtles as drying ditches and draining ponds force turtles out and newly flooded ditches and ponds encourage individuals to migrate in. Within these dynamic wetlands, the timing of sampling may have a very large effect on the number of turtles present and a study design focused on a smaller number of sites may have provided misleading estimates of density. Most studies of turtles have been restricted to one or a small number of focal wetlands (e.g., Beshara 2009, Elston et al. 2016, Konvalina et al. 2016, Mahoney and Lindeman 2016). We are unaware of any attempts to use mark-recapture estimates to extrapolate regional abundance to a geographical area as large as ours, although this approach has been used on a smaller scale for estimating turtle abundance in river sections (Shaffer et al. 2017) and on a larger scale for other taxa such as birds (Wiest et al. 2016).

The Mississippi Alluvial Valley was once almost entirely river floodplain (Twedt and Loesch 1999). The habitats we studied are novel to the region and the turtle communities and densities reported here almost certainly vary substantially from those that occurred historically in more natural aquatic habitats. What role they play in these novel ecosystems remains to be investigated, but when turtles attain high densities they likely play important roles as predators, prey, competitors, scavengers, and nutrient cyclers (Lovich et al. 2018, Mali 2014). Within aquaculture ponds, they are viewed as nuisances, as they compete with and potentially depredate the fish. As new habitats are created, novel communities are formed, which presents challenges to managers who must balance biodiversity conservation with the needs and desires of stakeholders who created and maintain these habitats.

Our estimation of total abundance of turtles across the region was in part in response to the commercial turtle harvest industry in Arkansas. Management cannot be effective without an understanding of current population levels. This is especially important for turtles because their delayed sexual maturity, longevity, and low fecundity make them susceptible to exploitation and slow to recover from harvest (Ceballos and Fitzgerald 2004, Congdon et al. 1994, Lovich et al. 2018, Rachmansah 2015, Rowe 2008). Yet very few rigorous estimates of abundance are available for harvested freshwater turtle populations. Commercial harvest occurs across our study region and currently is not restricted by any size limits, bag limits, closed seasons, or effort limits. In 2019 alone, commercial harvest of 39,840 *T. scripta* and 4,258 *A. spinifera* was reported from the Arkansas' MAV, with only 35% of harvest permit owners reporting (Irwin 2020). Most of these turtles were harvested from aquaculture ponds and irrigation ditches. Our study provides an important baseline for determining the sustainability of commercial turtle harvest in the MAV region.

Tables and Figures

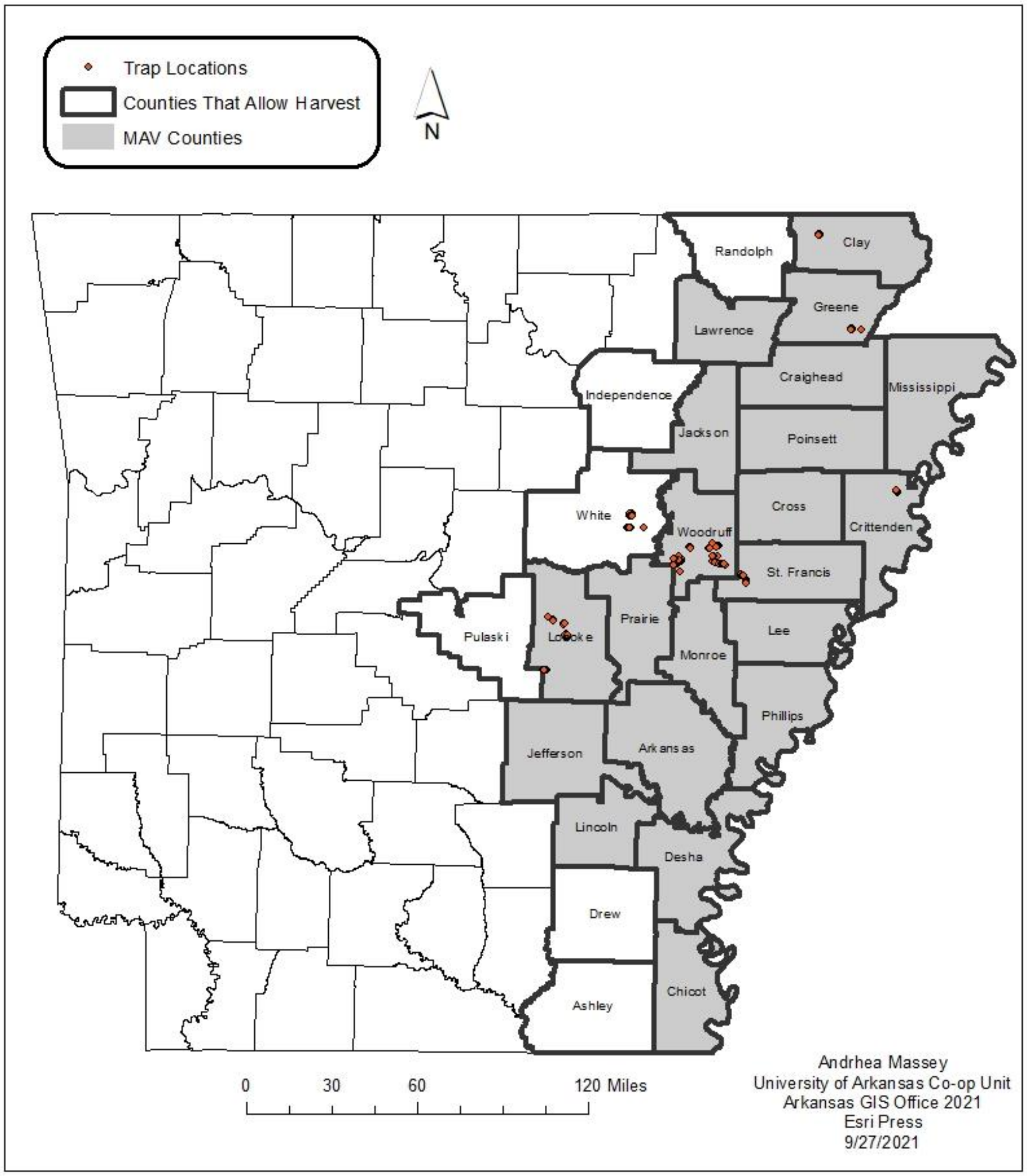

Figure 1: Study region within Arkansas represented by counties heavily outlined and labelled – shaded counties fall within the MAV. Points show locations of traps (2019-2021). Map created with ArcGIS using data downloaded from the Arkansas GIS office.

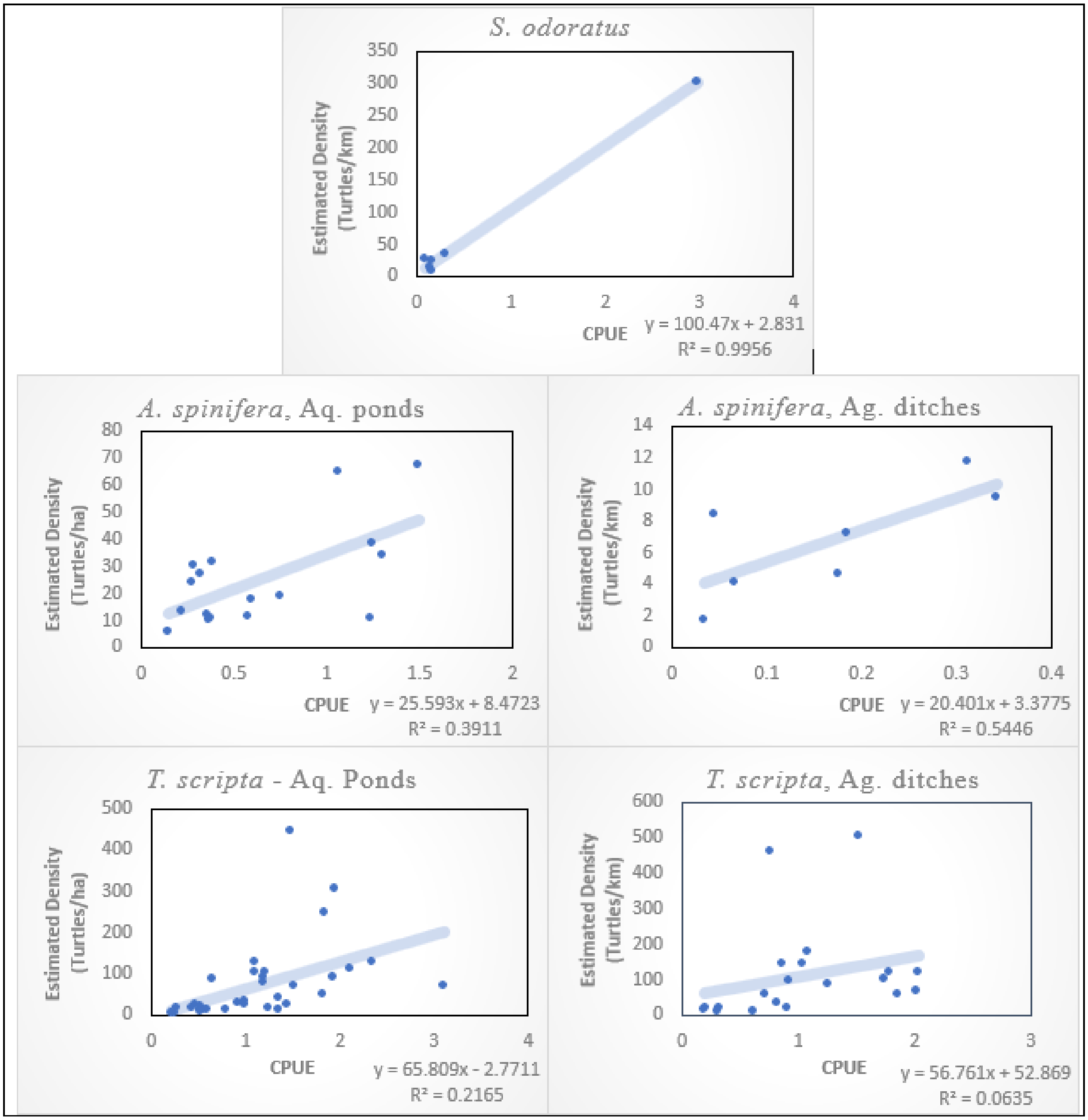

Figure 2. Linear regressions showing the relationship between catch per unit effort (CPUE; individuals captured per trap-day), and estimated density generated from capture-mark-recapture analyses of 3 species of freshwater turtle in Arkansas.

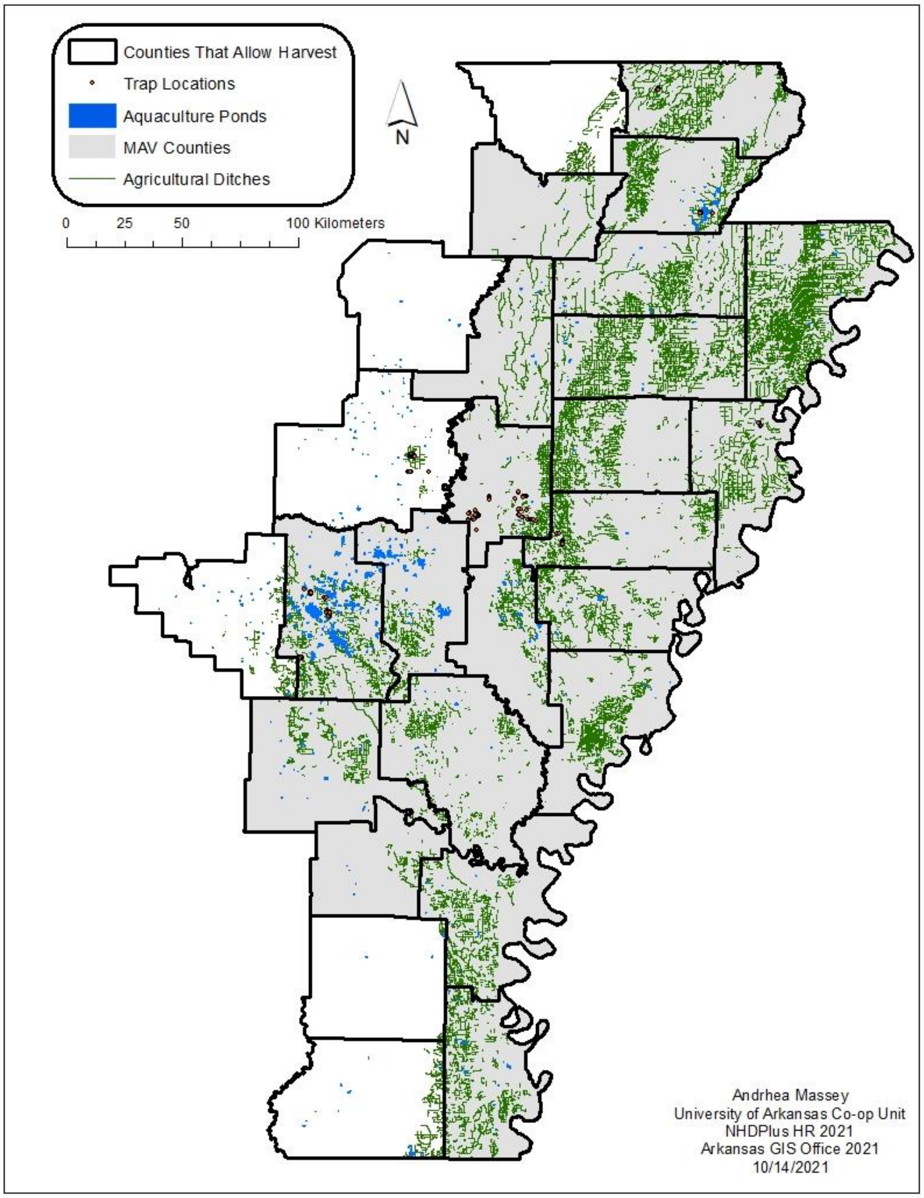

Figure 3. During three summers (2019 - 2021), we trapped turtles to estimate community composition and density in aquaculture ponds and agricultural irrigation ditches across 27 counties in Eastern Arkansas, primarily within the Mississippi Alluvial Valley Ecoregion which corresponds to areas where turtles are legally commercially harvested. We mapped aquaculture ponds and agricultural ditches across our study region with National Hydrography Dataset Plus High Resolution data.

Table 1. Summary of captures by species

Species	Individuals	Recaptures	Grand Total
Red-eared Slider (*T. scripta*)	2695	879	3574
Spiny Softshell Turtle (*A. spinifera*)	640	254	894
Common Musk Turtle (*S. odoratus*)	508	29	537
Eastern Mud Turtle (*K. subrubrum*)	81	19	100
Common Snapping Turtle (*C. serpentina*)	56	4	60
River Cooter (*P. concinna)*	11	0	11
Southern Painted Turtle (*C. dorsalis*)	7	0	7
Mississippi Map Turtle (*G. p. kohnii*)	7	0	7
Alligator Snapping Turtle (*M. temminckii*)	2	0	2
Grand Total	4007	1185	5192

Table 2. Average composition of captures by wetland type, and significance between percent composition of each species in each wetland type

Species	Agricultural Ditch	Aquaculture Pond	Overall	Two Tailed t-tests
Red-eared Slider (*T. scripta*)	66.30%	62.90%	63.91%	t = 0.46, P = 0.648
Spiny Softshell turtle (*A. spinifera*)	6.14%	26.43%	20.38%	t = -4.26, P > 0.001
Common Musk turtle (*S. odoratus*)	15.67%	5.18%	8.31%	t = 2.67, P = 0.015
Eastern Mud turtle (*K. subrubrum*)	5.80%	2.36%	3.38%	t = 1.38, P = 0.176
Common Snapping turtle (*C. serpentina*)	5.68%	2.71%	3.59%	t = 0.66, P = 0.513

Table 3. Range of density estimates for four species of freshwater turtles trapped in two wetland types in Arkansas determined via capture-mark-recapture: lowest - highest (median), number of estimates (n).

Species	Aquaculture Ponds (turtles/ha)	Agricultural ditches (turtles/km)
T. scripta	3.78 - 444.03 (15.65), n = 51	10.69 - 500.08 (59.58), n = 32
A. spinifera	5.60 - 67.19 (11.09), n = 51	1.70 - 11.75 (4.22), n = 32
S. odoratus	N/A	2.83 - 302.36 (8.53), n = 32

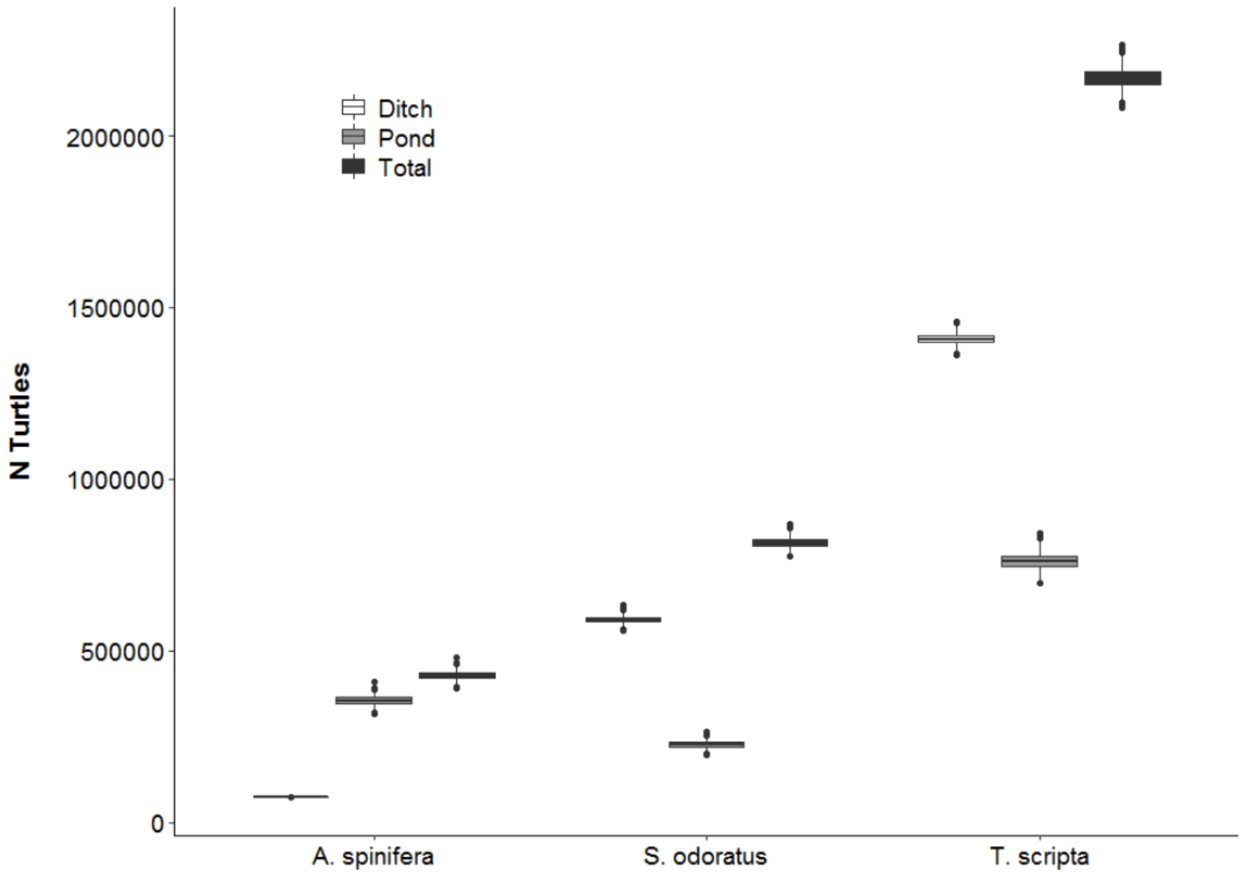

Figure 4. Total estimated abundance of the three most common turtle species in aquaculture ponds, agricultural ditches, and total, across the MAV of Arkansas. Box-whisker plots represent median, 25%/75% quartile, 95% quantile, and outliers, based on 1,000 simulations of a Monte-Carlo extrapolation of density estimates across available habitat in the region.

Table 4. List of densities in multiple habitat types reported in the literature for the three most commonly captured species

T. scripta

Density	Location	Habitat	Citation
7.04 turtles/ha	Arkansas	Lake	Konvalina et al. 2016*
28 turtles/ha	Texas	Pond	Ingold et al. 1986
31.75 turtles/ha	**Arkansas**	**Aquaculture Pond**	**This Study**
42 turtles/ha	S. Carolina	Farm ponds	DeGregorio et al. 2012, Congdon et al. 1986
43 turtles/ha	Texas	River and lake system	Munscher et al. 2020
57 turtles/ha	Illinois	Pond	Dreslik et al. 2005
58 turtles/ha	Chiapas, Mexico	Pond	Dean 1980
61.5 turtles/ha	S. Carolina	Carolina Bay	Congdon et al. 1986
129 turtles/ha	Oklahoma	Ponds	Beshara 2009
135 turtles/ha	Oklahoma	Ponds	Beshara 2009
190 turtles/ha	Panama	River	Moll and Legler 1971
205.8 turtles/ha	Missouri	Lake	Glorioso et al. 2010
247 turtles/ha	Texas	Farm Pond	Rose and Manning 1996
279.7 turtles/ha	Missouri	Lake	Nickerson et al. 2019
333 turtles/ha	Texas	Farm Pond	Rose and Manning 1996
361 turtles/ha	Florida	Pond	Auth 1975
362.5 turtles/ha	Oklahoma	Ponds	Beshara 2009
513 turtles/ha	Texas	Farm Pond	Rose and Manning 1996
983 turtles/ha	Texas	Farm Pond	Rose and Manning 1996
2,200 turtles/ha	N. Carolina	Golf Course Pond	DeGregorio et al. 2012
85.89 turtles/km	**Arkansas**	**Agricultural Ditch**	**This Study**
136.54 turtles/km	Arkansas	Urban Ditch	Elston et al. 2016*

**We converted reported data into a new unit for this table.*

Table 5. (Cont.)

A. spinifera

Density	Location	Habitat	Citation
5 turtles/km	Missouri	River	Shaffer et al. 2017
5 turtles/km	**Arkansas**	**Agricultural Ditch**	**This Study**
9 turtles/km	Missouri	River	Shaffer et al. 2017
9.5 turtles/km	Missouri	River	Shaffer et al. 2017
12.75 turtles/km	Arkansas	Urban Ditch	Elston et al. 2016
24.5 turtles/km	Missouri	River	Shaffer et al. 2017
36 turtles/km	Missouri	River	Shaffer et al. 2017
0.39 turtles/ha	Illinois	Lake	Dreslik et al. 2005
1.9 turtles/ha	Missouri	Lake	Nickerson et al. 2019
1.9 turtles/ha	Missouri	Lake	Glorioso et al. 2012
13.6 turtles/ha	Missouri	Lake	Nickerson et al. 2019
16.42 turtles/ha	**Arkansas**	**Aquaculture Pond**	**This Study**

S. odoratus

Density	Location	Habitat	Citation
2.67 turtles/ha	Illinois	Lake floodplain	Ernst & Lovich 2009, Dreslik et al. 2005
2.67 turtles/ha	Illinois	Lake	Dreslik et al. 2005
25.1 turtles/ha	Missouri	Lake	Nickerson et al. 2020
25.1 turtles/ha	Missouri	Lake	Glorioso et al. 2011
26.77 turtles/km	**Arkansas**	**Agricultural Ditch**	**This Study**
49.5 turtles/ha	Missouri	Lake	Nickerson et al. 2019
60.7 turtles/ha	Oklahoma	Creek	Mahmoud 1969
148.5 turtles/ha	Alabama	Lake	Ernst & Lovich 2009; Dodd et al 1989
150 turtles/ha	Oklahoma	Creek	Mahmoud 1969
1690 turtles/ha	Texas	river and lake system	Munscher et al. 2021

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CHAPTER III: DENSITY AND DEMOGRAPHY OF FRESHWATER TURTLES IN HARVESTED AND UNHARVESTED ANTHROPOGENIC HABITATS OF EASTERN ARKANSAS

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Abstract

In 2018, the Center for Biological Diversity, the IUCN SSC Tortoise and Freshwater Turtle Specialist Group, and dozens of leading turtle biologists urged Arkansas to ban or strictly regulate commercial harvest of its freshwater turtle species, yet baseline data required to make informed commercial turtle harvest management and regulation decisions is lacking and little is known about the effects the current level of harvest has on local turtle populations. We used a Capture-Mark-Recapture approach over three summers (2019-2021) to evaluate population demographics and densities of freshwater turtles in 5 agricultural ditches where harvest was known to occur, 16 agricultural ditches with no known harvest, 14 aquaculture ponds where harvest was known to occur, and 21 aquaculture ponds with no known harvest. We trapped at two aquaculture facilities that had never experienced turtle harvest, one that was commercially harvested 5 years ago (Farm C), and one that was commercially harvested the year prior to our study (Farm B). In total, we captured and marked 3,865 individual turtles and totaled 1,173 recaptures. We captured nine species of turtle including the red-eared slider (*Trachemys scripta;* N = 2612), spiny softshell turtle (*Apalone spinifera;* N = 601), common musk turtle (*Sternotherus odoratus;* N = 494), eastern mud turtle (*Kinosternon subrubrum;* N = 81), common snapping turtle (*Chelydra serpentina;* N = 53), river cooter (*Pseudemys concinna;* N = 11), southern painted turtle (*Chrysemys dorsalis;* N = 7), Mississippi map turtle (*Graptemys pseudogeographica kohnii*; N = 4), and alligator snapping turtle (*Macrochelys temminckii;* N = 2). We found significantly higher densities of red-eared sliders in unharvested ponds than in harvested ponds. Spiny softshell turtles in ponds harvested more than five years prior to our study (Farm C) comprised a larger percentage of the turtle community, and were larger on average than in unharvested ponds, possibly due to the reduction in competition after harvest

occurred. Spiny softshell turtles were never captured in ponds that had directly experienced harvest a year prior to sampling, although few spiny softshell turtles were also captured at an unharvested farm (Farm D), likely due to a short sampling period. While Farm C had missing size cohorts of red-eared sliders and spiny softshell turtles, Farm A, an unharvested farm, had no missing cohorts of either species and all red-eared slider size cohorts were represented at Farm D. Additionally, missing size cohorts of spiny softshell turtles and red-eared sliders at Farm C appear to coincide, further indicating that large scale removal of both species occurred simultaneously at this location and effects of harvest are detectable up to 5 years after the event. However, effects were much less pronounced in ditch habitats, with no differences in estimated densities and limited differences in sex ratios and body sizes, likely driven by the dynamic hydrology of this habitat. We conclude that the effects of commercial turtle harvest are obscured by the dynamic nature of ditches but have persistent effects in pond habitats.

Introduction

Harvest of freshwater turtles is especially detrimental to wild turtle populations due to their unique ecological and life history traits. Turtle life history traits include highly uncertain nest success, low juvenile survival probabilities, and poor resilience to adult mortality (Ernst and Lovich 2009, Wilbur 1975). While turtles have persisted for over 200 million years (Ernst and Lovich 2009), the modern anthropogenic world presents novel threats, such as commercial harvest for human consumption, which targets larger individuals, vehicular mortality, which may especially affect reproductive females searching for nesting sites, and widespread habitat loss and degradation (Ernst and Lovich 2009, Mali et al. 2014, Steen et al. 2006, van Dijk et al. 2000). With the loss of adults before successful reproduction, many populations have begun to decline dramatically. Of the 356 extant turtle species, approximately 61% are threatened or recently extinct. The primary threats to population persistence are often identified as commercial harvesting and/or habitat loss (Lovich et al. 2018, Kelly 2013, Rachmansah et al. 2020).

While humans have been consuming turtles for thousands of years, the rate at which this consumption occurs has increased with the rise in global human populations (van Dijk 2000). Countries in Asia are the leading consumers of turtles worldwide, because turtles are highly valued culturally as a food source and for traditional medicine (Mali et al. 2014). As human populations have skyrocketed and individuals have become more financially flexible, the demand for turtles has increased. Turtle populations in Asia have suffered under the increased pressure, leading to extinction or extirpation of many native turtle species (van Dijk et al. 2000). While local turtle populations have been reduced or eliminated, the demand for turtle meat persists and thus these Asian countries have begun importing wild caught turtles from elsewhere.

The United States became one of the leading exporters of freshwater turtles in the world shortly after the Asian market collapse in the 1990s (Mali et al. 2014) and since that time, commercial turtle harvest in the U.S. has become a contentious issue. Pressure from conservation groups and concern regarding overharvest has led numerous states to heavily regulate or close commercial turtle harvest. Currently, 31 states have completely closed (e.g., Missouri, Alabama) or strictly regulated (e.g., Connecticut, New Jersey, Iowa) commercial turtle harvest, while only nine states currently maintain little to no regulations. States allowing commercial turtle harvest face political pressure to initiate regulation and several states have commissioned studies to guide their regulation decisions. For instance, Missouri closed commercial harvest in response to a study concluding that even low annual harvest rates may be unsustainable and detrimental to turtle populations in Missouri's rivers (Zimmer-Shaffer et al. 2014). In 2008, Texas Parks and

Wildlife Department funded a 5-year study of their native freshwater turtle populations which demonstrated that turtles were highly sensitive to commercial harvest and susceptible to long-term population declines due to over-harvesting. Texas subsequently banned commercial harvest in 2018. As states begin regulating or closing their commercial industries (Brown et al. 2011, Zimmer-Shaffer et al. 2014), there is fear that harvest pressure will shift to the few remaining unregulated states, particularly the southeastern states where freshwater turtle species richness is highest and population densities can be high (Buhlmann et al. 2009; Iverson 1982).

For some species of turtle, any level of harvest may be unsustainable. For instance, for snapping turtles (*Chelydra serpentina*) and softshell turtles (*Apalone spinifera* and *Apalone mutica*) in Missouri, harvest was sustainable only when demographic rates were at maximum values, which is unlikely to occur for wild populations (Zimmer-Shaffer et al. 2014). A long-term study conducted on snapping turtles in Michigan found that an increase of just 10% in mortality of adult females would reduce the population by half in fewer than 20 years (Congdon et al. 1994). Brown et al. (2011, 2012) found that even native slider turtles (*Trachemys scripta*), a generalist species that often attains extremely high densities (Ernst and Lovich 2009), are negatively affected by over-harvesting.

Currently, Arkansas is one of the least regulated states regarding commercial turtle harvest. Arkansas regulations allow for issuance of 150 commercial turtle harvest permits per year to trap unlimited numbers of 13 species of aquatic freshwater turtles, with limited restrictions on equipment (hoop net or basking traps only), and no size or sex restrictions. According to the Arkansas Game and Fish Commission, 1.39 million freshwater turtles have been removed from Arkansas between 2004 and 2017, although this is an underestimate as much of the legal harvest goes unreported (Irwin 2020). In 2019 alone, approximately 48,026 freshwater turtles were commercially harvested from the region, comprised mostly of sliders (n = 39,840) and spiny softshell turtles (n = 4,258). Little is known about the impact this level of removal may be having on Arkansas' freshwater turtle populations and there is concern that regulations are needed to prevent population declines.

The objective of this study was to explore the potential impacts of commercial turtle harvest in Arkansas by comparing population densities and demographics of turtles at harvested and unharvested sites. In eastern Arkansas, much of the commercial turtle harvest occurs in anthropogenically modified habitat consisting of agricultural irrigation ditches and aquaculture ponds. Therefore, we focused our investigation on these widespread and abundant wetland types. Our specific objectives were to: 1) compare the population densities and demographics (i.e., size class distribution and sex ratios) of wetlands protected from harvest to those with known harvest histories, and 2) evaluate differences in the turtle community composition between harvested and unharvested sites.

Materials and Methods

Study sites - We trapped turtles in 7 counties across eastern Arkansas (Figure 5). We selected counties where commercial harvest is currently allowed, sampling two wetland types commonly targeted by local commercial harvesters: aquaculture ponds and irrigation ditches. We selected sites with and without known commercial harvest. We considered sites harvested if turtles had been commercially removed from the location within five years prior to our first sampling period, as reported by landowners, Arkansas Game and Fish Commission biologists, or local turtle harvesters themselves. We considered sites unharvested if they were located on protected land (i.e., state fish hatcheries or management areas explicitly forbidding turtle harvest). We selected sites that were representative of available habitat across our study region with the intention of selecting a broad range of habitats within the constraints of accessibility.

Trapping – We conducted short, intense trapping sessions, marking turtles to estimate populations at a large number of harvested (N = 19) and unharvested (N = 37) sites, some of which were visited repeatedly in a given year in order to better understand variation between years. We sampled turtle populations in 5 agricultural ditches, ranging in size from 0.08 – 2.55 linear km, where harvest was known to have occurred and 16 agricultural ditches, varying in size from 0.06 – 2.29 linear km, with no known harvest history. We sampled 14 aquaculture ponds, varying in size from 0.43 – 10.23 ha, located on two different private fish farms where harvest was known to have occurred, and 21 aquaculture ponds varying in size from 0.08 – 7.00 ha, on two different state-owned fish farms with no known harvest history. We re-sampled ten aquaculture ponds (4 with harvest history, 6 without) and seven ditch sites (4 with harvest history, 3 without) in different years. Given high inter-annual variation in habitat and environmental conditions, as well as turtle densities, we treated the site-year as the replicate in our analyses resulting in a total sample size of 46 pond sites (17 with harvest, 29 without) and 32 ditch segments (13 with harvest, 19 without). We treated individual aquaculture ponds within the farms as separate replicates. We conducted sampling from late April to early August for three consecutive summers (2019 – 2021), capturing turtles with baited hoop net traps (Memphis Net and Twine, Memphis, Tenn.) set approximately 15-30 m apart and left in place for five days (i.e., set Monday, pulled Friday). Number of traps set per site varied from 5 to 35 traps based on wetland size and water depth – water depth must be such that traps are only partially submerged. We used custom hoop nets replicating the trap most often used by local commercial turtle harvesters such that traps had two ‘Arkansas style’ funnel throats and mesh size of 3.175 cm and 50.8 cm diameter. The front of the trap was held open with a wooden stake while the back of the trap was held with PVC, hammered into the substrate so that a minimum of approximately 40% of the trap remained above water, allowing captured turtles to breathe. To avoid traps becoming submerged in the case of unexpected rapid water level rise, a common occurrence in agricultural ditches, we fitted each trap with a flotation device in the back section of the trap. In the case of rising water levels, the PVC allowed the trap to slide up, lifted by the float, avoiding the drowning of captured turtles. We baited traps with raw chicken or fish. To avoid spreading disease in aquaculture farms, we only used chicken as bait and decontaminated traps with a 1% Virkon® Aquatic solution between each trapping session (Stockton-Fiti 2017).

We checked traps daily to remove all captured turtles and to replace bait. We recorded the mass, straight-line carapace length (SCL) and straight-line plastron length (SPL) of each captured turtle. Mass was measured to the nearest gram using a Pesola® scale. We measured straight-line carapace length (SCL) and straight-line plastron length (SPL) to the nearest millimeter using a Mantax Blue® caliper. We recorded the sex of each turtle by examining secondary sexual characteristics specific to each species (Ernst and Lovich 2009). Turtles lacking secondary sexual characteristics and below size limits for sexual maturity as determined for each species (Ernst and Lovich 2009) we recorded as juveniles. We individually marked all captured turtles. We marked hard-shelled turtles using a Dremel or hand file to notch the marginal scutes with a unique code, a modified version of the Nagle et al. (2017) marking schematic. We marked soft-shelled turtles with a 2.5 cm metal clip tag displaying a unique number attached to the posterior edge of the carapace (Ostovar et al. 2021; National Band and

Tag Company, Newport, KY, USA). We released all turtles near the trap they had been captured in immediately after processing.

Diversity and Community Composition – To assess if turtle harvest altered the community composition and diversity of turtles, we calculated the species richness and Simpson's Diversity Index at each site, as well as the proportion composition of each turtle species at each site. Species richness was simply calculated as the number of turtle species captured for each site and proportion composition was the number of captured individuals of a given species divided by the total number of captured individuals of all species at that site. Simpson's Diversity Index is a measure of diversity which takes into account both the number of species present (richness) and the relative abundance of each species (evenness) through the following equation:

$$D = 1 – (\Sigma n(n\text{-}1)/N(N\text{-}1)$$

where n is the number of individuals of each species and N is the total number of individuals of all species (Simpson 1949). Values range from 0 (no diversity) to 1 (infinite diversity). We compared these three values between harvested and unharvested ditches and ponds using two-tailed t-tests.

Density Estimation – We used closed capture-mark-recapture (CMR) analyses in Program Mark (Cooch and White 2012) to estimate the density of the three most frequently captured turtle species; the spiny softshell turtle, red-eared slider, and common musk turtle (*Sternotherus odoratus*) at each site. Data (i.e., weekly encounter histories) assumed populations were closed for each of the 5-day trapping sessions. We ran three models (M_0, M_t, M_b) for the 5-day encounter histories for each site-year using the Closed Populations – Full Likelihood p and c data type. This data type assumes 1) the population is closed to births, deaths, immigration, and emigration during the (5-day) sampling period, 2) no tags are lost, 3) there is no human error in recording data, and 4) probability of capture during each sampling occasion is constant and equal. Model M_0, the "null" model, assumes capture probability is constant with respect to all factors; model M_t assumes capture probabilities vary over time or trapping occasion (day), and model M_b assumes capture probabilities vary due to behavioral responses (i.e., trap happy or trap shy response), with initial capture probability (p) allowed to vary from recapture probability (c) (Otis et al. 1978). Once we had run each model, we removed models that failed to converge or yielded nonsensical population estimates and conducted model averaging to estimate abundance of each species and the associated 95% CI for each site-year. To convert abundance to density, we divided each abundance estimate by the area that was trapped, which was calculated manually in ArcGIS v10.7.1 (ESRI Inc. Redlands, CA, USA) using measurement units of linear kilometers for agricultural ditches and hectares for ponds.

Some of our trapping sites had few captures and recaptures of some species, precluding CMR abundance estimation for some species at some sites. To estimate density at these sites, we extrapolated density based on that site's capture per unit effort (CPUE; total number of individuals captured per species divided by effort [trap-days]). We first used linear regression analyses to explore the relationship between CPUE and density estimated using CMR for each species within each habitat type. We then used the resulting species/habitat-specific regression equations to extrapolate density based on CPUE at sites where that species was captured, but CMR failed. We were unable to estimate density of musk turtles at any pond sites. Thus, we extrapolated density of musk turtles for pond sites using the CPUE-density regression from ditches, essentially assuming 1 ha of aquatic habitat per km of ditch, or an average ditch width of 10 m. We assumed zero densities for species at sites where no individuals were captured.

We then compared estimated densities of red-eared sliders, spiny softshell turtles, and common musk turtles between harvested and unharvested ditch sites using two-tailed t-tests. For aquaculture pond sites, we compared estimated densities of the three species between the four aquaculture facilities rather than simply grouping into harvested and unharvested. We chose this approach because the two harvested fish farm facilities experienced harvest at very different times prior to our sampling (~5 yrs. vs 1 yr.). We were unable to attain this level of resolution for ditch sites and thus grouped them as simply harvested or unharvested. We used Kruskall Wallis tests to look for differences in density among ponds at the four aquaculture facilities and then used post-hoc Wilcoxon Rank Sum tests to make pair-wise comparisons between the different aquaculture facilities.

Differences in demography –To assess differences in demography between harvested and unharvested sites we focused on the two most commonly captured species (red-eared slider and spiny softshell turtle) and considered male and female spiny softshell turtles separately due to their strong sexual size dimorphism (Ernst and Lovich 2009). For each species, sex, and habitat type, we placed each captured individual into SCL size categories ranging from 0 to >430 mm with 10- or 20-mm intervals. We then compared size class distributions between farms and harvested or unharvested ditch sites using contingency table analysis. We compared mean SCLs of males and females between harvested or unharvested sites using two-tailed *t*-tests. For all demographic analysis we only used data from the first capture of each individual turtle.

In addition, we explored whether harvest affected the sex ratios of turtles present at each site. For each site, we calculated the number of males per female, M:F, for each species. We used two tailed *t*-tests to compare the sex ratios present at harvested to unharvested ditches and at harvested to unharvested ponds. We conducted all statistical analysis in Microsoft Excel and Program R (v.3.5.3) (R Development Core Team, 2019).

Results

Trapping –We totaled 2,317 trap days in agricultural ditches (N = 1015 in harvested ditches, N = 1302 in unharvested ditches) and 1,571 trap days in aquaculture ponds (N = 326 in harvested ponds, N = 1245 in unharvested ponds). We captured 3,865 individual turtles and totaled 1,173 recaptures. We captured nine species of turtle, including the red-eared slider (*Trachemys scripta;* N = 2612), spiny softshell turtle (*Apalone spinifera;* N = 601), common musk turtle (*Sternotherus odoratus;* N = 494), eastern mud turtle (*Kinosternon subrubrum;* N = 81), common snapping turtle (*Chelydra serpentina;* N = 53), river cooter (*Pseudemys concinna;* N = 11), southern painted turtle (*Chrysemys dorsalis;* N = 7), Mississippi map turtle (*Graptemys pseudogeographica kohnii*; N = 4), and alligator snapping turtle (*Macrochelys temminckii;* N = 2; Table 5).

Diversity and Community composition – Red-eared sliders dominated the turtle community in three of the four categories, comprising a mean (± 1 SD) of 61% (± 22%) of individuals captured in harvested ditches, 68% (± 24%) of individuals captured in unharvested ditches, and 74% (± 27%) of individuals captured in unharvested ponds (Table 6). However, in harvested ponds, red-eared sliders comprised only 38% (± 35%) of individuals captured. Percent community composition of red-eared sliders was significantly higher at unharvested pond sites (t = -2.94, P < 0.01) than at harvested pond sites, but we detected no statistical significance between harvested and unharvested agricultural ditch sites (t = -0.56, P = 0.59).

Spiny softshell turtles were the most frequently captured turtle in harvested ponds, comprising 49% (± 38%) of individuals captured, which was significantly higher than that of unharvested ponds (16% (± 16%); t = 2.73, P = 0.02). While percent composition of spiny softshell turtles was not significantly different between harvested and unharvested ditches (t = -0.33, P = 0.75), spiny softshell turtles only comprised about 6.5% (± 6%) of captures in agricultural ditches overall. We found no difference in percent community composition between site categories for common musk turtles (Ditches; t = 1.41, P = 0.21; Ponds: t = 1.24, P = 0.23), eastern mud turtles (Ditches; t = 0.26, P = 0.80; Ponds: t = 0.21, P = 0.84), or common snapping turtles (Ditches; t = -1.13, P = 0.28; Ponds: t = -0.90, P = 0.38).

We found no significant differences in Simpson's Diversity Index or species richness between harvested and unharvested ponds (Table 7; Simpsons Diversity Index: t = 0.74, P = 0.47. Richness: t = -0.80, P = 0.43). However, harvested agricultural ditches had significantly higher mean diversity (mean = 0.41; t = 2.29, P = 0.04) and mean richness (mean = 3.46; t = 5.08, P < 0.001) than unharvested agricultural ditch sites (diversity mean = 0.28, richness mean = 2.56).

While most species were caught in both harvested and unharvested aquaculture ponds and agricultural ditches, alligator snapping turtles were only encountered in one agricultural ditch with recent harvest history. Likewise, ten of the eleven river cooters were captured at harvested sites. In contrast, southern painted turtles were only captured at sites with no prior harvest history.

Density Estimation – Through CMR and extrapolation, we were able to calculate 390 density estimates. For sites sampled multiple times within a year, the CMR estimate with the tightest CI was selected as our site-year estimate. After selecting one estimate per site-year for red-eared sliders, spiny softshell turtles, and common musk turtles, we had 234 estimates to use for our analyses (Table 8). When corrected for the area of wetlands trapped, density estimates for red-eared sliders ranged from 0 – 68.43 (mean = 13.43) turtles/ha in harvested aquaculture ponds and 0 – 127.77 (mean = 44.54) in unharvested aquaculture ponds. We found significantly higher densities of red-eared sliders in unharvested ponds (t = -3.41, P < 0.01) than in harvested ponds. Red-eared slider density estimates in unharvested ditches ranged from 0 – 500.1 (mean = 86.34) turtles/linear km and 0 – 177.48 (mean = 85.23) turtles/linear km in harvested ditches, but we found no significant difference between the two harvest categories (t = -0.04, P = 0.98).

When comparing density estimates among the four aquaculture farms, we found significant differences in red-eared slider density among farms (Table 9; χ^2 (3) = 12.57, P < 0.01), driven by extremely low density at Farm B, the aquaculture facility that had been harvested 1 year prior to our study (Table 8).

Estimated density of spiny softshell turtles did not differ between unharvested and harvested ponds (t = 1.69, P = 0.63) or ditches (t = 1.49, P = 0.15). However, when exploring differences among the four aquaculture facilities, we found that softshell densities varied among the farms (Table 10; χ^2 (3) = 26.09, P < 0.01), driven by low densities at Farm B (harvested one year prior) and Farm D (unharvested but sampled only once) (Table 8).

We found no significant difference in density of common musk turtles between harvested and unharvested agricultural ditches (*t* = 1.92, *P* = 0.08) or between harvested and unharvested aquaculture ponds (*t* = -1.08, *P* = 0.29), although we caught no common musk turtles in ponds at Farm D (unharvested). Capture-recapture events did not occur frequently enough to draw many conclusions for common snapping turtles – however, it is important to note that of our 23 captures in aquaculture ponds, only four (17%) occurred in ponds with history of harvest. Of our 30 captures in ditch systems, only eight (27%) occurred in ditches with a history of harvest.

Differences in Demography – Both male and female red-eared sliders captured in unharvested ditches had significantly larger mean SCL than those captured in harvested ditches (Table 11; Females: *t* = -5.77, *P* <0.01; Males: *t* = -2.79, *P* <0.01). We found no significant differences in size of male or female red-eared sliders captured in harvested or unharvested aquaculture facilities (Male: *t* = 0.83, *P* = 0.40; Female: *t* = -.053, *P* = 0.59). Red-eared slider size frequency distributions (Figure 6) in pond habitats were associated with harvest status (χ^2 (28) = 231.5, *P* < 0.01), and so were size frequencies for this this species in ditch habitats (Figure 7; χ^2 (14) = 94.71, *P* < 0.01). Larger (>230 mm SCL) and smaller (<120 mm SCL) size cohorts were largely absent for this species at Farm C, where harvest had occurred approximately five years prior. Farm B, where harvest had occurred the year before sampling, exhibited mostly absent larger size cohorts (>240 mm SCL) for this species. No size cohorts were absent for Farms A or D, where no harvest has occurred.

Female spiny softshell turtles captured in harvested aquaculture ponds had larger mean SCL than individuals captured in unharvested aquaculture ponds (t = 2.15, P = 0.03), but we found no significant difference in size for females in harvested vs unharvested ditches (Table 11; t = 0.18, P = 0.86). We found no significant difference in male SCL in aquaculture ponds (t = 1.76, P = 0.08) or ditches (t = 0.30, P = 0.76). Spiny softshell size frequencies in ponds were associated with harvest status (Figure 8 (χ^2 (87) = 191.39, P < 0.01), but the same is not true for ditch habitats (Figure 9; χ^2 (29) = 30.48, P = 0.39). Larger (> 410 mm SCL) and smaller (<171 mm SCL) size cohorts were largely absent for this species at Farm C, where harvest had occurred approximately five years prior. No size cohorts were absent for Farm A, where no harvest has occurred, but few spiny softshell turtles were captured at Farm B (N = 3) where harvest occurred the year prior, or Farm D (N = 2) where no harvest has occurred, but sampling was limited to a single session.

We found that sex ratios were not significantly different between harvested and unharvested wetlands in either habitat type for red-eared sliders (ponds: t = -0.77, P = 0.46; ditches: t = -1.50, P = 0.16), musk turtles (ponds: t = 1.21, P = 0.26; ditches: t = 1.35, P = 0.22), or mud turtles (ponds: t =1.63, P = 0.20; ditches: t = 0.42, P = 0.69). We found that sex ratios were more strongly male biased for spiny softshell turtles in unharvested ditches (.87M: 1F; t = 2.85, P = 0.02), but we found no significant difference in sex ratio between harvested and unharvested ponds (t = -1.94, P = 0.09). We captured too few common snapping turtles in harvested ponds (one male, two females) to yield meaningful results in sex ratio comparisons for ponds, and sex ratios between harvested and unharvested ditch habitats were not significantly different (t = 0.92, P = 0.43).

Discussion

During the course of this study, we most frequently captured red-eared sliders and spiny softshell turtles, the two most frequently harvested species in Arkansas (Irwin 2020). Our results indicate that harvest affects the density and size class distribution of both of these species but that the effects vary by habitat and are different between the species. Aquaculture facilities that had experienced recent harvest had lower densities and percent composition of red-eared sliders. We found that in ponds that had experienced harvest ~5 years prior, spiny softshell turtles had larger SCL on average and occurred in greater densities than in ponds with no harvest. However, spiny softshell turtles were absent in ponds that had directly experienced harvest one year prior to sampling. Curiously, few spiny softshell turtles were also captured at Farm D, which had not experienced harvest, but this is likely due to limited sampling at this site (N = 156 trap days, or 1 week).

We found relatively few consistent effects of commercial turtle harvest on density or demography of our focal species occurring in agricultural ditches, likely due to the dynamic nature and hydrology of ditches. In general, our results indicate that harvest has effects on populations that can be detected years after the initial harvest event, although these effects are more clearly seen in aquaculture ponds than in agricultural ditch systems.

By examining densities between our four aquaculture farm facilities, we were better able to interpret harvest effects because there were numerous sampling ponds that had all experienced the same or similar harvest pressure. Density of sliders varied among the four different aquaculture facilities (Table 8). We found that mean slider density at an aquaculture facility that was recently harvested (Farm B: 1 year prior to our study) was approximately 1/9th the density of unharvested sites (Table 8). Similarly, Farm C, a facility that was harvested approximately 5 years prior, still had only 1/3rd of the mean slider density than at unharvested aquaculture facilities. These results indicate that harvest directly reduces the density of sliders at aquaculture ponds and the effects are discernable for at least 5 years. Additionally, we identified missing slider size cohorts at both harvested farms (Farm B and C) but not at either of the unharvested facilities (Farm A and D; Figure 6). Although density was lower and size cohorts were missing in harvested ponds, the differences in sex ratios were not discernible, likely because harvesters keep all captured turtles rather than targeting specific sexes. This method of harvesting may be less likely to produce gaps in size distribution, but more likely to contribute to overall population decline due to all adults and subadults being targeted, resulting in overall decrease in reproductive adults (Dodd et al. 2016). While sliders are generalist species and may be capable of withstanding some removal, Brown et al. (2012) found that they are susceptible to harvest, and our results confirm that commercial turtle harvest severely reduces local populations.

Softshell densities also varied between the four aquaculture facilities (Table 8), although somewhat opposite the results for red-eared sliders. Few spiny softshell turtles were captured at either Farm B (harvested one year prior) or Farm D (unharvested but sampled only once). Mean density was highest at the aquaculture facility that had experienced harvest 5 years prior (Farm C), nearly double the mean density at the robustly sampled unharvested facility (Farm A). Spiny softshell turtles at the facility harvested five years prior comprised a higher percent of community composition, had higher density, and were larger than in unharvested ponds, suggesting a possible release from competition (Hill and Vodopich 2013, Pearson et al 2015). We identified missing size cohorts of female spiny softshell turtles at Farm C, but not at Farm A (Figure 8), suggesting this was an effect of harvest. Additionally, missing size cohorts of spiny softshell turtles at Farm C seemed to coincide with the missing slider cohorts at this facility, further indicating that large scale removal of both species occurred simultaneously at this location. Larger and smaller cohorts were missing for both species, suggesting that, during the harvest event approximately five years prior, the number of reproductive adults was severely reduced and smaller turtles that were not removed have only recently been recruited into reproductive adult cohorts.

In contrast to clear differences in turtle density and demography in ponds with and without recent harvest history, our results from ditch habitats were less consistent. While we found evidence that mean SCL of both male and female red-eared sliders in harvested ditches is smaller than in unharvested ditches, we detected no statistical differences in density and size class frequencies. Ditches within our study area are used mainly for irrigation and are subject to periodic flooding and drying as irrigation needs change throughout the year (Figure 10). This highly unpredictable and dynamic hydrology results in constantly shifting populations of turtles as drying ditches force turtles out and newly flooded ditches encourage individuals to migrate in. Our density estimates in ditches ranged from 0 turtles/km to more than 500 turtles/km, and we had trapping sessions where hundreds of individual turtles (N = 398) were captured one year and very few (N = 53) at the same site the next year and vice versa. Many of our ditch sites were connected to permanent stream or river systems and we hypothesize that turtles move freely into and out of these ditch segments as water allows and population dynamics at these sites are driven more by an immigration/emigration model, obscuring harvest effects. In addition, turtles (particularly red-eared sliders) have a remarkable ability to immigrate into depleted areas immediately following large scale removal events (Mali et al. 2016), have been known to utilize newly flooded habitats preferentially (Bodie and Semlitsch 2000), and have been noted moving between canals and larger waterbodies (Bodie and Semlitsch 2000, Glorioso et al 2010). Thus, while our results show that sex ratios and size class distributions for some species vary between harvested and unharvested ditches, we recognize that many of these results are likely due more to immigration and emigration patterns than they are reflective of harvest effects.

In conclusion, our results provide mixed evidence on the impact of commercial turtle harvest in anthropogenically altered aquatic habitats in eastern Arkansas. Our analysis of aquaculture farms indicates that harvest has effects on local populations of red-eared sliders and spiny softshell turtles that may be detected years after the initial event. These results bolster those of other studies that have detected body size differences and missing size cohorts in harvested populations of sliders (Brown et al. 2011, Close and Siegel 1997) and decreased density of spiny softshell turtles and sliders years after harvest (Brown et al. 2012). However, our analysis of agricultural ditches suggests that substantial movement may mask these effects in lotic systems or large water bodies. The dynamic nature of these aquatic habitats, and the fact that harvest pressure is generally unknown, especially on private lands, makes it difficult to say whether current harvest pressure across the region is sufficient to result in widespread declines over time. Our data provides a foundation for population modeling, which is an important tool for extrapolating these impacts and better informing management of Arkansas' freshwater turtle species.

Figures and Tables

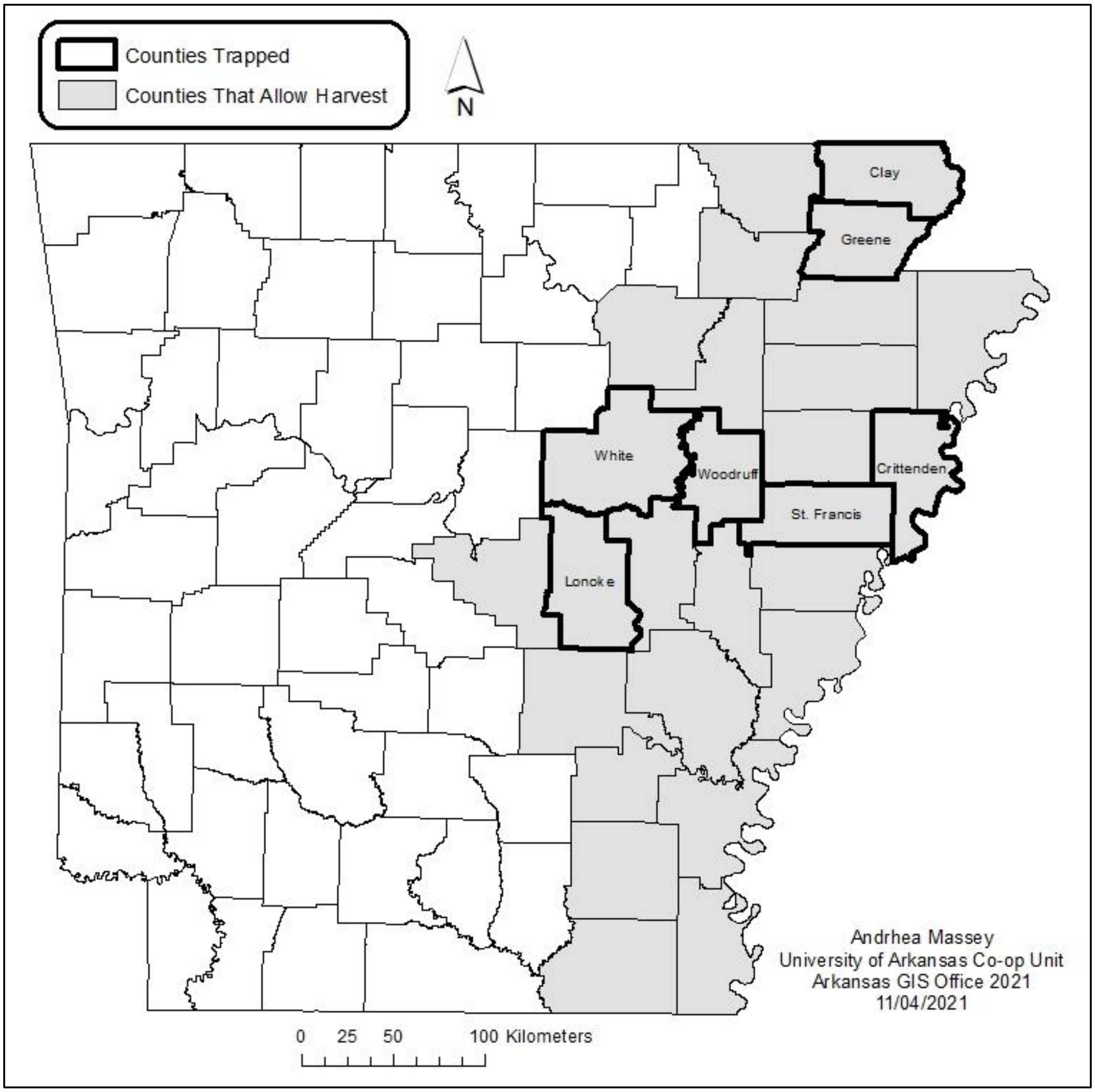

Figure 5. Throughout three summers (2019-2021), we sampled seven counties (outlined in bold) in Eastern Arkansas where commercial harvest of freshwater turtles is allowed (shaded counties). The study region makes up approximately one-third of the state.

Table 6. Count of individual captures of all encountered species in ditches and aquaculture ponds of eastern Arkansas that have and have not experienced commercial harvest of aquatic turtles.

	Harvested Ditches	Unharvested Ditches	Harvested Ponds	Unharvested Ponds	**Total Captures**
Trachemys scripta	759	617	131	1105	2612
Apalone spinifera	70	64	119	348	601
Sternotherus odoratus	294	124	18	58	494
Kinosternon subrubrum	29	33	6	13	81
Chelydra serpentina	8	22	3	20	53
Pseudemys concinna	9	1	1	0	11
Chrysemys dorsalis	0	2	0	5	7
Graptemys p. kohnii	3	0	1	0	4
Macrochelys temminckii	2	0	0	0	2
Total	1174	863	279	1549	3865

Table 7. Mean (± 1 SD) percent composition of the five most commonly captured species in ditches and aquaculture ponds of eastern Arkansas that have and have not experienced commercial harvest of aquatic turtles.

	Harvested Ditches	Unharvested Ditches	Harvested Ponds	Unharvested Ponds
Trachemys scripta	61% (± 22%)	68% (± 24%)	38% (± 35%)	74% (± 27%)
Apalone spinifera	6% (± 4%)	7% (± 7%)	49% (± 38%)	16% (± 16%)
Sternotherus odoratus	25% (± 19%)	12% (± 13%)	7% (± 8%)	4% (± 8%)
Kinosternon subrubrum	6% (± 9%)	5% (± 8%)	3% (± 7%)	3% (± 10%)
Chelydra serpentina	1% (±1.4%)	8% (± 2%)	1% (±2.6%)	4% (±1.4%)

Table 8. Mean (± 1 SD) Simpsons Diversity Index and Species Richness ditches and aquaculture ponds of eastern Arkansas that have and have not experienced commercial harvest of aquatic turtles.

	Harvested Ditches	Unharvested Ditches	Harvested Ponds	Unharvested Ponds
Simpsons Diversity Index	0.49 (± 0.15)	0.28 (± 0.25)	0.36 (± 0.18)	0.31 (± 0.21)
Species Richness	6.0 (± 1.0)	2.56 (± 2.03)	2.64 (± 1.34)	3.05 (± 1.63)

Table 9. Estimated density ranges(turtles/ha) and (mean density) of red-eared sliders and spiny softshell turtles in aquaculture farm facilities and ditch-harvest categories of eastern Arkansas. Farm A and Farm D have not had turtles removed, while Farm C and Farm B have. A commercial turtle harvester visited Farm C ~ 5 years prior to sampling, and Farm B <1 year prior to sampling.

	Apalone spinifera	*Trachemys scripta*	*Sternotherus odoratus*
Farm A (U)	0 - 67.20 (16.33)	0 - 127.77 (45.42)	0 – 74.59 (11.00)
Farm D (U)	0 - 9.75 (3.14)	0 - 112.41 (41.14)	*0*
Farm B (H)	0 - 11.89 (2.45)	0 - 15.64 (4.79)	0 – 23.39 (3.68)
Farm C (H)	0 - 43.46 (27.83)	0 - 68.44 (15.7)	0 – 27.95 (9.64)
Unharvested Ditches	0 – 10.44 (3.62)	0 - 500.1 (86.34)	0 -74.59 (8.52)
Harvested Ditches	0 – 11.75 (5.21)	0 – 177.48 (85.23)	0 – 27.95 (6.48)

*Table 9. Wilcoxon Rank Sum test of Red-eared slider density estimates between aquaculture farms. Farm B and Farm C have experienced commercial harvest of turtles, Farm A and Farm D have not. *significant results*

Trachemys scripta	Farm D	Farm B	Farm A
Farm B	0.051	-	-
Farm A	0.767	0.019*	-
Farm C	0.159	0.444	0.064

*Table 10. Wilcoxon Rank Sum test of Spiny softshell density estimates between aquaculture farms. Farm B and Farm C have experienced commercial harvest of turtles, Farm A and Farm D have not. *significant results*

Apalone spinifera	Farm D	Farm B	Farm A
Farm B	0.9397	-	-
Farm A	0.0036*	0.0025*	-
Farm C	0.0032*	0.0024*	0.0025*

Table 11. Range and mean (in parenthesis) straight carapace length (mm) for male and female red-eared sliders and spiny softshell turtles captured within harvested (H) and unharvested (U) aquaculture farms and ditches of eastern Arkansas, where commercial turtle harvest takes place.

	Male		Female	
	Spiny softshell	Red-eared slider	Spiny softshell	Red-eared slider
Farm A (U)	114 - 216 (175)	103 - 310 (189)	95 - 440 (300)	90 - 275 (201)
Farm D (U)	0	111 - 214 (168)	238 - 312 (275)	98 - 241 (162)
Farm B (H)	0	123 - 224 (186)	364 - 440 (405)	108 - 244 (198)
Farm C (H)	97 - 204 (180)	112 - 229 (189)	116 - 397 (323)	121 - 249 (181)
Unharvested Ditches	84 - 204 (163)	94 - 230 (175)	100 - 414 (279)	83 - 253 (198)
Harvested Ditches	135 - 194 (165)	84 - 220 (154)	124 - 407 (282)	89 - 252 (181)

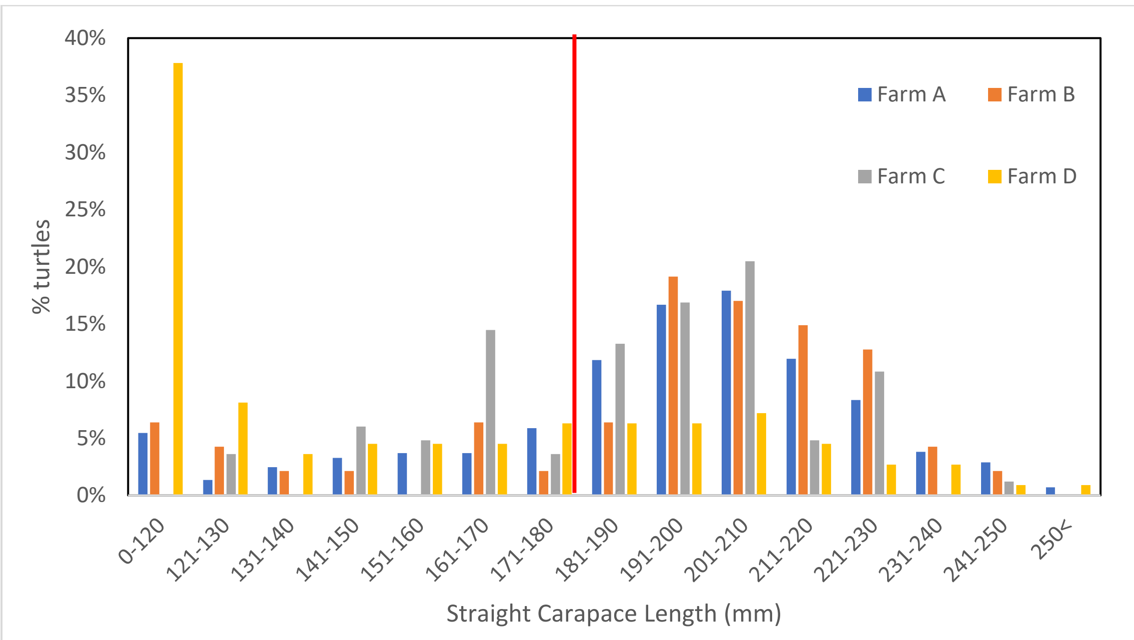

Figure 6. Red-eared slider size distributions (# turtles within 10 mm SCL increments) at four aquaculture ponds. Red line indicates size at sexual maturity (SCL > 177 mm). Note robust bell curve of the population at Farm A, an aquaculture farm with no prior commercial turtle harvest.

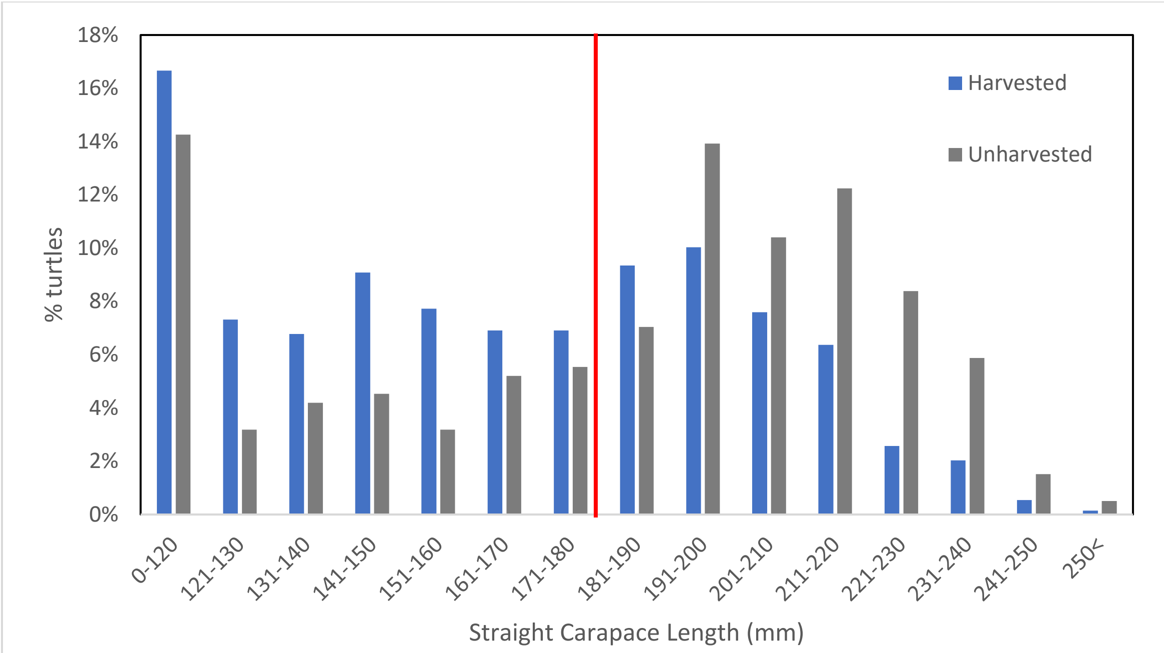

Figure 7. Size Distributions (# of individuals within 10 mm SCL increments) of red-eared sliders in agricultural ditch habitats that have and have not been commercially harvested. The red line indicates size at sexual maturity (SCL > 177 mm).

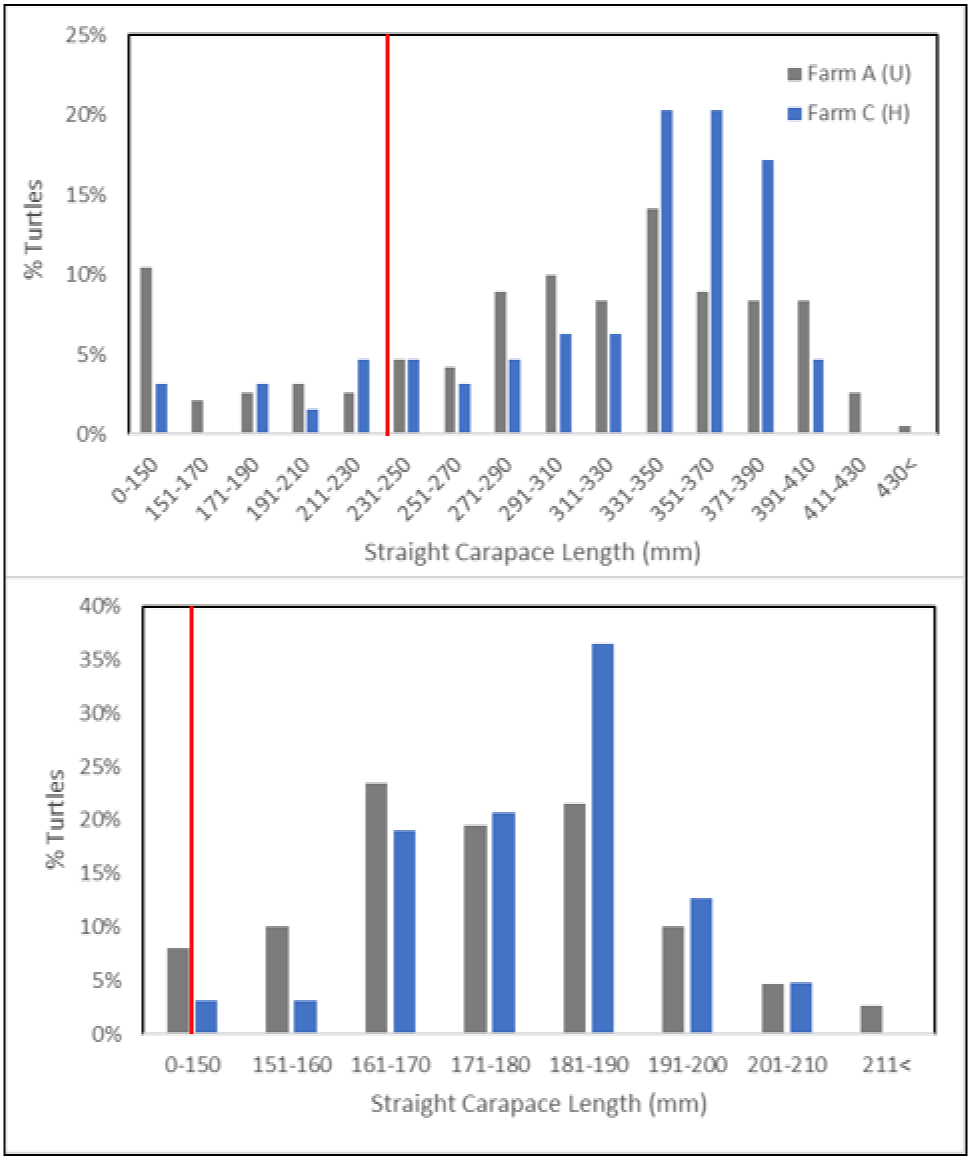

Figure 8. Size distributions (# of turtles within 10 and 20 mm SCL increments) of female(top) and male (bottom) spiny softshell turtles at two aquaculture farms, excluding Farms B and D due to too few captures. The red line indicates straight carapace length (SCL) at sexual maturity (Females = SCL >248mm, Males = SCL > 145 mm). Note the robust bell curve of the population, in particular for females, at Farm A, an aquaculture farm with no prior commercial turtle harvest. Note also the missing cohorts of Farm B, an aquaculture farm with prior harvest.

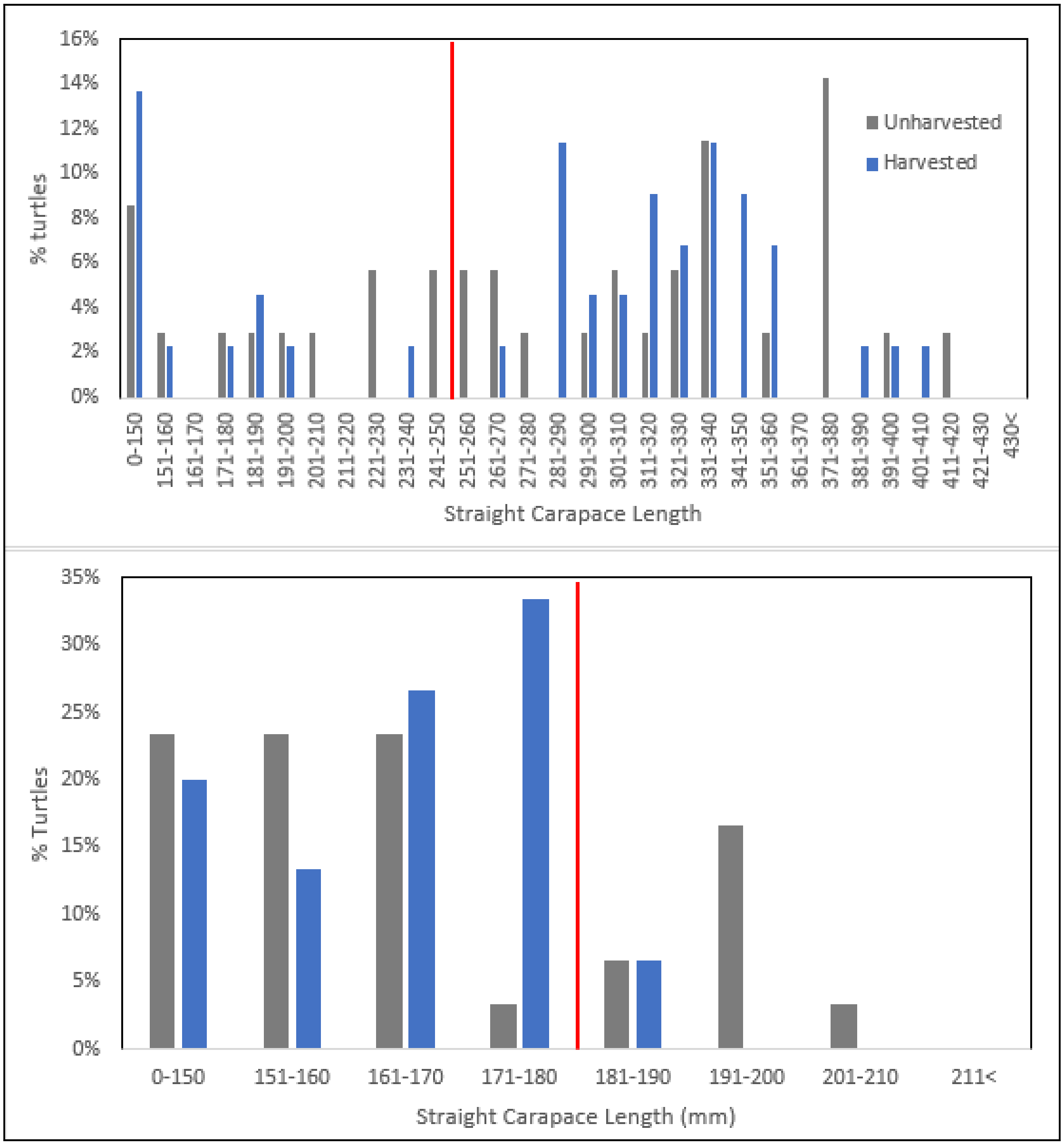

Figure 9. Size distributions (# individuals within 10 mm SCL increments) of female (top) and male (bottom) spiny softshell turtles in agricultural ditch systems that have and have not experienced commercial harvest. Note the variability in both categories and overall small sample size. The red line indicates size at sexual maturity (Females = SCL >248mm, Males = SCL > 177 mm).

Figure 10. Hydrology in agricultural ditches of eastern Arkansas is very dynamic. Water levels can fluctuate drastically overnight due to rain or changing irrigation needs.

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CHAPTER IV: CONCLUSIONS

In this thesis we provide baseline information important in making management decisions to state biologists. While our results show a clear difference in turtle assemblages between habitat types, they provide mixed evidence on the impact of commercial turtle harvest in these anthropogenically altered aquatic habitats. Our extrapolated estimates suggest that there are over two million red-eared sliders and 425,000 spiny softshell turtles occurring in the harvestable area of eastern Arkansas. This baseline population estimate is critical to management because these are the two most heavily harvested turtle species in Arkansas (Irwin 2007, 2020). However, our analysis of aquaculture farms indicates that harvest has effects on local populations of these species that may be detected years after the initial event, specifically reduced densities and shifts in community composition. These results agree with those of other studies that have detected body size differences and missing size cohorts in harvested populations of red-eared sliders (Brown et al. 2011, Close and Siegel 1997) and decreased density of spiny softshell turtles and red-eared sliders years after harvest (Brown et al. 2012). It is also important to note that the current level of harvest is only a fraction of what the state allows – in 2019, the state sold only 23 of the 150 commercial harvest permits allowed (Irwin 2020).

Our analysis of agricultural ditches suggests that substantial movement may mask effects of harvest in this system. The dynamic nature of these aquatic habitats, and the fact that harvest pressure is generally unknown, especially on private lands, makes it difficult to detect effects of harvest on local populations.

Our data provide a foundation for population and demographic modeling, which is an important tool for extrapolating these impacts and better informing management of Arkansas' freshwater turtle species. While our study has provided vital baseline information on the turtles and their populations in these habitats, future research is recommended. Specifically, further population estimates through the same or similar methods in larger waterbodies in order to estimate abundance in these habitats and compare densities and assemblages to aquaculture ponds and agricultural ditch systems. In addition, radiotelemetry within ditch systems is needed to better understand how different species are utilizing these habitats in relation to the larger, more permanent waterbodies they are connected to as well as to understand the movements of each species in response to drying events.

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