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Methods to Identify and Assess Suitability of Reintroduction Sites for the Alligator Snapping Turtle (Macrochelys temminckii)

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METHODS TO IDENTIFY AND ASSESS SUITABILITY OF REINTRODUCTION SITES FOR THE ALLIGATOR SNAPPING TURTLE (*MACROCHELYS TEMMINCKII*)

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In Partial Fulfillment

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Master of Science, Biology

By

Kameron Cheree Voves

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ABSTRACT

Reintroduction has been employed as a management strategy to combat population declines of the Alligator Snapping Turtle. The suitability of previous reintroduction sites was determined by researchers with years of experience in Alligator Snapping Turtle biology, following detection surveys and a visual habitat assessment. I developed methods that will facilitate the assessment of suitable release sites. First, I investigated the amount of effort needed to detect Alligator Snapping Turtles during surveys, then I compared methods to quantify submerged deadwood (a key habitat feature), and finally, I developed a standardized field survey and habitat suitability model that can be used to determine suitability of potential release sites. Following permutations, I determined that surveys for Alligator Snapping Turtles have a high risk of falsely concluding absence of small populations when survey effort is limited, and at least 100 net nights should be conducted to detect populations representing a catch-per-unit-effort of at least 0.02 turtles per net night. When quantifying deadwood density within long stretches rivers, sonar was found to be more useful compared to point-count methods because it provides the most exhaustive measure of deadwood. The accuracy of point counts was variable on a small scale, but the method may be viable following a transformation if sonar cannot be used. When using sonar, it may be beneficial to shorten processing time by sub-sampling sonar data. In which case, I recommend dividing sonar data into segments of 40 meters and counting pieces of deadwood within a random selection of segments comprising 40–60% of the river's length. Finally, the use of raw field data in a habitat suitability model was an effective way to determine suitability of rivers for Alligator Snapping Turtles. Sonar-based scores correctly predicted the presence of suitability habitat at sites with current populations of Alligator Snapping Turtles and deadwood density and human disturbance were the main drivers in determining score. The application of the methods I developed will aid in the identification of future reintroduction sites, contributing to continued recovery of this declining turtle.

KEYWORDS: *Macrochelys temminckii*, reintroduction, detection, side-scan sonar, remote-sensing, habitat suitability

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OVERVIEW

The Alligator Snapping Turtle (*Macrochelys temminckii*) has experienced population declines throughout its range, primarily due to historical overharvesting and habitat alterations such as channelization and the construction of impoundments (Reed et al. 2002, Pritchard 2006). This species is protected to some degree in all states where it occurs (Roman and Bowen 2000) and is currently being assessed for listing on the Endangered Species Act. In response to population declines, a captive breeding program was initiated at Tishomingo National Fish Hatchery in 2000 with the goal of restoring populations in rivers and other suitable wetlands where the species has been extirpated and to supplement small populations that are unlikely to persist without intervention. Reintroductions have taken place at sites in Oklahoma, Illinois, and Tennessee, and augmentation of existing populations has occurred in Louisiana. Even though head-started juveniles produced from the program have been released at numerous locations, standard methods to identify and evaluate potential release sites are not well established. The development of such a protocol could be a powerful tool for future reintroductions.

The first step prior to initiating reintroductions is to confirm the presence or absence of Alligator Snapping Turtles by trapping target sites with sufficient effort to confidently conclude that the species has been extirpated. After absence is confirmed, the habitat of each site needs to be evaluated to ensure the available habitat meets the needs of the Alligator Snapping Turtle. In the past, habitat assessments have been done by visual inspection by a researcher with years of experience conducting Alligator Snapping Turtle studies, but standardized methods to quantify habitat attributes would facilitate comparison among potential reintroduction sites. Because Alligator Snapping Turtles spend the majority of their lives underwater, abundant submerged

deadwood is essential (Sloan and Taylor 1987, Ewert et al. 2006, Riedle et al. 2006). Methods to measure density of submerged deadwood need to first be evaluated and compared to characterize such a critical habitat feature. Finally, after habitat attributes are directly measured in the field, a method needs to be developed to translate raw habitat measurements to a meaningful value representing site-specific habitat suitability for Alligator Snapping Turtles—and the accuracy of this method needs to be confirmed at sites currently supporting populations of Alligator Snapping Turtles. Finally, potential release sites can be compared and ranked to identify the most suitable habitats for future reintroduction.

In Chapter 1, I examined the amount of trapping effort conducted at survey sites in relationship to the likelihood of correctly concluding the absence of Alligator Snapping Turtles. In it, I report results of trapping surveys I conducted to detect Alligator Snapping Turtles at sites that could be suitable for reintroductions. Using permutations and capture data at sites where I detected Alligator Snapping Turtles, I determined the likelihood of falsely concluding absence of Alligator Snapping Turtles given different levels of sampling effort. With these permutations, I evaluated previous surveys to determine if sites were sufficiently sampled in order to correctly conclude absence of Alligator Snapping Turtles. Finally, I conducted additional permutations to determine the minimum level of effort needed to detect populations across a range of abundances.

Chapter 2 compares two methods to quantify submerged deadwood density. Here, I described a common field method used in turtle habitat studies in which the number of submerged logs is counted within a small radius around many sample points. I compared results of this field method to densities of logs seen on side-scan sonar images and determined whether the field method can reliably be used to assess submerged deadwood density within long

stretches of many rivers. I also discuss methods that can be applied to side-scan sonar data in order to shorten post-collection processing time while also accurately estimating deadwood density.

In Chapter 3, I present a habitat suitability model that can be used to determine habitat suitability of a potential reintroduction site following a standardized field survey. I conducted habitat surveys at 17 sites where Alligator Snapping Turtles were absent, present, or have been reintroduced, and calculate suitability scores using two methods to estimate submerged deadwood density. I determined which habitat features appeared to drive suitability scores and evaluated the accuracy of the models to predict suitable habitat at sites where alligator snapping turtles were present. Finally, I discussed the proper application of the model and ways the model could be used to prioritize reintroduction sites and aid recovery efforts of the Alligator Snapping Turtle.

DETECTING ALLIGATOR SNAPPING TURTLES: PROBABILITY OF CONCLUDING FALSE ABSENCE

Abstract

When surveying for a species with a potentially fragmented distribution of variably sized populations, such as the Alligator Snapping Turtle, it is important to use effective sampling methods. While numerous surveys have been conducted to assess the current distribution of Alligator Snapping Turtles, detection and likelihood of falsely concluding absence from limited trapping efforts is seldom addressed. I conducted trapping surveys for Alligator Snapping Turtles at four sites in Oklahoma where previous surveys had failed to detect the species. I then used a permutation analysis based upon real and theoretical trapping data to assess the robustness of previous surveys that have failed to detect Alligator Snapping Turtles and to determine a baseline level of effort needed to detect populations with different relative abundances. I detected Alligator Snapping Turtles at two of my four sample sites, highlighting the importance of increased sampling effort and multiple sampling bouts through time. I also found that some surveys conducted near the periphery of the geographic range of Alligator Snapping Turtles may have been insufficient to detect low density populations. Based upon the permutations conducted in this study, researchers can set a minimum level of effort needed in future studies to effectively detect populations above a predetermined density threshold.

Introduction

Rapid ecological assessments (REAs) were developed by The Nature Conservancy as a fast and flexible methodology to characterize flora and fauna communities that serve as baseline

data for use in conservation planning (Sayre et al. 2000). They were specifically designed to quickly determine biodiversity across large areas where little is scientifically known. Such assessments often result in the identification of areas of high conservation concern, establishment of protected areas and design of biological corridors, and the development of management plans, zonation, and threat abatement programs. REAs require careful planning and clearly defined field sampling protocols to achieve targeted conservation goals (Sayre et al. 2000).

The protocols that have been developed for REAs also can be applied to monitoring the status and distribution of species that are rare or otherwise difficult to detect. More broadly, the goals of REAs used to characterize biodiversity across large scales can be applied to characterizing biological trends in a single species. Determining the status of a population requires knowledge about density and demography (IUCN 2017), which often requires considerable effort to effectively assess just one potential locality. Assessing distribution, on the other hand, simply aims to determine absence or presence of a species across a geographic area. There are inevitable tradeoffs to consider when planning projects to assess a species' distribution. Arguably the most important consideration in study design is to collect data efficiently and in a manner that achieves the goals of the survey while also remaining within constraints (e.g., time, budget) imposed upon the project. This is commonly done by balancing the number of sites surveyed with the amount of effort expended at each site (Mackenzie and Royle 2005; Guillera-Arroita et al. 2010). Less effort invested per site usually allows investigators to survey a larger area; however, if the probability of detecting the targeted species is expected to be low, investing more effort at fewer sites (Moilanen 2002) and making multiple visits to sites (Bailey et al. 2007; Guillera-Arroita et al. 2010) may be necessary to confidently conclude presence versus absence.

The Alligator Snapping Turtle (*Macrochelys temminckii)* is a large, highly aquatic species that inhabits Gulf of Mexico river drainages from eastern Texas through the Florida panhandle and north into Kansas, Missouri, and Illinois. Surveys were initiated to establish the current distribution of the Alligator Snapping Turtle after a petition to list the species under the Endangered Species Act was denied in 1983 due to insufficient information regarding the species' status (Heck 1998). Declines in populations and overall range contractions were found to be more pronounced in the northern periphery of the range (Shipman et al. 1995; Riedle et al. 2005; Shipman and Riedle 2008; Bluett et al. 2011a; Lescher et al. 2013; Baxley et al. 2014) than in many southern states that represent the core of the species' range (Wagner et al. 1996; Jensen and Birkhead 2003; Folt and Godwin 2013; Huntzinger et al. 2019)

Despite the fact so many studies have been conducted to assess the distribution of Alligator Snapping Turtles, the probability of detecting individuals does not appear to be factored into the design of many surveys. Alligator Snapping Turtles are typically captured using baited hoop nets set near side channels, log jams, downed trees, or other locations with submerged cover—all preferred habitat features for the species. In Oklahoma and Louisiana, detection probabilities were found to be highest (a probability of nearly 0.40) when mean daily temperatures ranged 12–24 °C (Dreslik et al. 2017). However, the probabilities were low overall and confidence intervals surrounding parameter estimates were large and often bounded zero, indicating Alligator Snapping Turtles are potentially difficult to trap or, more likely, other meaningful covariates to estimate detection probabilities were not incorporated into models. Variables that could impact the detectability of Alligator Snapping Turtles—or turtles in general—include weather (Dreslik et al. 2017), season (Boundy and Kennedy 2006), type of trap (Bluett et al. 2011b), type and freshness of bait (Bluett et al. 2011b), and moving traps each night (Hollender 2019). Presence of logs

(Shipman and Riedle 2008), water temperature (Fitzgerald and Nelson 2011; Trauth et al. 2016), and proximity/saturation of traps (Boundy and Kennedy 2006) also may affect detectability but have not been empirically tested.

While detection is largely driven by environmental conditions and trapping methods, the amount of effort needed for detection also varies with abundance (Tanadini and Schmidt 2011; McCarthy et al. 2013), and researchers risk falsely concluding absence if insufficient effort is invested to detect small populations, even under ideal trapping conditions. Considerations for optimal survey protocols to detect Alligator Snapping Turtles and assessing the probability of falsely concluding absence *a priori* has seldom been addressed in previous studies. Assuming all of the aforementioned variables are controlled for and trapping is executed competently, then the consequences of survey effort can be considered. As such, a baseline amount of effort to detect populations of varying relative abundances can be determined.

During this study, I trapped five previously surveyed rivers (Riedle 2001) in eastern Oklahoma with the intent of verifying the reported absence of Alligator Snapping Turtles at each site. I then used the results to evaluate the probability of falsely concluding absence of Alligator Snapping Turtles during surveys; I used a permutation analysis to assess whether sufficient trapping effort was expended during previous surveys for Alligator Snapping Turtles in which the species was concluded to be absent. Finally, permutations were also conducted using theoretical data to determine the minimum amount of effort required to minimize false detection rate of Alligator Snapping Turtles given different relative abundances and assuming ideal trapping conditions.

Methods

Turtle surveys. Five rivers were surveyed in eastern Oklahoma to detect Alligator Snapping Turtles: the Neosho River and its tributaries Big Cabin Creek and Chouteau Creek, the Deep Fork River above Lake Eufaula, and the Poteau River near the city of Arkoma (Fig. 1). Alligator Snapping Turtles historically occurred at all study sites and were not detected during surveys conducted in the late 1990s (Riedle et al. 2005). All survey sites were located within the Arkansas River drainage and were upstream of reservoirs or were tributaries of dammed rivers. Sites were selected because Alligator Snapping Turtles were presumed to be absent, yet habitat appeared to meet requirements of the Alligator Snapping Turtle, making these sites potential candidates for future reintroduction efforts.

Each river segment was trapped in June–July 2018 for seven consecutive days, and a minimum of 15 traps were set per day, resulting in 100–120 net nights (NN) per site. One net night was equivalent to a single trap set overnight. I used single-throat 0.9-m diameter and double-throat 1.2-m diameter hoop nets with 2.5-cm mesh. Traps were set and georeferenced in the afternoon and checked the following morning. Traps were moved to a new location each night; each survey area was divided into an upper portion and lower portion, and traps were moved from one section to the other each night. Each trap was rebaited daily with frozen or freshly caught fish. Traps were set such that a portion of the net remained above water to allow turtles to breath. If inclement weather was forecast, either trapping was postponed or flotation devices were placed in traps to keep a portion of the net above the water as a safeguard against rising water. The segment of river that was sampled at each site differed in length, ranging from 6 to 13 km, and depended on access to boat ramps, navigability, and the amount of public use (corresponding to likelihood of nets being tampered with).

For all species of turtle captured, I recorded sex, mass, shell measurements, and shell scarring. Turtles were given a common scute notch to differentiate new captures from recaptures, and each turtle was only measured at the time of first capture. Softshell turtles received a small notch in the posterior edge of the carapace (Plummer 2008). Captured Alligator Snapping Turtles were given a common notch as a means of quickly identifying recaptured individuals, but also were injected with a Passive Integrate Transponder (PIT) tag in the right rear leg to ensure permanent individual identification. Prior approval for this project was obtained from the Missouri State University Institutional Animal Care and Use Committee (Approved: 04/2018; IACUC ID: 17-028.0) following proper Collaborative Institutional Training Initiative research compliance trainings (Appendix A).

Permutation analyses. I conducted a permutation analysis to estimate the probability of obtaining a false negative for a given an amount of survey effort. This was done by randomly re-sampling capture data for a given number of NN, and the proportion of 10,000 iterations that resulted in zero captures of Alligator Snapping Turtles—leading to a conclusion of false absence—was calculated. All analyses were implemented in R version 3.5.0 “Joy in Playing” (R Core Team 2018).

Permutations were conducted using trapping data that I obtained at sites where Alligator Snapping Turtles were captured despite prior suggestion they had been extirpated. Permutations using field data included the true number of captures per trap. I used this approach to calculate the probability of a false negative for a range of NN and for effort reported in previous survey efforts at sites where Alligator Snapping Turtles were not detected (Shipman et al. 1995; Riedle et al. 2009; Bluett et al. 2011a; Baxley et al. 2014). This application allowed me to evaluate the likelihood that the absences of Alligator Snapping Turtles found during previous survey efforts

were false negatives. The results of these permutations were interpreted as the probability of a false negative assuming surveyed sites supported populations that were comparable to those I surveyed in Oklahoma.

I also used a permutation analysis to investigate the minimum effort needed to detect at least one Alligator Snapping Turtle under scenarios with varying relative abundance. When Alligator Snapping Turtles occur in greater abundance, more individuals are expected to be captured assuming trapping effort remains constant, resulting in a higher detection-per-unit-effort (DPUE), which I define as the number of traps that captured at least one Alligator Snapping Turtle divided by the total number of traps set. I simulated trapping results for 51 theoretical Alligator Snapping Turtle surveys, each with a different DPUE. Simulated surveys consisted of 200 NN, and traps that failed to capture an Alligator Snapping Turtle were assigned a 0; traps that captured at least one Alligator Snapping Turtle were assigned a 1. The smallest simulated DPUE was 0.005 (1 detection in 200 NN); the remaining simulated surveys had DPUEs that ranged from 0.010 (2 detections) to 0.500 (100 detections), increasing by 0.010 (2 detections) with each new simulation. Permutation analyses were conducted for each of the 51 simulated surveys using effort ranging from 1 to 200 NN, and the minimum number of NN needed where the probability of a false negative was less than 5% was determined—which corresponds to a 95% confidence of detection assuming populations were equal to a density resulting in each given DPUE. Researchers can use the results of these permutations to set a minimum level of effort needed in future studies to effectively detect populations above a predetermined density threshold.

Results

Turtle surveys. I sampled turtles at five sites in Oklahoma over 475 NN, which yielded a total of 20 individual Alligator Snapping Turtles at two of the five locations—Big Cabin Creek and the Poteau River (Appendix B). The capture rate in the Poteau River (0.130 CPUE) was much higher than in Big Cabin Creek (0.024 CPUE). Of the 17 individuals captured in the Poteau River in 127 NN, 14 were juveniles, 2 were females, and 1 was male. In Big Cabin Creek, 2 juveniles and 1 female were captured in 124 NN. The resulting adult to juvenile ratios were 1:4.67 in the Poteau River and 1:2 in Big Cabin Creek. In both rivers combined, juveniles ranged in carapace length from 164 to 331 mm, and the adults measured 411–457 mm.

Permutation analyses. Trapping data for the Poteau River (18 captures in 127 NN—including 1 recapture) and Big Cabin Creek (3 captures in 124 NN) were used in permutation analyses to estimate probabilities of a false negative at different levels of effort (Fig. 2). The curve for Poteau River declined much more quickly than did Big Cabin Creek—the probability of a false negative fell below 5% with an effort of 19 NN in the Poteau River compared to 78 NN in Big Cabin Creek. Alligator Snapping Turtles were not captured at either of these sites when surveyed in 1997 by Riedle (2001). Assuming detectability during the previous survey attempts were comparable to that in this study, in 23 NN on the Poteau River and 25 NN at Big Cabin Creek, Riedle (2001) had a 2% and 51% chance, respectively, of falsely concluding absence of Alligator Snapping Turtles.

When effort from nine additional surveys in Oklahoma (Riedle et al. 2009) was compared to permutation analyses using Big Cabin Creek trapping results, I identified one site where the probability of a false negative was greater than 0.50, three sites that had a probability greater than 0.20, and two sites had a probability less than 0.05 (Fig. 3). When results were applied to permutations based on the Poteau River, only one site of nine had a probability of greater than

0.05. Considering survey efforts outside of Oklahoma, surveys in Kansas (Shipman et al. 1995) had the greatest likelihood of having concluded false absences, with 84% of 19 sampled sites resulting in a probability of a false negative greater than 0.75 if abundance were similar to that in Big Cabin Creek. In Illinois (Bluett et al. 2011a), 76% of 17 sites surveyed had probabilities above 0.50, and in Kentucky (Baxley et al. 2014), 38% of 24 sites surveyed had probabilities above 0.50. When data from these surveys were run using permutations based upon Poteau River (greater abundance of turtles), the proportion of sites with probabilities of false negatives greater than 0.5 decline to 37% in Kansas, 6% in Illinois, and 0% in Kentucky.

When abundance of Alligator Snapping Turtles in theoretical surveys was high (DPUE ≥ 0.26), fewer than 10 NN were needed before the probability of a false negative was less than 5% (Fig. 4). When DPUE was greater than 0.44, only 5 NN were needed to detect Alligator Snapping Turtles in at least one net. More than 50 NN were needed to detect a population resulting in a DPUE of 0.05 or less, with 190 NN required to detect presence when DPUE was 0.005.

Discussion

Overall, the detection of Alligator Snapping Turtles in rivers from which they have been declared extirpated highlights the importance of increased sampling effort for this and other species that exhibit low detectability, whether due to difficulty trapping or genuinely low population densities. Trapping was conducted by Riedle (2001) using methods that were very similar to those used in my study. Given the high chance Riedle (2001) had of falsely concluding absence of Alligator Snapping Turtles in Big Cabin Creek, it was likely that too few traps were set to detect the small population present at that site. The very small probability of a false

negative in the Poteau River, on the other hand, indicates there may have been other factors that reduced the likelihood of detecting Alligator Snapping Turtles. For example, the previous survey effort on the Poteau River was conducted in early August when water temperatures might have been high enough to suppress activity—as has been observed in Common Snapping Turtles (*Chelydra serpentina*) (Hollender in prep.)—or turtles moved into deeper, cooler water (Riedle et al. 2006). In either case, the likelihood of a turtle to enter a trap could have been lower than it was during the period that I conducted population surveys.

Alternatively, while 27 NN is statistically enough effort to detect an Alligator Snapping Turtle in a fairly dense population, it is possible the population in the Poteau River was smaller in 1997 than it presently is. Alligator Snapping Turtles reportedly reach sexual maturity after 11 to 13 years and with a minimum carapace length of 330 mm (Dobie 1971). As such, the majority of individuals captured in my study may have been juveniles that represent recruitment into the population after sampling took place in 1997. Additionally, Alligator Snapping Turtles are capable of moving long distances within a river network (Sloan and Taylor 1987; Harrel et al. 1996; Trauth et al. 1998) and could have moved upstream from the nearby Arkansas River in the last 23 years. The Arkansas River is dammed approximately 43 km and 17 km upstream from its confluence with the Poteau River by two locks and dams. The abundance of Alligator Snapping Turtles below these dams between their construction in 1970 and present day is unknown, and I therefore cannot confidently speculate on the chance of Alligator Snapping Turtles having recently colonized the Poteau River. However, previous studies have reported lower capture rates or complete failure to detect Alligator Snapping Turtles in large rivers compared to tributaries (Jensen and Birkhead 2003; Huntzinger et al. 2019), especially those with impoundments. Frequent releases of water from a dam results in unstable flow conditions and may reduce the

quality of the habitat (e.g., fewer submerged logs). These conditions may have stimulated emigration of Alligator Snapping Turtles out of the main channel of the Arkansas River and into lower-order streams that exhibit better hydrological stability.

Permutations proved useful in evaluating results of previous efforts to detect Alligator Snapping Turtles. Catch-per-unit-effort is often used as a proxy for population size in Alligator Snapping Turtle studies because so few surveys have been sufficient to generate robust population estimates. It is unlikely that Kansas, Illinois, and Kentucky support large populations of Alligator Snapping Turtles, but according to my permutations, not enough effort was invested at the majority of surveyed sites to confidently infer complete absence of the species. Instead, interpretation of surveys resulting in non-detection of Alligator Snapping Turtles should infer the potential presence of a population representing a certain DPUE. For instance, the average effort expended during surveys conducted in Kansas, Illinois, and Kentucky was 24 NN. Based upon my permutation analysis, at this level of effort, I can be 95% confident that detectability of Alligator Snapping Turtles would be less than 0.12 Alligator Snapping Turtles per NN. The maximum amount of effort expended during these surveys was 96 NN, for which I can be 95% confident that the detectability would be less than 0.03 Alligator Snapping Turtles per NN. Using previous surveys as a guideline, Baxley et al. (2014) determined that the level of effort expended at sites in Kentucky represents potential populations resulting from a CPUE of 0.06 at most. The average number of NN conducted during Baxley’s surveys was 34.5 NN per site, which corresponds to 95% confidence of detectability less than a DPUE of 0.07, according to my analysis. Future interpretation of surveys that result in zero Alligator Snapping Turtle captures should endeavor to make similar comparisons.

Based on the reported trapping results in Riedle et al. (2009) in rivers not included in my surveys, it seems that enough effort was previously invested in rivers in Oklahoma to confidently conclude absence of Alligator Snapping Turtles, but these results are misleading. Upon further inspection, results reported for some rivers are actually several sites pooled together (individual site data can be found in Riedle 2001). For example, 109 NN are reported for the Neosho River in Riedle et al. (2009), but only 30 NN are reported in the main stem of the river in Riedle (2001) while the remaining 79 NN were from smaller tributaries of the Neosho. Rivers and their tributaries are open systems and turtles could be moving among survey areas. Given this, pooling trapping effort in such a way reflects surveys of a single contiguous population. However, presenting the data this way makes further interpretation of effort and capture rates difficult, because it represents an average CPUE across a large area, whereas each survey site may have differed in habitat quality and therefore abundance of Alligator Snapping Turtles. When evaluating the probability of false absences in previous surveys, it was therefore important to consider how the data were presented. Other studies that have conducted distribution surveys for Alligator Snapping Turtles report absences with a varying degree of effort—and therefore varying probabilities of false negatives (Folt and Godwin 2013; Huntzinger et al. 2019)—pooled data making interpretation of false absence difficult (Wagner et al. 1996; Jensen and Birkhead 2003), or did not report amount of effort invested per site at all (Boundy and Kennedy 2006; Shipman and Riedle 2008).

It is important to note that the application of permutations in this study represent captures under ideal trapping conditions (hypothetical data) or under conditions in which my field data were collected (Big Cabin Creek and Poteau River). My method does not take into account variable detectability based upon environmental factors and should only be used as a baseline for

survey methodological design. However, considering the effect of detectability during study design could be used to correct for environmental factors that might reduce detectability. For instance, detectability of Alligator Snapping Turtles may be reduced during surveys when it is very hot or very cold. Sampling effort should therefore be increased past the baseline determined by my permutations in order to achieve comparable results if trapping were conducted during more moderate times of the year. Few studies have been conducted to investigate the effect of environmental variables and trapping methods on capture rates of Alligator Snapping Turtle specifically, and these variables should be included in occupancy models similar to Dreslik et al. (2017) to more accurately estimate the detection probability in this species.

In light of the results presented, future surveys to detect Alligator Snapping Turtles—or any elusive or rare species—should take care to consider optimal survey design. Overall, the small amount of effort needed to detect a large density of Alligator Snapping Turtles indicates that the species is relatively easy to detect. However, robust study design may be even more important in parts of the species' range where it has experienced substantial range contractions and population sizes are more variable. To detect small populations, I recommend trapping for at least 100 NN, which corresponds to a probability of a false negative below 0.05 given an expected CPUE of 0.02 or greater. The results of this study will provide a baseline for researchers to determine optimal sampling design during surveys for Alligator Snapping Turtles, but future studies should endeavor to further investigate the effects of environmental variables, abundance, and trapping effort in relationship to the detection of Alligator Snapping Turtles.

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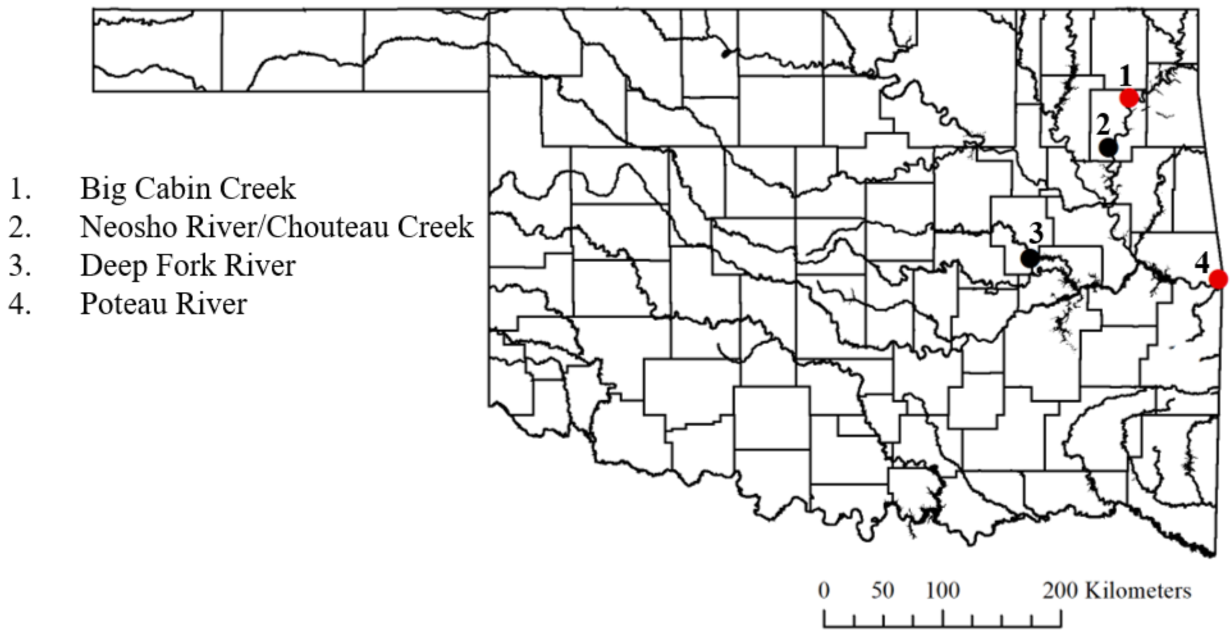

Fig. 1. Sampled rivers in Eastern Oklahoma, summer 2018. Red points indicate rivers where Alligator Snapping Turtles were captured.

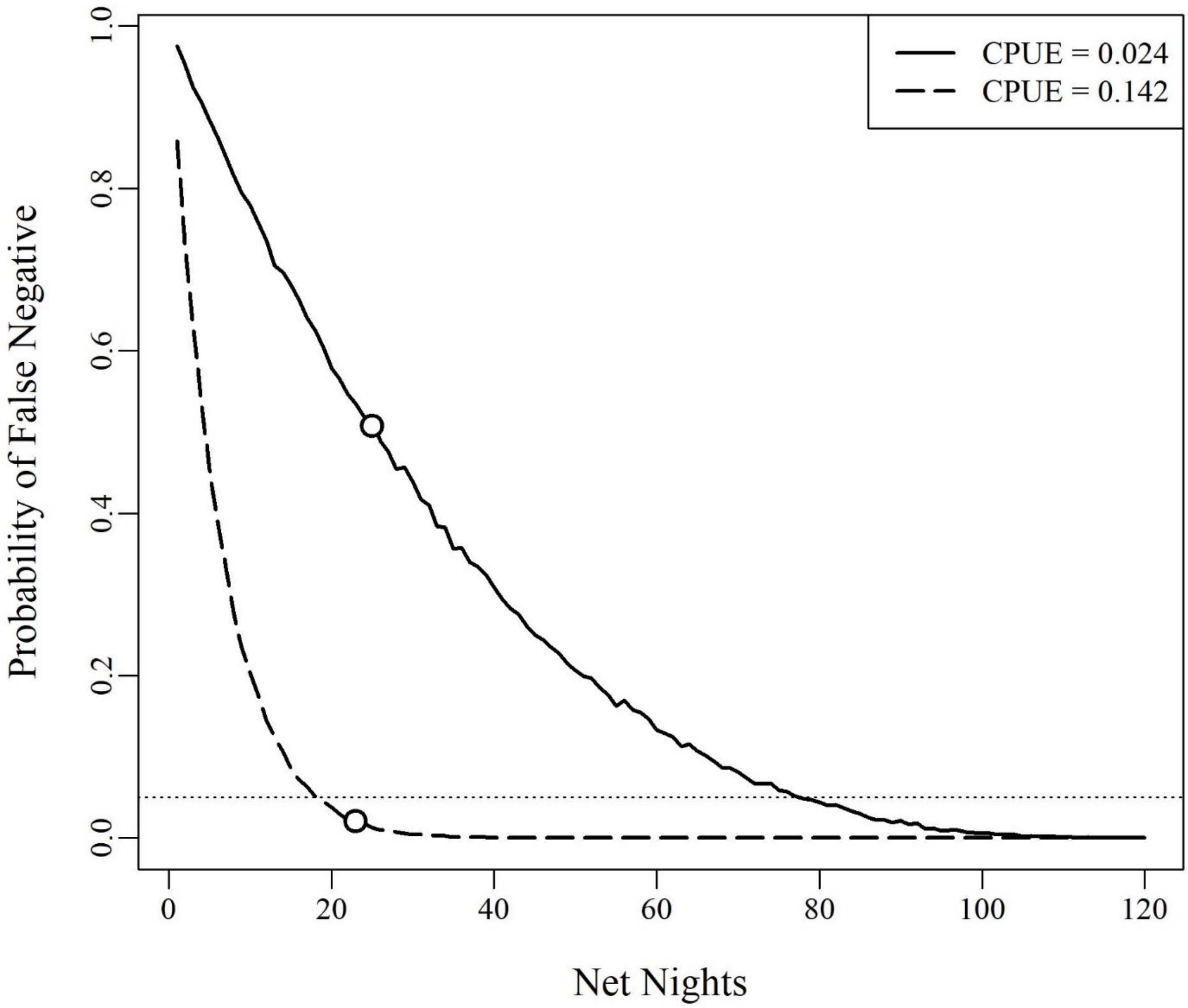

Figure 2. Relationship between trapping effort, in net nights, and probability of incorrectly concluding that *Macrochelys temminckii* is absent from a system. Curves are based upon permutations generated from populations in Big Cabin Creek (CPUE=0.024) and Poteau River (CPUE=0.142—includes 1 recapture). Points indicate the amount of effort expended at the same sites during a previous survey effort (Riedle 2001) and the generated probability of a false negative (0.51 at Big Cabin Creek and 0.02 at Poteau River). The dotted line denotes a probability of 0.05.

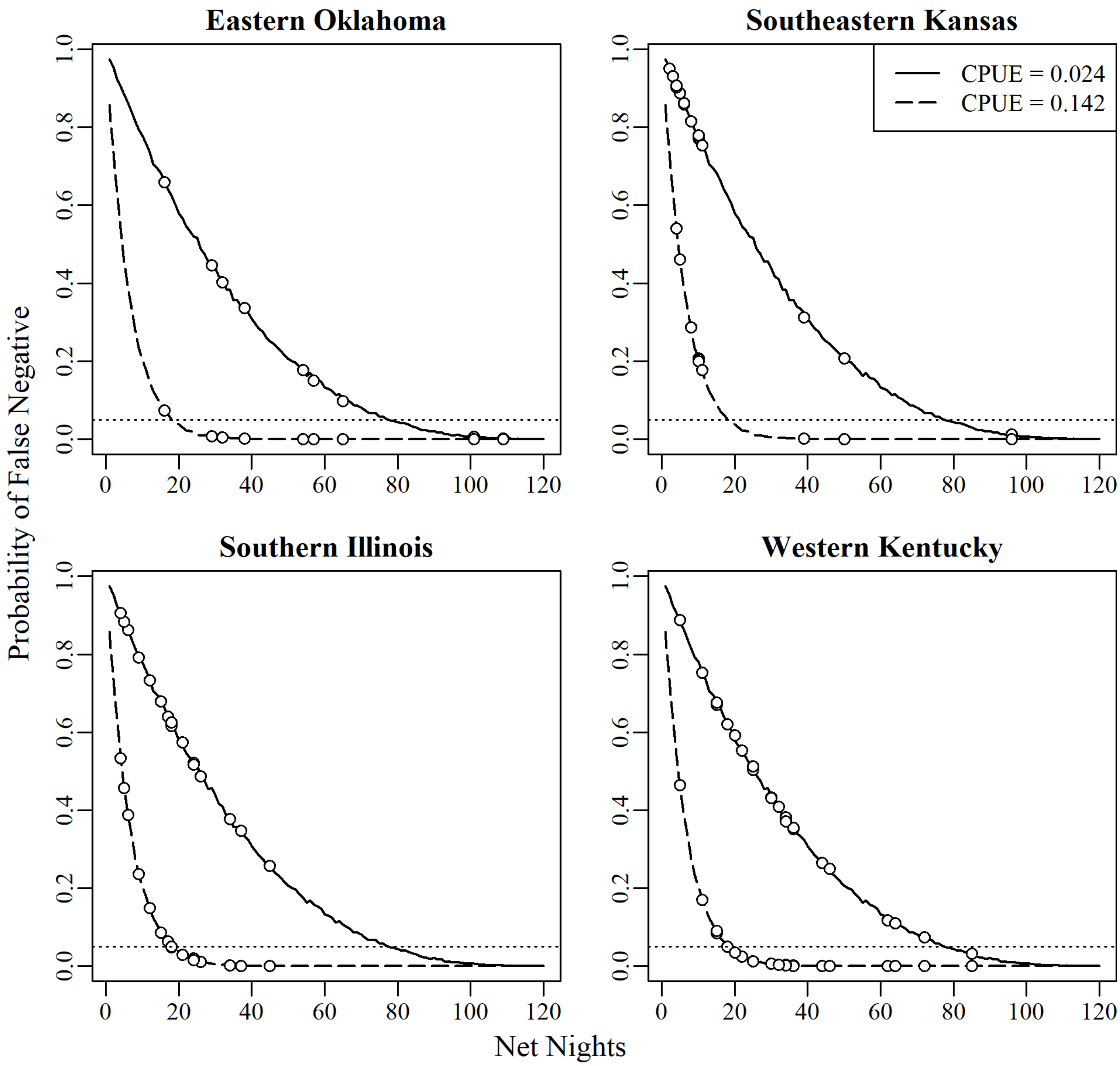

Figure 3. Probabilities of false negatives given reported effort expended at sites during previous surveys where Alligator Snapping Turtles were not detected. Curves are based upon permutations generated from populations in Big Cabin Creek (CPUE = 0.024) and Poteau River (CPUE = 0.142—includes 1 recapture). Each point corresponds to effort invested at one site. The dotted line denotes a probability of 0.05. Data included in the analysis are: Riedle et al. (2005) [Oklahoma], Shipman et al. (1995) [Kansas], Bluett et al. (2011a) [Illinois], and Baxley et al. (2014) [Kentucky].

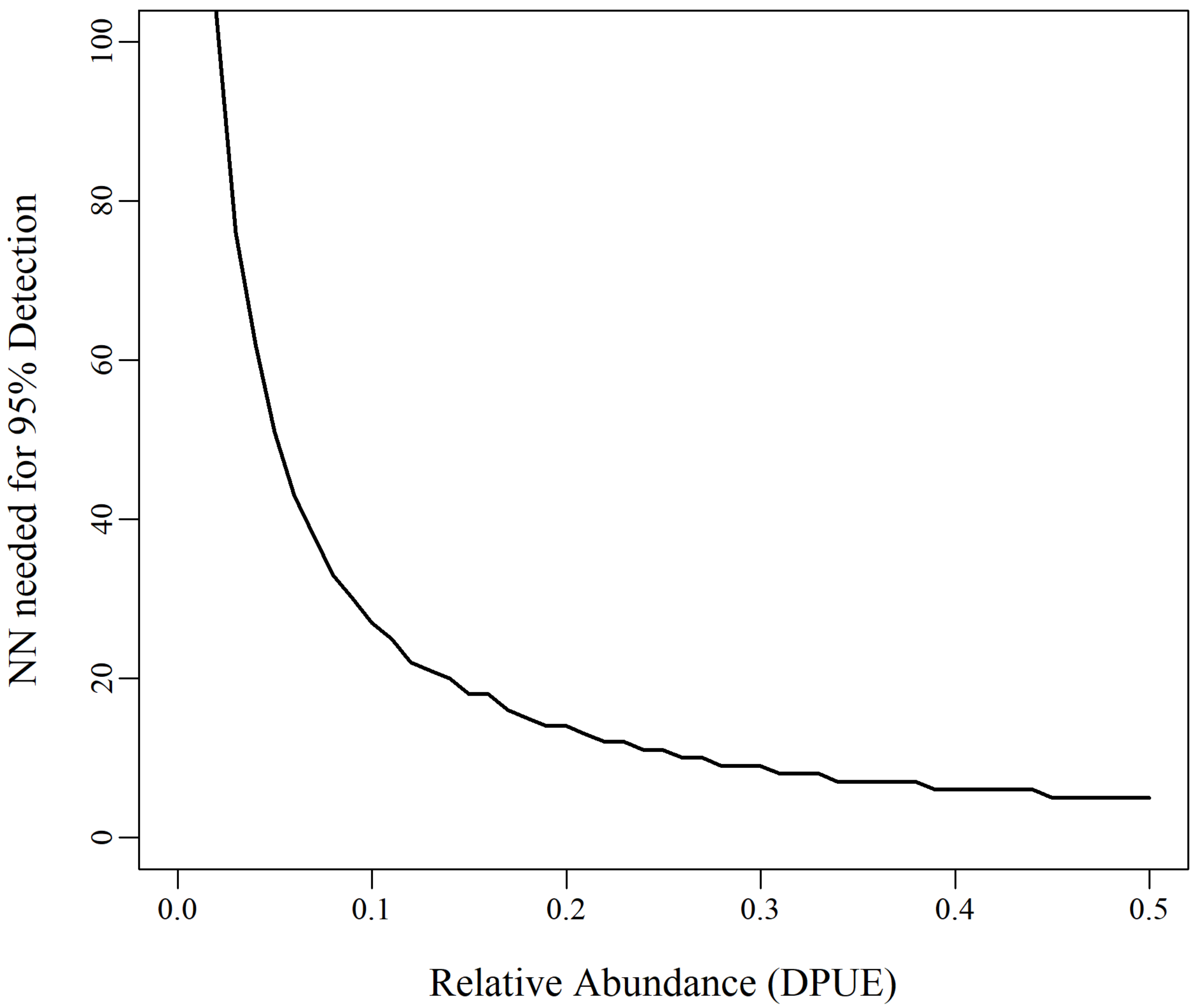

Figure 4. The minimum number of net nights needed with zero captures to conclude absence of *Macrochelys temminckii* with 95% confidence given different relative abundances. Abundance was estimated with increasing detections-per-unit-effort (DPUE) in theoretical capture data consisting of 200 net nights. Permutations were performed under these different DPUE scenarios to estimate minimum effort requirements.

THE APPLICATION OF SIDE-SCAN SONAR TO QUANTIFY UNDERWATER TURTLE HABITAT: A COMPARISON TO A STANDARD FIELD METHOD

Abstract

Habitat preferences of freshwater turtle species are often determined by comparing habitat use to available habitat. The density of submergent woody debris is a commonly measured variable in such studies and often is quantified within a small radius (typically < 25 m) around capture locations. However, methods to determine submerged deadwood density within a large spatial extent—such as across many kilometers of river—are not well established. The development of such a method is important to aid in identification and conservation of critical habitat for declining turtle species. In this study, I explored the application of side-scan sonar to quantify submerged deadwood density on a large scale and among many rivers. First, I compared the standard point-count method used in the field to methods utilizing side-scan sonar technology. Point counts were compared to three scales of sonar-generated deadwood densities: deadwood density within the same area as point counts, deadwood density along the banks of the entire surveyed area, and total deadwood density. At the smallest scale, the accuracy of point counts compared to sonar varied among rivers, but point counts tended to document a greater number of logs than seen on sonar in the same area. Point-count deadwood density was inconsistently related to sonar bank-side deadwood density. However, results indicated that the relationship between point-count density and sonar total density was consistent enough that point counts may be used to predict deadwood density within the entire sample reach following transformation. Second, I examined methods to sub-sample sonar data to decrease post-collection processing time. I determined that as the total length of the sample area increased, so did the amount of

processing effort needed to accurately estimate deadwood density. Additionally, the amount of required effort was determined by a proportion of the sample area rather than a fixed length. Finally, it was better to sub-sample rivers that had been divided into many small segments than few large segments. An improved understanding about how the scale at which habitat measurements are made compared to the scale of interest will aid researchers in collecting more meaningful field data.

Introduction

Emergent woody debris provides critical aerial basking structure for many species of freshwater turtle (Cagle 1950, Chessman 1987, Fuselier and Edds 1994, Lindeman 1999, Lindeman 2000). However, the presence of deadwood in lotic habitats provides other benefits related to the quality of turtles' underwater habitat (Fig.1). With its complex surface structure, submerged deadwood offers an excellent surface for growth of a range of bacteria, fungi, and algae (Davies and Storey 1998, Treadwell et al. 2007). Some turtles will forage directly on the resulting biofilm (Shively and Jackson 1985), but more often these biofilms support communities of organisms that are preyed upon by turtles—making submerged logs centers for turtles to forage for algae, invertebrates, fish, and other organic material such as detritus. As has been suggested for fish, the spatial complexity of submerged deadwood may also provide cover from predators, act to visually isolate turtles and reduce individual contact, and provide refuge from high flow rates, thereby reducing energetic costs (Crook and Robertson 1999). Underwater debris may also aid in navigation by serving as spatial reference points (Crook and Robertson 1999). Extending beyond submerged deadwood as provision of microhabitat to turtles, the presence of large woody debris in rivers influences geomorphic functions such as increasing the

hydraulic roughness of the river bed which provides an environment of varied flow patterns, increasing the retention of organic matter and sediment, and creating pools, bars, and islands—all which increase habitat heterogeneity at larger scales (Gurnell et al. 1995, Dudley et al. 1998, Treadwell et a. 2007).

Because deadwood plays such a central role in turtle ecology, its abundance and other characteristics are often measured to understand species' habitat associations and preferences. One species which the quantification of submerged structure is especially important is the bottom-dwelling Alligator Snapping Turtle (*Macrochelys temminckii*; Riedle et al. 2006). Studies investigating Alligator Snapping Turtle habitat preferences have often measured deadwood density within a small radius around capture locations and then made comparisons to deadwood density at random locations (Riedle et al. 2006, Bass 2007, Shipman and Riedle 2008, Riedle et al. 2009, Lescher et al. 2013, Riedle et al. 2016, Townsend 2016). As turtle populations decline (Lovich et al. 2018)—often due to habitat alteration (Ernst and Lovich 2009)—attention to large-scale habitat characterization and impacts of landscape-use on turtle populations has grown (Bodie 2001, Marchand and Litvaitis 2004, Rizkalla and Swihart 2006, Carrière and Blouin-Demers 2010. Sterrett et al. 2011, Limbaugh 2012, see also Chapter 3). Reintroduction has been employed as a management strategy to combat local extirpation of the Alligator Snapping Turtle populations in Oklahoma (Anthony et al. 2015, Dreslik et al. 2017). Assessing habitat quality at a large scale is advised prior to releasing head-started turtles (IUCN 2013), but methods to quantify deadwood density—an essential habitat feature for the Alligator Snapping Turtles (Riedle et al. 2006)—within a large extent are not well established in the literature.

Recreation grade side-scan sonar is a low cost, flexible, and time-efficient method to quantify instream habitat (Kaeser and Litts 2010) and has been used to survey habitat used by

fish (Cheek et al. 2015, Richter et al. 2016, Walker and Alford 2016), freshwater mussels (Garner et al. 2016, Smit and Kaeser 2016), as well as turtles (Thomas 2013, Sterrett et al. 2015, Jenkins and Godwin *public communication*). Few examples of the use of side-scan sonar to quantify submerged deadwood density in rivers exist, but the method is becoming more widely implemented (Kaeser and Litts 2008, Kaeser and Litts 2010, Kitchingman et al. 2013, Koeller 2014). Ground-truthing surveys have been conducted to assess the accuracy of the interpretation of sonar images at small scales (460–700 m; Kaeser and Litts 2008), but it is yet unknown how this accuracy may differ among many rivers with variable habitat conditions (e.g., water depth and clarity). Additionally, it is unclear how deadwood density estimated using side-scan sonar over long stretches of river (greater than 2 km) compares to estimations made using more traditional field methods.

The first objective of my study was to use side-scan sonar to characterize total deadwood density in multiple rivers and compare the results to those obtained by manually quantifying deadwood abundance in the field. Methods were designed using Alligator Snapping Turtles as a model organism with the intention that methods tested in this study be used to assess habitat quality of future reintroduction sites. My goal was to first determine how visually counting the number of logs around sample points compares to logs seen on sonar images in the same area. Then, I investigated whether this point-count method could reliably be used to estimate whole-river deadwood density compared to sonar.

The use of side-scan sonar provides a thorough quantification of deadwood density that is more time efficient than a full deadwood census in the field (2.5 man-hrs/km vs. 29 man-hrs/km) (Kaeser and Litts 2008). However, compiling and interpreting the sonar data after collection takes a considerable amount of time when the method is applied to long stretches of many rivers.

For example, post-collection time expenditure was estimated to be 138 min/km (Kaeser and Litts 2008). Using this estimation, the time required to processes the sonar data collected in my study was equivalent to eight 40-hour work weeks. Consequently, the second objective of my study was to investigate methods to sub-sample sonar data to shorten processing time while still accurately estimating deadwood density. To assess habitat suitability at potential reintroduction sites for Alligator Snapping Turtles, my goal was to determine the sonar sub-sample size required to estimate deadwood density within 15% of the true density using different ways to divide the river into segments. An error of 15% was chosen to be large enough to noticeably reduce sonar processing time but not so large as to vastly deviate from the true deadwood density. Additionally, I wanted to know how the length of the surveyed reach and variation in deadwood density along the surveyed reach impacted the required sub-sample size.

Methods

Data collection and processing. Deadwood density data (point counts and sonar surveys) were collected on 19 rivers in Kansas, Oklahoma, and Mississippi in 2018 and 2019 (Table 1). The length of river included in the sample reach depended on previous survey work conducted on the rivers, access to boat ramps, and navigability. Point counts were conducted in the field on rivers surveyed in 2019. For this procedure, deadwood was quantified at 25 sampling points along each bank of the river, for a total of 50 sample points at each study site. Paired sample points were located directly across the channel from one another and were equally spaced along the surveyed stretch of river. The number of submerged logs that could be visually detected was determined 20 m downstream and upstream of the sample point and within 10 m of the bank (Fig. 2A). In this study, a submerged log was included in point counts if it was large

enough to serve as cover for a juvenile Alligator Snapping Turtle—generally a structure with a diameter ≥ 10 cm and with a portion of the structure resting on the river bottom. Entire downed trees were counted as one log. Other types of structure defined as a log included downed limbs and vertical stumps—floating structure was not included in my counts.

Side-scan sonar surveys were conducted on all rivers using a Humminbird Helix 10 CHIRP Mega Side Imaging System (Johnson Outdoors, Inc., Racine, Wisconsin, USA) (Appendix C). The sonar transducer captured images from the bow of a 4.6-m flat-bottomed boat traveling 4–6 km/hr. Sonar was recorded with sensitivity and contrast settings of 10 and at a frequency of 1200 kHz. Multiple passes of the river were made to cover the full channel width—each pass covered a range of 20–35 meters on either side of the boat. Third-party analysis software was used to process the sonar data, and sonar mosaics were created from several sonar recordings of each river (Reef Master 2.0; ReefMaster Software, Ltd., West Sussex, UK). Two types of deadwood density (logs/km) were calculated with sonar data. Total deadwood density (TDD) was determined by counting all submerged structures seen on sonar along the entire length of sampled river (Fig. 2C)—dropping a waypoint at each log location. Bank-side deadwood density (BDD) only included deadwood within 10 m of the bank, but still along the whole survey reach (Fig. 2D). Deadwood densities calculated with sonar were considered to be the "true deadwood density," as this method is assumed to be unaffected by turbidity or depth. I reviewed all sonar mosaics to reduce observer-related bias in sonar interpretation (Kaeser and Litts 2008).

Data analysis. Deadwood density method comparison. To assess the accuracy of point counts, sample point locations were superimposed onto the sonar mosaic and the number of logs on the sonar were counted within the same sample area as counted during point counts (Fig. 2E).

Rivers were only included in this analysis if the point count and sonar surveys were conducted within 48 hours of each other. Shapiro-Wilk tests were conducted to assess normality, and either Pearson correlations (for normal data) or Spearman's rank correlations (for non-normal data) were used to determine the correlation between logs counted during point counts versus logs counted on sonar images. If logs counted during point counts were significantly correlated with logs on sonar, a regression analysis was used to determine if the ratio between the two methods was different than a 1:1 relationship.

Deadwood density (logs/km) was estimated from point-count data by first totaling the number of logs counted for each pair of sampling points. The average number of logs per 40 meters was calculated for the 25 pairs of sample points and scaled to number of logs per kilometer (Fig. 2B). Regression analysis was used to investigate the relationship between deadwood density estimated with point counts and sonar TDD. Regression analysis was also used to compare point-count-estimated deadwood density and sonar BDD. Data were natural log-transformed and confidence and prediction intervals were calculated. The resulting regressions were used to transform point-count-estimated deadwood densities to assess whether transformation could be used to increase accuracy of point-count estimations. Deadwood density measurements for Pond Creek and the Caney River were excluded from these analyses due to high water levels during point count surveys that may have affected point-count estimations. Measurements from the Old River Channel were also excluded because much of the structure at this site consisted of large piles of brush resulting from severe flooding, and discerning discrete pieces of deadwood in these conditions was difficult.

To further understand the utility of using point counts to estimate TDD, sonar mosaics were divided into 40-m segments and 25 randomly selected segments were used to estimate

deadwood density (matching the point-count method). Deadwood density was estimated first by counting all of the logs per segment (TDD) and then estimated by counting only the logs within 10 m of the bank in each segment (BDD). Boxplot distributions of 500 random samples per counting method were constructed, and deadwood densities estimated using point counts were compared to the generated distributions.

Sonar sub-sampling. To quantify the amount of processing effort needed to estimate TDD within a sample reach, river sonar data were sub-sampled in a number of ways. For example, using a mid-line track of a river, the sample reach was divided into 10 segments of equal length (Fig. 3A). The number of logs within each segment was counted (Fig. 3B). Two segments were randomly selected and the number of logs per kilometer was calculated from this random selection. Fifty samples were randomly selected, and deadwood density was calculated each time. Then, the same steps were repeated but after selecting three segments, then four segments, and so on (Fig. 3C). The proportion of estimated deadwood density to true deadwood density was calculated for each iteration (Fig. 3D). The minimum number of segments needed to estimate true deadwood density was determined by finding the number of segments counted in which 95% of the iterations were within 15% of the true density. The minimum number of segments needed in a sub-sample was multiplied by the length of each segment to obtain the minimum number of kilometers required, which was then divided by the total length of the sample reach to obtain the minimum proportion required. This procedure was repeated for 17 rivers and with 10, 15, 20, 25, 50, 75, and 100 segments to determine if the number of segments into which the river was divided changed the amount of processing effort that was required. Additionally, I divided each river into segments of the same length (500 m, 250 m, 150 m, 100 m, and 40 m) and repeated the above steps, again with 50 iterations.

Regressions were used to determine how the required number of kilometers counted and the required proportion of river counted changed with the total length of the sampled reach and the standard deviation of the number of logs per segment. Confidence intervals were used to determine if there was significant difference between the slopes and/or y-intercepts of the regression lines. All analyses were implemented in R version 3.5.0 (R Core Team 2018). The R packages *sp* (Pebesma and Bivand 2005, Bivand et al. 2013), *rgeos* (Bivand and Rundel 2019), *GISTools* (Brunsdon and Chen 2014), *sf* (Pebesma 2018), and *smoothr* (Strimas-Mackey 2020) were used to implement the sonar sub-sampling procedure (Appendix D).

Results

Deadwood density method comparison. Of the eight rivers used to determine the accuracy of point counts, seven of them resulted in a significant correlation between the number of logs counted around sample points during point count surveys and the number of logs counted in the same area on sonar images (Fig. 4; also Fig. 2A compared to Fig. 2E). Pond Creek was the only river in which the correlation was not significant ($r = 0.2242$, $P = 0.1175$). Significant correlations were highly variable ($r = 0.3715$–0.7796). The relationship between point counts and sonar counts followed a 1:1 ratio in the Pascagoula River and Washita River. In all other rivers, the ratio between the two counts differed from 1:1 in that the number of logs counted during point counts tended to be greater than the number of logs counted on sonar—with the exception of the Escatawpa River where the opposite was observed (Table 2). Collectively, the number of logs counted during point counts matched the number seen on sonar 25% of the time, 38% of the time logs were overestimated during point counts compared to sonar, and 37% of the time the number of logs was underestimated.

Deadwood density estimated with point counts was positively related to both sonar TDD (R^2 = 0.8578, P = <0.001; Fig. 2B compared to Fig. 2C) and sonar BDD (R^2 =0.5276, P = 0.025 Fig. 2B compared to Fig. 2D). The prediction interval surrounding the regression between point-count density and sonar TDD spanned about 125 logs/km when estimated density was low and over 300 logs/km when estimated density was high (Fig. 5A). The prediction interval was wider when point-count deadwood density was compared to sonar BDD, ranging from approximately 175 logs/km at low densities and over 500 logs/km at high densities (Fig. 5B).

When 25 random 40-m segments were used to estimate deadwood density, across all rivers an average of 19% of the 500 iterations calculated deviated from the sonar TDD by ≥ 15% (range: 0.08–0.33) and an average of 24% of the 500 iterations deviated from the sonar BDD by ≥ 15% (range: 0.10–0.45). Deadwood densities estimated from point counts always underestimated sonar TDD with more than a 15% difference (Fig. 6A). Deadwood densities were so greatly underestimated that in six of the nine rivers the estimations fell outside of the distribution of possible outcomes resulting from sonar sub-sampling. Transforming point-count deadwood density estimations using the sonar TDD regression equation increased estimations enough that only two rivers fell outside of their potential density distributions (Pond Creek and Caney River). Of the remaining seven rivers, five of the estimations were within 15% of the true density. Transformation of the deadwood estimate for Pond Creek resulted in a negative value and was not displayed on the plot. This result demonstrates a minimum density required for use in the regression—such a low deadwood estimate resulted at Pond Creek because it was flooded during point counts.

When point-count-estimated densities were compared to randomly generated densities of BDD, density was overestimated in four rivers and underestimated in five rivers (Fig. 6B). Point-

count estimations fell outside of the possible range for three of the nine rivers. Of the six rivers in which point-count density fell within the potential outcomes, two were within 15% of the true BDD. Transforming point-count density estimations using the sonar BDD regression equation slightly improved the accuracy of point-count estimations compared to random sonar densities doubling the number of estimates that were within 15% of the true value.

Sonar sub-sampling. Regardless of the number of segments a river was divided into, as the total survey length increased so did the minimum number of kilometers needed in a sub-sample to estimate TDD within 15% of the true value (Fig. 7A; Table 3). More divisions lead to a smaller slope, and confidence intervals surrounding slopes for treatments with 10, 15, 20, and 25 segments overlapped and were separate from the remaining division treatments (Fig. 7B). Dividing the river into 100 segments resulted in a slope that was different from all other treatments except for the 75-segment treatment. The minimum proportion of the river required in a sample did not significantly increase with total surveyed length (Fig. 7C; Table 4). In general, the proportion of river required in a sub-sample decreased when the number of divisions increased, but did not significantly differ between division treatments as indicated by the wide and overlapping confidence intervals surrounding the y-intercept (Fig. 7D). Sub-samples declined from counting approximately 60% down to 40% of the river as the number of segments increased. Overall, 55% or less of the river was required in a sub-sample to estimate deadwood density when the river was divided into 50 segments or more, and confidence intervals were tighter around the estimate in terms of both the minimum kilometers and minimum proportion of the river required in a sub-sample.

The same relationship between total survey length and minimum number of kilometers needed in a sub-sample was present when segments of equal size were sub-sampled (Fig. 8A;

Table 3). Again, as rivers were divided into a greater number of shorter segments, the slope of the regression decreased (Fig. 8B). The slope corresponding to the division treatment resulting in the greatest number of segments (40 m) was different from all other treatments except for segments of 100 m—with the upper bound of one confidence interval exactly matching the lower bound of the other. In all treatments, except segments of 500 m, the proportion of the river included in a sub-sample significantly decreased as the total length of surveyed river increased—unlike when the river was divided into an equal number of segments (Fig. 8C; Table 4). The slope of this relationship was not significantly different between division treatments; confidence intervals overlapped between all treatments (Fig. 9). The regressions between total length of surveyed river and the proportion of river needed in sub-samples resulted in y-intercepts ranging from just over 65% to just under 45% of the river being required in a sub-sample. Disregarding the 500-m treatment, confidence intervals surrounding the y-intercept of the proportion of river counted overlapped for treatments of 250, 150 and 100 meters and dividing the river into segments of 40 m was only different than segments of 100 m (Fig. 8D).

The standard deviation in the number of logs per segment was homogenized among rivers when sites were divided into many small segments, regardless of the method in which divisions were made (Fig. 10). When rivers were divided into the same number of segments, the minimum number of kilometers needed in a sub-sample increased with greater standard deviation in logs per segment (Fig. 10A; Table 5). This pattern may be an artifact of the way the rivers were divided into segments. Longer rivers had a higher standard deviation in logs per segment simply because the segments were also longer (Fig. 10B). When the river was divided into segments of the same length (Fig. 10C), the standard deviation in the number of logs was

not related to the length of the surveyed river (Fig. 10D) and also did not impact the minimum kilometers required in a sub-sample (Table 5).

Discussion

Deadwood density method comparison. Although previous studies have ground-truthed sonar-derived estimates of deadwood (Kaeser and Lits 2008, Koeller 2014), to my knowledge this is the first to compare the technology to a standard field estimator across a diverse range of sites. Given my results, the point-count method used in this study was variably accurate on a small scale. The number of logs counted during point counts was inconsistently correlated with the number of logs seen in the same area on sonar. While relationships between the two measurements were significant in the majority of rivers, the strength and nature of the relationship differed, indicating that the accuracy of point counts are, to some extent, river-dependent. In this study, the density of deadwood was estimated during point counts based upon the pieces of structure that could be visually detected—either because the structure was visible just below the surface of the water or because part of the structure was emergent. When most of the structure is completely submerged and not visible from the water's surface, point counts would underestimate the deadwood density compared to sonar—as likely occurred at the Escatawpa River. However, during sonar review, a single downed tree and its limbs were considered one piece of structure. In the field, it can be difficult to discern whether limbs emerging from the water are all part of a single downed tree or are distinct limbs from multiple trees. This uncertainty can lead to overestimating the density of deadwood compared to sonar—as observed for many of the study sites. Similar to my results, another study also found that greater numbers of logs were recorded in traditional field counts than were detected on sonar

(Kaeser and Litts 2008). Conversely, in a lentic system, interpretation of sonar data compared to field point sampling was 74% accurate and log abundance was underestimated in the field more often than overestimated (Koeller 2014). The mixed results from other studies are consistent with the variable agreement that I observed between the two methods on a small scale.

Because point counts only included logs within 10-m of the bank, I expected the point-count-estimated densities to correlate strongly to sonar BDD. While the correlation was moderately high, the prediction intervals were so wide that it is unlikely that point-count density is sufficiently indicative of true BDD. For example, assume a density of 100 logs/km was estimated using point counts. The prediction interval indicates that a true BDD with a value anywhere between 37 logs/km and 269 logs/km would be consistent with the regression line. Such a wide range reduces the utility of point-count estimates to assess BDD. The point-count method both overestimated and underestimated BDD when compared to the distribution of potential estimations of BDD generated by randomly sampling sonar (see Fig. 6B). This reflects the inconsistent accuracy of the point counts on a small-scale. However, two-thirds of the untransformed point-count estimations were within the distributions of potential densities. In this way, point counts appeared to better estimate BDD than TDD, as originally expected. Transforming data did not improve interpretation of point-count density estimations, because transformation increased some estimations while it decreased other estimations.

Nevertheless, point counts may be an appropriate method to estimate TDD rather than BDD. Even though point counts tended to document a greater number of logs compared to sonar on a small scale, point counts always underestimated TDD. Not only do transformed data points fall within the potential density distributions, 5 of the transformed estimations were within 15% of the actual TDD. The only 2 rivers in which estimated densities fell outside of the distributions

were those in which point count surveys took place during high water, obscuring much of the structure. Even though river-dependent error seems to act upon point-count density estimates on a small scale, the relationship between point-count estimates and TDD was consistent enough to result in a smaller prediction interval and a greater R-squared value. In other words, on a large scale (such as along an entire surveyed length) fine-scale differences in the estimated number of logs between the two methods are not as consequential because where deadwood may be underestimated in one area, it is likely overestimated in another—thereby obscuring the difference. Because of this, with additional transformation, deadwood density estimated with point counts may be an adequate measure of TDD for point-count estimations greater than 50 logs/km—estimations less than this threshold would result in a negative value. To further confirm this, additional point-count and sonar deadwood densities that were not used to generate the regression should be tested.

One previous study that compared field deadwood density measurements to sonar measurements conducted field surveys during low water levels and in a river with low turbidity (Kaeser and Lits 2008). Under these conditions, field measurements were more reflective of true deadwood density. Because point counts in this study were conducted in deep, turbid waters within a small radius of sample points, I would not consider my method to be a measure of ground-truthing in the same way other studies have compared traditional field methods to sonar methods. I would even argue that ground-truthing in such systems may be impossible aside from during occasional droughts. Nevertheless, the comparison between deadwood density estimated with points counts and sonar TDD resulted in an R-squared value that was comparable to a similar comparison in Kaeser and Litts (2008), indicating that even with poor visibility in the field, sonar and traditional field methods are highly predicative of one another.

While this study demonstrates that point counts can reasonably be used to estimate TDD in a river, using side-scan sonar to quantify deadwood density is a more comprehensive and effective method to cover long lengths of river. The primary advantage of sonar is that pieces of deadwood that are completely submerged and visually undetectable from the surface, in addition to deadwood that is emergent, can still be quantified. This advantage is particularly important when assessing habitat for a species that rarely leaves the water, such as bottom-dwelling turtles. Furthermore, the depth and turbidity of the water does not inhibit the measurement of deadwood density when using sonar (Kaeser and Litts 2013)—even in flooded conditions—but it does greatly affect the accuracy of deadwood that is visually detectable in the field, as demonstrated by the greatly underestimated point-count densities from Pond Creek and the Caney River. The use of side-scan sonar to quantify deadwood abundance in rivers also maximizes the amount and type of data collected without adding additional time in the field. The data are easily reviewed after collection and can be used to investigate a number of questions not addressed in this study, such as patterns in longitudinal (Kitchingman et al. 2013) and temporal distribution of deadwood, riverbed substrate composition (Kaeser and Litts 2010), and fish abundance (Flowers and Hightower 2015). Such questions are not easily investigated with standard field methods across large spatial scales

Side-scan sonar does come with some disadvantages, however. Many factors impact the quality of the sonar images and resolution of individual logs during review, such as flow conditions, suspended sediments within the water column, and position of structure in relation to the sonar transducer (Kaeser and Litts 2008). Additionally, large piles of logs and brush are often obscured on sonar with shadows, making the differentiation between individual pieces of structure difficult. The combination of these conditions could lead to potentially fewer logs seen

on sonar than there actually are. For example, when compared to a deadwood census during low water conditions, sonar was able to account for 80% of the TDD seen in the field (Kaeser and Litts 2008). Additionally, the interpretation of sonar images is subjective, and TDD may be reviewer-dependent depending, for example, on how complex piles of branches are scored (Kaeser and Litts 2008). However, when measuring deadwood density, differences in results generated by different interpreters were found to only be significant at small spatial scales (Kitchingman et al. 2013).

Sonar sub-sampling. The use of side-scan sonar to quantify underwater habitat can be useful but time-consuming to sample large areas (as in this study), but depending on the intended application of the data a larger degree of error may be acceptable. In such cases, saving time during the sonar processing and reviewing stage might be desirable, especially if a large number of sample sites are involved. In the case of quantifying deadwood density, the amount of effort required was determined by a proportion of the sample area rather than a fixed length. It was best to sub-sample rivers that had been divided into many small segments rather than fewer large segments. Sub-sampling smaller segments was not so critical when the length of surveyed river was short, but greatly reduced the amount of processing needed when the surveyed length was long. Because survey sites differed in length, it was better to use divisions that were the same size to avoid biases that may result from comparing rivers that have been divided into the same number segments but of different length. All of this together indicates a division treatment of less than 100 meters be used and the amount of river sub-sampled be customized to the length of the surveyed reach. It is important to note that the target precision (in this case 15%) may not apply to all study designs. A secondary outcome of the sub-sampling procedure was therefore

providing an easily replicated method for researchers to examine the effect of precision on sampling effort according to the conditions applicable to their own study system.

Conclusions. In light of my results, the methods used to quantify underwater turtle habitat should match the scale of interest. In the case of my study, TDD was determined within long stretches of river with the intention that methods tested in this study be used to assess habitat quality of future reintroduction sites for Alligator Snapping Turtles. In this context, side-scan sonar—the method resulting in the most comprehensive assessment of submerged structure—should be used in the future, and if time allows, TDD should be determined along the entire length of surveyed river. The best alternative method is to sub-sample sonar data by counting logs in a random selection of 40-m segments, with the number of segments counted comprising 40–60% of the total length of the surveyed area. If manual field measurements must be made in lieu of side-scan sonar, the point-count method may be applied and the resulting deadwood density can be transformed to convert the values to a scale that matches sonar TDD. However, the accuracy of results will be river-dependent. Randomly generated point-count densities using sonar images deviated from the true density by ≥15% as much as 33% of the time in one river and on average 24% of the time across all rivers. This variation in the accuracy of point counts indicates that conducting 25 pairs of point counts at all sample sites regardless of length is inferior to conducting point counts at fixed intervals. An alternative method would be to increase the number of point counts conducted as the total surveyed length increases (similar to sub-sampling sonar data), but the accuracy of such a method has yet to be tested. Comparison between methodologies such as presented in this study highlight the importance of study design and will aid researchers in gathering more meaningful data in the future.

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Table 1. State, year surveyed, length of study reaches, average channel width, and deadwood densities (DD). Total deadwood density (TDD) and bank-side deadwood density (BDD) were estimated using sonar. Reported numbers are in logs/km. Rivers are ordered from least TDD to greatest TDD.

River	State	Survey Year	Length (km)	Width (m)	Sonar TDD	Sonar BDD	Point Count
Poteau River—Wister	OK	2018	10.98	65	128	63	
Neosho River	OK	2018	3.58	177	129	36	
Fall River	KS	2019	5.33	48	141	65	97
Elk River	KS	2019	7.17	48	188	125	123
Verdigris River—Oologah	OK	2019	5.40	66	192	98	
Big Cabin Creek	OK	2018	6.83	55	197	130	
Chouteau Creek	OK	2018	2.57	47	211	169	
Washita River	OK	2019	9.40	46	218	161	140
Pearl River	MS	2019	8.11	88	227	91	141
Pond Creek	OK	2019	8.16	36	234	165	36
Poteau River—Arkoma	OK	2018	10.96	58	281	178	
Old River Channel	OK	2019	4.63	89	287	237	385
Caney River	OK	2019	17.88	64	303	181	87
Verdigris River—Toronto	KS	2019	6.22	37	320	242	187
Chickasawhay River	MS	2019	6.30	60	322	109	162
Deep Fork River	OK	2018	11.99	40	340	160	
Pascagoula River	MS	2019	4.53	123	241	109	159
Big Creek	OK	2019	4.48	37	448	181	
Escatawpa River	MS	2019	4.90	45	458	339	182

Table 2. Regression analysis results comparing number of logs counted around sample points during point counts and number of logs counted on sonar images in the same area. Reported P-value is of the slope.

River	Slope	R^2	P-value	df
Chickasawhay	0.7602	0.7270	0.0010	49
Pascagoula	0.8953	0.7755	0.1216	49
Pearl	0.7654	0.5612	0.0250	46
Escatawpa	1.5212	0.7054	0.0007	49
Elk	0.6349	0.5310	0.0001	49
Old River Channel	0.5296	0.7775	<0.0001	49
Washita	0.9248	0.5903	0.3106	49

Table 3. Regression analysis results for the effect of the length of sample reach on the minimum number of kilometers needed in a sub-sample to estimate deadwood density with under different division treatments.

Treatment	Slope	R^2	P-value	df
10 Segments	0.5673	0.9479	<0.0001	16
15 Segments	0.5541	0.9679	<0.0001	16
20 Segments	0.5576	0.9521	<0.0001	16
25 Segments	0.5094	0.9577	<0.0001	16
50 Segments	0.4092	0.9784	<0.0001	16
75 Segments	0.3629	0.9680	<0.0001	16
100 Segments	0.3214	0.9612	<0.0001	16
500 m	0.5052	0.9461	<0.0001	16
250 m	0.4140	0.9376	<0.0001	16
150 m	0.3522	0.8910	<0.0001	16
100 m	0.3183	0.8923	<0.0001	16
40 m	0.2253	0.8708	<0.0001	16

Table 4. Regression analysis results for the effect of the length of sample reach on the proportion of river needed in a sub-sample to estimate deadwood density under different division treatments. Reported P-value is of the slope.

Treatment	Intercept	Slope	R^2	P-value	df
10 Segments	0.6127	-0.0047	-0.0503	0.6355	16
15 Segments	0.5877	0.0030	-0.0544	0.6812	16
20 Segments	0.5156	0.0030	-0.0584	0.7371	16
25 Segments	0.5344	-0.0018	-0.0629	0.8201	16
50 Segments	0.4717	-0.0055	0.0347	0.2288	16
75 Segments	0.4316	-0.0059	0.0361	0.2254	16
100 Segments	0.4049	-0.0074	0.0935	0.1243	16
500 m	0.6270	-0.0106	-0.0103	0.3747	16
250 m	0.6644	-0.0230	0.3406	0.0082	16
150 m	0.6413	-0.0261	0.5556	0.0004	16
100 m	0.5559	-0.0213	0.4523	0.0019	16
40 m	0.4355	-0.0191	0.5635	0.0003	16

Table 5. Regression analysis results for the effect of standard deviation in logs per segment on the number of kilometers needed in a sub-sample to estimate deadwood density under different division treatments. Reported P-value is of the slope.

Treatment	Intercept	Slope	R^2	P-value	df
10 Segments	1.5372	0.0490	0.7239	<0.0001	15
15 Segments	1.4488	0.0700	0.7619	<0.0001	15
20 Segments	0.6582	0.1034	0.8163	<0.0001	15
25 Segments	1.2805	0.0915	0.6433	<0.0001	15
50 Segments	1.0596	0.1250	0.6874	<0.0001	15
75 Segments	1.0640	0.1409	0.6135	<0.0001	15
100 Segments	1.0873	0.1438	0.5414	0.0004	15
500 m	1.7346	0.0577	0.194	0.0437	15
250 m	1.9973	0.0597	0.1263	0.0889	15
150 m	2.1870	0.0508	0.0283	0.2445	15
100 m	2.2736	0.0363	-0.0389	0.5359	15
40 m	2.1690	-0.0346	-0.0532	0.6676	15

Fig. 1. A drone image of an Alligator Snapping Turtle utilizing a piece of submerged structure. Also note the growth of biofilms on the structure's surface and the presence of fish nearby. Photo Credit: Kristen Sardina.

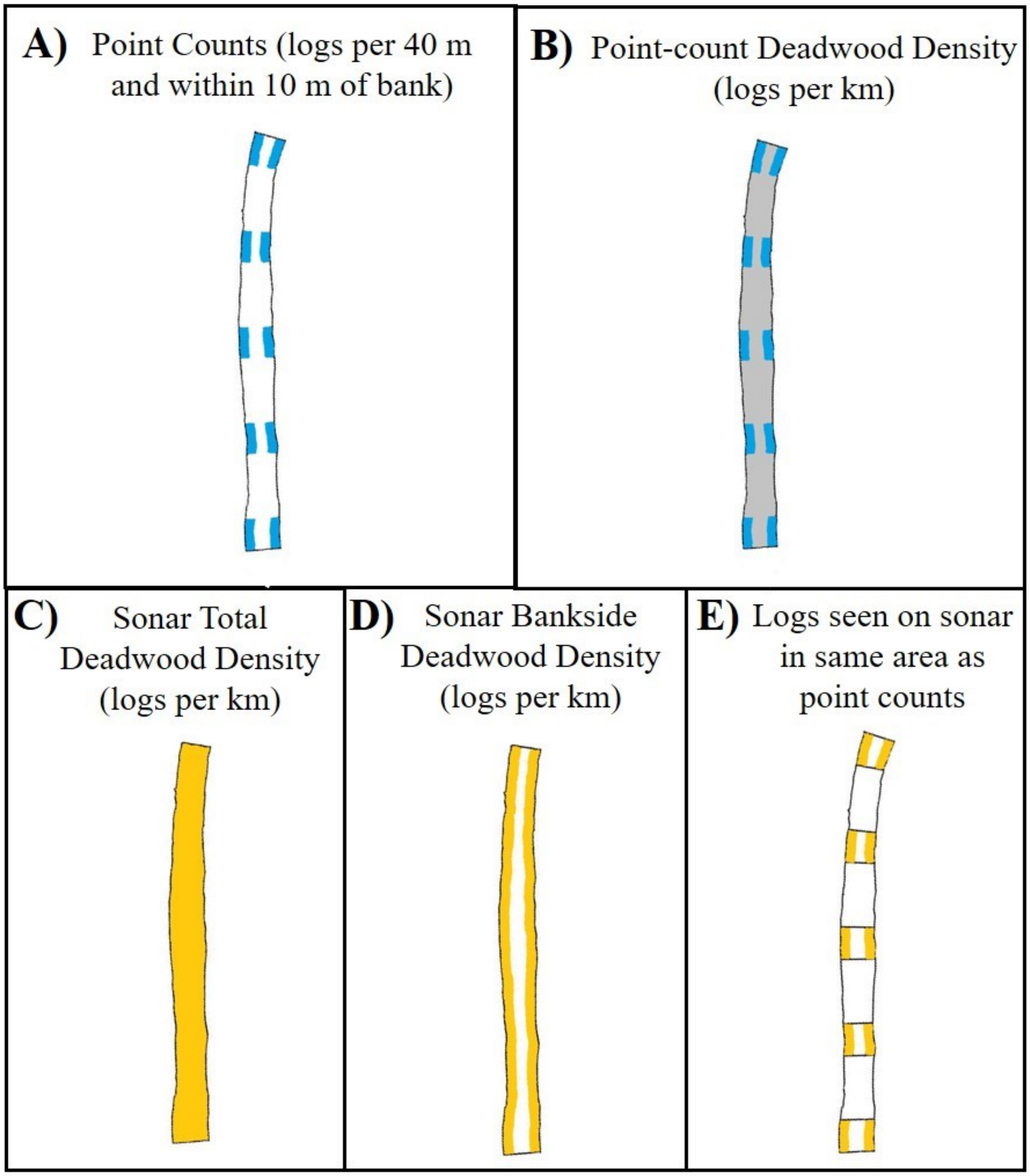

Fig. 2. A comparison of methods to estimate deadwood density. A) During point count surveys, the number of logs were counted 20 m upstream and downstream of samples points and within 10 m of the bank. Sample points were paired and evenly spaced along the sample reach. B) Point count surveys were scaled from number of logs per 40 m to number of logs per kilometer, resulting in point-count-estimated deadwood density. C) Total deadwood density (TDD) was logs counted within the sample reach on sonar. D) Bankside deadwood density (BDD) was logs counted on sonar within 10 m of the bank along the entire sample reach. E) The number of logs were counted on sonar within the same area as counted during point count surveys.

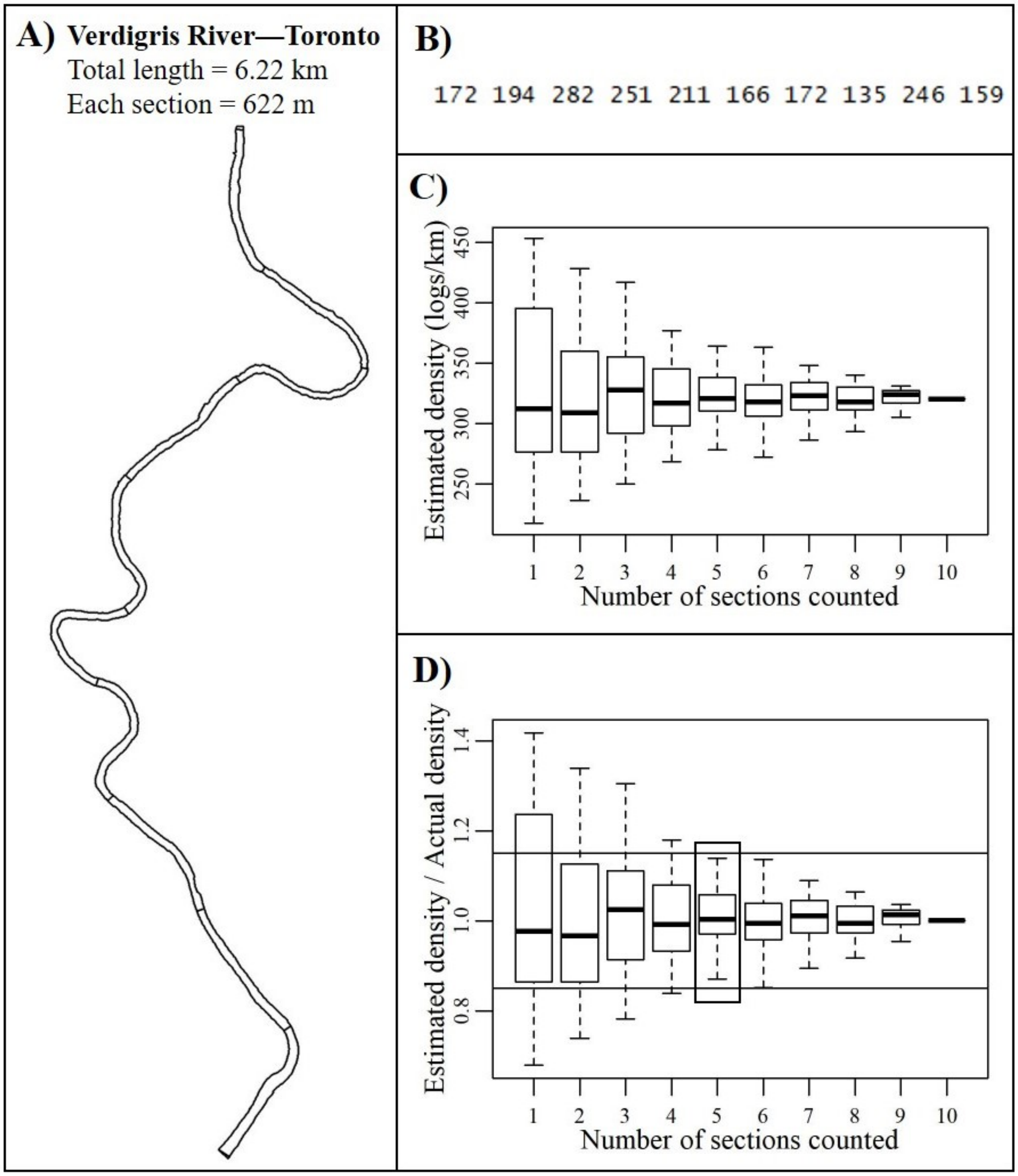

Fig. 3. An example of the sub-sampling procedure to determine the minimum amount of river needed to estimate deadwood density. A) The river was divided into 10 segments of equal length. B) The number of logs in each segment was counted. C) 50 random samples were selected and deadwood density was calculated each time. This was done for an increasing number of segments included in the sub-sample. D) The proportion between the estimated density resulting from each random sample and the actual density was calculated and plotted for each number of segments included in the sub-sample. The minimum number of segments needed in a sub-sample (indicated by the black box) was determined by finding the number of segments counted in which 95% of the 50 iterations were within 15% of the true density (indicated by the black lines).

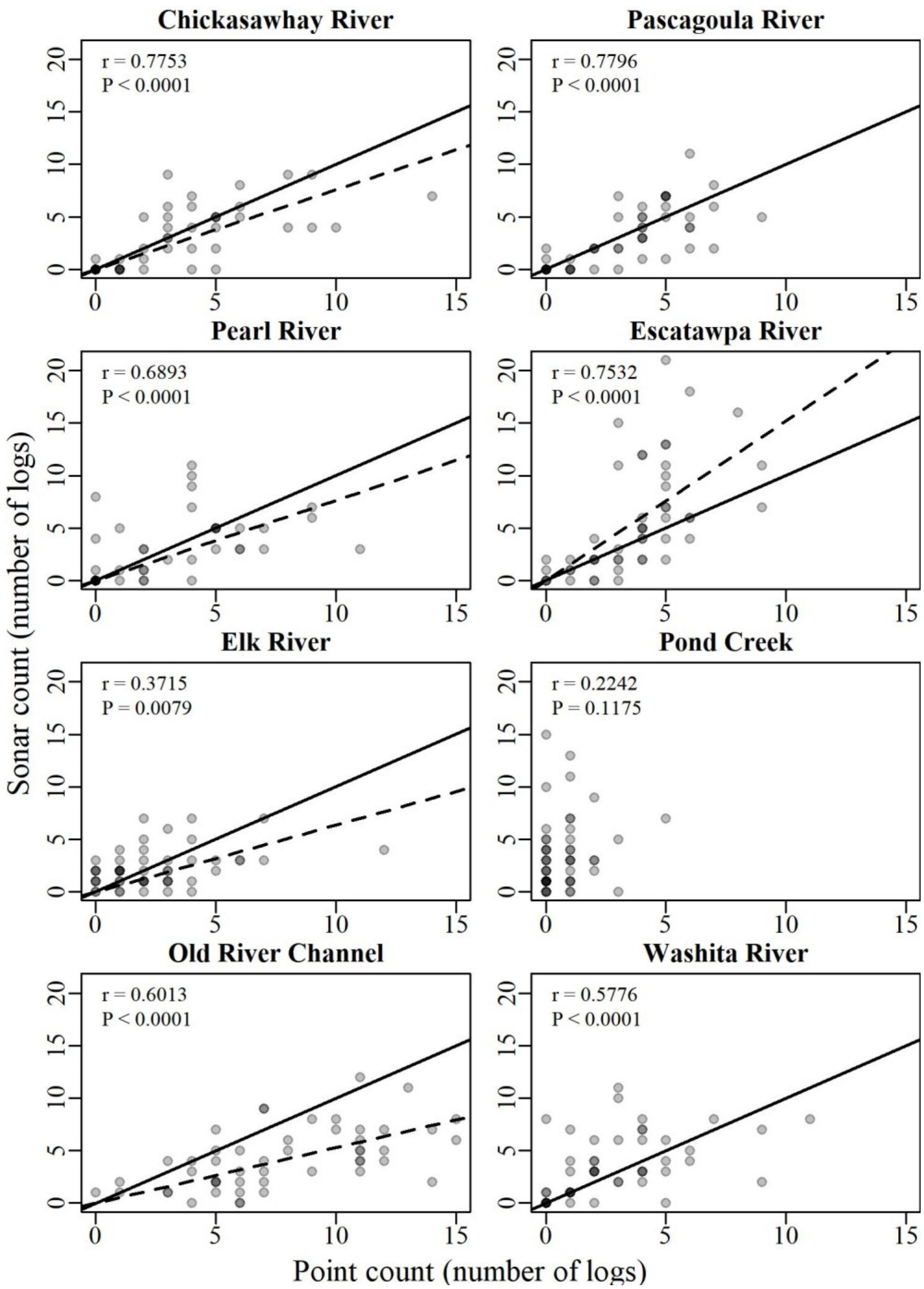

Fig. 4. The correlation between the number of logs counted in the field 20 m upstream and downstream of sample points and within 10 m of the bank versus the number counted on sonar images in the same area. Darker points indicate multiple data points overlapped. Black lines represent a 1:1 relationship whereas dashed lines indicate relationships that differed significantly from 1:1. Lines were excluded if the correlation was not significant.

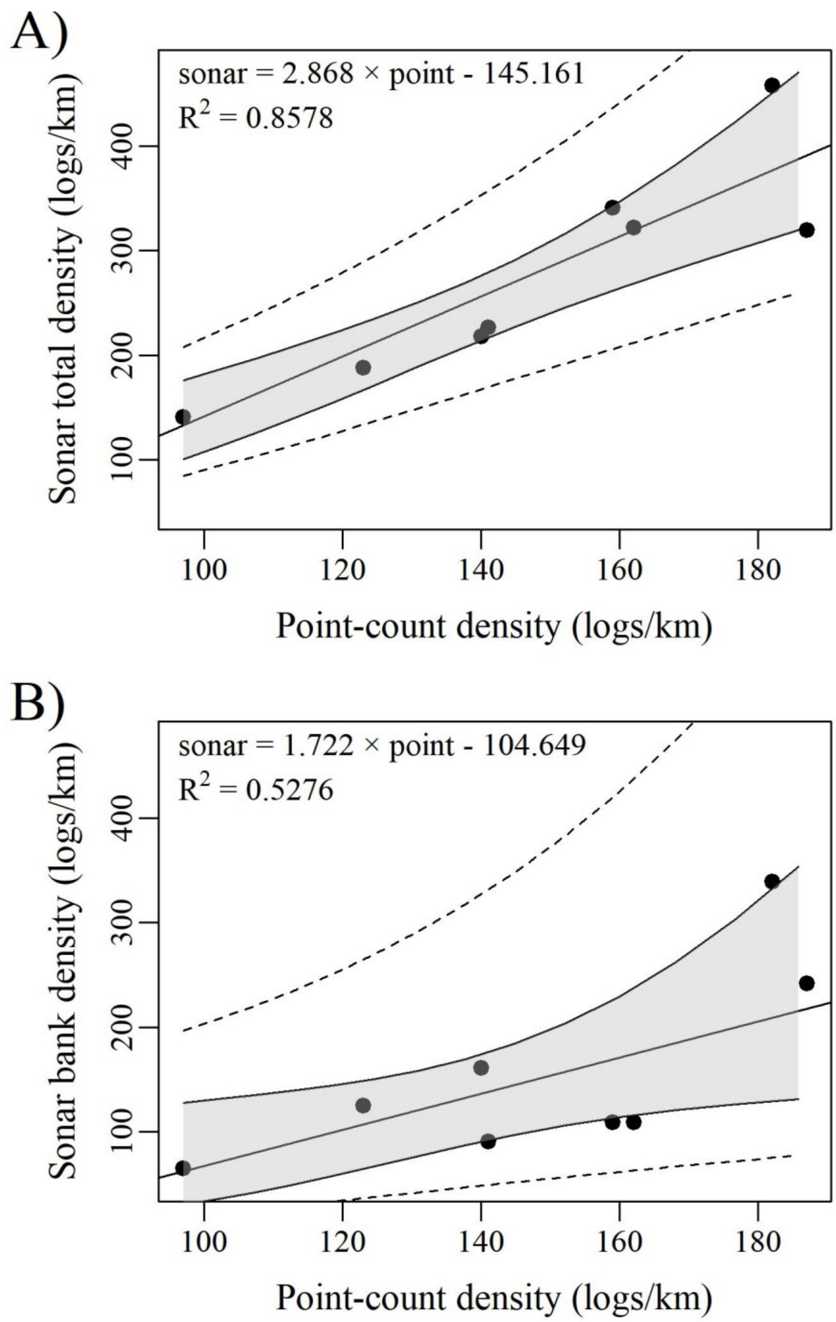

Fig. 5. Back-log transformed regression analysis between deadwood density estimated by counting logs around sample points and A) sonar total deadwood density and B) sonar bank-side deadwood density. Shaded region represents a 95% confidence interval and dashed lines represent a 95% prediction interval.

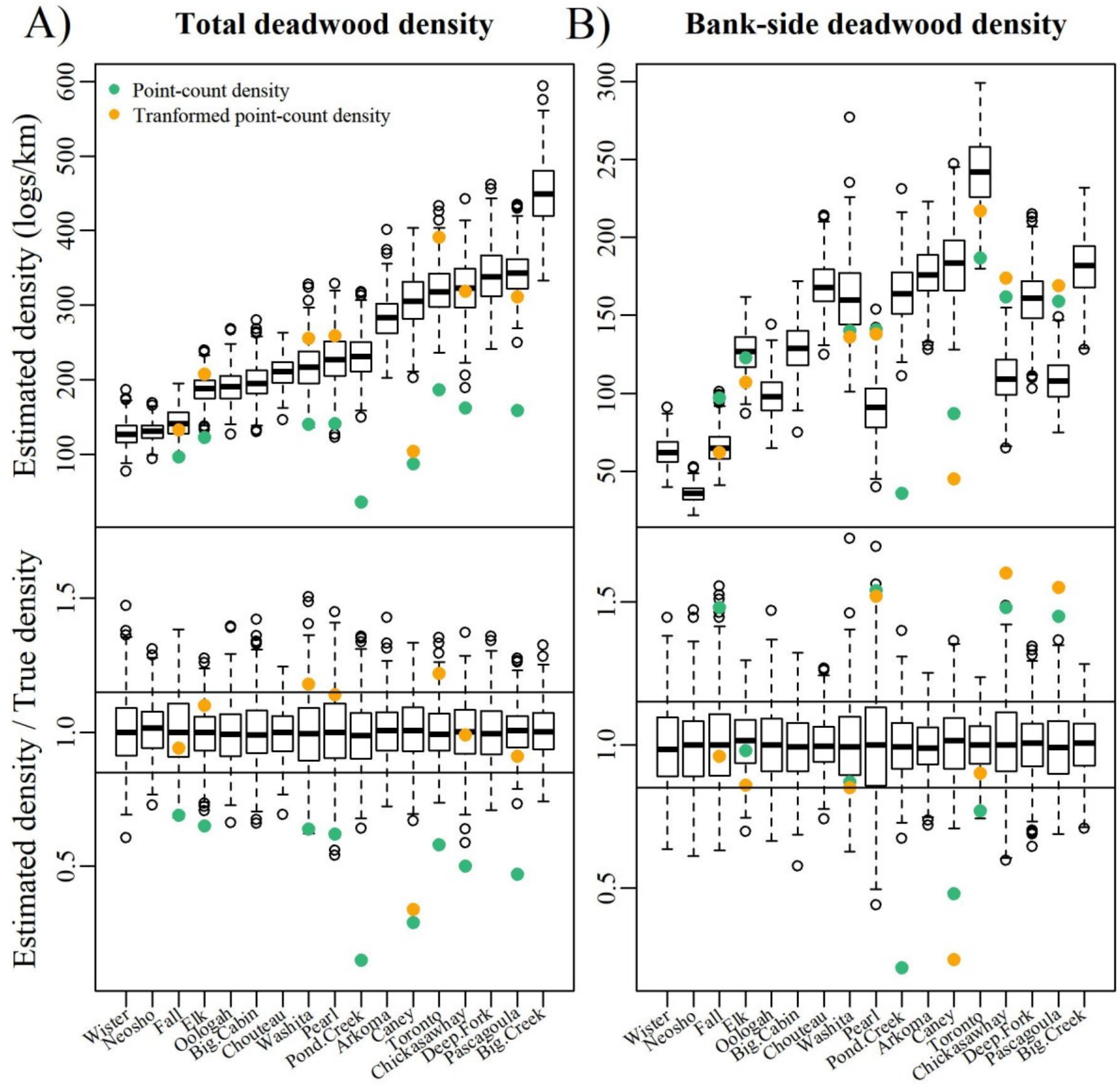

Fig. 6. Upper boxplot distributions display estimated deadwood density resulting from 500 random samples of 25 40-m long segments of river when A) all logs were included in sampling and B) only logs within 10 m of the bank were included in sampling. Lower boxplots represent the distribution of the ratio between the estimated density resulting from each sample and the true density. Horizontal lines indicate a 15% difference from the true density. Teal points indicate deadwood density estimated from point counts conducted in the field while orange points indicate point-count data transformed with either the regression relating to total deadwood density or bank-side deadwood density (see Fig. 5).

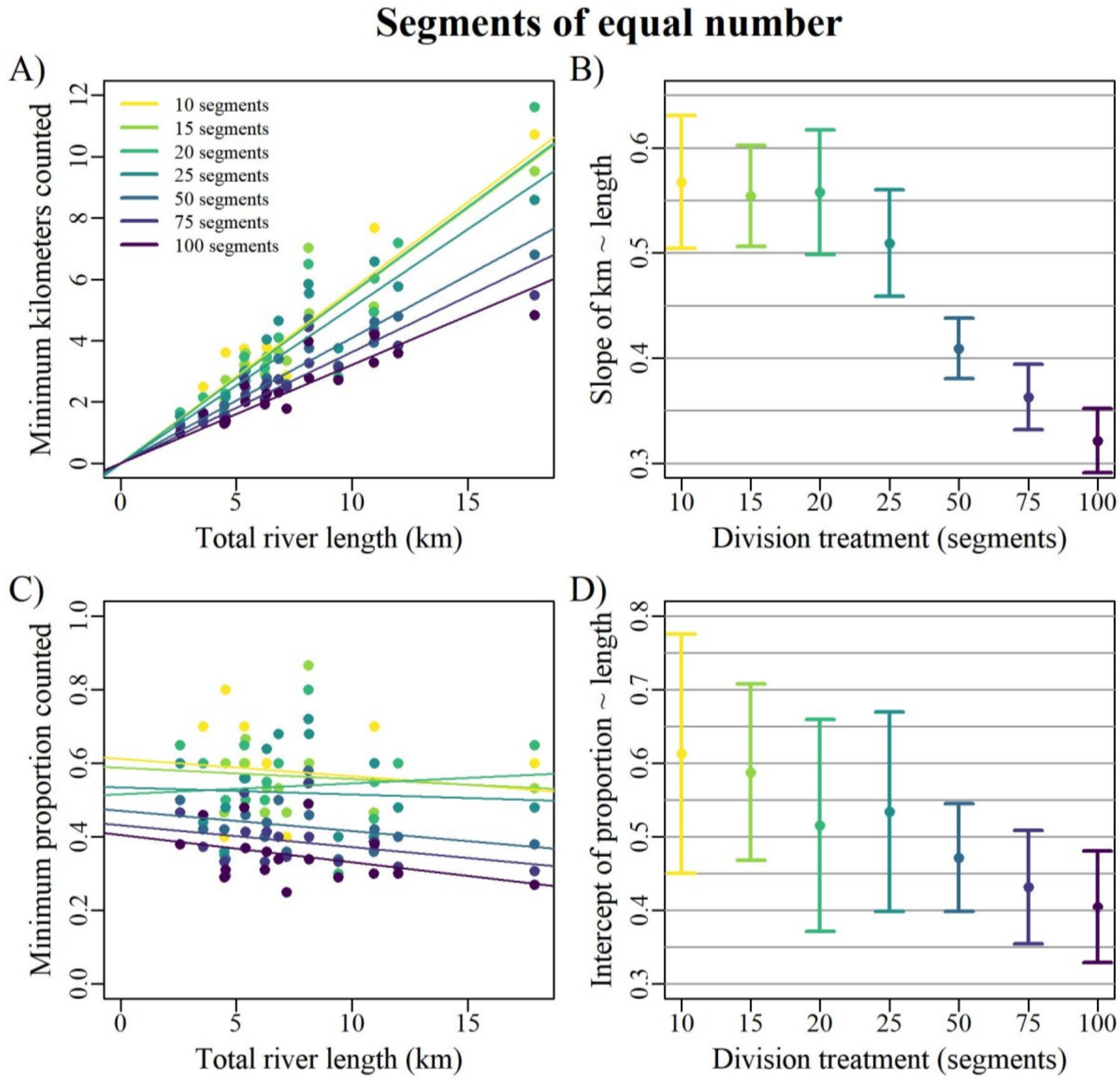

Fig. 7. River sub-sampling results when rivers were divided into an equal number of segments. A) the effect of the total surveyed length of river on the minimum kilometers needed in a sub-sample, B) the slopes of the regressions in A) and their confidence intervals, C) the effect of the total surveyed length of river on the proportion of river needed in a sub-sample, and D) confidence intervals surrounding the y-intercepts of the regressions in C).

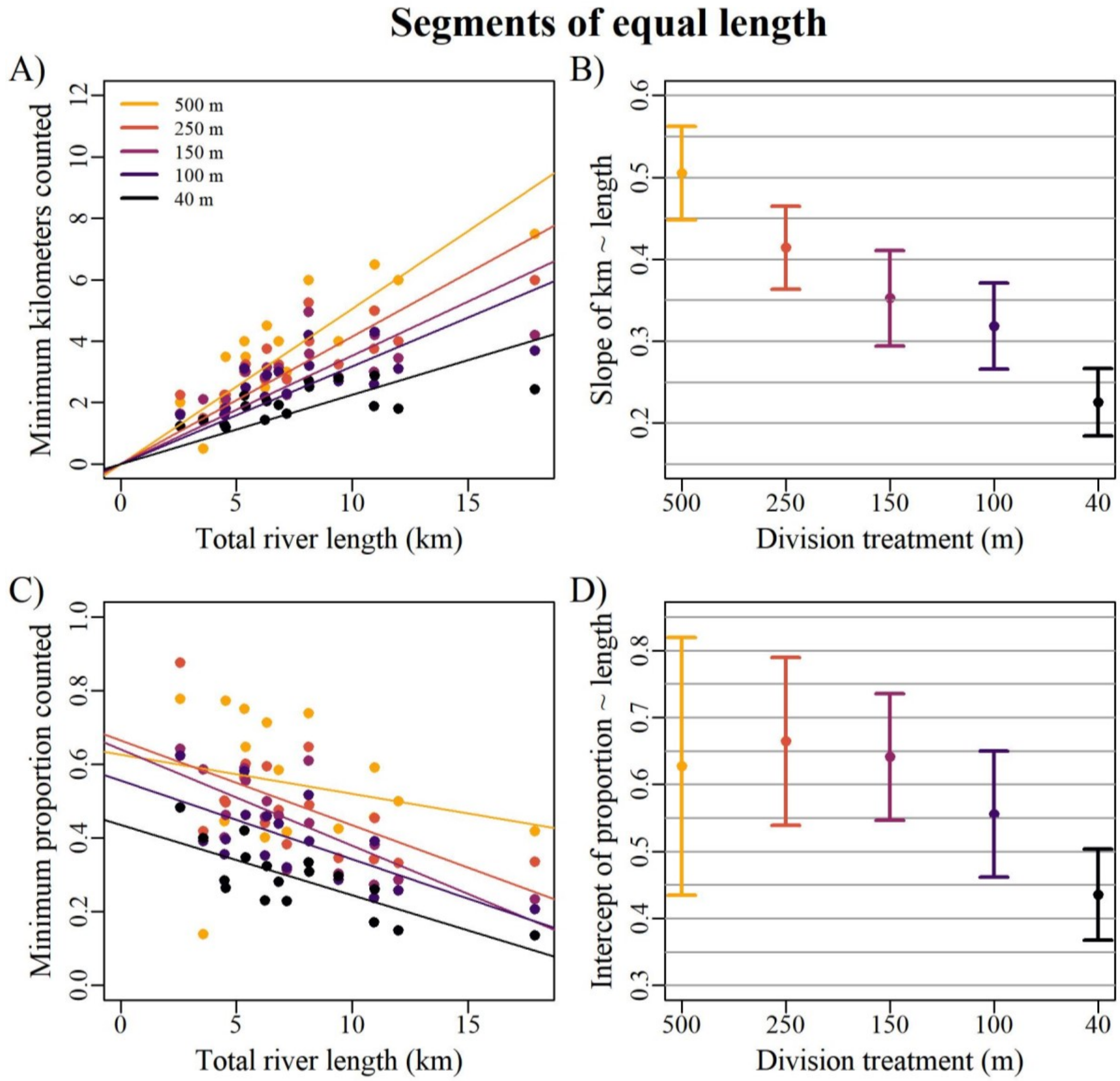

Fig. 8. River sub-sampling results when rivers were divided into segments of equal length. A) the effect of the total surveyed length of river on the minimum kilometers needed in a sub-sample, B) the slopes of the regressions in A) and their confidence intervals, C) the effect of the total surveyed length of river on the proportion of river needed in a sub-sample, and D) confidence intervals surrounding the y-intercepts of the regressions in C).

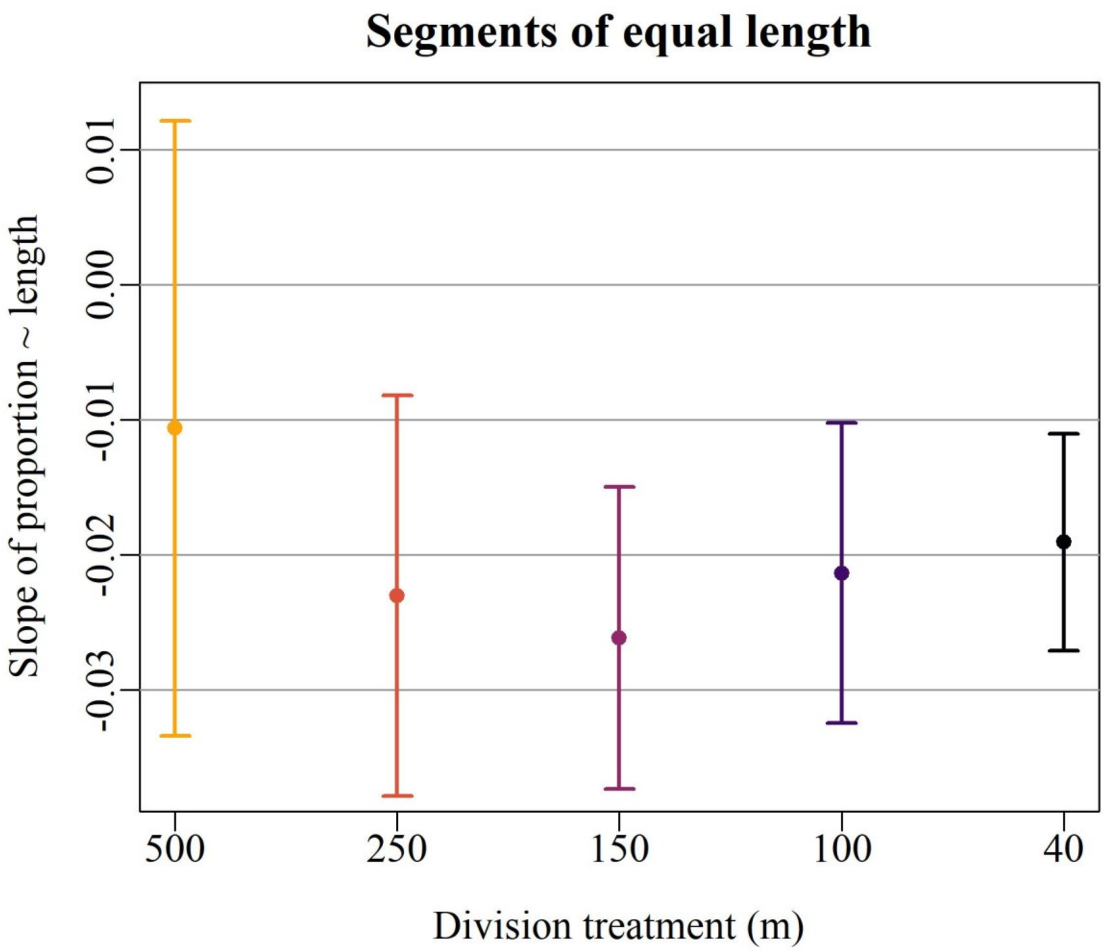

Fig. 9. Confidence intervals surround the slopes of the regressions between the length of surveyed river and proportion of river needed in a sub-sample (see Fig. 8C) for each of five division treatments when rivers were divided into segments of equal length.

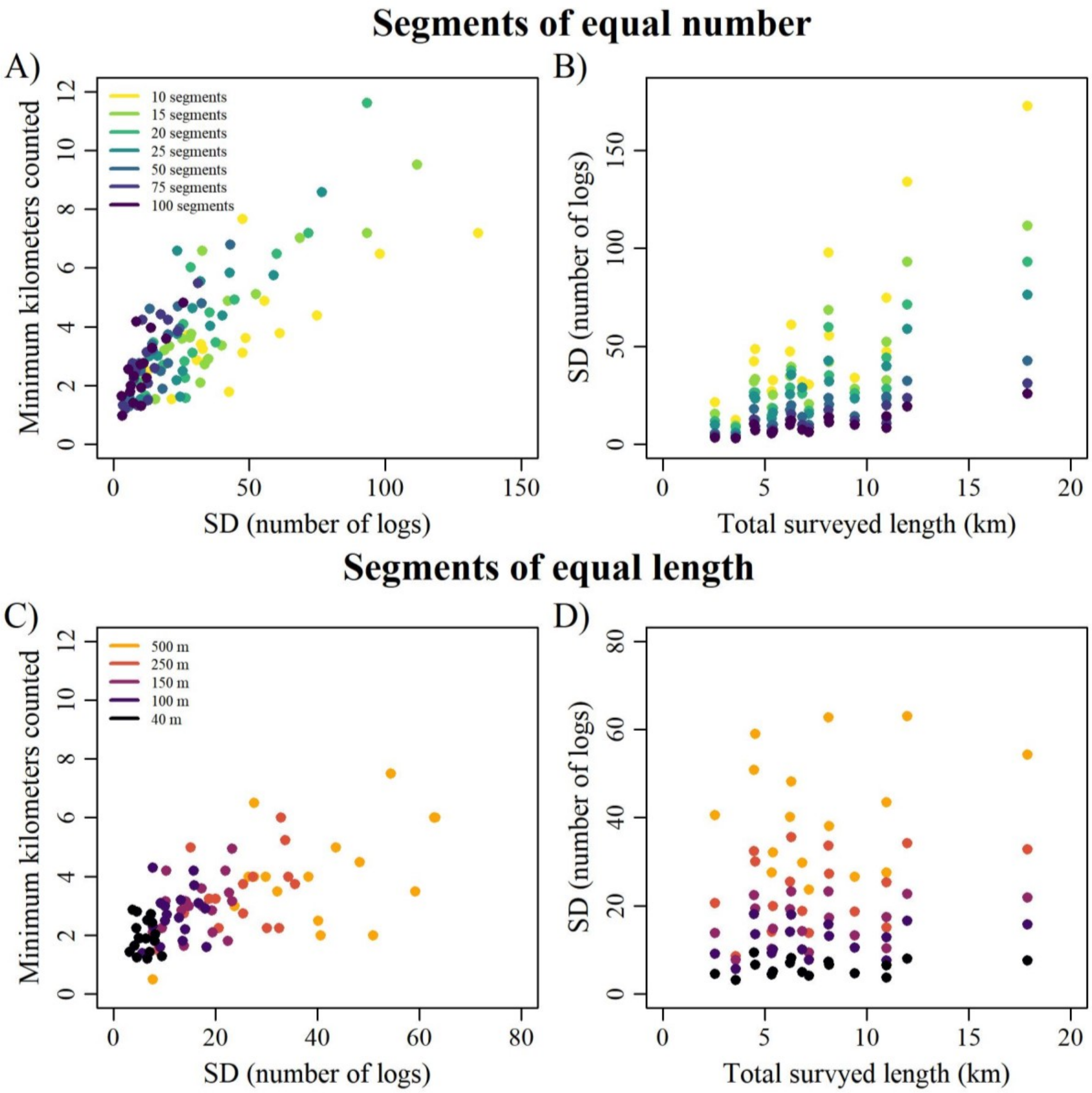

Fig. 10. The effect of standard deviation in the number of logs per segment on the minimum number of kilometers needed in a sub-sample (A and C), along with the effect of the total surveyed length on the standard deviation in the number of logs per segment (B and D) when rivers were divided into segments in two ways.

A STANDARDIZED FIELD METHOD AND HABITAT SUITABILITY MODEL TO ASSESS SUITABILITY OF REINTRODUCTION SITES FOR ALLIGATOR SNAPPING TURTLES

Abstract

A head-start program was initiated to aid in the recovery of the alligator snapping turtle (*Macrochelys temminckii)* throughout the Mississippi River drainage, and head-started juveniles have been released into rivers and associated wetlands in Oklahoma, Illinois, Tennessee, and Louisiana. As the success of the breeding program increases, there is a need to identify additional release sites where suitable habitat persists for reintroduction of alligator snapping turtles, but methods to evaluate habitat suitability remain loosely defined. I developed a standardized field survey and habitat suitability model that can be used to assess and prioritize potential release sites. Important habitat features were identified following a literature review and included submerged deadwood density, canopy cover, bank height, bank angle, bank substrate composition, and amount of human disturbance. These habitat attributes were measured in the field and indexed using a simple model that translated continuous data to an ordinal scoring system, resulting in an overall site suitability score ranging from 0 (poor suitability) to 27 (high suitability). I developed the model using two potential measurements of submerged deadwood density (a traditional field method and a method using side-scan sonar) and determined habitat suitability among 17 sites in Kansas, Oklahoma, and Mississippi where alligator snapping turtles were absent, present, and reintroduced. The sonar model performed more accurately than the field model and correctly predicted the presence of good habitat at sites where wild and reintroduced populations of alligator snapping turtles were present. Overall, the

sonar model predicted suitability ranging from fair to excellent habitat; no sites were assessed as poor habitat. The application of the suitability model removes the necessity of having years of experience and knowledge of alligator snapping turtle habitat requirements, facilitating the initiation of reintroductions throughout the species' range. Additional advantages include a method to evaluate population success in light of habitat suitability and identify potential site-specific habitat deficiencies. Continued assessment of the variety of habitats in which alligator snapping turtles can be found will provide much needed insight that will aid in the conservation of this declining reptile.

Introduction

In recent decades, the alligator snapping turtle (*Macrochelys temminckii*)—like many other turtle species—has experienced significant declines and extirpations throughout its range, largely due to habitat alteration and human exploitation (Ernst and Lovich 2009). The alligator snapping turtle is a large, highly aquatic species that inhabits Gulf of Mexico river drainages from eastern Texas through the Florida panhandle and north into Kansas, Missouri, and Illinois. Currently, it is listed as Vulnerable on the IUCN Red List of Threatened Species and is protected to varying degrees in all states where it occurs (IUCN 1996, Roman and Bowen 2000, Reed et al. 2002). Surveys have assessed the current distribution and abundance of alligator snapping turtles. Population declines and range contractions are most pronounced in the northern periphery of the distribution such as in Oklahoma (Riedle et al. 2005), Missouri (Shipman and Riedle 2008, Lescher et al. 2013), Tennessee (J. Ennen, unpublished data), Kansas (Shipman et al. 1995), Illinois (Bluett et al. 2011), and Kentucky (Baxley et al. 2014).

In response to the collapse of alligator snapping turtle populations, a head-start program was established at Tishomingo National Fish Hatchery in 2000 with the goal of restoring populations in rivers and other suitable wetlands where the species has been extirpated and to supplement small populations that are unlikely to persist without intervention. Head-started juvenile turtles produced from the program have been released at four sites in Oklahoma (Anthony et al. 2015, Moore et al. 2013), one site in Illinois (Dreslik et al. 2017), seven sites in Louisiana (Dreslik et al. 2017, J. Carr, personal communication), and at least one site in Tennessee (Ream 2008, Colvin 2015).

Despite these reintroduction efforts, the number of alligator snapping turtle populations across the species' range is still a small fraction of what was present historically (Riedle et al. 2005); therefore, continued restoration efforts depend on identifying additional sites where re-establishing the species appears feasible. However, rigorous methods for identifying and prioritizing release sites are lacking. Past reintroductions have occurred in rivers and associated wetlands within the historical range of the alligator snapping turtle where trapping surveys determined that a viable population was absent but that suitable habitat remained. Characterizations of suitable habitat were done qualitatively by visual inspection by researchers with years of experience conducting field studies of alligator snapping turtles. Because identifying suitable habitat is considered to be crucial step for a successful reintroduction program (IUCN 2013), broadly applicable and standardized methods developed to assess habitat at potential reintroduction sites for alligator snapping turtles could be a powerful tool for future reintroductions.

Habitat suitability index (HSI) models were first used in the 1980s by the United States Fish and Wildlife Service for ecological assessment, managing species of concern, identifying

and protecting critical habitat, and identifying suitable habitat for reintroduction efforts for a wide variety of taxa(U.S. Fish and Wildlife Service 1981, Brooks 1997). Habitat suitability models are typically designed to generate a score from 0 (poor or unsuitable) to 1 (maximally suitable) for a site based on essential habitat features—such as food, cover, and breeding sites—needed for various life stages and seasonal habitat use of a target species. Habitat features are typically selected based on literature reviews, opinions of researchers knowledgeable in the requirements of the focal species, targeted field studies, and/or mixed linear models, and then combined into a single equation to describe habitat suitability (Brooks 1997). The results of the model are often tested or validated against population data, rankings made by knowledgeable researchers, or another habitat suitability index model (e.g. Santos et al. 2006, Wakamiya and Roy 2008, Warren et al. 2016). Despite their theoretical utility, some HSI models have been criticized because they failed to account for the variability of input data, sampled habitat variables at a scale inappropriate for the species' biology, or did not demonstrate whether model output is responsive to changes in input data (Roloff and Kernohan 1999). Thus, future HSI models should address these potential shortcomings.

An added challenge is the application of a controversial method such as HSI models to a species capable of inhabiting a wide range of environmental conditions (e.g. Jurzenski et al. 2014). Alligator snapping turtles are considered to be habitat generalists as they are found in rivers, and associated lakes, ponds, canals, oxbows, swamps, and bayous (Ernst and Lovich 2009). They opportunistically forage on a wide range of food items and seem to thrive under diverse conditions provided there is low adult mortality, sufficient cover, and suitable nesting habitat. I attempted to incorporate the wide variety of habitat conditions where alligator snapping turtles can be found by adapting an HSI model (Macdonald et al. 2000) created for another

habitat generalist, the beaver (*Castor fiber*) (Hartman 1996, Müller-schwarze 2011, Baldwin 2013).

The goal of my study was to develop a standardized field survey protocol and a broadly applicable habitat suitability model that can be used to assess the suitability of potential reintroduction sites for alligator snapping turtles to improve future alligator snapping turtle reintroduction efforts. This method will facilitate habitat assessment by reducing the need for extensive knowledge about alligator snapping turtle habitat requirements, thereby increasing accessibility for wildlife managers to initiate reintroductions and make well-informed management decisions. Specifically, I address the following: 1) Can a suitability model following the format used by Macdonald et al. (2000) characterize a wide range of habitat suitability that accurately corresponds to presence of alligator snapping turtles? 2) Do the results of the model at sites where alligator snapping turtles have been reintroduced agree with previous assessment by researchers? 3) Which habitat attributes are the main drivers in determining suitability score? and 4) How does the method of quantifying submerged deadwood density (a generally important habitat characteristic) affect the output of the model?

Study Area

I surveyed sites that spanned a variety of conditions in three states within the range of the alligator snapping turtle (Fig. 1; Appendix E). Sites included rivers in Oklahoma and Mississippi currently supporting natural populations of alligator snapping turtles (n = 6 sites), sites in Oklahoma where the alligator snapping turtle has been reintroduced (n = 4 sites), and sites in Oklahoma and Kansas where alligator snapping turtles likely have been extirpated and reintroductions may be warranted (n = 7). Five sites in Oklahoma were surveyed in June–July

2018. Sites in Mississippi were surveyed 27 June–03 July 2019. Finally, sites in Kansas and reintroduction sites in Oklahoma were surveyed 23 August–09 October 2019. In Oklahoma and Kansas, surveyed sites were located in the Verdigris, Neosho, and Canadian watersheds—all sub-watersheds of the Arkansas River drainage. All of the sites were upstream of reservoirs or were tributaries of dammed rivers. Due to these hydrologic alterations, the rivers were prone to frequent and prolonged seasonal flooding and/or wide daily fluctuations in water level. Sites in Mississippi were in the Pearl and Pascagoula River drainages. The study site on the Pearl River was downstream of an impoundment. The Pascagoula river drainage, on the other hand, has few impoundments that are limited to low-order streams, and therefore exhibits a relatively natural flow regime. Rivers in Mississippi were characterized by faster flow than rivers in Oklahoma and Kansas and were dominated by sandbars (Pascagoula rivers) and gravel bars (Pearl River). The Escatawpa River was the only cypress-lined system I surveyed.

Methods

Habitat Variables and Field Assessment. Alligator snapping turtles spend the majority of their lives underwater, although females emerge onto land to build terrestrial nests. Therefore, suitable habitat must provide adequately abundant submerged deadwood and shade to act as cover as well as tall, steep banks for nesting. I identified six variables to describe suitable alligator snapping turtle habitat based upon a review of studies describing the species' habitat preferences (Table 1). I also considered factors related to anthropogenic impacts on rivers that may negatively affect alligator snapping turtles.

To measure habitat variables for inclusion in a habitat suitability model, I developed a standardized field survey (Appendix F and G). Habitat measurements were taken at 25 sampling

points along each bank of a river, for a total of 50 sample points at each study site. Paired sample points were located directly across the channel from one another and were equally spaced along the surveyed stretch of river. The length of river segments included in the assessment was determined by accessibility and navigability and encompassed the length of river that has been previously surveyed for alligator snapping turtles.

Channel characteristics were measured to assess the instream habitat availability for individual alligator snapping turtles (Fig. 2A). Overstory canopy was measured from a boat approximately 5 m from the bank using a concave densitometer. Overstory was taken in four directions in relation to the river: oriented upstream, downstream, towards the left bank, and towards the right bank. The four directions were averaged to obtain canopy cover. In 2018, submerged deadwood was visually estimated within a 10-m radius of the sample point using an ordinal scale of 0 (none) to 3 (high density). A slightly modified method to estimate submerged deadwood was used in habitat surveys conducted in 2019: the number of submerged logs that could be visually detected was determined 20 m downstream and upstream of the sample point and within 10 m of the bank. These counts were then converted to the same ordinal scale used in 2018, such that 0 corresponded to zero logs detected, low density (1–2 logs), medium density (3–4 logs) and high density (≥5 logs). In addition to manually estimating deadwood, deadwood density was quantified using side-scan sonar following methods described in Chapter 2. The presence of other, non-log submerged structure objects was also noted. Examples include beaver dens, rocks or boulders, aquatic vegetation, floodplain vegetation, trees rooted into the bank with branches overhanging the water, undercut banks, partially submerged root balls, and man-made structures such as docks, bridge pylons, and artificial fish structures.

Stream bank characteristics were measured to assess the availability of nesting habitat (Fig. 2B). Percent substrate composition was measured within a 1-m strip of the bank from the water to the riparian shelf. Substrates included sand (S), dirt (D), rock (R), gravel (G), clay (C), leaf litter (L), tall or dense vegetation (DV), short vegetation such as grasses and herbaceous vegetation (SV), and woody debris or roots (W); total cover did not exceed 100. Next, bank slope was measured by placing a dial gauge angle finder along a 1 m long board placed on a representative portion of the bank. If the slope appeared to be variable from the edge of the water to the top of the bank, more than one angle measurement was taken and the measurements were averaged. Finally, vertical height was measured from the water to the riparian shelf using a 2-m long pole with graduations every 0.1 m.

The anthropogenic impacts included in the model were width of the riparian corridor, degree of human disturbance surrounding the river, and boating activity. The width of the riparian corridor was measured at each sample point using satellite imagery. An intact riparian buffer along the river is not only more suitable nesting habitat, it may also act as a natural barrier to limit human use of the river. The degree of human disturbance was measured using a secondary ranking system (Table 2) that incorporates metrics such as evidence of recreational use, types of non-forested land present on the landscape surrounding the river, and the presence of roads and residential areas along the river. The result of this ranking system is an accumulation of points associated with types of disturbance present, where a larger value indicates a higher risk for human-related disturbance to alligator snapping turtles. Finally, the number of boats seen per day of survey work was recorded—more boats seen per day is likely to be concomitant with more frequent interactions between fishermen and alligator snapping turtles, potentially leading to increased mortality.

Habitat Suitability Model. The overall habitat suitability at each site comprised three aspects of the river—channel characteristics (turtle habitat), bank characteristics (nesting habitat), and anthropogenic disturbance (Table 3). Raw measurements of habitat variables were graded into ordinal categories where each category was assigned a score of 0, 1, 2, or 3. A score of 3 for each habitat attribute, signifying excellent suitability, is associated with published values for alligator snapping turtle habitat preferences. At each of 50 sample points, scores for channel characteristics, bank characteristics, and riparian buffer were summed to give sample point totals. The overall "habitat suitability score" for each site was calculated as the average of all 50 sample point total scores plus the score associated with human disturbance.

Two versions of the habitat suitability model were developed—one using point-count deadwood density and the other using sonar total deadwood density. Because abundant deadwood appears to be a crucial habitat characteristic for alligator snapping turtles (Sloan and Taylor 1987, Ewert et al. 2006, Riedle et al. 2006), this attribute was given a weight of three. This was done in the point-count model by tripling the score for submergent deadwood at each sample point before calculating the sample point total score. In the sonar model, the deadwood density score—as determined for the whole sample site in logs/km—was tripled before adding it to the overall site suitability score. The score given to presence of non-log submergent structure was not tripled. After weighting scores in this way, the maximum habitat suitability score possible was 27 and overall habitat suitability was divided into poor (<11), fair (11–15), good (16–21), and excellent (≥22) categories.

While harvest of alligator snapping turtles is restricted in most states, they are frequently caught on unattended fishing lines such as jug lines, limb lines, and trotlines and are killed either by drowning or gunshot wounds (Folt and Goodwin 2013, Howey and Dinkelacker 2013, Moore

et al. 2013, Huntzinger et al. 2019). There may be some situations in which good habitat is present at a potential release site, but a high level of boat traffic—and therefore a potentially high level of fishing—may lead to increased mortality of released turtles. Because of this, even if habitat suitability is high in other respects, this increased mortality will negatively impact the success of reintroduction efforts and resources may be better placed elsewhere. Boat traffic was added to the model to make the distinction between rivers that have both suitable habitat and are good candidates for reintroduction and rivers that have suitable habitat but are too heavily used by the public. Instead of contributing additional points to the overall suitability score, increased boat traffic penalizes the score by deducting the indicated number of points. This final score is termed “reintroduction suitability” and can be divided into the same ratings as the habitat suitability score. Future renditions of the model should endeavor to replace boat traffic with the number of unattended fishing lines present during surveys.

Suitability Score Analysis. Habitat suitability scores were calculated for each of 17 rivers using both the point-count model and sonar model (Appendix H). Habitat suitability scores calculated with sonar were considered to be the most representative of true habitat suitability, because sonar gives the most accurate measure of deadwood density (Chapter 2). Therefore, to identify the main drivers in determining HSI score, correlations were conducted between the sonar HSI scores and the main habitat attributes. HSI scores calculated using point counts were compared to sonar HSI scores using a spearman-rank correlation to assess how the method of quantifying deadwood density affected the output of the model. Finally, reintroduction suitability scores were calculated for rivers in which alligator snapping turtles were not detected during previous survey work and used as an example of management decisions based upon scores generated by the HSI models.

Results

Prior to model testing, sites with natural populations of alligator snapping turtles ($n = 6$), as well as sites where reintroductions have been initiated after assessment by researchers knowledgeable in habitat requirements of the alligator snapping turtle ($n = 4$) were assumed to be at least Good habitat suitability. Sonar HSI scores at these sites were 16.50–21.52, corresponding to a habitat suitability rating of Good in all cases (Table 3; Appendix I). The HSI scores at sites where I failed to detect resident populations of alligator snapping turtles ($n = 7$) were 14.77–24.66, which spans the habitat suitability ratings from Fair to Excellent (Table 3; Appendix I). No sites were scored as Poor. The most consequential determinants of HSI scores were sonar deadwood density ($r = 0.7946$, $P < 0.001$) and the level of human disturbance ($r = -0.4946$, $P = 0.0435$). Both correlated significantly with sonar HSI scores (Fig. 3), whereas variation among sites in canopy cover ($r = 0.2484$, $P = 0.3363$), bank height ($r = -0.1719$, $P = 0.5093$), bank angle ($r = -0.4138$, $P = 0.0987$), and bank substrate composition ($r = 0.2337$, $P = 0.3665$) was insufficient to cause substantial differences in scores among the sites that I surveyed (Fig. 4). However, surveys did reveal important trends in habitat characteristics between sites with reintroduced and wild populations of alligator snapping turtles (Fig. 5). Sites where reintroductions have occurred in Oklahoma tended to have less canopy cover, banks that were composed of less open substrates, and steeper slopes than sites with wild populations.

The choice of methods for estimating deadwood density notably affected HSI scores. Scores calculated using sonar were, on average, 2.7 points (10%) greater than scores calculated with the point-count method (Fig. 6A). The discrepancy in scores between the two models increased as sonar deadwood density increased (Fig. 6B). Even so, the ranks of the two scores

were significantly related to one another (Spearman's rank correlation: $r = 0.5858$, $P < 0.0152$). The differences in scores between the two deadwood density estimation methods were translated to differences in overall habitat suitability ratings as well. After scores were converted to suitability ratings, eight of the 17 sites (47%) received a lower rating using the point-count method. Five of the sites with lower ratings were rivers with natural or reintroduced populations of alligator snapping turtles that were presumed to be of Good quality but were scored as Fair when using the deadwood densities estimated with the point-count method, demonstrating that the results of this model were not as accurate as the model based upon sonar.

Two point-count methods were applied in this study. Point counts conducted in 2018 quantified logs within a much smaller radius of the sample point. Suitability scores for deadwood around point counts were considered low (corresponding to SI scores between 1.5 and 4.5) at all six sites sampled in 2018 (Fig. 7). The radius around sample points was increased in 2019 to better characterize variability in deadwood. In 2019, variation in suitability scores for deadwood increased—scores for six sites were considered low density, four were medium density, and one was high density.

Boat traffic was used to determine reintroduction suitability scores for both the sonar and point-count HSI models. Reintroduction scores ranged from 11.77 (Poor) to 23.66 (Excellent) when using sonar, and ranged from 11.88 (Poor) to 19.94 (Good) when using point counts (Table 5).

Discussion

In the case of the alligator snapping turtle, a habitat generalist, an HSI model following the format of Macdonald et al. (2000) and using sonar to estimate deadwood density was an

effective way to assess habitat suitability. The sonar HSI model predicted a wide range of suitability scores that corresponded to habitat supporting alligator snapping turtle populations and in rivers that were assessed and deemed suitable by researchers with years of experience conducting field studies of alligator snapping turtles. It is true that the majority of scores fell within the Good rating with few Fair and Excellent ratings. Sites included in these surveys were selected either because they support current populations of alligator snapping turtles or because they were identified to be potentially suitable for future reintroductions. Given that the sites I surveyed were pre-judged to be potentially suitable, the potential for any one site to have received a rating less than Fair was low. At the other extreme, to receive an Excellent rating a site would need to have high deadwood density along with low anthropogenic disturbance within and around the river. It may be difficult to identify habitat that meets both of these conditions given that many rivers within the range of alligator snapping turtles are modified by dams and channelization—leading to decreased deadwood—and/or are located within an anthropogenically disturbed landscape.

Habitat suitability scores calculated with point-count deadwood data rather than sonar-estimated deadwood densities were unreliable indicators of habitat suitability, and the error increased as deadwood density increased. At the smallest scale (10-m radius in 2018), point counts failed to detect variation in deadwood density and did not characterize sites with high deadwood density as such, leading to a larger discrepancy as deadwood density increased. Even after doubling the radius around sample points, the notable difference between HSI scores using sonar and the point-count method indicated that point counts are likely insufficient to accurately characterize deadwood density. One alternative is to measure habitat attributes at a scale similar to the minimum home range of the target species (Roloff and Kernohan 1999). The 2019 point-

count method estimated deadwood density within a 400-m^2 area, which may be a suitable scale for hatchling alligator snapping turtles (maximum reported home range = 568 m^{2}; Bass 2007). But alligator snapping turtles are known to inhabit much larger areas than measured during point counts (Sloan and Taylor 1987, Harrel et al. 1996, Riedle et al. 2006, Shipman and Riedle 2008). Harrel et al. (1996) reported a mean linear home range of adult male alligator snapping turtles to be 3,495 m, which is a greater distance than the entire segment of Chouteau Creek sampled during this study. Therefore, side-scan sonar is the most effective method to quantify deadwood density for use in an HSI model because it can be applied to a scale that reflects the biology of the alligator snapping turtle. If point counts must be used in lieu of sonar, transformation of point-count data may be performed following methods presented in Chapter 2, and the resulting deadwood densities can be used in the sonar HSI model.

Reintroduction may be warranted at sites where alligator snapping turtles are not present. Trapping was conducted alongside habitat assessments in Oklahoma and Kansas to confirm absence of alligator snapping turtles, and populations were detected at two sites that had been surveyed previously without detection (Riedle 2001; See also Chapter 1). Therefore, when using the HSI model to identify suitable reintroduction sites, it is essential to combine habitat surveys with population surveys to establish presence or absence of alligator snapping turtles. After absence of alligator snapping turtle is confirmed, a method for site selection should be determined prior to conducting habitat surveys and calculating reintroduction suitability scores. For example, in this study reintroduction was considered for sites that scored at least an 18. Three sites qualified for reintroduction under this criterion using the sonar HSI model (Chouteau Creek, Deep Fork River, and the Verdigris River above Lake Toronto). Results from the point-count HSI model indicate the Verdigris River as the only suitable site for future reintroduction.

This discrepancy again highlights the importance of methods used to estimate deadwood density when using suitability scores to make management decisions.

Additional variables that could be important in describing suitable habitat for alligator snapping turtles that were not incorporated into the model include food availability, water current (Shipman 1993, Riedle et al. 2009), water depth (Riedle et al. 2006, Howey and Dinkelacker 2009, Townsend 2016), flood frequency, dispersal barriers (Riedle et al. 2008), and climate (Thompson et al. 2017). Many of these variables have been included in large scale niche models spanning the current distribution of alligator snapping turtles in the United States (Thompson et al. 2017, U.S. Geological Survey 2017). All of these factors should be considered alongside the output of the HSI model before a site is deemed suitable for reintroductions. For instance, based upon niche modeling conducted by the U.S. Geological Survey, the Verdigris River above Lake Toronto is likely at the northern edge of suitable habitat (U.S. Geological Survey 2017; Fig. 1). Alligator snapping turtles would have very little habitat to colonize because downstream dispersal would be impeded by the impoundment creating Lake Toronto. Thus, while the sonar HSI model identifies the local habitat to be suitable, reintroduction at this site may not be warranted given other considerations. In the future, publicly available data from weather stations and water gauges could be incorporated into the HSI model with the use of software such as Indicators of Hydrological Alteration (The Nature Conservancy 2009).

An advantage to using an HSI model such as used by Macdonald et al. (2000) is that the assignment of suitability for each habitat attribute was step-wise rather than continuous (Fig. 8). For example, alligator snapping turtles prefer to nest atop steep banks. Low slopes are generally less suitable than intermediate slopes, but at some point, the slope becomes too steep for a turtle to climb, thereby decreasing suitability. A continuous HSI model, like commonly used following

U.S. Fish and Wildlife Service guidelines (U.S. Fish and Wildlife Service 1981), would assume an angle of 22° is slightly more suitable than 21°, even though there is no support for such a relationship in the alligator snapping turtle literature. If the suitability criteria for bank angle used in this study were translated to the scale used by U.S. Fish and Wildlife Service, the suitability index curve would resemble a histogram. Suitability index histograms are typically used when the habitat attribute is more easily defined by categories (U.S. Fish and Wildlife Service 1981)—such as is the case for water regime for slider turtles (e.g. permanently flooded, temporarily flooded, seasonally flooded; Morreale and Gibbons 1986)—but histograms may also be useful for variables in which incremental changes would not necessarily impact suitability. Continuous suitability indices result in suitability scores with variability matching that of the input data. Suitability histograms, on the other hand, convert continuous habitat data into suitability categories that lessen the variability of the resulting scores, which more accurately reflects the habitat adaptability of alligator snapping turtles. Similar methods were used to describe suitability of sea turtle nesting habitat (Santos et al. 2006).

Many of the input variables did not have significant correlations with sonar HSI score, but because suitability was determined with histograms, many of the relationships between habitat attributes and HSI scores would not be linear. Therefore, changes to these variables are still likely to contribute to HSI scores in a useful manner (Fig. 9). One such circumstance may be the cypress-lined Escatawpa River. The Escatawpa River supported the highest deadwood density but the lowest average bank height and bank angle. The poor nesting habitat resulted in an HSI score that was lower than expected for the high deadwood density. The Old River Channel provided another example of a site with a proportionally low HSI score given the deadwood density. In this case, the lack of canopy cover and open substrates on the bank, along

with a high level of human disturbance was the cause for a lower score. Although sonar HSI scores were most influenced by deadwood density and human disturbance, these examples demonstrate that HSI scores are a sum of all the parts—the other habitat attributes included in the model will remain consequential at some sites in which the model may be applied to in the future.

Future use of this HSI model should follow several additional considerations regarding limitations and relevant application. First, it is recommended that the model only be used in lotic systems or lentic systems that maintain a riverine shape, as it has not been tested in lentic habitats with irregular shorelines. If used in habitats such as swamps, lakes, ponds, and bayous, field methods and model parameters may need to be altered to better represent such environmental conditions. Second, the season in which field surveys are conducted may impact the model output. The substrate composition of the river bank is the only habitat attribute that is likely to substantially vary seasonally as vegetation cover increases. If possible, habitat surveys should be conducted in early spring, corresponding to nesting season (Dobie 1971), so that bank substrate composition reflects the conditions experienced by females during the nesting process. Surveys conducted in late summer or fall, such as was done for surveys at sites where reintroductions occurred in Oklahoma, could result in less open substrates on the river bank than would be available during nesting season, leading to a slightly lower HSI score. Finally, I calibrated model parameters as best as possible given the current knowledge of alligator snapping turtle habitat preferences, but the model may not perform as well in states that were not included in the model development and in parts of the species' range where the model has not been tested. This is because the habitat studies used in development were limited in the scope of the possible range where alligator snapping turtles can be found. Studies included in the model

development spanned seven of the 14 states where alligator snapping turtles historically occurred, but the majority of the studies took place in Louisiana, Oklahoma, and Florida. Furthermore, very few studies have reported specific habitat attributes of nesting habitat (Ewert 1976, Ewert and Jackson 1994, Woosely 2005, Miller et al. 2014). Because of this, the parameters for nesting habitat (especially bank height) were conservative and may need to be updated as new information becomes available. Overall, testing of the model should be continued and expanded across the entire range of the species.

The application of the suitability model reduces the necessity of having years of experience and knowledge of alligator snapping turtle habitat requirements, facilitating the initiation of reintroductions throughout the species' range. In addition to prioritizing future reintroduction sites, the sonar HSI model can be used to identify habitat suitability shortcomings of rivers with wild or reintroduced populations that may be cause for population declines. Capture rates, population demographics, and nesting success are all metrics that can be compared to habitat suitability to assess the success of a population. The HSI model could then be used to identify the habitat variable(s) that is/are likely related to population decline, and habitat restoration efforts could be used to improve habitat suitability. As the number of sites that are assessed using an HSI model increases, comparisons between rivers where alligator snapping turtles are absent, present, and reintroduced will provide much needed insight regarding the conservation needs of this declining turtle species.

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Table 1. Habitat variables identified to be important to the quality of alligator snapping turtle habitat in the USA based upon literature review.

Variable	Description	Supporting literature
Submerged deadwood	Density of submerged logs, stumps, and branches that provide underwater cover and serve as "core sites"	Sloan and Taylor 1987, Harrel et al. 1996, Ewert et al. 2006, Riedle et al. 2006, Bass 2007, Lescher 2010, Moore et al. 2013, Thomas 2013, Spangler 2017
Non-log submerged structure	Other types of submerged structure such as root balls, beaver dens, undercut banks, overhanging bushes, aquatic vegetation, and made-made structure that provide cover	Sloan and Taylor 1987, Harrel et al. 1996, Ewert et al. 2006, Riedle et al. 2006, Bass 2007, Ream 2008, Lescher 2010, Moore et al. 2013, Thomas 2013, Spangler 2017
Canopy cover	Presence of high canopy cover to provide shade	Shipman 1993, Riedle et al. 2006, Shipman and Riedle 2008, Howey and Dinkelacker 2009, Moore et al. 2013, Spangler 2017
Bank substrate composition	Lack of roots, woody debris, rock, and dense vegetation on banks that would impede nest site choice and construction	Ewert 1976, Ewert and Jackson 1994, Woosely 2005
Bank angle	The average slope of the bank to be climbed prior to nesting	Ewert 1976, Woosely 2005, Miller et al. 2014
Bank height	The height of the bank (i.e. the vertical distance above the water where a nest could be constructed)	Miller et al. 2014

Table 2. Scoring system to determine the potential for human disturbance at sites assessed for habitat suitability for alligator snapping turtles in Kansas, Oklahoma, and Mississippi, USA in 2018 and 2019. The indicated number of points for each type of disturbance present were summed to generate a score for total human disturbance with higher scores representing higher risk of impacts on alligator snapping turtles.

	3 points	2 points	1 point
River use	4 or more boat ramps	2–3 boat ramps	1 or fewer boat ramps
	Dramatic water level cycling from regular dam release	Active limb/jug lines	Abandoned limb/jug lines
	Frequent flooding	Campgrounds	Rowing boats
			Foot paths
			Recreational use of banks/sandbars
Land use	3 or more roads <100-m	2 Roads <100-m	1 Road <100-m
	Buildings/docks <100-m	Oil Wells <100-m	Pastures <100-m
			Crop Land <100-m

Table 3. Alligator snapping turtle habitat suitability model used to assess habitat suitability at sites in Kansas, Oklahoma, and Mississippi, USA in 2018 and 2019. Score for submerged structure was tripled. Total scores for every sample point were averaged to obtain overall suitability score for the site. The level of human disturbance was determined using Table 2. A total of 27 points was possible. Suitability was defined as follows Poor: <11, Fair: 11–15, Good: 16–21, and Excellent: ≥22. Only one type of submerged deadwood estimation should be used.

Habitat Attribute	Score 0	Score 1	Score 2	Score 3
Channel Characteristics				
Point-count submerged deadwood (# logs)	0 logs	1–2 logs	3–4 logs	5 or more logs
Sonar counted submerged deadwood (logs/km)[a]	<50	50–149	150–249	≥250
Non-log submerged structure	Absent	One type present	Two types present	Three or more types present
Canopy cover (%)	0–9	10–24	25–49	≥50
Bank Characteristics				
Substrate composition	≥50% rock/clay	0–29% dirt/sand	30–69% dirt/sand	≥70% dirt/sand
Angle (°)	<10°, >75° *or* ≥50% rock/clay	10°–19°	20°–29°, 46°–75°	30°–45°
Height (m)	<0.5 *or* ≥50% rock/clay	0.5–0.9	1–1.49	≥1.5
Anthropogenic Characteristics				
Riparian buffer (m)	<2	2–8	9–15	>15
Human disturbance[a]	Certain (≥12)	High (8–11)	Medium (4–7)	Low (<4)
Reintroduction suitability				
Boat traffic (boats/day)[b]	<1	1–2	2–3	>3

[a] indicates a score that is added after sample point scores are averaged.

[b] indicates a score that is deducted from the overall suitability score rather than added.

Table 4. Habitat suitability scores using the sonar HSI model for sites without, with natural, and with reintroduced populations of alligator snapping turtles in Kansas, Oklahoma, and Mississippi, USA. Suitability rating was determined as Poor: <11, Fair: 11–15, Good: 16–21, and Excellent: ≥22.

Site	MATE Status	Channel Score	Bank Score	Anthropogenic Score	HSI Score	Suitability Rating
Neosho	Absent	5.19	6.69	2.88	14.77	Fair
Fall River	Absent	5.08	6.10	3.64	14.82	Fair
Poteau—Wister	Absent	4.74	6.30	3.90	14.94	Fair
Chouteau	Absent	7.08	6.50	4.75	18.33	Good
Elk River	Absent	9.06	6.76	2.88	18.70	Good
Deep Fork	Absent	10.88	7.54	4.00	22.42	Excellent
Verdigris	Absent	11.80	7.86	5.00	24.66	Excellent
Big Cabin	Present	8.27	5.63	2.65	16.55	Good
Pearl	Present	7.78	6.52	3.98	18.28	Good
Poteau—Arkoma	Present	11.18	6.26	2.98	20.42	Good
Escatawpa	Present	11.51	4.98	3.98	20.47	Good
Pascagoula	Present	10.70	6.90	3.90	21.50	Good
Chickasawhay	Present	10.12	6.46	4.94	21.52	Good
Pond Creek	Reintroduced	7.68	5.78	3.04	16.50	Good
Old River Channel	Reintroduced	9.74	5.76	2.22	17.72	Good
Washita	Reintroduced	7.10	7.68	3.94	18.72	Good
Caney	Reintroduced	10.60	6.90	2.22	19.72	Good

Table 5. Reintroduction suitability scores and ratings compared between the sonar and point-count HSI models at sites with no population of alligator snapping turtle in Kansas and Oklahoma, USA. Suitability rating was determined as Poor: <11, Fair: 11–15, Good: 16–21, and Excellent: ≥22.

	Reintroduction suitability score		Reintroduction suitability rating	
Sites	Sonar count	Point count	Sonar count	Point count
Neosho	11.77	11.88	Poor	Poor
Fall	12.94	12.00	Fair	Fair
Poteau—Wister	13.82	13.78	Fair	Fair
Elk	15.70	14.30	Fair	Fair
Chouteau	18.33	14.83	Good	Fair
Deep Fork	22.42	16.84	Excellent	Good
Verdigris	23.66	19.94	Excellent	Good

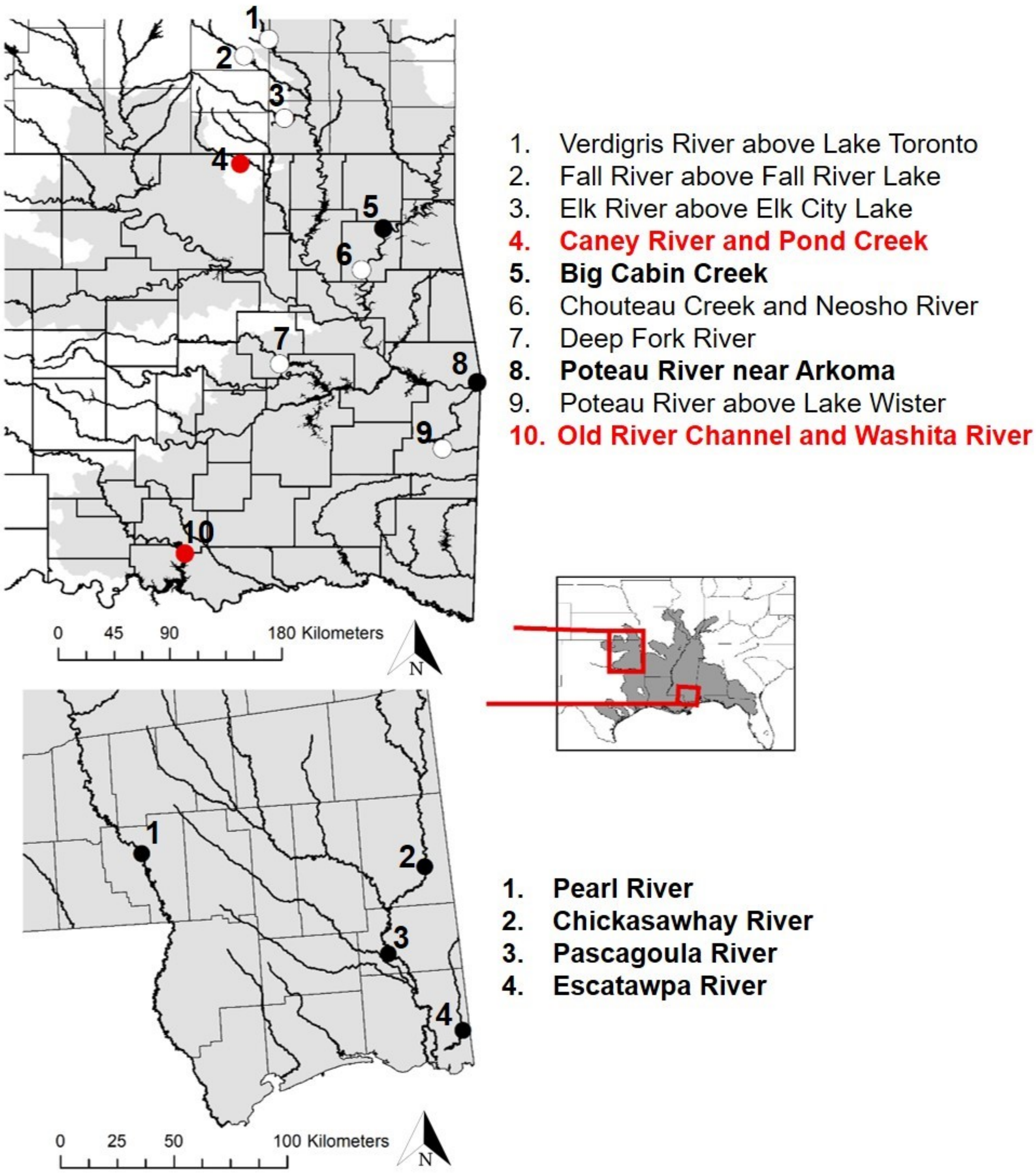
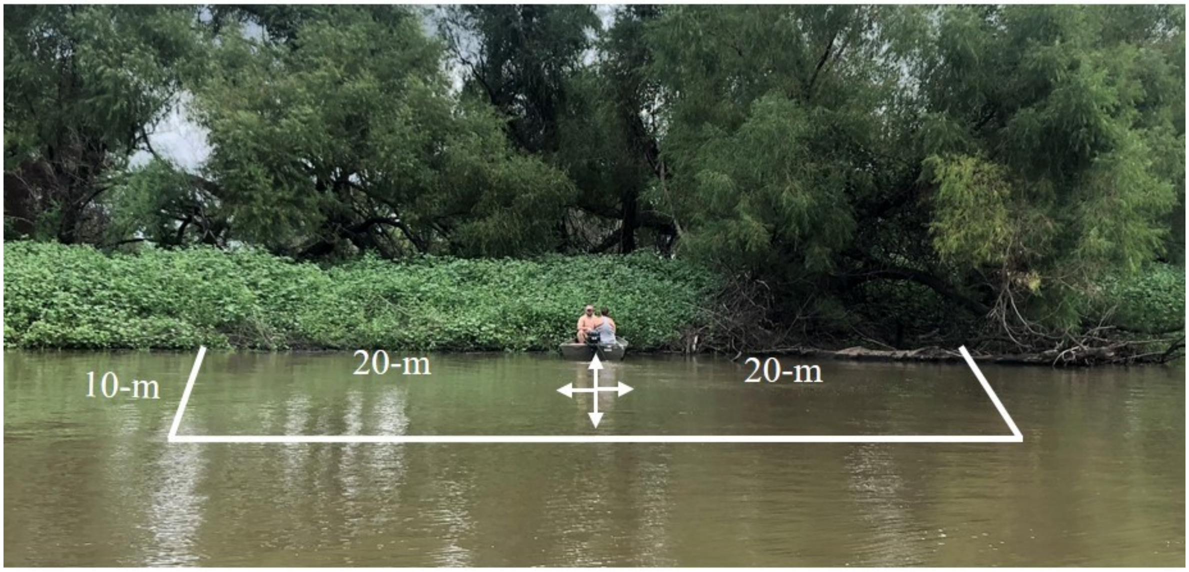


Figure 1. Locations sampled in Kansas, Oklahoma, and Mississippi, USA in 2018 and 2019 to assess habitat suitability for alligator snapping turtles. Gray shading is the extent of predicted habitat suitable for alligator snapping turtles according to niche models conducted for the U.S. Geological Survey Gap Analysis Project (U.S. Geological Survey 2017). Open, black, and red circles correspond to rivers where alligator snapping turtles were absent, present, and reintroduced respectively.

(A) Channel Characteristics



(B) Bank Characteristics

Figure 2. Demonstration of methods used to measure habitat attributes to assess suitability for alligator snapping turtles at all sample sites in Kansas, Oklahoma, and Mississippi in 2018 and 2019. (A) An example of channel characteristics where densitometer readings were taken in four directions shown by the arrows, and the number of logs were counted within the white indicator box. (B) Examples of bank characteristics where bank height was measured, substrate composition was estimated within the white indicator box, and bank angle was measured using a straightedge placed along the bank.

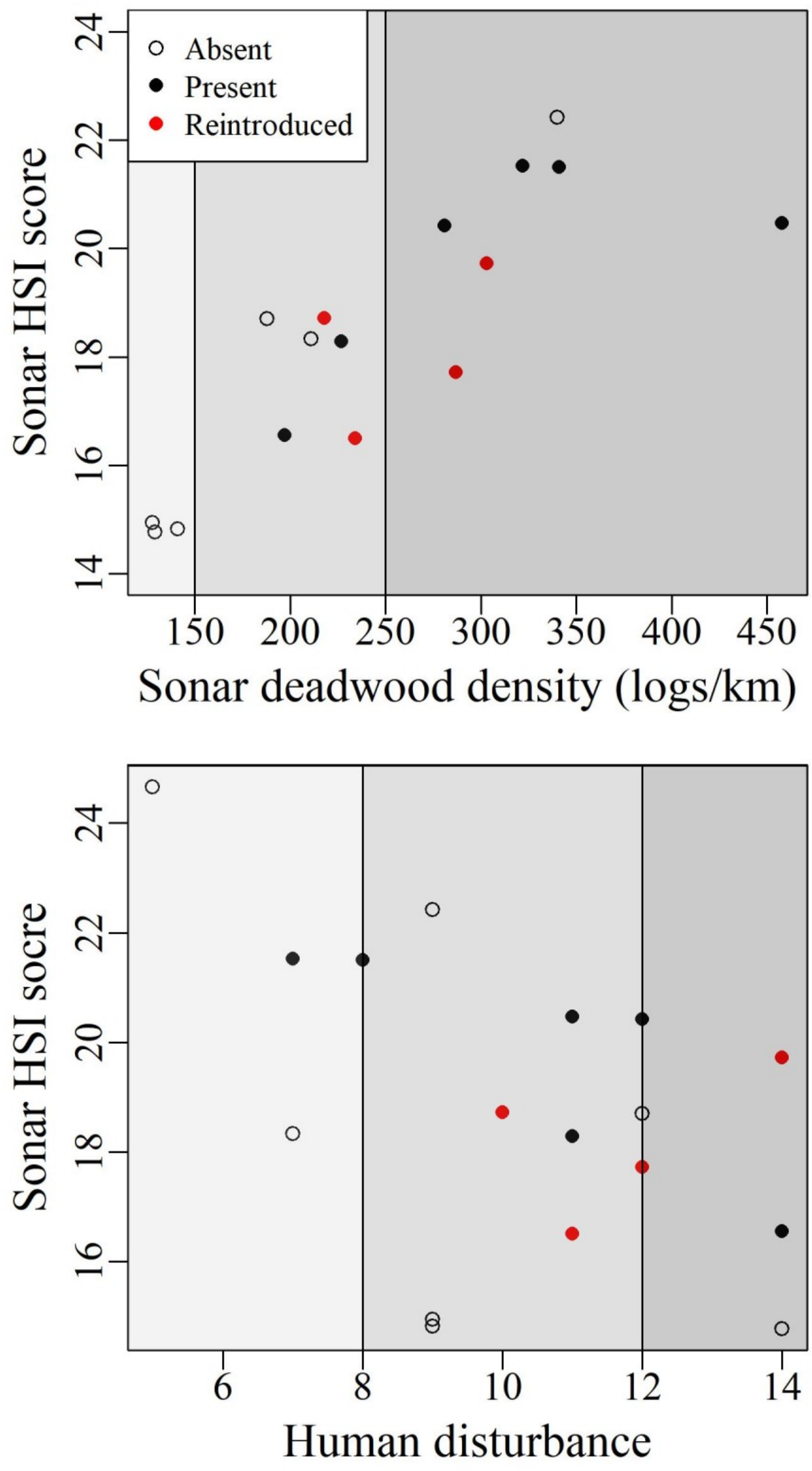


Figure 3. Sonar deadwood density and human disturbance as drivers of sonar habitat suitability index (HSI) scores among 17 sites in Kansas, Oklahoma, and Mississippi, USA in which habitat suitability for the alligator snapping turtle was assessed in 2018 and 2019. Shading indicates point values awarded for the habitat variable where dark gray corresponds to a score of 3; point values decline by one with each lighter shading.

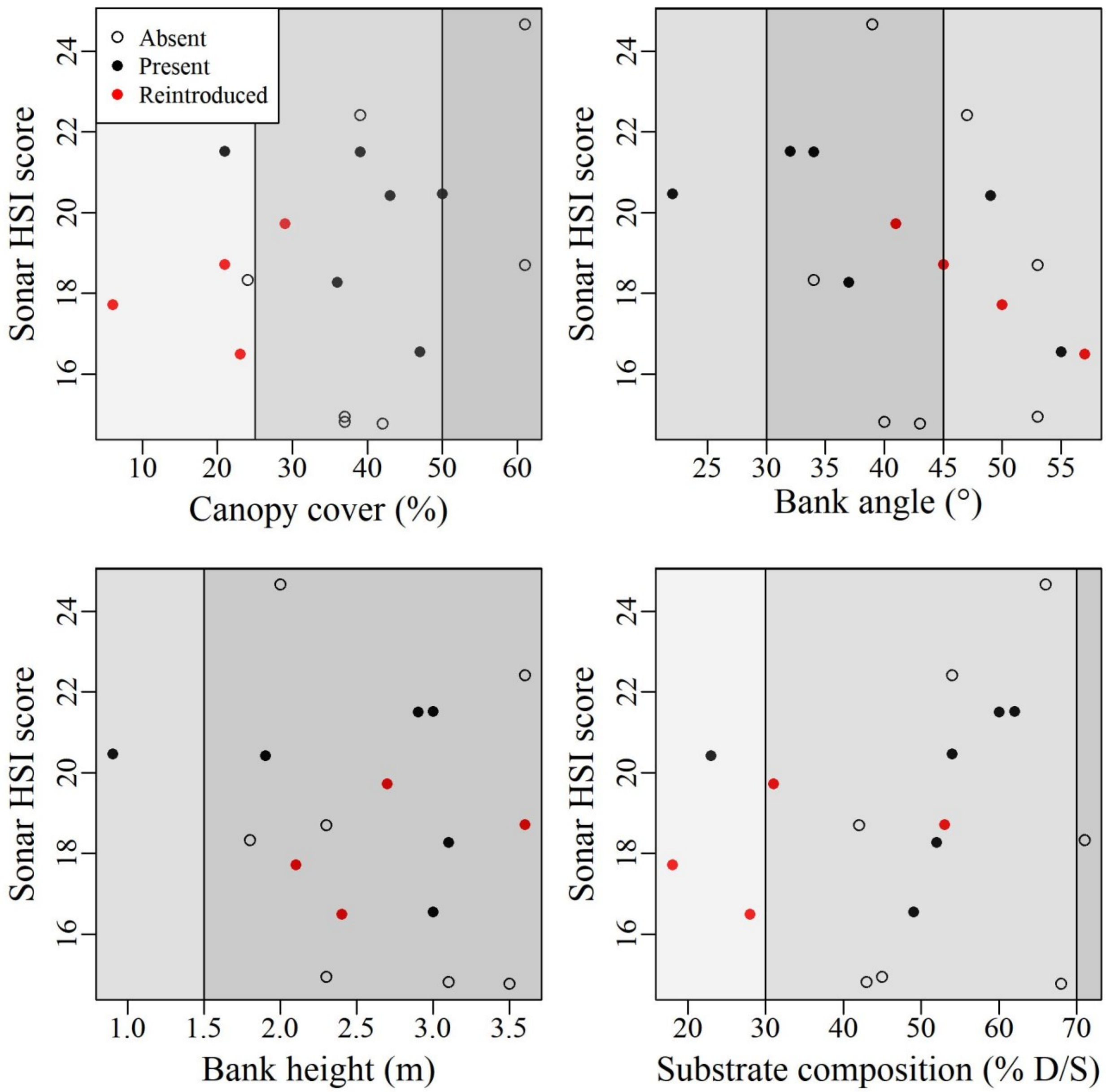


Figure 4. Average canopy cover, bank angle, bank height, and substrate composition in relation to sonar habitat suitability index (HSI) scores among 17 sites in Kansas, Oklahoma, and Mississippi, USA in which habitat suitability for the alligator snapping turtle was assessed in 2018 and 2019. Shading indicates point values awarded for the habitat variable where dark gray corresponds to a score of 3; point values decline by one with each lighter shading.

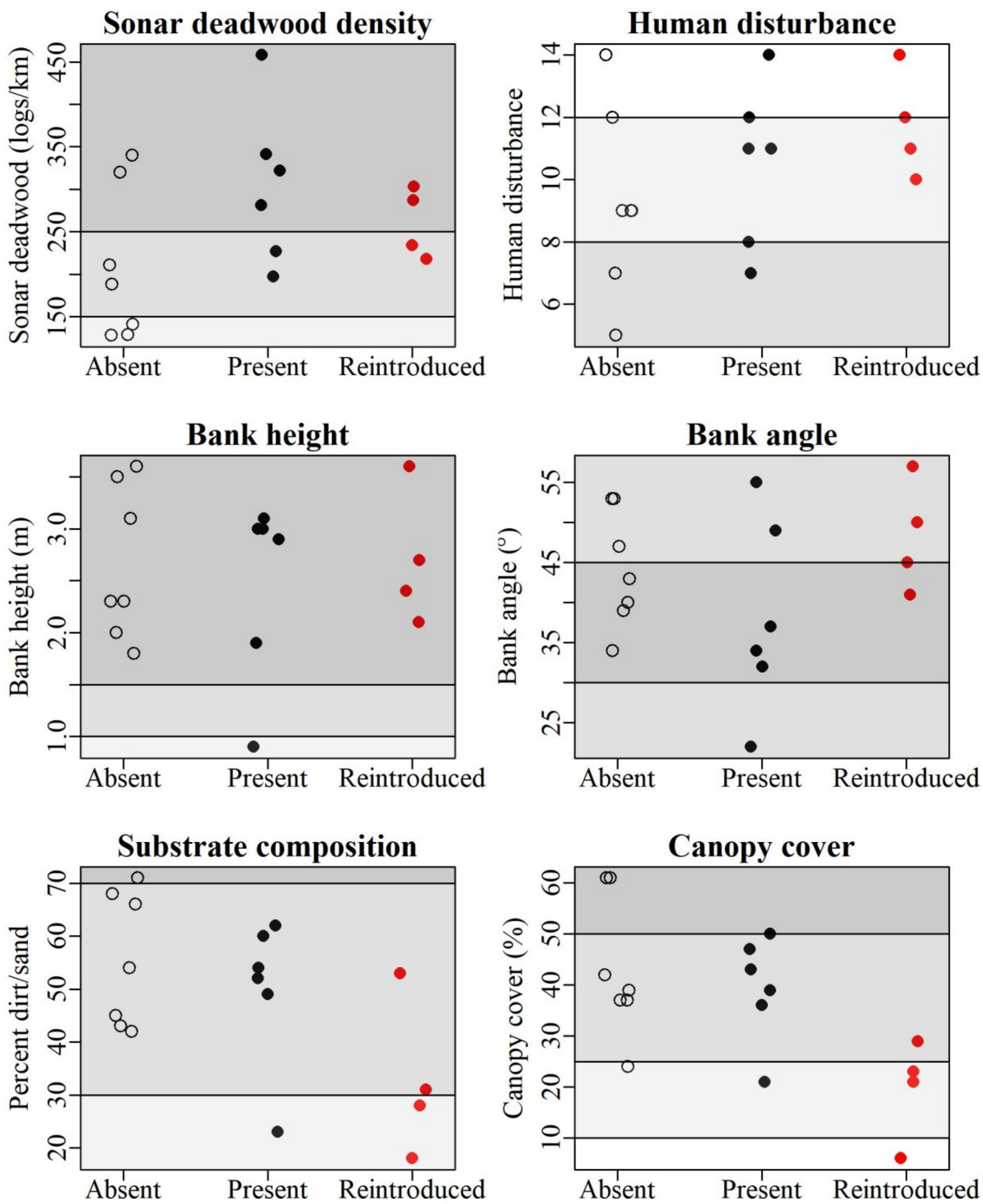


Figure 5. Strip chart distributions for averages of habitat attributes among sites where alligator snapping turtles were absent, present, and reintroduced in Kansas, Oklahoma, and Mississippi, USA. Shading indicates point values awarded for the habitat variable where dark gray corresponds to a score of 3; point values decline by one with each lighter shading.

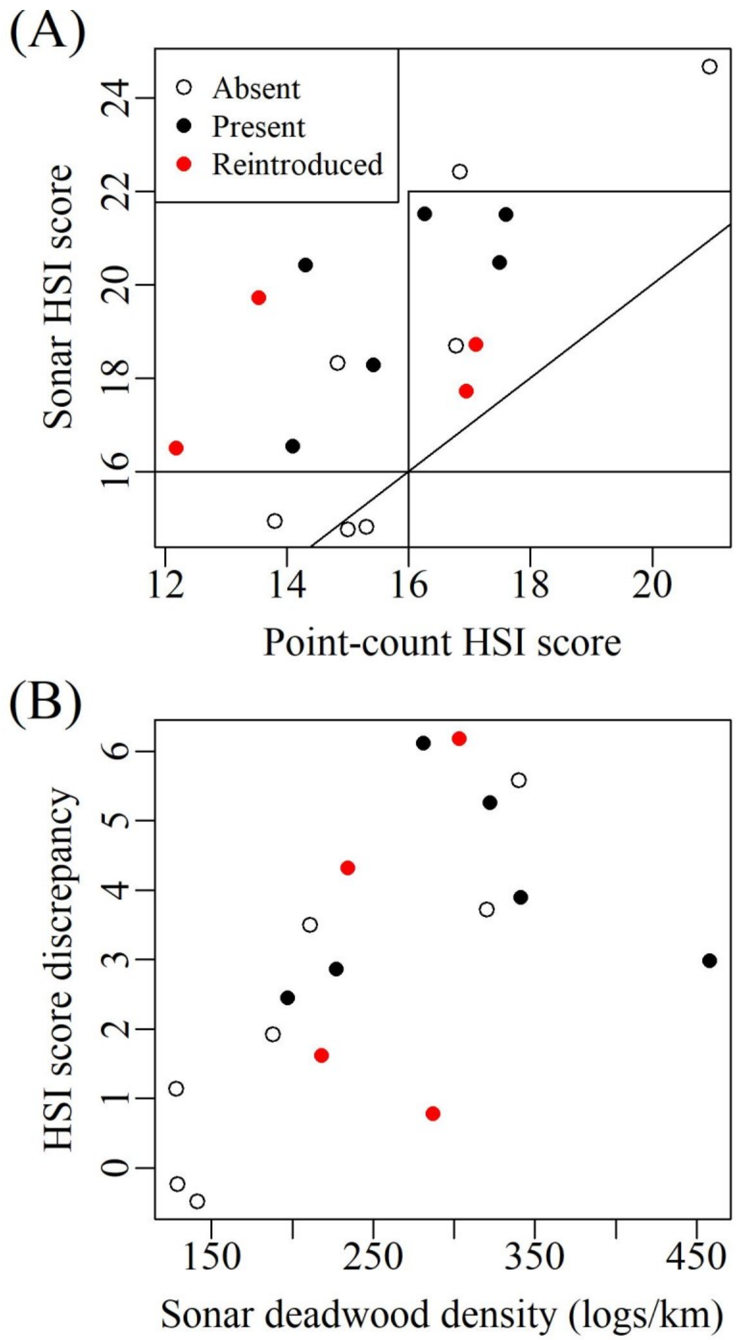


Figure 6. (A) Comparison between habitat suitability index (HSI) scores calculated using point counts and sonar among 17 sites in Kansas, Oklahoma, and Mississippi, USA in which habitat suitability for the alligator snapping turtle was assessed in 2018 and 2019. Data points above the line are sites where the sonar HSI score was greater than the point-count HSI score. Points outside the boxes are sites where the rating resulting from the scores were different between the two methods. (B) The relationship between sonar deadwood density and the discrepancy between point-count HSI scores and sonar HSI scores.

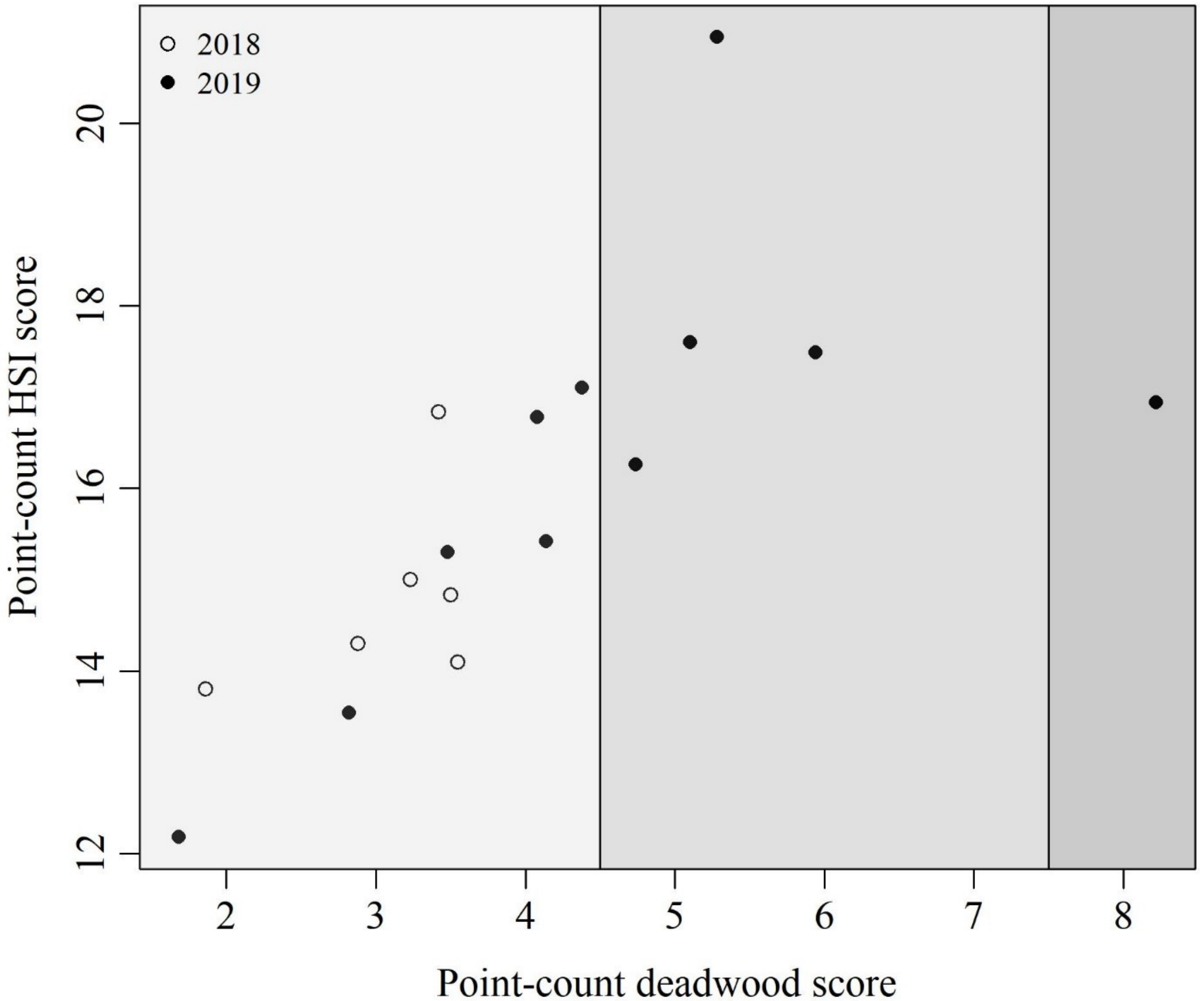


Figure 7. The relationship between scores assigned for point-count deadwood density and the final point-count habitat suitability index (HSI) score among 17 sites in which habitat suitability for the alligator snapping turtle was assessed. Open and black points correspond to point-count field methods applied in 2018 (10-m radius) and in 2019 (20-m upstream and downstream and within 10-m of the bank) respectively. Sites surveyed in 2018 were located in Oklahoma, and sites surveyed in 2019 were located in Kansas, Oklahoma, and Mississippi, USA. Shading indicates point values awarded for deadwood density where dark gray corresponds to high density; density declines with lighter shading.

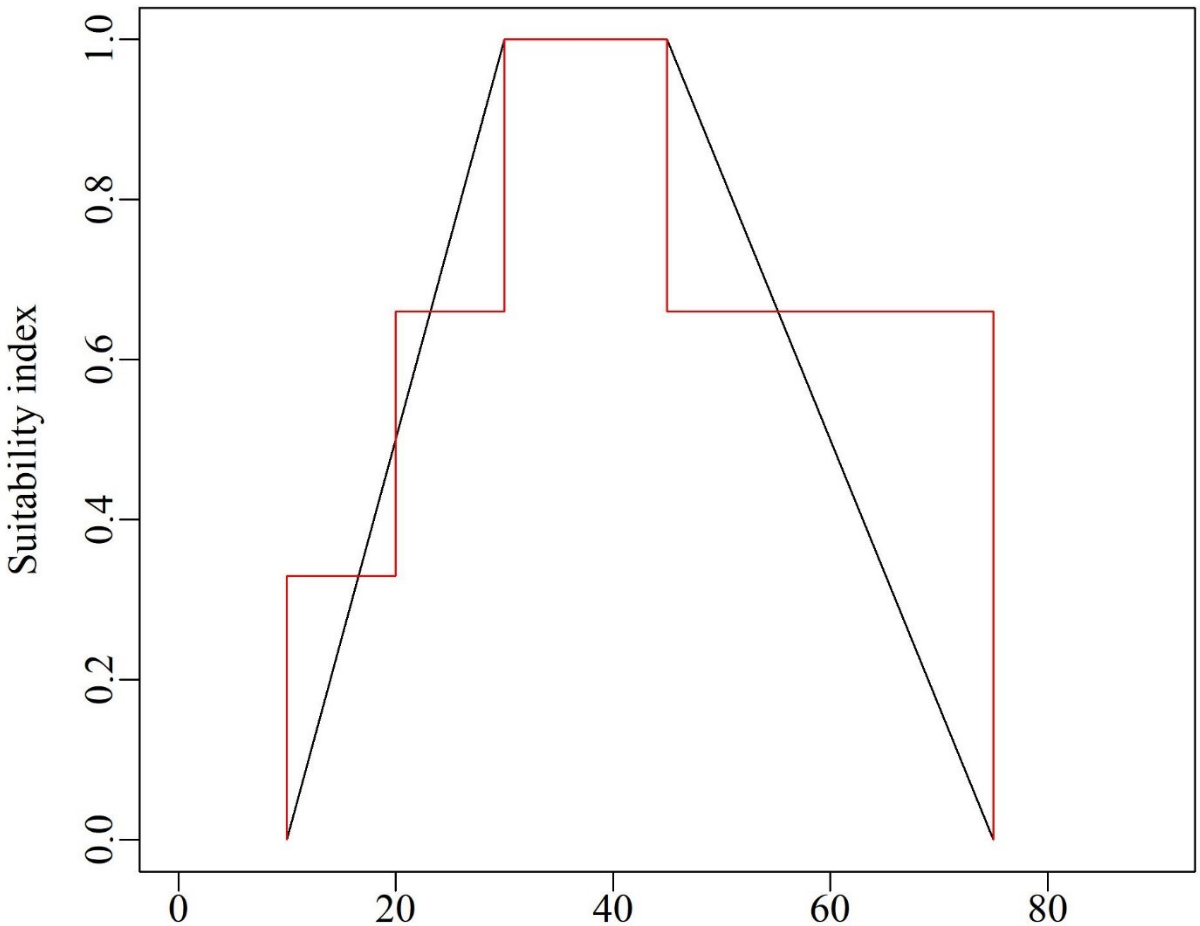


Figure 8. The comparison between a continuous suitability index (black) like commonly used in other studies and a step-wise suitability index (red) as used in this study.

Escatawpa River

Old River Channel

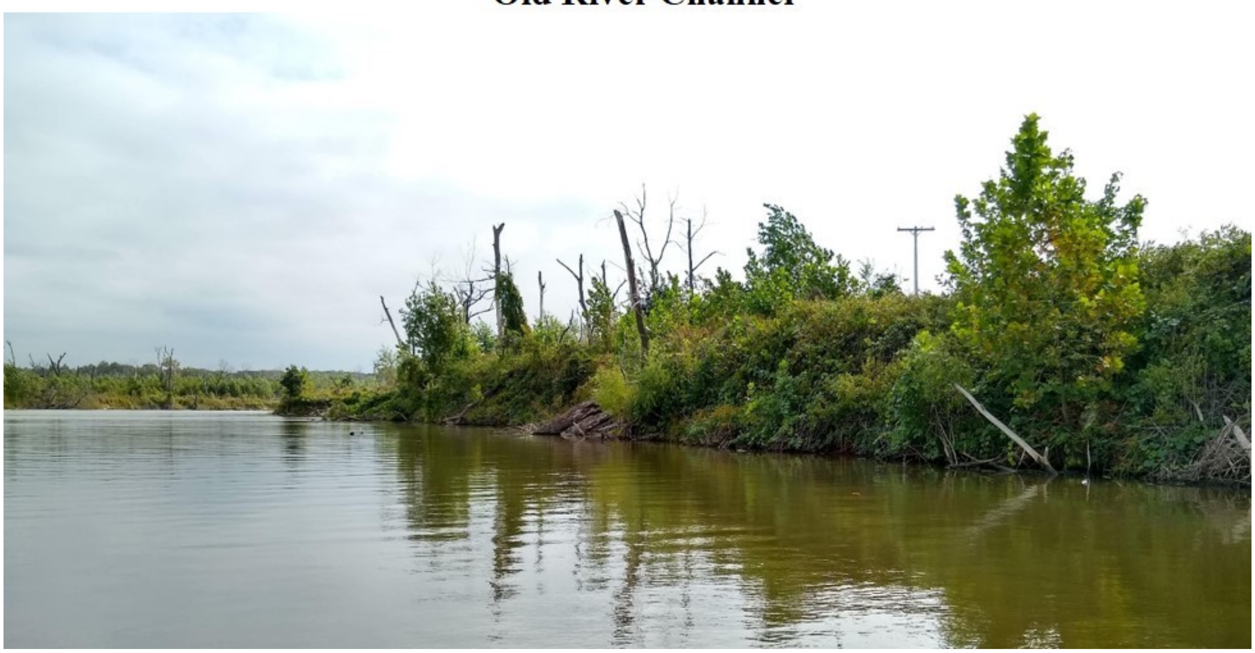


Figure 9. Examples of sites with lower habitat suitability scores than expected for high deadwood density. The Escatawpa River lacked tall and steep banks and the Old River Channel lacked canopy cover due to severe flooding in 2015 and banks were covered in dense vegetation.

SUMMARY

Surveys have been conducted throughout the range of the Alligator Snapping Turtle (*Macrochelys temminckii*) to determine the present status of this declining reptile. Although the boundaries of the species range have only contracted slightly, populations range-wide tend to be smaller and fewer in number than was the case historically (Riedle et al. 2005). Population declines are most pronounced in the northern periphery of the range (Shipman et al. 1995; Riedle et al. 2005; Shipman and Riedle 2008; Bluett et al. 2011; Lescher et al. 2013; Baxley et al. 2014). Reintroduction was identified to be a potential strategy to mitigate declines and restore populations into rivers and other suitable wetlands where the species is thought to be extirpated. The suitability of previous sites where reintroductions of Alligator Snapping Turtles have occurred was determined by researchers with years of experience in Alligator Snapping Turtle following detection surveys and a visual habitat assessment. Reintroductions were initiated following confirmation that a site did not currently support a population of Alligator Snapping Turtles and that suitable habitat remained. The results of my studies begin to define standardized methods that will facilitate the identification of suitable release sites for Alligator Snapping Turtles and remove the need for years of expertise about Alligator Snapping Turtle biology. First, I investigated the amount of effort needed to detect Alligator Snapping Turtles during surveys, then I compared methods to quantify submerged deadwood (a key habitat feature), and finally, I developed a standardized field survey and habitat suitability model that can be used to determine suitability of potential release sites.

In Chapter 1, I performed permutation analyses to investigate the effects of sampling effort on the probability of falsely concluding absence of Alligator Snapping Turtles during

surveys. Reintroductions are targeted in locations where Alligator Snapping Turtles historically occurred but have likely been extirpated. To confirm absence of a dense population, a potential reintroduction site must be surveyed prior to the release of turtles. According to my permutations, effort expended during detection surveys conducted in states within the northern periphery of the Alligator Snaping Turtle distribution may have been insufficient to detect the small populations that may remain in those states. In order to reduce the probability of falsely concluding absence of Alligator Snaping Turtle at a potential reintroduction site, I recommend trapping for at least 100 net nights in order to detect populations that result in a catch-per-unit-effort (CPUE) of 0.02 or more turtles per net night in ideal trapping conditions. To detect populations smaller than those resulting in a CPUE of 0.02, upwards to 190 NN are required.

In Chapter 2, I compared two methods that could be used to quantify submerged deadwood density at potential reintroduction sites. One method involved counting the pieces of submerged structure within a small radius around many sample points, which was compared to densities resulting from recreational side-scan sonar surveys. Field point count surveys were not as accurate as estimating deadwood density with sonar; however, this method may be used if necessary, following a transformation described in this chapter. Sonar was found to be most useful in quantifying submerged deadwood density within long stretches of many rivers because it provides the most exhaustive measure of deadwood. Methods were described in this chapter to decrease time expenditure to process sonar images while still estimating deadwood density within 15% error. Based upon my results, I recommend dividing sonar data into segments of 40 meters and counting pieces of deadwood within a random selection of segments comprising 40%–60% of the river's length.

Finally, in Chapter 3, I described a standardized field surveyed that was used to characterize habitat at sites where Alligator Snapping Turtles were absent, present, or reintroduced. I developed a habitat suitability model that translated raw habitat data into a score that could be used to assess suitability of habitat present at each site. Scores of the 17 sites in which I surveyed ranged from Fair to Excellent habitat suitability when using sonar to measure deadwood density, and I identified three potential sites where Alligator Snapping Turtles are thought to be absent as potentially suitable reintroduction sites. The habitat suitability model accurately predicted rivers with current or reintroduced populations of Alligator Snapping Turtles as having suitable habitat, and deadwood density measured with sonar and level of human disturbance were the main drivers of scores. To further confirm the accuracy of the model, sites expected to have poor habitat suitability should be surveyed.

The application of these methods will aid in future recovery of the Alligator Snapping Turtle throughout its range. By following methods presented in my thesis, researchers can confirm absence of Alligator Snapping Turtles more confidently, conduct habitat surveys and thoroughly measure key habitat features, and determine habitat suitability of potential reintroduction sites more easily.

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APPENDICES

Appendix A

Completion Date 18-Jan-2018
Expiration Date 17-Jan-2021
Record ID 25850049

This is to certify that:

Kameron Voves

Has completed the following CITI Program course:

Not valid for renewal of certification through CME. Do not use for TransCelerate mutual recognition (see Completion Report).

Animal Care and Use (Curriculum Group)
Researcher - Wildlife (Course Learner Group)
1 - Basic Course (Stage)

Under requirements set by:

Missouri State University

CITI Collaborative Institutional Training Initiative

Verify at www.citiprogram.org/verify/?w29732a91-fe70-4081-8559-498858771b5a-25850049

Completion Date 22-Jan-2018
Expiration Date N/A
Record ID 25850050

This is to certify that:

Kameron Voves

Has completed the following CITI Program course:

Not valid for renewal of certification through CME. Do not use for TransCelerate mutual recognition (see Completion Report).

Working With Fish in Research Settings (Curriculum Group)
Working with Fish (Course Learner Group)
1 - Lab Animal Research (Stage)

Under requirements set by:

Missouri State University

CITI Collaborative Institutional Training Initiative

Verify at www.citiprogram.org/verify/?wd6388707-a8a6-40e2-b875-2ddb5629fd03-25850050

Completion Date 22-Jan-2018
Expiration Date N/A
Record ID 25850050

This is to certify that:

Kameron Voves

Has completed the following CITI Program course:

Not valid for renewal of certification through CME. Do not use for TransCelerate mutual recognition (see Completion Report).

Working With Fish in Research Settings (Curriculum Group)
Working with Fish (Course Learner Group)
1 - Lab Animal Research (Stage)

Under requirements set by:

Missouri State University

CITI Collaborative Institutional Training Initiative

Verify at www.citiprogram.org/verify/?wd6388707-a8a6-40e2-b875-2ddb5629fd03-25850050

Appendix A. Approved CITI research compliance training to work with wildlife, amphibians, and fish.

Appendix B

Appendix B-1. Length of study reach, number of species, number of net nights (effort), catch per unit effort (CPUE), recapture rate (RR), Shannon Diversity Index (H'), and Equitability Index (E) for all river systems sampled in eastern Oklahoma in 2018. For H', higher values indicate higher relative diversity. Evenness values closer to 0 indicate disproportionate species abundance where value close to 1 signify equal abundance.

River	km	# Species	Effort	CPUE	RR	H'	E
Big Cabin	6.83	7	124	3.14	0.255	0.735	0.378
Chouteau	2.57	7	55	3.65	0.183	0.750	0.385
Neosho	3.58	3	49	2.00	0.059	0.892	0.811
Deep Fork	11.99	6	120	2.32	0.038	1.063	0.593
Poteau	10.96	6	127	2.26	0.130	0.598	0.334

Appendix B-2. Total number of individuals of each species captured on all river systems sampled in Oklahoma in 2018. Rivers are ordered from north to south. Species codes are as follows: APSP = *Apalone spinifera*, CHSE = *Chelydra serpentina*, GROU = *Graptemys ouachitensis*, GRPS = *G. pseudogeographica*, MATE = *Macrochelys temminckii,* PSCO = *Pseudemys concinna*, *STOD* = *Sternotherus odoratus,* and TRSC = *Trachemys scripta.*

River	APSP	CHSE	GROU	GRPS	MATE	PSCO	STOD	TRSC	Total
Big Cabin	25	13	24	5	3	1	-	318	389
Chouteau	15	2	16	1	-	2	3	162	201
Neosho	8	-	34	-	-	-	-	56	98
Deep Fork	27	-	166	7	-	2	3	73	278
Poteau—Arkoma	7	-	13	1	17	3	-	246	287

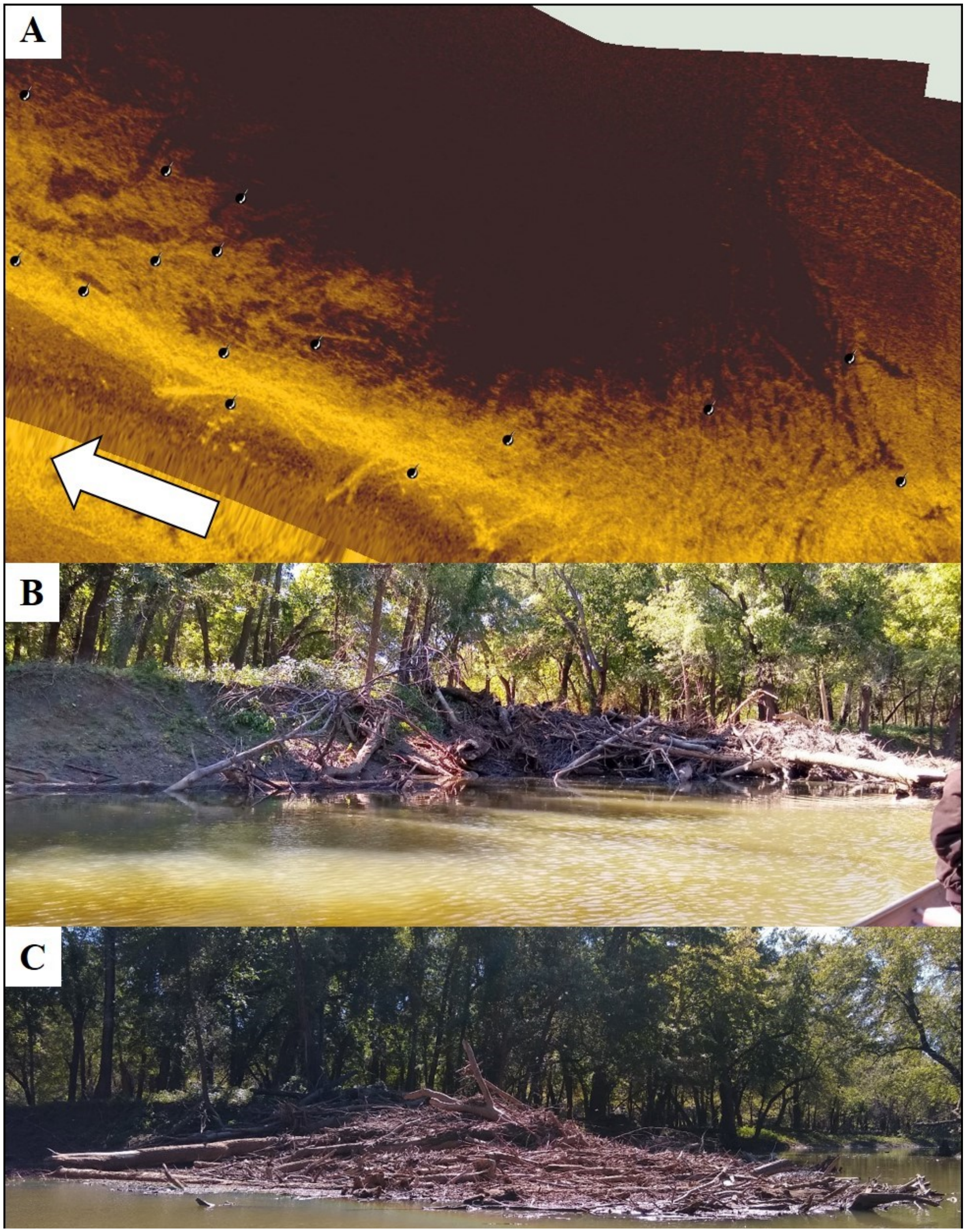
Appendix B-3. Morphometric data of all Alligator Snapping Turtles captured in Oklahoma in 2018 including carapace length (CL) and plastron length (PL).

River	CL (mm)	PL (mm)	Mass (g)	Sex	PIT Tag ID
Big Cabin Creek	457.0	357.0	24000	F	900118001212018
Big Cabin Creek	258.0	194.8	6950	J	900118001210575
Big Cabin Creek	282.2	218.1	5500	J	900254001568713
Poteau—Arkoma	411.9	319.8	15600	F	900254001568985
Poteau—Arkoma	421.0	329.0	18500	F	900254000252077
Poteau—Arkoma	428.0	325.5	18500	M	900254001560792
Poteau—Arkoma	164.0	130.5	1040	J	900254001398167
Poteau—Arkoma	178.9	138.8	1250	J	900254000254998
Poteau—Arkoma	202.0	151.6	1875	J	900254000289277
Poteau—Arkoma	215.2	165.0	2220	J	900254001569174
Poteau—Arkoma	219.0	172.0	2570	J	900254001578588
Poteau—Arkoma	233.0	184.2	2770	J	900254000258452
Poteau—Arkoma	238.0[1]	191.0	2115	J	900254000317736
Poteau—Arkoma	253.2	177.1	2930	J	900254000303757
Poteau—Arkoma	254.6	197.2	3845	J	900254001583727
Poteau—Arkoma	303.1	236.0	6750	J	900254001585518
Poteau—Arkoma	307.0	233.2	6150	J	90025400314193
Poteau—Arkoma	318.9	241.0	6670	J	900254001571246
Poteau—Arkoma	330.2	256.0	8470	J	900254001582285
Poteau—Arkoma	331.1	255.0	8470	J	900254000291816

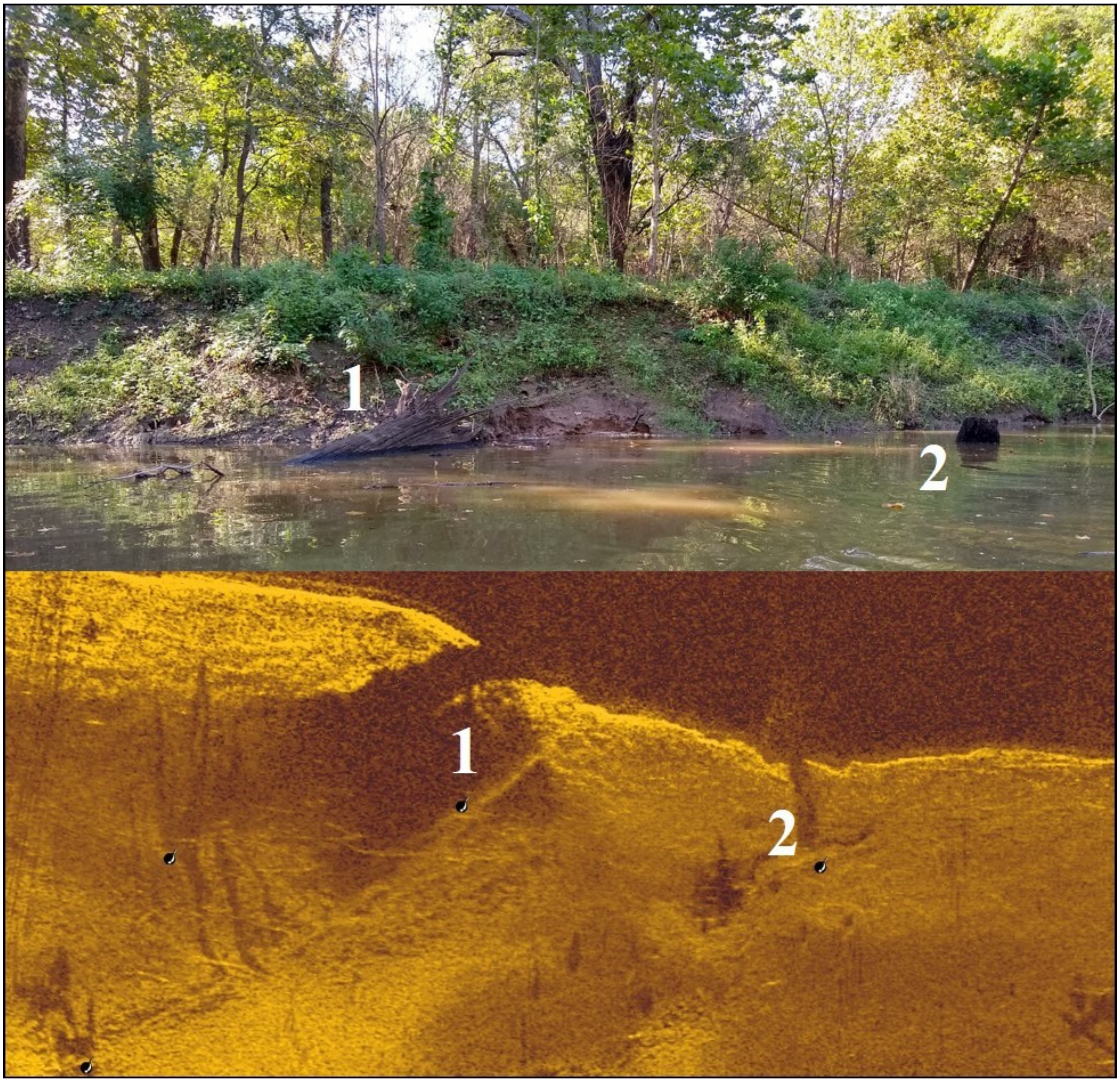
[1]Measurement affected by shell damage.

Appendix C.

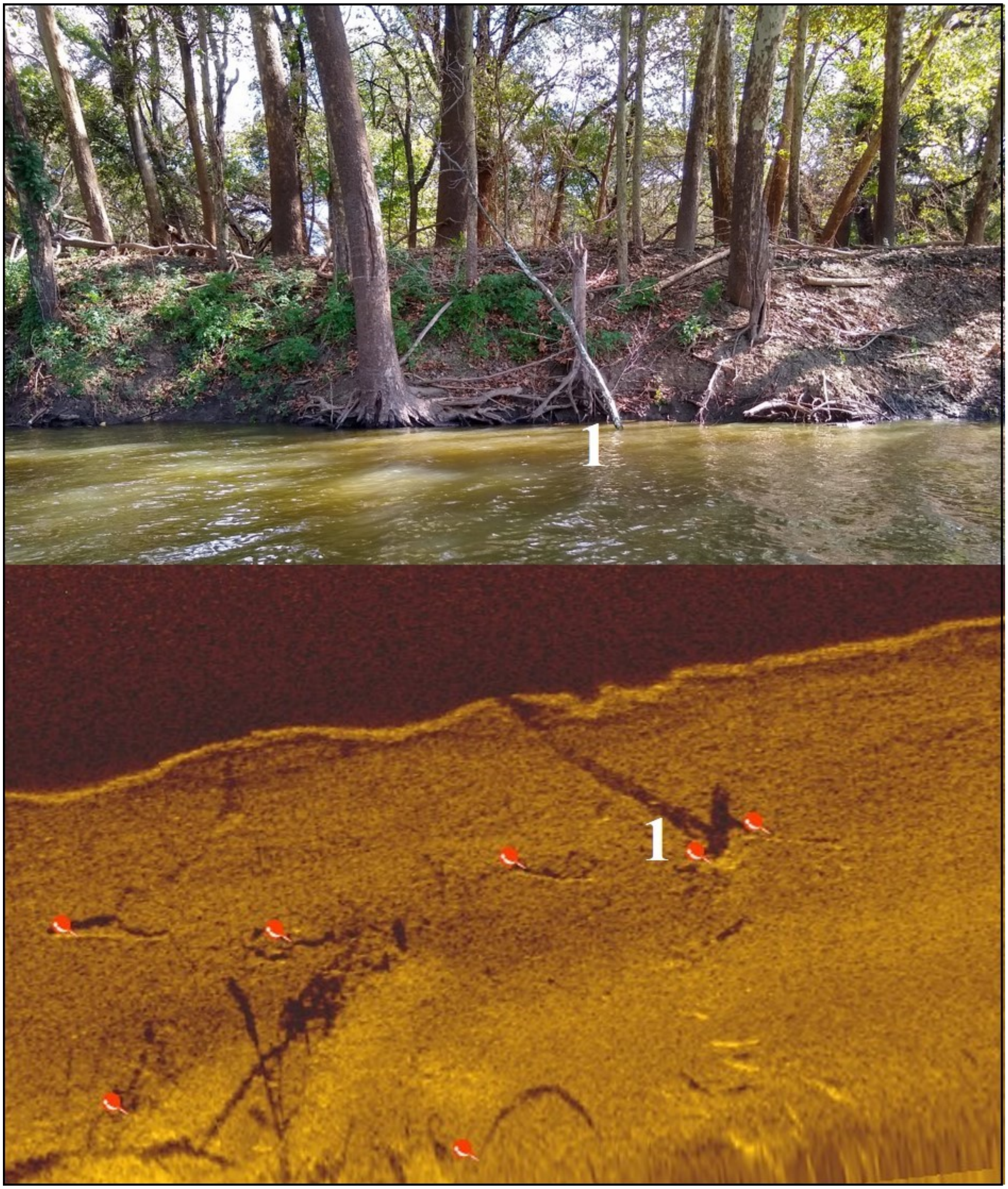
Appendix C-1. The side-scan sonar was mounted to the front of the boat and powered by a marine battery. The receiver was oriented parallel within the water column and lifted and lowered as needed.



Appendix C-2. A large pile of debris seen on sonar and in the field at Big Creek, OK. This example demonstrates the difficulty to differentiate individual logs in large accumulations. A) an example of shadows cast on sonar by a large pile of debris. Arrow shows direction of flow, B) view of debris from downstream, and C) view of debris from upstream. Black pins mark pieces of structure included in the total deadwood density estimation.



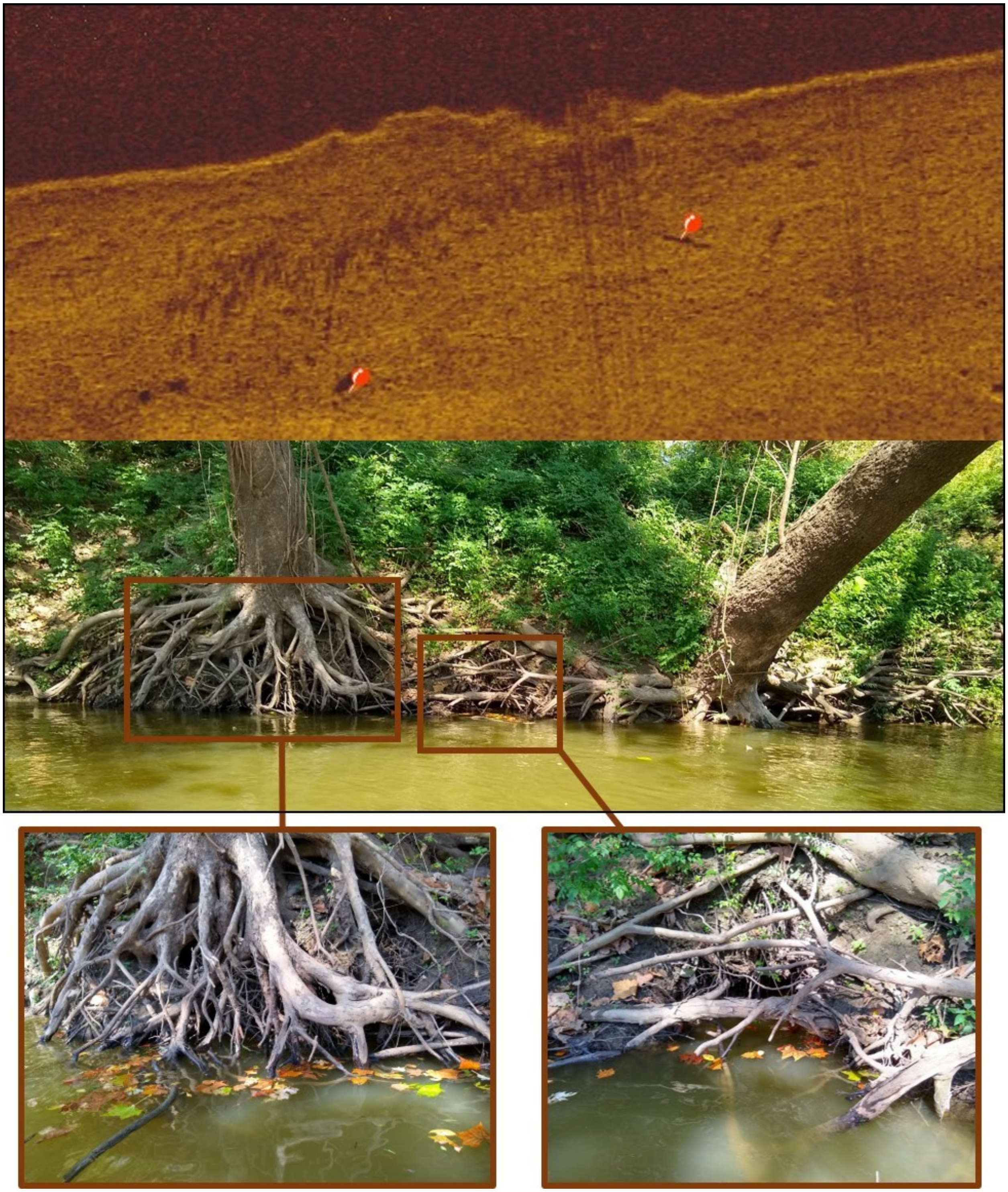
Appendix C-3. An example of the emergent structure surrounding a capture location of an Alligator Snapping Turtle on Pond Creek and the corresponding sonar image. Numbers indicate pieces of structure that can be seen in both images. Black pins mark pieces of structure included in the total deadwood density estimation.



Appendix C-4. An example of the emergent structure surrounding a capture location of an Alligator Snapping Turtle on the Caney River and the corresponding sonar image. Numbers indicate pieces of structure that can be seen in both images. This example demonstrates abundant submergent structure in the absence of emergent structure. Orange pins mark pieces of structure included in the total deadwood density estimation.

Appendix C-5. An example of the emergent structure surrounding a capture location of an Alligator Snapping Turtle on the Caney River and the corresponding sonar image. Numbers indicate pieces of structure that can be seen in all three images. Emergent structure seen from two angles demonstrates how difficult it can be to determine the number of discrete logs from emergent branches. Orange pins mark pieces of structure included in the total deadwood density estimation.

Appendix C-6. An example of the emergent structure surrounding a capture location of an Alligator Snapping Turtle on the Caney River and the corresponding sonar image. Numbers indicate pieces of structure that can be seen in all three images. Bottom image demonstrates the poor quality that can result from imperfect conditions during sonar recording. Orange pins mark pieces of structure included in the total deadwood density estimation.



Appendix C-7. An example of a capture location of an Alligator Snapping Turtle on the Caney River and the corresponding sonar image. Orange pins mark pieces of structure included in the total deadwood density estimation. This example demonstrates an instance of a turtle using a non-log structure type for cover rather than abundant submerged deadwood. Bottom images show prominent undercut of a potential beaver den.

Appendix D

Package 'logs'
Unofficial Documentation
April 7, 2020

Title Submerged Log Quantification and Sub-sampling Package
R Version 3.5.0 "Joy in Playing"
Author Kameron Voves
Packages required 'sp', 'rgeos', 'GISTools', 'sf', 'smoothr'
Description Methods to create evenly spaced transects along rivers, sub-sample rivers, and quantify submerged log density in different ways using gps locations of logs.

R topics documented:

---

logs-package   Submerged Log Quantification and Sub-sampling Package

---

Description

The **logs-package** provides tools to manipulate river shapefiles and xy coordinate information. The included functions were primarily created to determine processing effort required to estimate submerged log density from side-scan sonar data.

Details

While reviewing sonar mosaics, waypoints were created at locations of submerged logs. Coordinates of all submerged structure were exported as a csv file. Shoreline shapefiles were created by manually tracing the boundary of the river using sonar imagery as a guide. This procedure was completed in QGIS Version 3.4.3-Madeira (QGIS Development Team 2018) and exported as a shapefile. River mid-line tracks were obtained either in the field with a handheld GPS, from sonar recordings, or by manual creation.

The following projections are used throughout the package.

```
usgs<-("+proj=aea +datum=NAD83 +lat_1=29.5 +lat_2=45.5 +lat_0=23 +lon_0=-96
        +ellps=GRS80 +x_0=0 +y_0=0")
WGS84<-("+proj=longlat +datum=WGS84")
CRS.usgs<-CRS("+proj=aea +datum=NAD83 +lat_1=29.5 +lat_2=45.5 +lat_0=23
              +lon_0=-96 +ellps=GRS80 +x_0=0 +y_0=0")
crswgs84<-CRS("+proj=longlat +ellps=WGS84 +datum=WGS84 +no_defs")
```

References

QGIS Development Team. 2018 QGIS Geographic Information System. Open Source Geospatial Foundation Project. http://qgis.osgeo.org

evenspace	Create evenly spaced points along a line

Description

The function computes evenly spaced points along a set of xy coordinates for easy sampling of meandering streams and rivers.

Usage

evenspace(xy, sep, start = 0)

Arguments

xy	A matrix of x and y coordinates.
sep	Distance between generated points. See Details for units.
start	Position along the line in which to begin the function.

Details

The units are defined by the coordinate projection system. The default projection is WGS84<- ("+proj=longlat +datum=WGS84"). Under this projection, the units are in meters.

Output Value

A matrix of coordinates defining computed points. This matrix can be used as argument in transect.

Author(s)

Barry Rowlingson (http://r-sig-geo.2731867.n2.nabble.com/how-to-generate-perpendicular-transects-along-a-line-feature-td7583710.html#a7583721)

Code

```
evenspace <- function(xy, sep, start = 0, size) {
 dx <- c(0, diff(xy[, 1]))
 dy <- c(0, diff(xy[, 2]))
 dseg <- sqrt(dx^2 + dy^2)
 dtotal <- cumsum(dseg)

 linelength <- sum(dseg)
 pos <- seq(start, linelength, by = sep)

 whichseg <- unlist(lapply(pos, function(x) {
  sum(dtotal <= x)
 }))

 pos <- data.frame(
  pos = pos, whichseg = whichseg,
  x0 = xy[whichseg, 1],
  y0 = xy[whichseg, 2],
  dseg = dseg[whichseg + 1],
  dtotal = dtotal[whichseg],
  x1 = xy[whichseg + 1, 1],
  y1 = xy[whichseg + 1, 2]
 )

 pos$further <- pos$pos - pos$dtotal
```

```
  pos$f <- pos$further / pos$dseg
  pos$x <- pos$x0 + pos$f * (pos$x1 - pos$x0)
  pos$y <- pos$y0 + pos$f * (pos$y1 - pos$y0)

  pos$theta <- atan2(pos$y0 - pos$y1, pos$x0 - pos$x1)

  return(pos[, c("x", "y", "x0", "y0", "x1", "y1", "theta")])
}
```

---

transect — Create transects perpendicular to a line

Description

The function places transects perpendicular to a line centered at coordinates computed by evenspace.

Usage

transect (tpts, tlen)

Arguments

tpts	A matrix of coordinate information; the output of evenspace.
tlen	The length of the transect on either side of the point. See Details for units.

Details

The units are defined by the coordinate projection system. The default projection is WGS84<- ("+proj=longlat +datum=WGS84"). Under this projection, the units are in meters.

Output Value

A data frame of coordinates defining computed transects.

Author(s)

Barry Rowlingson (http://r-sig-geo.2731867.n2.nabble.com/how-to-generate-perpendicular-transects-along-a-line-feature-td7583710.html#a7583721)

Code

```
transect <- function(tpts, tlen) {
  tpts$thetaT <- tpts$theta + pi / 2
  dx <- tlen * cos(tpts$thetaT)
  dy <- tlen * sin(tpts$thetaT)
  return(
    data.frame(
      x0 = tpts$x + dx,
      y0 = tpts$y + dy,
      x1 = tpts$x - dx,
      y1 = tpts$y - dy
    )
  )
}
```

---

river.sections — Divide a river shapefile into sections of equal length

---

Description

The function uses transects created from evenspace and transect to divide a meandering river into sections of equal mid-line length.

Usage

river.sections (track, shoreline, n, width=100)

Arguments

track	A Formal Class SpatialPolygonsDataFrame defining the mid-line of the river.
shoreline	A Formal Class SpatialPolygonsDataFrame defining the outer boundary of the river.
n	The number of sections in which to divide the river.
width	The length of the generated transects; equal to the maximum width of the river. See details for units.

Details

The units are defined by the coordinate projection system. Input spatial data are projected in WGS84<- ("+proj=longlat +datum=WGS84"). Under this projection, the units are in meters.

Before used as an argument in this function, the track spatial polygon may need to be smoothed using 'smooth' in package{smoothr}. This function reduces the number of points forming the line, thereby eliminating jagged edges. Also, depending on the nature of the shoreline polygon, the number of sections resulting from the function may exceed the number specified by 'n'. A quality check is required to ensure the function performed properly before the function output can be used further. See example.

Output Value

A Formal Class SpatialPolygons containing individual polygons resulting from the function. The output can be used as an argument in section.count.

Author(s)

Kameron Voves

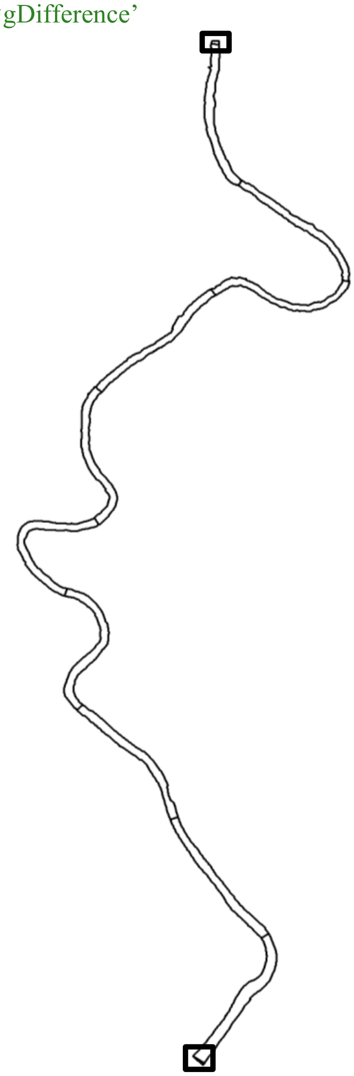
Code

```
river.sections<-function(track, shoreline, n, width=100){

 #Project shapefiles
 shoreline.proj<-spTransform(shoreline,CRS.usgs)
 track.proj<-spTransform(track,CRS.usgs)

  #Remove any part of the track that falls outside shoreline shapefile
 track<-gIntersection(track.proj, shoreline.proj)

 #Determine where to divide the shapefile
 length<-gLength(track)/(n+0.000001) #Determine length of n segments in meters
 xy<-cbind(track@lines[[1]]@Lines[[1]]@coords[, 1],
        track@lines[[1]]@Lines[[1]]@coords[, 2]) #Extract xy coordinates from track
 tspts<-evenspace(xy, length) #Generate evenly spaced points along track
 tslines<-transect(tspts,width) #Create transects at those points

 #Make transects a SpatialLines object type
 begin.coord<-data.frame(lon=tslines$x0, lat=tslines$y0)
 end.coord<-data.frame(lon=tslines$x1, lat=tslines$y1)
 #Create list of simple feature geometries (linestrings)
 l_sf <- vector("list", nrow(begin.coord))
```

```
for (i in seq_along(l_sf)){
  l_sf[[i]] <- st_linestring(as.matrix(rbind(begin.coord[i, ], end.coord[i,])))
}
#Create simple feature geometry list column
l_sfc <- st_sfc(l_sf, crs = usgs)
## Convert to `sp` object
lines_sp <- as(l_sfc, "Spatial")
lines.sp.proj<-spTransform(lines_sp, CRS.usgs)

#Split shoreline polygon using transects
 lpi<-gIntersection(shoreline.proj, lines.sp.proj)   #Intersect your line with the polygon
blpi <- gBuffer(lpi, width = 0.000001)  #Create a very thin polygon buffer of the intersected line
dpi <- gDifference(shoreline.proj, blpi)   #Split shoreline polygon using 'gDifference'

return(dpi)

}
```

Example

```
#Project track and smooth
track.proj<-spTransform(track, CRS.usgs)
track.smooth<-smooth(track.proj, method="ksmooth", smoothness=5)

##Quality check
plot(river.sections(track.smooth, shoreline, n=10, width=50))

# Enclosed in boxes to the right are tiny "dummy" polygons included
in the function output. Their creation is in excess of the number of
sections specified as an argument and is a bug of the function in its
current state.
```



section.count	Count the number of points in polygons

Description

The function counts the number of logs in multiple polygons.

Usage

section.count (sections, logs)

Arguments

sections — A Formal Class SpatialPolygons containing individual polygons; the output of river.sections.

logs — A csv file containing xy coordinates of log locations within the river.

Details

The function will count logs inside "dummy" sections created by river.sections. These rows of the output will need to be removed or combined with adjacent rows before output can be used further. The output can be used as an argument in processing.effort.

The function can be modified to use a Formal Class SpatialPoints object type as an argument in place of a csv by removing the code lines that project the log shapefile. After modification, the output of shore.logs can be used as an argument.

Output Value

A matrix with rows equal to the number of polygons present in the sections argument. Each row of the matrix represents the number of logs inside each polygon.

Author(s)

Kameron Voves

Code

```
section.count<-function(sections, logs){
 #Project shapefiles
 logs.sp<-SpatialPoints(logs [,c(2,1)], CRS(WGS84))
 logs.proj<-spTransform(logs.sp, CRS.usgs)

 #Create storage matrix
 result<-matrix(NA,nrow=length(section@polygons[[1]]@Polygons), ncol=1)

 #Extract each polygon and count logs
 for(i in 1:length(sections@polygons[[1]]@Polygons)){
  #Make matrix with xy coordinates
  pud.x<-sections@polygons[[1]]@Polygons[[i]]@coords[,1]
  pud.y<-sections@polygons[[1]]@Polygons[[i]]@coords[,2]
  pud<-cbind(pud.x, pud.y)

  #Create projected polygon
  pud.proj<-SpatialPolygons(list(Polygons(list(Polygon(pud)),1)), proj4string = CRS.usgs)

  #Count logs in section
  result[i]<-poly.counts(logs.proj, pud.proj)
 }
 return(result)
}
```

processing.effort	Determine minimum effort needed to quantify log density

Description

The function is used to determine the minimum amount of river required in a sub-sample in order to accurately estimate true log density (as determined if logs were counted along the whole river).

Usage

processing.effort (section.logs, track, shoreline, precision=0.2, iter=100, p=0.05)

Arguments

section.logs	A matrix of log abundance in each section of a divided river; the output of section.count.
track	A Formal Class SpatialPolygonsDataFrame defining the mid-line of the river.
shoreline	A Formal Class SpatialPolygonsDataFrame defining the outer boundary of the river.
precision	The desired error in which density estimations can differ from the true density.
iter	The number of random samples to perform.
p	The desired proportion of samples that deviate from the specified precision.

Details

First, the function draws random samples from the log abundance matrix and log density is calculated each time. This is done for an increasing number of sections included in the sub-sample, starting with one section and increasing until all of the sections are included in the sample. The proportion between the estimated density resulting from each random sample and the actual density is calculated. The minimum number of sections needed in a sub-sample is then determined by the smallest sample size in which the specified proportion of the iterations fall within the designated precision level of the true density.

The track and shoreline shapefiles are used to determine the length of the sampled river which is then used to determine log density.

Output Value

A matrix that includes the length of the river, minimum number of sections required in a sub-sample, and the mean, standard deviation, minimum, and maximum log density resulting from the random samples.

Author(s)

Kameron Voves

Code

```
processing.effort<-function(section.logs, track, shoreline, precision=0.2, iter=100, p=0.05){
 #Project shapefiles
 shoreline.proj<-spTransform(shoreline,CRS.usgs)
 track.proj<-spTransform(track,CRS.usgs)

  #Remove any part of the track that falls outside shoreline shapefile
 track<-gIntersection(track.proj, shoreline.proj)

 #Repeat random sample of different sizes
 sample.size.test<-matrix(NA,nrow=iter, ncol=length(section.logs))
 for(k in 1:iter){
  for(i in 1:length(section.logs)){
   sample<-sample(section.logs, i, replace=FALSE)
   logs.mean<-mean(sample)
   #Calcuate density
   sample.size.test[k,i]<-round((logs.mean*1000)/(gLength(track)/length(section.logs)),digits=0)
  }
 }
```

```
 #Calculate the difference between the random sample log density and actual log density
 sample.size.difference<-matrix(NA,nrow=iter, ncol=length(section.logs))
 sample.size.difference.abs<-matrix(NA,nrow=iter, ncol=length(section.logs))
 for(i in 1:iter){
  #Calculate actual density
  actual<-sum(section.logs)/(gLength(track)/1000)
  #Calculate proportion (sample/actual)
  sample.size.difference[i,]<-sample.size.test[i,]/actual
  #Calculate the proportion difference (1-sample/actual)
  sample.size.difference.abs[i,]<-abs((sample.size.test[i,]/actual)-1)
 }

 #Which sample size has less than 5% of iterations within 20% of the actual density?
 sample.size.threshold<-matrix(NA, nrow=1, ncol=length(section.logs))
 for(i in 1:length(section.logs)){
  sample.size.threshold[1,i]<-length(which(sample.size.difference.abs[,i]>precision))/iter
 }
 result<-length(section.logs)-length(which(sample.size.threshold[1,]<p))+1
 mean<-mean(sample.size.test[,result])
 sd<-sd(sample.size.test[,result])
 min<-min(sample.size.test[,result])
 max<-max(sample.size.test[,result])
 return(c(gLength(track),result, mean, sd, min, max,actual))
}
```

Example

```
#Create storage matrix
effort.sections<-matrix(NA, nrow=1, ncol=9)
colnames(effort.sections)<-c("River Length","Min Effort", "Density Mean",
                             "Density SD","Density Min","Density Max",
                             "Actual Density", "Logs Mean", "Logs SD")
#Run function
for(i in c(10,15,20,25,50,75,100)){
  #Divide river into different numbers of sections
 sections<-river.sections(track.smooth, shoreline, i, 50)
 #Count the number of logs per section for each division treatment
 section.logs<-section.count(sections, logs)
 #Fix dummy sections
 section.logs<-c((section.logs[1,]+section.logs[2,]), section.logs[3:i],
         (section.logs[i+1,]+section.logs[i+2,]))
 #Calculate mean and SD of the number of logs per section for each division treatment
 mean.logs<-mean(section.logs)
 sd.logs<-sd(section.logs)
 #Determine min processing effort for each division treatment
 sub.sample<-processing.effort(section.logs, track.smooth, shoreline, 0.15, 50)
 sub.sample<-c(sub.sample, mean.logs, sd.logs)
  #Store results in matrix for each division treatment
 effort.sections<-rbind(effort.sections, sub.sample)
}

#Remove empty row
effort.sections<-effort.sections[-1,]
#Label division treatment results
rownames(effort.sections)<-c(10,15,20,25,50,75,100)
```

	River Length	Min Effort	Density Mean	Density SD	Density Min	Density Max	Actual Density	Logs Mean	Logs SD
10	6222.125	6	320.18	21.23214	283	363	319.505	198.80000	47.39151
15	6222.125	7	319.40	23.63930	276	379	319.505	132.53333	34.84018
20	6222.125	9	316.28	23.81565	266	367	319.505	99.40000	29.09766
25	6222.125	10	319.80	24.81195	254	372	319.505	79.52000	25.46422
50	6222.125	19	320.10	25.55446	280	376	319.505	39.76000	17.34942
75	6222.125	25	323.70	28.73329	270	380	319.505	26.50667	12.57343
100	6222.125	29	319.02	23.27431	265	377	319.505	19.88000	10.07375

---

shore.logs Extract log locations that are along the shoreline

---

Description

The function creates a buffer inside a river's boundaries and extracts log locations that lie between the buffer and shoreline.

Usage

shore.logs(shoreline, logs, buffer=-10)

Arguments

shoreline	A Formal Class SpatialPolygonsDataFrame defining the outer boundary of the river.
logs	A csv file containing xy coordinates of log locations within the river.
buffer	The distance from the shore in which to extract log locations. See details for units.

Details

The units are defined by the coordinate projection system. The default projection is WGS84<- ("+proj=longlat +datum=WGS84"). Under this projection, the units are in meters.

The output can be used as an argument in a modified version of section.count.

Output Value

A Formal Class SpatialPoints object containing coordinates of log locations that lie outside of the buffer.

Author(s)

Kameron Voves

Code

```
shore.logs<-function(shoreline, logs, buffer=-10){
 #Project shapefiles
 shoreline.proj<-spTransform(shoreline,CRS.usgs)
 logs.sp<-SpatialPoints(logs[,c(2,1)], CRS(WGS84))
 logs.proj<-spTransform(logs.sp, CRS.usgs)
```

```
  #Create inner buffer
  buff<-gBuffer(shoreline.proj, width=buffer)

  #Find coordinates inside and outside of the buffer
  buff.logs<-point.in.polygon(logs.proj@coords[,1], logs.proj@coords[,2],
                              buff@polygons[[1]]@Polygons[[1]]@coords[,1],
                              buff@polygons[[1]]@Polygons[[1]]@coords[,2])

 #Extract coordinates that are outside buffer
 pud<-cbind(logs.proj@coords, buff.logs)
  shore.logs<-subset(pud, buff.logs==0)

  #Create SpatialPoints object type
  logs.sp<-SpatialPoints(shore.logs[,1:2], WGS84)
}
```

Appendix E

Appendix E-1. Photographs of Big Cabin Creek, in northeastern Oklahoma showing fairly narrow channel and abundant log and debris piles. Numerous pastures with grazing cattle occurred along the river stretch. Limestone banks were prominent on the downstream reach of the sampled area. Houses were also present along the entire river, often times with walkways to access the water.

Appendix E-2. Photographs of the Deep Fork River in eastern Oklahoma, showing fairly narrow channel with very tall, steep red clay banks and deep water near the river's edge. Woody debris was abundant but primarily towards the center of the river channel (farther than 10-m from the bank). Downstream, banks were shorter and more vegetated, and woody debris was closer to the banks. The upstream reach of the river was within the boundaries of Deep Fork National Wildlife Refuge and had multiple foot access areas.

Appendix E-3. Photographs of the Poteau River above Lake Wister in eastern Oklahoma, showing a moderately wide river channel. Banks were tall but heavily overgrown with dense vegetation. Bushes and trees often overhung into the water. Submerged logs occurred in patches in high abundance, but patches were interspersed with long stretches with little to no structure.

Appendix E-4. Photographs of the Poteau River near Arkoma, Oklahoma. The channel was moderately wide with heavily overgrown, tall banks. Water levels rose nightly with influx of water from the Arkansas River after water was released from Robert S. Kerr Reservoir. Counterintuitively, inundating water originated downstream of my study site. Logs and large boulders were numerous. Residential housing was common along the river.

Appendix E-5. Photographs of Chouteau Creek in northeastern Oklahoma, showing low water levels at the time of sampling. The channel was narrow and submerged and emergent structure were abundant.

Appendix E-6. Photographs of the main channel of the Neosho River in northeastern Oklahoma. The channel was very wide and lined with boat houses and marinas. The height of the banks varied, and some limestone banks were present. While some large pieces of woody debris were observed, vertical stumps comprised most of the submerged structure.

Appendix E-7. Photographs of the Chickasawhay River in southeastern Mississippi. This river was characterized by large gradually sloped sandbars on one side and tall, steep slopes on the other. The majority of submerged structure was condensed along the tall banks and consisted of large trees. The river was moderately wide and fast flowing.

Appendix E-8. Photographs of the Escatawpa River, a cypress lined river in southeastern Mississippi. Much of the submerged structure in this river was in the form of flooded cypress groves and aquatic vegetation, in addition to abundant woody debris. Both small and large sandbars were present and provided access to suitable nesting areas—as in bottom left. Banks not associated with sandbars, were lined with tree trunks and were unsuitable for nesting. In these cases, suitable nesting area was located several meters from the water and into the forest. Boathouses and sandbar camping were prevalent along the river.

Appendix E-9. Photographs of the Pascagoula River, in southeastern Mississippi. Much like the Chickasawhay, the Pascagoula River was characterized by high density of structure opposite large sandbars. The river channel was much wider than either the Chickasawhay River or the Escatawpa River. The banks were tall, steep, and mostly sandy, but some rocky outcrops were present. Stands of willows often overhung the water.

Appendix E-10. Photographs of the Pearl River in southeastern Mississippi. The river supported a variety of bank types such as tall sandy banks, sandbars, gravel bars, rocky outcrops, and hard clay banks. Sandbars were less prevalent than at other rivers in Mississippi. Some cypress and aquatic vegetation was seen, in addition to submerged woody structure.

Appendix E-11. Photographs of the Verdigris River above Lake Toronto, KS. Upstream, where the channel was narrow, canopy cover and density of submerged structure was high. Closer to the lake, the channel widened and became more open. Banks were tall throughout, and some rocky outcrops were present.

Appendix E-12. Photographs of the Fall River above Fall River Lake in southeastern Kansas. Rocks were abundant along this moderately wide river. Submerged logs were in low abundance, but banks were tall throughout. There were several primitive camping areas along the river.

Appendix E-13. Photographs of the Elk River above Elk City Lake, Kansas. Limestone outcrops were abundant in the upstream portion of the river. While submerged logs were moderately abundant, the Elk River had many examples of non-log type structure such as root balls and undercut banks. This river was heavily used by the public.

Appendix E-14. Photographs of Pond Creek in northern Oklahoma. At the confluence of Pond Creek with the Caney River, Pond Creek was wide with shallow sloped banks and low canopy cover. The floodplain was often inundated during times of high water. Canopy cover and bank height increased moving upstream. Above a large rocky outcrop, the river became too shallow to navigate by motorboat.

Appendix E-15. Photographs of the Caney River in northern Oklahoma. Just upstream of the Hulah Lake, the Caney was very wide with little canopy cover and isolated submerged structure. After the confluence with Pond Creek, the Caney narrowed and canopy cover, submerged structure, and prevalence of tall banks increased. Large piles of woody debris lined the river from recent floods.

Appendix E-16. Photographs of the Old River Channel. No longer a free-flowing river, the Old River Channel was disconnected from the main stem Washita River with dykes and was semi-separated into sections by shallow, vegetated areas or culverts below roads. The water channel was very wide and surrounded by dead or young trees. While submerged woody debris was plentiful, much of the structure was piles of small diameter brush. Much of the woody debris that was present likely resulted from two major flood events that occurred in 2015. Banks were tall and steep, but largely covered in debris or dense vegetation. Oil extraction activities were very prominent along the entire site.

Appendix E-17. Photographs of the Washita River in southern Oklahoma. Banks were tall, but heavily eroded. Submerged structure was moderately dense, and mostly in the form of large, whole trees.

Appendix F

SITE HABITAT MEASUREMENTS

SITE NAME____________________

DATE____________________

SIDE OF RIVER____________________

Circle characteristics present on river

Buildings/docks	Foot Paths	Pastures/ grazing	Crop fields	Oil wells	Active limb lines	Abandoned limb lines	# of Roads
Water cycling from dam release	Frequent flooding	Recreational use of the bank (Fishing/lounging)		Campgrounds	Rowing boats	Motorboat Tally	# Boat ramps

DISTANCE SAMPLED: DISTANCE BETWEEN PTS:		CHANNEL CHARACTERISTICS (TURTLE HABITAT)						BANK CHARACTERISTICS (NESTING HABITAT)									
		NON-LOG STRUCTURE (TYPE)	SUBMERGENT STRUCTURE (# LOGS)	CANOPY COVER				% SUBSTRATE								HEIGHT (m)	ANGLE
WAYPOINT	CHANNEL WIDTH (m)			U	D	L	R	S	D	R/G	C	HV	DV	L	W		

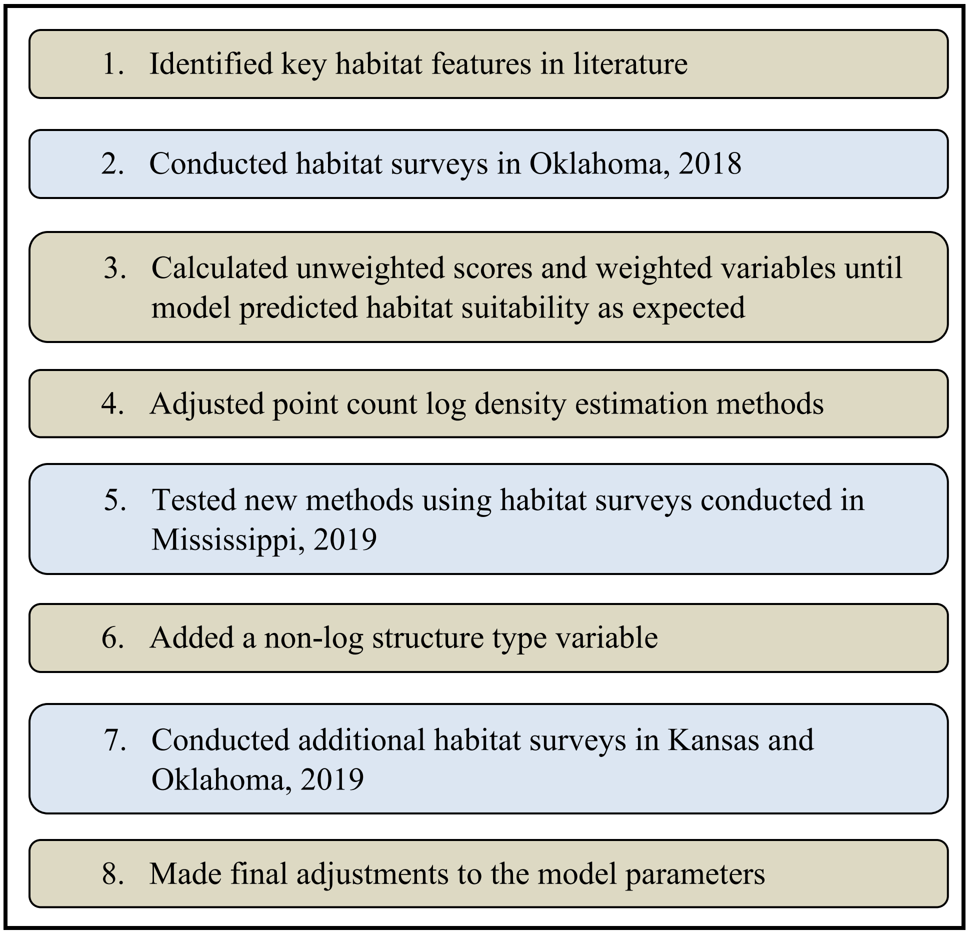
Measurements taken at 25 points on both sides of the stream
Distance: Distance between points is length of sampling area divided by 25

Channel Characteristics: taken 5-m from the bank
Non-log Structure: Types include **B**=beaver den, **R**=rocks or boulders, **V**=aquatic vegetation, **FV**= floodplain vegetation, **T**=tree rooted in bank with branches overhanging into water, **U**=undcercut bank, **RB**= root ball with no visible undercut, **M**=man-made structures (docks, bridge pylons, oil well pipes, artificial fish structures)
Submergent Structure: The number of emergent logs (whole logs/trees counts as 1) plus the number of submerged only logs, counted 20 m upstream and downstream of sample point and 10 m from bank
Overstory: taken in for directions, upstream, downstream, left bank, right bank. Full canopy cover=96 quadrats

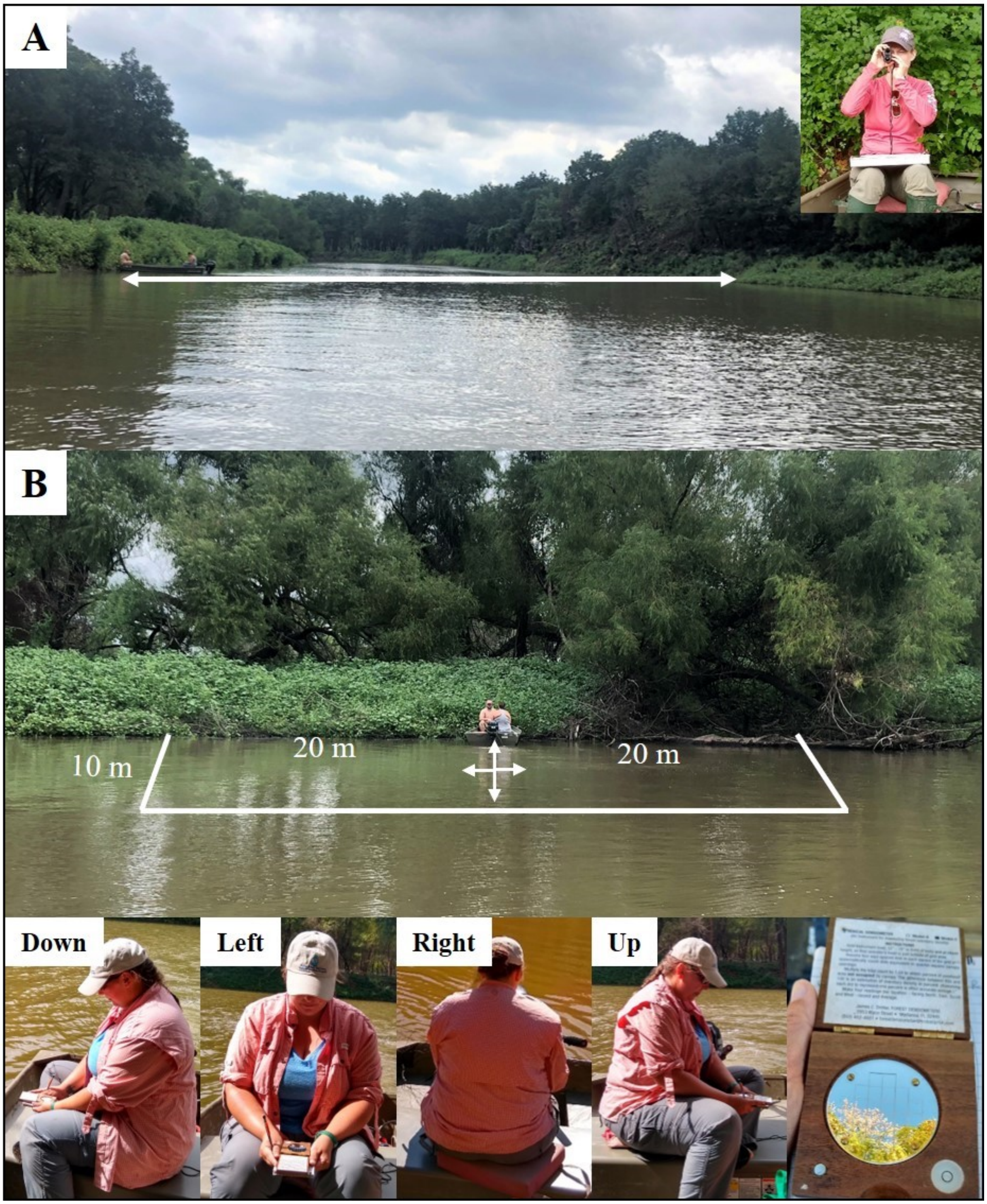
Bank Characteristics: taken perpendicular to river
% Substrate: Measured in a 1-m strip along the total height of the bank
S=sand, **D**=dirt, **R**=rock, **G**=gravel, **C**=clay,
HV=herbaceous vegetation/grasses, **DV**=dense vegetation,
L=leaf litter, **W**=woody debris/roots
Height: When would a turtle be done climbing the bank?
Angle: Place board on a representative slope of the bank

Appendix F. The datasheet used during habitat surveys.

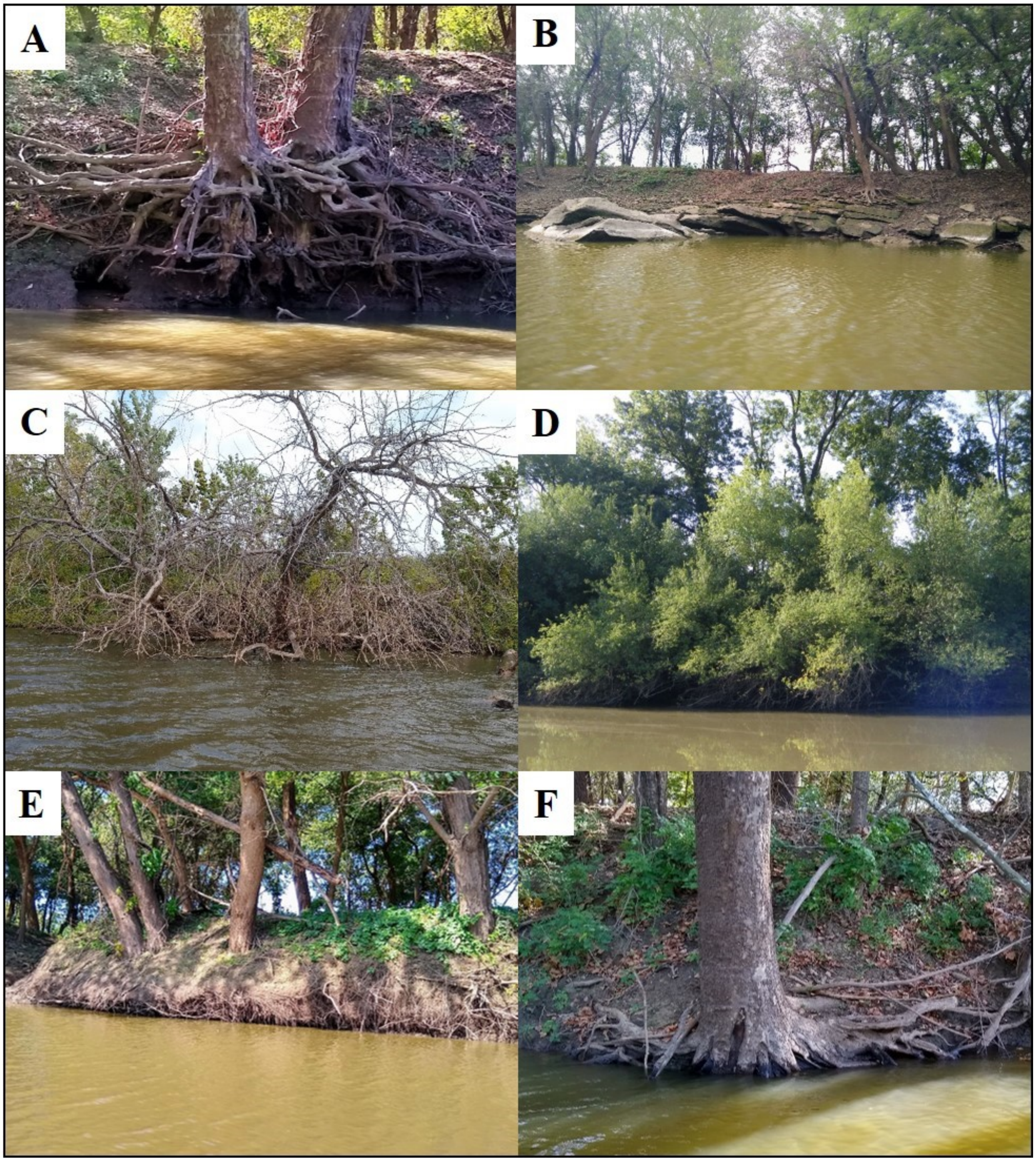
Appendix G



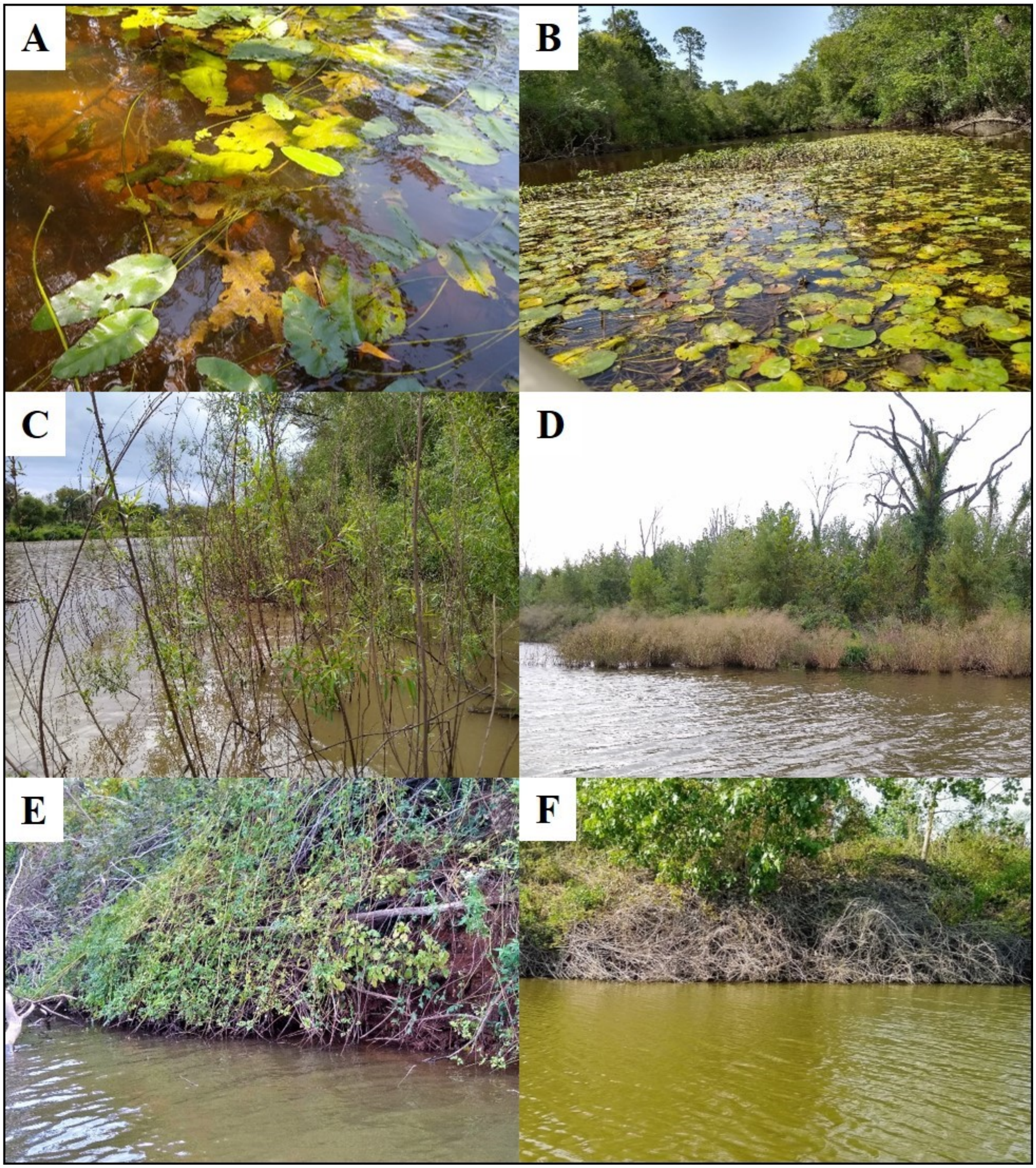
Appendix G-1. The process of model development, testing, and habitat surveys. Gray boxes indicate adjustments to the model while blue boxes represent field surveys to test adjustments.



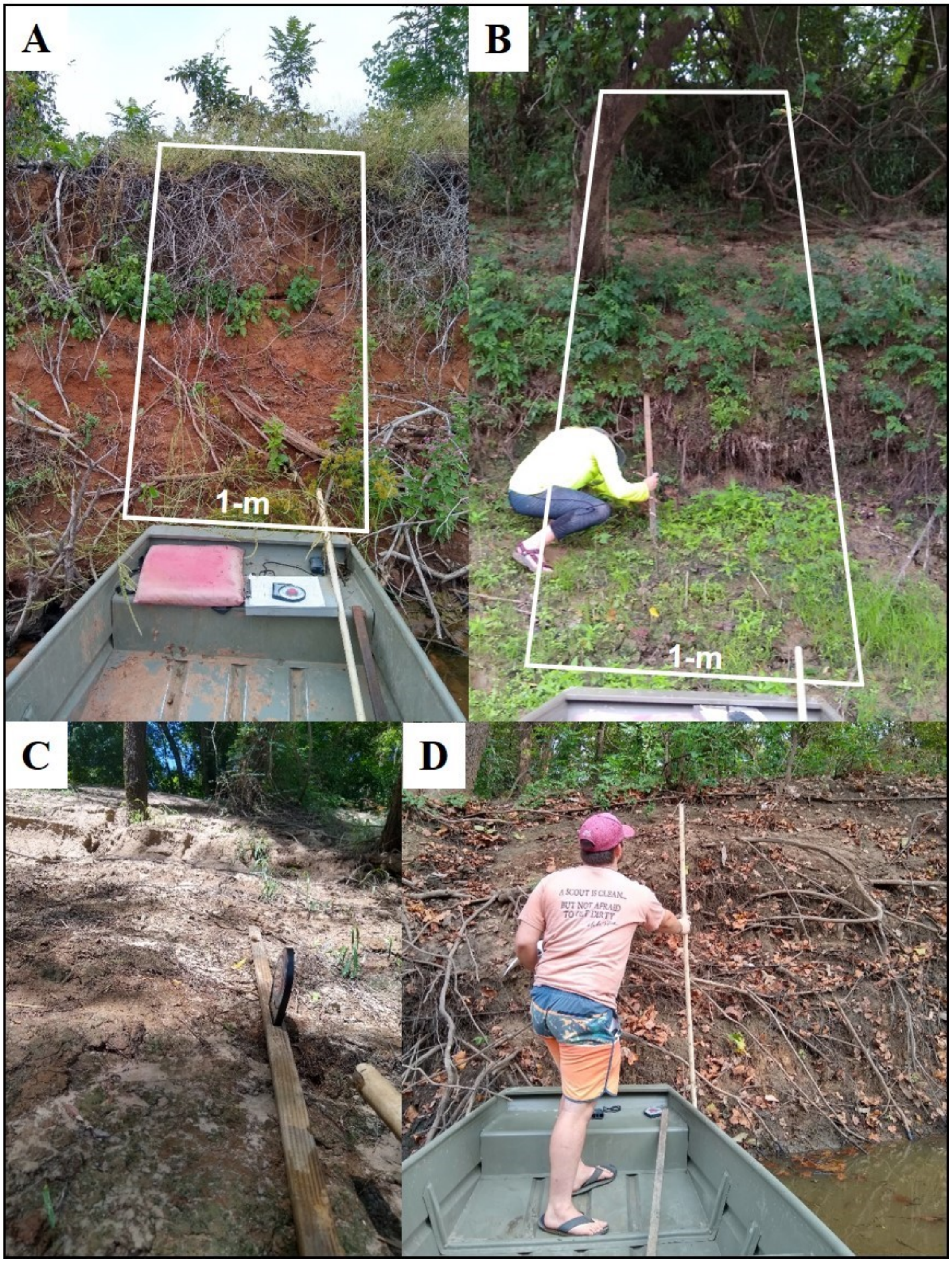
Appendix G-2. Demonstration of methods to measure channel characteristics. Measurements were taken from the boat while perpendicular to the bank. A) measurement of the channel width using a range finder, B) densitometer readings were taken at 5 m from the bank and in four directions shown by the arrows (upstream, downstream, right bank, and left bank). The number of logs were counted 20 m upstream and downstream of sample point, and within 10 m from the bank.



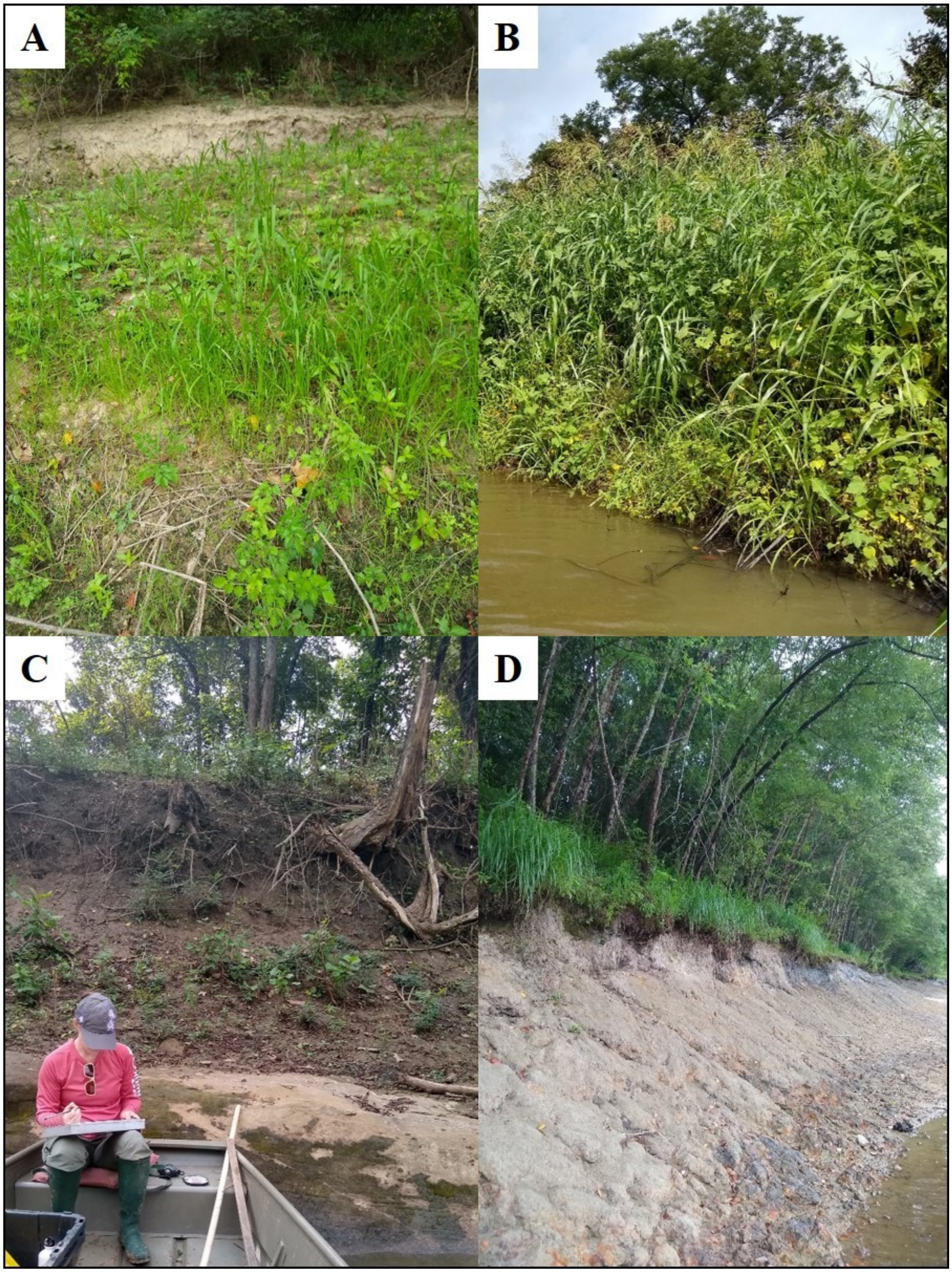
Appendix G-3. Examples of non-log submerged structure types used in habitat field surveys. A) beaver dens, B) large rocks, C) and D) trees rooted in the bank with branches overhanging into the water, E) undercut banks, and F) root balls with no visible undercut.



Appendix G-4. Examples of aquatic vegetation (A and B) and riparian vegetation (C–F) used as non-log submerged structure types in habitat surveys. A) *Nuphar* sp., B) *Nymphaea* sp. and *Justicia* sp., C) flooded willow (*Salix* sp.) thicket, D) submerged grasses, E) living vines growing over the water, and F) dead woody debris over hanging into the water.

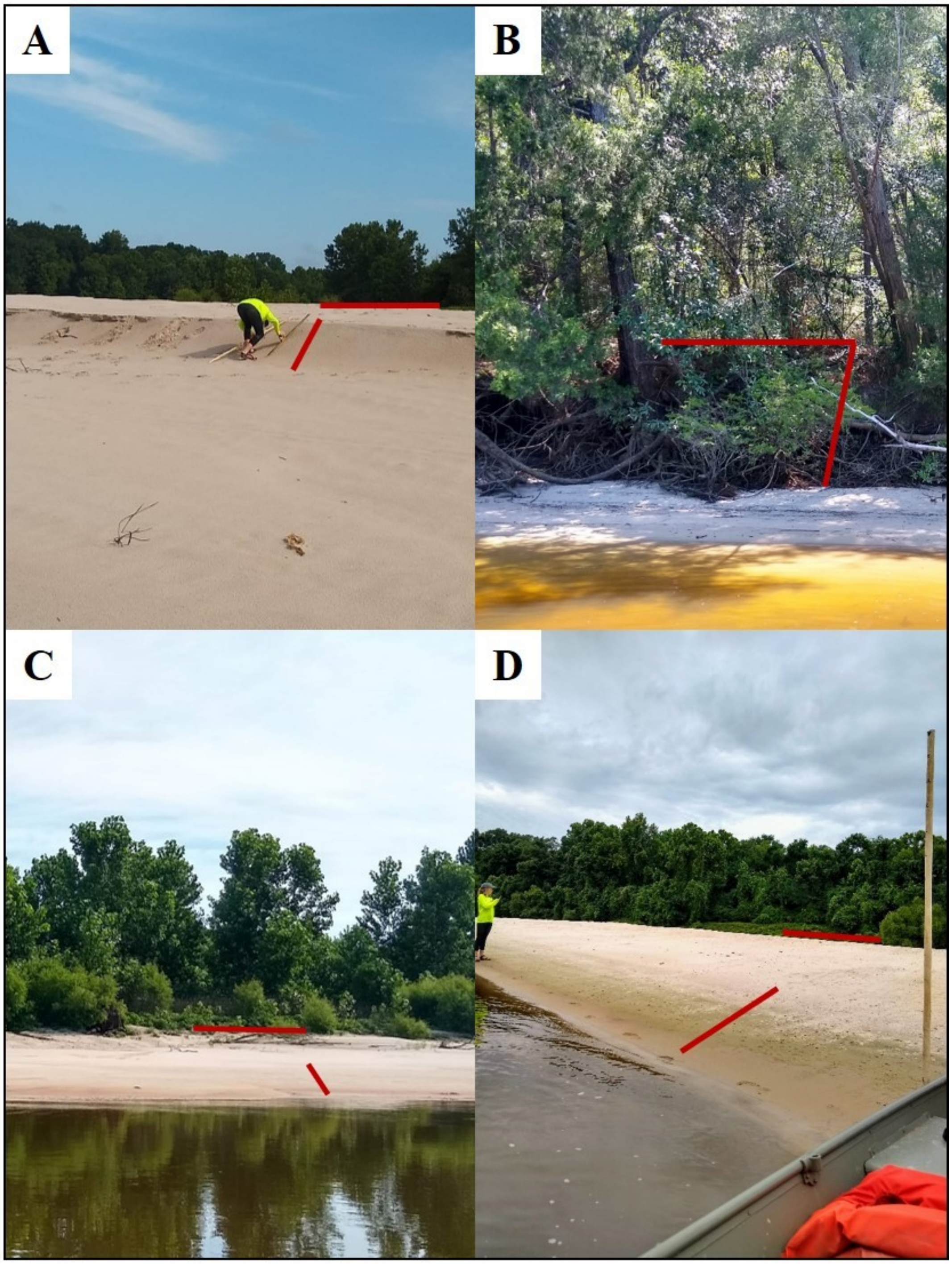


Appendix G-5. Demonstration of methods to measure bank characteristics. A) and B) substrate composition was measured in a 1-m strip along the height of the bank, B) and C) a wooden plank was placed along a representative portion of the bank to measure angle, and D) bank height was estimated using a 2-m long PVC pole with markings every 0.1-m.



Appendix G-6. Examples of substrate types used in habitat field surveys. A) sand and herbaceous vegetation, B) primarily dense vegetation and some rock, C) rock, dirt, herbaceous vegetation, and woody debris/roots, and D) hard clay and herbaceous vegetation.

Appendix G-7. Locations of habitat measurements during low water levels. 1) log density was estimated within the wetted channel, 2) canopy cover was measured 5-m from the base of the true bank—where the bank would be under normal conditions, 3) bank angle was measured along the slope of the true bank, and 4) bank height and substrate composition were measured from the base to the top of the true bank.



Appendix G-8. Examples of bank characteristics taken on sandbars. Locations of bank angle and height measurements are shown as red lines. A) when a large sand bar had multiple dunes, bank angle was taken on steep slopes between dunes, B) on small sand bars, bank angle was taken on the steep slope at the end of the sand, C) and D) on gradually sloping sandbars, angle was measured near the water, and bank height was measured at the top of the slope by viewing the PVC pole from a distance.

Appendix H.

scores	Calculate habitat suitability index scores

R Version 3.5.0 “Joy in Playing”

Author Kameron Voves

Description

The function uses raw habitat data from sample points to calculate habitat suitability index scores for the Alligator Snapping Turtle at each sample location.

Usage

scores(habitat, site)

Arguments

habitat	A matrix of raw habitat data for each sample point across all sample locations
site	A matrix of habitat characteristics for each sample location

Details

The required column names for the habitat matrix are as follows:

colnames(habitat)<-c(“Site”, “Non.log”, “Field.count”, “Canopy.cover”, “Sand”, “Dirt”, “Rock”, “Clay”, “Height”, “Angle”, “Buffer”)

The required column anmes for the site matrix are as follows:

colnames(site)<-c(“Site”, “Sonar.logs.km”, “Human.distrubance”, “Boat.traffic”)

Output Value

A list containing several matrices

- Suitability scores associated with habitat attributes for each site
- Suitability scores associated with site habitat characteristics for each site
- Means of the raw habitat attributes for each site
- Overall habitat suitability score for each site calculated using points count
- Overall habitat suitability score for each site calculated using sonar

Code

```
scores<-function(habitat,site){

 #Storage matrix
 scores.habitat<-matrix(nrow=nrow(habitat), ncol=7)
 colnames(scores.habitat)<-c("Point Count Logs", "Non Log", "Canopy Cover", "Substrate", "Angle", "Height",
"Buffer")
 scores.habitat<-cbind(habitat[,1:7],scores.habitat)

 #Point Counts
 for(i in 1:nrow(habitat)){
```

```
  scores.habitat$`Point Count Logs`[i]<-if(is.na(habitat$Field.count[i])){
    print(NA)
  }else if(habitat$Field.count[i]==0){
    print(0)
  } else if(habitat$Field.count[i]==1 | habitat$Field.count[i]==2){
    print(1)
  } else if(habitat$Field.count[i]==3 | habitat$Field.count[i]==4){
    print(2)
  } else
    print(3)
}

#Non-log Structure
scores.habitat$`Non Log`<-habitat$Non.log

#Canopy Cover
for(i in 1:nrow(habitat)){
  scores.habitat$`Canopy Cover`[i]<-if(is.na(habitat$Canopy.cover[i])){
    print(NA)
  }else if(habitat$Canopy.cover[i]<10){
    print(0)
  } else if (habitat$Canopy.cover[i]>=10 & habitat$Canopy.cover[i]<25){
    print(1)
  } else if (habitat$Canopy.cover[i]>=25 & habitat$Canopy.cover[i]<50){
    print(2)
  } else
    print(3)
}

#Substrate
for(i in 1:nrow(habitat)){
  scores.habitat$Substrate[i]<-if((habitat$Rock[i]+habitat$Clay[i])>=50){
    print(0)
  }else if((habitat$Sand[i]+habitat$Dirt[i])<30){
    print(1)
  } else if((habitat$Sand[i]+habitat$Dirt[i])>=30 & (habitat$Sand[i]+habitat$Dirt[i])<70){
    print(2)
  } else
    print(3)
}

#Angle
for(i in 1:nrow(habitat)){
  scores.habitat$Angle[i]<-if((habitat$Rock[i]+habitat$Clay[i])>=50){
    print(0)
  } else if(is.na(habitat$Angle[i])){
    print(NA)
  }else if(habitat$Angle[i]<10  | habitat$Angle[i]>75){
    print(0)
  } else if(habitat$Angle[i]>=10 & habitat$Angle[i]<20){
    print(1)
  } else if(habitat$Angle[i]>=20 & habitat$Angle[i]<30){
    print(2)
  } else if(habitat$Angle[i]>=46 & habitat$Angle[i]<=75){
    print(2)
  }else
```

```
    print(3)
}

#Height
for(i in 1:nrow(habitat)){
  scores.habitat$Height[i]<-if((habitat$Rock[i]+habitat$Clay[i])>=50){
    print(0)
  }else if(is.na(habitat$Height[i])){
    print(NA)
  }else if(habitat$Height[i]<0.5){
    print(0)
  }else if(habitat$Height[i]>=0.5 & habitat$Height[i]<1){
    print(1)
  }else if(habitat$Height[i]>=1 & habitat$Height[i]<1.5){
    print(2)
  }else
    print(3)
}

#Buffer
for(i in 1:nrow(habitat)){
  scores.habitat$Buffer[i]<-if(is.na(habitat$Buffer[i])){
    print(NA)
  }else if(habitat$Buffer[i]<2){
    print(0)
  }else if(habitat$Buffer[i]>=2 & habitat$Buffer[i]<9){
    print(1)
  }else if(habitat$Buffer[i]>=9 & habitat$Buffer[i]<15){
    print(2)
  }else
    print(3)
}
#Remove rows that are missing a habitat measurement
na.removed<-complete.cases(scores.habitat[,8:14])
scores.habitat<-scores.habitat[na.removed,]

#Calculate the mean of each habitat vairable for each site
means<-aggregate(scores.habitat[,8:14], by=list(scores.habitat$Site), mean)

#Full River Variables

#Storage matrix
scores.site<-as.data.frame(matrix(nrow=nrow(site), ncol=3))
rownames(scores.site)<-means[,1]
colnames(scores.site)<-c("Sonar.logs.km", "Human.Disturbance", "Boats.hr")

#Sonar logs/km
for(i in 1:nrow(site)){
  scores.site$Sonar.logs.km[i]<-if(site$Sonar.logs.km[i]<50){
    print(0)
  }else if(site$Sonar.logs.km[i]>=50 & site$Sonar.logs.km[i]<150){
    print(1)
  }else if(site$Sonar.logs.km[i]>=150 & site$Sonar.logs.km[i]<250){
    print(2)
  }else
    print(3)
```

```
  }

  #Human disturbance
  for(i in 1:nrow(site)){
   scores.site$Human.Disturbance[i]<-if(site$Human.disturbance[i]>=12){
     print(0)
   }else if (site$Human.disturbance[i]>=8 & site$Human.disturbance[i]<12 ){
     print(1)
   }else if (site$Human.disturbance[i]>=4 & site$Human.disturbance[i]<8){
     print(2)
   }else
     print(3)
  }

  ###Boat Traffic
  for(i in 1:nrow(site)){
   scores.site$Boats.hr[i]<-if(site$Boat.traffic[i]<1){
     print(0)
   }else if(site$Boat.traffic[i]>=1 & site$Boat.traffic[i]<2){
     print(1)
   }else if(site$Boat.traffic[i]>=2 & site$Boat.traffic[i]<3){
     print(2)
   }else
     print(3)
  }

  #Combine habitat and site scores into one object
  means<-cbind(means[,],scores.site[,])

  #Calculating final scores
  final.scores<-matrix(nrow=nrow(site), ncol=5)
  rownames(final.scores)<-means[,1]
  colnames(final.scores)<-c("Channel", "Bank", "Disturbance", "HS Total", "RS Total")

  final.scores[,1]<-(3*means$`Point Count Logs`)+means$`Canopy Cover`+means$`Non Log`
  final.scores[,2]<-means$Substrate+means$Angle+means$Height
  final.scores[,3]<-means$Buffer+means$Human.Disturbance
  final.scores[,4]<-final.scores[,1]+final.scores[,2]+fiNAl.scores[,3]
  final.scores[,5]<-final.scores[,4]-means$Boats.hr

  point.count<-fiNAl.scores

  final.scores[,1]<-(3*means$Sonar.logs.km)+means$`Canopy Cover`+means$`Non Log`
  final.scores[,2]<-means$Substrate+means$Angle+means$Height
  final.scores[,3]<-means$Buffer+means$Human.Disturbance
  final.scores[,4]<-final.scores[,1]+final.scores[,2]+final.scores[,3]
  final.scores[,5]<-final.scores[,4]-means$Boats.hr

  sonar<-final.scores

  output<-list(scores.habitat, scores.site, means, point.count, sonar)
  return(output)
}
```

Appendix I

Appendix I-1. Habitat characteristics for sites where alligator snapping turtles were present. Data are displayed as the average of each habitat attribute (average HSI point value assigned). Point values were summed to obtain HSI Score—only one type of deadwood density should be used for these calculations.

Habitat Attribute	Big Cabin Creek	Pearl	Poteau—Arkoma	Escatawpa	Pascagoula	Chickasawhay
Channel Characteristics						
Point count log density[1]	Low (3.55)	Low (4.14)	Low (2.88)	Low (5.94)	Med (5.10)	Med (4.74)
Sonar TDD (logs/km)[1]	197 (6)	227 (6)	281 (9)	458 (9)	341 (9)	322 (9)
Non-log structure[2]	N/A	0.04 (0.04)	N/A	0.36 (0.37)	0.00 (0.00)	0.02 (0.02)
Canopy cover (% occupied)	47 (2.27)	36 (1.74)	43 (2.18)	50 (2.14)	39 (1.70)	21 (1.10)
Bank Characteristics						
% dirt/sand	49 (1.86)	52 (1.92)	23 (1.44)	54 (2.08)	60 (2.14)	62 (2.20)
Angle (°)	55 (1.49)	37 (2.02)	49 (2.24)	22 (1.47)	34 (2.12)	32 (1.76)
Height (m)	3.0 (2.28)	3.1 (2.58)	1.9 (2.58)	0.9 (1.43)	2.6 (2.64)	3.0 (2.50)
Anthropogenic Characteristics						
Riparian buffer (m)	140 (2.65)	348 (2.98)	65 (2.98)	400 (2.98)	213 (2.90)	379 (2.94)
Human disturbance	14 (0)	11 (1)	12 (0)	11 (1)	8 (1)	7 (2)
Boat traffic (boats/day)[3]	1.22 (-1)	3 (-3)	0.50 (-0)	5 (-3)	4 (-3)	10 (-3)
Sonar HSI score	16.55	18.28	20.42	20.47	21.50	21.52

[1]Suitabilty score was tripled.
[2] Displayed as the proportion of 50 sample points where at least one type of non-log submerged structure was present. Point values for non-log structure exceeded this proportion when more than one type was present at any given sample point.
[3] Score is deducted from the sonar HSI score to obtain the reintroduction suitability score.

Appendix I-2. Habitat characteristics for sites where alligator snapping turtles were reintroduced. Data are displayed as the average of each habitat attribute (average HSI point value assigned). Point values were summed to obtain HSI Score—only one type of deadwood density should be used for these calculations.

Habitat Attribute	Pond Creek	Old River Channel	Washita	Caney
Channel Characteristics				
Point count log density[1]	Low (1.68)	High (8.22)	Low (4.38)	Low (2.82)
Sonar TDD (logs/km)[1]	234 (6)	287 (9)	218 (6)	303 (9)
Non-log structure[2]	0.54 (0.56)	0.32 (0.48)	0.04 (0.04)	0.22 (0.22)
Canopy cover (% occupied)	23 (1.12)	6 (0.26)	21 (1.06)	29 (1.38)
Bank Characteristics				
% dirt/sand	28 (1.46)	18 (1.28)	53 (2.16)	31 (1.68)
Angle (°)	57 (1.92)	50 (1.94)	45 (2.52)	41 (2.44)
Height (m)	2.4 (2.40)	2.1 (2.54)	3.6 (3.00)	2.7 (2.78)
Anthropogenic Characteristics				
Riparian buffer (m)	156 (2.04)	65 (2.22)	357 (2.94)	193 (2.22)
Human disturbance	11 (1)	12 (0)	10 (1)	14 (0)
Boat traffic (boats/day)[3]	0.5 (-0)	0 (-0)	1 (-1)	2 (-1)
Sonar HSI score	16.50	17.72	18.72	19.72

[1]Suitabilty score was tripled.
[2] Displayed as the proportion of 50 sample points where at least one type of non-log submerged structure was present. Point values for non-log structure exceeded this proportion when more than one type was present at any given sample point.
[3] Score is deducted from the sonar HSI score to obtain the reintroduction suitability score.

Appendix I-3. Habitat characteristics for sites where alligator snapping turtles were absent. Data are displayed as the average of each habitat attribute (average HSI point value assigned). Point values were summed to obtain HSI Score—only one type of deadwood density should be used for these calculations.

Habitat Attribute	Neosho	Fall	Poteau—Wister	Chouteau Creek	Elk	Deep Fork	Verdigris
Channel Characteristics							
Point count density[1]	Low (3.23)	Low (3.48)	Low (1.86)	Low (3.50)	Low (4.08)	Low (3.42)	Med (5.28)
Sonar TDD (logs/km)[1]	129 (3)	141 (3)	128 (3)	211(6)	188 (6)	340 (9)	320 (9)
Non-log structure[2]	N/A	0.26 (0.28)	N/A	N/A	0.44 (0.54)	N/A	0.08 (0.08)
Canopy cover (% occupied)	42 (2.19)	37 (1.80)	37 (1.74)	24 (1.08)	61 (2.52)	39 (1.88)	61 (2.72)
Bank Characteristics							
% dirt/sand	68 (2.38)	43 (1.68)	45 (1.96)	71 (2.58)	42 (1.86)	54 (2.16)	66 (2.42)
Angle (°)	43 (2.00)	40 (2.02)	53 (1.92)	34 (1.58)	53 (2.02)	47 (2.38)	39 (2.50)
Height (m)	3.5 (2.31)	3.1(2.40)	2.3 (2.42)	1.8 (2.33)	2.3 (2.88)	3.6 (3.00)	2.0 (2.94)
Anthropogenic Characteristics							
Riparian buffer (m)	508 (2.88)	151 (2.64)	682 (2.90)	355 (2.75)	169 (2.88)	696 (3.00)	212 (3.00)
Human disturbance	14 (0)	9 (1)	9 (1)	7 (2)	12 (0)	9 (1)	5 (2)
Boat traffic (boats/day)[3]	7.57 (-3)	1 (-1)	2.00 (-2)	0 (-0)	6 (-3)	0.44 (-0)	1.33 (-1)
Sonar HSI score	14.77	14.82	14.94	18.33	18.70	22.42	24.66

[1]Suitabilty score was tripled.
[2] Displayed as the proportion of 50 sample points where at least one type of non-log submerged structure was present. Point values for non-log structure exceeded this proportion when more than one type was present at any given sample point.
[3] Score is deducted from the sonar HSI score to obtain the reintroduction suitability score.

Appendix I-4. Summary of the types of human disturbances found at each sample site surveyed. Sites are ordered alphabetically. Types of disturbance highlighted in light gray received one point, dark gray received two points, and black received three points.

Site	Recreational use of banks/sandbars	Foot paths	Rowing boats	Abandoned limb/jug lines	Pastures	Crop land	Active limb/jug lines	Campgrounds	Oil wells	Dramatic water cycling	Frequent flooding	Buildings/docks	Number of roads	Number of boat ramps	Total score
Big Cabin													2	1	14
Caney													2	1	14
Chickasawhay													1	1	7
Chouteau													1	1	7
Deep Fork													2	1	9
Elk													1	1	12
Escatawpa													0	1	11
Fall													1	1	9
Neosho													2	2	14
Old River Channel													3	3	12
Pascagoula													2	1	8
Pearl													1	1	11
Pond Creek													2	1	11
Poteau–Arkoma													1	1	12
Poteau–Wister													2	3	9
Verdigris													1	1	5
Washita													0	1	10