Original Article

Size Matters: the Influence of Trap and Mesh Size on Turtle Captures

JOSHUA R. ENNEN [iD],[1,2] *Tennessee Aquarium Conservation Institute, 175 Baylor School Rd, Chattanooga, TN 37377, USA*

KRISTEN K. CECALA, *Department of Biology, University of the South, 735 University Ave., Sewanee, TN 37383, USA*

PHILIP GOULD, *School of Environment and Natural Resources, Ohio State University, 2021 Coffey Rd, Columbus, OH 43210, USA*

ROBERT COLVIN, *Tennessee Wildlife Resources Agency, Region I, 200 Lowell Thomas Drive, Jackson, TN 38301, USA*

JEREMY DENISON, *Tennessee Wildlife Resources Agency, Region I, 200 Lowell Thomas Drive, Jackson, TN 38301, USA*

DUSTIN F. GARIG, *Department of Biology, Southeast Missouri State University, One University Plaza, MS 6200, Cape Girardeau, Missouri 63701, USA*

SAIDEE HYDER, *Department of Biology, Southeast Missouri State University, One University Plaza, MS 6200, Cape Girardeau, Missouri 63701, USA*

LEWIS RECKER, *Department of Biology, Southeast Missouri State University, One University Plaza, MS 6200, Cape Girardeau, Missouri 63701, USA*

JON M. DAVENPORT,[3] *Department of Biology, Southeast Missouri State University, One University Plaza, MS 6200, Cape Girardeau, Missouri 63701, USA*

ABSTRACT Methods used in wildlife ecology can influence population- and community-level estimates, such as species richness, sex ratio, age and size structure, occupancy and detection probabilities, and community composition. Various trapping and sampling biases exist for freshwater turtles including bait and trap choice and survey technique. To date, no study has investigated the influence of hoop net and mesh size on various population- and community-level estimates. Here, we use detection models to determine if trap and mesh size influence detection probability of nine species of freshwater turtles over 3 consecutive years (2016–2018) in west Tennessee. Additionally, we use multivariate models to determine if freshwater turtle community composition was influenced by hoop trap and mesh sizes. Our results indicated that there was a bias related to mesh size in detection probabilities and community composition. Smaller mesh sized traps were better at detecting smaller-bodied turtle species, which then changed community species richness but not catch-per-unit-effort estimates (i.e., abundance). Additionally, larger mesh sized traps were better at detecting common snapping turtles (*Chelydra serpentina*), which supports previous research. Our results suggest that researchers should account for the variation in detection probabilities by mesh size when conducting mark-recapture and occupancy analyses. Moreover, erroneous inferences about population trends and changes in diversity within turtle communities through time could cause managers to misidentify population declines and conservation value of a site. © 2021 The Wildlife Society.

KEY WORDS biases, chelonian, comparability, conservation, diversity, Testudinidae.

As the 6th mass extinction strikes wildlife worldwide, conservation biologists must ensure that variability in estimates of population parameters are not due to human error. Several studies have pointed out challenges with detecting trends and declines in populations (e.g., Pechmann et al. 1991, Koper and Brooks 1998,

Sun et al. 2014, Fournier et al. 2019). For example, study sites chosen based on availability and abundance of focal species (i.e., site-selection bias) can lead to misleading inferences about population trends (Pechmann et al. 1991, Fournier et al. 2019). Another major source of potential bias in population estimates is survey method, in particular study design and trap deployment. Bias in survey method can produce very different estimates of population and community parameters (Senar et al. 1999, Brennan et al. 2005, Burton et al. 2015, Kordjazi et al. 2016, Bovendorp et al. 2017). Ultimately, incorrectly concluding that populations or communities are improving or declining could easily lead to misallocation of limited funding and conservation efforts. Yet, many studies only use one method or do not account for trap-type biases

when estimating the spatiotemporal variability in populations.

The wide variation in sampling designs and protocols across studies creates a lack of comparability (Gotelli and Colwell 2001, Alivisatos et al. 2015, Brown and Matthews 2016). This variation is especially striking across studies focusing on closely related taxa or even the same taxon (Bailey and Nichols 2009). Yet, an understanding of how variation influences population parameters (sex ratio, population abundance, age and size structure, survivorship), community parameters (diversity, evenness, richness), and detection probabilities for occupancy and capture-mark-recapture models is still needed for many wildlife species (Gotelli and Colwell 2001, Bisi et al. 2011, Alivisatos et al. 2015, Burton et al. 2015, Edwards et al. 2016). For example, multiple trapping methods produced varying sex ratios and size-class structure estimates within populations of the painted turtle, *Chrysemys picta* (Ream and Ream 1966, Tesche and Hodges 2015). Therefore, robust comparisons across studies encompassing potential biases are needed to provide managers with guidelines and quality assessments.

Turtles as a group are imperiled globally, with over half of the known species at risk of extinction (Rhodin et al. 2018). Recent reviews have highlighted the vital part of many ecosystems that imperiled turtle taxon can play (Lovich et al. 2018, Garig et al. 2020). Therefore, accurate and comparable population- and community-level parameters are critical for species conservation. However, similar to other taxonomic groups (e.g., arthropods, mammals, and amphibians; Gotelli and Colwell 2001, Bisi et al. 2011, Sun et al. 2014, Edwards et al. 2016), variation in methods (Tesche and Hodges 2015) accompanied by low and variable capture rates (Bluett et al. 2011), may influence population- and community-level parameters. For example, trap and bait type along with study design can influence the estimated abundance, sex ratios, and age structure within a freshwater turtle population and species richness within a freshwater turtle community (Mali et al. 2012, 2014*a,b*; Tesche and Hodges 2015; Mali et al. 2018). Additionally, some turtle studies (Howell et al. 2016, Gulette et al. 2019) have reported trapping biases related to hoop net diameter, where smaller diameters capture smaller species (e.g., spotted turtles [*Clemmys guttata*]; Howell et al. 2016) and larger diameters capture larger species (e.g., common snapping turtle [*Chelydra serpentina*]; Gulette et al. 2019). All the various trapping methods deployed for freshwater turtle studies have well-reported biases, which has led several authors to suggest the use of multiple capture methods to improve population estimates for turtles (Ream and Ream 1966, Koper and Brooks 1998, Tesche and Hodges 2015). However, the intense labor of turtle trapping and resource limitations (e.g., initial cost of traps) usually impede researchers from deploying multiple capture methods. Despite known shortcomings across methodologies, the need for evaluation of a common technique (hoop net traps for aquatic turtles) and potential bias is still necessary.

Here, we collected presence-absence and abundance data from 64 sites across western Tennessee using three variations of baited hoop nets. Our overall goal was to determine if variation in hoop net size and mesh size, however slight, influences detection probability, estimates of community diversity, and capture per unit effort for freshwater turtles. We hypothesized that: 1) smaller meshed traps would have higher detection probabilities for smaller bodied species of turtles, 2) the smaller meshed traps will lead to different estimates of a freshwater turtle community than larger mesh traps because smaller mesh sizes will prevent escape of small-bodied species; and 3) larger diameter hoop nets will capture larger bodied turtle species similar to other studies (Gulette et al. 2019). Finally, we hypothesized that there will be no difference among hoop net size and mesh size in catch per unit effort within individual species.

STUDY AREA

Tennessee was known to harbor at least 16 turtle species, most of which inhabited the western portion of the state (Scott and Redmond 2019). Our trapping sites were all located in western Tennessee (Fig. 1)—an area within the Mississippi Embayment portion of the Gulf Coastal Plain. Within the 21-county region, our specific sites were located within the alluvial floodplain of the Mississippi or within the following watersheds: Wolf, Hatchie, Forked Deer, and Obion. The latter watersheds were characteristic of slow-flow, meandering river systems that were prone to flooding, which produced numerous sloughs, swamps, and backwater areas (Etnier and Starnes 1993). The aquatic turtle community of the region included 11 species: spiny softshell turtle (*Apalone spinifera*); smooth softshell turtle (*A. mutica*); common snapping turtle, (*Chelydra serpentina*); southern painted turtle (*Chrysemys dorsalis*); false map turtle (*Graptemys pseudogeographica*); Ouachita map turtle (*G. ouachitensis*); eastern mud turtle (*Kinosternon subrubrum*); western alligator snapping turtle (*Macrochelys temminckii*); river cooter (*Pseudemys concinna*); stinkpot (*Sternotherus odoratus*); and pond slider (*Trachemys scripta*). Much of this region was disturbed by human activity with habitat destruction and modification via agricultural practices and channelization of most major river drainages (Etnier and Starnes 1993), and these were likely the main threats to freshwater turtles. The region was also home to the only natural lake, Reelfoot Lake, in Tennessee.

METHODS

Trapping Design
We trapped a total of 64 unique sites in 3 consecutive years (2016–2018) from April through October (Fig. 1). Each site was trapped once using 10 traps that were deployed for 3 consecutive trap nights (approximately 72 hours). We used 3-ringed hoop nets (Miller Net Company, Memphis, Tennessee, USA) with either small or large hoop sizes and either small or large mesh sizes: 1) 91.44 cm hoop nets with 3.81 cm mesh (i.e., small hoop-small mesh trap; 3 ft diameter with 1.5 in mesh), 2) 121.92 cm hoop nets with
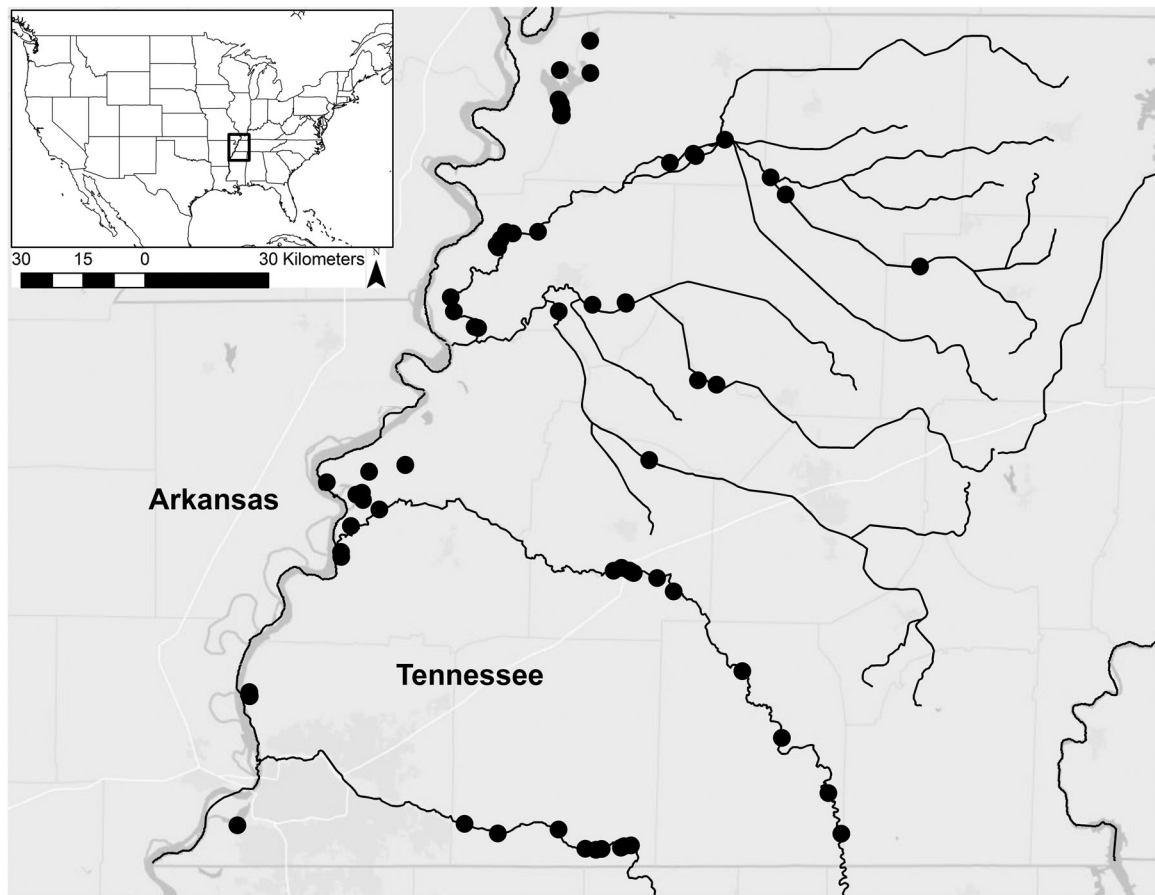
Figure 1. The locations of our 64 trapping sites (black circles) in western Tennessee, USA, that were sampled during April through October 2016–2018.

3.81 cm mesh (i.e., large hoop-small mesh trap; 4 ft diameter with 1.5 in mesh), and 3) 91.44 cm hoop nets with 10.16 cm mesh (i.e., small hoop-large mesh trap; 3 ft diameter with 4 in mesh). The traps had flat throats, and variation in throat height can influence capture and recapture rates (Mali et al. 2014*b*). We did not measure throat height before or during the study, therefore, we can not control for any variation in throat height across traps sizes or time. However, unlike Mali et al. (2014*b*), we did not notice loosening of the throats (i.e., increasing throat height) through time, which might be related to trap manufacturers or some other unknown factors. Traps were baited with various types of fish (e.g., sunfish, crappie, buffalo, gar, and carp) provided by the Tennessee Wildlife Resources Agency and local fish markets. Traps were re-baited every day regardless of whether the bait was missing or degraded. All bait was presented in a bait bag suspended from the top of the hoop net. Although not every site was baited with the exact same species of fish, similar fish species were used within a site, which should have minimized bait bias (Thomas et al. 2008). We acknowledge that variation in bait likely explains some of the differences observed between sites; however, in our study, we were interested in the effect of trap type and not variation among sites. All traps were set near submerged vegetation or structures and at water depths that allowed the trap to protrude from the water to ensure that all turtles would be able to surface for air.

All species of turtles were measured and sexed when captured; however, in this study, we only used presence-absence and abundance data in our analyses. For each species, our abundance data was converted into catch per unit effort (CPUE) using number of traps and trap nights for each of the 3 trap types per site. All descriptive statistics are means accompanied by standard deviations.

Analyses

Detection model.—We evaluated site-occupancy for all species of turtle concurrently in a multi-species occupancy model (Kéry and Royle 2015). However, we removed *Apalone mutica* from the detection model because of the low sample size (n = 1). In this framework, we estimated the effect of all parameters on each individual species. For each detection parameter, the defined prior distribution was normally distributed around a hyper-parameter for both the mean and precision. The priors for the community hyper-parameters were normally distributed with a mean of 0 and precision of 0.1. We estimated occupancy using an intercept-only occupancy model as our intention was to evaluate trap-type effects on detection. To model the effects of trap-type on turtle detection, we treated type-1 as the reference factor. For site-occupancy z_{ik} for species k at trap i, we modeled:

$$z_{ik} \sim \text{bernoulli}(\psi_{ik}),$$

$$\text{logit}(\psi_{ik}) = \text{beta0}_k,$$

$$\text{beta0}_k \sim \text{normal}(\text{logit}[\text{mu.psi}], \text{sig.psi}^2),$$

where beta0_k is a species-level intercept.

We modeled the observation process as:

$$y_{ijk} \sim \text{Bernoulli}(z_{ik} * p_{ijk}),$$

$$\text{logit}(p_{ijk}) = \text{alpha0}_{km} + \text{alpha1}_k * x_i,$$

$$\text{alpha1}_k \sim \text{normal}(\text{logit}[\text{mu.alpha.lpsi}], \text{sig.alpha.psi}^2),$$

where y_{ijk} is the observed presence of each turtle species and p is the probability of capture for each species k at trap i on visit j. Detection intercept and covariate estimates were modeled with similar hyper-parameters to occupancy.

We ran models for 60,700 iterations, with a burn-in of 700, a thinning rate of 10, and an adaptation of 6000. Models were fit with JAGS in R (R Core Team 2017) using package *jagsUI* (Kellner 2015). We evaluated model fit using a Freeman-Tukey diagnostic by fitting observations with estimates from the model and estimating a Bayesian P-value (BP). Models were deemed to have high predictive ability if BP values were approximately 0.5. We determined model convergence by visually inspecting chains for convergence and determining that parameters had adequate (>300) effective sampling.

Community Composition
We conducted two permutational multivariate analysis of variance (NPMANOVA; Oksanen et al. 2019) to determine if species composition differed among the trap types. In NPMANOVAs, we used either CPUE or presence-absence data to create species by site matrices. We used Jaccard and Bray-Curtis dissimilarity index as our distance measure for our presence-absence and CPUE data, respectively. We used 999 permutations and controlled for site via the strata option in both NPMANOVAs. By controlling for variation among sites, our analyses account for the variation in different baits. We used an alpha of 0.05 to determine significance.

Species-Specific Catch-Per-Unit-Effort Models
We conducted generalized linear mixed models (GLMM; Brooks et al. 2017) to determine if CPUE differed by trap type for each species. In the GLMMs, we used CPUE as our dependent variable, trap type as a fixed effect, and site as a random effect. Because our dataset consisted largely of zeros, we tested zero-inflated models with negative binomial distribution. The zero-inflated models failed to improve model fit (i.e., Akaike Information Criteria [AICc] values); therefore, we proceeded with the most parsimonious model for each species. In all cases, the most parsimonious model used a negative binomial distribution type 2 (i.e., nbinom2). We determined significance ($\alpha = 0.05$) of the models using

Anova.glmmTMB function, and we conducted post hoc analyses using *emmeans* function (Lenth et al. 2020). Due to low sample sizes (i.e., sites and captures) for several species, we only ran models for *Apalone spinifera*, *Chelydra serpentina*, *Graptemys pseudogeographica*, *Pseudemys concinna*, *Sternotherus odoratus*, and *Trachemys scripta*.

RESULTS
We captured 3,026 turtles and a total of 11 turtle species (i.e., *Apalone spinifera*, *A. mutica*, *Chelydra serpentina*, *Chrysemys dorsalis*, *Graptemys pseudogeographica*, *G. ouachitensis*, *Kinosternon subrubrum*, *Macrochelys temminckii*, *Pseudemys concinna*, *Sternotherus odoratus*, and *Trachemys scripta*) over a 3-year period. The total number of turtles captured at individual sites ranged from 1 to 278, with a mean of 45.9 turtles per site. *Trachemys scripta* was by far the most abundant turtle ($\bar{x} \pm$ SD CPUE across traps = 1.33 $\pm$ 2.19), followed by *Chelydra serpentina* (0.12 $\pm$ 0.18). Other species with moderate CPUE were *Graptemys pseudogeographica* ($\bar{x} \pm$ SD = 0.05 $\pm$ 0.21, *Apalone spinifera* (0.04 $\pm$ 0.10), *Sternotherus odoratus* (0.03 $\pm$ 0.11), and *Macrochelys temminckii* (0.01 $\pm$ 0.08). The lowest mean CPUE for any species was 0.0006 $\pm$ 0.008 for *Apalone mutica* followed by *Kinsternon subrubrum* (0.001 $\pm$ 0.015), *Graptemys ouachitensis* (0.003 $\pm$ 0.028), *Chrysemys dorsalis* (0.005 $\pm$ 0.025), and *Pseudemys concinna* (0.008 $\pm$ 0.036).
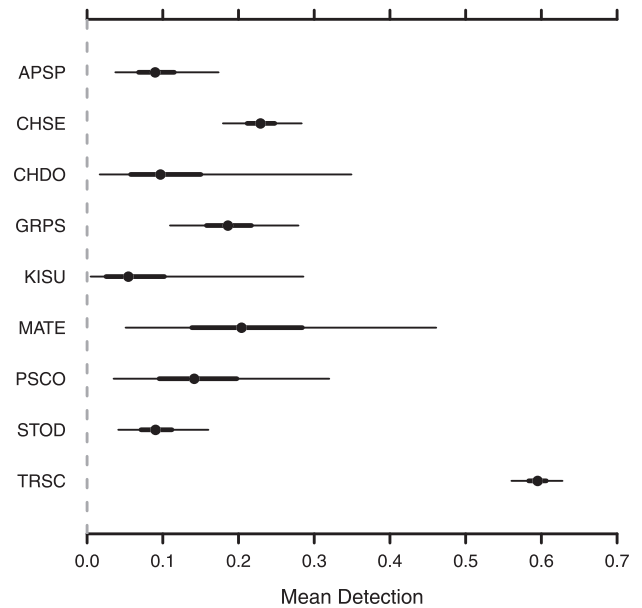
Figure 2. The mean detection probabilities for 9 species of freshwater turtles in western Tennessee, USA, April through October 2016–2018. Abbreviations are as follows: *Apalone spinifera* (APSP), *Chelydra serpentina* (CHSE), *Chrysemys dorsalis* (CHDO), *Graptemys pseudogeographica* (GRPS), *Kinosternon subrubrum* (KISU), *Macrochelys temminckii* (MATE), *Pseudemys concinna* (PSCO), *Sternotherus odoratus* (STOD), *and Trachemys scripta* (TRSC). Dots indicate the mean detection probability estimate. The stronger bolded line indicates the 50% credible interval, whereas the less bold line indicates the 95% credible interval range.

Detection Probability

Overall, mean detection varied by species with *T. scripta* having the greatest mean detection probability (Fig. 2). The effect of trap type on detection was species specific, although generally species were more readily captured using small hoop-small mesh traps (Table S1, available online in Supporting Information; Fig. 3). Small hoop-small mesh traps were the most effective trap type for detecting *Apalone spinifera* and *Sternotherus odoratus*. Large hoop-small mesh traps were as effective as small hoop-small mesh traps for 6 species (*Chelydra serpentina*, *Chrysemys dorsalis*, *Graptemys pseudogeographica*, *Kinosternon subrubrum*, *Macrochelys temminckii*, *Pseudemys concinna*), although no species was best detected by large hoop-small mesh traps. Small hoop-large mesh traps were the most effective trap for detecting *Chelydra serpentina*, however, they were significantly less effective than both small hoop-small mesh and large hoop-small mesh traps for detecting *Sternotherus odoratus* (Fig. 3). Small hoop-large mesh traps were also as effective as small hoop-small mesh traps for detecting *Pseudemys concinna* and *Trachemys scripta*. Across all trap types, mean species detection ranged from 0.09 to 0.60 and was greatest for *Trachemys scripta* ($\bar{x} = 0.60$, range = 0.56–0.63) whereas lowest for *Sternotherus odoratus* ($\bar{x} = 0.09$, range = 0.03–0.16). Posterior predictive check assessment indicated a BP value of 0.51, suggesting strong model predictive ability.

Community Composition

Our NPMANOVA models using CPUE and presence-absence data produced different results. There was a significant difference in estimated community composition among the trap types ($R^2 = 0.03$, $F_{(2,165)} = 2.35$, $P = 0.01$) using our presence-absence data, whereas there was no difference among

the trap types ($R^2 = 0.01$, $F_{(2,165)} = 1.09$, $P = 0.15$) using CPUE (Table 1). Only small meshed hoop nets (i.e., small hoop-small mesh and large hoop-small mesh traps) captured smaller-bodied turtle species, such as *Sternotherus odoratus* and *Kinosternon subrubrum*, which lead to higher species richness estimates.

Species-Specific CPUE Models

Our species-specific models comparing CPUE among trap type were largely not significant (Table 2). There was a significant difference in CPUE among the trap types ($\chi^2_{(2)} = 13.92$, $P \leq 0.01$) for only *Trachemys scripta*, and the pairwise post-hoc analysis found significant differences ($t_{(182)} = 3.701$, $P < 0.01$) between small hoop-small mesh ($\bar{x} \pm SD = 1.801 \pm 2.903$) and large hoop-small mesh traps (0.914 ± 1.754; Table 2). For all other species-specific models, χ^2 values ranged from 0.02 to 2.95 and p-values ranged from 0.23 to 0.99.

DISCUSSION

One of the most common trap types used for capturing freshwater turtles is the baited hoop net (Plummer 1979, Vogt 2016, Cecala et al. 2017). There are multiple factors (e.g., bait type, bait age, throat size, duration, and placement) that influence capture of freshwater turtles using hoop nets (Nall and Thomas 2009; Mali et al. 2014a, b; Long et al. 2017; Richardson et al. 2017). Our study is one of the first to provide support that variation in hoop net and mesh size, but mainly mesh size, influences population estimates of freshwater turtles. We hypothesized that smaller meshed traps would have higher detection probabilities for smaller bodied species of turtles. Our results supported this hypothesis because our trap types with smaller mesh size
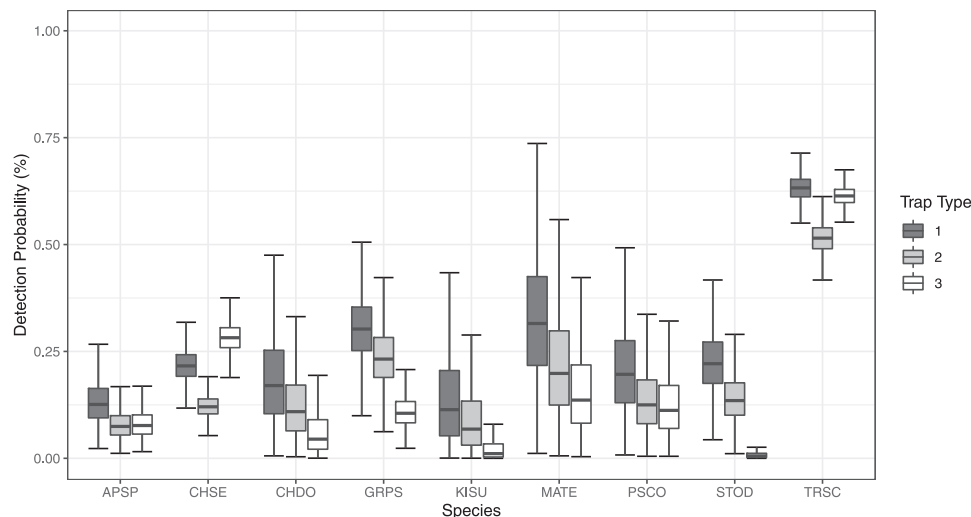
Figure 3. Boxplots showing detection probability as a function of trap type for 9 species of freshwater turtles in western Tennessee, USA, April through October 2016–2018. Trap types were as follows: 1) small hoop-small mesh (91.44 cm hoop nets with 3.81 cm mesh), 2) large hoop-small mesh (121.92 cm hoop nets with 3.81 cm mesh), and 3) small hoop-large mesh (91.44 cm hoop nets with 10.16 cm mesh). Both small hoop-small mesh and small hoop-large mesh traps (smaller hoop net diameter) were more effective than large hoop-small mesh taps (larger hoop net diameter) respectively, for *Chelydra serpentina*. Plots were generated using the MCMC sample distributions. Fifty percent of the posterior density is represented within the interquartile range, and the error bars represent the 5th and 95 percentiles. Abbreviations are as follows: *Apalone spinifera* (APSP), *Chelydra serpentina* (CHSE), *Chrysemys dorsalis* (CHDO), *Graptemys pseudogeographica* (GRPS), *Kinosternon subrubrum* (KISU), *Macrochelys temminckii* (MATE), *Pseudemys concinna* (PSCO), *Sternotherus odoratus* (STOD), and *Trachemys scripta* (TRSC).

Table 1. Mean catch per unit effort (CPUE) across all turtle species and species richness by the different hoop and mesh size. Individuals were captured from April through October 2016–2018 in western Tennessee, USA.

Trap type	Hoop size (cm)	Mesh size (cm)	Mean CPUE	Species richness
Small hoop-small mesh	91.44	3.81	0.195 ± 0.343	10
Large hoop-small mesh	121.92	3.81	0.102 ± 0.232	9
Small hoop-large mesh	91.44	10.16	0.140 ± 0.196	6

(i.e., small hoop-small mesh and large hoop-small mesh traps) were more effective at detecting small-bodied species than our large-meshed trap (i.e., small hoop-large mesh traps; Fig. 3). In general, CPUE did not vary among trap types for most species, except for *T. scripta*. We acknowledge that throat height variation across traps and through time could have influenced our CPUE results; however, this uncontrolled variation likely had little influence on our detection of smaller bodied species in our small-meshed hoop traps. If researchers do not account for variation in detection among these trap types, they may erroneously infer population and community trends among studies or sampling periods if they use various types of hoop traps—even if the trapping protocol otherwise remains consistent.

Detection probabilities for particular species varied among the trap types. Small hoop-small mesh traps were generally the most effective trap across species. Small hoop-small mesh traps have the smallest mesh size (i.e., 3.81 cm mesh) and consequently were the most effective at capturing small bodied species (*S. odoratus*), although they were also the most effective at capturing one of the largest species present in the study (*A. spinifera*). There was also some evidence that small hoop-small mesh traps are the most effective trap for other small bodied species (*C. dorsalis* and *K. subrubrum*),

however, due to limited observations ($n = 8$ and 2, respectively), we hesitate to make strong conclusions for these species. Except for *P. concinna* which was detected by all types with equal efficacy, most species present in the study showed some bias against at least one trap type. In general, our results indicated that using turtle traps with smaller mesh-size and smaller hoop diameter increases detection of small-bodied turtles, while still capturing larger bodied species as effectively as larger meshed traps. However, *C. serpentina*—a large-bodied species—was the only species whose detection probability was greater in larger meshed traps. Another study reported that larger traps (i.e., both mesh and diameter sizes) captured more and larger individuals of *C. serpentina* than smaller traps (Gulette et al. 2019). Although speculative, the higher density of twine in smaller mesh-sized traps might entangle the long claws and serrated shells of *C. serpentina* thus impeding trap entry in some instances.

Our study is one of the first to highlight the influence of mesh size on freshwater turtle detection and estimates of community composition and species richness. We supported our hypothesis that smaller mesh traps would capture a different community of freshwater turtles relative to larger mesh traps. For example, we only captured *Sternotherus odoratus* and *Kinosternon subrubrum* in traps with the smaller mesh size, likely because both species are small-bodied and can escape from the larger mesh used in some hoop traps (J. R. Ennen, Tennessee Aquarium Conservation Institute, personal observation). Although we found traps with smaller mesh size, regardless of trap diameter size, captured smaller bodied turtle species better than those with larger mesh, these results were based only on presence-absence data. The trap-type effect approached significance when we considered turtle abundance (i.e., CPUE) as our independent variable and showed a trend for small hoop-small mesh trap capturing more turtles per unit effort. We suspect that hoop net and mesh size might influence CPUE in other systems with different turtle species assemblages, and turtle biologists should test for this potential bias before drawing inferences from their data. Finally, we recognize our study was not fully designed to disentangle the effect of hoop net size and mesh size independently, and this issue warrants further investigation.

Turtles, in general, are imperiled globally and face multiple anthropogenic threats, such as climate change, habitat loss, and overexploitation (Rhodin et al. 2018). It is critical for managers to have accurate estimates of population- and community-level parameters for these imperiled species. Our results point to a simple, yet

Table 2. Mean ± standard deviation of catch per unit effort (CPUE) for each turtle species by trap type. Individuals were captured from April through October 2016–2018 in western Tennessee, USA. Abbreviations are as follows: *Apalone mutica* (APMU), *Apalone spinifera* (APSP), *Chrysemys dorsalis* (CHDO), *Chelydra serpentina* (CHSE), *Graptemys ouachitensis* (GROU), *Graptemys pseudogeographica* (GRPS), *Kinosternon subrubrum* (KISU), *Macrochelys temminckii* (MATE), *Pseudemys concinna* (PSCO), *Sternotherus odoratus* (STOD), and *Trachemys scripta* (TRSC). Trap sizes were as follows: small hoop-small mesh—91.44 cm hoop nets with 3.81 cm mesh, large hoop-small mesh—121.92 cm hoop nets with 3.81 cm mesh, and small hoop-large mesh—91.44 cm hoop nets with 10.16 cm mesh. Traps not sharing a letter are significantly different ($\alpha = 0.05$).

Species	Small hoop-small mesh	Large hoop-small mesh	Small hoop-large mesh
APMU	0.002 ± 0.014	0.000 ± 0.000	0.000 ± 0.000
APSP	0.051 ± 0.115	0.022 ± 0.073	0.037 ± 0.113
CHDO	0.006 ± 0.029	0.006 ± 0.029	0.002 ± 0.017
CHSE	0.125 ± 0.154	0.064 ± 0.126	0.173 ± 0.224
GROU	0.006 ± 0.043	0.004 ± 0.025	0.000 ± 0.000
GRPS	0.059 ± 0.164	0.072 ± 0.33	0.023 ± 0.061
KISU	0.002 ± 0.014	0.003 ± 0.021	0.000 ± 0.000
MATE	0.021 ± 0.133	0.005 ± 0.042	0.007 ± 0.033
PSCO	0.009 ± 0.047	0.009 ± 0.035	0.007 ± 0.024
STOD	0.060 ± 0.159	0.025 ± 0.116	0.000 ± 0.000
TRSC	1.801 ± 2.903	0.914 ± 1.754	1.293 ± 1.687

obvious, bias in turtle trapping methods that could misinform conservation efforts. Effectively, smaller mesh sized traps were better at detecting smaller-bodied species, which then changed community composition estimates for a site. Even within a species, larger mesh-sized nets have the potential to bias captures towards larger individuals (Lovich and Gibbons 1990, Gulette et al. 2019). Thus, management decisions for turtle communities or individual species, such as species of concern or harvested species, would be influenced disproportionately if larger mesh sized traps were the only method deployed. This is especially important for states such as Tennessee, Georgia, and Louisiana, USA, that still allow harvesting of certain turtle species. Therefore, it is vital for management that we understand how methodological differences could influence our estimates related to turtle populations and communities. We acknowledge that testing an implicit bias in sample methods is not a new concept, but is it often only acknowledged and never fully investigated. Many agencies do not provide the resources to evaluate different methods, thus undermining potential study outcomes. Therefore, an accurate assessment of methods, especially on imperiled taxa, should be required before management decisions are made on populations.

MANAGEMENT IMPLICATIONS

Managers need accurate and reliable data to make effective management decisions to protect species and their respective habitats. Without reliable data and inferences, managers might erroneously underestimate conservation value of habitats and misinterpret population-level trends in imperiled species. Our study complements other turtle studies (e.g., Ream and Ream 1966, Tesche and Hodges 2015) by demonstrating that bias in trap type, design, and now mesh size could produce different results across sites. Researchers and managers should carefully consider factors related to hoop net and mesh size before conducting field research or making comparisons across studies. For example, researchers might want to consider multiple hoop net and mesh sizes to capture freshwater turtles when the objective of the study is to characterize turtle reproduction or communities within a system. If the objective of the study is to estimate abundance and population trends of turtle species, researchers need to consider a standardized trapping protocol, including factors related to hoop net and mesh size. However, studies considering only a focal turtle species should also strongly consider mesh size as smaller individuals (hatchlings and juveniles) could be missed, resulting in underestimates of population size and skewed age or sex ratios. Models, such as mark-recapture and occupancy, are critical in understanding population trends and identifying habitat needs of threatened species. However, ignoring heterogeneity of detection caused by survey methods (i.e., trap and mesh size variation), especially in models estimating abundances and occupancy, could lead to underestimates of population size and bias occupancy rates (Royle 2006, Cubaynes et al. 2010). Failure to account for trap and mesh size potentially could lead to erroneous inferences related to population trends and changes in community diversity through time. For example, large meshed traps may underestimate species richness and recruitment potentially causing managers to overlook the conservation value of the site and habitat, or conclude diversity is declining within a community. Finally, wildlife managers are faced with many management challenges, such as lack of comparability across studies and heterogeneity of detection probability. Therefore, any range-wide population assessment of threatened or endangered turtle species should consider standardizing trap and mesh size; otherwise, caution is needed when comparing population and community parameters. Moreover, our data suggests that researchers should adapt trapping protocols to their focal species to maximize detection probabilities for more accurate population estimates.

LITERATURE CITED

Alivisatos, A. P., M. J. Blaser, E. L. Brodie, M. Chun, J. L. Dangl, T. J. Donohue, P. C. Dorrestein, J. A. Gilbert, J. L. Green, J. K. Jansson, R. Knight, M. E. Maxon, M. J. McFall-Ngai, J. F. Miller, K. S. Pollard, E. G. Ruby, S. A. Taha, and C. Unified Microbiome Initiative. 2015. MICROBIOME. A unified initiative to harness Earth's microbiomes. Science 350:507–508.

Bailey, L. L., and J. D. Nichols. 2009. Capture-mark-recapture, removal sampling and occupancy models. K. Dodd, editor. Amphibian Ecology and Conservation: A Handbook of Techniques. Oxford University Press, Oxford, UK.

Bisi, F., S. Newey, M. Nodari, L. A. Wauters, A. Harrison, S. Thirgood, and A. Martinoli. 2011. The strong and the hungry: bias in capture methods for mountain hares *Lepus timidus*. Wildlife Biology 17:311–316.

Bluett, R. D., E. M. Schauber, C. K. Bloomquist, and D. A. Brown. 2011. Sampling assemblages of turtles in Central Illinois: A case study of capture efficiency and species coverage. Transactions of the Illinois State Academy of Science 104:127–136.

Bovendorp, R. S., R. A. McCleery, and M. Galetti. 2017. Optimising sampling methods for small mammal communities in Neotropical rainforests. Mammal Review 47:148–158.

Brennan, K. E. C., J. D. Majer, and M. L. Moir. 2005. Refining sampling protocols for inventorying invertebrate biodiversity: influence of drift-fence length and pitfall trap diameter on spiders. Journal of Arachnology 33:681–702.

Brooks, M. E., K. Kristensen, K. J. van Benthem, A. Magnusson, C. W. Berg, A. Nielsen, H. J. Skaug, M. Maechler, and B. M. Bolker. 2017. glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modeling. The R Journal 9:378–400.

Brown, G. R., and I. M. Matthews. 2016. A review of extensive variation in the design of pitfall traps and a proposal for a standard pitfall trap design for monitoring ground-active arthropod biodiversity. Ecology and Evolution 6:3953–3964.

Burton, A. C., E. Neilson, D. Moreira, A. Ladle, R. Steenweg, J. T. Fisher, E. Bayne, S. Boutin, and P. Stephens. 2015. REVIEW: Wildlife camera trapping: a review and recommendations for linking surveys to ecological processes. Journal of Applied Ecology 52:675–685.

Cecala, K. K., J. M. Davenport, J. R. Ennen, and W. R. Fields. 2017. Amphibians and Reptiles. F. R. Hauer and G. A. Lamberti, editors. Methods in Stream Ecology. Academic Press, London, United Kingdom.

Cubaynes, S., R. Pradel, R. Choquet, C. Duchamp, J. M. Gaillard, J. D. Lebreton, E. Marboutin, C. Miquel, A. M. Reboulet, C. Poillot, P. Taberlet, and O. Gimenez. 2010. Importance of accounting for detection heterogeneity when estimating abundance: the case of French wolves. Conservation Biology 24:621–626.

Edwards, E., T. K. Pauley, and J. L. Waldron. 2016. Estimating spring salamander detection probability using multiple methods. Journal of Herpetology 50:126–129.

Etnier, D. A., and W. C. Starnes. 1993. Fishes of Tennessee. University of Tennessee Press, Knoxville, Tennessee.

Fournier, A. M. V., E. R. White, and S. B. Heard. 2019. Site-selection bias and apparent population declines in long-term studies. Conservation Biology 33:1370–1379.

Garig, D. F., J. R. Ennen, and J. M. Davenport. 2020. The effects of common snapping turtles on a freshwater food web. Copeia 108:132–139.

Gotelli, N. J., and R. K. Colwell. 2001. Quantifying biodiversity: procedures and pitfalls in the measurement and comparison of species richness. Ecology Letters 4:379–391.

Gulette, A. L., J. T. Anderson, and D. J. Brown. 2019. Influence of hoop-net diameter on capture success and size distribution of comparatively large and small freshwater turtles. Northeastern Naturalist 26:129–136.

Howell, H. J., D. G. McKnight, and R. A. Seigel. 2016. A novel method of collecting Spotted Turtles (*Clemmys guttata*). Herpetological Review 47:202–205.

Kellner, K. 2015. jagsUI: a wrapper around rjags to streamline JAGS analyses. R package version 1. <https://cran.r-project.org/web/packages/jagsUI>. Accessed 8 Feb 2021.

Kéry, M., and J. A. Royle. 2015. Applied Hierarchical Modeling in Ecology: Analysis of distribution, abundance and species richness in R and BUGS. Volume 1: Prelude and Statis Models. Academic Press, London, UK.

Koper, N., and R. J. Brooks. 1998. Population-size estimators and unequal catchability in painted turtles. Canadian Journal of Zoology 76:458–465.

Kordjazi, Z., S. Frusher, C. Buxton, C. Gardner, and T. Bird. 2016. The influence of mark-recapture sampling effort on estimates of rock lobster survival. PLoS One 11:e0151683.

Lenth, R., H. Singmann, J. Love, P. Buerkner, and M. Herve. 2020. emmeans: Estimated marginal means, aka least-squares means. R package version 1.4.8. <https://github.com/rvlenth/emmeans>. Accessed 9 Feb 2021.

Long, J. M., D. R. Stewart, J. Shiflet, D. Balsman, and D. E. Shoup. 2017. Bait type influences on catch and bycatch in tandem hoop nets set in reservoirs. Fisheries Research 186:102–108.

Lovich, J. E., and J. W. Gibbons. 1990. Age at maturity influences adult sex rati in the turtle *Malaclemys terrapin*. Oikos 59:126–134.

Lovich, J. E., J. R. Ennen, M. Agha, and J. W. Gibbons. 2018. Where have all the turtles gone, and why does it matter? BioScience 68:771–781.

Mali, I., D. J. Brown, J. R. Ferrato, and M. R. J. Forstner. 2014b. Sampling freshwater turtle populations using hoop nets: Testing potential biases. Wildlife Society Bulletin 38:580–585.

Mali, I., D. J. Brown, M. C. Jones, and M. R. J. Forstner. 2012. Switching bait as a method to improve freshwater turtle capture and recapture success with hoop net traps. Southeastern Naturalist 11:311–318.

Mali, I., A. Duarte, and M. R. J. Forstner. 2018. Comparison of hoop-net trapping and visual surveys to monitor abundance of the Rio Grande cooter (*Pseudemys gorzugi*). PeerJ 6:e4677.

Mali, I., D. Haynes, and M. R. J. Forstner. 2014a. Effects of bait type, bait age, and trap hours on capture success of freshwater turtles. Southeastern Naturalist 13:619–625.

Nall, I. M., and R. B. Thomas. 2009. Does method of bait presentation within funnel traps influence capture rates of semi-aquatic turtles? Herpetological Conservation and Biology 4:161–163.

Oksanen, J., F. G. Blanchet, M. Friendly, R. Kindt, P. Legendre, D. McGlinn, P. R. Minchin, R. B. O'Hara, G. L. Simpson, P. Solymos, et al. 2019. Vegan: Community ecology package. https://cran.r-project.org/web/packages/vegan/vignettes/intro-vegan.pdf. Accessed 9 Feb 2021.

Pechmann, J. H., D. E. Scott, R. D. Semlitsch, J. P. Caldwell, L. J. Vitt, and J. W. Gibbons. 1991. Declining amphibian populations: the problem of separating human impacts from natural fluctuations. Science 253:892–895.

Plummer, M. V. 1979. Collecting and marking. Pages 45–60 in M. Harless and H. Morlock, editors. Turtles: Perspectives and Research. J. Wiley and Sons, New York, New York, USA.

R Core Team. 2017. R: A language and environment for statistical computing. R Foundation for Statistical Computing. Vienna, Austria. https://www.R-project.org/. Accessed 7 Jun 2019.

Ream, C., and R. Ream. 1966. The influence of sampling methods on the estimation of population structure in painted turtles. American Midland Naturalist 75:325–338.

Rhodin, A. G. J., C. B. Stanford, P. P. V. Dijk, C. Eisenberg, L. Luiselli, R. A. Mittermeier, R. Hudson, B. D. Horne, E. V. Goode, G. Kuchling, et al. 2018. Global conservation status of turtles and tortoises (Order Testudines). Chelonian Conservation and Biology 17:135–161.

Richardson, C., T. Messa, and M. S. Mills. 2017. Testing the effectiveness of bait types on trapping freshwater turtles in Missouri. Herpetological Review 48:331–333.

Royle, J. A. 2006. Site occupancy models with heterogeneous detection probabilities. Biometrics 62:97–102.

Scott, A. F., and W. H. Redmond. 2019. Atlas of Reptiles of Tennessee. Austin Peay State University, The Center for Field Biology, Clarksville, Tennessee, USA. https://www.apsubiology.org/tnreptileatlas/. Accessed 9 Feb 2021.

Senar, J. C., M. J. Conroy, L. M. Carrascal, J. Domènech, I. Mozetich, and F. Uribe. 1999. Identifying sources of heterogeneity in capture probabilities: an example using the Great Tit *Parus major*. Bird Study 46:S248–S252.

Sun, C. C., A. K. Fuller, and J. A. Royle. 2014. Trap configuration and spacing influences parameter estimates in spatial capture-recapture models. PLoS One 9:e88025.

Tesche, M. R., and K. E. Hodges. 2015. Unreliable population inferences from common trapping practices for freshwater turtles. Global Ecology and Conservation 3:802–813.

Thomas, R. B., I. M. Nall, and W. J. House. 2008. Relative efficacy of three different baits for trapping pond-dwelling turtles in east-central Kansas. Herpetological Review 39:186–188.

Vogt, R. C. 2016. Freshwater Turtles. Pages 168–193 in K. Dood, editor. Reptile Ecology and Conservation: A Handbook of Techniques. Oxford University Press, Oxford, UK.

Associate Editor: Refsnider.

SUPPORTING INFORMATION

Additional supporting information may be found in the online version of this article at the publisher's web-site.