Transactions of the Illinois State Academy of Science
(2013) Volume 106, pp. 15-21

received 4/22/13
accepted 8/17/13

Estimating Occupancy of *Trachemys scripta* and *Chrysemys picta* with Time-Lapse Cameras and Basking Rafts: A Pilot Study in Illinois, USA

Robert D. Bluett[1] and Bradley J. Cosentino[2,3]
[1]Illinois Department of Natural Resources, One Natural Resources Way, Springfield, Illinois 62702, USA
e-mail: bob.bluett@illinois.gov
[2]Department of Natural Resources and Environmental Sciences, University of Illinois,
1102 S. Goodwin Avenue, Urbana, Illinois 61801, USA
[3]Department of Biology, Hobart and William Smith Colleges, 300 Pulteney Street, Geneva, New York 14456, USA
e-mail: cosentino@hws.edu

ABSTRACT

We evaluated time-lapse cameras aimed at man-made basking rafts (camera traps) by estimating probabilities of occupancy and detection for *Trachemys scripta* and *Chrysemys picta* at 15 isolated ponds or wetlands in three regions of Illinois. Evaluation of camera traps relied on comparisons with hoop nets and published accounts of relative abundances of target species. After accounting for imperfect detection, occupancy probabilities for *C. picta* were 0.75 (SE = 0.18) using hoop nets and 0.91 (SE = 0.09) using camera traps. Occupancy probabilities for *T. scripta* were 0.96 (SE = 0.42) using hoop nets and 0.71 (SE = 0.17) using camera traps. The most-supported model of detection with camera traps included region and date of survey for both species, whereas the top model of detection with hoop nets included region and trap effort for both species. Regional differences in occupancy and detection for both survey methods were consistent with reports of relative abundances of target species. Daily rates of detection with camera traps varied during the 20-day sampling period, but in a predictable manner described by a single covariate (date of survey). Environmental variables were uninformative for predicting detection probability. Costs of labor and travel were lower (at least half) for camera traps than hoop nets, which required three or more surveys per site given observed rates of occupancy and detection. Camera traps require more evaluation, but show promise as an efficient, relatively inexpensive, and minimally invasive method to assess presence-absence of species of freshwater turtles that bask aerially.

Key words: camera trap; basking; occupancy; *Trachemys scripta*; Red-eared Slider; *Chrysemys picta*; Painted Turtle; Illinois

INTRODUCTION

Camera traps are useful tools for studies of wildlife ecology (Cutler and Swann 1999). Applications are now commonplace because of technological improvements to cameras, reasonable costs, and innovative approaches for analyzing data (O'Connell et al. 2011; Cox et al. 2012). Logistical advantages are another attractive feature. For example, camera traps can collect data during a long period of time with few visits whereas capture devices require regular checks to uphold standards of animal welfare.

Few herpetological studies have employed camera traps because heat- and motion-sensitive triggers tend to perform poorly for cold-blooded, slow-moving and often diminutive subjects (Dorcas and Peterson 2012). Exceptions include observations of Gopher Tortoises (*Gopherus polyphemus*; Boglioli et al. 2003), Timber Rattlesnakes (*Crotalus horridus*; Sadighi et al. 1995) and Grassland Earless Dragons (*Tympanocryptis pinguicolla*; McGrath et al. 2012). Camera traps have also been used to study nesting ecology of crocodilians (Hunt and Ogden 1991; Kermeen and Lemnell 2000) and freshwater turtles (Doody and Georges 2000; Geller 2012).

We evaluated camera traps for detecting patterns of presence-absence of freshwater turtles that bask aerially. If successful, the method could be applied economically at large spatial scales to estimate geographic distribution, habitat needs, population trends, and metapopulation processes via occupancy modeling (e.g., Rizkalla and Swihart 2006; Cosentino et al. 2010). We compared camera traps to hoop nets by estimating probabilities of occupancy and detection for two species (*Trachemys scripta*, Red-eared Slider; *Chrysemys picta*, Painted Turtle) in three regions of Illinois. Our objectives were: 1) identify and correct causes of camera malfunctions, 2) evaluate protocols for placement of rafts, 3) determine whether detection probabilities were similar for camera traps and hoop nets by testing the hypothesis $\rho \geq 0.5$ for each species with both methods, and 4) evaluate effects of region, date of survey, capture effort, and environmental variables on detection probabilities. We chose a threshold of $\rho \geq 0.5$ because both target species are common in much of the state (Phillips et al. 1999). Comparison of survey methods allowed "soft validation" of camera traps (Rodda 2012). We discuss attributes of these and other survey methods relative to costs and statistical constraints for estimating occupancy at large spatial scales.

MATERIALS AND METHODS

Study areas. Richardson Wildlife Foundation is located in Lee County near West Brooklyn, Illinois, USA. Managers of this private, 719–ha property have restored much of the area, including 24 wetlands. We randomly chose seven of these wetlands for our study; an eighth was sampled because *Emydoidea blandingii*, an endangered species in Illinois, had been encountered there earlier in the year. Individual wetland areas varied from 0.81–4.73 ha; all had emergent vegetation (e.g., *Typha latifolia*, *Scirpus* spp.) around their perimeters and submergent vegetation (e.g., *Pot-*

amogeton spp.) in open-water areas. For brevity, we designate this area as "North".

Our second study area ("Central") was a 23-ha private property located near Springfield, Illinois. The area was developed for aquaculture and fee fishing but currently is idle. We chose four of 11 man-made ponds based on similarities in size (0.15–0.44 ha) and presence of natural rather than concrete shorelines. Creeping water primrose (*Ludwigia peploides*) grew near margins of ponds. Our last study area ("South") was in Union County near Ware, Illinois. This 12-ha private property has a man-made pond (0.12 ha) and two man-made wetlands (0.11–0.21 ha), all of which were used for our study. The pond was bordered by turf on one side and forest on the other; wetlands were bordered by native grasses and small trees (*Salix* sp. and *Acer* sp.). Features used for basking at our study areas included bare shoreline, deadwood, floating mats of vegetation, and concrete rubble.

Equipment. Tops of rafts were constructed from exterior plywood (Length [L] = 120.7 cm; Width [W] = 28.4 cm; Depth [D] = 1.9 cm). We drilled two 2.54 cm-diameter (Diam) holes in the top, which were centered and 10.2 cm from each end. Two cedar boards (D = 2.54 cm; W = 15.2 cm [nominal dimensions]; L = 120.7 cm) were cut at a 22.5° angle along one edge, which was affixed to the top of the raft with exterior screws and polypropylene glue to create two wooden ramps. We stapled a piece of charcoal-colored pet-resistant screening (New York Wire, Mt. Wolf, Pennsylvania, USA) to the top and wooden ramps to provide a good purchase for turtles climbing aboard the raft. A piece of high-density R-10 insulation (D = 5.1 cm) was cut, trimmed, and affixed to the bottom of each raft with foam-board adhesive (PL300; Henkel Corp., Rocky Hill, Connecticut, USA) for flotation. Two wire extensions (19-guage hardware cloth; 1.27 X 1.27-cm mesh; W = 25.4 cm; L = 95.3 cm) were affixed to tops of wooden ramps with poultry staples (L = 1.9 cm). When rafts were deployed, ends of extensions dipped 2–4 cm beneath the water line to assist turtles attempting to climb aboard rafts. We left enough room under the staples so wire extensions could be folded over the top of the raft for transport. During deployment, we used two elastic cords (L = 55.2 cm) to secure each lower corner of the wire ramp on one side of the raft to that on the opposing side. Approximate cost for all materials was $10 US per raft; two fiberglass fence poles (Diam = 1.27 cm; L = 1.82 m; $5 US each) were used to anchor rafts in place.

To monitor each basking raft (Fig. 1) we used Timelapse Plantcam™ (Wingscapes®, Alabaster, Alabama, USA) with four-megapixel resolution. We set cameras to take high resolution photos (2560 X 1920 pixels) at three-hour intervals between a daily wake-up time of 0900 h and daily sleep at 1600 h (i.e., 0900, 1200 and 1500 h). Cameras were set to imprint photos with lapse interval, location, date, and time. Focus distance was set to infinity. Each camera was mounted approximately 1.25 m above the surface of the water on a piece of metal conduit (Diam = 1.9 cm; L = 3 m) located 2 m from the leading edge of the raft. We used a ladder to mount cameras and adjust the field of view to capture the entire basking raft. Where needed, we used a post driver constructed from metal conduit (Diam = 2.54 cm; L = 1 m) with an end cap to anchor mounting poles in hard substrates. Cost of camera, mounting pole and memory card was approximately $90 US.

For comparative data to time-lapse cameras, we captured turtles at each pond using two single-throated hoop nets (D = 0.61 m) with 3.8 X 3.8-cm mesh. Each hoop net cost $68 US. Bait (fresh fish changed daily) was suspended in a mesh bag tied to the hoop farthest from the throat.

Sampling methods. Given our objectives, we modified protocols during the study to optimize performance of camera traps. At Central, we used a crossover design to evaluate models as decoys on basking rafts (Red-eared Slider, Safari Ltd., Miami Gardens, Florida, USA). Decoys did not appear to increase detection rates, so we quit this practice at other sites. At Central, two ball-and-joint camera mounts crept out of position during sampling. Later, we secured cameras in position by running a cable tie through two apertures on the back of the case and anchoring the fastened cable tie to the mounting pole with electrical tape. At Central, we positioned rafts perpendicular to and 2 m away from shore. At other sites, water was too shallow to deploy our gear effectively

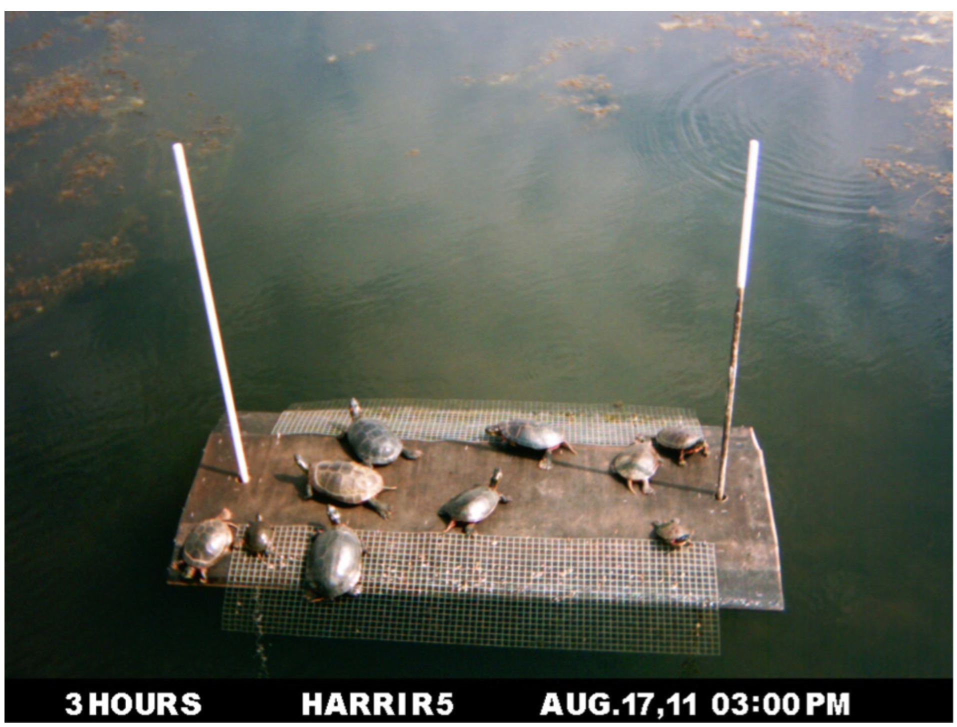

Fig 1. *Chrysemys picta* basking on a man-made raft and photographed with a time-lapse camera in Lee County, Illinois, USA, 2011.

at this distance, so we chose positions with deeper water regardless of distance from shoreline. At Central and South, we used two camera traps in each pond or wetland, placing them in quarters of the water body chosen randomly beforehand. At both locations, this restriction caused placement of two rafts where they were shaded by trees much of the day. We decided random placement of rafts within a water body was untenable, and use of two camera traps per wetland was unnecessary. At North, we used one raft per wetland, placing it opportunistically in full sunlight and open water deep enough for our gear (>40 cm).

Camera traps were deployed 13 May through 1 June 2011 (Central), 15 June through 4 July 2011 (South) and 10 through 29 August 2011 (North). At Central, hoop nets were set at each camera station (2 per pond) on 11 May 2011 and 2 June 2011 and checked the next day. We used the same protocol at South, but checked nets on three occasions (14 June, 6 July and 7 July 2011). At North, we set two nets per wetland and checked them twice (9 and 10 August 2011).

We used a drill to apply unique marks to marginal scutes of turtles captured in hoop nets. To avoid bias, a consultant with extensive herpetological experience (J. G. Palis, Palis Environmental Consulting, Jonesboro, Illinois) was hired to identify turtles in photographs using diagnostic features of heads, limbs, tails and carapaces. Individuals that could not be assigned confidently to species were classified as unknown.

We obtained data for weather variables (maximum temperature, total evaporation, total precipitation, total solar radiation) from meteorological stations of the Illinois Climate Network (http://www.isws.illinois.edu/warm/datatype.asp) at DeKalb, Illinois (North), Springfield, Illinois (Central), and Carbondale, Illinois (South).

Statistical analyses. We used occupancy modeling to estimate detection probabilities for camera trap and hoop net surveys at our study wetlands (N = 15). The program PRESENCE (v. 3.1) was used to build single-season models of detection probability (ρ) for each species and sampling method based on repeated surveys within wetlands (MacKenzie et al. 2006). Separate model sets were constructed for each sampling method. For camera trapping, a repeated survey was defined as all photos taken on a single day in a single wetland (North = 3 photos; Central and South = 6 photos). All wetlands were surveyed for 20 days. For trapping, a repeated survey was defined as a group of hoop traps in a single wetland open for one night. We surveyed wetlands for either two (North and Central) or three (South) days. Data for a third survey at wetlands in North and Central regions were coded as missing observations. All 15 wetlands were used in the analysis for *C. picta*, whereas only wetlands from the Central and South regions (N = 7) were used for *T. scripta*, as this species is generally not found in Lee County (B.J. Cosentino, unpublished data) and was not detected in North wetlands during our study.

Covariates were used to model variation in ρ among sites. We expected significant variation in abundance among regions, so we included an effect of region on ρ in all models. For camera trapping, we also evaluated models that included date of survey, maximum air temperature, total evaporation, total precipitation, and total solar radiation. For hoop nets, we evaluated models that included date of survey, trap effort, and maximum air temperature. Trap effort was calculated as the number of trap-hours for each wetland. For both camera trapping and hoop nets, we limited models of ρ to include only a single covariate beyond the inclusion of an effect of region. Occupancy probability (Ψ) was held constant in all models because of the limited number of wetlands in our study.

We used the Akaike Information Criterion corrected for small sample size (AIC_C) to rank the relative support of models for each combination of species and sampling method (Burnham and Anderson 2002). For each model, *i*, we estimated AIC_C differences ($\Delta AIC_C = AIC_{C,i}$ - minimum AIC_C) and Akaike weights (w_i). Models were considered to have competitive support when $\Delta AIC_C \leq$ two.

RESULTS

For *C. picta*, we captured 23 individuals (0.70 individuals/survey) at nine wetlands (naive Ψ = 0.6) using hoop nets, and

Table 1. Sampling effort, daily detections, and numbers of basking turtles observed with camera traps at 15 sites in Illinois, USA, 2011.

		Daily dectections (presence) and no. of turtles observed[a]					
		Trachemys scripta		*Chrysemys picta*		Unidentified emydids	
Site	No. cameras	Days	No. turtles	Days	No. turtles	Days	No. turtles
South	2	20	249	2	2	12	27
South	2	20	142	0	0	14	40
South	2	0	0	0	0	0	0
Central	2	10	58	8	14	6	6
Central	2[b]	0	0	0	0	0	0
Central	2[c]	12	23	4	9	0	0
Central	2	15	65	11	25	9	10
North	1	0	0	20	296	9	10
North	1	0	0	15	49	0	0
North	1	0	0	20	197	1	1
North	1	0	0	17	62	0	0
North	1	0	0	5	5	0	0
North	1	0	0	20	273	5	5
North	1	0	0	12	21	0	0
North	1	0	0	16	116[d]	0	0

[a]Observations of turtles are not independent; the same individual could have been photographed on multiple occasions during a day and on multiple days during a sampling session
[b]One camera inoperable for 9 days
[c]One camera inoperable for 2 days
[d]A portion of the basking raft (<25%) was out of view for 10 days after water level dropped; daily detections unaffected (all positive)

Table 2. Mean detection probabilities of *Chrysemys picta* and *Trachemys scripta* using camera traps and hoop traps in three regions in Illinois, USA.

Species	Trap Type	Region	Mean Detection Probability	SE
C. picta	Camera	North	0.78	0.01
		Central	0.38	0.02
		South	0.04	0.00
C. picta	Hoop	North	0.56	0.02
		Central	0.58	0.04
		South	0.13	0.01
T. scripta	Camera	Central	0.62	0.03
		South	1.00	-
T. scripta	Hoop	Central	0.69	0.01
	South	South	0.78	0.03

Table 3. Model selection statistics for detection probability of *Chrysemys picta* and *Trachemys scripta* using camera traps and hoop traps in 15 isolated ponds and wetlands in Illinois, USA. Main effects are included for each model. Summary statistics for each model include the relative difference between model AIC_C and AIC_C for the best model (ΔAIC_C), Akaike weights (w_i), the number of parameters estimated (K), and twice the negative log-likelihood (-2*l*).

Species	Trap Type	Model	ΔAIC_C	ω_i	K	2*l*
C. picta	Camera	R + D	0.00	0.98	5	255.28
		R + T	8.43	0.01	5	263.71
		R + Ev	12.04	0.00	5	267.32
		R	14.90	0.00	4	272.24
		R + S	15.16	0.00	5	270.44
		R + P	15.37	0.00	5	270.65
C. picta	Hoop	R + Ef	0.00	0.39	5	36.22
		R	0.08	0.38	4	39.09
		R + D	2.04	0.14	5	38.26
		R + T	2.85	0.09	5	39.07
T. scripta	Camera	R + D	0.00	0.98	4	63.49
		R + T	7.54	0.02	4	71.03
		R + Ev	14.74	0.00	4	78.23
		R + S	17.20	0.00	4	80.69
		R	22.65	0.00	3	88.26
		R + P	23.59	0.00	4	87.08
T. scripta	Hoop	R	0.00	0.50	3	20.11
		R + Ef	1.03	0.30	4	17.66
		R + T	2.87	0.12	4	19.50
		R + D	3.43	0.09	4	20.06

R = Region, D = Date, T = Temperature, Ev = Evaporation, P = Precipitation, S = Solar Radiation, Ef = Trap Effort

counted 1069 individuals (3.56/survey) at 12 wetlands (naive Ψ = 0.8) using camera traps (Table 1); in the latter case, some individuals were undoubtedly counted multiple times during the 20-day survey period. After accounting for imperfect detection, occupancy probabilities for *C. picta* were 0.75 (SE = 0.18) using hoop nets and 0.91 (SE = 0.09) using camera traps. For *T. scripta*, we captured 45 individuals (2.65 individuals/survey) at 6 wetlands (naive Ψ = 0.86) using hoop traps, and we counted 537 individuals (1.79/survey) at 5 wetlands (naive Ψ = 0.71) using camera traps (Table 1). After accounting for imperfect detection, occupancy probabilities for *T. scripta* were 0.96 (SE = 0.42) using hoop nets and 0.71 (SE = 0.17) using camera traps.

With a few exceptions, mean daily detection probabilities were high (> 0.5) for both methods and species (Table 2). The most-supported model for detection with camera traps included region and date of survey for both species (Table 3). Mean daily detection probabilities were greatest at North for *C. picta* and South for *T. scripta* (Table 2). Detection probability increased over time for *C. picta* (Fig. 2; beta estimate = 4.49, SE = 1.11) and *T. scripta* (Fig. 3; beta estimate = 11.56, SE = 2.47). At South, we detected *T. scripta* during every sampling occasion with camera traps (Fig. 3).

Using hoop nets, the most-supported model of detection probability included region and trap effort for *C. picta* and region for *T. scripta* (Table 3). However, trap effort was included in a competitive model for *T. scripta* (Table 3). Mean daily detection probabilities were equally high at North and Central for *C. picta* and South for *T. scripta* (Table 2). Detection probability was related positively to trap effort for both species (beta estimates: *C. picta* = 1.42, SE = 1.03; *T. scripta* = 0.73, SE = 0.56). Environmental variables were generally uninformative for both camera traps and hoop traps.

DISCUSSION

Estimates of occupancy are robust because they account for imperfect detection of target species (Mazerolle et al. 2007). As a rule of thumb, $\rho > 0.5$ is desirable while $\rho > 0.15$ is acceptable (MacKenzie et al. 2006; O'Connell et al. 2006). By these standards, camera traps and hoop nets performed

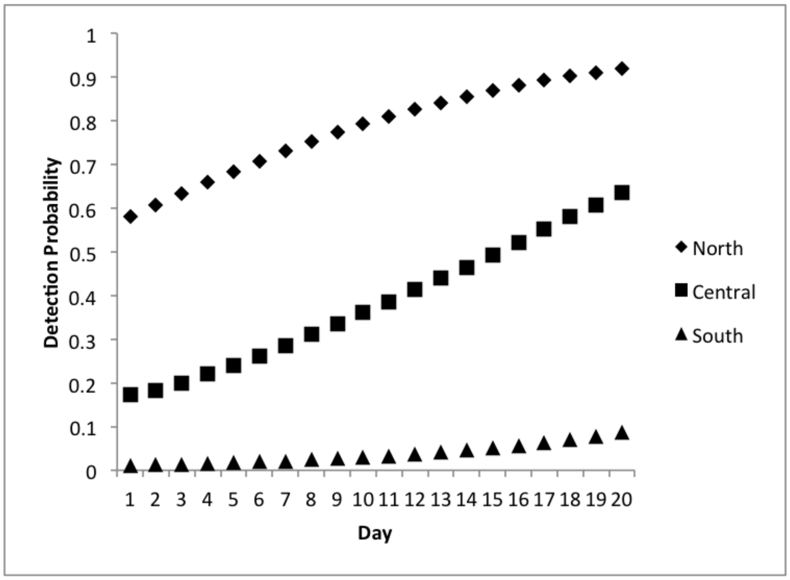

Fig. 2. Relationship of predicted detection probability of *Chrysemys picta* to date of survey (1-20). Julian dates for surveys were 13 May through 1 Jun 2011 for Central, 15 Jun through 4 Jul 2011 for South, and 10 through 29 Aug 2011 for North.

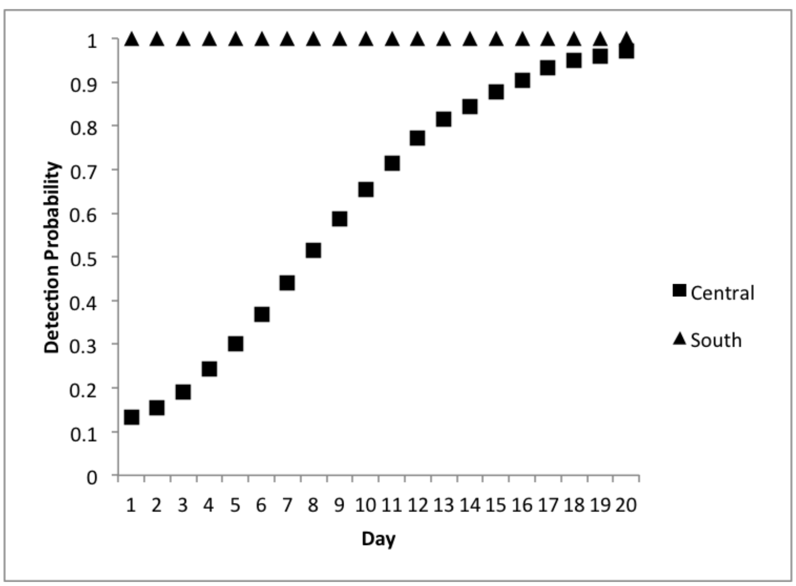

Fig. 3. Relationship of predicted detection probability of *Trachemys scripta* to date of survey (1-20). Julian dates for surveys were 13 May through 1 Jun 2011 for Central and 15 Jun through 4 Jul 2011 for South.

similarly for detection of target species.

Our data did not provide a direct test of the value of camera traps for describing spatial aspects of population ecology. However, we obtained a qualitative check on performance by comparing our results to reports of relative abundances of target species. Red-eared Sliders are rare or absent in northern Illinois, but abundant in the southern part of the state (Smith 1961; Readel et al. 2008). Painted Turtles occur throughout Illinois, but are more common in North than South (Dreslik and Phillips 2005). Thus, strong support for region in occupancy models was encouraging. We suspect regional differences in detection probabilities for camera traps also reflected our refinement of protocols. Presumably, this contributed to greater detection of *C. picta* at North, where we had one camera and three photos per wetland per day, than Central or South, where we had two cameras and six photos per wetland per day. In keeping with our protocol at North, we recommend placing one camera trap in deep (> 40 cm), open water with full sunlight most or all day when sampling small (< 5 ha) bodies of water.

Environmental variables were uninformative in models of detection with camera traps. This seems counter-intuitive, but we note varying degrees of support for effects of ambient conditions on basking behavior (Crawford et al. 1983; Enge and Wallace 2008; Selman and Qualls 2011). One possibility is that temporal differences in basking behavior were masked by overriding effects of acclimation to basking rafts during the 20-day surveillance period.

Our protocol of setting two hoop nets per wetland for two or three trap-nights affected detection of target species. This was not surprising, as observed rates of occupancy and detection suggested three or more surveys per site were best for sampling *T. scripta* and *C. picta* with hoop nets (MacKenzie and Royle 2005). Thus, hoop nets would have required at least twice as many trips (one to set gear and three to tend it) as camera traps (one to set gear and one to retrieve it). Savings on labor and travel must be weighed against costs of gear because hoop nets retrieved from one site after sampling could be deployed at four more during a 20-day period.

Variability caused by sampling methods can be problematic for occupancy modeling, which assumes rates of detection are constant among sites and visits unless heterogeneity is described by covariates (Pollock et al. 2002). Camera traps allowed collection of data simultaneously and consistently among sites and visits. This is a clear advantage over manual methods of sampling. For example, timing of visits to tend capture devices (and accrued effort) often varies with numbers of turtles processed earlier in the day. Camera traps also avoid heterogeneous rates of detection caused by multiple observers (e.g., differences in experience or acuity), behaviors of target species (e.g., differences in flushing distances), and other methodological sources of bias.

Our rate of non-identification (6.2%) was similar to Lindeman's (2000) surveys with spotting scopes (4.5%) at a site where *Graptemys* spp. and *T. scripta* dominated the assemblage. Gooley et al. (2011) identified 63% of emydid turtles to genus and 55% to species during visual surveys whereas Enge and Wallace (2008) identified 41% of turtles to genus and 58% to species. We suspect ability to identify species with photographs taken by camera traps would also vary with complexity of assemblages and subtlety of diagnostic traits of members. Therefore, we recommend using eight-megapixel cameras marketed after we purchased our gear (e.g., Wingscapes® TimelapseCam 8.0™). Other suggestions include securing cameras to mounting poles to maintain positioning (see methods section), using cleansers designed specifically for plastics when treating clear ports on camera housings, and possibly raising the height of cameras to obtain a better field of view of rafts. Portions of rafts were not visible when water levels changed > 25 cm after cameras were positioned; this could be problematic for some applications (e.g., tidal and possibly lotic habitats).

Our study appears to be the first to use camera traps to detect freshwater turtles in aquatic settings. This was possible, in part, because we used time-lapse triggers to obtain simultaneous estimates of presence-absence for *T. scripta* and *C. picta* at multiple sites. Use of basking rafts was not innovative (e.g., Alvarez 2006), but provided an effective and standard means of attracting turtles into range of cameras. Camera traps need more evaluation, but show promise as an efficient, relatively inexpensive, and minimally invasive method to assess presence-absence and other traits of turtles that bask aerially.

ACKNOWLEDGMENTS

We thank Jeane Boosinger, Terry Moyer and Dan Woolard for access to their properties. Dan Woolard and Mark Alessi assisted with sampling. Two reviewers provided helpful comments. Activities were funded by the Illinois Department of Natural Resources, authorized by state law (515 Illinois Compiled Statutes 5/20-100), and complied with standards of animal welfare adopted by the American Society of Ichthyologists and Herpetologists.

LITERATURE CITED

Alvarez, J.A. 2006. Use of artificial basking substrate to detect and monitor pacific pond turtles (*Emys marmota*). Western North American Naturalist 66:129–131.

Boglioli, M.D., C. Guyer, and W.K. Michener. 2003. Mating opportunities of female Gopher Tortoises, *Gopherus polyphemus*, in relation to spatial isolation of females and their burrows. Copeia 2003:846–850.

Burnham, K.P., and D.R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Second edition. Springer, New York, New York, USA.

Cosentino, B.J., R.L. Schooley, and C.A. Phillips. 2010. Wetland hydrology, area, and isolation influence occupancy and spatial turnover of the painted turtle, *Chrysemys picta*. Landscape Ecology 25:1589–1600.

Cox, W.A., M.S. Pruett, T.J. Benson, S.J. Chiavacci, and F.R. Thompson III. 2012. Development of camera trap technology for monitoring nests. Pp. 185–210 *In* Video Surveillance of Nesting Birds. Ribic, C.A., F.R. Thompson III, and P.J. Pietz (Eds.). Studies in Avian Biology (no. 43), University of California Press, Berkeley, California, USA.

Crawford, K.M., J.R. Spotila, and E.A. Standora. 1983. Operative environmental temperatures and basking behavior of the turtle *Pseudemys scripta*. Ecology 64:989–999.

Cutler, T.L., and D.E. Swann. 1999. Using remote photography in wildlife ecology: a review. Wildlife Society Bulletin 27:571–581.

Doody, J.S., and A. Georges. 2000. A novel technique for gathering turtle nesting and emergence phenology data. Herpetological Review 31:220–222.

Dorcas, M.E., and C.R. Peterson. 2012. Automated data acquisition. Pp. 61–68 *In* Reptile Diversity: Standard Methods for Inventory and Monitoring. McDiarmid, R.W., M.C. Foster, C. Guyer, J.W. Gibbons and N. Chernoff (Eds.). University of California Press, Berkeley, California, USA.

Dreslik, M.J., and C.A. Phillips. 2005. Turtle communities in the upper Midwest, USA. Journal of Freshwater Ecology 20:149–164.

Enge, K.M., and G.E. Wallace. 2008. Basking survey of map turtles (*Graptemys*) in the Choctawhatchee and Ochlokonee rivers, Florida and Alabama. Florida Scientist 71:310–322.

Geller, G. 2012. Reducing predation of freshwater turtle nests with a simple electric fence. Herpetological Review 43:398–405.

Gooley, A.C., H.J. Stanton, C.J. Bartkus, and T.K. Pauley. 2011. The distribution of aquatic turtles along the Ohio, Great Kanawha, and Little Kanawha rivers, West Virginia, with emphasis of *Graptemys ouachitensis* and *G. geographica*. Ohio Biological Survey Notes 3:21–28.

Hunt, R.H., and J.J. Ogden. 1991. Selected aspects of the nesting ecology of American alligators in Okefenokee Swamp. Journal of Herpetology 25:448–453.

Kermeen, Q., and P.A. Lemnell. 2000. On the use of camera traps to study crocodilian nesting behavior. Crocodile Specialist Group Newsletter 19:17–19.

Lindeman, P.V. 2000. Resource use of five sympatric turtle species: effects of competition, phylogeny, and morphology. Canadian Journal of Zoology 78:992–1008.

MacKenzie, D.I., and J.A. Royle. 2005. Designing occupancy studies: general advice and allocating survey effort. Journal of Applied Ecology 42:1105–1114.

MacKenzie, D.I., J.D. Nichols, J.A. Royle, K.H. Pollock, L.L. Bailey, and J.E. Hines. 2006. Occupancy Estimation and Modeling. Elsevier, Inc., Boston, Massachusetts, USA.

Mazerolle, M.J., L.L. Bailey, W.J. Kendall, J.A. Royle, S.J. Converse, and J.D. Nichols. 2007. Making great leaps forward: accounting for detectability in herpetological field studies. Journal of Herpetology 41:672–689.

McGrath, T., D. Hunter, W. Osborne, and S.D. Sarre. 2012. A trial use of camera traps detects the highly cryptic and endangered grassland earless dragon *Tympanocryptis pinguicolla* (Reptilia: Agamidae) on the Monaro Tablelands of New South Wales, Australia. Herpetological Review 43:249–252.

O'Connell, A.F. Jr., N.W. Talancy, L.L. Bailey, J.R. Sauer, R. Cook, and A.T. Gilbert. 2006. Estimating site occupancy and detection probability parameters for meso- and large mammals in a coastal ecosystem. Journal of Wildlife Management 70:1625–1633.

O'Connell, A.F., J.D. Nichols, and K.U. Karanth (Eds.). 2011. Camera Traps in Animal Ecology: Methods and Analyses. Springer, New York, New York, USA.

Phillips, C.A., R.A. Brandon, and E.O. Moll. 1999. Field Guide to the Amphibians and Reptiles of Illinois. Illinois Natural History Survey Manual 8. Illinois Natural History Survey, Champaign, Illinois, USA.

Pollock, K.H., J.D. Nichols, T.R. Simons, G.L. Farnsworth, L.L. Bailey, and J.R. Sauer. 2002. Large scale wildlife monitoring studies: statistical methods for design and analysis. Environmetrics 13:105–119.

Readel, A.M., C.A. Phillips, and M.J. Wetzel. 2008. Leech parasitism in a turtle assemblage: effects of host and environmental characteristics. Copeia 2008:227–233.

Rizkalla, C.E., and R.K. Swihart. 2006. Community structure and differential responses of aquatic turtles to agriculturally induced habitat fragmentation. Landscape Ecology

21:1361–1375.

Rodda, G.H. 2012. Statistical properties of techniques and validation. Pp. 197–203 *In* Reptile Diversity: Standard Methods for Inventory and Monitoring. McDiarmid, R.W., M.C. Foster, C. Guyer, J.W. Gibbons, and N. Chernoff (Eds.). University of California Press, Berkeley, California, USA.

Sadighi, K., R.M. DeGraaf, and W.R. Danielson. 1995. Experimental use of remotely-triggered-cameras to monitor occurrence of timber rattlesnakes (*Crotalus horridus*). Herpetological Review 26:189–190.

Selman, W., and C.P. Qualls. 2011. Basking ecology of the yellow-blotched sawback (*Graptemys flavimaculata*), an imperiled turtle species of the Pascagoula River system, Mississippi, United States. Chelonian Conservation and Biology 10:188–197.

Smith, P.W. 1961. The Amphibians and Reptiles of Illinois. Illinois Natural History Survey Bulletin 28. Illinois Natural History Survey, Urbana, Illinois, USA.