Multiscale Habitat Factors Influence the Occupancy and Turnover of the Suburban Herpetofauna of Chicago, Illinois, USA

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Abstract.—Effective management and conservation of urban herpetofauna is challenging because little is known about the distribution, population dynamics, or habitat use of most species in urban areas. To address this knowledge gap, we studied the occupancy dynamics of seven common species in the Chicago, Illinois, USA, metropolitan area using artificial cover objects and aquatic trapping. From 2009 to 2012, we detected seven focal species 899 times, ranging from 49 detections of Eastern Tiger Salamanders (*Ambystoma tigrinum*) to 223 detections of Painted Turtles (*Chrysemys picta*). We used single-species multi-season (dynamic) occupancy models and a suite of environmental covariates to determine how natural (e.g., canopy cover) and urban (e.g., road density) habitat features influence rates of occupancy and turnover at local (n = 159 monitoring points) and landscape (n = 27 preserves) scales while explicitly incorporating detectability. Our results suggest that natural habitat features were generally better predictors of occupancy and turnover than anthropogenic features. At the local scale, extinction was more common than colonization, while at the landscape scale, colonization was more common than extinction. This study provides important baseline information to guide future research and management decisions and we recommend managing urban preserves for habitat diversity and connectivity to maximize herpetofaunal diversity.

Key Words.—*Ambystoma*; *Chelydra*; *Chrysemys*; detection probability; *Lithobates*; *Thamnophis*; urban ecology

Introduction

Amphibians and reptiles are of conservation concern in urban landscapes because urbanization can greatly alter their distribution, diversity, movement, genetic structure, reproduction, and survival through habitat fragmentation, alteration, and degradation (Germaine and Wakeling 2001; Andrews et al. 2005; Noël et al. 2007; Barrett and Guyer 2008; Gangloff et al. 2017). Further, herpetofauna generally have low mobility and dispersal capabilities and can be sensitive to environmental pollutants, making them particularly suspectable to the effects of habitat fragmentation and degradation. Thus, areas of intense urbanization often have depauperate herpetofaunal communities due to fragmented, homogeneous, and degraded habitats (Cushman 2006; Andrews et al. 2008; Hamer and McDonnell 2008); however, where native habitats remain in urban areas, some species can persist, especially habitat generalists such as American Toads (*Anaxyrus americanus*) and

American Bullfrogs (*Lithobates catesbeianus*; Rubbo and Kiesecker 2005). Therefore, the challenge remains to identify how much and what kind of urban habitat is needed for native herpetofauna to persist and how the urban environment is impacting them.

Quality aquatic and upland habitats are critical for turtle, salamander, snake, and frog populations, particularly in urban environments, because both are needed for many members of these clades to complete their life cycles (Guerry and Hunter 2002; Gibbons 2003; Semlitsch and Bodie 2003). Aquatic turtles are particularly sensitive to urban environments due to their slow reproductive rates, coupled with high nest depredation and high road mortality (Congdon et al. 1994; Gibbs and Shriver 2002; Marchand and Litvaitis 2004a). Urban turtle assemblages, however, can persist in abundance if wetlands are productive, not highly fragmented, and are in proximity to adequate nesting habitat (Marchand and Litvaitis 2004b), although this is complicated by hydroperiods (Cosentino et

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al. 2010). Some habitats in urban areas, such as golf course wetlands, can support ostensibly healthier turtle populations than nearby non-golf course wetlands (Winchell and Gibbs 2016).

Similarly, the low mobility of salamanders makes them sensitive to fragmentation in human-dominated landscapes. Salamanders can quickly become isolated by fragmentation and genetic diversity can diminish (Noël et al. 2007; Greenwald et al. 2009). In addition, the permeable skin of salamanders makes them susceptible to environmental toxins and urban runoff. For example, embryonic and larval pool-breeding ambystomatids are at risk of reduced survival if road de-icing agents reach certain threshold concentrations (Karraker et al. 2008). The occurrence of pool-breeding salamanders is closely related to the amount of upland forested habitat available during non-breeding seasons (Semlitsch and Bodie 2003), while forest composition and microhabitat characteristics are important to their persistence (Belasen et al. 2013).

In human-dominated landscapes with remnant native habitats, snake species abundance and richness is positively correlated with patch size but can be confounded by patterns of species co-occurrence (Kjoss and Litvaitis 2001a; Steen et al. 2014). Road networks are largely detrimental to snake populations (Andrews et al. 2008; Fahrig and Rytwinski 2009). For an aquatic snake assemblage in the Midwestern U.S. (Indiana), roads reduced the most mobile species by as much as 21% (Roe et al. 2006) while some snake species may actively avoid roads (Shepard et al. 2008a; Robson and Blouin-Demers 2013). Because snake species cover a broad ecological breadth, landscape occupancy varies by the species in question (Steen et al. 2012) but both local- and landscape-scale habitat features are important to consider (Jenkins et al. 2009; Sutton et al. 2017).

Compared to other wildlife taxa inhabiting urban areas such as birds and mammals, herpetofauna are understudied (McKinney 2008), which limits the ability of managers to make the best conservation decisions (Bury 2006). Most studies of herpetofauna in urban areas focus on measures such as species diversity and richness as related to habitat types and proximity to urban development, while often failing to account for detection probabilities in their inference (Dickman 1987; Bodie et al. 2000). However, accounting for imperfect detection in herpetology has become increasingly common in recent years, allowing for explicit estimation of detectability (Durso and Seigel 2015), and therefore more accurate estimates of population parameters (MacKenzie et al. 2006; Mazerolle et al. 2007).

To address the lack of studies examining patterns of site occupancy and spatial turnover in human-dominated areas, we quantified local and landscape factors impacting seven species of herpetofauna in a suburban preserve system. Blue-spotted Salamanders

(*Ambystoma laterale*), Eastern Tiger Salamanders (*A. tigrinum*), Common Gartersnakes (*Thamnophis sirtalis*), Green Frogs (*L. clamitans*), Northern Leopard Frogs (*L. pipiens*), Painted Turtles (*Chrysemys picta*), and Snapping Turtles (*Chelydra serpentina*) are common species found throughout the Chicago metropolitan area, but it is poorly known how common species are impacted by urbanization (Andrews et al. 2008). Specifically, we used multi-season occupancy modeling (MacKenzie et al. 2003) to: (1) identify potential environmental, temporal, or survey design induced factors causing heterogeneity in detectability; (2) associate herpetofauna site occupancy and turnover rates with habitat features (e.g., canopy and herbaceous cover) and anthropogenic features (e.g., distances to urban structures and roads) at a local scale; and (3) identify landscape characteristics (e.g., native landcover metrics, patch isolation) and anthropogenic disturbances (e.g., road and urban density) most influential to occupancy dynamics and turnover rates of herpetofauna at the landscape scale.

MATERIALS AND METHODS

Study site.—We conducted our study approximately 40 km northwest of downtown Chicago in suburban Lake County, Illinois, USA (Fig. 1). Lake County is part of the Chicago metropolitan area, which is the third largest metropolitan area in the USA with nearly 10,000,000 inhabitants (U.S. Census Bureau. 2017. QuickFacts Lake County, Illinois. U.S. Census Bureau. Available from https://www.census.gov/quickfacts. [Accessed 25 July 2018]). Although Lake County maintains many of its historic natural communities, the county is highly developed and is the third most populous county in Illinois (population > 703,000) with a population density > 600 people/km² (U.S. Census Bureau. 2017. *op cit.*).

Within this suburban landscape, the Lake County Forest Preserve District (LCFPD) manages a network of 55 preserves totaling 126 km² for natural resources and outdoor recreation. Preserve size ranges from 8–585 ha, with a mean area of 198 ± (SE) 19 ha. Dominant vegetative landcover varies between preserves but consists of forests and woodlands (28%), wetlands (18%), old-field cool-season grasslands (16%), croplands (10%), shrublands (10%), prairies (7%), and savannas (5%) with the remaining 6% consisting of other cover types. Annual precipitation in nearby Chicago averaged 101 cm from 2009–2012 with an average annual temperature of 10.7° C (https://www.ncdc.noaa.gov/cdo-web).

Field methods.—We surveyed all 55 preserves for herpetofauna from 2009 to 2012 as part of a larger multi-taxon wildlife monitoring program. We sampled
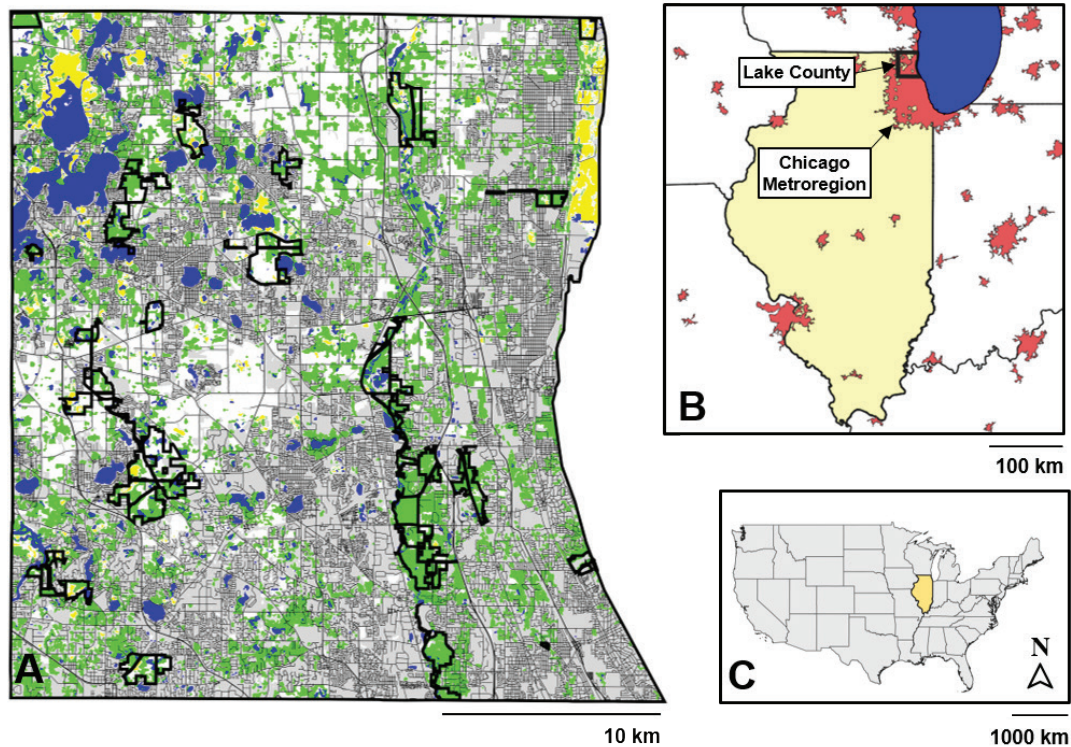
FIGURE 1. Lake County is a highly urbanized suburb of Chicago, Illinois, USA. (A) Boundaries of Lake County Forest Preserve District analyzed in this study (thick black lines), roads (dark gray lines) and major landcover classes pooled from the from the 2011 National Landcover Database (light gray = developed, green = forest, yellow = freshwater emergent vegetation, blue = open water, white = other, but mostly row crop agriculture; Homer et al. 2011). (B) Location of Lake County within Illinois (beige) and in relation to the Chicago metro region (labeled) and other major US Census designated urban areas (maroon). (C) Location of Illinois (beige) within the United States.

preserves every other year (n = 26) or every 4 y (n = 27) based on a priori habitat management and restoration goals (Gary A. Glowacki and Tim S. Preuss, unpubl. report). We only included preserves sampled every other year in our multi-season occupancy analyses (Table 1). We used ArcMap 10 (Esri, Redlands, California, USA) to randomly select monitoring points (n = 159) within each preserve at an average density of 1/37.5 ha, spaced ≥ 400 m apart, with a minimum of two points per preserve. Each preserve (and all points within) was surveyed every other year (starting in 2009 or 2010) for one week in spring (mid-April through June; Season 1) and one week in summer (mid-July through August; Season 2). We selected survey locations randomly using ArcMap to avoid issues of convenience sampling (Anderson 2001).

At each preserve, we used artificial coverboards to survey salamanders and snakes and used aquatic traps to survey turtles and frogs at sites with standing water. We deployed two sets of two coverboards at each point; each set contained one large (90 × 120 cm) and one small (30 × 90 cm) plywood board. To allow for acclimation, we deployed coverboards one season prior to initial sampling (Grant et al. 1992). We defined a survey as

a check of all four coverboards. Because we surveyed coverboards twice each season (never on consecutive days), there were four surveys per monitoring point per year (Table 2).

We deployed minnow traps (trap diameter < 25 cm), small hoop nets (entrance diameter < 50.8 cm), and/ or large hoop nets (entrance diameter > 50.8 cm) in standing water (> 50.8 cm deep) within a 100 m buffer of monitoring points. We baited hoop nets with canned sardines, and we replaced bait as necessary during trap checks. Depending on the available amount and depth of water within the buffer, we used variable combinations of the trap types; however, whenever possible, we deployed more than one trap of each type. During the sampling week, we set traps on Monday, checked traps daily, and removed them on Friday. We did not bait minnow traps. We considered that each daily check of all traps at a plot was a survey, resulting in four surveys per season and thus eight surveys per monitoring point per year (Table 2).

Modeling covariates.—We used 14 survey-specific covariates and 15 site-specific covariates in our single-species multi-season occupancy models (Supplemental

TABLE 1. Sampling (preserve name, years sampled, number of monitoring points) and site characteristics (preserve size and hectares of important landcover types) of Lake County Forest Preserves sampled for reptiles and amphibians from April-May and July-August 2009–2012 in Lake County, Illinois, USA. Monitoring points were randomly distributed at a density of 1/37.5 ha and spaced > 400 m apart with a minimum of two points/preserve. See Table 4 for landcover descriptions.

Preserve Name	Years Sampled	# Points	Size (ha)	Forest (ha)	Wetland (ha)	Prairie (ha)	Shrub (ha)	Urban (ha)
Ethel's Woods	2009/2011	5	189.1	86.3	15.7	0.0	11.1	0.0
Fort Sheridan	2009/2011	3	98.9	30.0	0.0	0.0	0.0	0.3
Grainger Woods	2009/2011	3	119.1	93.2	2.5	0.0	2.5	0.3
Grant Woods	2009/2011	11	439.5	19.3	33.3	6.5	155.8	0.5
Grassy Lake	2009/2011	7	273.8	63.9	90.3	9.4	51.7	0.4
Independence Grove	2009/2011	11	462.4	132.3	50.1	46.4	35.2	0.3
Lakewood North	2009/2011	12	522.2	118.7	0.7	0.0	7.9	0.3
Lakewood South	2009/2011	16	585.0	54.3	106.0	0.0	20.5	0.4
MacArthur Woods	2009/2011	5	199.7	197.9	0.0	0.0	0.0	0.1
Marl Flat	2009/2011	2	83.5	18.8	14.6	2.0	11.6	0.1
Old School	2009/2011	5	214.8	80.8	1.5	1.6	10.5	0.4
Ryerson	2009/2011	5	217.2	189.9	3.0	4.8	0.0	0.6
Singing Hills	2009/2011	3	296.9	19.8	92.6	0.0	10.9	0.3
Spring Bluff	2009/2011	2	91.7	0.0	58.4	8.5	0.0	0.2
Wilmot Woods	2009/2011	2	92.1	87.8	0.3	0.0	2.7	0.4
Wright Woods	2009/2011	7	277.9	237.1	0.5	0.0	0.7	0.3
Berkeley Prairie	2010/2012	2	7.5	1.1	0.0	4.5	1.0	0.7
Cahokia Flatwoods	2010/2012	2	88.9	76.0	5.1	0.0	0.7	0.5
Cuba Marsh	2010/2012	8	313.3	39.7	72.5	49.3	52.5	0.5
Gander Mountain	2010/2012	3	117.6	56.0	2.3	22.4	1.9	0.3
Lyons Woods	2010/2012	3	107.1	36.6	5.0	0.0	7.8	0.3
Middlefork Savanna	2010/2012	6	254.0	55.9	48.9	51.3	54.5	0.2
Nippersink	2010/2012	3	123.4	11.1	35.6	2.0	10.6	0.4
Rollins Savanna	2010/2012	12	499.6	47.1	162.6	159.7	20.5	0.4
Sun Lake	2010/2012	7	252.9	28.0	72.7	57.9	24.8	0.3
Tanager Kames	2010/2012	2	53.2	12.1	28.5	0.0	0.0	0.2
Wadsworth Savanna	2010/2012	12	483.4	105.8	157.9	28.7	43.3	0.2

Information). We used survey-specific covariates to explicitly model imperfect detection (MacKenzie et al. 2003) and we recorded covariates at each monitoring point during trap or coverboard checks. For landscape-scale site-specific covariates, we used the Patch Analyst extension (Centre for Northern Forest Ecosystem Research, Thunder Bay, Ontario, Canada) in ArcGIS 10 and 3 m resolution landcover classifications (developed by LCFPD ecologists) to calculate preserve characteristics of habitat and anthropogenic features. Prior to modeling, we z-transformed (proportions were arcsin square root transformed and then z-transformed) all continuous and proportion covariates.

To characterize habitat structure at the local scale, we assessed seven variables at each monitoring point using a percentage-based system similar to that used by Daubenmire (1959). We estimated percentages of cover

provided separately by trees and shrubs, tall (> 0.3 m) and short (< 0.3 m) herbaceous vegetation, litter (i.e., woody debris and leaf litter), and standing water within a 20 m buffer surrounding aquatic traps and a 100 m buffer surrounding coverboards. For aquatic trapping sites, we averaged values from both seasons within a year to obtain one value for each year then averaged years to obtain one value for each site, and for coverboards we averaged each year.

We then simplified these variables for modeling using Principal Components Analysis (PCA) on the seven ordinal habitat structure variables to avoid over-parameterization and collinearity while still permitting ecological inference of site occupancy, colonization, and extinction (McGarigal et al. 2000; Durso et al. 2011). We retained the first two orthogonal factors for aquatic trap sites and the first three orthogonal factors for terrestrial

TABLE 2. Survey effort used to detect reptiles and amphibians at 27 Lake County Forest Preserve District properties in Lake County, Illinois, USA, from April-May and July-August 2009–2012.

Group	Year	Preserves	Points with Coverboards	Points with Aquatic Traps	Coverboard Effort (total board checks)	Aquatic Trap Effort (total trap nights)
1	2009/2011	10	63	55	504	1760
2	2010/2012	17	96	68	768	2176
	Total	27	159	123	1272	3936

sites. Cumulative variability explained by the two aquatic and three terrestrial principal components totaled 66.2% and 67.1%, respectively. Principal components for aquatic habitat structure indicated increasing values of $PC1_{Aquatic}$ were indicative of woodland and forest ephemeral pools and increasing values of $PC2_{Aquatic}$ represented open canopy wetland marshes such as cattail marsh and sedge or wet meadows. Principal components for terrestrial habitat structure showed: (1) increasing values of $PC1_{Terrestrial}$ characterized woodland and forest habitat comprised of tree and shrub canopy with sparse vegetation in the understory; (2) increasing values of $PC2_{Terrestrial}$ were indicative of tall herbaceous plant communities with mesic characteristics; and (3) increasing values of $PC3_{Terrestrial}$ represented short herbaceous plant communities such as old fields.

Occupancy modeling.—We constructed occupancy models (MacKenzie et al. 2003) for each species using the single-species multi-season modeling framework in program PRESENCE 5.8 (USGS Patuxent Wildlife Research Center, Laurel, Maryland, USA). We used coverboard data to model Blue-spotted Salamanders, Eastern Tiger Salamanders, and Common Gartersnakes, and used aquatic trap data to model Painted Turtles, Snapping Turtles, Green Frogs, and Northern Leopard Frogs. Incidental observations were not included in our analyses. For each species, we used detection probabilities (p) to inform estimates of site occupancy (ψ), colonization (γ), and extinction (ε) at the local (monitoring point) and landscape scale (preserve) by ranking models using Akaike Information Criterion adjusted for small sample size (AIC$_c$; Anderson and Burnham 2002). We interpreted models that comprised the 0.90 cumulative Akaike weight (i.e., 90% confidence model set) and model-averaged covariate coefficients for inference at both local and landscape scales (Anderson and Burnham 2002). We considered model-averaged covariate coefficients significant if their 95% confidence interval did not contain zero. We did not assess model fit because no formal procedures exist to assess the multi-season occupancy model goodness-of-fit (MacKenzie et al. 2006).

Due to the relatively small number of preserves for landscape level analyses (n = 27), we considered only simple models that contained one covariate per model for the parameter of interest except for the global model (which often failed to converge due to lack of data). We removed any models that failed to calculate maximum likelihood estimates and we did not consider them in final model sets. Models containing survey or site covariates for both scales were founded on *a priori* hypotheses and/or previous findings in the literature concerning the behavior and ecology of the species in question (Supplemental Information). We used a three stage hierarchical approach to model p, ψ, and γ/ε at both scales; we first modeled p, then used the best model for p to model ψ, γ, and ε (MacKenzie et al. 2006).

When conducting wildlife surveys, especially for rare or elusive species, accounting for heterogeneity in detection probabilities is essential, and failing to do so can result in biased estimates of interested parameters (Anderson 2001, MacKenzie et al. 2002, MacKenzie et al. 2005). We estimated p by holding ψ, γ, and ε constant and considered a null model, models with covariates (Table 3), and a global model. We used the most parsimonious detection models for each species at each scale and fit varying models to estimate initial ψ and how it was influenced by site covariates. At the local scale, we determined whether the null or group model was best supported, which was then used for subsequent ψ modeling.

We then used the most parsimonious ψ and p models from both scales to model either γ or ε as functions of site covariates (Table 4), whichever had the higher probability of occurring based on the null model, unless the species was in a state of equilibrium ($\gamma \approx \varepsilon$), when neither variable was modeled. After determining which parameter to model, we used the same suite of models from each scale that were used for ψ, plus an additional model at the landscape scale to investigate the effect of preserve isolation. Except for this isolation covariate, variables believed to be influencing the site occupancy of a species also were thought to have the same influence on their colonization or extinction (Supplementary Information). We interpreted landscape scale colonization/extinction results as having occurred within a 2-y period during 2009–2012, regardless of their group association. We report only on variables that were significant (i.e., confidence intervals that did not overlap with zero).

TABLE 3. Local and landscape level explanatory variables used to model detectability for Blue-spotted Salamanders (*Ambystoma laterale*), Eastern Tiger Salamanders (*A. tigrinum*), Common Gartersnakes (*Thamnophis sirtalis*), Green Frogs (*Lithobates clamitans*), Northern Leopard Frogs (*L. pipiens*), Painted Turtles (*Chrysemys picta*), and Snapping Turtles (*Chelydra serpentina*) in Lake County, Illinois, USA.

Covariate	Description	Scale
Julian	Date in Julian days	Local
Julian^2	Date Julian days squared	Local
Minutes	Time past 0500	Local
Minutes^2	Time past 0500 squared	Local
Temp	Temperature (° C)	Local
Temp^2	Temperature (° C) squared	Local
Group	2009/2011 or 2010/2012	Local
Rain	Rain during survey	Local
Wind	Wind during survey > 2 m/s	Local
Response	Species detected during previous survey	Local
Minnow	# active minnow traps	Local
Medium	# active medium hoop traps	Local
Large	# active large hoop traps	Local
Effort	number survey points/preserve	Landscape
Survey	separate detection probability for each survey	Both
Season	separate detection probability for each survey	Both

TABLE 4. Local and landscape level explanatory variables used to model occupancy, colonization, and extinction rates for Blue-spotted Salamanders (*Ambystoma laterale*), Eastern Tiger Salamanders (*A. tigrinum*), Common Gartersnakes (*Thamnophis sirtalis*), Green Frogs (*Lithobates clamitans*), Northern Leopard Frogs (*L. pipiens*), Painted Turtles (*Chrysemys picta*), and Snapping Turtles (*Chelydra serpentina*) in Lake County, Illinois, USA.

Covariate	Description	Scale
PC1$_{Aquatic}$	indicative of closed canopy ephemeral ponds	Local
PC2$_{Aquatic}$	indicative of open canopy wetlands	Local
PC1$_{Terrestrial}$	indicative of closed canopy sparse understory	Local
PC2$_{Terrestrial}$	indicative of tall herbaceous growth, mesic	Local
PC3$_{Terrestrial}$	indicative of short herbaceous growth, old fields	Local
Urban	distance to nearest urban structure	Local
Road	distance to nearest paved road	Local
Bask	Presence (binary) of basking spots (logs, boulders, exposed banks with shallow slopes, etc.)	Local
Prairie	ha of prairie within a preserve	Landscape
Forest	ha of forest within a preserve	Landscape
Forest_ED	m/ha of forest edge within a preserve	Landscape
Shrub	ha of shrubland within a preserve	Landscape
Wetland	ha of wetland within a preserve	Landscape
Wetland_ED	m/ha of wetland edge within a preserve	Landscape
Urban_Buffer	% of 100 m buffer around preserve comprised of urban structures	Landscape
Road Buffer	m/ha of paved roads within a 100 m buffer around preserve	Landscape
Isolation	mean distance between each preserve and all other preserves	Landscape

RESULTS

We recorded 899 detections of focal herpetofauna species from 2009–2012. Painted Turtles were the most frequently detected species and Eastern Tiger Salamanders were the least frequently detected species (Table 5). Eastern Tiger Salamanders also had the lowest naïve occupancy rates at both spatial scales (naïve ψ_{local} = 0.13, naïve $\psi_{landscape}$ = 0.33). Green Frogs had the highest naïve occupancy rate (naïve ψ_{local} = 0.55) at the local scale and were tied for highest rate at the landscape scale ($\psi_{naïve}$ = 0.74) with Snapping Turtles and Common Gartersnakes (Table 5). Beta values for individual species covariate effects varied greatly by species, scale, and parameter (Supplemental Information). In addition to our focal species, we detected American Bullfrogs, Boreal Chorus Frogs (*Pseudacris maculata*), American Toads, Gray Treefrogs (*Hyla versicolor-chrysoscelis* complex), Red-spotted Newts (*Notophthalmus viridescens*), Plains Gartersnakes (*T. radix*), Smooth Greensnakes (*Opheodrys vernalis*), Brownsnakes (*Storeria dekayi*), Red-bellied Snakes (*Storeria occipitomaculata*), Graham's Crayfish Snakes (*Regina grahamii*), Eastern Foxsnakes (*Pantherophis vulpinus*), and Blanding's Turtles (*Emydoidea blandingii*). We did not analyze these species due to few detections during standardized surveys (e.g., Boreal Chorus Frog), extremely low naïve occupancy (e.g., Graham's Crayfish Snake) and/or high naïve occupancies (e.g., American Bullfrog) that were not conducive to occupancy modeling.

Blue-spotted Salamander.—At the local scale (n = 28 models; Table 6), detection was overwhelmingly influenced by season (w_i = 0.98 standard error) and increased from spring ($\hat{p}$ = 0.64 ± 0.09 standard error) to summer ($\hat{p}$ = 0.76 ± 0.09). Occupancy was 55% higher in preserves surveyed in 2010 and 2012 ($\hat{\psi}$ = 0.14 ± 0.05) than preserves surveyed in 2009 and 2011 ($\hat{\psi}$ = 0.09 ± 0.05), increased with forest canopy cover
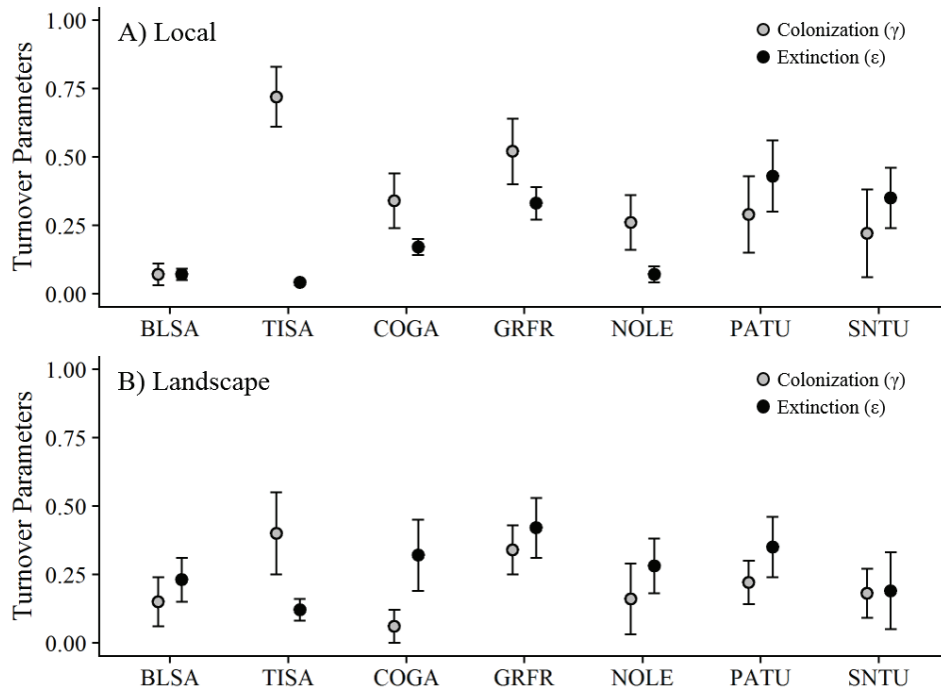
Figure 2. Mean estimates of colonization, extinction, and standard errors from null multi-season occupancy models (as unaffected by covariates) at the (A) local and (B) landscape scales for Blue-spotted Salamanders (*Ambystoma laterale*; BLSA), Eastern Tiger Salamanders (*A. tigrinum*; TISA), Common Gartersnakes (*Thamnophis sirtalis*; COGA), Green Frogs (*Lithobates clamitans*; GRFR), Northern Leopard Frogs (*L. pipiens*; NOLE), Painted Turtles (*Chrysemys picta*; PATU), and Snapping Turtles (*Chelydra serpentina*; SNTU) detected during April-May and July-August 2009–2012 on 27 Lake County Forest Preserve District properties in Lake County, Illinois, USA.

($\text{PC1}_{\text{Terrestrial}}$), and decreased with both tall herbaceous cover ($\text{PC2}_{\text{Terrestrial}}$) and short herbaceous cover and ($\text{PC3}_{\text{Terrestrial}}$). Naïve colonization ($\hat{\gamma} = 0.07 \pm 0.02$) and naïve extinction ($\hat{\varepsilon} = 0.07 \pm 0.04$) were similar at the local scale, so we did not model covariate effects (Fig. 2).

At the landscape scale (n = 21 models; Table 6), detection probability increased with the number of sites surveyed and occupancy increased with forest landcover (Fig. 3). Colonization ($\hat{\gamma} = 0.23 \pm .08$) was greater than extinction ($\hat{\varepsilon} = 0.15 \pm 0.09$) and increased with prairie, wetland, and forest landcover. In addition, colonization increased with road density and decreased with preserve isolation (Table 6).

Eastern Tiger Salamander.—At the local scale (n = 37 models; Table 6) detection probabilities were high ($\hat{p} > 0.9$) but there was no clear best model. Occupancy increased with increasing forest canopy cover ($\text{PC1}_{\text{Terrestrial}}$) and tall herbaceous cover ($\text{PC2}_{\text{Terrestrial}}$). Extinction was higher ($\hat{\varepsilon} = 0.72 \pm 0.11$) than colonization ($\hat{\gamma} = 0.04 \pm 0.01$; Fig. 2) and increased with forest canopy cover ($\text{PC1}_{\text{Terrestrial}}$) and short herbaceous cover ($\text{PC3}_{\text{Terrestrial}}$). Extinction decreased with tall herbaceous cover ($\text{PC2}_{\text{Terrestrial}}$) and proximity to roads.

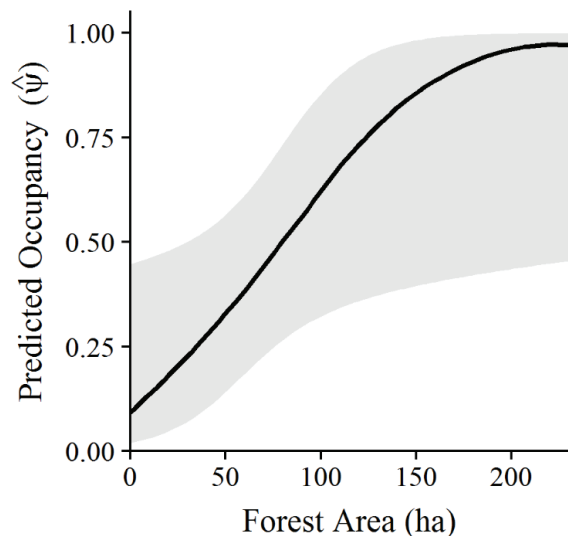At the landscape scale (n = 20 models; Table 6)

Figure 3. Blue-spotted Salamander (*Ambystoma laterale*) occupancy increased with the total amount of forest (ha) within a preserve at the landscape scale. Predicted occupancy probability and 95% confidence intervals were calculated from the top model (87% model weight) of the final Blue-spotted Salamander model set (Table 6) based on cover board checks conducted during April-May and July-August 2009–2012 at 27 forest preserves in Lake County, Illinois, USA.
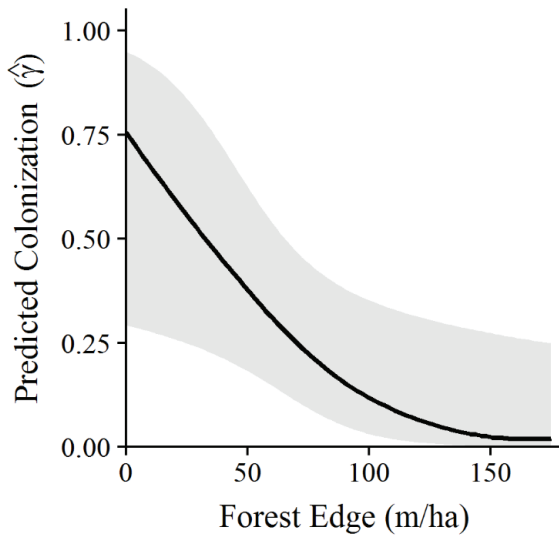
FIGURE 4. Northern Leopard Frog (*Ambystoma laterale*) colonization decreased with the amount of forest edge (m/ha) within a preserve at the landscape scale. Predicted occupancy probability and 95% confidence intervals were calculated from the top model (57% model weight) of the final Northern Leopard Frog model set (Table 6) based on aquatic trap checks conducted during April-May and July-August 2009–2012 at 27 forest preserves in Lake County, Illinois, USA.

detection was estimated to be perfect ($\hat{p} = 1.00 \pm 0.00$) and the models containing detectability covariates failed to converge; therefore, we used the null detectability model in subsequent analyses. Occupancy ($\hat{\psi} = 0.06 \pm 0.05$) increased with forest edge and wetland cover but decreased with wetland edge. Extinction ($\hat{\varepsilon} = 0.40 \pm 0.15$) was greater than colonization ($\hat{\gamma} = 0.12 \pm 0.04$; Fig. 2) and decreased with forest landcover and forest edge density (Table 6).

Green Frog.—At the local scale, (n = 34 models; Table 6), the global model for detection was best supported ($w_i = 0.53$), although a model containing half the number of parameters (an additive effect of years

surveyed and number of medium hoop traps) was nearly as competitive ($w_i = 0.45$). Factors affecting detection probability included a positive effect of Julian day and medium hoop trap effort. Occupancy varied by year surveyed ($\hat{\psi}_{2009/2011} = 0.40 \pm 0.16$, $\hat{\psi}_{2010/2012} = 0.33 \pm 0.18$), increased with distance from urban structures and open canopy wetlands ($PC2_{Aquatic}$), and decreased with proximity to roads. Extinction ($\hat{\varepsilon} = 0.52 \pm 0.06$) was greater than colonization ($\hat{\gamma} = 0.42 \pm 0.11$; Fig. 2) and increased with forest canopy cover ($PC1_{Terrestrial}$) and sites with more basking spots. Extinction probability decreased with open canopy wetlands ($PC2_{Aquatic}$) and distance from urban structures.

At the landscape scale (n = 22 models; Table 6), detection probability increased with the number of sites surveyed. Occupancy increased with forest landcover, road density, and wetland edge density, and decreased with prairie landcover and forest edge density. Colonization ($\hat{\gamma} = 0.42 \pm 0.11$) was greater than extinction ($\hat{\varepsilon} = 0.34 \pm 0.09$; Fig. 2), increased with wetland landcover, forest landcover, and wetland edge density, and decreased with forest edge density and urban landcover (Table 6).

Northern Leopard Frog.—At the local scale (n = 33 models, Table 6), detection was influenced by year surveyed, increased with Julian day, and decreased with Julian day² (a quadratic effect). Occupancy also varied by year surveyed ($\hat{\psi}_{2009/2011} = 0.32 \pm 0.13$, $\hat{\psi}_{2010/2012} = 0.03 \pm 0.03$), increased with open canopy wetland ($PC2_{Aquatic}$) and distance from urban structures, and decreased with distance from roads. Extinction ($\hat{\varepsilon} = 0.07 \pm 0.03$) was less than colonization ($\hat{\gamma} = 0.26 \pm 0.10$; Fig. 2), decreased with increasing basking opportunities and proximity to urban structures, and decreased with forest canopy cover ($PC1_{Terrestrial}$). At the landscape scale (n = 21 models; Table 6), detection increased with the number of sites surveyed and occupancy increased with wetland and prairie landcover. Colonization ($\hat{\gamma} = 0.28 \pm$

TABLE 5. Naïve occupancy rates and total number of detections for Blue-spotted Salamanders (*Ambystoma laterale*), Eastern Tiger Salamanders (*A. tigrinum*), Common Gartersnakes (*Thamnophis sirtalis*), Green Frogs (*Lithobates clamitans*), Northern Leopard Frogs (*L. pipiens*), Painted Turtles (*Chrysemys picta*), and Snapping Turtles (*Chelydra serpentina*) from April-May and July-August 2009–2012 on 27 Lake County Forest Preserve District properties in Lake County, Illinois, USA.

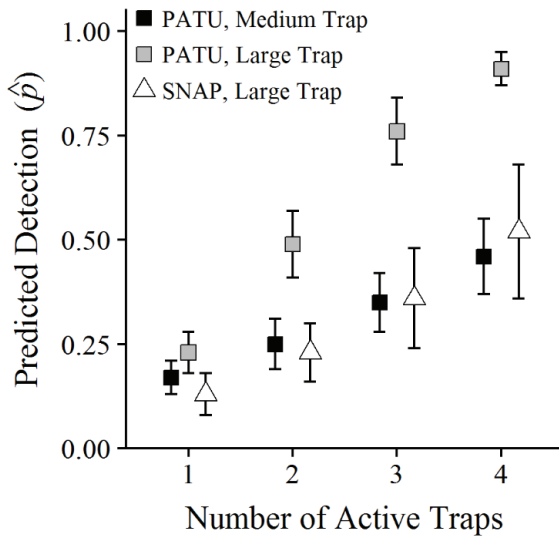| Species	Naïve Occupancy		Yearly Detections				
	Local	Landscape	2009	2010	2011	2012	Total
A. laterale	0.32	0.67	18	56	38	55	167
A. tigrinum	0.12	0.33	11	13	13	12	49
L. clamitans	0.56	0.74	46	50	38	49	183
L. pipiens	0.21	0.52	29	3	21	10	63
C. serpentina	0.38	0.74	16	30	24	28	98
C. picta	0.45	0.70	27	69	71	56	223
T. sirtalis	0.29	0.74	31	21	42	22	116
Total			178	242	247	232	899

FIGURE 5. Detection probability at the local scale increases with the number of active traps for both Painted Turtles (*Chrysemys picta*; PATU) and Snapping Turtles (*Chelydra serpentina*; SNTU). Model-averaged estimates of detection probabilities (± SE) as functions of the number of active medium and large hoop traps per survey were estimated from 55 monitoring points during April-May 2009 and 2011, and 68 points during July-August 2010 and 2012 at forest preserves in Lake County, Illinois, USA.
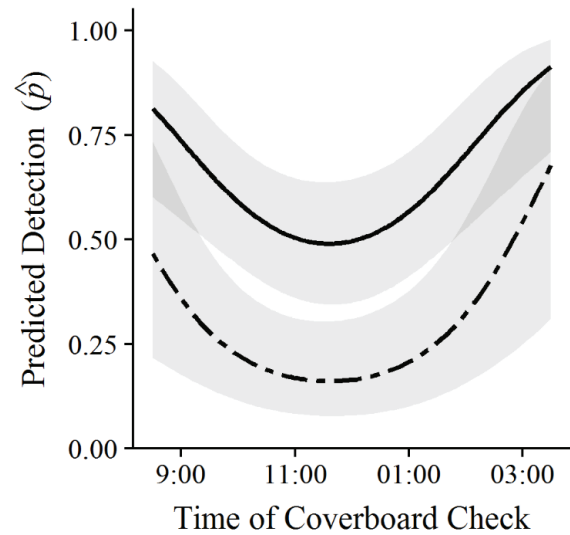
FIGURE 6. Detection probability at the local scale dips during mid-day for Common Gartersnakes (*Thamnophis sirtalis*). Model-averaged estimates of detection probabilities and 95% confidence intervals for Common Gartersnakes (*Thamnophis sirtalis*) found under coverboards during 2009 and 2011 (solid line) and 2010 and 2012 (dashed line) were estimated from 63 monitoring points during April-May 2009 and 2011, and 96 points during July-August 2010 and 2012 at forest preserves in Lake County, Illinois, USA.

0.10) was greater than extinction ($\hat{\varepsilon} = 0.16 \pm 0.13$; Fig. 2), increased with wetland landcover, and decreased with forest edge density (Fig. 4), urban landcover, and preserve isolation (Table 6).

Painted Turtle.—At the local scale (n = 34 models; Table 6), the global detection model was best supported. Detection increased with temperature and the number of medium and large hoop traps (Fig. 4). Occupancy decreased with forest canopy cover ($PC1_{Terrestrial}$) and increased with open canopy wetlands ($PC2_{Aquatic}$). Colonization ($\hat{\gamma} = 0.43 \pm 0.13$) was greater than extinction ($\hat{\varepsilon} = 0.29 \pm 0.14$; Fig. 2) and increased with open canopy wetland, distance to roads, and distance to urban structures.

At the landscape scale (n = 22 models; Table 6), detection increased with the number of sites surveyed. Occupancy was high and strongly influenced by increasing wetland landcover and wetland edge density. Colonization ($\hat{\gamma} = 0.35 \pm 0.11$) was greater than extinction ($\hat{\varepsilon} = 0.29 \pm 0.14$; Fig. 2), increased with prairie landcover, and decreased with forest edge density and road density (Table 6).

Snapping Turtle.—At the local scale (n = 29 models, Table 6), the global detection model was supported best, and detection increased with the number of larger hoop nets (Fig. 5). Occupancy decreased with forest canopy cover ($PC1_{Terrestrial}$) and increased with open canopy

wetlands ($PC2_{Aquatic}$). Colonization ($\hat{\gamma} = 0.35 \pm 0.11$) was greater than extinction ($\hat{\varepsilon} = 0.22 \pm 0.16$; Fig. 2), increased with distance from urban structure and roads, and decreased with forest canopy cover ($PC1_{Terrestrial}$) and open canopy wetland ($PC2_{Aquatic}$).

At the landscape scale (n = 14 models, Table 6), detection decreased from spring ($\hat{p} = 0.27 \pm 0.07$) to summer ($\hat{p} = 0.10 \pm 0.05$). Occupancy was strongly influenced by wetland landcover and decreased with forest edge density, prairie landcover, urban landcover, and road density. Colonization ($\hat{\gamma} = 0.19 \pm 0.14$) was similar to extinction ($\hat{\varepsilon} = 0.18 \pm 0.09$; Fig. 2) so we did not model covariate effects (Table 6).

Common Gartersnake.—At the local scale (n = 29 models, Table 6), the model including an additive effect of year surveyed and time of day² received most of the support (model weight = 0.94; Fig. 6). Occupancy decreased with forest canopy cover ($PC1_{Terrestrial}$) and increased with tall herbaceous ($PC2_{Terrestrial}$) and short herbaceous cover ($PC3_{Terrestrial}$). Extinction ($\hat{\varepsilon} = 0.34 \pm 0.10$) was greater than colonization ($\hat{\gamma} = 0.17 \pm 0.3$; Fig. 2), increased with forest landcover, and decreased with both tall and short herbaceous cover.

At the landscape scale (n = 14 models; Table 6), detection increased with the number of sites surveyed. Occupancy increased with wetland landcover and wetland edge density but decreased with forest landcover. Colonization ($\hat{\gamma} = 0.32 \pm 0.13$) was greater

TABLE 6. Top overall model for each species at the local and landscape scale, model weights (ω) for each parameter (detection, p; occupancy, Ψ; colonization or extinction, γ or ε), and number of models run for each parameter for Blue-spotted Salamanders (*Ambystoma laterale*), Eastern Tiger Salamanders (*A. tigrinum*), Common Gartersnakes (*Thamnophis sirtalis*), Green Frogs (*Lithobates clamitans*), Northern Leopard Frogs (*L. pipiens*), Painted Turtles (*Chrysemys picta*), and Snapping Turtles (*Chelydra serpentina*) detected from April-May and July-August 2009–2012 on 27 Lake County Forest Preserve District properties in Lake County, Illinois, USA. Covariate descriptions are in Tables 2 and 3 and full model sets with ranking criteria are presented in Supplemental Information. We did not model covariate effects for species where $\gamma \approx \varepsilon$.

| | | | | ω, # of models run | | |
Species	Scale	Top Model	Ψ	γ	ε	p
A. laterale	Local	$\Psi(\text{GROUP}+\text{PC1}_{\text{Terrestrial}}+\text{PC2}_{\text{Terrestrial}})$, $\gamma(.)$, $\varepsilon(.)$, $p(\text{SEASON})$	0.53, 13	n/a	n/a	0.98, 14
A. tigrinum	Local	$\Psi(\text{PC1}_{\text{Terrestrial}}+\text{PC2}_{\text{Terrestrial}})$, $\gamma(.)$, $\varepsilon(\text{PC3}_{\text{Terrestrial}})$, $p(\text{JULIAN})$	0.40, 12	n/a	0.45, 11	0.21, 14
L. clamitans	Local	$\Psi(\text{ROAD})$, $\gamma(.)$, $\varepsilon(\text{GROUP}+\text{PC1}_{\text{Aquatic}})$, $p(\text{GLOBAL})$	0.39, 11	n/a	0.20, 11	0.53, 12
L. pipiens	Local	$\Psi(\text{GROUP})$, $\gamma(.)$, $\varepsilon(\text{PC2}_{\text{Aquatic}})$, $p(\text{GROUP}+\text{JULIAN}+\text{JULIAN}^2)$	0.23, 11	n/a	0.42, 10	0.85, 12
C. picta	Local	$\Psi(\text{PC1}_{\text{Aquatic}}+\text{PC2}_{\text{Aquatic}})$, $\gamma(\text{GROUP}+\text{BASK})$, $\varepsilon(.)$, $p(\text{GLOBAL})$	0.29, 11	0.33, 11	n/a	0.99, 12
C. serpentina	Local	$\Psi(\text{PC2}_{\text{Aquatic}})$, $\gamma(\text{ROAD})$, $\varepsilon(.)$, $p(\text{GLOBAL})$	0.67, 6	0.41, 11	n/a	0.98, 12
T sirtalis	Local	$\Psi(\text{PC1}_{\text{Terrestrial}}+\text{PC2}_{\text{Terrestrial}})$, $\gamma(.)$, $\varepsilon(\text{PC1}_{\text{Terrestrial}})$, $p(\text{GROUP}+\text{MIN}+\text{MIN}^2)$	0.51, 13	n/a	0.41, 12	0.94, 14
A. laterale	Landscape	$\Psi(\text{FOREST})$, $\gamma(\text{ISOLATION})$, $\varepsilon(.)$, $p(\text{EFFORT})$	0.86, 8	0.53, 9	n/a	0.87, 4
A. tigrinum	Landscape	$\Psi(.)$, $\gamma(.)$, $\varepsilon(\text{URBAN})$, $p(\text{EFFORT})$	0.35, 7	n/a	0.46, 9	0.84, 4
L. clamitans	Landscape	$\Psi(\text{FOREST})$, $\gamma(\text{FOREST\_ED})$, $\varepsilon(.)$, $p(\text{EFFORT})$	0.26, 9	0.47, 9	n/a	0.86, 4
L. pipiens	Landscape	$\Psi(\text{PRAIRIE})$, $\gamma(\text{FOREST\_ED})$, $\varepsilon(.)$, $p(\text{EFFORT})$	0.46, 8	0.58, 9	n/a	0.75, 4
C. picta	Landscape	$\Psi(\text{WETLAND})$, $\gamma(\text{FOREST\_ED})$, $\varepsilon(.)$, $p(\text{EFFORT})$	0.72, 11	0.43, 7	n/a	0.80, 4
C. serpentina	Landscape	$\Psi(\text{WETLAND})$, $\gamma(.)$, $\varepsilon(.)$, $p(\text{SEASON})$	0.53, 9	n/a	n/a	0.58, 4
T. sirtalis	Landscape	$\Psi(.)$, $\gamma(\text{PRAIRIE})$, $\varepsilon(.)$, $p(\text{EFFORT})$	0.23, 9	0.45, 9	n/a	0.99, 4

than extinction ($\hat{\varepsilon} = 0.06 \pm 0.06$; Fig. 2), increased with prairie landcover and wetland edge density, and decreased with forest landcover and forest edge density.

DISCUSSION

Our results show that natural and anthropogenic features influenced the site occupancy, colonization, and extinction rates of herpetofauna at two spatial scales within a suburban forest preserve network of the Chicago region. At both scales of analysis, differences in occupancy, colonization, and extinction were generally due to features of habitat rather than anthropogenic influences. At the local scale, anthropogenic features were only included in the top occupancy model (lowest AIC$_c$) for Green Frogs; at the landscape scale, no anthropogenic features were included in the top occupancy model. When investigating reptile and amphibian ecology, consideration of multiple spatial scales is essential when inferring the spatial distribution of species and turnover rates as profoundly different conclusions can be drawn depending on the scale of analysis (Lowe and Bolger 2002; Bowne et al. 2006; Rizkalla and Swihart 2006; Jenkins et al. 2009). For example, extinctions were generally more common at

the local scale, whereas colonizations were common at the landscape scale.

In addition, we found that different species had different habitat requirements. For example, at the local scale, closed canopy woodlands with sparse understories (PC1$_{\text{Terrestrial}}$) were associated with increased Blue-spotted Salamander occupancy but decreased Common Gartersnake occupancy, reflecting both species conflicting natural history needs (avoiding desiccation versus thermoregulation). At the landscape scale, Blue-spotted Salamander occupancy was most heavily influenced by forest landcover, whereas Northern Leopard Frog occupancy was most heavily influenced by prairie landcover and Snapping Turtle occupancy was most heavily influenced by wetland landcover. These results are unsurprising but provide important empirical evidence that maximizing habitat diversity is important when managing a landscape for herpetofaunal diversity.

As expected, temporal and environmental factors and circumstances of survey design produced heterogeneity in detection probabilities. Overall, we obtained adequate precision in parameter estimates at the local scale, but confidence in landscape scale results was likely hindered by low sample size (n = 27 preserves

studied). At the landscape scale, the number of sites surveyed was important and positively associated with detection probability for all species and was the top ranked model for all species but Snapping Turtles. Coverboards and hoop nets were generally effective at targeting our species of interest. Coverboards offer a low-cost low-maintenance means of sampling terrestrial herpetofauna assemblages; however, coverboards are not a catch-all technique and other methods such as drift fences should be considered (Kjoss and Litvaitis 2001b; Ryan et al. 2002). When surveying for snakes, midday coverboard checks have the lowest likelihood of detecting target species. Thus, to maximize detection we suggest coverboards should be checked during the morning or evening hours. Similarly, we suggest maximizing the size and number of traps to increase the detection probabilities for aquatic turtles.

Blue-spotted Salamander.—As predicted, we found a strong seasonal effect in the detection probability of Blue-spotted Salamanders, which were 13% more likely to be detected in July-August than April-May at the local scale. This may be due to the influx of juveniles leaving breeding pools and thus being available for detection, or that adults in April are still at the breeding pools and thus less likely to be detected under terrestrial cover boards (Regosin et al. 2005). At the landscape scale, the amount of forest was important to Blue-spotted Salamander occupancy, as expected (Mierzwa 1998). Subtle differences in sub-canopy tree species composition and leaf litter compositions can encourage or discourage Blue-spotted Salamander presence (Silva et al. 2003; Belasen et al. 2013), but they are closely tied to ephemeral vernal pools of forest and woodland habitat (Lathrop et al. 2005). In addition, occupancy was higher in larger preserves, likely due to more available habitat (larger tracts of forest) and less fragmentation.

Blue-spotted Salamanders had equally low probabilities of colonization and extinction at the local scale but colonizations occurred more frequently than extinctions at the landscape scale. Purrenhage et al. (2009) found that Spotted Salamanders (*A. maculatum*) in a fragmented landscape had negligible genetic differences between disjunct populations and suggested that salamanders were moving freely between perceptually isolated populations. In Lake County, Blue-spotted Salamanders may be using floodplains or drainage ditches as corridors for dispersal between ostensibly isolated preserves. As this is a Species of Greatest Conservation Need in Illinois (State of Illinois. 2015. Wildlife Action Plan 2015 Implementation Guide. State of Illinois. Available from https://www.dnr.illinois.gov [Accessed 12 December 2017]) and a regional priority species (Chicago Wilderness. 2018. Priority Species. Chicago Wilderness. Available from https://www.chicagowilderness.org [Accessed 2 April 2019]), understanding the spatial ecology of Blue-spotted Salamanders will be an important component of their conservation, and we recommend research into their movement patterns and dispersal ability.

Eastern Tiger Salamander.—Detection probabilities for Eastern Tiger Salamanders were high ($p > 0.9$) at both scales, but we detected them less often than all other species. At the local scale, we were 5.5 times more likely to detect an Eastern Tiger Salamander on a subsequent survey if they had been detected during the first visit, suggesting Eastern Tiger Salamanders were using cover boards repetitively. While Eastern Tiger Salamanders have been reported to establish residency at known refugia (Madison and Farrand 1998), this has not been documented previously for artificial cover objects. Additionally, on six occasions during our study where we surveyed coverboards and aquatic traps at the same sites during the same season and year, we detected Eastern Tiger Salamanders in aquatic traps and not under coverboards (i.e., false absences).

Eastern Tiger Salamanders are widespread in the Great Lakes region (Anton 1999) and use a wide variety of habitats, including degraded habitats such as farm ponds (Knutson et al. 2004, Brodman 2010). Due to the high detection probabilities, known false absences (unpubl. data), and violation of the assumption of independence between surveys, our models likely underestimated site occupancy rates at both scales (MacKenzie et al. 2003, MacKenzie et al. 2006). As a result, these results should be considered preliminary, and we recommend increased monitoring to determine the status, distribution, and population parameters of Eastern Tiger Salamanders in the region.

Green Frog.—The global model was the best-supported detection model, and many factors influenced detectability of Green Frogs. After model-averaging, the squared term of temperature, Julian day, and number of medium traps all had statistically significant impacts on detection probability, but effect sizes (s) were weak. At the local scale, models including anthropogenic factors such as distance to roads (negative impact) and urban areas (positive impact) were ranked higher than habitat models. Roads negatively impact a wide variety of reptile and amphibian species (reviewed by Andrews et al. 2008). Green Frogs are known to be susceptible to high levels of road deicing agents (Karraker 2008) and Green Frog abundance decreases with increasing traffic volume (Gravel et al. 2012); however, it is unclear why Green Frog occupancy increased with proximity to urban structures. Future research should investigate this link further, as urban structures were not highly correlated with any relevant habitat variables.

At the landscape scale, forest landcover was the top-ranked model, similar to the findings of Mazerolle et al. (2005), who found that Green Frog occupancy increased with forest cover at the landscape scale (1,000 m buffer), but decreased with forest landcover at smaller spatial scales (250 m buffer). This is consistent with our results, as Green Frogs were more likely to occupy open-canopy wetland sites at the local scale. These same sites had lower probabilities of extinction suggesting high-quality wetlands were beneficial to the localized persistence of this species. Green Frogs, however, can undergo metamorphosis either during the same year they hatched, or over-winter as larva, then undergo metamorphosis the following spring (Martof 1956), making inference of spatial turnover difficult because our inferences of colonization and extinction were based on time periods of 2 y. During these time periods, sites could have experienced both localized extinctions and colonizations between sampling periods.

Northern Leopard Frog.—Unlike Green Frogs, detection probabilities for Northern Leopard Frogs depended on time of year and increased sharply during spring sampling (April-May) and decreased throughout summer sampling (July-August) as indicated by the quadratic effect of Julian day. We likely observed increased detection probabilities throughout the spring as adults moved to breeding pools, then saw a decline as adults departed, leaving only tadpoles which are likely too small to be detected by medium and large hoop traps in late spring (Ruth L. Hine et al., unpubl. report), and therefore only able to be detected by minnow traps. The use of dipnets or call surveys would likely increase detection of Northern Leopard Frogs (and other anurans), but we were unable to include these techniques due to logistic constraints.

At the local scale, Northern Leopard Frogs were more likely to occupy open canopy wetlands, consistent with recent studies of habitat selection in the Midwest (Knutson et al. 2018), although open canopy sites also had increased probabilities of extinction. However, at the landscape scale, colonization was more likely than extinction, suggesting that Northern Leopard Frogs were acting as a metapopulation, with local extinctions countered by landscape level colonizations (Carlson and Edenhamn 2000). In addition, preserves that were increasingly isolated from other preserves had decreased probabilities of colonization, but the effect size was weak. As Northern Leopard frogs in the Midwest have small home ranges, roads, agriculture, and other human-dominated features likely decrease the ability of Northern Leopard Frogs to disperse successfully (Knutson et al. 2018). Further, natural features, such as forest edge, may also prevent successful dispersal, suggesting that the persistence of Northern Leopard Frogs in urban

preserves depends on the careful consideration of many factors at large spatial scales.

Painted Turtle.—Increased likelihoods of Painted Turtle detection during warmer temperatures were likely a direct result of increased activity periods during warmer temperatures as they seek food and basking opportunities (Grayson and Dorcas 2004). Painted Turtles were more likely to occupy open canopy marsh wetlands and exhibited an avoidance of sites with closed canopies from trees and shrubs. Aquatic habitats were diverse in the study area; however, the most common wetland type was cattail (*Typha* spp.) marsh, a habitat type well documented as quality habitat for both Painted and Snapping Turtles (Bodie et al. 2000; Brown et al. 1994). Site occupancy was twice as high for sites with adequate basking spots relative to sites without basking spots. Lefevre and Brooks (1995) found that Painted Turtles bask for approximately 1–2 h per day, and if local turtle densities are high, basking opportunities become increasingly limited which can result in aggression between individuals (Lovich 1988). Wetland size is an important factor for Painted Turtle site occupancy (Cosentino et al. 2010) and is corroborated by our results.

Preserves with greater surrounding road density had decreased probabilities of colonization by Painted Turtles, thus, they may have experienced high mortality during the nesting season. Road mortality of turtles was not uncommon at our study site, and in general is greatest for aquatic turtles moving between wetland and upland sites during mating and nesting (Gibbs and Shriver 2002; Cureton and Deaton 2012). Further, sex ratios of Painted Turtles were highly male biased in Lake County preserves, indicative of road morality or other negative effects of urbanization, such as subsidized predators (Vanek and Glowacki 2019). Preserves with increased prairie landcover and preserves with decreased forest edge density proved advantageous to landscape scale colonization probabilities. If Painted Turtles experience high nest depredation, especially along forest edges by Raccoons (*Procyon lotor*; Marchand and Litvaitis 2004a), then it makes biological sense that preserves with decreased amounts of forest edge had increased colonization probability. Alternatively, Painted Turtles may have experienced nest success in prairie habitats, thus explaining colonization events as new turtles colonize previously unoccupied adjacent wetlands.

Snapping Turtle.—Only the number of large hoop nets increased detection for Snapping Turtles, likely because medium hoop nets excluded larger adults. The average curved carapace length during this study was 26.4 cm (standard deviation = 8.7 cm), thus most Snapping Turtles were unavailable for sampling via

medium hoop traps. Snapping Turtles had similar trends of occupancy and colonization to Painted Turtles and were more likely to occupy open canopy marsh wetlands and exhibited an avoidance of sites with high levels of canopy cover. At the local scale, Snapping Turtles were impacted negatively by roads, perhaps due to mortality events associated with road crossing behavior (Haxton 2000; Gibbs and Shriver 2002; Steen and Gibbs 2004). Declines in turtle populations can often go unnoticed, and the long lifespan of adult turtles can mask population declines based on demographic collapse (Lovich et al. 2018). Because Snapping Turtles have long cohort generation times of nearly 25 y and require high adult survival for population persistence (Congdon et al. 1994), we recommend further study into the demographics of Snapping Turtles in Lake County to ensure adult survival is high and enough juveniles are being recruited into the population.

Common Gartersnake.—Detection probability of Common Gartersnakes using coverboards was lowest during the middle of the day, a similar observation made without the consideration of detection probability by others (Kjoss and Litvaitis 2001b, Joppa et al. 2010). Although our top ranked model contained a group effect, the overall trend was similar between years. At the local scale, Common Gartersnakes avoided sites with high canopy cover while being positively impacted by tall herbaceous mesic and short herbaceous habitats. It is possible that sites with high amounts of canopy cover do not meet the thermoregulatory requirements of Common Gartersnakes relative to open canopy sites, as seen in other snake species (e.g., Eastern Massasaugas, *Sistrurus catenatus*; Shoemaker and Gibbs 2010). As with most of the other species studied, increased preserve isolation decreased colonization rates, supporting the notion that isolated preserves become increasingly harder to colonize due to increased road mortality of dispersing snakes (Andrews et al. 2008). The probability of landscape scale colonization increased for preserves with increasing amounts of prairie and wetland edge density, two habitat features important to snake survival (Cagle 2008).

Conservation implications.—Urbanization is often detrimental to many reptile and amphibian populations (Hamer and McDonnel 2008), and so the conservation and management of already existing habitat may be needed along with the restoration of new properties (Palmer et al. 1997). For example, we found that Blue-spotted Salamander occupancy increased with the total amount of forest in a preserve, and occupancy was also higher in larger preserves. As salamanders are susceptible to habitat fragmentation, increasing local and landscape-scale habitat connectivity can improve site occupancy rates and mitigate genetic isolation

(Eastman et al. 2007). Further, as temperate forests take many years to grow, effective conservation of Blue-spotted Salamanders in the Chicago region may require the protection of existing patches of forest.

Our study indicates the importance of conserving and managing for multiple habitat types. Although we report on relatively common and generally widespread species within Lake County, other species with more specific habitat requirements may not be able to persist without management of more specific habitat types (e.g., Spotted Salamanders; Sacerdote and King 2009). In addition, though our focal species were generally more influenced by natural features than anthropogenic features, as to be expected in preserves managed for biodiversity and natural resources, other species may be more susceptible to the negative impacts of urbanization, such as through increased depredation by subsidized predators (e.g., Blanding's Turtles; Urbanek et al. 2017) or road mortality (e.g., Eastern Massasaugas; Shepard et al. 2008b).

The Lake County Forest Preserve District maintains a nearly contiguous corridor of preserves along the Des Plaines River in the eastern half of the county while the Fox River connects four preserves as it meanders along the western edge of the county. This connectedness could facilitate terrestrial and aquatic movements, especially during annual flood events. In large river floodplain systems, Galat et al. (1998) and Bodie et al. (2000) explained that remnant wetlands associated with rivers that have extended hydroperiods from flooding are important to the conservation of abundant and diverse reptile and amphibian assemblages. Therefore, we recommend maintaining these important corridors, and suggest that land acquisition focus on building connectivity between isolated preserves, particularly for those not already connected by rivers and floodplains.

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Supplemental Information: http://www.herpconbio.org/Volume_14/Issue_2/Cassel_etal_2019_Suppl.pdf

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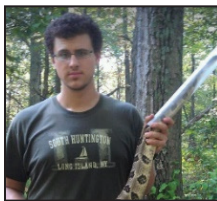
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