Vegetation Preferences of Blue-spotted Salamanders (*Ambystoma laterale*) Using Readily Accessible Monitoring Protocols

The Chicago Wilderness region contains ca. 150,000 ha of protected natural land that includes some of the world's best surviving remnants of prairies, oak savannas, oak woodlands, and other rare communities (Wang and Moskovits 2001; Sullivan 2003). Taken together, this land harbors more than 500 federally- or state-listed endangered or threatened species (Illinois Natural Heritage Database 2020; Indiana Natural Heritage Data Center 2020), as well as many species of particular conservation concern such as the Blue-spotted Salamander (*Ambystoma laterale*). Due to its restricted and vulnerable habitat, *A. laterale* has been designated a "priority species" in the Chicago Wilderness region and is considered a "species of greatest conservation need" in Illinois by the Department of Natural Resources and by Midwest Partners in Amphibian and Reptile Conservation, with a "medium-high" concern score for the entire Midwest region (https://www.chicagowilderness.org/page/PrioritySpecies; www.mwparc.org/species/). Conservation of pond-breeding salamanders such as *A. laterale* has broader implications for the ecosystem, as these animals may be an important means of nutrient transfer from aquatic to terrestrial food webs (Gibbons et al. 2006; Regester et al. 2006), and such energy transfer by animals is thought to play a critical role in the movement of biomass between trophic levels (Vanni 2002). Furthermore, *A. laterale* is the only salamander species known to occur at the study site (see below). Because functional redundancy is thought to be an important determinant of ecosystem stability, it has been argued that species representing the sole member of a functional group should receive priority conservation attention (Walker 1992; Palmer et al. 1997; Tilman et al. 1997).

Chicago Wilderness is home to thousands of land managers and volunteer stewards dedicated to improving the health of our natural areas. They do so primarily through techniques such as invasive species removal, prescribed fire, sowing seeds, and planting seedlings (CRBC 1999; Fahey et al. 2015). Culling White-tailed Deer is one of the few direct animal manipulations commonly practiced (CRBC 1999). Since most land management techniques focus on plants rather than animals, it is worth asking how management of the vegetation community is indirectly impacting wildlife species of concern.

However, the intensive time, expertise, and resources required for many amphibian monitoring methods make their use impractical for most practitioners (Heyer et al. 1994). For anurans, nighttime calling surveys have proven to be an easily-learned, reliable, and sustainable means of monitoring long-term population changes at large spatial scales (Droege and Eagle 2005), and they allow site stewards to track the number of anuran species breeding on site. Urodeles present a greater

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monitoring challenge due to their fossorial habits and non-vocal breeding behavior. The most accurate monitoring methods for these species (for example, dye tracking, drift fences with pitfall traps, and radio tracking; e.g., Regosin et al. 2005; Graeter et al. 2008; Hoffman et al. 2018) require extensive expertise and high resource investment. Simpler methods such as visual encounter surveys and cover board transect monitoring can be suitable substitutes for more intensive methods, provided their rigor and efficacy have been tested (Heyer et al. 1994). Site stewards aiming to restore and manage habitat for fossorial salamanders need monitoring methods that are practical in the short term and sustainable over time and that can be conducted with on-site training, small budgets, and limited time.

The first goal of the current study was to explore the relationship between *A. laterale* numbers and the composition of the vegetation community at a local oak woodland preserve that is undergoing ecological management. We know that adult

Fig. 1. "Flitch wood" pile (a) and cover board monitoring (b), showing partial transect layout in the background of photo 1b.
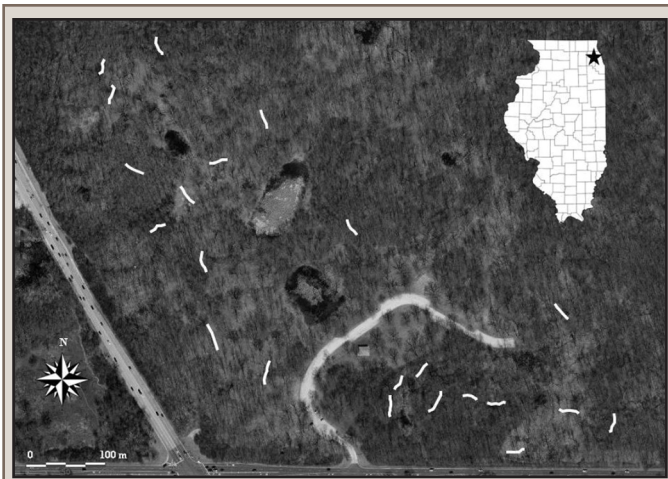
Fig. 2. Locations of transects. Inset: Map of Illinois with star showing approximate location of study site. (Map created with arcgis.com. Inset from Wikimedia Commons, labeled for noncommercial reuse with modification.)

amphibians preferentially select terrestrial habitat based on structural elements such as leaf litter depth, coarse woody debris, canopy cover, and vegetation cover (Graeter et al. 2008; Osbourn et al. 2014; Hoffman et al. 2018). However, very few studies have examined the extent to which vegetation composition, as opposed to simply vegetation cover, impacts amphibian abundance (but see Nuzzo and Mierzwa 2000; Belasen et al. 2013). Since increasing plant diversity and removing invasive species are stated goals of most land management plans in the local oak ecosystems where *A. laterale* occurs (Fahey et al. 2015), I wanted to learn the extent to which these factors influenced salamander counts. I was particularly interested in two invasive plant species that are abundant at the study site and that receive a great deal of management attention by site stewards, Common Buckthorn (*Rhamnus cathartica*) and Tall Goldenrod (*Solidago altissima*). The negative ecosystem impacts of the shrub *R. cathartica* have been widely documented (Heneghan et al. 2004; Warren et al. 2017), and its abundance within the woodlands of Chicago Wilderness has been estimated to exceed 26 million stems, or more than 500 sapling stems/acre (Glennemeier 2004). *Solidago altissima* is a native forb that forms monocultures often needing active control in order to support native plant diversity (Chicago Wilderness 2016). It exhibits dense rhizomatous growth and allelopathic activity and can remain dominant in natural areas for decades (Werner 1980; Abhilasha et al 2008). I quantified the vegetation community within salamander monitoring transects in order to identify any correlations between salamander counts and key measures of vegetative composition such as species richness and invasive species prevalence.

The second study goal was to determine the practicality and efficacy of my chosen monitoring method for use by site stewards and land managers who have limited time, resources, or herpetological backgrounds. The method was designed to not unduly disturb the habitat, to be readily learned and conducted by site stewards, and to be affordable, effective, and rigorous. While collecting data for the vegetation study (goal number one), I examined three components of the monitoring method toward goal number two: First, I tested a novel cover board material that closely mimicked natural fallen logs, due to concerns over possible (albeit untested) chemical leaching from plastics or preservative-treated plywood, as well as the desire to

maintain site conditions as close as possible to the presumed natural state. Using cover boards also ameliorated concerns over habitat disturbance from repeated sampling of natural cover objects (Goode et al. 2004; MWPARC 2012). Second, I determined the ability of photo mark-recapture methods to provide useful information with a reasonable investment of time and technology. Visual pattern identification has long been used to identify and track individuals, especially for large animals (Urian et al. 2014). Advances in digital cameras and pattern recognition software have made this technique more useful for smaller animals and more accessible for monitors (e.g., Sannolo et al. 2016; Speybroeck and Steenhoudt 2017). I wanted to test its practical efficacy for use by monitors with extensive ecological expertise but little specific training in wildlife population studies. Third, I worked alongside many site stewards and volunteers to collect the data, and I recorded the time required for training, data collection, and photo ID analysis, in order to determine the suitability of this monitoring method for other practitioners.

MATERIALS AND METHODS

The study site was an 87-ha parcel within the Forest Preserves of Cook County (FPCC) that historically was likely a mix of open oak woodland and savanna (Packard and Mutel 2005). It sits within the Village of Northbrook, Illinois and is a relatively isolated, suburban preserve surrounded by roads and developed land. For most of the last century, it was heavily degraded by invasive species, primarily *Rhamnus cathartica*, and was characterized by bare ground, dense shade, and low biodiversity. Since 1989, FPCC land managers and volunteer stewards have been removing invasive species, spreading the seeds of a diverse mix of native plants, returning prescribed fire to the landscape (McClain et al. 2010), and restoring the canopy cover to levels that support oak reproduction and that support herbaceous communities characteristic of oak woodlands. Oak woodlands within the Chicago Wilderness region are thought to have developed under a canopy cover of 50–80%, whereas most unmanaged woodlands currently have canopy cover well above this range, due to fire suppression and invasive brush encroachment (CRBC 1999). The site's current condition ranges from heavily degraded in some areas to very high quality (i.e., rich in plant and animal diversity) in others.

In November 2017, I established 22 transects of 30-m length, each containing 10 cover boards spaced ca. 3 m apart. Board sizes were 20–30 cm wide by 80–95 cm long, with the long edge oriented perpendicular to the direction of the transect. For the cover board material, I obtained undried ash (*Fraxinus* sp.) lumber harvested as end cuts, or "flitch wood," from local trees, and placed the outer surface (curved, with some bark attached) against the ground (Horigan Urban Forest Products, Inc., North Chicago, Illinois, USA; Fig. 1). I chose transect locations to represent a wide range of vegetation diversity and invasive species prevalence, based on qualitative meander surveys. Boards were placed 15–124 m away from nearby breeding ponds (Fig. 2), representing distances that salamanders are known to occupy (Semlitsch 1998; Regosin et al. 2005; Ryan and Calhoun 2014). Mean distance from one transect to the next nearest transect was 61 m (median = 58 m; range = 10–185 m).

Salamander surveys.—We (interns, volunteers, and I) checked boards approximately weekly from 2 April through 27 August 2018 by lifting the board and looking underneath for salamanders (median sampling interval = 8 d; range = 3–21 d).

Animals were weighed, measured for snout–vent length (SVL), photographed, and immediately released next to the cover board where they were found. We sterilized all equipment and boots with a 3% bleach solution before and after each site visit. While in the field, we sterilized equipment with isopropyl alcohol after each transect where an animal was encountered (NEPARC 2014) and wore gloves when handling animals.

Vegetation surveys.—During the first week in August 2018, I collected vegetation data within a ¼ m² quadrat placed next to each cover board, for a total of 10 quadrats per transect. This standard quadrat size is small enough to allow for rapid and complete species lists and reasonably precise vegetation cover estimates while being large enough to accurately represent the sample area (Dengler 2008; Spyreas 2016). Within each quadrat, I recorded the species and percent cover of all plants (including all herbaceous species and all woody species under one meter tall), as well as leaf litter depth. At each quadrat, I visually estimated canopy cover as a proxy for ground-level shade, using photographs taken with a 180° fish-eye lens attached to a smart phone and held at 1-m height. I recorded visual estimates of coarse woody debris (fallen logs or bark) within the 3-m sections between boards (and 3 m beyond board 10), for a total of ten estimates per transect.

Photo mark-recapture.—We (interns, volunteers, and I) used a non-invasive photographic mark-recapture technique to identify individual animals. Each animal was placed facing left on a small white board and then photographed with a Panasonic DMC-FZ200 Lumix camera using a macro lens. We over-exposed the photographs to admit as much light as possible, to help distinguish the light blue spots from the darker surrounding skin. Photographs were cropped to include only the area from the tip of the snout to the insertion point of the front left forelimb. We used the open-source, pattern recognition software Wild-ID (Bolger et al. 2012) to narrow down the list of potential matches for each individual and then used manual inspection (which identified matches clearly and unambiguously) to make the final determination. We also manually inspected all images, including all possible combinations of photos, to be certain we were positively identifying all re-captures and to test the accuracy of the Wild-ID program.

Statistical analysis.—Due to a large influx of first-year metamorphs in August, I focused the analysis on pre-August captures in case there were behavioral differences among cohorts. To avoid multiple counts of the same individual, re-captured animals were not included in the analysis. Values for vegetation metrics, leaf litter depth, canopy cover, and coarse woody debris were each averaged to provide a single value of each variable per transect, to be used in statistical analyses. I addressed the hypothesis that adult salamanders were non-randomly selecting habitat based on vegetation diversity or invasive plant prevalence, using linear regression of vegetation variables against the number of salamander captures per transect as the dependent variable. I did not hypothesize an interaction between these two vegetation components with regard to salamander counts. The two species making up the vast majority of invasive species cover, *R. cathartica* and *Solidago altissima*, were analyzed individually, using linear regression for *S. altissima* and Analysis of Variance (ANOVA) for *R. cathartica*, as described next.

Because *R. cathartica* grows as a shrub well over one meter tall, its prevalence was likely underestimated based on the vegetation quadrat sampling (which only counted woody species

Table 1. Capture totals and sizes for two cohorts of Blue-spotted Salamanders (*Ambystoma laterale*).

Capture dates	Mean weight ± SD (g)	Mean SVL ± SD (cm)	Total number of captures	Number of re-captures
2 April–16 July	4.7 ± 1.6	4.7 ± 0.6	142	39
6–27 August	1.6 ± 1.3	3.2 ± 0.7	119	10

under 1 m tall). I therefore also specifically compared salamander captures within *R. cathartica* thickets to captures within areas of varying degrees of floristic quality. *Rhamnus cathartica* thickets were easily distinguished from other areas by their nearly ubiquitous *R. cathartica* cover and extensive bare ground. I performed an ANOVA of the number of new captures versus floristic quality, with "buckthorn thicket" as one floristic quality group, to address the hypothesis that salamanders were avoiding buckthorn thickets in favor of areas with greater floristic quality. Habitat quality was defined by the Floristic Quality Index (FQI), which incorporates species richness and composition as follows. Each plant in the flora of the Chicago region has been assigned by collective, expert judgment a Coefficient of Conservatism ranging from 0 to 10 that indicates the degree to which the species is faithful to high-quality natural communities. The FQI = (Mean C) × √N, where C is the Coefficient of Conservatism and N is the number of native species (Swink and Wilhelm 1994; Taft et al. 1997). The FQI at the scale of the ¼ m² quadrat ranges from 9–15 for the highest quality habitats, with FQI of 7–9 indicating good habitat, and scores below 7 indicating fair or poor quality habitat (unpublished observations).

Statistical analyses were conducted using R version 3.5.1 (R Core Team 2018). For parametric tests, normality was assessed using visual analysis of frequency histograms and normal quantile plots (Whitlock and Schluter 2009). Square root transformation of count data was used to meet assumptions of normality. All habitat variables met the assumptions of normality necessary for correlation tests, with the exception of *S. altissima* cover, which was not amenable to transformation. I therefore used a categorical, binary dummy variable within regression to allow *S. altissima* cover to be analyzed along with the other quantitative variables (Quinn and Keough 2002). Visual inspection of the relationship between *S. altissima* and salamander abundance showed a non-linear relationship, with an apparent threshold value of about 7% *S. altissima* cover. I used this apparent threshold to assign a value of 0 or 1 to *S. altissima* cover for inclusion in the linear model. To address the risk of overfitting the data by this *post hoc* selection of a threshold value, I also tested the *S. altissima* model with the first year (August) metamorphs as an out-of-sample group.

To assess support for models representing alternative predictive habitat variables, I ranked models using Akaike's Information Criterion (AIC; Burnham and Anderson 2002) and adjusted R² values from linear regression.

Results

Our monitoring team recorded 261 total captures, with 212 new captures and 49 recapture events (Table 1). Twenty-nine individuals were recaptured, with seven captured more than once (thus accounting for the difference between the number of recapture events and the number of recaptured individuals). All
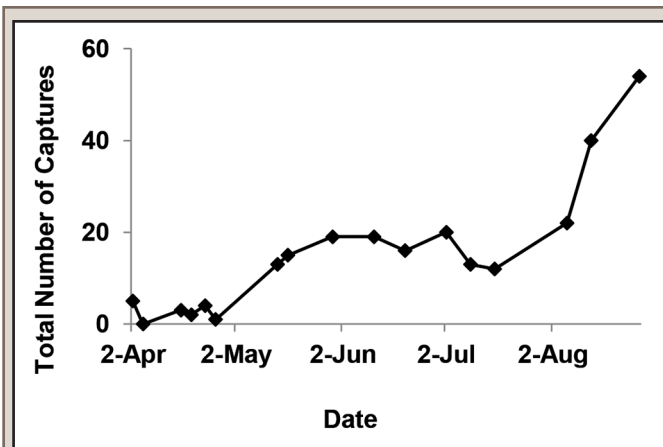
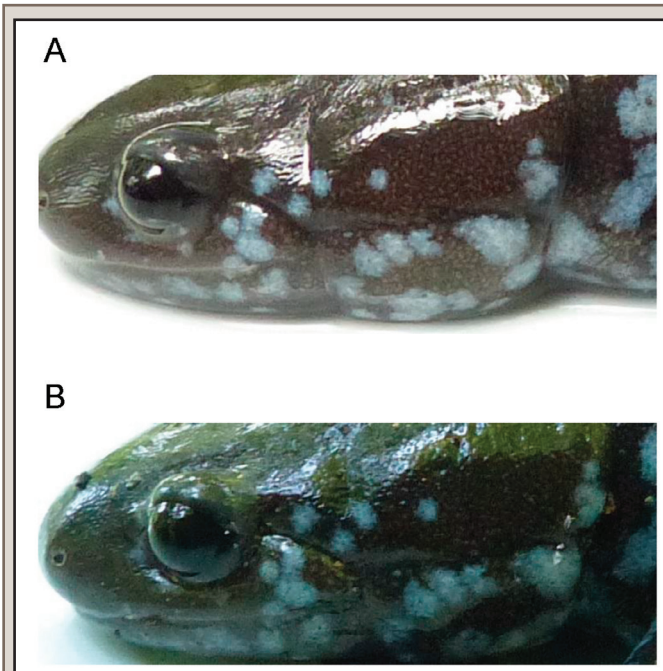FIG. 3. Total number of Blue-spotted Salamanders (*Ambystoma laterale*) captured on different dates, 2018.

TABLE 2. Regression table examining variation in salamander counts explained by number of plant species and invasive species cover.

Coefficient	Estimate	Standard Error	*t*-value	*P*-value
Intercept	2.12	0.527	4.03	0.0007
Number of plant species	0.0547	0.0788	0.694	0.496
Cover by invasive plant species	-0.0189	0.00899	-2.11	0.0488

Multiple R-squared = 0.1909; Adjusted R-squared = 0.1058
F = 2.242, df = 2 and 19, *P* = 0.1336

FIG. 4. A Blue-spotted Salamander (*Ambystoma laterale*) captured on 6 August 2018 (a) and again on 13 August 2018 (b). Spot patterns are clearly recognizable as the same individual. (Photographed by Karen Glennemeier.)

collection. Photo recognition software training and use required approximately three person-hours per visit, with manual photo inspection requiring an additional one person-hour per visit.

Capture numbers were low in early spring and then increased mid-May through July, followed by a dramatic increase mid-August (Fig. 3). Based on animal size, the vast majority of August captures appeared to be first-year metamorphs. There was no relationship between distance to the nearest pond and the first date at which animals were found within a transect (*t* = -0.699, *P* = 0.493; data not shown).

Fig. 4 shows an individual captured on two different dates and illustrates the confidence with which individual salamanders can be identified according to their spot patterns. Of the 49 manually identified recapture events, only 21 matches, or 43%, were identified by the Wild-ID pattern recognition program.

I tested the degree to which salamander counts were explained by vegetation species diversity or invasive plant prevalence, using linear regression of these two vegetation variables against the number of salamander captures per transect as the dependent variable. The regression model explained 11% of the variance in salamander numbers, with a significant effect of invasive species cover but not of vegetation species number (Table 2). Invasive species were primarily comprised of *S. altissima* and *R. cathartica*, so these species were next examined individually for their ability to explain salamander numbers. Cover by *S. altissima* (high vs. low) explained 29% of the variance in salamander numbers (*P* = 0.005, *t* = -3.11, Adj. R^2 = 0.293). There were no differences in salamander abundance among different habitats, with habitats defined as *R. cathartica* thicket or varying levels of floristic quality (Fig. 5; ANOVA: *F* = 0.539, *P* = 0.661).

I then tested whether the *S. altissima* effect might be explained by other habitat variables known to be associated with salamander abundance, including canopy cover, leaf litter depth, and coarse woody debris. *Solidago altissima* cover was somewhat correlated with canopy cover (Pearson's correlation: r = -0.639, df = 20, *P* = 0.001), but not with leaf litter depth (Pearson's correlation: r = -0.240, df = 20, *P* = 0.282) or coarse woody debris (Pearson's correlation: r = 0.0482, df = 20, *P* = 0.831). Canopy cover alone did not significantly affect salamander numbers (*P* = 0.218, *t* = 1.27, Adj. R^2 = 0.0968). Due to their strong auto-correlation, I was unable to include both *S. altissima* and canopy cover in a single regression. However, comparison of the two separate models suggests that *S. altissima* was a modestly stronger predictor than canopy cover, due to the former's higher adjusted R^2 value (*S. altissima* R^2 = 0.293, canopy cover R^2 = 0.0968) and

but one recaptured animal was found within the same transect as its original capture, and 23 of the 29 recaptured animals were found under the same cover board as their original capture. Of the six salamanders that moved between boards but within the same transect, three were found in the board adjacent to the original board (3 m distance), and three were found two boards (6 m) away from the original board. Time between recaptures ranged from 1 to 11 weeks, although animals may have remained in place after the final survey date in late August, meaning that the 11-week maximum may be an underestimate of the time animals spent in one location.

Each site visit averaged just under four hours in duration, with a team of 3–4 people covering all 22 transects each visit. Our team included the principle investigator, two interns, and seven volunteers or site stewards who alternated monitoring days. Training was conducted on site, concurrent with data

lower AIC value (*S. altissima* AIC = -10.5, canopy cover AIC = -5.17). Models that differ by more than 2 AIC units from the top model are generally considered to be less supported than the top model (Burnham and Anderson 2002). Figure 6 illustrates the relationships between salamander counts and each of these two habitat variables. The out-of-sample test of August captures against *S. altissima* cover also showed a significant effect, with 21% of the variation in salamander number explained by *S. altissima* cover ($P < 0.001$, $t = 8.96$, Adj. $R^2 = 0.206$).

<center>DISCUSSION</center>

Goal Number One.—Most land managers and stewards focus their direct efforts primarily on the vegetation community, with increased plant diversity being the most widely cited management goal for Midwestern ecosystems (e.g., Laatsch and Anderson 2000; Jackson 2009; Reid et al. 2020). Healthy wildlife populations are nonetheless a high priority for most ecological restoration efforts (CRBC 1999), and amphibians in particular are of high concern due to global population declines (Wake 1991), their potential for signaling the health of isolated wetlands (Simon et al. 2000; Crewe and Timmermans 2005), and their important ecosystem roles (Gibbons et al. 2006; Regester et al. 2006). Despite these concerns, the impacts of plant diversity on amphibian numbers remain poorly studied (but see Nuzzo and Mierzwa 2000).

A potential mechanism by which vegetation composition may affect amphibian populations is through an increase in the availability of prey items. Many studies have demonstrated that native plant species host a greater abundance and diversity of insects compared to non-native species (Herrera and Dudley 2003; Burghardt and Tallamy 2013; McCary et al. 2015), and some studies have found greater insect abundance and diversity within more diverse vegetation plots compared to less diverse plots (Haddad et al. 2001). Maerz et al. (2005a) found lower weight gain in Green Frogs (*Lithobates clamitans*) within areas invaded by Japanese Knotweed and with lower plant diversity, compared to un-invaded, higher diversity areas. Some studies also suggest that the composition of plant species within the terrestrial habitat impacts habitat choice or quality for amphibians. For example, Maerz et al. (2005a) found lower frog weight gain in plots invaded by *F. japonica*, based on a hypothesized reduction in arthropod abundance, and Belasen et al. (2013) found a preference in salamanders for Red Maple (*Acer rubrum*) versus Black Cherry (*Prunus serotina*) leaf litter, possibly mediated through chemical cues.

I found no relationship in the current study between plant diversity and salamander numbers, suggesting that, although plant diversity is vitally important for ecosystem health and function (Tilman et al. 1997; Isbell 2011; Cardinale et al. 2012), adult salamanders do not appear to be directly responding to terrestrial plant diversity when selecting burrowing sites. As a contrasting example, both bird abundance and bird diversity have shown positive relationships with native plant abundance and diversity (Burghardt et al. 2008; Bonifacio et al. 2011). Although avian prey such as caterpillars and other herbivorous insects show distinct preferences for native plant species with which they have co-evolved (Tallamy 2004; Burghardt and Tallamy 2013), salamander prey may be less sensitive to the composition of the herbaceous layer compared to other invertebrate prey (but see McCary et al. 2015). *Ambystoma laterale* prey consist largely of generalist herbivores such as slugs and earthworms, whose own diets include a significant proportion of decaying plant matter,
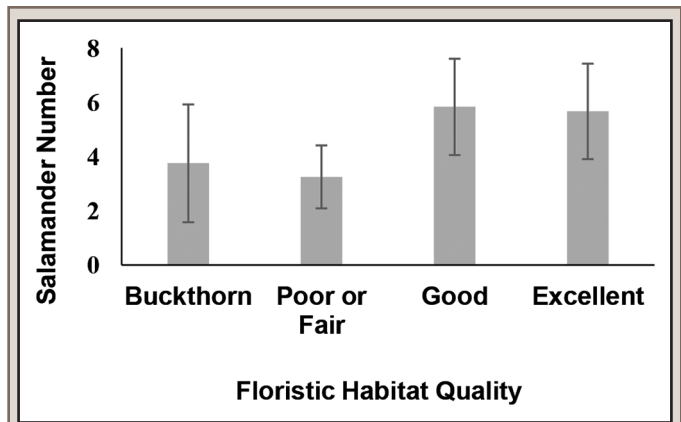

FIG. 5. Abundance of Blue-spotted Salamanders (*Ambystoma laterale*) versus floristic habitat quality. Abundance represents the mean total new captures per transect within each habitat group for salamanders captured before August. Error bars represent the standard errors of the mean. Sample size for each group, from left to right, was N = 4, 7, 5, and 6 transects, respectively.

fungi, and soil components such as bacteria (Chichester and Lowell 1973; Keller and Snell 2002). The indirect importance of a diverse plant community on salamanders is likely profound, through effects on ecosystem processes, stability, and productivity (Tilman et al. 1997; Haddad et al. 2011), and soil invertebrates may be affected by the presence of invasive plant species (Heneghan et al. 2007; McCary et al. 2015). However, salamanders may not be cueing into or making choices based on the living plant community at a fine scale. Whether they are maximizing growth and reproduction within these low-diversity plant communities or are simply "making due" within sub-optimal habitat (see Van Horne 1983) is unknown; to answer this question will require long-term studies that follow individuals over time.

Amphibians also can be negatively impacted by particular plant species and may avoid these species when selecting terrestrial habitat. Recent studies have found that invasive species such as *R. cathartica*, Amur Honeysuckle (*Lonicera maackii*), *F. japonica*, and Purple Loosestrife (*Lythrum salicaria*) may have negative impacts on amphibians. Most of these studies have focused on the egg and larval stages of anurans, finding direct, allelopathic effects on embryo survival and tadpole survival and behavior (Maerz et al. 2005b; Brown et al. 2006; Hickman and Watling 2014; Sacerdote and King 2014; but see Cohen et al. 2018).

I found a weak but significant effect of another invasive species, *S. altissima*, on *A. laterale* numbers. The negative effect did not appear to be wholly explained by the correlation between *S. altissima* and canopy cover, although this explanation cannot be entirely ruled out based on the current study. Although native to North America, *S. altissima* behaves similarly to the invasive species mentioned above by forming aggressive monocultures that land managers actively work to control (Chicago Wilderness 2016). A 2005 survey of protected grasslands throughout the Chicago Wilderness region found *S. altissima* to be the most abundant herbaceous species present (Glennemeier 2006), and woodlands undergoing restoration are also heavily impacted by this species (Glennemeier 2004; Cook County Land Audit 2008, unpubl. data).

If the negative relationship between *S. altissima* and *A. laterale* proves robust to further testing, it is worth asking why salamanders may avoid dense goldenrod patches. Ambystomatid adults use burrows, typically created by small mammals, to
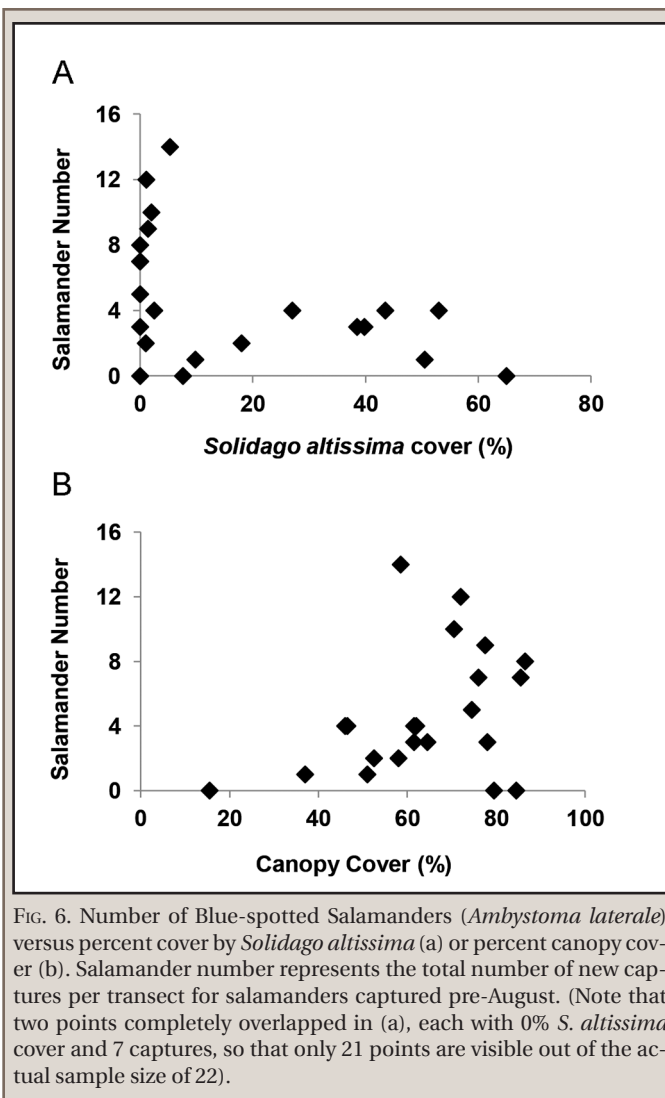
Fig. 6. Number of Blue-spotted Salamanders (*Ambystoma laterale*) versus percent cover by *Solidago altissima* (a) or percent canopy cover (b). Salamander number represents the total number of new captures per transect for salamanders captured pre-August. (Note that two points completely overlapped in (a), each with 0% *S. altissima* cover and 7 captures, so that only 21 points are visible out of the actual sample size of 22).

maintain skin moisture levels, and salamanders fare poorly where burrowing is inhibited (Semlitsch 1983; Osbourn et al. 2014). The dense, rhizomatous root masses created by *S. altissima* may make burrowing into the soil more difficult (Meyer and Schmid 1999) and thus result in fewer burrows within dense *S. altissima* stands. It also is possible that dense *S. altissima* patches may be difficult for salamanders to move through. Movement and dispersal behavior may be influenced by fine-scale habitat features and may have critical impacts on long-term population persistence (Cosentino et al. 2011; Pittman et al. 2014). *Solidago altissima* also emits allelopathic chemicals that may inhibit the growth of other plants (Bing-yao et al. 2006; Abhilasha et al. 2008). These chemicals have not been tested for impacts on amphibians, but given amphibians' sensitivity to secondary metabolites from other plant species, (Maerz et al. 2005b; Hickman and Watling 2014; Sacerdote and King 2014), this hypothesis is worth testing. Alternatively, any effects of *S.altissima* may be indirect, through impacts on prey species or physical habitat. Whether salamander prey species avoid *S. altissima* is unknown, but Cates and Orians (1975) found low palatability of *Solidago canadensis* (a closely related goldenrod species) to one species of slug (the Banana Slug, *Ariolimax columbianus*, native to the northwestern USA).

Future research should directly test the hypothesis that *A. laterale* adults avoid areas of dense *S. altissima* cover, using experiments that manipulate *S. altissima* density while holding canopy cover and other associated habitat variables constant. Behavioral choice experiments as in Patrick et al. (2008), Belasen et al. (2013), and Feuka (2017) would help determine whether salamanders are directly avoiding *S. altissima*, versus indirect effects mediated through prey items or physical habitat.

It is worth noting that other studies have found significant impacts of canopy cover on ambystomatid salamander abundance and survival (Rothermel and Luhring 2005; Rothermel and Semlitsch 2006; Osbourn et al. 2014). In the current study, I found no difference in salamander abundance with changes in canopy cover. However, the range of values for these variables in the current study was relatively small (37% to 87%, Fig. 6), as I confined this study to woodlands that were generally considered to be good habitat for salamanders (Phillips et al. 1999) but that differed in fine-scale habitat quality. Therefore, I may not have sampled across the range of these variables necessary to see an effect of canopy cover on salamander numbers.

Goal Number Two.—The successful use of affordable and locally sourced cover material, non-invasive monitoring and photo-mark-recapture methods, and reasonable time investments highlights the potential for increased monitoring of fossorial salamanders by site stewards and land managers. With approximately weekly half-day site visits, we were able to learn the distribution of *A. laterale* adults throughout the site, track the arrival of (presumed) first-year metamorphs in the terrestrial habitat, establish the typical adult summer range, and examine salamander habitat choices. Although the principal investigator was present for every visit during the current study, all monitors received sufficient training and experience after one half-day visit to enable them to conduct similar surveys on their own in the future.

The "flitch wood" cover boards, made from local, native ash species, were well-used by *A. laterale* adults, making it a suitable addition to the list of accepted cover board materials used to assess amphibian habitat preference and relative abundance (DeGraaf and Yamasaki 1992; Moore 2005; Dodd 2010). The use of flitch wood from native tree species may be particularly appropriate for high quality natural areas where stewards wish to minimize the introduction of foreign materials such as plywood, rubber, or metal boards.

Many authors have raised concerns over detection probabilities using surface counts of fossorial animals (e.g., Dodd and Dorazio 2004; Bailey et al. 2018), while other studies support the use of such counts as a correlate for relative abundance (Smith and Petranka 2000). The ability to account for imperfect detection continues to improve as new analytical methods are developed (O'Donnell and Semlitsch 2015). In the current study, the use of standardized cover objects and the spatially and temporally replicated approach should have ameliorated many of the concerns related to variation in counts over space and time, while the focus on a single salamander species removes concerns over species-specific biases in detection probabilities.

Photo mark recapture proved promising as a means for site stewards to track individual salamanders across space and time, as the technique was non-invasive and easily learned. Risks to animals were limited to those incurred by handling stress and potential disease transmission, both of which were minimized with appropriate training, protective gear, and decontamination techniques. Restricting the monitoring to site stewards who were only monitoring a single site further reduced the risk of disease transmission. The Wild-ID pattern recognition software was of

limited help in identifying matches, with failure rates similar to those reported elsewhere (Morrison et al. 2016). Closer cropping of the images to remove all extraneous shapes and colors might have improved program accuracy. However, if sample sizes are modest, stewards may find manual inspection of spot patterns to be more efficient, effective, and accessible than pattern recognition software, especially given the time required to learn and use the software.

Photo identification mark-recapture revealed that *A. laterale* adults did not move far after their initial foray away from breeding ponds, consistent with other studies showing small summer home ranges for adult ambystomatids (Semlitsch 1981). For site stewards wishing to monitor salamanders with minimal time commitment, this pattern suggests that future monitoring of terrestrial habitat might be adequately conducted with a handful of visits concentrated in late spring, to establish the locations adults have selected and where they presumably will remain for the duration of the season. If first-year metamorphs are also of interest, then the current results suggest that a second round of monitoring should be conducted in August. With a very manageable investment of time and resources, such targeted monitoring would provide critical information about adult salamander habitat use that can be incorporated into management planning.

<div align="center">CONCLUSION</div>

Given the important ecosystem role of *Ambystoma laterale* as the lone salamander species at the study site, increased understanding of its habitat needs is vital, particularly with regard to components of the vegetation community that can be influenced by site stewards and land managers. The current study suggests that rigorous data on salamander habitat choices can be obtained with reasonable investments of time, training, and resources, and that *A. laterale* adults may be sensitive to the presence of invasive species within otherwise suitable habitat.

<div align="center">LITERATURE CITED</div>

ABHILASHA, D., N. QUINTANA, J. VIVANCO, AND J. JOSHI. 2008. Do allelopathic compounds in invasive *Solidago canadensis s.l.* restrain the native European flora? J. Ecol. 96:993–1001.
BAILEY, L. L., T. R. SIMONS, AND K. H. POLLOCK. 2018. Estimating detection probability parameters for *Plethodon* salamanders using the robust capture-recapture design. J. Wildl. Manag. 68:1–13.
BELASEN, A, E. BURKETT, A. INJAIAN, K. LI, D. ALLEN, AND I. PERFECTO. 2013. Effect of sub-canopy on habitat selection in the blue-spotted salamander (*Ambystoma laterale-jeffersonianum* unisexual complex). Copeia 2013:254–261.

BING-YAO, S, T. JIAN-ZHONG, W. ZHI-GANG, G. FU-GEN, AND Z. MING-DE. 2006. Allelopathic effects of extracts from *Solidago canadensis L.* against seed germination and seedling growth of some plants. J. Env. Sci. 18:304–306.
BOLGER, D. T., T. A. MORRISON, B. VANCE, D. LEE, AND H. FARID. 2012. A computerassisted system for photographic mark–recapture analysis. Meth. Ecol. Evol. 3:813–822.
BONIFACIO, R. S., C. M. KINROSS, G. M. GURR, AND H. NICOL. 2011. The effect of woody plant diversity and other stand and landscape factors on the diversity and abundance of birds using farm shelterbelts. Pac. Conserv. Biol.17:22–35.
BROWN, C. J., B. BLOSSEY, J. C. MAERZ, AND S. J. JOULE. 2006. Invasive plant and experimental venue affect tadpole performance. Biol. Inv. 8:327–338.
BURGHARDT, K. T., D. W. TALLAMY, AND G. SHRIVER. 2008. Impact of native plants on bird and butterfly biodiversity in suburban landscapes. Conserv. Biol. 23:219–224.
———, AND ———. 2013. Plant origin asymmetrically impacts feeding guilds and life stages driving community structure of herbivorous arthropods. Div. Distrib. 19:1553–1565.
BURNHAM, K. P., AND D. R. ANDERSON. MULTIMODEL INFERENCE: UNDERSTANDING AIC AND BIC IN MODEL SELECTION. SOC. METH. RES. 33:261–304.
CARDINALE, B. J, J. E. DUFFY, A. GONZALEZ, D. U. HOOPER, C. PERRINGS, P. VENAIL, ET AL. 2012. Biodiversity loss and its impact on humanity. Nature 486:59–67.
CATES, R. G., AND G. H. ORIANS. 1975. Successional status and the palatability of plants to generalized herbivores. Ecology 56:410–418.
CHICAGO WILDERNESS. 2016. Native Landscape and Ecological Restoration Guide. Prepared by the Native Landscape and Restoration Contractor Selection Guide Working Group, Chicago, Illinois.
CHICHESTER, L. F., AND L. G. LOWELL. 1973. The terrestrial slugs of northeastern North America. Sterkiana 51:11–42.
COHEN, J. S., J. C. MAERZ, AND B. BLOSSEY. 2018. Traits, not origin, explain impacts of plants on larval amphibians. Ecol. Appl. 22:218–228.
COSENTINO, B. J., R. L. SCHOOLEY, AND C. A. PHILLIPS. 2011. Connectivity of agroecosystems: Dispersal costs can vary among crops. Landscape Ecol. 26:371–379.
CRBC (CHICAGO REGION BIODIVERSITY COUNCIL). 1999. Biodiversity Recovery Plan. Chicago Region Biodiversity Council, Chicago, Illinois.
CREWE, T. L., AND S. T. A. TIMMERMANS. 2005. Assessing Biological Integrity of Great Lakes Coastal Wetlands Using Marsh Bird and Amphibian Communities. Technical Report. Marsh Monitoring Program, Bird Studies Canada.
DEGRAAF, R. M., AND M. YAMASAKI. 1992. A nondestructive technique to monitor the relative abundace of terrestrial salamanders. Wildl. Soc. Bull. 20:260–264.
DENGLER, J. 2008. Pitfalls in small-scale species-area sampling and analysis. Folia Geobot. 43:269–287.
DODD, C. K. 2010. Amphibian Ecology and Conservation: A Handbook of Techniques. Oxford University Press, Oxford, UK.
———, AND R. M. DORAZIO. 2004. Using counts to simultaneously estimate abundance and detection probabilities in a salamander community. Herpetologica 60:468–478.
DROEGE, S., AND P. EAGLE. 2005. Evaluating calling frog surveys. *In* M. Lannoo (ed.), Amphibian Declines: The Conservation Status of United States Species, pp. 314–319. University of California Press, Berkeley, California.
FAHEY, R. T., L. DARLING, AND J. ANDERSON. 2015. Oak Ecosystems Recovery Plan: Sustaining Oaks in the Chicago Wilderness Region. Chicago Wilderness Oak Ecosystems Recovery Working Group, Chicago, Illinois.
FEUKA, A. 2017. Effects of light pollution on habitat selection in post-metamorphic wood frogs and unisexual blue-spotted salamanders. Herpetol. Conserv. Biol. 12:470–476.
GIBBONS, J. W., C. T. WINNE, D. E. SCOTT, J. D. WILLSON, X. GLAUDAS, K. M. ANDREWS, ET AL. 2006. Remarkable amphibian biomass and abundance in an isolated wetland: Implications for wetland conservation. Conserv. Biol. 20:1457–1465.

GLENNEMEIER, K. 2004. The state of our wooded lands: results from the Chicago Wilderness woods audit. Chicago Wilderness J. 2:16–22.

———. 2006. The state of our grasslands: results from the Chicago Wilderness grassland audit. Chicago Wilderness J. 4:35–40.

GOODE, M. J., D. E. SWANN, AND C. R. SCHWALBE. 2004. Effects of destructive collecting practices on reptiles: A field experiment. J. Wildl. Manag. 68:429–434.

GRAETER, G. J., B. B. ROTHERMEL, AND J. W. GIBBONS. 2008. Habitat selection and movement of pond-breeding amphibians in experimentally fragmented pine forests. J. Wildl. Manag. 72:473–482.

HADDAD, N. M., D. TILMAN, J. HAARSTAD, M. RITCHIE, AND J. M. H. KNOPS. 2001. Contrasting effects of plant richness and composition on insect communities: a field experiment. Am. Nat. 158:17–35.

HENEGHAN, L., C. RAUSCHENBERG, F. FATEMI, AND M. WORKMAN. 2004. European buckthorn (*Rhamnus cathartica*) and its effects on some ecosystem properties in an urban woodland. Ecol. Restor. 22:275–280.

———, J. STEFFEN, AND K. FAGEN. 2007. Interactions of an introduced shrub and introduced earthworms in an Illinois urban woodland: Impact on leaf litter decomposition. Pedobiologia 50:543–551.

HERRERA, A. M., AND T. L. DUDLEY. 2003. Reduction of riparian arthropod abundance and diversity as a consequence of giant reed (*Arundo donax*) invasion. Biol. Inv. 5:167–177.

HEYER, W. R., M. A. DONNELLY, R. W. MCDIARMID, L. C. HAYEK, AND M. S. FOSTER (eds.). 1994. Measuring and Monitoring Biological Diversity: Standard Methods for Amphibians. Smithsonian Institution Press, Washington, DC. 364 pp.

HICKMAN, C. R., AND J. I. WATLING. 2014. Leachates from an invasive shrub causes risk-prone behavior in a larval amphibian. Behav. Ecol. 25:300–305.

HOFFMAN, K., M. HUNTER JR., A. J. K. CALHOUN, AND J. BOGART. 2018. Post-breeding migration and habitat use of unisexual salamanders in Maine, USA. J. Herpetol. 52:273–281.

ILLINOIS NATURAL HERITAGE DATABASE. 2020. https://www2.illinois.gov/dnr/ESPB/.

INDIANA NATURAL HERITAGE DATA CENTER. 2020. https://www.in.gov/dnr/naturepreserve/.

ISBELL, F., V. CALCAGNO, A. HECTOR, J. CONNOLLY, AND W. S. HARPOLE. 2011. High plant diversity is needed to maintain ecosystem services. Nature 477:199–202.

JACKSON, P. W. 2009. Temporal and spatial vegetation dynamics in a remnant oak savanna: Middlefork Savanna, Lake County, Illinois. Erigenia 22:9–35.

KELLER, H. W., AND K. L. SNELL. 2002. Feeding activities of slugs on Myxomycetes and macrofungi. Mycologia 94:757–760.

LAATSCH, J. R., AND R. C. ANDERSON. 2000. An evaluation of oak woodland management in northeastern Illinois, USA. Nat. Areas J. 20:211–220.

MAERZ, J. C., B. BLOSSEY, AND V. NUZZO. 2005a. Green frogs show reduced foraging success in habitats invaded by Japanese knotweed. Biodiv. Conserv. 14:2901–2911.

———, C. J. BROWN, C. T. CHAPINS, AND B. BLOSSEY. 2005b. Can secondary compounds of an invasive plant affect larval amphibians? Funct. Ecol. 19:970–975.

MCCARY, M. A., C. M. MARTINEZ, L. UMEK, L. HENEGHAN, AND D. H. WISE. 2015. Effects of woodland restoration and management on the community of surface-active arthropods in the metropolitan Chicago region. Biol. Conserv. 190:154–166.

MCCLAIN, W. E., T. L. ESKER, B. R. EDGIN, G. SPYREAS, AND J. E. EBINGER JE. 2010. Fire history of a post oak (*Quercus stellata* Wang.) woodlands in Hamilton County, Illinois USA. Castanea 75:461–474.

MEYER, A. H., AND B. SCHMID. 1999. Experimental demography of the oldfield perennial *Solidago altissima*: the dynamics of the shoot population. J. Ecol. 87: 17–27.

MOORE, J. D. 2005. Use of native dominant wood as a new coverboard type for monitoring eastern red-backed salamanders. Herpetol. Rev. 36:268–271.

MORRISON, T. A., D. KEINATH, W. ESTES-ZUMPF, J. P. CRALL, AND C. W. STEWART.

2016. Individual identification of the endangered Wyoming toad *Anaxyrus baxteri* and implications for monitoring species recovery. J. Herpetol. 50:44–49.

MWPARC (MIDWEST PARTNERS IN AMPHIBIAN AND REPTILE CONSERVATION). 2012. Field Herpetology Etiquette. http://www.mwparc.org/

NEPARC (NORTHEAST PARTNERS IN AMPHIBIAN AND REPTILE CONSERVATION). 2014. Disinfection of Field Equipment to Minimize Risk of Spread of Chytridiomycosis and Ranavirus. Northeast Partners in Amphibian and Reptile Conservation Publication 2014–02.

NUZZO, V. A., AND K. S. MIERZWA. 2000. The effect of forest structure on amphibian abundance and diversity in the Chicago region. Report to Citizens for Conservation, Fox River Grove, Illinois.

O'DONNELL, K. M., AND R. D. SEMLITSCH. 2015. Advancing terrestrial salamander population ecology: The central role of imperfect detection. J. Herpetol. 49:533–540.

OSBOURN, M. S., G. M. CONNETTE, AND R. D. SEMLITSCH. 2014. Effects of fine-scale forest habitat quality on movement and settling decisions in juvenile pond-breeding salamanders. Ecol. Appl. 24:1719–1729.

PACKARD, S., AND C. F. MUTEL. 2005. The Tallgrass Restoration Handbook for Prairies, Savannas, and Woodlands – Science and Practice of Ecological Restoration. 2nd ed. English Island Press, Washington, D.C. 504 pp.

PALMER, M. A., R. F. AMBROSE, AND N. L. POFF. 1997. Ecological theory and community restoration ecology. Restor. Ecol. 5:291–300.

PATRICK, D. A., E. B. HARPER, M. L. HUNTER, AND A. J. K. CALHOUN. 2008. Terrestrial habitat selection and strong density-dependent mortality in recently metamorphosed amphibians. Ecology 89:2563–2574.

PHILLIPS, C. A., R. A. BRANDON, AND E. O. MOLL. 1999. Field Guide to Amphibians and Reptiles of Illinois. Illinois Natural History Survey Manual 8. Champaign, Illinois. 282 pp.

PITTMAN, S. E., M. S. OSBOURN, AND R. D. SEMLITSCH. 2014. Movement ecology of amphibians: A missing component for understanding population declines. Biol. Conserv. 169:44–53.

QUINN, G., AND M. KEOUGH. 2002. Experimental Design and Data Analysis for Biologists. Cambridge University Press, New York. 553 pp.

R CORE TEAM. 2018. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

REGESTER, K. J., M. R. WHILES, AND C. M. TAYLOR. 2006. Decomposition rates of salamander (*Ambystoma maculatum*) life stages and associated energy and nutrient fluxes in ponds and adjacent forest in southern Illinois. Copeia 2006:640–649.

REGOSIN, J. V. B., S. WINDMILLER, R. N. HOMAN, AND J. M. REED. 2005. Variation in terrestrial habitat use by four pool-breeding amphibian species. J. Wildl. Manag. 69:1481–1493.

REID, J. L., N. J. HOLMBERG, M. ALBRECHT, S. ARANGO-CARO, O. HAJEK, Q. LONG, AND J. TRAGER. 2020. Annual understory plant recovery dynamics in a temperate woodland mosaic during a decade of ecological restoration. Nat. Areas J. 40:23–34.

ROTHERMEL, B. B., AND T. M. LUHRING. 2005. Burrow availability and desiccation risk of mole salamanders (*Ambystoma talpoideum*) in harvested versus unharvested forest stands. J. Herpetol. 39:619–626.

———, AND R. D. SEMLITSCH. 2006. Consequences of forest fragmentation for juvenile survival in spotted (*Ambystoma maculatum*) and marbled (*Ambystoma opacum*) salamanders. Can. J. Zool. 84:797–807.

RYAN, K. J., AND A. J. K. CALHOUN. 2014. Postbreeding habitat use of the rare, pure-diploid blue-spotted salamander (*Ambystoma laterale*). J. Herpetol. 48:556–566.

SACERDOTE, A. B., AND R. B. KING. 2014. Direct effects of an invasive European buckthorn metabolite on embryo survival and development in *Xenopus laevis* and *Pseudacris triseriata*. J. Herpetol. 48:51–58.

SANNOLO, M., F. GATTI, M. MANGIACOTTI, S. SCALI, AND R. SACCHI. 2016. Photo-identification in amphibian studies: a test of I^{3}S pattern. Acta Herpetol. 11:63–68.

SEMLITSCH, R. D. 1981. Terrestrial activity and summer home range

of the mole salamander (*Ambystoma talpoideum*). Can. J. Zool. 59:315–322.

———. 1983. Burrowing ability and behavior of salamanders of the genus *Ambystoma*. Can. J. Zool. 61:616–620.

———. 1998. Biological delineation of terrestrial buffer zones for pond-breeding salamanders. Conserv. Biol. 12:1113–1119.

SIMON, T. P., R. JANKOWSKI, AND C. MORRIS. 2000. Modification of an index of biotic integrity for assessing vernal ponds and small palustrine wetlands using fish, crayfish, and amphibian assemblages along southern Lake Michigan. Aquat. Ecosyst. Health Manag. 3:407–418.

SMITH, C. K., AND J. W. PETRANKA. 2000. Monitoring terrestrial salamanders: Repeatability and validity of area-constrained cover object searches. J. Herpetol. 34:547–557.

SPEYBROECK, J., AND K. STEENHOUDT. 2017. A pattern-based tool for long-term, large-sample capture-mark-recapture studies of fire salamanders *Salamandra* species (Amphibia: Urodela: Salamandridae). Acta Herpetol.12:55–63.

SPYREAS, G. 2016. Scale and sampling effects on floristic quality. PLoS ONE 11:e0160693.

SULLIVAN, J. 2003. An Atlas of Biodiversity. Chicago Wilderness, Chicago, Illinois. 66 pp.

SWINK, F., AND G. WILHELM. 1994. Plants of the Chicago Region. 4th ed. Indiana Academy of Science, Indianapolis. 936 pp.

TAFT, J., G. WILHELM, D. LADD, AND L. MASTERS. 1997. Floristic quality assessment for vegetation in Illinois: A method for assessing vegetation integrity. Erigenia 1977:3–95.

TALLAMY, D. W. 2004. Do alien plants reduce insect biomass? Conserv. Biol. 18:1689–1692.

TILMAN, D., J. KNOPS, D. WEDIN, P. REICH, M. RITCHIE, AND E. SIEMANN. 1997. The influence of functional diversity and composition on ecosystem processes. Science 277:1300–1302.

URIAN, K., A. GORGONE, A. READ, B. BALMER, R. WELLS, ET AL. 2014. Recommendations for photo-identification methods used in capture-recapture models with cetaceans. Mar. Mamm. Sci. 31:298–321.

VAN HORNE, B. 1983. Density as a misleading indicator of habitat quality. J. Wildl. Manag. 47:893–901.

VANNI, M. J. 2002. Nutrient cycling by animals in freshwater ecosystems. Ann. Rev. Ecol. Syst. 33:341–370.

WAKE, D. B. 1991. Declining amphibian populations. Science 253:860.

WALKER, B. H. 1992. Biodiversity and ecological redundancy. Conserv. Biol. 6:18–23.

WANG, Y., AND D. K. MOSKOVITS. 2001. Tracking fragmentation of natural communities and changes in land cover: Applications of Landsat data for conservation in an urban landscape (Chicago Wilderness). Conserv. Biol. 15:835–843.

WARREN, R. J., A. LABATORE, AND M. CANDEIAS. 2017. Allelopathic invasive tree (*Rhamnus cathartica*) alters native plant communities. Plant Ecol. 218:1233–1241.

WERNER, P. A., I. K. BRADBURY, AND R. S. GROSS. 1980. The biology of Canadian weeds. 45. *Solidago canadensis* L. Can. J. Plant Sci. 60:1393–1409.

WHITLOCK. M. C., AND D. SCHLUTER. 2009. The Analysis of Biological Data. Roberts and Company Publishers, Greenwood Village, Colorado. 704 pp.

An Unanticipated Ecological Trap: Entrapment of the California Tiger Salamander in Technogenic Structures as a Confounding Stressor for a Threatened Species

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Numerous wildlife species have been captured or collected in natural and manmade ecological traps throughout the world (Hawken 1951; Brattstrom 1953; Delibes et al. 2001; Villa et al. 2018). The Rancho La Brea tar pits in Los Angeles County, California, provide a prime example of a natural trap, where millions of specimens and approximately 600 species were trapped in naturally occurring tar pits and have been preserved (Brattstrom 1953). Sites like Rancho La Brea collect indiscriminately, and mortality to individuals is nearly certain (Brattstrom 1953). These sites appear to represent natural habitats (i.e., aquatic features, retreat sites, etc.) to which species would normally be attracted (Lemon and Churcher 1971; Churcher 1966; Campbell 1979). Delibes et al. (2001) characterized these natural sites as attractive sinks and believed that this maladaptive selection of habitat or microhabitat is likely rare or goes unreported for most species. A more recent suite of potential population sinks is the ecological trap, whereby an individual or species mistakenly selects suboptimal habitat for its preferred habitat (Hale and Swearer 2010; Sih 2013). They