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Do species distribution models predict species richness in urban and natural green spaces? A case study using amphibians

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Research Highlights:

- Amphibian species richness maps significantly over-predicted species richness.
- Over-prediction may have partially been a result of undersampling during surveys.
- Over-prediction was likely due to poor model performance and undersampling.
- Despite over-prediction, models did project relative species richness well.

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1. Introduction

Much of global population growth is occurring in urban areas (United Nations, 2004; Wu et al. 2011) where over one-half of the United States population resides in urban areas (MacKun & Wilson, 2011). Nevertheless, human land use patterns are dynamic and some locations within urban areas are experiencing declining populations. Such declines can result in land abandonment and provide an opportunity to replace developed habitat with green infrastructure. It is well established that urbanization changes the biotic and abiotic properties of an ecosystem and these impacts can reach far outside the urban area (Gaston, 2010). To reduce these effects, there has been a movement to implement green infrastructure or incorporate green space to urban areas. The benefits of green space in urban systems include increased psychological well-being, recreational opportunities, and human health benefits (e.g., Barton & Pretty, 2010; Breuste & Qureshi, 2011; Tzoulas et al., 2007; van den Berg, Hartig, & Staats, 2007). These benefits are often predicated by ecosystem services and functions such green space in urban ecosystems provide (e.g., water filtration and quality; Bolund & Hunhammar, 1999; Faulkner, 2004; Gaston, Davies, & Edmondson, 2010). Whereas urban habitats may not act as smaller versions of undeveloped patches of land, they may still provide ecological and human-oriented benefits such as providing habitat connectivity, which helps to sustain regional biodiversity (Goddard, Dougill & Benton, 2009; Irvine et al., 2010; Luck & Smallbone, 2010), or providing permeable surface for stormwater infiltration, or water purification (Boyer & Polasky, 2004). Monitoring and management for biodiversity has inherent value (Connery, 2009) and biodiversity conservation within urban areas can help minimize extinction risk of some species and increase the value of biota to humans as they more frequently encounter wildlife (Goddard et al., 2009). Toward this

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end, metrics are needed to measure the degree to which urban green spaces sustain biota and subsequent biodiversity.

An ideal metric would use taxa that serve as indicators of overall biodiversity, provide an ecosystem service, and are a critical link to the biotic community within the green space (i.e., provide ecosystem functions). Amphibians are often the most abundant, diverse group of vertebrate organisms in forested and wetland systems, they serve as important food resources for higher trophic levels, and in many systems are considered the top-predators (Burton & Likens, 1975; Davic & Welsh, 2004; Gibbons et al., 2006). Amphibians are also considered to be indicators of environmental stress (DeGarady & Halbrook, 2006; Southerland et al., 2004; Welsh & Droege, 2001; Welsh & Ollivier, 1998), but see Kerby et al. (2010), and are known to provide a number of ecosystem functions in natural ecosystems (Davic & Welsh, 2004; Regester, Lips & Whiles, 2006; Regester & Whiles, 2006; Whiles et al., 2006). Moreover, amphibians in urban environments, like other biota, can enhance educational opportunities for human inhabitants (Pickett et al., 2001). Because of their importance to ecosystems, ability to indicate environmental stress, and education value, research involving amphibians in urban systems is warranted (Hamer & McDonnell, 2008; McDonnell & Hahs, 2008; Pickett et al., 2001; Smallbone, Luck & Wassens, 2011). Due to time and financial constraints associated with conducting biotic surveys, modeling methods may provide assistance in understanding the value of urban green space to this taxon.

Presence-only species distribution models (SDMs) are models that correlate species distribution records to environmental data to predict areas of suitable habitat for taxa (see review in Guisan & Zimmerman, 2000; Guisan & Thuiller, 2005; Elith et al., 2006). They are a group of approaches for identifying species distributions of undersampled species, predicting impacts

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of environmental change on distributions, and identifying areas of conservation importance (Elith & Leathwick, 2009). In recent years, validations of models in various forms have been increasing. For example, methods utilizing species occupancy or detection (Franklin et al., 2009; Rota et al., 2011), independent and non-independent data validation (Araujo et al., 2005), and the incorporation of field/survey data to inform or test model accuracy (Newbold et al., 2010; Pineda & Lobo, 2009; Trotta-Moreu & Lobo, 2010) have been examined. However, studies simply using field data to validate whether models are projecting species distribution correctly are rare.

Others have noted several limitations to SDMs including exclusion of biotic, geographical, or physiological constraints on species distributions, use of museum records that may be widely variable in both spatial and temporal quality, and issues relating to extrapolation of model predictions (see review in Elith & Leathwick, 2009). These limitations may be exacerbated when modeling within spatial extents that include urban environments, because species are sampled less in urban areas as ecologists tend to focus collections or research on natural areas (Gaston et al., 2010; Martin, Blossey & Ellis, 2012). Furthermore, lack of uniform sampling across gradients of development presents a challenge to using SDMs in an urban landscape because SDMs assume that biases in locality data (e.g., false absences) are not correlated with environmental gradients used to build projected distributions (Bean, Stafford & Brashares, 2012; Hijmans, 2012). In addition, error in the predictions of SDMs varies over large spatial scales (extent and resolution) due to increased spatial heterogeneity (Osborne, Foody & Suarez-Seoane, 2007; Smulders et al., 2010; Zhang & Zhang, 2007), such as variation of environmental, landscape, and habitat structure. This trend may be seen at smaller spatial scales (extent and resolution) when using fine-scale data to build models (e.g., 30 m resolution) such as in urban areas that have several classes of land use categories (e.g., habitat heterogeneity), as the

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increased heterogeneity in urban areas within a smaller spatial scale could pose similar prediction errors.

We tested whether species richness maps generated from SDMs can be used to prioritize areas of high biodiversity value in urban and non-urban green space. We asked if SDMs built using landscape variables associated with amphibian species richness could be used to project areas of suitable habitat. We addressed this question by comparing modeled species richness maps (based on accumulated individual SDMs) to field surveys across a number of urban and non-urban green spaces. In addition, we investigated the landscape-level predictors of observed amphibian species richness to determine what variables may be important to include or protect in the creation, management, or conservation of urban and non-urban green space.

2. Materials and Methods

2.1. Species Distribution Modeling using Maximum Entropy

We developed species distribution models using Maxent version 3.3.3a (Phillips & Dudik, 2008) for 23 species of amphibians with current distributions within Hamilton County, Ohio, U.S.A. Maxent is a software program that employs a machine learning method that is based on the principle of maximum entropy to model species distributions using presence-only data coupled with environmental data. Entropy is characterized by Shannon (1948) as "a measure of how much „choice" is involved in the selection of an event" and is utilized in the framework of maximum entropy to examine species geographic distributions (Phillips, Anderson & Schapire, 2006). The approach estimates habitat suitability based on an input set of environmental variables encompassing the region where a species is known to occur based on locality records. The program maximizes the entropy in the probability distribution of suitability across all areas

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92 of the distribution where empirical observations are lacking. For each species identified as
93 occurring in Hamilton County, OH, species presence data were obtained for the period of 1997–
94 2001from HerpNET (http://www.herpnet.org), Global Biodiversity Information Facility (Lane,
95 2003; GBIF; http://www.gbif.org), and personal collections of herpetologists (Appendix A). All
96 locality points were cross-referenced to each other and duplicate points were removed.
97 Furthermore, localities that fell outside the current species range (identified by county-level
98 distribution maps found in Lannoo, 2005) were not utilized to develop models. To maximize
99 model quality, each model was built using at least 20 point locations for each species (Wisz et
100 al., 2008).
101 We modeled the suitable habitat of each species across the National Hydrography Dataset
102 Plus (U.S. Geological Survey, 2005; NHDPlus; retrieved from http://www.horizon-
103 systems.com/nhdplus/) Region 05 Unit B watershed delineation. This delineation was necessary
104 to encompass the environmental variability of each species and provide predicted records from
105 across a larger geographic region adjacent to our primary study area. Additionally, amphibian
106 distributions can be restricted by geographic barriers (i.e., large rivers), and this region
107 encompassed a large portion of each species range, as well as the primary area of our field
108 surveys used to validate our models. This watershed delineation also encompasses a number of
109 similar adjacent urban areas (e.g., Lexington, KY, Indianapolis, IN, and Huntington, WV;
110 Appendix B) that could be examined in future studies.
111 We chose 11 initial environmental variables that others have shown to be important to
112 amphibians (e.g., Herrmann et al., 2005; Weyrauch & Grubb Jr, 2004) and had available data.
113 Geographic data layers were at a resolution of 30 m and variables extracted from these included
114 land cover, elevation, canopy cover, distance from stream, and slope/aspect. We created three

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115 layers using a moving window analysis to compute a ratio of cells classified as urban or water
116 within moving windows of 150, 300, and 500 m. Lastly, we calculated the number of cells
117 representing wetland habitat, as defined by the National Wetlands Inventory layer (U.S. Fish and
118 Wildlife Service, 2011; retrieved from http://www.fws.gov/wetlands/), within 300 m and 2 km
119 moving windows. We tested collinearity of the 11 layers by extracting environmental
120 information from 1000 randomly selected points (selected using the Random Point Generation
121 Tool within the Hawth"s Analysis Tool software for ArcGIS) within the watershed. A
122 correlation matrix was generated and correlations with $r \geq 0.70$ were considered highly
123 correlated. When pairs of variables exceeded this threshold ($r \geq 0.70$), we chose one variable
124 from the pair that we considered the most biologically relevant. Eight variables (listed below
125 with the data source) were chosen for inclusion in the final distribution model. Land cover
126 (Homer et al., 2007; retrieved from http://www.mrlc.gov/) was reclassified into one of seven
127 classes as follows (original land cover classes given in parenthesis): „forest" (41, 42, and 43),
128 water/wetlands (11, 90, and 95), „low intensity developed" (21 and 22), „medium intensity
129 developed" (23), „high intensity developed" (24), „natural non-forest" (31, 52, and 71), and
130 „agriculture" (81 and 82). Canopy cover values were taken from the National Land Cover
131 Database (NLCD; Homer et al., 2007; retrieved from http://www.mrlc.gov/). Elevation data
132 were derived from NHDPlus (National Hydrography Dataset; U.S. Geological Survey, 2005;
133 retrieved from www.horizon-systems.com/nhdplus/). The distance from stream (DSL) was
134 measured as distance (in meters) from the nearest drainage. Drainages were delineated from the
135 flow accumulation layer (downloaded from NHDPlus; retrieved from www.horizon-
136 systems.com/nhdplus/) as areas that drained 100 ha or more. The synthetic slope/aspect layer
137 (TASL; which represents aspect ranging from -1 [NE] to 1 [SW] and is weighted based on the

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steepness of the slope), was derived from NHDPlus elevation data following Pierce et al. (2005). Finally, we selected three layers from the moving window analysis: Sum of urban, water and wetland cells within a 300-m buffer. Urban and water cells were designated from our reclassified NLCD database; whereas wetland cells were designated from the National Wetlands Inventory layer (U.S. Fish and Wildlife Service, 2011; retrieved from http://www.fws.gov/wetlands/). We chose the 300-m buffer because a 290-m buffer is recommended for maintaining wetland and riparian habitat, thus this distance has been suggested to be important for encompassing the core habitat requirement for many amphibian populations (Semlitsch & Bodie, 2003). Moving window, DSL, and TASL layer calculations were performed using ArcGIS (version 9.3; ESRI, Redlands, CA).

Maxent was run from the command line using the default settings with exception of background points. A total of 620 target-group background data points representing localities of amphibians in the designated watershed were used to develop an initial environmental envelope to represent the range of environmental conditions within the modeled region. This method is used to reduce the bias inherent in sample locality data (Phillips et al., 2009). The target-group background approach uses background data (known as pseudo-absences) to develop the models. The background points are chosen with the same biases as occurrence data and produce an unbiased estimate of the geographic distribution of each species. The resulting background data provide an equitable sample of the environmental conditions within the region modeled. Due to the use of background points, we disabled the "addsamplestobackground" function. Further, we disabled the "extrapolate," "Do MESS with analysis when projecting," and "Do clamping" functions. Following methods in Milanovich et al. (2010), a multiple threshold approach was used to designate a location as environmentally suitable for a species. Because Maxent produces

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161 a continuous probability of suitable habitat for each species, it is logistically unfeasible to present
162 each location as a probability of occupancy; therefore, we converted the continuous suitability
163 surface [0–1 from Maxent to presence/absence (1/0)] using four model output thresholds applied
164 by Maxent; fixed cumulative value 10 (FC10), minimum training presence (MTP), 10 percentile
165 training presence (10% TP), and maximum training sensitivity plus specificity (MTSPS). Next,
166 we generated four species richness maps based on the accumulated binary modeled distributions
167 of each species using the four Maxent thresholds. This four-threshold approach makes our
168 results comparable to other studies that provide predictions based on strict environmental
169 distributions of species (i.e., thresholds that maximize the agreement between observed and
170 predicted distributions; Cramer, 2003). This approach allows us to present model predictions
171 that relax the assumption of strict environmental control on species" distributions, and provides a
172 range of scenarios that could influence the predicted suitable habitat at the species-level and
173 comprehensive species richness.
174 We used null models to test the significance of each species distribution model (Raes & ter
175 Steege, 2007). We generated five null data sets, each with 1000 sets of sample points that were
176 randomly drawn without replacement from the pool of 620 background points. We generated a
177 null data set with the number of random points per distribution equal to 20, 45, 75, 150 and 250
178 data points, which represented a range of presence points available to model each species.
179 Maxent was used to calculate the area under the curve (AUC) for the 1000 null data sets to create
180 an AUC frequency distribution. The calculated AUC for each species model was compared to
181 the 95th percentile AUC value of the null frequency distribution created from the representative
182 number of sample points (20, 45, 75, 150, or 250). A species model performs better than random

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183 (e.g., null model) and is considered significant if the calculated AUC is greater than the
184 corresponding 95th percentile AUC of the null-distribution (Raes & ter Steege, 2007).
185 *2.2. Examining differences in amphibian species richness*
186 *2.2.1. Amphibian surveys*
187 To evaluate the degree to which SDMs accurately predict amphibian richness, we conducted
188 amphibian surveys at 20 sites across the Cincinnati, OH metropolitan area following methods
189 proposed in Shaffer et al. (1994). Each site consisted of identified green space, spanned a range
190 of sizes (3–1758 ha; Appendix C), and were spread across a gradient of urbanization (e.g., within
191 and outside the designated Cincinnati, OH metropolitan area; Fig. 1; Appendix B). Surveys were
192 conducted March – June 2011 and each site was sampled three times with a 3–5 week period
193 between repeat samples. To standardize our search effort within each site, nine plots were
194 selected and surveyed at each site. We attempted to place three plots in each of three habitat
195 types associated with amphibian richness: Wetlands/ponds, streams, and terrestrial (forest or
196 non-forest natural [e.g., grassland/prairie]) habitat. In the event that three each of
197 wetlands/ponds or streams were not available, we increased the number of terrestrial plots to
198 reach nine total plots for each site. If more than three of an aquatic habitat type were available
199 (e.g., wetlands/ponds or streams) we sampled plots with the most suitable amphibian habitat (i.e.,
200 forested wetlands/ponds or streams). We employed three survey techniques at each site: area
201 sensitive dip-net surveys for pond and/or ephemeral wetland habitats, area sensitive terrestrial
202 (searching cover objects) surveys, and leaf litter bags and dip-netting area sensitive surveys for
203 headwater streams following Chalmers & Droege (2002) and Waldron et al. (2003). At each
204 wetland/pond, we dip-netted several 1 m^2 areas adjacent to the bank using a 40.6 X 40.6 cm (3.2
205 mm mesh size) net. Plots were placed every 10 m until the entire perimeter of the body of water

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206 was sampled. For terrestrial plots, we surveyed five, 5 m^2 sub-plots at each site. Plots were
207 chosen by placing a single 5 m^2 plot within the center of each habitat type occurring on the site
208 (forest or non-forest natural [prairie or grassland habitats]) and an additional 5 m^2 plot in each
209 cardinal direction 25 m from the center plot. Within each sub-plot, all cover objects and existing
210 leaf litter were searched. Within each stream, five 1 m^2 plots were created every 10 m starting
211 approximately 20 m from the confluence of a larger stream, road, or trail and continuing
212 upstream. Within each plot, we placed one leaf litter bag (55 x 25 cm made with 1.3 cm^2 mesh)
213 filled with deciduous leaf litter from the surrounding forest in a wetted portion of the stream
214 channel with a large rock on top to prevent dislodging. We followed the methods of
215 Nowakowski & Maerz (2009) and Peterman, Truslow, & Samuel (2008) to remove contents
216 from each bag; we thoroughly dip-netted the 1 m^2 plot by dislodging substrate parallel to the
217 stream bank across the width of the stream. In addition to our structured surveys, any animals
218 encountered (seen or heard) while travelling between habitat types or plots within a site were
219 noted. All animals captured or heard (frog calls) were identified to species.

220 *2.2.2. Validation of amphibian surveys and richness comparisons*

221 We examined how well predicted species richness matched observed species richness by
222 comparing the maximum species richness value from each site between observed and predicted
223 (from each Maxent threshold) values derived from our maps. We compared differences between
224 species richness within "urban" and "non-urban" sites by categorizing a site as "urban" if $\geq$ 50%
225 of the adjacent landscape within a 2-km buffer of the centroid of the site had urban cells (NLCD
226 categories 21 – 24) based on our NLCD 2001 reclassified maps. To test for differences between
227 predicted species richness from each model threshold, we used paired *t*-tests to compare
228 observed and predicted (from each threshold) species richness values from each site surveyed.

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229 To examine if we adequately sampled species richness, we estimated species richness using
230 EstimateS version 8.2 (Colwell, 2009) for six sites that had been surveyed on at least 10
231 occasions. Additional survey data were derived from reports summarized in Davis, Krusling,
232 and Ferner (1998). These reports summarized data from county-park-level surveys that took
233 place between 1988 and 1995, surveyed a similar level of habitat heterogeneity and utilized
234 similar survey methods to our study, and lasted approximately one year. We chose to make
235 comparisons between species richness at this group of sites using the Chao 2 estimator because
236 this algorithm produces species accumulation curves that approach maximum values with few
237 samples (i.e., the estimator was the least sensitive to undersampling; Colwell & Coddington,
238 1994). The Chao 2 algorithm inflates the observed species richness by a factor derived from the
239 number of species observed only once or twice within a total sample. The estimator is calculated
240 as $S_{Chao2} = S_{obs} + Q^2_1 / 2Q_2$, where S_{obs} is the observed species richness and Q_1 and Q_2 are the
241 number of species detected only once or twice per site, respectively. Thus, this estimator
242 accounts for the fact that species differ in detectability and uses the relative frequencies of
243 species that are rarely detected to estimate the number of taxa that are present but not detected
244 (Chao, 1987; Colwell & Coddington, 1994). For each additional site we calculated coverage
245 (number of species observed/number of species estimated X 100), exclusive species (percentage
246 of species only observed in a given site or category), and completeness of a sample (species
247 observed as a percentage of the total number of species expected in the site; Gardner et al.,
248 2007). This approach estimated the amphibian species richness at sites visited $\geq$ 10 times and
249 provided an estimate of the number of sampling periods needed to accurately capture the
250 estimated richness value. These values were then compared to the observed amphibian species

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richness values derived from the current surveys and the surveys of Davis, Krusling, and Ferner (1998).

2.2.3. *Identifying factors that influence observed species richness*

We used a general linear model and an information theoretic approach to examine which environmental factors were most predictive of observed amphibian species richness. This model is specific to existing green spaces, unlike the species distribution models, which provide a more general estimate of species richness patterns. As a result, the environmental variables used in this analysis are similar to those in the distribution model; however, some differences do exist. Observed species richness was the dependent variable. Independent variables included the percent of urban land within a 2-km buffer of the centroid of each site (square root transformed), the number of wetlands within a 2-km buffer of the centroid of each site (log transformed), percent forest within each site (square root transformed), and percent non-forest natural habitat (agriculture and non-forested natural habitat) within each site (square root transformed) as continuous predictor variables. Using the same variables as above, we evaluated the importance of landscape variables on observed species richness using Akaike Information Criterion (Burnham & Anderson, 2002). These variables were derived from our reclassified NLCD 2001 land-cover layer and a National Wetlands Inventory layer (U.S. Fish and Wildlife Service, 2011; http://www.fws.gov/wetlands/) and were chosen out of an initial set of 11 independent variables that we thought would have an influence on amphibian species richness (Table 3). We generated a correlation matrix of all 11 variables and chose four variables that were not correlated with each other ($r \leq 0.70$). Variables were transformed to meet normality assumptions (Sokal & Rohlf, 1995).

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273 Lastly, to determine the relationship between our observed species richness values and the
274 mean predicted species richness values across the urban gradient, we conducted an analysis of
275 covariance. Maxent predictions (predicted species richness) and field observations (observed
276 species richness) were factors in the analysis, percent urbanization within 2-km of each site
277 (square root transformed) was our covariate, and observed species richness and mean predicted
278 species richness values were used as our dependent variables. The significance of the interaction
279 term was evaluated to determine if the relationship between observed and predicted species
280 richness estimates differed across the urban treatments.
281
282 3. Results
283 The mean AUC for amphibian distribution models based on landscape variables was 0.83
284 (range = 0.70 – 0.96; median = 0.83; Appendix A) and the AUC for each species was better than
285 random (i.e., model AUC values exceeded the 95th percentile of the null AUC distributions).
286 Observed and predicted species richness varied across each site (Appendix C). Mean observed
287 species richness was triple in non-urban sites compared to the value in urban sites (mean species
288 richness values; overall = 3.2, urban = 1.9, non-urban = 6.2). There were significant differences
289 between observed and predicted species richness for each threshold (Table 1; all *t*-tests $P \leq$
290 0.001). Overall, mean differences between observed and predicted species richness within each
291 threshold ranged from 253 to 558% (mean range = 8 to 18 species). These differences increased
292 when "non-urban" sites were isolated from "urban" sites (mean range = 343 to 619%; mean
293 range = 8 to 17 species) and decreased when sites sampled greater than 10 times were isolated
294 from sites sampled three times (mean range = 14 to 98%; mean range = 2 to 11 species; see
295 "Additional" column in Table 1).

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296 The MTSPS threshold, our most conservative threshold, had the lowest levels of over-
297 prediction between observed and predicted species richness in both urban and non-urban sites;
298 whereas the MTP threshold had the greatest levels of over-prediction. The species accumulation
299 curves (Fig. 2) for all green spaces sampled > 3 times, except Farbach-Werner Nature Preserve,
300 suggest these sites had been (more or less) exhaustively sampled (i.e., the species accumulation
301 curve reaches a plateau). There was no site for which the final estimate of species richness from
302 the species accumulation curves was achieved by the third sample (based on, for example, Chao
303 2 richness estimator; Fig. 2; Table 2).
304 Observed amphibian species richness varied depending on the number of wetlands and
305 percent of urbanization within a 2-km buffer around the centroid of each site (Tables 3 and 4).
306 Percent of forest or non-forest natural habitat within each site did not predict observed species
307 richness (Table 3). The best model (based on lowest AICc and highest w_i) for observed species
308 richness also includes number of wetlands and percent of urbanization within a 2-km buffer
309 around the centroid of each site (Table 4). Percent of forest or non-forest natural habitat within
310 each site did not predict observed species richness (Table 3). Despite the significant
311 discrepancies in absolute richness values between the observed and predicted species richness
312 values, there was a significant trend of decreasing richness with increasing urban habitat ($P <$
313 0.001 for both observed and mean predicted richness). Further, the slopes of these two trends
314 did not differ ($P = 0.913$; Appendix D) between modeled and observed values and the
315 relationships between observed and predicted species richness was significantly correlated to the
316 level of adjacent urbanization (Appendix D).
317
318 4. Discussion and Conclusions

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319 The use of SDMs built with high-resolution landscape variables and real-time amphibian
320 surveys allows us to assess whether this technique is a valid method to identify areas of higher
321 and lower species richness across urban and non-urban green space. Our approach of modeling
322 followed by field surveys revealed differences between observed and predicted species richness
323 across urban and non-urban green space (Table 1). These differences can be interpreted as (1)
324 sites were undersampled, thus our observed measure of species richness could be inaccurate, or
325 (2) the SDMs are over representing actual species distributions – which accumulate to unrealistic
326 predictions of species richness across the landscape. Our results suggest the differences are
327 likely due to both factors. First, species accumulation curves derived from the Chao 2 estimator
328 based on six sites that were extensively surveyed (i.e., $\geq$ 10 sampling events over approximately
329 one year), indicated three sampling periods was not a sufficient effort to capture species richness.
330 In each of these six sites, species accumulation curves did not plateau until between six to 10
331 sampling periods (Fig. 2). We conducted an intensive single-season survey that reasonably
332 mimics the effort many agencies would implement to understand factors driving species
333 distributions of amphibians. We know from studies with a more intensive sampling effort that
334 this is not sufficient to capture all species, because often species are unavailable for capture for
335 entire breeding years (e.g., Barrett & Guyer, 2008).
336 However, undersampling alone did not explain all of the differences between observed and
337 predicted species richness. Chao 2 species richness estimates showed SDMs still over-predicted
338 species richness even in sites considered to be adequately sampled (Table 2). For that subset of
339 sites, we have reasonable confidence (empirically demonstrated via richness estimator) that all
340 differences between sampled and predicted species richness are not due to undersampling alone.
341 This suggests the differences are at least partially a consequence of SDM predictions.

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342 We propose the SDM influence on over-prediction is the result of the inherent nature of
343 SDMs. By solely using environmental variables to construct predictions of a species‟ suitable
344 habitat, SDMs fail to incorporate biological or geographic influence on species distributions
345 (e.g., Guisan & Thuiller, 2005; Heikkinen et al., 2006; Luoto et al., 2005), also referred to as
346 realized versus fundamental niche (Rodda, Jarnevich & Reed, 2011; Soberón & Peterson, 2005).
347 This in turn can lead to an overestimation of species suitable habitat, as only areas of suitable
348 habitat, not realized distribution, are projected. This overestimation may increase when adding
349 multiple individual models to create species richness maps (Hortal & Lobo, 2006; Pineda &
350 Lobo, 2009; Thullen, Sartoris & Walton, 2002; Vasconcelos, Rodriguez & Hawkins, 2011), as
351 was the case in this study. Furthermore, although the use of target-group background data is
352 suggested to reduce sampling bias (Phillips et al., 2009), use of this approach could be a possible
353 source of overestimation. Models developed utilizing replicate samples and random background
354 data could result in a greater or lesser over-estimation of species richness values. Further
355 investigation into the influence of various modeling approaches to overestimation of species
356 richness is warranted.
357 Using multiple thresholds to develop a range of scenario‟s of species predicted suitable
358 habitat was a definite strength of this study (Fig. 1; Appendices B and C), and the differences
359 between predicted species richness for different model thresholds should not be ignored. The
360 importance of presenting a range of model thresholds has been supported in other studies (Liu et
361 al., 2005; Milanovich et al., 2010; Pineda & Lobo, 2009; Thullen et al., 2002). We suggest
362 presenting a range of thresholds allowed the geographic range of suitable habitat for each species
363 to be both larger and smaller than realized ranges. This may account for some of the geographic
364 and biotic influences on species ranges; however, as our results indicate, threshold adjustment

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365 may be a necessary, but not entirely sufficient means for generating more precise SDMs.

366 Despite the over-prediction of our models and differences between model thresholds, we
367 argue this technique is a useful tool for management of green space by identifying areas suitable
368 for amphibians in urban ecosystems. Specifically, although the model richness estimate was
369 high, there was no difference between the trend in observed richness and modeled richness
370 across the urban treatments. Species distribution models are increasingly used to identify
371 potential areas for conservation and management of biodiversity (Lawler, Wiersma &
372 Huettmann, 2011; Loiselle et al., 2003). In this study, SDMs identified areas of higher
373 suitability, as green space with higher observed species richness generally had higher predicted
374 species richness (Appendix B and C). For example, the Cincinnati Nature Center and
375 Richardson Nature Preserve (the non-urban sites with the highest observed species richness) and
376 Farbach-Werner Nature Preserve and Triple Creek Park (the urban sites with the highest
377 observed species richness) each had some of the highest predicted species richness within each
378 threshold (Appendix C). One commonality between these sites is each site had a high number of
379 adjacent wetlands (within 2-km of the centroid), and therefore, a high number of aquatic habitats
380 surveyed (Appendix C). With respect to management of existing or development of new urban
381 green spaces, we found the number of wetlands adjacent to each site was a significant predictor
382 of observed amphibian richness (Tables 3 and 4). Our predictive models support the importance
383 of wetlands in urban green space to amphibian species richness, as the two urban sites with
384 surveyed wetlands (Farbach-Werner N.P. and Triple Creek Park) had the highest predicted
385 species richness of all urban sites. This highlights the importance of prioritizing the
386 maintenance, restoration, or mitigation of wetlands in urban green space. Furthermore, this is a
387 testament to the usefulness of SDMs for identifying areas of potential conservation importance,

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388 but emphasizes that models must be constructed with biologically relevant variables that
389 facilitate presence of a particular species or taxon (e.g., wetlands or forest for amphibians).
390 In conclusion, we tested whether species distribution models could be utilized to identify
391 areas of higher amphibian species richness in urban ecosystems. If one"s goal is to make rapid
392 decisions concerning the management or creation of green space or infrastructure based on
393 conserving areas of higher biodiversity, this method provides relative measures of richness. Our
394 results indicate substantial effort is needed to sufficiently survey and identify areas of suitable
395 habitat for amphibians in urban and non-urban green spaces. Field surveys often require
396 expertise, time, energy, and funds; whereas SDMs can be developed using existing, readily
397 available data (e.g., museum records) and can be conducted using a variety of freely available
398 programs (Elith et al., 2006). We offer two suggestions on how to improve SDM use if one"s
399 goal is to identify areas important to high biodiversity. First, the development of species-specific
400 models, particularly in cases where there may be a desire to increase the abundance of a
401 particular species of concern, could enhance predictive ability. These models could utilize key
402 (spatially explicit) variables important to the biology of a particular species. Second, we
403 recommend using SDM approaches that incorporate an estimation of probability of occurrence
404 and species detectability, where such data are available for species of interest. This method has
405 been successfully utilized and detailed in more recent studies (e.g., Franklin et al., 2009;
406 Newbold et al., 2010; Rota et al., 2011). Unfortunately, our data did not permit such analyses.
407 We suggest incorporating one or both of these approaches to strengthen the accuracy of SDMs in
408 predicting species richness.

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Table 1. Estimates of how much SDMs over-predicted species richness values in green spaces in and around Hamilton County, Ohio. Values in the table represent the percentage each threshold (MTP = minimum training presence threshold, MTSPS = maximum training sensitivity plus specificity threshold, 10% TP = 10 percentile training presence threshold, FC10 = fixed cumulative value 10 threshold) over-predicted species richness compared to surveyed values for all sites (overall), urban sites (sites with ≥ 50% urbanization within the 2-km buffer of centroid), non-urban sites (sites with ≤ 50% urbanization within 2-km buffer of centroid), and from eight sites where past species richness values (derived from county-park surveys) were added to our current species richness values derived from current surveys (labeled Additional).

Table 2. Summary species richness data for amphibians at sites sampled ≥ 10 times in the Hamilton County, Ohio area.

Table 3. Results from a general linear model investigating the factors that influenced the observed amphibian species richness within each site and associated parameter estimates and 95% confidence intervals (CI). Percentage data were square root transformed and number of wetlands within 2-km buffer was log transformed. Excluded correlated variables not chosen for analysis were as follows: size; standard deviation of land cover within parks and 2-km buffer of centroid; percent forest, non-forest natural, and urban cells within 2-km buffer of centroid, and number of aquatic habitats surveyed within each site.

Table 4. Best Akaike Information Criteria (AIC) models for observed species richness across all 20 green spaces. Shown are only the confidence set of candidate models (i.e., models with AIC w_i within 10% of highest model). Variables included were percent urbanization within 2-km buffer, number of wetlands within 2-km buffer, percent forest within site, and percent non-forest natural habitat within each site.

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Urbanization Category	MTP	FC10	10% TP	MTSPS
Overall (n = 20)	558	467	373	253
Non-urban (n = 6)	619	572	468	343
Urban (n = 14)	532	422	333	215
Additional (n = 8)	98	81	50	14

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Table 2. Summary species richness data for amphibians at sites sampled ≥ 10 times in the Hamilton County, Ohio area.

Site	S_{obs}[a]	Chao 2 est. richness[b]	95% lower CI[b]	95% upper CI[b]	% Coverage[c]	% Exclusive species[d]	% Completeness[e]
Miami	12	12	12	12	100	10	60
Shawnee	10	10	10	11	100	0	50
Farbach	5	10	6	46	53	0	25
Triple Creek	9	9	9	9.8	100	0	45
Richardson	14	14	14	20	98	5	70
Withrow	9	9	9	10	100	15	45

a Number of species observed.

b Number of species estimated to be present (and associated 95% confidence intervals) based on the Chao 2 estimator.

c Number of species observed as a percentage of the average estimated species richness (Chao 2).

d Number of species not found elsewhere as a percentage of the landscape total.

e Number of species observed as a percentage of the landscape total.

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Table 3. Results from a general linear model investigating the factors that influenced the observed amphibian species richness within each site and associated parameter estimates and 95% confidence intervals (CI). Percentage data were square root transformed and number of wetlands within 2-km buffer was log transformed. Excluded correlated variables not chosen for analysis were as follows: Size; standard deviation of land cover within parks and 2-km buffer of centroid; percent forest, non-forest natural, and urban cells within 2-km buffer of centroid, and number of aquatic habitats surveyed within each site.

Source	df	MS	F	*P*	Parameter Estimate	-95% CI	+95% CI
% urbanization within 2-km buffer	1	11.717	5.465	0.034	-5.66	-10.82	-0.50
No. wetlands within 2-km buffer	1	15.118	7.052	0.018	0.97	0.19	1.75
% forest within site	1	0.593	0.277	0.607	0.79	-2.40	3.98
% non-forest natural within site	1	0.111	0.052	0.823	0.52	-4.35	5.39
Error	15	2.144	–	–	–	–	–

Table 4. Best Akaike Information Criteria (AIC) models for observed species richness across all 20 green spaces. Shown are only the confidence set of candidate models (i.e., models with AIC w_i within 10% of highest model). Variables included were percent urbanization within 2-km buffer, number of wetlands within 2-km buffer, percent forest within site, and percent non-forest natural habitat within site.

Model	AICc	Δ AICc	w_i
No. Wetlands + % Urbanization	72.19	0.00	0.61
No. Wetlands + % Urbanization + % Forest	75.35	3.16	0.13
No. Wetlands + % Urbanization + % Non-forest	75.35	3.16	0.13
No. Wetlands + % Non-forest	76.26	4.07	0.08

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Figure 1. (a) Distribution of locality points (black dots) used to model species suitable habitat within the watershed boundary. The boundary is located across several states with Ohio and Kentucky (the states where the surveys were conducted) located in the northeast and southern border of the map; and (b) the three counties where each green space (site) was surveyed; metropolitan areas are outlined with a red line and surveyed green spaces are represented by filled green polygons. Surveys were conducted throughout various locations within each site. The four lower panels represent the predicted amphibian species richness maps within the three counties surveyed as derived from the maximum training sensitivity plus specificity threshold (MTSPS) (c), minimum training presence threshold (MTP) (d), 10 percentile training presence threshold (10% TP) (e), and fixed cumulative value 10 threshold (FC10) (f)

Figure 2. Sample-based species accumulation curves for the six heavily sampled sites in the Hamilton County, Ohio metropolitan area

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Appendix A. Area under curve (AUC) values.

Appendix B. Species richness maps.

Appendix C. Summary data.

Appendix D. Results from analysis of covariance.

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Appendix A. Area under curve (AUC) values for each species modeled.

Species	AUC	Sample size
Northern Cricket Frog (*Acris crepitans*)	0.776	89
Streamside Salamander (*Ambystoma barbouri*)	0.816	31
Jefferson Salamander (*Ambystoma jeffersonianum*)	0.808	48
Spotted Salamander (*Ambystoma maculatum*)	0.827	54
Marbled Salamander (*Ambystoma opacum*)	0.957	28
Small-mouthed Salamander (*Ambystoma texanum*)	0.932	20
American Toad (*Bufo americanus*)	0.738	121
Fowler"s Toad (*Bufo fowleri*)	0.782	23
Northern Dusky Salamander (*Desmognathus fuscus*)	0.831	20
Southern Two-lined Salamander (*Eurycea cirrigera*)	0.785	58
Long-tailed Salamander (*Eurycea longicauda*)	0.868	20
Cave Salamander (*Eurycea lucifuga*)	0.838	27
Spring Salamander (*Gyrinophilus porphyriticus*)	0.948	37
Cope"s Grey Tree Frog (*Hyla chrysoscelis*)	0.799	65
American Bullfrog (*Rana catesbeiana*)	0.701	45
Green Frog (*Rana clamitans*)	0.736	114
Northern Leopard Frog (*Rana pipiens*)	0.798	40
Wood Frog (*Rana sylvatica*)	0.887	23
Eastern Red-backed Salamander (*Plethodon cinereus*)	0.828	37
Northern Ravine Salamander (*Plethodon electromorphus*)	0.919	20

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Northern Slimy Salamander (*Plethodon glutinosus*)	0.901	21
Spring Peeper (*Pseudacris crucifer*)	0.778	99
Western Chorus Frog (*Pseudacris triseriata*)	0.883	24

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Appendix B. Depictions of: a) Cities of Indianapolis, Indiana (1), Cincinnati, Ohio (2) Lexington, Kentucky (3), and Huntington, West Virginia (4) located within the watershed boundary. Predicted amphibian species richness maps within the watershed boundary derived from the maximum training sensitivity plus specificity threshold (b), minimum training presence threshold (c), 10 percentile training presence threshold (d), and fixed cumulative value 10 threshold (e).

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Appendix C. Summary data including size (ha), species richness (SR; observed [Obs.] and predicted), and landscape variables (within 2-km buffer of the centroid of each site and within each site) for each site. MTP = minimum training presence threshold, MTSPS = maximum training sensitivity plus specificity threshold, 10% TP = 10 percentile training presence threshold, and FC10 = fixed cumulative value 10 threshold. FRS = Forested area, NFRS = Non-forested natural area, URB = urban area.

Site	Size (ha)	Obs. SR	SR threshold				% in 2-km buffer			% in site				
			FC10	MTP	MTSPS	10% TP	FRS	URB	NFRS	FRS	URB	NFRS	No. wetlands within buffer	No. aquatic habitats surveyed
Alms Park	38	1	17	21	13	17	26.7	64.5	6.6	81.4	16.2	2.4	6	0
Ault Park	89	2	20	23	14	20	23.4	57.9	7.6	76.4	21.4	2.3	6	2
Avon Woods	15	1	16	22	10	16	6.1	92.1	1.7	63.6	6.9	29.5	1	2
Bracken Woods	11	3	16	22	9	16	43.0	55.8	0.8	90.9	9.1	0.0	1	1
Burnet Woods	41	0	15	19	9	15	10.4	89.0	0.4	38.1	58.4	1.3	1	0
Cincinnati Nature Center	414	9	22	23	14	22	67.6	17.8	13.0	93.2	1.6	4.8	33	6
Devou Park	324	2	21	23	13	21	35.4	55.4	2.5	69.6	28.4	1.5	9	2
Farbach-Werner	9	5	14	17	7	14	16.0	83.6	0.3	72.7	27.3	0.0	24	3
French Park	115	2	20	23	13	20	25.6	72.8	1.4	83.3	15.7	0.8	6	2
Glenway Woods	8	2	18	21	10	18	26.2	73.1	0.1	98.7	1.3	0.0	2	0
Kennedy Park	3	1	11	16	7	11	19.3	79.4	1.4	34.4	65.6	0.0	2	1

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Koeing Park	5	0	12	14	7	12	7.7	91.6	0.2	0.0	100	0.0	4	0
Miami Forest	1758	9	22	23	15	22	53.4	7.9	35.8	54.4	6.9	36.0	13	4
Rapid Run Park	22	1	16	20	8	16	16.1	83.4	0.5	50.6	46.9	4.9	1	1
Richardson	159	5	21	23	14	21	60.1	21.5	17.1	71.0	7.7	21.3	24	4
Seymour	83	2	22	23	13	22	29.5	65.0	4.4	87.0	6.1	6.9	5	2
Shawnee Forest	613	6	21	23	14	21	40.7	4.2	45.2	45.2	1.8	43.5	11	5
Triple Creek Park	72	5	16	19	8	16	30.1	66.5	2.8	16.8	72.1	4.5	21	2
Withrow	109	3	21	23	14	21	45.6	32.9	5.2	94.1	2.2	3.7	4	2
Woodland Mounds	431	5	22	23	14	22	68.3	13.4	7.0	80.4	9.2	10.4	19	3

Appendix D. Results of ANCOVA and regression analyses to determine if species richness observations/estimates differed across the urban gradient. Species richness was the dependent variable, which was estimated either through field-observations (observed SR) or Maxent-predictions (predicted SR). Square root transformed percent urbanization within 2 km of site was the covariate in the analysis. The non-significant interaction term indicates that the relationship between species richness estimates and urbanization was not different. Estimates for each method differed significantly, and urbanization had a significant, negative effect on richness estimates. Linear equations for observed and predicted species richness and percent urbanization within 2-km are as follows: observed species richness, $r^2 = 0.627$, $P \leq 0.001$, Observed species richness = 9.597 – 0.896*% urbanization; predicted species richness, $r^2 = 0.491$, $P \leq 0.001$, Predicted species richness = 23.765 – 0.925*% urbanization.

Source	df	MS	F	*P*
Predicted SR	1	1949.510	521.920	≤ 0.0001
% urbanization within 2-km buffer	1	180.590	48.348	≤ 0.0001
Predicted SR * % urbanization within 2-km buffer	1	0.05	0.0121	0.913
% forest within site	36	3.740	–	–

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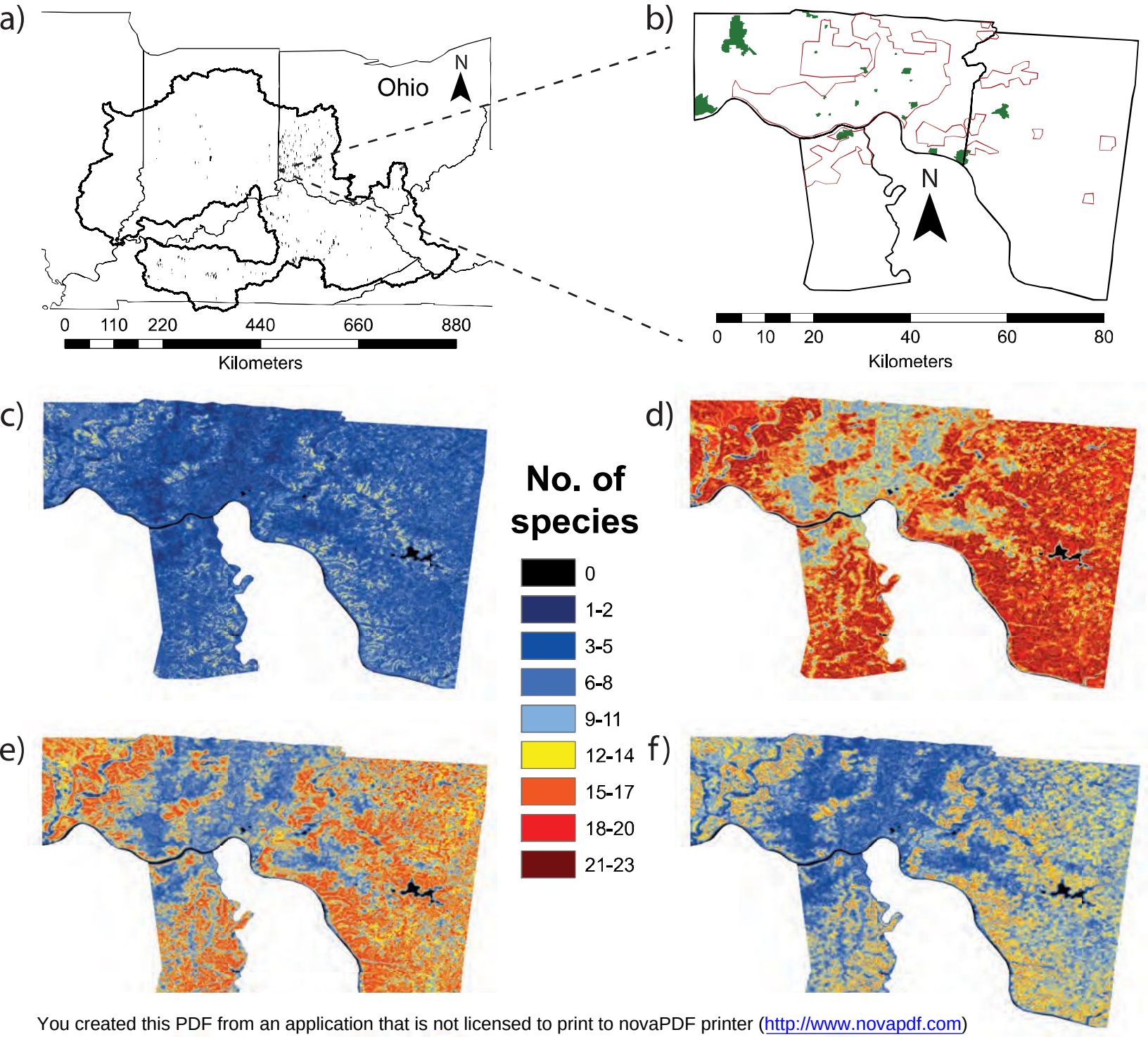
Figure 1.



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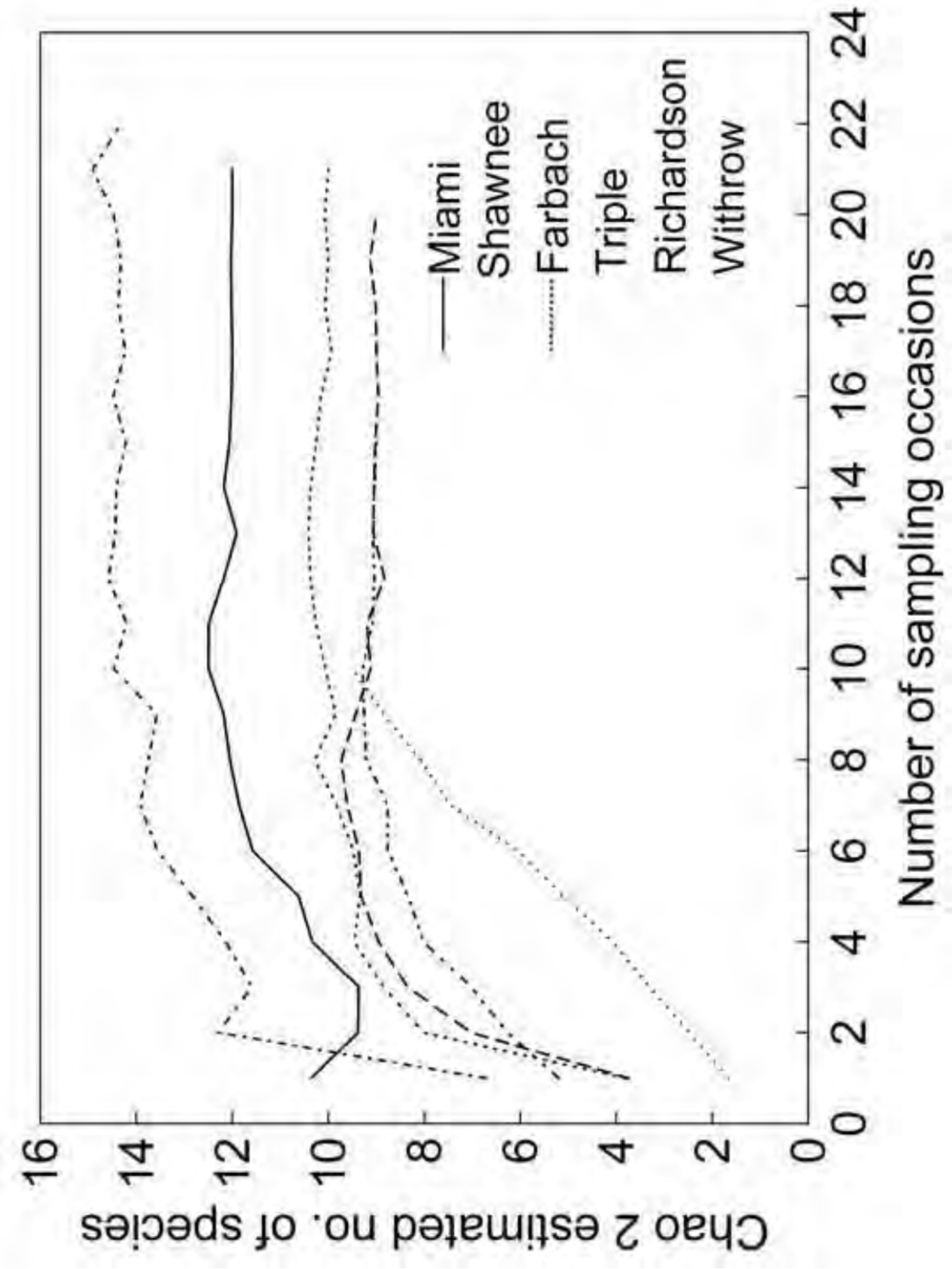

Figure 2.

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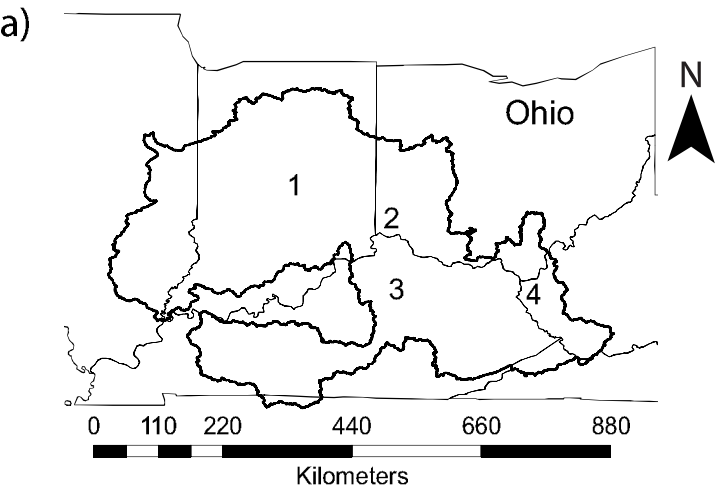
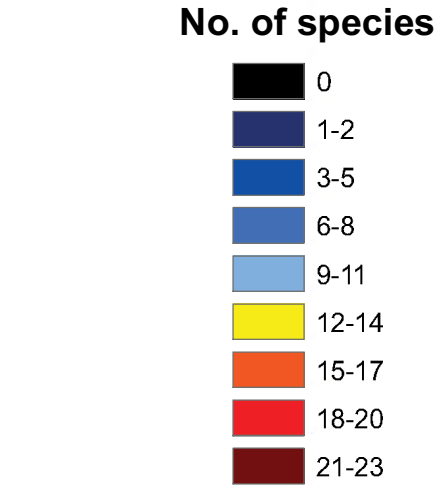
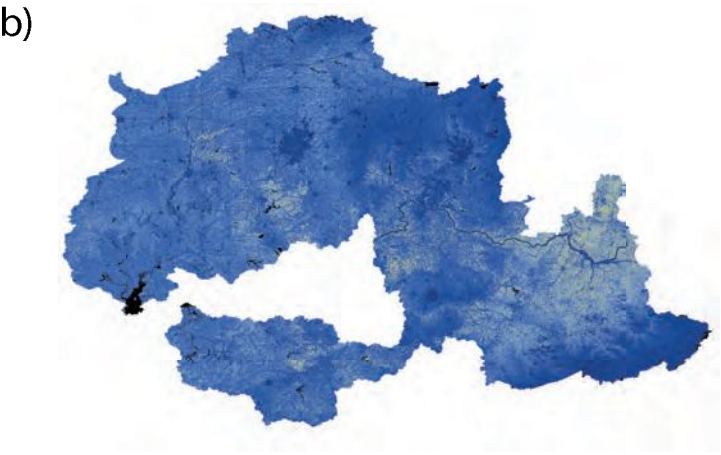
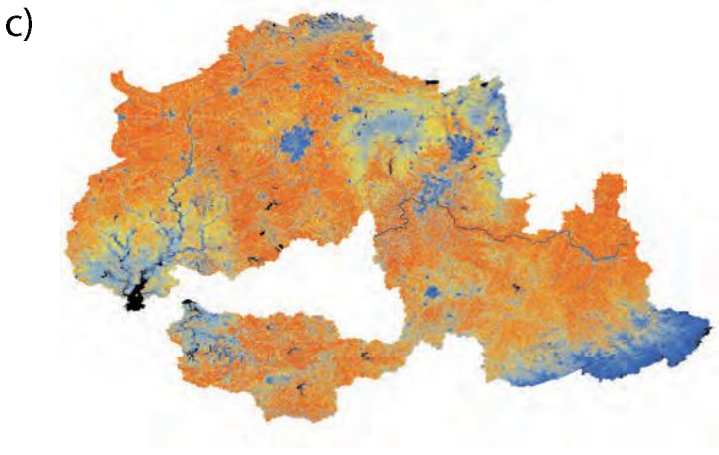
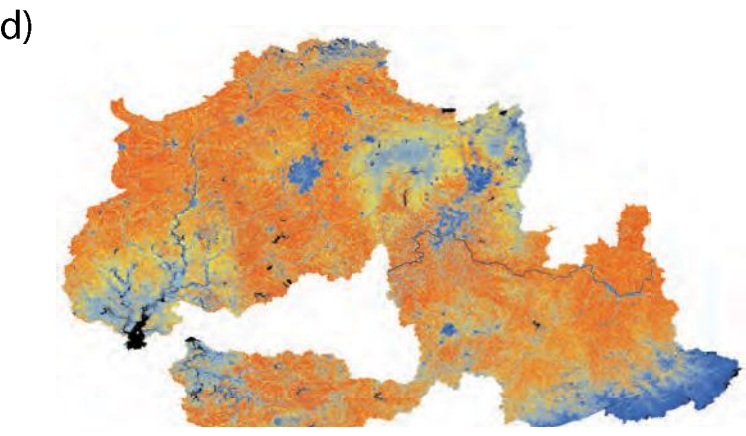
Appendix A.

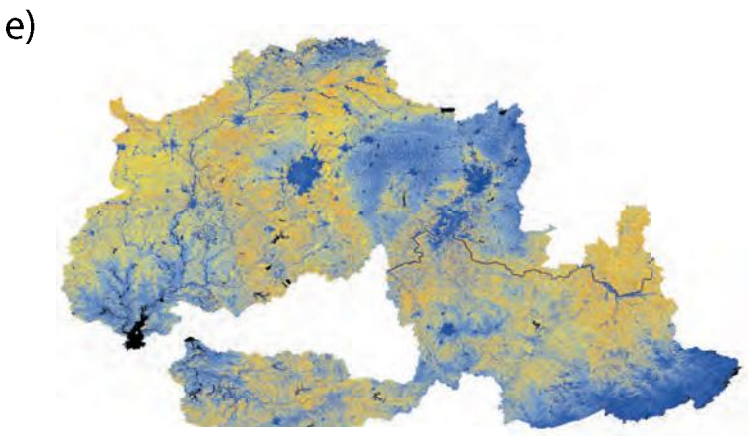












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