Landscape and Urban Planning 107 (2012) 409–418

Contents lists available at SciVerse ScienceDirect

Landscape and Urban Planning

journal homepage: www.elsevier.com/locate/landurbplan

Research Paper

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

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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.

ARTICLE INFO

Article history:
Received 13 February 2012
Received in revised form 9 July 2012
Accepted 18 July 2012
Available online 9 August 2012

Keywords:
Amphibians
Green infrastructure
Green spaces
Species distribution models
Urbanization
Validating

ABSTRACT

Urban green spaces are potentially important to biodiversity conservation because they could provide patches of high quality habitat or connectivity to nearby habitat. Presence-only species distribution models (SDMs) represent a potential tool for assessing the biodiversity value of urban green space; however, there is limited research to validate SDM results with field surveys to see if the predictions accurately represent observed species richness. We generated a range of SDMs using multiple suitability thresholds for 23 species of amphibians that occur in southwest, Ohio, USA. The distributions were overlaid to enumerate species richness. We surveyed 20 sites for amphibian species to evaluate model predictions. Our models over-predicted species richness relative to survey data. For example, we observed a mean pairwise difference of 14 species between models of species richness and observed values. Our results suggest either SDMs built with landscape variables we selected did not represent accurately amphibian richness, or the amphibian surveys did not detect all species present. Analyzing sites that had more than three sampling events suggests the explanation of inadequate sampling effort is only partially correct. Differences such as that between predicted and observed values of species richness is a challenge for land managers and conservation biologists that need a tool for modeling biodiversity. Species distribution models did project relative species richness well in urban and non-urban green space, which suggests this technique offers a spatially explicit way to identify more species rich areas and may help managers and conservation biologists manage systems with greater efficiency.

1. Introduction

Much of global population growth is occurring in urban areas (United Nations, 2004; Wu, Jenerett, Buyantuyev, & Redman, 2011) and 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.,

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0169-2046/$ – see front matter. Published by Elsevier B.V.
http://dx.doi.org/10.1016/j.landurbplan.2012.07.010

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 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, Richards-Hrdlicka, Storfer, and Skelly (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 Elith et al., 2006; Guisan & Thuiller, 2005; Guisan & Zimmermann, 2000). They are a group of approaches for identifying species distributions of undersampled species, predicting impacts 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, Wejnert, Hathaway, Rochester, & Fisher, 2009; Rota, Fletcher, Evans, & Hutto, 2011), independent and non-independent data validation (Araujo, Pearson, Thuiller, & Erhard, 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, Nelson, Jelinski, Nielsen, & Stenhouse, 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 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 of the distribution where empirical observations are lacking. For each species identified as occurring in Hamilton County, OH, species presence data were obtained for the period of 1997–2001 from HerpNET (http://www.herpnet.org), Global Biodiversity Information Facility (Lane, 2003; GBIF; http://www.gbif.org), and personal collections of herpetologists (Appendix A). All locality points were cross-referenced to each other and duplicate points were removed. Furthermore, localities that fell outside the current species range (identified by county-level distribution maps found in Lannoo, 2005) were not utilized to develop models. To maximize model quality, each model was built using at least 20 point locations for each species (Wisz et al., 2008).

We modeled the suitable habitat of each species across the National Hydrography Dataset Plus (U.S. Geological Survey, 2005; NHDPlus; retrieved from http://www.horizon-systems.com/nhdplus/) Region 05 Unit B watershed delineation. This delineation was necessary to encompass the environmental