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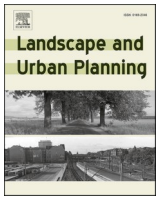

Research Paper

Managing yards for mammals: Mammal species richness peaks in the suburbs

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HIGHLIGHTS

- We deployed camera traps to assess backyard mammal diversity.
- Species richness peaked in in the suburbs and was lower in urban and rural yards.
- Urban gradient, not yard habitat, best predicted species richness and composition.

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ABSTRACT

Increased urbanization drives habitat loss, yet residential land-use represents significant habitat potential for mammals and could provide connectivity between patches of green spaces. Diverse mammal communities exist across urban gradients, but it is unclear how mammal community composition varies within residential lands. We conducted a camera trapping study in 36 residential yards across an urban gradient to assess the relative contributions of the degree of urbanization in the land-use context versus parcel habitat features, such as vegetation structure, on mammal community diversity and composition. We hypothesized that land-use context would more strongly influence mammal community metrics than parcel features, and that there would be species-specific differences in response. We detected 14 non-domesticated mammal species and found that species richness peaked in the suburbs and tapered off at the rural and urban ends of the gradient in accordance with patterns seen in other taxonomic groups, yet rarely quantified in mammals. Large-bodied interior forest species were associated with rural sites, urban-dwelling species were associated with urban sites, and suburban sites had an overlap of species types. Although composition of mammal species in residential yards appears to be strongly related to land-use context, which is often outside of residents' control, management of parcel habitat features such as retention of large mature trees may facilitate connectivity between patches of habitat across urbanizing landscapes. Informed residential yard management remains an important tool for urban wildlife management in an era of global change.

1. Introduction

The human population is growing and shifting away from rural areas and into cities and suburbs – by 2050, 66% of the world population will live in urban areas (UN, 2014). These trends drive worldwide patterns of habitat loss, fragmentation, and urban sprawl that result in changes to wildlife community composition (McKinney, 2002, 2006). Urban areas, particularly residential lands, are highly heterogenous at multiple scales, and habitat quality often varies based on land management on individual parcels (Daniels & Kirkpatrick, 2006; Goddard et al., 2010; Hansen et al., 2020). By understanding how patterns and processes spanning from the landscape to the parcel influence wildlife communities, households could manage residential lands so they can contribute to the connectivity of habitat fragments in human-dominated landscapes (Gallo et al., 2017; Lerman & Warren, 2011; Rudd, Vala, & Schaefer, 2002).

A multi-scalar approach is critical to understanding the factors influencing wildlife community composition (Aronson et al., 2016;

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Clergeau, Mennechez, et al., 2001; Hostetler, 1999; Jokimäki, 1999; Melles et al., 2003; Rega-Brodsky & Nilon, 2017; Shochat et al., 2006; Uchida et al., 2021). The impacts of connectivity and configuration of habitat patches vary in scale from 500 m for small to medium-sized mammals (e.g., racoons, *Procyon lotor*; Gallo et al., 2017) to multiple kilometers for large mammals (e.g., mountain lions, *Puma concolor*; Ordeñana et al., 2010). Within individual land parcels, high levels of heterogeneity represent the outcome of landscaping choices made by individual households (Cook et al., 2012), and result in different landscaping designs (e.g., lawn dominated versus tree dominated; Davies et al., 2009; Lerman & Warren, 2011; Pearse et al., 2018), as well as the presence of human-provided resources (e.g., bird feeders; Reed & Bonter, 2018) or human-caused disturbances (e.g., noise pollution; Warren et al., 2006). Further, body size or other species-specific traits influence how species interact with and use residential yards, which in turn is mediated by land-use configuration, habitat features, and resources/disturbances (Davies et al., 2009b; Kays & Parsons, 2014; Murray & Clair, 2017; Orros & Fellowes, 2015). By understanding the relative contributions of landscape and parcel features (Clergeau, Jokimäki, et al., 2001; Daniels & Kirkpatrick, 2006; Pennington & Blair, 2012), we can more fully understand the processes that filter and shape mammal communities across urbanizing landscapes.

Residential development varies continuously, thus, urban to rural gradients, rather than categorical classifications of urbanicity (urban, suburban, exurban), are often used to study shifts in biodiversity (e.g. Luck & Smallbone, 2010). Although many studies have described patterns of wildlife biodiversity across urban to rural gradients, most have targeted birds (e.g., Blair, 1996; Clergeau et al., 2001; Evans et al., 2018; Lepczyk et al., 2008; Shochat et al., 2010) and arthropods (e.g. Ahrné et al., 2009; Nagy et al., 2018; Sattler et al., 2010), and few have documented how mammals respond to urbanization (e.g., Crooks, 2002; Eötvös et al., 2018; Faeth et al., 2011; Mahan & O'Connell, 2005; McKinney, 2008; Nilon & Huckstep, 1998). For most taxonomic groups, as urbanization increases, species richness decreases monotonically (Ahrné et al., 2009; Crooks, 2002; McKinney, 2008). Species richness sometimes reaches a peak at intermediate levels of urbanization (McKinney, 2008; Shochat et al., 2010), particularly for plants (Deutschewitz et al., 2003; Godefroid & Koedam, 2003) and birds (Blair, 1996; Lepczyk et al., 2008; Marzluff & Rodewald, 2008), although this relationship could be scale-dependent (Pautasso, 2007). This intermediate peak is often driven by a mix of non-native and native plantings in suburban gardens and yards, which provides additional resources for a mix of native and generalist wildlife species, and adds to plant biodiversity (Donnelly & Marzluff, 2004; Faeth et al., 2011; Goddard et al., 2013).

Within a single land-use type, such as residential lands, households, through their land management and land-use, have the potential to influence mammalian species richness (Aronson et al., 2017; Beninde et al., 2015). Unlike landscape connectivity and configuration, individual householders exert direct control over an individual parcel and shape habitat structure either through active (e.g., planting trees) or inactive (e.g., leaving a wooded section of the parcel unmanaged) management practices. Once developed, residential lands provide relative stability and are rarely converted to new land-uses, although will sometimes be redeveloped or redesigned with ownership changes (Groffman et al., 2017; Murray & Clair, 2017). Therefore, households have the opportunity for long-term beneficial management practices for wildlife (Goddard et al., 2010, 2013; Lerman & Warren, 2011; Mumaw & Bekessy, 2017; Pardee & Philpott, 2014; Rudd, Vala, & Schaefer, 2002).

Most studies of mammal communities in urban areas have focused on non-residential urban green spaces, such as parks, golf courses, cemeteries, and forest patches (Gallo et al., 2017; Mahan & O'Connell, 2005). These studies often find that mammalian omnivorous generalists and *meso*-predators dominate these urban green spaces (Crooks, 2002; Crooks & Soule, 1999; Gallo et al., 2017; Gompper, 2002). Despite the dominance of residential land-use in urban areas (e.g., 21.8% – 26.8% of urban land-use in cities across the United Kingdom, which equates to 35–47% of total urban green space; Loram et al., 2007), few studies exist of mammal communities in residential yards (Hansen et al., 2020; Malpass et al., 2015; Van Helden et al., 2020). Although previous research has provided information on species-specific responses to resource subsidies in residential yards (Kays & Parsons, 2014; Murray & Clair, 2017) we lack information about mammal community biodiversity and composition in residential yards, which limits our ability to have a comprehensive picture of urban mammal communities in both public and private lands.

Residential yards often have human-provided resources that are either intentionally (e.g., bird feeders) or unintentionally (e.g., garbage) made available to wildlife (Goddard et al., 2013; Kays & Parsons, 2014; Lepczyk et al., 2004; Lerman & Warren, 2011; Loram et al., 2007; Murray & Clair, 2017). Kays and Parsons (2014) found that the mammal species typically observed in yards included those that occasionally took advantage of human-provided resources, such as eggs in chicken coops. Mammals, particularly squirrels, raccoons, skunks and foxes, often access these and other unintended wildlife resources, causing human-wildlife conflict (Gehrt et al., 2010; Ghert, 2004; Soulsbury & White, 2016).

Human-caused disturbances also influence mammalian use of residential yards. Light and noise pollution may drive some species away, while altering the time-of-day activity patterns of others (Ciach & Fröhlich, 2017; Grade & Sieving, 2016; Raap et al., 2015; Stone et al., 2009, 2015; Ware et al., 2015; Warren et al., 2006). Lawn-mowing and children playing in yards may disturb mammals, although the extent of impact from these activities is still understudied. Outdoor domestic pets, e.g., dogs (*Canus lupus familiarus*) and house cats (*Felis catus*), also serve as human-caused disturbances (Baker et al., 2005; Bonnington et al., 2013; Loss et al., 2013; Parsons et al., 2016; Silva-Rodríguez & Sieving, 2012). They may induce human-wildlife conflict (e.g., coyote, *Canis latrans*, depredating pets; Gompper, 2002), create opportunity for zoonotic disease transmission (Bradley & Altizer, 2007; Chalkowski et al., 2019; Daszak et al., 2000), and outdoor cats, in particular, are estimated to kill 6.2 – 22.3 billion mammals per year in the United States alone (Loss et al., 2013). Although impacts of cats and dogs on mammal populations are well-documented, it is unclear whether domestic pet presence in yards influences mammals at the community level.

Although previous studies have investigated parcel-specific land management (e.g. (Loram et al., 2007; Van Helden et al., 2020), human-provided resources and disturbances (e.g. (Kays & Parsons, 2014; Murray & Clair, 2017), as well as land-use context surrounding the yards (e.g. (Blair, 1996; Evans et al., 2015), we lack information about how all these features might interact to structure mammal communities. Using a multi-faceted approach, we conducted a camera trapping study to compare potential factors driving mammalian community structure across spatial scales to better understand which relevant scale(s) and features best inform management of mammalian diversity in human-dominated landscapes. We hypothesized that land-use context (summarized hereafter as "landscape") would strongly influence mammalian species richness and community composition (Aronson et al., 2016; Blair, 1996; Lepczyk et al., 2008; Nagy et al., 2018), while pertinent habitat features for most mammal species, such as tree size and shrub cover would influence species richness and community composition to a lesser extent (Kays & Parsons, 2014; Murray & Clair, 2017). Further, we hypothesized that species would respond differently to specific features depending on whether they are characterized as urban avoiders, utilizers, or dwellers (Aronson et al., 2016; Fischer et al., 2015; McKinney, 2008; Ordeñana et al., 2010). We also examined the extent to which the detection of outdoor domestic cats and dogs influenced mammalian species richness and community composition. We hypothesized that detection of cats would have a greater effect than detection of dogs since cats typically spend longer periods of time in yards unattended, including overnight, and are highly effective predators of small

mammals (Baker et al., 2005; Loss et al., 2013). Further, we hypothesized that yards in which cats were present would have significantly different mammalian species richness and community compositions from those without cats.

2. Methods

2.1. Study system

We introduced camera traps and surveyed mammals in 36 residential yards across an urban to rural gradient in western Massachusetts, United States (Fig. 1a). Sites were at least 250 m apart, and in most cases were

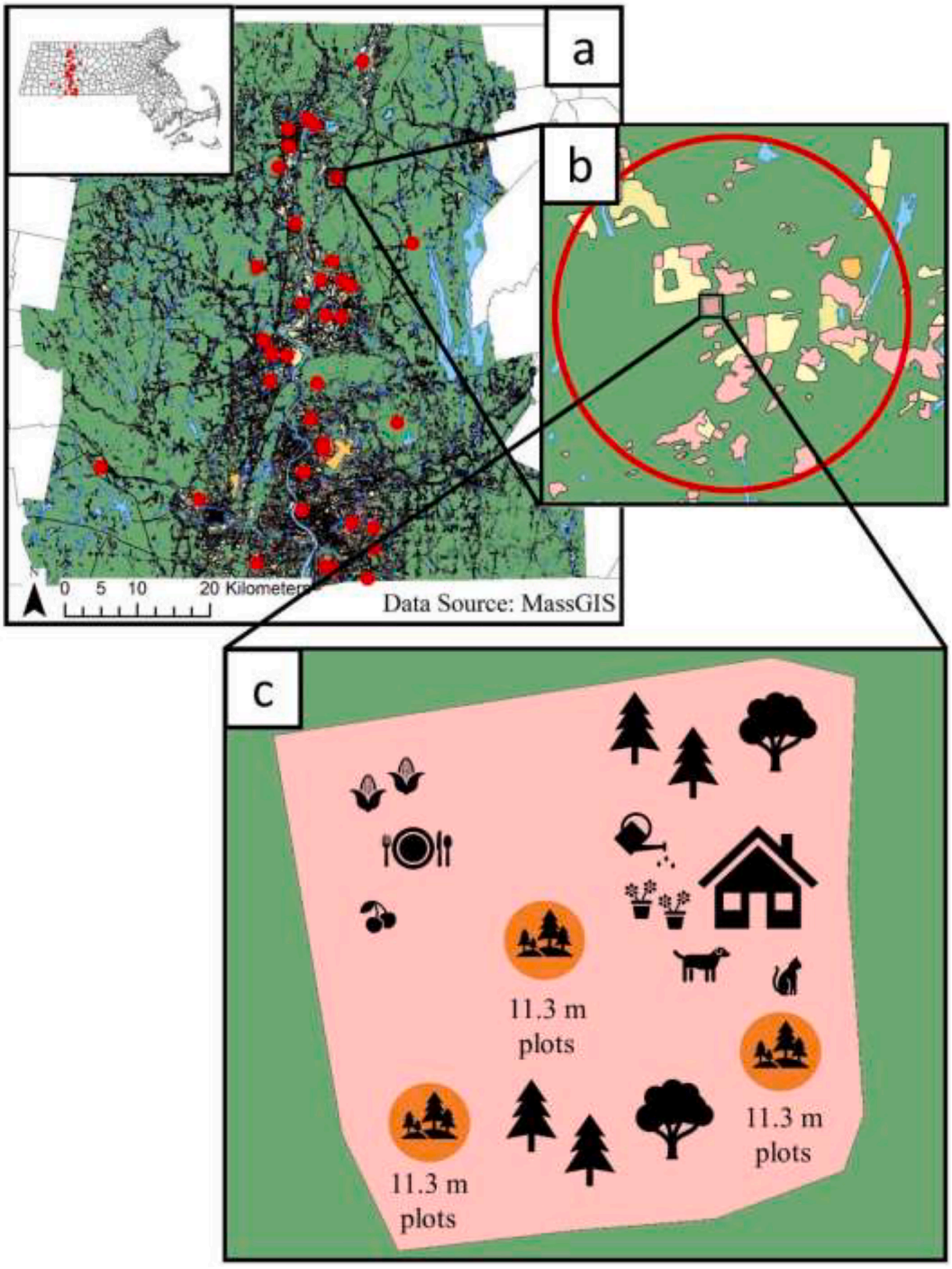

Fig. 1. (a) Cameras were deployed in residential yards across western Massachusetts, United States (Study Region), and data were collected from May – mid-September 2016–2018. Red circles indicate study yard locations. (b) Urban index was calculated based on land-use categories surrounding each residential yard. Pink areas indicate development, and we indicate one residential parcel with a black rectangle here. (c) Parcel features were measured with parcel-wide counts standardized by parcel-size in ha. These features included food resources, water resources, and gardens. Within each parcel, detailed vegetation and built-structural measures were assessed within 3 replicate 11.3 m plots (orange circles with vegetation symbol) within the parcel, with the mean of selected measures used for final analyses. Pictural symbols represent different yard features measured within individual parcels. Presented locations at the multiple scales were shifted from true locations to protect participant anonymity. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

over 1 km apart. The region has a temperate climate, and we conducted the study during late-spring and throughout the summer in 2016 – 2018. Survey efforts were in conjunction with an avian nesting ecology study on the effects of urbanization on house wrens (*Troglodytes aedon*) nesting in the same yards (Grade et al., 2021). Residential yards were situated throughout western Massachusetts and centered on Springfield, MA (Fig. 1a). These sites varied from low-density rural development to high-density suburban development. Land-use and land cover surrounding the sites included a mixture of residential housing, farmland, recreational areas, and mixed deciduous-coniferous temperate forest. Landscaping practices varied in residential yards. Many were dominated by traditional manicured turfgrass lawn, while others were dominated by either trees and shrubs, "low-mow" unmanaged grass, or gardens.

2.2. Camera trap surveys and photo processing

We deployed cameras twice per year (once in late spring to early summer, once mid- to late summer) for eight days rather than continuously due to equipment limitations. We used a stratified random sampling design to determine dates of deployment for even sampling across the urban to rural gradient. Each deployment consisted of two Bushnell Trophy Cam or Bushnell Trophy Cam HD cameras (Bushnell Corporation, Overland Park, KS) with infrared flash, three-picture burst, and 30 s delay between new picture burst triggers (Bondi et al., 2010; Erb et al., 2012). We treated the two cameras per each deployment as a single coordinated sampling effort (i.e., non-independent and combined) in the analysis. We mounted each camera on 1 m tall garden stakes and angled them slightly downward. In each yard and for each deployment, we situated one camera approximately 2.5 m from a house wren nest box, angled toward the nest box. The nest box was mounted on a 2.5 m metal garden stake, typically at the edge of the managed section of the backyard, less often in the side yard or front yard. We placed the second camera at least 10 m from the nest box camera at another location in the yard that was relatively open, had signs of mammal activity (e.g., scat), was adjacent to vegetation cover, and/or was near the edge of the yard boundary, and not aimed towards the nest box (O'Connell et al., 2010; Sollmann, 2018). Using two cameras per deployment allowed us to capture a larger area of the yard simultaneously, while also treating the nest box and its associated activity as a pseudo- "bait" station to assess potential nest predators.

We imported and processed all photos using CPW Photo Warehouse Software version 4.3.0.3 (Newkirk, 2016). At least two observers examined and identified each photo to species or the most accurate taxonomic level possible. A third observer with additional experience and training examined photos with conflicting identifications and determined the correct identification. We set independent detections, or the assumption that the detection of an animal of the same species was a new individual, at 60 min between detection events on the same camera (O'Connell et al., 2010; Sollmann, 2018). We exported and collated the data for import into R program version 3.6.2 for analysis (R Core Team, 2019). We logged a total effort of 119.91 camera-days. Metadata from each camera trapping deployment included site, deployment date, and effort per deployment (total camera trap-hours; hereafter "effort"). We accounted for effort in subsequent analyses.

2.3. Urban index

At each yard, we assessed land cover across the landscape (Fig. 1b) and condensed these into a single axis of urbanization following the methods of Padilla and Sutherland (2019). Defining landscape at a single buffer surrounding sites is difficult since there is wide variation in species' home ranges. Therefore we used a method that standardizes multi-dimensional gradients of human-influence and is robust across spatial scales from 500 m to 1.5 km, which is within the range of variation for home ranges of the species' surveyed (Bowler et al., 2017; DeGraaf & Yamasaki, 2001; Gallo et al., 2017). We chose this method over simplified measures of urbanization (e.g., building density, percent impervious surface) to reflect the real-world complexities of urbanization and land-use context surrounding the sites (Padilla & Sutherland, 2019; Rodewald et al., 2013).

For our urban index, we used a GIS layer (projection: Albers Equal Area; AEA, datum: North American Datum of 1983; NAD83) developed by Padilla and Sutherland (2019). They first extracted land cover data (30-m resolution, NLCD, https://www.mrlc.gov/national-land-cover-database-nlcd-2016) and computed spatially weighted averages for each pixel (land cover type) across the entire study area via Gaussian kernel at a 500 m smoothing distance, which resulted in a surface for each land cover type that ranged from 0 to 1 based on each pixel's 500 m radius neighbors. They also ran the analysis with a 1500 m smoothing distance and found no significant effect of varying scale on the index. They then assessed relationships and patterns between land cover types at each pixel via PCA, and condensed the land cover into an urban index (i.e., urban to rural gradient) based on the dominant land cover types across the main axis (PC1). In the PCA, PC1 explained 20.8% of the variance (Standard Deviation = 1.766). The resulting index is a point-by-point PCA-weighted average (i.e., raster of smoothed NLCD values multiplied by corresponding PC weight for each NLCD value). PC1 loaded positively on open (0.406), low (0.509), medium (0.493), and high (0.369) development classes, and negatively on evergreen (-0.292), deciduous (-0.155), and mixed forests (-0.261). The urban index scale was centered around 0 (i.e., suburban), and spanned from −1 (most rural) to +1 (most urban). The mean urban index value for our study locations was 0.19 (SD 0.19, range −0.22 to 0.47).

2.4. Parcel habitat measures

2.4.1. Plot surveys

We assessed vegetation, built structures, human resource subsidies, and human disturbances in each yard (Fig. 1c). We took detailed vegetative structural measures at three replicate 11.3 m radius plots at each parcel (Martin et al., 1996). Since vegetative structure is relatively stable in yards across a two-to-three-year period (Loram et al., 2007), we conducted the vegetation survey only once per yard. We centered the plot on the house wren nest box, the second in the geographic center of the backyard (but at least 20 m away from the nest box), and the third plot location was selected at random and separated from the other two plots by at least 50 m. In some cases, parcels were too small to encompass three plots that were 50 m apart, and in those cases, we took the third measurement out of the parcel boundary, but no farther than 50 m from the second plot. Following a protocol adapted from iTree version 5 (i-Tree software suite, 2018; Lerman et al., 2014), within each 11.3 m plot, we noted shrub and tree species names and abundance, measured shrub area (m^2) of each shrub and diameter at breast height (DBH) of each tree (Martin et al., 1996; Schmid-Holmes & Drickamer, 2001). Using visual estimation, we determined percent canopy cover and canopy height within the 11.3 m radius (Martin et al., 1996; Schmid-Holmes & Drickamer, 2001). Within a 5 m sub-radius of each plot center we estimated woody stem counts and percent ground cover of select categories (e.g., grass, brush, forbs). To analyze the replicate plot survey data, we aggregated variable measurements from each site by taking the mean of each variable.

2.4.2. Human resource subsidies and disturbances

We counted and listed parcel-wide human resource subsidies and disturbances, which included: built-structural components (e.g., houses, sheds), transportation features (e.g., roads, road types, vehicles), potential food and water resources (e.g., vegetable and flower gardens, bird feeders, unlidded trash cans, bird baths), and presence/absence of potential disturbance indicators (motion-activated lights, and children's toys; Kays & Parsons, 2014). Since human resource subsidies and human disturbances are more likely to vary over time, we conducted these surveys once per year in each yard while the study was active and used

the mean count of each variable, converted into density measures (per ha) for analysis (Lepczyk et al., 2004). We used our camera trap data to determine detection/non-detection of dogs and cats in each yard, assuming non-detection of outdoor dogs and cats at a yard when undetected by the camera traps (i.e., assumed naïve detectability; MacKenzie & Bailey, 2004).

2.5. Statistical analyses

2.5.1. Variable inclusion

Prior to our analysis, we conducted a variable inclusion procedure to reduce the number of variables. We examined correlation matrices of related numerical variables (e.g., ground cover variables) to assess collinearity (Zuur, et al., 2009). We considered two variables with correlation coefficients $> |0.5|$ collinear and included only one of the two collinear variables as covariates in future analyses (see Appendix A and Appendix A Table A.2). We chose which covariate to prioritize by reviewing literature on northeastern mammal habitat use (DeGraaf & Yamasaki, 2001). From this process, we selected a final set of 11 predictor variables plus their squared terms for inclusion as potential covariates in our analyses (Table 1).

2.5.2. Species richness

To assess biodiversity, we calculated estimated species richness for each site. Due to difficulties of visual identification, we combined all mice spp. and vole spp. (e.g., families Cricetidae, Dipodidae, and Muridae, representing nine possible species) into a mice/vole spp. category, and the two possible species of rabbits (*Sylvilagus floridanus* and *S. transitionalis*) into a rabbit spp. category. We did not include domestic cats or dogs in this analysis, although we considered them as potential human disturbances in later analyses. To estimate true species richness by site, we used raw species abundance data with the *iNEXT* function in the package *iNEXT* in program R (Hsieh et al., 2020), with individual-based abundance set as the data type, Hill number (q) as 0, endpoint sample size as double the reference sample size, and knots (K) as 40. We used the resulting estimated species richness by site in subsequent analyses.

We assessed the relationship between our predictor variables of interest (Table 1) and estimated species richness via maximum likelihood selection from a set of *a priori* chosen models (n = 47) and used corrected Akaike information criterion (AICc) for selection (Akaike, 1973; Burnham et al., 2011). Initial exploratory analyses indicated that relationships between most predictor variables and estimated species richness were non-linear (Eilers & Marx, 1996). Thus, we included the squared-terms of all predictor variables in models, although we also tested a linear-only urban index in a set of identical models (see Appendix A Table A.1 for full model set). We then used the *modavg* function in the R package *AICcmodavg* (Mezerolle, 2017) to generate model averaged estimates and 95% confidence intervals from the full *a priori* model set as a proxy for variable "significance" (Burnham et al., 2011; Mezerolle, 2017; Symonds & Moussalli, 2011). We generated plots using the R package *ggplot2* (Wickham, 2016).

Table 1
List of potential predictor variables selected for inclusion in the models. See Appendix A Table A.1 for full list of variables. Sites were located in western Massachusetts, United States, and data were collected from May – mid-September 2016–2018.

Variable	Units	Sampling extent
Urban Index	Index	1-km buffer
Number of vegetable gardens / acre	Count/ha	Parcel
Number of fruit trees / acre	Count/ha	Parcel
Number of bird feeders / acre	Count/ha	Parcel
Number of pet food receptacles / acre	Count/ha	Parcel
% log cover (on ground)	Percentage	Parcel (mean of 3 plots)
Canopy height	M	Parcel (mean of 3 plots)
Shrub cover	m^2	Parcel (mean of 3 plots)
Number of trees in plot	Count	Parcel (mean of 3 plots)
Mean tree diameter at breast height (DBH)	Cm	Parcel (mean of 3 plots)
Proportion hardwood versus softwood trees	Percentage	Parcel (mean of 3 plots)

2.5.3. Variance partitioning analyses

We quantitatively assessed the species-specific relative contribution of the urban index using the *Hmsc* function in the R package *Hmsc* (Tikhonov et al., 2019). The function uses Bayesian Hierarchical Modelling of Species Communities (HMSC) and Gibbs Markov chain Monte Carlo (MCMC) sampling to partition the variance and estimate the proportion of variance explained for each species by urban index and the remaining residual variance explained by other effects (i.e., random effects of site; Ovaskainen et al., 2017).

2.5.4. Community similarity

To test whether mammal communities varied along the urban to rural gradient, we conducted Jaccard's similarity index analysis. Jaccard's similarity considers presence-absence and not abundance, to estimate community similarity among sites (Jaccard, 1901). We calculated Jaccard's index with non-metric multidimensional scaling (nMDS) in the function *metaMDS* in R package *vegan* (Oksanen et al., 2019). We specified two dimensions and ran 100 iterations (Oksanen et al., 2019; Shafii et al., 2013). Based on the results of estimated species richness, we tested for overlap between Jaccard's community similarity and urban index using distance matrices to run a permutational multivariate analysis of variance (MANOVA). We used the function *adonis* in the R package *vegan* (Oksanen et al., 2019), specified 200 permutations, and fit a linear model with Jaccard's similarity value as the response and urban index as the predictor. For visualization, we overlaid species with the urban index in two-dimensional ordinal space using the Jaccard's nMDS results and the functions *ordisurf* and *orditorp* in the R package *vegan* (Oksanen et al., 2019). Significance was assessed at the $P < 0.05$ level.

2.5.5. Effects of domestic cats and dogs on mammal communities

To evaluate the effect of domestic pet detection on mammal communities across the urban to rural gradient, we split each site into either cats detected (i.e., at least one cat captured on cameras during all surveys at the site, although most sites in which cats were present had more than one individual cat) or cats not detected (i.e., no cats captured on cameras during all surveys at the site). We used a separate variable to indicate dogs detected versus dogs not detected based on the camera trap data. Collinearities between cat/dog detection and the urban to rural gradient could have confounded our results. Thus, we ran two generalized linear models that ruled out a significant relationship between urbanization and cat/dog detection/non-detection (cats, urban index: z = 1.72, $P = 0.0849$, urban index^2: z = -1.22, $P = 0.2238$, and dogs, urban index: z = 1.53, $P = 0.1265$, urban index^2: −1.81, 0.0701). We then tested for differences in species richness between sites with or without cats/dogs using generalized linear models (GLMs), setting categorical cat/dog detection/non-detection variables as the predictor variable and estimated species richness as the response variable. Finally, we used the categorical cat/dog detection/non-detection variables to assess the influence of cats and dogs on Jaccard's community similarity, using the same methods outlined above (*Community similarity*) for testing for the effects of urbanization.

We ranked the ratios of species detection when cats were also present to see whether a pattern existed between the predator–prey relationship of the mammalian species and what species were present in yards that also had cats. We calculated the ratio as:

$$\frac{\alpha_{smp}}{\alpha_{scp}} / \frac{\alpha_{sma}}{\alpha_{sca}} \quad (1)$$

where α_{smp} = proportion of sites in which the mammal species is present, α_{scp} = proportion of sites in which cats are present, α_{sma} = proportion of

sites in which the mammal species is absent, and α_{sca} = proportion of sites in which cats are absent. A higher ratio value indicates greater likelihood that a species is present when cats are also present. Since cats are known to depredate small to medium-sized mammals (Loss et al., 2013), we assessed cat detection versus other mammal species detection using chi-squared contingency tables.

3. Results

3.1. Relative effects of landscape and parcel features on species richness

In total, we identified 14 wild mammal species/species categories and two domestic species: cats and dogs (Table 2). Out of the 47 models testing the relationship between parcel- and landscape-level features and estimated species richness, one model was ΔAICc < 2 from the other models, and seven were ΔAICc < 4 (see Appendix A Table A.1). The top model predictors included only urban index (est. = 1.19, SE = 0.64, t = 1.86, P = 0.0719) and urban index^2 (est. = -4.59, SE = 1.81, t = -2.54, P = 0.016; Fig. 2), other models with ΔAICc < 4 included canopy height, logs, number of fruit trees, number of bird feeders, and their squared terms as predictors. When we model averaged all potential predictors from the full set of models, only urban index^2 had 95% confidence intervals that did not overlap zero (est. = -4.8, SE = 1.9, CI = -8.52, −1.08, see Table 3 for all predictor CIs). When we partitioned the variance of urban index and the remaining random effects for site for each mammal species/species categories, urban index generally explained a higher proportion of the variance than random effects, although the relative proportions differed by mammal species (Fig. 3). In the case of coyotes, random effects explained a higher proportion of the variance than urban index.

3.2. Effects of landscape and parcel features on community similarity

We found a significant relationship between Jaccard's similarity and urban index ($F_{1,33}$ = 3.30, R^2 = 0.94, P = 0.004975). A stress plot indicated good preservation of similarity/dissimilarity when the data were condensed into two-dimensional space (Oksanen et al., 2019; Shafii et al., 2013). Eastern gray squirrel (*Sciurus carolinensis*), opossum (*Didelphis virginiana*), and red squirrel (*Tamiasciurus hudsonicus*) occupied the more urban space in the ordination. Eastern chipmunk, (*Tamias striatus*), groundhog (*Marmota monax*), rabbits spp., and raccoon (*Procyon lotor*) occupied more of the suburban space, and black bear (*Ursus*

Table 2
Summary statistics of all mammal species that were detected by camera traps during the study, with common names and Latin names. Total detections at all sites for all years was under the assumption that a detection event >60 min apart was a new individual. Proportion of sites (out of 36) that species were present represents a raw detection proportion and does not account for false absences or total effort. Sites were located in western Massachusetts, United States, and data were collected from May – mid-September 2016 – 2018.

Species Common Name	Species Latin Name	Total Detections	Proportion of sites species were present
American red squirrel	*Tamiasciurus hudsonicus*	13	0.11
Black bear	*Ursus americanus*	5	0.11
Common raccoon	*Procyon lotor*	111	0.64
Coyote	*Canis latrans*	6	0.17
Domestic cat	*Felis catus*	223	0.72
Domestic dog	*Canis lupus familiaris*	196	0.36
Eastern chipmunk	*Tamias striatus*	1019	0.89
Eastern gray squirrel	*Sciurus carolinensis*	1248	0.94
Gray fox	*Urocyon cinereoargenteus*	19	0.31
Groundhog	*Marmota monax*	90	0.22
Mice/vole spp.	Families Cricetidae, Dipodidae, or Muridae	24	0.31
Rabbit spp.	*Sylvilagus floridanus* and *S. transitionalis*	223	0.78
Red fox	*Vulpes vulpes*	28	0.33
Striped Skunk	*Mephitis mephitis*	79	0.72
Virginia opossum	*Didelphis virginiana*	102	0.67
White-tailed deer	*Odocoileus virginianus*	19	0.22

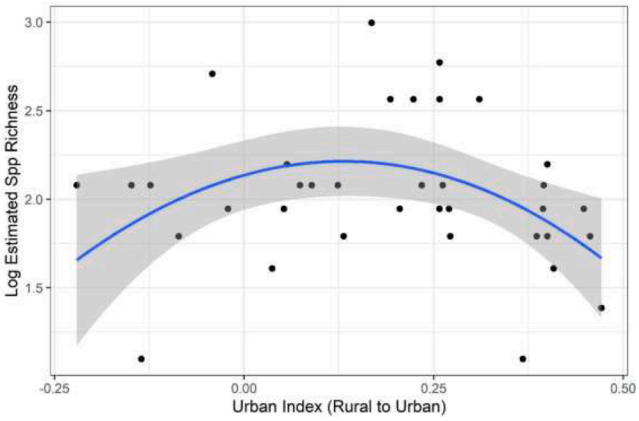

Fig. 2. Estimated species richness by urban index, from rural to urban. Sites were located in western Massachusetts, United States, and data were collected from May – mid-September 2016–2018.

Table 3
Model averaged parameters for all landscape- and parcel-level predictor variables, including beta = model-averaged beta estimates, SE = unconditional standard errors, and 95% CI = 95% unconditional confidence intervals of model averaged variables. Bolded variables do not overlap 0 at 95% confidence. Parameters rounded to nearest hundredth.

Predictor	Variable Name	beta	SE	95% CI
soft_hard	Urban index	−0.26	0.38	−0.9, 0.59
logs	Proportion of logs	−0.03	0.04	−0.1, 0.04
canopyht	Canopy height (m)	0.01	0.03	−0.05, 0.08
shrubcover	Proportion of shrub cover	0.02	0.03	−0.03, 0.07
numtrees	Number of trees	0.01	0.02	−0.03, 0.04
avgtreedbh	Average tree diameter at breast height (DBH, cm)	−0.01	0.03	−0.06, 0.05
perc_hwd	Proportion of hardwood versus softwood trees	0.99	1.46	−1.88, 3.86
veggiegardens_num	Number of vegetable gardens per ha	−0.01	0.02	−0.05, 0.03
fruittree_num	Number of fruit trees per ha	−0.01	0.00	−0.01, 0.00
soft_hard^2	**Squared term**	**−4.8**	**1.9**	**−8.52, −1.08**
logs^2	Squared term	0.00	0.00	−0.00, 0.00
canopyht^2	Squared term	0.00	0.00	−0.00, 0.00
shrubcover^2	Squared term	0.00	0.00	−0.00, 0.00
numtrees^2	Squared term	0.00	0.00	−0.00, 0.00
avgtreedbh^2	Squared term	0.00	0.00	−0.00, 0.00
perc_hwd^2	Squared term	−0.65	1.09	−2.78, 1.48
veggiegardens_num^2	Squared term	0.00	0.00	−0.00, 0.00
fruittree_num^2	Squared term	0.00	0.00	−0.00, 0.00

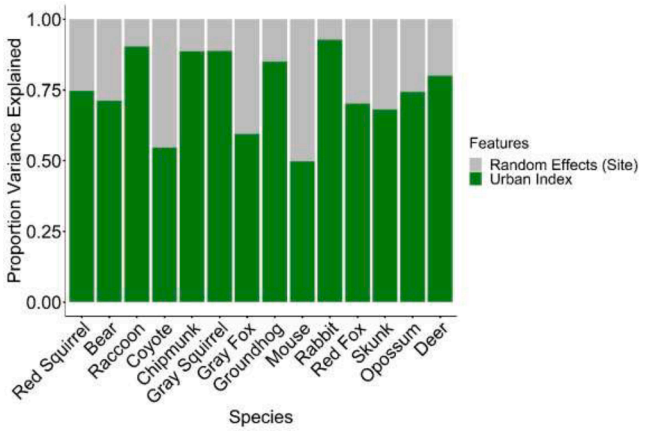

Fig. 3. Proportion of variance explained for species-specific occurrence at sites. Variance partitioned by: Random effects of site (gray) and urban index value (green). Sites were located in western Massachusetts, United States, and data were collected from May – mid-September 2016–2018. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

americanus), coyote (*Canis latrans*), gray fox (*Urocyon cinereoargenteus*), mice/voles spp., red fox (*Vulpes vulpes*), skunk (*Mephitis mephitis*), and white-tailed deer (*Odocoileus virginianus*) occupied the more rural space of the ordination (Fig. 4).

3.3. Effects of cats and dogs on mammal communities

Out of the 36 sites, we detected cat presence at 20 sites, dogs at 20 sites, and both cats and dogs at 12 sites. There was no significant relationship between estimated species richness and detection/non-detection of cats (t = 1.09, $P = 0.285$) or dogs (t = 0.99, $P = 0.329$). We documented no significant community dissimilarity at sites where we detected cats versus sites where we did not detect cats ($F_{1,35} = 1.16$, $P = 0.3035$). Generally, potential prey species of cats (e.g., eastern gray squirrel) were less likely to be detected in yards occupied with cats, while potential predator species (e.g., gray fox) were more likely to be present in yards occupied with cats (Table 4). This descriptive observation was only statistically significant for two mammal species via chi-square tests. We had no *a priori* prediction for bear association with cats, but they were significantly more likely to be present in yards in which cats were never detected ($\chi^2_{1,35} = 5.00$, $P = 0.025$), while red fox, a potential cat predator species, were significantly more likely to be present in yards occupied by cats ($\chi^2_{1,35} = 6.92$, $P = 0.009$).

4. Discussion

In this study we investigated the influence of urbanization on mammal community composition and diversity. We found support for our hypothesis that land-use context has a stronger influence on mammalian species richness and community composition than parcel habitat features. The best supported models included only urban index, and not features of habitat or anthropogenic resources and disturbances. Although we did not find evidence to support a species-specific response to habitat features based on a species' classification as urban avoiders, utilizers, or dwellers, we did find predictable species-specific differences in strength of response to urbanization (Fig. 3) and how species clustered across the urban gradient (Fig. 4). Finally, although presence of cats in yards was related to presence of some other mammal species, we found no compelling evidence of an overall impact on community

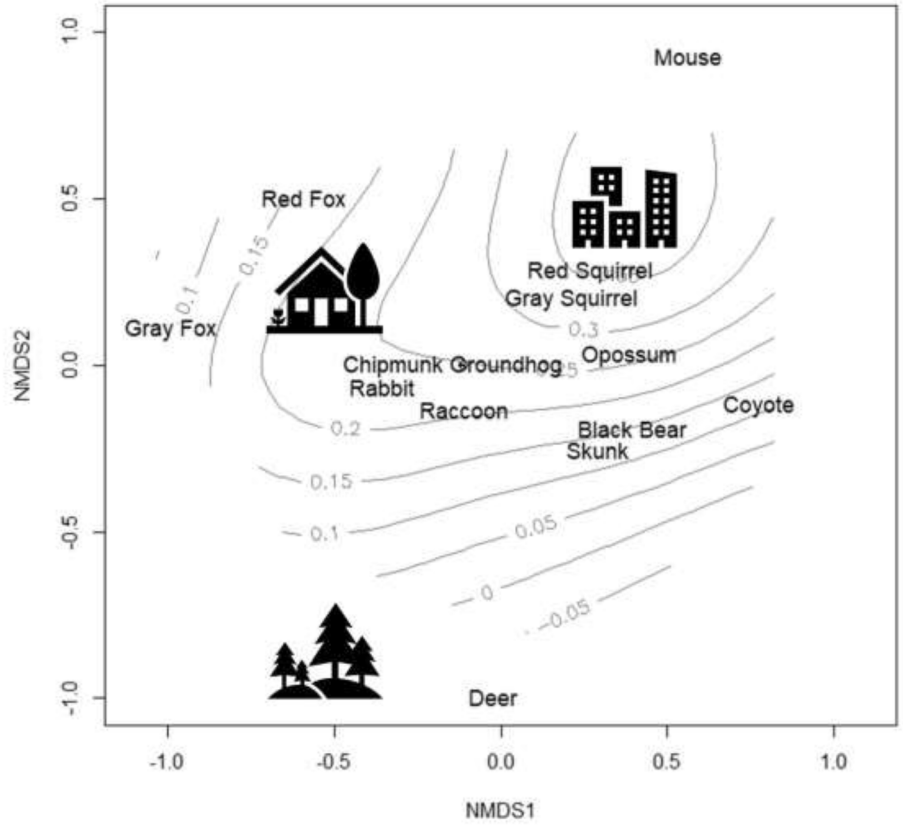

Fig. 4. Species disbursed across the urban gradient in two-dimensional ordinal space via NMDS. Isoclines indicate urban index values spanning from more urban (positive values) in the center and extending out into suburban and rural (negative values) on the edges. Sites were located in western Massachusetts, United States, and data were collected from May – mid-September 2016 – 2018.

Table 4
Mammal species surveyed, trophic relationship of species to cats (Baker et al., 2005; Loss et al., 2013), and ratio of species detection when cats are also detected versus non-detection when cats are also. Ratio = (proportion of sites that species is present / proportion of sites that cats are present) / (proportion of sites that species is absent/proportion of sites that cats are absent) Higher ratio values indicate greater likelihood that species is present when cat is also present. Species listed by shortened common names (see Table 2). Sites were located in western Massachusetts, United States, and data were collected from May – mid-September 2016 – 2018.

Species Common Name	Trophic relationship to cats	Ratio of species detection when cats were also detected
Bear	N/A	0.33
Red squirrel	Prey	1
Deer	N/A	1.67
Skunk	N/A	2.25
Raccoon	N/A	2.29
Mouse/vole	Prey	2.67
Gray squirrel	Prey	2.78
Chipmunk	Prey	3
Opossum	N/A	3
Rabbit	Prey	3.67
Coyote	Predator	5
Groundhog	N/A	7
Gray fox	Predator	10
Red fox	Predator	10

diversity and composition to presence of cats or dogs. Our findings do lend some initial support to our hypotheses that cats in yards may have a greater effect on mammal communities than dogs, we encourage future studies to investigate this question in more depth.

Urbanization generally, and not parcel-level management, was the major contributor to species richness and community composition. Mammal species richness peaked in the suburbs, a pattern that could represent the intersection of urban-dwelling and urban-avoiding species ranges. Taken together, these findings highlight the importance of land-use context as well as parcel-level management to mammal community structure. Decades of research indicates that land-use context features filter and shape wildlife community structure in human-dominated landscapes across many geographical regions and taxa (Aronson et al., 2016; Blair, 1996; Faeth et al., 2011; Gallo et al., 2017; Hostetler, 1999; McKinney, 2008; Rega-Brodsky & Nilon, 2017; Uchida et al., 2021), but there is little research that compares the relative contributions of management at different scales in residential areas.

In our study system, the most influential feature on mammal community structure included an urban index that had land-uses ranging from mostly forest in rural settings to mostly development in urban settings. Previous studies have highlighted that forest specialist mammals more likely use residential yard resources when they have access to large forest patches nearby (Kays & Parsons, 2014; Murray & Clair, 2017). In the northeastern United States (where our study took place), medium-to-large sized mammals, such as raccoons or coyotes, have home ranges that exceed the size of an average residential parcel (DeGraaf & Yamasaki, 2001). Forest specialists also require specific natural features, such as rocky outcroppings for dens for suitable habitation (DeGraaf & Yamasaki, 2001). Therefore, presence of a species in a western Massachusetts yard, especially when it requires intact forested habitat, may depend on landscape context, regardless of the yard's available resources.

The importance of landscape context is borne out by our finding that suburban yards had the highest level of predicted and measured species richness. Suburban yards in our study area are embedded within a heterogenous matrix of land-use that includes both wildland and more dense human development (Ehlers Smith et al., 2018; Goad et al., 2014). The more urban yards in our study area often had limited forest and agriculture nearby and surrounded by more residential and commercial land. In contrast, rural yards tend to be surrounded by other low-density residential developments, as well as larger tracts of forest and agricultural land. With this heterogeneity, suburbs in our study area have an intermediate combination of proximity to natural areas and high densities of anthropogenic resources, which may draw mammals occupying adjacent wildlands into the backyard (Kays & Parsons, 2014; Newsome et al., 2015).

Surprisingly, the more rural end of the gradient also exhibited a decline in species richness, perhaps by losing more urban-dwelling species such as opossum (Fig. 4). Available resources in rural areas are often limited in density since rural parcels are often separated by forest and agricultural patches (Kays & Parsons, 2014), which may preclude the need for mammals to disperse through or utilize the relatively "unfriendly" backyards at the rate seen in more developed areas. Communities were also significantly dissimilar based on degree of urbanization. Our results indicated that community composition varied within residential land-use across the urban to rural gradient. Predictably, large-bodied, wide-ranging, and interior forest species, such as gray fox, associated more with rural sites, while species such as eastern gray squirrel, were more urban-associated (Goad et al., 2014). This distribution of species indicates that while we documented few urban-dwelling species in rural areas and few urban-avoiding species in urban areas, both sets of species occasionally inhabit the suburbs. Though we did not directly quantify species interactions, this shift in community composition likely has implications for trophic dynamics, especially between predators and their prey, such as fox and rabbit (Fischer et al., 2012; Jokimäki et al., 2020; but see Eötvös et al., 2018). For example, our camera trap captured an instance of a red fox depredating a groundhog in a suburban yard (Appendix A Fig. A.1). Although we did not find a statistically significant effect of detection of cats (or dogs) on species richness and community dissimilarity, there was a higher mean species richness in yards occupied by cats. We also found a positive correlation between cat and red fox presence. Perhaps the mere presence of cats could attract predators, such as foxes (Kays et al., 2015; Ordeñana et al., 2010; Shochat et al., 2004). By design, our camera trap survey targeted medium to large-sized mammals (e.g., skunks and larger; Gallo et al., 2017; O'Connell et al., 2010), which cats rarely prey upon (Baker et al., 2005; Loss et al., 2013).

There are some key caveats for our study. Our study design prioritized maximizing the number of sites over increasing the amount of replicate surveys at each site (Erb et al., 2012). This sampling design allowed us better coverage across the gradient for estimated species richness at the expense of assessing abundance (Bowler et al., 2017; Ehlers Smith et al., 2018; Royle & Nichols, 2003; Sollmann, 2018), but consequently eliminated our ability to conduct occupancy analyses. Although our study included three years of data collection, we only focused our efforts during a single season (spring/summer). A continuous monitoring scheme across seasons would yield rich occupancy data of backyard mammals. Since we were unable to readily identify many small mammal species (e.g., mice and voles) via our camera traps, we suggest future research augment camera traps with live trapping in yards to better capture the influence of cat presence and other influences on small mammal communities and populations (Bondi et al., 2010; Elizondo & Loss, 2016). Although we were unable to obtain detailed population-level data, the community-level data we assessed gave us a previously unexplored insight into the multi-scale effects of residential development on mammalian biodiversity.

Our study demonstrated the conservation value of residential landscapes for mammals and highlights some possible management actions. Yards are part of the broader landscape and allow for greater connectivity between fragmented forest patches and more intact forested areas (Gallo et al., 2017; Goddard et al., 2010; Rudd, Vala, & Schaefer, 2002). Given the multi-scale effects of habitat use, management that benefits mammal biodiversity can be complex. In our study, the landscape context of residential lands, not the habitat structure of individual parcels, was the most important filter of mammalian species with regards to species composition and species richness. We suggest that management (or lack thereof) of surrounding residential lands, including zoning

regulations, land preservation and open-space, and implementation of conservation easements on farmlands and woodlands, all contribute more to mammalian community diversity than the landscaping decisions of individual households. Previous research has demonstrated that although land-use context and other landscape-level drivers of mammalian community composition are mostly out of householders' control, intentionally managing our yards for habitat, such as preserving and maintaining large mature trees (Rudd, Vala, & Schaefer, 2002), and mitigating disturbances such as outdoor cats (Loss et al. 2013), are additional ways to support mammalian populations in yards. These measures are valuable for conserving urban biodiversity, and they also enhance households' connections to the natural world.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.landurbplan.2021.104337.

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