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Adam D. Chupp[1,2,*], Amy M. Roder[2], Loretta L. Battaglia[1], and John F. Pagels[2]

Abstract - The primary objective of this study was to compare urban and peri-urban mammal assemblages and relate variation in these communities to local differences in vegetation. We surveyed 15 locations in both urban and peri-urban habitats ($n = 30$). Boundaries of our survey areas coincided with those of National Park Service (NPS) areas in central Virginia. Over a 14-month period, we used five trap-types to document species in three guilds. A total of 9 and 15 species were documented at urban and peri-urban locations, respectively. Top predators *Canis latrans* (Coyote) and *Felis rufus* (Bobcat) were undetected at urban sites, while mesopredators were consistently more abundant. The absence of four small prey species and reduced abundances of the most common native generalist, *Peromyscus leucopus* (White-footed Mouse), were also associated with urban locations. Multivariate analyses of relative abundance data indicated significantly dissimilar mammal communities in urban and peri-urban locations. Shrub cover was highest in peri-urban locations, while grass cover was highest in urban sites—a pattern that was only marginally significant due to greater variability among these sites. The exotic grass *Microstegium vimineum* (Japanese Stiltgrass) was present at several urban sites and contributed to the complex relationship between percent grass cover and the small-mammal assemblages that we surveyed. Our results suggest that disturbances that reduce the recruitment of shrubs and other native plants and promote the spread of invasive grasses may have severe consequences for small-mammal communities. In addition, culturally preserved areas within both survey sites (i.e., battlefields planted with fescue grasses) were inhospitable to most small-mammal species and wildlife in general. In many NPS areas, there is great opportunity for development of adaptive management strategies that integrate ongoing NPS efforts to control invasive plant species with the enhancement of wildlife habitat in both culturally and naturally preserved areas. There is an urgent need for the conservation of native habitat in NPS areas and non-park sites threatened by urbanization. The primary focus of these efforts should include the control of exotic species and mesopredators, facilitation of native shrub recruitment, and, in many of these areas, the ecological restoration of historic sites. NPS lands in urbanized areas offer unique conditions for wildlife management and abundant opportunities for conserving native communities.

Introduction

Of the 363 National Park Service (NPS) lands that reported visitation in 1985, 29.5% (107) were located in urban or suburban areas (NPS 1985). With population growth and increased exurban sprawl over the last 25 years, it is likely that many more NPS areas are now facing the effects of urbanization. In these fragmented and

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degraded landscapes, small parks often contain remnants of natural habitats that provide refugia and corridors that are vital to biodiversity conservation (Falkner and Stohlgren 1997, Ferguson et al. 2001). In fact, small national parks and other managed areas may support a disproportionately large portion of the regional species pool (Falkner and Stohlgren 1997). However, in fragmented urban or suburban landscapes, the continuing effects of land-use modification and other human activities often threaten the ecological value of these protected areas.

On remnant patches in fragmented landscapes, native small-mammal species often persist because of their small home ranges (Mahan and O'Connell 2005). However, these species respond to local variation in microhabitat structure (e.g., vertical vegetation structure, volume of coarse woody debris, canopy openness, and percent grass-cover), affecting the survival of individuals and the dynamics of local populations (Bellows et al. 2001; Bowman et al. 2000; Clark et al. 1987; Dickman 1987; Dickman and Doncaster 1987; Garden et al. 2007; Kaufman et al. 1983, 1985; Manning and Edge 2004; Washburn et al. 2000). Disturbances that alter the structure of native habitats may reduce species richness and result in compositional shifts toward more generalist species (DeStefano and DeGraaf 2003; Donnelly and Marzluff 2004; McKinney 2002a, 2006; Nilon and VanDruff 1987). The invasion of exotic generalist species (e.g., *Mus musculus* L. [House Mouse] and *Rattus norvegicus* Berkenhout [Norway Rat]), and associated reductions in abundance of native generalists (e.g., *Peromyscus leucopus* Rafinesque [White-footed Mouse]) may be indicative of highly disturbed areas (Huckstep 1996, Nilon and VanDruff 1987). Such compositional shifts have been correlated with increases in bare soil, tree-canopy loss, and litter and other anthropogenic refuse (Huckstep 1996, Nilon and VanDruff 1987, VanDruff and Rowse 1986).

In urban environments, many human activities may directly or indirectly disturb the structure of native habitat. For example, the spread of exotic grasses, which is facilitated by the presence of edge habitat, corridors for dispersal (e.g., roadways), heavy visitation/tourism, and ecologically unsound management, may homogenize local plant communities and degrade native habitat (Cilliers et al. 2008; Ehrenfeld et al. 2001; Lesica et al. 1993; Lonsdale 1999; McKinney 2002b, 2004, 2006; Tyser and Worley 1992; Whipple 2001). In addition, the culturally based management of historic sites may have similar effects. For example, historic battlefields are more easily maintained as dense mats of cool season grasses such as *Festuca* spp. (fescue grasses), but are likely inhospitable to many small vertebrate species (Pagels et al. 2005a). Such practices may promote novel communities that are detrimental to biodiversity conservation. Along urban-to-rural gradients, the importance of addressing these issues is increasing as human disturbances become more persistent and the structure of our native landscape is more severely degraded. Understanding compositional trends with respect to local habitat conditions is vital to planning for and prioritizing wildlife conservation in all types of managed areas (Andren 1994, Sauvajot et al. 1998).

Whereas small-mammal communities may be affected by local habitat conditions, larger carnivorous mammals have more expansive home ranges and respond

to larger scale variations in habitat suitability. Populations of large mammalian predators may respond negatively to any disturbance that reduces patch connectivity and increases the likelihood of encountering humans (Crooks 2002, Noss et al. 1996, Tigas et al. 2002, Woodroffe and Ginsberg 1998). While top predators may persist in urban areas (e.g., Tigas et al. 2002), declining numbers of large carnivores in remnant patches may have cascading effects on multiple trophic levels. Mesopredator-release theory suggests that mesopredators (i.e., medium-sized carnivores) benefit from the absence of larger predators and ultimately reduce the abundance of smaller species upon which they prey (Soule et al. 1988). However, like other generalist species, the success of mesopredators in urbanized landscapes may also result from their ability to take advantage of anthropogenic subsidies (e.g., human refuse, ornamental fruits and vegetables, and pet food). Carnivores of all sizes present unique conservation challenges in human-dominated landscapes (Gehrt et al. 2010).

Small national park areas may be extremely important for maintaining the richness of regional mammal communities, and in urbanized areas, remnant patches contained within protected areas should be prioritized for biological conservation (Croci et al. 2008). Improving the environmental conditions for native flora and fauna requires specific knowledge of local species assemblages and how they are affected by specific habitat characteristics. Empirical studies that identify relationships between local communities and their environment are of broad interest to managers and biologists who are concerned with maintaining the ecological value of natural resources and regional species pools.

The primary objective of this study was to compare mammal assemblages and vegetation characteristics at locations in urban and peri-urban habitat patches (see next section for definitions of urban and peri-urban). Patches were categorized into habitat types that were equally replicated in urban and peri-urban areas. Our study addressed the following questions: (1) Are there differences between the mammal assemblages of urban and peri-urban locations? (2) If the mammal assemblages differ, which species and trophic levels (prey species, mesopredators and/or top predators) account for the differences? (3) Are vegetation characteristics good predictors of prey-species abundances? (4) Is there a relationship between the abundances of prey and mesopredator species?

Study Areas

We defined an urban area as one that was located within city limits and included high-density development with impervious structures (e.g., buildings, sidewalks, parking lots) and roads with high traffic volume (>100 cars/day) (Forman 1995). For this study, we established urban locations within the Eastern Front Unit of Petersburg National Battlefield (Eastern Front). This area is located within the city limits of Petersburg, VA. Other surrounding metropolitan areas include Colonial Heights, VA to the northwest and Hopewell, VA and Fort Lee to the northeast (Figs. 1, 2). This NPS area contains remnant habitat near a center of urbanization and is located in the vicinity of the Fall Line near the juncture of the Piedmont and

Coastal Plain physiographic provinces. The Eastern Front covers an area of 585 ha that includes fields of fescue grasses and a mosaic of forested successional habitat types ranging from pine- to hardwood-dominated forests. Less-developed areas adjacent to the Eastern Front occur only on the southeastern boundary. As a result of its proximity to Petersburg, Colonial Heights, Hopewell, and Fort Lee (combined population ≈80,000), park visitation (≈150,000 visitors/yr), and surrounding commercial and residential development, the flora and fauna of this NPS area are likely heavily influenced by the effects of urbanization (see Fig. 2).

We define peri-urban as a rural area adjacent to a city or town (Clergeau et al. 2001, Snep et al. 2006). The peri-urban locations of our survey were established within the Five Forks Unit of Petersburg National Battlefield (Five Forks) in Dinwiddie County, 24 km southwest of the urban trapping area (Figs. 1, 3). Five Forks is also located in close proximity to the Fall Line. However, relative to urban trapping locations, the peri-urban locations are closer to the rural end of the urban-to-rural gradient. The area covers 452 ha, and includes mowed fields of fescue grasses and a variety of successional habitat types ranging from pine- to hardwood-dominated forests. The peri-urban sites lie within a rural setting that includes agricultural fields, forests of various ages, and scattered residences that are characteristic of
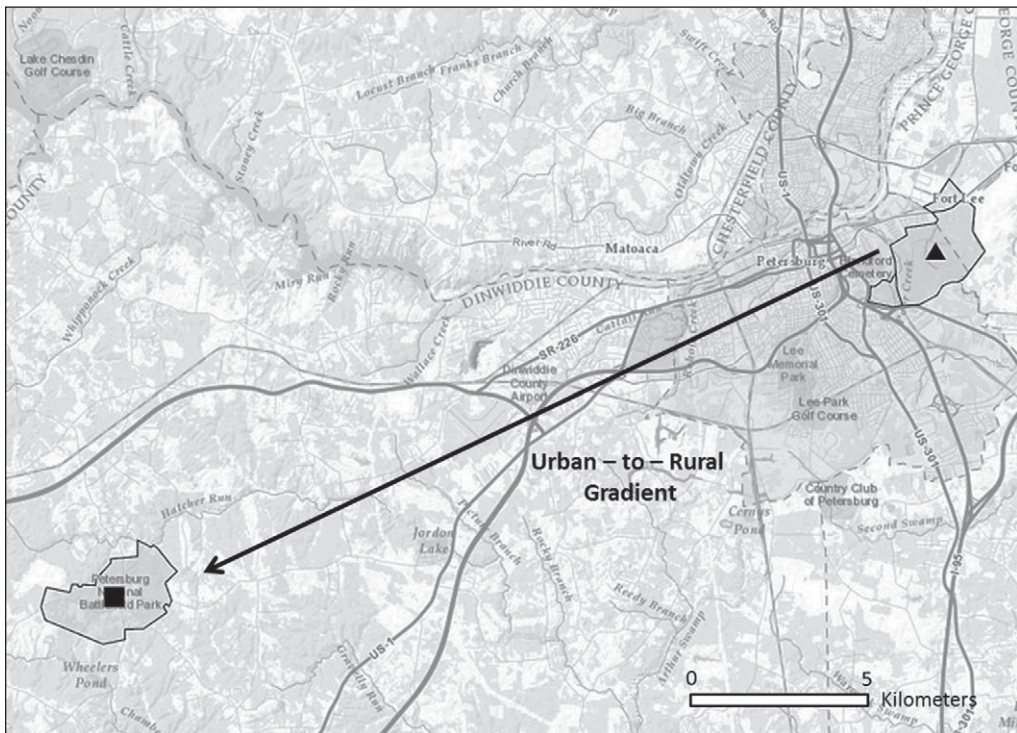

Figure 1. Map of urban and peri-urban survey areas and the surrounding landscape. Locations of these areas are shown with respect to the urban-to-rural land-use gradient that radiates outward from downtown Petersburg, VA. Urban and peri-urban survey areas are represented by a triangle and a square, respectively. The map was created with ArcGIS 10 software (Esri).

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much of present-day south-central Virginia. The landscape is less fragmented by impervious surfaces than our urban site and contains larger contiguous patches of habitat that overlap with the boundaries of our peri-urban survey area (Figs. 1, 3).

Methods

Trapping locations

Before selecting trapping locations, we first determined the major habitat types (i.e., plant communities) within our urban and peri-urban survey areas. With the initial help of natural resource managers and orthophotographs, we determined the available habitat types and scouted possible trapping locations within each. In all, 5 major habitat types were identified in both urban and peri-urban areas: field-forest
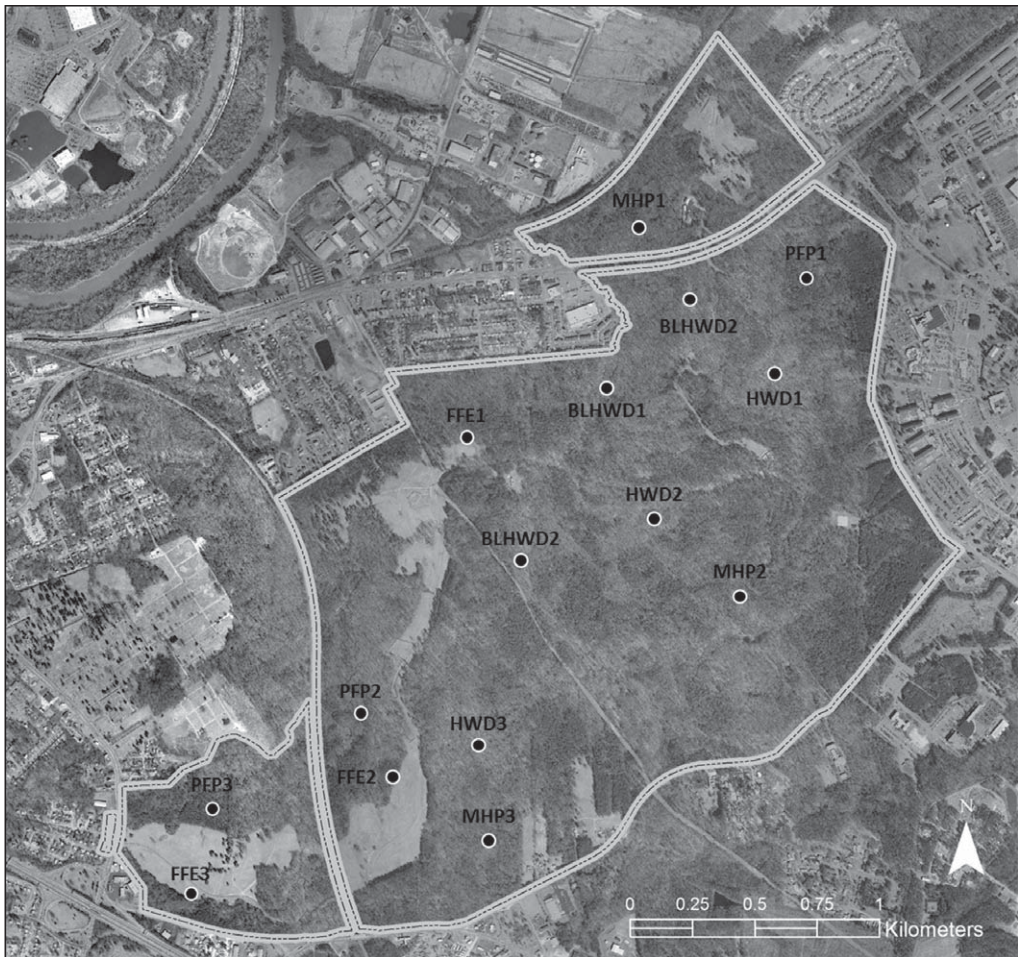

Figure 2. Orthophotograph of urban sites located within the city limits of Petersburg, VA. There are 15 sites total with 3 replicates for each of 5 habitat types: field-forest edge (FFE), pine-forest plantation (PFP), hardwood (HWD), mixed-hardwood pine (MHP), and bottomland hardwood (BLHWD). Image was taken in 2006 and used with permission from the National Park Service.

edge (FFE), pine-forest plantation (PFP), hardwood (HWD), mixed hardwood-pine (MHP) and bottomland hardwood (BLHWD). We then established three replicate trapping locations in each habitat type (total of 15 locations within both survey areas). We randomly selected trapping locations using a grid (overlain on orthophotographs) and random number generator. Boundaries of all trapping locations were at least 300 m apart and >30 m from edge habitat. Stratifying our trapping locations in this way safeguarded against sampling bias (i.e., incorporation of more habitat heterogeneity in one area than the other) (Figs. 2, 3).

Field-forest edge (FFE). This habitat type occurred where field maintenance or mowing created very abrupt or narrow contact areas along field-forest edges. In most cases, FFE sites were only 1–5 m wide. Vegetation along FFE habitat was typically a mix of field and forest vegetation, and was much more heterogeneous than that of either the field or forest habitats. This habitat type contained both coniferous and
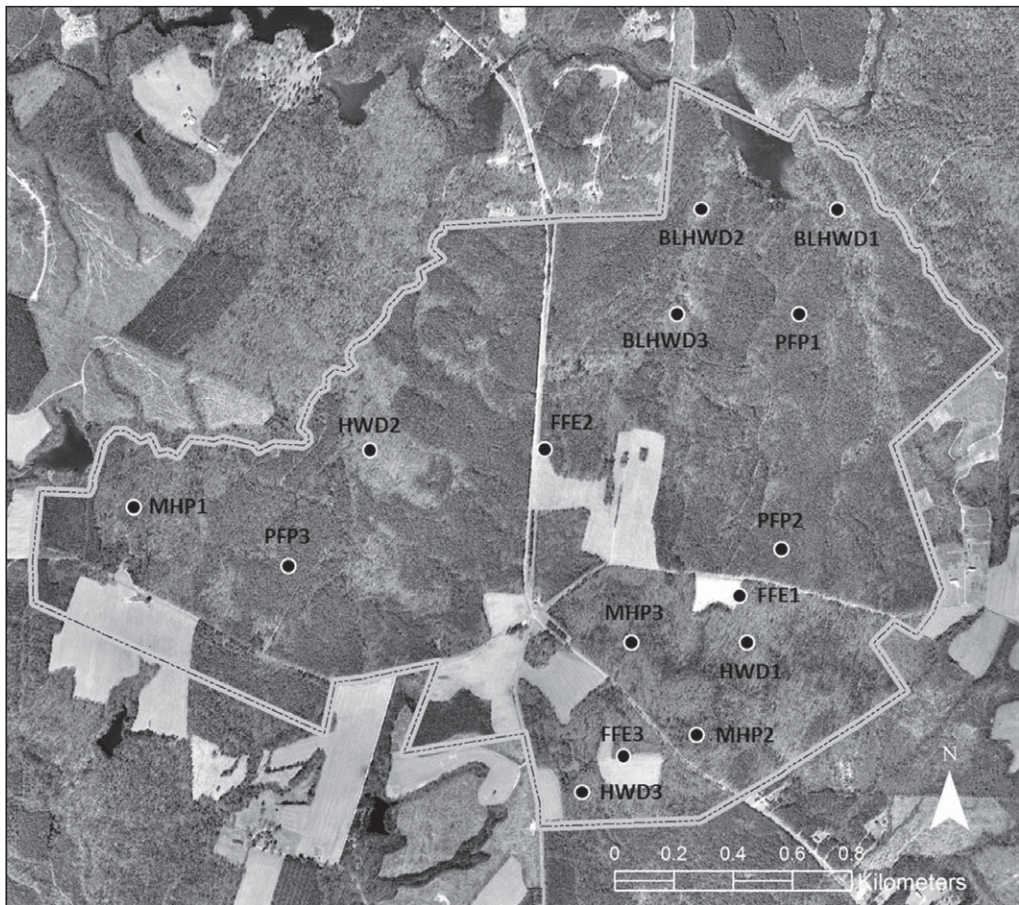

Figure 3. Orthophotograph of peri-urban sites located in Dinwiddie County, VA. There are 15 sites total with 3 replicates for each of 5 habitat types: field-forest edge (FFE), pine-forest plantation (PFP), hardwood (HWD), mixed-hardwood pine (MHP), and bottomland hardwood (BLHWD). Image was taken in 2006 and used with permission from the National Park Service.

deciduous species in the overstory, and the understory was comprised of saplings of overstory species. However, the understory was often dominated by shade-intolerant pioneer species such as *Juniperus virginiana* L. (Eastern Red Cedar). Fescue grasses and other exotic species including *Ailanthus altissima* Mill. (Tree of Heaven), *Ligustrum sinense* Lour. (Chinese Privet), and *Lonicera japonica* Thunb. (Japanese Honeysuckle) were more common here than in other habitats.

Pine forest plantation (PFP). The overstory of this habitat type was dominated by *Pinus taeda* L. (Loblolly Pine), and although tree age was not determined, it was estimated that these stands ranged from 20–25 y in age. The stands we sampled had begun a natural thinning process, and the understory was beginning to open at most locations. However, the ground layer still consisted of a thick layer of pine-needle litter. The understory of this habitat type was dominated by saplings of deciduous tree species, most commonly *Liquidambar styraciflua* L. (Sweetgum). Nearly all understory trees in this habitat type were small (diameter at breast height [DBH; breast height = 1.5 m] < 10 cm). Although grasses and herbs were relatively uncommon here, the invasive species *Microstegium vimineum* Trin. (Japanese Stiltgrass) was present in urban PFP sampling locations.

Hardwood (HWD). This habitat type was characterized by deciduous species in the overstory and understory. Overstory species common to both urban and peri-urban locations included Sweetgum, *Acer rubrum* L. (Red Maple), *Liriodendron tulipifera* L. (Tulip Poplar), *Nyssa sylvatica* Marshall (Blackgum), *Quercus alba* L. (White Oak), *Quercus rubra* L. (Northern Red Oak), and *Carya* spp. (hickory species). Subcanopy species included *Sassafras albidum* Nutt. (Sassafras), *Carpinus caroliniana* Walter (Ironwood), *Ilex opaca* Aiton (American Holly), *Cornus florida* L. (Dogwood), and Eastern Red Cedar. Understory tree species were mostly saplings of overstory species. Ground-cover consisted primarily of deciduous leaf litter. Herbs, grasses, and shrubs were relatively sparse in the HWD habitat type.

Mixed hardwood-pine (MHP). This habitat type included both deciduous and coniferous species in the overstory and is an intermediate successional stage between pine and hardwood forest types. Loblolly Pine was the dominant coniferous species. *Pinus virginiana* Mill. (Virginia Pine) was also present in this habitat type. Although deciduous trees were more abundant than conifers in the overstory, the average DBH of conifers was considerably larger than that of deciduous species. Hardwood species in this habitat type were a mix of those species found in the HWD habitat type. Understory tree species were mostly saplings of overstory hardwood species. Few coniferous saplings were found in the understory. At urban locations, Japanese Stiltgrass was observed in this habitat type.

Bottomland hardwood (BLHWD). This habitat type was restricted to flood-plain forests associated with streams or rivers. Overstory trees were primarily deciduous, and typical species included *Platanus occidentalis* L. (American Sycamore), *Fraxinus pennsylvanica* Marshall (Green Ash), and *Betula nigra* L. (River Birch). The understory was comprised of saplings of overstory species. The aromatic shrub *Lindera benzoin* L. (Spicebush) was also common. Grasses and herbs were observed in much greater abundance here than in any other habitat

type. Japanese Stiltgrass (exotic) and *Arundinaria gigantea* Walter (Switch Cane) were abundant at urban BLHWD locations. Ferns were also common in this habitat type. With locations so close to water sources, BLHWD sampling sites had very moist substrates, and washouts from flooding were not unusual at many of our BLHWD sampling sites.

Mammal surveys

Seven trapping sessions were partitioned into seasons that occurred between June 2003 and August 2004. During each session, urban and peri-urban locations were sampled for the same number of nights during the same two-week time period. Depending on season, trapping sessions ranged from 2–4 nights of continuous trapping. A trap-night was subtracted from the effort when a trap was found to be sprung but unoccupied (modified from Nelson and Clark 1973).

We used a circular-plot sampling scheme for all trapping locations except those located in field-forest edges. The circular design has been successfully used in studies focused on mammal population dynamics (Orrock et al. 2000), mammal communities (Bellows et al. 2001, McShea et al. 2003), endangered species surveys (Orrock et al. 2000), and determining new records of occurrence (Bellows et al. 1999). Each trapping plot (i.e., trapping location) consisted of a 30-m-diameter circle with markers at the center and 15 m from the center in each cardinal direction so that the plot was divided into four equal quadrants. We used 60-m-long transects instead of circular plots in the FFE habitat type because patches in this habitat type were often narrow, and circular plots would have overlapped into other habitat types. Based on trap type and trap number, the sampling effort on transects was equivalent to that of the circular plots, but traps were arranged in a linear fashion at 20-m intervals (i.e., points at 0, 20, 40 and 60 m).

At each location, we used a combination of four different trap types that would allow us to effectively sample for all small prey and mesopredator species. Three Sherman live traps (7.5 x 9 x 23 cm) were placed at likely capture spots within a 2-m-radius of each cardinal direction marker. Two small Tomahawk live traps (40.6 x 12.7 x 12.7 cm) were placed in opposite quadrants from each other, and one medium-size Tomahawk live trap (81.3 x 25.4 x 30.5 cm) was placed at or near the center of the plot. Sherman live traps were baited with an oatmeal-peanut butter mixture that was wrapped in wax paper and hung from the back door of the trap. Small amounts of peanut butter were also placed on the open front door. Small Tomahawk traps were baited with apples covered in peanut butter, and large Tomahawk traps were baited with apples and sardines.

For more effective sampling of the smallest prey species (i.e., shrews), two pitfall traps were placed in each of the four quadrants (Kirkland and Sheppard 1994, Mitchell et al. 1993). Sixteen-ounce (533 ml) beverage cups (i.e., pitfall traps) were placed in the ground at likely capture spots, i.e., where natural drift fences (logs and stumps) were found. We filled the bottom 5 cm of each pitfall trap with water, and we used plastic mesh lids (15 x 15 cm) elevated by nails to shield the pitfall traps from falling leaves and other debris. Because we observed poor capture success with shrews in 2003, we added two larger pitfall traps to each site for sampling in

spring 2004. We made the larger pitfall traps from 2-L plastic soda-bottles with the tops cut off (after Handley and Varn 1994) and lowered the plastic mesh lids over all pitfall traps to close them between sampling sessions.

Mammals smaller than or equal to the size of an adult *Sigmodon hispidus* Say and Ord (Hispid Cotton Rat; ≈100–225 g) were tagged with Monel ear tags (National Band and Tag Co., Newport, KY). Mammals larger than or equal to the size of an adult *Sciurus carolinensis* Gmelin (Eastern Gray Squirrel; ≈400–600 g) were marked with non-toxic spray paint and examined for distinguishing features and approximate age. Both marking techniques allowed us to identify recaptures within each trapping session. All mammals were released at the location of capture.

Using TrailMaster's Active Infrared Trail Monitor (Model # TM1550) and Camera Kit (Model # TM35-1) (TrailMaster Infrared Trail Monitors, Lenexa, KS), we conducted night-camera photography to document medium to large species (i.e., mesopredators and top predators). This method is appropriate for use in mammal inventories where larger mammals need to be surveyed (Silveira et al. 2003). We used three camera traps simultaneously. During five trapping sessions, cameras were active for the same number of nights as the trapping locations. However, camera traps were not placed near trapping locations and instead were most often in secluded wooded areas. Locations of camera traps varied among sampling sessions. We placed cameras in areas most likely to be frequented by medium to large nocturnal mammals (i.e., game trails and small unpaved roads or walking paths), and where vegetation and topography would not trigger the trail monitors. We situated trail monitors at a height of approximately 0.25–0.5 m and set them for a 2-minute delay after each photograph. These camera sites were active from dusk to dawn and were baited with sardines, peanut butter, apples, and raw chicken.

Vegetation surveys

Within each trapping location, DBH was recorded for all trees, defined as woody plants with a DBH ≥5 cm. For transect-style trapping locations, any tree less than 6 m from the transect line was considered to be within the trapping location; therefore, the total area sampled was nearly equivalent to that of circular plots. All trees were identified to species, with a few exceptions when we were only able to assign a genus. Groundcover was quantified using the line-transect method (Canfield 1941). For circular trapping locations, we established two 40-m transects that crossed in the plot center to divide the trapping location into four equal quarters. For transect-style trapping locations, we extended the same transect line established for mammal sampling (60 m) by 20 m and used it to quantify groundcover. Using the line-transect method, we recorded observations at 1-m intervals in the following categories: grass, non-grass herbaceous, shrub (woody plants <5 cm DBH), and downed wood. Downed wood was considered to be any portion of a woody stem or trunk >1 m in length. We recorded the diameter of any intercepted piece of downed wood that had a DBH ≥4 cm (i.e., coarse woody debris).

Data treatment and statistical analyses

For each trapping location, we calculated the relative abundance of prey and

mesopredator species as the total number of unique individuals captured (M_{t+1}, excludes recaptures) divided by the total number of trap-nights, for traps in which a species could be captured during that trapping session (Slade and Blair 2000). The relative abundance of each species captured was expressed per 100 trap-nights and averaged across all 7 trapping sessions for each trapping location; species abundances at each trapping location are expressed as mean relative abundances. Abundance estimated using M_{t+1} is an index of population size because the number of individuals captured is a function of population size as well as the likelihood that an individual will be captured (Pollock et al. 2002, Slade and Blair 2000). M_{t+1} was used because it performs as well as an estimator that incorporates capture probability (e.g., the Lincoln-Petersen estimator) when captures are low (Slade and Blair 2000), as was the case in this study and other short-term surveys. Although capture probability for a single species may vary depending upon habitat type, season, or trap type (Pollock et al. 2002), our trapping effort was designed to eliminate variation in these factors between survey areas (urban vs. peri-urban) and among habitats (FFE, PFP, MHP, HWD, BLHWD).

We used two-way ANOVA to test for effects of habitat type and landscape type (urban vs. peri-urban) on the mean relative abundances of species captured. In addition, we compared mammal species abundances among habitats within each landscape type using one-way ANOVA. We calculated the Shannon-Weiner index of diversity for each trapping location using the mean relative abundances of mesopredator and small-prey species and compared diversity between urban and peri-urban trapping locations using an independent samples *t*-test. Diversity of small-prey species alone was similarly compared. All univariate tests were conducted using SAS 9.2 (SAS Institute 2007).

Across all locations, trends in mammal community composition (excluding top predators) were examined using non-metric multidimensional scaling ordination (NMDS, McCune and Grace 2002); we used Bray-Curtis dissimilarities, based on the mean relative abundances of species captured, to generate the ordination. We also performed an NMDS ordination using only the mean relative abundances of small prey species (excluding mesopredators and top predators). We used analysis of similarity (ANOSIM, Clarke 1993) to test for significant differences between urban and peri-urban communities. The NMDS and ANOSIM were conducted using the DECODA software package (Minchin 1989).

We also conducted an indicator species analysis to identify species indicative of either landscape type (PC-ORD; McCune and Mefford 1999). For trapping locations within each landscape type, we calculated indicator values from the mean relative abundance and/or relative frequency of occurrence for individual species. Faithful occurrence and/or concentrated abundance within a single group of sample units (i.e., urban or peri-urban) produce a higher indicator value for that species (Dufrene and Legendre 1997, McCune and Grace 2002).

We used the number of night photographs as an estimate of relative abundance for species of mesopredator size or larger within each survey area. This was the sole indicator of top predator abundance, and a supplemental measure for the abundance

of mesopredators. Because individuals could not be discerned among night-camera photographs, we used the proportion of each species represented in photographs as estimates of relative abundance. Abundance estimates from camera traps were summarized but not used in any statistical analysis.

We calculated the percent cover of grasses, non-grass herbaceous plants, shrubs, and woody debris from line intercept data for each location. We also calculated the volume of coarse woody debris (after Van Wagner 1968, Warren and Olsen 1964) and basal area for all tree species (genus in some cases) at each site. All vegetation data were square-root transformed to satisfy statistical assumptions. We used two-way ANOVA to test for differences in vegetation variables with respect to landscape and habitat type. We used one-way ANOVA to test for significant differences in vegetation among habitat types within each landscape type.

We used vector fitting to test whether trends in the NMDS ordinations of mammal assemblages were significantly correlated with vegetation variables (vectors). For ordinations using only small-prey species abundances, the relative abundances of mesopredator species were also used as vectors. Vector fitting was conducted using the DECODA software package (Minchin 1989). Finally, for species with mean relative abundances that differed significantly between landscape types and/or among habitat types, we conducted regressions of their mean relative abundance against each vegetation variable. In addition, the mean relative abundances of prey species were regressed against the mean relative abundances of mesopredator species. We conducted all regressions using SAS version 9.2 (SAS Institute 2007). For all analyses, mean abundance data were square-root transformed to meet statistical assumptions.

Results

Across 7 seasonal trapping sessions (21 nights), we recorded a total of 7352 and 7596 trap-nights at urban and peri-urban trapping locations, respectively (Table 1). The small discrepancy in trap nights between the two focal areas is due to a greater number of sprung traps at urban trapping locations. Sprung traps were most likely the result of tampering by mesopredator species. Excluding results from camera trapping, a total of 145 unique individuals (excluding recaptures), representing 8 species, was captured at urban sites (Table 2); five species were small prey and 3 were mesopredator species. The White-footed Mouse was the most abundant species (32% of individuals captured), followed by *Procyon*

Table 1. Number of trap nights per trap type at trapping locations within each landscape type.

	Urban	Peri-urban
Pitfall	3013	3030
Sherman	3416	3603
Small Tomahawk	584	613
Medium Tomahawk	297	308
Camera	42	42
Total	7352	7596

lotor L. (Raccoon: 24%), *Blarina carolinensis* Bachman (Southern Short-tailed Shrew: 17%), and *Didelphis virginiana* Kerr (Virginia Opossum: 13%). A total of 193 individuals, representing 13 species, was captured at peri-urban trapping locations; 9 species were small prey and the remaining 4 were mesopredators (Table 2). The White-footed Mouse was again the most abundant species (64% of individuals captured), followed by the Southern Short-tailed Shrew (10%), and *Sorex longirostris* Bachman (Southeastern Shrew: 7%). Five small-mammal species were captured at peri-urban trapping locations that were not captured at urban sites: *Sorex hoyi* Baird (Pygmy Shrew), Southeastern Shrew, *Microtus pinetorum* LeConte (Pine Vole), *Ochrotomys nuttalli* Harlan (Golden Mouse), and *Oryzomys palustris* Harlan (Marsh Rice Rat). Based on the mean relative abundances of small prey and mesopredator species, the average Shannon-Weiner indices

Table 2. Number of captures (includes recaptures) per trap type of all mammalian species captured at trapping locations in the urban and peri-urban survey areas (camera captures = number of photographs). The number of unique individuals captured (excludes recaptures) is in parentheses. SM = small prey, MP = mesopredator, and TP = top predator.

				Tomahawk		
	Guild	Pitfall	Sherman	Small	Medium	Camera
Urban						
Peromyscus leucopus	SP		118 (47)			
Blarina carolinensis	SP	14 (14)	11 (11)			
Sigmodon hispidus	SP		13 (10)			
Tamias striatus L. (Eastern Chipmunk)	SP		1 (1)			
Sciurus carolinensis	SP			4 (3)		
Procyon lotor	MP				35 (35)	89
Didelphis virginiana	MP			3 (3)	16 (16)	11
Felis catus	MP				6 (5)	1
Urocyon cinereoargenteus	MP					89
Peri-urban						
Peromyscus leucopus	SP	3 (3)	274 (120)			
Blarina carolinensis	SP	6 (6)	13 (13)			
Sigmodon hispidus	SP		1 (1)	1 (1)		
Oryzomys palustris	SP		13 (10)			
Sorex longirostris	SP	11 (11)	2 (2)			
Sorex hoyi	SP	1 (1)				
Microtus pinetorum	SP	1 (1)	2 (2)			
Mus musculus L. (House Mouse)	SP		1 (1)			
Ochrotomys nuttalli	SP	1 (1)	5 (5)			
Didelphis virginiana	MP		1 (1)		9 (9)	25
Mephitis mephitis Schreber (Striped Skunk)	MP			1 (1)	1 (1)	
Felis catus	MP				1 (1)	
Procyon lotor	MP				2 (2)	5
Felis rufus	TP					1
Canis latrans	TP					5

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(± standard error) for urban and peri-urban trapping locations were 0.82 ± 0.11 and 0.76 ± 0.11, respectively. Using only the relative abundances of prey species, average indices were 0.46 ± 0.10 (urban) and 0.50 ± 0.08 (peri-urban). In both cases, an independent samples *t*-test verified that differences between urban and peri-urban trapping locations were non-significant ($P > 0.05$).

There was a total effort of 42 camera trap-nights within the boundaries of each survey area (Table 1). Photographs from urban camera traps indicated an abundance of Raccoon and *Urocyon cinereoargenteus* Schreber (Gray Fox). Each species was recorded on 89 photographs, and combined represented 94% of the photographs taken within this survey area. The remaining 6% of photographs identified *Felis catus* L. (Feral Cat) and Virginia Opossum (Table 2). Peri-urban camera traps took 36 photographs, revealing the presence of Virginia Opossum (69% of photos), Raccoon (14%), *Canis latrans* Say (Coyote: 14%), and *Felis rufus* Schreber (Bobcat: 3%). Five photographs of Coyote and one photograph of a Bobcat documented the presence of these top predators on the peri-urban landscape. Camera-trapping documented one additional species at urban sites (Gray Fox) and two additional species at peri-urban sites (Bobcat and Coyote) that other trap types failed to capture (Table 2).

Two-way ANOVA results indicated that White-footed Mouse and Raccoon abundances differed significantly between urban and peri-urban landscape types. The White-footed Mouse was significantly more abundant ($F_{1,20} = 23.62$, $P < 0.0001$) at peri-urban (mean = 5.21 ± 0.78) versus urban (mean = 1.62 ± 0.28) trapping locations. In contrast, the Raccoon was significantly more abundant ($F_{1,20} = 39.73$, $P < 0.0001$) at urban (mean = 12.02 ± 2.65) versus peri-urban (mean = 1.43 ± 1.03) sites. Indicator-species analysis suggested that the White-footed Mouse, Southeastern Shrew, and Raccoon were good indicator species. The White-footed Mouse and Southeastern Shrew were significant indicators of peri-urban locations (IV = 76.3, $P = 0.001$; IV = 46.7, $P = 0.007$) and the Raccoon was indicative of urban communities (IV = 83.4, $P = 0.001$). Between landscape types, differences in the relative abundance of all other species were non-significant ($P > 0.05$).

Two-way ANOVA results suggested the Southern Short-tailed Shrew (captured only at peri-urban sites) was the only prey species with significant differences in mean relative abundance among habitat types ($F_{1,20} = 4.94$, $P = 0.0062$). Abundance of this species was higher in MHP (mean = 0.67 ± 0.17) relative to HWD (mean = 0.12 ± 0.07) and PFP (mean = 0.06 ± 0.04), and significantly higher in BLHWD (mean = 0.46 ± 0.07) relative to PFP. Results from one-way ANOVA confirmed that across peri-urban sites, the mean relative abundance of the Southern Short-tailed Shrew was significantly higher ($F_{4,10} = 10.03$, $P = 0.0016$) in MHP (mean = 0.73 ± 0.20) and BLHWD (mean = 0.45 ± 0.15). On the urban landscape, the Raccoon was significantly more abundant ($F_{4,10} = 5.87$, $P = 0.0107$) in FFE (mean = 18.06 ± 7.73) and BLHWD (mean = 24.20 ± 4.03). No other species showed significant differences among habitat types across or within landscape types. In addition, there were no interactive effects between landscape and habitat type that were significantly related to differences in species abundances.

Two-way ANOVA suggested percent shrub-cover was significantly higher ($F_{1,20}$ = 37.20, P < 0.0001) at peri-urban locations (mean = 32.1 ± 5.3) compared to urban sites (mean = 6.0 ± 1.9). However, there were no significant patterns in shrub cover with respect to habitat type. Although percent grass-cover was higher in urban (mean = 39.3 ± 10.6) versus peri-urban sites (mean = 22.2 ± 7.1), the differences were not statistically significant ($F_{1,20}$ = 3.68, P = 0.0694). Results from one-way and two-way ANOVA suggested that grass cover was significantly higher in FFE and BLHWD, a trend that was observed across all sites in both landscape types. The highly invasive Japanese Stiltgrass was found at several urban trapping locations including BLHWD sites and was frequently observed throughout the survey area. Across habitat types, other significant trends in vegetation included a greater coverage of non-grass herbs and less coverage of coarse woody debris in FFE habitat.

Results from a two-dimensional NMDS ordination of trapping locations indicated differences between urban and peri-urban sites based on the mean relative abundances of mesopredator and small prey species (Fig. 4), and analysis of similarity (ANOSIM) confirmed this trend (R = 0.54, P < 0.0001). Vector fitting suggested that these dissimilarities were significantly correlated with the aforementioned
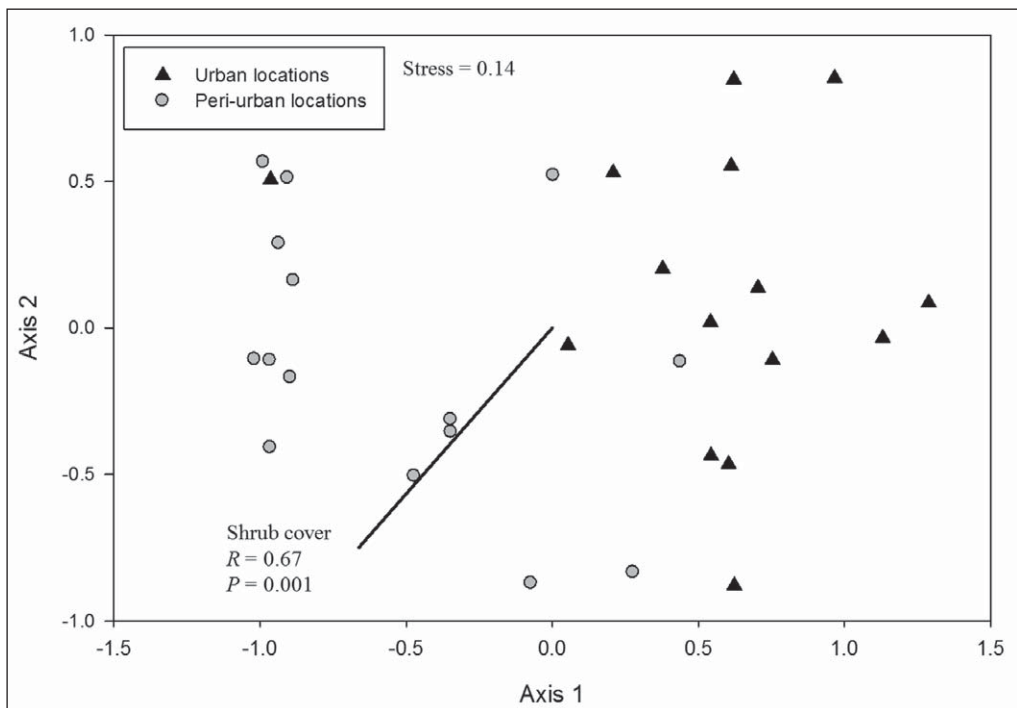

Figure 4. Non-metric multi-dimensional scaling (NMDS) ordination of the relative abundance of mesopredators and small-prey species at 30 trapping locations. Trapping locations are grouped by the areas within which they are located (urban and peri-urban). Analysis of similarity (ANOSIM) indicated significant dissimilarities between the two groups (R = 0.54, P < 0.0001). The vector represents the direction through the ordination space that is significantly correlated with shrub cover.

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differences in the percent cover of shrubs ($R = 0.67$, $P = 0.001$), where shrub cover was significantly higher in peri-urban locations. A two-dimensional NMDS ordination using only small-prey species abundances suggested a similar pattern of dissimilarity between urban and peri-urban communities (Fig. 5), a trend that was again statistically verified by ANOSIM ($R = 0.26$, $P = 0.0002$). Results from vector fitting also suggested that these dissimilarities were significantly correlated with differences in the percent cover of shrubs ($R = 0.65$, $P = 0.001$). In addition, percent grass-cover was correlated with trends in ordination space ($R = 0.47$, $P = 0.03$). However, the direction of the grass vector was not well aligned with dissimilarities between urban and peri-urban locations. Instead, this vector was more closely aligned with dissimilarities among urban sites (Fig. 5). Mesopredator species abundances and all other habitat variables were not correlated with trends in small-mammal composition and relative abundance ($P > 0.05$).

The White-footed Mouse and Raccoon were the most abundant prey and mesopredator species, respectively, captured during our surveys and the only species whose mean relative abundances differed between urban and peri-urban trapping
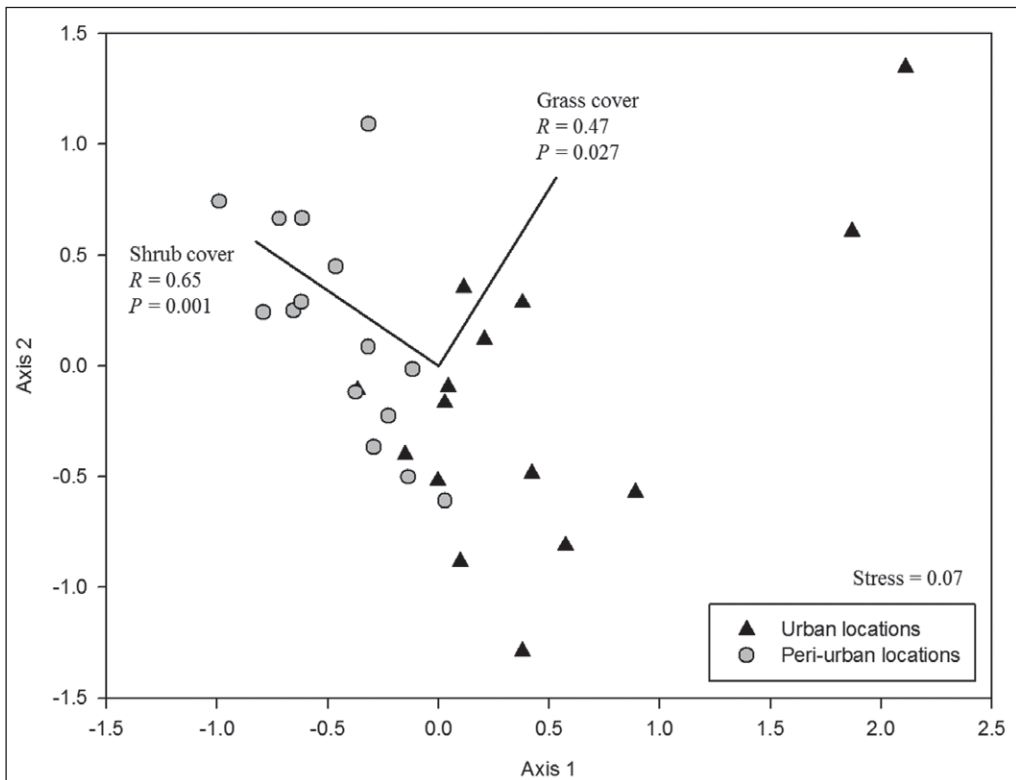

Figure 5. Non-metric multi-dimensional scaling (NMDS) ordination of the relative abundance of mesopredators and small-prey species at 30 trapping locations. Trapping locations are grouped by the areas within which they are located (urban and peri-urban). Analysis of similarity (ANOSIM) indicated significant dissimilarities between the two groups ($R = 0.26$, $P = 0.0002$). The vectors represent the directions through the ordination space that are significantly correlated with shrub and grass cover.

locations. Results of regression analyses show a significant positive and negative relationship between shrub cover and the relative abundances of White-footed Mouse and Raccoon, respectively ($R^2 = 0.37$, $P = 0.0004$; $R^2 = 0.27$, $P = 0.003$). Regression analyses failed to detect significant relationships between the relative abundances of these species and any other vegetation variable or mesopredator abundances. ($P > 0.05$).

Discussion

Of the 38 mammalian species with distributions that overlap our survey areas (Linzey 1998, Webster et al. 1985), 24 and 40% of these species were captured at urban and peri-urban trapping locations, respectively. These results are consistent with observations from much smaller urban and peri-urban woodlands where urban sites accommodated just over half the number of species found at peri-urban locations (Croci et al. 2008). However, small NPS areas are federally managed, and tend to be larger than other urban parks and woodlands. Therefore, with proper management we would expect the mammal assemblages of sites within urban NPS areas to more closely resemble their more rural counterparts. Nevertheless, we did not expect large-bodied carnivores, which have extensive home ranges, to be present on the urban landscape.

During our surveys, camera traps recorded the presence of Coyote and Bobcat only at peri-urban locations. Although top predators have been observed in urban habitats (Tigas et al. 2002), our results suggest at the very least a reduced presence of Coyote and Bobcat within the urban landscape we surveyed. Smaller patch sizes, reduced connectivity, and the negative consequences of encounters with humans may all be responsible for reducing populations of top predators (Crooks 2002, Riley et al. 2006, Tigas et al 2002, Woodroffe and Ginsberg 1998). In comparison, results from both live- and camera-trapping produced convincing evidence that three important mesopredator species (Raccoon, Gray Fox, and Feral Cat) are significantly more abundant at urban sites relative to peri-urban locations. We suspect that a release from predation is at least partly responsible for this trend (Soule 1988). However, we also agree that the total effect of mesopredator release is mediated by the productivity of the system (Elmhagen and Rushton 2007). If mesopredators can more thoroughly exploit the productivity of a system when top-down control (i.e., lethal and non-lethal effects of predation) is removed, then removal of top predators in more productive systems will have a greater positive effect on the abundance of mesopredators. In urban ecosystems, natural productivity is subsidized by food from anthropogenic sources (Prange et al. 2004). As such, omnivorous mesopredators (Raccoon, Gray Fox, and Feral Cat), which are free from predation and can exploit anthropogenic subsidies, may flourish under these circumstances (Sinclair et al. 1998, Soule et al. 1988). The Feral Cat is perhaps the best known example of this phenomenon (Sims et al. 2008). On our urban sites, the adjacent residential and commercial developments offer numerous anthropogenic subsidies including refuse from dumpsters and trash cans, pet food, and ornamental fruits and vegetables. Inside park boundaries, refuse from trash cans and scattered

randomly by the nearly 150,000 park visitors per year also contributes to the availability of these subsidies (John Pagels and Adam Chupp, pers. observ.). We argue that Raccoon, Gray Fox, and Feral Cat are benefiting from the exploitation of these subsidies and a release from predation. Managers should implement measures that focus on the control of these species, including educational outreach to local neighborhoods and businesses that address the need for reducing the availability of anthropogenic subsidies.

Relationships between mesopredator and prey species abundances were not evident at any level of analysis and thus we provide no evidence for the cascading effects of mesopredator release (Soule et al. 1988). In addition, we found no consistent trends with respect to prey species assemblages and habitat type, although abundances of the Southeastern Shrew and Southern Short-tailed Shrew were elevated at MHP sites on the peri-urban landscape. Instead, dissimilarities between urban and peri-urban small-mammal communities were highly correlated with shrub cover, which was significantly greater at peri-urban locations. Mean relative abundance of the White-footed Mouse, an indicator of peri-urban sites, was consistent with these trends. Although the White-footed Mouse is a ubiquitous generalist, it has been known to associate with certain habitat characteristics, specifically vertical vegetative structure (Kaufman et al. 1983, 1985). Other small mammal species that were unique to peri-urban trapping locations (Pygmy Shrew, Southeastern Shrew, Pine Vole, Golden Mouse, and Marsh Rice Rat) may also benefit from increases in percent shrub-cover. For example, greater abundances of the Golden Mouse, an arboreal nest-builder, and Southeastern Shrew have been correlated with higher frequencies of shrubs and resultant increases in the vertical structure of woody vegetation (Bellows et al. 2001, Wagner et al. 2000). In addition, forest specialists like the Pygmy Shrew and Pine Vole depend on moisture gradients and spatial heterogeneity that result from more complex assemblages of understory vegetation (Bellows et al. 2001). Furthermore, shrubs may reduce the possibility of detection from predators by increasing overhead cover and decreasing the density of herbaceous vegetation, which may hinder movements of small mammals and increase detection by acoustically oriented predators (Kotler 1984, Rice 1982, Simonetti 1989).

Percent grass cover was well aligned with dissimilarities in small-prey species communities, especially among urban sites (Fig. 5). We documented higher grass coverage at urban sites (marginally significant) and between FFE and BLHWD habitat types. The presence of Japanese Stilt-grass on the urban landscape contributed to higher grass coverage observed at several trapping locations in these areas. However, due to high variability, these observations were only marginally significant. At the Eastern Front (i.e., our urban survey area), efforts to control the spread of Japanese Stilt-grass are targeted towards BLHWD habitat and other communities that are heavily invaded and/or susceptible to invasion (Tim Blumenschine, Petersburg National Battlefield, Petersburg, VA, pers. comm.). While native grasses may be integral components of small mammal habitat, invasions by exotic grasses and other understory plants are a major

detriment to the function of native habitat (Ehrenfeld et al. 2001, McKinney, 2004). Dense populations of Japanese Stilt-grass may have widespread impacts on the recruitment of native woody species, and thus, limit the amount of shrub cover for small mammals (Flory 2010). Although several control methods can mitigate the spread of this species (Judge et al. 2005, Tu 2000), NPS areas are vulnerable to re-introductions due to heavy traffic of vehicles, recreational activities, and other avenues of propagule transport (Lesica et al. 1993, Lonsdale 1999, Whipple 2001). In addition, parks located near population centers and sources of exotic species dispersal (e.g., roadways) may be more vulnerable to invasions (Tyser and Worley 1992). Controlling the spread of Japanese Stilt-grass and other invasive grasses is a complex but crucial step towards preserving suitable habitat in these areas.

In NPS areas, the structure of native habitat may also be reduced by historic landscape restorations that focus on the cultural significance of historic sites. NPS areas designated as National Historical Parks, National Monuments, or National Battlefields may have areas that maintain the cultural appeal of the site at the expense of natural resource preservation (John Pagels and Adam Chupp, pers. observ.). For example, at Petersburg National Battlefield in Petersburg, VA, fields of cool season fescue grasses are used to maintain cultural vistas and "enhance the visitor's ability to interpret the siege that occurred around Petersburg" (NPS 2006). However, unlike warm season bunch-grasses which have good vertical structure and allow a greater freedom of movement for wildlife while also providing cover (DeLong and Brittingham 2001, Ryan and Marks 2006, Washburn et al. 2000), fescue grasses form dense mats and virtually eliminate native habitat structure in these culturally restored patches (IDFW 2006; Mengak 2004; John Pagels and Adam Chupp, pers. observ.). In addition, species of fescue that form symbiotic relationships with endophytic fungi may be even less attractive to herbivorous mammals (Coley et al. 1995). Conversion to warm season bunch-grasses and periodic maintenance (mowing and fescue control) would result in old-field habitat that is preferable to many small-mammal species. Not captured at any trapping location, *Cryptotis parva* (Say) (Least Shrew), *Reithrodontomys humulis* Audubon & Bachman (Eastern Harvest Mouse)*, Microtus pennsylvanicus* (Ord) (Meadow Vole)*,* and *Zapus hudsonius* Zimmerman (Woodland Jumping Mouse) are frequently found in old-field habitat in central Virginia (Bellows et al. 2001; Pagels et al. 1992, 2005b). Among both urban and peri-urban sites, several small-mammal species may benefit from the restoration and maintenance of old-field habitat with little-to-no impact on the cultural interpretation of these historic sites. Combining principles of cultural and ecological conservation could provide numerous opportunities for wildlife conservation in many NPS areas.

In urbanized landscapes, small NPS areas may contain relatively large remnant habitat patches with the potential for native species conservation. Efforts to manage these areas for robust mammal assemblages should seek to increase general habitat complexity for small-mammal species (forest- and field-dwellers, alike). Controlling the spread of exotic grass species and facilitating the recruitment of native shrubs may be crucial components of these management strategies. There is great opportunity for development of adaptive management strategies

that integrate ongoing NPS efforts to control invasive species with the enhancement of wildlife habitat in both culturally and naturally preserved areas. Linking these goals in a flexible management framework could provide an efficient and effective approach for achieving multiple desirable conservation outcomes. While balancing the conservation of natural and cultural resources may be challenging, there are great to the successful integration of these priorities. A great example that is relevant to this study is the conversion of 202 ha (500 ac) at Manassas National Battlefield Park (MNBP) in Manassas, VA to warm-season grasses. In addition to providing higher quality habitat for wildlife, resource managers planted warm-season grasses wherever possible because these species require no lime or fertilizer, are adapted to acidic soils, function as a good riparian buffer that preserves water quality, and maintain historic views of battlefields (NPS 2007). It should be noted that similar to our urban survey site, the MNBP is located in a densely populated area and thus influenced by surrounding urban development. This example is a great success story for managers and conservation biologists to ponder before making decisions about the direction of future management strategies. The ecological value of NPS areas is increasingly threatened by human disturbances. On urban lands that are often culturally preserved, NPS areas offer unique opportunities for wildlife management and conservation.

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