A Tale of Two City Parks: A Case Study of Mammalian Wildlife Impacts of Urbanization in Urban Green Spaces

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Abstract

Urban green spaces (UGS) have the potential to play an increasingly important role in biodiversity conservation. To realize this potential, fine-scale, site-specific, and easily accessible understanding of UGS habitat value is needed, as planning and management of UGS occur primarily at local levels. This camera-trap case study compared mammalian wildlife in two large, Seattle-area UGS, St. Edward State Park (SESP) and Juanita Woodlands Park (JWP). The parks are ecologically similar, but differ in the principal dimensions of urbanization-related habitat impacts: habitat size, fragmentation, and matrix quality, with JWP more heavily urbanized than SESP. Equal sampling effort in two habitat types (riparian forest, 1157 camera-days; upland forest, 1094 camera days) documented greater native mammalian species richness and diversity in SESP in both habitats. Two species, flying squirrel and Townsend's chipmunk, were detected only at SESP. Species occupancy patterns indicated that bobcat, Douglas squirrel and mountain beaver were more likely to be present in SESP, while raccoon, coyote, and black-tailed deer, as well as six non-native species, did not show consistent differences between parks. The predator community of SESP showed a higher overall level of activity, and differed from JWP in species' relative prominence. These results show in a straightforward and accessible manner the critical importance of urbanization impacts in determining the capacity of UGS to support wildlife—particularly small species and "urban avoiders"—and the range of habitat value across large UGS as they exist in a metropolitan area. Species we found to be largely restricted to SESP may be tractable indicators of UGS habitat quality in the urbanized Pacific Northwest region, and SESP provides a possible example of a regional reference site for forested UGS habitat quality. Similar studies can produce analogous applied outcomes in urban settings elsewhere.

Introduction

Urbanization is a major and increasing cause of biodiversity loss worldwide (Seto et al. 2011; McDonald et al. 2018), reducing, fragmenting, and degrading natural habitats on which native species depend (McDonald et al. 2008). Invasion by human commensals and other non-native species further degrade habitats and reduce native biodiversity in urban areas (McKinney 2006; Shochat et al. 2010) With ongoing and projected human population growth and migration to cities (United Nations 2019), the negative impacts of urbanization on biodiversity are likely to intensify in the future (Seto et al. 2012).

Against this backdrop, urban green spaces—parks and other undeveloped lands within urban and suburban areas—have the potential to play an increasingly important role in biodiversity conservation (Aronson et al. 2014, 2017; Ives et al. 2016), although this potential has been under-appreciated (Soanes et al. 2018). For some local endemics, urban habitats harbor the only remaining individuals of a species (McDonald et al. 2008; Soanes and Lentini 2019), the global persistence of which is entirely dependent on maintenance of the urban green space the species inhabits. More commonly, urban green spaces support populations of more widely distributed species, and thus provide a defense against local extinctions and population reductions, e.g., defaunation (Dirzo et al. 2014), that result in reduced local diversity, and can ultimately lead to species extinction (Ceballos et al. 2017).

Beyond the conservation of biodiversity itself, urban green spaces support valuable ecosystem services provided by the biodiversity that they host (Nowak and Dwyer 2007; Luederitz et al. 2015; Zari 2018). Furthermore, because of their proximity to human population centers, urban green spaces conserve the experience of nature and biodiversity (Pyle 1993; Miller 2005), including wildlife (Belaire et al. 2015; McCance et al 2017), for large numbers of people, with important implications for quality of life (Bertram and Rehdanz 2015; Cameron et al. 2020), environmental justice (Nesbitt et al. 2019; Schell et al. 2020), and the future of conservation (Stokes 2006; Soga and Gaston 2016; Hughes et al. 2018).

To maximize the conservation potential of urban green spaces, thorough understanding of their value as refugia for native biodiversity and the impact of urbanization on that value is needed (Knapp et al. 2021). Importantly, this understanding must correspond to the scales at which urban land-use planning and green space conservation and management occur. As the legal, regulatory, and cultural frameworks guiding urban development and green space protection and management—e.g., land-use decisions, ordinances, regulations, community action—primarily occur at the local (city, county, or other local jurisdiction) level (Lawrence 2005; Azerrad and Nilon 2006; Miller et al. 2009), fine-scale, local, and accessible understanding of biodiversity impacts of different degrees, forms, and extents of urbanization will be important in informing these planning and management processes (Stokes et al. 2010; Molina-Holgado et al. 2020; Allred et al. 2021). Relevant questions include: Are there meaningful differences in the habitat value of urban green spaces at the scales at which they vary within a metropolitan area? What is the range of species responses to the kinds of habitat changes that attend ongoing urbanization? Are there easily monitored species that can serve as indicators (Crooks 2002; McKenzie et al. 2018; Lopucki et al. 2019) of habitat value of a municipality's green spaces? Can we identify a regional standard for urban green space habitat quality, i.e., an urban green space reference site (Klaus and Kiehl 2021).

We addressed these questions through a case study using camera-trap data to compare mammalian wildlife diversity in two substantial Seattle-area urban green spaces that are similar in location, vegetation, topography, and anthropogenic history and use, but differ in degree of urbanization-related habitat alteration of the types expected to directly impact biodiversity (McDonald et al. 2020): patch size, fragmentation, and matrix quality. St. Edward State Park (SESP) is a large (128 ha; in the top 1% of urban Seattle-area parks in size), predominantly native forest habitat patch that is relatively unfragmented and bordered by a mix of developed and undeveloped lands. Nearby Juanita Woodlands Park (JWP), has similar habitat, but is smaller (15 ha; top 10% of urban Seattle-area parks in size), more fragmented, and is located in a matrix that is more completely converted to anthropogenic uses (Fig. 1, Table 1).

If the differences between these two sites are meaningful for wildlife, we expected to find greater presence and diversity of native mammal species in SESP. As some species are less tolerant of the impacts of urbanization than others (McKinney 2002; Eakin et al. 2018; Rodriguez et al. 2021), we also expected greater presence of less tolerant species ("urban avoiders," *sensu* Blair 1996) at SESP, while more tolerant species, "urban adapters," may not differ between the two green spaces, and species dependent on urbanized conditions, "urban exploiters," and non-native invasive species may be more common at JWP. This variation in tolerance of urbanization may produce differences in the species

composition of the wildlife communities of the two parks, including differences in the mammalian predator guilds (Bateman and Fleming 2012), with possible resulting impacts on ecological function (Crooks and Soulé 1999; Fischer et al. 2012; Jones et al. 2016).

By comparing the mammalian fauna of these two parks we hoped to shed light on the scale and nature of differences in habitat value of green spaces as they exist in an urban area. Faunal differences between sites, accessibly communicated, can inform efforts to reduce negative impacts of urbanization and to better conserve wildlife in urban settings.

Study sites

St. Edward State Park (SESP) and Juanita Woodlands Park (JWP) are large, mostly forested wildland parks located 2.1 km apart in the Seattle metropolitan area cities of Kenmore and Kirkland respectively (Fig. 1). Both parks are situated in a present-day landscape dominated by suburban residential and commercial land uses, including heavily trafficked arterial roads (City of Kirkland 2020), as well as other forested green spaces. The pre-settlement vegetation of both locations was native forest, which was cleared in the late 1800s (Harvey 1992; NPS 2006; City of Kirkland 2018). Following clearing, both sites appear to have undergone natural succession, resulting in a maturing, largely native forest typical in structure and species composition of maturing native forests of the western hemlock zone of lowland western Washington and Oregon (Franklin and Dyrness 1988). The two parks include both upland habitats and riparian habitats along small perennial streams.

Both parks receive high levels of use by the public (WSP 2020; Stokes 2021). The forested areas of both parks are passively managed for dispersed recreation, with anthropogenic alterations mainly consisting of official and unofficial recreational trails. Human visitation rates in forested areas of both parks range from high in heavily trailed areas to very low in less accessible areas (Stokes et al. 2020; Stokes 2021).

While the two parks are similar ecologically, they contrast sharply in the degree to which they are subject to the impacts of urbanization on habitat size, habitat fragmentation, and matrix quality (Table 1). Both parks are substantially larger than the mean size of parks in the urbanized Seattle area within King County ($\bar{x}$ = 6.2 ha, SD = 17.4, n = 773; King County 2021), but SESP is more than eight times larger than JWP. Two major arterial roads traverse JWP, fragmenting the park into a larger number of smaller patches and greater edge-to-area ratio than SESP, which has only one major road crossing one corner of the park (Fig. 1). The landscape matrix surrounding SESP is characterized by more forest and lower road and residential lot densities than JWP (Fig. 1, Table 1). Thus, by all of these measures, JWP is the more urbanized of the two parks.

Methods

We used unbaited camera traps to record mammal activity in SESP and JWP in two major habitat types of each park: riparian mixed forest and upland conifer forest. Comparison camera sites were chosen for similarity of vegetation type and microsite configuration (camera height, focal distance, microhabitat

type). Riparian sites were located along a small perennial creek in each park, and upland sites were widely distributed on minor unofficial trails in upland coniferous forest.

To control for possible seasonal differences in animal activity, each habitat type in the two parks was sampled on the same dates, with equal sampling effort in the same habitat type of each park. Riparian sites were sampled from April 2, 2019 – January 9, 2020 and from February 12 – June 22, 2020, a sample period of 410 days. Upland sites were sampled from April 8 – September 11, 2019 and February 4 – June 22, 2020, a sample period of 296 days. We checked cameras and downloaded data at 3-to-6-week intervals. Number and location of camera sites varied over the sample period due to interruptions in camera function resulting from camera failure, storm-caused blockage of camera view, and vandalism. For each of the two habitat types, number of active camera sites was the same in both parks on all sample days, with three (337 days) or two (73 days) active camera sites in riparian habitats, for a total of 1157 camera-days of sampling; and four (206 days) or three (90 days) active camera sites in upland habitats, for a total of 1094 camera-days.

We used three similar and widely available camera models: Bushnell Trophy Cam HD (model # 119537), Bushnell Trophy Cam HD Essential E3 (model # 119837C), and Browning Dark Ops HD Pro (model BTC-6HDP). The cameras are triggered by movement of a warm object (e.g., mammal or bird). They take color photographs in the daytime, and black and white photographs, using an infrared flash, at night. All camera types have similar trigger (detection) speeds (0.2 ± 0.1 second; Bushnell 2013, 2017; Browning 2018) and detection ranges (80 – 100 feet; Trailcampro 2019), a distance much greater than the distance at which the cameras in this study were aimed (< 10 m). Images recorded included a time and date stamp.

All cameras were set to take a burst of three photographs when triggered, and to delay for 5 seconds between triggers. Cameras were strapped to trees at a height of 1 – 1.8 m, depending local topography, and aimed at the ground at a distance of 4 – 8 m. Test deployments of paired Browning and Bushnell cameras as configured and deployed in this study indicate that detection rates of mammals larger than deer mouse *(Peromyscus spp.)* did not differ significantly between camera types, and weekly habitat occupancy, the primary response variable used in this study (see below), was the same for the two camera models (Stokes, unpublished data).

Analysis

To ensure thorough and consistent review of the photographic data and identification of species, all images recorded by the study cameras were independently visually inspected by two people, first by Samuelson or a senior wildlife seminar student, and second by Stokes. All photographs of the same species within 30 minutes of another photograph of the same species at the same site were considered the same detection. We assigned reliability ratings to species identifications as follows:

“Certain” image is clearly identifiable to species; there is no other reasonable possibility

“Highly probable” image is almost certainly the species but there is minor ambiguity

“Possible” image appears to be the species, but could reasonably be a different species

“Unknown” an animal is present, but the image provides no reliable indication as to species

We used only “certain” and “highly probable” species detections (91% of all potential wildlife detections; n = 3993) in our analysis.

To minimize bias resulting from spatial and temporal autocorrelation (Sollmann 2018), we used habitat occupancy as our primary response variable as follows: We divided the overall sample period into one-week periods (60 and 43 weeks for riparian and upland samples respectively), and considered a reliable (“certain” or “probable”) detection of a species by any of the cameras in a single habitat type and park (e.g., by any of the cameras in riparian habitat in SESP) to indicate the presence of the species in that habitat in that park for the week in which it occurred. We tested for significant difference in species habitat occupancy of the same habitat type in the two parks using chi-square tests for difference in number of one-week periods each species was detected, using the Holm–Bonferroni correction for multiple tests (Holm 1979).

Results

Camera trapping with equal sampling effort in the two study sites documented 10 native mammal species in SESP, and 8 native mammal species in JWP (Table 2). We detected mountain beaver (*Aplodontia rufa*), Douglas squirrel (*Tamiasciurus douglasii*), deer mouse, raccoon (*Procyon lotor*), long-tailed weasel (*Mustela frenata*), coyote (*Canis latrans*), bobcat (*Lynx rufus*), and black-tailed deer (*Odocoileus hemionus*) in both parks. Townsend’s chipmunk (*Neotamias townsendii*) and flying squirrel (*Glaucomys spp.*) were detected only in SESP. In the riparian habitat sample, of the 8 native species detected, Douglas squirrel was only recorded in SESP. In the upland habitat sample, Townsend’s chipmunk, flying squirrel, and bobcat were only recorded in SESP.

Shannon diversity indices for native species using number of days a species was detected as a measure of abundance were markedly higher in SESP than JWP, with effective species number (Jost 2007) 43% and 95% greater in SESP in riparian and upland habitats respectively (Table 2).

The same six non-native species were detected in both parks: eastern gray squirrel (*Sciurus carolinensis*), non-native rat (*Rattus spp.*), eastern cottontail rabbit (*Sylvilagus floridanus*), Virginia opossum (*Didelphis virginiana*), and domestic cat (*Felis catus*) and dog (*Canis lupus familiaris*) with no owner present. Most domestic dogs (with no owner visible) had visible collars and all were detected in daytime hours, suggesting that they were likely off-leash in the company of humans and were not feral animals inhabiting the parks. Collar presence could not be determined in most images of cats, all of which were detected after dark.

Occupancy of the two parks differed significantly for several species in both riparian and upland habitats (Fig. 2). In riparian habitats, bobcat, Douglas squirrel and mountain beaver had significantly higher occupancy levels in SESP, while coyote was significantly higher in JWP. In upland habitats, bobcat, Douglas squirrel and raccoon had higher occupancy levels in SESP, with mountain beaver marginally non-significant ($0.017 < p < 0.05$; Fig. 2b).

Occupancy patterns of non-native species were similar in the two parks, with only opossum occupancy in riparian habitats differing significantly (higher in SESP; Fig. 3). Both parks had high occupancy levels of eastern gray squirrel, eastern cottontail, and Virginia opossum, and low occupancy levels of non-native rat. Both domestic species, cat and dog, were more frequently recorded at JWP, but the difference was not significant.

Mammalian predators (raccoon, coyote, bobcat, opossum, long-tailed weasel, and cat) were more often present in SESP than JWP. The five predators detected in riparian habitats were present on average for 29 of the 60 sample weeks in SESP versus 25 of 60 weeks in JWP. During the 43-week sample period for upland habitat, predators were present for an average of 18 weeks on average in SESP versus 11 weeks in JWP.

Predator activity, as indicated by the sum of all sample days on which each predator species was detected, was also greater in SESP than in JW in both riparian (36% more total days of predator detections in SESP) and upland (12% more days of detections in SESP) habitats (Fig. 4). The species that constituted the predator communities of the two parks were similar; however, the relative activity of predator species, as indicated by number of days on which each species was detected, differed, with JWP characterized by greater activity of coyote and domestic cat, and less of raccoon and bobcat (Fig. 4). No bobcats were detected in the upland forest at JWP.

Discussion

The results of this case study illustrate the wildlife conservation potential and limitations of urban green spaces as they exist in a major metropolitan area. Both of our comparison green spaces supported substantial native mammalian diversity, including diverse native mesopredators: coyote, bobcat, raccoon, and long-tailed weasel. However, neither green space appeared to support the full array of native mammal species of the region (Ingles 1965), and we recorded no occurrences of any of the regionally widespread native apex predators—black bear (*Ursus americana*) and mountain lion (*Felis concolor*)—or largest herbivores—e.g., elk (*Cervus elaphus*). Additional widespread native mammals—e.g., striped skunk (*Mephitis mephitis)*—also appeared to be absent. Thus, while urban green spaces can support substantial diversity, even a large, relatively unfragmented urban green space such as SESP is likely to support only a subset of the regional species pool (Aronson et al. 2016; Lerman et al. 2021).

These results also demonstrate the differences in wildlife value of relatively large urban green spaces subject to different levels of urbanization impact. Our cameras detected a greater diversity of native mammal species, including a more diverse and robust native predator guild and significantly higher levels

of occupancy of several native species in SESP, which is larger, less fragmented, and situated amid more wildlife-friendly land uses than JWP. These differences were observed in both upland and riparian forest habitats. As these patterns did not obtain for the largest and most conspicuous species–coyote and deer—the wildlife impact of urbanization may not be apparent to the public and government officials charged with land-use planning and green space management.

Unlike native species, non-native species occupancy levels were largely similar in the two parks (Fig. 3). The sole exception was Virginia opossum, which showed higher riparian habitat occupancy at SESP, although its upland habitat occupancy did not differ significantly between parks. Opossum, eastern gray squirrel and cottontail rabbit activity levels were high in both parks. These results suggest that the range in urbanization of our two study sites had little influence on non-native species, and that some non-native mammal species are likely to be prominent faunal elements of even the largest and most ecologically intact urban green spaces.

Some native species were more sensitive to the impacts of urbanization than others. All native rodent species were more frequently detected at SESP, with Douglas squirrel and mountain beaver relatively common in SESP, but rarely present in JWP, and Townsend's chipmunk and flying squirrel, detected at low levels in SESP and never detected in JWP. This pattern contrasts with the non-native eastern gray squirrel's high occupancy levels in both habitat types in both parks. Eastern gray squirrel has proven to be highly invasive in a variety of locations where it has been introduced (e.g., British Columbia, United Kingdom, Italy), occupying diverse habitats, including all but the most intensely urbanized sites (Bruemmer et al. 2000; Bonnington et al. 2014).

Among the native mesopredators, bobcat occupancy was much greater in SESP, while coyote and raccoon occupancy patterns were mixed, with no consistent differences between the parks. These results parallel findings elsewhere that bobcats avoid more urbanized sites (Ordeñana et al. 2010; Lesmeister et al. 2015) while coyotes (Gehrt et al. 2010) and raccoons (Prange et al. 2004) are more tolerant of urban conditions (Bateman and Fleming 2012; Rodriguez et al. 2021). Differential sensitivity to urbanization across species resulted in substantially different predator communities in the two parks, with JWP characterized by greater presence of coyote and domestic cat, and lower presence of bobcat. This suggests that beyond the absence of apex predators, the predator communities and predator-prey dynamics may differ substantially in more and less urbanized green spaces (Crooks 2002; Fisher et al. 2012; Smith et al. 2018; Rodriguez et al. 2021), with possible ecosystem impacts, e.g., through trophic cascades (Jones et al. 2016; Wilson et al. 2020).

This study did not seek to identify the specific aspects of urbanization that were most important in producing the wildlife differences we observed. However, our results suggest the importance of urban features that are barriers to animal movement, such as roads and other hostile land uses around green spaces (matrix). The two largest and most mobile native species we observed, deer and coyote, were prominent in both parks, and in some cases were more active in JWP, while smaller native species—all of the native rodent species—were less active in JWP. Studies elsewhere (Rico et al. 2007; McGregor et al.

2008) indicate that small animals may be particularly vulnerable to the barrier effects caused by roads and other features of urbanization. The smallest non-native predator species we recorded—the opossum—also had lower occupancy in JWP, perhaps reflecting its vulnerability to barrier effects and road mortality (Glista and DeVault 2008; Smith-Patten and Patten 2008). The reduced presence of bobcat in JWP also may be, at least in part, a result of barriers, since this species was present in the riparian habitat of JWP where it can travel along stream corridors, but was absent in uplands where access requires road crossings. Other studies have found that bobcats favor riparian habitats for movement (Hilty and Merenlender 2004), avoid roads (Poessel et al. 2014), and are susceptible to road-related mortality (Tigas et al. 2002).

Applications

These results have implications for urban green space planning, management, and conservation. First, they provide clear and readily understandable evidence of the value of urban green spaces as wildlife habitat, and they illustrate the greater conservation value of larger, less fragmented, and more favorably situated green spaces as they exist in an urban area. By providing a measure of the wildlife impact of greater and lesser levels of urbanization, the comparative results of studies such as this can be used by local officials and planners to present accessible, site-specific information to the public, allowing informed decision making that includes biodiversity values, which are often not available and therefore are only included in decision making processes in general terms or left out altogether (Miller et al. 2009; Wilkinson et al. 2013; Apfelbeck et al. 2020). This is particularly important given that widespread occurrence of a small number of conspicuous urban adapter species—e.g., coyote and deer—which may obscure habitat value differences of green spaces for the public. Greater public awareness of less visible wildlife, and the charismatic nature of some of those species—e.g., bobcat and flying squirrel—may foster greater community interest and support for biodiversity conservation in land-use decisions (Stokes et al. 2010).

As one of the largest and most ecologically intact green spaces in the Seattle area (Stokes et al. 2014), SESP may approximate a best-case situation for urban wildlife, and our results offer a possible standard against which the potential and performance of other urban green spaces in the Pacific Northwest region can be assessed. As this study shows, even the most favorable urban habitats (e.g., SESP) do not support all of a region’s native species (Lerman et al. 2021). The faunal diversity of SESP, as opposed to the complete native diversity of the region, may be an appropriate standard of evaluation of the wildlife conservation value of other urban green spaces. Thus, SESP may serve as a regional urban reference site (Klaus and Kiehl 2021) for forested urban green spaces. A similar urban reference site function may be served by particularly high quality urban green spaces in other regions.

Clearly, even large green spaces such as SESP and JWP are too small, by themselves, to support viable populations of many of the mammal species that occur there. For example, the average territory size of a single female bobcat in western Washington is more than 300 ha (Link 2007), an area larger than SESP. Even small animals such as Douglas squirrel (Steele 1999) and mountain beaver (Arjo et al. 2007) are

unlikely to occur at densities high enough to permit long-term viability of populations solely within the boundaries of SESP. This implies that the future persistence of these and similar species in green spaces depends on their continued access to and use of habitats beyond green space boundaries. To maintain wildlife presence in urban parks, land-use planning must address wildlife habitat value outside those parks. Given the negative impact of barriers on wildlife diversity suggested by our results, planning, design, and management of roads and other potential barriers to animal movement should receive particular attention.

The prominence of non-native mammals in a very large urban green space such as SESP highlights the importance of gaining a better understanding of the impact of those nonnatives on native species in urban green spaces. While the negative effects of domestic cats and dogs are well documented (e.g., Loss et al. 2013; Doherty et al. 2017), the impacts of eastern gray squirrel, eastern cottontail rabbit, and Virginia opossum on native fauna are less investigated (but see Bruemmer et al. 2000; Bonnington et al. 2014) and merit study, with the goal of informing appropriate management of these nonnatives.

Finally, our study identifies several species that may serve as practical indicators of native wildlife habitat quality of forested green spaces in urban areas of the Pacific Northwest. Possible indicator species are Douglas squirrel, bobcat, and mountain beaver, which are both sensitive to urbanization impacts and relatively easily monitored with camera traps. An additional potential indicator is the flying squirrel, a species primarily associated with mature forests in the Pacific Northwest (Holloway and Smith 2011; Smith 2012). We detected few flying squirrels; however, our camera deployments were not designed to maximize detections of this species. With cameras positioned appropriately—aimed at tree trunks and downed logs—(Boulerice and Van Fleet 2016; DS *unpubl. data*]), or using acoustic methods (Diggins et al. 2016), this species may be sufficiently detectable to serve as a useful indicator. Analogous indicator species (e.g., McKenzie et al. 2018) can be used to evaluate habitat value of urban green spaces in other regions with different species pools.

Declarations

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Competing interests

The authors have no relevant financial or non-financial interests to disclose.

Author contributions

Stokes conceived and designed the study. Material preparation, data collection and analysis were performed by both authors. The first draft of the manuscript was written by Stokes. Samuelson produced the map figure and did the GIS work for Table 1. Both authors contributed to revisions of the manuscript. Both authors read and approved the final manuscript.

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Tables

Tables 1 to 2 are available in the Supplementary Files section.

Figures

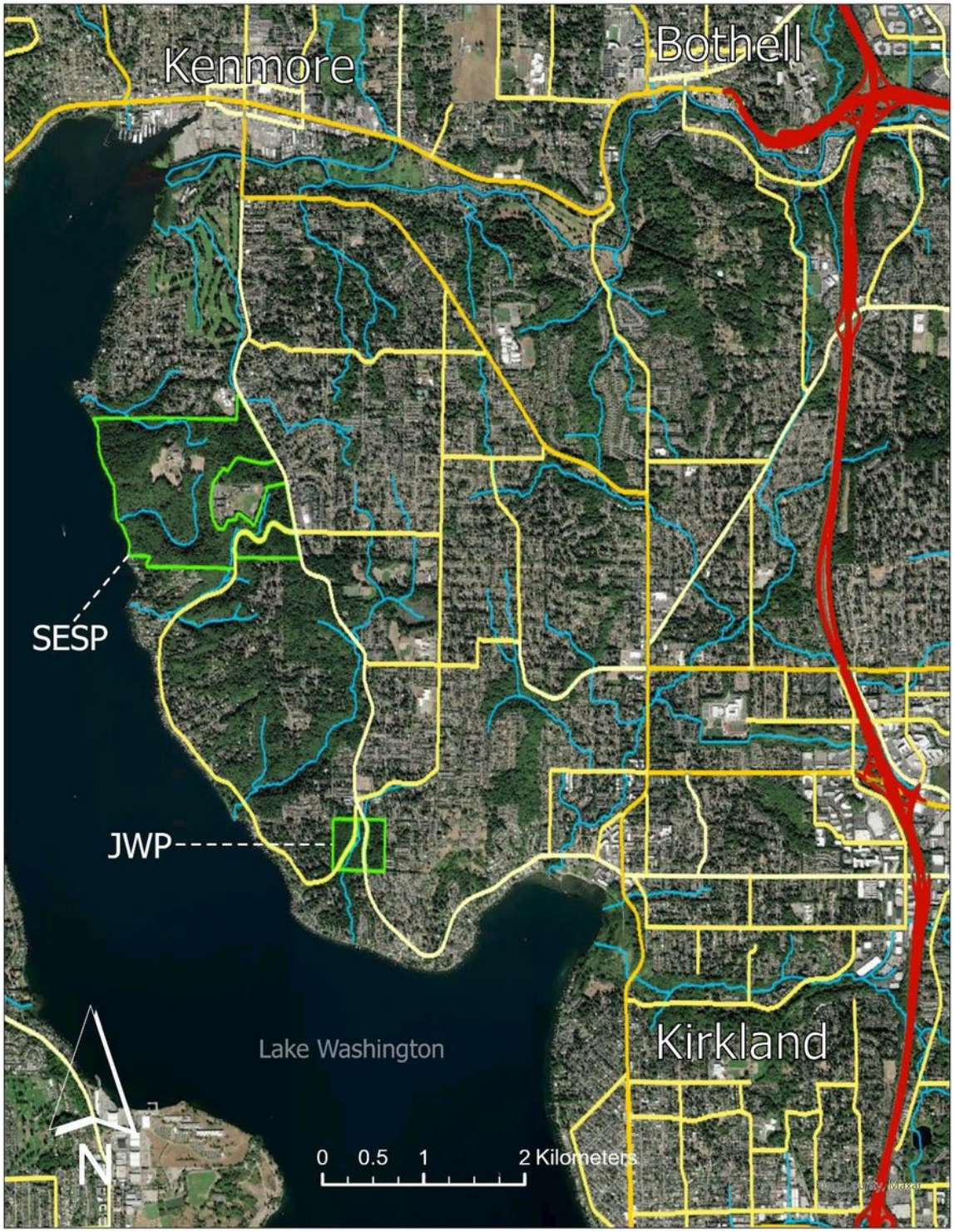

Figure 1

St. Edward State Park (SESP) and Juanita Woodlands Park (JWP) in the Seattle metropolitan area. SESP and JWP outlined in green. Suburban city centers of Kirkland (population 92,175), Kenmore (population 20,460), and Bothell (47,524) shown, along with arterial roads (yellow), interstate highway (red), and perennial streams (blue). Seattle (population 737,015) city center 15 km southwest of SESP. Aerial imagery and data from King County GIS Center (King County 2021)

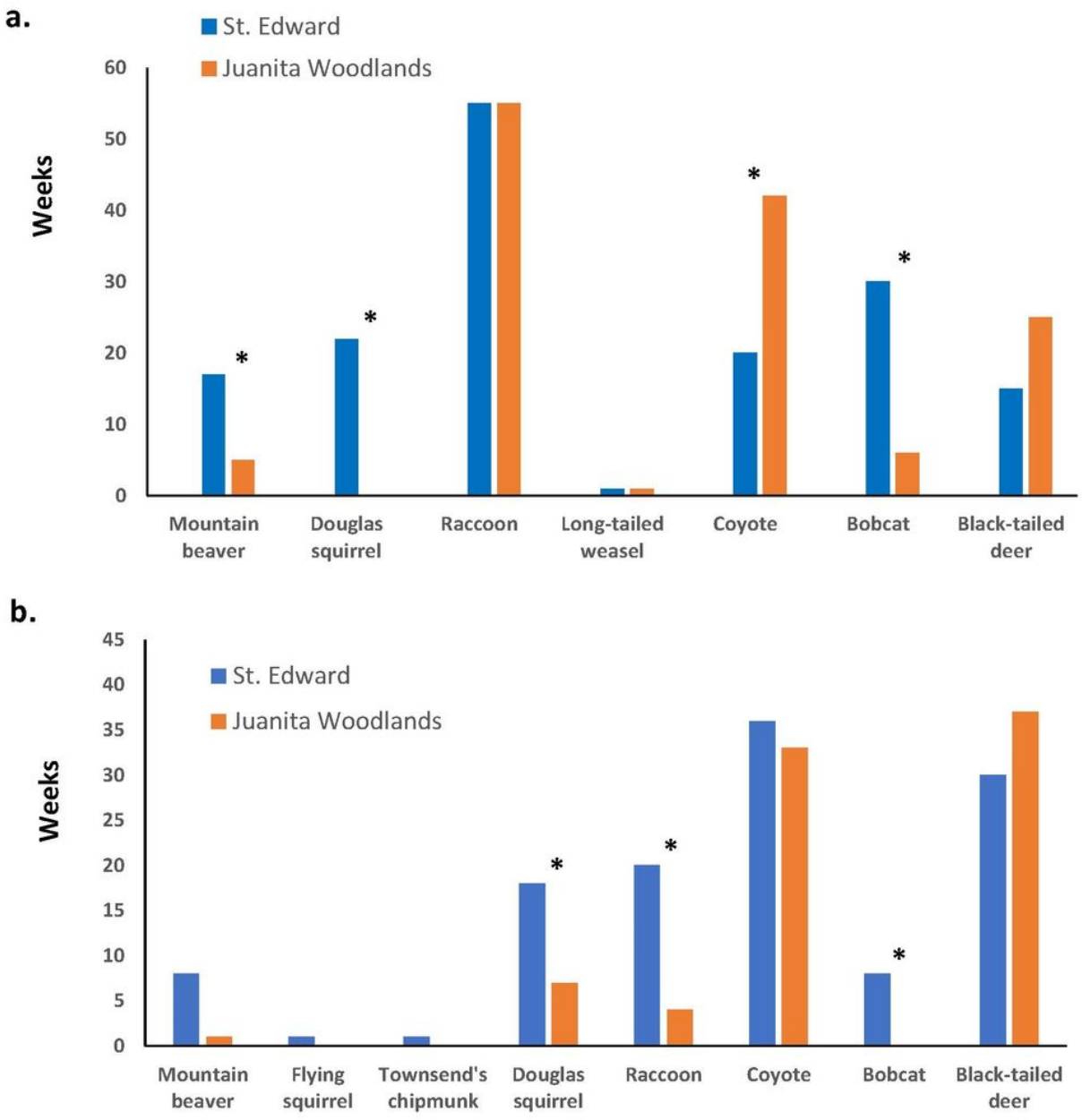

Figure 2

Occupancy of riparian (2a) and upland (2b) forest habitat by native mammal species in two urban greenspaces. Number of weeks species were recorded at study camera sites in SESP and JWP during sample period. **2a.** Riparian habitat: Sample period = 60 weeks. Equal numbers of cameras (2 or 3) in each park active on all sample days. 2**b**. Upland habitat: Sample period = 43 weeks. Equal numbers of cameras (3 or 4) in each park active on all sample days. Deer mouse, recorded at riparian sites in both parks, not shown because detectability of this species differed among sites. ***** indicates significant difference (< 0.05), chi-square test for all species with > 5 weeks total occupancy, Holm–Bonferroni correction for multiple tests

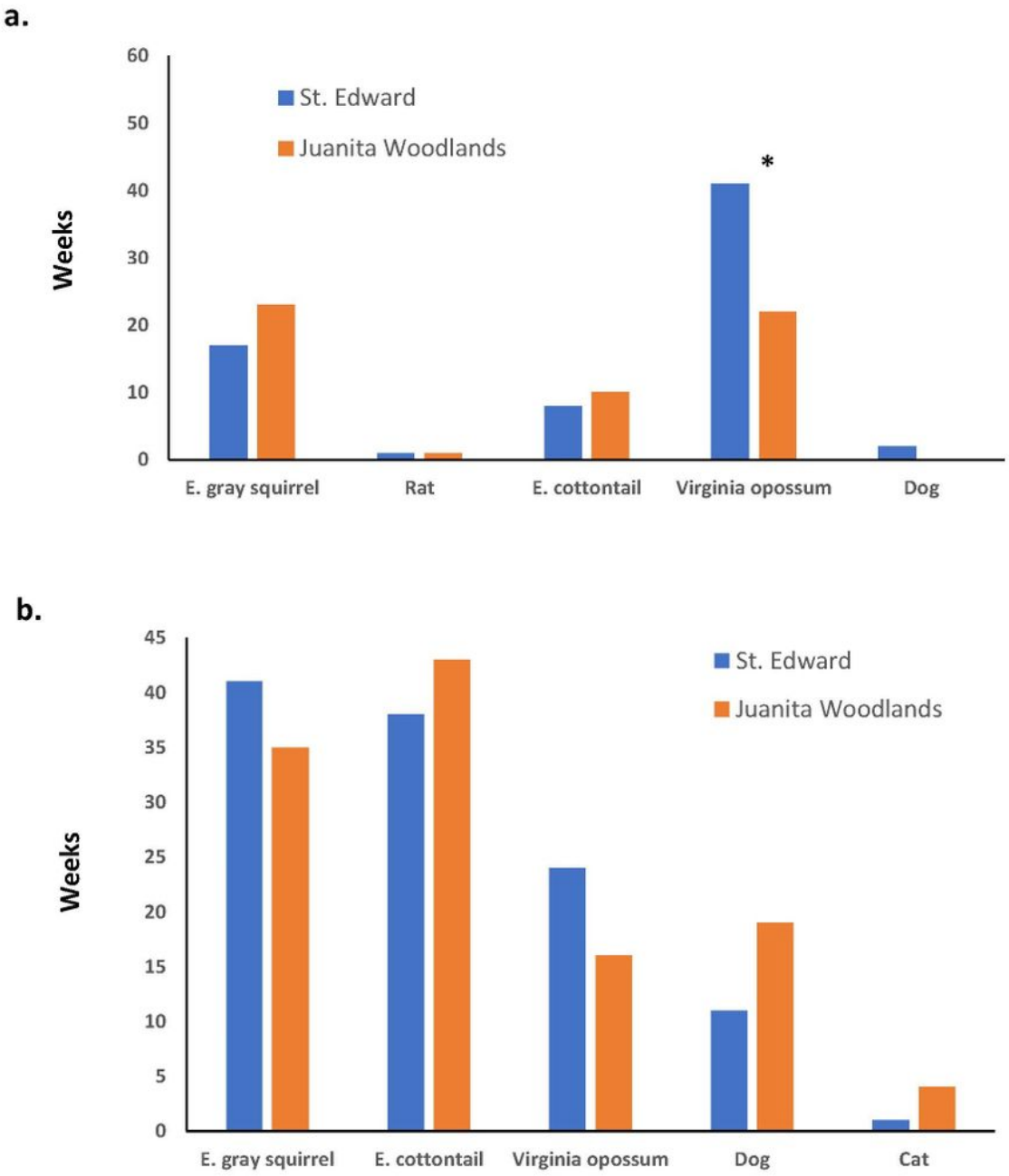

Figure 3

Occupancy of riparian (3a) and upland (3b) forest habitat by non-native mammal species in two urban greenspaces. Number of weeks species were recorded at study camera sites in SESP and JWP during sample period. **3a.** Riparian habitat: Sample period = 60 weeks. Equal numbers of cameras (2 or 3) in each park active on all sample days. **3b**. Upland habitat: Sample period = 43 weeks. Equal numbers of

cameras (3 or 4) in each park active on all sample days. * indicates significant difference (= 0.05), chi-square test for all species with > 5 weeks total occupancy, Holm–Bonferroni correction for multiple tests

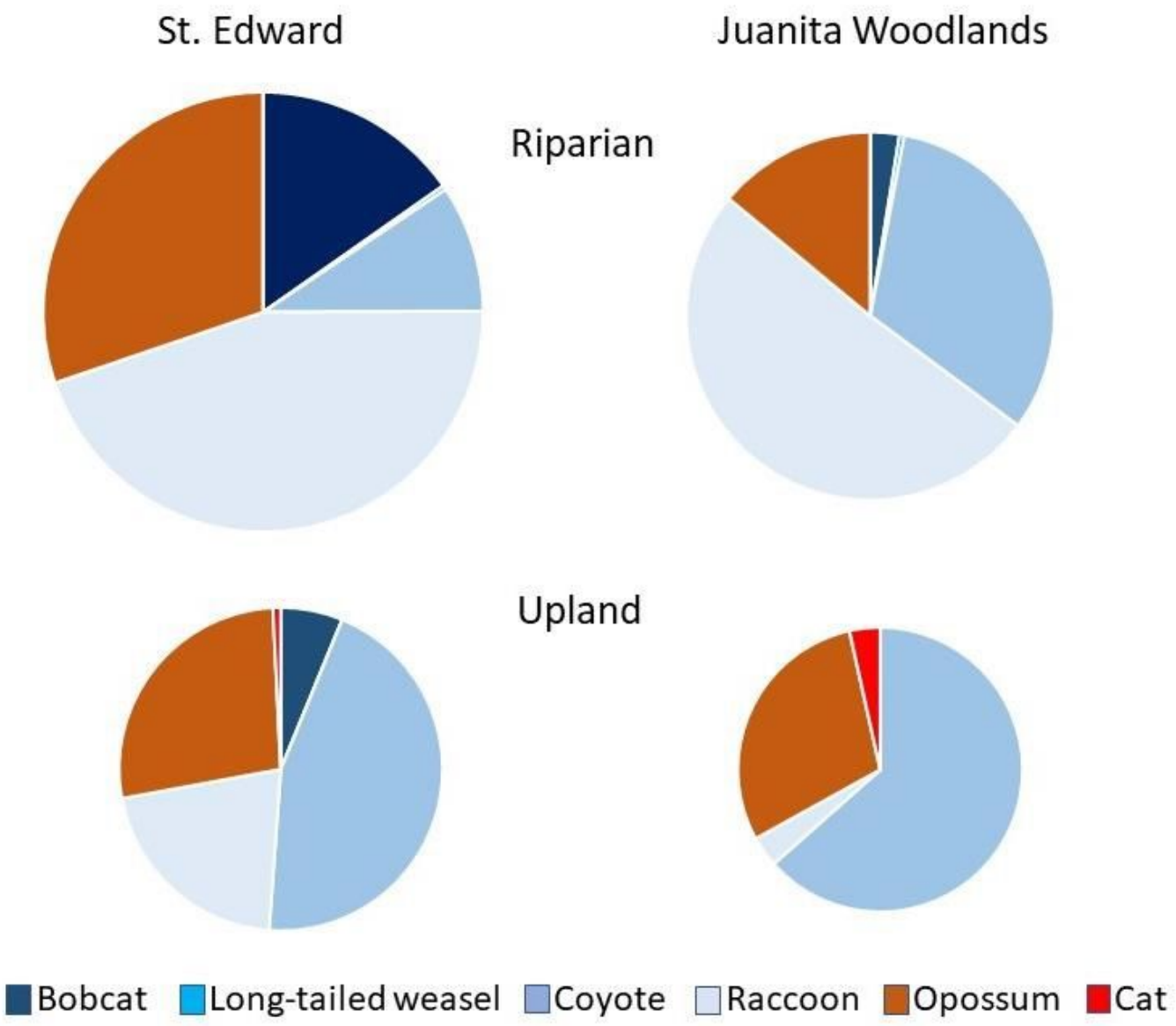

Figure 4

Relative activity of mammalian predator species in riparian and upland forest habitats in St. Edward and Juanita Woodlands parks. Activity measured in number of days a predator species was detected in each habitat in each park. Pie chart diameter proportional to sum of all predator activity/days sampled. Dogs are unlikely to be resident in the parks and were not included. Sum of days species were detected: SESP riparian: 321; JWP riparian: 236; SESP upland: 129; JWP upland: 115. Riparian sample period = 410 days; upland sample period = 296 days

Supplementary Files

This is a list of supplementary files associated with this preprint. Click to download.

- Table1.pdf
- Table2.pdf