

HABITAT FEATURES ASSOCIATED WITH BIRD COMMUNITIES OF PARKS IN
URBAN AND EXURBAN AREAS

A Thesis

Presented to the faculty of the Department of Biological Sciences
California State University, Sacramento

Submitted in partial satisfaction of
the requirements for the degree of

MASTER OF SCIENCE

in

Biological Sciences

(Ecology, Evolution and Conservation Biology)

by

Angela Rose Haas

SPRING
2018

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Abstract
of
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Bird species richness (BSR) is affected by vegetation composition and habitat structure at both local and landscape scales. Urban development plays a pivotal role in avian diversity by inducing changes in the landscape structure and available resources. However, it is unclear how bird communities are responding in areas undergoing urban development, particularly in the non-breeding season when fewer studies take place. I examined the avian communities at 17 parks in urban (within established urban matrix >50% built, n=8) and exurban (residential developments in the adjacent grassland 5-20% built, n=9) areas of Rancho Cordova in Sacramento County, California during the non-breeding season. I collected data on bird species and abundance, conducting five line-transect surveys per park (85 total surveys). I investigated the factors influencing avian assemblage, including local habitat features such as percent land cover, number of trees, fruiting trees and tree species, tree height and diameter at breast height (DBH), park size, park age; and landscape features including percent land cover within 500m of parks and distance to riparian habitat. Fifty unique bird species, including 15 migrants, were

observed in total. The BSR and avian abundance at urban and exurban parks was not significantly different, but community assemblages did differ significantly. Park area positively predicted BSR. Avian abundance was negatively predicted by park percent built cover, landscape percent water cover, and average number of fruiting trees. Species composition was associated with distance to river and landscape percent grass cover for all but one exurban park, and tree height, DBH and park and landscape percent tree cover for urban parks. The results suggest that in the non-breeding season, a variety of parks can support similar numbers, but different assemblages, of bird species.

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ACKNOWLEDGEMENTS

I would like to thank Dr. Jamie Kneitel and Dr. Timothy Davidson for their feedback and support, and Dr. Sara Kross for her feedback and expertise throughout this process. I would also like to thank Dr. James Baxter and Dr. Robert Crawford for serving on my graduate committee, Dr. Patrick Foley for conversations on statistics, Cristina James with the CRPD, the Albert Delisle family for a supporting scholarship, Jon Martindill for endless patience and kindness, Emily Pavia for invaluable feedback, and friends and family for their ongoing support as well.

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INTRODUCTION

Urbanization poses a great threat to biodiversity worldwide (Aronson et al. 2014, Leveau and Leveau 2016, Aronson et al. 2017). Specifically, land transformation and the introduction of exotic species have contributed to changes in biodiversity (Grimm et al. 2000) in every ecosystem on Earth (Vitousek et al. 1997). Cities are growing rapidly worldwide (Alberti 2005) as more people concentrate in urban areas (Shochat et al. 2006; Grimm et al. 2008). Simultaneously, suburban and exurban areas are spreading, impacting native species (Marzluff et al. 2001). In areas such as California's Central Valley where this study takes place, the spread and intensification of agriculture contributes to the adverse impacts on wildlife (Matchett and fleskes 2017). The resulting alterations in environmental structure and biotic interactions due to urbanization (Shochat et al. 2006) demand investigation (Chamberlain et al. 2009). Ecological research historically focused on more pristine environments (as noted by Blair 1996, Crooks et al. 2004, Grimm et al. 2008); however, there has been a surge in research on biotic communities in urban habitats (as reviewed by Pickett et al. 2011, also see the review by McDonnell and Hahs 2008). Despite a growing body of research, the loss of biodiversity due to urbanization remains poorly understood (Shochat et al. 2010), and the generally accepted effects of urbanization on species richness and abundance (e.g. Pickett et al. 2011) have been questioned (Saari et al. 2016).

Urbanization is defined as concentrated human presence in residential and industrial settings and its associated effects (Chace and Walsh 2004). Urban development fragments, isolates, and degrades natural habitats (Alberti 2005) while simplifying and

homogenizing species composition (Aronson et al. 2014). High levels of development reduce areas of primary productivity and increase exotic plant species (Blair 1996). Some of the highest extinction rates are driven by urbanization, as it often eliminates most of the native species (McKinney 2002). In the United States urbanization is a primary cause for species decline (Czech et al. 2000).

Urbanization plays a key role in species diversity of bird communities (Lee et al. 2004). Urban areas tend to have higher avian biomass and lower bird species richness (BSR) (Lancaster and Rees, 1979) than native environments (Emlen 1974, Beissinger and Osborne 1982, Aronson et al. 2014), and this is seen consistently across diverse habitat types (Chace and Walsh 2004). This biotic homogenization (Lockwood et al. 2000, Leveau et al. 2017) occurs because disturbance favors the spread of some species and the reduction of others (McKinney and Lockwood 1999), though these processes are scale-dependent (Leveau et al. 2017) and might not be apparent at the regional scale. Due to habitat changes (Marzluff 2005) and competitive exclusion (Shochat et al. 2010), the extinction of specialist and endemic species is promoted by urbanization, while higher proportions of invasive species and the dominance of few abundant species characterize urban areas (Blair 1996, Clergeau et al. 1998, Leveau et al. 2017). Specifically, the dominant species tend to be invasive (nonnative and usually introduced by humans) or synanthropic (native species highly associated with humans) (Shochat et al. 2010).

Factors associated with urban environments can adversely affect bird health and survival. Yet ultimately, urban assemblages are shaped by the differential responses of individual bird species (Jokimaki 1999, Crooks et al. 2004) or individual birds (Marzluff

1997) to development and habitat fragmentation. Species vary greatly in their responses to urbanization: most are unable to occupy new habitats created by urban development, while others persist or even increase in density because of their environmental tolerance (Rebolo-Ifran et al. 2015). For instance, traffic noise can alter grassland bird communities through avoidance (Chace and Walsh 2004) and artificial light can influence stopover sites used by migrants (McLaren et al. 2018).

Collision with man-made objects, competition, food supply, disease, and changes in the predator assemblage all influence avian survivorship in urban areas (Chace and Walsh 2004). Large buildings in regions of low urbanization result in high collision mortality rates in the US (Hager et al. 2017). Introduced predators such as domestic cats (*Felis catus*) are especially damaging (Marzluff 1997, Baker et al. 2005), hunting 1.4–3.7 billion US birds annually (Loss et al. 2013). A recent meta-analysis of 47 studies conducted on different continents, including Europe, North America and Australia, compared urban and non-urban bird populations: a consistent pattern of earlier lay dates, lower clutch size, lower nestling weight and lower productivity per nesting attempt was found across species in urban landscapes (Chamberlain et al. 2009).

Conversely, some species respond positively to food sources provided directly or indirectly by humans (Emlen 1974), such as bird feeders, spilled waste and waste collection centers (Marzluff 1997). Feeder use varies among individual birds and species, and so can have asymmetrical outcomes. For instance, in New Zealand, introduced species dominated this resource, suggesting a negative impact for native species conservation (Galbraith et al. 2017).

Some species also respond positively to increased habitat features, such as perch heights and nesting surfaces (Emlen 1974), or perhaps more importantly, holes and crevices (Lancaster and Rees 1979). The increase in habitat features due to urban development can increase spatial environmental heterogeneity (EH). Stein et al. (2014) conducted a meta-analysis of 1148 data points from 192 studies and provided robust evidence for a positive heterogeneity-richness relationship across taxa, biomes and spatial scales. Though not explicitly acknowledged in the literature, urbanization and environmental heterogeneity are not necessarily discrete: the complex mosaic of disturbance and land transformation caused by urban development (Blair 2004, Pautasso 2007) can generate habitat heterogeneity for biotic communities to respond to (McKinney 2008). This can also be scale dependent, depending on how species interact with their environment. In general, higher plant diversity means a more diverse array of resources for food; and higher structural complexity of vegetation should provide more microhabitats in which to take shelter, rest, forage, nest and breed (Lawton 1983, Marzluff and Ewing 2001). Furthermore, habitat heterogeneity may allow for greater coexistence between prolific urban species and rarer native species (Shochat et al. 2004).

High species diversity in areas where urban and rural habitat features intermix is often discussed in the context of the intermediate disturbance hypothesis (Jokumaki and Suhonen 1993; Blair 1996, 1999; Clergeau et al. 2001; Crooks et al. 2004). While this framework may be helpful, the concept of environmental heterogeneity applied to the same areas can be quantified in relation to bird assemblages in a meaningful way. For example, it has long been held that habitat diversity, as measured by

foliage height profile, is a good indicator of bird species diversity in North America (MacArthur & MacArthur 1961, 1964; Recher 1969). Additionally, birds respond to measurable habitat features, such as type and structure of vegetation (White et al., 2005). Trees and other plants layers were positively correlated with high bird diversity in urban open areas in the U.S (Mason et al. 2007) and species richness and abundance of migrating birds in Mexico (MacGregor-Fors et al. 2010). Tree size and species richness along streetscapes ameliorated the effects of urbanization by increasing BSR in Brazil (de Castro Pena et al. 2017). Water bodies and age of woody vegetation were beneficial to BSR across several studies in a 62-study review (Nielson et al. 2013).

Both local- and landscape-scale habitat factors play a role in shaping urban bird communities. Yet, in studies of urban park BSR, measures of the adjacent landscape are often absent. Local scale habitat diversity, within the area surveyed for bird-diversity (per Smith et al. 2014), has been well-studied, demonstrating a positive relationship with species richness (as reviewed by Nielson et al. 2013). Additionally, the landscape factors surrounding a local area have significant impacts on bird populations within that local area (Savard et al. 2000), and in urban areas this means the presence of suitable habitat does not guarantee particular species will be observed (Litteral and Shochat 2017). A study that measured the landscape structure surrounding urban parks found that the adjacent habitat was related to both BSR and individual species occurrences within parks (Jokimaki 1999). In other studies, landscape structure had only a small effect compared to other variables, such as park area (Murgui 2007), or was only more pronounced in young parks, which are colonized mainly from the regional species pool (Fernandez-

Juricic 2000). Interestingly, one study found the level of urbanization in the surrounding area to be more important than site size and plant structure in determining bird community composition (Huste & Boulinier 2011).

A basic challenge for conservation is to understand how urbanization affects biodiversity (McKinney 2002). Within the fragmented landscapes created by urban development, urban parks are refuges for many bird species (Fernandez-Juricic and Jokimaki 2001). They show higher bird species diversity and richness than other urban areas (Jokimaki 1999, Nielson et al. 2013), highlighting their potentially integral role in maintaining (Savard et al. 2000) and promoting (Nielson et al. 2013) biodiversity in urban landscapes. Because this area of study generally focuses on the conservation and enhancement of biodiversity in urban spaces, bird species richness (as reviewed by McKinney 2008 and Nielson et al. 2013), and the abundance of each species, are used as response variables. These measurements provide quantitative estimates for how many individuals of how many species utilize parks and can be used to inform conservation strategy and urban design.

Although urban parks have been well-studied within cities and city edges (Jokimaki 1999, Fernandez-Juricic 2000, Clergeau et al. 2001, Murgui 2007), exurban areas make up a small portion of the literature (Marzluff et al. 2001). According to Marzluff et al. (2001), exurban is the preferred terminology to describe city-adjacent areas that have approximately 5-20% built area and are surrounded by a natural matrix (as opposed to an agricultural matrix, which distinguishes rural areas). The term

“exurban” will therefore be used to describe the parks that have recently been built within the natural grassland matrix, as they meet these parameters.

Many of the world’s migratory bird populations are in alarming decline (McLaren et al. 2018). Studies of urban parks in North America often take place during or analyze data from the breeding season (Marzluff et al. 2001; also see Blair 1996, 2001; Crooks et al. 2004; Chocron et al. 2015), while the influence of urbanization on winter bird use is not as well understood (Smith 2007). Accordingly, fall migrant and overwintering species use of parks isn’t always considered (although see Carbo and Ramirez 2012 for their work in Mexico); yet, neotropical and other long-distance migrants are particularly sensitive to urbanization (Zhou and Chu 2012, Litteral and Shochat 2017). Further, parks within areas currently undergoing urban development have been largely overlooked (but see White et al. 2005 for their study in Australia). It will therefore be instructive to investigate the avian assemblages of parks within recently built areas during the non-breeding season.

The Central Valley of California has been subject to a growing human population, and the associated expansion and intensification of agriculture and urban development have impacted wildlife habitats (Marchett and Fleskes 2017). Sacramento County is likewise experiencing urban growth and an increase in population (USCB 2016). Urban development can be expected to continue in the foreseeable future, making this a model system for how expanding urban areas may influence the composition of avian communities. Information on the bird communities that are shaped by human endeavors are imperative to avian conservation and biodiversity management. More avian

population, density, and diversity data is needed to analyze and forecast how urban populations affect bird conservation (Chamberlain et al. 2009). Urban spaces will continue to become necessary habitat for species as urban development and sprawl continues. If environmental thresholds exist for rare, threatened, or native species to persist, such as percent tree cover, those quantifiable measures can be included in habitat design of urban parks, greenways or other urban-adjacent habitats.

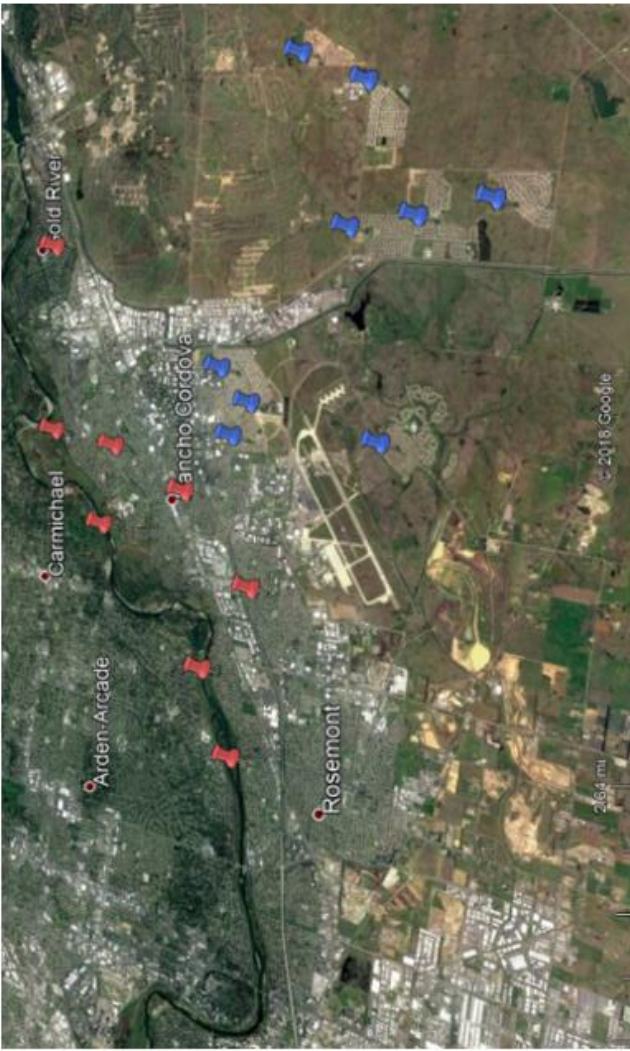
In this study, I assessed bird communities of parks in urban and exurban areas to investigate the role of park characteristics and adjacent habitat in shaping community assemblages during the non-breeding season, and to specifically investigate avian assemblages in an area undergoing transformation due to urban development. I conducted the study in Rancho Cordova, California because the city's growth pattern is not uncommon: houses and commercial buildings were constructed near the river and expanded outward over time, and the sprawl continues, with housing developments being built in the exurban grassland progressively further from the river. First, I looked for differences in the species richness and abundance of birds present at parks. I hypothesized that both species richness and abundance would be higher in the older, urban parks because they tend to have bigger trees and more habitat complexity and be surrounded by an urban matrix. I also expected the urban parks to have higher proportions of non-native species. I hypothesized that BSR and abundance would be lower at exurban parks where trees tend to be recently planted and the surrounding matrix is exurban. I also expected the community composition of parks to differ. I hypothesized that the exurban parks would have higher proportions of native species, and for the of the

urban parks to include more invasive and synanthropic species. I looked for associations between local and landscape habitat features and the bird species richness and abundance at parks. Finally, I explored the species assemblages at young versus mature parks to parse differences in species present at parks located in urban versus exurban areas during the non-breeding season.

METHODS

Study Sites

I collected field data at 17 parks in Sacramento County, CA within the City of Rancho Cordova or adjacent census-designated areas (Figure 1). The parks are all between 0-250 feet in elevation. Within the urban environment, parks are older, were built near the American River, and tend to have more mature trees and complex vegetation (Table 1). Parks in exurban areas undergoing residential development tend to be more recently built, and have fewer, smaller trees and less overall vegetation, as well as be located further from the river (Table 1). The locations of the urban and exurban parks share many similarities that helped control for spatial aggregation between the two groups, including similar elevation, temperature and rainfall. Exurban parks were often bordered on one side by grassland habitat, and the exurban matrix is primarily grassland with housing developments, a few commercial developments, a shooting range and a golf course intermixed. The most recent park was built in 2016, one year before the study took place, and construction in the study area is ongoing: during surveys, construction on houses or empty lots sometimes occurred within the development where an exurban park was located, or in an adjacent plot. The exurban area adjacent to Rancho Cordova that remains undeveloped or is undergoing urban development is primarily grassland, and some seasonal wetlands. This habitat is home to several bird species of conservation interest, including Swainson's hawk (*Buteo swainsoni*, California-threatened), short-eared owl (*Asio flammeus*, a CDFW species of special concern), loggerhead shrike



Sacramento
County, CA
From: Wikipedia



Figure 1. Site Map (red = urban parks (8), blue = exurban parks (9))

Table 1. Mean (\pm standard deviation) habitat characteristics for exurban (n=9) versus urban (n=8) parks
Transect Characteristics (50m)

Park Type	Fruiting Trees (#)	Tree spp. (#)	Trees (#)	Tree DBH (cm)	Tree Height (m)		
Exurban	1.02 \pm 1.36	2.11 \pm 1.45	7.22 \pm 6.01	12.85 \pm 12.58	5.51 \pm 3.47		
Urban	1.33 \pm 2.15	2.75 \pm 0.97	7.38 \pm 4.33	50.34 \pm 23.82	10.70 \pm 2.4		
Park Characteristics							
	Park Age (yrs)	Park Area (km ²)	Park % Tree	Park % Grass	Park % B. Ground	Park % Water	
Exurban	8.22 \pm 4.7	30.79 \pm 24.23	9.27 \pm 8.91	67.72 \pm 13.63	3.49 \pm 7.0	1.51 \pm 4.2	
Urban	43.13 \pm 11.5	69.68 \pm 72.44	29.39 \pm 10.2	55.31 \pm 12.17	1.43 \pm 2.04	0.52 \pm 1.3	
	Fruiting Trees (#)	Human Activity (ped/min)	Dogs (dogs/min)	Park % Built Cover			
Exurban	10.3 \pm 13.6	0.18 \pm 0.11	0.08 \pm 0.09	18.01 \pm 3.94			
Urban	3.7 \pm 4.9	0.23 \pm 0.19	0.11 \pm 0.10	13.36 \pm 8.73			
Landscape Characteristics (500m)							
	Land. % Tree	Land. % Grass	Land. % Bare Ground	Land. % Built	Land. % Water	Dist. to River (m)	
Exurban	5.54 \pm 3.89	44.1 \pm 12.11	7.47 \pm 4.81	40.4 \pm 11.45	0.08 \pm 0.17	6803 \pm 2174	
Urban	22.5 \pm 3.46	22.32 \pm 6.24	3.77 \pm 3.19	44.19 \pm 8.32	3.86 \pm 4.91	919.5 \pm 711.8	

(*Lanius ludovicianus*, CDFW species of special concern and USFWS bird of management concern) and tricolored blackbird (*Agelaius tricolor*, CDFW species of special concern and USFWS bird of management concern) (CDFW, 2017). A 3,828-acre site including hundreds of acres of grassland and seasonal wetland habitat has been slotted for development, which has begun, resulting in some of the newest parks in this study.

Data Collection

Bird Surveys

Single-year and single-visit surveys have been deemed appropriate for predicting bird abundances and species richness in urban spaces (Jokumaki and Suhonen 1998, Jokimäki & Kaisanlahti-Jokimäki 2003). Further, counts can be repeated if good estimates of the community of specific areas are desired (Ralph et al. 1992). I therefore performed five surveys at each park (e.g. Clergeau et al. 2001, Crooks et al. 2004) for a total of 85 surveys. Surveys were conducted once or twice weekly from September-November 2017, between sunrise and 11:00A.M. (Fernandez-Juricic 2000) to coincide with peak bird activity and singing. Parks were visited throughout the season at different times of day to avoid confounding day or time effects (Appendix A). Surveys were not conducted during rain or strong winds (around 12 km/hr per Ralph et al. 1995) or when parks were being watered or mowed. Birds were identified to species using Monarch 500 8x32 binoculars and Sibley's Field Guide to Birds of Western North America (2003) and assigned resident or migrant status based on their range map information.

The line-transect survey method was used to count birds (e.g. White et al. 2005). Transects were 50m long to: standardize transect length and area surveyed at all parks and randomize the direction each transect started from. A random number generator was used to determine the side of the park (1=N, 2=S, 3=E, 4=W) the transect started from, and the same transect was walked for each of the five surveys. Surveys for lasted five minutes and the timer app of the Apple iPhone 6 was used. Each individual bird seen or heard was identified to species and recorded. Data collected from within a 50-m radius can be used for among-species comparisons of abundance (Ralph et al., 1995); therefore, distance (<50m, >50m) was recorded. Care was taken to avoid counting the same individual twice (initial visual/aural detection recorded; if a bird of the same species flew from the same area subsequently during the survey, it was not counted). Birds that flew over but not within the habitat were labeled as flyovers. Behavior was opportunistically recorded.

Bird species were recorded using the appropriate four-letter Bird Banding Laboratory code. Birds that were not identified to species were recorded as unidentified (e.g. UNHU = “unidentified hummingbird”). A Garmin GPSMAP 64 unit was used to take geographic coordinates (UTMs) and elevations of each transect. On a Transect Survey Data Sheet, the following was recorded: bird species, abundance and distance; date, time, park name, UTM coordinates at the start and end of the transect, elevation, temperature, GPS waypoint numbers, and notes on behavior or other pertinent information.

Habitat Measurements

It has been postulated that in North American cities, native bird species abundance is correlated with the density of vegetation, while introduced species abundance is related to urbanization level (Lancaster and Rees 1979). In studying avian communities, the proportions of vegetated open space versus human made structures (e.g. percentage of built area) have been used as surrogates for urbanization (Blair 1996, Clergeau et al. 1998, Lee et al. 2004). Further, area covered by pavement and area covered by buildings have been shown to be redundant measures in evaluating bird communities (Blair 1996) and can be combined into a single measure of “built environment”. According to Stein et al. (2014), plant diversity and land cover are probably suitable for capturing the habitat requirements of many species. Prodon and Lebreton (1981) proposed the community composition of avifauna within a single bioclimatic level is almost completely determined by vegetation structure. To capture some environmental heterogeneity due to vegetation structure, land cover, tree DBH and tree height were measured.

I collected habitat measurements within 50m of the transect and within the park, and at the broader landscape scale of 500m (e.g. Murgui 2007; Smith et al. 2014) since this is an appropriate scale for meaningfully measuring landscape heterogeneity (Litteral and Shochat 2017) (Appendix B). Within 50m on either side of the transect, number of trees, number of fruiting trees, number of tree species, tree height and tree DBH were recorded. For each measurement a mean value was calculated for each transect (e.g. Husté and Boulinier 2011) which is the mean reported for each park (Table 1). The area and percent land covers for each park, including percent grass, trees, bare ground, water

and built cover, were determined using aerial photographs and Google Earth Pro software. For the 500m surrounding habitat, iTree Canopy was used to estimate percent cover based on the classifications tree, grass/shrub, built, bare ground or water. iTree Canopy software is peer-reviewed and can be used to generate cover percentages that stabilize between 600-1000 randomly generated points with a maximum predicted standard error of <3.0% (Jacobs et al. 2014). Because my area of interest was a fraction of the size usually considered (1km² versus thousands), I used 400 randomly generated points just to be thorough, and all standard errors were <3.0%.

Shortest distance to a river was measured, as well as to the closest park, and to native grassland habitat (>5 ha grassland patch; the estimated area requirement of the Western meadowlark (Herkert, 1994); a common grassland species according to preliminary data) using Google Earth. Rate of visitors to a park can decrease BSR (Clergeau et al. 2001). To gauge human disturbance, I recorded the number of pedestrians that entered the survey area per minute, as well as number of cats and dogs. Park age was determined by land acquisition dates as provided by the Cordova Recreation and Park District; dates were corroborated using Google Earth (Riviera Park was established before the acquisition date: labeled as mature and approximately 26 years of age since established prior to 1993, the oldest available imagery from Google Earth).

Statistical Analysis

Species Richness & Abundance

Analyses were carried out on bird species richness (cumulative number of species seen during five surveys) and on avian abundance (the sum of the average number of

individuals of each species seen in each park during five surveys) (e.g. Crooks et al. 2004, White et al. 2005, Biadun and Zmihorski 2011). Parks were labeled as urban (located in the established urban matrix, >25 years of age) or exurban (recently built within the exurban matrix of grassland habitat, <20 years of age). Flyovers were excluded from analyses (Jokimaki 1999, Kross et al. 2012), as were unidentified species. No cats were seen in the study area so this variable was excluded. Statistical significance was considered achieved when $p < 0.05$.

To investigate whether bird species richness varied between urban and exurban parks, I ran a Two Sample t-test on total BSR between urban and exurban parks. A Two Sample t-test was also conducted to investigate whether maximum avian abundance differed between the park groups. Then, to investigate associations between the avian richness and abundance and habitat variables, I created two linear models using a Gaussian distribution: one with BSR, and one with avian abundance, as the response variable. Dependent variables were tested for normality using the Shapiro–Wilk test (e.g. Huste and Boulinier 2011). I used frequency histogram and qq plots to assess whether the data fit the assumptions of normality and homoscedasticity of residuals. Abundance and park area were log-transformed to meet these assumptions (e.g. Jokimaki 1999, Fernandez-Juricic 2000). The independent variables measured were: park area; park age; park percent grass, trees, bare ground, water and built cover; average number of fruiting trees, transect average number of trees, fruiting trees, and tree species; average tree DBH and tree height; average number of pedestrians and dogs; and landscape percent trees, grass, bare ground, water and built cover because these are ecologically relevant variables

shown to predict bird richness and or abundance in previous studies (Blair 1996, Bolger et al. 1997a, Jokimaki 1999, Fernandez-Juricic 2000, Fernandez-Juricic and Jokimaki 2001, Melles et al. 2003, White 2005, Murgui 2007, Biadun and Zmihorski 2011, Zhou and Chu 2012, Leveau and Leveau 2016). Distance to river was also included to account for the fact that many of the urban parks are clustered closer to the river than exurban parks, and riparian habitat could introduce different bird species (Sabo et al. 2005) to adjacent parks.

I used multiple linear regression to investigate what habitat features were associated with BSR and abundance. To meet the assumptions of multiple linear regression, I looked at the Cook's distance to check for outliers. Hagan Community Park has a larger area than all parks and had the highest BSR. However, no outliers were found (all Cook's distances were <3) so Hagan was included in all analyses. To avoid issues of multicollinearity, I looked at the relationships and correlation between variables using plots and correlations in R. Several variables were correlated using Pearson's coefficient: park percent water and park percent bare ground ($r=0.92$, $p<0.001$), tree height and tree DBH ($r=0.88$, $p<0.001$), transect average number of number of trees and tree species richness ($r=0.734$, $p<0.001$); landscape percent grass cover and landscape percent tree cover ($r=-0.837$, $p<0.001$), landscape percent grass cover and landscape percent built cover ($r=-0.637$, $p=0.003$), and number of pedestrians and number of dogs ($r=0.69$, $p=0.002$). In each case, the variable that seemed more biologically relevant and/or had a higher correlation with the dependent variables was selected; transect average number of number of trees, average tree height, park percent bare ground cover, landscape percent

grass cover and average number of dogs were removed. Park number of fruiting trees was used in place of transect number of fruiting trees since the park count always included the transect count and more information about the park. I then formally checked the remaining independent variables for multicollinearity: any with variance inflation factors (VIF) >3 were sequentially removed starting with the covariate with the highest VIF, VIFs were recalculated, and this process was repeated until all VIFs were smaller than 3, a stringent pre-selected threshold (Zuur et al. 2010).

I ran stepwise multiple regression (e.g. Pino et al. 2000, Crooks et al. 2004, Carbo-Ramirez and Zuria 2011) on the simplified linear models. The independent variables were: log park area, park percent built cover, park percent water cover, landscape percent built cover, landscape percent water cover, number of fruiting trees, average tree DBH and average number of pedestrians at parks. Stepwise regression accounts for the contribution each of the independent variables makes to explaining total variance, and thus demonstrates the relative importance of these variables in the regression (Pino et al. 2000). I used a non-automatic variable search (e.g. Carbo-Ramirez and Zuria 2011) using backwards elimination techniques and the Akaike information criterion (AIC) to select the most informative model (e.g. Biadun and Zmihorski 2011). The model residuals were examined for deviations from assumptions and none were observed. All analyses were run in R, version 3.4.3 (R Core Team 2017).

Community Composition

I conducted nonmetric multidimensional scaling (NMDS) to examine differences in avian composition between parks (e.g. Leveau and Leveau 2016). The dissimilarity

matrix was calculated using the Bray–Curtis distance metric (e.g. Kneitel 2014, Leveau and Leveau 2016) with species abundance data. Two dimensions were specified. Environmental variables were fitted onto ordination, and only significant variables ($p < 0.05$) were kept and plotted. This analysis was conducted in R, version 3.4.3 (R Core Team 2017).

Species densities were used to conduct a one-way analysis of similarity (ANOSIM) to test for differences in community composition between urban and exurban parks. Density was calculated as individual birds per hectare (e.g. Zhou and Chu 2012): using the abundance of birds across the five surveys within 50m of the transect. Significant ANOSIM results were followed by a Similarity of percentages (SIMPER) test, using Bray–Curtis dissimilarity to determine the relative contribution of each species to differences in community assemblages (Kneitel 2014) between park types. ANOSIM and SIMPER were conducted using PAST, version 3.19 (Hammer et al. 2001).

RESULTS

A total of 437 observations of 50 bird species were made during 5 surveys at 17 parks (excluding flyovers and unidentified species, and an additional seven species seen at parks outside of surveys) (Table 2). Thirty-five unique species were observed at exurban parks, and 36 at urban parks. Exurban ($x=11.56$, $sd=1.71$, $n=9$) and urban ($x=12.75$, $sd=2.63$, $n=8$) parks did not differ significantly in total number of species observed (Two Sample Student's t-test, $t=-1.0531$, $df=15$, $p=0.3089$) (Figure 2). Similarly, exurban ($x=22.56$, $sd=10.42$, $n=9$) and urban ($x=24.4$, $sd=16.6$, $n=8$) parks did not differ significantly in the abundance of birds present (Two Sample Student's t-test, $t=-0.043$, $df=15$, $p=0.967$) (Figure 3). Age and area of parks were not correlated ($r=0.35$, $p=0.16$). The average number of species observed per survey across all parks varied between 3.4-6.6 species (Figure 4).

In multiple regression models, the best linear model for BSR explained 81% of the variance in the data and included park age, landscape percent bare ground cover, landscape percent water cover, and park area ($R^2=0.81$, $p<0.001$) (Table 3). Park area was a significant, positive predictor of bird species richness ($t=6.84$, $p<0.001$). Although non-significant, landscape percent build cover was positive predictor of bird species richness ($t=1.33$, $p=0.207$). Average tree DBH ($t=-1.3$, $p=0.218$) and landscape percent water cover ($t=-1.69$, $p=0.117$) were non-significant, negative predictors of BSR. The best linear model for avian abundance explained 75% of the variance in the data and included park percent built cover, landscape percent built cover and landscape percent water cover, average number of fruiting trees at park, and park area ($R^2=0.75$, $p=0.004$)

Table 2. Bird species observed across all surveys. Status: R=resident, M=migrant, U=unestablished. Number of exurban, urban, and total number of parks where a species was observed.

Common Name	Scientific Name	Status	Exurban Parks	Urban Parks	Total Parks
Acorn Woodpecker	<i>Melanerpes formicivorus</i>	R	0	1	1
American Crow	<i>Corvus brachyrhynchos</i>	R	1	2	3
American Goldfinch	<i>Spinus tristis</i>	M	1	1	2
American Pipit	<i>Anthus rubescens</i>	M	3	2	5
American Robin	<i>Turdus migratorius</i>	R	5	3	8
Anna's Hummingbird	<i>Calypte anna</i>	R	3	5	8
Black Phoebe	<i>Sayornis nigricans</i>	R	8	5	13
Blue-gray Gnatcatcher	<i>Polioptila caerulea</i>	M	0	1	1
Brewer's Blackbird	<i>Euphagus cyanocephalus</i>	R	7	1	8
Brown-headed Cowbird	<i>Molothrus ater</i>	R	1	0	1
Bushtit	<i>Psaltiriparus minimus</i>	R	2	0	2
California Gull	<i>Larus californicus</i>	M	0	1	1
California Scrub-Jay	<i>Aphelocoma californica</i>	R	5	8	13
Canada Goose	<i>Branta canadensis</i>	M	1	4	5
Common Merganser	<i>Mergus merganser</i>	M	0	1	1
Dark-eyed Junco	<i>Junco hyemalis</i>	R	1	0	1
Downy Woodpecker	<i>Picoides pubescens</i>	R	0	1	1
European Starling	<i>Sturnus vulgaris</i>	R	4	7	11
Great Egret	<i>Ardea alba</i>	R	1	1	2
Great-tailed Grackle	<i>Quiscalus mexicanus</i>	U	3	0	3
Greater White-fronted Goose	<i>Anser albifrons</i>	M	0	2	2
Horned Lark	<i>Eremophila alpestris</i>	R	3	0	3
House Finch	<i>Haemorhous mexicanus</i>	R	5	6	11
House Sparrow	<i>Passer domesticus</i>	R	2	0	2
Killdeer	<i>Charadrius vociferus</i>	R	8	5	13
Lesser Goldfinch	<i>Spinus psaltria</i>	R	3	5	8
Mallard	<i>Anas platyrhynchos</i>	R	0	1	1
Northern Flicker	<i>Colaptes auratus</i>	R	2	8	10
Northern Mockingbird	<i>Mimus polyglottos</i>	R	3	3	6

Table 2 continued

Common Name	Scientific Name	Status	Exurban Parks	Urban Parks	Total Parks
Orange-crowned Warbler	<i>Oreothlypis celata</i>	M	0	1	1
Purple Finch	<i>Haemorhous purpureus</i>	R	1	0	1
Red-tailed Hawk	<i>Buteo jamaicensis</i>	R	1	0	1
Red-winged Blackbird	<i>Agelaius phoeniceus</i>	R	3	0	3
Ring-billed Gull	<i>Larus delawarensis</i>	M	0	1	1
Ruby-crowned Kinglet	<i>Regulus calendula</i>	M	0	1	1
Savannah Sparrow	<i>Passerculus sandwichensis</i>	R	6	0	6
Say's Phoebe	<i>Sayornis saya</i>	M	3	0	3
Snowy Egret	<i>Egretta thula</i>	R	0	1	1
Swainson's Hawk	<i>Buteo swainsoni</i>	M	1	0	1
Turkey Vulture	<i>Cathartes aura</i>	R	1	0	1
Western Bluebird	<i>Sialia mexicana</i>	R	3	5	8
Western Meadowlark	<i>Sturnella neglecta</i>	R	6	1	7
White-breasted Nuthatch	<i>Sitta carolinensis</i>	R	0	1	1
White-crowned Sparrow	<i>Zonotrichia leucophrys</i>	M	1	0	1
White-tailed Kite	<i>Elanus leucurus</i>	R	2	1	3
Wild Turkey	<i>Meleagris gallopavo</i>	R	1	2	3
Yellow Warbler	<i>Setophaga petechia</i>	M	0	1	1
Yellow-billed Magpie	<i>Pica nuttalli</i>	R	0	6	6
Yellow-rumped Warbler	<i>Setophaga coronata</i>	M	4	6	10

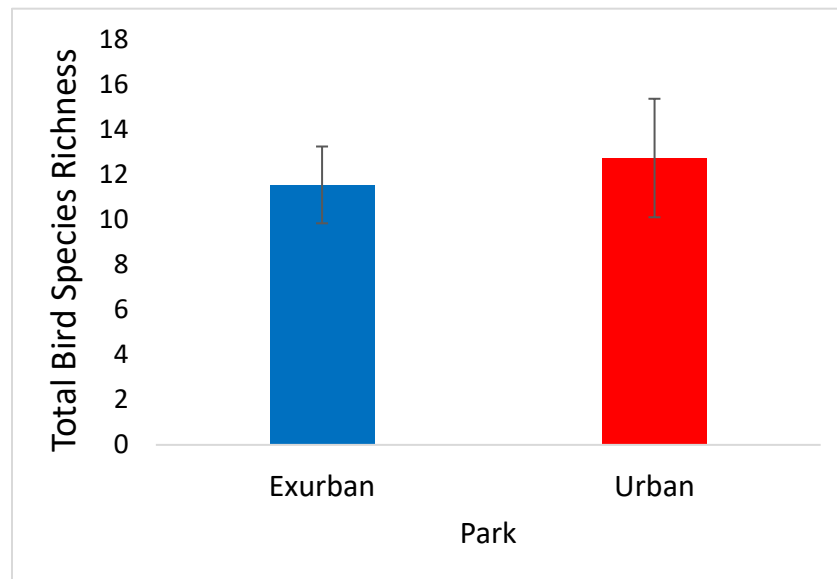


Figure 2. Comparison of total number of bird species seen at urban and exurban parks ($t=-1.0604$, $df=15$, $p=0.3057$). Error bars are \pm standard deviation.

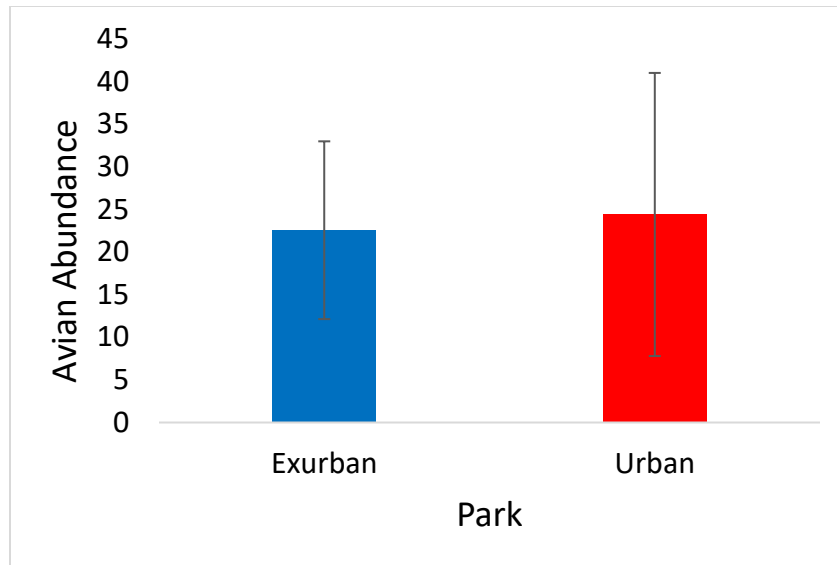


Figure 3. Comparison of avian abundance at urban and exurban parks ($t=-0.043$, $df=15$, $p=0.967$). Error bars are \pm standard deviation.

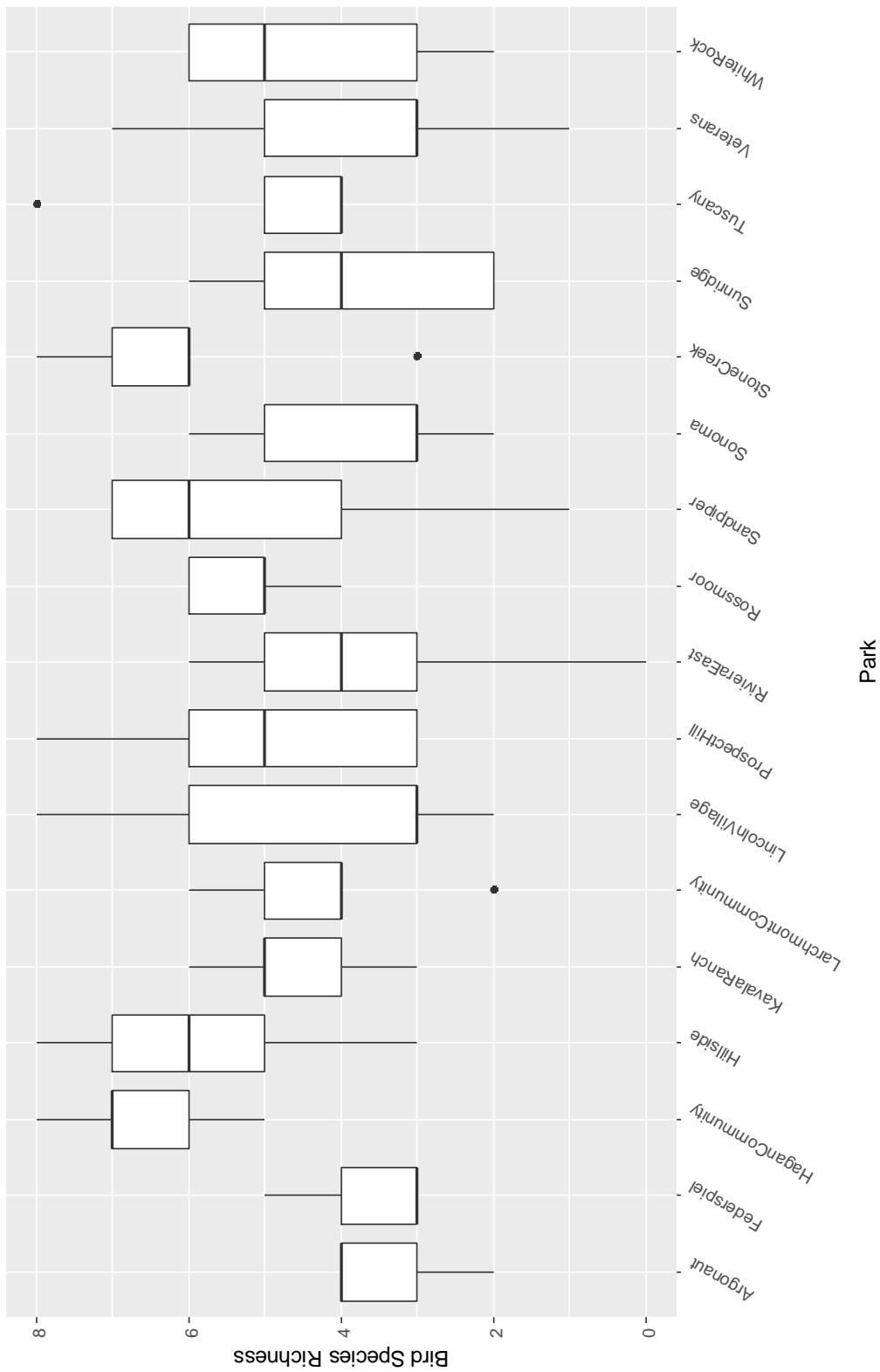


Figure 4. Boxplot comparison of mean number of species seen during 5 surveys at each of 17 parks.

(Table 3). Park percent built cover ($t=-3.67$, $p=0.003$), landscape percent built cover ($t=-2.09$, $p=0.06$), landscape percent water cover ($t=-4.74$, $p<0.001$), and average number of fruiting trees at park ($t=2.71$, $p=0.02$) were all negatively associated with abundance. Park area was the only positive, though non-significant, predictor of abundance ($t=1.98$, $p=0.073$).

Communities segregated in NMDS space according to park location type (Stress=0.15; Figure 5). The environmental factors that significantly drove dimension one were distance to river ($r^2=0.67$, $p=0.003$) and landscape percent grass cover ($r^2=0.55$, $p=0.002$), which were negatively associated with dimension one; and tree DBH ($r^2=0.57$, $p=0.007$), tree height ($r^2=0.76$, $p<0.001$), park percent tree cover ($r^2=0.51$, $p=0.005$) and landscape percent tree cover ($r^2=0.62$, $p=0.002$), which were positively associated with dimension one. When parks were ordinated into multidimensional space, urban and exurban parks tended to segregate along dimension one. Distance to river and landscape percent grass cover led to the grouping of the exurban parks, and tree DBH, tree height, park percent tree cover and landscape percent tree cover led to the grouping of urban parks. The exception to this pattern was Veterans Park, an exurban park which segregated in space with the urban parks (Figure 5).

Based on Bray-Curtis dissimilarity indices, species composition was significantly different between urban and exurban parks using ANOSIM ($R = 0.526$, $P = <0.001$). SIMPER indicated that the species that contributed the most to these differences were Brewer's blackbird (*Euphagus cyanocephalus*), Western meadowlark (*Sturnella neglecta*), Wild Turkey (*Meleagris gallopavo*), Killdeer (*Charadrius vociferous*) and

Table 3. Final models (lowest AIC) and R^2 values from stepwise multiple linear regression analyses between bird species richness or avian abundance of bird communities at exurban and urban parks, and park and landscape habitat characteristics.

Dependent Variable		Coefficient	Std. Error	<i>p</i> -value
BSR	(Intercept)	-1.280	2.454	0.612
	Average Tree DBH	-0.016	0.012	0.218
	Landscape % Built Cover	0.041	0.031	0.207
	Landscape % Water Cover	-0.142	0.084	0.117
	Log Park Area	3.487	0.510	<0.001 ***
	Model ($R^2=0.81$)			<0.001***
Abundance	(Intercept)	3.872	0.654	<0.001***
	Park % Built Cover	-0.045	0.012	0.003**
	Landscape % Built Cover	-0.017	0.008	0.060
	Landscape % Water Cover	-0.129	0.027	<0.001***
	Park Fruiting Trees	-0.021	0.008	0.020*
	Log Park Area	0.272	0.137	0.073
Model ($R^2=0.75$)			0.005**	

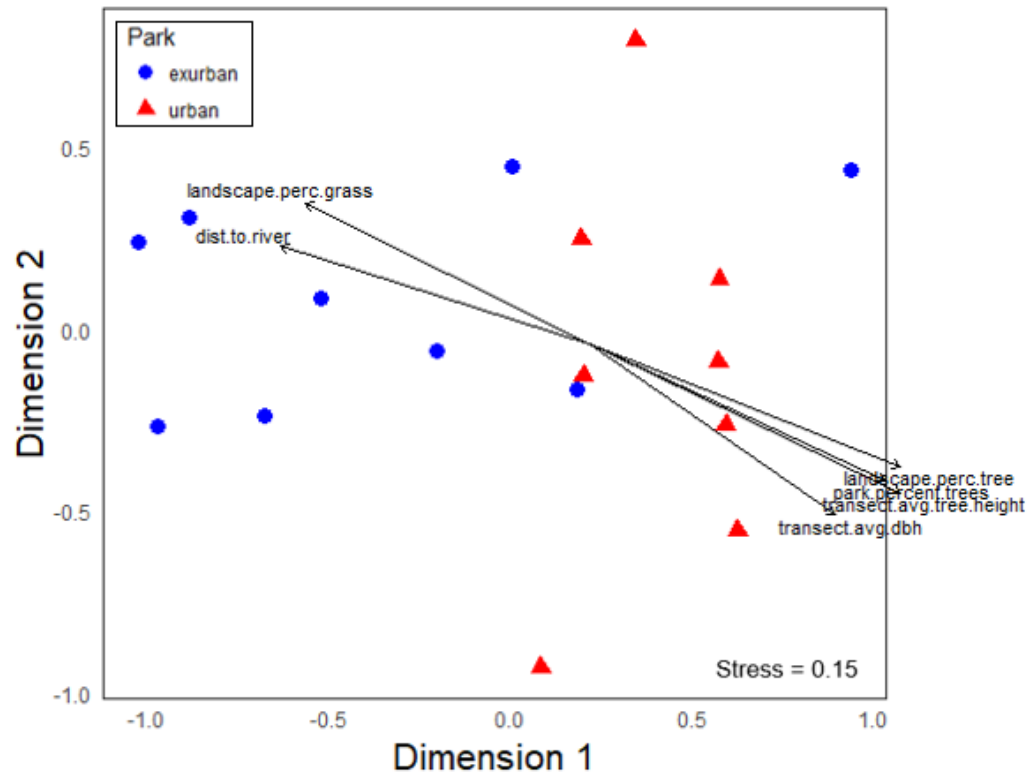


Figure 5. Nonmetric multi-dimensional scaling (NMDS) plot of avian communities at urban and exurban parks. Bray-Curtis dissimilarity used. Significant ($p < 0.05$) environmental vectors fitted onto ordination. Left arrows: landscape % grass cover, distance to river. Right arrows: tree height, DBH, park and landscape % tree cover.

American Pipit (*Anthus rubescens*) present in higher densities at exurban parks; and Canada Goose (*Branta canadensis*), European Starling (*Sturnus vulgaris*), Yellow-rumped Warbler (*Setophaga coronata*), House Finch (*Haemorhous mexicanus*) and Yellow-billed Magpie (*Pica nuttalli*) present in higher densities at urban parks (Table 4). Together, these species explained 75.92% of the dissimilarity in species composition between park groups.

Urban parks had a higher abundance of non-native species, owing to the presence of the invasive European Starling, sometimes in high numbers. For instance, Federspiel Park had an average of about 13 European Starlings per survey and a maximum of 63 during a single survey. The average abundance of European starlings was higher at urban ($\bar{x}=3.35$, $sd=4.12$) than exurban ($\bar{x}=0.78$, $sd=1.07$) parks.

There were several bird species observed at only exurban, or only urban, parks. Brown-headed Cowbird (*Molothrus ater*), Bushtit (*Psaltiriparus minimus*), Dark-eyed Junco (*Junco hyemalis*), Great-tailed Grackle (*Quiscalus mexicanus*), Horned Lark (*Eremophila alpestris*), Purple Finch (*Haemorhous purpureus*), Red-tailed Hawk (*Buteo jamaicensis*), and Red-winged Blackbird (*Agelaius phoeniceus*) were only observed at exurban parks. Acorn Woodpecker (*Melanerpes formicivorus*), California Gull (*Larus californicus*), Canada Goose, Common Merganser (*Mergus merganser*), Downy Woodpecker (*Picoides pubescens*), Greater White-fronted Goose (*Anser albifrons*), Orange-crowned Warbler (*Oreothlypis celata*), Mallard (*Anas platyrhynchos*), Ring-billed Gull (*Larus delawarensis*), Ruby-crowned Kinglet (*Regulus calendula*), Yellow

Table 4. Between park type dissimilarity (SIMPER, dissimilarity = 87.06) for comparisons of urban and exurban parks based on species density. Each species is listed with its contribution to the dissimilarity, the cumulative percentage dissimilarity, and the park type where it was observed in higher densities.

Species	Contribution %	Cumulative %	Higher Density
Brewer's Blackbird	20.32	20.32	Exurban
Canada Goose	18.89	39.21	Urban
European Starling	8.35	47.56	Urban
Western Meadowlark	6.60	54.16	Exurban
Yellow-rumped Warbler	4.40	58.55	Urban
House Finch	4.16	62.72	Urban
Yellow-billed Magpie	3.80	66.52	Urban
Wild Turkey	3.54	70.06	Exurban
Killdeer	3.25	73.30	Exurban
American Pipit	2.62	75.92	Exurban
California Scrub-Jay	2.20	78.12	Urban
Savannah Sparrow	2.14	80.26	Exurban
Northern Flicker	2.12	82.38	Urban
Lesser Goldfinch	1.81	84.19	Urban
Lark Sparrow	1.48	85.67	Urban
Black Phoebe	1.45	87.12	Exurban
American Robin	1.10	88.22	Urban
Western Bluebird	1.09	89.31	Urban
Horned Lark	1.08	90.39	Exurban
Bushtit	1.03	91.42	Exurban
Great-tailed Grackle	1.02	92.44	Exurban
Red-winged Blackbird	0.97	93.41	Exurban
Northern Mockingbird	0.73	94.15	Exurban
Anna's Hummingbird	0.73	94.87	Exurban
Mallard	0.62	95.49	Urban
American Goldfinch	0.48	95.97	Exurban
Brown-headed Cowbird	0.37	96.34	Exurban
American Crow	0.35	96.70	Urban
Ring-billed Gull	0.34	97.04	Urban
Turkey Vulture	0.30	97.34	Exurban
Blue-gray Gnatcatcher	0.27	97.61	Urban
White-tailed Kite	0.23	97.84	Urban
Say's Phoebe	0.21	98.05	Exurban
Snowy Egret	0.21	98.25	Urban
Ruby-crested Kinglet	0.21	98.46	Urban
Acorn Woodpecker	0.19	98.64	Urban
Great Egret	0.17	98.81	Urban
House Sparrow	0.17	98.99	Exurban
Common Merganser	0.14	99.12	Urban
Greater White-fronted Goose	0.13	99.25	Urban
Yellow Warbler	0.10	99.35	Exurban
White-crowned Sparrow	0.09	99.44	Urban
Orange-crowned Warbler	0.08	99.53	Urban
White-breasted Nuthatch	0.08	99.61	Urban
Purple Finch	0.07	99.69	Exurban
Swainson's Hawk	0.07	99.76	Exurban
Dark-eyed Junco	0.07	99.84	Exurban

Downy Woodpecker	0.07	99.90	Urban
Red-tailed Hawk	0.06	99.97	Exurban
California Gull	0.03	100	Urban

Warbler (*Setophaga petechia*) and Yellow-billed Magpie were only observed at urban parks.

DISCUSSION

The results of this study suggest that the avian species richness of parks during the non-breeding season in the study area is not significantly different between older parks in established urban areas, and recently built parks still surrounded in part by native grassland habitat. This is contrary to my hypothesis that BSR would be higher in urban parks. The results also suggest that the number of individual birds using parks did not differ significantly between park types, which is contrary to my hypothesis that avian abundance would be higher in urban parks. It should be noted that the exurban parks in this study (<20 years of age) are also uniformly younger than those labelled urban (>25 years of age), and that older parks tended to be closer to the river. Because these factors cannot be disassociated, the findings of this study should be interpreted with each of the park characteristics in mind. Several features of parks were measured and analyzed in this study to try and quantitatively and objectively compare these two park types (Table 1) so the findings might be generalized to parks in other areas during the non-breeding season.

Age of parks did not predict bird species richness. This is surprising, given that park age has been correlated with BSR in several studies (Fernandez-Juricic 2000, Fernandez-Juricic and Jokimaki 2001, Biadun and Zmihorski 2011, Carbo-Ramirez and Zuria 2011). The city of Rancho Cordova is not as old as some European cities where many studies of urban parks take place: the parks in this study ranged from one year to 58 years of age. Fernandez-Juricic (2000) conducted a study of urban parks in Madrid, which included parks as old as 367 years of age. However, a moderately young (27 years) and very old (367 years) park had the same species richness (Fernandez-Juricic 2000).

BSR differs broadly among cities (Aronson et al. 2014), which precludes direct comparisons. Yet, the youngest parks (<20 years) in their study had BSRs on the lower end of those observed (Fernandez-Juricic 2000); whereas the younger parks in this study did not show the same tendency. Potentially the ability to generalize a correlation between age and BSR is likewise limited by the specific locale and characteristics of parks.

Park age and the resulting complexity of habitat structure should increase bird diversity (Nielson et al. 2013; also see Fernandez-Juricic and Jokimaki 2001, Biadun and Zmihorski 2011); however, this is not always demonstrated. For instance, in their study of urban parks, Jokimaki (1999) found that certain species were predicted by particular vegetation variables; but that diversity of vegetation and foliage height was not correlated with BSR. Although distance to river and age were not included in the regression models, tree DBH indirectly captured this information: older parks were closer to the river and had older, larger trees; without the models violating the assumptions of non-multicollinearity.) Even though they tended to lack mature trees (five of the nine exurban parks had an average DBH of less than 10cm) the BSR of younger parks was comparable to that of parks with much higher vegetative complexity as measured by tree height, DBH and percent cover. The fact that park age was not a predictor of BSR indicates that the more recent, exurban parks have characteristics leading to comparable BSR to older, urban parks.

As expansion into habitats such as grassland homogenizes the landscape, urban adapter species from the regional species pool are seen (Blair 1996, Leveau et al. 2017).

The results of a study in Argentina and Finland suggested that within a particular biome, urbanization does not homogenize wintering bird communities at the regional scale (Leveau et al. 2017). Further, bird assemblages contain more species in the initial stages of urbanization (Jokimaki and Suhonen 1993) because moderate disturbance increases bird abundance and richness (Blair 1996, 2004). The younger parks in this study could therefore be experiencing a temporary boost in BSR because they were recently built in an exurban matrix. Fernandez-Juricic (2000) found that the incidence of species at young parks (<25 years) was correlated with regional densities of species, while the same was not true of older parks. Therefore, the grassland species seen at exurban parks in high densities (Brewer's Blackbirds, Western Meadowlarks) or seen only at exurban parks (Red-winged Blackbirds, Savannah Sparrows) could be contributing to a boost in BSR at young parks, at least for now. Such species might not be observed at these parks in the future when, presumably based on ongoing construction and plans for development, the surrounding grassland habitat is transformed by urban sprawl.

Larger parks tended to have higher species richness, corroborating the finding that park area is correlated with higher BSR (Jokimaki 1999, Fernandez-Juricic and Jokimaki 2001, Murgui 2007, Carbo-Ramirez and Zuria 2011). Greater area increases the availability of habitats for birds (Fernandez-Juricic 2000). Many measures of habitat heterogeneity, including land cover types, tree height and DBH, scale positively with area (Stein et al. 2014). Although park age and area have been correlated in other studies (Fernandez-Juricic 2000), the same was not true of the parks in this study. Park area was correlated with an increase in BSR regardless of park age. This suggests that even when

an increase in area doesn't directly translate to an increase in habitat complexity or heterogeneity, parks may still be able to support a larger number of species. For instance, at several younger exurban parks where trees were small, many birds were observed foraging in swaths of grass, often in mixed-flocks. Because this study took place in the fall after a Central Valley summer where temperatures regularly surpass 100°F, the lush green grass tended by humans may have presented an especially important foraging substrate for species during that time.

White et al. (2005) found that recently built streetscapes in Australia characterized by a lack of mature trees and limited structural diversity had lower BSR and abundance than parks and native streetscapes. Savard et al. (2001) stated that new residential developments have low bird diversity, but depending on building structure, can support higher bird densities. While neither of these assertions was supported by the results of this study, this demonstrates that clear patterns have not been established for bird communities of recently developed urban areas. The parks located in newer residential developments in this study were able to support comparable numbers of species and abundances of birds to established parks.

Interestingly, in this study, a surrounding habitat that had a higher percentage of built cover (including roads, pavement and buildings) positively predicted bird species. The landscape percent built cover between parks types are on average quite similar (Table 1). Several exurban parks were built in a new residential development where most of the cover is pavement because trees and shrubs are still immature, which is in turn surrounded by grassland. The urban parks were usually housed in developments with

mature, tree-lined streets. Both park types showed an average of about half the surrounding area being covered by built cover, but this likely has different a meaning depending on park type. The fact that BSR is predicted by the built cover of the adjacent landscape is somewhat perplexing. Studies have demonstrated an increase in abundance with increasing built cover (Blair 1996, 2004); however, a decrease in BSR is expected (Aronson et al. 2014). Perhaps the surrounding habitat with higher built cover was indicative of a higher diversity of structures for activities such as nesting and roosting (Emlen 1974).

In this study, two factors that tended to be characteristic of older, urban parks were associated with lower bird species richness: higher percentage of water cover in the adjacent landscape, and average tree DBH. Three of the urban parks (and none of the exurban parks) had landscape water cover that was greater than one percent of the total landscape cover, primarily due to the American River. The presence of riparian habitat was expected to increase BSR by attracting riparian species since the Central Valley is very important habitat for overwintering waterbirds (Matchett and Fleskes 2017). While more ducks and geese were seen at urban parks, perhaps in general those parks were not representative of the nearby riparian species. Average tree DBH was also negatively associated with BSR. Tree size has been linked with BSR in previous studies (Biadun and Zmihorski 2011, Carbo and Ramirez 2012). However, habitat heterogeneity could be lowered if larger trees uniformly provide higher cover. Additionally, several of the exurban parks with near average BSR had tree cover of less than 5% and an average DBH <10cm (Appendix B). This clearly demonstrates that other factors, including the

characteristics of the surrounding landscape, help to explain BSR in tandem with vegetative structure of trees. This may be especially true for the exurban parks where grassland species were seen in high numbers.

Abundance was predicted by a slightly different set of factors than BSR. Larger parks tended to have higher avian abundance. It is unsurprising that larger areas were able to support larger numbers of birds, since this can potentially mean more space to forage in and a larger number of habitats to take advantage of. Similar to BSR, abundance had the strongest negative association with landscape percent water cover. Although the effect sizes were small, park and landscape percent built cover were negatively associated with abundance (Table 3). An increase in built cover in the park and the adjacent landscape meant generally less habitable areas for birds. Together with landscape water cover, these three factors might have meant less of a rich food source in the early to late fall months for some birds (grass and associated insects) or safe areas to take shelter, roost and forage (trees) leading to lower abundance overall.

Abundance was also negatively predicted by the average number of fruiting trees, although the effect sizes was also small (Table 3). We would expect fruiting trees to attract more birds; however, it appears that this was not the case. Generally, exurban parks tended have higher numbers of fruiting trees (Table 1). The trees in exurban parks tended to be non-native and/or ornamental, and often less mature. Additionally, they tended to be a single species planted in close proximity or in rows. Perhaps these trees in particular were not attractive to birds, although they sometimes had brightly colored fruits (e.g. bright red berries on trees at Tuscany Park).

Community Composition

During the non-breeding season, a diversity of bird species utilizes urban and exurban parks in Rancho Cordova, CA. These include many resident and several migrant species that arrive in Fall, and/or overwinter in the area. Birds with a diversity of life-histories are seen across parks. Although bird species richness and abundance did not differ between the urban and exurban parks, considering these findings in isolation could lead to misleading results about the similarity of their respective avian assemblages.

Even in studies where no difference in BSR is detected, significant differences in community assemblage can be driven by urbanization (Litteral and Shochat 2017). Species composition between the parks in urban versus exurban areas during the non-breeding season was significantly different. The exurban parks tended to be dominated by grassland birds including Brewer's Blackbirds and Western Meadowlarks; whereas the urban parks were dominated by European Starlings and Canada Geese. Brewer's blackbirds and Western Meadowlarks have specific habitat requirements (reeds, grassland >5ha) that can't be met by a heavily urbanized landscape. Conversely, European Starlings and Canada Geese are well-adjusted to heavily urbanized areas; in the case of the European Starling this is evinced by their incredible numbers and ubiquity in their native habitat in Europe as well as here in the United States.

The prediction that urban parks would have higher proportions of non-native species was supported. This was due solely to the high density of the European Starling at urban parks (the second most abundant species at urban parks) compared to exurban parks. Surprisingly, the House Sparrow, though ubiquitous in the urban area of this study,

was only seen at a single exurban park. The House Sparrow is regarded as one of the most successful avian urban invaders (MacGregor-Fors 2017) and it has been suggested that they should be observed in high densities at urban parks in the non-breeding season (Leveau and Leveau 2016). In one study, maximum building height and human-related variables such as pedestrians were positively related to House Sparrow abundance, while tree density and cover were negatively related (MacGregor-Fors 2017). Perhaps the large, mature trees characteristic of the urban parks in this study led to the absence of House Sparrows; while the exurban parks were generally free of tall buildings and high levels of urbanization that might have led to their presence. Although there was only one nonnative species seen at parks, the high abundance of starlings suggests that summary statistics such as BSR and abundance could be misleading for conservation measures if native and nonnative species are both included without being considered separate from one another.

The majority of species seen across parks (48 out of 50 species) were native to California, suggesting that during the non-breeding season these parks are important habitat for native birds. The Great-tailed Grackle, which was seen at three exurban parks, historically had a range that did not include Sacramento or the surrounding area (in Sibley's Guide to Birds of North America from 2003, their presence in the Central Valley is absent or denoted as "rare"). However, the presence of these birds at multiple parks during the non-breeding season is indicative of their range spreading further north.

Interestingly, there were more fall migrant/overwintering species observed at urban parks (12 species) than exurban parks (7 species). This might have been influenced

by higher levels of artificial light near urban parks, which demonstrably increases autumnal migrant stopover density (McLaren et al. 2018). Three of the migrant species seen only at urban parks were ducks, and two were gulls. As ducks and gulls tend to associate near water, it is possible that closer proximity to the river contributed to these observations. Other migrants seen only at urban parks include arboreal species such as the White-breasted Nuthatch and Yellow Warbler. Observing a total of 15 migrant species at parks is notable because migratory birds are more highly disadvantaged in urban environments than species that are year-round residents (Huste and Boulinier 2011). Migrants must recover breeding sites that can be lost to sedentary species, while sedentary species have more opportunities to adapt to urban constraints. This underscores the potential importance of parks as refuges for migrant species, perhaps especially in more highly urbanized areas.

The key factors affecting species composition were identified as distance to river, landscape percent grass cover, tree DBH, tree height and landscape percent tree cover. The exurban parks were characterized by longer distances to river habitat and higher landscape percent grass cover. Together, these factors contributed to the species assemblages at exurban parks, which included grassland species such as Brewer's Blackbird and Western Meadowlark and excluded species such as warblers and ducks. The Western Meadowlark has a habitat requirement of grassland >5ha (Johnson and Igl 2001). Four exurban parks directly bordered grassland on one side, and all nine reside in housing developments that border grassland on one side or all sides (Figure 1). Although with continued urban development it is not always possible to have parks that directly

border native habitat, increasing habitat connectivity and the availability of parks to use as stepping stones (Fernandez-Juricic and Jokimaki 2001) could help maintain their presence at parks.

The groupings of urban versus exurban parks along the NMDS gradient as defined by the characteristics above was clear except for one exurban park that aggregated with the urban parks: Veterans Park. The housing development that Veterans Park is located in is older than all others of the exurban group. Although Veterans Park was established in 1990, the trees within the park are much more mature, having been planted much earlier. This resulted in an average tree DBH more than twice as large as any other exurban park, and an average tree height higher than any other exurban park. Despite these differences, the park had a total BSR near average for the exurban park group and didn't differ wildly in species composition; although it was the only park where the Dark-eyed Junco was observed.

In the study area, the results showed a spatial organization of bird species due to local and landscape habitat features of parks. To support a variety of bird species, no clear patterns emerged on a park by park basis. However, the fact that parks with divergent features supported comparable species richness and abundances suggests that parks with large swaths of grass, and parks with larger, mature trees and more tree cover, are both used by birds in urban spaces. Recently built parks with young saplings might not support as many migrants that forage or roost in large trees, but they can provide refuge to displaced grassland species whose native habitat was transformed by development. Future studies could expand on this study by considering the same parks in

the breeding season and comparing species richness, abundance, and community assemblages during various times of year. The habitat features associated with specific species in this study could also be investigated using presence absence data. Information derived from such an analysis could prove useful for specific conservation target species and goals. While urban environments pose great challenges to many bird species, they also present new opportunities for adaptation (Shanahan et al. 2014). Cities can simultaneously support biodiversity and people (Aronson et al. 2014). In providing a diversity of refuges within urban landscapes that are accessible to one another and to landscape features, hopefully the expectation of biotic homogenization can be slowed, or even evaded, and the greatest number of native species supported.

Appendix A. Survey number, start time, date and temperature for all transect surveys by park name (85 total)

Park	Survey	Start Time (AM)	Date	Temp (F)
Argonaut	1	7:04	9/24/2017	53
Argonaut	2	8:18	10/5/2017	56
Argonaut	3	10:42	10/21/2017	61
Argonaut	4	8:45	11/2/2017	55
Argonaut	5	9:58	11/19/2017	53
Federspiel	1	9:25	10/12/2017	56
Federspiel	2	8:30	10/21/2017	48
Federspiel	3	9:50	10/26/2017	69
Federspiel	4	9:17	11/10/2017	64
Federspiel	5	7:08	11/19/2017	45
Hagan Community	1	8:10	10/13/2017	53
Hagan Community	2	8:46	10/21/2017	51
Hagan Community	3	9:32	11/10/2017	64
Hagan Community	4	7:50	11/19/2017	45
Hagan Community	5	9:07	11/30/2017	50
Hillside	1	7:55	9/24/2017	54
Hillside	2	10:36	10/12/2017	60
Hillside	3	8:43	10/26/2017	62
Hillside	4	9:12	11/2/2017	57
Hillside	5	9:32	11/19/2017	53
Kavala Ranch	1	9:10	9/24/2017	64
Kavala Ranch	2	7:32	10/5/2017	53
Kavala Ranch	3	7:47	10/26/2017	55
Kavala Ranch	4	10:12	11/2/2017	59
Kavala Ranch	5	10:39	11/19/2017	57
Larchmont Comm.	1	7:06	9/28/2017	58
Larchmont Comm.	2	10:43	10/26/2017	75
Larchmont Comm.	3	7:38	11/2/2017	55
Larchmont Comm.	4	7:03	11/10/2017	64
Larchmont Comm.	5	8:07	11/30/2017	45
Lincoln Village	1	8:19	9/28/2017	65
Lincoln Village	2	7:25	10/13/2017	47
Lincoln Village	3	8:22	11/2/2017	55
Lincoln Village	4	7:25	11/10/2017	57
Lincoln Village	5	8:47	11/30/2017	48
Prospect Hill	1	8:06	10/12/2017	53
Prospect Hill	2	7:47	10/21/2017	47
Prospect Hill	3	8:41	11/10/2017	60

Prospect Hill	4	8:48	11/19/2017	48
Prospect Hill	5	9:36	11/30/2017	51
Riviera East	1	7:43	9/28/2017	60
Riviera East	2	9:14	10/21/2017	55
Riviera East	3	8:00	11/2/2017	55
Riviera East	4	10:08	11/10/2017	64
Riviera East	5	8:26	11/30/2017	46
Rossmoor	1	8:53	10/12/2017	55
Rossmoor	2	8:09	10/21/2017	47
Rossmoor	3	10:13	10/26/2017	71
Rossmoor	4	9:01	11/10/2017	64
Rossmoor	5	8:25	11/19/2017	47
Sandpiper	1	8:24	9/24/2017	57
Sandpiper	2	7:55	10/5/2017	56
Sandpiper	3	8:04	10/26/2017	58
Sandpiper	4	9:55	11/2/2017	59
Sandpiper	5	10:22	11/19/2017	57
Sonoma	1	7:21	9/16/2017	63
Sonoma	2	9:08	9/28/2017	72
Sonoma	3	10:03	10/21/2017	59
Sonoma	4	10:50	11/2/2017	60
Sonoma	5	10:30	11/30/2017	56
Stone Creek	1	8:00	9/16/2017	65
Stone Creek	2	9:32	9/28/2017	73
Stone Creek	3	9:09	10/26/2017	65
Stone Creek	4	7:52	11/10/2017	58
Stone Creek	5	10:16	11/30/2017	55
Sunridge	1	7:28	9/24/2017	54
Sunridge	2	10:25	10/12/2017	60
Sunridge	3	8:24	10/26/2017	62
Sunridge	4	9:30	11/2/2017	57
Sunridge	5	9:17	11/19/2017	52
Tuscany	1	8:41	9/16/2017	67
Tuscany	2	10:01	9/28/2017	78
Tuscany	3	10:23	10/21/2017	60
Tuscany	4	8:08	11/10/2017	59
Tuscany	5	9:58	11/30/2017	55
Veterans	1	9:45	9/24/2017	60
Veterans	2	10:15	9/28/2017	78
Veterans	3	8:42	10/13/2017	57
Veterans	4	10:33	11/2/2017	58
Veterans	5	10:48	11/30/2017	57

White Rock	1	8:43	9/28/2017	69
White Rock	2	9:45	10/12/2017	56
White Rock	3	9:36	10/26/2017	63
White Rock	4	9:49	11/10/2017	64
White Rock	5	7:30	11/19/2017	45

Appendix B. Habitat variables at each park (exurban=9, urban=8)

Transect Characteristics (50m radius)							
Park	Fruiting Trees (#)	Tree spp. (#)	Trees (#)	Tree DBH (cm)	Tree Height (m)	Humans (ped/min)	Dogs (dog/min)
<i>Exurban</i>							
Argonaut	0	3	7	7.78	4.73	0.36	0.16
Hillside	5.4	0	0	0.00	0.00	0.08	0.04
Kavala Ranch	4.4	1	2	8.44	4.25	0.12	0.00
Sandpiper	0	1	1	3.50	3.20	0.00	0.00
Sonoma	6.6	3	15	9.89	4.71	0.32	0.00
Stone Creek	45.4	5	19	17.67	7.07	0.28	0.28
Sunridge	0	1	9	3.57	2.04	0.20	0.08
Tuscany	16.4	2	5	21.33	7.64	0.08	0.08
Veterans	15.2	3	7	43.46	12.71	0.20	0.04
<i>Urban</i>							
Federspiel	15	3	16	69.98	14.28	0.08	0.12
Hagan C.	1.2	1	4	86.74	14.38	0.00	0.00
Larchmont C.	0	3	6	20.00	7.83	0.64	0.32
Lincoln Vill.	8	4	8	34.46	9.36	0.28	0.04
Prospect Hill	3.6	2	2	52.31	12.33	0.12	0.08
Riviera East	0	4	12	43.24	9.79	0.36	0.16
Rossmoor	0.8	3	4	76.19	9.38	0.20	0.00
White Rock	1	2	7	19.80	8.29	0.16	0.12
Park Characteristics							
Park	Park Age (yrs)	Park Area (km ²)	Park % Tree	Park % Grass	Park % B. Ground	Park % Water	Park % Built
<i>Exurban</i>							
Argonaut	9	25.23	4.12	72.60	4.57	0	18.71
Hillside	1	31.22	1.48	44.26	22.89	13.57	17.80
Kavala Ranch	4	20.19	1.59	81.55	0.00	0	16.86
Sandpiper	9	20.39	3.24	81.48	0.74	0	14.54
Sonoma	10	17.36	13.61	70.12	0.00	0	16.27
Stone Creek	10	98.30	12.41	71.79	1.21	0	14.59
Sunridge	3	21.88	0.42	79.27	1.73	0	18.58
Tuscany	10	17.12	19.30	64.12	0.25	0	16.34
Veterans	18	25.46	27.27	44.33	0.00	0	28.40
<i>Urban</i>							
Federspiel	58	15.26	42.73	52.40	0.22	0	4.66
Hagan C.	38	256.58	40.55	40.60	6.42	4.16	8.27
Larchmont C.	38	57.77	26.54	66.20	0.00	0	7.26
Lincoln Vill.	50	67.82	22.86	53.94	1.76	0	21.44
Prospect Hill	28	29.95	39.28	43.13	0.00	0	17.59
Riviera East	26	35.99	31.46	54.16	0.00	0	14.38
Rossmoor	52	36.94	13.58	81.18	2.13	0	3.11
White Rock	55	57.08	18.15	50.83	0.88	0	30.14

Appendix B continued

Landscape Characteristics (500m radius)						
Park	Land. % Tree	Land. % Grass	Land. % Bare Ground	Land. % Built	Land. % Water	Dist. to River (m)
<i>Exurban</i>						
Argonaut	6.91	45.34	8.90	36.57	0	8082
Hillside	0.46	62.16	13.81	20.96	0	7450
Kavala Ranch	2.97	51.22	4.75	39.17	0	9987
Sandpiper	4.94	39.43	1.24	52.52	0	8946
Sonoma	8.95	31.62	10.45	47.13	0.25	3660
Stone Creek	2.35	43.93	15.41	31.15	0	4565
Sunridge	1.99	60.76	4.46	30.59	0	8674
Tuscany	7.93	22.43	7.00	60.53	0.50	4499
Veterans	13.37	39.97	1.25	43.14	0	5365
<i>Urban</i>						
Federspiel	25.65	18.20	2.25	52.43	0.00	1457
Hagan C.	19.03	30.33	10.60	29.60	8.25	291
Larchmont C.	27.75	12.19	3.30	39.16	13.00	58
Lincoln Vill.	17.10	18.48	4.91	54.38	0.00	1384
Prospect Hill	20.43	27.03	0.00	48.82	1.00	1294
Riviera East	24.79	19.57	6.20	37.84	8.60	220
Rossmoor	24.56	31.39	1.43	39.40	0.00	474
White Rock	20.65	21.43	1.46	51.89	0.00	2178

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