# Black-tailed Deer Population and Spatial Responses to an Immuno-Contraception Trial in Oak Bay, British Columbia

Final Report: March 2025

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Prepared for: Urban Wildlife Stewardship Society, Oak Bay, British Columbia.



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#### ACKNOWLEDGEMENTS

This research was conducted by University of Victoria's ACME Research Lab in collaboration with the Urban Wildlife Stewardship Society (UWSS), the District of Oak Bay, and the Province of British Columbia, with funding provided by Oak Bay, the Provincial Urban Deer Cost-Share Program, the UWSS, MITACS Canada, and the Natural Science and Engineering Research Council of Canada (NSERC), through their Alliance Grants Program.

This project was conceived and spearheaded by the Urban Wildlife Stewardship Society. Kristy Kilpatrick, Steve McKerrell, Barb McKerrell, Steve Huxter, Brian Gates, Ralph Archibald, Annette Dehalt, and Don Eastman at UWSS were the driving force at that time and the existence of the project is owed to their fundraising and lobbying. David Budd, Holly Vear, and Susan Laberee joined their efforts and provided tremendous support. Thanks to Mayor and Council of the District of Oak Bay for their support, as well as Holger Bohm, Provincial Ungulate Specialist, Government of British Columbia.

Dr. Adam Hering led all deer captures and treatments, with the support of dozens of volunteers from University of Victoria and Camosun College. Dr. Lynette Brown, veterinarian, also spent substantial volunteer efforts on this project. Sandra Frey coordinated the research project. Wylie Fuller conducted the GPS collar analysis; thanks to Margo Supplies for providing some collars free of charge. Camera image tagging was shared among many people, but much was predominantly done by Katrice Baur with support from Isabel Giguere and Karina Lamy. Alina Fisher provided field help and scientific communication throughout. Isabel Giguere led the spatial camera analysis and contributed extensively to field work. Dr. Andrew Barnas led the density estimation, which had been started by Dr. Joanna Burgar. Andrew also provided the

updated fawning rate analysis, which had been started by Macgregor Aubertin-Young. JT Fisher led the scientific aspects of the program and is responsible for the reporting.

A sincere thanks to all property owners, private businesses, and clubs across Oak Bay who have hosted our deer cameras. Our deepest gratitude to the countless volunteers who spent hundreds of hours cataloguing deer images and assisted with locating and capturing deer for IC. We also thank all the Oak Bay landowners who granted access to their property to locate and dart deer to administer IC.

#### **GENERAL INTRODUCTION**

Black-tailed deer (Odocoileus hemionus columbianus, hereafter "BTD") are native to British Columbia (BC), and fill several important ecological roles<sup>1</sup>. The changing landscape of BC's suburban areas has benefited BTD populations, due to increased foraging opportunities<sup>2</sup> and likely due to reductions in predator presence. Native predators such as wolves (Canis lupus) and cougars (Puma concolor) are kept at low density from most urban areas, effectively excluding them from their ecological roles in deer-population control<sup>3</sup>. An additional benefit to BTD populations in suburban/urban environments is the presence of high-energy and highnutrient forage plants. Urban and suburban areas contain abundant backyard gardens and agricultural crops, which provide ample food resources for deer, potentially allowing BTD to breed more often and more successfully than in unaltered landscapes<sup>4</sup>. In BC's suburban environments, deer have been shown to select for areas with high vegetation greenness, a high proportion of large-sized residential lots, and proximity to parks and golf courses <sup>2</sup> suggesting that human-driven changes to the landscape are key to maintaining urban deer populations. As BTD populations are very sensitive to factors affecting recruitment<sup>5,6</sup>, the lack of natural predation coupled with the abundance of high-quality food resources in suburban and urban landscapes has likely contributed to greater reproductive output and subsequent population growth for BTD.

With BTD populations increasing in suburban and urban areas, various human-deer coexistence challenges may arise. Urban deer can be perceived as "pests" when foraging on gardens or agricultural crops<sup>7</sup>, may act as vectors of disease<sup>8</sup>, and lead to increased traffic accidents<sup>9,10</sup>, although human perceptions of wildlife impacts vary<sup>11,12</sup>. To address these challenges, some municipalities have implemented culling programs to directly reduce the

number of deer present in communities<sup>13</sup>. Unfortunately, these culls have thus far been based on little scientific data and results are highly variable<sup>14</sup>. Moreover, the culling of wildlife can be a contentious and politicized issue, leading to divided communities and even legal battles<sup>15,16</sup>.

One urban deer management strategy that is growing in popularity is the use of immunocontraceptive (IC) treatments to reduce reproductive output<sup>17-19</sup>. Immuno-contraceptive vaccines trigger an animal's immune system to prevent fertilization of the egg<sup>20</sup>, and vaccines such as porcine zona pellucida (PZP) have been applied to various urban deer populations as means of non-lethal deer population control<sup>18,21-23</sup>. Prior to the fall reproductive season (i.e., rut), IC vaccines can be delivered remotely to adult female deer to suppress their ability to produce young the following spring, thus eliminating the need for lethal removal of individuals in a growing population.

To estimate the effectiveness of any wildlife population management strategy, it is necessary to estimate population density before and after treatment. However, collecting precise estimates for deer population density in urban environments comes with challenges. Traditional surveys are based on herd counts - which provide useful distribution information but generally provide low-precision results with wide confidence limits <sup>24</sup> – or aerial surveys which are nearly impossible in suburban areas. Citizen-based herd count surveys engage the public, but do not produce reliable estimates; they lack the statistical rigor to justify management actions scientifically or legally. Collecting precise estimates of urban BTD population density requires the application of rigorous surveying approaches using a combination of remote cameras, satellite collaring, and novel statistical techniques.

Remote cameras are rapidly becoming a popular wildlife research tool because they produce large volumes of data at low cost<sup>25,26</sup>. Their reliability in surveying wildlife species can

be quantified <sup>27</sup> and cameras have been shown to have very high accuracy at detecting deer<sup>28,29</sup>. Remote camera surveys are an increasingly common approach for surveying ungulate occurrence<sup>30-32</sup>, and they have been used to monitor deer in urban areas but previously without the statistical ability to estimate density from the data<sup>33</sup>. The advent of novel statistical techniques has provided important opportunities to collect reliable population density estimates from camera data, allowing for the comparison of urban deer population response before and after management actions.

The first part of our research program was to estimate deer density We used camera traps placed across Oak Bay, and a class of models called Spatial Mark-Resight (SMR) models <sup>34-36</sup> to estimate deer density before, during, and after IC treatment. We also used camera traps to estimate the relative proportion of fawns across Oak Bay throughout this study period.

Population size is only one aspect of the possible outcomes of IC treatment. Deer, like all animals, distribute in space by selecting high-value resources and avoiding perceived risk<sup>37,38</sup>. Part of that decision-making is accounting for the other deer in the landscape, who are competitors for resources and which can defend territories, excluding outsiders<sup>39,40</sup>. Space-use is therefore density-dependent and changes according to how many animals are competing in a landscape<sup>41-43</sup>. We used two approaches to estimate deer space-use. First, we asked how urban deer without any treatment select space across Oak Bay by capturing and GPS collaring female deer. This gives us insights into how the landscape of Oak Bay allows urban deer to persistent and thrive. Second we used camera-trap data to inform species distribution models <sup>44,45</sup> that allow us to infer how spatial distributions changed before, during, after IC, to see how treatment with IC affects how deer select for or avoid the many types of landscape features in this urban area. Finally, we finish with a comparison of urban deer densities to non-urban environments. This report as packaged is a collection of stand-alone chapters, thus the repeating of some information will occur. One chapter has already been published as a scientific journal article and the remainder are all being prepared similarly for publication.

#### **GENERAL METHODS**

#### Study Area

This research took place in the 10.5km<sup>2</sup> District of Oak Bay, Vancouver Island, British Columbia (Figure 1). The study area is largely an urban environment, dominated by small and large residential lots, commercial development, golf courses, and district-managed natural park areas. The eastern and southern edge of the district is bordered by the Salish Sea.



Figure 1. Remote camera array (white dots) to monitor urban deer population response to immune-contraceptive treatment in Oak Bay, B.C.

#### Deer Capture, Marking, and Immuno-contraceptive Treatment

In February-March 2018, we captured 20 female BTD to mark as our control population. We applied GPS-collars fitted with coloured plastic tags to allow for individual identification of marked deer. Data collected by the GPS collars were used to examine urban deer habitat use<sup>2</sup>.

We commenced our IC program in September 2019 and administered the PZP IC-vaccine Zonastat-D to adult female BTD prior to the fall rut. We generally selected for mature (>1.5year-old) does based on body size and/or presence of fawns. We searched for deer in the early morning by conducting road surveys throughout the entirety of our study area to attempt an even coverage of treatment across Oak Bay. Chemical immobilization was delivered via telemetry darting by an experienced wildlife veterinarian (A. Hering) using current regulatory approvals and field protocols. On capture, each animal was injected with 100 ug of Zonastat-D. Captured deer were marked using a combination of coloured marker collars and/or numbered ear tags to allow for individual-level identification (Figure 2).

In September-October 2019, we administered IC to 60 female BTD. Two to six weeks after treatment with the primary PZP vaccine, we were able to locate 55 of our 60 initially treated deer to administer a booster of the same vaccine. Booster vaccination did not require live capture of our study deer but was delivered remotely via darting. In September-October 2020, we administered IC vaccines to an additional 60 female BTD that were not treated the previous fall. Of these 60 newly treated individuals, we administered booster vaccines to 57 individuals. We also administered booster vaccines to 48 of the deer treated with IC in 2019. In September-October 2021, we did not administer primary IC vaccines to any new female BTD but administered IC booster vaccines to 71 BTD previously treated in 2019 and 2020. This totalled to 120 deer marked for IC, plus an additional 19 "control" deer marked in spring 2020 (although

mortalities of marked deer were observed between 2019-2021). IC treatment was paused after fall 2021, although we have continued to monitor the population through 2024.

#### **Ethics approval**

Landowner permission was obtained for every occasion we used private land. Research was facilitated by the District of Oak Bay staff and conducted under permit NA22-687811 issued by the Province of British Columbia (BC), Ministry of Forests, Lands, Nature Resource Operations and Rural Development. BC provided animal care review and approval as part of the permitting process.



Figure 2. Remote camera image of a female black-tailed deer fitted with numbered ear tags and a marker collar.

#### Camera Monitoring and Image Review

We used a systematic study design to monitor deer across Oak Bay. In ArcGIS we overlaid a 400-m radius grid cell array over a spatial data layer of Oak Bay. We deployed a single camera in each grid cell, where logistically feasible, in a site where we aimed to maximize detection of deer (if present) but also reducing detections of people and vehicles. In August 2018, we deployed 39 Bushnell<sup>TM</sup> infra-red camera on both public and private properties across Oak Bay, secured to a tree about 0.5 - 1.5 m off the ground. Cameras were programmed to take 3 pictures in sequence when movement was detected by the infrared trigger, followed by a ten second trigger delay. Following a targeted theft of cameras in winter 2019, we moved some cameras from public properties to private and replaced older Bushnell models with Browning<sup>TM</sup> cameras using similar settings.

We serviced cameras regularly to refresh batteries and download the collected images. Trained technicians manually reviewed and catalogued collected camera imagery using Timelapse image software. For each deer detection on camera, we collected information on the camera location, date, time, the number and sex of adult deer, and the number of fawns present in the image. For marked deer, we also noted any information on ear tag number and marker collars to allow for individual identification, which provides the basis of our population density estimation models prior to- and after IC treatment.

Due to the significant time investment of manually processing camera imagery, we subset our dataset to only examine deer detections in the month of September. We selected this month as does are easily distinguished from bucks due to the presence of antlers, while fawns (<1 year) could be easily distinguished from yearlings (>1 year). Due to challenges with including ongoing marking of deer in September 2019 and 2020 (when female BTD were being marked and treated with IC), we selected a 35-day window from the date for which the last IC-treated deer was marked. For 2019 and 2020, our sampling period is therefore October 8<sup>th</sup> – November 11<sup>th</sup> when the marked population was stable following completion of the primary IC vaccines (and associated markings) being administered in the month of September. No primary vaccines or boosters have been applied since fall 2021.

We monitored the population 2018-2025, culminating in this report (Fig. 3, below).



Figure 3. Timeline of black-tailed deer treatment and monitoring in Oak Bay, British Columbia from 2018-2025. Numbers represent deer known to be alive based on observations of account for known deaths.

## CHAPTER ONE: HABITAT SELECTION BY (UNTREATED) URBAN DEER IN OAK BAY, BRITISH COLUMBIA.

Note: This chapter has been published as *Fisher*, J. T., Fuller, H. W., Hering, A., Frey, S., & Fisher, A. C. (2024). Black-tailed deer resource selection reveals some mechanisms behind the 'luxury effect' in urban wildlife. Urban Ecosystems, 27(1), 63-74.

https://link.springer.com/article/10.1007/s11252-023-01428-7

#### Abstract

The global urban population is expected to increase by 2.5 billion people over the next 30 years. Yet the doubling of urban landscapes in the last decades have already led to habitat loss and concomitant impacts to biodiversity. Nonetheless urban landscapes remain important for wildlife, and multi-city analyses have revealed that wealthy urban areas house more biodiversity (species richness), a 'luxury effect'. We researched some of the mechanisms for the luxury effect for urban black-tailed deer, a species of increasing concern in urban landscapes across the northwestern Nearctic. We hypothesized that deer were capitalizing on anthropogenic resource subsidies occurring in affluent housing and supported by urban green spaces such as parks and golf courses. In 2018-19 we satellite-collared twenty deer in an affluent urban landscape in British Columbia, Canada, with 13-hr fix rates. We used generalized models in an informationtheoretic framework to weigh evidence for competing hypotheses about the role of tree cover, productivity, public green spaces, and wealth in explaining deer selection. Wealth, manifesting as housing lot size, emerged as the dominant predictor of deer space-use, which is highly concentrated into very small home-ranges. Other landscape elements stemming from affluence, including golf courses and parklands, were also strongly selected by deer. We show postcolonization landscape conversion from dry semi-arid savannah to well-watered highproductivity landscapes is supporting deer, with ramifications for the rest of the biotic community. With urban landscapes becoming an increasingly important for biodiversity conservation, understanding these mechanisms and incorporating them into urban planning can help to promote wildlife-human coexistence.

#### Introduction

The global urban population is expected to increase by 2.5 billion people over the next 30 years<sup>46</sup>, following decades of continued urban growth<sup>47</sup>. Consequently, urban landscapes have doubled in the last few decades leading to direct and indirect forest loss<sup>48</sup>, and loss of grasslands and agricultural lands. By 2030, global urban land cover is projected to increase between 430,000 km<sup>2</sup> and 12,568,000 km<sup>246</sup>. With urbanization comes a loss of natural habitats – especially tree cover – and increase in impervious surfaces, of low value to biodiversity<sup>49</sup>. Thus, urbanization generally correlates to losses in species richness and hence biodiversity<sup>48,50</sup>. However, many anthrophilic species continue to coexist with humans in urban environments <sup>51,52</sup> and there is sustained advocacy, research, and planning for urban areas that promote wildlife-human coexistence<sup>53,54</sup>.

Although urban wildlife ecology as a discipline was advocated by Aldo Leopold in the 1930s it remains markedly understudied in academia<sup>55</sup>. The biodiversity outcomes of urban development have been varied, from local extirpation of undesirable species such as predators <sup>56</sup> to multi-taxic rapid phenotypic changes in urban centers, implicating urbanization as a mode of evolutionary change<sup>57</sup>. Species' responses vary within and among cities and contextualizing the mechanisms behind responses remains a key endeavor<sup>46,51,58</sup>. As urban areas are complex mosaics of impervious surfaces (buildings, roads, parking lots), natural or semi-natural greenspaces (parks), and heavily modified greenspaces (yards, gardens, golf courses), each offer

different resources and risks for different species. Those resources are the outcome of social and economic drivers within the human population<sup>59</sup>.

Globally, large carnivores are one of the first groups extirpated, as we seek to "make safe" urban places for humans. One of the outcomes of extirpating large carnivores from urban environments is providing prey species with refugia from predation<sup>52,60</sup>, often combined with substantial foraging subsidies for browsing and grazing herbivores<sup>61</sup>. These anthropogenic changes to landscapes and wildlife communities have led to the perception of an "urban deer (*Odocoileus spp.*) problem" <sup>62-64</sup>in wildlife management. Fifty-four percent of the world's population lives in urban areas and is expected to increase to 66% by 2050<sup>65</sup>. Within cities, low-medium density housing areas carry the highest likelihood of urban wildlife interactions due to high species richness and low species extinction rates <sup>66</sup> and the greatest areas of greenspace and diversity of landcover<sup>67</sup>. Yet people living within these low-medium density housing tend to react most negatively to human-wildlife conflict, reflecting a range of values around urban wildlife <sup>68</sup>.

One interesting outcome observed in urban ecology is the "luxury effect" wherein differences in affluence among neighborhoods generates differences in biodiversity<sup>69</sup>. Evidence for a luxury effect dates back thousands of years, arising from Egyptian archaeological records, and continue through the Anthropocene<sup>70</sup>. The luxury effect spans spatial scales, occurring both within and among cities<sup>66</sup>, albeit inconsistently. Among 20 North American cities studied <sup>71</sup> per capita income played a role in explaining vertebrate diversity in half; instead, species richness was highly (negatively) correlated with urban intensity<sup>72</sup>. Affluence is thus a proxy measure for biological properties associated with rich neighborhoods<sup>72</sup>: lower human density, energy subsidy, and especially greenness. Indeed, the luxury effect is generally amplified in arid environments<sup>70</sup>.

Most research on luxury effect uses species richness of plant or animal assemblages as the metric. For large mammals, individual behavior is a key mechanism explaining response to urban development<sup>8</sup>, so we examine luxury effect from this different angle.

We examine resource selection by urban black-tailed deer (*Odocoileus hemionus columbianus*; deer), a native to the western Nearctic including the Canadian province of British Columbia (BC). They are important prey for BC's diverse carnivore population <sup>1</sup> but the changing landscape has led to abundant urban deer. Predator persecution is an obvious culprit, but we suspect landscape change is an important driver. Deer select high-energy and high-nutrient plants as forage <sup>73</sup> and are very sensitive to factors affecting the recruitment of young deer into the breeding population<sup>5,74</sup>. The abundant backyard gardens of urban and suburban areas in affluent neighborhoods <sup>54</sup> provide ample deer food, potentially allowing deer to breed more often and more successfully than in 'natural' (non-urban) landscapes.

There exists a trade-off between security from predation and food resources which is not well understood even in natural, undeveloped systems<sup>75</sup>. How deer perceive risk in urban areas – and how they capitalize upon potential resource subsidies – remains unknown. Urban environments have been shown to impact wildlife behaviour, resulting in unique adaptations that differ from their non-urban counterparts<sup>76,77</sup>. Similarly animal behaviour and personalities influence the efficacy of behavioural tools for urban wildlife management such as hazing deterrents and culls<sup>8</sup>. A better understanding of urban deer resource selection, avoidance, and spatial landscape use would help determine if the luxury effect is impacting individual deer behavioural, and if so, what are some of the proximal mechanisms for this effect. This information is also an important tool for suburban deer management, both in terms of

minimizing the impacts on wildlife population processes as well as negative human-wildlife interactions<sup>78</sup>.

To evaluate our hypotheses about the luxury effect, we used resource selection function (RSF) analyses to make inferences about how black-tailed deer use different landcover features<sup>79,80</sup>. RSFs have been used extensively to assess animal movement patterns, response to novel anthropogenic features, and identify movement pathways<sup>81-86</sup>. We examined the role of (1) vegetation productivity and tree cover, (2) residential lot size, (3) road density, (4) golf courses and public green spaces, and combinations thereof. We hypothesized that if the luxury effect was apparent, then residential lot size would show the significantly positive effect size. We also predicted that road density was a risk deer avoided, and that native (parks) and non-native (golf courses) forage sources would be selected, but with smaller effect sizes.

#### Methods

#### Study Area

Our sampling frame is the western distribution of the Columbian black-tailed deer (*Odocoileus hemionus columbianus*) on the Nearctic Pacific coastline, within the dry Garry Oak (*Quercus garryana*) savannah<sup>87-90</sup>. This savannah's Canadian distribution includes the District of Oak Bay on Vancouver Island, British Columbia, characterized by small- and large-lot residential areas, urban development, golf courses, as well as managed and natural parks throughout. The district is bordered by the Salish Sea to the east and south (Fig. 1). In 2021, 17,990 people lived in its 10.5 km<sup>2</sup> and median household income exceeded \$100,000 CAN<sup>91</sup>. *Deer collaring* 

We deployed 20 LOTEK Lifecycle GPS collars (Newmarket, ON, CAN) collars on female black-tailed deer in Oak Bay in 2018. Collars were programmed to obtain a GPS location every 13 hours, providing 1-2 location fixes daily for six months. We opted for female deer as the reproductive component of the population is expected to be most susceptible both to risk and to available forage. Through February - March 2018 we conducted > 200 km of road-based surveys covering all of Oak Bay, every morning from sunrise to 11.00 hrs (when deer were most active), except in winds > 20 knots. We systematically searched the entire OB period multiple times over this span, ensuring that animals throughout the study areas had the opportunity for capture. However, captures are necessarily constrained to where deer occur (Fig. 2). Female deer were captured using chemical immobilization via darting with a Pneu-Dart - Model 389 Rifle with cartridge fired projector, 1cc Type C darts with a 1" needle, a wire barb, and a tri-port. Deer were immobilized by the wildlife veterinarian (AH) using 1ml of Wildpharm's BAM II Premix (27.3mg of Butorphanol, 9.1mg of Azaperone, and 10.9mg of Medetomidine) and reversed with 2ml of Atipamazole (50mg) administered intramuscularly (IM), and 0.5ml of Naltrexone (25mg) IM. Animals were moved to safety and blindfolded. Vital parameters including heart rate, respiratory rate, mucus membrane colour, and body temperature were monitored. Supplemental oxygen was administered via nasal cannula when needed. Collars were applied with unique coloured tags to allow field identification.



Fig. 1. We researched black-tailed deer resource selection in the District of Oak Bay, Vancouver Island, British Columbia, Canada. Oak Bay is bounded by the ocean (dark blue) to the south and east and by Greater Victoria to the north and west. Housing is depicted in grey, golf courses and private and public green spaces in green.

#### Deer home-range size

We used kernel density estimation (KDE) to estimate home-range size. We applied a smoothed "kernel" of equal-sized grid cells in a systematic pattern across a study region weighted by the density of observations and their locations<sup>92</sup>. We calculated the *97.5% isopleth* – the largest area used by the animal (within the chosen timeframe), excluding the extreme 2.5% of movements. Home range analyses were executed using ArcGIS v10.6 (ESRI, Redlands, CA, USA), using the Geospatial Modelling Environment package and the KDE tool.

#### Landscape Quantification

We quantified the landscape at deer locations in ArcGIS 10.6 and R (R Core Team, 2019) from publicly available digital datasets from BC provincial and municipal governments, Capital Regional District (CRD) Land Trust, Habitat Acquisition Trust (HAT) (Blyth 2013); and the US Geological Survey (USGS). Natural landcover features included vegetation greenness and tree cover. We derived Normalized Difference Vegetation Index (NDVI) from cloud-free USGS Landsat-8 2018 imagery for the summer growing season (April to September 2018) at 30m resolution. Landsat-8 imagery was downloaded from the USGS Earth Explorer (https://earthexplorer.usgs.gov/). We averaged NDVI values by pixel across images from each of the 6 months to determine one greenness value for each pixel on the finished dataset. NDVI measures vegetation greenness and can be used to assess vegetation location and density, infer water and nutrient levels, and infer vegetation structure (for example, irrigated grass is typically greener than coniferous forest). To represent vegetation structure in our models, we used a 100m resolution tree canopy cover raster dataset derived from high-resolution 2011 CRD aerial imagery by Habitat Acquisition Trust<sup>93</sup>. We used this percent tree-cover dataset in our RSF analysis.

We extracted residential zoned lots from zoning maps for the municipalities of Oak Bay, Saanich, and Victoria. We used a BC provincial dataset

(https://catalogue.data.gov.bc.ca/dataset/bc-parks-ecological-reserves-and-protected-areas) to quantify parkland and manually digitized golf courses. We discretized residential lots into three area classes – small, medium, and large –using the *Natural Breaks (Jenks) classification* algorithm in ArcMap. Small residential lots had mean area = 679 m<sup>2</sup>, s.d. = 176, min. = 202 m<sup>2</sup>, max = 1052 m<sup>2</sup>. Medium residential lots had mean area = 1495 m<sup>2</sup>, s.d. = 405, min = 1052 m<sup>2</sup>, max = 2523 m2. Large residential lots had mean area = 3601 m<sup>2</sup>, s.d. = 1609, min. = 2530 m<sup>2</sup>, max. = 14766 m2. We used a publicly available BC roads dataset

(<u>https://catalogue.data.gov.bc.ca/dataset/digital-road-atlas-dra-master-partially-attributed-roads</u>) to quantify roads, and a 100-m resolution impervious surfaces percent-cover dataset from Habitat Acquisition Trust (Blyth 2013) to quantify non-road urban features.

Following Zuur, et al. <sup>94</sup> we assessed all variables for collinearity (r < 0.7) and variance inflation (VIF < 3.0). Due to high correlation between impervious surfaces and other variables (e.g., small residential lots and roads), we excluded impervious surfaces from our analysis. We scaled all variables (mean=0, s.d. = 1) to allow comparison of effect sizes.

#### *Deer resource selection analysis*

We defined "used" locations as those where we received a successful location fix from any of our collared deer across the study period, for a total of 3,924 used locations. Available locations were randomly generated across the District of Oak Bay, bounded by the area for which we searched and captured deer. We generated available points in a 3:1 ratio, the number needed to achieve an asymptotic distribution of variable means<sup>95,96</sup>. For each used and available location, we calculated mean vegetation greenness (NDVI), and the percent area of tree cover within a 50-m radius buffer, a size selected to minimize error associated with GIS data resolution while also representing small-scale resource choice. We also calculated the percent area of three small, medium, and large residential lots, as well as road density (km/km<sup>2</sup>). Parks and golf courses were poorly represented inside buffer areas, so we measured proximity to these features.

We evaluated  $\beta$  coefficients from a single global model containing all selected landscape features. We chose not to do model selection as our goal was not to find the most parsimonious (reduced) model with a small subset of component variables, but rather to ascertain the selection strength of multiple variables<sup>97</sup>. We used a logistic regression in a generalized linear model (GLM with binomial errors and a logit link) with used locations (1's) and randomly selected available locations (0's) regressed against landscape covariates. We examined the second order of selection <sup>98</sup> which examines use within a group of animals: in this case the population of Oak Bay. We used k-fold cross-validation <sup>99</sup> to examine model fit and calculated odds ratios (OR) from  $\beta$  coefficients as  $e^{\beta}$ .

#### Results

#### Deer survival and data summary

GPS data collection ran March  $15^{\text{th}}$ , 2018, to December  $18^{\text{th}}$ , 2019. Not all deer were collared for the duration of the study - some deer whose collars transmitted to 2019.12.18 were collared part-way through the study after initial individuals died or had their collars fall off. In total 23 animals were collared during sampling; 20 individuals were retained for analysis with greater than or equal to 85 successful location fixes per animal, with mean fixes per animal = 271, min = 85, max = 495. "Of 20 individuals used in sampling, 5 individuals survived to winter 2018; 4 replacement collared individuals survived to spring 2019; 11 remaining individuals survived to winter 2019.

#### Space-use

Urban black-tailed deer occupied very small home ranges. Mean core home range size (50% isopleth) for the 20 collared does was  $0.14 \text{ km}^2$  (14 ha) (SD=0.07), and the mean outer home range size (95% isopleth) was  $0.64 \text{ km}^2$  (64 ha) (SD=0.31). Female deer displayed high site fidelity over the two-year period, staying centralized during that time (Fig. 2).

#### Deer resource selection analysis

Urban deer strongly selected large residential lots and areas of high vegetation productivity (or greenness; NDVI). Model validation did not indicate any model misspecification (Dsq = 10.21; overdispersion = 0.98; K-fold  $\Delta$  = 0.17). Deer were more than twice as likely to use an area with each unit increase in residential lot size ( $\beta$  = 0.89, s.d. = 0.04, OR = 2.44) and NDVI ( $\beta$  = 0.80, s.d. = 0.06, OR = 2.21) (Fig. 3). Deer also selected areas closer to public parks ( $\beta$  = 0.68, s.d. = 0.04, OR = 1.97) and golf courses ( $\beta$  = 0.68, s.d. = 0.05, OR = 1.97) (Fig. 3). Deer showed a weaker selection for small ( $\beta$  = 0.35, s.d. = 0.05, OR = 1.41) and medium-sized residential lots ( $\beta$  = 0.21, s.d. = 0.04, OR = 1.23). Deer strongly avoid areas with higher road densities ( $\beta$  = -0.03, s.d. = 0.04, OR = 0.74). Deer did not select or avoid area of treed cover ( $\beta$  = 0.01, s.d. = 0.05, OR = 1.01). Effect sizes varied among landscape features, and error was small (Fig. 4). Extrapolation of observed deer responses (i.e.,  $\beta$  coefficients) to natural and urban landcover covariates across our study area highlights affluent neighbourhoods in Oak Bay are most strongly selected by urban black-tailed deer, along with golf courses, followed by parks (Fig. 5).



Fig. 2. Home ranges of satellite-collared female black-tailed deer in the urban landscape of Oak Bay, British Columbia. Polygons are 97.5% kernel density estimates. Each individual deer is represented by a unique colour. Home ranges are based on satellite telemetry location fixes collected from February 2018 - March 2020. Main roads (black), residential roads (grey), and ocean (blue) are depicted. Roadless areas are golf courses and public parks.



Fig. 3. The probability of black-tailed deer selecting (or avoiding) natural and urban landscape features across Oak Bay, BC. Values greater than 0 represent selection for these features by deer (e.g., vegetation greenness, large residential lots) while values less than 0 indicate avoidance (e.g., roads). Values overlapping zero indicate neither avoidance nor selection (e.g., tree cover).



Fig. 4. Deer selection of landscape features in the urban landscape of Oak Bay, British Columbia, 2018-2022. Model  $\beta$  coefficients ("estimated selection", or effect size) are in blue; grey bands are standard error.



Fig. 5. Predicted black-tailed deer resource selection across the urban landscape of Oak Bay, BC based on extrapolated model  $\beta$  coefficients. Green areas are high use, fading to low use in red. Golf courses (purple) and parks (blue) are outlined. Roads are shown as red lines.

#### Discussion

The luxury effect – the positive relationship between urban biodiversity and human affluence – observed globally <sup>70,72</sup> manifests here in the behavioral resource selection by urban black-tailed deer. The greatest drivers of deer selection in this former savannah urban landscape are residential lot size and vegetation greenness (i.e., NDVI). Both are products of wealth. Oak Bay is a highly affluent neighborhood with 2022 house prices ranging from \$1-12 million CAD (mls.ca). Large lots (mean =  $3601 \text{ m}^2$ , s.d. = 1609, range 2530-14766) are highly manicured, with gardens offering abundant resource subsidies. Although the smaller "high density" lots in Oak Bay are not very small (mean =  $679 \text{ m}^2$ , s.d. = 176, range 202-1052) and still heavily gardened and watered they are still not as strongly selected as the large lots.

Deer selection for areas with high-productivity vegetation highlights the importance of high-quality forage availability in urban deer resource selection. Higher vegetation greenness is represented in areas with healthy and dense vegetation and is inversely linked to dry or drought conditions<sup>100</sup>. We observed a high correlation with vegetation greenness (measured as NDVI) and the normalized difference moisture index (Supplementary Information, Fig. 1), with higher soil moisture associated with higher vegetation greenness. The unirrigated regions of this landscape experience extended summer drought; Garry Oak meadows provide dry, nutrient-poor vegetation<sup>87-89,101,102</sup>. Parsing apart the effects of urban development is difficult in any system, as pre-development records are typically scant; but historically Cowan <sup>103</sup> observed "from the standpoint of deer the food potential of a west coast climax forest is so low that over vast areas deer are almost non-existent". At that time dense deer were only observed in regenerating forestry clearcuts. In the dry summer months, water was noted as particularly in demand, with a strong selection for plants in hygric areas<sup>103</sup>. Thus, following European colonization the

conversion of historically well-drained, drought-resistant Garry oak savannah ecosystems maintained by Indigenous peoples<sup>89,90,104</sup>, to modern watered lawns and high-productivity vegetation is a key contributor to urban black-tailed deer abundance. This supports other research that shows the luxury effect is most observed in arid and semi-arid landscapes, linked to irrigation and diverse plant communities<sup>70</sup>.

Beyond the effects of natural vegetation, large residential lots are also a highly significant predictor of urban deer habitat-use in our study area. Large-sized residential lots are embedded in neighborhoods of similar-sized lots, generating low housing density. Deer are therefore likely responding to the lower development density associated with these neighborhoods, as well as greater densities of high-productivity vegetation associated with larger residential lots, and vegetative cover on lots. Neighborhoods with larger lot sizes and high investment into landscaping – features generated by financial affluence – are therefore more likely to experience greater deer use, and to perceive these interactions negatively<sup>68</sup>.

Concentrated, high-quality resources mean deer can maintain smaller home ranges, which we observed here. Female urban deer home ranges were a quarter of the size of wild females in nearby Washington State<sup>105</sup>. Small home ranges in suburban environments have been noted by Happe <sup>106</sup> and Bender, et al. <sup>105</sup>. Ideal free distribution theory suggests animals occupy the smallest areas that provide the resources they require<sup>107,108</sup>, and this phenomenon has been noted for other deer species<sup>109</sup>. In natural (developed) landscapes black-tailed deer strongly select shrubs (as opposed to graminoids or forbs) in early successional conifer stands (14-20 years)<sup>110</sup>, and the abundance of hygric shrubs throughout large residential lots offers substantial subsidies that keep home-ranges small. People living in large residential lots tend to have the highest per capita income yet react most negatively to human-wildlife conflict<sup>68</sup>. Thus, citizen reports of

hyperabundant deer stem from peoples' negativity bias <sup>111,112</sup> and repeated sightings of the same deer.

Anecdotal citizen sightings suggested urban deer were strongly associated with golf courses and green spaces and our analysis corroborates this observation. Natural green spaces remain semi-arid oak savannahs but provide abundant escape cover; golf courses offer abundant well-watered grazing opportunities. Both parks and golf courses are elements of wealthy landscapes<sup>113,114</sup>, and the relationship between affluence and negativity bias towards urban wildlife <sup>68</sup> are likely to make these citizen sightings noteworthy.

Habitat selection by urban deer occurs on a predator-free backdrop. No wolves (*Canis lupus*), cougars (*Puma concolor*), or black bears (*Ursus americanus*) – primary prey of Columbian black-tailed deer – live in these urban landscapes, as none were ever detected on cameras. Wolves in particular regulate black-tailed deer; in wild landscapes on northern Vancouver Island wolf control was associated with increased deer numbers; modelling (and mortality observations,<sup>115</sup>) suggested that increased recruitment was the primary mechanism<sup>116</sup>. Therefore, a predator-free urban environment is expected to markedly increase recruitment (and hence abundance) over natural landscapes. However, predator regulation of black-tailed deer populations is mediated by forage availability and proximity to carrying capacity; dense deer populations are less affected by predators<sup>1</sup>. Even if urban environments did allow predators, the abundant resource subsidies are likely to sustain abundant deer populations.

#### Caveats

We collared female black-tailed deer as our goal was to monitor the space-use of deer before and after immunocontraception, the focus of a separate study. Male habitat selection may differ; however, we rarely observed males separate from females, even outside the breeding period, so male selection could be inferred with caution from our data. Seasonality in habitat use is not captured here, but seasonality is very mild in this system, ranging from rainy winters of mean lows of 5°C to dry summers with mean highs of 20°C. Our surveys did not note any marked changes in deer locations nor did our exploratory analysis. Finally, we did not examine spatially explicit attributes such as the juxtaposition of different habitat types, and this question is ripe for future investigation.

#### Conclusions

The luxury effect for urban wildlife manifests as strong selection by black-tailed deer for landscape features associated with wealth which provide resource subsidies. Large residential lots of the wealthy had the greatest positive effect on urban deer habitat selection, in combination with relatively undeveloped (uninhabited) green spaces and golf courses and with a smaller effect of smaller residential lots. Our research illuminates some of the mechanisms of the luxury effect of urban wildlife observed across the globe, which are driven by subsidies of water and vegetation<sup>70,72,113</sup>. Here, the conversion of historical drought-resistant Garry oak ecosystems into lush and landscaped urban environments have altered deer selection. The consequence for biodiversity more generally remains unknown; we can surmise greater biodiversity with luxury as observed elsewhere<sup>72,113</sup>. However, given observed the negative effects that highly abundant black-tailed deer have on plants and associated arthropod communities in nearby ecosystems <sup>117-</sup> <sup>119</sup> this is worth close examination. As urbanization continues to expand around the world, thrusting humans into higher densities and radically altering the habitats for millions of species, the mechanisms driving urban biodiversity should be a 21<sup>st</sup> century focus for wildlife ecology, so that future planning can effectively allow for coexistence of urban population and wildlife approximating as much as possible natural conditions.

## CHAPTER TWO: THE INFLUENCE OF REPRODUCTIVE STATUS ON URBAN BLACK-TAILED DEER HABITAT SELECTION

Isabel Giguère, Jason T Fisher, and Andrew Barnas

#### Introduction

In many species, reproductive females face unique trade-offs between accessing highquality forage and minimizing predation risk to offspring<sup>120,121</sup>. In urban species, this can result in increased human-wildlife conflict, necessitating population management interventions. However, understanding how reproductive status influences habitat selection is essential for predicting spatial distribution of reproduction success and informing useful management decisions<sup>120,121</sup>.

Habitat preference in deer is influenced by a combination of ecological pressures, including the availability of forage and the need for protective cover. Theories such as optimal foraging, landscape of fear, and central place foraging provide insight into how animals navigate these trade-offs in human-altered landscapes<sup>122-127</sup>. Research has shown that urban deer often select habitat that provides nutrient-rich forage with sufficient cover while avoiding areas to minimize risks like human-wildlife conflicts. This interplay between resource selection and avoidance should vary between reproductive groups because their demands for resources differ<sup>124,125</sup>. By understanding whether and how treated (non-reproductive) and untreated (reproductive) individuals select habitat differently, we can better anticipate potential changes to deer distribution and behavior following management interventions.

In this study, we investigate how reproductive status influences habitat selection patterns in Columbian black-tailed deer (*Odocoileus hemionus columbianus*), in Oak Bay, British

Columbia. We hypothesize that reproductive status drives key differences in habitat selection strategies, with mothers prioritizing habitats offering both forage and protective cover to support fawn survival, while non-mothers are likely to have \*riskier\* foraging behaviour, prioritizing forage availability over safety. We predict that mothers will more strongly avoid roads and areas with higher residential densities to reduce dangers associated with vehicles and human-wildlife conflicts. We predict that mothers will more strongly select for natural areas that contain vegetation types that provide concealment for fawns but nutrient rich forage for mothers. In this way, it satisfies both the high energetic demands of mothers while increasing fawn survival and recruitment. Since non-mothers do not have as strong energetic demands or pressure of raising offspring, we expect non-mothers to show weaker, less predictable selection patterns, avoiding only the most inhospitable habitats while capitalizing on a broader range of forage opportunities. By comparing habitat selection patterns between these two reproductive groups, this study aims to provide insight into the ecological impacts of IC management and inform future strategies for balancing deer population control with habitat conservation.

#### Methods

#### *Camera Trap Array*

The camera trap array consisted of 39 remote infrared cameras – Busnhnell (Trophy Cam HD Essential E2; L20 Prime Low Glow) and Browning (Dark OPs Pro 1080). Cameras were deployed in a systematic grid design to capture the entirety of the study area. The study area was divided into 400-m grid and a camera was placed within the confines of the grid cell. The systematic design was selected so that one individual could be detected on multiple cameras, which is necessary for density estimates<sup>128</sup>. In this urban environment, camera locations were highly dependent on landowner permissions, so cameras were not placed at the center of the cell in all cases. Cameras were often fixed to a tree, 0.5m - 1.5 m from ground level. This level allows the camera's field of view to capture the full body of an average-sized black-tailed deer. *Camera Image Processing* 

Images collected from cameras were processed using *Timelapse 2.0* software. We retained the same study periods of September for 2018 and 2021-2023 but used October-November for 2019 and 2020. September was primarily used as the focal period to optimize detectability and aging as fawns and adults are most mobile, and fawns still retain their spots<sup>120</sup>. Imagery was sampled in October-November for 2019 and 2020 due to logistic constrains associated with density estimates but are not expected to skew results for habitat selection as fawns are still distinguishable in October<sup>120</sup>.

The precise number of days that a camera was active was recorded to account for uneven sampling effort in models. For each deer(s) detected in an image, age (fawn or adult), sex (male or female), group size (number of deer), and unique tag identifications were documented (Figure. 2). We further categorized detections of deer (but not individuals: see next chapters) into two reproductive categories: non-mothers (lone female adults), and mothers (adult females with fawns). If a fawn was detected alone on a camera, this detection type was categorized as a mother event because it was assumed that the mother was close by, it just was not in the field of view of the camera. <sup>120</sup>We considered detections of the different demographic groups to be independent when separated by a threshold of 30 minutes<sup>25</sup>. For each camera site, we summarized the number of days that each demographic group was detected or not (i.e. present or absent that day).



Figure 2. Imagery of A) a PZP-treated doe without a fawn (non-mother) and B) Control doe with a fawn (mother).

#### Habitat Classification

Habitat features with varying degrees of forage quality and protective cover were classified to assess their influence on the presence of demographic groups. ARC GIS Pro v 3.2 software was used for all habitat feature extraction and geospatial statistics<sup>129</sup>. First, buffers were applied to both the study area and camera sites to define the boundaries for habitat feature extraction. A 400m buffer was applied to municipal Oak Bay boundary to ensure relevant habitat data was gathered for cameras at the periphery of the study area. A 150m circular buffer radius was applied around each camera and habitat values were extracted from within its boundaries. A 150m buffer radius is half the average distance between cameras, which minimizes buffer overlap from adjacent cameras while still being able to gather site-specific habitat values unique to the site. Landcover data was collected from a combination of municipal and provincial opensource geospatial databases. The *CRD 2019 LiDAR land cover dataset* was the primary source used (Table 2), chosen for its high-resolution accuracy in capturing current surface features
across the study area CRD, <sup>130</sup>. All habitat variables, except for residential lot density and proximity to green space was extracted from this dataset and represented as the proportion of surface area (m<sup>2</sup>) a given feature covered within the camera buffer.

Table 1. Definitions of habitat features hypothesized to influence habitat selection in reproductive groups.

| Predictor variable | Description  |  |  |  |  |  |
|--------------------|--|--|--|--|--|--|
| Bare ground        | Areas of exposed soil and bare land falling outside        |  |  |  |  |  |
|                    | agricultural land uses (e.g., construction sites, cleared  |  |  |  |  |  |
|                    | areas)   |  |  |  |  |  |
| Grass              | Grass land cover falling within residential and urban land |  |  |  |  |  |
|                    | uses, including lawns, gardens, playing fields and         |  |  |  |  |  |
|                    | institutional grounds. These areas represent lands subject |  |  |  |  |  |
|                    | to regular maintenance.                                    |  |  |  |  |  |
| Herbaceous         | Areas of natural herbaceous vegetation that is not         |  |  |  |  |  |
|                    | manicured (e.g., grasses, reeds, ferns, flowers and low-   |  |  |  |  |  |
|                    | lying vegetation).   |  |  |  |  |  |
| Shrub              | Shrub dominant areas (includes small trees) between        |  |  |  |  |  |
|                    | 50cm-3m in height. This does not include shrubs such as    |  |  |  |  |  |
|                    | hedges that are above 3m in height.                        |  |  |  |  |  |
| Deciduous tree     | Deciduous trees (and deciduous shrubs) above 3m in         |  |  |  |  |  |
|                    | height.  |  |  |  |  |  |

| Coniferous tree          | Coniferous trees (and coniferous shrubs) above 3m in   |
|--------------------------|--|
|                          | height.  |
| Pavement                 | Paved areas/impervious surfaces excluding buildings    |
|                          | (e.g., roads, sidewalks, driveways and parking lots).  |
| Road                     | All road networks excluding all other impervious       |
|                          | surfaces and paved areas.                              |
| Proximity to green space | The average distance to the closest three green spaces |
|                          | composed of 50% or more green area, excluding          |
|                          | residential green lots (e.g., parks, golf courses,     |
|                          | cemeteries, schools, regional trails).                 |
| Residential Lot Density  | Mean kernel density of residential lot parcels.        |

Mean residential lot densities were generated by kernel density estimation from residential lot parcel center points<sup>129</sup>. Center points were given to independent living units/dwellings. Civic addresses, satellite imagery, google street view and knowledge of the area were all used to guide decisions around defining a residential unit. For apartments and condos, the number of center points given were based on the number of dwellings on the first floor, that had their own entrance and/or green space. Consequently, every residential address was not assigned a center point to avoid over inflating residential lot density estimates for dwelling types that deer could not utilize. In this way, the ratio of living units and green space within a parcel was consistent among dwelling types. The kernel density tool was applied to residential center

points at a cell size of 10 and search radius of 200m. The mean lot density (residential dwelling/m<sup>2</sup>) values were calculated for each camera site.

The proximity to green space was generated by calculating distance from each camera to the nearest three green spaces. The green space layer was created by including all non-residential parcels that were 50% or more green space. To delineate these areas, an NDVI layer was created to enhance the visualization of vegetation productivity. The NDVI layer was then overlaid with zoning data and satellite imagery to refine polygon placement and define precise boundaries. The resulting green space layer included parks, golf courses, schools, cemeteries, and private water access points. The mean Euclidean distance was calculated for the nearest three green space polygons to each camera site and referred to as the proximity to green space.

#### Statistical Analysis

To evaluate the effect habitat has on site selection between mothers and non-mothers groups, candidate model sets were created for each reproductive group (Table 4). Generalized linear mixed-effect models (GLMMs) were used to test how reproductive status influenced habitat selection, based on a proportional binomial GLMM with a logit link function. All habitat variables were tested for Pearson correlation ( $|\mathbf{r}| > 0.6$  threshold)<sup>94</sup>. High correlation coefficients were found for roads and shrubs ( $\mathbf{r} = 0.62$ ), deciduous trees and roads ( $\mathbf{r} = -0.61$ ), and deciduous trees and pavement ( $\mathbf{r} = -0.74$ ), therefore these pairs were not included together in models to avoid multicollinearity<sup>94</sup>. All continuous predictor variables were scaled (mean=0, SD=1). Year was included as a fixed effect to control for the on-going IC treatment as well as annual variance in deer response to habitat variables. Site ID was included as a random effect in all models to account for multiple observations at each camera location (pseudoreplication).

Models were evaluated using Akaike's Information Criterion corrected for small sample sizes to gain empirical support for model hypothesis<sup>131</sup>, wherein the lowest AIC score represents the model that best explains the data. Models were fit separately for each reproductive group. Odds ratio (OR) were calculated to interpret effect size of predictor variables in each model and were plotted for the top 4 models for both groups. All analyses were conducted in R-Studio v4.4.1<sup>132</sup>.

Table 4. AICc scores of candidate models assessing factors expected to influence detections of mothers in the Oak Bay study area, British Columbia, Canada, from 2018–2023. The table includes the number of parameters (DF), log-likelihood and  $\Delta$ AIC, the difference in AICc scores between the best-supported model and all other models as well as model weight.

| Mother Models AIC Table       |   |    |           |          |        |      |  |  |
|-------------------------------|---|----|-----------|----------|--------|------|--|--|
| Model                         | Covariates DF Log-Likelihood A                  |    | AICc      | Δ AICc   | Weight |      |  |  |
| Danger                        | Road + Year                                     | 8  | -870.5812 | 1,757.89 | 0.00   | 0.61 |  |  |
| High-Quality Forage and Cover | Shrub + Deciduous Tree + Herbaceous + Year      | 10 | -869.4985 | 1,760.13 | 2.23   | 0.20 |  |  |
| Forage and Cover              | Shrub + Deciduous Tree + Year                   | 9  | -871.8148 | 1,762.55 | 4.65   | 0.06 |  |  |
| Residential                   | Lot Density                                     | 8  | -873.0095 | 1,762.75 | 4.86   | 0.05 |  |  |
| Low-Quality Forage            | Herbaceous + Grass + Year                       | 9  | -872.5552 | 1,764.03 | 6.14   | 0.03 |  |  |
| Vegetation Cover              | Coniferous Tree + Shrub + Deciduous Tree + Year | 10 | -871.6685 | 1,764.47 | 6.57   | 0.02 |  |  |
| No Forage or Cover            | Bare Ground + Pavement + Year                   | 9  | -873.1166 | 1,765.15 | 7.26   | 0.02 |  |  |
| Proximity to Green Space      | Nearest 3 Green Patches                         | 8  | -874.5542 | 1,765.84 | 7.95   | 0.01 |  |  |
| Null Model                    | -   | 2  | -971.5613 | 1,947.18 | 189.29 | 0.00 |  |  |

#### Results

#### Mother Habitat Selection Models

The model that best explains the presence of mothers is the Danger model (w=60%), followed by High Quality Forage and Cover ( $\Delta$ AIC=2.23, w=19%), Forage and Cover ( $\Delta$ AIC=4.65, w=6%) and Residential ( $\Delta$ AIC=4.86, w=5%) (Table 4). Together, these competitive models carry 90% of model weights in explaining the presence of mothers in Oak Bay (Table 4). Mothers strongly avoid danger, the odds in the form of roads. The odds of detecting mothers decreased by 41% for every unit increase in roads (OR: 0.59, CI:0.42-0.82, p=0.002) (Figure 3A). Mothers strongly select for High Quality Forage and Cover. The odds of detecting mothers increased by 35% and 43% for every unit increase in deciduous tree and herbaceous vegetation (OR: 1.35, CI: 0.96-1.90, p=0.087) and 43% (OR:1.43, CI:1.04-1.96, p=0.026) respectively (Figure 3A). The odds of detecting mothers (minimal effect of shrubs) decreased by 20% for every unit increase in shrubs (OR: 0.80, CI:0.56-1.14, p=0.211) (Figure 3A). Odds ratios were similar for deciduous tree and shrub predictors in the forage and cover model. Mothers avoid dense residential lots; the odds of detecting mothers decrease by 30% (OR = 0.70, CI: 0.49–0.99, p = 0.042) for every unit increase in lot density (Figure 3A).

Shrubs are a weak predictor of mother presence (compared to all covariates in competitive models), meaning that mothers do not strongly avoid or select this feature (Figure 3A). Therefore, shrubs may not be a reliable predictor of mother presence (Figure 3A). Alternatively, Herbaceous vegetation has a strong and predictable positive affect on the presence of mothers with fawns (Figure 4B, Figure 4C). The High-Quality Forage and Cover model showed superior performance with the additional herbaceous vegetation covariate, which further illustrates the strong influence of this predictor on mothers. Deciduous trees have a predictable positive effect on mother presence (Figure 4B, Figure 4C). Models that included deciduous tree and shrubs performed well, despite the weak predictor strength of shrub. This suggests that deciduous tree may be biologically more meaningful of a predictor of mother presence than numerically represented in models. The Forage and Cover model and the Residential model performed similarly, with comparable AIC scores (Table. 4). However, among the covariates in these two comparable models, lot density had the strongest influence on mother presence, with

the predicted probability of mother presence reliably decreasing along with lot density (Figure.4D).

#### Non-mother Habitat Selection Models

The model that best explains the presence of non-mothers is the Danger Model (w=36%), followed by No Forage or Cover ( $\Delta$ AIC=0.6, w=27%), Residential ( $\Delta$ AIC=1.95, w=14%), and Food and Cover Models ( $\Delta$ AIC=2.62, w=10%) (Table. 5). Together, these competitive models carry 87% of model weight in the candidate set that explain the presence of non-mothers in Oak Bay (Table. 5).

Non-mothers avoid danger. the odds of detecting non-mothers decreases by 21% for every unit increase in roads (OR: 0.79, CI: 0.63-0.99, p=0.039) (Figure 3B). Non-mothers also avoid areas void of resources with a similar trend. The odds of detecting non-mothers decreased by 15% and 14% for every unit increase in pavement (OR:0.85, CI:0.66-1.09, p=0.200) and Bare ground (OR:0.86, CI:0.67-1.11, p=0.252) respectively (Figure 3B). Non-mothers avoid dense residential lots as the odds of detecting a mother decreased by 15% for every unit increase in lot density (OR:0.84, CI:0.67-1.06, p=1.43) (Figure 3B). Non-mothers do select for vegetation that provides forage and cover as the odds of detecting mothers increases by 22% with deciduous trees (OR:1.22, CI;0.96-1.54, p=0.106), but the effect of shrubs is minimal (OR:0.94, 0.74-1.19, p=0.597) (Figure 3B).

Shrub is a weak predictor of non-mother presence, meaning that they do not strongly avoid or select this feature (Figure 3B). Most covariates in competitive models had a negative relationship with non-mother presence (Figure 5A-C) except for deciduous tree (Figure 5D). Non-mothers avoid most features but select for deciduous trees, there is a predictable positive effect on non-mother presence with deciduous tree (Figure 5D). Danger and No Forage or Cover model performed similarly with comparable AIC scores (Table 5). This suggests that they avoid the dangers associated with roads just as strongly as they avoid resource poor areas. The Residential and Forage and Cover model also performed similarly, with comparable AIC scores (Table 5). This suggests that they avoid dense residential areas just as strongly as they select for this resource subsidies. The strength of the relationship between non-mothers and all predictors was not strong. Although the results show that non-mother do prefer and avoid certain features, they do not have strong habitat preference compared to mothers.

#### Deer Occurrence Through Time

Time has a negative effect on deer presence across all years for both groups (Figure 3A, Figure 3B). Non-mother presence is significantly lower for the year 2020 (p<0.001) and 2023 (p<0.05) (Figure 3B,). The lowest detection counts for non-mothers in 2020 is consistent with the timing of IC administration and fawning estimates, indicating a sharp reduction in fawn numbers in 2019, thus less of these reaching adulthood in the following year (Table 1; Figure). Mother presence is significantly lower for every year (p<0.001) compared to the 2018 reference year, with more prominent reductions occurring in 2021 and 2022 (Figure 3A,). These results are consistent with the timing of IC administration as these years coincide with when most females had up to date vaccines.

Table 5. AICc scores of candidate models assessing factors expected to influence detections of non-mothers in the Oak Bay study area, British Columbia, Canada, from 2018–2023. The table includes the number of parameters (DF), log-likelihood, the difference in AICc scores between the best-supported model and all other models ( $\Delta$ AIC), and model weight.

| Non-mother Models AIC Table   |   |    |                |          |        |        |  |
|-------------------------------|---|----|----------------|----------|--------|--------|--|
| Model                         | Covariates                                      | DF | Log-Likelihood | AICc     | Δ AICc | Weight |  |
| Danger                        | Road + Year                                     | 8  | -954.5663      | 1,925.86 | 0.00   | 0.36   |  |
| No Forage or Cover            | Bare Ground + Pavement + Year                   | 9  | -953.7713      | 1,926.46 | 0.60   | 0.27   |  |
| Residential                   | Lot Density                                     | 8  | -955.5432      | 1,927.82 | 1.95   | 0.14   |  |
| Forage and Cover              | Shrub + Deciduous Tree + Year                   | 9  | -954.7829      | 1,928.48 | 2.62   | 0.10   |  |
| Proximity to Green Space      | Nearest 3 Green Patches                         | 8  | -956.5359      | 1,929.80 | 3.94   | 0.05   |  |
| High-Quality Forage and Cover | Shrub + Deciduous Tree + Herbaceous + Year      | 10 | -954.6934      | 1,930.51 | 4.65   | 0.04   |  |
| Vegetation Cover              | Coniferous Tree + Shrub + Deciduous Tree + Year | 10 | -954.7812      | 1,930.69 | 4.83   | 0.03   |  |
| Low-Quality Forage            | Herbaceous + Grass + Year                       | 9  | -956.4191      | 1,931.76 | 5.89   | 0.02   |  |
| Null Model                    | -   | 2  | -965.9239      | 1,935.91 | 10.04  | 0.00   |  |
|                               |   |    |                |          |        |        |  |

Odds Ratios for Models Predicting Habitat Selection in Mothers Shrub Roads Lot Density Herbaceous Mother Model Predictor Variables Danger Deciduous Tree Forage and Cover High Quality Forage and Cover 2023 Residental Year 2022 2021 2020 2019 0.5 1.5 2.0 1.0 Odds Ratio (95% CI)

A)



Figure 3. Odds ratio including 95% confidence intervals of habitat variables in top 5 models that predict habitat selection in A) mothers and B) non-mothers. The dotted red line represents no effect at a value of 1, values above 1 have a positive effect while values below have a negetative effect on the response variable. Values furthur from the red line indicate a stronger effect.



Figure 4. Marginal effect plot of covariates in top models for mothers. Plots represent the predicted probability of deer presence as a function of habitat type, while holding other variables constant at their mean values.



Figure 5. Marginal effect plot of covariates in top models for non-mothers. Plots represent the predicted probability of deer presence as a function of habitat type, while holding other variables constant at their mean values.

# Discussion

Habitat selection patterns differed notably between the reproductive groups, with mothers displaying stronger and more predictable preferences and avoidance behaviours than nonmothers. While both groups avoided roads, mothers exhibited a more pronounced avoidance, likely reflecting the need to reduce disturbance and perceived risk for their offspring. Similarly, both groups prefer larger residential lot sizes, but this preference was again more pronounced by mothers, suggesting mothers are utilizing the heightened forage and cover associated with larger lots that are attached to affluent neighbourhoods. In contrast, non-mothers demonstrated consistent avoidance of non-beneficial habitat features, regardless of danger, while mothers more reliably select for deciduous trees and herbaceous plants. This potentially indicates a more targeted strategy focused on securing high-quality resources. Mothers demonstrated a selection for resources that reduced risks to fawns and increased protection.

# CHAPTER THREE: BLACK-TAILED DEER POPULATION DENSITY ESTIMATE AND RESPONSE TO IMMUNO-CONTRACEPTION IN OAK BAY, BC

Andrew F. Barnas, Sandra Frey, Adam Hering, Joanna Burgar, Andrew Ladle, Alina C. Fisher, Macgregor Aubertin-Young, Jason T Fisher

# Introduction

Anthropogenic disturbance is a driver of global terrestrial biodiversity loss, but ensuing landscape change stemming from development can impact species in both negative and positive ways<sup>133-136</sup>. Some anthrophilic species, or human commensals <sup>137</sup> can benefit from anthropogenic activity which can create predator refuges <sup>138,139</sup> or provide increased resource availability<sup>140-142</sup>. Typically, the outcome of anthropogenic commensalism is population growth for those wildlife populations in developed environments<sup>137</sup>. Although in anthropogenic commensal relationships there is no loss to human populations, the increased population growth of commensals species can be perceived as threatening and translate to human-wildlife conflicts<sup>11,65,77</sup>. Conflicts are pronounced in urban and suburban areas of intermediate human population is expected to increase by 2.5 billion people over the next 30 years <sup>46</sup> and urban landscapes have doubled in the last few decades<sup>48</sup>. As such, there is a growing need to understand the population dynamics of anthropogenic commensals and explore strategies for mitigating human-wildlife conflict.

Population growth is the outcome of births, deaths, immigration and emigration, and substantial literature has been devoted to the primary mode of experimental wildlife population reduction<sup>143</sup>, specifically increased mortality through harvest <sup>144</sup> or culls<sup>145</sup>. Less studied are the population outcomes of reducing populations through non-lethal means, such as suppressing

reproductive output through fertility control. The paucity of such research is in part owing to fertility control being used often as a management tool without a solid ecological research framework<sup>17,146-148</sup>. Even less research has focused on monitoring population response when management intervention actions such as fertility control are paused or ceased entirely, allowing for a potential population rebound through immigration or increased births.

Here, we examine applied reproductive control and density estimation methods to evaluate population outcomes using Columbian black-tailed deer (Odocoileus hemionus columbianus) as a model organism. Black-tailed deer range across western North America with subspecies exhibiting marked morphological and behavourial differences<sup>5</sup>. Columbian blacktailed deer (hereafter "BTD") live in the western coastal regions of the Nearctic and are a native deer to British Columbia (BC), Canada. The changing landscape of BC's urban and suburban areas has generally benefited BTD populations where they live as anthropogenic commensals, exploiting increased foraging opportunities <sup>2</sup> and predator extirpations. Native predators such as wolves (*Canis lupus*), and cougars (*Puma concolor*) are kept at low density from most urban areas, effectively excluding them from their ecological roles in deer-population control<sup>3</sup>. An additional benefit of suburban/urban environments to BTD populations is the presence of highenergy and high-nutrient forage plants. Urban and suburban areas may contain abundant backyard gardens and agricultural crops, which provide ample food resources for deer, potentially allowing BTD to breed more often and more successfully than in unaltered landscapes<sup>4</sup>. In BC's suburban environments, deer have been shown to select for areas with high vegetation greenness, a high proportion of large-sized residential lots, and proximity to parks and golf courses<sup>2</sup> suggesting that human-driven changes to the landscape are key to maintaining urban deer populations. As BTD populations are sensitive to factors affecting recruitment<sup>5,6</sup>, the

lack of natural predation coupled with the abundance of high-quality food resources on (sub) urban landscapes has likely contributed to greater reproductive output and subsequent population growth for BTD.

Killing deer is the most direct and immediate method of reducing deer population density, including in urban and suburban environments<sup>13,149</sup>. However, the culling of wildlife can be a contentious and politicized issue<sup>150</sup>, leading to divided communities and even legal battles<sup>15,16</sup>. Fertility control may be used as a non-lethal strategy for reducing population size and immunocontraceptive (IC) treatments are used on free-ranging ungulates to reduce reproductive output<sup>17-</sup> <sup>19</sup>. Immuno-contraceptive vaccines trigger the female immune system to prevent fertilization of the egg<sup>20</sup>, and vaccines such as porcine zona pellucida (PZP) have been applied to various urban deer populations as means of non-lethal deer population management<sup>18,21-23</sup>. Prior to the fall reproductive season (i.e., rut), IC vaccines can be delivered remotely to adult does to suppress her ability to produce young the following spring, thus eliminating the need for lethally removing individuals in a growing population. Over time, reduced reproductive output should translate to reduced population density as adults die from natural and other causes (e.g. vehicular strikes) and are not replaced by recruited offspring. While fertility control is suggested to be a socially acceptable and effective method for wildlife population management compared to culls, the effectiveness of IC treatment to suppress population growth in BTD in urban environments has not yet been examined. The rarity of such research has likely partly been due to the past inherent difficulties in estimating population size and density for free-ranging populations<sup>34</sup>.

Remote wildlife cameras <sup>25,136</sup> offer a cost-effective opportunity for assessing population size and monitoring population response following intervention strategies such as fertility control. Their reliability in surveying wildlife species can be quantified <sup>27</sup> and cameras have been

shown to have very high probability of detecting deer when present (sensu MacKenzie, et al. <sup>151</sup> even in remote wild environments<sup>28,29</sup>. Camera surveys are an increasingly common approach for surveying ungulate occurrence <sup>30-32</sup> and they have been used to monitor deer in urban areas<sup>33,152</sup>. The concurrent advent of novel statistical techniques has provided important opportunities to collect reliable population density estimates from camera data, allowing for the comparison of urban deer population response before and after management actions.

Here, we apply IC to a free-ranging population of urban BTD and monitor via remote cameras the population response over a course of five years to understand the demographic and population outcomes of reduced reproduction in an open population with opportunities for immigration and emigration. Reducing reproductive output could result in a reduced overall population density wherein most animals stay in place and mortality exceeds replacement. Conversely, suppressing reproductive output could induce a functional response wherein increased immigration from adjacent areas results in an equilibrium population size. Likewise, a pause in management intervention could also result in a rapid rebound of the population size following increased reproductive output from deer previously treated with IC. Under natural conditions it is challenging to measure reproductive output as well as immigration and emigration of many individuals, but we can measure 1) the relative abundance of fawns and adults, and 2) the population density of adults. To examine these phenomena, we use an array of remote cameras to collect images of BTD adults and fawns, making use of unique identification applied during capture of control and IC treatment animals. We hypothesize that relative fawn abundance will rapidly decrease following IC treatment, as fawn production should be suppressed for IC treated females. However, we hypothesize adult population density reductions will be slower following IC treatment, as any reductions in density must be due to natural adult

mortality without replacement by incoming fawns. Following the pause in IC treatment, we expect a marked and rapid increase in fawn abundance but a slower and delayed increase in adult population density.

#### Methods

#### Study Area

This research took place in the 10.5 km<sup>2</sup> District of Oak Bay on Vancouver Island, British Columbia (Figure 1), one of 13 contiguous municipalities within the Greater Victoria region. The study area is largely urban environment, dominated by small and large residential lots, commercial development, golf courses, and district-managed natural park areas. The eastern and southern edge of the district is bordered by the Salish Sea, whereas the northern and western edge are bordered by the municipalities of Saanich and Victoria, respectively. The district is home to approximately 18,000 people as of 2021 (Statistics Canada 2021). The Oak Bay landscape is described comprehensively in Fisher, et al. <sup>2</sup>.

#### Deer Capture, Marking, and Immuno-contraceptive Treatment

In February-March 2018, we deployed 20 GPS collars on female BTD to collect movement data and mark a control (untreated) population to determine a baseline density estimate of deer prior to IC treatment. We applied Lotek Lifecycle satellite collars (Newmarket, Ontario, CA) fitted with coloured plastic tags to allow for individual identification of marked deer. Data collected by the satellite collars were used to examine urban deer habitat use<sup>2</sup>.

We commenced our IC treatment program in September 2019 and administered the PZP IC-vaccine Zonastat-D to adult female BTD prior to the fall rut. We selected for mature (>1.5 years old) does based on body size and/or presence of fawns (minimum ages confirmed using dental eruption). We searched for deer in the early morning by conducting road surveys

throughout the entirety of our study area to attempt an even coverage of treatment across Oak Bay, but targeted areas of known high deer congregations. Once a target female was selected, chemical immobilization drugs were delivered via telemetry darting. On capture, vaccination consisted of 0.5cc of lyophilized PZP solution reconstituted in sterile saline and then emulsified in 0.5cc modified Freund's complete adjuvant hand injected in a gluteal muscle. Captured deer were marked using a combination of coloured marker collars and/or numbered ear tags to allow for individual-level identification (Figure 1C). When possible, a follow-up booster dose of the same PZP solution emulsified in 0.5cc Freund's incomplete adjuvant was administered by remote drug delivery (darting) 2-6 weeks after initial capture.

In September-October 2019, we administered IC to 60 female BTD, and two to six weeks after treatment with the primary PZP vaccine, we were able to locate 55 of those 59 initially treated deer to administer a booster of the same vaccine. The primary and booster vaccine had different adjuvants as per the standard PZP protocol. Booster vaccination was delivered remotely via darting. In September-October 2020, we administered IC vaccines to an additional 60 female BTD untreated the previous fall: most were likely yearlings from 2019, with some putative immigrants. Of these 60 newly treated individuals, we administered booster vaccines to 57 individuals. We also administered booster vaccines to 48 of the deer treated with IC in 2019. In September-October 2021, we did not administer primary IC vaccines to any previously untreated female BTD but administered IC booster vaccines to 71 of the 120 deer treated in fall 2019 and 2020. This totalled to 120 deer marked for IC, plus an additional 19 "control" deer marked (or remarked) in spring 2020 following the pre-programmed release of the GPS collars 2-years after deployment.

Study Design and Camera Monitoring

We used a systematic study design wherein 39, 0.16-km<sup>2</sup> cells were overlaid on the entire  $10 \text{ km}^2$  landscape. In August 2018, one Bushnell camera (Trophy Cam HD Essential E2 or L20 Prime Low Glow) was deployed within each cell, as close to the centroid as possible while allowing for logistical difficulties on both public and private properties across Oak Bay. Cameras were secured to a tree approximately 0.5 - 1.5 m off the ground with no attractants used. Cameras were programmed to take three pictures in sequence when movement was detected by the infrared trigger, followed by a ten second trigger delay. Whenever possible, cameras were aimed towards areas where deer were evidenced or assumed to have travelled through to maximize detections of deer individuals present in the surrounding area. Following a targeted theft of cameras in winter 2019, we moved some cameras from public properties to private and replaced older Bushnell units with Browning Dark Ops Pro x 1080 cameras using the same settings. Thus, sampling locations differed slightly between years, but cameras remained within the same designated areas.

Species and demographic data were extracted from imagery using Timelapse Image Analyzer 2.0 <sup>154</sup> and reviewed by the same person for all years of data. For each deer detection on camera, we collected information on the camera location, date, time, the age, sex, and number of adult deer, and the number of fawns present. We classified deer as fawns if they were less than one year old, which we inferred by a combination of their body size, stature, presence of white spots, and snout-to-head length ratio. We classified yearling deer as adults because yearlings are reproductive by their second fall when we surveyed them. For marked deer, we also noted any information on ear tag number/colour and marker collars to allow for individual identification, which provides the basis of our population density estimation models prior to- and after IC treatment.



Figure 1 Monitoring of urban Black-tailed deer population in Oak Bay, British Columbia. A) Location of remote cameras. Note camera locations changed slightly within each year; see main manuscript for further details. B) Example remote camera setup. C) Example image of a marked deer with an identifiable ear tag and collar combination.

Due to the significant time investment of manually processing camera imagery and the substantial number of images containing deer collected across cameras each week, we subset our dataset to only examine deer detections for a single month each year in the fall. We initially selected September as does are easily distinguished from bucks due to the presence of antlers, while fawns (<1 year) are sufficiently mobile but can still be distinguished from yearlings (>1 year). However, as we were marking newly treated deer in September 2019 & 2020, we chose to review imagery from a 35-day period from October 8<sup>th</sup> to November 11<sup>th</sup> in 2019 and 2020, when the same advantages in age/class identification hold true, and when the marked population was stable following completion of the primary IC vaccines (and associated markings) being administered.

We considered detections of deer at a camera to be independent if consecutive images were > 30 minutes apart, or if a different individual could be distinguished (based on sex, age class, or markings). For example, if an unmarked deer was detected on a camera at 10:15am, and another detection of an unmarked deer occurred at 10:30am, these would be considered part of the same detection as there is no way to distinguish the individuals. Conversely, images collected at 10:15am and 10:50am would be considered two independent detections.

# Relative Fawn and Adult Abundance

To examine the impact of IC treatment on the relative abundance of fawns and adult deer, we examined the total independent detections of deer with and without at least one fawn present. Using the 30-minute independent detection threshold described above, across all sites each year we calculated the proportion of detections only containing adults (i.e., no fawn was detected in any images), and those which contained both adults and fawns. Some events did contain only

fawns in the images (i.e., no adult deer were observed), but we did not consider this as a separate category as we assumed an adult was nearby and just not detected.

Next, we examined independent detections of each group (adults only vs fawns present) at each camera site. We divided the number of detections at each site by the number of days the camera was operational. Due to clear influences of outliers in these data during initial exploration, we removed outliers exceeding 1.5 times the interquartile range separately for each group. These outliers were caused by one or few individual deer that tended to rest directly in front of the camera daily.

# Statistical Analysis – Adult Population Density Estimation

For the analysis of population density, we again used independent detections based on the 30-minute threshold described above, but we also considered information on marked individuals to delineate separate events. Images containing multiple identifiable individuals were recorded as unique events, as we could clearly identify the time and location for everyone. Further, images containing both marked and unmarked individuals were separated into distinct counts of a marked and unmarked detection. We did not consider independent events of fawns for density estimation.

We estimated population density of adult BTD using spatially explicit mark-resight (SMR) models with package *secr* v4.6.9 in R Studio 4.4.1<sup>155,156</sup>. SMR models have been developed to estimate density from repeated detections of known individuals in a partially marked population<sup>34,157,158</sup>. SMR models use repeated detections, or "resights", of both unmarked and marked individuals to estimate the density of a population<sup>36,158</sup>. We fit a single multisession model to estimate density separately for each study year. Each study year (session) contained 30 or 35 sampling occasions, but varying number of cameras (detectors). While this model assumes

individuals are available for detection in all years, initial density estimates from single-year models had poor precision due to less data available in any given year. We therefore elected to use a multisession model which can share information across all sessions. We specified a half-normal detection function, assuming detections of individuals are maximized closer to activity centers, but decay based on a scale parameter. We treated cameras as "count" detectors which are a class of "proximity" detectors which record animal presence at a point without restricting movement, allowing for >1 detection of animals per sampling occasion. Counts were modelled assuming a Poisson distribution.

We applied a discrete habitat mask for all years to avoid estimating density in areas of unsuitable habitat (i.e., the adjacent Salish Sea near the study area). We fit exploratory models using a 1000m buffer around the sampling area to examine the initial sigma estimates and suggested buffer size in relation to effective sampling area plots<sup>159</sup>, which reveal insensitivity of density estimates to changes in larger buffer size in all years (). Based on these retrospective checks, we chose the conservatively large buffer size of 945m for all years. This buffer was greater than 4 times the maximum initial estimate of sigma as suggested by Efford <sup>160</sup>. We then took this buffered area in each sampling year, and clipped out the adjacent ocean habitat, and applied the habitat mask to the model. We first fit models assuming no overdispersion to generate initial parameter estimates. We then used those initial estimates to simulate data 10,000 times using the "nsim" argument in the "secr.fit" function. We then refit the model using a pseudolikelihood adjusted for the estimated level of overdispersion from simulations. For each year we obtained a mean density estimate of adult deer  $\pm$  95% confidence intervals, converted to estimated deer per km<sup>2</sup>.

# Results

During the six-year study period, we collected data from 206 cameras representing 5,705 camera days in the autumn of each year (Table 1). Mean deployment periods and total number of active cameras varied each year due to camera failures.

| Year                     | Number of | Camera | Mean             | Range of | Independent | Independent |
|--------------------------|-----------|--------|------------------|----------|-------------|-------------|
|                          | active    | days   | deployment $\pm$ | days     | Detections  | Detections- |
|                          | cameras   | -      | sd               | -        | – adults    | with fawns  |
|                          |           |        |                  |          | only        |             |
| 2018*                    | 34        | 949    | $27.9\pm5.0$     | 1 - 30   | 706         | 592         |
|                          |           |        |                  |          |             |             |
| <b>2019</b> <sup>¥</sup> | 32        | 1047   | $32.7\pm7.0$     | 3 – 35   | 1166        | 434         |
|                          |           |        |                  |          |             |             |
| <b>2020</b> <sup>¥</sup> | 36        | 1022   | $28.4\pm9.6$     | 6-35     | 1049        | 248         |
|                          |           |        |                  |          |             |             |
| 2021*                    | 34        | 720    | $21.2 \pm 11.1$  | 1 - 30   | 810         | 187         |
|                          |           |        |                  |          |             |             |
| 2022*                    | 36        | 1004   | $27.9\pm6.7$     | 3 - 30   | 670         | 396         |
|                          |           |        |                  |          |             |             |
| 2023*                    | 34        | 963    | $28.3\pm4.5$     | 11 - 30  | 603         | 375         |
|                          |           |        |                  |          |             |             |

Table 1. Summary of camera deployment in Oak Bay, British Columbia, from 2018 to 2023.

\* September 1–30

<sup>¥</sup> October 8 – November 11

# Relative Fawn and Adult Abundance

Based on independent detections of adults and fawns from cameras, it appears that the relative abundance of fawns was initially higher at 0.46 fawns / camera trapping day in 2018 and 0.27 fawns / camera trapping day in 2019 prior to IC treatment (IC was first delivered in 2019 but there would not be an effect on reproduction until the following spring). Then, in 2020 and 2021, the two years following the initial IC treatment in the population, proportion fawn abundance was at its lowest at 0.19 fawns / camera trapping day in both 2020 and 2021. The proportion of fawns then increased in subsequent years to near pre-treatment level proportions of 0.37 fawns / camera trapping day in 2022 and 0.38 in 2023 (Figure 2A). Some variability is apparent when considering independent detections adjusted for camera trapping effort (i.e. relative abundance), but the same general pattern is retained (Figure 2B). In 2018 and 2019 prior to treatment we found the highest mean ( $\pm$  SD) number of fawn detections per camera day (0.34  $\pm$  0.34, and 0.27  $\pm$  0.25 respectively), and the two lowest in the two years following treatment (2020: 0.15  $\pm$  0.15, 2021: 0.13  $\pm$  0.19).

#### Adult Deer Density Estimates

Population density estimates prior to and during the first fall of IC treatments were initially high: 18.5 adult deer/km<sup>2</sup> [95% CI: 15.9 - 21.6] in 2018 and 23.6 adult deer/km<sup>2</sup> [95% CI: 20.6 - 27.0] in 2019. However, following the first year of IC treatment, population density estimates became progressively lower with 20.0 adult deer/km<sup>2</sup> [95% CI: 17.2 - 23.2] in 2020,

18.9 adult deer/km<sup>2</sup> [95% CI: 16.1 – 22.2] in 2021, 14.4 adult deer/km<sup>2</sup> [95% CI: 12.4 – 16.8] in 2022, and 13.2 adult deer/km<sup>2</sup> [95% CI: 11.2 – 15.6] in 2023 (Figure 3).



Figure 2. Independent detections of deer with and without fawns present. A) the proportion of detection types across all sites each year, B) violin plot of the number of detections at each camera site adjusted for the number of days each camera was active, with outliers removed.



Figure 3. Spatial mark-resight population density estimates (per km<sup>2</sup>) of urban black-tailed deer in Oak Bay, British Columbia, from 2018-2023.

# Discussion

Effective mitigation of human-wildlife conflicts requires balance between scientifically evidenced practices and values of local communities, which in the context of urban deer management may require the need to reduce population density using non-lethal approaches. Here, we observed an apparent reduction in relative fawn abundance with three years of fertility control, which contributed to a near 50% reduction in adult deer population density within four years. Reproductive output appeared to rebound immediately following a reduced effort in IC treatment in 2021 with an increase in relative fawn abundance in 2022 when a smaller proportion of the previously treated population received an IC booster, and no newly recruited adult does were treated. This indicates that intermittent fertility control administered in the absence of

ongoing treatment may have limited effectiveness as a long-term urban deer population management strategy. Moreover, in the absence of preventing immigration of deer from neighbouring environments, concurrent strategies of fertility control and management of deer attractants (e.g. irrigated landscapes<sup>161</sup>) may be required to maintain an overall suppressed deer population size. Nonetheless, as similar issues around population rebounds are expected across both lethal and non-lethal strategies for wildlife population control, our results highlight IC vaccines as a potentially effective method for decreasing urban deer population density in the near-term.

The effectiveness of IC treatment as a sole strategy for suppressing urban deer populations likely relies on an annual administration of booster vaccines to maintain the immune response in previously treated animals as well as primary vaccines to newly recruited or immigrated individuals. Booster vaccines are required to maintain the animal's immune response preventing fertilization of the egg. Gionfriddo, et al. <sup>162</sup>showed a single IC vaccine of a gonadotrophin-releasing hormone resulted in 88% of treated deer not becoming pregnant in the first year following treatment, but this dropped to 47% in the second year. Importantly those authors did not examine effects on reproductive output or population density, but our findings of increasing proportion fawn abundance to pre-treatment levels following a reduced IC treatment effort suggest that adult population density will increase in future years if IC intervention does not continue. Previous research on closed populations of white-tailed deer (Odocoileus virginianus) suggest if 25-50% of females are treated annually, reductions of 30-60% could be achieved over a 4–10-year period<sup>163</sup>. The sharp rebound in reproductive output observed in our study population indicates that a higher proportion (>50%) of females may need to be treated to maintain a suppressed overall population size. In open populations where immigration may lead

to increased overall population density through movement of adult deer as well the introduction of untreated adult females producing fawns for subsequent recruitment into the adult population, administering IC boosters to previously treated does in the absence of also administering primary vaccines to untreated individuals may be insufficient to maintain a suppressed overall population density. Ongoing monitoring of the Oak Bay urban deer population following the reduced effort in and subsequent pause of IC treatment will provide insights into the effectiveness of IC as a long-term population management strategy. However, in the absence of knowing the extent of immigration into our study area from neighbouring deer populations, disentangling the effects of immigration versus rebounding reproductive output will remain challenging.

# Caveats

Our camera array proved effective for monitoring urban BTD, providing high detections of both marked and unmarked individuals. As the number of marked individuals increased throughout the study, the number of detections of marked deer increased, providing greater precision during data exploration and model development. The decision to use a single multisession density model, as opposed to several single-year models, improved precision due to the greater availability of data but assumes individuals were available for detection in all years. We know this assumption was violated in several instances due to lost tags and natural mortalities. Although we recommend thoughtful interpretation of our results, given our trend in density estimates logically aligns with the proportion of adult and fawn detections on camera, we are confident our findings reflect the impacts of IC treatment on urban BTD.

# Conclusions

Our study indicates IC as an effective short-term strategy for reducing urban deer population density, as indicated by a sharp reduction in mean density estimates between 2019

and 2023. IC may be an effective long-term strategy if most of the female population receives treatment in the form of primary vaccines (for newly recruited/immigrated individuals) and/or boosters (for previously treated individuals) every three years. Maintaining suppressed deer densities in urban environments is likely best achieved through a combined effort of fertility control and attractant management, as elements of the urban landscape (e.g. landscapes and irrigated lawns) likely support higher reproductive output and incentivise immigration (*sensu Fisher, et al.* <sup>161</sup>). Future work will examine population responses following the several years since the ongoing pause of IC treatment, providing insights on how quickly populations may rebound following this management technique.

# Acknowledgments

This study was conducted by University of Victoria's ACME Research Lab in collaboration with the Urban Wildlife Stewardship Society, the District of Oak Bay, and the Province of British Columbia, with funding provided by Oak Bay, the Provincial Urban Deer Cost-Share Program, the UWSS, and the Natural Science and Engineering Research Council of Canada, through their Alliance Grants Program. A sincere thanks to all property owners, private businesses, and clubs across Oak Bay who have hosted our deer cameras for the past 6 years and continue to support this project. Our deepest gratitude to the countless volunteers who spent hundreds of hours cataloguing deer images and assisted with locating and capturing deer for IC. We also thank all the Oak Bay landowners who granted access to their property to locate and dart deer to administer IC. Figure 1 was created using the Canadian provinces and territories layers accessed via ESRI online<sup>164</sup>. Silhouette graphics in Figure 3 created by Gabriela Palomo-Munoz under CC license https://creativecommons.org/licenses/by-nc/3.0/. The writing of this manuscript was aided by the Lofi Hip Hop radio channel on Youtube.

# CHAPTER FOUR: USING CAMERA TRAPS TO EVALUATE EFFECTIVENESS OF IMMUNOCONTRACEPTIVE TREATMENT FOR REDUCING REPRODUCTIVE SUCCESS IN TREATED INDIVIDUALS

Andrew F. Barnas, Sandra Frey, Jason T. Fisher

# Introduction

Reducing reproductive output is a common goal of wildlife management programs, often used to mitigate human-wildlife conflict in urban settings while minimizing the need for contentious lethal removal programs<sup>15,16</sup>. Immunocontraception has emerged as a promising tool for reducing reproduction in urban deer populations, as initial treatments are less invasive than surgical sterilization <sup>165</sup> and follow up vaccines can be delivered remotely by darting. Previously, we evaluated the effectiveness of IC methods at the population level and demonstrated a reduction in adult deer density following treatment years. However, determining the efficacy of IC at the individual level remains unclear. While indirect measures such as declines in population density can suggest successful treatment, these do not confirm whether treated individuals failed to reproduce.

The objectives of this report are to examine the efficacy of IC at suppressing reproductive output in treated individuals. Specifically, we use camera traps to investigate whether IC-treated individuals are detected more frequently in images without fawns, than are untreated deer in the population. We hypothesize IC-treated individuals would be sighted more frequently without fawns than with fawns, which would provide evidence that treatment reduces fawn production in treated individuals.

#### Methods

The study design of this report mimics the previous section on estimating population density responses of black-tailed deer to IC treatment. Briefly, we used the same datasets of deer detections (same camera numbers, locations, and deployment periods) to examine detections of IC-treated individuals and untreated individuals.

Images of deer were classified into independent detections using a commonly accepted 30-minute classification threshold<sup>166,167</sup>. For example, a sequence of images collected at 10:15am and a sequence at 10:30am would be considered the same event and treated as the same bin of images. For the sequences of images collected within an independent event, we determined if an unmarked adult, a marked adult from the control group, or a marked adult from the treated group was present. Noting that the same event may contain multiple adults and thus multiple groups. We then determined if any fawns were present within that sequence of images. Independent detections with only fawns and no adults present were very rare, and we did not consider them for this report.

For independent events, we tracked known individuals from the year they were first marked, and then in all subsequent years throughout the study to compare how often they were detected with and without fawns. In 2018, a group of control deer were captured and marked, but no IC treatment was applied. These deer serve as a control group which provides baseline data, and no additional control deer were added in subsequent years. We compared the proportion of detections of lone adults versus adults with fawn for 2018, monitoring the same individuals from 2019 to 2023. In 2019 the first deer for IC treatment were captured, marked, and treated. We compared proportion of 2019 IC-treated adults and fawns from their first year of 2019, and then subsequent years from 2020-2023. For deer first marked and treated in 2020, we compared their

first year of 2020 to subsequent years from 2021-2023. Unmarked individuals were compared as a summed group for all study years (2018-2023) and then each year separately.

In some cases, marked individuals were known to have lost marks or were killed during the study period. This information was recorded but is not considered in our comparisons as we compare proportions for detections of marked and unmarked individuals in each year.

# Results

### Control vs. Treatment Individuals

The proportion of detections of adult deer with fawns present did not differ between control deer in 2018 and those same individuals from 2019 to 2023 (Figure 1). As expected, the control deer proportions also did not differ from the unmarked individuals from 2018 to 2023. Deer initially treated with IC had a lower proportion of fawn detections in both initial treatment years of 2019 and 2020 compared to control deer in 2018. While deer treated in 2019 showed a decrease in proportion detections of fawns in subsequent years (2020-2023), the deer initially treated in 2020 showed an increase in the proportion of fawn detections in later years (2021-2023).

### Unmarked Individuals

Although we expected no change in the proportion of fawn detections in unmarked individuals, we observed notable lower proportions in 2019- 2021 compared with the baseline measures from 2018. (Figure 2). Proportion fawn occurrence rebounded in unmarked deer in 2022 and 2023.



Figure 1. The proportion of independent detections of adults only (green) and adults with fawns present (yellow) across the pooled population of control, treated, and unmarked black-tailed deer from 2018 to 2023 in Oak Bay, British Columbia.



Figure 2. The proportion of independent detections of adults only and adults with fawns present for unmarked black-tailed deer from 2018 to 2023 in Oak Bay, British Columbia.

# Discussion

IC is effective at reducing fawn production in treated individuals, but this is a signal we could detect only at the population. This is satisfactory, as our goal was always to assess population-scale effects, and not the efficacy of the IC vaccine.

Camera traps are not an effective tool for tracing reproductive output at the individual level. This was due in large part to the sociality of deer, shared maternal care, and behavioural associations between breeding and non-breeding individuals, which obfuscated our ability to determine doe-fawn relationships when multiple individuals were present within an image (or suspected to be present outside the camera field of view). If IC-treated deer without fawns associate with breeding mothers, then by chance alone we would fail to detect a signal of decreased fawn production in treated individuals. Parsing different demographic groups through camera trap data can be challenging for social animals featuring young<sup>168</sup>. Misclassifications of marked individuals in multi-year camera trap data can also occur due to the loss of identifying collars or ear tags, introducing potential bias into results. However, we emphasize here that camera traps were effective at detecting a decreased proportion of fawns in the immediate years following IC treatment, which likely reflects true population trends of reduced reproductive output<sup>169</sup>.

# Conclusion

IC treatment influences reproduction output of the population, but camera trap data does not allow for tracking the reproductive output of specific individuals. This likely stems from the sociality of deer, whereby treated, non-reproductive individuals, still associate with untreated deer and are therefore still associated with fawns in camera trap data. IC is effective at reducing

fawn production, but we highlight analyses of population level trends as a more informative measure for evaluating IC treatment.


# CHAPTER FIVE: AN ESTIMATE OF BLACK-TAILED DEER POPULATION DENSITY IN NATURAL VANCOUVER ISLAND LANDSCAPES TO COMPARE TO OAK BAY, BC.

Emerald Arthurs and Jason T Fisher

#### Introduction

Deer populations are generally regulated by top-down processes (predation) and bottomup processes (forage availability)<sup>170</sup>. In many urban areas in North America, predators are exlcuded <sup>52 171</sup> and forage quantity and quality is increased, creating landscapes that may allow for unchecked population growth in deer. High deer populations in urban areas can have negative social and ecological consequences, such as increased human-deer conflicts <sup>172</sup> and overbrowsing of sensitive ecosystems <sup>119,173</sup>. The "urban deer problem" may arise when populations exceed the ecological carrying capacity (ability of the resources in the ecosystem to support the population) and/or the social carrying capacity (level of tolerance of humans to the impacts)<sup>174</sup>.

Based on increased public reports of human-wildlife conflicts in urban areas on southern Vancouver Island, deer are assumed to be overabundant but there are no estimates of rural areas to confirm this claim. Comparing urban and non-urban deer densities may be useful to determine if a population should be considered as 'overabundant' ecologically or socially <sup>175</sup> as a non-urban density can provide a population baseline under 'natural' conditions. We estimated the population density in the Sooke Hills to provide a 'reference' population that can be compared to the urban estimates from Oak Bay and Esquimalt.

#### Methods

An unmarked spatial capture-recapture (uSCR) model <sup>176</sup> was used to estimate the population density in the Sooke hills (Figure 1). uSCR models are an extension of traditional

capture-recapture models with extensions to account for the missing information (no individual identities). This modelling approach requires detections at each camera site, the locations of cameras, and an estimate of home range size for the species. The population of deer is treated as a collection of 'activity centers' which are estimated based on the spatial correlation between the detections at nearby camera locations<sup>176</sup>. Using the 'activity centers' and information about the home range size of the species, the model attempts to construct individual detection histories for each camera trap site.

Data are provided by the <u>Sooke Hills Wildlife Monitoring project</u>, led by Dr. Chris Bone from the SURREAL Lab at UVic. Data from October of 2022 was used to adult estimate deer density. This time frame was chosen as there will be increased movement due to the rut and no population growth. Cameras in the Sooke hills are located based on 0.75 km<sup>2</sup> grid cells, within which one camera was placed along a trail or road as close to the center of the cell as possible. Mean spacing between cameras is approximately 0.7 kilometers, which was considered acceptable for this density estimation method – as multiple cameras must be within one home range<sup>177</sup>. Each deer detection event was considered 'independent' if it was separated by at least 30 minutes from the last detection, and each individual present in the event was counted (except for fawns). The study area, or "state space" included 44 cameras and a 1 km buffer within which density was estimated, for a total of 62 km<sup>2</sup>.

#### Results

Consistent with our predictions, *preliminary results* demonstrate that population density for the rural area is lower than urban areas. We estimated 3.1 [BCI: 1.3 - 5.6] deer/km<sup>2</sup> in the Sooke hills, compared to the 2018 estimate from Oak Bay of 18.5 adult deer/km<sup>2</sup> [95% CI: 15.9 – 21.6].

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Figure 1. Map of Sooke Hills (rural), Oak Bay (urban), and Esquimalt (urban) camera trap arrays.

## Discussion

While these results are preliminary, the estimates indicate that urban deer populations are about 6 times the density of rural populations. This indicates that increases in forage quality or quantity, and release from predation pressure are allowing urban deer populations to exist at much higher densities than under 'natural' conditions, such as in the Sooke Hills. Urban deer on southern Vancouver Island select for high-quality vegetation <sup>178</sup> and large predators that decrease adult recruitment (fawns surviving into maturity) <sup>179</sup>are absent from urbanized landscapes. Increased forage is likely augmenting the 'ecological carrying capacity' or urban areas, wherein there are sufficient resources to support higher density populations. Therefore, while deer are existing within an amplified ecological carrying capacity due to human presence, they are 'overabundant' ecologically and socially.

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