USING BEHAVIORAL ECOLOGY TO MINIMIZE HUMAN-WILDLIFE CONFLICT IN A MAJOR METROPOLITAN AREA

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DEDICATION

This dissertation is dedicated to the flora and fauna of the Sonoran Desert,
who have served as a source of motivation and strength for me.
I am hopeful that this research will be used to help alleviate some of the challenges they face.

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ABSTRACT

Urbanization has transformed landscapes globally, fragmenting habitat and reducing natural resource availability for native wildlife populations. At the same time, the percentage of humans living in an urban environment continues to increase, bringing more humans in close contact with species facing limited resources. As a result, human-wildlife conflict is a growing concern worldwide. For many wildlife species, urbanization can result in habitat loss and population decline, as well as sickness, injury, or mortality resulting from conflict. I studied human-wildlife relationships in Tucson, AZ to better understand how wildlife in Tucson adapt to urbanization and gauge resident responses to wildlife in Tucson. My studies are focused on bobcat (*Lynx rufus*), coyote (*Canis latrans*), and collared peccary, or javelina (*Pecari tajacu*), due to government concern regarding human-wildlife conflict involving these three species. To identify root causes of human-wildlife conflict in urban environments and evaluate which strategies are most effective for reducing conflict, I conducted a literature review of non-lethal deterrent techniques used by urban residents and managers to mitigate human-carnivore conflict. I additionally deployed wildlife cameras in ephemeral stream corridors in and around Tucson to better understand how urbanization affects mammalian use of riparian corridors. Finally, I distributed a public survey assessing the efficacy of deterrent techniques promoted by state wildlife agencies. Human-wildlife conflict in Tucson overall appears to be low at present based on our study results; riparian corridors in and around the city may help alleviate habitat loss and fragmentation resulting from urbanization by providing natural resources and connectivity between natural areas. Restoration and preservation of riparian corridors may therefore offer a promising strategy for reducing human-wildlife conflict in urban centers.

CHAPTER 1

A review of non-lethal strategies to mitigate human-wildlife conflict on local and regional scales

ABSTRACT

How animals respond and adjust to human disturbance is determined by their biology; therefore, principles of behavioral ecology can be used to mediate human-carnivore relationships in urban environments. A variety of strategies exist that individuals or management teams can implement to resolve conflict between humans and wildlife at local and regional scales. We review the primary non-lethal techniques used to mitigate human-wildlife conflict at local scales, including property modifications, aversive conditioning techniques, and translocation, as well as at regional scales, discussing educational outreach, land management, and disease management strategies. Reducing attractants in areas of high conflict, including removing potential food sources and altering landscaping to be less attractive, may be especially effective at reducing conflict. Aversive conditioning and deterrent devices can also be effective at managing urban wildlife; however, deterrent strategies should be combined, alternated, or rotated to remain dynamic and prevent habituation. As a last resort, removal of animals that are repeatedly involved in conflict may be necessary. Outreach campaigns that educate citizens on how to effectively prevent and resolve conflict with wildlife will strengthen management efforts to address human-wildlife conflict at a broader scale and help maintain low levels of habituation in urban wildlife. Habitat and corridor restoration in designated urban areas where wildlife is desired may also be effective for reducing pressure on anthropogenic resources and facilitating wildlife movement while minimizing spatial overlap with humans. Other proactive management

strategies, such as disease management and rodenticide bans, can disrupt positive feedback loops that increase human-wildlife conflict. While adopting any one of these approaches may be effective at reducing human-wildlife conflict temporarily, a holistic approach that addresses human behavior, wildlife ecology, and land management is needed for long-term success.

INTRODUCTION

Although many cities were not designed with wildlife in mind, wildlife is commonly found in cities, and conflict between humans and wildlife is a growing problem worldwide. Urbanization degrades and fragments wildlife habitat, reducing availability of natural resources and landscape permeability (Grimm 2008, Hess et al. 2014), which in turn increases pressure on anthropogenic resources (Blackwell et al. 2016). Subsequent reliance on anthropogenic resources by wildlife includes not only the exploitation of food resources derived from humans, such as garbage, unsecured pet food, and landscaping shrubs (Soulsbury and White 2015), but also the use of infrastructure or areas designed for humans with high human traffic, such as residential properties or roads (Grubbs and Krausman 2009*a*, Hinton et al. 2010). The disruption of natural corridors needed for dispersal can also force animals to navigate through private property and/or use roads for movement (Tigas et al. 2002, Grubbs and Krausman 2009*a*), increasing the likelihood of human-wildlife encounters and vehicle collisions. Urbanization can also create habitat for species that are well-adapted to exploit anthropogenic resources (McKinney 2006) or that favor edge habitat (Crooks 2002, Dunagan et al. 2019). Shared use of space and resources leads to human-wildlife conflict (HWC), defined as a negative interaction between humans and wildlife (Madden 2004, Soulsbury and White 2015). HWC has been a concern for urban residents throughout human history (Nyhus 2016). As urban centers grow in population density and deplete available natural resources for native species, it is reasonable to expect that human-wildlife conflict will intensify. Indeed, many regions have reported an increase in human-wildlife conflict over recent decades (Baruch-Mordo et al. 2008, Soulsbury and White 2015, Anand and Radhakrishna 2017, Bombieri et al. 2018, Farr et al. 2023, Lajeunesse et al. 2023). Some wildlife species are adapting to survive and even thrive in urban environments (McKinney 2006, Lowry et al. 2013, Sol et al. 2013); in many regions, humans exert strong selection (e.g., poison, lethal removal) in response to conflict, hastening natural selection (Schell et al. 2021).

The relationship between humans and carnivores (Order Carnivora) is particularly contentious, and lethal removal of individuals as well as local extirpation of certain carnivore species has been prevalent in many cultures throughout their history (Lennox et al. 2018). However, attitudes toward lethal control have changed in recent decades and non-lethal strategies are increasingly favored by the general public and wildlife management agencies (Jackman and Rutberg 2015, Bergstrom 2017, van Eeden et al. 2017). Here, we review the efficacy of non-lethal techniques for mitigating HWC with carnivores, the challenges of implementing such techniques, and how the use of behavioral and ecological knowledge can be more broadly extended to wildlife management and city planning to address the root causes of conflict and reduce further HWC moving forward. We focus on urban areas because residents living in urban and rural areas each face a unique suite of challenges. Many techniques that are practical for urban residents are not necessarily feasible for farmers or ranchers due to larger tracts of land, livestock size, and/or livestock numbers. We refer readers to Smith et al. 2000*b* and Shivik 2006 for a review of deterrent techniques employed by ranchers.

WHY USE NON-LETHAL METHODS?

In some cases, lethal removal may be an appropriate tool to manage human-wildlife conflict if a problem behavior is shown to be highly heritable and managers wish to select a problem behavior out of the population (Schell et al. 2021). Lethal removal may also be an effective technique for "problem individuals" exhibiting a relatively rare, learned behavior (Baker 2007, Breck et al. 2017, Swan et al. 2017). Attempts to lethally control animal populations without adequate behavioral knowledge, however, are often inefficient or ineffective and may lead to greater human-wildlife conflict (Nowell and Jackson 1996, Nogueira et al. 2007, Blackwell et al. 2016, Much et al. 2018, Lennox et al. 2018). In some cases, lethal removal may be counterproductive and result in an increase in livestock depredation (Conradie and Piesse 2013, Lennox et al. 2018), driven by increases in litter sizes (Knowlton 1972, Conradie and Piesse 2013), lower juvenile mortality (Peebles et al. 2013), or younger population with limited experience (Lambert et al. 2006, Kerston et al. 2013)). Additionally, if the attractant is not removed, lethal removal may simply open up a territory for a new animal to recolonize the area (Holzman et al. 1992, Sagør et al. 1997, Marker and Dickman 2005, Lieury et al. 2015). Lethal removal is also time intensive, costly, and may have little public support in many cases (McManus et al. 2015, Jackman and Rutberg 2015).

Carnivores may also provide population control of herbivores (Beschta and Ripple 2009, Kilgo et al. 2010), which can reduce road collisions with wildlife (Gilbert et al. 2017) as well as prevent over browsing of vegetation and its associated consequences (Beschta and Ripple 2009, Warren 2011, Goetsch et al. 2011, Sakata and Yamasaki 2015). Larger carnivores such as coyotes may help suppress populations of invasive mesocarnivores, such as domestic cats, which in turn can increase avian biodiversity (Crooks and Soule 1999). Because competition with other wildlife species is inevitable and lethal control of carnivore populations often results in unintended consequences, discussions of human-wildlife relationships have shifted from eliminating conflict to encouraging coexistence (Madden 2004, Bergstrom 2017). However, citizens living in urban and suburban areas may have little experience living with wild animals and may lack the ecological knowledge to prevent or respond to conflict effectively (Elliot et al. 2016). Employing management strategies that are informed by behavioral ecology, including long-term public awareness campaigns, allows urban residents, managers, and planners to mitigate negative consequences of human-wildlife competition.

In many cases, adjusting human behavior or using non-lethal tools is easier, safer, and more effective at resolving HWC than attempting to control animal populations (Blackwell et al. 2016, Bergstrom 2017, Much et al. 2018, van Eeden et al. 2018). Because animal behavior has been shaped by ecology and evolution, humans can use ecological and evolutionary principles to encourage or deter wildlife behavior and influence movement. Many non-lethal management strategies take advantage of these principles to reduce human-wildlife conflict. Non-lethal techniques can be more effective and less costly than lethal control (McManus et al. 2015). Use of behavioral knowledge to effectively employ non-lethal techniques further allows humans to benefit from higher biodiversity in urban areas while minimizing human-wildlife conflict.

LOCAL MITIGATION OF HUMAN-WILDLIFE CONFLICT

In the following section, we review small-scale, localized strategies that an individual or management team can use to prevent conflict with wildlife, deter habituated animals that are already present, and remove problem animals (Figure 1). Many of these strategies take advantage of species ecology by employing aversive conditioning, in which humans illicit a natural fear response to condition the animal to expect a negative consequence in association with a certain behavior. These strategies may also increase the risk of foraging and/or the effort required to forage so that the use of anthropogenic resources is no longer energetically favorable (Haswell et al. 2019, Mazué et al. 2023). If conflict persists despite the employment of deterrents, translocation or lethal removal may be warranted, particularly if an individual becomes aggressive and attacks pets or people (Lajeunesse et al. 2023).

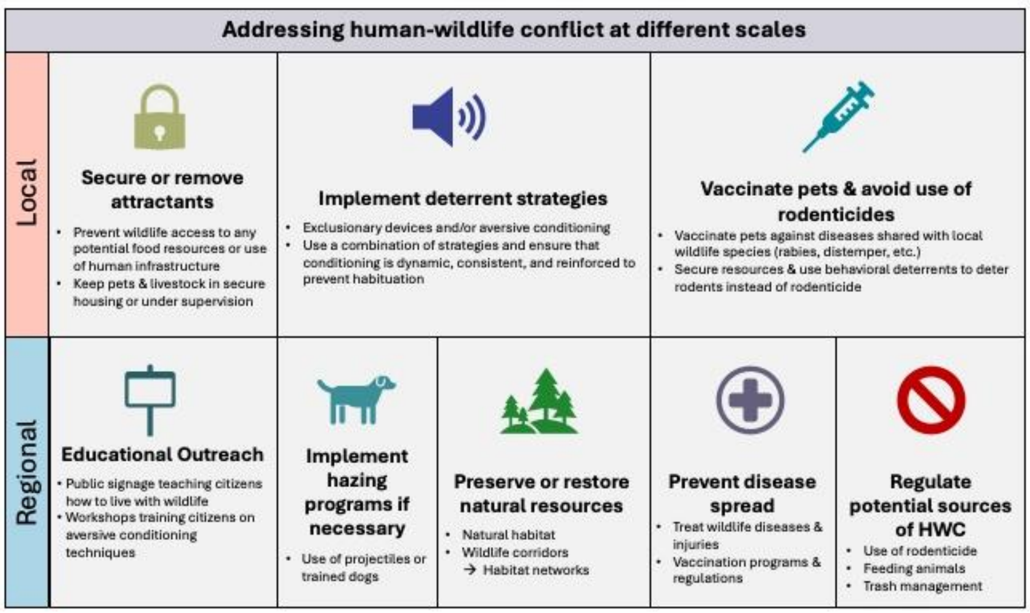

Figure 1: Recommendations for coexisting safely with wildlife at local and regional scales. Both individuals and government agencies can address human-wildlife conflict.

Securing or removing attractants

Urban residents can reduce the likelihood of wildlife visitation by making changes on their property. Removing or securing items that attract wildlife, such as trash or compost bins, is the most straightforward solution when feasible (Elliot et al. 2016, Murray and St. Claire 2017). When unsecured anthropogenic resources were reduced, chacma baboons (*Papio ursinus*) in South Africa decreased foraging in urban areas and increased foraging in natural areas (Mazué et al. 2023). Fallen fruit from trees can also represent a significant food source for omnivorous carnivores like coyotes (Grubbs and Krausman 2009*b*) and removing fallen fruit from the ground can be an effective solution for reducing property attractiveness (Baker 2007, Elliot et al. 2016). High shrub cover is selected by a number of wildlife species, including coyotes for resting and denning purposes (Grubbs and Krausman 2009*a*, Murray and St. Claire 2017). Clearing low-hanging canopy branches and larger understory shrubs in areas where carnivores are not desired can reduce the area's attractiveness as good habitat, influencing whether certain species use the area and how quickly they move through it (Hinton et al. 2010, Abouelezz et al. 2018).

Artificial barriers (e.g., fences) and natural barriers (e.g., vegetation) can reduce human-wildlife conflict in urban areas, including vehicular collisions (Sawyer et al. 2012 , van der Ree et al. 2015, Bombieri et al. 2018, Smith et al. 2020). Fencing is commonly used to prevent wildlife from accessing roadways (van der Ree et al. 2015) or potential food sources (human food, trash, pets, children; Bombieri et al. 2018). Barriers can be effective for some carnivore species (Murray and St. Clair 2017, Bombieri et al. 2018, Smith et al. 2020). Coyotes in Edmonton, Alberta avoided properties with barriers; backyards used by coyotes were 66.7 times less likely to be fenced (Murray and St. Clair 2017). However, barriers can have negative effects on both target and nontarget species (Smith et al. 2020), including a reduction in landscape connectivity for many species, which has cascading effects on genetic diversity and population health (Osipova et al. 2018). Temporary or semi-permeable barriers that target specific species but remain permeable to most wildlife may be a viable compromise (e.g., VerCauteren et al. 2007, Laskin et al. 2020).

Aversive conditioning and exclusionary devices

If preventative techniques such as securing resources are not feasible or fail to reduce unwanted animal visitations, animals can be conditioned to associate a human resource or area with danger or discomfort through the use of aversive conditioning techniques. Forming negative associations with problem behaviors such as raiding food bins (e.g., aversive conditioning, or “hazing”) can reduce an animal’s likelihood of repeating problem behaviors through learning, although effects may be temporary if habituation occurs (Leigh and Chamberlain 2008, Ausband et al. 2013). Aversive conditioning may also be used to train animals released from captivity into the wild to avoid humans (Zier-Vogel and Heurer 2022). Studies of aversive conditioning primarily focus on large mammals; more research is needed to determine whether aversive conditioning is appropriate and effective for smaller species as well.

Conditioning may employ visual, auditory, or olfactory stimuli to provoke fear and repel the animal. Shouting, waving arms, making loud noises, throwing small objects, or spraying water are all common hazing techniques that management agencies encourage residents to use to deter wildlife (Lajeunesse et al. 2023). Aversive conditioning tactics may be grouped by intensity, in which low intensity conditioning consists of yelling, making noise, or waving arms, moderate intensity conditioning consists of spraying water, use of chemical deterrent, or throwing projectiles, and high-intensity conditioning consists of device-deployed projectiles (e.g., rubber bullets, slingshot) or the use of dogs (Lajeunesse et al. 2023). Higher intensity conditioning may be more effective at increasing sensitization and preventing habituation than low intensity conditioning (Blumstein 2016, Honda et al. 2019, Lajeunesse et al. 2023). However, an upper threshold may exist in which high intensity conditioning is no longer effective. Moderate-intensity conditioning may therefore offer the most success (Found et al.

2018). Habituated individuals may be responsible for most conflicts in some populations (Baker 2007, Honda et al. 2019). Individuals that are less habituated may be more sensitive to aversive conditioning and require less resources to manage compared to those that are more habituated (Baker 2007). Habituated animals may react more boldly to aversive conditioning attempts, in some cases approaching or challenging wildlife professionals or their dogs (Lajeunesse et al. 2023). Ideally, aversive conditioning should be employed before populations become habituated (Baker 2007). Many populations used to test the effects of aversive conditioning already show moderate levels of habituation (e.g., short flight distance, approaching humans or attacking pets and livestock), which may help explain why aversive conditioning strategies often show only temporary success (Baker 2007). Aversive conditioning should also be employed immediately and consistently when an individual animal displays an undesired behavior. Latency in employing conditioning and/or inconsistent conditioning may prevent negative associations from forming. Similarly, if enough time has passed between conditioning events, learned responses may also become weaker (Lajeunesse et al. 2023). Timing, frequency, consistency, and intensity of aversive conditioning are thus all important components that affect success (Lajeunesse et al. 2023). The following sections discuss seven different methods used to deter or condition wildlife.

Projectiles

Trained professionals have used non-lethal projectiles, such as rubber bullets or buckshot, Airsoft guns, or bean-filled bags to induce pain and provoke fear, with varied success (Beckmann et al. 2004, Leigh and Chamberlain 2008, Mazur 2010, Honda et al. 2019, Alldredge et al. 2019). Projectiles may be very effective for smaller species; for example, Japanese macaques (*Macaca fuscata*) hazed with Airsoft guns engaged in less conflict with humans (Honda et al. 2019). Projectiles have had more limited success with large carnivores like bears and cougars, however (Rauer, Kaczensky, and Knauer 2003, Alldredge et al. 2019). Mazur (2010) found that rubber bullets were most effective at repelling black bears (*Ursus americanus)* in a study comparing different aversive conditioning techniques. However, some food-conditioned bears immediately returned to forage on anthropogenic resources. Rubber bullets were also temporarily effective against Louisiana black bears (*Ursus americanus luteolus)* on residential and industrial properties. However, the vast majority of bears returned to nuisance behaviors within five months (Leigh and Chamberlain 2008). Cougars (*Puma concolor)* living on the urban-wildland interface in Colorado generally did not respond to aversive conditioning using bean bags (Alldredge et al. 2019). Deploying projectiles against wildlife often requires specialized authorization and training (Shivik 2006) and so may not be feasible for urban residents to implement. Use of a chemical repellent such as pepper spray may be a more suitable technique for the average urban citizen and has shown success against repelling black bears (Mazur 2010).

Motion- or radio-activated devices

Installing certain deterrent devices such as motion-sensor lights, motion-activated sprinklers, or movement- or radio-activated guard (MAG/RAG) boxes may also act as a form of aversive conditioning that introduce negative, potentially aversive stimuli to approaching wildlife (Shivik 2006, McLellan and Walker 2021). These exclusionary devices allow property owners to employ aversive conditioning without physical human presence, which reduces effort needed to deter wildlife and increases the consistency of aversive conditioning for more effective behavioral modification. In Vancouver, British Columbia, motion-activated sprinklers significantly reduced the number of coyote visits when deployed, and coyote visits were significantly shorter even after sprinkler removal (McLellan and Walker 2021). Movement- or radar-activated guard boxes, which emit lights and sounds when triggered, have also been shown to be successful at reducing depredation of calves (Breck et al. 2002) and sheep (Linhart et al. 2002); however, these devices have not been tested widely in urban settings. RAG boxes rely on radio signals to activate the device and so would only be a feasible solution against individuals that are radio collared. Additionally, the cost of each device may be prohibitive.

Guardian animals

Guardian animals, including livestock-guarding dogs (LGD) and llamas (*Lama glama*), can be effective tools against predation (see Smith et al. 2000a for a review of LGD use focusing on larger livestock). Though most commonly used in rural areas, large dogs may also be trained to be effective at deterring wildlife from residential properties, including predators preying on urban livestock (e.g., goats, chickens). The presence of dogs can result in avoidance by some wildlife species in urban green spaces (Beasley et al. 2023) and natural areas (Lenth et al. 2008). Some manyattas, or Maasai settlements, in Kenya keep dogs to warn of and protect against predators, and manyattas with dogs experienced fewer depredation events than manyattas without dogs (Kruuk 1980). This practice varied by region, however, with members of one region perceiving dogs to be ineffective against predators (Kruuk 1980). Dogs are also used in Bhaktapur, Nepal to protect residents against leopards, martins, and mongoose (Gosai et al. 2014).

Dogs have been used as a method of aversive conditioning; however, the efficacy of aversive conditioning using dogs is limited for some carnivore species. Hazing of coyotes by urban residents was less effective when dogs were present (Bonnell and Breck 2017). Similarly, dogs were not effective at deterring Louisiana black bears from residential and industrial areas (Leigh and Chamberlain 2008). Black bears in Nevada were exposed to dogs in combination with aversive condition treatments to deter them from urban areas; however, dogs did not improve the efficacy of deterring bears (Beckmann et al. 2004). Dogs were temporarily effective at deterring habituated elk (*Cervus canadensis*) from Banff, Canada (Klopper et al. 2005), however, indicating that dogs may be a useful method of aversive conditioning for other species.

Visual deterrents

Visual deterrents may be used to confuse or frighten animals or mimic natural threats. Fladry, a string of flags, can be an effective deterrent by confusing predators attempting to cross it (Musiani et al. 2003). Animals may become habituated to fladry if the flags are left for extended time periods, and so fladry offers a mainly temporary solution or a solution to a temporary problem. Electrified fladry may help prevent habituation by reinforcing the visual deterrent of fladry with an unpleasant or painful physical response if the visual deterrent is ignored, and electrified fladry was found to be 2-10 times more effective than non-electrified fladry (Lance et al. 2010).

Sonic deterrents

Sonic deterrents include noise-making sensors that deter animals via pain, fear, communication ‘jamming’, disorientation, or the use of acoustic alarm or territorial signals (see Bomford and

O'Brien 1990 for review). Sonic deterrents may be biosonic, relying on biological signals with ecological relevance, or nonbiosonic, using perceived unpleasant sonic properties to deter animals. Nonbiosonic deterrents that inflict pain via high intensity sounds have been used in the past but are now rarely used. These deterrents are expensive to produce, likely ineffective in open environments, and objected to due to animal welfare issues (Bomford and O'Brien 1990). Nonbiosonic deterrents may also inflict fear at lesser intensities through the use of loud, acute, and unfamiliar noises. Animals typically habituate to nonbiosonic noises unless maintained in a random fashion (altering amplitude, frequency range, and position) or paired with an ecologically relevant stimulus (e.g., distress call or real threat; Bombford and O'Brien 1990). In studies that compared biosonic and nonbiosonic deterrents, biosonic alarm signals were more effective and resulted in less habituation compared to nonbiosonic deterrents (Langowski et al. 1969, Spanier 1980). Although studies of sonic deterrents have focused primarily on herbivores and birds (Gilsdorf et al. 2004, Klopper et al. 2005, Walter et al. 2010, Biedenweg et al. 2011), Nelson et al. (2006) showed that ultrasound was moderately effective at deterring cats.

Olfactory deterrents

Human-wildlife conflict may also be reduced by manipulating how anthropogenic resources smell (Price et al. 2022) and taste (Snijders et al. 2021). Olfactory cues can be used as a learning tool by pairing smells with negative or neutral experiences or preventing a positive association from forming (e.g., "conditioned taste aversion;" Massei et al. 2002). When an individual repeatedly has an unrewarding experience in association with the resource's olfactory cues, it may become habituated to the smell and/or no longer seek out the resource. "Olfactory misinformation" can be especially effective at deterring mammals, which rely primarily on their sense of smell when foraging (Price et al. 2022). Red foxes reduced consumption of bait after the addition of a nausea-inducing agent (Andrewartha et al. 2023). Olfactory cues can further be used to mask or camouflage the smell of attractants or can be used as a tool to habituate animals to the smell of target food so that it is no longer perceived as a resource (Price et al. 2022). Whereas other deterrent techniques deter animals after they have discovered the resource, olfactory cues can be used to prevent wildlife from finding the resource in the first place.

Biofencing, or the use of an ecologically relevant chemical substance to deter other animals around a perimeter, can be an effective deterrent for some carnivores (Ausband et al. 2013, Hansen et al. 2024). Because large carnivores maintain territory boundaries via scent marking, the use of chemicals mimicking a potential competitor can deter predators as well (Ausband et al. 2013). Biofencing can act as a temporary (on a monthly-yearly time interval) barrier for large carnivores and can be used effectively to limit depredation on sheep (Ausband et al. 2013). Carnivores may alter livestock predation rate based on seasonal resource availability (Melville et al. 2004). Thus, a temporary application only during seasons with high conflict could reduce habituation effects. Compounds that are identical to or that closely match the full chemical profile of a predator or competitor are more effective (Maestas-Olguin et al. 2021). The use of predator urine may be ineffective against mesopredator species that overlap spatially with the predator and are not a main prey source for the predator (Severud et al. 2011, Yocom-Russell and Verble 2020). Though wolf urine was an effective deterrent of beavers, a primary prey source of wolves (*Canis lupus)*, it was ineffective at reducing the activity of raccoons (*Procyon lotor)* and muskrats (*Ondatra zibethicus)*, which are less commonly preyed on by wolves (Severud et al. 2011). Coyote urine also did not deter raccoons from feeding at bait stations (Yocom-Russell and Verble 2020).

Removal

Removal of animals that repeatedly engage in conflict may be necessary if removing attractants and aversive conditioning are ineffective at reducing conflict (Baker 2007, Swan et al. 2017). Removal may be lethal or non-lethal, and each may result in unintended consequences (Fontúrbel and Simonetti 2011, Blackwell et al. 2016, Nyhus 2016, Lennox et al. 2018). Thus, removal should be carefully considered and ideally used only as a last resort after other strategies have been exhausted. Translocation, or "the human-mediated movement of living organisms from one area, with release in another (IUCN 2013)," is a popular strategy used to solve problems with wildlife; however, translocations are often not a sustainable solution for wildlife conflict (Spencer et al. 2007, Walter et al. 2010, Fontúrbel and Simonetti 2011). Trapping and releasing animals can be expensive and moving them does not necessarily resolve conflict. A survey of 48 management agencies in North America found that a majority of managers view translocation of black bears as ineffective (Spencer et al. 2007). Translocations may lead to increases in vehicular collisions or simply transfer conflict to another property or residence (Fontúrbel and Simonetti 2011, Nyhus 2016). In a review of nonlethal removal across North America, 80% of carnivore translocations were found to result in vehicular mortality (Fontúrbel and Simonetti 2011). Additionally, if the attractant is not removed or secured, another animal will likely fill the vacancy left by translocations, perpetuating the problem (e.g., Marker and Dickman 2005). If translocations are identified as the best possible solution, adhering to species ecology (e.g., ensuring that animal is not released in conspecific territory, translocating groups if social) may increase translocation success.

Challenges to implementing non-lethal techniques at a local scale

In the following section, we provide a brief overview of the challenges of implementing non-lethal techniques described above. Many benefits exist to employing techniques that take advantage of behavioral knowledge; however, reliance on a single deterrent technique will decrease in effectiveness over time (Blackwell et al. 2016). For example, animals can habituate to unpleasant stimuli if they are not paired with a real threat, or animals may adapt behaviorally to deterrents, as they do for lethal and nonlethal removal. Employing a suite of the aversive conditioning techniques described above by alternating and/or combining techniques, rather than relying on a single technique or solution, is most effective to prevent habituation and reduce HWC long-term (Madden 2004, Kloppers et al. 2005, Blackwell et al. 2016, Heim et al. 2022). Other deterrent strategies, such as the use of guardian animals, discourage habituation by employing a variety of threatening visual, olfactory, and auditory stimuli in a consistent manner, and reinforcing threat displays with chases, bites, or attacks when escalated (Brown 2011). Likewise, deterrent techniques that resemble predators and/or predator hunting strategies are effective at reducing habituation (Kloppers et al. 2005, Storms et al. 2022) There will likely never be a single permanent solution, and managers will always need to take learned and evolved behavioral changes into account when planning and implementing strategies to minimize human-wildlife conflict.

In addition to preventing habituation, the monetary cost of deterrent techniques is an additional challenge that property owners and management teams must face. Guardian animals are costly to train and care for, and certain individuals may occasionally kill livestock (Smith et al. 2000*a,* Brown 2011). Other behavioral techniques, such as biofencing, may additionally be costly to maintain. Cost can be prohibitive for managing large areas of land, and so these deterrent strategies may be better suited for more urban environments with smaller properties.

Finally, the values, attitudes, and behavior of the local community need to be taken into consideration, and management objectives should always seek to align closely with community values (Madden 2004, McCleery et al. 2014). Some non-lethal techniques may still risk the welfare of wildlife and be viewed as unethical by the general public. Projectiles such as rubber bullets, the use of dogs for aversive conditioning, and certain sonic deterrents may be viewed as unethical (Bomford and O'Brien 1990, Sampson and van Patter 2020). Wildlife professionals may therefore wish to avoid these techniques or carefully train employees to prioritize animal well-being if these techniques are used. Ultimately, adaptive management strategies that continuously monitor human-wildlife relationships and adjust course when necessary are needed to respond to behavioral changes and adaptations effectively.

REGIONAL MITIGATION OF HUMAN-WILDLIFE CONFLICT

The strategies outlined above can be used by urban residents to resolve conflict with wildlife on their private properties or by wildlife managers working with problem individuals. However, large-scale management actions may be needed to address the root causes of conflict where HWC is widespread or has reached problematic levels. In the following section, we review how wildlife and land managers can address human-wildlife conflict. Long-term solutions may potentially be met by reducing wildlife dependence on anthropogenic resources. Broad-scale management strategies that prevent HWC include educational outreach programs, habitat restoration and the creation of habitat networks, and disease prevention efforts.

Educational outreach

Educational outreach programs that inform urban residents of how to prevent and resolve conflict with wildlife is imperative for reducing urban conflict and facilitating healthy coexistence. Educational programs should emphasize the efficacy of proactive behaviors (e.g., securing attractants) in reducing conflict, reducing disease transmission, and conserving wildlife, as well as how and when to appropriately haze wildlife (Puri et al. 2024). Human-wildlife conflict mainly occurs in the absence of wildlife professionals, and thus much of the opportunity for habituation or sensitization lies with the general community. While many conflict-prone animals are sick or injured, in some circumstances, animals have developed nuisance behaviors in response to feeding by urban residents (Baker 2007, Gehrt et al. 2009). Garbage and food left unsecured is also commonly identified as a primary source of conflict (Baker 2007, Spencer et al. 2007, Morehouse and Boyce 2017, Puri et al. 2024). Educational campaigns encouraging citizens to secure food and trash and adopt predator-savvy behaviors for pets (e.g., keeping pets under close supervision, bringing pets inside between dusk and dawn) may be needed to reduce habituation or reliance on anthropogenic resources (Puri et al. 2024). Encouraging certain behaviors, such as taking out garbage cans the morning of trash pickup instead of the night before or purchasing wildlife-resistant receptacles, can additionally reduce anthropogenic resource availability fueling problem behaviors (Klees van Bommel et al. 2020, Puri et al. 2024). Educational campaigns that inform citizens on how to appropriately condition animals using aversive techniques (e.g., BearSmart) have shown success at reducing problem behaviors (Beckmann et al. 2004, Baker 2007, Bonnell and Breck 2017, Lajeunesse et al. 2023). While management agencies have employed aversive conditioning tactics temporarily, the general public is in a unique position to enforce conditioning long-term and across contexts. Lapses in educational outreach programs may allow wildlife to revert to problematic levels of habituation (Baker 2007), and so continuous outreach efforts may be necessary to prevent habituation. Where educational outreach is ineffective, local regulations may be necessary to curb human behaviors leading to conflict (e.g., New Hampshire Code of Administrative Rule Fis 310.01 *Control of Nuisance Black Bears*).

Habitat networks - wildlife corridors and crossings

Wildlife may increase consumption of anthropogenic food sources when natural food items are limited (Kolowski and Holekamp 2006, Newsome et al. 2015, Mukeka et al. 2019). Ensuring that natural habitat is available for movement, shelter, and foraging or hunting can therefore reduce pressure on anthropogenic resources (Blackwell et al. 2016, Cassaigne et al. 2021). For example, coyotes in Chicago, particularly those living in natural preserves, consumed mostly natural prey, and those that consumed more anthropogenic food inhabited areas of lower natural prey availability (Newsome et al. 2015). Similarly, black bears are less likely to seek out anthropogenic food in years of plentiful natural food (Beckmann and Berger 2003). Motivation influences the efficacy of deterrents in that aversive conditioning and barriers may be less effective when animals are highly motivated by malnourishment or nutrient deficiency (Walter et al. 2010). Preserving or restoring wildlife habitat within urban areas may therefore increase the efficacy of aversive conditioning and other deterrents by lowering motivation to tolerate unpleasant experiences or breach fences. Increasing green space and native species coverage in cities also increases the rate of positive interactions with wildlife, reducing negative attitudes toward wildlife (Schell et al. 2021).

In addition to providing habitat for natural food and shelter resources, a network of natural habitat is needed within cities to allow wildlife to navigate through cities without requiring animals to cross or use roads and other anthropogenic infrastructure. While many studies have identified the importance of natural parks and other green spaces for wildlife (e.g., Grinder and Krausman 2001, Dunagan et al. 2019), and the restoration of "pockets" of urban wildlife habitat have been encouraged (Miller 2005, McCleery 2014), a network of restored native vegetation may be necessary to provide a safe corridor for wildlife movement between green patches, rather than leaving wildlife to disperse between natural patches using roadways, ditches, and residential areas (e.g., Grubbs and Krausman 2009*a*, Hinton et al. 2010). At the same time, roads represent barriers that restrict wildlife movement and increase risk of injury or death during dispersal (Tigas et al. 2002, Riley et al. 2014, Serieys et al. 2021). Wildlife populations that are subject to urban development and/or that remain within city limits will inevitably require individuals to move through developed areas during dispersal. Connectivity between habitat patches bisected by roads is often limited, leading to vehicular collisions resulting in injury, death, and property damage (Bissonette et al. 2008, Rowden et al. 2008). In addition to acting as a direct source of mortality for wildlife populations, roads also fragment and degrade habitat, reducing overall wildlife abundance via indirect effects (see Fahrig and Rytwinski 2009 for review). Implementing or maintaining wildlife corridors during city expansion and development, including wildlife crossings, could dramatically reduce mortalities, injury, and property damage resulting from vehicular collisions with wildlife (Glista et al. 2009, Schell et al. 2021). Some cities and counties have implemented this proactive approach by strategically protecting open space from development. In Southeastern Arizona, Pima County has established the Sonoran Desert Conservation Plan (SDCP), which restricts future land development to protect critical habitat and wildlife corridors. Expanding these plans to not only prioritize the conservation and connectivity of wildlife habitat but also to minimize future HWC may offer an effective approach for coexisting with wildlife.

Disease prevention

Educational outreach and increases in habitat alone may help prevent disease by reducing or eliminating the consumption of anthropogenic resources by wildlife. However, if zoonotic diseases remain a known concern prevention and treatment of contagious diseases offer another long-term solution for reducing HWC. Animals involved in human-wildlife conflict are often reported to be unhealthy or injured (Baker 2007, Gehrt et al. 2009, Murray et al. 2015*a, b*). Diseased coyotes infected with sarcoptic mange (*Sarcoptes scabei*) consumed a greater proportion of anthropogenic food (Murray et al., 2015*b*) and were twice as likely to be detected at compost piles than in natural areas (Murray et al. 2016*a*). Sick individuals were also nine times more likely to select yards with anthropogenic food (Murray and St. Clair 2017). Consumption of anthropogenic food by coyotes can create a positive feedback loop as the animal's body condition is weakened due to poor nutrition, further limiting the ability of the animal to forage naturally and increasing disease susceptibility (Sugden et al. 2020; Figure 2). Rapid treatment of sick or injured animals is therefore vital for reducing human-wildlife conflict and spread of disease.

Rodenticide contributes to positive feedback loops, exacerbating HWC when rodenticide used to eradicate rodents is transferred through the food chain and becomes concentrated in larger carnivores upon consumption of the dead rodents. At high concentrations, rodenticide reduces immune function, leading to health problems in carnivores, including sarcoptic and notoedric mange (*Notoedres cati;* Riley et al. 2007, Poessel et al. 2015, Serieys et al. 2015, Serieys et al. 2018). Reduced mobility due to illness can subsequently increase conflict and may result in death (Riley et al. 2007, Poessel et al. 2015, Siereys et al. 2015). Regulating the use of rodenticides and treating wildlife infected with mange may prevent the foraging behaviors that lead to human-wildlife conflict. At detrimental levels, mange is clearly detectable in wildlife camera photos, allowing researchers to better understand patterns underlying disease and focus management efforts on individuals living in those areas (Murray et al. 2016*a*, 2021).

Oral bait vaccines can be an effective way for vaccinating wildlife against certain diseases, including mange, plague, and rabies (Marks and Bloomfield 1999, Estrada et al. 2001, Kapil and Yeary 2011). Vaccination programs targeting companion species (e.g., dogs, cats, and ferrets) can also prevent disease transmission to and spillback from wildlife, including canine distemper virus (CDV), for which wildlife serve as reservoirs (Kapil and Yeary 2011). Finally, if supplemental feeding is required for conservation purposes, managers can reduce potential negative health effects of supplemental feeding by following recommendations outlined by Murray et al. (2016*b*).

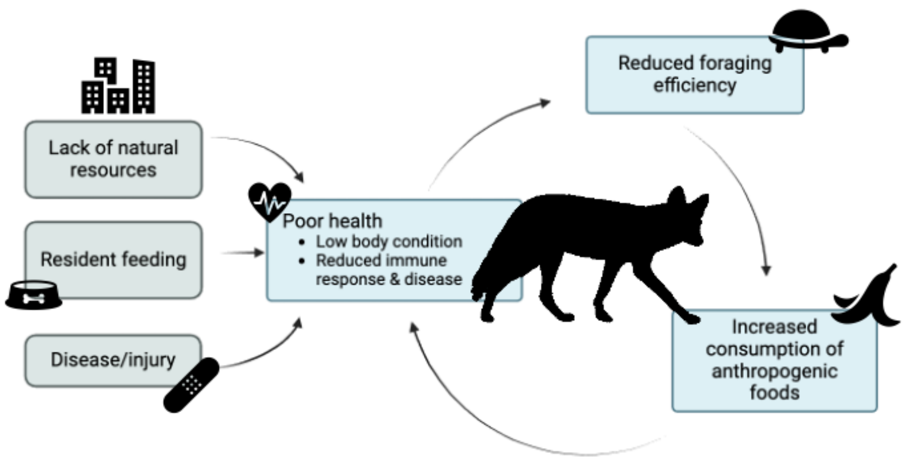

Figure 2: Wildlife may get caught in a positive feedback loop if they rely on anthropogenic resources to meet caloric or nutritional demands. Consumption of protein-poor resources can alter microbiome composition and increase exposure to contagious diseases like sarcoptic mange, leading to poor health, which reduces foraging efficiency and further increases reliance on anthropogenic food.

CONCLUSIONS

Animal populations have declined significantly as a result of habitat loss, and human-wildlife conflict further jeopardizes human and non-human populations through removal, poisoning, and in extreme cases, eradication. The potential of HWC to exacerbate local extinction highlights the importance of developing a wide range of sustainable solutions. Unless artificial selection of heritable behaviors in a certain direction is desired, solutions that take behavior into account are more likely to be effective than removal in many cases. Employing a suite of aversive conditioning tactics and exclusionary deterrents that disrupt or discourage behaviors leading to conflict is a promising solution for resolving direct conflict. Humans can also generate long term solutions by altering their own behavior on larger scales. Educational outreach that encourages citizens to adopt appropriate behaviors (e.g., securing attractants) and respond to conflict effectively (e.g., aversive conditioning) addresses the root causes of human-wildlife conflict to prevent conflict from reoccurring. Urban development projects that retain or restore a cohesive network of wildlife corridors, crossings, and green space may further reduce human-wildlife conflict by reducing wildlife reliance on anthropogenic food and infrastructure as resources. Finally, disease prevention and management are needed to disrupt positive feedback loops leading to HWC. The use of behavioral principles to reduce HWC therefore offers a number of promising solutions; however, the adoption of just one strategy is unlikely to have lasting success. A holistic, multi-faceted approach that addresses human behavior and resource availability in addition to wildlife behavior is needed to mitigate HWC long-term.

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CHAPTER 2

Suburban ephemeral streams support native mammalian communities in a semiarid, urban ecosystem

ABSTRACT

Ephemeral streams, which only flow for short periods of time after precipitation events, are important for the survival of upland and riparian species in arid ecosystems. These streams may be of particular importance in urban environments where natural resource availability is low and connectivity between natural patches is limited. We used a network of 62 wildlife cameras to study mammal species richness, community composition, and use of ephemeral streams across an urban-exurban gradient in the Sonoran Desert. We observed 26 species of mammals. Human disturbance (e.g., human presence, impervious surface), landscape connectivity, and streambed characteristics affected mammal species richness. Suburban and exurban communities in ephemeral streams exhibited high compositional overlap, but urban communities were distinct. Top models predicting bobcat (*Lynx rufus*), coyote (*Canis latrans*), and collared peccary (*Pecari tajacu*) probability of use and rate of use all included network distance and impervious surface, however, the spatial scale at which impervious surface had an influence varied by species. Human activity also reduced the rate of bobcat use and the probability and rate of coyote use. Our findings show that ephemeral streams represent a key habitat for Sonoran Desert mammals and may be particularly important for supporting biodiversity in suburban environments. Our results provide additional insight into how riparian corridors can be managed to minimize the effects of urbanization on wildlife and enhance connectivity between green spaces.

INTRODUCTION

Rivers and streams are important habitat for terrestrial species (Sabo et al. 2005). This is particularly true in arid environments where riparian environments provide a suite of resources upon which both desert upland and riparian species may rely upon for survival (Ticer et al. 1998; Levick et al. 2008, Grubbs and Krausman 2009, Acuña et al. 2014, Free et al. 2015). Ephemeral streams, which only flow for short periods of time after precipitation events, make up a large proportion of waterways in arid environments (Messager et al. 2021). Many perennial and intermittent streams will likely become ephemeral as climate change intensifies (Tonkin et al. 2018, Reid et al. 2019). Despite their ephemeral flow, these streams can still support important riparian habitat for myriad species (Hutmacher et al. 2014, Stromberg et al. 2009, 2017). Ephemeral streams can offer canopy cover, cooler microhabitats, unique food sources, and a water source during parts of the year (see Sánchez-Montoya et al. 2021 for review) in addition to serving as dispersal corridors for plants and animals (Sánchez-Montoya et al. 2016).

Riparian zones located in and around cities are particularly valuable, serving as habitat refuges and wildlife corridors that can allow wildlife to traverse or persist in urban environments (e.g., Grubbs and Krausman 2009). However, riverbanks and floodplains are often highly modified in urban settings, and high impervious surface coverage surrounding urban stream channels removes and degrades available habitat (Paul and Meyer 2001). This can cause shifts in community composition toward more tolerant or generalist species (Lussier 2006, Łopucki et al. 2013).

Because riparian zones are important for the survival of many species living in arid regions (e.g., Free et al. 2015), development of stream channels in urban environments may have substantial consequences for mammal species richness and community composition. Sensitive species, including larger carnivores that require large home ranges may be extirpated by urban development of riparian and upland environments (e.g. mountain lions: Ordeñana et al., 2010). However, for generalist species, urbanized stream channels may be important habitat that allow individuals to survive and navigate the urban matrix (e.g. coyotes: Grinder and Krausman 2001; Ordeñana et al., 2010). Examining how mammalian communities along ephemeral streams change across an urban-exurban gradient may provide insight into how urbanization affects the ability of ephemeral streams to support biodiversity. Assessing which landscape and channel characteristics are most important for explaining observed biodiversity trends would aid wildlife managers and urban planners in restoring or preserving stream channels in a way that benefits both humans and wildlife.

Some mammal species, such as coyote (*Canis latrans*, Grinder and Krausman 2001, Grubbs and Krausman 2009) and collared peccary (*Pecari tajacu*, Ticer et al. 1998), use urbanized ephemeral streams as travel corridors in the southwestern USA. Bobcats (*Lynx rufus*) are also known to use riparian zones (Serieys et al. 2021). While all three species are able to adapt to low levels of urbanization and are present in or on the outskirts of many southwestern cities, all three species avoid high density urban development or impervious surface cover (bobcats: Tigas et al. 2002, Riley et al. 2003, Dunagan et al. 2019, Schmidt et al. 2023; coyotes: Riley et al. 2003, Grubbs and Krausman 2009; peccaries: Ticer et al. 1998, Bellantoni and Krausman 1993). We additionally focused our analyses on these three species due to government concern regarding the proportion of complaints attributed to these three species (AZGFD, personal communication) and government and public concern of attacks on pets, urban livestock, and people.

Riparian corridors through the Sonoran Desert city of Tucson, Arizona, USA vary in their structure and surroundings, including channel and corridor width, vegetation structure, and percentage of impervious surface. This variability creates a natural experimental setting in which to investigate the characteristics that wildlife select when using ephemeral streams in urban habitat, as well as how urbanization affects wildlife use in general. While many small streambeds in Tucson have been channelized and narrowed into culverts, others remain partly or mostly intact, with natural substrate, xeroriparian vegetation, and wider channels to manage large seasonal floods that would otherwise threaten city infrastructure (Webb et al. 2014). Streambeds close to the city center are often channelized, paralleling roads, and in some cases, banks are cemented over to reduce erosion during storms.

In this study, we placed 62 wildlife cameras across a network of mainly ephemeral streams along an urban-exurban gradient in metropolitan Tucson to understand how urbanization affects mammal occupancy, abundance, and community composition. Occupancy and abundance analyses centered on our three focal species – bobcat (*Lynx rufus*), coyote (*C. latrans*), and peccary (*P. tajacu*), whereas richness and composition analyses included all mammals detected except for non-Sciuridae rodents. We expected that mammal richness would decrease and community composition would change as urbanization increases, particularly due to higher densities of impervious surface and lower connectivity between streambeds (i.e. network distance). Similarly, we expected our three focal species to respond negatively to impervious surface and network distance (e.g., lower probability of site use and lower rate of use).

METHODS

Study Area

Tucson, Arizona is the center of a large metropolitan area with a population of approximately 1 million people. Annual rainfall ranges from 300 to 500 mm and follows a bimodal pattern, with up to 50 percent of rain occurring as monsoons in the summer months (July to September; Sheppard et al. 2002). Summer high temperatures average 35º C, with yearly averages of 17º C (Sheppard et al. 2002). The Tucson metropolitan area is partially surrounded by public land preserved for wildlife habitat, including Saguaro National Park, Pusch Ridge Wilderness, and various state and county parks that are all engaged in a collective regional land management plan known as the Sonoran Desert Conservation Plan (Huckelberry 2002). Historically, the Tucson area featured a variety of vegetation including lower and upper Sonoran Desert types and riparian corridors, but urbanization has since increased man-made structures and exotic landscaping (Germaine and Wakeling 2001; Webb et al. 2014). Streambeds within and around Tucson exhibit great variety in structure, vegetation cover, and surrounding impervious surface (Fig 1).

Figure 1: Riparian corridors in Tucson, AZ vary in streambed width, substrate, and vegetation structure

Sampling Design

We deployed 62 motion-activated trail cameras (Bushnell Trophy Cam HD) from January 2021 through December 2022 across an urban-exurban gradient in primarily ephemeral streams; none of our sites had year-round surface water. We placed cameras along streams approximately 1.5 km apart except where private land ownership prevented regular spacing (Fig 2). The cameras were set to allow one photo per 30 sec, and sensitivity levels were set to normal. Cameras were enclosed in a protective metal casing and secured with steel locking cables to trees. Camera placement height ranged from approximately 0.5–2 m above the ground depending on the tree position and angle. All cameras were positioned to capture medium- or large-sized mammals (>1kg) with a viewshed of the main stream channel whenever possible.

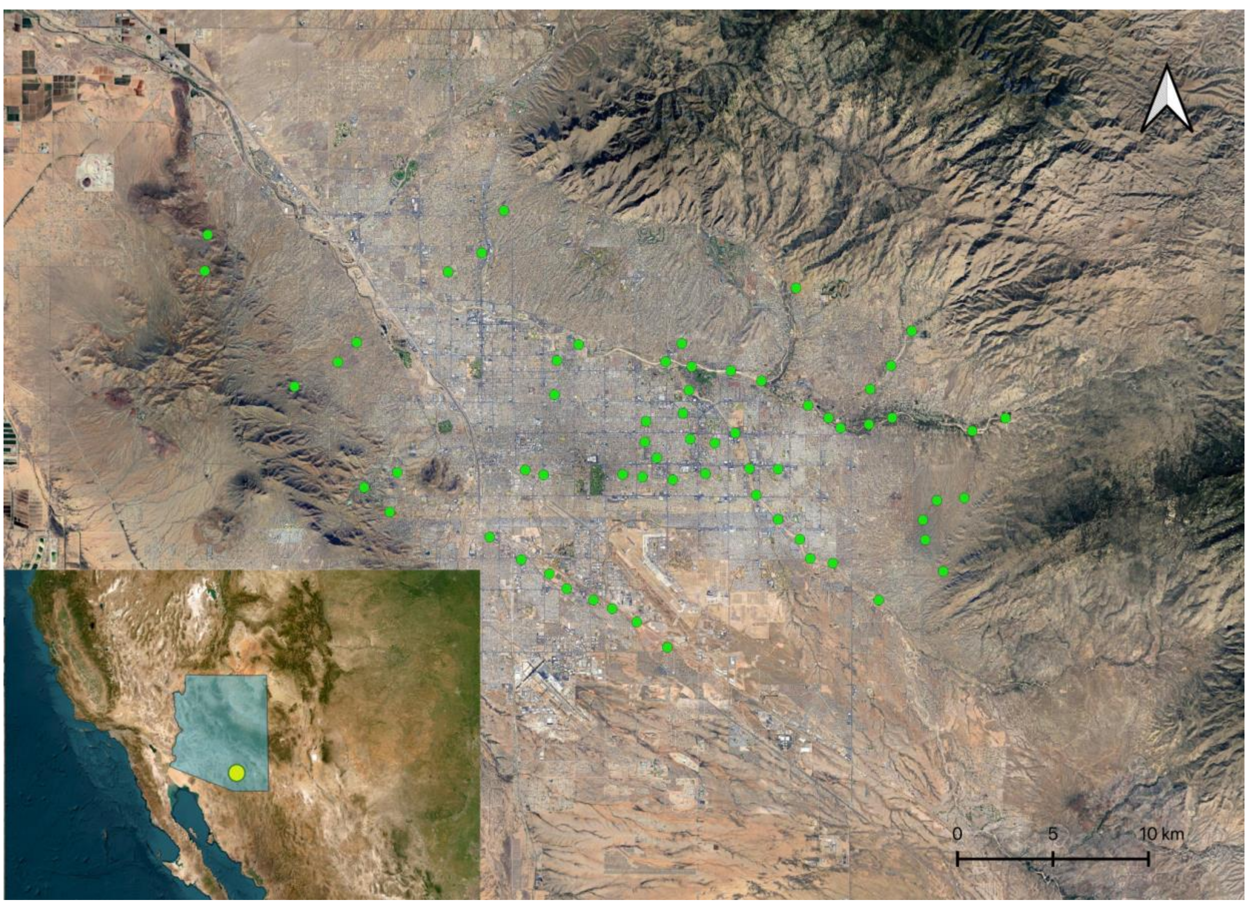

Figure 2: Map of wildlife camera locations in Tucson, Arizona, USA. Cameras were placed adjacent to ephemeral streams and spaced approximately 1-2 km apart except where private property prevented access

Data Processing and Urbanization Metrics

We used Timelapse (Greenberg and Godin 2015), a trail camera software, to manually sort and identify species and count data. Observations were omitted if the species had been documented within the previous 60 min to ensure independent observations and more accurate calculation of relative abundances. When identifying independent observations, we retained the observation with the highest number of animals captured in a single photograph within each hour. Sampling effort varied among cameras due to theft, camera damage, and battery or storage shortage. While most cameras were deployed for >1 year, we placed 7 cameras for a shorter period of time (April–June 2022) to include a better representation of highly urbanized streams while attempting to minimize risk of theft; cameras were often stolen in these locations within a few weeks. The precise placement of cameras at each site (in the main channel or on the upland bank) varied depending on site access permissions and the availability of a suitable camera placement site. Sampling effort and camera placement were therefore included in all statistical models to account for variation in effort and placement.

Rodents were largely omitted from species identification and counts to avoid misidentification. Some small mammal species require handling for positive species identification, and conventional camera traps can produce over-exposed images that make identifying small, nocturnal mammals difficult without modifications to the camera setup (De Bondi et al. 2010). Further, our cameras were not positioned to capture small mammals reliably.

Therefore, we omitted all rodents (Order Rodentia) with the exception of squirrels (Family Sciuridae) due to their larger size, diurnal habits, and ease of identification.

We used QGIS (v.3.12) to visualize and calculate percentage impervious surface and vegetation cover for each camera site. We used 2021 NLCD land cover data of impervious surface cover (Dewitz 2023) and created buffer zones of $10m^2$, $50m^2$, $100m^2$, $0.5km^2$, $1km^2$, $2km^2$, $3km^2$, and $5km^2$ around each camera site to estimate urbanization around each camera and identify which scale of impervious surface was most informative for modeling our observations of mammalian use of streambeds. To estimate shrub and tree cover within our streambed channels, we used Google Earth imagery and estimated by eye percent coverage of trees and shrubs within our $10m^2$ buffer zones around each camera site. We also measured network distance, the distance from each camera location to the nearest confluence with another stream channel of any size. Distances to confluences serve as a measurement of connectivity, where streambeds that are further away from a confluence represent more isolated streambeds, typically surrounded by a higher density of impervious surfaces. We additionally measured Euclidean distance to the nearest golf course to include in coyote occupancy and abundance models (see below), because coyotes were found to select for golf courses in Tucson in a previous study (Grubbs and Krausman 2009). We were interested in whether coyotes might be selecting for the perennial water sources often found on golf courses, and so we additionally measured the Euclidean distance to the nearest perennial drinking water source (e.g. in an urban park or golf course pond). We used Pearson’s correlation analysis to create a correlation matrix of all continuous variables. Variables that were strongly correlated with each other ($r \geq 0.6$) were not included in the same model.

Mammalian Richness and Community Composition

We ran linear regressions in R (R Core Team 2022) and Rstudio (v.2023.09.1 + 494) to model native mammalian species richness (excluding small mammals and domestic animals such as cats and dogs) with environmental characteristics, including impervious surface cover, vegetation cover, channel width, rate of human presence via relative abundance (O'Brien 2011), and proximity to an additional streambed. We quantified the rate of human presence (hereafter referred to as 'human rate of use') by calculating the relative abundance index (O'Brien 2011), dividing the number of independent observations by the total number of days the camera was deployed. We initially tested eight linear models of native species richness and impervious surface corresponding to the eight different scales of impervious surface to determine which size buffer zone to include in subsequent models. We selected the spatial scale used in the model with the lowest AIC. We then tested four linear models for native mammal richness corresponding to four hypotheses predicting mammal richness based on: 1) human disturbance (e.g., impervious surface, human rate of use), 2) landscape characteristics (i.e., percentage impervious surface, Euclidean distance to 2nd streambed), 3) local (site) characteristics (i.e., streambed width, canopy cover), and 4) a global model including all variables. All models included sampling effort (total number of nights the camera was deployed) and camera placement (i.e., main channel or upland).

We compared mammalian community composition among urban, suburban, and exurban locations using non-metric multidimensional scaling (NMDS) with the *vegan* package (Oksanen et al. 2022). We removed humans from our species composition dataset for this analysis in order to fit environmental variables, which included human rate of use, to our NMDS ordination plot. We defined urban, suburban, and exurban categories based on impervious surface levels using the same thresholds as those used by McKinney (2008). Locations with greater than 50% impervious surface within 1 km^2 buffer were defined as urban, and locations with less than 20% impervious surface within 1 km^2 were defined as exurban. We used the *vegan* package (Oksanen et al. 2022) to run a PERMANOVA test to determine whether any communities were distinct, followed with a post-hoc, pairwise test using the *pairwiseAdonis* package (Martinez Arbizu 2017) to determine which of the three communities were different. We used the *indicspecies* package (De Cáceres and Legendre 2009) to identify indicator species for urban, suburban, and exurban environments and calculate A and B components of indicator values for species with significant associations. We used the A and B components to calculate indicator values based on Dufrêne and Legendre (1997).

Occupancy and Abundance Models

To understand which characteristics may be influencing the use of washes at a finer scale, we used the "unmarked" package (Fiske and Chandler 2011) to estimate occupancy of sites for three main urban species: collared peccary (*Pecari tajacu;* hereafter referred to as "peccary"*),* bobcat (*Lynx rufus)*, and coyote (*Canis latrans).* We chose these species because they are the primary species of local concern in regard to human-wildlife conflict (Locana De Souza, Tucson Arizona Game and Fish Department, pers. comm.). Because our primary interest was in mapping habitat use rather than modeling colonization and extinction of sites, we ran single season occupancy models using stacked data for each species and interpreted occupancy probabilities as the probability of use (Mackenzie and Royle 2005, Latif et al. 2016). Peccary and coyote additionally exhibit ecologies that make delineating "open" and "closed" populations difficult, as peccary breed year-round in Tucson (Burnett, pers. observations) and transient coyotes display nomadic movements over a large area (Grinder and Krausman 2001). To relax the assumption of closure throughout the entire study period of two years, we stacked our encounter histories such that each year at each site was considered independently from the other. We included year as a random effect in all of our models to account for pseudoreplication. We delineated survey periods by month, resulting in 12 repeated surveys per site per year. We ran Moran's test for spatial autocorrelation for each species and removed three cameras from coyote occupancy models (Moran's I = 0.03: p = 0.06 after removal of 3 cameras) as well as 2 cameras from peccary occupancy models (Moran's I = -0.007; p = 0.45 after removal of 2 cameras) to prevent spatial autocorrelation. Bobcat observations showed no spatial autocorrelation when all 62 cameras were included (Moran's I = 0.03; p = 0.1).

For each species, we initially tested four occupancy models to model detection probability, keeping psi constant. We used AIC to select observer covariates (season and effort) to include in subsequent occupancy models. For all occupancy models, sampling effort was calculated as the proportion of each month that the camera was deployed (days active/total days in the month), creating an observer covariate for each survey at each site. After modeling detection probability, we tested 8 occupancy models, corresponding with the 8 scales of impervious surface measured. The scale used by the model with the lowest AIC was then used in subsequent model testing. We then tested a series of models for each species, corresponding to four hypotheses regarding how environmental and disturbance variables may affect species occupancy: 1) a model including all variables thought to affect species occupancy supported by previous literature, 2) a landscape model that included only landscape-level variables, 3) a local model that included only site-specific variables, and 4) a global model that included both local and landscape variables. Because distance to golf courses and distance to perennial water sources were correlated (*r* =

0.77), we tested two separate models each for the local and global coyote occupancy models to avoid including two or more correlated variables in one model. Overall, we tested four occupancy models for bobcats, six models for coyotes, and four models for peccaries. We additionally tested the same set of models with a relative abundance index (RAI; O'Brien 2011) for each species as a response variable to further understand how these environmental variables affect the rate of use by each species. We calculated RAI by dividing the total number of independent observations of each species over each year and dividing by the total number of nights the camera was deployed each year, resulting in a daily average number of independent observations per day for each species. All environmental variables were scaled for occupancy models. All occupancy and abundance models included camera placement, effort, and year (see Appendix S3 for list of models tested).

RESULTS

Mammal Richness and Community Composition

Sampling effort for each camera ranged from 64 nights to 729 nights, with a mean of 484.77 nights per camera. We observed 26 species of native mammals (Appendix S1). Impervious surface cover showed the greatest effects on species richness at $5km^2$. For this reason, the $5km^2$ impervious surface scale was used in subsequent models. Our model of human disturbance was selected as the top model based on ΔAIC; however, our global model (hypothesis 4) was within 2 ΔAIC (Appendix S2). We observed a decrease in native mammal richness in response to human rate of use (β = -0.39 ± 0.15) and mean percent impervious surface in a $5km^2$ buffer around camera locations (β = - 0.07 ± 0.02). In addition to these effects, the next top model showed mammalian richness increase in response to increasing network distance (β = 0.0002 ±

0.0004) and a decrease in response to streambed width (β = -0.02 ± 0.01). Cameras placed in upland areas captured fewer mammalian species than did those placed in streambed channels (β_{upland} = -0.99 ± 0.62). When data were stacked in a method similar to that of occupancy models (i.e., data were separated by year), the top model showed a decrease in species richness in the year 2022 compared to the year 2021 (β = -0.07 ± 0.35; see supplementary information for details of this analysis)

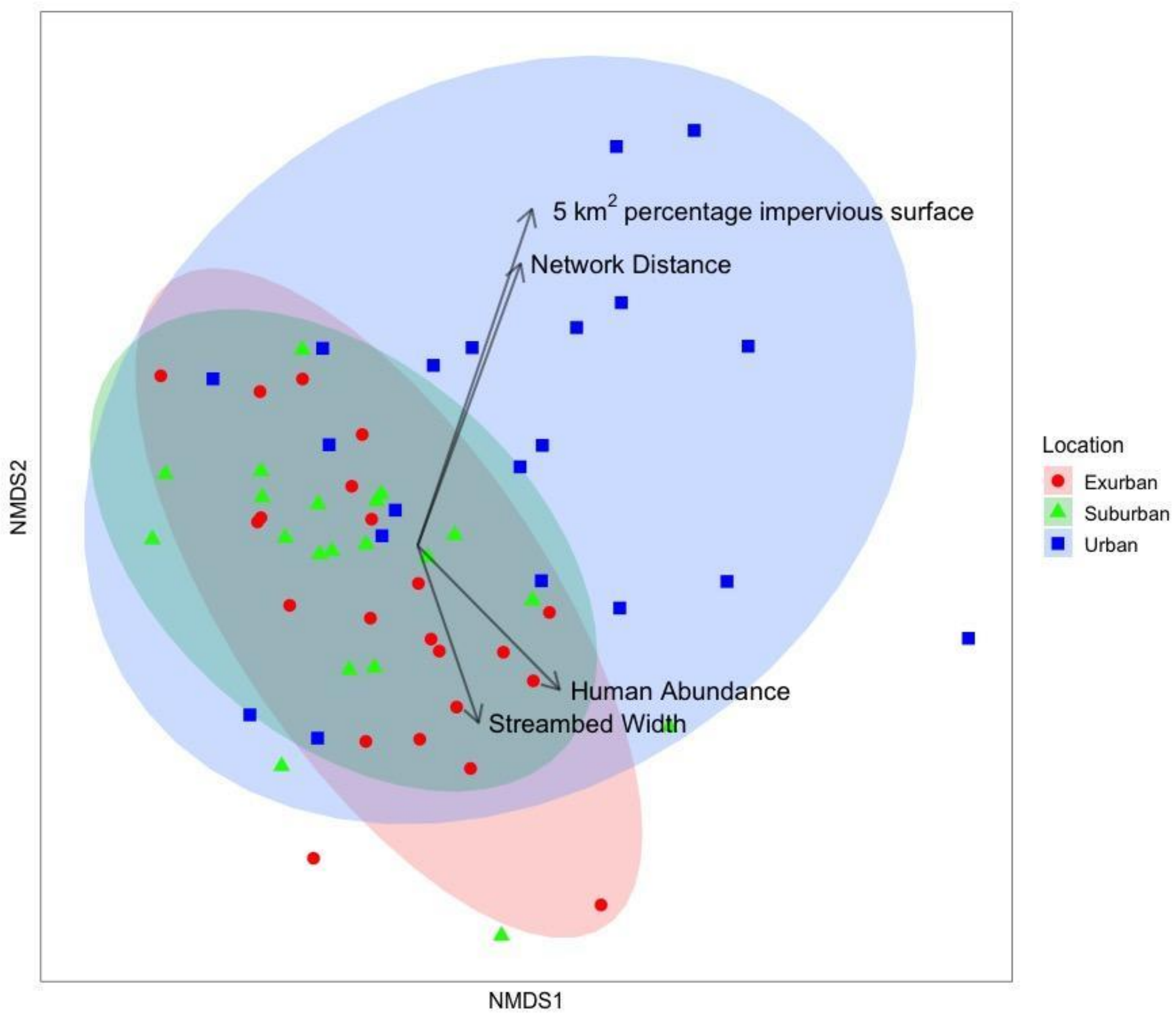

Figure 3: Non-metric Multidimensional Scaling (NMDS) ordination visualization of dissimilarity between urban (n = 21), suburban (n = 21) and exurban (n = 22) mammalian

communities in Tucson, Arizona, USA (stress = 0.18). Greater polygon overlap indicates greater similarity between communities.

The composition of mammals at suburban sites heavily overlapped with both urban and exurban sites in community composition as demonstrated in the NMDS visualization (stress = 0.18; Fig 3), but many urban sites were compositionally distinct from suburban and exurban sites (PerMANOVA, df=2, F = 3.74, R^2 = 0.11, p < 0.01). The urban community composition differed from both exurban (pairwiseAdonis, F = 5.26, R^2 = 0.12, p = 0.001) and suburban (pairwiseAdonis, F = 4.45, R^2 = 0.1, p = 0.001), whereas suburban and exurban communities were not statistically different (pairwiseAdonis, F = 1.19, R^2 = 0.03, p = 0.3) Mule deer (*Odocoileus hemionus;* IndVal = 49.6; p < 0.001)*,* domestic horses (*Equus ferus caballus;* IndVal= 45.5; p < 0.01), gray foxes (*Urocyon cinereoargenteus*; IndVal = 40.2; p < 0.01), and badgers (*Taxidea taxus;* IndVal = 22.5; p < 0.05) were indicators of exurban locations, whereas domestic cats (*Felis catus*) were indicator species for urban locations (IndVal= 82.29; p < 0.001). Coyotes (IndVal = 48.1; p < 0.01) and round-tailed ground squirrels (*Xerospermophilus tereticaudus*; IndVal = 24.7; p < 0.05*)* were indicators for suburban locations.

Occupancy and Relative Abundance Models

Detection probability models that included both season and effort emerged as a top model or within 2 AIC of the top model for all species, and so season and effort were included as covariates of detection probability in all subsequent occupancy models.

Bobcat Occupancy and Relative Abundance

Models including impervious surface and network distance both emerged as top models of both probability of site use (i.e., occupancy) and the rate of use (i.e., RAI) by bobcats. Landscape characteristics (i.e., impervious surface and network distance) were most influential for predicting bobcat presence, however, both landscape and local variables (e.g., shrub cover, wash width) influenced the rate at which bobcats use ephemeral stream channels. Initial model selection for scale of influence on bobcat presence showed a 1km^2 buffer around each camera to be the most influential scale. Subsequent testing identified our models based on landscape variables as those with the lowest AIC. The probability of bobcat presence at our sites decreased with increasing network distance ($\beta \pm SE = -0.72 \pm 0.33$) and increasing percentage of impervious surface ($\beta = -0.73 \pm 0.34$). The probability of bobcat use also decreased in adjacent upland areas ($\beta = -0.38 \pm 0.6$) compared to the main channel of the streambed. We also observed a decrease in the probability of bobcat presence in 2022 compared to 2021 ($\beta = -0.06 \pm 0.55$). In testing models of bobcat RAI, our landscape model including impervious surface and network distance was selected as the top model; however, our model including local variables (e.g., shrub cover, streambed width, and human rate of use) also fell within 2 AIC. The rate of bobcat use increased slightly with impervious surface cover ($\beta = 3.21e\text{-}5 \pm 1.37e\text{-}4$) and decreased with increasing network distance ($\beta = -4.29e\text{-}6 \pm 3.04e\text{-}6$). The rate of bobcat use decreased slightly in upland habitat ($\beta = -3.72e\text{-}3 \pm 5.47e\text{-}3$) compared to the main channel. Although the probability of bobcat use decreased from 2021 to 2022, we observed a slight increase in the rate of bobcat use in the year 2022 ($\beta = 6.27e\text{-}4 \pm 4.48e\text{-}3$). The next top model of bobcat RAI including local variables showed a decrease in bobcat rate of use in response to human rate of use ($\beta = -2.63E\text{-}03 \pm 1.14E\text{-}03$) and shrub cover ($\beta = -7.08E\text{-}03 \pm 1.00E\text{-}02$), and an increase in bobcat rate of use in response to wash width (β = 8.15E-06 ± 1.03E-04; see Appendix S2 for results).

Coyote Occupancy and Relative Abundance

Both local and landscape characteristics affected the probability and rate of coyote use, with human rate of use estimated to have the largest effect in both top models . Coyote presence was most influenced by impervious surface on the 0.5km^2 scale, and further model testing identified our global model as that with the lowest AIC. Impervious surface reduced the probability of coyote use (β = -38.84 ± 77.8) whereas network distance increased the probability of coyote use (β = 16.77 ± 33.3). Human rate of use (β = -0.69 ± 12.9), increasing distance to golf courses (β = -18.62 ± 42.2), and streambed width (β = -11.87 ± 39.8) all reduced the probability of use. The probability of coyote use was higher in upland areas (β = 23.06 ± 2097.3) compared to the main channel. We observed an increase in coyote use of streambeds in 2022 compared to 2021 (β = 16.92 ± 44.4). Variation in coyote RAI was best explained by our model based on literature; however, our models based on landscape and local variables both fell within 2 AIC. Coyote rate of use decreased where impervious surface was higher (β= -3.25e-4 ± 6.35e-4); however, this trend reversed in model 2, which was within 2 AIC. Coyote rate of use decreased with increasing network distance (β= -2.4e-5 ± 1.52e-5) and distance from golf courses (β= -6.2e-6 ± -5.9e-6), as well as in response to human rate of use (β = -0.01± 0.006). We also observed a decrease in coyote rate of use in upland habitat (β= -0.06 ± 0.03) compared to the main channel. The rate of coyote use decreased in the year 2022 compared to 2021 (β= -0.006 ± 0.02). Other models within 2 AIC showed a negative effect of network distance, human rate of use, distance from water, and wash width (see Appendix S2)

Peccary Occupancy and Relative Abundance

Our models that included landscape characteristics emerged as the top models for both probability of site use as well as rate of use by peccary, indicating that impervious surface cover and network distance have the largest influence on the use of ephemeral streams by peccaries. Peccary presence was most affected by impervious surfaces at the $2km^2$ scale, although other scales were within 2 ΔAIC of the top model. Our landscape model exhibited the lowest AIC and included impervious surface, network distance, placement, and year. Higher impervious surface cover reduced the probability of peccary use (β= -1.35 ± 0.47) whereas network distance increased the probability of peccary use (β= 1.44 ± 0.85). We observed lower site occupancy in main channels compared to upland habitat (β_{upland}= 1.59 ± 1.28), and we observed greater site occupancy in 2022 compared to 2021 (β_{2022}= 0.12 ± 0.68). During model selection for peccary rate of use, our landscape model was selected as the top model based on AIC. Impervious surface had a small positive effect on peccary rate of use (β= 6.8e-4 ±7.6e-4) whereas increasing network distance had a small negative effect on peccary rate of use (β= -8.5e-6 ± 1.6e-5). Rate of use by peccaries was also positively influenced by camera placement in upland habitat (β= 1.3e-4 ± 0.03) compared to the main channel. We observed an increase in peccary rate of use in the year 2022 compared to 2021 (β= 0.003 ± 0.02).

Table 1 Top occupancy and relative abundance models for bobcat, coyote, and peccary in Tucson, AZ based on AIC. Only models with a ΔAIC less than 2 were considered. Relative abundance index (RAI) is the rate of use, calculated by totaling the number of independent observations for each species and dividing the sum by the total number of nights the camera was deployed each year. IS refers to the average percentage impervious surface cover.

Species	**Model**	**Top models**	**AIC**	**ΔAIC**
Bobcat	Occupancy	Ψ (1km^2 IS + network distance + placement + year), p (season + effort)	1219.69	0.00
	Relative Abundance	1km^2 IS + network distance + placement + effort + year	-513.31	0.00
		Human RAI + 10m^2 shrub cover + streambed width + placement + effort + year	-515.11	1.81
Coyote	Occupancy	Ψ (0.5km^2 IS + network distance + Human RAI + Distance to golf course + streambed width + placement + year), p (season + effort)	1102.15	0.00
	Relative Abundance	0.5km^2 IS + Human RAI + Distance to golf course + placement + year + effort	-150.19	0.00
		0.5km^2 IS + network distance + placement + year + effort	-150.83	0.65
		Human RAI + Distance to water + streambed width + placement + year + effort	-151.62	1.44
Peccary	Occupancy	Ψ (2km^2 IS + network distance + placement + year), p (season + effort)	1279.99	0.00
	Relative Abundance	2km^2 IS + network distance + placement + effort + year	-135.3	0.00

DISCUSSION

Mammal Richness and Community Composition

Ephemeral streams in Tucson are used by a large number of terrestrial species, and may serve as vital habitat, particularly as movement corridors, for native mammals. Our findings are consistent with other studies showing that a variety of wildlife species use ephemeral streams (Sánchez-Montoya et al 2016). We observed lower mammalian richness and a different mammalian composition in locations with higher impervious surface coverage compared to locations with lower impervious surface coverage, consistent with our hypothesis that impervious surface would result in lower mammal richness. Contrary to our hypothesis; however, native mammal richness increased as network distance, or distance to an additional streambed, increased, which could be due to greater reliance on riparian corridors in more heavily urbanized areas.

Our findings add to a wide body of evidence indicating that urbanization can have significant ramifications on wildlife behavior (see Lowry et al. 2013 for review), community composition (Chace and Walsh 2006, Parsons et al. 2018), and species richness (Chace and Walsh 2006, McKinney 2008, Russo and Ancillotto 2015, Liang et al. 2022). Many species may avoid areas with high impervious surfaces due to the perception of high risk and/or low resource areas, such that urban and suburban environments can be highly impermeable for many native species (Chace and Walsh 2006, Moll et al 2020, Ordeñana et al 2010), potentially disrupting connectivity between populations. Mammalian species richness declines with higher impervious surface coverage (McKinney 2008, Magle et al. 2021). However, the effects of urbanization, including impervious surface, can vary depending on species (Larson and Sander 2022, Mueller et al. 2018, Tranquillo et al. 2024), individual animals (Graser et al. 2012), and spatial scales (Moll et al. 2020). Coyotes (Grubbs and Krausman 2009, Mueller et al. 2018) and bobcats (Schmidt et al. 2023) have been shown to exhibit avoidance of medium- and high-density impervious surface areas, whereas raccoons (*Procyon lotor;* Graser et al. 2012), foxes (Mueller et al. 2018), and some species of squirrel (Larson and Sander 2022, Tranquillo et al. 2024) may capitalize on such environments. Riparian corridors running through the city can reduce this impermeability; however, high impervious surface levels surrounding riparian corridors may disrupt the ability of wildlife to use the corridors by encompassing streambeds with potentially dangerous terrain offering little cover. Impervious surfaces surrounding ephemeral streams may therefore isolate streambeds from upland habitat and further decrease permeability to native species through urban areas.

Human presence during recreational activities (e.g. hiking, mountain biking, dog walking) can be considered another form of landscape disturbance, affecting the spatial ecology of wildlife at a different spatiotemporal scale via the ‘landscape of fear’ (Laundre et al. 2010, Palmer et al. 2022). We have documented a negative effect of human activity on mammal richness, a result consistent with several other studies reporting that some mammals either avoid highly developed areas or select these areas during nocturnal hours to avoid humans (Ticer et al. 1998, Tigas et al. 2002, Riley et al. 2003, Grubbs and Krausman 2009, Gaynor et al. 2018, Dunagan et al. 2019). Human activity can affect use or avoidance of habitat surrounding recreational trails for some mammals (George and Crooks 2006, Lewis et al. 2021). Human activity may have similar effects in streambeds, which are commonly used as walking trails by humans even if they are not explicitly designated to be so by municipal authorities (Webb et al. 2014).

Community composition changes with increased urbanization as hypothesized. Although the community composition of urban sites was distinct from suburban and exurban sites, suburban mammal communities exhibited surprising overlap with exurban communities, indicating that suburban areas may maintain native biodiversity and community composition. Suburban and exurban areas hosted a number of species that were not present in more urban areas, including American badgers (*Taxidea taxus),* black-tailed jackrabbits (*Lepus californicus*), and Harris' antelope ground squirrel (*Ammospermophilus harrisii*). A number of native species were restricted to minimally urbanized streambeds in exurban areas, including mountain lions (*Puma concolor)*, mule deer (*Odocoileus hemionus*), white-tailed deer (*Odocoileus virginianus*), all skunk species, and white-nosed coati (*Nasua narica*). Other, more generalist species, like bobcat, peccary, coyote, and racoons were present in some urban streambeds, but not others. Coyotes and peccary in the Southwest use these streambeds for travel (Bellantoni and Krausman 1993; Ticer et al. 1998, Grinder and Krausman 2001; Grubbs and Krausman 2009). We have established here that ephemeral streams serve as important habitat and connectivity corridors between and beyond suburbs for a wider variety of species. Łopucki and Kitowski (2017) found similar results, in which the small mammal community of Chełm, Poland resembled that of the surrounding rural community. Some suburban areas in the United States exhibit higher mammalian richness than surrounding urban and rural areas (Parsons et al. 2018, Grade et al. 2022). Suburban areas of Tucson typically have larger buffer zones surrounding streambeds compared to urban areas, and suburban streambeds are largely left with their natural streambed and riparian vegetation intact (Webb et al. 2014). Preserving floodplains and buffer zones around stream channels may provide additional benefits for native species in providing cover, food or prey, and thermal refugia, as well as ecosystem services such as reduction of flood damage, increasing connectivity between ground and surface water, and maintaining underground aquifers (Peipoch et al. 2015).

Occupancy and Abundance

We predicted that impervious surface cover would reduce both site occupancy and rate of use of our three focal species. Impervious surface did have this effect on site occupancy. Our findings are in line with other studies showing that all three species avoid highly developed areas, particularly when humans are present (bobcats: Tigas et al. 2002, Riley et al. 2003, Dunagan et al. 2019, Schmidt et al. 2023; coyotes: Riley et al. 2003, Grubbs and Krausman 2009; peccary: Bellantoni and Krausman 1993). Bobcats in California also exhibited lower probability of use as impervious surface cover increased (Schmidt et al. 2023). Contrary to our hypothesis; however, increasing impervious surface also increased the rate of use for all three species. This finding may indicate a greater reliance on streambeds in more well-developed areas, resulting in more frequent use of streambeds in more urban areas where these species are present. All three species use stream corridors for travel (Ticer et al. 1998, Grubbs and Krausman 2009, Hughes et al. 2019, Popescu et al. 2021) and may depend on these corridors for traversing heavily urban environments. We observed a reduction in the probability of bobcat use in more isolated streambeds (increasing network distance). Riparian corridors that are isolated from other riparian corridors may be difficult for bobcats to access or may be too developed to serve as adequate habitat. In contrast, both coyote and peccaries exhibited an increase in the probability of streambed use as network distance increased. Coyotes and peccary may therefore rely more heavily on riparian corridors in more well-developed areas in a manner similar to that of bobcats; however, coyotes and peccaries may be able to make use of isolated riparian corridors where bobcats are unable to. Still, the rate of use for all three species decreased with increased network distance, suggesting that these isolated washes are used more rarely than well-connected washes. Our results highlight the negative effects of high impervious surface coverage on wildlife habitat use and movement, as well as the importance of stream channel connectivity in allowing wildlife to navigate urban environments.

CONCLUSIONS

Urban sprawl represents a major source of habitat loss and fragmentation (Mcdonald, Kareiva, and Forman 2008, Seto, Güneralp, and Hutyra 2012, Jaureguiberry et al. 2022), and management of riparian corridors through urban and suburban areas may offer promising solutions to counteract the consequences of development. Riparian corridors can facilitate connectivity between wildlife populations that is essential for maintaining population health (Fremier et al. 2015, Correa Ayram 2016), and could play an important role in climate resilience by providing cooler microhabitats and allowing wildlife to move in response to environmental change (Nuñez et al. 2013, Fremier et al. 2015, Robillard et al. 2015). Our results demonstrate that these riparian corridors may be important for supporting native biodiversity in suburban areas. Maintaining connectivity with other streambeds and/or preserving streambeds to limit isolation can further support biodiversity and may limit animal movement across anthropogenic landscapes (e.g., roads, residential areas) by providing safer, more appealing corridors to move through. Where permeability through dense or large cities is a concern, limiting impervious surface cover around riparian areas and other potential corridors may be effective to maintain connectivity through dense urban areas. Existing river networks could therefore serve as a blueprint for restoration or preservation efforts (Fremier et al. 2015). Preserving the natural structure of streambeds (natural banks and floodplains) is important for maintaining native species richness and should be prioritized wherever possible. Restoring and preserving riparian areas also provides ecosystem services to the local community. Floodplains assist with flood control by allowing rivers space to absorb precipitation (Birkland et al. 2003, Peipoch et al. 2015), which will become increasingly important as climate change brings more extreme storms. Access to natural green spaces brings mental and physical health benefits to local residents (van den Bosch and Sang 2017, Tillmann et al. 2018, Rosales Chavez et al. 2023). Providing explicit walking trails for humans adjacent to streambeds may help restrict human activity within the streambed and limit the influence of human presence or activity on biodiversity and richness. Restricting human access to some streambeds, particularly those with high biodiversity, may be important for more sensitive species.

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Supplementary Material

Appendix A: Complete list of species 1) included in analyses and 2) observed on camera.

Appendix B: Linear regression modeling of native species richness – list of models tested, output of AIC tables, and summary output of top models

Appendix C: NMDS plot of species in relation to their communities

Appendix D: Occupancy and RAI models of bobcat, coyote, and peccary – list of models tested, output of AIC tables, and summary output of top models

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APPENDIX A: Lists of species observed across all wildlife cameras

Table S1 Complete list of mammalian species included in analyses. All species listed were included in ordination and indicator species analyses whereas only native species were included in modeling of native species richness

Scientific name	Common name
Ammospermophilus harrisii	Harris' antelope squirrel
Bos taurus	Domestic cow
Canis familiaris	Domestic dog
Canis latrans	Coyote
Conepatus leuconotus	American hog-nosed skunk
Didelphis virginiana	Virginia opossum
Equus ferus caballus	Domestic horse
Felis catus	Domestic cat
Lepus alleni	Antelope jackrabbit
Lepus californicus	Black-tailed jackrabbit
Lynx rufus	Bobcat
Mephitis macroura	Hooded skunk
Mephitis mephitis	Striped skunk
Nasua narica	White-nosed coati
Odocoileus hemionus	Mule deer
Odocoileus virginianus	White-tailed deer
Otospermophilus variegatus	Rock squirrel
Pecari tajacu	Collared peccary
Procyon lotor	Raccoon
Puma concolor	Mountain Lion
Spilogale gracilis	Western spotted skunk
Sylvilagus audubonii	Desert cottontail
Taxidea taxus	American badger
Urocyon cinereoargenteus	Gray fox
Xerospermophilus tereticaudus	Round-tailed ground squirrel

Table S2 Complete list of all species observed on wildlife cameras, including those that were excluded from statistical analyses (e.g. birds, reptiles), grouped by class

Scientific name	Common name
Amphibia	
Scaphiopus couchii	Couch's spadefoot toad
Aves	
Accipiter cooperii	Cooper's hawk
Amphispiza bilineata	Black-throated sparrow
Athene cunicularia	Burrowing owl
Auriparus flaviceps	Verdin
Bubo virginianus	Great-horned owl
Buteo jamaicensis	Red-tailed hawk
Callipepla gambelii	Gambel's quail
Cardinalis cardinalis	Northern cardinal
Cardinalis sinuatus	Pyrrhuloxia
Catharus guttatus	Hermit thrush
Calypte anna	Anna's hummingbird
Calypte costae	Costa’s hummingbird
Campylorhynchus brunneicapillus	Cactus wren
Chordeiles minor	Common nighthawk
Corvus corax	Raven
Empidonax traillii	Willow flycatcher
Geococcyx californianus	Greater roadrunner
Haemorhous mexicanus	House finch
Melanperpes uropygialis	Gila woodpecker
Meleagris gallopavo	Wild turkey
Melospiza melodia	Song sparrow
Melozone aberti	Abert's towhee

Mimus polyglottos	Northern mockingbird
Myiarchus cinerascens	Ash-throated flycatcher
Passer domesticus	House sparrow
Pyrocephalus obscurus	Vermillion flycatcher
Saynoris nigricans	Black phoebe
Spizella passerina	Chipping sparrow
Streptopelia decaocto	Eurasian collared dove
Taxostoma curviostre	Curve-billed thrasher
Turdus migratorius	American robin
Zenaida asiatica	White-winged dove
Zenaida macroura	Mourning dove
Zonotrichia leucophrys	White-crowned sparrow
Mammalia	
Ammospermophilus harrisii	Harris' antelope squirrel
Bos taurus	Domestic cow
Canis familiaris	Domestic dog
Canis latrans	Coyote
Chaetodipus baileyi	Bailey's pocket mouse
Chaetodipus penicillatus	Desert pocket mouse
Conepatus leuconotus	American hog-nosed skunk
Didelphis virginiana	Virginia opossum
Dipodomys merriami	Merriam's kangaroo rat
Equus ferus caballus	Domestic horse
Felis catus	Domestic cat
Homo sapiens	Human
Lepus alleni	Antelope jackrabbit
Lepus californicus	Black-tailed jackrabbit
Lynx rufus	Bobcat
Mephitis macroura	Hooded skunk
Mephitis mephitis	Striped skunk
Nasua narica	White-nosed coati

Neotoma albigula	White-throated woodrat
Odocoileus hemionus	Mule deer
Odocoileus virginianus	White-tailed deer
Otospermophilus variegatus	Rock squirrel
Pecari tajacu	Collared peccary
Peromyscus eremicus	Cactus mouse
Procyon lotor	Raccoon
Puma concolor	Mountain lion
Spilogale gracilis	Western spotted skunk
Sylvilagus audubonii	Desert cottontail
Taxidea taxus	American badger
Urocyon cinereoargenteus	Gray fox
Xerospermophilus tereticaudus	Round-tailed ground squirrel
Reptilia	
Aspidoscelis sonorae	Sonoran whiptail
Aspidoscelis tigris	Tiger whiptail
Crotalus sp.	Rattlesnake sp.
Gopherus agassizii	Desert tortoise
Heloderma suspectum	Gila monster
Sceloporus magister	Desert spiny lizard

APPENDIX B: Model selection of native species richness

Models in bold type indicate those with the lowest AIC.

Selection of spatial scale for subsequent models:

model1: lm(native_mammals ~ effort + IS(10m^2))

model2: lm(native_mammals ~ effort + IS(50m^2))

model3: lm(native_mammals ~ effort + IS(100m^2))

model4: lm(native_mammals ~ effort + IS(0.5km^2))

model5: lm(native_mammals ~ effort + IS(1km^2))

model6: lm(native_mammals ~ effort + IS(2km^2))

model7: lm(native_mammals ~ effort + IS(3km^2))

model8: lm(native_mammals ~ effort + IS(5km^2))

Hypothesis 1: Species richness is primarily dependent on human disturbance, including human abundance and impervious surface cover.

model1: lm(native_mammals ~ IS (5km^2) + Human RAI + effort + placement)

Hypothesis 2: Species richness is primarily dependent on landscape characteristics, such as connectivity (distance to an additional streambed) and impervious surface cover.

model2: lm(native_mammals ~ IS (5km^2) + network distance + effort + placement)

Hypothesis 3: Species richness is primarily dependent on local, site-level characteristics, such as whether the streambed is channelized or not (condition), streambed width, and shrub cover.

model3: lm(native_mammals ~ streambed width + tree canopy (10m^2) + effort + placement)

Hypothesis 4: Species richness is primarily dependent on a combination of local and landscape variables.

model4: lm(native_mammals ~ IS (5km^2) + Network distance + Human RAI + streambed width + effort + placement)

Table S3 Results of model selection of native mammalian richness based on AIC

	df	AICc	delta
model1	6	260.6	0
model4	8	261.2	0.51
model2	6	267.1	6.46
model3	6	280.8	20.19

Table S4 Beta estimates and standard error of top model for native mammal richness

	Estimate	Std. Error
Intercept	6.78	0.97
Effort	0.004	0.001
Upland placement	-0.99	0.62
Human RAI	-0.39	0.15
Impervious surface (5km^2)	-0.07	0.02

Table S5 Beta estimates and standard errors of 2nd top model for native mammal richness, which fell within 2 AIC

	Estimate	St. Error
Intercept	7.09	0.95
Effort	0.004	0.001
Upland placement	-1.04	0.62
Impervious surface (5km^2)	-0.08	0.02
Network distance	0.0002	0.004
Human RAI	-0.39	0.15
Streambed width	-0.02	0.01

Modeling native mammal richness with stacked data

To identify increases or decreases in native mammal richness between years, we stacked our data in a manner similar to the datasets used in our occupancy models (see Methods section). Data from each camera site was separated and analyzed by year, producing two rows of data for each site (one for each year). We found that a different scale of impervious surface – 0.5km^2 –

was favored over $5km^2$, although impervious surface cover on a $5km^2$ scale still fell within 2 AIC. We used 0.5km2 impervious surface cover for subsequent modeling of stacked data. Models of native mammal richness were otherwise identical to those tested with unstacked data. Our global model representing hypothesis 4 was selected as the top model based on AIC (see Table S4 below).

Table S6 Results of model selection of native mammalian richness based on AIC using stacked data

	df	AICc	delta
model1	9	260.6	0
model4	7	261.2	0.51
model2	7	267.1	6.46
model3	7	280.8	20.19

Table S7 Beta estimates and standard error of top model for native mammal richness using stacked data

	Estimate	St. Error
Intercept	6.34	0.59
Effort	0.005	1.00E-03
Upland placement	-1.06	4.60E-01
Impervious surface (0.5km2)	-7.00E-02	1.00E-02
Network distance	2.00E-04	2.00E-04
Human RAI	-3.90E-01	1.00E-01
Streambed width	-3.00E-02	8.00E-03
Year (2022)	-7.00E-02	3.50E-01

APPENDIX C: NMDS ordination results showing species mapped

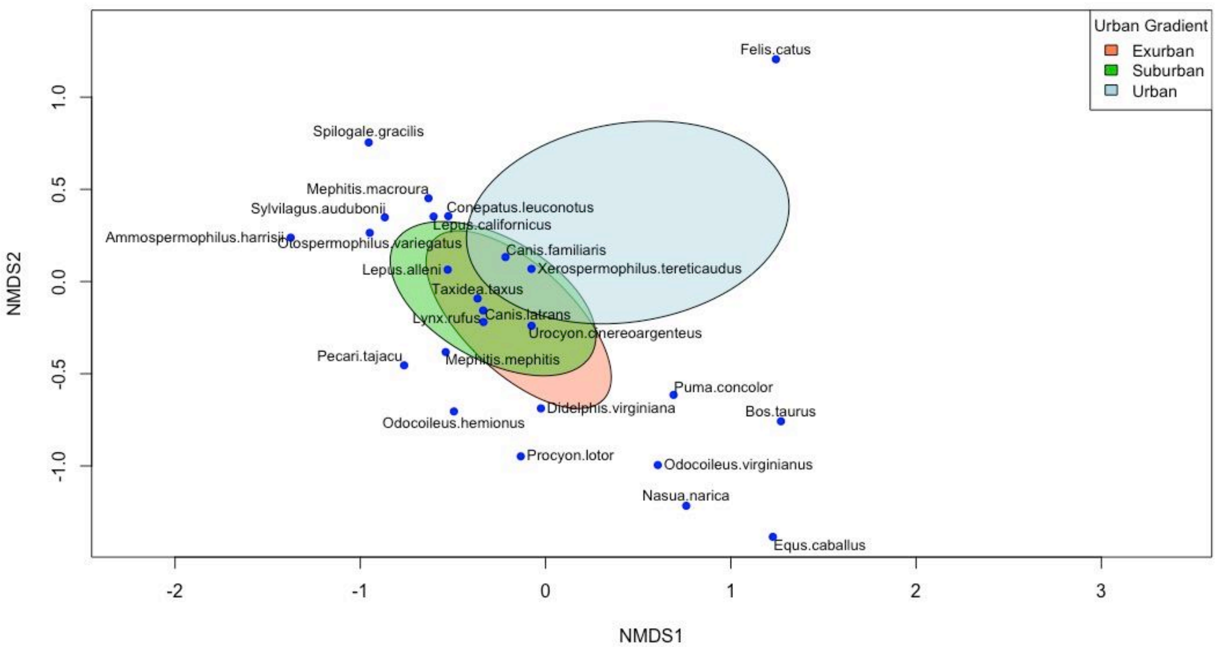

Figure S1 NMDS ordination plot showing where each species observed maps in relation to the different communities across level of urbanization.

APPENDIX D: Occupancy & abundance modeling of bobcat, coyote, and peccary

Note that dist.water (distance to nearest perennial water source) and dist.golf (distance to nearest golf course) are positively correlated ($r = 0.77$) and so we tested multiple models for some hypotheses to separate these variables. Bold models indicate those with the lowest AIC.

Bobcat detection probability model selection:

dp1 : occu(~ 1 ~ 1, bob.umf)

dp2 : occu(~ season ~ 1, bob.umf)

dp3 : occu(~ effort ~ 1, bob.umf)

dp4 : occu(~ season + effort ~ 1, bob.umf)*

*within 2 AIC of top model, and so we used both season and effort as site covariates.

Selection of spatial scale for subsequent occupancy models:

model1 : occu(~ season + effort ~ IS(10m^2) + year)

model2 : occu(~ season + effort ~ IS(50m^2) + year)

model3 : occu(~ season + effort ~ IS(100m^2) + year)

model4 : occu(~ season + effort ~ IS(0.5km^2) + year)

model5 : occu(~ season + effort ~ IS(1km^2) + year)

model6 : occu(~ season + effort ~ IS(2km^2) + year)

model7 : occu(~ season + effort ~ IS(3km^2) + year)

model8 : occu(~ season + effort ~ IS(5km^2) + year)

Bobcat occupancy model selection:

Hypothesis 1: *Bobcat occupancy influenced by impervious surface and human abundance (Tigas et al. 2002, Riley et al. 2003, Riley 2006, Dunagan et al. 2019, Schmidt et al. 2023)*

model1: occu(~ season + effort ~ IS(1km^2) + Human RAI + placement + year)

Hypothesis 2: *Bobcat occupancy is primarily shaped by surrounding landscape features like impervious surface cover and connectivity*

model2: occu(~ season + effort ~ IS(1km^2) + network distance + placement + year)

Hypothesis 3: *Bobcat occupancy is primarily shaped by local, site-level features like human abundance, streambed width, and shrub cover.*

model3: occu(~ season + effort ~ Human RAI + streambed width + shrub(10m2) + placement + year)

Hypothesis 4: *Bobcat occupancy is shaped by a combination of local and landscape features.*

model4: occu(~ season + effort ~ IS(1km^2) + network distance + Human RAI + streambed width + shrub(10m^2) + placement + year)

Table S8 Results of model selection bobcat occupancy based on AIC. *K* represents the total number of parameters in the model

	K	AICc	delta
model2	9	1241.94	0
model4	12	1244.98	2.93
model1	9	1245.2	3.25
model3	10	1260.78	18.84

Table S9 Beta estimates and standard errors of top bobcat occupancy model

Occupancy (logit-scale)		
	Estimate	St. Error
Intercept	1.47	0.45
Impervious surface (1km^2)	-0.73	0.34
Network distance	-0.72	0.33
Upland placement	-0.38	0.6
Year (2022)	-0.06	0.55
Detection (logit-scale)		
	Estimate	St. Error
Intercept	-1.21	0.38
Monsoon season	-0.25	0.17
Winter season	-0.03	0.17
Effort	1.2	0.38
Number of sites: 111		

Bobcat abundance model selection:

Base model:

model1: glm(Bobcat RAI ~ IS(1km2) + Human RAI + placement + effort + year)

Landscape model:

model2: glm(Bobcat RAI ~ IS(1km^2) + network distance + placement + year + effort)

Local model:

model3: glm(Bobcat RAI ~ Human RAI + shrub(10m^2) + streambed width + placement + year + effort)

Combination:

model4.1: glm(Bobcat RAI ~ IS(1km^2) + Human RAI + network distance + streambed width + shrub(10m^2) + placement + effort + year)

Table S10 Results of model selection of bobcat RAI model using AIC

	df	AICc	delta
model2	7	-513.31	0
model3	8	-515.11	-1.81
model4	10	-515.68	-2.37
model3	7	-517.45	-4.14

Table S11 Beta estimates and standard errors for top bobcat RAI model

	Estimate	St. Error
Intercept	1.68E-02	9.02E-03
Impervious surface ($1km^2$)	3.20E-05	1.37E-04
Network distance	-4.29E-06	3.03E-06
Upland placement	-3.72E-03	5.47E-03
Year (2022)	6.27E-04	4.48E-03
Effort	7.98E-03	9.16E-03

Table S12 Beta estimates and standard errors for bobcat RAI model that fell within 2 AIC of top model

	Estimate	St Error
Intercept	1.82E-02	7.38E-03
Human RAI	-2.63E-03	1.14E-03
Shrub cover ($10m^2$)	-7.08E-03	1.00E-02
Streambed width	8.15E-06	1.03E-04
Upland placement	-6.02E-03	5.52E-03
Year (2022)	3.43E-05	4.42E-03
Effort	9.28E-05	8.28E-03

Coyote detection probability model selection:

dp1 : occu(~ 1 ~ 1, coy.umf)*

dp2 : occu(~ season ~ 1, coy.umf)

dp3 : occu(~ effort ~ 1, coy.umf)

dp4 : occu(~ season + effort ~ 1, coy.umf)**

*Model did not converge

**within 2 AIC of top model, and so we used both season and effort as site covariates.

Selection of spatial scale for subsequent occupancy models:

model1 : occu(~ season + effort ~ IS(10m^2) + year)

model2 : occu(~ season + effort ~ IS(50m^2) + year)

model3 : occu(~ season + effort ~ IS(100m^2) + year)

model4 : occu(~ season + effort ~ IS(0.5km^2) + year)

model5 : occu(~ season + effort ~ IS(1km^2) + year)

model6 : occu(~ season + effort ~ IS(2km^2) + year)

model7 : occu(~ season + effort ~ IS(3km^2) + year)

model8 : occu(~ season + effort ~ IS(5km^2) + year)

Coyote occupancy model selection:

Note that distance to the nearest golf course (dist.golf) and distance to the nearest perennial freshwater source (dist.water) were correlated with each other (r = 0.77) and so were not included in the same model. Including vegetation metrics (shrub(10m^2) or tree(10m^2)) produced errors and so were not included in our coyote models.

Hypothesis 1: Coyote occupancy is primarily shaped by human abundance and distance to the nearest golf course (Grinder and Krausman 2001, Tigas et al. 2002, Riley 2006, Grubbs and Krausman 2009).

model1: occu(~ season + effort ~ IS(0.5km2) + Human RAI + dist.golf + placement + year)

Hypothesis 2: *Coyote occupancy is primarily shaped by surrounding landscape features like impervious surface cover and connectivity*

model2: occu(~ season + effort ~ IS(0.5km^2) + network distance + placement + year)

Hypothesis 3: *Coyote occupancy is primarily shaped by local, site-level features like human*

model3.1: occu(~ season + effort ~ Human RAI + dist.golf + streambed width + placement + year)

model3.2: occu(~ season + effort ~ Human RAI + dist.water + streambed width + placement + year)

Hypothesis 4: Coyote occupancy is shaped by a combination of local and landscape features.

model4.1: occu(~ season + effort ~ IS($0.5km^2$) + network distance + Human RAI + dist.golf + streambed width + placement + year)

model4.2: occu(~ season + effort ~ IS($0.5km^2$) + network distance + Human RAI + dist.water + streambed width + placement + year)

Table S13 Model selection results of coyote occupancy model using AIC. *K* represents the total number of parameters in the model

	K	AICc	delta
model4.1	12	1102.15	0
model2	9	1109.68	7.53
model1	11	1113.19	11.04
model4.2	12	1113.87	11.72
model3.1	10	1133.46	31.31
model3.2	10	1137.48	35.33

Table S14 Beta estimates and standard errors of top coyote occupancy model

Occupancy (logit-scale)		
	Estimate	St. Error
Intercept	41.17	84
Impervious surface (0.5km^2)	-38.84	77.8
Network distance	16.77	33.3
Human RAI	-0.69	12.9
Distance to golf course	-18.62	42.2
Streambed width	-11.89	39.8
Upland placement	23.06	2097.3
Year (2022)	16.92	44.4
Detection (logit-scale)		
	Estimate	St. Error
Intercept	-0.75	0.32
Monsoon season	-0.11	0.18
Winter season	0.24	0.19
Effort	1.98	0.33
Number of sites: 107		

Coyote abundance model selection:

Base model:

model1 : glm(Coyote RAI ~ IS(0.5km^2) + Human RAI + dist.golf + placement + year + effort)

Landscape model:

model2 : glm(Coyote RAI ~ IS(0.5km^2) + network distance + placement + year + effort)

Local model:

model3.1 : glm(Coyote RAI ~ Human RAI + dist.golf + streambed width + placement + year + effort)

model3.2 : glm(Coyote RAI ~ Human RAI + dist.water + streambed width + placement + year + effort)

Combination:

model4.1 : glm(Coyote RAI ~ IS(0.5km2) + network distance + Human RAI + dist.golf + streambed width + placement + year + effort)

model4.2 : glm(Coyote RAI ~ IS(0.5km2) + network distance + Human RAI + dist.water + streambed width + placement + year + effort)

Table S15 Model selection results of coyote RAI models using AIC

	df	AICc	delta
model1	8	-150.19	0
model2	7	-150.83	-0.64
model3.2	8	-151.62	-1.43
model4.2	10	-152.92	-2.73
model3.1	8	-154.12	-3.93
model4.1	10	-156.45	-6.26

Table S16 Beta estimates and standard errors of top coyote RAI model (model 1).

	Estimate	St. Error
Intercept	1.55E-01	4.88E+02
Impervious surface (0.5km^2)	-3.25E-04	5.52E-04
Human RAI	-1.06E-02	5.66E-03
Distance to golf course	-6.19E-06	5.89E-06
Upland placement	-5.84E-02	2.71E-02
Year (2022)	-6.01E-03	2.24E-02
Effort	4.87E-02	4.49E-02

Table S17 Beta estimates and standard errors of coyote RAI landscape model (model 2).

	Estimate	St. Error
Intercept	1.28E-01	4.22E-02
Impervious surface (0.5km^2)	2.10E-04	6.21E-04
Network distance	-2.50E-05	1.46E-05
Upland placement	-4.22E-02	2.81E-02
Year (2022)	3.96E-03	2.26E-02
Effort	4.06E-02	4.45E-02

Table S18 Beta estimates and standard errors of coyote RAI local model (model 3.2).

	Estimate	St. Error
Intercept	1.36E-01	3.74E-02
Human RAI	-1.08E-02	5.68E-03
Distance to permanent water	-1.69E-06	5.91E-06
Streambed width	-9.17E-04	5.42E-04
Upland placement	-5.44E-02	2.79E-02
Year (2022)	2.70E-04	2.24E-02
Effort	5.86E-02	4.23E-02

Peccary detection probability model selection:

dp1 : occu(~ 1 ~ 1)

dp2 : occu(~ season ~ 1)

dp3 : occu(~ effort ~ 1)

dp4 : occu(~ season + effort ~ 1)

Selection of spatial scale for subsequent occupancy models:

model1 : occu(~ season + effort ~ IS(10m^2) + year)

model2 : occu(~ season + effort ~ IS(50m^2) + year)

model3 : occu(~ season + effort ~ IS(100m^2) + year)

model4 : occu(~ season + effort ~ IS(0.5km^2) + year)

model5 : occu(~ season + effort ~ IS(1km^2) + year)

model6 : occu(~ season + effort ~ IS(2km^2) + year)

model7 : occu(~ season + effort ~ IS(3km^2) + year)

model8 : occu(~ season + effort ~ IS(5km^2) + year)

Peccary occupancy model selection:

Hypothesis 1: Peccary occupancy is primarily shaped by impervious surface and tree cover (Bellantoni and Krausman 1993, Ticer et al. 1998, Bender et al. 2014).

model1 : occu(~ season + effort ~ IS(2km^2) + tree(10m^2) + distance to perennial water + placement + year)

Hypothesis 2: Peccary occupancy is primarily shaped by surrounding landscape features like impervious surface cover and connectivity

model2: occu(~ season + effort ~ IS(2km^2) + network distance + placement + year)

Hypothesis 3: Peccary occupancy is primarily shaped by local, site-level features like tree cover, streambed width, and human abundance.

model3: occu(~ season + effort ~ tree(10m^2) + distance to perennial water + streambed width + Human RAI + placement + year)

Hypothesis 4: Peccary occupancy is shaped by a combination of local and landscape features.

model4: occu(~ season + effort ~ IS(2km^2) + network distance + distance to perennial water + Human RAI + tree(10m^2)+ streambed width + placement + year)

Table S19 Model selection results of peccary occupancy models. *K* represents the total number of parameters in the model

	K	AICc	delta
model2	9	1284.8	0
model4	13	1290.8	6
model1	10	1293.05	8.25
model3	11	1299.57	14.77

Table S20 Beta estimates and standard errors of top peccary occupancy model, our landscape model

Occupancy (logit-scale)		
	Estimate	St. Error
Intercept	1.94	0.66
Impervious surface ($2km^2$)	-1.35	0.47
Network distance	1.44	0.85
Upland placement	1.59	1.28
Year (2022)	0.12	0.68
Detection (logit-scale)		
	Estimate	St. Error
Intercept	-1.66	0.38
Monsoon season	0.44	0.17
Winter season	0.55	0.17
Effort	1.25	0.38
Number of sites: 107		

Peccary abundance model selection:

Base model:

model1: glm(Peccary RAI ~ IS($2km^2$) + tree($10m^2$) + placement + effort + year)

Landscape models:

model2: glm(Peccary RAI ~ IS($2km^2$) + network distance + placement + effort + year)

Local model:

model3: glm(Peccary RAI ~ tree($10m^2$)+ streambed width + Human RAI + placement + effort + year)

Combination:

model4: glm(Peccary RAI ~ IS(2km2) + network distance + Human RAI + tree($10m^2$)+ streambed width + placement + effort + year)

Table S21 Model selection results of peccary RAI models

	df	AICc	delta
model2	7	-132.55	0
model4	8	-138.79	-6.24
model1	11	-139.65	-7.1
model3	9	-141.65	-9.1

Table S22 Beta estimates and standard errors of the top peccary RAI model, our landscape model

	Estimate	St. Error
Intercept	-1.59E-02	5.05E-02
Impervious surface ($2km^2$)	6.76E-04	7.64E-04
Network distance	-8.48E-06	1.59E-05
Upland placement	1.27E-04	3.02E-02
Year (2022)	3.34E-03	2.44E-02
Effort	7.16E-02	4.98E-02

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CHAPTER 3

Use of a public survey to evaluate the efficacy of nonlethal wildlife deterrent techniques and model human-wildlife conflict in a semiarid, urban landscape

ABSTRACT

Human-wildlife conflict is a major conservation concern worldwide that is projected to increase in frequency and intensity as urban centers expand. Urban residents can employ a number of nonlethal strategies in order to deter wildlife from their property and reduce conflict; however, some strategies may be more effective than others. We distributed an online survey to residents in Tucson, AZ via neighborhood associations to 1) assess attitudes toward and conflict with three focal species, 2) identify which nonlethal deterrent strategies are most effective for each species, and 3) determine whether habitat fragmentation and degradation influence conflict. We focused our survey on three species, coyote (*Canis latrans*), bobcat (*Lynx rufus*), and collared peccary (*Pecari tajacu*), due to the percentage of resident complaints involving these three species. We found that attitudes toward each species were largely positive, and that a relatively small percentage of respondents attempted to deter each species from their property. Aversive conditioning was highly effective at causing all three species to immediately leave. Aversive conditioning also showed long-term efficacy for deterring bobcats but showed mixed long-term success for deterring coyote and peccary. Proactive strategies, such as securing resources or installing barriers, showed mixed long-term success for peccary and coyote, but were not effective for deterring bobcats long-term. We did not find any relationship between proportion of conflict (i.e., attempts to deter wildlife) in survey locations and percentage impervious surface cover, distance to nearest riparian corridor, or percentage impervious surface

within the riparian corridor. Our results provide Tucson residents with more detailed information regarding which strategies are most effective at deterring coyote, bobcat, and peccary in the short- and long-term. Our results also suggest that habitat degradation is not the main driver of human-wildlife conflict in Tucson; however, further study is needed.

INTRODUCTION

As human populations increase and urban environments expand outward into undeveloped areas, human-wildlife conflict (HWC) is predicted to increase in frequency and intensity (Soulsbury and White 2015). Urban growth increases the density of humans across a wider land area, such that wildlife are more likely to encounter humans. Furthermore, increases in habitat destruction and fragmentation due to development lead to decreases in the quality and availability of natural habitat providing shelter and food to animals, which can increase their reliance on anthropogenic resources (Kolowski and Holekamp 2006, Newsome et al. 2015, Mukeka et al. 2019). As more animals learn to take advantage of anthropogenic resources (e.g. trash, domestic animals), HWC could be further exacerbated (Schell et al. 2021).

While lethal removal of wildlife has traditionally been used as a response to HWC, the practice is increasingly controversial with the general public (Jackman and Rutberg 2015, Bergstrom 2017, van Eeden et al. 2017). Lethal removal may induce a number of unintended consequences that exacerbate HWC (see Nyhus 2016 for review). These trends and consequences have led wildlife managers and urban residents to search for ways of coexisting with wildlife and reducing HWC, including behaviorally based techniques such as aversive conditioning and exclusionary devices. Aversive conditioning uses associative learning principles to deter animals by inflicting an unpleasant experience, causing the animal to have a negative association with the property or the resource (see Smith, Linnell, and Odden 2000 for review). The use of aversive conditioning to deter urban wildlife may include making loud noises, spraying water, or using a repellent like mace (AZGFD 2024). Preventative strategies, such as securing resources or removing attractants, may prevent the animal from identifying a potential resource in the first place (Elliot et al. 2016, Sampson and van Patter 2020, Mazué et al. 2023), and chemical and physical barriers may prevent the animal from accessing the property (Apfelbach et al. 2005, Murray and St. Clair 2017).

Studies on the efficacy of aversive conditioning are typically focused on high-intensity conditioning orchestrated by wildlife professionals (e.g., Leigh and Chamberlain 2008, Mazur 2010, Lajeunesse et al. 2023). In contrast, few studies have evaluated the effectiveness of aversive conditioning when employed by residents (but see Bonnell and Breck 2017). Because most wildlife encounters do not occur in the presence of a wildlife professional, it is important to determine whether urban residents can effectively deter an animal engaging in problematic behaviors using the behavioral methods suggested by government personnel. If resident hazing is effective, encounters between residents and wildlife could further allow for more widespread and consistent aversive conditioning of local wildlife populations than government agencies are able to provide, which may be more effective at discouraging problem behaviors (Sampson and van Patter 2020, Lajeunesse et al. 2023). Finally, although deterrent strategies may be useful for preventing some conflict, investigation into the root causes of HWC is vital for mitigating conflict (Sampson and van Patter 2020). For example, conflict resulting from low natural resource availability likely requires further habitat management to adequately address conflict (Mukeka et al. 2019).

In Arizona, U.S.A, bobcat (*Lynx rufus*), coyote (*Canis latrans*), and collared peccary (*Pecari tajacu*) are among the largest wildlife species living in urban areas. Some individuals take advantage of human infrastructure, landscaping, and human-derived food sources, including pet food, trash, and compost (Bellantoni and Krausman 1993, Grubbs and Krausman 2009*a*). Complaints about bobcat, coyote, and peccary made up almost half of logged complaints (n = 10,003 total complaints) about urban wildlife in Arizona (*personal communication with AZGFD, 2019*). Many urban residents in Arizona have experienced loss of smaller pets due to predation by coyotes (Lawrence and Krausman 2011), which may also contribute toward their fear of bobcats. A number of Tucson residents also care for small livestock such as chickens, which may be vulnerable to predation by bobcat and coyotes. Peccary can cause property damage, and residents and their pets have been attacked occasionally by peccary that view canids as a threat (Wegner 2024).

In this study, we distributed an online survey to urban residents of Tucson, Arizona to gage human-wildlife conflict and effectiveness of deterrent strategies advertised by state agencies, as well as determine whether HWC is influenced by local habitat conditions. Specifically, our objectives were to: (1) assess attitudes toward, and intensity of conflict with, bobcats, coyotes, and peccaries, (2) identify which strategies are most effective at deterring each species in the short- and long-term, and (3) determine whether reported conflict is affected by habitat degradation and fragmentation. Our goal was to use this study to provide residents with more detailed information on how to reduce or prevent HWC so that humans and wildlife can more safely coexist in Tucson and other cities.

METHODS

Tucson has a population of approximately 550,000 people (U.S. Census Bureau 2024) and serves as the center of a metropolitan area with nearly 1 million residents. The Tucson metropolitan area is largely surrounded by protected natural areas, with two units of Saguaro National Park lying to the east and west of Tucson, and Coronado National Forest bordering the city on the north (Figure 1). At the time the survey was conducted, much of the area south of the city was also largely undeveloped Sonoran Desert. Ephemeral streams of varying width and length run throughout Tucson, forming riparian corridors that provide natural resources to wildlife and increase connectivity between natural areas despite urban development (Webb et al. 2014). The riparian corridors vary in their degree of urbanization however, in that some riparian corridors are completely cemented over whereas others have natural resources available (e.g., vegetation for food and shelter, exposed soil for foraging and burrowing). Tucson is therefore a good location to examine the efficacy of deterrent techniques provided by AZGFD and explore patterns between human wildlife conflict and habitat alteration or degradation.

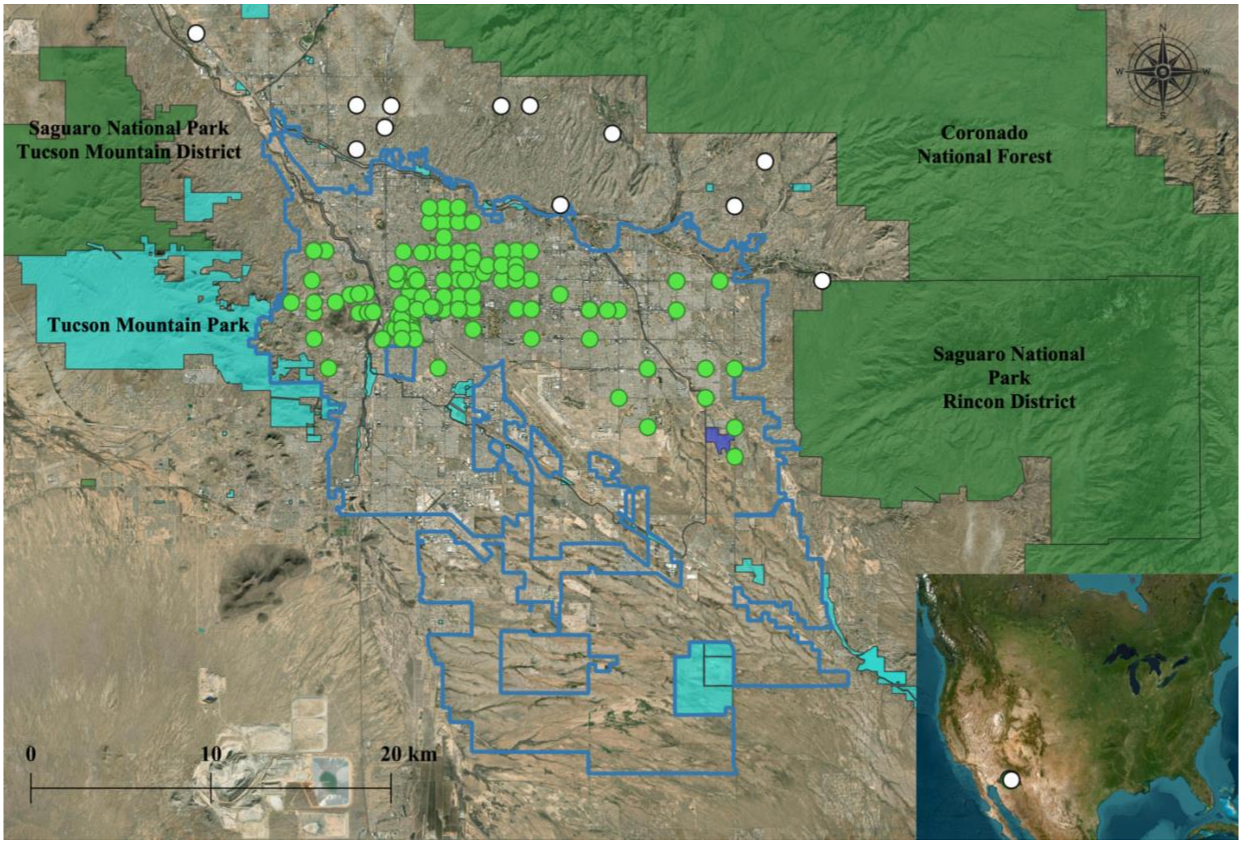

Figure 1: Survey locations in Tucson, AZ. Green points indicate survey responses within Tucson city limits that were retained in the dataset for analysis. The Tucson city jurisdiction is outlined in blue. Federal (dark green), state (dark blue), and county (light blue) public conservation lands are highlighted.

Survey Instrument

We distributed an online questionnaire to Tucson residents via Qualtrics software (IRB #2010122939) in March 2022. A questionnaire link was distributed over social media (NextDoor) and email listservs of Tucson neighborhood associations (see Supplemental Information for survey instrument). The questionnaire asked respondents about the number of years spent at their residence, location as defined as the nearest major crossroads, and whether respondents had seen any of the three main study species: coyote, bobcat, and peccary. If the respondent had seen any of these species on their property, we asked how often each species was seen – weekly, monthly, seasonally, annually, or rarely (species seen only a handful of times or less). If the respondent chose "seasonally," they were asked whether the animal was most likely to be seen during the winter season (November–January), dry season (February–June), or monsoon season (July–October). We then asked whether they attempted to deter that species. If they responded affirmatively, they received follow-up questions about the reasons for deterring (write-in) and which techniques were used (multiple choice). The multiple-choice question asked residents that attempted to deter the animal(s) to choose from a list of techniques that are provided on the Arizona Game and Fish Department (AZGFD) website as suggested methods for deterring wildlife (AZGFD 2024). Respondents were offered the opportunity to write a response if they chose "Other." If aversive conditioning was chosen (e.g., yelling, waving objects, etc.), the residents were asked whether the animal(s) left immediately (multiple choice options of yes, no, or unsure). All respondents that attempted to deter were asked whether any of the three focal species were subsequently detected. Responses were used to determine whether deterrent techniques were effective over short-term and long-term intervals. If respondents answered "yes" to the animal leaving the property, we considered the strategy to be effective short-term for that respondent. If the respondent did not see the same species on their property again, we considered the strategy to be effective long-term. All respondents were also asked to report their attitude toward each species (seven-point Likert scale ranging from extremely positive to extremely negative).

Before distributing our survey instrument to the public, we vetted the questionnaire with a panel of experts (wildlife professionals, including academic researchers and government personnel) to establish validity (Carmines and Zeller 1979). We also distributed the questionnaire to a pilot population separate from the study population to pinpoint any potential issues (Bowden et al. 2002). Because the primary purpose of the questionnaire was to evaluate the outcome of human-wildlife interactions rather than human opinions, we did not attempt to establish reliability (Carmines and Zeller 1979), as answers would change depending on whether the resident had interacted with wildlife between survey periods.

Data Analysis

We used R and RStudio to view and analyze the results (R Core Team 2022). To estimate efficacy of each deterrent strategy, we calculated the percentage of successful deterrent attempts. Deterrent attempts were considered successful short-term at deterring wildlife if respondents answered "yes" to the animal leaving their property. Deterrent attempts were considered effective long-term if those same respondents indicated that they had not seen the species on their property again. To estimate habitat degradation and fragmentation in the immediate vicinity of each response, we calculated percent impervious surface cover around the nearest major crossroads provided by survey respondents. We used the average percentage impervious surface (Dewitz 2023) at several spatial scales (see below) as an index for habitat degradation. Impervious surface plays a large role in decreasing habitat quality by reducing vegetation cover, increasing temperatures of air and water, and sealing soil with cement, which in turn disrupts nutrient cycling, filtration of pollutants, and recharge (Chithra et al. 2015, Ferreira et al. 2018, Zipperer, Northrop, and Andreu 2020). Further, we previously found that impervious surface coverage was one of the primary environmental factors influencing the presence of wildlife in Tucson (Burnett et al. *in press)* and impervious surface coverage has previously been identified and used as a measurement of "urbanness" (Arnold and Gibbons 1996, McKinney 2008). We used distance to extant riparian corridors as an index of habitat fragmentation because riparian corridors are known to function as wildlife corridors in Tucson (Grubbs and Krausman 2009*b*, Burnett et al. *in press)* and distance to riparian corridors varies across the urban-rural gradient.

We used QGIS v3.12 to map respondent locations, measure Euclidean distance from respondent locations to stream corridors, and sample impervious surface values within buffer zones around respondent locations and within the nearest stream corridors. We calculated the mean percentage of impervious surface within a $100m^2$, $1km^2$, and $2km^2$ circular buffer around each approximate respondent location (i.e., nearest crossroads). We used a GIS layer of washes provided by Pima County to measure Euclidean distance to the nearest wash. This layer shows heavy overlap between washes and roads where waterways move underground, and so we only measured distance to washes or sections of washes that were clearly discernible from satellite imagery. To measure impervious surface within the closest riparian corridor, we measured a 1km stretch upstream and downstream (2km stretch total when possible) from the point at which the wash was closest to the survey location and created a $15m^2$ riparian buffer around the main stream channel within the 2km stretch. We calculated the average percentage impervious surface within the $15m^2$ buffer zone based on 30m-resolution data provided by National Land Cover Database (Dewitz, 2023).

Using these calculated variables, we ran generalized linear regression models to test whether the proportion of conflict at survey locations (i.e. the proportion of people that attempted to deter any species at the crossroads listed) could be explained by percentage impervious surface around the crossroads identified, distance to the nearest riparian corridor, and average percentage impervious surface within the riparian corridor.

RESULTS

We received responses from 394 respondents between the months of March 2022 and November 2022. After removing responses from respondents who did not live within Tucson city limits and/or did not fully complete the survey, we analyzed responses from a total of 360 respondents (Figure 1). Although the survey was distributed to neighborhood associations throughout Tucson, the majority of survey respondents lived in central Tucson. Therefore, our conclusions are limited to residents in central Tucson.

Among our respondents, 23% had lived in their home for less than five years, 16% had lived in their home between five and ten years, 19% had lived in their home between 11 and 20 years, and 42% of respondents had lived in their home for over 20 years. Approximately 23% of respondents had seen a bobcat on their property (n = 82), 79% of respondents had seen a coyote on their property (n = 283), and 72% of respondents had seen a peccary on their property (n = 258). Overall attitudes towards all three species were largely positive (Figure 2). Attitudes toward bobcats were 75% positive (39% extremely positive, 30% mostly positive, 6% somewhat positive), 17% neutral and 8% negative (0.3% extremely negative, 2% mostly negative, 6% somewhat negative). Attitudes toward coyotes were 74% positive (26% extremely positive, 38% mostly positive, and 10% somewhat positive), 18% neutral, and 8% negative (1% extremely negative, 2% mostly negative, 5% somewhat negative). Attitudes toward peccary were 65% positive (18% extremely positive, 34% mostly positive, and 12% somewhat positive), 16% neutral, and 20% negative (2% extremely negative, 6% mostly negative, 12% somewhat negative).

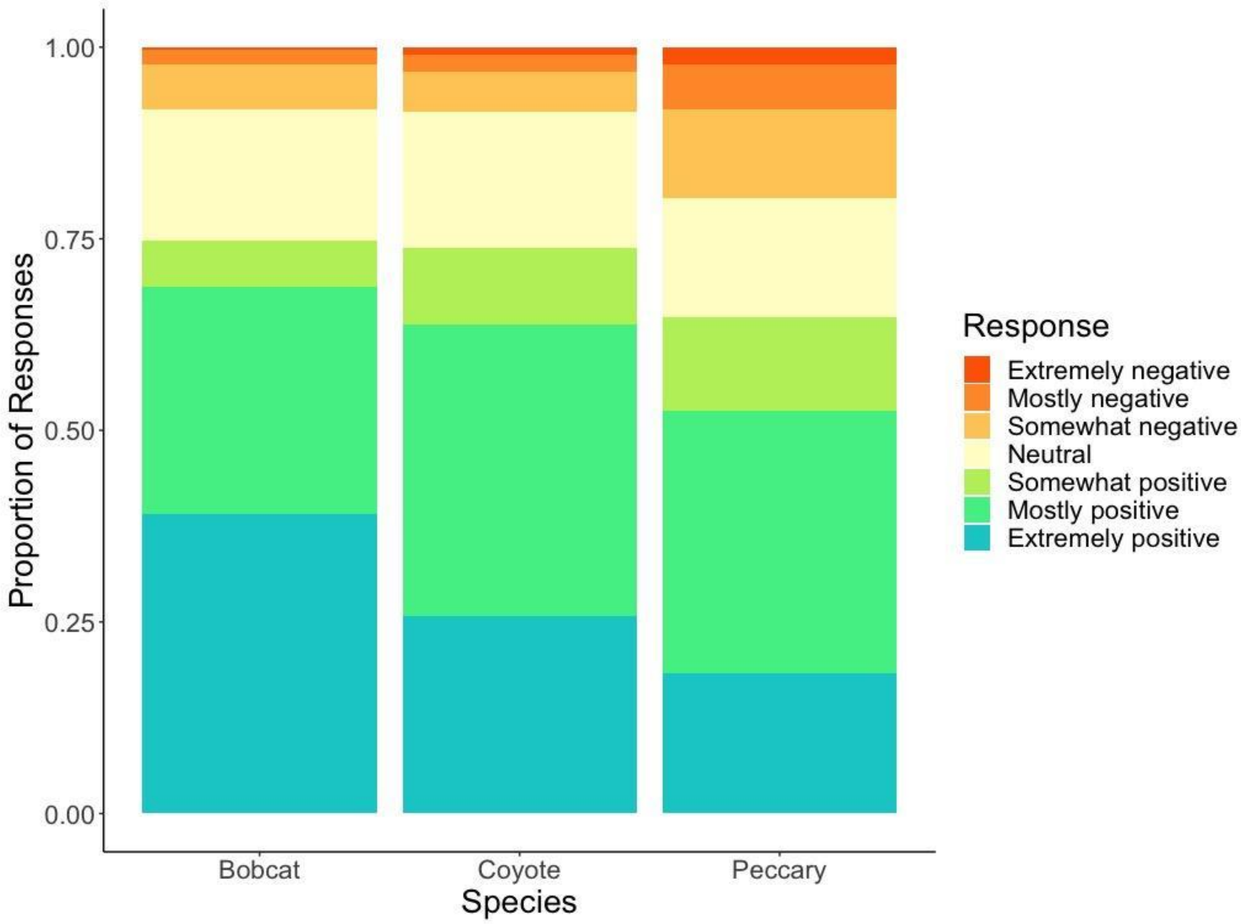

Figure 2: Residents' attitudes toward bobcats, coyotes, and peccaries within the city limits of Tucson, Arizona.

Bobcat observations and deterrence

Bobcats were never seen on a weekly basis (0% of respondents) but were seen on a monthly basis by 29% of respondents. Approximately 17% of respondents observed bobcats annually and 49% of respondents saw bobcats rarely. Bobcats were observed seasonally by 5% of respondents, of which 50% observed bobcats during the dry season and 50% observed bobcats during the monsoon season.

Of the residents that observed bobcats on their property (Figure 3), 11% attempted to deter them (n = 9) and 89% did not (n = 73). Two respondents yelled at bobcats, which left in response 100% of the time. Neither respondent saw a bobcat again. Five respondents installed physical barriers, and 80% of those respondents saw a bobcat again. One respondent secured resources but had since seen a bobcat on their property again. One respondent chose “other” and noted that they had a dog present, which may deter the bobcat, however the bobcat has been seen on the property again. All respondents who attempted to deter bobcats cited doing so at least in part to protect their pets (100%), and four respondents (44%) also cited concern for people.

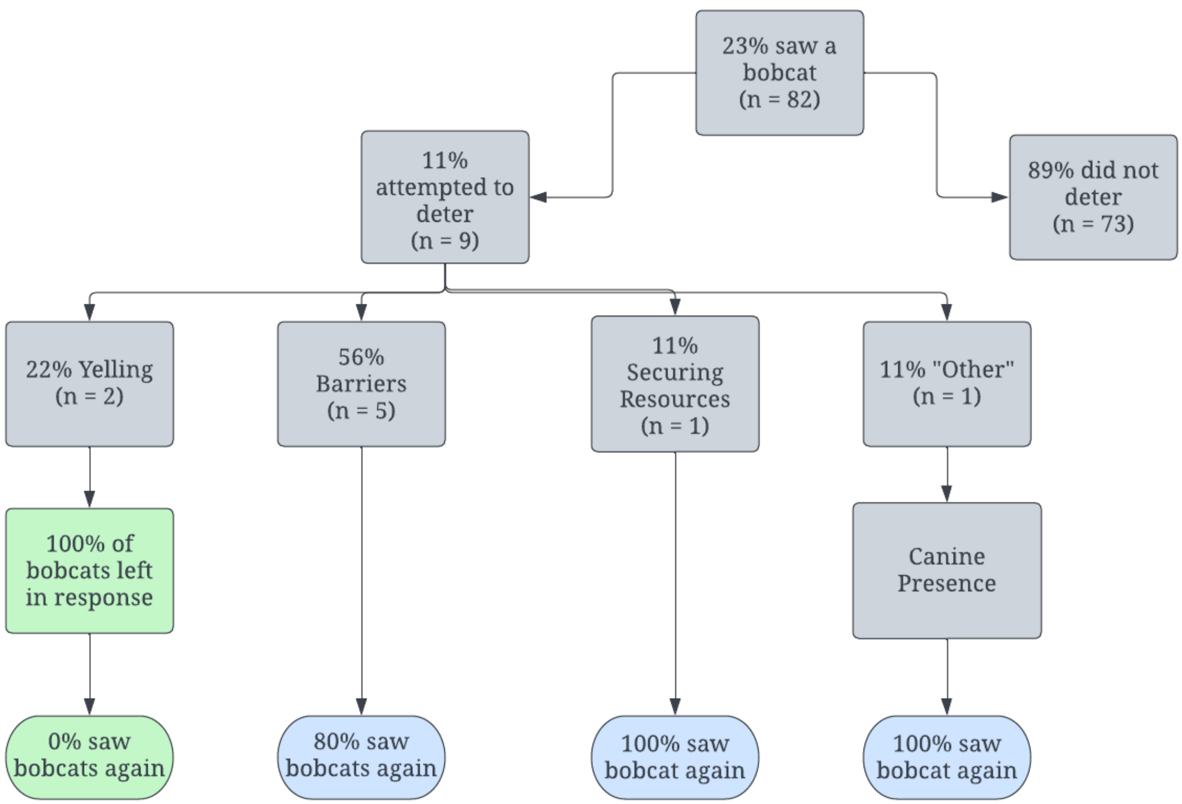

Figure 3: Flow chart showing how respondents responded to questions relating to bobcats in Tucson, Arizona. Green boxes indicate that the strategy was largely successful, and blue boxes indicate the strategy was largely unsuccessful.

Coyote observations and deterrence

The frequency with which residents observed coyotes varied, with 8% of respondents observing coyotes at least once weekly, 33% of respondents observing coyotes once monthly, 18% observing coyotes annually, and 34% that had seen coyotes more rarely. Some respondents saw coyotes seasonally (7%), of which 45% (n = 9) saw coyotes during the dry season and 45% saw coyotes during the monsoon season. Only 10% of respondents (n = 2) that observed coyotes seasonally noted them during the winter season.

Of the residents who observed coyotes on their property (Figure 4), 17% attempted to deter them (n = 46) and 83% did not (n = 232). Five responses were removed from this analysis because responses were unclear (e.g., respondents noted that they attempted to deter but when prompted to choose which strategy they used to deter the coyotes, they wrote “none” or left the box blank). Ten respondents (22%) yelled at coyotes, which left in response 80% of the time. Twenty percent of respondents had not seen a coyote on their property since. Four respondents (9%) waved objects, and 75% of those respondents did not see a coyote again. Unfortunately, there was a logic error in the survey instrument that prevented these respondents from receiving the question regarding whether coyotes left immediately, and so those data were not available. Of the 25 respondents who installed physical barriers (54%), 64% of those respondents did not see a coyote again. Three respondents attempted to secure resources, and two respondents (67%) had not seen a coyote since. Four respondents chose “other.” One respondent used a laser pointer and had seen a coyote since. The three other respondents used a partial barrier (e.g., cactus fence, hedge, or yard only partially fenced), and 67% had not seen a coyote since. Reasons for deterring coyotes were primarily due to fear for their pets (65% of respondents), people, especially children (11%), and livestock (e.g., chickens; 9%). Other reasons included fear of rabies (2%), protecting feral cats (7%) and wildlife (4%), appeasing neighbors (4%), and general fear of coyotes (4%).

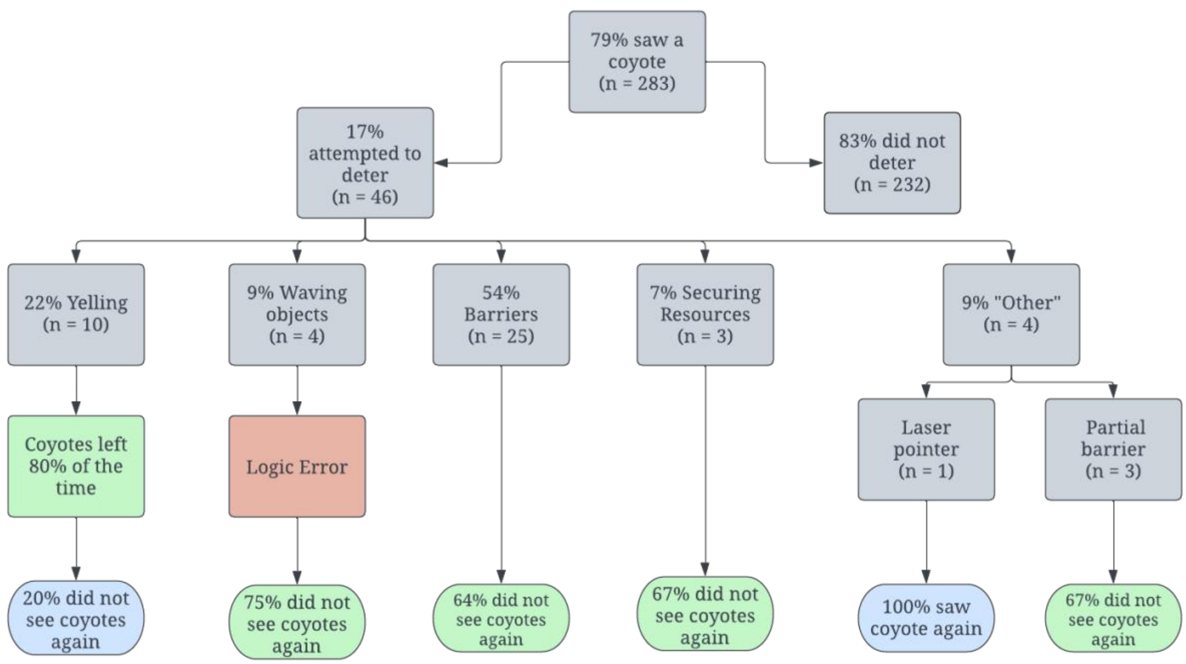

Figure 4: Flow chart showing how respondents responded to questions relating to coyotes in Tucson, Arizona. Green boxes indicate that the strategy was largely successful, blue boxes indicate the strategy was largely unsuccessful. An error in the logical flow of our survey instrument prevented respondents who selected “waving objects” from receiving the question about whether coyotes immediately left the property.

Peccary observations and deterrence

Peccaries were observed regularly (at least once a week) by 7% of respondents, monthly by 31% of respondents, and were observed annually or rarely by 15% and 41% of respondents, respectively. Respondents that observed peccaries seasonally (7%) saw them primarily during the monsoon season (59%) or the winter season (35%). Only one respondent that observed peccaries seasonally (6%) observed them during the dry season.

After removing three responses in which the deterrent strategy was unclear, we found that, of respondents that observed peccaries on their property, 21% attempted to deter them (n = 53) and 79% did not (n = 202; Figure 5). Five respondents yelled at intruding peccaries, and peccaries left in response 60% of the time. 60% of respondents had not seen a peccary on their property again. Four respondents (8%) waved objects, and peccaries left in response 75% of the time. Fifty percent of these respondents had not seen a peccary since. One respondent sprayed water, and the peccaries left the property in response; however, peccaries had been seen on the property since. Similarly, one respondent threw gravel, and the peccaries left the property in response, however, peccaries had been seen on the property since. Physical barriers were installed by 55% (n = 29) of respondents, and 52% of those respondents did not see peccaries again. Nine respondents (17%) attempted to secure resources, and all respondents had since seen a peccary. One respondent translocated the peccaries and had not seen a peccary on their property since. Three respondents chose “other.” Two respondents used a combination of methods. One person secured resources and sprayed unpleasant chemicals; peccaries were seen again on this property. The second person used loud noise, a physical barrier, and placed ammonia-soaked rags; peccaries were not seen there again. The remaining “other” response used a partial barrier (e.g., yard only partially fenced) and this respondent did not see peccaries on their property again. Reasons for deterring peccaries included concern for landscaping and property destruction (53%), concern for people (21%), concern for pets (17%) and concern that the animals would move into their crawlspace or under their decks (8%). Other concerns include concern about peccaries getting into garbage (9%), rabies (2%) and concern for the peccaries themselves (2%).

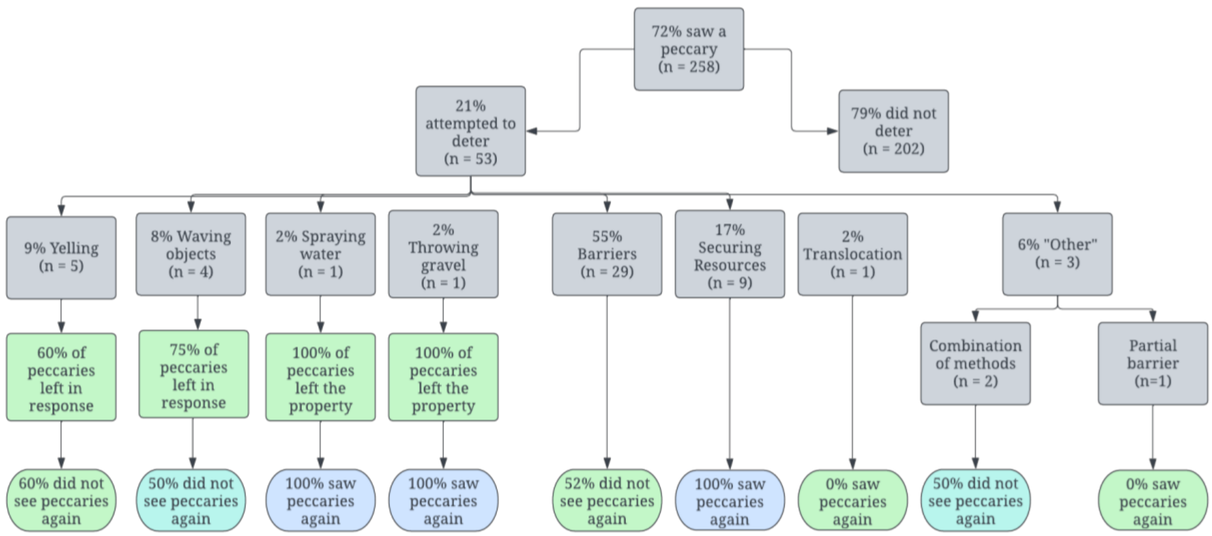

Figure 5: Flow chart showing how respondents responded to questions relating to peccaries in Tucson, Arizona. Green boxes indicate that the strategy was largely successful, blue boxes indicate the strategy was largely unsuccessful, and turquoise boxes indicate intermediate success.

Conflict prediction models

We found no effects of percentage impervious surface (GLM; df = 114; $p_{100m}{}^2 = 0.44$, $p_{1km}{}^2 = 0.44$, $p_{2km}{}^2 = 0.21$), distance to riparian corridor (GLM; df = 114, p = 0.75), or percentage impervious surface within the riparian corridor (GLM; df = 114, $p_{15m}{}^2 = 0.61$) on the proportion of conflict observed at each location (n = 120 locations).

DISCUSSION

Residents in central Tucson frequently observed bobcats, coyotes, and peccaries on their properties and held an overall positive view of the three species despite a high proportion of complaints about them compared to other wildlife species. Only a small percentage of people (~10-20%) attempted to deter each species. We observed more positive views toward coyotes in central Tucson (~74% positive, 8% negative) than observed in previous surveys conducted in Tucson (Lawrence and Krausman 2011) and compared to several other urbanized regions (Draheim et al. 2013, Elliot et al. 2016, Nardi et al. 2020, Puri et al. 2024). A similar study conducted in two of central Tucson's neighborhoods (El Encanto and Colonia Solona) in 2007 found that approximately 30% of residents felt that coyotes were a nuisance (Lawrence and Krausman 2011). Attitudes toward coyotes in Washington D.C. (Draheim et al. 2013) and North Carolina (Lute et al. 2020) were largely neutral, whereas attitudes toward coyotes in Madison, WI were split roughly evenly among positive, negative, and neutral (Nardi et al. 2020). Additionally, resident attitudes toward coyotes and bobcats in Atlanta, GA were primarily negative (Puri et al. 2024). In contrast, resident attitudes toward bobcats were largely positive in central New Mexico, in which the majority of surveyed residents liked the idea of nearby bobcats (Harrison 1998). Attitudes toward peccaries in towns outside of Iguaçu National Park in Brazil were similarly largely positive (Cascelli de Azevedo and Conforti 2008). Proximity to natural areas like national parks may be associated with more positive views of wildlife for urban residents, possibly due to people with more positive views of wildlife moving to areas that are within close proximity to natural areas or due to increased opportunities for neutral or positive interactions with wildlife (McCleery et al. 2014).

We found that attempts to deter bobcats and coyotes were driven largely by concern for the safety of pets and people in central Tucson and attempts to deter peccaries were driven largely by concern for property destruction. Residents in other regions reported similar reasons for concern; negative attitudes toward bobcats and coyotes were driven primarily by concern for the safety of pets, children, as well as their own personal safety (Harrison 1998, Elliot et al. 2016, Lute et al. 2020). A large percentage of surveyed residents in North Carolina were additionally concerned about the spread of rabies by coyotes (Lute et al. 2020). Negative attitudes toward peccaries in Brazil were driven largely by concern for agricultural crops (Cascelli de Azevedo and Conforti 2008). A follow-up study investigating whether bobcats in Tucson consume pets or other anthropogenic resources found that these anthropogenic sources are very unlikely to be a main dietary item (see Appendix S4), highlighting possible discrepancies between perceived human-wildlife conflict and actual conflict.

Deterrent techniques varied in effectiveness depending on species. Aversive conditioning worked well in the short-term for all three species but showed mixed long-term efficacy. Bobcats that were shouted at did not return to the property, indicating long-term efficacy; however, our sample size was limited to only two respondents. Aversive conditioning showed limited long-term efficacy for coyotes or peccaries. Although a number of management plans recommend low-intensity hazing (e.g., yelling, spraying water), few studies have compared between the efficacy of different low-intensity techniques (Lajeunesse et al. 2023). However, studies evaluating the overall efficacy of low-intensity aversive conditioning have found results similar to ours. A majority of coyotes encountered by residents in Denver, Colorado (Bonnell and Breck 2017) and Calgary, Alberta (Lajeunesse et al. 2023) retreated in response to low intensity hazing such as yelling. However, yelling at coyotes was more effective for residents in central Tucson than in North Carolina (Lute et al. 2020). Although to our knowledge, no studies have evaluated the effects of aversive conditioning on bobcats or peccaries, aversive conditioning may be more effective long-term for bobcats due to their general avoidance of humans (Hubbard et al. 2022, Reilly et al. 2022, Schmidt et al. 2023). Species that are less habituated to humans may view humans as a threat, such that strategies of yelling and waving objects are sufficient for deterrence. Other habituated species, such as coyote and peccary, may no longer view humans as a threat, and thus may require higher intensity strategies for deterrence (Blumstein 2016, Honda et al. 2019, Lajeunesse et al. 2023).

More preventative strategies, particularly physical barriers, worked more than 50% of the time for long-term deterrence of coyotes and javelina, but was largely ineffective for long-term deterrence of bobcats, which are good climbers. Fences, particularly those over 2m, can be an effective obstacle in preventing the use of urban backyards by coyotes (Murray and St. Clair 2017). In Edmonton, Canada, coyotes that used backyards primarily selected for backyards without fences (Murray and St. Clair 2017). Interestingly, we found that securing resources was relatively ineffective at deterring bobcats and peccaries, despite evidence demonstrating that removing attractants can reduce visitation by wildlife (Elliot et al. 2016, Murray and St. Claire 2017, Mazué et al. 2023). However, we obtained only very small sample sizes of residents that attempted this strategy overall. Residents may not have adequately secured resources to prevent exploitation by wildlife; however, it is likely that wildlife were attracted to the property due to landscaping or infrastructure rather than human-derived food resources. Peccaries and coyotes use landscaping plants for cover and food (Ticer et al. 1998, Grubbs and Krausman 2009*a*, Murray and St. Claire 2017), and bobcats are frequently observed resting in backyards and using flat roofs of suburban houses for raising offspring in Tucson (*C. Mollohan, personal* *communication).* Securing potential food resources therefore may not adequately deter wildlife in Tucson from accessing residential properties if the property still provides other resources. Further, wildlife may be attracted to the scent of some resources despite the resources being inaccessible to the animal. For residents seeking to deter wildlife, a combination of techniques may be most effective for targeting multiple species and avoiding habituation, and/or increasing long-term efficacy (Blackwell et al. 2016).

Given the low incidence of deterrence attempts and overall positive attitudes towards these species, human-wildlife conflict appears to be relatively low in central Tucson. We did not find a relationship between the proportion of respondents that attempted to deter the focal species at survey locations and impervious surface coverage, distance to riparian corridor, or impervious surface coverage within the riparian corridor. Our results suggest that low resource availability resulting from habitat degradation is not the main driver of conflict in central Tucson. This is supported by the lack of seasonality in observations of animals. Animals were most frequently seen either semi regularly (i.e., monthly) or rarely, with relatively few residents seeing animals weekly or seasonally. If conflict were driven by low resource availability, we would expect residents to observe animals on their properties either regularly, if resource availability were low year-round, or seasonally when natural resources are scarce (e.g., Mukeka et al. 2019, Thompson et al. 2021). Conflict may instead be driven by individuals that are sick (Baker 2007, Gehrt et al. 2009, Murray et al. 2015, Coogan and Raubenheimer 2016) or individuals that have learned nuisance behaviors (Baker 2007, Breck, Poessel, and Bonnell 2017, Swan et al. 2017). Further study on the wildlife populations within and around Tucson to determine which wildlife groups or individuals exhibit conflict-prone behaviors may provide insight into the causes of conflict.

CONCLUSIONS

Our study shows that aversive conditioning can be an effective solution for urban residents to resolve conflict of immediate concern with our three focal species without the use of translocation or lethal methods. Further, deterrent strategies can be effective in the long-term when species ecology is considered. For example, while household fences may be ineffective against climbing species such as bobcats, they can be effective against non-arboreal species such as coyote and peccary (Murray and St. Clair 2017). Similarly, aversive conditioning was effective at deterring bobcats, which are generally sensitive to human presence (Tigas et al. 2002, Riley et al. 2003, Dunagan et al. 2019). Our results thus provide residents in central Tucson with a more refined toolbox to pull from when encountering wildlife on their property that allows them to safely and effectively reduce HWC using nonlethal methods. We also found that the proportion of conflict at survey locations was not related to our indices of habitat degradation and fragmentation, suggesting that there may be other drivers of conflict in central Tucson.

Acknowledgements

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Supplementary Material

Appendix E: Survey instrument evaluating public attitudes and responses toward human-wildlife conflict. Survey was distributed in 2022 in the City of Tucson, AZ.

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APPENDIX E: Survey instrument evaluating public attitudes and responses toward human-wildlife conflict. Survey was distributed in 2022 in the City of Tucson, AZ.

A Survey of Deterrent Efficacy in Tucson, AZ

Tucson Urban Wildlife Survey
Principal Investigator: Alexandra Burnett

You are being asked to participate in a research study.
Your participation in this research study is voluntary and you do not have to participate. This document contains important information about this study and what to expect if you decide to participate. Please consider the information carefully. Feel free to ask questions before making your decision whether or not to participate.

Why is this study being conducted?
The purpose of this survey is to evaluate the short- and long-term effectiveness of deterrent techniques at reducing interactions between people and wildlife. The information gathered from this project can inform management measures intended to minimize human-wildlife conflict in Tucson, Arizona.

What will be required of me to participate in this study?
This questionnaire is an anonymous, online survey that can be taken in the privacy of your own home. No personal information is collected for this study. Your answers are part of a University of Arizona Ph.D. project aimed at preventing or resolving conflict between residents and urban wildlife. The survey uses a mix of open and closed questions to assess how wildlife respond to deterrent strategies used by residents and whether such efforts are effective long-term. You are not obligated to participate in this study and you may exit the survey at any time.

What risks or benefits can I expect from participating?
There are no expected risks to you as a result of participating in this study. You will not benefit directly from participating in this study, however, you will be helping to identify management techniques that minimize human-wildlife conflict in urban areas. Collective information from residents will be used to help develop future strategies relating to urban wildlife management.

How much time will it take to complete the survey?
This survey takes approximately 5-20 minutes to complete.

Will my study-related information be kept confidential?
The information that you give in the study will be anonymous and we do not ask for any personal information. The PI will be the only person with access to survey responses, which will be stored digitally on a secure server managed by the University of Arizona.

For questions, concerns, or complaints about the study you may contact the co-Principal

Investigator, Alexandra Burnett, Phone contact: (+1) 248 561 5892, Email: aburnett93@email.arizona.edu

For questions about your rights as a participant in this study or to discuss other study-related concerns or complaints with someone who is not part of the research team, you may contact the Human Subjects Protection Program at 520-626-6721 or online at http://rgw.arizona.edu/compliance/human-subjects-protection-program.

An Institutional Review Board responsible for human subject's research at The University of Arizona reviewed this research project and found it to be acceptable, according to applicable state and federal regulations and University policies designed to protect the rights and welfare of participants in research.

Authorization from participant
By proceeding with the survey, you indicate your agreement to participate in the study. You may exit the survey at any time if you no longer wish to participate in the study.

End of Block: Block 1

Start of Block: Default Question Block

Q2 Are you a resident of the Tucson Metropolitan Area?

- ○ Yes (1)
- ○ No (2)
- ○ Unsure (3)

Q3 Please describe the nearest major crossroads of your residence (this information remains anonymous and will not be published).

____

Q4 Approximately how many years have you lived at your current residence?

- ○ <5 years (1)
- ○ 5–10 (2)
- ○ 11–20 (3)
- ○ >20 years (4)

Q5 Which of the following wildlife have you detected at your residence (check all that apply)?

- ▢ Coyote (1)
- ▢ Javelina (2)
- ▢ Bobcat (3)
- ▢ None (4)

Skip To: End of Survey If Q5 = 4

Display This Question:

If Q5 = 1

Q25 Approximately how often do you see coyote on your property?

- ○ Regularly (at least once a week) (4)
- ○ Occasionally (once a month) (5)
- ○ Annually (once a year) (6)
- ○ Rarely (have only seen this species on my property a handful of times or less) (7)
- ○ Seasonally (only during certain seasons) (8)

Display This Question:

If Q25 = 8

Q33 Which season are you most likely to see coyote?

- ◯ Dry (Feb–June) (1)
- ◯ Monsoon (July–October) (2)
- ◯ Winter Rains (November–January) (3)

Display This Question:

If Q5 = 1

Q6 Have you ever attempted to deter the coyote from your property (hazing, fences, barriers, etc.)?

- ◯ Yes (1)
- ◯ No (2)

Display This Question:

If Q6 = 1

Q26 What were your reasons for deterring the coyote?

____

Display This Question:

If Q6 = 1

Q7 Which strategy did you primarily use to deter the coyote(s)?

- ◯ Yelling or making loud noises (1)
- ◯ Waving your hands or objects in the air (9)
- ◯ Spraying water (2)
- ◯ Throwing gravel (3)
- ◯ Commercial repellent (e.g., Mace) (11)
- ◯ Securing or removing attractants (garbage, food, small pets, pet food, etc.) (7)
- ◯ Physical barriers (6)
- ◯ Applying a biological deterrent (i.e. predator urine) (4)
- ◯ Removal/ translocation (5)
- ◯ Other (8) ____

Display This Question:

If Q7 = 3

Or Q7 = 2

Or Q7 = 1

Q8 Did the coyote(s) leave the property?

- ◯ Yes (1)
- ◯ No (2)
- ◯ Unsure (3)

Display This Question:

If Q5 = 2

Q27 Approximately how often do you see javelina on your property?

- Regularly (at least once a week) (4)
- Occasionally (once a month) (5)
- Annually (once a year) (6)
- Rarely (have only seen this species on my property a handful of times or less) (7)
- Seasonally (only during certain seasons) (8)

Display This Question:

If Q27 = 8

Q34 Which season are you most likely to see javelina?

- Dry (Feb–June) (1)
- Monsoon (July–October) (2)
- Winter Rains (November–January) (3)

Display This Question:

If Q5 = 2

Q9 Have you ever attempted to deter the javelina from your property (hazing, fences, barriers, etc.)?

- Yes (1)
- No (2)

Display This Question:

If Q9 = 1

Q29 What were your reasons for deterring the javelina?

____

Display This Question:

If Q9 = 1

Q10 Which strategy did you primarily use to deter javelina?

- ◯ Yelling or making loud noises (1)
- ◯ Waving your hands or objects in the air (9)
- ◯ Spraying water (2)
- ◯ Throwing gravel (3)
- ◯ Commercial repellent (e.g., Mace) (11)
- ◯ Securing or removing attractants (garbage, food, potted or landscaped plants, pet food, birdseed, etc.) (7)
- ◯ Physical barrier (6)
- ◯ Applying a biological deterrent (i.e. predator urine) (4)
- ◯ Removal/ translocation (5)
- ◯ Other (8) ____

Display This Question:

If Q10 = 1

Or Q10 = 2

Or Q10 = 3

Or Q10 = 9

Or Q10 = 11

Q11 Did the javelina leave the property?

- ◯ Yes (1)
- ◯ No (2)
- ◯ Unsure (3)

Display This Question:

If Q5 = 3

Q28 Approximately how often do you see bobcat on your property?

- ◯ Regularly (at least once a week) (4)
- ◯ Occasionally (once a month) (5)
- ◯ Annually (once a year) (6)
- ◯ Rarely (have only seen this species on my property a handful of times or less) (7)
- ◯ Seasonally (only during certain seasons) (8)

Display This Question:

If Q28 = 8

Q35 Which season are you most likely to see bobcat?

- ◯ Dry (Feb–June) (1)
- ◯ Monsoon (July–October) (2)
- ◯ Winter Rains (November–January) (3)

Display This Question:

If Q5 = 3

Q12 Have you ever attempted to deter the bobcat(s) from your property (hazing, fences, barriers, etc.)?

- ◯ Yes (1)
- ◯ No (2)

Display This Question:

If Q12 = 1

Q30 What were your reasons for deterring the bobcat(s)?

____

Display This Question:

If Q12 = 1

Q13 Which strategy did you primarily use to deter the bobcat(s)?

- ○ Yelling or making loud noises (1)
- ○ Waiving your hands or objects in the air (9)
- ○ Spraying water (2)
- ○ Throwing gravel (3)
- ○ Commercial repellent (e.g., Mace) (10)
- ○ Securing or removing attractants (pet food, birdseed, etc.) (7)
- ○ Physical barriers (6)
- ○ Applied a biological deterrent (i.e. predator urine) (4)
- ○ Removal/ translocation (5)
- ○ Other (8) ____

Display This Question:

If Q13 = 1

Or Q13 = 2

Or Q13 = 3

Q14 Did the bobcat(s) leave the property?

- ○ Yes (1)
- ○ No (2)
- ○ Unsure (3)

Display This Question:

If Q6 = 1

Or Q9 = 1

Or Q12 = 1

Q15 Have any of the following species been seen on the property since (check all that apply)?

- ▢ Coyote (1)
- ▢ Javelina (2)
- ▢ Bobcat (3)
- ▢ None (4)

Q24 What is your attitude toward coyotes?

- ○ Extremely positive (1)
- ○ Mostly positive (2)
- ○ Somewhat positive (3)
- ○ Neutral (4)
- ○ Somewhat negative (5)
- ○ Mostly negative (6)
- ○ Extremely negative (7)

Q31 What is your attitude toward javelina?

- ◯ Extremely positive (1)
- ◯ Mostly positive (2)
- ◯ Somewhat positive (3)
- ◯ Neutral (4)
- ◯ Somewhat negative (5)
- ◯ Mostly negative (6)
- ◯ Extremely negative (7)

Q32 What is your attitude toward bobcats?

- ◯ Extremely positive (1)
- ◯ Mostly positive (2)
- ◯ Somewhat positive (3)
- ◯ Neutral (4)
- ◯ Somewhat negative (5)
- ◯ Mostly negative (6)
- ◯ Extremely negative (7)

Q16 If you have any further comments or details you would like to share, please use the space below:

____

APPENDIX F: Dietary analysis of bobcats in Southeastern Arizona comparing rural and suburban bobcats.

METHODS

We collaborated with Bobcats in Tucson organization and Arizona Game and Fish Department (AZGFD) to obtain hair samples collected from collared bobcats as well as hair and feather samples collected from common prey items. Samples from prey items were collected from hunting check stations, during rodent trapping for other projects, as well as opportunistically from roadkill. All hair and feather samples were weighed, rinsed with a 2:1 chloroform-methanol solution and deionized water, and processed in the Environmental Stable Isotope Facility on the University of Arizona's campus.

We used R (v.4.4.1; R Core Team 2022) and RStudio to visualize and compare isotopic signatures and conduct a Bayesian mixing model to determine the dietary proportions for suburban bobcats. We used package *MixSIAR* (Stock et al., 2018) to create isospace plots and run Bayesian models to estimate the proportion of each source consumed by suburban bobcats. We used discrimination factors derived by Parng et al. (2014) for bobcats. The sources used included lagomorphs (n = 16; cottontails (*Sylvilagus audubonii)* and black-tailed jackrabbits (*Lepus californicus*)), rodents (n = 226; *Sigmodon, Peromyscus, Dipodomys,* and *Chaetodipus sp.*), non-domesticated birds (n = 17; doves (*Zenaida sp.*) and Gambel's quail *(Callipepla gambelii*), domestic chickens (n = 3; *Gallus domesticus*), domestic cats (n = 9; *Felis catus*), and domestic dogs (n = 12; *Canis familiaris*). We chose these sources based on literature stating that lagomorphs and rodents were the primary prey item of bobcats in the Southwest (Leopold and Krausman 1986, Delibes and Hiraldo 1987) as well as eyewitness accounts in which bobcats were observed feeding on doves. We additionally included domestic chickens, dogs, and cats because we were specifically interested in quantifying dietary proportions of these items due to resident concerns. We ran our model using uninformative priors and used Geweke and Gelman-Rubin diagnostic statistics to confirm model convergence (Geweke 1992, Stock and Semmens 2016).

RESULTS

Isotopic signatures for rural (n = 19) and suburban bobcats (n = 38) in Southeast Arizona were significantly different when comparing Carbon 13 (Wilcox test; W = 197.5, $p < 0.01$), Nitrogen 15 (Wilcox test; W = 612.5, $p < 0.001$), and the ratio of carbon to nitrogen (Wilcox test; W = 516, $p < 0.01$). Suburban bobcats consumed significantly higher protein ($d^{15}N = 10.58 \pm 0.23$‰) and consumed prey sources with lower C4 signatures ($d^{13}C = -17.47 \pm 0.16$‰) compared to rural bobcats (9.32 ± 0.23‰ and -16.11 ± 0.33‰, respectively; Figure S1).

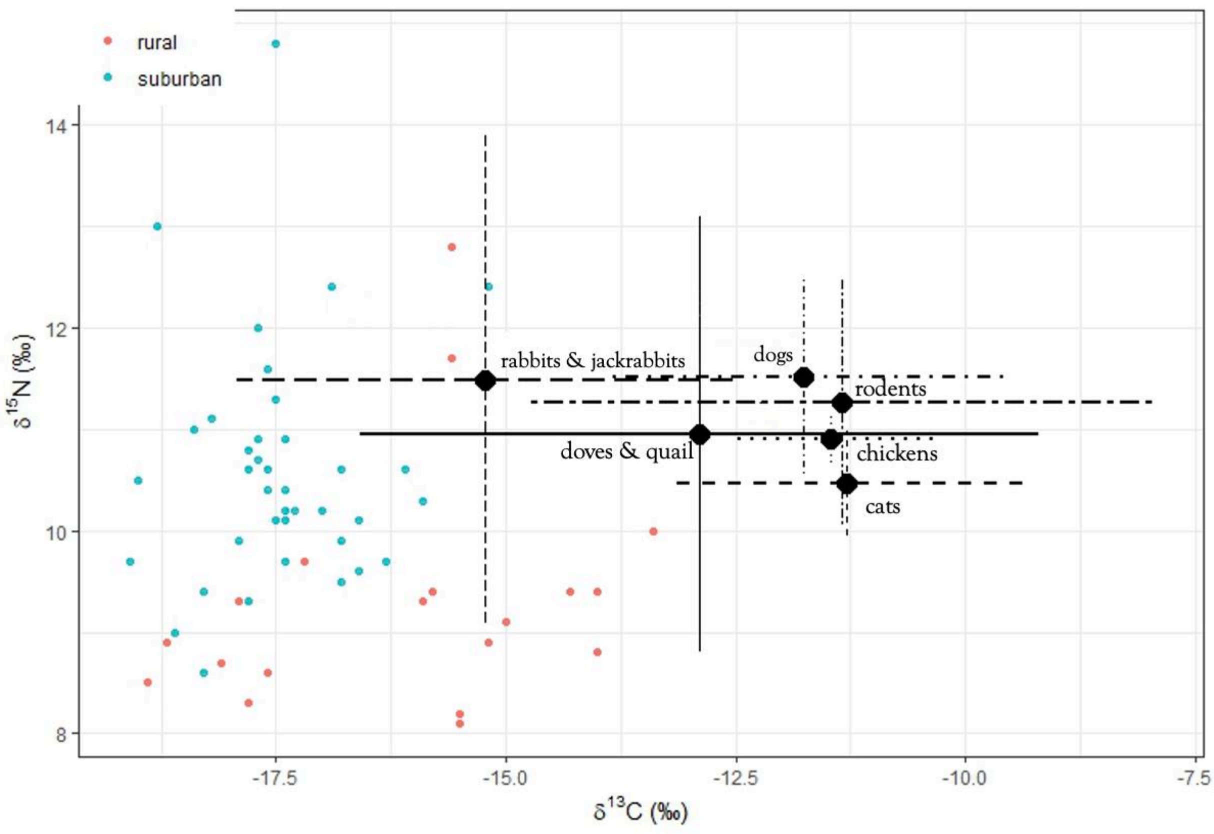

Figure S1: Isospace plot showing Tucson bobcat signatures (blue points), Altar Valley bobcat signatures (red points), and signatures of possible prey items included in the model (black points). Signatures of prey items are corrected with a trophic enrichment factor.

Our Bayesian mixing model indicated that the diet of bobcats in Tucson consisted of ~9.6 ± 9.1% non-domesticated avian sources (e.g., doves, quail) and 86 ± 9% lagomorphs (Figure S2). Dogs, cats, and chicken were each estimated to comprise less than 2% of their diet (Table S1). Our Bayesian mixing model for bobcats in Altar Valley yielded a bimodal result, suggesting that rural bobcats may most likely be consuming either very high proportions of lagomorphs and very low proportions of avian prey sources or approximately 25% lagomorph and 70% avian sources, with the remainder made up of rodents, cats, or chicken (Figure S2).

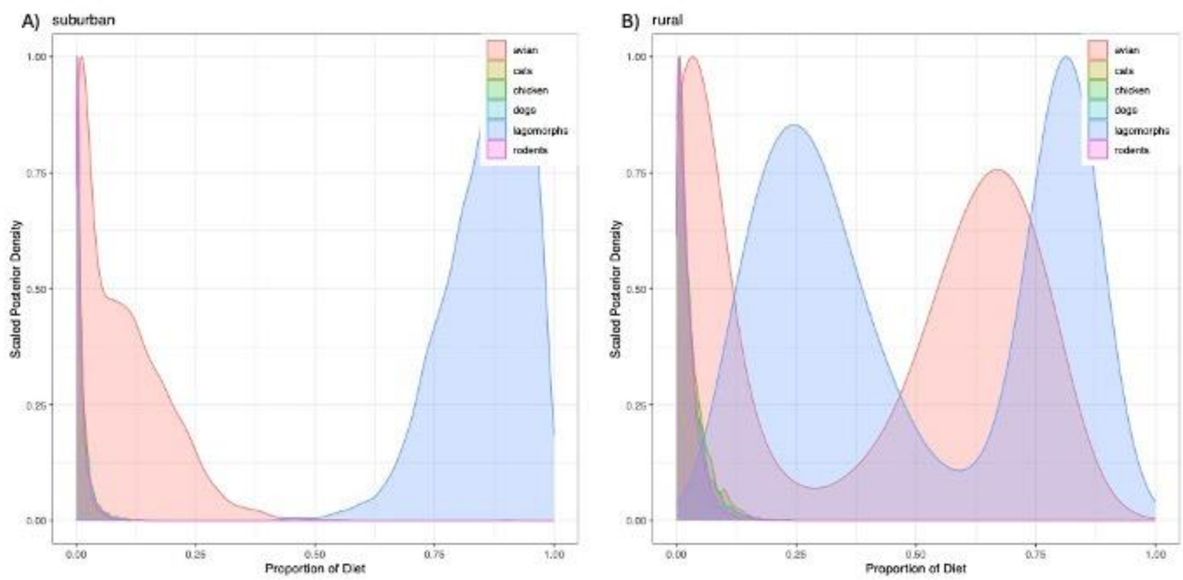

Figure S2: Posterior density plots derived from Markov chain Monte Carlo mixing models showing the predicted proportion of A) suburban bobcat diet (n = 38 bobcats) for each included prey item and B) rural bobcat diet (n = 19). The diet of suburban bobcats in Tucson, AZ is predicted to be comprised primarily of lagomorphs and non-domesticated birds (specifically doves and quail). Mixing models for rural bobcats yielded a bimodal result suggesting two alternative possibilities for the diet composition of bobcats in Altar Valley, AZ.

Table S1: Mean, standard deviation (SD), and credible intervals for each included prey source predicted by Bayesian mixing models for suburban bobcats sampled in Tucson, AZ (n = 38) and rural bobcats sampled in Altar Valley, AZ (n = 19).

Region	**Prey source**	**Mean**	**SD**	**2.5%**	**5%**	**25%**	**50%**	**75%**	**95%**	**97.5%**
Rural	Avian	0.394	0.306	0.003	0.005	0.045	0.514	0.674	0.791	0.816
Rural	Cats	0.03	0.032	0.001	0.001	0.008	0.019	0.043	0.099	0.117
Rural	Chickens	0.027	0.03	0.001	0.001	0.007	0.018	0.038	0.088	0.110
Rural	Dogs	0.024	0.023	0.001	0.001	0.007	0.017	0.033	0.069	0.087
Rural	Lagomorphs	0.5	0.278	0.106	0.128	0.243	0.412	0.794	0.874	0.894
Rural	Rodents	0.025	0.026	0.001	0.001	0.007	0.017	0.033	0.076	0.096
Suburban	Avian	0.096	0.091	0.001	0.001	0.016	0.076	0.152	0.264	0.306
Suburban	Cats	0.012	0.015	0	0	0.002	0.006	0.016	0.042	0.055
Suburban	Chickens	0.012	0.016	0	0	0.002	0.006	0.016	0.044	0.059
Suburban	Dogs	0.012	0.016	0	0	0.002	0.006	0.015	0.043	0.059
Suburban	Lagomorphs	0.857	0.09	0.655	0.697	0.804	0.871	0.926	0.973	0.98
Suburban	Rodents	0.011	0.015	0	0	0.002	0.006	0.014	0.04	0.055

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