THESIS

THE EFFECTS OF URBANIZATION AND ROAD DEVELOPMENT ON CARNIVORES IN

SOUTHERN CALIFORNIA

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ABSTRACT

THE EFFECTS OF URBANIZATION AND ROAD DEVELOPMENT ON CARNIVORES IN

SOUTHERN CALIFORNIA

Habitat destruction and degradation is a serious threat to biodiversity, and urbanization

and road development are driving factors in habitat loss. As the human footprint continues to

expand, the natural landscape becomes increasingly fragmented and human development can

create barriers to wildlife movement and gene flow. Carnivores in particular are sensitive to

fragmentation and may be used as model animals for understanding how roads and urbanization

can fragment wildlife populations. In this thesis, I investigate the effects of urbanization and

road development on carnivores in southern California. Since southern California is one of the

most populous areas of the country, coupled with high biodiversity, it is a unique area to study

the effects of road development and urbanization on carnivores.

In the first chapter, I estimate the density of bobcats in a coastal reserve isolated by

urbanization using mark-recapture and mark-resight techniques with camera trap data. The use

of camera trap data to estimate carnivore abundance is increasingly common, and to date many

such studies have utilized a mark-recapture framework and focused on carnivores with unique

pelage patterns. The recent improvement of mark-resight estimators, however, provides an

opportunity to estimate the abundance of carnivores without unique pelage patterns. We utilized

both the mark-recapture and mark-resight frameworks to estimate bobcat population sizes in a

geographically isolated urban reserve in southern California. Due to their sensitivities to urban

fragmentation, bobcats have been a focal species in several studies throughout southern

California, yet few population estimates exist for this region. Since bobcats are individually

identifiable, and a subset of the study population was physically marked with GPS telemetry collars, we were able to compare the utility of both the mark-recapture and mark-resight frameworks for carnivore population estimation with camera trap data. We deployed a sampling grid of 30 cameras throughout the study area and recorded 109 bobcat photos during 4,669 camera nights from July 2006 through January 2007. Density point estimates were reasonably consistent with prior studies and ranged from 0.40 to 0.55 bobcats per km^2 depending upon the estimator used, but the confidence intervals for all estimates overlapped suggesting that they were not significantly different. Percent confidence interval length ranged from 150% to 180% indicating a low amount of precision for all of our estimates. We conclude that mark-resight estimators performed comparably to the mark-recapture estimators and show promise for use with camera trap data to estimate carnivore population sizes. The low precision for both our mark-recapture and mark-resight estimators, however, highlights the sensitivity of both frameworks to small datasets typical of large carnivore studies. In future studies, it will be important to develop techniques to increase capture probabilities of target species to maximize the utility of camera traps for estimating population sizes.

In the second chapter, I evaluate the effects of a road expansion and mitigation project on underpass usage of three target species: bobcat, coyote and mule deer. Roads can negatively impact wildlife, particularly large mammals. In response, transportation agencies have implemented mitigation measures like the installation of wildlife fencing and wildlife crossing structures. The evaluation of these mitigation measures is crucial to determine the success of reducing road impacts. Herein, we evaluate a road expansion and mitigation project completed by the California Department of Transportation along State Route 71 (CA-71) through the Chino Hills southeast of Los Angeles. We designed a remote camera survey to study how the widening

of CA-71 and implementation of mitigation measures affected large mammal movement and underpass use. Based on camera detections, bobcat underpass use was higher in the construction and mitigation zone after construction than before, but there was no difference in use of underpasses in the impact compared to the control zone in either time period. Underpass use by coyotes was higher in the control zone than in the impact zone, but there was no difference in use between the before and after periods. Small numbers of mule deer detections at few underpasses precluded a comparison between the control and impact zones, but a comparison of before and after periods revealed that mule deer underpass use was slightly higher post-construction. We cannot fully attribute increased detections post-construction to mitigation efforts, and other factors, such as habitat availability, urbanization, or demography, may have also influenced underpass use along CA-71. Nonetheless, even with the expansion of the freeway and subsequent increase in traffic volume, mitigation structures along CA-71 did allow for continued movement and hence connectivity across the roadway for the target species.

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CHAPTER 1

MARK-RECAPTURE AND MARK-RESIGHT POPULATION ESTIMATION OF

BOBCATS IN AN URBAN COASTAL RESERVE USING REMOTE CAMERAS

INTRODUCTION

Reliably estimating abundance for carnivore populations can be difficult because many carnivores exist in low densities and are wide-ranging, nocturnal, secretive, and persecuted by humans (Noss et al. 1996, Crooks 2002, Balme et al. 2009). Consequently, traditional methods of physical capture and direct observation have been replaced by the use of non-invasive detection techniques (Cutler and Swann 1999, Ruell et al. 2009). Remotely-triggered cameras offer a viable option for researchers interested in non-invasively assessing carnivore abundance (Balme et al. 2009). Specifically, camera traps are low cost, low maintenance, and create minimal disturbance (Cutler and Swann 1999). Moreover, photo records offer concrete visual evidence to derive date, time, frequency of visits, and individual identification.

The use of camera trap data to estimate carnivore abundance has grown since initial efforts in the mid-1990s (e.g., Mace et al. 1994, Karanth 1995). To date, photo data have been utilized in both a mark-recapture and mark-resight framework to estimate population numbers. Under the photographic mark-recapture framework, researchers do not physically capture and mark animals. Instead, researchers non-invasively capture animals via photograph and identify individuals by their pelt pattern or other natural markings, i.e., "mark" animals based on unique natural characteristics. The initial photographing or capture occasion is followed by several recapture occasions where marked animals (those identified as unique individuals in previous photos) are recaptured by cameras, and animals not previously identified in photos are noninvasively marked by pelt identification (Karanth 1995). Most photographic mark-recapture

studies have focused on species with unique pelage patterns such as tigers (Karanth 1995), ocelots (Dillon and Kelly 2007), jaguars (Soisalo and Cavalcanti 2006), snow leopards (Jackson et al. 2006), and bobcats (Heilbrun et al. 2006).

When animals do not have unique pelage or other natural markings, it may be difficult to use camera data in a mark-recapture framework because individuals cannot be identified by photograph alone. However, if researchers can physically mark animals and individually identify the tagged animals with photographs, mark-resight models may be appropriate (White 1996, McClintock et al. 2009). In mark-resight studies, after the initial physical marking of individuals, there may be one or several resighting occasions in which physically marked animals are resighted but unmarked animals remain unmarked and are counted as such (White 1996). This distinguishes mark-resight from mark-recapture studies in which new marks are introduced during recapture occasions after the initial capture and marking session (Otis et al. 1978).

Herein, we use camera data to estimate population sizes of bobcat (*Lynx rufus)* within and around an urban coastal reserve in southern California. Due to their sensitivities to urban fragmentation, bobcats have been focal species in several studies throughout southern California (Crooks 2002, Tigas et al. 2002, Riley et al. 2003, George and Crooks 2006, Riley et al. 2006), yet few bobcat density estimates exist for this region (except see Ruell et al. 2009). Bobcats are individually identifiable by pelt patterns and thus photo data for this species can be used with mark-recapture models without physically marking animals (Heilbrun et al. 2006, Larrucea et al. 2007). In addition, we conducted our camera survey in conjunction with an ongoing GPS telemetry study where animals were physically marked by researchers, therefore also enabling a mark-resight framework. Since the flexibility of our photo data enables the use of both

mark-recapture and mark-resight models, we compare these approaches and evaluate the potential of mark-resight models for use with species that may not have unique pelage markings but are physically marked.

METHODS

Study Area

The San Joaquin Hills study area was located within the Coastal Reserve (33°36'N; 117°47'W) of the Nature Reserve of Orange County, south and west of two principal 10-lane freeways, Interstates 5 and 405, between the cities of Costa Mesa and Laguna Niguel, Orange County, California (Figure 1.1). The landscape contained a mix of urban and suburban development (housing developments, shopping centers, commercial centers, golf courses, urban parks, and greenbelts) as well as natural habitat, including undeveloped private property, nature reserves, state parks, and county parks. Natural habitat primarily consisted of coastal sage scrub, chaparral, riparian, coastal oak woodland, and annual grassland communities.

Camera Trap Survey

We created a sampling scheme by overlaying a square grid on a topographic map. The grid consisted of 30 sampling units that were each 2 km x 2 km (4 km^2), a unit size that represented the average bobcat home range in the North Irvine Ranch (Lyren et al. 2006), a useful approximation for the preferred cell size of camera grids (Zielinski et al. 1995) (Figure 1.1). For camera monitoring, we only considered grid cells that intersected open space parks and reserves, given that bobcats typically avoid urbanized areas (Tigas et al. 2002, Riley et al. 2003). To determine camera placement, each sampling unit was further subdivided into 16 grid cells measuring 500 m x 500 m each, one of which we randomly selected for installation of a camera station. We used the presence of bobcat sign (e.g., tracks, scat) and expert opinion to select the specific camera trap locations within each 500 m x 500 m cell to increase the probability of

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passively detecting bobcats via camera traps. In total, we placed 30 film camera traps (Camtrakker; CamTrak South Inc.) in 30 sampling units throughout the study area. Cameras were attached to a post and placed perpendicular to the travel route to capture the best possible photographs for pelt identification and matching (Heilbrun et al. 2003) and set with a 3 minute delay between successive photographs. We conducted camera surveys from July 2006 to January 2007.

Physical Capture

In addition to the sampling grid, bobcats were detected on the study area via physical captures for a concurrent GPS telemetry research project. Bobcats were captured in cage traps (61 x 43 x 109 cm) placed in locations based on sign and knowledge of bobcat movement. We anesthetized animals using a combination of ketamine (10mg/kg) and xylazine HCL (1mg/kg). Bobcats were fitted with a unique combination of an ear tag and colored taping on the GPS telemetry units, or an ear tag and a cat collar. During processing we photographed all animals on both sides of their body in a series of systematic poses to aid in identification of physically captured animals later photographed by camera traps. As described in Heilbrun et al. (2003), the poses included photographs of the fore legs, hind legs, torso, face, and tail of the captured animals. These capture photos were compared to the remotely-triggered camera photos to assist with individual identification. The United States Geological Survey (USGS) and Colorado State University (CSU) Animal Care and Use Committees approved all capture and handling procedures.

Photograph Identification

We individually identified animals by comparing bobcat photographs using the pelt-pattern identification protocol outlined by Heilbrun et al. (2003). Since our camera traps

consisted of a single camera, most useable photos were taken of either the left or right side of the body. Photographs of poor quality due to inadequate lighting, distance (e.g., too close or too far from the camera), and extreme angles (e.g., walking strait into or away from the camera) were not included because individual pelts could not be reliably identified. Following Heilbrun et al. (2003), individual bobcats were matched by confirming that three natural pelage features (e.g., groupings of leg spots, groupings of body spots, facial markings, and tail markings) or introduced marks (i.e., ear tags or GPS collar) were present in both photographs. The identification of a differing feature between pelt patterns of photographed bobcats indicated unique individuals.

Model Framework

To estimate bobcat abundance, we used closed capture mark-recapture and mark-resight estimators in program MARK (White and Burnham, 1999). Under the mark-recapture framework, there is an initial capture and marking occasion followed by several recapture occasions where new marks are added to the population (Otis et al. 1978). Marks added in either the initial capture occasion or subsequent occasions may be man-made introduced marks (e.g., tags, collars, dyes etc.) or the identification of natural marks (e.g., individually unique pelage patterns or other natural characteristics). In our mark-recapture study, we initially marked animals captured via camera traps by identifying individual pelt patterns. After the initial photographic capture and marking occasion, if we identified a new bobcat in subsequent recapture occasions that was previously unidentified, we then considered it marked. We then created individual capture histories for each bobcat captured via photograph. Although bobcats are uniquely spotted, they are bilaterally asymmetrical (Heilbrun et al. 2003). We were therefore unable to match left-side photos with right-side photos because we used a single camera at each

camera trap. Consequently, photo data for the mark-recapture models were split into left- and right-side datasets. In addition, the mark-recapture models in program MARK require sampling without replacement, meaning individuals can be recaptured a maximum of once per sampling occasion. As such, we defined sampling occasions as 23 day periods pooled across all cameras such that capture probabilities were >0.1 (Otis et al. 1978).

Under the mark-resight framework, the initial capture occasion adds new marks (natural or introduced) and is followed by resighting occasions where, unlike the mark-recapture framework, no new marks are added to the population. Thus, unmarked animals remain unmarked throughout the study (White 1996). In our study, the initial capture occasion occurred during the physical capture of bobcats for an ongoing GPS telemetry project. As described above, we physically marked the animals during the capture process with a unique combination of an ear tag and colored taping on the GPS collar, and we took a series of photos. Afterward, we used the remotely-triggered camera grid to resight those tagged animals. Bobcats that were never physically captured were considered unmarked throughout the study. Since we physically marked animals and photographed both sides of their body during the initial marking phase, we were able to combine both the left and right side photo datasets. We used the Poisson log-normal (PNE) mark-resight estimator in program MARK (McClintock et al. 2009). Unlike the mark-recapture models discussed above, sampling may be with replacement for the PNE so sampling occasions do not need to be delineated (McClintock et al. 2009); instead, all photo resightings were counted over the duration of the study. In addition, the number of marked animals in the population must be known.

Assumptions

Closed population abundance estimation, including both mark-recapture and mark-resight estimators, entails 4 key assumptions: 1) the study population is closed both geographically and demographically, 2) marks (both natural and introduced) are not lost, 3) marks are properly identified, and 4) unequal detection probabilities among animals in the population must be accounted for (McClintock and White 2010, Otis et al. 1978, White et al. 1982). We believe we satisfied the assumption of geographic closure because urbanization surrounded the study area to the north, east, and south, while the Pacific Ocean bordered the study area to the west. In addition, during the study we did not detect any bobcats moving onto or off of the study area via GPS telemetry or remotely-triggered cameras (Lyren et al. 2008b, Lyren et al. 2008a). While we did not document any births during the study period, the assumption of demographic closure was not met because we documented bobcat mortality during the study period, including six bobcat road kill mortalities. As such, our results must be interpreted as an estimate of the population at the beginning of the study period (McClintock and White 2010 , Otis et al. 1978, White et al. 1982).

Regarding the second and third assumptions, we observed retention of the introduced marks, and the bobcat pelt patterns used to identify individuals do not change over time, ensuring the retention of marks throughout the study (Heilbrun et al. 2003, Larrucea et al. 2007). In addition, the pelt pattern identification protocol outlined by Heilbrun et al. (2003) was designed to minimize incorrect matching of bobcat pelts by requiring that the three pelage features on the animals or one introduced man-made mark must match. Identifying at least one differing feature between animals confirmed unique individuals. Furthermore, several researchers in this study independently reviewed and confirmed the matches.

Regarding the fourth assumption, both mark-recapture and mark-resight estimators can account for unequal detection probabilities but do so differently. Mark-recapture models in program MARK were designed to explore sources of variation in capture probabilities due to time, behavior, or individuals (Otis et al. 1978, White et al. 1982). We ran models (M) that assumed no capture heterogeneity (Mo), explored capture heterogeneity due to time (Mt) based on wet (December-January) and dry (July-November) seasons, capture heterogeneity based on individual sources of variation using two mixtures (Mh2) (Pledger 2000), and capture heterogeneity due to seasons and individual variation (Mth2) (Otis et al. 1978). The PNE mark-resight estimator accounts for individually heterogeneous resighting probabilities if the marks are individually identifiable (McClintock and White 2010), as was the case in our study. We ran PNE mark-resight models with and without the individual heterogeneity parameter (sigma). To further explore individual heterogeneity in the PNE estimator, we associated three covariates with both the sigma parameter and the alpha parameter (the intercept for the mean resighting rate) (McClintock et al. 2009); the overall mean resighting rate can then be derived from these parameters (McClintock and White 2010). The three covariates assigned to the alpha and the sigma parameters were: known weight at the time of physical capture (W); sex (S); and Euclidian distance (D) of the physical capture location to the nearest camera station as measured in ArcGIS 9.3. To assess the effect of the covariates on the model parameters and overall mean resighting rate, we evaluated the directionality of the beta coefficient for each covariate present in the top PNE model set, and whether the confidence intervals around the coefficient overlapped zero. The PNE mark-resight estimator has one additional assumption that marked animals and unmarked animals have independently and identically distributed resighting probabilities. This assumption requires that the marked sample accurately represents the sighting probabilities of the

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entire population and that sighting probability is independent of marking status (Bowden and Kufeld 1995). We believe this assumption was satisfied by using a different method for physical capture and marking (i.e., cage traps) from the resighting method (i.e., camera traps).

Because we constructed multiple models within the model sets for each estimator (left-side mark-recapture, right-side mark-recapture, and PNE), we calculated abundance by model averaging over the entire model set for each estimator (Burnham and Anderson 2002).

Density Estimation

Density estimates can be sensitive to methods used to determine the size of a study area (Dillon and Kelly 2008, Balme et al. 2009). As such, it may be difficult to compare densities if methods of study area delineation vary across studies (Dillon and Kelly 2008). To determine the size of the effective study area, we used methods similar to Ruell et al. (2009), who estimated bobcat densities via non-invasive fecal DNA surveys in the Santa Monica Mountains north of Los Angeles. We used our GPS telemetry data to calculate the mean bobcat home range size (8.83 km^2), estimated with a fixed 95% kernel (Powell 2000), and 95% confidence intervals about that mean (5.26 km^2 to 12.39km^2) using the t-distribution. We then created buffers in ArcGIS 9.3 around each camera location for the radius (1.68 km) and the diameter (3.35 km) of the mean home range size. We also generated buffers around each camera for the radius and diameter derived from the lower confidence limit (radius = 1.29 km; diameter = 2.59 km) and the upper confidence limit (radius = 1.99 km; diameter = 3.97 km) of the mean home range size. The buffers were dissolved into one layer for the radius and one layer for the diameter measurements, thus eliminating overlap between the individual camera buffers. Following Ruell et al. (2009), we removed areas of urbanization from the buffer, which our GPS collared bobcats generally avoided; 7% (SEM = 2%, N=16 bobcats) of GPS locations per individual were located

in urban areas, as classified by GIS land-use layers from the Southern California Association of Governments (SCAG 2005). The buffer did include golf courses, regional parks, riparian strips, and natural habitat, which bobcats used more frequently (Figure 1.2).

Following Ruell et al. (2009), to calculate density point estimates and ranges about those estimates, we first divided our abundance estimate by the effective study area derived from the mean home range radius and again separately by the area derived from the mean home range diameter. To generate minimum density estimates, we divided the lowest limit of the abundance estimate confidence interval by the upper limit of the effective study area confidence interval derived from the radius and then the diameter of the home range. Conversely, to generate maximum density estimates, we divided the upper limit of the abundance estimate confidence interval by the lower limit of the effective study area confidence interval calculated from the home range radius and diameter.

RESULTS

We captured 109 bobcat photos in 4,669 camera trap nights. Seventeen of those photos were not used in the mark-recapture or mark-resight analysis due to poor photo quality. For the closed capture mark-recapture analysis, we organized the remaining usable photographs into two datasets consisting of 49 right- and 43 left-side photos. Since sampling is without replacement in the closed capture mark-recapture estimator, we were able to use only 35 right-side photos and 35 left-side photos because some individuals were captured multiple times within a sampling occasion. We identified 24 individual bobcats in the left-side photo data set and 23 individuals in the right-side photo data set.

Model-averaged closed capture mark-recapture point estimates slightly differed between right-side (44 bobcats) and left-side (39 bobcats) datasets, but confidence intervals overlapped substantially (Table 1.1). Within the closed capture mark-recapture models sets, right- and left-

side datasets supported differing models with respect to capture heterogeneity (Table 1.2). The right-side dataset most supported the models with an individual heterogeneity effect with two mixtures (model 1), as well as an individual heterogeneity effect and a seasonal time effect (model 2). Weaker support existed for the null model (model 3) and the model with only a seasonal time effect (model 4). Conversely, the left-side dataset most supported the null model with no capture heterogeneity (model 5). The model in which both captures and recaptures could differ depending on seasonal variation, but with no heterogeneity (model 6), garnered the second most support from the data. Finally, models including individual capture heterogeneity (model 7) and individual capture heterogeneity with a seasonal effect had weaker support (model 8).

Under the PNE mark-resight framework, sampling is with replacement so all 92 usable photographs were combined into one dataset. We physically captured and marked 27 bobcats on the study area and 15 of those animals were resighted with the remotely-triggered camera grid, accounting for 45 photographs. The model-averaged PNE mark-resight abundance estimate of 53 bobcats was slightly higher than the closed capture mark-recapture abundance estimates, but again, all confidence intervals overlapped (Table 1.1). PNE mark-resight models that included the body weight covariate (W) associated with the individual heterogeneity parameter (sigma) appeared in four of the top five models and held a variable importance weight of 0.917 (Table 1.3). Furthermore, the top model in the PNE model set included only the weight covariate associated with individual heterogeneity and had relatively strong support with a model probability of 60.6%. The effect of the weight covariate in the top model was positive (coefficient = 1.39; 95% CI = 0.98 - 1.96), suggesting that heavier animals displayed more variation in resighting rates than lighter animals and had a higher overall mean resighting rate. This same trend held for the weight covariate (W) in models 2 and 3 of the model set. There was

less evidence of gender (S; 0.210 variable importance weight) or distance (D; 0.190 variable importance weight) effects for explaining individual capture heterogeneity. All other mark-resight models showed relatively little support but were included in the model-averaged abundance calculation at their respective weights for each model set.

The effective study area size ranged from 96.2 km^2 (95% CI = 79.1 - 104.0), derived from the radius of the average bobcat home range, to 119.1 km^2 (95% CI = 112.9 - 122.1), derived from the diameter of the average home range. Density point estimates ranged from 0.30 – 0.55 and 0.24 - 0.45 bobcats per km^2 with the study areas defined by the radius and diameter of an average bobcat home range, respectively (Table 1.1). Although the density point estimates varied between estimators, the confidence intervals for each estimator were relatively large (119 - 143% CI length) and overlapped.

DISCUSSION

The flexible design of our remote camera study and its association with an ongoing GPS telemetry study enabled us to compare the use of mark-resight and mark-recapture frameworks to estimate the abundance and density of a fragmentation-sensitive carnivore, the bobcat in urban southern California. The PNE mark-resight estimator produced higher point estimates than the mark-recapture estimators, although the confidence intervals overlapped, suggesting they performed comparably.

In a non-invasive remotely-triggered camera study, both the mark-resight and the mark-recapture frameworks present advantages and disadvantages. If animals can be individually identified via natural pelage markings, the mark-recapture framework presents a clear advantage because animals may never need to be physically handled since they can be non-invasively identified via photograph. If researchers do plan to use the mark-recapture framework, we recommend using a dual camera trap setup, as has been deployed in several prior studies (e.g.,

Karanth 1995, Karanth and Nichols 1998, Dillon and Kelly 2007). In our study, we needed to split our data set depending upon whether the bobcat was photographed on the right- or left-side because we could not reliably match right- and left-side photos due to asymmetrical pelt patterns. If the full dataset could have been used, our mark-recapture results would have been more precise due to increased sample size.

In contrast, under the mark-resight framework, animals are initially marked using a different method than the resighting technique (Bowden and Kufeld 1995). This typically would involve physical capture and thus is more invasive and costly than the mark-recapture framework if animals can be individually identified via natural markings. The strength of the mark-resight models arises when animals may not have distinct pelage patterns that can be readily identified by observers. In such a case, if researchers are handling animals, for example to deploy telemetry collars, then the concurrent use of remote cameras to acquire non-invasive resightings presents a viable opportunity for the use of the mark-resight framework to estimate abundance. Additionally, the ability to use individual covariates to explore resighting rates and individual heterogeneity (McClintock and White 2010) in a remote camera study presents a distinct advantage over mark-recapture models. The physical capture of animals, which is the most invasive aspect of using a mark-resight framework, is typically what allows for the collection of individual covariate data such as weight and gender. Although the mark-recapture framework can support such covariates (White 2008), because captures in a mark-recapture camera study occur via non-invasive photography, such individual covariate data can be difficult to obtain. Further, the ability to use one camera to detect introduced man-made marks in a mark-resight study, as opposed to two cameras to detect bilaterally asymmetrical natural markings in a mark-recapture study, is another potential advantage of the mark-resight framework. In addition, the

process of matching marks (natural or introduced) among photos may be streamlined, because only those animals who were captured and marked in the initial capture session need to be identified, as opposed to the mark-recapture estimators which requires identification of every individual photographed. Finally, sampling can be with replacement for the PNE estimator, so delineation of secondary sampling occasions is not necessary, which is intuitive for camera trap data; generally cameras traps are placed on the landscape and run for the duration of the primary study period, with no designation for secondary sampling occasions. Sampling with replacement means that all identifiable photographs are used and no identifiable photos are discarded, a contrast with the mark-recapture estimators, in which multiple detections of the same individual within the same sampling occasion are discarded and only one detection of that individual is used in the estimator.

Our estimates of bobcat density (0.40 to 0.55 per km^2) were slightly higher than estimates for another study area in the region (0.33 to 0.42 bobcats/km^2) using non-invasive scat surveys in a mark-recapture framework in a similarly-defined study area north of Los Angeles, California (Ruell et al. 2009). Notably, Ruell et al. (2009) suggested that relatively low densities could be due to a recent notoedric mange epizootic in their bobcat population, and that prior densities in the area were suspected to be $\geq 0.6\ km^2$ as estimated from radio-telemetry data. This suggests that the population sizes we estimated in our study area using both the mark-recapture and mark-resight frameworks were reasonable and within the bounds of similar density estimates for bobcats in the region generated with other approaches. Although our density estimates were similar to Ruell et al. (2009), our confidence intervals were somewhat less precise. Confidence interval width for density ranged from 0.38 to 0.90 bobcats per km^2 in our study compared to 0.32 to 0.54 in Ruell et al. (2009) depending upon the method used to define the effective study

area (radius or diameter of mean bobcat home range). The lower precision in our study

highlights the sensitivity of both mark-recapture and mark-resight estimators to small data sets

often typical of large carnivore studies.

MANAGEMENT IMPLICATIONS

The non-invasive population estimation enabled by the mark-recapture and mark-resight

frameworks provides insight into the management and conservation of our study population,

which is largely isolated from other bobcat populations by urbanization and roadways. For

example, a potential cause for concern is the relatively high mortality we recorded in this

population, including six mortalities during our study due to roadkill alone. Using the broadest

confidence intervals from our estimates, these roadkill mortalities could represent anywhere from

7% to 22% of the population. It is unknown if this type of mortality is additive or compensatory,

but we recommend that management agencies in the study area consider monitoring this

population and implement actions to mitigate road mortality to prevent population decline.

Importantly, the mark-resight and mark-recapture estimators in program MARK can be used in a

robust design where the population can be monitored during periods of population closure as

well as open periods (McClintock and White 2010, White 2008). This type of design would

allow for the estimation of apparent survival and temporary emigration (Kendall et al. 1995,

Kendall et al. 1997, McClintock and White 2009), which would be important for monitoring

long-term population trends. The relatively low precision of our estimates, however, warrants

caution when designing a long-term study and interpreting trends. Although these

methodologies can account for the inability to perfectly detect carnivores, broad confidence

intervals reduce the ability to discern biologically important population fluctuations. Increasing

capture probabilities will help achieve more precise estimates, and one possible solution is to

increase the density of camera traps on the landscape (Larrucea et al. 2007). Another is to use a

dual camera setup, which would eliminate the need to split the data set for use with mark-

recapture estimators, thereby increasing capture probabilities and sample size.

Table 1.1 Camera survey model-averaged mark-recapture and mark-resight bobcat (*Lynx rufus*) abundance and density estimates in the San Joaquin Hills study area, Orange County, California. % CI length denotes the confidence interval length relative to the abundance and density estimates.

Estimators	Abundance			Density/km^2 (Radius of average home range)			Density/km^2 (Diameter of average home range)		
	N-hat	95% CI	% CI length	D-hat	95% CI	% CI length	D-hat	95% CI	% CI length
Mark-recapture (Right side)	44	30 – 87	130	0.45	0.28 - 1.10	180	0.37	0.24 - 0.77	143
Mark-recapture (Left side)	39	29 – 69	103	0.40	0.28 - 0.87	150	0.32	0.24 - 0.61	119
Mark-resight (PNE)	53	27 – 92	123	0.55	0.26 - 1.16	164	0.45	0.22 - 0.82	131

Table 1.2 Camera survey closed capture mark-recapture model results for bobcats (*Lynx rufus*) in the San Joaquin Hills study area, Orange County, California.

	Model	Capture Heterogeneity Description	Delta AICc	Model Weight	N-hat	SE	95% CI
		Mark-Recapture Results					
Right Side Results							
1	{Mh2}	Individual (2 mixtures)	0.0	0.491	44.5	13.1	30.1 - 87.6
2	{Mth2}	Individual (2 mixtures) and Seasonal	0.2	0.436	44.1	12.9	30.0 - 86.7
3	{Mo}	None	5.0	0.040	33.8	6.5	26.7 - 55.0
4	{Mt}	Seasonal	5.4	0.034	33.7	6.4	26.6 - 54.6
Left Side Results							
5	{Mo}	None	0.0	0.562	37.9	7.9	28.9 - 63.3
6	{Mt}	Seasonal	1.2	0.304	37.8	7.9	28.8 - 63.1
7	{Mh2}	Individual (2 mixtures)	3.7	0.088	43.7	16.2	28.8 - 104.3
8	{Mth2}	Individual (2 mixtures) and Seasonal	5.0	0.047	43.6	16.1	28.8 - 103.8

Table 1.3 Camera survey mark-resight (PNE) model results for bobcats (*Lynx rufus*) in the San Joaquin Hills study area, Orange County, California. See text for details and description of model parameters and covariates.

	Model	Covariate Description	Delta AICc	Model Weight	N-hat	SE	95% CI
		Mark-Resight (PNE) Results					
1	{alpha(.), sigma(W), U(.)}	Weight on Individual Heterogeneity	0.0	0.606	54.1	19.4	27.4 - 107.1
2	{alpha(.), sigma(W+D), U(.)}	Weight and Distance on Individual Heterogeneity	2.9	0.145	52.0	22.3	23.2 - 116.5
3	{alpha(.), sigma(W+S), U(.)}	Weight and Gender on Individual Heterogeneity	3.0	0.137	54.3	19.8	27.2 - 108.6
4	{alpha(.), sigma(S), U(.)}	Gender on Individual Heterogeneity	5.8	0.034	43.5	14.6	22.9 - 82.5
5	{alpha(.), sigma(W+D+S), U(.)}	Weight, Gender, and Distance on Individual Heterogeneity	6.1	0.029	51.9	23.3	22.4 - 120.2
6	{alpha(.), sigma(.), U(.)}	No Covariates	7.3	0.014	58.9	12.5	39.1 - 88.8
7	{alpha(.), sigma(S+D), U(.)}	Gender and Distance on Individual Heterogeneity	8.2	0.010	44.8	15.7	23.0 - 87.4
8	{alpha(W), sigma(.), U(.)}	Weight on the Mean Resighting Rate Intercept	8.5	0.009	54.1	11.4	36.0 - 81.3
9	{alpha(.), sigma(D), U(.)}	Distance on Individual Heterogeneity	9.4	0.006	39.7	14.4	20.0 - 79.0
10	{alpha(S), sigma(.), U(.)}	Gender on the Mean Resighting Rate Intercept	10.6	0.003	48.4	13.1	28.7 - 81.7
11	{alpha(D), sigma(.), U(.)}	Distance on the Mean Resighting Rate Intercept	10.7	0.003	46.5	12.9	27.3 - 79.3
12	{alpha(W+S), sigma(.), U(.)}	Weight and Gender on the Mean Resighting Rate Intercept	11.2	0.002	54.4	11.3	36.4 - 81.2
13	{alpha(W+D), sigma(.), U(.)}	Weight and Distance on the Mean Resighting Rate Intercept	11.4	0.002	54.5	11.3	36.4 - 81.6
14	{alpha(S+D), sigma(.), U(.)}	Gender and Distance on the Mean Resighting Rate Intercept	13.3	0.001	50.4	14.5	29.0 - 87.7
15	{alpha(W+D+S), sigma(.), U(.)}	Weight, Gender, and Distance on the Mean Resighting Rate Intercept	14.4	0.000	54.8	11.2	36.8 - 81.5
16	{alpha(.), sigma(0), U(.)}	No Individual Heterogeneity	40.1	0.000	54.9	5.8	44.7 - 67.5
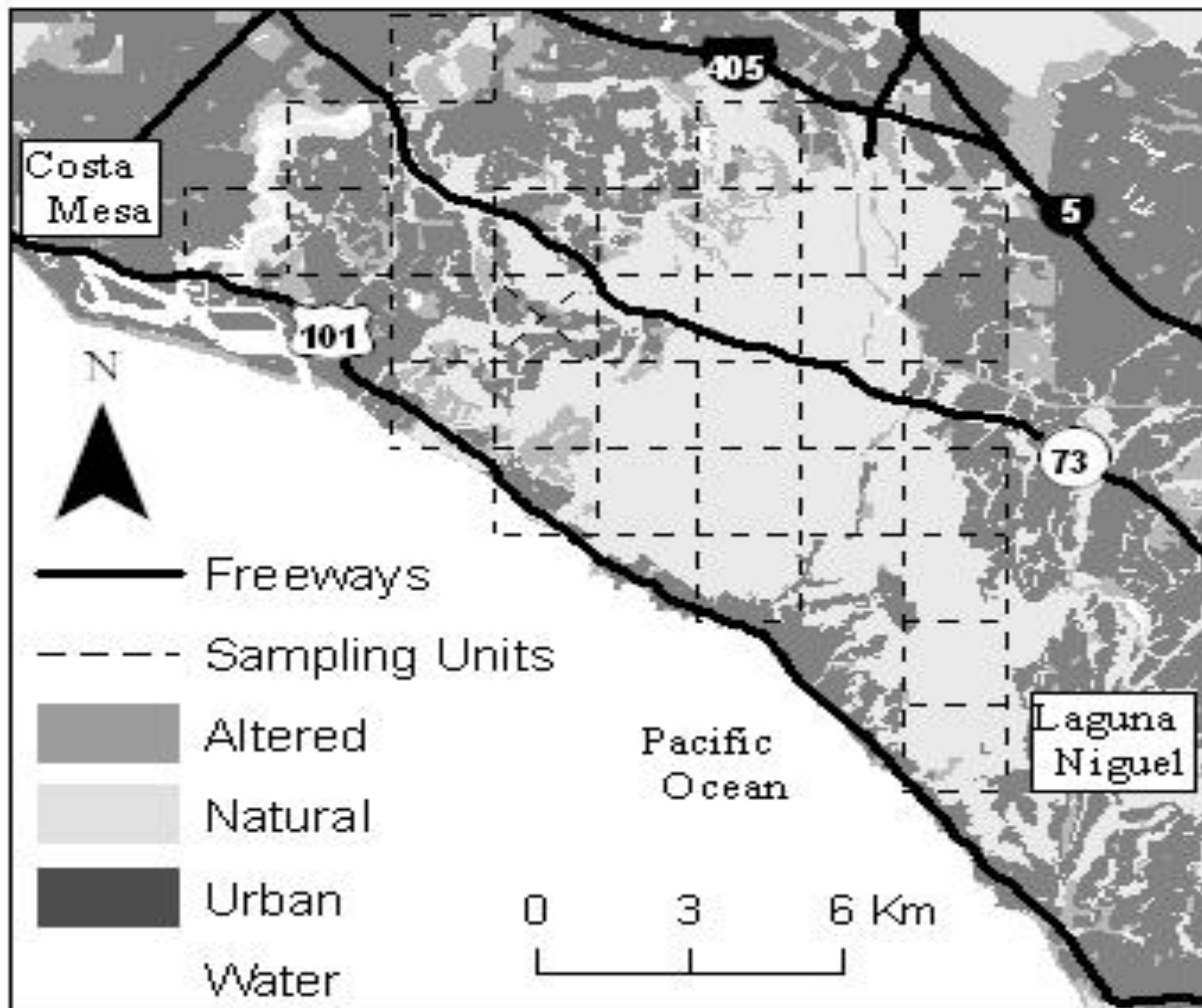
Figure 1.1 The San Joaquin Hills study area, Orange County, California. The sampling unit grid (dashed lines) was used to determine the locations of remote camera stations.
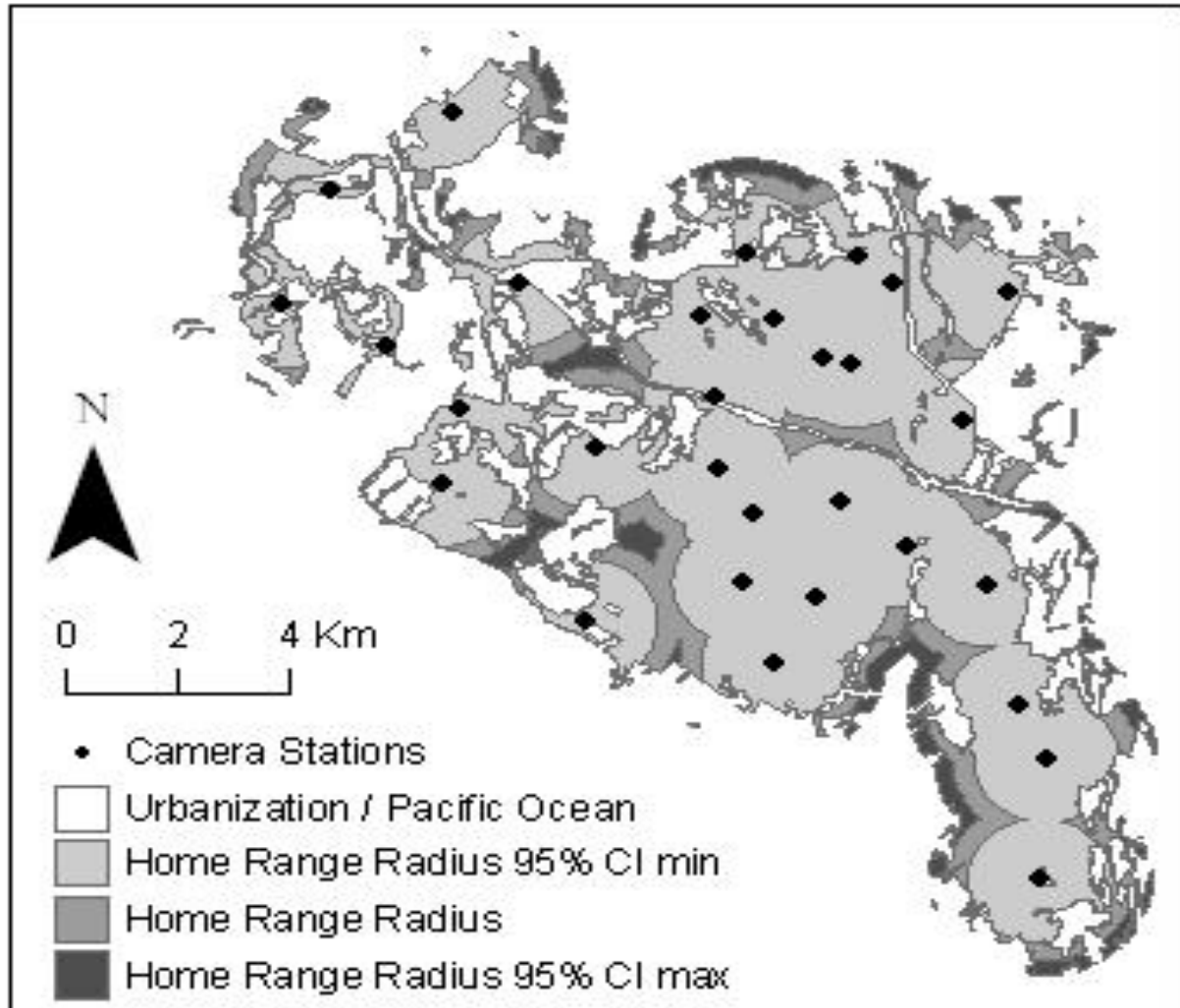
Figure 1.2 San Joaquin Hills effective study area and 95% CI estimated by buffering camera stations with the radius of the mean and 95% CI of bobcat home range size as determined from GPS telemetry data. Areas of high urbanization and the Pacific Ocean were removed because they were generally unavailable to bobcats.

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AN EVALUATION OF A ROAD EXPANSION AND WILDLIFE CONNECTIVITY

MITIGATION PROJECT ON A SOUTHERN CALIFORNIA FREEWAY

INTRODUCTION

Roads have both direct and indirect negative effects on biodiversity (Forman and Alexander 1998, Forman 2003, Coffin 2007). Direct effects of roads include habitat loss, a decrease in quality of adjacent habitat, wildlife mortality, and the creation of barriers to animal movement (Forman 2003). Since roads are the main network for human travel across the landscape, indirect effects include the facilitation of urban and agricultural development and in general, the expansion of the human network and associated anthropogenic disturbance. Large mammal species with broad area requirements, such as ungulates and carnivores, are especially susceptible to negative road effects (Fahrig and Rytwinski 2009). In addition, large mammals often have low rates of reproduction, slow population growth rates, exist in relatively low densities, and are particularly vulnerable to human persecution, which can further exacerbate the consequences of mortality and barrier effects of roads (Noss et al. 1996).

To offset the negative impact of roads on wildlife, a variety of mitigation measures have been implemented (Forman 2003). Specifically, wildlife crossing structures to facilitate the safe passage of animals across roads and wildlife fencing to help prevent animals from venturing onto roadways can be successful at reducing mortality and the barrier effect of roads (Clevenger and Waltho 2000, Clevenger et al. 2001, Forman 2003). Evaluation of mitigation measures is critical to determine their effectiveness for conserving connectivity, and it is important to maximize inferential strength of these evaluative types of studies (Roedenbeck et al. 2007, Fahrig and Rytwinski 2009). Surprisingly, few studies have assessed changes in wildlife movement before

and after installation of mitigation structures. One possible approach to achieve this goal is a before-after-control-impact (BACI) design, which has been applied in environmental impact studies but is uncommon in the field of road ecology (Roedenbeck et al. 2007, Fahrig and Rytwinski 2009).

Southern California is one of the most populous areas of the United States and the natural landscape is highly fragmented with urbanization and road development (Beier et al. 2006). This region has been identified as a biodiversity "hotspot" consisting of numerous endemic species juxtaposed with human development, thus creating a center of species endangerment and extinction (Myers 1990, Dobson et al. 1997, Myers et al. 2000). Previous research in the region has targeted large mammals as a focal group to study the effects of road development and urban fragmentation on animal movement and landscape connectivity (Crooks 2002, Tigas et al. 2002, Riley et al. 2003, Ng et al. 2004, Riley et al. 2006, Ruell et al. 2009). Along California State Route 71 (CA-71) through the Chino Hills southeast of Los Angeles, two studies in particular evaluated carnivore movement around and across the roadway from during1997 to 2000 (Haas 2000, Lyren 2001). Haas (2000) found that measurable characteristics of the road influenced frequency and probability of bobcat (*Lynx rufus*) and coyote (*Canis latrans*) usage of underpasses and culverts. In a radio-telemetry study focused on coyotes, Lyren (2001) showed that coyote use of culverts was negatively correlated to peak traffic periods.

In 2005, to facilitate increased traffic flow, the California Department of Transportation (Caltrans) added a northbound and a southbound lane to a 4 km segment of CA-71, thereby widening the highway and lengthening some culverts that had previously supported carnivore movement (Haas 2000, Lyren 2001). In addition, during the widening process, Caltrans incorporated multiple mitigation measures recommended by Haas (2000) and Lyren (2001).

These measures included the installation of two span bridges where culverts previously existed, wildlife fencing, and concrete center dividers, as well as native vegetation restoration around culverts and wildlife underpasses. Importantly, adjacent segments of CA-71 that were studied by Haas (2000) and Lyren (2001) were not modified during the 2005 construction.

We took advantage of the construction along CA-71 and the prior wildlife studies along this roadway (Haas 2000, Lyren 2001) to study an impacted area and a control area both before and after roadway construction. Our objective was to quantify how the widening of CA-71 and implementation of mitigation measures affected large mammal movement across the roadway. Using remote camera survey data collected prior to and following the 2005 construction project, we evaluate the possible impacts of road construction and success of roadway mitigation on underpass use of bobcats, coyotes, and mule deer (*Odocoileus hemionus*).

METHODS

Study Area and Study Periods

Our study area was located southeast of the greater Los Angeles area along an 8 km north-south stretch of CA-71 between Pine Avenue and California State Route 91 (CA-91), including a 4 km segment where CA-71 was widened in 2005. The freeway delineated two large blocks of habitat that differed in topography and vegetation type, and that changed between our two study periods of November 1997 - January 2000 (before highway widening and mitigation) and August 2008 - September 2009 (after highway widening and mitigation). To the east of CA-71, Prado Flood Control Basin was relatively flat and, during the November 1997 - January 2000 study period, dominated by riparian vegetation and non-native eucalyptus forest, with some smaller amounts of non-native annual grassland. By the August 2008 - September 2009 study period, habitat restoration projects in the Prado basin had removed much of the eucalyptus forest next to CA-71 and replaced it with native coastal sage scrub, leaving primarily riparian

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vegetation with some coastal sage scrub, eucalyptus forest, and non-native annual grassland. To the west of CA-71, steep hills and valleys characterized the Chino Hills. Before road widening and mitigation, habitat on the Chino Hills side of CA-71 was predominantly invasive annual grassland with some native coastal sage scrub. This habitat burned in the Freeway Complex wildfire in November 2008. Post-fire and during our 2008-2009 study period, much of the habitat in Chino Hills was barren, with a few pockets of invasive annual grassland and coastal sage scrub (Figure 2.1).

CA-71 Expansion, Mitigation, and Underpasses

In 2005, Caltrans widened a 4 km segment of CA-71 on the western side of the freeway to accommodate a new northbound and a new southbound lane, thereby expanding the freeway from two to four lanes. Before widening, there were 25 potential crossing structures under CA-71 in the 4 km impact-zone segment. During widening, 7 of those 25 structures were lengthened by 0.36 to 5.92 m (Table 2.1). As mitigation for highway widening, two span bridges (71-08, 71-14; Table 2.1) were installed during construction. Each bridge replaced a pair of culverts vertically stacked on top of each other in each location. Thus, the creation of the 2 span bridges removed 4 culverts, for a total of 23 underpass structures in the impact zone after construction. Most structures for the entire 8 km study area were reinforced concrete pipes (60.6% of 36 structures), but they also included corrugated metal pipes (11.3%), bridges (11.3%), reinforced concrete boxes (8.5%), and arch culverts (8.5%).

As further mitigation, Caltrans fenced the entire length of the impact zone using 3 m high, 10 x 15 cm mesh wildlife fencing, and restored native vegetation around crossing structures in the construction zone. Finally, highway center dividers were upgraded from guardrails to concrete dividers in the impact zone to prevent animals from attempting surface crossings. We

defined 3 km north and 1 km south of the impact zone on CA-71 as a control zone since the

roadway and the 13 possible crossing structures under it were not altered by the expansion or

mitigation project (Figure 2.1).

Monitoring Techniques

To evaluate wildlife use of CA-71 underpasses by large mammals, we sampled wildlife

activity at potential crossing structures with remotely-triggered cameras placed perpendicular to

the path of an animal entering or exiting the underpass. Target species were bobcat, coyote, and

mule deer. We considered detections of animals at underpass cameras as an indication of

underpass use. Since mule deer stand roughly 1 m tall at the shoulder (Anderson and Walmo

1984) and we detected deer at some underpasses too small to support deer movement, we

evaluated mule deer activity at underpasses >2.5 m in height, the minimum recommended

underpass height for mule deer use (Gordon and Anderson 2004, Clevenger and Huijser 2011).

Prior to road expansion and mitigation, remotely-triggered film cameras (Camtrakker;

CamTrak South Inc, Watkinsville, GA) were placed on the western side of CA-71 at 21 of 36

crossing structures, 18 located in the impact zone and 3 located in the control zone. Pre-

construction cameras sampled from November 1997 through January 2000 (Haas 2000, Lyren

2001). After road construction, remotely-triggered digital cameras (Cuddeback Expert;

NonTypical Inc, Park Falls, WI) were placed at underpass entrances on both the west (Chino

Hills) and east (Prado Basin) sides of the freeway. Cameras sampled 18 of 23 structures in the

impact zone and 10 of 13 structures in the control zone. Post-construction cameras sampled

from August 2008 through September 2009. Nineteen of these structures (16 in the impact zone,

3 in the control zone) were monitored both before and after construction and were included in the

analyses (see below). These 19 structures included the 2 span bridges that replaced 4 previous

structures in two locations and the 7 structures in the impact zone that were lengthened during road widening (Table 2.1).

Data Analysis

Since fewer crossing structures were sampled prior to road expansion and mitigation, and those structures were only sampled with a single camera on the western side of CA-71, we used the data from the same camera locations at those 19 structures that were sampled both before and after construction (Table 2.1). We calculated an index of relative activity from the camera data for each of the target species by dividing the number of detections of a species at a specific camera station by the number of nights sampled at that same camera station (George and Crooks 2006). We used this index as a measurement of underpass use for our analysis. Because our data did not meet the assumption of normality and our design was unbalanced, we could not use a two-way repeated measures ANOVA approach most commonly associated with the BACI design (Green 1993, Smith 2002, Roedenbeck et al. 2007). Instead, we used a series of non-parametric tests to evaluate differences in underpass use both before and after the expansion and mitigation project as well as between the control and impact zones. For before and after comparisons of bobcat and coyote underpass use, we conducted two analyses for each species using Wilcoxon signed-rank tests, first pairing on the underpass location for all 19 underpasses in the impact and control zone, and then restricting the analyses to the 16 underpasses in the impact zone; small sample size precluded before-after comparisons within the control zone. For comparisons of underpass use between the control and impact zones, we conducted two analyses for each species using Wilcoxon-Mann-Whitney rank sum tests, first comparing the control and impact zones before construction and then comparing the control and impact zones after construction.

For mule deer, only four sampled underpasses (control: 71-24; impact: 71-04, 71-14, 71-18; Table 2.1) met the minimum recommended height (> 2.5 m) to support deer movement (Gordon and Anderson 2004, Clevenger and Huijser 2011) prior to the expansion and mitigation project. Due to the construction process, one of those four underpasses (71-04) was lengthened by 5.3 m, and a span bridge (71-14) replaced another where two vertically stacked culverts previously existed; the other two underpasses (71-24, 71-18) were not modified. Two other vertically stacked underpasses too small to support deer movement pre-construction were replaced by the second span bridge (71-08), resulting in five underpasses sampled post-construction that were potentially large enough for deer use (control: 71-24; impact: 71-04, 71-08, 71-14, 71-18; Table 2.1). The small number of underpasses that were candidates for deer movement precluded comparisons of deer underpass use between treatment and control zones as well as the before and after comparisons separately for each zone. Consequently, we used a Wilcoxon signed-rank test to evaluate if deer camera indices differed before and after construction, pooling structures among the treatment and control zones and pairing on the five post-construction underpass locations.

RESULTS

Before road expansion and mitigation, remote cameras installed at the 19 focal underpasses recorded 415 coyote photos at 18 underpasses (15 impact, 3 control), 125 bobcat photos at 13 underpasses (12 impact, 1 control), and 30 mule deer photos at 1 underpass (1 impact, 0 control) in 3,442 camera trap nights (Table 2.1). After the expansion and mitigation project, cameras stationed at the same locations recorded 1,139 coyote photos at all 19 underpasses, 511 bobcat photos at all 19 underpasses, and 419 mule deer photos at 4 of the 5 underpasses considered as candidates for mule deer use (4 impact, 0 control) in 4,950 camera

trap nights (Table 2.1). Cameras also recorded 14 deer photos at 3 impact underpasses (71-05, 71-12, and 71-16; Table 2.1) that were 1.05 to 1.5 m high. However, these few detections were of approaches rather than evidence of underpass use (see Discussion).

For bobcats, camera indices were higher after construction than before it for all 19 underpasses in the impact and control zones (Wilcoxon signed-rank $W = 5$, $p < 0.001$) and for the 16 underpasses in the impact zone ($W = 4$, $p < 0.001$; Figure 2.2). Bobcat underpass use, however, did not differ between the impact or control in either the before (Wilcoxon-Mann-Whitney $U = 12.5$, $n = 16$ impact, $n = 3$ control, $p = 0.211$) or the after ($U=25$, $n = 16$ impact, $n = 3$ control, $p = 0.958$) time periods (Figure 2.2). In contrast, coyote camera indices were higher in the control than in the impact zone during both the before (Wilcoxon-Mann-Whitney $U = 46$, $n = 16$ impact, $n = 3$ control, $p = 0.008$) and after ($U = 42$, $n = 16$ impact, $n = 3$ control, $p = 0.047$) time periods (Figure 2.3). Coyote underpass use did not differ, however, between the time periods for all 19 underpasses pooled together (Wilcoxon signed-rank $W = 71$, $p = 0.353$) or for the 16 underpasses in the impact zone ($W = 43$, $p = 0.211$; Figure 2.3). A non-significant trend suggested that deer camera indices were higher after the road widening and mitigation project than before the project for all 5 underpasses available for deer use (Wilcoxon signed-rank $W = 0$, $P = 0.100$; Figure 2.4).

DISCUSSION

We detected an increase in bobcat underpass use after the CA-71 road widening project compared to before it, both for all underpasses pooled together and specifically those in the impact zone. We cannot, however, fully attribute this increased activity solely to the mitigation efforts, particularly because underpass use appeared to increase in the control zone as well (Figure 2.2). The increase in bobcat detections after construction could have resulted from a variety of factors. It may reflect an increase in bobcat population size in the vicinity since the

initial pre-construction survey, although population densities before and after construction are unknown. The Freeway Complex fire in November 2008 also could have contributed to increased bobcat activity along the roadway. This fire extended through the Chino Hills due west of CA-71 and our control and impact zones (Figure 2.1B), and it destroyed most of the available habitat throughout the area. As a result, quality bobcat habitat was limited to a relatively narrow area immediately adjacent to CA-71, potentially increasing bobcat movement around and across the roadway. Nonetheless, regardless of what factors contributed to increased movement along CA-71, the mitigation measures did at least support greater underpass usage and thus movement of bobcats between the Chino Hills and Prado Basin, despite increased road width and traffic volume along the roadway post-construction. Annual average daily traffic volume throughout our study area increased 59.4% after construction, from 34,500 pre-construction (CalTrans 1999) to 55,000 post-construction (CalTrans 2009). The fact that this large increase in the amount of traffic did not noticeably reduce bobcat underpass use and hence movement between the Chino Hills and Prado Basin could be considered a successful outcome of the mitigation project.

In contrast to bobcats, coyote underpass use did not considerably differ before and after the road construction and mitigation project. Results did suggest, however, that coyote underpass use was higher in the control zone, although sample size in the control was limited. In particular, coyotes used one underpass in the control zone more frequently than any other structure in the study area during both the before and after sampling periods. This underpass was located directly next to a golf course, which likely supported a large prey source for coyotes, and was a major contributor to the increased underpass usage in the control zone. Importantly, although we did not detect an increase in coyote movement in the impact zone after construction

and mitigation, we did not detect a decrease in coyote underpass usage either. This finding again suggests the mitigation project was at least partially successful in allow carnivore movement to continue across the roadway.

Mitigation efforts also seemed to facilitate increased mule deer movement across CA-71 between the Chino Hills and Prado Basin. Prior to the mitigation project, we documented deer using only one underpass out of four sufficiently large enough to support deer movement. After the mitigation project, we documented deer using four out of five underpasses large enough to support deer movement. Specifically, deer were recorded using the two span bridges (71-08 and 71-14) installed during the mitigation project, whereas deer movement had not been detected in the four culverts at those two locations prior to the project. Further, a large box culvert not used by deer prior to the road project was used after construction, and the underpass that deer did use prior to construction underwent an 85.5% increase in deer usage post-construction. Only one large underpass, located at the northern limit of our study area in the control zone, did not support deer movement before or after the expansion and mitigation project. A relatively high amount of urbanization around this underpass likely contributed to this pattern, as mule deer have been known to avoid areas with human development and favor underpasses with more natural habitat (Nicholson et al. 1997, Ng et al. 2004). In the period after construction, we detected deer at three underpasses that we expected were too small (< 2.5 m in height) for deer use a total of 14 times; these images showed deer near the underpasses but not entering or exiting these structures. We again cannot fully attribute the trend of increased deer activity to the mitigation project, and it is likely that a number of factors contributed to this pattern, including concurrent habitat restoration projects in the Prado Basin that may have improved deer habitat quality along the roadway.

In addition to apparently facilitating increased underpass use for our target species across CA-71, mitigation efforts also likely reduced wildlife mortality from vehicles on the roadway. Prior to the widening and mitigation project, Lyren (2001) documented 21 coyote mortalities and 1 bobcat mortality during ca. 30 months of road-kill monitoring on CA-71 in 1997-2000; most of these mortalities were in sections along the highway where wildlife fencing was not present. After construction and mitigation, we documented 7 coyote mortalities, 1 bobcat mortality, and 1 mule deer mortality during ca. 28 months of road-kill monitoring during 2008-2010. Interestingly, as was the case before mitigation, most of these mortalities were detected in areas without wildlife fencing, including near the CA-71/CA-83 and CA-71/CA-91 interchanges. This spatial pattern of mortality suggests the wildlife fencing was effective at reducing road-kill and that additional fencing should be considered in places where it is absent. We conclude that even with the widening of the freeway, and subsequent lengthening of seven underpasses and substantial increase in traffic speed and volume, the mitigation measures along CA-71 did allow for continued movement of large mammals, and hence connectivity, across the roadway. In this case, the expansion of CA-71 demonstrates that it is feasible to include wildlife connectivity mitigation projects within existing plans to upgrade or maintain roads.

Table 2.1 Characteristics, modifications, and large mammal use for the 19 underpasses along CA-71 sampled both before (1997-2000) and after (2008-2009) construction and mitigation in the impact (with road widening and mitigation) and control (without) zones along CA-71 southeast of Los Angeles, CA. Underpasses ordered to represent spatial arrangement on landscape from north to south. Bobcat, coyote, and mule deer underpass use assessed by detections with remote cameras stationed at underpass entrances, presented as indices of number of detections divided by sampling effort (see text).

Underpass ID	Height (m)	Length (m)	Width (m)	Underpass Type	Zone[1]	Underpass Modification	Bobcat		Coyote		Mule Deer[2]	
							Before	After	Before	After	Before	After
71-24	4.57	87.00	5.79	∩	Control	None	0	0.015	0.824	1.215	0	0
71-25	1.80	64.00	2.13	■	Control	None	0	0.108	0.472	0.300		
71-27	1.80	65.00	1.80	o	Control	None	0.022	0.249	0.322	0.237		
71-01	1.20	45.75	1.80	■	Impact	Extended 5.92 m	0.089	0.137	0.172	0.396		
71-02	1.05	40.83	1.05	o	Impact	Extended 4.25 m	0.010	0.081	0.034	0.136		
71-03	1.05	42.51	1.05	o	Impact	Extended 3.5 m	0	0.052	0.008	0.160		
71-04	3.77	35.06	4.30	□	Impact	Extended 5.3 m	0.022	0.161	0.565	0.440	0	0.416
71-05	1.05	61.02	1.05	o	Impact	Extended 2.5 m	0	0.047	0.013	0.040		0.020
71-06	1.05	63.76	1.05	o	Impact	Extended 0.36 m	0.021	0.069	0.051	0.052		
71-07	1.05	60.96	1.05	o	Impact	None	0.058	0.016	0.085	0.032		
71-08	13.00	25.20	21.71	=	Impact	Replaced 2 structures	0.054	0.076	0.126	0.109	0	0.058
71-10	1.05	60.96	1.05	o	Impact	None	0.007	0.038	0.081	0.011		
71-11	1.05	46.33	1.05	o	Impact	None	0	0.074	0.053	0.095		
71-12	1.05	48.77	1.05	o	Impact	None	0.031	0.21	0	0.004		0.029
71-13	1.05	33.44	1.05	o	Impact	Extended 1.75 m	0.043	0.068	0.043	0.004		
71-14	14.00	23.30	22.39	=	Impact	Replaced 2 structures	0.049	0.092	0.093	0.088	0	0.112
71-16	1.50	121.92	1.50	o	Impact	None	0.064	0.281	0.035	0.306		0.020
71-17	1.50	112.17	1.50	o	Impact	None	0.018	0.112	0.036	0.184		
71-18	4.57	44.94	5.84	∩	Impact	None	0	0.116	0.099	0.156	0.197	0.366

[1]span bridge (=), large arch culvert (∩), concrete box culvert > 2.5 meters (□), concrete box culvert < 2.5 meters (■), reinforced concrete pipe and corrugated metal pipe culvert (o).

[2]Data listed for 5 underpasses large enough for mule deer use and thus included in the analyses with the exceptions of underpasses 71-05, 71-12, and 71-16, which were detections of approaches rather than use. See text.
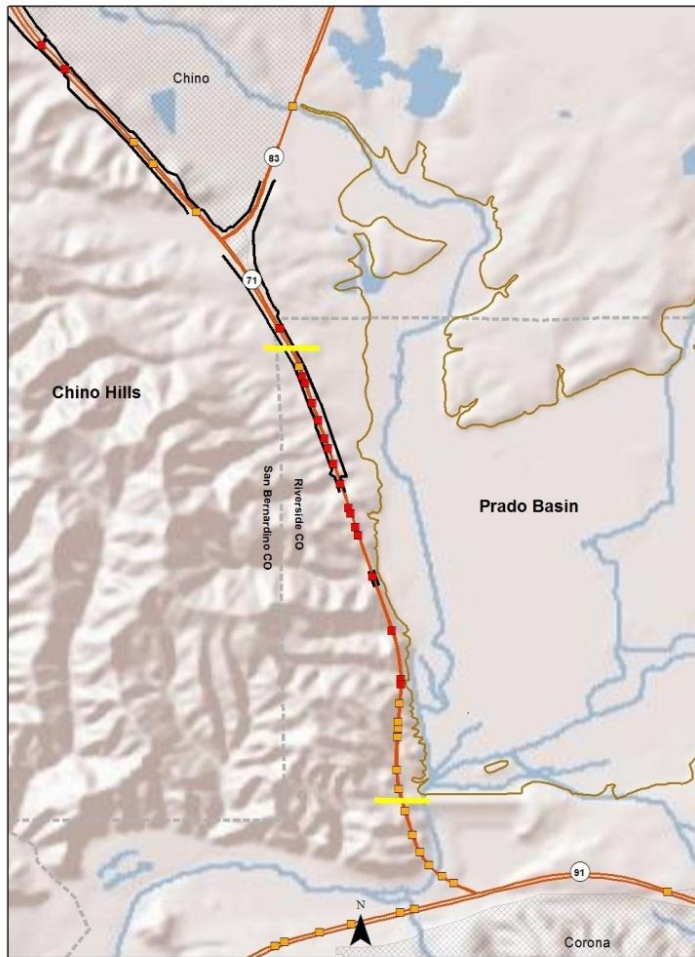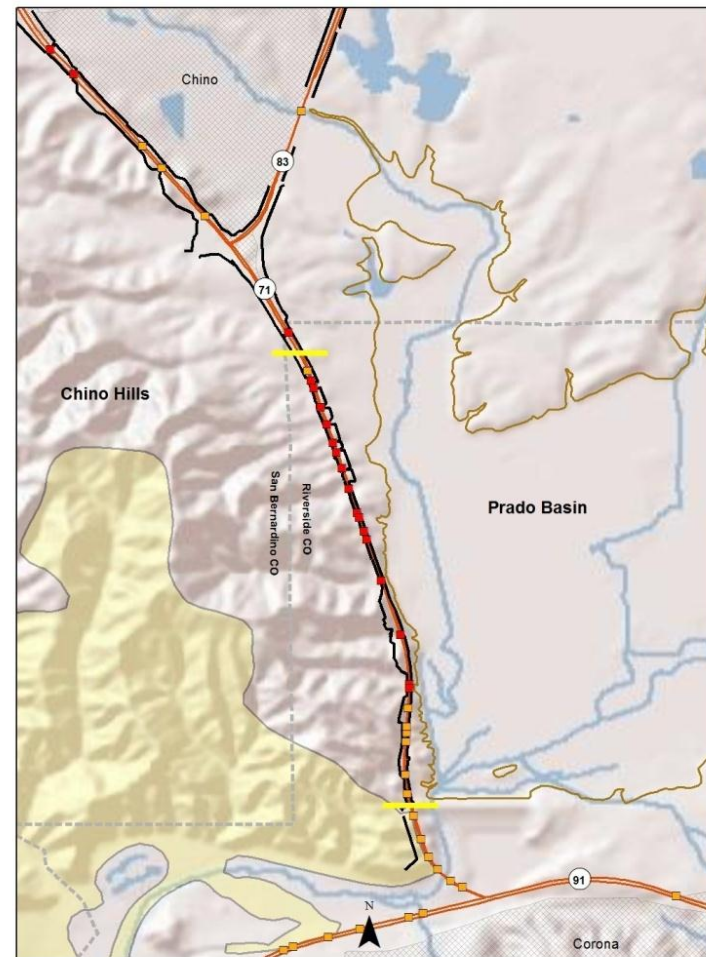
Figure 2.1 CA-71 study area showing conditions in A) 1998 before roadway construction and mitigation and B) 2008 after the construction and mitigation project. The section of road located between the horizontal yellow lines is the impact zone where the widening and mitigation project occurred. The sections of road located outside the yellow lines were considered the control zone. Dark lines next to the freeway represent the wildlife fencing that existed on the landscape. Squares represent underpasses, and those in red represent structures monitored both before and after the construction process. In B) the light yellow polygon represents the Freeway Complex wildfire (November 2008) boundary.
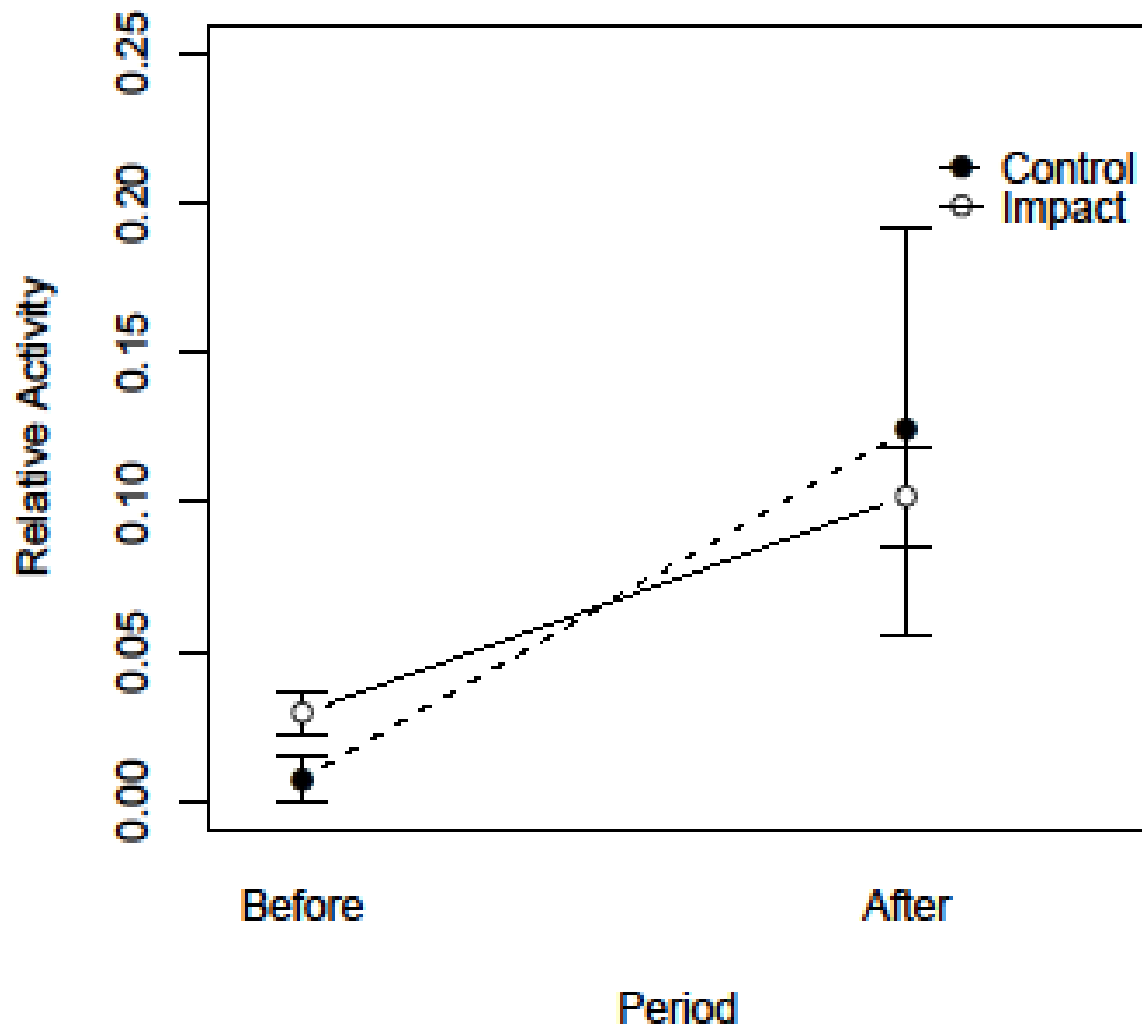
Figure 2.2 Bobcat (*Lynx rufus*) interaction plot showing an increase in underpass use in both the control and impact zones after construction and mitigation (2008-2009) compared to before construction and mitigation (1997-2000) along CA-71.
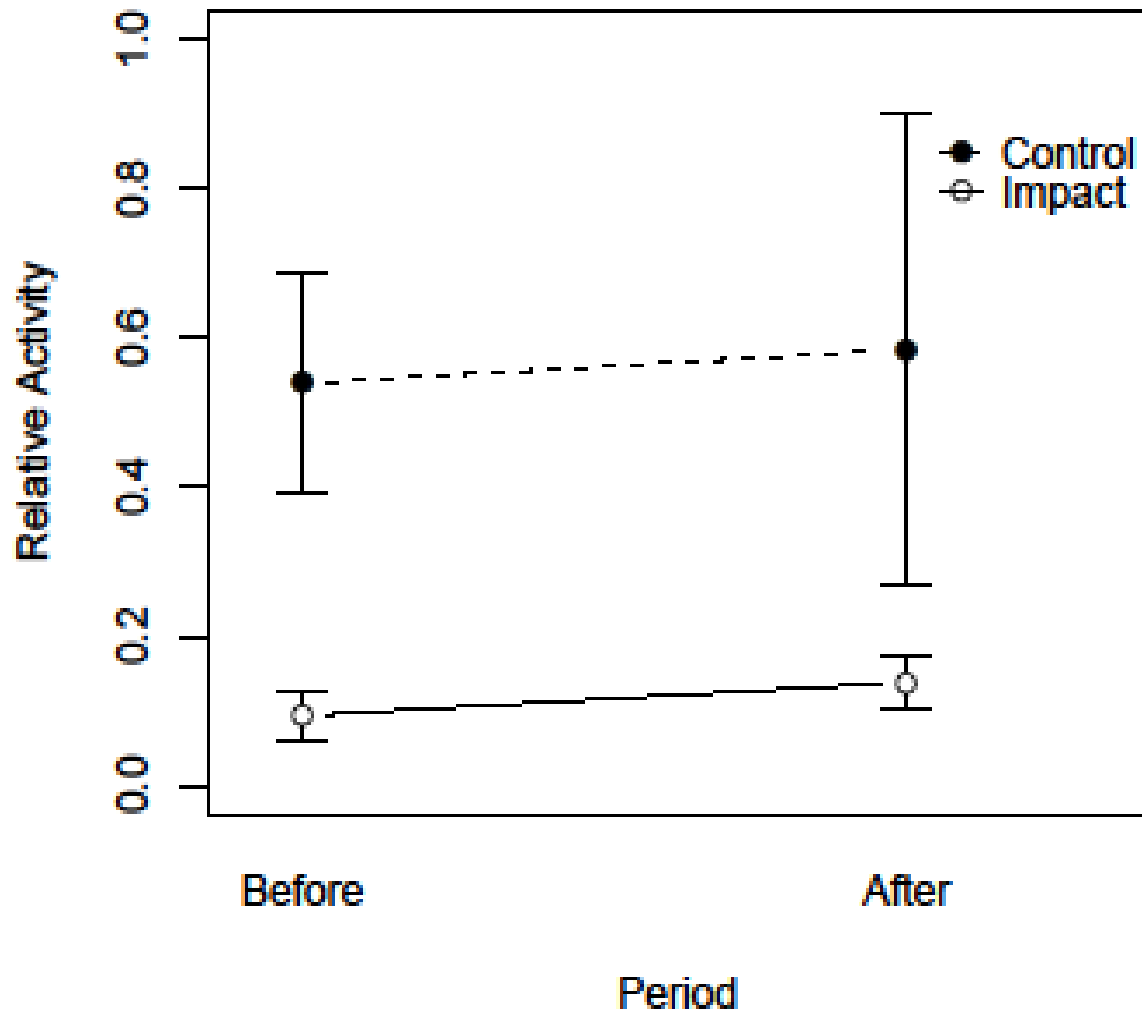
Figure 2.3 Coyote *(Canis latrans)* interaction plot showing greater underpass use in the control than in the treatment zone both before (1997-2000) and after (2008-2009) construction and mitigation along CA-71.
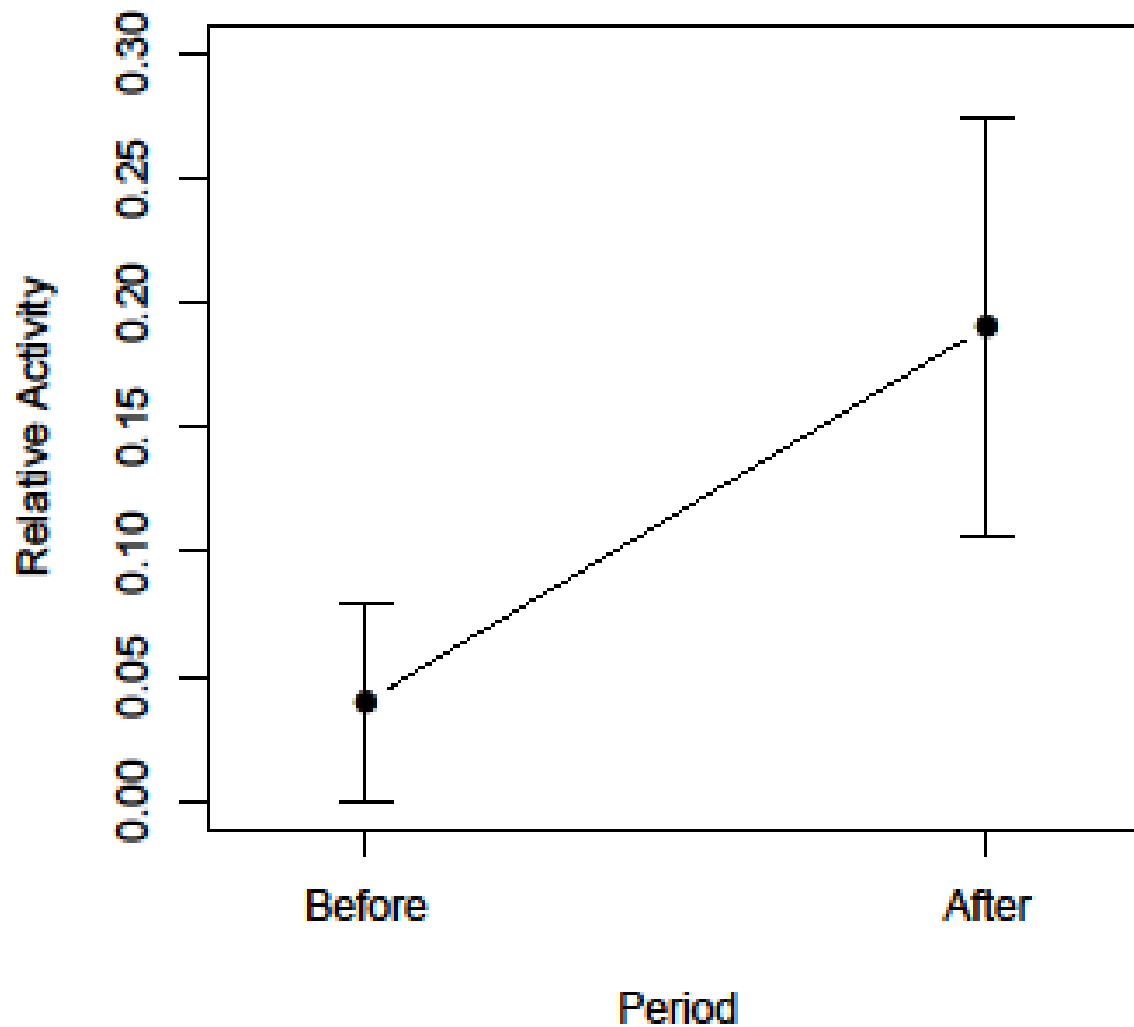
Figure 2.4 Mule deer (*Odocoileus hemionus*) before and after plot showing trend of increased underpass use after (2008-2009) construction and mitigation compared to before construction and mitigation (1997-2000) along CA-71.

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