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Highway Effects on Small Mammal Communities and Effectiveness of a Deer-Vehicle Collision Mitigation Strategy

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HIGHWAY EFFECTS ON SMALL MAMMAL COMMUNITIES AND EFFECTIVENESS OF A DEER-VEHICLE COLLISION MITIGATION STRATEGY

by

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ABSTRACT

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Effectiveness of a Deer-Vehicle Collision
Mitigation Strategy

by

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My work focused on the study of road effects and mitigation of negative impacts of roads on wildlife. Two different studies were conducted on Interstate 15, in southern Utah. My first study reported on road effects on small mammal communities. The results suggested that overall, there was no clear effect on small mammal populations relative to distance from the road. Most small mammal species did not appear to be negatively affected by the presence of the road. Instead, the road seemed to have either a neutral or a positive effect. The abundance and diversity of small mammals responded more markedly to micro-habitat than to the presence of the highway. I suggest that other factors such as water runoff during rainy periods may be responsible for the detected patterns by increasing primary productivity in areas close to the road. I conclude that roads may often provide favorable micro-habitat in the desert landscape for many small mammals and that the disturbance caused by the highway use (e.g., noise, road surface vibration) seemed to have a negligible effect on these organisms. My second study examined the effectiveness of a mitigation strategy to reduce mule deer (*Odocoileus hemionus*) road mortality. Mitigation included exclusion fencing, earthen escape ramps, and underpass crossing structures. Results comparing mortality data before and after the mitigation showed 76-96% reductions of deer-vehicle collisions. There was no evidence that the mitigation caused "end-of-the-fence" problems, i.e., higher mortality at the ends of the exclusion fencing. Results from underpass camera monitoring showed an increasing deer use of the underpasses over time. The volume of crossings recorded on new underpass structures approached the volume of crossings observed in a 20-year-old control underpass. My results suggest that human use and location of structures influenced deer use of underpasses. Overall results show that the mitigation strategy was effective and has reduced the number of deer-vehicle accidents while allowing easier wildlife movement across the landscape. I presented future maintenance recommendations to assure a long-term success for this strategy.

(86 pages)

DEDICATION

To my family in Lisbon and Logan. To all of those who have loved me back.

Silvia Rosa

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Silvia Rosa

CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1

INTRODUCTION

The continued expansion of human influence into natural areas has raised awareness of the importance of understanding and mitigating human impacts on ecosystems. Current estimates of the total landscape transformed or degraded by human influence have been estimated to be between 39-50% of total land surface of the earth (Vitousek et al. 1997). One of the most distinctive elements of human expansion on the landscape is the presence of roads. They constitute linear linkages between human populations allowing for the flow of people, resources, and information. However, natural areas crossed by road networks are heavily impacted (Forman 2000, Trombulak and Frissell 2000). The presence of the road infrastructure itself results in direct loss of habitat, realignment of watersheds, and direct mortality of wildlife (see Forman & Alexander 1998 for review). But road influences are not limited to the physical area occupied by the infrastructure; instead they extend into the adjacent landscape creating what is often referred to as a "road effect zone" (Forman and Deblinger 1998, Forman 2000). This zone, located beyond the road verge, is often characterized by high levels of disturbance (e.g., noise, vibration), altered habitat quality and hydrologic regimes, occurrence of exotic species, and a high probability of soil contamination (Forman 2000). Habitat can be affected hundreds of meters away from the verge, resulting in a "road effect zone." This is often a disproportionately large area when compared with the area physically occupied by the road *per se* (Forman 1998). In the United

States, an estimated15-20% of the total land area is ecologically affected by roads (Forman & Deblinger 1998, 2000; Forman 2000), while only 1% is physically occupied by road networks (Forman 1998). The extent to which this impacted area affects wildlife is highly variable and still uncertain in many cases. Research has documented several ecological and behavioral changes in populations close to roads, e.g., avoidance of roaded areas (Rost and Bailey 1979, Witmer and deCalesta 1985), interference with reproduction (Reijnen et al. 1995), and genetic isolation (Epps et al. 2005). But the influence of roads on organisms is not yet fully understood or satisfactorily studied. Noise and vibration are generally the most cited disturbances; but they only seem to selectively affect some wildlife (Adams and Geis 1983, Reijnen et al. 1995, Forman and Alexander 1998, Meunier et al. 1999; Goosem 2000). The barrier effect caused by roads is also considered an important influence on populations (Oxley et al. 1974, Swihart and Slade 1984, Gerlach and Musolf 2000, Epps et al. 2005), but some species are less susceptible than others depending on their mobility, tolerance to disturbance, and vulnerability to traffic collisions. The impact of roads is thus neither straightforward nor equal for all wildlife. A more accurate identification of road effects on wildlife of concern is thus essential to adjust management to real conservation needs. Furthermore, there is little information on the role that roads play on communities dynamics, e.g., on multi-species interactions. If a species avoids roads because of disturbance, it might leave free habitat for other species without competition or predation pressure (Trombulak and Frissell 2000). That would imply that habitat adjacent to the road is more suitable for generalist species (Gossem 2000). However, where habitat availability is scarce because of human development, road adjacent habitat may be the last natural refuge for wildlife, constituting premium habitat occupied by both generalists and specialists (Way 1977, Bennett 1988, Bellamy et al. 2000, Underhill and Angold 2000).

If a logical first step of research is to identify and understand the impacts of roads on ecosystems, then an effective second step is to mitigate its observed negative effects. Current research largely focuses on developing and testing effective mitigation strategies. Attention has been traditionally given to reduce wildlife mortality directly caused by vehicular traffic. This pervasive effect has received much attention from the public, not just because of its impact on wildlife populations, but primarily because of human safety concerns (Conover et al. 1995). Despite the fact that current estimates indicate that one million vertebrates are killed on the road every day in the United States (Forman and Alexander 1998), the main concern is centered on large-sized animals that can potentially cause serious accidents. North American and northern European large-sized ungulates and carnivores are generally perceived as major hazards on the roaded landscape (Bruinderink and Hazebroek 1996, Nielsen et al. 2003). However, the considerable impact that roads have on their population viability and dynamics has also concerned wildlife managers (Lehnert 1996, Seiler 2003). Conover et al. (1995) estimated that 1.5 million deer-vehicle collisions (DVCs) occur in the U.S. every year. In Utah, very conservative estimates suggest a median of 2,200 DVCs reported to authorities annually (Kassar 2005). Lehnert (1996) estimated that 5.6% to 17.4% of northern deer populations were killed by vehicle collisions every year.

The resolution of this problem has to reconcile two different perspectives: human safety and wildlife management (Sullivan and Messmer 2003).

As referred above, the science of road ecology has currently two major interests: identification of road impacts on ecosystems and mitigation of negative effects. My research addressed both fields of study: I studied road effects on small mammal communities and I tested the effectiveness of a mitigation strategy to reduce mule deer mortality on a heavily traveled Interstate highway. The following chapters describe these two studies.

In Chapter 2, titled "Impacts of Highways on Small Mammal Communities," I focused on the impact of a southern Utah highway (Interstate 15) on small mammals. I was interested to see if road disturbance affected this group, and if community composition changed at increasing distances from the road. I expected that habitat adjacent to the road would support different communities either because of disturbance, changes in habitat, or shifts in interspecific dynamics. Results are discussed in the context of the high altitude desert ecosystem of southern Utah.

In Chapter 3, entitled "Mitigation Strategy for Deer-Vehicle Collisions in Southern Utah: Evaluation of Effectiveness" I present the results of a mitigation strategy to reduce animal vehicle collisions. I monitored a 32.2 km (20 miles) stretch of I-15 where exclusion fencing, earthen escape ramps, and underpasses were constructed to reduce deer mortality on the road. I compared results of mortality before and after the intervention and analyzed whether the strategy was effective in reducing mortality. Ends of the fence and hotspot areas were analyzed for reductions in mortality Additionally, I report on mule deer use of underpass structures and compare use of new and control structures. I present migration periods, behavior crossing patterns, and changes in structure use with time. I analyze the comparative effectiveness of each crossing structure in allowing deer to cross the road. The results of this monitoring study will help evaluating the effectiveness of these types of interventions and contribute to its implementation in other problem areas.

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CHAPTER 2

IMPACTS OF HIGHWAYS ON SMALL MAMMAL COMMUNITIES

ABSTRACT

My study focused on road effects on small mammal communities in a high desert region of southern Utah. Specifically, I tested the idea that roads create adjacent zones characterized by lower habitat quality that cause lower small mammal densities. I sampled abundance of small mammals at increasing distances from Interstate 15 for 2 summers. I detected 11 genera and 13 species. My results suggested that overall, there was no clear effect on small mammal populations relative to distance from the road. The road by itself does not seem to influence abundance or diversity patterns. Most small mammal species did not appear to be negatively impacted by the presence of the road. Instead, abundance of organisms was either similar at different distances from the road or higher closer to the road. I found that 2 species were never detected near roaded areas, but their numbers were clear exceptions in the community context. I did not test other factors responsible for the distribution of small mammals, but suspect that micro-habitat features were important. The abundance and diversity of small mammals responded more markedly to habitat quality and complexity than to the presence of the highway. I suggest that water runoff during rainy periods may be responsible for the patterns we detected. In this arid environment, even sparse rains can have important effects on plant growth and structure. I conclude that roads may often provide favorable micro-habitat in the desert landscape for many small mammals

and that the disturbance caused by the highway use seems to have a negligible effect on these organisms.

INTRODUCTION

Roads present one of the greatest concerns for wildlife conservation across the earth (Forman and Alexander, 1998; Trombulak and Frissell, 2000; Jaeger et al., 2005). The most visible effect of roads on wildlife is direct mortality. However, road influences on landscapes extend much further than their physical boundaries (Reijnen et al., 1995; Forman, 2000; Forman and Deblinger, 2000; Bissonette, 2002). Several indirect effects of roads on wildlife communities have been reported such as habitat quality alteration, loss in landscape connectivity, and barrier effects (Forman et al., 2003; Jaeger et al., 2005). An effect zone of up to 100m on either side of the road has been described as causing visible impacts on ecological communities (Underhill and Anglod, 2000). Small mammal communities are good models to study such impacts. These communities are generally composed of species that use a wide variety of resources; have short generation times allowing for quick detection of environmental changes, and are permanent residents of a site responding directly to disturbance in a perceptible and measurable way (Steele et al., 1984).

Roads can impact small mammal communities by: 1) creating an edge with different habitat characteristics (Garland and Bradley, 1984; Tyser and Worley, 1992); 2) promoting the introduction of exotic species (Getz et al., 1978; Vermeulen and Opdam, 1995; Underhill and Anglod, 2000); 3) increasing stress and reducing survival (Benedict and Billeter, 2004) through disturbance and contamination (Jefferies and French, 1972; Williamson and Evans, 1972; Quarles et al., 1974); 4) blocking movement thus causing genetic barriers and home range rearrangements (Oxley et al., 1974; Garland and Bradley, 1984; Mader, 1984; Swihart and Slade, 1984; Merriam et al., 1989; Gerlach and Musolf, 2000); and finally 5) causing direct road mortality (Wilkins and Schmidly, 1980; Ashley and Robinson, 1996; Mallick et al., 1998).

While the main focus of studies on the impact of roads on small mammals has been on road barrier effects, less attention has been given to the effect of roads on density and diversity of local natural communities. Some have mentioned the importance of road verges to small mammal conservation but do not make reference to road effects on diversity or density in natural adjacent habitats (Bennett, 1988; Bellamy et al., 2000). Others have compared diversity and density between natural adjacent habitat and road verges/medians (Adams and Geis, 1983; Adams, 1984; Garland and Bradley, 1984; Meunier et al., 1999; Goosem, 2000), but do not describe community attributes in natural areas without road influences. Moreover, conclusions drawn from these studies are often based on the use of count indices instead of mathematically derived estimators of abundance or density that are corrected by capture probability estimates. In studies of this nature, capture probabilities may be radically affected at different levels of human disturbance. Animals not accustomed to human disturbance may be more prone to avoid traps than animals living in more disturbed areas, thus having a lower probability of capture. Therefore, numbers of animals captured need to be corrected by capture probability at different sites. Without correction for capture probabilities, the use of indices to estimate accurate population sizes is flawed (McKelvey and Pearson, 2001) preventing accurate conclusions about road effects.

Further analysis on the effect of roads on natural habitats is needed. My objective was to assess and compare density estimators and diversity of small mammal communities in areas influenced by roads with areas having no road influence. I tested the hypotheses that, in natural habitats, density and species diversity are the same at increasing distances from the road.

STUDY AREA

This study was conducted in the high elevation desert region of southwestern Utah. It is included in the Great Basin geographic region (Durrant, 1952; Barosh, 1960; Cronquist, 1978). The study area was located near Beaver, Utah (38°16'N latitude and 112°37'W longitude) adjacent to Interstate 15 (I-15) (Figure 2-1). Elevation in the study area ranged from 1,700 to 1,900m (Department of Natural Resources, 1978). I-15 is a 4-lane divided highway. The habitat was dominated by big sagebrush (*Artemisia tridentata*) with occasional patches of pinyon pine (*Juniperus osteosperma*) and juniper (*Pinus edulis*). The road verge was either covered by sagebrush and grass-like vegetation or non-vegetated. Weather was characteristic of high elevation Intermountain desert with below freezing temperatures and snow cover during the winter and high temperatures during the summer. Maximum temperatures rarely exceeded 100°F (38°C) and minimum temperatures were usually above -10°F (-23°C); the annual mean temperature was 47.4°F (8.6°C). Annual precipitation in the form of rain and snow was less than 12 in (305 mm), occurring primarily during winter, early spring, and late summer (Department of Natural Resources, 1978). Relative humidity was very low and evaporation potential was high (Durrant, 1952; Zeveloff and Collet, 1988). Prolonged periods of drought are frequent in this region (Durrant, 1952). The soil on trapping sites was composed mainly of fine sand deposits with occasional volcanic rocky areas (Chronic, 1990).

METHODS

Field Methodology

Small mammal sampling was conducted exclusively in sagebrush habitat on both sides of the road during the summer periods of 2004 and 2005. Trapping was conducted close to and distant from the road to sample communities with and without putative road influence. For the first year (2004), 2 trapping webs were placed on a perpendicular transect from the road at each site (Figure 2-2). The first webs were centered at 50 m from the road (Close) and the second webs centered on average 400 m from the road (Distant). Each web was composed of 8 arms extending 50 m outwards from a central point. Each arm had 6 trapping-stations (5, 10, 20, 30, 40, and 50 m) plus 1 trapping station located at the center of the web. In total, each web had a total of 98 traps. I used both lethal (snap traps) and non-lethal (Sherman) traps to maximize the number of species detected and to allow sampling during the diurnal period.

A different trapping design was used in 2005 to correct problems detected in the first trapping season. Closer webs in 2004 were thought to be sampling a considerably wide area away from the road (100m) that would probably confound presumed impacted and not impacted communities. Trapping lines were used instead in 2005. Three trapping lines were placed in a perpendicular transect from the road (Figure 2-2). Trapping lines were set at increasing distances from the road verge (0m - Close, 200m - Mid, 600m - Distant). Each line (150m) had a total of 30 traps (lethal and non-lethal). All traps were baited with a mixture of horse feed and peanut butter and checked for three consecutive mornings and intermediate afternoons (lethal traps only). Upon capture, all animals were identified, sexed, measured, marked, and released. Dead animals were removed from the study site. Trapping was conducted according to Utah State University IACUC protocol #1139 of animal welfare.

Due to differences in trapping design and areas sampled, data from 2004 are not directly comparable to data from 2005; hence data were analyzed by year. Yearly trends can be compared to assess if densities differed by proximity to the road.

Data Analysis

Diversity

I used the Shannon-Wiener diversity index (*H*) to compare community diversity at different distances from the road (Begon et al., 2006). The index was calculated for each web or trap-line in all transects. I tested diversity differences at different distance from the road using the Wilcoxon paired-sample test for 2004 data, and Friedman's test for 2005 data (Zar, 1996). A Least significance difference (LSD) multiple comparison test for Friedman's test (Sprent, 1989) was used with 2005 data to determine which pairs of distances (close vs. mid; close vs. distant; mid vs. distant) were significantly different.

Abundance and Density Estimation

Analysis for 2004 Web-based data employed a distance method described by Anderson et al. (1983) and accounted for first capture locations for each individual and their distances to the center. Program DISTANCE 4.1 (Buckland et al., 1993, 2001) was used to calculate densities and variance estimates. For analysis purpose, capture data in different transects were pooled in close webs and distant webs due to low number of animals sampled in each web. Estimation was possible for all small mammals combined (all species) and for the most abundant species (i.e., >30 captured individuals per pooled database). Additional analysis was performed for 2004 data by pooling groups of transects set in similar geographic areas (A, B, and C; see Figure 2-1). Grouping of these transects was done to account for biologically meaningful factors observed on the field (e.g., habitat and soil differences). I compared differences between areas and within areas at different distances. Density estimations in program DISTANCE were obtained by trying all available combinations of models (uniform, half-normal, hazard, and negative exponential), with adjustment terms (cosine, simple polynomial, or hermite polynomial). Final model selection was based on Akaike's Information Criterion (AIC) value and on model performance (i.e., models running without warnings). Because the amount of data was scarce, data sets were used in their entirety (i.e., no truncation was performed). Intervals used in DISTANCE (0, 7.5, 15, 25, 35, 45 m) were the midpoints between trap-stations. Resulting densities in Close and Distant Webs were tested for significant differences with a Wald test.

Analysis for 2005 trapping-line-based data was performed using a closed population mark-recapture method in Program MARK 4.3 (White and Burnham, 1999). Closure was assumed given that trapping occurred in a sufficiently brief interval and the removals were known and accounted for in the analysis (Williams et al., 2001). The Huggins Closed Capture estimator was used to obtain abundance estimates. Capture data was pooled into 3 groups representing increasing distances from the road (Close, Mid, and Distant). Estimates were obtained for the null model and other models that accounted for variability in capture probabilities due to behavior, heterogeneity, and time. Models that did not converge were discarded. Remaining models with the lowest AIC value were averaged to obtain final estimates of abundance. Differences in abundance estimates were tested using a Wald test.

RESULTS

Trapping

I completed a total of 8,406 trap nights (webs 7,056; trap-lines 1,350) and captured 484 small mammals (webs 420; trap-lines 58) comprising 13 species and 11 genera. The two species trapped most often were deer mice (*Peromyscus maniculatus*) and great-basin pocket mice (*Perognathus parvus*).

In 2004 I captured a total of 11 species (Table 2-1). Two of the species, rock squirrel (*Spermophilus variegatus*) and sagebrush vole (*Lemmiscus curtatus*), were captured exclusively in areas closer to the road, and 2 other species, pinyon mouse (*Peromyscus truei*) and white-tailed antelope squirrel (*Ammospermophilus leucurus*), were captured exclusively distant from the road. The remaining 7 species were captured at both distances. During 2005 I captured a total of 7 species (Table 2-1). Three of the species - desert cottontail (*Sylvilagus audubonii*), jackrabbit (*Lepus californicus*) and desert woodrat (*Neotoma lepida*) - were only detected closer to the road. No unique species were detected at mid or at distant classes. The number of species decreased as distance to road increased.

During the two years of sampling I noted that some species were only detected in areas with unique micro-habitat characteristics. For example, desert woodrats were only captured close to Pinyon-Juniper habitat or areas with rocky substrate; chisel-toothed kangaroo rats (*Dipodomys microps*) only were detected in the southern portion of the study area, near Beaver; cottontail rabbits and jackrabbits juveniles were only detected in road verge habitat; and rock squirrels and sagebrush voles were caught only at higher elevations in a transect with more structurally complex vegetation (area B). The transect done in area B was distinctly different from the others, not only because of its habitat features, but also because of the disproportionately high number of organisms captured (132 individuals) and the occurrence of 3 unique species.

Diversity Analysis

Results of Shannon-Wiener diversity index (H) analysis showed different trends in diversity according to different sampling years (Table 2-2). For 2004, Shannon-Wiener diversity indices were 43.2% higher in areas distant from the road (Z = -2.224, P = 0.026) as compared to results in 2005 in which diversity was 57-87% lower further from the road (Friedman test χ^2 = 6, P = 0.05).

Abundance and Density Analysis

Analysis to compare total small mammals distribution relative to road distance seems to indicate opposite trends for different years. In 2004 (Figure 2-3), I did not detect a significant difference in densities at distant and closer webs. Despite the fact that density was 28.9% higher at distant webs, there was not a significant difference (Z = -0.49, P = 0.63). In 2005, however, abundance comparisons between close, mid, and distant found lower abundance of small mammals at distant transects (Figure 2-4). An 87.3% difference between abundances at close and distant was highly significant (Z = 3.99, P < 0.001). The difference between mid and any other distance was non-significant because the low capture and recapture rates at mid resulted in less precise estimates (CV_{MID} = 0.84).

Despite the fact that trapping areas were chosen carefully to be consistent, observations in the field suggest that sites might have had relevant differences in micro-habitat characteristics. Observed differences such as volcanic rock substrate, Pinyon-Juniper proximity, higher elevation, and extensive existence of fallen trees may have influenced trapping outcomes in some transects (e.g., area B). Given this situation, the pooling of distant and close webs across the entire data set seemed inappropriate. When I pooled transects with similar characteristics (corresponding to similar geographic areas) and compared densities between areas, I was able to test if differences in habitat influenced density (Figure 2-5). Area B was located at higher elevation and had a higher abundance of fallen trees; It had higher densities of organisms both in webs near and farther from the highway compared with areas A or C. Densities at area B were significantly different from densities at area A (for both close [Z = -2.15, P = 0.03] and distant webs [Z = -3.07, P = 0.002]) and area C (for both close [Z = -2.84, P = 0.004] and distant webs [Z = -2.97, P = 0.003]).

When I compared close and distant abundances of organisms within each of the geographic areas I found no significant differences ($Z_{\text{Area A}} = 1.33$, P = 0.18; $Z_{\text{Area B}} = -1.61$, P = 0.11; $Z_{\text{Area C}} = -1.12$, P = 0.26).

For individual species *P. maniculatus* and *Perognathus parvus* (Figure 2-6), comparisons between densities close and distant failed to reject the null hypothesis, indicating no statistical difference in densities between close and distant trapping sites for either species (*P. maniculatus* Z = -1.06, P = 0.29; *Perognathus parvus* Z = 0.71, P = 0.48). However, results seem to indicate opposite trends for the two species. *P. maniculatus* density was 100.6% higher at distant webs while P*erognathus parvus* density was 31.8% lower.

DISCUSSION

The main objective of this study was to assess if roads had any zone effects on small mammal community abundance and density. The null hypothesis was that abundance and density would not vary significantly at increasing distances from the road if the road had a neutral effect. I expected effects, if any, to be constant throughout the length of the study. However, the results are contradictory in different sampling years and suggest that there is no clear effect on small mammal populations relative to distance to the road. Abundances of small mammals were similar close and distant in 2004, and higher closer to the road in 2005. Diversity was higher away from the road in 2004 and closer to the road in 2005. The road by itself does not seem to influence abundance or diversity patterns. I did not detect any negative impacts. Small mammal populations did not appear to be negatively affected by the presence of the road. Similar results have been reported in several other studies (Adams and Geis, 1983; Adams, 1984; Garland and Bradley, 1984; Meunier et al., 1999; Goosem, 2000). These studies (see Table 2-3) never report a road negative impact on total abundance or diversity. Roads appear to have either a neutral or a positive effect. A negative effect has only been reported for specific species that avoided roaded areas, but their numbers were in clear minority (see Table 2-3). My study also detected species that were never found near roads (*Peromyscus truei* and *Ammospermophilus leucurus*), but they were rare and constituted a very small proportion of the captured small mammals.

Because of the variable patterns of abundance in different years, I concluded that road effect zone by itself does not strongly influence small mammal community dynamics and patterns on the landscape. Roads may intervene in the landscape as distinctive structures causing barrier effects but do not appear to cause disturbance or habitat impoverishment for small mammals. The yearly differences in abundance and diversity recorded in my study suggest that other factors, possibly combined with road presence, may be influencing these patterns. Differences in areas sampled, sampling methods, or different trapping years, could have influenced the results. Differences between areas were clearly more important than differences between close and distant trapping sites. Results show that micro-habitat highly influenced organism abundances. Differences in sampling methods could also have influenced results. For example, trapping lines are more suitable to detect the diversity present at a site and are not as robust for estimating abundances (Stickel, 1948). Finally, yearly differences, such as variable precipitation regimes, could have influenced my results. There is a possible interaction between roads and precipitation, whereby road water runoff, a factor known to influence small mammal life cycles in desert ecosystems (Beatley, 1969), might help explain my results. In years with good precipitation, vegetation growth near roads is enhanced because roads may act as water collectors and as protection against evaporation (Huey, 1941). Increased water availability in the soil by these “linear watersheds” may influence primary productivity, increasing food and shelter availability for small mammals (Garland and Bradley, 1984), ultimately influencing their abundances and diversity (Swihart and Slade, 1990; Li et al., 2003). This scenario is likely in my study area because of the desert climate conditions. The first year of my study, 2004, registered the end of a severe multi-annual drought period in Utah. One can hypothesize that the 2004 precipitation could have easily been absorbed by the dried soils or immediately evaporated. The following year, 2005, was even wetter and precipitation would have been sufficient to result in runoff to the road verges. This may have promoted water retention in the soil, and increased primary productivity, which may have led to higher diversity and abundance of small mammals near the road. However, a multi-causal effect, such as this road vs. precipitation interaction, has not yet been tested so as to separate road effects from other environmental (spatial or temporal) factors.

If roads indeed promote water availability in desert habitats and induce higher abundance and diversity of organisms, their function may be viewed as similar to the function exerted by riparian vegetation zones. They both constitute linear components on landscape differing from the matrix by having distinct vegetation composition and higher water availability.

My results also suggest that the abundance and diversity of small mammals responds more markedly to habitat quality and complexity than to the presence of roads. The comparison of geographic areas in 2004 showed that higher densities of mammals existed with favorable habitat conditions (higher food and shelter availability in Area B than on other areas). Therefore, I suggest that management of roaded landscapes to increase small mammal populations would more profitably focus on roadside habitat improvement rather than on road disturbance mitigation.

One of the limitations of this type of study is that conclusions are based on the assumption that habitat modification induced by road presence will have the effect of altering species abundances. However, variation in abundance may not be a good indicator of habitat quality (van Horne, 1983). The number of animals present in an area depends on several other aspects other than habitat quality (e.g., source-sink dynamics; interspecific movements, and effects caused by coexistence relationships). Examples include situations where a high density of animals may occur in a low-quality habitat site due to a disturbance event, or, conversely, a low abundance of organisms may be found on good quality habitat due to competitive exclusion. Areas near roads, for example, could hypothetically be lower-quality areas (sinks) with a high density of animals because of high immigration rates from core areas (sources). In this case, the real impact of roads may not be reflected in abundance but in individual survival. According to van Horne (1983) survival should be a reliable indicator of habitat quality. Therefore, studies on small mammal survival at increasing distances from the road could provide a more reliable measure of the real impact of roads. Another inherent problem is that conclusions tend to be biased towards abundant species due to the difficulty of using statistical analysis with low abundances. This compromises the understanding of road effects on rare and probably the most sensitive species.

Despite these problems, this study supports the conclusion that the scientific predisposition to consider roads as negative landscape elements for all wildlife is not valid for small mammal communities. I documented the existence of few intolerant species, but most small mammal species in the community were indifferent or attracted to road areas. Road verges are often seen as refuges to preserve native wildlife in places where loss of natural habitat is an issue (Way, 1977; Bennett, 1988; Bellamy et al., 2000; Underhill and Angold, 2000). But even in this situation, where sagebrush habitat integrity was not a problem, immediately adjacent verge habitat seemed to be more suitable than matrix habitat beyond the right-of-way zone. Organisms may benefit from the abundance of water or even from the isolation provided by the fenced road verge from other human induced disturbances (e.g., cattle grazing, or deforestation). In conclusion, roads can create favorable micro-habitat in the desert landscape for many small mammals. The disturbance caused by the road use seems to have a negligible effect on these organisms.

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Table 2-1. Species detected at different distances from I-15 in 2004 and 2005 in southern Utah, USA. Species (number of individual captures). * = species uniquelly detected at certain distance.

DISTANCE TO ROAD	2004	2005
CLOSE	*Peromyscus maniculatus* (124) *Perognathus parvus* (39) *Tamias minimus* (27) *Dipodomys microps* (5) *Rethrodontomys megalotis* (4) *Peromyscus boylii* (3) *Neotoma lepida* (2) *Lemmiscus curtatus* (1) * *Spermophilus variegatus* (1) *	*Perognathus parvus* (12) *Peromyscus maniculatus* (10) *Dipodomys microps* (8) *Tamias minimus* (2) *Sylvilagus audubonii* (2) * *Lepus californicus* (1) * *Neotoma lepida* (1) *
MID	-	*Dipodomys microps* (11) *Perognathus parvus* (4) *Peromyscus maniculatus* (1) *Tamias minimus* (1)
DISTANT	*Peromyscus maniculatus* (120) *Perognathus parvus* (54) *Tamias minimus* (18) *Peromyscus boylii* (11) *Ammospermophilus leucurus* (4) * *Rethrodontomys megalotis* (3) *Peromyscus truei* (2) * *Neotoma lepida* (1) *Dipodomys microps* (1)	*Dipodomys microps* (2) *Perognathus parvus* (2) *Peromyscus maniculatus* (1)

Table 2-2. Comparison of small mammal diversity H (Shannon-Wiener Index) trends in 2004 and 2005 among different distances from the road in southern Utah, USA. Wilcoxon paired-ample test was used for 2004 data. Least significant difference (LSD) multiple comparisons test for Friedman test was used for 2005 data. (*) differences significant at $P < 0.05$; (NS) not significant.

YEAR	COMPARISON	TREND IN DIVERSITY	DIFFERENCE BETWEEN H ESTIMATES (%)	SIGNIFICANCE
2004	H_{close} vs. $H_{distant}$	Distant > Close	43.2%	*
2005	H_{close} vs. H_{mid}	Close > Mid	57.3%	*
	H_{close} vs. $H_{distant}$	Close > Distant	87.2%	*
	H_{mid} vs. $H_{distant}$	Mid > Distant	70.1%	NS

Table 2-3. Summary of main conclusions reported in the literature about road effects on small mammals.

STUDY	LOCATION	TOTAL ABUNDANCE/DENSITY	DIVERSITY	SPECIES REPORTED RESPONSE TO ROADS[1]	
Adams and Geis, 1983	USA (Oregon/California Illinois North/South Carolina)	Verge > adjacent habitat	Verge > adjacent habitat	*Reithrodontomys humulis*	A
				Microtus pennsylvanicus	A
				Microtus ochrogaster	A
				Sorex vagrans	A
				Microtus townsendii	A
				Peromyscus leucops	I
				Peromyscus maniculatus	I, I, R
				Ochrotomys nuttallii	R
				Microtus californicus	R
Adams, 1984	USA (North Carolina)	Median = adjacent habitat	-	-	
Garland and Bradley, 1984	USA (southern Nevada)	Verge = adjacent habitat	Verge = adjacent habitat	*Thomomys spp.*	A
				Spermophilus tereticaudus	A
				Onychomis torridus	I
				Dipodomys merriami	I
				Peromyscus eremiscus	R
Meunier et al., 1999	France	Verge = adjacent habitat	Verge ≥ adjacent habitat	*Crocidura russula*	A
				Apodemus sylvaticus	A
				Microtus arvalis	A
Goosem, 2000	Australia	-	Verge ≠ adjacent habitat	*Melomys cervinipes*	A
				Rattus spp.	R

[1]*Codes for species reported response to roads (A – Attracted; I – Indifferent; R – Repelled).*

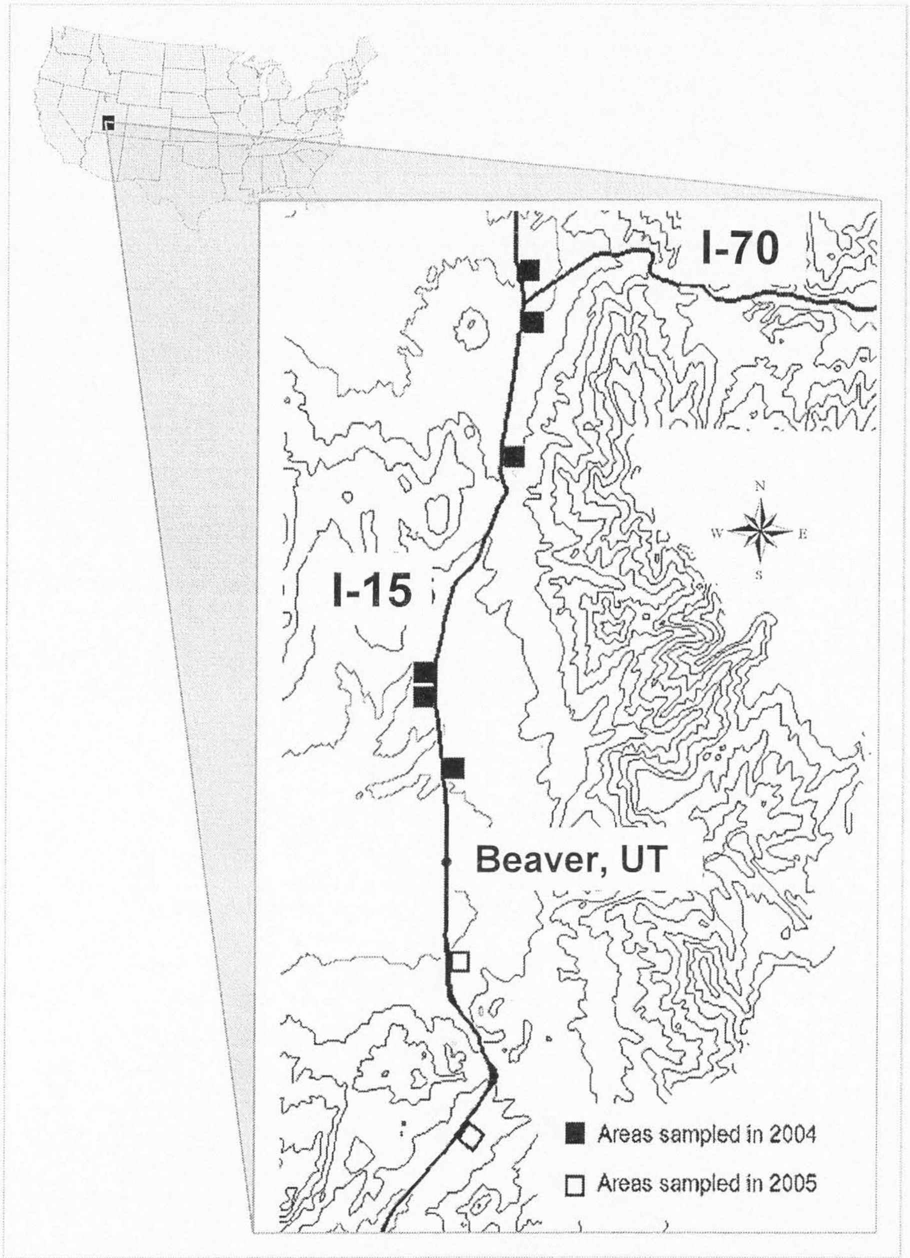

Figure 2-1. Study area map with trapping location in 2004 and 2005 and geographic areas (A, B, and C) used for comparison of densities in 2004 in southern Utah, USA.

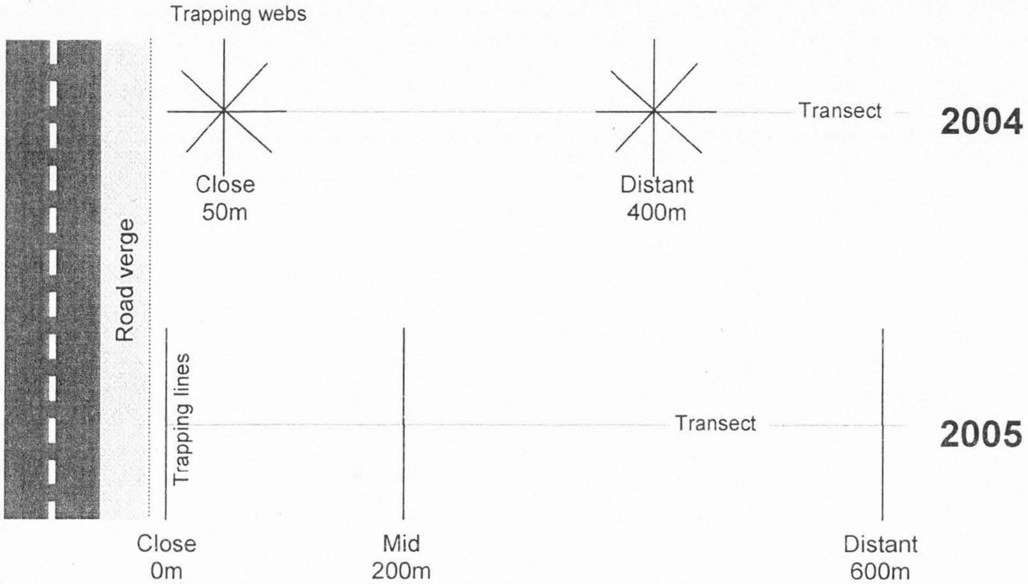

Figure 2-2. Schematic representation of trapping schemes in 2004 and 2005 used in southern Utah, USA.

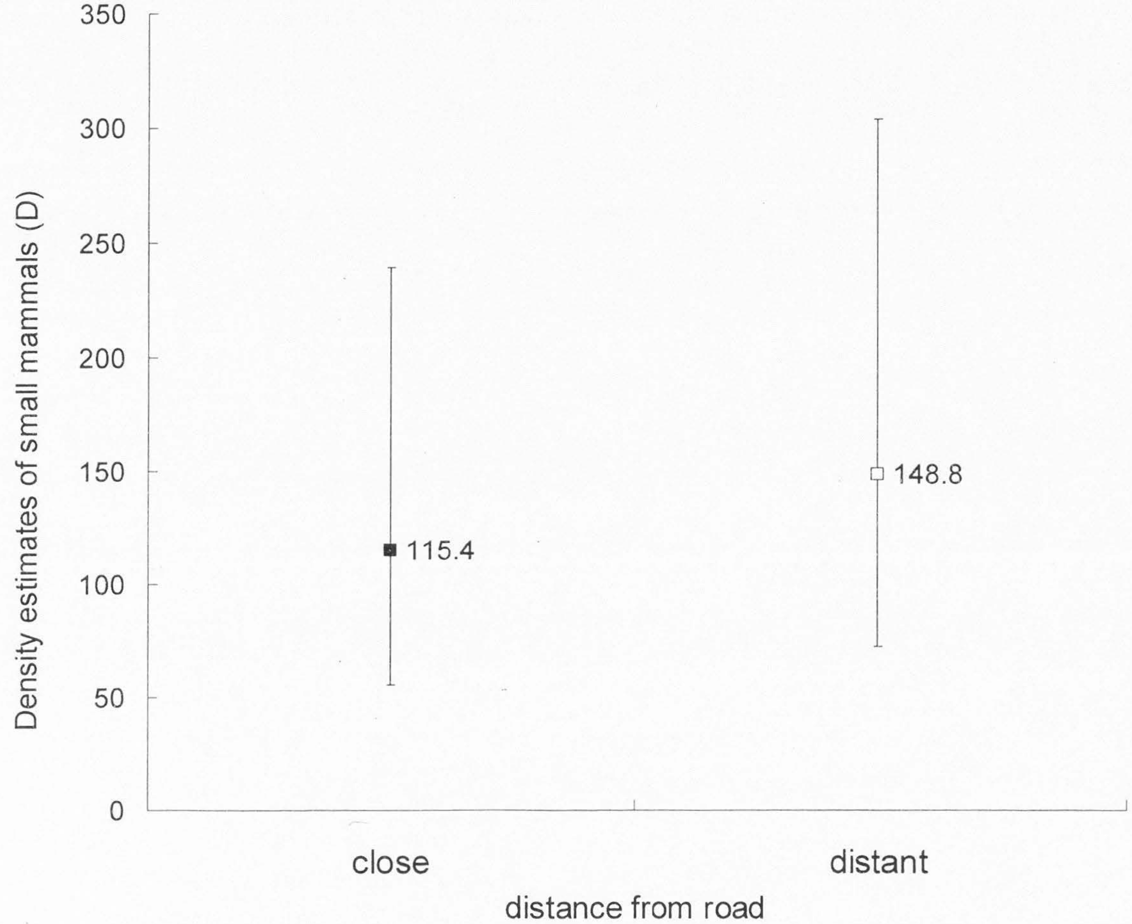

Figure 2-3. Density estimates of small mammals (and 95% Confidence Intervals) in 2004 at different distances from the road in southern Utah, USA.

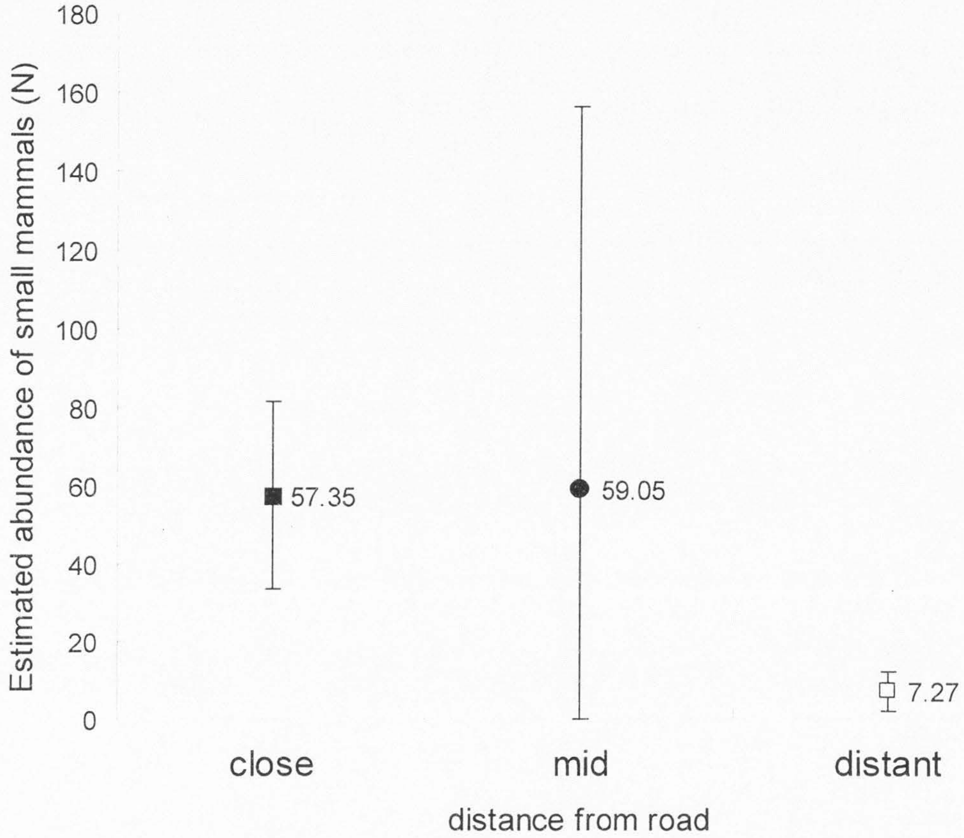

Figure 2-4. Abundance estimates of small mammals (and 95% Confidence Intervals) in 2005 at different distances from the road in southern Utah, USA.

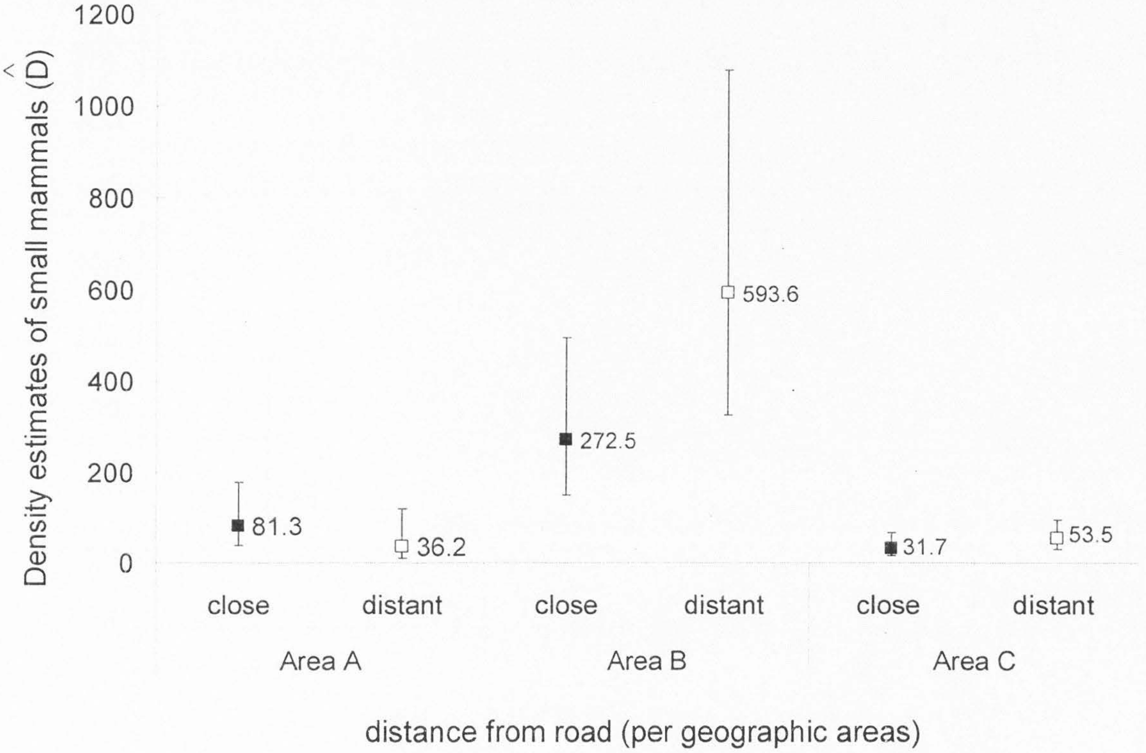

Figure 2-5. Density estimates of small mammals (and 95% Confidence Intervals) in 2004 at different distances from the road in three distinct geographic areas (A, B, C) in southern Utah, USA.

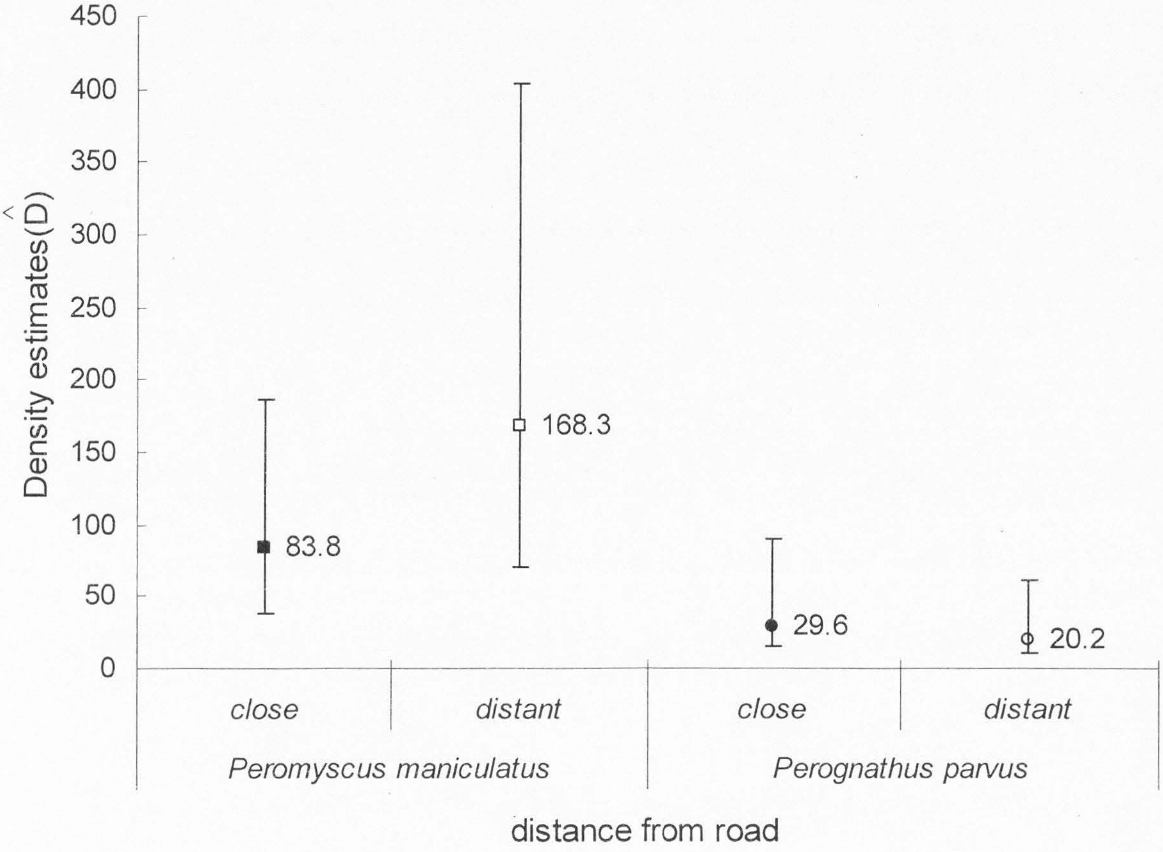

Figure 2-6. Density estimates of *Peromyscus maniculatus* and *Perognathus parvus* (and 95% Confidence Intervals) in 2004 at different distances from the road, southern Utah, USA.

CHAPTER 3

MITIGATION STRATEGY FOR DEER-VEHICLE COLLISIONS IN SOUTHERN UTAH: EVALUATION OF EFFECTIVENESS

ABSTRACT

I present the results of a study that examined the effectiveness of a mitigation strategy to reduce mule-deer (*Odocoileus hemionus*) mortality on Interstate 15 near Beaver, Utah, USA Historically, high wildlife mortality recorded in a 32.2 km (20 miles) stretch of I-15 south of its confluence with I-70 and north of the town of Beaver led to the establishment of a mitigation strategy with two major objectives: 1) a decrease wildlife-vehicle collisions, and 2) maintenance and improvement of landscape permeability that facilitates wildlife movement across the roaded landscape. The mitigation involved the construction of exclusion fencing, right-of-way escape ramps, and two underpasses designed primarily for large mammal passage. In this study, I: 1) assessed the effectiveness of the mitigation measures in reducing mule deer mortality; 2) evaluated the success of the new underpasses in allowing wildlife to cross the road safely; and 3) analyzed the "end-of-the-fence" problem, defined here as increased road mortality of mule deer at the ends of the exclusion fences. Carcass removal data for the study area were used to assess mule deer mortality changes. I compared two years of post-construction mortality data with 6 years of pre-construction mortality. Mitigation resulted in 76-96% reductions of deer-vehicle collisions. I used remotely sensed cameras to record deer passage

through the new underpasses during fall 2004, spring 2005, fall 2005, and spring 2006 migrations and compared results with a 20-year old "control" underpass structure. Results showed an initial low use of the structures during the first migration period (fall 2004), however, use increased with time, with more deer crossing in subsequent migrations (spring 2005, fall 2005, and spring 2006). Additionally, my results suggest that road noise and human use of structures interfered with deer use of underpasses. These results strongly suggest that the mitigation strategy has been effective and has reduced the number of deer-vehicle accidents while allowing easier wildlife movement across the landscape. I present future maintenance recommendations to assure a long-term success for this strategy.

INTRODUCTION

Current estimates of the total landscape transformed or degraded by human influence fall in the range of 39-50% of total land surface (Vitousek et al. 1997). One of the greatest challenges for wildlife managers today is to mitigate the negative influences of these changes on wildlife populations. Road networks have been an important contributor for this transformation and their effects on wildlife have been of growing concern because natural areas are progressively invaded by a continuously expanded and upgraded road system (Puglisi et al. 1974, Nielsen et al. 2003, Sullivan and Messmer 2003). The most evident direct effect of roads on wildlife is decreased survival due to wildlife-vehicle collisions. In the Intermountain West region of the United States, wildlife-vehicle collisions greatly affect ungulates in general and mule deer (*Odocoileus hemionus*) in particular. Nation-wide estimates of 1.5 million deer-vehicle collisions (DVCs) per year reveal the scale of the problem (Conover et al. 1995). DVCs are frequent and responsible for high costs associated with damage to vehicles, human injury and loss of life. From the animal perspective, roads create habitat fragmentation, disruption of migratory movements, as well as depletion of deer populations (Reilly and Green 1974, Lehnert et al. 1998, Sawyer et al. 2005). In Utah very conservative estimates point to an average of 2,000 DVCs reported to authorities annually (Kassar 2005) and a 5.6% to 17.4% reduction in northern deer populations due to vehicle collisions every year (Lehnert 1996), with several major roads being considered mortality hotspots (Kassar 2005).

This study focused on a 32.2 km (20 miles) stretch of Interstate 15 (I-15) in southern Utah (Figure 3-1) that historically has been considered an hotspot. In this area, deer have traditionally migrated east to summer ranges at higher elevations, and west to winter ranges at lower elevations, frequently crossing I-15. The upgrade of this road to an Interstate in the 1960s/1970s most likely blocked the traditional East to West migratory route and caused considerable deer mortality levels (B. Bonebrake, Utah Division of Wildlife Resources, personal communication). Similar situations of high levels of mortality or disruption of migration routes caused by the upgrade or realignment of roads have been described (Reilly and Green 1974, Lehnert et al. 1998, Sullivan et al. 2004, Braden 2005, Sawyer et al. 2005). The disruption of migratory routes in this study was likely due to road avoidance, and to high levels of mortality while crossing the highway. Both situations can cause a reduction of deer population sizes, either by direct kill or by lower survivorship of animals unable to reach traditional seasonal ranges. Despite the initial heavy impact on deer populations, migration was not disrupted and deer continued to use this route across the highway. As a consequence a 9.6 km (6 miles) stretch of road still recorded heavy mortality.

In 2003, in an attempt to increase driver's safety, reduce deer mortality, and provide deer access to their traditional seasonal ranges, three state agencies (Utah Division of Wildlife Resources, Utah Department of Transportation, and Bureau of Land Management) jointly created a mitigation strategy that included three integrated measures: 1) construction of deer-proof fencing; 2) installation of earthen escape-ramps; and 3) construction of wildlife underpass crossing structures. This strategy was planned to mitigate the negative impact of the highway on deer, but some of the measures were implemented to minimize adverse consequences of the mitigation itself. For example, underpasses were constructed to reduce the barrier effect intensified by fencing (Foster and Humphrey 1995, Bruinderink and Hazebroek 1996, Putman 1997, Jaeger and Fahrig 2004). Also, earthen escape ramps were constructed to allow deer to escape the fenced right-of-way (ROW). Data show that exclusion fencing is seldom if ever 100% effective, even with continued maintenance (Putman 1997). As a result, animals that enter the ROW often become trapped, increasing its probability of a collision (Reed et al. 1982). This situation is minimized by using escape-ramps (Hammer 2002) or one-way gates (Reed et al. 1974, Putman 1997).

I conducted a 2-year study following implementation of this mitigation. The overall objectives were to reduce DVCs and to assure landscape connectivity for deer across the highway. I was interested to see if fences and escape-ramps jointly reduced deer mortality on the road, and if underpasses allowed mule deer movement across the interstate.

For the mitigation strategy to be considered effective, I established 3 *a priori* criteria. First, DVCs on this stretch of I-15 needed to be reduced by 70%. This threshold was chosen according to mule deer population studies in northern Utah (Lehnert 1996), and according to reduction in mortality observed in successful mitigations strategies elsewhere (Clevenger et al. 2001, McDonald 1991). Second, there should not be an increase in DVCs at mitigation fence edges, i.e., the "end-of-the-fence problem" (Bellis and Graves 1971, Ward 1982, Clevenger et al. 2001). Third, there should be substantial underpass use by deer, which should increase with time (Ward 1982).

STUDY AREA

The study area was located on I-15 in southern Utah, between I-15 and I-70 interchange (Mile Post 132) and north of Beaver (Mile Post 112) (Figure 3-1). I-15 is a 4-lane divided pavement highway. Surrounding habitat included patches of big sagebrush (*Artemisia tridentata*), pinyon pine (*Juniperus osteosperma*) and juniper (*Pinus edulis*), agricultural fields, and urban areas.

METHODS

Mitigation Strategy Description

Mitigation construction started in spring 2004 and ended in fall 2004. A 2.44-m (8 ft) fence was erected on both sides of the road. During the summer of 2004, the highway was fenced from Mile Post 112 to 132. Additional fencing extended the northern end of the fence to Mile Post 133 during the summer of 2005 to prevent deer access to the highway intersection of I-15 with I-70. Earthen escape ramps (n = 64) were installed throughout the 32.2 km (20 miles) stretch of the study area especially around known deer crossing areas. Two wildlife specific underpasses were constructed. Underpass 1 (UP1) was located at Mile Post 126 and Underpass 2 (UP2) at Mile Post 124 (Figure 3-1). The structures were located based on prior mortality data. Both structures were oval-shaped double tunnels, made of corrugated metal, with large middle open areas. UP1 (Figure 3-2a) had an openness-ratio-score of 6.68 (6.55m (height) x 11.13m (width) / 19.82m (length)) in each tunnel section. This structure was crossed by a dirt road opened to construction and recreational traffic. UP2 (Figure 3-2b) had an openness-ratio-score of 1.62 (4.23m (height) x 8.12m (width) / 21.23m (length)) in each tunnel section and was designed solely for wildlife use. Because this crossing structure followed the topography of Wildcat Creek, the two tunnels were not aligned. The 2 new underpass structures were baited during the course of the study with hay, apples, and salt blocks to encourage deer use in early stages of underpass establishment.

Monitoring

Deer-Vehicle Collisions

To assess the joint effects of fencing and escape-ramps in reducing mule deer mortality on the road, I analyzed carcass removal counts before and after the mitigation. Data from carcass removal surveys from Utah Department of Transportation databases were available. Road carcass removal work was conducted an average of 4 times per month by contract personnel from 1998 to 2006. To distinguish between any effects due to the mitigation from the usual yearly fluctuation of road mortality, I monitored a similar control area located north of the study area (Mile Post 137 – 144) (Figure 3-1). This area had a similar mortality problem but no exclusion fencing. A BACI design (Before-After, Control-Intervention) was used to assess if variation on road mortality was due to the intervention (Eberhardt 1976, Green 1979). A drop in mortality on the treatment area would not necessarily be a consequence of the mitigation; but a higher proportional decrease of mortality when compared to a control area would reflect a successful intervention. I compared annual road-kill average for 6 years before and 2 years after. For each individual year I estimated difference in mortality counts between control and treatment areas. I estimated average of differences before and after and used a 2-independent sample t-test (Zar 1996) to test the null and alternative hypothesis. My null hypothesis was that the average change in mortality in the study area was equal to or lower than in the control area. My alternative hypothesis was that the average change in mortality was higher in the study area comparatively to control. The rejection of the null hypothesis would therefore indicate a successful mitigation strategy in reducing mortality. I was constrained to assume that difference data was normally distributed due to the difficulty of testing normality with the limited amount of data available (n = 8 years). T-test results were obtained according to Levene's Test for Equality of Variances. I compared before and after mortality by year, fall migrations (October-January), and spring migrations (April-July).

I used a t-test to assess whether the mitigation was effective in reducing mortality at the hotspot area (MP 120-126) by comparing annual deer mortality averages before and after the mitigation.

To test if end-of-the-fence problems existed I compared annual deer mortality averages before and after the mitigation at the northern (MP 131-134), and southern (MP 111-113) ends of the fence by using a t-test as described above. I accounted for mortality that occurred within 2.4 km (1.5 miles) on either side of the fence for a total of 4.8 km (3 miles). At the northern end of the fence, however, I analyzed a 6.4 km (4 miles) section of road because additional fencing extended the northern end of the fence by 1.6 km (1 mile) during 2005.

Underpass Use

I monitored a 20-yr-old control underpass (Control UP) to compare mule deer use between new and established structures. The Control UP was located south of the study area (Mile Post 103) in a similar mule deer migration area (Figure 3-1). The Control UP (Figure 3-2c) was composed of two double-span bridges with an openness-ratio-score of 4.43 each (4.12m (height) x 21.49m (width) / 20m (length)) and a large median area. This structure was also designed exclusively for wildlife, and mule deer use had been previously reported. Exclusion fencing was also present in the control area.

To record animal crossings, each underpass (UP1, UP2, and Control) was equipped with a Reconyx® camera (digital, triggered by motion and heat, with infrared illumination). All cameras were installed facing north, inside the median of each underpass. Camera placement was chosen to assure approximately equal photo capture probabilities in all the structures. Cameras were camouflaged and mounted inside urban electric boxes to reduce the probability of damage or theft. Cameras were equipped with 512Mb memory cards and checked an average of twice a month from October 2004 to August 2006. Cameras were set with maximum sensitivity, a 2-second lag between triggers, and took 1 picture per trigger. I sampled four migration periods (fall 04; spring 05; fall 05; spring 06).

I used camera data to 1) characterize overall use of the structures and 2) to estimate deer crossing volume and temporal variation.

To characterize overall use of the structures I categorized photos into classes (mule deer, humans, cattle, other wildlife, and blank) and computed percent use for all classes in each underpass by dividing the number of photos in each category by the total number of photos collected. Using a χ^2 test of homogeneity of proportions, I evaluated whether proportions of each class were similar in every underpass (Zar 1996).

Photos taken of mule deer were used to estimate deer volume use and changes with time. Because all cameras were fully functional for >90% of the monitoring period (678 days total) the results resemble census data more than sample data. Camera data provide a continuous monitoring of deer passage, i.e., a census, rather than a replication of independent samples. As a result, null hypothesis and significance testing have no theoretical interpretation (Berger 1985, Gill 2001). Thus, analysis of the volume of underpass use and changes with time are descriptive, using summary statistics.

Generally, mule deer are not individually identifiable by unique external characteristics (e.g., pelage markings). This impedes the estimation of the exact number of different individuals using the structures and the frequency at which the same individuals crossed. Because of this constraint, instead of counting the number of animals that used the structure, I counted the total number of crossings detected. I also noted direction of crossings: west, to winter ranges, and east, to summer ranges. Net number of crossings was inferred from the difference between crossings in each direction (west – east). By convention, net crossings towards the west were represented by positive values, and net crossings towards the east by negative values. I used net crossings to monitor changes in flux in either direction through time. This allowed the detection of migration periods, as well as changes in volume use of new underpasses. Crossing data were also used to identify different types of movement exhibited by deer.

RESULTS

Deer-Vehicle Collisions

The BACI analysis of carcass removal data indicates that reduction in DVCs was due to the mitigation and not to stochastic annual oscillations in mortality in the study area. I documented a significant decrease in annual DVC levels (t = 4.244, P = 0.004) that corresponded to a 77% reduction in mortality after the mitigation (Figure 3-3a). spring DVC levels were also significantly decreased (t = 2.903, P = 0.027) corresponding to a 96% mitigation-induced reduction in mortality (Figure 3-3b). Finally, fall DVC levels were equally significantly reduced (t = 2.463, P = 0.049) to levels that correspond to 76% of the original mortality (Figure 3-3c).

Analysis of mortality at the hotspot stretch suggests a reduction of mortality of 66% with average of DVCs before and after significantly different (t = 2.809, P = 0.023). Further results show that mitigation measures did not promote an increase of mortality at the ends of the fence. I observed lower levels of mortality at the northern end (t = 2.831, P = 0.022); and equal levels of mortality in the southern end (t = 1.274, P = 0.238); thus DVC levels were not higher at either end of the fence.

Underpass Use

There were considerable differences in underpass use between the three structures. From a total of 48,483 pictures (UP1: 18,829; UP2: 14,421; Control:

14,509) I noted similarities between UP2 and the Control UP, and a different pattern of use in UP1 (Figure 3-4).

UP1 registered the highest levels of human use, differing significantly from UP2 ($\chi^2 = 7910$, $P < 0.001$) and the Control ($\chi^2 = 8010$, $P < 0.001$). UP1 also had the lowest number of mule deer detections both in absolute and proportional terms when compared to UP2 ($\chi^2 = 5238$, $P < 0.001$) and to the Control ($\chi^2 = 1782$, $P < 0.001$).

In UP2 and the Control, I recorded a higher proportion of deer use and frequently detected other wildlife (e.g., coyotes, cottontail rabbits, and birds). Elk were only detected in the Control UP. UP2 and the Control differed in proportion of cattle ($\chi^2 = 1687$, $P < 0.001$) and deer ($\chi^2 = 906$, $P < 0.001$). Deer and cattle used the Control UP simultaneously several times. Occasionally, deer and elk were detected using the structure at the same time.

Deer exhibited similar crossing behavior in all the structures. Animals would either enter the structure to cross the road in a direct movement (i.e., without turning in the opposite direction), or they would remain in the proximity or interior of the structure crossing several times in either direction. In the new structures, some photos showed active use of bait, with deer groups frequently spending considerable amount of time feeding. In the Control UP, direct movements were generally the rule, but water and salt accumulations under the road often caused deer to remain inside the structure for some time. Photos also documented deer startling behavior when inside the structures as a reaction to traffic.

I identified two different types of deer movement by plotting the monthly number of crossings in east and west directions (Figure 3-5). During certain periods, deer exhibited daily movements, crossing in equal numbers east and west. During other periods, deer exhibited seasonal migratory movements, crossings disproportionately more in one particular direction. For example, in the Control UP (Figure 3-5a), I documented that in some periods of the year (e.g., December through March), deer displayed approximately the same number of crossings in each direction. However, during migration periods (namely, October or May), the number of crossings was much more pronounced in one particular direction. When I analyzed UP1 (Figure 3-5b) and UP2 (Figure 3-5c) I found that migratory periods were not as evident in either of the new structures. These underpasses showed mostly daily movements all year long, possibly because of the presence of bait in the structures. The flux of animals in one predominant direction at migratory times was not as obvious. Migration could only be assumed by the increase in deer activity (i.e., a higher number of crossings) in the new underpasses at migration periods. However, UP2 showed some evidence of spring migratory movements in May (2005 and 2006).

Results from the analysis of net crossings (Figure 3-6) allowed us to identify four migratory movements: fall 04 (October – November 2004), spring 05 (April – June 2005), fall 05 (September – November 2005), and spring 06 (April – June 2006). Migratory activity was not detected through UP1 and UP2 during the first migration monitored. However, subsequent migrations did occur through the new crossing structures and followed temporal patterns of the Control UP. The volume of crossings during spring migrations was generally higher than fall migrations, but fall migrations extended longer in time, with some migration movements occurring in later winter months (e.g., January – March).

Finally, net crossings during migration months indicated increasing use with time in UP1 and UP2 (Figure 3-7). Results show that the volume of use in the new structures gradually approximated the Control UP. For example, during the first migration period (fall 04), UP1 only registered 12.6% of the movement observed in the Control, whereas during the last migration sampled (spring 06) crossings increased to 33%. Similarly, UP2 increased from 5.9% in the first migration to 71.7% in the last migration, nearly matching movement volume of the Control UP.

DISCUSSION

The results suggest that the mitigation strategy was effective. The three *a priori* established criteria were met. DVC levels were significantly and satisfactorily reduced in the study area; the strategy did not create end-of-the-fence problems; and underpasses were heavily used by deer.

Annual mortality in the study area was reduced by 77%. This reduction was limited by results in 2004. High mortality levels registered were due to periods when mitigation was either not yet implemented or fully operational. Therefore 2004 can be considered a transition year. I observed a 76% reduction in mortality during fall migrations and a 96% reduction during spring migrations. The difference between the two periods was due to the influence of higher road mortality recorded during the first fall migration sampled in the 2004 transition year. During that period, fences had several gaps that were easily exploited by deer to gain access to the road as frequently reported in other studies (Falk et al. 1978, Feldhamer et al. 1986). Until fencing problems were detected and repaired (shortly after the first migration), deer continue to access the ROW. Fencing problems were more influential in the hotspot area, where I recorded a mortality reduction of only 66%.

The impact on deer population of the observed reduction in mortality cannot be fully understood with my results. Population dynamics data are needed to estimate how much reduction in mortality is required to reverse population declines or improve survivorship. According to Lehnert (1996), a 60-100% reduction in road kill was sufficient to stop the decline in mule deer population in central Utah. However, population information for the study area is scarce and insufficient to define a biologically meaningful reduction threshold. I argue that the *a priori* selected 70% reduction threshold is a justifiable goal for this area. I expect that 70% less mortality on the road has a high probability of translating in a significant increment in survival and a high chance of reversing declining local population trends. The study area overall results of 76-96% are satisfactory when compared to mortality reduction results published in the literature. For example, Clevenger et al. (2001) reported reductions of 80% in levels of ungulate-vehicle collisions in Banff National Park; Braden (2005) reported a reduction of 83-92% in Key deer-vehicle collisions in Florida; and McDonald (1991) described a 70% reduction in moose mortality in Alaska.

The results also confirm that the strategy did not cause an end-of-the-fence problem, suggesting that deer that crossed the road used the new available underpasses. To my knowledge, this was the only study where mitigation exclusion fencing did not cause end-of-the-fence problems (Bellis and Graves 1971, Reed et al. 1975, Clevenger et al. 2001, Braden 2005). I argue that the fences extended far enough from deer kill hotspot areas (11.3 km (7 miles) north and 19.3 km (12 miles) south of the underpasses) that deer were discouraged from moving around the end of the fence and encouraged to use the underpasses across the highway. I also observed evidences of deer using 3 other non-wildlife specific structures along the fenced area. Wildlife use of non-wildlife specific underpasses has often been reported in the literature (e.g., Ng et al. 2004, Krawchuk et al. 2005).

Overall use of the underpasses shows that these structures were appropriate for deer. UP2 registered the majority of deer use, probably due to topography (alignment with Wildcat Creek), location (traditional migratory pathway for deer), and virtually no human disturbance. Despite its unusual configuration, deer seemed to easily adapt to the structure, almost matching the volume of crossings in the Control UP later in the study. The short length of its tunnels appeared to facilitate crossings.

I was surprised that UP1 was less used than UP2. I expected a higher use than observed because of its greater openness ratio, and straight alignment. However, there were high levels of vehicular traffic, mostly related to recreational activities. Additionally, UP2 was located centrally at the hotspot area, and probably collected the majority of the migratory animals. However, deer did continue to use UP1 in considerable numbers.

Elk were never recorded in UP1 or UP2 even though they occurred in the area. Some studies suggested that elk avoid roaded areas more than deer (Rost and Bailey 1979, Witmer and deCalesta 1985, Rowland et al. 2005, Wisdom et al. 2005), and need longer periods of time to habituate to roads (Lyon 1983). These factors may explain why elk used the Control UP and not the new structures. Another potentially important factor may involve differences in the types of vegetation surrounding the Control (Pinyon-Juniper on both sides) and new structures (Pinyon-Juniper on just one side). Elk may be more prone to move through denser vegetation cover (Rost and Bailey 1979). I expect that with learning and habituation, these underpasses may be potentially usable by elk in the future.

In 2004, migration immediately following construction was very low and was probably delayed by reluctance to use the new crossing structures. Hesitation behavior has been reported for similar circumstances by Reed et al. (1975) and Ward (1982) in early and late stages of underpass use. Hesitation and avoidance are frequent wildlife reactions to recently-built human structures (Ward 1982, Merrill et al. 1994, Sawyer et al. 2005). However, subsequent movements through the new structures increased over time, suggesting a gradual learning process (Putman 1997). I also documented that deer use of the new structures approximated over time that of the use of the Control UP, which is also an evidence of gradual learning.

Two different types of movement were identified in this study. I recorded both daily and migratory movements. Daily movements occurred yearlong in the new structures, probably because of the presence of bait inside the structures. However, daily movements were also recorded in the Control UP outside of migration periods. Deer apparently forage near roads all year long potentially increasing the risk of accidents. DVCs related to daily movements have been reported to cause the highest impact on population size (van Langevelde and Jaarsma 2004), highlighting the importance of focusing on these types of movements for population management. A total of 4,658 crossing events were observed in new underpasses and thus prevented from occurring over the road.

Deer usually follow similar migratory routes to reach the same seasonal ranges (Garrot et al. 1987, Kucera 1992). Deviation from their traditional migratory routes through an underpass structure may be challenging for the animals and involves gradual learning. With the mitigation strategy fully operational, deer population should benefit by an increased probability of reaching their traditional winter and summer ranges without being killed on the road, thus having higher survival probability and increased fitness. I expect that the number of animals accessing seasonal ranges through the underpasses will increase with time, firmly establishing migratory routes through the mitigation structures.

Even though I considered this strategy effective in the first years following mitigation, I was unable to assess its real impact on the deer migratory population. It was beyond the scope of this study to obtain estimates of the proportion of the total migratory population that actually crossed the highway, or estimates of the proportion of deer that may have been blocked by the mitigation (Reed et al. 1975). These data are needed to fully evaluate the extent to which this population may be impacted by the mitigation (Hardy et al. 2003) and are desirable in future research. The available results, however, strongly suggest that exclusion fencing and escape ramps, combined with wildlife underpasses, were effective in reducing DVCs and maintaining landscape connectivity for mule deer.

CONCLUSIONS AND FUTURE CONSIDERATIONS

This mitigation action on southern Utah is a good example of effective cooperation between state agencies in mitigating human wildlife conflicts. Often Transportation and Wildlife agencies have divergent perceptions and solutions for this deer-vehicle collision problem (see Sullivan and Messmer 2003 for review) and do not communicate or cooperate to obtain combined solutions. Transportation agencies are primarily concerned with human safety while the goal of wildlife agencies is mainly wildlife management. It is clear that mitigation is less likely to solve the problem when it only accounts for traffic issues and disregards wildlife concerns (Foster and Humphrey 1995). Integrated mitigation approaches, such as the one described here, confer better results by allying scientific wildlife expertise with transportation engineering solutions. Often agency objectives can be met, but intelligent and authentic compromise is required. In this study, if mitigation remains effective through time, each agency involved will have accomplished their objectives, namely, increased driver safety, landscape connectivity, and higher deer population survival rates.

An integrated solution, however, does not liberate agencies from their long-term responsibilities in maintaining the strategy effective. Clevenger (2005) and Clevenger and Waltho (2005) clearly stated that mitigation for wildlife mortality on highways is a long-term process. Agency responsibilities do not cease with the implementation of mitigation measures but extend to long-term maintenance and assessment of effectiveness. Damaged fencing and non-functional crossing structures are frequent problems in years following mitigation (Falk et al. 1978, Hammer 2002). Reed et al. (1982) estimated that fence maintenance costs are about 1% of the initial fence cost. If long-term maintenance is planned in the early stages of a project, this would facilitate maintenance of project effectiveness. I expect that fencing at the study site will require regular maintenance in order to keep animals from accessing the ROW. Annual maintenance checks will help keep the mitigation effective.

Additionally, the use of underpasses should be monitored regularly, to assure that deer continue to use the structures. If a marked decrease in use is observed in the future, bait should be utilized to attract deer; but it should be used in low quantity and discontinuously in time. This will prevent the risk of artificial feeding habituation and avoid the association between structures and feeding areas. Deer should perceive underpasses as crossing corridors rather than as feeding sites.

Continued wildlife movement through the underpasses may also be facilitated if some degree of protection was conferred to underpass areas. Underpasses should be considered migration bottlenecks, as described by Sawyer et al. (2005), i.e., areas along migration routes that confine animal movements to narrow or limited regions. Continued evaluation and management of underpass areas is thus especially important and should aim to protect migratory deer from human disturbances, including both recreational and hunting activities.

This study documented a successful mitigation strategy where deer highway mortality was reduced without blocking migratory routes or generating end-of-the-fence problems. Reflection on the future of this strategy should be the next step for state agencies. Special attention should be given to fencing maintenance and to underpass use. State agencies should set joint goals to assure that this strategy constitutes a long-term solution for I-15 in southern Utah.

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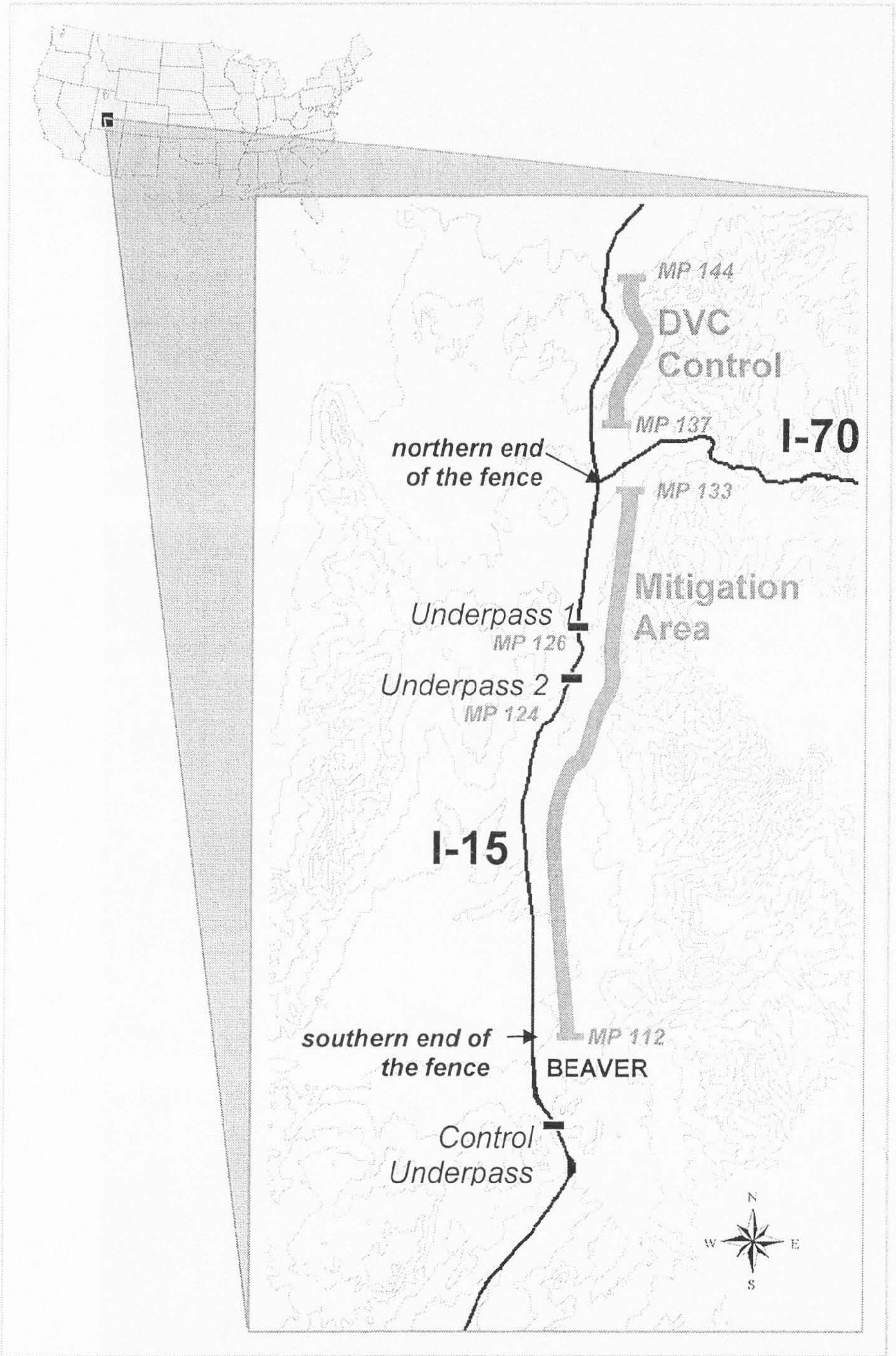

Figure 3-1. Study area location in southern Utah, USA. Location of deer-vehicle collision (DVC) control area, mitigation area, monitored underpasses, and fence extension in the study area.

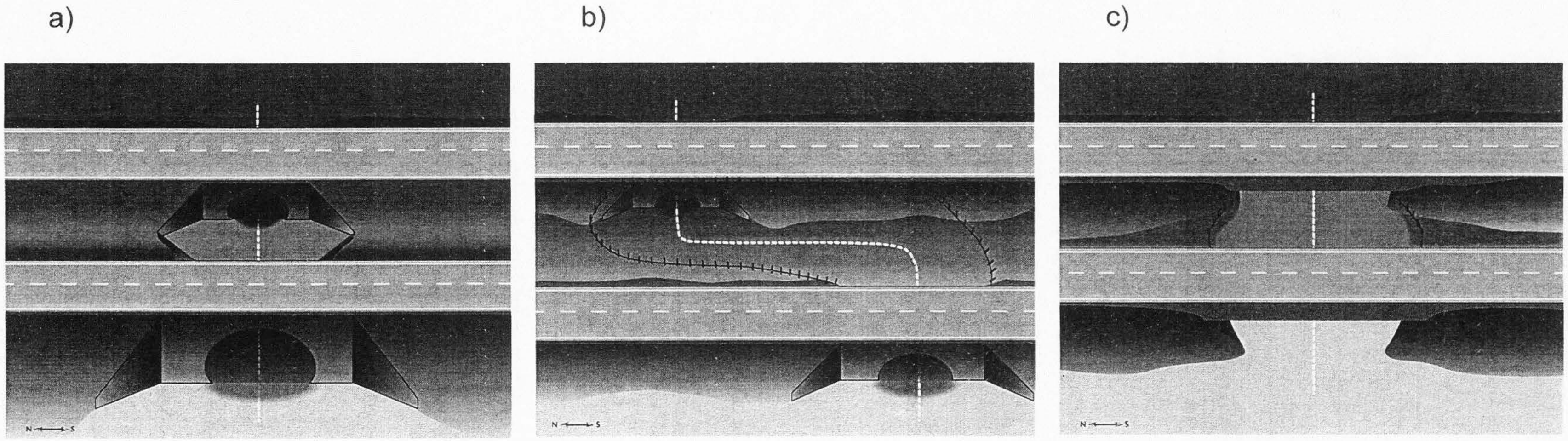

Figure 3-2. Schematic representation of underpasses monitored in the study area on I-15, southern Utah, USA.

a) Underpass 1; b) Underpass 2, c) Control Underpass. Images ©Diana Marques (www.dianamarques.com).

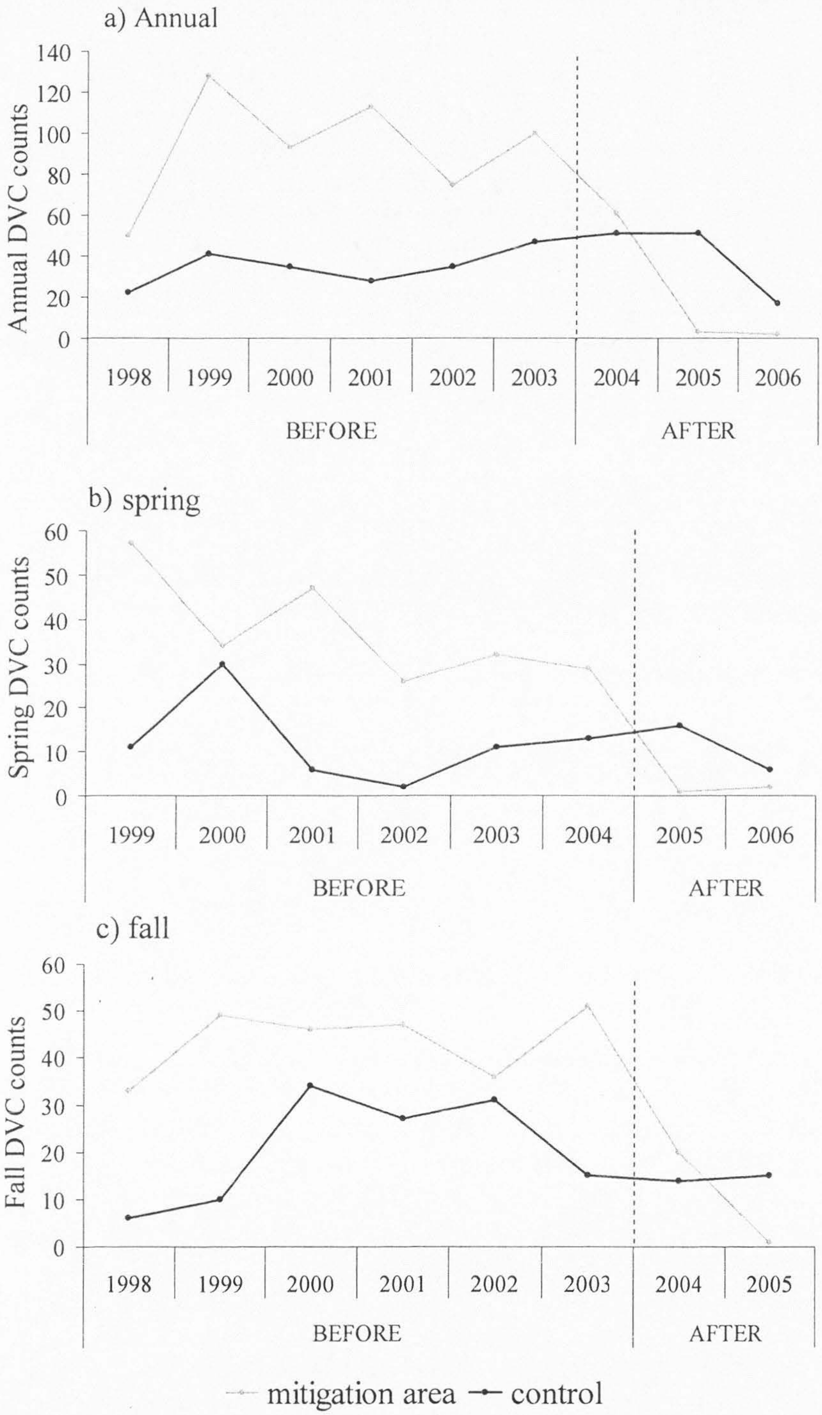

Figure 3-3. Deer vehicle collision counts in control and mitigated areas. Dashed line represents mitigation strategy implementation. Data from UDOT deer carcass removals on I-15 in southern Utah, USA.

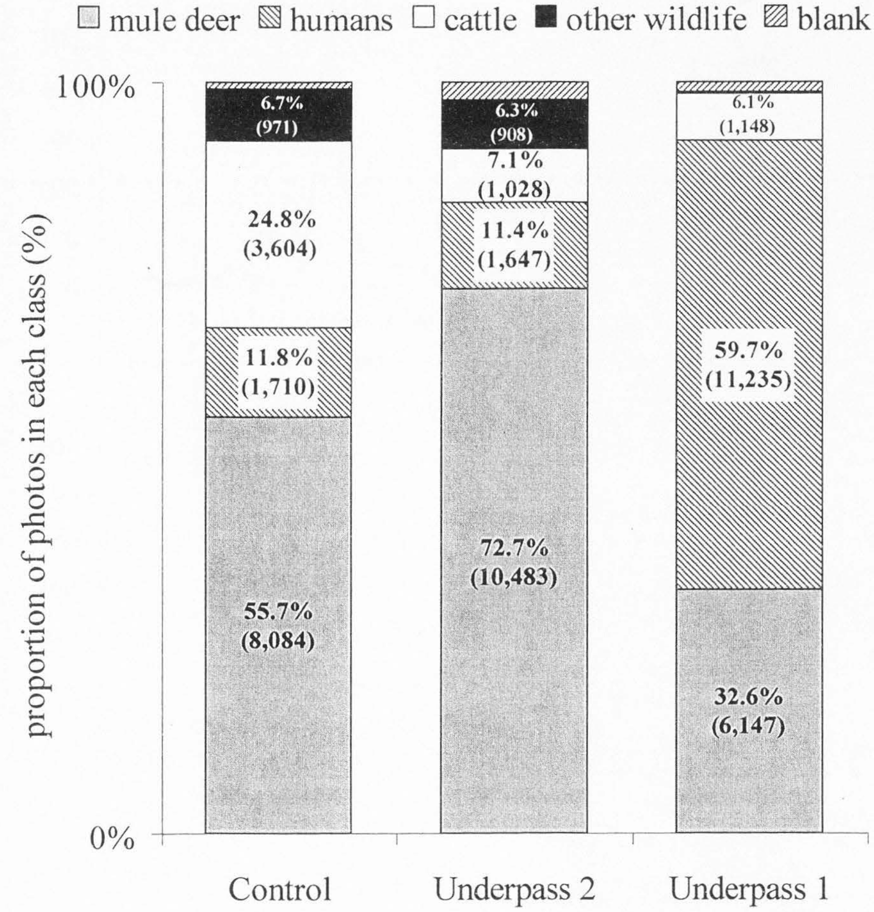

Figure 3-4. Comparison of deer use of monitored underpasses on I-15, in southern Utah, USA. Proportional values (%) and absolute values are presented. Photos grouped in: mule deer use, human use, cattle use, other wildlife use, and blank photos.

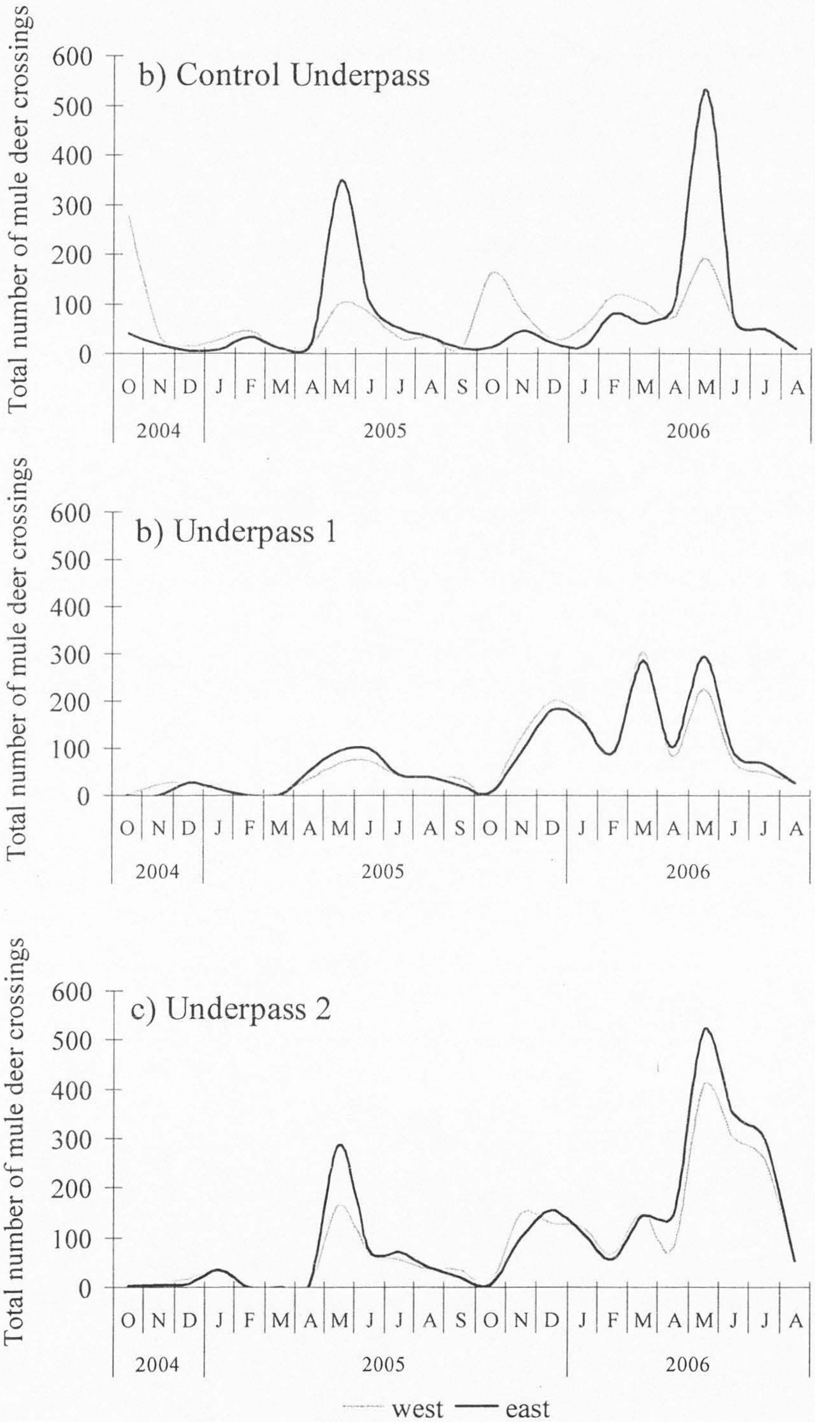

Figure 3-5. Monthly counts of deer crossings detected in both directions (east and west) on monitored underpasses on I-15, southern Utah, USA.

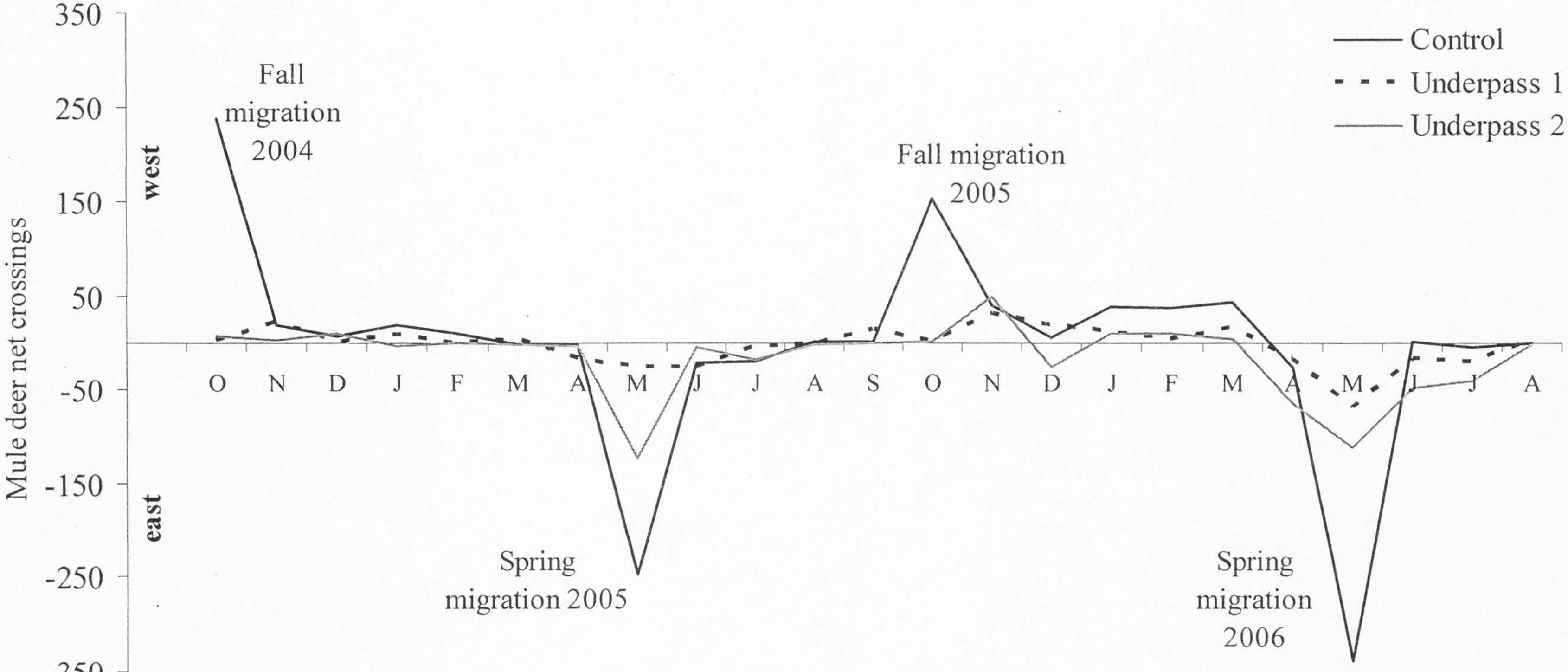

Figure 3-6. Net crossings through time for each underpass on I-15, in southern Utah, USA. Positive values represent movement towards the west; negative values represent movement towards the east.

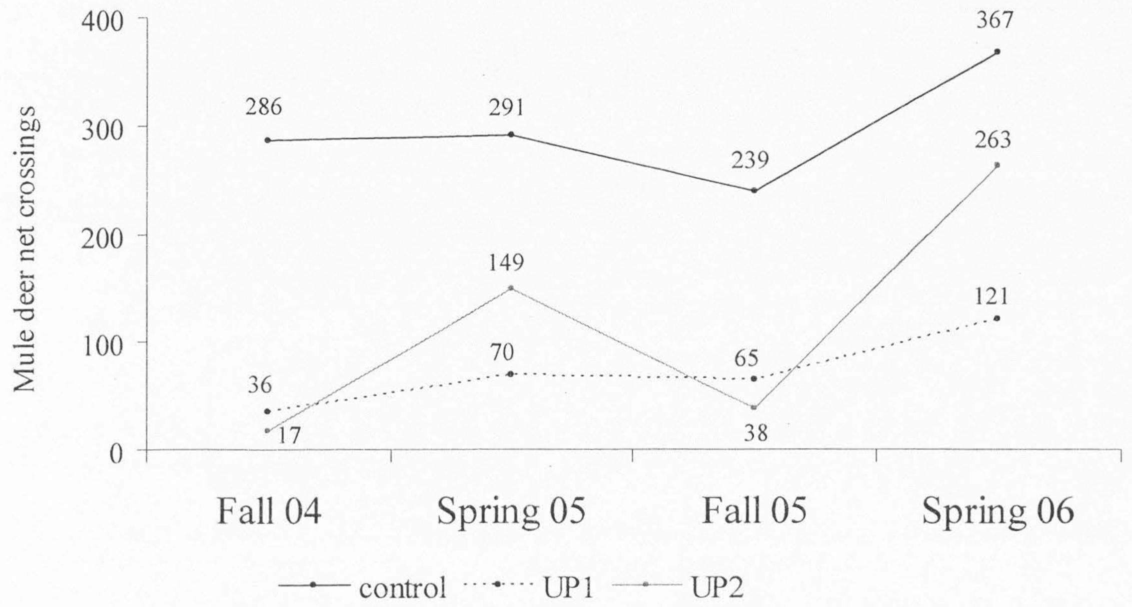

Figure 3-7. Net crossings in monitored migration periods (fall: October-January; spring: April-July) for each underpass on I-15, southern Utah, USA.

CHAPTER 4

CONCLUSION

One major focus of the developing science of Road Ecology is the study of impacts of roads on wildlife. The understanding of the impact of roads on wildlife has motivated extensive research with the general conclusion that natural areas crossed by road networks are heavily ecologically impacted and that those impacts are negative (Forman 2000, Trombulak and Frissell 2000). The awareness of the existence of extensive road impacts on wildlife began when researchers reported high levels of road-killed animals affecting a wide range of animal groups (Stoner 1925). Later research assessed in more detail the effects of roads on different animal groups and identified other impacts besides reduction in survival. Avoidance of roaded areas (Rost and Bailey 1979, Witmer and deCalesta 1985), interference with reproduction (Reijnen et al. 1995), and genetic isolation (Epps et al. 2005) were some of the identified effects on wildlife. Noise and vibration are frequently cited as major sources of disturbance (Forman and Alexander 1998).

However, the influence of roads on organisms is not yet fully understood or satisfactorily studied, especially for less visible organisms, e.g., small mammals. Roads potentially impact small mammal communities because they create edges with different habitat characteristics (Garland and Bradley 1984, Tyser and Worley 1992), promote the introduction of exotic species (Getz et al. 1978, Vermeulen and Opdam 1995, Underhill and Anglod 2000), induce disturbance and contamination (Jefferies and French 1972, Williamson and Evans 1972, Quarles et al. 1974), constraint dispersal movements causing genetic barriers and home range rearrangements (Oxley et al. 1974, Garland and Bradley 1984, Mader 1984, Swihart and Slade 1984, Merriam et al. 1989, Gerlach and Musolf 2000), and cause direct road mortality (Wilkins and Schmidly 1980, Ashley and Robinson 1996, Mallick et al. 1998).

The main focus of small mammals' studies has been on road barrier effects (Oxley et al.1974, Garland and Bradley 1984, Mader 1984, Swihart and Slade 1984, Merriam et al. 1989, Gerlach and Musolf 2000), while less attention has been given to the effect of roads on density and diversity of local natural communities. Little is known about communities living in natural habitats adjacent to roads and how they diverge from communities occurring in similar areas with no road influence. Furthermore, conclusions drawn from available studies (Adams and Geis 1983, Adams 1984, Garland and Bradley 1984, Bennett 1988, Meunier et al. 1999, Bellamy et al. 2000, Goosem 2000) are often based on the use of count indices instead of mathematically derived estimators of abundance or density corrected for capture probabilities. Without this correction, the use of indices to estimate accurate population sizes in this type of study is flawed (McKelvey and Pearson 2001) preventing accurate conclusions about road effects.

In my study, presented in Chapter 2, I used estimations of abundance and density and analyzed natural habitat communities in areas adjacent to the road and areas without road influence. My major conclusion was that roads seemed to have a neutral or positive effect on abundance and diversity of small mammal communities. Other studies with small mammals have reported analogous results

(Adams and Geis 1983, Adams 1984, Garland and Bradley 1984, Meunier et al. 1999, Goosem 2000). However, the idea that some wildlife is indifferent or attracted to roads has not yet been embraced by the scientific community nor incorporated into the larger theoretical framework of road ecology. The major contribution of this study is thus heuristic. It reinforces the need for debate about whether all road impacts are negative for wildlife. It also highlights some unstudied questions. For example, what situations make wildlife attracted or indifferent to roads? Are there specific habitat characteristics or climatic conditions that induce wildlife to be attracted to roads? Do roads impact survival of organisms rather than abundance, and if so, how? Are there other groups of animals that show the same abundance patterns than those observed for small mammals? Do we observe these patterns in other areas of the globe? The need for further research to fully understand the complex interaction of organisms and roads is still considerable.

Other major focus of Road Ecology is testing mitigation strategies to minimize negative impacts of roads, especially the impact of roads on wildlife mortality. Wildlife mortality on roads is a problem of increasing concern for transportation engineers and wildlife managers around the globe (Forman and Alexander 1998, Trombulak and Frissell 2000, Jaeger et al. 2005). Mitigation measures have traditionally been applied in problem areas, especially where, large-sized animals, like deer, elk, or moose can potentially cause serious accidents. However, not only are mitigation measures often ineffective, they may also cause additional problems for wildlife. Exclusion fencing, one of the most common measures used to prevent animals from entering the road, is seldom

100% effective and blocks animal migratory movements (Putman 1997). The need for multi-integrated measures has been acknowledged for some time (Feldhamer et al. 1986).

Several studies tested the effectiveness of integrated mitigative measures such as fencing and underpasses to reduce deer-vehicle collisions (Ward 1982, McDonald 1991, Clevenger et al. 2001, Braden 2005). Some of the conclusions drawn from these studies, however, do not provide reliable information because of poor experimental design (Hardy et al. 2003). Often studies cannot explain if reduction in wildlife-vehicle collision was due to the intervention or to confounding variables (e.g., fluctuation in wildlife abundances, traffic volumes). The inclusion of comparisons "before-after" and "control-treatment" is also not frequent and compromises the ability to explain observed patterns (Hardy et al. 2003). Those concerns were incorporated in my study and an integrated multi-measure strategy to minimize road associated deer mortality strategy was evaluated. Results clearly demonstrated that the mitigation was effective during the 2-year period following the mitigation. I observed an overall 77% reduction in collisions in the study area, prevention of "end-of-the-fence" increase in mortality, and increase use of the underpass structures. Future monitoring is important to assess if deer migratory routes through the underpasses become established, and to fully understand mitigation consequences on local populations. Also of concern is the long-term future sustainability of the mitigation strategy. Problems with maintenance of structures often result in decrease of effectiveness, and have to be addressed in order to maintain a long-term functionality and thus justify the initial investment.

Finally, this mitigation strategy was an example of the desirable cooperation between state agencies, and provides an effective model in other areas where similar problems exist.

In conclusion, my results contributed to the understanding of the impacts of roads on wildlife and tested an effective way to mitigate road negative effects on mule deer. My results for small mammals suggest that roads may have positive effects and raise important questions about what kinds of scientific research are needed to better understand the total range of road effects. For mule deer I suggest that further discussion and evaluation is needed for a long-term effective mitigation strategy. To the extent that road-wildlife interactions are openly evaluated, the results will be useful for the management of road-wildlife conflicts.

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