A Before-After-Control-Impact Study of Wildlife Fencing Along a Highway in the Canadian Rocky Mountains

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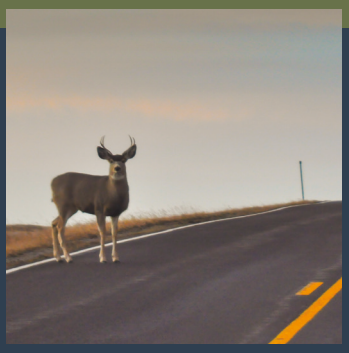

Wildlife Vehicle Collision Reduction and Habitat Connectivity Pooled Fund Study, TPF-5(358)

REDUCE Wildlife Vehicle Collisions

INCREASE Habitat Connectivity

IMPLEMENT Cost Effective Sollutions

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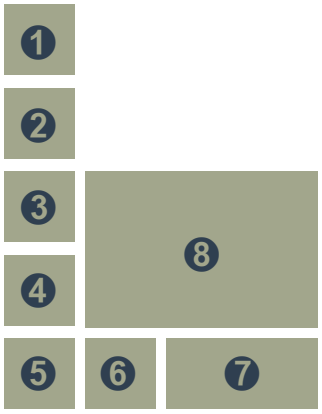

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NDOT Research Report

Report No. 701-18-803 TO 3

TPF-5(358)
A BEFORE-AFTER-CONTROL-IMPACT STUDY OF WILDLIFE FENCING ALONG A HIGHWAY IN THE CANADIAN ROCKY MOUNTAINS

June 2022

Nevada Department of Transportation
1263 South Stewart Street
Carson City, NV 89712

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16. Abstract

Wildlife exclusion fencing has become a standard component of highway mitigation systems designing to reduce collisions with large mammals. Past work on the effectiveness of exclusion fencing has relied heavily on control-impact (i.e., space-for-time substitutions) and before-after study designs. These designs limit inference and may confound the effectiveness of mitigation with co-occurring process that also change the rate of collisions. We used a replicated before-after-control-impact study design to assess fencing effectiveness along the Trans-Canada Highway in the Rocky Mountains of Canada. We found that collisions declined for common ungulates species (elk, mule deer and white-tailed deer) by up to 96% but not for large carnivores. The weak response of carnivores is likely due to combination of fence intrusions and low sample sizes. When accounting for background changes in collision rates observed at control sites, naïve estimates of fencing effectiveness declined by 6% at one site to 90% and increased by 10% at another to a realized effectiveness of 82%. When factoring in the cost of ungulate collisions to society as a whole, fencing provided a net economic gain within 1 year of construction. Over a 10-year period, fencing would provide a net economic gain of >$500,000 per km in reduced collisions. In contrast, control site may take upwards of 90 years before the background rates of collisions decline to a break even point. Our study highlights the benefits of long-term monitoring of road mitigation projects and provides evidence of fencing effectiveness for reducing wildlife-vehicle collisions involving large mammals.

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A before-after-control-impact study of wildlife fencing along a highway in the Canadian Rocky Mountains

FINAL REPORT

Prepared for:

Nevada Department of Transportation (NDOT)

Carson City, Nevada

Prepared by:

Anthony P. Clevenger, Western Transportation Institute

Adam T. Ford, University of British Columbia

Western Transportation Institute

Montana State University

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TABLE OF CONTENTS

List of Tables i
List of Figures ii
Abstract iv
1. Introduction 1
2. Methods 3
2.1. Field Data Collection 4
2.2. Data Analysis 4
3. Results 6
4. Discussion 12
5. References 15

LIST OF TABLES

Table 1: Summary of wildlife-vehicle collision data collected between 1988-2010. The duration and length of each road section and time sequence are described in the text. 6

Table 2: Summary of the mixed-effect negative binomial models to test for the interaction between road segment (control vs impact) and time sequence (before vs after) on elk, mule deer, and white-tailed deer. 8

Table 3: Summary of the mixed-effect negative binomial models to test for the interaction between road segment (control vs impact) and time sequence (before vs after) for all ungulate species pooled together and for all carnivores. Coefficients fixed effects shown with standard errors in brackets. 10

LIST OF FIGURES

Figure 1: Map showing the location of wildlife vehicle collisions along the Trans-Canada Highway in Banff National Park [top] and Dead Man's Flats [bottom], AB, Canada between 1998 and 2010. Green points represent the location of collisions in the control section 3

Figure 2: Predicted effects of the mixed-effects, Poisson regression showing the interaction between road segment (control vs impact) and time sequence (before vs after) on the collision rate of elk (left panel), white-tailed deer (center panel), and mule deer (right panel). For ease of illustration, only the main effects of the model are shown here. The complete statistical model included an AR1 autoregressive term for year nested within study area to account for temporal autocorrelation and a random intercept for study area. 7

Figure 3: Predicted effects of the mixed-effects, Poisson regression showing the interaction between road segment (control vs impact) and time sequence (before vs after) on the collision rate of all ungulates pooled together (left panel) and all carnivore species pooled together (right panel). For ease of illustration, only the main effects of the model are shown here. The complete statistical model included an AR1 autoregressive term for year nested within study area to account for temporal autocorrelation and a random intercept for study area. Note different scales on the y-axis..... 9

Figure 4: Estimated cost of collisions occurring within each road segment and time sequence for the two study areas, Dead Man's Flats and Banff National Park. Cost estimates per collision are based on Huijser et al (2009). 11

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ABSTRACT

Wildlife exclusion fencing has become a standard component of highway mitigation systems designing to reduce collisions with large mammals. Past work on the effectiveness of exclusion fencing has relied heavily on control-impact (i.e., space-for-time substitutions) and before-after study designs. These designs limit inference and may confound the effectiveness of mitigation with co-occurring process that also change the rate of collisions. We used a replicated before-after-control-impact study design to assess fencing effectiveness along the Trans-Canada Highway in the Rocky Mountains of Canada. We found that collisions declined for common ungulates species (elk, mule deer and white-tailed deer) by up to 96% but not for large carnivores. The weak response of carnivores is likely due to combination of fence intrusions and low sample sizes. When accounting for background changes in collision rates observed at control sites, naïve estimates of fencing effectiveness declined by 6% at one site to 90% and increased by 10% at another to a realized effectiveness of 82%. When factoring in the cost of ungulate collisions to society as a whole, fencing provided a net economic gain within 1 year of construction. Over a 10-year period, fencing would provide a net economic gain of >$500,000 per km in reduced collisions. In contrast, control site may take upwards of 90 years before the background rates of collisions decline to a break even point. Our study highlights the benefits of long-term monitoring of road mitigation projects and provides evidence of fencing effectiveness for reducing wildlife-vehicle collisions involving large mammals.

1. INTRODUCTION

Roads are a ubiquitous feature of human activity across the earth, with ~21 million km built as of 2018 and upwards of an additional 25 million km of new roads developed by 2050 (Dulac 2013, Meijer et al. 2018). Central to commerce, resource extraction, and human connectivity, roads also induce a number of negative impacts on biodiversity. These impacts include barrier effects to movement (Jaeger et al. 2005, Epps et al. 2005, Ford and Fahrig 2008), changes to ecological interactions (Whittington et al. 2011, Dickie et al. 2017), increased access by people to intact habitats (Laurance et al. 2009, Carter et al. 2020), and increase mortality from wildlife-vehicle collisions (Trombulak and Frissell 2000, Ford and Fahrig 2007, Clevenger et al. 2015). Wildlife-vehicle collisions not only harm biodiversity, they can also jeopardize human safety – particularly for larger species like ungulates. For example, 2-3% of WVCs involving deer result in human injury and this increases to ~20% for WVCs involving larger species like moose (Huijser et al. 2009).

To counteract the negative effects of WVCs on wildlife and people, transportation and wildlife agencies have employed a number of mitigation strategies and technologies. These strategies include reduced speed limits (Riginos et al. 2019), road closures (Lamb et al. 2018, Whittington et al. 2019), road density planning (Carter et al. 2020), and technologies such as warning signs for drivers (Huijser et al. 2015). One of the more common tools is wildlife-exclusion fencing, which is designed to prevent large animals from accessing the road right-of-way (but see (Ford and Clevenger 2019). Fence designs vary – and may include shorter 'drift' fences for amphibians (Boyle et al. 2021) and taller wire fences designed for large mammals (Ford et al. 2011). Fencing may be accompanied by wildlife crossing structures, which are intended to maintain or restore the safe movement of wildlife across the highway via underpasses or overpasses. Because of its cost, fencing is usually prioritized along particular road segments or 'hotspots' of WVCs (Ford et al. 2011, Huijser et al. 2016, Lee et al. 2020).

Despite the widespread use of technologies like fencing and crossing structures, monitoring and evaluation of mitigation often suffers from lack of experimental control. For example, study designs may compare WVC rates before and after the construction of mitigation (e.g., Clevenger et al. 2001) ; however, as wildlife populations fluctuate over time independently of road mitigation, such a before-after ("BA") design may confound changes in the wildlife population with mitigation effectiveness. Indeed the underlying road and biophysical process that contribute to WVCs are highly localized even for similar species within a region or watershed (Clevenger et al. 2015, Lee et al. 2020). Likewise, study designs that compare mitigation across nearby areas (i.e., control-impact or CI) may confound local variation in wildlife population size, road design, or driver behaviour with mitigation (Rytwinski et al. 2015).

To support evidence based approaches to the design and management of road mitigation, it is critical that more rigorous, quantitative approaches be used to assess mitigation effectiveness (Rytwinski et al. 2016). Upon experimental testing, putatively effective technologies may not work as intended at the scales or intensity needed to reduce WVC rates. For example, vehicle speed reductions did not affect WVCs involving deer in Wyoming (Riginos et al. 2019). Similarly, a meta-analysis indicated that wildlife reflector systems are equivocal in their effect on WVC reductions for large mammals, whereas crossings and fencing have a strong negative effect on WVCs across studies (Rytwinski et al. 2016). In addition, this meta-analysis revealed that studies

using BA and combined BACI (before-after control impact) designs had stronger effects on WVC rates than CI studies alone. This suggests that opportunities to assess road mitigation through BACI designs are not only more rigorous (Boyle et al. 2021), but likely to reduce the probability of a Type II error (i.e., there is an effect of mitigation on WVCs but it was not detected).

The Bow Valley of Alberta includes some of the most well-studied, long-term monitoring of WVCs and road mitigation (Ford et al. 2010, Lee et al. 2012, Clevenger et al. 2015).Spurned by the upgrade of Canada's major transportation corridor (the Trans-Canada Highway – TCH) in the 1970s, coupled with a hyperabundant elk population, Banff National Park undertook one of the earliest and most expansive wildlife crossing structure efforts in the world. Beginning with a series of underpasses and large mammal exclusion fencing, this system of mitigation expanded along a series of phases (~15-30km each) from the early 1980s to the present day (Ford et al. 2010). This phased approach has enabled managers to prioritize higher-risk road segments for mitigation (e.g., Ford et al. 2011, Lee et al. 2020), but it – along with long-term monitoring - has also facilitated opportunistic field experiments in road ecology. For example, Ford and Clevenger (2010), used a BACI design across highway mitigation phases to test the prey-trap hypothesis, i.e., that wildlife crossing structures facilitate predation by large carnivores.

Here, we used an opportunistic field experiment in a BACI-design to quantify the effect of highway fencing on large mammals WVCs. We integrate long-term monitoring of WVCs (Gunson et al. 2009, Clevenger et al. 2015) over a 12-15 year period covering adjacent highway segments in two regions of the TCH. We focused on the occurrence of ungulates and large carnivores in WVCs. Building on published values that assess the cost of a WVCs to society in terms of property damage, injury and lost opportunity costs of an animal (Huijser et al. 2009, Ford et al. 2011), we also measured the number of years from the completion of mitigation to reach a cost recovery in reduced WVCs.

2. METHODS

The Trans-Canada Highway (TCH) runs along the Bow River Valley and is a major east-west aligned transportation corridor spanning from Atlantic to Pacific coast. In Alberta, the TCH traverses the Bow River watershed on the east side of the Continental Divide. The TCH is four lanes wide with an estimated annual average daily traffic volume (AADT) of over 17,000 vehicles per day, with peaks of more than 30,000 vehicles per day during summer (Highway Service Center, Parks Canada, unpublished data). In addition to the TCH, the Bow River Valley includes portions of the Bow River, the Canadian Pacific Railway, outlying commercial areas (ski hills, golf courses), and the towns of Lake Louise, Banff, Canmore, and Dead Man's Flats (Ritchie and Brent 1998, Barrueto et al. 2014). In general, the Bow River Valley represents high quality, low elevation wildlife habitat in this mountain landscape. The region is a vital movement corridor for large mammals, including grizzly bears (*Ursus artcos*), wolves (*Canis lupus*) elk (*Cervus elaphus*), deer (*Odocoileus* sp.), moose (*Alces alces*), and Rocky Mountain bighorn sheep (*Ovis canadensis*) (Musiani et al. 2010).

Within the Bow River watershed we assess the performance of mitigation measures to reduce WVCs in two distinct sections of TCH based on biophysical transition. Banff National Park (BNP) and Dead Man's Flats (DMF) (Figure 1). BNP is characterized by higher elevation, greater precipitation, and lower human population density compared to the DMF section.

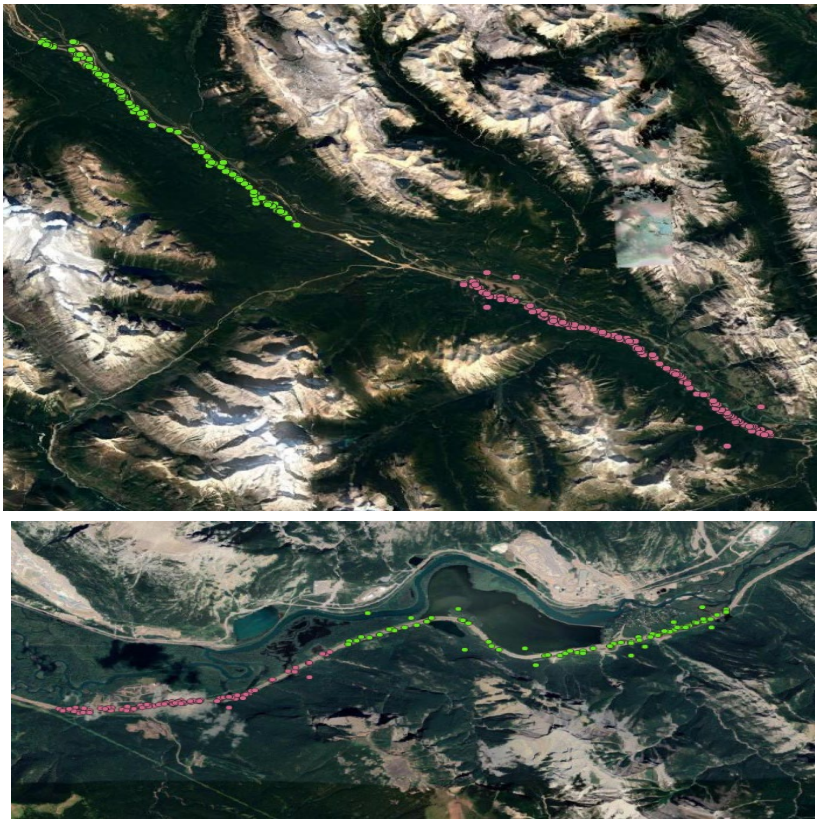

Figure 1: Map showing the location of wildlife vehicle collisions along the Trans-Canada Highway in Banff National Park [top] and Dead Man's Flats [bottom], AB, Canada between 1998 and 2010. Green points represent the location of collisions in the control section

2.1. Field Data Collection

Wildlife-vehicle collision (WVC) data were collected year-round as part of routine wildlife observation reporting by staff from BNP. Each accident site was visited and the date of the WVC was recorded along with information about the species and the number of individuals. For each WVC location a Universal Transverse Mercator (UTM) coordinate was obtained using a handheld, global positioning system (GPS) unit accurate to <5 m (Gunson et al. 2009, Clevenger and Barrueto 2014). For the DMF section, WVC data were synthesized from four provincial datasets over a 10-year period (Lee et al. 2012). Most of the data were not systematically collected but based on opportunist data entry from highway maintenance contractors, provincial biologists and the general public. The data were cleaned and a government biologist removed duplicates.

In BNP, both the unfenced (i.e., control) and impact (i.e., fenced) sections of highway were 17 km long. The impact section of highway was contiguous with existing fence on both the east and west ends. We did not analyze buffer zones around fence ends for potential fence end effects because the control section was at least 3 km from any fence end during the entire analysis period. As a result, 34 km of highway were analyzed during a 12-year period (six years before and six years after). Mitigation fencing required three years to completely install, so those years were excluded from the analysis for both the fenced and unfenced sections. For each of seven years prior to mitigation (Nov 1988 – Nov 1994) and six years following mitigation (Nov 1998 – Nov 2003), data were compiled by species and by kilometer.

In DMF, mitigation measures consisted of a wildlife underpass flanked by 1.5 km of fencing to the east and 4 km of fencing to the west, with fencing constructed on both sides of the highway (Lee et al. 2012). We compiled counts of WVCs for each of the six years prior to mitigation (1998 - 2004) and six years following mitigation (2005 - 2010). The treatment segment was defined as the portion of highway associated with the underpass and fencing described above as well as a buffer zone extending 1 km beyond the east end of the fence to account for potentially heightened WVCs at the fence end (Huijser et al. 2016). The western end is contiguous with an existing fence and therefore was not assigned as a buffer zone. WVC data were compiled for this 4-km treatment section as well as a contiguous 6 km unmitigated control segment to the east

2.2. Data Analysis

Collision data were annualized and assigned to one of four experimental groups: before-control, after-control, before-impact, after-impact. The control sections were not fenced during the study period; the impact sections were fenced in the 'after' period only. We separately analyzed elk, mule deer, and white-tailed deer, as well as separate analyses for pooled ungulates and pooled carnivores. The pooled ungulate analysis included unidentified deer species, moose, and bighorn sheep – all of which of occurred too infrequently to be analyzed independently. Similarly, we pooled carnivores because low numbers across experimental groups made species-specific analyses impossible in the mixed-effects modeling framework we used.

We used a Poisson mixed effects models to analyze these data. We first confirmed that the count data (collisions per year per section) were not over dispersed or had too many zeros using the functions testDispersion() and testZeroInflation() in the R package, DHARMa (v 0.4.5). We used a main-effects factor for the fenced vs unfenced road section (i.e., "road segment"), a main-effects factor for the before vs after period (i.e., "time sequence"), and an interaction term between road segment and time sequence. We included study area (i.e., BNP or DMF) as a random

intercept, with an autoregressive (AR1) term nested within study area to account for temporal autocorrelation. We included an offset term for effort – i.e., the number of kilometer-years sampled in each of the four experimental groups.

For the cost-benefit analysis, we assigned elk, moose, sheep, and deer species a cost-per-collision based on Huijser et al (2009). We summed these costs for each section. We then examined the cost of fencing based published values for this area ($60/meter; Ford et al 2011). We note that actual costs of building fencing will vary by project, so we offer these values as a proof of concept. We then calculated the annual reduction in collision costs caused by mitigation. Finally, we estimated the number of years to cost-recovery for fencing by dividing the total cost of fencing by the annualized cost reduction in collisions.

3. RESULTS

We compiled data from 376 wildlife-vehicle collisions (WVCs), including 151 from the two control road sections and 225 from the two impact sections (Table 1). Most WVCs (92%) involved ungulates – and 43% of all ungulate collisions involved elk. Among 31 carnivore WVCs, most (65%) were black bears. Both study areas had comparable total WVCs (i.e., BNP= 183; DMF = 193); however, when accounting for road length of each study area and years of monitoring, DMF had 1.06-3.63 collisions per km per year (for control and impact sections, respectively), while BNP had 0.33-0.85 collisions per km per year (for control and impact sections, respectively) prior to mitigation. The impact road sections had roughly twice the rate of WVCs compared with the control sections in both study areas. Following mitigation, there were 1.25 WVCs per km per year for fenced sections at DMF and 0.06 WVCs per km per year for fenced sections in BNP.

Table 1: Summary of wildlife-vehicle collision data collected between 1988-2010. The duration and length of each road section and time sequence are described in the text.

		Control		Impact	
Study area	**Species**	**Before**	**After**	**Before**	**After**
Banff	Black bear	0	2	2	2
	Cougar	0	0	0	0
	Unknown deer_spp	0	1	0	1
	Elk	18	14	57	2
	Lynx	0	0	0	0
	Moose	3	3	1	0
	Mule deer	10	6	23	1
	Sheep	0	0	0	0
	Unknown ungulate_sp	0	0	0	0
	Wolf	1	0	3	0
	White-tailed deer	9	8	16	0
Dead Man's Flats	Black bear	6	3	0	5
	Cougar	0	1	1	2
	Unknown deer_spp	4	11	5	7
	Elk	12	8	33	6
	Lynx	1	0	0	0
	Moose	0	0	7	1
	Mule deer	4	1	5	2
	Sheep	0	3	5	0
	Unknown ungulate_sp	0	0	2	0
	Wolf	0	0	2	0
	White-tailed deer	11	11	27	7

At a species-specific level, there was a significant effect of the interaction between road segment (control vs. impact) and time sequence (before vs. after) for elk, mule deer, and white-tailed deer (Figure 2, Table 2). Across all species and study areas, the timing of mitigation was associated with a 2% increase (from 0.50 to 0.52 WVCs per km per year) in WVCs for control sections and an 78% reduction for fenced sections (from 1.32 to 0.28 WVCs per km per year). For ungulates on unfenced control sections, mitigation was associated with an 6% decrease in BNP (0.33 to 0.31 WVCs per km per year) and a 9% increase in DMF (0.86 to 0.94 WVCs per km per year). For ungulates on fenced highway sections, mitigation was associated with an 96% decrease at the BNP study area (0.96 to 0.04 WVCs per km per year) and a 73% decrease in DMF study area (3.50 to 0.96 WVCs per km per year).

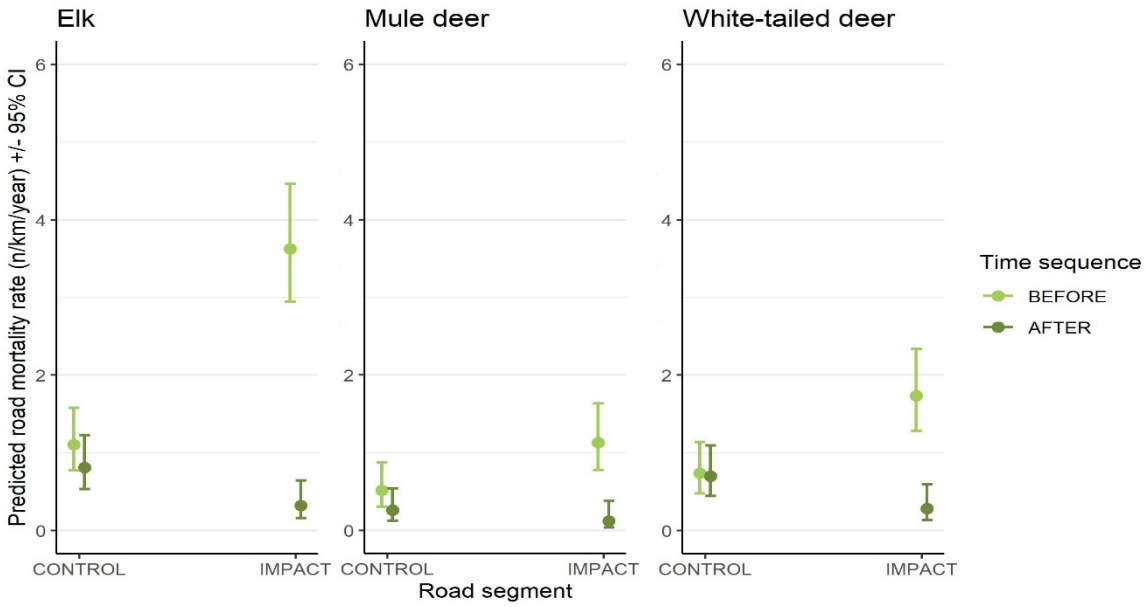

Figure 2: Predicted effects of the mixed-effects, Poisson regression showing the interaction between road segment (control vs impact) and time sequence (before vs after) on the collision rate of elk (left panel), white-tailed deer (center panel), and mule deer (right panel). For ease of illustration, only the main effects of the model are shown here. The complete statistical model included an AR1 autoregressive term for year nested within study area to account for temporal autocorrelation and a random intercept for study area.

Table 2: Summary of the mixed-effect negative binomial models to test for the interaction between road segment (control vs impact) and time sequence (before vs after) on elk, mule deer, and white-tailed deer.

Model term	Species		
	Elk	**Mule deer**	**White-tailed deer**
Intercept	-4.20 *** (1.25)	-4.366 *** (0.843)	-4.060 *** (1.179)
Road segment: Impact [relative to control]	1.26 *** (0.21)	0.785 * (0.328)	1.015 *** (0.272)
Time sequence: After [relative to before]	-0.05 (1.70)	-0.415 (1.085)	-0.028 (1.626)
Road segment x Time sequence	-2.10 *** (0.47)	-1.519 * (0.766)	-1.746 *** (0.520)
*** $p < 0.001$; ** $p < 0.01$; * $p < 0.05$.			

Reductions in WVCs were greatest for elk (81-96% decline for fenced sections vs. 9-33% decline in control, for BNP and DMF, respectively), white-tailed deer (100-74% decline for fenced sections vs 0-3% change in control, for BNP and DMF, respectively), and mule deer (96 to 60% decline for fenced sections vs 75-30% decline in control, for BNP and DMF, respectively) in the after period relative to the before period. Both elk and mule deer WVCs declined in unfenced control sections. At the taxa level, there was a significant interaction of road segment and time sequence for ungulates but not carnivores (Figure 3, Table 3).

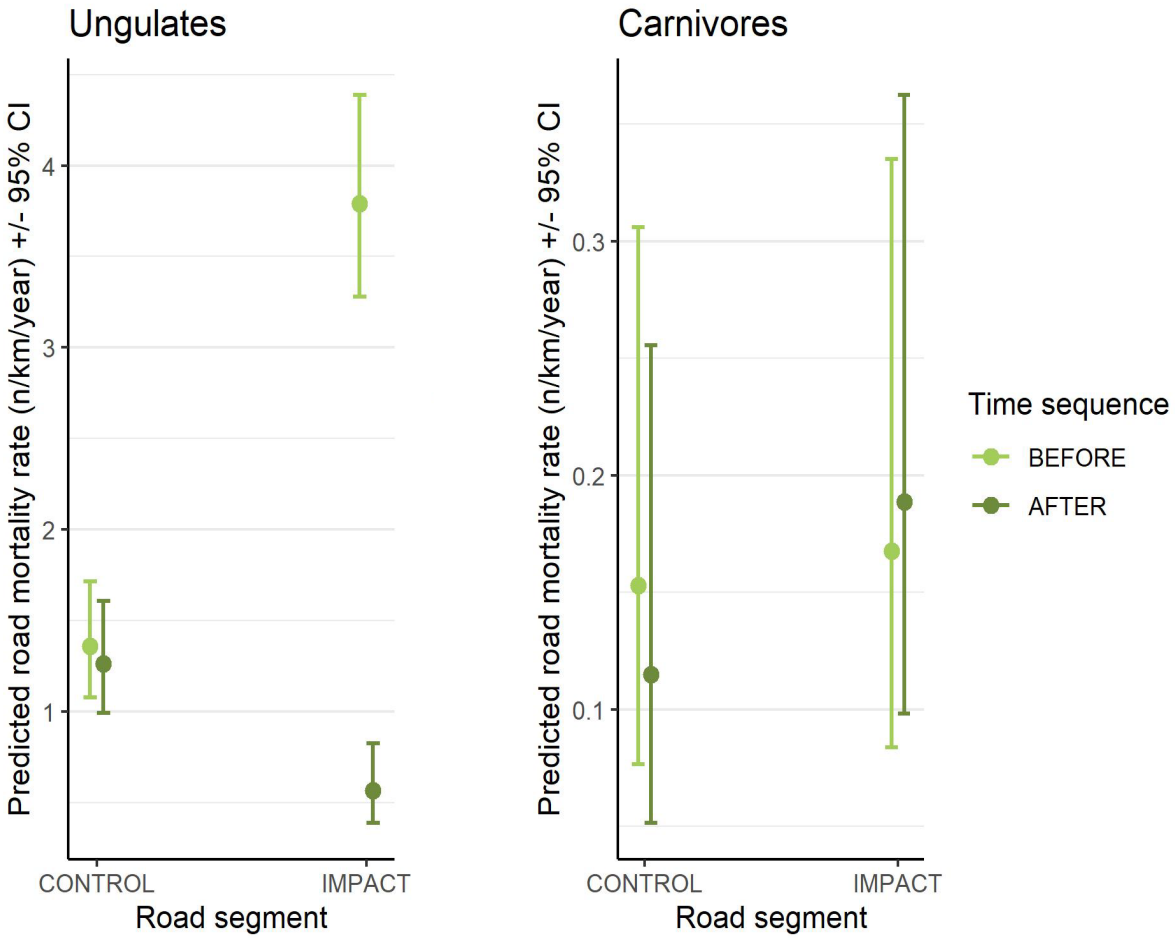

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Table 3: Summary of the mixed-effect negative binomial models to test for the interaction between road segment (control vs impact) and time sequence (before vs after) for all ungulate species pooled together and for all carnivores. Coefficients fixed effects shown with standard errors in brackets.

Model term	Taxa	
	Ungulates	**Carnivores**
Intercept	.658 ** (2.027)	-6.749 *** (1.428)
Road segment: Impact [relative to control]	1.130 *** (0.141)	0.248 (0.502)
Time sequence: After [relative to before]	0.051 (2.640)	-0.653 (1.687)
Road segment x Time sequence	-1.784 *** (0.269)	0.449 (0.729)
*** $p < 0.001$; ** $p < 0.01$; * $p < 0.05$.		

Prior to mitigation, WVCs incurred a total cost of $24,131 to $69,325 per km per year for control and treatment road segments (pooled across study areas), respectively. Following mitigation, WVCs incurred a cost of $21,387 to $8,783 per km per year for control and treatment road segments, respectively. In other words, the cost of WVCs across control sections declined by $2,743 per km per year even though these sections were never fenced during the study. This outcome is likely driven by the observed decline in elk and mule deer WVCs at control sections. Over the same period of time, the cumulative cost of WVCs in fenced sections declined by $60,541 per km per year (Figure 4).

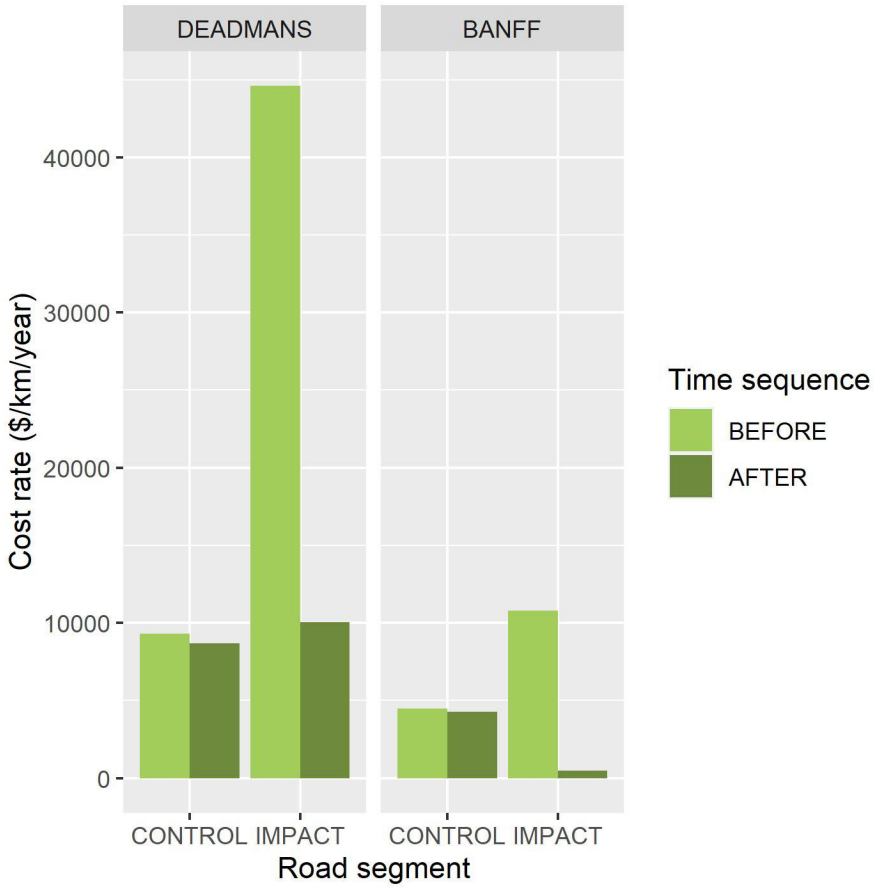

Figure 4: Estimated cost of collisions occurring within each road segment and time sequence for the two study areas, Dead Man's Flats and Banff National Park. Cost estimates per collision are based on Huijser et al (2009).

When accounting for both the cost of fencing and the benefits of WVC reduction, fenced segments delivered a net benefit of $3,287 in the first year. Assuming a one-time fixed cost of installing fences, mitigation is estimated to provide a net benefit of $572,872 per km at treatment sections over a 10-year period. Because there were generally fewer WVCs at control sections, fencing them will take longer to reach a net economic gain. When projecting the observed background declining cost of WVCs of 11% per km per year, it would take 88 years to reach a net cost of < $1 per km per year for control sections in the absence of fencing. Thus, the time required to reach a financial break even point for fenced sections (<1 year) and unfenced control sections (~70 years) is roughly 90x greater without fencing.

4. DISCUSSION

Through a combination of long-term monitoring and phased highway mitigation, we conducted an opportunistic, before-after-control-impact (BACI) study design in two adjacent areas of the Canadian Rocky Mountains. We found that highway fencing led to a significant (>80%) reduction in wildlife-vehicle collisions (WVCs) for common ungulate species. We did not detect a change in carnivore-vehicle collisions following mitigation. Our results indicate that that societal-wide economic benefits of mitigation can be realised in less than 2 years for high-risk WVC road segments and in 6 years for lower risk areas. While the effect sizes we observed are consistent with related studies (e.g., Clevenger et al. 2001, Rytwinski et al. 2015, 2016), the BACI study design we employed revealed confounding variation in WVC rates that more accurately portrays the costs and benefits of highway fencing.

Consistent with previous studies in this region (Clevenger et al. 2001, Gilhooly et al. 2019) and abroad (McCollister and van Manen 2010), wildlife exclusion fencing is highly effective at reducing vehicle collisions with ungulates. The reduction in WVCs was greatest in the BNP study area, likely because of the additional effects of staff at Banff National Park who will actively direct animals off the highway right-of-way through a series of large (4-m wide) swing gates and conduct fence repairs when damaged (Clevenger et al. 2009). In addition, the BNP section of the highway was more than twice as long at the DMF section, possibly indicating that 'fence end' intrusions could be reducing the effectiveness of this shorter section of fencing (see Huijser et al. 2016).

Unlike ungulates, carnivore mortality rates were low but did not change following mitigation. The lack of response by carnivores can be explained by both biological and statistical reasons. From a biological perspective, page wire fences can be climbed by black bears – the most common carnivore in our study (Serrouya 1999, Hebblewhite et al. 2003). Downed trees on top of the fence and soil erosion under the fences can also open gaps in the fences that lead to intrusions onto the right of way. Consequently, carnivore intrusions may remain an issue for the TCH – particularly if there are food attractants like road-killed ungulates and herbaceous plants (i.e., for bears). In addition to these biological explanations, our study may not have the statistical power to detect a change in carnivore collisions. For example, we detected no cougar mortalities in BNP and just 4 in DMF area. Of those 4, only 1 occurred in the control section. Similarly low observations occurred for wolves and lynx, making it impossible to conduct species-specific comparisons among carnivores. We know from other studies that carnivores are frequently observed near the highway and use nearby wildlife crossing structures many hundreds of times annually, thus significantly reducing mortality risk compared to a unfenced TCH (Clevenger et al. 2009, Ford and Clevenger 2010, Ford et al. 2010). From a road safety perspective our results suggest that carnivore-vehicle collisions are low risk. However, the per capita effect of even rare mortality, especially for adult females, can be profound for low fecund, low density species like carnivores (Lamb et al. 2020).

For the public, the return on investment from highway mitigation can be realized fairly rapidly providing that a few conditions are met. First, the fencing needs to be placed in area with high risk of collisions. While intuitive, identifying area of high risk also requires adequate data: long term, consistent, and accurate data collection protocols. Variation in such protocols, such as GPS vs landmark-located carcases can profoundly affect where and why roadkill hotspots occur (Gunson et al. 2009). Prior to mitigation in our study area, high-risk sections of the TCH incurred close to $70,000 per km per year in damage to vehicles, human health and lost opportunity costs

of wildlife mortality (sensu Huijser et al. 2009). With an estimated cost of $60,000 per km for fencing (Ford et al. 2011), benefits to society at large are realized almost immediately. For the return-on-investment analysis to function, we assumed that both funders and benefactors of fencing share a common economic pool. However, the BNP section was part of a national parks system which provides ecosystem services to visitors from all over the world, most of whom place transportation infrastructure as a highly-ranked part of their experience (Geng et al. 2021). In this case, the funders of the BNP mitigation costs are 'national' or international in scope, and benefactors include a largely non-resident population of national park visitors and people passing through Banff National Park. In contrast, the DMF section of the highway was funded through a mix of provincial and federal funding (Lee et al. 2012), with benefits accruing to the traffic and a smaller tourism sector (Hu et al. 2021). Thus, the payers and benefactors of the economic analyses we conducted can vary significantly at different sections of the same highway. Nonetheless, for nationally-significant infrastructure like the TCH, the assumptions we made based on (Huijser et al. 2009) for a net societal benefit seem reasonable. For more localized mitigation projects, such as secondary roads, such cost-benefit calculations of fencing require further exploration of these assumptions.

Following perspectives shared in Rytwinski et al. (2015), our study was one of few to conduct a BACI analyses to test for the effectiveness of fencing to reduce WVCs. This approach was made possible by long-term monitoring of WVCs conducted along sequential phases of highway mitigation. Such a phased approach can be very cost effective by targeting high-risk areas with more efficient and shorter road segments (Ford et al. 2011, Huijser et al. 2016). Indeed, the "before" periods for the fenced sections had much higher rates of WVCs than the before period for the non-fenced sections. This means that fencing was appropriately installed at higher-risk areas. High-risk locations can be identified through WVC monitoring, as in our study system (e.g., Clevenger et al. 2015) and through various connectivity algorithms that simulate likely road crossing locations (Clevenger et al. 2002, Quaglietta et al. 2019, Zeller et al. 2020).

Another important outcome of our study was the observed decline in WVCs rates among elk and deer along unfenced road sections. Given the short distance between fenced and unfenced sections, the mechanisms leading to the decline of WVCs at control sections was likely also present at the treatment sections. We speculate that the background decline in WVCs could be driven by a declining population abundance of elk (Hebblewhite et al. 2002), changing animal movement patterns near roads (Seiler 2005), or changing driver behaviour associated with other mitigation efforts in the area, like signage (Huijser et al. 2009). Traffic volume in this area has increased over the duration of the study area from 10,000 annual average daily traffic (AADT) in the early 1980s to ~20,000 AADT in 2014 (Gilhooly et al. 2019) – the volumes are much higher than reported thresholds of avoidance for ungulates (Seiler 2005). In other words, its is unlikely that the added traffic volume created a stimuli that led to fewer attempts by individual animals to cross the road. Thus, we do not believe that the increase in traffic volume is causing ungulates to avoid crossing the road more in the after period than during the before period. Changes in driver behaviour may be linked to fewer WVCs at unfenced sections; however it is difficult to imagine a mechanism that altered driver behaviour in such a significant manner over a short period of time, and for the BNP section of the TCH but not the DMF section.

Regardless of the mechanism for the decline in WVCS at control sites, this signal confounds the magnitude of the fencing effect *per se*. For example, it would be reasonable but inaccurate to conclude that fencing led to a 96% decline in ungulate WVCs in BNP (i.e., 0.96 to

0.04 WVCS per km per year). We argue that the decline in WVCs owing to fencing must consider background changes of WVC declines at control sections (i.e., 6% decline in ungulate WVCs in BNP). This means that the realized effect of fencing is closer to 90% in BNP for ungulates. At the DMF study area, controls had a 10% increase in ungulate WVCs and fenced areas had a 73% decline in WVCs. Thus, the realize effect of fencing per se on ungulate WVCs at DMF should be closer to 82%.

The BACI design we used is not only associated with stronger effect sizes 'before-after' or 'control-impact' (space for time substitutions) studies alone (Rytwinski et al. 2016), it is also critical for developing a rigorous understanding of how large, free-ranging animals interact with transportation infrastructure. In a few cases, such as road closures (Whittington et al. 2019) or lockdowns associated with the COVID-19 pandemic (Shilling et al. 2021, Abraham and Mumma 2021), there is limited opportunity to perform replicated field experiments in road ecology for large mammals. Investments in long-term monitoring of WVCs (Gunson et al. 2009, Gilhooly et al. 2019), animal movement at crossing structures (Clevenger et al. 2009, Ford et al. 2017), and predator-prey interactions (Ford and Clevenger 2010) are a key feature of BACI designs when infrastructure cannot be readily moved or altered following construction. As part of this long-term approach to monitoring, accounting for temporal autocorrelation will become a more reliable means to meet the assumptions of statistical analyses (Boyle et al. 2021). These approaches may not be reliable for short time series data even if the variances in the data are temporally structured in the statistical model.

Fencing is being used for a variety of purposes in conservation and science, including human-wildlife coexistence (Bauer 1964, Packer et al. 2013), disease mitigation (Mysterud and Rolandsen 2019), basic ecological research (Goheen et al. 2018), and ecological restoration (Laskin et al. 2020, Ford 2021). Indeed, with the growing use of fencing in conservation, specialised analytical tools are being used to assess the response of wildlife movement (Xu et al. 2021) amidst broader calls for developing the science of "fence ecology" (Jakes et al. 2018, McInturff et al. 2020). Our work extends these studies by showing how fencing technology (~2.4m high variable mesh size page wire fence with a buried chain link mesh apron) reduces ~70-90% of ungulate-vehicle collisions. Further research on fencing effectiveness for large carnivores and other species like small mammals (Ford and Clevenger 2019) will be needed, particularly given their infrequent occurrence in WVC studies (Ford and Fahrig 2007).

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Nevada Department of Transportation

Kristina L. Swallow, P.E. Director

Ken Chambers, Research Division Chief

(775) 888-7220

kchambers@dot.nv.gov

1263 South Stewart Street

Carson City, Nevada 89712