

FACTORS AFFECTING BEAR AND UNGULATE MORTALITIES
ALONG THE CANADIAN PACIFIC RAILROAD THROUGH
BANFF AND YOHO NATIONAL PARKS

by

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A thesis submitted in partial fulfillment
of the requirements for the degree

of

Master of Science

in

Land Resources and Environmental Sciences

MONTANA STATE UNIVERSITY

Bozeman, Montana

November, 2011

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November, 2011

ACKNOWLEDGEMENTS

I would like to thank Tawnya Hewitt for being a great friend, enormous source of support, and helpful critic of this thesis. I am grateful for the patient guidance of my advisors, Dr. Lisa Rew and Dr. Bruce Maxwell. You both have inspired me. Dr. Tony Clevenger provided enormous amounts of support and guidance, including a wealth of transportation-ecology specific direction and help writing. Dr. Mike Gibeau provided much of the initial support and direction to get this project started and helped identify the research questions, thanks Mike. The Western Transportation Institute (WTI) at Montana State University provided more than 80% of the financial support for this project, thank you WTI, Rob Ament and Susan Gallagher. Parks Canada provided financial support, housing and office space. Hal Morrison, Steve Michel, Tom Hurd, and Kris McCleary provided ideas and direction at various stages in the project. Shelagh Wrajez, Saundi Stevens, Jessie Whittington, Colleen Campbell, Partick Langdon, Karsten Heuer, Anne Forshner and others collected and processed data used in this thesis. Thank you all.

TABLE OF CONTENTS

1. REVIEW OF LITERATURE: IMPACTS OF RAILROADS ON TERRESTRIAL WILDLIFE	1
Introduction.....	1
Methods.....	3
Results and Discussion	3
Habitat.....	4
Movements.....	7
Mortality	10
Reducing Impacts.....	14
Conclusion	16
2. RELATIONSHIPS BETWEEN BEAR (<i>URSIDAE</i>) FORAGING RATES, MORTALITY AND TRAIN-SPILLED GRAIN IN BANFF AND YOHO NATIONAL PARKS.....	24
Introduction.....	24
Methods.....	29
Study area.....	29
Data Collected.....	31
Data Analysis	38
Results.....	41
Temporal.....	41
Spatial	43
Discussion.....	47
Management Implications.....	50
3. RELATIVE RISK AND VARIABLES ASSOCIATED WITH BEAR AND UNGULATE MORTALITIES ALONG A RAILROAD IN THE CANADIAN ROCKY MOUNTAINS.....	54
Introduction.....	54
Methods.....	58
Study Area	58
Data Collected.....	60
Data Analysis	65
Results.....	68
Strike rates	68
Relative abundance	72
Hotspots and relative risk.....	74
Variables associated with strikes	75
Discussion	79

TABLE OF CONTENTS - CONTINUED

Management Implications.....	83
LITERATURE CITED.....	85

LIST OF TABLES

Table	Page
1. Impacts of railroads on terrestrial vertebrates. Impacts (x) were classified as: habitat, movement or mortality based on studies in peer reviewed literature, professional conference proceedings, and technical reports between 1960 and 2011.....	18
2. Description of field collected and GIS derived spatially varying wildlife, train and railroad design variables and the hypothesized correlation to strike rates. (+) indicates a positive correlation and (-) indicates a negative correlation hypothesized for both ungulates and bears.....	69
3. Variation between years and railroad segments for a) elk, b) deer and c) bears along the Canadian Pacific Railroad within Banff and Yoho National Parks, 1989 to 2009.....	73
4. Significant ($p < 0.05$) parameter estimates explaining train strike rates with elk, deer and bear. Model fit statistics are provided from maximum likelihood fits.	78

LIST OF FIGURES

Figure	Page
1. Map of the 134 kilometers of the Canadian Pacific Railroad study area that traverses through Banff and Yoho National Parks in the Canadian Rocky Mountains. Each analysis segment (4.86 km) is shown denoted by a small perpendicular bar along the railroad.....	31
2. (a) Reconyx® time-lapse camera b) A time-lapse photograph of a bear (<i>Ursus</i> spp.) exhibiting the foraging behavior on the Canadian Pacific Railroad near Field, B.C.	34
3. Grain spill sampling methods used on rail bed. a) Grain spill monitoring screen covering a 0.37m ² area between the tracks, used to estimate the weekly weight of grain spilled at four locations. b) Spatial grain spill sampling frame (10cm ²), used to estimate the weekly weight of grain spilled at four locations. b) Spatial grain spill sampling frame (10cm ²), used with three replicate random throws to estimate the mean density of wheat and barley seeds within a 5 m zone.....	39
4. (a) Total weekly foraging time by bears on train-spilled grain by hour of day and, (b) mean foraging duration in minutes by bears on train-spilled grain by hour of day, recorded at four time-lapse camera stations on the Canadian Pacific railroad between between Field and Lake Louise between 12 May to 30 July 2009 and 2010 (c) Mean train arrival times over a 24 hour period for April to November 2010 on the Canadian Pacific Railroad determined from a motion activated camera located between Field and Lake Louise.....	42
5. (a) Weekly grain spill weight measured in grams per 0.37 m ² between Field and Lake Louise in spring/summer) during 2009 and 2010. b) Average foraging duration and c) total foraging time (log minutes) on train-spilled grain detected at four time-lapse camera stations on the Canadian Pacific Railroad between Field and Lake Louise between 5 May to 30 July in 2009 and 2010.....	44
6. (a) Total weekly foraging time (log minutes) on train-spilled grain compared to total grain weight (log grams) on a log-log scale. (b) Mean foraging duration (log minutes) on train-spilled grain compared to log grain weight. Foraging time and duration was recorded using four time-lapse camera stations, and grain spill was recorded at monitoring stations on the Canadian Pacific Railroad between Field and Lake Louise in 2009 and 2010.....	44

LIST OF FIGURES - CONTINUED

Figure	Page
7. (a) Foraging time in log minutes and (b) average foraging duration in log minutes estimated using time-lapse photography recorded at four camera stations on the Canadian Pacific Railroad between Field and Lake Louise between 5 May to 30 July in 2009 and 2010.....	45
8. Grain spill measured as mean counts of wheat and barley seeds within a 10cm ² frame at random locations and within 1 m of bear scats detected.	46
9. Spatial relationships between a) number of bear scats and grain density, b) train bear collisions (strikes) and grain density, and c) train bear strikes and bear scat density on the Canadian Pacific Railroad from the east entrance of Banff National Park, Alberta to the west boundary of Yoho National Park, BC in 2008 and 2009 summarized by 4.86 km long rail segments.	46
10. Grain spill sampling method used on rail bed. The grain spill sampling frame (10cm ²), used with three replicate random throws to estimate the mean density of wheat and barley seeds within a 5 m zone.	64
11. The number of train strikes (white bars) for a) elk, b) deer, c) and d) bears; compared to the expected number of strikes (grey bars, a positive value) based on the abundance of wildlife sign along each 4.86 km segment along perpendicular transects. d) compares on-track relative abundance for bears to number of strikes. Hotspots are segments with a strike count above the 95% confidence interval (grey dashed line) and high risk segments are those with a risk estimate significantly above 1.0 (black points and error bars). Grey points are risk estimates not significantly different from 1.0 (the number killed was close to the number expected based on wildlife abundance). Segments proceed west to east, where 0 is the first 4.86 km inside the west boundary of Yoho National Park. The Field town site is located at segment 6, the Lake Louise town site corresponds to segment 13 and the Banff town site segment 24.	70
12. Annual train strikes for a) elk b) deer and c) bears along 134 kilometers of the Canadian Pacific Railroad through Banff and Yoho National Parks, 1989 to 2009.	72

LIST OF FIGURES - CONTINUED

Figure	Page
13. Segments with high strike risk labeled for (E) elk, (D) deer, and (B) bears during the 21-year period (1989 to 2009) along the Canadian Pacific Railroad through Banff and Yoho National Parks.....	75

ABSTRACT

Railroads, roads and associated traffic have been shown to adversely affect ecosystems by killing wildlife and altering the landscape. Relatively little research has been conducted along railroads. Given the probable growth of railroads, it is imperative that we understand the impacts railroads exhibit on wildlife. In this thesis, I reviewed the documented impacts of railroads on wildlife then conducted analyses on data collected along the Canadian Pacific Railroad (CPR) through Banff and Yoho National Parks (~134 km). In the study area, over 1000 train strikes with 26 mammal species have been recorded between 1990 and 2010, which included 579 elk (*Cervus elaphus*), 185 deer (*Odocoileus spp.*) and 79 bears (*Ursus spp.*). The goal of this research was to provide an initial assessment of the factors affecting strikes with ungulates and bears along the CPR. To accomplish these goals, I studied four general factors that have been hypothesized to affect the rate and spatial distribution of strikes. These are: wildlife abundance, anthropogenic foods, and railroad design. I compared strike rates along three mile long rail segments to train spilled grain, train and railroad design variables. I developed an estimate of risk using line transect data so that I could determine if there was evidence for nonconstant strike risk. Statistical models were used to identify which factors best explained strike rates. I detected correlations between the density of train-spilled grain and bear foraging rates but not with bear strikes. I identified locations where corrective measures or mitigation solutions may be needed and identified railroad designs and landscape variables associated with those locations. Hotspots were identified for elk and deer but not bears. Relative abundance was generally correlated with strike rates. High risk locations, where more strikes occurred than were expected, were identified. Train speed limit and right-of-way width was positively associated with strikes for elk and/or deer. For bears, the number of structures (e.g. highway overpasses) and bridges were positively associated with strikes. These results were used to suggest management recommendations including train speed reductions, habitat modifications and railroad design alterations to reduce the risk of strikes.

REVIEW OF LITERATURE: IMPACTS OF RAILROADS ON TERRESTRIAL WILDLIFE

Introduction

Understanding of the impacts of railroads on terrestrial wildlife is needed because new railroads and upgrades are planned (Horsley 2007; Welbes 2011). Many nations are attempting to develop environmentally sustainable transportation systems by increasing utilization of railroads. Railroads can be used to help reduce the overall impact of transportation on the environment because trains are efficient. Moving goods and people by rail can reduce carbon dioxide emissions and congestion on highways (Horsley 2007; Welbes 2011). Trains can transport up to 500 ton-miles per gallon (0.54 L/100 km for 0.89 tonnes), resulting in 2 to 5 times higher fuel efficiency than truck transport and use 21% less energy per passenger mile (Welbes 2011). Thus future development of railroads may yield some environmental benefits but impacts to terrestrial wildlife (hereafter: wildlife) should be acknowledged when constructing or upgrading railroads.

Railroads and roads have been shown to impact wildlife (van der Grift 1999; Forman et al. 2003). The primary impacts are: habitat alteration (e.g. habitat fragmentation), reduced wildlife movements (e.g. barrier effects) and wildlife deaths due to vehicle collisions (van der Grift 1999; Forman et al. 2003; Davenport and Davenport 2006). Human activity and related disturbances (e.g. land use change and hunting) are considered a secondary impact, although they may be the primary threat to some wildlife populations in some regions (Laurance 2004). These activities and other indirect impacts

may be affected by railroads (Rodgers et al. 1980) but are outside the scope of this review.

Most research on the ecological impacts of surface transportation infrastructure has been conducted on roads (Foreman et al 2003; National Research Council 2005; Davenport and Davenport 2006). Less is known about the ecological affects of railroads, although it seems logical that railroads and trains cause similar impacts to roads and vehicles and it has been suggested that railroads impact wildlife similar to low volume roads (Forman et al. 2003). However, some differences in the degree of impact and solutions to reduce those impacts likely exist. For example, habitat fragmentation effects due to railroads may be less than roads because railroad widths are narrower, railroad networks are generally less dense, and rail traffic volume less frequent. Furthermore, the solutions to reduce rail impacts will likely differ from those established along roads. For example, wildlife warning signs are probably ineffective along railroads because trains are unable to evade wildlife or stop quickly. Because of these differences, a review of railroad specific wildlife studies was needed. Although an initial literature review was conducted in 1999 by van der Grift, additional studies have been published since then (van der Grift 1999). The objective of this paper was to provide an current review of the documented impacts of railroads and rail traffic on terrestrial wildlife. This review aims to help direct and inform future wildlife research and to help identify rail specific mitigation solutions.

Methods

Peer reviewed literature, professional conference proceedings, and technical reports were identified by searching: the “Web of Science”, Google scholar and the “Road Ecology” eScholarship site at UC Davis in September 2011. Each bibliographic database was queried using the words: railroad, kill, strike, mortality, barrier, habitat, fragmentation, movement, crossing and travel. Queries were repeated using synonyms and translations. Additional literature, published prior to 1999, were obtained from one railway review paper (van der Grift 1999). Publications were evaluated based on the following questions: 1) what species were described? 2) In what country was the research conducted? 3) What impacts were described? Impacts were classified as: habitat, movement or mortality. 4) Last, each paper was examined to determine if causal factors or measures to reduce impacts were tested or proposed.

Results and Discussion

Publications from 19 countries in 50 studies were identified documenting railroad impacts on terrestrial wildlife (Table 1). A total of 115 wildlife species were described in the literature, ranging from the eastern box turtle (*Terrapene carolina*) to Asian elephants (*Elaphus maximus*). Wildlife impacts were studied on or along high-speed railways (n=7) in Spain, Mongolia and China (Rodriguez et al. 1996; Ito et al. 2005; Yin 2006)) with the remainder being freight lines with speeds less than 200 km/hr (124 mph).

Most publications originated from Canada (n=19) which has a rail network of approximately 48000 km (Transport Canada 2011). Other countries have more extensive

rail networks such as China and Russia (>91000 km), but only three papers were found from China and none from Russia. Colleagues at the Chinese Academy of Transportation Sciences helped identify studies from China, but no such measures were used to identify studies in Russia. Thus information from Russia and other countries may have gone undetected and their global distribution may be biased by the electronic search methods used.

Habitat

Wildlife are impacted by the changes to habitat that occur during the construction and operation of railroads. These changes include the removal of vegetation and installation of a surface suitable for trains (Carpenter et al. 1994). Train movements, noise and other factors also affect the ability of wildlife to use the landscape near railroads (Waterman et al. 2002; Ito et al. 2005). The amount of habitat impacted along a railroad, has been based on the width of a standard gauge rail line, 1,435 mm (4 ft 8.5 in), plus the right-of-way (ROW), totaling 15 m (49.2 ft) (Carpenter et al. 1994). Most railroads maintain a ROW or strip of semi-natural vegetation kept at retarded heights for safety and operational needs. In this paper “railroad” refers to both the foot-print and ROW (following Carpenter et al. 1994). However, this definition is problematic because the extent of habitat impacted is likely to differ by species, extend out past the ROW and/or operate at much larger spatial scales that could cause habitat fragmentation (Forman and Deblinger 2000).

Habitat impacts vary depending on the wildlife species considered. When land in a natural or semi-natural state is modified through railroad construction it is generally

considered impacted or destroyed because railroads persist on the landscape over long time frames (>100 yr). However, some species can continue to use modified habitats such as railroads (Li et al. 2010). The unique characteristics created by railroads may match some species habitat requirements or “niche” (van Bohemen 2005). For example, two species of lizard (*Lacertidae*) have been documented using railroads as habitat (Langton et al. 1997) and for reproduction (Blanke 1999). Thus, the habitat changes created by the studied railroad reported a positive effect on *Lacertidae* populations. Other species, that utilize edge habitats may also respond in ways that suggest railroads can benefit wildlife or have no impact.

Railroads are long linear features that often create habitat conditions and vegetation preferred by wildlife (Havlin 1987, van der Grift 1999). Studies have documented moose (*Alces alces*) using railroads to access seasonally available vegetation and habitat (Child 1983; Modafferi 1991; Andersen et al. 1991). Other species may use railroads as habitat due to a lack of predators (Harris 1977; Kolb 1984; Li et al. 2010). Forage and lack of predators was thought to be partially responsible for increased avian density and richness observed within 300 m of a high-speed railroad in China (Li et al. 2010). The species in this study appeared to successfully exploit the semi-natural habitat near a railroad and highway (Li et al. 2010). However, studies on birds from other regions have shown the opposite trend (decreasing density and richness) near railroads (Waterman et al. 2002) suggesting impacts may be habitat and species specific.

Wildlife use of railroads cannot simply be regarded as a positive effect. Use may indicate an impact because wildlife do not always choose “ideal” habitats defined as

those that maximize reproduction and survival (Beckmann and Berger 2003). On the contrary, some individuals may choose habitat sinks, where the rate of mortalities is greater than recruitment (Dias 1996). In these cases immigration from source habitats may maintain populations in sinks (Dias 1996). Individuals may choose these types of habitat to avoid competition or predation because optimal habitats are already occupied by dominant individuals or other species (Beckmann and Berger 2003; Mattson et al. 1990; Allen and Sargeant 1993). For example, less dominant grizzly bears (*Ursus arctos*) have been shown to use sub-optimal habitats to avoid more dominant individuals (e.g. adult males) and thus avoid predation but incur higher mortality rates (Mattson et al. 1990). Problematically, none of the studies identified in this review attempted to determine if wildlife preferred or selected railroads as habitat (by comparing use to availability) nor were able to compare demographic effects based on use of railroads as habitat. Thus it cannot be determined if wildlife were selecting railroads as habitat or if railroad habitat acted as a source or sink to populations. One approach to assess whether habitat use along railroads is impacting populations is to compare survival rates between areas with railroads to those without. Although difficult and costly, these types of studies are needed.

When the impacts of railroads on wildlife habitat are considered at larger spatial scales (e.g. national or international), it becomes clear that railroads divide unaltered habitat “patches” or fragment habitat (Forman et al. 2003). Habitat fragmentation results in two types of impact. First, the dividing of habitat into patches fundamentally alters the patch by decreasing the total patch area and increases the relative amount of edge habitat

for each patch (Forman et al. 2003). Secondly, fragmentation can affect the ability of wildlife to move between patches of habitat (Forman et al. 2003).

Generally, some wildlife species appear able to at least partially use the land altered by railroads. This may be due to the semi-natural substrate and/or narrow width and often low train traffic density. However, studies documenting habitat use have often documented other impacts such as increased mortality or altered movements associated with use of railroads as habitat, thus the benefits of habitat use needs to be considered relative to other impacts.

Movements

Reduced wildlife movements between patches of habitat have been shown to affect some species ability to access resources, migrate, and disperse (Skogland 1986; Dyer et al. 2002). Reduced movements have been shown to result from: species specific reactions to design (e.g. steel rails or fencing), lack of natural cover (e.g. gaps in forest canopy or other vegetation), or other conditions (e.g. sounds, sights, or human presence) (Fagan et al. 1999; Bartoszek and Greenwald 2009). Movement impacts have been documented for wary or limited mobility species, e.g. Mongolian gazelle (*Procapra gutturosa*) (Ito et al. 2005; Kornilev 2006; Bartoszek and Greenwald 2009). For example, Eastern box turtles are physically unable to cross standard gauge rails due to the design, which clearly blocks most movements across railroads (Kornilev 2006). Other wildlife groups such as ungulates, arboreal primates or amphibians may be physically able to cross but do not because of exposure to risky conditions during crossing (predation or thermal maximums) or perceived risk (e.g. wariness) (Groot Bruinderink and Hazebroek

1996). These behavioral factors were likely responsible for large scale movement impacts to Mongolian gazelles which did not cross a high-speed railroad during a two year study (Ito et al. 2005; Ito et al. 2007). Although Mongolian gazelles are highly mobile and no fencing prevented crossings they utilized only a portion of the possible habitat on one side of a railroad (Ito et al. 2005). For other species it is unclear what factors are responsible for movement impacts or if railroads impede movements.

Wildlife may utilize railroads for movements, which could be considered a positive impact. Wildlife move on or along railroads while foraging (Dyer et al. 2002), accessing critical resources (e.g. water) (Singh 2001), migrating (Dyer et al. 2002) and dispersing (Hedeen and Hedeen 1999; Gherghel et al. 2009). For example, foxes (*Vulpes vulpes*) have been documented using railways for quickly traversing their home range (Kolb 1984) and dispersing (Trehwella and Harris 1990). Wildlife tend to move along railroads for at least three reasons. First, railroads are often co-aligned with high quality habitats and natural movement corridors (e.g. valley bottoms and mountain passes) (Child 1983; Andersen et al. 1991; Wells et al. 1999). Secondly, wildlife may move along railroads because foods are available along ROW or on the rail bed (Wells et al. 1999). These include edge vegetation, carrion from strikes (Wells et al. 1999) and agricultural grains spilled from hopper cars (Wells et al. 1999; Waller and Servheen 2005). Lastly, wildlife may move along railroads because the flat rail bed provides an easily traversable route particularly apparent in regions receiving significant amounts of snowfall where railroad beds may offer a relatively snow free travel path (Child 1983; Andersen et al. 1991; Modafferi 1991).

In some cases wildlife movements may have been affected by the interaction between railroads and other landscape features such as parallel transportation infrastructure (Reh and Seitz 1990; Berthoud and Antoniazza 1998; Waller and Servheen 2005). These interactions may result in reduced movements across transportation corridors due to multiple physical barriers to movement, visual and auditory stimuli or increased chance of mortality (Skogland 1986; Vos et al. 2001). Multiple transportation features contributed to the genetic differences found for moor frogs (*Rana arvalis*) among breeding sites (Reh and Seitz 1990). The authors found the number of linear features including railroads and roads better explained the genetic structure in moor frogs than geographic distance (Reh and Seitz 1990). A study in Montana, USA also provides some evidence that transportation features interact in ways that affect wildlife movements. The authors found that grizzly bears crossed a transportation corridor when highway traffic volume was lowest but railroad traffic was highest (Waller and Servheen 2005). They noted that 23 grizzly bear mortalities have occurred on the railroad compared to two along a parallel highway in 22 years along 109 km of the Burlington-Northern Santa Fe railroad (Waller and Servheen 2005). In this case, the avoidance of highway traffic likely resulted in increased movements and mortalities to grizzly bears along the railroad.

Possible interactions between railroads and roads should not be ignored. However, co-aligned transportation features are likely preferred to limit habitat fragmentation in addition to gains in transportation efficiency. Co-aligned features likely fragment habitat less and enable mitigation, such as fencing and wildlife crossings, to encompass transportation corridors (Skogland 1986). Based on my review, wildlife

movements are not the largest impact of railroads on wildlife. Rather because many wildlife species move along or across railroads they are present to be struck by trains.

Mortality

Wildlife mortalities along railways typically occur when trains “strike” wildlife (Child 1983). However, other causes of mortality can be significant (e.g. electrocution). For example, over 100 badger (*Meles meles*) deaths were recorded in one month along a newly electrified railroad (Seiler 2005). Mortalities also occur when wildlife collide with wires or are trapped in equipment (Seiler 2005; Kornilev 2006). Strikes were the most common impact documented for 84 different species in 32 studies (Table 2.1). Of these studies, 34% described moose or grizzly bear strikes. Most studies reported strike rates per length of railroad over a given amount of time (Hunt 1987, Rodriguez et al. 1996, Wells et al. 1999). For example, 19 bird species were reported to be struck by trains at a rate of 0.47 km⁻¹ and 0.29 km⁻¹ annually along a railroad in the Czech Republic (Havlín 1987) but it was unclear what effect these mortalities had on individual species or overall populations in the region. It is difficult to determine the impact of strikes on populations without estimates of per capita mortality rates or population size. Results such as these are further complicated by other factors such as unknown detection probabilities and removal rates by scavengers (Hunt 1987; Rodriguez et al. 1996; Wells et al. 1999).

Lacking mortality rates relative to population size, some studies have reported the number of strikes relative to other sources of mortality (e.g. hunting or road-kills). This has been the case in numerous studies on bears, which have shown bear deaths along railways sometimes equal or exceed death rates on parallel highways (Boscagli 1987;

Kaczensky et al. 2003; Waller and Servheen 2005, Bertch and Gibeau 2010). Bear strikes were twice as frequent on railroads compared to roads and were second only to poaching kills in Italy (Boscagli 1987). In a study from Croatia, 70% of transportation related mortality for grizzly bears were due to strikes (Huber et al. 1998). However, this approach still fails to provide an answer to the question, do strikes impact populations? But it does enable identification of the relative importance of strikes on species that are detectable and reported.

Studies have demonstrated that some wildlife populations were affected by strike rates (Child 1983; Modafferi 1991; Andersen et al. 1991; Schwartz and Bartley 1991). For example, moose strikes resulted in a 70% decline of a moose population in Alaska (Schwartz and Bartley 1991). Moose strike rates as high as $5.5 \text{ km}^{-1} \text{ yr}^{-1}$ ($n=523$) have been recorded (Modafferi 1991). Similar strike rates in Alaska and British Columbia caused moose strikes to be incorporated into population based harvest quotas (Modafferi 1991; Andersen et al. 1991) to ensure viable populations were retained. These studies demonstrate train strikes can affect wildlife populations and show the utility of recording strike rates relative to local population abundance which is often missing in mortality reports (Wells et al. 1999).

Train strike rates in a given location are affected by multiple factors including both wildlife (abundance, behavior and foraging) and railroad (alignment, design and traffic volume) (Seiler 2005). The relative abundance of wildlife along a railroad may be the primary factor affecting strike rates. If more animals are present (higher abundance), more are likely exposed to be struck. Additionally, if a species selected railroads as

habitat or for movements, local abundance would be higher and strikes more likely. Strikes for ungulates and carnivores were most common where high quality wildlife habitat and railroads intersected (Lösekruug 1982; Modafferi 1991). High quality habitat appeared to be an indicator of higher abundance in these studies. Likewise, most studies have documented more herbivore than carnivore mortalities which reflects their relative abundance in most landscapes (Wells et al. 1999; Jêdrzejewski 2005; Joshi and Singh 2009; Joshi 2010). Overall mortality rates may be higher for ungulates because more are exposed. But carnivores may encounter railways more commonly because of their large home ranges, thus the relative impact to carnivore populations may be obscured by multi-species reports of strike rates (Wells et al. 1999; Joshi 2010).

Foods found on and along railroads may also be a factor affecting strikes by increasing the time wildlife spend directly on or adjacent to railroads. Foods found along railroads may consist of natural vegetation, carrion and agricultural products spilled from train cars. Agricultural products such as wheat, corn and peas (hereafter: grain) have been spilled during train derailments, leaked from grain hopper cars due to poor maintenance or spilled on the exterior of cars during loading (Huber et al. 1998; Wells et al. 1999; Waller and Servheen 2005). In Canada's Banff National Park, grizzly and black bears (*U. americanus*) have been noted foraging on grain during summer months (Gibeau and Heuer 1996; Wells et al. 1999; H. Morrison, Human-wildlife conflict specialist, Parks Canada. pers. com. 2008). In this region edge vegetation, such as that found along roads and railroads, has become scarce due to large scale vegetative changes (forest succession) resulting from long-term fire suppression (Gibeau and Heuer 1996; Pengelly and Hamer

2006). Wildlife species in this area, such as bears, that forage on vegetation, carrion and anthropogenic foods may be particularly susceptible to increased strike risk due to time foraging on multiple food sources found along or on railroads (Huber et al. 1998).

Some individuals within a species have been recorded to behave in ways that increase their likelihood of being struck (Rea et al. 2010). For example, moose have been recorded exhibiting a naïve flight response to trains (Child 1983; Rea et al. 2010). Some individuals have been recorded to flee from trains down track until they are overcome or attempt to charge on-coming trains (Child 1983; Child 1991; Rea et al. 2010). This behavior typically results in a strike. The behavioral response of some individuals may vary based on railroad or train variables. Studies have hypothesized railroad or train variables increase the probability of a strike by: reducing reaction times, e.g. train speed (Andersen et al. 1991; Modafferi 1991), constrain flight paths, e.g. bridges, (Andersen et al. 1991; Modafferi 1991; Kaczensky et al. 2003; Van Why and Chamberlain 2003) or reducing detectability, e.g. rock-cuts (Van Tighem 1981; Kaczensky et al. 2003). However, these variables may be associated with increased relative abundance and no study has clearly documented causal relationships between any of these variables and increased strike rates.

Understanding of the impacts of railroads on wildlife is just starting to emerge. Most studies simply report the presence/absence of wildlife use, movements or relative rates of strikes. These studies fail to clearly quantify the impacts to populations by conducting before-after control-impact (BACI) studies or measuring demographic affects associated with wildlife behavior near railroads. More research is urgently needed to

establish under what conditions significant ecological impacts occur and warrant mitigation.

Reducing Impacts

Four studies were found that tested reduction measures. All attempted to reduce moose strikes. Only two studies documented valid reductions in strike rates. The successful strategies included habitat alteration and supplemental feeding (Jaren et al. 1991; Andreassen et al. 2005). Habitat alteration was shown to result in a 56% reduction in moose-train strikes in an experiment where vegetation was mowed within 25 m of the rail line (Jaren et al. 1991). The decrease was recorded over an eight year period after which the authors conducted a cost-benefit analysis for all of Norway's railroads. Jaren et al. (1991) estimated vegetation removal should be conducted along railway segments with more than 0.3 km^{-1} moose strikes annually. Andreassen et al. (2005) also found similar reductions in moose-train collision rates (40–50%) due to habitat alteration (forest clearing) and supplemental feeding of moose. These studies may have been successful because they effectively reduced moose exposure to strikes by decreasing the probability moose would use habitat near railroads.

Other studies failed to find a decrease in moose strike rates due to a mitigation measure or had methodological problems. These methods included ultrasonic whistles, reduced train speeds and scent marking (Muzzi and Bisset 1990; Becker and Grauvogel 1991; Andreassen et al. 2005). In Alaska, reduced train speeds were tested but failed to reduce the likelihood of strikes. Speeds were reduced from 79 to 40 kmph (Becker and Grauvogel 1991). The authors noted that at some speed, strikes may decrease but these

speeds were cost prohibitive and were not tested further. Results from these studies suggest that reducing moose presence may be the most effective strategy to reduce moose strikes.

Others studies documented the use of wildlife crossing structures (WCS), including structures designed for wildlife, and culverts placed to maintain hydrologic flows under railroads (Hunt 1987; Rodríguez et al. 1996, 1997; Xia et al. 2007). Studies have shown that wildlife will use appropriately sized and designed crossing structures under railroads (Rodríguez et al. 1996, 1997; Pelletier et al. 2005). Crossing rates at WCS's are affected by other variables such as fencing, substrate and vegetation around entrances (Hunt 1987; Rodríguez et al. 1996, 1997; Xia et al. 2007). Authors have recommended crossing structures to reduce movement and mortality impacts (Rodríguez et al. 1997; Singh 2001). However, no study has evaluated the long-term effectiveness of WCS to reduce the movement or mortality impacts to wildlife along railroads. Only one study provided data on movements or strike rates with and without crossing structures or as part of a BACI study (Xia et al. 2007). Therefore it is difficult to determine if crossing structure use reduced impacts in other studies. A BACI study is underway along a high-speed railroad in China, thus this information should be available in the future for at least one species (Xia et al. 2007). Sections along the Qinghai-Tibet railway in China have been elevated where chiru (*Pantholops hodgsonii*) migrate. Initial data suggest these measures have provided sufficient movement to maintain annual chiru migrations (Xia et al. 2007). Studies similar to Xia et al. (2007) are needed that evaluate the effectiveness of mitigation measures.

Conclusion

Railroads may be a more environmentally sustainable form of transportation compared to roads (e.g. lower green house gas emissions). However, railroads and trains impact wildlife in the ecosystems they pass through. These impacts include direct wildlife mortality, habitat alteration and reduced wildlife movements. The changes railroads exhibit to the landscapes they pass through may result in positive effects for some species. Wildlife are able to persist near railroads, use railroads as habitat and for movements. However, determination of whether impacts are positive or negative may be more complex than some studies indicate by simply documenting presence or absence. Impacts such as direct mortality may result because wildlife use railroads and thus are exposed to strikes. Strikes were documented to affect population sizes for moose, but it is not clear if other species are impacted to the same degree. Strikes were the most common impact documented. However, the literature could be biased to documenting strikes because they are easily observed and may damage trains (Andersen et al. 1991). There may be a species reporting bias too. Wildlife that have economic or conservation value may be overly represented in the published literature, e.g. moose and bear. Therefore, it is difficult to determine if strikes are the primary impact or if these two species are most commonly affected. Mitigation measures have been directed at reducing strike rates for moose. Successful strategies reduced moose presence along railroads by modifying habitat. Additional mitigation strategies are needed for other species who may be using railroads for movements or as habitat.

More research is needed to fully understand the impacts of railroads on wildlife. Future studies need to provide a clear indication of the degree to which railroads impact wildlife populations. To accomplish this, future studies need to document demographic effects and survival rates for wildlife using railroads as habitat or for movements. Future studies need to provide an indication of the significance of strike rates on populations as a whole not just individuals along railroads. Railroads commonly occur near roads, however, few studies have documented the differences or potential interactions between different types of transportation infrastructure such as roads and railroads. Impacts along high-speed rail lines are of particular interest because these are the focus of new rail initiatives. It is unclear if impacts will differ along high-speed lines but some differences are likely due to differences in noise levels, speeds, and the common practice of fencing high-speed lines. Thus further research is needed to more clearly understand the impacts of railroads on wildlife, particularly along high-speed railroads.

Table 1. Impacts of railroads on terrestrial vertebrates. Impacts (x) were classified as: habitat, movement or mortality based on studies in peer reviewed literature, professional conference proceedings, and technical reports between 1960 and 2011.

Common name	Latin name	Habitat	Movement	Mortality	Reference
Reptiles and Amphibians					
Marbled salamanders	<i>Ambystoma opacum</i>		x		Bartoszek and Greenwald 2009
Spotted turtle	<i>Clemmys guttata</i>	x			Pelletier et al. 2005
Bank vole	<i>Clethrionomys glareolus</i>				Gerlach and Musolf 2000
Ladder snake	<i>Elaphe scalaris</i>		x		Rodríguez et al. 1996
Sand lizard	<i>Lacerta agilis</i>	x			Blanke 1999
		x			van Bohemen 2005
Ocellated Lizard	<i>Lacerta lepida</i>		x		Rodríguez et al. 1996
Common lizard	<i>Lacerta vivipara</i>		x		Avery 2009
Montpellier snake	<i>Malpolon mlonspessulanus</i>		x		Rodríguez et al. 1996
Iberian wall lizard	<i>Podarcis hispanica</i>		x		Rodríguez et al. 1996
Common wall lizard	<i>Podarcis muralis</i>				Gherghel et al. 2009
Psammodromus	<i>Psammodromus algirus</i>		x		Rodríguez et al. 1996
Moor frog	<i>Rana arvalis</i>		x		Allen and Sargeant 1993
			x		Vos et al. 2001
Common frog	<i>Rana temporaria</i>		x		Reh and Seitz 1990
Eastern box turtle	<i>Terrapene carolina</i>		x	x	Kornilev 2006
Snub-nosed viper	<i>Vipera latasti</i>		x		Rodríguez et al. 1996
Birds					
Northern goshawk	<i>Accipiter gentilis</i>			x	Leiva and Palacios 1997
Northern saw-whet owl	<i>Aegolius acadicus</i>			x	Wells et al. 1999
Mallard duck	<i>Anas platyrhynchos</i>			x	Leiva and Palacios 1997
Common swift	<i>Apus apus</i>			x	Leiva and Palacios 1997
Long-eared owl	<i>Asio otus</i>			x	Havlin 1987
Little owl	<i>Athene noctua</i>			x	Leiva and Palacios 1997
Ruffed grouse	<i>Bonasa umbellus</i>			x	Wells et al. 1999
Great horned owl	<i>Bubo virginianus</i>			x	Wells et al. 1999
Common hawk	<i>Buteo buteo</i>			x	Havlin 1987
				x	Leiva and Palacios 1997

Table 1 - Continued

Common name	Latin name	Habitat Movement Mortality	Reference
Red-necked nightjar	<i>Caprimulgus ruficollis</i>	x	Leiva and Palacios 1997
Killdeer	<i>Charadrius vociferus</i>	x	Wells et al. 1999
White stork	<i>Ciconia ciconia</i>	x	Leiva and Palacios 1997
Great spotted cuckoo	<i>Clamator glandarius</i>	x	Leiva and Palacios 1997
Rock pigeon	<i>Columba livia</i>	x	Havlin 1987
Stock pigeon	<i>Columba oenas</i>	x	Leiva and Palacios 1997
Wood pigeon	<i>Columba palumbus</i>	x	Leiva and Palacios 1997
Carrion crow	<i>Corvus corone</i>	x	Leiva and Palacios 1997
Rook	<i>Corvus frugilegus</i>	x	Havlin 1987
Jackdaw	<i>Coloeus monedula</i>	x	Leiva and Palacios 1997
Quail	<i>Coturnix coturnix</i>	x	Leiva and Palacios 1997
House martin	<i>Delichon urbica</i>	x	Leiva and Palacios 1997
Great spotted woodpecker	<i>Dendrocopos major</i>	x	Leiva and Palacios 1997
horned lark	<i>Eremophila alpestris</i>	x	Li et al. 2010
European robin	<i>Erithacus rubecula</i>	x	Leiva and Palacios 1997
Common kestrel	<i>Falco tinnunculus</i>	x	Havlin 1987
Pied flycatcher	<i>Ficedula hypoleuca</i>	x	Leiva and Palacios 1997
Bald eagle	<i>Haliaeetus leucocephala</i>	x	Wells et al. 1999
Barn swallow	<i>Hirundo rustica</i>	x	Leiva and Palacios 1997
Common wryneck	<i>Iynx torquilla</i>	x	Havlin 1987
Great grey shrike	<i>Lanius excubitorides</i>	x	Leiva and Palacios 1997
Black-headed gull	<i>Larus ridibundus</i>	x	Havlin 1987
Tibetan lark	<i>Melanocorypha maxima</i>	x	Li et al. 2010
Black kite	<i>Milvus migrans</i>	x	Leiva and Palacios 1997
Red kite	<i>Milvus milvus</i>	x	Leiva and Palacios 1997
White wagtail	<i>Motacilla alba</i>	x	Havlin 1987
		x	Leiva and Palacios 1997
White-winged snowfinch	<i>Montifringilla nivalis</i>	x	Li et al. 2010
Plain-backed snowfinch	<i>Montifringilla blandfordi</i>	x	Li et al. 2010
White-rumped snowfinch	<i>Montifringilla taczanowskii</i>	x	Li et al. 2010

Table 1 - Continued

Common name	Latin name	Habitat	Movement	Mortality	Reference
Rufous-necked snowfinch	<i>Montifringilla ruficollis</i>	x			Li et al. 2010
Eurasian scops owl	<i>Otus scops</i>			x	Leiva and Palacios 1997
Blue tit	<i>Parus caeruleus</i>			x	Havlin 1987
Great tit	<i>Parus major</i>			x	Leiva and Palacios 1997
House sparrow	<i>Passer domesticus</i>			x	Havlin 1987
				x	Leiva and Palacios 1997
Eurasian tree sparrow	<i>Passer montanus</i>			x	Leiva and Palacios 1997
Grey partridge	<i>Perdix perdix</i>			x	Havlin 1987
Common pheasant	<i>Phasianus colchicus</i>			x	Havlin 1987
				x	Leiva and Palacios 1997
Chiffchaff	<i>Phylloscopus collybita</i>			x	Leiva and Palacios 1997
European magpie	<i>Pica pica</i>			x	Leiva and Palacios 1997
Green woodpecker	<i>Picus viridis</i>			x	Leiva and Palacios 1997
Tibetan ground tit	<i>Pseudopodoces humilis</i>	x			Li et al. 2010
Collared dove	<i>Streptopelia decaocto</i>			x	Havlin 1987
Turtle dove	<i>Streptopelia turtur</i>			x	Havlin 1987
Tawny owl	<i>Strix aluco</i>			x	Havlin 1987
				x	Leiva and Palacios 1997
Spotless starling	<i>Sturnus unicolor</i>			x	Leiva and Palacios 1997
Blackcap	<i>Sylvia atricapilla</i>			x	Leiva and Palacios 1997
Common whitethroat	<i>Sylvia communis</i>			x	Havlin 1987
				x	Leiva and Palacios 1997
Sardinian warbler	<i>Sylvia melanocephala</i>			x	Leiva and Palacios 1997
Eurasian blackbird	<i>Turdus merula</i>			x	Havlin 1987
				x	Leiva and Palacios 1997
Song thrush	<i>Turdus philomelos</i>			x	Havlin 1987
Fieldfare	<i>Turdus pilaris</i>			x	Havlin 1987
Hoopoe	<i>Upupa epops</i>			x	Leiva and Palacios 1997
Mammals					
Moose	<i>Alces alces</i>			x	Andersen 1991
				x	Andreassen 2005
				x	Becker and Grauvogel 1991

Table 1 - Continued

Common name	Latin name	Habitat	Movement	Mortality	Reference
				x	Belant 1995
				x	Heerschap 1982
				x	Bertwistle 2001
				x	Child 1983
				x	Child 1991
				x	Child 1991
				x	Jaren et al. 1991
				x	Modafferi 1991
				x	Muzzie and bisset 1990
				x	Schwartz and Bartley 1991 1991
				x	Wells et al. 1999
				x	Rea et al. 2010
Wood mouse	<i>Apodemus sylvaticus</i>	x			Rodríguez et al. 1996
Coyote	<i>Canis latrans</i>			x	Bertwistle 2001
				x	Wells et al. 1999
Wolf	<i>Canis lupus</i>			x	Bertwistle 2001
				x	Wells et al. 1999
Roe deer	<i>Capreolus capreolus</i>			x	Havlin 1987
				x	Rodríguez et al. 1996
				x	Rodríguez et al. 1996
Camel	<i>Camelus dromedarius</i>			x	Rea et al. 2010
Beaver	<i>Castor canadensis</i>			x	Wells et al. 1999
Elk	<i>Cervus elaphus</i>			x	Bertwistle 2001
				x	Wells et al. 1999
				x	Rea et al. 2010
White-toothed Shrew	<i>Crocidura russula</i>		x		Rodríguez et al. 1996
Asian elephant	<i>Elaphus maximus</i>	x	x	x	Singh 2001
			x	x	Joshi and Singh 2009
Hedgehog	<i>Erinaceus europaeus</i>			x	Rodríguez et al. 1996
				x	Huijser et al. 2000
				x	Havlin 1987
Cougar	<i>Felis concolor</i>			x	Wells et al. 1999
Wildcat	<i>Felis silvestris</i>			x	Rodriquez et al. 1997

Table 1 - Continued

Common name	Latin name	Habitat	Movement	Mortality	Reference
Common genet	<i>Genetta genetta</i>	x			Rodriguez et al. 1997
Wolverine	<i>Gulo gulo</i>			x	Wells et al. 1999
European hare	<i>Lepus europaeus</i>			x	Havlin 1987
Brown hare	<i>Lepus granatensis</i>	x			Rodriguez et al. 1996
Iberian lynx	<i>Lynx pardinus</i>	x			Rodriguez et al. 1997
Badger	<i>Meles meles</i>			x	Bekker and Dekker 2009
Algerian mouse	<i>Mus spretus</i>	x			Rodriguez et al. 1996
European polecat	<i>Mustela putorius</i>			x	Havlin 1987
Mule deer	<i>Odocoileus hemionus</i>			x	Bertwistle 2001
				x	Wells et al. 1999
White-tailed deer	<i>Odocoileus virginianus</i>			x	Bertwistle 2001
				x	Wells et al. 1999
	<i>Odocoileus spp.</i>			x	Rea et al. 2010
Muskrat	<i>Ondatra zibethica</i>			x	Havlin 1987
Porcupine	<i>Ondatra zibethicus</i>			x	Wells et al. 1999
Mountain goat	<i>Oreamnos americanus</i>			x	Wells et al. 1999
European rabbit	<i>Oryctolagus cuniculus</i>			x	Havlin 1987
				x	Rodriguez et al. 1996
Bighorn sheep	<i>Ovis canadensis</i>			x	Bertwistle 2001
				x	Van Tighem 1981
				x	Wells et al. 1999
Leopard	<i>Panthera pardus</i>			x	Joshi 2010
Tibetan antelope	<i>Pantholops hodgsonii</i>	x			Xia et al. 2007
Mediterranean pine vole	<i>Pitymys duodecimcostatus</i>			x	Rodriguez et al. 1996
Mongolian gazelle	<i>Procapra gutturosa</i>			x	Ito et al. 2005
				x	Ito et al. 2007
Caribou	<i>Rangifer tarandus</i>			x	Wells et al. 1999
				x	Skogland 1986
Brown rat	<i>Rattus norvegicus</i>			x	Havlin 1987
Red squirrel	<i>Sciurus vulgaris</i>			x	Havlin 1987
Black bear	<i>Ursus americanus</i>			x	Bertwistle 2001
				x	Van Why and Chamberlain 2003

Table 1 - Continued

Common name	<i>Latin name</i>	Habitat	Movement	Mortality	Reference
Brown bear	<i>Ursus arctos</i>			x	Wells et al. 1999
				x	Bertch and Gibeau 2010
				x	Boscagli 1987
				x	Huber et al. 1998
Grizzly bear	<i>Ursus arctos</i>			x	Kaczensky et al. 2003
				x	Bertwistle 2001
				x	Gibeau and Heuer 1996
Red fox	<i>Vulpes vulpes</i>		x	x	Waller and Servheen 2005
				x	Wells et al. 1999
			x	x	Kolb 1984
		x	x		Trewhella and Harris 1990
			x		Rodriquez et al. 1997

RELATIONSHIPS BETWEEN BEAR (*URSIDAE*) FORAGING RATES, MORTALITY
AND TRAIN-SPILLED GRAIN IN BANFF AND YOHO NATIONAL PARKS

Introduction

Surface transportation features (e.g. roads and railroads) have been shown to negatively affect wildlife populations through direct mortality, habitat alteration, landscape fragmentation, and increased human access (Spellerberg 1998; van der Grift 1999; Forman et al. 2003; Farhig 2009). Most studies have described the adverse affects of roads on large mammals (Forman et al. 2003; Davenport and Davenport 2006). It has been suggested that railroad impacts on wildlife are similar to roads (Forman et al. 2003; Davenport and Davenport 2006). However, railroad specific studies are needed because railroads may impact wildlife differently (Spellerberg 1998; van der Grift 1999) and new railroads and upgrades are planned in North America (Horsley 2007; Welbes 2011).

Compared to roads, railroads may affect wildlife populations via direct mortality more than expected. Bears (*Ursidae*) have been struck at rates equal to, or higher than, those along roads in Canada, Croatia, Slovenia and the United States (Boscagli 1987; Huber et al. 1998; Waller 2005; Bertch and Gibeau 2010a). In Canada's Rocky Mountain National Parks train strikes are the leading source of human-caused grizzly bear (*U. arctos*) mortality averaged over the past 20 years (Bertch and Gibeau 2010a). Trains strike and kill bears at a rate of 0.35 year⁻¹ for grizzly bears and 1.95 year⁻¹ for American black bears (*U. americanus*) within Banff and Yoho National Parks along 134 km of the Canadian Pacific Railroad (CPR) (Bertch and Gibeau 2010a; Bertch and Gibeau 2010b).

Based on the frequency of train strikes, grizzly bear populations are likely negatively impacted within the Rocky Mountain National Parks (Benn and Herrero 2002; Hebblewhite et al. 2003).

In order to maintain persistent bear populations it is crucial to minimize human causes of mortality, particularly in the Rocky Mountains (Craighead et al. 1974; Benn and Herrero 2002) because bears in this region have lower than average reproductive rates (Garshelis et al. 2005). Across their range in North America, bear populations are strongly affected by mortality rates (Mattson et al. 1996; Weaver et al. 1996) because they are long lived and have low reproductive rates (Craighead et al. 1974; Garshelis et al. 2005). Ursids have some of the lowest reproductive rates of all terrestrial mammals in North America (Craighead et al. 1974;). Grizzly bears in the Rocky Mountains have lower reproductive rates compared to populations elsewhere partially due to food scarcity (Beckmann and Berger 2003; Garshelis et al. 2005). Banff and Yoho National Parks have been considered to contain low to moderately productive habitat for grizzly bears compared to other areas where high protein foods like salmon or large ungulates are more abundant (Mattson 1996; Hilderbrand 1999; Gibeau et al. 2001). Food scarcity in this region has been shown to cause reproductive stress in grizzly bears as demonstrated through decreased litter size, increased age of first litter and increased time interval between litters (Elowe 1989; Hilderbrand 1999; Garshelis et al. 2005). Therefore, understanding is needed of bear foods and mortality sources to maintain persistent bear populations (Herrero et al. 2000).

Bears have been observed consuming train-spilled grain along the CPR (Wells et al. 1999; H. Morrison, Human-wildlife conflict specialist, Parks Canada. pers. com. 2010). Agricultural grains leak from grain hopper cars onto the rail bed (Wells et al. 1999). These grains include: wheat (*Triticum aestivum*), barley (*Hordeum vulgare*), canola (*Brassica napus*), pea (*Pisum sativum*) and corn (*Zea mays*). The grain products are shipped from the interior of Canada and the United States to ports on the west coast each year. Although grain is a valuable commodity, hopper cars leak due to improper loading, operation, or faulty discharge gates (Wells et al. 1999; Waller and Servheen 2005). These three factors result in a slow release of grain from hopper cars as they travel through the Rocky Mountains. Additionally, large point accumulations of grain or “piles” occur at temporary stops, such as road crossings and sidings due to the stopping and starting of leaking cars. Wheat and barley are the primary grains spilled and have been called “high quality food” for bears based on average crude protein (9-13%) and starch (54 to 73%) content (Mattson 1990).

Studies in Banff and Yoho National Parks have described the foraging habits of both grizzly and black bears. However, no study has documented or quantified bears feeding on wheat or barley, possibly due to a general lack of spatial overlap between bear distributions, study areas and major grain cultivation areas (Mattson 1990). Both species in this area have been shown to maximize energy intake by foraging on many different foods including vegetation, insects, mammals and fish (Hamer and Herrero 1987; Raine and Kansas 1990). Bears switch foraging efforts between different foods primarily because the availability of preferred forage items such as buffaloberry (*Shepherdia*

canadensis), vary in quantity and quality throughout each year. Some highly preferred foods are usually consumed when encountered (e.g. carrion or fish) but other foods may only be consumed when they are highly abundant. Bears switching between food types can be detected by several indicators including scat contents or visual observations (Hammer and Herrero 1987; Raine and Kansas 1990; Munro 2006).

Three observations suggest bears have used train-spilled grain as a food source along the CPR. Bear scats have been observed that contained grain along the right-of-way (S. Michel, Human-wildlife conflict specialist, Parks Canada. pers. com. 2008). Bears have been observed slowly moving between the rails in a head-down posture indicative of foraging and the stomach contents of bears killed by trains have contained grain (H. Morrison, Human-wildlife conflict specialist, Parks Canada. pers. com. 2008). However, no study has assessed bear foraging rates on grain, nor the relationships between grain and mortality events (train strikes). To conclude that bear mortality rates are affected by foraging on grain assumes the likelihood of trains striking a bear is altered by one or more of the following. Exposure time: the time bears are exposed (standing between the rails) is longer when grain is present, thus strikes are more probable (Waller 2005). Vigilance: when a bear is feeding on grain, it is less vigilant making a strike more probable (Dyck 2004). Tolerance: bears become more tolerant of on-coming trains over time, in which case flight responses or reaction distances may decrease making a strike more probable (Herrero et al. 2005). These three factors may be contributing to mortality events. However, exposure time is the clearest and easiest measure of both bears response to grain and risk of being struck by trains.

The time bears spend on the CPR was used to test for bears response to grain, and to test for a relationship between grain and strike events because previous observations of bears foraging on grain have been incidental and could be misleading. For example, the perception that bears are foraging on grain could be biased. Railroads generally run parallel to highways, as a result bears are more easily seen by passing motorists on railroads than away from railroads. Bias could also exist due to the low digestibility of grains. Thus grain would appear common in scats relative to other food items (Bunnell 1983).

Based on current understanding of bear foraging behavior, three different relationships between grain spill and foraging rates may be detectable (MacArthur and Pianka 1966). If bears were not switching to foraging on grain, one might expect no relationship between grain density and the rate bears consume grain (no relationship). If switching was occurring, more grain would likely be consumed when more grain was available (positive relationship). Alternatively, if grain was the only or most preferred food, bears may forage longer when grain was less abundant because bears may attempt to maintain a constant energy intake although search and handling time increased (negative relationship) (MacArthur and Pianka 1966).

The goal of this research was to increase understanding of the role train-spilled grain plays in some grizzly and black bears foraging habits and determine whether foraging behavior might explain the spatial pattern of strikes. To accomplish this goal, the relationships between the amount of grain spilled on the CPR rail bed was compared to the time bears spent foraging on grain, and to bear mortality using an historic data set,

over multiple temporal and spatial scales. Three specific hypotheses were evaluated: H₁) Bears foraged for longer durations when grain spill volume was higher (temporal). H₂) Bears foraged more where grain spill volume was higher (spatial). H₃) Train strikes were more frequent where bears were observed feeding on grain (spatial). The results of these tests were used to suggest management options to reduce bear foraging on grain and possibly train related bear mortalities.

Methods

Study Area

The study was conducted along a 134 km section of the Canadian Pacific Railroad (CPR) running from the west boundary of Yoho National Park, British Columbia (116°39' W, 51°14' N) to the eastern boundary of Banff National Park, Alberta (115°25' W, 51°8' N) (Figure 1). An average of 19 trains day⁻¹ pass through the study area at an average speed of 60 km hour⁻¹ (CPR unpublished data 2010). The Trans-Canada Highway runs parallel to the CPR throughout the study area at an average distance of 416 (±325 m) from the CPR. A 2.4 m high wildlife exclusion fence, to prevent wildlife vehicle collisions, exists along the highway for 48 km starting at the eastern extent of the study area (Clevenger et al. 2001). There were 27 wildlife crossing structures passing over or under the highway at the time of this study, with additional fencing and structures under-construction east of Lake Louise, Alberta (Figure 1), one of the structures was a shared wildlife crossing and railroad underpass. An additional section (28 km) of fencing

was completed in 2010 along with an additional shared wildlife-rail underpass (Clevenger et al. 2009).

The CPR crosses the Continental Divide at 1,670 m above sea level west of Lake Louise (9 km). The CPR right-of-way is approximately 60 m wide consisting of one track except for an 8.3 km long section near Lake Louise, Alberta where two tracks exist which diverge to a maximum of 500 m. Due to large elevation gradients in the area (> 2000 m), vegetation types vary and average annual rainfall decreases from west to east due to a rain-shadow affect. Approximately 39% of the land area in Banff and Yoho National Parks is at or above tree line (~2300m). Where vegetation was present, it consisted of closed and open canopy forest, dominated by lodgepole pine (*Pinus contorta*), white spruce (*Picea glauca*), and subalpine fir (*Abies lasiocarpa*) (Holland and Coen 1983). Vegetation within the CPR right-of-way was kept in an early successional stage by mechanical and herbicide treatments and consisted primarily of: herbaceous vegetation (grasses, sedges and rushes) (*Poaceae*), dandelions (*Taraxacum* spp.), bearberry (*Arctostaphylos uva-ursi*) and horsetail (*Equisetum* spp.). Numerous species of shrub including buffaloberry, and tree species including lodgepole pine and white fir were present at reduced heights (<2 m). Vegetation between the rails was almost non-existent due to herbicide treatments and the rocky substrate. Where vegetation was present between the rails there was small herbaceous growth of native and non-native grasses or young sprouts of train-spilled agricultural products (cereal grains).

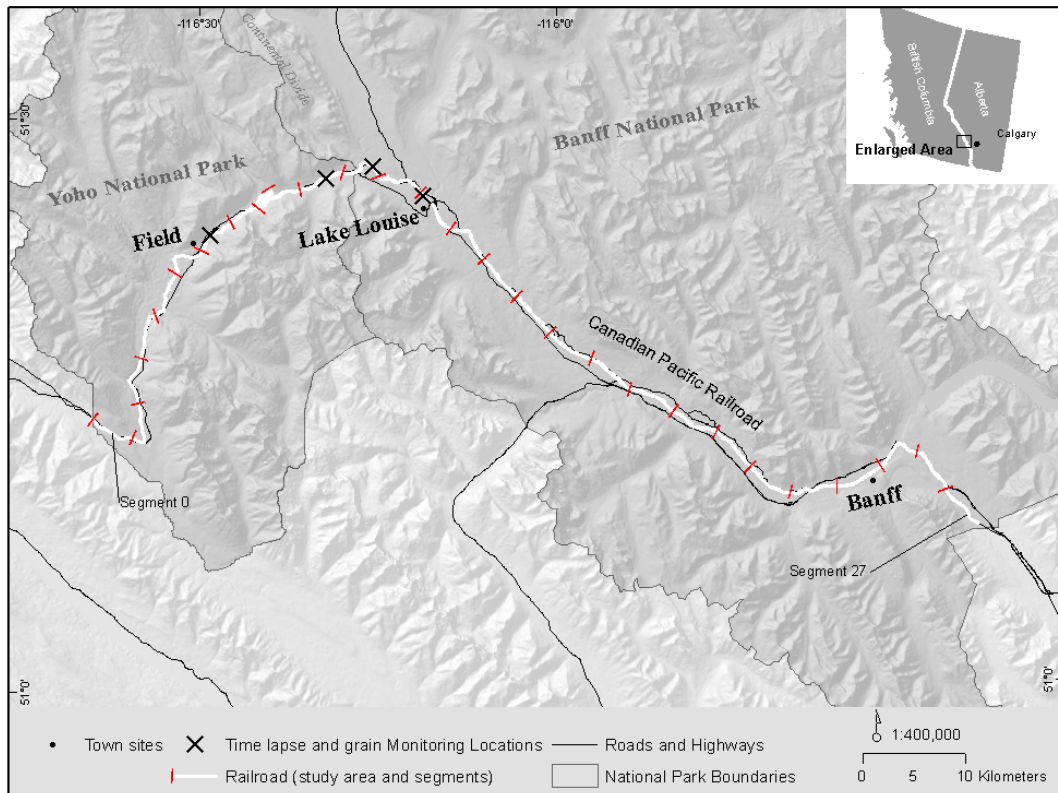


Figure 1. Map of the 134 kilometers of the Canadian Pacific Railroad study area that traverses through Banff and Yoho National Parks in the Canadian Rocky Mountains. Each analysis segment (4.86 km) is shown denoted by a small perpendicular bar along the railroad.

Data Collected

Two data sets were collected. One was a repeated measures dataset collected from a small geographic portion of the study area (<25 km) that included weekly train traffic, bear foraging rates, and the amount of grain spill information. The second dataset was spatially explicit and was collected over the entire study area (134 km). The spatial dataset included the number of bear strikes, bear foraging rates and amount of grain.

However, the spatially explicit data could not be collected within a sufficiently short timeframe to eliminate all temporal fluctuations in the response (<1 month). Therefore, the temporal data were used to correct for seasonal trends in the spatial data collected over a four month period. Consequently, each spatial observation was weighted according to the date it was observed in time.

A spatial dataset representing the railroad was acquired from the CPR Company then imported into a geographic information system (GIS). The layer was separated into 30 roughly equal segments approximately 4.86 km (3 miles) in length. The segment length was based on a conservative estimate of the minimum mapping accuracy of the wildlife strike records. Wildlife strike records were tallied to the appropriate 4.86 km long segment in a GIS. Segments inside of tunnels were omitted. A total of 28 segments were used in the analysis.

Wildlife Strike Records. Counts of train-struck black and grizzly bears that occurred between 1989 and 2009 were acquired from Parks Canada (Parks Canada unpublished data 2010). These records included all strikes reported by CPR and Parks Canada personnel. Strikes were visited and removed to avoid double counting. Records after 2001 were spatially referenced with a Garmin® global positioning system (GPS) but previous to 2001, records were referenced to permanent mile-marker posts or to geographic features along the ROW. As a result some strike records were spatially inaccurate by as much as 1.6 km (1.0 mile) thus an error of plus or minus 1.6 km was assumed for all records. Date and time of each strike was present in the data but the accuracy of strike times varied: some were recorded accurately as the date and time of the

strike, while others were recorded as the time and date the report was completed within Parks Canada, which may have lagged behind the actual strike by days (S. Wrazej, Resource Management Technician, Parks Canada. pers. com. 2010). Therefore, these data were excluded from temporal analyses.

Temporal Foraging Data. Temporal foraging data were collected using four Reconyx® color-infrared and true color still cameras (Figure 2a). Each camera took a photograph every three minutes between the hours 0400 h to 2300 h (Figure 2b) and recorded the time and date of the image. The five hour period between 2300 h and 0400 h was omitted because at night the infrared flash was unable to illuminate objects greater than 10 m away, which would underestimate night-time use. Bears were identified in the camera images when they were within 200 m of the camera. However, it was difficult to differentiate between black and grizzly bears at greater distances. Therefore, the two species were grouped for the analysis. The cameras were active from 12 May to 30 July in both 2009 and 2010. Each camera was attached to a tree at least 2 m off the ground, adjacent to the CPR such that it had an unobstructed view of the railroad tracks. Cameras were located between Lake Louise and Field, BC within 2 km of a grain monitoring station (Figure 1).

Two variables were derived from the images: the number of foraging events and the length of each event (Figure 2b). A foraging event was classified as the date and time when a bear was first photographed between the rails in a head-down posture which was assumed to indicate the bear was feeding on grain.

a)



b)



Figure 2. a) Reconyx® time-lapse camera b) A time-lapse photograph of a bear (*Ursus* spp.) exhibiting the foraging behavior on the Canadian Pacific Railroad near Field, B.C.

A foraging event was considered to end when the bear stopped exhibiting the foraging position or left the field of view for at least 15 minutes because a bear re-entering the field of view could not be confirmed to be the same individual. The duration of each foraging event was defined as the number of images that the bear remained in the field of view exhibiting behavior consistent with feeding on grain multiplied by three (time between images). When multiple bears e.g. adult bears with cubs, were photographed they were treated as one bear.

Temporal Grain Spill Data. The weight of grain spill over time was measured at four monitoring stations established in a systematic sampling pattern across the target portion of the study area (Figure 1). These data were part of a larger grain spill monitoring study (unpublished data, Parks Canada 2010). At these monitoring stations, the grain sampling device consisted of a plastic rectangular screen placed on the rail bed that covered the entire width between the tracks (0.37 m² area; 1 x 4 ft) (Figure 3a) held down with four metal weights. Monitoring was initiated in 2008 and conducted for all months of the year. However, for these analyses only data collected between 1 May and 30 August 2009 to 2010 were used because they were compared to temporal foraging data collected only during this period. At least once a week, the grain that accumulated on each screen was collected, sifted to remove debris, dried, and weighed in grams. Multiple tracks occurred at two of the monitoring stations, thus data were collected from both and summed. The total grain weight per week was used in analysis. Grain weights were converted to a weekly deposition rate for correction of spatial grain data.

Temporal Train Traffic Data. The frequency and arrival time of trains were collected to evaluate fluctuations in train traffic over time using a Reconyx® motion activated digital camera aligned perpendicular to and 5 m from the railroad. The date and time of each passing train was recorded from 1 May to 30 November for the years 2008 to 2010. The camera operated 24 hours a day during the entire sampling period and was located near the center of the study area west of Lake Louise (Figure 1). Data from the camera was representative of the entire area because the railroad has no branches along the 134 km study area.

Spatial Foraging Data. Bears foraging on grain was estimated by counting scat found on the railroad along strip transects. Scats were counted on the rail bed, a 3 m wide area centered between the two rails. Randomly selected 4.86 km long segments of track were walked by the same observer between 10 May and 24 September and repeated each in 2008 and 2009. When a scat was encountered within a randomly selected segment, the location of each scat was recorded using a GPS unit (Trimble Navigation Ltd., Sunnyvale, USA) and the totals binned to the appropriate 4.86 km long segment in a GIS. A visual estimate of the age, contents, diameter and distance to the next nearest scat was evaluated in the field to determine when multiple scats likely represented one foraging event by one individual. When the diameter, age and contents were similar for scats found within 5 m, only one scat was recorded. All scats detected were removed from the CPR to avoid double counting. Three scats were left *in situ* early in the sampling period that were verified to be detectable at the end of sampling year 2008. Where multiple tracks were encountered, e.g. sidings, all tracks were surveyed and the mean number of

scats detected per track were used in the analysis. All scat that did not contain grain were removed from the statistical analysis.

Scats were weighted based on the date they were detected. The first day when bears have been historically sighted in the study area is 15 April. Although the date of bears emergence varies each year, it was used as a start date for when bears could have been foraging on grain and left scats. As each day passed after 15 April there were cumulatively more days when bears could have visited and left scats. Therefore, each scat was given a weight inversely proportional to the day of year it was detected after 15 April, which is the 105th Julian calendar day of the year. For example, the weighted value of one scat detected on 30 September, (day 273 of the year) would be $[(273-105)^{-1} = 0.006]$. The final value was the sum of all weighted scats detected for each track segment stored as BearTRACKDEN_{total}.

Spatial Grain Spill Data. The density of grain spilled was measured at: 1) random points across the study area and 2) within 1 m of each detected bear scat. Two hundred random points were generated at the beginning of each year and navigated to using a GPS during transect surveys. The sampling device consisted of a 10 cm² frame placed on the rail bed (Figure 3b). The frame was tossed blindly between the rails, and then the visible wheat and barley seeds inside the frame were counted. Three replicate tosses and counts were conducted within a 5 m segment of track. At locations with multiple tracks, the sampling process (three subsamples) was repeated for each track. The average wheat and barley count was calculated from all tosses. The grain count was affected by the date it was measured because grain spill decreased over the course of a

summer. Using the temporally explicit grain monitoring data described above, counts were weighted by the week of year they were observed. The weighted value was calculated as the mean seed count multiplied by the average rate of decrease (3%) for each additional week after 15 April. Because grain spill weights were 97% of the previous week for each week after 15 April for both years, a weighted seed count of 10 seeds for 20 June (four weeks after 15 April) was the seed count multiplied by $[10 * (0.97^4) = 38.8]$. All grain spill data were binned to the appropriate 4.86 km long segment in a GIS and averaged.

Data Analysis

The temporal relationships between bears foraging on grain, spilled grain weight, train volume, and train strikes were compared by hour of the day and week of year. X^2 tests were used to compare distributions of hourly foraging and train traffic. To evaluate if grain spill or foraging rates changed by sampling year, analysis of variance (ANOVA) was used.

The first of three primary hypotheses (H_1), bears respond to increasing amounts of grain by increasing foraging rates, was tested by evaluating the fit and parameter estimates of two linear mixed models (Pinheiro and Bates 2000). The first model compared the average foraging event duration to weekly grain spill weight. This model theoretically assessed the length of time bears foraged once they discovered grain. A second model fit the weekly total foraging time (sum of all event times) to weekly total grain spill weight, which may better represent overall foraging effort.

a)



b)



Figure 3. Grain spill sampling methods used on rail bed. a) Grain spill monitoring screen covering a 0.37m^2 area between the tracks, used to estimate the weekly weight of grain spilled at four locations. b) Spatial grain spill sampling frame (10cm^2), used to estimate the weekly weight of grain spilled at four locations. b) Spatial grain spill sampling frame (10cm^2), used with three replicate random throws to estimate the mean density of wheat and barley seeds within a 5 m zone.

The second model was not as strongly affected by the biologically arbitrary but logistically necessary use of 15 minutes to define separate foraging events. Year was incorporated into the models as a fixed effect and week as a random effect (Zurr 2010). The null hypothesis implied by model fitting was that there was no relationship between total foraging time and quantity of grain. The null hypothesis was rejected when significant fit was apparent in a model and all predictors as indicated by a significant F test ($p < 0.05$). Both models used log-log transformations of the predictor and response, thus medians were produced by taking the exponent of model coefficients.

The two remaining hypotheses utilized spatially explicit data to determine if relationships between bear foraging behavior, grain spillage and strikes could be detected. The hypothesis (H_2), bear foraging was positively correlated to grain spill, was evaluated at two spatial scales. At the fine scale (5 m) grain spill density was compared between locations where scats were found and random locations using ANOVA. Secondly, at the rail segment scale (4.86 km), grain spill density was compared to the density of scats using generalized linear models (GLM). The final hypothesis (H_3), bear strikes were positively correlated to grain spill, was tested at the rail segment scale by evaluating the fit and parameter estimates of a linear model that compared the number of train strikes (collected over 21 years) to grain density and density of scats. Spatial auto-correlation was assessed but not accounted for in either model, because no spatial correlation was apparent in the residuals (Zuur 2010). The statistical software program, R (2005) was used for all analysis with an alpha level of 0.05 for significance tests. Linear

mixed models were fit using the “lme()” function in the nlme package version 3.1 (Pinheiro and Bates 2009) in R (2005).

Results

Temporal

Bears were detected in the foraging position by at least one of the four time-lapse cameras on 110 days out of a possible 632 camera days over both years in 2009 and 2010. A total of 368 foraging events were recorded and the mean duration was 28.7 ± 1.48 minutes with foraging events lasting a maximum of 3 hours and 18 minutes, and minimum of 3 minutes (photo lapse time). Bears were detected foraging a total of 176 hours out of 12008 hours of camera time. Total foraging time was not evenly distributed over the day-time hours sampled $X^2(1, n=18) = 38.99, p = 0.003$ with more total time apparent in mornings and evenings, and longer durations in morning hours (0400 to 1200) relative to the other hours cameras were active (Figure 4a and 4b).

It was unclear if train volumes or arrival times affected foraging duration or total foraging time. A motion activated camera revealed that trains arrived in the study area at all hours of the day $X^2(1, n=24) = 4.94, p > 0.1$ (Figure 4c). Between 0400 h and 1100 h, arrival times were uncorrelated to total foraging time $r(17) = -0.32, p = 0.18$ and to foraging duration $r(17) = -0.08, p = 0.74$ (Figure 4a and 4c). The average interval between trains was 85 ± 3.39 minutes and foraging events lasting this long only occurred 5% of the time. Thus, bears were foraging at the same times that trains passed through the

study area. Approximately 50% of trains passed within three minutes, thus it could not be determined how often bears returned to foraging after a train passed.

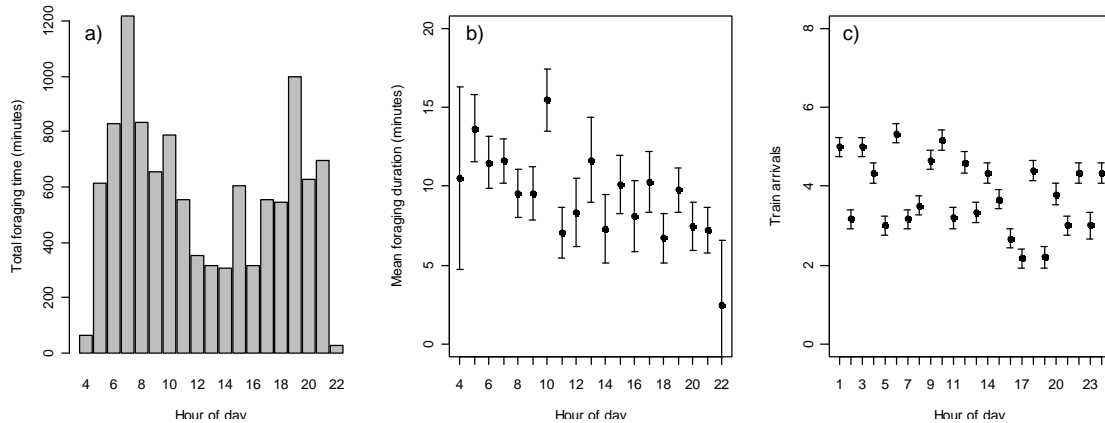


Figure 4. (a) Total weekly foraging time by bears on train-spilled grain by hour of day and, (b) mean foraging duration in minutes by bears on train-spilled grain by hour of day, recorded at four time-lapse camera stations on the Canadian Pacific railroad between Field and Lake Louise between 12 May to 30 July 2009 and 2010 (c) Mean train arrival times over a 24 hour period for April to November 2010 on the Canadian Pacific Railroad determined from a motion activated camera located between Field and Lake Louise.

Between 2009 and 2010 average weekly grain spill ($n=29$) decreased 62% from 0.26 ± 0.04 to 0.10 ± 0.02 g/0.37m² ($F(1,27) = 10.64$, $p < 0.005$) (Figure 5a). During this time, there was no detectable decrease in total foraging time (Figure 5c) and only a marginal decrease in mean foraging duration of 2.76 ± 1.13 min ($F(1,21) = 3.98$, $p = 0.056$) (Figure 5b). No correlations were found between foraging duration and grain amount ($r(27) = 0.38$, $p = 0.07$) nor between foraging time and grain weight ($r(27) = 0.21$, $p = 0.33$) (Figure 6a and 6b). After accounting for week and year effects neither model, total foraging time ($F(3, 25) = 0.47$, $p < 0.49$) nor foraging duration ($F(3,25) = 0.04$, $p <$

0.84) indicated a significant relationship with grain spill. Thus these data did not provide sufficient evidence to support the hypothesis (H_1) that bears foraged longer when more grain was available. The lack of significance may have been because total foraging time and average foraging duration were confounded with week of year. For this reason the amount of grain spill was removed from the models to explore correlations between the two response variables and week of year. Week of year was important because it allowed estimation of foraging relative to the onset of summer and possibly the availability of other forage items (spring green up of vegetation). Both models showed relationships with week and year. However, only total foraging time was linearly related to week within years ($F(3,25) = 12.11, p < 0.001$). The parameter estimate for week of year in the log scale was $\beta = -0.19 \pm 0.05$ ($p < 0.001$) with a y-intercept of 9.19 for 2009 and 15.81 for 2010 ($p < 0.05$). The rate of decrease differed significantly by year (for 2010 $\beta = -0.51 \pm 0.12$ ($p < 0.05$)) (Figure 7a and 7b).

Spatial

Grain spill and bear scats were found across the entire study area with at least one scat detected on all but four rail segments. On average there were 12 ± 3.98 scats per rail segment, of which 90% were observed to contain wheat or barley. Grain spill density varied spatially across the study area ($X^2(1, 27) = 220.29, p < 0.001$) and was positively correlated with bear scats at two spatial scales. The mean grain density was 39% higher within 5 m of bear scats containing wheat and barley than at random sample points ($F(1, 374) = 61.04, p < 0.001$) (Figure 8).

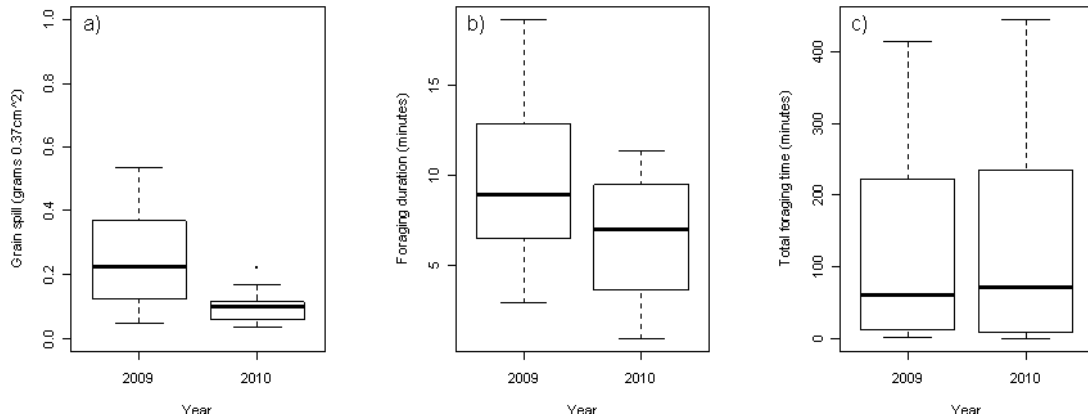


Figure 5. a) Weekly grain spill weight measured in grams per 0.37 m² between Field and Lake Louise in spring/summer) during 2009 and 2010. b) Average foraging duration and c) total foraging time (log minutes) on train-spilled grain detected at four time-lapse camera stations on the Canadian Pacific Railroad between Field and Lake Louise between 5 May to 30 July in 2009 and 2010.

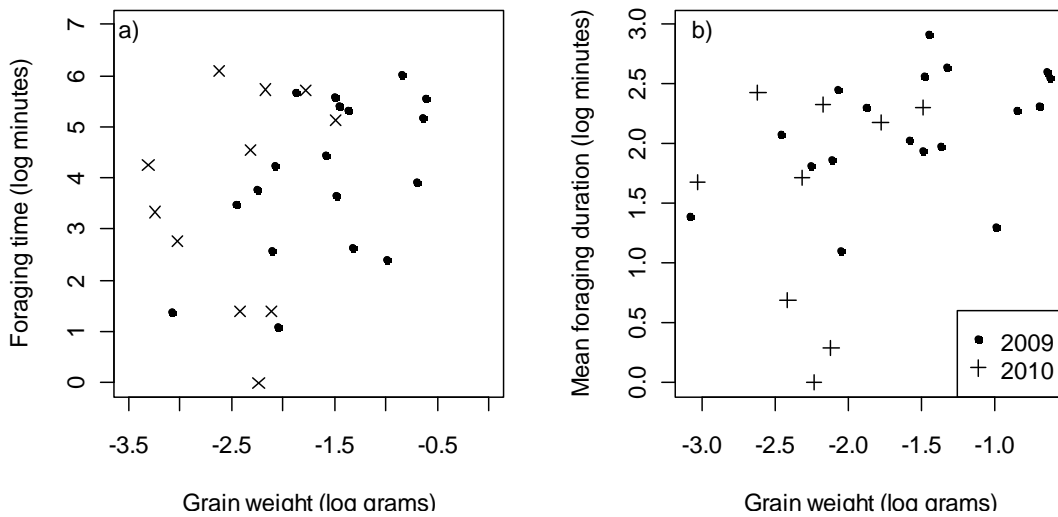


Figure 6. (a) Total weekly foraging time (log minutes) on train-spilled grain compared to total grain weight (log grams) on a log-log scale. (b) Mean foraging duration (log minutes) on train-spilled grain compared to log grain weight. Foraging time and duration was recorded using four time-lapse camera stations, and grain spill was recorded at monitoring stations on the Canadian Pacific Railroad between Field and Lake Louise in 2009 and 2010.

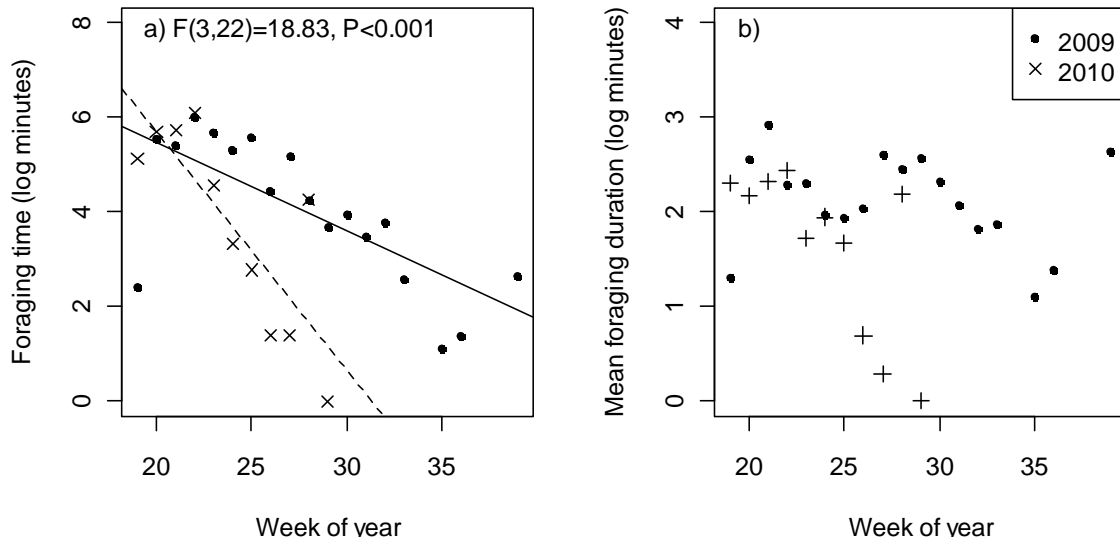


Figure 7. (a) Foraging time in log minutes and (b) average foraging duration in log minutes estimated using time-lapse photography recorded at four camera stations on the Canadian Pacific Railroad between Field and Lake Louise between 5 May to 30 July in 2009 and 2010.

At the rail segment scale (4.86 km), there was a positive relationship between the quantity of scats and grain ($r^2_{adj}=0.48, F(1,26) = 10.9, p < 0.001$) (Figure 9a). The linear (log-log) model estimated that one additional scat was detected when the median density (10cm^2) of wheat and barley seeds increased by 2.80 ± 0.73 ($t(18)= 4.29, p < 0.001$) (Figure 9). These data supported the second hypothesis (H_2) that grain spill density was positively correlated to bears foraging on the CPR as indicated by the density of scat.

A total of 79 bear strikes were recorded over the 21-year period from 1989 to 2009 along the 134 km study area. These included 9 grizzly and 69 black bear and one unknown bear. At the rail segment scale, these strike locations were not correlated to either grain density ($r(26)=0.30, p > 0.1$) nor bear scat density ($r(26)=0.13, p > 0.5$). These

data did not support the final hypothesis (H_3), that bears were more commonly struck by trains where more grain was spilled or where more foraging occurred (Figure 9b and 9c).

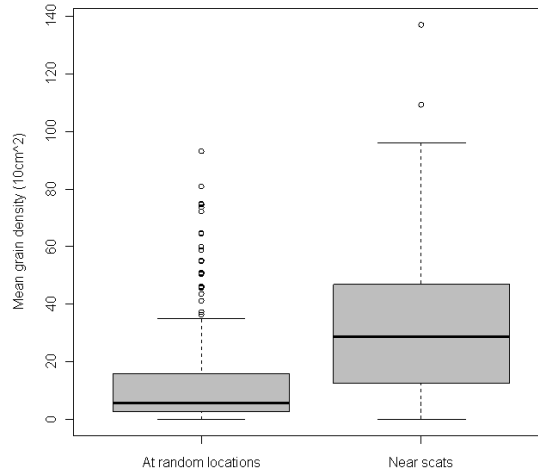


Figure 8. Grain spill measured as mean counts of wheat and barley seeds within a 10cm² frame at random locations and within 1 m of bear scats detected.

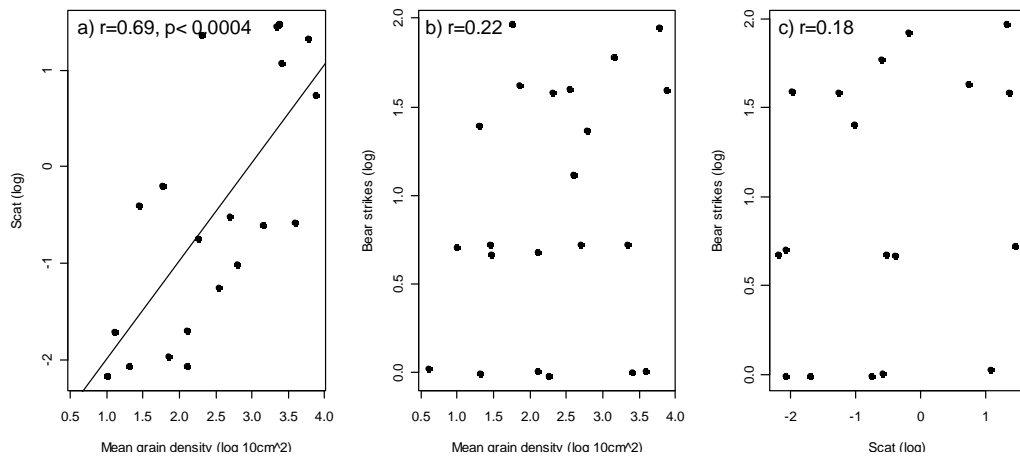


Figure 9. Spatial relationships between a) number of bear scats and grain density, b) train bear collisions (strikes) and grain density, and c) train bear strikes and bear scat density on the Canadian Pacific Railroad from the east entrance of Banff National Park, Alberta to the west boundary of Yoho National Park, BC in 2008 and 2009 summarized by 4.86 km long rail segments.

Discussion

Two separate data sets revealed that bears were foraging on grain both frequently in time and over a large geographic area along the CPR. A 62% decrease in grain spill and 30% decrease in foraging duration were detected between 2009 and 2010. The decrease in grain spilled could be explained by the repair of over 1,000 grain hopper cars during 2009. If the decrease in foraging duration in 2010 was due to reduced grain spill then further hopper car repairs could further reduce the time bears spend on the CPR. Alternatively, this may have been due to random variation in the sampling period. Further research is needed to confirm this possible relationship.

Total foraging was highest during weeks of the year when grain spill was highest. However, within a year (e.g. weekly) neither overall foraging time nor the mean duration of foraging events was significantly related to grain spill. Foraging on grain generally decreased for each successive week of the year, although some foraging continued in the highest spill locations through-out the year. It was unclear if bear foraging time decreased within a year because of a decrease in spilled grain or because other foods became available that were associated with spring vegetation green up. These results suggested that grain was not the most preferred food item. Other studies have documented the natural foods consumed during spring, which included *Hedysarum* spp., *Equisetum* spp., graminoids, dandelion spp. and ants (Hamer and Herrero 1987; Raine and Kansas 1990; Auger 2004; Munro 2006). *Hedysarum* has been shown to be consumed by Ursids in the study area during the weeks of highest foraging on grain (Hamer and Herrero 1987). It is possible that some bears in this study selected grain over *Hedysarum* which may have

been because of lower search time and lower energy expenditure needed to consume grain (compared to digging needed to consume *Hedysarum* roots), but we did not evaluate this. *Hedysarum* has also been shown to appear less in bear scats as summer progressed (Hamer and Herrero 1987; Raine and Kansas 1990). Other studies have suggested bears preferred other foods to *Hedysarum* that became available as summer progressed (Hammer and Herrero 1987). These results suggest that although some bears preferred grain over the other foods available in early spring; either more preferred foods became available or grain decreased to a level that other foraging opportunities were preferred as summer progressed.

Spatially, increased signs of foraging (scat) were found with increased grain density at two spatial scales (5 m and 4.86 km). However the density of grain filled bear scats was not correlated with train strikes over the previous 21 years at the 4.86 km scale. This may have been because other factors affect the probability of strikes or because bears foraging in one location in early spring adopted foraging and movement behavior that resulted in a strike along a different segment. One other factor that likely affected strike probability was train speed. At one location west of Lake Louise where both grain spill and foraging rates were highest, trains move no faster than 15 km hour^{-1} . Train strikes may be unlikely at this location because bears could escape from the railroad before being struck. *Post hoc*, examination of the data indicated other spatially explicit variables could also have effected foraging intensity. For example, the six segments with the highest number of bear scats (Figure 9a) coincide with areas of dense forest between the railway and highway. High foraging intensity in these areas was likely related to

decreased traffic visibility, noise or other disturbance factors. Also along three rail segments passing through town-sites, bears were hazed from the town-sites and in some cases the railroad to prevent human-bear and bear-human conflict. These factors may have obscured relationships between grain, foraging and strikes if they existed.

Both grain sampling methods had methodological faults that likely underestimated grain spill. The primary fault of both methods was that neither prevented wildlife from removing grain from the sampling area before it could be measured. Second, when grain spill was high, CPR employees occasionally used a rail mounted vacuum to remove grain. The sampling devices also failed to capture large point accumulations of grain or “piles”. These piles occurred in different locations over time but never on the sampling device. No data were collected to determine grain removal rates by CPR or the frequency of grain piles. The spatial estimate of grain density also tended to underestimate grain spilled but in a slightly different way compared to the temporally explicit data. At least three other crop seeds were usually visible inside the sampling frame. Using the count of only wheat and barley seeds underestimated the total amount of grain available but helped resolve other methodological problems. Other seeds had different detectability, for example canola seeds were usually present at a higher density but were difficult to count accurately because they were smaller and dark colored and similar to soil and stone particles on the rail bed. These factors made canola difficult to count accurately. Other seeds such as pea and corn were large and highly visible which may have increased their detectability but also likely affected their removal rates by wildlife. Using only wheat and barley seeds which were average in size relative to the

other seeds present and easily visible both on the rail bed and in bear scats was an attempt to control for different seed detectability. These methodological problems increased sampling efficiency but may have introduced measurement error known to bias significance testing and obscure relationships (Lahor 1999).

The differences between species and individual behavior of bears must be acknowledged. Although some generalities exist for both species, individual grizzly and black bears exhibit different habitat selection and behavior between as well as within species. Further, differences in behavior and habitat selection vary based on: sex, age, reproductive status, or past foraging experiences. Regardless of species, some bears may exploit high quality habitats close to transportation features or human use areas that other individuals may use less frequently or avoid. Access to these areas may provide reduced competition and high food rewards, but incur a higher likelihood of mortality (Gibeau et al. 2001; Benn and Herrero 2002; Nielson 2004). Relative to detecting foraging responses to grain, behavioral heterogeneity is problematic and was acknowledged but could not be accounted for in this study.

Management Implications

This study revealed that both black and grizzly bears were foraging on grain along the CPR. Bears foraged more in locations with higher grain density in the two years data were collected. Bears also spent more time foraging on grain when grain density was highest and few other forage items were available. Together these relationships indicate grain is key component resulting in some bears moving and foraging on the CPR. The marginal decrease in foraging duration detected over two years may indicate bears spend

more time on the CPR in years when more grain is available. However, other factors could have also affected foraging durations such as the onset of spring (i.e. availability of other forage). Further reductions of train-spilled grain would likely reduce foraging, which may lessen the probability of bears becoming food conditioned or passing on unnatural foraging behavior to offspring (Mazur and Seher 2008). Although these analyses could not directly relate the relationship between grain foraging and mortality, other studies have shown foraging on human foods increased mortality for bears (Hammer and Herrero 1987; Hebblewhite et al. 2003). Reductions in grain spillage may reduce the probability of individual bears becoming tolerant of humans and transportation infrastructure because the CPR is co-aligned with multiple roads and areas of high human use (Benn and Herrero 2002). If bears spend less time in these areas known to be associated with increased mortality, they may be less likely to be killed through management operations and/or road-kills (Martinka 1974; McLellan 1999). These important factors in bear behavior may affect overall mortality rates but not strike rates, thus making success difficult to measure or detect (Cole 1974; Herrero 1976; Mazur and Seher 2008).

Continued monitoring of grain spill both temporally and spatially along with an indicator of foraging effort by bears is needed. These data could be used to evaluate the effectiveness of future management efforts and to verify short-term trends observed in these analyses. Some modifications and additions to the methods described above are recommended. First, grain sampling methods need to exclude wildlife foraging. This could be accomplished by using a tray or box like device with a wire mesh covering.

Secondly, annual and seasonal monitoring of natural bear forage items is needed to compare the relative importance of grain to natural forage items (see Latham 2011). This could be accomplished by using line transect methods or monitoring known foraging sites and species as recommended by Latham (2011). Lastly, individual and species specific response rates to grain are needed to evaluate if response rates differ by species or life stage (e.g. reproductive age females). These data need to be combined with survival and fecundity to determine the role of grain on bear population dynamics. Tracking of known individuals such as reproducing females via GPS collars may be the best approach to determine how grain affects a bears foraging strategy and subsequent survival.

These methods and data were only able to account for one type of mechanism, increased exposure (time spent on the rail bed) and could not account for possible alterations of vigilance or habituation over time. When a bear feeds on grain, it is receiving a food reward (food conditioning). For food stressed bears in the Canadian Rocky Mountains, unnatural foods are likely a pathway to increased tolerance to noise, movements of trains and other human infrastructure (see Hopkins et al. 2010). As a result, the CPR may act as a pathway for exposure to numerous other conditioning and learning stimuli that increases the likelihood of mortality. The strength of this pathway is unknown, but may be an important component of continued bear mortality and human food conditioning in bears (Hopkins et al. 2010).

Natural bear foraging behavior has been restored by reducing or controlling access to anthropogenic foods in other national parks and protected areas (Craighead et al. 1974; Cole 1974). However, rapid elimination of unnatural foods has been associated

with decreased survival and reproduction of bears (Knight and Eberhardt 1985; Stringham 1986), thus a gradual reduction over time is needed. If grain spill is reduced, creation of alternative foraging habitat may be necessary to reduce the possibility of negative nutritional consequences. Alternative foraging habitat created through land management operations (e.g. prescribed burning) could provide two benefits. First, it could act as a precautionary measure due to the chance that bears may simply increase foraging time to maintain a constant intake of grain, particularly when few other forage items are available (MacArthur and Pianka 1966). Secondly, it could relieve food stress in this region and help reinforce natural foraging habits away from transportation features. This habitat would need to provide temporally relevant foraging opportunities, ideally during early spring (April-May) when this study observed higher foraging rates on the CPR.

RELATIVE RISK AND VARIABLES ASSOCIATED WITH BEAR AND
UNGULATE MORTALITIES ALONG A RAILROAD IN THE CANADIAN ROCKY
MOUNTAINS

Introduction

Like roads, railroads affect wildlife through direct mortality, habitat loss and habitat fragmentation (van der Grift 1999; Forman et al. 2003; Davenport and Davenport 2006). Direct mortalities result when trains strike wildlife. Strikes can be a significant source of mortality for some wildlife populations and have been reported for decades (Child 1983; Gundersen et al. 1998; Bertch and Gibeau 2010a). Studies have reported high strike rates for large mammals such as grizzly bears (*Ursus arctos*) (Bertch and Gibeau 2010a; Bertch and Gibeau 2010b) and moose (*Alces alces*) (Child 1983; Child et al. 1991; Modafferi 1991). Ungulate species may be struck more commonly due to their higher densities relative to other species on most landscapes. However, carnivores such as bears (*Ursus spp.*) are particularly susceptible to negative impacts from transportation infrastructure such as railroads and highways because they are highly mobile, occur at low densities and have low reproduction rates (Weaver et al. 1996; Fahrig and Rytwinski 2009). Thus per capita or population level impacts may occur although strike rates are lower, compared to ungulates.

Bears may be particularly susceptible to mortality from trains but it is unclear why. Bears have been struck at rates equal to, or higher than, those along roads (Boscagli 1987; Huber et al. 1998; Waller and Servheen 2005; Bertch and Gibeau 2010a). In

Canada's Rocky Mountain National Parks train strikes are the leading source of human-caused grizzly bear mortalities averaged over the past 20 years (Bertch and Gibeau 2010a). Trains strike and kill bears at a rate of 0.35 year⁻¹ for grizzly bears and 1.95 year⁻¹ for black bears within Banff and Yoho National Parks (Bertch and Gibeau 2010a; Bertch and Gibeau 2010b). Here, train strikes were the second largest source of mortality for black bears, deer (*Odocoileus spp.*), elk (*Cervus elaphus*) and moose. These species were most commonly killed on roads (unpublished data, Parks Canada 2009; Bertch and Gibeau 2010a). The true mortality rates due to train strikes are likely higher than reported. For example, as little as 50% of strikes with large mammals were reported by standard observers (train engineers) along the Canadian Pacific Railroad (CPR) during a six year period (Wells 1999). In other cases, strikes may not be reported except when large groups (>450) of wildlife are killed (Chaney 2011) or when strikes occur near protected areas (Waller and Servheen 2005). Long-term data of train strikes along the CPR exist because strikes have been reported to Parks Canada for at least 30 years. However, other railroads may not report or record strikes. Therefore, it is important to identify the variables affecting the number of animals killed where data does exist.

Studies on roads have analyzed the spatial pattern of road-kills, which showed that road-kills occurred in clusters or hotspots (Finder et al. 1999; Clevenger et al. 2003; Malo et al. 2004). The spatial pattern of road-kills has been explained by landscape, environmental or infrastructure variables (Finder et al. 1999; Hubbard et al. 2000; Gunson et al. 2005). These studies have helped inform management actions targeted at reducing road-kills (Clevenger et al. 2001; Grilo et al. 2009). At least five general factors

are thought to affect the spatial pattern of road-kills and train strikes (Seiler and Helldin 2006). Along railroads these include: animal variables (e.g. wildlife abundance and behavior), train variables (e.g. train speed and frequency), railroad design variables (e.g. curvature or alignment), landscape variables (e.g. vegetation type and habitat quality) (Huber et al. 1998; Bashore et al. 1998; Finder et al. 1999; Seiler and Helldin 2006). Driver behavior variables are largely removed from train strike analyses because trains generally cannot stop or swerve to avoid animals. In some cases, landscape variables consisted of land cover or vegetation type data that are associated with habitat quality (Bashore et al. 1985; Finder et al. 1999; Roger and Ramp 2009); these have been the best predictors of road-kill rates which has suggested estimates of relative abundance are needed when assessing the spatial pattern of road-kills and train strikes (Roger and Ramp 2009).

Along the CPR, wildlife abundance, train and railroad design varies which have likely affected the spatial distribution of train strikes. Clearly, where wildlife abundance is high, strikes over time may also be high because more individuals have been exposed to train strikes. Studies have shown strikes coincided with locations where wildlife abundance was high along railroads such as at moose wintering areas or on migration routes (Gundersen et al. 1998; Ito et al. 2007). Train variables and railroad design vary along the CPR and may also affect strike rates. If train or railroad variables altered the probability of a strike; strike rates would vary spatially along with differences in these variables even if wildlife abundance were constant. There are at least three theoretical mechanisms that could alter the spatial probability of strike occurrence. These

mechanisms, based on wildlife behavior relative to train or railroad variables: 1) constrained flight paths - some design features like bridges may restrict wildlife movements on the rail bed surface instead of out of the path of oncoming trains; 2) reduced detectability - some design features may impede the sight and/or sound of oncoming trains, therefore wildlife are less likely to detect trains resulting in increased strike rates; 3) reduced reaction time - higher train speeds may reduce the time available to wildlife to successfully cross or flee before being struck. These mechanisms likely interact. For example, with increasing train speeds both the time to detect and appropriately flee may be reduced.

There is evidence that these mechanisms affect the rate of train strikes with ungulates and bears along railroads. Bears have been recorded to be struck at bridges and rock cuts (constrained flight paths) and in locations where the sound of oncoming trains was reduced (reduced detectability) (Van Why 2003; Kaczensky et al. 2003). Some ungulate strikes occurred in Alaska because moose did not flee off of the railroad tracks. The authors suggested deep snow alongside the railroad constrained moose to flee down the track (Andersen et al. 1991; Modafferi 1991). Similarly for bears, engine mounted video footage suggested some bears were struck while fleeing from trains where track-side slopes or other design variables appeared to obligate flight paths on the tracks instead of off and out of danger (CPR unpublished data). Other associations are unclear, for example bighorn sheep (*Ovis canadensis*) strikes have been associated with rock cuts and may be a result of fine-scale habitat selection or multiple design mechanisms acting concurrently (Van Tighem 1981). As a result, it may be difficult to determine the

mechanism(s) driving strike rates. However, identifying the animal, train or railroad variables associated with strikes and the direction of the effect (e.g. increased mortality at a particular design feature) may be a more tractable approach and could indicate ways to reduce strikes.

The purpose of this research was to identify locations where species specific mitigation solutions are needed and identify variables associated with those locations. Four ungulate species (elk, mule deer [*Odocoileus hemionus*], white-tailed deer [*O. virginianus*] and two bear species (grizzly and black bear) were used for this analysis. Strike rates were hypothesized to be correlated to relative abundance for each species along the CPR. These rates were hypothesized to also be affected by nine train and railroad design variables (Table 2).

Methods

Study Area

The study was conducted along a 134 km section of the CPR running from the west boundary of Yoho National Park, British Columbia (116°39' W, 51°14' N) to the eastern boundary of Banff National Park, Alberta (115°25' W, 51°8' N) (Figure 1). An average of 19 trains day⁻¹ pass through the study area at an average speed of 60 km hour⁻¹ (unpublished data, this study 2009). The Trans-Canada Highway runs parallel to the CPR throughout the study area at an average distance of 416 (±325 m) from the CPR. A 2.4 m high wildlife exclusion fence, to prevent wildlife vehicle collisions, exists along the highway for 48 km starting at the eastern extent of the study area (Clevenger et al. 2001).

There were 27 wildlife crossing structures over or under the highway at the time of this study, one of the structures was a shared wildlife crossing and railroad underpass. An additional section of fencing (28 km) was completed in 2010 along with wildlife crossing structures and a shared wildlife railroad underpass (Clevenger et al. 2009).

The CPR crosses the Western Continental Divide at 1670 m above sea level 9.0 km west of Lake Louise. The CPR consists of one track except for an 8.3 km long section near Lake Louise, Alberta where two tracks diverge to a maximum of 500 m. Due to large elevation gradients in the area (> 2000 m), vegetation types vary and average annual rainfall decreases from west to east due to a rain-shadow effect. Approximately 39% of the land area in Banff and Yoho National Parks is at or above tree line (~2300 m). Where vegetation was present, it consisted of closed and open canopy forest, dominated by lodgepole pine (*Pinus contorta*), white spruce (*Picea glauca*), and subalpine fir (*Abies lasiocarpa*) (Holland and Coen 1983). Vegetation within the CPR right-of-way (ROW) was kept in an early successional stage by mechanical treatments and consisted primarily of herbaceous vegetation (grasses, sedges and rushes) (*Poaceae*), dandelions (*Taraxacum* spp.), bearberry (*Arctostaphylos uva-ursi*) and horsetail (*Equisetum* spp.). Numerous species of shrub including buffaloberry (*Sheperdia canadensis*) and tree species including lodgepole pine and white fir were present at reduced heights (<2 m). Vegetation between the rails was almost non-existent due to herbicide treatments and the rocky substrate. Where vegetation was present between the rails there was small herbaceous growth of native and non-native grasses or young sprouts of train-spilled agricultural products (cereal grains).

Data Collected

Wildlife strike records Counts of train-struck ungulates and bears were obtained from Parks Canada for a 21-yr period (1989 to 2009) (Parks Canada, unpublished data, 2010). These records included all strikes reported by CPR and Parks Canada personnel. Strikes were visited and removed by Parks Canada personnel to avoid double counting. Records after 2001 were spatially referenced with a Garmin® global positioning system (GPS), but previous to 2001, records were referenced to permanent mile-marker posts or to geographic features along the CPR. As a result some strike records were spatially inaccurate by as much as 1.6 km (1.0 mile) thus an error of plus or minus 1.6 km was assumed for all records. A spatial data layer representing the railroad was acquired from CPR then imported into a Geographic information system (GIS) data . The spatial layer was separated into 30 roughly equal segments 4.86 km (3 miles) in length. The segment length was based on a conservative estimate of the minimum mapping accuracy of the wildlife strike records. Wildlife strike records were tallied to the appropriate 4.86 km (3 mile) segment in a GIS. The lengths of track inside of two tunnels and one secondary track west of Lake Louise were omitted from analyses. A total of 28 segments were used in the analysis.

Relative abundance data were collected by sampling transects oriented parallel and perpendicular to the CPR. These were called “rail bed transects” and “perpendicular transects” respectively. Both methods were binned to 4.86 km long rail segments, thus data were analyzed at the same spatial scale. The relative abundance of bears was sampled using both methods. However, ungulates were sampled only on perpendicular

transects. The relative abundance of ungulates could not be estimated on rail bed transects because fecal pellets were either disturbed via snow clearing on the rail bed before they could be counted or did not represent their distribution when they were most commonly struck. More than 80% of ungulate strikes occurred in December and January (Wells 1999). Thus summer pellets on the rail bed did not represent relative abundance during winter because ungulate distributions shifted in summer months.

Parallel transects were centered directly on the rail bed. The same observer walked each segment (n=28), on the rail bed, once in 2008 and 2009 between 10 May and 24 September. Segments were treated as a strip transect 3 m wide and 4.86 km long, centered on the rail bed along which the locations of bear scats were recorded using a GPS unit (Trimble Navigation Ltd., Sunnyvale, USA). Black and grizzly bear scats were classified as “bear” because they could not be differentiated in the field. When the diameter, age and contents were visibly similar for scats found within 5 m, only one scat was recorded. All scats detected were removed from the CPR to avoid double counting. Three scats were left *in situ* early in the sampling period. These were verified to be detectable at the end of the sampling period in year 2008. Where multiple tracks were encountered, e.g. sidings, all tracks were surveyed and the mean number of scats detected per track was used in the analysis. Differing from Chapter 2, scats were used in this analysis regardless of grain contents. In Chapter 2, only scats containing grain were used.

Perpendicular transects were used to estimate relative abundance for bears and ungulates within the railroad corridor (< 250 m). To determine where to sample, two random points were generated within each rail segment in a GIS for each sampling year.

Each point was used as the starting location for a perpendicular transect. At each point, a transect extended out perpendicularly away from the CPR in both directions for 250 m. Transects were 3 m wide. Pellet groups and scat found more than 1.5 m from the transect center were not recorded. Perpendicular transects were truncated at the edge of impassable features such as rock cliffs, rivers and open water. Along each transect ungulate pellet groups were classified into those of elk or deer based on differences in size and shaped described by Elbroch (2003). White-tailed deer and mule deer pellets were combined into one class “deer” because they could not be reliably differentiated between in the field. The number of pellet groups was counted for each transect. Bear scats were recorded using the same criteria as the rail bed transects. Pellets and scat that appeared to be more than one year old based on visible signs of decay (bleached color or unconsolidated) were excluded as were amorphous ungulate or bear feces (Elbroch 2003). All transects were walked by one observer from 1 May to 31 August during 2008 and 2009.

The count of bear scats were weighted based on the date they were detected (same as chapter 2). The first day when bears have been historically sighted in the study area is 15 April. Although the date of bears emergence varies each year, it was used as a start date for when bears could have been foraging on grain and left scats. As each day passed after 15 April there were cumulatively more days when bears could have visited and left scats. Therefore, each scat was given a weight inversely proportional to the day of year it was detected after 15 April, which is the 105th Julian calendar day of the year. For example, the weighted value of one scat detected on 30 September, (day 273 of the year)

would be $[273-105^{-1} = 0.006]$. The final value was the sum of all weighted scats detected for each track segment stored as $RA_{\text{rail bed}}$.

The relative abundance for each species was defined as the number of pellet groups or weighted value km^{-1} within each 4.86 km long segment (i) (Eq. 1). For example the relative abundance of elk, $RA_{\text{rail corridor}}(i)$ was the number of elk pellet groups km^{-1} averaged over all transects within each rail segment (i).

$$RA_{\text{rail corridor}}(i) = \frac{\sum \left(\frac{\text{pellet groups}_t}{\text{transect length}_t} \right) * 1000}{\sum_{t=1}^t} \quad (\text{Eq. 1})$$

Railroad Design and Train Spill Grain Data. Three additional variables were measured at the starting location of each perpendicular transect. The mean right-of-way width (ROWmean) was measured using a Bushnell® Yardage Pro® 1000 range finder (Bushnell Corporation, Denver, CO) which extended to the forest edge or open water. A maximum distance of 500 m was imposed in locations where open habitat and level ground resulted in an unidentifiable ROW corridor. Topography adjacent to the railroad was classified into three categories similar to those used to assess wildlife-vehicle collisions on the parallel Trans-Canada Highway (Clevenger et al. 2003; Gunson et al. 2005) (VERGESLOPE). The categorical variable consisted of: (0) flat, (1) raised and partly raised, and (2) buried, buried-raised, and part buried. Train-spilled grain was also measured at each transect starting location on the rail bed. Grain density was estimated visually by counting the wheat and barley seeds inside a frame (10 cm^2) placed randomly three times on the rail bed within a 5 m segment of track (Figure 10). The sampling process was repeated for each track where sidings were present. Grain density was

calculated as the average of all subsamples within a rail segment ($\text{GRAIN}_{\text{mean}}$). Grain density was affected by the date it was measured because grain spill decreased over the course of a summer (see chapter 2). Therefore, grain density was weighted by the week of year it was measured. For both years, the weighted value was calculated as the mean seed count multiplied by the average rate of decrease (3%) for each additional week after 15 April. For example, a weighted seed count of 80 wheat and barley seeds measured on 20 June (four weeks after 15 April) was $[80 \times (0.97 \times 4) = 310.4]$. Grain density data were binned to the appropriate 4.86 km long segment in a GIS.



Figure 10. Grain spill sampling method used on rail bed. The grain spill sampling frame (10cm^2), used with three replicate random throws to estimate the mean density of wheat and barley seeds within a 5 m zone.

The GIS layer representing the railroad contained additional information on train speed limits, bridges, sidings and track grades. The highest posted train speed limit ($SPEED_{max}$) and mean track grade ($GRADE_{mean}$) were calculated for each segment. The variable *SINUOSITY* was calculated using the “sinuosity” function in Hawth’s analysis tools extension for ArcGIS 9 (ESRI Redlands, CA 2004) . The number of bridges were counted and summed for the variable hereafter termed $BRIDGE_c$. A count of highway vehicle overpasses, tunnels, snow sheds or rock cuts occurring along each segment were each given a value of “1” and summed for the variable hereafter called $BARRIER_c$.

Data Analysis

The following analyses were conducted independently for elk, deer and bears. Strikes that occurred over the 21-yr period were compared to relative abundance estimated using data covering a 2-yr period (2008 and 2009). This comparison assumes relative abundance across the study area remained relatively stable during the preceding years. To assess this assumption, I looked for changes in the distribution of strikes over time and assessed the stability of analysis results using a 10-year period. A better approach would be to assess changes in relative abundance over time, but these data were not available for more than half of the study area for all species studied. Changes in the distribution of strikes over time were assessed by conducting ANOVA on year and rail segment. Next, to determine if strikes and/or wildlife abundance were evenly distributed along the CPR, chi-squared tests were used. To identify hotspots (Malo et al. 2004), the upper 95% confidence interval of strikes per segment was used, which assumed the counts followed a Poisson distribution (Bivand 2008; Malo et al. 2004).

Risk estimates for each rail segment were developed through three steps. First, the $RA_{\text{rail corridor}}$ or $RA_{\text{rail bed}}$ estimates for each segment (i) were converted to a percent of the total from all rail segments. Next, the expected number of strikes for each rail segment ($\text{Expected}_{(i)}$) was calculated by multiplying the percent of total scat on that segment by the overall mortality rate for that species (Eq. 2). Risk was estimated by assessing the ratio of the number of strikes to the expected number of strikes for each segment (Eq. 3). Both values were assumed to equal at least 1.0 for all rail segments because all species are present in the study area. In cases where no strikes or no pellets were detected the risk estimate was set to 1.0 without confidence intervals. Otherwise the risk estimate would have equaled zero or infinity in these cases.

$$\text{Expected}_i = \frac{\sum \text{STRIKES}_i}{\sum RA_{\text{rail corridor}_i}} * RA_{\text{rail corridor}_i} \quad (\text{Eq. 2})$$

The risk estimate, evaluates relative risk between segments. It has been widely used in spatial epidemiology studies and is referred to as a standardized mortality ratio where the ratio is expected to equal 1.0 because internal standardization was used (Mantel 1959; Banerjee et al. 2004; Bivand 2008). In these data, the average risk (risk = 1.0), meant the number of strikes was proportional to the relative abundance for that species. When the estimated ratio was above 1.0 it indicated relatively high risk (more strikes occurred than expected). Internal standardization reduced the number of degrees of freedom to $n-1$ which is important when conducting chi-squared tests (Kim and Wakefield 2004).

$$\text{Risk}_i = \frac{\text{Strikes}_i}{\text{Expected}_i} \quad (\text{Eq. 3})$$

To test for non-constant risk, the number of strikes observed was compared to the number expected using X^2 tests (Bivand 2008) and the relationship between observed and expected was evaluated with generalized linear models. To identify which sections had unusually high risk, a bootstrap function was used to resample the wildlife abundance estimates from scat counts for each rail segment. A new relative risk estimate was computed for each of 1000 resampled abundance estimates. Then confidence intervals were assessed from the distribution of the 1000 estimates. High risk segments were defined as those where 95% of bootstrapped estimates of risk were above 1.0. All analyses were performed in R (R Development Core Team 2009). Analyses were originally conducted at a 4.86 km scale then repeated at coarser scales 9.72, 19.44, 34.0 km (6, 12, and 21 miles) to determine the stability of the results.

A set of variables hypothesized to affect train strikes were tested using generalized linear mixed models with a log link function (Table 2). Models were evaluated for each species by fitting the count of strikes per segment (summed over the 21-yr period) to ten predictors including the appropriate wildlife abundance estimate (which included two for bears due to evaluation at the rail bed and landscape scales). Initial fits used restricted maximum likelihood estimation and likelihood ratio tests to assess whether a negative binomial distribution was needed (Zuur et al. 2009). Each model was subjected to a drop one approach where the least significant parameter was dropped until all remaining parameters were significant. If models were reduced to a single predictor, the model with the minimal residual deviance was selected as the final

model. Final models were refit with maximum likelihood and inspected using standard diagnostic plots. Semivariograms and residuals from the final model were plotted to assess remaining spatial trend. Prior to performing the analysis covariates were tested for collinearity (Menard 1995). When correlated variables ($r > 0.6$) were found, the one with higher correlation to the response remained in the analysis (Guisan and Zimmerman 2000). Based on this cut-off, track grade was significantly correlated with train speed and was removed from the analysis.

Results

Strike Rates

Over the 21-yr period (1989-2009), 862 strikes were recorded along 134 km of the CPR, consisting of 579 elk, 185 deer, 69 black bear, 9 grizzly bears and 1 unidentified bear sp. The bear data were combined to one class. The spatial distribution of strikes per rail segment were non-uniform for elk $X^2(1, 27) = 834.4, p < 0.001$, deer $X^2(1, 27) = 252.5, p < 0.001$, and bears $X^2(1, 27) = 37.1, p = 0.03$ (Figure 11).

During the 21-yr period strike rates varied annually for elk, deer, and bears (Figure 12). A decreasing trend was detected for elk strikes ($\beta_1 = -1.04, t(19) = -2.29, p = 0.03$). However, an increasing trend was found for deer ($\beta_1 = 1.13, t(19) = 8.18, p < 0.001$) and bears ($\beta_1 = 1.12, t(19) = 4.58, p < 0.001$). These changes were likely due to

Table 2. Description of field collected and GIS derived spatially varying train and railroad design variables and the hypothesized correlation to strike rates. (+) indicates a positive correlation and (-) indicates a negative correlation hypothesized for both ungulates and bears.

VARIABLE	Description	Hypothesis	Source
Continuous variables			
RA _{rail corridor}	Mean number of weighted pellets or scats standardized by distance searched along perpendicular strip transects	+	Field
RA _{rail bed}	Total of weighted bear scats detected on the CPR track bed	+	Field
GRAIN _{mean}	Mean number of train spilled grains	+	Field
GRADE _{mean}	Mean percent grade of the railroad bed	+	GIS
ROW _{mean}	Mean right-of-way width	-	GIS
SINUOSITY	Ratio of rail segment length to the distance between the start and end locations	+	GIS
SPEED _{max}	Maximum speed limit in miles per hour	+	GIS
Categorical variables			
BARRIER _c	Count of overpass, snow shed, rock cut, or tunnel	+	Field
BRIDGE _c	Count of bridges	+	Field
SIDING	Presence/absence	+	Field
VERGESLOPE*	Railroad verge slope classes*	+	Field

*(0) flat (1) at least one down slope (2) at least one up slope or water

population fluctuations, but could have also been due to changes in wildlife distributions or movement patterns. Although annual mortality rates fluctuated for all three species, the spatial distribution of strikes accounted for more of the variability in strikes for elk and bears but not deer (Table 3b).

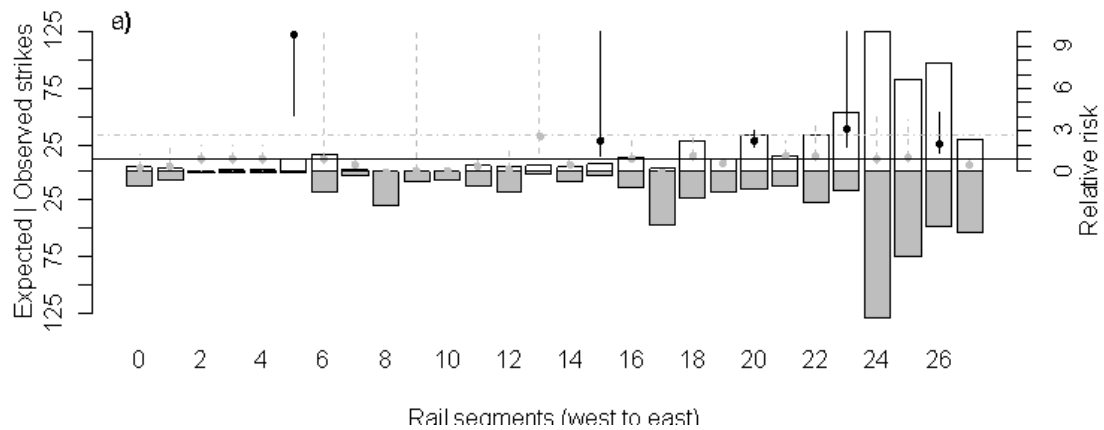
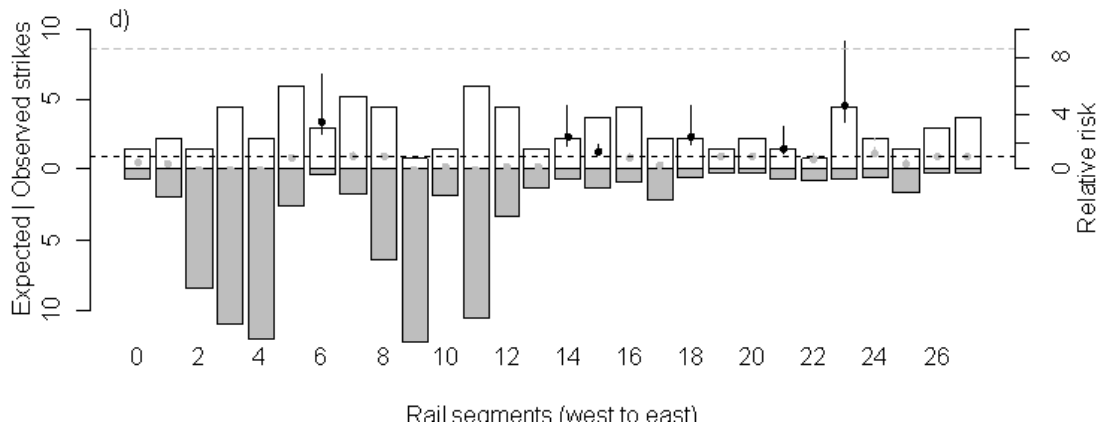
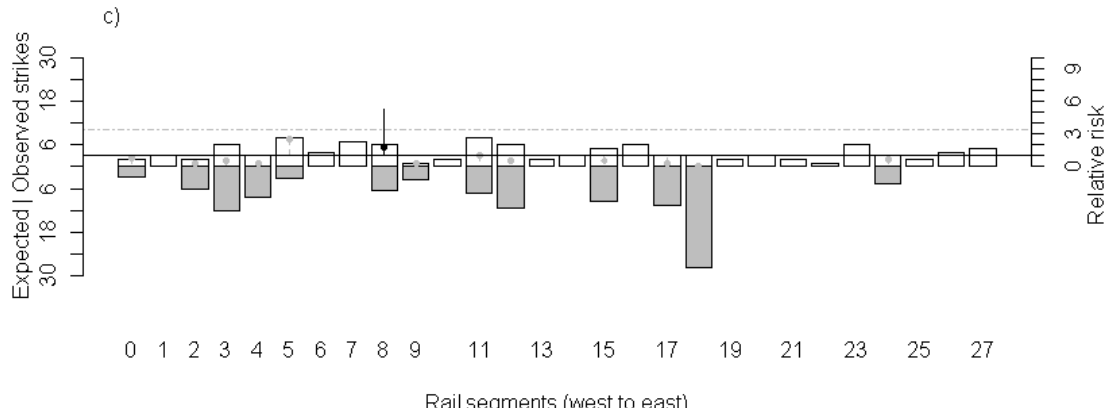
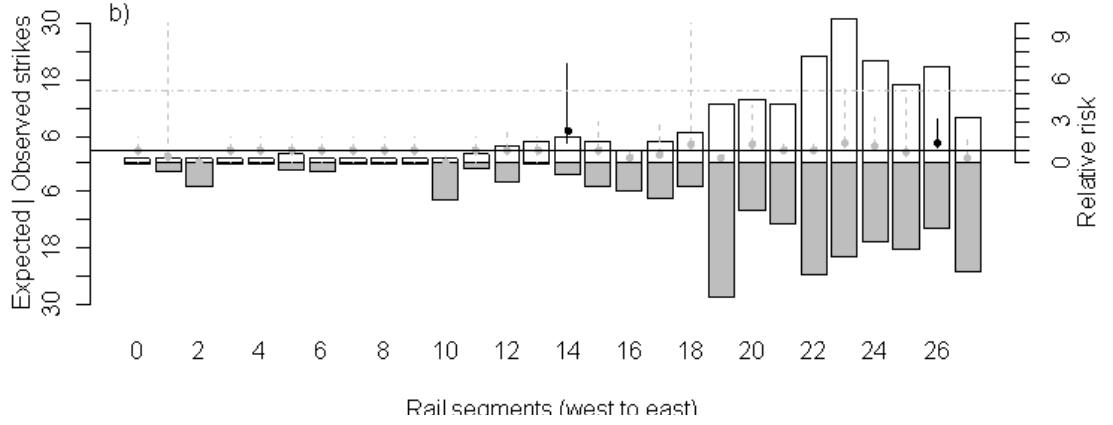


Figure 11. The number of train strikes (white bars) for a) elk, b) deer, c) and d) bears; compared to the expected number of strikes (grey bars, a positive value) based on the abundance of wildlife sign along each 4.86 km segment along perpendicular transects. d) compares on-track relative abundance for bears to number of strikes. Hotspots are segments with a strike count above the 95% confidence interval (grey dashed line) and high risk segments are those with a risk estimate significantly above 1.0 (black points and error bars). Grey points are risk estimates not significantly different from 1.0 (the number killed was close to the number expected based on wildlife abundance). Segments proceed west to east, where 0 is the first 4.86 km inside the west boundary of Yoho National Park. The Field town site is located at segment 6, the Lake Louise town site corresponds to segment 13 and the Banff town site segment 24.

Figure 11. Continued.



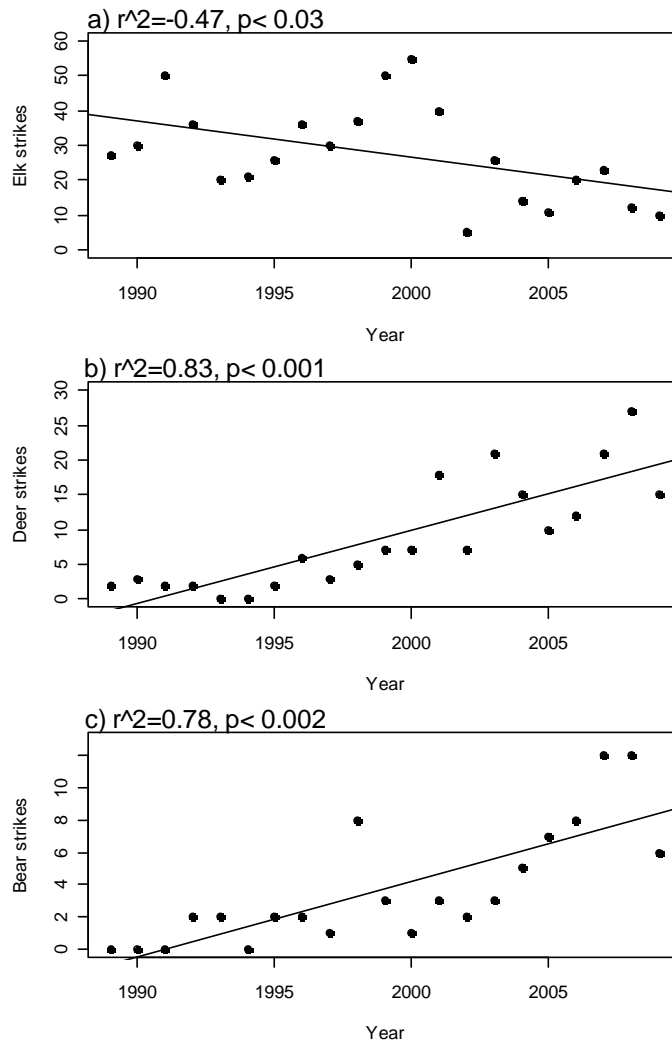


Figure 12. Annual train strikes for a) elk b) deer and c) bears along 134 kilometers of the Canadian Pacific Railroad through Banff and Yoho National Parks, 1989 to 2009.

Relative Abundance

A total of 313 km of rail bed transects were surveyed and 129 perpendicular transects were surveyed in 2008 and 2009. A total of 341 bear scats were detected on the CPR resulting in a detection rate of $0.46 \text{ km}^{-1} \text{ yr}^{-1}$. On perpendicular transects, 599 elk pellet groups, 212 deer pellet groups, and 39 bear scats were detected.

Table 3. Variation between years and railroad segments for a) elk, b) deer and c) bears along the Canadian Pacific Railroad within Banff and Yoho National Parks, 1989 to 2009.

a)					
Factor	Df	elk Sum Sq.	Mean Sq.	F	<i>p</i>
Year	20	311.31	15.57	2.4	<0.001
Railroad Segment	26	760.81	29.26	4.6	<0.001
Residuals	234	1489.85	6.37		

b)					
Factor	Df	deer Sum Sq.	Mean Sq.	F	<i>p</i>
Year	20	74.19	3.71	3.7	<0.001
Railroad Segment	26	67.65	2.60	2.6	<0.001
Residuals	234	233.37	1.00		

c)					
Factor	Df	bears Sum Sq.	Mean Sq.	F	<i>p</i>
Year	20	17.19	0.86	3.2	<0.001
Railroad Segment	26	22.34	0.86	3.2	<0.001
Residuals	234	63.27	0.27		

These data resulted in average detection rates of $5.98 \text{ km}^{-1}\text{yr}^{-1}$ for elk, $2.23 \text{ km}^{-1}\text{yr}^{-1}$ for deer and $0.33 \text{ km}^{-1}\text{yr}^{-1}$ for bears. Relative abundance varied for elk ($X^2 (27) = 940.12, p < 0.001$) and deer ($X^2 (27) = 294.2, p < 0.001$).

Bear scats were unevenly distributed both on the rail bed ($X^2 (27) = 263.3, p < 0.001$) and on perpendicular transects ($X^2 (27) = 125.4, p < 0.001$) (Figure 11c and 11d). The rail bed relative abundance transects likely better represented relative risk for bears because few bear scats were detected along the perpendicular transects ($n=39$) relative to

the 341 scats detected directly on the rail bed. Therefore, further analyses for bears were based on rail bed scats.

Hotspots and Relative Risk

Hotspots were detected for elk and deer, but not bears. Six elk hotspots were identified (21% of the CPR segments) which were segments with 29 or more strikes (Figure 11). Likewise, five deer hotspots (17 % of the CPR segments) were identified, which averaged 6.54 ± 1.73 per segment. Bears were struck on average 2.86 ± 0.44 per segment, but no segment was identified as a hotspot because no single segment incurred more than seven strikes.

After estimating the ratio of strikes to wildlife abundance for each segment, overall non-constant risk was apparent for elk ($X^2 = 286.078$, $df = 27$; $p < 0.0001$), deer ($X^2 = 182.0247$, $df = 27$; $p < 0.0001$), and bears ($X^2 = 7,424,041$, $df = 27$; $p < 0.0001$). High risk segments were identified for each species (Figure 11). Five, two and six high risk segments were determined for elk, deer and bears, respectively.

Hotspots and high risk segments differed in number and location (Figure 11). Hotspots were segments that had strike rates significantly higher than the overall mean. High risk segments had significantly higher strike rates than was expected based on estimates of relative wildlife abundance. Although in some cases, segments were identified as both hotspots and high risk, fewer segments were identified for elk and deer as high risk. For bear, no hotspots were identified but six high risk segments were detected (Figure 11d and 13).

Variables Associated with Strikes

The variability in train strikes were evaluated for each species by fitting strikes per segment to relative abundance and nine train or railroad design variables (Table 2). Initial fits and likelihood ratio tests indicated a negative binomial distribution for all species models. Initial models did not indicate autocorrelation was present between rail segments. The most parsimonious model for each species were reported in Table 4.

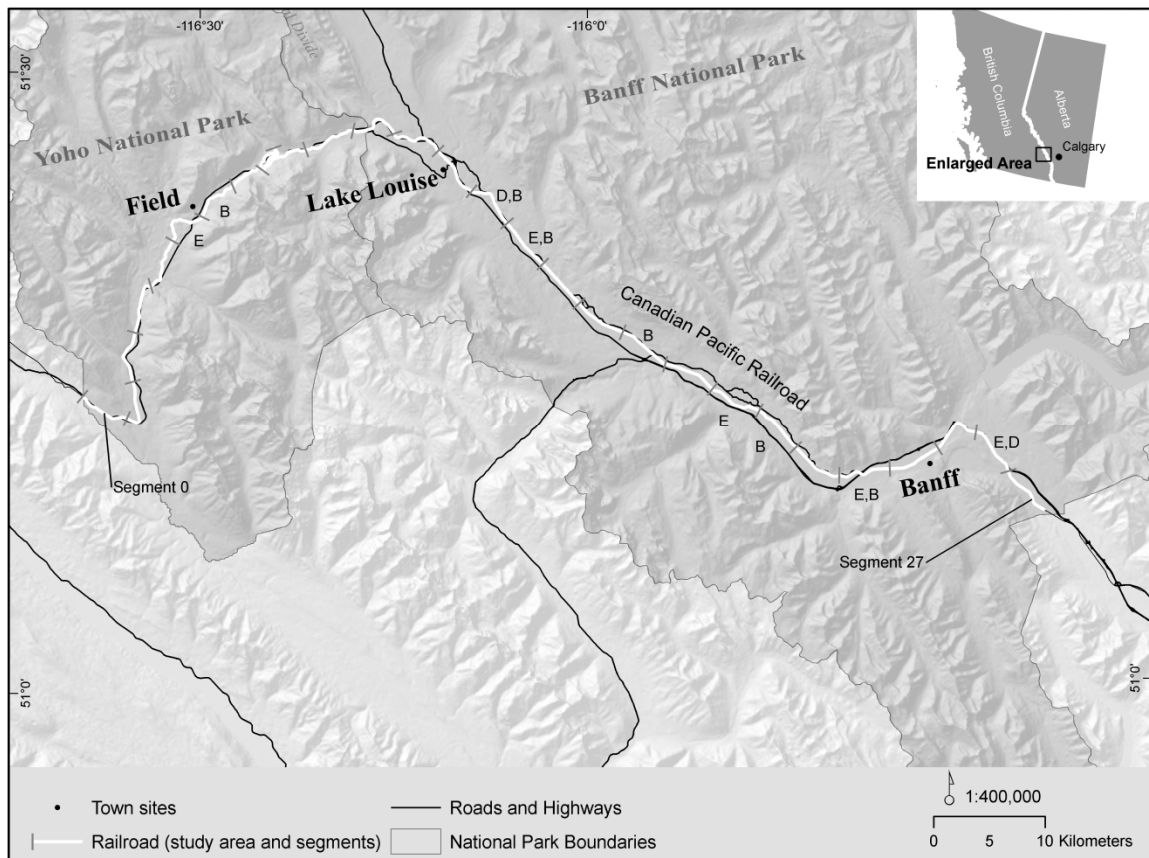


Figure 13. Segments with high strike risk labeled for (E) elk, (D) deer, and (B) bears during the 21-year period (1989 to 2009) along the Canadian Pacific Railroad through Banff and Yoho National Parks.

Elk. Train speed and relative abundance explained the variability in elk strikes (Log likelihood= -113.0 with 2 df, $X^2 = 26.3$, $p < 0.001$). Models used a log link function thus parameter estimates could be back transformed to estimate changes in strikes for each variable given that the other variables in the model were held constant. For example, elk strikes increased on average $e^{0.07} = 1.07$ per segment with each 1.0 mph increase in maximum posted train speed when elk relative abundance was held constant (Table 4). Likewise when the $SPEED_{max}$ was held constant strikes increased $e^{0.03} = 1.03$ for each one unit increase in elk $RA_{rail\ corridor}$. The variable $SPEED_{max}$ averaged 37.5 ± 1.85 and ranged from 20 to 50 mph.

Deer. The variables deer relative abundance ($RA_{rail\ corridor}$), $SPEED_{max}$ and ROW_{mean} best explained the variability in deer strikes using the model selection process (Log likelihood= -78.46 with 5 df, $X^2 = 48.89$, $p < 0.001$). The parameter coefficients from a maximum likelihood fit indicated that $e^{0.06} = 1.06$ additional deer strikes were observed on average with an 1.0 mph increase in the posted train speed limit when deer abundance and ROW width were held constant. Likewise $e^{0.009} = 1.009$ additional deer strikes were observed for each 1.0 m increase in ROW width when speed and deer abundance were held constant. The variable ROW_{mean} was on average 79.78 ± 7.68 m across all segments.

Bears. A single predictor model including the variable $BARRIER_c$ best explained bear strikes (Log likelihood= -57.233 on 4 df, $X^2 = 14.686$, $p < 0.001$). The parameter estimated for $BARRIER_c$ ($\beta = e^{0.458} = 1.58$) revealed that for each additional barrier feature per segment bear strikes increased on average 1.58 (95% CI 1.13 to 2.20)

(Table 4). A second single predictor model has almost equal explanatory power. This model included the variable $BRIDGE_c$. However, $BRIDGE_c$ and $BARRIER_c$ were not correlated ($r = 0.23$, $t = 1.1968$, $df = 26$, $p = 0.24$). The variable $BRIDGE_c$ was positively correlated to bear strikes ($r = 0.44$, $p < 0.02$).

For bears, neither estimate of relative abundance were significant. Thus the final model did not include either variable describing bears relative abundance. This changes the modeling results from assessing risk to incidence rate. For this reason, the model selection procedure was repeated with the variable $RA_{rail\ bed}$ log transformed and held in the model as an offset to assess relative risk (Zuur et al. 2010). This model selection procedure resulted in a single predictor model that included $SPEED_{max}$. The parameter estimated for $SPEED_{max}$ indicated that as posted train speed increased so did “risk” of bear strikes ($\beta = e^{0.053} = 1.054$). However, train speed explained only 17% of the deviance compare to 82% explained by the $BARRIER_c$ model (Table 4). Therefore this model may not be as useful except to point out the potential differences when modeling risk versus incidence rate.

Table 4. Significant ($p < 0.05$) parameter estimates explaining train strike rates with elk, deer and bear. Model fit statistics are provided from maximum likelihood fits.

Explanatory variables	Elk			Deer			Bears		
	β	SE	p	β	SE	p	β	SE	p
Y-intercept	-			-					
$RA_{\text{rail corridor}}$	1.063	0.782	<0.001	1.917	0.751	<0.001	0.825	0.153	<0.001
$RA_{\text{rail bed}}$	0.024	0.007	<0.001	0.069	0.014	<0.001			
$GRAIN_{\text{mean}}$									
$GRADE_{\text{mean}}$									
ROW_{mean}				0.009	0.003	0.005			
SINUOSITY									
$SPEED_{\text{max}}$	0.080	0.020	<0.001	0.055	0.017				
Categorical variables									
BARRIERc							0.458	0.170	0.007
BRIDGEc									
VERGETOPO*									
Dispersion (θ)	1.5			3.8			20.0		
Deviance Explained (%)	41.7			28.3			82.042		

Discussion

This research was the first attempt to account for unevenly distributed wildlife populations when assessing strikes along a railroad. This information was used to help understand the spatial pattern of train strikes along the CPR. Based on strike data alone, hotspots were identified for elk and deer, but not bears. Hotspots were generally associated with higher relative abundance for elk and deer at the analysis scale. Although there was disagreement between relative abundance and strike rates for bears at a scale of approximately 5 km (Figure 11d), general correlation was apparent at coarser scales of 10 and 20 km (data not shown). Overall, bear strikes and relative abundance were relatively higher in the western half of the study area (Figure 11d segments 0 to 13) compared to the eastern half (Figure 11d segments 14 to 27).

Risk estimates provided a way to evaluate the importance of other variables and indicated potential problem areas that may have been overlooked using strike data alone. For example, six high risk segments were detected but no hotspots were identified for bears. Risk estimates revealed additional information for assessing problem areas or attempting to develop mitigation solutions because relative abundance for these species varied along the CPR and likely varies across other transportation features (Figure 12).

There was a significant relationship for elk, deer and bears with at least one train or railroad design variable. These relationships indicated higher train speeds (elk and deer) and larger ROW widths (deer) were both associated with increased strike rates for

these ungulates. For bears, the number of barriers and presence of bridges was also positively correlated with strike rates.

Results from multivariate analyses supported the hypothesis that there are at least three general variables that affect the spatial pattern of train strikes: 1) the relative abundance of wildlife either on the rail bed or on the adjacent landscape 2) train speed 3) railroad designs such as highway overpasses, rock-cuts, tunnels, snow sheds and bridges. Other railroad studies have observed similar associations. In Slovenia, Eurasian brown bears (*U. arctos*) were struck at rock cuts and on bridges (Kaczensky et al. 2003). At least one road study has noted the effect of bridges on strikes with bears (Van Why 2003). Similar variables to this study were measured by Huber et al. (1998) in Croatia at sites where brown bears were struck by trains. They found no difference in verge slope, longitudinal or perpendicular visibility but did detect a difference in the presence of bear foods at strike locations compared to random locations and noted a slight difference in the distance at which trains were first audible at strike locations (Huber et al. 1998). Field observations suggest that train volume varied greatly depending on train direction within the study area. These data were not collected systematically in this study but should be examined in future research.

There are at least four possible explanations for the lack of correlation between bear strikes and relative abundance found in this study. The first is based on the strength of nonconstant risk phenomena and three others deal with limitations of the methods and analysis. First, railroad design or other variables may have strongly affected strike probabilities. If these variables had strong causal or probabilistic effects, the spatial

pattern of strikes would be a function of the spatial pattern of these variables and not bears relative abundance. However, it is unclear if this is the case because the sample size is relatively low ($n=69$) and no validation of these results has been attempted. Therefore other explanations for the lack of agreement between strikes and bears relative abundance could be due to the field and/or analysis methods used. The field data collected may have failed to adequately represent the true distribution of bears along the CPR. Bears may have been moving or utilizing the railroad without leaving scat, which would have been undetectable using these methods. Additionally, scat sampling methods may have only represented individuals who regularly encountered the CPR and have learned to coexist, to some degree, with the railroad. Bears that utilized grain as a food item in this study area may have developed behaviors that allowed them to forage on grain without being struck by trains. Although train-spilled grain was shown to be correlated with bear scats, grain filled scats (90% of all detected) were also shown to be unrelated to strike rates in Chapter 2 of this thesis. In this analysis scats were unrelated to strike rates regardless of grain contents. Lastly, it is possible that there is a more optimal analysis scale. The analysis scale was chosen due to the accuracy of the mortality data and management needs. Bears have large home ranges thus a larger analysis scale may have been more appropriate. However, from the perspective of management solutions, a finer scale may have been more appropriate to generate solutions targeted at specific locations and features. The analysis scale used was a necessary compromise based on the data available. Future research should use more precise data or alternative methods that allow multi-scale analyses.

The data collected may have failed to fully describe relative abundance of deer along the CPR. This may have been why ROW width was positively correlated to strikes for deer. The hypothesized influence of ROW width was that narrower ROW's (a negative relationship) would increase strike risk to wildlife. This variable may have accounted for spatial or temporal variability in deer abundance that the transect data failed to capture.

An animal's behavior when a strike occurs is only partially influenced by location specific variables. Other variables likely affect an animal's behavior that are difficult to measure over long time frames and large geographic areas. These variables include: fine scale temporal variables (individual wildlife behavior, landscape and train) and behavioral heterogeneity between individual animals. For this study strikes spanning 21-yr was used to provide adequate sample sizes for comparison to railroad variables that have not changed during that time period. However, it is clear from Figure 12 that strikes have changed over this period and thus relative abundance may have also changed during this time period. To address this concern analyses were repeated using a 10-yr period. Results showed some shifts in the locations of hotspots and high risk segments. However, these did not alter the general conclusions. The longer term 21-yr analysis was likely better because of the larger sample size provided which provided more information for hotspots, multivariate modeling and stability to risk estimates.

Management Implications

This study found wildlife abundance, train speed, right-of-way widths, barriers and bridges likely affect strike rates for ungulates and/or bears. Based on these results a variety of approaches could be used to reduce strike rates. These include those that affect wildlife abundance near railroads, reduced train speeds, or modified railway designs. Much of the CPR is co-aligned with productive montane habitat, thus altering the abundance of deer or other species near the CPR may not be feasible. A two step approach may be needed that both reduces forage quality and availability along the CPR and increases habitat quality away from the CPR. One of the few studies to show a decrease in train strikes, developed ungulate habitat away from a railroad (Jaren et al. 1991; Modafferi 1991).

Train speed was also clearly important. Reduced train speeds may be an effective measure to reduce strike risk, particularly in problem areas. Train speed is a spatially and temporally explicit management solution and likely interacts with the other mechanisms described above, thus could be implemented where or when strikes are most likely to occur. However, reduced speeds may be ineffective if other mechanisms are the primary driver in strike occurrences. For example, a study in Alaska observed a possible mechanism (constrained flight paths) resulting in strikes with moose (Becker and Grauvogel 1991). The study then evaluated reduced train speeds (reaction time mechanism) on reducing moose strike rates. The authors found reduced train speeds to be ineffective in this case (Becker and Grauvogel 1991). If moose were being struck as the authors hypothesized, because they did not leave the railroad. A logical experiment

should alter variables enabling moose to move off the railroad such as snow removal, vegetative or other terrain modifications. The authors suggested that at some level of speed reduction the approach may have been effective but these speeds are likely economically cost prohibitive (Becker and Grauvogel 1991). Although moose were not analyzed in this study, speed was positively associated to increasing strike risk for elk and deer along the CPR. For this reason, reduced train speeds are recommended in problem areas.

For bears, it may be most important to evaluate design modifications or mitigation solutions targeted at barriers including: highway vehicle overpasses, tunnels, snow sheds, rock cuts and bridges. However, multiple approaches are likely warranted in high risk segments including those that: decrease the probability bears will be exposed to strikes, increase the detectability of trains, increase the opportunities for safe flight paths off-track and increase the time bears have to successfully avoid trains.

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