EFFECTIVENESS OF ROAD MORTALITY MITIGATION IN A NORTHERN COMMUNITY OF SNAKES

by

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ABSTRACT

Roads and vehicular traffic are one of the most wide-spread threats to wildlife globally. They are particularly detrimental to long-lived species with lower population growth rates that are less able to naturally compensate for increases in anthropogenic mortality. Road crossing structures and fencing often are implemented to mitigate the impacts of road mortality on wildlife. However, follow-up studies assessing their efficacy are limited, especially for reptiles who are particularly vulnerable to road effects. Although numerical (population) responses to mitigation structures are strong indicators of the overall effectiveness of the mitigation structures, they require intensive and/or long-term data. However, functional (behavioural) responses also may provide good indicators of the response to mitigative efforts; thus, functional and numerical responses used in tandem may provide a more robust assessment of the impact that the mitigation structures, including those designed to reduce road impacts, are having on populations.

I conducted a detailed assessment on the short-term, immediate impact of recently-installed ecopassages and directional fencing on a threatened Western Rattlesnake (*Crotalus oreganus*) population south-central British Columbia, Canada. Using road surveys, traffic monitoring, and mark-recapture methods, I analyzed trends in roadkill rates and population size during a 6-year period (2015-2020) that encompassed the periods before, during, and immediately after (two years) the installment of ecopassages under a highway. Roadkill rates appeared to decrease after mitigation installation (0.06 ± 0.03 SE deaths/km/day before, 0.03 ± 0.01 SE deaths/km/day after) despite an increase in traffic (302 vehicles/day before, 454 vehicles/day after), yet population trends for adults did not indicate a clear trajectory towards recovery.

Wildlife cameras were installed in the ecopassages, and I used photograph data to quantify spatial and temporal usage patterns and assess the immediate post-installation effectiveness of drift fencing for the Western Rattlesnake (*Crotalus oreganus*) and two other threatened species in the same community: the Great Basin Gophersnake (*Pituophis catenifer deserticola*), and Western Yellow-bellied Racer (*Coluber constrictor mormon*). I quantified "appearances" (any time an individual appeared on camera), and "passages" (where a snake was documented travelling through an ecopassage within a 30 minute time span). Across two years (2019 and 2020) appearances and passages were, respectively, 115 and 12 for rattlesnakes, 183 and 29 for gophersnakes, and 3061 and 748 for racers. Although racers clearly used the ecopassages significantly more than the other two species, both appearances and passages increased for rattlesnakes, gophersnakes, and racers from 2019 to 2020. The proportion of entrances with fences showed increased usage from 2019 to 2020, however these changes only were significant for rattlesnakes. Temporal and spatial use patterns differed among species, and I postulate that this was due to differences in movement patterns and habitat preferences.

This study highlights the short-term yet complex response of snake communities to the effects of roadway mitigation, as the animals presumably encounter, adjust, and respond to the new structures in their environment. Despite the three snake species inhabiting the same environment, different responses were demonstrated towards the new mitigation structures. Short-term assessments of the response to mitigation efforts, such as applied in this study, likely illustrate a 'shock phase' in the wildlife community, and should be coupled with longer term monitoring to gauge the full effect of the conservation actions.

Keywords: road ecology, road mortality, mitigation, ecopassages, drift fencing, population estimate, camera monitoring, conservation, reptile, snake

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TABLE OF CONTENTS

LIST OF FIGURES

List of Tables

CHAPTER 1

INTRODUCTION

Wildlife Mortality Factors

Although the mortality of individuals within animal populations may be attributable to a single cause, often a number of natural and anthropogenic factors are working in tandem with one another. Broadly put, these interactions may be additive or compensatory in nature. Compensatory mortality is when mortality from one cause is reduced but the spared individuals still end up dying from other causes (Mills 2013, Péron 2013), whereas additive mortality is described as mortality compounded on to natural mortality rates, meaning individuals would have otherwise survived if the proximate cause had been removed (Mills 2013, Péron 2013). Whether road mortality acts as a compensatory or additive force depends on the population, but the latter is more likely for species with high adult survival rates and delayed sexual maturity (Gibbs and Shriver 2002, Moore et al. 2023, Winton 2018).

Road Mortality and Mitigation

The presence of roads and vehicular traffic are one of the most wide-spread, global threats to wildlife (Forman et al. 2003, Boyle et al. 2021). They are considered a main driver of habitat degradation and destruction, posing a tremendous risk to wildlife both directly and indirectly (Dillon et al. 2020, Trombulak and Frissell 2000**).** Direct mortality from vehicle collisions can result in reductions in population persistence and viability (Boyle et al. 2021, Fahrig and Rytwinski 2009).

In response to growing concern for wildlife conservation, mitigation measures are being increasingly implemented to prevent road mortality (Glista e al. 2009, Jarvis et al. 2019, van der Grift et al. 2013). Some mitigation measures intended to reduce road-related wildlife mortality are designed to influence motorist behaviour (e.g. warning signs, reduced speed limits), while others seek to influence animal behaviour (e.g. fencing, crossing structures), with the latter thought to be more effective (Huijser et al. 2007, Rytwinski et al.

2016). More broadly, wildlife crossing structures are intended not only to prevent direct mortality by vehicles, but aid in maintaining population connectivity (Dillon et al. 2020). They can take the form of under- or over-passes, and vary significantly in design depending on the intended species, with some examples being amphibian tunnels, badger pipes, wildlife bridges, rope bridges, and glider poles (Glista et al. 2009, Rytwinski et al. 2016, van der Grift et al. 2013). 'Ecopassage' is a term used for under-road tunnels that often are used to facilitate road-crossing movements in smaller animals, and they are becoming an increasingly common mitigation tool for the conservation of reptile species (Baxter-Gilbert et al. 2015, Boyle et al. 2021, Dillon et al. 2020).

Despite ecopassages now being used more commonly for reptile conservation, in-depth analyses of their effectiveness for this taxon remain rare (Baxter-Gilbert et al. 2015, Boyle et al. 2021, Dillon et al. 2020). The research to date most often assesses only the amount of roadkill or the amount the ecopassage is used, but these metrics do not necessarily equate to effectiveness (van der Grift et al. 2013). Changes in individual behaviour can affect survival and reproduction, which in turn ultimately affects populations (French et al. 2018). For a more robust and accurate assessment of the effectiveness of these mitigation structures for reptiles, roadkill, ecopassage use, and population-level impacts should be considered (Boyle et al. 2021, Fahrig and Rytwinski 2009).

Snakes and Roads

Reptiles are among the fastest-declining taxa globally, facing numerous wide-spread threats including habitat loss, degradation, and fragmentation (Böhm et al. 2013, Cox et al. 2022, Saha et al. 2018). Roads are an example of these threats, and reptiles are particularly vulnerable to their effects due to their small size and slow rates of movement and reproduction (Boyle et al. 2021, Brehme et al. 2018, Gigeroff and Blouin-Demers 2023).

Within the reptiles, snakes are particularly at-risk of direct road mortality for a number of reasons. Seasonal migrations to and from hibernacula put snakes at risk of encountering roads, especially if hibernacula lie in close proximity to a road (Gunson and Schueler 2019). Snakes also are drawn to roads to thermoregulate (Baxter-Gilbert et al. 2015, Mccardle and Fontenot 2016) and morphological factors, like an elongated body shape, creates roadkill risk for snakes, such as when motorists may fail to detect small or juvenile animals or mistake them for sticks or twigs. (Gunson and Schueler 2019). Unfortunately, snakes also are intentionally run over by motorists (Ashley et al. 2007). Species with life histories characterized by low reproductive rates and low adult mortality, like rattlesnakes, are particularly vulnerable to demographic consequences of road mortality (Brehme et al. 2018, Forman et al. 2003).

These effects are amplified in northern snake species. In British Columbia for example, rattlesnakes face shorter cooler active seasons, that limit their annual growth rate and reproductive frequency even further (Macartney and Gregory 1988). They also are restricted to the southern latitudes of the political borders of Canada while being at the northern limits of their geographic range more than any other taxa (Currie and Marconi 2020). Species that are not limited by density dependence, and instead limited by intrinsic factors like growth rate and reproductive frequency, will not experience a strong compensatory response to anthropogenic mortality since the presence of other individuals is not restricting their ability to survive (Péron 2013). Therefore, road mortality generally is an additive factor that can be detrimental to northern snake populations (Winton et al. 2020).

Study Site

The Okanagan is home to the fastest-growing metropolitan area in Canada (Statistics Canada 2022), and is a tourism hotspot (Destination British Columbia 2017). As a result, there has been immense expansion of infrastructure and therefore, loss or degradation of wildlife habitat. This is concerning since the South Okanagan is where the northern edge of the Great Basin Desert reaches into Canada, making it an extremely rare environment that is home to many species occurring nowhere else in the country (B.C. Ministry of Environment 1998, Parks Canada 2022).

Within the South Okanagan river valley is the White Lake Basin, which includes part of the White Lake Grasslands Protected Area. This landscape is home to many species at risk (BC Parks 2023, Figure 1.1), including three federally Threatened snake species: Western Rattlesnakes (*Crotalus oreganus*), Great Basin Gophersnakes (*Pituophis catenifer* *deserticola*), and Western Yellow-bellied Racers (*Coluber constrictor mormon*). Historical work by Winton et al. (2018, 2020) focused on Western Rattlesnake road mortality and its impacts on the population in the area. This area and the species make an ideal combination for such a study, since there are very few anthropogenic features on the landscape apart from roads, and the communal denning behaviours of rattlesnakes make an extensive mark-recapture study achievable (Macartney et al. 1990). More specifically, the Winton research accurately quantified rattlesnake road mortality by testing and correcting for associated sources of error, characterized the rattlesnake population affected by road mortality, and assessed the long-term persistence of the population under the threat of roadkill. They found ~6.6% of the rattlesnake population dying each year from this road mortality, and using a Population Viability Analysis they determined that even a slight increase in road mortality could result in the species becoming extirpated from the area within 100 years (Winton et al. 2020). They recommended establishing mitigation measures in the area as an effort to conserve the species, which resulted in the installation of ecopassages and drift fencing.

Research Objectives

In this thesis I assessed the short-term impact of ecopassages on a Western Rattlesnake (*Crotalus oreganus*) population in British Columbia, Canada. I conducted intensive surveys to monitor roadkill and assess changes in population size. By repeating the same methodology used by Winton et al. (2018, 2020) to estimate roadkill occurrences, population size, and annual survivorship, I was able to compare these metrics for the two years immediately following the establishment of the ecopassages and associated drift fencing to the three years before in which Winton et al. conducted the same measurements.

I also quantified and compared the use of eight ecopassages in the area by Western Rattlesnakes (*Crotalus oreganus*), Great Basin Gophersnakes (*Pituophis catenifer deserticola*), and Western Yellow-bellied Racers (*Coluber constrictor mormon*). I positioned wildlife cameras at either end of each ecopassage to monitor the occurrence of each species throughout their active season, enabling me to compare use both spatially and temporally.

The overarching research objectives of my thesis were to:

1. Compare Western Rattlesnake pre-mitigation roadkill and population estimates to those in the years immediately following the installation of ecopassages and fencing; and
2. Quantify ecopassage use of three at-risk snake species, considering spatial and temporal differences between species.

In the remaining portion of this chapter, I provide an overview of my study site and more information about the ecopassages and drift fencing that were installed there. I collected data for this research in 2019 and 2020, but data from all years (since 2015) of this long-term project were incorporated into this thesis. In Chapter 2, I replicate the work done by Winton et al. (2018, 2020) to assess the immediate impact of the newly installed mitigation structures on the road mortality rates and population size of rattlesnakes in the area. In Chapter 3, I use photos from wildlife cameras to quantify and compare the ecopassage use of Western Rattlesnakes, Great Basin Gophersnakes, and Western Yellow-bellied Racers. I also conduct a secondary assessment of fence effectiveness. Finally, in Chapter 4, I summarize the findings of my research and discuss the resulting management implications. I conclude with highlighting the importance of both short- and long-term data sets in fully understanding the response of these animals to mitigation efforts.

Site Description

My study took place in the White Lake Basin of the South Okanagan region of British Columbia, Canada (latitude 49.318°N, longitude 119.638°W). The basin consists of open shrub-steppe grassland habitat, and is managed with the objective to "integrate livestock management with conservation of habitat for species at risk" by The Nature Trust of British Columbia (TNTBC 2022). The paved, undivided two-lane road (BC Class 5 highway) traversing the bottom of the basin bisects the grassland and has an unposted speed limit of 80 km/h, in part due to the presence of a Canadian federal astrophysical observatory. Two roads – one running largely East-West (White Lake Road) and one running North-South (Willowbrook Road) – meet at a T-intersection (Figure 1.1). There is barbed-wire fence

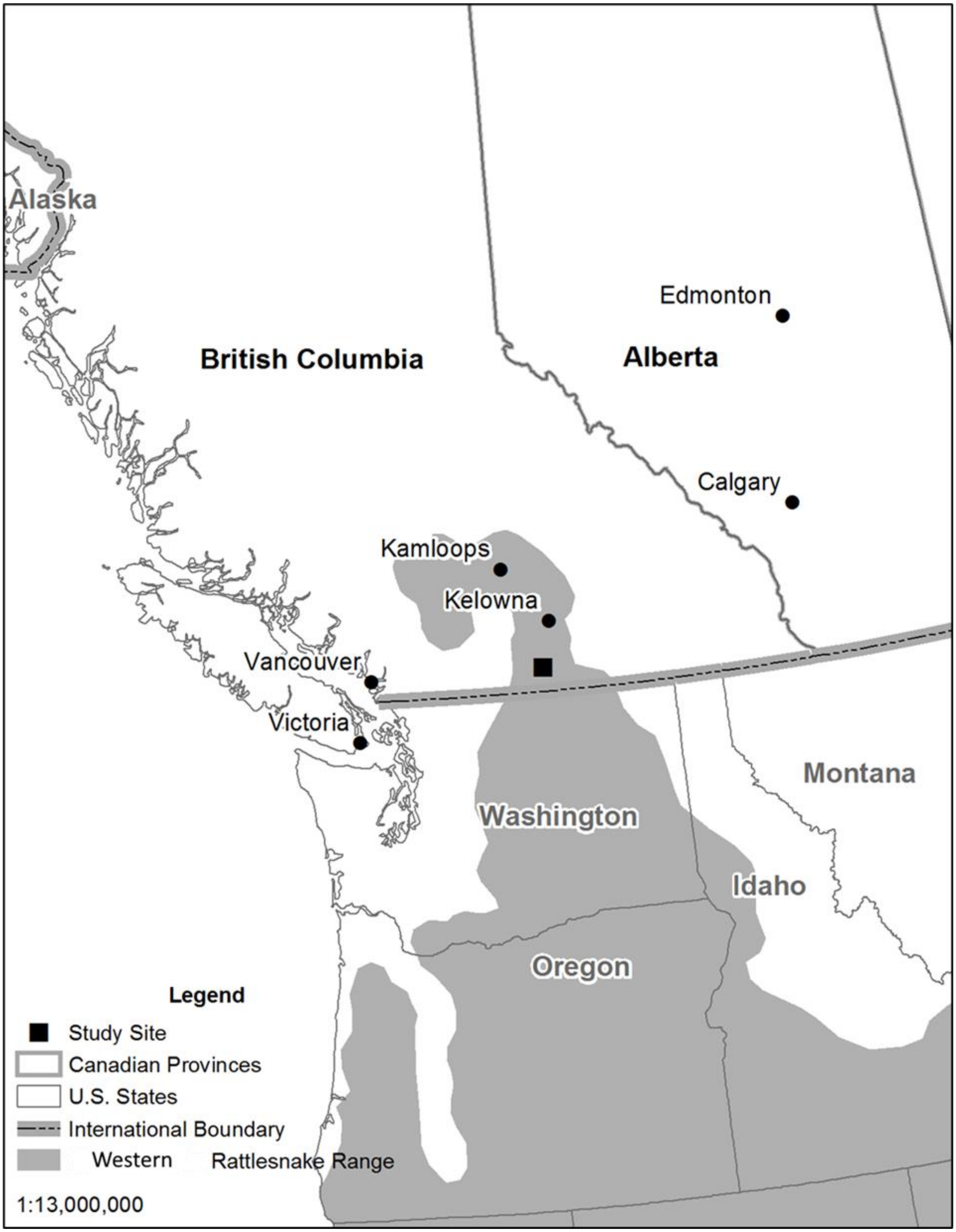

Figure 1.1. Study site location within the Western Rattlesnake (*Crotalus oreganus*) range in western North America (from Winton 2018).

running ≈ 5m parallel to the road, and each year the vegetation along the road shoulder is mowed to a height of less than 0.5 m for a distance of 1.8 m from the road edge. The basin largely lacks infrastructure apart from the roads and the Dominion Radio Astrophysical Observatory (run by the National Research Council Canada). The basin is surrounded by rolling hills, steep bluffs, and two golf courses with small residential communities and ponds lie just outside the basin. Elevation within the basin ranges from 500 – 1100 m.

The South Okanagan valley is characterized by hot, dry summers and long, cold winters with temperatures reaching below freezing (Meidinger and Pojar 1991). The average temperature patterns were consistent throughout the years of the study, although February 2019 was colder than usual. The total monthly precipitation patterns also were relatively similar with more variation, particularly in 2017, which had high amounts of precipitation in May and none in July or August (Figure 1.2).

Initial research on Western Rattlesnake road mortality in the White Lake Basin began in 2015, focusing on road surveys and extensive mark-recapture efforts (Winton et al. 2018, 2020, Figure 1.3). New ecopassages were installed in 2017, and cameras were mounted inside the entrances in 2018. The first two years to have cameras deployed in the ecopassages throughout the entire active season were 2019 and 2020. Drift fencing was installed at ecopassage entrances in early 2019 (Figure 1.4). Details of these structures can be found in the Methods section of Chapter 3.

The second year of this study took place in the summer of 2020 during the height of the Covid-19 pandemic, which admittedly may have altered traffic patterns in the White Lake Basin (Destination British Columbia 2017). However, field methods and the collection of traffic data were conducted consistently with other years of the study.

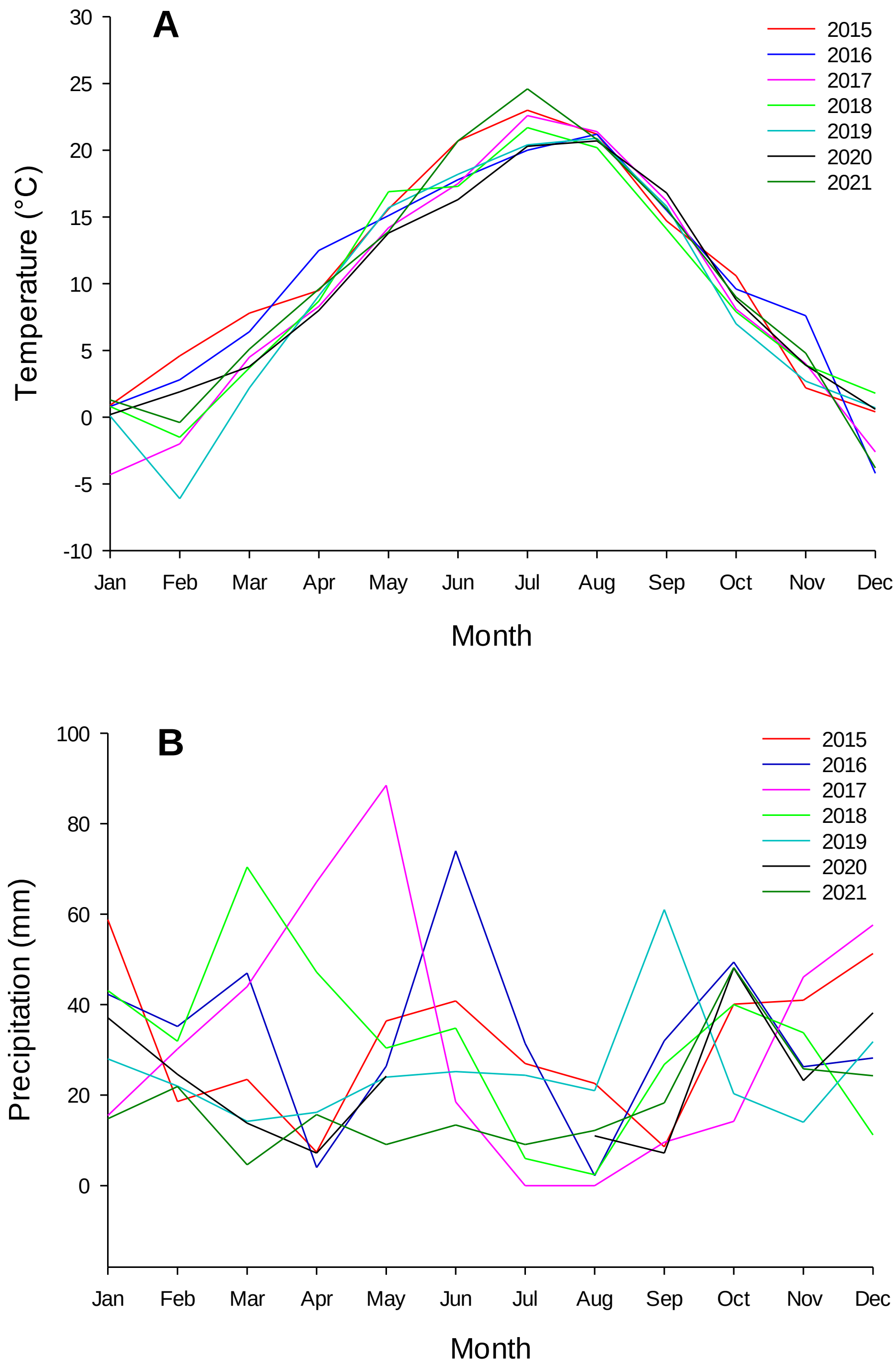

Figure 1.2. A) Mean monthly temperature (°C) and B) total monthly precipitation (mm) in the South Okanagan during the study years (2015-2021) as measured at the Penticton Regional Airport, British Columbia, Canada (49°N, 119°W; Environment and Climate Change Canada 2024).

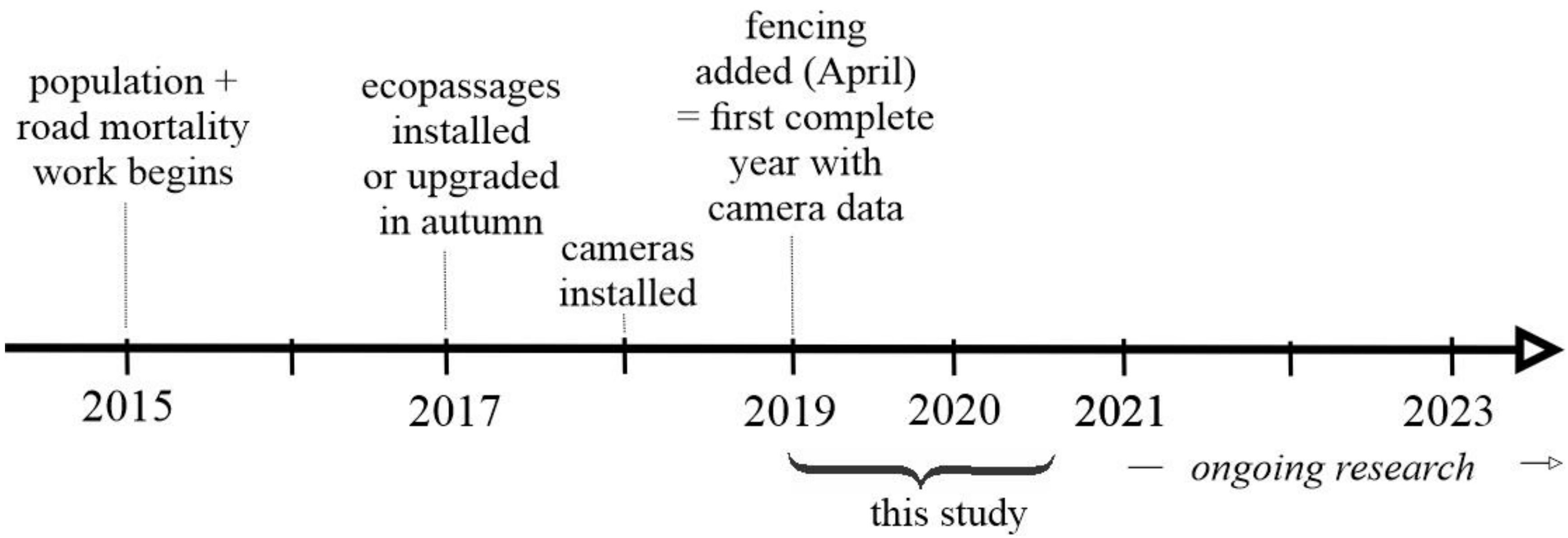

Figure 1.3. Timeline of snake research in the White Lake Basin, British Columbia.

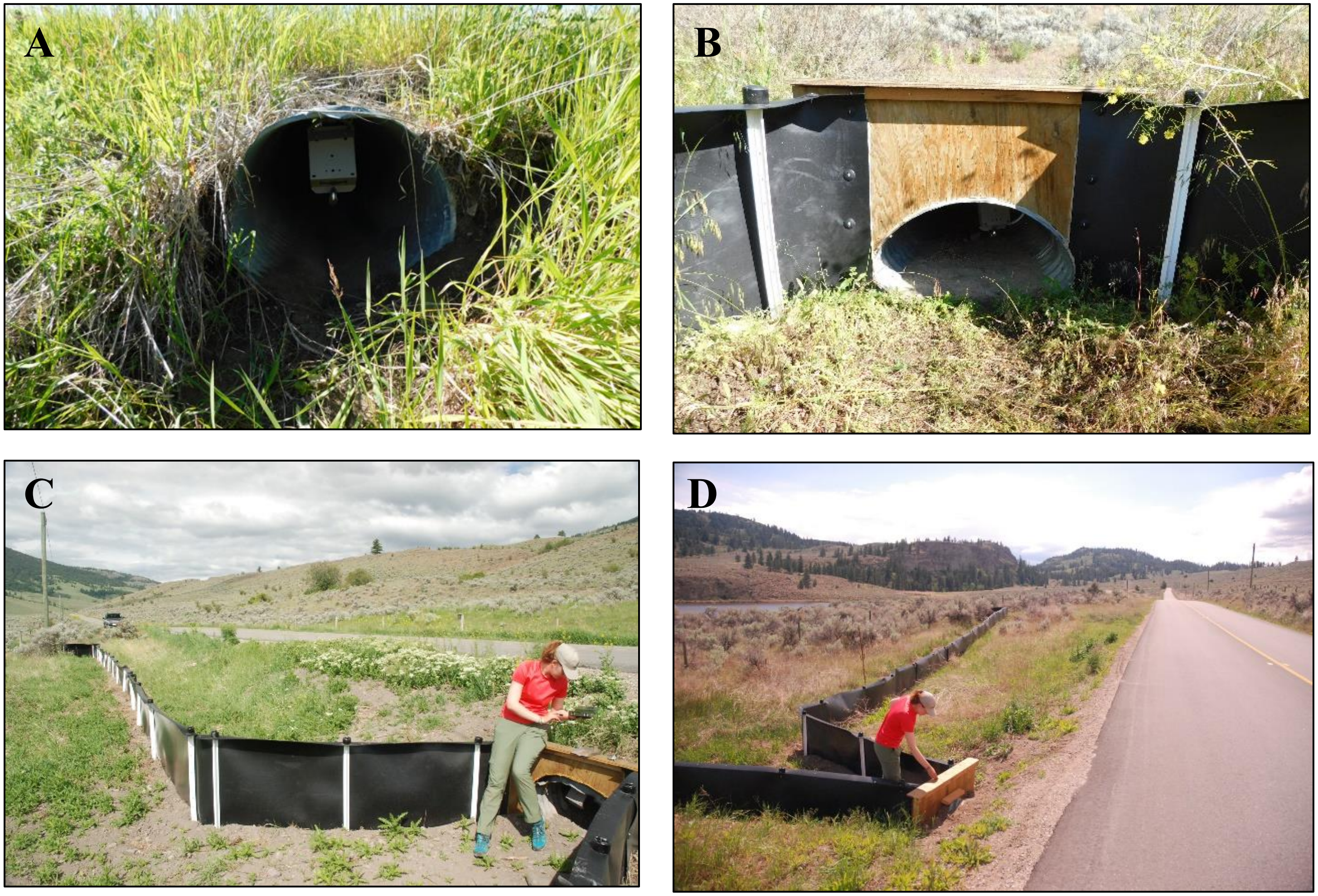

Figure 1.4. Examples of snake road mortality mitigation measures in the White Lake Basin, British Columbia, including A) a modified culvert equipped with a camera, with the entrance unaccompanied by fencing, B) a newly-installed semi-circular ecopassage equipped with a camera and fencing surrounding the entrance, and C & D) drift fencing positioned to funnel snakes into under-road ecopassages. (Photo credits: Karl Larsen)

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CHAPTER 2

TRANSITIONAL RESPONSE OF WESTERN RATTLESNAKES (*CROTALUS OREGANUS*) TO THE INSTALLATION OF ROAD ECOPASSAGES AND DRIFT FENCING

INTRODUCTION

Roads and traffic are a major threat to wildlife populations across many taxa, both through direct mortality and the fragmentation of habitat, the latter creating barriers within populations (Baxter-Gilbert et al. 2015, Fahrig and Rytwinski 2009). Reptiles are one of the fastest-declining taxa globally, being particularly vulnerable to road effects due to their small size, attraction to roads for thermoregulation, and low vehicle avoidance (Baxter-Gilbert et al. 2015, Fahrig and Rytwinski 2009). Reptile species with life histories characterized by low reproductive rates and low adult mortality are even more vulnerable to the demographic consequences of road mortality (Boyle et al. 2021, Forman et al. 2003). These impacts can be exacerbated in reptile populations at higher latitudes that experience additional constraints due to climate (Macartney et al. 1990).

To reduce road mortality while improving habitat connectivity for vulnerable reptile species, the implementation of mitigation measures has become increasingly common (Jarvis et al. 2019, Dillon et al. 2020). Ecopassages are commonly installed under roads to facilitate the crossing of roads, and often are coupled with drift fencing meant to direct wildlife into the passageways. Such structures usually are targeted towards recovering a specific species at risk. However, despite the increasing application of these mitigative tactics for reptile survival, in-depth analyses of their effectiveness remain rare, both in terms of their direct impact on roadkill rates as well as the persistence of populations or communities (Boyle et al. 2021, Fahrig and Rytwinski 2009).

Intuitively, the response of wildlife to newly-installed mitigation structures will vary in time following the moment of installation. Whelan et al. (2002) outlined how population changes following wildfire will depend on the organism, and their species-specific mortality, reproductive rates, immigration, and emigration. These responses also are likely to be age- or stage- specific, being highly linked with the life cycles of the organism as well as the timing and severity of the disturbance. In referring to the effects of wildfire, Warren et al. (1987) termed the period immediately following disturbance the ‘shock phase’ of ecosystem recovery, followed by the recovery phase. Analogous effects may be associated with the implementation of anthropogenic structures designed to reduce road impacts on wildlife: their sudden presence on the landscape likely will impact species differently, leading to ‘stages’ in the recovery of populations and communities. For example, crossing structure use increases over time for large mammals, as animals become acclimatized and learn to start using them (Ford et al. 2017, Gilhooly et al. 2019, Seidler et al. 2018).

Similar research on the response of reptiles and other small-bodied animals to the addition of road crossing structures is lacking (Baxter-Gilbert et al. 2015, Boyle et al. 2021, Dillon et al. 2020). Initiating monitoring programs immediately after mitigation is affected allows the ‘shock phase’ response to be examined, while providing a foundation for longer-term monitoring. Knowledge of if and when these structures become effective at reducing roadkill, or how responses will vary through time, becomes essential for designing monitoring programs and/or interpreting results at different stages of response.

I postulate that snakes are a taxa where initial responses to new structures, such as mitigative ecopassages, may pass through a ‘shock phase’. Conspecific scent trailing is important for snakes, in particular communally aggregating snakes (Brown and MacLean 1983, Muellman et al. 2018). Therefore, it may take time for snakes to initially find these new corridors and establish repetitive and frequent use of structures and pathways. Snakes demonstrating large, annual migration distances, fidelity to their overwintering and summer habitat, and a tendency to take the same path and re-visit specific habitat features from year to year likely are especially reliant on scent trails (Duval et al. 1990, Gomez et al. 2015, Parker and Anderson 2007). These conditions likely will result in an even more pronounced ‘shock phase’.

As rattlesnakes exhibit the characteristics mentioned above, they may demonstrate a muted response to mitigation structures during the ‘shock phase’. Establishing scent trails in the ecopassages likely is important for other rattlesnakes to begin using them since neonates of some rattlesnake species follow scent trails of conspecifics (Brown and MacLean 1983, Muellman et al. 2018, Scudder et al. 1988). Furthermore, since vipers take relatively longer to transform energy from prey into offspring than similar-sized endotherms (i.e. longer generation times), the result is a slower population-level response to changes in their environment (Nowak et al. 2008). This slow response time likely is amplified in northern viper populations that face shorter and cooler active seasons, such as rattlesnakes living in Canada (Macartney and Gregory 1988).

Winton et al. (2018, 2020) collected baseline data on the impacts of traffic on a rural community of Western Rattlesnakes (*Crotalus oreganus*) in British Columbia, including demographic changes. Their work documented significant road mortality in the population (≈ 7% annually) despite low traffic (350 vehicles/day) on roads bisecting an otherwise reasonably-intact ecosystem. The Winton study also identified specific roadkill 'hotspots' along the two roads in the area, where relatively high mortality appeared linked to migratory movement corridors to and from hibernacula (Winton 2017). In an effort to curb road mortality, eight ecopassages were established within the study area at these locations (see Figure 3.1).

Using identical survey methods to Winton et al. I assessed the immediate impact (during what is likely a 'shock phase') of the ecopassages on the road mortality rates of rattlesnakes in the same study area. By repeating the same walking surveys and methodology used by Winton et al. (2018, 2020), I examine changes in road mortality and population demographics during the years immediately following the establishment of the ecopassages and associated drift fencing. I predicted I would not detect any significant changes in roadkill rates or population size during this two year time period post-implementation of ecopassages and drift fencing, due to the limited amount of time available for the snakes to adjust to the structures.

METHODS

Study Site

This study was conducted in the White Lake Basin (latitude 49.318°N, longitude 119.638°W) in the South Okanagan region of British Columbia, Canada (Winton et al. 2018, 2020). Here, near the northern limit of the Western Rattlesnake's range (see Figure 1.1) the typical annual activity period of the snake generally spans April – October. The basin consists of open shrub-steppe grassland habitat, surrounded by rolling hills and steep bluffs. The area is characterized by Big Sagebrush (*Artemisia tridentata*), Bluebunch Wheatgrass (*Agropyron spicatum*), and Ponderosa Pine (*Pinus ponderosa*) (Meidinger and Pojar 1991). The area is managed with the objective to "integrate livestock management with conservation of habitat for species at risk" by The Nature Trust of British Columbia (TNTBC 2022). Traversing the valley bottom is a paved, undivided two-lane road (BC Class 5 highway) with an unposted speed limit of 80 km/h.

In September 2017, four ecopassages were newly installed through the study area at identified snake roadkill hotspots (Winton 2017), and modifications were made to an additional four existing drainage culverts. This produced eight ecopassages within a 6.5 km stretch of road, with the intended purpose of lowering Western Rattlesnake road mortality. All eight ecopassages were made from corrugated metal, although, the newly-installed ecopassages were more oval-shaped than the older, rounder drainage culverts (See Figure 1.4. A & B). The ecopassages had an average length of 12 ± 1 SD m, height of 45 ± 8 SD cm, and width of 67 ± 12 SD cm, with an average openness (= height * width / length) of 2.5 ± 0.5 SD cm. A substrate of sand (≈ 5 cm deep) lined the bottom of the culverts.

In April 2019, drift fencing was added to some of the entrances of the ecopassages, with a design specifically targeted towards directing snake movement (model AMX-SP40, Animex Wildlife Fencing Solutions, see Figure 2.1) into the ecopassages (Baxter-Gilbert et al. 2015, Colley et al. 2017). Construction of the fences was hampered by bedrock, drainage patterns, and cattle use, thus preventing a robust experimental design, such as altering fencing on upslope versus downslope entrances of the ecopassages. Still, drift fences (varying lengths, and ≈ 75 cm in height) were established at one end of each of seven ecopassages (See Figure 1.4. C & D, and Figure 3.1), while one received no fencing. Fence lengths flanking the entrances of the ecopassages ranged from a total length of 43-63 m, with an average length of 51.5 ± 8.3 SD m. A total of 0.4 km of the 23.4 km of road edge within the study area were fenced.

Calculating Road Mortality Rates

To determine roadkill rates, I duplicated the survey method established by Winton et al. (2018). An 11.7 km route along White Lake Road and Willowbrook Road was surveyed approximately every 3 days during the active season, from April to October. Road surveys were conducted on foot: two observers walked along opposite edges of the road, scanning for dead and alive snakes on the road and 1.8 m vegetation control zone on the road shoulder. Throughout the duration of the research at this site road survey effort (total kilometers surveyed) varied from year to year, with a minimum of 465 kms (2015), a maximum of 865.8 kms (2017), and an average of 625.7 kms surveyed.

Also following Winton et al. (2018), I conducted four observer-bias experiments to calculate the detection probabilities of dead snakes during road surveys. A third researcher planted dead snakes (previously roadkilled) of a variety of sizes on the road and shoulder at sites along the road survey route, and revisited these sites immediately following the survey (within approximately 30 minutes) to determine whether the observers (hereon known as surveyers) had detected or missed each carcass. Four observer detection trials were completed during walking road surveys from July-September, 2020. Detection probability was calculated using the proportion of planted snakes observed by the surveyers, which then were used in the model developed by Winton et al. (2018) to provide a more accurate roadkill estimate.

Detections of dead rattlesnakes on the road survey route outside of official survey times also were recorded, and combined with the gross number of road survey detections to provide a number of total detections for that year. Two traffic counters (TRAFx, G4) – one on each of the two roads within the survey route – were provided by the BC Ministry of Transportation and Infrastructure (MoTI) to provide daily traffic volumes. Willowbrook Road was consistently busier than White Lake Road, so data from the traffic counter on Willowbrook Road was used to calculate the mean number of vehicles per day in the study site (maximum daily average) for the time period June 1 – August 31 of each year, when traffic volumes typically peak in the region due to school closures and tourist travel.

Mortality rates due to roadkill were modelled following Winton et al. (2018): the density of dead rattlesnakes on the road, observer detection error, rate of carcass removal by scavengers, and time since the last survey were combined to calculate a rate with units of deaths/km/day. I conducted a preliminary analysis of the trend in snake mortality rates using linear regression.

Monitoring Population Trajectory

To track the rattlesnake population in the basin, as per Winton et al. (2020), I conducted mark-recapture work at 6 focal hibernacula located within 400 m of the road. Repeated visits to these hibernacula occurred during spring egress and fall ingress from Spring 2015 to Autumn 2021. Marking of snakes was done using implantable PIT (passive integrated transponder) tags (Mini HPT8 PIT Tag, Biomark, Inc.). The general area also was arbitrarily surveyed throughout the active season for snakes from these dens.

Appending my extended mark-recapture data with that collected by Winton et al. (2020), I produced updated Jolly-Seber population and survival estimates of the rattlesnake population using data from the 2015-2021 capture sessions. Jolly-Seber population estimates cannot be calculated for the first and final sampling periods in a data set, and additionally, survival estimates cannot be calculated for the second last and final periods (Krebs 1989): therefore, the total of seven capture sessions provided five Jolly-Seber population estimates (2016-2020) and 4 survival estimates (2015-16 to 2018-19). Estimates were generated using the *Rcapture* package (Baillargeon and Rivest 2007) for RStudio (version 4.1.1; R Core Team, 2021), with each active season (April-October) being considered a 'capture session'. I used linear regression to conduct preliminary analyses of trends in the yearly population estimates.

RESULTS

Road Surveys

A total of 53 road surveys were conducted in 2019 and 49 in 2020, with an average of 2.6 and 3.2 days between surveys, respectively. Between road surveys and incidental observations outside of the surveys, a total of 35 dead rattlesnakes were found within the survey route in 2019, and 21 in 2020. Analogous detections from research done in 2015-2018 were 36, 56, 28, and 29, in each year respectively (2015-2016 from Winton et al. 2018, 2017-2018 from Larsen, unpublished data).

Traffic Rates

Traffic rates shown in Figure 2.1 represent the daily average number of vehicles on Willowbrook Road (the busier of the two roads within the survey route) from June-August. 2015 was the year with the lowest traffic rate of 266 vehicles per day, and 2020 was the highest with a rate of 505 vehicles per day. Unfortunately, data for 2018 were lost when the traffic counters were removed prematurely by government personnel.

Roadkill Rates

Roadkill rates show a trend suggestive of a decline over the study years, despite traffic rates increasing (Figure 2.1). The linear regression results indicate that this trend is insignificant, but this analysis should be considered preliminary due to the insufficient sample size of six years. The year 2018 had low roadkill rates but unfortunately the missing traffic data makes it impossible to comment on how the rates in that year were related to traffic.

Mark-Recapture

Total snakes captured (new individuals + recaptures) ranged from 172 to 324 individuals per year across all study years. In the pre-mitigation years of the study (2015/2016) mid-sized adults in the 60-70cm SVL size category were the greatest proportion of captures, with neonates in the 20-30 cm category having about half as many captures. In the post-mitigation years of the study (2019/2020) the proportion of neonate captures was higher than any other individual size category, resulting in the size distribution of snake captures being inconsistent in the pre- and post-mitigation years (Figure 2.2. A & B, Two-sample Kolmogorov-Smirnov test, $D = 0.191$, $P < 0.001$). Also, there were many fewer roadkilled neonate carcasses detected post-mitigation despite the high proportion of neonate captures in that time period (Figure 2.2 B & D). Roadkilled neonates were classified more broadly in age categories (versus SVL size categories), as carcasses are often so damaged that they are much more difficult to accurately measure than live snakes. For these reasons, I limit the comparison over time to only adult snakes, where there was less variation in the proportion of snakes captured from year to year.

Population Estimate

Jolly-Seber population estimates for all snakes and adult snakes were the highest in 2019 (900 ± 96 SE and 405 ± 47 SE, respectively) and lowest in 2017 (526 ± 54 and 272 ± 25). Population and survival estimates for all snakes appear in Figure 2.3 (top), alongside the original population estimate from Winton et al. (2020). Figure 2.3 (bottom) displays the Jolly-Seber population and survival estimates for adult rattlesnakes only (≥55 cms). In both cases population estimates were higher during the period when mitigation structures are being installed, but then lower in 2020. The linear regression results indicate that the population trends for all snakes and adult snakes are insignificant, but these results should be considered preliminary due to the insufficient sample size of five years.

DISCUSSION

This study highlights the necessity of monitoring mitigation effects both before and after implementation, given the difficulty of drawing firm conclusions based on short-term data. Although there was a suggestion of decline in rattlesnake roadkill rates despite an increase in traffic rates post-mitigation, it may be too early to detect concrete shifts in population size and survival rates. Still, this study provides an important snapshot of the 'shock phase' response to mitigation structures, and a strong foundation for monitoring beyond it (Wong *in prep*).

The two post-mitigation years have some of the lowest road mortality rates, but the highest traffic rates. Camera data (see Chapter 3) confirms that rattlesnakes were using the ecopassages in some capacity, and therefore these results could be due to ecopassage use by the snakes. However, road avoidance by rattlesnakes is another possible explanation. Other taxa have demonstrated increased road avoidance with increased traffic volume (Jacobson et al. 2016, Loraamm et al. 2021). Western Rattlesnakes in British Columbia display smaller home ranges and shorter range lengths in disturbed areas compared to those in areas with no human disturbance (Lomas et al. 2019). Another Canadian rattlesnake species (*Sistrurus catenatus catenatus*) moves shorter distances and less frequently with increased human disturbance, defined by a higher number of vehicles on roads and more hikers (Parent and Weatherhead 2000). If higher volumes of traffic are causing limited movement of Western Rattlesnakes around the road, it could lead to barrier effects that can have significant impacts on populations. These include reduced gene flow, fragmentation of populations close to the road, and altered habitat use that increases energetic costs (Forman et al. 2003, Paterson et al. 2019). Since northern snake populations already face environmental factors that limit growth and reproductive frequency, the added energetic costs of road avoidance could be detrimental to their persistence (Macartney and Gregory 1988). Western Rattlesnake populations in BC have significant genetic differentiation and limited population connectivity in areas with major highways (Schmidt et al. 2019), so future development around the basin could further increase traffic volumes, exacerbating the situation.

The positive Jolly-Seber population and survivorship estimate trends represent a relatively short period, but are still encouraging. Although these data suggest an increase in rattlesnake population size after the ecopassage installation in 2017, it is unlikely that a population-level change of that magnitude in such a short time frame is directly attributable to the mitigation measures, particularly when a population drop occurs again in 2020. Snake populations can naturally fluctuate over time due to both density dependence and weather-induced factors affecting juvenile survival or prey availability (Altwegg et al. 2005, Madsen et al. 2006, Shine et al. 2021), but these factors were not measured in this study. Northern vipers exhibit more “K-selected” life-history traits than other snakes, including delayed maturation, infrequent reproduction, and low fecundity (Macartney and Gregory 1988). This results in more density-dependent population growth that typically would not express itself as rapid and drastic fluctuations (Shine et al. 2021). Therefore, environmental factors also are likely to be contributing to these changes in population. Although our data set is relatively extensive for this type of assessment, the variation in these population parameters suggests long-term monitoring will be necessary to conclude if there are population-level impacts of these mitigation measures. Currently, a generation for Western Rattlesnakes in British Columbia is estimated at approximately 13.7 years (Maida et al. 2018). Therefore, one would expect a detectable response in the population to take longer to manifest than the time period encompassed in this study. Increasing use of crossing structures over time has been documented in mammals (Ford et al. 2017, Gilhooly et al. 2019, Seidler et al. 2018), but remains to be clearly shown in reptiles.

The population estimates calculated using only adult rattlesnakes are a better representation of the trend throughout the duration of the study than those using all the snakes. The higher proportion of neonates captured in the post-mitigation years (2019 and 2020) is possibly due to a difference in sampling effort or marking method from the pre-mitigation years. Initially scute clipping was used to identify neonate rattlesnakes because PIT tag needles were too large to be used on them. Scute clips are visible for a short time, but as the snake grows and sheds they become difficult to distinguish (Fitch 1987). As smaller PIT tags became available in the later years of the study, we began PIT tagging all neonate rattlesnakes, which is a more reliable form of identification (Jeminson et al. 1995). Therefore, a scute clipped neonate caught later as an adult may have been considered a new individual, contributing to inaccuracies in the ‘all snakes’ survivorship and population estimates. Regardless, since neonate survival appears low (6 – 46%; Macartney 1985), using only adults for this type of analysis likely is a more accurate predictor of functioning population size. The capture and marking of relatively larger numbers of neonates, such as seen in 2019 and 2020, would inflate the population estimate, given age class survival rate is not factored into a Jolly-Seber population estimate.

Should a more pronounced decline in roadkill and an increase in the population size be expected as monitoring continues beyond the ‘shock phase’? In other species, snakes

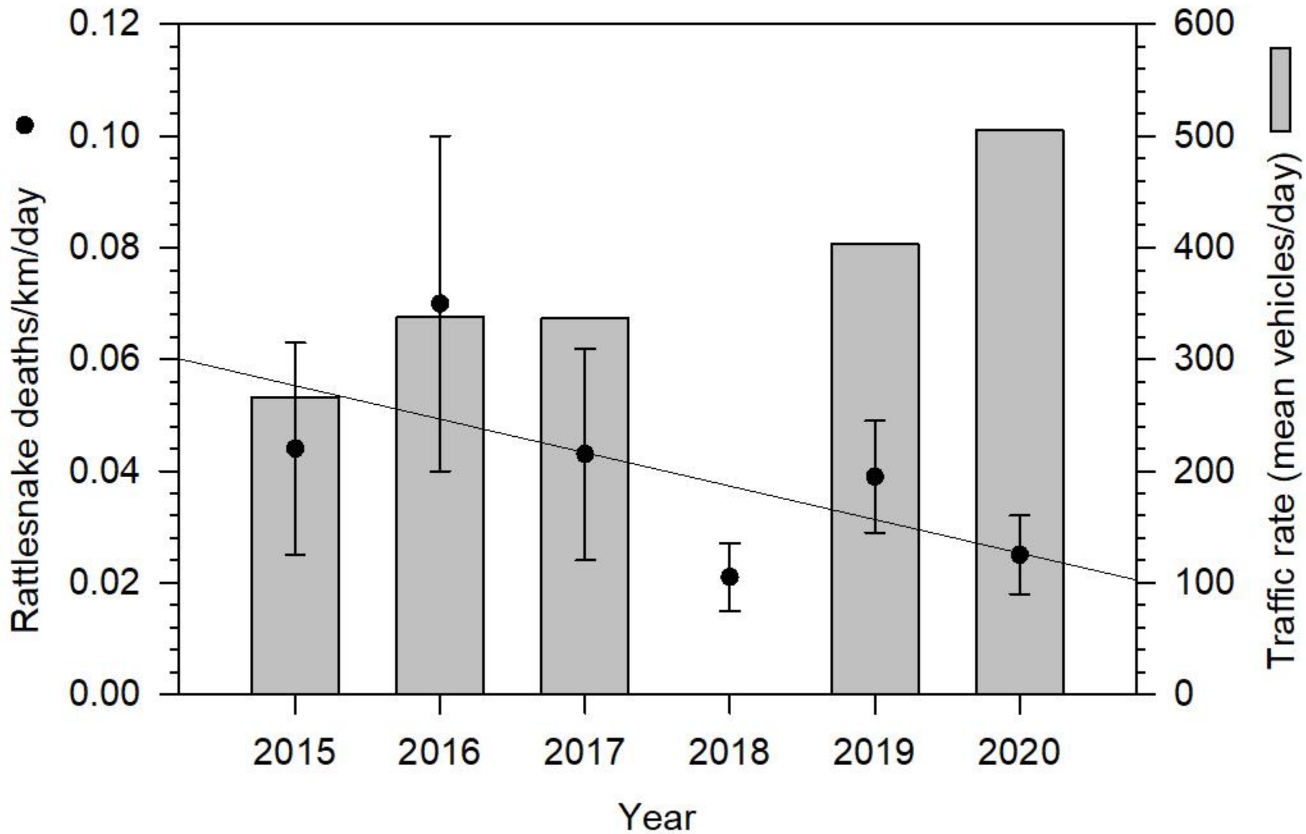

Figure 2.1. Calculated Western Rattlesnake (*Crotalus oreganus*) roadkill rates (± 1 SE) (left ordinate, black data points) with a linear regression line (y = -0.006x + 0.0613, $R^2 = 0.42$, $P = 0.17$), and average traffic rates (right ordinate, bar graph) from 2015-2020. Traffic data were unavailable for 2018. Solid arrow on bottom indicates when ecopassages were installed in 2017; broken arrow indicates the installation of drift fencing prior to the 2019 active season.

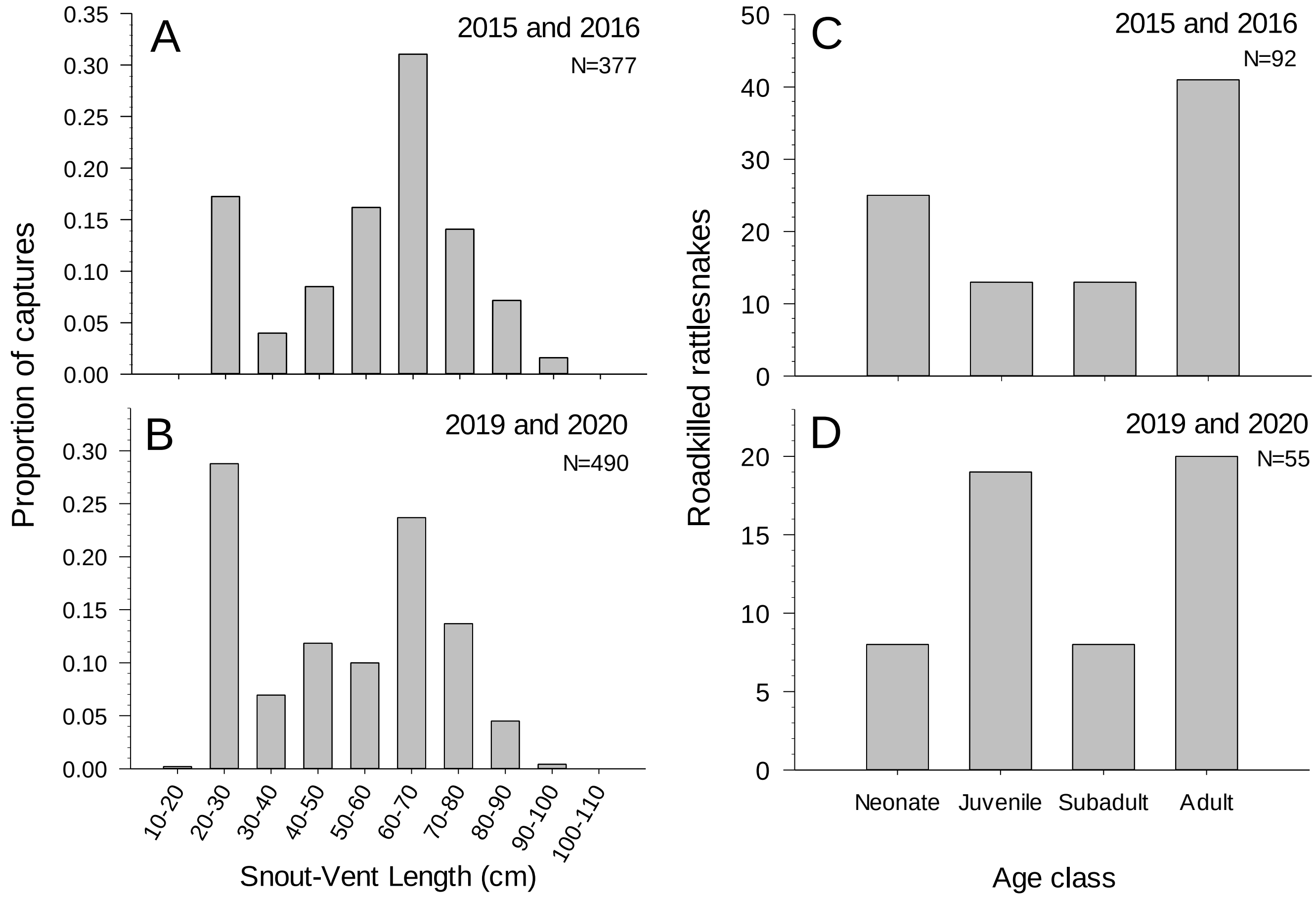

Figure 2.2. A & B) Proportion of Western Rattlesnakes captured across size classes at six focal dens in the White Lake Basin before (A) and after (B) ecopassages were installed. Snakes with measured lengths falling exactly on an interval cut-off were included in the larger category. C & D) Number of roadkilled Western Rattlesnakes in broad age categories as detected in the White Lake Basin before (C) and after (D) ecopassages were installed. Neonates are defined as rattlesnakes with snout-vent lengths of < 30 cm, juveniles are 30-40 cm, subadults are 40-55 cm, and adults are > 55 cm.

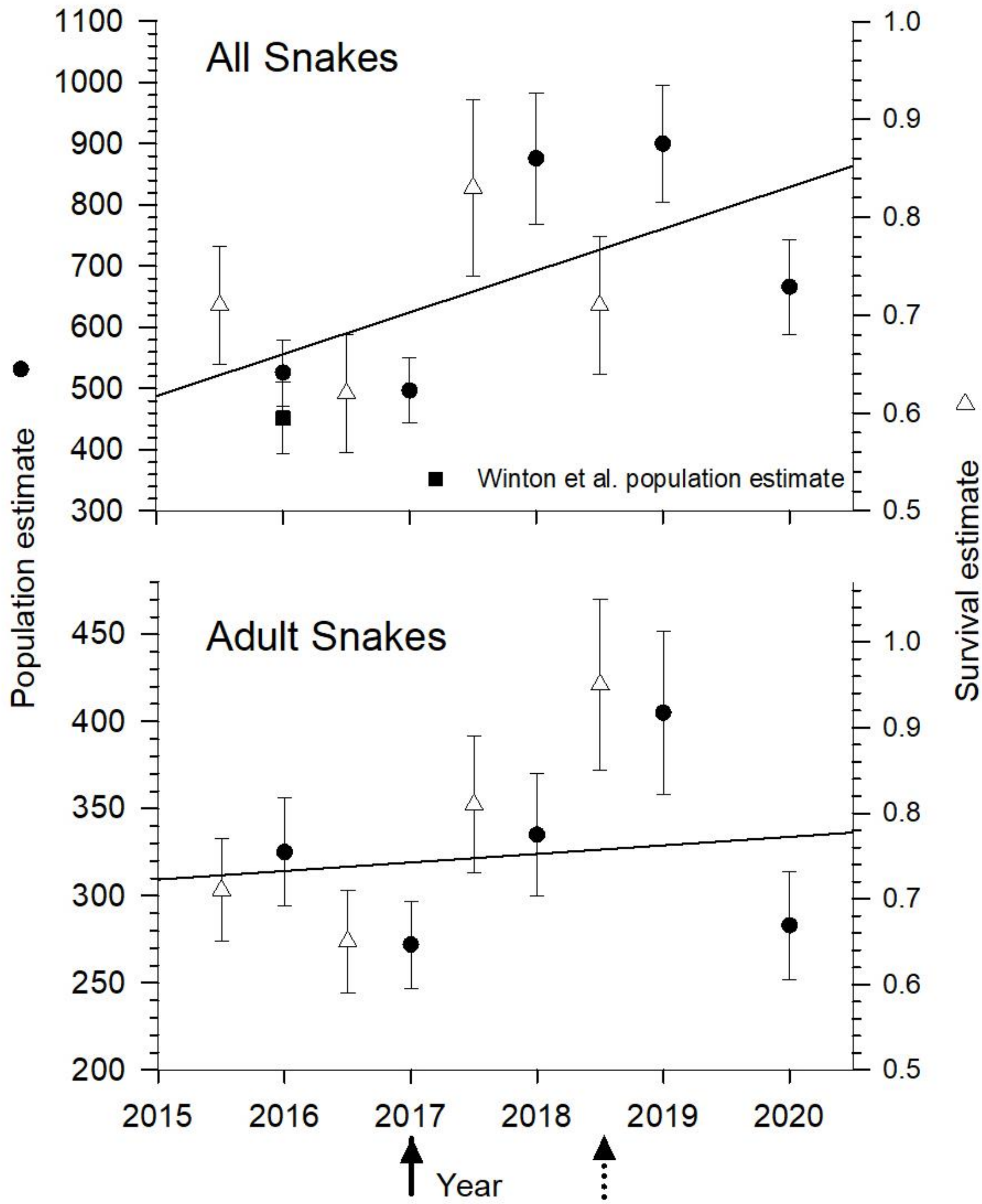

Figure 2.3. Jolly-Seber estimates (± 1 SE) for population size and survival of all (top) and adult (bottom) Western Rattlesnakes (*Crotalus oreganus*) at focal dens in the White Lake Basin, BC from 2015-2020. Also shown in the top graph is the original Winton et al. (2020) population estimate for 2016 (■) as calculated using the shorter mark/recapture history available during their study. Linear regression lines for the population estimates of all snakes ($y = 68.3x + 488.1$, $R^2 = 0.33$, $P = 0.32$) and adults snakes ($y = 4.9x + 309.3$, $R^2 = 0.02$, $P = 0.81$) are also shown. The solid arrow represents the installation of snake-targeted ecopassages and the dashed arrow indicates the installation of directional fencing as road mortality mitigation.

follow scent trails of conspecifics to locate both mates and hibernacula (Costanzo 1989, LeMaster et al. 2001). Thus, increased usage may even be more exponential than linear. Given the duration of this study is considerably less than the estimated generation time for Western Rattlesnakes in British Columbia, this suggests that a numerical response of the population would take more time than the two years post-mitigation period to show clear positive effects. However, at this point there is insufficient data to determine how long the 'shock phase' will persist, or if this species will exhibit different recovery phases. Before longer-term data are available, the conservation Precautionary Principle should be invoked, namely that "when an activity raises threats of harm to human health or the environment, precautionary measures should be taken even if some cause and effect relationships are not fully established scientifically" (Raffensperger and Tickner 1999). However, this must be balanced against the fact that the deployment of ecopassages is costly, both in terms of installation and maintenance.

Mark-recapture success, and the subsequent confidence in Jolly-Seber estimates, is affected by a number of factors under varying levels of control by the researchers. Changes in field personnel in 2021 likely affected survey effort, as will weather patterns and other seasonal variants that influence emergence patterns and capture success of snakes. These factors may have contributed to the differences in population estimates I calculated across the latter years of my study, but this non-conclusive outcome would be hard to eliminate given the nature of this study. I believe the main conclusions made herein remain sound.

Understanding the rate at which mitigation efforts, including ecopassages and fencing, generate behavioural and numerical responses in smaller vertebrates remains important for assessing their efficacy. As with natural disturbances, immediate responses may differ considerably from longer-term ones, but together they paint a complete picture of the response which can aid in the conservation of populations. This study reveals a potential decline in Western Rattlesnake roadkill in the 'shock phase' immediately following ecopassage installation, although longer-term work beyond the 'shock-phase' is needed. This study shows that immediate, obvious effects from mitigation efforts on target populations should not be expected, while providing a cogent argument for ensuring commitment to projects like this are made for the long term.

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CHAPTER 3

DIFFERENT SNAKE SPECIES SHOW VARYING RESPONSES TO ECOPASSAGE INSTALLATION

INTRODUCTION

Crossing structures often are used to lessen the impacts of roads on wildlife, as they reduce roadkill and aid in habitat connectivity (Glista et al. 2009, Jarvis et al. 2019, van der Grift et al. 2013). There has been extensive research done on their efficacy for large mammals, since collisions with such wildlife have safety and economic impacts on humans (Donaldson and Elliott 2021, Lee et al. 2020, Rytwinski et al. 2016). However, smaller animals face the same road effects and, although collisions with vehicles cause minimal damage to humans, these species can benefit from mitigation structures just the same (Fahrig and Rytwinski 2009, McGregor et al. 2007, Plante et al. 2019).

Reptiles are one of the fastest declining taxa globally, and particularly vulnerable to road effects due to their small size, attraction to roads for thermoregulation, and low vehicle avoidance (Baxter-Gilbert et al. 2015, Fahrig and Rytwinski 2009). The implementation of mitigation measures to reduce road mortality while improving habitat connectivity for vulnerable reptile species has become increasingly common (Jarvis et al. 2019, Dillon et al. 2020). To facilitate the crossing of road corridors, 'ecopassages' are installed under roads and coupled with barrier fencing designed to direct wildlife into the passageways. Such initiatives often are prompted by a recognition or concern that roads are threatening the viability of wildlife populations (Aresco 2005, Colley et al. 2017, Polak et al. 2019). Although the application of these mitigative tactics is increasing, few studies have fully assessed their use and resulting effect on road mortality for the target species (van der Grift et al. 2013, Baxter-Gilbert et al. 2015).

Intuitively, one of the most efficient and cost-effective methods of monitoring crossing structures are wildlife cameras (Ford et al. 2009, Welbourne et al. 2017). Cameras may be installed inside ecopassages and set to take movement-activated photos and/or timed photos at selected intervals, enabling a considerable amount of data to be collected with minimal effort, and without disturbing the animals (Wearn and Glover-Kapfer 2019). Photos from wildlife cameras provide information on species' use, and allow comparisons between species (Pagnucco et al. 2011, Wearn and Glover-Kapfer 2019). Although research using camera traps has primarily been focused on mammals, fish, and birds, with improving technology it can be a useful monitoring tool for reptiles and amphibians as well (Molyneux et al. 2017, Wearn and Glover-Kapfer 2019, Welbourne et al. 2017).

Ecopassages for snakes were established in southern British Columbia, Canada, after research (Winton et al. 2018, 2020) documented significant road mortality and a population decline in Western Rattlesnakes (*Crotalus oreganus*). Cameras were used to monitor wildlife use, principally the three at-risk snake species in the area: Western Rattlesnakes (F. Viperidae - *Crotalus oreganus*), and two species in the Family Colubridae: Great Basin Gophersnakes (*Pituophis catenifer deserticola*), and Western Yellow-bellied Racers (*Coluber constrictor mormon*). In addition, drift fences were installed at ecopassage entrances prior to the snakes' active season in 2019 (see Chapter 1). In this chapter I compare camera detections in the ecopassages between the three snake species immediately upon their installation (the 'shock phase' – See Chapter 2).

I predicted that the general response to the installation of the new ecopassages and drift fencing would vary among the three target species in the White Lake Basin. Despite occupying the same ecosystem, the three species differ in habitat requirements and foraging strategies. Rattlesnakes leave communal hibernacula on rocky, warm slopes in the spring for summer foraging grounds, where they spend much of their time under cover of rocks and shrubs. They are ambush predators (Clark 2004, Theodoratus and Chiszar 2000), largely remaining concealed so that they can forage successfully (B.C. Ministry of Water, Land and Air Protection 2004c, COSEWIC 2015). In contrast, gophersnakes and racers appear to use a broader range of locations for hibernating and are active foragers (B.C. Ministry of Water, Land and Air Protection 2004a and 2004b). Gophersnakes search rodent holes and climb to reach bird nests, and racers will approach and chase their prey (B.C. Ministry of Water, Land and Air Protection 2004a and 2004b, COSEWIC 2013). Because of this, these two members of the colubrid family of snakes move more often and with more tortuosity than rattlesnakes (Rouse 2006, Wong *in prep*).

I predicted that ecopassage use would increase over time for all three species of snakes. This trend has been observed in large mammals, where the animals become acclimatized and learn to adopt the structures (Ford et al. 2017, Gilhooly et al. 2019, Seidler et al. 2018). In addition, snakes rely on olfactory cues for navigation, and scent trails from conspecifics develop over time (Brown and MacLean 1983, Costanzo 1989, Muellman et al. 2018). Although the timeframe of my project was relatively short, it focused on a very important period when an assessment of mitigation success is highly desirable, particularly if other similar work is being contemplated. All three of my target species are listed as Threatened under the Canadian Species at Risk Act (2002) and recognized as 'Special Concern' within the province of British Columbia (Racer Management Team Working Group 2013, Southern Interior Reptile and Amphibian Working Group 2016a and 2016b). Thus, the two main objectives of my study were to (1) quantify the short-term response of these three species to newly installed mitigation structures, and (2) compare differences in temporal and spatial ecopassage use between species. A secondary objective of this study was to assess the immediate efficacy of drift fence associated with ecopassages.

METHODS

Study Site

This study was conducted in the White Lake Basin (LAT 49.318°N, LONG 119.638°W) in the South Okanagan region of British Columbia, Canada. As suggested above, research at this site has been taking place since 2015, providing pre-mitigation roadkill data. This location lies near the northern limit of the three target species' respective ranges (Fig. 1, Winton et al. 2018), and the annual activity period of the snakes here generally extends from April – October of each year. The basin consists of open shrub-steppe grassland habitat surrounded by rolling hills and steep bluffs, with two golf courses and small residential communities and ponds lying just outside the basin. Two roads – one running largely East-West (White Lake Road) and one running North-South (Willowbrook Road) – meet at a T-intersection (Figure 3.1).

In September 2017, 4 ecopassages were installed throughout the study area at locations where snake roadkill occurrences were the highest in 2015-2016 (Winton unpubl.). In April 2019 drift fencing was added to 3 of these new ecopassages and 4 existing drainage culverts to funnel snakes into the openings (Figure 1.3). In total, eight ecopassages were created with the specific intent of lowering rattlesnake road mortality within a 6.5 km stretch of road (Figure 3.1). Some of the ecopassages are in close proximity to known rattlesnake dens while others lie further away.

Ecopassages were made of corrugated metal, with an average height of 45 ± 8 SD cm and an average width of 67 ± 12 SD cm, with an average openness of 0.025 ± 0.005 SD m (= width x height / length; Clevenger and Waltho 2001). The ecopassages that were fully new installations (n=4) were more oval-shaped than the rounder drainage culverts that were retained and refurbished (Figure 1.4. A & B). A substrate of sand (≈ 5 cm deep) lined the bottom of all eight ecopassages.

Drift fencing is recommended to increase ecopassage use by reptiles (Fahrig and Rytwinski 2009, Forman et al. 2003), so in April 2019 drift fencing was added to the entrances of the ecopassages to putatively increase snake usage (Figures 1.3 and 2.1). Bedrock, drainage patterns (including snowmelt), potential danger to cyclists, and cattle use affected a controlled, uniform experimental design for the drift fences. As a result, 7 drift fences were established, with only one fence being situated at a single entrance to each of 7 ecopassages. Drift fencing was not installed at one ecopassage because the surrounding terrain was deemed too steep and rocky. The drift fencing was made of black, 2mm-thick recycled High Density Polyethylene (model AMX-SP40, Animex Wildlife Fencing Solutions), and the top of the drift fences were folded over away from the road to create a lip, making it more difficult for snakes to climb over (See Figure 1.4. B, C & D). Fences were ~75cm in height, although this varied (± 5 cm) within and between fences, depending on the depth of the substrate around the base of the fence section. When constructed, the fence lengths flanking the entrances of the ecopassages had an average length of 52 ± 8.3 m, representing 0.4 km out of 23.4 km of road edge within the study site (Fig. 1, Winton et al. 2018).

Camera Monitoring

I used Bushnell HD Natureview cameras (model numbers 119439, 119440, and 119740) equipped with 46 cm focal lenses to monitor snake use of the ecopassages. The cameras were located immediately inside the entrances of all eight ecopassages, resulting in a total of 16 deployed cameras. Each camera was mounted facing inwards using a lock box secured to either the top or side of the ecopassage with a large bolt screw (See Figure 1.4. A & B). Two trigger methods were used to secure pictures inside the culverts: the cameras were set to high sensitivity and took 3 consecutive photos (each 1 second apart) when activated by movement, as well as timed captures each hour on the hour. The cameras were active from early April to early October of 2019 and 2020. Every two weeks the Secure Digital (SD) cards were switched out and the batteries replaced if necessary. Photos were manually processed to identify the presence of snakes. Data recorded for each animal captured on camera included date, time, species, culvert entrance ID, field scan/ motion sensor activation, direction of travel, and whether the appearance resulted in a passage (see below).

An ***appearance*** was recorded any time a snake was detected in an ecopassage. In some instances, an individual was captured on camera multiple times within one appearance event. For example, if a snake remained coiled in the entrance of an ecopassage for multiple hours, it was only considered one appearance despite being photographed multiple times throughout its stay. If an image of a snake was linked to a ***passage*** (e.g. photographed twice, once at each end), then that snake was only considered as one appearance. (e.g. total detections – passages = appearances).

Photo quality generally made it impossible to be 100% confident that the same individual was entering and exiting an ecopassage. However, the infrequent passage of snakes (especially within species) allowed me to designate a passage as having occurred when a snake of the same species and appearance (size, skin patterns, etc.) was photographed entering and exiting opposite ends of an ecopassage within a 30-minute time interval.

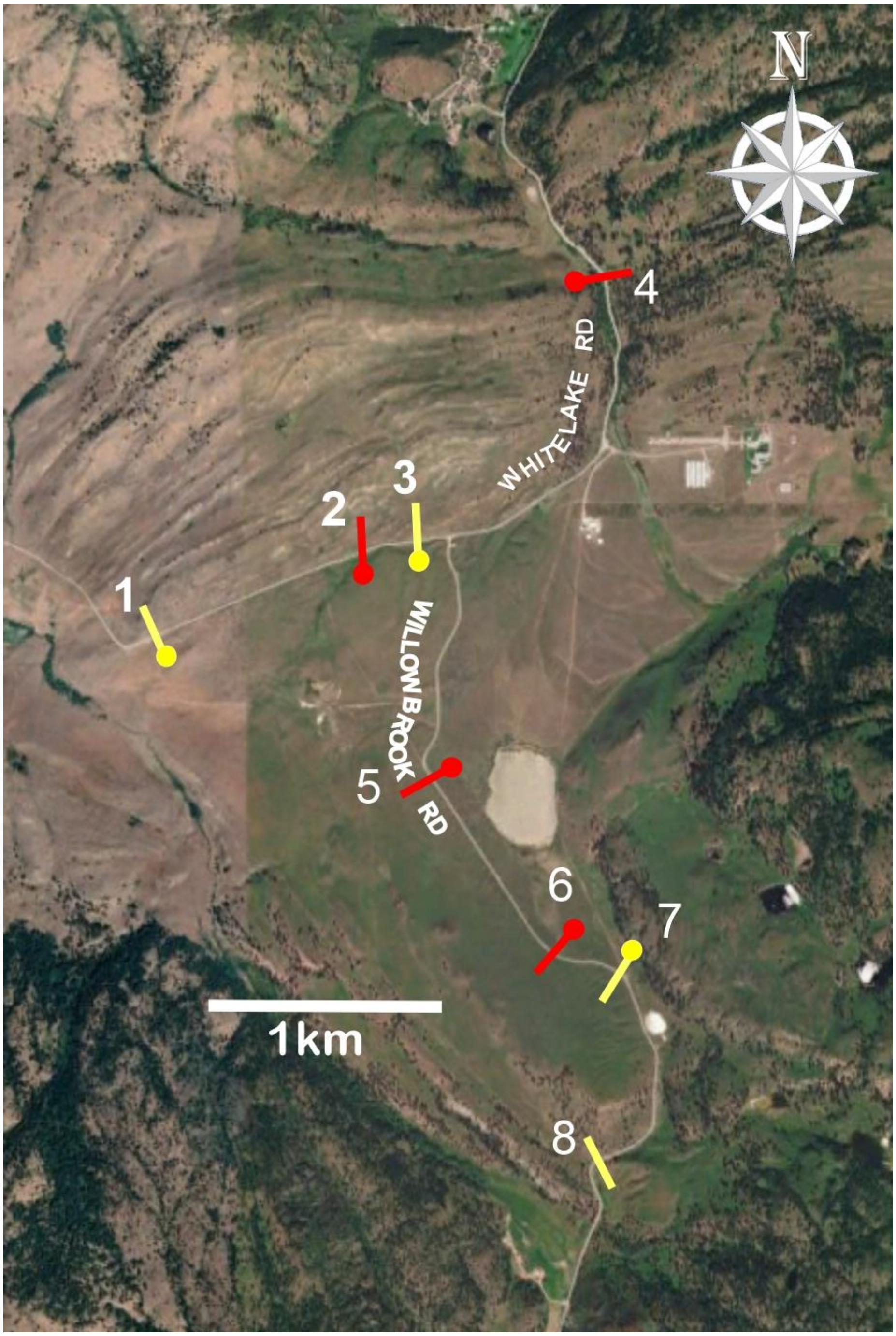

Figure 3.1. Map of camera-monitored ecopassage locations on White Lake and Willowbrook Roads, in the White Lake Basin, British Columbia, Canada. Ecopassages labelled in red are existing, modified drainage culverts, and ecopassages labelled in yellow were newly-created in September 2017 at snake roadkill hotspots. Circles indicate the side of the road that drift fencing was on. Lines representing ecopassages are not drawn to scale. (Base map from Google Earth)

An ***entrance*** was recorded when a snake was photographed moving inwards from an ecopassage opening. This metric was necessary to assess the efficacy of the drift fences. Entrances from a fenced end of an ecopassage indicate that a fence may have aided in directing a snake towards an opening, whereas exits from a fenced end would not.

Data Analysis

Paired T-tests were used to compare entrances of snakes through an ecopassage opening with a drift fence versus the opposing opening lacking a drift fence, with the ecopassage being the basis of the pairing. I did 6 tests in total, quantifying differences within species (rattlesnakes, racers, gophersnakes) for both years of the study (2019 and 2020).

I used a χ^2 test to determine if ecopassage preference differed between species by comparing the proportion of appearances of each species in each ecopassage.

RESULTS

In each of my two study years, I recorded over 1000 appearances of racers, whereas gophersnakes and rattlesnakes provided less than 110 appearances each (Table 3.1). Passages for all species were far less common, but I detected over 300 racer passages in both years while gophersnakes and rattlesnakes produced less than 25 in either year. However, in 2020 all three species increased in appearances and passages (Table 3.1). Overall, gophersnakes showed slightly more ecopassage use (appearances and passages) than rattlesnakes, and racers showed substantially more use than either of the other two species. Of the two metrics, I opted to use appearance data for the χ^2 test comparing ecopassage preference since there was significantly less passage data for each species (Table 3.1).

The three species differed in their ecopassage use both temporally and spatially. There was an obvious lack of any temporal patterns both among species and between years, apart from rattlesnake use peaking mid-season in both 2019 and 2020 (Figure 3.2). The relative use of the different ecopassages was strikingly different across species (Figure 3.3 -

$\chi^2 = 497.3$, df = 14, $P < 0.001$), with rattlesnakes, gophersnakes, and racers most frequently being detected in ecopassages 1, 8, and 4, respectively.

The only instance in which the frequency of entrances through ecopassage ends with drift fences and non-fenced ends differed significantly was with rattlesnakes in 2020, in which they entered through fenced ends more often (Table 3.2). However, the percent of entrances through fenced ends increased for all three species from 2019 to 2020 (Table 3.2). Both rattlesnakes and gophersnakes entered a new ecopassage most frequently, whereas for racers the four ecopassages with the most entrances all were the modified ones (Fig 3.4).

DISCUSSION

As predicted, there was substantial variation among the species in their use of the ecopassages quantitatively, temporally, and spatially. Nonetheless, snakes of all three species were found to be transiting the ecopassages within the two years immediately after their installation. This is an early sign of the ecopassage effects, particularly if one assumes each time a snake uses an ecopassage it avoids a road surface crossing. Of interest is the fact that rattlesnakes used the ecopassages the least of the three at-risk snake species, while racers used them considerably more often than the other two snake species. I cannot conclude how these numbers relate to the relative abundance of the three species: although population sizes and densities are available for the rattlesnakes in this area (owing largely to their denning behaviour), comparable data for gophersnakes and racers are much more difficult to obtain. However, the overwhelming disproportionate use by racers relative to the other two species suggest that these species are showing markedly different responses to the ecopassages.

Despite major variation in temporal use of the ecopassages throughout the active seasons, one interesting commonality is that in both 2019 and 2020 the peak of rattlesnake ecopassage use was around the midway point of the summer. In British Columbia rattlesnakes migrate away from their dens in the spring (April-June) and back in the fall (September-October) (Macartney 1985, Maida et al. 2020), so those times would be when

Table 3.1. Ecopassage camera appearances and passages of three ‘at-risk’ snake species in 8 ecopassages in the White Lake Basin, British Columbia, Canada. Percentages show how many appearances resulted in passages (Passages/Appearances*100). By way of example, in 2019 there were 53 appearances recorded for rattlesnakes, and of those, only 3 (or 6%) resulted in a documented passage. See text for working definitions of ‘appearances’ and ‘passages’.

	Appearances		Passages			
Year	2019	2020	2019	%	2020	%
C. oreganus	53	62	3	6	9	15
P. c. deserticola	74	109	7	9	22	20
C. c. mormon	1219	1842	346	28	402	22

Table 3.2. Ecopassage entrances of three ‘at-risk’ snake species detected by wildlife cameras in 7 ecopassages in the White Lake Basin, British Columbia, Canada. One end of each ecopassage is flanked by drift fencing, whereas the opposite entrance is not. Paired T-tests were used to compare the number of fenced vs. non-fenced entrances for each species in each of the two years of the study, and the resulting *P* values are displayed here, with an asterisk (*) denoting a significant difference.

	Entrances through fenced ends		Entrances through non-fenced ends		% Entrances through fenced ends		*P*	
Year	2019	2020	2019	2020	2019	2020	2019	2020
C. oreganus	12	22	10	12	55	65	0.680	**0.025***
P. c. deserticola	17	31	22	19	44	62	0.140	0.439
C. c. mormon	295	488	422	482	41	50	0.196	0.977

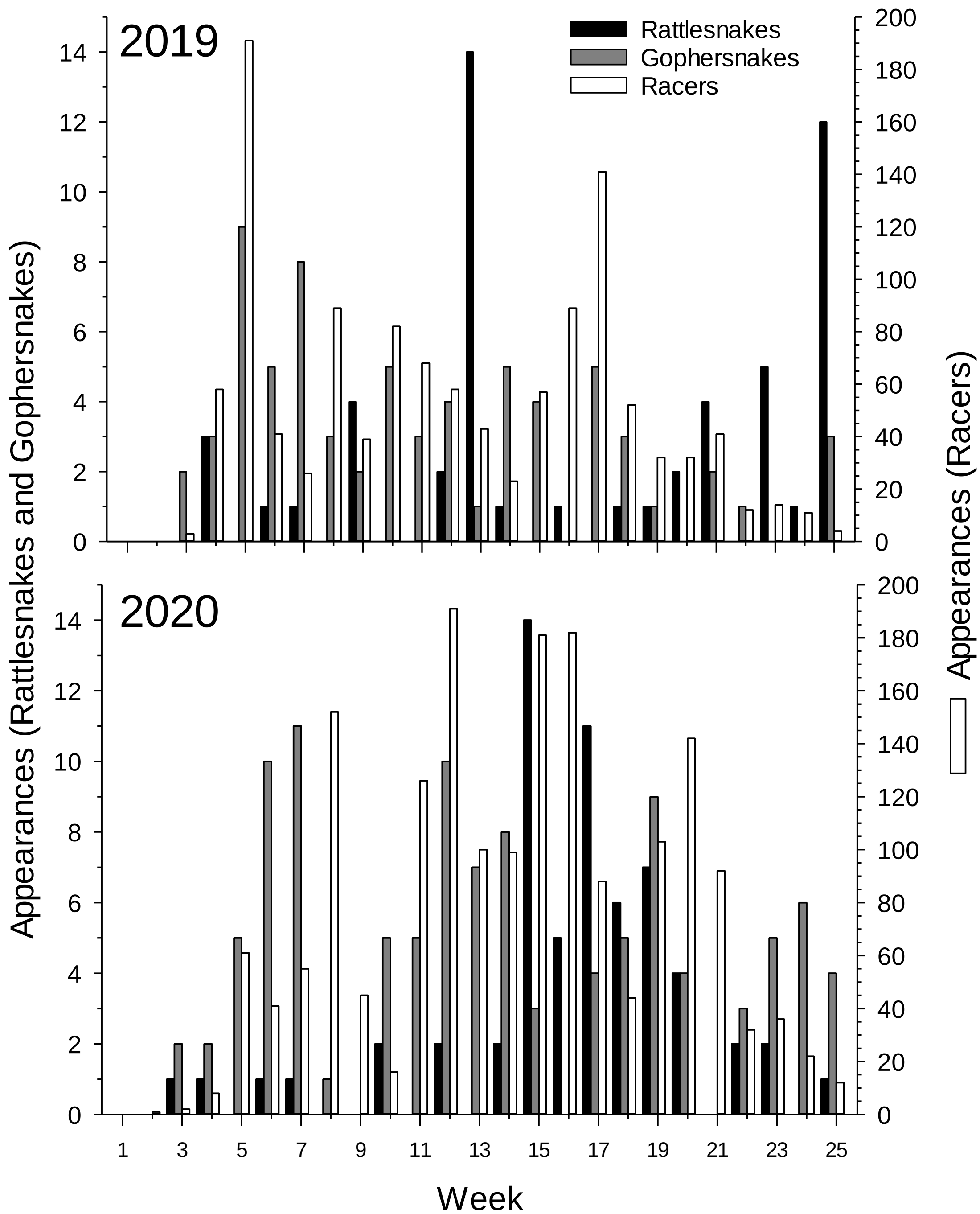

Figure 3.2. Weekly ecopassage (n=8 ecopassages) camera appearances of Western Rattlesnakes, Great Basin Gophersnakes, and Western Yellow-bellied Racers from April 15 to October 6, 2019 and 2020 in the White Lake Basin of British Columbia, Canada.

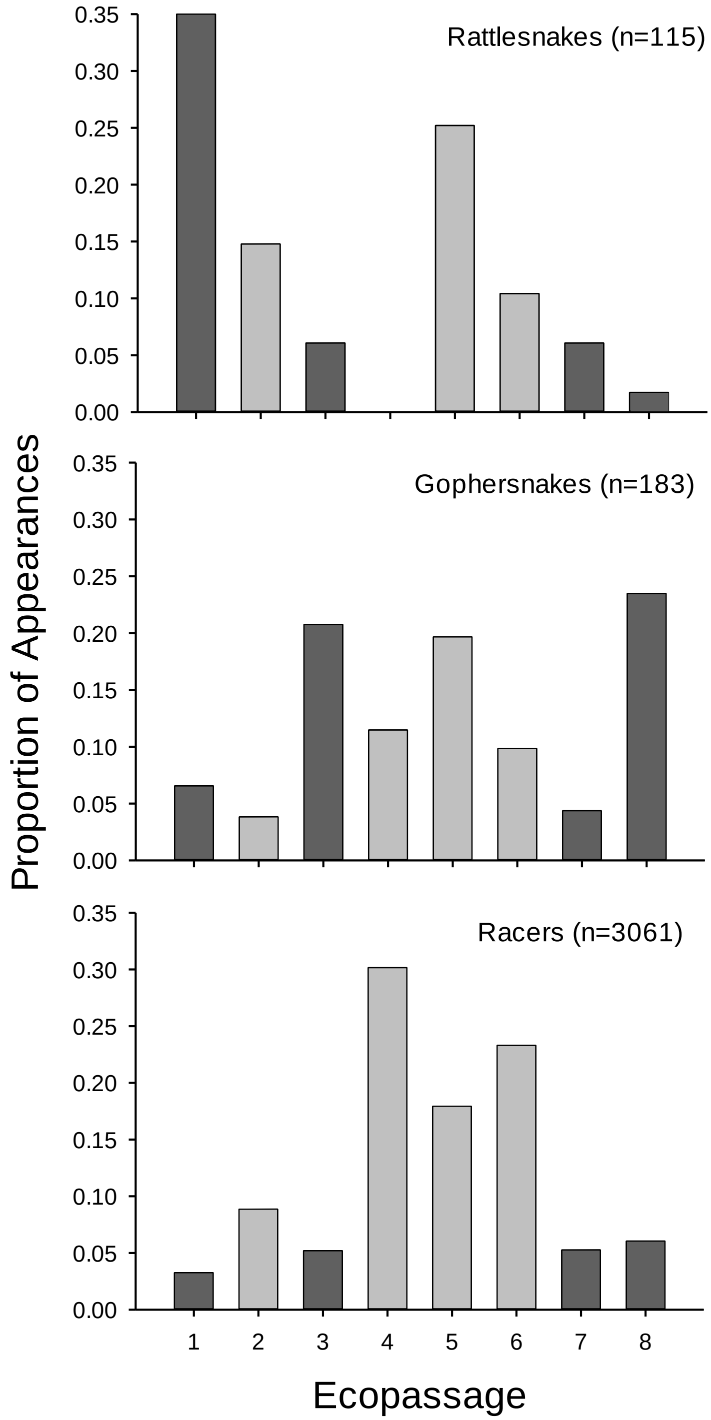

Figure 3.3. Proportion of appearances of Western Rattlesnakes, Great Basin Gophersnakes, and Western Yellow-bellied Racers in 8 ecopassages in the White Lake Basin, British Columbia, Canada in 2019 and 2020 combined. Dark-coloured bars represent newly installed ecopassages, and light-coloured bars represent pre-existing culverts that were modified with drift fencing. See Figure 3.1 for ecopassage locations.

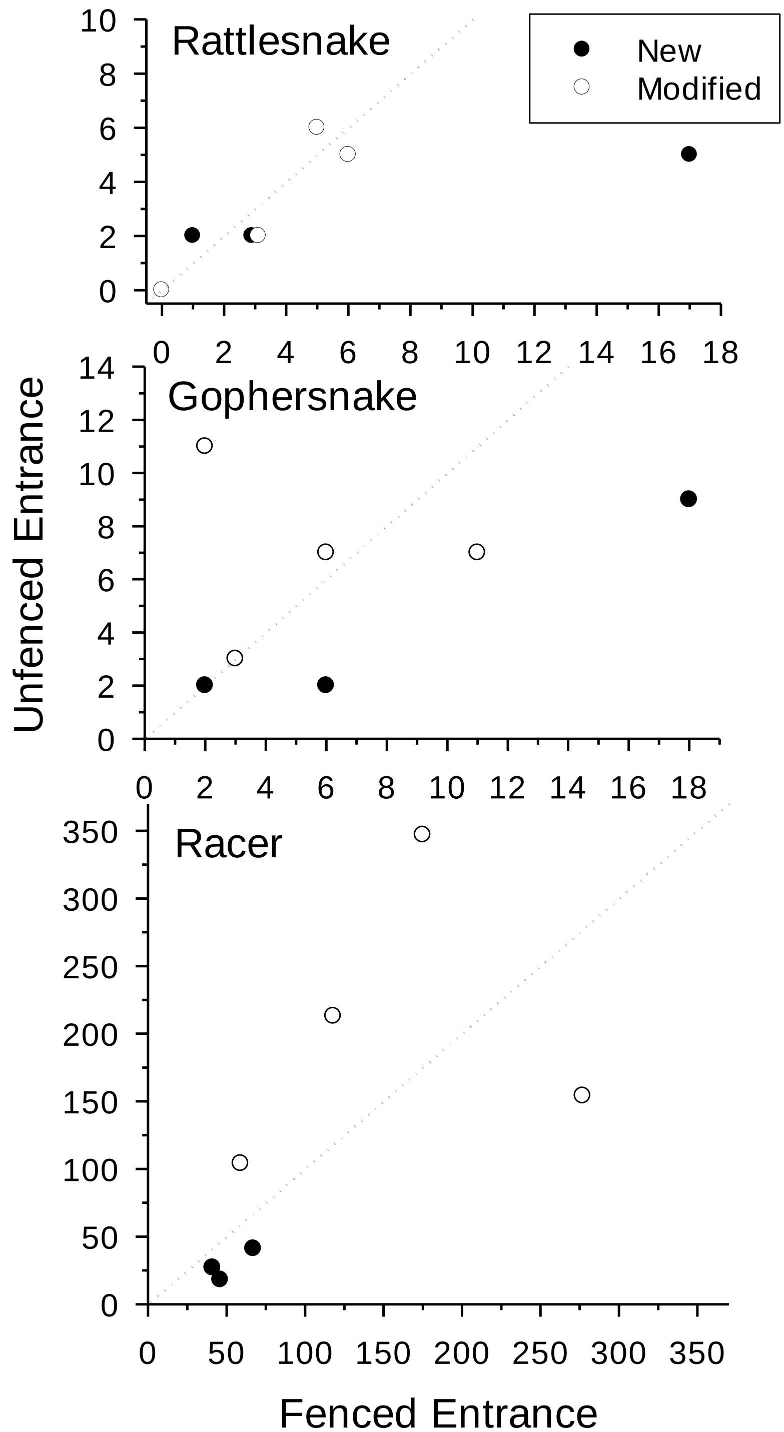

Figure 3.4. Snake entrances through drift fenced versus unfenced ends of seven ecopassages in 2019 and 2020 combined. The dotted line represents hypothetical equal entrances from both ends. Closed circles represent newly installed ecopassages, and open circles represent pre-existing culverts that were modified with drift fencing.

snakes are most likely to be detected in association with ecopassages and/or roads. However, the ecopassage camera data did not demonstrate strong shoulder-season spikes in activity. Perhaps rattlesnakes are attracted to the road for thermoregulatory reasons or continue moving mid-season to find mates or food (Harvey and Larsen 2020). Conversely, gophersnakes and racers appeared to lack consistency in temporal usage patterns between years. Since the denning and egg-laying locations of racers and gophersnakes are more variable than that of rattlesnakes (B.C. Ministry of Water, Land and Air Protection 2004a, 2004b and 2004c, McKelvey *in prep*) individuals of these species may spend more of the active season near the ecopassages, and therefore encounter them more frequently and sporadically.

The spatial dissimilarities in ecopassage use among these species likely reflect, to some degree, differences in habitat selection, although without more detailed study this statement is largely conjecture. Still, the ecopassage with the most rattlesnake detections lied relatively close to two rattlesnake communal hibernacula on a roadkill hotspot corresponding to a migratory corridor. Conversely, the ecopassages used most often by racers were in close proximity to a pond offering, relatively moist habitat in a near-desert ecosystem. Racers are highly insectivorous and forage in areas of abundant herbaceous vegetation, like riparian habitat (Fleet et al. 2009). Ideally, the number and placement of ecopassages would stratify habitat believed important to a multitude of species, but admittedly this may be a lofty and expensive task.

The increase in detection rates of snakes in ecopassages from 2019 to 2020 is consistent with the notion that snakes are particularly responsive to olfactory cues (see below), allowing them to build up familiarity with new passageways (Brown and MacLean 1983, Costanzo 1989, Muellman et al. 2018). It is also consistent with a reduction in rattlesnake roadkill rates in the area from 2019 to 2020 (Chapter 2). However, without longer term data, it would be premature to conclude this pattern was at work.

Similarly, snake entrances through fenced ends of the ecopassages also increased from 2019 to 2020. The year of drift fence installation (2019) showed a muted immediate response by the snakes in terms of fenced versus unfenced entrances to culverts. However, the following year (2020), the percentage of entrances through fenced ecopassage entrances increased for all three species, and the number of rattlesnakes entering fenced ends was significantly higher than the number at non-fenced ends. Previous research at other locations, on other species, suggests that road mortality would be reduced by expanding the fencing (Baxter-Gilbert et al. 2015, Colley et al. 2017, Markle et al. 2017, Plante et al. 2019). Although installation constraints impacted my research design, in one year of this study rattlesnake ecopassage use was indeed higher according to the presence of fencing. Currently the fences surrounding the culvert entrances provide relatively short, inconsistent barriers to snake movement within the study area. Intuitively, their overall impact on snake road mortality would be enhanced if the fences were expanded.

Olfactory cues and conspecific scent trailing are important for snakes, particularly communally aggregating snakes like rattlesnakes (Brown and MacLean 1983, Muellman et al. 2018). Thus, ecopassage appearances and passages likely will continue to increase as scent trails develop. The impact of increased familiarity/scenting may be linked to the relatively higher entrance counts for pre-existing, modified ecopassages than the newly installed ones, most notably for racers. Possibly racers rely more heavily on scent trails, or their movement patterns result in repeated trips more often in the same area (Wong *in prep*) In the case of rattlesnakes and gophersnakes, there were obvious outlying ecopassages (both newly-installed) that experienced more entrances both in general and through fenced ends. This may be because these were ideal locations for the ecology of that species. The ecopassage experiencing the most rattlesnake entrances has a known den on either side of the road, so in this case the number of snakes nearby and/or the ideal location of the ecopassage likely outweighs the unfamiliarity of it.

Although the timeframe encompassed in this study is relatively short, this initial period following the installation of mitigation measures provides an important snapshot of the immediate response of snakes to these structures. In Chapter 2, I likened this period to that described by Warren et al. (1987) and Whelan et al. (2002) for the recovery of ecosystems and species immediately after a wildfire (the so-called ‘shock phase’). Conducting assessments of mitigation effects through this time period will prove valuable for understanding long-term patterns (which also should be monitored), including adding to our knowledge of how quickly target species respond to new structures of this type. In this study, I have shown clearly that even three closely-related species respond differently to the same crossing structures, although they all used ecopassages to cross roads, presumably in lieu of crossing on the road's surface. Differences in crossing structure use of similar taxa has been reported elsewhere, for example mule deer more readily use underpasses, and in general use crossing structures twice as often as sympatric pronghorn, which prefer overpasses (Sawyer et al. 2016). In this case, snake crossing varied by frequency, time, and location among species. To accommodate this variation and direct more efficient use of conservation funds, ecopassages intended for multi-taxa use may be installed (Polak et al. 2019, Santini et al. 2016). Photos from the cameras at this study site have shown at least 31 other species using the ecopassages – many being of conservation concern in British Columbia - including the American Badger (*Taxidea taxus jeffersonii*) and Western Tiger Salamander (*Ambystoma mavortium*) (Matson 2021, Appendix D). Both of these species often fall victim to roadkill (Crosby 2014, Sunga et al. 2017). Facilitating the use of ecopassages by a wide range of species should also be considered when designing fencing along the entrances (Aresco 2005, Baxter-Gilbert et al. 2015, Cunnington et al. 2014).

It is important to recognize that although all three snake species in this study used the ecopassages to cross roads, this metric alone cannot be used to conclude the mitigation efforts are having a significant effect. Ideally work of this nature needs to be combined with demographic and roadkill monitoring (as done for rattlesnakes in Chapter 2). In particular, the short timeframe of this study, and lack of population data for racers and gophersnakes, makes it premature to conclude on the merits of the mitigation work. But, in the absence of noticeable negative effects, the Precautionary Principle and the continued application of these techniques appears warranted.

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CHAPTER 4

CONCLUSION

The overarching goal of my thesis was to understand the immediate impacts of road mortality mitigation actions on snakes. Specifically, I assessed the numerical and behavioural (functional) responses of three at-risk grassland snake species in the years immediately following the installation of ecopassages and drift fencing in the South Okanagan Valley of British Columbia. The three focal species were Western Rattlesnakes (*Crotalus oreganus*), Great Basin Gophersnakes (*Pituophis catenifer deserticola*), and Western Yellow-bellied Racers (*Coluber constrictor mormon*), with a primary focus on rattlesnakes due to long-standing conservation concerns and the relative ease of their population monitoring. I conducted this research by: (a) comparing rattlesnake roadkill and population estimates pre- and post-mitigation, (b) quantifying and comparing ecopassage use of three snake species, and (c) assessing the efficacy of drift fences flanking the ecopassages.

The main findings of my thesis were:

- Rattlesnake roadkill rates have a decreasing trend over the years from 2015 to 2020, despite traffic rates increasing.
- There were no obvious population or survivorship trends for Western Rattlesnakes throughout the course of the study.
- In general, the post-mitigation years had less rattlesnake road mortality and a larger population size than the pre-mitigation years.
- Gophersnakes showed 1.6x and racers 26x the number of appearances of rattlesnakes in the ecopassages. Gophersnakes completed 2.4x and racers 62x the number of passages through ecopassages that rattlesnakes did.
- The timing of ecopassage use throughout the active season lacked consistency among species, and within species between years.
- The use of different ecopassages was not consistent across species, and I postulate this was based on environmental factors.
- Rattlesnakes entered fenced ends of ecopassages more than unfenced ends only in the last year of the study.

- Measuring the usage of ecopassages through camera data is a complimentary and valuable tool to augment the monitoring of demographic trends in the target population to assess their response.
- Taken together, the three main components of this study (road mortality, population/survivorship, and camera data) work together to provide a well-rounded assessment of the of the response of snakes to mitigation measures, and the subsequent conservation implications.

Overall, these findings demonstrate that functional responses to newly-installed mitigation structures can be detected sooner than numerical responses, especially for animals with relatively long life spans. This study also highlights the importance of (i) monitoring the response to mitigation efforts (such as ecopassages) immediately after their introduction, but also (ii) developing long-term data sets that continue well past the 'shock phase'. The results of my study depict stark variation in ecopassage use between species, which may be important to consider when developing species-specific conservation strategies.

Management Implications

Initial management recommendations for the snakes in the study region were outlined by Winton (2018) following her earlier work on the road ecology of rattlesnakes. She suggested that fences paired with underpasses be installed at high priority locations to reduce roadkill while maintaining habitat connectivity. She also emphasized the importance of continued monitoring of road mortality and the rattlesnake population in order to detect any changes that result from these mitigation structures. Fortunately, these recommendations came to fruition and drove the research presented in this thesis. However, there are some recommendations by Winton (2018) that have not been implemented, and I believe these bear repeating. Firstly, a reduced speed limit and traffic signs in the basin (as suggested by Farmer and Brooks 2012, Valero et al. 2015) would slow traffic, raise awareness about snakes' presence in the area, and draw attention to the impacts of road mortality. Fortunately, ongoing research in the area (Wong *in prep*) is assessing the comparative movement patterns of rattlesnakes, gophersnakes, and racers to better understand their susceptibility to road mortality and affinity to crossing structures. Conducting population-level studies (Chapter 2) to assess mitigation efforts are more difficult for species that do not easily lend themselves to population monitoring (i.e. racers and gophersnakes versus rattlesnakes). Examining movement patterns provides a metric that allows more direct comparisons between the three species, however it is not a reliable indicator of the conservation efficacy of the mitigation structures. Additionally, similar research should be done in other areas of concern throughout the province.

Based on my own research (particularly that described in Chapter 3), I recommend expanding the length of the fences to better direct snakes into the ecopassages. Partial fencing has proven to be ineffective, and compromises the success of connectivity structures (Baxter-Gilbert et al. 2015, Markle et al. 2017). It also results in an issue known as the "fence-end effect", where animals are found on the road around fence ends (Harman et al. 2023, Markle et al. 2017). Unfortunately the slope and/or substate of the areas surrounding the ecopassages at my study site prevented fences from being built on both sides of the ecopassages. Although extending fences at this site still would result in partial fencing, there are initial signs of fence success. Therefore, this is a situation where the Precautionary Principle should be invoked. I also recommend that fences are built with long-term structural integrity in mind, because hiring personnel to keep up with long-term maintenance will be expensive and logistically difficult, and compromised fencing has been shown to be ineffective (Baxter-Gilbert et al. 2015).

In my work, the extreme discrepancy between ecopassage use across the three species makes it clear that snakes, despite taxonomic and morphological similarity, should not be expected to respond in similar fashion to mitigation structures. The type of ecopassages installed were designed for general snake use (BC Ministry of Environment and Climate Change Strategy 2020, Ontario Ministry of Natural Resources and Forestry 2016), and therefore provide an ideal structure to use in this study area and other locations where snake road mortality is of concern. Although the declining population of rattlesnakes at this site (Winton et al. 2020) was the impetus for the installation of mitigation structures, I have shown that multiple snake species now use the ecopassages – some more so than rattlesnakes. These snake-targeted ecopassages also happened to accommodate a variety of other taxa (Jaccard 2024, see Appendix D), which has obvious advantages.

Future Research Considerations

Future research should not only focus heavily on mark-recapture of racers and gophersnakes, but attempt to use a different capture method, like traps, to increase evenness in catchability among the sympatric species. This study was principally framed around rattlesnake conservation, and as a result population estimates for racers and gophersnakes were unavailable, given a shortage of mark-recapture data for these two species in my study area. This is largely due to the fact that our mark-recapture survey efforts were focused on communal rattlesnake hibernacula, and even when present in rattlesnake hibernacula, it appears they spend comparatively little time there in spring, making capture and enumeration very difficult (McKelvey *in prep*). Therefore, it is not possible to determine if the extreme discrepancy in ecopassage use between species is a result of a corresponding variation in population size. Since my fieldwork, there have been stronger mark-recapture efforts for these species in the basin, however the catchability of these snakes (particularly racers) remains a major obstacle to assessing the status of these animals. Similarly, wildlife camera detection probablity can vary interspecifically (Caravaggi et al. 2020). Experiments could be done testing snakes of each species upon ecopassage transit, and identifying the proportion of times that the cameras are successfully triggered. This would enable the calculation of correction factors that could be applied to camera capture rates for each species. Finally, differences in movement and behaviour patterns could be affecting how often each species encounters ecopassages or fences. As mentioned earlier, a comparison of the habitat use and movements of rattlesnakes, gophersnakes, and racers is ongoing, in an attempt to relate the results to the frequency of their ecopassage use (Wong *in prep*). This research also continues to monitor rattlesnake roadkill rates and population size, which will add substantially to the work in this thesis.

Conclusion

My research considered both numerical and behavioural (functional) factors to assess the immediate response of Western Rattlesnakes to ecopassages and drift fencing, along with the behavioural response of Great Basin Gophersnakes and Western Yellow-bellied Racers. Although the short timeframe of the ‘shock phase’ following ecopassage installation prevents my ability to draw strong conclusions from the data, the rattlesnake population estimates gathered at this time still provide an excellent basis for continued monitoring to generate longer-term post-mitigation results. Nevertheless, the post-mitigation rattlesnake roadkill rates are promising, as road mortality appears to be trending downwards despite an increase in traffic. The behavioural response of the snakes is desirable, as ecopassage use was immediate and appears to be increasing. Therefore, it is possible that we could see a clear and positive numerical response in the coming years.

The understanding of the numerical and behavioural responses of a species to alterations to their environment, both in the short- and long-term, will aid conservation practitioners when developing recovery strategies for wildlife populations. The results highlight the importance of long-term monitoring pre- and post- mitigation, and the extreme variation in responses that can occur across species of the same taxa. I achieved my research objectives of comparing Western Rattlesnake pre-mitigation roadkill and population estimates to those in the years immediately following the installation of ecopassages and fencing, and quantifying and comparing the use of ecopassages by Western Rattlesnakes, Great Basin Gophersnakes, and Western Yellow-bellied Racers. Based on my results, I make the following recommendations:

- Where funding permits, the Precautionary Principle should be applied and ecopassages and drift fencing should be installed to reduce roadkill of snakes and other small taxa.
- Since this site now has the unique and valuable history of intensive monitoring data pre- and post-mitigation, it should remain a focal research site for snakes in the Southern Interior of BC. Monitoring of snake roadkill and rattlesnake population size should be continued long-term.

- Analyzing camera data is labour intensive, and given my study shows quite clearly that snakes are using the ecopassages, the value of ongoing camera monitoring is dubious, particularly if it deflects money and energy away from other investigations. Software for analyzing wildlife photos is currently not effective for distinguishing snakes (K.W. Larsen, pers. obs.). For these reasons, I would recommend ceasing camera monitoring in the ecopassages in order to redirect resources to other areas of the project, or at least move away from a yearly monitoring scheme.
- Monitor Great Basin Gophersnake and Western Yellow-bellied Racer populations in a way that can be compared among species, accounting for differences in catchability.
- Monitor movement patterns of all three species to gain insight into the differences in their ecopassage use.

In closing, I have met the objectives of my thesis, and I trust that the findings of my research will contribute to the conservation and persistence of the snake populations in the White Lake Basin, and to wildlife conservation in general.

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APPENDIX A

JOLLY-SEBER ESTIMATES FOR RATTLESNAKES IN THE WHITE LAKE BASIN

Table A.1. Jolly-Seber population estimates for Western Rattlesnakes (*Crotalus oreganus*) in the White Lake Basin, BC (2015-2020; estimates not available for 2015 or 2021 based on method of calculation).

		All Captures		Focal Dens	
	Year	Population Size	SE	Population Size	SE
All age classes	2015	NA	NA	NA	NA
	2016	644	70	526	54
	2017	730	86	497	53
	2018	1038	128	876	107
	2019	1365	152	900	96
	2020	834	90	666	77
	2021	NA	NA	NA	NA
	MEAN	922	105	693	77
Adults only	2015	NA	NA	NA	NA
	2016	382	39	325	31
	2017	393	41	272	25
	2018	475	55	335	35
	2019	611	72	405	47
	2020	386	40	283	31
	2021	NA	NA	NA	NA
	MEAN	449	49	324	34

Table A.2. Jolly-Seber survivorship estimates for Western Rattlesnakes (*Crotalus oreganus*) in the White Lake Basin, BC (2015–2021; estimates not available for 2020 or 2021 based on method of calculation).

		All Captures		Focal Dens	
	Year	Survival Estimate	SE	Survival Estimate	SE
All age classes	2015 → 2016	0.72	0.07	0.71	0.06
	2016 → 2017	0.64	0.07	0.62	0.06
	2017 → 2018	0.73	0.08	0.83	0.09
	2018 → 2019	0.75	0.08	0.71	0.07
	2019 → 2020	NA	NA	NA	NA
	2020 → 2021	NA	NA	NA	NA
	MEAN	0.71	0.08	0.72	0.07
Adults only	2015 → 2016	0.72	0.06	0.71	0.06
	2016 → 2017	0.66	0.06	0.65	0.06
	2017 → 2018	0.73	0.07	0.81	0.08
	2018 → 2019	0.93	0.10	0.95	0.10
	2019 → 2020	NA	NA	NA	NA
	2020 → 2021	NA	NA	NA	NA
	MEAN	0.76	0.07	0.78	0.08

APPENDIX B

SNAKE ROAD MORTALITY IN THE WHITE LAKE BASIN

Table B.1. Frequency of roadkill detected for at-risk snake species during surveys and incidentally along 11.7 kms of road within the White Lake Basin, BC, Canada.

Year	CROR	PICA	COCO
2015	36	32	49
2016	56	52	79
2017	28	47	77
2018	29	60	87
2019	35	56	38
2020	21	56	47

Table B.2. Detections of dead Western Rattlesnakes (*Crotalus oreganus*) and densities per km surveyed (unadjusted for observer error or scavenging rates) by road survey method in the White Lake Basin, BC, Canada, 2015-2018.

Survey method	Time of day	Total km surveyed						Number of dead rattlesnakes detected						Unadjusted dead rattlesnake density (dead rattlesnakes/km)					
		2015	2016	2017	2018	2019	2020	2015	2016	2017	2018	2019	2020	2015	2016	2017	2018	2019	2020
Walk	Day	116.2	253.5	485.6	274.8	386.1	538.2	6	13	8	10	20	16	0.052	0.051	0.016	0.036	0.052	0.030
Drive	Day	62.4	51.6	157.9	81.1	210.6	35.1	3	4	2	5	7	1	0.048	0.078	0.013	0.062	0.033	0.028
Drive	Night	233.3	292.5	222.3	81.9	0	0	12	9	2	5	-	-	0.051	0.031	0.009	0.061	-	-
Bike	Day	53.1	129.9	0	64.5	23.4	0	1	8	-	1	1	-	0.019	0.062	-	0.016	0.085	-
Surveys total		465	727.5	865.8	502.3	620.1	573.3	22	34	12	21	28	17	0.047	0.049	0.014	0.042	0.045	0.030
Incidental		1633.8	2382.4	-	-	-	-	14	22	16	8	7	4	0.009	0.009	-	-	-	-
Surveys + Incidental		2098.8	3109.9	-	-	-	-	36	56	28	29	35	21	0.017	0.018	-	-	-	-

Table B.3. Calculated road mortality rates (deaths/km/day) and number of Western Rattlesnake (*Crotalus oreganus*) deaths based on walking survey results and accounting for scavenger-removal and observer error during the active season (April-October) along the 11.7 km survey route in the White Lake Basin, BC, Canada, 2015-2018.

				After culvert installation	After fencing installation	
	2015	2016	2017	2018	2019	2020
Mean mortality rate (±SE) (rattlesnake deaths/km/day)	0.044 (0.019)	0.070 (0.030)	0.043 (0.019)	0.021 (0.006)	0.039 (0.010)	0.025 (0.007)
Active season length (days)	176	188	183	182 *	182	181
Calculated rattlesnake deaths per year (11.7 km survey route)	91	154	93	45	83	53
Number of dead rattlesnakes detected	36	56	28	29	35	21
Correction factor	2.5	2.8	3.3	1.6	2.4	2.5

*Average of 2015-2017 due to late spring sampling in 2018

Table B.4. Adjusted dead Western Rattlesnake (*Crotalus oreganus*) road densities (dead rattlesnakes/km) for five types of survey methods, accounting for scavenger-removal and observer error, in the White Lake Basin, BC, Canada, 2015-2018.

Survey method	Time of day	Adjusted dead rattlesnake density (dead rattlesnakes/km)					
		2015	2016	2017	2018	2019	2020
Walk	Day	0.130	0.143	0.054	0.058	0.125	0.075
Drive	Day	0.120	0.218	0.044	0.099	0.079	0.070
Drive	Night	0.128	0.087	0.031	0.098	-	-
Bike	Day	0.048	0.174	-	0.031	0.204	-
Surveys total		0.118	0.137	0.046	0.068	0.108	0.075
Incidental		0.023	0.025	-	-	0.007	0.005
Surveys + Incidental		0.043	0.050	-	-	0.026	0.018

APPENDIX C

SNAKE ECOPASSAGE USE IN THE WHITE LAKE BASIN

Table C.1. The distribution of ecopassage use of Western Yellow-bellied Racers, Great Basin Gophersnakes, and Western Rattlesnakes in the White Lake Basin, British Columbia, Canada, with the proportion of the species total in brackets. An appearance was defined as any camera detection of the species inside the ecopassage and a passage occurred when a snake of the same species entered and exited opposite ends of the ecopassage within 30 minutes.

		Racer		Gophersnake		Rattlesnake	
Year	Ecopassage	Appearance	Passage	Appearance	Passage	Appearance	Passage
2019	1	4 (0.003)	1 (0.003)	0 (0)	0 (0)	19 (0.358)	0 (0)
	2	145 (0.119)	44 (0.127)	3 (0.041)	0 (0)	6 (0.113)	2 (0.667)
	3	22 (0.018)	1 (0.003)	16 (0.216)	0 (0)	1 (0.019)	0 (0)
	4	335 (0.275)	161 (0.465)	8 (0.108)	3 (0.428)	0 (0)	0 (0)
	5	203 (0.167)	2 (0.006)	15 (0.203)	2 (0.286)	15 (0.283)	0 (0)
	6	377 (0.309)	124 (0.358)	13 (0.176)	2 (0.286)	6 (0.113)	0 (0)
	7	60 (0.049)	6 (0.017)	4 (0.054)	0 (0)	4 (0.075)	1 (0.333)
	8	73 (0.060)	7 (0.020)	15 (0.203)	0 (0)	2 (0.038)	0 (0)
2020	1	96 (0.052)	4 (0.010)	12 (0.110)	0 (0)	22 (0.355)	2 (0.222)
	2	126 (0.068)	34 (0.085)	4 (0.037)	2 (0.091)	11 (0.177)	2 (0.222)
	3	137 (0.074)	19 (0.047)	22 (0.202)	2 (0.091)	6 (0.097)	0 (0)
	4	588 (0.319)	157 (0.391)	13 (0.119)	5 (0.227)	0 (0)	0 (0)
	5	346 (0.188)	137 (0.341)	21 (0.192)	6 (0.273)	14 (0.226)	3 (0.333)
	6	336 (0.182)	37 (0.092)	5 (0.046)	1 (0.045)	6 (0.097)	2 (0.222)
	7	101 (0.055)	1 (0.002)	4 (0.037)	0 (0)	3 (0.048)	0 (0)
	8	112 (0.061)	13 (0.032)	28 (0.257)	6 (0.273)	0 (0)	0 (0)

APPENDIX D

ECOPASSAGE USE BY NON-TARGET SPECIES IN THE WHITE LAKE BASIN

Table D.1. Ecopassage camera appearances and passages of non-target species in 8 ecopassages in the White Lake Basin, British Columbia, Canada in 2019 and 2020. (Adopted from Jaccard 2024)

	Appearances		Passages	
	2019	2020	2019	2020
American Badgers	1	0	0	0
Birds	22	99	0	0
Burrowing Owls	1	0	0	0
Bushy-tailed Woodrats	33	1	2	0
Frogs & Toads	22	12	0	0
Muskrats	2	0	0	0
Porcupines	2	0	0	0
Rabbits	72	10	0	0
Salamanders	29	12	4	0
Skunks	3	0	2	0
Weasels	17	29	4	16
Western Painted Turtles	3	0	2	0
Yellow-bellied Marmots	0	6	0	2
Yellow-pine Chipmunks	7	1	0	0
Mice/ Voles	>200	>200	-	-