

*Survival of chestnut-collared longspurs
(*Calcarius ornatus*) and Baird's sparrows
(*Centronyx bairdii*) on the breeding
grounds in southeastern Alberta*

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Abstract

The grasslands of Alberta have suffered extreme destruction and habitat loss for over 150 years, including destruction that continues from energy industry development. Oil wells that are built through this development have an ongoing impact on wildlife from the physical footprint of the infrastructure and the additional footprint of noise emitted from oil extraction. Chronic noise from oil extraction may impede conspecific communication during the critical post-fledging stage, which may result in lower survival of fledglings and could be a source of population declines. To investigate the effects of oil extraction noise on chestnut-collared longspurs adult (*Calcarius ornatus*) and fledgling survival, and adult Baird's sparrow (*Centronyx bairdii*) survival, I used an experimental design that isolates noise recorded from active oil wells. I used handheld, very high frequency radio-telemetry to track tagged individuals daily. Adult survival for both adult chestnut-collared longspurs and Baird's sparrow was close to 100% in 2017 and 2018. I found no effect of oil well infrastructure or noise on post-fledging survival of chestnut-collared longspurs. Older, heavier fledglings had a higher likelihood of survival. These results suggest that population declines are not coming from the adult life stage for these species on the breeding grounds of southeastern Alberta. The post-fledging survival results support previous studies that fledgling weight and age are positively correlated with survival. Ongoing research of the full annual cycle of these two species should be prioritized to inform conservation and management decisions.

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Chapter 1 Introduction

1.1 The perils faced by North American grasslands

North American grassland habitat is one of the most threatened habitats in the world. Grassland species continue to suffer declines because of the numerous ways that habitat has been destroyed, lost, or degraded (Rosenberg et al. 2019). In North America, conversion to agricultural land, and oil and gas exploration and extraction, have had the largest combined impacts on prairie habitat, resulting in both habitat loss and degradation (Roch and Jaeger 2014). Habitat loss results from development activities such as land conversion and roads. Habitat is further degraded from invasive plant species (Olden et al. 2005), habitat fragmentation from roads and fences (e.g. Thompson et al. 2015), and introduced noise from energy development. Land conversion, energy development, and additional disturbances from urbanization have reduced the Great Plains by close to 70% with only 29% of mixed grass prairie remaining (Samson et al. 2004). Habitat loss and degradation in the North American prairies have negatively affected numerous species across many taxonomic groups. Among those, avian species have been heavily affected by changes to grassland habitat, particularly habitat specialist songbird species (Correll et al. 2019). Since 1970, a staggering 300 million birds have disappeared from the grasslands (Rosenberg et al. 2019). Grassland bird populations have likely been suffering population declines since before large-scale surveying efforts began, so declines could be much larger.

The intensity and capacity of research, policy, funding, and management of avian, grassland species has increased as more evidence from long-term monitoring confirms populations are declining. Certain species have fared worse than others; for example, species that are tolerant of large-scale agriculture have experienced declines of 39% while grassland dependent species as a group have experienced cumulative population declines of 87% (North American Bird Conservation Initiative Canada 2019). Research has steadily increased around grassland conservation to investigate which disturbances may be contributing to population declines, and to better understand the mechanisms that explain why these disturbances are putting species at risk. The Species at Risk and Migratory Bird Convention Act are federal legislation that were

established to protect individuals of declining species and their residences. Additionally, federal and provincial recovery strategies for chestnut-collared longspurs (*Calcarius ornatus*) describe recommendations for industry such as applying buffers around active nests (Cerney and Calon 2015; Environment and Climate Change Canada 2018). Although policy and recovery plans help species at risk, these types of conservation effort are reactionary and do not address root causes of decline.

Conversion to agricultural land, energy development, and the physical footprint of well pads and roads lead to direct habitat loss. Relatively recent research has also begun to show the effects of less obvious habitat loss through altered soundscapes (Francis et al. 2009; Ware et al. 2015; Potvin 2017). For example, noise from vehicle traffic has been shown to further degrade habitat by reducing species abundances and foraging time of individuals in proximity to noisy roads (Ware et al. 2015). Currently, there is limited legislation protecting grassland birds from noise partially due to limited evidence demonstrating the impacts of noise. It is critical to understand specific, causal factors of decline in and near anthropogenically, disturbed habitat to reduce or cease the decline of grassland avian populations. More evidence is needed for grassland birds, specifically at-risk species.

Additionally, it is critical to understand the source of these causal factors from both a historical and social context to better facilitate consequence mitigation and conservation management initiatives. Conservation is multifaceted and requires perspectives from sources outside of ecology for meaningful results. Biological conservation often takes place in isolation of other aspects that affect species at risk. Managers and conservationists often look at conservation problems and look at the direct biological mechanisms that affect species. Although this is important in guiding how to protect species at risk and habitat, focusing solely on biological mechanisms does not consider the additional intersections of anthropogenic and natural resource dimensions with regards to conservation issues. To this end, providing context for other researchers and managers allows for broader understanding and insight into conservation dilemmas that many species are currently experiencing. The understanding of anthropogenic dynamics and both current and historical situations can provide managers with the ability to approach conservation issues with more tact and, ideally, more support from the surrounding communities. It is critical for scientists to acknowledge that empirical evidence is

not the sole method for understanding the physical environment. Scientists and conservation managers need to acknowledge the history of a study area. Management solutions need to integrate an understanding of the economic and social context in which they would be implemented. The historical social context is important because it is the real-world conditions where management is applied. Having this context and perspective for the study area will improve understanding of problems that are a result of historical events that occurred 150 years ago. Although mitigation efforts need to confront current policy, some legislation is from decades ago, if not longer. Because of the lasting impacts and influence policy has on the environment, it is critical for managers to understand the history.

1.2 Historical and social context of North American grasslands

Human actions have been significant in shaping ecosystems and therefore, should be acknowledged in biological conservation and management (Szabó and Hédl 2011). Providing biological evidence and mechanisms is required for justifying management decisions; however, the social and historical contexts of management choices are often not included (Charnley et al. 2017). Large scale conservation efforts can often benefit from community support, as shown from experiences in conservation related to marine protected areas (Christie et al. 2017). Social sciences, including the historical context, can provide more thorough reasoning to help facilitate conservation policies, actions and outcomes making them more legitimate, salient, robust and effective (Bennett et al. 2017).

The historical context includes understanding how the current perilous situation in the North American grasslands came to be and requires the incorporation of perspectives from people who currently live in these areas and those that were displaced. Descriptions of the recent, negative consequences that have occurred within the grassland habitat is one area where it is important to distinguish the difference between anthropogenic and settler-colonial activity. Indigenous human activity is not responsible for the destruction the grasslands have experienced and which continues in present day. Furthermore, First Nations management of North American grasslands is generally recognized as playing a key role in promoting regeneration of vegetation and increasing diversity of grasslands (Schuster et al. 2019). Understanding the historical and social context of destruction that has occurred in the prairies would better prepare decision makers and,

ideally, lead to conservation action that has greater support from people outside of the conservation domain. Just as the ecological background is traditionally provided for conservation management, the historical background of the study area should also be presented. It is well known that North American grassland habitat is currently one of the most threatened ecosystems and that the cause of the habitat loss and destruction is from anthropogenic activity (Samson and Knopf 1994). The most destructive changes occurred during the past two centuries and have resulted in widespread habitat loss, habitat degradation and have affected populations of numerous plant and animal species (Samson et al. 2004). Using intentional and accurate terminology when describing the changes that have occurred in the grasslands should be a deliberate part of conservation action and reconciliation. This has been a problem in the context of North American grasslands because research and management articles and reports frequently use the term anthropogenic to describe habitat destruction in this area.

Large-scale changes that have occurred in the prairies are explained by the presence of colonial and settler people (Samson et al. 2004). The federal government of Canada is responsible for the first and largest amount of habitat alterations and disturbance in the prairies including the Dominion Lands Act in 1872, the extirpation of bison (*Bison bison*) around 1879, encouraging the conversion and development of natural prairie, the creation of Prairie Farm Rehabilitation Administration in 1935 to help farmers manage the effects of drought from the decade long “dust bowl” of 1928-1938 (with management efforts including seeding crested wheatgrass to help reduce erosion). The Dominion Lands Act was one of the most damaging acts on the Canadian prairies (Ostlie et al 1997). Historical classifications of grassland habitat demonstrate federal priorities and the relationship that settler-colonial governance has had with this landscape. An example of this is found in definitions used by Statistics Canada where “improved pasture” and “unimproved pasture” refer to rangeland seeded with introduced grasses and native grassland, respectively. Therefore, the term anthropogenic is inaccurate within the context of North American grassland habitat because destructive activities that have occurred and continue to occur in the prairies are primarily rooted in economic and national development led by settler-colonial people.

When studying effects of settler-colonial disturbance on wildlife populations it is important to contextualize their impacts on the landscape so that as conservation measures are implemented

the impact of those changes does not repeat past errors that originally caused decline of environmental health. Empirical evidence is necessary before the implementation of management action; however, the addition of historical and social context and multiple perspectives can improve longevity of conservation actions. Indigenous nations of Siksika, Crowfoot, and Dakota, have lived in the Great Plains for at least 13,000 years (Friesen 2006). Acknowledging that the negative impacts of anthropogenic activity in the North American prairies was not caused by Indigenous peoples provides opportunity for accountability and reconciliation for the impacts that settler-colonial people have had in the Canadian Great Plains. The lack of specification in the source of disturbances and habitat loss can easily result in further erasure of Indigenous participation in the implementation of mitigative solutions (Loring and Moola 2021). Additionally, conservation already has a past rooted in negative actions towards Indigenous peoples through their displacement for protected land and national parks (Brockington and Igoe 2006; Agrawal and Redford 2009). The current lens that conservation operates through is centric to western science and European ideology and prioritizes nature without humans instead of nature in cooperation with humans (Mason et al. 2018).

The prairies, specifically, have played a key role in the development and expansion of Canada due to their high agricultural potential and because they joined the East and West coasts of the nation under the ambitious plan of Prime Minister Macdonald. Settler-colonial habitat destruction in the grasslands began around 1846 as European settlers moved west with the colonization of Canada (Blackstock and McAllister 2004). Canadian grasslands that once covered close to 61 million ha have been reduced to closer to 11 million ha (Samson et al. 2004). Through settlement, population expansion, and subsequent urban development, large tracts of natural prairie have been converted to agricultural cropland. Bison (*Bison bison*) herds were intentionally extirpated, rangeland was unknowingly overstocked, and energy development infrastructure further fragmented and degraded habitat. Research and land management practices have attempted to protect areas of the prairies and return habitat to pre-colonization status through practices such as prescribed burns (Brockway et al. 2002), lower cattle-stocking rates (Sliwinski and Koper 2012), the reintroduction of bison herds (Steenweg et al. 2016), along with the creation of protected areas and parks. Although these are positive conservation actions, these

results continue to act as justification for western-oriented management that has historically excluded Indigenous people (Mamers 2020).

One of the top reasons for habitat loss in the prairies has been conversion to agricultural cropland. Agricultural practices in this region started in the mid-1800s and became more common in the 1880s when Canadian naturalist, John Macoun, sent a report back to federal government officials that the prairies would be productive for farming purposes (Waizer 2003). The region that Macoun described as having productive potential for cultivation spanned from south-western Manitoba to south-eastern Alberta. Settlers who moved out west to cultivate and farm this region experienced success for the first few decades after their arrival (Lewis 1981); however, the prairies had much less potential than Macoun had observed because of an unusually heavy rainfall cycle during his expedition (Waiser 2003). Triggered in part by Macoun's misinformed enthusiastic report of cultivatable land (Waiser 1976) and motivated by the expansion and completion of the Canadian Pacific Railway (McKee and Klassen 1983), the prairies entered a phase of heavy disturbance that is ongoing to this day.

As the country developed, the destruction that settlers brought on the prairies expanded from agriculture and ranches to energy infrastructure (Friesen 2017). This industrial growth was an attempt to find similar economic success to the initial wheat production boom, and to follow suit with the economic success the United States was having from oil discoveries in the early 1900s. The discovery of oil in Leduc, Alberta, in 1946 created a surge in economic activity (Stanford 2014). That first discovery in Alberta has led to an economic dependence on oil extraction that still largely exists to this day. The subsequent events of oil extraction in Alberta has followed a boom-and-bust trend that is typical of extractive staples economies (Stanford 2014). The reduction in conventionally, easily available oil then led toward a focus on bitumen, a resource that involves an extensive process to extract and refine. Ultimately, Alberta has continuously moved towards resources that are increasingly difficult to extract while leaving habitat disturbed from past resource extraction activity such as empty wells abandoned by energy companies (commonly referred to as orphaned wells), fences, roads, and planted invasive species like crested wheatgrass (*Agropyron cristatum*). The dependency on extractive resources that exists within Alberta has created a divisive atmosphere with regards to financial government investment, how land is treated, and among people where perspectives tend to be more extreme.

Due in part to the normalization of habitat destruction and precedent set by colonial and settler activity, destruction is likely to continue from development and expansion of urban areas and the ongoing extraction and unsustainable use of natural resources. The prairies currently require intense and urgent conservation effort and rehabilitation, and at-risk grassland species conservation need urgent management.

1.3 Effects of oil development on grassland songbirds

The changes that continue to occur due to energy development in the North American Great Plains have resulted in intense ecological disturbance. Oil development disturbance, including the process of drilling oil wells and extracting of oil, can last years (Pipher et al. 2016). Despite the push towards renewable energy, the oil and gas sector of the energy industry continues to drill and extract oil and natural gas from new wells each year (Alberta Energy Regulator 2021) and has experienced an increase in the past year. Thus, it is still necessary to study effects of oil infrastructure and its associated noise on avian species, especially while settler-colonial disturbances remain present on the landscape and when there is limited enforcement of removing inactive oil infrastructure.

Disturbance from energy development includes the footprint of the oil or gas well infrastructure and accessory development like roads and powerlines leading to the well. In southeastern Alberta, the focal location of this study, energy development has steadily continued to present day with infrastructure from many retired oil and gas wells remaining on the landscape. Energy development can have a varying footprint of habitat loss depending on species (Thompson et al. 2015; Ware et al. 2015). For example, natural gas extraction disturbance can affect occupancy and abundance of species like Savannah sparrows (*Passerculus sandwichensis*) and Sprague's pipit (*Anthus spragueii*) while having no apparent effect on chestnut-collared longspurs (*Calcarius ornatus*) (Hamilton et al. 2011). However, chestnut-collared longspurs were found to have larger clutches farther from gas wells (Yoo and Koper 2017), and sensitive species may avoid wells by up to 350 m (Thompson et al. 2015). Areas with oil wells have a higher abundance of brown-headed cowbirds (*Molothrus ater*) (Ludlow et al. 2015) in addition to an increase in nest parasitism (Bernath-Plaisted et al. 2017). The physical footprint alone of oil and gas infrastructure can have a negative effect on grassland songbird nesting success, even in

the absence of noise and human activity (Bernath-Plaisted and Koper 2016). In addition to disturbance related to active oil wells, many abandoned, inactive oil wells exist on the landscape. These orphaned oil wells do not emit any noise associated with oil extraction but well heads and associated roads often remain structurally intact, potentially negatively affecting many species (Bernath-Plaisted and Koper 2016).

Oil wells are accompanied by accessory infrastructure such as fences, powerlines, and roads. Fences almost always surround the oil wells to protect the infrastructure from cattle interference. Electrical powerlines are associated with many types of oil extraction pumps, and dirt and gravel access roads are required for regular maintenance and vary in vehicular traffic intensity. Workers also visit wells for maintenance and safety checks and for oil collection from wells that are not connected to a pipeline. Roads and other linear disturbances associated with energy infrastructure can further disturb habitat demonstrated through, and resulting in, reduced abundance, altered behaviour, or reduced nest densities. Chestnut-collared longspur (*Calcareous ornatus*) are less abundant near roads but more abundant near trails (Sutter et al. 2000) and female chestnut-collared longspurs have shown reduced parental care near roads (Ng et al 2019). Sprague's pipits (*Anthus spragueii*) and Baird's sparrows (*Centronyx bairdii*) avoid nesting within proximity of trails (Ludlow et al. 2015). Proximity to roads affects fledging timing and the number of successfully fledged offspring. Nests close to roads fledged later and fledged fewer nestlings (Ng et al 2019). Other linear disturbances that can directly affect the avian community include various types of fencing that exist around energy infrastructure and that create pastures for cattle. Perches that exist as fencing and powerlines decrease the abundance of chestnut-collared longspurs (Nenninger 2016). Linear disturbances like roads and powerlines have fragmented much of the remaining grassland habitat creating edge effects that can result in additional habitat loss for species that avoid edges. Edge effects increase with habitat fragmentation (Fletcher 2005) and greatly decrease habitat quality for species that exhibit sensitivity to habitat area (Herkert 1994; Sliwinski and Koper 2012; Thompson et al. 2015). Species that are sensitive to patch size suffer from additional habitat loss through habitat fragmentation (Davis 2004). Species can also experience an ecological trap when reduced habitat quality is not perceived but individual condition is negatively affected. For example, crested tit

(*Parus cristatus*) nestlings had lower body mass in habitat with increased fragmentation, which results in a delay in post-fledging dispersal (Lens and Dhondt 1994).

Additional habitat degradation can result from the introduction of exotic vegetation associated with energy development. Crested wheatgrass is strongly associated with oil and gas infrastructure (Dale et al. 2007) because in the late 1980s and early 1990s it was often planted after pipelines had been installed (Koper et al. 2014). Introduced vegetation can cause habitat degradation from structural differences; for example, crested wheatgrass can have a negative effect on nest success, defined by at least one young fledgling, and abundance of certain grassland songbird species (Lloyd and Martin 2005), for example, Sprague's pipit nest success decreases with increasing amount of crested wheatgrass. Habitat degradation from the introduction of exotic vegetation creates a perceptual trap for Savannah sparrows which have lower abundance near crested wheatgrass but have nests that fledge more young (Ludlow et al. 2015).

Peripheral infrastructure from energy development can have varying and sometimes unpredictable effects. The presence of fences and powerlines can alter ecological systems through the increase in predator abundance (Hethcoat and Chalfoun 2015). This may increase predation risk, but the impacts of predator abundance on nesting success is complex. For example, infrastructure increases the abundance of some avian predators by increasing the availability of perch sites (Bylo et al. 2014), but avian predators are likely to prefer small mammals, which are also nest predators, as a primary prey source (Ackerman 2002). Additionally, Richardson's ground squirrel populations fluctuate with hawk populations (Schmutz and Hungle 1989). So, increased abundances of avian predators can sometimes lead to increased survival rates for fledglings by reducing small mammal abundances (Schmutz and Hungle 1989).

1.4 Effects of noise on adult and fledgling songbirds

Anthropogenic noise resulting from energy development can have direct and indirect effects on species interactions, both positive and negative, and can change ecological interactions. The main difference between most active and inactive oil wells is noise and human activity. Noise is emitted from active wells extracting oil and can affect species within range of this noise to

varying extents. Noise degrades habitat, as indicated by reduced bird communities and body condition of individuals in proximity to noise (Ware et al. 2015). Ovenbirds (*Seiurus aurocapilla*) are negatively impacted by the presence of chronic industrial noise as shown by reduced pairing rates and younger, less experienced males being more abundant near noise-generating sites (Habib et al 2007). Oil well noise also affects female chestnut-collared longspur distribution, as heavier, older, females nest closer to oil well infrastructure (Des Brisay 2018). Proximity to noise produced from gas well compressors reduced abundance of gray flycatchers (*Empidonax wrightii*) and western scrub jays (*Aphelocoma californica*); however, nesting success was higher for flycatchers near noise (Francis et al. 2011). Noise also has negative effects on abundance of Greater sage-grouse (*Centrocercus urophasianus*) at leks (Blickley et al. 2012), and on many boreal species (Bayne et al. 2008).

Anthropogenic noise has been thoroughly studied in the past decade and continues to gain interest because noise has such a large potential footprint (Rosa and Koper 2018). Noise characteristics, such as amplitude, vary among different types of energy infrastructure which result in different effects on avian species. For example, noise from natural gas compressor stations is louder than screw pumps because of the additional motors, turbines, fans, and human activity (Rosa 2019). The noise from compressor stations is loud enough to distract from alarm signals coming from conspecifics resulting in reduced feeding latency at Savannah sparrow nests (Antze and Koper 2018). In this same study, no effect of noise from screw pumps was found on feeding latency, demonstrating that noise intensity and type can have varying effects. Some variation in the impact of oil extraction noise from wells observed among previous studies might be explained in part by the fact that noise intensity varies among types of energy development infrastructure. Two of the main types of oil extraction infrastructure used across Alberta's prairies are pumpjacks powered by the provincial electrical grid and generator-powered pumpjacks. The main difference between the two is that the former is connected to the electrical grid and emits 62 dB(C) (53 dB(A)) of noise at 10 m while the latter is powered by a generator and emits 73 dB(C) (57 dB(A)) (Rosa 2019). Generator-powered screw pumps are frequently used in the energy industry to extract oil. Under normal circumstances, these wells are constantly extracting oil and emitting noise because of the generator. Noise from energy development can reduce habitat quality or result in habitat loss for some species (Nenninger and Koper 2018). For

a group of species in steep decline, such as grassland songbirds, further habitat loss or a reduction in habitat quality could result in intensified population declines and, potentially, extirpations.

Chronic noise can cause changes in behaviour that directly affect the fitness of an individual and indirectly affect the offspring of an individual. Noise can alter behaviours, such as increasing vigilance, which may result in a lower individual body condition (Ware et al. 2015), or reduce parental care of nestlings, which results in decreased reproductive output (Ng 2019). Changes in behaviour such as increased parental vigilance (Ware et al. 2015) and altered song (Curry et al. 2017) has been observed in some species. Behaviour changes like increased vigilance results in overall less time spent foraging and fewer provisioning visits to young. Noise may affect growth and survival of fledglings from noise masking communication between parents and fledglings. Tree swallow (*Tachycineta bicolor*) nestlings miss feeding detections in the presence of noise due to masked signals (Leonard and Horn 2012).

Nesting success of ground-nesting birds appears to be independent of operating noise from oil wells (Bernath-Plaisted and Koper 2016); however, fledgling survival may not follow the same pattern. Nest success is not an equivalent metric to fledgling survival and there are different factors that affect fledglings and not nestlings. For example, nestlings often rely on adult presence for thermoregulation and protection from environmental conditions while fledglings do not. One key difference between nestling and post-fledging stages is mobility of young. Because of this difference, noise may be one of the factors that has varying effects between the nestling and post-fledging stages. It is important to determine whether noise from oil wells per se impacts survival of birds at each life stage because mitigation strategies may differ by stage.

Noise could result in a decrease in parental care of fledglings, as has been observed during the nestling stage of chestnut-collared longspurs (Ng et al. 2018). Parental care could continue to be affected by noise into the post-fledging period as juvenile birds gradually move towards independence (Ribic et al. 2019). The length of period where fledglings receive parental care can have direct effects on fledgling survival; a longer period of parental care during the post-fledging dependent period is positively correlated with survival of fledglings in barn swallows (*Hirundo rustica*) (Gruëbler and Naef-Daenzer 2010). Fledglings remain dependent on parental care after

leaving the nest (Harris 1944) and gradually become independent. Individuals can sustain flight at approximately 12 days old. Before that time fledglings rely on alarm calls from parents and remaining hidden from predators. Fledglings continue to rely on feeding from parents and, therefore, also on continued communication after leaving the nest (Bleho et al 2020). However, noise could have other indirect effects on fledgling survival. Predation directly affects survival, but effects of noise could be amplified by masking alarm calls or resulting in missed detections for feeding which could result in lower body condition. There has been no evidence of reduced nest success near oil wells. My research is the first to study the effects of oil well noise on survival during the post-fledging period in the chestnut-collared longspur.

1.5 Post-fledging and adult survival

Full annual life cycle research provides important survival estimates and allows inferences to be made about why populations might be experiencing significant declines. In migratory avian species, it is important to locate declines in specific life stages within the full annual cycle (Marra et al. 2015). For migratory species this is particularly important because life stages within the full annual cycle occur in multiple geographic areas. For example, the breeding grounds, first migration south, non-breeding grounds, and first migration north each occur over different geographic ranges. Understanding survival rates on the breeding grounds can provide information into population sources and sinks, and factors that may be contributing to overall population declines. Negative effects that occur on the breeding grounds could reduce the overall reproductive output of a population as lower reproductive success in general would result in lower recruitment. Adult survival on the breeding grounds can be vastly different compared to survival during migration or on the wintering grounds.

Nest outcomes have often been used as a proxy for gauging reproductive success as it is easier to monitor success of nests than survival of fledglings. However, Streby et al. (2014) argue that reproductive success should be redefined to include the post-fledging period. The post-fledging period is increasingly being studied due to the feasibility and affordability given advances in technology. Studying this life stage is important because the post-fledging period requires different resources compared to the nestling stage and can provide further insight into understanding species population trends (Jenkins et al. 2016).

All grassland songbird species, including my two focal species, have to endure particularly difficult conditions to survive the first breeding season. Grassland songbird eggs, nestlings, and fledglings are all extremely susceptible to predation and intense weather events that occur in the prairies. During the post-fledging life stage, individuals are still dependent on parental care such as feeding and alarm calls. Grassland songbirds leave the nest before having fully developed wings and flight abilities making them extremely susceptible to predation. Individuals can sustain flight at approximately 12 days old. Before that time fledglings rely on alarm calls from parents and remaining hidden from predators. Nestlings that stay in the nest longer will benefit from more parental care and feeding but might incur higher risk from staying in the same place for a longer time (Ribic et al. 2019). Fledglings also still rely on their parents for food, which requires continued communication after leaving the nest (Bleho et al 2020). Mortality is highest in the first three to four days post-fledging and survival may vary most strongly with phenotype during that time (Jones et al. 2017a). Post-fledging survival rates for other grassland species range from 21% over 14 days for Grasshopper sparrows (Hovick et al. 2011) to 54% over 29 days for Dickcissel (Suedkamp Wells et al. 2007).

Age is often a key predictor of fledgling survival, and this could be due to the correlation with growth and development. Heavier individuals with better wing development have varying benefits during the post-fledging stage. Benefits for individuals with better body condition are two-fold. Firstly, individuals are less likely to die from exposure to environmental conditions due to increased efficiency in thermoregulation (Jones et al. 2017b), and secondly, advanced wing development has been shown to increase survival of fledglings (Martin et al. 2018) likely because it improves mobility and avoidance of predators. Fledgling body condition may only result in increased survival because it is correlated with wing musculature (Veasey et al 2000). Species with higher predation risk fledge earlier at relatively smaller mass – suggesting greater predation risk favors a strategy of prioritizing wing growth over mass (Cheng and Martin 2012). Although increased activity and movement is likely to attract predators, fledgling movement tends to occur in intervals and not continuously. Movement that occurs only periodically is less likely to attract predators compared to constant movement. It could be beneficial for fledglings to not move much within the first week after leaving the nest to allow for easier detection by parents and, therefore, likelihood of increased feedings. Alternatively, it might be a more

successful strategy to move farther from the nest within the first week to reduce the likelihood of detection by a predator. Moving farther from the nest is a strategy to decrease predation risk, but environmental noise could reduce the ability of parents to locate fledglings by masking calls.

Weather events are likely to have a strong effect on fledgling survival and nest success. Survival of western meadowlark fledglings was positively affected by ambient temperatures (Giovanni et al 2015). Exposure to colder temperatures could lead to mortality events in the absence of adequate cover. Grassland songbirds are especially vulnerable to climate change (Jarzyna et al. 2016; Nixon et al. 2016; Wilsey et al. 2019) and ongoing anthropogenic disturbance resulting in additional habitat loss on breeding and wintering grounds. Chestnut-collared longspurs have not been found to be affected by drought, while lark buntings are (Wilson et al 2018). Investigating effects of weather is particularly important now, because as consequences of climate change intensify there will be an increasing likelihood of stochastic weather events (Sauchyn 2010). Ambient temperatures are predicted to vary widely within seasons as climate change progresses. More intense weather such as droughts and flooding are likely to affect fledgling survival. Climactic effects are also likely to continue to affect population sizes of these species. Climate change may greatly affect species on non-breeding or breeding grounds (SÆther et al. 2004). Understanding population recruitment of the full annual cycle can provide insight to where climate change might have the greatest affects (Culp et al. 2017).

1.6 Natural history of focal species

I studied two species of grassland songbirds, chestnut-collared longspurs and Baird's sparrow. Both species are grassland obligate species. My main focal species for this study was the chestnut-collared longspur. Chestnut-collared longspurs are grassland obligates that breed in the Great Plains and migrate to the southern U.S. and Mexico for the non-breeding season (Bleho et al 2020). Males are thought to be first to arrive on the breeding grounds to secure territories that are retained throughout the breeding season. Chestnut-collared longspurs prefer naturally disturbed habitat that occurs primarily from grazing (Bleho et al 2020) because foraging is likely facilitated due to reduced vegetation and bare ground. Similar environments are created through energy development, in some cases from increased dirt roads, two-tracks, and grazing from cattle

as cattle tend to be neophilic and are frequently attracted to fencing which is a prominent structure associated with energy development. There has been no evidence of reduced nest success near wells, and my research is the first to study the post-fledging period in the chestnut-collared longspur. Longspurs may have up to three broods within a single breeding season (Hill and Gould 1997). The first nesting attempt begins around mid-May while second broods often occur close to July. Like most songbirds, longspurs are an altricial species and fledge approximately 11.1 days after hatching (range 7 – 15 days) (Jones et al. 2010) although the average age of fledging per nest may be lower farther from roads (Ng 2016). The primary cause of nest failure in chestnut-collared longspurs is predation (see Davis 2003) as is the case with most grassland songbirds. Males increase parental care as nestlings age and will continue feeding fledglings during the post-fledging period. Fledglings will stay with parents while learning to fly and while still dependent on food delivery for at least two weeks after fledging (Harris 1944).

I also studied Baird's sparrow as overall population trends are declining and are listed as Special Concern under the Species At Risk Act. Baird's sparrow is a grassland obligate songbird species whose range covers the North American Great Plains and extends to the southern United States and northern Mexico. Baird's sparrow prefer a minimum vegetation height of 20 cm but less than 100 cm (Dechant et al. 2002) on their breeding grounds in mixed grass prairies. After arrival on the breeding grounds around mid- to late May, males will choose and defend a territory. Baird's sparrow populations have experienced significant declines since the 1960s, as have many grassland songbird species.

Conservation concerns exist for both Baird's sparrows and chestnut-collared longspurs. Baird's sparrow are listed federally under the species at risk as sensitive and chestnut-collared longspurs are listed as threatened. These species have breeding ranges that overlap within southeastern Alberta, although their ranges have reduced considerably from what would have existed historically. The population health of these two species depends on understanding how anthropogenic disturbances affect them in their breeding range.

This study aims to contribute survival rates to the full annual cycle of both Baird's sparrows and chestnut-collared longspurs. To accomplish this, I studied these two species on their breeding grounds. Chestnut-collared longspur adults and fledglings were radio-tagged to track

daily for survival rates. Only Baird's sparrow adults were radio-tagged to track daily because nests in the study area were too difficult to find and would not have resulted in a sufficient sample size. Baird's sparrow adults were only tagged and tracked for the first of two field seasons because many individuals were unable to be relocated after being tagged and were the most time consuming to track. For a more efficient use of the radio-tags to get a larger sample size and because chestnut-collared longspur fledglings were able to be tagged, chestnut-collared longspurs were prioritized between the two species for the second field season.

1.7 Research Objectives

The objectives of this study were to investigate the effects of oil wells and the associated noise on survival of two different grassland songbird species on their breeding grounds. To accomplish this, I used radio-telemetry and tagged adult and fledgling chestnut-collared longspurs and adult Baird's sparrows. To isolate the effects of noise from the physical infrastructure of oil wells, I used an experimental design that broadcasted oil extraction recordings. I used an experimental design that included four treatment types so that I could compare effects of noise from oil extraction to that of active oil wells. These treatment site types included: 1) active oil well emitting noise, 2) noise-broadcasting unit, 3) equivalent infrastructure from noise-broadcasting unit that remained silent, and 4) a control where there was minimal unnatural disturbance and no noise. The constructed, noise-broadcasting units and the active oil wells were emitting noise continuously for the entirety of the breeding season

Results from this thesis will contribute to full annual cycle information for the chestnut-collared longspur and Baird's sparrow. Additionally, these results can be used to inform future management decisions within the breeding range of these species, specifically in southeastern Alberta.

1.8 Predictions for post-fledging and adult survival

If oil infrastructure and the associated noise have a negative effect on survival, then I expect survival to be lowest on noisy, active oil well sites. If noise alone has a negative effect on survival, then I expect survival to be lowest on sites with active oil wells and active noise broadcasting systems. If the infrastructure from oil wells alone has a negative effect, then I expect survival to be lowest on active oil well sites and not the noise-broadcasting sites.

Chapter 2 Methods

This study was accomplished under the University of Manitoba animal care protocol F15-005, Canadian bird banding permit 10840E, and Canadian Wildlife Service permit #11-MB/SKL/AB- SC007.

2.1 Study area

Research for this study was conducted in Newell County in southeastern Alberta (Brooks, Alberta 50° 33' 51"N 111° 53' 56" W). This land was traditionally the land of Niitsítpiis-stahkoií ᓃ'ᓂ-ᓂ ᓃ"ᓃ', Michif Piyii (Métis), and Očeti Šakówiŋ (Sioux) peoples. It is currently managed by the Eastern Irrigation District where ranchers and energy companies lease parcels of land for use.

The study sites were chosen within dry, mixed-grass prairie habitat. The soil type in this area is brown chernozemic. Common grass species in this ecoregion include western porcupine grass (*Hesperostipa curtiseta*), needle-and-thread (*Hesperostipa comata*), northern wheatgrass (*Agropyron cristatum*), western wheatgrass (*Pascopyrum smithii*), blue grama (*Bouteloua gracilis*), and porcupine grass (*Stipa spartea*) but also including introduced species such as crested wheatgrass (*Agropyron cristatum*). Additional vegetation species such as rough fescue (*Festuca scabrella*), sedge (*Carex obtusata*), club moss (*Selaginella densa*), spike oat (*Helictotrichon hookeri*), silver sage (*Salvia argentea*), and golden bean (*Thermopsis rhombifolia*) are found in mixed-grass prairie.

Chestnut-collared longspurs are found at high density within this study area as well as other passerine grassland species like Baird's sparrow (*Ammodramus bairdii*), Sprague's pipit (*Anthus spragueii*), grasshopper sparrow (*Ammodramus savannarum*), horned lark (*Eremophila alpestris*) and Savannah sparrow (*Passerculus sandwichensis*). Common predators of nests in this system are northern grasshopper mouse (*Onychomys leucogaster*), deer mouse (*Peromyscus maniculatus*), prairie rattlesnake (*Crotalus viridis viridis*), plains garter snake (*Thamnophis radix*), Richardson's ground squirrel (*Urocitellus richardsonii*), northern harrier (*Circus cyaneus*), and Swainson's hawk (*Buteo swainsoni*) (Renfrew and Ribic 2003; Kirkham and Davis 2013; Bernath-Plaisted 2015; Pietz and Granfors 2017).

Daily minimum temperatures during the 2017 and 2018 breeding seasons ranged from 1.4 – 17.1 and 3.5 – 14.4 degrees Celsius. Daily maximum temperatures during 2017 and 2018 ranged from 16.4 – 36.6 and 10.2-36.5 degrees Celsius. Daily precipitation in 2017 and 2018 ranged from 0.3 – 20.1 and 1.4 12.7 mm, respectively.

2.2 Experimental design

To investigate the effects of noise on adult and fledgling survival, we used active oil well infrastructure and a sound broadcasting system that emits active oil extraction noise from wells without the infrastructure of a real well (Rosa et al. 2015). This experimental design had four treatment types with three replicates of each 800 m by 800 m site. The experimental infrastructure was placed at the center of each site. Site placement, shown in Figure 1 and Figure 2, was chosen based on previous studies that occurred in this area and where land access was granted by Eastern Irrigation District. Site location remained the same between years but the treatment assigned to each site changed as shown in Figure 1 and Figure 2.

The four treatment types assigned to sites were 1) active, generator-powered screw-pumps, 2); noise broadcasting unit, 3); silent broadcasting unit; and 4) control sites where no oil extraction infrastructure was present. Active, generator-powered screw-pumps are common oil extraction infrastructure that emit sound while extracting oil throughout the day and night and are often surrounded by a gravel well pad, fencing, and a dirt access road. The experimental noise broadcasting unit had 5 solar panels and a small enclosure housing the electrical equipment and four 6V batteries (**Error! Reference source not found.**). High fidelity noise recordings were taken of generator-powered screw-pumps. Once the broadcasting system was set up, sound pressure level (SPL) transects were conducted following protocol from Rosa et al (2015) to confirm accuracy of reproduced soundscape. One of the treatment types for this experiment were sites with silent broadcasting units that emitted no noise but had the same infrastructure as the noisy units. This was done to control for any impact of the noise broadcasting system infrastructure on birds. Control sites were areas that were the same size as other sites and contained no active or inactive oil wells or playback infrastructure and had limited anthropogenic disturbance. In the second year of the study (2018), the same site locations were used; however, experimental treatments were randomly reassigned to control for differences among sites. Real

oil infrastructure sites were used in both 2017 and 2018, except if a well was inactive in the subsequent field season, in which case a replacement oil well was used.

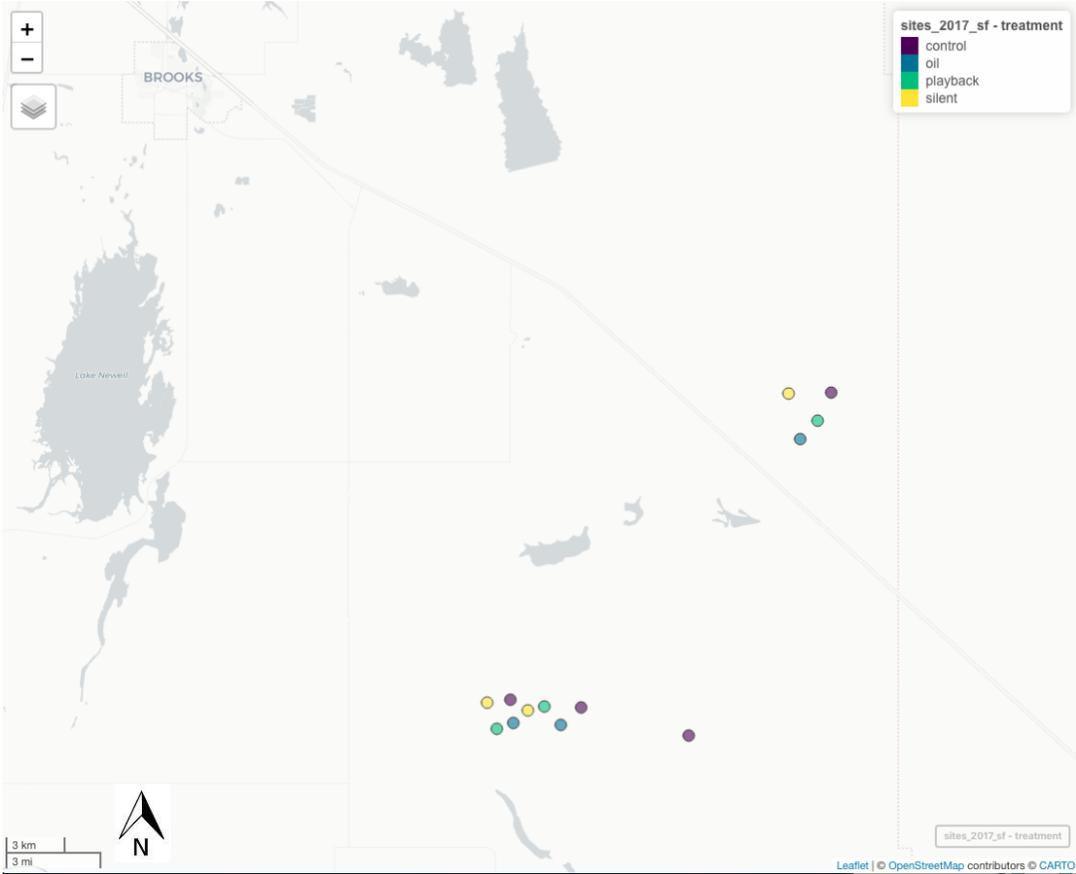


Figure 1 Site locations for 2017, colour-coded by treatment type and shown in relation to Brooks, AB

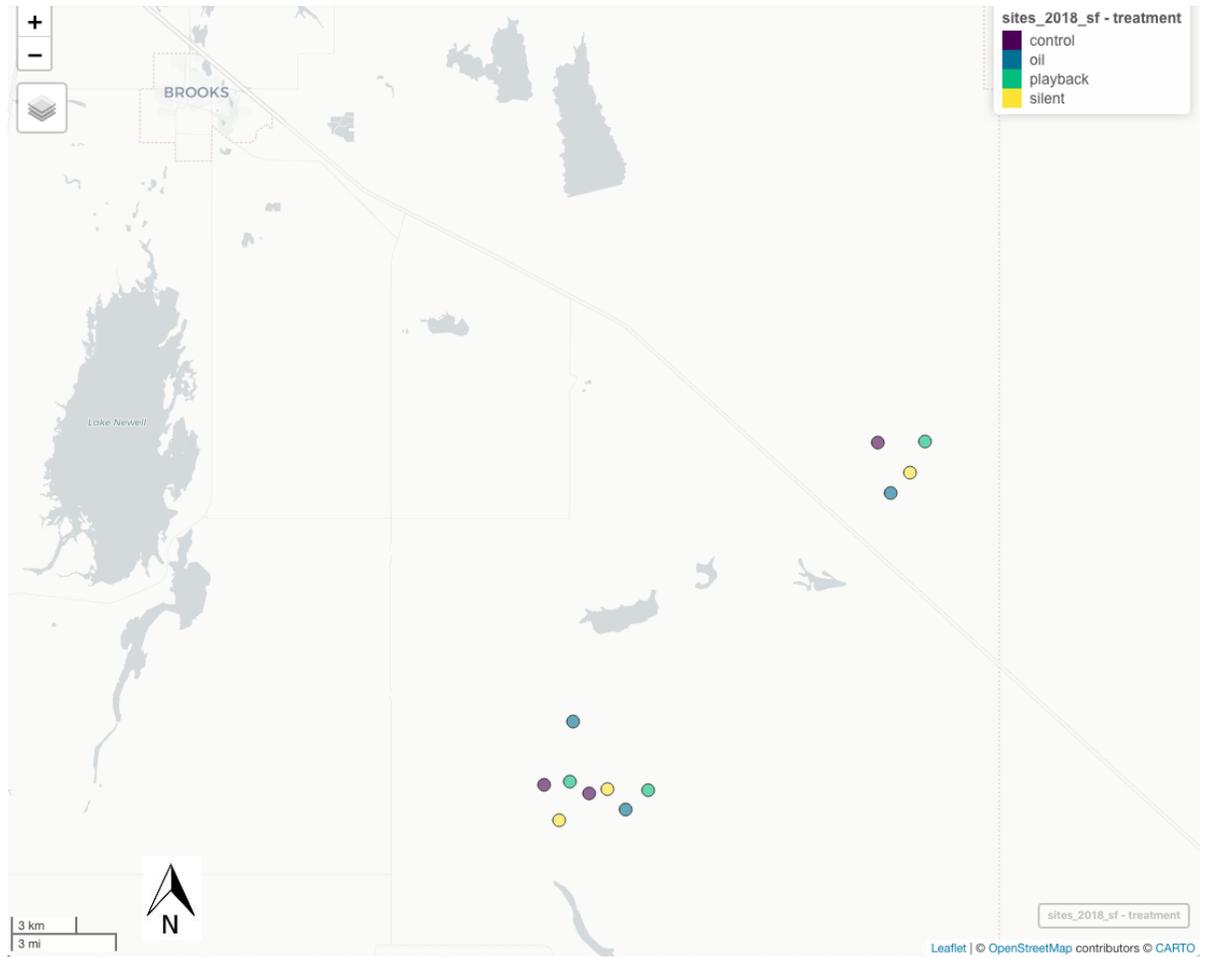


Figure 2 Site locations for 2018, colour-coded by treatment type and shown in relation to Brooks, AB



Figure 3 This image shows one active, noise broadcasting unit used for the experimental design. The infrastructure includes a wooden box housing batteries and electrical equipment, five solar panels, and metal fencing. Image from Hannah Carey 2018.

In addition to the experimental noise set-up, all non-naturally occurring disturbances within 1 km of the center of all study sites were mapped. These disturbances included linear features (electrical powerlines, fences), roads, and additional oil and gas well infrastructure commonly found in the surrounding landscape.

2.3 Field methods

We monitored 130 chestnut-collared longspur nests in 2017 and 72 in 2018. Nest searching took place in the sites mentioned above. Nests were primarily found using behavioural observations, while a few were found through incidental flushing of females off nests. Nests were marked with a small wooden stake wrapped in flagging tape 10 m south of the nest cup and a whisker flag 10 m west and were monitored every 1-4 days. Most nest searching took place

between 07h00 and 10h00 when incubating females were likely to be on nests (Kirkham and Davis 2013). A GPS point was taken for every new nest and monitored until the nestlings reached banding age. Photos of nest contents were also taken when a nest was found and for every life stage change (i.e. eggs to nestlings), and when ageing nestlings.

Nestlings were banded and fitted with a radio-tag at approximately 8 days old (Jongsomjit et al. 2007). The radio-tags used for fledglings were PicoPicAG339 (Lotek Wireless Inc., New Market Ontario). These tags have minimum battery life of 18 days. The radio-tag and all harness material represented less than 4% of an individual's weight. Radio-tag harnesses were pre-fitted to reduce the time the individual spent in-hand and followed a leg-loop design (Rappole and Tipton 1991). Harnesses were made with elastic, nylon cord (0.5 mm) and a skin-safe strong adhesive (Vet Bond). Nestlings were aged as day 8 when they appeared fully feathered with pin feathers partially unsheathed. Nestlings were banded and tagged during the same nest visit, if possible. Nestlings were banded with a USGS size 1 aluminum band. At the time of banding, we measured nestling mass, wing chord, length of the ninth primary feather (p9), length of the unsheathed portion of the ninth primary feather, and tarsus length. Within a brood, only nestlings that weighed 12 g or more were fitted with a radio-tag. We radio-tagged two randomly-selected nestlings that were above the weight requirement from broods to ensure that sample sizes would be sufficient (Suedkamp Wells et al. 2007). If no nestlings met the minimum weight requirement then the nest was revisited the follow day. Radio-tags were fitted with a nylon harness following an adapted method by Rappole and Tipton (1991).

Adult male and female chestnut-collared longspurs were captured using either mist nets and song playback (males only) or walk-in traps over nests (either male or female). Once captured, adults were banded with a USGS size 1 aluminum band and a unique three-coloured combination of plastic Darvic leg bands for identifying individuals. Radio-tags for adults were the PicoPip AG379 (Lotek Wireless Inc., New Market Ontario) and had minimum battery life expectancy of 40 days. Adult tag range using omni, 3-prong and 5-prong antennas were 220 m, 250 m and 350 m, respectively. Locations were recorded where an individual was first observed.

Once radio-tags were deployed, tagged individuals were relocated daily using receivers by Communications Specialist Inc. and Advanced Telemetry Systems Inc. until either the radio-tag

battery died, the individual died, or the individual went missing. Individuals, specifically fledglings, frequently went missing. Most individuals that were unable to be located initially were never relocated. Based on communication with LOTEK and previous VHF radio-tag studies, the likelihood of radio-tag failure was minimal. Alternative reasons for an individual not being relocated within the first days post-fledging are predation by predators, such as snakes, ground squirrels and birds of prey because fledglings lack mobility to travel long distances and avoid predators. When an individual went missing, protocol required search efforts to continue every day for one week after the individual went missing and then once per week until the minimum battery life of the tag was reached. Our efforts for looking for missing individuals included scanning 200m from the nest location using a 5-prong antenna and driving with an omni antenna while scanning through frequencies in case the tag frequency had drifted. Tag frequencies often drifted within the first few days of being deployed. If a fledgling was unable to be located within seven days of fledging, we assumed the individual was depredated.

Relocation data were collected for fledgling and adult chestnut-collared longspurs (*Calcarius ornatus*) and adult Baird's sparrows (*Ammodramus bairdii*) between May and August 2017 and for adult and fledgling chestnut-collared longspurs between May and July 2018. For this study I did not collect any data of Baird's Sparrow fledglings as our we were unable to find Baird's sparrow nests in the first field season.

Nestlings and adults were monitored using very-high frequency (VHF) radio-telemetry. Each individual is tagged with a radio-tag set to a unique frequency and relocated using hand-held antennas and receivers. Individuals were relocated until either the radio-tag battery died, the individual died, or the individual could not be found. We used sequential sampling to reduce autocorrelation with approximately 24hrs between relocations except for when weather did not permit (e.g. thunderstorms and high winds).

When a bird was relocated, protocol adapted from Bird Conservancy of the Rockies was followed to assign a status for the individual (BCR 2016). There were five options for individual status that could be assigned: confirmed alive, confirmed dead, no signal, signal moving, in nest. When a signal could not be detected, the individual was classified as missing. If a tag did malfunction, technicians were still able to relocate adult individuals using their unique colour

band combination. Adult tag range using the omni, 3-prong and 5-prong antennas were 220 m, 250 m and 350 m, respectively (using an alternate ATS receiver the distance increased detectable distance to 290 m for the 3-prong). Fledgling tag range was 178 m when using an omni and 193 m when using the 3-prong antenna and <350 m when using the 5-prong antenna.

2.4 Fate analyses

Fate analysis uses binary data to model factors that affect survival. The fate of an individual is recorded as 0 (observed dead or presumed dead) or 1 (observed alive) for each time an individual is relocated. To conduct known fate analysis, the probability of an individual being found has to be 1. When an individual could not be relocated, the two likely options are that the individual emigrated from the study area or was taken out of range by a predator. To address this issue, I developed a protocol to determine whether individuals that went missing could be confidently assumed to be dead or whether the status had to remain as unknown. For each individual, I calculated the distance between relocation points (i.e. step length) to find the furthest a fledgling travelled after leaving the nest. I used age and step length to determine that only individuals that were at least 8 days post-fledging traveled over 250 m between days, suggesting that individuals that could not be located within the first 8 days after fledging were more likely to have been carried away by a predator than to have dispersed out of the study area, assuming there was no radio-tag failure. Similarly, the majority of the 18-day-old individuals had step lengths under 350 m. Therefore, individuals that could not be detected within 18 days since fledging were assumed to have died. Survival is lowest during the first two weeks after fledging, so we are confident that assessing fledgling survival within 18 days of fledging provides an appropriate estimate of survival during the post-fledging period. Individuals that could not be detected after 18 days post-fledging were considered to have an unknown fate.

2.5 Statistical methods

I used logistic exposure models (Shaffer 2004) in R (R Core Team 2021) to investigate post-fledging survival. I used linear mixed models (Bates et al. 2015) to test three main models, biological, weather, and management, using a hierarchical approach such that significant variables from the first tested model, biological, were subsequently included in the next models,

weather and then management. The response variable for all models was the outcome of the individual in each daily observation period: dead or alive.

The biological model was tested first because nestling age at fledging, weight, and brood size could affect survival. Independent variables included in the biological model were nestling weight, number of nestlings within same nest, date as an ordinal value, and number of days since fledging. There was only one case where an individual returned to the nest cup after apparently fledging. The first day an individual left the nest and did not return was considered the fledge date. I used AIC to assess whether number of days since fledging should be modeled as a linear or log value. For subsequent models I used the logarithmic value of days since fledging, as models including this variable had lower AIC values. To account for seasonality I included quadratic date as a term in the models in case survival peaked in the middle of the breeding season (Lusk and Koper 2013). Significant variables from the biological model were kept in all subsequent models.

I next evaluated effects of weather on survival which can affect survival of individuals that could not find shelter from sun, wind, or rain. These data were collected from weather stations within 13km of sites, which were Rolling Hills AGM and Tide Lake stations (<https://weather.gc.ca/>), and downloaded using weathercan package in R (Lazerte and Albers 2018). The weather model included daily minimum and maximum temperatures. Because certain days were missing precipitation data, a separate model was run to test whether precipitation had any effect alone on fledgling survival. For individuals that were located in the morning, average weather data from the previous day were used; for individuals that were located in the evening, average weather data from the current day were used to investigate effects of precipitation and temperature on fledgling survival.

Lastly, the management model included significant variables from both the biological and weather models, and distance from each relocation point to the nearest treatment type. Effects of management were determined using null hypothesis significance testing as the study was designed to address this question. An interaction term for distance from each relocation point to site center, and site type was included in this model.

Chapter 3 Results

3.1 Adult survival of chestnut-collared longspurs and Baird's sparrow

In 2017 we radio-tagged 38 chestnut-collared longspur adults and 14 Baird's sparrow adults while in 2018 we radio-tagged 36 chestnut-longspur individual adults. In 2017, apparent survival for chestnut-collared longspur adults was 96% and 100% for Baird's sparrow adults. In 2018, only chestnut-collared longspurs were radio-tagged and there was 100% apparent adult survival. Of the longspur adults that died in 2017, one adult was found intact and looked emaciated (band number 263175723). This individual was a female with a nest that had been monitored and successfully fledged four nestlings. No nestlings from this individual's nest were tagged for this study because the nestlings were carrying a species of blowfly ectoparasite. In the other mortality case, the tagged female was presumed dead because flight feathers were found outside her nest and there were remains found from one of the two nestlings.

Although there were no known fatalities of the tagged Baird's sparrows, some individuals were assumed to have emigrated from the area because the signal was undetectable and multiple attempts at relocation were unsuccessful. This was a more common occurrence with Baird's sparrow individuals than chestnut-collared longspurs. It is possible that Baird's sparrows would fly out of range of our receivers, or that individuals were disturbed by the event of capturing and tagging and left the area.

3.2 Fledgling survival of chestnut-collared longspurs

In 2017 ($n = 123$) and 2018 ($n = 39$), we radio-tagged 162 chestnut-collared longspur nestlings. I analyzed data from a total of 107 and 32 fledglings from 2017 and 2018, representing a total of 61 and 22 broods, respectively. Individuals that were unable to be relocated but could have dispersed based on age (i.e. > 18 days old) were excluded from the analysis. Known causes of mortality included environmental exposure, and depredation events by snakes and mammals. One deployed radio-tag was tracked to a snake burrow for consecutive days before locating the tag in a pellet that had passed through a snake. In two instances, fledglings were tracked to inside or near the entrance of a ground squirrel burrow. Many fledglings were found with small bite marks that were indicative of a mammal smaller than a ground squirrel. Predation events by

larger avian predators likely occurred; however, we found no evidence during either breeding season of these events. This could be due to larger avian predators carrying tagged fledglings outside of the range of the receivers or the radio-tag itself being destroyed. For many fledglings the cause of death was not easily determined so it was not included as a factor in the analysis. It is worth noting that in 2018, we had multiple nests that failed on the day when nestlings reached banding age due to nest flooding caused by a rainstorm.

Towards the end of June and for the remainder of the breeding season older fledglings had larger step lengths compared to earlier in the season (Figure 4). This is indicated by older fledglings travelling farther within the span of a day and towards the end of the season (Figure 5). However, there was no correlation between date and individual age (Figure 6, Figure 7, Figure 8) perhaps because our survey period coincided with the nesting period, so this effect was not a function of fledglings being older at the end of our field season. This means that season did affect step length, and the apparent effect of season on step length was not caused by fledgling age. Using this information during the breeding season, I was able to confidently determine when individuals that were unable to be relocated could have dispersed based on age. These individuals were excluded from the analysis.

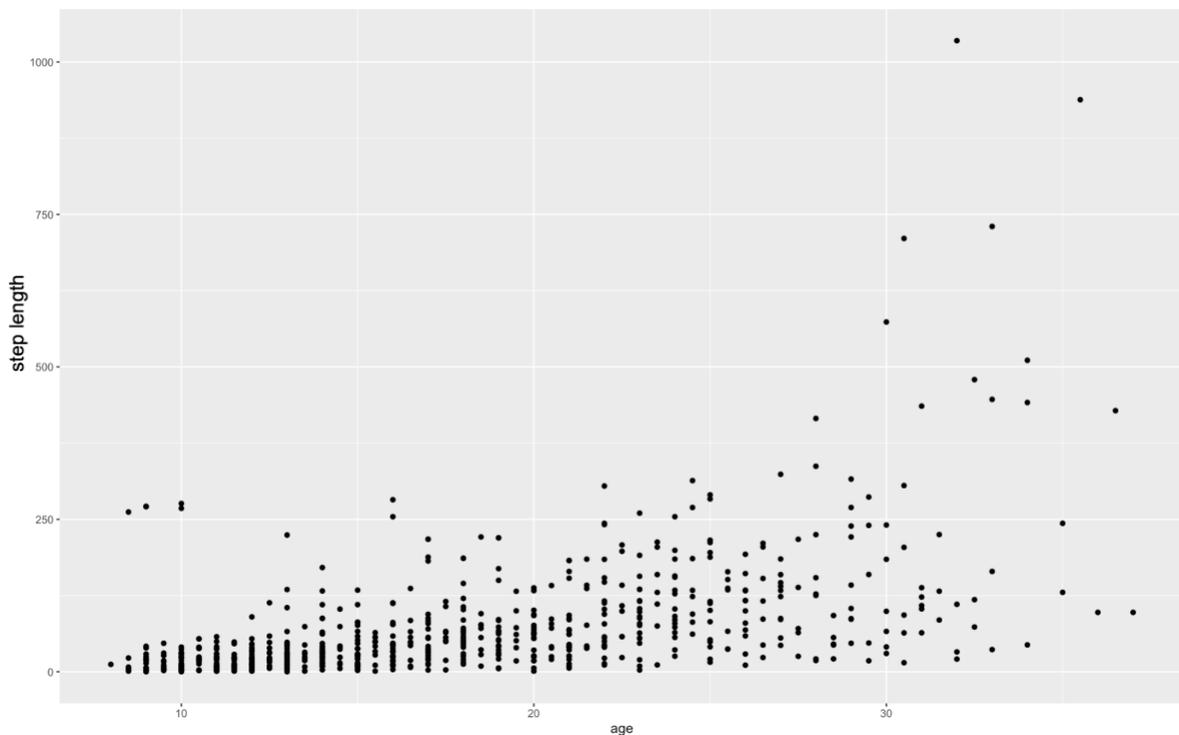


Figure 4 Fledgling step length by age showing that older fledglings moved farther than younger fledglings.

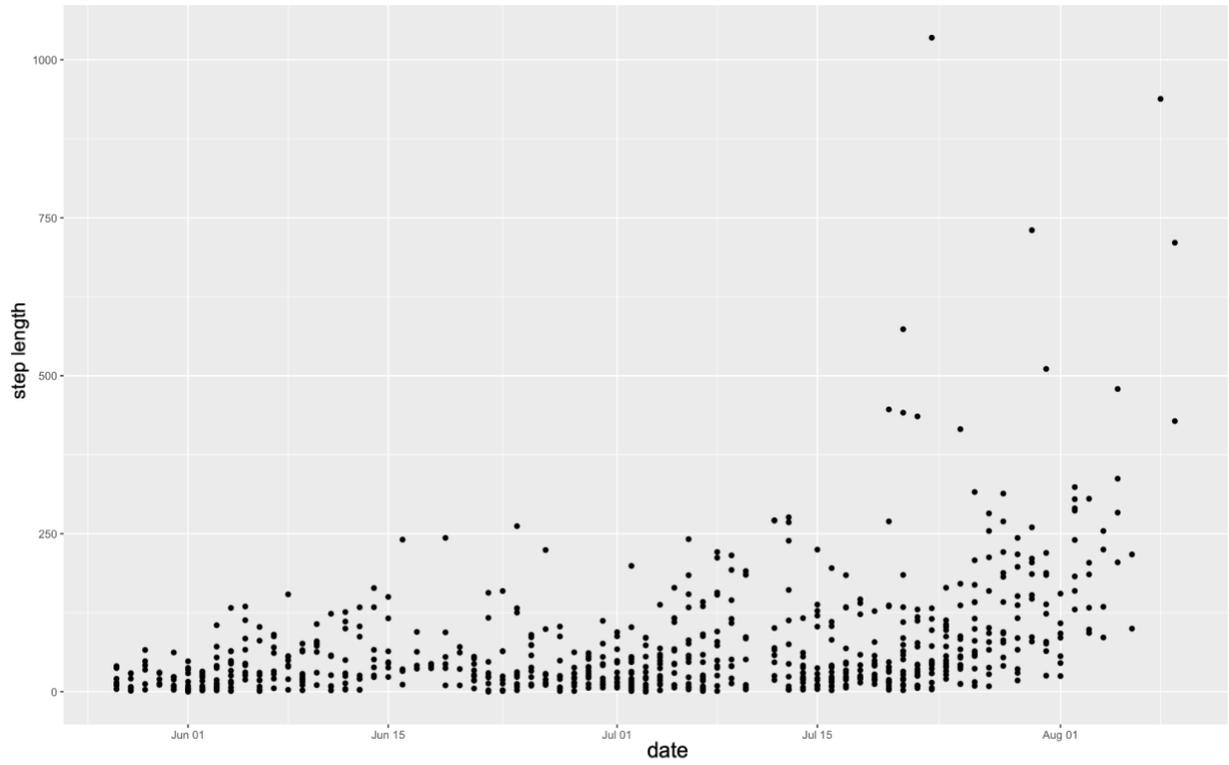


Figure 5 Fledgling step length over breeding season. Fledglings took longer steps towards the end of the breeding season.

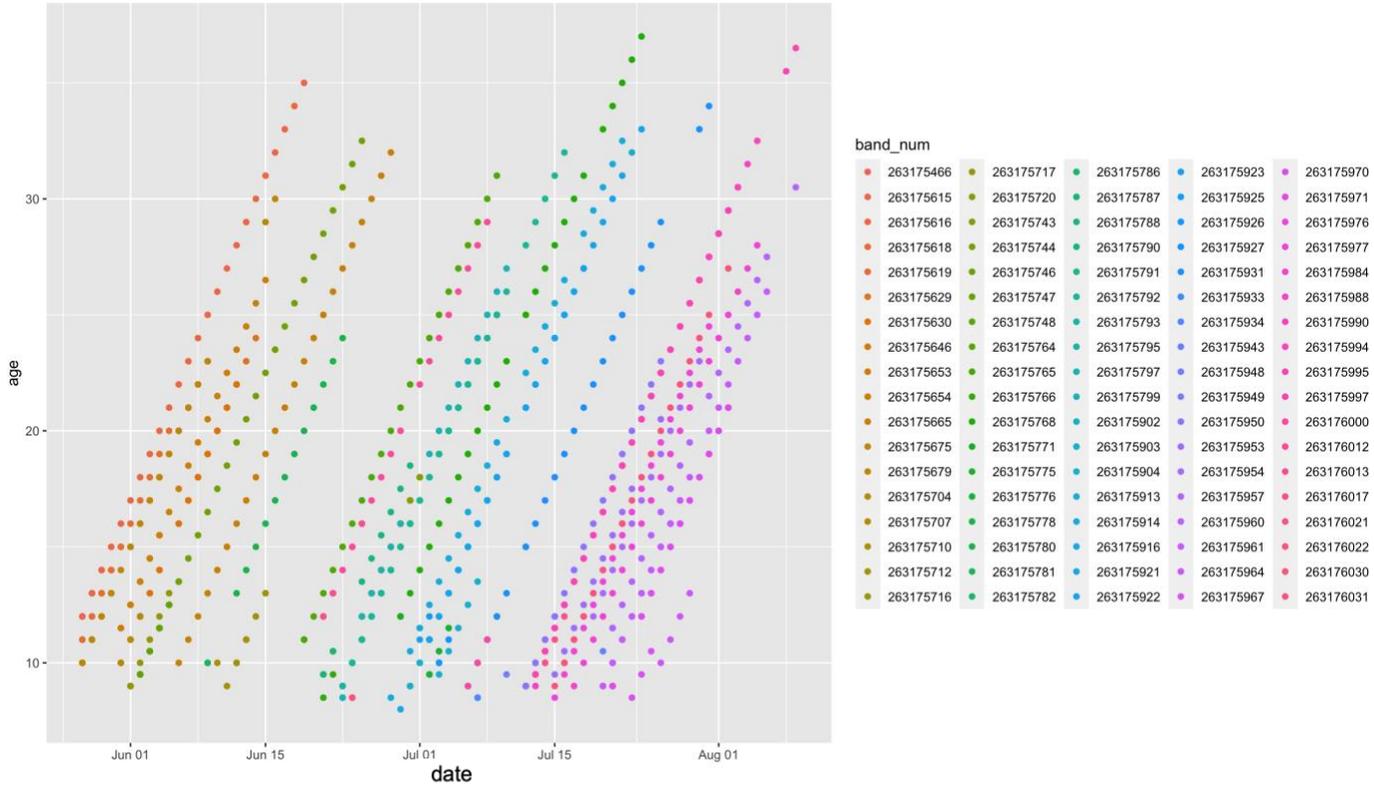


Figure 6 Age of active fledglings over season demonstrating that older fledglings did not bias longer step lengths at end of season.

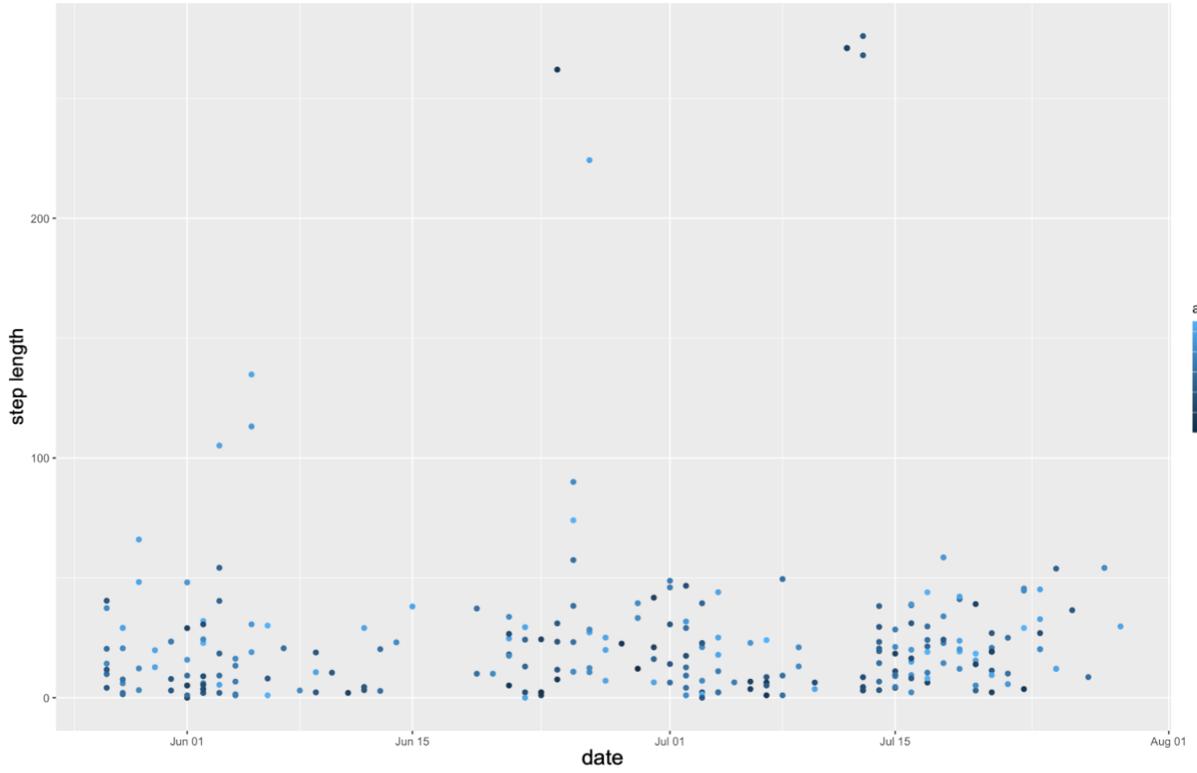


Figure 7 Age of active radio-tagged fledglings throughout 2017 breeding season remained evenly dispersed.

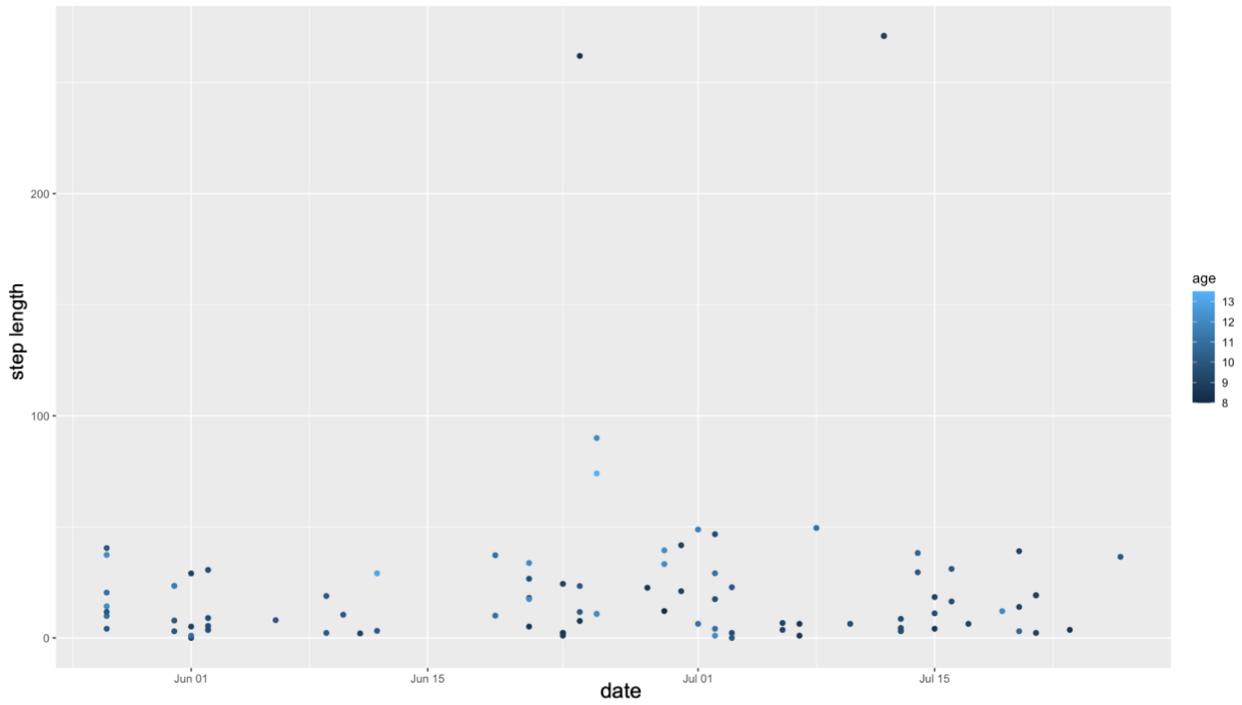


Figure 8 Step length over the breeding season with age of individuals

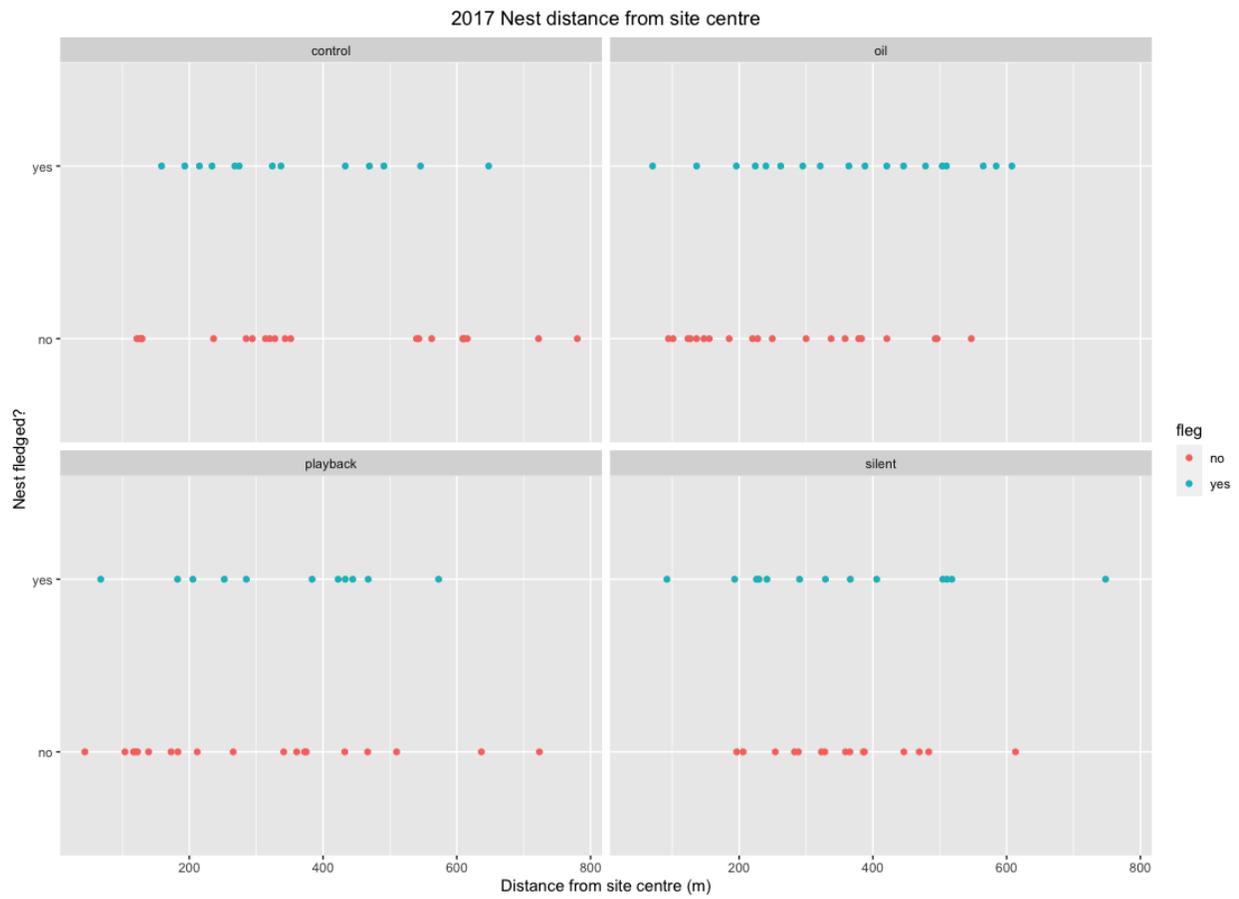


Figure 9 Distances between successful nests and site centers of control, active oil, active noise-broadcasting, and silent treatments in 2017. The y-axis shows a categorical variable of nest that fledged young, yes or no.

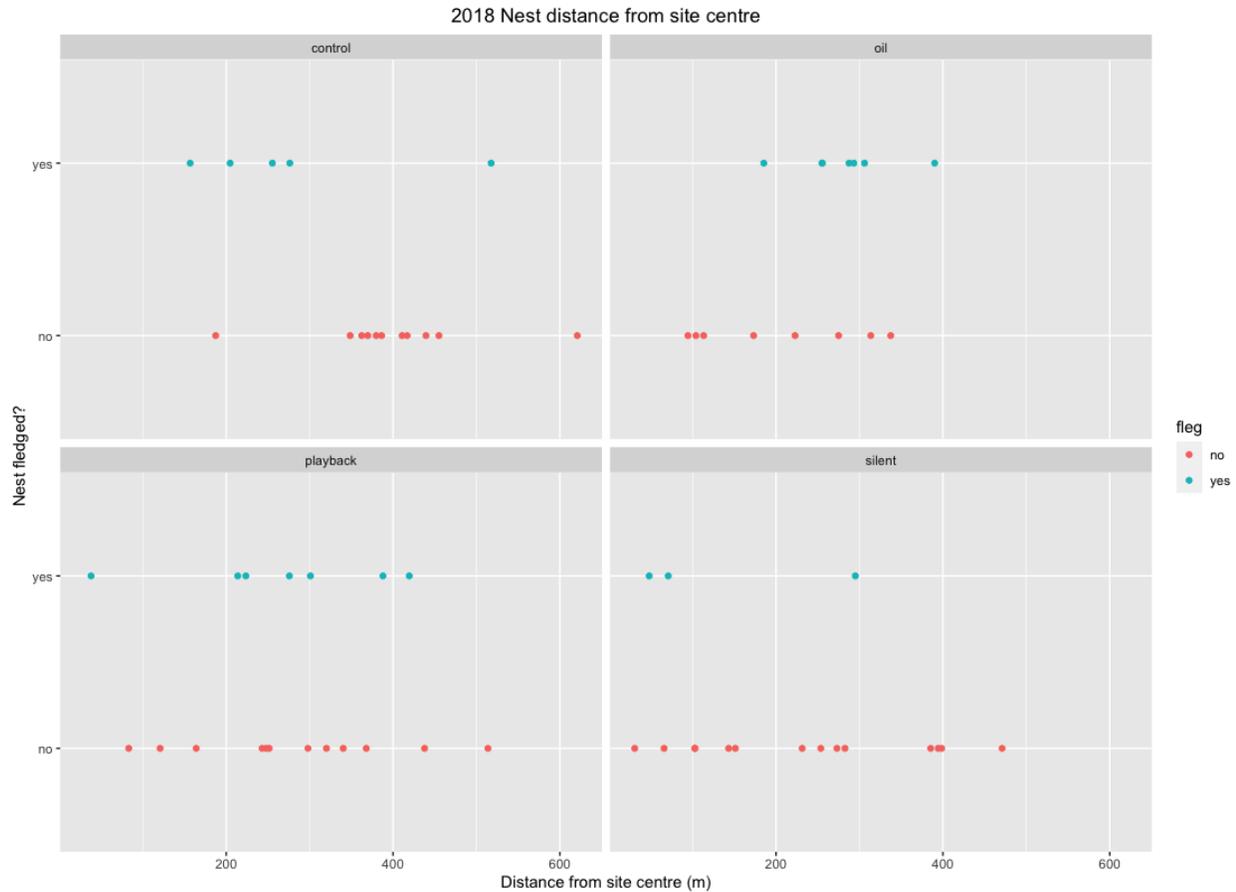


Figure 10 Distance from nests to site center of control, active oil, active noise-broadcasting, and silent treatments in 2018 and the outcome of fledging. This year had particularly high rates of nest failures, especially at silent treatment sites.

3.3 Effects of biological and climactic variables on fledgling survival

Of the variables included in models, days since fledging and brood size were the only significant predictors of fledgling survival in 2017 (Table 1). Survival increased the longer a fledgling was outside of the nest, although this was only significant in 2017. In 2017, but not 2018, larger brood size had a negative effect on fledgling survival (Table 1). Nestling weight at banding was a significant predictor of survival in 2018 alone (Table 1), such that nestlings that were heavier on day 8 had a higher likelihood of survival once they were fledglings.

In 2017, daily maximum and minimum temperatures were significant predictors of survival; however, they had opposite effects on fledgling survival (Table 2). Lower minimum temperatures were correlated with lower survival, whereas higher maximum temperatures

correlated with higher survival. Daily average precipitation had no effect on fledgling survival in either year, so it was not included in subsequent model.

Table 1 Biological models reporting estimate, (standard error), and p value. Observations in each year represent the total number of relocations for all individuals.

Biological Models		
	<i>Dependent variable:</i>	
	Survival	
	2017	2018
Intercept	-106.682 (162.536)	-682.670 (469.718)
	p = 0.512	p = 0.147
Fledgling age	0.051 (0.082)	1.038 (0.531)
	p = 0.537	p = 0.051*
log(days since fledging)	0.876 (0.379)	-1.588 (1.557)
	p = 0.021**	p = 0.308
Weight	-0.002 (0.102)	1.328 (0.469)
	p = 0.982	p = 0.005***
Brood size	-0.611 (0.253)	-0.074 (0.872)
	p = 0.016**	p = 0.933
Date	0.006 (0.009)	0.037 (0.026)
	p = 0.500	p = 0.161
Observations	659	75
Log Likelihood	-158.043	-22.013
Akaike Inf. Crit.	328.086	56.027
<i>Note:</i>	* p<0.1; ** p<0.05; *** p<0.01	

Table 2 Weather models reporting estimate, (standard error), and p value. Observations in each year represent the total number of relocations for all individuals.

Weather Models		
	<i>Dependent variable:</i>	
	Survival	
	2017	2018
Intercept	-80.849 (190.451)	-864.308 (538.221)
	p = 0.672	p = 0.109
Fledgling age	0.053 (0.090)	1.268 (0.565)
	p = 0.555	p = 0.025**
log(days since fledging)	1.028 (0.412)	-1.774 (1.793)
	p = 0.013**	p = 0.323
Weight	0.081 (0.103)	1.567 (0.491)
	p = 0.431	p = 0.002***
Date	0.005 (0.011)	0.047 (0.030)
	p = 0.677	p = 0.122
Minimum daily temperature	-0.098 (0.058)	0.488 (0.214)
	p = 0.091*	p = 0.023**
Maximum daily temperature	0.061 (0.043)	-0.160 (0.108)
	p = 0.156	p = 0.138
Total daily precipitation	-0.040 (0.040)	0.237 (0.252)
	p = 0.311	p = 0.348
Observations	616	61
Log Likelihood	-144.239	-18.041
Akaike Inf. Crit.	304.478	52.081

Note: * p<0.1; ** p<0.05; *** p<0.01

3.4 Effects of oil infrastructure and noise on fledgling survival

Fledgling survival on oil well and noise broadcasting sites did not differ from control sites in either year (Table 3). I investigated whether distance from each relocation point to site center had any effect on fledgling survival as noise amplitude is not constant over the site. Distance from each relocation point to site center did not significantly affect fledgling survival in either year (Figure 11, Figure 12). Sample size for tagged fledglings was evenly distributed among the four site types although nest success varied by site (Figure 6 and Figure 7).

Table 3 Management models reporting estimate, (standard error), and p value. Observations in each year represent the total number of relocations for all individuals.

Management Models		
	<i>Dependent variable:</i>	
	Survival	
	2017	2018
Intercept	-4.854 (4.737)	-7.598 (6.006)
	p = 0.306	p = 0.206
log(distance from site center	0.682 (0.736)	-1.055 (0.793)
	p = 0.355	p = 0.184
Oil site	3.234 (5.096)	0.515 (1.219)
	p = 0.526	p = 0.673
Noise site	4.724 (5.452)	-0.399 (1.180)
	p = 0.387	p = 0.736
Silent site	8.179 (5.593)	-0.197 (1.429)
	p = 0.144	p = 0.891
log(days since fledging)	1.142 (0.174)	1.954 (0.611)
	p = 0.000***	p = 0.002***
Weight	0.051 (0.093)	0.953 (0.361)
	p = 0.582	p = 0.009***
Minimum daily temperature	-0.127 (0.056)	0.218 (0.145)
	p = 0.024**	p = 0.134
Maximum daily temperature	0.086 (0.033)	-0.076 (0.081)
	p = 0.010***	p = 0.351
Distance to Oil*Oil site	-0.566 (0.875)	
	p = 0.518	
Distance to Noise*Noise site	-0.761 (0.946)	
	p = 0.421	
Distance to Silent*Silent site	-1.437 (0.955)	
	p = 0.133	
Observations	659	75
Log Likelihood	-154.575	-25.369
Akaike Inf. Crit.	333.149	68.737
<i>Note:</i>	*p<0.1; **p<0.05; ***p<0.01	

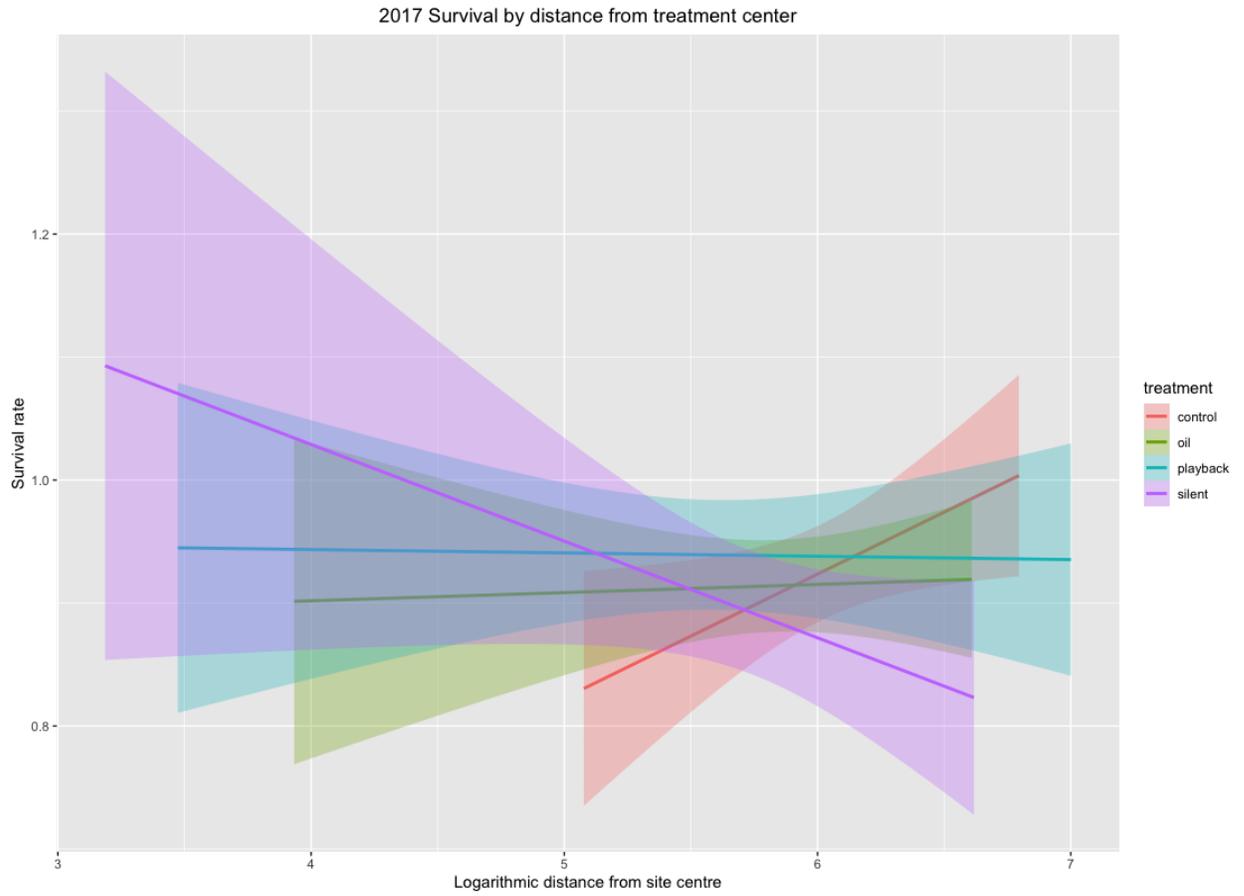


Figure 11 Fledgling survival in 2017 with increasing distance from site center. Distance fitted a logarithmic scale with survival based on AIC value, although this variable was not significant.

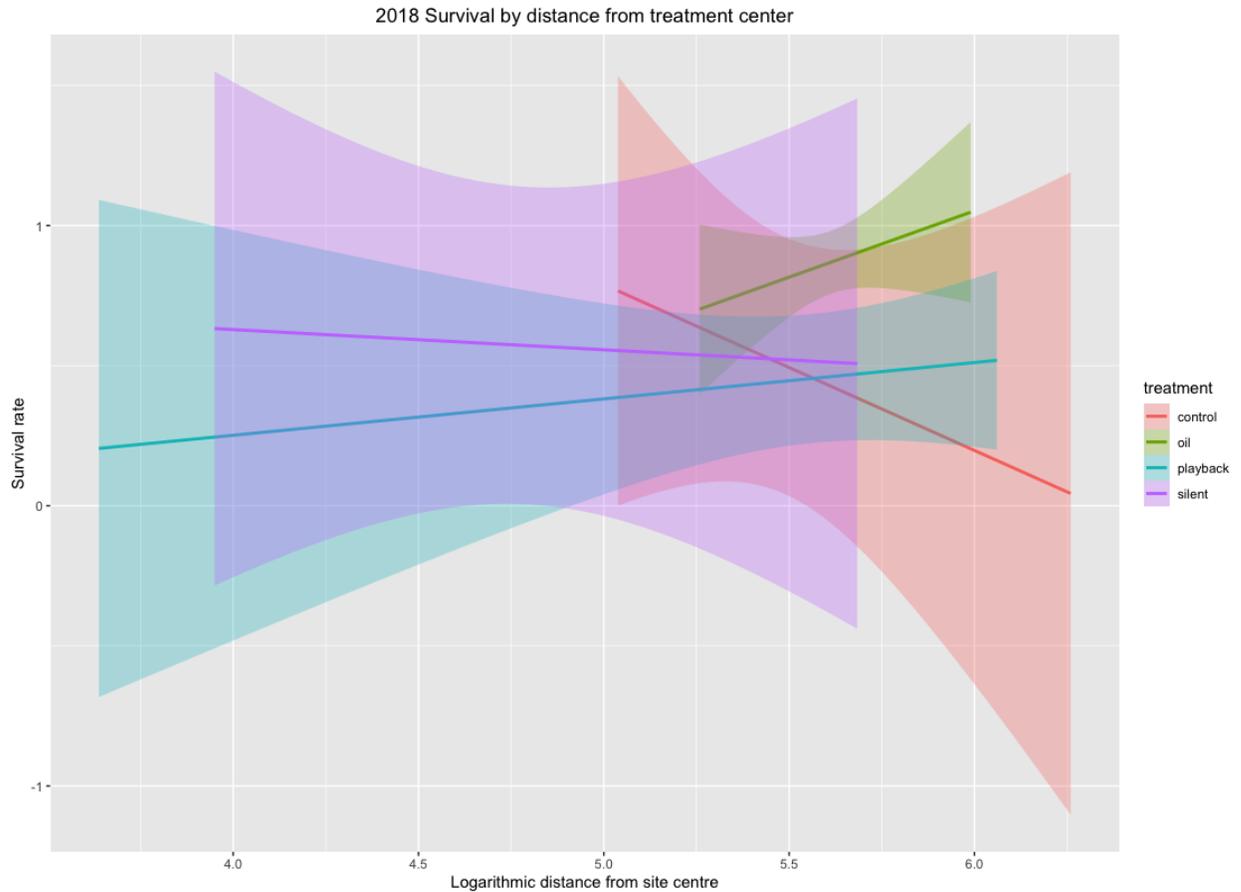


Figure 12 Fledgling survival in 2018 with increasing distance from site center. Distance fitted a logarithmic scale with survival based on AIC value, although this variable was not significant.

Chapter 4 Discussion

4.1 Adult survival of chestnut-collared longspurs and Baird’s sparrow

Adult survival for both chestnut-collared longspurs and Baird’s sparrow was close to 100% in both 2017 and 2018. Based on these results, adult mortality for these populations on their breeding grounds is likely not a current source of population decline for either species. Instead, survival during other life stages on the breeding grounds may be a source of population decline. Additionally, effects from the breeding ground may carry over to migration, the non-breeding ground, or subsequent breeding seasons. Increased parental care through longer post-fledging dependent periods of common kestrels (*Falco tinnunculus*) improved survival of young;

however, in the subsequent breeding season adult males had lower clutch sizes (López-Idiáquez et al. 2018) representing a potential reproductive cost in additional parental care. My results support the need for additional research into the full annual life cycle (Marra et al. 2015), specifically of the chestnut-collared longspur and Baird's sparrow. These species would benefit from research monitoring the same individuals over multiple years to investigate potential carryover effects.

4.2 Fledgling survival of chestnut-collared longspurs

I found no effect of oil wells or nest distance to oil wells on fledgling survival. Although infrastructure is associated with an increase in predator abundance through perch sites (Bylo et al. 2014), fledgling survival was not affected by nest proximity to infrastructure. These results could be due to parents of fledglings mediating any negative effects of oil wells. Alternatively, any local effects of energy infrastructure might be overwhelmed by other environmental factors that affect fledgling survival.

The results demonstrating no effect of oil extraction noise from oil wells on fledgling survival are unexpected. Noise has been shown to increase time to fledge and reduce nestling size (Injaian et al. 2018). Results from only studying the nestling stage would lead to interpreting these results as a negative effect of noise. My study demonstrates that negative effects of noise may not be present in all life stages and highlights the importance of studying multiple life stages to assess productivity. There have been no other studies investigating fledgling survival in the context of noise. Studies focusing on effects of noise on nestlings may easily extrapolate negative consequences of exposure to noise when it is not the case. The lack of an observable effect of noise might occur because acoustic communication is more important in some life stages but not others. Alternatively, birds might be able to compensate behaviourally for the presence of noise, or operating noise from oil wells might be below the biologically influential threshold for this species. There may also be additional methods of communication or signaling between fledglings and adults that allow compensation for any impacts of noise (Rosa and Koper 2018). Because of noise, nestlings may be unable to perceive risk in the nest, therefore resulting in a decreased stress response (Heathcote 2019). However, if noise continued to reduce detection

of risks once in the post-fledging phase I would expect to see a reduced survival from active oil wells and the noise broadcasting sites and this was not the case.

The absence of an effect of noise or active oil well treatments on fledgling survival may be because the maternal parent mediates negative effects from noise on nestlings. Results from Heathcote (2019) also support the hypothesis of maternal mediation in stressful environments. This could indicate that other factors have a strong effect on fledgling survival and that negative effects of infrastructure and noise are mitigated by parents. Parental mediation may continue through the post-fledging stage in disturbed habitat. Grassland songbirds might also be relatively tolerant of environmental noise, having adapted to wind that creates a natural, noisy environment due to the lack of physical barriers in the grassland (Curry et al. 2017). Species using this habitat during the breeding season would have adapted to the level of noise produced by wind (Curry et al. 2017). Although noise may not directly influence survival, it may affect additional behaviours such as movement of fledglings (Carey unpubl. data). Chestnut-collared longspurs consistently reduced parental care in the presence of conventional oil and gas infrastructure and industrial noise (Ng et al. 2019); however, I did not find an effect of noise on fledgling survival. This suggests that parents are still able to communicate with fledglings after leaving the nest regardless of the presence of noise, or that communication is not a determining factor in survival.

Fledging age directly impacts flight ability (Martin et al. 2018). I expected age to be a significant predictor of fledgling survival because as fledglings age and develop, their ability to thermoregulate increases, their ability to move increases, and their ability to forage increases. The physiological development of endothermy increases with body mass and age (Pereyra and Morton 2001) Thermoregulation allows fledglings to avoid the negative effects of exposure to weather and climatic variables. By moving to covered areas, fledgling grassland songbirds reduce chance of mortality from exposure (Giovanni et al. 2015). Becoming more agile and capable of moving longer distances allows fledglings to escape predators; however, increased movement can also attract predators.

Heavier longspur nestlings also had higher rates of survival. Heavier nestlings might be more tolerant of being fed less after leaving the nest. Heavier nestlings also have lower corticosterone, which may be beneficial in terms of post-fledging survival (Heathcote 2019). Individuals that

experience lower stress responses earlier in life have greater reproductive success and survival (Blas et al. 2007). Lower corticosterone levels may result in less movement post-fledging which could reduce detection by predators. Chronic exposure to corticosterone can have negative impacts on growth and development (Sapolsky et al. 2000; Kitaysky et al. 2003; McEwen and Wingfield 2003) and increases the mobilization of energy which depletes glucose stores (McEwen and Wingfield 2003) which would carry over into the post-fledging stage. However, I found no effect of oil infrastructure or oil extraction noise on mass or the length of the ninth primary feather on nestlings, thus demonstrating that growth rates were similar in the presence and absence of oil extraction noise.

This could have a negative effect on adult females, specifically on survival due to increased stress and energy mediating negative effects. However, survival for adults was almost 100% throughout the breeding season demonstrating that any negative effects incurred by females do not have immediate impacts, although it could affect migration and long-term survival. The positive effect of nestling weight on fledgling survival demonstrates the importance of development in the nest before entering the post-fledging period. I found that larger broods had decreased fledgling survival. This could be because of reduced resources limiting growth rates and weight gain while in the nest.

Grassland songbirds are particularly sensitive to environmental conditions because of the high level of vulnerability of juveniles after leaving the nest. I found effects of temperature on fledgling survival in 2017 and no effect of temperature in 2018. It is likely that there are different mechanisms that would be affected by minimum or maximum daily temperatures. Fledgling survival of great tits (*Parus major*) was affected by higher temperatures (Greño et al. 2008). Higher temperatures reduce body size in zebra finches (Andrew et al. 2017) and mass gain in Cape rockjumpers (*Chaetops frenatus*) (Oswald et al. 2021). Temperature has been shown to have a negative effect on nestling growth in older nestlings but not younger ones (Siikamäki 1996). Nestlings have temperature thresholds and when temperatures exceed those thresholds physiological/morphological effects can have an increased impact (Cunningham et al. 2013). Chestnut-collared longspur nestling thresholds are unknown and represent an area of research that could benefit the species considering the likelihood of increasing temperatures due to climate change.

These results could suggest that longspurs may be resilient to climate change. However, climate change is expected to change many environmental conditions other than mean and maximum temperatures. Species that overwinter in Mexico, both chestnut-collared longspurs and Baird's sparrow, are likely to be vulnerable to increased drought (Culp et al. 2017). Climate change is likely to impact the amount of precipitation within the grassland system that occurs over winter and during the spring and summer months (Skagen and Adams 2012). Precipitation in the Great Plains can have direct effects on nests and fledglings as large, daily rain events that can cause drowning or indirectly by seasonal precipitation that affects vegetation and food resources (Skagen and Adams 2012). In June of 2018, a large storm flooded five chestnut-collared nests that were ready to be banded and radio-tagged and all nestlings drowned as a result of the rain. In addition, climate change is likely to intensify weather events, like storms and periods of drought, and this may pose a risk to grassland songbirds because extreme weather events may drive this system, and survival in general (Wiens 1977). Extreme precipitation influences productivity in the environment, including predators. Wet-dry cycles also affect the population cycles of ground squirrels and other key predators, which can lead to decreased productivity (Rotenberry and Wiens 1989). Winter precipitation levels also affect reproductive output (clutch size, brood size, fledgling survival) of Brewer's sparrow (Rotenberry and Wiens 1989). This demonstrates that grassland ecosystems present a complex system of cycles and links between predators, prey, and climatic events. Because there are other confounding factors like ground saturation, vegetation changes, variation in soil, and erosion, it is difficult to determine how climate change will affect ground nesting birds.

The habitat surrounding the nest could have a strong effect on fledgling survival due to limited mobility after leaving the nest likely being an important factor. To facilitate foraging, fledglings need access to bare ground or short, sparse vegetation until individuals are capable of longer flight. Conversely, for protection from environmental conditions and predators, fledglings would require vegetation cover through either shrubs or taller, dense vegetation. These two reasons demonstrate the necessity and benefit of heterogenous vegetation in mixed grass prairies. Vegetation homogeneity has occurred from an increase in crested wheatgrass (Dale et al 2009) and other introduced species that are used to cover buried pipelines and other infrastructure. Certain life stages such as the post-fledging dependent period are likely to experience negative

effects from homogenous vegetation. Surprisingly, though, a review of 31 studies suggested that breeding habitat vegetation had no effect on post-fledging survival (Cox et al. 2014). Fledglings would have adapted to navigating in mixed vegetation types with portions of bare ground. I witnessed fledglings using areas of dried wetland, similarly to how adult chestnut-collared longspurs typically use burned ground for easier foraging, and walking, especially for fledglings that do not yet fly.

Chapter 5 Summary and Management Implications

Active oil wells and noise each had no significant effect on post-fledging survival. This indicates that exposure to noise and oil infrastructure are not primary factors influencing survival during the post-fledging period or that other factors are more influential during this period. Adult survival for both Baird's sparrow and chestnut-collared longspur was close to 100% throughout the breeding season. Estimating survival during the first few weeks post-fledging can provide important life stage estimates.

My results suggest that operating well noise should not be the focus of management decisions or mitigation effort. Evidence of an effect, or lack thereof, of noise is important because mitigation actions to reduce noise from oil wells differ from actions to reduce impacts from physical infrastructure. For example, noise mitigation includes sound barriers, while mitigation for effects from physical infrastructure might include burying infrastructure or removing it from the landscape. Current management strategies vary for adult birds, migrating birds, and nests. Nests can be easily protected through policy guidelines because nests can be located and marked and are immobile. Juvenile birds in the post-fledging phase would require protection that covers larger amounts of land and would require more resources to track and monitor success.

Policy makers need to broaden justification of management decisions to include species populations that are not yet at risk. Unfortunately, conservation managers are dependent on policy to defend and protect endangered species. This creates a negative feedback loop and leads to species only receiving the most protection when their population is at the most risk of extinction. Incorporating perspectives from conservation biology and traditional ecological

knowledge yields conservation programs that are more effective for species protection (Drew and Henne 2006).

Federal funding from the COVID-19 Economic Response Plan includes \$1.7 billion toward cleaning up orphaned oil and gas wells in British Columbia, Alberta, and Saskatchewan (Kemball 2020). In Alberta, funding from the federal government went directly to the Orphan Wells Association and to the provincial government. The provincial government created a site rehabilitation program (<https://www.alberta.ca/site-rehabilitation-program-overview.aspx>) and a liability management framework. This will reduce the amount of infrastructure on the landscape through IOGC's [environmental requirements](#), including Remediation Action Plan and Reclamation and Surrender processes. Contract holders are responsible for returning the land to the First Nations in a state that is equivalent to the pre-disturbance surrounding land use. The contract holder is responsible for contacting the First Nation to discuss end land use and reclamation goals. For example, if there are any facilities to be left in place or if there is a change in land use, a First Nation Band Council Resolution will be required to be secured by the contract holder. This demonstrates one opportunity where the Canadian federal government, provincial government and industry should collaborate with First Nations to restore destroyed prairie habitat. Additionally, land reclamation can also contribute to carbon sequestration (Wang et al 2014). Although my research suggests that fledgling survival is independent of the presence of infrastructure, removal of infrastructure and remediation will help adult Baird's sparrow and chestnut-collared longspurs (Bernath-Plaisted 2015; Nenninger 2016).

However, removal of oil and gas infrastructure does not include the removal of roads and invasive vegetation, which will continue to fragment and degrade habitat. This is an issue because roads reduce parental care in chestnut-collared longspurs (Ng 2018). The landscape needs to be managed at a broader scale to include roads. Grassland needs to be restored to reduce fragmentation and create larger tracts of native grassland habitat.

Reducing habitat fragmentation could be achieved through the removal of roads and then subsequently protecting remaining areas of grasslands in North America. Samson et al. (2004) summarize part of the historical context of the Great Plains and the colossal mistake of selling land for agricultural conversion over the past decade and a half. The article also points out the

inefficiencies of government agencies working toward conservation action. Approximately 11% of the mixed grassland ecoregion is conserved in Canada (*Ecoregional summary – Mixed Grassland* 2019) and includes federal and provincial pastures, land managed by Nature Conservancy of Canada, Ducks Unlimited, and Key Biodiversity Areas. The Migratory Birds Convention Act was established in 1917 and sought to protect species from hunting which was the main cause of decline at the time. From section 5.1, the Migratory Birds Convention Act prohibits depositing harmful substances and any environmental take; however, disturbances such as habitat fragmentation and noise production altering the soundscape affect the same species but are not covered within the Act. The Species At Risk act became legal in 2002 and represented a huge stride toward protecting endangered and threatened species in Canada.

Management for specific species can be a beneficial strategy although managing specific areas would help to conserve multiple species. Increased land protection would be beneficial to grassland songbirds as a broad form of management, specifically for chestnut-collared longspurs and Baird's sparrows. Protected parcels of land are also beneficial during non-migratory phases of these species' life cycles compared to the difficulty in protecting habitat over full migration routes. The ultimate goal should be to convert parcels of land back to native prairie instead of managing for specific species and only as a reaction to declining population sizes. One way to achieve this in the North American grasslands is by returning to grazing and management practices and capacities that were once held by Indigenous peoples (Kemp and Michalk 2007). Prescribed burns are currently practiced management actions and were recorded in the late 18th century being used by Piegan, Blood, and Siksika groups (Fidler 1793, Lewis 1985). Additionally, Indigenous people need to have increased access to land and management decisions. This has already started through the creation of Indigenous Protected Areas and from land sharing networks where farmers and landowners can provide access to Indigenous people for traditional harvesting (<https://treatylandsharingnetwork.ca/about/>). Indigenous lands in Canada currently support more threatened species than protected areas (Schuster et al. 2019). Indigenous protected areas can be beneficial especially due to the high degree of overlap between Indigenous lands and the ranges of threatened species (Artelle et al. 2019). Co-management practices do not provide much improvement in the way of reconciliation but Indigenous protected areas do (Finegan 2018). Indigenous protected areas would also be more aligned with the precautionary principle.

Precautionary conservation should be prioritized to preserve biodiversity (Myers 1993, Cooney 2004). Precautionary actions are often resisted by industry because they are more costly up front even though reactionary solutions to habitat destruction and the series of consequences this causes are more expensive long term. There is a lack of motivation and willingness to take action before scientific evidence demonstrates environmental harm and destruction. However, long term effects on habitats, society, and economy are often not considered, especially with regards to the extraction of natural resources that poses the risk of irreversible damage and destruction.

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