

**The Effects of Oil and Gas Development On Songbirds of
the Mixed-grass Prairie:
Nesting Success and Identification of Nest Predators**

by
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ABSTRACT

Over the past century, populations of North American grassland songbirds have declined sharply as a consequence of habitat destruction. Alberta's mixed-grass prairie constitutes Canada's largest remaining tract of native grassland. However, this region has recently undergone a rapid expansion of conventional oil and natural gas development, and few studies have documented its effects on songbird nesting success. During the 2012-2014 breeding seasons, I monitored 813 nests of grassland songbirds located at sites that varied with respect to presence/absence, distance from, and types of oil and gas infrastructure (pump jacks, screw pumps, compressor stations) and gravel roads. Nest survival was significantly lower at infrastructure sites relative to controls for both Savannah sparrow and vesper sparrow. Additionally, vesper sparrow nest density was greater within 100 m of structures. These findings suggest that habitat disturbance caused by infrastructure may result in increased frequencies of nest predation at multiple spatial scales.

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CHAPTER 1: INTRODUCTION

1.1 Background

Over the past half-century, recorded declines in population sizes of grassland songbirds have been more severe than those of any other North American habitat group (Herkert 1995; Sauer et al. 2014). In Canada, grassland songbird populations have decreased by an estimated 40% since 1970 (NABCI 2012). These declines are primarily attributed to conversion of grassland to cropland (e.g., Peterjohn and Sauer 1999; Brennan and Kulvesky 2005; Askins et al. 2007), which has resulted in the almost complete destruction of the Northern Tallgrass prairie and reduced mixed-grass prairie to 29% of its historic extent (Samson et al. 2004). Today, only 25-30% of Canada's original grasslands exist (Gauthier and Wiken 2003). Furthermore, much remaining grassland habitat is highly degraded (Askins et al. 2007). Habitat fragmentation (e.g., Herkert et al. 1994; Davis 2004; Winter et al. 2006), woody encroachment (e.g., Coppedge 2001; Grant et al. 2004; Ellison et al. 2013), and the disruption of historical fire and grazing regimes (e.g., Samson et al. 2004; Fuhlendorf et al. 2006) continue to reduce habitat quality for grassland songbirds. In Canada, grassland habitat degradation is further exacerbated by the expansion of oil and natural gas extraction across the mixed-grass prairie (Askins et al. 2007; Nasen et al. 2011; Riley et al. 2012).

Western North America has been subject to rapid energy development over the past century, and it is estimated that 21% of its grasslands are currently affected by energy development activities, the second highest proportion of any terrestrial biome (Copeland et al. 2011). Alberta possesses the world's third largest crude oil reserves (Government of Alberta 2015), and is the world's second largest exporter of natural gas

(Riley et al. 2012). Combined, conventional oil and natural gas extraction account for the largest surface footprint of any energy source in North America, and Alberta's mixed-grass prairie is located in the epicenter of this activity (Copeland et al. 2011). Alberta contains over 300,000 oil and natural gas wells and 350,000 km of pipeline, occupying a total land area of approximately 1,000,000 ha (Government of Alberta 2014). Oil and gas extraction have already resulted in substantial increases in fragmentation and surface disturbance across the mixed-grass prairie region (Nasen et al. 2011), and in Alberta, land continues to be developed for conventional oil and gas extraction at approximately 120 ha/day (Government of Alberta 2014).

Although the lease size of an individual oil well is typically only 1 ha (Van Wilgenburg et al. 2013), wells are numerous in regions of high oil and gas activity, and are often associated with extensive access road networks (Schneider et al. 2003; Thompson et al. 2006; Nasen et al. 2011; Allred et al. 2015). It is estimated that for each well built, an additional 100 ha of land is disturbed by the construction of access roads (Riley et al. 2012). These linear features dissect the landscape, creating a high proportion of edges (Bayne and Dale 2011, Nasen et al. 2011), and potentially resulting in edge effects for nesting songbirds (Riley et al. 2012). Edge effects occur at the junction of two habitat types where structural heterogeneity produces increased predator diversity and activity (Gates and Gysel 1978). Grassland songbirds nesting near edges frequently experience decreased nesting success (e.g., Johnson and Temple 1990; Winter et al. 2000) and increased brood parasitism by brown-headed cowbird (*Molothrus ater*) (e.g., Patten et al. 2006). Energy infrastructure may also provide nesting substrate and perches for raptors and corvids, as well as anthropogenic food subsidies for generalist predators

(Liebezeit et al. 2009; Bui et al. 2010). Oil and natural gas extraction is also a source of chronic anthropogenic noise (Habib et al. 2007; Bayne et al. 2008). There is increasing concern about the numerous negative effects of chronic noise from traffic and industrial sources on wildlife (Barber et al. 2010). Noise from energy infrastructure may negatively affect songbird reproductive success by impairing communication (Habib et al. 2007), interfering with vigilance behavior and predator detection (Barber et al. 2010, Chan et al. 2010), and triggering stress responses that can reduce nestling provisioning rates (Schroeder et al. 2012). Additionally, active oil infrastructure sites are typically visited daily by personnel, resulting in increased human presence and vehicle traffic, which may affect breeding songbirds (Reijnen and Foppen 2006).

Energy infrastructure has the potential to alter ecosystems in many ways that may impact songbird nesting success. Predators respond to changes in habitat at multiple scales (Klug et al. 2009), predator response to habitat characteristics varies by species (Ellison et al. 2013), and different edge types may have different effects (Ries et al. 2004). Energy development fragments the landscape (Nasen et al. 2011), alters habitat structure through the introduction of perches (Bui et al. 2010) and exotic vegetation (Nasen et al. 2011; Ludlow et al. 2015), and increases chronic noise levels (Bayne et al. 2008; Blickley et al. 2012). However, few studies have directly examined the effects of oil and natural gas development on reproduction and nest survival in grassland songbirds.

1.2 Problem statement

Alberta's mixed-grass prairies constitute an estimated 43% of Canada's remaining grassland (Gauthier and Wiken 2003), making this a critical area for high concentrations of more than 8 species of grassland-obligate birds with limited distributions (Sauer et al.

2014), including non-passerines such as the endangered burrowing owl (*Athene cunicularia*) (COSEWIC 2006), and long-billed curlew (*Numenius americanus*), a species of special concern in Canada (COSEWIC 2011). Songbird species that commonly breed in mixed-grass prairie include Savannah sparrow (*Passerculus sandwichensis*), vesper sparrow (*Pooecetes gramineus*), and western meadowlark (*Sturnella neglecta*), all of which are currently declining across North America (Sauer et al. 2014). Additionally, the mixed-grass prairie is important habitat for two threatened species in Canada, chestnut-collared longspur (*Calcarius ornatus*) (COSEWIC 2009) and Sprague's pipit (*Anthus spragueii*) (COSEWIC 2010). These two species have experienced total population declines of approximately 93% and 83%, respectively, since the 1960s (COSEWIC 2009, 2010). Though concentrations of oil and natural gas infrastructure are high in the mixed-grass prairie ecosystem (Askins et al. 2007; Copeland et al. 2011; Government of Alberta 2014), little is known about the effects of energy development on these species.

Existing research suggests that oil and natural gas extraction have a variable, but predominantly negative effect on avian species (Bayne and Dale 2011; Riley et al. 2012). A number of studies have recorded decreased avian abundance near energy infrastructure, suggesting that birds avoid these disturbances on the landscape (e.g., Doherty et al. 2008; Carpenter et al. 2010; Gilbert and Chalfoun 2011). Additionally, there is evidence that songbirds avoid roads and traffic noise (Sutter et al. 2000; Inglefinger and Anderson 2004; McClure et al. 2013), as well as seismic lines (Machtans 2006). In grasslands, the effects of oil and natural gas activity on avian abundances vary among species and studies. Nonetheless, there is some consensus that the Sprague's pipit exhibits decreased

abundance near oil and gas wells (Linnen 2008; Hamilton et al. 2011). Both chestnut-collared longspur and Sprague's pipit appear to be edge sensitive (Koper et al. 2009; Sliwinski and Koper 2012), and it is likely that habitat fragmented by infrastructure is less suitable for them. Most studies have focused on avian abundance responses rather than reproductive success (e.g., Linnen 2008; Hamilton et al. 2011, Gilbert and Chalfoun 2011; Kalyn-Bogard and Davis 2014). Yet, songbird density may not be an accurate indicator of habitat quality (Van Horne 1983) or nesting success (Vickery et al. 1992). This is especially true in areas of high anthropogenic disturbance where birds may fail to recognize ecological traps (Bock and Jones 2004).

Only a handful of studies have examined the effects of oil and gas infrastructure on reproductive success in avian species, but these findings have been inconsistent. For example, greater sage-grouse (*Centrocercus urophasianus*) experience increased chick mortality near energy infrastructure (Aldridge and Boyce 2007), and Sprague's pipit nesting success is reduced in vegetation altered by oil and natural gas development (Ludlow et al. 2015). Conversely, the nesting success of lesser prairie-chicken (*Tympanuchus pallidicinctus*) is unaffected by proximity to infrastructure (Pitman et al. 2005). Furthermore, Ball et al. (2009) found that ovenbirds (*Seiurus aurocapilla*) in the boreal forest nesting near forest edges associated with roads and pipelines experience improved nesting success. Given these variable findings and the extent of habitat affected by conventional oil and natural gas development across North America (Copeland et al. 2011), further research is needed to determine whether there is a link between these activities and the reproductive success of declining grassland songbirds.

1.3 Rationale

An understanding of factors driving reproductive success is critical to the conservation of species (Martin 1992). Predation is the primary cause of nest failure in grassland songbirds (Gates and Gysel 1978; Martin 1993a, 1995), including those nesting in northern mixed-grass prairie (e.g., Davis 2003; Jones et al. 2010, Kerns et al. 2010; Ludlow et al. 2014), and the interaction between habitat and predators is an important factor limiting population growth in avian species (Martin 1993b).

Knowledge of common predators and specific behavior patterns is an important factor in successful management and conservation (Thompson and Ribic 2012). Determining if and how oil and natural gas infrastructure affect predation rates and alter the predator community can assist land managers in mitigating the ecological effects of existing and further developments. However, traditional methods of assessing nest predation are no longer considered reliable, as research has shown that predators cannot be inferred from evidence left at the nest (Larivière 1999; Pietz and Granfors 2000; Staller et al. 2005). Video surveillance offers an alternative method that can definitively identify predators and avoid the biases of observer and inference-based approaches (Thompson et al. 1999; Pietz and Granfors 2000; Pietz et al. 2012). Camera studies are now well accepted and have been successfully conducted in forest, shrubland, and grassland systems (e.g., Pietz and Granfors 2000; Thompson and Burhans 2003, Ribic and Renfrew 2003; Schaefer 2004; Renfrew et al. 2005; Ellison et al. 2013). However, only one study thus far has attempted to identify predators in Canada's northern mixed-grass prairie using video-surveillance techniques (Davis et al. 2012).

1.4 Objectives

The object of this study was to determine whether structures and roads associated with conventional oil and natural gas extraction impact the reproductive success of songbirds nesting in Alberta's mixed-grass prairie. This study included four site types: 1) pumpjacks and 2) progressive cavity pumps, or screw pumps, both associated with oil extraction; 3) compressor stations associated with natural gas extraction; and 4) control sites that contained no extraction structures or compressor stations. Although this study is primarily concerned with the effects oil infrastructure on nesting songbirds, I chose to include natural gas compressor stations in the study because they have a large footprint and produce a high volume of noise (Koper et al. 2015), and, therefore, are one of the most likely structures to produce an effect. This study was also intended to assess the effects of noise and disturbances, such as mechanical noise, traffic, and human visitation associated with energy infrastructure. Sites that were in use and producing noise during the breeding season were designated as active and sites that were turned off and not producing noise or receiving site visits from company personnel were considered inactive.

To meet these objectives I:

- 1) Analyzed nesting success as a function of (a) site type, (b) site activity, (c) distance to nearest road and nearest structure, and (d) vegetation characteristics.
- 2) Deployed 24-hour video surveillance systems on active songbird nests to document the identity and behavior of nest predators.

1.5 Hypotheses and predictions

1. *Hypothesis:* Oil wells and associated infrastructure may concentrate predator activity by attracting or providing food and/or habitat subsidies to certain predators.

Prediction 1: Frequency of nest predation will be elevated at infrastructure sites compared with control sites

Prediction 2: Frequency of nest predation will increase with decreasing proximity to infrastructure features

2. *Hypothesis:* Chronic anthropogenic noise and disturbance associated with oil and gas structures may reduce efficacy of adult songbirds to communicate, detect predators, or provision young.

Prediction 1: Nesting success of songbirds will be lower at active infrastructure sites than at either controls or inactive sites.

Prediction 2: Nesting success will be lowest at the loudest sites, such as those with compressor stations or those that are generator powered, as these structures produce the greatest volume of noise.

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CHAPTER 2: LITERATURE REVIEW

2.1 Grassland songbird population declines and loss of grassland habitat

Population declines—Historically, the conservation of grassland songbirds has been neglected relative to more frequently studied forest and wetland species (Herkert 1995). However, the persistent decline of grassland songbirds over the latter half of the 20th century is well recognized (e.g., Knopf 1994; Peterjohn and Sauer 1999; Askins et al. 2007; NABCI 2012). Recent analysis of Breeding Bird Survey (BBS) data, spanning the years of 1966-2013, found that 75% of grassland songbirds in North America are presently in decline (Sauer et al. 2014). Over 50% of these species are considered at risk in Canada (NABCI), and both chestnut-collared longspur (*Calcarius ornatus*) and Sprague's pipit (*Anthus spragueii*) are threatened (COSEWIC 2009, 2010). In Alberta, these two species are currently declining at average per annum rates of 6.6% and 3.8%, respectively. Observed declines in grassland songbirds are most likely explained as a response to massive habitat losses over the past century (Knopf 1994; Herkert 1995).

Status of grassland ecosystems—Grasslands are among the most endangered ecosystems on Earth (Hoekstra et al. 2005). Worldwide, 45% of historic grassland habitat has been converted and less than 5% of the world's remaining grasslands are currently protected (Hoekstra et al. 2005). In the United States, 55% of critically endangered ecosystems, (i.e., those that have experienced 98% or greater reductions from historical extent), are classified as grassland or savanna (Noss et al. 1995). In Canada, nearly 75% of historic grasslands have been destroyed (Gauthier and Wiken 2003). Virtually no Tallgrass prairie remains in North America today, while the mixed-grass and Shortgrass

prairies have been reduced by approximately 70% and 50%, respectively (Samson et al. 2004).

Agricultural land conversion—There is little doubt that row crop agriculture has been the primary source of grassland habitat destruction over the past two centuries (e.g., Knopf 1994; Peterjohn and Sauer 1999; Samson et al. 2004; Brennan and Kulvesky 2005; Askins et al. 2007). European settlement of the North American prairies began in 1830 (Samson and Knopf 1994), and was accelerated by the Homestead Act of 1862 in the United States and the Land Dominion act of 1872 in Canada (Samson et al. 2004). During this time, governments of both countries actively encouraged the conversion of all fertile land to farmland (Askins et al. 2007). The eastern Tallgrass prairies were well-suited for crops and consequently they have been almost completely developed for crop production (Knopf 1994; Samson et al. 2004). For example, between 1810 and 1850, farming reduced Illinois' prairies by 75% (Herkert 1991). Today, 80% of Illinois' landmass is occupied by cropland (Iverson 1988), and only 4-square miles of Illinois' native Tallgrass prairie remain (Mlot 1990). Similarly, <1% of the original prairies remain intact in other eastern states and provinces where conditions are optimal for crop production (Samson and Knopf 1994). Conversely, agricultural conversion of Shortgrass and mixed-grass prairie has been less severe, though still significant (Samson et al. 2004), most likely tempered by arid climate conditions and nutrient poor soils in these regions (Askins et al. 2007; Bailey et al. 2010).

There is a strong link between the growth of row crop agriculture and the decline of grassland songbirds; land use change explains 25-30% of the variation in population trends among North America's grassland and shrubland songbirds (Murphy et al. 2003).

For most species, population size is positively correlated with rangeland and negatively correlated with cover crop (Murphy et al 2003). In Alberta, the tilling of mixed-grass prairie results in the loss of all grassland birds except for horned lark (*Eremophila alpestris*) (Owen and Myres 1973). Similarly, grassland songbird abundances decline in planted grasslands and cattle forage, such as crested wheatgrass (*Agropyron cristatum*) (McMaster et al. 2005) and hayfields (Dale et al. 1997; Sutter and Brigham 1998). In regions where the conversion of grasslands to cropland has been extensive and well documented, such as Illinois, the positive relationship between the growth of row crop agriculture and songbird decline is especially evident (Herkert 1991; Warner 1994).

Additionally, the intensification of agricultural practices on existing cultivated pasture has negatively affected species that have successfully transitioned to breeding in planted grasslands (Rodenhouse et al. 1993; Herkert 1995; Askins et al. 2007). The increasing prevalence of large monoculture farming operations has led to frequent mechanical mowing, tilling, and harvesting of planted grasslands during the breeding season (Tews et al. 2013). This practice is known to destroy approximately 50% of ground nests (Davis et al. 1999), and to kill juvenile birds (Bollinger et al. 1990) and reduce nesting productivity (Frawley and Best 1991; Dale et al. 1997). It is estimated that agricultural mowing and tilling kill 941,000 Savannah sparrows (*Passerculus sandwichensis*) annually (Tews et al. 2013).

Today, 52% of the United States' landmass, and 11% of Canada's, are devoted to cropland (Rodenhouse et al. 1993). The loss of grassland habitat to cropland has slowed since the 1980s (Knopf 1994), and in some regions of the United States it has been fractionally reversed as a result of the Conservation Reserve Program (Johnson and

Schwartz 1993). Similarly, Canada's Permanent Cover Program has also restored relatively minor areas of cropland back to grassland-like habitat (McMaster and Davis 2001). Nonetheless, grassland habitat losses to row crop agriculture continue; since 1997, 47.4 million acres of grassland have been converted to cropland, and 50% of this conversion has occurred in the Great Plains region (Turner 2013).

2.2 Degradation of mixed-grass prairie habitat

Woody encroachment and afforestation—Although the mixed-grass prairies of Alberta have suffered relatively little destruction compared to other grassland regions in Canada (Samson and Knopf 1994; Samson et al. 2004), much of this habitat is highly degraded (Askins et al. 2007). The encroachment of woody vegetation on the Great Plains has contributed to this reduction in habitat quality, as fire suppression, altered hydrology, and human cultural preferences have all favored the introduction of trees and shrubs into grasslands (Knopf 1994; Brennan and Kulvesky 2005). During the settlement of the prairies, homesteaders were sometimes granted land on the condition that they would plant up to 40 acres of trees and shrubs (Baer 1989). Direct planting of trees by farmers and settlers has continued and the result is a network of forested shelterbelts estimated to occupy 3% of the Great Plains (Knopf 1994).

Grassland songbirds respond negatively to woody vegetation (Coppedge et al. 2001; Grant et al. 2004; Thompson et al. 2014), and trees and shrubs facilitate the invasion of potential competitors from forest and shrubland systems (Knopf 1986; Coppedge et al. 2001). Furthermore, woody vegetation alters predator communities (Ellison et al. 2013), and may increase the foraging efficiency of some predators (Klug et al. 2010). Songbirds nesting near trees and shrubs experience both elevated predation

(e.g. Johnson and Temple 1990; Winter et al. 2000) and brood parasitism (e.g. Hauber and Russo 2000; Patten et al. 2011). Grassland songbirds appear to avoid nesting near wooded edges for this reason (O’leary and Nyberg 2000, Winter et al. 2000).

Disruption of fire and grazing—Historically, grasses were maintained as the dominant vegetation on the Great Plains by the interaction of frequent wildfires and grazing by native, free-roaming bison (*Bison bison*) and prairie dogs (*Cynomys* spp.) (Knopf 1994; Samson et al. 2004; Brennan and Kulvesky 2005; Askins et al. 2007). However, human activities have disrupted both of these processes. Prior to fire suppression by European settlers, wildfires are thought to have occurred once every 3-4 years in typical mixed-grass prairie, but today an average of only 33 km² is affected by fire each year (Samson et al. 2004). Bison were almost completely eradicated by the end of the 19th century (Brennan and Kulvesky 2005), with populations plummeting from approximately 30 million to 280 individuals (Knopf 1994). Similarly, farmers and ranchers actively attempted to eliminate prairie dog colonies (Brennan and Kulvesky 2005) and today’s populations are estimated to represent only 2% of historic levels (Knopf 1994).

Although cattle have replaced native grazers on the Great Plains, cattle have different grazing preferences than bison (Leuders et al. 2006), and current rangeland-management guidelines promote even, rotational grazing (Holecheck et al. 1998; Fuhlendorf and Engle 2001). This practice, combined with cattle fencing and overgrazing, creates a homogenous and deteriorating rangeland (Knopf 1994; Brennan and Kulvesky 2005). The interaction between patchy fires and free-roaming grazers is essential to maintaining a shifting mosaic of diverse habitat (Fuhlendorf and Engle 2004;

Brennan and Kulvesky 2005). Songbirds in the mixed-grass prairie have diverse vegetation structure preferences and microhabitat features can affect reproductive success (Davis 2005). Successful multi-species management of grassland songbirds requires the maintenance of habitat patches with varying intensities of both fire and grazing (Fuhlendorf et al. 2006; Richardson et al. 2014).

Fragmentation—Crop production in the Great Plains has not only reduced available habitat for grassland songbirds, but it has also fragmented remaining contiguous habitat (Samson et al. 2004; Askins et al. 2007). Many grassland songbirds exhibit area sensitivity and are absent from smaller habitat patches (e.g., Samson et al. 1980; Herkert et al. 1994, Winter and Faaborg 1999; Johnson and Igl 2001; Winter et al. 2006). Area sensitivity may occur as a consequence of territory-size thresholds (Vickery et al. 1994; Burke and Nol 2000), or greater edge to core habitat ratio in small habitat patches (Helzer and Jelinski 1999; Davis et al. 2004), resulting in increased frequencies of predation and brood parasitism (Herkert et al. 2003). Additionally, there is evidence that food availability may be limited in small patches (Zanette et al. 2000).

In mixed-grass prairie, Davis et al. (2004) found that grasshopper sparrow (*Ammodramus savannarum*), Baird's sparrow (*Ammodramus bairdii*), and chestnut-collared longspur densities decline in small patches (Davis et al. 2004). However, a similar study concluded that the effect of patch size on mixed-grass prairie songbirds is more variable; although Sprague's pipit densities increase with patch size, there is little effect of patch size on Savannah sparrow, Baird's sparrow, chestnut-collared longspur and western meadowlark (*Sturnella neglecta*) (Davis et al. 2006). In eastern prairie fragments, 50% reductions in the density of Savannah sparrow and vesper sparrow

(*Poocetes gramineus*) were observed at patch-size thresholds below 10 ha and 20 ha, respectively (Vickery et al. 1994).

2.3 Energy development and avian species

Energy development in western North America—World demand for energy increased by 50% over the latter half of the 20th century and this trend is projected to continue (National Petroleum Council 2007). Western North America possesses immensely valuable energy resources, including hydrocarbon deposits, oil sands, oil shale, and potential for wind, solar and geothermal energy (Copeland et al. 2011). Leases associated with all energy sources combined occupy approximately 20% of total habitat in this region (Copeland et al. 2011). Among these sources, conventional oil and natural gas is currently most extensive, with nearly 59 million ha of public and crown land leased in the United State and Canada (Copeland et al. 2011). The Western Canadian Sedimentary Basin is the largest known hydrocarbon deposit in the world; between 1995 and 2009, Canada's conventional oil sector drilled a minimum of 5,000 new wells per annum and produced no less than an average of 800,000 barrels of crude oil each day (Petroleum Resources Branch 2010). Canada's upstream oil and gas industry produces more annual revenue than Canadian auto-manufacturing, forestry and logging, and wheat and barley production combined (Petroleum Resources Branch 2010).

Oil and gas extraction in Alberta's mixed-grass prairie—Oil was first discovered in Alberta in 1914 (Government of Alberta 2015a) and development has been rapid (Copeland et al. 2011; Government of Alberta 2014). Today, Alberta contains thousands of oil and natural gas wells, thousands of kilometers of access roads and pipeline (Government of Alberta 2014), and over 5,000 compressor stations (Riley et al. 2012).

An additional 10,000-15,000 new wells are constructed each year (Government of Alberta 2014). Annually, a combined average of 34,057 ha of Alberta's prairie is disturbed by seismic lines, pipelines, and wells (Van Wilenberg et al. 2013). This development is likely to continue into the near future; Alberta produces 71% of Canada's total oil output (Petroleum Resources Branch 2010), and in 2013 the province earned 2.5 billion dollars in royalties from the conventional oil and gas sector (Government of Alberta 2015b). In 2012, Alberta's upstream oil and gas industry, (not including oil sands and unconventional sources), received 25 billion dollars in investments, and currently, an estimated 19.7 billion barrels of oil remain un-extracted in the province (Government of Alberta 2015b).

Conventional oil and natural gas extraction—Hydrocarbons, such as oil and natural gas, are produced by the decomposition of organic matter deposited beneath the Earth's surface by the process of sedimentation (Copeland et al. 2011). Natural gas is primarily composed of methane (CH₄), but it may also contain ethane (C₂H₆), propane (C₃H₈), butane (C₄H₁₀) and pentane (C₅H₁₂) (CAPP 2012). Conventional crude oil is a mixture of hydrocarbons and impurities, such as Sulphur (S), and is liquid at normal surface temperature and pressure. It is differentiated from unconventional oil by ease of extraction, geology, and carbon density (IEA 2015).

Conventional oil and natural gas deposits are found in porous rock formations, typically 3000-4000 m below the surface (Devold 2013). These deposits can be accessed by drilling vertically from above, or horizontally beneath the ground from an existing well (CAPP 2012). Once a well has been drilled, conventional natural gas flows freely and is gathered at wellheads where it is fed into pipelines for distribution (CAPP 2012);

pressure in these pipelines is maintained by compressor stations (CAPP 2012), which are large, relatively loud surface structures that use turbines or motors to pump gas (Koper et al. 2015).

Oil extraction often requires the use of an artificial lift device, the most common of which is a sucker rod pump (Devold 2013). This structure is more colloquially known as a pumpjack. Pumpjacks are piston-like structures that draw oil vertically from the ground (Koper et al. 2015). In deep reserves, another type of pump, called a progressive cavity pump (PCP), is inserted further down into the well (Devold 2013). These pumps are also known as screw pumps. Screw pumps function by creating positive displacement pressure, forcing oil from the ground into pipe; both pumpjacks and screw pumps produce less noise than compressor stations (Koper et al. 2015). Pumpjacks and screw pump may be powered by the local electric grid, or by onsite propane-fueled generators (Koper et al. 2015). Once at the surface, oil may be stored onsite in batteries for collection by vehicle at a later time, or it may be transported by pipeline (Devold 2013).

Energy development and avian mortality—Collision with infrastructure and traffic are significant sources of avian mortality associated with energy development (Bayne and Dale 2011; Riley et al. 2012). In Canada, an estimated 3,462 birds are killed by vehicle collision each breeding season per every 100 km of road (Bishop and Brogan 2013). Conventional oil and natural gas wells are typically accompanied by numerous access roads (Copeland et al. 2011), which undoubtedly contribute to avian traffic mortality figures. Furthermore, many energy development structures are electrically powered, and increased densities of transmission lines associated with oil and gas infrastructure are likely detrimental to songbirds. Nationally, transmission lines are the

third largest anthropogenic source of avian mortality in Canada, and interestingly, southern Alberta has been identified as a region of disproportionately high avian mortality attributed to human activities (Calvert et al. 2013).

Additional sources of avian mortality associated with oil and gas development include flaring by natural gas facilities (Riley et al. 2012), toxic waste fluid, which kills at minimum 500,000 birds each year in the United States (Trail 2006), and light entrapment, a phenomena in which migrant songbirds are attracted to lights on infrastructure during poor weather conditions (Ramirez et al. 2015). Construction and maintenance activities associated with oil and natural gas exploration in Alberta's prairie alone result in the destruction of an estimated 3,919 nests annually, and a subsequent loss in annual recruitment of 2,913 individuals (Van Wilgenburg et al. 2013).

Energy development and avian abundance—Research on Galliformes suggests that these species avoid oil and gas infrastructure (Hagen 2010). Greater sage-grouse (*Centrocercus urophasianus*) not only avoid wintering habitat near infrastructure (Doherty et al. 2008; Carpenter et al. 2010), but male lek attendance is also reduced (Walker et al. 2007; Blickley et al. 2010, Harju et al. 2010). Furthermore, sage grouse appear to avoid nesting near infrastructure (Lyon and Anderson 2003; Aldridge and Boyce 2007; Holloran et al. 2010). Similar responses have been documented in prairie chicken (*Tympanuchus* spp.) and sharp-tailed grouse (*Tympanuchus phasianellus*) (Hagen 2010).

Songbirds in several ecosystems also display an avoidance of oil and natural gas infrastructure. In boreal forest, songbirds are rarely detected singing or nesting near pipelines (Bayne and Dale 2011), and ovenbird (*Seiurus aurocapillus*) abundances

decline 19% for every 1 km/km² of seismic lines (Bayne et al. 2005). Ovenbirds also move their territories to avoid crossing seismic lines (Machtans 2006). In sagebrush-steppe, sage-obligate songbird abundances decline with proximity to oil and natural gas infrastructure (Gilbert and Chalfoun 2011) and energy access roads (Ingelfinger and Anderson 2004).

The response of grassland songbirds to oil and natural gas infrastructure appears to be more variable. One study found that Savannah sparrow, Baird's sparrow, chestnut-collared longspur and Sprague's pipit all decline in abundance near oil and gas wells, while horned lark abundances increase (Linnen et al. 2008). Similarly, grassland songbirds appear to avoid unconventional oil infrastructure and roads as well (Thompson et al. 2015). Conversely, another study found that though Sprague's pipit respond negatively to natural gas wells, chestnut-collared longspur are unaffected and Savannah sparrow abundances actually increase in regions of high well density (Hamilton et al. 2011). Grasshopper sparrow, chestnut-collared longspur, and McCown's longspur decline with increasing proximity to natural gas wells at the local scale, but abundances of these species are higher overall at high well density sites (Kalyn Bogard and Davis 2014). Savannah sparrow abundance has been reported to increase with proximity to gas wells, but only below certain well density thresholds (Kalyn Bogard and Davis 2014). Similarly, another study found that density of Savannah sparrow territories increase with proximity to wells (Ludlow et al. 2015).

Energy development and avian reproductive success—Little is known about the effects of oil and gas development on avian nesting success. In greater sage-grouse, the presence of infrastructure is correlated with increased chick mortality (Aldridge and

Boyce 2007) and decreased annual survival of yearlings (Holloran et al. 2010). Conversely, greater prairie-chicken (*Tympanuchus cupido*) nesting near infrastructure experience no decrease in nesting success (Pitman et al. 2005). A study examining the effects of clear-cut forest edges associated with oil extraction on ovenbird nesting success found that nesting success improves with increasing proximity to edge (Ball et al. 2009). However, this study was conducted in remote, relatively undisturbed habitat, and did not measure other infrastructure variables. In sage-obligate songbirds, nest survival does not vary by natural gas well density (Hethcoat and Chalfoun 2015a). Nonetheless, it is influenced by an index of surrounding habitat modification driven by energy development, where nesting success decreases with increasing habitat alteration (Hethcoat and Chalfoun 2015a). This result suggests that energy development may affect birds at the landscape scale even when the effect is not apparent at the local scale. Additionally, activity of small mammal predators was elevated in more disturbed areas (Hethcoat and Chalfoun 2015b).

Only one study has examined the effects of oil and natural gas infrastructure on songbirds nesting in Alberta's mixed-grass prairie (Ludlow et al. 2015). Although nesting success was not affected by proximity to well for any species, Sprague's pipit reproductive success declined with increased cover of crested wheatgrass (*Agropyron cristatum*), a species propagated by surface disturbance (Ludlow et al. 2015). Additionally, Sprague's pipit and Baird's sparrow avoided nesting within 100 m of trails, and nests near trails fledged fewer young. Conversely, vesper sparrow (*Pooecetes gramineus*) displayed the opposite trend, frequently nesting near trails and fledging more young from nests in close proximity to trails (Ludlow et al. 2015).

Edge effects—High densities of oil and natural gas infrastructure create a fragmented landscape with a high proportion of edge habitat (Bayne and Dale 2011; Nasen et al. 2011; Riley et al. 2012). Edge habitat is found at the boundary between habitat types and often has different characteristics than either core habitat; these differences can include changes to microclimate, vegetative structure, composition of plant and animal community, and songbird nest density (Murcia 1995). Collectively, changes in habitat features near edges are referred to as edge effects (Gates and Gysel 1978; Murcia 1995). Edge effects are an important factor in avian reproductive success because elevated predation and brood parasitism of songbird nests are often reported near edges (e.g., Gates and Gysel 1978; Brittingham and Temple 1983; Johnson and Temple 1990; Paton 1994; Manolis et al. 2002). Edge effects on nesting songbirds may extend well beyond immediate habitat boundaries and have the potential to affect large areas of habitat (Flaspohler et al. 2001; Renfrew and Ribic 2003).

The cause of increased nest predation near edges remains controversial, though a number of theories have been proposed (Lariviere 2003). One likely possibility is that the structural heterogeneity of edge habitat creates high predator diversity (Oehler and Litvaitis 1996; Dion et al. 2000), and concentrates the activity of generalist mesopredators with broad feeding niches (Heske et al. 1999; Dijak and Thompson 2000; Winter et al. 2000). Reptilian predators also appear to be attracted to edges, likely because they gain thermoregulatory benefits from having simultaneous access to shaded and open habitat (Blouin-Demers and Weatherhead 2010). Another possible explanation for edge effects on nesting songbirds in grassland systems is that woodland predators use forested edge habitat as a travel lane through open habitat (Johnson and Temple 1990;

Ribic et al. 2012; Ellison et al. 2013). Additionally, edges created by human alterations to the landscape, known as anthropogenic edges, often provide food subsidies for generalist species, as in agricultural landscapes (Oehler and Litvaitis 1996). However, the causes of edge effects are variable and often predator or system specific (Lahti 2001; Ries et al. 2004)

Edge effects may be more pronounced in open systems (Miller et al. 1998), and in grassland songbirds, there is strong support for both increased nest predation (Miller et al. 1998; Winter et al. 2000; Perkins et al. 2003), and brood parasitism (Johnson and Temple 1990; Patten et al. 2006) with increasing proximity to edge. In response to elevated predation and brood parasitism near edges, some species of grassland songbirds appear to avoid nesting near edges altogether (Winter et al. 2000; Bollinger and Gavin 2004; Patten et al. 2006), though it is not clear that this strategy is effective (Renfrew et al. 2005).

Predator facilitation—In addition to edge effects, energy development infrastructure may alter predator density and behavior by providing resources to potential predators (Liebzeit et al. 2009). For example, unconventional natural gas extraction infrastructure in Wyoming provides artificial nesting substrate for common raven (*Corvus corax*), and subsequent increases in raven nest densities are correlated with elevated predation on greater sage-grouse nests (Bui et al. 2010). Similarly, ravens in the Mojave Desert have benefited from human infrastructure, leading to spillover predation on desert tortoise (*Gopherus agassizii*) (Kristan and Boarman 2003). The presence of perches on the landscape, such as buildings, fences, and transmission lines, may facilitate hunting by raptors (Lammers and Collopy 2007), and perch availability has also been implicated in increased brood parasitism (e.g., Freeman et al. 1990; Saunders et al. 2003).

Furthermore, roads may be attractive to mammalian predators that use them as travel corridors (Frey and Conover 2006), and roads can also provide carrion food subsidies to some predators (Kristan et al. 2004).

Anthropogenic noise—Chronic anthropogenic noise produced by traffic, industry and other human activities has a multitude of negative effects on wildlife (Barber et al. 2010). Noise associated with both energy infrastructure (Bayne et al. 2008) and traffic (McClure et al. 2013) negatively impacts songbird abundances. Songbirds also avoid nesting near noise producing structures (Francis et al. 2009), and lower songbird abundances are found in urban settings, particularly for species that sing at lower frequencies (Proppe et al. 2013). This trend may be in part a response to acoustic masking, which occurs when high ambient noise levels render normal avian vocalization amplitudes insufficient for detection (Barber et al. 2010).

Acoustic masking has fitness consequences for songbirds, as it causes individuals to sing louder in order to be heard (Dowling et al. 2011) thereby increasing the energetic cost of vocalization (Thomas 2002). Acoustic masking may also interfere with communication directly. For example, quality and complexity of male singing is thought to represent honest signaling in avian species (Spencer et al. 2003), and acoustic masking could disrupt this process. Chronic noise associated with oil and gas extraction has been shown to reduce ovenbird mate-pairing success in the boreal forest (Habib et al. 2007). Furthermore, though not yet demonstrated in avian species, acoustic masking may also reduce the ability of organisms to detect predators (Chan et al. 2010). Similarly, it has the potential to disrupt the ability of adult birds to communicate predator information to offspring through vocal signals (Haff and McGrath 2010). Finally, in the U.K., house

sparrows (*Passer domesticus*) nesting near electric power generators exhibit smaller brood sizes, poor nestling body condition, and reduced provisioning rates by adults, possibly as a consequence of increased stress levels (Schroeder et al. 2012).

Vegetation alteration—Energy development infrastructure disturbs vegetation and may introduce exotic species (e.g., Berquist et al. 2007; Hamilton et al. 2011, Nasen et al. 2011; Allred et al. 2015). Roads alter vegetation as well (Forman and Alexander 1998; Sutter et al. 2000; Simmers and Galatowitsch 2010), and vehicle traffic facilitates the dispersal of exotic plants (Von Der Lippe and Kowarik 2007). In mixed-grass prairie, reduced cover and diversity of native grasses and herbaceous plants was documented within 20-25 m of natural gas wells, and these effects persisted up to 50 years following initial disturbance (Nasen et al. 2011). Oil infrastructure in the mixed-grass prairies also increases cover of crested wheatgrass (Koper et al. 2014; Ludlow et al. 2015), a common alien species that is associated with reduced plant-community diversity in mixed-grass prairie (Henderson and Naeth 2005).

Mixed-grass prairie songbirds are sensitive to vegetation, and changes in structure and composition of vegetation can affect songbird abundance and reproductive success (Davis 2004, 2005; Koper and Schmiegelow 2006). Chestnut-collared longspur and Sprague's pipit suffer reduced nesting success in crested wheatgrass (Lloyd and Martin 2005; Ludlow et al. 2015), and Sprague's pipit also experiences reduced juvenile survival (Fisher and Davis 2011a). Additionally, exotic vegetation and structural changes to vegetation can result in reduced arthropod availability (e.g., Fielding and Brusven 1995; DeBano 2006, Flander et al. 2006; Hickman et al. 2006).

2.4 Nest predation and potential predators

Nest predation—Predation is the single greatest source of nest failure in most songbirds (11-78% of nest mortality) and consequently it is an important selective force in shaping avian life histories (Martin 1993a, 1995). In systems where nest predation is high, nest site selection (Martin 1993b), nestling development periods (e.g., Ricklefs 1969; Martin 1995), and clutch size (e.g., Martin 1993a; Zanette et al. 2013) may all be regulated by predation. Nesting songbirds have varied and complex responses to perceived predation risk (Lima 2009), but some commonly observed behaviors include: avoidance of unsafe nesting habitat (Winter et al. 2000, Forstmeier and Weiss 2004; Monkkonen et al. 2009), varying clutch size (Monkkonen et al. 2009; Zanette et al. 2013), and increased vigilance behavior (Geig-Smith 1980; Marzluff 1985). Nest predation is typically higher in shrub and grassland systems than in forests (Martin 1993a), and in some regions of Canada's mixed-grass prairie, nest predation may account for up to 88% of nesting failures (Davis 2003).

Nest predators in grassland ecosystems—The advent of video surveillance technology has revealed a wide array of mammalian, avian, and reptilian nest predator species (Pietz et al. 2012). In grasslands, mammals are the most frequently documented predator group, especially in the northern Great Plains where snakes comprise only 5% of recorded predators (Pietz et al. 2012). Some of the most commonly reported predators of grassland songbird nests include: thirteen-lined ground squirrel (*Ictidomys tridecemlineatus*), Franklin's ground squirrel (*Poliocitellus franklinii*), American badger (*Taxidea taxus*), weasel (*Mustela* spp.), raccoon (*Procyon lotor*), striped skunk (*Mephitis mephitis*), gartersnake (*Thamnophis* spp.), northern harrier (*Circus cyaneus*), and brown-

headed cowbird (*Molothrus ater*) (e.g., Pietz and Granfors 2000; Renfrew and Ribic 2003, Renfrew et al. 2005; Davis 2012, Pietz et al. 2012; Ribic et al. 2012).

2.5 Accounts of mixed-grass prairie songbirds

Savannah sparrow (*Passerculus sandwichensis*)—The Savannah sparrow is widespread throughout North America, but its populations in Canada have declined by 1.3% per annum since 1966 (Table 2.1) (Sauer et al. 2014). Savannah sparrows are open-habitat generalists and vary in phenotype by geographic region; they are small sparrows distinguished by a yellow lore, short, notched tail, and heavily streaked breast (Wheelright and Rising 2008). Plumage is monomorphic between sexes, though there is slight size-dimorphism in some populations (Weatherhead 1980). Savannah sparrows are found in a wide variety of open and grassland habitats; they are ground foragers and will consume grasses, seeds, fruit, insects, spiders, and insect eggs and larvae (Wheelright and Rising 2008).

The Savannah sparrow is a long distance migrant that breeds throughout all of Canada and the Northern United States and winters in Mexico and the southern United States (Wheelright and Rising 2008). Territories range from 0.11 to 1.25 ha depending on region; some males are polygynous while others are monogamous (Wheelright and Rising 2008). Males arrive on the breeding grounds 1-3 weeks before females, usually beginning in mid-April to early May (Wheelright and Rising 2008), and peak nest-initiation occurs in late May in Montana (Jones et al. 2010), and early to mid-June in Saskatchewan (Davis 2003).

Savannah sparrows select nest sites with minimal bare ground and high litter-depth (Davis 2005), and nests are usually positioned in a depression or at the base of

vegetation (Wheelright and Rising 2008). Females construct nests from coarse grasses that form the exterior, and line the interior with finer grasses; they seldom use other materials (Wheelright and Rising 2008). Eggs are sub-elliptical, smooth, pale blue, green, or cream in color, and speckled or splotched in brown with markings often heavily concentrated at the larger end (Baicich and Harrison 1997). Nestlings have dark gray or brown down and pink mouth with yellow gape-flanges (Baicich and Harrison 1997). Females typically brood for a greater proportion of time than males, but one study recorded males brooding at approximately 74% of nests and contributing equally to provisioning (Bedard and Meunier 1983). In Saskatchewan's mixed-grass prairie, incubation and nestling periods are approximately 11.5 d and 9 d, respectively (Davis 2003). In Alberta, mean clutch size is 4.5 and nest predation and brood parasitism frequencies are approximately 47% and 11%, respectively (Ludlow et al. 2014). Savannah sparrows in mixed-grass prairie rarely reject cowbird eggs (Klippenstine and Sealy 2008).

Vesper sparrow (Pooecetes gramineus)—Like many grassland species, vesper sparrow has been in decline for the past half-century, though not steeply (0.9% per annum) (Table 2.1) (Sauer et al. 2014). The vesper sparrow is a large, stocky sparrow with pale coloration, and distinguishing markings include a pale auricular with a dark border, white outer-tail retrices, and rusty lesser wing-coverts (Jones and Cornely 2002). The vesper sparrow inhabits grassland, shrubland, and early-successional habitat (Jones and Cornely 2002), and unlike many grassland songbirds it responds positively to shrubs, likely because males require singing perches (Best and Rodenhouse 1984). Similarly, the vesper sparrow often establishes territories in cropland or near fencerows and is known to

sing frequently from fences and other available perches (Rodenhouse and Best 1983). It is a ground forager and consumes a variety of insects, seeds, grains, and vegetation (Jones and Cornely 2002).

Vesper sparrow breeds throughout most of the northern United States and southern Canada, and winters in the southern United States and Mexico (Jones and Cornely 2002). Vesper Sparrow arrives on its breeding grounds in Alberta in mid-April to early May and territory size ranges from 0.29 to 8.19 ha (Jones and Cornely 2002). Vesper Sparrow is seasonally monogamous, though polygyny does occur; both sexes incubate, brood, and provision young (Jones and Cornely 2002).

Little is known about the nest construction process, but the female is thought to construct nests alone; the exterior of the nest is made from coarse vegetation while the interior is finer and may include animal hair or pine needles, (where available), (Jones and Cornely 2002). Nests are generally built at the base of vegetation or in a depression (Jones and Cornely 2002). One study found that nests in areas with greater vegetation cover and less bare ground are more successful (Wray and Whitmore 1979). Eggs are smooth, white in color, and covered in indistinct mottling with sparse bolder marks concentrated on the larger end (Baicich and Harrison 1997). Nestlings have gray down and a deep-pink mouth with yellow flanges (Baicich and Harrison 1997). Incubation and nestling periods are 11-13 d and 9-13 d, respectively (Baicich and Harrison 1997). In Alberta's mixed grass prairie, the average clutch size is 3.8, predation is approximately 54%, and only 2% of nests are parasitized (Ludlow et al. 2014).

Chestnut-collared longspur (Calcarius ornatus)—The chestnut-collared longspur has declined heavily over the past century and has been extirpated from parts of its

original range (Bleho et al. 2015). Chestnut-collared longspur is the smallest longspur, and can be distinguished from the similar McCown's longspur (*Rhynchophanes mccownii*) by a dark black triangle in the center of the tail (Bleho et al. 2015). Chestnut-collared longspur displays strong sexual dimorphism during the breeding season; males are distinguished by a bold black breast and belly, buffy throat, black eye line, and rusty nape, while females are drab and possess few clear markings besides a white necklace, though some may also develop a faint collar (Bleho et al. 2015). Chestnut-collared Longspur is a prairie specialist, and its breeding habitat is restricted to mixed-grass prairie with short vegetative structure; its feeding habits are similar to other ground-foraging grassland species and diet consists primarily of insects and seeds (Bleho et al. 2015).

Chestnut-collared longspur is a long-distance migrant and it breeds in the Northern Great Plains and winters in the southern United States and Mexico (Bleho et al. 2015). It arrives on its breeding grounds in Alberta during mid to late April (Bleho et al. 2015), and peak nest-initiation occurs in mid-May in Montana (Jones et al. 2010) and early to mid-June in Saskatchewan (Davis 2003). Male chestnut-collared longspurs are highly territorial. They perform aerial song displays and will defend a territory even without a mate; territories in Alberta are typically 1 ha (Bleho et al. 2015). The Chestnut-collared Longspur is a monogamous double-brooder but frequently engages in extra-pair copulation (Bleho et al. 2015). Both parents brood and provision young (Bleho et al. 2015), and the presence of the male is critical to reproductive success (Lynn and Wingfield 2003).

Vegetation at nest sites is generally short and sparse (Davis 2005), and nests may be located in a depression or near a cow patty (Bleho et al. 2015). The Chestnut-collared Longspur will not nest in cropland (Owen and Myres 1973), and though it will use crested wheatgrass, nesting success is reduced in this habitat (Lloyd and Martin 2005). The nest is constructed by females and primarily composed of woven grasses, but it may also include hair, feathers, and other materials; the rim of the nest is flush with the ground (Bleho et al. 2015). Eggs are short, smooth, and creamy in color with irregular, variably colored splotches (Baicich and Harrison 1997). Nestlings have gray down concentrated on the head and back, and pink legs and bill (Baicich and Harrison 1997). In Saskatchewan's mixed-grass prairie, incubation and nestling periods are approximately 11.5 d and 10 d, respectively, average clutch size is 4.2, and nest predation is approximately 64% (Davis 2003). Brood parasitism of chestnut-collared longspur nests is variable in mixed-grass prairie, as Davis (2003) reported 16% parasitism while Klippenstine and Sealy (2008) reported only 5%. In a study of egg rejection behavior in grassland songbirds, rejection of cowbird eggs was not observed at any chestnut-collared longspur nests (Klippenstine and Sealy 2008).

Western meadowlark (Sturnella neglecta)—The western meadowlark is an abundant and widely distributed open country species, but it has declined at 1.3% per annum since 1966 (Table 2.1) (Sauer et al. 2014). It is distinguished from other mixed-grass prairie songbirds by its large size, bright-yellow throat and breast, and disproportionally short tail; sexes are similar but females are marginally smaller with less distinct markings (Davis and Lanyon 2008). Individuals can also be sexed by vocalization as the rattle and roll calls are specific to females and males, respectively

(Davis and Lanyon 2008). Western meadowlark is found most frequently in native grasslands and grassland converted back from cropland (Davis and Lanyon 2008). It is a granivorous species during the winter and insectivorous in the spring and summer (Davis and Lanyon 2008). Western meadowlark is also a documented nest predator of other grassland songbirds (Davis et al. 2012).

Unlike other species discussed here, the western meadowlark is a short-distance migrant in parts of its range and resident in much of the western United States (Davis and Lanyon 2008). In Alberta, males arrive on the breeding grounds as early as late March and begin to defend territories prior to the arrival of females; territory sizes vary by region and territories as large as 13 ha have been reported (Davis and Lanyon 2008). Western meadowlarks are polygynous and males often have two mates simultaneously (Davis and Lanyon 2008). Peak nest-initiation occurs in late May in Montana (Jones et al. 2010) and early June in Saskatchewan (Davis 2003).

Western meadowlark prefers to nest in dense vegetative cover, but avoids woody growth and tall grass (Davis and Lanyon 2008). Females construct ground nests without assistance from males, and nests are often built in natural depressions and made from grass, fine bark and other vegetative material (Davis and Lanyon 2008). As in Sprague's pipit, nests are characterized by roofs and runway entrances (Davis and Lanyon 2008). Females provide most parental care (Davis and Lanyon 2008), and eggs are oval, large, glossy, and white in color with reddish-brown speckles concentrated at the larger end (Baicich and Harrison 1997). Nestlings are told from those of other grassland birds by size and red gape with ivory flanges (Baicich and Harrison 1997). In Saskatchewan, incubation and nestling periods are approximately 13 d and 11 d, respectively (Davis

2003). In Alberta, average clutch size is 5.4, nest predation is approximately 46%, and brood parasitism is approximately 4%. As with other species, higher parasitism is reported in Saskatchewan (Davis 2003; Klippenstine and Sealy 2008). Unlike most grassland species, western meadowlark is a relatively proficient egg rejecter and ejects cowbird eggs approximately 50% of the time (Klippenstine and Sealy 2008).

Sprague's pipit (Anthus spagueii)—Sprague's pipit is an endemic of North America's grasslands, and like chestnut-collared longspur, its populations have declined extensively over the previous century (Davis et al. 2014). It is a slender, streaked bird with a pale breast, buffy face, white eye-rings, and white outer-tail retrices; sexes are similar (Davis et al. 2014). Sprague's Pipit is notable for its protracted aerial displays (the longest of any known avian species in the world) in which the male circles and sings up to 100-m above the ground for an average of 11-30 minutes (Robbins 1998). Sprague's pipit is found in dry, native grasslands with low shrub-densities (Davis et al. 2014). It avoids heavy grazing (Davis et al. 1999) and edges (Koper et al. 2009), and subsists primarily on arthropods and seeds (Davis et al. 2014).

Sprague's pipit breeds almost exclusively in the Northern Great Plains and winters in the southwestern United States and central Mexico (Davis et al. 2014). It typically arrives on its northern breeding grounds in late April to early May (Davis et al. 2014), with peak nest-initiation occurring in late May in Montana (Jones et al. 2010), and in early to mid-June in Saskatchewan (Davis 2003). Only females appear to participate in nest construction (Sutter 1997). Females provide most parental care (Davis et al. 2014), though provisioning by male Sprague's pipit has been observed (Dohms and Davis

2009). Similarly, while Sprague's pipit is thought to be generally monogamous (Davis et al. 2014), polygyny does occur (Dohms and Davis 2009).

Sprague's Pipit selects nest sites with intermediate vegetation height and litter depth, minimal bare ground, and low forb-density (Sutter 1997). Although it is considered a native-grass specialist, Sprague's Pipit will make use of structurally similar planted fields (Fisher and Davis 2011b). However, planted grasslands may act as habitat sinks for this species (Fisher and Davis 2011a). Nests are located on the ground, constructed from woven dried grasses, and characterized by dome, or tunnel-like canopies over the entrance (Sutter 1997). Eggs are smooth, pale gray to buff in color and heavily mottled with variably colored, evenly distributed markings (Baicich and Harrison 1997). Nestlings have long, light-gray down (Baicich and Harrison 1997), and diagnostic bright-orange gape with yellow flanges (personal observation). In Saskatchewan, incubation and nestling periods last approximately 13 d and 11 d, respectively (Davis et al. 2003). In Alberta, average clutch size is 4.7 and nest predation is approximately 29% (Ludlow et al. 2014). Sprague's pipit appears to be an infrequent cowbird host in Alberta with 0% ($n = 21$) parasitism documented in one study (Ludlow et al. 2014). However, parasitism as high as 16% ($n = 19$) has been reported in mixed-grass prairie in Saskatchewan (Davis 2003; Klippenstine and Sealy 2008).

Table 2.1. Total population size estimate, annual change in population trend, and percent of world population held in Canada for five species of grassland songbird in Canada. Population size estimates are taken from Partners in Flight Science Committee 2013 and annual trends are taken from Sauer et al. (2014).

Species				
Common name	Scientific name	Population estimate	Annual trend (%)	Percent world (%)
Savannah sparrow	<i>Passerculus sandwichensis</i>	110,000,000	-1.3	61.1
Vesper sparrow	<i>Pooecetes gramineus</i>	10,000,000	-0.9	35.7
Western meadowlark	<i>Sturnella neglecta</i>	7,500,000	-1.3	8.8
Sprague's Pipit	<i>Anthus spragueii</i>	700,000	-3.8	77.6
Chesnut-collared longspur	<i>Calcarius ornatus</i>	600,000	-6.6	20.0

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CHAPTER 3: INFRASTRUCTURE NOT AMPLITUDE: PHYSICAL FOOTPRINT OF OIL AND GAS INFRASTRUCTURE, NOT ANTHROPOGENIC NOISE, REDUCES NESTING SUCCESS OF GRASSLAND SONGBIRDS

Abstract

Alberta's mixed-grass prairies have recently undergone a rapid expansion of conventional oil and natural gas development, the effects of which are largely unknown for nesting songbirds. Assistants and I monitored 813 grassland songbird nests located at a variety of infrastructure and control sites to determine if the presence of infrastructure affects nesting success. Nest survival was significantly lower at infrastructure sites relative to controls for both Savannah sparrow and vesper sparrow, but there was no difference in nesting success for any species between sites with varying noise intensities; similarly, roads had no immediate impact on nest survival for any species. Finally, apparent vesper sparrow nest density increased with proximity to infrastructure features. The findings of this study suggest that management strategies focusing on reducing anthropogenic noise and traffic disturbance may be ineffectual for grassland songbirds. Instead, managers should seek to reduce the physical footprint of disturbances on the landscape

3.1 Introduction

Grasslands are among the most imperiled ecosystems in the world, with nearly 50% of grassland ecosystems already converted to other habitat types, and only 5% occurring in protected areas (Hoekstra et al. 2005). In North America, loss of grassland habitat to cover crop and row crop agriculture has been extensive (Herkert 1991; Knopf 1994; Brennan and Kulvesky 2005). Over the past two centuries, grasslands of the Great Plains have been reduced to 30% of their historic extent (Samson et al. 2004). This habitat loss has had a direct and negative impact on grassland songbird populations (Herkert 1995; Peterjohn and Sauer 1999), which are declining more rapidly than birds of any other North American biome (Sauer et al. 2014). These declines have been exacerbated by degraded rangeland conditions (Knopf 1994; Askins et al. 2007) and fragmentation of remaining grasslands (Samson et al. 2004).

The rapid expansion of energy development across western North America has contributed to the loss and degradation of grasslands, and may have significant conservation implications, as energy leases currently occupy an estimated 21% of all grassland habitats in this region (Copeland et al. 2011). Alberta's mixed-grass prairie comprises 43% of Canada's remaining grasslands (Gauthier and Wiken 2003). However, this region is imperiled as it is located above Canada's Western Sedimentary Basin (Mossop and Shetsen 1994), the largest confirmed hydrocarbon deposit in the world (Petroleum Resources Branch 2010). Alberta contains at least 300,000 oil and natural gas wells and over 350,000 km of pipeline, occupying a total land area of approximately 1,000,000 ha (Government of Alberta 2014a). Annually, energy development activity disturbs an average of 34,057 ha of Alberta's mixed-grass prairie (Van Wilgenburg et al. 2013).

Energy development can have a variety of effects on avian species, and may pose a further threat to declining grassland songbirds (Askins et al. 2007; Riley et al. 2012). Direct mortality is among the most obvious of these effects. For example, waste fluid in tailing ponds created by unconventional oil extraction kills between 500,000 and 1,000,000 birds annually in the United States (Trail 2006). Similarly, flaring by natural gas facilities and collision with infrastructure and traffic are additional sources of avian mortality associated with energy development activities (Riley et al. 2012; Bishop and Brogan 2013; Calvert et al. 2013). In Alberta's mixed-grass prairie, surface disturbance caused by oil and gas development is estimated to result in the direct mortality of 3,919 songbird nests annually (Van Wilgenburg et al. 2013).

However, energy infrastructure also disturbs large areas of habitat at multiple scales and may have additional indirect effects on avian species (Bayne and Dale 2011; Copeland et al. 2011; Riley et al. 2012; Allred et al. 2015). For instance, avoidance of oil and gas infrastructure by some species of birds is well documented. Sage-obligate songbird densities decline with proximity to energy infrastructure (Gilbert and Chalfoun 2011) and associated access roads (Inglesinger and Anderson 2004). Densities of grassland specialists also decline near oil, natural gas wells, and roads (Linnen 2008; Hamilton et al. 2011; Thompson et al. 2015). Additionally, there is strong evidence that greater sage-grouse (*Centrocercus urophasianus*) actively avoid infrastructure on the landscape (e.g., Doherty et al. 2008; Carpenter et al. 2010, Harju et al. 2010, Holloran et al. 2010). However, density and habitat use may not accurately reflect habitat quality or nesting success (Van Horne 1983; Vickery et al. 1992a).

Little research has been conducted on the effects of energy infrastructure on avian reproductive success, and the results of these studies have been mixed. Red-winged blackbird (*Agelaius phoeniceus*) and blue-gray gnatcatcher (*Poliioptila caerulea*) nesting near wind turbines experience no decrease in nest survival or reproductive success (Bennett et al. 2014, Gillespie and Dinsmore 2014). Likewise, a study of lesser prairie-chicken (*Tympanuchus pallidicinctus*) found no effect of proximity to oil and gas infrastructure on nesting success (Pitman et al. 2005). Ovenbirds (*Seiurus aurocapilla*) nesting near clear-cut forests associated with unconventional oil development in the boreal forest experience improved nesting success (Ball et al. 2009). Conversely, other studies have observed negative effects of energy infrastructure on reproductive success. For example, in sagebrush steppe, songbird nesting success decreases as the proportion of

habitat on the landscape disturbed by oil and gas extraction increases (Hethcoat and Chalfoun 2015a); additionally, nest predation and small mammal activity increased in these disturbed areas (Hethcoat and Chalfoun 2015b). Another study reported high chick mortality in greater sage-grouse territories near energy infrastructure (Aldridge and Boyce 2007). There is also evidence that energy infrastructure may alter the nest density of breeding songbirds, attracting some species and displacing others (Shaffer and Buhl 2015). These diverse findings suggests that energy infrastructure has the potential to impact reproductive success in a number of ways, and that these effects may be system or species specific.

Reproductive success plays a critical role in maintaining stable populations (Martin 1992). Therefore, successful conservation requires an understanding of the parameters that govern reproductive success. Nest predation is the largest source of nest failure in most songbirds (Gates and Gysel 1978; Martin 1993a, 1995), including mixed-grass prairie species (Davis 2003; Kerns et al. 2010; Ludlow et al. 2014), and predation pressure plays an important role in shaping avian life histories (Ricklefs 1969; Martin 1993a,b, 1995).

Oil and gas infrastructure and associated linear features, such as roads, transmission lines, and seismic lines may impact reproductive success through the fragmentation of landscapes and introduction of high proportions of anthropogenic edge (Bayne and Dale 2011, Nasen et al. 2011). Grassland songbirds are sensitive to edges and patch size (Davis 2004; Koper et al. 2009; Sliwinski and Koper 2012), and alterations to landscape structure may create edge effects, which can alter frequencies of nest predation and brood parasitism (Gates and Gysel 1978; Johnson and Temple 1990). Edges can

alter predator community composition and behavior as well (e.g. Winter et al. 2000; Blouin-Demers and Weatherhead 2001; Renfrew and Ribic 2003), and energy infrastructure may provide food and habitat subsidies to some predator species (Kristan and Boarman 2003; Liebezeit et al. 2009; Bui et al 2010). Additionally, energy development produces anthropogenic noise (Habib et al. 2007; Bayne et al. 2008; Barber et al. 2010), which can alter predator-prey dynamics (Francis et al. 2009; Chan et al. 2010) and may reduce songbird fitness (Schroeder et al. 2012) and productivity (Knight et al. 2012). As the proportion of grassland habitat affected by energy development increases, it is important to evaluate how these changes may affect the reproductive success of grassland songbirds.

The purpose of this study was to determine if and to what extent oil infrastructure affects the reproductive success of declining songbird species in Alberta's mixed-grass prairie. The study was conducted primarily at site with one of two types of oil extraction well pumps: pumpjacks and screw pumps. Both are noise producing structures designed to extract oil from wells beneath the surface (Koper et al. 2015). However, I also chose to include natural gas compressor stations sites in the study, because they have a large footprint and are loudest class of infrastructure in the study area (Koper et al. 2015). The inclusion of multiple structure types in the study allowed me to determine whether potential effects were proportional to footprint size and noise production levels of structures. The study also included some facilities that were active during the breeding season (i.e. producing noise and receiving regular visits from site operators), and some that were not. This allowed me to isolate the effects of noise and disturbance from effects caused by the physical structure. From a management perspective, it is critical to separate

the mechanisms driving potential effects, as mitigation strategies intended to reduce noise will differ markedly from those intended to address the physical impacts of infrastructure.

To assess the effects of infrastructure on nesting songbirds, I monitored nesting success at infrastructure and control sites as a function of proximity to structures and associated roads, and the presence of noise and human disturbance. I hypothesized that infrastructure creates edge effects and may alter the predator community. Consequently, I predicted that songbird nests located at infrastructure sites would be depredated with greater frequency than those at control sites. Additionally, I hypothesized that songbirds are negatively affected by noise and disturbance generated by active infrastructure, and therefore I predicted that nesting success would be lower at with active infrastructure than at either controls or inactive structures. I also employed video-surveillance systems to identify common predators of songbird nests in this system.

3.2 Methods

3.2.1 Study site—Research activities were conducted on 64-ha plots (quarter sections), of native mixed-grass prairie surrounding Brooks, AB (50° 33' 51" N 111° 53' 56" W) (Figure 3.1). Brooks is located in southeastern Alberta at an elevation of 758 m above sea level, and receives an average annual precipitation of 348 mm (Government of Canada 2015). This area was chosen because it is located in a region of high conventional oil and natural gas resources (Government of Alberta 2014b). In Brooks, AB, average oil well and natural gas well densities are 0.5/km² and 5.9/km², respectively. The landscape surrounding Brooks is approximately 61% grassland, 36% cropland and forage, and 3% water (Koper et al. 2009). The native grasslands in this area are considered mixed-grass prairie (Bailey et al. 2010), and are characterized by predominantly native-grass species

including blue grama (*Bouteloua gracilis*), needle-and-thread (*Hesperostipa comata*), western wheatgrass (*Pascopyrum smithii*), and junegrass (*Koeleria macrantha*). Goatsbeard (*Tragopogon dubius*) and crested wheatgrass (*Agropyron cristatum*) are common exotic plant species throughout the study area; the former occurs as isolated flowers while the latter may be found in monotypic stands.

A total of 73 sites were used during the 2012, 2013, and 2014 breeding seasons (May-July), including 27 control sites and 49 infrastructure sites. Not all sites were used during all years, and site activity at some infrastructure sites varied among years (Appendix I-II). Study sites were located within 50 km of Brooks, AB (Figure 3.2) and were selected based on information and legal-land descriptions regarding availability of infrastructure sites (Cenovus Energy 2012). Well location layers and Alberta Township Grid layers were provided by Divestco and IHS Energy, respectively. This information was then overlaid on satellite imagery to allow me to select grassland sites within suitable areas. All sites were ground validated to ensure that vegetation was predominantly native mixed-grass prairie, accuracy of infrastructure locations, and that sites with tall, non-native vegetation, trees, large areas of wetland, extreme topography, or sites in close proximity to paved roads were not used. All sites were surrounded by mixed-grass prairie and did not border other systems (e.g., shelter belts, cropland) to minimize edge effects (Koper et al. 2009). The center point of each site was at least 400 m away from the center point of any other, and control sites were at least 800 m away from focal infrastructure features to ensure that noise from active infrastructure did not influence control sites. Some control sites contained minor infrastructure with small, local footprints, such as

shallow gas wells, dirt roads, fences, and transmission lines, all of which are typical of prairies in the region.

Each nest-searching plot was defined as an 800-m by 200-m area containing an oil well, natural gas compressor station, or a control point at the center. Control sites were characterized by open, native grassland and the absence of infrastructure (Figure 3.2a). Infrastructure sites included pumpjacks, screwpumps, or compressor stations. Pumpjacks and screwpumps are both varieties of artificial lift device used to extract oil from active wells, and each is also known by an alternate technical name, sucker rod pump and progressive cavity pump, respectively (Devold 2013). Pumpjacks extract oil from wells via vertical motion while screwpumps force oil from the ground by positive displacement (Koper et al. 2015). Compressor stations are large facilities that function to pressurize pipelines for the transport of natural gas following extraction (CAPP 2012) (see Table 3.1. for noise amplitudes of different structures). All infrastructure sites were located on leases characterized by bare ground, barbed wire fences at the perimeter, technical cattle fences around infrastructure, and permanent gravel access roads (Figure 3.2b-d). Although the typical lease size for oil and gas infrastructure in this region is 1 ha (Van Wilgenburg et al. 2013), the size of fenced and non-vegetated area varied among sites; the average disturbed area at pumpjack and screwpump sites was 3,787 m² ($n = 4$) and 10,636 m² ($n = 3$), respectively. Compressor sites were generally the largest structure, but height and footprint size were highly variable among sites.

I further classified oil infrastructure sites by power source; some structures were powered by the local electrical grid via transmission lines while others were powered by propane-fueled generators. I also classified each infrastructure site as either active or

inactive, depending on whether it was operating and therefore producing noise and receiving visits from site operators at regular intervals during the breeding the season; active oil wells are typically visited by company personnel once each day. At inactive sites, infrastructure is present but machinery is turned off and site visits are infrequent. This allowed me to use infrastructure activity as a variable to capture potential effects of anthropocentric noise and human disturbance (i.e. traffic, human presence). Noise levels differed among structure types and between power sources (Table 3.1). On average, compressor stations were the loudest structure, and generator-powered structures were louder than grid-powered structures; noise levels at all active structures were higher than at controls (Table 3.1).

Roads were the primary linear feature associated with oil and gas development in this region. The majority of the roads near my study plots consisted of gravel and dirt-packed range, township, and oil access roads of similar structure; roads were not paved. I used GPS units (Garmin etrex 10) to map the location of roads at each site.

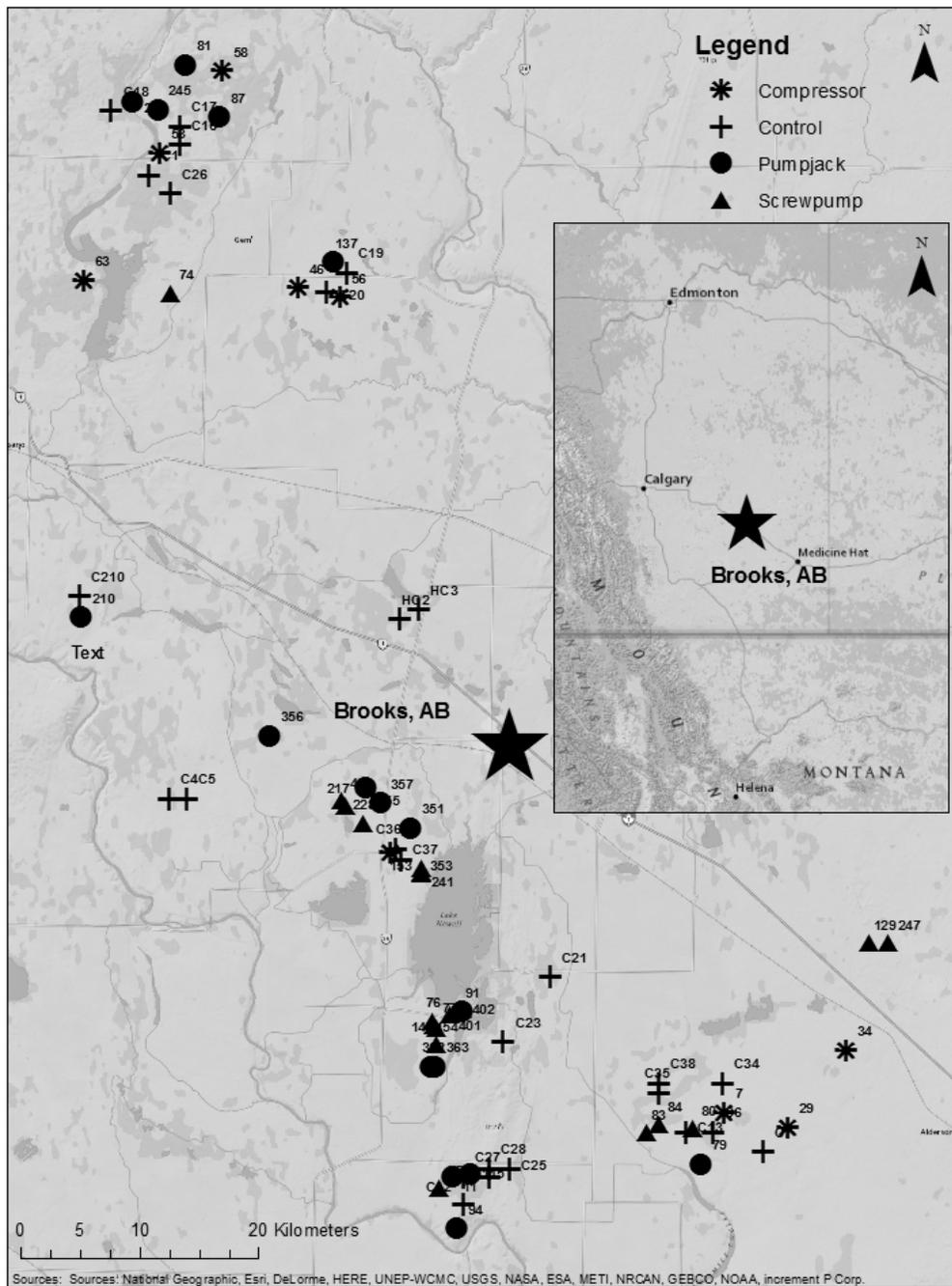


Figure 3.1. Distribution of control and infrastructure study sites around Brooks, AB, 2012-2014, and location of Brooks, AB in southern Alberta.

(a)



Photo by J. Bernath-Plaisted, 2014.

(b)



Photo by J. Bernath-Plaisted, 2013.

(c)



Photo by H. Nenninger 2013

(d)



Photo by B. Antze 2013

Figure 3.2. Photo (a)—Typical control site in Brooks, AB. Photo (b)—Generator-powered pumpjack and technical fence. Photo (c)—Grid-powered screw pump and technical fence. Photo (d)—Compressor station with traditional fence.

Table 3.1. Time-averaged L_{Ceq} sound-pressure levels measuring noise produced within 10 m of infrastructure types ± standard deviation. Units are in C-weighted decibels (dB). Description of sound level measurements is given in Koper et al. (2015). Decibels are logarithmic units; for reference, the human speaking voice is approximately 60 dB and a motorcycle at 15 mph is approximately 80 dB (Barber et al. 2010).

Structure type	Power source	dB(C)
Control (n=11)	N/A	52 ± 5.0
Pumpjack (n=4)	Generator	73 ± 5.2
Pumpjack (n=4)	Grid	62 ± 2.8
Screwump (n=5)	Generator	79 ± 3.5
Screwump (n=5)	Grid	59 ± 2.0
Compressor station (n=4)	N/A	82 ± 3.1

3.2.2 *Collection of nesting data*—Technicians and I searched for nests using rope-drag surveys (Winter et al. 2003). In each year, we conducted three rounds of rope drag searches over the course of the season beginning in the last week of May and ending in the third week of July. To maximize efficiency, and minimize temporal bias, we conducted rope surveys each day as soon as the grass was dry (e.g., between 0630 and 0800 MT) and concluded surveys by 1200 MT each day. This practice ensured that our nest-searching efforts were relatively equal among plots, and that they occurred when songbirds are more likely to be on the nest (Davis and Holmes 2012). We did not survey when the temperature was below 10° C or when the grass was excessively wet.

When nests were detected, we marked their locations with a bamboo stake 10 m to the south and a nail and high-visibility pink chaser 10 m to the west. We took GPS coordinates at the nests and also sketched a brief map of the microhabitat surrounding the nest, to assist in relocating nests on later visits. During all initial nest visits and subsequent checks, we photographed the contents of the nest, recorded the number of eggs and or nestlings, aged nestlings when present, noted the status of adult birds, and recorded any evidence of depredation, abandonment or brown-headed cowbird (*Molthrus ater*) parasitism. We returned to monitor nests at 2-5 d intervals for the duration of nest activity. We visited more frequently (2-3 d) when nests were thought to be near fledging age, but we visited less frequently during times of inclement weather and when roads were not passable. We did not visit nests daily to avoid causing abandonments and trampling nesting areas (Winter et al. 2003). To improve confidence in our ability to correctly distinguish between successful or failed nests, we conducted 30-minute fledge

checks on empty nests that were suspected of having fledged (i.e., nests that would have contained 8 d or older nestlings the day the nest was found to be empty). During fledge checks, we observed adult behavior and watched for evidence of fledgling provisioning activities, or agitation. We also searched the immediate area for the fledglings. I did not quantify the number of young fledged at each nest, as this number could not be reliably collected without daily visits. Instead, I considered nests successful if they fledged at least one host young and failed if they did not, assigning a “1” to successful intervals and a “0” to failed intervals. All research activities involving wildlife adhered to guidelines specified by the University of Manitoba Animal Care Committee (ACC; protocol number F12-010).

3.2.3 *Nest cameras*—During the 2013 and 2014 breeding seasons, I placed video-surveillance systems (Cox et al. 2012) on active chestnut-collared longspur (*Calcarius ornatus*) and Sprague’s pipit (*Anthus spragueii*) nests. Video systems consisted of the following components (Figure 3.4a-b): 12-V 3.66-mm lens Speco Technologies® infrared, weatherproof security cameras (model number: CVC 628M) mounted on small blocks of wood secured to the ground with screws; 704 by 480 resolution miniature digital-video recorder (DVR) and 32GB removable SD cards; 12V-35ah rechargeable lead-acid battery; waterproof ammo-case housing; 30.5 m combined power-video BNC (Bayonet Neill–Concelman) cable. I followed procedures similar to those described in Thompson et al. (1999) and Davis et al. (2012), and I disguised cameras with burlap and native vegetation to reduce abandonment risk. I also secured cables in the ground with u-nails to prevent cattle and other wildlife from disturbing them, and I covered cables under vegetation where possible. Boxes containing batteries and DVR’s were always placed as

far from the nest as possible (30 m with cable extended) and hidden behind shrubs or in ground depressions. Additionally, cameras were placed no closer than 20 cm to nests to minimize risk of abandonment, based on preliminary experiments with the cameras. Once in place, cameras were not removed until termination of the nesting attempt, and footage was collected without interruption. Batteries and SD cards were changed every 3 d at nest checks

(a)



Photo by J. Bernath-Plaisted 2014

(b)



Photo by J Bernath-Plaisted 2013

Figure 3.3. Photo (a)—Nest camera disguised with native vegetation. Photo (b)—Nest camera system with ammo-case housing (cable not extended in photo).

3.2.4 Vegetation surveys—Vegetation sampling protocols were adapted from Wiens (1969). Technicians and I conducted vegetation surveys at each nest upon termination of the attempt. We conducted surveys throughout the season immediately following termination of nesting attempts to ensure that vegetation characteristics reflected conditions around the nest during its activity. We conducted vegetation surveys by placing two meter sticks so that they intersected across the center of the nest to create a 1-m² area divided into four sampling quadrats (NE,NW,SE,SW). We used a Wien's pole (Wiens 1969) to measure stem height, density, and litter depth, and we estimated percent cover of grasses, forbs, lichens, moss, bare ground, and exotic species. I averaged vegetation characteristics across quadrats for each nest during analysis. Vegetation data were collected inconsistently in 2012, and these data were not included in analysis.

3.2.5 Statistical analysis—I conducted all statistical analyses in Program R version 3.1.2 (R Core Team 2014). I accepted 0.10 as the critical alpha value to evaluate statistical significance, as this reduced the probability of incorrectly concluding that there was no effect of variables (Type II error). In conservation biology, the costs of Type II errors are often greater than falsely concluding that there is an effect when there is not (Type I error) (Taylor and Gerrodette 1993); therefore, a lower threshold for significance was appropriate for my study.

3.2.6 Preliminary analysis—I used a binomial probability distribution to model nesting success, where each nesting interval was a Bernoulli trial with two possible outcomes, 1 (success) and 0 (failure). I modeled the effects of infrastructure proximity on nest density using negative binomial and Poisson distributions; I determined which of

these was most appropriate for each dataset by calculating the ratio of deviance to degrees of freedom.

For nesting success analyses, I ran a preliminary temporal model—including year, Julian date, and a quadratic Julian date term—to determine which fixed-effect nuisance variables should be included in subsequent models. Additionally, because I collected data from the same sites over multiple years, I included the site variable as a random effect in GLMM models. I used AIC (Akaike 1974) to determine if inclusion of the random variable improved model fit. To allow for consistency in assessing model fit among the diversity of my different analytical models, I used Δ AIC to provide a relative measure of goodness of fit for my models, when compared with the appropriate null model (i.e., models that included nuisance variables only). I considered models acceptable when main effects improved model fit by $>2 \Delta$ AIC (Arnold 2010). I subscribed to a frequentist approach (null hypothesis significance testing) in all other aspects of analysis (Mundry 2011; see also for comparison of frequentist and information theoretic approaches), and thus evaluated effects of independent variables based on significance. I did not use AIC to select model fixed-effects or compare models.

3.2.7 Analysis of nesting success—To assess the nesting success among the species of the mixed-grass prairie songbird community, I combined nests of all species. I did not include nests that were abandoned (unless abandonment was triggered by partial predation), or nests that failed as a consequence of weather events. I also analyzed species-specific nesting success for Savannah sparrow (*Passerculus sandwichensis*), vesper sparrow (*Pooecetes gramineus*), chestnut-collared longspur, western meadowlark (*Sturnella neglecta*), and Sprague's pipit. I did not analyze nesting success of clay-

colored sparrow (*Spizella pallida*), horned lark (*Eremophila alpestris*), Brewer's blackbird (*Euphagus cyanocephalus*), Baird's sparrow (*Ammodramus bairdii*), or grasshopper sparrow (*A. savannarum*) separately because there were too few nests of these species ($n < 25$); however, I included them in combined models. Only one Sprague's pipit nest was found in 2012, so I dropped this nest from species-specific analysis to allow the inclusion of the year variable in this species-specific analysis. Julian date variables were not significant in preliminary analysis of nesting success for any species, and consequently were not included in any global survival models. Year was included for all species combined, Savannah sparrow, and Sprague's pipit models. The random effect, site, did not improve model fit in any species and was not included.

I modeled nesting success using logistic exposure (Shaffer 2004). This was accomplished in R using the lme4 package (Bate et al. 2014) in combination with modified logit link-function code provided by Shaffer (2004). For each species, I developed three models; I chose to keep site scale, local scale and vegetation variables in separate models to reduce collinearity and avoid over-parameterizing small datasets, and because sample sizes differed among some variables.

Site-scale infrastructure—I ran three models that included a single main effect each (in addition to nuisance variables): either site type (control or infrastructure), activity (control, inactive infrastructure or active infrastructure), or structure type (e.g., control, pumpjack, screw pump, or compressor). To determine the potential effect of structure power source, I also ran a model that contained a structure power variable (generator or grid) and a structure activity variable (active or inactive). This analysis was conducted only on nests at screw pump sites, because there were no active electric

pumpjacks or generator-powered compressor stations in my study for comparison. I could not conduct the structure type or power source analyses for species with small sample sizes (i.e. chestnut-collared longspur, western meadowlark, and Sprague's pipit), because replicates were insufficient at some levels of variables. I also ran a model including camera (camera or no camera) and nest stage (egg or nestling) for chestnut-collared longspur (as cameras were deployed primarily on CCLO nests) to determine if cameras negatively impacted their nest survival.

Local-scale infrastructure—I analyzed nesting success at the local scale with a model that tested whether proximity of nests to infrastructure features influenced nesting success. This model included two main effects (in addition to nuisance variables): distance to nearest road and distance to nearest infrastructure (i.e. structure). For this analysis, I trimmed the dataset to include only nests that were within 1 km or less of infrastructure, as data were sparse above this distance.

Vegetation and nesting success—I analyzed the effects of vegetation characteristics on nesting success and I included six fixed variables (in addition to nuisance variables): average stem-height, average stem-density, average litter-depth, percent bare ground cover, percent shrub cover, and percent exotic species cover. I dropped the shrub variable from smaller datasets (as listed above), as shrub cover was minimal at nesting sites for all species except vesper sparrow and clay-colored sparrow.

3.2.8 Analysis of nest density—I analyzed nest density using generalized linear models and analyzed only all species combined, Savannah sparrow and vesper sparrow, as sample size was insufficient for other species. I excluded clay-colored sparrow from the combined model because they exclusively nested in shrubs and could have biased the

results, as distribution of nests for this species would likely have been determined by the presence of large shrubs at study sites and not model main effects. I ran a single nest density model for each dataset and it contained four variables: year, site type, distance bin, and the interaction of site type and distance bin. The distance bin variable described the density of nests in four distance bins (0-100 m, 101-200 m, 201-300 m, and 301-400 m) extending from the center of study plots, and was treated as a continuous variable during analysis. Nests beyond 400 m from center points were excluded from this analysis because of sparse data beyond this distance. Random effects did not improve model fit for any species. Only nests found during nest surveys on specified nesting plots were included in this analysis ($n = 744$), because inclusion of incidental nests would have violated search effort assumptions.

3.3 RESULTS

3.3.1 General results—During the 2012-2014 breeding seasons, 813 songbird nests of 10 species were located and monitored. Nests of Savannah sparrow comprised 59% of all nests found. All species were included in combined analyses, but five species were analyzed individually as well. These included: Savannah sparrow (SAVS) ($n = 481$), vesper sparrow (VESP) ($n = 118$), chestnut-collared longspur (CCLO) ($n = 78$), western meadowlark (WEME) ($n = 39$), and Sprague's pipit (SPPI) ($n = 31$).

Across all years, 53% of nests were successful (apparent success), 41% were depredated, and 3% were abandoned (Table 3.2). Nesting success differed among years and was highest in 2013 (58%) and lowest in 2012 (42%) (Table 3.2). Abandonment was notably higher for chestnut-collared longspur (10%) than in other species. However, in five cases during 2013 and 2014, abandonments may have been triggered by placement of nest cameras (Table 3.2); all of these occurred during egg stage. For chestnut-collared longspur nests with cameras, abandonment was 12% (Table 3.3). Consequently, we did not place additional cameras on nests during egg stage. During 2013 and 2014, brood parasitism by brown-headed cowbirds was 8% overall, with clay-colored sparrow ($n = 19$) appearing to experience the highest frequency of parasitism (32%) (Table 3.4). Parasitism data were not collected in 2012.

During 2013 and 2014, video surveillance systems were placed at 41 chestnut-collared longspur nests and 4 Sprague's pipit nests (Table 3.3). Cameras captured a total of 8 nest predation events by 6 different predator species (Table 3.5). Three predations occurred at egg stage and 5 at nestling stage. The most frequently documented predator appeared to be northern grasshopper mouse (*Onychomys leucogaster*) ($n = 3$) (Table 3.5).

Daily precipitation and air temperature for May-August 2012-2014, and cumulative annual precipitation data for 2006-2014 were obtained from the Brooks, AB weather station (Government of Alberta 2015). Overall, daily precipitation ranged from 0-42 mm, reaching its maximum in June 2012 (Appendix IV), and average daily temperature ranged from 1- 25°C (Appendix V). Total precipitation received during the months of May-August was highest in 2012 (275 mm), intermediate in 2013 (231 mm), and lowest in 2014 (186 mm). The cumulative annual precipitation of years sampled ranged from 273.4-387.9 mm with the highest precipitation occurring in 2012 (Appendix III). Mean daily temperatures during the months of May-August were consistent among years and averaged $16.1^{\circ}\text{C} \pm 0.37$ over the three months combined.

Table 3.2. Fates and abundances of grassland songbird nests found in southeastern Alberta, 2012-2014. Percentages refer to proportion of total nests found of species in given year

	Savannah sparrow (n = 481)			Vesper sparrow (n = 118)			Chestnut-collared longspur (n = 78)			Western meadowlark (n = 39)			Sprague's pipit (n = 31)		
	2012	2013	2014	2012	2013	2014	2012	2013	2014	2012	2013	2014	2012	2013	2014
Successful (%)	23 (56)	107 (60)	132 (50)	7 (50)	33 (70)	31 (54)	5 (45)	20 (47)	14 (58)	1 (33)	7 (42)	9 (47)	1 (100)	11 (65)	8 (62)
Depredated (%)	10 (24)	61 (34)	120 (46)	6 (43)	13 (28)	24 (42)	5 (45)	18 (42)	7 (29)	2 (67)	8 (47)	9 (47)	0	5 (29)	5 (38)
Weather/other (%) ¹	0	2 (1)	5 (2)	0	0	1 (2)	0	0	0	0	0	0	0	0	0
Abandoned (%)	1 (2)	8 (4)	5 (2)	0	0	0	0	5 [†] (12)	3 [‡] (13)	0	2 (12)	1 (5)	0	0	0
Unknown fate (%)	7 (17)	0	0	1 (7)	1 (2)	1 (2)	1 (9)	0	0	0	0	0	0	1 (6)	0
Total nests found	41	178	262	14	47	57	11	43	24	3	17	19	1	17	13
Yearly % ²	42.3	54.4	67.4	14.4	14.4	14.7	11.3	13.1	6.2	3.1	5.2	4.9	1.0	5.2	3.3
	Clay-colored sparrow (n = 19)			Horned lark (n = 6)			Other ³ (n = 7)			Unknown (n = 34)			Total (n = 813)		
	2012	2013	2014	2012	2013	2014	2012	2013	2014	2012	2013	2014	2012	2013	2014
Successful (%)	0	3 (30)	5 (56)	0	2 (67)	1 (33)	0	3 (75)	1 (100)	4 (16)	3 (38)	1 (100)	41 (42)	189 (58)	201 (52)
Depredated (%)	0	7 (70)	4 (44)	0	1 (33)	1 (33)	1 (50)	1 (25)	0	17 (68)	5 (62)	0	41 (42)	119 (36)	170 (44)
Weather/other (%) ¹	0	0	0	0	0	0	0	0	0	0	0	0	0	2 (1)	6 (2)
Abandoned (%)	0	0	0	0	0	1 (33)	0	0	0	0	0	0	1 (1)	15 (6)	10 (3)
Unknown fate (%)	0	0	0	0	0	0	1 (50)	0	0	4 (16)	0	0	14 (14)	2 (1)	1 (0)
Total nests found	0	10	9	0	3	3	2	4	1	25	8	1	97	327	389
Yearly % ²	0.0	3.1	2.3	0.0	0.9	0.8	2.1	1.2	0.3	25.8	2.4	0.3	-	-	-
Total nests	97	327	389	813											
% of overall ⁴	11.9	40.2	47.8	-											

1 Nest failure caused by weather, trampling, or unknown causes
2 Percentage of total nests found in given year belonging to given species
3 Brewer's blackbird, Baird's sparrow, and grasshopper sparrow nests
4 Percentage of all nests found in given year out of all nests found in all years
† Three abandonments triggered by nest cameras
‡ Two abandonments triggered by nest cameras

Table 3.3. Chestnut-collared longspur and Sprague's pipit nests monitored with cameras in southeastern Alberta, 2013-2014.

	Chestnut-collared longspur			Sprague's pipit			Total		
	2013	2014	Total	2013	2014	Total	2013	2014	Total
Nests monitored	43	24	67	17	13	30	60	37	97
Predation events (%)	18 (42)	7 (29)	25 (37)	5 (29)	5 (38)	10 (33)	23 (38)	12 (32)	35 (36)
Cameras deployed (%) ¹	20 (47)	21 (88)	41 (61)	4 (24)	0	4 (12)	24	21	45 (46)
Abandonments (%) ²	3 (15)	2 (5)	5 (12)	0	0	0	3 (15)	2 (5)	5 (12)
Predations recorded (%) ³	3 (17)	4 (57)	7 (28)	1 (20)	0	1 (10)	4 (7)	4 (11)	8 (8) [†]

1 Percentage of nests at which cameras were deployed

2 Percentage of nests with cameras that abandoned

3 Percentage of predation events recorded by cameras

† In two additional cases, predator was not recorded because of camera malfunction

Table 3.4. Frequencies of brood parasitism in grassland songbird nests in southeastern Alberta, 2013-2014. Western meadowlark parasitism may be underestimated, as this species is known to reject cowbird eggs.

	Nests parasitized (%)					
	2013	<i>n</i>	2014	<i>n</i>	Both years ¹	<i>n</i>
Savannah sparrow	21 (12)	178	17 (6)	262	38 (9)	440
Vesper sparrow	7 (15)	47	2 (4)	57	9 (9)	104
Chestnut-collared longspur	2 (5)	43	1 (4)	24	3 (2)	67
Western meadowlark	0	17	2 (11)	19	2 (6)	36
Sprague's pipit	0	17	0	13	0	30
Clay-colored sparrow	5 (50)	10	1 (11)	9	6 (32)	19
Horned lark	0	3	0	3	0	6
Other ²	0	4	0	1	0	5
Unknown	1 (13)	8	0	1	1 (11)	9
Total	36 (11)	327	23 (6)	389	59 (8)	716

1 Parasitism data was not collected in 2012

2 Brewer's blackbird, Baird's sparrow, and grasshopper sparrow

Table 3.5. Predator species recorded depredeating grassland songbird nests in southeastern Alberta, 2013-2014.

Predator				
Common name	Scientific name	2013	2014	Total
Northern grasshopper mouse	<i>Onychomys leucogaster</i>	3	0	3
American Badger	<i>Taxidea taxus</i>	1	0	1
Domestic cattle	<i>Bos taurus</i>	0	1	1
Garter snake spp.	<i>Thamnophis spp.</i>	0	1	1 [†]
Northern Harrier	<i>Circus cyaneus</i>	0	1	1
Swainson's Hawk	<i>Buteo swainsoni</i>	0	1	1

† In two other cases, garter snakes were recorded at nests following predation or fledging

3.3.2 *Nesting success*—Nesting success was significantly lower at infrastructure sites relative to controls for all species combined ($\beta = -0.3259$, $SE = 0.1298$, p -value = 0.0120), Savannah sparrow ($\beta = -0.4188$, $SE = 0.1677$, p -value = 0.0125), and vesper sparrow ($\beta = -1.2734$, $SE = 0.6006$, p -value = 0.0340) (Appendix VI; Figure 3.4a-c); nest survival at infrastructure sites was reduced by 11%, 12%, and 23%, respectively, for each of these groups. However, survival did not vary significantly between active and inactive sites for any species (Appendix VI). For all species combined and Savannah sparrow, nest survival was lowest at screw pump sites and significantly lower than at controls (17% reduction) ($\beta = -0.5014$, $SE = 0.1413$, p -value = 0.0004; $\beta = -0.5338$, $SE = 0.1848$, p -value = 0.0039) (Appendix VII; Figure 3.5a-b). For all species combined, nests at pumpjacks and compressor stations displayed non-significant reductions in survival of 4% and 5% relative to controls, respectively ($\beta = -0.1075$, $SE = 0.1668$, p -value = 0.5193; $\beta = -0.1514$, $SE = 0.2056$, p -value = 0.4616) (Appendix VII). Similarly, for Savannah sparrow, nest survival was non-significantly reduced at pumpjacks and compressor stations relative to controls by 9% and 13%, respectively ($\beta = -0.2590$, $SE = 0.2079$, p -value = 0.2127; $\beta = -0.3880$, $SE = 0.2764$, p -value = 0.1604) (Appendix VII). For vesper sparrow, structure type did not improve model fit (Appendix VIII).

For all species combined ($\beta = -0.4624$, $SE = 0.1928$, p -value = 0.0165) and Savannah sparrow ($\beta = -0.8474$, $SE = 0.2768$, p -value = 0.0022), nest survival was significantly lower at grid-powered infrastructure sites compared to generator-powered sites (reductions in survival of 16% and 28%, respectively) (Appendix IX; Figure 3.6a-b). For all species combined, model fit was marginal ($\Delta AIC = 1.9$), but I chose to include these results anyway, because they are consistent with Savannah sparrow results and

because model fit was only slightly below my ΔAIC threshold of >2 . For vesper sparrow, model fit was not improved (Appendix VIII). Infrastructure variables at the site scale did not affect survival for chestnut-collared longspur, western meadowlark, and Sprague's pipit (Appendix VI and Appendix VII), and infrastructure variables did not improve model fits for these species (Appendix VIII).

Vesper sparrow was the only species to respond to local-scale infrastructure variables, and nest survival for this species decreased significantly with increasing proximity to structures ($\beta = 0.0035$, $SE = 0.0016$, $p\text{-value} = 0.0289$) (Appendix X; Figure 3.7). Nest survival was invariant with respect to proximity of both structures and roads for all other species (Appendix VIII).

The presence of nest cameras did not affect nesting success in chestnut-collared longspur (Appendix VI). Overall, vegetation characteristics had little effect on nesting success (Appendix XI), and vegetation variables did not improve model fit relative to null models for any species (Appendix VIII).

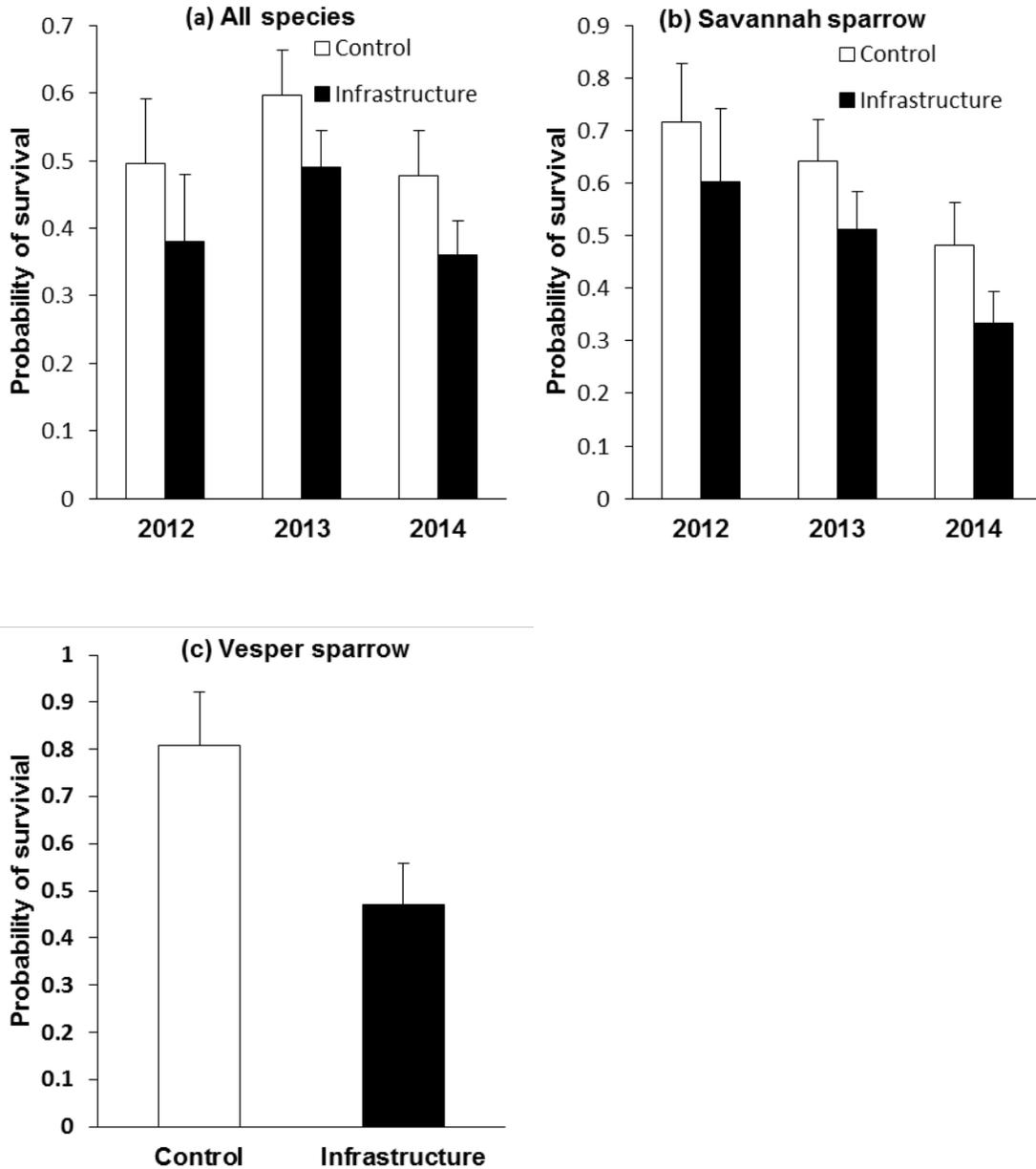


Figure 3.4. The effects of oil and natural gas infrastructure on songbird nesting success in southeastern Alberta, 2012-2014: (a) all species combined; (b) Savannah sparrow (*Passerculus sandwichensis*); (c) vesper sparrow (*Pooecetes gramineus*). Vesper sparrow nesting success did not differ among years. Probability of survival is significantly lower at infrastructure sites in all groups pictured. Error bars denote upper 90% confidence intervals.

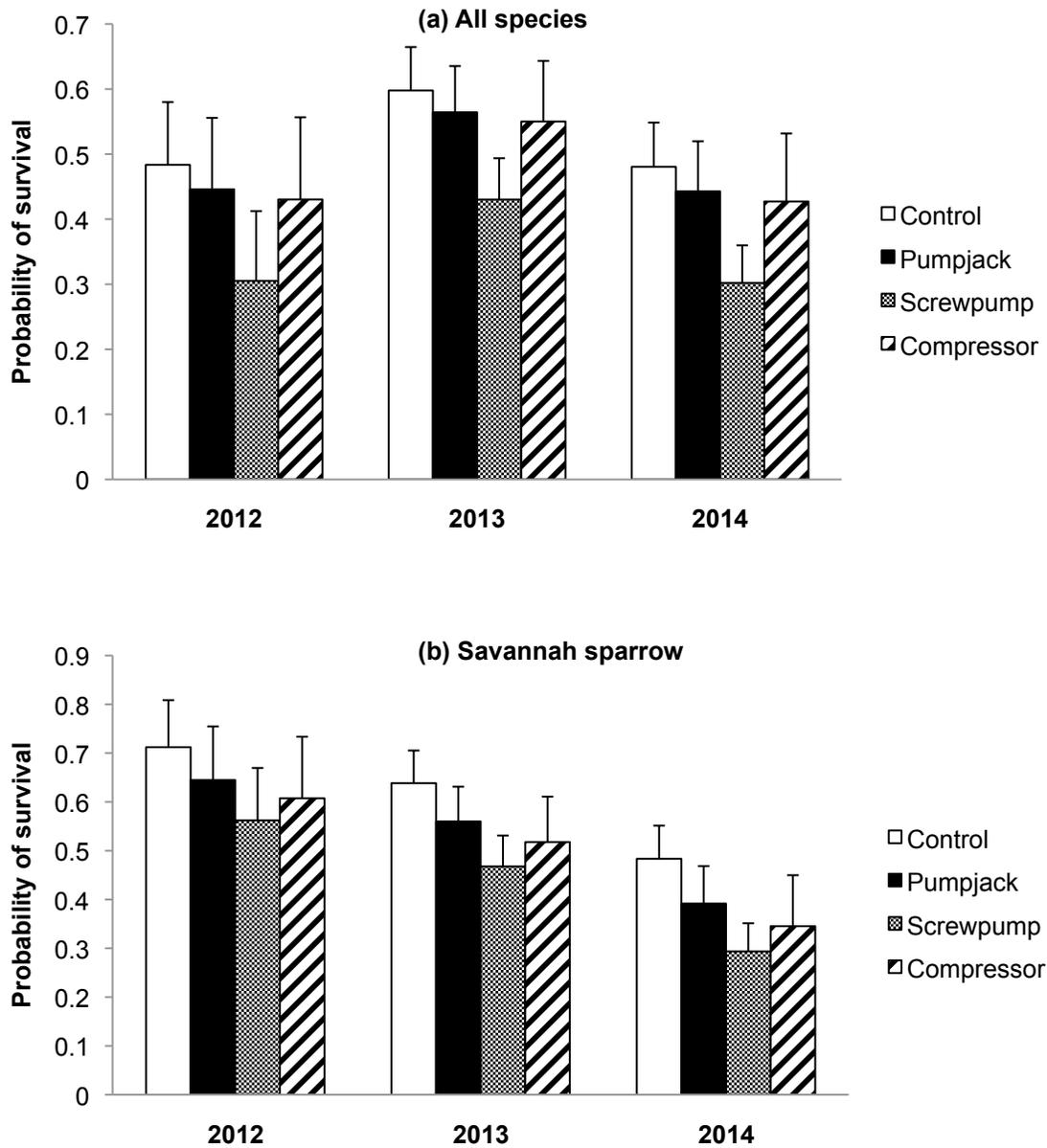


Figure 3.5. The effects of structure type on songbird nesting success in southeastern Alberta, 2012-2014: (a) all species combined and (b) Savannah sparrow. Probability of survival is significantly lower at screw pump sites relative to controls for both figures. Error bars denote upper 90% confidence intervals.

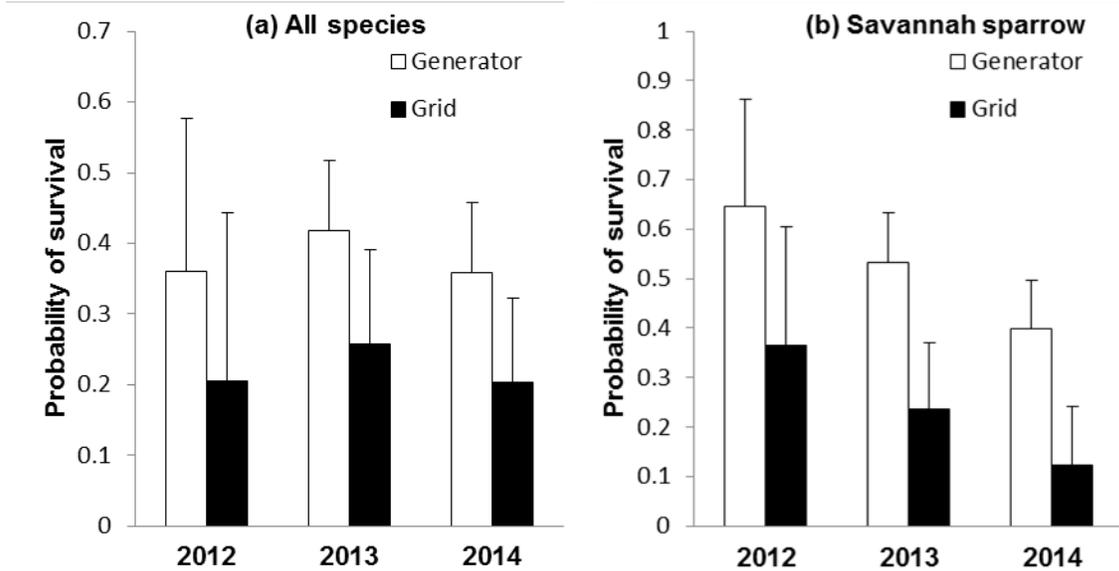


Figure 3.6. The effects of structure power source on songbird nesting success in southeastern Alberta, 2012-2014: (a) all species combined and (b) Savannah sparrow (*Passerculus sandwichensis*). Probability of survival is significantly lower at grid-powered sites compared to generator powered-sites. Error bars denote upper 90% confidence intervals.

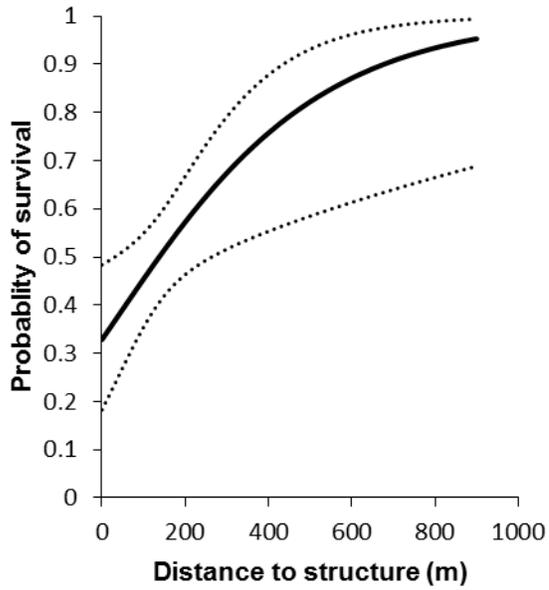


Figure 3.7. The effects of structure proximity on vesper sparrow (*Pooecetes gramineus*) nesting success in southeastern Alberta, 2012-2013. Nesting success was significantly reduced with increasing proximity to structures, and did not differ among years. Dotted lines denote 90% confidence intervals.

3.3.3 *Nest density*—Vesper sparrow was the only species to display a difference in nest density trends between control and infrastructure sites (Appendix XII; Appendix XIII). Vesper sparrow density was significantly higher with increasing proximity to infrastructure center points relative to control center points ($\beta = -0.4601$, $SE = 0.2457$, p -value = 0.0612) (Figure 3.8). I consider nest density results to be apparent nest density only, as rope-drag surveys likely did not detect 100% of nests presents (Giovanni et al. 2011).

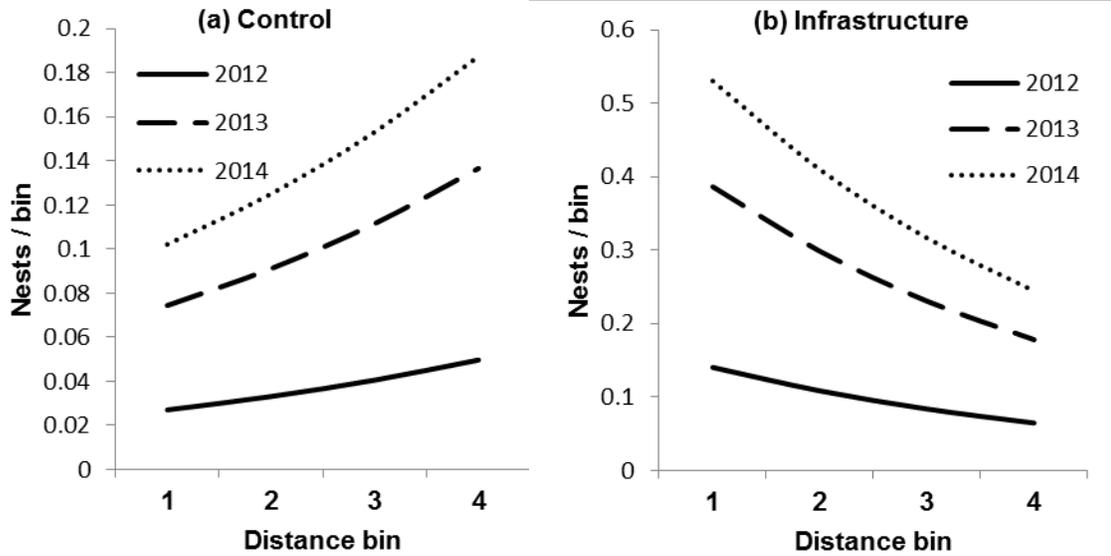


Figure 3.8. The effects of proximity to infrastructure on apparent vesper sparrow (*Pooecetes gramineus*) nest density in southeastern Alberta, 2012-2014: (a) control sites and b infrastructure sites. Distance bins 1,2,3 and 4 correspond to distance intervals of 0-100, 101-200, 201-300, and 301-400 m from site center points, respectively. Nest density differed significantly between control and infrastructure sites.

3.4 DISCUSSION

My results support the hypothesis that oil infrastructure negatively impacts the nesting success of grassland songbirds by increasing frequencies of nest predation. However, this effect varied by both species and infrastructure type, with screw pumps and grid-powered structures correlating with greater reductions in nesting success. Contrary to predictions, noise and human activity associated with infrastructure did not explain effects of wells on nesting success, suggesting that effects were driven by the physical impact of infrastructure. Additionally, the presence of infrastructure resulted in changes in nest density of vesper sparrows.

Nesting success—The elevated predation frequencies observed at the site scale in my study are best explained as a consequence of broad changes to the predator community driven by high overall habitat disturbance in these areas. Generalist predators often reach peak abundance in highly fragmented landscapes (Heske et al. 1999; Dijak and Thompson 2000), and these predators may take advantage of linear features and fragmentation associated with energy infrastructure. It is estimated that for each oil well built, an additional 100 ha of road is constructed (Riley et al. 2012). This network of access roads may provide predators from nearby agricultural and urban systems access to interior mixed-grass prairie habitat. Roads can facilitate the movement of mammalian predators (Frey and Conover 2006), and increases in human development on the landscape may result in increased predation pressure on nesting songbirds (Klug et al. 2009). A recent study of nest survival in sage-obligate songbirds nesting near unconventional natural gas infrastructure showed that while proximity to infrastructure did not affect predation rates at the local scale, nest predation frequencies were greater in

areas where a high proportion of habitat had been disturbed by energy infrastructure at the landscape level (Hethcoat and Chalfoun 2015a). Furthermore, these high-disturbance areas also displayed increased small-mammal predator activity (Hethcoat and Chalfoun 2015b).

Energy infrastructure is also known to provide food and habitat subsidies to certain predators. For example, unconventional natural gas exploration in Wyoming facilitated the expansion of common raven into previously unoccupied sage flats by providing additional nesting substrate, resulting in increased predation of greater sage-grouse nests (Bui et al. 2010). Similarly, research shows that artificial substrates such as buildings and transmission lines provide additional nesting locations for ferruginous hawk (*Buteo regalis*), and that these nests experience greater success and productivity than those built on natural substrates (Wallace et al. 2015). Similarly, in mixed-grass prairie, a gps tracking study of burrowing owl (*Athene cunicularia*) found that this species spends a large proportion of time nears fences and posts, likely because it makes use of these artificial perches (Scobie et al. 2014). At my study sites, several instances of ravens and raptor species nesting and perching on infrastructure were observed (J. Bernath-Plaisted, per. obs). Furthermore, roads may provide carrion food subsidies for generalist predators (Kristan et al. 2004).

However, evidence of increased nest predation near infrastructure at my study sites is in contrast to a similar study, also conducted at oil infrastructure sites in Alberta's mixed-grass prairie, which found no direct effect of infrastructure on the nesting success of the same species (Ludlow et al. 2015b). It is possible that the smaller sample size analyzed by Ludlow et al. (2015b) reduced power, or that the presence of an effect was

diluted because Ludlow et al. (2015b) combined survival data collected from oil wells and conventional natural gas wells. Shallow natural gas wells in this region are small, relatively quiet structures, usually without connection to permanent access roads (Koper et al. 2014). Previous research suggests that shallow natural gas wells may not reduce nesting success, while the larger footprint, vertical structure, and road networks associated with oil wells can lead to a decrease in the productivity of grassland songbirds (Yoo 2014).

Nest survival was generally lowest at screw pump sites, and at facilities powered by the electric grid. However, because survival did not differ between active and inactive sites, these patterns cannot be explained by differences in noise or human activity among structures. Instead, they may be driven by the degree of surrounding habitat disturbance, including the presence of linear features, and size of infrastructure footprint. Many of the screw pump facilities in my study were grid powered, and require transmission lines. Transmission lines may serve as perches for raptor species and may subsequently increase nest predation frequencies (Lammers and Collopy 2007).

Nonetheless, this does not explain why survival at screw pump sites was lower than at compressor stations, which are also grid powered and have the largest physical footprint of any structure included in the study. One possible explanation is that compressor stations may be loud enough, or large enough, to deter some predator species (Francis et al. 2009). Francis et al. (2009) found that not only was songbird nesting success greater at sites with higher-amplitude noise treatments, but abundance of common avian nest predators was also reduced at these locations. There is evidence that some mammals may be deterred by anthropogenic noise as well (Francis et al. 2012). If

predator deterrence as a consequence of noise occurred at my study sites, infrastructure with intermediate footprint and noise production levels, such as screw pumps, might have the largest impact on nest survival; intermediate disturbances, the presence of linear features, and a significant infrastructure footprint may facilitate movement of and attract nest predators of several taxa, but at even higher disturbances, excessive noise, traffic, and activity might deter them (Francis et al. 2009).

Vesper sparrow was the only species that displayed decreasing nesting success with increasing proximity to structures. It is unclear why local scale edge effects were detected for vesper sparrows but not other species. However, vesper sparrow nest density was higher within 100 m of infrastructure than at control points (see below). Therefore, the effects of infrastructure may have been more apparent for this species, or nest predators may have responded positively to high nest density (Gates and Gysel 1978). Alternatively, it is possible that an edge effect was also present for Savannah sparrow, but it occurred at too large a spatial scale to be detected within 800-m by 200-m nest plots. The spatial extent of edge effects is highly variable and can affect large areas of habitat or may be relatively limited (Winter et al. 2000; Flaspohler et al. 2001; Renfrew and Ribic 2003). Furthermore, edge effects are often driven by specific predator behaviors (Lahti 2001; Ellison et al. 2013), and these behaviors may have affected different species of nesting songbird differently. For example, some predators may prey disproportionately on vesper sparrow nests relative to Savannah sparrow nests, as vesper sparrow nests are often located on exposed bare ground and therefore may be less concealed than nests of other grassland songbird (Wray and Whitmore 1979; Ludlow et al. 2015b).

It is also possible that higher quality vesper sparrow males defend territories further from infrastructure, forcing lower quality individuals to nest near infrastructure (Fretwell and Lucas 1970); in this case, increased nesting failure near infrastructure could be a consequence of individual quality. The possibility of demographic effects such as this is currently being investigated by fellow researchers at the University of Manitoba. Regardless, further research is required to determine exactly how energy infrastructure alters the predator community, and what predators are responsible for increased predation of nests belonging to different species.

Infrastructure did not appear to have any impact on the nesting success of chestnut-collared longspur, western meadowlark, or Sprague's pipit. It is possible that nest predators of these species are generalists that encounter nests opportunistically, rather than through directed search efforts (Vickery et al. 1992b). Consequently, patterns in nest predation may be difficult to predict. Alternatively, both chestnut-collared longspur and Sprague's pipit are grassland specialists (Bleho et al. 2015; Davis and Robbins 2014) and may be edge-sensitive (Koper et al. 2009; Sliwinski and Koper 2012). Therefore, it is possible that these species select high-quality habitat farther from infrastructure disturbance, thus negating its effects. Sprague's pipit's low use of habitat near natural gas wells and unconventional oil wells supports this idea (Hamilton et al. 2011; Thompson et al. 2015). Concurrent research conducted at my study sites also suggests that Sprague's pipit abundances decline near conventional oil wells (H. Nenner, University of Manitoba, unpublished data).

It is also possible I failed to detect effects as a consequence of small sample sizes in these species. Lastly, it is important to note that I did not investigate all potential

effects of infrastructure on reproductive success, such as post-fledging survival, annual fecundity, and lifetime reproductive success (Martin 1995). To my knowledge, there is no existing research on the potential long-term effects of energy infrastructure on productivity and recruitment in songbirds.

Nest density—Nest density was not affected by infrastructure for most species. Although nesting success was reduced at infrastructure sites, it is possible that many species do not recognize these areas as poor habitat. Research on edge effects in nesting grassland songbirds suggests that while some species attempt to avoid nesting in habitat where predation frequencies are elevated, others do not (Winter et al. 2000). Furthermore, given that elevated predation frequencies occurred at the site scale for some species, it may not be surprising that nesting densities were invariant at the local scale, as avoidance is ineffectual when predator effects extend beyond immediate edges (Renfrew et al. 2005).

Only the nest density of vesper sparrow appeared to be affected by the presence of infrastructure, and contrary to expectations, this species showed a preference for nesting in close proximity to structures. Though it is possible that this nest density pattern is an artifact of high nesting failure for vesper sparrow nests near infrastructure resulting in repeated nesting attempts and thus inflating density estimates, adult vesper sparrow abundances on my study sites were also higher at infrastructure sites relative to controls (H. Nenninger, University of Manitoba, unpublished data). Furthermore, this finding is consistent with existing research in which vesper sparrow was attracted to recently constructed wind turbines in grassland habitat (Johnson et al. 2000). Vesper sparrow's attraction to infrastructure may be driven by its tendency to nest on bare ground and

incorporate perches into its territories (Rodenhouse and Best 1983; Best and Rodenhouse 1984; Jones and Cornley 2002). Both of these features are readily available near infrastructure, as lease sites are frequently characterized by exposed ground (Nasen et al. 2011), and structures themselves and associated fences provide numerous perches.

It may seem surprising that vesper sparrow would select nesting habitat where predation rates are extremely elevated, but songbird nesting preferences sometimes reflect selection of certain habitat features more strongly than fitness consequences (Chalfoun et al. 2007). At my study sites, energy infrastructure appears to act as an ecological trap (Gate and Gysel 1978) for nesting vesper sparrow by providing features that individuals perceive as attractive in low quality habitat (Patten and Kelly 2010). Ecological traps have been previously documented in grassland songbirds, including notable cases involving nesting in grasslands where haying occurs during the breeding season (Gilroy and Sutherland 2007) and increased predation pressure on managed grasslands (Shochat et al. 2005). Grassland songbirds appear to be susceptible to such traps as a consequence of strong preferences for certain vegetation structures (Schlaepfer et al. 2002; Gilroy and Sutherland 2007). Ecological traps in which individuals select low quality habitat over high can have serious implications for the persistence of avian populations, even when the proportion of poor habitat on the landscape is relatively low (Donovan and Thompson 2001). Given that vesper sparrow nests suffer a reduction in survival of 21% at infrastructure sites, to which they are apparently attracted, high densities of infrastructure features on the mixed-grass prairie could pose a serious threat to local populations of this species.

Predators—Video identification of nest predators at my study sites revealed that the predator community in Alberta’s mixed-grass prairie is diverse and that, as in other northern grassland systems, mammals and avian species may be more important nest predators than snakes (Pietz et al. 2012). Additionally, cattle may also play a role in nest predation at infrastructure sites, as they were among nest predators captured on video in this study, and there is evidence that cattle may concentrate near infrastructure (Koper et al. 2014). Further research should attempt to quantify the behavior and densities of predators in areas affected by energy development.

3.5 CONCLUSION

The results of this study suggest that oil extraction infrastructure and natural gas compressor stations negatively impact the reproductive success of some species of grassland songbird. Nest survival of both Savannah sparrow and vesper sparrow was reduced substantially by the presence of infrastructure, and it is possible that infrastructure may also act as an ecological trap for nesting vesper sparrows. The effects of infrastructure varied by structure type and species, and overall, grid-powered screw pumps appear to be the most detrimental structure, perhaps because they are associated with transmission lines, or because their footprints are larger than those of other wells. Interestingly, the activity status of structures did not appear to affect nesting success, suggesting that the physical footprint of energy infrastructure, and not the anthropogenic noise it produces, is responsible for observed declines in nesting success.

The most likely explanation for increased predation frequencies of grassland songbird nests near energy infrastructure is alterations to predator composition and behavior as a result of physical infrastructure and associated fragmentation and habitat

disturbance. Although roads had little impact on nest survival at the local scale for any species, roads fragment and disturb large amounts of habitat in areas of high infrastructure density and may contribute to landscape level changes in the predator community. Currently, it remains unknown what predators are responding positively to infrastructure on the landscape. Video identification of nest predators at study sites revealed a diverse predator community in Alberta's mixed-grass prairie, and further sampling of nest predators in areas highly fragmented by infrastructure is essential to understanding the mechanism driving elevated predation rates and in selecting appropriate management strategies.

Regardless, nesting success is a key life-history trait that affects population trends (Martin 1992; Martin 1993a), and given the imperiled status of grassland songbirds in North America (Sauer et al. 2014), managers should take immediate steps to address declining nest survival at infrastructure sites. Successful mitigation will require a systematic attempt to reduce the total habitat area affected by infrastructure on the landscape. For example, this could be accomplished to some degree without cessation of oil production by increased use of horizontal drilling to minimize new surface infrastructure (CAPP 2012), reclamation of access roads no longer in use, and burying transmission lines to reduce perch availability for raptor predators.

Lastly, it should be noted that nests of Savannah sparrow and vesper sparrow combined comprised the vast majority of nests monitored in this study, and the lack of effects from infrastructure detected for remaining species may reflect small sample sizes rather than genuine lack of effect. Further large sample-size studies are needed to

determine how infrastructure is affecting these species, especially for the threatened Sprague's pipit and chestnut-collared longspur.

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CHAPTER 4.0: MANAGEMENT IMPLICATIONS

The presence of oil extraction infrastructure and natural gas compressor stations on the landscape appears to result in elevated predation risk for grassland songbird nests. Both Savannah sparrow and vesper sparrow experienced reduced nesting success at infrastructure sites, and vesper sparrow nesting success also declined within sites with increasing proximity to infrastructure. Noise and activity associated with infrastructure did not appear to affect nesting success, and effects of infrastructure on nesting success are most likely caused physical footprint on the landscape. However, surprisingly, vesper sparrow nest density increased with proximity to infrastructure, and it is possible that energy infrastructure may act as an ecological trap for this species.

Given that physical infrastructure is likely responsible for observed increases in predation frequency, managers should seek to reduce the impact of existing and future developments on the surface environment. Mitigation of landscape level habitat disturbance could be partially achieved by measures such as increased use of horizontal drilling techniques to minimize surface disturbance when accessing new oil and gas wells (CAPP 2012), burying transmission lines, and reclaiming abandoned wells and access roads that are no longer in regular use. Following these steps will help to reduce the total number of simultaneous disturbances on the landscape at any given time.

In Alberta, leaseholders are required to reclaim abandoned oil and natural gas wells and all associated surface infrastructure (Government of Alberta 2013; 2014), but there is little regulation of the timeframe in which reclamation must take place. Although site operators are liable for surface reclamation for up to 25 years in Alberta (Government of Alberta 2014), there is often economic incentive for energy companies to suspend dry wells while waiting for new technologies to develop and continue to make lease

payments instead of proceeding with reclamation (Prescott 2013). Furthermore, Alberta faces a growing backlog of abandoned wells that are out of compliance with provincial regulations (Robinson 2010). Mitigation of negative effects oil and gas infrastructure on grassland songbirds will require stronger legislation to ensure that reclamation of abandoned lease sites is timely and properly done.

Additionally, though leaseholders are required to restore vegetation of native grassland lease sites, composition of reclaimed sites is only required to be 70% similar to surrounding habitat (Government of Alberta 2013). It is essential that reclaimed sites are properly re-seeded with native plant species, as disturbances to vegetation can persist for decades following the construction of roads and energy facilities (Simmers and Galatowitsch 2010; Nasen et al. 2011). In particular, managers should continue to make efforts to control the spread of crested wheatgrass (*Agropyron cristatum*), a species commonly associated with energy development (Koper et al. 2014, Ludlow et al. 2015) that has also been linked to decreased nesting success and juvenile survival in Sprague's pipit and chestnut-collared longspur (Lloyd and Martin 2005; Fisher and Davis 2011; Ludlow et al. 2014). Finally, energy companies should consider favoring the use of generator-powered oil extraction structures in place of grid-powered structures wherever possible, as generators do not require transmission lines and appear to have a lesser negative impact on nest survival.

Further research is essential to determine what predators are driving observed trends, and whether these changes to the predator community occur primarily at the landscape or local scale. For example, it is possible that structures and roads attract or subsidize particular predators (Liebezeit et al. 2009; Bui et al. 2010). If this is the case,

managers may be able to take direct steps to deter or control these species. Alternatively, changes to the predator community may occur predominantly as a consequence of increasing fragmentation and habitat disturbance at the landscape level as densities of roads and energy infrastructure continue to increase (Hethcoat and Chalfoun 2015). In this case, the only effective management strategies will be those that reduce the presence of infrastructure on the landscape.

4.1 Literature cited

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APPENDIX I

List of infrastructure study sites in Brooks, AB and associated site use and activity statuses during 2012, 2013, and 2014 breeding seasons, and lengths of road contained.

Project site name	Structure type	Power source	Site use and activity			Road length (m)
			2012	2013	2014	
7	Compressor station	N/A	Inactive	Not used	Not used	1898
29	Compressor station	N/A	Inactive	Not used	Not used	2433
34	Compressor station	N/A	Active	Active	Active	1801
45	Compressor station	N/A	Active	Active	Active	1078
46	Compressor station	N/A	Active	Active	Active	1555
53	Compressor station	N/A	Active	Active	Active	2471
56	Compressor station	N/A	Active	Active	Active	1635
58	Compressor station	N/A	Active	Not used	Not used	751
63	Compressor station	N/A	Active	Not used	Not used	446
11	Pumpjack	Generator	Active	Active	Active	659
79	Pumpjack	Generator	Active	Inactive	Inactive	1830
81	Pumpjack	Generator	Active	Active	Active	696
87	Pumpjack	Generator	Not used	Active	Active	1395
94	Pumpjack	Generator	Active	Active	Active	957
137	Pumpjack	Generator	Not used	Active	Active	868
145	Pumpjack	Generator	Not used	Inactive	Inactive	572
210	Pumpjack	Generator	Active	Active	Not used	1409
220	Pumpjack	Generator	Not used	Active	Active	501
245	Pumpjack	Generator	Active	Active	Active	1122

351	Pumpjack	Generator	Inactive	Inactive	Inactive	682
355	Pumpjack	Generator	Inactive	Not used	Not used	483
356	Pumpjack	Generator	Active	Not used	Not used	1682
357	Pumpjack	Generator	Active	Not used	Not used	1155
362	Pumpjack	Generator	Inactive	Not used	Not used	1661
363	Pumpjack	Grid	Not used	Inactive	Inactive	1543
402	Pumpjack	Grid	Not used	Inactive	Inactive	2297
74	Screwump	Generator	Active	Active	Not used	1449
75	Screwump	Generator	Active	Inactive	Inactive	1138
76	Screwump	Grid	Active	Active	Active	1301
77	Screwump	Grid	Active	Active	Active	1404
80	Screwump	Generator	Active	Active	Active	1255
83	Screwump	Generator	Not used	Not used	Inactive	861
84	Screwump	Generator	Not used	Not used	Inactive	1089
91	Screwump	Generator	Inactive	Not used	Not used	2867
129	Screwump	Generator	Not used	Inactive	Inactive	1094
146	Screwump	Grid	Active	Not used	Not used	2214
147	Screwump	Grid	Active	Active	Active	3030
153	Screwump	Grid	Active	Not used	Not used	1128
154	Screwump	Grid	Not used	Inactive	Inactive	3081
217	Screwump	Generator	Active	Active	Active	1433
228	Screwump	Generator	Active	Not used	Not used	1792
241	Screwump	Grid	Not used	Active	Active	943
247	Screwump	Generator	Not used	Active	Inactive	1693
353	Screwump	Grid	Active	Active	Active	1123
401	Screwump	Grid	Not used	Active	Active	2074
404	Screwump	Grid	Not used	Active	Active	2210

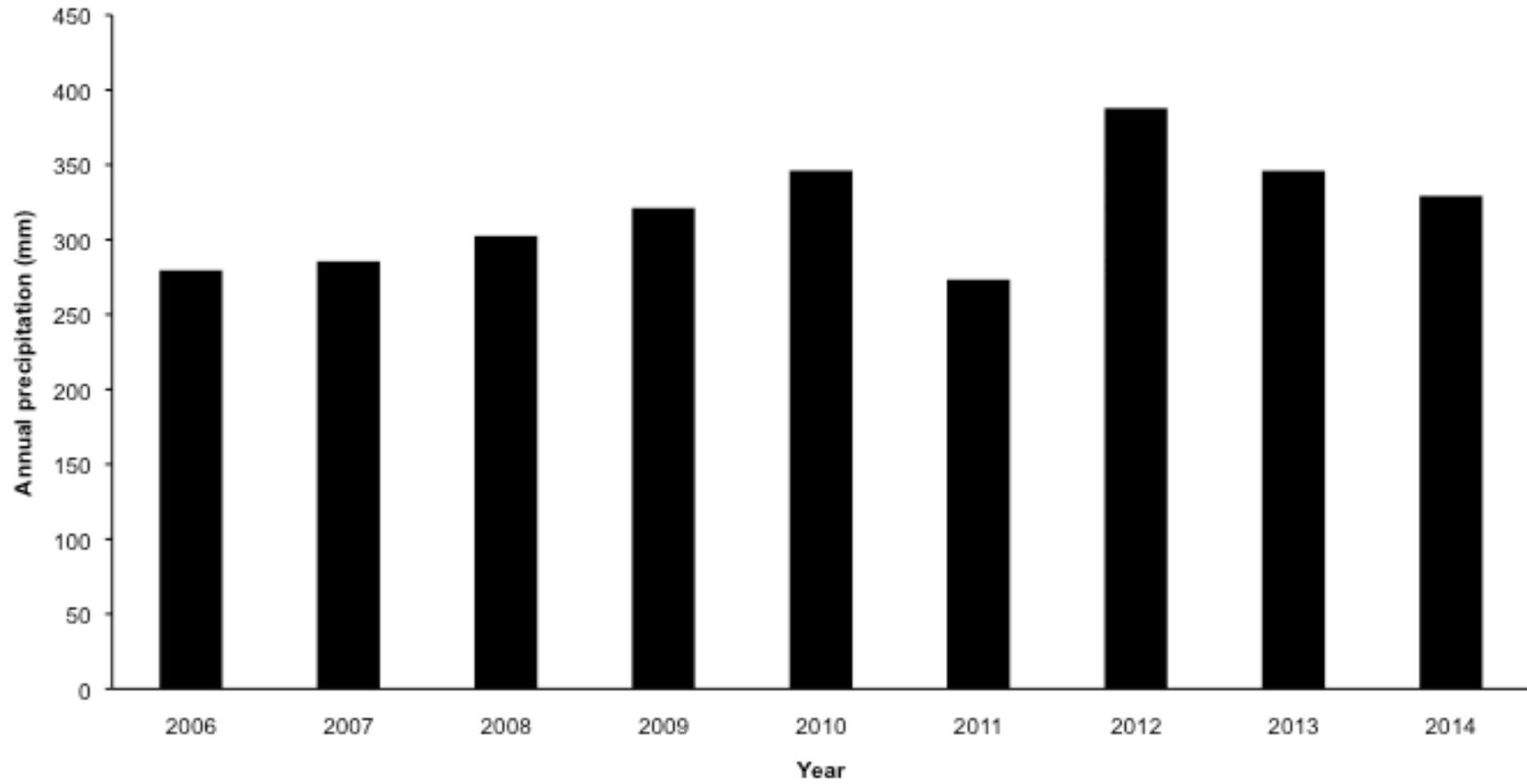
APPENDIX II

List of control study sites in Brooks, AB and site use statuses during 2012, 2013, and 2014 breeding seasons, and lengths of road contained within 500-m radius buffer zone.

Project site name	Site type	Site use			Road length (m)
		2012	2013	2014	
HC2	Control	Used	Not used	Not used	529
HC3	Control	Used	Not used	Not used	1957
C1	Control	Not used	Not used	Used	441
C4	Control	Used	Not used	Not used	359
C5	Control	Used	Not used	Not used	1377
C6	Control	Used	Not used	Not used	1426
C7	Control	Used	Not used	Not used	339
C12	Control	Used	Not used	Not used	1224
C13	Control	Used	Not used	Not used	491
C16	Control	Used	Used	Used	802
C17	Control	Used	Used	Used	491
C18	Control	Used	Used	Used	1019
C19	Control	Not used	Used	Used	1284
C20	Control	Not used	Used	Used	1007
C21	Control	Used	Not used	Not used	513
C23	Control	Used	Not used	Not used	0
C25	Control	Used	Not used	Not used	430
C26	Control	Not used	Used	Used	1717

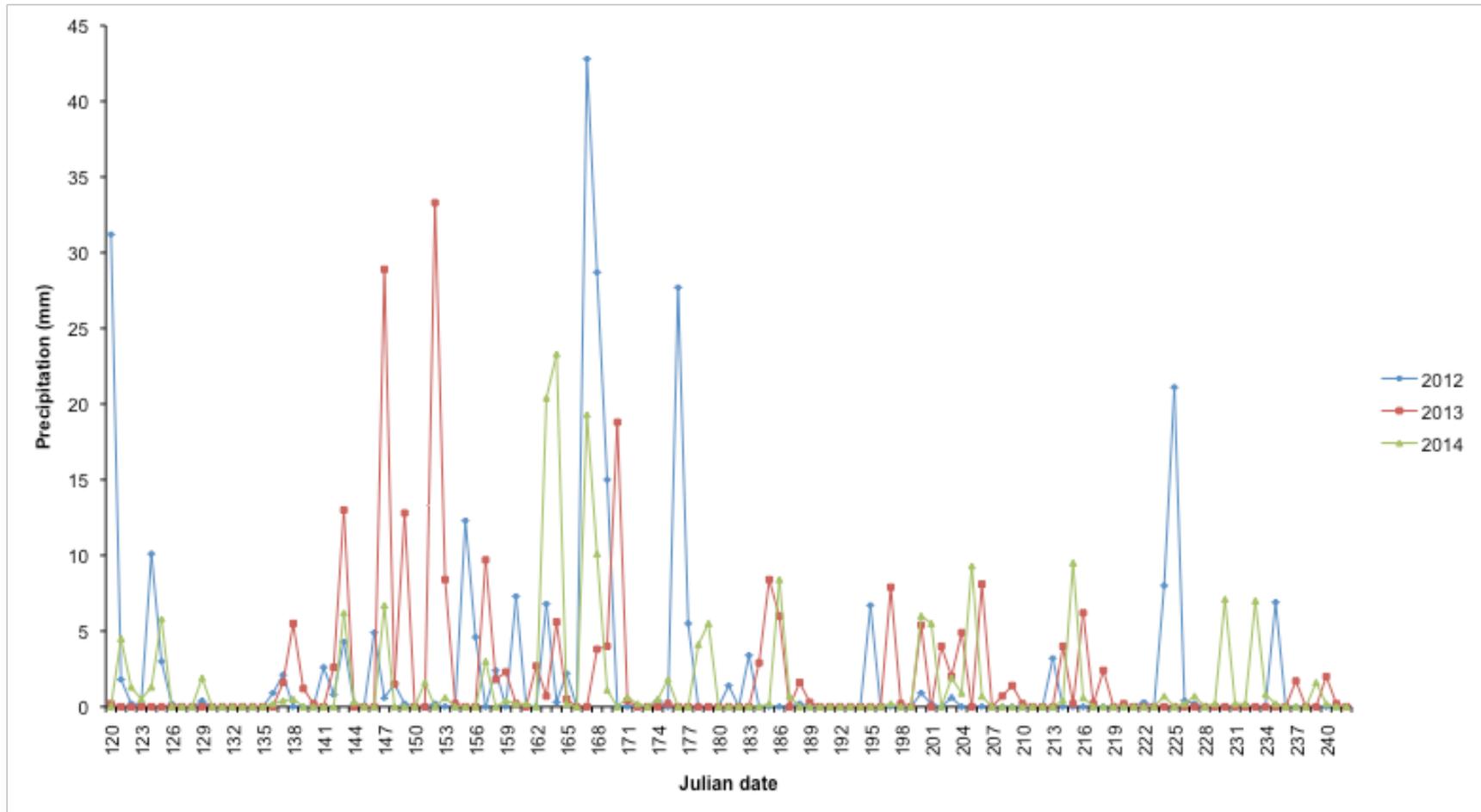
C27	Control	Not used	Used	Used	0
C28	Control	Used	Not used	Not used	464
C34	Control	Not used	Used	Used	432
C35	Control	Not used	Used	Used	600
C36	Control	Not used	Used	Used	0
C37	Control	Not used	Used	Used	913
C38	Control	Not used	Used	Used	600
C210	Control	Not used	Used	Not used	426
JBPC	Control	Not used	Used	Used	736

APPENDIX III



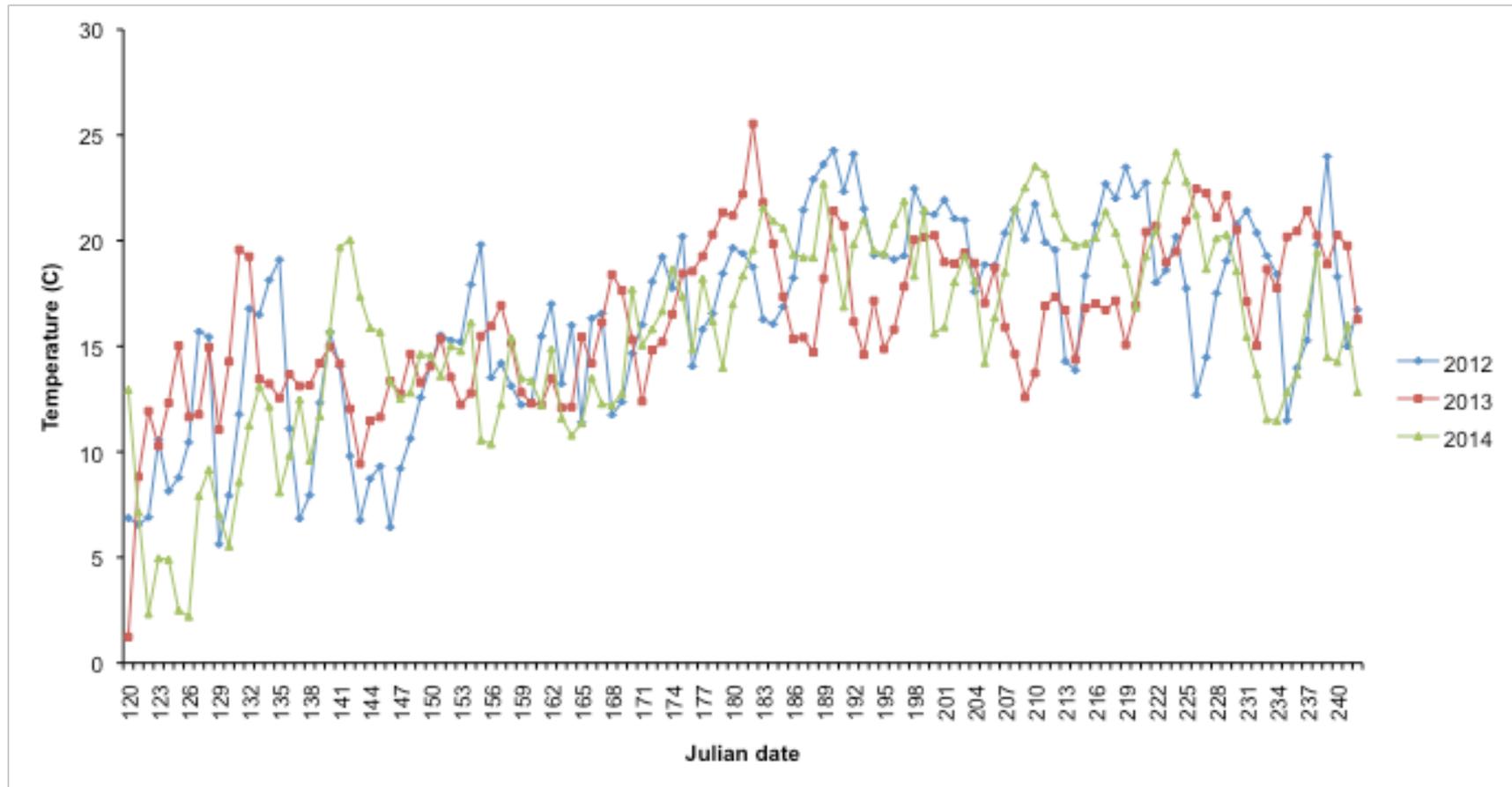
Cumulative annual precipitation in Brooks, AB, 2006-2014.

APPENDIX IV



Daily precipitation in Brooks, AB by Julian date during the months of May-August, 2012-2014.

APPENDIX V



Daily air temperature in Brooks, AB by Julian date for the months of May-August, 2012-2014.

Appendix VI

The effects of year, site type, site activity, and nest cameras on songbird nesting success in southeastern Alberta, 2012-2014.

Species ¹		Site type model			Activity model					Camera model	
		Year ²		Site type ³	Year ²		Activity ³			Camera ⁵	Nest stage ⁶
		2012	2013	Infrastructure	2012	2013	Active	Inactive	Active vs. inactive ⁴	Yes	Nestling
All species (<i>n</i> = 3387)	β	0.0555	0.3668	-0.3259	0.0512	0.3659	-0.3023	-0.3696	-0.0673	N/A	N/A
	SE	0.1784	0.1213	0.1298	0.1786	0.1213	0.1383	0.1556	0.1337	N/A	N/A
	LCL	-0.2295	0.1684	-0.5429	-0.2342	0.1675	-0.5324	-0.6261	-0.2853	N/A	N/A
	UCL	0.3566	0.5675	-0.1158	0.3526	0.5667	-0.0771	-0.1140	0.1547	N/A	N/A
	<i>p</i> -value	0.7555	0.0025	0.0120	0.7745	0.0026	0.0289	0.0175	0.6148	N/A	N/A
Savannah sparrow (<i>n</i> = 2020)	β	0.7925	0.5073	-0.4188	0.7669	0.5084	-0.3361	-0.5743	-0.2383	N/A	N/A
	SE	0.3338	0.1602	0.1677	0.3344	0.1602	0.1802	0.2008	0.1764	N/A	N/A
	LCL	0.2817	0.2471	-0.7007	0.2551	0.2482	-0.6367	-0.9065	-0.5260	N/A	N/A
	UCL	1.3847	0.7746	-0.1481	1.3600	0.7758	-0.0432	-0.2435	0.0558	N/A	N/A
	<i>p</i> -value	0.0176	0.0016	0.0125	0.0218	0.0015	0.0621	0.0042	0.1768	N/A	N/A
Vesper sparrow (<i>n</i> = 553)	β	N/A	N/A	-1.2734	N/A	N/A	-1.3385	-1.1547	0.1839	N/A	N/A
	SE	N/A	N/A	0.6006	N/A	N/A	0.6108	0.6403	0.3333	N/A	N/A
	LCL	N/A	N/A	-2.4281	N/A	N/A	-2.5051	-2.3549	-0.3507	N/A	N/A
	UCL	N/A	N/A	-0.4014	N/A	N/A	-0.4438	-0.1939	0.7498	N/A	N/A
	<i>p</i> -value	N/A	N/A	0.0340	N/A	N/A	0.0284	0.0713	0.5812	N/A	N/A
Chestnut-collared longspur (<i>n</i> = 283)	β	N/A	N/A	0.0587	N/A	N/A	-0.0853	0.2558	N/A	0.6622	0.0435
	SE	N/A	N/A	0.4054	N/A	N/A	0.4418	0.4938	N/A	0.4469	0.4171
	LCL	N/A	N/A	-0.6436	N/A	N/A	-0.8359	-0.5618	N/A	-0.0413	-0.6734
	UCL	N/A	N/A	0.7054	N/A	N/A	0.6354	1.0857	N/A	1.4487	0.7088
	<i>p</i> -value	N/A	N/A	0.8850	N/A	N/A	0.8470	0.6040	N/A	0.1380	0.9170

	β	N/A	0.0010	-0.4086	N/A	0.9054	-0.5591	-0.2879	N/A	N/A	N/A
	SE	N/A	0.6442	0.6683	N/A	0.6954	0.8027	0.7802	N/A	N/A	N/A
Sprague's pipit (n = 166)	LCL	N/A	-0.2442	-1.5706	N/A	-0.2321	-1.9143	-1.5710	N/A	N/A	N/A
	UCL	N/A	1.9124	0.6658	N/A	2.0938	0.7817	1.0700	N/A	N/A	N/A
	p-value	N/A	0.2020	0.5410	N/A	0.1930	0.4860	0.7120	N/A	N/A	N/A
	β	N/A	N/A	0.1457	N/A	N/A	0.2503	-0.0071	N/A	N/A	N/A
	SE	N/A	N/A	0.6455	N/A	N/A	0.6835	0.7090	N/A	N/A	N/A
Western meadowlark (n = 146)	LCL	N/A	N/A	-1.0639	N/A	N/A	-1.0035	-1.2903	N/A	N/A	N/A
	UCL	N/A	N/A	1.1188	N/A	N/A	1.3087	1.1084	N/A	N/A	N/A
	p-value	N/A	N/A	0.8210	N/A	N/A	0.7140	0.9920	N/A	N/A	N/A

1 Sample sizes are given in nesting intervals for logistic exposure

2 2014 is the reference level

3 Control is the reference level

4 Comparison of active and inactive parameter estimates when active is the reference level

5 No camera is the reference level

6 Egg stage is the reference level

Appendix VII

The effects of year and structure type on songbird nesting success in southeastern Alberta, 2012-2014

Species ¹		Year ²		Structure type ³		
		2012	2013	Pumpjack	Screwump	Compressor
All species (<i>n</i> = 3387)	β	0.0088	0.3586	-0.1075	-0.5014	-0.1514
	SE	0.1796	0.1214	0.1668	0.1413	0.2056
	LCL	-0.2785	0.1600	-0.3811	-0.7361	-0.4827
	UCL	0.3118	0.5595	0.1686	-0.2713	0.1944
	<i>p</i> -value	0.9609	0.0031	0.5193	0.0004	0.4616
Savannah sparrow (<i>n</i> = 2020)	β	0.7703	0.4888	-0.2590	-0.5338	-0.3880
	SE	0.3343	0.1606	0.2079	0.1848	0.2764
	LCL	0.2587	0.2279	-0.6013	-0.8417	-0.8288
	UCL	1.3632	0.7568	0.0843	-0.2323	0.0810
	<i>p</i> -value	0.0212	0.0024	0.2127	0.0039	0.1604
Vesper sparrow (<i>n</i> = 553)	β	N/A	N/A	-1.1024	-1.3598	-1.2078
	SE	N/A	N/A	0.6708	0.6133	0.6957
	LCL	N/A	N/A	-2.3382	-2.5291	-2.4713
	UCL	N/A	N/A	-0.0702	-0.4598	-0.1192
	<i>p</i> -value	N/A	N/A	0.1003	0.0266	0.0825

1 Sample sizes are given in nesting intervals for logistic exposure

2 2014 is the reference level

3 Control is the reference level

APPENDIX VIII

Goodness of fit quantified by ΔAIC for all models analyzing songbird nesting success in southeastern Alberta, 2012-2014.

Species	Model	Parameters	ΔAIC
All species	Site 1	Year + site type	4.1
	Site 2	Year + activity	2.9
	Site 3	Year + structure type	9.2
	Site 4	Power + activity	1.9
	Local	Year + structure distance + road distance	+ 2.9
	Vegetation	Year + stem height + stem density + litter depth + bareground + shrub + exotic	+ 6.5
Savannah sparrow	Site 1	Year + site type	4.7
	Site 2	Year + activity	4.4
	Site 3	Year + structure type	2.8
	Site 4	Year + power + activity	6.1
	Local	Year + structure distance + road distance	+ 3.4
	Vegetation	Year + stem height + stem density + litter depth + bareground + shrub + exotic	+ 4.1
Vesper sparrow	Site 1	Site type	4.5
	Site 2	Activity	2.8
	Site 3	Structure type	0.9
	Site 4	Power + activity	+ 3.1
	Local	Structure distance + road distance	4.9
	Vegetation	Stem height + stem density + litter depth + bareground + shrub + exotic	+ 4.8
Chestnut-collared longspur	Site 1	Site type	+ 2.0
	Site 2	Activity	+ 3.4
	Local	Structure distance + road distance	+ 3.1
	Camera	Camera + nest stage	+ 1.4
	Vegetation	Stem height + stem density + litter depth + bareground + exotic	+ 3.9
Sprague's pipit	Site 1	Year + site type	+ 1.6
	Site 2	Year + activity	+ 3.5
	Local	Year + structure distance + road distance	+ 1.6
	Vegetation	Year + stem height + stem density + litter depth + bareground + exotic	+ 0.8
Western meadowlark	Site 1	Site type	+ 2.0
	Site 2	Activity	+ 3.7
	Local	Structure distance + road distance	+ 3.4
	Vegetation	Stem height + stem density + litter depth + bareground + exotic	+ 6.5

Appendix IX

The effects of year, structure power source, and structure activity status on songbird nesting success in southeastern Alberta, 2012-2014.

Species ¹		Year ²		Power source ³	Activity ⁴
		2012	2013	Grid powered	Active
All species (<i>n</i> = 1222)	β	N/A	N/A	-0.4624	0.2516
	SE	N/A	N/A	0.1928	0.1992
	LCL	N/A	N/A	-0.7825	-0.0789
	UCL	N/A	N/A	-0.1470	0.5780
	<i>p</i> -value	N/A	N/A	0.0165	0.2066
Savannah sparrow (<i>n</i> = 643)	β	0.7575	0.3879	-0.8474	0.4015
	SE	0.6137	0.2513	0.2768	0.2724
	LCL	-0.1329	-0.0182	-1.3137	-0.0540
	UCL	1.9226	0.8111	-0.4015	0.8438
	<i>p</i> -value	0.2171	0.1228	0.0022	0.1406
Vesper sparrow (<i>n</i> = 261)	β	N/A	N/A	-0.6329	0.3839
	SE	N/A	N/A	0.7738	0.7785
	LCL	N/A	N/A	-2.1445	-0.7579
	UCL	N/A	N/A	0.4980	1.9003
	<i>p</i> -value	N/A	N/A	0.4130	0.6220

1 Sample sizes are given in nesting intervals for logistic exposure

2 2014 is the reference level

3 Generator powered is the reference level

4 Inactive is the reference level

Appendix X

The effects of year, distance to nearest structure and distance to nearest road on songbird nesting success in southeastern Alberta, 2012-2014.

Species ¹		Year ²		Distances	
		2012	2013	Nearest structure	Nearest road
All species (<i>n</i> = 2971)	β	0.0016	0.3744	0.0003	0.0000
	SE	0.1998	0.1257	0.0003	0.0005
	LCL	-0.3161	0.1686	-0.0002	-0.0008
	UCL	0.3406	0.5824	0.0009	0.0007
	<i>p</i> -value	0.9936	0.0029	0.3319	0.9518
Savannah sparrow (<i>n</i> = 1768)	β	0.6472	0.5082	0.0003	0.0000
	SE	0.3538	0.1658	0.0004	0.0006
	LCL	0.1082	0.2388	-0.0004	0.0011
	UCL	1.2761	0.7847	-0.0009	0.0011
	<i>p</i> -value	0.0674	0.0022	0.4997	0.9426
Vesper sparrow (<i>n</i> = 524)	β	N/A	N/A	0.0035	-0.0006
	SE	N/A	N/A	0.0016	0.0019
	LCL	N/A	N/A	0.0011	-0.0037
	UCL	N/A	N/A	0.0064	0.0026
	<i>p</i> -value	N/A	N/A	0.0289	0.7317
Chestnut-collared longspur (<i>n</i> = 260)	β	N/A	N/A	0.0000	-0.0012
	SE	N/A	N/A	0.0009	0.0012
	LCL	N/A	N/A	-0.0014	-0.0031
	UCL	N/A	N/A	0.0016	0.0010
	<i>p</i> -value	N/A	N/A	0.9810	0.3350
Western meadowlark (<i>n</i> = 135)	β	N/A	N/A	-0.0009	0.0001
	SE	N/A	N/A	0.0013	0.0024
	LCL	N/A	N/A	-0.0028	-0.0034
	UCL	N/A	N/A	0.0014	0.0045
	<i>p</i> -value	N/A	N/A	0.4740	0.9780

	β	N/A	1.1051	0.0012	0.0041
	SE	N/A	0.7576	0.0016	0.0037
Sprague's pipit (<i>n</i> = 127)	LCL	N/A	-0.1897	-0.0012	-0.0008
	UCL	N/A	2.3785	0.0044	0.0118
	<i>p</i> -value	N/A	0.1450	0.4470	0.2650

1 Sample sizes are given in nesting intervals for logistic exposure
2 2014 is the reference level

Appendix XI

The effects of year and vegetation characteristics on songbird nesting success in southeastern Alberta, 2013-2014.

Species ¹		Year ²	Stem height (m)	Stem density	Litter depth (mm)	%Bareground	%Shrub	%Exotic
All species (<i>n</i> = 3080)	β	0.2931	1.9620	-0.0273	-2.0742	-0.0015	-0.0151	-0.0043
	SE	0.1246	1.0974	0.0252	4.3779	0.0093	0.0111	0.0033
	LCL	0.0893	0.1886	-0.0678	-9.1774	-0.0155	-0.0316	-0.0095
	UCL	0.4994	3.7974	0.0150	5.3082	0.0156	0.0046	0.0015
	<i>p</i> -value	0.0186	0.0738	0.2781	0.6356	0.8741	0.1732	0.1974
Savannah sparrow (<i>n</i> = 1895)	β	0.4013	1.8213	-0.0069	-4.6635	-0.0335	-0.0152	-0.0095
	SE	0.1659	1.5636	0.0317	5.5678	0.0254	0.0785	0.0038
	LCL	0.1316	-0.6975	-0.0578	-13.7178	-0.0709	-0.1314	-0.0155
	UCL	0.6779	4.4466	0.0467	4.8216	0.0123	0.1276	-0.0028
	<i>p</i> -value	0.0155	0.2441	0.8274	0.4023	0.1868	0.8468	0.0135
Vesper sparrow (<i>n</i> = 496)	β	N/A	1.4696	0.0377	6.7781	-0.0162	-0.0596	-0.0011
	SE	N/A	2.9619	0.0983	14.0202	0.0145	0.0306	0.0095
	LCL	N/A	-3.3211	-0.1165	-15.4091	-0.0390	-0.1062	-0.0148
	UCL	N/A	6.4259	0.2054	30.7847	0.0104	-0.0064	0.0171
	<i>p</i> -value	N/A	0.6198	0.7017	0.6288	0.2630	0.0512	0.9085
Chestnut-collared longspur (<i>n</i> = 250)	β	N/A	1.1280	-0.1824	-13.4797	0.0205	N/A	-0.0961
	SE	N/A	4.7306	0.0946	23.0303	0.0361	N/A	0.3104
	LCL	N/A	-6.8454	-0.3375	-50.5845	-0.0206	N/A	-0.5829
	UCL	N/A	9.2364	-0.0176	25.9765	0.1170	N/A	0.4683
	<i>p</i> -value	N/A	0.8115	0.0538	0.5583	0.5695	N/A	0.7568

Sprague's pipit (<i>n</i> = 160)	β	N/A	0.0372	-0.1587	-62.4042	0.0432	N/A	-0.0424
	SE	N/A	0.0157	0.1646	42.7681	0.0905	N/A	0.0481
	LCL	N/A	0.0135	-0.4406	-140.3846	-0.0820	N/A	-0.1093
	UCL	N/A	0.0671	0.1228	3.2756	0.2245	N/A	0.0631
	<i>p</i> -value	N/A	0.0180	0.3350	0.1450	0.6340	N/A	0.3780
Western meadowlark (<i>n</i> = 132)	β	N/A	13.6567	-0.1653	-6.9058	0.0088	N/A	-0.0140
	SE	N/A	7.6220	0.1879	22.4533	0.0622	N/A	0.0156
	LCL	N/A	1.3656	-0.4747	-42.5859	-0.0782	N/A	-0.0394
	UCL	N/A	26.7619	0.1443	31.0972	0.1370	N/A	0.0152
	<i>p</i> -value	N/A	0.0732	0.3788	0.7584	0.8878	N/A	0.3688

1 Sample sizes are given in nesting intervals for logistic exposure
2 2014 is the reference level

Appendix XII

The effects of year, site type, distance bin, and interaction on songbird nest density in southeastern Alberta, 2012-2014.

Species ¹		Year ²		Site type ³		Interaction ³
		2012	2013	Infrastructure	Distance bin	Infrastructure * distance bin
All species (<i>n</i> = 744)	β	-1.4210	-0.2198	0.3395	0.1481	-0.1102
	SE	0.2011	0.1260	0.2306	0.0684	0.0811
	LCL	-1.6318	-0.3657	-0.0397	0.0352	-0.2453
	UCL	-1.2170	-0.0743	0.7256	0.2620	0.0243
	<i>p</i> -value	0.0000	0.0130	0.1410	0.0302	0.1740
Savannah sparrow (<i>n</i> = 465)	β	-1.8899	-0.4172	0.1908	0.2044	-0.0819
	SE	0.1779	0.1083	0.2902	0.0835	0.0999
	LCL	-2.1922	-0.5962	-0.2835	0.0677	-0.2479
	UCL	-1.6061	-0.2394	0.6760	0.3430	0.0830
	<i>p</i> -value	0.0000	0.0001	0.5109	0.0143	0.4122

	β	-1.3274	-0.3164	2.1071	0.2023	-0.4601
	SE	0.6803	0.2907	0.7108	0.2275	0.2457
Vesper sparrow ($n = 114$)	LCL	-1.8303	-0.6531	1.0135	0.1676	-0.8744
	UCL	-1.4495	0.0155	3.3717	0.5881	-0.0600
	p -value	0.0000	0.1187	0.0030	0.3738	0.0612

1 Sample sizes are given in nests

2 2014 is the reference level

3 Control is the reference level

APPENDIX XIII

Goodness of fit quantified by ΔAIC for all models analyzing songbird nest density in southeastern Alberta, 2012-2014.

Species	Model	Parameters	ΔAIC
All species	Site distance	Year + site type + distance bin + interaction	+ 0.3
Savannah sparrow	Site distance	Year + site type + distance bin + interaction	4.9
Vesper sparrow	Site distance	Year + site type + distance bin + interaction	19.0
