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A recent goal set by the Department of Energy to promote expansion of wind-generated electricity in the United States has resulted in a rapidly growing wind energy industry. Current research assessing the impacts of wind energy development to wildlife has largely focused on those impacts to bats, raptors, and other bird species despite the fact that suitable sites for wind energy production in the western United States often coincide with crucial habitat for ungulates including winter range used by pronghorn (Antilocapra americana). Pronghorn on winter range already experience harsh environmental conditions where energetic costs are high. Exposure to further disturbance, such as wind energy development, could result in modifications to pronghorn behavior leading to net energy losses that may impact survival and overall health of these populations. Avoidance of oil and gas development by several ungulate species has been documented; however, how these same species will respond to wind energy development is largely unknown. In order to understand the potential impacts of wind energy development to pronghorn on winter range, I evaluated the response of a wintering pronghorn population to the Dunlap Ranch wind energy facility over 3 winters (2010, 2010–2011, 2011–2012) in southcentral Wyoming, USA. My objectives were to 1) evaluate displacement of pronghorn in relation to wind energy infrastructure components to identify changes in movement rates influenced by vicinity to wind energy development, 2) develop a resource selection function (RSF) for pronghorn exposed to wind energy at the population level to gain insights into winter resource selection on this landscape, 3) apply the overall best fit population level model to individuals with home ranges that overlapped wind energy to isolate potential avoidance behavior relative to each individual within its own home range, and 4) to identify environmental

and anthropogenic predictor variables (inclusive of wind energy) influencing pronghorn mortality risk on winter range. In addition, I modeled resource selection and estimated survival for a neighboring population of pronghorn near Walcott Junction, Wyoming (see appendix A).

Dunlap Ranch was developed for wind energy production in 2010 and was located approximately 11.8 km north of Medicine Bow, in Carbon County, Wyoming. I obtained location data from 47 female pronghorn equipped with GPS-transmitters at Dunlap Ranch. I modeled frequency of use as a continuous response variable to predict pronghorn resource selection across the Dunlap Ranch both at the population and individual levels. I estimated survival for pronghorn using the Kaplan-Meier product limit estimator. Finally, I modeled mortality risk for pronghorn at Dunlap Ranch using the Cox proportional hazards model inclusive of cumulative, weekly, and monthly temporal scales.

At the population level, pronghorn at the Dunlap Ranch selected for areas closer to wind energy facilities and with lower slopes, standard deviation in snow depth, and density of fences. At the individual level, coefficients for distance to nearest wind energy facility did not differ from zero (P > 0.05) across all 3 winters, indicating no effect of wind energy development on pronghorn resource selection on winter range in my study area. In addition, pronghorn daily net displacement did not increase closer to wind energy development ($r^2 = 0.001-0.012$) during each winter. Twenty-four pronghorn from the Dunlap Ranch study area died with the majority of deaths (n = 13 or 54.2% of deaths) occurring in winter 2010–2011. Survival (\hat{S}) at the end of my 2.5-year study period was 0.30 (90% CI: 0.17–0.43) at Dunlap Ranch. Survival (\hat{S}) during winter 2010 was 0.97 (90% CI: 0.92–1.00), winter 2010–2011 was 0.53 (90% CI: 0.37–0.70), and winter 2011–2012 was 0.91 (90% CI: 0.82–1.00). Pronghorn winter mortality risk for Dunlap Ranch individuals decreased with increasing average distance from major roads (hazard ratio = 0.80, 90% CI: 0.66–0.98), average time spent in sagebrush (*Artemisia tridentata* spp.; hazard ratio = 0.96, 90% CI: 0.95–0.98), and terrain ruggedness (hazard ratio = 0.08, 90% CI: 0.01–0.67).

Overall, I did not detect an influence of wind energy development on pronghorn movement behavior, resource selection, or mortality risk at the Dunlap Ranch. Most avoidance behaviors documented in ungulate populations are associated with human presence and increased traffic. Lower traffic rates observed within the Dunlap Ranch paired with less overall length of access roads and less habitat developed than in oil and gas fields may explain why avoidance and increased movement rates were not observed in pronghorn. My results are valuable in providing guidance for wildlife managers considering future wind energy development on pronghorn winter range. For example, identifying sagebrush as influential for pronghorn winter mortality risk should lead to greater conservation of sagebrush stands in areas where development and pronghorn winter range coincide. My results also illustrate that fence densities and variability in snow depth are important contributors to habitat selection by wintering pronghorn in southcentral Wyoming and must be considered when implementing further modifications (energy and other human developments) to these harsh environmental landscapes to minimize impacts to pronghorn. Regardless, caution must be taken when generalizing these results across pronghorn populations. Although pronghorn were not impacted negatively by wind energy on the Dunlap Ranch, my results cannot be directly applied to populations exposed to wind energy development at larger scales and on other seasonal ranges where traffic levels and environmental conditions may differ.

PRONGHORN (Antilocapra americana) RESPONSE TO WIND ENERGY DEVELOPMENT ON WINTER RANGE IN SOUTH-CENTRAL, WYOMING

By

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CHAPTER 1. Introduction

WIND ENERGY

In 2007, the Energy Information Administration (EIA) predicted the need for electricity within the United States will increase by 39% from 2005 to 2030, reaching a high of 5.8 billion megawatt/hours (EIA 2010). A new energy program was initiated by the United States Department of Energy (DOE) in 2008 to meet these demands. The objective of this program was to develop renewable resources domestically, in a cost effective manner and to do so while reducing carbon emissions (DOE 2008). Wind energy possesses these qualities and, has therefore, taken precedence over other sources of renewable energy. The Department of Energy's goal for wind energy is to contribute 20% of the total United States electricity supply by 2030. However, currently, wind energy is projected to contribute 4.6% of total U.S. electricity generation by 2015 (EIA 2014).

Renewable portfolio standards have also been set by several western states requiring that renewable energy resources contribute a given percentage to overall utilities in each state. Inexpensive production of wind resources throughout the United States could assist several of these states to reach their renewable portfolio standard goals. Wind resources in Wyoming alone could provide up to 116-times the current state's electricity needs. In addition, Wyoming is ranked eighth for highest wind resources in the United States (AWEA 2013). As of June 2013, Wyoming had 24 wind energy projects installed with 5,742 MW of wind energy awaiting production (AWEA 2013). Transmission line development could distribute additional energy produced by Wyoming's wind resources to surrounding regions (Lantz and Tegen 2011). Exporting Wyoming's wind energy will assist the United States in meeting renewable energy goals while increasing electricity demands.

ENERGY DEVELOPMENT AND WILDLIFE

Anthropogenic disturbances such as wind energy development may directly and indirectly impact wildlife. Increasing urbanization, infrastructure, human presence, and seismic activity may lead to extensive habitat loss and fragmentation for wildlife populations and in turn affect the overall ecological structure of the environments they inhabit (Mortberg 2001, O'Neill and Boutin 2002, Doherty et al. 2008; Sawyer et al 2006, 2009). Quantifying these effects to wildlife can be complex and are usually analyzed by evaluating either direct or indirect effects. Direct effects to wildlife by energy development are measured unambiguously by quantifying habitat loss or direct mortalities caused by standing infrastructure. For example, greater sage-grouse (*Centrocercus urophasianus*) population persistence may be negatively impacted by the direct loss of sagebrush cover vital for nesting, brood rearing, and winter habitat (Aldridge et al. 2008, Doherty et al. 2010, Smith et al. 2014). In addition, direct effects may be quantified by measuring direct mortalities caused by collisions with novel infrastructure. The blades of wind turbines, associated with wind energy production, have been reported to cause direct mortalities to raptors, passerine birds, and bats (Erickson et al. 2001, Piorkowski and O'Connell 2010).

Indirect effects to wildlife are usually quantified by identifying behavioral changes within an affected population. Measuring changes in time spent foraging, resting, or being vigilant can offer insights into how wildlife may respond to anthropogenic disturbance. Time allocated to feeding by woodland caribou (*Rangifer tarandus caribou*) increased in response to petroleum exploration, suggesting an increase in energy expenditure by these animals (Bradshaw et al. 1997). In addition, habitat selection patterns of wildlife populations are commonly used as a surrogate to measure avoidance in anthropogenically-disturbed landscapes. Evaluating habitat selection patterns in boreal forest environments developed for oil and gas production showed that

woodland caribou exhibited avoidance behavior within distances of 1,000 m of wells and 250 m of roads (Dyer et al. 2001). In Oklahoma, lesser prairie-chickens (Tympanuchus pallidicinctus) avoided areas beneath transmission lines within 100 m, suggesting indirect habitat loss (Pruett et al. 2009). Sawyer et al. (2006) found that mule deer (Odocoileus hemionus) altered their habitat use during and after development of gas fields in western Wyoming. High use habitats were avoided by mule deer during and after construction of well pads and other infrastructure related to development. Mule deer also avoided well pads where human presence was elevated (Sawyer et al. 2009). Shifting habitat selection from high to low use areas indicates that mule deer may be relegated to using lower quality habitats, potentially prompting long-term impacts to these populations. Avoidance of human used, structurally altered, and disturbed areas related to oil and gas development has caused substantial habitat loss to wild ungulate populations. Solely measuring direct habitat loss to wildlife may result in an underestimation of total habitat loss. For example, in gas fields, approximately 0.012–0.016 km² of habitat is disturbed by a single conventional natural gas well pad; however, Sawyer et al. (2006) found that areas used the most by mule deer in western Wyoming were 2.7-3.7 km away from well pads during development. In addition, oil and gas roads are typically permitted to be no more than 12-24 m wide on public lands in the United States (BLM 2007); however, woodland caribou were avoiding roads up to 250 m in a petroleum-developed landscape in northern Alberta, Canada (Dyer et al. 2001). Although direct habitat loss of both well pads and roads may be minimal, overall habitat loss to wildlife through avoidance of these disturbed areas may be much greater.

The demand for renewable energy is relatively new and few studies have been published documenting the indirect effects of wind energy development on wildlife. LeBeau et al. (2014) found that mortality risk increased for greater sage-grouse nests and broods within 5 km of wind

turbines. In addition, a study in Oklahoma evaluating the response of elk (*Cervus elaphus*) to installation of 45 wind turbines and associated infrastructure found that diet quality and displacement of elk did not change or occur across pre-development, construction, and production periods encompassing a 3-year period (Walter et al. 2006). These findings suggested elk did not alter their behavior or forage selection in response to wind energy development and production; thus, Walter et al. (2006) found no evidence supporting negative impacts stemming from wind energy development to the elk population in that area.

PRONGHORN AND WINTER HABITAT

Pronghorn (*Antilocapra americana*) are endemic to North American sagebrush (*Artemisia* spp.)steppe and grasslands, where they comprise the only currently extant species of the family Antilocapridae. In 1997, Wyoming was home to 55% of the North American and 57% of the United States populations of pronghorn (Yoakum and O'Gara 2000). In Wyoming, pronghorn obtain resources from habitats seasonally influenced by stochastic weather patterns. Seasonal resource availability has long influenced pronghorn habitat selection and seasonal migration patterns in the arid regions they inhabit. The relative success of pronghorn populations to negotiate rugged terrain and anthropogenic influences to access food supplies and milder climatic conditions influences survival and concomitant population productivity.

Pronghorn winter range is impacted by harsh weather conditions and reduced forage quality (Schwartz et al. 1977, Byers 1997). Costs of increased thermoregulation and maneuvering through snow result in net energy expenditure and, thus, reduced body condition (Parker et al. 1984, Miura and Muruyama 1986). Temperate ungulates cope with the increased demand for energy by utilizing fat reserves and increasing their time spent foraging (Miura and Muruyama 1986, Franzmann et al. 1987, Tyler and Blix 1990). Pronghorn forage more during winter months than any other time of the year, spending up to 55% of their time feeding to make up for this increased demand in energy (Byers 1997).

Forage quality and diet composition on winter range differs largely from other seasonal ranges. Forbs, shrubs, grasses, and sedges form pronghorn diets in spring (Mitchell and Smoliak 1971), but diets shift to browse as herbaceous species senesce during summer (Bayless 1969). Availability and nutritional quality of grasses, sedges and forbs are limited during the winter, thus restricting pronghorn diets to mainly woody browse. For example, silver sagebrush (*A. cana*) and fringed sagewort (*A. frigida*) composed 82% of the winter diet of pronghorn in southern Alberta (Mitchell and Smoliak 1971). In central Montana, pronghorn winter diets are composed of 78–90% shrubs with big sagebrush (*A. tridentata*) making up 45% of shrub consumption (Bayless 1969).

Snow levels may also dictate winter forage availability. During winter, pronghorn may paw craters with their forelegs to reach more desirable forage when snow cover is sparse; however, when snow levels are elevated pronghorn may be restricted to eating shrubs and grasses that extend above the surface of the snow (Byers 1997). In addition, pronghorn populations may also suffer a decrease in reproductive success during harsh or extended winters (Mortinka 1967, Barrett 1982). During long winters, pregnant females may adjust to reduced resources by increasing gestation periods; however, birthrates may still be reduced significantly by harsh conditions (Mortinka 1967, Barrett 1982).

Pronghorn and other ungulate seasonal ranges are often accessed by way of migratory corridors. Pronghorn may remain in seasonal ranges year round or migrate various distances depending on geographic location. Pronghorn migrations between seasonal ranges have been documented at 258.0 km in western Wyoming (Sawyer et al. 2005), 70.6 km in North Dakota

(Kohlar et al. 2011), 23.1 km in South Dakota (Jacques et al. 2009), and 32.9–53.6 km in Idaho (Hoskinson and Tester 1980). Variation in migration distances provides evidence for conditional strategies by which individuals migrate when environmental conditions are favorable. When migration does occur, most pronghorn in the Yellowstone region display fidelity to a particular range and migratory corridor (White 2007). Sawyer et al. (2005) identified the 1.6 km-wide Trapper's Point bottleneck as the most used migratory corridor for pronghorn and mule deer populations in western Wyoming. In a species with high fidelity to seasonal ranges, the destruction or disturbance of small migratory corridors could result in pronghorn being unable to access vital winter or summer ranges imperative for annual subsistence and survival of individuals.

Energy development, urbanization and agriculture, have brought about several changes to pronghorn habitats. Development of roads, fences, buildings, and energy infrastructure not only modifies the appearance of the landscape, but also has potential to change the function and stability of local wildlife populations. For example, pronghorn have evolved specialized limbs built for speed and their physiology does not adequately afford them the ability to jump fences (Byers 1997). Inadequacies to jump fences result in direct mortalities as well as avoidance of fenced areas (Sheldon 2006). Pronghorn are already constrained by food deprivation, thermodynamics, and energetic limitations on winter range. Additional anthropogenic disturbance to pronghorn winter range may result in negative impacts to populations during construction or production phases. Evaluating these impacts can lead to improved mitigation measures that lessen pressures for pronghorn on crucial winter range.

STUDY DESIGN AND OBJECTIVES

I examined the response of pronghorn to wind energy development and production near the Dunlap Ranch wind energy facility north of Medicine Bow in the Shirley Basin of south-central Wyoming, USA. Dunlap Ranch was developed by PacifiCorp for wind energy production beginning in 2009 on pronghorn crucial winter range (Figure 1.1). Oil and gas resources were never developed at the Dunlap Ranch, eliminating confounding effects of wind energy with other sources of energy development.

Development occurring on both pronghorn and elk crucial winter range in the Dunlap Ranch prompted PacifiCorp and the WGFD to identify the need to study and monitor big game species in this area (PacifiCorp Energy 2009). Monitoring projects are intended to provide WGFD managers with data to evaluate the degree to which pronghorn may be impacted by wind energy construction and production (PacifiCorp Energy 2009). As such, I monitored pronghorn in the Dunlap Ranch from January 2010 to May 2012. I examined the response of pronghorn to wind energy development over three winters (2010, 2010–2011, and 2011–2012). Specific objectives of my study were to 1) evaluate displacement of pronghorn in relation to wind energy infrastructure components to identify changes in movement rates influenced by vicinity to wind energy development, 2) develop a resource selection function (RSF) for pronghorn exposed to wind energy at the population level to gain insights into winter resource selection on this landscape, 3) apply the overall best fit population level model to individuals with home ranges that overlapped wind energy to isolate potential avoidance behavior relative to each individual within its own home range, and 4) to identify environmental, anthropogenic (exclusive of wind energy), and wind energy predictor variables influencing pronghorn mortality risk.

The influence of oil and gas development on pronghorn habitat selection has only recently been evaluated. Beckmann et al. (2012) found that pronghorn in the Upper Green River Basin of Wyoming were avoiding patches of high level disturbance in an oil and gas field. Further, probability of use for pronghorn in the Upper Green River Basin decreased with decreasing distance to wells and increasing habitat disturbance (Beckmann et al. 2012). Currently, no studies have been conducted to evaluate the influence of wind energy development on pronghorn habitat selection and mortality risk. Evaluating the influences of energy development to pronghorn on winter range is particularly critical given that they encounter elevated energetic demands during this time of year (Sawyer et al. 2002). Understanding these relationships will be beneficial in providing information to assist in making future conservation and management decisions.

STUDY AREAS

Dunlap Ranch Wind Energy Treatment Area

The Dunlap Ranch wind energy treatment area encompasses the 62.2 km² Dunlap Ranch wind energy facility (106°11'0.290"W, 42°01'43.060"N) and pronghorn crucial winter range located in Carbon County in Township 23 North and Ranges 78 and 79 West (Figure 1.1). PacifiCorp owns and operates the Dunlap Ranch wind energy facility located approximately 11.8 km north of Medicine Bow, Wyoming. Within the Dunlap Ranch wind energy facility PacifiCorp owns 56.7 km² of private fee land, leases 6.2 km² of Wyoming state land, and 2.6 km² is owned by a separate private entity. The Dunlap Ranch project site was selected for wind energy development by PacifiCorp based on historic wind data and data acquired from currently installed meteorological towers (PacifiCorp Energy 2009).

Plans for development of the Dunlap Ranch wind energy facility were split into two phases. Phase I (Dunlap I; Figure 1.1) facilitated the construction of wind energy infrastructure on pronghorn crucial winter range from September 2009 to September 2010. Phase II (Dunlap II) will be constructed on elk crucial winter range at a later date (PacifiCorp Energy 2009). Highway 487 extends north and south through the middle of the Dunlap Ranch and pronghorn crucial winter range (Figure 1.1). Installation of the Dunlap I wind-powered generation facility began with civil construction and the installation of 28.3 km of access roads in fall 2009. The PacifiCorp operation and maintenance building offers office facilities for employees as well as storage for vehicles and equipment used on the Dunlap Ranch. The operation and maintenance building is located just east of highway 487 and is constructed in the early phases of Dunlap I. Output from wind turbine generators is delivered to an onsite 34.5/230-kV substation constructed directly east of the operation and maintenance building. Approximately 18.6 km of transmission lines were installed on the Dunlap Ranch from November 2009 to June 2010. The transmission lines parallel highway 487 and extend 14.5 km north of the Dunlap Ranch wind energy facility. Weather and climate data were obtained from 3 meteorological towers reaching heights of 79.9 m. One meteorological tower existed before the initiation of Dunlap I. Construction of the Dunlap I wind-powered generation facility led to the erection of 74 General Electric Company 1.5-MW sle¹ wind turbine generators (119 m tall) from April, 2010 to September 30, 2010 (PacifiCorp Energy 2009).

In addition to Dunlap I, three wind energy facilities in the surrounding Carbon and Albany County areas were potentially used by pronghorn (Figure 1.2). The Seven Mile Hill wind energy facility includes 79, 1.5-MW wind turbines (119 m tall) that became operational in December 2008 and is located approximately 14 km southwest of Dunlap I between Hanna and

Medicine Bow, Wyoming. The High Plains and McFadden Ridge wind energy project encompasses 85, 1.5-MW (119 m tall) wind turbines since 2009 and is located 38 km southeast of Dunlap I in Albany and Carbon Counties near McFadden, Wyoming. Lastly, the Foote Creek Rim wind energy project encompasses 182, 600 kW wind turbines (61 m tall) since 1999 and is located approximately 35 km south of Dunlap I near Arlington, Wyoming (Rocky Mountain Power 2010, BLM 2011b).

The Dunlap Ranch occurs in the southern portion of the broad, intermontane Shirley Basin, dominated by arid shrublands and grasslands. The region is marked by flats, high hills, and low mountains with Wyoming big sagebrush (*A. t. wyomingensis*) being the most prevalent cover type. The Freezeout Mountains to the west, support populations of alderleaf mountain mahogany (*Cercocarpus montanus*), aspen (*Populus tremuloides*), and limber pine (*Pinus flexilus*). Elevations within the Dunlap Ranch range from 2,000 to 2,530 m with annual precipitation of 30.5 cm and annual snowfall of 109 cm. I acquired climate data from the nearest Western Regional Climate Center weather station located 11.8 km south of the study area in Medicine Bow, Wyoming. Average minimum and maximum temperatures ranged from –11.4°C to 12.8°C for the winter months (November–April) and –2.6°C to 28.7°C for the summer months (May–October) from 1981 to 2010 (WRRC 2011). The Little Medicine Bow and Medicine Bow Rivers are located just south of the Dunlap Ranch boundary with Muddy Creek, a tributary of the Medicine Bow River, flowing within the eastern border of the wind energy facility (PacifiCorp Energy 2009).

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Figure 1.1. Wind construction at the Dunlap Ranch wind energy facility, located in Carbon County, Wyoming, USA approximately 11.8 km north of Medicine Bow, is planned to take place in two phases. During phase I (Dunlap I), a contractor for PacifiCorp constructed 74 wind turbines, 28.3 km of access roads, 2 meteorological towers, 1 substation, 18.6 km of transmission lines, and on site storage facilities on pronghorn crucial winter range from September 2009 to September 2010.



Figure 1.2. In addition to Dunlap I, three other wind energy facilities were in production phases in nearby Carbon and Albany Counties, Wyoming prior to initiation of my study. The Seven Mile Hill wind energy facility maintained production of 79 (1.5 MW) wind turbines, the High Plains and McFadden Ridge wind energy project maintained production of 85 (1.5 MW) wind turbines, and the Foote Creek Rim wind energy project maintained production of 183 (600 kW) wind turbines throughout the study period from Jan 2010 to May 2012.

CHAPTER 2. Habitat Selection and Displacement of Pronghorn Exposed to Wind Energy Development on Winter Range in South-Central Wyoming

In the format for manuscripts submitted to Ecosphere

ABSTRACT

Few studies have evaluated the impacts of wind energy development on ungulate populations. The recent demand for wind-generated electricity coupled with a tendency for suitable sites for wind-energy facilities in some areas to occur within pronghorn (Antilocapra americana) winter range, make this a critical issue, especially given that pronghorn experience harsh environmental conditions during winter. I evaluated pronghorn response to wind energy development over 3 winters (2010, 2010–2011, 2011–2012) in south-central Wyoming, USA. I obtained data from 47 adult female pronghorn equipped with GPS-transmitters and developed resource selection functions for pronghorn at both the population and individual level. At the population level, pronghorn selected for areas closer to wind energy facilities and with lower slopes, standard deviation in snow depth, and density of fences. At the individual level, coefficients for distance to nearest wind energy facility did not differ from zero (P > 0.05) across all 3 winters. In addition, pronghorn daily net displacement did not increase closer to wind energy development $(r^{2} = 0.001 - 0.012)$ during each winter. For individual pronghorn exposed to wind energy, I did not identify avoidance and increases in movement behavior, which have previously been documented for pronghorn and other ungulate species exposed to oil and gas development. Most avoidance behaviors documented in ungulate populations are associated with human presence and increased traffic. Thus, lower traffic rates observed within my study area paired with less overall length of access roads and less habitat developed than oil and gas fields may explain why avoidance and increases in movement rates were not observed in these pronghorn. Nevertheless,
caution must be taken when considering impacts to pronghorn populations exposed to wind energy development in different environments and scales than those explored in my analysis.

INTRODUCTION

Current research assessing the impacts of energy development to ungulates has largely focused on impacts of oil and natural gas to these iconic species. Responses to oil and natural gas development by woodland caribou (Rangifer tarandus caribou), elk (Cervus elaphus), mule deer (Odocoileus hemionus), and pronghorn (Antilocapra americana) include avoiding developed areas and shifting movement patterns (Bradshaw et al. 1997, Dyer et al. 2001, Sawyer et al. 2006 and 2009, Webb et al. 2011, Beckmann et al. 2012). These avoidance behaviors are commonly associated with human presence and intensity of energy development. For example, woodland caribou and mule deer have demonstrated greater avoidance of oil and gas developments in areas with elevated traffic (Dyer et al. 2001, Sawyer et al. 2009). Similarly, mule deer in western Wyoming increased movement rates through areas of intense oil and gas development (Sawyer et al. 2013). In addition, elk in Colorado and New Mexico increased movement distances and complexity of movement patterns closer to development areas in an oil and gas field (Webb et al. 2011). With malnutrition being a primary cause of winter mortality and reduced pregnancy rates for several of these ungulate species (Barrett 1982, Pyrah 1987, Boertje and Gardner 1998, Bishop et al. 2005), exposure to higher levels of human activity associated with oil and gas fields could further exacerbate body condition deterioration and overall population viability on winter range. In order to understand overall impacts of energy development to wintering ungulates predisposed to harsh environments, it is imperative that we consider how different types of energy development will affect these populations given that varying levels of human activity

associated with them play a key role in elicitation of avoidance and changes in movement behavior by ungulates.

Whether impacts to ungulates brought on by oil and gas development are comparable to those impacts brought on by wind energy development remains largely unknown. A recent goal set by the Department of Energy to promote expansion of wind energy in the United States has resulted in a rapidly growing industry with wind energy projected to contribute 4.6% of total U.S. electricity generation by 2015 (EIA 2014). Wyoming alone has the wind energy potential to provide up to 116-times the state's current electricity needs (AWEA 2010). Differences in physical infrastructure between wind energy (wind turbines, transmission lines, meteorological towers, sub-stations, etc.) and oil and gas development (well pads, compressor stations, pumps, pipelines, etc.) are apparent; however, the potential impacts to wildlife behavior caused by these infrastructure components, along with the varying levels of human activity associated with them, are not clearly understood. Oil and gas and wind energy developments have similar densities of infrastructure and habitat loss per unit area (Jones and Pejchar 2013). Both developments are responsible for increased truck traffic and human activity during construction and production phases; however, direct comparisons of traffic levels and human activity between oil and gas and wind energy developments have not been made. Whereas most avoidance behaviors of oil and gas development elicited by ungulates are associated with high levels of traffic and human activity (Dyer et al. 2001, Sawyer et al. 2009, Beckmann et al. 2012), responses of ungulate populations to wind energy development are nearly impossible to predict. Pronghorn exposed to oil and gas development on winter range in western Wyoming abandoned highly disturbed areas and experienced significant losses of high quality habitats (Beckmann 2012). If pronghorn exposed to other forms of energy development such as wind shift from using high quality

habitats as has been documented with oil and gas development, negative impacts to overall population health may result.

Pronghorn winter range is characterized by harsh environmental conditions where pronghorn must balance energy intake with overall energy costs (Schwartz et al. 1977, Parker et al. 1984, Byers 1997). Pronghorn have higher energy demands relative to body mass than most domestic ruminants, resulting in higher stress during periods of food deprivation (Wesley et al. 1970). Pronghorn may respond by increasing time spent foraging during winter months (Byers 1997); however, high mortality rates are common among wintering pronghorn populations (Martinka 1967, Barrett 1982, Pyrah 1987, O'Gara 2004). Winter range for pronghorn may coincide with suitable sites for wind energy development. For pronghorn existing under already stressful conditions on winter range where mortality rates are generally elevated, further development of wind energy associated infrastructure may result in changes in pronghorn behavior that are energetically costly. While impacts of oil and gas development to pronghorn have been explored, no studies have evaluated the impacts of wind energy development on this species.

Along with the recent expansion of wind energy development comes uncertainty related to the response of pronghorn populations in the face of this rapidly growing industry (EIA 2014, AWEA 2010). Development of the Dunlap Ranch wind energy facility on pronghorn crucial winter range in south-central Wyoming, USA created a unique opportunity for me to evaluate the influence of wind energy development on wintering pronghorn. My objectives were to 1) evaluate displacement of pronghorn in relation to wind energy infrastructure components to identify changes in movement rates influenced by vicinity to wind energy development, 2) develop a resource selection function (RSF) for pronghorn exposed to wind energy at the

population level to gain insights into winter resource selection on this landscape, 3) apply the overall best fit population level model to individuals with home ranges that overlapped wind energy to isolate potential avoidance behavior relative to each individual within its own home range, and 4) monitor traffic and calculate daily traffic loads in the Dunlap Ranch to gain insights into the amount of traffic existing within a wind energy production facility. These objectives assisted me in better understanding habitat-specific responses of wintering pronghorn exposed to wind energy development.

METHODS

Study area

I evaluated resource selection and displacement from wind energy development for pronghorn associated with the Dunlap Ranch, 11.8 km north of Medicine Bow in Carbon County, Wyoming, USA. My study focused on a 36.5-km² wind energy facility completed on the Dunlap Ranch in spring 2010 sited within a larger area that includes 1452.3 km² of rangeland (22.6% Bureau of Land Management, 7.6% state of Wyoming, and 69.7% private ownership) delineated by the Wyoming Game and Fish Department as crucial winter range for pronghorn. PacifiCorp owns and operates the Dunlap Ranch wind-powered generation facility. Plans for development of the Dunlap Ranch wind energy facility were split in two phases. Phase I (Dunlap I) facilitated the construction of wind energy infrastructure on pronghorn crucial winter range from September 2009 to September 2010. Construction of the Dunlap I wind-powered generation facility included installation of 74 General Electric Company 1.5-MW sle¹ wind turbine generators (119 m tall), 28.3 km of access roads, 3 meteorological towers (79.9 m tall), an onsite 34.5/230-kV substation, and onsite maintenance buildings. Construction began in fall 2009 and concluded with the erection of wind turbines from April 2010 to September 2010 (PacifiCorp Energy 2009). Highway 487 extends north and south through the center of the Dunlap Ranch and pronghorn crucial winter range. We began studying pronghorn in winter 2010 before wind turbines were erected at Dunlap I. In addition to Dunlap I, pronghorn winter range also includes the Seven Mile Hill I and II wind energy projects inclusive of 66 and 13, 1.5-MW wind turbines (119 m tall), respectively, and is located approximately 14 km southwest of Dunlap I between Hanna and Medicine Bow, in Carbon County, Wyoming.

The Dunlap Ranch is located in the southern portion of the broad, intermontane Shirley Basin, dominated by arid shrublands and grasslands. The Little Medicine Bow and Medicine Bow Rivers run just south of the Dunlap Ranch boundary with Muddy Creek, a tributary of the Medicine Bow River, flowing within the eastern border of the wind energy facility (PacifiCorp Energy 2009). The region is marked by flats, high hills, and low mountains with Wyoming big sagebrush (Artemisia tridentata wyomingensis) being the most prevalent cover type. The Freezeout Mountains to the west, support alderleaf mountain mahogany (Cercocarpus montanus), aspen (Populus tremuloides), and limber pine (Pinus flexilus). Elevations within the Dunlap Ranch range from 2,000 to 2,530 m. I acquired climate data from the nearest High Plains Regional Climate Center weather station located 11.8 km south of the study area in Medicine Bow. Total snow fall for winter 2010 (November 2009–April 2010) was 136.6 cm, for winter 2010–2011 (November 2010–April 2011) was 212.6 cm, and for winter 2011–2012 (November 2011–April 2012) was 90.7 cm. Average minimum and maximum temperatures for winter 2010 (November 2009–April 2010) were -10.2°C and 3.2°C, for winter 2010–2011 (November 2010–April 2011) were –8.8°C and 2.9°C, and for winter 2011–2012 (November 2011–April 2012) were –9.7°C and 5.4°C (HPRCC 2012).

Capture and monitoring

We contracted with Leading Edge Aviation, LLC (Lewiston, Idaho 83501) to capture 35 and 17 adult female pronghorn in early January 2010 and December 2011. Protocols were approved by the University of Wyoming Institutional Animal Care and Use Committee (protocol 01012010) and Wyoming Game and Fish Department (Chapter 33 Permit ID 742) to capture, handle, and mark female pronghorn. Each captured animal was examined to estimate age and fitted with an Advanced Telemetry System (Isanti, Minnesota 55040) store-on-board GPS neck collar (model G2110B) in or within the general vicinity of each study area. Collars were programmed to record locations every 7 hours from 16 November to 15 May and every 11 hours from 16 May to 15 November. Locations were collected from January 2010 through May 2012. Aerial relocation flights were conducted 4 times annually to record the status of study animals. I recovered collars transmitting a mortality signal to download locational data collected up to time of death. I assessed the cause of mortality for each dead pronghorn; however, the delayed recovery of collars made this assessment difficult in most cases. The remaining collars detached in late April 2012 and were recovered by late May 2012.

Displacement

I calculated daily net displacement for each individual pronghorn on the Dunlap Ranch as a function of step length and turning angle using Program R (Vanak et al. 2010). For each set of daily locations for an individual, I used the minimum distance from wind turbines to analyze displacement for each given day. Distance of pronghorn from all wind infrastructure components (i.e., wind turbines, wind turbine access roads, wind facility buildings) were highly correlated, thus, I used distance to wind turbines to represent vicinity to wind energy development. I plotted displacement values of pronghorn for each winter as a function of

minimum distance to nearest wind turbine. Although wind turbines did not exist in the Dunlap Ranch wind energy facility during winter 2010, construction activities were taking place around wind turbine pads. In addition, wind turbines were in production phases within the nearby Seven Mile Hill wind energy facility during winter 2010.

Population level resource selection

Thomas and Taylor (2006) identified two key issues when approaching resource selection modeling: 1) pooling data across animals, and 2) temporal or spatial autocorrelation. To address issues of pooling data across individuals, recent studies have utilized the individual animal as a random effect in a mixed modeling approach to account for individual variation (Gilles et al. 2006, Hebblewhite and Merrill 2008, Beckmann et al. 2012); however, designs using logistic regression in combination with mixed modeling do not address issues of autocorrelation of finescale GPS locational data. A recent modeling approach has addressed both issues identified by Thomas and Taylor (2006) by modeling intensity of use rather than a binomial response (i.e., used or unused; Nielson and Sawyer 2013). In this approach, each radio-marked animal is treated as an experimental unit and the intensity of use (number of locations) is quantified within randomly generated circular sampling units placed across study area of interest (Sawyer et al. 2006, Sawyer et al. 2009, Nielson and Sawyer 2013). The frequency of locations within each sampling unit is modeled as the dependent variable using a negative binomial regression. To account for individual variation, individuals are then bootstrapped to estimate standard errors for coefficients in the top model. To address issues of autocorrelation while accounting for variation across individuals, I used methods described by Nielson and Sawyer (2013) to create a resource selection model for pronghorn at the population level (Manly et al. 1993). I modeled relative frequency of use as a continuous response variable using a generalized linear model and negative

binomial distribution (glm.nb in package Mass, Program R; Venables and Ripley 2002). I estimated resource selection using the following equation:

$$\ln(E[li/total_{pop}]) = \beta_0 + \beta_1 X_1 + \dots + \beta_{last} X_{last}$$

where li is the number of locations within a given sampling unit, $total_{pop}$ was the total count of pronghorn locations for each population, β_0 was the intercept, and β_1 was the coefficient for predictor variable X₁ (Nielson and Sawyer 2013). I used an offset term ($E[li/total_{pop}]$) to convert location counts within each sampling unit into relative frequency of use. Using the offset term allowed me to predict pronghorn use on a probability scale rather than a frequency scale (Nielson and Sawyer 2013). Statistical modeling was carried out using ArcGIS 10.0 and the packages boot, car, MASS, and RMS in Program R (Davison and Hinkley 1997, Venables and Ripley 2002, Fox and Weisberg 2011, Harrell 2013).

Predictor variables.—I evaluated predictor variables within 3 variable classes 1) environmental 2) anthropogenic and 3) wind energy (Table 2.1). I acquired spatial layers for landcover data from the USGS GAP Landcover Data Set (U.S. Geological Survey GAP 2011). Because sagebrush is the primary component of winter diets for many pronghorn populations in northern latitudes (e.g., Bayless 1969, Beale and Smith 1970, Mitchell and Smoliak 1971), I focused on sagebrush as the primary landcover category. I categorized each 30 x 30 m cell as 1) sagebrush or 2) other (including agriculture, marshland, mixed grassland, riparian, salt desert shrub, and rocky soils). I then calculated percent sagebrush within each sampling unit. A recent study also considered agricultural lands and riparian areas as important landcover types for predicting pronghorn resource selection in southwest Wyoming (Beckman et al. 2012); however, riparian and agricultural lands composed <0.02% of the land coverage across the Dunlap Ranch study area, so they were not considered separately in my analyses. I acquired snow depth for the Dunlap Ranch study area from the Snow Data Assimilation System (SNODAS; National Operational Hydrologic Remote Sensing Center. 2004). SNODAS is an assimilation model that integrates satellite, airborne platforms, and ground stations to provide consistent estimates of snow cover; thus, snow depths derived from SNODAS models do not represent direct snow accumulation estimates. I calculated the standard deviation of winter snow depth values within each 30 x 30 m cell. Snow depth is positively correlated with the standard deviation of snow depth.

I developed spatial layers in ArcGIS 10.0 for percent slope, aspect, and vector ruggedness measure (VRM; Sappington 2007) using a 1–arc–second (30 m) National Elevation Dataset (NED; U.S. Geological Survey). VRM accounts for variation in both slope and aspect to measure heterogeneity in landscape terrain (Sappington et al. 2007); I referred to this landscape covariate as terrain ruggedness. I categorized aspect as 0 = flat, 1 = north, and 2 = south facing. Aspect was considered flat (0) if slope was $\leq 2^{\circ}$ and was used as a baseline category for dummy coding. I developed spatial layers for distance to nearest major road (Interstates, US Highways and State Highways) and minor roads (county roads). I digitized fences using 2009 National Agriculture Imagery Program (NAIP) imagery and used digitized fences to develop a spatial layer for distance to nearest fence and density of fences. To represent the impact of wind energy development on pronghorn, I used spatial layers for distance to nearest wind turbine, access road, and wind energy facility building (i.e., substations and employee and maintenance buildings; Table 2.1).

Defining availability.—McClean et al. (1998) suggested that the study area boundary, or the distribution of radio-collared animals, was best suited for defining availability in resource

selection studies regardless of statistical methods. Thus, I defined habitat availability for Dunlap Ranch pronghorn by combining individual home ranges for each winter to characterize availability at the population level. I used Program R to calculate 90% Kernel Utilization Distributions (KUD; Worton 1989) to delineate home range boundaries for each pronghorn during each winter. I then combined all individual 90% KUD boundaries resulting in a single polygon for the Dunlap Ranch (4447.7 km²).

Home ranges for individual pronghorn were defined by their respective locations within winter range. Winter ranges for resident individuals were defined by the majority of movements that occurred in a given year and season. For example, the majority of pronghorn that displayed seasonal movements moved in early April 2010 from winter to summer range (Figure 2.1). Consequently, for resident (i.e., no movement to seasonal ranges) Dunlap Ranch pronghorn, home ranges were defined by individual locations occurring between January 2010–Apr 16, 2010 for winter 2010, December 1, 2010–April 1, 2011 for winter 2010–2011, and Nov 1, 2011–Apr 30, 2012 for winter 2011–2012. The majority of pronghorn movements from winter to summer range occurred in late March–early May and varied from 2010 to 2011 (Figure 2.1).

Sampling units.—I used sampling units with 200 m radii to estimate frequency of use for pronghorn in Dunlap Ranch. Sampling units should be small enough to detect changes in animal movements, large enough to ensure multiple locations can occur in each unit, and should be larger than spatial error in locations and spatial variables (Sawyer et al. 2006, Nielson and Sawyer 2013). Previous studies have used sampling units with 100 m radii for mule deer with 2 hour fix rates; however, locations for pronghorn captured on the Dunlap Ranch fixed every 7 hours resulted in greater distances moved between fixes. Thus, I chose to use a larger sampling unit of 200 m radii previously used for elk studies (Nielson and Sawyer 2013). I generated

random points where 85% of available habitat (KUD population home ranges) was covered by sampling units with 200 m radii, similar to the coverage used by Nielson and Sawyer (2013). I averaged distance (Dist) to nearest access road (DistAccrd), wind energy facility (DistFac), wind turbine (DistTurb), fence (FenceDist), major road (MajorRd), minor road (MinorRd), and elevation, slope, standard deviation in snow depth (SnowdSD), and terrain ruggedness (VRM) predictor variables within each sampling unit. In addition, I calculated the majority category for aspect (AspectMaj; 0 =flat, 1 =south, and 2 =north), density of fences (FenceDen), and percent sagebrush within each sampling unit (Table 2.1).

RSF modeling.—I pooled locations across winters (2010, 2010–2011, 2011–2012) and individuals to identify an overall best fit model for pronghorn captured at the Dunlap Ranch. Prior to model development, I tested predictor variables for collinearity (r > | 0.6 |) and did not allow correlated variables to be included in any same model. I first screened individual variables by determining whether 85% confidence intervals for coefficients overlapped 0 (Arnold 2010). Variables with coefficients that overlapped 0 were not included in model building. In addition, I evaluated instability of coefficients for paired predictor variables in the same model. Moderately correlated variables (| 0.3 | < r < | 0.6 |) may result in instability of coefficients where signs switch and standard errors are inflated (Doherty 2008). If paired variables resulted in sign switching of coefficients, I did not allow these variables to be included in any same model.

I used a 2 step, sequential modeling approach. First, I identified a best fit model for each of the 3 variable classes separately: 1) environmental 2) anthropogenic and 3) wind energy. I tested all model combinations of predictor variables that passed initial screening for each variable class. I identified the best model for each of the 3 variable classes using Akaike's Information Criterion (AIC; Burnham and Anderson 1998, 2004). I calculated differences in

AIC from the top model (Δ AIC). The model with the lowest AIC score was identified as being the best fit model; however, models within 4 AIC points of the best fit model were considered competitive (Arnold 2010). Second, I allowed individual predictor variables in top models for each variable class to compete in all possible combinations to determine the overall best fit model for pronghorn winter resource selection on the Dunlap Ranch. I calculated the variance inflation factor for coefficients in all top models, which measures variance inflation caused by collinearity between variables. Variance inflation factor values ≤ 5 indicate that variance of coefficients is not inflated as a result of collinearity between variables in the model (Heiberger and Holland 2004).

Once a best fit model was identified for the Dunlap Ranch, I bootstrapped the model 1000 times across 77 randomly selected individuals (total individuals used in the resource selection modeling) to derive a standard error from each coefficient in the model, which accounted for individual variation. Individuals were represented by a given pronghorn ID and winter; thus, each pronghorn may have been represented up to 3 times (1 for each winter). I used an offset term equal to the natural log of the total number of locations unique to each bootstrap iteration. Each bootstrap iteration consisted of randomly selecting 77 individuals (total possible individuals across 3 winters) with replacement and estimating the intercept and coefficients for the top model. From the 1000 bootstraps, I then calculated the standard error and 95% CI for each coefficient (Nielson and Sawyer 2013).

I developed a predictive map of pronghorn winter resource selection in the Dunlap Ranch. I first averaged predictor variable values within circular units of 200 m radii to ensure predictions were carried out at the same scale as modeling (Nielson and Sawyer 2013). I then applied top model equations to predictive variable layers. Each cell was assigned to a use

category delineated by 4 quantiles: 1 = high use, 2 = medium high use, 3 = medium low use, and 4 = low use.

I validated the best fit model for the Dunlap Ranch using a 5-fold cross validation (Koper and Manseau 2010). I randomly assigned pronghorn winter locations to 5 folds. Four folds were used as the predictive set and one fold was retained as the validation set. I ran the model for the predictive set, separated model output into 10 equal area use bins (1 = low use, 10 = high use), and quantified the frequency of validation locations within each bin. I tested correlation between frequency of validation locations and bins using the Spearman's rank correlation coefficient (r_s) and averaged across the 5 correlation coefficients to get an overall correlation value for the 5fold cross validation. A strong positive correlation would indicate that my model had good predictive performance because more pronghorn locations would fall into higher-ranked probability bins (Boyce et al. 2002).

Individual level resource selection

To evaluate the influence of wind energy development on individuals within their respective home ranges, I only considered individuals with home ranges that overlapped wind energy and employed methods outlined by Nielson and Sawyer (2013). I first applied an independent set of sampling units to each individual home range and modeled relative frequency of use for the individual as a continuous response variable using a generalized linear model and negative binomial distribution (Nielson and Sawyer 2013). Rather than developing an overall best fit model for each individual, I applied the best fit model identified at the population level (see above) to each individual on its respective home range. To identify potential avoidance of wind energy, I tested coefficients for distance to wind energy against zero using a one-sample *t*-test (Marzluff et al. 2004, Sawyer et al. 2006).

Traffic monitoring

I monitored traffic at six locations along access roads at the Dunlap Ranch using TrailMaster TM1550 infrared trail monitoring system (Goodson and Associates, Inc., Lenexa, KS) from 6 December 2011–30 January 2012 and from 29 February–27 April 2012. I attached each trail monitor to a steel fence post approximately 1 m off the ground. I calculated average daily traffic events at each location.

RESULTS

Displacement

The deaths of 2 pronghorn in 2010 and 2 in 2011 were attributed to capture stress, and were not included in my analyses. Thus, I evaluated daily net displacement as a function of distance to nearest wind turbine for 47 adult female pronghorn on the Dunlap Ranch. I used observations acquired over 3 winters with 10,468 locations in winter 2010 (n = 32 females, mean locations per animal = 334, range: 92–466), 7,712 locations in winter 2010–2011 (n = 27 females, mean locations per animal = 293, range: 31–485), and 8,606 locations in winter 2011–2012 (n = 23 females, mean locations per animal = 381, range: 32–577). I found no relationship between daily net displacement and distance to wind energy infrastructure for pronghorn on the Dunlap Ranch during winter 2010 ($r^2 = 0.012$), winter 2010–2011 ($r^2 = 0.007$), or winter 2011–2012 ($r^2 = 0.001$; Figure 2.2).

Population level resource selection

I used 24,339 locations from 47 individuals (average locations per animal = 518; range: 85– 1,171) during winters 2010, 2010–2011, and 2011–2012 to develop a population level resource selection function for pronghorn in the 4447.7 km² of winter habitat identified within a 90% KUD for the Dunlap Ranch study area (note: number of locations used in the RSF analysis is less than that of the displacement analysis due to exclusion of locations outside of 90% KUD boundaries). There were 2 competitive models ($\Delta AIC = 2$) for the environmental variable class inclusive of percent sagebrush (Sage), percent slope (Slope), and standard deviation in snow depth (SnowdSD); however, model fit for percent slope (Slope) and standard deviation in snow depth (SnowdSD) did not improve with the addition of percent sagebrush (Sage; AIC increased by 2 points; Table 2.2). Thus, I identified the model inclusive of standard deviation in snow depth (SnowdSD) and percent slope (Slope; $w_i = 0.731$) as the top environmental model and brought it forward to compete with variables from other variable classes. The top anthropogenic model included predictor variables for density to fences (FenceDen), distance to nearest major road (MajorRd), and distance to nearest minor road (MinorRd; $w_i = 0.881$, Table 2.2). The top model for the Wind Energy variable class was not competitive with any other models (Δ AIC = 15, $w_i = 0.999$) and included distance to nearest wind energy facility (DistFac; Table 2.2). Both distance to major road (MajorRd) and distance to minor road (MinorRd) were moderately correlated with distance to nearest wind facility (DistFac; r = 0.30-0.55) thus, these variables were not permitted to occur in any same model when brought forward to compete for overall best fit model.

The overall top model included predictor variables from environmental (Slope and SnowdSD), anthropogenic (FenceDen), and wind energy (DistFac) variable classes (Table 2.3). Pronghorn selected areas with lower percent slope, lower standard deviation in snow depth, lower densities of fences, and closer to wind energy facilities (Figure 2.3). The high use category (top 25%) averaged 2.6 percent (95% CI: 2.6–2.6) slope, 7.9 cm (95% CI: 7.9–8.0) standard deviation in snow depth, 32.6 km/km² (95% CI: 31.9–33.2) of fences, and 9.7 km (95% CI: 10.8–10.8 km) from wind energy facilities (Figure 2.4).

Individual level resource selection

I applied the overall best fit population level model (Slope, SnowdSD, FenceDen, and DistFac) to 37 individuals (total locations = 13205; average locations per animal = 348; range: 31–499) whose available habitat (based on 90% KUD's) encompassed wind energy facilities during winters 2010, 2010–2011, and 2011–2012. Coefficients for distance to nearest wind energy facility did not differ from zero during winter 2010 (mean = -0.053, $t_{10} = -1.19$, P = 0.260), winter 2010–2011 (mean = 0.014, $t_{10} = 0.48$, P = 0.640), and winter 2011–2012 (mean = -0.042, $t_{15} = -1.89$, P = 0.079; Figure 2.5).

Traffic assessment

Due to traffic monitor malfunction, I was unable to use traffic data collected from 1 traffic monitor during 29 February–27 April 2012. Thus, I evaluated 1094 traffic events collected at 5 traffic monitor locations during 6 December 2011–30 January 2012 (718 traffic events) and 29 February–27 April 2012 (376 traffic events). Daily traffic ranged from 0.9–3.6 with an average of 1.8 (95% CI: 1.4–2.3) traffic events per day during 6 December 2011–30 January 2012 and 1.9 (95% CI: 1.5–2.4) during 29 February–27 April 2012.

DISCUSSION

My study provided a unique opportunity to assess the impacts of wind energy on wintering pronghorn and to gain insights into the differences in behavioral responses between wind and oil and gas development. Contradictory to previous research documenting the influence of oil and gas energy development on ungulate avoidance and changes in movement behavior (Bradshaw et al. 1997, Dyer et al. 2001, Sawyer et al. 2006 and 2009, Webb et al. 2011, Beckmann et al. 2012), I did not detect avoidance or behavior shifts in movement by pronghorn in response to wind energy development on the Dunlap Ranch. At the population level, pronghorn selected for

habitat with lower slopes, lower standard deviation in snow depth, lower densities of fences, and distances closer to wind energy facilities. When applying the best fit population level model to pronghorn at the individual level and only evaluating individuals with home ranges that overlapped wind energy, I found that distance to wind facilities did not influence pronghorn frequency of use of sampling units. In addition, pronghorn displacement was not influenced by vicinity to wind energy infrastructure.

Pronghorn selection for lower density of fences on the Dunlap Ranch is consistent with previous research documenting pronghorn avoidance of fences (Sheldon 2006). In addition, fences are known to contribute to direct mortalities and act as barriers sometimes restricting access to vital wintering areas utilized during harsh weather events (Oakley and Riddle 1974, O'Gara and Yoakum 2004, Harrington and Conover 2006). Previous studies have documented pronghorn selection for less snow accumulation during winter (Beckmann et al. 2012). Selection for lower snow depth is a beneficial survival strategy considering pronghorn mortality rates are elevated during winters with increased snow accumulation (Barrett 1982, Pyrah 1987). Whereas selection for low snow depths is important for pronghorn use of winter range, my results suggest variation in snow depth may be just as critical. Pronghorn selection for habitat encompassing lower standard deviations of snow depth may illustrate their affinity for areas with more predictable weather conditions. By avoiding areas with greater variations in snow depth, pronghorn may be able to prevent exposure to unforeseen weather events that may contribute to higher energy costs and ensuing lower survival on winter range. Overall, my results illustrate that fence densities and variability in snow depth are important contributors to habitat selection by wintering pronghorn in south-central Wyoming and must be considered when implementing

further modifications (energy and other human developments) to these harsh environmental landscapes to minimize impacts to pronghorn.

The population level model indicated pronghorn were selecting for areas closer to wind energy on the Dunlap Ranch; however, results from my evaluation of individual pronghorn do not suggest that pronghorn actually selected for wind energy infrastructure components. Rather, I believe this relationship may have resulted from capture occurring at a localized level within the central portion of the study area. While pronghorn were initially captured within 12 km of the Dunlap Ranch, individual home ranges sometimes extended up to 80 km beyond capture locations. Despite their ability to move large distances, pronghorn were generally found in the central portion of their home range showing some fidelity to the habitat near their origin of capture. As a result of available habitat being large and the majority of pronghorn locations being located near the origin of capture, it appeared that pronghorn selected for areas closer to wind energy. Thus, it was necessary to focus on individuals directly exposed to development (i.e., home ranges for animals that overlapped wind energy development) to isolate the effects of wind energy on wintering pronghorn.

The lack of avoidance and lack of increasing movement rates closer to wind energy development exhibited by pronghorn in the Dunlap Ranch study area may be a result of the overall extent and difference in human activity associated with wind energy development. Previous studies illustrate that ungulate avoidance of oil and gas development is most often associated with increased traffic and human presence (Dyer et al. 2001, Sawyer et al. 2009, Beckmann et al. 2012). My analysis of traffic loads on the Dunlap Ranch did not act as a direct comparison to oil and gas traffic levels; however, it did provide a rough assessment for understanding potential differences in wind energy and oil and gas traffic loads. Average daily

traffic on the Dunlap Ranch (1.8 (95% CI: 1.4–2.3) and 1.9 [95% CI: 1.5–2.4] detections per day) was lower than that at LGS (liquid gathering system; 3.3–3.6 detections per day) well sites, non–LGS wells (7.3–8.4 detections per day), and active drilling pads (85.3–112.4 detections per day) on the Pinedale Anticline Project Area in southwestern Wyoming during January–March 2006 and 2007 (PAPA; Sawyer et al. 2009). Mule deer avoided all types of well pads but avoided non–LGS and active drill pads (areas with higher traffic) more than LGS (lower traffic) well pads. Thus, traffic loads in the Dunlap Ranch study area were generally lower than those that facilitated avoidance behavior by mule deer in the PAPA (Sawyer et al. 2009). In addition, oil and gas fields tend to be larger, result in greater length of access roads, and greater proportion of habitat changed than wind energy production facilities (Jones and Pejchar 2013). Paired with less overall length of access roads and less habitat developed, lower traffic rates observed within the Dunlap Ranch may explain why avoidance and increases in movement rates were not observed by pronghorn.

For a species already predisposed to high mortality rates on winter range due to harsh environmental conditions and high energy demands (Wesley et al. 1970, Schwartz et al. 1977, Parker et al. 1984, Byers 1997), special care must be taken when considering further development to winter landscapes used by pronghorn. For managers to better assess the impacts of new developments such as wind energy to pronghorn populations, direct assessment of the energy development of interest is necessary given differences in infrastructure, surface disturbance, overall size, traffic levels, and human presence between different types of energy development. Pronghorn exposed to wind energy development on winter range in my study did not elicit the same behaviors as pronghorn and other species exposed to oil and gas development. Lower levels of human activity in wind energy facilities are potentially responsible for the lack

of avoidance demonstrated by pronghorn on the Dunlap Ranch; however, direct and more detailed comparisons between human disturbance in oil and gas fields and wind energy facilities are needed to more accurately inform managers of potential impacts to wildlife across these types of energy developments. Further, caution must be taken when considering pronghorn populations exposed to wind energy developments encompassing extents and intensities larger than those evaluated within the scope of my analysis. My results suggest pronghorn may not avoid wind energy infrastructure at larger scales of development; however, pronghorn response may deviate from results reported here if larger scale wind energy developments encompass higher levels of human activity (i.e., increased traffic rates and longer construction periods) than those associated with our study.

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Table 2.1. Predictor variables considered in resource selection models for pronghorn exposed to wind energy development on winter range in Dunlap Ranch, south-central Wyoming, USA, winters 2010, 2010–2011, and 2011–2012.

Variable class/variable	Description
Environmental	
AspectMaj	Majority category of aspect derived from a 30 m Digital Elevation Model (DEM; $0=$ flat [$\leq 2^{\circ}$], 1= south, 2= north)
Elevation	Elevation of the landscape derived from a 30 m DEM
Sage	Majority category of landcover $(1 = \text{sagebrush}, 0 = \text{other})$ derived from classified land cover data (sagebrush or other; US Geological Survey GAP 2011; 30 m cell size)
SnowdSD	Standard deviation of daily snow depth for all winters (NOHRSC 2004, SNODAS 2013; 30 m cell size)
Slope	Slope of the landscape derived from a 30 m DEM
VRM	Vector Ruggedness Measure (VRM; Sappington et al. 2007) scaled to 1000 (VRM x 1000) calculates terrain ruggedness within a 3 cell size neighborhood
Anthropogenic	
FenceDen	Density of fences (m/m^2) , digitized using NAIP imagery (2009)
FenceDist	Euclidean distance to nearest fence (km; digitized using NAIP imagery [2009])
MajorRd	Euclidean distance to nearest major road (km; U.S. highways, State highways, and Interstate highways) derived from a 30 m DEM
MinorRd	Euclidean distance to nearest county road (km) derived from a 30 m DEM

Variable class/variable	Description
Wind Energy	
DistAccrd	Euclidean distance to nearest wind facility access road (km) derived from a 30 m DEM
DistFac	Euclidean distance to nearest wind energy facility building (km; substations and employee and maintenance buildings) derived from a 30 m DEM
DistTurb	Euclidean distance to nearest wind turbine (km) derived from a 30 m DEM

Table 2.2. Model fit statistics for environmental, anthropogenic, and wind energy variable classes used to evaluate pronghorn resource selection in the Dunlap Ranch, south-central Wyoming, USA, winters 2010, 2010–2011, and 2011–2012. Number of parameters in each model (*K*), log likelihood (LL), Akaike's Information Criteria (AIC), difference in AIC from the top model (Δ AIC), and Akaike's weights (*w*_i) are also reported.

Variable class model	Κ	LL	AIC	ΔΑΙϹ	Wi
Environmental					
Slope + SnowdSD	3	-26031.00	52068.000	0	0.731
Sage + Slope + SnowdSD	4	-26031.00	52070.000	2.000	0.269
Slope	2	-26046.00	52096.000	28.000	0.000
Sage + Slope	3	-26045.50	52097.000	29.000	0.000
Sage + SnowdSD	3	-26636.00	53278.000	1210.000	0.000
SnowdSD	2	-26638.00	53280.000	1212.000	0.000
SagePerc	2	-26641.00	53286.000	1218.000	0.000
Null	1	-26643.50	53289.000	1221.000	0.000
Anthropogenic					
MajorRd + MinorRd + FenceDen	4	-26371.50	52751.000	0	0.881
MajorRd + MinorRd	3	-26374.50	52755.000	4.000	0.119
MajorRd + FenceDen	3	-26407.50	52821.000	70.000	0.000
MajorRd	2	-26410.00	52824.000	73.000	0.000

Variable class model	K	LL	AIC	ΔΑΙϹ	Wi
MinorRd + FenceDen	3	-26551.00	53108.000	357.000	0.000
MinorRd	2	-26553.00	53110.000	359.000	0.000
FenceDen	2	-26642.00	53288.000	537.000	0.000
Null	1	-26643.50	53289.000	538.000	0.000
Wind Energy					
DistFac	2	-25292.00	50588.000	0	0.999
DistTurb	2	-25299.50	50603.000	15.000	0.001
DistAccrd	2	-25327.50	50659.000	71.000	0.000
Null	1	-26643.50	53289.000	2701.000	0.000

Table 2.3. Model fit statistics for all possible combinations of variables within the top models for each variable class used to evaluate pronghorn resource selection in the Dunlap Ranch, south-central Wyoming, USA, winters 2010, 2010–2011, and 2011–2012. Number of parameters in each model (K), log likelihood (LL), Akaike's Information Criteria (AIC), difference in AIC from the top model (Δ AIC), and Akaike's weights (wi) are also reported.

Model	K	LL	AIC	ΔAIC	wi	Rank
Slope + SnowdSD + DistFac + FenceDen	5	-24701.00	49412.00	0.00	1.000	1
Slope + SnowdSD + DistFac	4	-24710.00	49428.00	16.00	0.000	2
Slope + DistFac + FenceDen	4	-24766.50	49541.00	129.00	0.000	3
Slope + DistFac	3	-24782.50	49571.00	159.00	0.000	4
SnowdSD + DistFac + FenceDen	4	-25119.50	50247.00	835.00	0.000	5
SnowdSD + DistFac	3	-25123.00	50252.00	840.00	0.000	6
DistFac + FenceDen	3	-25284.50	50575.00	1163.00	0.000	7
DistFac	2	-25292.00	50588.00	1176.00	0.000	8
Slope + SnowdSD + MajorRd + MinorRd	5	-25819.00	51648.00	2236.00	0.000	9
Slope + SnowdSD + MajorRd + MinorRd + FenceDen	6	-25816.00	51644.00	2232.00	0.000	10
Slope + MajorRd + MinorRd + FenceDen	5	-25820.00	51650.00	2238.00	0.000	11
Slope + MajorRd + MinorRd	4	-25829.50	51667.00	2255.00	0.000	12
Slope + SnowdSD + MajorRd + FenceDen	5	-25853.00	51716.00	2304.00	0.000	13

Model	K	LL	AIC	ΔΑΙϹ	wi	Rank
Slope + SnowdSD + MajorRd	4	-25855.00	51718.00	2306.00	0.000	15
Slope + MajorRd + FenceDen	4	-25863.50	51735.00	2323.00	0.000	14
MajorRd + Slope	3	-25872.00	51750.00	2338.00	0.000	16
Slope + SnowdSD + MinorRd + FenceDen	5	-25947.00	51904.00	2492.00	0.000	17
Slope + MinorRd + FenceDen	4	-25950.00	51908.00	2496.00	0.000	18
Slope + SnowdSD + MinorRd	4	-25949.50	51907.00	2495.00	0.000	19
MinorRd + Slope	3	-25958.00	51922.00	2510.00	0.000	20
Null	1	-26643.50	53289.00	3877.00	0.000	40

Table 2.4. Model-averaged parameter estimates for the top model predictive of pronghorn winter resource selection in the Dunlap Ranch, Wyoming, USA, winters 2010, 2010–2011, and 2011–2012. Standard errors (SE) for coefficients (β) were estimated from bootstrapping across individual pronghorn.

Variable class			Coefficient 95% CI		
variable	β	Bootstrap (SE)	Lower	Upper	
Environmental					
Slope	-0.105	0.0004	-0.106	-0.104	
SnowdSD	-0.053	0.0009	-0.054	-0.051	
Anthropogenic					
FenceDen	-0.088	0.0007	-0.089	-0.087	
Wind Energy					
DistFac	-0.070	0.0005	-0.070	-0.069	



Figure 2.1. Occurrence of pronghorn seasonal movements from winter to summer range (n = 31 in 2010 and n = 13 in 2011; A) and summer to winter range (n = 27 in 2010 and n = 9 in 2011; B) in the Dunlap Ranch, south-central Wyoming, USA.



Figure 2.2. Daily net displacement plotted as a function of distance to nearest wind turbine for pronghorn during winters 2010, 2010–2011, and 2011–2012 in the Dunlap Ranch study area, south-central Wyoming, USA.



Figure 2.3. Relative probability of use for the best fit model (slope [%], standard deviation in snow depth [cm], density of fences [km/km²], and distance to wind energy facility[km]) predictive of adult female pronghorn resource selection in the Dunlap Ranch, south-central, Wyoming, USA, winters 2010, 2010–2011, and 2011–2012.



Figure 2.4. Categories of habitat use (high, medium high, medium low, and low) predicted from top models for pronghorn winter resource selection on the Dunlap Ranch, Wyoming, USA, winters 2010, 2010–2011, and 2011–2012. The area depicted in white corresponds to Seminoe Reservoir, which was excluded from pronghorn resource selection analysis.


Figure 2.5. Boxplots of coefficients for distance to nearest wind facility (wind energy predictor variable) for pronghorn with home ranges that overlapped wind energy for each winter season. Boxplots depict medians (black horizontal line within interquartile range), quartiles, and ranges for 11 pronghorn in winter 2010, 11 pronghorn in winter 2010-2011, and 16 pronghorn in winter 2011-2012.

CHAPTER 3. Winter Mortality Risk for Pronghorn Exposed to Wind Energy Development

In the format for manuscripts submitted to The Journal of Wildlife Management

ABSTRACT

The demand for renewable energy has increased substantially in recent years. Landscapes with high potential for wind energy development in the western United States often coincide with suitable wintering habitat for pronghorn (Antilocapra americana). Evaluating the influence of energy development on pronghorn winter survival is particularly critical given that they encounter elevated energetic demands during this time of year. The purpose of my study was to evaluate pronghorn mortality risk on a landscape developed for wind energy production (Dunlap Ranch) over 3 winters (2010, 2010–2011, 2011–2012) in south-central Wyoming, USA. I captured and attached GPS-transmitters to 52 female pronghorn in January 2010 and December 2011 at the Dunlap Ranch and monitored these animals via fixed-wing flights approximately 4 times per year. Twenty-four pronghorn from the Dunlap Ranch study area died with the majority of deaths (n = 13) occurring in winter 2010–2011. Survival (\hat{S}) at the end of my 2.5-year study period was 0.30 (90% CI: 0.17–0.43) at Dunlap Ranch. Survival (Ŝ) in the Dunlap Ranch study area during winter 2010 was 0.97 (90% CI: 0.92-1.00), winter 2010-2011 was 0.53 (90% CI: 0.37-0.70), and winter 2011-2012 was 0.91 (90% CI: 0.82-1.00). I modeled mortality risk for pronghorn in the Dunlap Ranch using the Cox proportional hazards model inclusive of cumulative, weekly, and monthly temporal scales. I specifically modeled anthropogenic, environmental, and wind energy classes of predictor variables separately, which included timestatic (e.g., distance to wind turbine, percent of locations in sagebrush [Artemisia spp.] cover) and time-dependent covariates (e.g., temperature and snow depth) and allowed top models to

compete with each other in all possible combinations to identify an overall top model. Pronghorn winter mortality risk for Dunlap Ranch individuals decreased by 20% with every 1.0 km increase in average distance from major roads (hazard ratio = 0.80, 90% CI: 0.66-0.98), decreased by 4.0% with every 1% increase in average time spent in sagebrush (hazard ratio = 0.96, 90% CI: 0.95-0.98), and decreased by 92% with every 1 unit (VRM × 1000) increase in terrain ruggedness (hazard ratio = 0.08, 90% CI: 0.01-0.67). Pronghorn winter mortality risk was not influenced by exposure to wind energy infrastructure; however, survival of pronghorn exposed to larger-scale wind energy development needs further study.

INTRODUCTION

The recent increase in the demand for energy has raised concerns about impacts of renewable energy development to wildlife (Erickson et al. 2001, Piorkowski and O'Connell 2010). Response to novel infrastructure and increased human activity may have short and long term impacts to wildlife populations. Caribou (*Rangifer tarandus*), mule deer (*Odocoileus hemionus*), and pronghorn (*Antilocapra americana*) are known to avoid oil and gas developments (Dyer et al. 2001, Sawyer et al. 2006 and 2009, Beckmann et al. 2012); however, studies evaluating the impacts of wind energy development to ungulate populations are limited. A recent study using 10 marked animals found no evidence supporting decreased nutritional status or increased displacement of elk (*Cervus elaphus*) in response to wind energy development (45, NEG Micon 1.65 MW turbines) in Oklahoma (Walter et al. 2006). Although some aspects of elk response to wind energy development were evaluated in this Oklahoma study, impacts of wind energy development for other ungulate species remain unknown.

In northerly regions, pronghorn winter range is typified by harsh environmental conditions resulting in increased thermoregulatory costs and overall energy expenditure

(Schwartz et al. 1977, Parker et al. 1984, Miura and Muruyama 1986, Byers 1997). Temperate ungulates cope with the increased demand for energy by utilizing fat reserves and increasing their time spent foraging (Miura and Muruyama 1986, Franzmann et al. 1978, Tyler and Blix 1990). Pronghorn forage more during winter than other times of the year, spending up to 55% of their time feeding to make up for this increased demand in energy (Byers 1997). In addition, pronghorn have higher energy demands relative to body size than most domestic ruminants, resulting in higher stress during periods of food deprivation (Wesley et al. 1970).

Previous studies have identified malnutrition (Barrett 1982, O'Gara 2004a), severe weather (Martinka 1967, Barrett 1982, Pyrah 1987, O'Gara 2004a), barriers to movements (O'Gara 2004b, Harrington and Conover 2006, Jacques et al. 2007), hunting (Jacques et al. 2007, Kolar et al. 2012), predation (Einarsen 1948, Connolly 1978, Byers 1997, Phillips and White 2003, Jacques et al. 2007, Brown and Conover 2011), and old age (O'Gara 2004a) as factors contributing to pronghorn mortality. In winter, deterioration of pronghorn body condition and depletion of fat stores play a key role in pronghorn survival (Wesley et al. 1970, Barrett 1982). Pronghorn may reduce chances of mortality by utilizing more rugged terrain (ridges, draws, and swales) during local weather events (Kitchen 1974, Richardson 2006) or areas with less snow during harsh winters (Barrett 1982, Parker et al. 1984, Miura and Muruyama 1986). In addition, availability and nutritional quality of forage may influence survival for pronghorn during winter months. Pronghorn mortality on winter range may also be impacted by fences and roads acting as barriers inhibiting pronghorn from reaching areas utilized during harsh weather events (Oakley and Riddle 1974, O'Gara 2004b). Vicinity to roads may also dictate risk of predation (James and Stuart-Smith 2000, Dyer et al. 2001). Coyotes (*Canis latrans*) are the primary predator of pronghorn fawns (O'Gara 2004*a*, Jacques et al. 2007, Brown and Conover 2011).

Moreover, coyote predation affects overall pronghorn population survival (Phillips and White 2003, Brown and Conover 2011) and predation on adult pronghorn by coyotes is also well documented (Einarsen 1948, deVos and Miller 2005, Jacques et al. 2007). Further, pronghorn may perceive high levels of disturbance around roads as a predation risk resulting in tradeoffs between fitness behavior and avoidance of disturbance (Frid and Dill 2002). Additional energy expended on avoidance behaviors could result in energy deficits and negative consequences to pronghorn survival.

I evaluated mortality risk for pronghorn exposed to wind energy development on winter range over the course of three winters (2010, 2010–2011, 2011–2012) at the Dunlap Ranch, Wyoming. I hypothesized that mortality risk for pronghorn on winter range was associated with greater exposure to risky anthropogenic and environmental landscape features. My specific objectives were to identify: 1) environmental and anthropogenic predictor variables influencing pronghorn mortality risk, and 2) whether wind energy infrastructure influenced pronghorn winter mortality. By synthesizing findings from other studies in the literature I evaluated five predictions relative to pronghorn winter mortality risk in my study: 1) Because pronghorn experience high energetic demands on winter range and incur high mortality associated with winter weather events (Martinka 1967, Schwartz et al. 1977, Barrett 1982, Parker et al. 1984, Miura and Muruyama 1986, Pyrah 1987, Byers 1997, O'Gara 2004a), I predicted that mortality risk would increase with increasing frequency of days below lower critical temperatures (-12 °C; Wesley et al. 1973) and increasing snow depth; 2) Because fences may prevent pronghorn from accessing areas vital to survival during severe winter weather (Oakley and Riddle 1974), I predicted that mortality risk would increase with decreasing distance to fences or increasing density of fences; 3) Because roads may increase predation and perceived predation risk (James

and Stuart-Smith 2000, Dyer et al. 2001, Frid and Dill 2002, Gavin and Komer 2006, Whittington et al. 2011) as well as increase access for hunters and poachers, I predicted that mortality risk would decrease with increasing distance from roads; 4) Because woody browse, in particular sagebrush (*Artemisia* spp.), is the primary dietary component for pronghorn on winter range in northern climates (Bayless 1969, Beale and Smith 1970, Mitchell and Smoliak 1971), I predicted that pronghorn spending greater proportions of time in sagebrush-dominated habitats would decrease mortality risk; and 5) Because pronghorn may avoid infrastructure associated with energy development (Beckmann et al. 2012), I predicted that mortality risk would increase with increasing exposure to wind energy development.

STUDY AREAS

I evaluated mortality risk for pronghorn in the vicinity of the Dunlap Ranch, 11.8 km north of Medicine Bow in Carbon County, Wyoming, USA. My study focused on a 36.5-km² wind energy facility completed on the Dunlap Ranch in spring 2010 sited within a larger area that included 1452.3 km² of rangeland (22.6% Bureau of Land Management, 7.6% state of Wyoming, and 69.7% private ownership) delineated by the Wyoming Game and Fish Department as pronghorn crucial winter range (Figure 3.1). PacifiCorp owned and operated the Dunlap Ranch wind-powered generation facility. Plans for development of the Dunlap Ranch wind energy facility were split into two phases. Phase I (Dunlap I) facilitated the construction of wind energy infrastructure on pronghorn crucial winter range from September 2009 to September 2010. Construction of the Dunlap I wind-powered generation facility included installation of 74 General Electric Company 1.5-MW sle¹ wind turbine generators (119 m tall), 28.3 km of access roads, 3 meteorological towers (79.9 m tall), an onsite 34.5/230-kV substation, and onsite maintenance buildings. Construction began in fall 2009 and concluded with the erection of wind

turbines from April 2010 to September 2010 (PacifiCorp Energy 2009). Highway 487 extended north and south through the center of the Dunlap Ranch and pronghorn crucial winter range (Figure 3.1).

In addition to Dunlap I, nine nearby wind energy projects in Carbon and Albany Counties were potentially accessible by pronghorn in my study (Figure 3.1). The Seven Mile Hill I and II wind energy projects include 66 and 13, 1.5-MW wind turbines (119 m tall) that became operational in December 2008 and is located approximately 14 km southwest of Dunlap I between Hanna and Medicine Bow, Wyoming. The High Plains I and McFadden Ridge I wind energy projects include 66 and 19, 1.5-MW (119 m tall) wind turbines installed in 2009 and is located 38 km southeast of Dunlap I in Albany and Carbon Counties near McFadden, Wyoming. Seven Mile Hill I and II, High Plains I, and McFadden Ridge I are owned and operated by Rocky Mountain Power, a subsidiary of PacifiCorp. Lastly, the Foote Creek Rim wind energy project is located approximately 35 km south of Dunlap I near Arlington, Wyoming (Rocky Mountain Power 2010, BLM 2011*a*). The Foote Creek Rim wind energy project encompasses a total of 182, 600-kW wind turbines (61 m tall) installed from 1999 to 2001 over five phases of development: Foot Creek I (68 wind turbines), Foot Creek II (3 wind turbines), Foot Creek III (33 wind turbines), Foot Creek IV (28 wind turbines), and Rock River I (50 wind turbines).

The Dunlap Ranch occurs in the southern portion of the broad, intermontane Shirley Basin, dominated by arid shrublands and grasslands. The region is marked by flats, high hills, and low mountains with Wyoming big sagebrush (*Artemisia tridentata wyomingensis*) being the most prevalent cover type. The Freezeout Mountains to the west support alderleaf mountain mahogany (*Cercocarpus montanus*), aspen (*Populus tremuloides*), and limber pine (*Pinus flexilus*). Elevations within the Dunlap Ranch range from 2,000 to 2,530 m. I acquired climate

data from the nearest High Plains Regional Climate Center weather station located 11.8 km south of the study area in Medicine Bow, Wyoming. Total snow fall for winter 2010 (November 2009–April 2010) was 136.6 cm, for winter 2010–2011 (November 2010–April 2011) was 212.6 cm, and for winter 2011–2012 (November 2011–April 2012) was 90.7 cm. Average minimum and maximum temperatures for winter 2010 (November 2009–April 2010) were –10.2°C and 3.2°C, for winter 2010–2011 (November 2010–April 2011) were –8.8°C and 2.9°C, and for winter 2011–2012 (November 2011–April 2012) were –9.7°C and 5.4°C (HPRCC 2012). The Little Medicine Bow and Medicine Bow Rivers are located just south of the Dunlap Ranch boundary with Muddy Creek, a tributary of the Medicine Bow River, flowing within the eastern border of the wind energy facility (PacifiCorp Energy 2009).

METHODS

Capture and Monitoring

The University of Wyoming contracted with Leading Edge Aviation, LLC (Lewiston, Idaho 83501) to capture 35 adult female pronghorn using the helicopter net gun technique in early January 2010 and 17 adult female pronghorn on 1 December 2011 in the Dunlap Ranch. Protocols were approved by the University of Wyoming Institutional Animal Care and Use Committee (01012010) and Wyoming Game and Fish Department (Chapter 33 Permit ID 742) to capture, handle, and mark female pronghorn. Each captured animal was examined to estimate age and fitted with an Advanced Telemetry System (Isanti, Minnesota 55040) store-on-board GPS neck collar (model G2110B) in or within the general vicinity of each study area. Collars were programmed to fix locations every 7 hours from16 Nov to 15 May and every 11 hours from16 May to 15 Nov. Locations were collected from January 2010 through May 2012. Aerial re-location flights were conducted 4 times annually to record the status of study animals. I

recovered collars transmitting a mortality signal to download locational data collected up to time of death. I assessed the cause of mortality for each dead pronghorn; however, the delayed recovery of collars made this assessment difficult in most cases. The remaining collars detached in late April 2012 and were recovered by late May 2012.

Fawn Surveys

Pronghorn have the highest energy output and peak milk volumes, relative to maternal weight, of any big game species in North America (Martin 1995). For pronghorn females, body condition in the fall may be representative of reproductive success for that spring. The high cost of lactation may greatly influence body condition throughout the reproductive period. Thus, females that have incurred the costs of reproduction may experience reduced survival on winter range (Dunn and Byers 2008). To establish a pre-existing body condition for pronghorn females on crucial winter range during winter 2011–2012, I conducted fawn surveys for all collared females on the Dunlap Ranch in August 2011. I also conducted fawn surveys for collared pronghorn in a neighboring population near Walcott Junction, Wyoming (see Appendix A for study area description). Fawn surveys were not conducted prior to winters 2010 and 2010–2011 due to timing constraints. I used aerial surveys with a Maule M7-234b fixed-wing aircraft (Owyhee Air, Murphy, Idaho 83650) and ground observations to identify fawns with collared females from 9 to 11 August 2011. In 10 cases I used ground surveys on 9-11 August 2011 to observe collared pronghorn too difficult to observe from the air. While flying, I used imagestabilized binoculars to increase the elevation of flight above animals, and, in turn, reduce herding pronghorn. It was important for the aircraft to avoid herding animals to prevent collared females from forming groups with other females and their fawns. Observers identified collared females and noted the presence or absence of fawns and number of fawns with each animal.

Each observer assessed their own ability to pair a fawn with each collared female on a scale of 1 to 3, where 1 was very confident and 3 was unsure.

Mortality Risk

Cox's (1972) proportional hazards regression (Cox PH) is a widely used method for analyzing risk through survival analysis (Johnson et al. 2004, Farmer et al. 2006, Chung et al. 2013, Gutierrez et al. 2013, Schelleman-Offermans et al. 2013). Cox PH can be used to predict occurrence of an event within a known time scale dependent on measured predictor variables (Cox 1972). Cox's model is capable of estimating regression models with censored data and utilizes estimation methods that allow for time-dependent covariates, which are predictor variables that may change in value across time and space (Allison 2010:153). I used the Andersen–Gill (A–G) formulation of the Cox PH model to evaluate mortality risk for pronghorn in Dunlap Ranch, which evaluates risk of mortality over time where each individual may be represented by multiple observations (Cox 1972, Andersen and Gill 1982). In addition, I estimated survival rates of collared pronghorn in the Dunlap Ranch by month over the course of the study period utilizing the Kaplan-Meier product-limit estimator (Kaplan and Meier 1958), modified for staggered entry (Pollock et al. 1989). I computed the variance for Kaplan-Meier survival estimates following Greenwood (1926) and used a log-rank test to compare survival rates between winters (Cox and Oakes 1984:105).

I included a spatial scale to capture potential movement by pronghorn between location fixes (every 7 hours from 16 Nov–15 May and every 11 hours from 16 May–15 Nov). Average movement for pronghorn between 7-hour locations was 980 m and was 1085 m for 11-hour locations. To account for variable exposure to time-dependent covariates through space, I thus

summarized time-dependent covariates within 980 m of locations collected every 7 hours and 1085 m of locations collected every 11 hours.

Cox's model assumes that the influence of time-dependent covariate X on probability of mortality at time t is dependent on the value of X at time t only (Allison 2010:153); further, the value of X at any prior time has no influence on probability of mortality at time t as affected by covariate X. For example, mortality of an individual may be influenced by the cumulative effects of low winter temperatures for an entire week prior to death; however, temperature at the exact time (t) mortality occurred will solely influence probability of mortality in the model. The use of large data sets presents a challenge when adopting a method to summarize time-dependent covariates to best represent their influence on mortality at time t of each observation in the model. Previous studies, inclusive of small sample sizes or VHF data, have used cumulative exposure to summarize time-dependent covariates (Johnson et al. 2004, Kirol 2012). Kirol (2012) averaged spatial time-dependent covariates from t = 0 (beginning of the study period) to the time of each observation; however, for large data sets often spanning several years and inclusive of thousands of locations per individual, averaging spatial covariates across study periods may not be the most efficient method to evaluate the effects of covariates on mortality risk. I thus evaluated the effects of time-dependent covariates over weekly and monthly scales in addition to cumulative scales (beginning of each winter, t = 0). For each observation in the model (time = t), I summarized (mean, standard deviation, accumulation) time-dependent covariates (snow depth and temperature) from the 1) beginning of each winter, 2) week prior to, and 2) month prior to time t. Evaluating several scales of time-dependent covariates allowed me to identify the most influential time period when these covariates were affecting mortality risk.

Covariates of Survivorship

The Cox PH model permits the use of both continuous and categorical covariates. I predicted pronghorn mortality risk in the Dunlap Ranch using time–independent and time–dependent covariates representative of environmental, anthropogenic, and wind energy variable classes. Time–independent predictor groups included age of pronghorn, fences, landcover, major roads, minor roads, movement rate of pronghorn, percent exposure to wind energy development, and terrain (Table 3.1). Time–dependent covariates included snow depth, temperature, and a winter severity index (Brinkman et al. 2005; Table 3.1).

O'Gara (2004*a*) identified old age as one of seven principal mortality factors for pronghorn; however, previous studies have found no evidence for this relationship (Kolar et al. 2012). Nonetheless, I evaluated the potential influence of age on mortality risk. I classified age to distinguish old from young pronghorn. The average lifespan for pronghorn in Wyoming and Montana has been observed to be about 9 years (O'Gara 2004*c*); however, Byers (1997) observed lifespans up to 15 years for adult females on the National Bison Range in Montana. Regardless, determining age of pronghorn via tooth wear often results in inaccurate age estimations beyond 5.5 years of age (Jensen 1998). Thus, I classified 'old' pronghorn as ≥ 5.5 years and 'young' pronghorn as 1.5–5.5 years old, based on tooth wear examination at time of capture.

Because sagebrush is the primary component of winter diets for many pronghorn populations in northern latitudes (e.g., Bayless 1969, Beale and Smith 1970, Mitchell and Smoliak 1971), I categorized landcover types as: 1) sagebrush or 2) other (including agriculture, marshland, mixed grassland, riparian, salt desert shrub, and rocky soils). I used the USGS GAP Landcover Data Set to categorize landcover types for mortality risk modeling (U.S. Geological

Survey GAP 2011). A recent study also considered agricultural lands and riparian areas as important landcover types for predicting pronghorn resource selection in southwest Wyoming (Beckman et al. 2012); however, riparian and agricultural lands composed <0.02% of the land coverage across the Dunlap Ranch study area, so they were not considered separately in my analyses.

I acquired daily snow depth measurements for the Dunlap Ranch study area from the Snow Data Assimilation System (SNODAS; NOHRS 2004). SNODAS is an assimilation model that integrates satellite, airborne platforms and ground stations to provide consistent estimates of snow cover; thus, snow depths derived from SNODAS models do not represent direct snow accumulation estimates. I modeled snow depth as a time-dependent covariate. I accumulated, averaged, and took the standard deviation of daily snow depth values for each individual from the beginning of each winter, across every week, and across every month prior to time t for each observation (Table 3.1). I acquired daily temperature lows for Chugwater, Medicine Bow, Rawlins, and Shirley Basin, Wyoming weather stations to represent weather conditions that pronghorn captured in the Dunlap Ranch study area may have encountered (HPRCC 2012) and paired each individual's locations with the daily low temperature for the nearest weather station. I only considered temperatures at or below pronghorn lower critical temperatures identified as – 12 °C by Wesley et al. (1973) and assigned them a value of 1. I modeled temperature as a timedependent covariate. I accumulated the temperature value for each individual from the beginning of each winter, across every week, and across every month prior to time t for each observation (Table 3.1). I also calculated a winter severity index (WSI) value using a combination of snow depth and temperature. Each day was given a value of 1 if the snow depth was \geq 35 cm or the temperature was \leq -12 °C. A value of 2 was given to days where both

simultaneously occurred (Brinkman et al. 2005). I then accumulated the WSI value for each individual from the beginning of each winter, across every week, and across every month prior to time t for each observation (Table 3.1).

I developed spatial layers in ArcGIS 10.0 for distance to nearest major road (Interstate Highways, US Highways and State Highways), distance to nearest minor road (county roads), slope, and vector ruggedness measure (VRM; Sappington 2007) using a 1-arc-second (30 m) National Elevation Dataset (NED; U.S. Geological Survey). VRM accounts for variation in slope and aspect to measure heterogeneity in landscape terrain (Sappington et al. 2007); I referred to this landscape covariate as terrain ruggedness. I digitized fences using 2009 National Agriculture Imagery Program (NAIP) imagery and used digitized fences to develop a spatial layer for distance to nearest fence and density of fences.

To represent the impact of wind energy development on pronghorn, I quantified percent exposure to energy infrastructure components. I calculated the percent of total locations per individual within 1 km and 2 km of wind turbines, access roads, and wind energy facility buildings (i.e., substations and employee and maintenance buildings; Table 3.1). Previous studies have used 1 km as an impact boundary for evaluating elk habitat selection in an oil and gas field (Harju et al. 2011); thus, I used a 1 km buffer to represent the impact of wind energy development on pronghorn. I also included a 2 km buffer of impact to account for potential impacts to pronghorn beyond 1 km, possibly resulting from the influence of tall vertical structures (wind turbines 61–119 m tall) associated with wind energy development on pronghorn.

Mortality Risk Modeling

I examined hourly risk of mortality for pronghorn on winter range in the Dunlap Ranch study area (winter 2010, 2010–2011, 2011–2012) as a function of covariates of survivorship. I utilized intervals of risk, rather than a single value for time, to accommodate discontinuous intervals of risk (Guo et al. 2008). Because I focused on winter locations specifically, time spent on summer 2010 and 2011 ranges were represented as discontinuous intervals of risk in the model. In addition to accommodating discontinuous intervals of risk, the A-G model allows for left, right, and interval censoring (Andersen and Gill 1982, Johnson et al. 2004). Pronghorn that survived the duration of the study period were right censored and pronghorn that died while on summer range were interval censored (Allison 2010:103). I used staggered entry for left-truncated data (Allison 2010:183) to accommodate pronghorn not introduced into the study until December 2011. Several locations were recorded daily for each pronghorn; thus, I evaluated mortality risk on an hourly time scale to retain all locational data for use in the model. I conducted statistical modeling using the packages AICcmodavg, Dynpred, Regression Modeling Strategies (RMS), and Survival in Program R version 2.12.2 (Putter 2011, R Core Team 2012, Therneau 2012, Harrell 2013, Mazerolle 2013).

I tested covariates of survivorship for collinearity (r > | 0.6 |) and did not allow correlated covariates to be included in any same model. In addition, multiple scales of the same time-dependent covariates were not considered in the same model to avoid biased model estimates. I calculated the variance inflation factor for coefficients in all top models, which measures variance inflation caused by collinearity between variables. Variance inflation factor values ≤ 5 indicate that variance of coefficients is not inflated as a result of collinearity between variables in the model (Heiberger and Holland 2004). I first screened individual variables by

determining whether the 85% confidence intervals for hazard ratios overlapped 1 (Arnold 2010). Variables with hazard ratios that overlapped 1 were not included in model building. I tested all model combinations for each hierarchical category of predictor variables. I identified the best model for each of the 3 variable classes using Akaike's Information Criterion (AIC; Burnham and Anderson 1998, 2004). I calculated differences in AIC from the top model (Δ AIC). The model with the lowest AIC score was identified as being the best fit model; however, models within 4 AIC points of the best fit model were considered competitive models (Arnold 2010). All competitive models within each variable class were considered top models and allowed to compete against top models for each of the other categories in all possible combinations (Carpenter et al. 2010) to determine the overall best fit model for pronghorn mortality risk in my study area. Because my final top models were competitive ($\Delta AIC < 4$), I model averaged across the 95% confidence sets to estimate coefficients and standard errors for variables occurring in all competitive model subsets (Burnham and Anderson 2002). I used the c(concordance) index to assess discrimination for all models within the 95% subset. The c index is capable of evaluating the predictive ability of risk models using time-dependent covariates and censored data (Harrell et al. 1996, Pencina and D'Agostino 2004). A c value equal to 0.5 indicates no concordance between observed and predictive responses. A c value equal to 1.0 indicates perfect concordance between observed and predictive responses (Harrell et al. 1996); however, models with c values ≥ 0.8 are considered to have good predictive discrimination (Pencina and D'Agostino 2004).

Cox's proportional hazard regression assumes that hazard ratios are proportional over time (Cox 1972); further, the effect of each covariate in the model should be the same throughout time (Allison 2010:172). The proportional hazards assumption is satisfied when Schoenfeld

residuals for a given covariate are not correlated with time (Schoenfeld 1982). Schoenfeld residuals have a separate residual for each covariate and each individual that experiences an event (Allison 2010:175). To test the proportional hazards assumption, I calculated Schoenfeld residuals for each covariate in the top model and plotted them against time (Hess 1995). I inspected residuals for uniform distribution and tested the fitted line for a non-zero slope (Johnson et al. 2004). Covariates with a zero slope for respective Schoenfeld residuals did not violate the proportional hazards assumption.

RESULTS

Survival Estimates

During my study, 24 of 52 (46.2%) female pronghorn died at the Dunlap Ranch (7 in 2010, 15 in 2011, and 2 in 2012). One female pronghorn died during winter 2010, 13 during winter 2010–2011, and 3 during winter 2011–2012 (Table 3.2 and Figure 3.2). Cause of mortality was attributed to hunter kills in 2 cases and unknown in 22 cases at Dunlap Ranch. I censored locational data in my survival analysis for 4 pronghorn where mortality was attributed to capture stress. The collars of 2 females from the Dunlap Ranch detached early due to low battery. In addition, I was unable to recover the collars of 1 pronghorn in the Dunlap Ranch. I thus estimated survival with 47 females from the Dunlap Ranch.

Kaplan-Meier (KM) product-limit survival estimates (\hat{S}) at the end of 2010 and 2011 were 0.81 (90% CI: 0.69–0.93) and 0.33 (90% CI: 0.19–0.47), respectively. Estimated survival (\hat{S}) at the end of the study period (April 2012) was 0.30 (90% CI: 0.17–0.43) at Dunlap Ranch (Figure 3.3). Survival (\hat{S}) in the Dunlap Ranch study area during winter 2010 was 0.97 (90% CI: 0.92–1.00), winter 2010–2011 was 0.53 (90% CI: 0.37–0.70), and winter 2011–2012 was 0.91 (90% CI: 0.82–1.00). Survival in the Dunlap Ranch was lower during winter 2010–2011 than

winters 2010 ($X_1^2 = 14.46$, P < 0.001) and 2011–2012 ($X_1^2 = 5.37$, P = 0.021); however, survival did not differ between winters 2010 and 2011–2012 ($X_1^2 = 0.81$, P = 0.368).

Body Condition

I observed fawns with 1 of 8 (12.5%) collared females in the Dunlap Ranch and with 11 of 21 (52.4%) collared females at Walcott Junction (Table 3.3). I was unable to locate 4 collared does at Dunlap Ranch due to collar malfunction or pronghorn movement out of the area. The collars of 2 does (1 with a fawn and 1 without a fawn) at Dunlap Ranch and 2 does (both without fawns) at Walcott Junction either detached early or individuals died prior to winter 2011–2012; thus, survival during winter 2011–2012 was unknown resulting in these individuals being censored from K-M survival analyses. Observer confidence was rated as "1" in 24 (82.8%) observations and as "2" in 5 (17.2%) observations. Winter 2011–2012 survival ranged from 0.83 to 1.00 and was not higher for does observed without fawns than does observed with fawns at Dunlap Ranch $(X_1^2 = 0.25, P = 0.617)$ and Walcott Junction $(X_1^2 = 0.80, P = 0.80;$ Table 3.4).

Mortality Risk Modeling

I evaluated mortality risk for 47 doe pronghorn from Dunlap Ranch using 26,772 observations and 17 mortality events. Observations were acquired over 3 winters with 10,468 observations in winter 2010 (n = 32 does, mean locations per animal = 334, range: 92–466), 7,712 observations in winter 2010–2011 (n = 27 does, mean locations per animal = 293, range: 31–485), and 8,606 observations in winter 2011–2012 (n = 23 does, mean locations per animal = 381, range: 32– 577). There were 3 top models (Δ AIC ≤2.982) for the environmental variable class inclusive of predictor variables for variation in snow depth within prior month (SnowdSD_mnth), percent time spent in sagebrush (SagePerc), number of days ≤–12 °C within prior month (Temp_mnth), and terrain ruggedness (VRMAvr; Table 3.5). Anthropogenic variables representing fences (distance to and density) and distance to minor roads (Table 3.1) were uninformative (85% CI for hazard ratios overlapped 1; Arnold 2010) and were not included as candidate variables. The top two models for the anthropogenic variable class were univariate and included average distance to major roads (MajorRdAvr) and decay distance to major roads at 3000 m (MajorRd_Decay3000; Table 3.5). Although models for average distance to major roads and decay distance to major roads were competitive (Δ AIC = 1.158), these two variables were highly correlated (r = -0.71) and their effect in combination with variables in other variable classes was similar. Thus, I selected the anthropogenic model with the lowest AIC value (MajorRdAvr) to compete with the top models in other variable classes. Similarly, all variables considered in the wind energy variable class were highly correlated (r > 0.95) and had similar effects in combination with variables. Thus, I selected the univariate wind energy model with the lowest AIC value (ExpAccrd_1km) to compete with the top models in the other variable classes. Thus, I selected the univariate wind energy model with the lowest AIC value (ExpAccrd_1km) to compete with the top models in the other variable classes (Table 3.5).

There were 9 competitive models ($\Delta AIC \leq 3.829$) for pronghorn winter mortality risk inclusive of predictor variables for variation in snow depth within prior month (SnowdSD_mnth), percent time spent in sagebrush (SagePerc), number of days ≤ -12 °C within prior month (Temp_mnth), terrain ruggedness (VRMAvr), average distance to major roads (MajorRdAvr), and percent of locations within 1 km of wind facility access roads (ExpAccrd_1km; Table 3.6). Relative importance was highest for percent time spent in sagebrush (1.00) and terrain ruggedness (1.00) and lowest for locations within 1 km of wind facility access roads (0.37). In addition, relative importance was similar for variation in snow depth within prior month (0.67), number of days ≤ -12 °C within prior month (0.63), and average distance to major roads (0.791). Model fit for environmental, anthropogenic, and environmental

+ anthropogenic models did not improve with the addition of wind energy (Table 3.6); rather, AIC values increased 0.033–1.517 with the addition of wind energy to these models (Table 3.6). Model-averaged parameter estimates from 95% confidence sets indicated predictor variables for variation in snow depth within prior month (SnowdSD mnth), number of days $\leq -12 \,^{\circ}$ C within prior month (Temp_mnth), and percent of locations within 1 km of wind energy facility access roads (ExpAccrd_1km) were uninformative (85% CI's for hazard ratios overlapped 1; Arnold 2010; Table 3.7). For model-averaged 95% model subsets, the hazard ratio for average distance to nearest major road was 0.80 (SE = 0.12, 90% CI: 0.66–0.98), for time spent in sagebrush was 0.96 (SE = 0.01, 90% CI: 0.95–0.98), and for terrain ruggedness (VRM \times 1000) was 0.08 (SE = 1.31, 90% CI: 0.01–0.67). Pronghorn winter mortality risk decreased by 20.0% with every 1 km increase in distance from nearest major road. In addition, mortality risk for pronghorn on winter range decreased by 4.0% with every 1% increase in average time spent in sagebrush and decreased by 92.0% with every 1 unit (VRM \times 1000) increase in terrain ruggedness (Table 3.7, Figure 3.4). All variance inflation factor values for coefficients within the 9 competitive models were ≤ 5 , indicating variance of coefficients did not increase as a result of collinearity between variables in the model (Heiberger and Holland 2004). In addition, all c index values for the 9 competitive models were ≥ 0.8 indicating my models had good concordance between observed and predictive mortality risk for pronghorn on winter range (Pencina and D'Agostino 2004).

DISCUSSION

I examined mortality risk for pronghorn on a winter landscape that included wind energy development in south-central, Wyoming. My results were consistent in supporting my hypothesis that mortality risk for pronghorn on winter range would be associated with greater exposure to riskier anthropogenic and environmental conditions. Consistent with three of my

predictions, I identified time spent in sagebrush habitats, average distance to major road, and terrain ruggedness as influential predictors for pronghorn winter mortality risk. Contrary to my predictions, exposure to wind energy infrastructure did not influence pronghorn mortality on winter range. Pronghorn on winter range already experience high energy demands and increased mortality rates (Barrett 1982, Pyrah 1987, Byers 1997, O'Gara 2004*a*); thus, the introduction of novel developments to these landscapes could lead to increased stress and avoidance behavior by pronghorn potentially resulting in further energy losses and increased mortality risk. Assessing the impacts of wind energy development to pronghorn survival is particularly critical given the recent growth of the industry and the potential for these sites to coincide with suitable wintering habitats.

Currently, few studies have evaluated the response of ungulates to wind energy development (Walter et al. 2006) and no studies have focused specifically on pronghorn. Previous studies have documented ungulate avoidance of infrastructure associated with oil and gas energy, especially where human presence was elevated (Dyer et al. 2001, Sawyer et al. 2009). Beckmann et al. (2012) did not detect avoidance of oil and gas wells by pronghorn but did document abandonment of highly disturbed areas. If pronghorn respond to wind energy development at the Dunlap Ranch as has been documented for other ungulate species exposed to oil and gas development (Dyer et al. 2001, Sawyer et al. 2009, Webb et al. 2011), abandonment of high quality habitats or avoidance of high levels of human activity could result in overall energy losses for individuals exposed to higher levels of wind energy development. I would have expected mortality risk to increase for individuals at higher wind energy exposure levels if they were experiencing higher levels of stress and allocating more energy towards avoidance behavior rather than fitness behaviors; however, pronghorn mortality risk was not influenced by

exposure to wind energy development at the Dunlap Ranch. Furthermore, pronghorn did not avoid wind energy development within their respective home ranges and pronghorn movement rates were not influenced by distance to wind energy infrastructure (see Chapter 2). Thus, wind energy development did not appear to negatively impact pronghorn behavior and survival on winter range at Dunlap Ranch, Wyoming.

The lack of avoidance and changes in movement behavior in response to wind energy development illustrated here may be a result of differences between wind energy and oil and gas developments. Generally, density of wind turbines within wind facilities (2.03 turbines/ km^2 at Dunlap Ranch) may be greater than well pads within oil and gas fields (Pinedale Anticline Project Area [PAPA] may allow up to 0.75 well pads/km²; BLM 2008); however, differences in human activity such as traffic levels may be more likely to influence avoidance and movement behavior of pronghorn than density of infrastructure alone. Mule deer and woodland caribou avoided oil and gas wells and roads where traffic levels were elevated (Dyer et al. 2001, Sawyer et al. 2009). If pronghorn respond similarly to human presence as has been documented with other ungulates, low traffic rates observed within the Dunlap Ranch may explain why pronghorn did not elicit an avoidance response (see Chapter 2). Alternatively, pronghorn may respond differently than other ungulates to energy development and increased human presence associated with it. Although pronghorn may have abandoned highly disturbed areas in an oil and gas field, pronghorn did not avoid wells (Beckmann et al. 2012). The lack of avoidance behavior documented for pronghorn exposed to oil and gas development in southwest Wyoming and wind energy development at the Dunlap Ranch may illustrate the ability of this species to tolerate or adapt to disturbance associated with energy development; however, Beckmann et al. (2012) did not report traffic levels associated with wells evaluated in their study. If traffic levels were

similar to those reported at the Dunlap Ranch, traffic rates may not have been high enough to elicit an avoidance response in either case. Lastly, the extent of development at the Dunlap Ranch may primarily be responsible for the lack of negative impacts reported in my study. Wind energy project areas are generally much smaller than oil and gas fields (e.g., the PAPA is 22times larger than the Dunlap Ranch wind energy facility). The Dunlap Ranch is nearly 6-times smaller (36.5 km²) than the average pronghorn winter home range size (208.1 km²) for individuals at the Dunlap Ranch. Pronghorn may not need to solely rely on winter range within the Dunlap Ranch if they can easily use areas beyond that developed by wind energy. However, pronghorn still did not avoid wind energy within their respective home ranges (see Chapter 2), suggesting pronghorn may not avoid wind energy developments extending over larger areas if disturbance (i.e., traffic and density of infrastructure) is similar to that at the Dunlap Ranch. Nevertheless, caution must be taken when considering pronghorn populations exposed to wind energy developments encompassing extents and intensities of development larger than those evaluated within the scope of my analysis. My results suggest pronghorn may not avoid wind energy infrastructure developed over larger areas; however, pronghorn response may deviate from results reported here if larger scale wind energy developments encompass higher levels of human activity (i.e., increased traffic rates and longer construction periods) than those associated with my study.

Although previous studies have documented avoidance of fences by pronghorn and high mortality rates for animals using habitat closer to fences during harsh weather events (Oakley and Riddle 1974, deVos and Miller 2005, Sheldon 2006), I found no evidence for distance to fences or density of fences being influential predictors for pronghorn mortality risk in my study. Distance to major roads, however, was an important anthropogenic and overall predictor variable

for pronghorn mortality risk (Tables 3.5 and 3.6). Pronghorn mortality risk was elevated for individuals closer to major roads (Table 3.7). Although I did not determine that any mortalities of collared pronghorn were due to collisions with vehicles, risk of mortality may have been elevated closer to major roads as a result of greater exposure to predators (Yoakum 1957, Phillips and White 2003, Jacques et al. 2007, Brown and Conover 2011) or pronghorn perceiving high levels of disturbance around highways as a predation risk (Frid and Dill 2002). Perceived risk may result in tradeoffs being made between behavior leading to higher fitness and avoidance of disturbance. For example, Gavin and Komer (2006) documented more time spent being vigilant and less time spent feeding by pronghorn near roads with high traffic levels as compared to those near roads with lower traffic levels. If pronghorn on the Dunlap Ranch selecting habitats closer to major roads elicited these same behaviors, reduced foraging time and increased vigilance could have resulted in increased energy expenditure and increased mortality risk.

In addition to major roads, I noted the importance of sagebrush to pronghorn mortality risk. Individuals that spent more time within sagebrush had lower mortality risk. This finding is supported by past work that identified shrubs as the primary food item forming pronghorn winter diets (Bayless 1969, Beale and Smith 1970, Mitchell and Smoliak 1971) and by resource selection analyses that documented pronghorn selection for sagebrush on winter range (Beckmann et al. 2012). Sagebrush has been documented to be important for survival of other sagebrush obligates (Oyler-McCance et al. 2001, Connelly et al. 2011, Smith et al. 2014); however, no studies have documented sagebrush as an influential predictor of adult pronghorn mortality risk. A decrease in mortality risk for pronghorn spending more time within sagebrush not only denotes the prevalence of consumption of this shrub by pronghorn, but more importantly denotes its intrinsic value for survival. Further, pronghorn exposed to less rugged

terrain on average were at greater risk of mortality than pronghorn exposed to more rugged terrain. Assuming more rugged terrain encompassed steeper slopes, these findings are consistent with previous research that identified higher survival rates for migrant pronghorn in areas with steeper slopes (Barnowe-Meyer et al. 2010). My results further support previous observations that pronghorn may utilize ridges, draws, and swales to avoid exposure to harsh weather events (Richardson 2006), thus potentially reducing mortality risk.

Lower survival rates during winter 2010–2011 on the Dunlap Ranch likely resulted from harsh weather conditions. Total snow fall for Medicine Bow, Wyoming from December to February was 1.4–1.9-times (30.2–53.1 cm) greater during winter 2010–2011 (114.6 cm) than winters 2010 (61.5 cm) and 2011–2012 (84.4 cm; HPRCC 2012). Similarly, previous studies have identified lower survival rates for pronghorn during winters with increased snow accumulation (Barrett 1982, Pyrah 1987). In contrast, more recent studies documented high survival rates during winter months and attributed the leading cause of annual mortality to hunter kills; however, low snow accumulation and mild winter weather documented during these study periods may have resulted in high survival for these populations (Jacques et al. 2007, Kolar et al. 2012). Over the course of my study I documented 2 hunter kills and no unknown mortalities occurring during the hunting season. Although variables associated with winter severity (temperature or snow depth) were not supported in my mortality risk models, the timing of deaths (11 of 17 pronghorn that died did so in January–March 2011) suggests that winter severity, rather than hunting, was the primary factor limiting pronghorn survival in my study.

Conflicting with my predictions, pronghorn mortality risk was not influenced by snow depth, temperature, or winter severity index. High mortality rates experienced during a winter with increased snow depth (winter 2010–2011) and in previous studies (Barrett 1982, Pyrah

1987), suggest there may be an underlying relationship between weather conditions and mortality risk. Also, time scales other than weekly, monthly, and from the beginning of each winter that I examined may better represent the effect of snow depth and temperature on pronghorn mortality risk. Due to large variations in climatic conditions across years and seasons, examining survival separately by winter may offer more insights into the effects of weather conditions on pronghorn mortality; however, in the case of my study, limited mortality events during winters 2010 (n = 1) and 2011–2012 (n = 3) would have prevented me from adequately evaluating pronghorn mortality risk for these winters.

Preexisting body condition may influence mortality risk for pronghorn on winter range (Dunn and Byers 2008); however, my mortality risk analysis did not include predictors of body condition. Specifically, adult female pronghorn that incurred energetic costs associated with lactation may have been in poorer body condition entering winter than females that did not successfully reproduce. Thus, we would have expected reduced survival on winter range for pronghorn that successfully produced fawns the prior summer. However, fawn surveys conducted prior to winter 2011–2012 showed no difference in winter survival between females with and without fawns on the Dunlap Ranch and a neighboring population of pronghorn (unpublished research). Although reproductive success of individuals may provide insights into mortality risk for other pronghorn populations (Dunn and Byers 2008), we deemed it unnecessary for our particular study based on the results of our fawn survey results; however, environmental condition associated with winter 2011–2012 were particularly mild, which may explain why survival rates did not differ between does with and without fawns. If fawn surveys

particular winter may have resulted in reduced survival rates for pronghorn does that incurred the costs of reproduction within the prior summer.

MANAGEMENT IMPLICATIONS

Understanding how environmental and anthropogenic variables influence pronghorn mortality risk can offer critical insights to formulate appropriate management strategies for populations experiencing harsh environmental conditions and increasing levels of landscape development. Exposure to wind energy did not influence pronghorn survival on winter range. My results suggest wind energy development did not impede pronghorn behavior and winter survival; rather, pronghorn survival on winter range was largely influenced by environmental (average time spent in sagebrush habitat and terrain ruggedness) and non-wind energy anthropogenic (distance to major roads) variables. Mitigating the effects of major roads for pronghorn survival may be difficult; however, maintaining large, continuous sagebrush stands in areas of development may be more realistic management strategies. I suggest wildlife managers prioritize sagebrush conservation on pronghorn winter range, especially in areas being developed for energy resources. In addition, I suggest maintaining corridors to wintering areas utilized during harsh weather events.

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Table 3.1. Predictor variables considered in modeling pronghorn survival risk using Cox proportional hazards regression for pronghorn exposed to wind energy development in south-central Wyoming, USA, 2010–2012.

Variable class	Covariate	Description
Environmental		
Age	Ageclass	Two age categories: pronghorn ≤ 5.5 years old (0) and pronghorn > 5.5 years old (1)
Movement rate	MovRateAvr	Distance moved between sequential locations for individual pronghorn averaged over all winters (meters/hr)
Terrain	VRMAvr	Mean Vector Ruggedness Measure (vector ruggedness index [VRM]; Sappington et al. 2007) scaled to 1000 (VRMx1000) calculates terrain ruggedness within a 3 cell size neighborhood
	SlopeAvr	Mean slope of the landscape derived from a 30 m DEM
Temperature	Temp	Number of observations from the beginning of each respective winter to time <i>t</i> where daily low temperatures were $\leq -12^{\circ}$ C (HPRCC 2012)
	Temp_wk	Number of observations within week prior to time t where daily low temperatures were $<-12^{\circ}C$ (HPRCC 2012)
	Temp_mnth	Number of observations within month prior to time t where daily low temperatures were $<-12^{\circ}C$ (HPRCC 2012)
Snow depth	Snowd	Daily snow depth average, accumulation and standard deviation (suffix: avr, acc, sd) from the beginning of each respective winter to time <i>t</i> (NOHRS 2013; 30m cell size)
	Snowd_wk	Daily snow depth average, accumulation and standard deviation (suffix: avr, acc, sd) from the week prior to time <i>t</i> (NOHRS 2013; 30m cell size)

Variable class	Covariate	Description
	Snowd_mnth	Daily snow depth average, accumulation and standard deviation (suffix: avr, acc, sd) from month prior to time <i>t</i> (NOHRS 2013; 30 m cell size)
Winter Severity Index (WSI)	WSI	Accumulation of WSI points from beginning of respective winter to time <i>t</i> , where 1 point was given when daily low temperatures were $<-12^{\circ}C$ (HPRCC 2012) and 1 additional point was given when daily snow depth was \geq 35 cm for each day
	WSI_wk	Accumulation of WSI points within week prior to time <i>t</i> , where 1 point was given when daily low temperatures were $<-12^{\circ}$ C (HPRCC 2012) and 1 additional point was given when daily snow depth was \geq 35 cm for each day
	WSI_mnth	Accumulation of WSI points within month prior to time <i>t</i> , where 1 point was given when daily low temperatures were $<-12^{\circ}$ C (HPRCC 2012) and 1 additional point was given when daily snow depth was \geq 35cm for each day
Landcover	SagePerc	Percent of total locations for individual pronghorn found within sagebrush, derived from classified land cover data (sagebrush or other; USGS GAP 2011)
Anthropogenic		
Fences	FenceDenAvr	Density of fences (km/km ²), digitized using NAIP imagery (2009)
	FenceDistAvr	Mean Euclidean distance to nearest fence (km; digitized using NAIP imagery [2009])
Major roads	MajorRdAvr	Mean Euclidean distance to nearest major road (U.S. highways, State highways, and Interstate highways) derived from a 30 m DEM
	MajorRd_Decay	Decay distance at 3000 m of mean Euclidean distance to nearest major road (U.S. and State highways and Interstate highways) derived from a 30 m DEM

Variable class	Covariate	Description
Minor roads	MinorRdAvr	Mean Euclidean distance to nearest county road derived from a 30 m DEM
Wind energy		
Percent exposure	ExpAccrd_1km	Percent of total locations for individual pronghorn found within 1 km of access roads associated with wind energy infrastructure
	ExpFac_1km	Percent of total locations for individual pronghorn found within 1 km of facilities (employee and maintenance buildings and substations) associated with wind energy infrastructure
	ExpTurb_1km	Percent of total locations for individual pronghorn found within 1 km of wind turbines associated with wind energy infrastructure
	ExpAccrd_2km	Percent of total locations for individual pronghorn found within 2 km of access roads associated with wind energy infrastructure
	ExpFac_2km	Percent of total locations for individual pronghorn found within 2 km of facilities (employee and maintenance buildings and substations) associated with wind energy infrastructure
	ExpTurb_2km	Percent of total locations for individual pronghorn found within 2 km of wind turbines associated with wind energy infrastructure

Table 3.2. Seasonal mortality and sample size (*n*) for collared pronghorn from the Dunlap Ranch, south-central Wyoming, USA, 2010–2012. Mortalities caused by capture stress or collars that detached early from study animals were not included. Numbers in parentheses are percentages of pronghorn in each sample that died.

Study Area	Season	Year	Mortalities	n
Dunlap Ranch	Winter	2010	1 (3.1)	32
		2010-2011	13 (48.1)	27
		2011-2012	3 (12.5)	24
	Summer	2010	4 (12.9)	31
		2011	3 (23.1)	13

Study Area	Does with fawns	Does without fawns
Dunlap Ranch		
Air	1	6
Ground	1	0
Total	2 (25.0%)	6 (75.0%)
Walcott Junction		
Air	6	6
Ground	5	4

11 (52.4%)

Total

10 (47.6%)

Table 3.3. Collared female pronghorn with and without fawns, 9–10 August 2011, Dunlap Ranch and Walcott Junction, Wyoming, USA.

Table 3.4. Kaplan-Meier product-limit survival estimates (Ŝ) during winter 2011–2012 (October 2011–April 2012) for collared females evaluated for fawn production during 9–11 August 2011 in the Dunlap Ranch (DR) and Walcott Junction (WJ) study areas, south-central Wyoming, USA, 2010–2012.

					Survival	90% CI
Study area	Fawns	п	Mortalities	Survival (Ŝ[t])	Lower	Upper
DR	yes	1	0	1.00	1.00	1.00
DR	no	5	1	0.83	0.58	1.00
WJ	yes	11	1	0.91	0.77	1.00
WJ	no	8	0	1.00	1.00	1.00

Table 3.5. Model fit statistics for environmental, anthropogenic, and wind energy variable classes used to evaluate pronghorn winter mortality risk at the Dunlap Ranch, south-central Wyoming, USA, 2010–2012. Number of parameters in each model (K), log likelihood (LL), Akaike's Information Criteria (AIC), difference in AIC from the top model (Δ AIC), and Akaike's weights (w_i) are also reported.

Variable class model	Κ	LL	AIC	ΔΑΙϹ	Wi	Rank
Environmental (16 candidate models)						
SagePerc + SnowdSD_mnth + VRMAvr	3	-36.09	78.176	0.000	0.502	1
SagePerc + SnowdSD_mnth + Temp_mnth + VRMAvr	4	-35.98	79.954	1.778	0.206	2
$SagePerc + Temp_mnth + VRMAvr$	3	-37.58	81.159	2.982	0.113	3
SnowdSD_mnth + VRMAvr	2	-39.44	82.888	4.712	0.048	4
SagePerc + SnowdSD_mnth	2	-39.99	83.983	5.807	0.028	5
SagePerc + VRMAvr	2	-40.03	84.069	5.893	0.026	6
SnowdSD_mnth + Temp_mnth + VRMAvr	3	-39.19	84.377	6.200	0.023	7
SagePerc + SnowdSD_mnth + Temp_mnth	3	-39.41	84.814	6.638	0.018	8
Null	0	-61.68	123.354	45.178	0.000	16
Anthropogenic (3 candidate models)						
MajorRdAvr	1	-44.04	90.078	0	0.641	1
MajorRd_Decay3000	1	-44.62	91.236	1.158	0.359	2

Variable class model	K	LL	AIC	ΔΑΙΟ	Wi	Rank
Null	0	-61.68	123.354	33.276	0.000	3
Wind Energy (5 candidate models)						
ExpAccrd_1km	1	-47.29	96.572	0	0.281	1
ExpTurb_1km	1	-47.35	96.704	0.131	0.263	2
ExpTurb_2km	1	-47.05	96.941	0.369	0.234	3
ExpAccrd_2km	1	-47.53	97.054	0.482	0.221	4
Null	0	-61.68	123.354	26.782	0.000	5

Table 3.6. Model fit statistics for subsets or combinations of environmental, anthropogenic, and wind energy variable classes predictive of pronghorn winter mortality risk in the Dunlap Ranch wind energy study area, south-central Wyoming, USA, 2010–2012. Predictor variables forming each variable class model appear in parentheses. Number of parameters in each model (*K*), log likelihood (LL), Akaike's Information Criterion (AIC), difference in AIC from the top model (Δ AIC), and Akaike's weights (*w*_i) are also reported.

Class model (predictor variables)	Κ	LL	AIC	ΔΑΙϹ	Wi
Anthropogenic + Environmental 3	4	-34.06	76.124	0.000	0.198
Anthropogenic + Environmental 2	5	-33.22	76.439	0.315	0.169
Anthropogenic + Environmental 1	4	-34.36	76.717	0.593	0.147
Anthropogenic + Environmental 3 + Wind	5	-33.72	77.432	1.308	0.103
Anthropogenic + Environmental 1 + Wind	5	-33.79	77.584	1.460	0.095
Anthropogenic + Environmental 2 + Wind	6	-32.98	77.956	1.832	0.079
Environmental 1 (SagePerc + SnowdSD_mnth + VRMAvr)	3	-36.09	78.176	2.052	0.071
Environmental 1 + Wind	4	-35.32	78.648	2.523	0.056
Environmental 2 (SagePerc + SnowdSD_mnth + Temp_mnth + VRMAvr)	4	-35.98	79.954	3.829	0.029
Environmental 2 + Wind	5	-35.29	80.576	4.451	0.021
Environmental 3 (SagePerc + Temp_mnth + VRMAvr)	3	-37.58	81.159	5.034	0.016
Environmental 3 + Wind	4	-36.60	81.192	5.067	0.016

Class model (predictor variables)	K	LL	AIC	ΔAIC	Wi
Anthropogenic (MajorRdAvr)	1	-44.04	90.078	13.953	0.000
Anthropogenic + Wind	2	-43.53	91.054	14.930	0.000
Wind (ExpAccrd_1km)	1	-47.29	96.572	20.448	0.000
Null	0	-61.68	123.354	47.229	0.000

Table 3.7. Model-averaged parameter estimates for the 95% model subset predictive of pronghorn winter mortality risk in the Dunlap Ranch wind energy study area, south-central Wyoming, USA, 2010–2012.

			Hazard ratio	Hazard ratio	90% CI
Variable	Coefficient	SE	[exp(coefficient)]	Lower	Upper
SagePerc	-0.04	0.01	0.96	0.95	0.99
SnowdSD_mnth ^a	-0.21	0.17	0.81	0.61	1.07
Temp_mnth ^a	-0.04	0.03	0.96	0.91	1.01
VRMAvr	-2.55	1.31	0.08	0.01	0.67
MajorRdAvr	-0.22	0.12	0.80	0.66	0.98
ExpAccrd_1km ^a	0.02	0.02	1.02	0.99	1.05

^aUninformative predictor variable.



Figure 3.1. In addition to Dunlap I, three other wind energy facilities were in production phases in Carbon and Albany Counties, Wyoming, USA near the Dunlap Ranch. The Seven Mile Hill wind energy facility maintained production of 79 (1.5 MW) wind turbines (66 in Seven Mile Hill I and 13 in Seven Mile Hill II), the High Plains and McFadden Ridge wind energy project maintained production of 85 (1.5 MW) wind turbines (66 in High Plains I and 19 in McFadden Ridge I), and the Foote Creek Rim wind energy project maintained production of 183 (600 kW) wind turbines (69 in Foot Creek I, 3 in Foot Creek II, 33 in Foot Creek III, 28 in Foot Creek IV, and 50 in Rock River I) throughout the study period from Jan 2010 to May 2012



Figure 3.2. Monthly distribution of female pronghorn mortalities during the study period (Jan 2010–Apr 2012) in the Dunlap Ranch, south-central, Wyoming, USA.



Figure 3.3. Monthly Kaplan-Meier product-limit survival ($\hat{S} \pm 90\%$ CI) curves for pronghorn captured in the Dunlap Ranch study area, south-central, Wyoming, USA, 2010–2012.



Figure 3.4. Hazard rates for the best fit model (average distance to major roads [km], percent of locations within sagebrush [%], and terrain ruggedness) predictive of doe pronghorn mortality risk in the Dunlap Ranch, south-central, Wyoming, USA. Hazard rates were plotted with 90% confidence intervals. Terrain ruggedness (Vector Ruggedness Measure) was rescaled by 1000 to adjust for small units.

APPENDIX A. Use of an Offsite Reference Area as a Comparison for Pronghorn Response to Wind Energy on the Dunlap Ranch

Studies lacking pretreatment data commonly establish offsite reference areas to use as controls for treatment conditions (Harju et al. 2011, Kirol et al. 2012). Reference sites are often carefully selected to encompass the same environmental characteristics, but exclude the treatment conditions of interest. The Walcott Junction was originally identified as a suitable offsite reference area encompassing similar environmental characteristics to the Dunlap Ranch, but lacked wind or other energy development. I initially planned to use the Walcott Junction pronghorn population as an offsite reference area for the Dunlap Ranch pronghorn population in my resource selection, survival, and mortality risk analyses; however, apparent differences between behaviors of the two study populations and habitat of the study areas observed over the course of my study suggested Walcott Junction was not a suitable control site for the Dunlap Ranch. Thus, I did not use Walcott Junction as a comparison in my analyses. Here, I report results for resource selection and survival analyses for a sample of 35 adult female pronghorn from Walcott Junction population. I also report results for using habitat selection models developed for pronghorn in Walcott Junction to predict habitat selection by pronghorn in Dunlap Ranch. The results from this analysis provide further evidence that data from Walcott Junction should not be used as an offsite reference area for Dunlap Ranch. Rather than being used as a direct comparison for pronghorn in Dunlap Ranch, these results are meant to provide insights for wildlife managers in regards to habitat selection and survival patterns observed for pronghorn in Walcott Junction during the course of my study.

Study Area

The Walcott Junction reference study area occurs within a larger area south of Interstate 80 that includes 330.4 km² of pronghorn crucial winter range delineated by the Wyoming Game and Fish Department (33.3% Bureau of Land Management, 4.3% state of Wyoming, and 62.3% private ownership) and is located 47.5 km southwest of the Dunlap Ranch in Carbon County, Wyoming (Figure A.1). Walcott Junction, like Dunlap Ranch, is classified as desert and semiarid steppe dominated by arid grasslands and sagebrush (BLM 2011*b*). Elevation within Walcott Junction ranges from 1,979 to 2,331 m. I acquired climate data from the High Plains Regional Climate Center weather station located 14.3 km east of the Walcott Junction in Sinclair, Wyoming. Total snow fall for winter 2010 (November 2009–April 2010) was 153.2 cm, for winter 2010–2011 (November 2010–April 2011) was 140.5 cm, and for winter 2011–2012 (November 2010–April 2011) was 92.9 cm. Average minimum and maximum temperatures for winter 2010 (November 2009–April 2010–2011 (November 2010–April 2011) were –11.3°C and 2.6°C, for winter 2010–2011 (November 2010–April 2011) were –11.4°C and 3.7°C (HPRCC 2012).

A crew captured and affixed GPS transmitters to 35 adult female pronghorn in the Walcott Junction study area in January 2010 (see Chapter 2 for capture details) according to protocols approved by the University of Wyoming Institutional Animal Care and Use Committee (01012010) and Wyoming Game and Fish Department (Chapter 33 Permit ID 742).

Survival Estimates

During my study, 10 of 35 (28.6%) females from Walcott Junction died (4 in 2010, 5 in 2011, and 1 in 2012; Figure A.2). Cause of mortality was attributed to hunter kills in 1 case and unknown in 9 cases at the Walcott Junction. I censored locational data in my survival analysis

from 3 pronghorn where mortality was attributed to capture stress in 2010, as mortality occurred within 1 week of capture. The collar of 1 female from the Walcott Junction detached early due to low battery. In addition, I was unable to recover the collars from 3 pronghorn in Walcott Junction. I thus estimated survival of 29 females from the Walcott Junction study area.

Kaplan-Meier (KM) product-limit survival estimates (\hat{S}) at the end of 2010 and 2011 were 0.87 (90% CI: 0.76–0.97) and 0.70 (90% CI: 0.56–0.83), respectively. Survival estimates (\hat{S}) at the end of the study period (April 2012) were 0.66 (90% CI: 0.51–0.80) at Walcott Junction (Figure A.3). Survival (\hat{S}) in the Walcott Junction study area during winter 2010 was 0.97 (90% CI: 0.91–1.00), winter 2010–2011 was 0.96 (90% CI: 0.90–1.00), and winter 2011– 2012 was 0.95 (90% CI: 0.86–1.00).

RSF Validation

For resource selection analyses, reference area models may be applied to treatment (impacted) study areas to predict habitat selection for treatment populations prior to treatment conditions or retrospectively to predict habitat selection if treatments were absent; however, caution has often been suggested when applying models to landscapes beyond the study area they were developed for because environmental conditions and species behavior vary widely across landscapes (Morrison 2008, Fielding and Haworth 1995, Whittingham et al. 2007). In this approach, an overall best fit RSF model, encompassing environmental and anthropogenic variables not associated with wind, would represent pronghorn resource selection in the Dunlap Ranch prior to wind energy development. This approach would assume pronghorn on winter range in the Dunlap Ranch selected for the same suite of environmental and anthropogenic variables (excluding variables associated with wind energy) as pronghorn in Walcott Junction. The value of this comparison would be in confirming that the only difference in resource selection between

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study areas was due to wind energy development. Prior to using the Walcott Junction as a surrogate for pre-treatment data, I checked this assumption by: 1) identifying an overall best RSF model for the Walcott Junction pronghorn on winter range (I followed the same methodology used in Chapter 2 to identify the best fit model) and 2) using a cross fold validation (standardizing rasters for both study areas to control for difference in availability of environmental and anthropogenic variables) to test the ability of the top Walcott Junction RSF model to predict habitat selection of pronghorn within Dunlap Ranch (Boyce et al. 2002). The overall top RSF model identified for Walcott Junction pronghorn illustrated that selection for environmental and anthropogenic variables (excluding variables associated with wind energy) differed between the two study areas (Tables A.1 and A.2). Specifically, pronghorn in Walcott Junction selected for areas with lower slopes, closer to minor roads (maintained county and BLM roads), and closer to fences (Figure A.4). Dunlap Ranch pronghorn selected for lower slopes, lower densities of fences, lower standard deviations of snow depth, and areas closer to wind energy development (see Chapter 2). Excluding wind energy, both pronghorn populations selected for lower slopes, but selection differed in regards to fences (sign change between study areas), snow depth variation (did not influence Walcott Junction pronghorn resource selection), and distance to minor roads (did not influence Dunlap Ranch pronghorn resource selection. When Dunlap Ranch locational data were used to validate the top RSF model from the Walcott Junction within the Dunlap Ranch landscape, the Spearman's rank correlation was moderately high ($r_s = 0.77$). Although the resulting Spearman's rank correlation may have suggested Walcott Junction would act as a suitable surrogate for pretreatment data, the top models for each study area included different environmental variables. These results confirmed that selection for environmental and anthropogenic variables between the Walcott Junction and Dunlap Ranch

study areas was different. Thus, I could not justify using the Walcott Junction as a surrogate for pretreatment data in a resource selection study.

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Table A.1. Model fit statistics for all possible combinations of variables within the top models for each variable class used to evaluate pronghorn resource selection in the Walcott Junction reference area, south-central Wyoming, USA, 2010–2012. Number of parameters in each model (K), log likelihood (LL), Akaike's Information Criteria (AIC), difference in AIC from the top model (ΔAIC), and Akaike's weights (w_i) are also reported.

Model	K	LL	AIC	ΔΑΙϹ	Wi
FenceDist + MinorRd + Slope	4	-16722.50	33453.000	0.000	1.00
MinorRd + Slope	3	-16726.00	33458.000	5.000	1.00
FenceDist + MajorRd + Slope	4	-16757.00	33522.000	69.000	0.00
MajorRd + Slope	3	-16766.00	33538.000	85.000	0.00
FenceDist + Slope	3	-16848.50	33703.000	250.000	0.00
Slope	2	-16864.50	33733.000	280.000	0.00
FenceDist + MinorRd	3	-17161.00	34328.000	875.000	0.00
MinorRd_km	2	-17182.00	34368.000	915.000	0.00
FenceDist + MajorRd	2	-17187.50	34379.000	926.000	0.00
MajorRd_km	2	-17222.50	34449.000	996.000	0.00
FenceDist_km	2	-17399.50	34803.000	1350.000	0.00
Null	1	-17473.00	34948.000	1495.000	0.00

Table A.2. Model-averaged parameter estimates for the top model predictive of pronghorn winter resource selection in the Walcott Junction reference area, Wyoming, USA, 2010–2012. Standard errors (SE) for coefficients (β) were estimated from bootstrapping across individual pronghorn.

Variable class			Coefficient 95% CI	
variable	β	Bootstrap (SE)	Lower	Upper
Environmental				
Slope	-0.147	0.005	-0.157	-0.136
Anthropogenic				
FenceDist	-0.101	0.021	-0.142	-0.061
MinorRd	-0.152	0.009	-0.170	-0.134



Figure A.1. The Walcott Junction reference study area was located 47.5 km southwest of the Dunlap Ranch wind energy facility in Carbon County, Wyoming, USA. The Walcott Junction study area had not been developed for wind or oil and gas resources and included 330.4 km² of pronghorn crucial winter range delineated by the Wyoming Game and Fish Department.



Figure A.2. Monthly distribution of female pronghorn mortalities during the study period (Jan 2010–Apr 2012) in Walcott Junction, south-central Wyoming, USA.



Figure A.3. Monthly Kaplan-Meier product-limit survival ($\hat{S} \pm 90\%$ CI) curves for pronghorn captured in the Walcott Junction reference area, south-central, Wyoming, USA, 2010–2012.



Figure A.4. Relative probability of use for the best fit model (slope [%], distance to nearest fence [km], and distance to minor roads [km]) predictive of adult female pronghorn resource selection in the Walcott Junction, south-central, Wyoming, USA, winters 2010, 2010–2011, and 2011–2012.

APPENDIX B. Pronghorn Movement Behavior in Response to a

Harsh Winter in the Dunlap Ranch

Weather conditions during the second winter of my study (2010–2011) were characterized by increased snow accumulation and lower temperatures (see Chapter 3–Discussion). During this winter, pronghorn at the Dunlap Ranch study area displayed unique movement behavior not observed during the prior or proceeding winter. During winter 2010, individuals remained within and up to 22 km outside of Dunlap Ranch (Figure B.1). During winter 2010–2011, 2 pronghorn moved 75 and 115 km outside of the Dunlap Ranch (1 southwest across Interstate 80 and 1 north to Casper, WY). Most notably, 13 pronghorn moved to winter south and southwest of Seminoe Reservoir (Figure B.2), an area not used during winters 2010 or 2011–2012 (Figure B.3). Previous studies have documented similar behavior where pronghorn utilize new areas during severe winters or severe weather events (Oakley and Riddle 1974, Barrett 1982, O'Gara 2004). Winter 2010–2011 received approximately 2 and 3 times more snow fall than winters 2010 and 2011–2012, respectively (see Chapter 3–Discussion). Movement patterns illustrated by pronghorn during winter 2010–2011 suggests the habitat south and southwest of Seminoe Reservoir may offer relief for Dunlap Ranch pronghorn during winters with increased snow accumulation. Thus, it may benefit pronghorn populations if critical habitats, like those identified south and southwest of Seminoe Reservoir, are managed as priority areas including conservation of sagebrush integrity and connectivity to other winter ranges.

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Figure B.1. Winter 2010 locations (n = 10,468) collected from 32 female pronghorn from the Dunlap Ranch in south-central, Wyoming, USA.



Figure B.2. Winter 2010–2011 locations (n = 7,712) collected from 27 female pronghorn from the Dunlap Ranch in south-central, Wyoming, USA.



Figure B.3. Winter 2011–2012 locations (n = 8,606) collected from 23 female pronghorn from the Dunlap Ranch in south-central, Wyoming, USA.