

**University of Alberta**

Measuring wildlife response to seismic lines to inform land use planning  
decisions in northwest Canada

by

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## **ABSTRACT**

Development of hydrocarbon resources across northwest Canada has spurred economic prosperity but also generated concerns over impacts to biodiversity. To balance these interests, comprehensive land use plans have been used to match targeted management strategies to ecological components deemed valuable by society such as wildlife. I used remote wildlife cameras to measure the response patterns of American marten and black bear to seismic lines, a ubiquitous linear feature in western Canada. Relative to undisturbed forest locations, martens avoid open and wide seismic lines, but not narrow and recovered lines; occupancy at the home range scale also declines with increasing seismic line density. By contrast, black bears use most types of seismic lines relative to forest locations, but habitat use at broad spatial scales is influenced by the amount of available upland forest rather than line density. This research provides information to develop policies capable of meeting intended management objectives.

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## **CHAPTER 1: GENERAL INTRODUCTION**

### **BACKGROUND**

The exploration and production of hydrocarbon resources across boreal Alberta (AB), British Columbia (BC), and the Northwest Territories (NWT) has increased dramatically over the past several decades. Development activity in this region has led to considerable economic growth (GNWT 2007, BC MEMPR 2009, Tertzakian and Bayton 2011), but also to concerns over the impacts on boreal biodiversity (Schneider 2002, Nitschke 2008, Festa-Bianchet et al. 2011). This is particularly true in remote northern jurisdictions of northeast British Columbia (NE BC) and the NWT where economies are defined by the energy sector (GNWT 2007, BC MEMPR 2009), but operations occur in otherwise undeveloped locations (Global Forest Watch 2000) where traditional land uses still occur (DCLUPC 2006, Nitschke 2008, SLUPB 2010, Strimbu and Innes 2011).

Energy development and wildlife conservation are consistently identified as important long term goals in the region (Salmo Consulting et al. 2003, SENES Consultants 2005), but development is poised to expand rapidly buoyed by massive unconventional energy reserves and promising geology (BC MEMPR and NEB 2006, Hamblin 2006). In an attempt to balance economic and ecological sustainability, comprehensive land use planning efforts that aim to identify, prioritize and manage social, economic, and ecological goals using a

series of targeted and properly informed regulatory mechanisms have become more common (SENES Consultants 2005, DCLUPC 2006, Antoniuk, 2009, SLUPB, 2010). In theory, land use planning efforts focus targeted management strategies and regulatory actions around the needs of specific ecosystem services or components of social value (or Valued Ecological Components (VEC); see Beanlands and Duinker, 1983) (Ananda and Herath 2009, Weber et al. 2012), ultimately meeting conservation goals while also facilitating resource extraction (Margules and Pressey 2000, Bettinger et al. 2003, Kato and Ahem 2011). While land use planning efforts across northwest Canada have successfully identified conservation goals (e.g. maintenance of those mammal populations important for sustenance hunting and trapping (Salmo et al., 2003; Salmo et al., 2004, SENES Consultants 2005; DCLUPC, 2006; SLUPB, 2010)), the science required to develop specific management decisions capable of meeting those identified conservation goals is often lacking.

Development of hydrocarbon resources in northern Canada involves the creation of numerous linear disturbances including seismic lines, roads, and pipelines for the exploration, access, and transportation of energy resources, respectively (Schneider 2002). Consequently, regulatory concerns often focus around the management of linear features, either by mandating construction techniques and line widths (Tamarack Solutions 2003, OGC 2011, AANDC 2011) or by limiting linear feature density (Salmo Consulting et al. 2003, Salmo Consulting et al. 2004). A key component of most land use plans are management thresholds (Kennett 2006). This approach aims to limit new

development until past activities have sufficiently recovered in order to prevent undesired impacts to wildlife (DCLUPC 2006, Sorensen et al. 2008). In some instances all linear features ever created are included in threshold calculations while in others certain features are excluded (Salmo Consulting et al. 2003, Salmo Consulting et al. 2004, DCLUPC 2006). To properly inform management strategies for linear features, it is important to understand the impacts of different feature types; a failure to accurately account for inherent differences in linear feature types or their ecological impacts precludes the development of effective management strategies (Kennett 2006).

### **SEISMIC LINES AND CURRENT KNOWLEDGE**

Particularly contentious in threshold and management discussions are the effects of seismic lines, narrow linear features used to explore for and delineate hydrocarbon resources. Although seismic lines are the narrowest, they are by far the most numerous disturbance feature associated with the energy sector (Schneider, 2002; Lee and Boutin, 2006). Seismic-based exploration, typically the first step in the hydrocarbon development cycle, uses energy waves produced from small explosions or vibrations to profile subterranean rock strata (Lee and Boutin, 2006). The imaging and analysis of subsurface strata is achieved by the systematic placement of “source” and “receiver” points, or origin and recording points for energy waves, respectively (Schneider, 2002; Lee and Boutin, 2006). Where the energy sector operates in forested regions of the world, such as in



northwestern Canada, this systematic placement is facilitated using lines (i.e., “seismic lines”) that are cut into the forest.

A detailed understanding of seismic line impacts on wildlife is lacking because most studies to date have lumped all seismic lines into a single disturbance class (James and Stuart-Smith, 2000; Dyer et al., 2001; Dyer et al., 2002; but see Bayne et al., 2005a; Ashenurst and Hannon, 2008). In reality, however, seismic lines exist at a variety of widths and recovery states. Seismic exploration across northwest Canada began in earnest in the 1950s (Morrell et al. 1995), and until the mid-1990s seismic lines were constructed as straight-line features between 6 and 10 m wide using bulldozers (hereafter conventional lines; Schmidt 2004, Lee and Boutin, 2006; Jordaan et al., 2009). Concerns over the impacts of conventional lines to timber supply and wildlife led the energy sector to begin constructing narrower and meandering lines as a series of so called low-impact seismic (LIS) techniques by the late 1990s (Schmidt 2004, Schneider 2002, AECOM 2009). Currently LIS lines range in width to between  $\leq 2$  and 5 m and their use is widespread (Mike Doyle, president of the Canadian Association of Geophysical Contractors, pers. comm., 2010), even mandated in some jurisdictions (OGC 2011, AANDC 2011). Whether narrower seismic lines actually mitigate impacts to wildlife, however, is poorly understood (Weclaw and Hudson 2004, but see Bayne et al. 2005a, Latham et al. 2011).

Seismic lines are considered temporary disturbance features by the energy sector. Although some lines may be converted to other development features such as roads or pipelines (Schneider 2002) and others may be kept open for

recreational use (Lee and Boutin 2006), most lines are assumed to recover naturally after their intended use and abandonment. However, seismic line recovery is inconsistent and highly variable (Jorgenson et al. 2010, Bayne et al. 2011). Some lines recover to heavy shrub and sapling growth over time (Revel 1984, Machtans 2006) while others remain in open or semi-open states for decades (MacFarlane 2003, Lee and Boutin 2006, Kemper and Macdonald 2009). Unfortunately, recovery is not well tied to line age *per se*, but is influenced by surrounding habitat type (Felix et al. 1992, Drawe and Ortega 1996, Lee and Boutin 2006, Ashenhurst and Hannon, 2008), the severity of ground and soil disturbance during line construction (de Grosbois et al. 1991, de Grosbois and Kershaw 1993, Emers et al. 1995), and often by complex interactions between those factors (Emers et al. 1995, Kemper and Macdonald 2009, Jorgenson et al. 2010). Anecdotal evidence suggests that wildlife respond to older or reclaimed seismic lines differently than open ones (Oberge, 2001; Neufeld, 2006), but again this is poorly understood for almost all species.

Past wildlife – seismic line research has focused on species' behavioural responses to individual lines (James and Stuart-Smith 2000, Dyer et al. 2001, Dyer et al. 2002, Ashenhurst and Hannon 2008, but see Bayne et al. 2005b). While important, this approach has limited value in informing specific land use planning strategies when societal values revolve around the conservation of wildlife populations (Richter et al. 1996, Griffin et al. 2007). For example, whether changes in organism behaviour actually translate into population level impacts is uncertain for most species (Caro 1999, Gill et al. 2001). Further,

because the ecological relationships between species are often complex, response of a single species at a single scale may not accurately reflect the ecological consequences of seismic lines. Of greater consequence may be whether seismic lines act to influence expected ecological relationships between species by triggering changes in broad-scale behaviours or habitat use (Levin 1992, Gustafson 1998).

## **THESIS OBJECTIVES**

The objective of this thesis is to fill a knowledge gap around wildlife response to seismic lines. Specifically, my aim is to provide a more detailed understanding of wildlife response to different types of seismic lines and seismic line density to help fill current knowledge gaps around line management with respect to identified wildlife-VECs in northern Canada. All field work for this thesis was conducted in northern British Columbia and Alberta, and in the southern Northwest Territories (Figures 1.1 and 1.2).

In Chapter 2, I measure the behavioural and population responses of the American marten (*Martes americana*) to seismic lines and seismic line density, respectively. Marten are a valued furbearer and a specified VEC in the region (INAC 2007). Although past research shows marten behaviour and populations are sensitive to habitat disturbance from timber harvest (Thompson 1994, Chapin et al. 1998, Potvin et al. 2000, Andruskiw et al. 2008, Godbout and Ouellet 2010), marten response to energy development is unstudied. To assess behavioural

response to seismic lines I compare the probability of use at seismic lines that vary by width and recovery state to use at undisturbed interior forest stands (Fulé et al. 1997, Stoddard et al. 2006, Nielson et al. 2008). To assess population response to line density I compare the probability of home range occupancy across a continuum of seismic line and linear feature density (Gibbs 1998, Schmiegelow and Mönkkönen 2002, Davies and Jackson, 2006). By coupling behavioural and population measurements, I measure whether observed behavioural responses scale to population level impacts. I also compare the efficacy of different definitions of linear feature density in predicting marten home range occupancy.

In Chapter 3, I measure the behavioural response of black bears (*Ursus americanus*) to seismic lines at two spatial scales to understand whether seismic lines influence bear use of lowland forest types important for boreal woodland caribou (*Rangifer tarandus caribou*). Caribou are an important VEC in the region (INAC 2007), and they are listed as a threatened species in Canada (COSEWIC 2002). Plans for their recovery and conservation are ongoing and include aims to prevent increased predation (Environment Canada 2011). Black bears are a recognized caribou predator (Rettie and Messier 1998, Faille et al. 2011, Pinard et al. 2012), and bear use of habitat disturbances associated with natural resource development has been implicated in higher bear use of caribou habitat (Brodeur et al. 2008, Mosnier et al. 2008, Pinard et al. 2012) and higher predation rates of caribou by bears (Faille et al. 2011, Pinard et al. 2012). As in chapter 2, I assess bear behavioural response to seismic lines that differed by width and recovery

state as the probability of line use to that expected at undisturbed interior forest stands. I then measure the likelihood that cumulative seismic line density increases black bear use of lowland forest types at a 5 km<sup>2</sup> scale. By coupling behavioural measurements at two scales, I measure whether seismic lines trigger changes in expected habitat use by black bears and in the expected predator-prey relationship between bears and caribou.

My thesis is organized as two independent manuscripts. Chapter 2 was submitted to *Biological Conservation* and Chapter 3 was submitted to the *Journal of Wildlife Management*; both manuscripts are currently in review. References and section breaks in those chapters match the requirements of specific journals listed. Otherwise, the general formatting of this thesis is consistent with the guidelines set forth by the University of Alberta.

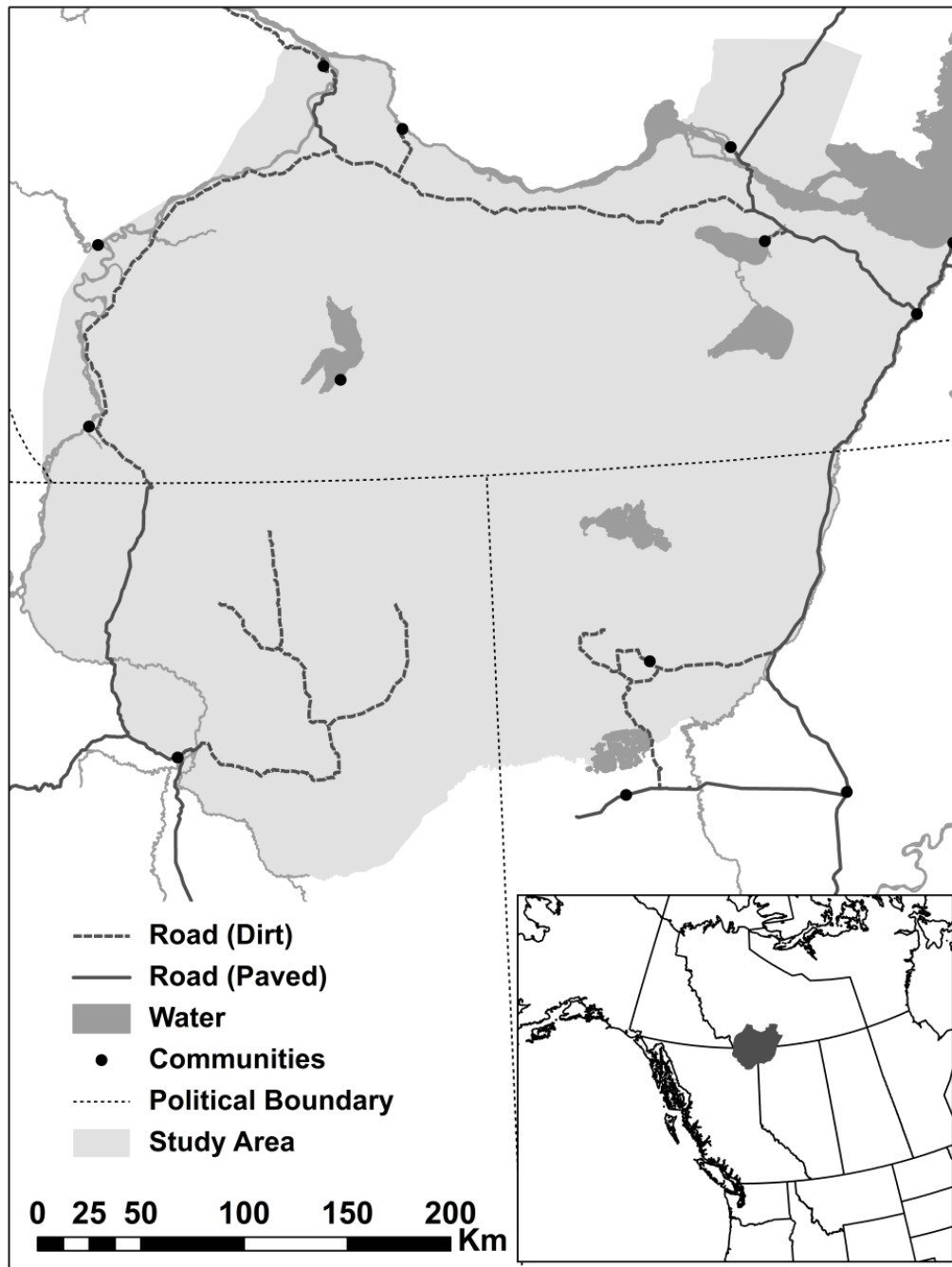


Figure 1.1. Map of the study area in northwest Alberta, northeast British Columbia, and southwest Northwest Territories.

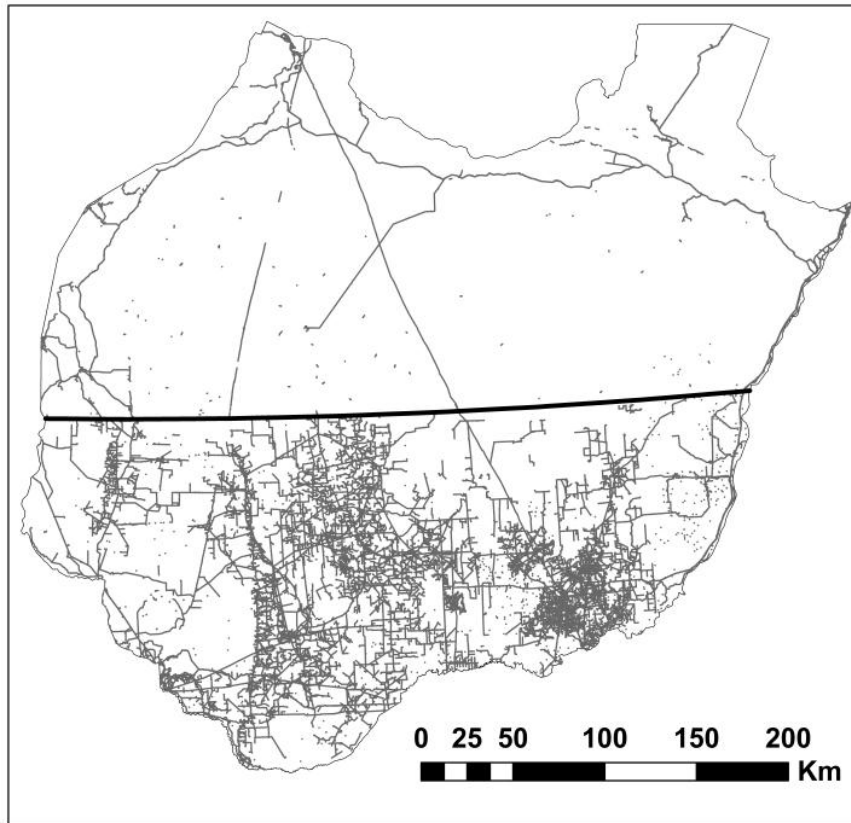


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## **CHAPTER 2: BEHAVIOURAL AND POPULATION RESPONSES OF THE AMERICAN MARTEN (MARTES AMERICANA) TO ENERGY SECTOR DEVELOPMENT IN NORTHERN CANADA**

### **INTRODUCTION**

Across the northwestern boreal forest of Canada, exploration and production of hydrocarbon resources has resulted in significant economic growth over the last several decades (GNWT, 2007; MEMPR, 2010; Tertzakian and Bayton, 2011). However, rapid development has also led to concerns over the impacts of energy sector activity on boreal biodiversity (Schneider, 2002; Nitschke, 2008; Festa-Bianchet et al., 2011). Energy development and wildlife conservation are consistently identified as important long term goals in the region, and a threshold-based management system has been suggested to balance these goals. The threshold approach aims to limit new development until past activities have sufficiently recovered in order to prevent undesired impacts to wildlife (Kennett, 2006; DCLUPC, 2006; Sorensen et al., 2008).

Development of hydrocarbon resources in northern Canada involves the creation of linear disturbances such as seismic lines, roads, and pipelines (Schneider, 2002). Consequently, regulatory thresholds focus around limiting the density of linear disturbances. However, recommended thresholds vary considerably in the region ranging from 0.6 km/km<sup>2</sup> to 2.4 km/km<sup>2</sup> using all linear features ever created, to using all features excluding certain seismic line types, to

using only roads and trails (Salmo Consulting et al., 2003; Salmo Consulting et al., 2004; DCLUPC, 2006). Critics of the threshold approach contend that all linear features are not equal and that a failure to account for these differences will confound threshold calculations and ultimately, management objectives (Kennett 2006). Particularly contentious are seismic lines, which although the narrowest, are by far the most numerous disturbance feature associated with the energy sector (Schneider, 2002; Lee and Boutin, 2006).

A detailed understanding of seismic line impacts on wildlife is lacking because most studies lump all seismic lines into a single disturbance class (James and Stuart-Smith, 2000; Dyer et al., 2001; Dyer et al., 2002; but see Bayne et al., 2005; Ashenurst and Hannon, 2008). Prior to the mid-1990s seismic lines were constructed between 6 and 10 m wide (hereafter conventional lines; Lee and Boutin, 2006; Jordaan et al., 2009). In response to a myriad of concerns, the energy sector began constructing narrower and meandering lines as a series of so called low-impact seismic (LIS) techniques; currently LIS lines range in width to between  $\leq 2$  and 5 m. Whether narrower seismic lines actually mitigate impacts to wildlife is poorly understood (Weclaw and Hudson, 2004; but see Bayne et al., 2005; Latham et al., 2011).

In addition, the energy sector has invested little effort in actively recovering vegetation along seismic lines because lines are considered temporary disturbances that will recover naturally after use. However, natural recovery on conventional seismic lines differs dramatically from recovery following timber harvest (Revel, 1984). On lines, recovery is highly variable (Bayne et al., 2011);

while some lines recover to heavy shrub or sapling growth over time (Revel, 1984; Machtans, 2006), others remain in open or semi-open states for decades (MacFarlane, 2003; Lee and Boutin, 2006; Kemper and Macdonald, 2009). Anecdotal evidence suggests that wildlife respond to older or reclaimed seismic lines differently than open ones (Oberg, 2001; Neufeld, 2006), but again this is poorly understood for almost all species.

Behavioural responses of wildlife to anthropogenic habitat disturbances play an important role in the development of management actions (Caro, 1999; Gill and Sutherland, 2000). To date, a number of studies on wildlife response to energy development have interpreted a “statistically significant” change in organism behaviour as inherently negative (Dyer et al., 2001; Sawyer et al., 2006; Habib et al., 2007; Doherty et al., 2008), and such interpretations have played a crucial role in the develop of linear feature thresholds in boreal Canada. However, whether behavioural changes actually translate into population level impacts is uncertain for most species (Caro, 1999; Gill et al., 2001a). Without a clear link between scales, it is uncertain whether a threshold system can achieve its intended management objectives (Sutherland, 1998; Gill et al., 2001b; Griffin et al., 2007).

Our goal was to measure the behavioral and population responses of the American marten (*Martes americana*), a valued furbearer in the region (INAC, 2007), to seismic lines. While behavioural and population impacts of timber harvest are well documented for marten across North America, (Thompson, 1994; Chapin et al., 1998; Potvin et al., 2000; Andruskiw et al., 2008; Godbout and

Ouellet, 2010), marten response to energy development is unstudied. First, we used remote cameras to compare marten behaviour between seismic lines of different widths and states of recovery to undisturbed forest interior locations. Next, we evaluated whether home range occupancy was influenced by seismic line density to determine if observed behavioral responses scaled to a measurable population level impact. Finally, we evaluated whether seismic line density is the best metric for setting linear feature thresholds by comparing predicted home range occupancy relative to different definitions of linear feature density (i.e. seismic line versus road versus total linear feature density).

## **METHODS**

### **Study Area**

We measured marten response to seismic line type and density across 200,000 km<sup>2</sup> of northern boreal forest in northwest Alberta (AB), northeast British Columbia (BC), and southwest Northwest Territories (NWT) between 61°48' and 58°48' latitude and 122°41' and 117°39' longitude in 2008 and 2009. Human density was extremely low, averaging < 1 person/km<sup>2</sup> (Weiss et al., 2008). Industrial land use was almost entirely limited to the energy sector, and was widespread south of the 60<sup>th</sup> parallel (in AB and BC), but uncommon to the north. Forestry occurs in parts of AB and BC, but we excluded these areas from our study to avoid confounding effects. Trapping of marten is economically

important throughout the region, and mainly occurs around communities where there is little or no energy development (Poole, 1991; Nitschke, 2008).

The study area was characterized by a cold continental climate (Downing and Pettapiece, 2006). Topography was flat or slightly undulating. Extensive lowland forests and peatland complexes occurred on flat poorly drained sites with organic soils. Upland forests were found on sloped or undulating terrain with well drained mineral soils. Lowlands contained numerous small water bodies and were dominated by black spruce (*Picea mariana*), sometimes mixing with trembling aspen (*Populus tremuloides*) or tamarack (*Larix laricina*). Understory was sparse, comprised of willows (*Salix* spp.), dwarf birch (*Betula glandulosa*), and Labrador teas (*Ledum* spp.). Uplands were dominated by mixed stands of white spruce (*Picea glauca*) and trembling aspen or balsam poplar (*Populus balsamifera*), but pure conifer or deciduous stands did occur. Understory was dense and mainly low-bush cranberry (*Viburnum edule*), green alder (*Alnus viridis*), red osier dogwood (*Cornus sericea*), and rose (*Rosa* spp.). Jack pine (*Pinus banksiana*) stands occurred sporadically on xeric sites and eskers.

### **Forest Type and Disturbance Footprint**

Forest type was obtained from Earth Observation for Sustainable Development (EOSD), a 25 m resolution raster based land cover classification (Wulder et al., 2003). In a Geographic Information System (GIS; ArcGIS 9.3, ESRI, Redlands, California), we reclassified these data into upland and lowland forests, shrub dominated and naturally open habitats, water bodies, and

anthropogenic disturbances. Where cloud cover, shadow, or missing data existed we mosaiced land cover from a Ducks Unlimited Canada Earth Cover Classification where available in the western NWT, and MODIS data elsewhere reclassified to the above specifications.

We compiled energy sector footprint data from a number of sources to generate individual layers for seismic lines, roads, pipelines, well-pads, and oil field facilities. In the NWT, hand-digitized data from Indian Remote Sensing satellite imagery (5 m resolution) were provided by the Dehcho First Nation, and were supplemented with operations records from the National Energy Board of Canada and by digitizing Google Earth imagery. In British Columbia, relic (pre-1997) seismic lines hand-digitized from air photos (1:20,000 scale) were provided by the British Columbia Ministry of Environment (TRIM data; MoE, 1997); all other footprint data were from operations records obtained from the British Columbia Oil and Gas Commission. All Alberta data were compiled and provided by IHS Energy Services. Because footprint features often evolve from one feature type to another (e.g. seismic lines are converted to roads or pipelines; Schneider, 2002), we deleted or spatially adjusted overlapping features from different datasets as required to maintain footprint accuracy at the time of all field sampling. Once updated, we calculated feature type densities using a roving window approach in a GIS.



## Site Selection

To isolate marten response to seismic lines and cumulative line density, we controlled for dominant forest type and cumulative habitat disturbance *a priori* using a GIS. Within a 5 km<sup>2</sup> roving window, (equates to the average marten home range in our region (Buskirk and MacDonald, 1989; Latour et al., 1992; Powell, 1994; Poole et al., 2004)), we stratified the study area into upland and lowland forest types and measured cumulative seismic line density as a continuous variable. Cumulative seismic line density was used as a proxy of total habitat disturbance because it was highly correlated to both total linear feature density and total footprint density (using 10,000 randomly generated points across the study area, pairwise correlation coefficients were 0.982 and 0.892, respectively). We randomly generated a set of candidate sites across a continuum of cumulative seismic line density ranging from < 0.1 km/km<sup>2</sup> to >26 km/km<sup>2</sup> within each forest type. Candidate sites were constrained to be  $\geq 5$  km from one another to maintain independence between sampled home ranges, but within 15 km of tracked or water access so they could be reached. In the field, if selected sites were inaccessible or incorrectly categorized, we chose the next closest site until suitable.

## Camera Trap Protocol

Around each selected site we established a cluster of remote cameras in a nested design to measure marten behavioral response and occupancy simultaneously (Figure 2.1) (University of Alberta Animal Care Protocol No. 476705,

Government of the Northwest Territories Wildlife Research Permit No. WL-005752). Marten behaviour was measured at the camera scale and population was measured at the cluster scale. All cameras were baited with 150 g of canned dog food and 50 g of tinned sardines packed in water once at camera set-up, programmed to collect data 24 hours/day, and retrieved after 10 trap nights. Cameras were paired on and off seismic lines for ease of deployment in the field; within the pair, cameras were spaced by 450 m to maintain independence between sampling locations. Interior cameras were set as far from disturbances as possible, (ranging from 450 m - < 50 m). Even in high disturbance landscapes, interior cameras were always set away from major footprint features such as roads, pipelines, or well-pads. To form a cluster, three camera pairs were grouped together and spaced by 900 m along a single seismic line. This design was chosen in concordance with the average marten home range in our region (5 km<sup>2</sup>) and the requisite survey effort to sample effectively for home range occupancy (Raphel, 1994). Sampling independence between clusters was maintained via site selection as described above.

## **Data Collection**

### ***Behavioural response***

Behavioural response was evaluated as the probability of seismic line use relative to the forest interior by pairing cameras on and off lines. Line cameras were pointed along a seismic line at one of the following treatments: 1) line  $\geq$  6 m wide and open (hereafter open conventional); 2) line  $\geq$  6 m wide and partially

recovered (partial conventional); 3) line  $\geq 6$  m and recovered (closed conventional); 4) line open and  $\leq 2$  m in width (open 2m); 5) line open and 3-4 m wide (open 3-4m); and 6) line open and 5 m wide (open 5m) (Figure 2.2). These treatments explicitly measured marten response to conventional seismic lines at varying recovery states (treatments 1-3) and to typical LIS construction widths (treatments 4-6). Line recovery was quantitatively assessed in the field using vegetation structure and composition attributes (Table 2.1) (see also Bayne et al. 2011 for full protocol description); attribute values are summarized in Table 2.2. LIS line widths were related to construction techniques: lines  $\leq 2$  m wide were constructed by hand, lines 3 – 4 m wide were cut with small tractors fitted with mulching blades, and lines 5 m wide were cut with small bulldozers.

At each camera location we classified the surrounding forest as upland deciduous, upland mixed wood, upland conifer, lowland conifer, or shrub dominated; and we estimated age on a rank scale from 1 to 4 corresponding to stand initiation, stem exclusion, mature forest, and old-growth forest, respectively (Chen and Popadiouk, 2002). Cameras were not set in open wetlands or in forested areas with  $< 10\%$  canopy cover where possible, or in recent burns. We also searched for evidence of trapping activity (actual traps or trap boxes, cabins, or tracked access) along sampled seismic lines, and within and near to camera clusters.

### ***Occupancy***

The population response of marten was evaluated as home range occupancy relative to linear feature density at the cluster scale. For our purposes occupancy was met with a single detection at any camera station within the cluster (at least 1 of 6 cameras within 60 trap nights) (Raphel, 1994). Previous work on marten shows high survey effort, such as this, provides consistent and high detection probabilities (Hargis et al., 1999; Slauson et al., 2007; Baldwin and Bender, 2008; Zielinski et al., 2008; Moriarty et al., 2011) and is sufficient to directly compare occupied and non-occupied home ranges assuming non-detections accurately reflected true absence (Zielinski and Stauffer, 1996; Kirk and Zielinski, 2007). We sampled home ranges evenly across forest type and geographic location in both years.

Because recommended threshold density metrics in the region vary considerably, we compared the efficacy of the following common metrics to predict marten home range occupancy: 1) cumulative seismic line density; 2) cumulative seismic line density excluding narrow seismic lines (corrected seismic line density); 3) non-seismic linear feature density; 4) road density, 5) total linear feature density; and 6) total linear feature density excluding narrow seismic lines (corrected total line density). It was not possible to calculate a series of density models excluding “recovered” seismic lines because recovery is not well correlated with line age in our region (Bayne et al., 2011), nor are remotely sensed products available to differentiate recovery states along lines. Narrow seismic lines were removed from corrected estimates based on metadata,

measurements in the field, Light Detection and Ranging (LiDAR; Lefsy et al., 2002) imagery, and inference from operational parameters.

## **Statistical Analysis**

### ***Behavioural response***

We assessed marten use (marten photographed at a camera = 1 vs. not photographed = 0) of seismic lines using a population averaged generalized estimating equation (GEE) with a logit link and binomial error family in Stata 11.1 IC (Stata-Corp, College Station, Texas). A modification of generalized linear models, GEE's account for potential correlation within panels of hierarchically structured data, in this case individual cameras nested within clusters (Hardin and Hilbe, 2003). We assumed an exchangeable correlation structure to our data meaning correlation of marten detections between cameras within a cluster was constant (Fieberg et al., 2010), but used a semi-robust estimator of variance to generate standard errors robust to potential misspecification (StataCorp, 2009). Because the failure to distinguish between unavailable and unused points could induce error in our estimate of marten response to different seismic line treatments (Aarts et al., 2008; Beyer et al., 2010), we evaluated used – unused data only from those clusters where at least one marten was photographed (i.e., occupied clusters) for this analysis. We assumed all cameras within occupied clusters were equally available to the sampled marten population so use accurately reflected a behavioral decision. Some cameras failed before the complete 10-night sampling period; if failure

occurred at set up (< 24 hours), the camera was removed from analysis. For remaining cameras, we calculated the natural log of total sampling duration (in minutes), constrained the coefficients to 1, and modeled this adjusted weight to control for the actual amount of opportunity each camera had to detect a marten. We assumed the likelihood of detection at a given camera remained constant over the sample period.

We included seismic line type, forest type and age at each camera, occurrence of trapping within the camera cluster, the number of other species photographed and total number of photographs capturing another species at each camera, and Julian date at camera set up in the global model. The number of other species and pictures recorded were included because we thought loss of bait to non-target species may have reduced camera attractiveness to marten. We also tested for an interaction between line type and the cumulative density of seismic lines, the cumulative disturbance footprint, and forest type at the cluster level to measure whether response to seismic line types was constant or was influenced by those surrounding variables. We removed non-significant variables using a backward stepwise procedure, and retained all variables significant at  $P < 0.15$  in the final model (Hosmer and Lemeshow, 2000).

Behavioural response is reported as an odds-ratio using the forest interior as the reference condition to show the directional change and the magnitude of difference in use between seismic line types from expected use in the forest interior (Nielson et al., 2007). Line treatments with an odds-ratio  $> 1$  were used more frequently than the interiors, while those with an odds-ratio  $< 1$  were used

less frequently (Larsen et al., 2000); when the 95% confidence interval included 1, any difference in use was not statistically significant (Hosmer and Lemeshow, 2000).

### ***Occupancy***

We assessed marten occupancy (photograph of marten in a home range = 1 vs. no photograph of marten = 0) relative to a range of linear feature densities using the same GEE procedure and parameters described above. By design we sampled home ranges across a continuum from low to high line density, however, disturbance was clustered such that it covaried with political jurisdiction; within each jurisdiction we sampled across the available continuum of line densities. Further, although upland and lowland forests occurred throughout the study area, we encountered some structural and compositional variation induced by latitude- and longitude-related climatic differences. To control for the potential influence of geographic location and variation in habitat types in occupancy analyses, we nested home ranges within ecological districts (i.e., ecological districts were panels and home range clusters replicates within those panels). An ecological district is a Canada-wide land cover classification describing average forest stand composition and productivity based on biotic and abiotic attributes including soil, geology, climate, and vegetation communities (ESWG, 1996). We assumed constant correlation between home range occupancy within each ecological district.

In all models we controlled for Julian date at camera set up, as well as the proportion of lowland forest within the home range. Elsewhere across the distribution of marten, home range selection is influenced by the relative proportion of forest types (Kirk and Zielinski, 2007, Baldwin and Bender, 2008). Univariate analyses of our data showed marten used lowland forests less than upland forests at the home range scale (lowland:  $\beta = -0.487$ ,  $P < 0.001$ , upland:  $\beta = 0.361$ ,  $P = 0.002$ ). Home ranges of mostly upland and mostly lowland occurred within each ecological district. Initially we tested for a significant influence of unequal sample duration (due to random camera failure) across clusters as the total number of sample nights and the number of fully functional cameras per cluster. Neither variables were significant predictors of occupancy ( $\beta = -0.018$ ,  $P = 0.111$ ; and  $\beta = -0.088$ ,  $P = 0.539$ , respectively) so were not included in final models. Instead sample period was controlled for as the natural log of total sampling minutes per cluster.

To compare the efficacy of each calculated line density metric to predict occupancy, we substituted density calculations across different models and compared model fit using a quasiliikelihood information criterion (QIC; Pan, 2001). QIC explicitly accounts for the GEE model link function and correlation structure when comparing variance rather than assuming data independence as do other information criteria (Burnham and Anderson, 2002).



## RESULTS

Cameras were deployed at 1035 unique locations across 173 clusters between May and September in 2008 and May and October in 2009. Sixty-three cameras sampled < 24 hours (failed) and were removed from analyses; camera failure occurred randomly across line treatments and political jurisdiction. Twenty-two camera clusters were established differently than described above in order to attain sufficient line treatment replication; these non-home range clusters were not used in occupancy analyses, but were used to group camera availability for behavioral analyses. Marten were detected at 143 out of 972 remaining locations, and in 66 of 151 home range clusters (75 of 173 total clusters).

### *Behavioural response*

A total of 423 unique camera locations across the 75 total clusters with marten detections were used in this analysis (Table 2.3). Marten response to seismic lines was significantly affected by the type of line encountered. Open seismic lines  $\geq 3$  m in width were used up to 90% less often than forest interiors (i.e., avoided). However, use of open lines  $\leq 2$  m wide and conventional lines supporting at least some regeneration of woody vegetation showed no difference in use relative to forest interiors (Table 2.4). Although not significant, use at closed lines was almost 50% greater than in interiors. The rank age of forest stands also had a significant effect on marten use (odds ratio = 1.473,  $P = 0.001$ ); marten were more likely to use older stands and this is well documented in the literature. The number of other species photographed at a camera was retained in

the model as well (odds ratio = 0.749,  $P = 0.126$ ), but did not change our conclusions about observed marten behavior. Visitation by other species likely did not result in the complete loss of attractant (i.e., scent or bait); Grey Jay (*Perisoreus canadensis*) and red squirrels (*Tamiasciurus hudsonicus*) were detected at 152 and 177 cameras evenly across habitat types, but rarely did either species consume all bait. Black bears (*Ursus americanus*) were detected at 225 cameras and while they typically did consume all or most bait, bears were highly selective of upland stands while marten were ubiquitous in upland and lowland forests. No interactions were significant, suggesting marten response to a given line type is constant relative to surrounding habitat and anthropogenic disturbance.

### ***Occupancy***

Predicted marten occupancy fell from almost 60% where seismic line density was low within a sampled home range to less than 10% in home ranges with the highest density of seismic lines. Both cumulative and corrected seismic line, and cumulative and corrected total linear feature densities significantly predicted home range occupancy (cumulative seismic:  $\beta = -0.083$ ,  $P = 0.005$ ; corrected seismic:  $\beta = -0.116$ ,  $P = 0.003$ ; cumulative total:  $\beta = -0.073$ ;  $P = 0.008$ ; corrected total:  $\beta = -0.100$ ,  $P = 0.003$ ) (Table 2.5, Figure 2.3). Julian date was not significant in any models; the proportion of lowland forest was significant in the corrected models (corrected seismic line density:  $\beta = -0.575$ ,  $P = 0.036$ ; corrected total line density:  $\beta = -0.544$ ,  $P = 0.042$ ), but not in those with a cumulative

estimate of density. Non-seismic linear feature density was not a significant predictor of home range occupancy ( $P = 0.519$ ), nor was road density ( $P = 0.155$ ). The corrected seismic line density estimate best fits our data, but  $\Delta QIC$  between that and corrected total linear feature density is  $< 2$  suggesting either are plausible predictors of occupancy (Burnham and Anderson, 2002). Where seismic lines  $\leq 2$  m wide were removed, average seismic line density fell from 9.436 km/km<sup>2</sup> to 6.531 km/km<sup>2</sup>, and total linear feature density fell from 10.031 km/km<sup>2</sup> to 7.369 km/km<sup>2</sup>. Within group (ecological district) correlation of model residuals for the corrected seismic line model were extremely low (0.006) suggesting any spatial autocorrelation between occupied home ranges was accounted for in the model. Correlation among corrected total linear feature density residuals was similarly low (0.007).

The mean rate of home range occupancy across the study area was 0.437. However, that differed significantly across jurisdictions. In the NWT, where total footprint is minimal, occupancy was 0.571; in BC where footprint is high but recently constructed using numerous industry best practices, it was 0.436; and in Alberta where footprint is high and development has been continuous for more than 5 decades, it was 0.115. Relative to the NWT, occupancy was not statistically different in BC (OR = 0.754,  $\beta = -0.282$ ,  $P = 0.472$ ), but was in Alberta (OR = 0.282,  $\beta = -2.351$ ,  $P = 0.002$ ). Within jurisdiction correlation between model residuals was -0.013.

## **DISCUSSION**

Marten behavioural response to seismic lines is strongly influenced by line width and recovery. While martens avoided open seismic lines  $\geq 3$  m wide (i.e., both conventional and some types of LIS) compared to forest interiors, their use of open lines  $\leq 2$  m wide and conventional lines with at least partial recovery of woody vegetation was similar to their use of forest interiors. Previous work on martens shows strong behavioural responses to fine scale habitat disturbances (Poole et al., 2004; Godbout and Ouellet, 2010). Although openings and structural simple stands are avoided, given sufficient overhead and lateral cover, recovering disturbances are readily reused (Chapin et al., 1997; Payer and Harrison, 2000; Poole et al., 2004). A similar response pattern seems to have occurred in our system as well with the specific attributes of different line types influencing marten behaviour.

Three hypotheses may explain the behavioural response patterns observed: incidence of trapping, prey availability, or a perceived or realized risk of predation. Trapping occurs extensively along open seismic lines, and trappers often exhaust martens along a given line segment before moving traps to a new location. However, marten response to line type was tested only within home ranges where martens occurred, and although evidence of trapping was detected both along open lines and within home range clusters where martens were detected, it was not a significant predictor of line avoidance in our data. Further, our study was conducted outside of the trapping season. For trapping to explain the observed behavioural pattern would require martens to comprehend the risk of

trapping along lines and to have some knowledge of specific trapping locations; this seems implausible.

For prey availability to explain the avoidance of seismic lines, total prey density on open lines would need to be dramatically reduced relative to the forest interior. Open linear features in the Northwest Territories trigger a compositional shift in prey communities, but provide approximately equal total prey density to adjacent forested stands (Darling, 2009). Further, marten diet is highly plastic (Douglass et al., 1983; Ben-David et al., 1997; Cumberland et al., 2001) and prey switching is common especially in northern latitudes where vole, mice, and hare populations fluctuate (Poole and Graf, 1996; Simon et al., 1999). While prey availability likely does not explain open line avoidance by marten, it may explain increased use on recovered lines. Recovered lines have higher stem density and lateral cover at and near the ground compared to forest interior locations (Table 2.2), habitat features linked to *Microtus* spp. and snowshoe hare use and abundance (Litvaitis et al., 1985; Vanderwel et al., 2010; Hodson et al., 2011). Near ground structural complexity may also give marten a competitive advantage in obtaining prey items (Potvin et al., 2000). However, high use of recovered lines may be a function of reduced predation risk.

Marten are killed by a variety of mammalian and avian predators (Bull and Heater, 2000; McCann et al., 2010), and the rate of predation is highest in open habitats (Thompson and Colgan, 1994; Ruggiero et al., 1994). Both empirical and experimental evidence show marten will not travel far from cover despite high prey availability in open areas (Hargis et al., 1999; Andruskiw et al., 2008),

indicating some perception of predation risk. We suspect that open lines  $\geq 3$  m are sufficiently wide to result in a perceived and or realized predation risk. This is somewhat surprising given the widest previously reported openings martens actively avoided were 50 m canopy gaps (Hargis and McCullough, 1984), and suggests martens may be more sensitive to openings than previously thought. In addition to increased near-ground structural complexity, recovering and recovered conventional seismic lines also provide increased overhead and lateral cover (Table 2.2), likely reducing a perceived or actual predation risk along lines (Thompson, 1994) and triggering their renewed use.

By comparing observed behavioural changes in use along seismic lines to a forest interior reference category, we show how regulatory guidelines and management considerations for seismic lines can be informed using an ecologically-derived recovery metric (Nielsen et al., 2007). Our findings are in stark contrast to current guidelines that suggest any LIS lines are capable of mitigating behavioural impacts for wildlife, or that seismic lines constitute permanent disturbance features. Based on our study, martens make no distinction between 3 m and  $\geq 6$  m wide lines when open; for LIS line types to meet management expectations for martens they must be  $\leq 2$  m in width. However, the specific width required to mitigate impacts for other species may be different, and is likely related to the ecology of the species in question (Bayne et al., 2005; Latham et al., 2011).

We also show that use of lines by martens increases as the amount of woody vegetation on lines also increases (Table 2.2). With respect to martens, line

recovery occurs at levels well below what would be considered recovered from a forestry perspective. Unfortunately, as has been shown elsewhere, line recovery in our study area is inconsistent and poorly tied to line age (Bayne et al., 2011). Instead, recovery appears to be a function of a combination of factors including disturbance intensity at construction, surrounding forest type, and continued recreational use of lines (Lee and Boutin, 2006; Kemper and Macdonald, 2009; Jorgenson et al., 2010; Bayne et al., 2011). Our results provide an ecological context for determining which lines should be viewed as recovered. The use of new remote sensing applications such as LiDAR may be an important tool for documenting line recovery remotely given the lack of other remote sensing products or the predictive success of other measurable variables.

The predicted probability of home range occupancy for marten declined precipitously with increasing seismic line density. Thus, the behavioural tendency for individual marten to avoid seismic lines does seem to scale up to the population level. In our best model, corrected seismic line density where lines  $\leq 2$  m wide were removed from density calculations, the predicted probability of home range occupancy in the least impacted home range (0.58 km/km<sup>2</sup> line density) was 0.719 while in the most impacted (24.8 km/km<sup>2</sup>) was 0.048, representing a 93% reduction in occupancy. Moriarty et al. (2011) showed a similarly dramatic decline in marten occupancy relative to forest harvest over time; with a 39% increase in harvest over 28 years, occupancy fell from 65% to 4% of survey locations. A number of other studies have also linked declines in marten occupancy to increases in timber harvest (Chapin et al., 1998; Hargis et

al., 1999; Potvin et al., 2000), although those studies report clear threshold responses (e.g. a sudden and dramatic decline) beyond 25% to 49% harvest in the study landscapes.

Our results are surprising for two reasons. First, the total disturbance footprint in our system is relatively small compared to forestry operations; only ~15 % of the total land base within the most heavily disturbed home range was directly disturbed. Seemingly disproportionate changes in occupancy relative to habitat disturbance may be indicative of a functional habitat loss surrounding line features as is suspected for caribou (Dyer et al., 2001). However, we feel this is unlikely as marten use clearcut edges for travel and hunting at least proportionally to availability elsewhere (Chapin et al., 1998; Cushman et al., 2011; Vigeant-Langlois and Desrochers, 2011). Second, our data show no indication of a non-linear response related to increasing habitat disturbance; instead occupancy declines steadily, almost linearly, with increasing line density. With predicted occupancy at less than 10% in high line density home ranges, it is hard to imagine a further threshold response in occupancy with additional development in our system short of extirpation.

Occupancy is a crude measure of a population's viability relative to increasing energy development because it provides no demographic information. However, occupancy has been clearly linked to population size in marten (Soutiere, 1979; Payer, 1999; Smith et al., 2007; Moriarty et al., 2011), as have changes in occupancy been linked to demographic shifts in marten populations favoring younger and less productive individuals (Thompson, 1994). Regardless,



because marten are territorial (Powell, 1994), reduced occupancy translates to fewer individual marten in a given system. Thus, by measuring home range occupancy along a continuum of linear feature density, we clearly show that increasing line density translates to fewer marten, and we present a robust prediction of home range occupancy at discrete points along that continuum (Fulé et al., 1997; Schmiegelow and Mönkkönen, 2002; Schneider et al., 2003).

Several linear feature density metrics predicted declines in occupancy similarly, however, QIC scores showed a shift in model accuracy between cumulative and corrected definitions of line density (i.e., the exclusion of seismic lines  $\leq 2$  m wide based on behavioural analyses) (Burnham and Anderson, 2002). Although fit was improved, improvement was not as substantial as expected given the clear behavioural responses to seismic line types. Likely, the removal of recovered lines would further improve model fit. It is also possible that line density was sufficiently high to affect occupancy regardless of how density metrics are informed. Narrow LIS lines are typically used to explore for unconventional resources (i.e. shale formations) (Cartwright and Huuse, 2005; AECOM, 2009), and although energy development has occurred in northern Canada for decades, development of unconventional resources is recent. In home ranges containing LIS lines  $\leq 2$  m wide the average change between cumulative line density and reduced line density was only 2.73 km/km<sup>2</sup> (9.439 km/km<sup>2</sup> and 6.709 km/km<sup>2</sup>, respectively). Current development in far northern Canada revolves around shale resources and presents several opportunities to test the impacts of high narrow line densities in isolation.

A complete understanding of the cumulative impact of linear disturbances to marten at a population scale is predicated on a thorough understanding of behavioural responses to individual disturbance types. Although behavioural responses alone may not be capable of developing robust management strategies, robust management strategies cannot be developed without incorporating behavioural metrics. With respect to management thresholds for linear features in northern Canada, our data suggest a proper linear feature density metric is likely one that includes only certain types of seismic lines rather than only roads or all linear features ever created. In isolating seismic lines from other disturbances in this study, we were capable of identifying many home range locations where road density was low or roads were completely absent, but where seismic density was high. If threshold metrics were based on road density alone, management would fail to prevent the significant impacts related to seismic lines. Conversely, we also show that a metric based on any line ever created will artificially inflate threshold calculations thereby limiting development before an undesirable reduction to a marten population actually occurs.

Our intention with this research is not to suggest that seismic lines are the most influential component of an energy sector disturbance footprint. On the contrary, access roads, pipelines, and well pads have all been shown to alter species behaviours in different ways. However, just as a statistically significant change in organism behaviour is not an inherently negative impact, attributing declines in species populations to a cumulative disturbance footprint (Dyer et al., 2002; Walker et al., 2007; Sorensen et al., 2008; Harjou et al., 2010; Gilbert and

Chalfoun, 2011; Strimbu and Innes, 2011) may overemphasize the impacts of certain disturbance features thereby precluding efficient management strategies. Our findings clearly show that all seismic lines do not constitute equal disturbances and may warrant different management considerations. These findings are mirrored in other, similarly in depth studies of species response to disturbance features that vary by attribute (Habib et al, 2007; Bayne et al., 2008; Sawyer et al., 2009; Francis et al., 2011; Wasser et al., 2011).

From a management perspective, there is a critical distinction between a mapped footprint and an ecologically relevant one. The application of an effective management threshold depends in part on accurately making that distinction, and in part on setting an appropriate density limit. Herein rests the true utility of these data from a land use planning perspective. If a single value, in this case either the full development of energy resources or the total preservation of the marten population, were the long term societal goal in a region, the formulation of a land use plan would be easy: either development at the expense of marten, or conservation at the expense of development. However, when the goals of a region are more nuanced, the decision making process requires weighting choices against likely outcomes. This research provides the ecological basis required to contextualize the cause and effect relationships between the competing social, economic, and ecological goals that land use planning seeks to balance. By plotting the relationship between the probability of marten occupancy and corrected seismic line density (as in Figure 2.3), the shape of that

relationship provides a strong ecological rationale for setting appropriate threshold limits on the appropriate density metric.

Table 2.1. Vegetation attributes measured to categorize the recovery of conventional seismic lines in northwest Canada and attribute relevance to marten behaviour.

<b>Vegetation Attribute</b>	<b>Description</b>	<b>Measure</b>	<b>Metric</b>	<b>Relevance for marten</b>
<b>Coarse Woody Debris</b>	Dead wood (if > 50 cm in length and $\geq$ 8 cm in width) at widest point, on or above the ground leaning > 45°	Counted pieces intersecting line transect (22.6 m)	Count of pieces and average width at widest point	Cover
<b>Horizontal Cover</b>	Visual obstruction along line of sight	Visually ranked obstruction using cover board for 5 height increments (0-0.5m, 0.5-1m, 1-1.5m, 1.5-2m, 2-3m) at 10m	Rank of visual obstruction per height increment	Cover
<b>Shrub Stem Density</b>	Density of all woody stems at 0.5 m	Counted stems within belt transect (1 X 22.6 m)	Density / m <sup>2</sup>	Prey
<b>Canopy Height</b>	Mode height of canopy	Measured or estimated height with meter tape or clinometer	Mode height in m	Cover & Line recovery
<b>Canopy Closure</b>	Overhead closure	Estimated closure using convex (20°) densitometer	Number of spaces reflecting unobstructed sky.	Cover
<b>Average DBH</b>	Average diameter at 1.43 m above ground for any woody stem ( $\geq$ 8 cm)	Average DBH (in cm) of all counted trees using prism (2 Factor)	Average DBH (of all trees)	Line recovery
<b>Basal Area</b>	Total volume of wood for live trees	Calculated volume (m <sup>3</sup> ) on above tree count	Volume (m <sup>3</sup> ) of wood from all live trees	Line recovery
<b>Trees on Line</b>	Tree along seismic line footprint	Presence / absence of trees	Count of lines with any trees	Line recovery

Table 2.2. Mean values (+/- **standard errors**) of selected vegetation attributes along conventional seismic lines at different stages of recovery in northwest Canada.

Location	n	Vegetation Attributes											Online Trees	
		Woody Debris		Horizontal Cover					Shrub density	Canopy		Average DBH		Basal area
		Count	Width	0.5m	1m	1.5m	2m	3m		Height	Closure			
Open	62	<b>0.43</b> ± 0.12	<b>12.42</b> ± 1.01	<b>3.64</b> ± 0.14	<b>2.27</b> ± 0.17	<b>1.12</b> ± 0.16	<b>0.84</b> ± 0.11	<b>0.59</b> ± 0.11	<b>2.27</b> ± 0.23	<b>1.29</b> ± 0.13	<b>59.93</b> ± 3.23	n / a	n / a	<b>0</b>
Partial	73	<b>0.91</b> ± 0.14	<b>13.30</b> ± 0.93	<b>4.29</b> ± 0.08	<b>3.36</b> ± 0.11	<b>2.17</b> ± 0.17	<b>2.16</b> ± 0.13	<b>1.83</b> ± 0.13	<b>3.67</b> ± 0.20	<b>3.21</b> ± 0.19	<b>54.80</b> ± 2.61	n / a	n / a	<b>15</b>
Closed	71	<b>1.22</b> ± 0.14	<b>15.98</b> ± 1.06	<b>4.59</b> ± 0.06	<b>4.11</b> ± 0.09	<b>3.29</b> ± 0.19	<b>3.38</b> ± 0.11	<b>3.04</b> ± 0.12	<b>4.00</b> ± 0.21	<b>5.12</b> ± 0.31	<b>25.16</b> ± 2.76	n / a	n / a	<b>39</b>
Interior	206	<b>3.46</b> ± 0.24	<b>16.50</b> ± 0.85	<b>4.47</b> ± 0.04	<b>3.79</b> ± 0.06	<b>3.00</b> ± 0.18	<b>3.13</b> ± 0.07	<b>2.99</b> ± 0.07	<b>3.15</b> ± 0.15	<b>18.64</b> ± 0.55	<b>21.30</b> ± 1.65	<b>20.94</b> ± 0.71	<b>21.59</b> ± 0.91	n / a

Table 2.3. Final sample sizes, per seismic line treatment, used to the test behavioural response (i.e. use) of American marten (*Martes americana*) to seismic line types in northwest Canada. Sample sizes include only those cameras that were functional (operational > 24 hours) and occurred within clusters where marten were detected.

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Treatment	n
<i>Seismic Lines</i>	
Open Wide	39
Partial Wide	49
Closed Wide	46
Open Narrow	23
Open 3-4 m	15
Open 5 m	22
Forest Interior	229

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Table 2.4. Behavioural response of American Marten (*Martes americana*) to tested seismic line types as compared to undisturbed forest interior locations in northwest Canada. Results are reported as an odds ratio showing both the direction and magnitude of marten response; interior cameras serve as a reference category and the likelihood of use there is set at 1. Where the odds ratio is  $> 1$ , use was more than expected; when  $< 1$ , use was less than expected. Compared to forest interior locations, marten use open conventional seismic lines almost 80% less than forest interior locations, however, use rebounds with line recovery. Further, use does not differ between interior locations and narrow LIS lines  $\leq 2$  m in width.

Seismic Line Type	Odds Ratio	SE	<i>P</i>	95 % Confidence Interval	
				Lower	Upper
Open Wide	0.223	0.113	0.003	0.083	0.603
Partial Wide	0.819	0.253	0.519	0.448	1.500
Closed Wide	1.488	0.493	0.231	0.777	2.850
Open Narrow ( $\leq 2$ m)	0.717	0.373	0.523	0.259	1.990
Open 3-4 m	0.350	0.158	0.020	0.144	0.848
Open 5 m	0.100	0.092	0.013	0.016	0.610



Table 2.5. Final models comparing predicted American marten (*Martes americana*) occupancy at the home range scale to different definitions of linear feature density in northwest Canada. Each model has 3 parameters in addition to the dependent variable (marten response): the specified linear feature metric, the proportion of lowland forest types within the home range, and Julian date at remote camera set up. Corrected seismic line and corrected linear feature definitions exclude low-impact seismic (LIS) lines  $\leq 2$  m. Corrected metrics perform best in predicting occupancy.

Linear feature definition	$\beta$	SE	QIC	$\Delta$ QIC
Cumulative seismic line	-0.083	0.030	204.219	3.401
<b>Corrected seismic line</b>	<b>-0.116</b>	<b>0.039</b>	<b>200.818</b>	<b>0.000</b>
Cumulative linear feature	-0.073	0.028	205.137	4.319
Corrected linear feature	-0.100	0.034	202.265	1.447
Roads	0.449	0.316	212.829	12.011
Roads and pipelines	0.130	0.202	214.176	13.358

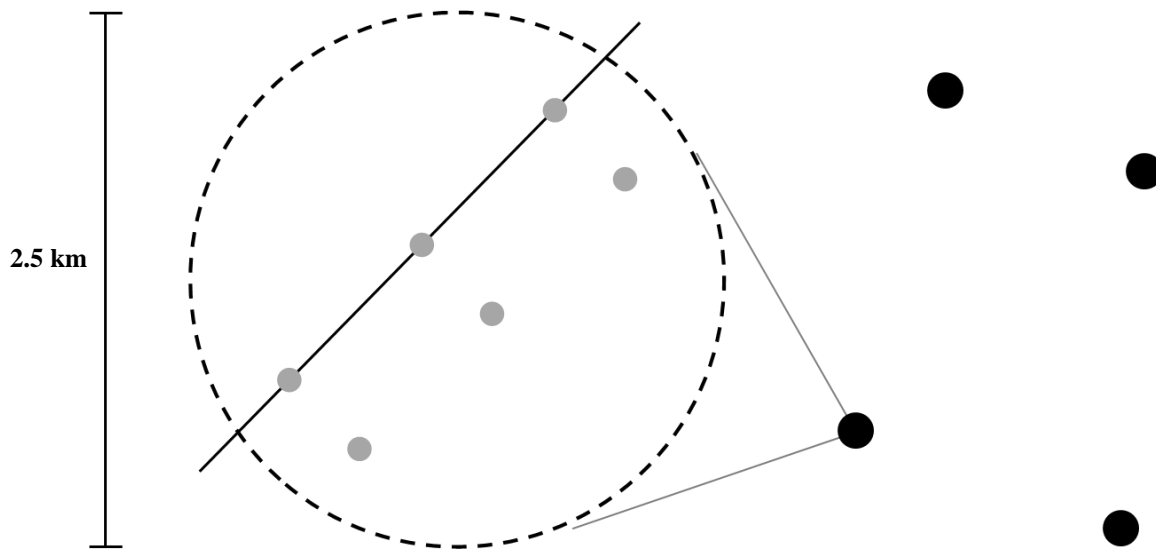


Figure 2.1. American marten (*Martes americana*) response to seismic line types and linear feature density was measured using remote cameras set in a nested design. Response to seismic line types (i.e., use) was measured relative to undisturbed forest interior locations at the camera scale (shown as small grey circles) by pairing cameras on and off seismic lines (shown as solid black line). Cameras were left in place to sample for ten trap nights, and were spaced within pairs by 450 to maintain independence between sampling locations. Response to linear feature density was measured as home range occupancy at the cluster scale by comparing occupied to unoccupied clusters. A cluster was comprised of a group of three camera pairs spaced at 900 m intervals along a single seismic to create a 5 km<sup>2</sup> (1.25 km diameter) effective sampling area (shown as large dashed black circle), the average marten home range in our region. Occupancy was met with a single detection at any camera station within the cluster (at 1 of 6 cameras within 60 trap nights; Raphael 1994). Clusters (shown as small black circles) were spaced by a minimum linear distance of 5 km to maintain sampling independence.



Figure 2.2. American marten (*Martes americana*) behavioural response (i.e., use) to seismic lines was measured at six different seismic lines treatments and compared to expected use measured at undisturbed forest interior locations. Seismic line treatments shown are as follows: (T, from L to R) open conventional, partially recovered conventional, and closed conventional lines; (B, from L to R) open low impact seismic (LIS) line  $\leq 2$  m wide, open LIS 3 – 4 m wide, and open LIS 5 m wide. (See also Table 2.2). Photo credits: J. Tigner.

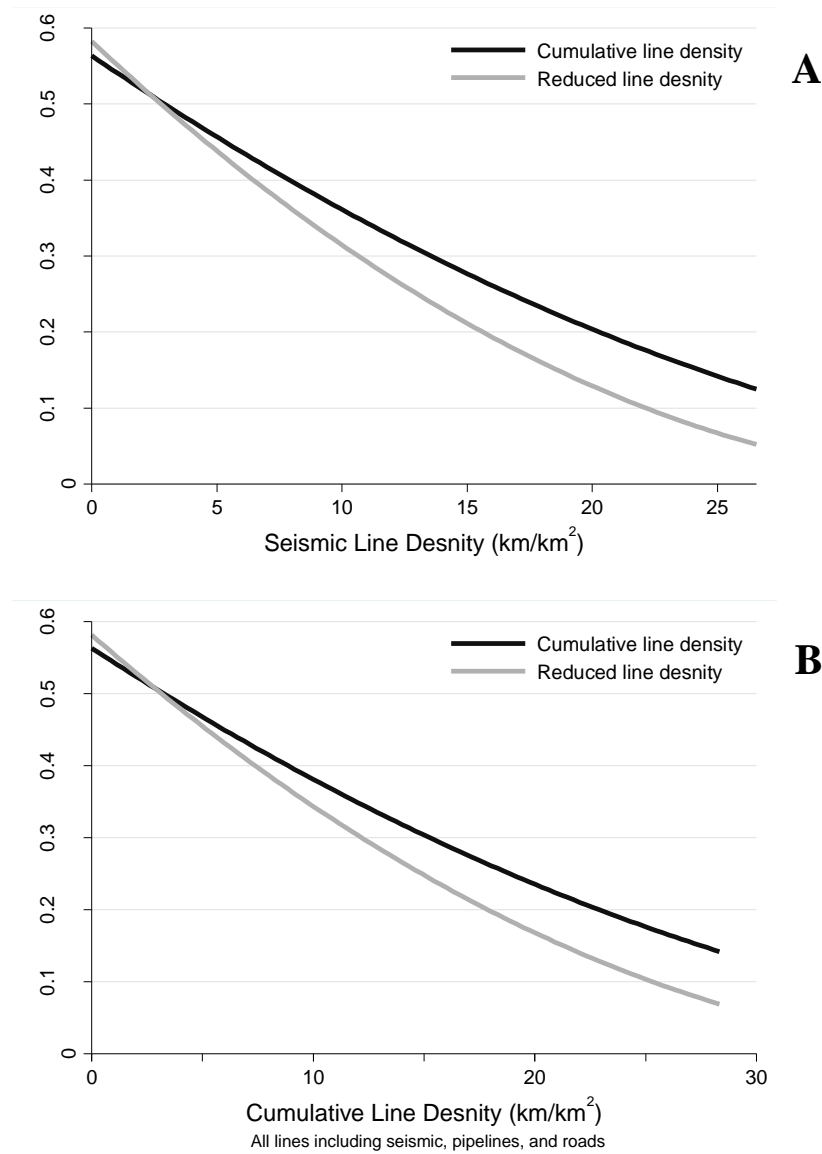


Figure 2.3. There is a negative relationship between the predicted probability of home range occupancy by American marten (*Martes americana*) and the linear feature density within a sampled home range in northwest Canada. The shape of that relationship is similar for seismic line density (A) and for cumulative linear feature density (including seismic lines, roads, and pipelines) metrics (B). In both panels, occupancy between cumulative (black line) and corrected (grey line; excluding LIS seismic lines  $2 \leq m$ ) is shown.

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## **CHAPTER 3: BLACK BEAR USE OF SEISMIC LINES IN NORTHERN CANADA: IMPLICATIONS FOR BOREAL WOODLAND CARIBOU**

### **INTRODUCTION**

Understanding wildlife response to anthropogenic disturbance is a principal goal of ecological field studies. Behavioural responses of animals to linear features have been widely studied, but research has focused mainly on edge effects (Donovan et al. 1997, Dijak and Thompson 2000, Ries and Sisk 2004) and the effects of habitat fragmentation (With and Crist 1995, Fahrig 1997, Chalfoun et al. 2002). Less emphasis has been placed on how linear features facilitate the movement of species and the associated changes in ecological processes that results from altered movement patterns. In western Canada, boreal woodland caribou (*Rangifer tarandus caribou*) populations have declined over the last several decades, especially where natural resource development is extensive (Sorensen et al. 2008, Festa-Bianchet et al. 2011). One cause of this decline is thought to be increased predation (Latham 2009, Environment Canada 2011). Previous research on wolves (*Canis lupus*) has shown use of linear features has facilitated wolf access to and movement through caribou habitat (Latham et al. 2011a). The resulting change in predator behaviour has increased caribou-wolf encounter rates (Whittington et al. 2011) and generally dissolved the spatial separation caribou have from their predators (James and Stuart-Smith 2002, James et al. 2004).

Although wolves are the primary predators of caribou across northern Canada (McLoughlin et al. 2003), black bears (*Ursus americanus*) can also be important predators, particularly for calves (Rettie and Messier 1998, Pinard et al. 2012). While bears are less predatory than wolves, black bear density in the northern boreal forest is an order of magnitude higher than that of wolves (Latham et al. 2011*b*). In eastern Canada, predation of caribou by bears is thought to have a greater effect on caribou population dynamics than predation by wolves (Faille et al. 2010, Pinard et al. 2012). In northwest Canada, black bears generally use upland habitats (Czetwertynski 2007, Latham et al. 2011*b*), but individual variation results in some bears using lowland forest within caribou range (Latham et al. 2011*b*). Such individuals may develop a search image for caribou, or simply predate caribou by chance while foraging for other food items (Latham et al. 2011*b*). Whether linear features increase bear habitat use of lowland forests used by caribou is unknown, however, several studies from across North America show black bears use linear disturbances for travel and foraging (Czetwertynski 2007, Mosnier et al. 2008*a*, Schwartz et al. 2010, Switalski and Nelson 2011).

The energy sector footprint in northern Canada is a network of linear features including roads, pipelines, and seismic lines. Used to explore for and delineate hydrocarbon formations, seismic lines are the most abundant linear feature in the region (Schneider 2002, Lee and Boutin 2006) and, as a result, are a key management challenge in northern jurisdictions (AANDC 2011, OGC 2011). However, our understanding of seismic line impacts on animal movements and

habitat use is lacking in two fundamental ways. First, little attention has been paid to how animals respond to different types of seismic lines. Prior to the mid-1990s, seismic lines were constructed between 6 and 10 m wide (hereafter conventional lines; Lee and Boutin 2006). In response to slow, incomplete, and inconsistent line recovery (Revel et al. 1984, Lee and Boutin 2006, Kemper and Macdonald 2009, Jorgenson et al. 2010), construction practices shifted to narrower seismic lines (hereafter low-impact seismic, or LIS) and by the early 2000s LIS was widespread (Weclaw and Hudson 2004, Tamarack Solutions 2003, AECOM 2009). Most LIS lines range from  $\leq 2$  to 5 m wide. Despite a variety of line recovery states and widths, past research on wildlife response to seismic lines has typically treated all seismic lines as a single disturbance type (James and Stuart-Smith 2000, Dyer et al. 2001, Dyer et al. 2002, but see Bayne et al. 2005a, Latham et al. 2011a). Understanding how specific seismic line attributes influence species habitat use and selection is critical for the development of effective regulatory decisions and reclamation strategies.

The second limitation of past wildlife – seismic line research is the focus on species' use or avoidance of individual seismic lines (Ashenurst and Hannon 2008; Machtans 2006; Dyer et al. 2001; Latham et al. 2011a, but see Bayne et al. 2005b). Of greater consequence for determining whether seismic lines alter ecological relationships is whether changes in wildlife behaviour at the scale of an individual line are manifest at larger scales whereby animal use is more or less likely in areas with a higher density of seismic lines. A general increase in use of areas with many seismic lines at a broad scale for a ubiquitous predator like the

black bear could have significant direct (i.e., increased encounter rate) and indirect (i.e., decreased spatial separation) consequences for caribou.

The goal of this study was to measure the behavioral response of black bears to seismic lines at two spatial scales. First, we used remote cameras to measure whether black bears used seismic lines more than forest interiors at the level of an individual line. Specifically, we compared the use of undisturbed forest interior locations to seismic lines that varied by width and vegetation recovery state. Second, we evaluated whether black bear habitat use was influenced by seismic line density at a 5 km<sup>2</sup> scale based on a cluster of six cameras. We measured the probability of bear occurrence in upland and lowland forest types across a continuum of cumulative seismic line density.

## **STUDY AREA**

We measured black bear response to seismic lines across 200,000 km<sup>2</sup> of boreal forest in northwest Alberta, northeast British Columbia, and southwest Northwest Territories between 61°48' and 58°48' latitude and 122°41' and 117°39' longitude in 2008 and 2009. Energy sector activity within the area was widespread south of the 60<sup>th</sup> parallel (in Alberta and British Columbia), but less common further north. Forestry occurred within some parts of Alberta and British Columbia, but we avoided these areas when designing our sampling strategy. Human density averaged < 1 person/km<sup>2</sup> (Weiss et al., 2008) and was

clustered in several small communities. Bear hunting was uncommon in the study area.

Area topography was flat or slightly undulating. Extensive peatland and fen complexes (collectively lowland forests) occurred on flat poorly drained sites with organic soils; upland forests were found on sloped or undulating terrain with well drained mineral soils. Lowlands were dominated by black spruce (*Picea mariana*), sometimes mixing with trembling aspen (*Populus tremuloides*) or tamarack (*Larix laricina*); the lowland understory was sparse and composed mainly of willows (*Salix* spp.), dwarf birch (*Betula glandulosa*), and Labrador teas (*Ledum* spp.). Uplands were dominated by mixed stands of white spruce (*Picea glauca*) and trembling aspen or balsam poplar (*Populus balsamifera*), but pure conifer or deciduous stands did occur. Dense upland understory was mainly low-bush cranberry (*Viburnum edule*), alder (*Alnus* spp.), red osier dogwood (*Cornus sericea*), and rose (*Rosa* spp.).

## **METHODS**

### **Forest Type and Disturbance Footprint**

In a Geographic Information System (GIS; ArcGIS 9.3, ESRI, Redlands, California), we reclassified Earth Observation for Sustainable Development (EOSD; Wulder et al. 2003) into upland and lowland forests, shrub-dominated stands, naturally open habitats, water bodies, and anthropogenic disturbance. We filled gaps in the EOSD layer with Ducks Unlimited Canada Earth Cover

Classification and MODIS data reclassified to the same classes. Energy sector footprint was compiled from satellite and aerial imagery, and from government and industry shapefiles. We generated individual layers for seismic lines, roads, pipelines, well-pads, and oil field facilities. Duplicate features between data products were removed from each layer. Because footprint features often evolve from one feature type to another (e.g. a seismic line is converted to a pipeline or road; Schneider 2002), we also deleted and spatially adjusted overlapping feature types as required to maintain overall footprint accuracy.

### **Site Selection and Data Collection**

To isolate bear response to seismic lines and to cumulative line density we controlled for dominant forest type *a priori* based on our GIS layers. Within a 5 km<sup>2</sup> roving window we stratified the study area into upland and lowland forest types ( $\geq 50$  % of habitat with 5km<sup>2</sup>), and measured cumulative seismic line density as a continuous variable. We randomly generated a set of candidate sites across a continuum of cumulative seismic line density ranging from  $< 0.1$  km/km<sup>2</sup> to  $>26$  km/km<sup>2</sup> within each forest type; candidate sites were generated at a minimum spacing of 5 km from one another or a community, but within 15 km of tracked or water access so they could be reached. In the field, if selected sites were inaccessible or incorrectly categorized, the next closest site was chosen until suitable.

At each site we used remote wildlife cameras (Bushnell Scout, Bushnell Corp. Overland Park, Kansas; and Reconyx P85, Reconyx, Inc., Holmen,



Wisconsin) to sample the behavioural response of bears to seismic lines and the probability of use as a function of cumulative seismic line density. All camera trapping was approved by the University of Alberta Animal Care Protocol No. 476805 and the Government of the Northwest Territories Wildlife Research Permit No. WL-005752.

## **Behavioural Response**

### *Seismic line type*

The behavioural response of black bear to seismic line types was evaluated as the probability of line use relative to use of the forest interior by pairing cameras on and off seismic lines. Line cameras were pointed along a seismic line at one of the following treatments: 1) line  $\geq 6$  m wide and open (hereafter open conventional); 2) line  $\geq 6$  m wide and partially recovered (partial conventional); 3) line  $\geq 6$  m and recovered (closed conventional); 4) line open and  $\leq 2$  m in width (open 2m); 5) line open and 3-4 m wide (open 3-4m); and 6) line open and 5 m wide (open 5m). This explicitly measured bear response to conventional seismic lines at varying recovery states (treatments 1-3) and to typical LIS construction widths (treatments 4-6).

The recovery state of conventional seismic lines was differentiated using field measurements of vegetation structure (Bayne et al. 2011). Specifically, we measured: horizontal cover (i.e., visual obstruction) to 1 of 6 rankings (0, open; 1,  $<10\%$  obstruction; 2,  $\geq 10$  to 25%; 3,  $\geq 25$  to 50%; 4,  $\geq 50$  to 75%; or 5,  $\geq 75\%$ ) from 10 m at five height classes (ground to 0.5 m, 0.5 to 1 m, 1 to 1.5 m, 1.5 to 2

m, and 2 to 3 m) using a cover board (Nudds 1977); shrub stem density to stems per hectare for all woody stems  $\geq 0.5$  m and  $< 8$  cm at diameter breast height (D.B.H) within belt transects; mode canopy height in to half meter increments; overhead cover to a relative ranking of closure (0 as no overhead cover and 96 as 100% closure) using a 20° concave densitometer (Cook et al. 1995, Korhonen et al. 2006); count and width of woody debris  $\geq 50$  cm in length and  $\geq 8$  cm in width intersecting line transects; and finally a tally of trees (any woody stem  $\geq 8$  cm at D.B.H.) growing on lines. All measures were also taken in undisturbed forest plots adjacent to line treatments to provide a reference vegetation state. Average attribute values are summarized in Table 3.1.

All paired cameras were spaced by 450 m to maintain independence between sampling locations. Interior cameras were set as far from disturbances as possible ranging between 50 m and 450 m. Cameras were always set away from major disturbance features such as roads, pipelines, or well-pads. Cameras were baited with approximately 150 g of canned dog food and 50 g of tinned sardines packed in water once at camera establishment, programmed to collect data 24 hours/day, and retrieved after 10 trap nights. At each camera location we classified the surrounding forest as upland deciduous, upland mixed wood, upland conifer, or lowland conifer; and we estimated stand age on a rank scale from 1 to 4 corresponding to initiation, stem exclusion, mature, and old-growth, respectively (Chen and Popadiouk 2002). Cameras were not set in open wetlands or recent burns, or in areas with  $< 10\%$  canopy cover.

### ***Cumulative seismic line density***

The behavioural response of black bear to cumulative seismic line density was evaluated as the probability of use (i.e., occurrence) at the cluster scale (5 km<sup>2</sup>) across a continuum of seismic line densities by comparing used and unused clusters. Each site consisted of three camera pairs that were spaced by ~900 m and collectively yielded a total of 60 trap nights; use was satisfied with a single photographic detection within the cluster. As 5 km<sup>2</sup> is far smaller than a typical black bear home range in the boreal forest (Bertram and Vivion 2002; Mosnier et al. 2008b; Brodeur et al. 2008), we did not assume this spatial unit was closed during sampling and therefore consider a photograph to reflect the probability of bear use at the cluster scale, not bear occupancy (Kendall and White 2009) or abundance (Gardner et al. 2010).

### **Statistical Analysis**

#### ***Seismic line type***

We assessed black bear use (bear photographed at a camera = 1 vs. not photographed = 0) of seismic lines using a population averaged generalized estimating equation (GEE) with a logit link and binomial error family in Stata 11.1 IC (Stata-Corp, College Station, Texas). GEEs are a modification of generalized linear models that account for potential correlation within panels of hierarchically structured data, in this case individual cameras nested within clusters (Hardin and Hilbe 2003). We assumed correlation of bear detections between cameras within a cluster was constant (i.e., exchangeable), but used a

semi-robust estimator of variance to generate standard errors robust to potential misspecification.

A failure to distinguish between unavailable and unused points could induce error in our estimate of bear response to different seismic line treatments (Aarts et al. 2008, Beyer et al. 2010). Therefore, we used only cameras within clusters where a bear was detected for this analysis. We assumed all cameras within a cluster were equally available to the sampled population such that use would reflect a behavioral decision. Some cameras failed prior to the full 10-night sample period; if failure occurred at set up (< 24 hours), that camera was removed from analysis. For each remaining camera, we calculated the natural log of the total sampling duration (in minutes), constrained the coefficients to 1, and modeled this adjusted weight to control for the actual amount of opportunity each camera had to detect a bear. We assumed the likelihood of detection at a given camera remained constant over the sample period.

In our global model, we included seismic line type, forest type, stand age at the camera station, whether a camera detected other species, the total number of other species photographed, distance from camera to closest adjacent upland stand in meters and as a squared term, distance from camera to closest minor road (secondary and tertiary combined), the proportion of upland and lowland forest types at the cluster scale, and Julian date at camera set up. The detection of other species at a camera was included because we thought a loss of bait to non-target species may have reduced camera attractiveness to bears. Prior to fitting a global model we screened for collinearity among variables using a  $|r| = 0.7$  cut-off. We

removed all non-significant variables using a backward stepwise procedure, and retained all variables significant at  $P < 0.1$  in the final model (Hosmer and Lemeshow 2000). We also tested for an interaction between line type and the cumulative density of seismic lines and forest type at the camera and cluster levels to measure whether bear response to seismic line types was constant or was influenced by large scale processes. To test whether observed patterns of use changed seasonally, we also ran the described analyses in 2 distinct time periods: caribou calving season (May through 30 June; Dyer et al. 2001; Latham et al. 2011b), and post calving (1 July through October).

Behavioural response to line types is reported as an odds-ratio using the forest interior as the reference condition. The odd-ratio shows both the direction and magnitude of difference in bear use of line treatments from what is expected at undisturbed, interior locations (i.e., controls). Where the odds ratio for a line treatment is  $> 1$ , the probability of bear use of that treatment is higher than that of the interior; where the odds ratio is  $< 1$  the probability of use is lower (Larsen et al. 2000). When the 95% confidence interval of the odds ratio includes 1, any difference in use is not statistically significant between treatments (Hosmer and Lemeshow 2000).

### *Cumulative seismic line density*

We assessed black bear response to cumulative line density as the probability of use at the cluster scale (photograph of bear in a cluster = 1 vs. no photograph of bear = 0) relative to forest type and cumulative seismic line density using the

same GEE procedure described above. Upland and lowland forests occurred throughout the study area, but varied to some degree in structure and composition by latitude- and longitude-related climatic differences. To control for this potential influence, we nested our 5 km<sup>2</sup> sampling clusters within ecological districts. An ecological district is a Canada-wide land cover classification describing average forest stand composition and productivity based on biotic and abiotic attributes including soil, geology, climate, and vegetation communities (ESWG 1996). Clusters of mostly upland or lowland forest occurred within each ecological district, and we assumed constant correlation between cluster use within each district.

Forest type and cumulative seismic line density were included in analyses as continuous variables; forest type as the proportion of upland and lowland forest within the cluster and line density as km/km<sup>2</sup> calculated at the cluster scale. Additionally, we included the distance to main roads, communities, areas of oil field activity (i.e., plants, drilling, completions, etc.), and Julian date. Prior to fitting a global model we screened for collinearity among variables using a  $|r| = 0.7$  cut-off value, and then removed all non-significant variables using a backward stepwise procedure. In the final model, we retained all variables significant at  $P < 0.1$ . To control for camera failure, sample duration at the cluster was accounted for as described above. Finally, we tested for an interaction between habitat and cumulative seismic line density to measure whether the probability of use of lowland forest type was influenced by the density of seismic lines at the cluster scale.

## RESULTS

Cameras were deployed at 1035 unique locations across 173 clusters during field work between 22 May and 3 September in 2008 and between 18 May and 4 October in 2009. Sixty-three cameras sampled < 24 hours (failed) and were removed from analyses; camera failure occurred randomly across line treatments and political jurisdiction. Black bears were detected at 225 out of 972 remaining locations and in 99 of 173 clusters. Twenty-two camera clusters were established differently than described above (i.e., +/- 6 cameras and or different camera/pair spacing) to attain sufficient line treatment replication and were not included in broad scale (5 km<sup>2</sup>) analyses. Black bears were detected in 87 of those remaining 151 clusters.

### *Seismic line type*

A total of 579 unique camera locations across the 99 total clusters with black bear detections were used in this analysis (Table 3.2). The probability of black bear use was higher at all seismic line types compared to use of undisturbed forest interior locations, except along lines  $\leq 2$  m wide where use did not significantly differ from interiors (Table 3.3). Use of open conventional seismic lines was five times higher than use in forest interiors. The likelihood of line use decreased with increasing recovery of woody vegetation (Table 3.1), however use along closed lines was still twice as likely as that of the forest interior. Although construction of narrower seismic lines is the principle strategy behind LIS techniques, only

lines  $\leq 2$  m did not receive higher use by bears; open seismic lines 3-4 m and 5 m wide were used 2.6 and 3.5 times more often than forest interiors, respectively.

Forest type and stand age at the camera station also significantly influenced bear use (forest type  $\beta = -0.163$ ,  $P = 0.006$ ; stand age  $\beta = -0.188$ ,  $P = 0.107$ ), but were difficult to interpret as categorical variables because there was no evidence of a linear change. We collapsed these variables into upland and lowland habitats based on field observation and into young and old stands where rank ages 1 and 2 were considered young, and 3 and 4 old; both remained significant (upland stands odds ratio = 1.878,  $P = 0.002$ ; young stands odds ratio = 1.541,  $P = 0.025$ ). At the stand scale, bears showed a preference for young upland stands. However, the proportion of upland or lowland forest types at the cluster scale did not significantly influence the detection of bears (upland odds ratio = 0.862,  $P = 0.693$ ; lowland odds ratio = 1.176,  $P = 0.656$ ), nor did the distance from upland stands (odds ratio = 0.999,  $P = 0.197$ ). The number of non-target species photographed at a camera influenced bear detection (odds ratio = 0.727,  $P = 0.033$ ), but after controlling for this influence the probability of use was still higher on seismic lines. Distance to minor roads was not significant, but was retained in the final model (odds ratio = 0.978,  $P = 0.061$ ).

No interaction between bear response to line type, forest type at the camera, proportion of upland or lowland forest type, or cumulative seismic line density at the cluster scale was significant. This suggests that once a bear was present in an area their response to seismic line types was constant; bears simply used upland stands more frequently than lowland stands, and lines more



frequently than forest interiors. In uplands we detected bears 92 times on lines and 57 times in the interior; in lowlands we detected bears 51 times on lines and 25 times in the interior. Bear detection at the cluster scale was not influenced by Julian date (odds ratio = 1.003,  $P = 0.222$ ) or caribou timing periods.

### *Cumulative seismic line density*

Of the 173 established clusters, 151 were used in this analysis; of the 151 clusters used, black bears were detected at 87. The probability of black bear use at the cluster scale was predominantly driven by the proportion of forest type within the cluster, rather than the cumulative seismic line density (Figure 3.1 A). Further, probability of use declined with an increased amount of lowland forest types ( $\beta = -1.697$ ,  $P < 0.001$ ). Where lowland forests comprised  $\leq 50\%$  of a sampled cluster, bears were detected at 54 of 107 clusters (50.47% occurrence); where lowlands comprised  $\geq 90\%$  of the cluster, occurrence dropped to 37.93% (11 detections of a possible 29). By contrast, bears were detected at 33 of 44 clusters (75% occurrence) where uplands forests comprised  $\geq 50\%$  of the sampled cluster, and at 9 of 10 clusters (90% occurrence) where uplands comprised  $\geq 90\%$  of the cluster.

We found no evidence that black bear use at the cluster scale was influenced by the cumulative density of seismic lines ( $\beta = -0.009$ ,  $P = 0.645$ ) (Figure 3.1 B), or evidence for an interaction between seismic line density and dominant forest type ( $\beta = -0.030$ ,  $P = 0.775$ ). Although insignificant, the relationship between bear use at this scale and line density was weakly quadratic, suggesting some increase in use of areas with moderate line densities. No other

variables included in the global model were retained in our final model. Julian date was not significant ( $\beta = -0.028$ ,  $P = 0.360$ ) suggesting the pattern of typical bear habitat use did not vary across seasons or relative to caribou calving. Distance to main roads, communities, and areas of oil field activity occasionally displayed evidence of quadratic relationships with bear detections, but often caused model convergence issues precluding robust interpretations of these effects (Hosmer and Lemeshow 2000). Within group (ecological district) model residuals were extremely low (- 0.014) suggesting any spatial autocorrelation between used clusters was accounted for in the model.

## DISCUSSION

At the individual seismic line scale, black bears show clear use of most line types. The likelihood of use does dissipate with line recovery, but even heavily regenerated lines were used more than twice as often as forest interiors. Only seismic lines  $\leq 2$  m wide were not used differently from interior locations. The observed use of open line types is not surprising as previous work shows strong selection by black bears for recent disturbances and linear features (Czetwertynski 2007, Brodeur et al. 2008, Mosnier et al. 2008a, Schwartz et al. 2010, Carter et al. 2010). However, persistent use of closed lines is inconsistent with other studies that show use of roads and seismic lines subsides with reclamation (Neufeld 2006, Switalski and Nelson 2011).

Determining the mechanisms driving bear use of seismic lines was beyond the scope of this study, however, both food- and movement-based hypotheses likely contribute. Black bears are omnivorous and highly opportunistic, routinely taking advantage of clustered food resources during spring and fall resource pulses (Welch et al. 1990, Mosnier et al. 2008a, Garneau et al. 2008, Brodeur et al. 2008). For example, during spring green-up bears forage along road sides and linear features to take advantage of early spring growth (Czetwertynski 2007, Mosnier et al. 2008a, Bastille-Rousseau et al. 2011), and in the summer and fall use recent clear cuts with high berry productivity (Mosnier et al. 2008a, Brodeur et al. 2008). In our study area, plant community composition on open seismic lines varied significantly from forest interior plots (Bayne et al. 2011), and supported increased densities of *Carex* spp. and herb species, known food resources for black bears (MacHutchon 1989, Welch et al. 1990, Partridge et al. 2001, Mosnier et al. 2008a). We observed anecdotal evidence of bears grazing and digging for plant roots on lines during green-up, and evidence of digging for roots and insects in the spring and fall, respectively. Intuitively, movement along open line features should be easier than movement through undisturbed boreal forest for large bodied animals like black bears (it was for us human field ecologists) and we observed several examples of bears simply walking past camera stations along open lines in captured photographs (Figure 3.2).

On older seismic lines in our study area, metrics of vegetation structure and composition became more similar to those of forest interior plots (Bayne et al. 2011). This reduces the likelihood that black bear use of older seismic lines is

related to increased food resources. If line use was entirely explained by food availability, probability of use at older lines and forest interior locations should have been equal based on vegetation composition. However, use on closed lines was more than twice as likely as use in the interior suggesting that closed lines still facilitate movement. Along half of the sampled seismic line locations in our system we found distinct game trails (on 231 of 458 lines; 50.44%). On closed lines this percentage was even higher with game trails on 64% (on 52 of 81 closed lines). Thus, even though seismic lines recover based on vegetation structure and composition attributes, game trails seem to persist and resulted in continued use by bears.

Low impact seismic (LIS) line construction techniques were developed as a way for the energy sector to mitigate potential impacts of resource exploration on wildlife before those impacts actually occur (Weclaw and Hudson 2004, AECOM 2009). In some locations, significant effort has also been allotted toward physically reclaiming existing seismic lines to reduce or reverse known impacts to wildlife (Neufeld 2006). Testing the assumptions linking these strategies to actual species response patterns is critical to ensure management actions achieve their intended goals. However, these assumptions are rarely tested (Weclaw and Hudson 2004). Our data shows that the construction of narrower seismic lines, a primary LIS technique, may prevent line use by bears, but that lines must be constructed to  $\leq 2$  m in width to prevent use. Wider seismic line widths may be considered LIS line types by regulatory standards, but fail to meet the objective of reducing predator movement.

A second component of LIS construction techniques is to minimize ground disturbance during line preparation by elevating bulldozer blades or using mulching blades where possible (AECOM 2009, AANDC 2011). While this may facilitate more rapid recovery of vegetation along ageing seismic lines (although we are not aware of research assessing this claim), our data show that for black bears, line recovery does not equate with the recovery of vegetation. Instead, the establishment of game trails along younger seismic lines and the persistence of those trails along heavily regenerated lines contribute to the long term use of lines well beyond when lines may be considered recovered based on other metrics. Intensive experimental reclamation has shown that physical line closure activities can successfully dissuade black bears from using at least short line segments (Neufeld 2006). However, there are currently hundreds of thousands of kilometers of seismic lines in western Canada (Morrell et al. 1995, Schneider 2002, Nitschke 2008) and intensive reclamation to close all of these lines will be difficult.

Black bears clearly use seismic lines when they are encountered, however, our data do not suggest this alters bear use of forest types. At the line scale, the rate of seismic line use was consistent in both upland and lowland forest types. If lines facilitated bear use of lowland forests, we would have expected the odds of line use in lowland forests to have been much higher than in upland forests. However, we found no evidence of an interaction between these variables; the likelihood of use was comparable in both forest types. Further, despite evidence of bears traveling along seismic lines in lowlands, those movements were likely

not long distance ones. For example, along 32 open seismic lines where cameras were set in succession, only 2 instances showed likely movement between locations (based on time stamps and visual inspection of bears in photographs); in both instances movement was between adjacent line cameras (900 m spacing).

At the cluster scale our data shows that the probability of use by bears is driven by the proportion of forest type rather than seismic line density. Black bear use of individual seismic lines does not result in a higher likelihood of use of areas with high line densities. Instead, at broader spatial scales, bears are more likely to use upland forests than lowland forests regardless of seismic line density. That black bears use upland forest types more than lowland ones is well supported in the literature (Czetwertynski 2007, Carter et al. 2010, Latham et al. 2011*b*), and this makes sense for a species with a largely plant based diet in our system where lowland forests are of very low productivity (Bonan and Shugart 1989, Mosnier et al. 2008*a*).

By contrast, in eastern Canada, widespread timber harvest in has triggered increased black bear use of or easy transit through lowland stands typically used by caribou (Brodeur et al. 2008, Mosnier et al. 2008*a*, Pinard et al. 2012). Our cluster scale data suggest the ecological impact of the energy sector disturbance footprint (oil sands mining and in situ operations notwithstanding) is fundamentally different from that of forestry from the perspective of black bears. Forestry operations disturb large amounts of a working landscape and ultimately cluster plant-food resources sufficiently to increase bear use of those areas. However, the energy sector does not. Although open conventional or wider LIS

lines may cluster food resources for bears to some degree, the overall disturbance footprint associated with energy sector development directly impacts only a small proportion of a landscape (Schneider 2002), and it is not likely able to cluster food resources similarly to forestry operations. Ecologically speaking, this is a critical distinction because bears are not believed to be drawn to recovering cut blocks to hunt caribou, but to take advantage of new plant-food resources (Bastille-Rousseau et al. 2011). The resulting increase in bear presence within caribou range is what leads to increased caribou predation (Faille et al. 2010, Pinard et al. 2012). It is possible later stages of energy development where high densities of roads, pipeline, well pads, compressors, and camps may sufficiently alter lowland habitats to become attractive to black bears, however, those landscapes were not the target of this study and were largely excluded from our sampling regime.

As demonstrated here, anthropogenic habitat disturbance changes the ways species use their environment and often does so in different ways at different scales (Levin 1992). To interpret how behavioural changes in one species might alter ecological relationships with other species, interpretation should be at a scale concordant with the ecological or management question of concern (Werner 1992; Gustafson 1998). The goal of this study was to assess whether black bears use seismic lines more than the forest interior at different scales and what implications this might have for changing spatial overlap between black bears and caribou. Although we found strong evidence black bears use seismic lines, we did not see evidence of behavioural changes at large enough or

ecologically relevant scales to trigger changes in black bear-caribou spatial relationships. Therefore, we caution against selectively using portions of this study to say that seismic lines will increase black bear impacts on caribou. If seismic lines increase black bear hunting efficiency or increase the interaction rate between caribou and bears as has been demonstrated for wolves (Whittington et al. 2011) then such a statement would be warranted. However, the fact that black bears use seismic lines does not necessarily increase the number of caribou that are being killed by bears. Further studies are needed to evaluate why some black bears use lowland environments, if bears kill caribou closer to seismic lines than expected, and if areas with higher seismic line density can support a higher density of black bears irrespective of the probability of bear use.

### **MANAGEMENT IMPLICATIONS**

Effective management of predator prey relationships pertaining to boreal woodland caribou in Canada requires the flexible application of policy strategies that incorporate detailed knowledge of species' behaviour. Because it is likely infeasible to reclaim all seismic lines once they are constructed, we recommend that where new seismic lines are required they be cut as narrow as possible to limit use by black bears. However, because neither seismic lines nor line density appear to trigger black bear use of lowland forest types, the strict regulation of very narrow line widths (i.e.,  $\leq 2$  m wide) should be reserved for those areas where line use by bears could have the biggest potential impacts on caribou. The



application of targeted rather than blanket management actions may better balance industrial and conservation interests thereby increasing the palatability and ultimately the success of conservation initiatives in regions where natural resources are an economic mainstay like in most boreal woodland caribou habitat in western Canada.

Table 3.1. Mean values (+/- **standard errors**) of vegetation attributes on conventional seismic lines at different stages line of recovery in northwest Canada. Mean values were used to categorize line closure to quantify black bear (*Ursus americanus*) response patterns to seismic line recovery between 2008 and 2009.

Location	n	Vegetation Attributes										Tree Tally <sup>1</sup>
		Woody Debris		Horizontal Cover					Shrub density	Canopy		
		Count	Width	0.5m	1m	1.5m	2m	3m		Height	Closure	
Open	62	<b>0.43</b> ± 0.12	<b>12.42</b> ± 1.01	<b>3.64</b> ± 0.14	<b>2.27</b> ± 0.17	<b>1.12</b> ± 0.16	<b>0.84</b> ± 0.11	<b>0.59</b> ± 0.11	<b>2.27</b> ± 0.23	<b>1.29</b> ± 0.13	<b>59.93</b> ± 3.23	<b>0</b>
Partial	73	<b>0.91</b> ± 0.14	<b>13.30</b> ± 0.93	<b>4.29</b> ± 0.08	<b>3.36</b> ± 0.11	<b>2.17</b> ± 0.17	<b>2.16</b> ± 0.13	<b>1.83</b> ± 0.13	<b>3.67</b> ± 0.20	<b>3.21</b> ± 0.19	<b>54.80</b> ± 2.61	<b>15</b>
Closed	71	<b>1.22</b> ± 0.14	<b>15.98</b> ± 1.06	<b>4.59</b> ± 0.06	<b>4.11</b> ± 0.09	<b>3.29</b> ± 0.19	<b>3.38</b> ± 0.11	<b>3.04</b> ± 0.12	<b>4.00</b> ± 0.21	<b>5.12</b> ± 0.31	<b>25.16</b> ± 2.76	<b>39</b>
Interior	206	<b>3.46</b> ± 0.24	<b>16.50</b> ± 0.85	<b>4.47</b> ± 0.04	<b>3.79</b> ± 0.06	<b>3.00</b> ± 0.18	<b>3.13</b> ± 0.07	<b>2.99</b> ± 0.07	<b>3.15</b> ± 0.15	<b>18.64</b> ± 0.55	<b>21.30</b> ± 1.65	<b>205</b>

<sup>1</sup> Refers to the total number of seismic lines that had measureable trees (Diameter Breast Height ≥ 8 cm), not the average stem count per treatment. All but 1 interior plot contained measureable tree stems; the average number of stems per plot was 30.

Table 3.2. Total number of unique camera locations, by treatment, used to test the probability of black bear (*Ursus americanus*) use of seismic lines that varied by width and vegetation recovery state in northwest Canada, 2008-2009.

Treatment	n*
<b>Seismic Lines</b>	
Open Conventional	65
Partial Conventional	65
Closed Conventional	46
Open 2 m	34
Open 3-4 m	28
Open 5 m	34
<i>Total Seismic Lines</i>	272
Forest Interior	307

Table 3.3. Comparative behavioural responses of black bears (*Ursus americanus*) to different seismic line types in northwest Canada, 2008-2009. Results are reported as an odds ratio to show the direction and magnitude of bear response patterns. Interior cameras served as a reference category. The likelihood of use at interior locations is set at 1; where odds are  $> 1$ , the probability of use was higher than expected; where odds are  $< 1$ , use was less than expected. Response to line types is reported for pooled data for all line types during the leaf-on season, and for caribou calving (May – June 30) and non-calving (July 1 - October) seasons relative to collected bear data for conventional line types as indicated.

Seismic Line Type	Odds Ratio	SE	P	95 % Confidence Interval	
				Lower	Upper
Open Wide	5.458	1.805	$< 0.001$	2.855	10.436
Calving	6.539	3.432	$< 0.001$	2.338	18.290
Non-calving	5.201	2.130	$< 0.001$	2.335	11.610
Partial Wide	3.585	1.020	$< 0.001$	2.053	6.261
Calving	3.317	1.832	0.030	1.124	9.789
Non-calving	3.257	1.048	$< 0.001$	1.734	6.120
Closed Wide	2.328	0.808	0.015	1.179	4.595
Calving	6.614	4.351	0.004	1.822	24.011
Non-calving	1.478	0.577	0.318	0.687	3.178
Open Narrow	1.206	0.466	0.629	0.565	2.572
Open 3-4 m	2.636	0.890	0.004	1.360	5.120
Open 5 m	3.543	1.436	0.002	1.601	7.843

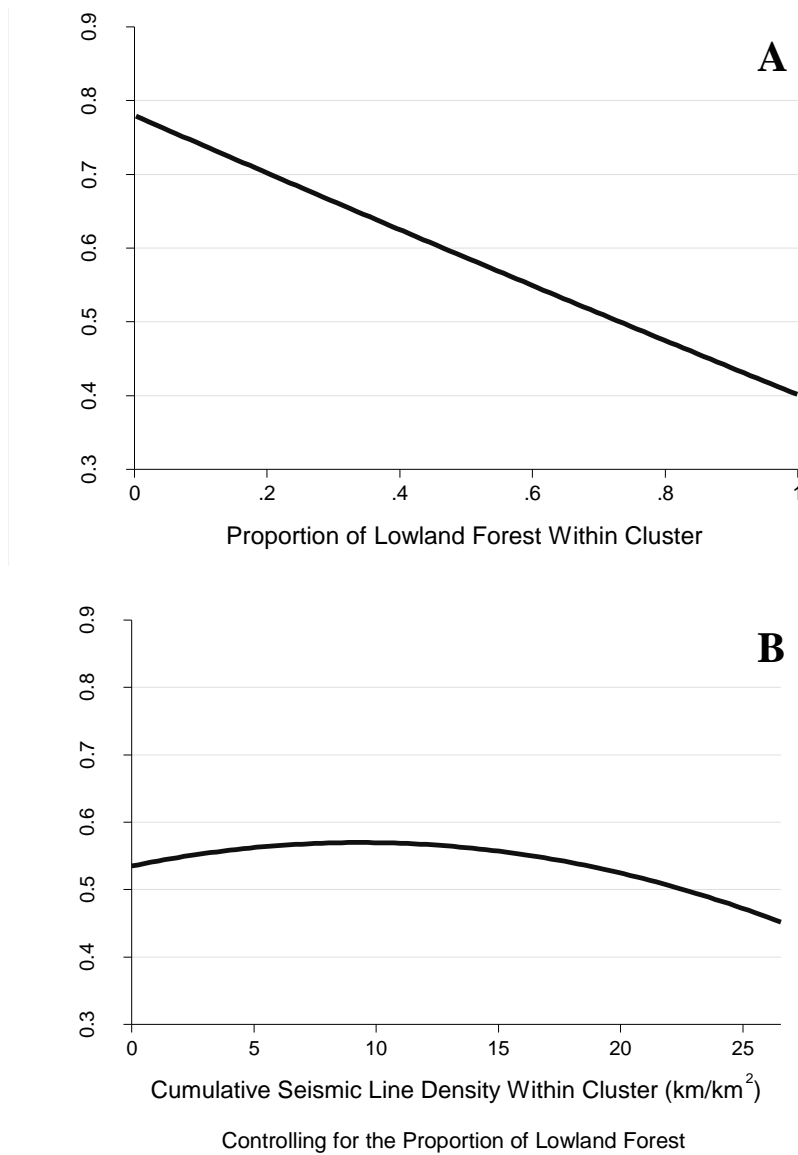


Figure 3.1. The probability of black bear (*Ursus americanus*) use at a 5 km<sup>2</sup> spatial scale relative to the proportion of lowland forest type (A) and cumulative seismic line density (B) in northwest Canada, 2008-2009. Black bear use was measured as the probability of use (i.e., occurrence) using six (6) individual remote cameras deployed for a ten (10) trap night period during the snow free seasons; the proportion of forest type and total seismic line density were calculated within a 1250 m buffer (4.91 km<sup>2</sup>) centered around the remote cameras.



Figure 3.2. Black bears walking along open seismic lines as captured by remote wildlife cameras. The image on the left shows a bear on a 6 m wide conventional seismic line in lowland forest within the Maxhamish caribou range in the Horn River Basin in northeast British Columbia. On the right the image shows a bear walking along an 8 m wide conventional seismic line in a mature upland mixed wood stand also in the Horn River Basin. In both cases the individuals passed the camera stations, but returned shortly afterward and were photographed investigating and eating the bait (dark spot on trees in foreground of each image).

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## **CHAPTER 4: GENERAL CONCLUSIONS**

### **CONCLUSION**

Effective land use planning depends on matching management strategies to clearly identified conservation priorities (Bettinger et al. 2003, Weber et al. 2012). This, in turn, requires a sufficient understanding of how human land use activities impact identified conservation priorities; in the absence of detailed information there is little ability to tailor management actions appropriately to meet conservation goals with a high level of certainty. In far northwest Canada the development of energy resources is set to expand rapidly (BC MEMPR and NEB 2006, GNWT 2007, BC MEMPR and NEB 2011) and several modeling exercises predict sweeping changes to boreal ecosystems as a result (Schneider et al., 2003; Nitschke, 2008; Strimbu and Innes, 2011). Although land use planning efforts in northern British Columbia and the Northwest Territories have identified the maintenance of healthy wildlife populations as long term conservation goals (Salmo Consulting et al. 2003, SENES Consultants 2005, DCLUPC 2006, SLUPB, 2010), the development of specific policies capable of balancing those goals with future energy development has lagged due to a general lack of data.

In this thesis I attempt to fill a portion of the current knowledge gap by providing a detailed understanding of wildlife response to seismic lines, the largest component of the energy sector disturbance footprint (Schneider 2002, Lee

and Boutin 2006), in three ways. First, I measure the response patterns of two species, the American marten and black bear, to seismic lines. Both species are sensitive to habitat disturbance, but they respond to disturbance features in opposite ways; martens avoid disturbed areas and forest stands with limited structural complexity (Potvin et al. 2004, Poole et al. 2004, Godbout and Ouellet 2010), whereas black bears use disturbances for travelling and foraging opportunities (Czetwertynski 2007, Mosnier et al. 2008, Switalski and Nelson 2011). Second, I measure whether these species' response to seismic lines is consistent across all encountered seismic lines or is influenced by specific line width and recovery attributes. Third, I measure species response to seismic lines at multiple spatial scales to better understand how responses to encountered lines scale up to larger ecological processes.

Wildlife response to encountered seismic lines is driven by the ecologies of those species in question, as well as the specific width and recovery attributes of the seismic lines encountered. While martens avoided open seismic lines  $\geq 3$  m in width, use of open lines  $\leq 2$  m wide and conventional lines supporting at least some regeneration of woody vegetation showed no difference in use relative to forest interiors. By contrast, black bears used all seismic line types, except for lines  $\leq 2$  m wide where use did not significantly differ from interiors. By comparing observed behavioural changes in use along lines to a forest interior reference category, we show how species use at seismic lines differs from expected use in undisturbed locations (Nielson et al., 2007). This provides a clear, ecologically-derived metric from which to develop regulatory guidelines

and consider management options for seismic lines (Richter et al. 1996). Further, by measuring the response patterns of marten and bear, I provide strong evidence for interpreting which seismic lines are may be considered disturbance features requiring management action and which may not.

Current management decisions regarding seismic lines do not adequately capture the importance and influence of specific line attributes. For example, LIS line types are suggested by governments and widely used by industry to limit the impacts of seismic lines on wildlife (Weclaw and Hudson 2004, OGC 2011, AANDC 2011). However, this research shows that all LIS lines are not capable of mitigating behavioural impacts for wildlife; to achieve that goal, lines, at least for marten and black bear, must be constructed to  $\leq 2$  m in width. Similarly, the recovery of seismic lines is often portrayed as an either-or alternative by industry and government; government regulations typically consider seismic lines permanent disturbance features, whereas industry often assumes lines readily recover after use and abandonment. From the perspective of wildlife seismic line recovery is possible, but is species-dependent. While marten readily re-use lines with even low levels of recovery, use of lines by bears continues well after the recovery of high volumes of woody vegetation. Further, the attributes that appear to contribute to line recovery for marten, overhead and later cover, can be provided by dense shrub growth rather than the robust regeneration of merchantable tree species making recovery metrics dramatically different from those typical in a forestry-based context.

Measuring species response to seismic lines at an ecologically relevant scale for interpreting impacts is important for the development of sound management policies. Although the incorporation of behaviour at the line scale is important for the development of detailed policies, when conservation priorities are focused at the population level, behaviours alone may not sufficiently capture the scale at which seismic lines impact wildlife. The ramification of behavioural responses at the seismic line scale to broader scales was also species dependent. Relative to increasing seismic line density, the likelihood of marten occupancy at the home range scale declines precipitously. Because martens are a territorial species (Powell 1994), I believe this translates to a population reduction (Smith et al., 2007; Moriarty et al., 2011) and a clear scaling between behavioural and population responses. Further, by considering marten response to individual line types when calculating seismic line density the estimate of home range occupancy is improved.

Black bears, however, do not show a similar scaling of response to seismic lines. Although individual seismic lines are used relative to undisturbed forest locations, use of larger spatial habitat patches was driven by the proportion of forest type, rather than cumulative seismic line density. Black bears typically use upland forest (Czetwertynski 2007, Latham et al. 2011) and although habitat disturbance has triggered increased bear use of lowland forest in other parts of North America (Brodeur et al. 2008, Mosnier et al. 2008, Pinard et al. 2012), high densities of seismic lines do not seem to trigger the same behaviour in northwest Canada. Thus, seismic line use by bears may alter short distance movements or

change localized patterns of habitat use, but it does not seem to translate to major shifts in movement or the use of available habitats at broad scales.

This research provides an ecological basis required to contextualize the cause and effect relationships between the competing social, economic, and ecological goals that land use planning seeks to balance. I show how management decisions can be informed by ecologically-relevant metrics (i.e., which seismic lines constitute disturbances) and how conservation actions can be matched to management decisions (i.e., which lines to include in threshold calculations and how to target linear feature densities). However, this work does not provide an ecological basis for determining specific management decisions or measuring the ecological validity of those decisions. As discussed above, effective land use planning matches science to clearly identified conservation objectives based on societal desires and demands. Ultimately those conservation objectives depend on society's choice and may range from preventing any change in species behaviours or probability of habitat use to maintaining a specific rate of species occupancy or occurrence.

The intent of this thesis is not to suggest that seismic lines should be constructed to a specified width, nor is it to suggest line densities are capped at a particular density. Rather, I show how data can be used to weigh conservation objectives against competing land use decisions, and to inform management decisions once they are made that link policies to predictable ecological patterns. Further, I provide some of those requisite data with regard to seismic lines and wildlife. It is critical, however, that for this information to be used effectively,

society must set clear and specific conservation priorities. The accurate interpretation of seismic line impacts on wildlife is species specific and scale dependent. Thus, the application of these data to land use planning challenges is likely dependent of the specific context of the management challenges in question.

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