ASSESSMENT OF BATHURST CARIBOU MOVEMENTS AND DISTRIBUTION IN THE SLAVE GEOLOGICAL PROVINCE

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DEPARTMENT OF RESOURCES, WILDLIFE AND ECONOMIC DEVELOPMENT

GOVERNMENT OF THE NORTHWEST TERRITORIES

YELLOWKNIFE, NT
2004

Manuscript Report No. 158

ABSTRACT

Caribou (*Rangifer tarandus groenlandicus*) from the Bathurst herd come into contact with mineral developments during their annual movements within mainland Northwest Territories and Nunavut. The influence of these developments on caribou movements and habitat use is poorly understood. Here we examine Bathurst caribou distribution during the summer post-calving season relative to three diamond developments. We used satellite collar and aerial survey data to analyze movement data of Bathurst caribou herd relative to vegetation type, phenology and productivity, local weather, insect activity, and proximity to the mine sites. Our goal was to investigate caribou distribution relative to mine sites and explore how weather, vegetation and other factors affect caribou distribution relative to the mines.

We used a number of data sources for our analyses. Caribou location data included satellite data from 6–19 collared Bathurst caribou monitored annually from April 1996 to December 2003, and systematic aerial survey data collected within study areas surrounding the Ekati (1998–2003), Diavik (2002–2003) and Snap Lake (1999–2003) diamond developments. Satellite collar location intervals ranged from weekly during the 1990s, to daily in 2002 and 2003. Aerial surveys generally occurred on a weekly basis at all mines. We examined caribou distribution using vegetation classification data, weather data, indices of insect abundance derived from weather data, a measure of plant productivity (Normalized Difference Vegetation Index; NDVI), and distance to development. AICc model selection methods were generally used in most analyses to determine the models most supported by the data sets.

We used satellite collar data to examine the probability of caribou movements in and out of strata or areas surrounding the mine sites. The mean and minimum distances of caribou from mine sites were relatively similar each year, and caribou were only close to the mine sites briefly for the pre-calving migration and for most of the post-calving through early fall period. We detected a trend of increasing rates of movement of caribou from the vicinity of the Ekati and Diavik mine sites using the multi-strata analyses, but a weak trend of movement into Snap Lake mine buffer area.

We examined how comparable satellite collars and systematic aerial transects were in estimating the distribution of caribou herds relative to the mine sites by examining the relative proportion of collared caribou within the aerial survey study areas compared to the proportion of the entire herd within the survey study areas. We found the estimated proportion of the population in mine areas as estimated by satellite collars was consistently higher than that estimated by aerial surveys on transects. However, there was correspondence between each estimate even though the collar-derived estimate was often higher than transect-derived estimate for the Ekati and combined Ekati/Diavik areas. No relationship was detected for Snap Lake mine site. We suggest that the estimated proportion of the population as estimated by aerial surveys might be an underestimate of caribou occurring within the mine areas for Ekati and Diavik.

We next modeled caribou habitat selection using vegetation data, NDVI, and insect activity indices as covariates, and examined the influence of distance from mine

developments on caribou habitat selection using first satellite collar data, then data from the aerial surveys. We used resource selection functions to assess habitat selection of caribou and the effects of mine sites on caribou distribution. Results were evaluated using AICc model selection methods; we also assessed overall fit and the predictive ability of the most supported models. Results demonstrated that caribou selection of habitat appeared to be affected by distance from mine site development. The large-scale (weekly) analysis of caribou satellite collar locations suggested an influence of 50–65 km from mine sites, although this influence was not strong. Although hampered by comparatively low sample sizes, analysis of fine-scale (daily) satellite data suggested a smaller influence distance, in the range of 20–25 km from mine development. The aerial survey data suggested a measurable influence of mine sites on probability of caribou occurrence for the Ekati study area that increased with time out to about 20 km. The combined Ekati/Diavik data yielded a slightly weaker model that also suggested an influence of distance from mine site. The Snap Lake data set yielded the weakest model, which showed a weak and decreasing influence of mine site on probability of caribou occurrence.

Our analyses suggest a trend of increasing rates of caribou movement from the vicinity of the Ekati and Diavik mine sites, and selection of habitat by caribou at further distances from these mine sites over time. Trends in the influence of mine development on caribou distribution and habitat use in Snap Lake area were weaker. Future analyses and monitoring of the influence of mine development on caribou distribution would be enhanced by larger sample sizes of collared caribou with frequent (daily) location intervals. The aerial survey data provided a complimentary method to model caribou distribution. Survey data would be made more robust by increasing location accuracy of observations, and by using line transect distance sampling to help estimate sightability of caribou within study areas. We suggest that both satellite collar deployment and aerial surveys be continued for the near future to utilize two independent methods to monitor the influence of mineral developments on movements of the Bathurst caribou herd.

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INTRODUCTION

Barren-ground caribou (*Rangifer tarandus groenlandicus*) are a keystone species of the Canadian North, and are of significant subsistence and cultural importance to aboriginal peoples. Two caribou herds primarily occur within the Slave Geological Province (SGP), an area that extends from Great Slave Lake north and northeast to the arctic coast on both sides of Bathurst Inlet:

- Bathurst herd – estimated at 349,000 ± 95,000 (SE) animals in 1996 (Gunn et al. 1997) and 186,000 ± 28,000 animals in 2003 (Gunn et al. In prep.).
- Ahiak (Queen Maud Gulf) herd – estimated at roughly 200,000 animals in 1996 (Gunn et al. 2000).

The Ahiak herd has only recently been confirmed as a separate herd (Gunn and D'Hont 2002), and while generally moving east of the Bathurst range, may overlap with the migration corridor and winter ranges of the Bathurst herd. However, few caribou from this herd have been collared, thus we restrict this analysis to movements of Bathurst caribou.

Since the early 1990s, considerable mineral development activity has taken place on the barrenlands in the SGP. These developments are primarily associated with diamondiferous kimberlite deposits, and have resulted in unprecedented levels of mineral exploration and development occurring in an area where little activity has occurred previously. Currently, two diamond mines are in production near Lac de Gras (Ekati began production in 1998 and Diavik in 2003), and two others are have had activity associated with bulk exploration but are pre-construction for underground mining (Snap Lake and Tahera Jericho) (Fig. 1). Exploration within this area continues (Johnson and Boyce 2004). Additional human activity in this region is confined primarily to seasonal fishing and hunting camps (Johnson and Boyce 2004).

The intense level of mineral exploration in the early 1990s led to accelerated collection of baseline data, which included data on caribou movements since 1996 using satellite-collared cows (Gunn et al. 2001, Gunn and D'Hont 2002). As well, environmental assessments and monitoring related to diamond mine activity in this area have generated descriptions of caribou relative abundance and distribution mostly based on aerial surveys.

Analyses of the locations of the satellite-collared cows 1996–2000 suggested inconsistent trends in the probability of caribou encountering mine sites during the post-calving and summer seasons, and in some years, a weak increasing trend for caribou movement out of a 50-km buffer zone surrounding the three diamond development sites (Ekati, Diavik and Snap Lake; Gunn et al. 2002). Applying different analytical methods but also using the same satellite collar dataset, Johnson and Boyce (2004) concluded that during the post-calving and summer season (to 31 Aug), caribou showed weak selection for vegetative resources, and weak avoidance of major developments at distances less than 33 km. BHP Billiton (2004) found a significant but weak relationship ($r^2 = 0.13$, $P < 0.01$) between the distribution of satellite-collared caribou from the Bathurst herd (mean distance from the Ekati study area boundary) and the number of caribou observed within the Ekati study area. Additional analysis suggested that the distribution of caribou did not change with distance from the mine within the Ekati study area (BHP Billiton 2004).

Aboriginal elders are reporting that the Bathurst herd is shifting its seasonal migrations and have suggested that the shift is due to avoidance of the operational diamond mines (Ekati and Diavik). These concerns were raised during environmental assessment hearings and led the Mackenzie Valley Environmental Impact Review Board in the 2003 report on De Beers Snap Lake diamond mine to recommend that: “The GNWT shall, within 24 to 36 months, develop a model that detects and evaluates the effects of development on caribou movements and populations in the Slave Geological Province.”

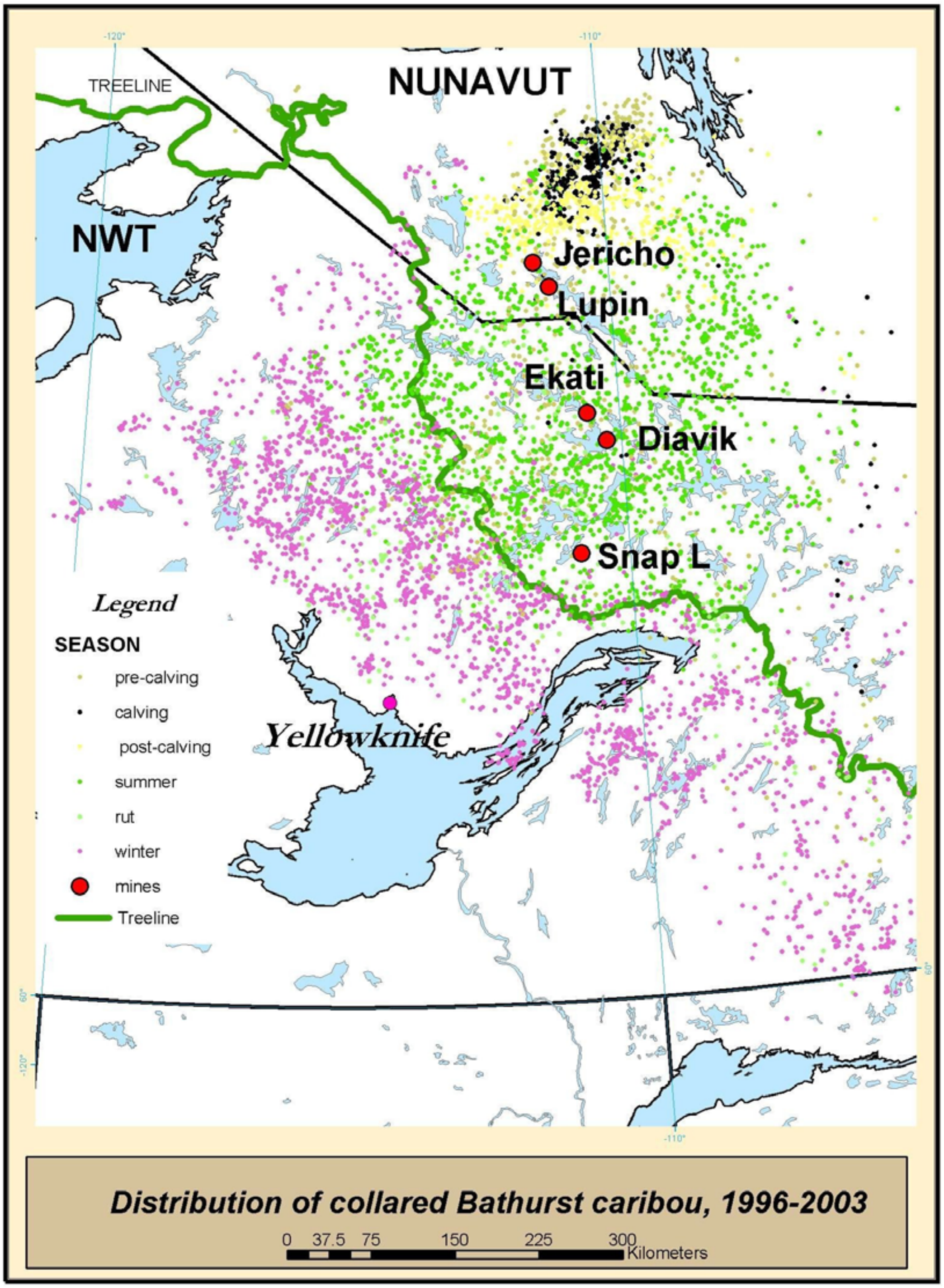


Figure 1. Seasonal distribution of satellite-collared Bathurst caribou in relation to current and proposed mineral development within the Slave Geological Province, 1996–2003.

In response to the aboriginal elders concerns and the Mackenzie Valley Environmental Impact Review Board recommendation, we initiated a study to analyze movement data of Bathurst caribou herd. As well as describing caribou movements and distribution, we needed to also consider factors that could affect movements (insect harassment and the search for forage, for example). We designed the analysis to use available data, including satellite collar locations and the diamond mine's monitoring of caribou relative to vegetation types, plant phenology and productivity, local weather, insect activity, and proximity to the mine sites. We conducted analyses at local and regional spatial scales, and examined, among other things, correlations between data provided by the satellite collars and aerial surveys conducted in the vicinity of the mine sites. We present the overall objectives and clarify data sources, study area, and seasons of interest. We then present individual analyses, and summarize our findings in a wrap-up discussion.

OBJECTIVES

The goal of this study was to investigate caribou distribution relative to mine sites and explore how weather, vegetation and other factors affect caribou distribution.

Overall objectives for this study were:

1. Use mathematical models to assess trends in fidelity of caribou to areas surrounding mine sites.
2. Conduct a descriptive analysis of caribou distances from mine site as determined by satellite collars and aerial survey data.
3. Use resource selection functions (RSF) to determine if vegetation, distance from mine site, or other variables (e.g., weather) are suitable predictors of caribou distribution.
4. Contrast distribution of caribou from satellite collars versus aerial surveys conducted in the vicinity of mine sites.
5. Produce a database and commentary on caribou annual and seasonal distribution at the local and regional scale, with commentary on the reliability, comparability and relevance of each data source.

Survey databases were amalgamated into a merged meta-database in a spreadsheet that can be imported into ArcView or similar GIS program) indexed to maps.

STUDY AREA

We primarily restrict our analysis to caribou movements relative to the two existing diamond mine developments (Ekati and Diavik) and the Snap Lake project, which has been in the exploration and permitting phase since 1999 (Fig. 1). Caribou can come into contact with these three developments during the northern migration to calving grounds near Bathurst Inlet, and during subsequent post-calving through fall seasons (Gunn et al. 2001). The main Ekati mine site and Diavik are 30 km apart, but a 29 km all-weather road runs between the main Ekati site and the Misery camp and pit, which is located 7 km from the Diavik mine site. Snap Lake is a further 105 km south of Diavik. These three projects are the largest developments in this area, with the exception of the Lupin gold mine on Contwoyto Lake, Nunavut. The Tahera Jericho project on the north end of Contwotyo Lake is at a preconstruction phase.

Bathurst caribou move hundreds of kilometres during their seasonal migrations (Fig. 1). For the most part, our analyses were conducted at two scales, regional and local. The regional scale was generally confined to a 200-km radius around the mines for practical reasons of obtaining digital databases. We reasoned that a regional study area of this size would undoubtedly encompass the area within which any potential regional scale influences of the mine sites on caribou movements and distribution could be detected and examined. The local scale was generally defined as the aerial transect study areas associated with each mine site. These areas encompassed 1,200–3,000 km^2 around each of the three main diamond mines (Appendix 1).

SEASONS OF INTEREST

Seasonal movements of the Bathurst herd and annual variation in these movements (Gunn et al. 2001) result in large fluctuations in the number of caribou in the vicinity of the mines each year. Very few to no caribou are found in the vicinity of the mines during winter (Fig. 1). Relative to the mines, caribou tend to move through rapidly and in a synchronous fashion during their northward spring migration to the calving grounds with less variation on the timing and route than during the post-calving migration (Gunn et al. 2001, BHP Billiton 2004). Variation in the route of the northern migration appears to be related primarily to annual variation in the location of wintering herds. Few caribou are found in the vicinity of the mines during calving and immediately post calving. During the post-calving period caribou tend to return rapidly in a south and southeast direction, and then disperse for the summer; movement rates decline during this period (Gunn et al. 2001). Movement out of the vicinity of the mines occurs after the fall rut, and by October, few caribou occur in the area. It is during the post-calving through summer periods when movement rates are reduced that the potential influence of the mines would be expected to be the greatest. We have therefore restricted our analyses to primarily consider the post-calving (16–31 June) through summer (1 July to 15 October) seasons to the beginning of the fall rut. Depending upon the analysis being conducted, data from portions of this period may be used.

DATA SOURCES

The following data were examined:

Caribou data

Satellite data from 6–19 collared Bathurst caribou monitored annually from April 1996 to December 2003 have been shown in numerous figures in annual reports from the mines and RWED publications (Gunn et al. 2001, Golder Associates Ltd 2003, BHP Billiton 2004, DDMI 2004). Duty cycle varied during the study from every 7 days to every 1 day, and became more frequent during latter years (Gunn et al. 2001; A. Gunn, unpublished data). Where multiple locations were obtained for an individual caribou each day, the best location each day was used as classified by on-board collar software (Table 1). Locations with class code 0 were not used in the analysis.

Data on caribou movements and distribution were collected as baseline data for the environmental assessments. As mine construction started, further data on caribou movements and distribution were collected during monitoring (Table 2). Methodology, transect spacing and width, data recording, study area size, and frequency of data collection varied within and among mines (Table 2; Appendix 1). Transect routes did not always remain consistent over time. Data sources were widely scattered and were often difficult to obtain. We were unable to obtain original digital data from some of the baseline work, and in some cases baseline and monitoring data were not spatially referenced. Most data obtained were original GPS location files, but in an attempt to utilize more descriptive baseline data we did digitize non-systematic observations from maps of caribou distribution in baseline studies from Diavik (Appendix 2). A merged database and a file of flight dates for all projects associated with the Bathurst range are included in this report.

Several of the analyses we conducted required use of systematically collected data, i.e., to determine where caribou were and were not at a given time period. For example, the Diavik project collected substantial and detailed descriptive baseline data, especially in 1996 and 1997, which was only available to us on paper maps (Penner and Associates Ltd. 1998a). We attempted to utilize data from July 1996 to July 1997 by over-laying aerial transects used for the combined Ekati/Diavik monitoring on top of the mapped locations. However, we abandoned this effort because 1) the range of dates of caribou movements each map provided was too coarse (mean = 12 days, range 5–30 days, n = 8 maps) to integrate into the finer-scale temporal digital data, 2) the arrows of movement on each map could not always be narrowed down to shorter time periods, and 3) the broad nature of the movement arrows and polygons of observed groups meant that spatial accuracy was not the same as data from subsequent monitoring.

Table 1. Satellite collar class codes.

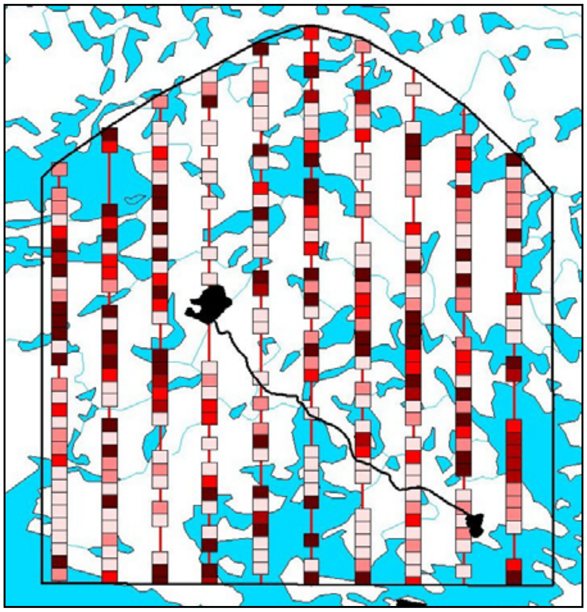
Class	Location accuracy
3	<150 m
2	150–350 m
1	350–1,000 m
0	>1,000 m

Table 2. Summary of caribou distribution data collected by mineral and associated developments in the Slave Geological Province. Most of the baseline and monitoring work was conducted between April and October, when caribou were found in the vicinity of the developments. More details are provided in Appendix 1.

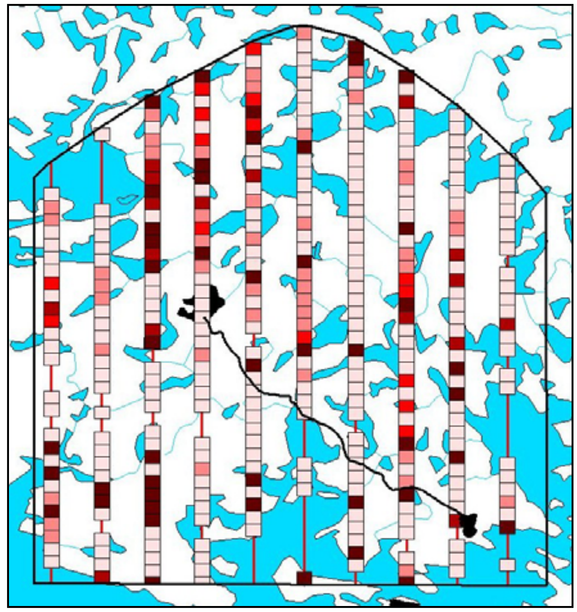
Project	Caribou baseline data	Caribou monitoring data
Ekati (BHP Billiton)	1994–1995: Systematic transects; 1996: Reconnaissance	1997–2003: Systematic transects; GPS locations not obtained in 1997
Diavik (Diavik Diamonds Mines Inc.)	1995–1997: Transects, then extensive reconnaissance flight patterns	1998–2001: Reconnaissance surveys only, primarily limited to the East Island; no GPS location data 2002–2003: Systematic transects, linked with Ekati flights
Snap Lake (De Beers Canada Mining Inc.)	1999–2000: Systematic transects during Apr–May and July–Sept	2001–2003: Systematic transects during Apr/May–Sept/Oct
Bathurst Inlet Port and Road (Bathurst Inlet Port and Road Project)	2001–2002: Transects (3 designs) between July 2001 and July 2002	

The distribution of caribou observed during systematic surveys by the three diamond mines has been shown in annual monitoring reports (e.g., Golder Associates Ltd. 2003, BHP Billiton 2004, DDMI 2004). We pooled data among years to show the relative density of caribou for the northern migration (up to 30 June) and the post-calving and summer seasons (after 30 June) for each mine (Figs. 2 and 3). Transect lines were divided into 1-km divisions with 600 m strip width on each side of the aircraft, forming 1.2 km^2 segments. Caribou groups were associated with each segment to determine an average density for that segment. Data were averaged by the number of years covered, but the point is not to show trends over time but simply the relative distribution and spatial pattern of caribou within each study area over the time for the northern and southern migrations. Numbers of caribou present and frequency of surveys would affect among-year differences in densities.

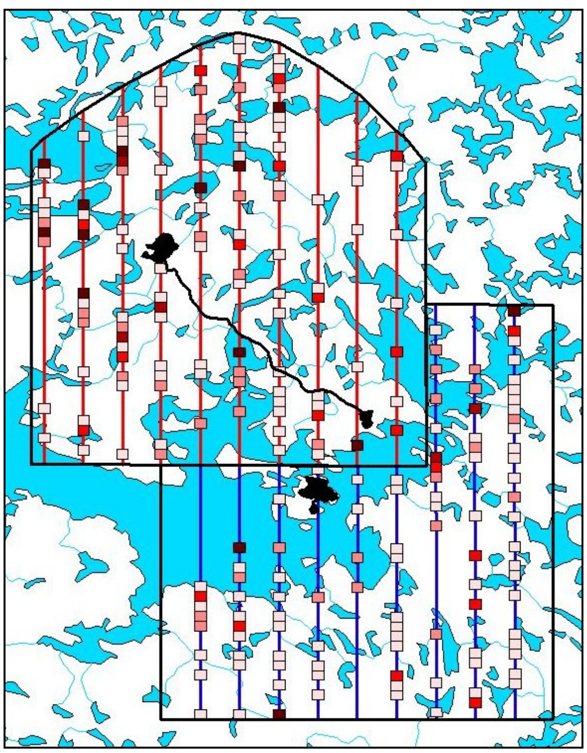
Ekati northern migration 1998–2001



Ekati post-calving migration 1998–2001



Ekati/Diavik northern migration 2002–2003



Ekati/Diavik post-calving migration 2002–2003

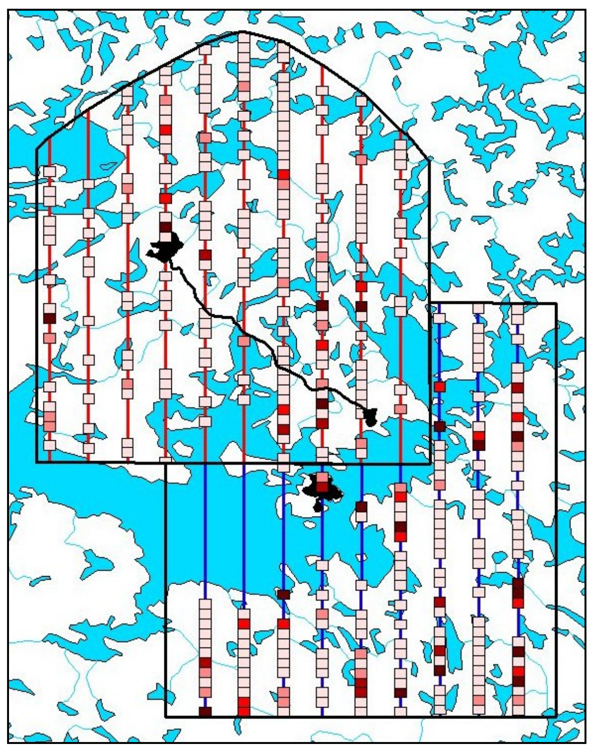


Figure 2. Density and distribution of caribou (number of caribou/km^2/year) observed during systematic aerial surveys within the Ekati and Diavik study areas. Transects (red and blue lines) are spaced at 4-km intervals. Ekati, Diavik and Misery mine sites and Misery Road are shown. See Fig. 3 for legend.

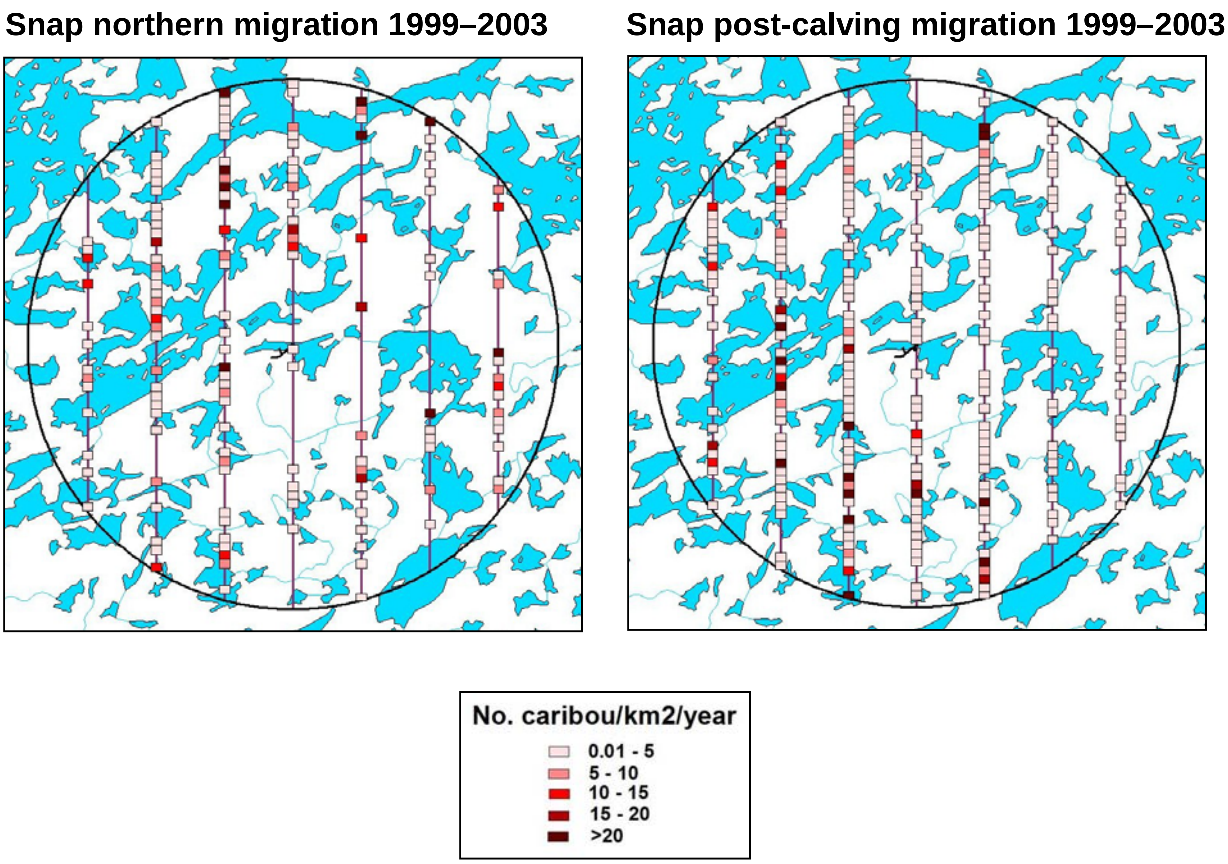


Figure 3. Density and distribution of caribou (number of caribou/km^2/year) observed during systematic aerial surveys within the Snap Lake study area, 1999–2003. Transects are spaced at 8-km intervals. The Snap Lake mine site is in the centre of the circular study area.

Vegetation maps

Matthews et al. (2001) developed a supervised classification of Landsat TM data to produce a vegetation classification map for much of the SGP region. The southern boundary of this classification cuts through the Snap Lake mine site. Although both Ekati and Diavik conducted their own vegetation assessments during baseline studies, both these mines currently use the Matthews et al. (2001) classification (BHP Billiton 2004, D. Panayi, Golder Associates, personal communication).

Snap Lake produced a vegetation classification within a regional study area 80 km (north-south) by 73 km (east-west), centred on the mine site (De Beers 2002). This classification closely matched the Matthews et al. (2001) classification (also known as the West Kitikmeot/Slave Study [WKSS] classification or RWED's Remote Sensing vegetation classification).

Weather data

Weather data were obtained from Ekati (Oct 1996, and Feb 1997–Dec 2003), Diavik (Oct 1997–May 2003), and Snap Lake (Jan 1998–Dec 2003). Problems with Ekati's weather collection in 2000 resulted in use of the Diavik dataset for that year (BHP Billiton 2004).

Plant productivity (NDVI)

Estimates of percent snow cover in the Ekati study area during spring 1997–2003 indicate that the timing of snowmelt can vary by up to one month (BHP Billiton 2004). Such differences among years would influence plant phenology and productivity, which could influence caribou use of habitats and movement patterns (Skogland 1980, Russell et al. 1993, Van der Wal 2000). We used Normalized Difference Vegetation Index (NDVI) imagery to track plant phenology and productivity within the study area. NDVI is related to the proportion of photosynthetically absorbed radiation, and is calculated from atmospherically corrected reflectance from the visible and near infrared channels from Advanced Very High Resolution Radiometer (AVHRR) flown on NOAA-series satellites. NDVI estimates the relative total green biomass, but not measures of lichen biomass. Annual changes in June NDVI values have been correlated with location of extent of calving and calf survival (Russell et al. 2002). We used 8-km resolution NDVI (NOAA 14 data) amalgamated by 10-day composite periods to examine caribou movements at the broad scale from May to September 1996 to 2001 (within a roughly 200 km radius around the three main mines). We also obtained 8-km resolution data for 2002 from 1-km resolution NDVI (NOAA 16) data amalgamated into 10-day composites. Data from the 21 July 2002 composite were removed from analysis because of unusually high values in many of the pixels. Mean NDVI values from 2002 were considerably lower than values from 1996–2001, and were likely related to poor calibration of post-2001 data and use of different satellite platforms, limiting comparison between data sets (Brad Griffith, University of Alaska, Fairbanks, personal communication). However, trends in relative green-up within years were likely still valid. NDVI data from 2003 were beyond the limitations of our budget. Details on the process used to develop the NDVI coverage are provided in Appendix 3.

We examined trends in plant productivity and green-up within and among years by selecting a subset of the 8-km resolution NDVI data from a 200 km (east-west) by 264 km (north-south) area centred on the typical summer range used by the Bathurst herd (822 NDVI cells; 3 entirely over water removed). This area encompassed the Ekati and Diavik mines and included to south of Snap Lake. All NDVI values <0 were given a value of 0; these are generally snow or ice covered. NDVI values on summer range increase rapidly between late May and late June in most years, although the onset of green-up occurred much earlier in 1998 (Fig. 4). Snowmelt during 1998 was also much earlier than normal (BHP Billiton 2004). NDVI values generally peak in mid to late July, with absolute values variable among years. The increase in NDVI values in September 2001 and associated high variance is suspect and may be unreliable (Thomas Naughten, Parks Canada, personal communication; Brad Griffith, University of Alaska, Fairbanks, personal communication).

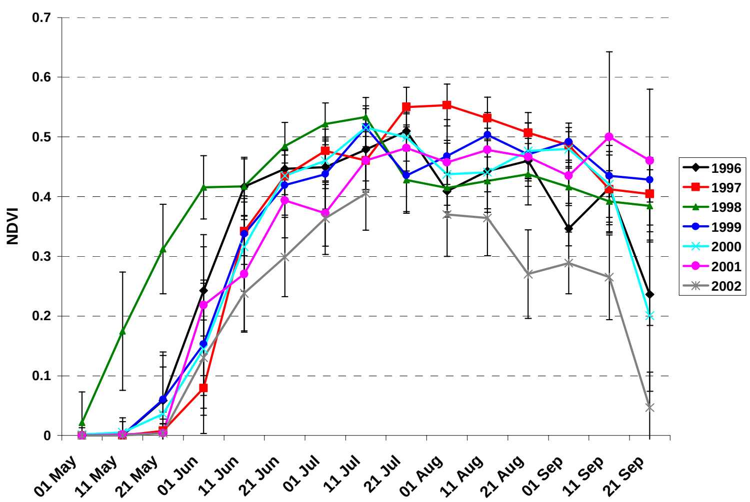


Figure 4. Trend in NDVI (± SD) within and among years for the Bathurst caribou herd summer range, 1996 to 2002. NDVI values based on 10-day composites from 8-km resolution data, except for 2002, which were obtained from 1 km resolution data modified to 8-km resolution (Appendix 3).

RELIABILITY, COMPARABILITY AND RELEVANCE OF DATA SOURCES

It was difficult to amalgamate all caribou baseline and monitoring data collected by the three largest diamond mine developments. Cooperation from the mines and especially their environmental consultants was good, but we found that some of the older data, especially data emanating from the baseline reports, could not be located or provided, or were either not collected or not available in a digital format. Lack of digital data hampered use of these data at the broad scale.

During both baseline and monitoring programs for all mines, data collection design changed often. The different flight patterns likely resulted from changing objective during the studies. Flights were either systematic on fixed transects (which sometimes changed between baseline and monitoring), or less rigid on flight path (e.g., flew the perimeter of the study area looking for tracks, then flew to count and map larger groups seen). Search width during flights on transects varied from unbounded to 600 m on each side of the aircraft. For the purpose of comparing survey data with satellite collar data, the most useful data collected by the mines were from systematic surveys on fixed transects. Surveys conducted non-systematically both temporally and spatially were useful for describing seasonal movements of caribou through the mine areas, but were of limited quantifiable value in this analysis.

To our understanding, GPS locations of caribou groups observed on transect during aerial surveys conducted by all three main diamond mines were not corrected for distance and direction to the caribou group from the aircraft where the GPS location was taken. This means that the point associated with the caribou group may be up to 600 to 1,000 m off in either direction from the actual group location (assuming the GPS location was taken when the caribou were perpendicular to the transect line). This has implications regarding most analyses requiring geo-referenced data (e.g., habitat or satellite data from digital databases), and also influences the resolution of analysis for distance to mine development.

SECTION 1: Distance, Distribution, and Movement Of Caribou Relative to Mine Sites for the Bathurst Caribou Herd 1996–2003

Introduction

This portion of the analysis was to summarize the distance and distribution of collared caribou relative to the Ekati, Diavik, and Snap Lake developments. The objectives for the analysis are:

1. Describe mean and minimum distances of individuals from mine sites as a function of season for 1996–2003.
2. Describe caribou distribution relative to mine sites for the summer season (here defined as July 14 to October 14).
3. For the summer season, determine the probability of caribou moving within encounter range of mine sites, and determine if there are temporal trends in probability of encounter of mine sites.

Methods

Description of mean distance of individuals from mine sites

The mean distance of each collared caribou from the Ekati, Diavik, and Snap Lake mine sites was estimated for all years and all seasons. In addition, the distance of the closest individual from each mine site was also determined. These data were summarized graphically to determine the dates in which most individuals were in the vicinity of mine sites.

Description of sample sizes of collared caribou relative to mine sites

The number of caribou locations that were within successive distance intervals from the mine sites were tallied to determine the relative sample size of points available within the proximity of mine sites. This information was used to evaluate sample sizes of various potential geographic strata for the multi-strata analysis (described next).

Trends in movement in the vicinity of mine sites

Our question is whether the probability of an individual caribou approaching the mine site changed for seasons in which the herd was in the vicinity of mine sites over the course of the study. The successive locations of an individual caribou are autocorrelated and form a time series for each week of collar data. In addition, the movements of animals within the Bathurst herd were not likely to be entirely independent, although the herd was reasonably dispersed during the post-calving and summer periods and therefore it could be assumed that individuals were somewhat independent. Regardless, pooling locations of individuals for analyses would most likely constitute pseudoreplication, therefore potentially biasing variance estimates of hypothesis tests (Otis and White 1999).

The problem of autocorrelation of GPS locations and non-independence of locations was confronted using multi-strata models (Hestbeck et al. 1991; Brownie et al. 1993) in program MARK (White and Burnham 1999). The multi-strata model is a generalization of the Jolly Seber mark-recapture model. The multi-strata model approach avoids the issue of autocorrelation as the individual caribou is considered the sample unit, rather than other approaches which pool individual animal location.

The multi-strata modeling approach involves defining different geographic strata, such as areas within and beyond a buffer radius of the mine site. The weekly caribou locations for individuals are then categorized as being within or outside of a given strata. The probability of movement of an individual into or out of the strata is then estimated using these records (Powell et al. 2000). For this exercise, an area within 50 km of a mine site was declared as one stratum, and the area outside the 50 km zone as the other stratum. The 50 km radius zone was chosen to allow suitable sample sizes for caribou points within the 50 km radius zone. In addition, the weekly movement distance for caribou was approximately 53 km (SD = 40.6 km) (Gunn et al. 2001) and therefore a 50 km radius corresponded to the measured weekly scale of movements. Data from the summer season only (July 14 to October 14) for 1996 to 2003 were used, as July 14 is the date analysis suggests that caribou have arrived in the vicinity of the mines.

Potential avoidance or response to the mine site can be explored by observing yearly differences in probabilities of movement into or out of the mine sites. The probability of movement into the 50 km zone is simply the probability that a caribou will move within 50 km of the mine site for any given week during the period of the analysis. The probability of movement out of the 50 km zone is simply the probability that a caribou will leave the 50 km zone once it is in the zone. For example, an increase in probability of movement out of the 50 km buffer zone might suggest avoidance of the buffer area. Because the 50 km zone is small relative to the overall range of the caribou, the probability of entry is low, and therefore this analysis allows us to examine trends in probability, and cannot be interpreted absolutely.

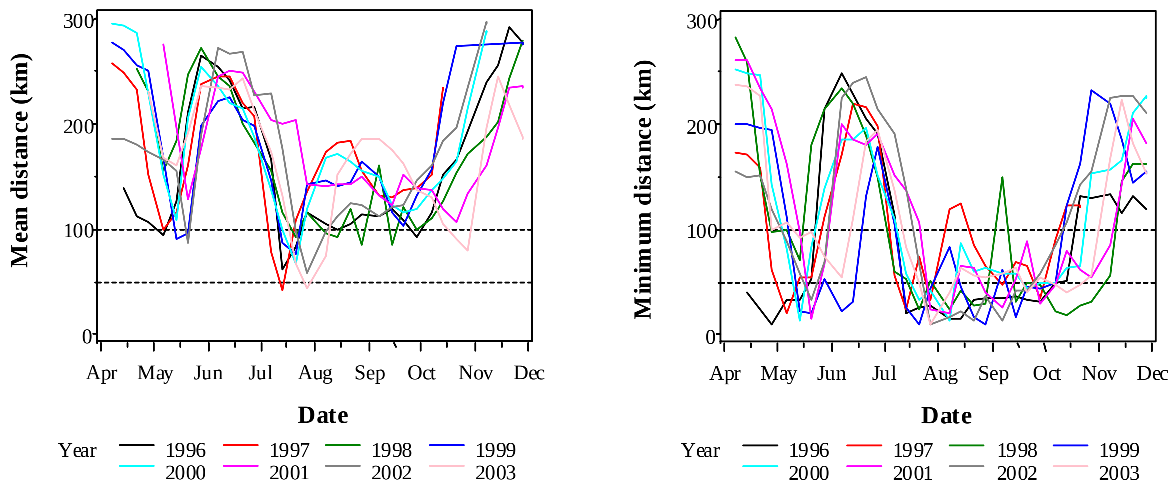
Multi-strata models were built which estimated the probability of movement of caribou between these two strata for each week the collars reported. The weekly estimates were averaged to provide estimates for the summer season of movement between the strata for each year. The multi-strata model also estimates survival that was fixed at 1 for the analysis. Recapture rate of caribou, the probability that a collared caribou returned a location each week, was estimated to account for missing weekly satellite collar locations for some caribou. Recapture rate was was pooled for all years since it was unlikely to vary temporally. Models were built which allowed year-specific movement probability between strata, linear trends between strata, and a constant rate of movement between strata. Results were evaluated using AICc model selection methods (which indexes the model most supported by a data set). Models with the lowest AICc scores were considered further (Burnham and Anderson 1998). Models that differed by less than 2 AICc units from the most supported models (as indicated by ΔAICc scores of less than 2) were also considered. Program MARK (White and Burnham 1999) was used for all modeling and estimates.

Results

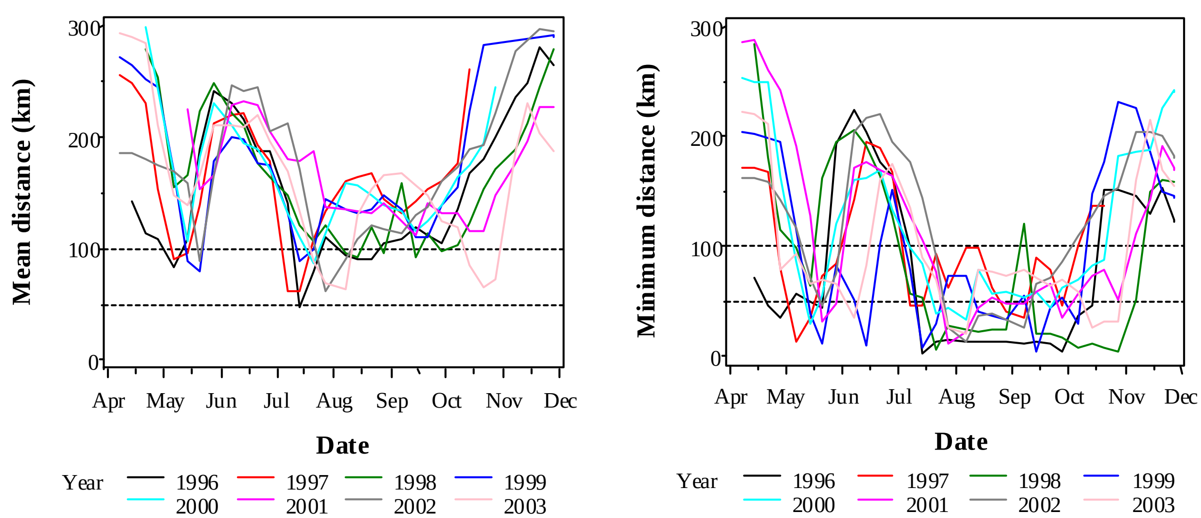
Description of mean individual distances of individuals from mine sites

Collared caribou were greater than 200 km from the mine sites for the winter, calving, and fall seasons. They passed within 50–100 km of the mine sites during the pre-calving migration, and were in the vicinity of the mine sites for the post calving and summer seasons for all years of the study (Fig. 5). Inspection of mean and minimum distances of caribou from the mine sites shows a relative degree of synchrony for all years of the study. Distances and trends in distance among mine sites were similar given their relatively close proximity, although because of the location of Snap Lake further south, caribou during summer were on average further and during fall were closer from the mine site compared with the other two mines. The data from 1998 were sparse due to poor satellite locations (Gunn et al. 2001), and therefore this year was not considered further in the analysis.

Ekati



Diavik



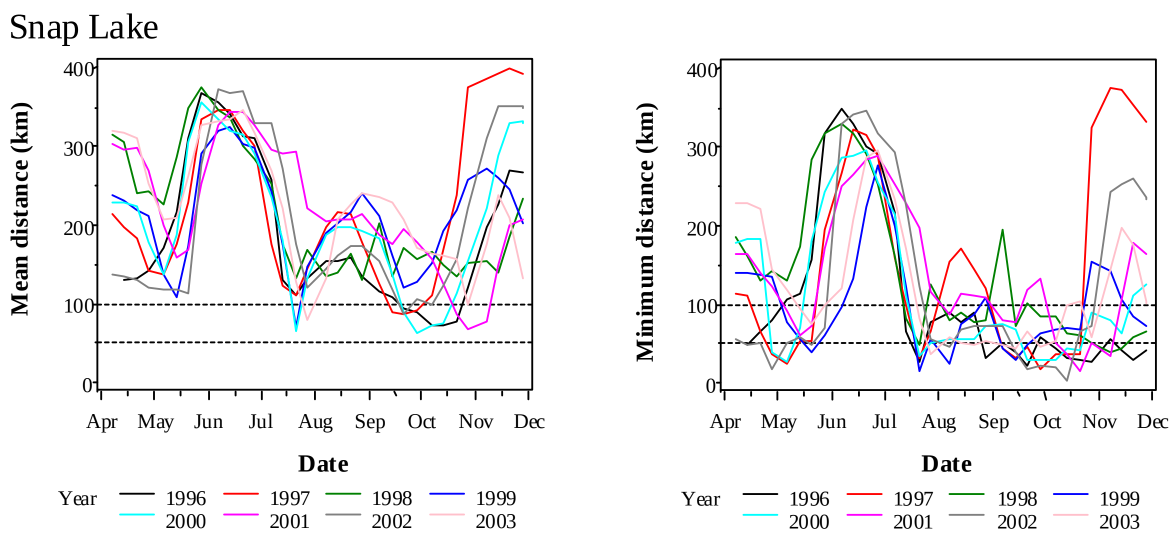


Figure 5. Mean and minimum distance for satellite-collared caribou from Ekati, Diavik and Snap Lake mine sites, NWT 1996-2003

Sample sizes of collared caribou relative to mine sites

Sample sizes of caribou relative to mine sites were assessed by estimating the mean number of locations that occurred within specified distances from the mine sites across all years of the analysis (Fig. 6). For example, on average there were two locations that were >10 km and four locations that were within 10–20 km of from the Diavik mine site for any given year of the analysis. As discussed later, low sample sizes of caribou have implications in terms of multi-strata and other analyses.

Trends in movement in the vicinity of mine sites

Thirteen weekly periods were used for the multi-strata analysis. Sample sizes were 10, 8, 14, 13,13, 11, and 10 collared caribou for 1996 to 2003, respectively (1998 excluded). Note that sample sizes are the total number of unique caribou that were collared for a given year.

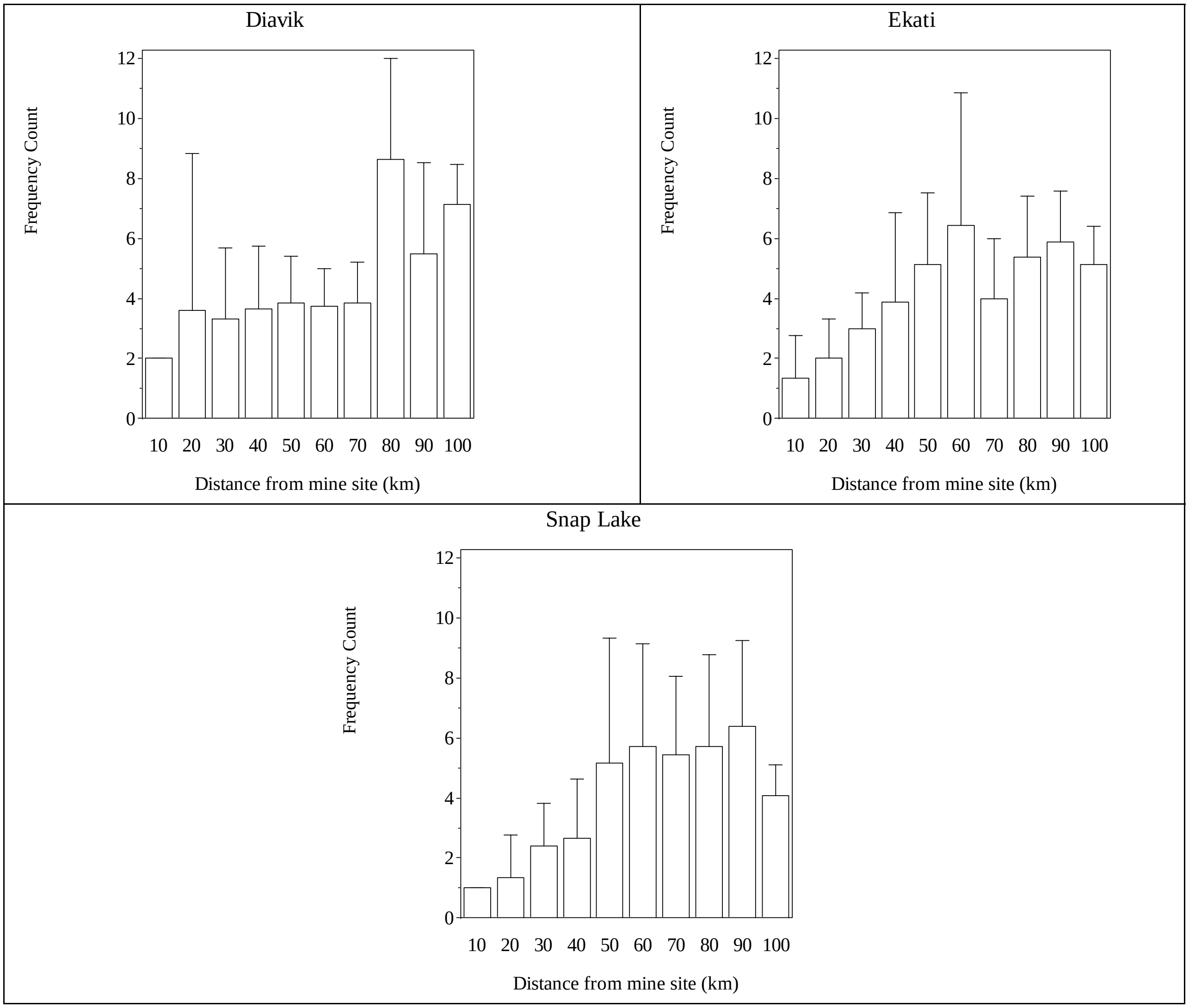


Figure 6. Mean number of collared caribou locations within specified distance intervals from mine sites. Sample sizes correspond to the periods from July 14 to October 14 for each year of the analysis. Locations >100 km from mine sites are not displayed.

Diavik

The most supported model suggested that probabilities of movement out of the buffer zone were constant and movement probabilities into the buffer zone were year-specific. The second most supported model suggested that there was a linear trend in movement out of the buffer zone but movements into the mine site was year specific (Table 3). A linear trend model assumes that the change in movement probability is the same for each year of the analysis so that a linear trend line can be fit through the data. These models were theoretically tied in terms of support because the difference in AICc scores was less than 2 (ΔAIC). Model averaged estimates of movement probabilities out of the buffer zone suggest they did increase over the duration of the study. However, this trend is relatively weak as shown by the large confidence intervals (Fig. 7).

Table 3. AIC model selection results from the Diavik mine analysis.

Movement		Model Selection Results			
Out	**In**	**AICc**	**ΔAICc**	**AICc Weights**	**k**
Constant	Year	832.7	0.00	0.617	9
Linear trend	Year	833.9	1.18	0.342	10
Year	Year	839.5	6.80	0.021	16
Constant	Year	841.1	8.36	0.009	5
Linear trend	Linear trend	842.3	9.56	0.005	6
Constant	Constant	843.4	10.69	0.003	4
Linear trend	Constant	844.7	11.94	0.002	5
Year	Linear trend	845.8	13.12	0.001	11
Year	Constant	848.2	15.45	0.000	10

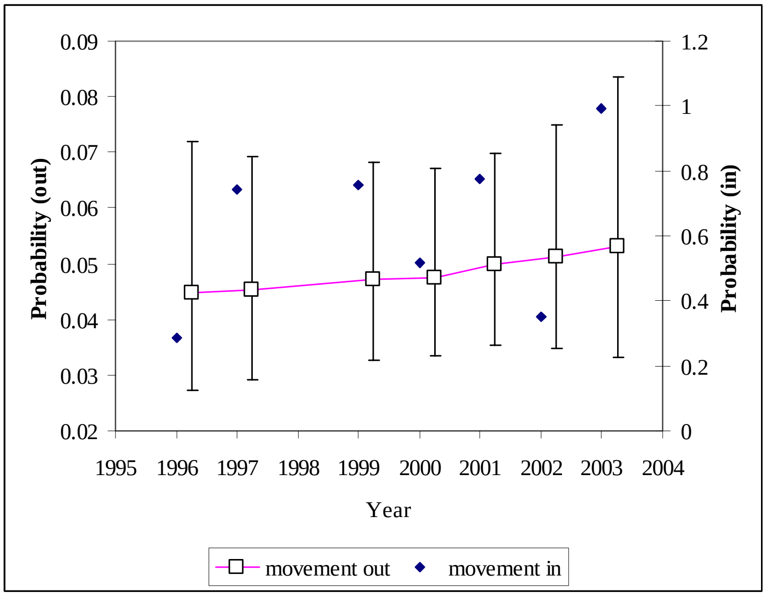


Figure 7. Estimates of movement probabilities of Bathurst caribou in and out of a 50 km buffer zone around the Diavik mine site. Confidence intervals are shown for movement probabilities out of the buffer zone since these estimates are the most meaningful in terms of analysis results.

Ekati

The most supported model suggested that there was a linear trend in movements out of the 50 km buffer zone and movements into the buffer zone were year-specific (Table 4). A model with constant movement probabilities out of the buffer zone was less supported, as suggested by ΔAICc values of >2. Inspection of model averaged estimates from the linear trend model revealed an increase in movements out of the 50 km zone for the duration of the study with a large degree of variability in movement into the buffer zone (Fig. 8). This suggests that caribou that came within the 50 km zone were more likely to move out within one weekly period in latter years of the study.

Table 4. AIC model selection results from the Ekati mine analysis.

Movement		**Model Selection Results**			
Out	**In**	**AICc**	**ΔAICc**	**AICc Weights**	**k**
Linear trend	Year	869.8	0.00	0.734	11
Constant	Year	872.3	2.50	0.210	10
Year	Year	874.9	5.15	0.056	16
Linear trend	Constant	885.1	15.36	0.000	5
Linear trend	Linear trend	887.1	17.29	0.000	6
Constant	Constant	887.5	17.77	0.000	4
Constant	Year	889.5	19.71	0.000	5
Year	Constant	890.3	20.48	0.000	10
Year	Linear trend	892.2	22.43	0.000	11

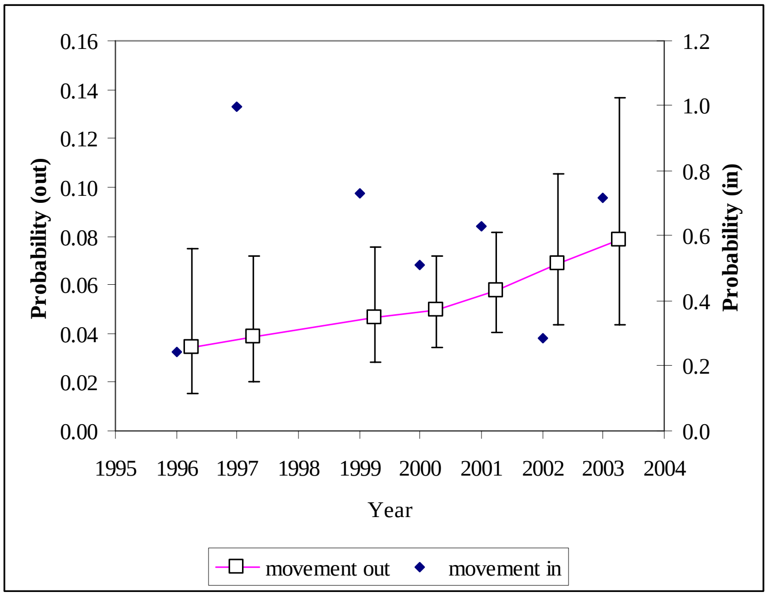


Figure 8. Estimates of movement probabilities of Bathurst caribou in and out of a 50 km buffer zone around the Ekati mine site. Confidence intervals are shown for movement probabilities out of the buffer zone since these estimates are the most meaningful in terms of analysis results.

Snap Lake

The most supported model suggested movements out of the 50 km buffer zone were year-specific and movements into the 50 km zone were constant (Table 5). The second most supported model suggested a year-specific trend in movement out and a linear trend in rates for movement into the 50 km radius zone. Both of these models were supported as indicated by ΔAICc scores of less than 2. Inspection of model averaged estimates suggests a slight increase in movements into the buffer zone with no apparent trend in movements out of the buffer zone (Fig. 9).

Table 5. AIC model selection results from the Snap Lake analysis

Movement		**Model Selection Results**			
Out	**In**	**AICc**	**ΔAICc**	**AICc Weights**	**k**
Year	Constant	874.6	0.00	0.718	10
Year	Linear trend	876.6	2.03	0.260	11
Year	Year	881.8	7.21	0.020	16
Constant	Constant	888.1	13.46	0.001	4
Linear trend	Constant	888.7	14.14	0.001	5
Constant	Year	890.0	15.45	0.000	5
Linear trend	Linear trend	890.7	16.14	0.000	6
Constant	Year	892.9	18.35	0.000	9

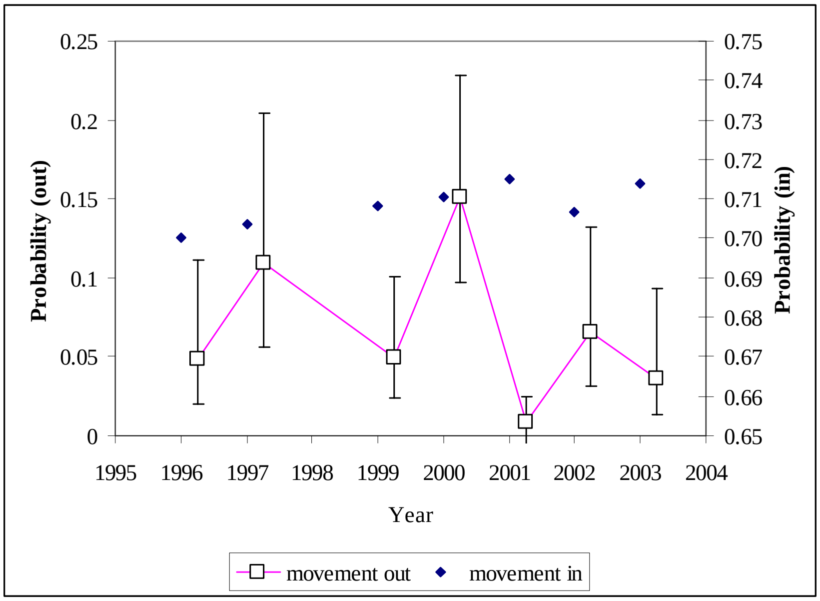


Figure 9. Estimates of movement probabilities of Bathurst caribou in and out of a 50 km buffer zone around the Snap Lake mine site. Confidence intervals are shown for movement probabilities out of the buffer zone since these estimates are the most meaningful in terms of analysis results.

Discussion

This analysis is the first step in what will be a substantive analysis of caribou distribution relative to mine sites. The results of this analysis document that the mean and minimum distances of caribou from mine sites are relatively similar each year, and that caribou are only close to the mine sites briefly for the pre-calving migration and for most of the post-calving through early fall period. The multi-strata results suggest some change in fidelity of caribou relative to the 50 km area around the Ekati mine site, a weaker trend around the Diavik mine, and no trend with the Snap Lake data. The level of activity supports this pattern; Ekati is a larger site with road traffic, Diavik is restricted to an island, and Snap Lake has not seen comparable activity. These results will be explored further in subsequent analyses.

There are some important assumptions that should be considered when interpreting the results of this study. The main assumption is that the distribution of collared caribou is representative of the entire Bathurst herd. This assumption may be violated due to the low sample sizes of collared caribou for each of the yearly periods. This assumption will be partially tested in future analyses by comparison of the distribution of collared caribou with the distribution of caribou from aerial surveys conducted by the mine sites.

A limitation of this study is the low resolution of satellite collar locations (primarily 5–7 days) to determine finer scale caribou movements and distribution relative to mine sites. For example, the 50 km buffer zone is quite large relative to the actual footprint of any of the mine sites. However, this buffer reflects the amount of resolution that is possible with weekly fixes. For example, the mean caribou weekly movement rate is about 50 km, and therefore using a smaller buffer zone would most likely be beyond the scale resolution possible from weekly fixes. Another assumption of the multi-strata model is that all movements across strata are detected. This assumption is probably violated to some degree given the long duration between satellite locations. However, it is still possible to compare movement probabilities between years and overall trends, if it is assumed that the bias caused by undetected movements is constant for all years of the study.

One other potential issue with this analysis is the low number of yearly caribou locations in close proximity to the mine sites (Fig. 6). For example, there were less than 10 yearly locations within 20 km of most of the mine sites for the summer season considered in the multi-strata analysis. It is difficult to infer whether the low number of points is due to the influence of mine sites, given that no pre-development data were collected. Low sample sizes potentially limit the multi-strata analysis to detect trends within smaller buffer areas around mine sites. Upcoming individual based analyses (as discussed later) that consider year-specific data collectively and habitat may increase the overall power to detect changes in caribou distribution relative to mine sites.

This analysis shows the potential utility of multi-strata models to confront biases associated with movement and distribution data, and allow detailed inference concerning response of caribou to mine sites. Of most interest is the weekly probability that a caribou will move out of the 50 km zone. A positive change in this probability over time would suggest less fidelity to or avoidance of the area. A positive trend is suggested with the Ekati mine site, which suggests that caribou are displaying a decreasing degree of fidelity to the 50 km buffer area. Of the mine sites, Ekati has seen the most intense development throughout the study, which might be associated with this change. This is an observational rather than controlled study and therefore correlation rather than causation can be inferred from these results.

A recent study by Johnson and Boyce (2004) suggested that caribou displayed avoidance of major developments at a distance of 33 km. However, this relationship was weak as determined by confidence intervals on model parameters that overlapped 0. The results of this study partially support the conclusion of some degree of avoidance by caribou to Ekati, but not other mine sites. As discussed in the following sections, an analysis similar to Johnson and Boyce (2004) will be conducted on an individual mine site basis.

SECTION 2: Comparison of distribution of caribou relative to mine sites from satellite collars and aerial transects

Introduction

The primary purpose of this analysis was to determine how comparable satellite collars and systematic aerial transects are in estimating the distribution of caribou relative to the mine sites. Both data sources have limitations that could potentially cause them to not reflect the true distribution of caribou. For example, the number of satellite collars was relatively low compared to the size of the herd. Aerial survey monitoring only covered a relatively small area around mine sites (1,200–3,000 km^2 around each mine; Appendix 1).

If estimates of the distribution of caribou relative to mine sites from aerial surveys and collars are similar then it would be expected that the relative proportion of collared caribou within the aerial survey study areas would be equal to the proportion of the entire herd within the survey study areas. To test this we used population estimates for the entire Bathurst herd derived from a trend analysis of calving ground surveys (Gunn et al. in prep) to determine the relative proportion of caribou detected in systematic aerial transects.

Methods

Data were analyzed separately for the Ekati mine (1997–2001), the combined Ekati/Diavik survey area (2002–2003), and the Snap Lake mine (1999–2003). Aerial surveys at each mine were conducted on a regular (generally weekly) to infrequently basis (Snap Lake), generally between April and October each year, and therefore it was often difficult to match the date in which satellite collars returned data with any particular aerial survey. For this reason, the data sets were broken up into weekly periods with both satellite collar data and transect survey data being summarized for each week. For each week, an estimate of the proportion of satellite-collared caribou on the survey grid ($\hat{p}_{collar}$) was obtained by $n_{on grid} / n_{total}$, where $n_{on\ grid}$ was the number of collared caribou that occurred on the grid and n_{total} was the number of collared caribou transmitting locations for the given week. For the weekly summary analysis, the mean proportion of locations on the grid was used if a caribou reported more than one location within a given survey week. For aerial survey data, the estimate of the proportion of the herd in the survey grid ($\hat{p}_{transect}$) was estimated as $\hat{N}_{grid} / \hat{N}_{total}$, where $\hat{N}_{grid}$ was the estimated population size within the grid area for the survey week and $\hat{N}_{total}$ was the estimated population size for the Bathurst herd for the year of survey (Gunn et al. in prep). If more than one aerial survey occurred within a week, then the mean estimate from the surveys was used. Two seasons were considered, the northern migration and calving season (up to and including 30 June), and the post-calving/summer season (after 30 June).

Two analysis were used to compare $\hat{p}_{collar}$ and $\hat{p}_{transect}$. First, each estimate was plotted and examined visually to determine yearly and seasonal-based variability in agreement between each estimate. Second, logistic regression was used to determine if $\hat{p}_{transect}$ could predict $\hat{p}_{collar}$. This analysis explored whether the relative number of caribou from aerial survey would influence the probability of any collared caribou being on the survey grid. For each mine survey data set, $\hat{p}_{transect}$, year, and season were used to predict the proportion of collared caribou in the aerial transect study areas, therefore determining if year or season influenced the ability of $\hat{p}_{transect}$ to predict $\hat{p}_{collar}$.

The individual caribou was used as the sample unit for this analysis. Basically, each satellite fix was considered a “success” if the caribou was within the transect area and a “failure” if it was not. Weekly estimates of proportion of the population on the sampling grid from aerial survey data were used as a predictor covariate. Year and season of study were also considered. One potential issue with this analysis was non-independence of repeated survey points from collared caribou that potentially violated assumptions of logistic regression. To confront this, a repeated measures generalized estimating equation model (Ziegler and Ulrike 1998) was used to estimate correlations between successive observations of the same caribou. An exchangeable correlation matrix structure in which an individual caribou was the sample unit also was used to provide an estimate of overdispersion. An exchangeable correlation matrix was used because it could accommodate non-uniform timing of locations and working correlation matrices of different sizes (Ziegler and Ulrike 1998).

Results

The mean number of satellite-collared Bathurst caribou monitored weekly from 1997 to 2003 was 10.5 (SD = 2.16, range 7–16). Satellite collar fix acquisition intervals varied by season and year, ranging from weekly to daily (Gunn et al. 2002).

The Bathurst herd has declined by approximately 5% each year between 1986 and 2003 as estimated from spring calving surveys (Fig. 10; Gunn et al. in prep). Population estimates from each transect survey were divided by the total population estimate to account for the decrease in population size of the Bathurst herd over time.

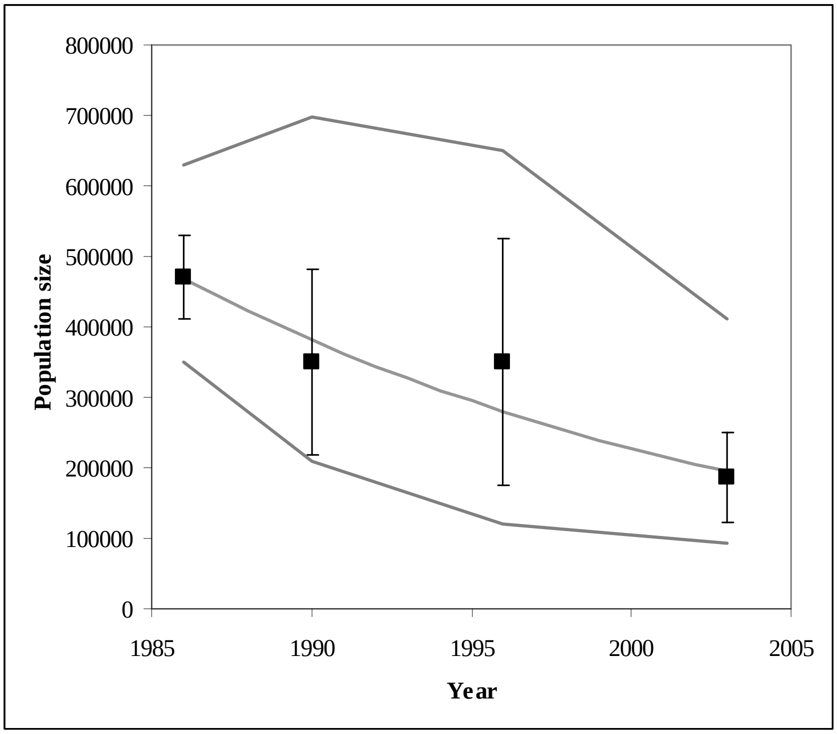


Figure 10. Population trend of the Bathurst caribou herd from calving ground surveys. Each point represents the herd size estimate and associated 95% confidence intervals. The trend line represents estimated trend in population size of the Bathurst herd and associated 95% confidence limits (Gunn et al. in prep). Estimates from the trend analysis were used to estimate the proportion of the herd in transect areas.

Ekati (1997–2001)

Of 1,365 satellite collar locations recorded during time periods of Ekati aerial surveys, 9 occurred within the Ekati study area. Comparison of $\hat{p}_{collar}$ and $\hat{p}_{transect}$ suggested that the proportion of caribou in the study area as estimated by satellite collared caribou was higher than the proportion of the population in the study area from aerial transects for both seasons (Fig. 11). One notable exception was from July 12, 1997 when the proportion of caribou estimated from aerial transects was 1.1 whereas the proportion of collars was 0.29. During that flight over 143,000 caribou were counted on transects in the study area, resulting in an extrapolated estimate higher than the projected herd estimate at that time (Fig. 10).

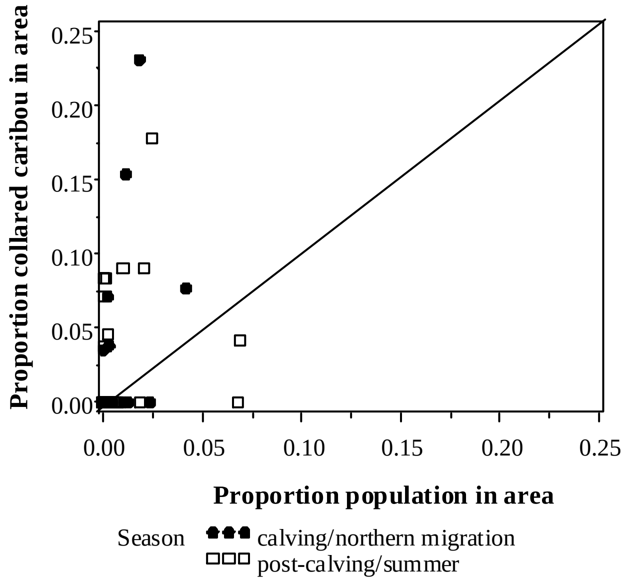


Figure 11. Proportion of satellite collars ($\hat{p}_{collar}$) versus proportion of population As estimated from aerial surveys on transect ($\hat{p}_{transect}$) study area for Ekati from 1997 to 2001 as a function of season. If both estimates of the proportion of the herd in the study area are equal they should be symmetrical around the 1:1 line of agreement. One observation in which the proportion of the population from surveys was 1.1 and the proportion of collars was 0.29 is not shown. Also, there were 11 observations in which both the proportion of collared caribou and proportion of the population was zero (77 observations total).

The observation of July 12, 1997 had a large degree of influence on the logistic regression analysis and therefore was eliminated as an outlier point. Results suggested that $\hat{p}_{transect}$ was a marginal predictor of $\hat{p}_{collar}$ (χ^2 = 3.94, df = 1, P = 0.047). The underlying relationship or slope between $\hat{p}_{transect}$ and $\hat{p}_{collar}$ was influenced by season ($\hat{p}_{transect}$ X season, χ^2 = 3.3, df = 1, P = 0.069). Year of study did not substantially affect the proportion of collared caribou in the transect study area (χ^2 = 5.03, df = 3, P = 0.17). Note that this analysis mainly tests how well $\hat{p}_{transect}$ predicts $\hat{p}_{collar}$ rather than if the two methods produce similar estimates.

Ekati and Diavik (2002–2003)

Of 1,065 satellite collar locations recorded during time periods of combined Ekati and Diavik aerial transects, 45 occurred within the combined study area. Comparison of $\hat{p}_{collar}$ and $\hat{p}_{transect}$ for the Ekati and Diavik transect study area suggests that the proportion of satellite collars was almost always greater than the estimated proportion of the total population size in the study area as derived from aerial surveys (Fig. 12). This comparison is most applicable to the post calving/summer season since few caribou were observed using either method for the calving/northern migration period.

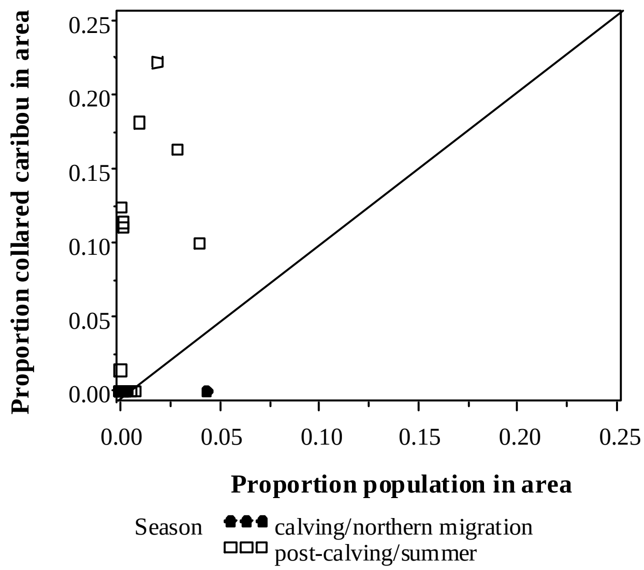


Figure 12. Proportion of satellite collars ($\hat{p}_{collar}$) versus proportion of population as estimated from aerial surveys on transect ($\hat{p}_{transect}$) study area for Ekati and Diavik transect areas from 2002 and 2003 as a function of season. If estimates are equal they should be symmetrical around the 1:1 line of agreement. There were 7 observations in which both the proportion of collared caribou and proportion of the population was zero (38 observations total).

Logistic regression results suggested that $\hat{p}_{transect}$ was a marginal predictor of $\hat{p}_{collar}$ (χ^2 = 3.38, df = 1, P = 0.05). The significance of $\hat{p}_{transect}$ as a predictor of $\hat{p}_{collar}$ was related to season ($\hat{p}_{transect}$ X season; χ^2 = 3.63, df = 1, P = 0.057). Year of study had no effect on the proportion of satellite collars in the study area (χ^2 = 0.02, df = 1, P = 0.88).

Snap Lake (1999-2003)

Of 484 satellite collar locations recorded during time periods of Snap Lake aerial transects, 12 occurred within the Snap Lake study area. Comparison of $\hat{p}_{collar}$ and $\hat{p}_{transect}$ for the Snap Lake study area suggested weak correspondence between the proportion of collared caribou and estimated population size on the transect study area (Fig. 13). There was a slight tendency for the proportion of satellite-collared caribou to be higher than the proportion population, however, this trend was not as strong as other mine sites.

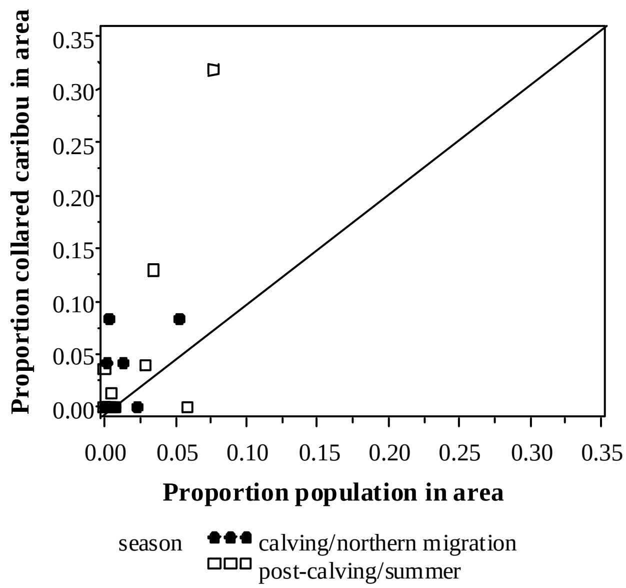


Figure 13. Proportion of satellite collars ($\hat{p}_{collar}$) versus proportion of population in transect ($\hat{p}_{transect}$) study area for Snap Lake transect areas from 1999-2003 as a function of season. If estimates are equal they should be symmetrical around the 1:1 line of agreement. There were 3 observations in which both the proportion of collared caribou and proportion of the population was zero (28 observations total).

Logistic regression results suggested that $\hat{p}_{transect}$ was not a significant predictor of $\hat{p}_{collar}$ (χ^2 = 1.74, df = 1, P = 0.19). Year of study had no effect on the proportion of satellite collars in the study area (χ^2 = 3.63 df = 3, P = 0.30). Season also did not influence the relationship between $\hat{p}_{transect}$ and $\hat{p}_{collar}$ (χ^2 = 0.43, df = 1, P = 0.51).

Discussion

One of the most apparent results from this analysis was that the estimated proportion of the population in mine areas as estimated by satellite collars ($\hat{p}_{collar}$) was consistently higher than that estimated by aerial surveys on transects ($\hat{p}_{transect}$). However, the estimated proportion of caribou in the mine areas from aerial surveys was associated with the proportion of collared caribou for the Ekati, and Ekati/Diavik studies as revealed by logistic regression. Therefore, there was correspondence between each estimate even though $\hat{p}_{collar}$ was often higher than $\hat{p}_{transect}$.

No relationship was detected between $\hat{p}_{collar}$ and $\hat{p}_{transect}$ for Snap Lake mine site. The estimates of $\hat{p}_{transect}$ and $\hat{p}_{collar}$ .are lower for the Snap area suggesting less caribou are in this area than around Ekati and Diavik. However, even if there were less caribou in the Snap area there still should have been a relationship between $\hat{p}_{transect}$ and $\hat{p}_{collar}$. One potential reason for the apparent lack of relationship was lower aerial survey coverage and lower number of collared caribou in the area that potentially reduced the precision of both estimates making any correspondence harder to detect.

The general trend in which $\hat{p}_{collar}$ was higher than $\hat{p}_{transect}$ does not necessarily mean that either of the methods is biased given that the true proportion of caribou within the transect study areas was unknown. Various circumstances might produce the observed results:

1. *Baseline population size from trend analysis was biased.* One potential cause for lower estimates of the proportion of caribou in the population from aerial transects would be if the population size estimates from calving ground surveys were biased. Estimates of total population size were based upon four spring calving ground surveys taken over 19 years. The trend estimated was exponential, which may have been a simplification of actual trend. However, the precision of the most recent 2003 survey was relatively high. Therefore, it is doubtful that the degree of bias in herd size estimates is large.

2. *Population estimates from transect surveys are biased low.* Factors such as sightability could cause estimates of population size for transects to be biased low. For example, in most cases the assumed strip width of surveys was 600 m on both sides of the aircraft, however, in up to 1999 at Ekati and 2000 at Snap Lake it was assumed to be 1 km, which would result in higher bias. If the actual strip width (area where most or all caribou were seen) was smaller, then the actual coverage of transects would be overestimated, leading to an underestimate of population size within the transect area. Narrower strip width would result in a lower visibility bias. As discussed later, it would be useful to estimate sightability from surveys to ensure this source of bias is minimized.
3. *Satellite collared caribou are not representative of herd.* An extremely small proportion of the Bathurst herd is collared, which could potentially lead to the collared caribou having poor coverage of the actual distribution of the herd. Intuitively, it would be expected that small sample sizes of collared caribou would affect the precision (repeatability) of estimates of distribution, but would not necessarily bias estimates unless the collared caribou were not a random sample of the entire herd of caribou.
4. *Low sample sizes of collars reduce the sensitivity of weekly estimates of proportion collars on grid.* On average, only 10.5 caribou were collared at any one time between 1996 and 2003. Therefore, the number of measurement intervals from collars was limited. For example, assuming 10 collars were available, the proportion of caribou in a sampling area could be 0, 0.1, 0.2, 0.3 to 1.0 for any given time period. Aerial transect measurements estimate the proportion of caribou in transect areas to be between 0 and 0.1 for many of the survey occasions at Ekati (Fig. 11) and Ekati/Diavik (Fig. 12). Therefore, given the sample size of collars, it would be difficult for the estimate of collars to match this estimate since measurements of 0 or 0.1 were only possible. Twice as many collars would be needed to allow corresponding estimates of 0.05. This problem was somewhat circumvented by the logistic regression analysis that estimated the probability of individual caribou occurring in transect areas. In this case the

sample size was the number of locations that individual caribou reported, eliminating the issue of weekly sample sizes of collars.

The consistency in which $\hat{p}_{collar}$ was higher than $\hat{p}_{transect}$ for Ekati and Ekati/Diavik data sets compared to Snap Lake suggests that $\hat{p}_{transect}$ might be an underestimate of caribou occurring within the mine areas for Ekati and Diavik. One plausible reason for this is that sightability of caribou has not been accounted for adequately, leading to an underestimate of population size around these areas. Distance sampling could be used to estimate the true strip width of surveys. This methodology has been successfully applied to pronghorn populations in Wyoming (Guenzal 1997). It would be most applicable to the summer season when caribou are dispersed and sightability is generally less than during periods of snow cover.

SECTION 3: Analysis of habitat selection by caribou relative to mine sites using data from satellite collars

Introduction

The objective of this analysis was to explore the potential effect of mine sites on caribou distribution using data from satellite collared caribou. Here we model caribou habitat selection using vegetation data, NDVI, and insect activity indices as covariates. We then examine the influence of distance from mine developments on caribou habitat selection. From this we infer whether mine sites have a measurable effect on caribou distribution, and if so, at what distance this effect occurs. The model’s main object was to create the simplest model that explained caribou distribution in summer range areas. Of primary interest was whether selection for habitats changed as a function of distance from mine sites.

Methods

The main season of interest for this analysis was the summer season after the post-calving migration, which we defined as between July 14 and October 15 based upon arrival of caribou in summer range areas. It is during this season that the assumptions of resource selection modeling, such as independence between animals, were most likely met. We contend that selection during the relative rapid spring migration to calving areas, and to a lesser extent the initial post-calving migration, probably occurs at the herd level, therefore making it problematic to model selection for resources based upon individual collared caribou.

Vegetation data

We used vegetation data from the WKSS remote sensing study (Matthews et al. 2001) to define caribou habitat. We used the Snap Lake vegetation study data (De Beers 2002) where available, since the Snap Lake data were developed with greater ground checks and extends into areas south of Snap Lake not covered by the WKSS study. We condensed categories based upon low frequency of some types, previous work by BHP Billiton (2004), and logical assumptions about caribou biology (Table 6).

Table 6. Vegetation categories used in the analysis and corresponding WKSS/Snap Lake vegetation types.

Pooled habitat association	Vegetation (habitat) types
Bare	Bare ground
	Gravel deposit
Boulder	Bedrock association
	Boulder association
Esker	Lichen veneer
	Esker complex
Forest	Spruce forest
	Mixed forest
	Old burns
	Young burns
Tundra	Heath tundra
	Heath/boulder
	Heath/bedrock
Other	Unclassified
	Ice and snow
Shrub	Tall shrub
	Birch seep
	Low shrub
Sedge wetland	Wetland (sedge meadow)
	Tussock/hummock
	Peat bog
Water	Deep water
	Shallow water

Seasonality

One potential issue in the analysis was that certain vegetation types such as sedge wetland and tundra may have not been selected for until after green-up. To index seasonality in vegetation we used the Normalized Difference Vegetation Index (NDVI). NDVI is highly correlated with vegetation parameters such as green-leaf biomass and green-leaf area and, hence, is of considerable value for vegetation discrimination (Justice et al. 1985). Mean NDVI values were estimated using an 8 x 8 km pixel size for the larger summer range area of caribou. These values were then modeled as an interaction term with tundra and sedge wetland to emulate changes in these habitats over a given season and among years.

From 1 x 1 km resolution NDVI data for 2002 we used data classified for the 11–21 July composite to create an NDVI-based habitat surrogate (Hab_{NDVI}). NDVI tends to peak during mid-July in most years (Fig. 4). The rationale behind this was that caribou should mainly select for areas of comparatively high greenness (and forage) value. Therefore, NDVI might provide a parsimonious substitute to categorical habitat values.

Insect activity

The degree of selection for a given habitat type is probably influenced by insect activity. For example, caribou appear to be found in esker and other windy, sparsely vegetated areas during time of high insect abundance (Murphy and Curatolo 1987, Russell et al. 1993, Gunn et al. 2002). To account for this, we used Russell et al.'s (1993) index of insect abundance. Russell et al. (1993) developed a functional relationship between insect abundance, and temperature and wind speeds by sampling mosquitoes and oestrid flies during different temperatures and wind speeds. The equations are listed below:

Mosquito index

if temperature>=18 then tim=1

if temperature<=6 then tim=0

if 18>temperature>6 then tim=(1-(18-temperature)/18)

if wind speed >6 then twin=0

if wind speed <6 then twin=(6-minwind)/6

index =tim*twin

Oestrid fly index

if temperature>=18 then tio=1

if temperature<=13 then tio=0

if 18>temperature>13 then tio=(1-(18-temperature)/10);

if meanwind>9 then owin=0

if meanwind<9 then owin=(9-meanwind)/9

ostindex=tio*owin

These indices were based on the wind speed in which no insects were caught (>6 km/hr for mosquitoes, >9 km/hr for oestrid flies), the minimum temperatures in which insects were caught (<6 and <13º C for mosquitoes and oestrids, respectively) and the temperature in which insect abundance did not increase (>18º C for both mosquitoes and oestrids). Data from the Ekati, Diavik and Snap Lake weather stations were used for these indices. Point locations of collared caribou were assigned insect abundance scores based upon data from the closest weather station.

Distance from mine sites and roads

One issue with the Diavik and Ekati analysis was that distances of any point location of caribou from the Diavik, Ekati, and Misery road and camp were highly correlated given the close proximity of the sites. For example, Pearson correlation coefficients were above 0.90 for all combinations of variables. Therefore, it was impossible to independently determine the influence of each of the mine sites or roads on caribou distribution. Given this, the mean distance of caribou to any mine site and road was used to describe proximity to mine developments. In addition, the closest distance of a caribou point location was used to determine the sensitivity of the analysis to how distance from mine site was defined. The Misery road was only considered during or after 2000, which was when it was completed and fully operational. Analysis for the Snap Lake mine was conducted independently given that it was further south than the Ekati and Diavik mines and may have different habitat relationships than Ekati and Diavik.

Statistical methodology

Resource selection functions (Manly et al. 1993) were used to assess habitat selection of caribou and the effects of mine sites on caribou distribution. For this analysis the vegetation types in a 1 km buffer radius of each caribou point were classified. These were compared with 6 random points that were within a circle around the previous location with an "availability radius" defined by the 95th percentile of the distanced moved for caribou for the interval between successive point locations (Arthur et al. 1996; Johnson and Boyce 2004) (Fig. 14). This method of defining availability assumes that caribou select habitat in a temporally narrow window of time so that availability of resources changes through time.

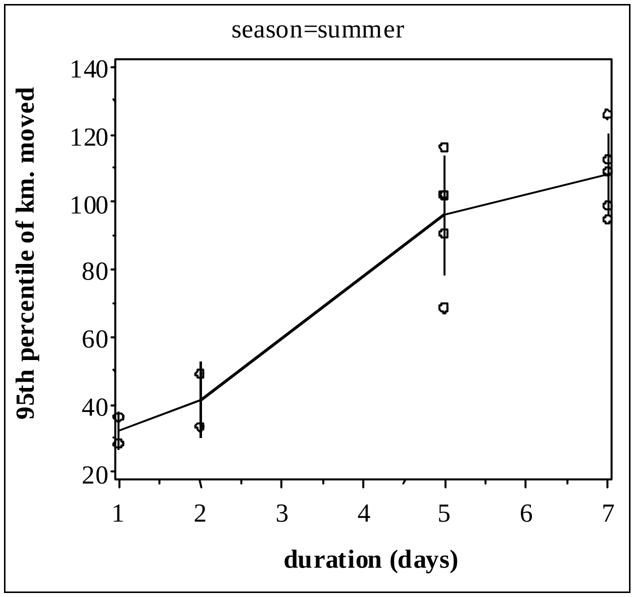


Figure 14. The 95th percentile of distances moved by Bathurst caribou between successive satellite collar fixes, 1996–2003. Each data point represents a given year in the analysis (interval between fixes changed among years), and error bars depict 1 standard error. The 95th percentile was used to define the availability of habitats for subsequent locations.

One issue with this analysis was determination of the range of distances from mine sites to be considered in the analysis. Johnson and Boyce (2004) only considered collar locations for caribou where the mine site was within the available area as determined by the circle of availability. This approach is intuitive, however, one issue that it caused was that the consideration of distance from mine site became a function of the duration of time between fixes rather than the distance of the caribou from the mine site. For example, a caribou that was 100 km from the mine site would be considered in the distance from mine site analysis if the interval between fixes was 7 days (with an availability radius of 110 km; Fig. 14). However, a caribou with an interval of 1 day between fixes would not be included in the analysis unless it was within 30 km of the mine site. Actual consideration of the effect of mine sites most likely depends on the scale in which caribou are selecting habitat features. It might be argued that caribou select habitat at a finer scale than that reflected by the availability radius from longer durations between fixes. For this reason, we conducted analysis at a larger scale in which data with an availability radius that intersected mine sites were included regardless of availability radius or duration between fixes. This approach was similar to that of Johnson and Boyce (2004). We also conducted the analysis at a finer scale in which only data from collars in 2002 and 2003 that returned daily fixes were used. This analysis assumed caribou were selecting features on a daily basis. We predicted that if a threshold of habitat selection occurred for the large or small-scale analysis then a model with a quadratic (i.e., distance from mine site squared) term would be supported by the data. If the effect of mine sites extended beyond the scale considered, then a linear increasing distance from mine site term would be supported by the data.

Conditional logistic regression (Hosmer and Lemeshow 2000) was used to compare used and random points, where the availability radius defined availability. The conditional logistic regression model defined each used and 6 accompanying random points as a cluster, therefore centring each comparison on the resources available to the caribou at the time in which the location was taken. This approach avoided issues with psuedoreplication caused by pooling telemetry data from different caribou (Pendergast et al. 1996, Johnson and Boyce 2004)

Logistic regression based upon comparison of used and random locations cannot estimate true probability of occurrence since the actual sampling proportion of random locations that may have been used is unknown (Manly et al. 1993). However, it is possible to estimate the odds ratio or selection coefficient (w_x) for a given model parameter. The odds ratio estimates the degree in which a given model variable (i.e., tundra) was selected relative to an incremental change in its value. The odds ratios of variables of interest were plotted as a function of their range of values to determine the relative importance of each variable. Analyses were conducted using procedure PHREG in SAS (SAS Institute 2000).

Results were evaluated using AICc model selection methods (which indexes the model most supported by a data set). Models with the lowest AICc scores were considered further (Burnham and Anderson 1998). Models which differed by <2 AICc units from the most supported model (as indicated by ΔAICc scores <2) were also considered. Akaike weights (w_i) were also used to determine proportional support for models in the analysis. Hypothesis tests of resource coefficient estimates were used to further evaluate individual model parameters. In addition, applicable relationships between the odds ratios and ranges of covariates were plotted to allow evaluation of the biological relationship between covariates and model predictions.

Receiver Operating Characteristic (ROC) curves were used to compare the predictive ability of models (Cummings 2000). A ROC curve basically considers how well a model predicts presence or absence through a range of probability cutpoints. The ROC score varies between 0.5 and 1. A score of 0.5 would correspond to a model with no predictive ability and a score of 1 would correspond to a model with perfect predicative ability. Models with scores >0.7 are considered to be of "useful" predictive ability (Boyce et al. 2002). In addition, the goodness of fit test of Hosmer and Lemeshow (2000) was used to evaluate overall model fit.

We were most interested in the distance from mine site that caribou occurrence was influenced. If an effect of mine site was occurring it would be expected that probability of occurrence would be low adjacent to the developments, and then increase and asymptote at a critical distance from the mine site. This relationship was approximated by a quadratic distance from mine term in the logistic regression equation. However, there is uncertainty in where the quadratic curve would asymptote due to sampling error and heterogeneity of landscape conditions. To estimate this uncertainty we used a bootstrap randomization procedure where a dummy data set was created in which all of the habitat and population terms in the most supported AIC model were set to mean values, except for distance from mine site that was varied across the range of observed values. The field data set was then randomly re-sampled 1,000 times and the data run through the most supported AIC model. Each model run created an estimate of the quadratic curve and estimate of distance from mine site where a potential effect was occurring (using the dummy data set values). The mean distance, standard error, and 2.5th and 97.5th percentiles (termed lower and upper confidence limits, respectively) were then used as an estimate of error around the distance at which the mine site affected caribou habitat selection.

Results

Habitat, greenness, and insect activity

Proportions of habitat associations were reasonably similar for points used by caribou and random points (Fig. 15). Tundra comprised over 40% of habitat used by caribou. The bare habitat association was of equal and relatively low proportion between used and random points, and was pooled with the boulder habitat association.

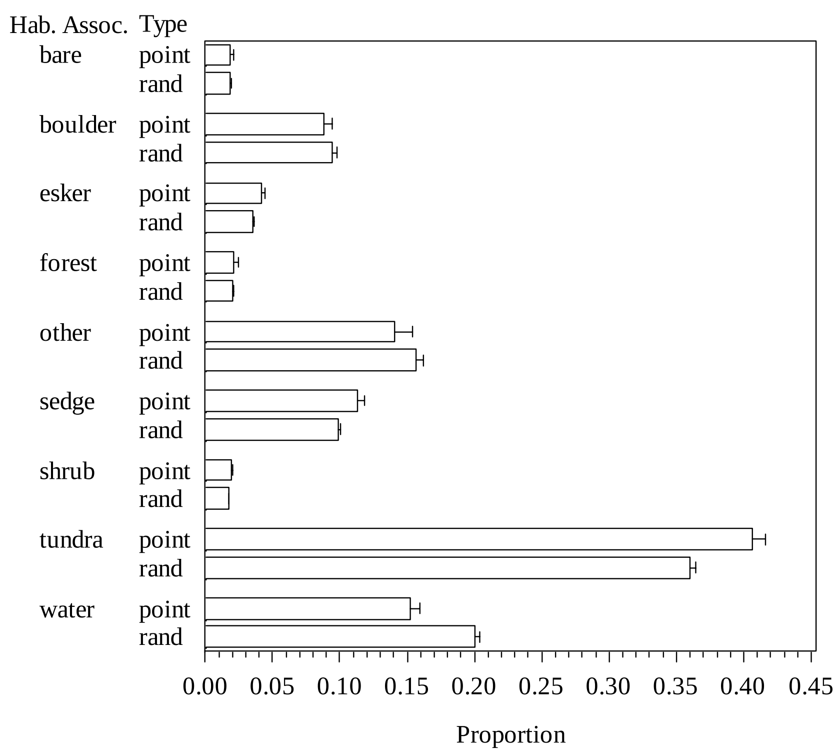


Figure 15. Proportion of habitat associations in 3.14 km^2 circles that buffered locations used by caribou during summer (point) and random (rand) points. Standard errors for mean estimates are shown as error bars.

Trends in NDVI index for the summer range revealed a reasonable degree of similarity among years. An exception was an earlier green-up period in 1998 compared with other years (Fig. 4).

Mosquito and oestrid indices displayed a large degree of temporal variability regardless of year or weather station (e.g., Fig. 16). This was most likely due to extreme variability in wind speeds and temperature.

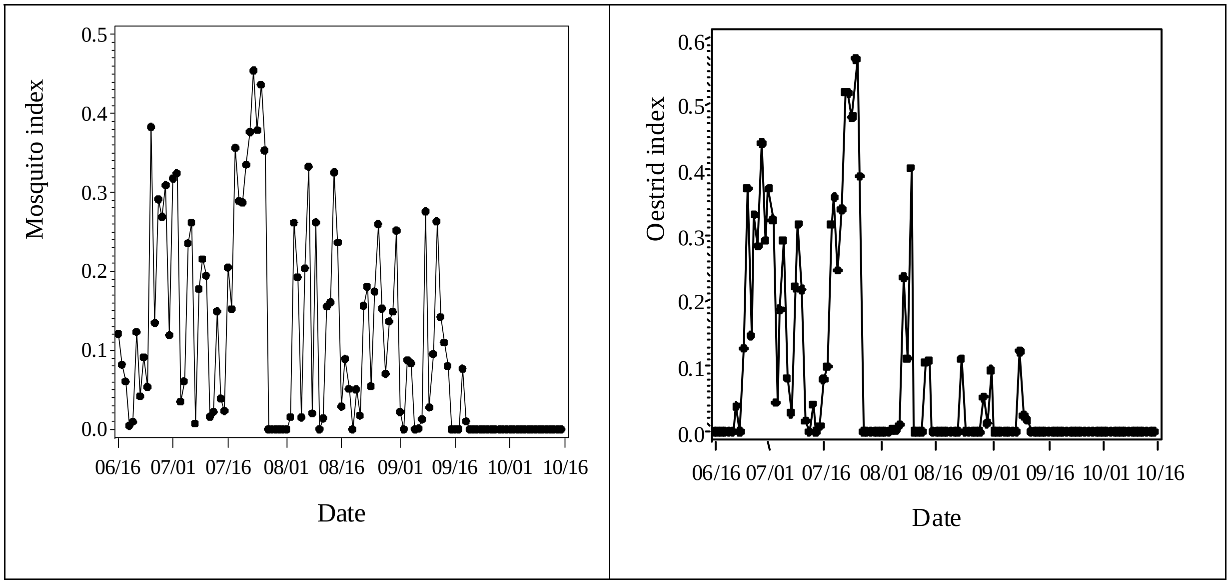


Figure 16. Example mosquito and oestrid indices from the Ekati weather station in 2002

Habitat selection analysis

Ekati and Diavik

Two hundred and sixty three points (each with 6 corresponding random points) from 49 caribou-year combinations were used for the large-scale habitat analysis of the combined Ekati/Diavik data. Data from 1996 to 2002 were used in the initial analysis. Data from 2003 could not be used because NDVI data were not available for that year. Because points had different time intervals between successive collar locations, the actual mean distance from mine development of points was 86.9 km. (SD = 47.4, range 10.3–283.6, n = 1820) for used and random points combined (Fig. 17).

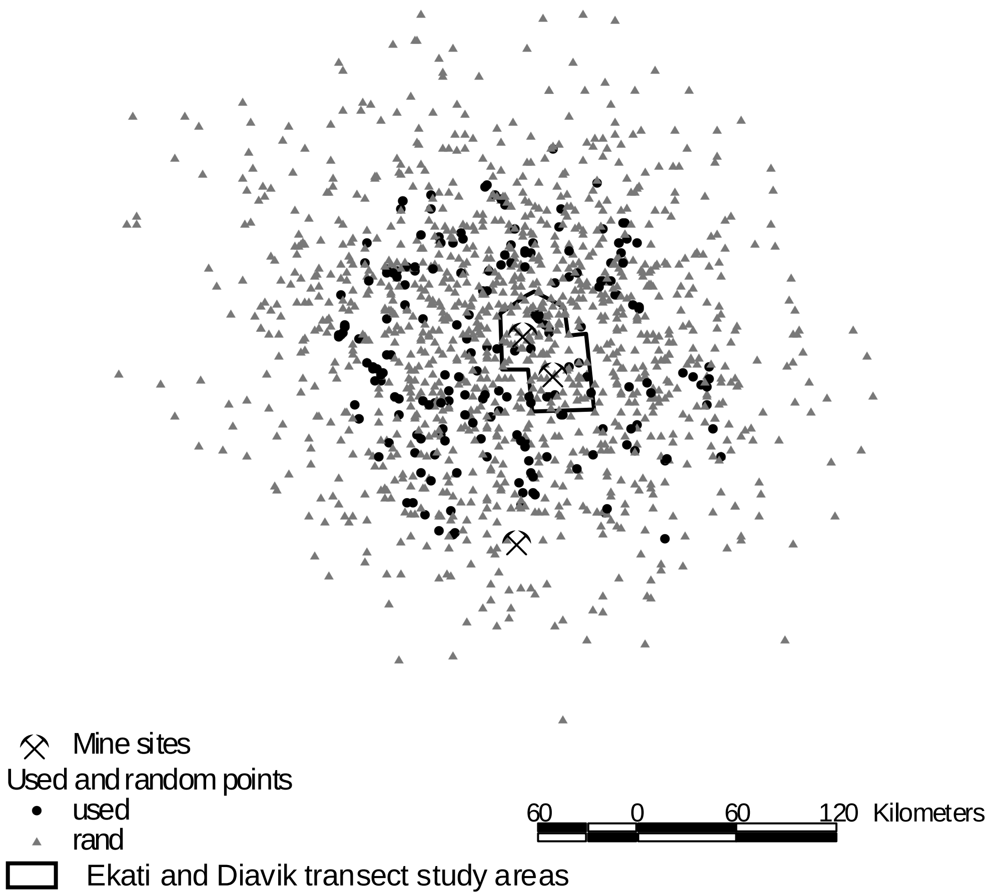


Figure 17. The spatial distribution of points used by satellite-collared caribou and associated random points for the large-scale analysis of the Ekati/Diavik area, Bathurst caribou herd, 1996–2002.

A preliminary analysis was conducted to determine if interactions between insect activity indices and esker and boulder habitats were significant predictors of caribou distribution. We hypothesized that caribou might choose eskers or bare ground in times of high insect activity given that these areas would be more exposed to wind. Hypothesis tests failed to detect significant interactions between insect activity indices (mosquito index X esker: $\chi^2 = 0.097$, df = 1, $P = 0.75$; mosquito index X boulder, $\chi^2 = 13.7$, df = 1, $P = 0.18$; oestrid index X esker: $\chi^2 = 0.20$, df = 1, $P = 0.66$; oestrid index X boulder $\chi^2 = 0.11$, df = 1, $P = 0.74$) and relevant habitat types. For this reason, insect activity indices were not considered further. In addition, including these indices in the

AIC analysis would have further reduced sample sizes since weather data were not available for all the dates considered in the analysis (primarily in 1996 and 1997).

AICc model selection results suggested that a model with a quadratic distance from development term as well as interaction tundra X NDVI and esker terms was most supported by the data (Table 7). Models with sedge, and sedge X NDVI terms were also supported by the data. A model with an interaction of year and distance from mine site was also potentially supported by the data, however, the interaction term was not significant ($\chi^2 = 0.60$, df = 1, $P = 0.44$).

Table 7. AICc model selection for Ekati and Diavik analysis for the large-scale analysis, Bathurst caribou herd, 1996–2002.

Habitat variables	Mine variables	AICc	ΔAICc	w_i	k	Log L
Esker, tundra X NDVI	Dist, dist^2	831.5	0.00	0.35	4	-411.7
Esker, tundra X NDVI	Dist, dist^2, dist X yr	833.0	1.47	0.17	5	-411.4
Esker, sedge, tundra X NDVI	Dist, dist^2	833.1	1.62	0.16	5	-411.5
Esker, tundra X NDVI, water	Dist, dist^2	833.3	1.76	0.15	5	-411.5
Esker, sedge X NDVI, tundra X NDVI	Dist, dist^2	833.6	2.07	0.13	5	-411.7
Esker, sedge, tundra	Dist, dist^2	836.6	5.04	0.03	5	-413.2
Esker, tundra, sedge, water	Dist, dist^2	838.6	7.11	0.01	6	-413.2
Boulder, esker, forest, other, sedge X NDVI, shrub, tundra X NDVI, water	Dist, dist^2	840.4	8.92	0.00	10	-409.8
Esker, tundra X NDVI	Dist,	857.7	26.13	0.00	3	-425.8
	Dist, dist^2	877.6	46.10	0.00	2	-436.8
Boulder, esker, forest, sedge, shrub, tundra		893.6	62.09	0.00	6	-439.6
Esker, tundra X NDVI		904.1	72.54	0.00	2	-450.0
Hab_{NDVI}		959.9	128.40	0.00	1	-479.0

Chi-square tests for parameters were significant for the most supported model (Table 8). Results from the Hosmer and Lemenshow (2002) goodness of fit test suggested adequate model fit for the most supported model ($\chi^2 = 6.79$, df = 1, $P = 0.56$). The ROC score for the most supported model was 0.72, which suggested that the model displayed useful predictive ability.

Table 8. Chi-square tests for parameters in the most supported model for the Ekati/Diavik large-scale analysis, Bathurst herd, 1996–2002 (Table 7).

Variable	df	β	Std Err (β)	χ^2	*P*
Dist	1	0.0330	0.0115	8.19	0.004
$Dist^2$	1	-0.0003	0.0001	17.98	0.000
Esker	1	6.5951	1.4013	22.15	0.000
Tundra X NDVI	1	7.1006	1.6209	19.19	0.000

Bootstrap analysis and plots of parameters revealed an apparent effect of mine development on probability of occurrence by caribou up to distances of 51.9 km (CI 40.4–60.2) (Fig. 18). Large confidence intervals on odds ratio estimates suggested that this relationship was statistically weak. Plots also suggested that areas with higher proportions of eskers and tundra (when NDVI was higher) were also selected for (Fig. 18).

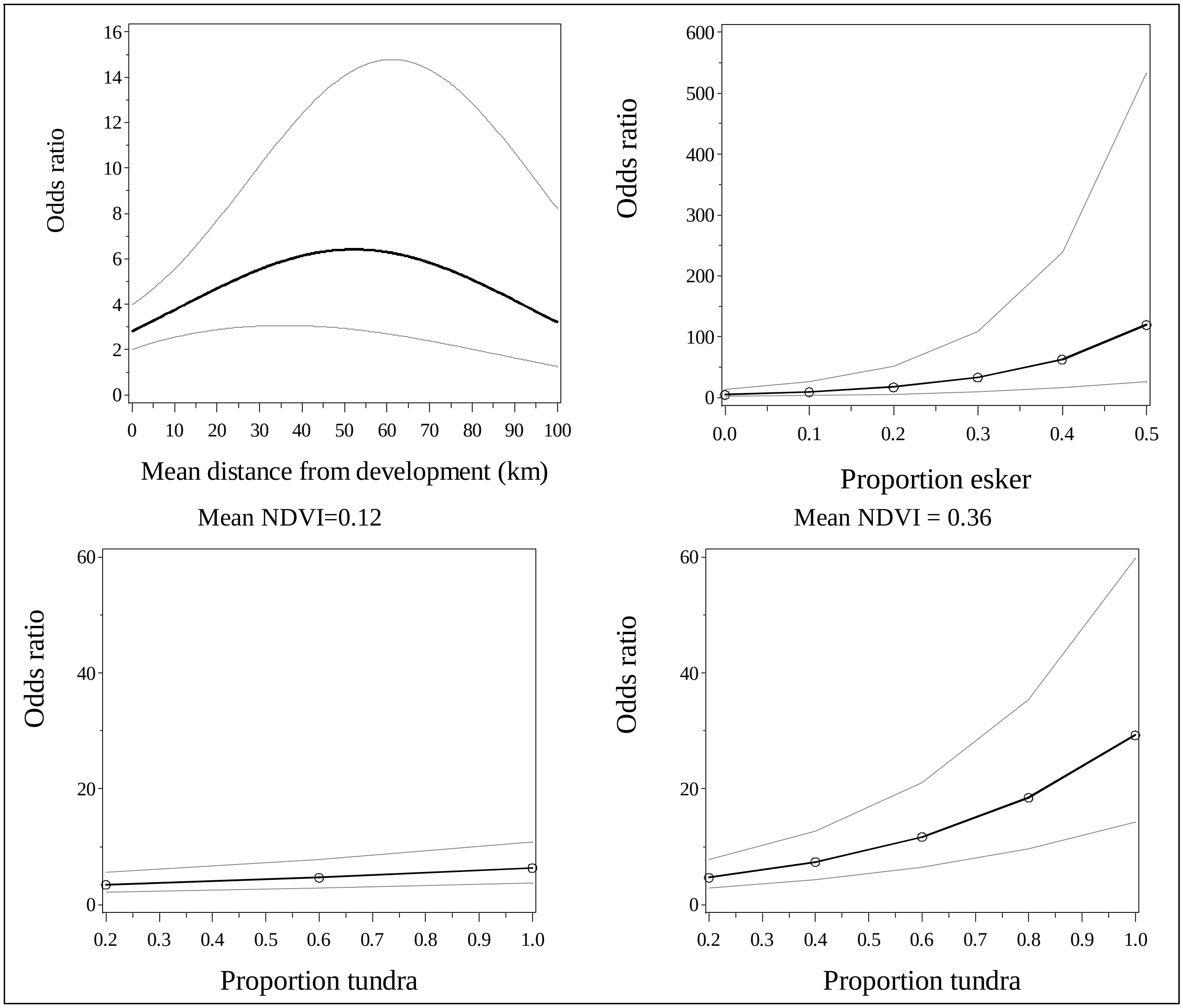


Figure 18. Predicted odds ratios from the large-scale analysis for the Ekati/Diavik area, Bathurst caribou herd, 1996–2002. All other covariates were held constant at their mean values. Ninety-five percent confidence intervals are given.

The fine scale analysis for the Ekati/Diavik area only included 2002–2003 collared caribou that reported daily satellite locations considered 57 used points (each with 6 accompanying random points) from 10 caribou-year combinations (Fig. 19). NDVI data were not available in 2003 and therefore NDVI-based models were not included in the analysis. The average distance of caribou used and random points from mine developments was 35.2 km, (SD = 16.17, range=10.2–122.9, *n* = 399)

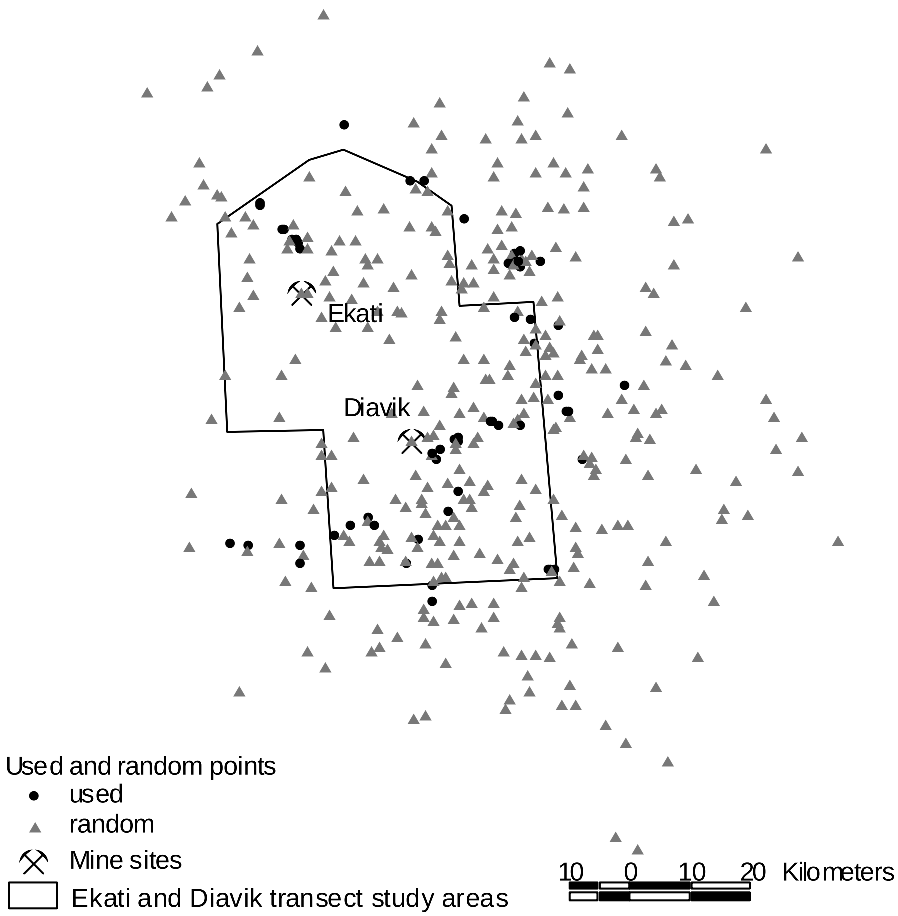


Figure 19. Used and random point spatial distribution for the fine scale analysis, Ekati/Diavik area, Bathurst caribou herd, 2002–2003.

Model selection results at the fine scale were different than the larger scale with a model with proportion water, and distance from development as a quadratic term being most supported (Table 9). A model with an interaction between year and distance from development was less supported (ΔAICc=2.25) suggesting that a change in the relationship between probability of caribou occurrence and distance from mine site was not detected. A model with a linear distance from development term was not supported by the data (ΔAICc = 19.49), further supporting the existence of a threshold distance in which mine sites affect caribou habitat selection (Table 9). The slope terms for all model parameters were significant for the most supported AICc model (Table 10).

Table 9. AICc model selection results for fine scale analysis (2002-2003 daily fix data), Ekati/Diavik area, Bathurst caribou herd.

Habitat variables	Mine variables	AICc	ΔAICc	w_i	k	Log L
Water	Distdev, distdev^2	155.20	0.00	0.56	3	-74.38
Tundra, water	Distdev, distdev^2	157.44	2.24	0.18	4	-74.34
Water	Distdev, distdev^2, Distdev X year	157.45	2.25	0.15	4	-74.35
Tundra	Distdev, distdev^2	158.22	3.02	0.12	3	-75.89
Esker, tundra	Distdev, distdev^2	160.53	5.33	0.04	4	-75.89
Esker, tundra	Distdev, distdev^2	160.53	5.33	0.04	4	-75.89
Esker, sedge, tundra	Distdev, distdev^2	160.69	5.49	0.04	5	-74.77
Esker, sedge, tundra, water	Distdev, distdev^2	161.25	6.05	0.03	6	-73.80
Water	Distdev	174.69	19.49	0.00	2	-85.23
	Distdev, distdev^2	179.83	24.62	0.00	2	-87.80
Boulder, esker, forest, other, sedge, shrub, tundra		187.25	32.05	0.00	7	-85.51

Table 10. Significance tests for the most supported AICc model for the small-scale analysis (Table 9), Ekati/Diavik area, Bathurst caribou herd, 2002–2003.

Variable	df	β	Std Err (β)	χ^2	*P*
Dist	1	0.410	0.131	9.71	0.0018
Dist^2	1	-0.008	0.002	12.09	0.0005
Water	1	-4.863	1.237	15.47	0.0001

Results from the Hosmer and Lemenshow (2000) goodness of fit test suggested adequate model fit for the most supported model (χ^2 = 12.78, df = 8, P = 0.12). The ROC score for the most supported model was 0.79, which suggested that the model displayed useful predictive ability. Bootstrap analysis and plots of the odds ratio of distance from development suggest a potentially large effect up to 26.5 km (CI 25.6 to

26.8) from the mine sites. However, the degree of precision of the magnitude of the effect, presumably due to lower sample sizes of data points, prevents definitive conclusions from this graph (Fig. 20)

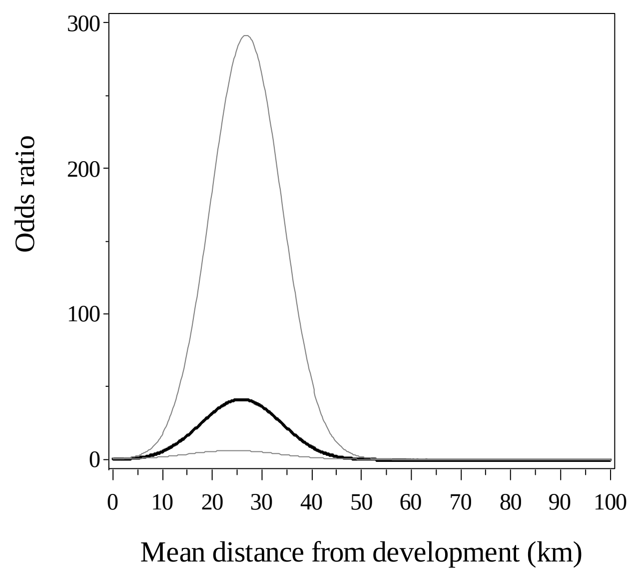


Figure 20. Odds ratios for selection of habitat by Bathurst caribou as a function of distance from Ekati/Diavik mine developments for the small-scale analysis, 2002–2003. All other covariates were held constant at their mean values. Standard errors are given around estimates as grey lines.

Snap Lake

For the large-scale analysis of the Snap Lake area there were 125 used locations (and 6 random locations for each) from 34 caribou-year combinations from 1999–2002. The average distance of used and random points from mine site was 92.7 km, (SD = 51.2. range 3.3–320.3, n = 2099). As with the Ekati and Diavik analysis, none of the interactions between insect activity and habitat were significant (at α=0.05) and therefore insect indices were not considered further in the analysis. AICc model selection suggested that the most supported model contained linear and quadratic distance from development terms, an interaction between year and distance from development, interaction between mean NDVI with tundra, and a linear water term (Table 11). A model that removed the interaction between year and distance was also supported by the data.

Table 11. AICc model selection for large-scale analysis of the Snap Lake area, Bathurst caribou herd, 1999–2002.

Habitat variables	Mine variables	AICc	ΔAICc	w_i	k	Log L
Tundra X NDVI, water	Distdev, distdev^2, Distdev X year	336.57	0.00	0.55	5	-163.05
Tundra X NDVI, water	Distdev, distdev^2	338.22	1.65	0.24	4	-164.95
Sedge X NDVI, tundra X NDVI, water	Distdev, distdev^2	340.26	3.69	0.09	5	-164.89
Tundra X NDVI	Distdev, distdev^2	341.78	5.21	0.04	3	-167.79
Esker, tundra X NDVI	Distdev, distdev^2 Distdev X year	342.12	5.55	0.03	5	-165.82
Esker, sedge X NDVI, water	Distdev, distdev^2	342.44	5.87	0.03	6	-164.88
Esker, sedge, tundra, water	Distdev, distdev^2	344.36	7.79	0.01	6	-165.84
Esker, sedge, tundra X NDVI	Distdev, distdev^2	345.49	8.92	0.01	5	-167.50
Boulder, esker, forest, other, sedge X NDVI, shrub, tundra X NDVI, water	Distdev, distdev^2	347.74	11.17	0.00	10	-162.95
Esker, sedge, tundra	Distdev, distdev^2	348.94	12.37	0.00	5	-169.23
	Distdev, distdev^2	349.07	12.50	0.00	2	-172.49
Esker, tundra X NDVI	Distdev	386.90	50.33	0.00	3	-190.35
Tundra X NDVI		449.78	113.21	0.00	2	-222.84
Hab_{NDVI}		453.93	117.36	0.00	1	-225.95
Hab_{NDVI} X NDVI		455.81	119.24	0.00	2	-225.86

Significance tests suggested that all parameters in the most supported AICc model were significant (Table 12)

Table 12. Significance tests for the most supported AICc model for the large-scale (Table 11), Snap Lake mine area, Bathurst caribou herd, 1999-2002

Variable	df	β	Std Err (β)	χ^2	*P*
Tundra X NDVI	1	4.98	1.75	8.05	0.005
Distdev	1	0.14	0.04	14.70	<0.001
Distdev^2	1	0.00	0.00	28.72	<0.001
Distdevyr	1	-0.01	0.00	3.23	0.072
Water	1	-1.16	0.50	5.31	0.021

The most supported ROC model displayed reasonable fit to the data as determined by the Hosmer-Lemenshow (2000) goodness of fit test (χ^2 = 7.93, df = 8, *P* = 0.44). The ROC score for the model was 0.78, which suggested reasonable predictive ability. Plot of the change in odds ratio as a function of distance from mine site revealed a similar relationship to the Ekati and Diavik analysis (Fig. 21). In 1999, precision of odds ratio estimates was low. In 2002, the odd ratios shifted to a closer distance suggesting lessened disturbance compared to 1999 (Table 13). In addition, odds ratio estimates also decreased, suggesting less avoidance of the Snap Lake mine site area.

Table 13. Distance (km) of mine site disturbance as determined from bootstrap analysis of the most supported AICc model.

Year	Distance	Std. Error	CV (%)	Lower CI	Upper CI
1999	66.1	3.76	5.7	57.5	73.0
2000	61.2	2.99	4.9	54.5	66.5
2001	56.3	3.20	5.7	49.5	62.5
2002	51.4	4.24	8.2	41.5	59.0

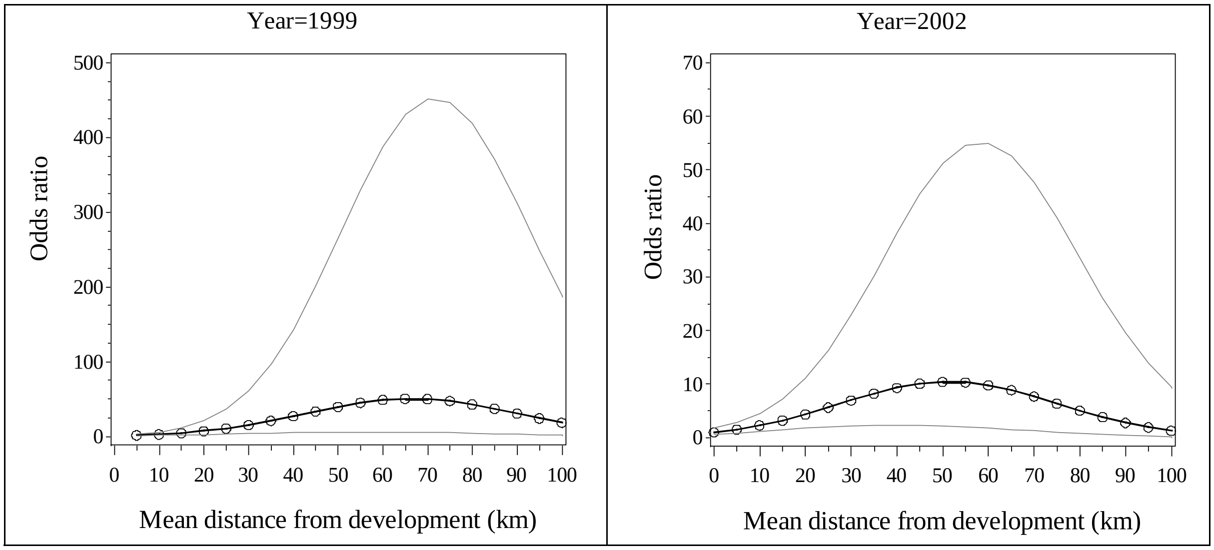


Figure 21. Relationship between distance from Snap Lake mine site and odds ratio for the large-scale analysis, Bathurst caribou, 1999–2002. All other covariates were held constant at their mean values.

For the fine-scale analysis, only 11 points (from 4 caribou) were within the availability radius of the Snap mine site during 2002 and 2003. The mean distance of used and random points was 30.9 km., (SD = 16.71, range 3.3–89.0, n = 77). We therefore did an exploratory analysis given the low sample size. Despite low sample size, AICc model selection still suggested that the Snap Lake mine site influenced probability of caribou occurrence (Table 14). As with the Ekati and Diavik analysis, a model with only a linear distance from mine site term was substantially less supported, further suggesting a threshold distance in which mine sites affect caribou.

Table 14. AICc model selection for the fine scale analysis for the Snap Lake area, Bathurst caribou, 2002–2003.

Habitat variables	Mine variables	AICc	ΔAICc	w_i	k	Log L
	Dist, dist^2	35.68	0.00	0.52	2	-15.17
Tundra	Dist, dist^2	36.18	0.50	0.40	3	-19.42
Esker, tundra	Dist, dist^2	40.54	4.86	0.05	4	-13.41
Tundra, water	Dist, dist^2	41.50	5.82	0.03	4	-13.89
Esker, tundra	Dist	44.89	9.21	0.01	3	-17.95
Esker, sedge, tundra	Dist, dist^2	46.66	10.98	0.00	5	-13.33
Esker, tundra		47.68	12.00	0.00	2	-21.17
Esker, sedge, tundra, water	Dist, dist^2	55.44	19.76	0.00	6	-13.32
Boulder, esker, forest, sedge, shrub, tundra		75.84	40.16	0.00	7	-16.92

Significance tests of model parameters for the most supported AICc model suggested that both parameters were marginally significant (distdev, $\beta = 0.419$, Std Err (β) = 0.29, $\chi^2 = 2.07$, $P = 0.15$; distdev^2, $\beta = -0.011$, Std. Err (β) = 0.001, $\chi^2 = 2.86$, $P = 0.09$). A plot of the odds ratio versus distance from the Snap mine site suggested a similar relationship to the fine scale Ekati/Diavik analysis. Bootstrap analysis suggests an effect of distance from mine site of up to 20.2 km (CI 16.6–25.0 km). As with the Ekati/Diavik analysis, low precision of estimates limits interpretation of the magnitude of the effect of distance from mine sites (Fig. 22).

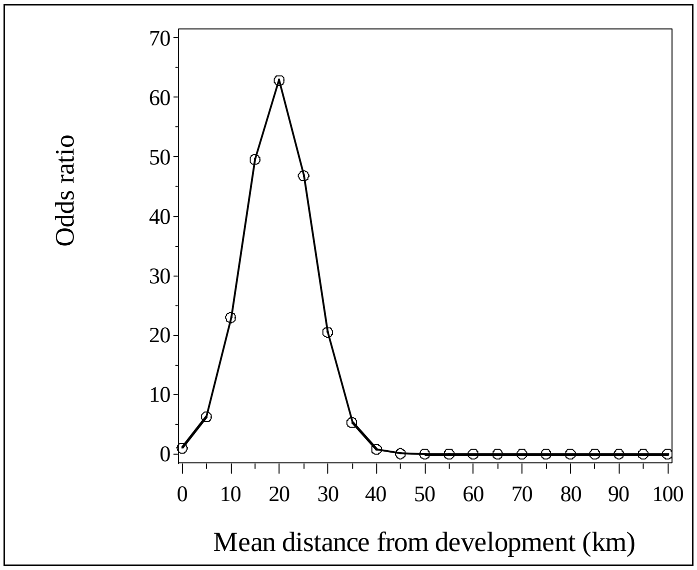


Figure 22. Estimated odds ratio as a function of distance from the Snap Lake mine site, Bathurst caribou, 2002–2003. Large standard errors prevent the plotting of confidence limits.

Discussion

Results of this analysis demonstrate that caribou selection of habitat appears to by affected by distance from mine site development. The large-scale analysis of caribou satellite collar locations suggest an influence of 50–65 km from mine sites, although this influence is not necessarily strong. Although hampered by comparatively low sample sizes, analysis of fine-scale satellite data suggest a smaller influence distance, in the 20–25 km from mine development range. However, in general, the precision of odds ratios estimates of habitat selection is relatively low and therefore it is difficult to conclusively determine the actual magnitude of change in habitat selection at varying distances from mine site. In addition, the degree of effect is sensitive to assumptions made about the temporal scale that caribou select habitat. The habitat selection analysis of Arthur et al. (1996) assumes that caribou habitat selection changes temporally and spatially. Availability, therefore, is defined by the 95th percentile of the area the caribou might potentially cover in the interval between successive fixes. Intuitively, this approach may be better than a home range based approach that assumes that caribou select habitat over their entire summer range at any given time. However, we argue that caribou may select or avoid anthropogenic features at a finer temporal scale than the duration between some of the satellite collar fixes. For example, many authors (Cameron et al. 1992, Nellemann et al. 2000, 2001, Dyer et al. 2001, Mahoney and Schaefer 2002) argue that industrial developments and roads affect *Rangifer* distribution and behavioural responses were detected at distances of <5-10 km. At these distances, the corresponding availability scale that matches this scale of selection is only for collared caribou that returned daily fixes (Fig. 14). Larger-scale analyses still might detect avoidance of mine sites; however, the ability to estimate actual avoidance distances (as estimated by the distance from mine site covariate) would be compromised if selection were occurring at a finer scale.

The results of this analysis suggest that the influence of mine sites on caribou does occur at finer scales, as suggested by the results of the fine-scale analysis (Figs. 20 and 22). If selection were occurring at a larger scale, then a linear termed model with an increasing slope across the range of distance from mine sites would most likely have been more supported by the data. The precision of the finer scale analysis is low and therefore it is difficult to conclusively evaluate these findings. However, we also note that a finer scale relationship with a closer (i.e. 20 km) distance threshold could occur within the estimated confidence limits of the larger scale distance from mine site curves (Figs. 19 and 21). Therefore, it is plausible that the estimated finer scale relationships might occur but not be detectable when the analysis is conducted at larger scales.

Interactions between insect activity indices and habitat features were poor predictors of caribou habitat selection. We suspect this is due to the fact that insect activity and caribou response occurs at a finer temporal scale than can be discerned by intermittent satellite telemetry locations. For example, we found a great degree of daily variation in insect activity (Fig. 16), and therefore it would be expected that caribou might choose habitats to avoid insects on a daily or even hourly basis. This type of short-term movement would not be readily documented by daily or weekly caribou locations.

NDVI provided a useful way to model the seasonality of habitat types such as heath tundra. It is quite likely that vascular plant forage may be influenced by green-up, the timing of which would vary on a yearly basis. Therefore, the NDVI index provides a parsimonious way to model within-year and among year variation. We found that NDVI at the pixel size classified from 2002 (1 x 1 km) was a poor surrogate for habitat type, probably because the landscape is more heterogeneous than can be discerned at the 1 x 1 km scale. In addition, caribou probably select features for reason other than greenness (i.e., insect avoidance, topography, adjacency to water bodies, etc.) and therefore the NDVI variable cannot account for this type of habitat selection.

More investigation into the scale in which availability is defined may be warranted. The method of Arthur et al. (1996) provides a useful way of considering caribou habitat availability at biologically relevant scales. However, as discussed previously, the ability of this type of analysis to discern finer scale relationships will be limited by the interval between fixes. Daily fixes were returned in 2002 and 2003 which proved useful, however, sample sizes were still marginal given the lower number of collared caribou. We suggest continued programming of caribou collars to provide at least daily fixes may help discern the scale at which caribou might be selecting or avoiding mine site areas. Ideally, the analysis might be stratified at different scales (i.e. hourly, daily, etc.), allowing assessment of model fit to the data as a function of scale.

SECTION 4: Analysis of caribou distribution relative to mine sites as measured by aerial surveys

Introduction

All three main diamond developments in the SGP have conducted systematic aerial surveys for caribou (Table 2, Appendix 1; Golder Associates Ltd 2003, BHP Billiton 2004, DDMI 2004). Limited analysis has been conducted on these data sets to examine the distribution of caribou in relative to mine infrastructure. Examining caribou survey data from 1998 to 2003, BHP Billiton (2004) concluded that the relative distribution of caribou did not change with distance from the Ekati mine and associated development during either the northern or post-calving migrations. Here we examine caribou distribution relative to the mines and associated development using a suite of habitat, biological and population parameters.

Our approach focuses on the distribution of caribou relative to mine sites as indicated by presence and absence of caribou rather than absolute or relative abundance of caribou. The rationale behind this approach is that caribou typically form social groups and usually will make decisions as a group rather than as individuals. Therefore, it is not biologically reasonable that caribou would respond to mine sites in terms of abundance. However, it is more reasonable to assume they might respond in terms of occurrence (presence or absence) at the group level (a group consisting of 1 or more animals) (Millspaugh et al. 1998). Modeling caribou response to mine sites using absolute or relative abundance as the response variable potentially adds extraneous variation into the analysis that may not be caused by the actual mine site when compared with modeling of occurrence.

Methods

As with the analysis of the satellite collar data, the objective for this analysis was the summer season when caribou are more sedentary in movements relative to mine sites. We defined this season as between July 14 and October 15 based upon arrival of caribou in summer range areas.

Data screening

When caribou were spotted during aerial transect surveys, observers recorded the GPS location of the helicopter and did not correct for distance to caribou group (Appendix 1). In addition, transect flight lines were not always strictly followed. Therefore, because the recorded locations associated with each observation were approximate, it was difficult in some cases to link an observation to an actual transect line. To confront this we estimated the distance of all observations to the closest transect line. We then observed the distribution of observations relative to the line. Theoretically, all observations should have been within 600 m or 1,000 m of the line (dependent on the assumed strip width for the given project and year; Appendix 1). In some cases observations were further than 1,000 m from the nearest transect line. We decided to not consider observations greater than 1.2 km from the centre of the line. This degree of filtering balanced the need to ensure the quality of data of the analysis without eliminating valuable observations from the analysis. Raw data were also compared with summary data files to ensure that only data from systematic monitoring surveys (as opposed to incidental sightings) were included in the analysis. Surveys in which no caribou were seen in the entire transect survey area were not included in the analysis.

Detection of disturbance

Analyses were conducted for Ekati (1998 to 2002), Ekati and Diavik combined (2002 to 2003) and Snap Lake (1999 to 2003). Using three separate analyses allowed testing for longer-term trends in the Ekati and Snap Lake areas as well as consideration of the 2 years of coordinated Ekati and Diavik surveys.

Transects segments were subdivided into successive 1 km cells that were 1.2 km wide. This width, which was 600 m on either side of the line, was the strip width associated with most of the aerial surveys. We determined the distance from mine site for all transect segments used in the analysis using the distance from the centroid of each transect cell to the centroid of each mine site. In the Ekati and Diavik transect areas, caribou could be affected by one or any of the mine developments (Ekati and Diavik mine sites, and the Misery camp) as well as the Misery road. Unlike the satellite collar analysis, distances of caribou from the various mine sites were not strongly correlated, largely because of the small scale of the analysis. Therefore, it was possible to consider models that had distances from multiple mine sites as parameters. However, it was possible that the mean distance of transect cells from mine sites or mine developments (i.e. Misery Road) might be as good a predictor of disturbance than the individual mine site distances (or the distance to the closest development). Therefore, models with mean distance (symbolized as *distdev*) from mine sites were also considered; a similar approach to the satellite telemetry analysis. In addition the closest or minimum distance (symbolized as *mindistdev*) of a caribou to a mine or road was also considered in the analysis.

We classified distance from mine site as the distance of a centroid of a transect cell to the centroid of the mine site. This did not take into account the actual footprint of mine sites, or change in the footprint of mine sites over time. Therefore, our analysis may have been more sensitive to mines such as Ekati where the footprint size increased over time. For example, the Ekati footprint has expanded over the last 5 years. Year of survey was entered as a continuous variable in the analysis for the Snap and Ekati analyses to estimate potential linear trends in the relationship between mine site distance and probability of occurrence. Year was entered as a categorical variable for the combined Ekati and Diavik analysis given that coordinated systematic surveys had only occurred for 2 years.

Use of the quadratic curve provides a simple and efficient way to approximate distances in which mine sites influence caribou occurrence. However, one attribute of the quadratic curve is that it forms a characteristic “bell shaped curve” therefore decreasing after its maximal asymptote value. Therefore, predictions from the quadratic curve should mainly be interpreted from the y-intercept of plots to the asymptote of the curve.

Habitat classification

Each cell was classified in terms of WKSS vegetation type (Ekati and Diavik) or Snap Lake vegetation type (Snap Lake; Fig. 23). Proportions of vegetation types were estimated using pixel or area counts. Vegetation types were pooled into habitat associations using the same methodology as the satellite collar analysis (refer to Table 6). Proportions of habitat types were generally similar among study areas with the exception of greater proportions of forest and less sedge wetland in the Snap Lake area.

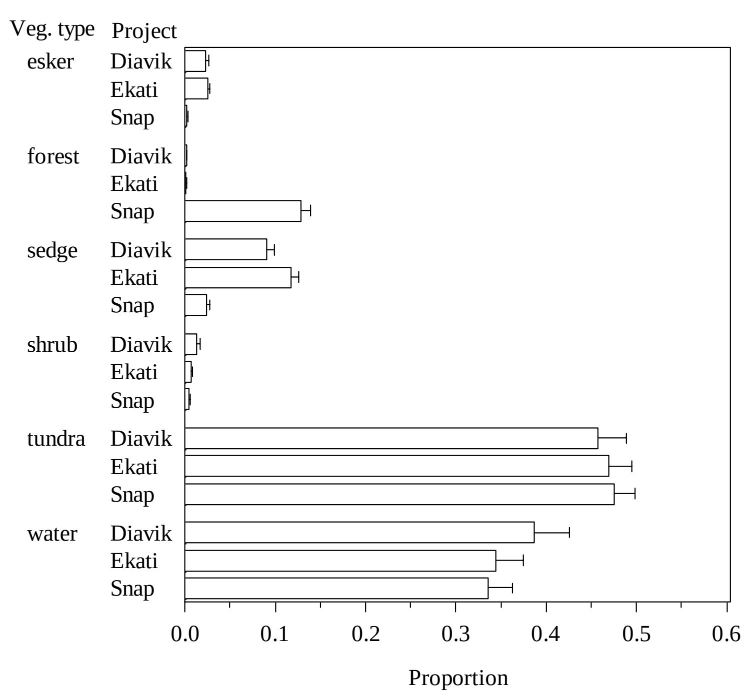


Figure 23. Mean proportion of habitat types (with associated SE) in each 1.2 km^2 cell for each of the transect study areas.

As with the satellite collar analysis, NDVI (greenness) was used to assess seasonality of vegetation types. NDVI was classified using an 8 x 8 km pixel size for 1996 to 2001 and a 1 x 1 km pixel size for 2002. Mean NDVI for pixels that occurred within the aerial survey study areas were used to model change in greenness or among years, by modeling the interaction of NDVI with tundra and sedge habitat classes. NDVI data were not available for 2003. NDVI was therefore used only with the Ekati data set (1998–2002).

Other covariates

Insect abundance was considered using methods identical to the satellite collar analysis. In this case, data from the closest weather station to each segment was used to estimate insect abundances at that particular time a survey was conducted.

The abundance of caribou in the aerial survey areas would have an obvious effect on the probability of occurrence of caribou in any of the cells; we expect that probability of occurrence would increase as population size increased. Therefore, the estimated population size of caribou on the grid was entered as a covariate. Population size was used (rather than raw counts) to account for differing strip widths for surveys in different years of the studies. Population size within each study area was estimated from counts using Jolly’s formula as presented in (Thompson 1992). Population size data were log transformed to reduce the influence of higher outlier observations.

Habitat modeling

The transect cell segments were classified with presence or absence of caribou for each survey. The presence and absence data were then analyzed using logistic regression. The objective of the analysis was to predict the potential effect of mine sites on caribou occurrence by assessing the relationship between probability of occurrence and distance of each transect cell from mine sites. Habitat variables, population size, and insect indices were then used as covariates to help separate the relationship between distance to mine sites and caribou occurrence with other natural causes of variation in caribou distribution.

We developed a-priori hypothesis about predictor variables, which allowed the building of several “candidate” models that potentially explained caribou distribution. AIC was used to evaluate the sets of candidate models. Models were evaluated using the sample size corrected AICc index of model fit. The model with the lowest AICc score was considered the most parsimonious thus optimizing the tradeoff between bias and precision (Burnham and Anderson 1992). The difference between any given model and the most supported (ΔAICc) was also used to evaluate the relative fit of models when their AICc scores were close (ΔAICc<2). SAS PROC GENMOD and LOGISTIC were used for this analysis (SAS Institute 2000).

The degree of significance of model parameters as well as AIC criteria do not allow absolute evaluation of how well the most supported model predicts caribou distribution. However, Receiver Operating Characteristic (ROC) curves can be used to compare the predictive ability of models (Cummings 2000). A ROC curve basically considers how well a model predicts presence or absence through a range of probability cutpoints. A cutpoint was the probability level in which presence or absence was declared in each cell. The ROC score varies between 0.5 and 1. A score of 0.5 would correspond to a model with no predictive ability and a score of 1 would correspond to a model with perfect predicative ability. Models with scores of greater than 0.7 are considered to be of “useful” predictive ability (Boyce et al. 2002). In addition, the goodness of fit test of Hosmer and Lemeshow (2000) was used to evaluate overall fit.

One potential issue with the data set was spatial autocorrelation due to the proximity and continuous nature of the sampling design. Two strategies were used to confront this issue. First, a generalized estimating equation model (GEE) (Ziegler and Ulrike 1998) was used to estimate correlations between successive observations on the same transect line for the most supported AIC model, and this information was used to provide adjusted variances. An exchangeable correlation matrix structure was used, which also provided an estimate of overdispersion. Type 3 chi-square tests, which are less sensitive to order of parameters in the model, were used to test for significance (SAS Institute 2000). If the overdispersion parameters were much greater than 1 then QAICc rather than AICc methods were used to select models (Burnham and Anderson 1998). Second, the most supported AICc model was further evaluated for significance using bootstrap methods (Manly 1997). For this, the data were randomly re-sampled 1,000 times to allow estimation of standard errors of regression parameters, and further evaluation of parameter significance.

We were most interested in the distance from mine site that caribou occurrence was influenced. If an effect of mine site was occurring, it would be expected that probability of occurrence would be low adjacent to the developments, and then increase and asymptote at a critical distance from the mine site. This relationship was approximated by a quadratic distance from mine term in the logistic regression equation. However, there is uncertainty in where the quadratic curve would asymptote due to sampling error and heterogeneity of landscape conditions. To estimate this uncertainty we used a bootstrap randomization procedure where a dummy data set was created in which all of the habitat and population terms in the most supported AIC model were set to mean values, except for distance from mine site that was varied across the range of observed values. The field data set was then randomly re-sampled 1,000 times and the data run through the most supported AIC model. Each model run created an estimate of the quadratic curve and estimate of distance from mine site where a potential effect was occurring (using the dummy data set values). The mean distance, standard error, and 2.5th and 97.5th percentiles (termed lower and upper confidence limits, respectively) were then used as an estimate of error around the distance at which the mine site affected caribou habitat selection.

Interpretation of model results was potentially difficult given the large degree of heterogeneity of landscape conditions. We produced maps of model predictions by imposing a grid of 1 x 1 km cells on the survey transects and immediately surrounding areas for each mine study area. Vegetation type, NDVI, distance from mine site, and other variables were classified for each cell. Model predictions were then generated for these cells to derive a probabilistic map for each study area.

Results

The interaction of insect activity indices and proportion esker habitat features was not a significant predictor of caribou distribution for any of the mine data sets (at α=0.05), and therefore these covariates were dropped from further analyses. The interaction of NDVI with tundra and sedge was a significant predictor of caribou distribution with some of the data sets.

Ekati analysis

The Ekati analysis used data from 1998 to 2002 in which 73 separate aerial surveys occurred (that counted at least 1 caribou). Presence or absence of caribou was determined for each of 387 cells for each survey. On average, caribou occurred in 9.3 (SD = 11.6; range 1–64, n = 73) of the cells each survey.

Results from the AICc model selection suggested that a model with distance from Ekati as a quadratic term, interactions between year and distance to Ekati, and distance to Ekati and Diavik as linear terms, , esker, and the log of population size, and interactions between NDVI and tundra and sedge as predictors was most supported (Table 15). A model with distance from Diavik as a quadratic term was also supported. The support for the interaction between year and distance from Ekati was most pronounced as shown by low support for models without this term. Estimates of model parameters and associated bootstrap confidence intervals and GEE type 3 chi-square tests suggested all parameters were statistically significant (Table 16).

Table 15. AICc model selection for Ekati analysis, 1998–2002.

Habitat variables	Mine variables[1]	AICc	ΔAICc	w_i	k	Log L
LogN, esker, sedge X NDVI, tundra X NDVI, water	Diavik, Ekati, Ekati^2, Ekati X year	5211.56	0.00	0.73	10	-2595.77
LogN, esker, sedge X NDVI, tundra X NDVI, water	Diavik, Diavik^2, Ekati, Ekati^2, Ekati X year	5213.52	1.96	0.27	11	-2595.75
LogN, sedge X NDVI, tundra X NDVI,	Diavik, Ekati, Ekati^2	5264.14	52.58	0.00	8	-2624.07
LogN, esker, sedge X NDVI, tundra X NDVI, water	Diavik, Ekati, Ekati^2	5265.69	54.13	0.00	9	-2623.84
LogN, esker, sedge X NDVI, tundra X NDVI, water	Diavik, Ekati, Ekati^2, misroad	5267.33	55.78	0.00	10	-2623.66
LogN, esker, sedge X NDVI, tundra X NDVI, water	Diavik, Diavik^2 Ekati, Ekati^2	5267.67	56.11	0.00	10	-2623.83
LogN, esker, sedge X NDVI, tundra X NDVI, water	Diavik, Ekati	5268.47	56.92	0.00	8	-2626.23
LogN, esker, sedge X NDVI, tundra X NDVI, water	Diavik, Ekati	5282.01	70.46	0.00	8	-2633.00
LogN, esker, sedge X NDVI, tundra X NDVI	Distdev, distdev^2	5285.93	74.37	0.00	7	-2635.96
LogN, esker, sedge X NDVI, tundra X NDVI	Distdev, distdev^2	5287.90	76.34	0.00	8	-2635.95
LogN, esker, sedge X NDVI, tundra X NDVI, water	Ekati	5290.61	79.05	0.00	7	-2638.30
LogN, esker, sedge X NDVI, tundra X NDVI, water		5292.85	81.29	0.00	6	-2640.42
LogN, esker, sedge X NDVI, tundra X NDVI, water	Mindistdev, mindistdev^2	5296.74	85.18	0.00	8	-2640.37
LogN, esker, sedge X NDVI, tundra X NDVI, water	Diavik, Ekati, Ekati^2	5299.87	88.31	0.00	9	-2640.93
Esker, forest, sedge, water		5406.23	194.68	0.00	6	-2697.12

[1] Terms Ekati and Diavik denote distance to Ekati and Diavik, respectively.

Table 16. GEE type 3 χ^2 tests and bootstrap confidence intervals for the Ekati most supported AICc model.

Parameter	Estimate	Std. Error	Lower CI	Upper CI	χ^2	*P*
Intercept	-7.092	0.374	-7.851	-6.391	13.09	0.0003
Diavik	0.019	0.005	0.010	0.028	38.33	<0.0001
Ekati	-32.264	5.060	-42.237	-22.319	5.70	0.0169
Ekati^2	-0.003	0.001	-0.004	-0.001	38.41	<0.0001
Ekati X year	0.016	0.003	0.011	0.021	15.56	<0.0001
Esker	5.144	1.029	3.060	7.125	56.87	<0.0001
LogN	0.162	0.019	0.124	0.200	17.83	<0.0001
Sedge X NDVI	8.495	1.199	6.081	10.839	41.30	<0.0001
Tundra X NDVI	4.407	0.706	3.087	5.849	8.24	0.0041
Water	0.840	0.303	0.262	1.461	13.09	0.0003

Results from the ROC analysis suggested that the model had adequate predictive ability with a ROC score of 0.71 (bootstrap confidence limits 0.69–0.73). Results of the Hosmer-Lemenshow goodness of fit test suggested adequate model fit (χ^2 = 7.24, df = 8, P = 0.47).

The quadratic curves for distance from Ekati were plotted for each year considered in the analysis (years 1998, 2000, and 2002 are shown; Fig. 24). These figures suggest that the mine influenced caribou distribution at increasing distances over time. For example, minimal impact of the mine was evident in 1998 compared to a more pronounced response curve in 2002.

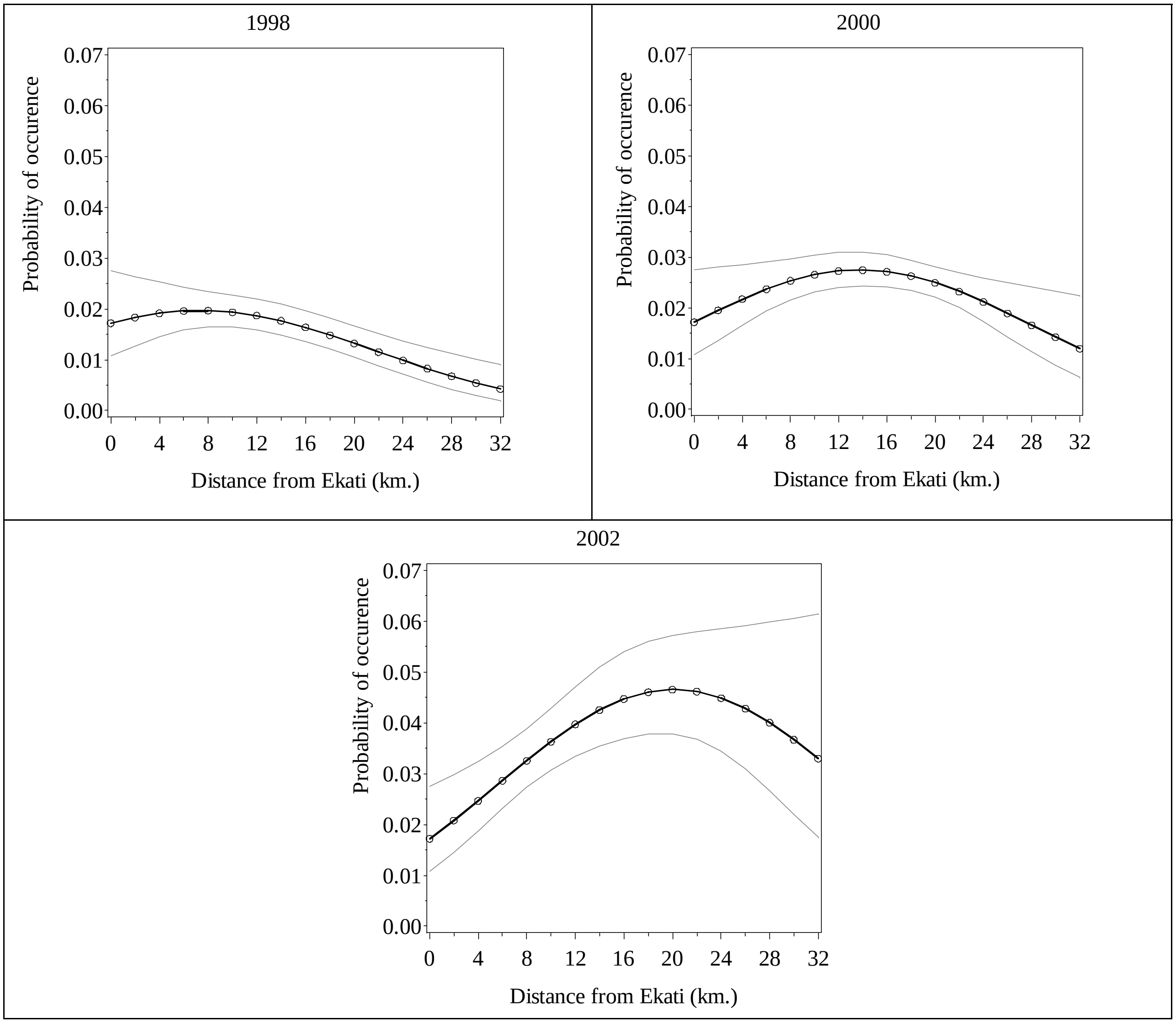


Figure 24. Probability of caribou occurrence as a function of distance from mine site as predicted by most supported AICc model in Table 15, Ekati study area, 1998, 2000, and 2002. All other parameters were standardized to mean values. Confidence intervals on curves are shown as grey lines.

Bootstrapped estimates of distance from mine influence also suggest an increasing distance of influence ranging from 7.3 in 1998 to 21.0 km in 2002 (Table 17). Coefficients of variation (CV) suggest increasing precision of distance estimates through time. This is intuitive given the more distinct shape of response curves in the latter years of the analysis (Fig. 24).

Table 17. Estimate of distance (km) from the Ekati mine effect on caribou occurrence from bootstrap analysis of AICc model, 1998–2002.

Year	Distance	Std. Error	CV (%)	Lower CI	Upper CI
1998	7.3	2.7	37.4	1.2	11.7
1999	10.1	2.6	25.3	3.5	13.9
2000	13.5	2.0	15.1	9.2	16.9
2001	17.4	2.8	15.9	13.8	24.9
2002	21.0	3.6	17.3	16.5	32.0

Model predictions were plotted for 1998 and 2002 to the mine study area. In both years, avoidance of water bodies is evident (Figs. 25 and 26). Predicted probabilities of occurrence for 1998 are homogenous or slightly clustered in the area of mine sites. Predictions for 2002 suggest higher probabilities of occurrence for areas at further distances from the mine sites. As discussed later, lower predicted probabilities at further distance from mine site could potentially be an artefact of the quadratic curve used to generate predictions.

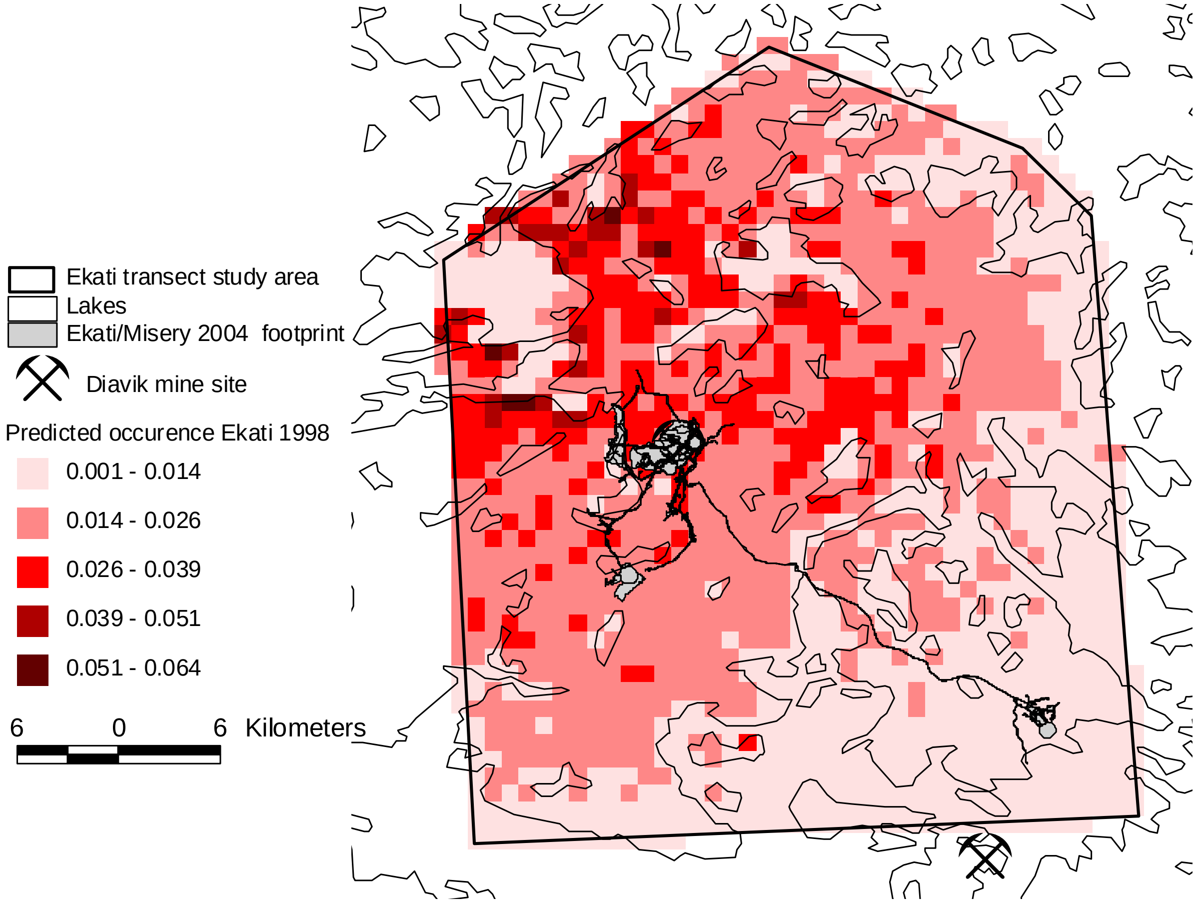


Figure 25. Predicted probabilities of occurrence for caribou for 1998 in the Ekati study area. The predictions assume a population size within the study area of 151 caribou and a mean NDVI value of 0.42.

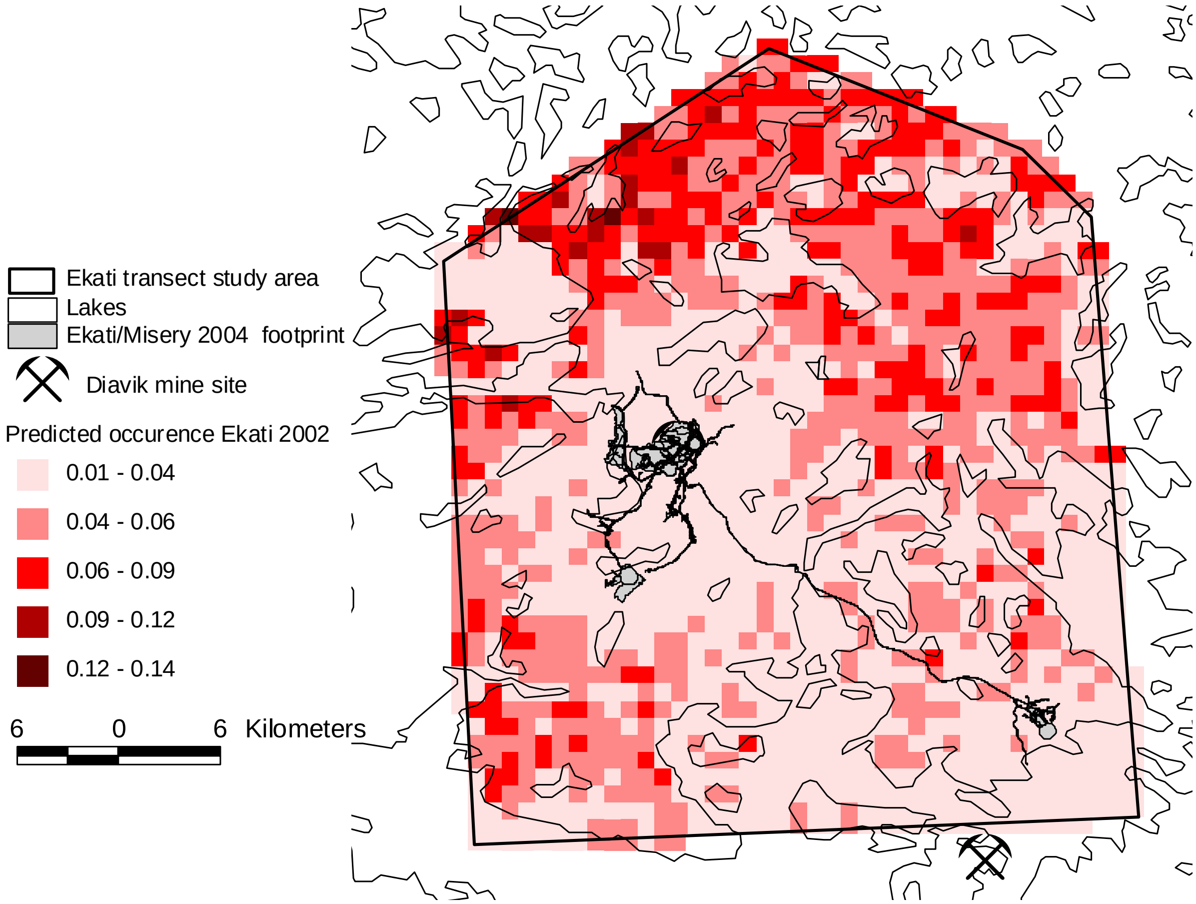


Figure 26. Predicted probabilities of occurrence for caribou for 2002 in the Ekati study area. The predictions assume a population size within the study area of 151 caribou and a mean NDVI value of 0.42.

Ekati and Diavik 2002-2003

The combined Ekati and Diavik analysis used data from 2002 and 2003 in which 17 separate surveys occurred over both areas. The transects were subdivided into 669 cells in which presence and absence was classified. On average, caribou occurred in 14.7 (SD = 14.7, range 1–32, n = 17) of the 669 cells.

The results of AICc model selection suggested several potential models were supported by the data (Table 18). The most supported models contained logN and water as habitat predictors, and linear and quadratic terms for distance from Diavik and Ekati mine sites. In addition, an interaction between year and distance from Ekati was evident in the most supported models. The most supported model was also the simplest in terms of the number of parameters. Therefore, it was considered primarily compared to other more complex supported models. Estimates of model parameters suggest all were significant with the exception of the quadratic Diavik term (Table 19). Bootstrap confidence intervals include 0 for this term, also suggesting that it is was not significant.

Table 18. AICc model selection results for Ekati and Diavik, 2002–2003.

Habitat variables	Mine variables	AICc	ΔAICc	w_i	k	Log L
LogN, water	Diavik, Diavik^2, Ekati, Ekati^2, Ekati X year	2323.99	0.00	0.45	8	-1153.99
LogN, water	Diavik, Diavik^2, Ekati, Ekati^2, misroad, misroad2, Ekati X year	2325.09	1.10	0.26	10	-1152.53
LogN, water	Diavik, Diavik^2, Diavik X year, Ekati, Ekati^2, Ekati X year	2325.74	1.75	0.19	9	-1153.89
Esker, sedge, tundra, water, LogN	Diavik, Ekati, Ekati^2, Ekati X year	2329.36	5.37	0.03	10	-1154.67
Esker, sedge, tundra, water, LogN	Diavik, Ekati	2330.28	6.30	0.02	8	-1157.14
Esker, sedge, tundra, water, LogN	Diavik, Ekati, Ekati^2	2330.56	6.57	0.02	9	-1156.27
Esker, sedge, tundra, water, LogN	Diavik, Ekati, Ekati^2, misroad, misroad^2, Ekati X year	2331.19	7.20	0.01	11	-1154.58
Esker, sedge, tundra, water, LogN	Diavik, Diavik^2, Ekati, Ekati^2, misroad, Ekati X year	2331.27	7.28	0.01	10	-1155.62
Esker, sedge, tundra, water, LogN	Distdev, distdev2	2335.40	11.41	0.00	8	-1159.69
Esker, sedge, tundra, water, LogN		2339.47	15.48	0.00	6	-1163.73
Esker, sedge, tundra, water, LogN	Diavik, Ekati, Ekati^2, Ekati X year	2340.73	16.75	0.00	9	-1161.36
Esker, sedge, tundra, water, LogN	Ekati	2341.46	17.47	0.00	7	-1163.72
Esker, sedge, tundra, water	Ekati, Ekati^2	2343.34	19.35	0.00	8	-1163.66
Esker, sedge, tundra, water, LogN	Diavik, Ekati, Ekati^2	2344.27	20.28	0.00	8	-1164.13
Esker, sedge, shrub, forest, tundra, water, LogN		2353.09	29.10	0.00	6	-1170.54
LogN	Diavik, Ekati, Ekati^2	2373.07	49.09	0.00	5	-1181.53

[1] Terms Ekati and Diavik denote distance to Ekati and Diavik, respectively.

Table 19. Parameter estimates, bootstrap confidence intervals, and significance tests for most supported Ekati and Diavik model, 2002–2003.

Parameter	Estimate	Std. Error	Lower CI	Upper CI	χ^2	*P*
Intercept	-6.068	0.650	-7.348	-4.786		
Diavik	0.067	0.033	0.003	0.133	4.32	0.0376
Diavik^2	-0.001	0.001	-0.002	0.000	1.32	0.2510
Ekati	0.050	0.023	0.004	0.096	4.90	0.0268
Ekati^2	-0.001	0.000	-0.001	0.000	4.11	0.0426
LogN	0.188	0.037	0.114	0.258	16.41	0.0001
Water	-1.806	0.258	-2.330	-1.344	46.38	0.0000
Year X Ekati	-0.005	0.002	-0.010	0.000	4.18	0.0410

ROC scores for the most supported model suggested marginal predictive ability with a ROC score of 0.68 (Bootstrap confidence interval 0.65-0.70). The Hosmer-Lemenshow goodness of fit test suggested adequate model fit (χ^2 = 10.9, df = 8, *P* = 0.21). Inspection of predicted probabilities of occurrence versus distance from Ekati and Diavik suggest a more pronounced response for Ekati in 2003 versus 2002 (Table 20 and Fig. 27). Predicted probabilities of caribou occurrence as a function of distance from Diavik suggest a response at further distances. However, this estimate should be viewed cautiously given the non-significance of the quadratic distance from Diavik term.

Table 20. Estimate of distance (km) of the Ekati and Diavik mine effects on caribou from bootstrap analysis of AICc model, 2002–2003.

Mine	Year	Distance	Std. Error	CV (%)	Lower CI	Upper CI
Ekati	2002	30.4	6.1	20.2	18.3	48.5
Ekati	2003	38.1	5.9	15.5	29.0	50.0
Diavik		44.1	6.4	14.5	31.8	50.0

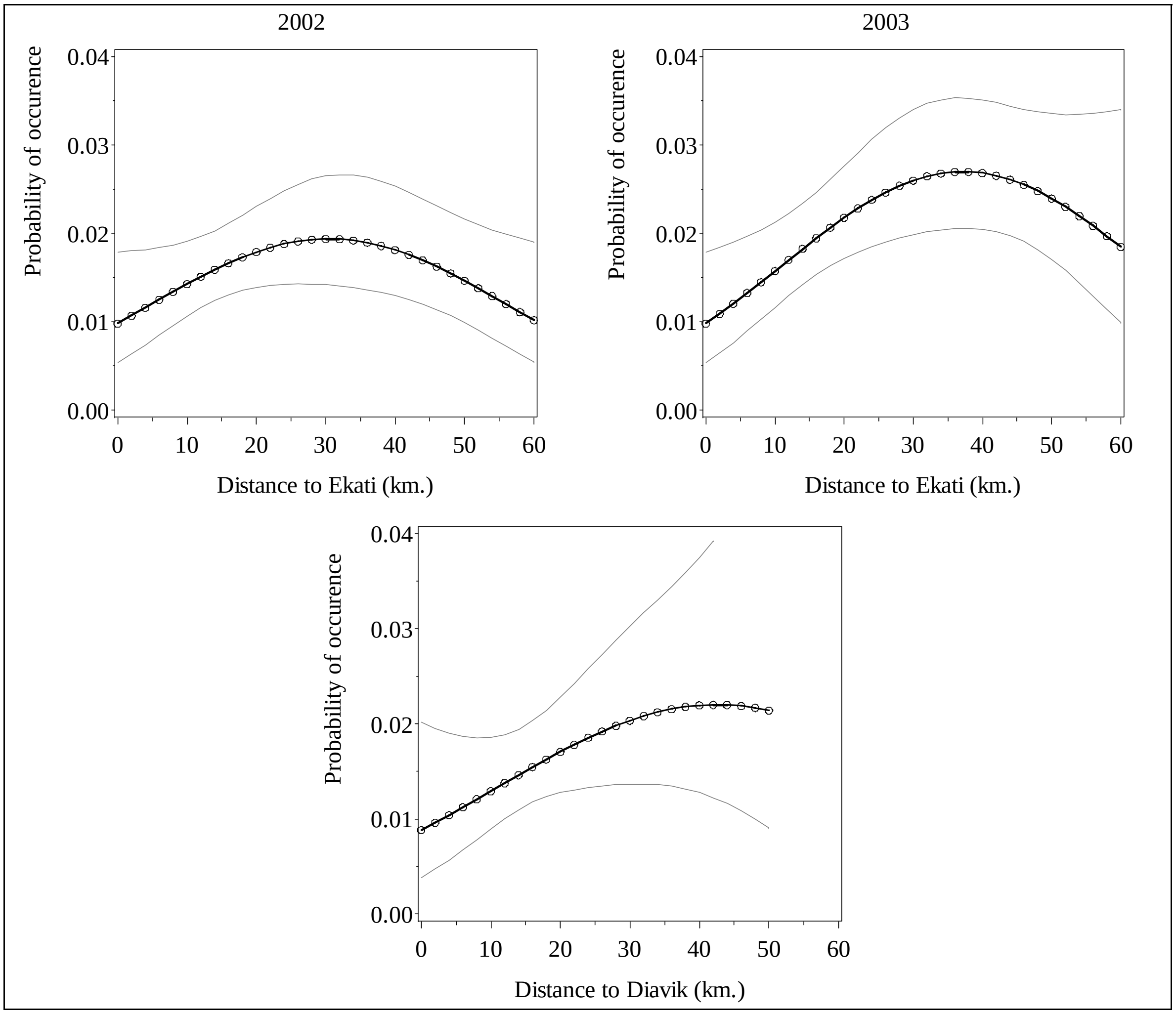


Figure 27. Probability of caribou occurrence as a function of distance from mine site as predicted by most supported AICc model in Table 19, Ekati and Diavik, 2002–2003. All other parameters were standardized to mean values. Confidence intervals on curves are shown as grey lines.

Inspection of maps of predicted probabilities suggested lower occurrences of caribou around mine sites and the Misery Road. It is possible that the Diavik term in the model partially absorbed some avoidance from the Misery Road. This is suggested when probabilities are viewed spatially. Namely, caribou occurrence is lower in the proximity of the Misery road, which comes within 7 km of the Diavik mine site. Much of the low caribou occurrence around the Diavik mine site is also due to the close proximity of large lakes (Fig. 28).

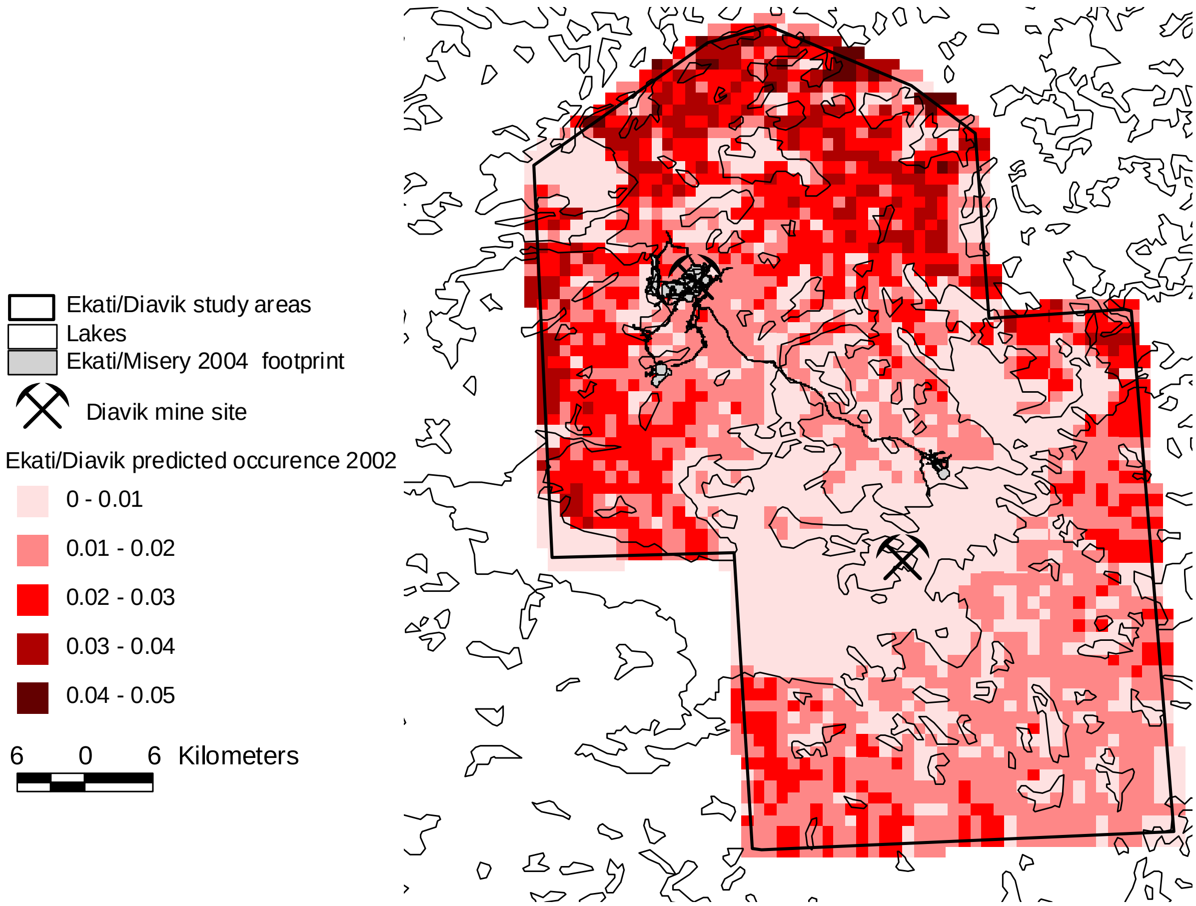


Figure 28. Predicted probabilities of occurrence of caribou for Ekati and Diavik in 2002. Predictions standardized for mean population size on grid of 302 caribou in the combined study areas.

Snap Lake 1999–2003

The Snap Lake analysis used data from 1999 to 2003 in which 15 separate surveys (in which caribou were detected) occurred. The transects were subdivided into 402 cells in which presence and absence was classified. On average, caribou occurred in 8.7 (SD = 8.21, range 1–24, *n* = 15) of the 402 cells.

AICc model selection suggested a model with logN and water with a linear and quadratic distance from Snap and a Snap with year interaction was most supported (Table 21). This simpler habitat model was substantially more supported than models that contained other habitat terms.

Table 21. AICc model selection for Snap Lake analysis, 1999–2003.

Habitat variables	Mine variables[1]	AICc	ΔAICc	w_i	k	Log L
logN, water	Snap, $Snap^2$, Snap X year	1232.55	0.00	0.93	6	-610.27
logN, esker, sedge, tundra, water	Snap, $Snap^2$, Snap X year	1237.79	5.24	0.07	9	-609.88
LogN, water	Snap, $Snap^2$	1249.74	17.19	0.00	5	-619.86
LogN, water	Snap, $Snap^2$, Snap X LogN	1251.07	18.53	0.00	6	-619.53
LogN	Snap, $Snap^2$	1252.63	20.08	0.00	4	-622.31
	Snap, $Snap^2$	1254.17	21.62	0.00	3	-624.08
LogN, esker, sedge, tundra, water	Snap, $Snap^2$	1254.98	22.43	0.00	8	-619.48
Esker, forest, logN, sedge, shrub, tundra, water		1261.22	28.67	0.00	8	-622.60

[1] The term Snap denotes distance to Snap Lake mine site.

Bootstrap confidence intervals and GEE type 3 tests suggested significance for all parameters except the log of population size (Table 22). ROC scores suggested poor predictive ability for the most supported AIC model with a ROC score of 0.65 (Bootstrap CI = 0.61–0.69). The Hosmer-Lemenshow goodness of fit test suggested adequate model fit (χ^2 = 3.02, df = 8, *P* = 0.93).

Table 22. Parameter estimates, bootstrap confidence intervals, and significance tests for most supported Snap model, 1999–2003.

Parameter	Estimate	Std. Error	Lower CI	Upper CI	χ^2	*P*
Intercept	-5.23	0.61	-6.469	-4.105		
Snap	29.69	5.28	19.467	40.290	26.10	<0.001
Snap2	-0.01	0.00	-0.009	-0.002	7.56	0.006
Snap X year	-0.01	0.00	-0.020	-0.010	25.86	<0.001
LogN	0.01	0.04	-0.065	0.094	0.07	0.791
Water	-0.80	0.34	-1.518	-0.162	6.27	0.012

Inspection of predicted probabilities as a function of distance from mine site indicate a much more pronounced response in 1999 compared to 2003 (Fig. 29). A trend of decreasing distance (suggesting less effect of the mine over time) is suggested by bootstrap distance from mine site estimates (Table 23).

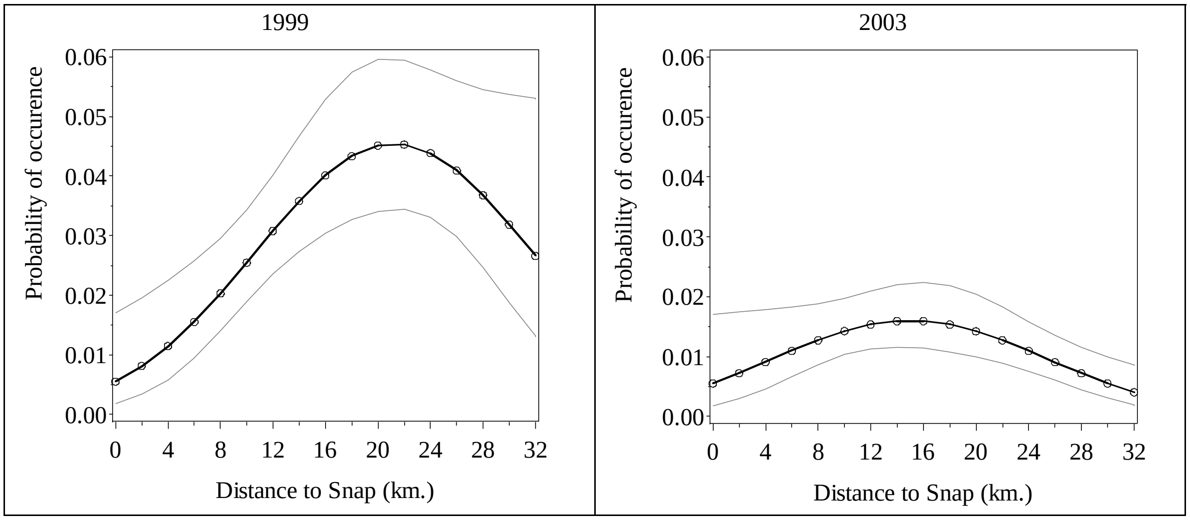


Figure 29. Probability of occurrence of caribou as a function of distance from Snap Lake mine site as predicted by most supported AICc model in Table 22. All other parameters were standardized to mean values. Confidence intervals on curves are shown as grey lines.

Table 23. Estimate of distance (km) of the Snap Lake mine effects from bootstrap analysis of AICc model, 1999–2003.

Year	Distance	SE	CV (%)	Lower CI	Upper CI
1999	21.5	2.4	11.4	18.3	28.0
2000	19.9	2.1	10.4	17.0	24.3
2001	18.1	1.8	9.9	15.0	21.3
2002	16.4	2.0	12.3	11.8	19.4
2003	14.7	2.5	16.9	7.8	18.3

Predicted map for 1999 suggests lower probabilities of occurrence for caribou around the immediate mine site area (Fig. 30). Inspection of observations suggests that there were also less observations of caribou around the immediate mine site area. Lower probabilities of occurrence on the fringe of the study area are potentially due to quadratic distance from mine effect model. A lower ROC score for this model suggests it has poor predictive ability so predicted probabilities should be interpreted cautiously.

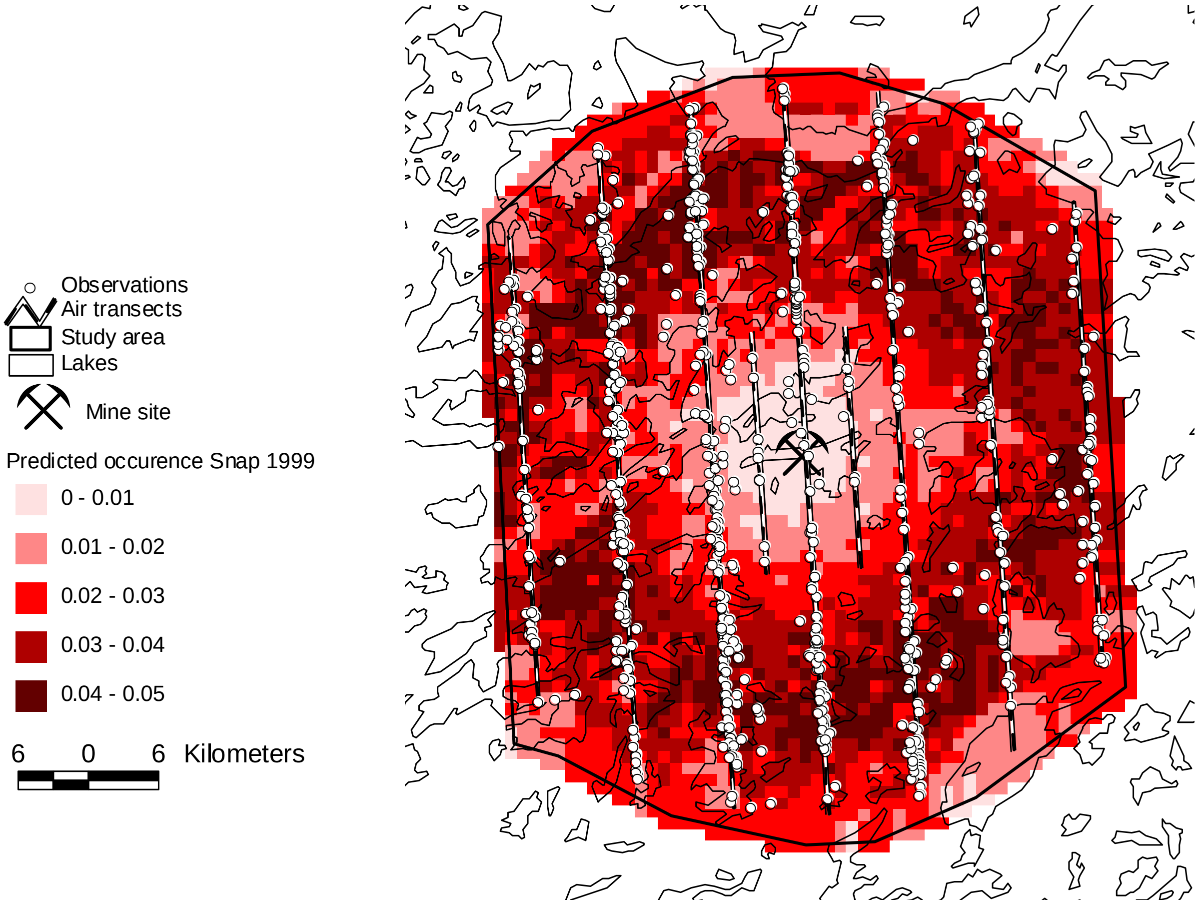


Figure 30. Predicted probabilities of occurrence of caribou for Snap Lake. Aerial transect lines and observations are also shown. Relatively few observations in the area of Snap Lake mine site are evident.

Discussion

Our analyses suggest a measurable influence of mine sites on probability of caribou occurrence (based on aerial surveys) for the Ekati study area that increased with time. The Ekati data set yielded the strongest model. An influence of the mine site was detected out to 21 km from development for 2002. The combined Ekati/Diavik data yielded a slightly weaker model that also suggested an influence of distance from mine site. The Snap Lake data set yielded the weakest model, which showed a weak and decreasing influence of mine site on probability of caribou occurrence.

It is important to consider these finding in terms of precision of estimates as well as overall fit of model. For example, the Ekati 1998–2002 analysis estimated a distance of mine effect of 21.0 km (CI = 16.5–32.0) in 2002, however the Ekati and Diavik analysis estimated this distance at 30.4 km (CI = 18.3–48.5). These may seem different, however, neither estimate is precise and their confidence intervals overlap, suggesting the difference in estimates could be contributed to chance. The estimates of distance of mine effect for Snap Lake suggest a decreasing effect of mine. However, the ROC score for this model is 0.65 (CI = 0.61–0.69), which suggests that the model is a poor predictor of caribou occurrence. Therefore, the estimates from this model should be interpreted cautiously.

An inherent assumption of this modeling effort is that the model adequately accounts for the principal factors determining caribou distribution. If this assumption is met then changes in probability of caribou occurrence as a function of distance from mine sites can be deduced to be caused by the mine site and not other factors. This assumption is difficult to test. Ideally, pre-development aerial surveys using the same methodology would provide a control data set to allow more rigorous inference. Given the changes in survey design during baseline research (Appendix 1), suitable control data were not available. Instead, we used modeling of change in probability of occurrence as a function of year of analysis as a surrogate for control data. In this case the main assumption was that change in habitat selection by caribou over time is driven by mine development. This assumption is reasonable in the case of mine sites such as Ekati, where activity and footprint size have increased over time. Analysis of the Snap Lake data set suggests less influence of the mine site on caribou distribution.

We developed a method to produce confidence intervals around estimates of distance from mine site in which caribou habitat selection was affected. This analysis suggests moderate precision of distance estimates. We suggest that this technique provides a useful way to frame estimates within the actual level of precision in the analysis.

SUMMARY DISCUSSION

In this report we have attempted to estimate effects of mine sites on caribou habitat selection and distribution. One of our main focuses was to study this problem using aerial survey data and satellite collars coupled with different methods of analyses to determine if similar conclusions could be reached from these independent data sources and statistical modeling methods. This approach resulted in more robust findings as well as insight into the strengths and weaknesses of each data type.

There were many challenges to this analysis. One of the primary challenges was lack of pre-development data. As a surrogate to pre-development data we attempted to observe how caribou movement and habitat selection changed over time as a function of distance from mine site. The inherent assumption behind these analyses is that prior to development the distribution of caribou was primarily determined by habitat factors that could be accounted for by covariates in our model. We assume that changes in caribou distribution or habitat selection as a function of proximity of mine site would be caused by mine sites as opposed to other factors. In terms of our analyses, furthest distance from mine site of mine effect (abbreviated as distance from mine site) is therefore analogous to the effect of mines. The closer the distance, the less the effect and the further the distance, the more pronounced the effect is.

We focused our analyses on the summer season when caribou herds are less migratory relative to mine sites. Caribou are most likely to behave independently and be affected by mine sites for measurable amounts of time during the summer season. During migration, large groups of non-independent animals move relatively quickly through mine areas (Gunn et al. 2001). We contend that measurement of habitat selection during this time would be problematic given that caribou movements are oriented towards or away from calving grounds as opposed to immediate habitat areas. However, this should not be interpreted to mean that mine sites may not influence caribou movements during the spring and post-calving migration (as further studied by Johnson and Boyce (2004)). We simply contend that it is difficult to measure this form of mine impact in the context of the methods we used in this paper.

Several general trends emerge from each of the analyses we conducted (Table 24). First, a trend of increasing rates of movement of caribou from the vicinity of the Ekati and Diavik mine sites was detected by the multi-strata analyses. In addition, the aerial survey habitat selection analysis suggested that caribou may be selecting habitat at further distances from these mine sites over time. Both results suggest increasing effects of the Ekati mine site. Second, assumptions regarding the temporal and spatial scale in which caribou select habitat influences the degree in which effects from mine sites are detected. For example lessened effects of mines were detected when data from caribou nearby to mine sites which returned daily satellite fixes, or aerial survey analyses were used for analyses. In contrast, greater effects of mine sites were detected when data from caribou with weekly intervals between fixes were used for the analysis; however, the precision of the magnitude of change in habitat selection caused by mine sites was low (Table 25). It is difficult to conclusively determine the most applicable scale in which caribou select habitat. Habitat selection is hierarchical, and in this study corresponds to third order selection (Johnson 1980). Caribou may select or avoid habitats at several spatial scales (Bradshaw et al. 1995, Stuart-Smith et al. 1997). However, we argue that caribou may select or avoid anthropogenic features at a finer temporal scale than the interval between some of the (earlier) weekly satellite collar fixes. For example, many authors (Cameron et al. 1992, Nellemann et al. 2000, 2001, Dyer et al. 2001, Mahoney and Schaefer 2002) argue that caribou respond behaviourally and modify their distribution relative to industrial developments and roads at distances of <5-10 km. At these distances, the corresponding availability scale that matches this scale of selection is only for satellite-collared caribou that returned daily fixes (Fig. 31) and the aerial survey data. Larger-scale analyses still might detect avoidance of mine sites; however, the ability to estimate actual avoidance distances (as estimated by the distance from mine site covariate) would be compromised if selection were occurring at a finer scale.

Table 24: Summary of results from principal analyses

Analysis/Objective	Results
Multi-strata analysis: Detect trends in caribou movement within 50 km buffer zones of mine sites (Section 1)	*Ekati/Diavik*: Weak and moderate trends in movement out of mine buffer areas for Diavik and Ekati, respectively. *Snap*: Weak trend of movement into mine buffer areas.
Comparison of distribution from satellite collars and aerial surveys. Proportion collared caribou in mine site areas estimated using estimates of total Bathurst herd (Section 2)	*Ekati/Diavik*: Estimates of proportion caribou in mine areas higher from satellite collars than aerial surveys. *Snap*: Minimal correspondence between proportion caribou from satellite collars and aerial surveys. *Low sample sizes* of collared caribou compromise analysis. *Aerial transects underestimate* numbers of caribou in mine site areas.
Satellite collar analysis: Analysis of the effect of distance from mine sites on habitat selection using RSF analysis of satellite collar data. Analysis conducted at larger (weekly) temporal (and spatial) scale and finer (daily) temporal and spatial scale. (Section 3)	*Large scale and fine scale analyses* produced different estimates of the distance in which mines affect caribou habitat selection. We argue smaller scale analysis is most applicable. *All models displayed* low precision for magnitude of effect of mine sites on habitat selection but higher precision for distance from which mine site affected habitat selection. *Ekati/Diavik*: Distance from mine sites correlated making it not possible to separate individual effects of mines. Small-scale model estimated distance of mine effect at 26.5 km. *Snap*: Small-scale model suggested distance from mine site effect at 20.2 km. A weak trend that distance was decreasing suggesting lessened mine effect over time.
Aerial survey analysis: Exploration of distance from mine effect on distribution (presence/absence) of caribou. (Section 4)	*Ekati/Diavik*: A trend of increasing distance from mine effect on caribou distribution for Ekati mine site. Distance of mine site effect increased from 7.3 in 1998 to 21.0 km in 2002. *Snap*: A trend of decreasing distance from mine site effect over time. *All aerial survey models* had low precision in terms of estimated distance of mine effects but higher precision in terms of magnitude of distance from mine effects. *General correspondence* between small-scale satellite collar and aerial survey results.

Results of analyses highlight the relative strengths of satellite collar and aerial survey data for the monitoring and detection of trends in caribou habitat selection and distribution relative to mine site. Models from the satellite collars were better at predicting caribou habitat selection as evidenced by higher ROC scores for most of the models (Table 25). In addition, use of individual caribou as the sample unit allow direct modeling of individual variation in habitat selection, and the assessment of habitat selection across multiple temporal and spatial scales (through the use of the Arthur et al. (1996) method of defining available habitat). However, the analyses were compromised by low numbers of satellite collars, and longer intervals between successive fixes for 1996–2001 years of the analysis. As a result, the precision of odds ratios estimates from models was low, making it difficult to discern the magnitude of effects that mine sites had on caribou habitat selection. The most useful years for modeling were 2002–2003 where caribou returned daily locations, allowing the modeling of habitat selection across finer temporal and spatial scales. Ideally, more years of data could be collected to allow a more direct comparison of habitat selection across many spatial scales with larger sample sizes of caribou. As with any habitat analysis, the primary limiting factor was the number of individual collared caribou, and we suggest that collar sample sizes be enhanced to allow further refinement of the finer scale habitat models.

Table 245: Predictions of distance from mine effect on habitat selection from aerial survey and satellite collar analyses

Data type	**Model predictions**						**Model diagnostics**	
	Model	**Mine site**	**Year**	**Distance (km)**	**CV (%)**	**CI**	**ROC**	**Effect CV (%)[2]**
Ekati/Diavik								
Air survey	Ekati 1998-2002	Ekati	2002	21.0	17.3	16.5-32.0	0.71	11.0
Air survey	ED[1] 2002-2003	Ekati (ED)	2002	30.4	20.2	18.3-48.5	0.68	14.6
Air survey	ED 2002-2003	Diavik (ED)	2002	44.1	14.5	31.8-50.0	0.68	26.2
Sat. collar	ED fine scale	ED	Pooled	26.5	1.6	25.6-26.8	0.79	>50
Sat. collar	ED large scale	ED	Pooled	51.9	9.9	40.4-60.2	0.72	29.5
Snap Lake								
Air survey	Snap	Snap	2002	16.4	11.8	11.8-19.4	0.65	14.5
Sat. collar	Fine scale	Snap	Pooled	20.2	16.6	16.6-25.0	0.68	>50
Sat. collar	Large scale	Snap	2002	51.4	8.2	41.5-59.0	0.78	>50

[1]Model in which data from Ekati and Diavik were both considered.
[2]Coefficient of variation of predicted odds ratio at estimated distance of mine site.

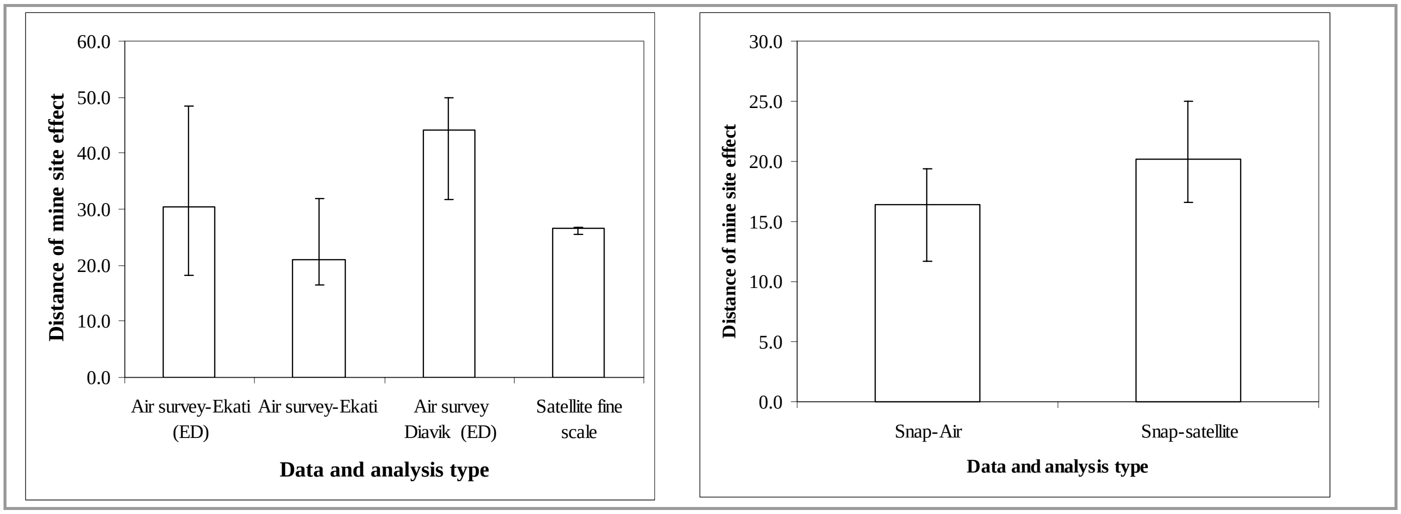


Figure 31: Comparison of estimates of mine site effect for Ekati/Diavik (left) and Snap Lake (right) analyses (Table 25). Confidence intervals are given as error bars.

The aerial survey data provided a complimentary method to model caribou distribution. We used the data to model presence and absence of caribou, which was not the original intention of survey efforts (Golder Associates Ltd 2003, BHP Billiton 2004, DDMI 2004). We contend that caribou would respond to disturbance in terms of distribution rather than abundance. By modeling presence and absence the modeling process was simplified and potentially made more powerful. One of the main issues with aerial surveys was the fixed scale of the analysis (when compared with satellite collar data). Basically, responses of caribou that went beyond the borders of transect areas were impossible to detect. We partially tested this issue by assessing the support of linear distance from mine models (that assumed caribou probability of occurrence would change evenly as a function of distance from mine site) with quadratic models (that assumed a threshold in habitat selection). In all analyses, quadratic models were more supported than the linear models, suggesting the scale of aerial survey study areas was adequate. In general, when precision of estimates was considered, aerial survey and satellite collar data did agree in terms of the effect of distance from mine sites on habitat selection at finer scales (Fig. 31).

One peripheral issue revealed by comparison of distribution of caribou from satellite collars versus aerial survey was that proportion of caribou in mine site areas was lower with aerial surveys compared to satellite collars. Many potential reasons for this are discussed of which most are difficult to control. One potential reason was that aerial surveys did not adequately count all caribou within the fixed strip widths of surveys. If this is occurring, then estimated population sizes of caribou around mine sites, as well as estimates of caribou occurrence relative to mine sites, could be negatively biased. We suggest the use of line transect distance sampling (Buckland et al. 1993) to help estimate sightability of caribou within study areas, rather than the implicit assumption that all caribou are sighted within strip width areas.

The quantification of uncertainty in terms of the effects of distance from mine site was a challenging aspect of this analysis. There are two principal types of uncertainty that should be considered. First, there is uncertainty in terms of the estimated maximum distance at which mine sites affect caribou distribution or habitat selection. The degree of error in estimates was potentially due to the shape of the quadratic curve used to estimate distance as well as sample sizes in the analysis. We used a bootstrap method to estimate this form of error and calculate confidence intervals for distance from mine site estimates (Table 25). The second source of uncertainty is the actual magnitude in which mine sites might affect caribou habitat selection or distribution. We indexed this by the coefficient of variation of estimates of odds ratios or probability of occurrence at the estimated distance of mine effect (Table 25 and Fig. 31). These sources of error were very particular to the type of analysis and type of data used in the analysis. For example, the satellite collar analysis provided reasonably precise estimates of distances of mine site affect but imprecise estimates of the magnitude of effect. In contrast, the aerial survey data provided less precise estimates of distance of mine effect but more precise estimates of the magnitude of mine effect.

Our analyses did not focus on differences in sensitivity to disturbance between sexes. The aerial survey data we used considered all caribou observed, while the satellite collar data represented females (most presumably with calves). Several studies have noted that maternal groups were most sensitive to disturbance (Murphy and Curatolo 1987, Nellemann and Cameron 1998, Nellemann et al. 2000, BHP Billiton 2004). These differences in sex and age cohorts considered in the various analyses may have contributed an unknown degree of differences to the conclusions.

Johnson and Boyce (2004) conducted a similar study of the effect of mine sites on caribou habitat selection using satellite-collared caribou data up to 2000. They considered developments in general across mainland Northwest Territories and Nunavut as opposed to individual mine sites considered in this analysis. Using this approach, habitat selection was estimated across broader geographic scales rather than areas in the proximity of individual mine areas. For example, we separated the Snap Lake mine site from Ekati and Diavik since this mine site is over 100 km away in the southern extent of caribou summer range. The analysis of Johnson and Boyce (2004) considered the effects of mine sites along with mines under development and communities in a “major development” category. In addition they considered mineral development areas, which were presumably areas of mineral exploration. They found a relationship between distance from major developments and caribou habitat selection for the post-calving season at approximately 130 km. They also found a weak relationship between mineral development areas and caribou habitat selection at 33 km. In both cases, confidence intervals for some of the disturbance model parameters overlapped 0, suggesting that relationships were relatively weak. It is difficult to compare the results of this study to those of Johnson and Boyce (2004) given that the major disturbance category considered disturbance types other than mine sites. However, our results do suggest that broader scale analyses potentially detect effects of mine sites at greater distances than smaller scale analyses.

We detected an influence of mine sites on caribou distribution out to approximately 20-25 km from mine footprints (Table 25), primarily in the Ekati/Diavik area, with minimal influence in the Snap Lake area. It is difficult to determine causative factors driving this impact, for example noise, dust on forage, etc. Changes over time may also be a result of learned behaviour. Regardless, the methods we present in this paper provide a means to measure potential impacts across multiple scales. No monitoring technique is perfect; each has inherent assumptions, strengths, and weaknesses. For this reason, analyses using multiple data sources such as aerial survey and satellite collars provide a way to contrast conclusions and determine the robustness of conclusions from analyses from any particular data source. Mine sites such as Ekati are currently expanding their footprint in areas traversed by caribou, and our analyses suggest an increase in zone of influence as development expands. A static impact of mine sites cannot be concluded. We suggest that managers may wish to continue monitoring caribou distribution using both independent data sources.

ACKNOWLEDGEMENTS

We thoughtfully acknowledge Tom Gaines' contribution to this manuscript. Tom became seriously ill after the initiation of the project, and despite severe health issues, contributed in many ways long past the time he needed to. His enthusiasm will be missed.

The NWT Department of Wildlife, Resources, and Economic Development (RWED) and the Department of Indian and Northern Affairs (DIAND) funded this study. We thank David Livingstone, DIAND, for his continued support and interest in this research. The data we analyzed were obtained from a vast and varied number of sources, and we thank the following for their assistance: Chris Hanks, Holly Newton, Matthew Otto, and Nahum Lee, BHP Billiton Diamonds Inc.; Cheryl Wray and Scott Wytrychowski, Diavik Diamond Mines Inc.; Robin Johnstone, De Beers Canada Mining Inc.; Damian Panayi, Adam Smith, and John Virgl, Golder Associates Ltd.; Francois Landry, Rescan Environmental Services Ltd.; and Vivian Banci, Banci Consulting. We greatly appreciate the assistance of RWED staff Judy Williams, and Adrian D'Hont with caribou-related data, and Alison Welsh with library knowledge. We thank Helmut Epp, Cindy Taylor, and Dave Taylor, Northwest Territories Centre for Remote Sensing, for their GIS expertise. We give special thanks to Adrian D'Hont for conducting additional GIS extractions under tight time lines. We thank Patricia Hurlburt, Geomatics Branch, Manitoba Conservations, for providing 2002 NDVI data. Brad Griffith, University of Alaska, Fairbanks, Pippa McNeil, Canadian Wildlife Service, and Tom Naughten, Parks Canada, kindly provided assistance with NDVI-related queries. Data used by the authors in this study include data produced through funding from the Earth Observing System Pathfinder Program of NASA's Mission to Planet Earth in cooperation with the National Oceanic and Atmospheric Administration (NOAA). The Earth Observing System Data and Information System (EOSDIS), Distributed Active Archive Center at Goddard Space Flight Center, which archives, manages and distributes this data set, provided the data.

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APPENDIX 1. Details of caribou location data collected during baseline and monitoring related to mineral developments in the Slave Geological Province near the range of the Bathurst caribou herd.

EKATI

Baseline data – 1994–1996.

Conducted by: Rescan Environmental Services Ltd. Detailed in BHP Diamonds Inc. (1995a, b) and BHP (1996).

Data source: V. Banci (formerly of Rescan). Some original Excel spreadsheets.

Objectives: Primary objectives were to determine occurrence, numbers and movement patterns within the Lac de Gras area, and the importance of habitats to caribou.

Time period: April to Sept–Oct 1994 to 1996.

Study area: Local, Koala area (100 km^2) and regional wildlife study area (1,900 km^2). Rectangular in shape, with longer E–W axis, with the Koala area in the southwest quadrant. Slight shifts in study area among years.

Aircraft: Helicopter and fixed-wing. Variable design. In 1994, reported to have flown at 30 m elevation and 44–50 kph, with a strip width 500 m each side of helicopter. For 1995 and 1996 flown at 60–90 m (200–300 feet) and 130 kph. Spring 1995 Cessna flights were flown at 185 kph and 150 m altitude. Two observers and one navigator/recorder.

Survey design: In 1994 surveys consisted of flying 11 linear parallel transects, 18–26 km in length, established east to west across the study area. Surveys were conducted using a helicopter during summer and fall, while a Twin Otter was used for spring surveys. Observations mapped onto topo maps. No GPS data available.

In 1995, regional study area had 15–18 transects 40 km long, arranged E – W, spaced 2 km (12 transects in the local area) or 4 km (3 transects in the regional area in the spring) apart. After spring migration, all transects were spaced 2 km apart, for a total of 18 transects. Spring surveys were flown using a Cessna 185, with 1 km strip width each side of aircraft. Flights were daily when caribou were present. Helicopter surveys used a 0.5 km width either side of aircraft. All data were geo-referenced.

In 1996, to be consistent with work occurring in the adjacent Diavik property and to follow recommendations of the Department of Renewable Resources to best capture main distribution and movement patterns, a “spaghetti” style survey path was used. This consisted of flying the perimeter of the study area, and when sign of caribou was noted, such as trails or tracks or caribou, the flight route was changed to intercept and count the caribou. Spring surveys were flown using a Cessna 185, with flights daily when caribou were present. Summer monitoring for mid-July only. All data were geo-referenced.

Survey dates: 1994: 2–3 May Twin Otter flight of the regional study area. From 21 July to October, flew helicopter transects.

1995: Spring survey using Cessna 185, flown daily when caribou were present. Remaining surveys by helicopter: June 15: Complete survey (18 transects); Aug 16: First 10 northern transects only (weather); Aug 17: Completion of survey (8 transects); Sep 5: First 10 northern transects only; Sep 6: Completion of survey (8 transects); Sep 13: no survey, incidental observation; Sep 18: First 4 southern transects only, survey cancelled due to fog; Sep 24: First 14 transects; helicopter down; Sep 25: Survey initiated where caribou first observed previous day; none observed in northern half of study area. See attached caribou flight date spreadsheet (Summer and fall data only provided; spring data are available).

1996: June 17–July 9 – few caribou; July 10–11 – large group of cows and calves arrive (10,000); location described, no GPS locations; July 12 – medium to large groups; July 13 – medium to large groups, location described; **some** GPS locations; July 14–Sept 9 – scattered caribou, small to medium groups; no GPS locations; Sept 10–Oct 2 – incidental groups described, no GPS locations; Oct 3 – one herd location; Oct 4 and 5 – several group locations. See attached caribou flight date spreadsheet (Mid-July data only provided. No GPS records after 13 July. Spring data are available).

Comments: Unable to obtain baseline data from BHPB. The data from 1994 surveys do not appear to be available. Data from 1995 and 1996 were fortuitously obtained from V. Banci, who was involved with the original baseline work. During this period, periodic summer and fall surveys were conducted, but not all GPS locations are available.

Caribou survey design differed considerably among years and often within years.

Used field studies and traditional knowledge to define two major migration routes: 1) north shore of Lac de Gras and 2) between Lac de Gras and north end McKay Lake.

Monitoring – 1997–2003

Conducted by: BHPB, Golder Associates Ltd. (during 1997, 1998, and 2000–2003), and Rescan Environmental Services Ltd. (during 1999). Detailed in Golder (1998, 1999), Rescan (1999), BHP (2001), BHP Billiton (2002, 2003, 2004).

Data source: Golder (D. Panayi, A. Smith, and J. Virgl), with permission from C. Hanks, BHPB. Original Excel spreadsheets from Golder.

Objectives: To determine the effect of mine development on relative abundance and seasonal movement patterns of caribou, and to identify major caribou movement corridors that may be impacted by the Misery road.

Time period: April/May to September/October 1997 to 2003.

Study area: Approximately 1,600 km^2, square-shaped with an extension to the north. Combined with the 1,200-km^2 Diavik study in 2002 to produce a larger regional study area surveyed using the same study design.

Aircraft: Hughes 500 and Bell 206B helicopter. Flown at 120–180 m (400–600 feet) and 145–160 kph. Two observers and one navigator/recorder.

Survey design: In 1997 the spaghetti survey design (described above) used in 1996 was used until September, at which time they switched to a systematic transect coverage. GPS locations were not recorded until 1998. From 1997 to 2001, surveys were conducted weekly from late April or May to late September, and twice weekly during peak migration periods, including when over 1,000 caribou were seen. In 2002 and 2003, surveys were flown weekly. Surveys were flown along 10 north-south transect lines, 4 km spacing apart. Transect length totalled 391 km. Survey width varied among years. In 1997–1998 they were unbounded surveys (estimated post hoc at 1 km either side of helicopter) conducted giving 50% coverage. In 1999, the unbounded survey was used for the northern migration and the survey was adjusted to 600 m either side of the helicopter for the post-calving migration (after 22 July 1999). From 2000 on, surveys were conducted with 600 m either side of helicopter (resulting in 30% coverage). Three partial transects were flown east of the main BHP study area during 2002 (extending Diavik transect lines to the north), but these are not reported in the annual monitoring report. Caribou locations were recorded using the GPS location of the helicopter (generally on transect), and not corrected for distance of caribou group off of line (up to 600 to 1,000 m on either side of the helicopter).

Survey dates: 189 surveys, including 1997, detailed in attached caribou flight date spreadsheet.

1997 – Late-Apr through early Oct (no GPS locations).

1998 to 2001 – Mid-Apr through mid-Oct.

2002 – Mid-Apr through late Sept (meshed with Diavik surveys)

2003 – Mid-Apr through late Sept, but every second transect flown from 6 June to 4 July (*n* = 5 surveys, 3 which recorded no observations). Also meshed with Diavik surveys.

Comments: No digital data from 1997.

Caribou data points not corrected for distance from the helicopter (generally on transect); therefore, point data has ± 600 m (1999 to 2003) to 1,000 m (1998 and part of 1999) accuracy.

Observer skill and training varied over the years of monitoring. Up to 10 to 12 personnel each year were involved in collecting the aerial survey data with varying degrees to knowledge and expertise. No one person has managed the caribou survey data over time.

DIAVIK

Baseline data – 1995–1997

Conducted by: Penner and Associates Ltd. Detailed in Penner and Associates Ltd. (1998a, b).

Data source: Maps and graphs in Penner and Associates Ltd. (1998a, b). GPS data not recorded or not available (see survey design below).

Objectives: Primary objectives to document the seasonal occurrence, distribution, relative abundance and movement patterns of caribou within the Regional Study Area (RSA; see below). Secondary objectives were to gather information on caribou habitat use and behavioural response to human-caused disturbances. The emphasis during caribou surveys was the documentation of caribou occurrence and movements within the Local Study Area (LSA) relative to the relative abundance, distribution and movements of caribou within the RSA.

Time period: June 1995 to August 1997.

Study area: RSA of approximately 14,000 km^2, bounded to the north by Duchess Lake and Coppermine River, to the west by the Coppermine River and Courageous and MacKay lakes, to the south by MacKay and Outram lakes and to the east by Afridi Lake, to the NE by Lac du Sauvage to Gloworm Lake. LSA of 805 km^2, primarily the islands and adjacent mainland of Lac de Gras.

Aircraft: Hughes 500 or Bell 206B helicopter. Flown at 150–300 m (500–1,000 feet) and 100–140 kph. Two observers and one navigator/recorder.

Survey design: In 1995 they used combinations of transect surveys over the LSA and more extensive surveys over the RSA. For 1996 and 1997 they switched to extensive flight patterns over the RSA, rather than transects in order to estimate the extent of widely distributed groups. Extensive survey flight paths generally included broad loops, flights parallel to shorelines, and spaghetti pattern. All caribou seen from the flight line during the surveys were counted. When caribou were encountered moving in columns or across a broad front, these movements were followed to estimate the number of animals in the movement and to map the movement route on 1:250,000 topographic maps. Trail mapping was done through ground based and aerial surveys. Coordinates of caribou locations were apparently taken, but we were unable to obtain to locate them.

Researchers also conducted 18 surveys to census the number of caribou associated with 7–10 satellite-collared cows in the area ("density-association" surveys).

Survey dates: Variable; frequency depended on number of caribou in area.

Comments: Descriptions of caribou movements within the RSA are detailed and comprehensive. Correlation with the straight-line travel routes of collared caribou showed a high degree of agreement with the caribou travel routes and movement corridors that were observed and synthesized from aerial survey data (Penner and Associates Ltd. 1998b).

We digitized information from the maps from July 1996 to July 1997 onwards to explore integrating this information into the overall digital database (see Appendix 2 for digitizing protocol). The maps consist of polygons of caribou concentrations associated (not always) with an estimated number. The maps also have arrows showing broad corridors (with estimated numbers), narrow travel routes (est. numbers) and small green arrows indicating 'small group movements'. Survey dates are given as a range, so that estimated survey dates were determined with the digitized caribou locations from these maps. There are 'total counts -all surveys' and 'estimated passage' numbers for the given date range.

To our knowledge, no systematic caribou flights and only a few reconnaissance surveys were conducted during 1998 and 1999 (A. Smith, Golder Associates, personal communication).

Monitoring – 2000-2003

Conducted by: DDMI and Golder Associates Ltd. Detailed in DDMI (2003, 2004).

Data source: Golder (D. Panayi, A. Smith, and J. Virgl), with permission from C. Wray and S. Wytrychowski, DDMI. Original Excel spreadsheets.

Objectives: Objectives of the caribou portion of the monitoring program are to determine if the zone of influence from mining activities is greater than the 3–7 km originally predicted.

Time period: April to September 2000–2003. Revisions to monitoring occurred in 2002, to conduct joint monitoring with BHPB Ekati.

Study area: 2002–2003: Approximately 1,200 km^2, square-shaped and abuts the BHPB wildlife survey area. A regional study area was defined by the combined boundaries of BHPB wildlife study area (1,600 km^2) and the DDMI wildlife study area (1,200 km^2). This regional study area was divided into four quadrants for the purpose of a combined study. The northern quadrants A and B were the Ekati Wildlife study area (east and west halves, respectively) and the southern Diavik Wildlife study area was called quadrant C. Quadrant D contains the East Island where the Diavik mine footprint is situated.

Aircraft: Helicopter. Flown at 120–180 m (400–600 feet) and 145–160 kph. Two observers and one navigator/recorder.

Survey design: During 1998 to 2001 only reconnaissance surveys conducted and primarily of the East Island, with no location data collected, or at least available to this review (2001 annual report). Systematic surveys (30% coverage) were started in 2002 (matched with BHPB) where surveys were flown weekly. In 2003, every second transect (even numbered) were flown from 5 June to 10 July, resulting in 15% coverage. As noted, for 2002–2003 the same survey spacing (parallel lines 4 km apart), survey width (600 m each side of aircraft) and standards were used as BHPB Ekati monitoring (above). Transect length totalled 284 km. Caribou locations were recorded using the GPS location of the helicopter (generally on transect), and not corrected for distance of caribou group off of line (up to 600 m on either side of the helicopter).

Survey dates: 42 surveys in total, detailed in attached caribou flight date spreadsheet.

2002 – mid April – mid September

2003 – April – September. Every second transect flown between 6 June and 4 July (minimum numbers of animals within the study area).

Comments: As best as we could determine, no systematic surveys were conducted beyond the East Island from 1998 to 2001. Caribou data points not corrected for distance from the helicopter (generally on transect); therefore, point data has ± 600 m accuracy. As with Ekati, observer skill and training varied during monitoring.

SNAP LAKE

Baseline data – 1999-2000

Monitoring – 2001-2003

Conducted by: Golder Associates Ltd. Detailed in De Beers (2002), Golder (2003).

Data source: Golder (D. Panayi, A. Smith, and J. Virgl), with permission from R. Johnstone, De Beers Canada Mining Inc. Original Excel spreadsheets.

Objectives: Objectives were to determine the migratory movements, abundance, distribution, and behaviour of caribou within the study area during the peak northern and post-calving migration periods, and to document composition of caribou groups during the post-calving migration periods. Surveys were timed to coincide with the peak movement of caribou through the area. Migration periods divided by 30 June.

Time period: April 1999 to October 2003.

Study area: Regional study area is a 31 km radius circle around the mine site (approximately 3,000 km^2). A temporary more intensive study was conducted in an 11 km radius around the mine site in 2000 (380 km^2).

Aircraft: Helicopter, primarily Bell 206B. Flown at 120–180 m (500 feet) and 145 to 160 km/h. Two observers and one navigator/recorder.

Survey design: *Baseline – 1999–2000*. Transects were spaced at 8 km, aligned north-south, resulting in seven transects of different lengths. Transect length totalled 366.5 km. To provide additional caribou coverage, aerial survey transects for caribou were doubled within an 11 km radius of the mine in 2000 only (thus spaced at 4 km in this inner area, and adding two extra 11 km long transects). Survey width was unbounded, but estimated post-hoc at 1 km on either side of the aircraft (giving 25% coverage of the study area). Off transect caribou are not recorded (i.e., on turns at end of transects). Survey started at the south end of the western-most transect, and worked east. All caribou locations were geo-referenced, but caribou locations were recorded using the GPS location of the helicopter (generally on transect), and not corrected for distance of caribou group off of line (up to 600 m on either side of the helicopter).. The dominant composition and behaviour of each caribou group and direction of movement were also recorded. In addition to caribou, snow tracks were recorded in spring 2000–2002 as a subjective estimate (high, moderate, low, none) of relative caribou tracks within 4 km sections of the transects.

Monitoring – 2001–2003. Surveys of main transects spaced at 8 km. Regular survey transects resumed in 2001 (extra local study area transects dropped), at which time survey width was set at 600 m on each side of the helicopter (giving 15% coverage of the study area). Caribou behaviour and spring snow tracks were recorded. All locations were geo-referenced.

Survey dates: 34 surveys in total, detailed in attached caribou flight date spreadsheet:

1999 – 30 Mar; 2 Apr; 21, 22 and 23 July. No caribou were seen on the 30 Mar flight and only 1 was seen on 2 Apr.

2000 – 11 and 14 Apr; 4, 7, and 10 May; 21 July; 17 Aug; no caribou seen on Apr flights.
2001 – 11 and 21 May; 8, 11 and 16 Aug; 24 Oct.
2002 – 4 and 25 Apr; 6, 9, 14 and 21 May; 23 July; 2 and 10 Aug; 30 Sept (no caribou during July flight).
2003 – 1 and 8 May; 25 and 29 July; 27 Sept; 17 Oct (no caribou observed during Sept flight).

Comments: Caribou data points not corrected for distance from the helicopter (generally on transect); therefore, point data has ± 1,000 m (1999 and 2000) to 600 m (2001 to 2003) accuracy.

Golder apparently has conducted most of the caribou flights, providing more consistent observer skill and expertise.

BATHURST INLET PORT AND ROAD (BIPR)

Baseline data –2001-2002

Conducted by: Rescan. Detailed in Rescan (2003).

Data source: Rescan (F. Landry), with permission from T. Keen (Bathurst Inlet Port and Road Project). Original Excel spreadsheets that were provided in Rescan (2003).

Objectives: To obtain information on the distribution, habitat use and ecology of caribou in the project area.

Time period: July 2001 to July 2002.

Study area: Project corridor defined as 5 km strips on both sides of the proposed road alignment, which covers a 290 km route from Bathurst Inlet, west to Contwoyto Lake, across Contwoyto Lake, and to Izok Lake.

Aircraft: Fixed-wing: Single Otter, Twin Otter, Turbo Beaver. Flown at 150 m (500 feet) and 80-100 knots. Two observers and one navigator/recorder.

Survey design: Type 1: July and August 2001; 58, 10-km long transects laid out at 5 km intervals centred on and running perpendicular to the proposed road alignment. Total transect length 580 km. Survey strip width 500 m each side of aircraft. Wildlife sightings plotted according to time and distance flown (i.e., not geo-referenced at the time).

Type 2: September 2001; three survey lines roughly parallel to the road alignment, one over the alignment, and two inside the outer limits of the 10 km wide corridor (about 4.5 km out from the road alignment). Total transect length 870 km. Same survey strip width, all points were geo-referenced.

Type 3: November 2001 and March-July 2002; two outer survey lines only (refuelling issues). Total transect length 580 km. Same survey strip width, all points were geo-referenced.

Survey dates: 11 surveys in total: 21 Jul, 29 Jul, 9 Aug, 21 Aug, 7 Sep, 20 Sep, and 8 Nov 2001, 22 Mar, 28 May, 7 Jun, and 7 Jul 2002.

Comments: Survey flight path changed due to availability of aircraft, difficulty of refuelling, and to minimize time off transect (lots of turning with survey type 1). Survey blocks designated by dividing the project corridor into 10-km sections. Caribou sightings converted to density within each block based on survey coverage within that block. This worked out to an average of 21–22% coverage for survey types 1 and 3, and 30% coverage for type 2.

Rescan (2003) noted that although only one caribou was observed during the two July 2001 aerial surveys, incidental sightings by ground crews reported 11 sightings totalling 760 animals.

TAHERA JERICHO

Baseline data – ??–??

Conducted by: Hubert and Associates? Tahera Corp.

Comments: Requested survey data. Received reworked satellite collar data with calculated distance moved between locations. Survey data was promised.

GAHCHO KUE DIAMOND PROJECT (De Beers)

Baseline data – field investigations initiated 1998, 1999, 2002.

Conducted by: AMEC

Data source: Timothy Bekhuys, AMEC

Objectives: To conduct aerial surveys to collect local and regional information on the seasonal distribution, abundance, and habitat use of caribou within the vicinity of the mine footprint.

APPENDIX 2. Diavik Baseline Maps July 1996–July 1997 – Digitizing guidelines

Geo-referenced databases depicting caribou sightings during non-systematic surveys conducted for the Diavik project baseline work are not available. Numerous maps depicting temporal changes in caribou distribution were available from a variety of Diavik reports (e.g., Penner and Associates Ltd. 1998a, 1998b). To explore incorporating these data and produce a method of recording that could be integrated with other spatially explicit data, we derived a set of guidelines to standardize data interpretation:

Survey flight lines and map annotation checked for clues to main direction of movement and point of caribou concentrations

1. Polygons: large polygons-split up and estimated number given was divided among smaller areas, small polygons- used centroid and mid-point of date range
2. More than one polygon associated with a number – divided number up between polygons
3. Travel arrow with number – use mid date and mid-point of arrow
4. Small arrows surrounding a number – points for arrows and divided number between points
5. Large arrows with number – split by date (using date range), but checked survey lines, major movements – staggered dates based on time period
6. Small green arrows with no number = 25 caribou
7. Scattered groups – 25
8. Ignored “?”
9. Direction of movement to 8 cardinal directions
10. Added more fields to database to include date range, data type, source

APPENDIX 3. Steps taken in developing AVHRR-derived Normalized Difference Vegetation Index (NDVI) 8-km resolution dataset

1. Download NOAA 14 AVHRR Channel 1, Channel 2, and NDVI files from the following web page: http://daac.gsfc.nasa.gov/data/dataset/AVHRR/01_Data_Products/03_Tenday/index.html
2. Byte-swap Ch1 and Ch2 unsigned integer values (conversion from the Unix 2 byte representations to PC 2 byte representations)
3. Convert Hierarchical Data Format (HDF) to GIS format
4. Apply gain (0.002) and offset (10) values to re-scale integer values to percent reflectance values (1 to 100%)
5. For a sample of scenes (10-day composites) calculate NDVI using the following formula: (Ch2-Ch1)/(Ch2+Ch1)
6. For the sample of scenes, apply gain (0.008) and offset (128) to the 1 byte NDVI data (downloaded). For 1 byte data no byte swapping is necessary
7. Statistically compare the generated NDVI with downloaded NDVI (11 scenes)[1]
8. Process remaining NDVI scenes (application of gain and offset to rescale NDVI values to the –1 to +1 range)
9. Re-project data from Goode projection to Lambert Conformal Conic projection, using supplied projection specifications
10. Convert raster NDVI composites to vector representation
11. Extract NDVI values matching date of provided satellite and random locations with the 10-day NDVI composite periods (15 periods per year, May 1st till September 31st)
12. Convert vector data to ArcView shape files
13. Convert extracted data file to Excel

We developed the 1 km NDVI maps by applying the re-scaling coefficients to the scaled (16-bit unsigned) raw NDVI data provided by Patricia Hurlburt, Geocomp Program Manager, Geomatics Branch, Manitoba Conservation. The raw NDVI values were calculated from Top of Atmosphere reflectance data (TOA).

In re-scaling, the following formula was used:

NDVI = A0 + DSL * A1

Where

A0 = -1

[1] We found almost perfect correlation between the tested scenes (r^2 between 0.995 to 1.0)

DSL = Digital Signal Level

A1 = 0.0001

After re-scaling, the raster NDVI maps (10-day composites, in PCI format) were converted to ArcView shape format.

Notes about AVHRR data and NDVI index (all information from AVHRR web pages):

The first AVHRR channel is in a part of the spectrum (0.58 to 0.68 um) where chlorophyll causes considerable absorption of incoming radiation, and the second channel is in a spectral region (0.73 to 1.10 um) where spongy mesophyll leaf structure leads to considerable reflectance. This contrast between responses of the two bands can be shown by a ratio transform; i.e., dividing one band by the other. Several ratio transforms have been proposed for studying different land surfaces (Tucker 1979). The Normalized Difference Vegetation Index (NDVI) is one such ratio, which is highly correlated with vegetation parameters such as green-leaf biomass and green-leaf area and, hence, is of considerable value for vegetation discrimination (Justice et al. 1985).

NDVI Relationships With Geophysical Variables:

A ratio between bands is of considerable use in reducing variations caused by surface topography (Holben and Justice 1981). It compensates for variations in radiance as a function of sun elevation for different parts of an image. The ratios do not eliminate additive effects caused by atmospheric attenuation, but the basis for the NDVI and vegetation relationship holds generally. The soil background contributes a reflected signal apart from the vegetation, and interacts with the overlying vegetation through multiple scattering of radiant energy. Huete (1988) found the NDVI to be as sensitive to soil darkening (moisture and soil type) as to plant density over partially vegetated areas.

NDVI 10-day composites:

The Composite Data Set contains the same data layers as the Daily Data Set. The Composite Data Set is similar to the daily data in structure and is derived from the Daily Data Set, however the process of “compositing” removes much of the cloud cover present in the Daily Data Set. The composite is generated by comparing the NDVI values for each 8 km bin from 10 consecutive Daily Data Sets. Because data at the edge of a scan may contain distortion and bidirectional effect biases, only data within 42 deg of nadir are used in the composite. **The pixel with the highest NDVI for the 10 days is chosen as the date for the inclusion in the composite**, and all 12 data layers are updated with data from that date. This compositing process is effective for removing most of the clouds and atmospheric contaminants, thus providing as close to a cloud free field in each of the data layers as is possible. However, in areas of persistent cloudiness, cloudy pixels will remain.

There are three composites per month. The first composite of each month is for days 1 to 10, the second composite is for days 11 to 20 and the third composite is for the remaining days. This convention was chosen so that these data could be used with many climatologies and meteorological data, which are provided in monthly averages.

Literature cited

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