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ESTIMATING THE ZONE OF INFLUENCE OF INDUSTRIAL DEVELOPMENTS ON WILDLIFE: A MIGRATORY CARIBOU AND DIAMOND MINE CASE STUDY

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Abstract. Wildlife species potentially respond to industrial development with changes in distribution, however, discerning a response to development from differences in habitat selection is challenging, and often differences in methodologies make comparison of studies problematic. Since the early 1990's, the summer range of the migratory tundra Bathurst caribou (*R. tarandus groenlandicus*) herd in the Canadian Arctic were exposed to the construction of three diamond mines. We used an innovative statistical approach to directly estimate the zone of influence (area of reduced caribou occupancy) of the mines during mid-July to mid-October. We used data from aerial surveys, and locations of satellite collared cow caribou as inputs to a model to account for patterns in habitat selection. We then constrained the zone of influence curve to asymptote, such that the average distance from the mine complex where caribou habitat selection was not affected by the mine could be estimated. Around the Ekati-Diavik mine complex during the operation period for both mines we detected a 14 km zone of influence from the aerial survey data, and a weaker 11 km zone from the satellite-collar locations. Caribou were about four times more likely to select habitat at greater distances from the mine complex than within the zone of influence. The implications are that caribou are responding to industrial developments at greater distances than shown in other areas, possibly related to dust deposition from mines. The methodology we developed provides a standardized approach to estimate the spatial impact of stressors on caribou or other wildlife species.

Key words: Arctic, barren-ground caribou, diamond mining, industrial disturbance, Rangifer tarandus groenlandicus*, resource selection functions, likelihood, zone of influence.*

1 Introduction

The impact of industrial development on wildlife is a frequent and worldwide concern, and this is especially true for long-distant migrants whose traditional routes can be threatened by industrial developments (Berger, 2004). Of particular interest is the relative spatial displacement of wildlife caused by a response to human activities. Many methods have been used to measure displacement, but often comparing findings is complicated by different methodologies and scales of disturbance considered (Starikowich, 2008). Differences in results from analyses of the same data sets can trigger controversy (Noel et al. 2004, Joly et al. 2006), which detracts from effective conservation and mitigation measures for species that are potentially impacted by industrial development.

We became interested in measuring potential displacement of migratory tundra caribou when investigating the impact of mine development on the Bathurst caribou herd on the central Canadian tundra (Northwest Territories). Migratory tundra caribou are a gregarious and migratory ungulate with ecological similarities to other open habitat, gregarious ungulates in Africa and Asia that face industrial developments on their ranges (e.g., Mongolian gazelles [*Procapra gutturosa*]; Ito et al., 2004).

The Bathurst caribou herd has declined since 1996 at an average annual rate of 5% (Nishi et al., 2008) and therefore assessment of cumulative effects of industry, harvest, and other stressors is of immediate concern. From the mid-1990s onward, Bathurst caribou have been exposed to a boom in mining exploration, which culminated in the construction of two open-pit and one underground diamond mine within the Northwest Territories. During environmental assessment hearings for the diamond mines and subsequent public meetings, strong concerns were expressed about how the mines would affect caribou movements and distribution, and the overall health of the herds (Boulanger et al., 2004; Johnson et al., 2005).

The distance where caribou change their behavior, habitat selection, and distribution relative to disturbance, which we term "zone of influence", has implications for measuring the cumulative effects of various stressors on caribou populations, especially where there are multiple mines and associated exploration activities (Duinker and Greig, 2006). Previous estimates of the zone of influence were mainly based on frequencies of caribou relative to distance from disturbance (e.g., Nellemann et al., 2000; Mahoney and Schaefer, 2002; Joly et al., 2006) or polynomial-based estimates (Boulanger et al., 2004; Johnson et al., 2005; Golder Associates Ltd., 2008a, 2008b). Each of these approaches has limitations. The frequency approach does not necessarily account for habitat factors that might influence distribution, and can be influenced by the choice of frequency classes. Polynomial-based methods, which fit a curvilinear curve to observed caribou selection or occurrence, do account for differences in habitat selection, but the polynomial curves only approximate the hypothesized asymptote in habitat selection caused by reduced caribou occurrence. For example, it would be expected that caribou selection should increase with distance from mine then asymptote where the mine has no impact. Polynomial methods allow selection to change non-linearly with distance but do not exactly asymptote, and often zone of influence is measured as the peak of a quadratic or cubic curve. Estimates of displacement for the Bathurst caribou herd using satellite collar and aerial survey data using polynomial-based methods ranged from 17 km (Boulanger et al., 2004; Golder Associates Ltd., 2008a, 2008b) to 130 km (Johnson et al., 2005).

We suspected that the large difference in zone of influence reported by studies was due to the effect of scale (ranges of distances considered in the analysis), and uncertainty in the exact distance due to the curvilinear nature of polynomial curves. We therefore developed a likelihood-based approach that fit the hypothesized asymptotic relationship, therefore estimating the exact distance at which mines affected caribou distribution while accounting for variation caused by differential habitat selection within the vicinity of mines. We also explored a possible mechanism for the zone of influence by considering the effects of dust deposition from mine activities on caribou distribution.

We note that this general methodology is applicable to the measurement of response to disturbance of any wildlife species given that it is based upon general habitat selection methods and likelihood based analysis models. We suggest that our methodology may help conservation measures by allowing a standardized zone of influence shape to be fit, therefore making results among different studies more equitable.

2 Materials and methods

2.1 Study area

The study was centered on the tundra of the central Arctic (~64°30′ N, 110°30′ W), approximately 300 km northeast of Yellowknife, Northwest Territories, Canada (Fig. 1). The area occupied by the caribou 15 July–15 October is about 100,000 km^2, with a high use area (70% kernel) of about 53,000 km^2, and a core (50% kernel) of about 33,000 km^2. The study area is within the Southern Arctic ecozone, an area of continuous permafrost (Ecological Stratification Working Group, 1996). Glaciers have largely shaped the landscape, which has esker complexes, boulder moraines, raised ridges of ancient beaches, and numerous lakes. Riverine habitats and seepage areas are the most productive habitats. Shrub communities of willow (*Salix* spp.), shrub birch (*Betula* spp.), and Labrador tea

(*Ledum decumbens*) dominate areas with adequate soil development. Mats of lichens, mosses, and low shrubs are found across exposed rocky and gravel sites. The climate is semi-arid with annual precipitation of approximately 300 mm. Summers are short and cool with average temperatures of ~12°C whereas winter temperatures are commonly <–30°C (BHP Diamonds, 1995).

The Bathurst herd of migratory tundra caribou annually moves hundreds of kilometers from wintering ranges below treeline, to calving and summer range on the open tundra (Gunn et al., 2001). Between 1996 and 2006, the herd declined from an estimated 349,000 (± 95,000 [SE]) to 128,000 (± 27,300) caribou (Nishi et al., 2008). The seasonal migrations of the Bathurst herd annually varies (Gunn et al., 2001), which causes the number of caribou in the vicinity of the mines to fluctuate. The northward spring migration to the calving grounds is usually rapid. During post-calving and summer, caribou either move rapidly in response to parasitic insect harassment (Russell et al., 1993) or movements are less while caribou feed. Movements away from the vicinity of the mines occur after the fall rut, and by October, few caribou generally occur in the area. It is during the post-calving through summer seasons that the potential influence of the mines is expected to be the greatest. We have therefore restricted our analyses to 15 July to 15 October (hereafter termed the summer season).

We analyzed caribou distribution relative to three existing diamond mines within the Northwest Territories: Ekati (BHP Billiton Diamonds Inc.), Diavik (Diavik Diamond Mines Inc.), and Snap Lake (De Beers Canada; Fig. 1). The main Ekati mine and Diavik are 30 km apart. Both mines are open pit mines with accommodation complexes and ore-processing buildings (the mines are fly-in operations). Ekati has a separate camp and open pit (Misery) which is connected by a 29 km all-weather road to the main Ekati site. The Misery camp and pit are 7 km from the Diavik mine, which is restricted to an island in Lac de Gras. Snap Lake mine is a more recent and an underground mine (Table 1) 105 km south of Diavik. Because of the juxtaposition of the Ekati and Diavik operations, we modeled these mines as a combined unit. Analysis of mines separately resulted in zone of influences that overlapped the two areas suggesting that the zone of influences of the two mines were confounded (J. Boulanger, unpubl. data).

The scale of our analyses was based on satellite collar data (ENR, unpubl. data). Most caribou cows occur within 100–150 km of the Ekati and Diavik mines near Lac de Gras during this period, while caribou distribution are generally not distributed more than 40 km south of Snap Lake (Fig. 1).

2.2 Caribou data sources

The first source of location data was from weekly aerial surveys using systematically spaced strip transects from Ekati (1998–2008), Diavik (2002–2008), and Snap Lake (1999–2008) (Table 2). Transect route, spacing and width, study area size, and frequency of data collection varied within and among mines, but mostly was a systematic (4- or 8-km spacing) coverage of 15–30 km radius study areas out from mine sites, flown by helicopter at 150 m altitude and 145–160 kph. Transect width was 600 m on both sides of the aircraft. Number of aerial surveys with caribou present varied annually and among mines (Table 2). For analysis we considered surveys where >1 cell had caribou present (>0.2% relative occupancy per survey), resulting in 168 useable aerial surveys flown between 1998 and 2008. For these surveys the mean relative occupancy (number of cells where caribou were detected/number of cells surveyed) was 5.1% (SD = 6.4%, range 0.3–41.0%).

The second source of caribou locations was from satellite transmitters attached to collars fitted to adult cow caribou tracked from April 1996 to October 2008 (Gunn et al., 2001; Environment and Natural Resources, unpublished data). The number of collared caribou available annually for analysis (that potentially encountered the mine sites [see *Treatment of satellite collar data*, below]) ranged from 4 to 19 (Table 2). The satellite collars varied from transmitting every 7 days beginning in 1996, to every 5 days beginning in 1998, with the addition of daily duty cycle for mid-July to mid-August beginning in 2002. We used 3,705 point locations during our period of interest (57.1% daily, 36.9% 5-day, and 6.0% 7-day) from an annual average of 11.5 (±1.25) individual cows.

2.3 Habitat classes

To provide seamless coverage of habitat classes over our study area we used the Land Cover Map of Northern Canada (NLC; Olthof et al., 2008), and Earth Observation for Sustainable Development of Forests (EOSD; http://cfs.nrcan.gc.ca/subsite/eosd/mapping) land cover classification. Esker coverage was extracted from 1:250,000 scale National Topographic Data Base maps (Natural Resources Canada; http://geogratis.cgdi.gc.ca/geogratis/en/product/search.do?id=8147). We used 12 habitat classes pooled between the NLC, EOSD and eskers coverages. We converted linear eskers into polygons with standardized width of 100 m. (Descriptions of the habitat classes are in Appendix A.)

2.4 Plant productivity

Plant phenology and productivity annually vary which could influence caribou use of habitats and movement patterns (Russell et al., 1993). 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. We used 1-km resolution NDVI amalgamated by 10-day composite periods for 1996 to 2006 (Latifovic et al. 2005), and calculated the mean values for each 1 x 1 km cell within the study area.

2.5 Dustfall

Caribou respond to and avoid vehicle and aircraft traffic, and the presence of people, machinery and buildings – a generalized response to predators (Frid and Dill, 2002). Additionally, aboriginal elders have repeatedly identified dustfall from mine activities as a concern for caribou through deposition on forage plants (Independent Environmental Monitoring Agency 2006–07 annual report, Yellowknife, NWT). Most of the larger dust particles are deposited within 100s of meters from the sources and affect vegetation composition (Myers-Smith et al., 2006). However, CALPUFF dispersion modeling in the Ekati and Diavik areas predict that smaller particles (total suspended particles [TSP] ~10 μm in size) will be deposited over a wider area and only reach background deposition rates (15 kg/ha/yr) 14–20 km from source (Rescan, 2006). Given that scale of effect, we included dustfall as a covariate in our analyses. The model generated isopleths of dust deposition, and we interpolated the grid values between successive contours (20 to 5000 kg/ha/yr). A value of 0 was assumed to occur 5 km outside of the 20 kg/ha/yr contour based on the average distance between contours 20 and 50 and adjusted for the interval increment.

2.6 Treatment of aerial survey data

We applied resource selection functions (Manly et al., 2002) to assess habitat and the effects of mine sites on caribou distribution from both aerial survey and satellite collar data. We treated the aerial survey observations as presence and absence of caribou rather than absolute abundance to minimize the effect of contagious behavior and group size (Millspaugh et al., 1998). We compiled the observations of presence or absence into successive 1 km cells that were 1.2 km wide, and calculated the proportion of habitat classes within each cell. We determined the distance from mine site for all transect cells used in the analysis using the distance from the centroid of each transect cell to the centroid of each mine site. When outlying components of the Ekati development were added (Misery and Fox pits), the distance to the nearest development component was used.

A potential issue of the sequential cells was spatial autocorrelation. We used a generalized estimating equation model (GEE) (Ziegler and Ulrike, 1998) to estimate correlations between successive observations on the same transect line for the most supported base habitat model, and produce empirical robust standard error estimates. We used an exchangeable correlation matrix structure to account for spatial autocorrelation. Type 3 chi-square tests, which are less sensitive to order of parameters in models, were used to test for significance (SAS Institute, 2000). We used ROC curves to estimate the goodness of fit for 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). We used SAS (SAS Institute, 2000) PROC GENMOD or PROC LOGISTIC for all analyses.

The abundance of caribou varied annually and seasonally, which created variation in habitat selection. We therefore used the relative abundance of caribou on the survey area, as indexed by the number of cells where caribou were detected relative to the number of cells sampled, as a “nuisance” predictor variable. This essentially eliminated the influence of abundance on habitat selection.

The design of the aerial surveys (survey area, coverage, flight details) varied among the mines. We explored the effect of survey design by estimating the interaction of different designs (as a categorical variable) and the estimated zone of influence (βzoi) predictor variables.

2.7 Treatment of satellite collar data

We determined the proportion of habitat types in a 1 km buffer radius (the maximum error of the satellite collar locations) around collar locations. Then we compared each buffered point with the buffered area around six random points that were within a circle around the previous location of the collared caribou. The circle was the “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 et al., 2005). Caribou possibly select habitat at a finer scale than that reflected by the availability radius, as the radius depends on the time between successive telemetry fixes. For this reason, we considered the interaction of each habitat variable with the scale of availability. This accounted for potential scale effects and allowed all the data to be simultaneously considered in a single analysis. Locations from caribou that potentially encountered the mine sites (as indicated by the availability radius) at least once in a given year were included in the analysis.

We compared caribou location points (used) and random points using conditional logistic regression (Hosmer and Lemeshow, 2000). The analysis defined each used and six accompanying random points as a cluster. This cluster centered each comparison on the habitat available to the caribou at the time at which the location was taken. This approach avoided issues with psuedoreplication caused by pooling telemetry data from different caribou (Pendergast et al., 1996; Johnson et al., 2005). We used k-fold cross validation to test goodness of fit of the used-random satellite collar data (Boyce et al., 2002). For this analysis, we subdivided the data into training and testing data sets based on Huberty's rule of thumb (Huberty, 1994). The goodness of fit of a model developed with the training data set was then tested with the testing data set. We estimated the Pearson correlation (Zar, 1996) of successive RSF score bins with the frequency of used locations in each bin (adjusted for availability area of each bin). If the model fitted the data then the RSF bin score and area-adjusted frequencies should be positively correlated (Boyce et al., 2002).

2.8 Base habitat model fitting procedure

We used logistic regression for the aerial survey and satellite collar data to estimate habitat selection. The response variable was binary corresponding to use/nonuse (aerial survey) or used/random (satellite collar). Firstly, we applied univariate tests to determine the statistical significance of individual habitat predictor variables (Hosmer and Lemeshow, 2000). The general form of the model was:

> Binary response = habitat variable + habitat variable^2 + habitat variable*movement rate + habitat variable*season + habitat variable*mean NDVI score + buffer scale*habitat variable (satellite collar analysis only).

The quadratic term (habitat variable^2) tested for situations when stronger associations with habitat values were likely to occur in the midpoint of the habitat variable value as opposed to a linear relationship. The interaction between movement rate and habitat variables was tested for cases when a habitat was used transitionally as indicated by a significant relationship between movement rate and the given habitat variable. We used the interactions among seasons (early summer, late summer, fall, and rut/late fall) and NDVI to test for seasonal selection of habitats. We also tested the satellite collar data for interactions between availability radius (duration between fixes [duty cycle] which determined the size of the buffer where available locations were placed) and habitat variables as discussed previously. Habitat variables were standardized to allow easy interpretation of slope coefficients and to minimize potential issues with varying measurement scales.

Significant variables from univariate tests were then added into a multivariate model in the same order as the univariate model (i.e., linear habitat variable, then habitat variable*movement rate etc). The fit of individual terms was evaluated by Type 3 chi-square tests and empirical standard error estimates (SAS Institute, 2000). From this, a base habitat model was derived, which was then used to test for the zone of influence of mine sites.

We entered TSP as a covariate to the base model for the Ekati/Diavik area, generated predictions of the odds ratio of habitat selection relative to TSP levels, and contrasted these results with zone of influence predictions. Data from 2003–2008 were used for this analysis under the assumption that this corresponded best to the time in which TSP levels were measured (i.e., both mines were in operation).

2.9 Estimation of the zone of influence of mine areas

To test for zone of influence, we used the base habitat model with a “zone of influence” predictor variable (symbolized as ZOI) and associated regression coefficient (β_{ZOI}). We sequentially tested increasing zones of influence by allowing the zone of influence to equal the distances of present/not detected (aerial survey data) or used and random (satellite collar data) locations up to a hypothesized zone of influence distance by

0.5 km increments (i.e., 0.5 km, 1.0 km, etc.) after which point the zone of influence variable was set equal to the hypothesized zone of influence for further distances. For example, when a 1.5 km distance was tested, all presence or used locations beyond 1.5 km were set to 1.5 km, regardless of how far out they were. By doing this, the odds ratio of selection relative to the mine site (as estimated by distance from mine*β_{ZOI}) was allowed to change linearly up to the hypothesized zone of influences at which point it would asymptote and remained constant for distances greater than the zone of influence (as estimated by ZOI*β_{ZOI}) (Fig. 2). The overall fit of each sequential zone of influence distance model was assessed by its log-likelihood. If fit was improved by the β_{ZOI} term, then the log-likelihood should increase to an optimum at the statistically most probable zone of influence before decreasing at larger distances (Fig. 2). If there were no zone of influence, then the log-likelihood would remain constant across the range of distances. The distance at which in which the log-likelihood was maximized was, therefore, the estimate for the zone of influence (i.e., the maximum distance where an influence of the mine on caribou distribution could be detected). In addition, the relative magnitude of the difference in habitat selection caused by the mine could be estimated by the odds ratio of habitat selection at the estimated zone of influence ($OR_{ZOI} = e^{(\beta_{ZOI} * ZOI)}$). The odds ratio in this case was the relative increase in habitat selection at distances further than the zone of influence relative to habitat selection within the zone of influence.

The relative shape of the likelihood curve assessed the strength of the zone of influence. For example, an irregular shaped likelihood curve, or a curve without a peak indicates that other spatial factors were influencing caribou selection relative to the mine (and that were not already accounted for in the base habitat model). Confidence intervals for the likelihood curve were constructed from the range of zone of influence distances in which the log-likelihood was within 1.92 of the maximum likelihood zone of influence (Hudson, 1971; Hillborn and Mangel, 1997).

We also analyzed the effect of temporal changes in mine activity by grouping years into periods of broad mine development. To retain sample size, we combined data for 1996–99 (1998–99 for aerial survey analysis), 2000–02, and 2003–08 (when Ekati and Diavik were both in operation) (Table 1). We also accounted for the expanding footprints of mines by adding the Misery pit and road to the footprint in 2000 and the Fox Pit to the footprint in 2003. We conducted a sensitivity analysis on both data sets to examine the influence of Misery road construction and operation on the zone of influence by comparing zone of influence estimates with and without Misery road for the 2000–08 time period.

Some studies have suggested that groups with calves (nursery groups) are more sensitive to disturbance than groups without calves (non-nursery groups) (Nellemann and Cameron, 1998; Nellemann et al., 2000; Cameron et al., 2005; Joly et al., 2006). To explore this we used the aerial survey data for Ekati and Diavik collected from 2003–08 and compared the estimated zone of influence between nursery and non-nursery groups. We assigned groups as nursery where composition was noted, and where no composition was noted, assumed all groups ≥50 caribou were nursery groups.

3 Results

3.1 Aerial survey analysis

Ekati-Diavik mine complex.—The multivariate base habitat model overall fitted the data with a ROC score of 0.793 (See Appendix B for results on the base habitat modeling). We initially estimated a zone of influence corresponding to all of the years of data collection (1998–2008). The zone of influence model terms were significant for the pooled Ekati-

Diavik complex (Z = 8.85, P < 0.002) and the overall fit of the model was adequate (ROC = 0.795). The asymptote of the likelihood curve corresponded to an estimated of zone of influence of 14 km (CI = 12.0–15.5 km) (Fig. 3).

Survey design also affected zone of influence estimates as suggested by a significant interaction of design and zone of influence term (χ^2 = 20.25, df = 2, P < 0.0001). We set all predictions to correspond to the aerial design in which both Ekati and Diavik were simultaneously surveyed under the assumption that this was the best data set to estimate zone of influence for the pooled mine complex. We estimated odds ratios of the zone of influence effect for the Ekati-Diavik mine sites, which suggested caribou were 4.2 times (SE = 1.08, CI = 3.60–4.85) more likely to select habitat at distances greater than 14 km from the mine areas (Fig. 3).

The zone of influence predictor terms (β_{zoi}) were significant (combined Ekati and Diavik) but differed among the three periods of mine development (Table 3, Fig. 4). In the initial time period (1998–99: Ekati construction) a weak zone of influence was evident at 4 km. In the middle period (2000–02: Ekati operation and Diavik construction) no zone of influence was evident, as indicated by a lack of peak in the likelihood curve. In the final period when both mines were in operation (2003–08; seven pits in total), a zone of influence was evident at 14 km (CI = 13.0-15.0 km) from the mine site, which was similar to the pooled estimate (Table 3, Fig. 3).

Of caribou groups observed in the Ekati-Diavik area from 2003–08, 271 were nursery groups and 1,453 were non-nursery groups. We did not detect a statistically significant difference between zone of influence for nursery groups (ZOI = 12 km, CI = 10.5–16.0 km) and non-nursery groups (ZOI = 14 km, CI = 12.5–15.5 km). Odds ratios

(OR) were also not significantly different between groups (nursery: OR = 3.32, CI = 2.08–5.31; non-nursery: OR = 5.21, CI = 4.37–6.20).

Removing Misery road as part of the mine area effectively increased the distance from mine area for caribou groups sighted on transects that were between the core Ekati and Diavik mine areas. As a result, zone of influence estimates during 2000–08 without Misery road (ZOI = 18 km, CI = 15.5–20.0 km) were slightly increased compared to estimates with Misery road (ZOI = 15 km, CI = 13.0–16.0 km).

Snap Lake mine.—The habitat base model for Snap Lake was significant with a good fit to the data (ROC = 0.80) (See Appendix B for results on the base habitat modeling). The pooled analyses among years suggested a weak zone of influence of 6.5 km (CI = 1–25 km) with a relatively weak odds ratio of 2.4 (CI = 1.88–3.12). Although the zone of influence term was marginally significant ($\chi^2 = 2.57$, df = 1, $P = 0.085$), the outer confidence limits of the zone of influence almost encompassed the mine aerial survey area (31 km radius). None of the period-specific zone of influence terms were significant (α = 0.1). Sample size was limited: Only four aerial surveys detected caribou during 2005–2008.

3.2 Satellite collar analysis

The base habitat model displayed adequate fit to the data as determined by Pearson correlation of area-adjusted frequencies and ordinal odds ratio bins ($\rho = 0.902$, $P < 0.0001$). (See Appendix B for results on the base habitat modeling.) This base habitat model was used for both the combined Ekati-Diavik and Snap Lake zone of influence analyses.

Ekati-Diavik mine complex.—The proportion of daily fixes for the satellite collar locations increased after 2001, which resulted in higher densities of used points during 2003–08 (Fig. 5). Although the caribou satellite collar locations were fewer near mine areas and then peaked from 25–50 km from the mines before decreasing at further distances, habitat influences such as lakes were affecting the distribution as well as the mine activities.

Analysis of zone of influence by time period suggested changes in the zones of influence over time (Fig. 6, Table 4). A zone of influence of 23 km (CI = 19-35 km.) was evident for the early period (1996–99) of the Ekati-Diavik complex development, however, the odds ratio of the zone of influence was considerably less than 1, indicating attraction to the mine areas rather than avoidance. Inspection of the raw data revealed congregations of caribou near mine areas in August–September 1996 and July–August 1999 that may have caused this trend. A zone of influence of 3 km (CI = 1-39) was evident for the middle period (2000–02), with an odds ratio of 2.26 (CI=1.32-225.7) suggesting avoidance, however, the confidence limits on the ZOI estimate were large (1–39 km). A zone of influence of 11 km (CI = 1–17 km) was evident for 2003–08 when both mines were in operation, with an odds ratio of 3.9 (CI= 1.6 - 10.1) also suggesting avoidance of the mine areas.

The precision of zone of influence estimates and odds ratio estimates were generally lower for satellite collar data (Table 4) than for aerial survey data (Table 3). Years 2003–08 had the highest sample size of collars (Fig. 5) and may be the best representation of the current zone of influence of the Ekati-Diavik mine areas.

Snap Lake mine.—Estimation of zone of influence for the Snap Lake area was challenged by low sample sizes of collared caribou. On average, the availability radius of 8.5 caribou (SD = 4.76, range 1–16, n = 11 years) was within the Snap Lake mine site given that the area is on the southern fringe of caribou summer range. For all years combined, the zone of influence likelihood curve suggested a zone of influence at 37 km (CI = 19–56 km); however, the odds ratio for the zone of influence was 1.4 with the confidence interval overlapping 1 (CI = 0.77–2.86), suggesting either aversion or attraction to the mine site. These results suggested that the zone of influence was not statistically different than random variation in habitat selection. Period-specific analysis for the Snap Lake mine area was not conducted because of low sample sizes.

3.3 Dustfall and the zone of influence

The CALPUFF model generates isopleths of dust deposition, which predicted that TSP declines rapidly >2 km from mine development and were indistinguishable from background deposition rates at a distance of 14–20 km from the Ekati-Diavik mine complex.

Using aerial survey data, the log of TSP as a covariate for the base Ekati-Diavik habitat model was a significant predictor (χ^2 = 117.13, df = 1, P < 0.0001) and the resulting model had a ROC score of 0.795, which suggested predictive ability. Plots of predictions suggested a steep decline in the odds ratio of caribou occurrence at relatively low levels of TSP (i.e., 100–200 kg/ha/yr) (Fig. 7). A similar analysis for the satellite collar data using only caribou locations that were within 50 km of the Ekati-Diavik mine complex indicated the log of TSP was also a significant predictor (χ^2 = 13.88, df = 1, P = 0.0002). This suggests that caribou will avoid areas with even low levels of TSP, which can occur at distances up to 14–20 km from mine areas.

4 Discussion

A large number of studies have attempted to address anthropogenic impacts on ungulates (Nellemann et al., 2003; Stankowich, 2008), but often results vary based upon methods used and scale of the sampling design. We developed an adaptable methodology that should allow better comparison among studies by the fitting of the exact hypothesized zone of influence curve that is not influenced by how the data are binned, and less influenced by scale of analysis. Our method can be applied to any procedure that estimates likelihood scores. It therefore allows the estimation of a zone of influence using underlying flexible, robust, habitat modeling procedures, such as conditional logistic regression or generalized estimating equations, that account for potential sampling biases (such as autocorrelation) and other habitat and population factors that might influence distribution. However, our approach still requires that a range of distances are sampled that encompass both anthropogenic impacts as well as natural habitat variation to allow an estimate of the asymptote of the zone of influence curve. We argue that the requirement of adequate survey scale to measure both impact and non-impact is fundamental to the design of any study that is attempting to estimate anthropogenic impact.

Our analyses suggest that caribou respond to disturbance at a large spatial scale, and that this response can be estimated using both aerial survey and satellite collar data. Strengths of our analyses compared to other published accounts of caribou and other ungulate species being displaced by industrial development were that firstly, we used two independent data sets (aerial surveys and satellite collars) that came up with similar results. Secondly, our analyses used base habitat models that accounted for patterns in habitat selection, as we tested the goodness of fit of the base habitat model without the zone of influence variables. Thirdly, we used a mathematical technique that constrained the zone of influence curve to asymptote, such that the average distance from mine complex could be estimated. A fourth strength of our approach was that we considered collar frequency of transmission in the analysis; more frequent (daily) locations allowed a more fine-grained analyses. Finally, our analysis suggests potential mechanisms for aversion to mine areas at larger distances in the form of dust (total suspended particle) deposition.

4.1 The overall impact of mines on the Bathurst caribou herd

The zones of influences that were detected in this study suggest that mines have a biologically significant impact on the distribution of caribou on their summer range, and the magnitude of the zone of influence is related to the relative level of activity at mine areas. A zone of influence around the Ekati-Diavik mine complex was detected based on aerial survey data, such that probability of caribou occurrence and selection of habitat were reduced close to mine development. This reduced occurrence was most evident during the operation phase of both mines (14 km, CI = 13-15, 2003–08), and less evident during initial operation of Ekati and construction of Diavik. Caribou were about four times more likely to select habitat at distances greater than 14 km from the mine complex (Table 3). A weak zone of influence of 6.5 km (CI = 1 – 25 km) was detected at the more recently constructed Snap Lake mine using the aerial survey data. Satellite collar data produced similar results; an 11 km (CI = 1-17 km.) zone of influence for the Ekati-Diavik complex, but no significant zone for Snap Lake. However, we note that Snap Lake is on the edge of typical caribou summer distribution, which reduced the sample size for the analyses.

Caribou habitat selection scales from fidelity to the overall summer range down to finer scales within that overall fidelity. We conducted model runs to ensure that we had not confounded the different scales of habitat selection. For example, we ran a model with satellite collars data that extended up to 100 km from the Ekati-Diavik area and found that log likelihoods initially peaked at the estimated mine zone of influence (~11 km), but then peaked again at larger distance from mine values (~70 km) with negative odds ratios suggesting selection for the larger area around the mine. In the case of larger distances, the zone of influence model was estimating the core of summer range, as also indicated by the highest used point densities (Fig. 5) rather than the zone of influence of the mine area. An inherent assumption of the zone of influence model is that the base habitat model accounts for any spatial variation in habitat selection, and that the primary factor influencing habitat selection relative to mine sites is the effects of mines. Inspection of likelihood plots and associated odds ratios of β_{ZOI} can provide an assessment of the overall adequacy of the zone of influence model and the presence of other gradients or factors that confound zone of influence estimates.

The area of reduced caribou occurrence from the Ekati-Diavik mine complex is ~6.7% of the 33,000 km^2 core and ~4.2% of the high use area of summer range of the Bathurst herd; cumulative impacts from other sources of disturbance on the landscape (Johnson et al., 2005) could have wider implications to the ecology and health of the herd (Nellemann et al., 2000; Cameron et al., 2005; Vistnes and Nellemann, 2008). In addition, we are unable to estimate the proportion of the herd that is affected by development, and thus the population-scale costs are unknown (Wolfe et al., 2000). We suggest, however, that our results depict clear separation of the effects of development from natural variation in habitat use.

4.2 Aerial survey versus satellite collar data

The aerial survey data provided the strongest analysis of zone of influence. However, although less influenced by larger summer range selection gradients, these surveys were constrained by the extent of survey area. Our modelling assumed that the areas surveyed encompassed both the zone influenced by the mine and areas beyond the influence of the mine to allow an estimate of the asymptote of the zone of influence curve. Even in the early years of the Ekati-Diavik monitoring, and in all surveys from Snap Lake, there was reasonable coverage out from development (~22 km for Ekati-Diavik, and 31 km for Snap Lake). The aerial survey data were not corrected for sightability bias (Buckland et al., 2004), but we assumed this had little impact on the analyses, as we used presence-absence rather than absolute numbers.

The satellite collars provided less precise estimates of zone of influence, largely due to limited sample sizes (resulting in less data available for areas near the mine) and less frequent duty cycles for the early years of study. Thus, contrary to suggestions by Vistnes and Nellemann (2008), we propose that satellite collars may not provide for the most effective analyses of habitat use on temporal and spatial scales relative to human activity and infrastructure in open, Arctic environments. This is because sample size (number of individuals) is usually low in telemetry studies relative to the large areas covered. In contrast, aerial survey transects sample areas adjacent to mine sites uniformly, therefore providing a more consistent indication of presence and absence of caribou relative to mine areas.

4.3 Limitation of analysis

The spatial arrangement of the Ekati and Diavik mines and Misery road limited our ability to estimate feature-specific zones of influence. For example, the Misery road connects the main Ekati mine site and Misery pit, which is 7 km from the Diavik mine (Fig. 1). The zone of influence estimates for the Ekati-Diavik mine complex effectively included the entire Misery Road. Therefore, it was difficult to determine if caribou aversion of the Misery road area was due to the road, or the overall effects of the Diavik and Ekati mine areas.

Infrequent aerial surveys (≤3 per year) and low numbers of satellite-collared caribou hampered analyses of the zone of influence around the Snap Lake mine, which is a consequence of the mine being near the southern edge of late summer and early fall range for Bathurst animals (Gunn et al., 2001). We were unable to subset the Snap Lake aerial survey or satellite collar analysis to shorter time periods, for instance to 2005–08 when the mine was constructed and the beginning of operations in 2008. We suspect that an underground mine operation would have less impact on surrounding caribou distribution compared with larger open-pit operations. Golder Associates Ltd. (2008b) used polynomial techniques on the same aerial survey data to conclude a 17 km zone of influence around the Snap Lake mine; however, the confidence interval (95% CI = 6.6–42.3 km) suggested weak support from the data.

4.4 Comparison of results with other caribou studies

Most regional studies reveal that *Rangifer* reduce their use of areas within 1–10 km of development (Murphy and Curatolo, 1987; Wolfe et al., 2000; Nellemann et al., 2001; Mahoney and Schaefer, 2002; Cameron et al., 2005; Joly et al., 2006; Weir et al., 2007; Vistnes and Nellemann, 2008). We suspect that it is the scale of our analyses that allowed us to detect a larger zone of influence than previously published responses distances. However, our study addressed the effects of large open pit mines, which would present a very different configuration of stimuli to caribou than, for example, a road or tourist lodge. The open tundra habitat likely allows caribou to respond at a greater distance, however, other studies such as at the Prudhoe Bay oilfield were also on tundra post-calving ranges (Wolfe et al., 2000; Vistnes and Nellemann, 2008).

Earlier analyses of the Bathurst herd using polynomial methods suggested larger zones of influence around diamond mines (~17–30 km, out to 130 km; Boulanger et al., 2004; Johnson et al., 2005; Golder Associates Ltd., 2008a, 2008b). One potential issue with the polynomial approach is that other habitat selection gradients, which occur beyond the zone of influence, can potentially influence the overall shape of the curve. For example, satellite collar data indicate a steep gradient of habitat use evident at distances past 50 km from mines as indicated by declining point densities (Fig. 5). A quadratic curve fit to these data would be influenced by both the gradient from mine zone of influence but also the other gradients, which would cause the peak of the curve to be shifted to the middle of the gradient. A zone of influence based on the peak of the quadratic curve would therefore be over-estimated due to the influence of the other gradient. We suspect this issue may have caused the relatively large zone of influence estimates of Johnson et al. (2005).

4.5 Potential mechanistic causes for zone of influence

How wildlife such as caribou respond to human activity is likely patterned as a response to predation risk (Frid and Dill, 2002), which includes the trade-offs between countering predation risk without risking other behaviors. Overall, response distances vary as the nature of the disturbances, methods to describe the responses, and environmental variables such as insect harassment or foraging conditions differ among and within studies. Some studies have suggested the greatest incremental impacts of development occur during initial construction of roads and related facilities (Nellemann and Cameron, 1998). Our analyses suggest less detectable impacts during construction and initial operation, which may be attributable to a learned behavior or accumulation of factors causing the avoidance behavior.

The scale at which caribou are selecting habitat relative to the imposed scale of measurement is also likely a mechanistic factor in determining the extent of influence of mines. Mayor et al. (2009) concluded that in winter, Newfoundland caribou were selecting against snow conditions and for lichens at distances of up to 15 km, which the authors related to the perceptual abilities of caribou. Haskell and Ballard (2008) suggested that caribou habituate to roads on an annual basis; however, these results were based upon 1 km roadside surveys of caribou abundance. The small scale of distances considered in their study make it difficult to evaluate potential larger scale shifts in caribou distribution caused by oilfield activities.

Our results suggested that the zone of influence of mines was at a greater distance (14 km) than was explainable by a predation risk patterned response. A factor that fits the scale of the response is dustfall. Although dustfall has been described for its effects on vegetation (Meyers-Smith et al., 2006), little is known about the response of herbivores to dust on forage. The mines use an atmospheric transport model (CALPUFF) to predict TSP deposition rates in excess of 5000 kg/ha/yr (1360 mg/m^2/day) close to mine activity in summer. Deposition rates decrease rapidly with increasing distance from mine activities, however, our analyses suggest that caribou avoid habitats with even lower levels of TSP. While caribou distribution around the immediate mine area may also be affected by sensory disturbance, we suggest that the larger zones of influence for caribou (i.e., 14 km) does correlate with the predicted geographic scale of dustfall.

4.6 Conclusions

Our results suggest a quantifiable zone of influence from diamond mines on caribou distribution that may be related to both behavioral disturbance and possibly the effect of dustfall on vegetation. These results suggest that researchers studying impacts of anthropologic development on caribou and other wildlife species should consider a larger range of scales than those caused by immediate behavioral responses to noise or other smaller-scale disturbances. In addition, alternative larger-scale impacts, such as dust deposition on forage, should be considered in addition to behavioral responses that have been the main focus of past ungulate studies (Stankowich 2008).

The methods developed in this manuscript can be further applied to explore the effects of anthropogenic disturbance on other wildlife species by allowing a robust estimate of displacement while accounting for variation in habitat selection and scale effects. We suggest that this standardized robust approach for assessment of anthropogenic impact will allow further development of monitoring and mitigation measures to manage the impact of mines and other developments on wildlife species.

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References

Arthur, S.M., Manly, B.F.J., McDonald, L.L., Garner, G.W., 1996. Assessing habitat selection when availability changes. Ecology 77, 215–227.

Berger, J., 2004. The last mile: how to sustain long-distance migration in mammals. Conservation Biology 18, 320–331.

BHP Diamonds., 1995. Ecological mapping: 1995 baseline study update. BHP Diamonds Inc. Yellowknife, NWT.

Boulanger, J., Poole, K., Fournier, B., Wierzchowski, J., Gaines, T., Gunn, A., 2004. Assessment of Bathurst caribou movements and distribution in the Slave Geological Province. Northwest Territories Department of Resources, Wildlife and Economic Development, Manuscript Report 158. 108 pp. http://www.enr.gov.nt.ca/_live/documents/documentManagerUpload/caribou_movements.pdf

Boyce, M.S., Vernier, P.R., Nielsen, S.E., Schmeigelow, F.K.A., 2002. Evaluating resource selection functions. Ecological Modelling 157, 281–300.

Buckland, S.T., Anderson, D.R., Burnham, K.P., Laake, J.L., Borchers, D.L., Thomas, L., 2004. Advanced distance sampling - estimating abundance of biological populations. Oxford Press.

Cameron, R.D., Smith, W.T., White, R.G., Griffith, B., 2005. Central Arctic Caribou and petroleum development: distributional, nutritional, and reproductive implications. Arctic 58, 1–9.

Duinker, P.N., Greig, L.A., 2007. Scenario analysis in environmental impact assessment: improving explorations of the future. Environmental Impact Assessment Review 27, 206–219.

Ecological Stratification Working Group., 1996. A national ecological framework for Canada. Environment Canada, State of the Environment Directorate, Ecozone Analysis Branch, Ottawa, Ontario, Canada.

Frid, A., Dill, L.M., 2002. Human-caused disturbance stimuli as a form of predation risk. Conservation Ecology 6, 11. [online] http://www.consecol.org/vol6/iss1/art11/

Golder Associates Ltd., 2008a. Analysis of environmental effects from the Diavik Diamond Mine on wildlife in the Lac de Gras region. Submitted to Diavik Diamond Mines Inc., Yellowknife, NWT.

Golder Associates Ltd., 2008b. Snap Lake mine: analysis of environmental effects on wildlife 1999 to 2007. Submitted to De Beers Canada Inc., Yellowknife, NWT.

Gunn, A., Dragon, J., Boulanger, J., 2001. Seasonal movements of satellite-collared caribou from the Bathurst herd. Final report to the West Kitikmeot Slave Study Society, Yellowknife, NWT. 80pp

Haskell, S.P., Ballard, W.B., 2008. Annual re-habituation of calving caribou to oilfields in northern Alaska: implications for expanding development. Canadian Journal of Zoology 86, 627–637.

Hillborn, R., Mangel, M., 1997. The ecological detective: confronting models with data. Volume 28.Princeton Academic Press, Princeton, New Jersey.

Hosmer, D.W., Lemeshow, S., 2000. Applied Logistic Regression Analysis. Second Edition. Wiley and Sons, New York.

Huberty, C.J., 1994. Applied discriminant analysis. Wiley Interscience, New York.

Hudson, D.J., 1971. Interval estimation from the likelihood function. Journal of the Royal Statistical Society 33, 256–262.

Ito, T.Y., Miura, N., Lhagvasuren, B., Enkhbileg, D., Takatsuki, S., Tsunekawa, A., Jiang, Z., 2005. Preliminary evidence of a barrier effect of a railroad on the migration of Mongolian gazelles. Conservation Biology 19, 245–248.

Johnson, C.J., Boyce, M.S., Case, R.L., Cluff, H.D., Gau, R., Gunn, A., Mulders, R., 2005. Cumulative effects of human developments on arctic wildlife. Wildlife Monograph 160, 1–36.

Joly, K., Nellemann, C., Vistnes, I., 2006. A reevaluation of caribou distribution near an oilfield road on Alaska's North Slope. Wildlife Society Bulletin 34, 866–869.

Latifovic, R., Trishchenko, A.P., Chen, J., Park, W.B., Khlopenkov, K.V., Fernandes, R., Pouliot, D., Ungureanu, C., Luo, Y., Wang, S., Davidson, A., Cihlar, J., 2005. Generating historical AVHRR 1 km baseline satellite data records over Canada suitable for climate change studies. Canadian Journal of Remote Sensing 31, 324–346.

Mahoney, S.P., Schaeffer, J.A., 2002. Hydroelectric development and the disruption of migration in caribou. Biological Conservation 107, 147–153.

Manly, B.F.J., McDonald, L.L., Thomas, D.L., Mcdonald, T.L., Erickson, W.P., 2002. Resource selection by animals: statistical design and analysis for field studies, second edition. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Mayor, S.J., Schaefer, J.A., Schneider, D.C., Mahoney, S.P., 2009. The spatial structure of habitat selection: a caribou’s-eye-view. Acta Oegologica 35, 253–260.

Myers-Smith, I.H., Arnesen, B.K., Thompson, R.M. Chapin, F.S., 2006. Cumulative impacts on Alaskan arctic tundra of a quarter century of road dust. Ecoscience 13, 503–510.

Millspaugh, J.J., Skalski, J.R., Kernohan, B.J., Raedeke, K.J., Brundige, G.C., Cooper, A.B., 1998. Some comments on spatial independence in studies of resource selection. Wildlife Society Bulletin 26, 232-236.

Murphy, S.M., Curatolo, J.A., 1987. Activity budgets and movement rates of caribou encountering pipelines, roads, and traffic in northern Alaska. Canadian Journal of Zoology 65, 2483–2490.

Nellemann, C., Cameron, R.D., 1998. Cumulative impacts of an evolving oil-field complex on the distribution of calving caribou. Canadian Journal of Zoology 76, 1425–1430.

Nellemann, C., Jordhøy, P., Støen, O-G, Strand, O., 2000. Cumulative impacts of tourist resorts on wild reindeer (*Rangifer tarandus tarandus*) during winter. Arctic 53, 9–17.

Nellemann C., Vistnes, I., Strand, P.J., 2001. Winter distribution of wild reindeer in relation to power lines, roads, and resorts. Biological Conservation 101, 351–360.

Nellemann, C., Jordhøy, P., Vistnes, I., Strand, O., Newton, A., 2003. Progressive impacts of piecemeal development. Biological Conservation 113, 307–317.

Nishi, J.S., Croft, B., Williams, J., Boulanger, J., Johnson, D., 2008. An estimate of breeding females in the Bathurst herd of barren-ground caribou, June 2006. Northwest Territories Department of Environment and Natural Resources, File Report 137. 72 pp. http://www.enr.gov.nt.ca/_live/documents/documentManagerUpload/137.pdf

Noel, L.E., Parker, K.R., Cronin, M.A., 2004. Caribou distribution near an oilfield road on Alaska's North Slope, 1978–2001. Wildlife Society Bulletin 32, 757–771.

Olthof, I., Latifovic, R., Pouliot, D., 2008. Circa-2000 Northern Land Cover of Canada. Earth Sciences Sector, Canada Centre for Remote Sensing, Natural Resources Canada.

Pendergast, J.F., Gange, S.J., Newton, M.A., Lindstrom, M.J., Palta, M., Fisher, M.R., 1996. A survey of methods for analyzing clustered binary response data. International Statistical Review 64, 89–118.

Rescan., 2006. EKATI diamond mine CALPUFF air dispersion modelling assessment report. Prepared for BHP Billiton Diamonds Inc., Rescan Environmental Services Ltd., October 2006.

Russell, D.E., Martell, A.M., Nixon, W.A.C., 1993. Range ecology of the Porcupine caribou herd in Canada. Rangifer, Special Issue No. 8, 120–125.

SAS Institute., 2000. The SAS System for Windows. SAS Institute, Cary NC.

Stankowich, T., 2008. Ungulate flight responses to human disturbance: a review and meta-analysis. Biological Conservation 141, 2159–2173.

Vistnes, I., Nellemann, C., 2008. The matter of spatial and temporal scales: a review of reindeer and caribou response to human activity. Polar Biology 31, 399–407.

Weir, J.N., Mahoney, S.P., McLaren, B., Ferguson, S.H., 2007. Effects of mine development on woodland caribou *Rangifer tarandus* distribution. Wildlife Biology 13, 66–74.

Wolfe, S.A., Griffiths, B., C.A.G. Wolfe., 2000. Response of reindeer and caribou to human activities. Polar Research 19, 63–73.

Zar, J. H., 1996. Biostatistical analysis. Third edition. Prentice-Hall, London.

Ziegler, A., Ulrike, G., 1998. The generalised estimating equations: A comparison of procedures available in commercial statistical software packages. Biometrical Journal 40, 245–260.

TABLE 1. Time line of development of three diamond mines in the Canadian Arctic between 1996 and 2008.

Mine site	Footprint in 2008 (km^2)	Baseline	Pre-construction	Construction	Operation
Ekati†	20.6	–	–	1996–98	1998–2008
Diavik	9.7	–	1996–99	2000–02	2003–08
Snap Lake	1.4	1996–98	1999–2004	2005–07	2008

† Within the Ekati mine development, the Misery Road was constructed starting in 2000, with work on the Misery Pit starting in 2001. The Fox Pit, a large pit 6 km south of the main Ekati mine site began development in 2003.

TABLE 2. Number of aerial surveys where caribou were observed in >1 cell, and the number of collared caribou used for analysis. Satellite collar data include only caribou that had a mine area within their availability radius at least once in a given year.

	Aerial surveys			No. of collared caribou
Year	Ekati	Diavik	Snap Lake	all mines
1996				9
1997				7
1998	17			-†
1999	18		3	14
2000	12		2	13
2001	11		3	9
2002	8	8	3	11
2003	9	9	2	10
2004	9 combined		2	4
2005	10 combined		2	18
2006	10	8	0	14
2007	9	10	2	19
2008	10	10	0	10

† Satellite collars in 1998 provided sporadic and unreliable data, and were removed from analysis.

TABLE 3. Zone of influence estimates for Ekati-Diavik mine areas as a function of time period from aerial survey data. The zone of influence estimate (ZOI), relative precision (% CI width), significance of zone of influence model term (β_{ZOI}), goodness of fit (GOF; ROC score), and the magnitude of zone of influence effect as described by the odds ratio (OR_{ZOI}) are given.

	ZOI			Significance of β_{ZOI}		GOF	OR_{ZOI}		
Period	(km)	CI	%CI width	*Z*	*P*	ROC	Est.	SE	CI
1998–99	4	3.0–7.0	100.0	9.12	<0.001	0.786	5.80	1.09	4.92–6.84
2000–02	-†								
2003–08	14	13.0–15.0	14.3	-9.91	<0.001	0.786	9.90	1.08	8.53–11.48
Pooled	14	12.0–15.5	25.0	10.94	<0.001	0.795	4.18	1.08	3.60–4.85

† No peak in the likelihood curve was observed making estimation of zone of influence not possible.

TABLE 4. Summary of zone of influence estimates for the Ekati-Diavik mine complex based on used/random analyses of satellite collar data. The zone influence estimate, relative precision (% CI width), significance of zone of influence model term(χ^2), goodness of fit (ρ) and the magnitude of zone of influence effect as described by the odds ratio are given.

				Significance of β_{ZOI}		GOF		Odds ratios		
Period	ZOI (Km)	CI	CV-CI (%)	χ^2	*P*	ρ	*P*	Est.	SE	CI
1996–99	23	19–35	69.6	18.30	<0.001	0.93	0.0007	0.09	0.02	0.07-0.22
2000–02	3	1–39	1266.7	0.07	0.80	0.97	<0.0001	2.26	0.07	1.32-225.7
2003–08	11	1–17	145.5	18.27	<0.001	0.94	0.0003	3.85	1.46	1.64–10.13
Pooled	3	1.5–12	350.0	2.48	0.1148	0.95	0.0002	26.2	3.92	1.2–420.2

Figure legends

FIG. 1. Location of the Ekati, Diavik, and Snap Lake diamond mines in the Canadian Arctic. The larger polygon represents the area of high use (70% kernel) of the distribution of collared caribou, 15 July–15 October, 1996–2008. The largest extent of the aerial survey study areas is also shown around each mine. Treeline represents the northern extent of continuous forests.

FIG. 2. The model used to estimate the zone of influence and the magnitude of the zone of influence. If a zone of influence exists (the grey area), habitat selection (as reflected by odds ratio of selection compared to the immediate mine area) should increase until the distance where the mine has no influence on selection. At this point, the model should best fit the data as indicated by the highest log-likelihood value. The slope of the increase in odds ratio is estimated by β_{ZOI}. At distances beyond the zone of influence, the zone of influence predictor variable was set constant (i.e., all distances beyond 10 km were set to 10 km), therefore creating an asymptote in the zone of influence curve.

FIG. 3. Predicted change in odds ratio (solid line with confidence limits as grey lines) and likelihood curve (dashed line) as a function of distance from the pooled Ekati-Diavik mine complex as determined from aerial survey data (1998–2008). Estimates are modelled upon the aerial survey design that flew both Ekati and Diavik mine sites in the same survey (2004–05).

FIG. 4. Likelihood curves as a function of time periods for the Ekati/Diavik pooled mine complex analysis, 1998–2008).

FIG. 5. Used satellite collar point densities for the Ekati-Diavik mine complex by period. The number of collared caribou was different for each time period, therefore each

curve should be interpreted in terms of relative distribution rather than actual densities of caribou near mines.

FIG. 6. Likelihood curves based on satellite collar analysis for time periods of the Ekati-Diavik mine complex.

FIG. 7. Predicted odds ratio of caribou occurrence as a function of TSP level for the Ekati-Diavik mine complex area from aerial survey data (a) and satellite collar data (b).

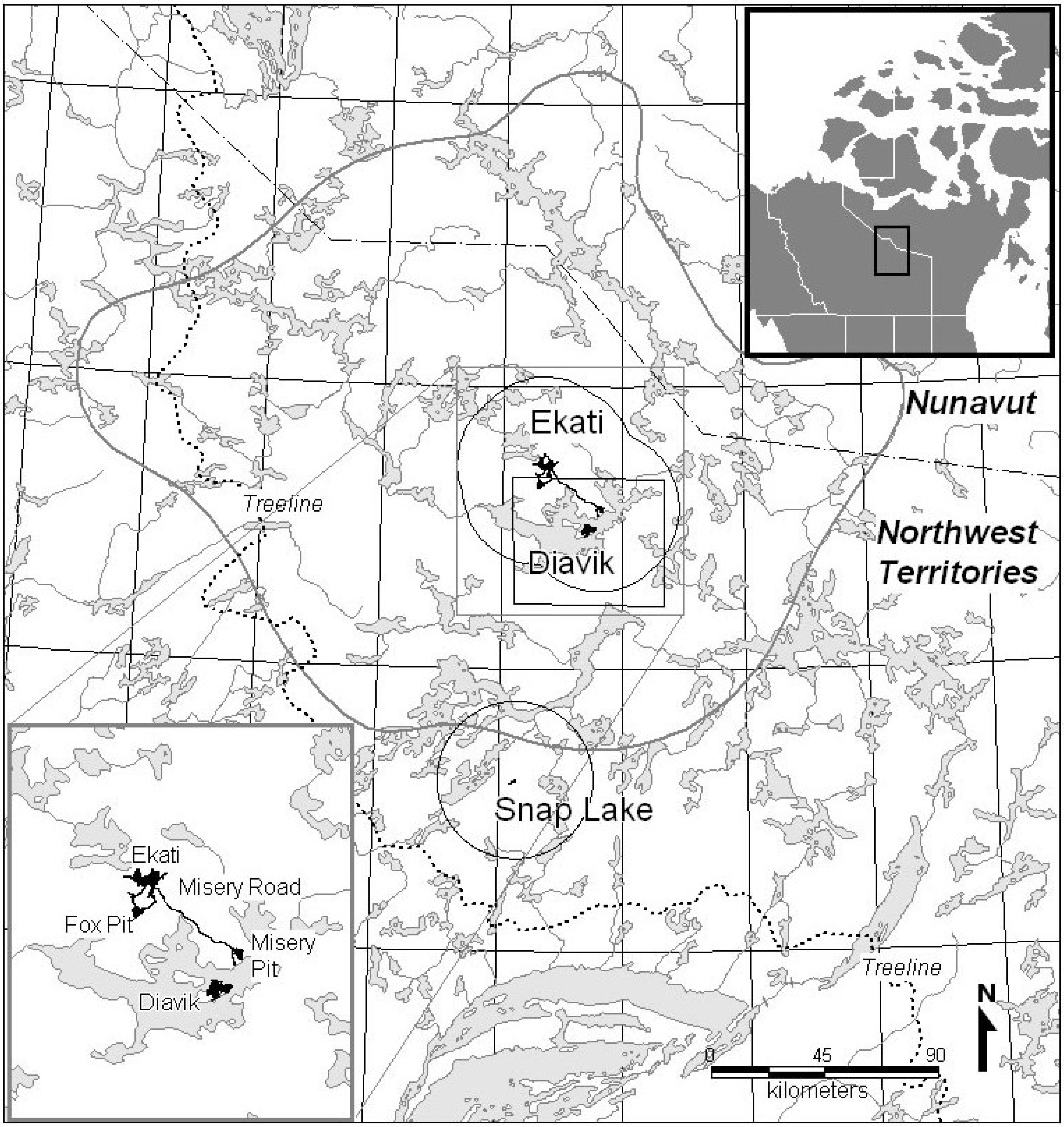
Nunavut
Northwest Territories
Ekati
Diavik
Snap Lake
Treeline
Treeline
Ekati
Misery Road
Fox Pit
Misery Pit
Diavik
N
0
45
90
kilometers

Figure 1

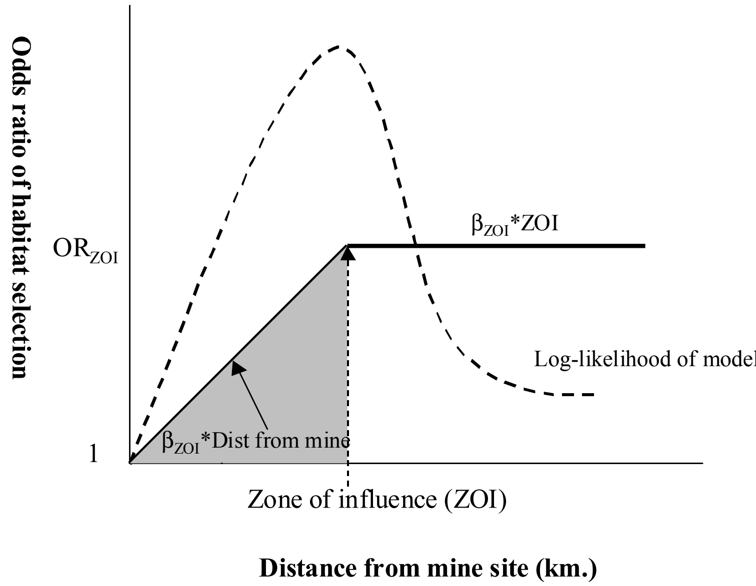
Odds ratio of habitat selection
OR$_{ZOI}$
1
β_{ZOI}*ZOI
Log-likelihood of model
β_{ZOI}*Dist from mine
Zone of influence (ZOI)
Distance from mine site (km.)

Figure 2

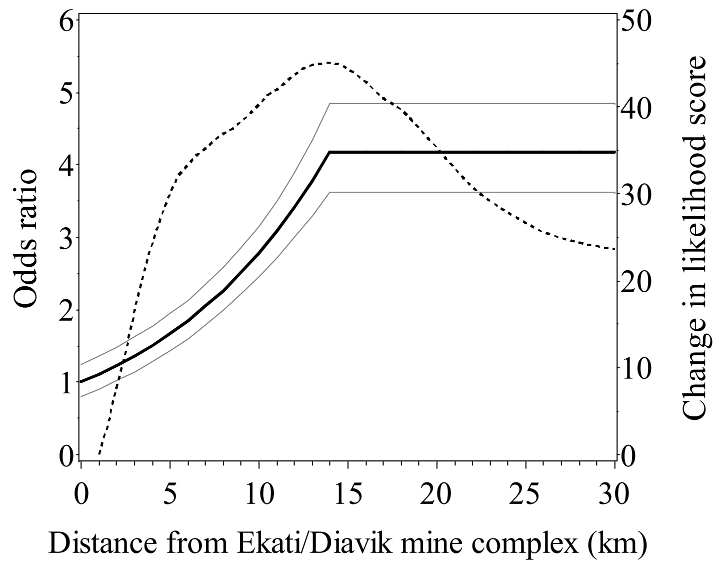
Odds ratio
Change in likelihood score
Distance from Ekati/Diavik mine complex (km)
0
1
2
3
4
5
6
0
10
20
30
40
50
0
5
10
15
20
25
30

Figure 3

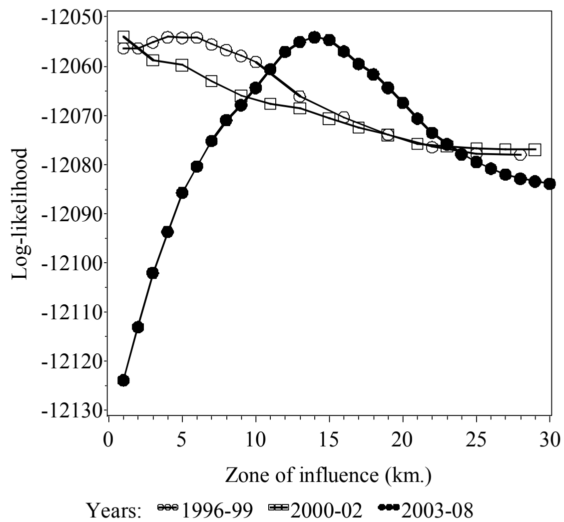
-12050
-12060
-12070
-12080
-12090
-12100
-12110
-12120
-12130
Log-likelihood
0
5
10
15
20
25
30
Zone of influence (km.)
Years: 1996-99 2000-02 2003-08

Figure 4

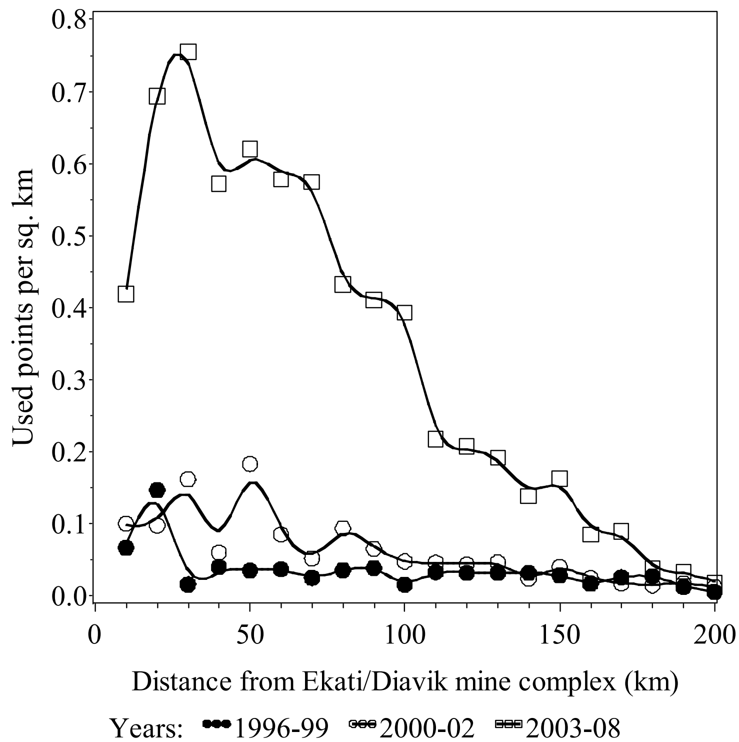
Used points per sq. km
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0
50
100
150
200
Distance from Ekati/Diavik mine complex (km)
Years: 1996-99 2000-02 2003-08

Figure 5

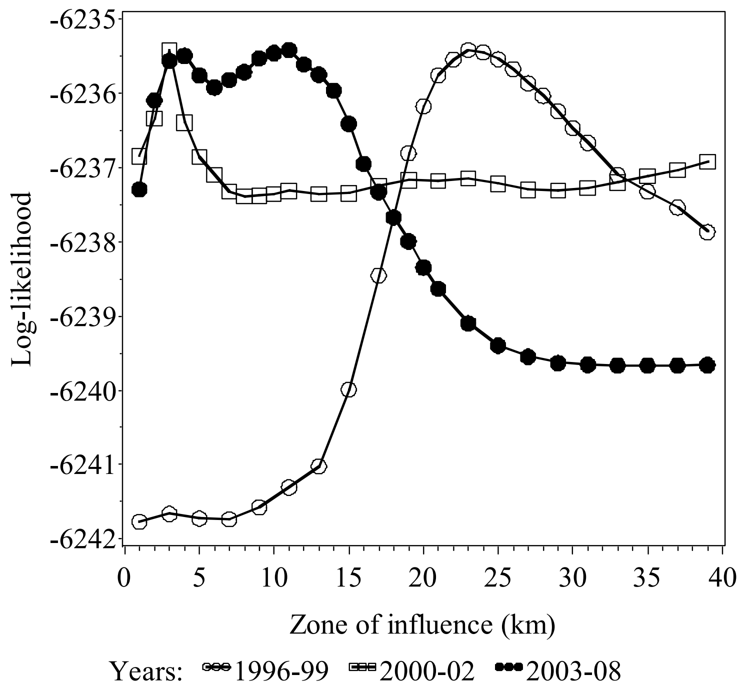
-6235
-6236
-6237
-6238
-6239
-6240
-6241
-6242
Log-likelihood
0
5
10
15
20
25
30
35
40
Zone of influence (km)
Years: 1996-99 2000-02 2003-08

Figure 6

a) Aerial survey data b) Satellite collar data

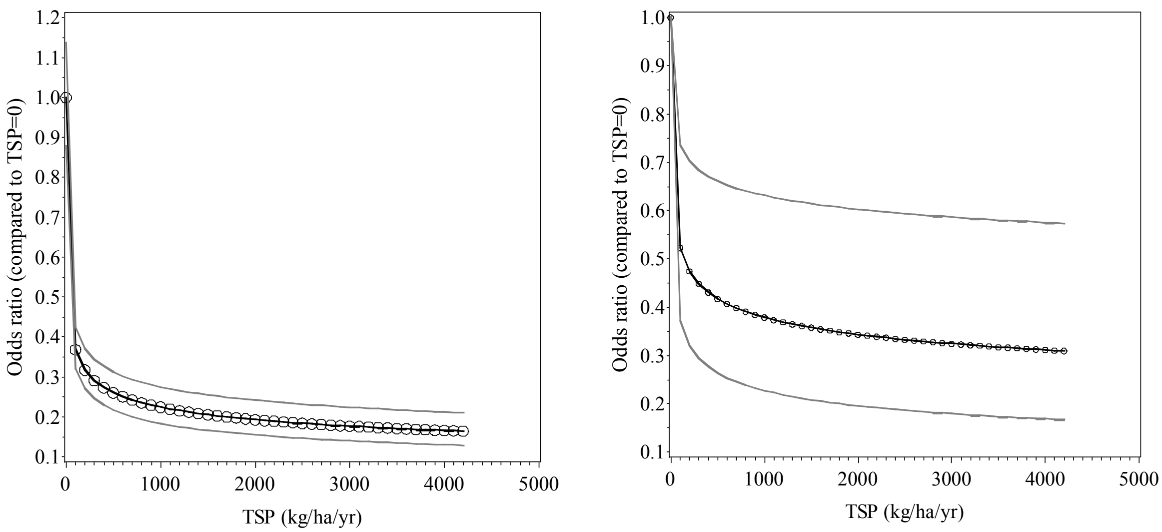

Figure 7

Appendix A

Habitat classification used in the analysis of zone of influence

We condensed habitat categories by blending two sources to provide complete coverage of the study areas, based on similarities in descriptions, low frequency of some types, and logical assumptions about caribou biology (Table A-1). We pooled habitat classes using the Land Cover Map of Northern Canada (NLC; Olthof et al. 2008), and Earth Observation for Sustainable Development of Forests (EOSD; http://cfs.nrcan.gc.ca/subsite/eosd/mapping) land cover classification. The NLC classification coverage was generally north of treeline, and was given precedence where coverage from both products overlapped. Esker coverage was obtained from 1:250,000 scale National Topographic Data Base maps (Natural Resources Canada; http://geogratis.cgdi.gc.ca/geogratis/en/product/search.do?id=8147).

TABLE A-1. Habitat associations used in base habitat models (Appendix B).

Pooled habitat associations	Acronym	Description
Bedrock-boulder	Bedbould	Exposed bedrock or boulders, barren, or sparsely vegetated
Moss-lichen	Mosslichen	Bryophytes or lichen
Tundra	Tundra	Non-tussock graminoids, prostrate dwarf shrubs
Tussock	Tussock	Tussock graminoid tundra
Sedge wetland	Sedgewet	Wet sedge and wetlands
Low shrub	Lowshrub	Low shrub (<40cm; >25% cover)
Tall shrub	Tallshrub	Tall shrub (>40cm; >25% cover)
Treeline herb	Treeherb	Wetland herb near forests
Forest	Forest	Conifer, broadleaf and mixed forests of all crown closures
Esker	Esker	Esker features from NTDB
Water	Water	Lakes, rivers, streams
Other	Other	

5 Appendix B

Results of base habitat models using aerial survey and satellite collar

5.1 Aerial surveys

Ekati and Diavik.—Univariate tests revealed linear relationships between caribou distribution and relative occupancy, esker, sedge wetland, and water predictor variables (Table B-1). Quadratic relationships were suggested between low shrub, tundra and water predictor variables. In addition, an interaction between tundra and NDVI suggested a positive seasonal influence of the use of tundra.

Snap Lake.—A unique base habitat model was developed for the Snap mine site given its location on the southern end of the summer range. The base habitat model for Snap suggested linear relationships with relative occupancy, bedrock-boulder, forest, moss-lichen, and tall shrub habitat classes (Table B-2). Quadratic relationships were suggested with water and forest habitat categories. In addition, seasonal use of water, bedrock-boulder, and tall shrub categories was suggested. The overall ROC score of the model was 0.80 suggesting adequate fit to the data.

5.2 Satellite collars

The base habitat model displayed adequate fit to the data as determined by Pearson correlation of area-adjusted frequencies and ordinal odds ratio bins ($\rho = 0.902$, $P < 0.0001$). The base habitat model analysis revealed linear or quadratic selection of forest, tall shrub, tundra, and water habitat variables (Table B-3). Seasonal selection was evident for bedrock-boulder, low shrub, treeline herb, tundra, and forest (interaction with NDVI) habitat categories. The selection of forest treeline herb and low shrub was also dependent on scale as determined by the availability buffer width and corresponding fix interval.

TABLE B-1. Base habitat model for aerial survey analysis for the Ekati and Diavik mine area aerial surveys. Standardized slope estimates are given for habitat variables (Appendix A).

Parameter	Estimate	Std Err	CI	χ^2	*P*
Intercept	-3.33	0.04	-3.40– -3.26	8737.26	<0.0001
Esker	0.04	0.02	0.01–0.07	5.52	0.0188
Reloccupancy	0.58	0.01	0.56–0.61	2656.08	<0.0001
Lowshrub^2	-0.06	0.03	-0.11– -0.01	6.28	0.0122
Sedgewet	0.15	0.04	0.08–0.23	15.71	<0.0001
Tundra^2	-0.10	0.02	-0.14– -0.06	28.18	<0.0001
Tundra*NDVI	0.49	0.25	0.00–0.97	3.87	0.0492
Water	-0.14	0.08	-0.29–0.02	2.97	0.0848
Water^2	-0.23	0.05	-0.32– -0.14	25.70	<0.0001

TABLE B-2. Base habitat model for aerial survey analysis for the Snap Lake mine aerial surveys. Standardized slope estimates and empirical standard errors are given for habitat variables.

Parameter	Group	Estimate	Std Err	CI	Z	*P*
Intercept		-4.812	1.072	-6.912– -2.712	-4.490	<0.0001
Reloccupancy		0.880	0.064	0.754–1.006	13.700	<0.0001
Bedbould		0.185	0.113	-0.037–0.407	1.630	0.102
Forest		0.434	0.129	0.182–0.686	3.370	0.001
Forest^2		-0.059	0.037	-0.132–0.013	-1.600	0.109
Mosslich		0.333	0.150	0.039–0.627	2.220	0.026
Tallshrub		0.131	0.084	-0.035–0.296	1.550	0.122
Water^2		-0.182	0.066	-0.310– -0.053	-2.760	0.006
Water*ndvi		0.561	0.524	-0.466–1.587	1.070	0.285
Bedbould*seasFall		-0.594	0.153	-0.893– -0.294	-3.880	<0.0001
Tallshrub*seasFall		0.225	0.096	0.037–0.413	2.340	0.019

TABLE B-3. Base conditional logistic regression habitat model used to estimate zone of influence from satellite collar data for the Snap Lake and combined Ekati-Diavik mine sites.

Parameter	Group	Estimate	Std. Err.	χ^2	*P*
bedbould*season	Fall Migration	-0.222	0.050	19.836	<0.0001
	Rut/Late fall	-0.137	0.072	3.549	0.0596
	Early Summer	-0.489	0.301	2.641	0.1041
Forest		0.948	0.132	51.665	<0.0001
Forest^2		-0.146	0.022	44.379	<0.0001
Forest*scale	1	0.044	0.094	0.217	0.6413
	5	-0.203	0.080	6.474	0.0109
Forest*NDVI		-1.032	0.178	33.476	<0.0001
Forest*movement rate		-0.016	0.004	13.657	0.0002
Lowshrub*scale		-0.039	0.011	11.723	0.0006
Lowshrub*season	Fall Migration	0.075	0.070	1.147	0.2842
	Rut/Late fall	0.158	0.091	2.988	0.0839
	Early Summer	0.148	0.096	2.382	0.1227
Tallshrub		-0.061	0.024	6.241	0.0125
Treeherb*scale	1	0.021	0.033	0.410	0.5222
	5	-0.141	0.055	6.627	0.01
Treeherb*Summer/Fall	Fall	0.137	0.051	7.213	0.0072
Tundra		-0.043	0.054	0.641	0.4233
Tundra*rate		-0.011	0.004	9.811	0.0017
Tundra*season	Fall Migration	-0.102	0.068	2.281	0.1309
	Rut/Late fall	-0.139	0.144	0.926	0.336
	Early Summer	-0.380	0.131	8.447	0.0037
Water		-0.649	0.034	365.589	<0.0001