Proceedings of the 13th
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Yellowknife, Canada
22-26 August, 2011

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and other Northern Ungulates

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2011
Arctic
Ungulate
Conference

Challenges of Managing Northern Ungulates

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Preface

The 13th International Arctic Ungulate Conference was held in Yellowknife, Northwest Territories, Canada, from 22-26 August 2011. This biennial conference is the primary venue for meetings of the International Arctic Ungulate Society. Over 200 delegates attended the conference with representation from Canada, the United States, Norway, Sweden, Greenland, Finland, Russia, and Japan.

The theme of the conference was "Challenges of Managing Northern Ungulates". The goal of the conference organizers was to address the difficulties of managing ungulate populations that are faced with the unpredictable effects of climate change and an ever-increasing human presence on the land. The conference also focussed on the challenges associated with developing recovery actions for declining caribou and reindeer populations that are an integral part of Aboriginal cultures and ways of life.

Prior to the conferences technical presentations, a number of pre-conference seminars and workshops were held including an Indigenous Talking Circle, Wildlife Necropsy and Health/ Condition Sampling Techniques, Conservation Education, and Caribou Demographic Modelling. Post-conference field trips were also held including a tour of Yellowknife, fishing on Great Slave Lake, and a tour of the Canadian Shield. During the conference, a number of events open to the public were also held including a storytelling session on the importance of Arctic ungulates to northern people (hosted by Paul Andrew) and a visual demonstration of how caribou populations are counted (John Nishi).

Opening the technical presentations was a plenary session chaired by Don Russell (Circum-Arctic Rangifer Monitoring Network) and Joe Tetlichi (Porcupine Caribou Management Board) related to the conference theme and titled "What are the challenges of managing caribou and reindeer?" Following this a panel session was convened comparing caribou and reindeer management across the circumpolar North. Over 60 oral presentations and over 30 posters were presented during the three days of technical sessions. Session topics ranged included Health and Disease, Management, Range and Habitat, Status and Population Ecology, Behaviour, Genetics and Evolution, Predators and Prey, Disturbance and Climate Change, Nutrition and Physiology, Incorporating Aboriginal Traditional Knowledge into Management, and a session presenting find-

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ings from CARMA. To conclude the conference Monte Hummel (World Wildlife Fund Canada) delivered a presentation titled "A circum-arctic conservation plan for caribou and wild reindeer" and the technical session chairs participated in panel sessions related to research and management priorities for northern ungulates.

The national and cultural diversity of the delegates to this conference greatly enhanced the exchange of information and brought new approaches and perspectives to the management and conservation of northern ungulates. On behalf of the organizing committee I would like to thank all the delegates to the conference for their participation, as well as the numerous volunteers who were instrumental to the overall success of the conference.

I look forward to seeing you at the 14th International Arctic Ungulate Conference in 2015 in Røros, Norway.

Troy Hegel

Department of Environment, Government of Yukon

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In Memoriam

Robert John “Bob” Hudson
December 16, 1946 – August 17, 2011

Bob Hudson passed away peacefully on August 17, 2011, surrounded by his loving family. Bob was an exceptional person, a truly international, Canadian and northern academician whose many contributions to the conservation, productivity and management of wildlife knew no geographic boundaries.

Bob was born and raised in Hamiota, Manitoba, the second of four children. His father,‘Doc Ed’,was a distinguished rural physician, his mother a nurse. Both of Bob’s advanced degrees–B.Sc. in Range Management (1967) and Ph.D in Animal Science (1971)–were completed at the University of British Columbia (UBC). Following his graduate research on immunology of lungworm infections of bighorn sheep, Bob did a postdoc (reindeer range assessment) at the Institute of Ecological Botany, Uppsala, Sweden. He was named Assistant Professor at UBC before moving to the University of Alberta (Faculty of Agriculture and Forestry, now Agricultural, Life and Environmental Sciences [ALES]) in 1974 where he established a long-term and stellar program in Wildlife Productivity and Management. In the mid-to-late 1970s, Bob became interested in rangeland monitoring, management of elephants and rhinos and issues of wildlife production policy and research in Kenya, doing a sabbatical there from 1980-1981. It was not all work for Bob while in Kenya, as he met the love of his life there, Yasmin, whom he married in 1981. Daughters Suraya and Tasreen were born in 1986 and 1992; Bob’s family was truly the center of his life.

Bob had a brilliant mind, with breadth and depth, and became an international leader in several research areas including multi-species grazing systems, bioenergetics of wild herbivores, system dynamics and more recently, sustainability science. He established the Ministik Wildlife Research Station (MWRS) in 1977 in a rural aspen parklands setting east of Edmonton where he and graduate students did some brilliant research (e.g., classic studies of bioenergetics) using captive bison, wapiti, mule and white-tailed deer and moose as study animals. Along the way he edited eight books and authored two books and many book chapters and extension articles, and over 160 peer-reviewed scientific articles.

While Bob was doing his thing at MWRS, Elk Island National Park and other sites in Western Canada, Kenya, India (sabbatical), Korea, Malaysia, China, South America, etc, his graduate students (n=55) were becoming leaders in Canada’s north, Western Canada, Africa, China and elsewhere. These students are a crucial part of Bob’s legacy, carrying forward his positive and visionary approaches, always thinking about what is to come or what might be with some critical, synthetic thought. He was the model of how to escape current dogma and advance toward new paradigms.

Bob was dedicated to collaboration and service. As one example, in 2010 he published a 300 page e-book, edited a 400 page book and co-authored a paper. Also, during his illnesses, he was Associate Dean International (ALES), Founding Director, Alberta Veterinary Research Institute, Acting Director, Canadian Circumpolar Institute and Chair, University Animal Policy and Welfare Committee. During the same time he was Theme Editor UNESCO Encyclopedia of Life Support Systems, Editor and Chief Open Conservation Biology Journal, Associate Editor Journal of Wildlife Management, and on the Editorial Board and Advisory Board of several journals and research centres.

In conclusion, Bob Hudson was a brilliant visionary in contemporary wildlife management issues; a person who shared that vision in quiet modest ways we all came to admire. He will be missed.

Bill Samuel, Professor Emeritus, Department of Biological Sciences, University of Alberta, Edmonton
Brad Stelfox, Forem Technologies and the ALCES group, Calgary
Lee Foote, Professor, Department of Renewable Resources and Director Devonian Botanic Garden, University of Alberta, Edmonton

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A road in the middle of one of the last wild reindeer migration routes in Norway: crossing behaviour and threats to conservation

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Abstract: The development of roads and associated infrastructure has interrupted several traditional migrations of wild reindeer *(Rangifer tarandus tarandus)* in Norway. The population in the Setesdal Austhei wild reindeer area still migrates from winter to calving grounds by semi-annually crossing a cabin-lined road through a narrow corridor, in which further anthropogenic development is planned. To understand if and how infrastructure affected reindeer migration patterns we studied the movements of 10 female reindeer equipped with GPS collars between 2002 and 2010. First, we identified the start and end of the migration period, and then we compared trajectory parameters (net displacement, step length and turning angles) recorded during migration with those recorded in proximity of the road. The analysis of the net displacement indicated that during spring migration reindeer moved at a constant pace towards the calving ground covering a net linear distance of 25 km in 40 days. In the middle of migration, reindeer changed travel direction and roamed parallel to the road for ca. 5 days without approaching further, possibly searching for an undisturbed place and time to cross. Reindeer finally crossed the road before daylight with highly directed movements, increasing their travel speed up to the highest values recorded during migration (4 km/3 hrs vs. 0.5 km/3 hrs). After crossing, reindeer moved quickly toward their calving ground covering the remaining 25 km net distance in less than a week. Migration patterns were markedly affected by disturbance during spring, as the road crossings occurred in the period characterized by the high traffic volume and intense human activities related to Easter holidays; during autumn, on the contrary, the hampering effect of the road was minimal. The results suggest that the current disturbance associated to the road hampers spring migration and might delay the arrival to the calving ground. The planned construction of a large number of recreational cabins in the migration corridor has the potential to threaten the migration and obstruct the access to the calving ground.

Key words: Calving ground; human disturbance; migration; *Rangifer;* roads; trajectory.

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Introduction

Movement is one of the most striking features of the ecology of reindeer *(Rangifer tarandus tarandus).* The species inhabits the highly seasonal circumpolar environment, and typically follows greening vegetation by migrating from lichen-rich winter habitat with relatively little snow cover to highly productive and insect-free calving and summer pastures. The importance of the search for the optimal seasonal habitat to fitness is such that some *Rangifer* populations perform the longest overland movements in the world (Bergman *et al.,* 2000). However, the ongoing expansion of human-dominated areas

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and the rapid development of transportation infrastructure interfere with the persistence of large-scale animal movements; many of the most spectacular migrations worldwide have either disappeared, or are in steep decline (Wilcove & Wikelski, 2008; Harris *et al.,* 2009).

Before industrial development Norwegian wild reindeer were grouped into two to three large population units that performed seasonal migrations between adjacent mountain systems, with summer ranges typically located in the west and winter ranges in the east. Nowadays, Norwegian wild reindeer are divided into 23 more or less isolated populations. Due to the development of roads, infrastructures and human disturbance, mainly along valley bottoms, several of the most important migration corridors were lost in the past centuries, and two of the most important remaining key corridors were abandoned in the 1980s (Skogland, 1986; Vistnes *et al.,* 2004). Recently, global positioning system (GPS) radio-collar data of reindeer from several parts of Norway show that most of the remaining movement corridors connecting different sub-populations are being used progressively less often or are being abandoned (*e.g.,* Vistnes *et al.,* 2004; Bevanger *et al.,* 2005; Strand *et al.,* 2006; Dahle *et al.,* 2008). The population of Setesdal Austhei is one of the few that still perform distinct seasonal migrations between non-overlapping summer and winter ranges. However, a road surrounded by cabins lies in the middle of the migration route, and reindeer have to cross it through a narrow corridor in which further anthropogenic development is planned. We studied the effect of this road on migration behavioral and movement patterns of GPS radio-collared females from the Setesdal Austhei population to understand if and how the proposed land development plan might affect the persistence of this migration.

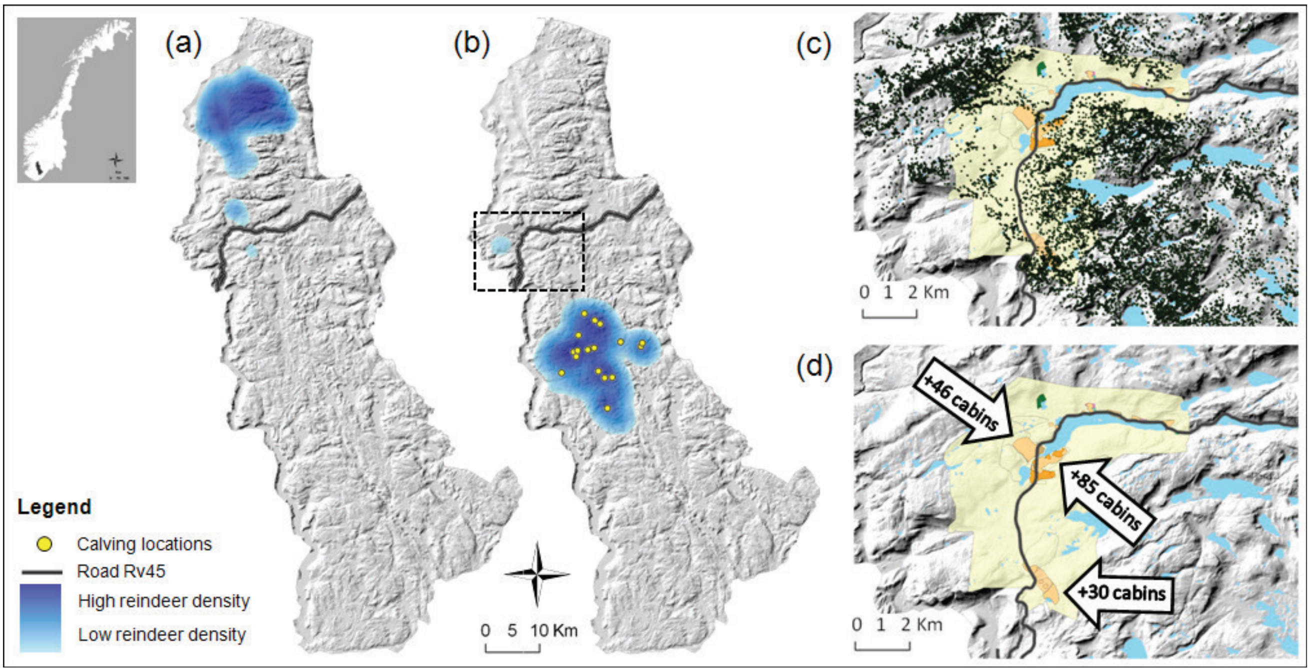

Fig. 1. The Setesdal Austhei study area. The location of a) winter (February) and b) calving range May15th – June 30th are represented by a 95% kernel density of the GPS radio-collar locations of 10 female reindeer monitored from 2002 to 2010. In (b) the yellow dots represent the locations of the calving events ($n = 20$) estimated for each individual each year. The dashed rectangle in b depicts the location of the area illustrated in (c), reindeer GPS radio-collar locations indicating the location of the two migration corridors, and (d), details of the land development plan for Bjørnevatn, indicating the planned location of the construction of several cabins for recreational use.

Methods

Study area

The Setesdal Austhei wild reindeer area (7° 78' E, 59° 23' N) is one of the two southernmost remaining populations of wild reindeer in Europe. The area was defined as a wild reindeer area in 1980 and opened for hunting in 1981; earlier, the reindeer inhabiting Setesdal Austhei were semi-domesticated. The area is confined by roads, which impede virtually any movement from or to other neighboring reindeer areas, and is bisected into a northern and southern part by a 45 km segment of the public road Rv 45, running parallel to a long dammed lake, Store Bjørnevatn, and cabins (Fig. 1). The average weekly daily traffic (AWDT) on Rv 45 is rather low but varies with time, being higher (AWDT ≈ 500 vehicles/day; Statens vegvesen, 2011) in spring during Easter holidays, which occur most often in the 8th week of the year, thus coinciding with the spring reindeer road crossing, and lower during the road crossing in autumn (AWDT ≈ 300). The northern area (ca. 770 km^2) is dominated by mountains up to 1537 m asl, while the southern area (ca. 1600 km^2) is characterized by low altitude hills and is mostly covered by forests. Further information regarding the study area can be found in Strand *et al.* (2011).

Reindeer migrate from their winter pastures in the northern area to the calving and summer areas in south. The reindeer GPS radio-collar data (see below) indicates that there are two migration corridors (Fig. 1c, d). One is approximately 1 km wide, and it is located adjacent to the western edge of the dammed lake, between Store Bjørnevatn and Lisle Bjørnevatn; the other is ca. 4 km wide, and is located south west of Lisle Bjørnevatn. However, we suggest that to characterize the location of the migration corridor more precisely further studies and data from a larger sample of animals, collected at finer temporal scales, are needed. The land development plan for Bjørnevatn includes the construction of a total of 161 cabins for recreational use (Kommuneplan for Bjørnevatn, 2008) in the migration corridors (Fig. 1c, d).

GPS radio-collar data

We captured 10 reindeer females in Setesdal Austhei and studied their movement pattern from 2002 to 2010. The animals were darted from helicopter, anesthetized using a mix of Salopine and Ketalar, equipped with GPS collars (Vectronics) with drop-off systems, and reversed using the antidote Narketan. The collars were programmed to record 1 GPS location every 3 hours. Outliers were identified and removed using the script provided in Bjørneraas *et al.* (2010).

Trajectory analyses

To understand whether and how Rv 45 affects reindeer migration patterns we first identified the migration period for each individual, calculated trajectory parameters during migration, and compared them with trajectory parameters calculated in proximity of the road. For each individual we selected 1 location every 3 hours and calculated the following trajectory parameters: step length (SL), turning angles (TA), net squared displacement (NSD), and net displacement (ND; Kareiva & Shigesada, 1983; Calenge *et al.,* 2009) using the package *adehabitat* (Calenge, 2006) in R 11.1 (R Development Core Team, 2011). ND is the mean distance between the first relocation of the trajectory and the last relocation of the current step (NSD is the same measurement, squared; Kareiva & Shigesada, 1983); hence, it describes the position of a step relative to other steps of the trajectory, and can be viewed as illustrating the movement towards or away from a starting point. Even though the analysis of ND throughout the year well describes an animal's movement patterns, the annual NDs of different individuals can be compared only if a common starting point is used for its calcula-

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tion. As a starting point for the calculation of reindeer displacement we used calving sites, all situated in the southern part of the range. To investigate differences in ND before and after the road crossing, for each individual we performed linear regressions between ND and the number of locations before and after crossing. We compared the slope of the regression before and after crossings with a Wilcoxon paired signed-rank test. SL and TA during migration in different seasons, 5 days before/after crossing, and during crossing were compared using t-tests. Finally, we calculated summary statistics of the time of crossing using circular statistics (Jammalamadaka & SenGupta, 2001), which is commonly used when analyzing directional data such as time of day. In particular, we calculated the mean crossing time, and *rho,* which is a measure of the spread of the data; *rho* ranges from 0 (all values are concentrated in one point; *i.e.*, zero variance) to 1 (uniform distribution, with values uniformly scattered throughout the circular range; *i.e.*, in our case the clock).

Identifying calving sites

We did not know the exact calving dates of the GPS-collared females in the area. Hence, we estimated their calving dates by analyzing variations in trajectory parameters during the calving season (Van Moorter *et al.*, unpublished data.). As calving occurs after reindeer reach the calving area south from Rv 45, for each individual we analyzed movement patterns during the period starting after the road crossing and terminating on July 15th. In particular, for each female we calculated the residence time (RT) which quantifies the time (*i.e.*, number of locations) spent within a patch of a given radius, in our case the estimated calving site, before moving further (Barraquand & Benhamou, 2008). RT was calculated within 8 buffers of different radii (10, 25, 50, 75, 100, 150, 200, 500 m), chosen based on information on the specie's calving behavior. We assumed that calving occurs on the day with the highest peak in RT. As the mean observed calving dates of wild reindeer in mainland Norway range between May 7 and May 28, and calving typically occurs very synchronously within each wild reindeer area (Reimers *et al.* 1983; Skogland, 1984; Reimers, 2002), we assumed that the best calving model was the one providing the most synchronous calving season (*i.e.*, the one providing the lowest variance in calving dates). The location of each individual during the average calving date was taken as a starting point for the calculation of its annual NSD, to investigate migration patterns.

Identifying the migration period

We calculated the average start and end date of the migration period by investigating the NSD of those individuals for which a full year of data, from one calving season to the next, was available (n = 21 reindeer-years). When animals move from a summer area to a winter area and back, the plot of their annual NSD appears like a double sigmoid: NSD increases during the autumn migration, reaches an asymptote when the animals reach the winter ground, and decreases again during the spring migration approaching zero if the animal comes back to the starting point (*i.e.*, the calving site). Hence, following Bunnefeld *et al.* (2011), we fitted a double sigmoid to the annual NSD of each individual:

$$NSD = \frac{\delta}{1+\exp(\frac{\theta_s - t}{\varphi_s})} + \frac{-\delta}{1+\exp(\frac{\theta_a - t}{\varphi_a})} \quad ;$$

(Equation 1) where δ is the asymptotic height, θ_s and θ_a are the timing at which the migration reaches half its asymptotic height in spring and autumn, respectively, φ_s and φ_a models the timing elapsed between reaching half and $\frac{1}{1+e^{-1}} \cong \frac{3}{4}$ of migration in spring and autumn, respectively, and t as time. Hence, the first sigmoid represents the autumn migration and the second the return to the calving ground. The inflection point of each sigmoid indicates the time at which an individual has reached the middle distance between summer and winter areas, while the distance between the upper and lower asymptotes reflects the distance between the start and the end point of migration. Equation 1 was fitted by non-linear least squares using the nls function in R. For each individual, we defined the start and the end of the seasonal migration as the number of days required to move from 5% to 95% of each sigmoid (Fig. 2). All trajectory parameters (SL, TA, and NSD) were calculated within the migration period.

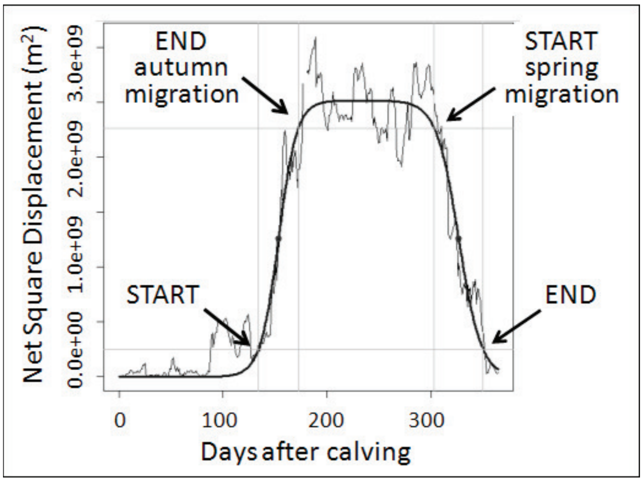

Fig. 2. Example of the migration parameters estimated for one focal individual based on its annual Net Square Displacement (NSD). The smooth line represents the model fitted to the focal individual to describe its migratory movement from the calving range to the winter range and back (Bunnefeld *et al.,* 2011). Grey lines identify the start and end of migration, defined as 5% and 95% of the sigmoid.

Results

We monitored 10 reindeer for a total of 21 reindeer-years; for each reindeer-year we obtained a minimum of 1085 locations, and a maximum of 2921 locations, for a total of 48 256 GPS positions, which can be viewed at http://www.dyreposisjoner.no.

Road crossing

All radio-monitored reindeer crossed Rv 45 twice a year during the seasonal migration. All crossings occurred during one single time bout, except for one case in autumn. All spring crossings occurred between April 11th and May 8th (average ± SD: April 23rd ± 6 days, n =18) and during autumn/winter between Sept. 9th and Jan. 2nd (Oct. 9th ± 23 days, n = 22; note: in only two cases reindeer crossed after Oct. 13th). Circular statistics (Jammalamadaka &

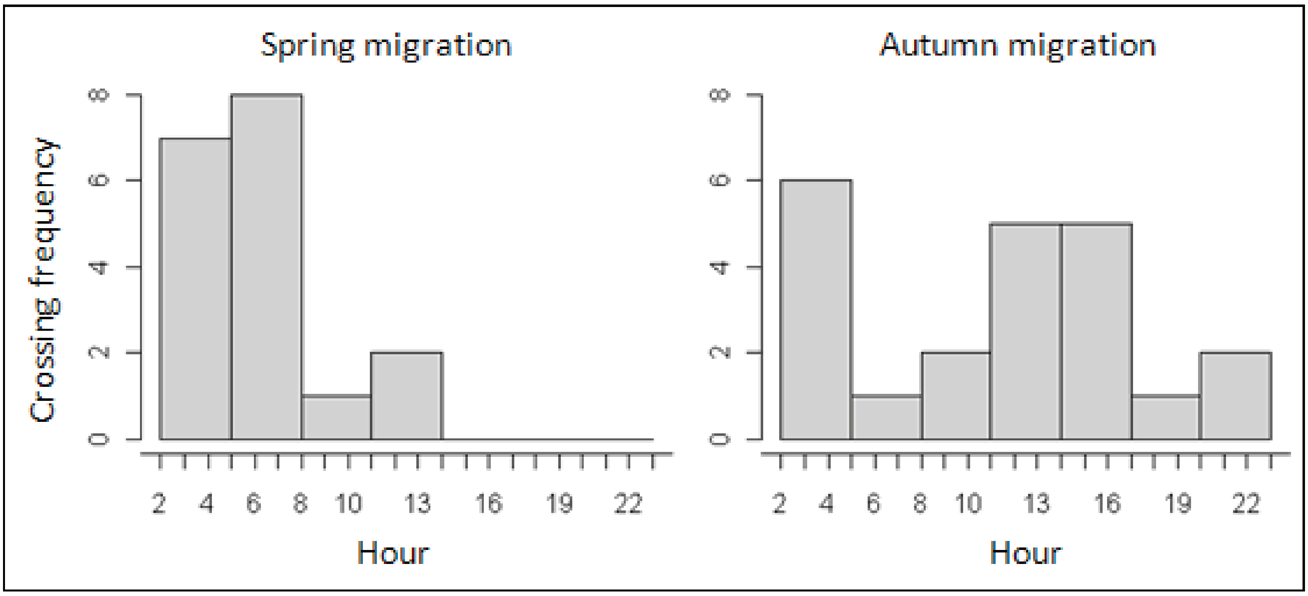

Fig. 3. Reindeer crossing frequency of road Rv 45 in Setesdal Austhei with respect to time of day during spring and autumn migration.

Table 1. Summary of the estimated start and end dates of migration, estimated calving dates, and observed dates and times of road crossing (± SD).

Period	Variable	Average	Range
Spring migration	Start date	Mar 15 ± 21 dd	Feb 22 - Apr 5
	Road crossing date	23 Apr ± 6 dd	Apr 11 - May 8
	Road crossing time	07:37 am (rho: 2.9) *	
	End date	3 May ± 9 dd	Apr 24 - May 12
	Calving	May 13 ± 5 dd	May 6 - May 23
Autumn migration	Start date	Sep 20 ± 15 dd	5 Sep - Oct 5
	Road crossing date	Oct 9 ± 23 dd	Sep 9 - Jan 2
	Road crossing time	18:18 pm (rho:0.7) *	
	End date	Nov 28 ± 23 dd	Nov 5 - Dec 20**

* Mean crossing time calculated with circular statistic (Jammalamadaka & SenGupta, 2001).

** Note: one individual that crossed on Jan 2nd was not included in the analyses of the migration period as we did not have enough data; however, for that individual the end of the autumn migration falls after Jan 2.

SenGupta, 2001), required for directional data, such as time of day, showed that most crossings occurred around 8:00 am (rho = 0.98). Seasonal differences in crossing time were detected (Fig. 3), with a higher frequency of crossings in the first hours of the day in spring (mean: 7:37 am) with very little variability (rho = 2.9), and no differences in autumn (mean: 6:18 pm, large variability: rho = 0.7); the difference between spring and autumn in the observed number of crossings in each of the eight 3-hour time bins was significant ($X^2 = 27.943$, df = 7, $P < 0.001$).

Calving dates

The method which performed best was the RT calculated within a radius of 150 m. Based on this method we estimated 20 calving dates (*i.e.*, for 10 adult females followed in 1 to 3 years). Calving occurred from May 6th to May 23rd (average: May 13th ± 4.6 days, SD). Calving occurred on average 21.6 ± 7.1 days (n = 17; range: 9 – 32 days) after the spring road crossing. In two cases the precision of our estimate was low, and did not allow us to estimate calving dates. This means that either the animals did not calve, or that the mother did not change movement patterns markedly during calving days.

Migration time

The double sigmoid model described well the reindeer migratory movements, as it explained on average 94.2 % (± 1.7, n = 10) of the variance in individual annual NSD. However, the method did not always perform well in identifying with high precision the start and end day of migration. The reason is that NSD is strongly dependent on the location of the starting point, and as common starting point we chose the calving sites, which are not too spatially concentrated within the summer range. Hence, calving sites located closer to the road led to a slightly delayed estimated start of migration, while calving sites located at the extreme south of the distribution range led to a slightly early start of migration. However, our results indicate that autumn and spring migration started on average on Sept. 20th (± 15 days) and March 15th (± 21 days), lasted 49 and 69 days, respec-

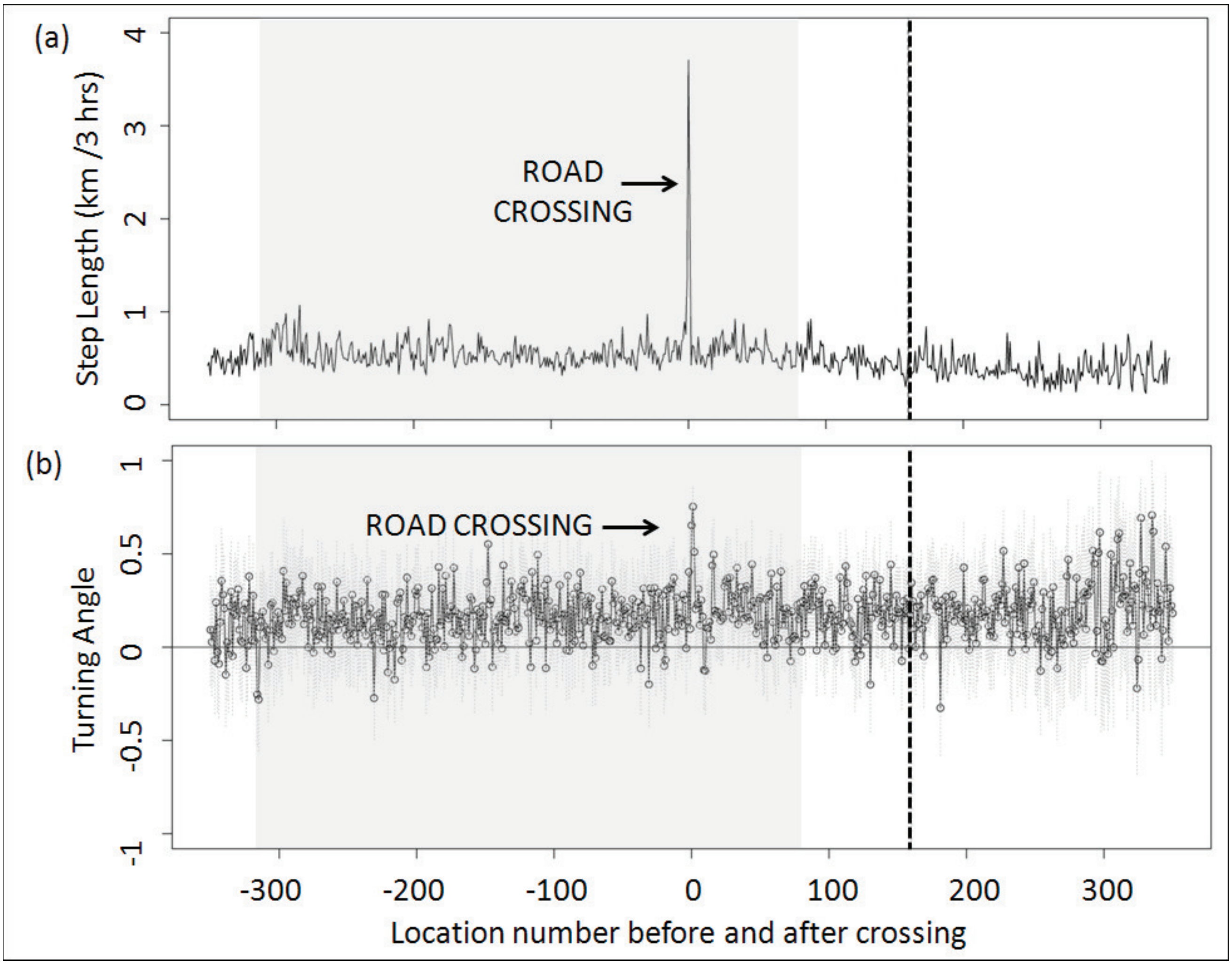

Fig. 4. Step length (upper graph) and turning angles (lower) of reindeer during 45 days (350 radio-locations) before and after the crossing of road Rv 45. The crossing occurs at the location coded 0, in the centre of the figure. The shaded area indicates the estimated migration period; the dashed line indicates the average estimated calving date.

tively, and covered 56.4 ± 5.4 km. A summary of the estimated migration periods, calving periods, and of the observed crossing dates is presented in Table 1.

Step length and turning angles

The average year-round SL was 0.493 km/3hrs (± 0.632, n = 47 866). When the animals crossed the road the average travel speed increased up to 3.706 km/3hrs (± 1.578, n = 37), with no difference between spring and autumn, and was much higher than the speed recorded during the 5 days (*i.e.*, 40 locations) before or after the crossing (0.613 km/3hrs ± 0.722, n = 2917; $t_{36.191}$ = 11.905, $P < 0.001$) or during the whole migration ($t_{110.296}$ = -10.398, $P < 0.001$; Fig. 4). Similarly, during the crossing in spring and autumn the average cosine of the turning angles was much higher than the one recorded during the 5 days before and after the crossing (0.515 ± 0.565, n = 93 vs. 0.162 ± 0.719, $t_{104.254}$ = -5.844, $P < 0.001$), or during migration (0.132 ± 0.717, n = 48256; $t_{113.048}$ = -6.286, $P < 0.001$), indicating very directional movements.

Net displacement

If the presence of the road had no effect on the

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spatial behavior during migration, we would expect the ND calculated from the crossing point to decrease linearly while the reindeer were approaching the road, and increase linearly after the road crossing (Fig. 5a). This is what we observed in autumn (Fig. 5b), as the slopes of the regression lines calculated for each individual between ND and the number of locations did not differ before and after crossing ($V = 87$, $n = 18$, $P = 0.644$). On the contrary, during spring the slopes of the regression lines were much higher after crossing compared to when approaching the road ($V = 165$, $n = 18$, $P < 0.001$). Indeed, during spring ND decreased slowly as the reindeer approached the road, it suddenly dropped before the road cross, and increased sharply after the crossing, with very little individual variability (Fig. 5c), due to the fact that after crossing the road movements became much more directional, with reindeer heading straight and quickly to their calving ground. While during spring reindeer migrated slowly towards the road covering a net linear distance of ca. 25 km in ca. 40 days, after the crossing they quickly drifted away from the road covering the same net distance in a week, and after 20 days they were displaced 35 km south of the road (Fig. 5b).

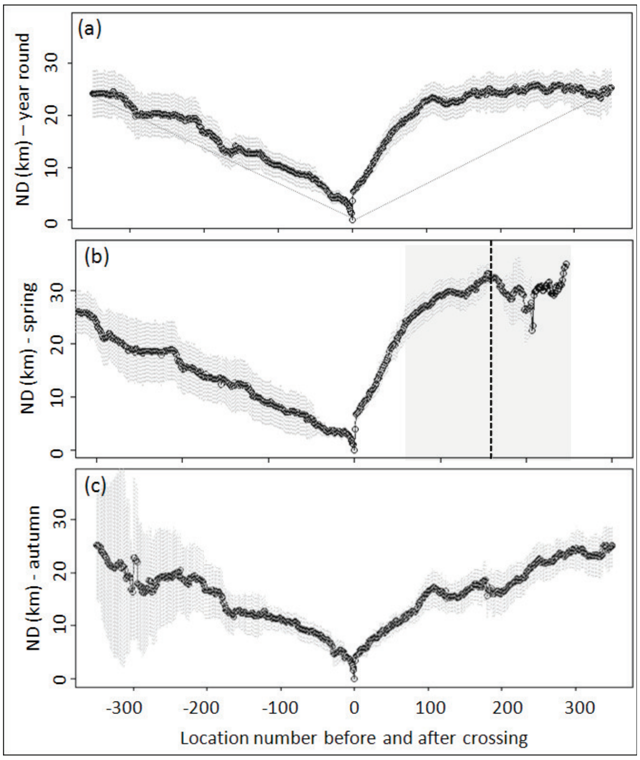

Fig. 5. Average Net Displacement (ND, km), which illustrates the speed of the movements towards and away from the road, calculated from the crossing point (coded 0) during 350 locations (ca. 45 days) before and after the crossing of road Rv 45 during both spring and autumn migration (a), and separately during spring (b) and autumn (c); dashed bars indicate standard errors. The dotted line in (a) indicates a theoretical ND if reindeer moved between the summer and winter areas at a constant speed (i.e., if the presence of the road had no effect on the spatial behavior during migration). The shaded area and the dashed line in (b) indicate, respectively, the estimated calving range and average estimated calving date.

Discussion

Several wild reindeer populations in Norway have abandoned their historical migration routes due to the expansion of human activities, namely transportation infrastructure and recreational facilities. We present the case of the population in Setesdal Austhei, which still performs distinct seasonal migrations between winter and calving grounds by crossing a road located in the middle of the migration corridor. Our results, however, show that all trajectory parameters are markedly altered in proximity of the road during both spring and autumn, as while crossing the road reindeer increase their travel speed to reach the highest values recorded during migration, perform highly directional movements, and cross during hours when the traffic is minimal. Most importantly, as the spring road crossings occur in the period characterized by the high traffic volume and intense use of private cabins related to Easter holidays, the overall spring migration patterns change as reindeer approach the road, and remain altered up to the arrival to the calving ground. During spring reindeer migrate at a constant pace towards the calving ground covering a net linear distance of 25 km in 40 days. As reindeer approach the road, they change travel direction and roam parallel to the road for ca. 5 days without approaching further, possibly searching for an undisturbed place and time to cross. As the pregnant reindeer cross the road, they speed up and quickly head toward calving ground covering the remaining 25 km net distance in less than a week. This suggests that the current disturbance associated with the road hampers spring migration and may delay the arrival to the calving ground. Interestingly, during autumn we could only identify short-term effects of the road (*i.e.*, travel speed increased and movements were highly directional during crossing), but no longer-term alteration of the migration patterns could be detected. This is probably due both to the lower traffic volume during autumn and to the lower physiological urgency of reaching the winter range.

The impact of infrastructure on *Rangifer* distribution and habitat use has been extensively documented worldwide during the last decades (Mahoney & Schaefer, 2002; Vistnes *et al.,* 2004; Johnson *et al.,* 2005; Nellemann *et al.,* 2010). Several studies reported range shifts in response to the construction of infrastructure (*e.g.*, Nellemann *et al.*, 2001; Dyer *et al.,* 2002), or documented the avoidance of potentially suitable habitat for several kilometers around disturbed areas, with little or no sign of habituation in the following decades (Vistnes & Nellemann, 2008; Nelleman *et al.,* 2010) or centuries (Schaefer, 2003). Not all studies, however, reached similar conclusions, and it has become clear that the responses to disturbance can vary greatly spatially and temporally depending on the type and size of the infrastructure, its location with respect to the core area of reindeer distribution range, variations in the intensity of disturbance, winter harshness, food availability and population density, and the degree of wildness of the population (Skogland, 1986; Carruthers & Jakimchuk, 1987; Duchesne *et al.,* 2000; Mahoney & Schaefer, 2002; Reimers *et al.*, 2007).

The list of confounding factors complicating the assessment of reindeer responses to human disturbance likely increases if the infrastructure is located in the middle of a migration corridor, as during migration reindeer may be forced to tolerate, up to a certain degree, higher levels of disturbance to avoid potentially serious population consequences. Dahle *et al.* (2008), for example, documented a clear avoidance of a buffer area surrounding a road, but no aversion could be detected along the same road in proximity of a traditional migration corridor. Hence, the drive for migration is such that it is often difficult to recognize the tipping point turning a permeable barrier into an effective barrier to migration in time useful for implementing mitigation measures. The construction of a railway and a parallel road in Norway, for example, obstructed a key migration corridor for decades. As the population, trapped in one of the seasonal ranges, reached high densities and faced overgrazing and starvation, reindeer started crossing the infrastructure occasionally during harsh winters. However, coincident with the increase in traffic and a reduction in deer numbers, all crossings finally stopped during the 1980s (Skogland, 1986). Hence, if the drive for migration is high, a given infrastructure can represent a barrier under some circumstances but not others, and the long-term population response may become evident only when the implementation of mitigation measures would be more complex and costly.

Previous analyses (Panzacchi *et al.,* 2011) showed that the motivation pushing the Setesdal Austhei population to continue migrating is high, as the optimal calving and winter grounds are located on the opposite sides of the road, and none of seasonal ranges seem to contain preferred resources year-round. Our study shows that even though reindeer still migrate through a narrow bottleneck on the west side of the road, migratory patterns are markedly altered by the disturbance associated with the road, especially during the spring migration to the calving ground. Considering the high drive for migration, our results suggest that the Setesdal Austhei population is already forced to tolerate relatively high levels of disturbance during spring migration to reach the preferred calving ground. The long-term consequences of the present intensity of disturbance associated with the road, cabins and dam are difficult to predict. However, based on previous experiences (*e.g.*, Skogland, 1986), it is plausible that the planned construction of additional 161 private cabins in the migration bottleneck (Strand *et al.,* 2011; Fig. 1c, d) will have critical consequences for the persistence of migration and for population viability. The data for the 10 GPS radio-collared females indicate that the migration corridors are very small: one is located in a very small geographic bottleneck of ca 1 km between two lakes, and the other covers an area of ca. 4 km (Fig. 1c, d). This implies that the implementation of mitigation measures at this stage might still be relatively cost-efficient. We are currently collecting GPS data for more animals on a finer temporal scale (*i.e.*, 15 min) to be able to assess more precisely the location of the migration corridor and analyze the relative effect of factors affecting the time and location of crossing, in order to assist the development of efficient mitigation measures. We recommend developing urgent, targeted and sound conservation plans to secure a future for the migration of the Setesdal Aushtei population.

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Efficacy of calf:cow ratios for estimating calf production of arctic caribou

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Abstract: Caribou (*Rangifer tarandus granti*) calf:cow ratios (CCRs) computed from composition counts obtained on arctic calving grounds are biased estimators of net calf production (NCP, the product of parturition rate and early calf survival) for sexually-mature females. Sexually-immature 2-year-old females, which are indistinguishable from sexually-mature females without calves, are included in the denominator, thereby biasing the calculated ratio low. This underestimate increases with the proportion of 2-year-old females in the population. We estimated the magnitude of this error with deterministic simulations under three scenarios of calf and yearling annual survival (respectively: low, 60 and 70%; medium, 70 and 80%; high, 80 and 90%) for five levels of unbiased NCP: 20, 40, 60, 80, and 100%. We assumed a survival rate of 90% for both 2-year-old and mature females. For each NCP, we computed numbers of 2-year-old females surviving annually and increased the denominator of the CCR accordingly. We then calculated a series of hypothetical "observed" CCRs, which stabilized during the last 6 years of the simulations, and documented the degree to which each 6-year mean CCR differed from the corresponding NCP. For the three calf and yearling survival scenarios, proportional underestimates of NCP by CCR ranged 0.046–0.156, 0.058–0.187, and 0.071–0.216, respectively. Unfortunately, because parturition and survival rates are typically variable (*i.e.*, age distribution is unstable), the magnitude of the error is not predictable without substantial supporting information. We recommend maintaining a sufficient sample of known-age radiocollared females in each herd and implementing a regular relocation schedule during the calving period to obtain unbiased estimates of both parturition rate and NCP.

Key words: composition counts; errors; fecundity; herd productivity.

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Introduction

Herd composition counts are commonly used in ungulate management (Bender, 2006), but the ratios obtained are the subject of ongoing debate (Caughley, 1974; McCullough, 1994; Bonenfant *et al.,* 2005; Harris *et al.,* 2008). Given the potential to provide ambiguous information on population dynamics (Caughley, 1974), these ratios require a number of restrictive assumptions for proper usage (McCullough, 1994). Composition counts may also be affected by unequal mixing of herd components (Bonenfant *et al.,* 2005). Nonetheless, late winter composition counts may adequately index overwinter calf survival and herd growth where 1) biased sampling is absent, 2) precise calf:cow ratios are available, 3) adult survival is precisely estimated, 4) the variance in juvenile survival swamps variance in subadult and adult survival, and 5) parturition is relatively constant (Hatter & Bergerud, 1991; Harris *et al.,* 2008; DeCesare *et al.,* 2012). However, it is not clear that these fairly restrictive assumptions can be consistently met for long-term field monitoring.

Within a few weeks after calving, the female segment of caribou (*Rangifer* tarandus granti) herds that calve north of the Arctic Circle (hereafter referred to as "arctic caribou") consists of sexually-mature individuals (with and without calves), sexually-immature 2-year-olds, and yearlings. Whereas yearling females typically are distinguishable from older females, it is nearly impossible to accurately classify 2-year-olds. Hence, 2-year-old and older females are combined in a single class called "cows" or, inappropriately, "adult females" for the calculation of CCRs.

In caribou herds that calve south of the Arctic Circle (hereafter called "subarctic caribou") up to 48% of 2-year-old females may be pregnant (*i.e.*, breed as yearlings at ca. 16 months of age; c.f. Bergerud *et al.,* 2008). In arctic herds, however, females rarely breed as yearlings and calve as 2-year-olds. For example, in the Porcupine herd (PCH) only 2 of 41 (5%) radiocollared and monitored 2-year-old females, 2004–2011, were observed to be parturient; and in only two of these years were any 2-year-olds parturient (J. Caikoski, ADF&G, pers. comm.). For the Central Arctic herd (CAH), only 3 of 94 (3%) radiocollared 2-year-old females monitored, 1994–2010, were parturient; and in only three of these years were any 2-year-olds parturient (Lenart, 2011). For the Teshekpuk caribou herd (TCH), only 3 of 46 (7%) radiocollared 2-year-old females monitored, 1993–2012, were parturient; and in only three of these years were any 2-year-olds parturient (Parrett, 2009). These values would be overestimates if less effort was invested in monitoring any nonparturient females located outside of the calving ground.

Caribou CCRs remain commonly used to index the product of parturition rate and calf survival (proportion of all ≥ 2-year-old cows that gave birth and retained calves by the survey date) of caribou herds in Alaska (*e.g.*, Boertje *et al.,* 1996; Valkenburg *et al.,* 2004). Typically, stratified random or systematic surveys are conducted on the calving grounds shortly after the peak of calving, and caribou are classified as bulls, cows, yearlings, and calves. The CCR for any particular survey is estimated as the number of calves observed per 100 cows observed.

A more explicit and ecologically relevant term than CCR is "net calf production" (NCP, the proportion of sexually-mature females observed with surviving neonates ~1–4 weeks after calving). For any arctic herd, NCP can be confidently estimated as the proportion of a sample of radiocollared females, known to have been fecund previously or at least 3 years old, that are accompanied by calves (Whitten *et al.,* 1992; Cameron *et al.,* 1993; 2005).

A CCR derived from survey counts is an underestimate of NCP for arctic herds, owing to observer inability to distinguish between 2-year-old females that are rarely parturient and sexually-mature females with parturition rates typically in the range of 70–90% (PCH, 2005–2011, J. Caikoski, pers. comm.; CAH, Cameron *et al.,* 2005). Including 2-year-olds artificially increases the CCR denominator, resulting in an underestimate of NCP. High survival rates of calves (prior-year) and yearlings would further inflate the CCR denominator, exacerbating underestimates of NCP by CCR. In contrast, for subarctic caribou herds in which a highly variable proportion of yearlings may breed (*e.g.*, 8–48%; Bergerud *et al.,* 2008), underestimation of NCP by CCR may be reduced in some years. However, quantifying such bias in CCRs would require precise annual estimates of 2-year-old female parturition rates, and these data are rarely available.

Dilution of the CCR denominator by sexually-immature 2-year-old females is intuitively obvious; yet status assessments of herds based on CCRs seldom make reference to the bias, perhaps because the data necessary for calibration (*i.e.*, proportion of all ≥ 2-year-old females that are sexually- immature 2-year-olds) are rarely available. In some cases, potential bias in CCRs has been acknowledged in the context of demographic modeling when the age composition of the female component of the population fluctuated with cohort-specific changes in survival. For example, Boulanger et al. (2011) encountered difficulties interpreting a decline in apparent calf recruitment in the Bathurst herd (Canada) based on CCRs and appropriately recognized the possibility of biased estimates of fecundity when the denominator in CCR included a substantial proportion of females in the relatively unproductive young and old age classes. Regardless, sexually-mature females are the most appropriate denominator for calculating NCP because they are the most numerically stable of the sex-age classes (McCullough, 1994; Harris *et al.,* 2008) and the only class of arctic caribou females likely to give birth.

With the increasingly widespread use of very high frequency (VHF) radio collars, NCPs can now be estimated by relocating sexually-mature females during the calving period and observing their maternal status. Even so, some biologists continue to routinely conduct separate calving ground and/or postcalving surveys to estimate CCRs, particularly when the number of radio collars deployed is considered insufficient.

Our objectives were to 1) document the magnitude and range of underestimates of NCP by CCR for arctic caribou using deterministic simulation modeling based on three survival scenarios for calves and yearlings, and 2) evaluate the utility of NCP as an estimator of herd growth status, given variable survival; and, by implication, the additional uncertainty introduced by CCR underestimates.

Methods

We conducted 10-year deterministic simulations using three different scenarios of overwinter calf and annual yearling survival (respectively: low, 60 and 70%; medium, 70 and 80%; high, 80 and 90%). We assumed 50% females among yearlings and assigned a constant 90% annual survival for females ≥ 2 years old. For each scenario, we projected annual numbers of surviving yearlings and 2-year-old females at five levels of NCP (20, 40, 60, 80, and 100%) and sequentially totaled the 1) number of females ≥ 3 years old and assumed to be sexually-mature (*i.e.*, the denominator for calculation of NCPs) and 2) number of sexually-mature females plus sexually-immature 2-year-old females (*i.e.*, the denominator of CCRs).

We began each simulation with 100 sexually-mature females, calculated numbers of current-year calves for each NCP level, and calculated future totals for yearlings, 2-year-old females, and sexually-mature females based on the set of survival rates specified. We totaled the accrued sexually-mature females with the surviving previous-year sexually-mature females; the sum was then used as the basis for calculating numbers of calves present at the specified NCP and for projecting numbers of mature females surviving through the following winter. We repeated that same procedure for each successive year through year 10. For each level of NCP and survival scenario, we summed sexually-mature and 2-year-old females to obtain a denominator typically used for CCR calculations. We then produced a series of hypothetical "observed" CCRs, computed a geometric mean CCR for the last six years of the simulation, and documented the degree to which each of those means differed from the corresponding NCP.

Finally, for each NCP, we computed a geometric mean annual rate of change in the number of sexually-mature females generated

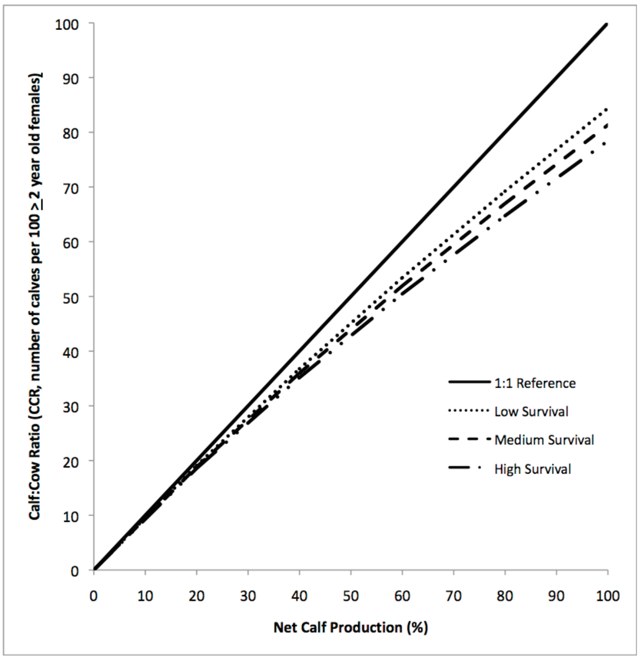

Fig. 1. Calf:cow ratios (CCR; *i.e.*, where the denominator includes sexually-immature 2-year-olds) in relation to net calf production (NCP; *i.e.*, where the denominator includes only ≥ 3-year-old sexually-mature females) at low (60, 70%), medium (70, 80%), and high (80, 90%) levels of overwinter calf and annual yearling survival, respectively (see Table 1).

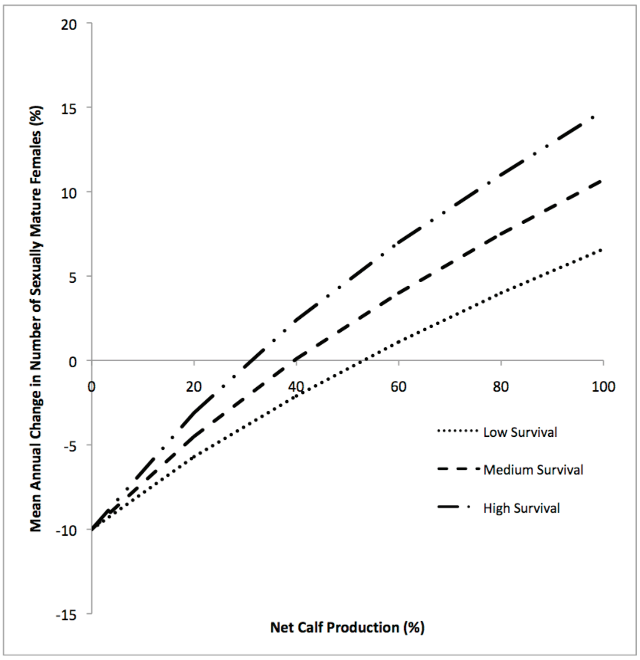

Fig. 2. Mean annual rates of change in the number of sexually-mature females in relation to net calf production at low (60, 70%), medium (70, 80%), and high (80, 90%) levels of overwinter calf and annual yearling survival, respectively (see Table 1).

by the simulations and described those trends within each of the three survival scenarios. We restricted our analyses to the last six years of the simulations in order to attain a stable age distribution.

Results

CCR underestimates of NCP increased with the progressive addition of 2-year-old females to the simulated population, from 0.046, 0.058, and 0.071 at 20% NCP to 0.156, 0.187, and 0.216 at 100% NCP for low, medium, and high rates of survival, respectively (Table 1, Fig. 1). Annual rates of change in the number of sexually-mature females were positively correlated with NCP for each of the three survival classes, increasing from ca. –3 to –6% at 20% NCP to ca. 7 to 15% at 100% NCP (Table 1, Fig. 2).

Discussion

Our intent was to illustrate the source, direction, and magnitude of biases incurred in the estimation of CCRs from survey observations at calving for arctic herds specifically. It was not our objective to offer a practical means of correcting those biases. Doing so would require annual overwinter survival estimates for calves, yearlings, and females ≥ 2 years old from a comprehensive radiotracking program. With those data, CCR underestimates could be calculated, and NCPs derived with some confidence. If one did have access to such data, however, NCPs could be estimated directly, and correcting CCRs would become a superfluous exercise.

Nonetheless, for the manager limited to the use of CCRs and interested in approximating the attendant errors, we do identify the component variables and structure of a simple model suited for those calculations. One could, for example, further align one or more of the age class survival rates to the estimated or suspected value(s) and compute a herd- and year-specific

Table 1. Underestimates of net calf production (NCP) by calf:cow ratios (CCR) and changes in the number of sexually-mature females derived from 10-year simulations.

Survival scenario	Net calf production (NCP, calves per 100 ≥ 3-year-old females) (%)[a]	Calf:cow ratio (CCR, calves per 100 ≥ 2-year-old females) (%)	Proportion of CCR denominator that is 2-year-old females	Proportional underestimate of NCP by CCR[b]	Annual change in number of sexuall mature females (%)[c]
Low[d]	20	19.1	0.053	0.046	−5.7
	40	36.8	0.105	0.081	−2.1
	60	53.4	0.154	0.110	1.1
	80	69.2	0.201	0.135	4.0
	100	84.4	0.245	0.156	6.6
Medium[e]	20	18.8	0.067	0.058	−4.5
	40	36.0	0.129	0.101	0.1
	60	51.9	0.186	0.135	4.0
	80	67.0	0.238	0.163	7.5
	100	81.3	0.287	0.187	10.7
High[f]	20	18.6	0.083	0.071	−3.1
	40	35.2	0.154	0.121	2.4
	60	50.5	0.217	0.159	7.0
	80	64.8	0.274	0.191	11.0
	100	78.4	0.325	0.216	14.8

[a] Percentage of sexually-mature (≥ 3 years old) females accompanied by calves ca. 3 weeks postpartum.
[b] Differences between each mean calf:cow ratio (CCR) and the corresponding net calf production (NCP).
[c] Mean percentage change between successive years; sexually-mature females are ≥ 3 years old.
[d] Calves, 60%; yearlings, 70%; females ≥ 2 years old, 90%.
[e] Calves, 70%; yearlings, 80%; females ≥ 2 years old, 90%.
[f] Calves, 80%; yearlings, 90%; females ≥ 2 years old, 90%.
NOTE: At NCP = 0, all mean CCRs and proportional underestimates = 0, and all mean annual changes in the number of sexually-mature females = –10% (*i.e.*, reflecting a constant survival rate of 90%).

estimate of bias for guidance. Or the fecundity of yearling females (*i.e.*, 2 years of age at calving), here assumed to be zero, could be raised if there was appropriate evidence for a significant parturition rate for this age class. The bias we document here would be reduced in direct proportion to the proportion of 2-year-old females that are parturient in a particular situation; and our use of adult survival of 90%, admittedly optimistic, indicates that our documentation of the underestimation of NCP by CCR is conservative. Likewise, our use of 60% as the low value for overwinter survival of calves implies much lower annual survival of calves (early calf mortality is subsumed in NCP) and minimizes bias in CCR.

The constraints imposed by our deterministic simulation are clearly artificial because both fecundity and survival are inherently variable; and for calves and yearlings, there are 2- and 1-year lags, respectively, before survivors become 2-year-old females that are incorporated into the CCR denominator. Thus, an investigator will rarely know the magnitude of bias in a CCR estimated from field surveys without substantial supplemental information.

Annual rates of change in the number of sexually-mature females (here, a surrogate for herd size) relative to NCP (Fig. 2) include a plausible range of herd growth rates, lending credence to the choice of numerical input variables. In reality, however, NCPs < 40% are unlikely, unless parturition rate is low and early postnatal mortality is extreme due to disease, starvation, or heavy predation. A 1-year reproductive pause is sufficient for mature females to regain body condition and resume breeding (Cameron, 1994; Cameron & Ver Hoef, 1994); hence, alternate-year breeding imposes a lower limit of 50% on parturition rate of sexually-mature females. An NCP of 100%, indicating parturition status of all sexually-mature females and no early calf mortality, is improbable as well.

Our calculated underestimates of NCP by CCR are not trivial. For NCPs of 40 and 100%, which easily encompass the range commonly observed in arctic caribou, CCRs projected over the range of survival rates evaluated (high to low) were 35–37 calves/100 cows and 78–84 calves/100 cows, respectively, yielding proportional underestimates ranging 0.081–0.216 (Table 1). Hence, CCRs are reasonable estimates of NCP when parturition and/or calf survival rates are low (ca. <40%), but the bias increases progressively thereafter, with CCRs becoming little more than approximations of NCP at the upper end of the scale (Fig. 1).

Counterintuitive results may also occur. For example, very low overwinter survival of yearlings would reduce the proportion of sexually-immature 2-year-old females subsequently entering the population at calving, which would reduce the negative bias that these animals typically impose and yield a relatively inflated CCR. This might falsely suggest that increased productivity had compensated for earlier reduced survival of younger age classes when, in fact, the increase in observed CCR was an artifact of previously-reduced survival.

Even direct estimates of NCP are of limited value for monitoring population trend because of uncertainty introduced by the range of plausible, but unknown survival rates for the various age classes. At an NCP of 40%, estimates of annual herd growth ranged −2 to 2%, depending upon the survival values applied in the simulations; while at an NCP of 90%, the range in growth rate increased to 5–13% (Table 2). Through a broad range of NCP, ~30% to ~60%, herd growth or decline is ambiguous if the mean and variance in survival are not known (Fig. 2).

Clearly, even NCPs offer only generalizations on herd trend; parturition rate and early calf survival are confounded in NCP, making it difficult to identify the limiting season without substantial supplemental information. Relying solely on CCRs would exacerbate the problem by biasing estimates of NCP downward, thereby underrating herd productivity and suggesting more conservative management strategies than warranted, increasingly more so as calf production increased. For the same reasons, CCRs are unsuitable as response variables in the evaluation of ecological processes or for population modeling.

Although our analysis was restricted to bias in CCRs, it should be noted that yearling:cow ratios are subject to identical problems with the ratio denominator. In addition, representative sampling may be quite challenging. Surveys conducted during the calving period must be broadened sufficiently beyond the calving grounds to include any yearlings located elsewhere, commonly in groups with bulls (R.D. Cameron & L.S. Parrett, unpubl. data). Similarly, sex-age composition counts of large postcalving aggregations and rutting concentrations, daunting enough under the best of circumstances owing to non-uniform distribution, are further complicated by increasing difficulty distinguishing between yearlings and 2-year-old females as the season progresses.

Despite problems with sampling and interpretation, CCRs are routinely used as indices of early calf survival (*e.g.*, Boertje *et al.,* 1996; Valkenburg *et al.,* 2004) and subsequent recruitment (*e.g.*, Dau, 2009; Parrett, 2009). In both of the above applications, making inferences on either long- or short-term trends in population "productivity" is hampered by the presence of an unknown proportion of sexually-immature females in the CCR denominator. Further, estimating the relative contributions of fecundity and survival to recruitment is not possible because the CCR numerator is a product of those two variables.

For a monitoring program, we recommend maintaining a sample of known-age radiocollared females, marked as yearlings (collars replaced as necessary, where practical), and implementing a relocation schedule appropriate to study objectives or management needs. If resources are insufficient for a sample size that yields adequate analytical power, we suggest a target of ~25 collars, the minimum required by many peer-reviewed journals to report proportions. Relocations during the calving period would provide estimates of parturition rate and NCP (and, by difference, early calf survival) without the bias introduced when using CCRs. Additional relocations during autumn and early spring would be desirable to estimate subsequent survival of calves, yearlings, and adult females.

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Have geographical influences and changing abundance led to sub-population structure in the Ahiak caribou herd, Nunavut, Canada?

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Abstract: We examined the premise that changing abundance and environmental conditions influence the seasonal dispersion and distribution of migratory tundra caribou (*Rangifer tarandus groenlandicus*). The Ahiak herd's (north-central Nunavut, Canada) calving shifted from dispersed on islands to gregarious calving on the mainland coast. As abundance further increased, the calving ground elongated east and west such that we proposed a longitudinal climate gradient. As well, the calving ground's east and west ends are different distances from the tree-line, which dips south closer to Hudson Bay. We proposed that whether caribou winter on the tundra or within boreal forest and the different climate across the long calving ground could contribute to differential survival and productivity such that sub-population structure would result. At the scale of the individual cows (identified through satellite-collars), we did not find inter-annual spatial fidelity to either the western or eastern parts of the calving ground. At the population scale (aerial surveys of calving distribution), we also did not find discontinuities in calving distribution. The spatial association of individual cows during calving compared with their association during the rut was inconsistent among years, but overall, cows that calve together, rut together. At this time and with the available evidence, we could not infer sub-population structure from shifts in dispersion and distribution as influenced by geography and changes in abundance for the Ahiak herd.

Key words: Ahiak herd; calving; geography; *Rangifer tarandus groenlandicus;* rutting; spatial fidelity; sub-population; tree-line.

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Introduction

The general premise for this paper is that physical geography (landforms and climate) and abundance influence migratory tundra caribou's (*Rangifer tarandus groenlandicus*) seasonal and annual distribution (Bergerud *et al.,* 2008). Seasonal ranges contract or expand with changes in abundance within the constraints imposed by how physical geography influences distribution.

The Ahiak herd (previously named as the Queen Maud Gulf herd; Gunn *et al.,* 2000) seasonally ranges mostly in north central Nunavut, Canada (Fig. 1). We first describe how abundance and physical geography influence the herd's calving and winter distribution. Sec-

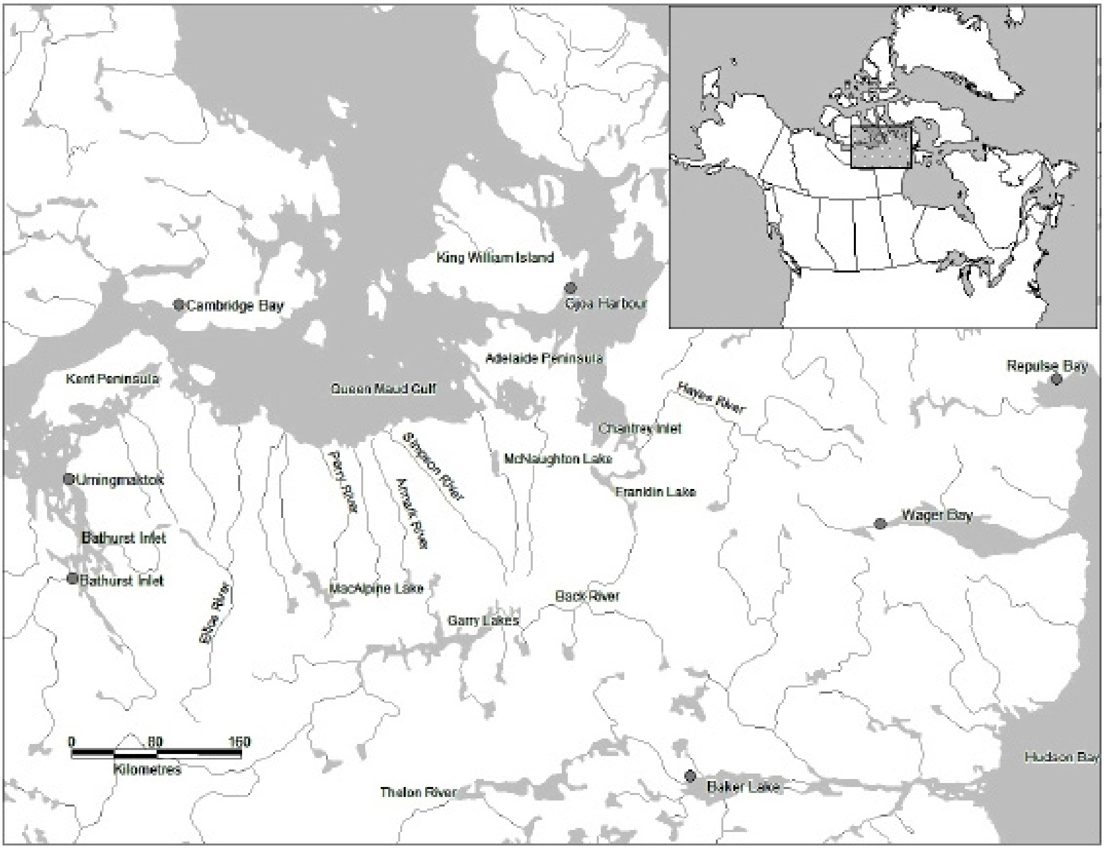

Fig. 1. Queen Maud Gulf and surrounding area, north central mainland, Nunavut.

ondly, we examined whether those influences contribute toward sub-population structure within the Ahiak herd. Wells & Richmond (1995) defined a subpopulation as *"an arbitrary spatially-delimited subset of individuals from within a population"*.

The geographical influences are firstly, the configuration of Queen Maud Gulf's coast line which stretches in a relatively straight east-west direction for 300 km and has 100s of islands and islets. Compared to other caribou calving grounds, the Ahiak herd's annual calving grounds (*sensu* Russell *et al.,* 2002) are unusually long and narrow (~340 km x ~75 km).

The second geographical influence is how the extent of the low-lying coastal plains of the northeast Nunavut mainland affects the position of the tree-line and in turn, caribou winter and pre-calving migration distribution. Within mainland Canada, from west to east toward Hudson Bay, the low elevation arctic coastal plain widens out and consequently the tree-line (Timoney *et al.,* 1992) dips south and widens (Fig. 2). The tree-line is more accurately described as a forest-tundra biome (Timoney *et al.,* 1992; Payette *et al.,* 2001) lying between the southern limit (<0.1% cover) of upland tundra and the northern limit (<0.1% cover) of trees >3-4 m tall. From Mackenzie Delta to north of Yellowknife, the forest-tundra biome is 60 -150 km wide compared to 230–340 km for the zone from the Dubawnt River to central

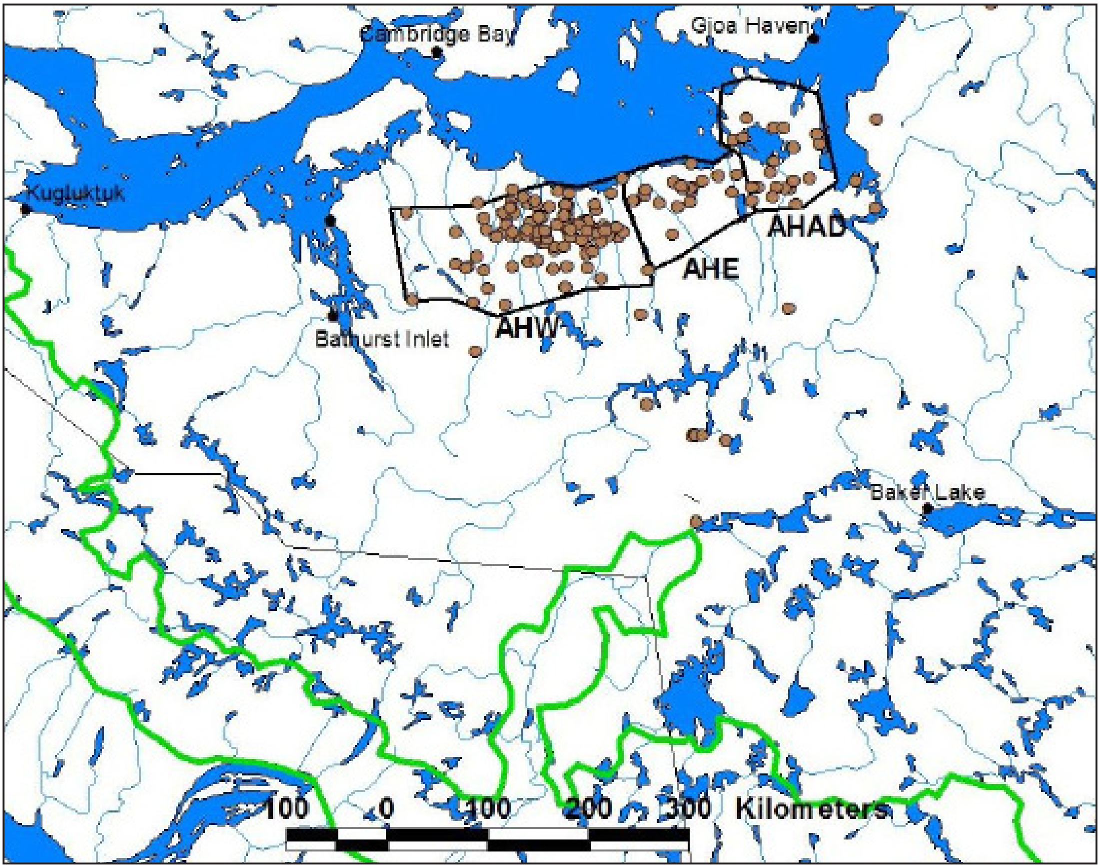

Fig. 2. Location of the Ahiak West (AHW), East (AHE), and Adelaide Peninsula (AHAD) blocks of the Ahiak calving grounds, assigned primarily based on aerial survey data 2006-09. Brown dots are June calving locations from collar data, 1995-2010. The tree-line and forest-tundra biome is shown within the green polygon (Timoney *et al.,* 1992; Payette *et al.,* 2001).

Manitoba-Keewatin (Timoney *et al.,* 1992). The southeast extension of the tundra west of Hudson Bay increases the likelihood for some Ahiak caribou to be wintering on the tundra.

In the paper's second part, we examine how the geographical influences (long straight coast line and position of the tree-line) could lead to sub-population structure. A longitudinal environmental gradient in the timing of snowmelt or plant green-up across the east-west length of the calving ground could influence early calf survival. If individual cows had fidelity between years to either the western or eastern calving ground and the environmental gradient is marked; sub-population structure could develop if the association of cows on the calving grounds is similar to the pattern of association during the rut. Additionally, if individual cows wintering on the tundra or within the forests are more likely to calve at the east or west end of the elongated coastal calving ground, then sub-population structure could develop or be accentuated in response to different environmental conditions during winter and pre-calving migration. Cows wintering within forest-tundra and lichen woodland communities of the boreal forest (Payette *et al.,* 2001), versus those wintering on the tundra, will face different ecological conditions during winter and pre-calving migration, which could affect their reproductive success and survival.

Sub-population structure development

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would depend on inter-annual fidelity by individual breeding females within the calving ground. While annual fidelity of individual cows to calving grounds is well documented, fidelity to specific sites has not been reported (for example, Fancy & Whitten, 1991). Our premise was that the length of the calving ground would be sufficient for cows to possibly exhibit annual fidelity to part of the calving ground.

Our objective is to review evidence for use patterns of the calving grounds at the individual scale (based on satellite-collared cows) and at the population-scale (using information from aerial surveys of calving distribution). Our hypothesis is that the Ahiak calving grounds are comprised of two or more calving sub-populations.

By focusing on a relatively detailed description of space use by the Ahiak herd, we aim to contribute to better understand the spatial dynamics of migratory caribou. The Ahiak herd is unusual in its long narrow calving ground and if it contributes to sub-population, the information will be useful to management. We need to be aware of similarities and differences between herds especially in the context of longer-term changes in space use as herds change in abundance.

Methods

We compiled information from government reports (Table 1) to derive descriptive statistics and spatial analyses for telemetry (1996-2008) and aerial survey data (2006-10) (Government of Northwest Territories, Department of Environment and Natural Resources). We also used 11 calving and rut locations from 2008-10 from a cooperative project for monitoring and baseline studies, Agnico-Eagle Mines Limited, AREVA Ltd and Government of Nunavut (M. Campbell, pers. comm.; Gebauer *et al.,* 2011). We tested four predictions that followed from our negative hypothesis:

1) Distribution of calving cows would be discontinuous across the calving ground and individual breeding females would show inter-annual fidelity to discrete areas within annual calving grounds observed in consecutive years.

2) Based on the sample of collared cows, 5% or less of the collared breeding females would switch calving locations in subsequent years between the western, central and eastern areas of an annual calving ground. (We selected 5% based on annual rates of switches of breeding females).

3) Based on aerial surveys of the Ahiak calving grounds, 2006-10, there would be an observable pattern of discontinuity along the east-west axis in the distribution of breeding females and newborn calves (measured at a grid scale interval of 10 km).

4) Cows that calved together on an annual calving ground (at the scale of the east or west halves of the calving ground), will not solely associate together during the subsequent fall rut. Thus, within a year, collared breeding females that calved within i) the western portion of the calving ground, or ii) the combined central and eastern portions would not be associated with other cows that calved within the same area of the coastal calving ground during the subsequent fall rut (temporal index is 20 October) at a rate greater than would be expected by chance (*i.e.*, >50% probability).

Distribution and abundance of caribou calving in the Queen Maud Gulf

To describe caribou calving along the coast of Queen Maud Gulf relative to changes in abundance, we compiled earlier accounts (Gavin, 1945; Appendix G in Gunn *et al.,* 2000), pre-calving surveys from 1983 and 1993 (Heard *et al.,* 1987; Buckland *et al.,* 2000), and systematic aerial calving ground surveys in 1986, 1996, and 2006-2010 which were strip transect surveys designed to map the distribution and estimate densities of caribou (Gunn *et al.,* 2000; D. Johnson, unpubl. data). We also include

Table 1. Sources and type of data used in analysis.

Type	Source	No. cows	Publication or project leader[a]	Project objectives or rationale
1996-98 telemetry	WMIS[b]	4	Gunn et al., 2000	Herd identity of wintering caribou east of Bathurst Inlet
2001-02 telemetry	WMIS	9	Gunn & D'Hont, 2002	Bathurst herd project (overlaP on winter ranges)
2006-08 telemetry	WMIS	33	D. Johnson	Beverly distribution
2009-10 telemetry	WMIS	21	A. Kelly	Beverly distribution
2009-10 telemetry	NIRB Public Registry	7	M. Campbell; Gebauer et al., 2011	Monitoring for Meadowbank mine; baseline data for AREVA Ltd.
2006-08 calving surveys	WMIS		D. Johnson unpubl. data; Poole et al., 2013; A. Kelly	Assess distribution and trend in density breeding females

[a] Use of data does not imply agreement with the interpretations presented in this paper.

[b] Wildlife Management Information System, Government of Northwest Territories

information from wildlife baseline surveys for mining development (Calef & Hubert, 2002; Rescan, 2011).

The 2006-10 aerial systematic surveys used transects aligned north-south and on-transect sightings were recorded as sex age classes and allocated to 10 km segments to quantify both dispersion and distribution of caribou on the calving grounds (D. Johnson, unpubl. data; Poole *et al.,* 2013). To compare the 2006 to 2010 systematic calving distribution (D. Johnson, unpubl. data), we created 90% fixed kernel polygons on the un-weighted distribution of breeding females each year and examined overlap between sequential calving grounds; where percent overlap in polygons = (*Common area in year i and year i+1)/((∑ area year i and year i+1)/2*).

Climate gradients across the calving ground

Russell et al. (2013) used a retroactive spatial climate database to create climate descriptions for the seasonal ranges of circumpolar migratory tundra caribou. The database consisted of 22 "caribou-relevant" climate indicators that covered the period 1979-2011. The actual derivations of these 22 variables are presented in Russell et al. (2013). We used this database to compare climate in the three Ahiak calving ground blocks (see next section for a description of the blocks). We tested for differences in climate variables and decadal trends among the blocks using an Analysis of Variance and Duncan's-Waller multiple range tests (SAS, version 9.10). Seasons presented in the analysis were: calving (June), summer (July and August), autumn (September – November), winter (December – March) and spring (April – May).

Queen Maud Gulf calving and rut associations

To examine the relationship between calving and rut associations, we used both individual and population-scale information. The individual scale information was from 74 adult cows fitted with conventional satellite or GPS collars and we used those data to describe the dispersion of individuals and their association. The various projects fitting satellite collars on adult female caribou had different objectives and, with the exception of 2007, all caribou were collared on winter range (Gunn *et al.,* 2000; Gunn & D'Hont, 2002; Gunn *et al.,* 2013). In July 2007, the collaring was conducted on the post-calving range (Johnson & Fleck, 2009). We acquired the data from the Government of Northwest Territories through their Wildlife Management Information System (WMIS). We accessed the calving and rutting locations of caribou cows collared near Baker Lake in 2008 and 2009 from Gebauer et al. (2011; available on the Nunavut Impact Review Board's Public Registry; accessed December 2011). We compiled a database using collared cows with at least one calving location on Queen Maud Gulf coastal calving ground 1996-2010. We also included those cows with a history of calving on the traditional Beverly calving grounds (south of Garry Lakes) but which had shifted to the Queen Maud Gulf coastal calving ground (Nagy *et al.,* 2011; Gunn *et al.,* 2012). Calving date was based on examination of daily movement rates coupled with spatial movements in GIS, such that calving was determined from a rapid drop in daily movement rate (generally to <2 km/day) and localization, followed by a 5-10 day period of reduced movement (Fancy & Whitten, 1991). The rut and winter distributions were indexed from collar locations on 20 October and 1 February each year, respectively (Gunn *et al.,* 2000).

For the population-scale analyses, Poole & Nishi (unpubl. data) had divided the calving ground into three blocks based on a qualitative assessment of caribou density observed during systematic aerial surveys from 2006-10 (GNWT ENR WMIS): Ahiak West (AHW) – from the western boundary of calving as far east as the Armark River (roughly the eastern boundary of the high or medium density zones in the past); Ahiak East (AHE) – from Armark River east to just east of McNaughton Lake; and Ahiak Adelaide (AHAD) – Adelaide Peninsula to the western shore of Chantrey Inlet (Fig. 2). The east-west width of the blocks varied: AHW was 235 km wide; AWE 120 km and AHAD 85 km. We used these three blocks to compile descriptive statistics on the consecutive calving locations to describe inter-annual patterns of calving location fidelity for individual calving cows and distances to rut and winter locations. We tabulated the distances between consecutive annual calving locations to describe the frequency of distance classes between consecutive calving year i to year i + 1. For some analyses, we combined the blocks as two calving associations (see below).

To compare whether cows that calved together were also associated during the rut based on the locations of collared cows, we combined AHE with AHAD to increase sample sizes. This gave us two sets of calving locations which we termed calving associations, the western and central/eastern. First we described the dispersion of the collared cows during the rut relative to each other. For each year, we calculated for the two calving associations and the combined rut locations the mean (average X and Y coordinate) centroid and the mean distances (and standard deviations) from the centroids to the individual caribou locations; the distances be-

Table 2. Density of caribou in 1986, 1996 and 2007-08 during aerial calving distribution surveys over the Ahiak calving grounds.

Year	Aerial coverage (%)	Survey area (km^2)	Density (caribou/km^2)	Reference
1986	23.2	7,320[a]	1.4	Gunn et al., 2000
1996	5.2	21,901[a]	3.9	Gunn et al., 2000
2006	4.5	25,379[b]	3.1	D. Johnson, unpubl. data
2007	4.6	23,929[b]	3.0	D. Johnson, unpubl. data
2008	6.7	23,696[b]	1.1	D. Johnson, unpubl. data

[a] Stratum area (Gunn *et al.,* 2000).

[b] Approximate size of the 2006, 2007 and 2008 calving areas at peak of calving (D. Johnson, unpubl. data)

tween centroids for the calving associations and combined rut locations were also calculated. Distances calculations were performed using in-house developed programs that use MapInfo Professional libraries (Pitney Bowes Software, Troy, NY).

Our second approach to compare whether cows that calved together were associated during the rut was to determine whether the overlap in the area used by the two rutting associations was due to chance or were in disagreement. To describe the area used, we calculated minimum convex polygons (MCP) buffered with one standard deviation for the rut locations based on the associations of individual cows during calving. Mapping was done in MapInfo Professional software. We compared the two rut MCPs to test if their differences are due to 'chance' or disagreement using the Kappa Index of Agreement (KIA) (http://www.spatialanalysisonline.com/output/). We used Idrisi GIS software (Clark Labs, Worcester, MA) to calculate the KIA for all pairings of the calving associations (by year) and combined rut locations and MCPs of the western and central/eastern calving associations. To judge the strength of the agreement, we used the classification by Landis & Kock (1977) (< 0 – poor; 0 to 0.2 – slight; 0.21 to 0.4 – fair; 0.41 to 0.6 – moderate; 0.61 to 0.8 – substantial; 0.81 to 1.0 – almost perfect).

For the spatial analysis of rut association, we removed three outliers as their rut distribution was highly distant (>400 km) from the other collared cows. We had no way of knowing the reproductive status, physical condition or health of these caribou which are possible reasons for the geographic isolation of single cows (or to the extent they were with other uncollared cows).

Results

Distribution and abundance of caribou calving in the Queen Maud Gulf

Caribou numbers were low in the Queen Maud Gulf coastal area and Adelaide Peninsula in the 1920s and increasing by the 1960s although population estimates were not obtained (Gavin, 1945; Banfield, 1950; Kelsall, 1968; Pelly, 2000; Gunn *et al.,* 2000). The rate of increase between the 1950s and 1980s was qualitative until the first systematic estimates in 1986 and 1996, which estimated that the herd increased 2-3 fold. The mean density of caribou on survey in 1996 was almost three times higher than in 1986 (Table 2), suggesting abundance had increased. Subsequently, during the period 1996–2007, the calving ground sur-

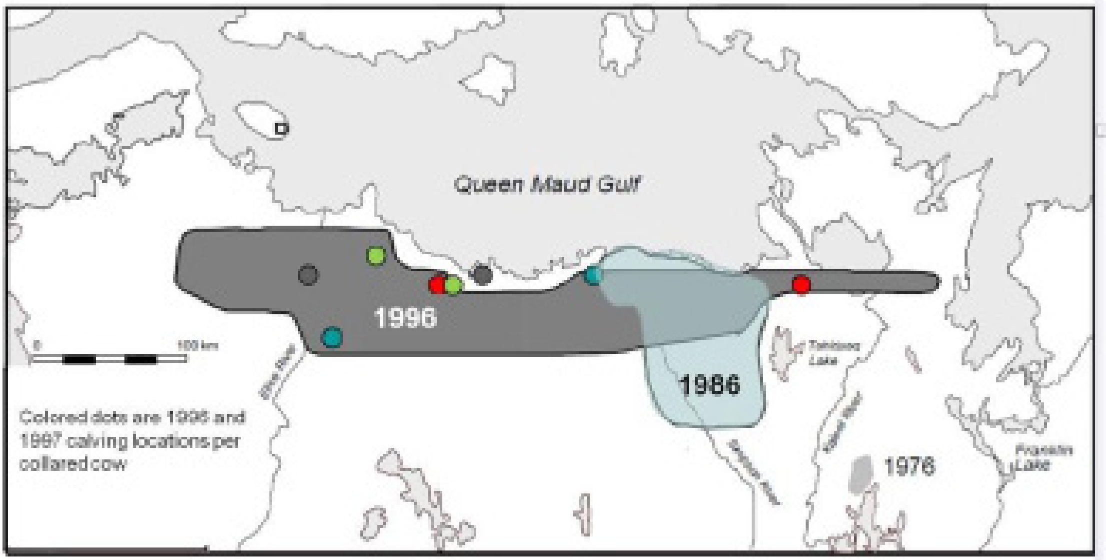

Fig. 3. The approximate extent of the Ahiak herd's calving distribution in June 1986 and 1996 (Gunn *et al.,* 2000), and 1996–97 calving locations from collared cows.

veys indicated no apparent trend in the density of caribou on the calving ground (Table 2). The lower densities in June 2008 may have reflected an unusual year as pregnancy rates were low and not all cows may have reached the calving ground or were later than the timing of the survey (D. Johnson, unpubl. data).

Gavin (1945) describes caribou calving dispersed on the numerous islands and islets along the shallow coast of Queen Maud Gulf. In June 1985 and 1986, cows and newborn calves were seen on the coastal islands (Gunn *et al.,* 2000; A. Gunn, unpubl. field notes). However, gregarious calving was observed on the adjacent mainland coast and during a systematic transect survey in June 1986, calving density was measured and distribution mapped with the western boundary of calving cows being the Simpson River (Gunn *et al.,* 2000).

By June 1995, the distribution of cows and calves extended to west of the Perry River and north of MacAlpine Lake (Gunn, 1996). The June 1995 survey overlapped a narrow survey area south of Kent Peninsula where baseline information on caribou distribution was collected from 1994 to 2002 for the Hope Bay mine project (Calef & Hubert, 2002). In June 1996, east-west systematic transects east of Bathurst Inlet mapped calving distribution east to the coast of Chantrey Inlet (Gunn *et al.,* 2000). The southern boundary of the calving ground was not delineated. Compared to 1986, the 1996 calving distribution had extended east and west of the 1986 distribution and was continuous across the width of the Queen Maud Gulf coastal area from Chantrey Inlet west to the Hope Bay area east of Bathurst Inlet (Gunn *et al.,* 1997; Gunn *et al.,* 2000) (Fig. 3).

Additionally, the basis for assuming that the 1986 calving ground had extended west was Inuit observations of pre-calving cows heading east of Bathurst Inlet (Gunn *et al.,* 2000). The capture sites east of Bathurst Inlet and movements of the collared caribou in April-June 1996 also supported an eastward pre-calving migration toward coastal Queen Maud Gulf. The calving locations for both 1996 and 1997 revealed use of the length of coastal calving grounds between the Ellice and Simpson rivers (Gunn *et al.,* 2000). Between 1996 and 2006,

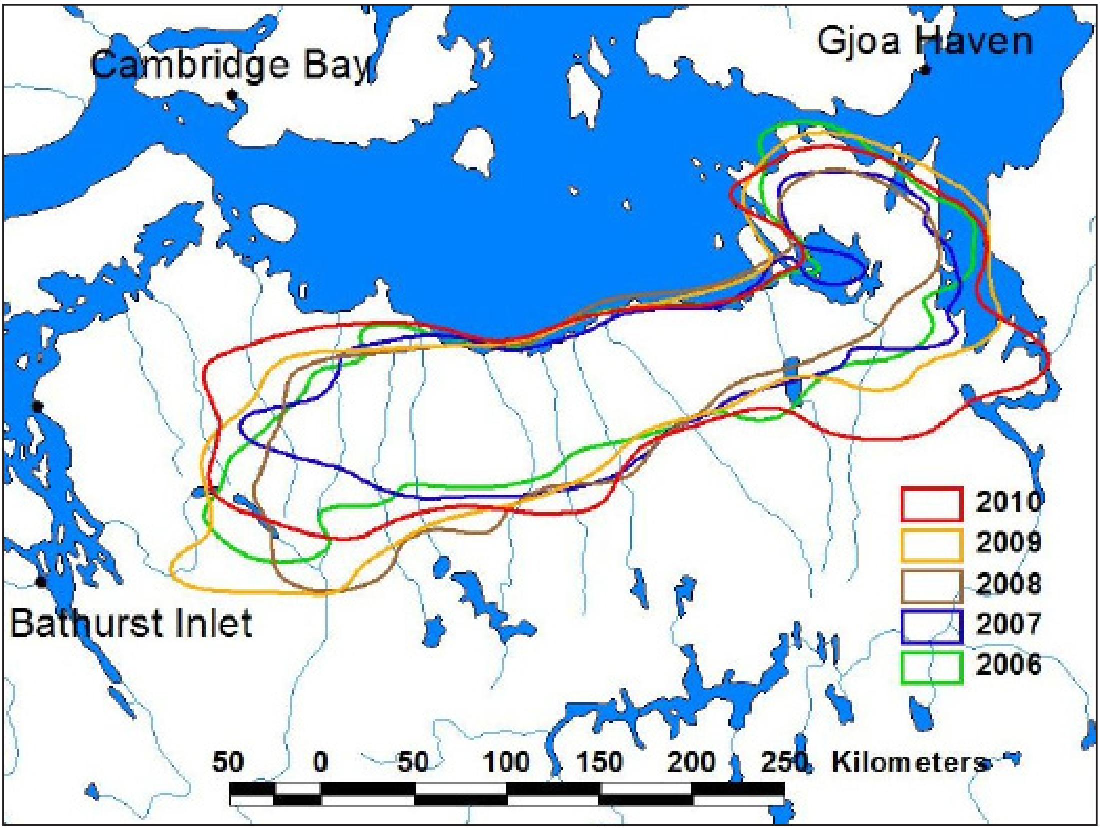

Fig. 4. Distribution of breeding females on the Ahiak calving grounds, 2006-10 (data source was D. Johnson, unpubl. data; Poole *et al.,* 2013).

the western boundary of calving distribution contracted from approximately the area just east of the Hope Bay-Spyder Lake corridor in 1996 to the vicinity of the Ellice River in 2006 (Gunn *et al.,* 2000; D. Johnson, unpubl. data). The comparison of the 2006-10 calving distributions along the Queen Maud Gulf showed an average 82% overlap (range 81–84%) during 2006-10, with changes among years primarily at eastern and western ends rather than changes in width (Fig. 4).

Climate gradients across the calving ground

There was a pattern in both winter and spring snow depth among blocks. In winter the AHE block had consistently highest snow 40.5 ± 2.1 (SE) cm and AHW consistently lowest snow depths (34.6 ± 2.0 cm). For the spring period leading up to calving the AHE block had significantly deeper snow than AHW (41.9 ± 2.7 cm versus 34.2 ± 2.2 cm, respectively) in every decade and higher than AHAD (35.3 ± 1.9 cm) in the 1990s. In the 2000s decade, this difference between AHE and AHW extended into the calving period (Table 3). Snow density in spring was consistently higher in AHE than AHW in all decades, significantly so in the 1980s and 1990s (0.272 ± 0.007 gm•cm-3 versus 0.254 ± 0.008 gm•cm-3, respectively).

If we consider decadal differences (Table 4),

Table 3. Significant differences for decadal snow depth, daily precipitation and daily temperature (ANOVA and Duncan's-Waller multiple range tests) among blocks based on MERRA climate variables.

Decade	Variable	Season	*P*	F value	Block Differences
1980s	snow depth	spring	0.012	4.58	AHE > AHW
1980s	snow depth	winter	0.008	5.27	AHE > AHW
1980s	snow density	spring	0.06	2.90	AHE > AHW
1980s	mean daily temperature	winter	0.0002	9.50	AHAD > AHE, AHW
1980s	daily precipitation	winter	0.0025	6.62	AHE, AHAD > AHW
1990s	snow depth	spring	0.0003	5.51	AHE > AHAD > AHW
1990s	snow depth	winter	0.0004	9.13	AHE > AHAD > AHW
1990s	snow density	spring	0.06	2.90	AHE > AHW
1990s	mean daily temperature	winter	<0.0001	43.39	AHAD > AHE > AHW
1990s	daily precipitation	winter	<0.0001	12.60	AHE > AHAD > AHW
2000s	snow depth	calving	0.09	2.65	AHE > AHW
2000s	snow depth	spring	0.06	2.95	AHE > AHW
2000s	mean daily temperature	winter	0.0003	9.16	AHAD > AHE > AHW
2000s	daily precipitation	winter	0.01	4.50	AHE > AHW

spring snow depths were consistently higher in the 1990s while the 1980s had the lowest spring snow depths. Spring snow depths were significantly higher in the 1990s compared to the 2000s in AHAD (40.6 ± 2.6 cm versus 35.3 ± 1.9 cm, respectively) and AHE (45.9 ± 2.7 cm versus 38.1 ± 2.0 cm, respectively). For all three blocks, drought index was higher in the 1990s compared to the other 2 decades during the calving (4.6 ± 0.91 versus 2.5 ± 0.46, respectively) and summer (19.3 ± 5.5 versus 9.7 ± 2.4, respectively) seasons. We did find differences in summer precipitation in the AHW and AHE blocks. In the AHE the mean daily precipitation in summer was significantly higher in the 1980s compared to the 2000s (46.6 ± 6.4 cumulative mm. month versus 36.6 ± 3.6, respectively) while the AHW had significantly higher summer precipitation in the 1980s compared to both the 1990s and the 2000s (46.3 ± 6.5 versus 35.1 ± 4.6, respectively). Further during the calving period in AHW the 2000s were the wettest decade, significantly higher than the 1990s and the 1980s (46.6 ± 6.4 versus 36.6 ± 3.6, respectively). Although there was a decadal and spatial pattern in respect to growing degree days (GDD) to July 15th, the relatively high annual variation meant that differences between decades and among blocks were not significant. Comparing decades, the 1990s experienced the highest GDD, while the 2000s were the coolest. Among blocks the AHW was consistently warmer (higher GDD) in every decade while the AHE had the lowest GDD in all decades. Mean GDD to July 15th were 388 ± 94.5 (AHW), 356 ± 76.9 (AHAD) and 345 ± 80.2 (AHE).

Queen Maud Gulf calving and rut associations

Spatial distribution of calving locations

We tested our prediction that 5% or less of the collared breeding females would switch calving locations in subsequent years between the western, central and eastern areas of an annual calving ground. We used 74 individual cows and 157 collar-years for all cows with at least one calving location along the Queen Maud Gulf coastal (Ahiak) calving ground (2001-10). We

Table 4. Significant differences for snow depth, daily precipitation and daily temperature (ANOVA and Duncan's-Waller multiple range tests) among decades based on MERRA climate variables.

Block	Variable	Season	*P*	F value	Decadal Differences
AHE	drought index	calving	0.008	5.84	1990s > 1980s, 2000s
AHE	drought index	summer	0.01	4.60	1990s > 1980s, 2000s
AHE	snow depth	spring	0.0009	5.01	1990s > 2000s
AHE	snow depth	winter	0.01	4.99	1990s > 1980s, 2000s
AHE	snow density	spring	0.09	2.45	1990s > 2000s, 1980s
AHE	snow density	winter	0.02	3.96	1990s, 2000s > 1980s
AHE	mean daily precipitation	summer	0.05	3.01	1980s > 1990s
AHE	mean daily precipitation	winter	0.01	4.84	1990s > 1980s, 2000s
AHAD	drought index	calving	0.02	4.54	1990s > 2000s
AHAD	drought index	summer	0.05	3.16	1990s > 2000s
AHAD	snow depth	spring	0.05	3.10	1990s > 2000s
AHAD	snow depth	winter	0.02	4.44	1990s > 2000s
AHAD	snow density	winter	0.03	3.72	1990s > 1980s
AHW	drought index	calving	0.02	4.28	1990s > 2000s
AHW	drought index	summer	0.0004	8.90	1990s > 2000s, 1980s
AHW	snow density	spring	0.01	4.40	2000s, 1990s > 1980s
AHW	snow density	winter	0.009	4.95	2000s > 1990s, 1980s
AHW	mean daily precipitation	calving	0.03	3.78	2000s > 1990s, 1980s
AHW	mean daily precipitation	summer	0.009	5.15	1980s > 2000s, 1990s

excluded six cows (8.1% - 6/74 cows; 5.7% - 6/105 collar-years) which calved on the Ahiak calving ground and switched to either the north east mainland (NEM) or Bathurst (BA) calving ground (1 cow AHW – BA – BA; one cow AHE – AHE – NEM – AHAD – AHE; two cows AHAD – NEM and one cow NEM – AHE). However, we included the eight cows whose initial known calving ground was the traditional Beverly ground and which subsequently calved on the Ahiak calving ground for at least 1 year. Agnico-Eagle Mines Limited and Government of Nunavut (M. Campbell, pers. comm.) deployed nine collars near Baker Lake, Nunavut in May 2008 and together with support from AREVA Ltd., deployed 21 collars on the tundra north of Baker Lake were in November 2009 (Gebauer *et al.,* 2011). Four collared cows calved on Adelaide Peninsula in 2008; two in 2009 and five in 2010 calved south of Adelaide Peninsula (Gebauer *et al.,* 2011). The other cows calved northeast of Chantrey Inlet within previously recorded calving distributions (Gunn & Fournier, 2000a; b).

We had only 1 year of calving locations for 18 cows on the Ahiak herd's calving ground and 15 of those (83%) calved within AHW. Of the 49 cows with 2 or more years of calving locations, 27 cows (55%) did not switch among the three blocks covering the length of the Ahiak calving ground and 24 (89%) of those cows calved in AHW. Conversely, 22 of 49 cows (45%) with more than 1 year's calving locations switched among blocks. Of these 22 cows that switched blocks, 13 of 14 (93%) cows with their initial calving location on the Ahiak calving ground calved 1 year in AHW then shifted to AHE or AHAD. Most shifts were west to east but two cows did shift from AHE or AHAD to AHW. Eight cows which initially calved on the traditional Beverly calving ground switched to the Ahiak calving ground: five of those cows with 2 or more years calving locations on the Ahiak calving ground switched among the blocks.

To test whether individual breeding females would show inter-annual fidelity to discrete areas within annual calving grounds observed in consecutive years, we examined the frequency of distance classes between consecutive calving year 1 to year 2; 72 paired calving locations. We restricted the period to 2006-10 as annual sample sizes were largest. Most (82%) were 25-150 km apart and the 10% tail end of the frequencies were separated by >150 km, which suggests use of the extent of the calving ground (Fig. 5).

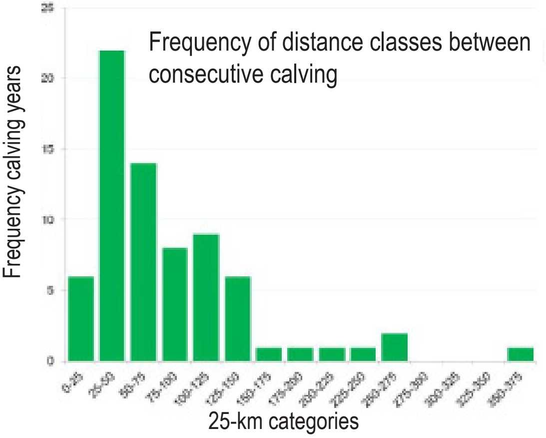

Fig. 5. Frequency classes based on distances between consecutive annual calving locations of satellite-collared breeding females, Ahiak herd, 1996–2010.

Calving to rut distances

The calving to rut distances varied but were not significantly different among AHW, AHE, and AHAD for 2006-10 (PROC GLM, F = 0.46, df = 2,120, *P* = 0.63; Table 5). The mean distances were 411 ± 14.6 km (*n* = 91) for the cows calving in AHW; 429 ± 28.2 km (*n* = 18) in AHE, and 379 ± 56.1 km (*n* = 14) for AHAD calving.

Table 5. Mean (standard errors) straight-line distances between calving and rut locations for collared cows that calved in either the Ahiak west (AHW), Ahiak east (AHE) and Ahiak Adelaide Peninsula (AHAD), blocks of the Queen Maud Gulf coastal calving area, 1996-2010.

	AHW			AHE			AHAD		
Year	$\bar{x}$ km	SE	*n*	$\bar{x}$ km	SE	*n*	$\bar{x}$ km	SE	*n*
1996	380	26.5	2	438		1			
1997	511	60.6	3						
2001	311	52.0	4						
2002	318	103.0	4	449	41.0	2			
2003	228		1				152		1
2004	644		1						
2005	265	88.7	5				412		1
2006	439	40.2	10	416	70.9	3			
2007	548	46.6	10	568	61.7	3	676		1
2008	359	23.7	19	310	58.0	3	153	41.3	4
2009	481	19.9	20	431		1	527	67.7	4
2010	339	11.1	12	412	56.6	5	541	0.1	2
Total	411	14.6	91	429	28.2	18	379	56.1	14

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Table 6. Mean distances (standard deviation) between individual rut locations and centroids for western and central/eastern calving association.

	Western calving association			Central/eastern calving association		
Year	$\bar{x}$ km	SE	*n*	$\bar{x}$ km	SE	*n*
2001	183	30.2	3	nda[a]	nda	
2002	200	44.1	4	nda	nda	
2005	138	38.4	4	nda	nda	
2006	137	28.6	10	177	37.2	4
2007	72	12.6	9	58	10.4	3
2008	100	20.0	20	141	38.2	6
2009	150	12.3	20	143	15.3	5
2010	64	10.4	12	138	21.7	10

[a] nda = no data available.

Rut distribution relative to calving associations

We predicted that cows that calved together on an annual calving ground (at the scale of the east or west halves of the calving ground), will not solely associate together during the subsequent fall rut. Thus, within a year, collared breeding females that calved within i) the western portion of the calving ground, or ii) the combined central/eastern portions would not be associated with other cows that calved within the same area of the coastal calving ground during the subsequent fall rut (temporal index is 20 October) at a rate greater than would be expected by chance (*i.e.*, >50% probability). Our results were that the degree of dispersion of the collared cows during the rut (20 October) as measured by the mean distance of individual locations relative to the centroid for the two calving associations did not differ annually between the two calving associations (western: $\bar{x}$ = 118 ± 8.8 SE; central/eastern: $\bar{x}$ = 136 ± 14.3; t = 1.98, df = 108, $P = 0.29$; Table 6; Fig. 6). In 2007 the dispersion was less than the other years (individual locations were more tightly clustered around the centroid) for both calving associations. In 2010, the individual locations were more clustered for cows from the western calving association).

The distribution of the rut varied between years based on the geographic location and respective position of the centroids of western and central/eastern calving associations to all rut locations (Fig. 6). The distribution suggested that the centroids of western calving associations during the rut are shifted to the southwest and central/eastern calving associations to the northeast (centres of activity ranging 117–276 km apart), except in 2007 when the cows from both the east and west calving associations had least dispersion (centroids ~32 km) and were furthest to the southwest (Fig. 6, Table 7). The straight-line distances between the centroids of the rut distribution based on the two calving associations varied among years (33 – 276 km). The greatest distance between the centroids was in 2008 (276 km) which was likely an atypical year as pregnancy rates were low during 2007-08 winter (D. Johnson, unpubl. data). The distance was least in 2007 (33 km) compared to 202 km (2006), 138 km (2009) and 117 km (2010).

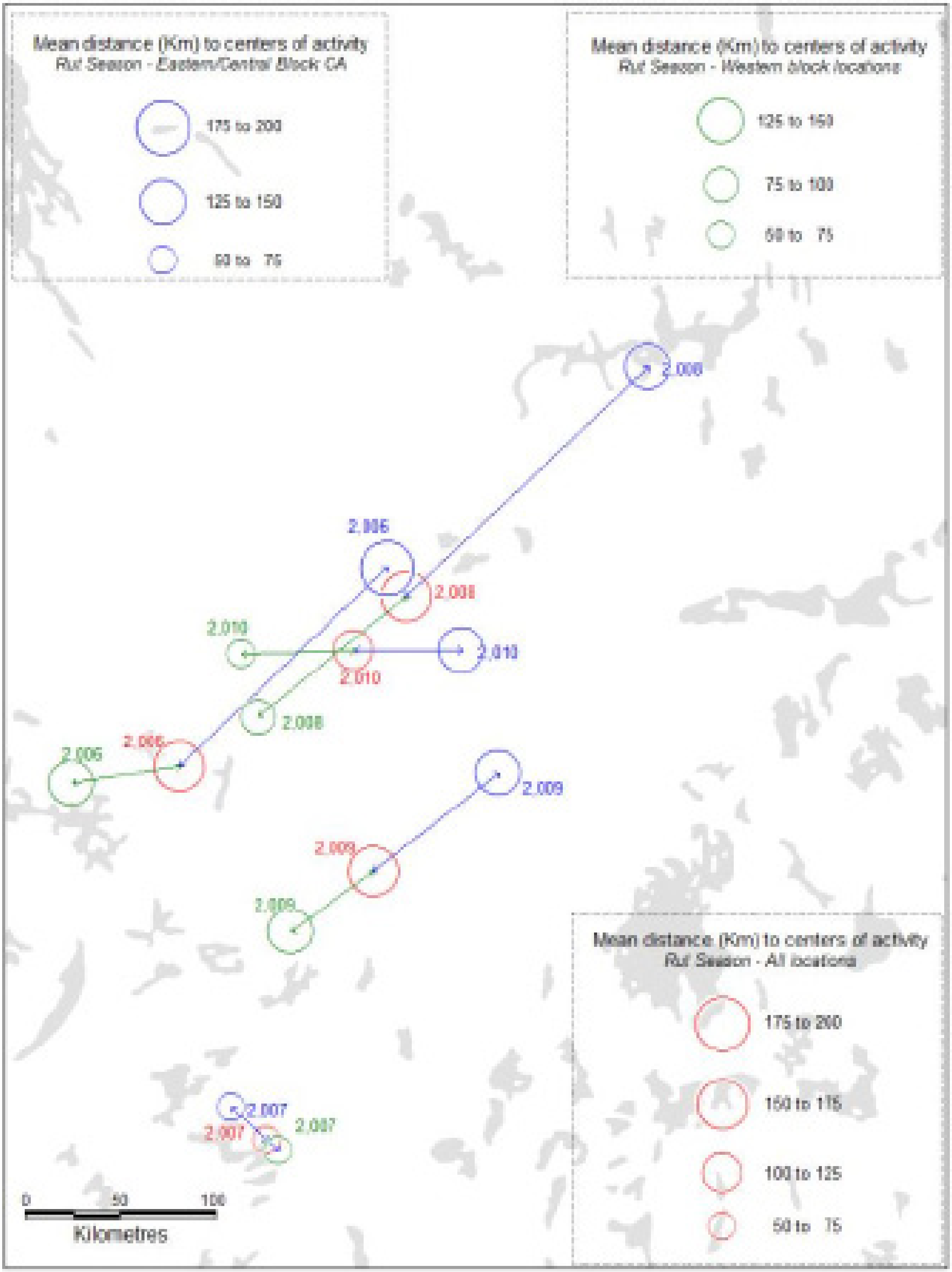

Fig. 6. Location and mean distance between centroids for rutting locations based on western, eastern/central and combined calving associations, 2006-10.

Table 7. Kappa Index of Agreement calculated for Minimum Convex Polygons (MCP) buffered with 1 standard deviation of distances from centres of activity to constituent locations, western block calving areas, eastern and central block calving areas, and all calving areas (<0–poor; 0 to 0.2 –slight; 0.21 to 0.4 – fair; 0.41 to 0.6 – moderate; 0.61 to 0.8 – substantial; 0.81 to 1.0 – almost perfect).

	Kappa Index of Agreement		
Year	MCP of AHW calving as reference	MCP of AHE/AHAD calving as reference	Overall Kappa
2006	0.19	0.32	0.24
2007	0.22	0.97	0.36
2008	0.00	0.00	0.00
2009	0.31	0.74	0.43
2010	0.57	0.14	0.22

Between years, cows from the two calving associations overlapped to varying degrees during the rut (2006-10; Fig. 7, Table 7). In 2007 and 2009, the agreement between the two calving associations and rut overlap was high to moderate suggesting that cows which calved together were together (in closer association) during the rut. However, the agreement was slight in 2008 and fair in 2010 (Table 7).

Calving distribution relative to the tree-line

To describe the pre-calving migration distance travelled across tundra relative to tree-line, we measured the distance from calving locations to the closest point on both the northern and southern edges of forest-tundra biome (Timoney *et al.,* 1992; Payette *et al.,* 2001). The mean distances differed significantly among calving blocks (Table 8; PROC GLM, F > 43.0, df = 2,146, *P* < 0.0001; Duncan's multiple range test, *P* < 0.05). For both the distance to northern and southern edges of the forest-tundra biome (Table 8), caribou calving in the western part of the Ahiak calving ground (AHW) were closer than caribou from the eastern portion (AHE), which was closer than those calving locations on the Adelaide Peninsula (AHAD). Most distances to the northern edge of tree-line were to the tongue of the forest-tundra biome that follows the Thelon River north to Aberdeen Lake (Fig. 1). Caribou that calved on the Adelaide Peninsula would have travelled an average 31% greater distance if they had wintered within the boreal forest compared with caribou that calving in the AHW calving block.

Winter distribution relative to the tree-line

Caribou wintered mostly on the tundra and forest-tundra biome. In total, 57% of February locations were on tundra; 24 % in forest-tundra biome, and 18% of locations were in the boreal forest (south of forest-tundra biome). The percentages changed with the destination of pre-calving migration as 52%, 64% and 84% of the cows calving in AHW, AHE, and AHAD, respectively, wintered on the tundra. For caribou wintering in the boreal forest, 21%; 14% and 0% of the caribou returned to calve in AHW,

Table 8. Distance (km) from calving locations within 3 calving blocks on the Ahiak calving ground, 1997-2010, to the north (N) and south (S) of forest – tundra biome (Timoney *et al,*. 1992, Payette *et al.,* 2001).

		Dist. to N edge of tree-line		Dist. to S edge of tree-line	
Calving block	*n*	$\bar{x}$ km	SE	$\bar{x}$ km	SE
AHW	107	310	2.5	588	3.4
AHE	19	336	3.4	687	5.5
AHAD	23	366	7.9	761	7.0

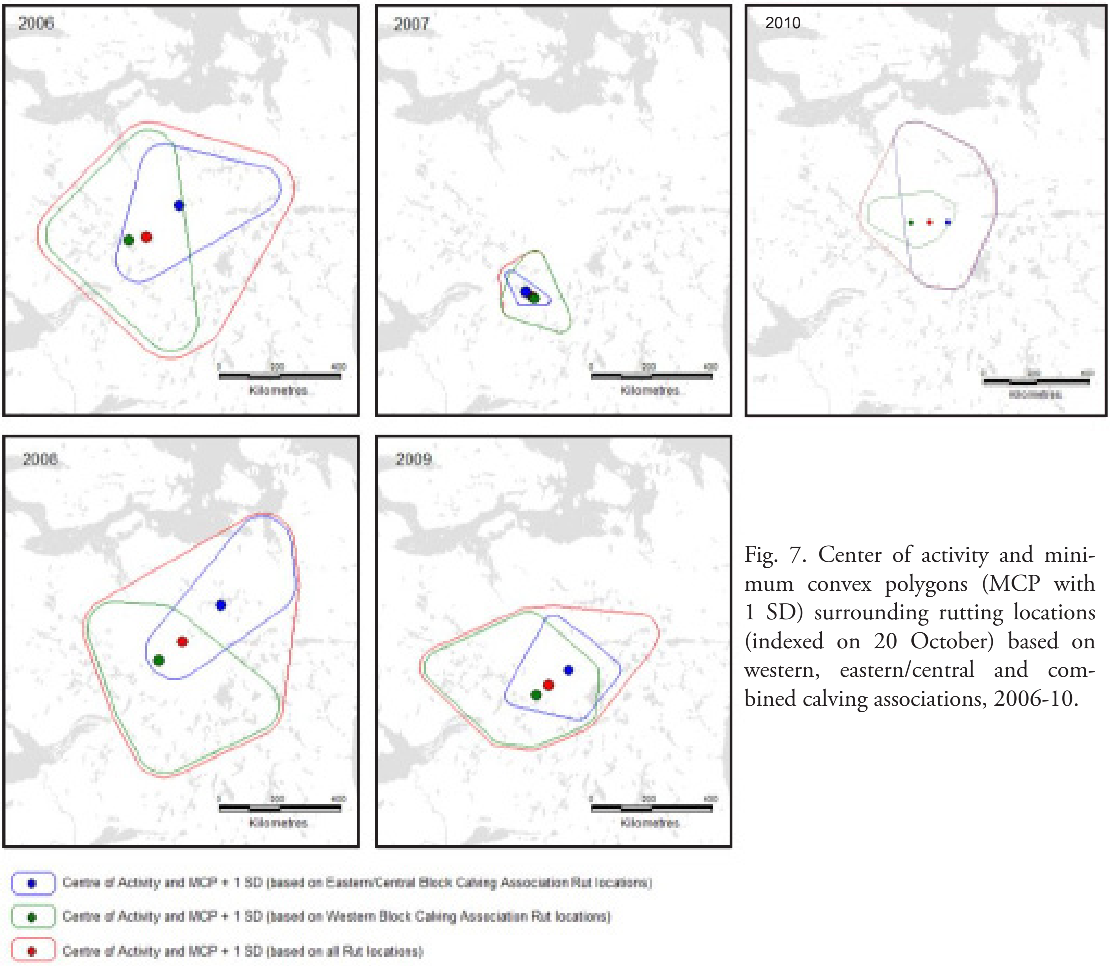

Fig. 7. Center of activity and minimum convex polygons (MCP with 1 SD) surrounding rutting locations (indexed on 20 October) based on western, eastern/central and combined calving associations, 2006-10.

AHE, and AHAD, respectively. The pre-calving migration straight-line distance (distance between winter location and calving) differed among calving blocks (PROC GLM, $F = 3.17$, df = 2.95, $P = 0.047$) and was similar (Duncan's multiple range test, $P < 0.05$) for AHW (525 ± 17.4 km; $n = 75$) and AHE (513 ± 60 km; $n = 14$), but significantly shorter for AHAD (397 ± 32.7 km; $n = 11$) (Table 9). Individual cows did not show high fidelity to wintering within tundra, tree-line, or taiga. Nearly three quarters (73%) of cows with ≥2 years of wintering locations switched among blocks ($n = 33$). Similarly, 68% ($n = 47$) sequential pairs of wintering areas switched between years.

Population-scale: Calving distribution based on 2006-10 calving ground surveys

To test our prediction that the distribution of calving cows would be discontinuous across the calving ground, we used the 2006-10 aerial systematic surveys with transects spaced at 10 or 20 km intervals (D. Johnson, unpubl. data; Poole *et al.*, 2013). We examined for a pattern of discontinuity along the east-west axis in the distribution of breeding females and newborn calves (measured at a grid scale interval of 10 km).The distribution data do not reveal any obvious spatial breaks in the distribution of cows and calves along the east-west axis of the coastal calving area. There were no north-south tran-

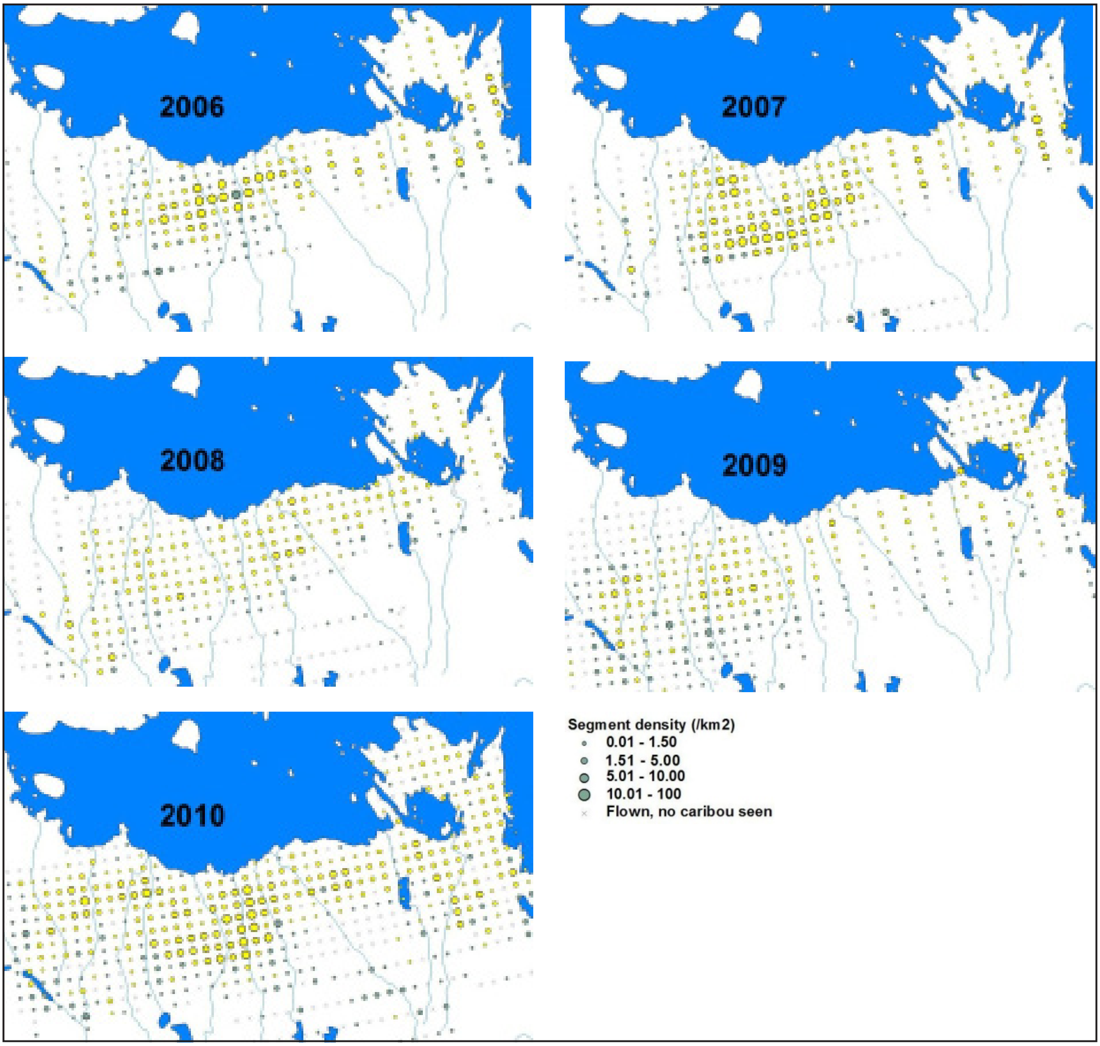

Fig. 8. Density of caribou observed by 10-km segment, Ahiak calving ground distribution surveys, 2006-10. Yellow circles denote density of 1+-year-old caribou where breeding females were present within a segment, regardless of the proportion of non-breeders. Grey circles are density of non-breeders where no breeders present within that segment. Crosses are segments flown but no caribou observed. Scale and graduated symbols constant among years.

sects with no breeding cows and/or calves, or only bulls and juveniles (Fig. 8).

Discussion

Changing abundance and geography (the long relatively straight coastline) influenced the dispersion and distribution of the Ahiak herd's calving. When abundance was low, caribou calved dispersed across the many islands along the Queen Maud Gulf coast. As abundance increased by the mid-1980s through mid-1990s, calving shifted to gregarious calving on a long narrow coastal calving ground. The main elongation of the calving ground was between 1986 and 1995 – a period when densities on the calving ground increased three-fold. The tree-line shifts south as width of the Hudson Bay coastal plain increases, and its orientation relative to

Table 9. Mean (standard errors) for straight-line distances between calving and winter locations for the Ahiak west (AHW), east (AHE) and Adelaide Peninsula (AHAD), 1997-2010.

	AHW			AHE			AHAD		
Year	$\bar{x}$ km	SE	*n*	$\bar{x}$ km	SE	*n*	$\bar{x}$ km	SE	*n*
1997	324	42.1	3						
2001	609		1						
2002	767		1	509	151.7	2			
2003	519	164.5	2				492		1
2004	337		1				294		1
2005	812		1						
2006	514	73.4	2	424	290.9	2	618		1
2007	482	24.8	10	613	113.9	3	414		1
2008	639	29.1	13	550		1			
2009	433	25.7	22	280		1	338	32.0	5
2010	596	31.3	19	529	112.2	5	429	66.2	2

the elongated calving ground increased the likelihood that cows calving on the eastern part had wintered on the tundra. However, individual cows annually varied as to whether they wintered in the boreal forest or tundra.

We did not find support for our four predictions about the use patterns of the calving grounds at the individual scale (based on satellite-collared cows) and at the population-scale (using information from aerial surveys of calving distribution). Thus our hypothesis (the Ahiak calving grounds are comprised of two or more calving sub-populations) was not supported. We had projected that the climate gradients across the elongated calving ground and its location relative to whether cows were more likely to winter on the tundra or boreal forest would affect survival and productivity and lead to sub-population structure.

At the population scale, we found no evidence that the distribution of calving cows or non-breeders was discontinuous in any one year although there were variations in density. For example, Poole et al. (2013) applied an Inverse Distance Weighted (IDW) interpolation to map the spatial pattern of density clumping using 2007 (Fig. 9).

The areas of higher density of breeding cows were connected by areas with lower density of cows and calves which coincides with the area of deeper snow in winter and spring. This pattern of spatial variation in the high densities is similar to, for example the Central Arctic herd, which has two areas of higher calving densities linked by areas of lower density and the cows and calves mingle during the other seasons of the year (Arthur *et al.,* 2009).

At the individual scale of satellite-collared cows, while most collared cows used the western third of the calving ground, almost half the individual cows shifted their annual calving locations within the length of the calving ground. Satellite-collared cows annually varied in the degree to which they associated during the rut and there was no clear-cut pattern of cows calving in one half of the calving ground remaining as a consistent (inter-annual) association during the rut. The cows mainly wintered on

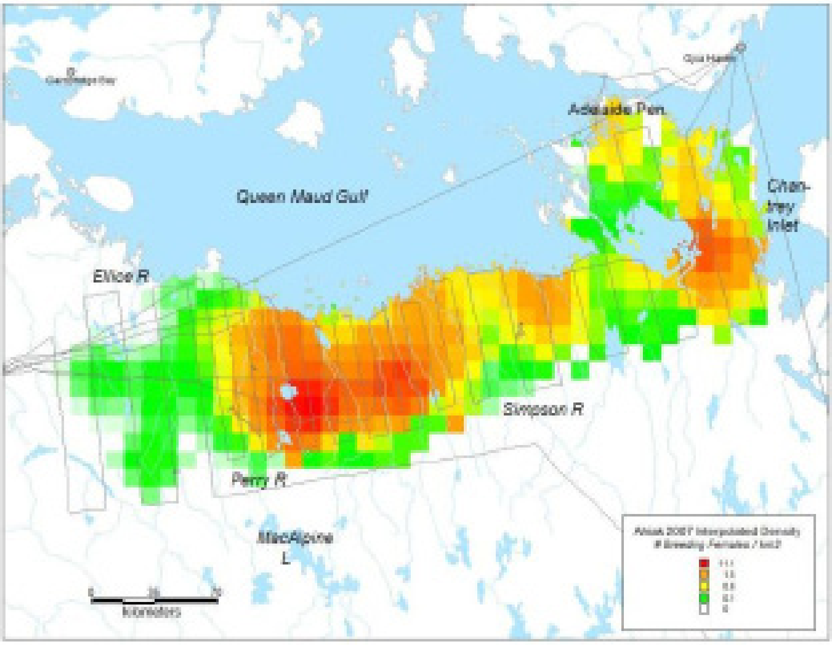

Fig.9. Distribution of breeding female caribou observed during the Ahiak survey flown in June 2007. Relative density conducted using Inverse Distance Weighting (IDW) mapping interpolation using a 10 x 10 km pixel, exponent of 2, and a 15 km search radius (Poole *et al.,* 2013: Fig. 23).

the tundra and within the forest-tundra biome, which partly reflects the geographical extent of the tundra. Individual cows varied among years as to whether they wintered in the forest-tundra biome or above tree-line on the tundra.

We used three methods to estimate calving relative to rut associations. The overlap of the individual rut locations have a stronger pattern of clustering of the calving associations during the rut than the distances between the centers of the rut distribution. This is because MCPs do not consider the dispersion of constituent caribou locations that lie within them (Nilsen *et al.,* 2008), while the mean distances that form the centroids are sensitive to the dispersion of the individual locations. The dispersion of individual locations likely has both process (biological) and measurement error (sample size). Inter-annual comparisons are limited because the membership of the calving associations varied among years (*i.e.*, the scale of calving spatial fidelity was not at the block scale). The overall results suggest that for 2006–10, cows that calve together also rut together. Low sample size and uncertainty about how the collared cows represent the seasonal distribution of all caribou at finer geographical scales are possible limitations of our conclusions.

The level of information available on the Ahiak herd over time varied, with the least amount of information for the earlier years. The frequency of aerial surveys of the coastal calving area to describe the calving distribution and abundance of breeding females was uneven over time. Similarly, the annual sample of collared cows occurred after the time when abundance was increasing (1986-96) and was low during 2001-06, which weakens inferences that can be drawn from the telemetry data. We also acknowledge that we have described the caribou dispersion and distribution over a 25 year period when abundance of the Ahiak and neighbouring herds changed.

Changing abundance in neighbouring herds may play a role in the likelihood of sub-population structure. An extreme decline in the neighbouring Beverly herd (J. Adamczewski, unpubl. data) may have caused a loss of gregarious calving on the traditional Beverly range, and subsequent movement of cows to the Ahiak herd although documentation of this through satellite-collared cows only started in 2007 (Gunn *et al.,* 2012). Alternatively, Nagy et al. (2011: Fig. 3, p. 234) proposed that the Queen Maud Gulf (*i.e.*, Ahiak) herd's long narrow calving ground could be the consequence of adjacent calving by two different sub-populations: the Beverly herd shifting from its traditional calving ground in the mid-1990s to the western part of Queen Maud Gulf and the caribou calving on the eastern Queen Maud Gulf coast in the 1980s. However, we could not infer sub-population structure for the coastal calving area based on the currently available evidence.

Another possibility for both a historical influence on the Ahiak calving ground and future sub-population structure is the caribou calving on Adelaide Peninsula. Until about the 1920s and 1930s, caribou migrated across Adelaide Peninsula in May and crossed to King William Island where they calved and summered before returning to Adelaide Peninsula in the fall (summarized in Appendix G of Gunn *et al.,* 2000). In June 1976, a few cow-calf pairs were counted south of Adelaide Peninsula (although the flight-lines did not extend further west; Fischer *et al.,* 1977). It is speculative whether the 1976 calving distribution shifted northwest (to the area mapped as calving in 1986), northeast of Chantrey Inlet, or disappeared. The 1976 calving area was surveyed in 1986 and no caribou were seen (Gunn & Fournier, 2000a). The possibility that calving shifted northeast of Chantrey Inlet is suggested from observations of calving caribou recorded in 1975, 1985, 1986, 1989 and 1991 (Gunn & Fournier, 2000a) and from seasonal movements of satellite collared cows in 1991-93 (Gunn & Fournier, 2000b). The four satellite-collared cows in 1991-93 showed that caribou wintering south of the Boothia Peninsula toward Baker Lake would calve northeast of Chantrey Inlet (and not Adelaide Peninsula) (Gunn & Fournier, 2000b).

The caribou which calved on Adelaide Peninsula mostly wintered on the tundra and reduced the length of their pre-calving migration by just over 100 km relative to the caribou which calved west of Adelaide Peninsula; caribou calving west of Adelaide Peninsula were more likely to winter within the closer forest and forest-tundra biome. We suggest that the shorter pre-calving migration distance for caribou that calved on Adelaide Peninsula may be from an energetic trade-off between the costs of foraging on the tundra, travel, and predation risk (Couturier *et al.,* 2010).

Currently, the high proportion of cows shifting calving locations along the length of the calving ground among years and the switching of cows wintering on the tundra or taiga/forest-tundra biome suggests a high degree of plasticity – the ability to change biology or behavior to respond to changes in the environment – in individual behaviour within a population (for example, Couturier *et al.,* 2010). However, variance in inter-annual selection of calving and wintering areas by individual adult females may be reduced if changing environmental conditions increase energetic costs and reduce fitness of cows and calves. While we could not infer population sub-structure for the Ahiak herd from the currently available evidence, it is not a future impossibility given the Ahiak herd's particular geographical attributes – the long narrow calving ground and the southeast tilt and width of the tree-line.

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Status of northern mountain caribou (*Rangifer tarandus caribou)* in Yukon, Canada

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Abstract: Caribou (*Rangifer tarandus)* are an important ecological, cultural and economic resource in Yukon, Canada. Three caribou ecotypes occur within Yukon: Grant's *(R. t. granti)*, northern mountain *(R. t. caribou)*, and boreal *(R. t. caribou).* Northern mountain caribou are classified as a species of special concern under Canada's *Species at Risk Act,* and a national management plan for northern mountain caribou was recently completed. Twenty-six northern mountain caribou herds occur at least partially within Yukon, representing approximately 30,000 – 35,000 animals. Active monitoring of Yukon's northern mountain caribou began in earnest in the early 1980s. To date, over 200 fall composition surveys have been carried out, over 1000 animals have been fitted with radio-collars, and nearly 40 formal population estimates have been completed. Disease and contaminant monitoring of these caribou has indicated relatively low disease prevalence and contaminant loading. Northern mountain caribou are harvested in Yukon, with an average of 230 caribou harvested per year by licensed hunters (1995 – 2012) and an unknown number by First Nation hunters. Future challenges related to caribou management and conservation in Yukon include increasing levels of industrial development primarily through mineral exploration and development, ensuring harvest of these herds is conducted sustainably given the absence of total harvest information, inter-jurisdictional management of shared herds, existing uncertainty surrounding herd distribution and delineation, and dealing with vehicle-related mortality of caribou for certain herds. Overall, the population status (*i.e.*, trend) of eight herds is known, with two increasing, two decreasing, and four stable.

Key words: management; monitoring; northern mountain caribou; *Rangifer tarandus caribou;* status; Yukon.

Rangifer, 33, Special Issue No. 21, 2013: 59–70

Background

Caribou (*Rangifer tarandus)* are an iconic species across Canada (Hummel & Ray, 2008). They are an important cultural, ecological, and economic resource in Yukon, Canada, and have been used by First Nations for thousands of years (Hare *et al.,* 2004). Two caribou subspecies occur in Yukon: Grant's *(R. t. granti)* and woodland *(R. t. caribou).* Within the woodland subspecies, two ecotypes are present: northern mountain and boreal. Twenty-six northern mountain caribou herds occur at least partially in Yukon in the southern 2/3 of the territory, roughly south of 66°N. (Fig. 1). Boreal caribou occur in a small, remote area of northeast Yukon (Fig. 1; Environment Canada, 2008; Nagy, 2011); while the large migratory Grant's caribou herds (Porcupine and Fortymile) occur in the northern and west-central portions of the territory.

Northern mountain caribou differ from both the more sedentary boreal ecotype and the large migratory barren-ground (*i.e.*, Grant's caribou) herds. Northern mountain caribou generally migrate elevationally between winter and summer ranges, but may also migrate longer distances between these ranges (*e.g.*, Weaver, 2006). Herds may winter on windswept alpine slopes or in lower elevation forested areas (Kuzyk *et al.,* 1999; Florkiewicz *et al.,* 2007) where they forage on terrestrial lichens. At calving, the peak of which occurs roughly around 20 May (*e.g.*, Chisana Caribou Recovery Team, 2010), parturient females disperse to higher elevations (Barten *et al.,* 2001) away from conspecifics and other prey species such as moose *(Alces alces;* Bergerud *et al.,* 1984). This is in sharp contrast to the more well-defined calving grounds associated with barren-ground herds. Following calving, animals aggregate into small groups in alpine areas, often occurring on snow patches for thermoregulation and insect avoidance (Ion & Kershaw, 1989). The summer, or post-calving, season lasts until roughly late-September at which time males and females begin to aggregate on alpine plateaus during breeding (*i.e.*, the rut) which lasts until approximately the middle of October at which time breeding groups break up and animals prepare to move to their winter ranges.

Caribou management is becoming increasingly challenging (Festa-Bianchet *et al.,* 2011) due to, among other factors, increasing land-use pressures, an increasing human population in Yukon (primarily in Whitehorse), and the uncertainty of the effects of climatic change (*e.g.*, changing forest fire regime, effects on predator and alternative prey species, and changes in parasite prevalence). In Canada, northern mountain caribou are federally designated as a species of special concern (COSEWIC, 2002) under the federal *Species at Risk Act.* A status reassessment of this ecotype is planned to begin in 2012 (J.C. Ray, COSEWIC, pers. comm.). As mandated following their listing as a species of special concern, a national management plan for northern mountain caribou was recently completed (Environment Canada, 2012).

The purpose of this report is to update the status of northern mountain caribou in Yukon including the best available information on herd sizes and trends, levels of monitoring, and conservation and management issues related to these herds. The last status assessment of Yukon mountain caribou is over 10 years old (Farnell *et al.,* 1998) and an update is warranted.

Population monitoring

A number of tools are used to monitor Yukon's northern mountain caribou herds, with monitoring efforts beginning in earnest in the early 1980s (Farnell *et al.,* 1998). The herd (*i.e.*, population) is the basic management unit for northern mountain caribou and radio-collar programs have been used extensively to track the distribution of individuals and subsequently map herd range boundaries (Fig. 1). Typically adult females have been collared and to date over 1000 animals have been fitted with radio-collars (Table 1). There were two peaks in collaring activity (Fig. 2); one in the mid-1990s associated with the Aishihik caribou recovery program (Hayes *et al.,* 2003) and the second in the mid-2000s associated with the Chisana caribou captive-rearing program (Chisana Caribou Recovery Team, 2010). Most of these collars were very high frequency (VHF) collars, but more recently both global positioning system (GPS; Klaza, Carcross, and Laberge herds) and satellite (Argos) collars (South Nahanni, Coal River, and La Biche herds) have been deployed. These collaring efforts have resulted in over 16,000 VHF relocations, and thousands more GPS and Argos relocations.

While many animals have been fitted with radio-collars and tracked, there remains uncertainty regarding herd "definition" in some areas. For example, data from four GPS-collared caribou in the Laberge herd in 2011, and the existing GPS radiocollar dataset for the Car-

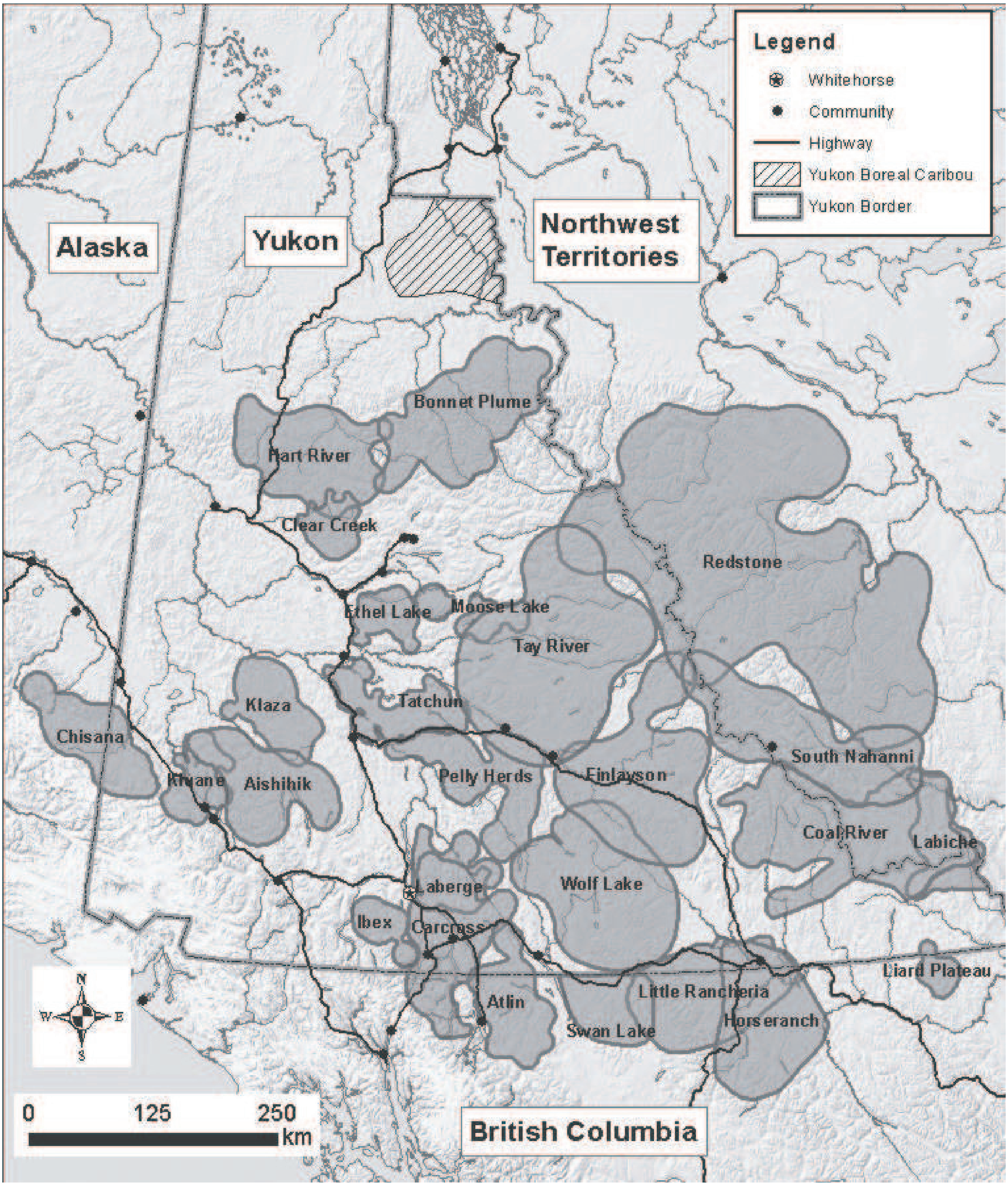

Fig. 1. Distribution of woodland caribou in Yukon, Canada. Northern mountain herds are individually labelled. The hatched area in the northeast Yukon represents the general range of boreal caribou in Yukon.

cross herd, will be used to assess whether these two herds are distinct. Distinguishing the Klaza and Aishihik herds is one objective of a recently initiated inventory study. GPS radio-collar relocations from animals collared in the Swan Lake area indicate their occurrence in Yukon just north of the Yukon-BC border (M. Williams, BC Ministry of Forests, Lands and

Table 1. Monitoring summary of northern mountain caribou herds occurring in the Yukon, Canada.

Herd	Size (Assessment Year)	Survey Method[b]	Current Known Trend	Number of Formal Population Estimates	Number of Collared Animals[c]	Number of Fall Composition Surveys[c]
Aishihik	2050 (2009)	MR	Increasing	2	91	21
Atlin[a]	800 (2007)	SPQ	Stable	2	11	1
Bonnet Plume	5000 (1982)	EO	Unknown	0	25	0
Carcross	800 (2008)	SRQ	Stable	3	72	18
Chisana	680 (2010)	MR	Stable	4	332	23
Clear Creek	900 (2001)	SRQ	Unknown	1	22	7
Coal River	450 (1997)	EO	Unknown	0	17	24
Ethel Lake	300 (1993)	SRQ	Unknown	1	12	17
Finlayson	3100 (2007)	SRQ	Decreasing	5	55	29
Hart River	2200 (2006)	MR	Unknown	1	79	2
Horseranch[a]	600 (1999)	SRQ	Unknown	1	-	-
Ibex	850 (2008)	SRQ	Increasing	3	23	25
Klaza	1180 (2012)	MR	Unknown	2	75	12
Kluane	180 (2009)	MR	Descreasing	2	36	21
La Biche	450 (1997)	EO	Unknown	0	4	1
Laberge	200 (2003)	SRQ	Unknown	0	29	4
Liard Plateau[a]	150 (2011)	MC	Unknown	0	3	1
Little Rancheria	1000 (1999)	EO	Unknown	2	11	6
Moose Lake	300 (1991)	SRQ	Unknown	1	4	1
Pelly Herds	1000 (2002)	EO	Unknown	0	29	4
Redstone[a]	10000 (2012)	EO	Unknown	0	-	-
South Nahanni	2100 (2009)	MR	Stable	2	86	8
Swan Lake[a]	400 (2005)	MC	Unknown	0	1	-
Tatchun	500 (2000)	MC	Unknown	0	24	17
Tay River	3750 (1996)	SRQ	Unknown	1	26	1
Wolf Lake	1400 (1998)	SRQ	Unknown	3	73	9

[a] Herds not typically monitored by Environment Yukon.
[b] MR – mark-resight, SRQ – stratified random quadrat, MC – minimum count, EO – expert opinion.
[c] Collaring/surveys by, or in collaboration with, Environment Yukon.

Natural Resources Operations, unpubl. data), and ambiguity exists around the discreteness of three herds in this border region: Little Rancheria, Horseranch, and Swan Lake. Whether or not there are two or three distinct herds in the Pelly "herds" remains a question for managers. Finally, spatial data from radio-collared caribou and genetic information (Zittlau, 2004) in the southeast portion of Yukon and into the Northwest Territories (NWT; Finlayson, South Nahanni, Coal River, La Biche, and Redstone) has led to questions surrounding herd designations there. Future analysis of these data will be conducted to address this question (*e.g.*, Roffler *et al.,* 2012).

A second tool used to monitor these herds has

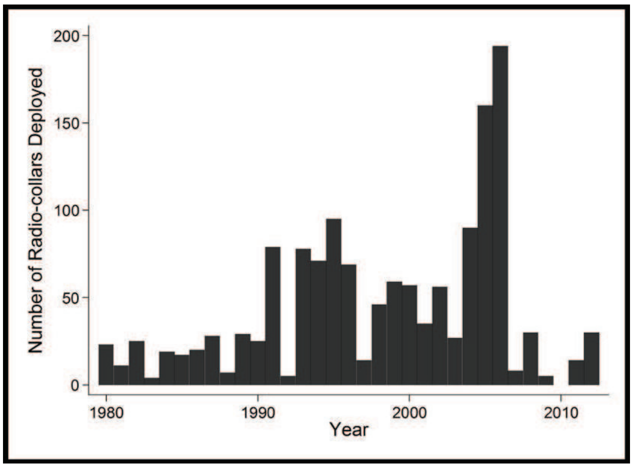

Fig. 2. Number of radio-collars deployed on Yukon northern mountain caribou from 1980 – 2012.

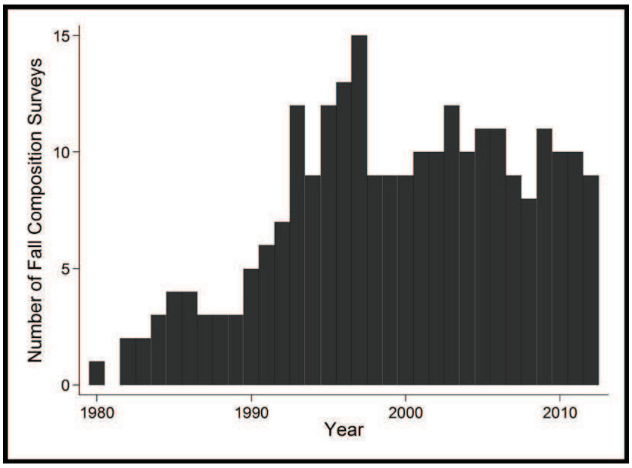

Fig. 3. Number of fall composition surveys conducted on Yukon northern mountain caribou from 1980 – 2012.

been fall composition surveys (*i.e.*, rut counts). These surveys are intended to sample a portion of the herd during breeding when males and females are aggregated on high alpine plateaus. They are not intended to estimate herd abundance. During these surveys, habitats deemed to have a high probability of breeding caribou are flown via helicopter. Once observed, animal groups are counted and subsequently classified into five categories: calves, adult females (*i.e.*, non-calf), immature bulls, mature bulls, and unclassified. The ratio of calves to cows is used as an index of recruitment into the herd, and the ratio of total bulls to cows used as an estimate of its adult sex ratio. From 1980 – 2012, 252 separate fall composition surveys have been conducted on Yukon's northern mountain caribou herds (Table 1; Fig. 3). Adult sex ratios of Yukon herds roughly average 45 bulls per 100 cows (Environment Yukon, unpubl. data) and generally there is little concern that there are too few bulls to limit breeding potential of oestrous cows.

Recruitment rates are much more variable (Hegel *et al.,* 2010) and have ranged from < 10 to > 50 calves per 100 cows. This is typical of the high variability observed in juvenile survival of most ungulates (Gaillard *et al.,* 2000). Due to this variability, generating inferences regarding a herd's dynamics from only one recruitment rate is problematic and multiple years of data should be considered. While some overwinter mortality of caribou calves inevitably occurs, research from mountain caribou in other areas with similar systems has documented that the majority of mortality occurs within the first few weeks of life (Adams *et al.,* 1995; Gustine *et al.,* 2006). Fall and subsequent late-winter (*i.e.*, March/April) recruitment rates are available for the Aishihik herd from five years. Four of these years indicated a decrease in recruitment from fall to late-winter. One year indicated a slight increase which is likely an artefact of sampling error and thus we assumed no decrease in recruitment in that year. The average absolute decrease in recruitment for these five pairs of fall and subsequent late-winter surveys was 4.3 calves per 100 cows (range: 0 – 8.7; Environment Yukon, unpubl. data).

A third key monitoring tool is the estimation of herd abundance. Due to the increased cost associated with estimating abundance compared to composition, abundance estimates have been carried out much less frequently.

Table 2. Disease prevlence (1988-1997) in Yukon northern mountain caribou (from Farnell et al., 1999).

Disease	Prevalence (%)	Sample Size
Brucellosis	0	408
Infectious Bovine Rhinotracheitis	0.9	440
Bovine Viral Diarrhea	0	435
Parainfluenza 3	0	434
Respiratory Syncytial Virus	0	402
Bluetongue	0	272
Epizootic Hemorrhagic Disease	1.2	416
Leptospirosis	0.8	253

Abundance of herds that were the focus of specific management actions have been estimated as well as those herds with specific conservation concerns (*e.g.*, small size, increasing industrial development). The first abundance estimates in Yukon were minimum counts, but given the issues with this approach (*e.g.*, Caughley & Goddard, 1972) and their lack of sightability estimates, a stratified random quadrat approach (Farnell & Gauthier, 1988) was used during most of the 1980s and 1990s. More recently, mark-resight approaches, using either temporary dyes or radio-collars as marks, have been used to estimate herd abundance (Hegel *et al.,* 2012). In cases where only limited information is available, minimum counts from a composition survey, for example, or expert opinion have been cautiously used to provide a crude indication of the herd's size (Table 1), while acknowledging the limitations of these approaches. However, in determining the trend for a given herd (Table 1), only formal population estimates are used (*i.e.*, those accounting for sightability and having an associated measure of precision). We also avoid making assessments of current trends for herds with abundance estimates that are deemed too old (*i.e.*, > 10 years). Thirty-seven formal population estimates have been conducted on 18 separate herds (Table 1).

Animal health

From the 1980s to the present, diseases, parasites, and contaminants have been assessed in Yukon's northern mountain caribou herds. Animal health issues are important both for the potential impact on population dynamics (*e.g.*, Albon *et al.,* 2002) and because caribou are an important food resource for Yukoners. Serological surveys of 11 herds conducted from 1988 to 1997 indicated a low prevalence of infectious diseases (Farnell *et al.,* 1999; Table 2). Kutz (2002) reported relatively low parasitic prevalence and intensity in a preliminary survey of three herds (Finlayson, Little Rancheria, and South Nahanni). Hoar et al. (2009) reported near 100% prevalence of Trichostrongylidae species in the Chisana herd, but with low levels of intensity.

Contaminants have been monitored by the Northern Contaminants Program (NCP; *e.g.*, Braune *et al.,* 1999). Generally, contaminant levels in Yukon northern mountain caribou are low and within safe levels for human consumption. Cadmium levels in the Tay River and Finlayson herds are elevated relative to other herds; however this is likely a result of greater background cadmium levels occurring naturally in the herd's range (Braune *et al.,* 1999; Gamberg *et al.,* 2005). Due to these low levels, the NCP has ceased their broad-scale survey of contaminants in northern mountain caribou in Yukon. The NCP will assess contaminant levels in northern mountain herds when specifically requested to do so by a community.

Overall, Yukon's northern mountain caribou are considered healthy; however, continued monitoring is warranted in light of potential changes in future environmental conditions. For example, with a warming temperature trend, host-parasite dynamics in northern latitudes may change, with warmer temperatures potentially resulting in increased prevalence and/or intensity of parasitic infections (Kutz *et al.,* 2005).

Harvest

In Yukon, licensed hunting of northern mountain caribou is limited to bulls, with the season occurring from 1 August to 31 October. All licensed hunters in Yukon are required to report their kill to an Environment Yukon office; a requirement which began in 1994. Licensed harvest of caribou is managed by the Yukon government under regulations outlined in the Yukon Wildlife Act and described in the Yukon hunting regulations summary (*e.g.*, Environment Yukon, 2012). For regulatory purposes hunters are classified into two categories: licensed and First Nation (*i.e.*, aboriginal). Licensed hunters may be either residents or non-residents of Yukon. All non-residents must be guided when hunting in Yukon. Non-Canadian non-residents (*i.e.*, alien) must be guided by a registered Yukon outfitter. Non-residents who are Canadian citizens must be guided by either a registered Yukon outfitter or by a Yukon resident under a special guiding license.

Subsistence harvest rights of members of individual First Nations are constitutionally entrenched and are not subject to Yukon hunting regulations when hunting within their individual traditional territory or in areas of overlap between the traditional territories of > 2 First Nations. First Nation members hunting within the traditional territory of another First Nation with a signed land claim agreement are subject to Yukon harvest regulations and are thus considered licensed hunters in this case. As First Nation harvest is not regulated by the Yukon government, formal statistics (*e.g.*, harvest rates, sex ratio of harvested animals) describing subsistence harvest of mountain caribou are not available for all herds.

Licensed harvest of most northern mountain caribou herds in Yukon is open in the sense that it is not under a limited-entry or lottery system; however, for a few herds harvest is either closed or managed under a permit hunt authorization (PHA). A PHA is a lottery-based system in which a pre-determined number of permits are awarded to drawn licensed hunters. PHAs are authorized under the *Wildlife Act* and are initiated where a conservation or management concern has been identified. PHAs require a regulation change under the *Wildlife Act* and thus go through a formal public review process with the Yukon Fish and Wildlife Management Board. Currently (*i.e.*, as of 2012) the Finlayson, Klaza, and Aishihik herds are harvested under a PHA. Due to its small size (Table 1), the Kluane herd is closed to all licensed hunting, and a voluntary harvest closure, for all hunters, is requested for the Ethel Lake herd by the Yukon government and the First Nations in this area. As part of their recovery program, the Southern Lakes herds (Atlin, Carcross, Laberge, and Ibex; Farnell *et al.,* 1998) are closed to all licensed harvest and the First Nations in these areas have also implemented a voluntary harvest closure. The Chisana herd will be harvested under a PHA beginning in 2013.

From 1995 to 2012 (n = 18), annual licensed harvest (*i.e.*, non-First Nation) of northern mountain caribou in Yukon averaged 230.4 animals/year (SE = 7.6, range: 196 – 306). The average annual resident and non-resident harvest during this time was 108.2 (SE = 4.6, range: 83 – 151) and 122.3 animals/year (SE = 3.7, range: 100 – 155), respectively. There was a negative trend in the number of northern mountain caribou harvested by licensed hunters from 1995 – 2012 (Fig. 4), with the decline being greater in resident hunters over non-residents. This may be due to the increasing urban population of Yukon, but it also generally follows hunter participation trends in other jurisdictions (*e.g.*, Boxall *et al.,* 2001). One coarse metric of hunter participation rates is the number of caribou seals sold prior to the hunting season. All licensed hunters require a seal which must be immediately attached to a harvested animal. From 1995 – 2011, there was an increase in the number of seals sold to non-residents (β = 7.7, SE = 2.0) and a decrease in the number of seals sold to licensed resident hunters (β = -15.9, SE = 5.1; Environment Yukon, unpubl. data). Caribou seals are not differentiated between northern mountain caribou and barren-ground caribou (*i.e.*, Porcupine herd) which makes drawing inferences from their sales, with respect to northern mountain caribou, challenging. Seal sales do not necessarily have a strong relationship with success rates or the number of animals harvested, but the decline in the number of seals sold to Yukon residents may be an indication of a decreasing level of interest in harvesting caribou.

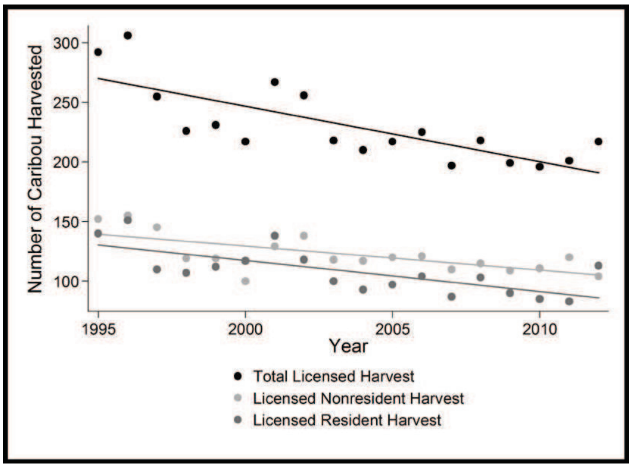

Fig. 4. Licensed harvest of northern mountain caribou (1995 – 2012) in Yukon, Canada. Trend lines indicate decreasing total licensed harvest (β = -4.65, SE = 0.96) for both non-resident (β = -2.02, SE = 0.53) and resident (β = -2.60, SE = 0.63) hunters.

Land-use

Human land-use within caribou ranges is a management concern for a number of herds in Yukon. For example, a large portion of the Carcross herd's winter range is occupied by the footprint of the City of Whitehorse and surrounding rural residential subdivisions which have reduced the effectiveness of this winter habitat (Florkiewicz *et al.,* 2007). Summer and winter habitat effectiveness has also been reduced for the Atlin herd from human activities on the landscape (Polfus *et al.,* 2011). While the direct habitat lost through human activities may be small in some cases, the indirect losses due to caribou avoidance may be greater (Weir *et al.,* 2007; Polfus *et al.,* 2011). Additionally, increased development and activities in caribou range often results in increased access which may result in caribou being more vulnerable to harvest pressure.

The recent rise in metals prices has preceded a substantial increase in mineral exploration activity in Yukon and a number of new operational mines are proposed over the next few years. This increase in mineral exploration and development will undoubtedly influence future research and monitoring. For instance, significant advanced exploration activity is occurring in the Klaza herd's range (Yukon Geological Survey, 2011) where a large inventory study was recently initiated, and a four-year inventory program of the South Nahanni herd, now finished, was initiated in part due to exploration activity and a potential mine and access road along the Yukon-NWT border. An intensive study of the Liard Plateau herd was initiated in 2010 due to proposed development within its range (S. McNay, Wildlife Infometrics, pers. comm.). A number of exploration activities are also occurring in the ranges of the Tay River, Finlayson, Clear Creek, and Hart River herds. The increase in mineral exploration and development brings a number of management issues including direct and indirect habitat loss, increased access potentially increasing harvest pressure, and the cumulative effects of natural (*e.g.*, fire) and anthropogenic effects in caribou ranges.

Highway mortality

A number of herd ranges are bisected by major Yukon highways which pose a mortality risk for animals crossing roads, or aggregating on them to take advantage of road salts used during winter highway maintenance. Two herds in particular, Carcross and Little Rancheria (Fig. 1), experience the most vehicle related mortality. On average (2001 – 2010) 15.4 (SE = 2.2) northern mountain caribou are killed on Yukon highways each year. These data represent minimum numbers of mortalities as they only include carcasses documented by Environment Yukon and do not take into account those animals which may have been injured through a collision and which subsequently succumbed to their injuries away from the road.

While the number of animals killed on highways may appear small, this likely represents a source of additive mortality. In particular for the Little Rancheria herd which is harvested in both Yukon and British Columbia, the addition of road-kills could have the potential to impact herd growth. This may be especially influential if females are killed. Unfortunately, data regarding the sex ratio composition of road-killed animals are unavailable.

The use of caribou deterrents, such as lithium chloride (Brown *et al.,* 2000), and additional road signage is currently being explored. Given the increase in the number of operating mines and mineral exploration the volume of heavy truck traffic carrying ore and other equipment is expected to increase on Yukon highways which may further increase caribou-vehicle collisions. With this increase in traffic volumes, road mortalities could become an even greater conservation concern in the future.

Summary

Of the 26 herds occurring in Yukon, population trend is known for eight (~ 31%; Table 1). Of these eight herds, two are increasing, two are decreasing, and four are stable. This variability in trend somewhat contrasts the general pattern of decline in *Rangifer* populations described by Vors & Boyce (2009); however, recent estimates of large barren-ground herds (*e.g.*, Porcupine herd) also indicate some are recovering from low levels. Additionally, while trend is known for a number of herds, it is unknown for nearly 70% of Yukon's northern mountain herds.

Radio-collar studies of caribou are ongoing, albeit at reduced levels than observed historically (Fig. 2). Recent radio-collaring efforts have largely been in response to specific management concerns and information needs. Fall composition surveys are also ongoing. Eight herds (Aishihik, Carcross, Chisana, Ethel Lake, Finlayson, Ibex, Kluane, and Tatchun; Fig. 1) have been identified for annual monitoring, when feasible, with the aim that results, particularly with respect to recruitment, provide a general indication (*i.e.*, above or below average) of the condition across all herds. Maintaining long-term time series of these data also provides the basis for analyses into the drivers of these demographic patterns (*e.g.*, Hegel *et al.,* 2010). Lack of information on trend, herd size, total harvest levels, and other vital rates will increase the challenges associated with management of Yukon's northern mountain caribou herds, particularly with the increasing land-use pressures facing them and the uncertainty of future climatic conditions. A number of herds also cross jurisdictional boundaries. Coordinated management and monitoring of these herds will likely be required into the future. For example, a multi-jurisdictional management plan (Chisana Caribou Herd Working Group, 2012) for the Chisana herd was recently formally approved, the signature page of which includes six parties representing multiple countries, agencies, and First Nations. A tri-agency research program on the South Nahanni herd is now complete. Such multi-agency partnerships increase the ability to carry out expensive research and monitoring programs.

Acknowledgements

Research and monitoring of Yukon's northern mountain caribou are a result of the many biologists and technicians who have carried out this work over the past three decades. In particular, Rick Farnell who managed the Caribou program for the Yukon Government during much of this time. We also acknowledge our partners in the various First Nations and Renewable Resources Councils who collaboratively participate in Yukon caribou management. J. Carey, R. Florkiewicz, and two anonymous reviewers provided valuable comments on an earlier draft of this manuscript. J. Adamczewski accepted editorial responsibilities for this article.

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Zittlau, K.A. 2004. *Population Genetic Analyses of North American Caribou (Rangifer tarandus).* Ph.D. dissertation. University of Alberta, Edmonton. 187pp.Table 1. Monitoring summary of northern mountain caribou herds occurring in the Yukon, Canada.

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Arctic Borderlands Ecological Knowledge Cooperative: can local knowledge inform caribou management?

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Abstract: While quantitative analyses have traditionally been used to measure overall caribou herd health, qualitative observational data can also provide timely information that reflects what people on the land are observing. The Arctic Borderlands Ecological Knowledge Co-op (ABEKC) monitors ecological change in the range of the Porcupine Caribou Herd (PCH). The community-based monitoring component of the Co-op's mandate involves the gathering of local knowledge through interviews with local experts in a number of communities.

We analyzed the responses to interviews collected during 2000–2007 related to caribou availability, harvest success, meeting needs and caribou health during fall and spring. Interviews revealed 1) caribou greater availability during the survey period, 2) an increasing trend in the proportion of harvesters that met their needs 3) no trend in animals harvested or proportion of successful hunters and 4) improving overall caribou health throughout the period.

There was no population estimate for the herd between 2001 and 2010. In 2001, 123,000 caribou were estimated in the herd. Based on an estimated 178,000 in 1989, a declining trend of ~ 3% annually occurred at least until 2001. In the interim agencies and boards feared the herd continued to decline and worked towards and finalized a Harvest Management Plan for the herd. In contrast, from the Co-op interviews all indications suggested improving herd conditions throughout most of the decade. A successful survey in 2010 determined the herd had grown to 169,000 animals. We conclude that the community-based interviews provided a valid, unique information source to better understand caribou ecology and express community perceptions of overall herd status and could provide a valuable contribution to management decision making. We recommend that ABEKC results become standard input into Porcupine Caribou harvest management decisions and serve as a model of integrating community based monitoring data into resource management decision making throughout the north.

Key words: caribou; community interviews; harvest; local knowledge.

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Introduction

Monitoring the status of migratory caribou populations remain challenging given that current methods to estimate population size often fall victim to inclement weather and caribou distributions that are difficult or impossible to survey because of mountainous terrain and/or fragmentary groups. The Porcupine Caribou Herd (PCH) (*Rangifer* tarandus granti) increased in the early 1970s from 100,000 to over 178,000 by 1989 (PCMB, 2012). Five population estimates after 1989 indicated the herd declined by ~3% per year to 123,000 by 2001http://www.taiga.net/pcmb/population.html). Between 2001 and 2010 there were no successful population estimates of the PCH. Fearing continued decline of the herd, governments and the Porcupine Caribou Management

Board (PCMB) worked on a Harvest Management Strategy (HMS). The Strategy was ratified in early 2010 and management actions revolved around imposing restrictions or relaxing restrictions based on the current population levels in the herd (PCMB, 2010). A major action proposed in the HMS is the monitoring of a number of indicators on an annual basis to help guide annual management actions and to assess the status of the population in the absence of a population estimate (PCMB, 2011). The HMS was developed in the absence of a recent population estimate of the herd, while assuming a continuing population decline due to presumed high harvest and high adult cow mortality. Using these conservative assumptions, ecological models indicated the herd was likely still declining and estimates of around 100,000 were suggested. A successful estimate was obtained in late 2010 which indicated the herd increased since the last census and numbered 169,000 animals.

In the last few decades the involvement of local communities in better understanding caribou status, ecology, and management needs has received considerable attention (*i.e.*, Ferguson & Messier, 1997; Berkes *et al.,* 2000; Kofinas *et al.,* 2000; Kofinas, 2002; Kendrick & Manseau, 2008). Aboriginal involvement in Canada has been driven largely by the legal need to consider traditional knowledge as stipulated in current land claim settlements, the desire by co-management boards to make decisions that reflect everyone at the table, and uncertainty that conventional monitoring systems accurately reflect current caribou status (Kofinas, 2002; Lyver & Dene First Nation, 2010). Moreover, efforts to integrate both western science and community knowledge has often resulted in a better understanding of current environmental conditions than would have been possible if either source of knowledge were considered in isolation (Berkes, 1999; Kofinas, 2002).

Initiated in the mid 1990s, the Arctic Borderlands Ecological Knowledge Cooperative (ABEKC) was designed to monitor changes, from a community perspective, within the range of the PCH (Eamer, 2006). The annual interviews in a number of communities were one avenue to inform the ABEKC of what people on the land were observing. Although the focus was the range of the PCH, interview questions also asked about observations regarding other land resources and weather. Kofinas (2002) described the experience with ABEKC's ongoing ecological monitoring program. He concluded that the monitoring program provided a richly detailed holistic account of environmental conditions that extended beyond the single community to represent a regional picture. Further, the results of the community-based monitoring can serve to fill the knowledge gap left by the limitations of western monitoring program methodologies, particularly with respect to the status of the PCH (Kofinas, 2002).

In the absence of a PCH population estimate from 2001–2011, we analyzed the responses to a number of caribou-related questions from 2000-2001 onward to determine if the results of the community interviews could have helped inform managers and the PCMB regarding the status of the herd from the perspective of availability, meeting subsistence needs, harvest levels, and caribou health and condition. More importantly, we aimed to determine if the continuation of these interviews could be valuable as an integral part of the annual monitoring of the herd, in support of the HMS.

Methods

Interview Process

Kofinas (2002) described the process in conducting the community interviews for the ABEKC. Interviews were conducted in the spring by locally hired individuals selected by the local organizations. The reporting period for each interview included the previous spring, fall, and current winter observations.

Table 1. Number of interviews analyzed by interview year.

Year	Interviews
2000-01	57
2001-02	56
2002-03	43
2003-04	57
2004-05	37
2005-06	85
2006-07	78

Thus, interviews in spring 2001 represented the 2000-2001 interview year. A three-day training session prepared the interviewers to conduct interviews and report on their work. Annually, between 10 and 15 local experts were selected by the local organization to be interviewed in each community. Although not all communities participated throughout the study period, communities included in the analysis were Aklavik, Fort MacPherson, Tsiigehtchic, and Tuktoyuktuk in the Northwest Territories, Old Crow in Yukon, and Arctic Village and Kaktovik in Alaska. Local experts were identified as those with the most extensive and current knowledge of conditions on the land. Thus, for example, elders who no longer went on the land were not selected. Interviews took place in person at the most convenient location and questions were both closed and open-ended with experts allowed to elaborate on their categorical answers when necessary. Between 2000 and 2007 a total of 413 people were interviewed. There were on average 59 interviews conducted each year with a low of 37 (2004-2005) and a high of 85 (2005-2006) that provided responses related to the data analyzed in this paper (Table 1).

Data analysis

The basis of this summary and comparative analysis is the percent frequency of responses to questions posed to the interviewees. Therefore it was necessary to determine a limited number of categorical (*e.g.*, "good", "average", "bad") responses. The frequency of responses in each class were converted to a percentage and plotted for each year of the study. Trends during the survey period were tested with a Spearman correlation (SAS version 9.1; SAS, 2006)

Caribou availability

Interviewees were asked how available caribou were to their community using categories "close", "not close", or "not available" for fall, winter, and spring. In many instances, the respondents qualified their answer often related to weather factors, personal ability to travel or difficult terrain. Thus the answer cannot directly be interpreted as a quantifiable distance from community, but rather a synthesis of distance with meaning to the interviewees themselves. To directly compare from one interview session to the next, an index of caribou availability was developed (*i.e.*, collapsing all responses into one metric). The caribou availability index (CAI) was calculated as:

CAI = 3*(%close) + 2* (%not close) + (%not available);

where "close", "not close", and "not available' were the percentage of those responses for an interview session.

Meeting needs

The interviewees were asked whether they met their needs for caribou for the fall and spring hunting periods with answers of "yes" or "no". What that question meant to the interviewee was captured when they qualified their answers. Their qualifications ranged from personal ability to hunt, to caribou availability, to whether they were able to share some of their kill.

Hunting activity

For the fall and spring periods, the interviewees were asked whether they hunted or not and, if not, why they didn't hunt. The percent of respondents that actively hunted was compared among periods.

Harvest

Those hunters that responded that they hunted were asked how many animals they killed. An

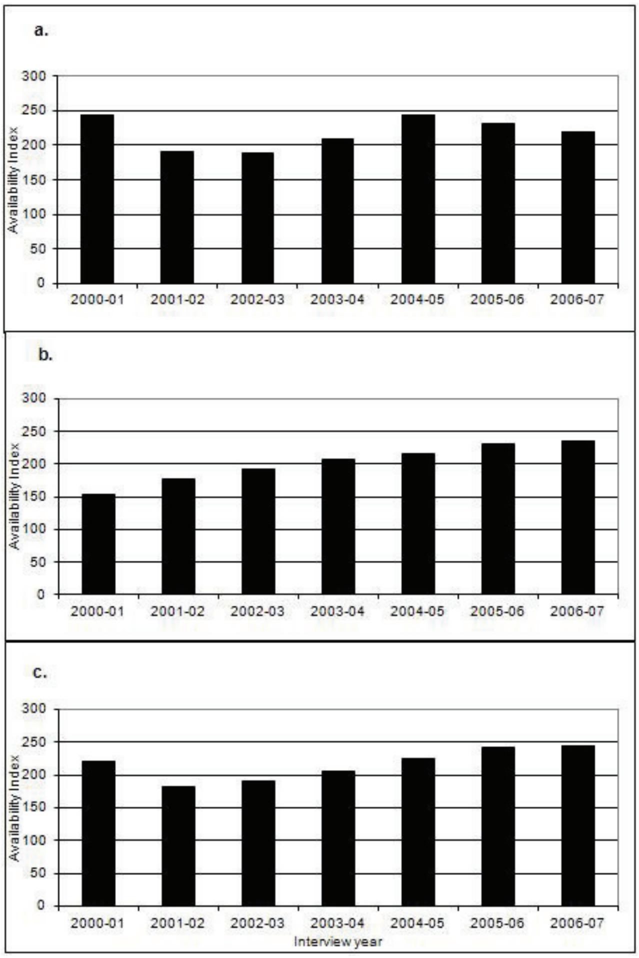

Fig. 1. Annual (2000–2007) fall (a), winter (b), and spring (c) availability index values of the Porcupine caribou herd based on community interviews.

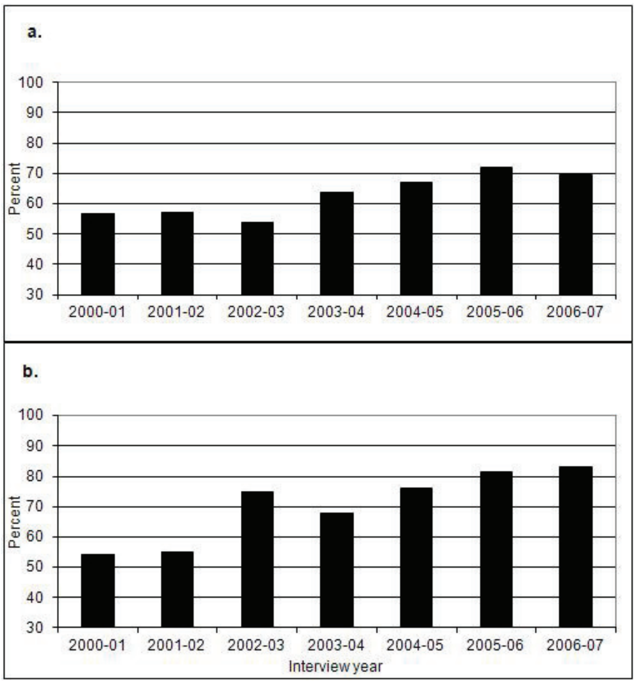

Fig. 2. Percent of respondents that met their Porcupine caribou needs in fall (a) and spring (b), 2000–2007, based on interviews in user communities.

index of harvest was calculated by multiplying the average number of caribou taken by the percent of respondents that indicated they hunted. Thus if the average take was 5 animals and the proportion of respondents indicating they hunted was 0.70, then the harvest index (HI) was 5*0.70 = 3.5. This index was used to compare HI among years.

Unusual health

During interviews, people were asked whether or not there was anything unusual in the health of the PCH in the previous fall and spring. We have used the response to this question as an annual index of the health of individuals in the PCH. If the interviewees responded "yes" to unusual health, respondents were asked what was unusual. Although most indicated negative reasons (*e.g.*, disease sign, skinny) some of those interviewed indicated positive conditions (*e.g.*, many fat animals). The health index is simply the percent of "no" responses and positive plus neutral "yes" responses, the higher the index the better the health of the herd.

Results

Caribou availability

Caribou availability in the fall showed no noticeable pattern (r = 0.07, P = 0.88) in contrast to winter (r = 0.99, P < 0.0001) and spring (r = 0.78, P = 0.04) when caribou were increasingly available beginning in 2000-2001 in winter and 2001-2002 in spring (Fig. 1). Availability based on combining all three periods shows a steady increase from 2000-2001 for the three periods (r = 0.75, P = 0.05).

Meeting needs

On average, 63% of respondents indicated that they met their needs for caribou in the fall hunting season and 70% met their needs

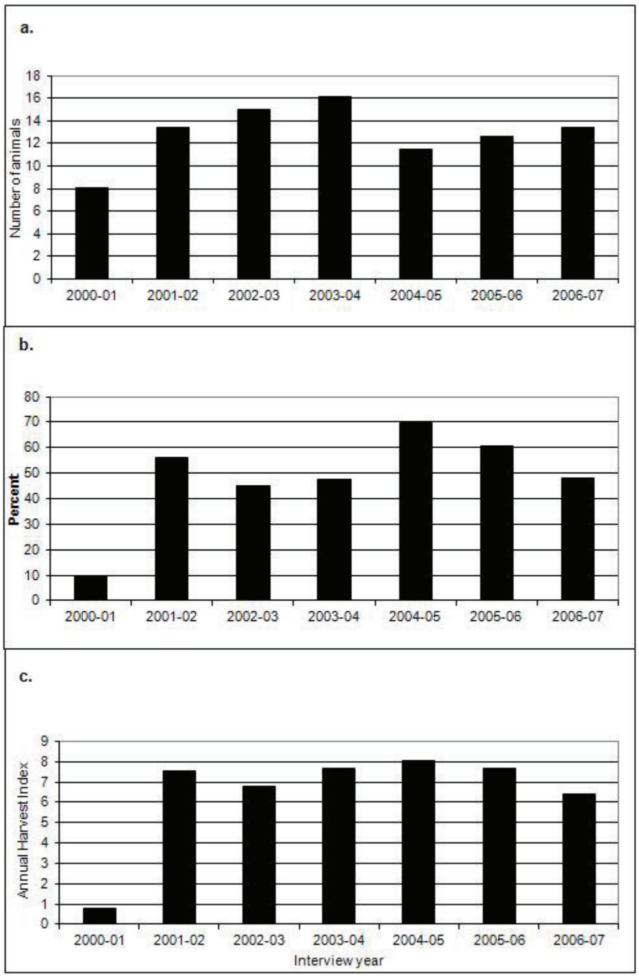

Fig. 3. Summary of harvest information of Porcupine caribou, 2000–2007, based on interviews in user communities: average number harvested per successful hunter (a), percent of interviewees that harvested caribou (b), and annual harvest index – the proportion of interviewees that hunted/average number of caribou harvested, HI (c).

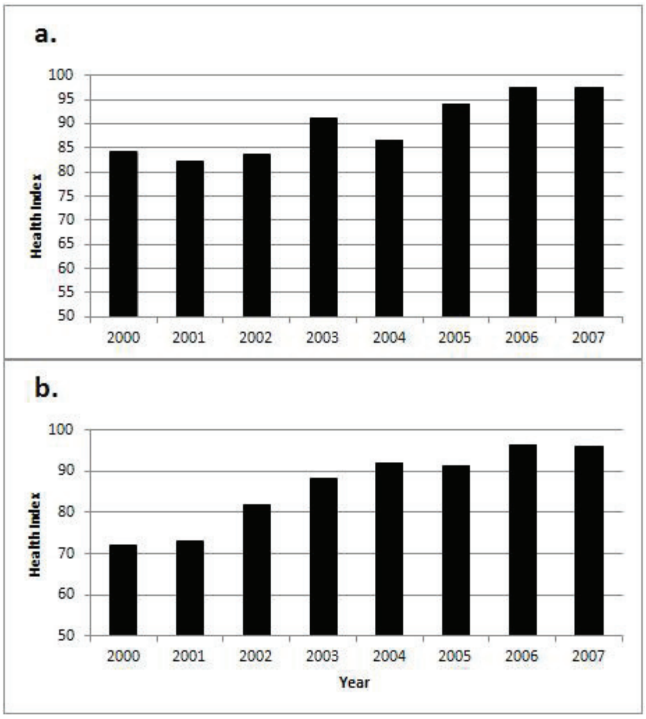

Fig. 4. Annual index of fall (a) and spring (b) Porcupine caribou herd health from 1999–2007 based on Arctic Borderlands interviews.

in the spring hunting season. There was an increasing general trend from 2001–2007 during both spring (r = 0.96, P = 0.005) and fall (r = 0.86, P = 0.01). In the latter three years of the surveys, on average, 70% and 80% of the respondents met their needs in fall and spring, respectively (Fig. 2). We noted a positive correlation between CAI and the percent of hunters that met their needs in spring (n = 7, r = 0.83).

Hunting activity

On average, 48% of the respondents annually reported that they harvested animals and of those successfully taking animals, the average number of animals taken was 13.2+12.8, range 1-120. There was no significant trend in average number of animals harvested during the study (r = 0.17, P = 0.79; Fig. 3a). Additionally, there was no trend in the percent of respondents indicating they harvested animals (r = 0.57, P = 0.18; Fig. 3b) and no trend in HI (r = 0.32, P = 0.48; Fig. 3c). Of those that did not hunt, the majority indicated they did not hunt because animals were unavailable to them, based on their ability to access areas occupied by caribou. Beyond availability, responses were equally scattered among other response categories (*e.g.*, no means to hunt, meat obtained elsewhere, no time to hunt, never hunt in the particular season, and bad weather). The latter reason, bad weather was much more important during the spring season than the fall season.

Unusual health

For the fall period, of 454 interviews, 99 (22%) respondents indicated there was something unusual, of which 11 (11%) gave neutral, 27

(27%) positive, and 61 (62%) negative reasons. The latter three years (2005–2007) had the highest health index among the years analyzed and there was a general increasing trend since 2001 ($r = 0.90$, $P = 0.002$; Fig. 4a). Of the 439 people who answered regarding any unusual health of caribou in spring between 2001 and 2007, 94 (21%) indicated there was something unusual and 345 (79%) said there was nothing unusual. Of those who answered "yes" to unusual, 14 (15%) indicated a positive reason (*e.g.*, unusually fat, very good condition), 53 (57%) gave a negative reason (*e.g.*, skinny, looked unhealthy, disease), and 27 (29%) gave a neutral reason (*e.g.*, many wounded animals, animals didn't come close). There was an increasing trend in the spring ($r = 0.95$, $P = 0.0003$) health index from 2000–2007 (Fig. 4b).

Discussion

During the period when aerial photo-census surveys to determine population estimates for the PCH were ineffective, community people reported an increasing availability, increased ability to meet their harvest needs, a stable harvest, and an increasing trend in caribou health. We conclude therefore that based on these interviews with community members between 2001 and 2007, conditions for the PCH and the ability of communities to access caribou apparently improved, especially for the spring period. Similarly there was a positive trend in the number of respondents indicating they had met their needs, which was related to caribou availability.

Although caribou tended to be more available throughout the period and a greater proportion of hunters met their needs, these factors did not translate into a higher harvest of caribou. The results of the percent of successful hunters and the average number of caribou taken indicate that HI remained constant throughout the period, with the exception of a low harvest in 2000-2001. In that year fewer people hunted (9.8% versus 48%) and those that did harvested fewer animals (8.1 versus 13.0) compared to the long-term average. This also suggests that this HI may not be sensitive enough to detect subtle changes in caribou availability, because the group targeted for interviews were community experts who may be able to compensate for reductions in 'herd availability' while other less experienced hunters may be affected by changes in availability.

There is a myriad of factors that can influence the health and condition of caribou in fall and spring. Fall condition is related to lactation status and probability of pregnancy (Cameron *et al.,* 1993; Gerhart *et al.,* 1997; Russell & White, 2000), condition entering in the summer, timing of green-up, level of summer insects (Weladji *et al.,* 2003) as well as parasite load and disease (Albon *et al.,* 2002). Snow characteristics play a primary role in the condition of caribou in the spring (Weladji & Holand, 2003), although lactation status and pregnancy (primarily in late-spring) also are important. Thus, community perception of caribou health integrates all these factors and more (Lyver & Dene First Nations, 2010). Parlee et al. (2005) noted a number of indicators that aboriginal hunters use to determine the health and physical condition of caribou, including not only overall appearance and chest girth, for example, but also behavioural characteristics.

Existing knowledge about caribou is frequently uncertain. The learning process involved in making management decisions includes mutual acknowledgement among co-management participants of the limitations of what is known about caribou systems (Kendrick, 2003). Co-management boards cannot make effective management decisions when information about population levels and harvest rates are lacking. The only way to address this lack of information is to develop multiple methodologies for collecting information about herd status from the multiple perspectives and knowledge that are held by people sitting at the co-management table (Kendrick, 2003).

The HMP calls for an annual assessment of herd status by considering a number of biological indicators, as well as several caribou-related questions from the ABEKC questionnaire. However, for the first annual assessment, results of the ABEKC interviews were not requested so ABEKC information was not available for consideration during those discussions. One of the objectives of this manuscript was to identify information sources to the PCMB and management agencies, especially during the period when little scientific data are available on harvest levels and trends in the population.

The development of ABEKC's indicators of caribou population condition presented here are significant for resource managers. First, the requirement to integrate community-based knowledge into decision making has been hindered by our ability to monitor local knowledge and integrate results to address management concerns. This paper is a start to address that challenge. Second, knowledge gaps related to conventional ecological monitoring in the north (*e.g.*, population estimates), leave decision makers with no alternative or parallel monitoring information. In fact, at the writing of this manuscript, the planned 2012 photocensus was cancelled due to poor weather and failure of the PCH to congregate during the calving/post calving period. At the same time, preliminary results from the ABEKC interviews for 2011 were already shared with government agencies, aboriginal governments, and co-management boards at the ABEKC data validation gathering in March 2012.

The next logical step would be to better understand and integrate ABEKC interview results with climate data, vital rates, and satellite collar movement and distribution data. These observations by expert community members need to be considered as an integral component in understanding the status of the PCH and thus should be presented, circulated, and utilized based on their own merit.

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Calf mortality of semi-domesticated reindeer (*Rangifer tarandus tarandus)* in the Finnish reindeer-herding area

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Abstract: During 1999-2008 calf mortality was studied in six reindeer-herding cooperatives in Northern Finland, where 3942 semi-domesticated reindeer (*Rangifer tarandus tarandus)* calves were equipped with radio mortality collars. The calves were weighed and earmarked mostly at 2-5 days of age, or at 2-8 weeks of age. Altogether 460 dead radio-collared calves were found from calving in May until winter round-ups in October-January. In northern mountain herding cooperatives, the average mortality of calves varied between 7-12%. On average, 39-54% of calves found dead were attributed to predation. Golden eagles killed 0-3.5% of calves in different years and areas in Ivalo and Käsivarsi cooperatives. Golden eagles were responsible for 33-43% of the cases and 84-93% of all identified predation. Most calves killed by golden eagles were found in July-August and in open areas. Calves killed by golden eagles were significantly ($P<0.01$) lighter than those not predated. No predation occurred in the Poikajärvi cooperative, but the annual mortality of calves varied between 0-35% in cooperatives near the Russian border. In Oivanki cooperative brown bears killed on average 2% of the radio-collared calves. Most predation (87%) occurred at the end of May and in early June. In the Kallioluoma cooperative, predator-killed calves found comprised 53% and wolf-killed 45%. Predation was 70% of total mortality in the Halla cooperative, and predation by wolf, bear, lynx and wolverine comprised on average 38%, 20%, 9% and 2.3%, respectively. The sex and pelt color did not significantly affect survival of calves. Birth weight of calves killed by bears was significantly ($P<0.01$) lighter than those not killed, but those calves killed by lynxes were significantly ($P<0.05$) heavier than that survived. Bears killed calves mainly in May-July, wolves in July-October and lynx in August-December.

Key words: calf mortality; predation; radio telemetry; semi-domesticated reindeer; survival.

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Introduction

Several studies have shown the major role of large terrestrial carnivores in neonate and juvenile mortality of different ungulate species (see Linnell *et al.,* 1995). Mortality during the first summer is an important factor in the population dynamics of many reindeer (*Rangifer tarandus tarandus)* and caribou *(R. tarandus)* herds, and predation is usually suspected as the primary cause of mortality (Miller & Broughton, 1974; Page, 1985; Mahoney *et al.,* 1990; Skogland, 1991; Whitten *et al.,* 1992; Adams *et al.,* 1995; Valkenburg *et al.,* 2004). Apart from many herding activities and supplementary feeding during winter months, reindeer are free-ranging most of the year. In the northern parts of Fennoscandia semi-domesticated reindeer comprise an important source of prey

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for many predators. A similar situation occurs in Finland, especially during the summer and snow-free periods. These areas share parts of their range with different predators including wolverine *(Gulo gulo),* Eurasian lynx (Lynx lynx), brown bear *(Ursus arctos),* wolf *(Canis lupus),* and golden eagle *(Aquila chrysaetos).* Large carnivores may cause substantial losses in semi-domesticated reindeer by preying on both adults and juveniles (Bjärvall *et al.,* 1990; Nybakk *et al.,* 2002; Danell *et al.,* 2006; Nieminen, 2010). Furthermore, golden eagle and red fox *(Vulpes vulpes)* also prey upon reindeer calves (Nybakk et al. 1999; Tveraa *et al.,* 2003; Norberg *et al.,* 2006).

The aim of this study was to investigate the survival, timing, extent, and causes of reindeer calf mortality in six reindeer-herding cooperatives in the Finnish reindeer husbandry area. Studies have been necessary to assess the role of predation in calf losses and the feasibility of current compensation regimes for predator-killed semi-domesticated reindeer. Some results are published earlier in Finnish reports (Norberg & Nieminen, 2004; 2007; Norberg *et al.,* 2005) and in the journal *Rangifer* (Nieminen, 2010; Nieminen *et al.,* 2011).

Study areas

The reindeer-herding cooperative of Ivalo, situated in the municipality of Inari, northern Finland, covers a total land area of 2626 km^2 (see Fig. 1). In Ivalo, there are two herding-groups, Nellim and Southern area partly separated from each other by a fence. The reindeer-herding cooperative of Käsivarsi, situated in the municipality of Enontekiö, covers a total land area 4658 km^2, and is the second largest among the 56 cooperatives in Finland. The cooperative of Käsivarsi is divided into three separate herding-groups by fences: 1) Palojärvi, 2) Kova Labba, and 3) Raittijärvi. The reindeer-herding cooperative of Poikajärvi (2414 km^2) is situated in the municipality of Rovaniemi. These study cooperatives are situated in the north boreal vegetation zone (Ahti *et al.,* 1964).

In Ivalo cooperative, range is dominated by rolling hills with different aged forest stands, mainly Scots pine *(Pinus sylvestris).* Mountain birch *(Betula pubescens czerepanowii)* grows on the slopes of the highest hills. In Käsivarsi, mountains dominate the landscape. The area of coniferous forests is relative small and located in the southern part of the cooperative. In Poikajärvi, the main landscape types are Scots pine and Norwegian spruce *(Picea abies)* forests. Oivanki (1361 km^2) and Kallioluoma (1369 km^2) cooperatives are located in Kuusamo municipality near the Russian border. The reindeer-herding cooperative of Halla in Kainuu, also located near Russian border and north of the wild forest reindeer *(R. t. fennicus)* area in

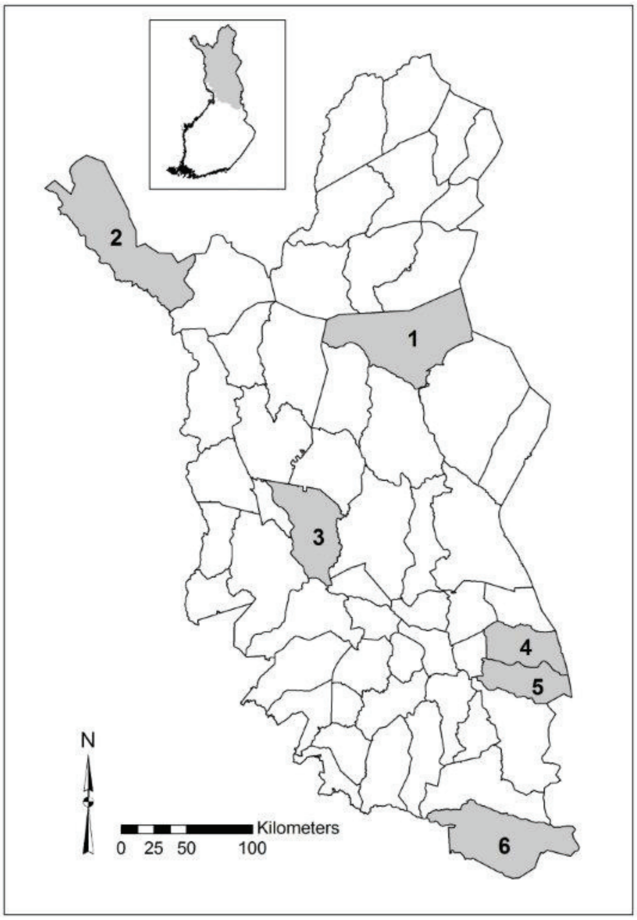

Fig.1. Reindeer-herding cooperatives in the study and Finnish reindeer husbandy area: 1. Ivalo, 2. Käsivarsi, 3. Poikajärvi, 4. Oivanki, 5. Kallioluoma and 6. Halla.

Finland (municipalities of Suomussalmi, Puolanka and Hyrynsalmi), covers a total land area 3592 km^2. It is the eighth largest cooperative in Finland. The cooperatives of Oivanki, Kallioluoma and Halla belong to the middle boreal vegetation zone (Ahti *et al.,* 1968) (Fig. 1). The main landscape types are Norway spruce and Scots pine forests with ericaceous heather and lichen and boggy areas.

There were on average 5660 reindeer (adults and calves after slaughtering) in the reindeer-herding cooperative of Ivalo during the study in 1999-2001, 10690 reindeer in Käsivarsi during 2002-04, 4670 in Poikajärvi during 2001-03, 2440 in Oivanki during 1999-2004, and 2270 in Kallioluoma during 2005-06. In Oivanki, the actual study area was the eastern part of the cooperative confined to Russian border. In Kallioluoma, the study area was between main road 5 and the border with Russia. There were approximately 1200 to 1600 adult (>1 year old) reindeer in the cooperative of Halla during the study 2006-08, and yearly about 500-700 calves were born. Reindeer densities (winter stock/km^2) were similar at 2.1 in Ivalo and 2.3 in Käsivarsi. In Poikajärvi, reindeer density was 1.9, in Oivanki 1.8, in Kallioluoma 1.7, and significantly lower in Halla, at 0.6-0.8 reindeer/km^2.

In the entire Finnish reindeer husbandry area, there were an estimated >160 bears, >75 wolverines, >50 lynx, and 15-25 wolves (RKTL, 2008). During 2000-03 in the entire northern reindeer husbandry area (including the 13 cooperatives) the minimum number of large carnivores was: 45-50 brown bears, 40-45 wolverines, 5-10 lynx, and 3 wolves (Kojola & Määttä, 2004). The most abundant among the large carnivores was brown bear in Ivalo and wolverine in Käsivarsi cooperative. In the eastern reindeer husbandry area, the minimum number of brown bears was 80-85 and 15-20 lynx. Most wolves of the Finnish reindeer husbandry area are found in Kainuu (23 000 km^2). In winter 2008, the size of the wolf population there was estimated at 29-37 animals, a decrease of about 50% compared with the previous year. The population size of the lynx was estimated at 140-190 individuals, including 23-31 litters, and the size of the wolverine population at 36-53 animals (Siira *et al.,* 2009). Faeces samples collected during summer 2005 in Kainuu (Näljänkä and Halla cooperatives) were genetically analysed (DNA), and 46 unique brown bears were identified (RKTL, 2008). In winter 2007, 55-62 wolves were estimated in wild forest reindeer area of Kuhmo (RKTL, 2008).

The golden eagle population in Finland is about 440 pairs or territories. About 80% of all golden eagles occur in Lapland (at most 350 pairs in 2006; Large Carnivore Working Group, 2008) and 90% in whole reindeer herding area. In the Kainuu area, 11 territorial pairs of golden eagles were estimated in 2009 (Ollila, 2009).

Material and methods

We fitted 3942 reindeer calves with mortality indicating radio-transmitters (Televilt Inc., Lindesberg, Sweden) fixed on expandable neck collars in six reindeer-herding cooperatives, in five herding-groups and areas during 1999-2008. Radio collars weighed about 100 grams, about 0.3-2.5% of the body weight of the calves at marking. Calf survival and cause-specific mortality were studied two years in the herding groups of Nellim (1999-2000) in Ivalo and Raittijärvi (2002-03) in Käsivarsi, and three years in other herding-groups (Southern area of Ivalo 1999-2001, Palojärvi and Kova-Labba 2002-04) in these cooperatives and also in cooperatives of Poikajärvi, Kallioluoma and Halla. The Oivanki cooperative study continued six years from 1999 to 2004.

Calves were weighed and marked at 2-5 days of age in calving corrals in May/June. Females were fed in corrals for 1.5 months with silage and concentrates during the spring and calving periods. Mid-summer earmarking took place in the last weeks of June and the first two weeks of July, when calf age was 2-8 weeks. Calves were marked in different corrals of cooperatives. Calves were sexed and weighed, and pelt color recorded upon which the collared calves were reunited with their mothers and released to summer pastures.

Dead collared calves were located by tracking mortality sensor signals from the air (fixed-wing aircraft and helicopter) and by ground triangulation (sensors activate after 2.5 hrs of being motionless). Tracking was performed in 2-3 day intervals during summer until the end of August, and once per week in September and October. Mortality activated radio-collars were located by using hand receivers (Televilt RX-8910®, Televilt Inc., Lindesberg, Sweden and Tracker Maxima® and hound radars, Tracker Inc., Oulunsalo, Finland). Field observations of the site and carcass were recorded and photographed. Cause of death was first investigated in the field (*e.g.*, evidence supporting presence of predator/scavenger species, such as tracks, scats and feathers/downs) and then augmented by necropsies conducted by biologists in the laboratory of the Reindeer Research Station in Kaamanen. The presence of hemorrhaging and perforations, both in the skin and soft tissues of the dead calf, were critical for determining the cause of death by depredation. If the combined evidence from the field site and the necropsy was inconclusive, usually due to late discovery of carcass, the cause of death was classified as unknown (see also Bjärvall *et al.,* 1990; Norberg *et al.,* 2005; 2006).

Physiological condition of dead reindeer calves in Halla cooperative was determined by using the oven-dry method of metatarsal marrow fat. Condition was expressed as percent of marrow fat (see Nieminen & Laitinen, 1986).

Statistical analysis

Due to the difference in the marking time and age of calves, the weight of calves at marking ranged between 4.2-32 kg, and therefore for statistical analysis the weights were adjusted to 1st June and to 1st July using a daily growth rate of 270 grams (see Timisjärvi *et al.,* 1982) for all calves weighed in the calving corrals in May. A daily growth rate of 302 grams was used for female calves and 315 grams for male calves weighed later during earmarking (Norberg *et al.,* 2005; 2006).

The daily survival estimates and ´reindeer days´ (one ´reindeer day´= one radio-collared reindeer out for one day) for the radio-collared calves were calculated using the Kaplan-Meier product/limit method (Kaplan & Meier, 1958) and using the computer program ´Kaplan-Meier survivorship analysis version 1.0´ (Pollock *et al.,* 1989) to obtain daily and total survival estimates for the study periods. Daily survival estimates were used to present survivorship curves between May/June and October. For calculating monthly survival estimates, cause specific mortality rates and 95% confidence limits, the program ´Micromort version 1.3´ (Heisey & Fuller, 1985) was used.

Survival estimates for this analysis were calculated based on calves that were: 1) found dead, 2) had dropped their radio-collars during the study, or 3) were recovered in the autumn/winter round-ups (survivors) when radio-collars were taken off. The statistical differences in calf weights in different groups were tested using t-test and stepwise logistic regression. In addition to weight, the effect of sex, pelt color, study year, and possible interactions on survival probability were investigated using logistic regression. Statistical tests were carried out by use of SPSS ver. 7.0 for Windows. The data were examined for statistical significance at $P<0.05$.

Results

During calving, 460 radio-collared calves were found dead in the six cooperatives studied. Pooled survival estimates in the mountain cooperatives of Ivalo (including all radio-collared calves during years 1999-2001) and of Käsivarsi (years 2002-04) were 0.937 (SE=0.011) and 0.885 (SE=0.011), respectively.

Of 806 reindeer calves radio-collared in Ivalo reindeer-herding cooperative during 1999-2001, 4.6% (37 calves) were found dead, 90.4% survived, and 5.0% were not recaptured until the end of the study (annual monitoring from marking until the end of October). Of 919 radio-collared calves in Käsivarsi cooperative during 2002-04 in total 5.2% (48 calves) were found dead, 87.4% survived, and 7.4% were not recaptured. Highest area-specific annual mortality occurred in Ivalo in 2000, when in total 19 (51.4% of all dead calves) were found dead, and in Käsivarsi in 2004, when 20 radio-collared calves (41.7%) were found dead.

In the northern mountain herding cooperatives of Ivalo and Käsivarsi the average mortality of calves varied between 7-12% (Fig. 2). On average, 39-54% of the calves found dead were attributed to predation, and golden eagles killed 0-3.5% of calves in different years and areas in these cooperatives. Golden eagle predation accounted for 33-43% of the cases and 84-93% of all identified predation. The most calves killed by golden eagles were found during July-August and in the open areas.

In Ivalo cooperative, 8% of all calves found dead were killed by brown bear and 3% by red fox, and in Käsivarsi 6% by wolverine and 2% by unidentified predators. Mortality rates caused by predators other than golden eagle were on average less than 1% in both study cooperatives. The share of other identified causes of death (accidents, traffic, others) was 19% in both cooperatives, while 27-40% of dead calves were associated with unknown causes of mortality. In Ivalo during 1999-2001, a total of 8% of all calves found dead were from accidents and collisions with vehicles, while 6% were by accidents in Käsivarsi. Other causes (11-13%) included disease, stress, and poor condition of calves in both cooperatives. When calves with unidentified causes of death were excluded, predation comprised on average 69% of the observed mortality in Ivalo and 74% in Käsivarsi.

Of 404 reindeer calves radio-collared in Poikajärvi reindeer-herding cooperative during 2001-03 only 2.5% (10 calves) were found dead, 93% survived, and 4.5% were not recaptured until the end of the study. The average mortality from calving period in May to the end of October was 5.7% and from 15th June to the end of October 1.4%. No predation occurred, and causes of death included poor condition of calves and traffic.

Annual mortality of calves varied between 0-35% in cooperatives near the Russian border. Of 580 reindeer calves radio-collared in the eastern part of Oivanki during 1999-2004, 7.2% (42 calves) were found dead, 86.6% survived and 6.2% were not recaptured until the end of the study. The average mortality from the calving period in May to the end of October was 9.7% and from 15th June to the end of October was 2.3%. In Oivanki, most mortality (87%) occurred in May and June. The most prominent cause of death was predation by brown bear comprising on average 2% of all radio-collared calves during 2000-04. When calves with unidentified causes of death were excluded, predation comprised on average 50% of the observed mortality in Oivanki. From all identified causes of death (*n*=18) 33.3% were killed by brown bear and wolverine, while lynx and wolf killed 5.6%.

In the eastern part of Kallioluoma, 139 of 587 radio-collared calves (23.7%) were found dead within the study period 2005-06. Mortality of radio-collared calves was on average 18-19% by the end of June, and 28-29%, 36-39% and 42-46% by the end of October, December, and mid-January, respectively. Predator-killed calves comprised 53% and wolf-killed calves were 45% of all the dead calves found. Wolf predation was on average 18% while the total

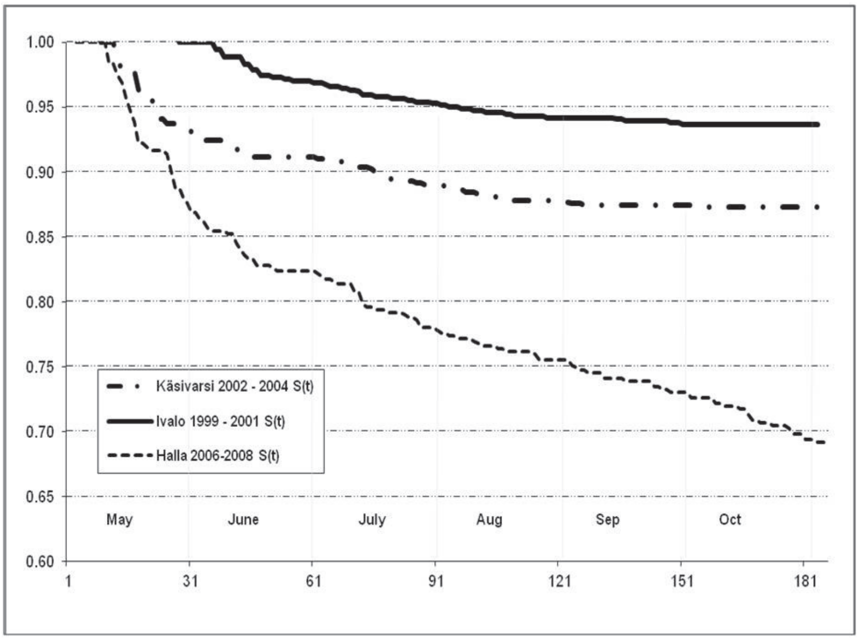

Fig. 2. Total survival curves for radio-collared reindeer calves in Ivalo (in 1999-2001), Käsivarsi (in 2002-04) and Halla (in 2006-08) reindeer-herding cooperatives, expressed as days after 1 May (day 1). (Mortality (M) = 1 – Survival(S)).

rate of all predation was at least 21%. Large carnivores comprised 92-97% of the total predation. The mortality rate due to other sources of mortality than predation was on average 10%. From all identified causes of death (*n*=100 calves) brown bear killed 62%. Bear and lynx killed 3%, and golden eagle 1%.

Of 546 radio-collared reindeer calves during 2006-08 totally 177 (32.4%) were found dead during the research period until mid-January in Halla cooperative in Kainuu area. The total mortality in 2006-08 was 30.7% at the end of October and increased to 34.6% by mid-January (see Fig. 2). Predation was 70% of total mortality, and predation by wolf, bear, lynx, and wolverine comprised on average 38%, 20%, 9%, and 2.3%, respectively. The mortality of reindeer calves was slightly higher in Suomussalmi near Russian border than in Hyrynsalmi/Puolanka area, because of bear and wolf predation. The sex and pelt color did not significantly affect survival of calves.

The average adjusted weight of those radio-collared calves that survived in Ivalo cooperative was significantly higher (mean 9.8 kg, SD=2.0 kg, *n*=169) than weight of dead calves (mean 7.8 kg, SD=2.2 kg, *n*=12; t=2.79, df=179, P=0.006). Calves killed by golden eagle were significantly lighter (mean 7.2 kg, SD=2.2 kg, *n*=5; t=2.47, df=172) than surviving calves, and also lighter than those calves that died from other causes (mean 8.2 kg, SD=2.4 kg, *n*=7). In Kova Labba herding-group (Käsivarsi cooperative) calves killed by golden eagle were 1.7 kg lighter (mean 10.7 kg, SD=1.4 kg, *n*=5) than those calves that survived (mean 12.4 kg, SD=1.9 kg, *n*=259; t=1.9, df=262, P=0.053).

The average adjusted weight (on 1st July) of those radio-collared calves killed by golden eagles in Ivalo cooperative was significantly lower (mean 12.8 kg, SD=1.7 kg, *n*=11) than mean weight of survivors (16.6 kg, SD=2.5 kg, *n*=560; t=760, df=10.84, *P*<0.001). Also, mean weight of calves killed by all predators was significantly lower (13.1 kg, SD=1.7 kg, *n*=13) than mean weight of survived calves (t=7.05, df=13.17, *P*<0.001). The weight of calves killed by golden eagle in Palojärvi herding-group in Käsivarsi was significantly lower (mean 13.0 kg, SD=1.7 kg, *n*=5) than mean weight of survived calves (19.3 kg, SD=3.4 kg, *n*=280; t=409, df=283, *P*<0.001). Mean weight of calves killed by all predators was significantly lower (12.2 kg, SD=2.2 kg, *n*=7) than mean weight of survived calves (t=5.46, df=285, *P*<0.001). Weight of calves killed by golden eagle in Kova-Labba herding-group was very low and significantly lower (mean 7.2 kg, SD=3.2 kg, *n*=6) than mean weight of survived calves (13.4 kg, SD=4.3 kg, *n*=28; t=3.03, df=31, P=0.005). No significant effect of pelt color on calf survival in Ivalo and Käsivarsi cooperatives was found.

The average adjusted weight (on 1st June) of those radio-collared calves that were found dead in Poikajärvi cooperative was slightly lower (mean 8.0 kg, SD=2.6 kg, *n*=10) than mean weight of survivors (9.3 kg, SD=2.1 kg, *n*=155; *P*<0.05). The average adjusted weight of those radio-collared calves that were found dead also in Oivanki cooperative was slightly lower (mean 9.3 kg, SD=2.5 kg, *n*=38) than mean weight of survivors (11.0 kg, SD=2.1 kg, *n*=348; *P*<0.001). The weight of calves killed by predators (8.6 kg, SD=1.2 kg, *n*=8) were significantly lower than weight of survivors (11.0 kg, SD=2.1, *n*=347; t=3.23, df=353, P=0.001) (see also Norberg *et al.,* 2005). In contrast to previous calf mortality, the weight of the calves in Kallioluoma cooperative did not have significant effect on the mortality.

Birth weight of calves killed by bears in Halla was significantly (*P*<0.01) lighter, but killed by lynx significantly (*P*<0.05) heavier than that of survivors. Bears killed calves mainly in May-July, wolves in July-October and lynx in August-December. Causes not associated with predation comprised 11.9% of total mortality, and included traffic accident and other accidents. Metatarsal fat content was < 25% in calves dead by bad condition and disease. Excluding the deaths from unknown causes (*n*=11), and if unknown calves eaten by different predators, mainly by bears (*n*=18), were also killed by these predators, total predation was very high, 83.1%.

Birth weight of calves that were lost or killed by predators during the study in Halla was on average 0.2 kg lower (mean 6.3 kg) than that of survivors. Birth weight of calves killed by bears was significantly (*P*<0.05) lower (mean 5.8 kg), but those killed by lynx was significantly (*P*<0.05) higher (mean 6.7 kg). Condition of calves was, however, fair or good (metatarsal fat content > 30%). The birth weight of male and female calves and pelt color did not affect survival differently, as there were no significant interaction between weight, sex and color on calf survival.

Discussion

According to Reimers (1983) mortality rates in calves of Svalbard reindeer *(R. t. platyrhynchus),* existing in an environment almost free of predators, are approximately 1% and 19% in the age intervals 0-6 and 6-12 months. The mortality rates among 0-6 month old reindeer calves is expected to be within the range of 6-21% found also in the predator-free South-Georgia (Leader-Williams, 1980), and lower than 45-60% found in many *Rangifer* herds subject to predation (Rehbinder, 1975; Bergerud, 1980). Wolves, bears, and golden eagles have been the most important predators of radio-tagged reindeer/caribou calves in both North America and Russia, as 80-89% of mortalities were caused by predation (see Bergerud, 1980).

In a study conducted in northeastern Finnish Lapland, mortality from golden eagles comprised annually 3-4% of the radio-collared cohort in 1997-98 (Norberg *et al.,* 2006). In studies conducted in central Norway, Nybakk et al. (1999) found golden eagle predation to account for 1-2% among radio-collared calves, while Kvam et al. (1998) observed a total mortality of 8%, and calves killed by golden eagle comprised 40% of all calves found dead. In the present study golden eagle was also the most significant cause of death both in mountain cooperatives of Ivalo and Käsivarsi, causing up to 3.5% annual mortality rate among radio-collared calves.

The majority of semi-domesticated reindeer calves are born in northern Finland in May with peak calving occurring from 18-23 May (Eloranta & Nieminen, 1986; Weladji *et al.,* 2006). In this study, 52% of calf mortality in Nellim and 87% in Kova-Labba reindeer herding group took place before the end of June. In Oivanki, most mortality (87%) occurred in May-June. According to Linnell et al. (1995) predation generally comprises the major share of the total mortality of juvenile ungulates. Many studies of reindeer/caribou have also demonstrated that calf mortality is usually highest during the first days and weeks after calving and then decreases considerably during summer and autumn (see Eloranta & Nieminen, 1986; Whitten *et al.,* 1992; Adams *et al.,* 1995; Norberg *et al.,* 2005).

In a study conducted in central Norway, 89% of the total mortality of calves from August to April was due to predation, and 60% of calves with identified cause of death were killed by lynx. Predation comprised an even higher proportion, 94% of all identified mortality, when examined from August to mid-November (Nybakk *et al.,* 2002). Also in northern Norway, predation accounted for 75% of the calf losses during summer and winter, and lynx was the main predator (55%) (Mathisen *et al.,* 2003). In Halla, predation caused 70% of all calves found dead and 87% of all identified mortality. Predation by wolf, bear, lynx, and wolverine comprised 38.4%, 20.3%, 9.0%, and 2.3% of all radio-collared calves found dead, respectively (see also Nieminen, 2010).

Highest mortality in mountain areas occurred in this study during July and August and was caused mainly by golden eagles. We conclude that access to, and use of alpine highlands and other open areas influenced the risk of reindeer calves to predation by golden eagles, and subsequently the temporal survival distribution in study cooperatives. We also emphasize the relative importance of golden eagles as a mortality factor in the northern part of the Finnish reindeer husbandry area, where the proportion of open alpine landscape is much higher than in the southern area (see also Nieminen *et al.,* 2011).

The design of the present study in Halla cooperative was similar to that of the four-year program that monitored reindeer calf mortality in Sweden in the 1980s (Bjärvall *et al.,* 1990), and also studies in 1995-96 in central Norway (Nybakk *et al.,* 2002) and in 1997-98 in northeastern Finnish Lapland (Norberg *et al.,* 2006). The total mortality recorded in the present study was, however, much higher (32.4%) than total mortality (14.3%) recorded in Umbyn, Sweden and in Lappi reindeer-herding cooperative in Finland (8.5%). The total mortality was also slightly higher than that in North-Trøndelag in Norway (31.0%) from August to April.

During 2006-08 in the Halla cooperative large predators (mainly wolf and lynx) killed 380 to 455 reindeer yearly, and compensation for predator-killed reindeer to reindeer owners was 5-6.5 times more than slaughter incomes. According to reindeer owners many wolves from Russia and Kuhmo are visiting Halla cooperative and killing reindeer mainly during summers and autumns. Predation accounted for a higher part of total mortality recorded in the present study (70%) than in studies in Sweden (65%) and in Finland (53%). In the Norway study, predation was higher (75.3%) than in the present study. Indeed, caribou herds exposed to predation may lose usually 50% of the annual calf crop (Bergerud, 1980), and predation can constitute up to 93% of total annual mortality in calves (Mahoney *et al.,* 1990). Nevertheless, the annual mortality recorded in the present study was higher than earlier reported from Sweden and Finland. If unknown calves eaten by different predators, mainly by bears, were also killed by these predators, total predation was highest in Halla at 83%. In central Norway 89.3% of the total mortality in calves was also due to predation, and predation by lynx was the dominant cause (42.4%) (Nybakk *et al.,* 2002).

A common perception is that animals preyed upon are either smaller or in poorer nutritional condition compared to survivors, and several studies (*e.g.*, Haukioja & Salovaara, 1978; Eloranta & Nieminen, 1986; Tveraa *et al.,* 2003) have shown that body weight of reindeer calves at calving and also during the first summer is positively correlated with survival. Although golden eagles are capable of killing ungulates up to the size of an adult reindeer in certain conditions (Bergo, 1987), they usually kill smaller than average calves (Nybakk *et al.,*1999; Norberg *et al.,* 2005; 2006). In the radio-collar study conducted in central Norway (Nybakk *et al.,* 1999), calves were marked during July and early August, and calves killed by golden eagles weighed on average 2.7-4.1 kg less than surviving calves. Also in the present study in mountain cooperatives, the weights of calves (adjusted to 1st June and to 1st July) killed by golden eagles were 1-3.8 kg lower than weights of survived calves. Mean weight of calves killed by all predators was also lower than mean weight of calves that survived, but there was no significant difference in weights of predator-killed calves compared to calves that died on other causes.

Birth weight of calves that were lost or killed by predators in Halla cooperative was slightly lower, but birth weights of calves killed by brown bears were significantly ($P<0.05$) lower than surviving calves. In our earlier study in nine reindeer herding cooperatives in Finland, birth weight of the lost calves was on average 0.4-0.5 kg lower than birth weight of the survived calves. In Oivanki cooperative, calves killed by bears also had 0.5 kg lighter birth weight compared to those that survived (Norberg *et al.,* 2002). Calves killed by lynx in Halla had, however, significantly ($P<0.05$) higher birth weight (mean 6.67 kg) than surviving calves. Most small calves were lost or killed by bears during early and mid-summer, and lynx killed bigger reindeer calves mainly during autumn. Also in central Norway predation by lynx peaked in autumn and early winter (Nybakk *et al.,* 2002).

In the present study, golden eagles were responsible for 33-43% of the cases and 84-93% of all identified predation in mountain cooperatives in the northern Finland. Most calves killed by golden eagles were found during July-August and in open areas. Predator-killed calves comprised >50% and wolf-killed calves 45% of all the dead calves found in Kallioluoma, in the southeastern cooperative near border of Russia. Our results showed, however, highest predation and calf mortality in Halla, in the southern cooperative, also near the Russian border and wild forest reindeer area of Finland. The total mortality was >30% at the end of October and reached 35% by mid-January. Predation was 70% of total mortality, and predation by wolf comprised on average 38% and by brown bear 20%. The economic consequences for reindeer husbandry of this area makes it questionable to what extent reindeer husbandry in its present form can be continued in Kainuu, southeastern reindeer-herding region of Finland.

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Standardized monitoring of *Rangifer* health during International Polar Year

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Abstract: Monitoring of individual animal health indices in wildlife populations can be a powerful tool for evaluation of population health, detecting changes, and informing management decisions. Standardized monitoring allows robust comparisons within and across populations, and over time and vast geographic regions. As an International Polar Year Initiative, the CircumArctic Rangifer Monitoring and Assessment network established field protocols for standardized monitoring of caribou and reindeer *(Rangifer tarandus)* health, which included body condition, contaminants, and pathogen exposure and abundance. To facilitate use of the protocols, training sessions were held, additional resources were developed, and language was translated where needed. From March 2007 to September 2010, at least 1206 animals from 16 circumpolar herds were sampled in the field using the protocols. Four main levels of sampling were done and ranged from basic to comprehensive sampling. Possible sources of sampling error were noted by network members early in the process and protocols were modified or supplemented with additional visual resources to improve clarity when needed. This is the first time that such broad and comprehensive circumpolar sampling of migratory caribou and wild reindeer, using standardized protocols covering both body condition and disease status, has been done.

Key words: body condition; caribou; disease; health; monitoring; parasites; *Rangifer tarandus;* reindeer; standardized protocols.

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Introduction

Understanding and tracking the health status of individual animals provides valuable information for wildlife management (Jean & Lamontagne, 2004; PCMB, 2010; GNWT, 2011). Changes in body condition or infectious disease indices of individuals may reflect shifts in population health and serve as early warning signals for wildlife managers, wildlife users, and public health officials concerned with zoonotic diseases. Examining interrelationships between various health indicators, as well as their trends over time and across herds and geographic regions, provides new insights into the ecology of a species and functioning of an ecosystem. Foundational to these activities, however, is securing a comprehensive baseline of the existing health status of the population (Karesh & Cook, 1995), including but not limited to: body condition, genetic diversity, physiological health, contaminants, and pathogen diversity and abundance. Understanding the relationships among various health indicators, and defining what is 'normal' and the variability around that normal within a host population, through population cycles and across the range of a species, is an important first step for identification of 'abnormal' and early detection of and response to changes in health.

Establishing baselines and maintaining monitoring programs for wildlife populations is not easy. Financial constraints, political and/or philosophical differences, transboundary issues, and the general elusive nature of wildlife make it difficult to establish and maintain successful programs (Kofinas *et al.,* 2002; Witmer, 2005). In the Arctic, widely scattered communities, a vast landscape, high costs of field access for research and hunting, and logistical constraints, add to the difficulties of wildlife monitoring. This is particularly true for caribou and reindeer *(Rangifer tarandus* sspp.), a very mobile keystone species (Geist, 1998) that is highly valued for cultural and economic reasons across its circumpolar range (Nuttal *et al*., 2010). Nevertheless, a variety of *Rangifer* monitoring programs of different types have existed over time (Kofinas *et al.,* 2002; Couturier *et al.,* 2004; Gunn *et al.,* 2005; Campbell, 2006; Campbell *et al*., 2010; Lyver & Nation, 2010).

Monitoring of a sort began with aboriginal people who, for thousands of years, have been keenly aware of the health of caribou or reindeer that they harvested for food, clothing, and tools (Cruikshank, 1981; Ferguson & Messier, 2010). A technical approach to monitoring body condition began in the mid-1900s, with emphasis on describing and validating condition indices which had allometric relations to the status of muscle, bone and fat, and their chemical constituents: water, protein, lipid, and ash (Dauphiné, 1976; Langvatn, 1977; Ringberg *et al.,* 1981a; b; Reimers & Ringberg, 1983; Huot & Goudreault, 1985; Adamczewski *et al.,* 1987a; Huot & Picard, 1988; Allaye Chan-McLeod *et al.,* 1995; Gerhart *et al*., 1996). Monitoring body size (*e.g.*, lower jaw and metatarsal bones) and shape (*e.g.*, heart girth) also provides insight into population trends (Parker, 1981; Crête & Huot, 1993; Mahoney & Schaefer, 2002; Couturier *et al.,* 2010). For the most part, this type of monitoring has been scientist-driven, typically initiated and conducted by wildlife managers or researchers, and often focused on a particular herd intermittently or for a finite length of time.

Opportunistic monitoring of infectious diseases of *Rangifer* by scientists in partnership with aboriginal hunters has also occurred since the mid-1900s, often initiated by hunter reports or submissions of abnormal tissues (*e.g.*, Canadian Cooperative Wildlife Health Centre http://www.ccwhc.ca/; Choquette *et al.,* 1967). More recently, targeted programs for contaminants and disease surveillance have developed across many jurisdictions (Elkin & Bethke, 1995; Zarnke *et al.,* 2000; Robillard *et al.,* 2002; Brook *et al.,* 2009; Stieve *et al.,* 2010).

Increasingly, through the wildlife co-management process, *Rangifer* users themselves are driving the context for, and implementation of, monitoring (Lyver & Gunn, 2004; Brook *et al*., 2009; Lyver & Nation, 2010). People who depend on *Rangifer* for food and income have expressed concerns about how this species, and those who depend on it, will cope with the increasing rate of environmental and political change (Kofinas *et al.,* 2003; Brook *et al.,* 2009). The impacts of climate change, resource development and other stressors on the health of *Rangifer,* and on food safety and security (*i.e.*, population sustainability) as it relates to *Rangifer,* are major concerns for subsistence hunters (Brotton & Wall, 1997; Brook *et al.,* 2009).

In light of global climate changes and conservation efforts, the Arctic Council launched a number of monitoring programs for Arctic species through the Conservation of Arctic Flora and Fauna Group (CAFF, 2010). The Circum-Arctic *Rangifer* Monitoring and Assessment (CARMA) network, positioned within CAFF, was launched in 2004 in response to increasing concern for *Rangifer* and the need for circumpolar collaboration. The network is a forum to exchange ideas, observations and data, and coordinate *Rangifer* monitoring activities around the Arctic.

A key objective of the CARMA network was to develop and implement standard methods for monitoring *Rangifer* health. Although monitoring activities were already in place for many herds, the methods, frequency, and type of data recorded varied greatly (Kofinas *et al.,* 2002). Integration of data across disciplines (*e.g.*, disease and body condition data), even within a herd, was sometimes lacking. Standardized approaches, both within and among herds, allow comparisons across space and time and, therefore, can provide a much deeper understanding of *Rangifer* health. As standardized baselines become established, links between the various health indicators can be examined and the costs, or benefits, of pathogens and pollutants at the individual, population and community levels can be evaluated. Such information provides the necessary foundation to assess *Rangifer* vulnerabilities and responses to environmental and anthropogenic changes. Importantly, for standardized protocols to be adopted and effective, the procedures need to be clear and without ambiguity, but flexible enough to accommodate differences in monitoring programs and objectives. At the same time, they must include enough indices to allow predictions: for example, to infer the probability of pregnancy from the fat and protein reserves (Kofinas *et al.*, 2003).

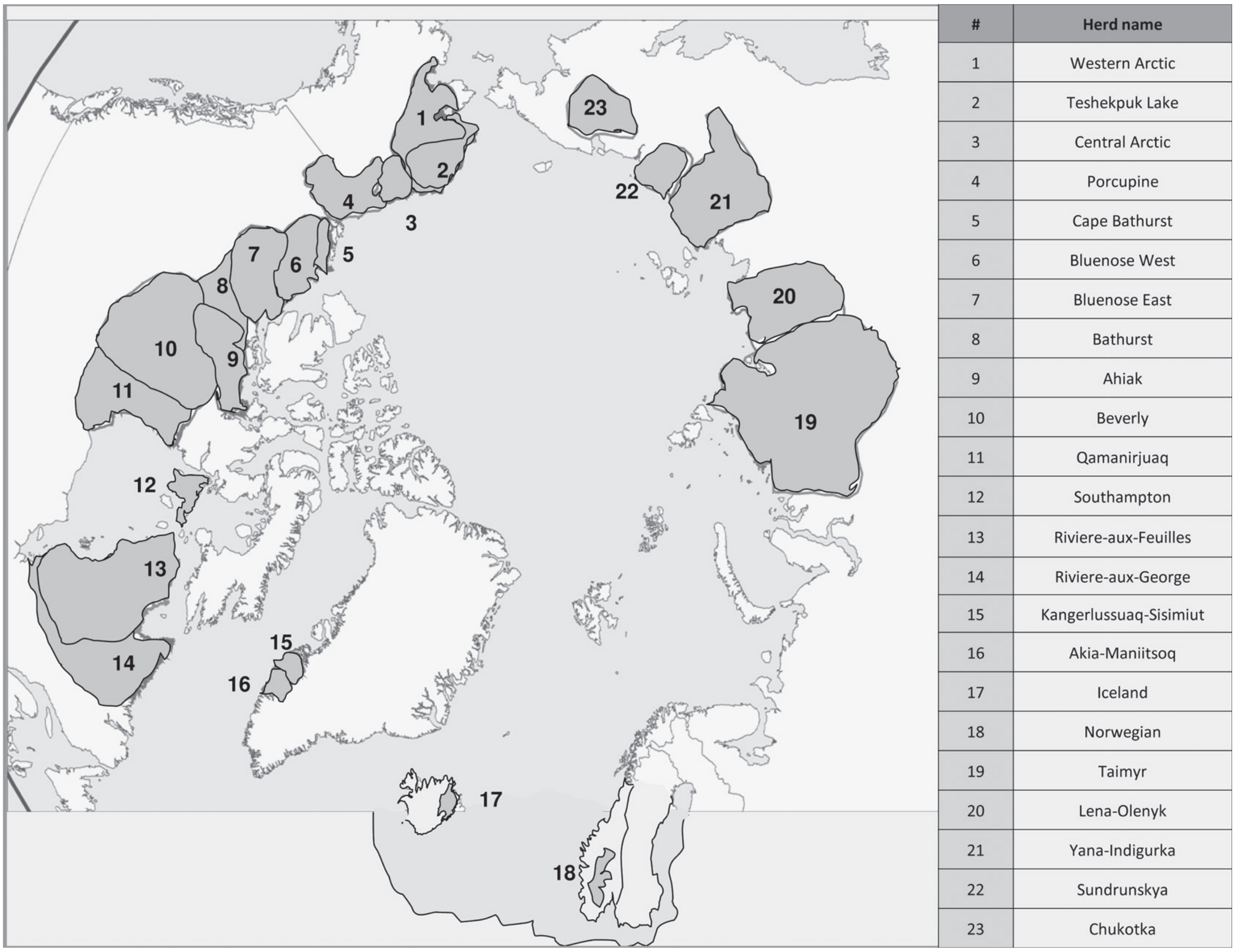
Supported by the International Polar Year (IPY) initiative (2005 – 2011), the CARMA network developed a standardized approach to circumpolar monitoring for *Rangifer*. In this paper we provide an overview of the CARMA protocols and the implementation of these protocols for CARMA-supported sampling of circumpolar caribou herds. We summarize the IPY sampling efforts, and discuss the successes and hurdles to such broad monitoring activities.

Methods

Standardized protocol development and application

Starting in 2006, a sub-group of the CARMA network developed: (i) a manual that described monitoring indicators, the rationale for each indicator, and the relevant literature, and (ii) standardized sampling protocols for collection and measurement of each indicator. The selection of indicators and sampling protocols were developed in consultation with network collaborators. These were based on published and unpublished literature and experiences from previous and ongoing *Rangifer* monitoring programs. Indicators were selected to provide data on age, diseases, physiological condition, short and long-term nutritional status, and maternal investment in reproductive fitness. Some indicators were compatible with, and could be used in energy-protein and body frame size models that CARMA developed to assess and predict *Rangifer* responses to environmental changes (Murphy *et al.,* 2000; Russell *et al.,* 2005).

Recognizing the value and constraints of both hunter and scientist-based monitoring, CARMA initially developed two levels of protocols. The Level 1 protocol was developed for subsistence hunters or community-based sampling and provided basic information on age, frame size, body condition, and a few pathogens. The Level 2 protocol was more com-



#	Herd name
1	Western Arctic
2	Teshekpuk Lake
3	Central Arctic
4	Porcupine
5	Cape Bathurst
6	Bluenose West
7	Bluenose East
8	Bathurst
9	Ahiak
10	Beverly
11	Qamanirjuaq
12	Southampton
13	Riviere-aux-Feuilles
14	Riviere-aux-George
15	Kangerlussuaq-Sisimiut
16	Akia-Maniitsoq
17	Iceland
18	Norwegian
19	Taimyr
20	Lena-Olenyk
21	Yana-Indigurka
22	Sundrunskya
23	Chukotka

Fig. 1. Distribution of migratory *Rangifer tarandus* herds.

prehensive and typically required at least one well-trained individual to collect and process samples. Additionally, this level required partnerships with commercial or research laboratories to do further analyses on samples. Level 2 included additional indicators for body condition and morphometrics that could be used to quantitatively predict whole body fat and protein reserves, and in-depth sampling methods for assessing contaminants and determining pathogen presence, abundance, and/or exposure.

In the protocols, the sampling procedures and data requirements for each indicator were described and references were provided for further details on more specialized procedures such as tooth sectioning or parasite isolation from tissues. The manual and protocols included less detail on pathogens and pathogen sampling than was applied in the CARMA-supported field collections so these procedures are described in greater detail in this paper.

CARMA network partners and participants were encouraged to follow the protocols when handling *Rangifer* during IPY-funded collections. The initial selection of herds and the level of sampling was discussed at the annual CARMA meetings and aimed to identify representative reference herds around the Arctic (Fig. 1). The final selection of the reference herds was determined by the priorities and support of management agencies, communities, and co-management boards. Further, the sampling intensity for each individual herd was influenced by the availability and engagement of regional biologists or managers and by the goals of individual research and monitoring projects.

Formal feedback on protocol implementation was gathered during the 2010 annual CARMA workshop after network members had an opportunity to use the protocols in field collections from 2007 – 2009. Network members (63 people representing 11 government agencies, 12 organizations, eight universities, and six countries) divided into three groups (community people, wildlife managers, and university researchers) and had guided open discussions on whether the protocols were useful and how content and accessibility could be improved.

Data management

All participating herd biologists or managers were asked to submit their data in a Microsoft Excel® datasheet format and all entries were imported into a newly created CARMA Excel datasheet. When possible, an automatic importation function (*e.g.*, "vlookup") was used to decrease human error. When needed, original datasheets were consulted to validate the data. In the absence of field datasheets, herd biologists or managers were asked to verify and validate their data once it was imported into the common database. For most collections, metadata records, documenting the date, location, number of animals, and types of data collected, were created by individual researchers and archived with the ArcticNet data portal *(www.arcticnet.ulaval.ca).*

Results

Standardized protocol development

Draft protocols were discussed with network members before and during IPY at the CARMA annual meetings as well as between meetings as needed (*e.g.*, manager/researcher requiring specific advice on sampling). Protocols were refined periodically based on these discussions and ongoing feedback and in some cases, based on the results of research studies. For example, the initial protocol for *Besnoitia* included sampling multiple tissues, but based on results from this sampling it was determined that metatarsal skin alone was a suitable index of infection (Ducrocq *et al.*, 2012). The manual and field protocols are available on the CARMA web site (CARMA, 2011a; b; c). An early version of the protocols was also translated into Russian to be tested by CARMA's Russian collaborators. Indicators and samples collected, the information gained, and some of the potential sources of error associated with collections are summarized in Table 1.

Field collections and application of the protocols

Between March 2007 and September 2010, CARMA collaborators collected body condition and health samples from at least 1206 individual caribou and reindeer from 12 North American, two Greenlandic, and two Russian herds (Fig.1; Table 2). The Chukotka herd was also sampled as part of the IPY initiative but data from this herd were not available at the time of writing.

Sampling intensity for each herd varied depending on the objectives of the monitoring project. Although only two levels of protocols were initially designed (see methods), in practice, sampling could be categorized into four overlapping, categories.

Level 1 (n = 152 animals)

This was the simplest type of sampling, required the least amount of training and data recording, and was generally done by subsistence hunters. Often hunters were paid for samples and data submission. Hunters were asked to collect information on a minimum number of indicators that were selected to provide important basic data while interfering minimally with meat handling practices (Table 1). Hunters filled in tags (Fig. 2a) to identify

Table 1. Levels of sampling done by CARMA with the samples and data gathered (indicators) at each level, the information that each indicator provides on animal body condition, disease, or contaminants, and observed or anticipated sources of error or sample quality compromise associated with collection or transport of the data or samples [modified from CARMA health and body condition manual (CARMA 2013].

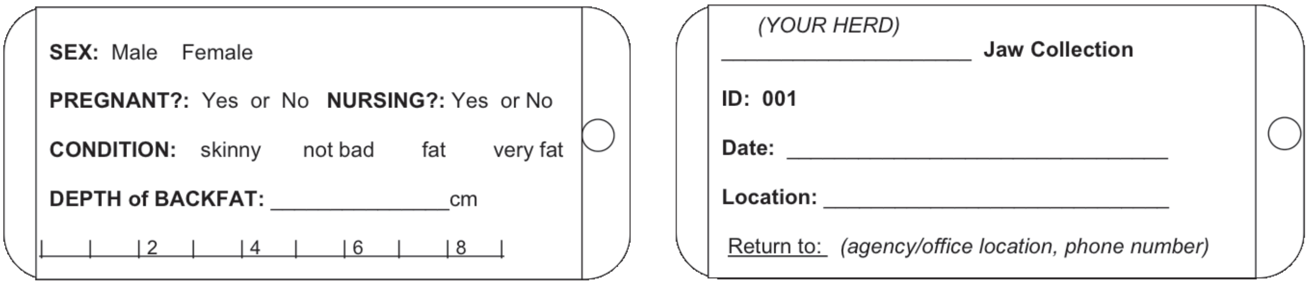
Level of sampling and information and samples collected	**Animal information (condition, morphology, physiology)**	**Other health measures (pathogens, abnormalities, contaminants)**	**Possible sources of sampling error in the field or processes that may compromise sample quality during storage and transportation**
Level 1 (subsistence hunter collection):			
Animal information recorded on identification tags or data sheets (Fig. 2)	Location Field age Sex Back fat depth	Abnormalities	General – Data are not recorded; tags not appropriately attached to samples; data from multiple animals on one tag/datasheet; writing implement smudges when wet.
	Pregnancy		Pregnancy – observer may be unable to detect early pregnancy.
	Qualitative fatness assessment		Hunter experience may influence their assessment of age and fatness.[1] Assessment of condition is done in context of the season (*i.e.,* 'good' in spring has less backfat then 'good' in fall).
Mandible			
Morphometrics	Body size	Lumpy jaw[3]	Measured jaw lengths may differ if measured wet versus dry.
Marrow fat	Body condition		
Tooth eruption and tooth wear	Age class	Dental disease	
Incisor I cementum	Age		Incisor root damaged during extraction.
Molars	Enamel hypoplasia (previous stress[2])		
Metatarsus (+/- foot)			
Hair and skin	Cortisol levels	Skin – *Besnoitia tarandi* cysts, Filarioidea microfilaria Foot rot (*Fusobacterium* sp.[4])	
Morphometrics	Body size		Small tarsal bone not removed and included in metatarsal bone length measurement. Measured length may differ if measured wet versus dry.
Marrow	Fat		Desiccation of bone may influence results.

Level of sampling and information and samples collected	**Animal information (condition, morphology, physiology)**	**Other health measures (pathogens, abnormalities, contaminants)**	**Possible sources of sampling error in the field or processes that may compromise sample quality during storage and transportation**
Level 2 (trained hunter and community hunts). All of the above plus the following:			
CARMA body size and condition measures (detailed in protocols)	Body condition Size (body, leg, foot lengths, height, chest girth)		Units of measure not recorded in data sheets; tail length included in body length; hoof length may vary due to wear; units of measurement not recorded.
Feces*	Diet Nitrogen balance Pregnancy Hormones	Macro (helminths) and microparasites (protozoa, bacteria, viruses) that are shed in feces Chronic wasting disease[5]	Freeze/thaw cycles compromise recovery of macro and micro parasites and hormone stability. Deep freeze (-80°C) may reduce recovery of some parasite eggs.
Blood on filter paper	Hormones (e.g., progesterone, cortisol)	Serology for various pathogens Blood-borne pathogens may be detectable by PCR in good quality samples	Incomplete saturation of filter paper strips. Inadequate desiccation of filter papers. Freeze-thaw cycles, excessive heat will compromise antibody, hormone, and DNA quality.
Kidney	Riney kidney fat – body condition	Contaminants[6]	Riney fat not cut in the field. Storage temperature for contaminants needs to be -80°C
Liver		Contaminants *Taenia* sp., *Fascioloides, Echinococcus*	Storage temperature for contaminants needs to be -80°C. Detection varies with search effort, must standardize.
Testicles	Confirmation of sex	*Brucella suis* [7] *Besnoitia tarandi*[8]	Repeat freeze-thaw cycles will reduce viability of *Brucella.*
Hide		*Hypoderma tarandi*	Inaccurate counts of warbles if heavy infestation where larvae are layered.
Head/Pharyngeal sacs		*Cephenemyia trompe*	Season of collection influences detectability. Larvae may be overlooked if small or in the nasal turbinates.
Gastrocnemius/Cranial Crural	Protein DNA	*Taenia* sp., *Sarcocystis* sp., *Toxoplasma*	Identification of the correct muscles and associated tendons; search effort for parasite cysts will influence results.

Level of sampling and information and samples collected	Animal information (condition, morphology, physiology)	Other health measures (pathogens, abnormalities, contaminants)	Possible sources of sampling error in the field or processes that may compromise sample quality during storage and transportation
Level 2 detailed (scientific collection). All of the above, plus:			
Heart		*Taenia* sp., *Sarcocystis* sp., *Toxoplasma*	Search effort for parasite cysts will influence results.
Abomasum and first 3 meters of small intestine		Gastrointestinal parasites of abomasum and proximal small intestine	First 3 meters estimated; parasites will migrate in guts post mortem and time since death may influence recovery.
Ileum and ileo-caecal and mesenteric lymph nodes		*Mycobacterium avium paratuberculosis*	Unable to find lymph nodes; freeze-thaw cycles will reduce organism viability.
Obex and retro-pharyngeal lymph nodes		Chronic wasting disease[9]	Incomplete removal of brainstem and obex.
Urine	Nitrogen balance		Contamination with blood or other material.
Serum*	Serum chemistry, trace vitamins, and minerals	Pathogen serology	Contamination with rumen contents, hair or dirt if not careful when cutting jugular or other blood vessels. Hemolysis.
Whole blood*	Nitrogen balance DNA	Blood-borne helminths, protozoa, bacteria	Contamination with rumen contents, hair or dirt if not careful when cutting jugular or other blood vessels.
Rumen content	Diet		
Milk sample	Fat and protein content		Inexperience in expressing milk may result in poor recovery.
Ovaries	Current pregnancy status / reproductive history[10]		Difficult to find; lymph nodes mistaken for ovaries.

* Collected during live sampling; [1](Loison *et al.*, 2001); [2](Wu *et al.*, 2012); [3](Wobeser, 2001); [4](Handeland *et al.*, 2010); [5](Haley *et al.*, 2009); [6] (Elkin & Bethke, 1995; Robillard *et al.*, 2002); [7](Tessaro & Forbes, 1986); [8] (Wobeser, 1976); [9](Williams, 2005); [10](Cuyler & Østergaard, 2005)

Figure 2 a



SEX: Male Female

PREGNANT?: Yes or No **NURSING?:** Yes or No

CONDITION: skinny not bad fat very fat

DEPTH of BACKFAT: ______________cm

(YOUR HERD)
____________________ **Jaw Collection**

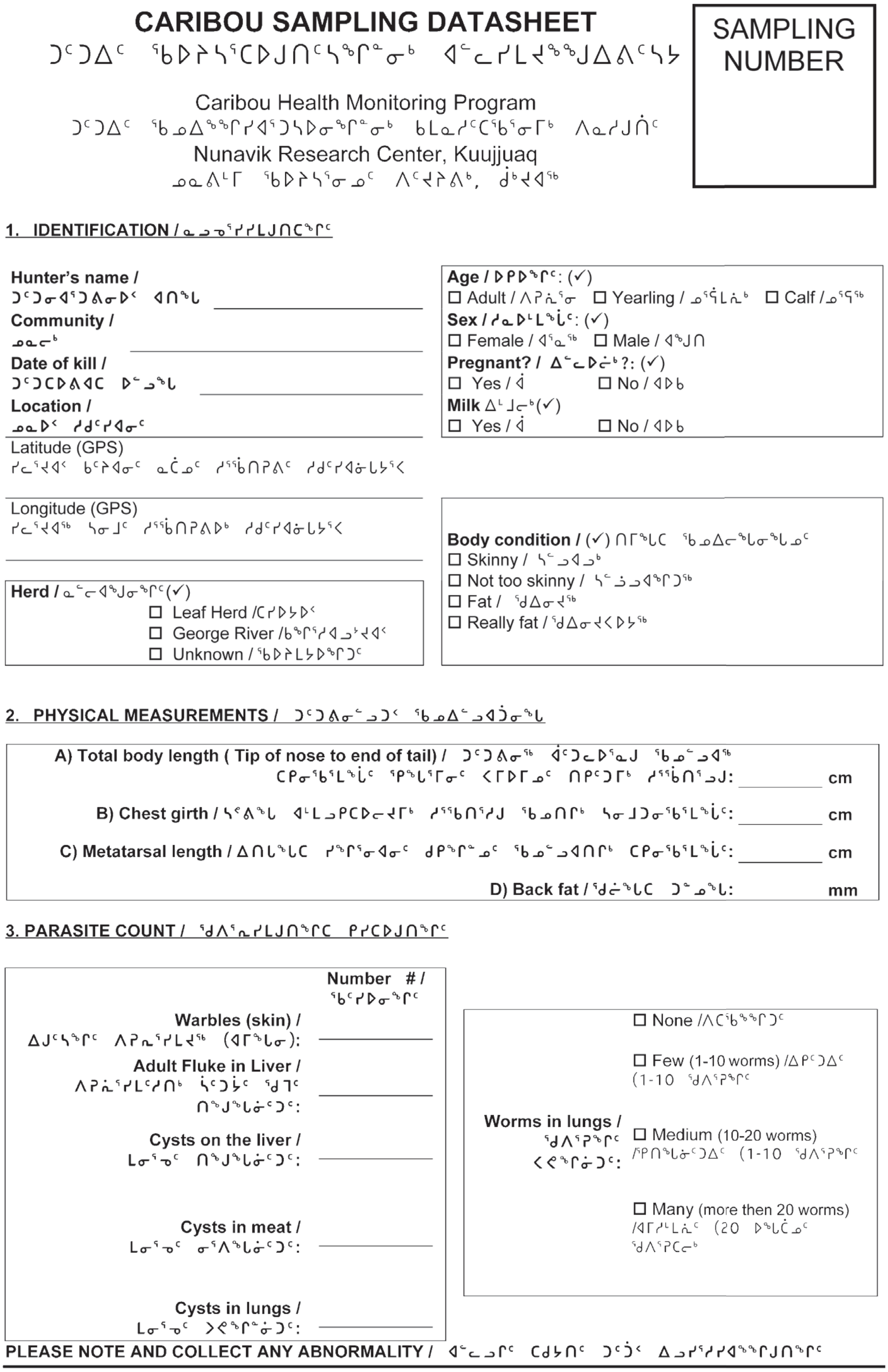
ID: 001

Date: ____________________________

Location: __________________________

Return to: *(agency/office location, phone number)*

Figure 2 b



CARIBOU SAMPLING DATASHEET
ᑐᑦᑐᐃᑦ ᖃᐅᔨᓴᕐᑕᐅᒍᑎᑦᓴᖏᓐᓂᒃ ᐊᓪᓚᓯᒪᔪᖅᖅᒍᐃᕕᑦᓴᔭ

Caribou Health Monitoring Program
ᑐᑦᑐᐃᑦ ᖃᓄᐃᖅᖏᓯᐊᕐᑐᓴᐅᓂᖏᓐᓂᒃ ᑲᒪᓇᓯᑦᑕᕐᑲᕐᓂᒥᒃ ᐱᓇᓱᒍᑏᑦ
Nunavik Research Center, Kuujjuaq
ᓄᓇᕕᒻᒥ ᖃᐅᔨᓴᕐᓂᓄᑦ ᐱᑦᔪᔨᕕᒃ, ᑰᒃᔪᐊᖅ

SAMPLING NUMBER

1. IDENTIFICATION / ᓇᓗᓈᕐᓯᓯᒪᒍᑎᑕᖏᑦ

Hunter's name / ᑐᑦᑐᓂᐊᕐᑐᕕᓂᐅᑉ ᐊᑎᖓ ______________________
Community / ᓄᓇᓕᒃ ______________________
Date of kill / ᑐᑦᑐᑕᐅᕕᐊᑕ ᐅᓪᓗᖓ ______________________
Location / ᓄᓇᐅᑉ ᓱᑯᑦᓯᐊᓂᑦ ______________________
Latitude (GPS) ᓯᓚᕐᔪᐊᑉ ᑲᑦᔨᐊᓂᑦ ᓇᑖᓄᑦ ᓱᕐᕐᑳᑎᕈᕕᑦ ᓱᑯᑦᓯᐊᓃᒐᔭᕐᐸ ______________________
Longitude (GPS) ᓯᓚᕐᔪᐊᖅ ᓴᓂᒧᑦ ᓱᕐᕐᑳᑎᕈᕕᐅᒃ ᓱᑯᑦᓯᐊᓃᒐᔭᕐᐸ ______________________

Herd / ᓇᓪᓕᐊᖑᓂᖏᑦ(✓)
☐ Leaf Herd /ᑕᓯᐅᔭᐅᑉ
☐ George River /ᑲᖏᕐᓱᐊᓗᒃᔪᐊᑉ
☐ Unknown / ᖃᐅᔨᒪᔭᐅᖏᑐᑦ

Age / ᐅᑭᐅᖏᑦ: (✓)
☐ Adult / ᐱᕈᓃᕐᓂ ☐ Yearling / ᓄᕐᖀᒪᓃᒃ ☐ Calf / ᓄᕐᖁᖅ
Sex / ᓱᓇᐅᒻᒪᖔᑦ: (✓)
☐ Female / ᐊᕐᓇᖅ ☐ Male / ᐊᖑᑎ
Pregnant? / ᐃᓪᓚᐅᓃᒃ?: (✓)
☐ Yes / ᐋ ☐ No / ᐊᐅᑲ
Milk ᐃᒻᒧᓖᒃ(✓)
☐ Yes / ᐋ ☐ No / ᐊᐅᑲ

Body condition / (✓) ᑎᒥᖓᑕ ᖃᓄᐃᓕᖓᓂᖓᓄᑦ
☐ Skinny / ᓴᓪᓗᐊᓗᒃ
☐ Not too skinny / ᓴᓪᓘᓗᐊᖏᑐᖅ
☐ Fat / ᖁᐃᓂᔪᖅ
☐ Really fat / ᖁᐃᓂᔪᐸᐅᔭᖅ

2. PHYSICAL MEASUREMENTS / ᑐᑦᑐᕕᓂᓪᓗᑉ ᖃᓄᐃᓪᓗᐊᑑᓂᖓ

A) Total body length (Tip of nose to end of tail) / ᑐᑦᑐᕕᓂᖅ ᐋᑦᑐᓚᐅᕐᓇᒍ ᖃᓄᓪᓗᐊᖅ ᑕᑭᓂᕐᑲᕐᒪᖔᑦ ᕿᖓᕐᒥᓂᑦ ᐸᒥᐅᒥᓄᑦ ᑎᑭᑦᑐᒥᒃ ᓲᕐᕐᑳᑎᕐᓗᒍ: ________ **cm**
B) Chest girth / ᓴᕝᕕᖓ ᐊᒻᒪᓗᑭᑕᐅᓕᔪᒥᒃ ᓱᕐᕐᑲᑎᕐᓯᒍ ᖃᓄᑎᒃ ᓴᓂᒧᑐᓂᕐᑲᕐᒪᖔᑦ: ________ **cm**
C) Metatarsal length / ᐃᑎᒐᖓᑕ ᓯᖏᕐᓂᐊᓂᑦ ᑯᑭᖏᓐᓄᑦ ᖃᓄᓪᓗᐊᑎᒥᒃ ᑕᑭᓂᕐᑲᕐᒪᖔᑦ: ________ **cm**
D) Back fat / ᖁᓃᖓᑕ ᑐᓐᓄᖓ: **mm**

3. PARASITE COUNT / ᖁᐱᕐᓇᓯᒪᒍᑎᖏᑕ ᑭᓯᑕᐅᒍᑎᖏᑦ

Number # / ᖃᑦᓯᐅᓂᖏᑦ

Warbles (skin) / ᐃᒍᑦᓴᖏᑦ ᐱᕈᕐᓯᒪᔪᖅ (ᐊᒥᖓᓂ): ________
Adult Fluke in Liver / ᐱᕈᓃᕐᓯᒪᑦᓯᑎᒃ ᕿᑦᑐᔫᑦ ᖁᒣᑦ ᑎᖑᖓᓃᑦᑐᑦ: ________
Cysts on the liver / ᒪᓂᕐᓄᑦ ᑎᖑᖓᓃᑦᑐᑦ: ________
Cysts in meat / ᒪᓂᕐᓄᑦ ᓂᕐᐱᖓᓃᑦᑐᑦ: ________
Cysts in lungs / ᒪᓂᕐᓄᑦ ᐳᕙᖏᓐᓃᑦᑐᑦ: ________

Worms in lungs / ᖁᐱᕐᕈᖏᑦ ᐸᕙᖏᓃᑦᑐᑦ:
☐ None /ᐱᑕᖃᙱᑐᑦ
☐ Few (1-10 worms) /ᐃᑭᑦᑐᐃᑦ (1-10 ᖁᐱᕐᕈᖏᑦ
☐ Medium (10-20 worms) /ᒥᑭᑎᓪᓗᓂᑦᑐᐃᑦ (1-10 ᖁᐱᕐᕈᖏᑦ
☐ Many (more then 20 worms) /ᐊᒥᓱᒻᒪᓃᑦ (20 ᐅᖓᑖᓄᑦ ᖁᐱᕐᕈᑕᓕᒃ

PLEASE NOTE AND COLLECT ANY ABNORMALITY / ᐊᓪᓚᓗᒋᑦ ᑕᑯᔭᑎᑦ ᑐᑦᑑᑉ ᐃᓗᓯᕐᓯᓯᐊᖅᖏᒍᑎᖏᑦ

Fig. 2. Field data recording: (a) example of a Level 1 double-sided Tyvek® tag used to collect basic information and identify samples, and (b) example of a Level 2 field datasheet used for the hunter-based sampling activities in Nunavik, Québec.

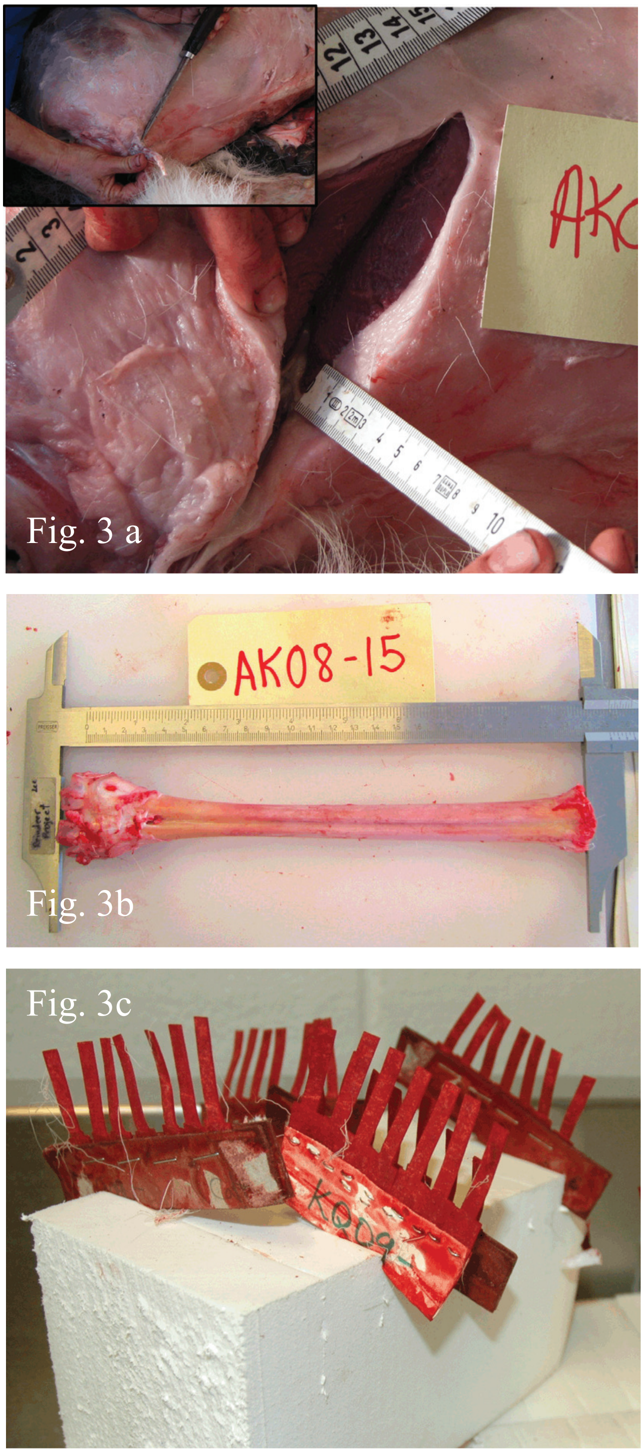


Fig. 3 Examples of samples taken and data acquired from caribou sampling activities. (a) Back fat depth is measured at a 45° angle from the base of the tail (see inset) with a measuring tape, a ruler, or the tag (see Fig. 2a). Backfat is measured at the deepest point. (Image credit: Wendy Nixon). (b) Metatarsal bone length measured by calipers (Image credit: Wendy Nixon). (c) Drying of Nobuto® filter paper strips that have been dipped in clean blood (Image credit: Karin Orsel).

location hunted, sex, pregnancy, a subjective assessment of body condition, and observations of any abnormalities. Back fat depth (Parker, 1981; Fig. 3a) was measured using the provided tags or rulers. Samples submitted included the lower jaw and metatarsal bone (Fig. 3b) with skin on, and core data collected included age (using tooth cementum annuli) (Miller, 1974; Matson, 1981), pregnancy, body condition (back fat, marrow fat, and a subjective hunter assessment; Lyver & Gunn, 2004), body size (jaw and metatarsal length; Parker, 1981; Couturier *et al.*, 2010), and presence and intensity of the parasite *Besnoitia tarandi* in metatarsal skin (Ducrocq *et al.*, 2012). *Besnoitia* cysts were quantified in the lab on gross observation using categories of number of cysts per square centimeter [0 = no cysts; 1 = very occasional (< 4 cysts/cm^2); 2 = easily observed (4-10 cysts/cm^2); 3 = high density (> 10 cysts/cm^2)]. Histological examination, however, was determined to be much more reliable both for presence and intensity (Ducrocq *et al.*, 2012).

Level 1 provided key samples and data that could be analyzed to evaluate and compare body condition and size, demographics (recognizing that hunter-based sampling may have strong, but identifiable biases), and *Besnoitia* infection status.

Level 2 sampling (n = 166 animals)

This type of sampling was typically done by trained subsistence hunters or during community organized hunts that were attended by biologists and veterinarians. Samples included those from Level I as well as blood collected on filter paper (Fig. 3c; Curry *et al.*, 2011), the left kidney with fat, liver, and feces (Table 1). Semi-quantitative observations were done for grossly visible parasites and, depending on the collection, some body measurements were recorded (Fig. 2b; CARMA, 2013).

Collection of blood samples allowed for more in-depth examination of pathogen diver-

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sity (Curry *et al.,* 2011). Blood was extracted from the filter paper in the lab and tested for exposure to a variety of pathogens using standard serological tests (Curry *et al.,* 2011). Additional testing for *Trypanosoma* spp. and *Setaria* sp. was attempted using polymerase chain reaction-based approaches. DNA was extracted from filter papers using a modified PurGene® (Qiagen, Canada) extraction procedure for compromised blood samples. Primers described by Noyes *et al.* (1999) were used to test for the presence of *Trypanosoma* and the primers described by Laaksonen *et al.* (2009) were used to test for the presence of *Setaria.* Unfortunately, DNA quality was variable and DNA was not reliably extracted from the filter papers (D. Schock & S. Kutz, unpubl. data). Thus, although the presence of *Trypanosoma* and *Setaria* was confirmed from some samples (Kutz *et al.,* 2012), the absence of these parasites could not be confidently established and quantitative assessment was not possible. It was, however, possible to sequence several *Trypanosoma* detected in caribou samples and compare genotypes among caribou herds and among ungulate species (D. Schock & S. Kutz, unpubl. data).

The kidney with surrounding fat was collected and used to establish the Riney kidney fat index (Riney, 1955). The kidney, together with a piece of liver, usually the caudate lobe, was frozen for future analyses. Feces were stored frozen and later tested for helminth and protozoan parasites and the bacteria *Mycobacterium avium paratuberculosis* (Forde *et al.,* 2012).

Observations of grossly visible parasites including: *Echinococcus granulosus* (cysts in lungs; Rausch, 2003), *Fascioloides* (Choquette *et al.,* 1970), *Taenia hydatigena* (cysticerci in liver) and *Taenia* spp. (cysticerci in skeletal or cardiac muscle; Thomas, 1996), *Hypoderma tarandi* (scars or bot larvae on underside of skin; Cuyler *et al.,* 2012), *Cephenemya trompe* (larval bots in the pharynx, examined once head was removed; Cuyler *et al.,* 2012), *Besnoitia tarandi* (cysts in metatarsal skin and bulbar conjunctiva; Ducrocq *et al.,* 2012), and any other abnormalities were also recorded on the field datasheet. The lungs and heart were examined grossly for parasites, and the gastrocnemius and/or the cranial crural muscles [referred to elsewhere as 'peroneus' (Allaye Chan-McLeod *et al.,* 1995); Fig. 4a, b], were collected and, later in the laboratory, weighed and examined grossly for *Taenia* cysticerci and *Sarcocystis* cysts. The presence of rumen flukes (*e.g.*, *Paramphistomum cervi* or *P. skrjabini;* Nikander & Saari, 2007), were noted only in the Russian Lena-Olenek herd.

This dataset provided substantially more quantitative and semi-quantitative information on body condition and pathogen presence, exposure, and abundance than Level 1. It therefore allowed for more extensive investigations of pathogen/parasite occurrence in relation to body condition and other physiological and demographic parameters.

Level 2 detailed sampling (n = 835 animals)

These were the most comprehensive collections. They were planned hunts done primarily for scientific purposes and/or management monitoring, and in some cases were done in collaboration with community hunts. Typically, these collections involved local hunters and the meat was given to the local communities. In addition to the data and samples listed above, more detailed body size measurements and in-depth pathogen assessments were done (Table 1; CARMA, 2013). Project specific sampling was done during these collections, varied across herds, and samples may have included sampling: the brain stem ventral to the obex and/or lymph nodes for chronic wasting disease (Williams, 2005), conjunctiva and skin from the rostrum, scrotum, and inner thigh for *Besnoitia* research (Ducrocq *et al.,* 2012), abomasum and first three meters of small intestine for gastrointestinal parasite analyses, a 5-10cm section of ileum and mesenteric lymph nodes for *Mycobacterium avium paratuberculosis,* ovaries (Dauphiné, 1978), various tissue sections in formalin for describing normal histology, weight of the rumen contents (Huot, 1989), and additional body measurements. Whole (EDTA tubes) and clotted blood for serum were often collected in addition to blood on filter paper. Lungs were dissected along the bronchi and major bronchioles to assess *Dictyocaulus* infection (Anderson & Prestwood, 1981) and livers were cut into 1 cm slices to examine for the liver fluke, *Fascioloides magna*, and *Taenia hydatigena* cysts (Lankester & Luttich, 1988). Transverse cuts through both the heart and the gastrocnemius muscle, and sometimes additional muscles, were done to examine for cysts of *Taenia* spp. Feces, muscle, fur, and urine were also collected for protein/nitrogen balance studies (Barboza & Parker, 2006).

This sampling level allowed for more detailed quantitative analyses of a broader range of pathogens. The broader dataset on body condition indices (back fat, kidney fat, marrow fat, empty and dressed body weight, and cranial crural muscles) provided the data necessary to predict body condition, fat, and protein (Ring-

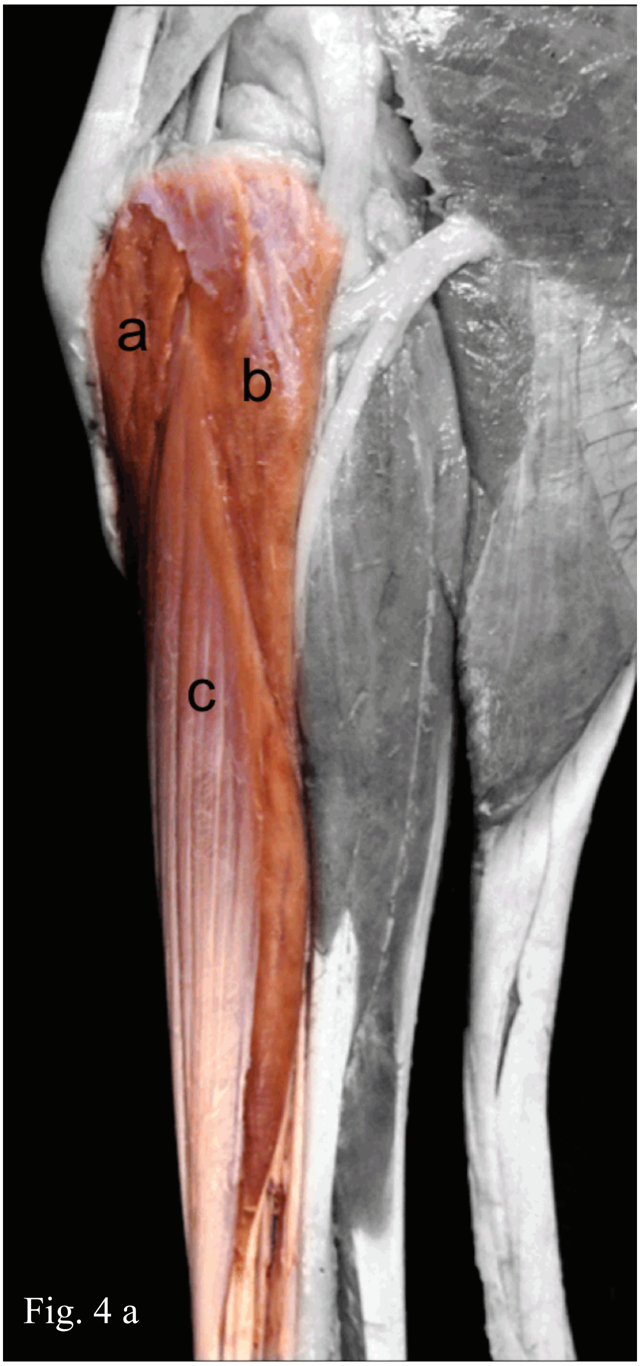

Fig. 4 a

Fig. 4 b

Fig. 4 The cranial crural muscle group of a caribou leg. (a) Lateral view of the hind leg showing the anatomical position of the cranial crural muscle group (spotlit). The cranial crural muscle group is composed of four muscles: a. the cranial tibial (tibialis cranialis), b. the long fibular (fibularis longus), and c. the fused third fibular (fibularis tertius) and long extensor (extensor digitorum longus). When consulting standard veterinary anatomy texts, the reader should be aware that fibularis and peroneus are synonyms and that both have been widely used; it has recently been agreed that fibularis should be the standard term (b) The cranial crural muscles separated. The long extensor and the third fibular are extensively fused at their upper ends and the two tendons of the long extensor are visible behind the third fibular. There is no need to separate these muscles in the field as they should all be weighed together. (Image credits: *Rangifer* Anatomy Project, P. Flood and C. Muelling).

Table 2. Collections done by the CARMA network during International Polar Year activities, 2007 – 2010. The types of collections, data, samples, and the number of animals from each herd that were sampled (n) are indicated. Blank cells indicate that no animals were sampled.

		Herds[1]															
	n	WAH	TSH	POR	CB	TUK	BW	BE	BT	B/A	B/Q	RAF	RG	AM	KS	TAY	LO
Type of collection																	
Level 1 (Subsistence)	152			30			44	10		38							30
Level 2 (Trained hunter/community)	166			25		13	32				56	40					
Level 2 (Scientific)	835	20	21				20		150	73		176	178	47	50	100	
Live-capture	53				15	6	32										
Total	1206	20	21	55	15	19	128	10	150	111	56	216	178	47	50	100	
Body condition indicators																	
Full body mass	474	17	19	20					145	72				41	40	100	20
Dressed body mass	250								67	62				41	40	40	
Total body length	456	18	21	25			19		146	72		35		41	40	39	
Chest girth	440	9	21	15			20		147	73		35		41	40	39	
Hind leg	105	9	20	10						22		19				25	
Metatarsal length	237	18	21	25						73		19		41	40		
Metatarsal bone length	761	14	10	53		9	74		147	27	48	263		46	40	30	
Total jaw bone length	365	9		13			28		49	59	51	39		44	48	25	
Cranial crural m. weight	255			13			19		95	22				41	40	25	
Gastrocnemius weight	103			13					50							40	
Hunter fatness estimate	421	19	20	48		13	12	8	90	12	53	34		27	50	35	
Back fat depth	586	18	17	45		12	81		139	102	25	22		45	50	30	
Kidney fat index[2]	451			23			26		139	76	50			47	40	30	20
Metatarsal bone marrow fat	549	1	18	49		13	73		141	36	49	36		47	50	36	
Jaw marrow fat	70		19											41		10	
Femur marrow fat	39	20	19														
Gross parasitism																	
Hypoderma tarandi	721	20	19	11			11		150	68	2	134	145	47	50	34	30
Cephenemyia trompe	444	20	21	11			10		149	73				47	50	34	29
Taenia hydatigena	748	20	20	12			17		150	72		148	144	45	50	40	30
Fascioloides magna	761	20	21	11			19		150	88		148	144	40	50	40	30
Echinococcus granulosus	701	20	20	11			17		149	49		148	145	47	50	15	30
Dictyocaulus	310	20	20	0			17		109	49				45	50		
Taenia spp.	738	20	20	11			20		150	73		148	145	41	50	30	30
Setaria spp.	359			6			19		149	88				47	50		
Fecal parasitism																	
Fecal parasitology (Wisconsin flotation)	572			23	15	6	52		109	36	14	91	90	47	49	40	
Fecals parasitology (Baermann)	566			23	14	6	49		106	35	23	91	89	46	49	35	
Giardia and *Cryptosporidium* (IFA)	522	10		22	15	6	51		97	35	39	58	58	47	49	35	
Pathogen presence or exposure																	
Besnoitia tarandi[3]	1045			13			25		129			636	146	46	50		
Mycobacterium avium paratuberculosis[4]	544			23	22	28	52	20	109	36	46	61	60	47	40		
Brucella spp [5]	690			33			80		147			147	144	49	50	40	
Neospora caninum [5]	647			33			80		145			147	143	49	50		
West Nile Virus[5]	645			33			79		144			147	143	49	50		
Toxoplasma gondii[5]	684			33			77		144			148	143	49	50	40	
Bovine Herpes Virus-1[5]	636			32			76		143			147	139	49	50		
Bovine Respiratory Syncytial Virus[5]	636			32			76		143			147	139	49	50		
Para-influenza-3[5]	636			32			76		143			147	139	49	50		
Pestivirus[5]	636			27			57		144			147	144	49	68		

		Herds[1]															
	n	WAH	TSH	POR	CB	TUK	BW	BE	BT	B/A	B/Q	RAF	RG	AM	KS	TAY	LO
Contaminants and heavy metals																	
Kidney[6]	187								40					47	40	30	30
Liver[6]	110													40	40	30	
Muscle[6]	80								10					20	20	30	
Liver – PFOS[7]	55								5					10	10	30	

[1] Herds: WAH=Western Arctic, TSH=Teshekpuk, POR=Porcupine, CB=Cape Bathurst, TUK= Tuktoyaktuk Peninsula, BW=Bluenose West, BE=Bluenose East, BT=Bathurst, B/A=Beverly/Ahiak, B/Q=Beverly-Qamanirjuaq, RAF= Rivière-aux-Feuilles, RG= Rivière-George, AM= Akia-Maniitsoq, KS= Kangerlussuaq-Sisimiut, TAY=Taymir, and LO=Lena-Olenek; [2] Russian collections had potentially an alternate measure of kidney fat that has yet to be calibrated to the KFI in the CARMA protocols; [3] From metatarsal skin sections; [4] From ileum and ileo-caecal lymph nodes, and fecal and serum samples; [5] From blood sampled by filter papers or serum samples; [6] A combination of any of the following heavy metals: Aluminum, Antimony, Arsenic, Barium, Beryllium, Bismuth, Cadmium, Calcium, Cesium, Chromium, Cobalt, Copper, Gallium, Iron, Lanthanum, Lead, Lithium, Magnesium, Manganese, Mercury, Molybdenum, Nickel, Palladium, Platinum, Potassium, Rubidium, Selenium, Silver, Strontium, Thallium, Tin, Uranium, Vanadium, and Zinc; [7] Perfluorooctane Sulfonate.

berg *et al.,* 1981b; Huot & Goudreault, 1985; Adamczewski *et al.,* 1987a; b; Taillon *et al.,* 2011) for comparison with reproductive status and individual and group abundance and diversity of pathogens and pathogen exposure.

Live animal sampling (n = 53 animals)

This sampling was done by biologists as a routine component of animal handling during radio-collaring. Data collected included an estimate of age (based on body size and tooth wear), body condition, feces, blood, and hair. Depending on season these animals were allotted in reproductive category (pregnant/non-pregnant, lactating/non-lactating, and in some cases weaning status). Body condition and reproduction status could also be examined relative to status of pathogens that could be grossly observed (*e.g.*, *Besnoitia*), or those that could be assessed serologically or through fecal examination.

CARMA network feedback and protocol evaluation

Overall, the protocols were well received and consistently applied. Formal feedback from the network, gathered during three breakout groups in 2010, was positive with helpful comments to improve the protocols and their utility (CARMA, 2013). Network members emphasized the importance of the written protocols and visual resources for sampling and laboratory processing (images and directions in the protocols, sampling video, anatomy website), and suggested that these be translated into the languages of the various user groups around the Arctic. Aboriginal members of the network indicated that their view and observations of *Rangifer* 'health' may differ from the scientific perspectives and that a community-developed protocol that incorporated this view for health monitoring would be valuable. Additional suggestions included: development of advanced protocols for non-lethal sampling, improved guidance on necessary sample sizes for power analysis, and additional guidance on interpretation of results.

Network members did identify some issues with interpretation of the field protocols (Table 1). One concern was identifying the anatomical limits of the 'peroneus' muscle, the weight of which could be used to estimate body protein. This is actually a complex of four muscles, better referred to as the cranial crural muscles. They are found on the front (cranial aspect) of the shinbone (tibia) in the crus or crural part of the leg; hence the name. They form a coherent functional group that, taken together, flex the hock and extend the digit. The cranial crural muscles are relatively fibrous when compared with other muscles and are not much sought after for food. They can be quickly and consistently removed as a group, and weighed without further dissection. To clarify the muscle group to be sampled, a series of images were produced by dissections of reindeer and caribou by Peter Flood, Christoph Muelling, and others (Fig. 4a, b). A step-by-step instructional Microsoft PowerPoint® presentation describing the appropriate anatomy and sampling process for this muscle group was also produced and is available at *http://www.ucalgary.ca/caribou/Sampling.html).*

Other potential sources of error were identified when datasets were merged into a single database and it became clear through this process that, to avoid errors, complete descriptions of any modifications from the standard protocols needed to be included with all metadata files. For example, the total body length measured for some herds included the tail whereas the tail was excluded for others. Occasionally confusion arose over the units of measurement. In some cases these were not specified on datasheets. For example, back fat could be measured as 1 mm versus 1 cm, or 1 inch versus 1 cm. This was particularly an issue if measurement tapes included both imperial and metric units. Some datasets contained blank spaces that left ambiguity as to whether an animal had been examined for that specific indicator/pathogen and zero observed, or if the indicator had not been examined at all. Errors, blanks, or unspecified deviations from protocols were rectified by clarification from contributors.

Sampling and data collection for pathogens were generally standardized and consistent, however, some clarifications to the protocols were also needed to prevent loss of data. For example, quantification for serology requires that filter paper strips are fully saturated. In some cases the blood strips were only partially saturated and results were not quantifiable or comparable. Quantification of *Taenia* cysticerci differed between protocols. Specifically, at level 2, the hunters were only asked if they saw any *Taenia* cysts in the carcass. In contrast, in the detailed level 2 protocol, two horizontal cuts were made through the gastrocnemius and the heart, and more intensive sampling (*e.g.*, more cuts or organs examined) was done for some herds. Once samples for pathogens were collected, standardization in laboratory analyses was achieved by ensuring similar storage of samples among herds and the use of the same laboratory for each pathogen. This limited sources of error and variability among herds.

In response to network member feedback and queries, early in the process two products were developed to supplement the protocols and provide additional visual resources. A DVD on basic and advanced sampling was produced in cooperation with hunters from the communities of Fort Good Hope and Colville Lake, Northwest Territories, Canada (CARMA, 2009). The video was distributed to CARMA network collaborators, participating communities, as well as local and regional wildlife groups and agencies. It is available on the CARMA website (CARMA, 2013). In addition, the *Rangifer* Anatomy Project was initiated to produce high quality anatomical images of *Rangifer* and instructional PowerPoint presentations to supplement the protocols and serve as teaching aids for a variety of audiences. The site is located at the University of Calgary *(http://www.ucalgary.ca/caribou/index.html),* and can be accessed through the CARMA website.

Discussion

CARMA's goal was to bring together the vast knowledge and expertise in the circumarctic so that sampling protocols incorporating local knowledge and science could be developed and provide reliable and comparable information on health and body condition of *Rangifer* across its range. This goal was achieved in that a series of standardized protocols with increasing levels of complexity were developed and implemented in varying degrees across herds. However, as evidenced by feedback in 2010, the CARMA protocols did not adequately incorporate the aboriginal views on health. Protocol development and implementation was a learning process, with challenges and limitations being identified and addressed on an ongoing basis. An advantage of CARMA's web-based approach was that it was responsive to identified needs. The protocols and manual were easily updated and clarified and supplementary resources could be made readily available on the web. This process is ongoing.

An innovative aspect was the partnership with infectious disease specialists, which brought a strong emphasis on monitoring techniques for pathogens and non-infectious diseases. This emphasis was in recognition of two points. Firstly, body condition and disease monitoring has not always been well integrated in the past. Body condition and disease are intricately linked and neither can be well understood in isolation from the other. Secondly, the emergence of new diseases, re-emergence of old, and pathogen spill-over among wildlife, domestic animals, and people has highlighted the need for effective and responsive wildlife disease monitoring and surveillance systems today (Kuiken *et al.*, 2005).

It became clear during IPY that when dealing with a species that is widespread across vast geographical remote regions there is a need to develop flexible yet scientifically robust strategies for data collection. Although the protocols were originally developed as two defined categories of collection, they were adapted to meet the monitoring, logistical, research, and human resource constraints for each herd. This highlights the importance of providing a prioritized continuum that allows useful and comparable information to be gathered even at the most basic level. A frequent limitation to monitoring is inadequate sample size, which can limit statistical power to detect trends in concomitant variables (Nickerson & Brunell, 1998). Importantly, "required" sample size will vary for each specific pathogen depending on its expected prevalence. Several authors emphasize that working with hunters to monitor health and condition of harvested caribou increases sample sizes (Kofinas *et al.,* 2003; Lyver and Gunn, 2004; Brook *et al.,* 2009; Curry, 2010). The protocols accommodate the trade-off between the greater sample sizes available from hunter-based collections compared to the more detailed dataset from fewer animals collected during intensive monitoring and research projects. For example, a level 1 collection (*i.e.*, the jaw, metatarsal, and other basic data) can provide reliable information on late term pregnancy, age and sex structure, body size, body condition, and abundance of a few select pathogens. Additional information on other physiological parameters can also be gathered from hair (*e.g.*, cortisol and other hormones; Ashley *et al.,* 2011) while teeth can be examined for dental enamel hypoplasia as an indicator of past stress events (Wu *et al.,* 2012). Hunter-based sampling was done prior to IPY across a number of jurisdictions and has, in general, broad acceptance amongst hunters (Gunn *et al.,* 2005; Brook *et al.,* 2009). With widespread hunter involvement, this type of monitoring could provide ongoing, reliable and affordable information on several indices of *Rangifer* health across a broad geographic range and across seasons (Kofinas *et al.,* 2003). As the complexity of data and sample collection increases (*e.g.*, through assisted community hunts and biologist led collections) more information is acquired and more comparisons are possible. However, such comprehensive monitoring is typically done on fewer herds and over shorter time frames, limiting the number of herds or years that can be compared. Thus, establishing a 'bare minimum' guideline for sample and data collection facilitates ongoing broader (geographically and temporally) comparisons at a basic level. Importantly, all sampling strategies have biases specific to that strategy, for example, subsistence hunters may select for animals in better condition, and these must be considered in the final data analyses.

The additional spatiotemporal sampling and increased sample sizes offered by hunter-based sampling may be particularly valuable for disease surveillance as it can increase detection of infectious diseases that may be present at a low prevalence and/or with a patchy or clumped distribution (Zhang *et al.*, 2011). Working with hunters, however, offers important advantages beyond simply increasing sample size. These include two-way exchange of knowledge, recognition of different ways to assess health and condition, and growing mutual respect and trust developed from working together (Kofinas *et al.*, 2002; Brook *et al.*, 2009; Lyver & Nation, 2010).

Emerging diseases are of increasing global importance, and are also of considerable concern for northern ungulates (Kuiken *et al.*, 2003; Kutz *et al.,* 2004; Laaksonen *et al.,* 2010). Northward range expansion of domestic and wild animals, together with various pathogen vectors, may allow introduction of new pathogens into *Rangifer* range. Concurrently, a warming climate is removing some of the environmental constraints on existing arctic pathogens, resulting in range expansion of pathogens and emergence of disease (Hoberg *et al.,* 2008; Kutz *et al.,* 2009; Laaksonen *et al.,* 2010). Establishing comprehensive baselines for pathogen biodiversity and abundance is necessary if we are to detect emergence of new pathogens and disease syndromes. Sophisticated molecular diagnostic techniques in combination with the advanced level of collection described here allows, through either physical recovery of the organism or serological evidence of exposure, detection of all known pathogens of *Rangifer* and the quantification/semi-quantification of most.

Monitoring programs must ensure accuracy and precision (*i.e.*, repeatability) of measurements. The CARMA network approached this through the detailed descriptions in the manual and protocols, hosting specific training sessions, and having biologists participate in collections before running collections of their own. Use of the same diagnostic and research laboratories in many cases helped to ensure standardization.

Preservation of biological specimens and data are critical elements of any monitoring program. In particular, appropriately preserved physical specimens allow for investigation of new questions, or re-evaluation of old studies, as new information and techniques emerge (Hoberg *et al.,* 2008). Archiving of CARMA tissues was limited to the capacity of individual researchers and, for those specific caribou populations, provides a rich source of information. Importantly, to ensure appropriate preservation of these materials, and to promote increased rates of archiving in the future, centralized, permanently curated facilities for specimen and DNA archiving are essential.

Data management in large-scale monitoring programs adds complexity at several levels. Data ownership and management is an important issue within any large network. Metadata for CARMA were managed through the ArcticNet portal, however, mechanisms for long-term storage of full datasets in a central database and subsequent access remains to be established. To date, CARMA has drafted a data policy as well as data submission and request forms. Ideally, in the spirit of open access that IPY promoted, all data, tissues, and specimens should be made available to the broader scientific community after a reasonable time period. Sharing data also raises questions of format and transcription errors when inputting from data sheets to a database. Exploring the use of a computer program for data mining would reduce transcription errors while transferring data from original databases. CARMA continues to discuss the development and implementation of solutions.

Conclusion

CARMA is a network of *Rangifer* users, biologists, scientists, and managers who have worked together to implement the first broad scale standardized body condition and health assessment of migratory caribou. To our knowledge, this is the most comprehensive sampling effort for a terrestrial mammal across its entire range. Network members were actively and willingly engaged in improving and implementing protocols, and although some challenges were encountered, network members philosophically supported the protocols and applied them. This was a clear reflection of the shared vision of how a comparative approach across space (herds) and time (seasons and years) can substantially improve our current scientific knowledge of *Rangifer* and enables us to monitor and detect changes more rapidly and confidently.

Flexibility and adaptability of sampling regimes are essential to ensure that specific research and monitoring objectives can be addressed. Still, there was consensus that a minimum of standard data and samples should be done for all herds (*e.g.*, Level 1). Easily accessible and up to date protocols provided key resources for field and laboratory personnel. Where needed, these were supplemented with additional visual aids such as the anatomy resources and sampling DVD. Ongoing in-person and practical training and mentorship of hunters and researchers are essential to ensure accurate and consistent sample and data collection into the future. Engagement of communities and local hunters is critical to promote knowledge sharing and mutually acceptable approaches for long-term health and condition monitoring in caribou.

Data management and ownership, together with authorship on scientific articles, had the potential to become problematic. However, a transparent approach and regular in-person discussions enabled the growth and maintenance of productive collaborations among network members. To be effective, the data and knowledge gained must be transferred back to communities and caribou managers in a timely and effective manner. This has been done directly within the CARMA network as community representatives are key and active network participants, and has also occurred in many regions in the form of co-management meetings. Additional researcher-initiated posters and community meetings presenting preliminary results have occurred. Unfortunately, as the funding from IPY has ended, there are severe financial constraints to returning to communities in person to present final results, many of which will be coming out over the next several years.

Rangifer populations co-exist with increasing human abundance around the Arctic, and like many parts of the world, anthropogenic modifications in parallel with natural phenomenon are influencing their health and sustainability (Balmford *et al.,* 2003). To understand these complex systems, wildlife research needs to focus on large-scale monitoring activities (Pollock *et al.,* 2002; Balmford & Bond, 2005). The CARMA network and the standardized sampling protocols that it has developed, will hopefully enable a more comprehensive understanding of migratory *Rangifer* populations, and provide new insights into the resilience of these animals under the current regime of environmental, social, and political change.

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Brief Communication

Observation of Arctic island barren-ground caribou (*Rangifer tarandus groenlandicus)* migratory movement delay due to human induced sea-ice breaking

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Abstract: The seasonal migration of the Dolphin and Union caribou (*Rangifer tarandus groenlandicus)* herd between Victoria Island and the mainland (Nunavut/Northwest Territories, Canada) relies on the formation of sea-ice that connects the Island to the mainland from late-October to early-June. During an aerial survey of the Dolphin and Union caribou herd in October 2007 on southern Victoria Island, Nunavut, Canada, we documented the short-term effects of the artificial maintenance of an open water channel in the sea-ice on caribou migratory movements during staging along the coast.

Key words: caribou; ice breaking; migration; movements; *Rangifer*; sea-ice.

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Introduction

In the context of observed changes in sea-ice conditions and the increase in maritime shipping activity, the impact of sea-ice breaking is of great concern for the movements and viability of Arctic wildlife and particularly of caribou (*Rangifer tarandus)* populations using sea-ice to migrate from one island to the other (Miller *et al.,* 1977; Miller & Gunn, 1978) or between an island and the mainland (Gunn *et al.,* 1997; Poole *et al.,* 2010). Connectivity between islands or between islands and the mainland is key to the viability of Arctic island caribou populations (Miller et al. 2005; 2007), and maritime traffic and sea-ice breaking has been a concern for local communities who rely on caribou and other wildlife for subsistence, cultural, and economic activities. While caribou are good "swimmers", and water-crossings between islands have been documented (Miller, 1995), it is unlikely that it happens when the distance to cross is more than a few kilometres or when air temperature is well below freezing. Although the documentation of sea-ice crossings (Miller *et al.,* 2005; Poole *et al.,* 2010) and seasonal habitat and forage use (Hughes, 2006) provide some support for the assumptions of demographic effects if the connectivity between the islands or between islands and the mainland is affected (COSEWIC, 2004), there is limited information on verified and quantified impacts on caribou movements due to anthropogenic alteration of the sea-ice.

Changes in sea-ice conditions and increased maritime traffic were part of the justification for the Dolphin and Union caribou (R. t. groen-

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landicus) herd to be currently listed as Special Concern under the Species at Risk Act (Canada Gazette, 2011).The Dolphin and Union herd is somewhat unique as its annual range encompasses Victoria Island for calving, post-calving, and rut, and the mainland as its main winter ground (Gunn & Nishi, 1998; Poole *et al.,* 2010).This implies annual migrations (fall and spring) across the sea-ice between Victoria Island and the mainland. The fall migration across the sea-ice is preceded by a staging period (Poole *et al.,* 2010) when caribou aggregate along the south coast of Victoria Island waiting for the ice to form between the Island and the mainland. As soon as the ice is strong enough, the crossing happens very rapidly with, in some areas, pauses on islands along the way (Poole *et al.,* 2010). We describe the short-term effects on the Dolphin and Union caribou herd migratory movements due to the temporary maintenance of an open channel through the sea-ice in the fall 2007, at Cambridge Bay, Victoria Island, Nunavut.

Materials and Methods

The reported observation was made during an aerial stratified strip transect survey of the Dolphin and Union caribou herd using a Helio-Courier H-295 on wheel skis at an altitude of 100 meters above the ground at a speed of 160 km/h between October 24 and October 30, 2007, following the method described in Nishi & Gunn (2004). Transects extended 500 m on each side of the plane with the 500-meter line indicated by a streamer below each wing. The calculation of caribou density was estimated using the caribou counted within the 500 m strip on each side of the plane. A channel in the sea-ice between Cambridge Bay (N69.13, W105.07) and the open water at the mouth of the bay was maintained artificially by a tug-boat every 12 hours between October 20 and October 28, 2007, in an attempt to allow the tug-boat and barges to navigate back to their base in the Northwest Territories. All aerial caribou observations on and off transects and during ferry flights were recorded and areas on both side of the channel were flown during the ice breaking period on October 26, 2007, and the day after it stopped. We used a Mann–Whitney U test (corrected for large sample size; Sokal & Rohlf, 1995) for a local change in caribou density observed on the east of the channel between October 26 and October 29, 2007. Flight tracks were divided into 1 km transect segments to obtain a number of 1 km^2 blocks with associated caribou density as the sample unit.

We calculated average daily temperature based on the hourly temperature data for Cambridge Bay from October 20 to October 31, 2007 (Environment Canada, 2008).

Results

On October 26, 2007, we estimated 1000 stationary caribou on the point of land and on the ice at the west edge of the open water channel maintained by the barge tug-boat (Fig. 1). Only a few caribou trails were observed on the east side of the boat channel and very few caribou were observed on the land or ice east of the channel (Fig. 2).

On October 29, 2007, we flew over the area again (Fig. 2) and only observed three caribou near the frozen boat channel but many caribou trails were going east past the frozen channel. Only a few caribou were still present on the point west of the channel. Local density of caribou on the land east of the channel increased significantly between October 26, 2007, and October 29, 2007, (ts = 3.284, P = 0.001) from 5.9 ± 7.0 caribou/100 km^2 on October 26, 2007, to 54.2 ± 92.7 caribou/100 km^2 on October 29, 2007 (Fig. 2).

Average daily air temperature started to fall between October 23 and 24, 2007, and remained between -14°C and -18°C from October 24 to October 31, 2007 (Fig. 3).

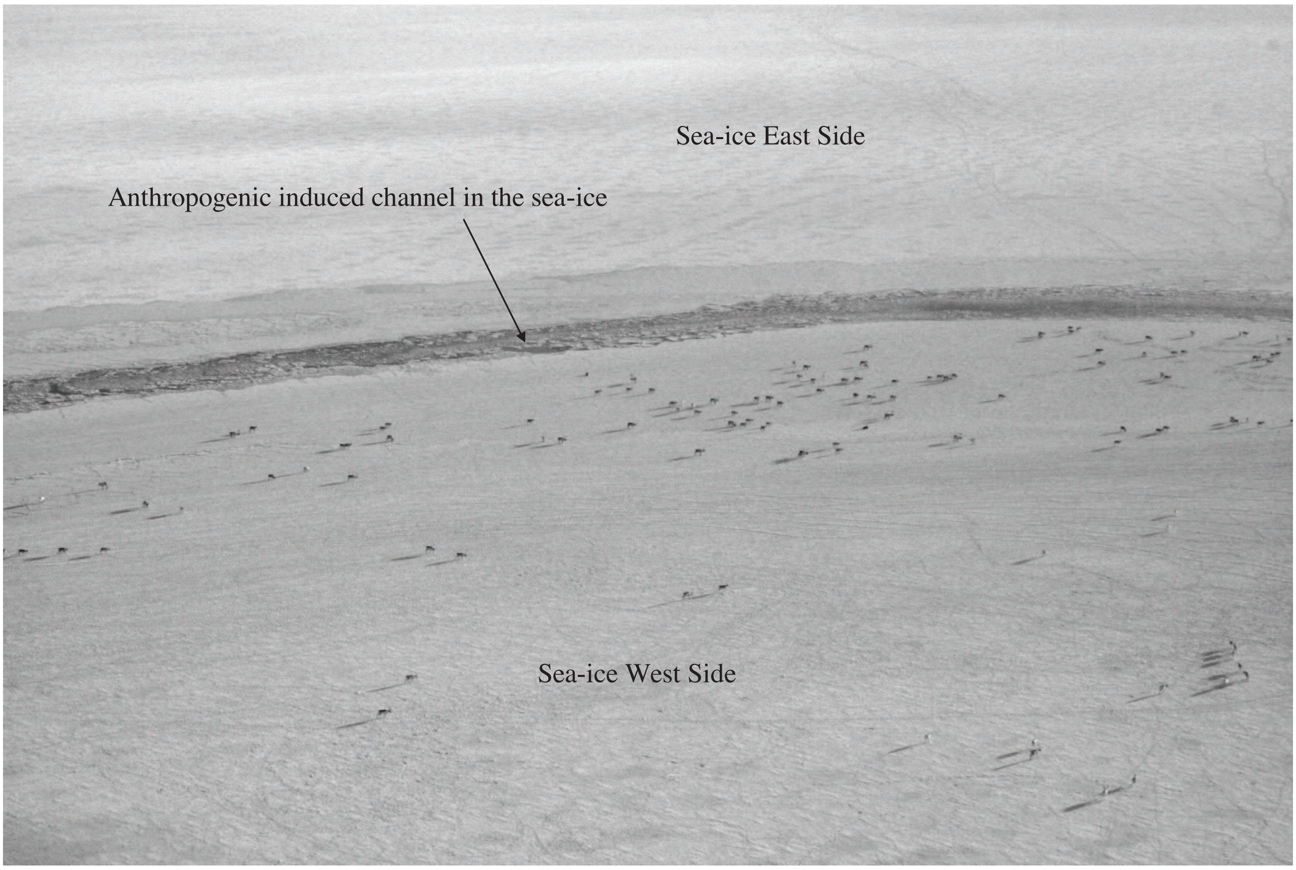

Figure 1. Dolphin and Union caribou aggregated on the west edge of an anthropogenic ice breaking channel near Cambridge Bay, NU, on October 26, 2007.

During our two flights over the channel we did not observe any caribou drowned or dead on the ice or in the water. It is possible that between ice breaking, the ice had the time to get thick enough for some caribou to cross the channel but it remained a barrier to the movement eastward for the majority of the caribou at least between October 22 and October 26, 2007. During the whole survey, we observed only three caribou that had drowned through the ice along the coast.

Discussion

It seems that overall, because it was stopped rapidly, the impact of the ice breaking resulted only in a few days delay for caribou movements. The cold temperature likely decreased the risk of caribou breaking through thin ice. It is unknown what level of impact it had on the condition of the animals or would have had if the ice breaking had resulted in a wider channel or if temperature would had been milder (*i.e.*, ice does not thicken as fast).

Caribou do die naturally from going through the ice (Miller & Gunn, 1986) and, during the fall migration of the Dolphin and Union caribou herd, reports of individuals that have gone through the sea-ice are common. The low number observed during our survey compared to the 1997 survey (Nishi & Gunn, 2004) and local observations could be related to the late but very fast ice formation this year which reduced the likelihood of caribou breaking through the ice during staging. In recent years, hunters also reported several animals on the mainland with a thick coat of ice on their fur indicating that a portion of the animals going through the ice managed to survive at least to reach the mainland. The impact of falling through the ice on the overall survival rate is unknown. Never-

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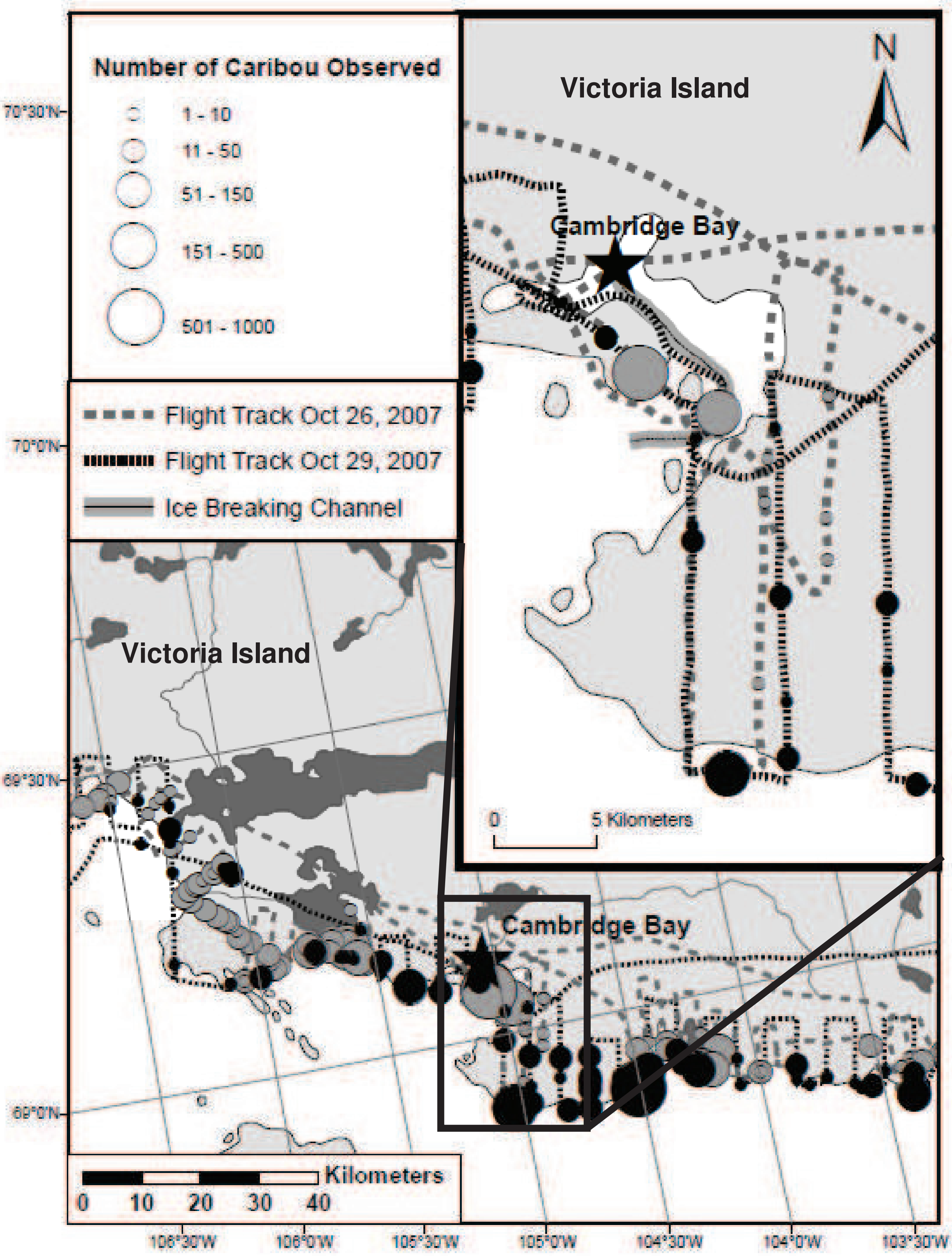

Figure 2. Flight tracks around Cambridge Bay, NU, on October 26 and 29, 2007, with caribou observations (dot size is proportional to caribou group size; October 26 in grey and October 29 in black) and the approximate barge tug-boat ice breaking channel.

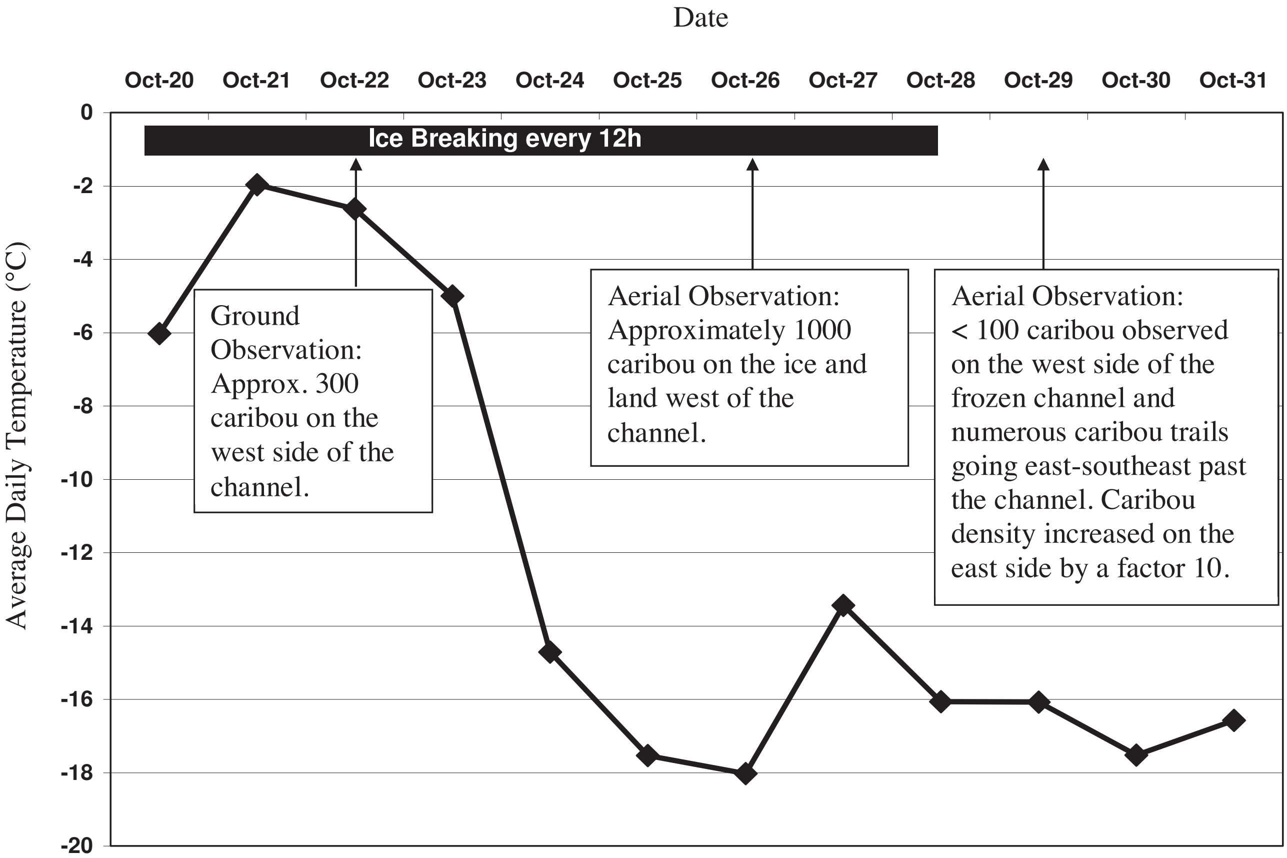

Figure 3. Chronology of observed events and average daily air temperature at Cambridge Bay, NU (from Environment Canada, 2008).

theless, in the spring of 2010, over two dozen caribou carcasses were found frozen in the ice around small islands near the mainland as well as on the islands, obviously having died shortly after getting out of the water (Allen Niptanatiak and Dustin Fredlund, pers. comm.; M. Dumond, pers. obs.). During the survey in October 2007 the ice was just starting to form away from the shore and this may explain the small number of caribou that were observed drowned (the number likely increased as they ventured further out).

Fall migration and winter are the periods of highest natural mortality in adult females of the Dolphin and Union caribou herd (Poole *et al.,* 2010). While causes are likely linked to individuals falling through the ice and predation, data are insufficient to rank natural mortality causes. The addition of new stress during the fall migration through anthropogenic disruption of the sea-ice formation could have cumulative impacts on the herd with unknown consequences for the herd survival.

Past low density and seclusion on Victoria Island resulted in the Dolphin and Union caribou herd being the most genetically differentiated of the barren-ground caribou herds (Zittlau, 2004) and may become more isolated as sea-ice formation is delayed, preventing migration to the mainland (Poole *et al.,* 2010). Increased disturbance of the sea-ice could precipitate the isolation of the Dolphin and Union caribou herd on Victoria Island. Forage availability, quality, and biomass are generally lower on Victoria Island than on the mainland (Hughes, 2006), and the seclusion of the herd on Victoria Island would likely result in a lower viable population size.

The increasing number of resource extraction projects and the domestic needs of Arctic communities are calling for an increase in maritime traffic and increasing pressure to extend the shipping season through ice breaking. Already some resource extraction projects are proposing year around maritime shipping. Information on the short- and long-term effects of sea-ice breaking on wildlife (including terrestrial wildlife) is crucial for the management of this source of impact, especially in the context of observed and projected changes in sea-ice formation and degeneration.

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13th Arctic Ungulate Conference
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Brief Communication

Moose *(Alces alces)* population size and density in the Inuvik Region of the Northwest Territories, Canada

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Abstract: Responding to community concerns, the Gwich'in Renewable Resources Board (GRRB) and the Government of the Northwest Territories Department of Environment and Natural Resources (ENR) conducted an aerial moose *(Alces alces)* survey in the Inuvik region of the Northwest Territories, Canada to estimate moose density and distribution. The survey was flown in March 2011 and a random stratified sample design was used. Local knowledge was incorporated in to the stratification of survey cells. Moose density in survey blocks ranged from 9.66 moose/100 km^2 in the Ikhil Pipeline block to 0 in the Peel River block with a coarse overall moose density 2.24 moose/100 km^2. Densities found were low but within expected range for the species in this region of North America based on past surveys.

Key words: *Alces alces;* moose; Northwest Territories; population; survey.

Rangifer, 33, Special Issue No. 21, 2013: 123–128

Introduction

In the Gwich'in Settlement Area (GSA) and the adjacent Inuvialuit Settlement Region (ISR) of the Northwest Territories (NWT), Canada (Fig.1), management of moose (Alces alces) populations is primarily the responsibility of co-management boards and of the territorial government. The Gwich'in Renewable Resources Board (GRRB) is the co-management board for wildlife in the GSA while the Wildlife Management Advisory Council (Northwest Territories) [WMAC (NWT)] is the co-management board for wildlife in the ISR.

A 2006 survey in GSA reported low and declining (from past surveys between 1980 and 2000) moose densities ranging from 0 to 3.78 moose/100 km^2 (Lambert, 2006). Moose in the ISR have not been surveyed since the mid-1980's (Jingfors & Kutny, 1989). Current local knowledge suggests moose numbers have increased in the Mackenzie Delta (Fig. 1). Local barren-ground caribou *(Rangifer tarandus groenlandicus)* herd population numbers have been low, in particular the Cape Bathurst herd, resulting in a harvest closure of that herd in 2007. So, despite a perception of current healthy moose population the declining caribou led to community concerns about impacts of possible predators and harvesters switching to moose. In response to these concerns and in order to inform possible management decisions by GRRB and WMAC (NWT), we conducted

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an aerial survey of moose population density and distribution in northwest NWT in March 2011.

Methods

Population estimates followed the stratified random sampling methods of Gasaway *et al.,* (1986) and was analyzed using the GeoSpatial Population Estimator Software (Delong, 2006). Density estimates (moose/100 km^2) were calculated for each of the eight survey blocks based on total number of moose sighted in selected cells (# moose/area surveyed * 100).

We held workshops with local Renewable Resources Councils (RRCs) and Hunter and Trapper Committees (HTCs) to define the survey region and map areas of expected high and low moose density in the survey period. Local experts were used as a cost-effective way to stratify the survey area while ensuring the involvement of local indigenous people.

Aerial survey methods generally followed those described by Kellie & Delong (2006). The survey region was divided into 2’ latitude by 5’ longitude (~ 4 km by 4 km) cells using ArcGIS 9.2 (ESRI, 2006). The cells were then stratified as high or low moose density using workshop classifications and habitat data. If cells were not classified as high or low moose density during the workshops vegetation cover maps were used to classify the cells. Area classified as open deciduous, closed deciduous, shrubs, wet herbaceous, emergent vegetation were considered areas were high density of moose would be expected. Areas with closed needle leaf, open needle leaf, non-vegetated soil, sparse vegetation or rock/gravel were considered low moose density classes.

Eight areas of interest were identified based on past surveys and input from HTCs, RRCs, GRRB, and WMAC (NWT) (Fig. 1). Survey blocks in the GSA were similar to the 2006 survey with slight modifications to the Peel River and Arctic Red River survey blocks based on input from the RRCs. This includes adjusting the Arctic Red River survey block (Fig. 1) into a discontinuous block with a portion near the community of Tsiigehtchic and a portion up river. New survey blocks were created in the ISR. Cells were randomly selected for surveying, with 2% of cell selections made manually to ensure good coverage, such as the inclusion of both high and low survey blocks. Surveyed cells represented 4,368 km^2 and 16.1 % of all survey blocks (Table 1).

We flew the survey in March 2011 using a Cessna 206 and Cessna 185 fixed-wing aircraft. Surveyed cells were to be covered in their entirely with the intent to detect all moose in the cell. Search intensity varied by block based on block vegetation cover; heavily treed areas were covered more intensely than open/tundra areas. Snow tracks were circled to determine if the moose was still located in the block. A pilot, navigator, and two observers spotted and classified moose inside each selected cell and noted any moose observed outside selected cells. Locations were recorded using GPS. Wolves and other wildlife observations inside or outside selected cells were also recorded.

Results & Discussion

Survey flights were conducted from March 16-24, 2011 with a total of 61.9 hours flown. We observed a total of 168 moose: 79 within surveyed grid cells and 89 moose outside surveyed cells. We classified 63% of observed moose: 40 cows, 32 calves and 34 bulls, resulting in bull to cow and calf to cow ratios of 85:100 and 80:100, respectively. Composition estimates may be biased as the presence of calves aided in classification of cows, such that cows without calves may have been more often unclassified than by chance. Other wildlife observed included; moose, 33 sheep, 38 wolves, and five caribou.

A total area of 3519 km^2 was surveyed

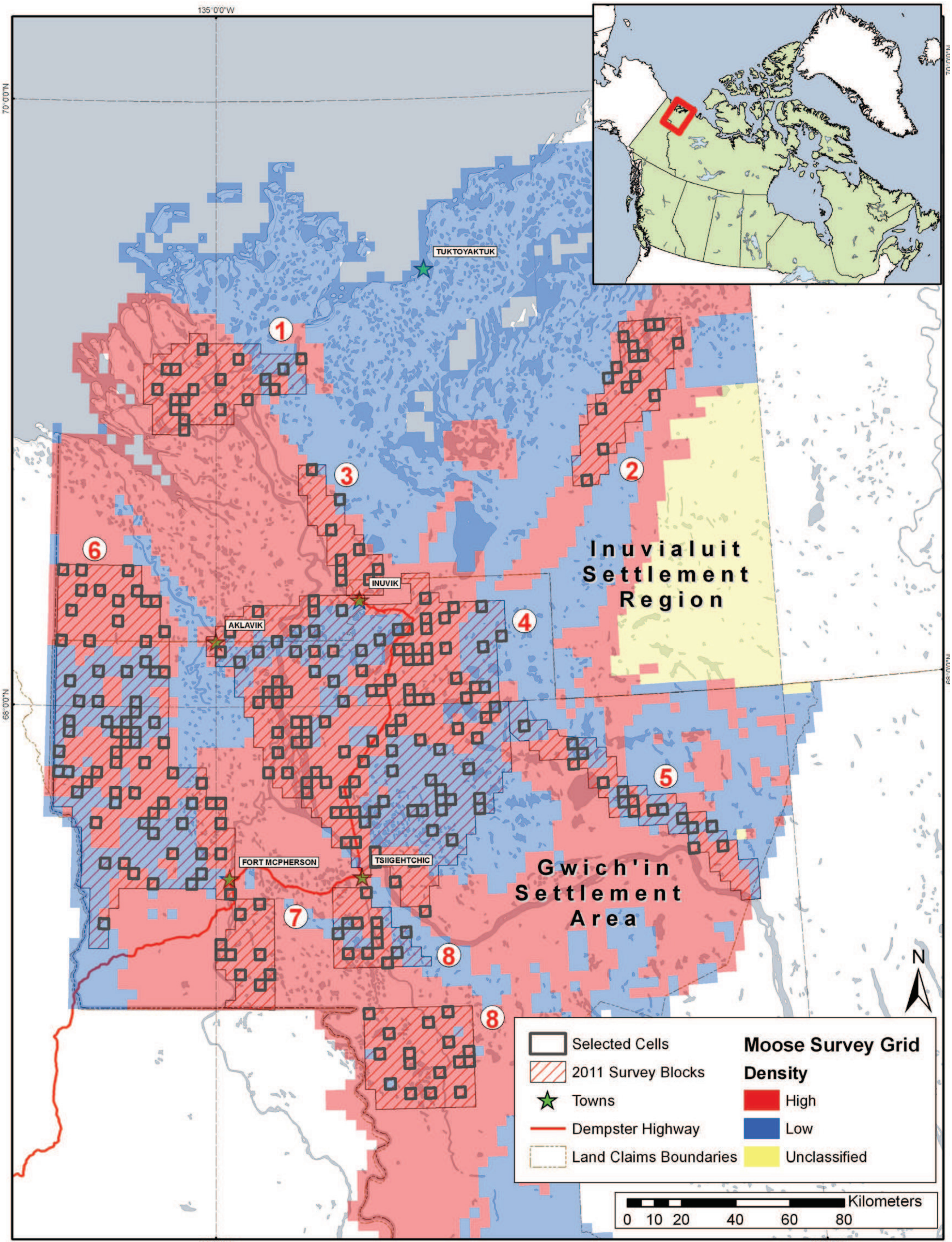

Fig. 1. Stratification and surveyed cells in eight survey blocks: 1) Delta North, 2) Kugaluk-Miner Rivers, 3) Ikhil Pipeline, 4) Inuvik-Tsiigehtchic, 5) Mackenzie Gas pipeline route, 6) Richardson Mountains, 7) Peel River, and 8) Arctic Red River.

Table 1. Number (#) of moose observations, moose densities, and population estimates by survey block.

Survey Block	Survey Block Area (km²)	Percent Surveyed	# of Moose observed	Population estimate	SE[1]	Density (Moose/ 100 km²)	2006 Density (Moose/ 100 km²)
Delta North	1448.1	15.5	10	61.21	28,19	4.49	NA
Kugaluk-Miner Rivers	1155.9	16.0	2	NA[2]	-	1.08	NA
Ikhil Pipeline	671.6	17.0	11	NA[2]	-	9.66	NA
Inuvik-Tsiigehtchic	8611.5	16.0	27	170.88	43.30	1.94	1.62
Mackenzie Gas Pipeline Route	1286.2	16.5	7	41.47	28.40	3.33	2.31
Richardson Mountains	5705.6	15.8	20	126.24	64.68	2.23	3.54
Peel River	704.1	17.0	0	-	-	0.00	0.84
Arctic Red River	2196.3	17.2	2	12.19	7.70	0.53	0.00

[1] Standard Error.

[2] Could not compute estimate – insufficient samples in one stratum.

making the coarse overall moose density 2.24 moose/100 km². The highest density, 9.66 moose/100 km², was found in the Ikhil Pipeline survey block (Table 1). The Arctic Red River block was found to have very low densities with 0.53 moose/100 km². No moose were observed in the Peel River block; however we do not believe there are no moose in the area, as there were tracks observed. The Peel River block was the smallest survey area and we believe that the sample size was too small to detect moose at the low densities they occur in this area. The densities found in the other blocks were: 4.49 moose/100 km² in Delta North, 1.08 moose/100 km² in Kugaluk-Miner Rivers, 1.94 moose/100 km² in Inuvik-Tsiigehtchic, 3.33 moose/100 km² in the Mackenzie Gas Pipeline Route, and 2.23 moose/100 km² in the Richardson Mountains (Table 1).

A sightability correction was not determined for our survey. Moose sightability varies by season, snow cover, habitat, and size of the survey unit (Gasaway *et al.*, 1986). Habitat in the study region ranged from alpine/tundra to semi-open coniferous forest with sightability higher in more open habitats. It is possible to estimate a sightability correction factor using radio-collared moose (Gasaway *et al.,* 1986). However, since there were no collared moose in our study area we could not obtain a correction factor.

The Delta North and Ikhil Pipeline survey blocks had not been previously surveyed. The Kugaluk-Miner river area was surveyed in 1988, with an overall density of 6 moose/100 km^2 (Jingfors & Kutny, 1989) which is higher than the density found in this survey (1.08 moose/100 km^2). It is not known if this is a real trend because we only have two data points that were obtained using different survey methods. The survey block area in 2011 was not as large as the 1988 survey. Compared to the 2006 survey in the GSA, the Richardson Mountains and Peel River blocks were found to have lower densities in 2011. The densities for the Richardson Mountain block were also lower than a 2000 helicopter survey of the Richardson Mountains that included the Yukon where a density of 4.8 moose/100 km^2 was found (Yukon Government, unpublished data). Methods for the 2000 survey were quite different, as optimal habitat was flown instead of randomly sampling areas. The Inuvik-Tsiigehtchic, Mackenzie Gas Pipeline route, and Arctic Red River survey blocks had higher densities in 2011 compared to the 2006 survey (Table 1).

Moose in the ISR and GSA are at the northern edge of their range. As such, environmental factors and range conditions may partially explain observed lower densities than in other portions of the species' range. Observed densities appear generally consistent with those reported for other subarctic regions. Franzmann & Schwartz (1998) summarized general densities (moose/100 km2) across the species range as < 12 in subarctic areas, 12-31 in better ranges, and 40-100 in excellent ranges.

Densities in the Inuvik Region appear lower than other areas of the NWT, except the North Slave region where density ranged from 2.0 to 3.5 moose/100 km2 in 2005 (Cluff, 2005). The highest densities recorded in the NWT have been 17 moose/100 km^2 around Fort Good Hope and Norman Wells (Maclean, 1994; Veitch *et al.,* 1995).

Participation of knowledgeable community members and harvesters in the stratification of survey areas was important to improve accuracy of population estimates. Natural low densities make it difficult to detect trends between surveys. Composition estimates could be improved if surveys were conducted before moose bulls shed their antlers. We advocate that information on moose habitat, recruitment, and mortality, as well as increased coverage of future surveys would help to increase precision and confidence of estimates and would help to explain changes in moose distribution, density and number.

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Brief Communication

Peary caribou (*Rangifer tarandus pearyi*) and muskoxen (*Ovibos moschatus*) on northwest Victoria Island, Northwest Territories

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Abstract: An aerial population survey of Peary caribou (*Rangifer tarandus pearyi)* and muskoxen *(Ovibus moschatus)* on Victoria Island, Northwest Territories, was conducted in July and August 2010. The population estimate of adult Peary caribou was 150 ± 104 (95% Confidence Interval [CI]) was not significantly different than the 2005 estimate of 66 ± 61 ($P < 0.05$). There was also an estimate of 430 ± 214 (95% CI) adult Dolphin-Union caribou (R. t. groenlandicus x pearyi) in the study area. However, these caribou represent only a small portion of the Dolphin-Union herd. The population estimate of 11 442 ± 1637 (95% CI) adult muskoxen is not significantly different than the 2005 estimate of 12 062 ± 2156 ($P < 0.05$).

Key words: muskoxen; *Ovibos moschatus*; Peary caribou; population survey; *Rangifer tarandus pearyi;* Victoria Island.

Rangifer, 33, Special Issue No. 21, 2013: 129–134

Introduction

Peary caribou (*Rangifer tarandus pearyi)* are endemic to Canada, occurring in the high Arctic islands of the Northwest Territories (NWT) and Nunavut. Peary caribou have declined across their range since the first surveys were done in the 1960s and 1970s; however, surveys of Peary caribou have not been consistent throughout the range. There is little trend information regarding some populations while other populations such as the Prince of Wales – Somerset Island Group have undergone steep declines without recovery (Jenkins *et al.,* 2011). Other islands seem to have small, yet stable numbers of caribou. Peary caribou were listed as endangered under the Canadian Species at Risk Act in February 2011.

Peary caribou are found year-round on northwest Victoria Island and are known as the Minto Inlet herd. At the southern end of Victoria Island, Dolphin-Union caribou *(R. t. groenlandicus x pearyi)* spend summer on the island and migrate to the NWT mainland across Dolphin and Union Strait in the fall to winter on the mainland. They migrate back to the south end of Victoria Island to calve in May/June. Muskoxen are resident throughout Victoria Island.

Caribou surveys on northwest Victoria Island have been done over the same area using the same methods since 1992; however, there were periodic surveys with varying coverage previous to this (Gunn, 2003). The first survey was done for a proposed Polar Gas Pipeline in 1980 (Jakimchuk & Carruthers, 1980). The Minto Inlet herd and Dolphin-Union herd ranges were first documented by a satellite telemetry program from 1987 to 1989 (Gunn & Fournier, 2000). There appeared to be a decline of Peary caribou in the 1980s and 1990s with harvesting, predation, and weather cited as possible causes (Nishi & Buckland, 2000; Gunn, 2003). There is currently a voluntary moratorium on harvest of Peary caribou on northwest Victoria Island.

Muskoxen on northwest Victoria Island are harvested by aboriginal subsistence harvesters from Ulukhaktok. The total annual allowable harvest of muskoxen is 1000; however, reported harvest has ranged from 208 to 270 muskoxen per year over the last 5 years.

Study area

Victoria Island is the second largest island in the Canadian Arctic Archipelago and is split between the Inuvialuit Settlement Region (ISR), in the NWT, and Nunavut. Wildlife on the NWT portion of Victoria Island is co-operatively managed by the Wildlife Management Advisory Council (WMAC [NWT]), made up of members appointed by the Inuvialuit, Government of the Northwest Territories and the Government of Canada, and whose mandate includes advising appropriate ministers on all matters relating to wildlife management within the ISR.

Methods

Survey lines, spaced 5 km apart, were flown with a Helio Courier fixed-wing aircraft over northwest Victoria Island. The survey crew consisted of a pilot, two rear seat observers, and a front seat recorder/navigator. The island was divided into 3 blocks for survey purposes (A, B, and C). Survey lines could be flown in less than 30 minutes to reduce observer fatigue. The survey was flown at 120 m above ground level at an average approximate speed of 160 km/hr, and markers were placed on the aircraft windows to delineate a strip width of 500 m on each side of the aircraft. The pilot maintained altitude using an aircraft altimeter. No correction factor was used for missed or over-counted animals due to inability to maintain survey altitude due to steep terrain. This type of terrain is rare on northwest Victoria Island, with most of the area being flat except for some rolling hills near Minto Inlet. Past surveys also did not have a correction factor and covered the same terrain with similar methods and would have similar biases. Observations within 500 m were considered 'on' transect. Muskoxen on transect were classified as adults or calves based on body size as calves are noticeably smaller than adult animals. All caribou observations (on or off transect) were classified as bulls, cows/young bulls, or calves. Mature bulls were identified by their antlers, and cows and young bulls were grouped as distinguishing them from the aircraft is more difficult. If necessary, the aircraft would leave the transect line to allow for classification of animals. Animals directly under the aircraft, and not visible from the side window, were considered 'off' transect as transect width were measured using the area visible to the observer from the side of the aircraft. Sightings of other wildlife were also recorded.

Movement data from collared caribou collected from 1987 to 1989 for Peary caribou and 1996 to 2005 for Dolphin-Union caribou were used to assign which survey blocks were utilized by each type of caribou (Gunn & Fournier, 2000; Nagy *et al.,* 2009c; Fig. 1).

Population estimates for adult caribou and muskoxen were calculated using a ratio method for unequal-sized units sampled without replacement (Krebs, 1999; Ecological Methodology, Version 7.0). Population estimates were calculated for the adult portion of the population because of the high variability of caribou calf production year-to-year and higher mortality rate in their first year (Larter & Nagy, 1999).

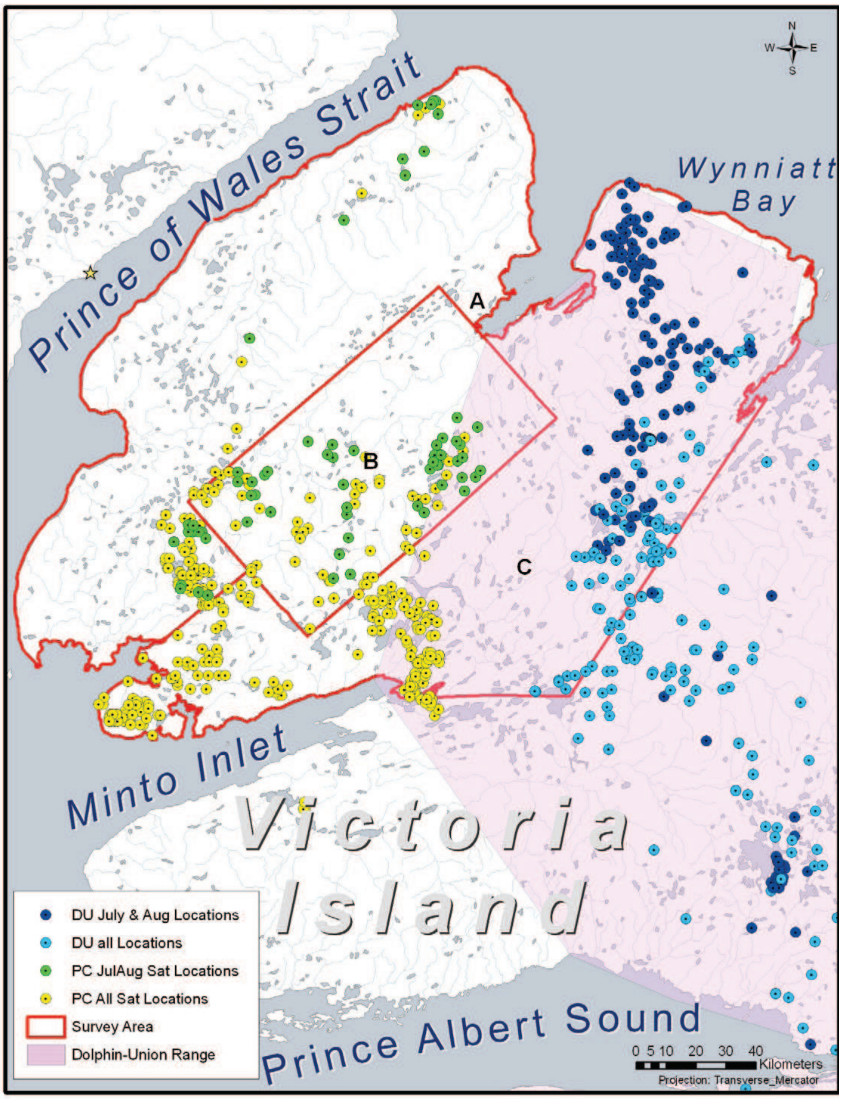

Fig. 1. Locations of collared Minto Inlet Peary caribou (PC) (1987 – 1989) and Dolphin-Union caribou (DU) (1996 – 2005) during the time of the survey (July-August) and throughout the year rest of the year (September – June: all locations). (Peary caribou data are from Gunn & Fournier [2000] and Dolphin-Union caribou data are from NWT & Nunavut [unpublished data])

This is also consistent with historic population estimates. A two-tailed t-test was used to determine if the population estimates of adult muskoxen and Peary caribou were significantly different from the 2005 estimates of Nagy et al. (2009a) for the same area, following statistical methods of Gasaway et al. (1986).

Results & Discussion

The survey was flown over 11 days between July 28th and August 15th, 2010. The survey was delayed intermittently by poor weather, and short line segments in the southwest end of Block A and northeast end of block C were missed due to fog (Fig. 2). The number of weather days increases the chance that caribou could have moved during the survey. There were 140 transects totalling 7155.7 km in length flown on the survey. With a transect width of 1 km this is a 19.9% coverage of the 36 021.2 km2 study area.

Caribou in blocks A and B were considered to be Peary caribou (Fig. 1). A total of 30 adult caribou and 4 calves were seen on transect in blocks A and B resulting in a population estimate of 150 ± 104 (95% Confidence Interval [CI], CV= 0.693) adult caribou for survey blocks A and B. The 2010 population estimate of adult Peary caribou is not significantly different than the 2005 estimate of 66 ± 61 ($t = 1.39$, $P < 0.05$, $df = 80$). The average group size for Peary caribou observed on transect was two animals with the largest group being six and the smallest group being one. The population trend for adult Peary caribou from 1998 – 2010 has been stable (Fig. 3).

Caribou observed in block C were considered to be of the Dolphin-Union herd (Nagy et al. 2009c; Fig.1). A total of 85 adult caribou and 14 calves were seen on transect in block C giving an estimate of 430 ± 214 (95% CI) adult Dolphin-Union caribou. However, these caribou represent only a small portion of the Dolphin-Union

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herd; therefore historic counts do not represent population trend but just use of the study area. The average group size for Dolphin-Union caribou observed on transect was 2.4 animals with the largest group being nine and the smallest group being one. Caribou distribution was concentrated in the east side of the study area (Fig. 2).

There were a total of 2273 adult and 31 calf muskoxen seen on transect, yielding a population estimate of 11 442 ± 1637 (95% CI, CV=0.143) adult muskoxen. The 2010 population estimate of adult muskoxen is not significantly different than the 2005 estimate of 12 062 ± 2156 (t = -0.463, P < 0.05, df = 128) for this part of Victoria Island. However, the number of calf muskoxen seen in 2010 was lower than the number of calves in 2005 (321 calves). The average group size for muskoxen observed on transect was 6.4 animals with the largest group being 23 and the smallest group being one. Muskoxen were distributed throughout the study area, with the highest concentration on the western portion (Fig. 4). The population of muskoxen had declined between the 2001 and 2005 survey (Nagy *et al.,* 2009c; Fig. 5).

The results of the 2010 survey indicated that the population of Peary caribou on Northwest Victoria Island is not recovering further from its low in the 1990s. It is not known the reason for this lack of recovery; however harvest is not a factor. Although the population of muskoxen in the survey area is

Fig. 2. Transects flown during the 2010 survey and distribution of adult caribou found 'on' and 'off' transect on northwest Victoria island, NWT.

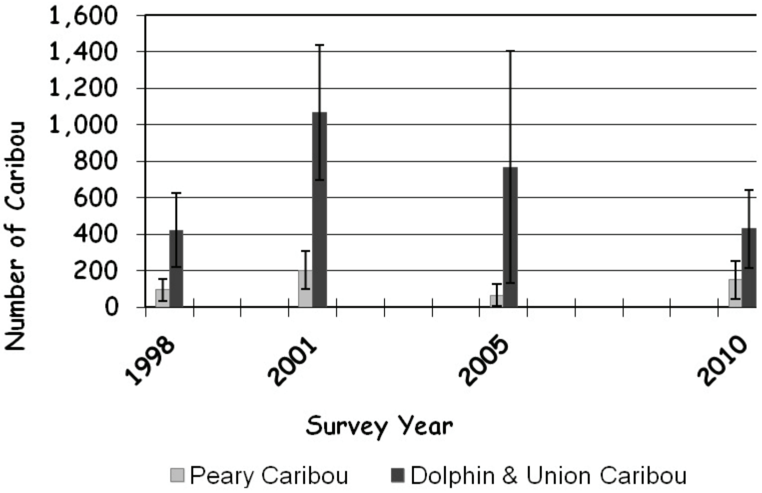

Fig. 3. Population counts of adult Minto Inlet Peary caribou and Dolphin-Union caribou on northwest Victoria Island, NWT with 95% confidence intervals around the population estimates.

stable from the 2005 estimate, the low number of calves observed is a concern with possible implications for the population in further years.

Fig. 4. Transects flown during the 2010 survey and distribution of adult muskoxen found 'on' transect on northwest Victoria Island, NWT.

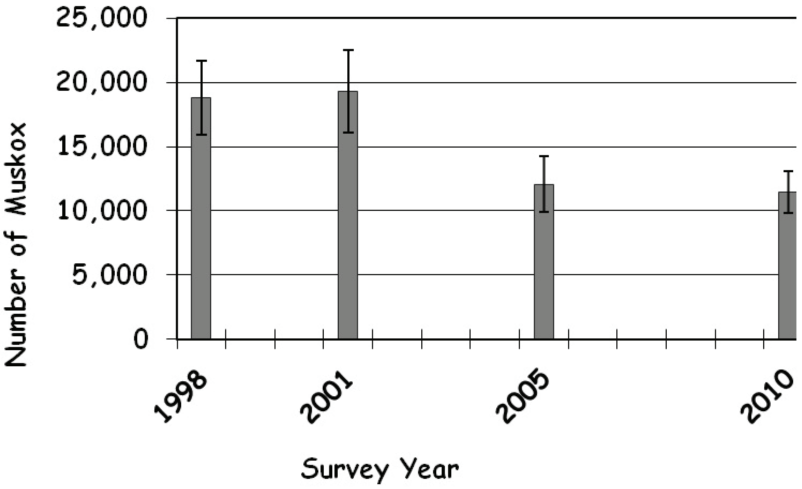

Fig. 5. Population trend of adult muskoxen on northwest Victoria Island, NWT, with 95% confidence intervals around the population estimates.

Acknowledgments

The authors would like to thank Allan Pogotak (Ulukhaktok) for his assistance as an observer and Perry Linton (North-Wright Airways Ltd.) for piloting the fixed-wing aircraft. This study was jointly funded by Inuvialuit Wildlife Studies Funds, Polar Continental Shelf Program, and Environment and Natural Resources.

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Brief Communication

Population survey of Peary caribou (*Rangifer tarandus pearyi)* and muskoxen *(Ovibos moschatus)* on Banks Island, Northwest Territories, July 2010

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Abstract: We conducted a systematic aerial transect survey of Peary caribou (*Rangifer tarandus pearyi)* and muskoxen *(Ovibus moschatus)* on Banks Island, Northwest Territories, in July 2010. The population estimate of adult Peary caribou was 1097 ± 343 (95% Confidence Interval: CI), which is not significantly different from the 2005 estimate of 929 ± 289 (95% CI; $P < 0.05$). The current number, however, is a 4- to 9-fold decrease since the 1980s. The adult muskoxen population estimate was 36 676 ± 4031 (95% CI), which is significantly lower than the 2005 estimate of 47 209 ± 3997 (95% CI; $P < 0.05$).

Key words: Banks Island; muskoxen; *Ovibos moschatus;* Peary caribou; population survey; *Rangifer tarandus pearyi.*

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Introduction

Peary caribou (*Rangifer tarandus pearyi)* are endemic to Canada, occurring in the high Arctic islands of the Northwest Territories (NWT) and Nunavut. Peary caribou have declined across their entire range since the first surveys were done in the 1960s and 1970s, however, surveys of Peary caribou have been conducted infrequently, which makes determining trend of the overall Peary caribou population difficult. The Peary caribou of Banks Island, however, have a long term aerial strip-transect survey data set. Surveys were first conducted on Banks Island in 1973, and systematic island-wide surveys using fixed-wing aircraft have been done every 2 – 5 years since 1982. These have documented the number and population trend of Peary caribou and muskoxen on Banks Island. Between 1982 and 1998, the Peary caribou population declined from about 9038 to 451 non-calf animals. The population estimate of adult Peary caribou in 2005 was 929 ± 289 (95% CI). The adult muskoxen population estimate was 47 209 ± 3997 (95% CI) in 2005.

Peary caribou on Banks Island are harvested by aboriginal subsistence hunters from Sachs Harbour. A quota for Peary caribou has been in place since 1991 and is currently 72 animals (bulls only). Reported harvest of Banks Island Peary caribou for the period 2005 – 2010 has ranged from 1 to 12 caribou per year. Peary caribou was assessed by COSEWIC as endangered in Canada in May 2004 and legally listed under the Species at Risk Act in February 2011.

For muskoxen, there is both a commercial and aboriginal subsistence harvest. Although the total annual allowable harvest of muskoxen is 10 000, reported harvests have ranged from 60 to 987 muskoxen per year for the period 2005 – 2010. Here we present the 2010 population estimates for Peary caribou and muskoxen on Banks Island in relation to past estimates.

Study area

Banks Island covers an area of 70 579 km^2, making it the 4th largest island in the Canadian Arctic Archipelago. Located in the Inuvialuit Settlement Region (ISR), Banks Island is home to both Peary caribou and muskoxen. Wildlife on Banks Island is co-operatively managed by the Wildlife Management Advisory Council (WMAC [NWT]), made up of members appointed by the Inuvialuit, Government of the NWT, and the Government of Canada and whose mandate includes advising appropriate Ministers on all matters relating to wildlife management within the ISR.

Methods

Survey lines, spaced 5 km apart, were flown over the entire island. The survey was conducted with two aircraft (Helio Courier and a Cessna 206) and each crew consisted of a pilot, two rear seat observers, and a front seat recorder/navigator. The island was divided into 11 blocks for survey purposes (A – K). This ensured that survey lines could be flown in less than 30 minutes to reduce observer fatigue. Transects were run perpendicular to drainages. The survey was flown at an average speed of approximately 160 km/hr, 120 m above ground level and markers were placed on the aircraft struts or windows to delineate a strip width of 500 m on each side of the aircraft. The pilot maintained altitude using an aircraft altimeter. No correction factor was used for missed or over-counted animals due to inability to maintain survey altitude due to steep terrain. This type of terrain on Banks Island is minimal and occurs mostly in the northwest portion of the island. Past surveys also did not have a correction factor and covered the same terrain with similar methods so would have similar biases. Observations within 500 m were considered ‘on’ transect. Muskoxen and caribou, on or off transect, were classified as adults (≥ 1-year old) or calves based on body size as calves are noticeably smaller than adult animals. Further, all caribou observed were classified as mature bulls, cows/young bulls, or calves. Mature bulls were identified by their antlers, and cows and young bulls were grouped as distinguishing between them from the aircraft is more difficult. Animals directly under the aircraft, and not visible from the side window, were considered ‘off transect’ as transect widths were measured using the area visible to the observer from the side of the aircraft. Sightings of other wildlife were also recorded.

Population estimates for adult caribou and muskoxen were calculated using a ratio method for unequal-sized units sampled without replacement (Krebs, 1999; Ecological Methodology, Version 7.0). Population estimates are calculated for the adults because of the high variability of caribou calf production year-to-year and higher mortality rate in their first year (Larter & Nagy, 1999b). This is also consistent with historic population estimates. A two-tailed t-test was used to determine if the population estimates of adult muskoxen and Peary caribou were significantly different than the 2005 estimates (Gasaway *et al.,* 1986; Nagy *et al.,* 2009a).

Results & Discussion

The survey was conducted between July 17 and 26, 2010. One survey crew flew blocks A, B, C, H, I, J, and K in 9 days, with one weather day (19th July). The other survey crew flew blocks D, E, F, and G in 7 days with no weather days once the survey started. All planned survey lines were flown except a short segment on the north shore of the island, which was obscured by fog (Fig. 1). There were 229 transects totaling 13 826.8 km in length flown on survey. With a transect width of 1 km this results in a coverage of 19.59% of the island.

A total of 285 Peary caribou (215 adults, 70 calves) were seen on transect, and a further 75 caribou (63 adults, 12 calves) were seen off transect (Fig. 1). On transect observations resulted in a population estimate of 1097 ± 343 (95% Confidence Interval, [CI]; CV = 0.313) adult Peary caribou on Banks Island. The proportion of calves among all caribou observed (on and off transect as all observations of caribou were classified) was 22.8%. The proportion of calves varies by survey, reported in 2005, 2001, and 1992 as 19.4%, 26.3% and 28.8%, respectively (Nagy *et al.,* 2006; 2009a; b). Eighty-one mature bulls were observed (on and off transect), which was 22.5% of all caribou observed. The proportion of mature bulls varies by survey, reported in 2005 and 2001 as 29% and 19%, respectively (Nagy *et al.,* 2006; 2009a). Distribution was concentrated in the northwest, northeast, and central portions of Banks Island (Fig. 1). The average group size for Peary caribou observed on transect was 4.2 with the largest group being 37 and smallest being one. The 2010 population estimate of Peary caribou was not significantly different from the 2005 estimate of 929 ± 289 (95% CI) (t = 0.74, $P < 0.05$, df = 45). Peary

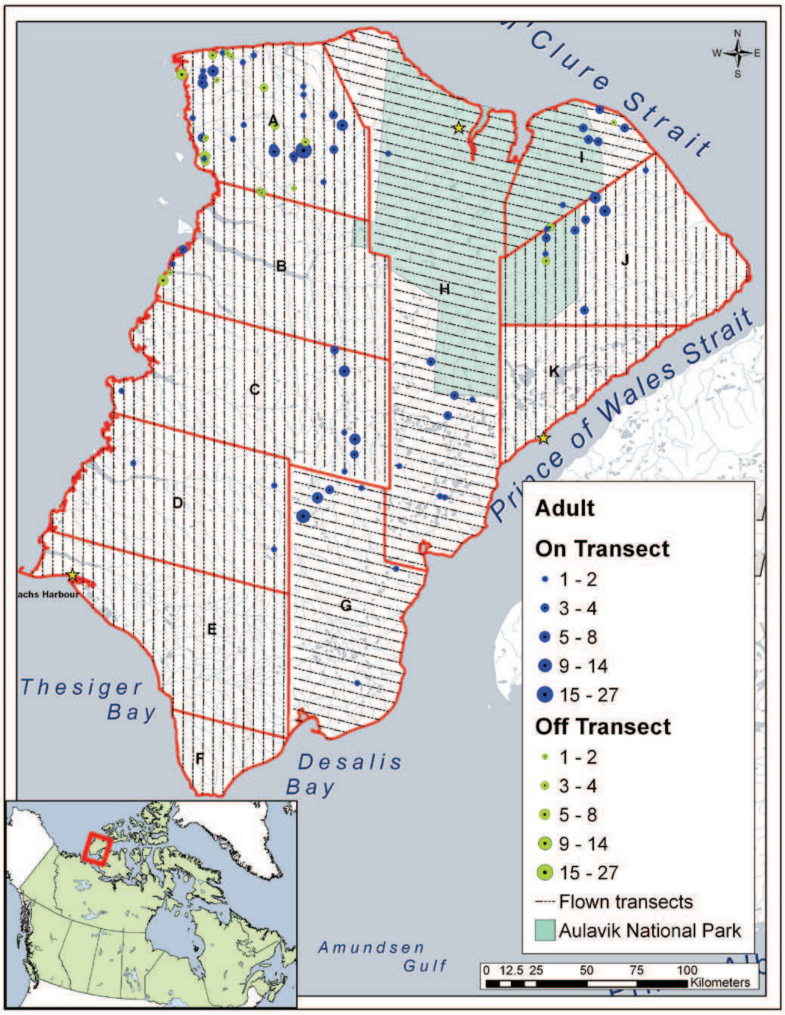

Fig. 1. Distribution of adult Peary caribou observed both on and of transect during an aerial servey flown from July 17 – 26, 2010, on Banks Island, NWT.

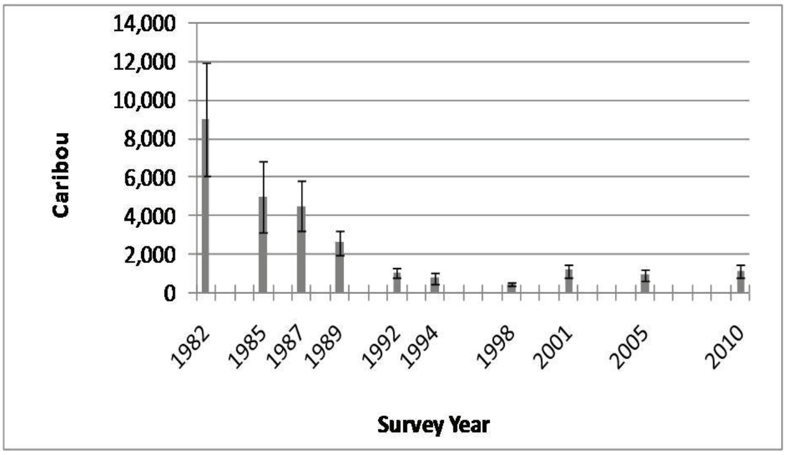

Fig. 2. Transects flown during the 2010 survey and distribution of adult caribou found 'on' and 'off' transect on northwest Banks Island, NWT.

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caribou numbers on Banks Island have decreased 9-fold since 1982; however, the current low abundance appears to have been stable for the past decade (Fig. 2). At small numbers populations become more vulnerable (Lacy, 2000) and at low numbers Peary caribou populations are particularly influenced by severe weather and changes in their environment (Gunn *et al.,* 1980).

The average group size for muskoxen observed on transect was 8.5 with the largest group being 38 and smallest being one. There were a total of 8054 muskoxen (7185 adults, 869 calves) observed on transect, resulting in a population estimate of 36 676 ± 4031 (95% CI; CV=10.99) adult muskoxen on Banks Island and a calf percentage of 10.7%. Muskoxen were spread throughout Banks Island, with concentrations in the southwest and north-central areas (Fig. 3). The 2010 population estimate of adult muskoxen was significantly lower ($t = 3.699$, $P < 0.05$, $df = 90$) than the 2005 estimate of 47 209 ± 3997. Muskoxen abundance on Banks Island from 1982 to 2010 has fluctuated, and since 2001 has exhibited a decline (Fig. 4).

The reasons for the decline are not clear. In 1996 a die-off of muskoxen on Banks Island was attributed to disease (Larter & Nagy, 1999a). During

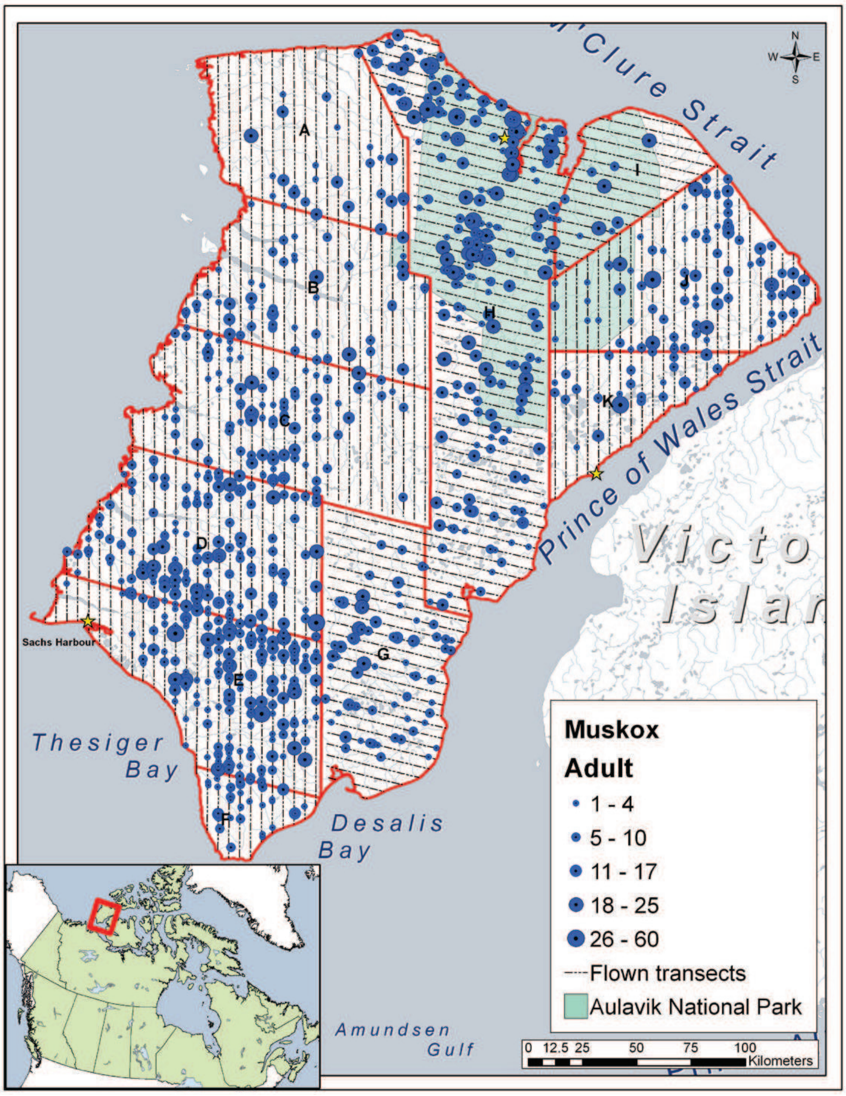

Fig. 3: Population counts of adult Minto Inlet Peary caribou and Dolphin-Union caribou on northwest Banks Island, NWT with 95% confidence intervals around the population estimates.

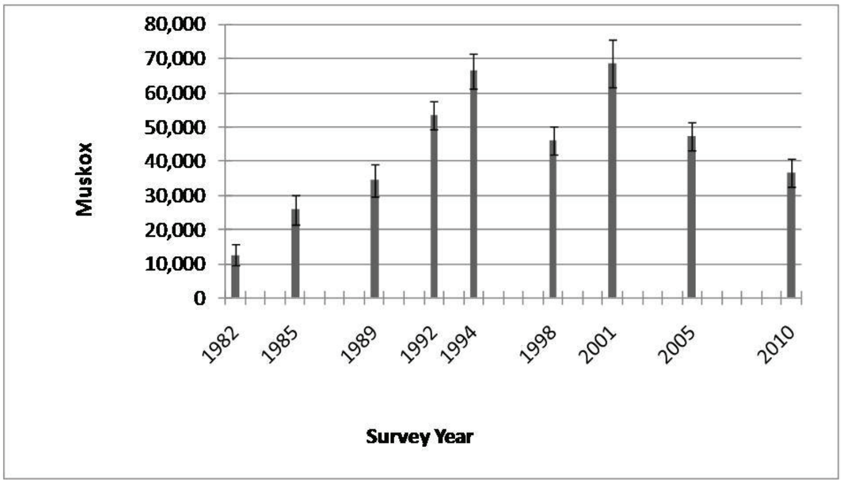

Fig. 4. Transects flown during the 2010 survey and distribution of adult muskoxen found 'on' transect on northwest Banks Island, NWT.

the July 2004 survey a large number of muskoxen carcasses were observed and calves made up only 4% of total muskoxen observed during the survey. This was assumed due to severe icing events in the 2003/2004 winter (Nagy & Gunn, 2009). Our 2010 survey observed only four carcasses, and 10.7% calves is similar to the recovered productivity of 12.6% observed in 2006 (Gunn & Williams, unpublished data, Government of the NWT July 2006). As designed, the surveys provided abundance estimates and trends, while explanations behind trends must be sought elsewhere.

Acknowledgments

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Brief Communication

CircumArctic *Rangifer* monitoring and assessment (CARMA) network – origins, goals, accomplishments and future

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Origins

The CircumArctic *Rangifer* Monitoring and Assessment (CARMA) network is an informal group of scientists, community representatives, and management agencies who formed to better understand the impacts of global changes on migratory tundra caribou. In this report we outline how that network evolved, discuss some of the accomplishments of the group, and look forward to CARMA's future. CARMA was formally launched in Vancouver in 2004. This launch was preceded by several events. In 1998 the Arctic Council ministers directed the Conservation of Arctic Flora and Fauna (CAFF) to "identify elements of a program to monitor circumpolar biodiversity" and to "assess the effects of climate change on Arctic ecosystems". Concurrent with that initiative, in 1999 in Rovaniemi, Finland, the International Arctic Science Committee (IASC) and the US National Science Foundation funded an interdisciplinary conference of *Rangifer* managers, users, and scientists to discuss elements of a circumpolar monitoring and assessment network for human-*Rangifer* systems and the formation of a community to implement the plan (Russell *et al.,* 2000). After a meeting to implement the Arctic Council's directive, *Rangifer* was confirmed as a key indicator species and the CARMA network was officially endorsed by CAFF. With that endorsement and the plan from the Rovaniemi conference in mind, the IASC human-*Rangifer* study group met in Minary, New Hampshire, in 2001 and fleshed out the elements of a *Rangifer* network, leading to the official launch of the CARMA network. Subsequently, in 2005, CARMA was invited to become an official network under the Circumpolar Biodiversity Monitoring Program (CBMP). The CBMP reports to the CAFF.

Goals

CARMA has held annual meetings since 2004. At the initial meeting CARMA adopted the mission to: "monitor and assess the impacts of global change on human-*Rangifer* (reindeer, caribou) systems across the circumArctic, through cooperation, both geographically and

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across disciplines". In 2005, two large grants provided support. One was through the Canadian International Polar Year (IPY) program and the other through the US National Science Foundation. CARMA's Canadian IPY project, "Starting the clock for the CARMA Network: Global Change, Resilience and Human-*Rangifer* Systems of the CircumArctic", aimed to developed protocols for monitoring at the individual and population level, and funded a number of projects to either initiate standardized monitoring programs or develop monitoring methodologies that could be incorporated into monitoring manuals. The funding also gave us an opportunity to: 1) develop a communications website (www.caff.is/carma), 2) develop data sharing and handling policies, 3) initiate programs to better understand how change is affecting communities across the north, 4) document local knowledge about changes in human-caribou relationships, 5) provide educational tools to schools, 6) provide materials and training for hunters to participate in monitoring the health of caribou, and 7) further our capability to synthesize data and assess herd-specific vulnerability and resilience to global change through modeling.

In the December 2010 meeting, collaborators of the program discussed the products produced by CARMA, highlighting the strengths and weaknesses so that we could improve and make these resources more accessible to our collaborators. Below is the summary of feedback received after reviewing the specific tools and resources.

Accomplishments

CARMA standardized protocols

CARMA has developed two protocol manuals. The body condition and health manual (Kutz *et al.,* 2013) presents three levels of field procedures to assess and monitor *Rangifer* health and body condition (http://caff.is/resources/field-protocols). Level 1 is designed for hunters in the field to make a quick assessment of the condition of caribou, collect very few samples, and record a limited amount of information. The advantage of using this method is that it greatly expands the sample size and geographic scope of collections and provides communities with capacity-building and educational tools. Level 2 is designed for hunters, accompanied by trained technicians, to provide an expanded amount of information and collect more samples. Level 3 is the most detailed and thorough sampling, conducted by veterinarians or biologists, and expands our knowledge of parasites and disease in populations.

The demographic manual (http://caff.is/resources/field-protocols) reviews population estimation techni-ques and presents tools to be used to report trends and variance in population parameters. The manual offers an overview of monitoring methods applied to migratory tundra caribou. The protocols associated with this manual give details on how to report on the monitoring of demographic indices that differs from the approach used in the CARMA health and condition manual and protocols.

The Rangifer Anatomy Project

The *Rangifer* anatomy project (RAP) grew out of a need for better resources to enhance *Rangifer* health monitoring, to promote responsible hunting, and to facilitate knowledge exchange across generations among *Rangifer* users including northern community members, wildlife managers, scientists, and educators. RAP was initiated at the Faculty of Veterinary Medicine, University of Calgary, in December 2007 and has since grown into a much larger collaboration in the *Rangifer* world. The project revolved around the detailed dissection of two domestic reindeer which produced hundreds of high quality images of *Rangifer* anatomy. A webpage has been created, and posters produced and distributed to a number of schools (http://www.ucalgary.ca/caribou/index.html).

Hunter training video

The hunter training video (http://caff.is/index.php?option=com content&view=article&id=661&ltemid=1310) complements the body condition and health protocol manual. The video presents a step by step procedure for recording information and collecting samples for Level 1 and Level 2 protocols. Although the target audience is primarily hunters, the video has also been used for education purposes in a number of schools in the north.

Voices of caribou people

The Voices Project (http://caff.is/caribou-people-ii) is a video-based project to document the knowledge of indigenous peoples who traditionally have a close relationship with caribou. The project was carried out in partnership with indigenous peoples of North America who subsist on caribou and identify themselves as "caribou people". The project: 1) captured local perspectives on global changes (*e.g.*, climate, industrial, and institutional change) and local understanding of the impacts to the environment and their way-of-life, 2) documented community strategies for coping with these changes, 3) shared knowledge and information with other northern communities, researchers, and the general public, and 4) took a snapshot of current concerns regarding caribou and caribou hunting during the IPY as a legacy for future researchers. In 2007 over 100 hours of video footage of more than 95 caribou people were recorded and included youth, elders, harvesters, and community leaders from six communities across North America: Anaktuvuk Pass (Alaska), Old Crow (Yukon), Wekweeti (Snare Lake; Northwest Territories), Lutsel K'e (Snowdrift; Northwest Territories), Arviat (Nunavut), and Kawawachikamach (Quebec). Two video features were produced and a website that makes all videos accessible is in production.

Climate database

CARMA has developed a climate database (Russell *et al.,* 2013) based on NASA's Modern Era Retrospective Analysis for Research and Applications (MERRA) dataset. The database contains daily data for 25 variables that are relevant to caribou ecology and is spatially explicit, includes 22 herds and 5 seasonal periods (calving, summer, fall, winter, and spring) and two habitat types (taiga and tundra). The period covered is 1979 – 2010 and will be updated annually. Currently the dataset has been entered into a Microsoft Access® database and is being made available through the CARMA website.

Energy/protein model

CARMA has provided funds to further develop an energy model (White *et al.,* 2013) for caribou that has been in development for the last 20 years (Russell *et al.,* 2005). The model will be used to help identify data gaps and research priorities and assess the impacts of development and climate change on individual caribou.

Future

Following an intensive four years under the International Polar Year program, CARMA is now in the data synthesis and herd assessment phase and is developing a strategy on how it should function in the future. At the 2010 meeting, collaborators discussed a number of future priorities for CARMA. Collaborators stressed that CARMA should continue to produce products that are relevant to managers, academics, co-management boards, and local communities. Providing products that can be used as educational materials was also felt to be a key role for CARMA. To enhance CARMA's engagement, collaborators suggested CARMA: 1) strengthen its profile and role in circumpolar affairs, 2) design and promote products to engage target audiences, 3) identify missing collaborators, and 4) broaden CARMA's scope by including work on boreal caribou and domestic reindeer. The group also recommended new "product lines" for CARMA to enhance its relevance. CARMA should develop tools to help management and renewable resource boards cope with rapid change in their herds including: 1) protocols to prepare for herd declines, 2) an action framework for communities and resource managers to use throughout fluctuating caribou population abundance, but especially during population declines, and when conducting herd management with less-than-adequate data, and 3) an easy-to-access system for sharing the collective experience of management actions under different situations. Collaborators also identified the need for CARMA to use its tools and models to develop a generalized, user-friendly cumulative effects model for resource managers and users. Finally collaborators identified four possible options for CARMA's future: 1) as an information source to academics, managers, and user communities, 2) as a think tank for cutting edge basic and applied research, 3) as a place for inter-regional and international exchange, and 4) as an educational node for training future managers and scholars. In the near future CARMA will have to decide on which of these options to adopt over the next five years.

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Brief Communication

CARMA's MERRA-based caribou range climate database

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Introduction

The CircumArctic *Rangifer* Monitoring and Assessment (CARMA) network's primary goal is to monitor and assess impacts of global change on caribou (*Rangifer tarandus*). One core approach is conducting cross-herd comparisons and contrasts to gauge how herds are similar and how they differ in their responses to climate. By understanding regional climates in which seasonally migratory tundra caribou herds have evolved, we can better assess strategies and mechanisms that *Rangifer* employ to cope with environmental stress. Climate has a strong influence on caribou ecology through its effects on forage growth and availability, its influences on snow conditions, and on insect abundance that can harass caribou and cause changes in caribou movements and redistribution (Griffith *et al.,* 2002; Bergerud *et al.,* 2008; Couturier *et al.,* 2009). We therfore need regional climate datasets that allow direct comparison of environmental attributes across and between continents. Although climate data are available from meteorological stations, those stations are relatively few, unevenly distributed across herd ranges, and often measure climate using different protocols.

An alternative to assessing regional climate based on meteorological stations is NASA's Modern Era Retrospective Analysis for Research and Applications (MERRA) dataset (http://gmao.gsfc.nasa.gov/research/merra/). MERRA was undertaken by NASA's Global Modeling and Assimilation Office with the objectives of placing the observations from NASA's Earth Observing System satellites in a climate context, and improving upon the hydrologic cycle represented in earlier generations of reanalyses (Rienecker *et al.,* 2011). The resolution of the MERRA grid is 1/2 degrees latitude by 2/3 degrees longitude and data are provided on a daily time step for most variables. MERRA was chosen over other datasets because it covers the modern era of remotely sensed data (from 1979 through the present), attempts to address problems with previous reanalysis products, and is focused on the hydrological cycle. Other long-term reanalyses of the Earth's climate have high levels of uncertainty in precipitation and inter-annual variability. MERRA also has better coverage north of 60° than other datasets such as NCEP and data are normally publicly available within a few months.

Methods

Climate data were summarized for 22 herds into as many as 8 polygons for each herd. For most herds separate shapefiles were constructed for five seasons (calving, summer, fall, winter, and spring), for tundra and taiga portions of the range, and for the annual range. For some herds, the two Greenlandic herds and the Iceland reindeer, only the annual range was used as their distribution is small and there is no taiga. For most North American herds, all Greenlandic herds and Iceland reindeer, polygons were determined from radio-collar data (Table 1). Fixed kernel polygons (90% utilization distributions) were produced, using standard settings (href, raster resolution set to 120) from the Rodgers and Carr (1998) Home Range extension for ArcView 3.2a (Environmental Systems Research Institute, Redlands, California, U.S.A.). Collar data were not made available to CARMA for the Beverly and Qamanirjuaq herds. Therefore for those herds older data (pre-1995) were obtained with permission from GNWT-ENR. The basis for the resultant polygons was from work done pulling together surveys, maps, and collar data. Leslie Wakelyn, on behalf of the Beverly-Qamanirjuaq Caribou Management Board (BQCMB), produced seven overall seasonal ranges that were a general amalgamation of the data for 10-25 years over time periods that generally began in the late 1950s and extended to the early 1980s to mid-1990s. Wakelyn's seven seasons were amalgamated to produce polygons for the five seasons used by CARMA. Because there is no history of collaring reindeer in Russia, for the five Russian herds, seasonal polygons were developed by combining seasonal maps produced from aerial surveys by Russian management agencies combined with personal contact to finalize seasonal distributions. To produce the taiga and tundra polygons, a global treeline shapefile was taken from the Circumpolar Arctic Vegetation Map from the Alaska Geobotany Center (http://www.geobotany.uaf.edu/), with the original coverage supplied from (http://www.arcticatlas.org/).

Table 1. The source of range information and time periods they represent for CARMA's 22 migratory tundra herds.

Herd	Source of ranges	Timeframe
Bathurst	Collar data	1996-2009
Ahiak	Collar data	2001-2010
Beverly	Dated survey maps, BQCMB	~1957-1995
Qamanirjuaq	Dated survey maps, BQCMB	~1957-1995
Cape Bathurst	Collar data	1996-2010
Bluenose West	Collar data	1996-2010
Bluenose East	Collar data	1996-2010
Dolphin and Union	Collar data	1996-2006
George River	Supplied Polygons	2006-2009
Leaf River	Supplied Polygons	2006-2009
Teshekpuk	Supplied Polygons	1990-2009
Western Arctic	Collar data	1987-2010
Central Arctic	Collar data	1986-2006
Porcupine	Collar data	1985-2010
Kangerlussuaq-Sisimiut	Collar data	1998-1999
Akia-Maniitsoq	Collar data	1997-1999
Iceland	Collar data	2006-2008
Taimyr	Mapped ranges from figures	Unknown
Yana-Indigurka	Mapped ranges from figures	1980-1990
Sundrunskaya	Mapped ranges from figures	Unknown
Lena-Olenyk	Mapped ranges from figures	Unknown
Chokotka	Mapped ranges from figures	Unknown

While MERRA provides a large number

of climate variables, we chose to download 36 variables to compare between herd ranges (Table 2). These were selected based upon discussions that considered the views of a number of disciplines; they include variables that, at a daily time step, are considered relevant to caribou and also to climatology. MERRA variables were downloaded using the “wget.exe” program from NASA’s website (http://disc.sci.gsfc.nasa.gov/daac-bin/FTPSubset.pl). Mapping boundary box coordinates to download MERRA variables were set to be west -170°, north 82°, south 45°, and east 180° as to cover all possible ranges of the 22 caribou herds. Thirty variables were downloaded from three different MERRA products. Variables (see Table 2 for a description) ps, u850, v850, t850, q850, h1000, h850, u10m, u2m, v10m, v2m, t2m, qv10m, qv2m, and disph were obtained through “Atmospheric Single-Level Diagnostics (tavg1_2d_slv_Nx)”. Variables ts, albedo, albnirdf, albnirdr, albvisdf, and albvisdr were extracted from “Surface and TOA Radiation Fluxes (tavg1_2d_rad_Nx)”. Variables grn, lai, tpsnow, precsno, prectot, snomas, snodp, smland, and frsno were downloaded from “Land Surface diagnostics (tavg1_2d_lnd_Nx)”. The six wind speed and direction variables, ws2m, wd2m, ws10m, wd10m, ws850, and wd850 (Table 2), were calculated from the “u2m” and “v2m” wind component vectors listed above (see

Table 2. MERRA variables downloaded into CARMA’s climate database.

Short Name	Long Name	Unit
albedo	Surface albedo	fraction
albnirdf	Diffuse beam NIR surface albedo	fraction
albnirdr	Direct beam NIR surface albedo	fraction
albvisdf	Diffuse beam VIS-UV surface albedo fraction	fraction
albvisdr	Direct beam VIS-UV surface albedo	fraction
disph	Displacement height	m
Frsno[1]	Fractional snow-covered area	fraction
grn[1]	Vegetation greenness fraction	fraction
h1000	Height at 1000 hPa	m
h850	Height at 850 hPa	m
lai[1]	Leaf area index	m ^ 2 m ^ (-2)
precsno	Surface snowfall rate	kg m ^ (-2) s ^ (-1)
prectot[1]	Total surface precipitation rate	kg m ^ (-2) s ^ (-1)
ps	Time averaged surface pressure	Pa
q850	Q850 Specific humidity at 850 hPa	kg kg ^ (-1)
qv10m	Specific humidity at 10m above displacement[2]	kg kg ^ (-1)
qv2m	Specific humidity at 2m above displacement	kg kg ^ (-1)
smland[1]	Snowmelt rate	kg m ^ (-2) s ^ (-1)
snodp[1]	Snow depth	m
snomas	Snow mass	kg m ^ (-2)
t2m[1]	Temperature at 2m above displacement	°C
t850	Temperature at 850 hPa	°C
tpsnow	Top snow layer temperature	°C
ts	Surface skin temperature	°C
u10m	Eastward wind component at 10m above displacement	m s ^ (-1)
u2m	Eastward wind component at 2m above displacement	m s ^ (-1)
u850	Eastward wind component at 850 hPa	m s ^ (-1)
v10m	Northward wind component at 10m above displacement	m s ^ (-1)
v2m	Northward wind component at 2m above displacement	m s ^ (-1)
v850	Northward wind component at 850 hPa	m s ^ (-1)
wd 850[3]	Wind direction at 850hPa (to North)	degree
Wd10m[3]	Wind direction at 10m above displacement	degree
wd2m[3]	Wind direction at 2m above displacement	degree
ws10m[3]	Wind speed at 10m above displacement	m s ^ (-1)
ws2m[3]	Wind speed at 2m above displacement	m s ^ (-1)
ws850[3]	Wind speed at 850 hPa	m s ^ (-1)

[1] Variables also included in the CARMA dataset
[2] Displacement is the height in meters above the ground at which zero wind speed is achieved because of obstacles such as vegetation
[3] Variable not downloaded from MERRA, but calculated based on MERRA wind components (u,v)

http://mst.nerc.ac.uk/wind vect convs.html for an explanation). The "mean" option box was checked to download daily averaged values. All downloads were saved as NetCDF (Network Common Data Form) files.

In order to extract information stored in these downloads, readings of each variable of each grid points were first extracted using "ncdf" package (Pierce, 2011) in the open source statistical software R (R Development Core Team, 2012). The procedures of extraction are listed as the following: 1) coordinates of the qualified MERRA grid points were obtained through overlaying herd ranges to MERRA grids, 2) median values for each variable among the qualified grid points were further extracted by ranking and locating the 50th percentile of all gridded data, and 3) daily climate files for each year, each herd, and each range were constructed and written into comma-separated values (csv) formatted files. Therefore, each MERRA reading in the database represents the median daily averaged value with its MERRA grid point falling inside or close to the herd range in study. Using median readings instead of mean values within each range avoids making normality assumptions and reduces bias.

All snow variables were considered in snow-years only. A snow-year is defined as starting from the 184th day of the year prior to the 183rd day of the current year. Thus, fall and winter periods of a snow-year consist of 182 days (or 183 days if the year prior is a leap year), and spring and summer periods of a snow-year consist of 183 days. Daily minimum and maximum temperature variables were extracted from the hourly MERRA assimilated temperature data with the following procedures. Hourly maximum and minimum readings were first extracted for each MERRA grid point within the study range. Median readings were then extracted by ranking to locate the 50th percentile.

Based on these 36 variables (Table 2) we then produced a "caribou-relevant" dataset that includes 25 variables for each herd range. Some MERRA variables could be directly used and for others we derived variables based on the MERRA variables for each herd and seasonal range. Equations used to derive the variables were written into R source codes.

1. Snow depth (snodp, m): Equal to MERRA variable snodp.

2. Snow density (snowdensity, g·cm^{-3}): Snow mass over snow depth. Snow mass and snow depth are equal to MERRA variables snomas and snodp, respectively.

$$\text{snowdensity} = \frac{snomas}{snodp}$$

3. Temperature at 2 meters above ground (t2m, °C): Equal to MERRA variable t2m.

4. Daily minimum temperature at 2 meters above ground (t2m_min, °C): Median reading of the daily minimum temperature from 0:30 to 23:30.

5. Daily maximum temperature at 2 meters above ground (t2m_ max, °C): Median reading of the daily maximum temperature from 0:30 to 23:30.

6. Total surface precipitation (prectot, kg·m^{-2} sec^{-1}): Equal to MERRA variable prectot.

7. Daily total surface precipitation (precip_24hr kg·m^{-2} d^{-1}): Total surface precipitation accumulated over 24 hours. Uses MERRA total surface precipitation variable prectot.

$$\text{precip\_24hr} = \text{prectot} \times 24 \times 60 \times 60$$

8. Snowmelt rate (smland, kg·m^{-2} sec^{-1}): Equal to MERRA variable smland.

9. Fractional snow covered area (frsno, fraction): Equal to MERRA variable frsno.

10. Surface snowfall (i.e., all frozen precipitation; precsno kg·m^{-2} sec^{-1}): Equal to MERRA variable precsno.

11. Number of days with freeze/thaw events (nday_fzthaw, day): Accumulated days from January to December, when t2m_max above 0°C and t2m_min below 0ºC.

$$\text{nday\_fzthaw} = \sum_{\text{Jan.1}}^{\text{Dec.31}} I_i\,,\ I_i = \begin{cases} 1 & \text{if t2m\_max} > 0 \text{and t2m\_min} < 0 \\ 0 & \text{otherwise} \end{cases}$$

12. Cumulative growing degrees above 0ºC (GDD0_cum, oC): Accumulated daily-averaged values of t2m if t2m > 0°C.

$$\sum_{i=1}^{n} t2m_i$$

13. Cumulative growing degrees above 5ºC (GDD5_cum, oC): Accumulated daily-averaged values of t2m if t2m > 5°C.

$$\sum_{i=1}^{n} t2m_i$$

14. Leaf area index (lai, m^2 m^{-2})): Equal to MERRA variable lai.

15. Vegetation greenness fraction (grn, fraction): Equal to MERRA variable grn.

16. Oestrid index (OI, unitless):

$$T = \begin{cases} 1 & \text{if t2m} > 18 \\ 0 & \text{if t2m} < 13 \\ 1 - \dfrac{18 - t2m}{10} & \text{otherwise} \end{cases} \quad W = \begin{cases} 0 & \text{if ws10m} > 9\text{m/s} \\ \dfrac{9 - ws10m}{9} & \text{otherwise} \end{cases}$$

17. Cumulative oestrid index (OI_cum, unitless): Accumulated daily OI from January 1 to December 31.

$$\text{OI\_cum} = \sum_{\text{Jan.1}}^{\text{Dec.31}} \text{OI}$$

18. Mosquito index (MI, unitless):

$$\text{MI} = \text{T} \times \text{W, where,}$$

$$T = \begin{cases} 1 & \text{if t2m } > 18 \\ 0 & \text{if t2m } < 6 \\ 1 - \dfrac{18 - t2m}{13} & \text{otherwise} \end{cases} \quad W = \begin{cases} 0 & \text{if ws10m } > 6\text{m/s} \\ \dfrac{6 - ws10m}{6} & \text{otherwise} \end{cases}$$

19. Cumulative mosquito index (MI_cum, unitless): Accumulated daily MI from January 1 to December 31.

$$\text{MI\_cum} = \sum_{\text{Jan.1}}^{\text{Dec.31}} \text{MI}$$

20. *Surface rainfall* (rainfall, mm·sec^{-1}): Total surface precipitation subtracted by all frozen precipitation, where use MERRA variables prectot and precsno.

$$\text{rainfall} = \mathit{prectot} - \mathit{precsno}$$

21. *Keetch Byram drought index* (KBDI, unitless): (today) = (yesterday) + (today), where DF (drought factor) is calculated following (Keetch & Byram, 1968);

$$\begin{cases} 0 & \text{if } P_t = 0 \text{ and t2m\_max }_t \leq 6.78; \\ \dfrac{(800 - KBDI_{t-1})(0.968e^{0.0875t2m\_max_t + 1.5552} - 8.3) * 10^{-3}}{1 + 10.88e^{-0.0174 * mean\ annual\ rainfall(cm)}} & \text{if } P_t > 0 \text{ and t2m\_max }_t > 6.78; \\ \dfrac{(800 - KBDI_{t-1})(0.968e^{0.0875t2m\_max_t + 1.5552} - 8.3) * 10^{-3}}{1 + 10.88e^{-0.0174 * mean\ annual\ rainfall(cm)}} & \text{if } P_t > 0 \text{ and} \sum P_t \leq 0.51cm \\ \dfrac{(800 - KBDI_{t-1})(0.968e^{0.0875t2m\_max_t + 1.5552} - 8.3) * 10^{-3}}{1 + 10.88e^{-0.0174 * mean\ annual\ rainfall(cm)}} - 39.37 \sum P_t & \text{if} P_t > 0 \text{ and} \sum P_t \geq 0.51cm. \end{cases}$$

where, P_t is the 24-hour precipitation, and KBDI (Jan. 1) is set to zero.

22. *Cumulative rain-on-snow* (rainsnow_cum, mm): Accumulated 24-hr rainfall if there is rainfall and snow depth is greater than 0.01 m.

$$\text{rainsnow}_{\text{cum}} = \sum_{i=1}^{t} \text{r ainfall}_i \times 24 \times 60 \times 60, \text{if } (\text{prectot} - \text{precsno}) > 0 \text{ and snodp} > 0.01$$

23. *Number of days of rain-on-snow* (nday_rainsnow, day): Accumulated number of days with rain-on-snow events.

$$\text{nday\_rainsnow} = \sum_{i=1}^{t} I_i, \text{where} I_i = \begin{cases} 1 & \text{if rain on snow happens} \\ 0 & \text{otherwise} \end{cases}$$

24. *Cumulative freezing rain* (fzrain_cum, mm): Accumulated 24-hr rainfall if there is rainfall and temperature is above 0°C.

$$\text{fzrain\_cum} = \sum_{i=1}^{t} \text{r ainfall}_i \times 24 \times 60 \times 60 \text{if } (\text{prectot} - \text{precsno}) > 0 \text{ and t2m} < 0$$

25. *Number of days with rain-on-snow* (nday_fzrain, day): Accumulated number of days with freezing rain event.

$$\text{nday\_fzrain} = \sum_{i=1}^{t} I_i, \text{where} I_i = \begin{cases} 1 & \text{if freezing rain happens} \\ 0 & \text{otherwise} \end{cases}$$

Discussion

Currently, CARMA has compiled the herd-specific data at the scale of seasonal herd ranges and has distributed the datasets to caribou management agencies. Additionally, CARMA has initiated summary analysis of the key variables. The MERRA dataset, although only including post-1979 data, will allow us to select climate variables to assess large-scale global oscillations which switch between climate phases at approximately decadal timescales. Over decades these large-scale climate patterns are indexed as switches between positive and negative phases and influence caribou ecology (Couturier *et al.,* 2009; Joly *et al.,* 2011).

To make the dataset readily available for CARMA members, these data have been organized into a menu-driven Microsoft Access® database. CARMA's intention is to provide the original dataset and annual updates through its website (www.caff.is/carma). CARMA is undertaking a series of validations by comparing datasets, for example, growing degree-days derived from MERRA data were compared to the Normalized Difference Vegetation Index (NDVI), a variable that has been used to measure "green-up" patterns on caribou range (Griffith *et al.,* 2002). The climate database has also been used to provide driving variables for CARMA's energy-protein model (Russell *et al.,* 2005) and will be central in CARMA's cumulative effects modeling program.

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Brief Communication

CARMA's integrative modeling: historical background of modeling caribou and reindeer biology relevant to development of an energy/protein model

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Introduction

The objective of this brief communication is to review how development of spreadsheet and computer simulation models of *Rangifer* biology/ecology has influenced construction of the CircumArctic *Rangifer* Monitoring and Assessment (CARMA) energy/protein model, which simulates body weight and condition and reproduction characteristics of a female caribou (*Rangifer tarandus*) in response to environmental inputs and reproductive history. A full description of input variables, driving algorithms and output variables of the CARMA energy/protein model is being written for a peer-reviewed publication. This publication also will be the basis of a manual to assist users as they exercise the model. In this publication we give rationale for algorithms and we justify the hierarchy used to allocate energy and protein resources throughout the model. Also in preparation is a publication that addresses verification of key algorithms and performs a sensitivity analysis of key components of the model. This review is restricted to models specific to *Rangifer* and published since the early 1970s. It covers the scope of input that influenced our modeling process and has import to understanding modeling of caribou biology and ecology in the last 40 years.

Initial models

Two simulation models presented at the first and second International Reindeer/Caribou Symposia were important to opening our appreciation for the potential modeling can play in understanding *Rangifer* ecology by quantifying biologically important relationships. At the first symposium a population-centric model specific to caribou (Bunnell *et al.,* 1975) was presented by Eoin McEwan who discussed how the model simulated caribou population responses to a suite of environmental variables. The objective was to use population trends in decision-making relative to caribou management. Input data were based on driving variables recommended by caribou biologists and managers who attended model development workshops. The workshop approach, led by Fred Bunnell of the University of British Columbia, initially used these experts to provide algorithms that constituted internal "mechanisms" of caribou responses to environmental drivers. To reflect the collaborative nature of its development, the model was fondly termed the "Buda Himimi McPapescaw Model", a title reflecting initials of participating experts. The primary impetus for the modeling approach was a need to better understand drivers of the "bottom-up" components of caribou ecology in order to assess the role of natural predation and hunter harvests on the population. The modeling exercise pointed to a need to better understand physiological responses of an animal to forage availability and digestion of dietary constituents relative to intake, and the energetic costs of migration, foraging, and harassment by flies and mosquitoes. Further, realistic assessments of maintenance and production costs were needed to add systematic responses imposed by environmental variables. These requirements were identified generally for modeling most cervid management systems. A contribution to advance our thinking on some of these identified physiological responses was addressed in a model presented at the second reindeer/caribou symposium in Norway by Swift et al. (1980), which focused on current aspects of rumen function from the cervid perspective and laid out a useful mechanism for modeling *Rangifer* rumen function.

Energy balance models

A significant step forward toward current models was provided by the intensive studies in the tundra biome program, within the International Biological Program (IBP) (Brown, 1975; Brown *et al.,* 1980; Bliss *et al.,* 1981). This program focused on quantifying controls over primary and secondary production in the Arctic and resulted in publications of a range of models. Within the biome programs, energy was an important currency for comparing ecological transactions. Based on their biome work, White et al. (1975) used an energy balance sheet (*i.e.*, a spreadsheet model that balances energy intake with expenditure) to test the hypothesis that coastal Arctic tundra in the vicinity of Prudhoe Bay, Alaska, was sufficiently low in biomass of forage species to limit individual productivity of female caribou. They concluded that to be reproductively successful, caribou were likely dependent on access to upland tundra south of the coastal plain, an area of greater species richness and higher biomass. A spreadsheet model approach was subsequently used to address energy relations in a number of herds. Using a spreadsheet model of energy balance, Boertje (1985) suggested there was no nutritional limitation on caribou of the Denali herd in interior Alaska. Also this form of model analysis was used by L. Camps (in Bergerud *et al.,* 2008) to estimate nutritional influence over forage intake to evaluate calving and early summer range of the George River herd (GRH) for which degradation of the range had likely nutritional consequences (Manseau, 1996).

Energy simulation model

A limitation of spreadsheet energy balance models is that they limit the user's ability to incorporate a larger number of variables and to project outcomes over long time frames using short time steps. Simulation modeling can fill these requirements. Thus, concurrently with development of the energy spreadsheet model for caribou at Prudhoe Bay, Russell (1976) formulated a simulation model that converted behavioral activity through decision-based modeling of caribou feeding cycles to determine energetic consequences of insect harassment superimposed on foraging strategies, again simulating an individual female caribou.

Following IBP funded research through the early 1970s, an evaluation of the modeling approach was made by a team of caribou biologists/ecologists/managers to determine if it could provide a linkage between habitat quality, body condition, and reproductive effort (Klein & White, 1978). To better understand these linkages, more research was recommended, as was the development of a comprehensive model. To that end and based on new energy expenditure estimates of reindeer and caribou, Fancy (1986) developed a Fortran based model for energy balance of female caribou in the Porcupine caribou herd (PCH) and concluded that an animal's control over energy input had a greater impact on balance than controls over energy expenditure. Subsequently, based on the potential for scenario building of the Bunnell et al. (1975) model and expanded algorithms in the Fancy (1986) model, a new model was developed to examine consequences for caribou of possible industrial development associated with drilling for oil in the Arctic National Wildlife Refuge, Alaska. Financial support from US and Canadian governments enabled this caribou modeling effort under the leadership of Fred Bunnell (Hovey *et al.,* 1989; Kremseter *et al.,* 1989). This energy-based model used the most advanced understanding of forage intake (White & Trudell, 1980; Trudell & White, 1981), ruminant physiology, biochemistry, and nutrition to simulate a female caribou driven by environmental variables measured in the range of the PCH (Russell *et al.,* 1993). The resultant PCH energy model was driven by an intake sub-model that produced metabolizable energy input to drive an energy allocation submodel that accounted for expenditures associated with maintenance and deposition in body reserves, gestation, and lactation (Russell *et al.,* 2005). Protein-N inputs and transactions associated with changes in body composition were tracked as a bookkeeping component linked to energy through known stoichiometry (ARC, 1980; Torbit *et al.,* 1985). Components of the model have been verified through applications that emphasize energy expenditure such as energy consequences of low flying fighter jet aircraft (Delta caribou herd: Luick *et al.,* 1996), road and pipeline effects at Prudhoe Bay [Central Arctic herd (CAH): Murphy *et al.,* 2000], integration of nutritional components to determine responses to climate change (PCH: Griffith *et al.,* 2002; Kruse *et al.,* 2004), effects of climate change (PCH: Russell *et al.,* 1996; CAH: Murphy *et al.,* 2000), summer range assessment (GRH: Manseau, 1995), and full integration of components for application to development (*e.g.*, Bathurst caribou herd: environmental assessment of Diavik mine, cumulative effects pilot project, Gunn *et al.,* 2011).

Energy/protein simulation model

A limitation of the PCH energy model was that it did not mechanistically simulate protein and nitrogen dynamics and their interactions with energy when inputs are uncoupled. In particular the ability to explore more flexible use of energy and protein through seasonal changes in nutrition and to produce microbial protein from recycled nitrogen using metabolizable energy derived from highly digestible forages (ARC, 1980; NRC, 2007), like lichens in winter, was lacking. Questions addressing the ability of mushrooms to provide a flush to body reserves in late summer-autumn required a more mechanistic linkage between energy and protein-N dynamics. As in many studies, the energy and protein drain of parasites to the individual caribou needed to be simulated (Gunn & Irvine, 2003).

With the support of the CARMA network the original energy model was modified and expanded to integrate protein transactions. Thus, we now simulate separate but coordinated partitioning of energy and protein-N. The model consists of three sub-models: 1) forage intake (diet selection, logistic controls over eating rate, time allocation); 2) metabolic transactions (rumen/post-ruminal digestion and absorption to predict daily intake of metabolizable protein-N in parallel with metabolizable energy); and 3) energy and protein allocation (partition metabolic nitrogen and energy to meet the animal's protein-N and energy requirements for maintenance, growth, and reproduction). Partitioning of metabolic protein and energy is a complex hierarchical process as shown diagrammatically in Fig. 1. The model simulates maternal protein and fat reserves at the beginning of winter and tracks them through seasonal changes in intake and environmental effects such as snow and icing conditions in winter and forage availability on the calving grounds and post-calving ranges. Although not formally tracked by the model, the simulations take into account use of protein reserves required for intermediary metabolism that could become limiting to fetal growth and milk production (White *et al.,* 2013).

In previous versions of the model, we have not simulated seasonal requirements for growth of antlers and coat. Based on simulations of antler growth by Moen & Pastor (1998) we have now included the energy and protein transactions for growth of both antlers and coat. For the first time nutrient requirements of deer (*Odocoileus* spp.), elk (*Cervus elaphus*), and reindeer/caribou as well as New World camelids are now available (NRC, 2007) and the calculations provide data for validating model outputs. Another validation source is provided by Barboza et al. (2007) in their book on integrative wildlife nutrition. The authors have used a spreadsheet approach to assist the reader gain a quantitative understanding of nutrient interactions in a wide array of animals. This book and reviews by Parker (2003) and Parker et al. (2009) provide further in-depth understanding of nutritional underpinnings of ruminant wildlife ecology independent of the algorithms driving the CARMA energy/protein model.

One of our priorities is to simulate energy and protein-N dynamics of infestation by parasites. Although energy costs of exposure to biting insects is well simulated by the model, metabolic costs of hosting larval stages of them is not well known. Analysis to date suggests the over-winter cost by warble fly larvae to Greenland caribou could be significant (Cuyler *et al.,* 2012). Likewise, we plan to simulate metabolic costs of hosting intestinal parasites that are almost ubiquitous in *Rangifer* (Gunn & Irvine, 2003; Kutz *et al.,* 2004).

Integrating remotely accessed data and scenario building

An added objective of the restructuring of the CARMA energy/protein model is to better support "what-if" scenario analyses applicable to assessment of cumulative effects of climate change and industrial development (Gunn *et al.,* 2013). To more easily enter extant data sets, we use abiotic data from regionally downscaled sites such as NASA's Modern Era Retrospective Analysis (MERRA) website (Russell *et al.,* 2013). MERRA-derived data sets drive seasonal and year-specific abiotic variables such as temperature, wind, and precipitation. From these data users can infer biologically important variables such as snow depth, rain-on-snow and icing events in winter, and incidence of mosquito and warble/nasal bot flies in summer. By calculating growing degree days (GDD) above 0^{o}C from MERRA data users can now derive plant biomass, protein concentration, and fiber levels of dietary important species (Finstad, 2008) throughout summer. A new "dashboard" was added to the model that will allow users to view the entered MERRA data appropriate to the region or the herd of interest. Also, the user can enter new data in order to exercise or "game" with the model. Thus, by driving the model through this dashboard we anticipate it will be easier to determine how climate might induce changes in abundance and quality of *Rangifer* forage plants and how this will affect body weight and body composition. The objective is to provide basic drivers in sufficient detail that

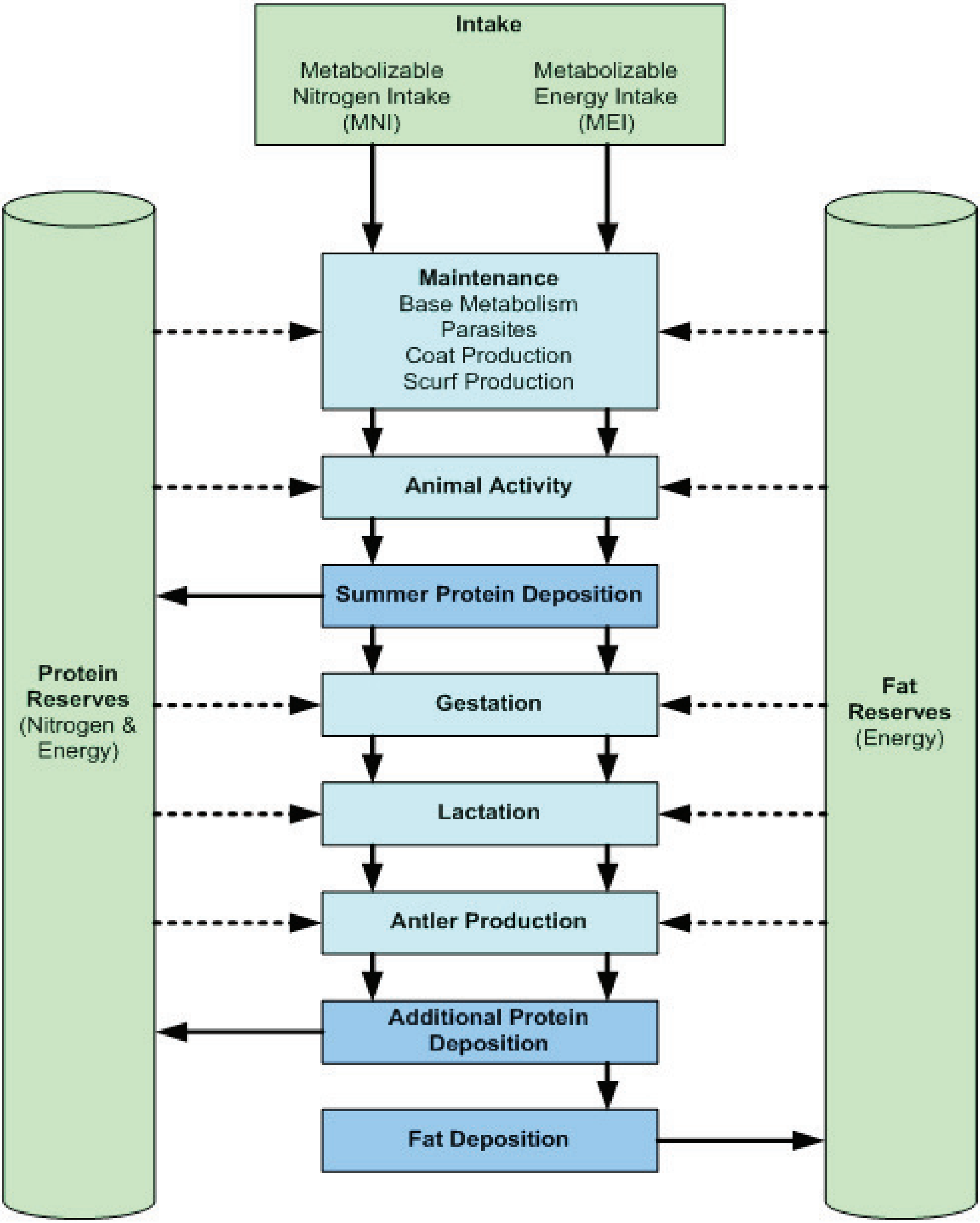

Fig. 1 Daily allocation of protein-N and energy showing the order of priority of each allocation stage.

biologists and managers of most circum-Arctic *Rangifer* populations will be able to drive the model either independently or with CARMA modeling staff. Regionally specific inputs with a 32 year historical record are currently available (Russell *et al.,* 2013).

We have operated under the assumption that the centerpiece of the model is a female caribou or reindeer whose reproductive performance and survival drives population dynamics. Thus, the female must be simulated in sufficient detail so as to provide insight into her responses to environmental variables as well as to emulate measures of body mass, body composition, and reproduction obtained from the field. By making multiple runs of the model with cohorts of varying reproductive history, the user can gain insight into population responses that may not be detectable in the field. From a CARMA perspective, the model will allow biologists and managers to compare caribou productivity in separate populations, it should allow the analysis of changes in female productivity in response to year-to-year variability in environmental drivers, and users of the model will be able to analyze the relative importance of drivers on classes of females in populations undergoing variable abundance.

Acknowledgements

We thank the many caribou and reindeer biologists who have added intellectually to the development of all of these models. Funding from the Canadian International Polar Year program allowed the development of the CARMA energy/protein model. During this funding period many attendees at CARMA annual workshops provided ideas as a component of CARMA's emphasis on free exchange between circum-polar groups to improve comparative analysis of monitoring and to assess whether local trends have circum-arctic relevance. Also important to our revamping of the model was feedback from CARMA participants on what should be the capabilities of the model. The list of contributors to the modeling process is too long to list here, but of special importance to us was input from our colleagues Anne Gunn, Susan Kutz, Christine Cuyler, Brad Griffith, Gary Kofinas, Kathy Parker, Perry Barboza, and Craig Nicolson.

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Brief Communication

CARMA's approach for the collaborative and inter-disciplinary assessment of cumulative effects

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Introduction

One of the most frequent concerns about the future of migratory tundra caribou, *Rangifer tarandus groenlandicus or granti,* are the impacts of the cumulative effects of changing climate and land-use activities across herd's ranges. Assessing cumulative effects is typically a requirement in environmental assessment of industrial developments but policy and technical limitations have hindered development of assessment methods (Duinker & Greig, 2006). Johnson & St.-Laurent (2011) commented on the lack of a methodological framework as one of the reasons for slow progress on cumulative effects. They suggested a framework based on the scaling from individual to population, the relative frequency, and magnitudes of effects and their regulation.

We know quite a bit about individual caribou responses to human activities – interruptions to foraging and displacement of individuals at various distances from the disturbance (Aastrup, 2000; Cameron *et al.,* 2005; Boulanger *et al.,* 2012). However, to scale up from the behavioral responses of individual caribou to the population scale (Johnson & St.-Laurent, 2011) requires baseline information on the 'state' of the individual and population giving consideration to, for example, climate, population density, and genetic structure. At both the individual and population scale, we also have to consider environmental influences, especially weather and climate, which will be additive or compensatory to impacts imposed by human activities.

To scale up the individual's behavioral responses to the population requires being able to estimate the costs to the individual and whether those costs will affect its reproduction and survival. Estimating the costs of a behavioral response is not straight forward; as well as the energy costs of movement and interruption in foraging time, there may also be an effect on diet (energy protein intake) if a displacement puts the individual in a different habitat. Understanding and integrating those relationships between behavior, habitat selection, energy and protein intake relative to reproduction and sur-

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vival is data intensive and interdisciplinary as the understanding is based on ecology, nutritional ecology, and modeling.

Collaborations among researchers and an interdisciplinary focus are among the strengths of the CircumArctic *Rangifer* Monitoring and Assessment (CARMA) which is a network of shared expertise (Russell *et al.,* 2013). CARMA has worked to develop an approach and associated tools for cumulative effects assessment. The principle tools are currently a spatial climate database scaled to herd seasonal ranges, an individual-based energy/protein (E/P) model (Russell & White, 2000; Russell *et al.,* 2005; White *et al.,* 2013) and a population model. CARMA, through international cooperation and collaboration, has also compiled herd-specific databases on caribou condition and health that is essential as input for modelling cumulative effects.

We have two objectives in this brief communication. Firstly, to describe the conceptual approach of using CARMA's tools and secondly, to briefly describe how the different types of input feed into the models and how the two models work together. The model generates corresponding outputs which are subsequently used to project cumulative effects.

Conceptually, the approach is to track how environmental conditions and movements affect the energy and nitrogen intake of a female caribou. The model tracks energy/protein input (*i.e.,* diet and foraging time) and then the model projects how a cow allocates her energy and protein balance for the probability of pregnancy, fetal growth during gestation, and calf growth during lactation. The pregnancy rates and calf survival are linked to a population model, which in turn tracks vital rates and trends in abundance.

Methods

The energy-protein model integrates the state of an individual caribou (*e.g.,* body size and condition) on a particular landscape which has specific attributes (*e.g.,* vegetation type, forage biomass, snow cover, and insect harassment). The approach accommodates responses to human activity as measured through displacement and/or daily activity budgets (*i.e.,* behavioral responses). Those responses can include a reduction in foraging time for caribou close to the development based on measured activity budgets, increased activity costs (*e.g.,* due to avoidance of human activity), and displacement away from the development that may result in foraging in different plant communities which affects diet quantity and quality for the individual caribou. The energy-protein model converts the diet to protein and energy reserves by tracking the physiological steps of digestion and metabolism and then allocates protein and energy to maintenance, protein and fat reserves, body growth, fetal growth, and calf growth (based on milk production).

To describe the different types of input we use the example of a population on its post-calving summer range. The first set of data input relate to the landscape. To describe forage quantity, the model input starts with the relative abundance of plant cover types derived from a vegetation classification typically based on Landsat satellite imagery. The frequency of the caribou's use of those plant cover types is derived from habitat selection modelling such as resource selection functions (Manly *et al.,* 2002). Estimates of above-ground green biomass available in the plant cover types during the growing season are available from satellite imagery (*i.e.,* the normalized vegetation difference index (NDVI)). The energy-protein model tracks 10 plant groups in the caribou diet (moss, lichens, mushrooms, horsetails, deciduous shrubs, evergreen shrubs, forbs, graminoids, standing dead graminoids, and cotton grass heads). The relative abundance of those plant groups among the plant cover types has been described using field measurements which can then be applied to other landscapes on the tundra.

The inputs to the model for forage quality (*e.g.,* nitrogen concentration, digestibility including secondary compounds of shrubs) are based on a relationship that associates published plant nutrients with phenological stage based on growing degree-days and biomass. The model can use as input field measurements of diet or if those data are unavailable, the model can generate a likely diet based on known nutrient requirements, forage biomass, and forage quality.

Growing degree-days, as well as other climatic variables that affect caribou activity patterns (*e.g.,* index of insect harassment) or diet (*e.g.,* mushroom growth index), as input to the energy-protein model are derived from one of CARMA's other tools. We downloaded the retrospective spatial data at the scale of seasonal ranges for all circumpolar caribou herds and developed caribou-relevant variables (Russell *et al.,* 2013). The climate data are available as a spreadsheet and a searchable database organized at the level of seasonal ranges for individual herds. The herd database is available on request to CARMA. The climate data themselves are from NASA's Modern Era Retrospective-analysis for Research and Applications (MERRA) website (http://gmao.gsfc.nasa.gov/merra/).

The next set of model inputs include daily activity budget, which can include those budgets when caribou are responding to disturbances by reduced time spent foraging and increased energy costs of walking or running away. The model inputs also require an assessment of the individual caribou's initial body condition. This is an advantage offered by CARMA which has compiled from historic sources, extensive herd-specific data and metadata on condition. The same databases are also useful as a validation of the model's projected probabilities of pregnancy which the model generates from fall body condition of the cow.

With these inputs, the energy-protein model can run scenarios to examine the possible range of effects of industrial development and or climate. The scenarios can include the degree of changes in distribution as a result of displacement which are tracked through shifts in habitat type (thus diet quantity and quality) and changes in density (tracked by plant biomass) if the displacement changed the relative density of the caribou.

Results and Conclusion

There is complexity in the modelling approach but the integration of spatial data using the habitat selection models has been successfully incorporated into the energetic model during a demonstration project for the Bathurst herd (Nishi *et al.,* 2009; Gunn *et al.,* 2011; Adamczewski *et al.,* In press). A significant advantage of the ability to integrate spatial data is that it allows the inclusion of longer-term datasets such as those held by aboriginal elders. For the demonstration project on the Bathurst herd's range, we were able to work with the Tlicho to include longer-term information on caribou distribution across the landscape based on knowledge from the elders (Legat *et al.,* 2001).

The original energy model (Russell *et al.,* 2005) was linked to a population model to explore the impacts of climate change and development on the Porcupine caribou herd (Kruse *et al.,* 2004). We are presently in the process of linking the current energy-protein model to a "Caribou Estimator", a model that projects populations into the future with the focus on assessing the impacts of harvest policies on the productivity of herds. That linkage will allow decision-makers to consider the health of populations into their harvest management planning.

In the context of cumulative effects, CARMA's approach offers four key features. Firstly, it allows the scaling up from individual to population responses to environmental changes including climate and industrial exploration and developments. Secondly, the energy-protein model is flexible in data input. Predicting the body condition of the individual uses monitoring data or values through literature review and expert opinion. For those inputs with significant uncertainty in their values, a range (or distribution) of values can be provided. Thirdly, the model is adaptive in that it incorporates recent data about typical caribou responses to human development (*e.g.,* a large open pit mine). Fourthly, during an environmental assessment, the approach can assist with decisions about cumulative effects by allowing the relative ranking of the relative effects of different response scenarios based on, for example, degrees of displacement across seasonal and annual ranges (Russell, 2012).

The adoption of individual- to population-scaled modelling in cumulative effects is a recent development although the need for the approach has been long recognized. There are other energy-based models for caribou (White *et al.,* 1975; Boertje, 1985; Camps & Linders, 1989; Fancy, 1986; Bergerud *et al.,* 2008), and Boertje's (1985) model is being used in the recent environmental assessment for a diamond mine in the Northwest Territories (Mackenzie Valley Environmental Impact Review Board public registry http://www.reviewboard.ca/). However, we are not aware of any model that tracks complete energy-protein balance and no other modeling approach designed to address both the cumulative effects of climate change and incremental human activity.

CARMA's tools for cumulative effects assessment work together to couple the state of an individual or population to the cumulative effects of climate change, industrial development, and harvest on circumpolar *Rangifer* herds. To be useful as tools, CARMA's models have to be relatively available and so CARMA is working to ensure that the models are web-based and accessible. Steps such as graphical comparison of alternative model scenarios, modular approach for sharing parameters between herds, built-in capability to edit model inputs in Microsoft Excel®, the ability to make multi-year runs, and the capability for stochastic Monte Carlo simulations are all underway.

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Brief Communication

Modeling energy and protein reserves in support of gestation and lactation: glucose as a limiting metabolite in caribou and reindeer

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Key words: body condition; CARMA; energy/protein model; gluconeogenesis.

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Introduction

In the CARMA (CircumArctic *Rangifer* Monitoring and Assessment) network we account for factors that affect reproduction and growth of an individual caribou through an energy/protein model, which simulates body weight and condition of an individual female caribou (*Rangifer tarandus*) over time (Russell *et al.,* 2005; Gunn *et al.,* 2013; White *et al.,* 2013). However, modeling mobilization of maternal fat and protein to support fetal growth and lactation is complex because energy and protein transactions occur through intermediary substrates (*e.g.*, fatty acids, glucose, amino acids). In the energy/protein model, however, we do not overtly deal with intermediary substrates; rather, we model the amount of body fat and protein that can be measured in the field. CARMA has a large body condition data set for Arctic caribou *(Rangifer tarandus)* populations that allow us to set limits on seasonal fat and protein mobilization and accretion rates. Within the model we estimate daily energy and protein balances, and if negative, we estimate the amount of fat and protein that needs to be mobilized to satisfy demands. During gestation this exercise controls fetal growth, and during lactation it controls milk production and calf growth. As validation we rely on seasonal changes in body composition of female caribou *(R. t. granti)* of the Porcupine caribou herd (PCH) (Gerhart *et al.,* 1996) and experimental data derived from caribou and reindeer *(R. t. tarandus)* fed a known diet and intake during late gestation (Barboza & Parker, 2006; 2008).

Defining labile fat and protein reserves

Body fat constitutes the major energy reserve in *Rangifer* (Reimers *et al.,* 1982; Adamczweski *et al.,* 1987; Gerhart *et al.,* 1996) and we have termed this the labile fat reserve (LFR). A labile protein reserve (LPR) is protein that is

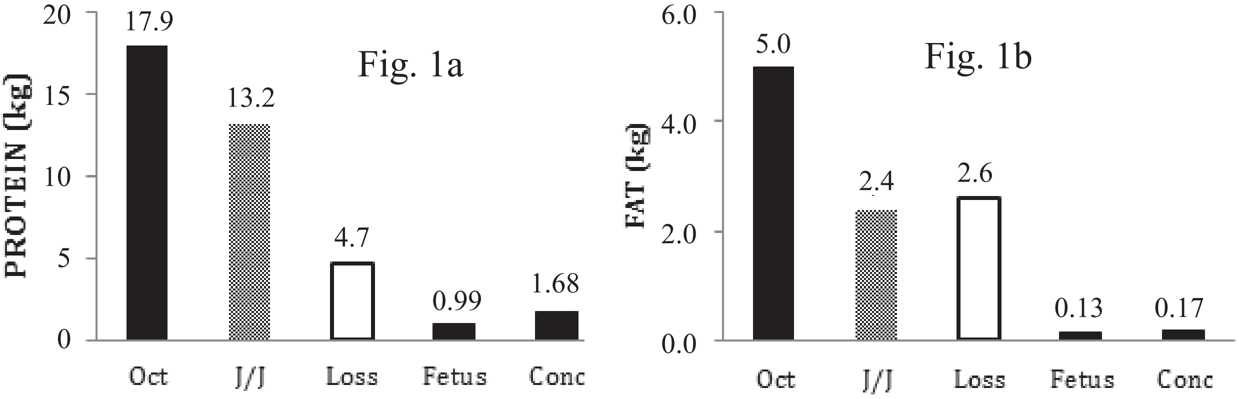

Fig. 1. Mean overwinter loss in body protein (a) and body fat (b) based on body composition measured on females of the Porcupine Herd in October (Oct) and June-July (J/J) (Gerhart *et al.,* 1996). The amount of fetal and conceptus (Conc.) protein (a) and fat (b) were made in the same females in J/J. Note the differences in scales for protein and fat.

mobilized over winter or early lactation. The LPR is difficult to define anatomically or at the tissue level; therefore we quantify it from seasonal changes in body composition. Studies on reindeer and caribou suggest that about 78% of mobilized protein is derived from skeletal muscle and 22% from the viscera (*e.g.*, liver, kidneys, alimentary tract; Reimers *et al.,* 1982; Gerhart *et al.,* 1996). Repeated measures of body composition calculated from estimates of body water using isotope studies (Cameron & Luick, 1972; Holleman *et al.,* 1982; Chan-McLeod *et al.,* 1994; Barboza & Parker, 2008) have import for use with live animals, and these studies on reindeer and caribou (Barboza & Parker, 2008; Parker *et al.,* 2005; 2009) challenge the assumption that the ability of ruminants to catabolize body protein is limited (Reid *et al.,* 1966; Nolan & Leng, 1968). Mobilized body protein as amino acids is used to satisfy gestation and lactation demands, and when oxidized, amino acids also are a source of energy. In the model, we only allow fat and protein to be mobilized down to lower limits ensuring that fat (F*m*) and protein (P*m*) are retained in sufficient amounts for life processes.

In the CARMA energy/protein model, we define F*m* as 2-3% of body weight (Adamczewski *et al.,* 1987; Huot 1989; Gerhart *et al.,* 1996) and consider this to be a fat reserve defended during gestation (FR*g*) and lactation (FR*l*). We set the P*m* defended in gestation (PR*g*) at 13 kg because this value approximates the minimum body protein required for conception (calculated from Gerhart *et al.,* 1997), and it approximates that for puberty in caribou. Several studies show the P*m* during early lactation is lower than at calving by about 1 kg. Actual LPR and LFR vary annually and depend on the levels of body protein and fat of females entering winter. Fat reserves vary independent of protein among populations and are dependent on reproductive history of the female (Parker *et al.,* 2005, 2009). Although body protein is often thought to be only poorly mobilized, data from *Rangifer* populations would challenge this assumption since the seasonal change in body protein is about 26-42% (Huot 1989; Gerhart *et al.,* 1996; Chan-McLeod *et al.,* 1999) giving a LPR of 3-6 kg. The 4.6 kg protein loss (Fig. 1a) exceeds that for fat (2.6 kg, Fig. 1b).

Capture of mobilized reserves by the conceptus

Important to modeling gestation is determining the fractions of LFR and LPR that are captured by the conceptus (fetus plus tissues) in contrast to that used in metabolic processes. Based on data for the PCH we calculated that 36% of the LPR was retained by the conceptus (Fig. 2a), but only 6.5% of the LFR was

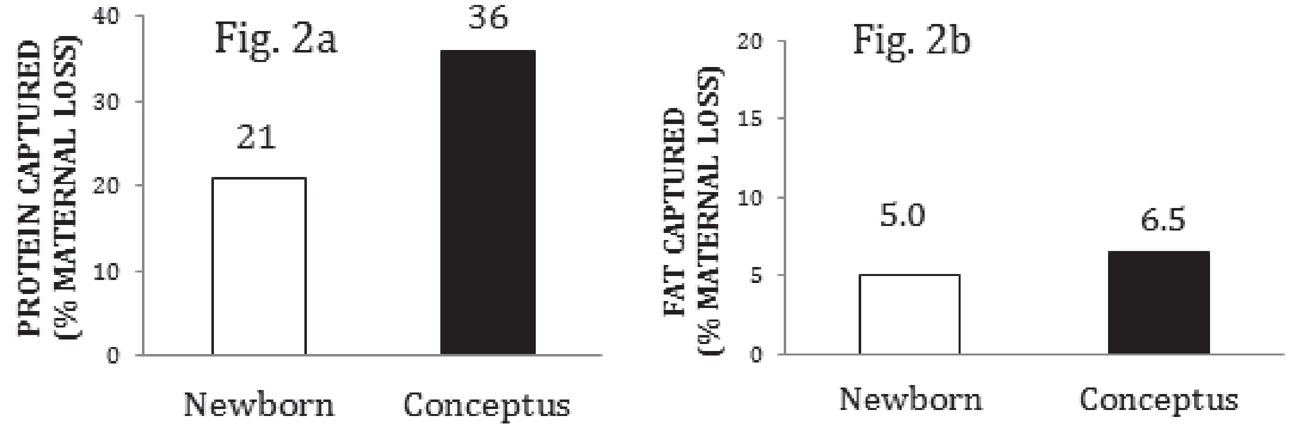

Fig. 2. Percent capture of protein (a) and fat (b) by the newborn fetus and conceptus of caribou of the Porcupine Herd based on the overwinter changes in body composition (Gerhart *et al.,* 1996). Note the differences in scales for protein and fat.

captured (Fig. 2b). The low capture rate for the LFR was expected given that body fat is a major metabolic reserve of the animal. These estimates represent the integral of the entire gestation period and are probably low compared with late gestation when fetal demands increase exponentially. Barboza & Parker (2008) report reindeer and caribou protein and fat balances in females and their offspring at near term and the third week of lactation. We converted their data to absolute values (Fig. 3). Daily, 42% of maternal protein loss (26 g) was captured by the fetus. Of the daily loss of fat, 15% (8.1 g) was captured. Both data sets (Fig. 1, 3) indicate considerable protein-N and fat is not captured and these differences must be accounted for in the model. Although we do not model intermediary substrates in calculating energy and protein-N metabolism, we must include their fundamental processes within the energy-protein algorithms of the model. For the fetus, glucose but not fatty acids constitutes the source of energy for metabolism (Barcroft, 1946), and demand for glucose increases dramatically during the last trimester.

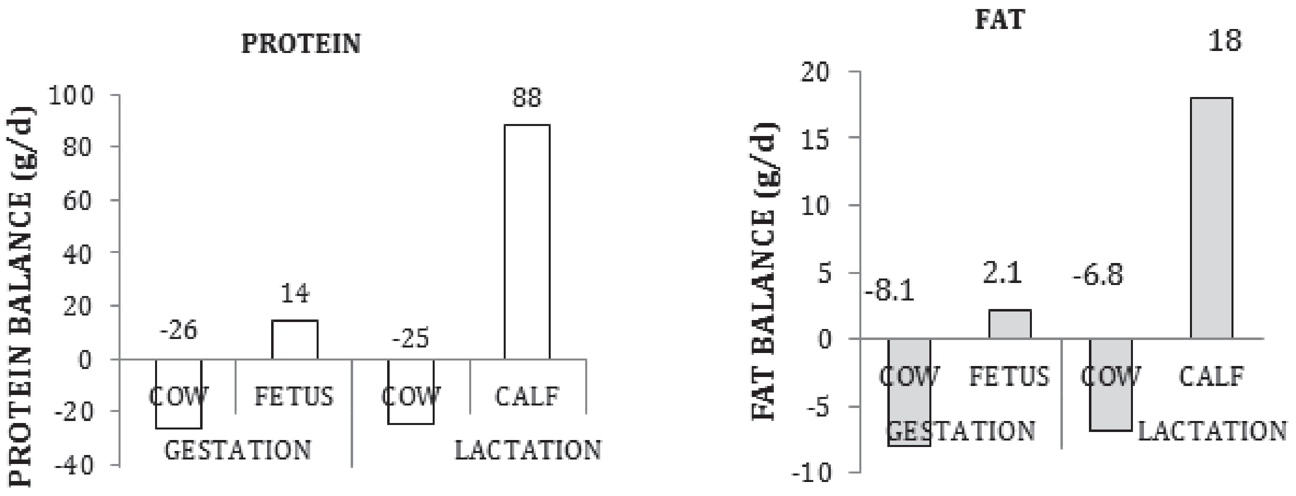

Fig. 3. Balance of protein (a) and fat (b) in caribou at term gestation and at three week lactation in relation to protein and fat deposition of the fetus and calf in caribou given diets of known composition. Calculations are based on data of Barboza & Parker (2008). Stable isotope analysis was used by Barboza and Parker to deduce the maternal versus dietary sources of protein and fat deposited in the offspring.

Table 1. Estimation of glucose synthesis by a caribou at term gestation based on isotope estimates of urea production. At 58 g/d this synthesis would require the metabolism of 105 g protein. Comparison is made with the likely demand for glucose by the fetus. An estimate made from the irreversible loss (IL) of glucose measured in near-term reindeer and caribou minus the glucose IL measured in non-pregnant animals fed at maintenance. A second estimate of fetal demand is calculated from the glucose uptake of the conceptus at term in sheep.

Body weight (kg)	Urea Entry Rate (g/d)	Possible Glucose Synthesis from Amino Acids (g/d)	Fetus Weight (kg)	Fetal Glucose Requirement (g/d)
105	36.8	57.7	6.4	83^1-110^2
Barboza & Parker, 2008	518 mgN/ $kg^{0.75}$.d and BW = 105 kg. Barboza & Parker, 2008	Gluconeogenesis from 100 g protein gives 55 g glucose and 35 g urea. Krebs, 1964	Barboza & Parker, 2008	

[1] 13 g/kg fetus per d based on glucose IL at term minus maintenance glucose IL (McEwan *et al.*, 1976).

[2] 17 g/kg fetus per d (Hodgson *et al.,* 1991).

Gluconeogenic demand

If glucose cannot be met by dietary precursors, it must be met mainly by gluconeogenesis from glucogenic amino acids of the LPR. Krebs (1964) has shown that gluconeogenesis from 100 g of protein results in the formation of 35 g urea (25.4 g urea-N) and 55 g glucose. Thus estimates of urea production can be used to estimate gluconeogenesis during pregnancy (Nolan & Leng, 1968). Barboza & Parker (2008) have made estimates of urea production, termed urea entry rate when isotopes are used. From their estimates we calculate that gluconeogenesis is 58 g glucose/d for this study (Table 1). Synthesis of 58 g glucose uses about 105 g of protein daily. Daily fetal glucose requirements vary from 83-110 g/d for a 6.4 kg fetus (Table 1), suggesting as much as 50% of fetal glucose requirements could be met by protein at terminal pregnancy.

During early lactation glucose is required for lactose synthesis. As an example, for an 85 kg caribou that produces daily about 1.2 L milk, 52 g lactose is synthesized. At higher rates of milk secretion, approaching 2 L/d, the supply of glucose precursors for lactose synthesis can limit lactose synthesis (White & Luick, 1976). In turn, the rate of lactose synthesis exerts a strong control over the level of milk production in ruminants (Annison *et al.,* 1968). Thus, whenever dietary glucose precursors cannot meet glucose demands the LPR is tapped. The minimum LPR, about 1 kg, could supply as much as 550 g glucose that could last for several days as a supplement to dietary precursors. However, this potential glucose supplement would be less because of the competing demand on body protein for synthesis of milk protein. Important to modeling is accounting for the contribution made by dietary protein to protein deposition (NRC, 2007) and to glucose synthesis during gestation and lactation (McEwan *et al.,* 1976; White & Luick, 1976).

Adaptive and ecological influences over energy and protein reserves: work in progress

Interactions between energy and protein must account for animal responses (Chan-McLeod *et al.,* 1994) and for nutrition as a fundamental integrator of *Rangifer* responses to the environment (Parker *et al.,* 2009). Given that reindeer express a greater dependency on body reserves than do caribou in late gestation and early lactation (Barboza & Parker, 2006), the question as to whether caribou and reindeer respond with similarly evolved responses to the use of body reserves relative to resource availability remains to be studied. We assumed the relative use of reserves for the same resource availability would be the same for both subspecies. Finally, the modeling process needs to reflect interacting components of ecological drivers (Russell *et al.,* 1993) while preserving our ability to analyze the implications for intermediary metabolism. Temporal and spatial aspects of phenological and biomass changes on calving grounds of caribou and reindeer that are reported as responses to climate change are complex, but nevertheless phenological mismatches are proposed (Post *et al.,* 2009). A thorough analysis of these environmental effects on intake and reproduction in relation to body reserves is warranted (Gunn *et al.,* 2009; Griffith *et al.,* 2010) given the fundamental role of meeting glucose requirements as we have detailed above.

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Brief Communication

Long-term patterns of abiotic drivers of mosquito activity within summer ranges of Northern Alaska caribou herds (1979–2009)

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Key words: barren-ground caribou; climate change; mosquito activity; NARR data.

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Introduction

Harassment by mosquitoes is an important factor influencing caribou (*Rangifer tarandus*) activity and movement patterns during summer (Kelsall, 1968; White *et al.,* 1975; Russell *et al.,* 1993). Incidence of mosquito activity, and consequently harassment to caribou is spatially and temporally dynamic in response to temperature and wind (Thomson, 1971; White *et al.,* 1975; Dau, 1986). Our knowledge pertaining to mosquito harassment for northern Alaskan herds is limited to the studies done within the ranges of Central Arctic herd from 1973–74 (White *et al.,* 1975) and 1982-83 (Dau, 1986), and the Porcupine herd from 1984–85 (Nixon, 1991). These assessments were done more than quarter of a century ago in only a small portion of these herds' summer ranges. Consequently, those results may not be representative of spatial conditions across entire summer ranges or long-term inter-annual variability, as well as the present conditions. Additionally, there are concerns regarding how warming summer temperatures (Wendler *et al.,* 2010) might influence the conditions conducive for potential mosquito activity that results in harassment to caribou within the summer ranges. Therefore, long-term estimates of mosquito activity that are spatially represented within the entire summer range of a herd are warranted. Objectives of this study were to (1) assess changes in potential mosquito activity in response to climate variability over space and time in northern Alaska, and (2) compare projected intensity of potential mosquito activity across the four Alaskan barren-ground caribou herds of Arctic Alaska.

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Table 1. Algorithm for computing Mosquito Activity Index (MAI) are adapted from Russell *et al.* (1993). MAI is computed for every 3-hourly period using instantaneous surface air temperature and wind speed data from the North American Regional Reanalysis (NARR) gridded climate dataset using this algorithm.

Mosquito Activity Index MAI = TI × WI,
Where TI is Temperature Index (range between min 0 – max 1),
and WI is Wind Index (range between min 0 – max 1)

If instantaneous temperature $T > 18^\circ$ C	→ $TI = 1$
If $T < 6^\circ$ C	→ $TI = 0$
If 6° C $\geq T > 18^\circ$ C	→ $TI = 1 – ((18-T)/ 13)$
If instantaneous wind speed $W > 6$ m/s	→ $WI = 0$
If $W \leq 6$ m/s	→ $WI = (6-W)/6$

Methods

We used the North American Regional Reanalysis data (NARR, Mesinger *et al.,* 2004), a long-term, gridded (0.33°latitude/longitude) climate dataset to estimate patterns in abiotic drivers of mosquito activity over space and time in northern Alaska. NARR surface air temperature and wind speed data were used to estimate potential mosquito activity within the summer ranges of four caribou herds: Western Arctic herd (WAH), Teshekpuk caribou herd (TCH), Central Arctic herd (CAH), and Porcupine caribou herd (PCH).

We computed a "Mosquito Activity Index (MAI)", a theoretical measure of mosquito harassment potential based on instantaneous air temperature and wind speed (Russell *et al.,* 1993; Table 1). We applied this relationship between mosquito activity and its abiotic drivers to every grid cell in the study region using 3-hour climate data, for every day of an a priori defined season (*i.e.*, 1 June to 31 August) for every year over our 31-year study period (1979–2009) to compute daily MAI for each grid. We used the software GrADS Ver 2.0 (Doty, 2011) to visualize and manipulate the climate dataset. The MAI is represented on a scale of zero to one, where zero is absence of mosquito activity and one is the highest potential for mosquito activity if mosquitoes are present.

To meet the first objective, we computed spatially explicit, long-term trends in MAI for northern Alaska. To address the second objective, we averaged the MAI over all grids within the entire summer range of each herd, and within each season (1 June to 31 August) for every year to compute an annual mean, and we compared the annual patterns of MAI between the herds.

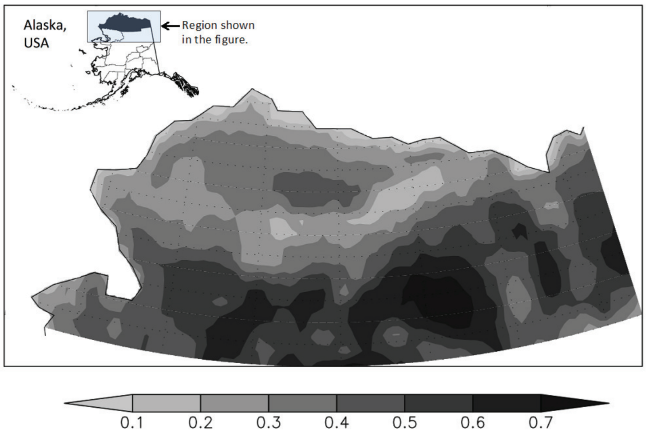

Fig. 1. Average annual MAI for northern Alaska for the period 1979–2009 showing regions of high (dark) and low (light) potential mosquito activity.

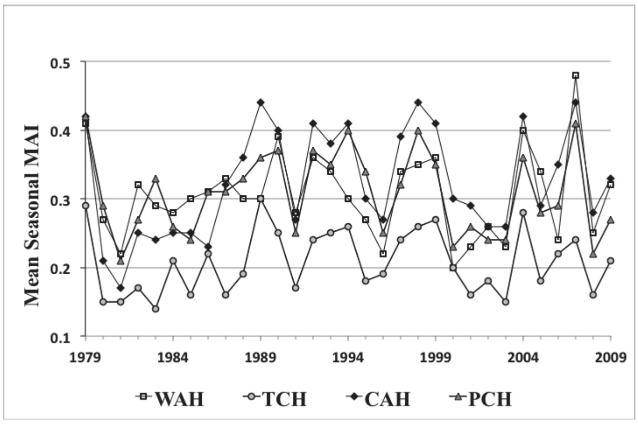

Fig. 2. Mean annual mosquito activity (daily MAI averaged for every season over 92 days, and entire summer range) from 1979–2009 for Western Arctic herd (WAH), Teshekpuk lake caribou herd (TCH), Central Arctic herd (CAH), and Porcupine caribou herd (PCH).

Results

The long-term average of daily MAIs depicts the spatial patterns of potential mosquito activity, over the 31-year period in Northern Alaska (Fig. 1), and highlights the regions of average high and low MAI during 1979–2009. Overall the coastal regions experienced the lowest MAI while regions south of the Brooks Range experienced relatively higher MAI. Comparison of MAI among herds' summer ranges shows similar temporal patterns in peaks and troughs in mean annual MAIs, although differing in magnitude (Fig. 2). The lowest 31-year average annual MAI was for the TCH (0.21 ± SD 0.04); followed by the WAH (0.31 ± 0.06) and PCH (0.31 ± 0.06); and CAH (0.33 ± 0.08). The lower MAI values for the TCH are primarily attributed to consistently higher winds in the summer range for this herd. The highest range of inter-annual variability in MAI was for the CAH.

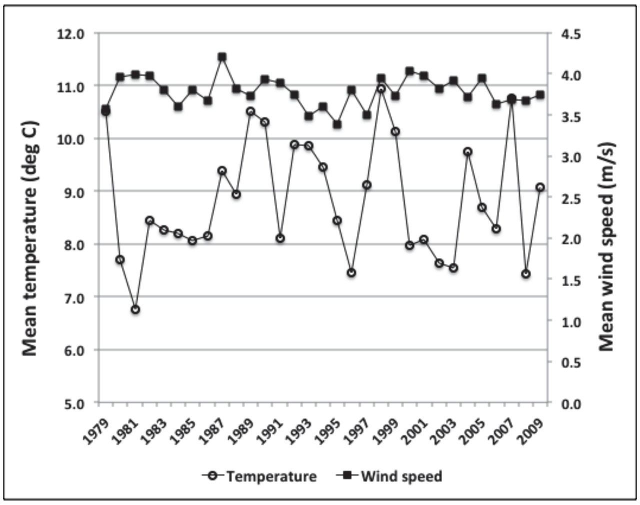

Fig. 3. Mean temperature and mean wind speed for the period 1 June to 31 August for the study region (1979–2009).

Mean summer temperature (June-July-August) was more variable than mean summer wind speed among years (Fig. 3). Changes in temperature contributed most to overall inter-annual variability in the magnitude of MAI for all four herds from 1979–2009. Variability in wind was more related to the spatial aspects such as distance from the coast and topography, and at any given location wind patterns were more constant than temperature over the entire analysis period. In our analysis the differences in spatial wind patterns drove the differences in magnitude of MAI among the four herds for any given year.

Discussion

Our analysis indicated a marked variability in mean seasonal MAI among the 31 years and the relative magnitude of this variability was consistent among the four herds. Overall the TCH summer range hosted the least conducive weather conditions for mosquito activity. This was due to coastal proximity, where cooler temperatures and higher winds prevailed. While TCH and CAH summer ranges are very similar in characteristics, with proximity to coast and absence of upland habitats, the wind patterns within the CAH range experienced higher seasonal variability; hence the difference in overall MAI values for the two herds (Fig. 2).

Although the MAI estimates were based on a theoretical model derived from empirical relationship between abiotic drivers of mosquito activity (Russell *et al.,* 1993), the potential "hot spots of mosquito activity" (*i.e.*, the areas showing highest mean MAI in the summer ranges for 1979–2009), and the potential "mosquito relief areas" (*i.e.*, areas of low MAI that we identified), warrant further verification with field data. Our analysis demonstrates a novel way for using climatology datasets to model long-term spatial and temporal distribution of potential mosquito harassment for caribou. We suggest these results could be useful in understanding seasonal distribution and movement of herds, implications for energetics and body condition from year to year, and comparison of relative effects on different herds.

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Brief Communication

Differences in parasite diversity, prevalence, and intensity assessed through analyses of fecal samples from two West Greenland caribou populations

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Key words: caribou; *Eimeria;* fecal analysis; Greenland; *Marshallagia;* nematodirine; *Ostertagia;* parasites; *Rangifer*; *Teladorsagia.*

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Introduction

Many northern communities depend on caribou (*Rangifer tarandus groenlandicus)* as a dietary staple and for their contributions to northern economies and cultures. In *Rangifer* sp., experimental removal of gastrointestinal helminth parasites has been associated with increased fat reserves and pregnancy rates, and it is generally accepted that the effects of these parasites on individuals can influence population dynamics and herd sustainability (Albon *et al.,* 2002; Stien *et al.,* 2002).

Study Area

West Greenland is home to several populations of caribou, with Akia-Maniitsoq (AM = 24, 000) and Kangerlussuaq-Sisimiut (KS = 98, 300) the two largest (Cuyler *et al.,* 2011). The ranges of these two populations extend east-west from the Greenland ice cap to the coast and from 62°N to 68°N, with KS the more northerly. They are isolated from other caribou populations by extensive fjord systems and from each other by the Sukkertoppen icecap, although movement between herds is theoretically possible (Cuyler *et al.,* 2011). While AM and KS are genetically related (Jepsen *et al.,* 2002), have adjacent ranges and lack predators (Melgaard, 1986), they are unique in their exposure to other ungulate species through human-mediated translocations and importations. The KS population shares its range with muskoxen *(Ovibos moschatus wardi)* translocated from east Greenland (Pedersen & Aastrup, 2000), whereas AM has seen transient introductions of cattle, horses, sheep, and goats

(Norlund, 1936; Rose *et al.,* 1984), and the importation of semi-domestic Norwegian reindeer *(R. t. tarandus)* into a neighbouring range (Cuyler, 1999). The presence of introduced ungulates may have influenced parasite diversity in these native caribou populations.

Gastrointestinal parasite diversity is poorly defined for animals in west Greenland, with the two previous studies focusing solely on ungulates from the KS range. Raundrup (2005) sampled muskoxen from this area and reported the presence of 'trichostrongyle' eggs and larvae which she attributes to *Ostertagia (Teladorsagia) circumcincta,* however she provides no morphological or molecular data to support this. Korsholm and Olesen (1993) did a smaller, but more detailed study not only on muskoxen (n = 5), but also caribou (n = 5) from the KS range. In caribou, these authors reported the abomasal nematodes *Teladorsagia circumcincta* and *Marshallagia marshalli,* as well as *Eimeria* oocysts. Along with these parasites, muskoxen were also positive for *Nematodirella longissimespiculata, Nematodirus helvetianus,* and *Moniezia expansa.* To date, no similar work has been done for AM, although warbles *(Hypoderma tarandi)* and nose bots *(Cephenemyia trompe)* have been evaluated in both populations (Cuyler *et al.,* 2012). The purpose of our work was to do a fecal survey of gastrointestinal parasites in the AM and KS caribou populations. We expected that, due to their common ancestry and neighboring ranges, there would be no differences between parasite species present in the two populations.

Methods

Collections for this study took place during International Polar Year as part of an initiative by the CircumArctic *Rangifer* Monitoring and Assessment (CARMA) network (Kutz *et al.,* in press). Adult female caribou (≥ 3 years), subadults (1–2 years) and their calves-at-heel (<1 year) were collected opportunistically through scientific hunts from Mar. 29 – Apr. 13, 2008 (AM) and Mar. 3 – 17, 2009 (KS). Rectal fecal samples were collected, frozen, and transported to the University of Calgary where they were processed using a modified Wisconsin double centrifugation-sugar flotation technique (Hoar *et al.,* 2009). Eggs and oocysts were identified to family or genus based on morphology (Foreyt, 2001). Eggs identified as having typical 'strongyle-egg' morphology can belong to a number of different genera, however in arctic and subarctic environments the abomasal nematodes *Ostertagia gruehneri* and *Teladorsagia boreoarcticus* are the most common in *Rangifer* sp. (Hoberg *et al.,* 2001; Kutz *et al.,* 2012). Also producing 'strongyle-type' eggs, *Teladorsagia circumcincta* has been previously reported in KS by Korsholm & Olesen (1993). This identification should be revisited as it preceded the description of the morphologically similar *T. boreoarcticus* and it is now recognised that most descriptions of *T. circumcincta* in arctic hosts are mistaken (Hoberg *et al.,* 1999; Kutz *et al.,* 2012).

Results

We observed differences between AM and KS in regards to the diversity of parasites present, prevalence (percent of samples positive), and intensity (eggs per gram feces from positive samples) (Table 1). 'Strongyle-type', Nematodirinae and Anoplocephalidae eggs, and *Eimeria* oocysts were found in both populations. *Marshallagia* spp. eggs were common in KS samples, but absent entirely from AM.

Prevalence and intensity of 'strongyle-type' eggs were much higher in AM than KS, which may reflect differences in species diversity, host genetics or health status, timing of sampling, or sampling years (Baker *et al.,* 2001; Baker *et al.,* 1998; Dunn, 1969; Irvine *et al.,* 2001). Nematodirinae eggs, produced by the small intestinal nematodes *Nematodirus* spp. or *Nematodirella* spp., were found in both populations, but with

Table 1. Prevalence [% positive samples] and intensity [EPG/OPG; Median (Min. – Max.)] of gastrointestinal parasite eggs and oocysts of fecal samples from animals of the Akia-Maniitsoq and Kangerlussuaq-Sisimiut caribou populations. Significant differences between herds ($p \leq 0.05$), within an age class, are indicated by *, Fisher's Exact (prevalence) or Wilcoxon (intensity) tests.

		Strongyle-type		*Marshallagia* spp.		Nematodirinae		Anoplocephalidae		*Eimeria* spp.	
Herd	*n*	Prevalence	Intensity	Prevalence	Intensity	Prevalence	Intensity	Prevalence	Intensity	Prevalence	Intensity
Akia-Maniitsoq (2008)	47										
Calves	6	83.3*	5.0 (0.2-11.7)*	0.0*	N/A	33.3	0.5 (0.2-0.8)	33.3	35.3 (18.9-51.6)	0.0	N/A
Subadults	7	71.4*	1.2 (0.2-2.5)	0.0*	N/A	0.0	N/A	28.6	8.4 (1.5-15.4)	0.0	N/A
Adults	34	50.0*	0.8 (0.2-2.8)*	0.0*	N/A	0.0*	N/A	5.9	3.5 (0.3-6.8)	2.9	32.1
Kangerlussuaq-Sisimiut (2009)	48										
Calves	9	22.2*	0.3 (0.3-0.3)*	55.6*	0.5 (0.3-1.0)	44.4	1.8 (0.5-4.4)	22.2	30.5 (13.6-47.4)	0.0	N/A
Subadults	4	0.0*	N/A	75.0*	0.8 (0.3-1.3)	50.0	0.8 (0.5-1.0)	0.0	N/A	0.0	N/A
Adults	35	11.4*	0.3 (0.3-0.5)*	62.9*	0.5 (0.2-1.0)	54.3*	0.3 (0.2-2.2)	11.4	9.5 (2.3-42.0)	2.9	0.3

very different patterns of distribution. In AM, these eggs were only present in samples from calves, as is expected for ruminants, however they were common in all age classes of KS. This suggests that there may be different species of nematodirines present in these two populations, which requires further investigation.

The presence of *Marshallagia* spp. in KS, but not AM, is likely the result of spill-over from introduced muskoxen (Pedersen & Aastrup, 2000), however the effects of climatic conditions cannot be discounted. *Marshallagia* spp. in particular is more commonly found in arid regions (Meradi *et al.,* 2011), and the drier climate of the KS region may be more suitable for this parasite than the AM range (Tamstorf, 2001). At least two varieties of *Eimeria* oocysts were present in each population, possibly *E. hreindyria, E. mayeri,* or *E. rangiferis* as these have all been reported previously in *Rangifer* sp. (Guðmundsdóttir, 2006). Differences in prevalence and intensity of *Eimeria* spp. between the populations were non-significant, which may be a true result, or be due to our small sample size. This may also apply to our Anoplocephalidae results.

Conclusions

Our research has demonstrated that, although the AM and KS caribou populations are closely related genetically (Jepsen *et al.,* 2002) and geographically, there are significant qualitative (species diversity) and quantitative (prevalence, intensity, and distribution across age classes) differences in their parasites. Identification of parasites in this study was limited to morphological examination of eggs and, as such, we are only able to report to genus or family level. Species-level identifications, currently underway, are expected to reveal greater differences in diversity and provide deeper insight into the influence of historical and contemporary factors on the parasite community of these caribou populations.

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