De Beers

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APR 15 2003

MACKENZIE VALLEY ENVIRONMENTAL IMPACT REVIEW BOARD

April 11, 2003

Mackenzie Valley Environmental Review Board
Attn: Mr. Gordon Wray
P.O. Box 938
Yellowknife, NT
X1A 2N7

BY HAND

Dear Gordon

Re: The record for the De Beers Canada Mining Inc. Snap Lake Diamond Project Environmental Assessment- Socio-economic and Wildlife Questions

De Beers hereby submits further information to address and alleviate the Review Board's concerns with respect to the public record and the hearing for the Snap Lake Diamond Project.

Relevant information related to the Boards wildlife questions are provided in this submission. De Beers hereby notifies the Board of its intention to submit the outcome of the regional Elders Caribou Workshop to be held at Snap Lake for their consideration. Due to community input, this workshop is confirmed to be held during May 12-15, 2003, and De Beers anticipates submitting key outcomes of the Board on May 16th, 2003. De Beers considers that this workshop may provide important information from holders of traditional knowledge that will serve to further reduce uncertainty in predicted impacts to wildlife and mitigation measures.

A written response to all outstanding socio-economic questions will be provided on Monday, April 14th.

Please contact the undersigned if you have any questions.

Yours truly,

Robin Johnstone
Senior Environmental Manager

encl.

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Caribou

Q (6): Request 1 and 2

A number of factors influence the Bathurst caribou herd. Those factors include both natural factors (e.g., weather, range conditions, insect abundance) and human activities (e.g., hunting morality, mine disturbance). The relationships of those factors and how they cumulatively affect the Bathurst caribou herd population are not clearly understood. A lack of information about the factors affecting caribou populations, as well as an incomplete understanding of the relationships among those factors and caribou populations, has resulted in the MVEIRB requesting additional modelling to reduce uncertainty related to cumulative effects to the Bathurst caribou herd.

De Beers concurs with the MVEIRB's interest in decreasing the uncertainty related to impact predictions. However, De Beers also agrees with the Lutsel K'e Dene First Nation that, *"it is extemely difficult to accurately predict the bahaviours and movements of animals (particularly with the spatial and time scales of data with which De Beers is making predictions)."*(Information Request 2.2.13)._Their analysis was that scientific predictions are "limited in their utility". In this response De Beers addresses the merits of the modeling requested, provides an alternative, and addresses uncertainty in caribou impact predictions.

De Beers and their consultants acknowledge that models can be useful to educate scientists about the potential outcomes of different hypothetical scenarios, and provide an additional tool for making decisions regarding potential impacts to populations or systems. However, if the assumptions and inputs of a model contain little or no empirical data, then the predictions are highly questionable. Arguments about the validity of unsubstantiated models typically result in casting suspicion about the credibility of scientific methods among the public, and do not help decision makers with issues related to the significance and uncertainty of impacts. In addition, using models with little or no validation has a high risk of undermining the credibility of scientific analysis with First Nation communities and holders of Traditional Knowledge.

There a number of issues related to the applicability and reliability of the CWS energetic (Russell et al. 2000) and Diavik friction models (Wierzchowski et al. 1998). The energetic model is based on the behaviour, habitat and forage conditions of the Porcupine caribou herd, which are not directly applicable to the ecology and diet of the Bathurst herd. In addition, the model is not accessible for review (except as an abstract: Russell et al. 2000) or broadly available. The Diavik friction model may reasonably capture the effect of disturbance on caribou during the northern migration when caribou are moving deliberately towards the calving grounds. However, the model likely does not reflect the change in energetics during the post-calving (southern) migration as caribou movement is less predictable and not in a straight north-to-south direction. In addition, there are a number of key assumptions in the model which have no empirical support, which include:

- arbitrary values for friction modifiers associated with terrain types;
- migration distance is a function of straight-line distance multiplied by 1.5;

- we estimated the mean percentage of the Bathurst herd that may be within a hypothetical zone of influence of any of the mine sites at any point in time, based on observations at Ekati;
- we determined the reduction in foraging opportunity related to a caribou within the zone of influence, based on monitoing programs at Ekati; and
- we compared the predicted foraging opportunities (assuming impact) to the opportunities of a circumstance with no mines.

The first step was to estimate the fraction (percentage) of the Bathurst herd that may be within the zone of influence of any project at any point in time. Figure 1 shows the annual mean percentage the herd per two-week interval within the Ekati study area (1,600 km^2) from mid-April through mid-October during the past six years. These data reflect the estimated number of caribou located within 20 km of the Ekati mine footprint during surveys flown approximately once per week (BHP Billiton 2003). Although the zone of influence on caribou behaviour from the Ekati mine appears to be within 5 km of the footprint (BHP Billiton 2003), we will consider the area within 20 km of a project to be the zone of influence for the Ekati mine and each of the other three mines (again, we have likely overestimated the cumulative effects). From 1997 to 2002, the average fraction of the Bathurst herd (estimated at 350,000 animals) within the Ekati study area over this six month period was 0.6% (solid horizontal line in Figure 1). From mid-April through mid-October caribou have the greatest probability of encountering mining projects within their annual home range as this is the period of their northern and post-calving migrations. During the winter, caribou are typically distributed near and within the treeline (Gunn et al. 2002), and we will assume (based on empirical results from Gunn et al. 2002) that caribou exposure to mining activities from early through late winter is negligible.

which is linked to nutritional condition (energetics), survival and reproduction (Robbins 1993). To accomplish this we first need to define a foraging (or feeding) unit for caribou that is based on range conditions from mid-April through mid-October (spring – autumn) with no mine-related disturbance. For the purposes of this discussion we define one "feeding unit" as the amount of food that one caribou will consume during spring – autumn with no impact from mining. Under this condition, there are 350,000 feeding units available for caribou during this six month period of the year.

Four years of monitoring activity budgets of caribou within the Ekati study area have suggested that caribou within 5 km of the footprint spend approximately 10 - 15% less time feeding than animals greater than 5 km from the mine (BHP Billiton 2003). These data include the effect of natural annual variation in insect abundance. Based on this information, we assumed that the feeding unit for individuals within the zone of influence of each mine (i.e., within 20 km from the footprint) would be reduced by 10%. Therefore, under this condition, a caribou foraging while within the zone of influence of a mine will ingest food at a rate equivalent to 90% of the original value assuming no disturbance.

Scenario 1 and Scenario 2 were then used to predict the loss of feeding units for the Bathurst caribou herd due to the cumulative effects of the four mines on caribou behaviour.

Scenario 1 (based on mean number of caribou)

Mean fraction of herd not exposed to mines = 97.6%
Mean fraction of herd exposed to four mines = 2.4%
Number of feeding units for caribou not exposed to mines:
97.6% x 350,000 units = 341,600 feeding units
Number of feeding units for caribou exposed to mines:
2.4% x 350,000 units x 0.90 = 7,560 feeding units
Overall reduction of feeding units = 840 feeding units (or 0.23% of caribou food intake assuming no mines; see Figure 2).

Scenario 2 (based on mean + 2 SE number of caribou)

Mean fraction of herd not exposed to mines = 93.6%
Mean fraction of herd exposed to four mines = 6.4%
Number of feeding units for caribou not exposed to mines:
93.6% x 350,000 units = 327,600 feeding units
Number of feeding units for caribou exposed to mines:
6.4% x 350,000 units x 0.90 = 20,160 feeding units
Overall reduction of feeding units = 2,240 feeding units (or 0.64% of caribou food intake assuming no mines).

Assumed in Analysis	Likely Condition
effects from Ekati mine would be the same at the other three mines	effects at other mines are likely less than at Ekati because of smaller footprints and less activity (e.g., underground mining, shorter haul roads)
assumed that any caribou within 20 km of the mines would be affected	observed zone of influence of mine on caribou behaviour appears to be not more than 5 km
Scenario 2 uses 2 standard errors above mean to estimate the fraction of the herd exposed to the mines	fraction of the herd exposed to the mine would be the observed mean

There are a number of natural environmental factors that can influence the foraging behaviour, energetics, survival and reproduction of the Bathurst caribou population. Food abundance and quality on summer and winter ranges have been determined to be important elements in tundra caribou population dynamics (Reimers 1983; Skogland 1990; Post and Klein 1999). Extreme weather events such as late spring snowfall or late snowmelt can influence access to food and result in lower calf weights or delayed parturition which influences survival of young (Skogland 1984; Adamczewski et al. 1987; Cameron et al. 1993). High insect abundance can also decrease forage intake, milk production and calf growth and survival (Helle and Tarvainen 1984; Russell et al. 1993). Factors that influence adult female food intake from summer through winter also determine pregnancy rate and parturition rate. These factors include insect abundance, frequency of predation risk (natural and human) and adverse weather conditions (time and amount of snowfall, snow hardness).

Based on this information, we consider that the amount of variation in year-to-year calf production is much more likely explained by natural environmental factors than by the overall reduction (0.24-0.64%) of food intake for caribou that are exposed to mining activity. Although the analysis indicates that the cumulative effect of the mines on caribou feeding behaviour could have some influence on the nutritional condition of 2 – 6% of the Bathurst herd each year, the effect is unlikely to be detectable from the cumulative influences of winter and summer forage quality, weather conditions, insect abundance and predation.

Q (6): Request 3

The MVEIRB requested De Beers to undertake a population modeling exercise to predict the long-term probability of fluctuations in caribou population size using three different scenarios that include harvest rates and mine-related effects. De Beers believes that we can not proceed with this type of modeling for the following reasons.

As discussed above (Request 1 and 2), there are a number of natural environmental factors that can influence the dynamics of the Bathurst caribou population. In addition, there is a complex interaction between habitat and caribou foraging and movement

Ferguson, M.A.D., L. Gauthier, and F. Messier. 2001. Range shift and winter foraging ecology of a population of Arctic tundra caribou. Canadian Journal of Zoology 79:746-758.

Gunn, A., J. Dragon, and J. Boulanger. 2002. Seasonal movements of satellite-collared caribou from the Bathurst herd. Final Report to the West Kitikmeot Slave Study Society.

Helle, T., and L. Tarvainen. 1984. Effects of insect harassment on weight gain and survival in reindeer (*Rangifer tarandus*) calves. Rangifer 4:24-27.

Messier, F., J. Huot, D. Le Hanaff, and S. Luttich. 1988. Demography of the George River caribou herd: evidence of population regulation by forage exploitation and range expansion. Arctic 41:279-287.

Post, E. and D.R. Klein. 1999. Caribou calf production and seasonal range quality during a population decline. Journal of Wildlife Management 63:335-345.

Reimers, E. 1983. Reproduction in wild reindeer in Norway. Canadian Journal of Zoology 61:211-217.

Robbins, C.T. 1993. Wildlife feeding and nutrition. Academic Press, New York, NY. 352 pp.

Russell, D.E., A.M. Martell, and W.A.C. Nixon. 1993. Range ecology of the Porcupine caribou herd. Rangifer Spec. Issue No. 8. Tromso, Norway: Nordic Council for Reindeer Research.

Russell, D.E., R.D. Cameron, R.G. White, and K.L. Gerhart. 2000. Mechanisms of summer weight gain in northern caribou herds [abstract]. Pp. 148 in Proceedings of the 8th North American Caribou Workshop, Whithorse, Yukon, Canada. R. Farnell, D. Russell, and D. van de Wetering (eds.). Rangifer Spec. Issue 12. Tromso, Norway: Nordic Council for Reindeer Research.

Skogland, T. 1984. The effects of food and maternal conditions on fetal growth and size in wild reindeer. Rangifer 4:39-46.

Skogland, T. 1990. Density dependence in a fluctuating wild reindeer herd: maternal vs. offspring effects. Oecologia 84:442-450.

Wierzchowski, J., C. Gates, and K. Lloyd. 1998. Evaluation of the effects of human activities on caribou migration in the Lac de Gras area, Northwest Territories, with the use of friction models. Technical Report for Diavik Diamonds Project Environmental Effects Report for Wildlife.

The variation in natural annual mortality of grizzly bears within the Regional Study Area ranges from 0.00 to 0.232 males per year and from 0.008 to 0.180 females per year. For both sexes, the natural variation in grizzly bear mortality is equal to the combined annual mortality rate for both sexes and would range from 0.008 to 0.412 individuals per year.

Q (7) Request 2.

The following is a response to the request by MVEIRB to provide an indication of the rate of mine-related mortality of grizzly bears that would fall within the range of natural variation.

Rates of mine-related mortality within the Snap Lake Diamond Project regional study area would have to be less than 0.412 individuals per year (or less than 10.7 bears over the 26 year life of the mine) in order to fall within the natural range variation estimated by McLoughlin et al. (2003a).

Q (7) Request 3.

The following is a response to the request by MVEIRB to provide estimates of mine-related mortality of grizzly bears from other projects in the NWT.

Since the beginning of construction in January 1997 (about 6.5 years), there has been one mine-related grizzly bear mortality at the Ekati mine. This amounts to a mine-related mortality rate of 0.15 individuals per year.

At Diavik, in the 3 years since the beginning of construction in January 2001, there have been no mine-related grizzly bear mortalities.

At Lupin, from 1986 – 2002, there have been 8 mine-related grizzly bear mortalities over a 16-year period. This amounts to a mine-related mortality rate of 0.5 individuals per year.

Q (8) Request 1.

The following is a response to the request by MVEIRB to provide an estimate of how cumulative effects from existing industry-related mortality may influence the population viability of grizzly bears in the SGP (based on McLoughlin et al. 2003b)

The annual removal rate of grizzly bears (i.e., the combination of all industry, subsistence, non mine-related problem kills, and harvest related mortality) in the study area used by McLoughlin *et al.* (2003b), which is approximate to the area of the SGP, was 13.4 bears per year. Population modelling suggested that at that removal rate there was a 10% risk of the population showing a 25% decline over the next 50 years. The risk of a 25% decline in the grizzly population in the next 50 years increased to over 40% in a scenario where another 6 bears were removed by the hunting or industry from the population annually (i.e., 13.4 or 19.4). McLoughlin *et al.* (2003b) stipulated that an annual removal rate above 15 bears would be detrimental to the population.

It is noteworthy that this analysis is based on the total annual removal rate (i.e. industry and non-industry related kills). However, of the annual removal rate of 13.4 bears per

McLoughlin, P.D., M.K. Taylor, H.D. Cluff, R.J. Gau, R. Mulders, R.L. Case, and F. Messier. 2003b. Population viability of barren-ground grizzly bears in Nunavut and the Northwest Territories. Arctic 56:177-182.