A literature review for monitoring rare and elusive species, and recommendations on survey design for monitoring boreal caribou

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FINAL REPORT

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1.1. Introduction

Effective wildlife management requires monitoring changes in the spatial distribution of species, their population size and their population trend (Williams *et al.* 2002; Sinclair *et al.* 2006). Reliable estimates of these population characteristics are necessary for determining current population status and providing a basis for evaluating management decisions in an adaptive management framework (Holling 1978). Obtaining reliable estimates of population distribution, size or trend, however, is not a trivial task. Surveys designed to collect the relevant data are often costly and challenged by environmental factors (e.g., weather, land cover) and animal behaviours that can cause imperfect detection of all individuals, leading to estimates that are biased and/or imprecise (Williams *et al.* 2002). Obtaining reliable estimates is particularly challenging for rare and elusive species. Low densities typically confound standard monitoring methods causing low encounter rates and the resulting estimates are usually too imprecise to effectively inform management (Thompson 2004). Yet, rare or elusive species are frequently a primary concern for management because many such species are often designated as threatened or endangered and/or deemed to be data-deficient (Drever *et al.* 2012). Because of their conservation concern, significant research effort continues to be directed at developing reliable, cost-effective monitoring methods for rare and elusive species (Thompson 2004; Conroy *et al.* 2008; Johnson *et al.* 2013; Royle *et al.* 2013). Here, we review recent advances in methods for monitoring rare and elusive species. We begin with an overview of common sampling designs and methods for collecting relevant data. We then examine statistical methods for estimating the population characteristics of spatial distribution, size and trend.

1.2. Sampling Designs and Data Collection

Effective and cost-efficient monitoring of rare and elusive species requires survey designs and data collection methods aimed at increasing encounter and detection rates to achieve estimates with acceptable levels of precision (i.e., coefficients of variation [CV] ≤ 20%; Pollock *et al.* 1990). Standard survey designs generally involve defining a study area that contains the population of interest, delineating sample units (e.g., grid cells, line transects) within this area, then surveying sample units to collect the relevant data (e.g., presence of sign; counts of individuals). For monitoring species distribution, this type of design may be sufficient, particularly in sampling frameworks that employ repeated site visits to explicitly model the probability of species detection (e.g., occupancy modelling, see *Section 1.3*). For estimating population size and trend, this design may provide acceptable estimates for species that have a relatively uniform distribution within the study area, even for species considered to be at low density (e.g., grizzly bears [*Ursus arctos*], Mowat & Strobeck 2000, Boulanger *et al*. 2004; jaguars [*Panthera onca*], Sollmann *et al.* 2011; tigers [*Panthera tigris*], Karanth *et al.* 2011). In many of these successful cases, primarily carnivores, the low detection rates usually associated with low-density species are overcome by the use of attractants or bait stations within sample units (Schlexer 2008; see *Data Collection* below). However, for those species where the use of attractants is impractical (e.g., rare, wide-ranging herbivores), standard survey designs generally produce population estimates with low precision, particularly when detection rates fall below 0.2 - 0.3 (i.e., the probability of detecting a species at a site given that it is present is <20-30%; Williams *et al.* 2002; McDonald 2004; Proctor *et al.*

2010; Wingard *et al.* 2011). Moreover, standard survey designs generally yield imprecise estimates of population size and trend for rare and elusive species that are spatially clustered (Thompson 2004). This imprecision arises when the survey data contains a high number of zeroes (i.e., sample units with no detections), although this problem may be minimized by stratification or post-stratification of the survey area (see below). In the following section, we focus on sampling designs aimed at efficiently estimating population size and trend for rare, spatially clustered species.

Sampling Designs for Estimating Population Size and Trend in Rare, Spatially Clustered Species

A number of alternative sampling designs have been developed to overcome the known challenges of surveying rare, spatially clustered species. Most of these designs focus on directing survey effort to where species are, or are predicted to be, to increase detection rates. Model-based stratification is one such approach, where the study area is partitioned based on habitat suitability models (Edwards Jr *et al.* 2005; Rachlow & Svancara 2006; Allen *et al.* 2008; Le Lay *et al.* 2010). Survey effort is then directed to areas with the highest predicted habitat quality for the target species. This approach is most successful when the species-habitat relationship is well understood (i.e., model prediction is high) and species distribution with their preferred habitat is relatively uniform. Conversely, it is less successful for species that are further spatially clustered within in their preferred habitat and well below the carrying capacity of their habitat (Rachlow & Svancara 2006).

Other survey approaches for rare and elusive species are variations of adaptive sampling designs (Thompson 2012). In these designs, an initial subset of sample units in the study area is selected by a probability-based process (e.g., simple random sampling). Observations on this initial subset then inform the selection of further units in a second phase of sampling. In adaptive cluster sampling, if units in the first phase of sampling meet some condition (e.g., species presence), then additional units in close proximity are selected in the second sampling phase (Smith *et al.* 2004; Brown *et al.* 2013). This design is thus most effective when the target species are highly clustered. While adaptive cluster sampling has shown promise in simulations, the few field applications have shown mixed results (Smith *et al.* 2003; Noon *et al.* 2006; Sullivan *et al.* 2008), primarily due to difficulties in defining the proximity neighbourhood (Salehi & Smith 2005). Thus, the design requires further empirical testing, particularly on terrestrial vertebrate populations. Another form of adaptive sampling is sequential sampling where units are sampled in a probabilistic process until an *a priori* criterion, or stopping rule, is met (Green 1970; Christman 2004; Salehi & Smith 2005). An example of a typical stopping rule is reaching a minimum number of units where the species is detected to reach a desired level of precision (Green 1970). As with adaptive cluster sampling, sequential sampling has few practical applications with respect to surveying vertebrate populations, with most examples limited to surveys for insects (e.g., Kolodnyhirsch 1986; Serra *et al.* 2013). Moreover, sequential sampling may not be cost-efficient over a large geographic area if the stopping rule results in a large number of surveyed units.

Recently, Conroy *et al*. (2008) suggested a two-phase adaptive sampling approach for surveying rare and spatially clustered species. In the first phase, a subset of sample units are selected in a probabilistic process and then surveyed to determine occupancy (e.g., presence / absence of the target species). In the second phase, a subset of the “occupied” units are intensively surveyed to determine per-unit abundance. The first-phase detection data and the second-phase abundance data are used to model the detection rate and overall population size is estimated by using the abundance-detection relationship. The design is not necessarily restricted to estimating population size and can be adapted to efficiently estimate species occupancy patterns (Pacifici *et al.* 2012). In empirical testing, Mathewson *et al*. (2012) used the design to generate precise estimates [CV < 15%] of endangered warblers. We do note, however, that the design is strongly dependent on sample units being "closed" (i.e., no immigration / emigration) during the first phase of sampling (DeMars & Boutin 2013).

Data Collection

Methods for collecting data necessary for estimating population size, trend and distribution generally fall into one of two categories: direct and indirect methods. Direct methods are those where the target species is either seen or heard. Examples of direct methods include aerial surveys, trapping devices for small mammals, and point count for birds. In general, direct methods have had limited effectiveness in collecting sufficient data for many rare and elusive species. As a consequence, recently developed indirect methods have been increasingly used to detect difficult-to-observe species (Taberlet *et al.* 1999; Waits & Paetkau 2005; Long *et al.* 2008). Indirect methods include camera-trapping (Karanth & Nichols 1998; Sollmann *et al.* 2011), observing tracks and other natural sign (Smallwood & Fitzhugh 1995; Linnell *et al.* 2007), and the collection of hair or scat to identify unique individuals using DNA genotyping methods (Woods *et al*. 1999; Harris *et al.* 2010; Hedges *et al.* 2013). These methods are designed to increase detection rates of the target species (versus direct methods) and all are broadly applicable for surveys designed to assess spatial distribution or relative abundance. Estimating true abundance, however, generally requires methods that can identify unique or previously "marked" individuals. Thus, track surveys typically cannot be used for estimating population size. In species where individuals can be identified by unique markings, camera data have been successfully used to estimate population size (Karanth & Nichols 1998; Plhal *et al.* 2011; Zero *et al.* 2013). Recent statistical advances in camera trapping have also resulted in models where individual recognition is not required (Rowcliffe *et al.* 2008), although this approach has not been tested on rare species where low detection rates may confound parameter estimation (Foster & Harmsen 2012; Rowcliffe *et al.* 2013). In recent years, population estimation for rare and elusive species has increasingly relied on data from DNA-based methods. This type of estimation has been used on a wide variety of species including grizzly bear (*Ursus arctos*, Woods *et al.* 1999), European wild cat (*Felis silvestris*, Kéry *et al.* 2010), Asian elephants (*Elaphas maximus*, Hedges *et al.* 2013), mountain goat (*Oreamnos americanus*, Poole *et al.* 2011), and Sitka black-tailed deer (*Odocoilues hemionus sitkensis*, Brinkman *et al.* 2011). DNA methods, however, are not infallible as genotyping errors can lead to biased estimates if these errors are not explicitly taken into account (Lukacs & Burnham 2005; Lampa *et al.* 2013). In the next sections, we review how data derived from direct and indirect methods have been used in analyses for assessing species distribution, population size and trend.

1.3. Species Distribution

Accelerating rates of landscape change and the predicted effects of global climate change have placed an increased emphasis on monitoring changes in species distribution (Tilman 2001; Staudinger *et al.* 2013). As a result, many approaches have been developed to model species distribution. Generally, most approaches rely either on 'presence/absence' data or 'presence-only' data. For presence/ absence data, occupancy modelling has become the dominant paradigm in the last decade (MacKenzie *et al.* 2002; Mackenzie & Royle 2005; Kéry 2011). In the typical framework of occupancy modelling, the study area is first partitioned into sites (commonly grid cells) and a random subset of sites is selected for survey. Thus, a key advantage to occupancy designs is that the sample unit becomes a site rather than an individual animal (as in mark-recapture studies – see *Section 1.5* below), which allows the scale of the survey to easily be expanded to larger extents. Selected sites are repeatedly surveyed for animal presence within a short time such that the sites are considered closed (i.e., no births, deaths, immigration or emigration). This design yields a detection history for each site and the detection data is used in a likelihood-based model to estimate both the probability of detection and the probability of site occupancy. Habitat covariates for predicting heterogeneity in detection and occupancy can easily be incorporated into the model (MacKenzie *et al.* 2002; Royle 2006). Critical to the design is approximating site size to the seasonal home range of the target species and selecting a sampling period where the population is closed (Bailey *et al.* 2013). While occupancy modelling has been applied to a wide range of taxon such as birds (Dorazio *et al.* 2006), fish (Comte & Grenouillet 2013), marine mammals (D'Souza *et al.* 2013), carnivores (Karanth *et al.* 2011) and ungulates (Poley *et al.* 2014), parameter estimation may be difficult for rare species when detection rates are below 0.2 (MacKenzie *et al.* 2006). For rare species, a more successful approach may be to incorporate occupancy modelling into a two-phase sampling design (see above; Pacifici *et al.* 2012).

Species distribution has also been assessed using models requiring ‘presence-only’ data. This suite of models includes resource selection functions (RSFs; Manly *et al.* 2002), maximum entropy models that link the distribution of species occurrences to environmental features (program Maxent, Phillips *et al.* 2006) and environmental niche factor analysis (Hirzel *et al.* 2002). From a monitoring perspective, these models have generally been used to generate predictive maps of a species’ space use based on modelled habitat suitability. Because habitat suitability is a gradient and absence is not explicitly incorporated into these models, the inferences from these models fundamentally differ from those of occupancy. Whereas as occupancy models estimate the probability of occurrence at a given site during a specific time period, presence-only models estimate the probability a site is selected given that it is encountered (Lele *et al.* 2013). The sampling framework in which the data are collected for many of these models also prevents robust inferences beyond the areas where the data were collected. That is, the data are collected as a function of individual animal movement and therefore are not collected in a sampling framework based on probability theory. Distributional inferences are further complicated by the validation process for many presence-only models. Validation is usually performed by assessing whether species occurrence is proportional to the predicted suitability of their habitat (Boyce *et al.* 2002) rather than assessing whether predicted areas of high suitability actually contain the target species. These key differences make presence-only models less powerful than occupancy models for monitoring changes in species distribution.

Recent advances in spatially explicit capture-recapture (SECR) models offer a further promising approach to monitoring species distribution. Under the SECR paradigm, inference can be expanded beyond the traditional sampling grid area by sampling sub grids or using two stage sampling approaches where “core areas” with higher coverage are sampled to estimate population size and density (see below) and secondary areas are sampled to assess distribution and broader-scale density (Conroy *et al*. 2008, Efford & Fewster 2013). This approach allows for an assessment of distribution which is similar to occupancy, but without the subjectivity and potential issues with defining plot sizes in occupancy models (Efford & Dawson 2012). The main advantage of SECR models in this context is that scale of movement and detection probabilities are estimated directly from the underlying mark-recapture data as opposed to occupancy where only detection is estimated. Therefore, the underlying sample unit for a SECR model is the estimated home range of detected individuals rather than the sampling plot of the occupancy model. Through the use of two-stage sampling methods, SECR methods are used to estimate movement and detection for sub-areas with less-intensive sampling for larger scale inference across regional areas. If habitat covariates exist it is also possible to develop RSF-type models that model density surfaces to therefore assess factors that might influence distribution and densities of the target species within survey extents (Miller *et al*. 2013; Royle *et al.* 2013). Finally, radio-telemetry data can also be incorporated into SECR analysis to assist in assessing animal movements during sampling (Royle *et al.* 2013). Much of the research into SECR methods has been oriented toward single-session estimates, although recent work has generalized the SECR method to allow estimates of apparent survival and trend from multi-year surveys (Chandler & Clark 2014). We further note that the development of SECR methods is relatively recent and therefore there are no published studies of its use with ungulates; however, it has been used extensively with carnivores and other species.

1.4. Population Size

Developing effective methods for efficiently estimation population sizes of wild animals has long been an active area in ecological research. Because wild animal populations can rarely be censused (i.e., the entire population counted), particularly species that are rare and/or elusive, methods have focused on accounting for the imperfect detection of all individuals. Here, we focus on two major modelling approaches currently used for estimating population size: distance sampling and mark-recapture. We also briefly review mark-resight models, forward-looking infra-red approaches and sightability models, particularly as they pertain to ungulate populations. We note that all of these approaches become problematic when detection rates are low and therefore we further emphasize the importance of sampling design in population size estimation (Couturier *et al.* 2013).

Distance Sampling

Abundance and/or density estimation from distance sampling primarily uses designs employing line transects, point stations, or trapping webs (Buckland *et al.* 1993; Thomas *et al.* 2010). Within these designs, the distance from the line, point or web centre to a detected individual is measured and a detection function is estimated to determine the size of the area sampled. A primary assumption of distance sampling is that individuals located on the line, point or center of the trapping web are detected perfectly (i.e., detection = 1.0). Combination mark-recapture and distance sampling approaches allow unbiased estimates when sightability on the line is less than zero (Borchers *et al*. 1998, Laake *et al*. 2008). Distance sampling has had mixed results when estimating population sizes of rare and elusive species, primarily due to the effects of small sample sizes. When sample sizes are sufficient (i.e., > 60 detections) then acceptably precise estimates can be achieved (e.g., de Tores & Elscot 2010); however, when sample sizes are low, estimates can suffer from low precision (Seddon *et al.* 2003; Wegge & Storaas 2009; Williams & Thomas 2009) or the detection function cannot be estimated (Thiessen 2009). Combining multiple years of data can potentially offset low sample sizes from single surveys as long as methods are standardized and appropriate covariates are collected (Buckland *et al*. 2004).

Mark-recapture

Capture-mark-recapture (CMR) methods have a long history in population size estimation dating back to the development of the Lincoln-Petersen estimator almost a century ago. Significant advances have been made since its inception, including the development of an omnibus software program for running the most current modelling approaches (program MARK; White & Burnham 1999). In its basic formulation, a sample of individuals is marked and released back into the population. After a period of mixing, the population is resampled and total population size is estimated by dividing the total number of marked individuals by the proportion of marked individuals collected in the second sample (Williams *et al.* 2002). Model formulations have progressed beyond simple closed population models to include 'robust' designs that incorporate demographic data such as survival (Pollock *et al.* 1990; Williams *et al.* 2002). Design is critical to the success of CMR experiments as precision is influenced by the proportion of the true population that is marked, the number of animals recaptured and the number of sampling sessions (Pollock *et al.* 1990; Rees *et al.* 2011). For estimating population size of rare and elusive species, CMR methods have been particularly promising when used with DNA-based sampling methods (Kéry *et al.* 2010; Gopalaswamy *et al.* 2012; Hedges *et al.* 2013). Using multiple methods to collect DNA data has helped offset sample size issues with species that occur at lower densities (Boulanger *et al*. 2008, Kendall *et al*. 2009, Gervasi *et al*. 2012).

One of the main recent developments in CMR has been the advent of spatially explicit capture-recapture methods (SECR; Efford *et al*. 2009, Efford 2011, Efford & Fewster 2013). The advantage that SECR has over classic mark-recapture methods is that the spatial coordinates of capture sites are taken into account. From this data, a detection function is fit to the mark-recapture data to estimate the probability of detection of a species at its home range center and the distance from the home range center where a species has a detection probability that is greater than 0. Using this information, population density on the study area is estimated without the issues of "closure violation" that challenge estimation with traditional closed CMR models (Otis *et al*. 1978). Early empirical testing suggests that SECR models have similar or better statistical performance than CMR models (e.g., higher precision; Blanc *et al.* 2013; Efford & Fewster 2013). The main potential issue with SECR is highly non-circular home ranges, which can create bias (Ivan*et al* 2013ab); however, this can be offset if covariates are used to account for variation in movement rates and/or densities due to habitat or geographic features. The SECR approach has been adapted to line transect sampling and it therefore can be used even when sampling is not at discrete traps or sites (Efford 2011).

Mark-resight

Mark-resight models have been used extensively for estimation of ungulate populations (Bear *et al*. 1989, Neal *et al*. 1993, White 1996, Wittmer *et al*. 2005). The usual approach has been the radio-collaring of a segment of the population followed by aerial surveys where marked and unmarked animals are observed; however, other approaches such as paint-ball marking have been used to create a marked population for surveys (e.g., Mahoney *et al*. 1998). Mark-resight methods differ from mark-recapture methods in that animals are marked usually in an initial session (i.e., new unmarked animals encountered during subsequent surveys are not marked as with mark-recapture methods). Mark-resight methods have been generalized to include estimators that allow movement to and from the survey area between sampling sessions (Neal *et al*. 1993) as well as unequal detectability of animals in the surveyed population (Minta & Mangel 1989, Bowden & Kufeld 1995). All of these estimators are contained in program NOREMARK (White 1996). Recently, estimators that allow flexible modelling of sightability based on individual covariates (when it is possible to identify individual marked animals during the survey) and temporal covariates (McClintock &White 2009; 2010) have been developed and incorporated into program MARK (White & Burnham 1999). The main requirement for mark-resight methods is having enough marked animals, and high enough resighting probabilities, to obtain adequate precision of estimates. This usually requires multiple surveys to obtain adequate sample sizes of resightings of marked animals.

Estimators that utilize information from radio-collared animals and mark-recapture data have been developed to estimate density. Ivan *et al*. (2013a) developed an estimator that uses estimates of the mean location of animal detection on the sampling grid (using mark-recapture methods) to estimate residency (using information from radio-collared animals). This approach provides an estimate of density that can be compared with SECR methods. Simulation studies suggest that it provides robust inference; however, it does require that a substantial number of animals are collared in the study area.

Forward-looking Infrared Aerial Surveys

Aerial surveys conducted with a forward-looking infrared (FLIR) camera have also been used to estimate population size in ungulates, particularly those species inhabiting areas where sightability is low (Bernatas & Nelson 2004; Kissell & Nimmo 2011; Franke *et al*. 2012; Carr *et al*. 2013). These surveys generally use a line-transect sampling design where the FLIR camera is mounted on the underside of the aircraft to detect thermal heat emitted from animals situated along each transect. The cameras generally have sufficient resolution to differentiate among species of large mammals within a given study area and the FLIR field of vision usually yields a transect width of ~100-m (Kissell & Nimmo 2011; Franke *et al*. 2012). Estimates of detection rates for FLIR surveys have generally been high (e.g., 89% for bighorn sheep [*Ovis canadensis]*, Bernatas & Nelson 2004; 95% for white-tailed deer [*Odocoileus* virginianus], Kissel & Nimmo 2011), which has facilitated its use with distance sampling to achieve acceptably precise estimates (Carr *et al*. 2013). However, the utility of FLIR surveys in estimating population size of rare and patchily distributed species over wide geographic areas has not been tested. For such species, extensive flying time – and hence survey cost – may be necessary to achieve sufficient samples sizes due to the narrow transect width.

Sightability Models

Developed primarily for aerial surveys of ungulate populations, sightability models use a sample of marked individuals (e.g., radio-collars or tags) in a mark-resight framework to estimate a correction factor to adjust raw counts of population size (Gasaway *et al*. 1986; Samuel *et al.* 1987; Steinhorst & Samuel 1989; Gilbert & Moeller 2008). In simple formulations, a single correction factor is applied to the raw counts; however, more complex models using logistic regression have been developed to account for differential sightability due to environmental and behavioural variation (Fieberg & Giudice 2008). Sightability models have had some success for estimating size for low-density ungulate populations (McIntosh *et al.* 2009) but have been problematic for species that live in heavily forested environments where detectability is low (DeMars & Boutin 2013).

1.5. Population Trend

For this section, we predominantly focus on monitoring population trend as a directional change in population size over time. We do note that management objectives often include monitoring trends in species distribution. Changes in distribution can be monitored through repeated occupancy surveys and recent advancements in occupancy methods include dynamic multi-state models that incorporate estimates of site colonization and extinction (McKenzie *et al*. 2006; Bailey *et al*. 2013). Changes in distribution often reflect changes in population size (He & Gaston 2000). This relationship however, is not straight-forward and is particularly problematic across large landscapes and for species that are spatially clustered (He & Gaston 2007; Hui *et al*. 2009). For example, for group-living species the number of animals per group may decline while the number of groups on the landscape may stay relatively constant; thus, a population could decline while its spatial distribution remains unchanged (McLellan *et al*. 2010). This process would result in a high-risk strategy of monitoring population change because change may not be detected until a rapid contraction in distribution is observed. Occupancy models that consider counts of animals rather than presence-not detected (Royle and Nichols 2003) may be more sensitive to changes in group size; however, the use of this approach assumes that individual groups can be counted adequately during surveys. For wide-ranging species with annual home ranges much larger than occupancy plot sizes, short-term changes in occupancy may reflect temporal variation in annual home range use rather than distributional changes related to changes in abundance. Also, with respect to plot size, unbiased occupancy estimation requires a large "plot size" (Efford & Dawson 2012) and thus inference from occupancy will indicate larger temporal changes in distribution. Therefore, tracking changes in occupancy as a surrogate for population size may result in limited power to detect smaller, short-term change.

Monitoring population trend can be directly estimated by changes in population size estimates obtained from sequential surveys. Imprecision in size estimates, however, can make this approach less than straightforward. Open CMR models provide a more powerful approach by simultaneously estimating yearly population size, trend (λ), apparent survival, and emigration / immigration (Pollock 1982). New developments in model fitting such as AIC model selection (Burnham and Anderson 1998), and estimation models such as the Pradel model (Pradel 1996) in program MARK (White and Burnham 1999) have provided potential ways to confront issues of precision through parsimonious model fitting. The Pradel model estimates apparent survival, recapture rate, rate of additions, and population rate of change (λ). Apparent survival estimates losses from the population (emigrants or deaths) and rate of additions estimates the number of births and immigrants at time j+1 per individual at time j. Population rate of change is the population size at time j+1 divided by the population size at time j. If λ is equal to 1 the population is stable, if it is negative the population is decreasing, and if it is greater than 1 the population is increasing. It is estimated as the summation of apparent survival and rate of additions. Each of these parameters can be time varying (producing estimates for approximately each sample session) or constant (not changing for the duration of sampling). Holding some parameters constant while allowing others to vary with time can test different hypotheses about why a population changes through time. For example, holding apparent survival constant while allowing the rate of additions to vary with time tests the hypothesis that population rate of change is being driven by additions as opposed to apparent survival (Schwarz 2001, Franklin 2002). If all parameters are constant, then a stable age distribution is assumed in a similar fashion to telemetry-based estimates of trend using the Lotka equation (Eberhardt *et al*. 1994, Hovey & McLellan 1996). In addition, parameters of the Pradel model can be constrained to be a function of individual covariates or temporal covariates. Franklin (2002) provides an excellent discussion of the Pradel model including examples of its various uses.

Besides CMR methods, population trend can be estimated using demographic models such as those used in population viability analyses (Boyce 1992; Morris & Doak 2002) or life table analyses (Krebs 2008). Many of these approaches, however, require estimates of vital rates which may not be easily attainable for rare and elusive species.

Other less data-intensive methods have been used to indirectly model population trend. Hatter & Bergerud (1991) used the ratio of adult survival to recruitment to estimate trend in moose and the method has been applied to other ungulate populations (Hervieux *et al.* 2013). While this approach is appealing in the terms of simplicity, it does make a set of assumptions regarding the symmetry of survival rate and recruitment estimates. For example, it assumes that annual female:juvenile ratios are an unbiased estimate of annual recruitment so that recruitment from this measure is directly comparable to annual rates of survival. The other issue with indirect estimates of trend is that λ can be difficult to interpret if there is not a baseline estimation of population size for reference. Consequently, initial efforts should be made to estimate population size – and periodically thereafter – to corroborate indirect trend measurements. Because the Hatter & Bergerud equation has been used for monitoring boreal caribou populations, we further discuss its use in the *Boreal Caribou Literature Review*.

We note that if there are baseline estimates of population size, survival estimates, and recruitment rate estimates then it is possible to fit multiple-data source models to further model demography and population trends (Buckland *et al*. 2004, Johnson *et al*. 2010, Boulanger *et al*. 2011). These approaches do not require annual surveys or annual measurements from any of the demographic indicators. They can accommodate sample biases with indicators, such as the effects of differential survival of calves and cows on calf-cow ratios, and can also incorporate harvest data (Boulanger *et al*. 2011). This approach utilizes all the data sources in a unified analysis therefore maximizing inference when compared to stand-alone interpretation of single data sources.

1.6. Traditional and Local Ecological Knowledge for Monitoring Rare and Elusive Species

Traditional knowledge systems employ a variety of approaches for monitoring wildlife populations though none are specifically targeted to monitoring rare and elusive species *per se* (Berkes *et al.* 2000; Berkes 2008). Rather, traditional monitoring methods are primarily aimed at understanding population changes in harvestable species (Moller *et al.* 2004). These methods include catch-per unit effort (CPUE; Moller *et al.* 2004), body condition indices (Kofinas *et al.* 2003; Parlee *et al.* 2014), evaluations of breeding success, and qualitative assessments of population size (Moller *et al.* 2004). We review these methods in terms of their applicability to monitoring species distribution, population size and trend.

Species Distribution

Traditional and local knowledge can be a valuable source of information for understanding and monitoring species distribution. Because traditional knowledge extends back in time over generations, this information has been successfully used to track historical changes in species distribution (Huntington 2000; Huntington *et al.* 2011; Santomauro *et al.* 2012). Further, traditional knowledge has been an important source of information for species that are little understood by modern science due to the difficulty in monitoring these species with contemporary methods (e.g., Greenland shark [*Somniosus microcephalus*]; Idrobo & Berkes 2012). Traditional knowledge of species distribution has also compared favourably to distributional models derived from modern scientific methods. Polfus *et al.* (2014) demonstrated that a distributional map for the northern ecotype of woodland caribou generated from traditional knowledge produced similar predictions to one generated from an RSF model. This relationship suggests that traditional knowledge can be a valuable source of distributional information for areas currently deficient in data derived from modern methods (e.g., for ungulates, areas where animals have not been radio-collared nor formally surveyed). Moreover, traditional knowledge could be integrated into survey designs that incorporate pre-stratification (i.e., pre-identification of areas of high and low habitat quality) to increase the precision of parameter estimates.

Population Size

CPUE has been used by indigenous hunters as a proxy for population size or density of the target species (Moller *et al.* 2004). This approach assumes that CPUE increases with increasing prey density, an assumption that may hold true at low prey densities. However, this assumption likely fails at high prey densities as the number of animals harvested per hunter will eventually reach a satiation point. CPUE is also likely more applicable to species with a relatively uniform distribution on the landscape. For species that are spatially clustered with considerable variation in animal density, CPUE may not reflect actual changes in species distribution, particularly if hunter effort is primarily directed at areas of high density while distributional changes are more noticeable outside these high density core areas (Moller *et al.* 2004).

Population size is also monitored by many indigenous cultures through the use of qualitative assessments (Moller *et al.* 2004). These assessments include visual estimates of animal numbers and aural estimates determined by the amount of sound generated by group-living species (e.g., the number of geese in a flock). As with CPUE, qualitative assessments of population size may be biased if estimates are derived from a species core areas (i.e., areas of high density), which may be more insulated from numerical changes than outlying areas (Moller *et al.* 2004).

Population Trend

Body condition measures such as body fat, animal size, and subjective judgements of movement ability (e.g., walking with a limp) have been used by indigenous cultures to infer information about habitat conditions (i.e., food availability), which are then linked to population dynamics (Kofinas *et al.* 2003; Parlee *et al.* 2014). For example, Gwich'in hunters assess the quality of female caribou by the "fullness" of the animal's rump, whiteness of its mane, and the quantity of back and stomach fat (Kofinas *et al.* 2003). Dene hunters assess the condition of internal organs to assess for diseases that if widespread, give an indication that the ecosystem is "out of balance" (Parlee *et al.* 2014). Information gained from body condition, however, may be biased because hunters target individuals deemed to be in good condition (Kofinas *et al.* 2003; Wray & Parlee 2013). Moreover, body condition may not directly relate to whether a population is increasing or decreasing (Moller *et al.* 2004), because of time lags or lack of synchrony between changes in body condition (from density dependence) and population responses.

Breeding success (i.e., the number offspring produced in a given year) has also been used to index population trend (Berkes 1982 cited in Moller *et al*. 2004). This index, however, is often correlated with maternal body condition (Parker *et al.* 2009) and thus it is subject to the same potential biases as body condition. That is, breeding success does not reliably track an increasing or decreasing trend in population size.

2. Monitoring Boreal Caribou: Literature Review

2.1. Introduction

The boreal ecotype of woodland caribou (*Rangifer tarandus caribou*) typifies the difficulties associated with monitoring rare and elusive species. Throughout their distribution, boreal caribou occur at low densities (<3 /100 km^2; Thomas & Gray 2002; Courtois *et al.* 2007; Arsenault & Manseau 2011) and are often clustered into small groups for most of the year (~3-10 individuals; Stuart-Smith *et al.* 1997; Rettie & Messier 1998; Culling & Culling 2013). They generally inhabit mature conifer forests where their cryptic colouration and tendency to remain motionless at the approach of aircraft (Ruttan 1960 *cited in* Rock 1988; C. DeMars, *personal observation*) interacts with canopy cover to make caribou difficult to detect visually (Cumming & Beange 1987). This lack of detection is compounded by low densities and spatial clustering, which further reduces encounter rates. These factors make survey methods typically used for other ungulates (e.g., stratified random block; Gasaway *et al.* 1986) impractical because they produce estimates with very low precision (e.g., Rowe 2006). In the last few years, more research has been directed at developing reliable methods for effectively monitoring boreal caribou abundance (e.g., Carr *et al.* 2012; Hettinga *et al.* 2012; DeMars & Boutin 2013) and this avenue of research has been designated a high priority in the recently released federal recovery strategy (Environment Canada 2012).

In this section, we review historical and current methods used to monitor boreal caribou across their distribution. We follow the same framework established in the previous section, first reviewing methods used to delineate and monitor the distribution of boreal caribou populations then reviewing methods for estimating population size and trend.

2.2. Monitoring Caribou Distribution

Boreal caribou populations are predominantly managed at the range scale (Alberta Woodland Caribou Recovery Team 2005; Environment Canada 2008; Environment and Natural Resources 2010a; BC Ministry of Environment 2011). Here, we assess monitoring caribou distribution in terms of the ranges of individual populations. In the federal review of critical habitat for boreal caribou, a range is defined as “a geographic area occupied by individuals of a local population that are subjected to the same influences affecting vital rates over a defined time frame” (Environment Canada 2008). In practice, caribou ranges in most jurisdictions have been delineated based on available radio-collar data (predominantly from females), reported sightings and local and expert opinion. Range boundaries have largely been determined by geographic barriers such as major rivers (Culling *et al.* 2004; McLoughlin *et al.* 2004), which are thought to limit movement – and hence gene flow – between adjacent populations. Because of a paucity of long-term telemetry data for a number of populations, range boundaries in many cases are considered to be rough estimates that will likely need updating when additional information becomes available (Arsenault & Manseau 2011; Environment Canada 2012; Culling & Culling 2013).

Across Canada, most jurisdictions do not have a formal monitoring program in place to track changes in the distribution of boreal caribou populations. The exception is Ontario, which has moved toward formally adopting an occupancy-type program to monitor boreal caribou distribution at a province-wide level (Ontario Ministry of Natural Resources 2011; Arthur Rodgers, *pers. comm.)*. This lack of formal monitoring in most jurisdictions likely reflects that a rigorous baseline distribution (e.g., range boundary) has not been established for many populations (Environment Canada 2012).

Where sufficient data exist, attempts have been made to assess changes in boreal caribou distribution. In northwestern Ontario, Racey & Armstrong (2000) compared caribou location data from the 1990s (telemetry, aerial survey and incidental sightings) to historical records (archeological data, trapper diaries and other written records, pre-1990 government survey data) to determine temporal changes in range occupancy. Using a 100-km^2 sample unit, they documented a northward recession of caribou range over the past 150 years with major pulses in range recession coinciding with periods of rapid landscape change (e.g., forest harvesting). Vors *et al.* (2007) expanded on the Racey & Armstrong analysis by linking caribou occupancy to landscape disturbance metrics to identify spatiotemporal thresholds predictive of local caribou extirpation. Schaefer (2003) provided an extended analysis across Ontario using a comparative mapping approach rather than a formal occupancy design to document a 34-km northward range retraction per decade since 1880. Arsenault & Manseau (2011) used a similar comparative mapping approach to assess range contraction in Saskatchewan. In the NWT, Gunn *et al*. (2003) used presence-not detected data to relate boreal caribou distribution to larger-scale habitat features in the Dehcho region. This approach was limited in that it did not estimate detectability but instead assumed it was constant across all habitat types. However, it was able to identify areas of potentially higher use by boreal caribou based on associations with habitat types. This approach could be useful as a "first-pass" to determine likely areas of abundance especially if data from RSF models or other sources are not available to assist in mapping distribution. Currently, occupancy models for caribou continue to be an active area for research in Ontario, particularly in northern regions (Poley *et al.* 2014).

Beyond occupancy designs, other modelling approaches have been used to assess boreal caribou distribution. DeCesare *et al.* (2012*a*) developed a multi-scale RSF model to generate a predictive map for caribou populations in west-central Alberta. Their model included first-order selection (*sensu* Johnson 1980), which evaluated selection of population ranges in comparison to the larger study area. Interestingly, their objective at this scale was simply to evaluate resource selection differences between the pre-defined ranges (based on historical telemetry data) and the study area, not to actually predict caribou distribution *per se*. We further note that many RSF models generate predictions of resource selection based on 'use / availability' designs, where caribou locations are a subset of available locations and thus sample units lacking caribou locational data do not equate to areas of absence. Moreover, the sampling scale of RSFs (most commonly pixels of 30 - 250-m) differs from the sampling scale most appropriate for assessing occupancy (e.g., 100-km^2; Vors *et al.* 2007). Consequently, inferences from RSF models differ from those derived from the 'presence / absence' designs of occupancy models (Lele *et al.* 2013). RSF models are therefore less suitable for monitoring long-term changes in caribou distribution.

For the NWT, boreal caribou distribution is considered continuous with no population-specific individual ranges designated (NWT Environment and Natural Resources 2010a; Environment Canada 2012). Nevertheless, developing range management plans in areas of potential resource extraction and monitoring caribou response to such activities have been identified as priorities for conserving boreal caribou in the NWT and is required under the federal *Species At Risk Act* (NWT Environment and Natural Resources 2010b, Environment Canada 2012). Monitoring long-term changes in caribou distribution with an occupancy-type sampling program could be used to meet such objectives. We note three important caveats when considering long-term monitoring of caribou distribution by occupancy modelling. First, assessing occupancy is most effectively accomplished during the winter by using aerial or ground-based (e.g., snowmobile) surveys to determine caribou presence by their tracks and feeding craters (Poley *et al.* 2014). Boreal caribou, however, are most dispersed during the spring and summer (Bergerud & Page 1987) and thus a winter-based occupancy program may not reflect the full extent of annual caribou distribution. Second, occupancy monitoring in and of itself is inadequate for monitoring other population parameters such as size or trend. For group-living species such as boreal caribou, initial population declines can be masked from occupancy monitoring when the number of individuals per group decreases while the number and distribution of groups on the landscape stays relatively constant (McLellan *et al.* 2010). Third, cell sizes used in occupancy designs will necessarily be smaller than the home ranges of a wide-ranging species like boreal caribou. Short-term changes in regional occupancy may reflect variation in annual range use and not necessarily reflect changes in regional animal abundances, a notion corroborated by traditional knowledge of caribou distributional movements (see *Section 2.5* below).

2.3. Estimating Population Size and Density

Reliable estimates of population size or density are rare for most boreal caribou ranges, primarily due to the difficulty of surveying caribou with methods that have been successfully applied to other ungulates (Bradshaw & Hebert 1996). In recent years, more research has been directed at developing reliable and cost-efficient methods for estimating boreal caribou population size (Carr *et al.* 2012; Hettinga *et al.* 2012; DeMars & Boutin 2013), including a workshop held in October 2012 to discuss current approaches (Alberta Biodiversity Monitoring Institute [ABMI], Edmonton, AB). In this section, we review past and current approaches by jurisdiction for estimating population size and/or density.

Northwest Territories

Boreal caribou distribution within the NWT is not partitioned into distinct herd ranges and instead is considered one continuous distribution (Environment Canada 2011; NWT Species at Risk Committee 2012). The population size for the NWT is estimated to be ~6000-7000 based on professional judgement of NWT biologists and local knowledge (NWT Species at Risk Committee 2012). Where empirical estimates of population size for regional jurisdictions exist, estimates have been primarily counts of minimum number alive derived from aerial surveys (Nagy *et al.* 2005). These surveys were conducted by flying to radio-collared animals and counting the total number of individuals encountered. Because of the difficulty in estimating population size, some NWT jurisdictions have established monitoring programs that rely on indirect methods for monitoring population trend using collared animals, rather than estimating population size *per se* (see below; Nagy *et al.* 2005; Nagy 2011; Larter & Allaire 2013).

British Columbia

Initial estimates of population size within each of BC's six boreal caribou ranges were derived from density estimates based on counts from winter aerial surveys conducted within two wildlife management units (Culling *et al.* 2004). Extrapolating habitat-based density estimates (*sensu* Boyce & McDonald 1999), however, can be problematic due to the spatial clustering of caribou during the winter and the inability of most habitat-based models to capture variability that may be important in dictating caribou-habitat relationships (e.g., lichen abundance; Johnson & Seip 2008). For the most part, BC has since relied on counts of minimum number alive to track relative changes in population size, with surveys conducted either during the fall rut (Thiessen 2009; Thiessen & DeMars 2010) or during late winter (Culling & Culling 2013).

More formal efforts to estimate population size were attempted by Rowe (2006) for the Maxhamish range and by DeMars and Boutin (2013) for the Parker and Prophet ranges. Rowe (2006) conducted a winter aerial survey using a stratified random block design (Gasaway *et al.* 1986), which is the current provincial protocol for surveying boreal caribou (BC RIC 2002). This survey produced an estimate with a coefficient of variation (CV) of ± 70%, a level of error considered to be too high to effectively inform management decisions (BC RIC 2002). DeMars and Boutin (2013) tested a new aerial survey approach which used an occupancy-abundance framework in a two-phase adaptive sampling design (Conroy *et al.* 2008). Briefly, this approach entailed flying fixed-wing transects through 16-km^2 sample units to determine occupancy status then used a helicopter to intensively search occupied units to determine abundance. This new method failed to produce reliable abundance estimates due to poor snow conditions creating ambiguity in assessing occupancy status, low encounter rates with caribou groups, and violations of closure assumptions between the first and second phase of sampling. In addition to testing the new method, DeMars and Boutin (2013) also calculated mark-resight estimates of abundance by considering radio-collared animals to be marked. Although sample sizes were small (n = 8 radio-collared animals), mark-resight estimates (resight = 4) were considered more reasonable than estimates derived from the occupancy-abundance method. They suggested further research into mark-resight methods, particularly given the province's recent collaring initiative (see below) which will substantially increase the number of collared individuals in all ranges.

Alberta

For the past two decades, Alberta has placed more emphasis on monitoring population trend rather than directly estimating population size as the prevailing view was that standard survey methods were impractical and unreliable for boreal caribou (Dzus 2001; ASRD & ACA 2010; Hervieux *et al.* 2013). Up to 2010, government estimates of population size for all boreal caribou ranges within the province had primarily been derived from counts conducted during late winter compositional surveys, which were further augmented by professional judgement (ASRD & ACA 2010). Two notable exceptions were formal survey efforts conducted by Fuller & Keith (1981) and Stuart-Smith *et al*. (1997). Fuller & Keith (1981) used mark-resight methods to generate a point estimate for a 1,400-km^2 study area in the northeastern part of the province but the estimate had no corresponding confidence interval. Stuart-Smith *et al*. (1997) used a stratified block survey to estimate caribou density but the derived estimate had a high coefficient of variation (>70%) due to low sightability (40%). In 2006 and 2009, Wasser *et al.* (2011) tested a novel approach that departed from traditional aerial survey methods. They used a mark-recapture approach using fecal DNA to estimate population sizes for their study area in the East Side Athabasca Range (ESAR). In their design, the study area was divided into 64-km^2 grid cells which were repeatedly surveyed from mid-December to mid-March by dogs trained to locate caribou pellets. While their 2006 estimate had marginally acceptable precision (CV=26%), the 2009 estimate had high precision (CV = 7.5%) and suggested a population size of 330 caribou, an estimate over twice the size of the previous estimate for the entire ESAR range. The authors reasoned that the differences in precision were due to differences in sampling between the two years: in 2006, within-cell surveys were structured to maximize habitat diversity while 2009 sampling was guided by a resource selection probability function to maximize pellet detection probabilities.

The Wasser *et al*. (2011) estimate sparked considerable debate as to whether long-term population monitoring requires reliable estimates of population size (Wasser *et al.* 2012). Following the ABMI population estimation workshop in late 2012, the Alberta government began testing a new fecal DNA-based, mark-recapture method this past year for reliably estimating population size (D. Hervieux, *pers. comm.*). In contrast to the Wasser *et al*. (2011) approach of using scat-sniffing dogs to locate caribou pellets in only a portion of the ESAR range, this new method uses a two-phase aerial-based approach to efficiently collect caribou pellets over the entire ESAR range. In the first phase, fixed wing transects are flown to locate caribou feeding craters. In the second phase, a helicopter is used to fly to cratering sites to collect caribou pellets. The initial year of data collection was completed in 2013 but preliminary results from this new method are still pending.

Saskatchewan

Aerial surveys to count boreal caribou date back to 1960 in Saskatchewan (Ruttan 1960 *cited in* Rock 1988) though the surveys conducted to date have not been part of a formal monitoring program *per se*. Survey designs have generally been transect-based and the end focus of many of the surveys was to derive density estimates (Godwin & Thorpe 2000). In the mid-1980's, Rock (1988) evaluated three survey techniques: a ground-based survey using snowmobiles to look for caribou tracks along a regularly maintained grid of snowmobile trails; a fixed-wing survey along transects spaced at 2-3-km apart; and a two-phase approach where a fixed-wing aircraft flew 2-km transects to locate fresh caribou sign then a helicopter was used to locate animals where sign was encountered. Ground-based surveys were found to be impractical and costly due to the time required to survey a sufficiently sized area. The two-phase approach generated the highest number of caribou observations. All three surveys produced population size estimates based on the number of animals observed augmented by professional judgement regarding the amount of sign encountered in areas where animals were not visually detected. Within this study, aerial surveys were conducted at different times during the winter months (December to April) over the course of two years. Rock (1988) suggested that the optimal survey period was early winter (December) as caribou at this time continue to be highly aggregated after the rut (e.g., bulls are still present in cow groups) and the darker summer/fall pelage provides a better contrast against snow-covered landscapes.

Trottier (1994) also used a transect-based design to inventory boreal caribou in an area adjacent to Key Lake, SK. Transects were 70-km long, spaced 2-km apart and flown entirely by helicopter. Similar to Rock (1988), a point estimate (with no confidence interval) of population density was derived from counts of observed caribou augmented by estimates of herd size based on sign (e.g., number of beds or number of trails where tracks split) where caribou were not detected.

Recently, mark-recapture methods using fecal DNA have been used to estimate caribou population size in Prince Albert National Park and adjacent areas (Arsenault & Manseau 2011). The resulting estimate yielded a coefficient of variation of < 15%, which is an acceptable level of precision for informing management strategies for threatened and endangered species (BC RIC 2002). Going forward, new projects to estimate population sizes using fecal DNA mark-recapture techniques are being planned for herds in both the Boreal Shield and Boreal Plains ecoregions (Tim Trottier, SK Ministry of Environment, *pers. comm.*).

Manitoba

The history of monitoring boreal caribou population size in Manitoba has been similar to that of Saskatchewan. Observation records of boreal caribou from transect-based aerial surveys date back to 1967 in Manitoba and the majority of historical surveys were not directed at surveying caribou but rather moose or deer (i.e., caribou were recorded as incidental observations; Manitoba Model Forest 2005). Starting with the deployment of VHF radio-collars in the mid-1990's, more dedicated monitoring was directed at boreal caribou with minimum number alive estimates derived from flying to collared animals and counting all individuals in the associated group (Manitoba Conservation 2011). Because of the lack of a systematic sampling design, these historical surveys did not produce any confidence intervals around their point estimates and the extent of survey coverage with respect to current caribou ranges is largely unknown (Environment Canada 2008).

In the mid-2000's, Manitoba became one of the first jurisdictions to evaluate the use of fecal DNA-based mark-recapture methods for estimating population size of boreal caribou. Hettinga *et al.* (2012) estimated population size of the North Interlake range over a five year period. During the winter months (January – March), they flew fixed-wing transects spaced at 3-km intervals 3-4 days after a snowfall to locate areas of caribou activity, then returned to these areas with a helicopter to collect fecal pellets. One sampling occasion was completed annually from 2004-06 with the rest of the years receiving two sampling occasions. To ensure sufficient per-site sampling, they collected a minimum of 10 pellets per pellet group and attempted to collect ~ 1.4 times more samples than the estimated number of animals thought to have been present at a particular site. After genotyping, they used a robust mark-capture statistical design, which allowed for an open population between years but a closed population during within-year sampling. Interestingly, the authors chose to focus on population trend rather than annual population size estimates *per se*, perhaps owing to the wide 95% confidence intervals surrounding the annual size estimates. The width of the confidence intervals is likely due to the relatively small number (≤ 2) of per-year sampling occasions (Rees *et al.* 2011). One unique aspect of the study was its estimation of population size separately for females and males. In aerial surveys using radio-collared animals to derive minimum number alive counts, males are likely under-represented because females are predominantly collared.

Ontario

In general, Ontario has placed more emphasis on determining caribou distribution than estimating caribou numbers (Art Rodgers, Research Scientist, Ontario Ministry of Natural Resources, *pers. comm.*). The recently released provincial recovery strategy supports this notion, with a proposed monitoring program focused on assessing caribou occupancy patterns and indirectly evaluating population trend (see next section) but not on determining population sizes (Ontario Ministry of Natural Resources 2011). Still, efforts have been made to estimate boreal caribou numbers within the province. In 1996, Cumming (1998) used a meta-analysis approach, gathering survey information and professional judgement from district offices, to estimate a population of 20,000 caribou in the province. When assessing the survey data, Cumming noted that the methodologies differed greatly, with some counts derived from aerial photography while most were incidental caribou observations collected during random block aerial surveys focusing on other ungulates. Thus, the resulting estimate from Cumming's approach should be viewed as a coarse estimate at best.

Recognizing that reliable population size estimates may be necessary to inform management actions, Carr *et al.* (2012) provided a direct test of three survey methods for boreal caribou in the Slate Islands along the north shore of Lake Superior. The study compared the efficacy of: *i*) transect-based aerial surveys using forward looking infrared (FLIR) technology; *ii*) ground-based transect surveys conducted in the spring; and *iii*) fecal DNA-based mark-recapture methods. The ground-based technique, which entailed observers walking transects averaging 4.2-km in length, was a feasible approach because of the small study area (37.2-km^2). Ground observers recorded all caribou seen or heard, the latter being possible because of a lack of other large mammals on the islands. For the FLIR technique, the fixed-wing aircraft flew transects 200-m apart, a relatively tight spacing driven by a FLIR sensor window width of 110-m when flying at 305-m above ground level. For both the ground-based transects and FLIR transects, abundance estimates were derived using distance sampling analyses (Thomas *et al.* 2010). Fecal pellet sampling was conducted during the winter (January – February) over two sessions in 2007 and three sessions in 2009. The study used standard closed-population mark-recapture models to analyze the fecal DNA data. Overall, the three methods produced abundance estimates with overlapping confidence intervals; however, only the FLIR technique and the three-session fecal DNA approach produced CVs < 20%. The lower CV's for the mark-recapture analysis may not represent the true precision for the methods given issues with modelling unequal detection rates with only 3 sessions of sampling (Chao 1989). While the FLIR results were promising we note that they did not provide details or commentary on whether some of the underlying assumptions of distance sampling, such as detection rates of 1.0 near the plane (Buckland *et al*. 1993), were met in their analysis. Therefore, it is difficult to evaluate the complete utility of the FLIR approach. We note that extrapolation of the efficacy of these techniques to other caribou ranges is not straightforward as the density estimates from these methods were 1-2 orders of magnitude larger than typically reported for other caribou ranges. In addition, sampling was conducted in island areas and therefore issues with estimation to larger study extents were not addressed. The authors further compared the cost of each technique and the three-session fecal DNA had the highest cost, being ~ 2.5 times more expensive than the FLIR method.

Quebec

For reviewing historical efforts to monitor boreal caribou population size in Quebec, we rely on the summary contained in Courtois *et al.* (2003) as government technical reports are not readily available. As with other jurisdictions, early techniques were primarily transect-based aerial surveys which generated either counts of minimum number alive or estimates that were likely biased and with low precision. In a few instances, complete censuses were attempted in small control areas. Density estimates from these controls were used to extrapolate to larger areas but as noted above, this approach can be problematic.

Given the lack of a reliable survey method, Courtois *et al*. (2003) tested a new two-phase aerial-based design in northern Quebec. In the first phase, a fixed-wing aircraft flew transects spaced at 2.1-km apart through 40-km^2 sample blocks to locate caribou tracks. In the second phase, completed the day after phase one, a helicopter returned to identified track networks to locate and count the animals. The method used radio-collared animals to calculate a sightability correction factor for each phase, which was then averaged across the two phases and used to adjust the raw counts from phase two. Despite producing a relatively precise estimate (CV = 15%), the Courtois *et al*. method has not been widely adopted, perhaps for the following reasons. First, the method had a relatively high cost ($160,000 in 1999 dollars for a ~42,000 km^2 study area), which may have prevented it from being tested in other jurisdictions lacking sufficient funding. Second, Courtois *et al*. (2003) estimated a detection rate of 85% based on one resighting occasion (90% in phase one, 94% in phase two), which is likely much higher than would be expected in other forested areas across the distribution of boreal caribou (Stuart-Smith *et al.* 1997; Rowe 2006; DeMars & Boutin 2013).

Labrador / Newfoundland

Of all jurisdictions, Labrador has perhaps had the most consistent historical monitoring of population sizes within its three boreal caribou ranges (Schmelzer *et al.* 2004). However, as with other areas, variation in survey methodology and uncertainty in survey results has made tracking population changes through time difficult. In the 1970's to 1980's, transect-based aerial surveys were the most commonly used method, with transects primarily flown by fixed-wing aircraft or occasionally by helicopter (Schaefer *et al.* 1999; Schmelzer *et al.* 2004). These surveys generally produced counts of minimum number alive or point estimates – often augmented by professional judgement – with no accompanying confidence intervals. In the late 1980s and 1990s, the use of stratified random block (SRB; Gasaway *et al.* 1986) and mark-resight surveys (Schaefer *et al.* 1999) became more prominent. Both methods were commonly conducted by helicopter with mark-resight methods relying on previously radio-collared animals as marks. In general, mark-resight methods produced better precision (Schaefer *et al.* 1999) than SRB surveys but mark-resight methods did not consistently produce estimates with CVs < 20% (Schmelzer *et al.* 2004). The lack of consistency with mark-resight methods is likely driven by the small number of sampling occasions (generally 2; Schaeffer *et al*. 1999) and variation in detectability among surveys (Pollock *et al.* 1990). As noted previously, the low density and spatial clustering of caribou into groups influences encounter rates and because groups can contain multiple marked individuals, variability in detection is therefore affected by the number of groups encountered. Further, detectability is influenced by canopy cover. These sources of variation may result in large differences in detectability among surveys. We note that variation in detection rates of caribou based on group sizes can create bias in estimates. Modern mark-recapture analysis methods allow use of group size as a covariate if it is possible to identify individual collared caribou during the resighting survey.

While the island of Newfoundland is not considered within the distribution of boreal caribou (Environment Canada 2008), we consider one further study that tested a novel method for estimating population sizes for six of Newfoundland's caribou herds. Mahoney *et al.* (1998) used a mark-resight method whereby animals were marked by an oil-alkalyde paint applied from a helicopter and directed toward the animal's back. Animals were located for marking by flying transects spaced 3-km apart and 150-m above ground level. When a caribou group was encountered, attempts were made to mark 25% of the individuals. For five of the six herds, one resight survey was flown 2-3 weeks after the marking flight using the same transects. For the other herd, three resight surveys were conducted in one year – twice by helicopter and once on the ground via snowmobile – and two resight surveys in another year (once by helicopter, once by fixed-wing). Population estimates were derived by the Chapman estimator and chi-square goodness-of-fit tests were used to assess for differences in per-resight size estimates for the herd receiving multiple resight surveys. Of the 12 population estimates generated, nine had CVs of < 20% with the other three estimates having CVs of 22-23%. Differences in precision were positively correlated to differences in detectability among surveys. The authors found no correlation between precision and the initial number of animals marked. For the herd receiving multiple resight surveys, there was no difference in size estimates among resight surveys; however, estimates from helicopter surveys had greater precision. Estimates derived by using all resight data (i.e., > 1 resight) had greater precision than those calculated from only one resight session.

2.4. Monitoring Population Trend

Fundamental to any monitoring program is reliably estimating population trend, which can be estimated through direct or indirect measures (Morris & Doak 2002). Due to the difficulty of obtaining reliable estimates of population size for boreal caribou (see previous section), monitoring population trend of boreal caribou herds has primarily been accomplished through indirect measures (Rettie & Messier 1998; Schaefer *et al.* 1999; Hervieux *et al.* 2013). The recent emphasis on developing reliable methods for estimating population size, however, may make direct measures of trend a viable management strategy. In this section, we review both approaches and suggest that if current research initiatives to develop reliable, cost-effective methods for estimating population size prove successful, then an effective monitoring program should employ information from both types of measures.

Direct Measures

Considerable variability in the methodologies and reliability of population estimates for boreal caribou has prevented rigorous evaluation of population trend through direct measures (e.g., N_{t+1}/N; Bradshaw & Hebert 1996). While acknowledging these limitations, some jurisdictions (e.g., Labrador) have used repeated abundance estimates to make qualitative judgements as to whether populations are stable, increasing or decreasing through time (Schmelzer *et al.* 2004). With the recent advances in population estimation methods, a few more quantitatively rigorous attempts have been made to directly assess population trend. Courtois *et al.* (2008) used their two-phase aerial survey method to assess changes in caribou density over a seven-year time period. They estimated five annual densities and evaluated trend based on whether 90% confidence intervals overlapped among survey years. Wasser *et al*. (2011) evaluated population trend in their study area from 2006 – 2009 using two estimates of population size; however, the short time period of monitoring combined with only two data points – one accompanied by a wide confidence interval – is likely insufficient to effectively evaluate population trend because there is considerable chance of a type II error (i.e., failure to detect a decline if in fact there is one; Thomas & Gray 2002; Boutin *et al.* 2012). Similar to Wasser *et al*. (2011), Hettinga *et al*. (2012) also used a fecal DNA mark-recapture approach to repeatedly estimate annual population size from 2005 – 2009 and consequently derive an estimate of trend. Again, this interval may be too short to effectively evaluate population trend (Thomas & Gray 2002); nevertheless, reliable estimates of population size could be used to augment inferences gained from indirect trend measures.

Indirect Measures

Among jurisdictions, Alberta has been the most prominent in monitoring caribou population trend using indirect measures (McLoughlin *et al.* 2003; Hervieux *et al.* 2013). For almost two decades, Alberta has monitored population trend using the so-called “R/M equation” developed by Hatter & Bergerud (1991). In its original formulation, the R/M equation calculates λ, the finite annual rate of population change by

$$\lambda = \frac{(1-\mathrm{M})}{(1-R)}$$

where M is the finite annual mortality rate and R is the finite annual recruitment rate. The numerator is usually derived from Kaplan-Meier estimates of survival (1 – M) from radio-collared females. Recruitment rates are estimated from late winter aerial surveys to determine calf:cow ratios. Recently, the R/M equation came under criticism for being potentially biased low, particularly over long time frames, due to errors in detecting and classifying calves, non-random censoring, and collared animals misrepresenting true population survival rates (Wasser *et al.* 2012). However, DeCesare *et al.* (2012*b*) demonstrated that by estimating R as a ratio of female calves-to-total females, i.e.,

$$R = \frac{female\ calves}{(female\ calves + adult\ females)}$$

λ calculated from the R/M equation is equivalent to λ values calculated from matrix population projection models (Morris & Doak 2002), including matrix models accounting for the relatively delayed age-at-first-reproduction of caribou. Similar to the R/M equation, population trend has also been measured using Caughley's (1977) survival-fecundity rate of increase (r_s; Rettie & Messier 1998; Schaefer *et al.* 1999).

Changes in the percentage of calves in the population has also used as an index of population trend (Fuller & Keith 1981; Schmelzer *et al.* 2004). Bergerud (1996) suggested that caribou populations are generally stable when calves constitute 15% of the population. Similarly, the federal scientific review of critical habitat for boreal caribou suggested that stable or increasing populations were generally associated with calf:cow ratios equal to or greater than 28 calves /100 cows (Environment Canada 2008). These indices, however, should only be used as rough guides to population trend as population growth rates are influenced by adult female survival as well as calf recruitment (Coulson *et al.* 2005; DeCesare *et al.* 2012*b*).

2.5. Traditional and Local Ecological Knowledge for Monitoring Boreal Caribou

In Section 1.6, we reviewed traditional knowledge systems for monitoring the distribution, population size and trend of wildlife species. A central concept to these systems was their reliance on feedback from indigenous hunters (Moller *et al.* 2004); thus, these systems are more generally applicable to species commonly targeted and harvested. For the majority of First Nations communities, boreal caribou are not – and historically have not been – a species specifically targeted for harvest and instead are generally harvested opportunistically (Benson 2011; NWT Species at Risk Committee 2012). Reasons for not specifically targeting boreal caribou include their low density and the difficulty of hunting in the muskeg conifer forests that constitute their primary habitat. Because boreal caribou are not specifically targeted, low rates of encounter between indigenous hunters and caribou preclude the gathering of reliable information on changes in caribou distribution, population size and trend (NWT Species at Risk Committee 2012). Nevertheless, here we review traditional and local knowledge of the current and historical status of boreal caribou populations.

Boreal Caribou Distribution

Within the NWT, First Nations' perception of boreal caribou distribution closely aligns with distributional maps produced by territorial and federal governments (Gunn 2009; Benson 2011; NWT Species at Risk Committee 2012; Legat & Chocolate 2013). Historical changes in boreal caribou distribution based on traditional knowledge, however, are difficult to discern. This difficulty is primarily due to the relative irregularity of boreal caribou sightings as a consequence of these animals not being specifically targeted for hunting. Traditional knowledge does give insight into the difficulty in monitoring boreal caribou distribution. A consistent theme in interviews with Gwich'in and Dene elders is that boreal caribou are difficult to predict in time and space (Benson 2011; Legat & Chocolate 2013). This unpredictability was thought to be due to animals having to move to keep from over-grazing a particular area, which resulted in animals cycling between areas on a 4-6+ years basis (Benson 2011). This knowledge suggests that monitoring the distribution of boreal caribou will need to be done on a long-term basis and that reliably predicting caribou distribution will require data on the quality of the food resource base (e.g., lichen abundance).

Boreal Caribou Population Size and Trend

Because boreal caribou are seldom targeted for hunting, traditional knowledge regarding current population size is limited (Gunn 2009; Benson 2011; NWT Species at Risk Committee 2012) and indices such as catch-per-unit-effort are not generally applicable. Traditional and local knowledge instead is more focused on subjective assessments of whether populations are increasing or decreasing. These judgements are often made with respect to the frequency of encounters while hunting other species or barren-ground caribou or while checking trap lines in the winter (Benson 2011). In general, the available traditional knowledge literature is inconclusive as to trend direction for boreal caribou populations at both local and regional scales (Gunn 2009; Benson 2011; NWT Species at Risk Committee 2012). Thus, integrating traditional and local knowledge into programs for monitoring boreal caribou population size and trend may be less informative than for understanding distribution (Gunn 2009).

2.6. Summary

The rarity and cryptic nature of boreal caribou has hampered the development of a comprehensive monitoring program for any population within their geographic distribution. Such a program would include monitoring the distribution, size, and trend of individual populations. Below, we summarize each of these monitoring program attributes and suggest ways they could be incorporated into a caribou monitoring program for the NWT. We further discuss specific advantages and disadvantages of proposed monitoring methods in our *Recommendations Report*.

1. *Caribou Distribution*
   The monitoring of caribou distribution is becoming increasingly relevant in light of current rates of anthropogenic landscape disturbance and global climate change. Based on our review, monitoring of caribou distribution can be accomplished through an occupancy-style program, similar to what has been developed in Ontario (Poley *et al.* 2014), or by incorporating recent advances in spatially explicit mark-recapture [SECR] methods that use a two-phase sampling design (see section 1.4 in the *General Literature Review*; Efford & Fewster 2013). Developing a monitoring program for assessing distribution could be complementary, rather than stand alone, to other monitoring initiatives. For example, most procedures for estimating population size rely on aerial-based transects flown through caribou range as part of their methodology, including those relying on recent advances in fecal DNA mark-recapture approaches (Courtois *et al.* 2003; Hettinga *et al.* 2012). By stratifying the range into appropriately sized grid cells (e.g., 100-km^2; Poley *et al.* 2014) and recording the spatial location of caribou sign during these flights, data to monitor changes in caribou distribution (i.e., presence / absence within grid cells) could be simultaneously collected with data relevant for population size estimation. Alternatively, a SECR-based program for estimating population size could be expanded to a broader scale second phase of sampling to explicitly assess distributional changes in density (Conroy *et al*. 2008; Efford & Fewster 2013). As we noted in the *General Literature Review*, SECR-based methods may be better at detecting smaller-scale changes in distribution compared to occupancy programs, which for caribou would necessarily require large sample units (Efford & Dawson 2012).

2. *Caribou Population Size*
   Recent methodological advances in population estimation suggest that reliable estimates of boreal caribou population size may be obtainable. Our review points to three approaches that are particularly promising: mark-resight using radio-collared animals, fecal-DNA mark-recapture and FLIR aerial surveys. Comparing the three, FLIR surveys are the least tested – particularly at the large survey scales necessary for boreal caribou – but this approach holds promise for being a cost-

effective method. Comparing mark-resight and fecal DNA methods, mark-resight is likely more cost-effective if the number of resight occasions is similar to the number of sampling occasions required by fecal DNA methods. Also, a major advantage of mark-resight methods is the concurrent collection of cause-specific mortality and survival data from collared caribou, which is an important component of learning how to remove factors limiting population growth and to enable recovery. Mark-resight methods further provide an immediate estimate of population size, or immediate estimates of precision, that can influence whether additional sampling is required in that same season. This immediate feedback is not possible with fecal DNA methods, which require post-survey genetic analyses. Mark-resight methods may also yield more reliable information on population structure (e.g., female: male ratios, female: juvenile ratios) although the increasing sophistication of genetic analyses is starting to yield this type of information from fecal DNA methods, including the potential to infer pregnancy rates (Hettinga *et al*. 2012; M. Manseau, *pers. comm*., ABMI Boreal Caribou Population Estimation Workshop, October 2012). A potential drawback to mark-resight methods is that a significant proportion of the population may require collaring, particularly if low detectability remains an issue. However, collaring can be invasive and can cause concern to some local communities. To that end, SECR methods (see the *General Literature Review*; Efford *et al*. 2004; Blanc *et al.* 2013), which have not yet been tested on boreal caribou, may help to improve detectability (or recapture) rates for both mark-resight and fecal DNA methods. Regardless of the approach, power analyses should be performed to determine the number of sampling sessions required to reach an acceptable level of precision (e.g., < 20%). We provide preliminary power analyses to assess sample sizes for SECR methods and mark-resight methods in the sampling design section.

3. *Caribou Population Trend*

   Historically, population trend for boreal caribou has best been monitored through indirect measures (Hervieux *et al.* 2013). Indirect approaches such as the one used by Alberta are advantageous because they can provide cost-effective estimates of annual population change (λ) compared to costs associated with repeated direct measures (i.e., annual estimates of population size). Moreover, aerial survey data of radio-collared animals can give more direct information regarding the population structure, the demographic rates influencing λ (e.g., recruitment) and factors causing mortalities. As yet, fecal DNA approaches are unproven in terms of providing reliable information on population structure, although they have been proven as a method to estimate population trend (Hettinga *et al* 2012). Nevertheless, for effective monitoring indirect measures should be periodically corroborated by direct measures of population size (Hatter & Bergerud 1991; Thomas & Gray 2002) to help prioritize conservation planning among populations. To that end, we suggest that robust inferences into population trend will best be attained by a unified approach where multiple sources of data (i.e., population size, survival, and recruitment) are utilized in an integrated modelling approach (see section 1.5 in the *General Literature Review*).

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4. PART B: Study design for monitoring surveys

4.1. Introduction

In this section we provide recommendations for survey designs for pilot studies to further develop monitoring methodologies for boreal caribou in the NWT. We stress that the most appropriate methodology depends on the objectives of the monitoring program. As discussed in the literature review section, occupancy-based methods are most appropriate for inference about broad-scale distribution but are less suitable to monitor population trend. For monitoring trend, we suggest methods that estimate population size, density, and trend using marked individuals (DNA or radio telemetry) are best since they allow direct estimation of demographic parameters. Occupancy is not as suited for estimation of population status or trend given that it is an index of distribution that may not be directly related to population size or demography. However, mark-recapture and mark-resight methods are limited in terms of study area size given the logistical challenge of collecting both sighting data and pellet group data as well as having adequate sample sizes of collared caribou that span entire regions. In Table 1 below, we summarize all methods presented in the literature review, but in this section we focus most on occupancy, aerial mark-resight (using collared caribou) and mark-recapture (using pellet group DNA methods).

If the monitoring program is to provide regional estimates of abundance that can be used to estimate population trend over time, we advocate a 2-phase sampling approach to the surveys that combine both broad- and fine-scale approaches to allow estimates of density for smaller survey areas that are then extrapolated to larger areas using broader-scale surveys (Conroy *et al*. 2008). In this case, phase 1 would be broad scale occupancy surveys and phase 2 would be more intensive survey to estimate population size and density. The degree of sampling that would be needed for the first phase would depend on the prior knowledge of distribution of caribou within the regional areas. If prior knowledge from RSF models or traditional/local knowledge is high then surveying for the initial phase could be stratified by likely areas of higher habitat value with an emphasis on cross-validating RSF model predictions. In this case, more effort is placed on the 2nd phase of sampling to estimate population size and density.

We also advocate an approach to surveys that combines data from multiple sources to maximize inference. It is possible to combine telemetry data and pellet-based mark-recapture data into a joint design that utilizes both the information from telemetry and fecal DNA samples. In particular, spatially explicit methods have been developed to allow telemetry data to estimate the area in which a caribou moves and is detectable with pellet-based sampling (parameterized as sigma in SECR models) (Royle *et al*. 2013). In addition, it is possible to include information about detections of radio collared caribou using multi-data source methods for a-spatial mark-recapture (Boulanger *et al*. 2008) and for spatially explicit methods (Efford *et al*. In prep). This approach should allow a robust estimate of the detection rates of caribou from fecal samples as well as more insight into the sightability of caribou (through mark-resight methods).

Table 1: Options presented in this report for monitoring the distribution, abundance, and trend of rare and elusive species.

Monitoring Goal	Possible Methods	Brief Description	Advantages	Disadvantages	Relative cost
Distribution	Occupancy Logistic regression Zero-inflated binomial regression	Study area is partitioned into grid cells that are surveyed to determine caribou presence / absence. Repeated surveys are required to model detectability.	Straightforward study design and non-invasive. May be accomplished using a fixed wing aircraft only. Useful for assessing long-term changes in distribution and larger-scale patterns of distribution.	Sample unit (cell size) is subjective and can be confounded with home range size. May be insensitive to changes in population if change mainly affects group sizes rather than distribution of groups.	Low, helicopter and collaring not required
Distribution/density	Spatially explicit capture-recapture (SECR)	Results from spatial distribution of individuals detected using DNA are used to assess movement/detection and distribution	Uses density rather than occupancy as the response variable and takes into account spatial scale and detection in a single analysis. Simultaneously estimates distribution and density	Requires DNA mark-recapture data (see below) and sufficient sample sizes of individuals detected to model density surface.	High, helicopter required to collect DNA from pellets
Population Size	Aerial Mark-resight	A group of caribou is collared with proportion sighted used to estimate sightability in aerial surveys	Provides nearly instant estimates of population size. Collars can provide information on factors limiting population abundance and information collected during live capture (pregnancy, body condition)	Requires collaring a sufficient sample size of caribou and also requires suitable sightability of caribou for precise estimates. Communities may not support large-scale collaring efforts.	High, Collaring costs and helicopter costs to locate individuals. Helicopter costs likely higher than DNA method, but no genotyping required
	Pellet group Mark-recapture/SECR	Aerial transects find crater areas which are systematically searched for	Identifies individuals, allowing estimation of detection even when	Is time consuming and costly to collect pellet data and requires DNA analysis	High, helicopter required to collect DNA from pellets

Monitoring Goal	Possible Methods	Brief Description	Advantages	Disadvantages	Relative cost
		pellets in repeated sessions	sightability of individuals is lower. Estimates density based upon repeated spatial recaptures of individuals	of pellet group which results in delays in estimating population size	
	Forward looking infrared (FLIR)	Study area is partitioned into transects which are flown in a fixed wing aircraft. A FLIR camera is used to detect animals.	May increase detectability compared to visual methods.	Lack of broad-scale testing prevents an evaluation of the method's performance over the large landscapes used by caribou	Low – potentially no helicopter required
Population Trend	Indirect estimation ("R/M equation")	Estimates λ using a simple equation (S / 1 – R)	Collars may provide information on factors limiting population abundance (e.g. causes of mortality)	No information on abundance, just trend. Community concerns about collaring caribou	Moderate, Requires many years of collaring, but not intensive surveying. Iridium collars can eliminate FWing costs.
	Sequential population size estimates	Sequential population estimates from mark-resight are used to estimate trend with regression methods	Relatively easy to analyze and to communicate results graphically. Collars may reveal factors limiting population abundance	Lacks power unless population estimates are precise	High, due to repeated surveying
	Open mark-recapture models	The records of individual caribou detected using pellet groups or collaring is used to estimate population trend.	Can infer demographic mechanisms in addition to estimation of population trend using the Pradel model	Study area must remain constant in size and requires longer time series of data	High, due to repeated surveying

One of the greatest challenges to designing a study is determining the detection rate of caribou with the various survey methodologies. Detection probability for aerial surveys would be the proportion of caribou in the target population that are detected during each sample session using resighting methods (for mark-resight surveys) or fecal-based DNA mark-recapture methods. For occupancy estimation, detection rate would be the proportion of times that a survey plot or site is occupied and the caribou are detected. Detection rate in general is not well known for boreal caribou due to the fact that many of the methods to survey caribou are new (fecal DNA combined with SECR methods) or have not been implemented previously (mark-resight methods). The other related issue is that sightability of boreal caribou is quite variable across boreal caribou range and therefore it is difficult to apply results from other studies to those that will occur in the Northwest Territories.

Given the lack of prior knowledge about detection rate, we focus our study design recommendations on single year studies to provide robust estimates of detection rate which can in turn be used to optimize future studies that focus on population distribution, size, and/or trend. We argue that it is best to first collect a rich data set that will maximize inference about detection rates, sources of variation in detection rates, as well as patterns in abundance and distribution before estimating other demographic parameters. We note that detection rate can be estimated concurrently for occupancy-type sampling, mark-resight sampling, and fecal DNA sampling since all utilize transect sampling based survey methods.

4.2. Methods

An overview of the general approach for a pilot study

One of the main objectives of the pilot study will be to assess detection probabilities of caribou or caribou sign. This approach involves multiple components for the phase 1 and phase 2 as summarized in Table 2. We note that the extent of phase 1 surveys will depend on prior knowledge of caribou distribution in the regional study area. If there are RSF maps that are developed for the regional area then phase 1 would consist of a reduced survey to cross-validate predictions about caribou distribution. If there is no knowledge then phase 1 would be a systematic survey of the regional area.

Table 2: Summary of phase 1 and phase 2 objectives, methodologies, and analyses

Phase/Objective	Method	Analysis/comments
1. Phase 1-broad scale distribution	Aerial surveys/double observer	Extent of this phase will depend on prior knowledge of study area
a. Determine broad-scale occupancy	Aerial survey of regional study area to note signs of occupancy	Logistic regression/occupancy analysis
b. Determine sightability of tracks and sign	Double observers during aerial survey	Double observer Occupancy models
c. Determine extent of strip width below plane to determine effective area of sampling	Have observers note relative distance of craters, tracks from plane as well as cover class	Distance sampling methods
2. Phase 2-Estimate population size and density	Aerial survey/double observer DNA mark-recapture Mark-resight	Will use the area defined to have higher concentrations of caribou in phase 1.
a. Estimate population size/density *and detection probability of caribou based on pellet sampling.*	Initial aerial survey to locate tracks (same as 1b above) with helicopter to pick up pellet groups at noted cratering sites.	Double observer to estimate for initial survey to find tracks/craters (same as 1b) Spatial mark-recapture for DNA pellet data
b. Estimate population size, density, and *detection probability of caribou based on sighting from plane/helicopters*	Mark-resight where collared caribou are used to assess sightability of caribou in forested areas.	Mark-resight models.

Broad-scale surveys to estimate population distribution as related to habitat covariates (Phase 1)

For initial “Phase 1” sampling to determine occupancy or document broad-scale habitat associations, the main parameters of interest are the detection rate of caribou in “sites” that caribou are occurring. Transect spacing for this phase is usually meant to ensure an adequate sampling of habitat types to allow assessment of habitat associations and define areas of higher caribou abundance. To estimate detection rates, replicate surveys of the same transect areas need to be conducted. In addition, independent observers on the same side of the airplane can be used to estimate sightability based upon whether both observers or one observer sight caribou or caribou sign (Buckland *et al*. 2010, Boulanger *et al*. 2014). The resulting data set can be analyzed with each observer constituting a session there allowing 2 sessions of sampling to be conducted from a single flight. We note that this component also occurs for fecal-based DNA mark –recapture methods given that the first component of pellet-based detection is the ability of observers to see crater spots or other signs of caribou during the aerial survey. We therefore suggest that a double observer approach be used to estimate detection rate of caribou sign on transect to inform both occupancy models as well as provide a component of sightability for fecal-based DNA detection rate.

The precision of occupancy estimates depends on the actual level occupancy, the detection probability at sites, the number of sampling sessions conducted and the method in which occupancy is estimated. We used formulas of MacKenzie and Royle (2005) to estimate precision of surveys across a range of occupancy and detection levels. We assumed that “sites” would consist of 10-km transect segments which would mean that there would be 400 sites available using the survey design dimensions discussed later for mark-recapture methods (Table 3). We considered designs in which there were 2 and 4 sampling sessions. If 2 aerial surveys were conducted then 4 sampling sessions would result if double independent observers are used on surveys (as discussed earlier). We note that at this time the actual strip width of the surveys that would be used to define the actual plot size of each segment is not known. Gunn *et al.* (2004) estimated that they surveyed 10% of a 10x10 km cell using aerial survey methods, which would mean an approximate 1 km strip width under the survey plane. We suggest that this actual area could be better estimated in pilot work by having observers record approximate distances of observations (craters, tracks, or caribou) from the survey plane with the resulting data analyzed using distance sampling methods (Buckland *et al*. 1993, Buckland *et al*. 2004, Thomas *et al*. 2009). This approach would allow a firm estimate of effective area that the plane is surveying.

We note that the assumption being made is that if data are sufficient for a precise occupancy estimate then it should also be sufficient to use the data to refine and stratify the phase 2 sampling efforts as well as aid in extrapolating estimates from smaller survey areas to larger regional areas. This is an approximate assumption which should be investigated further using simulation modelling once pilot data are collected (as discussed later).

Surveys to estimate population size and density (Phase 2)

For “Phase 2” surveys that estimate population size and density, the main parameters of interest for determining optimal sampling design are the area that the caribou traverse during sampling and the detectability of caribou in their home range. Transect spacing and sampling intensity is based upon ensuring that caribou have adequate detection and redetection rates, which will ensure precise population estimates using mark-recapture (pellets) or mark-resight (collars) methods. Detectability can be estimated either by DNA sampling from pellets or resighting of collared caribou using aerial surveys. An additional assumption of most survey methods is that caribou do not exhibit net directional movement from or to the study area during sampling, beyond normal activity within the home range area. Meeting this assumption ensures that a single estimate of population size or density will apply to the survey area. For spatially explicit mark-recapture methods it is possible for caribou to have home ranges that only partially overlap the survey area as long as the home range is “stationary” during sampling. It is also possible to use mark-resight models that allow a proportion of the collared caribou outside the study area during sampling. However, these models add more complexity and therefore will have less precise estimates.

We first used existing telemetry data to estimate the home range area of caribou during the winter period (February to April) when sampling would occur. This data was supplied by NWT Environment and Natural Resources with samples coming from the Dehcho and South Slave regions. These data allowed us to estimate the area covered and whether there was net movement from home range areas in late spring as a result of migration to calving areas.

We estimated movement rates as the distance between daily locations of individual caribou. If more than one location was taken on a caribou per day then the mean location for the day was used. We screened data for outliers by estimating the daily distance moved between points. We eliminated locations that had daily movement rates of greater than 50 kilometers under the assumption that this rate of movement was likely due to erroneous locations. To estimate home ranges we used the minimum convex polygon (Mohr 1947) estimator which is simply the polygon that is derived by the outermost telemetry points of an individual animal. Home ranges were generated for locations collected from February 1 to April 31, February 1 to April 15, and February 1 to March 31. Caribou had to have at least 20 locations for the given time period and had to have locations for all the months considered to be included in the analysis. Home ranges and movement rates were then compared to assess relative size and if net movement occurred during the time period that the home range areas were collected.

Spatially explicit mark-recapture methods

We assumed that the general study design used in Manitoba would be followed where a fixed-wing transect survey first locates tracks and crater sites followed by collection of pellets using helicopters (Hettinga *et al*. 2012). We determined transect spacing needed for spatial mark-recapture methods based on general relationships between home range area and transect spacing needed to ensure adequate sampling for spatially explicit mark-recapture models (Murray Efford, per-comm). More exactly, assuming a circular home range, the radius (r) of the home range can be derived as $= \sqrt{MCP\ area/\pi}$. The spatially explicit scale parameter (σ) is then related to home range radius as σ= r/2.45. A general rule of thumb is that detector spacing should be between 1.5 σ and 2.5 σ for use of spatially explicit models. Using these relationships it is then possible to relate home range area to transect spacing.

We emphasize that the relationship between home range area and σ is approximate especially given that transect sampling is continuous and linear as opposed to a discrete detector or traps. In addition, there are no data to estimate detection rate at the center of the home range and it is likely that this parameter will also influence the optimal transect spacing. Basically, if caribou are very detectable at their home range center then optimal transect spacing could be wider. We therefore conducted simulations to explore the effects of linear transect sampling and assess sensitivity of predictions to assumptions about detection rates. Simulations were conducted in a beta-version of the statistical software 'secr.design' in collaboration with Murray Efford (University of Otago, New Zealand). Secr.design conducts simulations using modules in program secr (Efford 2011) in the R software package (R Development Core Team 2009).

For these simulations transect spacing was varied from 2 to 5 kilometers. A density of 0.02 caribou per km^2 was assumed with a conservative home range area of 50 km^2 during sampling. Forty 100 kilometer transects were simulated which resulted in different study area sizes and population sizes of caribou within the study area (Table 3). Given the low detectability of boreal caribou noted in the literature review, we simulated conservative detection rates of 0.05 to 0.2 at home range center with 2 sessions of sampling.

We note that these simulations address the question of optimal transect spacing to ensure adequate detection and redetection of caribou within the areas traversed during sampling. In addition, they also address the approximate size of a study area that can be sampled given 4100-4400 kilometers of flying per session (in addition to ferrying kilometers to the survey area). Once detection probability data and habitat data (to better define study areas) is available more in-depth simulations that address optimal sampling strategies for specific areas could be undertaken.

Table 3. Dimensions of hypothetical study area(s) used for spatially explicit simulations. Transect spacing was varied with a fixed number of 100 kilometer transects which resulted in different study area sizes and different population sizes of caribou within the survey area.

Transects	spacing (km)	width (km)	length (km)	area (km^2)	Plane (km flown)[a]	Density ($caribou/km^2$)	N caribou	Plane Cost $[b]
40	2	80	100	8000	4158	0.02	160	25,200
40	3	120	100	12000	4237	0.02	240	26,700
40	4	160	100	16000	4316	0.02	320	26,700
40	5	200	100	20000	4395	0.02	400	27,100

[A] Approximate kilometers flown from one corner returning to the same corner including ferrying between transect lines

[b] Costs provided by GNWT. Based on a Husky FW aircraft, for one session of sampling.

Mark-resight methods

We assumed that a mark-resight study could be conducted concurrently with the fecal DNA study given that both utilize aerial transect sampling. The study design for mark-resight methods also relies on transect spacing that will allow adequate sighting and resighting rates for collared and non-collared caribou (Neal *et al*. 1993, White 1996). In addition, precision of estimates will be determined by the population size of caribou, number of collared caribou, and sightability of collared caribou. We conducted a general set of simulations to illustrate sample sizes needed for mark-resight. For these simulations we assumed a moderate number of caribou within the study area of 150 and 300 which would result from a study area with transect spacing of approximately 2 and 4 kilometers in Table 3 with an assumed density of 0.02 caribou per km^2. Per-session resighting rates ranged from 0.1 to 0.3 with 3 to 5 sessions of sampling. We used the joint-hypergeometric estimator and simulation module in program NOREMARK (White 1996).

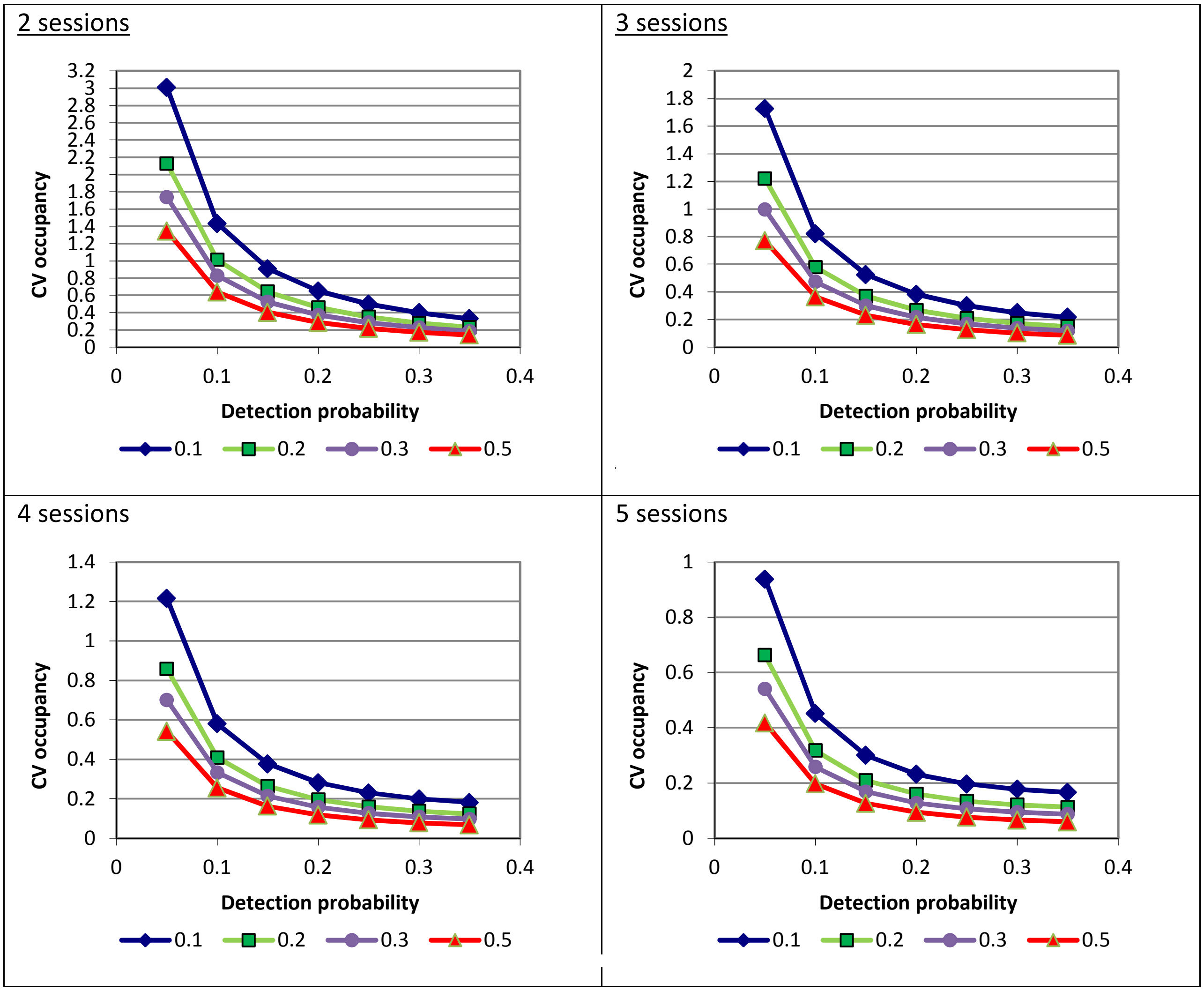
4.3. Results

Phase 1 sampling for broad-scale distribution

The precision of occupancy estimates depended on assumptions made about both occupancy and detection probability (Figure 1). For 2 sessions of sampling, only higher occupancy levels (0.5) and higher levels of detection (0.3) would provide precise occupancy estimates (CV<0.2). With 4 sessions, a detection probability of 0.2 and occupancy levels of 0.2 would result in precise occupancy estimates. From this exercise, it seems likely that **2 sessions (with a double observer platform which would result in 4 sessions of occupancy data) would be required unless detection rates and/or occupancy rates are high** (Figure 1).

It is difficult to apply the results of these simulations to surveys, because the detection rate is not known. The one boreal caribou aerial survey that did occur in the Dehcho (Gunn *et al*. 2004) found that

33% of the cells had caribou craters or sign of feeding, however this proportion would be the product of detection rate and occupancy since detection rate was not estimated (since only one sampling session was conducted during the survey). Therefore, the 33% could mean high occupancy and low detection or high detection and low occupancy. In addition the 33% was based upon 210 of 618 cells having sign of caribou of which 72 had craters, 137 had tracks, and 1 cells with an observation of 15 caribou. The survey was not conducted immediately after snowfall so it is likely the number of tracks sighted would be different dependant on when the last snowfall occurred. In addition, this survey was conducted using a stratified approach based upon a preliminary habitat model. Therefore, the percentage cells would likely be lower unless the survey was pre-stratified using habitat models.



2 sessions

3 sessions

4 sessions

5 sessions

Figure 1: Predicted precision of occupancy estimates as a function of detection probability (x-axis), occupancy (levels of 0.1-0.3) and the number of sampling sessions assuming 4000 kilometers of sampling and 10 km segments (400 cells total) .

Surveys to estimate population size and density (phase 2)

We initially assessed movement rates and home ranges to determine survey extents for mark-resight methods and spatially explicit mark-recapture. We then synthesized the results for each method.

Movement rates

Daily movement rates indicated that movements were relatively similar up to mid-April, at which time the mean and degree of variance in movements increased (Figure 2). Some of the outlier movements (as delineated by open circles) may have been due to collar errors. We chose to eliminate movement rates of greater than 50 kilometers per day from the home range analysis based on the movement rate analysis.

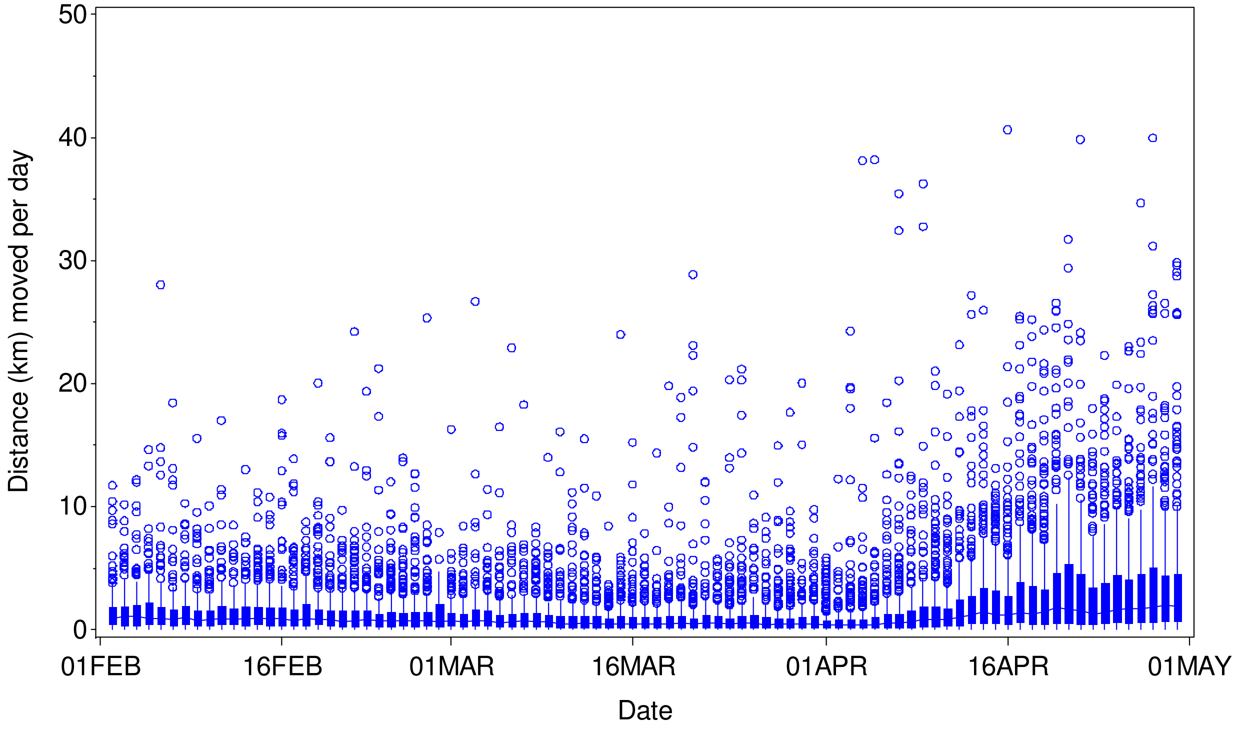


Figure 2: Movement rate (distance moved per day) as described by box plots. The box indicates the 25th and 75th percentile of movements, the whiskers indicate the range of data up to 1.5 x inter-quartile ranges, and the lines join the median rates for each day. Any point beyond 1.5 interquartile range is indicated by an open circle.

Home ranges

Assessment of home range areas during sampling allowed us to determine the area likely to be traversed during the duration of sampling. Boxplots of the distributions of MCP home range for the full range of dates (Feb 1-April 30) compared to restricted ranges of date demonstrated that home range areas increased especially in the last 2 weeks of April (Figure 3). We suspect that some of the larger home ranges for each of the intervals may be due to location errors and therefore we mainly considered median home ranges (the central bar in each of the boxplots below) rather than mean home ranges for each of the data series. In this case, median home ranges were 62.24, 105.0, and 303.3 km^2 for Feb 1-March 31, Feb 1-April 15, and Feb 1-April 30 respectively.

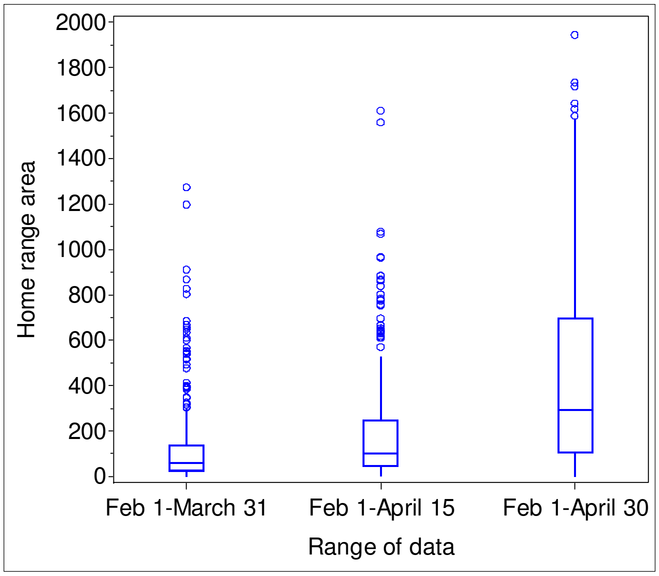


Figure 3: Boxplots of MCP home range areas for 3 ranges of data for boreal caribou. Sample sizes of home ranges were 336 for the 3 ranges of data.

Plots of MCP home range areas also revealed a much larger area in later April (Figure 4). An assumption of the SECR sampling method as well as any method that defines a study area is that home ranges are stationary with minimal directional movement from the area during sampling. The SECR method can accommodate partial overlap of home range areas off the study area as long as movement is within the home range area during sampling. **Therefore, we suggest that the sampling period from Feb 1 to April 15 is optimal** to avoid potential bias caused by caribou emigrating from the sampling area in later April.

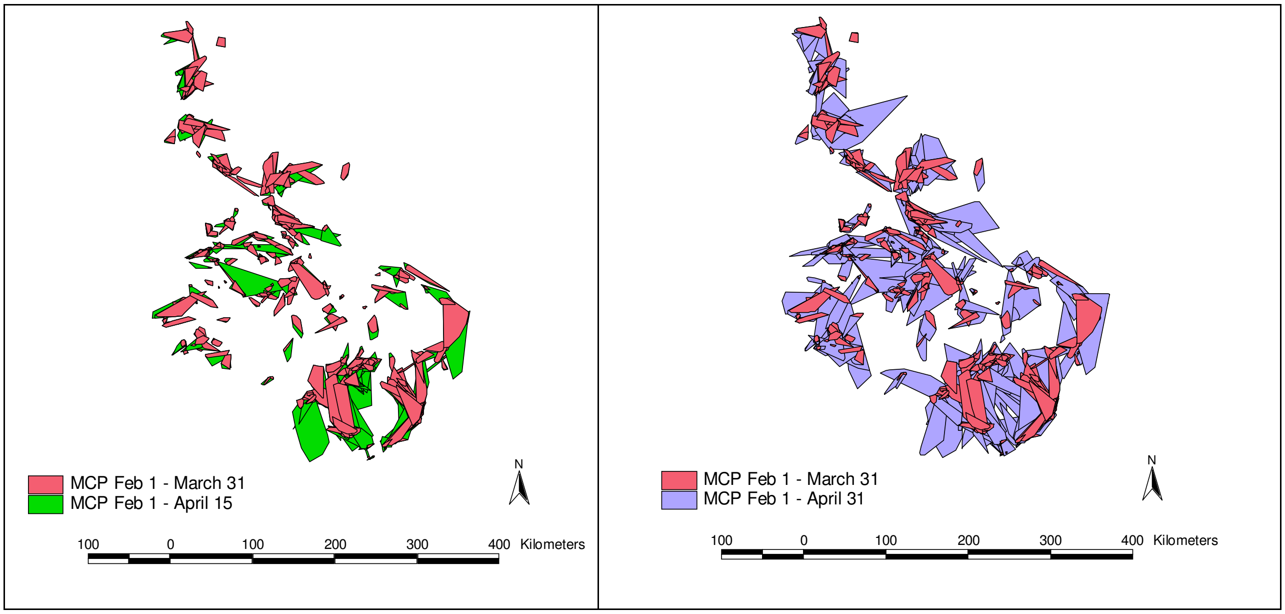


Figure 4: Comparison of plotted home ranges for Feb 1-March 31 compared to Feb 1- April 15 (left) and Feb 1 to April 31 (right).

Spatially explicit methods

Approximate transect spacing needed for spatially explicit methods was then estimated using the home range sizes. Using the ranges, the median transect spacing needed was 3.6, 4.7, and 8.0 for Feb 1-March 31, Feb 1-April 15, and Feb 1-April 30 respectively (Figure 5).

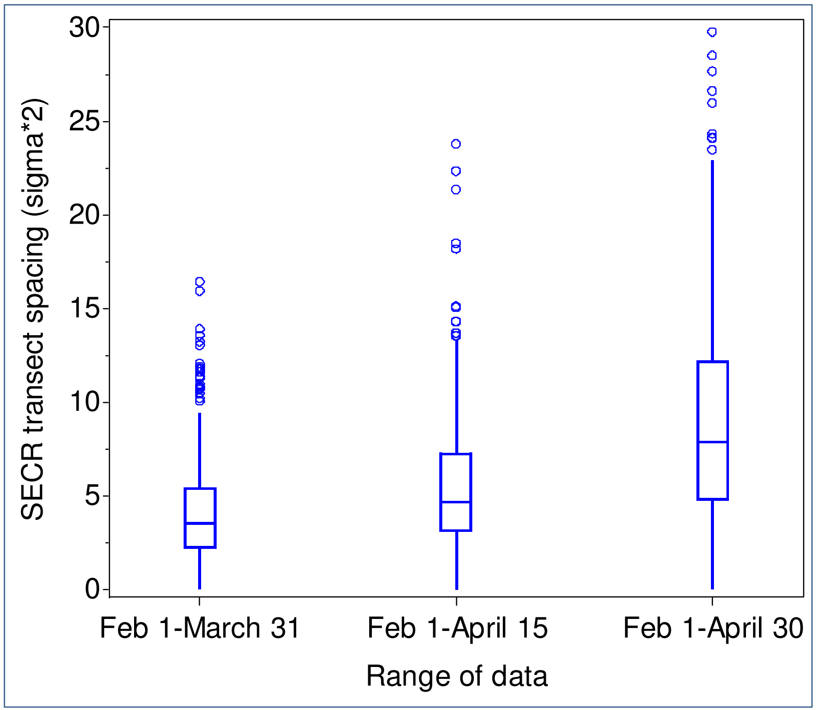


Figure 5: Approximate spatially explicit transect spacing needed based on home range sizes (Figure 2).

Simulations of SECR with transect sampling revealed that transect spacing of 2 to 5 kilometers had little effect on the precision of estimates, however detection probability at home range center (g_0) had a much higher effect (Figure 6) on precision. In addition, at lower detection rates (g_0=0.05), transect spacing had a higher influence on estimates. As transect spacing increased, the relationship between transect spacing and precision of estimates became more pronounced (Murray Efford, per. Comm). We note that these simulations basically demonstrate the trade-off between increasing study area size (which resulted in a larger population size on the study area given density was fixed) and increasing transect spacing (which would reduce detection probabilities of caribou). The basic conclusion can be interpreted that the effects of increasing study area size compensated for reduced detection rates at higher transect spacing. **The take-home message from the simulations was that transect spacing of 2 to 5 kilometers is likely to be adequate, but it will require pilot data, and estimates of detection rate at the home range center to further optimize transect spacing**. If detection rates are higher than it is likely that spacing can be wider. If detection rates are low, then closer transect spacing will be needed to ensure adequate numbers of detections and redetections of caribou pellet groups.

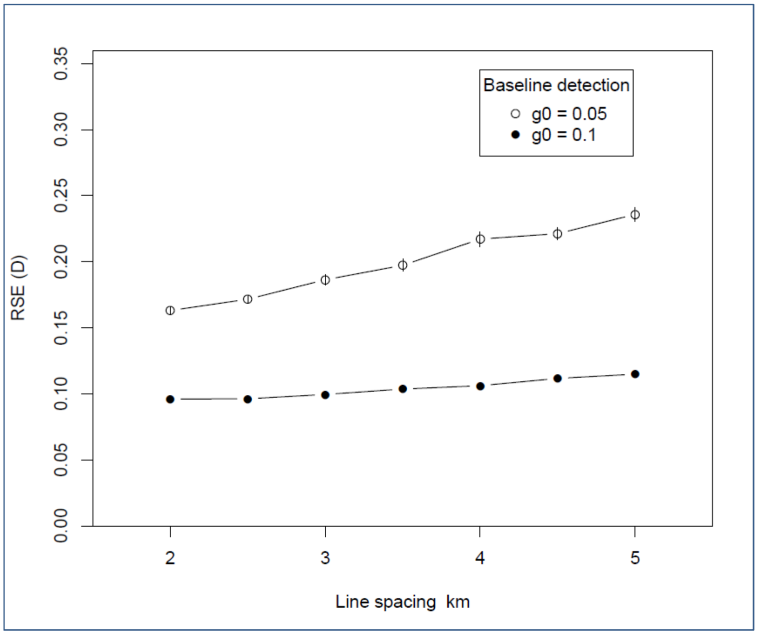


Figure 6: Relationship between transect line spacing, precision of SECR estimates, and detection rates at home range center for simulations conducted in secr.design.

Mark-resight methods

Results of simulations suggested that the precision of estimates is very dependent on assumed sighting probabilities with sighting probabilities of at least 0.2 needed for precise estimates especially when 3 sessions of sampling were conducted (Figure 6).

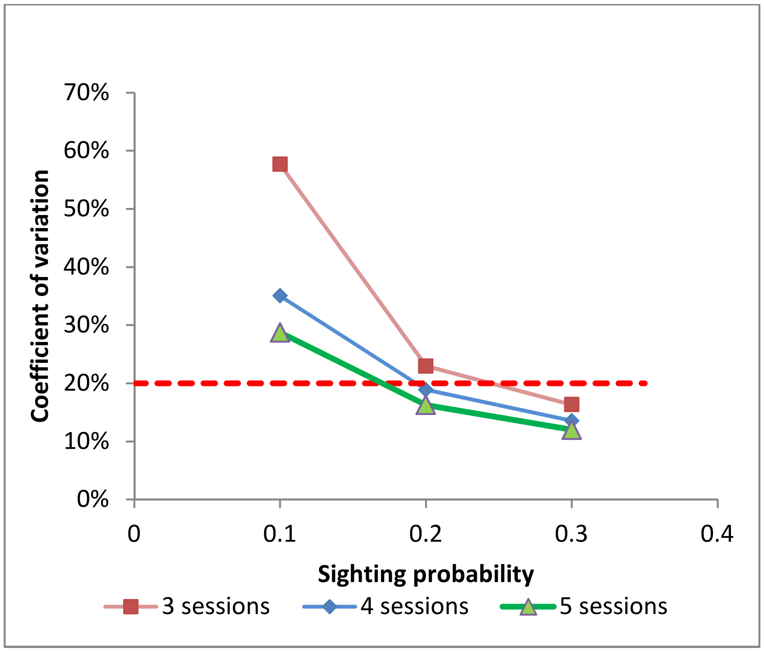


Figure 7: Mark-resight simulations with assumed population size of 150 caribou with 3-5 sessions of sampling and sighting probabilities of 0.1 to 0.3

Precision of estimates was not as affected by the assumed population size given that the number of radio collars influences the precision of sighting probabilities rather than the number of caribou in the population (Figure 7).

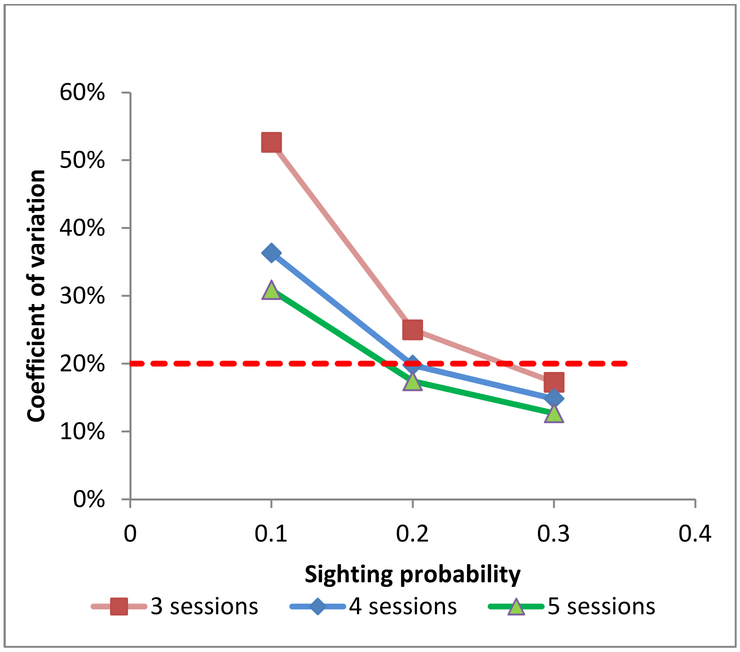


Figure 8: Mark-resight simulations with assumed population size of 300 caribou with 3-5 sessions of sampling and sighting probabilities of 0.1 to 0.3

Increasing the number of collars to 40 helped offset low precision when sighting probabilities were at 0.2 with 3 sessions of sampling (Figure 8).

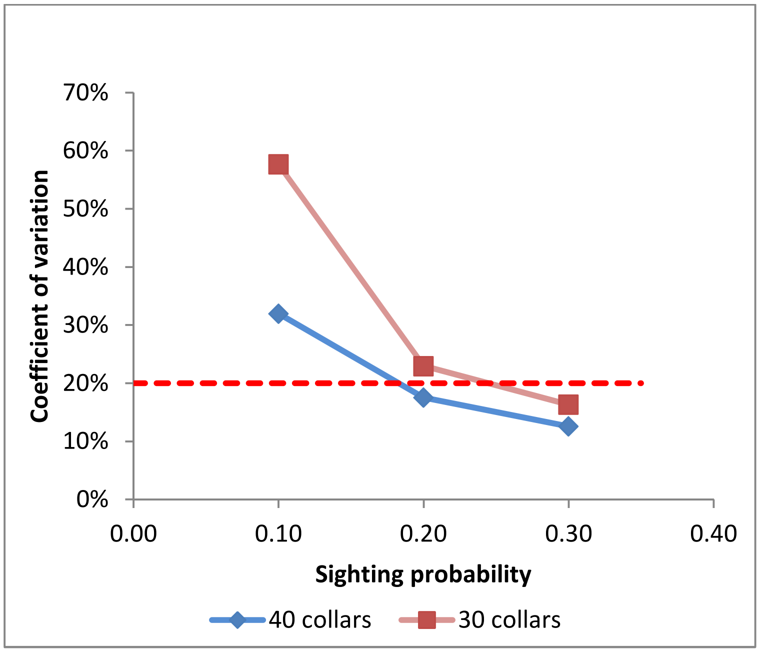


Figure 9: The influence of the number of collars on the precision of estimates assuming a population size of 150 caribou and 3 sessions of sampling

In conclusion, like the SECR simulations, estimates were greatly influenced greatly by detection rates. With likely sample sizes of collars (30-40) and sampling sessions (3-5), sighting probabilities needed to be at least 0.2 for precise estimates. This result can be also conceptualized in terms of the number of caribou that needed to be observed per session of sampling. If population size was 150, then 30 caribou would need to be observed per sampling session of which approximately 6 would be collared (if there were 30 collared caribou). If population size was 300, then on average 60 caribou would need to be observed per session of which 8 would be collared (assuming 30 collars total).

A fundamental assumption of mark-resight methods is that caribou have roughly equal sighting probabilities and collared caribou and uncollared caribou have equal sighting probabilities. Factors such as group size, vegetation/habitat type, snow conditions (how recent was the snowfall) will affect this assumption. **We strongly suggest that data is collected for mark-resight so that individual collared caribou can be identified in the mark-recapture sample.** If data is collected in this fashion then it is possible to model sighting probabilities of individual caribou using covariates (group size, dominant habitat class or other factors influencing sightability) with mark-resight models in program MARK (McClintock and White 2009;2010).

Survey costs

Fixed-wing flight costs were estimated to be approximately $26,000 per session and were relatively invariant to the study area size because the number of transects was held constant for our scenarios (Table 3). Clearly, the greatest multiplier of costs will be the **number of sampling sessions** that are chosen (Figure 1). The costs we estimated are limited to the fixed wing because helicopter costs to collect scat or to locate animals for resighting will again depend greatly on encounter rates. However, if we assume that helicopters have to travel approximately 75% the total distance of the fixed-wing, then helicopter costs will be approximately $30,000 per session. This estimate is based on a Bell Long Ranger ($1550/hr) with a range of 495 km (3 hour range). Of the two methods described here, helicopter costs will likely be lower for fecal DNA surveys than mark-resight surveys because fecal DNA surveys only require helicopter transport of ground crews to known caribou crater locations while mark-resight surveys will require additional search time to visually locate the animals. Therefore, total aerial costs will be approximately $60,000 per session, though a buffer of $10,000 per session is likely warranted. Again though, stratification of the study area could greatly reduce these costs. Assuming 3 sampling sessions, a conservative estimate of aerial costs for a pilot study would be $210,000. These costs do not include any genotyping, personnel, processing, or analyses.

4.4. Discussion

We note that in general the requirements for detection probability for occupancy (Phase 1) and population and density estimation (Phase 2) were all influenced by assumptions made about detection rates. **This general result further justifies a pilot study to estimate detection rates**. We emphasize that detection probability estimates for all methods can be obtained concurrently. We also note that the actual area covered by the Phase 1 occupancy phase of the pilot study will depend on whether the determination of larger-scale occupancy is an objective. If the main objective of the pilot study is to mainly estimate detection rate then the Phase 1 and Phase 2 sampling area can be the same area. If broader scale occupancy is an objective of the pilot study then a larger survey extent will need to be covered.

We suggest that information about occupancy based upon previous studies (Gunn *et al* 2004) and ongoing RSF modelling be evaluated to determine the need for a larger survey extent for Phase 1 sampling for the pilot study. We provide an example of this approach in the Deh Cho case study later in this document.

Simulations of spatially explicit and mark-resight highlight the need for pilot data to further refine sampling programs. In both cases, simulation results were very sensitive to assumptions about the detectability of caribou from pellet samples, or the sightability of collared caribou. For spatially explicit methods, detection rates at the home range center needed to be close to 0.1 to ensure precise estimates. For mark-resight methods, resighting probabilities needed to be close to 0.2. In both cases, it is difficult to determine optimal survey effort to achieve these targets until pilot study data is collected. We suggest that a design that is more intensive (closer transects) will allow firm evaluation of the best strategy.

We also note that once pilot data is collected and analyzed it will be possible to determine optimal survey intervals and frequencies for longer-term monitoring studies. Two approaches are possible for monitoring. First, regression-based methods can be used to estimate change in population size from successive surveys. With estimates of precision of initial surveys it is possible to conduct simulations to determine survey interval using methods similar to those applied to barren ground caribou (Boulanger 2011). Second, if individual genotypes of caribou are available from pellet-based DNA mark-recapture methods, then open mark-recapture models can be used to estimate trend, apparent survival and other demographic parameters from the mark-recapture data including the exploration of mechanisms and covariates that drive demography (Pradel 1996, Nichols and Hines 2002, Boulanger *et al*. 2004, Chandler and Clark 2014). Simulation modelling can also be used to determine optimal design for mark-recapture-based demographic studies (Boulanger 2005, Stetz *et al*. 2010).

The analysis of caribou home ranges suggested that transect spacings of 3 to 5 kilometers will ensure adequate detection rates of caribou. If detection rates are high then it is likely that transect spacing can be broadened to reduce survey effort. If available, habitat data could be used to stratify sampling to ensure the most coverage for areas of higher habitat value. However, we still recommend a more systematic sampling approach to ensure that all caribou within the study area have a non-zero probability of being sampled. A post-stratification exercise would allow evaluation of potential stratification strategies using the pilot data set.

The analysis of home range areas and movements did define an optimal time frame of sampling of February 1 to April 15 when home ranges are relatively stationary and there is less risk of emigration from the study areas. We suggest that monitoring of collared caribou movements during sampling will also help assess if net movement is occurring especially if sampling occurs in April.

An example of the eventual two-phase sampling design (to be refined once pilot data is collected)

We provide a brief example to illustrate the 2-phase sampling design using a historic boreal caribou aerial survey data set collected in the Deh Cho region (Gunn *et al*. 2004). In this study, data from aerial surveys were used to develop a preliminary boreal caribou habitat model. This data was then used to stratify presence-not detected surveys based upon transect surveys (10 kilometer spacing) in relation to dominant habitat classes as defined based upon 10x10 km cells in which a single transect was flown through the center of the cell to sample for caribou sign. This model identified areas that had higher probabilities of caribou occurrence (Figure 10) that corresponded to proportion spruce-lichen habitat.

Figure 10: Example of the first phase of a 2-phase sampling design based upon aerial survey of boreal caribou within the Deh Cho region (Gunn *et al*. 2004). The 10 x 10 km grid, the high (dark), moderate (medium) and low (light) probability of occurrence based on the preliminary model, flight lines for March 2002 and caribou sightings (circles).

We suggest that a similar approach could be used as the initial phase of sampling based on previous aerial survey studies or RSF habitat modelling. In this case, aerial survey sampling would be stratified based on dominant habitat classes. Unlike the Gunn *et al*. (2004) approach, we would suggest a double observer platform for aerial sampling to provide an estimate of detection probabilities to allow the use of occupancy models or zero inflated regression models. This would insure that habitat classes where sightability is lower would not be under-represented in the habitat model. The intensity of phase 1 sampling will be determined by prior knowledge of boreal caribou distribution in the regional study area. For example, for the Deh Cho study area it would be useful to reconsider the habitat model of Gunn et al (2004) based upon radio telemetry data and updated habitat cover data. Once an initial phase 1 sampling is conducted it would be likely that the overall intensity of sampling could be reduced or stratified to the most likely areas where caribou occur. This was the general approach that Gunn et al (2004) used. The initial habitat model was based on reconnaissance data from bison surveys. This data was then used to stratify sampling in 2002 to develop the habitat model displayed in Figure 11. If further data collection were to be collected for this grid it would focus on cross-validation of the existing habitat model.

Figure 11 provides a hypothetical location of the phase 2 pilot study area based upon the habitat model of Gunn et al (2004). This first year pilot study area would utilize 2 or 3km grid spacing. The size of the study area based upon dimensions given in Table 3 is shown in Figure 11. A grid with 5 km spacing would occupy the area of the 2 grids combined. This further illustrates the need for pilot data to determine optimal grid spacing. If 5 km spacing achieve adequate detection rates then it will allow sampling of a much larger area than the grids with closer spacing.

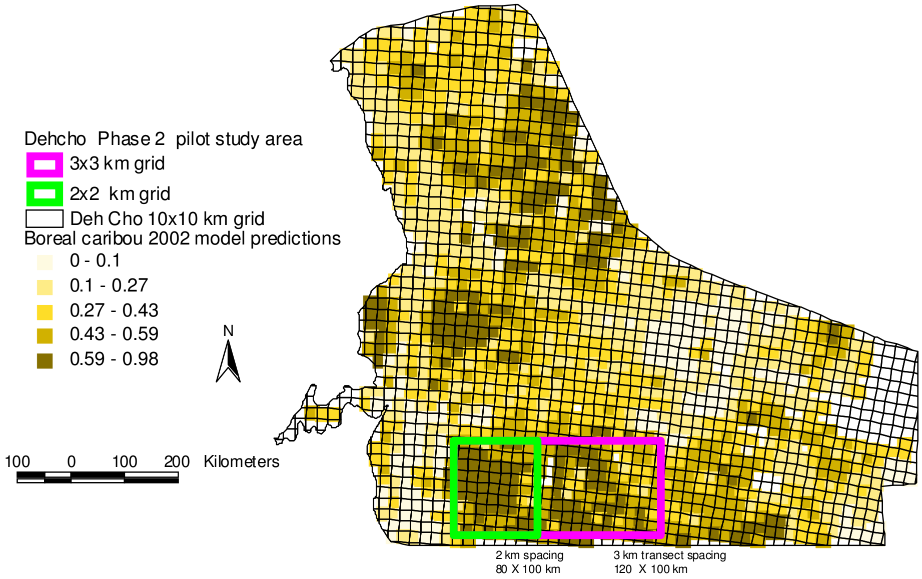


Figure 11: Hypothetical phase 2 pilot study areas based on 2002 boreal caribou habitat model

Subsequent years of sampling would use the pilot data collected in year 1 to determine optimal transect spacing and from this adequate study area size as summarized in Table 4. We note that some of the methods used in the pilot year such as distance sampling to define effect survey areas of the plane will not be needed in subsequent years. Furthermore, we suggest that use of telemetry-based mark-resight and pellet-based mark-recapture may not be needed unless a side objective of monitoring is to collect detailed survival rate and movement data from collared caribou. We suggest that a set of subgrids within the larger sampling area would be surveyed at higher intensity with concurrent collection of pellets for DNA mark-recapture analysis to allow estimates of density. In this case, subgrids would be placed in areas of higher probability of occupancy but also a range of habitat classes where boreal caribou occur. The basic premise is that with SECR inference to larger scales is more robust using subgrids rather than a single grid (Efford and Fewster 2013). The larger-scale aerial survey data would then be used to extrapolate estimates to larger areas using spatially explicit mark-recapture methods using RSF-based SECR analysis (Royle *et al*. 2013).

Table 4: Differences between pilot study and subsequent studies on boreal caribou. The rows are based upon Table 3.

Phase/Objective	Pilot year	Years after pilot study
3. Phase 1-broad scale distribution	Extent of aerial survey depends on prior knowledge of distribution	Survey extent reduced if sufficient knowledge on distribution
a. Determine broad-scale occupancy	Aerial survey of regional study area to note signs of occupancy	Cross-validation of previous year. Stratified and reduced design
b. Determine sightability of tracks and sign	Double observers during aerial survey	Double observer to note differences between years
c. Determine extent of strip width below plane to determine effective area of sampling	Have observers note relative distance of craters,tracks from plane as well as cover class	Not necessary after pilot year of study
4. Phase 2-Estimate population size and density	Single study area to estimate detection rates to optimize transect spacing	Multiple subgrids stratified to allow inference on regional study area.
a. Estimate population size/density *and detection probability of caribou based on pellet sampling.*	Intensive aerial transect spacing to collect rich data set Initial aerial survey to locate tracks (same as 1b above) with helicopter to pick up pellet groups at noted cratering sites.	Optimized transect spacing based on pilot data Pellet mark-recapture or mark-resight but not both.
b. Estimate population size, density, and *detection probability of caribou based on sighting from plane/helicopters*	Mark-resight where collared caribou are used to assess sightability of caribou in forested areas.	Mark-resight models if detection probabilities >0.2 from pilot study

The main challenge to the implementation of this design currently is lack of knowledge of detection rates of caribou with aerial survey methods as well as DNA mark-recapture methods. This knowledge is critical for determination of optimal survey intensity especially for the smaller subgrids that would be used to estimate density. With pilot data results, simulation modelling that utilizes estimates of detection at home range center and scale (from SECR analysis) can be used to further assess the optimal transect spacing as well as investigate the use of smaller subgrids to sample the larger survey extent. This data can also be used to further design longer-term monitoring studies.

4.5. Acknowledgements

We thank Nicole McCutchen and James Hodson for helpful comments on this report, and the Government of NWT for providing us with caribou telemetry data. We also thank Judy Williams for providing detailed estimates of survey costs.

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