Linking Landscape Structure to Woodland Caribou Density Using Non-Invasive Spatial Capture-Recapture

by

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ABSTRACT

Assessing how landscape composition and anthropogenic disturbance influence spatial variation in abundance is essential for effective conservation planning in threatened species. For woodland caribou (*Rangifer tarandus caribou*), there remains limited knowledge of spatially explicit population density, potential differences in density and space use between males and females, and the specific habitat and disturbance features that structure their distribution across managed forest landscapes. We applied spatial capture-recapture (SCR) modeling to non-invasive fecal DNA samples collected in the Churchill Range of northwestern Ontario to estimate caribou density, space-use parameters, and evaluate how landscape composition and disturbance influence spatial variation in abundance. A total of 589 genotyped samples identified 245 unique individuals (132 females, 113 males). The best-supported SCR model estimated a population density of 43.6 individuals per 1,000 km^2 (95% CI: 31.8 – 59.6) and a mean space-use parameter (σ) of 5.4 km, corresponding to an estimated home range radius of 25.6 km. Sex-specific models revealed no strong differences in density or space use between males and females. Landscape models indicated that caribou density was positively associated with conifer forest and greater distance to roads, and negatively associated with deciduous cover, highlighting the influence of forest management on caribou distribution in this region. This study provides the first SCR-based population density estimate for woodland caribou in Ontario. Our findings offer critical baseline data for conservation planning and recovery efforts and demonstrate the value of non-invasive SCR for linking population abundance to habitat features in wide-ranging, low-density species.

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CAUTION TO THE READER

This MScF thesis has been through a formal process of review and comment by two faculty members and an external examiner. It is made available for loan by the Faculty of Natural Resources Management for the purpose of advancing the practice of professional and scientific forestry.

The reader should be aware that the opinions and conclusions expressed in this document are those of the student and do not necessarily reflect the opinions of either the thesis advisor, external examiner, committee members, the faculty or Lakehead University.

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INTRODUCTION

Assessing spatial variation in wildlife population density is essential to effective conservation planning. Especially for at-risk species, identifying which landscape features promote or constrain density is critical to setting recovery thresholds, prioritizing conservation areas, and evaluating the effectiveness of habitat-based management (Pereira et al. 2013). Anthropogenic disturbance alters landscape composition and connectivity, affecting how animals use space and often reducing local population densities (Luskin et al. 2017). However, for many wide-ranging, low-density species, spatially explicit density estimates remain rare, limiting the ability of resource managers to make informed decisions.

Boreal woodland caribou (*Rangifer tarandus caribou*, hereafter referred to as caribou), listed as threatened under the Canadian *Species at Risk Act* and Ontario *Endangered Species Act*, are emblematic of this challenge. Forest harvesting indirectly reduces caribou populations by increasing deciduous forest conditions, resulting in greater moose (*Alces alces*) and wolf (*Canis lupus*) densities, leading to higher predation risk (Darby and Duquette 1986, Wittmer et al. 2007, Fryxell et al. 2020). Road networks also increase predation risk by allowing easier movement of wolves, increasing their encounters with caribou (Fortin et al. 2013). Additionally, forest fires reduce preferred habitat with potentially cumulative effects on caribou ecology, particularly in areas with existing high levels of anthropogenic disturbance and habitat alteration (Konkolics et al. 2021). Despite the well-documented impacts of disturbance on caribou habitat selection, few studies have been conducted to determine the extent to which disturbance and habitat conditions impact caribou population densities.

Most ecological research on caribou has relied on telemetry to understand seasonal habitat selection and movement (Ferguson and Elkie 2004*a*, O'Brien et al. 2006, Beauchesne et al. 2013, Hornseth and Rempel 2016, Fryxell et al. 2020). These studies have been instrumental in showing that woodland caribou tend to avoid disturbed areas and deciduous stands, instead preferring mature conifer forest and peatlands that reduce exposure to predators (Environment Canada 2011*a*, Fortin et al. 2013, Hornseth and Rempel 2016, Fryxell et al. 2020, Konkolics et al. 2021). While telemetry can be used to inform resource selection models, it does not yield estimates of density and the high cost of capturing and collaring typically means that a small number of individuals, often only adult females (Fortin et al. 2013, Hornseth and Rempel 2016), are sampled. As a result, inferences about space use may not represent the broader population (Poutanen et al. 2023), particularly for males and calves. For caribou, whose large home ranges are a central consideration in conservation planning, understanding broader spatial requirements across all demographic groups is essential.

Because telemetry cannot be used to estimate population density and typically samples only a narrow subset of individuals, spatial capture-recapture (SCR) modelling fills a critical gap in caribou monitoring. SCR enables robust estimation of density while simultaneously assessing the influence of landscape features on spatial variation in abundance (Efford 2004, Royle et al. 2013). When paired with non-invasive genetic sampling, such as fecal DNA, SCR can detect variation in density across age and sex classes, allowing for more representative inferences. Importantly, SCR also estimates the spatial scale parameter (σ), which reflects the extent of individual space use and is closely related to home-range size. This makes SCR particularly valuable for conservation planning in species like caribou, where large home ranges and exposure to disturbance impact population density and management needs. Sex-specific SCR using non-invasive fecal DNA sampling has been successfully applied to various ungulate species, including caribou in Saskatchewan and Alberta (McFarlane et al. 2020, 2022), red deer (*Cervus elaphus*) (Tourani et al. 2023), white-tailed deer (*Odocoileus virginianus*) (Poutanen et al. 2023) and moose (Koitzsch et al. 2022).

To date, no SCR based density estimates are currently available for caribou in Ontario. Existing studies have been conducted in western Canada, where the anthropogenic footprint is dominated by oil and gas development and associated linear features that heavily fragment the landscape (McFarlane et al. 2020, 2022). Among these, only one study incorporated spatially heterogeneous density modelling using habitat quality and distance to linear features as covariates (McFarlane et al. 2022). It found that both high- and low--quality habitat contributed to explaining caribou density, with a strong positive association between density and distance to roads (McFarlane et al. 2022). In contrast, the boreal landscape of Ontario is primarily shaped by industrial forestry, characterized by harvest blocks and associated road networks (OMECP 2020), in addition to regional ecological differences. These distinctions raise important questions about whether patterns of habitat selection, space use, and population density observed in western ranges also apply in Ontario, as caribou are shown to exhibit plasticity in space use across landscape conditions (Fortin et al. 2008). In particular, the effects of natural and anthropogenic disturbance on caribou density in Ontario remain unknown.

In this study, we used non-invasive fecal DNA sampling and SCR modeling to estimate density and abundance of caribou in the Churchill Range of northwestern Ontario, while also evaluating how landscape composition, anthropogenic disturbance, and natural disturbance influence spatial variation in abundance. Our objectives were to: (1) estimate sex-specific caribou density and space-use parameters using SCR, allowing assessment of potential differences in home-range size between males and females; and (2) evaluate the relative influence of habitat and disturbance variables on spatial patterns in density across the study area. We predicted that male and female densities would be similar, but that winter home range size would differ by sex. We also predicted that caribou density would be positively associated with coniferous and wetland cover types, and negatively associated with deciduous cover, forest harvesting, linear features, and natural disturbances. By linking sex-specific density and movement patterns with landscape features, this study provides new insights into how landscape structure shapes the distribution of a threatened boreal species. Our results offer a spatially explicit understanding of how habitat and disturbance influence caribou populations in Ontario and can support the refinement of habitat-based management strategies aimed at recovery and long-term persistence.

METHODS

Study Area

This study was conducted in the Churchill Range of Northwestern Ontario, which spans approximately 21,300 km^2 just north of Sioux Lookout (Figure 1). The region lies at the southern limit of continuous caribou distribution and the northern extent of Ontario’s Area of the Undertaking (AOU), where commercial forest harvesting is permitted (MNRF 2014*a*). As a result, the Churchill Range represents a transitional zone between relatively intact boreal landscapes and areas subject to forest management.

Ecologically, the Churchill Range is typical of the boreal forest region in northern Ontario (MNRF 2014*a*). The landscape is dominated by black spruce (*Picea mariana*) and jack pine

(*Pinus banksiana*) forest types, shaped by a relatively short fire return interval. Large areas of wetlands and numerous lakes also contribute to the region's ecological complexity. Natural disturbances in the area are primarily driven by wildfire, with localized effects from insect outbreaks, flooding, and blowdown events.

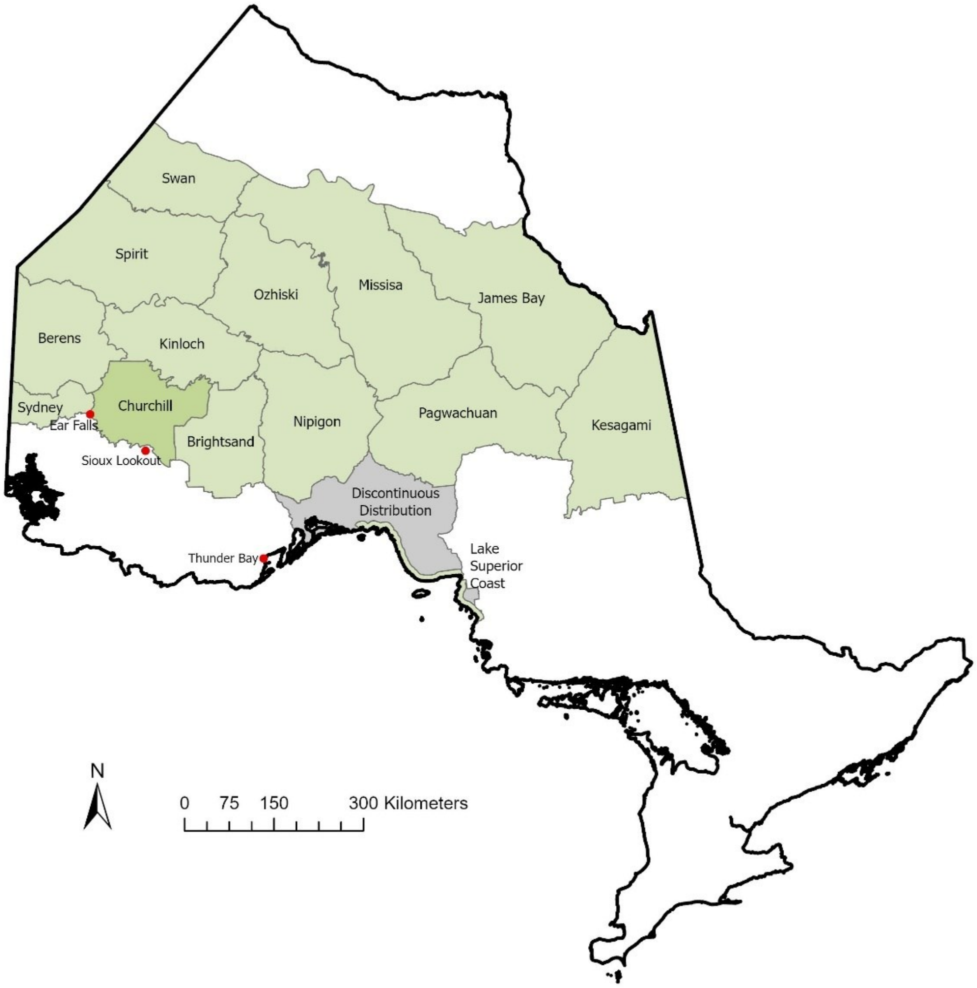

Figure 1. Location of the Churchill Range within the Canadian province of Ontario.

Anthropogenic disturbance in the Churchill Range includes forest harvesting, mineral exploration and development, and linear infrastructure (e.g. roads, trails, pipelines, and railways).

As of the last range assessment in 2012 (MNRF 2014*a*), 41.3% of the range was classified as disturbed with 35.9% attributed to anthropogenic disturbance, with disturbed areas concentrated in the southern and western portions of the range (Figure 2).

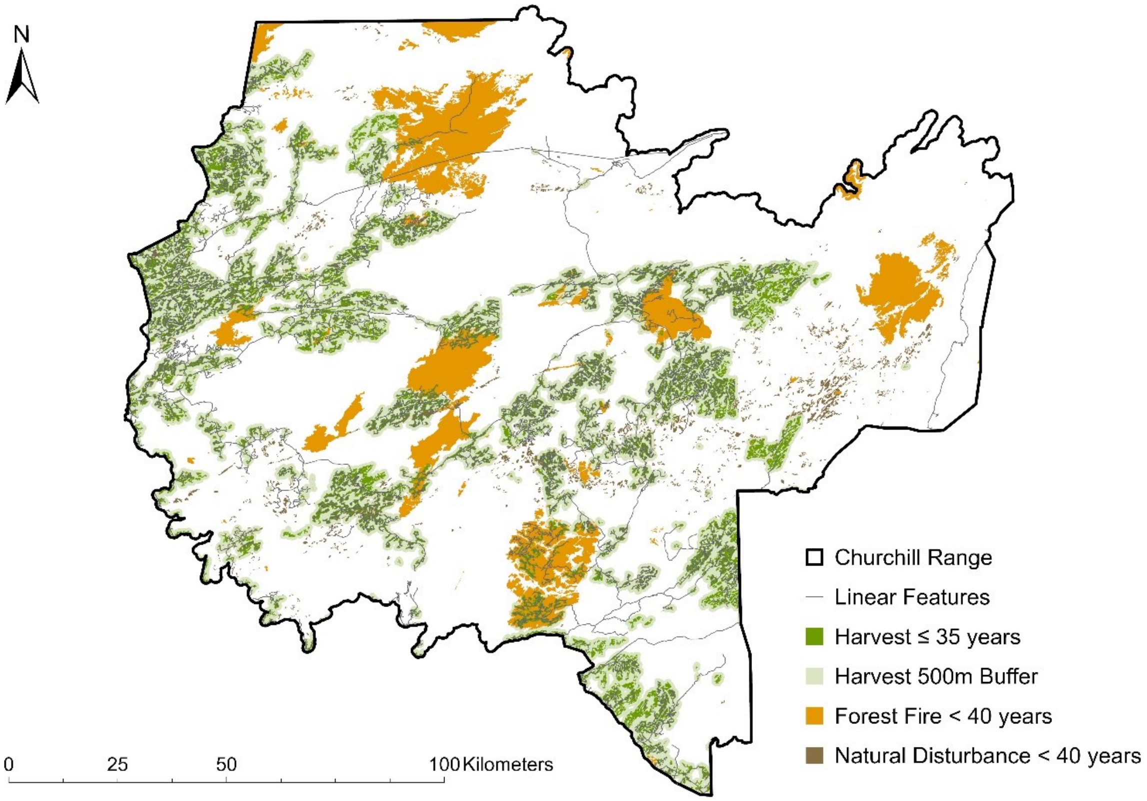

Figure 2. Map of disturbances by type in the Churchill Range of Ontario (see Supporting Table S2 for data sources).

At the time of the most recent habitat assessment, the total area of mature forest available for winter and refuge habitat was within the range of predicted historical estimates based on natural disturbance regimes. Additionally, the spatial arrangement of forest, in primarily large, contigous blocks, remained relatively intact meaning that at both a 6,000 and 30,000 ha scale, the range is not considered fragmented compared to what is expected of a natural landscape for that region.

Despite their threatened status, the most recent population monitoring occurred during the 2012 Integrated Range Assessment (IRA) which placed the minimum animal count, derived from aerial surveys, at 262 individuals with a total population estimated to exceed 300. Caribou occupancy was highest in the north-central portion of the range, while evidence of range retraction was noted in the south. Recruitment rates were low in both 2012 and 2013, with estimated population growth rate below replacement level ($\lambda = 0.96$) (MNRF 2014*a*). A declining population growth trend is also supported by an independent study which reported $\lambda = 0.837$ (Fryxell et al. 2020). Although Canada's Woodland Caribou Recovery Strategy (Environment Canada 2011*b*) classified the range as self-sustaining with moderate risk based on total disturbance levels, the 2012 IRA (MNRF 2014*a*) concluded that the Churchill Range was at an intermediate risk, with uncertainty about whether current habitat conditions are sufficient to maintain a self-sustaining population.

Fecal Sampling

Sampling of caribou fecal pellets was conducted during the winter of 2024 using a three-window design with surveys spaced approximately one month apart in January, February, and March (Hettinga et al. 2012, McFarlane et al. 2020). For each sampling occasion, reconnaissance flights were conducted at 3 km transect intervals using a fixed-wing aircraft (Figure 3). During these flights, two observers recorded the locations of caribou or signs of caribou activity such as tracks, cratering sites, or slushing pits with a global positioning system (GPS). One to two days later, a ground crew accessed these sites by helicopter to locate and collect fecal pellets.

The sampling protocol required a minimum of 10 fecal pellets from each discrete pellet group that could be assigned to an individual. A sample was defined as a group of 10 or more pellets. To ensure that all individuals from a given location were sampled, the number of samples collected was targeted to be approximately 1.4 times the estimated number of caribou present. For example, 14 samples would be collected from a location where 10 animals were estimated to be present. Each sample was labelled with a unique identifier and its GPS location recorded. Samples were kept frozen during field collection and stored at -20°C until DNA extraction.

A total of 601 fecal pellet samples were collected from 53 unique locations in the Churchill Range during the winter of 2024. Of these, 132 were collected in January (occasion 1), 107 in February (occasion 2), and 362 in March (occasion 3). Two sampling locations just beyond the eastern boundary of the Churchill Range were included, as surveyors had tracked caribou from within the range and were confident that the same individuals had moved approximately 1.5 km beyond the boundary, where the best cratering signs and fecal samples were observed.

Sampling efforts were hindered by poor survey conditions. Warm weather with low snow accumulation made it difficult to detect tracks and cratering in January and February. Field observations also suggested that these atypical conditions impacted caribou behaviour, with fewer individuals grouping together to create large cratering sites.

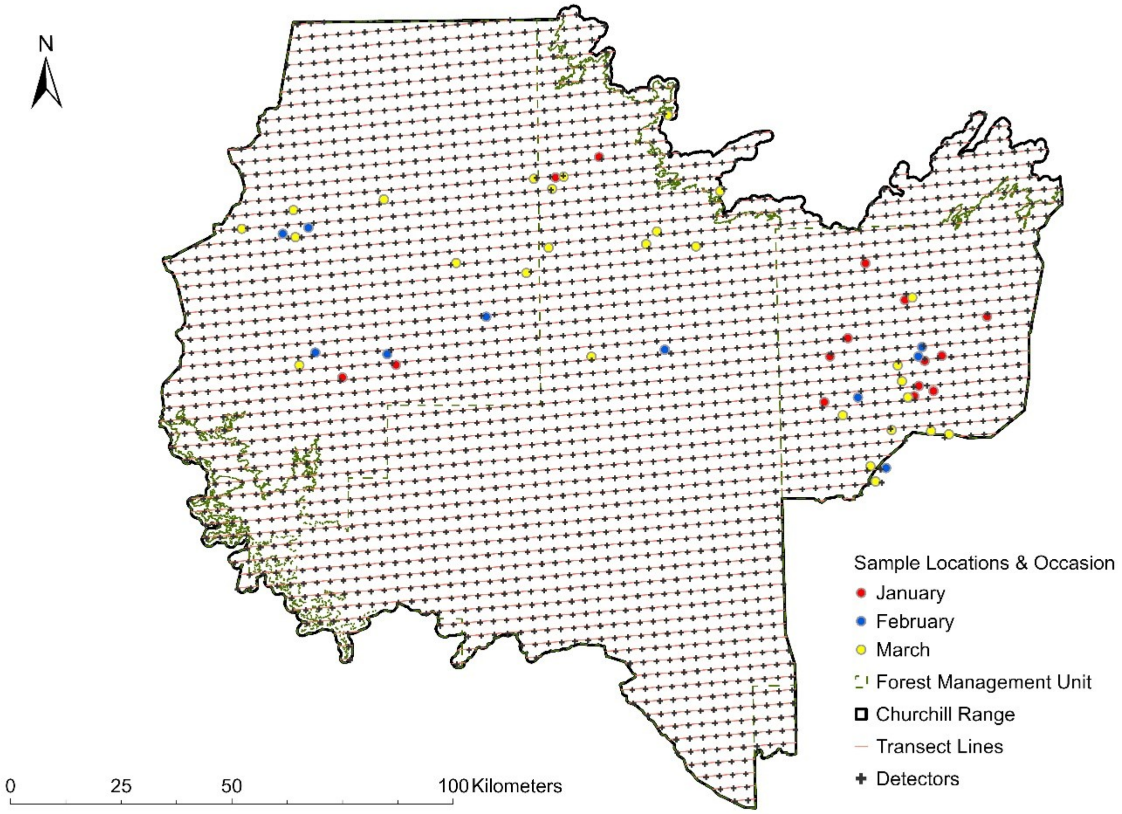

Figure 3. Map of the Churchill Range fecal sample locations by occasion with flight transect lines and SCR detectors.

DNA Extraction and Genotyping

From each sample, four individual fecal pellets were swabbed to remove the mucosal coat. DNA was extracted following a modified version of the protocol described by Ball et al. (2007). Extracted DNA was amplified across three multiplex PCRs targeting a total of 15 polymorphic microsatellite loci (Supporting Table S1). PCR reactions contained 1x Qiagen Multiplex PCR Master Mix (Qiagen, CA), approximately 50 -ng of DNA, -0.1 to 0.4 µM of each primer, and nuclease free water in a total volume of 10 µl. Thermocycling followed the protocol described by Yannic et al. (2016). Sex was determined using the KY1/KY2 primers (Yamauchi et al.

2000), which amplify a portion of the amelogenin gene located on both the X and Y chromosomes in ungulates, following the reaction mixture and thermocycling conditions described by Anderson and Thomson (2024). PCR products were analyzed using a SeqStudio™ Flex Genetic Analyzer (Applied Biosystems, Thermo Fisher Scientific, USA) and fragment sizes were scored using GeneMarker® v3.0.1 (SoftGenetics LLC, State College, PA, USA).

Alleles were scored following standardized guidelines to ensure consistency in genotyping. Samples were separately scored by at least three individuals so that allele calls could be compared to identify potential genotyping errors. Scoring inconsistencies were rectified by re-examining the peak profiles and poor-quality allele profiles were reamplified and genotyped as required. To further reduce potential error, any samples that did not have a genetic match or those that were close genetic matches (i.e. differences of one or two alleles) were re-amplified and rescored.

To assess genotyping error rates, a randomly selected set of 96 samples was reamplified and genotyped. Any mismatch in scoring, except for missing data, was considered an error. Final identification of unique individuals was conducted using the R package *Allelematch* (Galpern et al. 2012), allowing for up to two mismatches between multilocus genotypes.

Population Density Estimation Using SCR

Abundance and population density were estimated using SCR analysis with spatial covariates in the R package *secr* (Efford, 2011). This package uses a maximum likelihood approach to jointly estimate density and detection parameters. The density model expresses the distribution of home ranges centers, while the detection model describes the decrease in detection probability with increasing distance from an individual's home-range centre. Spatial detectors were generated in a 3000 m x 3000 m grid (Figure 3) aligned with the spacing of the aerial reconnaissance transects. Individual detection histories were constructed by recording the presence or absence of each identified individual at each detector during each sampling occasion (McFarlane et al. 2020). The SCR detector type "proximity" was used, as it permits multiple detections of the same individual within a single occasion.

Detection Model Specification

The first stage of model selection involved fitting a set of detection models under the assumption of homogeneous density across the study area (*D* ~ 1). These models allowed g0 and σ to vary between sampling occasions and sex (Poutanen et al. 2023). The parameter g0 represents the probability of detecting an individual at its home-range centre during a given sampling occasion, while σ describes space-use by defining the rate at which detection probability declines with distance from an individuals' home range centre.

A null model was fit, along with models incorporating combinations of covariates effects on g0 and σ, including time (t, T), site (k, K), behaviour response (b, bk), and sex as a group-level covariate. All models used the hazard exponential detection function, which has a longer tail and is suited to capturing occasional long-distance movements characteristic of caribou (Efford 2011, McFarlane et al. 2020, 2022). A mask spacing of 1 km was used with a state-space buffer of 30 km applied to all mark locations to ensure full inclusion of potential home range centers. This value was based on the recommendation to buffer the state space by approximately four times the average home range radius (Efford 2025), with average winter home range radius of caribou in northern Ontario estimated at 7 km (Ferguson and Elkie 2004*b*).

The *secr* package automatically collapses multiple sampling locations into a single occasion using "proximity" detectors, unless time- or behaviour-specific covariates are specified. To ensure consistent model structure and allow valid AICc comparisons across all candidate models, including those with temporal or behavioural covariates, all detection models were initially fit using three distinct occasions. Additional models were fit following *secr*'s default approach using a collapsed-single occasion structure for all models that did not include occasion specific covariates (e.g. time or behaviour). AICc values were used to determine the top ranked detection model for use in subsequent heterogeneous density modelling.

Spatial Covariates for Density Modelling

To evaluate spatial variation in caribou density, we compiled a set of landscape covariates informed by previous studies on caribou habitat selection and distribution. These included forest stand composition (Wittmer et al. 2007, Courbin et al. 2009, Hornseth and Rempel 2016), linear features such as roads (Fortin et al. 2008, Galpern et al. 2012, Beauchesne et al. 2013), natural disturbance including forest fires (Joly et al. 2003, Konkolics et al. 2021), forest harvest (Vors et al. 2007, Courtois et al. 2008, Fryxell et al. 2020), and wetlands (Hornseth and Rempel 2016)-. All spatial data were sourced from the Ontario GeoHub (Ontario 2024), Sustainable Forest License (SFL) holders, and the Ontario Ministry of Natural Resources and Forestry (Supporting Table S2). Raster surfaces were developed using ArcGIS Pro and R and were projected in NAD83 / Ontario MNR Lambert (EPSG:3161).

Land cover data were derived from the Ontario Land Cover Compilation Version 2 (OLCCv2) (Supporting Table S2). We reclassified OLCCv2 into binary rasters representing the presence or absence of three cover types: coniferous (Supporting Figure S1), deciduous (Supporting Figure S2), and wetland (Supporting Figure S3). Moving window analyses were then used to calculate the proportion of each cover type surrounding each 1 km grid cell. Two radii were used: 1km, consistent with previous SCR study on caribou (McFarlane et al. 2022) and 7km, based on the average area used by caribou in winter (Ferguson and Elkie 2004*b*).

We generated Euclidean distance rasters for three categories of linear features to capture variation in access and potential effects on predator movement: (1) all linear features (roads, railways, utility lines), (2) all road types only, and (3) primary roads only (Supporting Figure S4 and Figure S5). Road use, accessibility, and decommissioning status vary by class and may influence the degree to which these features act as barriers or conduits to caribou and their predators (Whittington et al. 2011, Newton et al. 2017).

Recent harvest areas were identified from Operational Planning Inventories (OPIs) and defined as stands ≤35 years old, consistent with the Churchill Range IRA (MNRF 2014*a*). Two raster layers were developed: one representing harvest alone, and another incorporating a 500 m buffer to account for edge effects on habitat use (Environment Canada 2012) (Supporting Figure S6 and S7).

Natural disturbance polygons <40 years old, including fire and windthrow, were extracted from both OPIs and provincial data following Environment Canada (2012). Two disturbance layers were created: (1) a composite of all natural disturbances, and (2) forest fire only (Supporting Figure S8 and Figure S9). Where harvest and natural disturbance overlapped, areas were assigned to the harvest layer only, consistent with MNRF (2014a). Proportional coverage within a 1 km and 7 km moving window was calculated for each disturbance surface.

Covariate Screening and Model Framework

We used a two-stage modeling approach to identify spatial drivers of caribou density. In the first stage, we fit a series of univariate spatial density models to determine the most appropriate spatial representation for each covariate. For example, we compared 1 km and 7 km moving window surfaces for conifer, deciduous, and wetland cover to identify the scale that best captured variation in caribou density. Similarly, we evaluated alternative representations of linear features—including distance to all roads, primary roads only, and all linear features combined—as well as different disturbance layers, such as harvest with and without a 500 m buffer, and forest fire alone versus all natural disturbance types. The goal of this stage was to select the most ecologically meaningful and interpretable surface for each variable. In some cases, multiple land cover variables were retained (e.g., conifer, deciduous, and wetland cover at 7 km) because they represented distinct and complementary aspects of caribou habitat use. Where necessary, correlation among covariates was assessed using Pearson's r, and highly correlated variables ($r > 0.6$) were not included in the same multivariate models.

In the second stage, retained covariates were used to construct 14 multivariate candidate models reflecting competing hypotheses about caribou habitat selection and disturbance avoidance. These models evaluated the relative importance of habitat selection (e.g., conifer and wetland cover), avoidance of anthropogenic disturbance (e.g., roads, harvest), and natural disturbance (e.g., forest fire). A total of 14 models were developed, some with sub-models, to compare specific hypotheses using additive combinations of covariates. Hypotheses included, for example, whether harvest has a stronger effect than forest fire (Model 8), whether habitat features or disturbance better explain variation in density (Model 13), and whether wetlands add explanatory power beyond conifer (Model 7). Model structure, covariates, and hypotheses are detailed in Supporting Table S3. Models were compared using AICc.

RESULTS

Fecal DNA Extraction and Genotyping

Quality control protocol required the removal of 9 samples due to the presence of multiple peaks, suggesting that the sample included DNA from more than one individual. Three samples were also removed due to high missing data. This led to a final dataset of 589 samples that were successfully genotyped for the identification of unique individuals. Genotyping error rates per locus ranged from 0 to 3.2%, with an average of 0.9% across loci. Genotyping analysis revealed a minimum animal count of 245 unique individuals, of which 132 were females and 113 were males.

Homogeneous Density Estimates

There were 35 recaptures across all sampling occasions, 10 were from female samples and 25 were from male samples. This provided an overall recapture rate of 14.3%, or 7.6% and 22.1% for females and males, respectively. Spatial recapture distances ranged from 3 km to 18.9 km, with a mean of 13 km.

The best fitting detection model assuming homogeneous density based on AICc was the null model ($g0 \sim 1$, $\sigma \sim 1$) (AICc = 2604.39) (Table 1). Two other models where g0 and σ varied by occasions (*K*, site transient response) were also highly supported, with ΔAICc <2. As the outputs between these three top fitting models were nearly identical, the top ranked null model structure was selected for the subsequent density modelling with spatial covariates. Density was estimated at 43.6 individuals per 1000 km^2 (95% CI: 31.8 – 59.6), with CV of 16.1%. This equates to an estimated population size (*N*) across the Churchill Range of 937 individuals (95% CI: 682 – 1287). The space-use parameter, σ, was estimated at 5393.5 m (95% CI: 4368.5 to 6658.8) and the detection probability at 0.0049 (95% CI: 0.0029 to 0.0082). The detection models where g0 and σ were allowed to vary by sex were not well supported (AICc = 2824.39 to 2826.04) (Table 2). A model where density varied by sex did not perform well based on AICc (ΔAICc >2 compared to the null model) (Table 3).

Table 1. Detection model selection results from SCR analysis in the Churchill Range.

Model	**Occasions**	**AICc**	**D**	**D 95% CI**	**SE (D)**	**CV (D)**	**g0**	**g0 95% CI**	***σ***	***σ* 95% CI**	***N***	***N* 95% CI**
D ~ 1, g0 ~ 1, σ ~ 1	1	2604.39	43.6	31.8 - 59.6	7.02	16.1%	0.0049	0.0029 - 0.0082	5393.5	4368.5 - 6658.8	928	678 - 1270
D ~ 1, g0 ~ 1, σ ~ K	1	2606.46	43.6	31.8 - 59.6	7.02	16.1%	0.0049	0.0029 - 0.0082	5393.2	4368.4 - 6658.6	928	678 - 1270
D ~ 1, g0 ~ K, σ ~ 1	1	2606.46	43.6	31.8 - 59.6	7.02	16.1%	0.0049	0.0029 - 0.0082	5393.3	4368.4 - 6658.6	928	678 - 1270
D ~ 1, g0 ~ K, σ ~ K	1	2608.55	43.6	31.8 - 59.6	7.02	16.1%	0.0049	0.0029 - 0.0082	5393.3	4368.4 - 6658.6	928	678 - 1270
D ~ 1, g0 ~ K, σ ~ 1	3	2937.34	90.2	65.4 - 124.4	14.88	16.5%	0.0064	0.0041 - 0.0102	4504.9	3853.7 - 5266.1	1921	1394 - 2649
D ~ 1, g0 ~ 1, σ ~ K	3	2938.55	90.1	65.3 - 124.2	14.86	16.5%	0.0063	0.0040 - 0.0100	4545.9	3888.2 - 5314.8	1919	1392 - 2645
D ~ 1, g0 ~ K, σ ~ K	3	2939.40	90.2	65.4 - 124.4	14.88	16.5%	0.0065	0.0041 - 0.0103	4495.8	3836.0 - 5269.0	1922	1394 - 2649
D ~ 1, g0 ~ t, σ ~ t	3	3124.23	41.7	32.0 - 54.3	5.65	13.6%	0.0015	0.0008 - 0.0029	8685.9	6597.6 - 11435.3	887	681 - 1156
D ~ 1, g0 ~ t+b, σ ~ 1	3	3148.24	107.3	46.4 - 248.1	48.08	44.8%	0.0015	0.0006 - 0.0039	5436.3	4404.2 - 6710.2	2285	988 - 5285
D ~ 1, g0 ~ t, σ ~ 1	3	3156.24	43.5	31.8 - 59.6	7.02	16.1%	0.0038	0.0022 - 0.0066	5389.3	4364.6 - 6654.5	927	677 - 1269
D ~ 1, g0 ~ 1, σ ~ k	3	3157.09	56.6	39.5 - 81.0	10.45	18.5%	0.0041	0.0025 - 0.0066	5113.3	4212.1 - 6207.5	1206	841 - 1725
D ~ 1, g0 ~ 1, σ ~ t+b	3	3157.25	79.3	41.4 - 151.8	27.01	34.1%	0.0041	0.0023 - 0.0073	3938.9	2909.5 - 5332.5	1689	882 - 3233
D ~ 1, g0 ~ t+bk, σ ~ 1	3	3157.75	43.2	31.6 - 59.1	6.95	16.1%	0.0038	0.0022 - 0.0067	5359.0	4335.7 - 6623.8	920	673 - 1259
D ~ 1, g0 ~ T σ ~ T	3	3158.31	43.3	31.8 - 59.2	6.93	16.0%	0.0012	0.0006 - 0.0023	8411.6	6472.3 - 10932.1	922	677 - 1260
D ~ 1, g0 ~ k, σ ~ k	3	3158.53	59.5	40.3 - 87.8	11.95	20.1%	0.0036	0.0020 - 0.0064	5292.4	4278.2 - 6547.2	1267	858 - 1870
D ~ 1, g0 ~ 1, σ ~ t	3	3161.85	43.3	31.6 - 59.2	6.98	16.1%	0.0052	0.0031 - 0.0085	4796.5	3869.0 - 5946.5	922	673 - 1261
D ~ 1, g0 ~ 1, σ ~ t+bk	3	3163.35	43.1	31.5 - 58.9	6.94	16.1%	0.0053	0.0032 - 0.0087	4769.6	3843.6 - 5918.7	918	671 - 1255
D ~ 1, g0 ~ k, σ ~ 1	3	3171.04	60.1	40.6 - 88.9	12.12	20.2%	0.0028	0.0016 - 0.0049	5946.4	4812.7 - 7347.2	1280	865 - 1894
D ~ 1, g0 ~ T, σ ~ 1	3	3173.03	43.5	31.8 - 59.6	7.02	16.1%	0.0029	0.0017 - 0.0051	5390.0	4365.3 - 6655.1	927	677 - 1269
D ~ 1, g0 ~ b, σ ~ b	3	3178.59	131.8	56.6 - 307.0	59.60	45.2%	0.0009	0.0003 - 0.0022	7050.5	5778.5 - 8602.4	2807	1206 - 6538
D ~ 1, g0 ~ 1, σ ~ T	3	3180.45	43.2	31.6 - 59.2	6.98	16.1%	0.0047	0.0029 - 0.0078	4419.8	3556.3 - 5492.9	921	673 - 1261
D ~ 1, g0 ~ b, σ ~ 1	3	3187.91	134.5	57.9 - 312.7	60.69	45.1%	0.0014	0.0005 - 0.0037	5445.2	4411.3 - 6721.4	2865	1233 - 6661
D ~ 1, g0 ~ 1, σ ~ b	3	3192.99	97.8	51.6 - 185.3	32.77	33.5%	0.0038	0.0022 - 0.0068	3923.1	2898.0 - 5310.9	2082	1098 - 3948
D ~ 1, g0 ~ 1, σ ~ 1	3	3205.84	43.6	31.8 - 59.6	7.02	16.1%	0.0049	0.0029 - 0.0082	5393.3	4368.5 - 6658.7	928	678 - 1270
D ~ 1, g0 ~ 1, σ ~ bk	3	3207.45	43.3	31.7 - 59.3	6.98	16.1%	0.0050	0.0030 - 0.0083	5364.2	4341.8 - 6627.4	922	675 - 1263
D ~ 1, g0 ~ bk, σ ~ 1	3	3207.47	43.3	31.7 - 59.3	6.98	16.1%	0.0050	0.0030 - 0.0083	5364.2	4341.7 - 6627.4	922	675 - 1263
D ~ 1, g0 ~ bk, σ ~ bk	3	3209.52	43.3	31.7 - 59.3	6.97	16.1%	0.0050	0.0030 - 0.0083	5370.2	4346.3 - 6635.2	923	674 - 1263

D: estimated density (individuals per 1000km^2), g0: baseline detection probability, σ: space use parameter (m), N: estimated total population, K: site (detector) transient response, k: site (detector) learned response, t: time (occasion), T: time (occasion) trend, b: learned behavioural response, bk: behavioural & site learned response.

Table 2. Detection model selection results from SCR analysis in the Churchill Range, with variation in g0 and σ by sex.

Females												
Model	**Occasions**	**AICc**	**D**	**D 95% CI**	**SE (D)**	**CV (D)**	**g0**	**g0 95% CI**	***σ***	***σ* 95% CI**	***N***	***N* 95% CI**
D ~ 1, g0 ~ 1, σ ~ g	1	2824.39	21.6	15.8 - 29.5	3.47	16.1%	0.0151	0.0089 - 0.0257	5285.96	4168.8 - 6673.7	460	682 -1278
D ~ 1, g0 ~ g, σ ~ 1	1	2824.45	21.6	15.8 - 29.6	3.48	16.1%	0.0149	0.0088 - 0.0254	5353.92	4331.5 - 6617.7	460	682 - 1278
D ~ 1, g0 ~ g, σ ~ g	1	2826.04	21.7	15.9 - 29.7	3.49	16.1%	0.0195	0.0078 - 0.0483	4675.99	3050.6 - 7167.4	462	682 - 1257
Males												
Model	**Occasions**	**AICc**	**D**	**D 95% CI**	**SE (D)**	**CV (D)**	**g0**	**g0 95% CI**	***σ***	***σ* 95% CI**	***N***	***N* 95% CI**
D ~ 1, g0 ~ 1, σ ~ g	1	2824.39	21.6	15.8 - 29.5	3.47	16.1%	0.0151	0.0089 - 0.0257	5375.3	4338.1 - 6660.5	460	682 -1278
D ~ 1, g0 ~ g, σ ~ 1	1	2824.45	21.6	15.8 - 29.6	3.48	16.1%	0.0151	0.0088 - 0.0259	5353.9	4331.5 - 6617.7	460	682 - 1278
D ~ 1, g0 ~ g, σ ~ g	1	2826.04	21.7	15.9 - 29.7	3.49	16.1%	0.0137	0.0075 - 0.0251	5593.6	4369.0 - 7161.4	462	682 - 1257

D: estimated density (individuals per 1000km^2), g0: baseline detection probability, *σ*: space use parameter (m), *N*: estimated total population, g: group (sex).

Table 3. Density model results with variation by sex as a group.

Females										
Model	**AICc**	**D**	**D 95% CI**	**CV (D)**	**g0**	**g0 95% CI**	***σ***	***σ* 95% CI**	***N***	***N* 95% CI**
D ~ g, g0 ~ 1, σ ~ 1	2822.99	23.3	16.68 - 32.53	17.2%	0.0150	0.0089 - 0.0252	5354.11	4331.64 - 6617.94	496.08	355.28 - 692.89
Males										
Model	**AICc**	**D**	**D 95% CI**	**CV (D)**	**g0**	**g0 95% CI**	***σ***	***σ* 95% CI**	***N***	***N* 95% CI**
D ~ g, g0 ~ 1, σ ~ 1	2822.99	19.9	14.17 - 28.05	17.5%	0.0150	0.0089 - 0.0252	5354.11	4331.64 - 6617.94	424.72	301.82 - 597.47

D: estimated density (individuals per 1000km^2), g0: baseline detection probability, σ: space use parameter (m), N: estimated total population, g: group (sex).

Spatial Density Estimates

All spatial covariates tested during univariate analysis improved model fit, with ΔAICc values at least 10 points lower than the null model. All cover and disturbance variables performed better when calculated using a 7 km moving window compared to a 1 km moving window. Unless otherwise stated, all cover and disturbance covariates discussed from this point forward are with a 7 km moving window.

The conifer cover model had the lowest AICc value (2250.25), greatly outperforming the null model (AICc = 2604.39). Among linear disturbance variables, distance to all roads had the strongest support (AICc = 2519.14). Among forest harvest covariates, the model including buffered (500 m) forest harvest was the top performer (AICc = 2525.68). The model including forest fire alone was better supported than the model including all natural disturbance types (AICc = 2569.65). Based on these results, the following covariates were selected for multivariate analysis: conifer cover, wetland cover, deciduous cover, harvest (500 m buffer), forest fire, and distance to all roads (Table 4). Supporting Table S4 includes the full univariate model selection results for land cover, linear features, harvest, and natural disturbance covariates.

Table 4. Spatial univariate model selection results from the Churchill Range.

Model	**AICc**	**D CV**	**β**	**β CV**
D ~ conifer cover (7), g0 ~ 1, σ ~ 1	2250.25	16.0%	3	7%
D ~ deciduous cover (7), g0 ~ 1, σ ~ 1	2511.91	16.0%	-1.3	16%
D ~ distance to all roads, g0 ~ 1, σ ~ 1	2519.14	16.1%	1.8	10%
D ~ harvest with 500m buffer (7), g0 ~ 1, σ ~ 1	2525.68	16.0%	-0.9	14%
D ~ forest fire (7), g0 ~ 1, σ ~ 1	2569.65	16.0%	-8.4	308%
D ~ wetland cover (7), g0 ~ 1, σ ~ 1	2593.08	16.1%	0.4	28%

D: density, g0: detection parameter, σ: space use parameter, (7): 7 km moving window.

All multivariate models produced stable density estimates (ranging from 42 to 46 individuals per 1000km^2), with coefficient of variation around 16% (Table 5). Although the full model including 6 covariates had the lowest AICc (2215.51), it appeared overfit. In this model, the estimate of the effect of forest fire was highly uncertain (β = -7.91, SE = 19.63, CV = 308%), and the effect of harvest was negligible (β = 0.01), suggesting little explanatory power (Supporting Table S5). The effect from harvest changed drastically from the univariate analysis, where its β estimate was -0.9 (Supporting Table S5). Therefore, spatial patterns in caribou density were interpreted based on the remaining multivariate models. The second-best model, which included four covariates and an AICc of 2237.76 was selected as the top performing model. This model yielded both reliable density estimates and well supported β coefficients, providing the most robust basis for evaluating spatial heterogeneity in density.

In the top multivariate model, conifer cover emerged as the strongest predictor of caribou density, followed closely by deciduous cover and distance to all roads. Conifer cover had a strong positive effect on caribou density (β = 2.63), greater distance to roads was associated with higher density (β = 1.17), deciduous cover had a negative effect (β = -0.86), and wetland cover had a moderately positive effect (β = 0.50).

Predicted caribou densities under this model generally ranged from 0 to 300 individuals per 1000 km^2. However, a few small clusters exhibited extremely high predicted densities (~4,600 individuals per 1000 km^2) (Figure 4). The average predicted density was 42.3 individuals per 1000 km^2, slightly lower than the null model estimate of 43.6 individuals per 1000 km^2. Given the uncertainty in predicted spatial heterogeneity, the estimates of density, detection probability (g0), and space-use (σ) from the top detection model were considered more reliable than predictions from the spatial covariate model.

Table 5. Multivariate density model selection results from SCR analysis in the Churchill Range.

Model	**AICc**	**ΔAICc**	**D**	**D CV**
D ~ conifer cover (7) + wetland cover (7) + deciduous cover (7) + distance to all roads + harvest with 500m buffer (7) + forest fire (7)	2215.51	0.00	45.0	16.0%
D ~ conifer cover (7) + wetland cover (7) + deciduous cover (7) + distance to all roads	2237.76	22.25	42.3	16.3%
D ~ conifer cover (7) + deciduous cover (7)	2241.19	25.68	40.2	16.0%
D ~ conifer cover (7) + wetland cover (7)	2249.31	33.80	40.7	16.0%
D ~ distance to all roads + deciduous cover (7) + harvest with 500m buffer (7)	2400.91	185.40	46.3	16.1%
D ~ distance to all roads + deciduous cover (7)	2403.62	188.11	46.1	16.1%
D ~ forest fire (7) + harvest with 500m buffer (7)	2467.24	251.73	45.6	16.0%
D ~ distance to all roads + harvest with 500m buffer (7)	2515.46	299.95	46.1	16.0%
D: average estimated density (individuals per 1000km^2), (7): 7 km moving window.				

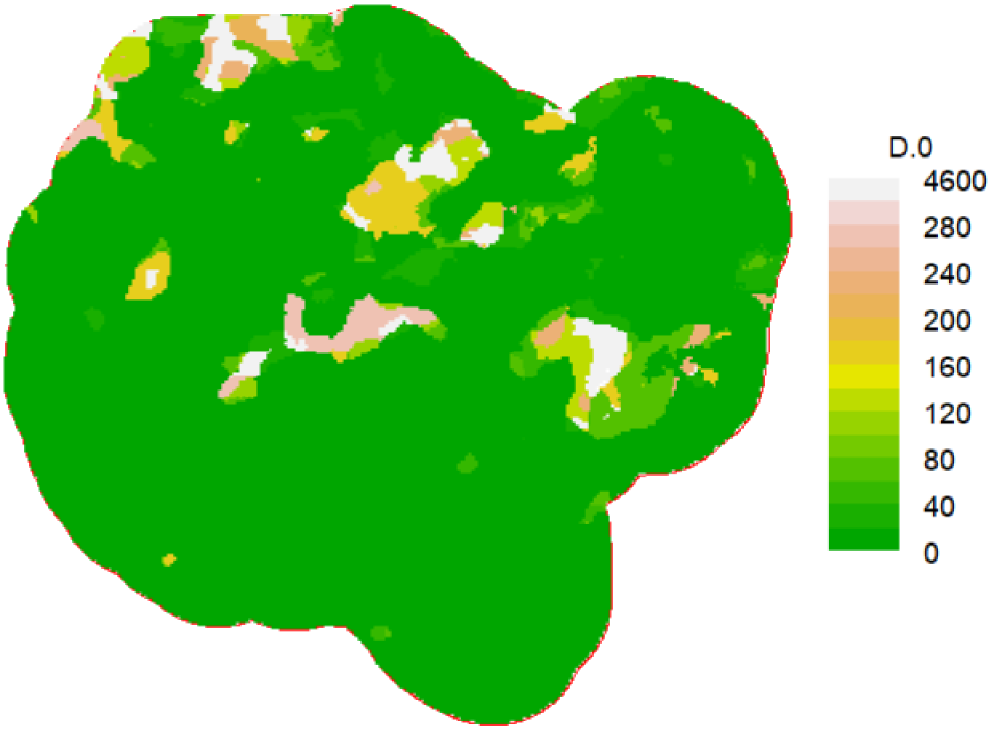

Figure 4. Estimated density surface map from the Churchill Range SCR analysis including a 30 km state-space buffer (D.0 = individuals per 1000 km^2).

DISCUSSION

Model Support and Movement Ecology

The strong support for the null detection model, which assumed constant g0 and σ, suggests that caribou detection probability and space use remained relatively stable across the study area. Although models allowing g0 and σ to vary by site received comparable AICc support, the similarity in their estimates further supports the use of the simpler model. In contrast, models incorporating behavioural or temporal effects were clearly less supported, indicating little evidence that detection or movement patterns varied over time or in response to the survey protocol. The lack of behavioural variation in detection probability may be attributed to the nature of the sampling methods itself; because detection was based on fecal pellets located along aerial transects, animals had no opportunity to exhibit “trap happy” or “trap-shy” behaviour (Pradel and Sanz-Aguilar 2012). Similarly, the absence of temporal variation in the space-use parameter suggests consistent caribou movement patterns throughout the survey period. This contrasts with previous findings in northern Ontario, where early winter home ranges were observed to be larger than those in late winter (Ferguson and Elkie 2004*a*). One possible explanation is the unusually low snow accumulation during the 2024 survey period, which may have reduced the energetic costs of movement and limited the need for caribou to alter their movement patterns in response to changing snow depth (Pereira et al. 2024).

The hypothesis that home range size would vary between sexes was not supported. While some studies have found larger home ranges for male ungulates (Ofstad et al. 2016), models allowing σ and g0 to vary by sex were among the lowest ranked in our analysis. For comparison, McFarlane et al. (2022) estimated winter home range radii of 12 km for females and 6 km for males in Saskatchewan. In contrast, Ferguson and Elkie (2004*b*) reported no significant sex-based differences in northern Ontario, a finding further supported by our results. The nearly equal representation of males and females in our dataset may have contributed to similar detection probabilities between sexes. However, the relatively low number of spatial recaptures in this study may have limited our ability to detect more subtle sex-based differences in movement parameters.

Despite this limitation, our study provides valuable insight into winter space use in the Churchill Range. Spatial recapture distances extended up to ~18 km, suggesting that some individuals used relatively large areas during the sampling period. The estimated σ value of 5.4 km (95% CI: 4.4 – 6.7 km) indicates moderate variation in individual movement, with a coefficient of variation below 20%. Compared to other SCR-based estimates, σ in the Churchill Range is larger than in Alberta (McFarlane et al. 2020), possibly reflecting lower disturbance levels as the amount of disturbed area in the Alberta ranges studied varied from 62 – 95% (Environment Canada 2012), compared to 41.3% in the Churchill Range (MNRF 2014*a*).

Although σ is not a direct measure of home range size, it provides a robust index of space use and can be used to approximate the radius or area of an animal's activity during the study period (Royle et al. 2014). Notably, our σ estimate of a 5.4 km, equating to an estimated home range radius of 25.6 km or home range area of 2058.9 km^2, falls within the range of home range sizes reported in the literature, typically ranging from 200 to 4000 km^2 depending on disturbance levels (Racey et al. 1999, Brown et al. 2003, OMECP 2020). However, it is larger than home range size radius previously reported for caribou in northern Ontario at 7 km (Ferguson and Elkie 2004*b*) and larger than an area of 828 km^2 previously found in the Churchill Range (Wilson et al. 2019).

Comparison of Density Estimates

Our density estimate of 43.6 individuals per 1,000 km^2 aligns well with values reported in other boreal regions. In Alberta, caribou density estimates range from 16.1 to 61.9 individuals per 1,000 km^2, with most ranges supporting between 30 and 50 individuals per 1,000 km^2 (McFarlane et al. 2020). A preliminary population density estimate from the neighbouring Brightsand Range of Ontario was 36.6 individuals per 1000 km^2, (McFarlane et al., in preparation). Thus, our population density estimate is well within the expected range for caribou in Canada. Compared to the Brightsand Range, the slightly higher density estimate obtained in this study could reflect the importance of the spatial arrangement of suitable winter habitat. The two ranges contain nearly the same proportion of winter habitat, with 41.2% of its total area as suitable habitat in the Churchill Range (MNRF 2014*a*) and 40.1% of its total area as suitable winter habitat in the Brightsand Range (MNRF 2014*b*). However, the Churchill Range has more favourable spatial arrangement of habitat with less fragmentation of winter habitat areas than the Brightsand Range (MNRF 2014*a*, *b*). Although there was some variation in survey design between the Brightsand and Churchill Range studies (5 km transect spacing and two sampling occasions in the Brightsand Range, compared to 3 km transect spacing and three sampling occasions within the Churchill Range), the detection probability, total samples, and number of recaptures were lower in the Churchill Range than the Brightsand Range. Taken together, this suggests that the higher density in Churchill is not due to greater sampling efforts but may be attributable to the arrangement of suitable winter habitat as previously mentioned.

Landscape Drivers of Caribou Density

This study supports the hypotheses that both land cover and disturbance features influence caribou density. Univariate models with cover type predictors received stronger AICc support than those with disturbance variables, indicating that habitat composition explained more variation in caribou density within the study area. This finding is consistent with a previous SCR study that found a strong influence of habitat quality on caribou density (McFarlane et al. 2022). However, in our analysis, multivariate models that included both habitat and disturbance variables were better supported than univariate models, suggesting that a combination of habitat and anthropogenic factors best captures spatial patterns in caribou density. Notably, the strong positive association between caribou density and greater distance to roads in our study is consistent with findings from Saskatchewan (McFarlane et al. 2022).

Among disturbance-related covariates, distance to all roads has greater explanatory power than distance to linear features or to primary roads alone. This suggests that non-road linear features had limited influence in this study, although this may reflect their relatively low abundance in the study area. Furthermore, the stronger effect of all roads compared to primary roads suggests that operational and branch roads still contribute to habitat fragmentation and predator movement.

Although harvest alone was associated with a strong negative effect on caribou density in the univariate models, this effect diminished when distance to roads was included in multivariate models. This pattern suggests that apparent impact of harvest may, in part, reflect the creation of access roads rather than the direct effect of vegetation removal. In managed forests, harvest activities are accompanied by the construction of roads, which can increase predator access and thereby elevate predation risk for caribou (Whittington et al. 2011). Nevertheless, harvest remains an important indirect factor, as it alters forest composition by reducing mature conifer cover (Darby and Duquette 1986).

Consistent with McFarlane et al. (2022), we found that caribou density was higher in contiguous, conifer-dominated areas located farther from roads, and lower in areas dominated by deciduous cover. These patterns suggest consistent habitat responses across boreal populations. The support for multivariate models in this analysis also agree with earlier resource selection studies, which found that caribou respond to multiple factors, including competition, disturbance, and forage availability (Hornseth and Rempel 2016). Collectively, these findings underscore the cumulative influence of both habitat composition and anthropogenic disturbance on caribou space use and density.

Survey Efficiency, Detection Challenges, and Design Implications

Despite a lower than ideal recapture rate (14%), density estimates in this study were precise, with a coefficient of variation of 16%. This outcome is consistent with species-specific simulation indicating that a single season of well-designed sampling can produce reliable estimates for caribou (McFarlane et al. 2020). In contrast, general SCR simulations have found that multiple years of sampling are required to achieve comparable precision (Schmidt et al. 2022). These results highlight that while reliable mean density estimates can be achieved under modest recapture rates, model performance may still be limited in other respects. In particular, low recapture rates reduce the ability to detect behavioural or temporal variation in detection (Schmidt et al. 2022), and limit power to model spatial heterogeneity in density (Royle et al. 2018).

This study yielded relatively low detection probabilities (g0) compared to Alberta caribou studies, where estimates ranged from 0.013 to 0.053 (McFarlane et al. 2020). The lower value observed here likely reflects the unusually warm, low-snow conditions during the winter 2024 survey, which made sample detection more difficult. These effects are also evident in the low number of samples collected in January and February when snow depth was much lower than in March. Such conditions might become more frequent as climate change increases variability in in winter severity and snow cover (Contosta et al. 2019). To buffer against this unpredictability, future efforts might consider extending monitoring across multiple years to ensure adequate sample sizes and improve model reliability.

Typically, detector spacing up to 2 x σ is recommended to balance sampling effectiveness with associated costs (Royle et al. 2013, Sun et al. 2014). In this study, the estimated movement parameter (σ) was 5.4 km, suggesting future surveys in similar landscapes could adopt a more cost-efficient transect spacing of 5 km to align with the recommended detector spacing of 1 x σ (Sollmann et al. 2012). While this contrasts with SCR simulations in Alberta where σ was 2.3 km and a 3 km spacing was recommended (McFarlane et al. 2020), the larger σ in the Churchill Range justifies a wider transect spacing. The utility of 5 km spacing is further supported by its use in the neighbouring Brightsand Range, which also produced reliable density estimates (McFarlane et al., in preparation). However, maintaining finer spacing (e.g. 3 km) may be advantageous under poor winter conditions, as experienced during the 2024 survey. When detection probability is low (e.g., $g0 \leq 0.05$), finer detector spacing has been shown to enhance model performance (Sun et al. 2014). Thus, while 5 km spacing may be adequate under normal conditions, denser coverage may be warranted in years with unfavourable snow conditions.

The predicted density surface from the top multivariate model included localized outliers with unrealistically high values, reaching up to 4,600 individuals per 1,000 km². These extreme estimates likely resulted from extrapolation into habitat conditions that were rare in the Churchill landscape—specifically, areas with nearly 100% conifer cover, which are uncommon at the 7 km scale. To improve the biological realism of spatial predictions, future studies should consider constraining model predictions to the range of covariate values observed during sampling. Additionally, using composite habitat quality indices—such as percent suitable habitat—instead of individual land cover types (e.g., conifer or deciduous cover) may help reduce overfitting and improve predictive performance. A similar approach was used in McFarlane et al. (2022) where habitat quality was represented as a composite metric and modeled alongside disturbance variables to explain broad-scale variation in caribou density. Despite these limitations, most local density estimates fell within the range of 0 to 300 individuals per 1,000 km^2, which is comparable to values reported in Saskatchewan (McFarlane et al. 2022). Moreover, the mean estimate from the multivariate model closely matched that of the homogeneous model. This consistency suggests that while fine-scale spatial patterns should be interpreted with caution, overall density estimates remain robust when sampling is extensive and well-distributed—even for species that exhibit spatial clustering (Royle et al. 2014, McFarlane et al. 2020).

MANAGEMENT IMPLICATIONS

Spatial clustering of caribou in the northern, central, and northeastern portions of the Churchill Range, areas also highlighted in the 2012 Integrated Range Assessment (MNRF 2014*a*), suggests these regions may warrant priority for habitat protection and disturbance mitigation. This spatial consistency across time reinforces their ecological importance for maintaining caribou occupancy in the range.

The strong positive association with conifer cover and greater distance to roads reinforces management strategies focused on road decommissioning and the maintenance of large, contiguous coniferous forest areas (OMNR 2014, OMECP 2019). Notably, distance to all roads was a stronger predictor of density than distance to primary roads or linear features alone, highlighting that secondary and tertiary roads also contribute to habitat fragmentation and facilitate predator movement (Newton et al. 2017). Recovery strategies should therefore extend beyond primary road management to include decommissioning or restoring older access roads and addressing the cumulative effects of all road types.

The minimum animal count of 245 obtained in this study closely aligns with value reported in the 2012 IRA (minimum animal count = 262; MNRF 2014*a*), suggesting stability in overall range-level use. However, our SCR approach produced a population estimate that was substantially higher than expected based on past monitoring efforts. This discrepancy does not likely reflect true population growth, but rather the ability of SCR to account for undetected individuals when modelling population size compared to the aerial survey method applied in the 2012 minimum animal count estimate that relied on visually identifying individuals. These findings offer confidence that caribou abundance in the Churchill Range may be greater than previously assumed, though further monitoring is needed to assess trends over time.

To improve population monitoring in Ontario and across Canada, we strongly recommended continued use of SCR paired with non-invasive genetic sampling. Our results show that precise density estimates can be obtained from a single season of well-designed sampling, even under suboptimal winter conditions. Based on the estimated movement parameter (σ = 5.4 km), future surveys could adopt a 5 km transect spacing to increase efficiency and reduce costs. However, finer spacing may be needed during years with poor detection, and multi-year sampling is advisable to buffer against interannual variation and improve recapture rates. This would support more detailed modeling of behavioural, temporal, or sex-based variation in detection and space use.

Overall, these findings emphasize that effective caribou conservation will require an integrated approach that combines robust, standardized monitoring with targeted landscape management. Strategic road network mitigation, habitat conservation, and long-term planning tailored to regional landscape conditions will be essential for supporting caribou persistence across the Churchill Range.

ETHICS STATEMENT

This research involved non-invasive sampling and did not require animal handling or capture; therefore, no permits were necessary. All fieldwork was conducted by contracted professionals in accordance with established safety protocols. Laboratory work involving fecal pellet analysis was approved under a biosafety application by Lakehead University.

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SUPPORTING INFORMATION

Table S1. Microsatellite primer information.

Locus	Size Range	Multiplex	Primer (µM)	5' Dye	Forward (5'-3')	Reverse (5'-3')	Reference
RT6	93-109	1	0.4	6FAM	TTCCTCTTACTCATTCTTGG	CGGATTTTGAGACTGTTAC	Wilson et al. 1997
BM6506	196-209	1	0.2	6FAM	GCACGTGGTAAAGAGATGGC	AGCAACTTGAGCATGGCAC	Bishop et al. 1994
NVHRT30	147-174	1	0.15	VIC	GTGGAGCATTGTGTATGTGT	GCCCCCACTGTGTTTT	Røed & Midthjell 1998
FCB193	90-114	1	0.2	NED	TTCATCTCAGACTGGGATTCAGAAAGGC	GCTTGGAAATAACCCTCCTGCATCCC	Buchanen and Crawford 1993
BMS1788*-PET	113-143	1	0.2	PET	ATTCATATCTACGTCCAGATTCAGATTTCTTG	GGAGAGGAATCTTGCAAAGG	Yannic et al 2014 (modified from Cronin et al. 2005)
RT27	131-160	2	0.1	6FAM	CCAAAGACCCAACAGATG	TTGTAACACAGCAAAAGCATT	Wilson et al. 1997
RT7	217-234	2	0.15	VIC	CCTGTTCTACTCTTCTTCTC	ACTTTTCACGGGCACTGGTT	Wilson et al. 1997
RT9	103-130	2	0.3	VIC	TGAAGTTTAATTTCCACTCT	CAGTCACTTTCATCCCACAT	Wilson et al. 1997
RT5	142-170	2	0.1	NED	TGGTTGGAAGGAAAACTTGG	CCTCTGCTCCTCAAGACAC	Wilson et al. 1997
Map2C-PET	90-113	2	0.2	PET	TGGTTGGAAGGAAAACTTGG	CCTCTGCTCCTCAAGACAC	Bishop et al. 1994
RT1	222-237	3	0.2	6FAM	AGGCCATATAGGAGGCAAGCTT	CATCTTCCCATCCTCTTTAC	Wilson et al. 1997
BM888	172-195	3	0.1	VIC	CACTTGGCTTTTGGACTTA	CTGGTGTATGTATGCACACT	Bishop et al. 1994
OheQ*	260-300	3	0.2	VIC	AGACCTGATTACAATGTGTCAGTGAAGGTCTTC	GATGGACCCATCCAGGCAACCATCTAG	Yannic et al. 2014 (modified from Jones et al. 2000)
BM848	361-384	3	0.3	NED	TGCCTTCTTTCATCCAACAA	CATCTTCCCATCCTCTTTAC	Bishop et al. 1994
RT24	207-234	3	0.2	NED	TGTATCCATCTGGAAGATTTCAG	CAGTTTAACCAGTCCTCTGTG	Wilson et al. 1997
RT30	193-210	3	0.2	PET	TCAGCAATTCAGTACATCACCC	GCGCAAGTTTCCTCATGC	Wilson et al. 1997
KY1/KY2		4	0.2				
zfX/zfY		4	0.2				

Table S2. Spatial covariate data sources.

Data Type	Source	Download Date	Last Updated
Roads, harvest, and natural disturbance in the Trout Lake, Wabigoon, and Lac Suel Forest Management Units	Dryden Fibre Canada ULC	October 25, 2024	Roads: 2023 Disturbance: 2021, 2019, and 2024, respectively.
Roads, harvest, and natural disturbance in the Red Lake Forest Management Unit	Red Lake Forest Management Company	October 28, 2024	Roads: 2023 Disturbance: 2016
Roads, harvest, and natural disturbance in the Dryden Forest Management Unit	Dryden Forest Management Company	October 28, 2024	Roads: 2021 Disturbance: 2018
Roads, harvest, and natural disturbance in the Caribou, English River, and Black Spruce Forest Management Units	Domtar Forest Products, Thunder Bay, On	November 8, 2024	Roads: April 2024 Disturbance: April 2024
Roads, harvest, and natural disturbance in the White Feather and Whiskey Jack Forest Management Units	Ontario Ministry of Natural Resources, NW Regional Resources Unit	October 29, 2024	Roads: 2023 Disturbance: 2019 and 2021, respectively.
Supplementary roads data and roads north of Area of Undertaking: Ontario Road Network (ORN)	https://geohub.lio.gov.on.ca/datasets/2fd52bccdb77479da0133c86545503f8_0/explore?location=50.448486%2C-89.651726%2C6.41	October 24, 2024	2019
Supplementary primary roads data: Canada Road Network File	https://www150.statcan.gc.ca/n1/en/catalogue/92-500-X	November 19, 2024	2024
Utility Lines	https://geohub.lio.gov.on.ca/datasets/cd6832d6a7cd4f6c8b615732e5c6c965_11/explore?location=49.874140%2C-84.745000%2C3.97	October 23, 2024	2020
Railway Lines: Ontario Railway Network - Track	https://geohub.lio.gov.on.ca/maps/mnrf::ontario-railway-network-orwn/about	October 23, 2024	2017
Wetlands	https://geohub.lio.gov.on.ca/datasets/5216a770ef684d2fae8bcc13ee9c4357_15/explore	December 12, 2024	2024
Forest and Water Cover: Ontario Land Cover Compilation Version 2 (OLCC v2)	https://geohub.lio.gov.on.ca/documents/7aa998fdf100434da27a41f1c637382c/about	November 5, 2024	2016
Forest Fire: Fire Disturbance Area	https://geohub.lio.gov.on.ca/datasets/439675582ff2409ab759c793d85a6a3b_28/explore?location=48.577684%2C-84.732487%2C4.34	October 28, 2024	2023
Range Boundaries	https://geohub.lio.gov.on.ca/datasets/f1cf5d1b5ef346feafa386ba626d536b_8/explore?location=49.874140%2C-84.745000%2C3.97	October 16, 2024	2014
Forest Management Unit Boundaries	https://geohub.lio.gov.on.ca/datasets/c5f284168fe7443c9dc108552ce1b47f_20/explore?location=49.874140%2C-84.745000%2C3.97	October 16, 2024	2024

Table S3. List of model structures and hypotheses.

Model Name		Model	Hypothesis
M1		conifer cover (7)	Caribou select conifer-dominated areas for forage and cover
M2		wetland cover (7)	Wetlands are positively associated with caribou density due to reduced predator access
M3		deciduous cover (7)	Caribou avoid deciduous-dominated areas due to increased moose presence and predator risk
M4		distance to all roads	Roads increase predation risk and are negatively associated with caribou density
M5		harvest with 500m buffer (7)	Caribou avoid harvested areas due to indirect increase in predation risk
M6		forest fire (7)	Caribou avoid recently burned areas due to early successional conditions
M7			M7b has greater explanatory power than M7a
	M7b	conifer cover (7)	
	M7b	conifer cover (7) + wetland cover (7)	
M8			M8b has a stronger negative effect than M8a. M8c determines relative effect sizes of forest fire and harvest in a combined model.
	M8a	forest fire (7)	
	M8b	harvest with 500m buffer (7)	
	M8c	forest fire (7) + harvest with 500m buffer (7)	
M9			M9a has a stronger effect than M9b. M9c determines relative effect sizes of conifer and deciduous in a combined model.
	M9a	conifer cover (7)	
	M9b	deciduous cover (7)	
	M9c	conifer cover (7) + deciduous cover (7)	
M10			M10a has stronger effect than M10b. M10c determines relative effect size of roads an harvest in a combined model.
	M10a	distance to all roads	
	M10b	harvest with 500m buffer (7)	
	M10c	distance to all roads + harvest with 500m buffer (7)	
M11			M11b will improve model fit from M11a. Tests combined avoidance of deciduous cover due to competition and roads due to predation.
	M11a	distance to al roads	
	M11b	distance to all roads + deciduous cover (7)	
M12			M12b will improve model fit from M12a. Testing additional risk from harvest.
	M12a	deciduous cover (7) + distance to all roads	
	M12b	deciduous cover (7) + distance to all roads + harvest with 500m buffer (7)	
M13			M13c will improve model fit from M13a and M13b. Tests the combined trade off between preferred habitat and risks.
	M13a	conifer cover (7) + wetland cover (7)	
	M13b	deciduous cover (7) + distance to all roads	
	M13c	conifer cover (7) + wetland cover (7) + deciduous cover (7) + distance to all roads	
M14		conifer cover (7) + wetland cover (7) + deciduous cover (7) + distance to all roads + harvest with 500m buffer (7) + forest fire (7)	Each covariate has an important influence on caribou density and will explain variation in density better in a combined model than any subset alone.

Table S4. Univariate model selection results for land cover, linear features, harvest, and natural disturbance covariates.

Model	AICc	D CV	Beta	Beta CV	Category
D ~ conifer cover (7), g0 ~ 1, σ ~ 1	**2250.25**	**16.0%**	**3**	**7%**	**Land Cover**
D ~ conifer cover (1), g0 ~ 1, σ ~ 1	2303.58	16.0%	3.1	10%	Land Cover
D ~ deciduous cover (7), g0 ~ 1, σ ~ 1	**2511.91**	**16.0%**	**-1.3**	**16%**	**Land Cover**
D ~ deciduous cover (1), g0 ~ 1, σ ~ 1	2561.58	16.0%	-8.9	314%	Land Cover
D ~ wetland cover (7), g0 ~ 1, σ ~ 1	2593.08	16.1%	0.4	28%	Land Cover
D ~ wetland cover (1), g0 ~ 1, σ ~ 1	2594.26	16.0%	0.4	28%	Land Cover
D ~ distance to all roads, g0 ~ 1, σ ~ 1	**2519.14**	**16.1%**	**1.8**	**10%**	**Linear Features**
D ~ distance to primary roads, g0 ~ 1, σ ~ 1	2523.80	16.0%	1.8	10%	Linear Features
D ~ distance to linear features, g0 ~ 1, σ ~ 1	2529.50	16.1%	1.7	5%	Linear Features
D ~ harvest with 500m buffer (7), g0 ~ 1, σ ~ 1	**2525.68**	**16.0%**	**-0.9**	**14%**	**Harvest**
D ~ harvest (7), g0 ~ 1, σ ~ 1	2534.83	16.0%	-0.8	16%	Harvest
D ~ harvest with 500m buffer (1), g0 ~ 1, σ ~ 1	2547.98	16.0%	-8.6	351%	Harvest
D ~ harvest (1), g0 ~ 1, σ ~ 1	2563.65	16.0%	-8.3	310%	Harvest
D ~ forest fire (7), g0 ~ 1, σ ~ 1	**2569.65**	**16.0%**	**-8.4**	**308%**	**Natural Disturbance**
D ~ forest fire (1), g0 ~ 1, σ ~ 1	2580.83	16.0%	-4.5	128%	Natural Disturbance
D ~ natural disturbance (1), g0 ~ 1, σ ~ 1	2593.95	16.0%	-0.7	59%	Natural Disturbance
D ~ natural disturbance (7), g0 ~ 1, σ ~ 1	2595.31	16.0%	-0.6	46%	Natural Disturbance

D: density (individuals per 1000 km^2), g0: detection parameter, σ: space use parameter (m), (1): 1 km moving window, (7): 7 km moving window. Selected covariates from each category are bolded.

Table S5. Summary table of multivariate density model results and covariate β estimates.

Model	AICc	ΔAICc	D	CV	Conifer cover (7)		Deciduous Cover (7)		Wetland Cover (7)		Distance to All Roads		Harvest + 500m buffer (7)		Forest Fire (7)	
					β	**β SE**	**β**	**β SE**	**β**	**β SE**	**β**	**β SE**	**β**	**β SE**	**β**	**β SE**
D ~ conifer cover (7) + wetland cover (7) + deciduous cover (7) + distance to all roads + harvest with 500m buffer (7) + forest fire (7)	2215.51	0.00	45.0	16.0%	1.62	0.24	-0.84	0.30	0.53	0.19	1.49	0.34	0.01	0.13	-7.91	19.63
D ~ conifer cover (7) + wetland cover (7) + deciduous cover (7) + distance to all roads	2237.76	22.25	42.3	16.3%	2.63	0.46	-0.86	0.55	0.50	0.21	1.17	0.39				
D ~ conifer cover (7) + deciduous cover (7)	2241.19	25.68	40.2	16.0%	3.03	0.20	-1.76	1.22								
D ~ conifer cover (7) + wetland cover (7)	2249.31	33.80	40.7	16.0%	3.31	0.28			0.23	0.13						
D ~ distance to all roads + deciduous cover (7) + harvest with 500m buffer (7)	2400.91	185.40	46.3	16.1%			-2.60	0.65			2.50	0.29	0.30	0.12		
D ~ distance to all roads + deciduous cover (7)	2403.62	188.11	46.1	16.1%			-2.06	0.41			1.93	0.17				
D ~ forest fire (7) + harvest with 500m buffer (7)	2467.24	251.73	45.6	16.0%									-0.96	0.13	-6.21	6.86
D ~ distance to all roads + harvest with 500m buffer (7)	2515.46	299.95	46.1	16.0%							1.01	0.32	-0.43	0.17		

D: density (individuals per 1000 km^2), (7): 7 km moving window.

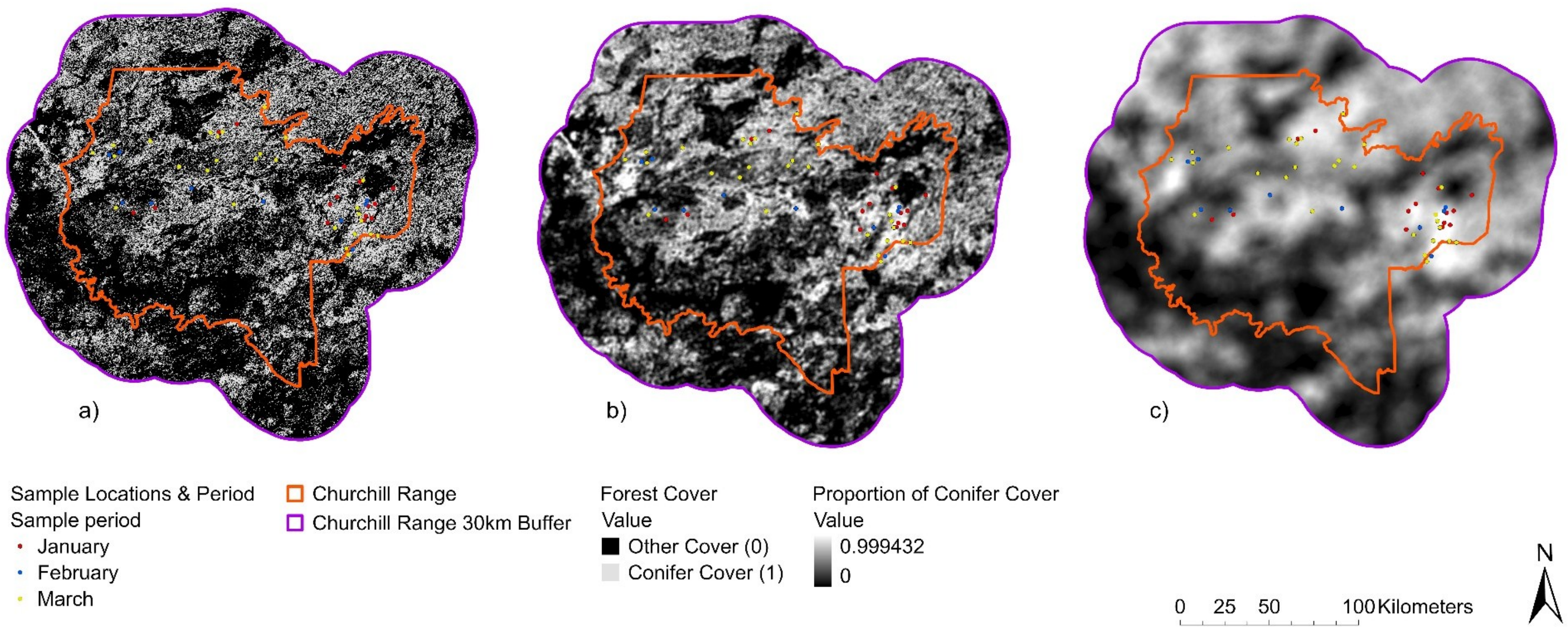

Figure S1. Conifer cover at different moving window radii with sample locations in the Churchill Range: a) raw conifer cover data, b) 1 km moving window radius, c) 7 km moving window radius.

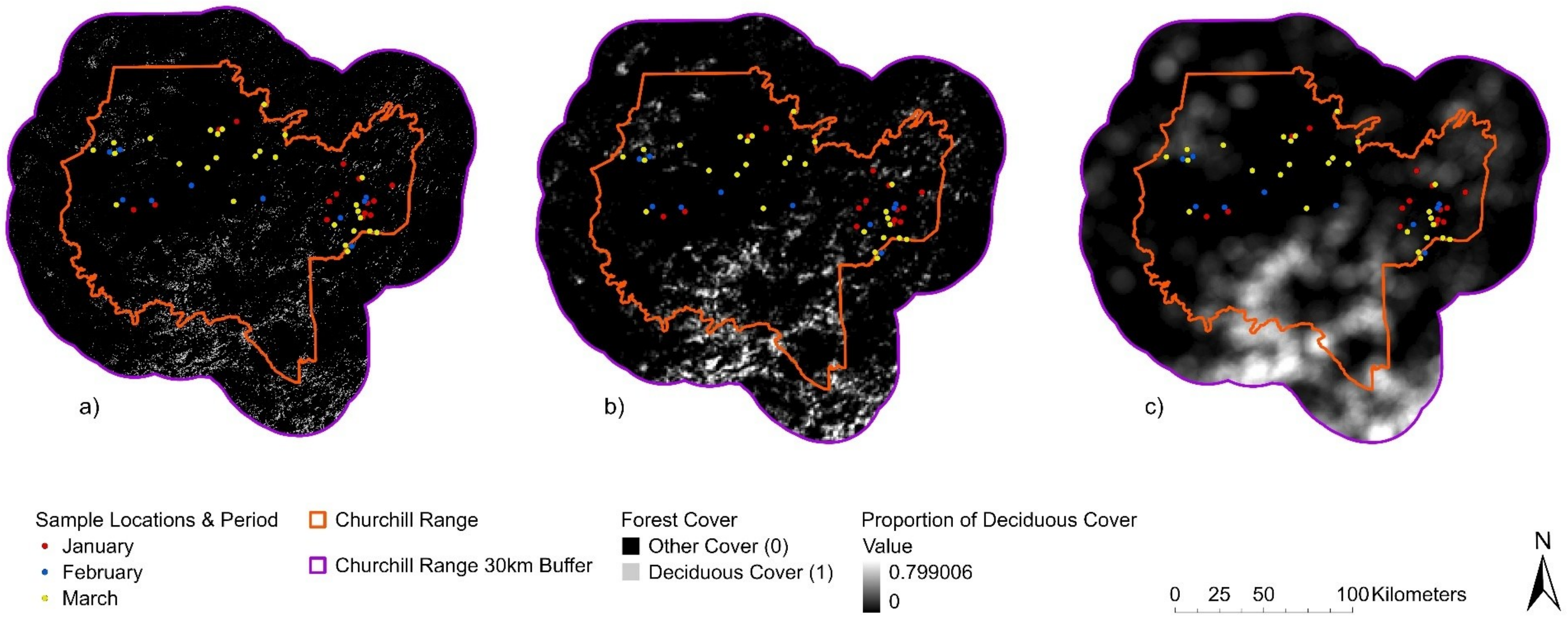

Figure S2. Deciduous cover at different moving window radii with sample locations in the Churchill Range: a) raw deciduous cover data, b) 1 km moving window radius, c) 7 km moving window radius.

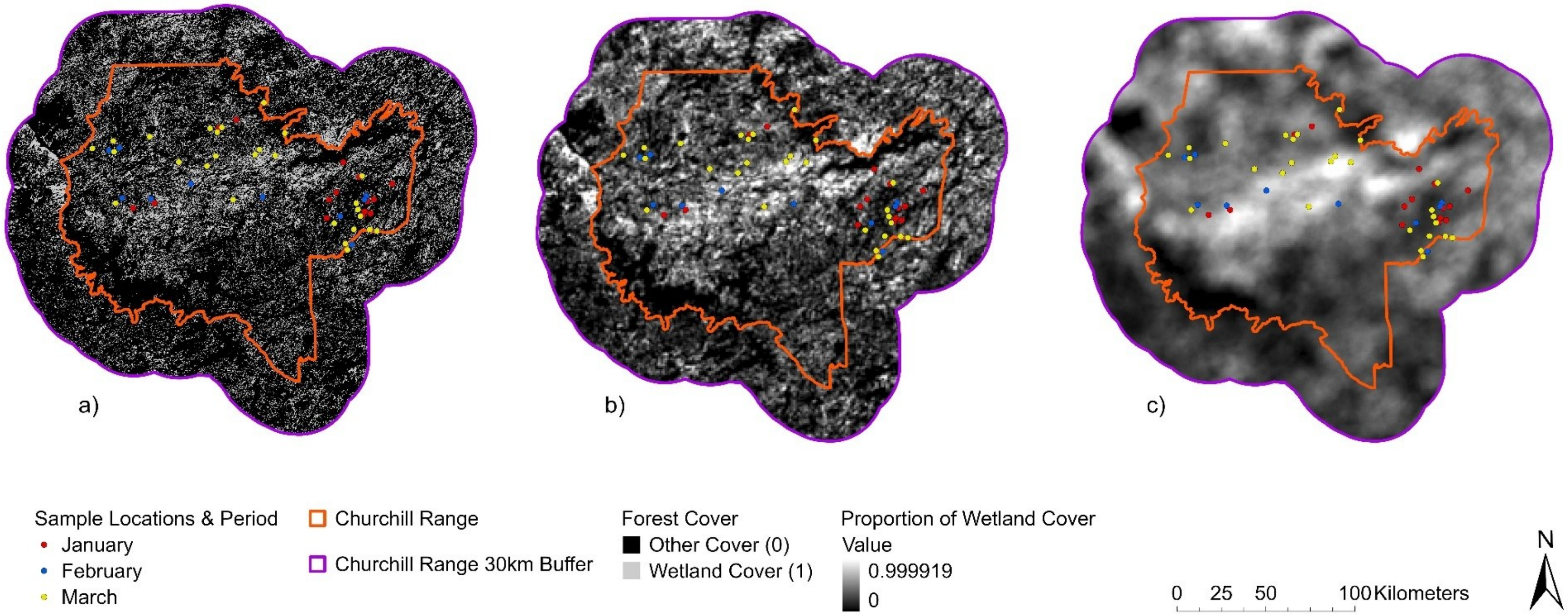

Figure S3. Wetland cover at different moving window radii with sample locations in the Churchill Range: a) raw wetland cover data, b) 1 km moving window radius, c) 7 km moving window radius.

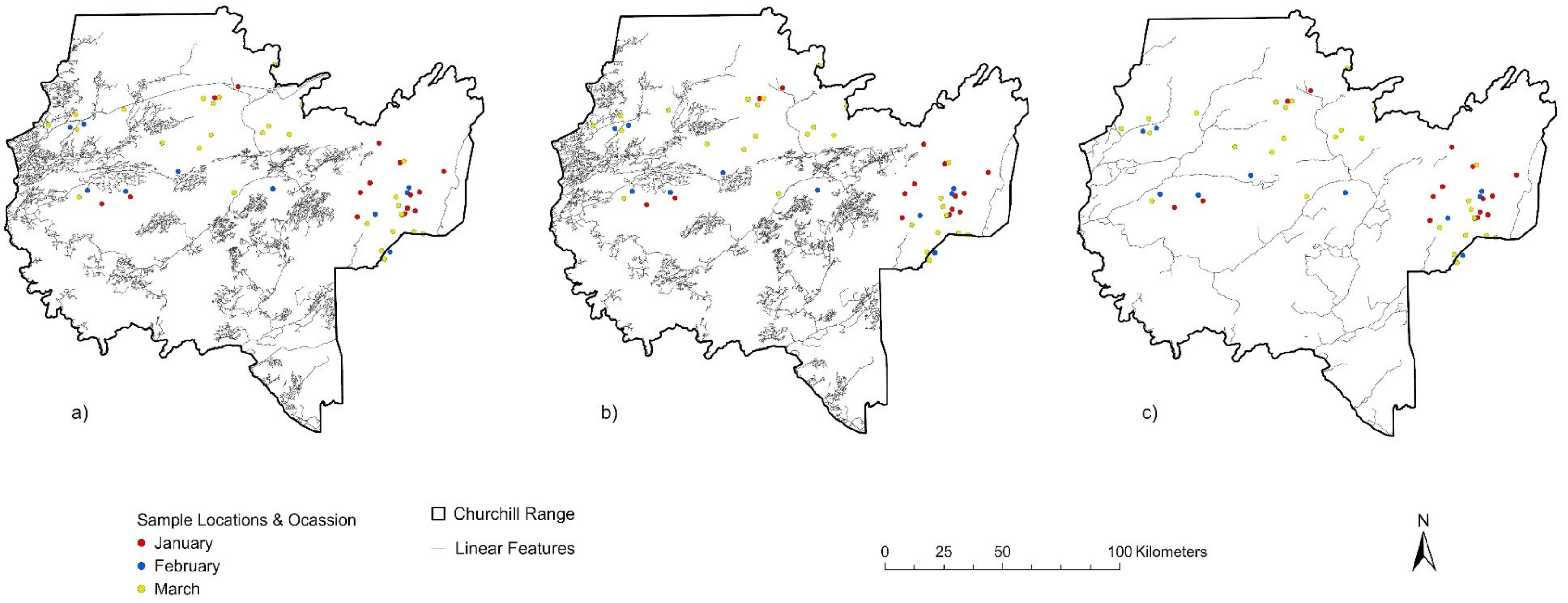

Figure S4. Different linear feature types with sample locations in the Churchill Range: a) all linear feature types, b) all roads only, c) primary roads only.

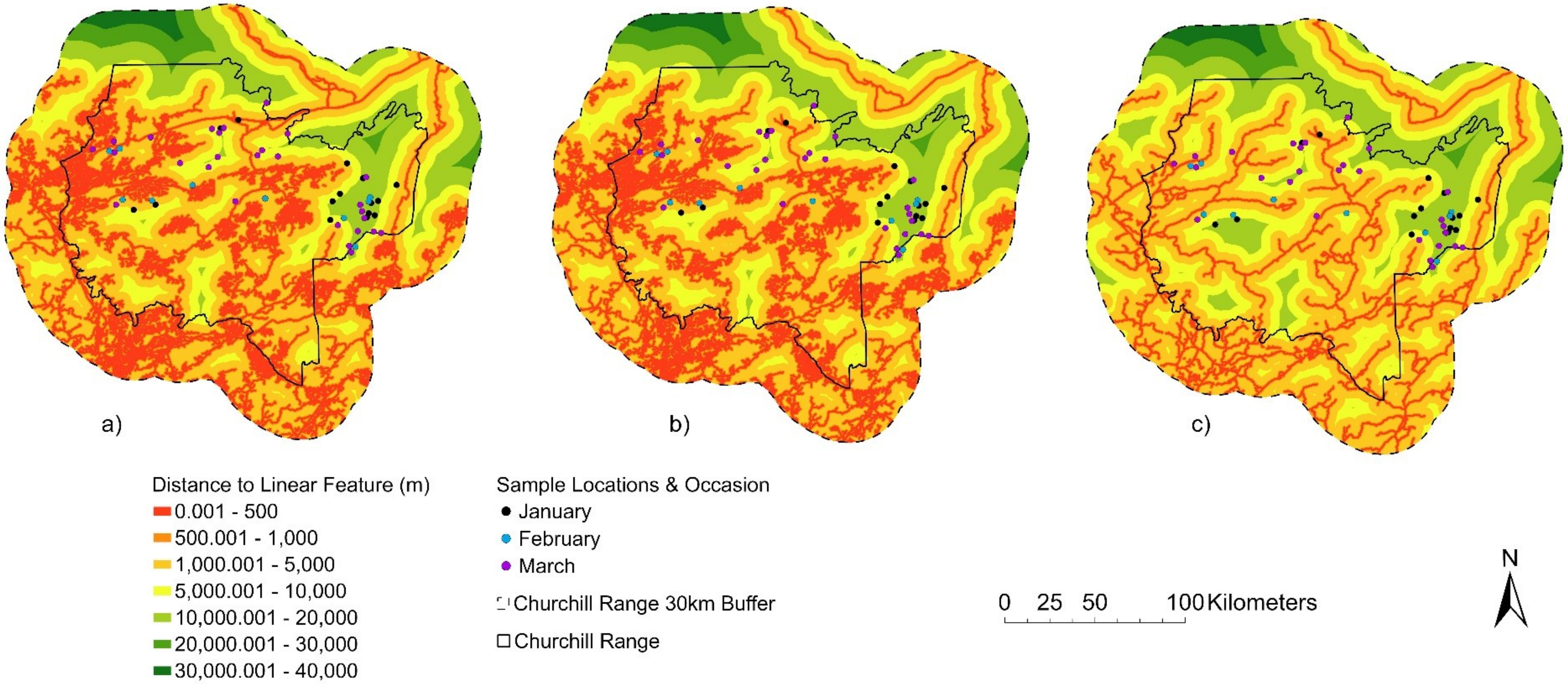

Figure S5. Euclidean distance to different linear feature types with sample locations in the Churchill Range: a) all linear feature types, b) all roads only, c) primary roads only.

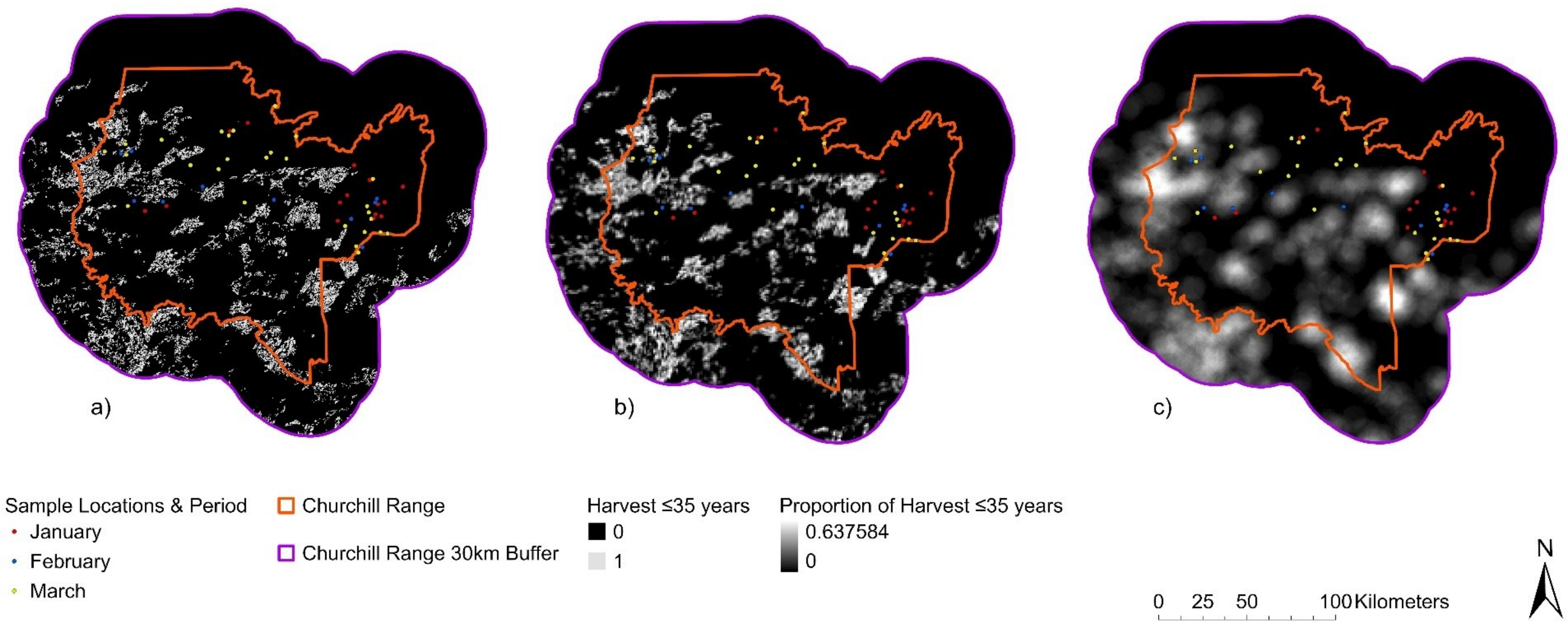

Figure S6. Harvest ≤35 years at different moving window radii with sample locations in the Churchill Range: a) raw harvest data, b) 1 km moving window radius, c) 7 km moving window radius.

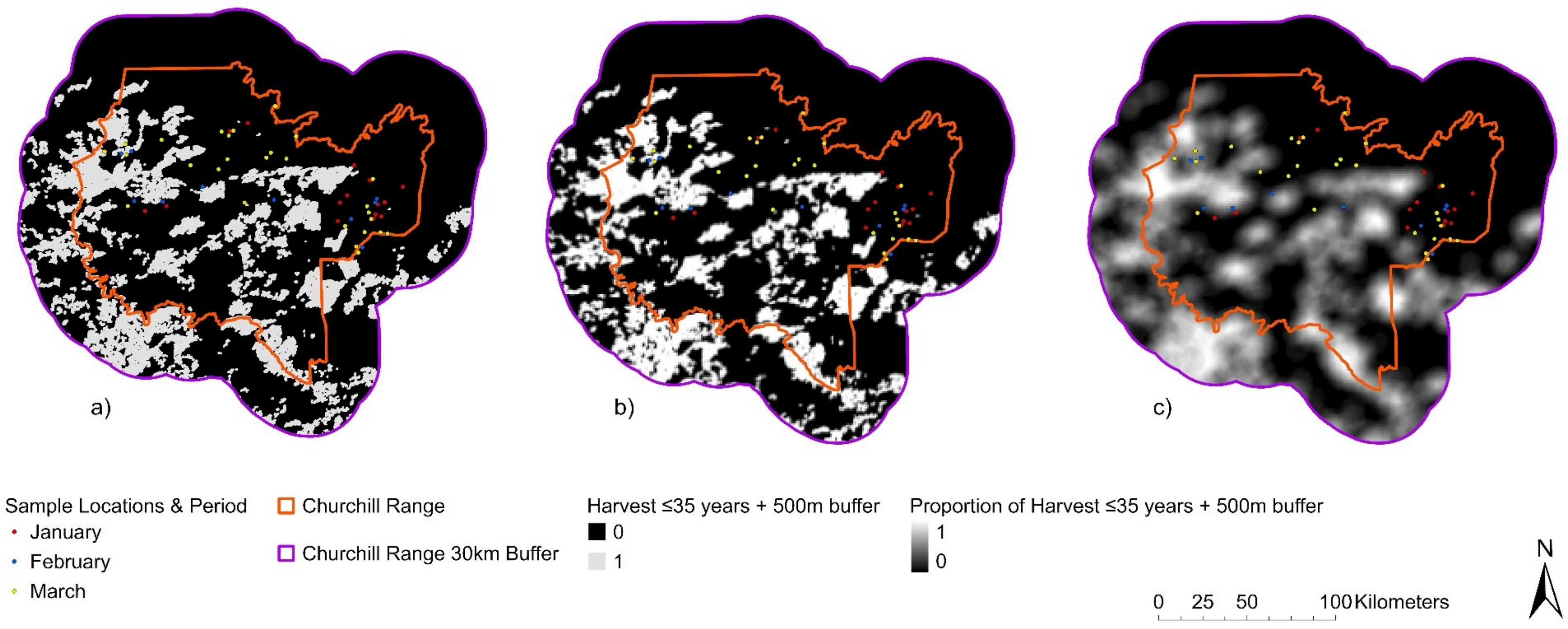

Figure S7. Harvest ≤35 years with a 500m buffer at different moving window radii with sample locations in the Churchill Range: a) raw harvest with 500m buffer data, b) 1 km moving window radius, c) 7 km moving window radius.

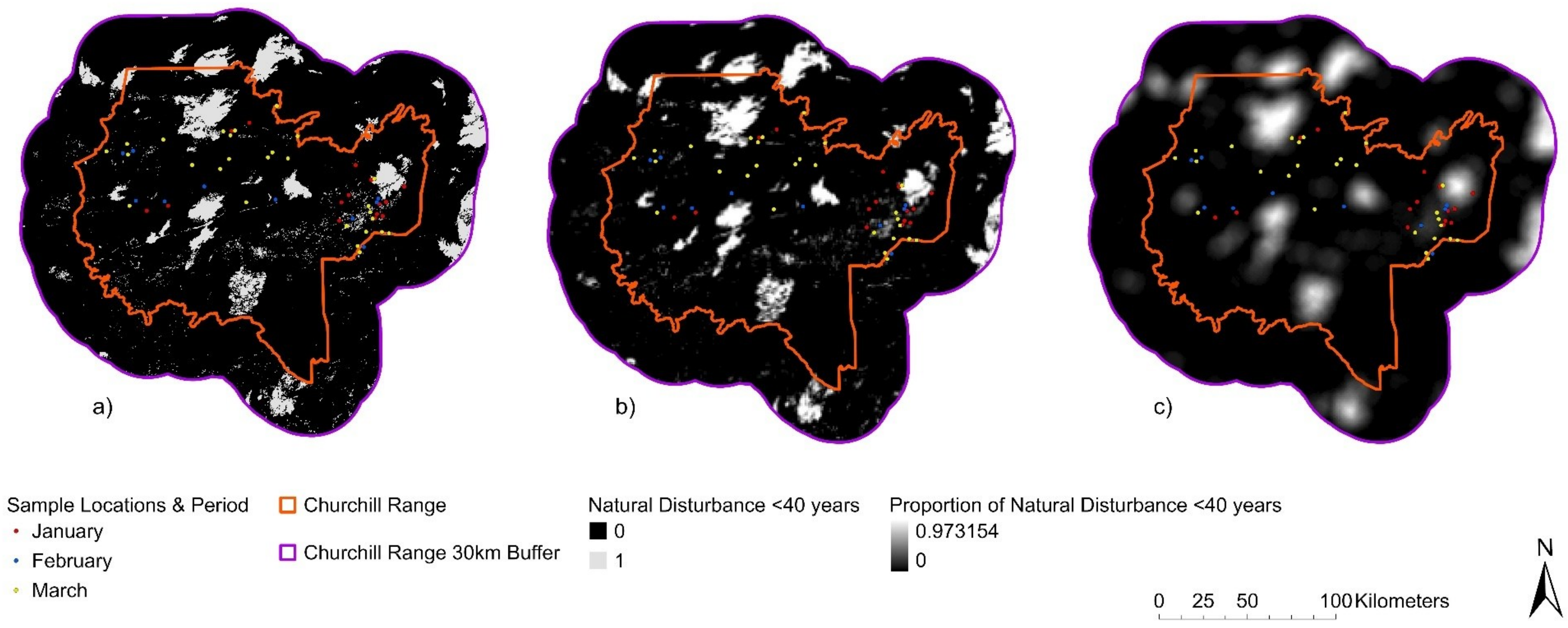

Figure S8. Natural disturbance <40 years at different moving window radii with sample locations in the Churchill Range: a) raw natural disturbance data, b) 1 km moving window radius, c) 7 km moving window radius.

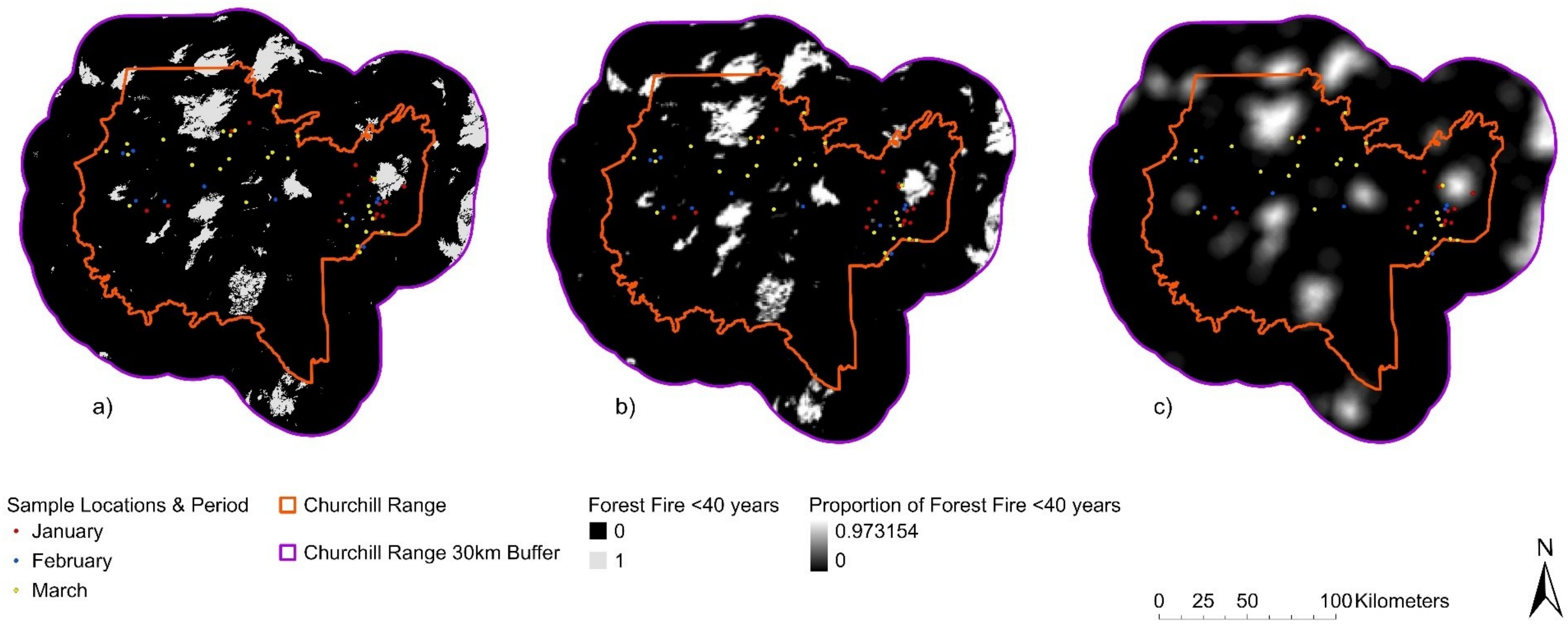

Figure S9. Forest fire <40 years at different moving window radii with sample locations in the Churchill Range: a) raw forest fire data, b) 1 km moving window radius, c) 7 km moving window radius.