Existing caribou habitat and demographic models are poorly suited for Ring of Fire impact assessment: A roadmap for improving the usefulness, transparency, and availability of models for conservation

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ABSTRACT

Environmental impact assessments often rely on best available information, which may include models that were not designed for purpose and are not accompanied by an assessment of limitations. We reproduced available models of boreal woodland caribou resource selection and demography and evaluated their suitability for projecting impacts of development in the Ring of Fire (RoF) on boreal caribou in the Missisa range (Ontario, Canada). The specificity of the resource selection model limited usefulness for predicting impacts, and high variability in model coefficients among ranges suggests responses vary with habitat availability. The aspatial demographic model projects decreasing survival and recruitment with increasing disturbance, but high variability among populations implies the importance of these impacts depends on the status of the Missisa population, which is not known. New models that are designed for forecasting the cumulative effects of development and climate change are required to inform RoF decisions. To demonstrate how open-source tools and reproducible workflows can improve the transparency and reusability of models we developed an R package for data preparation, resource selection, and demographic calculations. Open-source tools, reproducible workflows, and reuseable forecasting models can improve our collective ability to inform wildlife management decisions in a timely manner.

Keywords: *Rangifer tarandus caribou*; Open science; Decision-support; Resource selection function; Demographic model; environmental impact assessment, cumulative effects

INTRODUCTION

Decisions about natural resource development should be informed by sound environmental impact assessment (Fortin et al., 2020; Hebblewhite, 2017; Johnson et al., 2019; Johnson and St. Laurent, 2011). Increasingly, these assessments mandate quantitative appraisals of cumulative effects (e.g. ECCC, 2021). Models that quantitatively link wildlife species responses to project impacts can influence policy and natural resource development (Wilson et al., 2021). However, information is not always available or accessible to support decisions. Practitioners are often required to use tools and models that were not designed to support impact assessment and may be unaware of their limitations. There is a need to develop open source, and easily reproduced, wildlife management decision-support tools (Eacker et al., 2019; Nagy-Reis et al., 2020; Nowak et al., 2018; Stewart et al., 2020) that increase the speed, scope, and quality of science that informs impact assessment, particularly for Species at Risk (McIntire et al., 2022; Roche et al., 2020).

Boreal woodland caribou (*Rangifer tarandus caribou*; hereafter 'boreal caribou') inhabit large portions of Canada's boreal forest (ECCC, 2011; Festa-Bianchet et al., 2011), where anthropogenic disturbance from the natural resource sector threatens population persistence (Fortin et al., 2020; Fryxell et al., 2020; Hebblewhite, 2017; Johnson et al., 2020). Nationally, only 29% of population ranges are considered self-sustaining and boreal caribou are listed as threatened under the federal Species at Risk Act. In the province of Ontario, boreal caribou are also listed as threatened due to development from forestry and mineral exploration (ECCC, 2011; MNRF, 2014a). Commercial logging and associated road development have contributed to landscape disturbance and are predicted to cause continued population declines (Fryxell et al., 2020).

The Hudson Plains in Northern Ontario, Canada, is one of the largest intact wetlands and peatland carbon stores on the planet (Ibisch et al., 2016; Poley et al., In Press; Sothe et al., 2022; Tootchi et al., 2019). This region is rich in valuable minerals and is a target for resource extraction and economic development (Carlson and Chetkiewicz, 2013; Far North Science Advisory Panel (Ont.) et al., 2011; IAAC, 2021). The region is home to Indigenous people and wildlife, including threatened boreal caribou, and is largely inaccessible by road. There is a need to assess the potential environmental impacts of proposed mining of 'Ring of Fire' (RoF) mineral deposits and associated development in this globally significant ecosystem. Development decisions in the RoF will impact caribou and tools are required to quantify this impact (Far North Science Advisory Panel (Ont.) et al., 2011; IAAC, 2021; Fig. 1).

Information to support impact assessments comes in many forms, including models of wildlife-habitat relationships and demography (Beyer et al., 2010; Matthiopoulos et al., 2020). Wildlife-habitat relationships are commonly represented by resource selection functions (RSFs), which can produce maps to predict habitat suitability and infer the consequences of habitat loss (Boyce et al., 2002; Harju et al., 2011; Johnson et al., 2004; Matthiopoulos et al., 2019). However, RSFs represent descriptive associations between animal behaviour and habitat under current, scale-specific conditions, limiting their predictive capacity and transferability to novel conditions (Boyce, 2006; Decesare et al., 2012; Harju et al., 2011; Mysterud and Ims, 1998; Yates et al., 2018). Aspatial demographic models that provide insight into drivers of population growth, such as survival and recruitment, are also used for characterizing wildlife responses to landscape alteration (Johnson et al., 2020; Sorensen et al., 2008; Stewart et al., 2020). However, high uncertainty about demographic parameters, drivers of change, and vital rates can limit the utility of these models for impact assessment (Chaudhary and Oli, 2020; Sleep and Loehle, 2010). Regardless of their potential limitations, RSFs and demographic models often represent the best available information to be used to support conservation decision making..

In Ontario, Canada, existing RSF (Hornseth and Rempel, 2016) and demographic models (Johnson et al., 2020) are used to support decisions provincially (Rempel et al., 2021) and federally (ECCC, 2019). These models represent the best available information for the RoF, but their utility for projecting impacts has not been formally evaluated, nor are they easy to scrutinize or apply. We reproduced the available RSF and demographic models and evaluated their suitability to predict some of the anticipated impacts of proposed mining in the RoF. To demonstrate the utility of open source tools and reproducible workflows in this context, we used a R package framework (Wickham, 2015) to create an R package that reproduces the existing RSF and demographic models. Our package can be integrated into predictive frameworks to support resource use decisions or modified and advanced by other practitioners for other uses (Bodner et al., 2021; McIntire et al., 2022; Micheletti et al., 2021; Miller and Frid, 2022). We identify and discuss challenges with using existing models for decision-support and opportunities for improving the usefulness, transparency, and availability of environmental impact assessment tools for SAR.

Methods

Study Area

Our study area included caribou ranges as delineated by Ontario's provincial government that contain and surround the RoF mining claims: Missisa, James Bay, Pagwachuan, and Nipigon, with most mineral claims being situated in the Missisa range (Far North Science Advisory Panel (Ont.) et al., 2011; IAAC, 2021; Fig. 1). The Missisa and James Bay ranges occur in Ontario's Far North (Far North Science Advisory Panel (Ont.) et al., 2011), where anthropogenic disturbance has thus far been limited to communities and exploration camps, winter roads, and the now inactive Victor Diamond Mine. The Nipigon and Pagwachuan ranges are almost entirely within the more productive southern portion of the Boreal Shield ecozone (Crins et al., 2009) where industrial forestry is common (the Managed Forest Zone or MFZ; Fig. 1). The Boreal Shield consists of mixed and coniferous forests dominated by black spruce (*Picea mariana*), and containing jack pine (*Pinus banksiana*), balsam fir (*Abies balsamea*), white spruce (*Picea glauca*), trembling aspen (*Populus tremuloides*), and balsam poplar (*Populus balsamifera*) (Crins et al., 2009). The Pagwachuan range also includes a portion of the Boreal Plains and the Great Clay Belt. The Missisa range includes the transition between the Boreal Shield and Hudson Plains, which is mainly composed of peatland complexes with poor drainage, with trees including stunted black spruce and tamarack (*Larix laricina*) (Crins et al., 2009). The James Bay range is almost entirely within the Hudson Plains. We compare the Missisa range to adjacent ranges to investigate the transferability of range-specific RSFs to different landscape conditions.

Ring of Fire Development Scenarios

We considered three simple development scenarios (Fig 2); a 'base' scenario with no change in development footprint, a 'roads-only' scenario that includes proposed RoF access roads (MNDMNRF, 2022), and a 'roads-and-mines' scenario which includes the proposed access roads and mining claims associated with the RoF (Table S1.2) . Our intention is not to speculate about the scale of future developments, but to assess the suitability of existing models for projecting impacts of development on boreal caribou.

RSF Model Reproduction

We used tables of published boreal caribou range- and season-specific RSF coefficients for Ontario (Hornseth and Rempel, 2016; Rempel et al., 2021), hereafter referred to as the original RSFs. The authors referred to these models as resource selection probability functions (RSPFs), but their approach differs from what is commonly understood as an RSPF (Johnson et al., 2006) so we use the term RSF. The data to produce the original RSFs were collected from GPS collared caribou across a variety of companion studies within the ranges of interest occurring between 2009 and 2013 (Hornseth and Rempel, 2016; MNRF, 2014b, 2014b; Rempel and Hornseth, 2018). We obtained the published coefficients for range-specific seasonal RSFs of each reported top model (Table S1.1) for the Nipigon, Pagwachuan, Missisa, and James Bay range, but associated error or uncertainty information is not available. The predictors used in the original RSFs were developed from land cover, forest fire and harvest disturbance history, linear features, and esker datasets but the original data was not available (Table S1.2, S1.3). To reproduce the original RSFs, we acquired data sets for the predictor variables that were publicly available and, where possible, used information on the timing of the event or the feature's construction to recreate the data that was available when the original model was produced (Table S1.2). We used current versions of these same datasets to project the RSFs on the landscape.

RSF Model Validation

To assess our ability to reproduce the original model, we acquired the original projected surfaces for comparison from the authors (R. Rempel pers. comm, 2021). We validated our models visually and quantitatively. The original model used a nested hexagon grid to generate predictions. We opted to approximate this approach with distance-weighted moving windows on a rectangular grid, which is easier to implement using standard raster-processing tools. To compare these approaches, we extracted the values from our rectangular grid to the hexagonal grid using the weighted mean. We mapped the difference in predictions and explanatory variables between the models and used scatterplots to compare the value of the model response for each grid cell produced by the two models. A perfect reproduction would produce a Pearson correlation coefficient of 1, and any deviation from the original prediction would reduce this coefficient. We expected some differences in the predictions as a result of the type of grid and predictor data sets used.

RSF Model Projection Under Disturbance Scenarios

The RSF includes road density as a predictor and would require additional information on roads within mining areas that we do not have; for the RSF we only compare the base and road-only scenarios. To understand how the 'roads only' scenario would affect the caribou RSF, we projected the original RSFs across updated landscape conditions as represented by i) temporal changes in forest structure between 2010 and 2020 (e.g., fires; Table S1.2) and ii) the new proposed roads only scenario (Fig. 2). We also assessed the potential for borrowing information from other ranges by using the coefficients from the top original RSFs from the adjacent James Bay, Nipigon, and Pagwachuan ranges transferred to Missisa. We visualized the spatial transferability of the models and examined their sensitivity to changing habitat availability using scatterplots to compare the value of the model response for each grid cell produced by the Missisa model and each respective adjacent range.

Demographic Model

Canada's national demographic boreal caribou models were developed from adult female survival (*S*), calf recruitment (*R*), and landscape data across 58 boreal caribou study areas, including 13 study areas in Ontario (Johnson et al., 2020; Table S1). We used these models (Johnson et al., 2020) to predict changes in *S* and *R*; in the roads-only and roads-and-mines development scenarios. The demographic model is aspatial and all types of anthropogenic disturbances are combined into a single measure of disturbed area, so the 'roads-and-mines' footprint is sufficient and there is no need to specify the location of roads within mining claim areas. We calculated the relevant predictor variables for the Missisa range (e.g., % anthropogenic disturbance buffered by 500 m; % wildfire within the last 40 years; Table 1), and calculated expected *R* and *S* as a function of disturbance according to the beta regression models with highest support (M4 and M1 respectively from Johnson et al., 2020):

$$R \sim Beta(\mu^R, \phi^R); log(\mu^R) = \beta_0^R + \beta_a^R\ anthro\ + \beta_f^R\ fireExclAnthro, \quad \text{(eq 1a)}$$

$$S \sim (46 \times Beta(\mu^S, \phi^S) - 0.5)/45; log(\mu^S) = \beta_0^S + \beta_a^S\ anthro, \quad \text{(eq 1b)}$$

Where ϕ^R ~ *Normal(19.862,2.229)* and ϕ^S ~ *Normal(63.733,8.311)* are precisions of the Beta distributed errors (Ferrari and Cribari-Neto, 2004), and survival rates are back transformed as in Johnson et al. 2020. Table 3 of Johnson et al. (2020) provides the expected values and 95% confidence intervals of all regression coefficients (β_0^R, β_α^R, β_f^R, β_0^S, β_α^S), which are assumed to be Gaussian distributed. To evaluate our recruitment and survival models, we sampled expected demographic rates across a range of anthropogenic and fire disturbance (0-100%) to reproduce expected values and 95% predictive intervals from Fig. 3 and Fig. 5 of Johnson et al. (2020).

In areas of low anthropogenic disturbance, substantial among-population variability in recruitment existed (Fig. 3 in Johnson et al., 2020). To model this variation, regression model parameter values for each sample population were selected at the beginning of simulations and each population was assigned to quantiles of the error distributions for survival and recruitment. Sample populations remained in their quantiles as the landscape changed, allowing us to distinguish the effects of changing disturbance from variation in initial population status. To show these effects, we projected population growth for 35 sample populations across a wide range of anthropogenic disturbance levels (0-90%) using a two-stage demographic model with density dependence and interannual variability that differed from the model used by Johnson et al. (2020) in the method used to ensure whole numbers of animals (Supplementary Material Part 2). We characterized the distribution of outcomes for our three disturbance scenarios (Fig. 2, Table 1) more thoroughly by projecting the dynamics of 500 sample populations. Population growth rate each year is given by $\lambda_t=N_{t+1}/N_t$ when $N_t>0$ and $\lambda_t=0$ when $N_t=0$. For each sample population, population growth was projected for 20 years from an assumed initial population size of 373 females, which we derived from the minimum animal count of 745 caribou between 2009 and 2011 and assumed 50% were female (MNRF, 2014b), and we report the average growth rate λ over that time. To verify our reproduction of the demographic model used by Johnson et al. (2020), we compared our outputs to those from model code supplied by the authors.

R Package Framework: caribouMetrics

We incorporated RSF and demographic model components into an R package with documentation and vignettes explaining their use, and used GitHub (github, 2020) to promote version control and transparency of the development process: https://github.com/LandSciTech/caribouMetrics. The standardized nature of packages allows them to easily integrate into the larger R ecosystem, allowing them to be extended and adapted towards tasks beyond their original design (e.g. prediction or forecasting; McIntire et al., 2022).

caribouMetrics also contains functions that automate the geospatial data preparation process to facilitate application to new landscapes. The original RSFs were developed using a closed scripting language called Landscape Scripting Language (LSL; Kushneriuk and Rempel, 2011), which we re-implemented in R. For the aspatial demographic model, we began by borrowing demographic rate sampling code from a SpaDES (Chubaty and McIntire, 2022) module (*caribouPopGrowth*; https://github.com/tati-micheletti/caribouPopGrowthModel; Stewart et al., in review), and modified the method to include precision and quantiles. We extracted the two-stage demographic model with density dependence and interannual variability from unpublished code (Johnson et al., 2020), and modified the code to improve transparency and ensure whole numbers of animals (see Supplementary Material Part 2).

Results

RSF Model Reproduction and Validation

We were able to produce a reasonably accurate representation of the original RSFs for the Missisa range, as evidenced by high Pearson's r values across all seasons ($r > 0.935$; Fig. S1.1). Maps highlighting the differences between predictions and some variation in the input data layers are provided in Supplementary Material (Fig. S1.2, S1.3). In the Missisa range, the model predicted the highest relative use probabilities in the northwest of the study area during winter (Fig. 3), consistent with the original RSFs. During the summer and spring, the eastern portion of the range was used more than the northwest (Fig. 3).

RSF Model Projection

We observed a lower relative probability of use in areas associated with proposed roads (Fig. 4). Lower relative probability of use along the road corridors was strongest in the spring and summer, consistent with seasonal changes in the response to roads described in the original RSFs (Fig. 4). There was high variability in the estimated response to roads among ranges (Fig. 4). The James Bay range prediction appeared the most similar to the Missisa range prediction (Fig. 4); however it varied by season (Fig S1.4). The Nipigon and Pagwachuan range projections, which visually differed substantially from the Missisa projection, did not show a strong response to proposed roads (Fig. 4; Fig S1.4).

Demographic model

Comparison with Johnson et al. (2020) indicates a good reproduction of the regression models for survival and recruitment (Fig. 5). Anthropogenic disturbance remains relatively low in all our development scenarios (Table 1), and the corresponding range of variability in demographic rates among sample populations is high (Fig. 5). The model predicts increasing anthropogenic disturbance will decrease both survival and recruitment (Fig. 5), but the importance of that decrease for self-sustainability of the population is highly uncertain and depends on initial population status (Fig. 5). A 2014 assessment informed by data from 2008-2012 indicated lower than expected recruitment, survival, and population growth rate in the Missisa range (diamonds in Fig. 5; MNRF, 2014a); however, we lack recent information on the survival, recruitment and status of this population to project the impacts of disturbance.

Discussion

We examined the applicability of two existing caribou models for projecting impacts of development in the RoF on boreal caribou in the Missisa range. We highlight limitations of existing tools and point out possible solutions. We found that the original RSFs are poorly suited for projecting the impacts of development in the RoF because they are specific to current conditions. In contrast to the RSFs, the existing aspatial demographic model is too general to project the impacts of development on this particular population. Ideally, forecasting models used to inform environmental decisions should be designed for the purpose, identify and account for uncertainty, be updateable with new information, and be transparent and reproducible (Bodner et al., 2021). However, in practice, environmental impact assessments often rely on best available information, which may include tools and models that were not designed for the purpose. We highlight limitations of existing tools and point out possible solutions.

The RSF models for each range were fit independently using data from that range. This is a reasonable approach when the objective is to characterize current habitat use, but not for projecting responses to changing landscape conditions. There has been very little anthropogenic disturbance in the Missisa range, so projecting impacts of disturbance on boreal caribou requires use of information from more disturbed areas. However, as formulated, the RSFs do not allow borrowing of information. There was high variability in the effect of linear features among range-specific RSFs, suggesting either that the behavioral response of caribou to linear features may vary with the amount or type of disturbance in a range (i.e. functional response; Mysterud and Ims, 1998), or that effects of linear features were confounded with other correlated predictors. Hierarchical regional models that include functional responses to disturbance (Matthiopoulos et al., 2011; Muff et al., 2020; Olson et al., 2021; Teitelbaum et al., 2021) could help distinguish between these possibilities, and integrated step-selection analysis approaches could yield models that are better suited for projection (Avgar et al., 2016; Prokopenko et al., 2017). The original RSFs used road density as a proxy for other anthropogenic disturbances (e.g., harvest) and did not distinguish among road types (Hornseth and Rempel, 2016). Evidence from other locations and contexts suggests that road type and traffic volume is important for wildlife, and metrics that include this information may be more informative (D'Amico et al., 2016; Jaeger et al., 2005; Leblond et al., 2013; Loosen et al., 2021). Although projecting impacts in a unique and previously undisturbed region is inherently challenging, we are optimistic that more informative models can be developed. In the meantime, these limitations should be considered when using or interpreting the original RSFs.

Among caribou ranges where anthropogenic disturbance was low, we saw that the demographic model predicted high variation in recruitment and survival among populations (Fig. 3 and Fig. 5 of Johnson et al., 2020), leading to high variability in projections of the impact of increasing disturbance on population growth rate (Fig. 5). This variability highlights the need for local information about the current status of caribou populations in the RoF and ongoing monitoring of development impacts in the area. An assessment of the status of the Missisa population based on 2008 – 2012 data (MNRF, 2014b) suggested survival, reproduction, and growth rate in this range was lower than predicted by the aspatial national demographic model of Johnson et al. 2020 (see black diamond in Fig. 5). More frequent and consistent data to estimate survival, reproduction, or population size are required to understand population trends and status. The national demographic model (Johnson et al., 2020) predicts that increasing anthropogenic disturbance will decrease both survival and recruitment, but the importance of that decrease for population viability depends on population status. Even small changes in adult female survival can affect the sustainability of a population in ranges with low adult survival (Johnson et al., 2020). Without information on the current status of the population, it is difficult to predict impacts of disturbance associated with RoF development on the viability of this particular population with any degree of confidence.

Our method of modelling variation in survival and recruitment among populations (Fig. 5) adequately reproduced the observed variation among populations (Fig. 3 and Fig. 5 in Johnson et al., 2020), but differed from the range-specific scenario approach taken by Johnson et al. (2020). We also opted to use a different method for ensuring whole numbers of animals, which lead to different estimates of population growth rate for small populations (see Supplementary Material Part 2). We assumed that demographic rates (i.e., recruitment and survival) vary with disturbance according to the best supported national models (Table 2 of Johnson et al., 2020) but note that other competing models in that candidate set were nearly as well supported by the data. This simple population model also assumed no variation in recruitment or survival with age or other parameters (Supplementary Material Part 2). Most female caribou reproduced at 2-3 years of age rather than at 1 year, as assumed in the model. This likely results in overestimating demographic parameter values under changing conditions. A more thorough investigation of the sensitivity of results, parameters, and representation of uncertainty and stochasticity in aspatial caribou demographic models is beyond the scope of this paper but seems warranted.

We lack region-specific information on disturbance history, vegetation, and vegetation recovery trajectories post-disturbance that are required for assessing impacts of climate change and anthropogenic disturbance in the RoF (McLaughlin and Webster, 2014; McLaughlin and Packalen, 2021). There may also be variability in caribou responses to fire (Konkolics et al., 2021; Palm et al., 2022) and interactions with predators (Bergerud, 1974) as has been found in other ranges. Disturbance-mediated apparent competition is generally the accepted mechanism for boreal caribou declines across Canada (Festa-Bianchet et al., 2011; Serrouya et al., 2016; Wittmer et al., 2013), but recent studies have noted that apparent competition may be less important in environments where above ground productivity is low because of the absence or lower abundance of other ungulate competitors (Superbie et al., 2022). The James Bay Lowlands, which make up a considerable portion of the Missisa range (Fig. 1), are a unique low productivity peatland ecosystem that currently experiences little fire (Abraham and Keddy, 2005). Climate change is predicted to alter fire regimes, but even if fire risk increases, this area will likely remain relatively wet and resistant to fire (Balshi et al., 2009; Wang et al., 2020). Given the uniqueness of this area, area-specific information is required; we demonstrate that data and models from other regions is not sufficient for understanding the effects of disturbance on wildlife-habitat relationships and boreal caribou demography in the RoF (MNRF, 2014a). Even where relevant data exists, political and social barriers to data access make it difficult or impossible to do synthetic work that would yield better insight into consistencies and differences among regions (Rutz et al., 2020; Tucker et al., 2018).

There are various efforts underway to collect data needed to inform resource development and wildlife management decisions in the RoF, yielding opportunities for collaborative development of region-specific open source forecasting models. We hope that our investigation of the limitations of existing models, and demonstration of the usefulness of implementing models an R package such as *caribouMetrics*, can help inform the development of new models and tools. Hierarchical regional models that include functional responses to disturbance (Matthiopoulos et al., 2011; Muff et al., 2020; Olson et al., 2021; Teitelbaum et al., 2021), integrated step-selection analysis (Avgar et al., 2016; Prokopenko et al., 2017), and hierarchical integrated population models (Moeller et al., 2021; Nowak et al., 2018) would yield more reliable projections of impacts in this previously undisturbed region. Open, transparent, reproducible workflows should be designed to enable ongoing incorporation of new data into models (Dietze, 2017; McIntire et al., 2022; Micheletti et al., 2021). Models should also be designed for flexibility and nimbleness, allowing decision makers to quickly adapt to specific contexts without duplicating efforts (Bodner et al., 2021; McIntire et al., 2022; Travers et al., 2019); an R package framework provides a practical option for achieving these goals.

We reiterate a call for transparent, quantitative decision-support tools to assess industrial development impacts on wildlife, and encourage developers of wildlife response models and collectors of relevant wildlife data to work together toward this goal (Davidson et al., 2021; Russell et al., 2021). We also recognize that the development of usable decision-support tools requires time and skills that are not possessed by everyone interested in doing open science. One solution is to work in multi-disciplinary teams (Bodner et al., 2021). Another is to recognize that steps toward transparency and reproducibility are valuable, even if the result is not always an easily usable tool. In this project, we were able to reproduce existing models because the developers of those models were willing to share their code. Code does not have to be flawless to enable others to build on previous work, and there are many ways that researchers can shift to more open and reproducible workflows that reduce the chance of errors, increase efficiency, improve reproducibility, and increase our ability to generalize across studies (Alston and Rick, 2021; Lewis et al., 2018). We are hopeful that improving the transparency, reproducibility and decision-relevance of wildlife response models will improve our collective ability to inform decisions for SAR and improve conservation outcomes.

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Data availability statement: The caribouMetrics R package is available at https://github.com/LandSciTech/caribouMetrics, and analysis code is available at https://github.com/LandSciTech/MissisaBooPaper. Data required for analysis is available in this OSF repository: https://osf.io/r9mkp/?view_only=fb71321265d14dbeb3d932e4de66be0c

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Table 1. Summary of changes in buffered anthropogenic disturbance and fire excluding anthropogenic disturbance (as defined in Johnson et al. 2020) in the Base Scenario without additional development, Roads Only scenario, and a Roads-and-Mines scenario that included the proposed roads and mining claims within the Ring of Fire area.

Scenario	Anthropogenic (%)	Fire (%)	Total Disturbance (%)	Fire Excluding Anthropogenic (%)
Base	0.41	4.27	4.58	4.17
Roads Only	1.11	4.27	5.22	4.12
Roads-and-Mines	6.95	4.27	11.02	4.71

Figure 1. The Missisa boreal caribou range (black outline), which includes the proposed Ring of Fire (RoF) mining claims (dark red), is the focus of this study. The blue dashed line distinguishes the Managed Forest Zone (MFZ) from the Far North, and shading distinguishes the Boreal Shield ecozone (pale red) from the Hudson Plains (blue). Inlay map (top-right) indicates the location of the Ontario boreal caribou ranges (grey) relative to Canada.

Figure 2. Maps of the Missisa range with the extent of linear features included in (a) the original model developed by Hornseth and Rempel (Base), (b) the roads only scenario used to project the RSF model, and (c) the roads-and-mines scenario used for demographic modelling. Existing roads are represented as green lines, proposed roads as orange lines, and mines are coloured in purple.

Figure 3. Reproduction of the seasonal RSF for the Missisa range from Hornseth and Rempel (2016) using caribouMetrics and the published coefficients to reproduce the relative probability of use (0-1) by boreal caribou during spring, summer, fall, and winter. The predictor variables used are approximations of those used by Hornseth and Rempel (2016) based on currently available data. Scale ranges from dark blue to yellow with yellow representing a higher relative probability of use.

Figure 4. Seasonal RSF predictions from *caribouMetrics* in the Missisa range under the roads only scenario using the coefficients from Hornseth and Rempel (2016) from the Missisa, James Bay, Nipigon, and Pagwachuan range to estimate the relative probability of use (0-1) by boreal caribou during spring, summer, fall, and winter. Scale ranges from dark blue to yellow with yellow representing a higher relative probability of use.

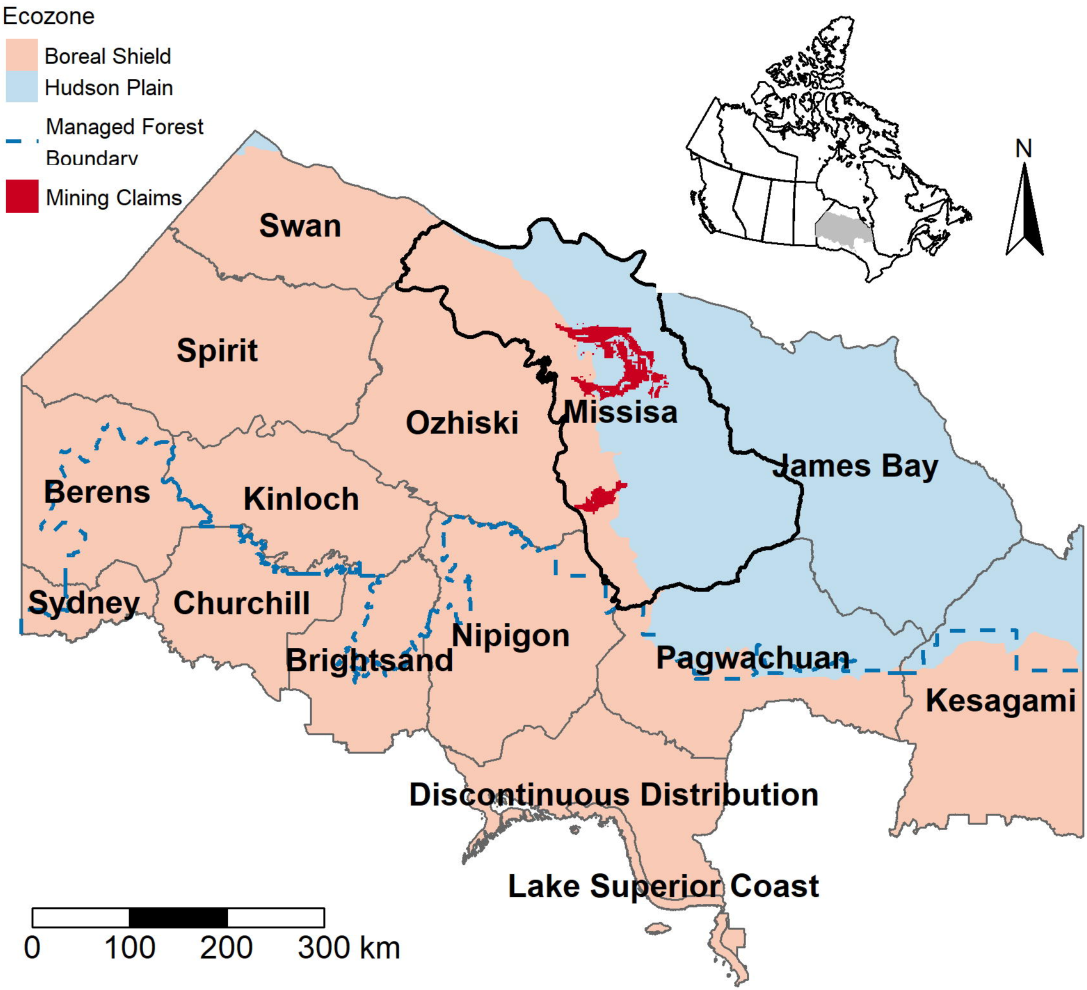
Figure 5. Demographic rate simulations derived from regression models in Johnson et al. (2020) for (a) Adult female survival (S), (b) Recruitment (calves per 100 cows), and (c) Average

population rate (λ). Overlap of grey and black dotted lines in (a) and (b) indicates a good match between expected values from our model (grey) and Johnson et al. 2020 (black). Coloured lines show effects of changing anthropogenic disturbance on demographic rates in 35 sample populations, assuming sample populations are randomly distributed among quantiles of the beta distribution, and each population remains in the same quantile of the beta distribution as disturbance changes. Alignment of these sample trajectories with 95% predictive intervals from Johnson et al. 2020 (pale grey bands in panels a and b) indicates that we have adequately reproduced variability in that model. Bars show the 2.5th and 97.5th percentiles for 500 sample populations under the three disturbance scenarios for the Missisa range (from left to right, Base, Road Only, and Roads and Mines). The black diamond indicates the demographic rates for the Missisa range according to a 2014 assessment (MNRF, 2014a). Populations with an average population trend of less than 0.99 are considered not self-sustaining.

Supplementary Material

Supplementary material is available in this OSF repository:

https://osf.io/r9mkp/?view_only=fb71321265d14dbeb3d932e4de66be0c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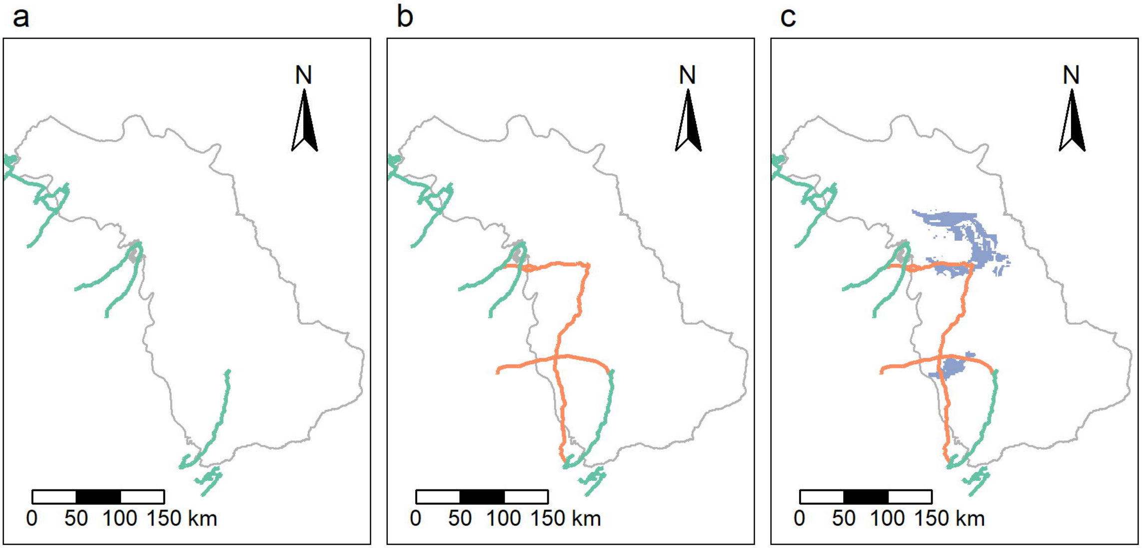
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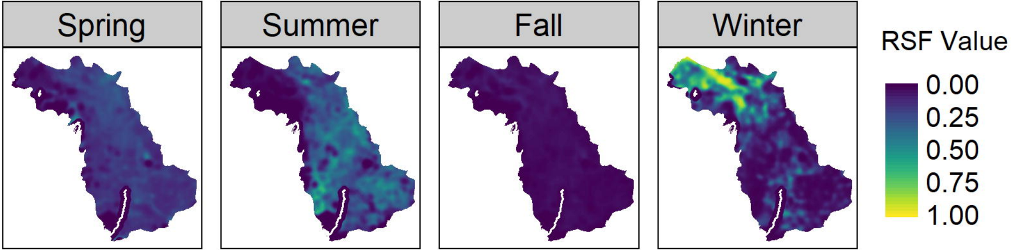

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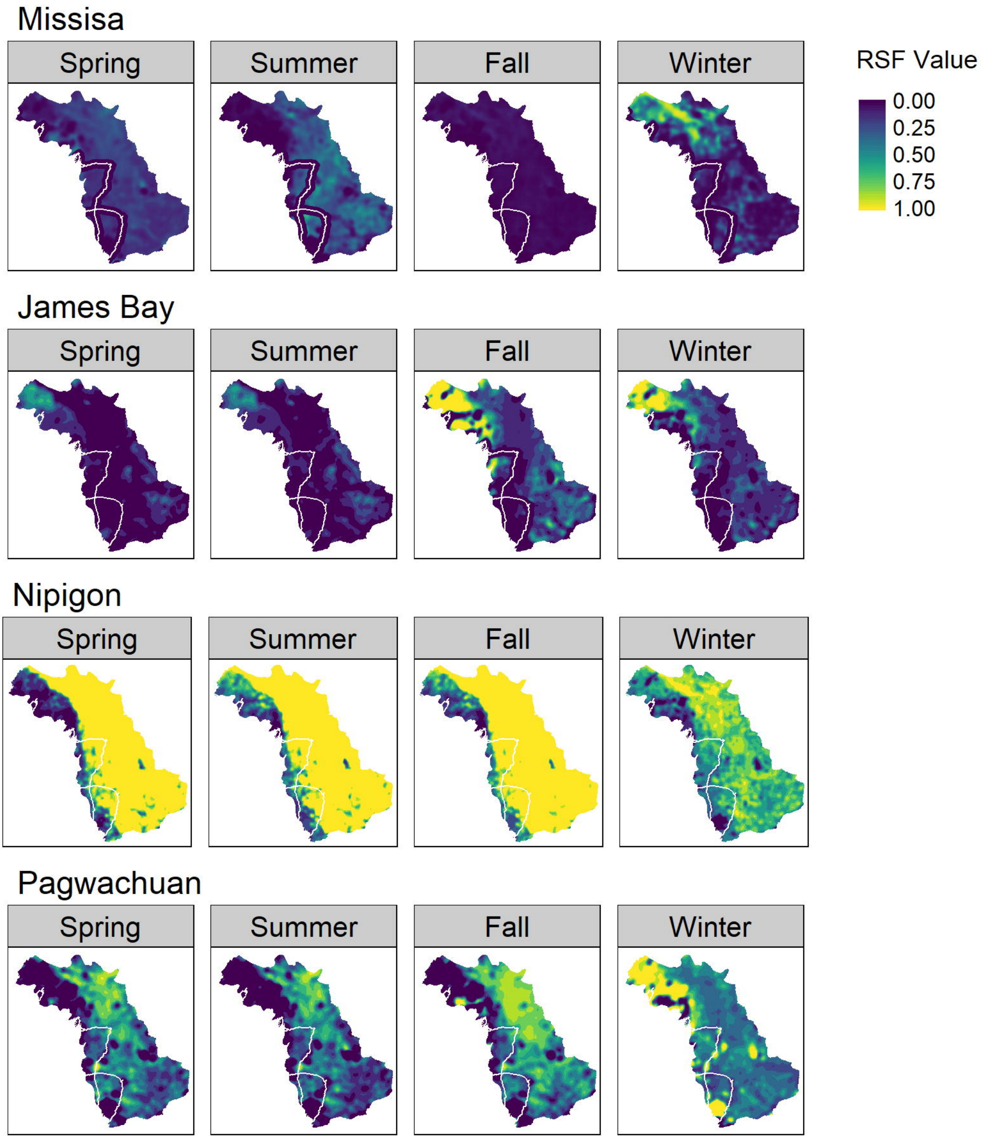
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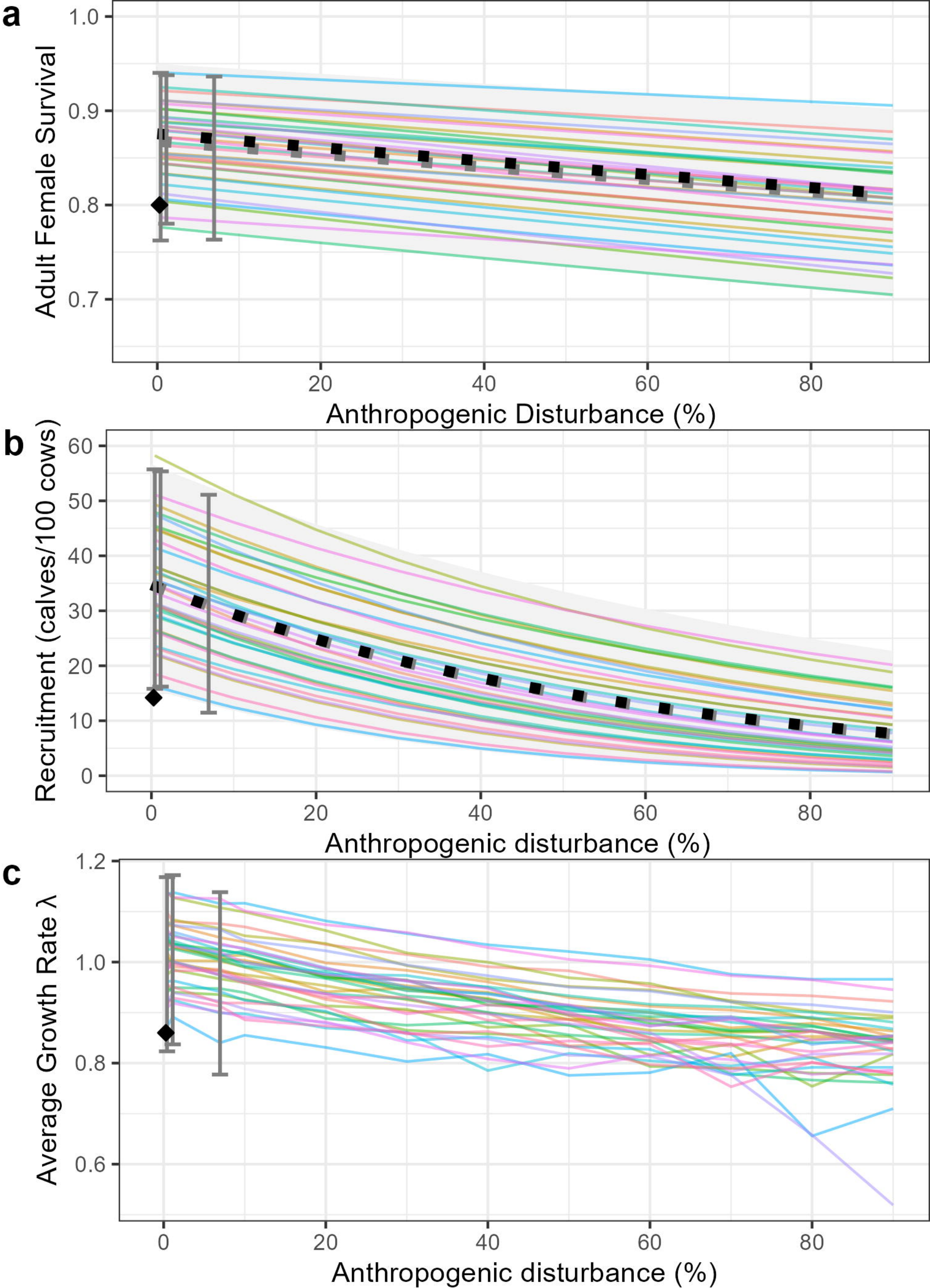
Fall
Winter
Pagwachuan
Spring
Summer
Fall
Winter
RSF Value
0.00
0.25
0.50
0.75
1.00



a
Adult Female Survival
1.0
0.9
0.8
0.7
0
20
40
60
80
Anthropogenic Disturbance (%)
b
Recruitment (calves/100 cows)
60
50
40
30
20
10
0
0
20
40
60
80
Anthropogenic disturbance (%)
c
Average Growth Rate λ
1.2
1.0
0.8
0.6
0
20
40
60
80
Anthropogenic disturbance (%)