

Vegetation characteristics of forest stands used by woodland caribou and those disturbed by fire or logging in Manitoba

Juha M. Metsaranta^{1*}, Frank F. Mallory¹ & Dale W. Cross²

¹ Department of Biology, Laurentian University, Sudbury, Ontario, Canada P3E 2C6 (fmallory@nickel.laurentian.ca).

² Manitoba Conservation, Northwest Region, P.O. Box 2550, The Pas, Manitoba Canada R9A 1M4.

* Current address: Canadian Forest Service, Northern Forestry Centre, 5320 122 Street, Edmonton, Alberta, Canada T6H 3S5.

Abstract: This study examined woodland caribou (*Rangifer tarandus caribou*) in an area known as the Kississing-Naosap caribou range in west central Manitoba. The vegetation characteristics of areas used by caribou and areas disturbed by fire or logging were measured in order to develop a model to estimate habitat quality from parameters collected during standard resource inventories. There was evidence that habitat index values calculated using a visual score-sheet index could be used as the basis to relate parameters commonly collected during resource inventories to habitat suitability. Use of this model to select long and short-term leave areas during forest management planning could potentially mitigate some of the negative impacts of forest harvesting. Abundance of arboreal lichen and wind-fallen trees were important predictor variables in the suitability model, but their inclusion did not explain more variance in habitat suitability than models that did not include them. Extreme post-fire deadfall abundance may play a role in predator-prey dynamics by creating habitat that is equally unsuitable for all ungulates, and thus keeping both moose and caribou densities low.

Key words: arboreal lichen, deadfall, disturbance, environmental impacts, forest management, habitat suitability, mitigation, predator-prey.

Rangifer, Special Issue No. 14: 255-266

Introduction

Woodland caribou (*Rangifer tarandus caribou*) are believed to be the late-successional forest ungulate in the boreal forest biome of North America. Numerous researchers have investigated the microhabitat characteristics of areas used by woodland caribou in boreal forest ecosystems over an entire year, as well as during specific seasonal periods, and across various disturbance regimes (e.g. Schaeffer & Pruitt, 1991; Racey et al., 1996; Rettie et al., 1997; Antoniak & Cumming, 1998; Martinez, 1998; Hirai, 1998; Wilson, 2000).

Studies have generally emphasized the importance of the abundance of arboreal and terrestrial lichen as a primary factor associated with microhabitat used by caribou during winter (Antoniak & Cumming, 1998; Martinez, 1998; Wilson, 2001). Snow conditions have also been considered important during winter, particularly factors contributing to lower

levels of snow accumulation (Schaeffer & Pruitt, 1991; Wilson, 2000). The proximity of escape habitat in the form of water or open bogs and wetlands is thought to be important during the calving and post-calving periods in the spring and summer (Bergerud, 1985; Cumming & Beange, 1987; Bergerud et al., 1990). Spring and summer habitat characteristics also reflect changes in diet from terrestrial and arboreal lichens in winter to emergent herbs, grasses, and deciduous shrubs during the snow-free period (Bergerud, 1972; Darby & Pruitt, 1984). Characteristics of habitats used during the breeding season (fall rut) are less known, although it is thought that sparsely treed and open upland areas or open muskeg swamplands are the primary habitat types used (Fuller & Keith, 1981; Cumming, 1992). Two other factors are thought to be important at all times of year. Deadfall can be a barrier to movement, potentially making habitat unsuitable for caribou,

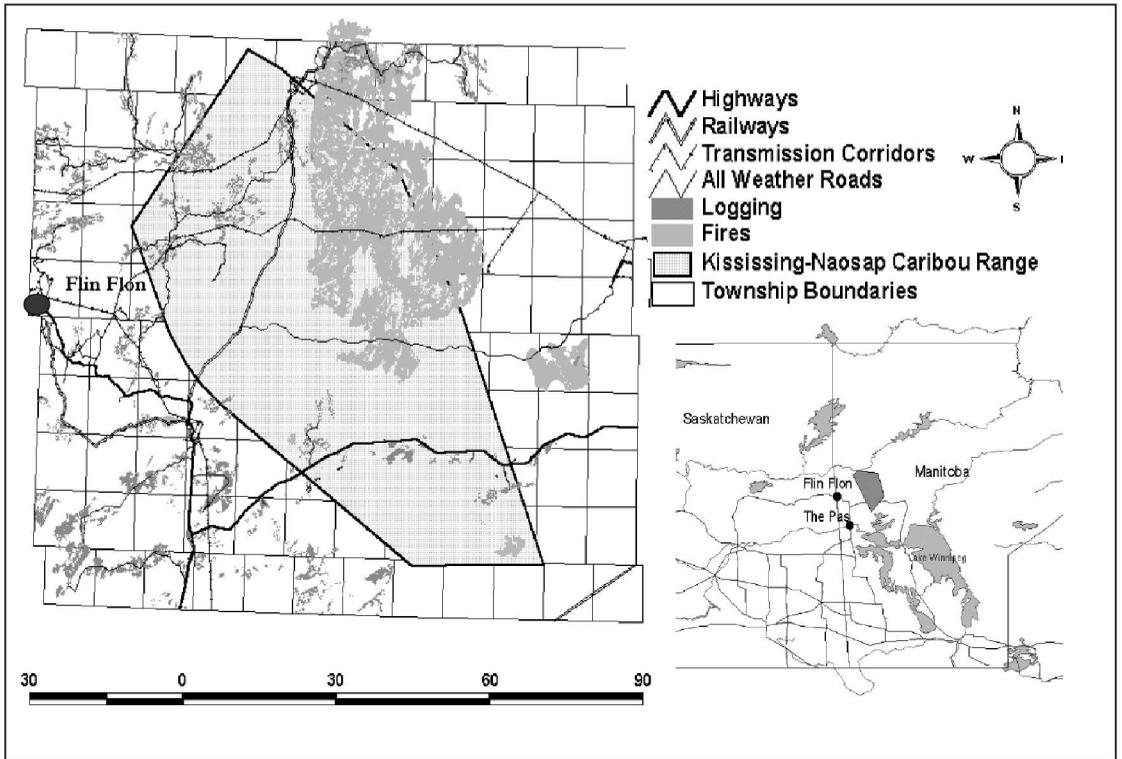


Fig. 1. Location of the Kississing-Naosap caribou range in west central Manitoba. The study area intersects the boundary of the Boreal Shield and Boreal Plains ecozones.

other ungulates, or predators depending upon its abundance. Shrub density can affect the ability of caribou to see and escape from predators, and can also change the suitability of the habitat to favour other ungulate species (e.g. moose, Van Dyke et al., 1995; Courtois et al., 1998).

Natural and human habitat disturbance can cause a number of changes in microhabitat characteristics. Natural disturbance (fire) is thought to reduce the abundance of lichens and increase accumulations of snow and deadfall, which reduce residual forage availability (Klein, 1982; Schaeffer & Pruitt, 1991). Human disturbance (logging) is thought to alter the structure of floral communities to favour deciduous early successional forest species (Carleton & MacClennan, 1994). These habitat types are generally thought to be unsuitable for caribou, as arboreal and terrestrial lichen forage species abundance is lower and the density of competing ungulates and predators is higher (Rettie & Messier, 2000). The tendency for caribou to segregate themselves spatially from other ungulates (particularly moose) in spring and summer is well known (Bergerud et al., 1984; Bergerud, 1985; Poole et al., 1999). This ability is potentially compromised by habitat disturbance (Smith et al., 2000).

It is believed that woodland caribou habitat selection varies with spatial scale in response to variations in impact of limiting factors at different spatial scales (Bergerud et al., 1990; Rettie & Messier, 2000). Johnson (1980) describes four scales or orders of habitat selection extending from the selection of geographic ranges (first order) down to the selection of components within a daily area (fourth order). Models of habitat suitability for caribou exist at various orders in this hierarchy. The majority are based on the general format for habitat suitability models (United States Fish and Wildlife Service, 1981), where a number of habitat variables are weighted in a mathematical relationship to give a defined habitat area a suitability score that ranges from 0 to 1. Such models are generally developed as decision support tools for resource management agencies. The source data are generally derived from forest resource inventory databases, as these are the standard vegetation databases used for forest management planning. Examples of such models have been developed in Manitoba (Palidwor & Schindler, 1995), British Columbia (Apps & Kinley, 1998), and Ontario (Antoniak & Cumming, 1998).

Most habitat suitability models apply at the level of the forest stand, and relate to the third order of

habitat selection – the characteristics of habitats within an individual's home range (Johnson, 1980). A potentially useful model of habitat suitability that uses a visual score sheet assessment technique and relates to the third and fourth (the microhabitat characteristics within forest stands) order scales was previously developed in this study area (Storey & Storey, 1980). This method has been proposed in the past as a tool to evaluate forests in Manitoba for their suitability for woodland caribou, but has not been widely implemented (Hristienko, 1985). This visual score sheet method assesses seven habitat index components (stand type; tree size class and canopy closure; cover; food plant diversity; food plant percent ground cover; deadfall; and area diversity) on a scale of 1 to 10. These components are then weighted to derive an aggregate habitat index ranging from 1 to 10 for that area. The visual assessment is based on how closely a site's characteristics for each of the habitat index components resembles ideal conditions, which in this case was defined as ideal winter habitat conditions (Storey & Storey, 1980).

Study area

This investigation was conducted in west central Manitoba and studied woodland caribou in the Kississing-Naosap range (approx. 4500 km²), located northeast of the towns of Flin Flon and The Pas (Fig. 1). Though precise range definitions have changed, a population estimate for the general area of the Kississing-Naosap range in 1992 varied from 150 to 300 individuals (Johnson, 1993). An aerial survey of the general area conducted in the winters of 1993 and 1994 observed between 90 and 164 animals (Cross & Smith, 1995). Though it could be debated, the data available indicate that the population in this region remained fairly stable or declined slightly throughout the last fifteen years.

The Kississing-Naosap range intersects the boundary of the Churchill River upland and the mid-boreal lowland eco-regions of the boreal shield and boreal plains eco-zones. The boreal shield landscape consists of rolling uplands and lowlands with many bedrock outcrops. This contrasts with the boreal plains landscape, which is topographically level to gently rolling, consisting of lacustrine or organic parent materials. Tree species include black spruce (*Picea mariana*), white spruce (*Picea glauca*), jack pine (*Pinus banksiana*), tamarack (*Larix laricina*), trembling aspen (*Populus tremuloides*), and white birch (*Betula papyrifera*). The climate is continental, characterized by short warm summers and cold snowy winters. Mean daily temperatures in the study area range from +17.7 °C in July to -21.4 °C in

January. Mean annual rainfall and snowfall range from 323.3 mm and 170.2 cm in The Pas to 345.3 mm and 143.9 cm in Flin Flon. Snowfall accumulation is typically present from mid-November to early April, with maximum mean depths of 40 cm to 45 cm occurring in January and February.

Logging began in the area in the early 1970s (Anon., 1996). The Kississing-Naosap caribou range is within the provincial Forest Management License Area No. 2 and is currently actively managed by Tolko Industries, Ltd. under a Forest Management Plan approved until 2009 (Anon., 1996). Fires are a natural component of ecosystems in the area, although they are currently actively suppressed. The most significant recent burn was the Webb Lake fire in 1989, which burned approximately 125 000 hectares in the central portion of the study area and a significant portion (approx. 25%) of the range of the Kississing-Naosap caribou (Anon., 1996). Caribou in the Kississing-Naosap range are also affected by highway and rail transportation corridors, road development associated with forestry operations, hydro transmission line construction, and various recreational activities. Because of the potential cumulative impacts of all of these disturbances and the potential for an increase in development pressure in the near future, the Kississing-Naosap range is being considered for designation as a high-risk range under a provincial woodland caribou conservation policy.

Study objectives

To mitigate the potential impacts of logging on woodland caribou, natural resource development agencies have proposed a number of management techniques. In the Kississing-Naosap range in west-central Manitoba, Canada, managers have established short and long term leave areas and undisturbed travel corridors that allow access to known calving sites and other important habitats (Anon., 1999). Qualitative information on the presence of caribou sign is sometimes collected as part of pre-harvest forest investigation surveys and operational timber cruises for forest management (Anon., 1996). However, quantitative relationships between parameters collected during these surveys and microhabitat suitability for woodland caribou have not been established in the area.

In order to establish such relationships, microhabitat characteristics thought to be important to woodland caribou were investigated in undisturbed areas used by this subspecies and in areas disturbed by fire or logging. Quantitative ecological and Forest Resource Inventory (FRI) parameters were related to habitat index scores calculated from the visual score

sheet method previously developed in the study area (Storey & Storey, 1980). A visual arboreal lichen abundance index to quantify the relative biomass of arboreal lichen at the sites sampled was also developed. The assumption that the visual score sheet index could distinguish habitats during different calendar seasons and disturbance regimes was tested. It was also hypothesized that a small number of easily measured quantitative FRI and ecological variables could be related to the visual index and predict habitat suitability for woodland caribou at the micro-site level.

Material and methods

Data collection

Sampling areas were located in undisturbed seasonal ranges known to be used by individual animals, as well as human disturbed (logged) sites, and naturally disturbed (burned) sites. Seasonal ranges (100% MCP) were defined from radio-telemetry locations collected from February 1996 to January 2000 as part of a concurrent study of range use and habitat selection. Six home ranges in each of the four calendar seasons (spring, summer, fall, and winter) were selected for use as sampling areas. These were chosen using a random number generator. The first six radio transmitter frequencies to be generated were selected. Fourteen disturbed sites of varying ages and disturbance types were also sampled. A random number generator was used to select ten by ten kilometre township squares as sampling areas for transects located at disturbed sites.

Line transects, a minimum of 500 m in length with plots located at 50 m intervals, were conducted within each sampling area. Transect start points within individual sampling areas were randomly selected using GIS, but limited to areas within 500 m of roads or lakes in order to provide access. Transect bearings were selected to intersect at least one location where the animal was known to have been present. When transportation resources were not available to access a randomly selected home range during the data collection period, an alternate site was selected. A similar procedure was used to select the starting point for transects located at logged and fire disturbed sites. Data collection occurred between July 29th and August 4th, 1999 and June 5th and July 26th, 2000. A total of 38 areas were sampled, consisting of 393 individual sample plots.

Data collected at each sample plot included scores for each habitat index component, Forest Resource Inventory (FRI) characteristics (species composition, age, height, canopy closure, diameter), and ecological characteristics (shrub/herb species composition),

including stand attributes thought to be important to caribou (deadfall density, visual density, and arboreal lichen abundance).

Data were collected hierarchically, using a number of nested plot sizes. Habitat index components were calculated by observing the characteristics of a 50 m x 50 m plot. The seven habitat index components (stand type; tree size class and canopy closure; cover; food plant diversity; food plant percent ground cover; deadfall; and area diversity) were each assessed on a scale of 1-10 (Storey & Storey, 1980). The individual habitat index components were also weighted and averaged (Storey & Storey, 1980) to give an aggregated habitat index score ranging from 1-10 for each plot. A brief description of how the habitat index components were scored is provided in the appendix.

Forest Resource Inventory and ecological characteristics were observed in a 10 m x 10 m plot located at the centre of the 50 m x 50 m visual index plot. Tree species composition, density, and diameter distribution at each plot was measured using the point-centred quarter method (Cottam & Curtis, 1956). Height and age were measured from a single representative tree selected at each plot. Heights were measured using a Suunto clinometer. Cores were extracted using an increment borer and collected for aging with the aid of a dissecting microscope. Crown closure was estimated visually to the nearest ten percent.

Shrub species composition was recorded from the same 10 m x 10 m plot as the FRI characteristics, while herb species composition was recorded on a randomly selected 2 m x 2 m plot within the 10 m x 10 m plot. Shrub and herb species composition was determined by visually estimating the percent cover (Daubenmire, 1959). Cover percentages were recorded to the nearest five percent. Shrubs were considered to be any woody plants between 1 and 5 m in height and less than 10 cm in diameter. Herbs included all non-woody plant species and woody shrubs less than 1 m in height. Deadfall amounts were determined by counting the number of fallen trees greater than 0.5 m from the ground along the 50 m line walked between each plot. Visibility was quantified by measuring the distance at which an individual walking along the transect between plots was no longer visible to an observer located at the previous plot.

Arboreal lichen abundance was assessed using a four level abundance index ranging from zero to three. To quantify this index, arboreal lichens were collected from sites throughout the study area. Nine sites representing each of the levels of abundance, with the exception of sites given an index value of 0,

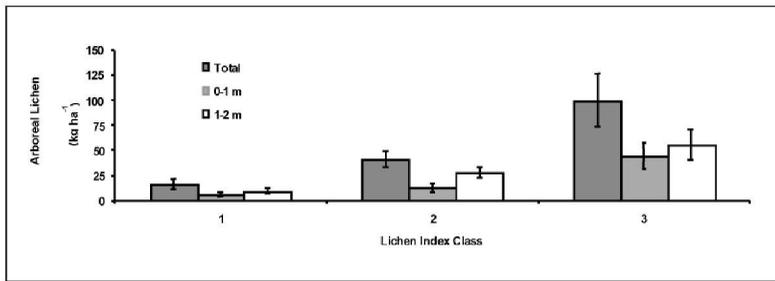


Fig. 2. Mean (SE) arboreal lichen biomass in each arboreal lichen abundance index class at heights of 0-1 m, 1-2 m, and total available height (0-2 m).

were sampled using a modification of the procedure first described by Van Daele & Johnson (1980). Lichens were collected from heights of 0-1 m and 1-2 m at each sample site, as maximum snow accumulation in this study area averaged less than 50 cm (Anon., 1998). For sampling purposes, each of the sample heights were divided into quarters. One quarter was then randomly selected and all arboreal lichens were stripped and collected. Samples were air dried for 48 h and oven-dried at 70 °C for one hour and measured to the nearest 0.01 g. This was then multiplied by four times the tree density at the plot to determine arboreal lichen abundance in kg ha⁻¹ in each height stratum.

Data analysis

The hypothesis that the total amount of arboreal lichen in each abundance index class did not differ was tested using a one-way analysis of variance and a Tukey's HSD multiple range test. Lichen biomass values were log-transformed to correct for skewness and stabilize variance. One-way analysis of variance and Tukey's HSD multiple range tests were used to explore the hypothesis that each of the individual visual index components, as well as the final habitat index did not differ among seasons and disturbance regimes. Where variance homogeneity assumptions were not met, the ANOVA results were confirmed with non-parametric (Kruskal-Wallis) tests.

Relationships between the habitat index score and measured habitat variables were modeled using stepwise multiple linear regression. Two regression models were developed. The first used standard FRI data as predictors (Model 1). The second used a combination of FRI data and ecological data as predictors (Model 2). Plant species percent cover estimates were reduced to presence/absence data to simplify the data requirements for applying the models. Only plants present in a minimum of 15% of plots were used to eliminate skewness.

Approximately half of the 393 plots sampled were randomly selected to develop the regression models.

To examine the internal reliability of the regression models, the regression predicted habitat values were correlated with measured habitat values at the sites not used to develop the regression model. The predicted habitat values were also regressed onto the measured habitat values. A one standard error of the estimate prediction interval was then applied to the measured/pre-

dicted relationship starting from the lowest possible habitat value that the regression models (1 or 2) could estimate in order to determine three statistical habitat quality categories.

All statistical analyses were done with SPSS 9.0 for Windows and results were considered statistically significant with $P < 0.05$.

Results

Arboreal lichen abundance index

Significant differences in the relative biomass of arboreal lichen were found among the four abundance index classes (Fig. 2). This was true for total lichen biomass ($F(2,24)=9.93$, $P < 0.01$), lichen biomass from 0-1 m above ground ($F(2,24)=8.36$, $P < 0.01$), and lichen biomass from 1-2 metres above ground ($F(2,24)=7.31$, $P < 0.01$). Post-hoc testing (Tukey, $P < 0.05$) showed that, in all cases, lichen biomass in each index class differed significantly from the others. Mean total lichen biomass was 15.7, 41.1, and 100.1 kg ha⁻¹ for abundance classes one, two, and three respectively (Fig. 2). Since the abundance index classes as defined reflected true differences in lichen biomass, this variable was considered in the rest of the analysis.

Habitat index component values

Each of the components in the habitat index was analyzed individually. Means and standard deviations for the habitat index components by season and disturbance type are listed in Table 1. In each case, there were significant differences ($F(5,387)=9.81$ to 58.82 , $P < 0.01$) between seasonal use areas and disturbance regimes. Using post-hoc testing (Tukey $P < 0.05$), it was found that used sites scored higher than disturbed sites in most cases, although there were some exceptions (Table 1). Burned sites were more likely than logged sites to not be different from used sites. An exception to this was deadfall, where burned sites scored much lower (indicating higher deadfall accumulation) than any of the used sites or

logged sites, which did not differ significantly from each other.

For each habitat index component, sites used in the winter were found in the highest value subset, with the exception of food plant diversity, where it scored less than fall sites. Both fall and winter sites had significantly higher food diversity than spring and summer sites. Summer sites also had significantly lower food plant ground cover than all the other seasons, but did not differ significantly from burned sites for this parameter (Table 1). Besides these variables, there was homogeneity in the habitat index component scores among seasons. All sites tended to be either closed or open stands of mature softwoods or mixed softwoods with more than 75% food plant ground cover and some (rather than no) deadfall, usually within 100 m of at least one or more other distinct habitat types.

Aggregate habitat index values

After analyzing each of the seven components individually, the individual scores were weighted to determine an aggregate habitat index score ranging from 0 to 10 for each plot. Significant differences were again noted between season and disturbance classes ($F(5,390)=86.2$, $P<0.01$) (Fig. 3). Post-hoc testing (Tukey, $P<0.05$) found that within seasonal use areas, winter sites scored significantly higher than summer sites, neither of which differed from spring or fall sites. All used sites as a group scored significantly higher than disturbed sites of either type. Within disturbed sites, burned areas scored significantly higher than logged areas (Fig. 3). At used sites, mean habitat index scores were 8.0, 7.5, 7.2, and 7.5 for winter, spring, summer, and fall areas respectively. At disturbed sites, mean habitat index scores were 5.7 and 5.0 for burned and logged areas, respectively.

Regression models to predict aggregated habitat index values Habitat index scores were related to FRI variables only (Model 1, Table 2) and a both FRI and ecological variables (Model 2, Table 3) using multiple linear regression to create a statistically significant relationship ($F(4,193)=67.79$ and 78.20 for Models 1 and 2, respectively, $P<0.01$). Presence of spruce trees, age, and mean tree diameter were positive predictors, while the presence of trembling aspen trees was a negative predictor for Model 1 (Table 2). Presence of spruce trees and arboreal lichen index were positive predictors, while presence of trembling aspen shrubs and deadfall density were negative predictors for Model 2 (Table 3). Both models explained approximately 60% of the variance in habitat index

Table 1. Comparison of habitat index components scores among sampling categories. Scores are presented as means ranging from 0-10, and (standard errors). Letters represent homogeneous subsets calculated using Tukey's HSD post-hoc testing. Subset A represents the highest scoring subset, B the second highest, etc.

Sampling category	Habitat index component						
	Stand type	Tree size class and crown closure	Cover	Food plant diversity	Food plant ground cover	Deadfall	Area diversity
Spring (n=63)	9.8 (0.12) A	7.5 (0.21) A	7.4 (0.23) A	5.1 (0.23) C	8.1 (0.27) A, B	9.0 (0.21) A	6.0 (0.26) A, B
Summer (n=68)	9.3 (0.22) A	7.7 (0.22) A	6.7 (0.28) A, B	5.2 (0.23) C	7.1 (0.25) B, C	9.5 (0.19) A	6.0 (0.28) A, B
Fall (n=60)	9.7 (0.18) A	6.9 (0.27) A	6.9 (0.32) A, B	7.1 (0.19) A	8.8 (0.18) A	8.9 (0.13) A	5.3 (0.26) B, C
Winter (n=60)	9.9 (0.10) A	7.7 (0.23) A	7.5 (0.26) A	6.0 (0.20) B	9.0 (0.21) A	9.5 (0.12) A	6.8 (0.23) A
Burned (n=52)	7.9 (0.38) B	3.7 (0.35) B	6.1 (0.25) B	4.6 (0.29) C, D	6.3 (0.39) C, D	7.3 (0.51) B	5.0 (0.29) B, C
Logged (n=90)	6.6 (0.31) C	4.2 (0.20) B	4.6 (0.25) C	3.9 (0.11) D	5.3 (0.24) D	8.8 (0.18) A	4.2 (0.26) C

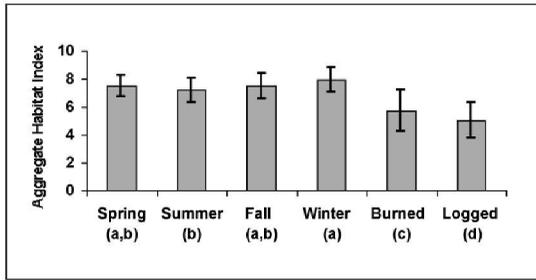


Fig. 3. Comparison of the aggregate habitat index scores among sampling categories. Scores are presented as means (± 1 s). Letters represent homogeneous subsets calculated using Tukey's HSD post-hoc testing. Subset A represents the highest ranking subset and D the lowest ranking subset.

scores ($r^2=0.58$ and 0.62 , $P<0.01$, for Models 1 and 2, respectively).

Measured habitat index scores were significantly correlated with the regression model predicted habitat index scores for plots not used to calculate the

models ($r^2=0.58$ and 0.51 , $P<0.05$ for Models 1 and 2, respectively). A regression relationship between measured scores and model predicted scores was developed by regressing predicted habitat index values onto measured habitat index values (Table 4). A prediction interval of \pm one standard error of the estimate was then applied to the relationship between actual and predicted values to calculate three statistical habitat quality categories for both models (Table 4). The bottom limit for the low quality habitat category was set to the lowest possible predicted value from the Model 1 and Model 2 relationships (4.6 and 5.0 for Models 1 and 2 respectively (Table 2 and 3)). Using the Model 1 and Model 2 equations to predict habitat index values, any sites scoring more than 8.7 for Model 1 or 8.4 for Model 2 represent high quality woodland caribou habitat.

Discussion

The aggregated habitat index was successful in distinguishing between seasonal habitat preferences,

Table 2. Regression model (Model 1) for predicting the aggregated habitat index score from Forest Resource Inventory parameters only.

Variable	Regression coefficient (a)	Multiple R ² (b)	% change in R ² (b)
(1)Presence of Spruce (0,1)	1.13	0.48	--
(2)Age (Years)	0.009	0.53	+10.4%
(3)Mean diameter (cm dbhob)	0.006	0.57	+7.5%
(4)Presence of Trembling Aspen (0,1)	-0.47	0.58	+1.8%
Constant	4.96	--	--

(a) Refers to final model with all four variables included.

(b) Refer to interim stepwise models, with the first, first and second, first, second, and third, first, second, third, and fourth variables included respectively.

Table 3. Regression model (Model 2) for predicting the aggregated habitat index score from a combination of Forest Resource Inventory and ecological parameters.

Variable	Regression coefficient (a)	Multiple R ² (b)	% change in R ² (b)
(1)Presence of Spruce (0,1)	1.54	0.51	--
(2)Presence of Trembling Aspen shrubs (0,1)	-0.78	0.57	+11.8%
(3)Arboreal lichen index (0,1,2,3)	0.34	0.61	+7.0%
(4)Deadfall density (stems m ⁻¹)	-0.02	0.62	+1.6%
Constant	5.72	--	--

(a) Refers to final model with all four variables included.

(b) Refer to interim stepwise models, with the first, first and second, first, second, and third, first, second, third, and fourth variables included respectively.

Table 4. Regression relationships between actual and predicted values for Models 1 (FRI parameters only) and 2 (FRI and ecological parameters in combination), and the statistical habitat quality classes calculated from these relationships based on +/- standard error of the estimate prediction intervals. The bottom limits of the low quality category are based on the smallest possible value that either the Model 1 or 2 relationships could predict.

Habitat Quality	Model 1 - FRI parameters Predicted Value = 0.57 (Actual Value) + 2.76 SE Estimate = 0.71	Model 2 - FRI and Ecological parameters Predicted Value = 0.59 (Actual Value) + 2.72 SE Estimate = 0.86
Low	4.6 to 7.3	5.0 to 6.7
Medium	7.3 to 8.7	6.7 to 8.4
High	8.7 to 10.0	8.4 to 10.0

particularly winter and summer areas. The relationship was not as clear for spring and fall areas, which did not differ from either winter or summer use areas. Migrations between winter and summer ranges are a factor influencing where animals are located during the spring and fall, resulting in the sampling of areas that animals may have only been temporarily associated with. A closer look at the spatial distribution of study animals revealed that, in many cases, areas sampled as spring or fall use areas were often used at other times of the year, usually summer or winter. It is also possible that there are no differences in the vegetation characteristics of habitat used by this species in the spring and fall relative to other times of the year. Both of these possibilities suggest that summer and winter are the only two times of year that woodland caribou select habitats with specific characteristics at the micro-site scale in this area. At larger spatial scales, such a conclusion is supported by habitat and movement studies elsewhere in Canada that have shown a bimodal pattern of range use, with distinct summer and winter use areas (Edmonds, 1988). However, such a pattern is not constant, with some populations showing much overlap between seasonal ranges (Ouellet et al., 1996; Stuart-Smith et al., 1997).

Considering all factors, the visual score sheet worked as it was intended. Numerically the trend was appropriate. Within use areas it generated the highest scores for winter sites and the lowest scores for summer sites, with intermediate scores for spring and fall (Fig. 3). Scores for disturbed areas were lower than for all used areas. Statistically, it was able to generate significantly different aggregate habitat index scores for the two seasons (summer and winter) that caribou seem to recognize in this area. It was

also able to generate significantly different aggregate scores for used areas and areas disturbed by fire or logging.

Habitat index prediction relationships

The results support the conclusion that the habitat index could successfully be related to both standard FRI data and FRI data in combination with ecological data, to distinguish three habitat quality classes. In each case, approximately 60% of the variance in habitat value could be explained using the quantitative data set under consideration.

Although the addition of ecological data created a different model than the FRI data alone, it did not explain any more variance and created similar habitat quality categories. This analysis provides evidence that the relative abundance of arboreal lichen and deadfall density are important ecological parameters influencing micro-habitat suitability for woodland caribou. However, it does not necessarily support the need to measure these parameters during pre-harvest forest assessments or during FRI data collection since they did not help to create a better assessment of habitat suitability in this case.

With the exception of arboreal lichen, no other individual forage plant could be considered a predictor of habitat suitability. This is not surprising considering the variety and seasonal fluctuation in the forage plants that woodland caribou are known to consume (Bergerud, 1972). Generally, arboreal lichen is most important as a forage plant during winter (Bergerud, 1972). Mean scores for the arboreal index in this study were 1.4 ($s=0.9$, $n=60$) for winter sites, 1.3 ($s=0.9$, $n=191$) for other seasons, and 0.2 ($s=0.5$, $n=141$) for disturbed sites (J. Metsaranta, unpubl. data). Thus, in this study area, arboreal lichens appear to be equally abundant at sites used during all seasons, and much less abundant at disturbed sites.

Many studies have postulated the importance of arboreal or terrestrial lichen in determining habitat suitability for woodland caribou. However, few studies have considered deadfall density. The only habitat index component that did not differ between used sites and logged sites was the abundance of deadfall (Table 1). Measured deadfall density at burned sites was 17.3 stems m^{-1} ($s=16.5$, $n=50$), while measured deadfall density at all other sites was

5.2 stems m^{-1} ($s=6.4$ $n=325$) (J. Metsaranta, unpubl. data). Schaeffer & Pruitt (1991) also examined the influence of deadfall on woodland caribou, and found that deadfall densities were much higher in burned areas than in corresponding unburned areas that continued to be used by caribou. They found continued use of unburned or crown burned residual stands and unburned lowlands during summer, with an apparent lack of use only in winter (Schaeffer & Pruitt, 1991). Similarly, in this study, a number of animals continued to use unburned residual portions and unburned islands in lakes within the extensive area burned by the Webb Lake fire in 1989.

Previous telemetry studies in the Reed Lake area southeast of this fire documented traditional caribou use of the burned area prior to 1989, but generally during summer only (Shoesmith & Storey, 1977; Benoit, 1996). Although data for determining if the current level of use is less than before the fire is lacking, the results would still suggest that a certain level of habitat suitability remains within the burned area since it has not been completely abandoned. Previous studies have suggested that the reduction in caribou habitat suitability following fire occurs as a result of lichen combustion or increased snow accumulation reducing residual forage availability (Klein, 1982; Schaeffer & Pruitt, 1991). The results of this study suggest that deadfall accumulation may also be an important factor, with the degree of impediment to travel imposed by the accumulation of deadfall possibly limiting the use of burned uplands at all times of year.

Lichens begin to re-establish in both logged and burned areas anywhere from 10 to 50 years after disturbance, with regeneration potentially occurring sooner after logging than after fire (Harris, 1996; Webb, 1998). Although disturbance by fire may have short-term detrimental effects on caribou, the long-term effects tend to be positive in terms of forage productivity (Klein, 1982; Schaeffer & Pruitt, 1991). However, in the interim period, the post-fire accumulation of deadfall might concurrently discourage use of this habitat by all ungulates, by creating areas that are virtually impassable to travel.

The literature has generally emphasized the positive impacts of fire on moose habitat by its tendency to create more abundant forage for this species (e.g. Schwarz & Franzmann, 1989; Loranger et al., 1991). However, it is also acknowledged that this may not universally be the case, depending on such factors as fire intensity and pre-fire moose density (e.g. Peek, 1974; Gasaway et al., 1989). Some have suggested that moose appear to avoid blowdown areas, where presumably deadfall densities would be high (Cumming, 1980). If this was in fact true, and con-

tinued to be the case during the 15 to 40 year post-fire period when moose populations are thought to be highest (Schwarz & Franzmann, 1989), then changes in the composition of the faunal community (i.e. increases in moose populations, and as a consequence increases in wolf populations (e.g. Bergerud & Elliot, 1986; Seip, 1992)) that potentially have a negative effect on woodland caribou might not occur.

The proposed mechanism for preventing this faunal shift is the post-fire accumulation of deadfall that could occur under certain fire intensity and vegetation combinations. Such circumstances may create habitat that, for a certain period of time, is equally unsuitable for both woodland caribou and moose. In the 12 years after the Webb Lake fire in 1989, two surveys of moose populations in this area have been conducted (Cross, 1991; Cross, 2000). An initial survey in 1990 predicted that moose density in this burned area should increase. However, a second survey in 2000 concluded that these increases had not yet occurred, and that areas where deadfall accumulations were high were nearly devoid of moose activity (Cross, 2000).

Conclusions

The analysis provides evidence for the efficacy of the visual score sheet method (Storey & Storey, 1980) for assessing microhabitat suitability for woodland caribou. It also shows that habitat index values calculated using the visual score-sheet index could be used as the basis to relate FRI and ecological parameters to microhabitat suitability. Any of the three methods presented here (i.e. the score sheet method or Model 1 and Model 2) could be used to select forest stands for inclusion into short and long-term leave areas and travel corridors with more confidence that they contain habitat that will be suitable for use by woodland caribou. This would potentially, at least at the microhabitat scale, mitigate some of the negative impacts that forestry and other development has had on this species.

However, it is known that different selective pressures act at different spatial scales to limit woodland caribou populations (Bergerud et al., 1990; Rettie & Messier, 2000). It is also known that there are no simple solutions to natural resource management problems, and that unless an ecosystem approach is applied, actions can have unwanted and unpredicted consequences (Thompson & Welsh, 1993). Management actions to conserve woodland caribou populations need to consider all the factors that potentially act to limit this species (Cumming, 1992). Ecologically, it is known that these act at

both small and large spatial scales. Applying the models presented here could conserve habitat characteristics needed by woodland caribou at small spatial scales. However, a further investigation quantifying the pattern and arrangement of suitable habitat types at a larger spatial scale would further increase the confidence in the ability to mitigate the impacts of development on this species.

Acknowledgements

Thanks to Becky Ferguson, Marcel Paré and one anonymous reviewer. Funding was provided by Manitoba Conservation, Tolko Industries Ltd., Manitoba Hydro, Laurentian University and Natural Sciences & Engineering Scholarship to SMM.

References

- Anonymous.** 1996. Repap Manitoba 1997-2009 Forest Management Plan. Repap Manitoba, Inc. The Pas, Manitoba.
- Anonymous.** 1999. Forest management/woodland caribou mitigation plan: Naosap/Peterson operating areas. Tolko Industries, Ltd. Manitoba Woodlands. Unpubl. Report.
- Antoniak, K. & Cumming, H.G.** 1998. Analysis of forest stands used by wintering woodland caribou in Ontario. – *Rangifer Spec. Issue No. 10*: 157-168.
- Apps, T.A. & Kinley, C.D.** 1998. Development of a preliminary habitat assessment and planning tool for mountain caribou in southeast British Columbia. – *Rangifer Spec. Issue 10*: 61-72.
- Benoit, A.D.** 1996. A landscape analysis of woodland caribou habitat use in the Reed-Naosap Lakes region of Manitoba (1973-1985). University of Manitoba, Natural Resources Institute practicum, Winnipeg, Manitoba. 109 pp.
- Bergerud, A.T.** 1972. Food habits of Newfoundland caribou. – *J. Wildl. Manage.* 36: 913-923.
- Bergerud, A.T.** 1985. Antipredator strategies of caribou: dispersion along shorelines. – *Can. J. Zool.* 63: 1324-1329.
- Bergerud, A.T., Butler, H.E., & Miller, D.R.** 1984. Antipredator tactics of calving caribou: dispersion in mountains. – *Can. J. Zool.* 62: 1566-1575
- Bergerud, A.T. & Elliot, J.P.** 1986. Dynamics of caribou and wolves in northern British Columbia. – *Can. J. Zool.* 64: 1515-1529.
- Bergerud, A.T., Ferguson, R. & Butler, H.E.** 1990. Spring migration and dispersion of woodland caribou at calving. – *Anim. Behav.* 39(2): 360-368.
- Carleton, T.J. & MacLennan, P.** 1994. Woody vegetation responses to fire versus clear-cutting logging: A comparative survey in the central Canadian boreal forest. – *Ecoscience* 1: 141-152.
- Cottam, G. & Curtis, J.T.** 1956. The use of distance measures in phytosociological sampling. – *Ecology* 37: 451-460.
- Courtois, R., Ouellet, J.P., & Gagne, B.** 1998. Characteristics of cutovers used by moose (*Alces alces*) in early winter. – *Alces* 34: 201-211.
- Cross, D.W.** 1991. Moose population estimation survey and age-sex data for Grass River Provincial Park and age-sex data for 5 preselected areas in GHA 7: 1989-90. Manitoba Natural Resources, Wildlife Branch. Manuscript Report No. 91-11. 15pp.
- Cross, D.W.** 2000. Moose Population Survey: GHA 7A – Northwest Region, January 2000. Manitoba Conservation, Northwest Region. Unpubl. Rep. 28pp.
- Cross, D.W., & Smith, C.E.** 1995. Woodland caribou winter distribution, habitat use assessment, and population estimate survey: Kisissing Lake area February 1993 and 1994. Manitoba Natural Resources/Repap Manitoba Inc. Unpubl. Rep. 29pp.
- Cumming, H.G.** 1980. Relation of moose track counts to cover types in northcentral Ontario. – *Proc. North. Am Moose Conf. Workshop* 16: 444-462.
- Cumming, H.G.** 1992. Woodland caribou: facts for forest managers. – *For. Chron.* 68(4): 481-491.
- Cumming, H.G. & Beange, D.B.** 1987. Dispersion and movements of woodland caribou near Lake Nipigon, Ontario. – *J. Wildl. Manage.* 51(1): 68-79.
- Darby, W.R. & Pruitt, W.O., Jr.** 1984. Habitat use, movements, and grouping behaviour of woodland caribou, *Rangifer tarandus caribou*, in southeastern Manitoba. – *Can. Field Nat.* 98(2): 184-190.
- Daubenmire, R.F.** 1959. A canopy-coverage method of vegetational analysis. – *Northwest Sci.* 33: 43-64.
- Edmonds, E.J.** 1988. Population status, distribution, and movements of woodland caribou in west central Alberta. – *Can. J. Zool.* 66: 817-826.
- Fuller, T.K. & Keith, L.B.** 1981. Woodland caribou population dynamics in northeastern Alberta. – *J. Wildl. Manage.* 45(1): 197-213.
- Gasaway, W.C., DuBois, S.D., Boertje, R.D., Reed, D.J., & Simpson, D.T.** 1989. Response of radio-collared moose to a large burn in central Alaska. – *Can. J. Zool.* 67: 325-329.
- Harris, A.** 1996. Post-logging regeneration of reindeer lichens (*Cladina* spp.) as related to woodland caribou winter habitat. Northwest Science and Technology, Ontario Ministry of Natural Resources, Thunder Bay, Ontario. Technical Report No. 69.
- Hirai, T.** 1998. An evaluation of woodland caribou (*Rangifer tarandus caribou*) calving habitat in the Wabowden area, Manitoba. University of Manitoba, Natural Resources Institute practicum, Winnipeg, Manitoba. 119pp.
- Hristienko, H.** 1985. The impact of logging on woodland caribou (*Rangifer tarandus caribou*): a literature review.

- Manitoba Natural Resources, Wildlife Branch, Winnipeg Manitoba. Techn. Rep. No. 85-3.
- Johnson, D.H. 1980. The comparison of usage and availability measurements for evaluating resource preference. – *Ecology* 61(1): 65-71.
- Klein, D.R. 1982. Fire, lichens, and caribou. – *J. Range Manage.* 35: 390-395.
- Loranger, A.J., Bailey, T.N., & Larned, W.W. 1991. Effects of forest succession after fire in moose wintering habitats on the Kenai Peninsula, Alaska. – *Alces* 27: 100-109.
- Martinez, I.M. 1998. Winter habitat use by woodland caribou (*Rangifer tarandus caribou*) in the Owl Lake region of Manitoba. Univeristy of Manitoba, Natural Resources Institute practicum, Winnipeg, Manitoba. 130pp.
- Ouelett, J.P., Ferron, J., & Sirois, L. 1996. Space and habitat use by the threatened Gaspé caribou in southeastern Quebec. – *Can. J. Zool.* 74: 1922-1933.
- Palidwor, K. & Schindler, D. 1995. Habitat suitability index models within the Manitoba model forest area: woodland caribou (*Rangifer tarandus caribou*). Version 2.0. Manitoba Model Forest, Pine Falls, MB. 112 pp.
- Peek, J.M. 1974. Initial response of moose to a forest fire in northeastern Minnesota. – *Am. Mid. Nat.* 91: 435-438.
- Poole, K.G., Heard, D.C., & Mowat, G. 1999. Habitat use by woodland caribou near Takla Lake in central British Columbia. – *Can. J. Zool.* 78: 1552-1561.
- Racey, G.A., Harris, A.G., & Foster, R.F. 1996. Caribou winter habitat in the new forest: lessons from Lucy Lake. Northwest Science and Technology, Ontario Ministry of Natural Resources, Thunder Bay, Ontario. Technical Report No.103.
- Rettie, W.J., Sheard, J.W., & Messier, F. 1997. Identification and description of forested vegetation communities available to woodland caribou: relating wildlife habitat to forest cover data. – *For. Ecol. Man.* 93: 245-260.
- Rettie, W.J. & Messier, F. 2000. Hierarchical habitat selection by woodland caribou: its relationship to limiting factors. – *Ecography* 23: 466-478.
- Schaeffer, J.A. & Pruitt, W.O., Jr. 1991. Fire and woodland caribou in southeastern Manitoba. – *Wildl. Monogr.* 116: 1-39.
- Schwarz, C.C. & Franzman, A.W. 1989. Bears, wolves, moose and forest succession: some management considerations on the Kenai Peninsula, Alaska. – *Alces* 25: 1-10.
- Seip, D.R. 1992. Factors limiting woodland caribou populations and their interrelationships with wolves and moose in southeastern British Columbia. – *Can. J. Zool.* 70: 1494-1503.
- Shoesmith, M.W. & D.R. Storey. 1977. Movements and Associated Behaviour of Woodland Caribou in Central Manitoba. – *Proc. Int. Congr. Game Biol.* 13: 51-64.
- Smith, K.G., Ficht, E.J., Hobson, D., Sorensen, T.C., & Hervieux, D. 2000. Winter distribution of woodland caribou in relation to clear-cut logging in west-central Alberta. – *Can. J. Zool.* 78: 1433-1440.
- Storey, D.R. & Storey, K.M. 1980. Woodland caribou habitat evaluation – a score sheet method (revised). Manitoba Natural Resources, Wildlife Branch, Winnipeg, Manitoba. MS Report No. 80-24.
- Stuart-Smith, A.K., Bradshaw, C.J.A., Boutin, S., Hebert, D.M., & Rippen, A.B. 1997. Woodland caribou relative to landscape patterns in northeastern Alberta. – *J. Wildl. Manage.* 61(3): 622-633.
- Thompson, I.D. & Welsh, D.A. 1993. Integrated resource management in boreal forest ecosystems – impediments and solutions. – *For. Chron.* 69(1): 32-39.
- United States Fish and Wildlife Service. 1981. Standards for the development of habitat suitability index models. United States Fish and Wildlife Service, Release No. 1-81, 103 ESM.
- Van Daele, L.J. & Johnson, D.R. 1983. Estimation of arboreal lichen biomass available to caribou. – *J. Wildl. Manage.* 47(3): 888-890.
- Van Dyke, F., Probert, B.L., & Van Beek, G.M. 1995. Seasonal habitat use characteristics of moose in south-central Montana. – *Alces* 31: 15-26.
- Webb, E.T. 1998. Survival, persistence and regeneration of the reindeer lichens, *Cladina stellaris*, *C. rangiferina*, and *C. mitis* following clearcut logging and forest fire in northwestern Ontario. – *Rangifer Spec. Issue No.* 10: 41-47.
- Wilson, J.E. 2000. Habitat characteristics of late wintering areas used by woodland caribou (*Rangifer tarandus caribou*) in northeastern Ontario. MSc Thesis, Laurentian Univ., Sudbury, Ontario. 103pp.

Appendix

Habitat index component scores

The following is a description of the factors that were assessed at a site when assigning values to each of the seven habitat index components that comprised the aggregated habitat index. A full description of each of the habitat index components and how they are weighted to determine an aggregate score can be found in Storey & Storey (1980).

I – Stand Type:

- Newly Cleared Land - 1
- Pioneer Shrubs - 4
- Hardwoods (>70%) - 6
- Mixed Hardwoods (50-70%) - 7
- Mixed Softwoods (50-70%) - 10
- Softwoods (>70%) - 10
- Overmature forest (>50% softwoods) – 6

II – Tree Size Class/Canopy Closure:

- Semi-open mature - 10
- Semi-open mixed - 9
- Closed or open mature - 8
- Closed or open mixed - 7
- Semi-open or open poles - 5
- Closed poles - 4
- Regeneration - 2
- No trees present – 1

III – Cover (includes tree trunks and branches between 0.5 and 2.0 metres height):

- 0% - 1
- 25% - 5

- 50% - 10
- 75% - 8
- 100% - 5

IV – Food Plant Diversity (only species in significant amounts):

- No food species present - 1
- Only herbs - 3
- Mosses, lichens and herbs - 6
- Deciduous shrubs, mosses, lichens, and herbs - 8
- Evergreen shrubs, deciduous shrubs, mosses, lichens, and herbs - 10

V – Food Plant % Ground Cover:

- 0% - 1
- 25% - 4
- 75% - 8
- 100% - 10

VI – Deadfall:

- Impossible to walk - 1
- Difficult to walk - 3
- Some deadfall, but no problem walking - 10
- No deadfall - 8

VII - Area Diversity:

- Homogeneous, far from different stand type - 1
- Within 100 m of 1 different stand type - 5
- Within 100 m of 2 different stand types - 8
- Within 100 m of >2 different stand types - 10