Historical perspectives on woodland caribou (*Rangifer tarandus caribou*) and landscape changes in the Prince Albert Greater Ecosystem

By

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A thesis submitted to the Faculty of Graduate Studies.
In partial fulfillment of the requirements for the Degree of

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Department of Environment and Geography
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FACULTY OF GRADUATE STUDIES

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Abstract

In central Saskatchewan, boreal woodland caribou population declines have been documented. In order to contribute to the conservation efforts, historical changes in caribou distribution, land cover, and habitat quality were documented in the Prince Albert Greater Ecosystem (PAGE), Saskatchewan, for the period of 1950 to the present. To examine changes in caribou distribution, survey observations, incidental sightings and telemetry data were collated. To quantify landscape changes, land cover maps were created for 1966 and 2006 using current and historic forest resources inventories, fire, logging, and roads data.

Results indicate that woodland caribou are still found throughout the study area although their distribution has changed and their use of Prince Albert National Park is greatly limited. Transition probabilities and landscape composition analyses point to an ageing landscape for both the National Park and provincial crown land portions of the PAGE. In addition, increased logging and the development of extensive road and trail networks on provincial crown land has resulted in significant landscape fragmentation and reduced functional attributes of caribou habitat.

To assess how these changes have affected the spatial arrangement of caribou habitat, a resource selection function (RSF) using GPS telemetry data and generalized estimating equations was developed. Results showed selection for treed muskeg and mature jack pine dominated stands and avoidance of hardwood, young coniferous stands, logging and linear features. The best model was applied to the 1966 and 2006vegetation maps to produce a predictive habitat map. Results showed greater area covered by high quality habitat in 2006 however the high quality habitat was clustered. In 1966 there was

less high quality habitat but it was distributed throughout the landscape. To assess changes in habitat connectivity between 1966 and 2006, spatial graph theory was used to create minimum planar graphs. The results showed the 1966 landscape was connected at much lower cost distance thresholds. Although there is presently a greater amount of caribou habitat on the landscape, the high level of anthropogenic activities on the PAGE area reduces the overall potential of this landscape for caribou.

To ensure the viability of boreal caribou on this landscape, habitat connectivity should be maintained throughout the area with larger clusters of habitat present at all times and adequate connectivity between these areas. To ensure caribou use of the range, habitat connectivity within and beyond Park boundaries should be maintained. Understanding historical landscape changes will assist with ongoing provincial and federal recovery efforts for boreal caribou, forest management planning activities, and landscape restoration efforts within and beyond the Park boundaries.

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1. Introduction

1.1 Background

Boreal caribou (*Rangifer tarandus caribou*) are one of five populations of woodland caribou in Canada (COSEWIC, 2002). This boreal population of woodland caribou was listed as threatened by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) in 2000; this status was reevaluated and confirmed in 2002. COSEWIC (2002) defines threatened as "a species likely to become endangered if limiting factors are not reversed". Boreal caribou have the largest range of all woodland caribou ecotypes and, because they occur in southerly areas, human encroachment is a problem (Thomas and Gray, 2002). The listing of boreal woodland caribou was based on habitat loss and increased predation thought to be facilitated by human activities (COSEWIC, 2002).

Although caribou have a varied diet, habitat rich in lichen is preferred because the high carbohydrate composition of select lichen species provides energy benefits (Coxson and Marsh, 2001; Thomas and Gray, 2002). Favourable lichen conditions make mature forests and wetland complexes preferable for woodland caribou (Thomas and Gray, 2002). Many studies throughout Canada have shown that woodland caribou select for treed muskeg, wetlands, and mature coniferous stands (Stuart-Smith et al., 1997; Hirai, 1998; Brown et al., 2000a; Brown et al., 2000b; Rettie and Messier, 2000; Schneider et al., 2000; Mahoney and Virgl, 2003; O'Brien et al., 2006). Habitat use is more variable in summer, which corresponds to a more diverse diet (Chubbs et al., 1993). In winter, snow thickness, hardness, and density influence habitat selection, which leads to caribou selecting sites with higher canopy cover (Schaefer and Pruitt, 1991; Schaefer, 1996).

Anthropogenic disturbances such as logging and the construction of linear features have a negative impact on caribou and their habitat (Cumming and Beange, 1993; Ferguson et al., 1988; Rettie and Messier, 1998; Smith et al., 2000; Rettie and Messier, 2000; Dyer et al., 2002; Thomas and Gray, 2002; Cameron et al., 2005; Whittington et al., 2005; Wittmer et al., 2005). These activities fragment and destroy habitat (Smith et al., 2000) and may lead to increased predation, primarily by wolves (*Canis lupus*) (James et al., 2004).

Relative to other boreal ungulates, woodland caribou have greater daily and seasonal movement rates (Johnson and Gilligham, 2002). Although some herds of boreal caribou show seasonal migrations, others do not migrate (Cumming and Beange, 1987). Often, studies delineate seasons, usually 5-8, based on movement rates (Bergman et al., 2000; Brown et al., 2003; Rettie and Messier, 2001; Ferguson and Elkie, 2004). In general, boreal caribou have the highest movement rates in the autumn, during the rut, and the lowest movements in the spring, during calving (Bergman et al., 2000; Rettie and Messier, 2001; Brown et al., 2003; Ferguson and Elkie, 2004). Various factors influence home range size of boreal caribou. In general, individual annual home range sizes are 150km^2 to 5900km^2 (Brown et al., 2000a; Poole et al., 2000; Rettie and Messier, 2001; Brown et al., 2003).

Predation is the principal factor that limits the size of most boreal caribou populations (Bergerud and Elliot, 1986; Bergerud, 1988; Post et al., 2003; Wittmer et al., 2005). To decrease predation risk, boreal caribou are found at low densities, on large tracts of mature coniferous forest, as an antipredator strategy (Cumming and Beange, 1993; Bergerud and Page, 1987). Disturbance leads to early successional forests that attract other ungulate species such as moose (*Alces alces*), elk (*Cervus elaphus*), and white-tailed deer (*Odocoileus virginianus*) (Brown et al., 2000a; Thomas and Gray, 2002; James et al., 2004). This increase in competitor populations is a concern as it decreases their spatial separation from caribou, often leading to further increases in predator populations, through reliance on these alternate prey species (Bergerud and Elliot, 1986; James et al., 2004). Overlap with other ungulate species may also increase disease transmission, particularly brainworm (Cumming and Beange, 1993; Pitt and Jordan, 1994; Thomas and Gray, 2002).

Caribou require habitat that is of suitable quality and quantity. The suitability of the landscape, however, is not determined only by the amount of suitable habitat available but also by the spatial configuration of areas of suitable habitat (Ferreras, 2001; Cook et al., 2003; Haynes and Cronin, 2004; O'Brien et al., 2006). Connectivity affects the ease with which an organism can move through the landscape (Taylor et al., 1993; Tischendorf and Fahrig, 2000). The impact of a decrease in connectivity for a given species depends on the hostility of habitat types in the matrix, the area surrounding the high quality patches, and the ability of a particular organism to cross unfavourable habitat types (With et al., 1999). Related to connectivity is fragmentation which is often caused by human land use activities (Andrén, 1994). This leads to decreased patch size and increased number of patches ultimately isolating populations from one another (Bélisle and Desrochers, 2002). Patch size is important to caribou. O'Brien et al., (2006) found that woodland caribou were associated with large clusters of high quality habitat patches likely to allow for predator avoidance and adequate forage.

1.2 Objectives

There are four objectives to this project

- Assess changes in caribou distribution in the Prince Albert Greater Ecosystem over the past 50 years,
- Quantify landscape changes between 1966 and 2006,
- Assess changes in habitat quality, between 1966 and 2006 using predictive mapping,
- Assess changes in habitat quality, between 1966 and 2006 using connectivity analyses.

1.3 General Methods

In order to examine changes in caribou distribution over time, woodland caribou occurrence data and associated survey efforts were collated for the period of 1950 to present. Data were obtained from Parks Canada and Saskatchewan Ministry of Environment and primarily consisted of survey observations, incidental sightings and telemetry data.

Using current and historic forest resources inventories, fire, logging, and roads data, the PAGE landscape was reconstructed for the period of 1966 to 2006. To quantify landscape change throughout the study area, landscape metrics and connectivity analysis were used. Landscape metrics were selected to describe patch size and shape, edge effects, fragmentation, and isolation. Statistical tests were used to assess changes in the landscape throughout the study period based on these landscape metrics.

To develop predictive habitat maps for 1966 and 2006, the influence of habitat types on the habitat selection of caribou was modeled by assessing differences between random and telemetry locations. A global model composed of biologically relevant variables was created. Then Proc GENMOD in SAS 9.1 (SAS Institute Inc., 2003) was used to develop generalized estimating equations (GEEs) for analysis. The distance layers generated in SELES were combined to generate a predictive habitat maps for 1966 and 2006.

To demonstrate changes in the connectivity of the landscape, spatial graph theory (as per Fall et al., 2007) was used. After identifying high quality patches using a resource selection function (RSF), an expected cluster size (ECS) was calculated and a corresponding x-y graph was produced. From this x-y graph, thresholds were identified and then used to assess minimum planar graphs (MPG) at a variety of thresholds. Comparing threshold at MPGs enabled me to identify areas where changes in connectivity have occurred.

Using this series of analyses, I assessed the changes occurring in this population of boreal caribou in terms of distribution and habitat, and quantified changes to this landscape.

1.4 Research Rationale

Any change in the use of protected areas by threatened species is concerning (COSEWIC, 2002). With species at risk, protected areas such as national parks play a role in ensuring the long-term survival of the species. However, woodland caribou occur on large landscapes and cannot be managed solely within parks and protected areas (Armstrong et al., 2000). Park borders are not barriers to movement and, thus, habitat surrounding protected areas is also of great significance (Mosnier et al., 2003). Overall, it is critical to ensure that connectivity exists between the Park and the surrounding areas, thereby allowing caribou to move throughout the landscape. To guide management decisions by federal and provincial government and forestry companies, it is important to quantify historical changes to the landscape and understand the reasons for some of these changes.

In the future, it may be necessary to restore specific areas of the landscape to improve caribou habitat and ensure their survival. In conclusion, understanding changes to the landscape over time can also help to identify areas with potential for restoration.

2. Literature Review

To understand this research, it is important to first provide background information on woodland caribou and landscape changes. Thus, this chapter provides information on woodland caribou, their physical characteristics, diet, habitat requirements, populations throughout the country, their movements and home ranges and finally limiting factors. Next, I discuss the status of woodland caribou in Canada. Then, I review natural and anthropogenic landscape disturbance including the effect these types of disturbance have on caribou. I conclude this chapter with a review of landscape configuration including landscape connectivity and fragmentation and its effect on woodland caribou.

2.1 Biology of Woodland Caribou

2.1.1 General Characteristics

Woodland caribou (*Rangifer tarandus caribou*) are members of the deer family (Cervidae) along with white-tailed deer (*Odocoileus virginianus*), elk (*Cervus elephas*) and moose (*Alces alces*) (Thomas and Gray, 2002). Caribou have large crescent shaped hooves and males average 600 pounds while females average 300 pounds (Burt and Grossenheider, 1998). Caribou have unique antlers in that they have a flattened appearance (Pattie and Hoffman, 1999). In addition, they are the only cervids where both sexes grow antlers (Pattie and Hoffman, 1999). Woodland caribou breed in the fall and, during the rut, males defend a harem of females (Burt and Grossenheider, 1998). Woodland caribou have a gestation period of approximately eight months (Burt and Grossenheider, 1998). They have a low reproductive rate as females generally have one calf per year and don't reproduce until three to four years of age (Burt and Grossenheider, 1998). For young of the year the survival rate is 30-50% and, for calves greater than one year, the survival rate is 5-15% (Thomas and Gray, 2002)

2.1.2 Diet

Lichen is an important part of the diet of woodland caribou; however, their diet varies throughout the year (Burt and Grossenheider, 1998). During the winter, their diet consists primarily of terrestrial and arboreal lichen (Rominger and Oldemeyer, 1990; Rettie et al., 1997; Johnson et al., 2000; Johnson et al., 2001a; Thomas and Gray, 2002; Johnson et al., 2003). Terrestrial lichen is located through the snow by smell (Kinley, 2003) and obtained by cratering - digging through the snow (Johnson et al., 2000; Johnson et al., 2001a; Johnson et al., 2003). When snow thickness is too great caribou utilize arboreal lichen (Johnson et al., 2000; Johnson et al., 2001a; Johnson et al., 2003). Lichen are low in protein but high in carbohydrates, which provides energy during the winter months (Cumming and Beange, 1993). During summer, woodland caribou diets are variable and include sedges, grasses, forbs, shrubs, fungi, moss and lichen (Rettie et al., 1997; Thomas and Gray, 2002).

2.1.3 Habitat

Various studies involving woodland caribou habitat selection at different scales and trends have occurred. At the fine scale, habitat with an abundance of lichen is preferred by woodland caribou (Terry et al., 2000; Coxson and Marsh, 2001; Johnson et al., 2001a; Thomas and Gray, 2002; Mosnier et al., 2003). Coarser scale habitat selection is based on presence of preferred lichen species and heavy lichen loads (Darby and Pruitt, 1984; Chubbs et al., 1993; Bradshaw et al., 1995). Favourable lichen conditions lead caribou to select mature forests and wetland complexes (Thomas and Gray, 2002).

In Alberta, Brown et al. (2000b) and Schneider et al. (2000) found caribou preferred wetland habitat types. Brown et al. (2000b) looked at habitat preference of woodland caribou over a two year period. They determined that wetlands with low tree cover were selected over other types of wetlands. A preference for wetlands was also observed when compared to uplands and anthropogenically altered areas. Schneider et al. (2000) found caribou selected for peatlands. In addition, caribou utilized bogs more frequently than fens throughout the study area. As the Schneider et al. study was conducted at the regional scale, the authors could not discount the use of upland islands within the wetland complexes.

In Manitoba, Brown et al. (2000a) and Hirai (1998) found that caribou utilized wetlands or treed muskeg regularly. Mahoney and Virgl (2003) conducted a study in Newfoundland that compared habitat preference between sexes and ages and found no difference in habitat preference. Differences did occur in habitat preference during different times of year. Mature conifer and scrubland were preferred throughout the year. Use of bogs increased in summer and fall and use of unmanaged stands increased in winter and spring.

In the Saskatchewan boreal plains ecozone, caribou show a preference for wetland complexes and associated black spruce dominated stands (Rettie and Messier, 2000). Rettie and Messier (2000) found differences in preferred habitat types between seasons. They also found avoidance of young coniferous stands and disturbed areas.

Several studies have shown that other variables also affect habitat selection. In winter, snow thickness, hardness, density and sinking depth influence habitat selection, which leads to caribou selecting sites with higher canopy cover (Schaefer and Pruitt, 1991; Schaefer, 1996; Mosnier et al., 2003). Habitat use is more variable in summer, which corresponds to a more diverse diet (Chubbs et al., 1993) and when forage species may be selected based on abundance or conspicuousness (Johnson et al., 2001a). In addition, habitat selection is influenced by woodland caribou's anti-predator strategies, which lead them to select open areas to minimize this predation risk (Bergerud and Page, 1987; Rettie and Messier, 2000).

2.1.4 Populations and Range

Woodland caribou (*Rangifer tarandus caribou*) in Canada occur in five populations: Atlantic-Gaspésie, Southern Mountain, Northern Mountain, Boreal and Newfoundland (COSEWIC, 2002). Boreal caribou occur across much of Canada in seven provinces and one territory (Thomas and Gray, 2002).

2.1.5 Movements

Several factors influence animal movements including reproduction, predation and habitat structure (Brown et al., 2003). Compared to other boreal ungulates, such as moose, elk and deer, woodland caribou have higher movement rates and seasonal migrations (Johnson et al., 2002). Although woodland caribou are not as gregarious as barren-ground caribou, they do form groups of 3-10 animals (Thomas and Gray, 2002). Adult bulls are solitary except during the breeding season (Pattie and Hoffman, 1999).

Some groups of woodland caribou show seasonal migrations while others remain in their home ranges throughout the year (Cumming and Beange, 1987). During the winter, movements are usually due to snow conditions (Brown et al., 2003). Brown et al. (2003) found that movement rates of a migratory woodland caribou group in northwestern Ontario increased during the fall and late winter. Their results also showed that female caribou migrated between summer and winter ranges with a mean distance between the centers of these two areas of 54.3 km. Ferguson and Elkie (2004) also studied movement rates in northwestern Ontario and delineated five seasons based on these rates; late winter, spring, calving, post-calving and early winter. They found the greatest movement rates in early winter (2.5km/day) and spring (1.8 km/day), and attributed these increased movement rates to migration between summer and winter ranges. They found distance between the center of their winter and summer home ranges to be 15.7 km. Poole et al. (2000) studied a migratory woodland caribou group in British Columbia and found the greatest movement rates during the summer months and lowest movement rates in the winter months. This study also determined the distance between summer and winter ranges to be <15km. In Labrador, Bergman et al. (2000) found an average summer movement rate to be 1.1km/day with increases in fall and spring resulting in an annual travel rate of 2.1km/day.

A non-migratory woodland caribou herd in Saskatchewan was studied by Rettie and Messier (2001) and they delineated five seasons based on movement rates. They observed highest movement rates during autumn, low movement rates in spring and their data indicated that the population was non-migratory. They hypothesized that the movements of this population are minimal as a predator avoidance technique. Using this same group of caribou, with a different set of telemetry data, Dyke (2008) identified 8 seasons based on movement rates- late winter, spring, calving, early summer, late summer, fall, rut, early winter. They found the lowest travel rates during calving (2.04 km/day) and the highest travel rate in early winter (4.08 km/hr).

2.1.6 Home Ranges

The most commonly used definition of home range is that stated by Burt (1943) "the area traversed by the individual in its normal activities of food gathering, mating and caring for young". General factors influencing home range size include reproduction, forage and habitat requirements (Brown et al., 2003). Factors specific to ungulate home range size include population size, temperature, insects, snow thickness, cover type availability, predation and anthropogenic disturbance (Edge et al., 1985; Downes et al., 1986; Sweaner and Sandegren, 1989; Kilpatrick and Lima, 1999; Kilpatrick et al., 2001).

Various studies have been conducted regarding the size of woodland caribou home ranges. Brown et al. (2000a) used 100% minimum convex polygons (MCP) and showed a population near Wabowden, Manitoba had an overall population range of 4600 km^2. In the winter and fall range size was 3200 km^2, in the summer the range was 2500 km^2 and in the spring a range of 1700 km^2 was observed. Brown et al. (2000a) also found average individual home range sizes in summer of 83 km^2 and year round range of 521 km^2. Brown et al. (2003) also used 100% MCP's and found an average annual home range size 4026 km^2 in northwestern Ontario. They found an individual annual home range size of 593km^2 to 5985km^2 over a study period of 3 years. In British Columbia, Poole et al. (2000) used 100% MCP's and found average annual home range sizes of 151 km^2. Arsenault and Manseau (2009) compared the current range sizes of caribou in Saskatchewan to Rettie and Messier's study (2000) which took place in the 1990's. Both studies used 100% MCPs. Arsenault and Manseau (2009) found range sizes have decreased significantly in the past 10 years.

2.2 Limiting Factor

Although forage availability, snow conditions, insect harassment and disease have been proposed as factors limiting woodland caribou populations (Rettie and Messier, 2000), predation is the currently accepted primary limiting factor (Bergerud and Elliot, 1986; Bergerud, 1988; Post et al., 2003; Wittmer et al., 2005). Grizzly bears, black bears and eagles are all reported to predate upon caribou adults and calves (Mahoney and Virgl, 2003, Valkenburg et al., 2004) but wolves are generally considered be the major predator (Bergerud, 1988; Rettie and Messier, 2000; Whittington et al., 2005; Wittmer et al., 2005). To decrease predation risk, boreal caribou have adapted various anti-predator strategies (Bergerud and Page, 1987). They occur at low densities on the landscape, which leads to increases in predator search time thus decreasing the benefit to predators (Cumming and Beange, 1993; Bergerud and Page, 1987). Caribou also actively avoid areas where other ungulate densities are high (Bergerud and Page, 1987; James et al., 2004; Whittington et al., 2005) as these areas have increased predator densities.

2.3 Status of Woodland Caribou

Within Canada, the population of boreal caribou is estimated at 33,000 over 64 populations (Thomas and Gray, 2002). The boreal population of woodland caribou was listed as a threatened species by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) in 2000 and Species at Risk Act in 2002. COSEWIC (2002) defines threatened as "a species likely to become endangered if limiting factors are not reversed". The listing of boreal woodland caribou was based primarily on range retraction and overall population decline (COSEWIC, 2002). Woodland caribou are not yet listed provincially as a Species at Risk.

2.4 Disturbance

2.4.1 Fire

In the boreal forest, fire is a source of natural disturbance and has long term benefits to the landscape (Bergeron, 1991; Klein, 1992; Johnson et al., 2001b). After a forest reaches a certain level of maturation, its productivity decreases and fire becomes necessary to regenerate plant growth (Klein, 1992; Johnson et al., 2001b). The landscape following fire is favourable for species with a preference for early successional forests, such as moose and deer (Schaefer and Pruitt, 1991; Klein, 1992).Woodland caribou's preference for mature forests means burned areas are avoided for 50 years or more following a fire (Schaefer and Pruitt, 1991; Klein, 1992; Thomas and Gray, 2002). Preventing fire (fire suppression) can however be an issue as it may lead to large tracts of old growth forest that, when burned, create huge areas of habitat inappropriate for caribou use (Schaefer, 1988). Woodland caribou can co-exist with fire if suitable habitat is available in adjacent areas (Schaefer, 1988; Schaefer and Pruitt, 1991).

2.4.2 Anthropogenic

Anthropogenic disturbances such as logging activities (clear-cutting) and the construction of linear features (e.g. roads and transmission lines) have significant negative effects on caribou and their habitat (Ferguson et al., 1988; Cumming and Beange, 1993; Rettie and Messier, 1998; Rettie and Messier, 2000; Dyer et al., 2002; Thomas and Gray, 2002; Cameron et al., 2005; Whittington et al., 2005; Wittmer et al., 2005; Lander, 2006). Logging has direct consequences for woodland caribou. Lichen, an important food source for caribou, depends on old growth forests (Mosnier et al., 2003). Logging disturbs the forest floor and removes trees thus destroying terrestrial and arboreal lichen (Johnson et al., 2003). In addition, herbicides are sometimes used as a treatment following harvest which may destroy any remaining lichen (Cumming and Beange, 1993).

Logging affects the ability of caribou to access food resources (Johnson et al., 2003). Decreased canopy cover leads to increases in snow depth which makes terrestrial lichen more difficult to access (Brown and Theberge, 1990; Johnson et al., 2000). Woodland caribou are proficient at cratering (digging through the snow with hooves and antlers) to reach lichens beneath the snow (Brown and Theberge, 1990; Johnson et al., 2001a) but there are thresholds at which snow depth is too great (Brown and Theberge, 1990). As logging removes canopy closure, snow depth increases which can be detrimental to caribou (Brown and Theberge, 1990; Johnson et al., 2001a).

In general, woodland caribou avoid areas recently logged (Chubbs et al., 1993; Cumming and Beange, 1993; Johnson et al., 2003; Smith et al., 2000; Lander, 2006). In Newfoundland, Chubbs (1993) found that caribou abandoned habitat when logging occurred in proximity. The same situation occurred in Ontario as found by Cumming and Beange (1993). Smith et al. (2000) found woodland caribou in Alberta to avoid recently cut areas as did Lander (2006), in Manitoba. This avoidance may displace caribou into marginal habitat, which will have consequences for their survival (Dyer et al., 2001). Logging leads to early successional forests that attract other ungulate species such as moose, elk, and white-tailed deer (Brown et al., 2000a; Thomas and Gray, 2002; James et al., 2004). This increase in competitor populations is a concern as it decreases their spatial separation from caribou often leading to further increases in predator populations through reliance on these alternate prey species (Bergerud and Elliot, 1986; James et al., 2004). Overlap with other ungulate species, particularly white-tailed deer, may also increase disease transmission (Cumming and Beange, 1993; Pitt and Jordan, 1994; Thomas and Gray, 2002). Brainworm (*Parelaphostrongylus tenuis*) is a parasite carried by white-tailed deer with little affect to their health, however, it is fatal to caribou (Pitt and Jordan, 1994).

Although roads cover a small area, they have major consequences for caribou. Caribou avoid roads (Nellemen and Cameron, 1996; Cameron et al., 2005) as they increase human and predator access to isolated areas (Dyer et al., 2001). This can increase mortality due to increased human activity, such as hunting, poaching, and caribou-vehicle accidents (Cumming and Beange, 1993) and predation by wolves (Dyer et al., 2001). McGarigal et al. (2001) suggested that the effects of roads may last longer than clear cuts. The negative impacts of roads often lead to range retraction (Bradshaw et al., 1997).

2.5 Landscape configuration

Recent studies have shown that the suitability of the landscape is not determined only by the amount of suitable habitat available but also the spatial configuration of areas of suitable habitat, particularly in fragmented landscapes (Ferreras, 2001; Cook et al., 2003; Haynes and Cronin, 2004; O'Brien et al., 2006). The landscape consists of patches, relatively homogenous areas occurring on the landscape (Forman and Godron, 1986). Changes to patch sizes and distribution can have substantial effects on the connectivity of a landscape, which affects the ease with which an organism can move through the landscape (Taylor et al., 1993; Tischendorf and Fahrig, 2000; Van der Ree et al., 2003). Structural connectivity refers to the linkages among adjacent habitat patches (Keitt et al., 1997), while functional connectivity refers to the movements of animals among these habitat patches (With et al., 1997; Goodwin and Fahrig, 2002; Brooks, 2003). Many organisms see the areas surrounding patches as varying in quality and the level of impedance to movement (Gustafson and Gardner, 1996; Ricketts, 2001; Shriver et al., 2004; Whittington et al., 2004). Therefore, the impact of a decrease in connectivity on a species depends on the amount and distribution of habitat patches and the nature of the area surrounding these habitat patches (the matrix) as well as the ability of a particular organism to traverse these areas (With et al., 1999).

Habitat fragmentation is considered one of the greatest threats to wildlife diversity making it an important conservation issue (Forman and Godron, 1986). Habitat fragmentation is generally defined as "the breaking up of a large habitat into smaller, more isolated, patches" (Andrén, 1994; Fahrig, 1997). Habitat fragmentation is highly influenced by human land use and, therefore, leads to significant changes in the landscape (Andrén, 1994). Human caused fragmentation as described by Forman and Godron (1986) occurs in four stages. The first stage, dissection, occurs through the building of linear features, such as roads. The second stage, perforation, occurs when holes in the landscape are created by landscape conversion, such as logging activities. Thirdly, fragmentation occurs as these converted areas merge. Finally, as the natural patches become smaller in size and further apart, the last stage, attrition, occurs.

In general, as habitat fragmentation increases, connectivity decreases (Forman and Godron, 1986). In addition to causing a loss of native habitats, habitat fragmentation leads to an increase in the number of patches and a decrease in patch size (Andrén, 1994; Fahrig, 1997), which ultimately can isolate populations from one another (Bélisle and Desrochers, 2002). Fall et al. (2007) used spatial graph theory to assess the connectivity of the landscape for woodland caribou. O'Brien et al. (2006) investigated the importance of the overall habitat mosaic (spatial distribution of habitat and non-habitat patches) and found that woodland caribou were associated with large clusters of high quality habitat likely to allow for predator avoidance and adequate forage. In conclusion, maintaining the connectivity of a landscape by minimizing habitat fragmentation helps to maintain biodiversity and prevent local extinction (Forman and Godron, 1986).

2.6 Summary of Literature Review

Woodland caribou are adapted to living in mature forests where they exist at low densities to maintain spatial separation from other boreal ungulates and their primary predators, wolves. Boreal caribou are sensitive to disturbance both natural and anthropogenic. Changes to the landscape may be detrimental to caribou as they prefer connected areas with low fragmentation levels. The boreal ecotype of woodland caribou is listed as a threatened species by the Species at Risk Act and their survival is important to maintaining biodiversity throughout the country.

3. Historical changes in caribou distribution and land cover in and around Prince Albert National Park: land management implications

3.1 Introduction

Human land use through settlement, recreation or industrial development may cause habitat fragmentation leading to significant changes in the landscape. Habitat fragmentation is generally defined as "the breaking up of a large habitat into smaller, more isolated, patches" (Andrén, 1994; Fahrig, 1997). Habitat patches are part of the landscape and the use of a patch by wildlife is not only a function of the patch attributes but also of the characteristics of neighboring patches (Andrén, 1994; Fahrig, 1997). In highly fragmented landscapes, the decline of wildlife populations is greater than that expected by habitat loss alone (Andrén, 1994) and ultimately, these changes to the landscape can isolate groups of animals (Bélisle and Desrochers, 2002). Habitat fragmentation is considered one of the greatest threats to biodiversity making it an important conservation issue (Harris, 1984; Forman and Godron, 1986; Saunders et al., 1991).

In the boreal forest, the main factors leading to habitat loss and habitat fragmentation are: changes in natural and anthropogenic disturbance patterns, increased commercial and industrial activities, increased road access to remote areas and recreational activities (Harris, 1984; Forman and Godron, 1986). Fire is a natural disturbance and has long-term ecological benefits (Klein, 1982; Bergeron, 1991; Johnson et al., 2001b). In the boreal mixedwood forest of North America, the fire return interval ranges from 30 to 150 years (Johnson, 1992). Changes in fire frequency can be caused by shifts in climate, land use pattern and land management strategies (Clark, 1988; Bergeron, 1991; Johnson and Larsen, 1991; Larsen, 1997). At the time of human settlement, fires were frequent as deliberate burns were set to clear land for agricultural purposes (Williams, 1989; Whitney, 1994; Weir, 1996). After an area is settled, fire frequency tends to decrease as forested areas become fragmented and cannot support the spread of fire (Weir, 1996).

Following settlement of the boreal forest, roads were constructed to provide access for industrial development, primarily forestry (Walker, 1999). Forest harvesting is an important commercial activity across the boreal forest and usually targets coniferous stands older than 50 years (Walker, 1999). To be sustainable, logging practices attempt to maintain stands of a variety of ages within the forest management area (Walker, 1999). In Saskatchewan, fire is suppressed over areas of commercial forest tenures or in proximity to communities; natural forest pattern standards and guidelines for the forest industry aim to produce landscapes and harvest areas that emulate the patterns created by fire (Saskatchewan Environment, 2009). Areas managed for logging are however still affected by fire so it may be difficult to retain stand composition that is comparable to a natural disturbance regime (Walker, 1999). Occurrences of fire on a landscape where logging activities are prevalent often lead to a young age structure (Reed and Errico, 1986).

Landscape changes, natural or anthropogenic, have significant impacts on the boreal population of woodland caribou (*Rangifer tarandus caribou*), a threatened species under the Species at Risk Act (2004). Boreal caribou are habitat specialists, dependent on old growth forests for survival (Rettie and Messier, 2000; Smith et al., 2000; Mahoney and Virgl, 2003). Due to this habitat specialization, natural or anthropogenic disturbance can be detrimental to caribou (Thomas and Gray, 2002). Due to increased abundance of other ungulate species (moose, deer and elk) and associated predators in stands of younger age classes, boreal caribou tend to avoid logged areas (Chubbs et al., 1993; Cumming and Beange, 1987; Smith et al., 2000; Johnson and Gilligham, 2002; Lander, 2006) and areas near roads and trails (Cameron et al., 2005; Nellemen and Cameron, 1996). Caribou also tend to avoid recent burns (Schaefer and Pruitt, 1991; Klein, 1992; Thomas and Gray, 2002; Lander, 2006). Caribou have persisted in the boreal forest for thousands of years in the presence of fire, provided suitable habitat is available in adjacent areas (Schaefer, 1996; Schaefer and Pruitt, 1991). Logging and road development also often displace caribou (Chubbs et al., 1993; Dyer et al., 2001) and since these activities lead to more permanent landscape changes, they can result in range retraction (Bradshaw et al., 1997; Thomas and Gray, 2002).

The Prince Albert National Park (PANP) and Greater Ecosystem are located in the boreal mixedwood forests of Canada, in the province of Saskatchewan and form a part of the Smoothstone-Wapaweka Woodland Caribou Management Unit (SW-WCMU). The fire frequency of this area has decreased following settlement (Johnson, 1992; Weir et al., 2000) and, in the past 40 years, significant logging and road development surrounding the Park has occurred. This ecosystem has traditionally been used by a resident population of boreal caribou (Banfield, 1961) but there are concerns over the long-term viability of the population (Arsenault, 2003; Saskatchewan Environment, 2007). In central Saskatchewan, population declines have been documented in the 1940s and again in the 1980s. The first decline led to a ban in sport hunting and an increase in caribou population in the 1950s was attributed to wolf control and hunting closure (Rock, 1988; Rock, 1992). In 1987, another population decline was confirmed and sport hunting was again banned (Rock, 1988; Rock, 1992). Subsistence harvesting still occurs, although only opportunistically (Trottier, 1988). Work conducted by the University of Saskatchewan (Rettie and Messier, 1998) and more recently through a collaborative effort between Parks Canada, Saskatchewan Environment, the Prince Albert Model Forest, Weyerhaeuser Canada Ltd. and the University of Manitoba (Arsenault and Manseau, 2009) suggests that the population may not be increasing. The Park and surrounding area are managed separately and under different legislation. The management of the National Park centres on the maintenance or restoration of ecological integrity while also providing opportunities for public education and enjoyment (Parks Canada, 1986). Logging has not been permitted within the Park in the past 60 years and, fire has been suppressed; however, a prescribed burning program has been put in place to reinstate a natural fire cycle (Prince Albert National Park, 2008). The area outside of the National Park is managed primarily for forestry purposes by Saskatchewan Ministry of Environment (MoE) (Government of Saskatchewan, 2002). Waskesiu is a community located on the east side of the National Park and Montreal Lake First Nation is situated to the east of the National Park, at the southern tip of Montreal Lake.

Our main objectives were to assess changes in caribou distribution and landscape composition over a period of 40 years, between 1966 and 2006. We predicted an ageing landscape for the Park area and significant habitat change due to commercial forestry activities on the provincial crown land portion of the Prince Albert Greater Ecosystem.

We hope that an historical representation of the landscape will assist in the recovery efforts and guide current and future forestry management and land-use planning activities.

3.2 Methods

3.2.1 Study Area

The Prince Albert Greater Ecosystem (PAGE) is a 20,000 km^2 area located in central Saskatchewan, Canada (Figure 3-1). Prince Albert National Park was established in 1927 to represent the southern boreal forest region of Canada. The portion of the Park within the PAGE, is a 2,688 km^2 area. The remaining part of the PAGE is provincial crown land. This includes the communities of Weyakwin and Waskesiu, the reserve community of Montreal Lake First Nation, Ramsey Bay Subdivision on Weyakwin Lake, and a few private properties. The dominant tree species are white spruce (*Picea glauca*), black spruce (*Picea mariana*), trembling aspen (*Populus tremuloides*), and jack pine (*Pinus banksiana*) (Ecological Stratification Working Group, 1998) and the main commercial activities are forestry, trapping and outfitting. A variety of recreational pursuits also occur (snowmobiling, fishing, hunting, trapping, and cross-country skiing).

Historically, when fires started in the National Park they were extinguished before much of the landscape burned. In recent years, controlled burns and clearing has been initiated to create a fire barrier along the Park boundaries with the objective of letting non-threatening fires burn in the Park and restoring the natural fire frequency (Prince Albert National Park, 2008). Although there is no overall land use plan for the provincial crown land area surrounding the National Park, the Saskatchewan Provincial Government manages the area for various industrial activities such as mining and forestry

(Government of Saskatchewan, 2002). Every five years a forest management plan is produced for the area surrounding the Park (Government of Saskatchewan, 2002) and the Park produces its own management plan, both with significant public consultation. The Prince Albert Model Forest was established in 1992 and has conducted significant research to guide land management and develop ideas and solutions for community sustainability (Prince Albert Model Forest, 2008).

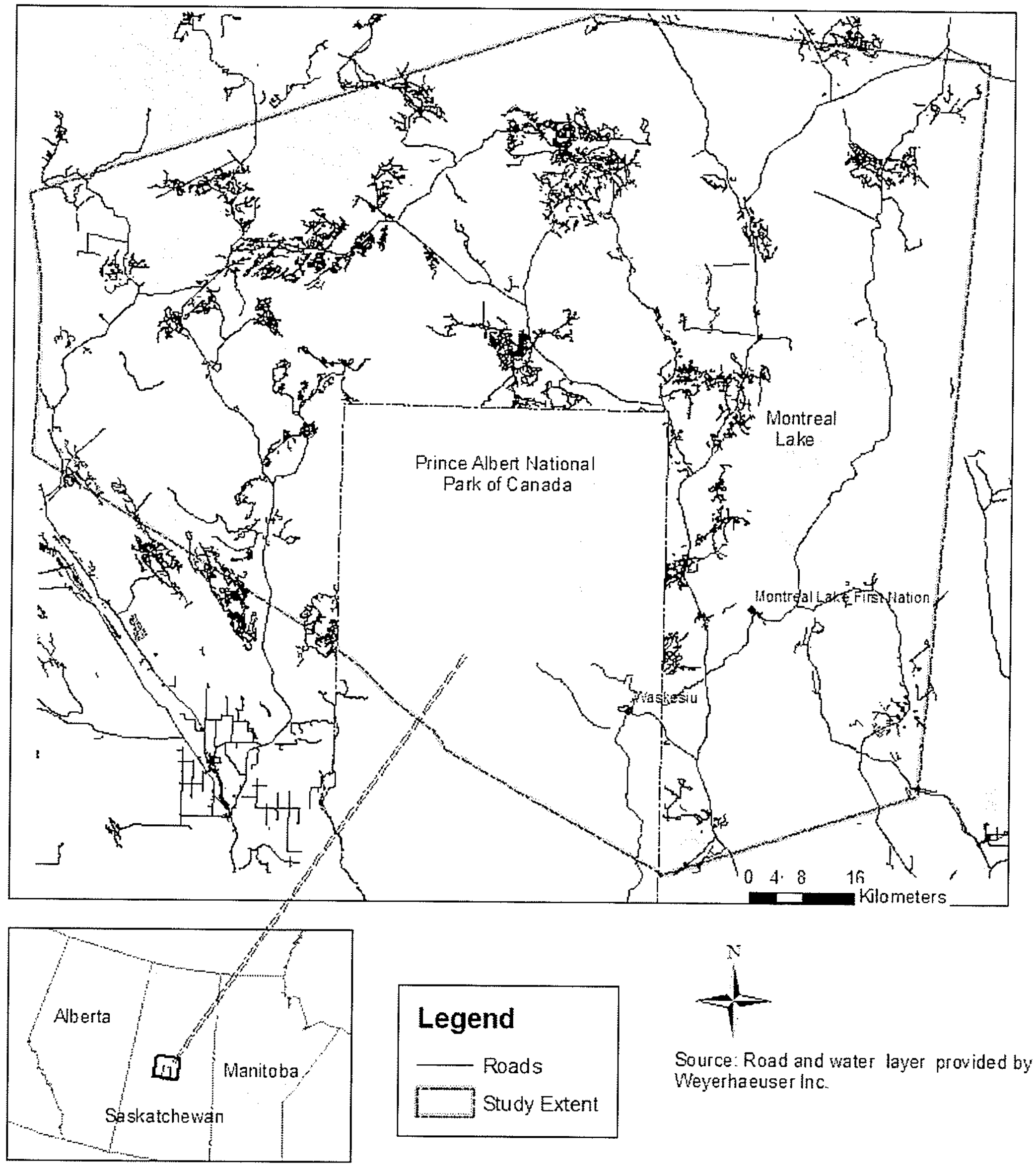


Figure 3-1. Prince Albert Greater Ecosystem, Saskatchewan

3.2.2 Smoothstone-Wapaweka Woodland Caribou Management Unit

Arsenault (2003, 2005) has defined seven Woodland Caribou Management Units (WCMUs) within the Province based on clusters of caribou observations, areas of similar ecological characteristics (Acton et al., 1998) and peatland distribution. The PAGE is part of the Smoothstone-Wapaweka WCMU that comprises an estimated 350 animals (Arsenault, 2003). This herd is considered high risk due to anthropogenic activity (logging and road development) and habitat loss (Arsenault, 2005).

3.2.3 Caribou Past and Present Distribution

In order to examine changes in caribou distribution over time, woodland caribou occurrence data and associated survey efforts were collated for the period of 1950 to present. Data were obtained from Parks Canada and Saskatchewan Ministry of Environment and primarily consisted of survey observations, incidental sightings and telemetry data.

3.2.4 Landscape Reconstruction

Map layers for the National Park and provincial crown land portion of the PAGE were created separately since the type and extent of data available for the two areas differed. Although we tried to create seamless layers for the PAGE area, map resolution issues could not be resolved and prevented us from directly comparing landscape changes between the two areas. For both the Park and the provincial crown land portion of the PAGE, we created map layers for 1966 and 2006 to assess historical landscape changes.

For the National Park area, the map layers consisted of a vegetation layer based on aerial photos taken in the 1960's (Parks Canada, 1986), a road layer and a burn polygon layer produced by Parks Canada, and a time since fire map produced by Weir (1996). Since the time since fire map was based on data collected in the 1990's, we subtracted 30 years from each forest stand to obtain a stand age for the 1966 layer. For the 2006 layer, we used the same vegetation type as for the 1966 layer (we did not account for forest succession) and added 10 years to the stand ages obtained from Weir (1996) and the time since fire map. To account for natural disturbances that occurred in the past 10 years, after the creation of the time since fire map, we overlaid the burn polygon layer and assigned a recent burn class to all forest stands that fell under those polygons.

For the provincial crown land portion of the PAGE, we used the most recent forest resource inventory (FRI), a road and a cut block layer developed by Weyerhaeuser Canada Ltd. and a burn layer from the Province. The FRI was based on aerial photos from 2004 and for each forest stand, it contains a large number of attributes including cover types (species, height and density), soil types, topography, history of disturbance and stand age. For the current layer, we used the data layers provided by Weyerhaeuser Canada Ltd. Since a burn class was not available in the FRI, we overlaid the burn polygon layer and assigned a recent burn class to all forest stands that fell under those polygons if the year of origin corresponded to the year of the fire ± 5 years. The cut block layer lacked a harvest year or a stand age for a number of polygons. To determine those stand ages, we sampled 10% (142 polygons) of the cut block polygons lacking a harvest year and estimated age by using ring counts on increment cores (Cook, 1990). Cut block polygons that were not sampled were assigned an age based on proximity to sampled cut block polygons, on the assumption that stands in a general area were harvested at approximately the same time. For the 1966 layer, we subtracted 40 years from the stand age. Since the FRI was current, stand composition and stand age prior to fire was not available. To obtain this information, we used older provincial FRI and hard copy maps from the 1960s. We digitized and georeferenced the maps and manually entered the composition and age of forest stands that burned over the 40 years.

To prepare the map layers for analyses, we reclassified the vegetation layers using a simplified classification scheme often used in the production of woodland caribou habitat maps (Rettie et al., 1997). Vegetation classes of similar composition were combined to produce 7 habitat classes (Table 3-1). Each map layer was rasterized at a 100 m grid and filtered using Spatially Explicit Landscape Event Simulator (SELES; Fall and Fall, 2001) to remove patches of less than 2 ha. Patches of this size are smaller than the minimum mapping unit and are often artifacts from the vector to raster conversion.

Table 3-1. Habitat classes used in the mapping and analyses of the provincial crown land and National Park portion of the PAGE.

Habitat Class	Provincial Crown Land - Land Cover	National Park Land Cover	Age (years)
Mature Coniferous	Jack Pine Mature	Jack Pine Mature	≥40
Mature Coniferous	Jack Pine/Black Spruce Mature	Jack Pine/Black Spruce Mature	≥40
Mature Coniferous	Black Spruce Mature	Black Spruce Mature	≥40
Mature Coniferous	White Spruce Mature	White Spruce Mature	≥40
Mature Coniferous	Coniferous Mixedwood Mature	Coniferous Mixedwood Mature	≥40
Treed Muskeg	Brushland	Brushland	na
Treed Muskeg	Closed Treed Muskeg	na	na
Treed Muskeg	Black Spruce/Larch	Black Spruce/Larch	All Ages
Treed Muskeg	Open Treed Muskeg	na	na
Treed Muskeg	Open Muskeg	na	na
Treed Muskeg	Fen, marsh, bog	Meadow, marsh, bog	na
Hardwood Mixedwood	Hardwood Mixedwood	Hardwood Mixedwood, Aspen Mixedwood	All Ages
Hardwood Mixedwood	Hardwood	Hardwood	All Ages
Coniferous Young/Recent Burn	Coniferous Young	Coniferous Young	<40
Coniferous Young/Recent Burn	Recent Burn	Recent Burn	<40
Recent Logged	Recent Logged	na	<40
Road	Road	Road	na
Water	Water	Water	na

3.2.4 Validation

For the 1966 map produced for the provincial crown land portion of the PAGE, we performed a validation of the resulting habitat types to assess accuracy. Forest

Resource Inventory maps from the 1960s were used in the analysis and 7451 systematically distributed points were generated in ArcGIS 9.2 (Environmental Systems Research Institute, 2006) using the Hawth's tools extension (Beyer, 2004). Stand attributes were derived for each point and compared. The results indicated that more than 70% of the points on the 1960 generated map corresponded to the classes extracted from the 1960 hard copy maps. This overall accuracy level is above the accepted standard of 70% (Burnside et al., 2003). Accuracy levels of 72% were obtained for coniferous mature and 84% for coniferous young and recent burns. Some of the differences may be attributed to different classification schemes, differences in map resolution or differences in the boundaries drawn (limits of the polygons) for each forest stand.

3.2.5 Transition Probabilities Analyses

Transition probabilities measure the likelihood of one habitat type transitioning into another within a given time period (Burnside et al., 2003). We calculated the transition probability of each habitat class between 1966 and 2006 by quantifying changes of each pixel in the two layers using SELES (Program written by A. Fall).

3.2.6 Landscape Composition and Configuration

Landscape metrics are commonly used when assessing fragmentation (e.g. Hargis et al., 1998; Southworth et al., 2002; Burnside et al., 2003; and Jackson et al., 2005). Total area, patch number, area-weighted mean patch size, mean nearest neighbor, mean shape index and amount of linear features were computed for each habitat type on the 1966 and 2006 map layers for the National Park and provincial crown land portions of the PAGE using Fragstats (McGarigal and Marks, 1995). Differences in landscape metrics between 1966 and 2006 were tested for statistical significance using t-tests in SAS 9.1 (SAS Institute Inc., 2003).

To further assess changes in landscape configuration, we used results from resource selection analyses presented in Dyke (2008) that pointed to the importance of neighboring habitat types in the selection of a given habitat patch. Dyke (2008) documented a greater selection of mature coniferous and treed muskegs away from avoided habitat types such as hardwood/mixedwood. We used ArcGIS 9.2 to measure distances from a source patch, either coniferous mature or treed muskeg, to the nearest hardwood/mixedwood patch. We tested for statistical differences of these distance metrics using t-tests in SAS (SAS Institute Inc. 2003).

3.3 Results

3.3.1 Caribou Past and Present Distribution

Although the survey efforts varied greatly between decades (particularly on provincial crown land), our results indicate that the extent of caribou use of the National Park portion of the PAGE has changed over the last 50 years, with very limited use detected since the 1980s (Figure 3-2). Despite multiple surveys conducted throughout the Park in recent years and large radio-collaring programs, only one observation was made over the last 14 years, in 2007. Caribou are still present over most of the provincial crown land portion of the PAGE despite their low density and clustered distribution (see below).

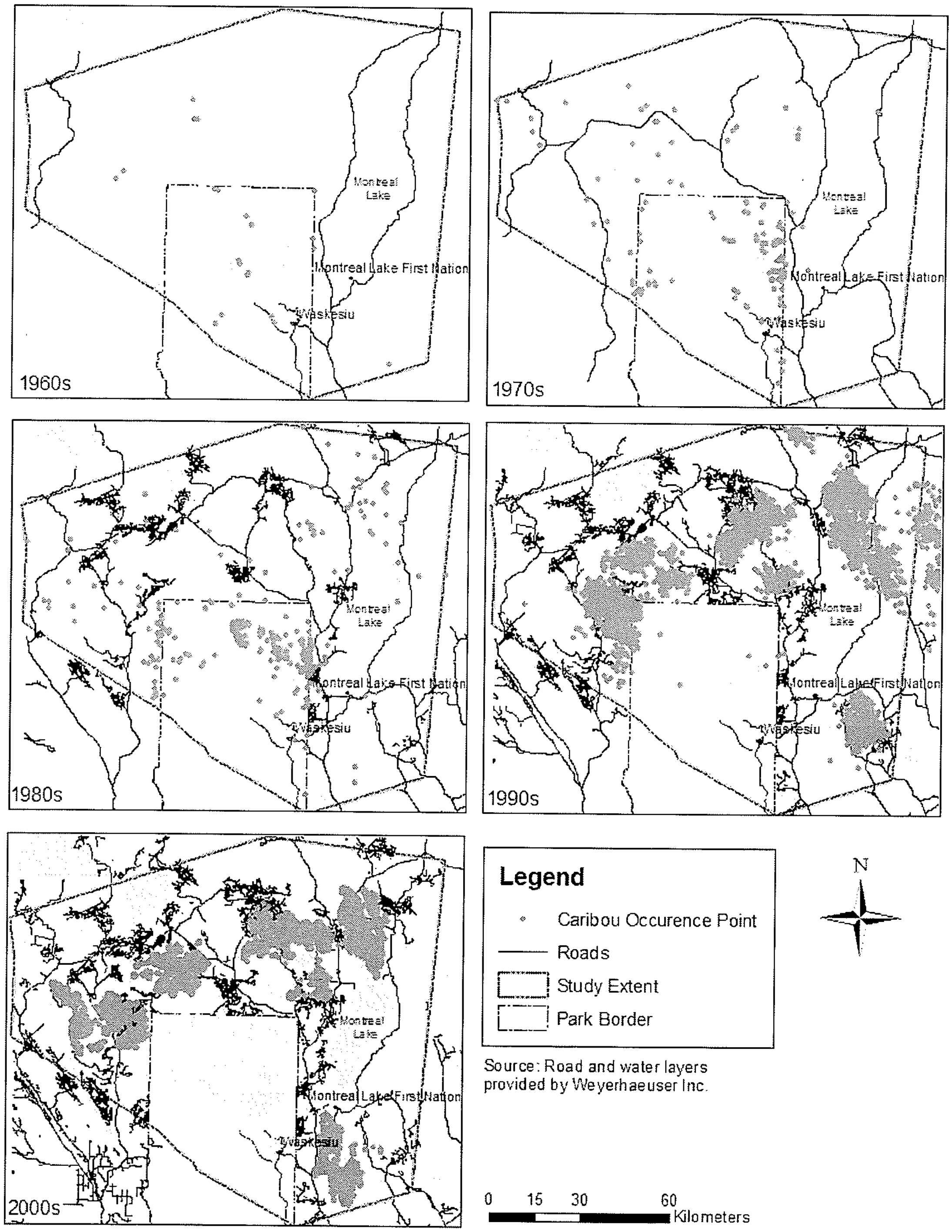
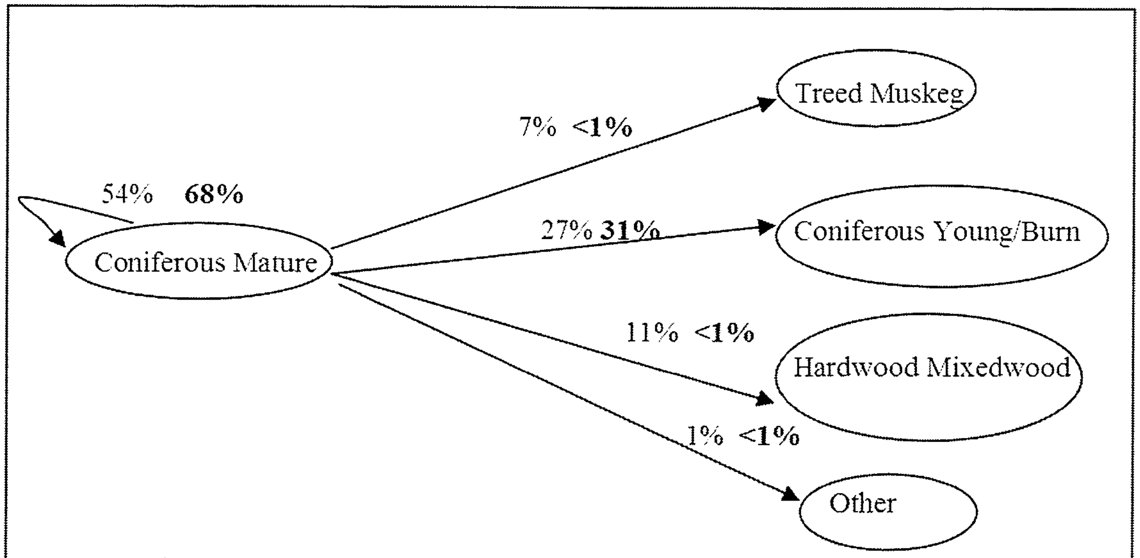


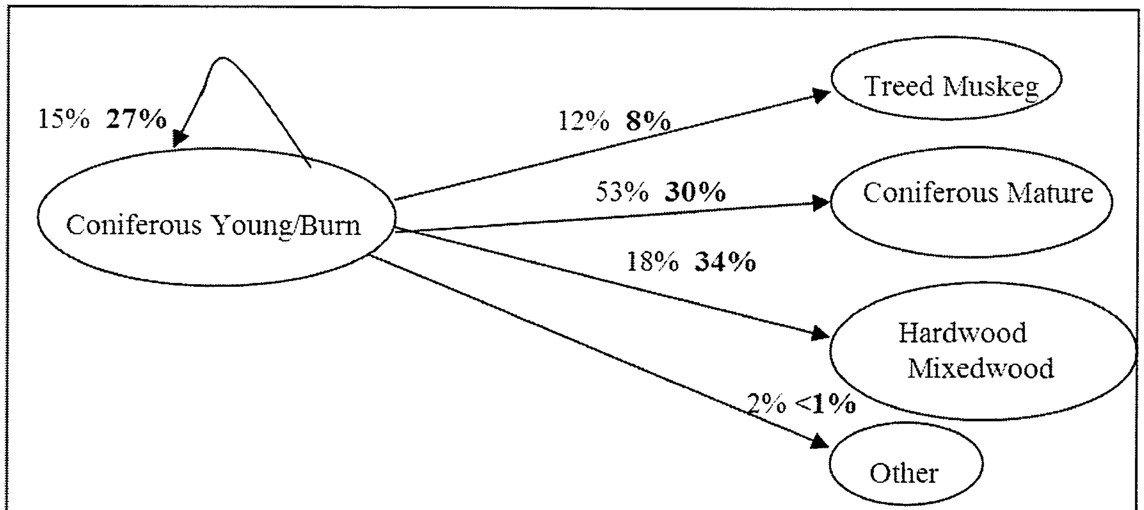
Figure 3-2. Compilation of boreal caribou occurrences in the Prince Albert National Park for the period of 1960 to the present.

3.3.2 Transition Probabilities

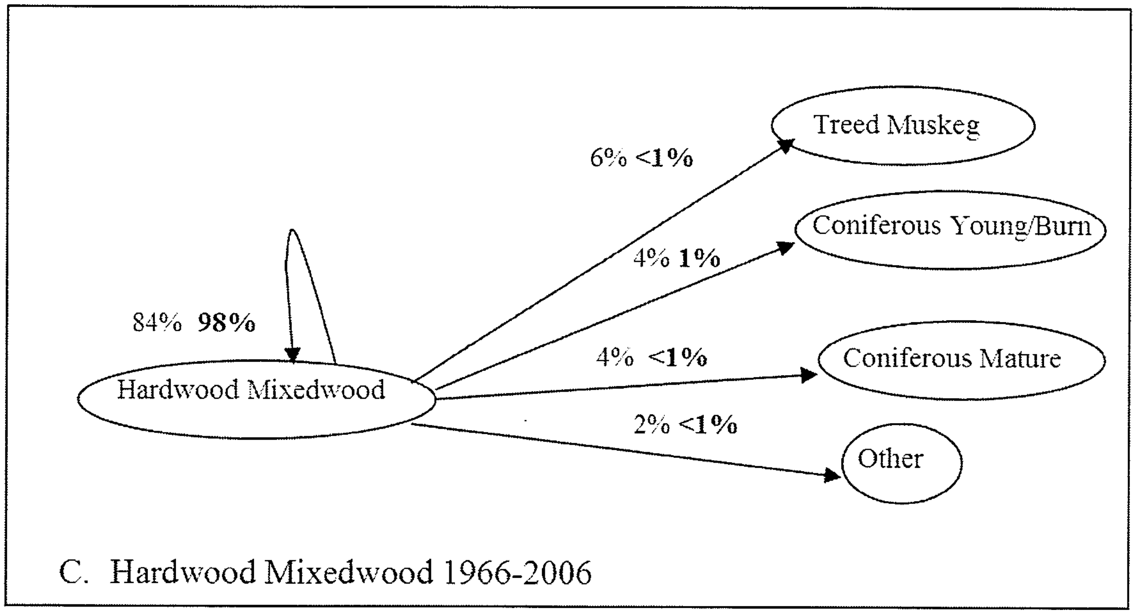
Transition probabilities showed similar trends in the National Park and provincial crown land portion of the PAGE. The most notable changes were with forest stands in the coniferous mature and coniferous young/burn classes (Figure 3-3 A and B). Less than 27% of the coniferous young/burn class remained in that class. A large portion of these stands aged to coniferous mature or to hardwood/mixedwood; the transition to a hardwood/mixedwood class was higher for the National Park area. Fifty four percent of National Park land and 68% of provincial crown land remained in the coniferous mature class. A substantial portion of land within the PAGE as a whole also transitioned to coniferous young/recent burn class. Of all habitat types, hardwood mixedwood and treed muskeg had the highest probability of remaining as the same habitat type (Figure 3-3 C and D). For hardwood/mixedwood, 84% on provincial crown land and 98% on National Park land remained in the same class between 1966 and 2006. Similarly, 86% of treed muskegs on provincial crown land and 99% in the Park area remained treed muskegs.



A. Coniferous Mature 1966-2006



B. Coniferous Young/Burn 1966-2006



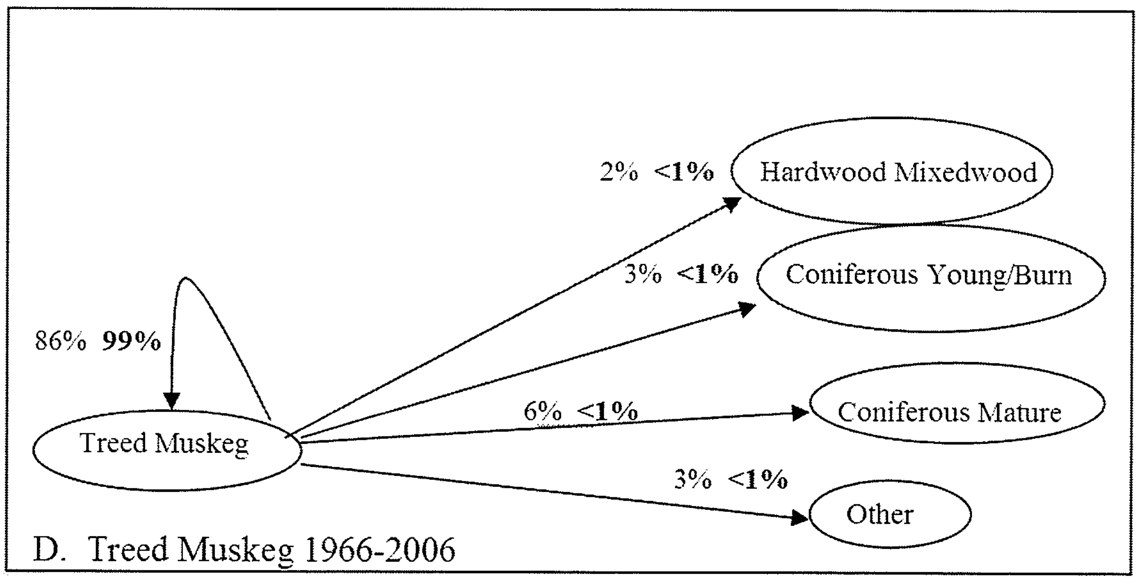


Figure 3-3. Habitat transition probabilities between 1966 and 2006 for the Provincial crown land (normal font) and National Park portions (bold font) of the Prince Albert Greater Ecosystem. The main habitat types consisted of coniferous mature (A), coniferous young and burn (B), hardwood mixedwood (C) and treed muskeg (D).

3.3.3 Landscape Changes

The predominant change to older aged stands suggests an ageing forest over the PAGE landscape as a whole. However, the transitioning of large tracts of crown land in the PAGE to coniferous young/burn stands is coincident with a noticeable increase in the number of cut blocks and the development of road and trails network (Figure 3-4, 3-5 and 3-6). The first mill was built in 1966 and the amount of area logged increased from 0 ha logged in 1966 to 58211 ha logged in 2006. The road network remained the same in the National Park but increased 14-fold on the provincial crown land portion of the PAGE, from 342 km to 4730 km over the same 40-year period. A majority of changes to roads resulted from building of logging roads to access cut blocks. Seventy five percent of the roads on the landscape in the present were constructed primarily for logging purposes. Major highways were also constructed to improve access to the communities of La Ronge, Montreal Lake First Nation, Sled Lake, and Dore Lake. In addition, highways and logging roads were built as travel corridors to the pulp mills at Prince Albert, to the south, and saw mills at Big River and Nipawin, to the southwest and southeast, respectively. Finally, land was converted from forest to commercial/residential with the moving of Molanosa residents from the east side to the west side of Montreal Lake and formation of a new community, Weyakwin. This change was further augmented with expansion of residential areas on reserve lands of Montreal Lake First Nation and the Lac La Ronge Indian Band, and with development of the Ramsey Bay subdivision at Weyakwin Lake.

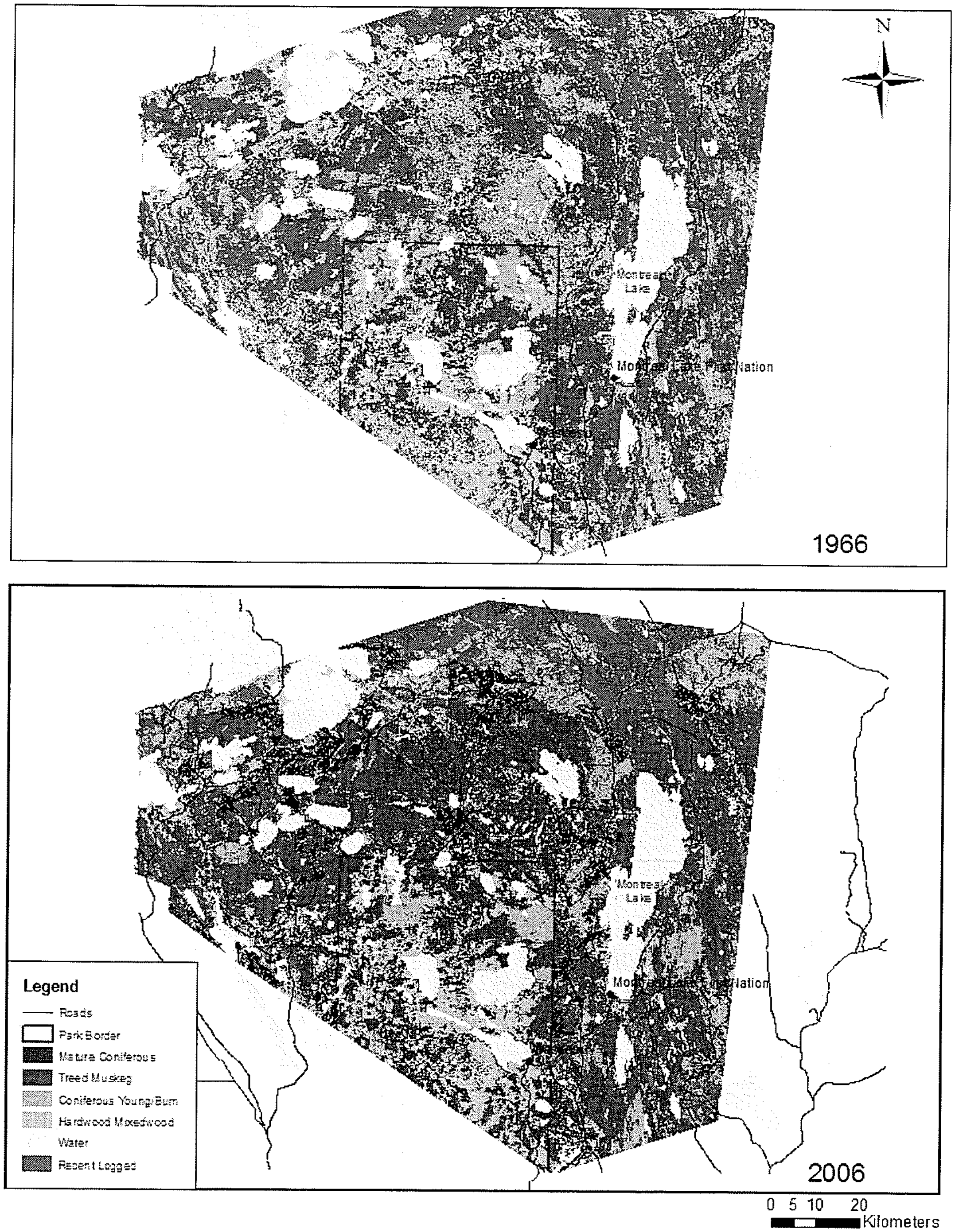


Figure 3-4. Landcover, natural and anthropogenic disturbances in the Prince Albert Greater Ecosystem in 1966 and 2006.

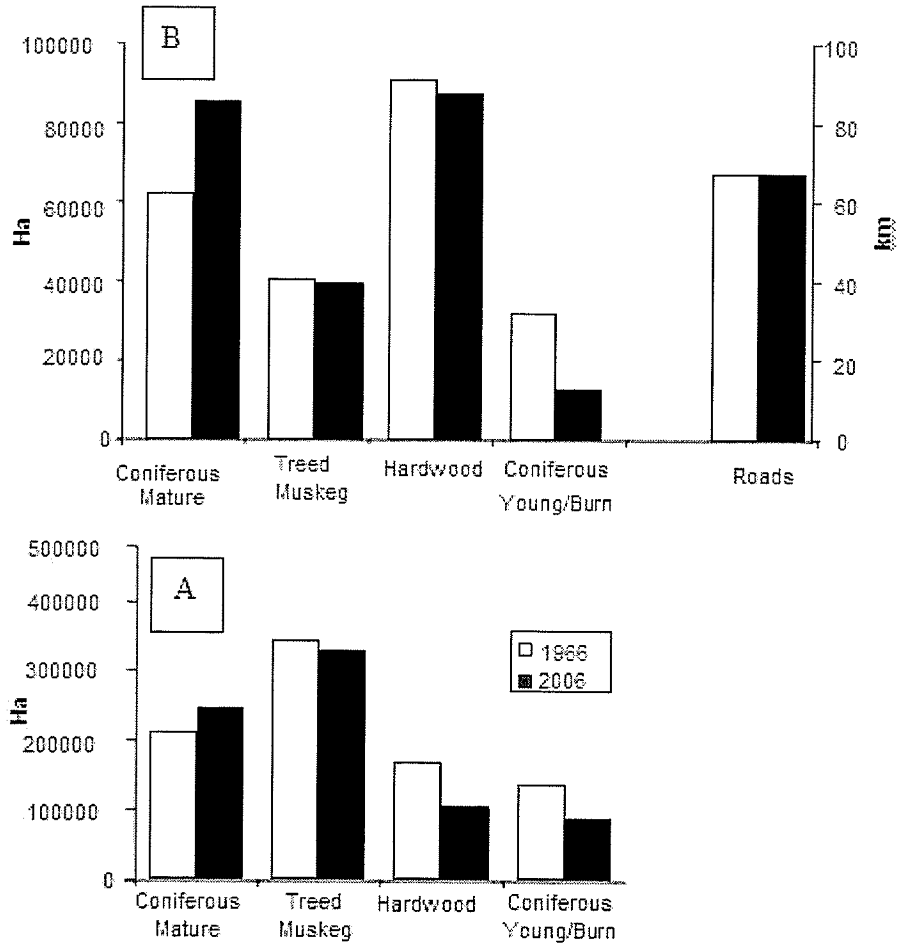


Figure 3-5. Area covered by the main habitat types and linear features in 1966 and 2006 on the Provincial Crown Land (A) and National Park (B) portions of the Prince Albert Greater Ecosystem.

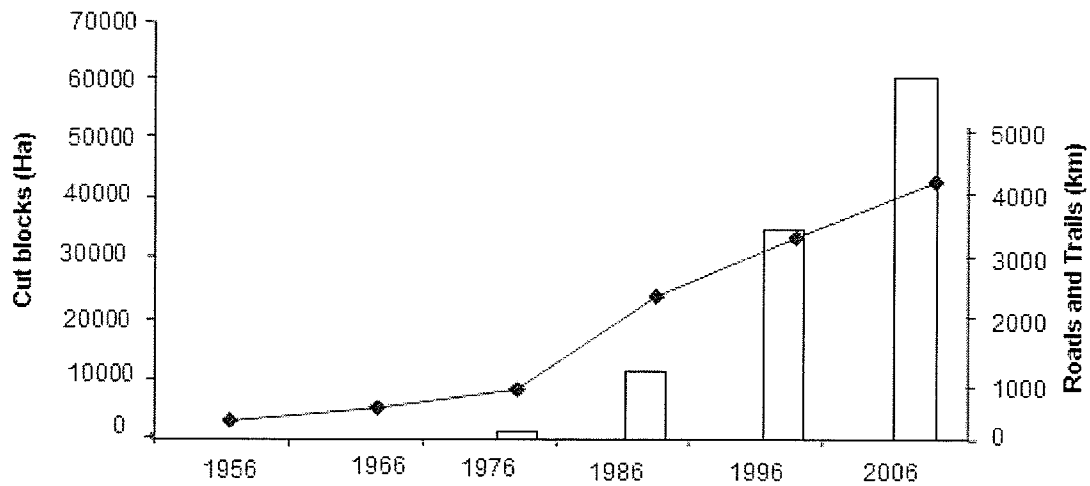


Figure 3-6. Area covered by cutblocks and linear features in 1966 and 2006 on the Provincial Crown Land portion of the Prince Albert Greater Ecosystem.

Landscape metrics include various measures of the distribution, spacing, types, sizes and shapes of forest stands. These metrics characterized an increase in mature coniferous stands over the study timeframe, both in the National Park and on the provincial crown land portions of the PAGE (Table 3-2 and 3-3). The increased number of mature coniferous patches, from 436 to 544 (Park) and 4874 to 5398 (Province), is probably also a reflection of an overall increase in the amount of mature coniferous forest in 2006 (Figure 3-4 and 3-5). Area-weighted mean patch size increased significantly and mean nearest neighbor distance decreased significantly indicating larger patches occurring closer together. Mean shape index describes patch shape and complexity. A significant decrease in the mean shape index only occurred outside the Park, indicating a drop in shape complexity, which often occurs following logging activities.

Change in coniferous young/burn stands between 1966 and 2006 also followed a similar trend in both portions of the PAGE, with the exception of area-weighted mean patch size (Table 3-2 and 3-3). Again, the decreased number of patches between 1966 and 2006 was likely a reflection of changes in the amount of young coniferous/burn forest on the landscape. The change in area-weighted mean patch size of coniferous young/burn differs between the Park and provincial crown land; the observed increase in the National Park and decrease on crown land is likely due to natural disturbance in the Park and a combination of natural and anthropogenic disturbance on crown land. An increased mean nearest neighbor distance was also detected for both areas indicating patches of the same cover type occurred farther from one another.

Table 3-2. Changes in habitat patch metrics ($\bar{x} \pm$ s.e.) between 1966 and 2006 for the National Park portion of Prince Albert Greater Ecosystem.

Habitat Class	1966	2006	P
Area-weighted mean patch size (ha)			
Coniferous Mature	7186 ±1000 (19652)	8317.14 ± 1128 (11864)	<0.0001
Treed Muskeg	647.44 ± 159 (2730)	624.37 ± 155 (2730)	0.0014
Hardwood	4895.15 ± 980 (15533)	4905.42 ± 958 (15514)	0.887
Coniferous Young/Burn	2009.78 ± 464 (6385)	4770.86 ± 1413; 6274	<0.0001
Mean nearest neighbour (m)			
Coniferous Mature	224.5 ± 3808 (3300)	165.51 ± 192 (2308)	0.0007
Treed Muskeg	182.63 ± 232 (2701)	187.77 ± 241 (2702)	0.6451
Hardwood	182.34 ± 176 (1503)	192.49 ± 211 (2302)	0.4499
Coniferous Young/Burn	258.17 ± 455 (3808)	1322.74 ± 2292 (8065)	0.0208
Mean shape index			
Coniferous Mature	1.98 ± 1.64 (23)	1.98 ± 1.7 (24)	0.898
Treed Muskeg	1.91 ± 1.03 (10)	1.91 ± 1.02 (10)	0.9535
Hardwood	1.97 ± 1.44 (16)	1.95 ± 1.4 (16)	0.9217
Coniferous Young/Burn	1.96 ± 1.1 (10)	2.11 ± 1.16 (7)	0.2536
Number of patches			
Coniferous Mature	436	544	
Treed Muskeg	954	945	
Hardwood	443	447	
Coniferous Young/Burn	279	28	

Treed muskeg was the habitat type exhibiting the least overall change throughout the study area. Area covered and number of patches remained static in the Park and on crown land and there were no significant changes in mean nearest neighbor or mean shape index (Table 3-2 and 3-3). The only noticeable change in treed muskeg stands was a significant decrease in area-weighted mean patch size, both in the Park and on the provincial crown land.

Similar to treed muskeg, limited changes were observed for hardwood/mixedwood stands between 1966 and 2006. The only changes detected were: a decrease in area covered by hardwood/mixedwood stands (174643 ha to 108063 ha): an increase in number of patches (Table 3-2) and a decrease in area-weighted mean patch size, all on the crown land portion of the PAGE. These changes were coincident with a history of logging that accelerated over the study period along with fragmentation of the landscape.

Table 3-3. Changes in habitat patch metrics ($\bar{x} \pm$ s.e.) between 1966 and 2006 for the provincial crown land portion of Prince Albert Greater Ecosystem.

Habitat Class	1966	2006	P
Area-weighted mean patch size (ha)			
Coniferous Mature	4013 ± 449 (19635)	4043 ± 450 (22132)	<0.0001
Treed Muskeg	14822 ± 1232 (41285)	8880 ± 902 (27475)	<0.0001
Hardwood	3886 ± 497 (17986)	340 ± 95 (1752)	<0.0001
Coniferous Young/Burn	3353 ± 426 (18368)	2154 ± 372 (8457)	<0.0001
Cutblocks	n/a	121 ± 47 (683)	n/a
Mean nearest neighbour (m)			
Coniferous Mature	217 ± 220 (1844)	184 ± 162 (2002)	<0.0001
Treed Muskeg	200 ± 170 (8261)	196 ± 171 (1844)	0.3154
Hardwood	265 ± 354 (5001)	254 ± 354 (7940)	0.2514
Coniferous Young/Burn	275 ± 392 (4800)	333 ± 636 (8746)	0.0015
Cutblocks	n/a	169 ± 295 (2956)	n/a
Mean shape index			
Coniferous Mature	1.81 ± 1.08 (24.84)	1.77 ± 1.19 (32.37)	0.0453
Treed Muskeg	1.83 ± 1.24 (26.11)	1.79 ± 1.19 (22.64)	0.0697
Hardwood	1.69 ± 1.02 (18.91)	1.79 ± 0.66 (7.44)	0.0991
Coniferous Young/Burn	1.76 ± 1.07 (22.23)	1.79 ± 1 (13.45)	0.6443
Cutblocks	n/a	1.90 ± 1.2 (13.5)	n/a
Number of patches			
Coniferous Mature	4874	5398	
Treed Muskeg	3543	3760	
Hardwood	2830	3845	
Coniferous Young/Burn	3064	1426	
Cutblocks	0	2526	

Finally, changes in landscape configuration measured through distance metrics were only significant on the provincial crown land portion of the PAGE. Distances between habitat classes selected by boreal caribou (mature coniferous and treed muskeg) and those avoided (hardwood/mixedwood) were significantly less on provincial crown land in 2006 when compared to 1966 (Table 3-4). The juxtaposition of selected and avoided habitat types and the recently constructed roads and trails network clearly point to a reduction in the functional attributes of selected habitat types for caribou.

Table 3-4. Landscape configuration changes: Distances ($\bar{x}$ ± s.e., (max)) between key habitat types in 1966 and 2006 for the provincial crown land and Prince Albert National Park portions of the Prince Albert Greater Ecosystem

Distance Variables	1966	2006	P
Provincial Crown Land			
Coniferous Mature to Hardwood	72.99 ± 299 (2955)	80.41 ± 330 (2955)	0.7121
Treed Muskeg to Hardwood	69.26 ± 269 (3477)	73.92 ± 237 (1783)	0.6888
National Park			
Coniferous Mature to Hardwood	344 ± 714 (6099)	283 ± 721 (5272)	<0.0001
Treed Muskeg to Hardwood	280 ± 507 (5233)	191 ± 620 (4758)	<0.0001

3.4 Discussion

The historical compilation of caribou observations demonstrated that the southern boundary of caribou distribution (in central Saskatchewan) has not changed over the last 50 years, although range retraction has occurred in other parts of the Province (Arsenault, 2003; Arsenault, 2005; Saskatchewan Environment, 2007). Also, very few observations have been made in the National Park since the 1980s despite significant survey and collaring efforts. In 2007, a group of caribou was observed in the northeast region of the Park, north of Crean Lake, fecal pellets were collected and 3 unique genotypes profiled (Manseau, unpublished results). Other observations have occurred east of the Park along Highway 2, near the Crean River. This suggests that the animals likely used the Park area as part of their larger range and recent landscape changes are affecting this sporadic or seasonal range use pattern. Overall their distribution is also more confined and compared to telemetry work done between 1992-1995 (Rettie and Messier, 2001), the current home ranges of caribou are significantly smaller (Arsenault and Manseau, 2009).

For both the National Park and the provincial crown land portions of the PAGE, our results suggest an ageing landscape; this has also been reported in other regions of the boreal forest (Johnson et al., 1998; Walker, 1999; Harvey et al., 2002.) and is most often attributed to changes in fire incidence and fire management strategies (Walker, 1999). As observed in other regions of the boreal forest, anthropogenic activities also increased over the last 40 years and particularly over the last 20 years. As expected, the changes primarily occurred on the provincial crown land portion of the PAGE and are the direct response of commercial forestry activities and the associated roads and trails network. Interestingly, both the results from the 1992-1995 and the 2004-2009 collaring work showed that animals north Montreal Lake, west of Bittern Lake and near Weyakwin Lake never crossed Highway 2. Animals west of Lawrence Lake never crossed Highway 922. In both locales, the animals moved within a few meters of the road but did not cross the road.

Changes in the PAGE included the creation of a national park in 1927, the establishment of commercial forestry and overall increased human activities that have lead to significant shifts in land use management policies. The presence of commercial timber, the development of roads, cottaging areas and settlements has all contributed to the current fire suppression efforts (Arsenault and Manseau, 2009). In the early 1940's, many fires burned unsuppressed in both the Park and surrounding area (Weir, 1996) as fire prevention and fire suppression were not practiced (R. Davies, pers. comm.). Changes in fire interval following settlement and industrial development have been observed in other regions including Ontario and Quebec (Bergeron, 1991), Minnesota (Clark, 1988), British Columbia (Johnson and Larsen, 1991) and Alberta (Larsen, 1997).

In the Park, fire suppression still occurs to protect residences, neighboring communities, park facilities and adjacent provincial forests (Prince Albert National Park, 2008). Research on fire interval in Prince Albert National Park conducted by Weir (1996, 2000) demonstrated a shorter fire interval in 1890 to 1950. At that time, fires were set to clear land and often fell out of control and burned forested areas. Weir also showed that from 1950 to the present, the fire interval was longer, and likely due to fire suppression. Extending fire intervals beyond long-term norms is detrimental in the boreal forest because fire is a natural disturbance and essential to maintaining lichen rich coniferous stands by regenerating plant growth (Klein, 1992; Johnson et al., 2001b). Recently, the Park has initiated controlled burns to create a fire barrier along the Park boundaries with the goal of reestablishing the natural fire frequency of the area (Prince Albert National Park, 2008). Our results clearly showed that habitat fragmentation primarily occurred on the provincial crown land portion of the PAGE. Fragmentation is characterized by an increased number of patches and mean nearest neighbor distances, a decreased patch size and mean shape index (Forman and Godron, 1986; Heggem et al., 2000; Turner et al., 2001). A decrease in mean shape index suggests a drop in complexity of patch shape that is often associated with fragmentation (McGarigal and Marks, 1995). Our results demonstrated all four of these indicators of fragmentation.

Boreal woodland caribou are sensitive to landscape change and the long-term persistence of local populations is essential for the conservation of this species (Thomas and Gray, 2002; Environment Canada, 2007). The ageing landscape and associated increase in mature coniferous habitat, in both the Park and the provincial crown land, should be favorable for woodland caribou as these are habitat types selected by the animals (Hirai, 1998; Brown et al., 2000a; Brown et al., 2000b; Rettie and Messier, 2000; Schneider et al., 2000; Thomas and Gray, 2002; Mahoney and Virgl, 2003; Lander, 2006; O'Brien et al., 2006). The increased anthropogenic disturbance and resulting patchwork of selected and avoided habitat types on provincial crown land may however be counteracting those benefits and reducing the functional attributes of the coniferous mature habitat patches. O'Brien (2006) found that woodland caribou selected large clusters of high quality habitat patches over the high quality habitat patches themselves. These large clusters of well-connected habitat patches or the resulting habitat mosaic were important in providing both food and cover. This translates into separation from higher densities, both of other ungulate species and associated predators.

Short-term effects of fire are detrimental to caribou as it creates significant amount of deadfall and affects the overall quality of those stands (Schaefer and Pruitt, 1991; Klein, 1992). Disturbance due to fire has however decreased in both the National Park and provincial crown land through increased fire suppression efforts (Prince Albert National Park, 2008). In general, fire is not a concern to caribou if suitable habitat is available in adjacent areas (Klein, 1992). However, if the trend toward decreasing levels of natural disturbance continues, it may lead to abnormally large tracts of old growth forest. When burned, they may result in huge areas of habitat unsuitable for caribou use (Klein, 1992; Johnson et al., 2001a).

Anthropogenic disturbance, such as logging and access development, have a variety of detrimental impacts on caribou populations (Cumming and Beange, 1987; Rettie and Messier, 1998). Increased number of patches of recently logged areas may attract greater number of other ungulate species such as moose, elk and white-tailed deer (Brown et al., 2000a; James et al., 2004) and subsequently, higher concentrations of predators such as wolves (Bergerud and Elliot, 1986; Rettie and Messier, 1998). Ultimately, increased area logged can lead to range retraction (Bradshaw et al., 1997) as caribou actively avoid disturbance (Cumming and Beange, 1987; Chubbs et al., 1993; Smith et al., 2000; Johnson and Gilligham, 2002). A developed roads and trails network may also facilitate access to formerly isolated areas increasing mortality, from hunting and predation (Dyer et al., 2001; Whittington et al., 2005) and from caribou-vehicle accidents (Cumming and Beange, 1987).

3.5 Conclusion

Both the National Park and provincial crown land portions of the PAGE can be characterized as an older forest when compared to the 1960s. The two areas are managed differently; the provincial crown land being accessible to forestry, offering transport corridors among communities and diverse recreational activities. This is a working landscape with significant levels of fragmentation. Restoring the natural fire process to the Park and surrounding area and limiting future anthropogenic activities outside the Park will ensure critical habitat is available for boreal woodland caribou in the future.

Protected areas such as national parks are important to ensure long-term survival of boreal caribou; however, they very rarely encompass an entire range and their conservation requires more integrated, landscape level management strategies (Armstrong et al., 2000). Park boundaries do not present movement barriers to caribou as it is a long ranging species and thus, habitat surrounding protected areas is integral to their survival (Mosnier et al., 2003). Overall, it is critical to ensure that sufficient habitat and adequate connectivity within and between clusters of habitat exist, thereby allowing caribou to move freely throughout their range.

4. Using predictive habitat modeling to assess changes in winter caribou habitat and landscape connectivity in the Prince Albert Greater Ecosystem

4.1 Introduction

Many studies throughout Canada have shown that woodland caribou (*Rangifer tarandus caribou*) select for treed muskeg, wetlands, and mature coniferous stands (Stuart-Smith et al., 1997; Hirai, 1998; Brown et al., 2000a; Brown et al., 2000b; Rettie and Messier, 2000; Schneider et al., 2000; Mahoney and Virgl, 2003; O'Brien et al., 2006). The suitability of the landscape, however, is not determined only by the presence or amount of suitable habitat but also by the spatial configuration of areas of suitable habitat (Ferreras, 2001; Cook et al., 2003; Haynes and Cronin, 2004; O'Brien et al., 2006). Connectivity affects the ease with which an organism can move through the landscape (Taylor et al., 1993; Tischendorf and Fahrig, 2000). The impact of a decrease in connectivity on a species depends on the hostility of habitat types in the matrix and the ability of a particular organism to cross unfavourable habitat types (With et al., 1999). The inverse of connectivity is fragmentation which is often caused by human land use activities (Andrén, 1994).

Habitat fragmentation is considered one of the greatest threats to biodiversity making it an important conservation issue (Harris, 1984; Forman and Godron, 1986; Saunders et al., 1991). In highly fragmented landscapes, the decline of wildlife populations is greater than that expected by habitat loss alone (Andrén, 1994) and ultimately, these changes to the landscape can isolate groups of animals (Bélisle and

Desrochers, 2002). Woodland caribou are sensitive to fragmentation; O'Brien et al (2006) found that woodland caribou were associated with large connected habitat patches.

In the boreal forest, the main factors leading to habitat loss and habitat fragmentation are: changes in natural and anthropogenic disturbance patterns, increased commercial and industrial activities, increased road access to remote areas and recreational activities (Harris, 1984; Forman and Godron, 1986). Fire is a natural disturbance and has long-term ecological benefits (Bergeron, 1991; Klein, 1992; Johnson et al., 2001b). In the boreal mixedwood forest of North America, the fire return interval ranges from 30 to 150 years (Johnson, 1992). Changes in fire frequency can be caused by shifts in climate, land use pattern and land management strategies (Clark, 1988; Bergeron, 1991; Johnson and Larsen, 1991; Larsen, 1997). At the time of human settlement, fires were frequent as deliberate burns were set to clear land for agricultural purposes (Williams, 1989; Whitney, 1994; Weir, 1996). After an area is settled, fire frequency tends to decrease as forested areas become fragmented and cannot support the spread of fire (Weir, 1996).

Following settlement of the boreal forest, roads were constructed to provide access for industrial development, primarily forestry (Walker, 1999). Forest harvesting is an important commercial activity across the boreal forest and usually targets coniferous stands older than 50 years (Walker, 1999). To be sustainable, logging practices attempt to maintain stands of a variety of ages within the forest management area (Walker, 1999). In Saskatchewan, fire is suppressed over areas of commercial forest tenures or in proximity to communities; natural forest pattern standards and guidelines for the forest industry aim to produce landscapes and harvest areas that emulate the patterns created by fire

(Saskatchewan Environment, 2009). Areas managed for logging are however still affected by fire so it may be difficult to retain stand composition that is comparable to a natural disturbance regime (Walker, 1999). Occurrences of fire on a landscape where logging activities are prevalent often lead to a young age structure (Reed and Errico, 1986).

Landscape changes, natural or anthropogenic, negatively affect boreal caribou, which, as habitat specialists, are dependent on old growth forests for survival (Rettie and Messier, 2000; Smith et al., 2000; Mahoney and Virgl, 2003). Due to increased abundance of other ungulate species (moose, deer and elk) and associated predators in stands of younger age classes, boreal caribou tend to avoid recently logged areas (Chubbs et al., 1993; Cumming and Beange, 1987; Smith et al., 2000; Johnson and Gilligham, 2002; Lander, 2006) and areas near roads and trails (Cameron et al., 2005; Nellemen and Cameron, 1996). Caribou also tend to avoid recent burns (Schaefer and Pruitt, 1991; Klein, 1992; Thomas and Gray, 2002; Lander, 2006). Caribou have persisted in the boreal forest for thousands of years in the presence of fire, provided suitable habitat is available in adjacent areas (Schaefer, 1996; Schaefer and Pruitt, 1991). Logging and road development often displace caribou (Chubbs et al., 1993; Dyer et al., 2001) and since these activities lead to more permanent landscape changes, they can result in range retraction (Bradshaw et al., 1997; Thomas and Gray, 2002).

The Prince Albert National Park (PANP) and Greater Ecosystem are located in the boreal mixedwood forests of Canada, in the province of Saskatchewan and form a part of the Smoothstone-Wapaweka Woodland Caribou Management Unit (SW-WCMU). The fire frequency of this area has decreased following settlement (Johnson, 1992; Weir et al., 2000) and, in the past 40 years, significant logging and road development surrounding the Park has occurred. This ecosystem has traditionally been used by a resident population of boreal caribou (Banfield, 1961) but there are concerns over the long-term viability of the population (Arsenault, 2003; Saskatchewan Environment, 2007). In central Saskatchewan, boreal caribou population declines have been documented in the 1940s and again in the 1980s. The first decline led to a ban in sport hunting and an increase in caribou population in the 1950s was attributed to wolf control and hunting closure (Rock, 1988; Rock, 1992). In 1987, another population decline was confirmed and sport hunting was again banned (Rock, 1988; Rock, 1992). Subsistence harvesting still occurs, although only opportunistically (Trottier, 1988). Work conducted by the University of Saskatchewan (Rettie and Messier, 1998) and more recently through a collaborative effort between Parks Canada, Saskatchewan Environment, the Prince Albert Model Forest, Weyerhaeuser Canada Ltd. and the University of Manitoba (Arsenault and Manseau, 2009) suggests that the population may not be increasing. The distribution of this herd is more confined now than 40 years ago (Arlt and Manseau, 2009) and the individual home range sizes of these animals has decreased significantly in the past decade (Arsenault and Manseau, 2009).

Logging has not been permitted within the Park in the past 60 years and, fire has been suppressed; however, a prescribed burning program has been put in place to reinstate a natural fire cycle (Prince Albert National Park, 2008). The area outside of the National Park is managed primarily for forestry purposes by Saskatchewan Ministry of Environment (MoE) (Government of Saskatchewan, 2002). Waskesiu is a community located on the east side of the National Park and Montreal Lake First Nation is situated to the east of the National Park, at the southern tip of Montreal Lake.

Creating an historical representation of the landscape will assist in the recovery efforts and guide current and future forestry management and land-use planning activities. The main objectives were to develop a predictive habitat model for boreal caribou using GPS telemetry data and assess changes in the amount of spatial configuration of habitat quality for 1966 and 2006. In the first chapter, an ageing landscape and significant landscape changes were demonstrated due to commercial forestry activities on the provincial crown land portion of the PAGE. We predicted an ageing landscape and significant habitat change due to commercial forestry activities on the provincial crown land portion of the PAGE.

4.2 Methods

4.2.1 Study Area

For study area details and map refer to previous chapter 3.2.1.

4.2.2 Smoothstone-Wapaweka Woodland Caribou Management Unit

Arsenault (2009) has defined eight Woodland Caribou Management Units (WCMUs) within the Province based on clusters of caribou observations, areas of similar ecological characteristics (Acton et al., 1998) and peatland distribution. The PAGE is part of the Smoothstone-Wapaweka WCMU that comprises an estimated 350 animals (Arsenault, 2003). This herd is considered high risk due to anthropogenic activity (logging and road development) and habitat loss (Saskatchewan Environment, 2007).

4.2.3 Data description

Adult female caribou were fitted with GPS collars (Lotek Wireless Inc., 115 Pony Drive, Newmarket, Ontario) following capture by net gun in winter. Eighteen animals were collared in 2005 and 2006 and collars were set with a 4-hour location frequency. The analyses focused on the late winter season (January 16 to March 28) defined based on changes in movement rates for this herd (Dyke, 2008). This season was selected because during this period habitat selection by caribou is strongest and resources are most scarce (Brown et al., 2007). Using one year of data per caribou resulted in 188 to 610 telemetry points per animal following Koper and Manseau (2009).

To calculate the home range of this herd, the location data was split into two groups – one group occupying the area north of PANP and another group occupying the Bittern Lake area east of PANP. This technique was used to avoid including the northeast portion of PANP in the home range as no location points occurred in this area. The home range for this herd was generated separately for both groups of location points by calculating 100% minimum convex polygons and resulted in a total home range size of 200,501 ha.

4.2.4 Habitat Maps

Map layers for the National Park and provincial crown land portion of the PAGE were created separately since the type and extent of data available for the two areas differed. Although we tried to create seamless layers for the PAGE area, map resolution issues could not be resolved and prevented us from directly comparing habitat quality between the two areas. For both the Park and the provincial crown land portion of the PAGE, we created map layers for 1966 and 2006 to assess changes in habitat quality through predictive mapping and connectivity analyses.

For details on creation of the habitat maps refer to the previous chapter 3.2.4.

To prepare the map layers for analyses, we reclassified the vegetation layers using a simplified classification scheme often used in the production of woodland caribou habitat maps (e.g. O'Brien et al., 1996; Rettie et al., 1997). Vegetation classes of similar composition were combined to produce 13 habitat classes (Table 4-1). Each map layer was rasterized at a 100 m grid and filtered using Spatially Explicit Landscape Event Simulator (SELES; Fall and Fall, 2001) to remove patches of less than 10 ha which are below the minimum mapping unit and are likely artifacts from the raster conversion process.

Table 4-1. Habitat class composition and definitions used in creating map layers

Vegetation Subclass	Description	FRI Land Cover Classification	Age (years)
SprM	Mature black and white spruce dominated stands	Black Spruce Mature	≥61
		White Spruce Mature	≥61
JpM	Mature jack pine dominated stands	Jack Pine Mature	≥61
JpBsM	Mature jack pine black spruce dominated stands	Jack Pine/Black Spruce Mature	≥61
Ci	Coniferous intermediate dominated stands	White Spruce Intermediate	41-60
		Black Spruce Intermediate	41-60
		Jack Pine Intermediate	41-60
		Jack Pine/Black Spruce Intermediate	41-60
BSL	Mature black spruce larch dominated stands	Black Spruce Larch Mature	≥61
TMsk	Closed or open treed muskeg	Closed Treed Muskeg	n/a
		Open Treed Muskeg	n/a
Mw	Mixedwood dominated stands	Coniferous Mixedwood Mature	≥61
		Coniferous Mixedwood Intermediate	41-60
		Hardwood Mixedwood	n/a
Hw	Hardwood dominated stands	Hardwood	n/a
OpMsk	Open muskeg or meadow or fen	Open Muskeg	n/a
		Brushland	n/a
		Meadow	n/a
		Fen	n/a
RbCy	Recently burned or coniferous young stands	Recent Burn	≤40
		Coniferous Young	≤40
Logged	Recently logged	Recent Logged	≤40
Road	Roads	Roads and Trails	n/a
Water	Water	Water	n/a

For the 1960 map produced for the provincial crown land portion of the PAGE, we performed a validation of the resulting habitat types to assess accuracy. Forest Resource Inventory maps from the 1960s were used in the analysis and 7451 systematically distributed points were generated in ArcGIS 9.2 (Environmental Systems Research Institute, 2006) using the Hawth's tools extension (Beyer, 2004). Stand attributes were derived for each point and compared. The results indicated that more than 70% of the points on the 1966 generated map corresponded to the classes extracted from the 1960 hard copy maps. This overall accuracy level is above the accepted standard of 70% (Burnside et al., 2003). Accuracy levels of 72% were obtained for coniferous mature and 84% for coniferous young and recent burns. Some of the differences may be attributed to different classification schemes, differences in map resolution or differences in the boundaries drawn (limits of the polygons) for each forest stand.

4.2.5 Predictive Habitat Maps

The influence of habitat types on the habitat selection of caribou was modeled (Brown et al., 2007) by assessing differences between random and telemetry locations. SELES (Fall and Fall, 2001) was used to generate 8,991 random points on the herd home range, equivalent in number to the telemetry points. Distances from telemetry and random points to each habitat type was calculated. Distance metrics were selected because they capture how the proximity of certain stand types influences the selection of an individual while located in a given patch (Johnson and Gilligham, 2005).

Mean distances to habitat types and correlations between distance variables were calculated. We created a global model composed of biologically relevant variables. The model included distance to variables and interaction terms.

Global model: Habitat use=intercept+SprM+JpM+JpBsM+BsL+TMsk+Hw+RbCy+

Logged+Water+JpM*Tmsk+JpM*Hw+Tmsk*Hw

We used Proc GENMOD in SAS 9.1 (SAS Institute Inc., 2003) to develop generalized estimating equations (GEEs) for analysis. Correlation structure for GEEs cannot be correctly specified in this case because random and telemetry points have differing structures (Koper and Manseau, 2009) so, we used an empirical variance estimator which is robust to misspecification of correlation structure (Hardin and Hilbe, 2003; Fitzmaurice et al., 2004; Overall and Tonidandel, 2004). We used an independent correlation structure as recommended by Koper and Manseau (2009) when dealing with random and telemetry locations.

Following the assessment of the global model using GEEs, with all animal data included, another model was constructed using only significant variables, based on empirical standard errors, from the global model. This second model was analyzed using a GEE to obtain parameter estimates for each of the retained variables. Positive parameter estimates indicate avoidance while negative parameter estimates indicate selection by animals. These parameter estimates were used to construct a predictive habitat map (Mladenoff et al., 1995) for the late winter season using the 2006 habitat layer. The distance layers generated in SELES were combined in ArcGis as per the models created by the GEEs to generate a predictive habitat map for 2006.

A k-fold cross validation (Boyce et al., 2002) was used to validate this model. GEEs predict habitat selection of a population so for each run of the k-fold validation data points from 3 animals were withheld and the model constructed using the remaining 15 animals (or 83% of the data) then the fit of the model was tested using the withheld data. Ten RSF bins with equal observations in each bin were created and a Spearmann Rank correlation analysis was performed on the area adjusted frequencies of the bins (Koper and Manseau, 2009).

The model used to develop the 2006 predictive map was applied to the 1966 habitat layer to create another predictive habitat map. To assess changes in habitat quality between 1966 and 2006 transition probabilities were used. Transition probabilities measure the likelihood of one habitat type transitioning to another within a given time period (Burnside et al., 2003). We calculated transition probabilities of each habitat quality level (low, medium-low, medium, medium-high, high) between 1966 and 2006 by quantifying changes of each pixel in the two layers using SELES.

4.2.6 Connectivity

Georeferenced mathematical graphs were used to assess the connectivity of the landscape (Harary, 1969; Marcot and Chin, 1982; Keitt et al., 1997). These graphs consisted of nodes to represent high quality habitat patches and a series of links which connect patches to one another (O'Brien et al., 2006). A minimum planar graph (MPG) was used to provide a visual interpretation of landscape connectivity. MPGs use a Delauney triangulation (Okabe et al., 2000) to create a spatial graph of the patches (nodes) and paths (links) between patches (Manseau et al., 2002; O'Brien et al. 2006; Fall et al. 2007). At a given distance threshold, movement is assumed to be restricted and links longer than this distance no longer exist (Keitt et al., 2007). Increasing the threshold adds links and at each threshold, patches that are connected by links are called clusters. Expected cluster size, a landscape level metric, (ECS; O'Brien et al., 2006) was computed at several distance thresholds on the 1966 and 2006 map layers.

There are two types of links between habitat patches - euclidian (straight line) or least cost (O'Brien et al., 2006). To calculate a least cost path, cost must be assigned to each habitat type based on the likelihood of caribou using that patch relative to a high quality patch. We used Proc GENMOD in SAS 9.1 (SAS Institute Inc., 2003) to develop generalized estimating equations (GEEs) for analysis to calculate probability of occurrence relative to a high quality habitat type (Hosmer and Lemeshow, 2000; Manley et al., 2002) in this case mature jack pine dominated stands. The same set of random points used in the predictive mapping was used for this analysis. Buffers of 150 meters were added to telemetry and random points and points were classified based on mode habitat type within each buffer. Cost values were computed for all habitat types as the inverse of the odds ratio (O'Brien et al., 2006). SELES (Fall and Fall, 2001) was used to create minimum planar graphs and to calculate ECS. The ECS was plotted over a variety of distance thresholds for both the 1966 and 2006 map layers.

4.3 Results

4.3.1 Predictive mapping

A total of 8991 telemetry points were used in developing the model and an equal number of random points were generated. Telemetry points occurred in most habitat classes except recently logged areas and roads. Random points occurred in each habitat class including one point each for recent logged and roads. Recent logged and roads cover the least area of any habitat type which may account for lack of points. Treed muskeg contained the greatest proportion of telemetry and random points, 61% and 28% respectively. Seventeen percent of the telemetry points and 18% of the random points fell in the black and white spruce mature habitat class. The greatest difference between mean distances of telemetry and random points occurred in the logged, road and treed muskeg habitat classes (Table 4-2). Telemetry points occurred closer to treed muskeg and farther from recent logged and roads. There were correlations between roads and recent logged and between mixedwood and hardwood ($r > 0.6$). Due to this mixedwood and roads were not included in the global model.

Table 4-2. Mean distance (m) of woodland caribou telemetry points and random points to various habitat types in the Prince Albert Greater Ecosystem during the late winter season (n=17982)

	Telemetry (m)	Random (m)
Variable	Mean (sd)	Mean (sd)
SprM	307.2 (264.9)	381.6 (405.3)
JpM	637.9 (571.9)	977.4 (917.3)
JpBsM	1400.7 (1311.9)	1429.0 (1239.2)
Ci	2948.4 (1996.9)	2519.9 (2034.1)
BSL	852.6 (688.2)	999.3 (894.6)
TMsk	60.16 (106.7)	338.4 (471.4)
Mw	1763.2 (1172.3)	1267.0 (992.7)
Hw	1611.4 (1096.7)	1206.2 (1080.8)
OpMsk	1162.7 (738.5)	1170.9 (952.0)
RbCy	2626.8 (1289.5)	2031.8 (1477.8)
Logged	4434.6 (1736.4)	2762.4 (1988.8)
Road	3593.8 (1725.2)	2358.8 (1981.5)

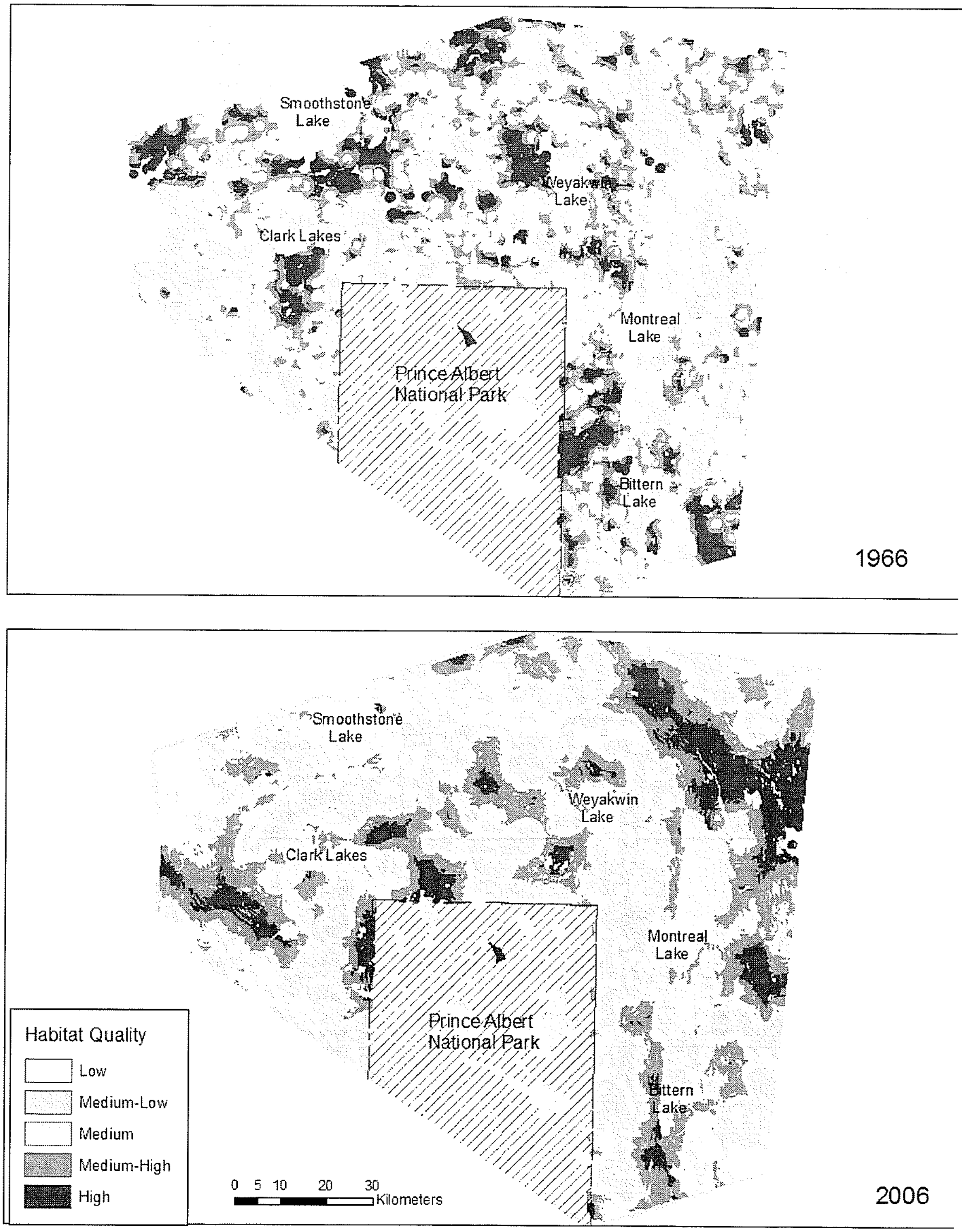
Within the Prince Albert Greater Ecosystem, during the late winter season, caribou showed a significant response for distance to treed muskeg, recent burn/coniferous young, cutblocks and the interaction term of jack pine*hardwood and treed muskeg*hardwood (Table 4-3). The interaction terms were included in the model due to the documented influence of hardwood stands on habitat selection (Thomas and Gray, 2002). Individuals on this home range showed selection for distance to jack pine (β=-0.1864) and distance to treed muskeg (β=-6.9334) and avoidance for distance to recent burn/coniferous young (β=0.2897), recent logged (β=0.3711) and hardwood (β=0.2554). The interaction terms, jack pine*hardwood and treed muskeg*hardwood had β values of -0.2456 and 1.3149, respectively.

In predicting habitat selection of the animals the k-fold cross validation revealed that the model had an average r of 0.762 (SE=.092).

Table 4-3. Identification of significant variables based on generalized estimating equations for the late winter season. Values in bold were used in the corresponding selection model

Variable	P
SprM	0.1264
JpM	0.6574
JpBsM	0.6946
BSL	0.5563
TMsk	**<0.0001**
Hw	0.1454
RbCy	**0.0205**
Logged	**<0.0001**
JPM*TMsk	0.0995
JPM*Hw	**0.0205**
TMsk*Hw	**0.0336**

The predictive habitat map produced for 2006 shows isolated clusters of high quality habitat throughout the study area (Figure 4-1). In 2006, the largest cluster of high quality habitat was north, northeast, and east of Montreal Lake. High quality habitat also occurred west and south of Bittern Lake, north of the park near Weyakwin Lake and also along the north-northwest border of the Park. Additionally, there was a cluster of high quality habitat west of Clark Lake. The predictive map for 1966 showed high quality habitat distributed more evenly across the landscape (Figure 4-1). In general, high quality habitat occurred east and northwest of Bittern Lake, northwest of Weyakwin Lake, south and north of Clark Lake and along the north and northwest borders of the study area.



$$\omega = \mathrm{Exp}(-1.3341+(-0.1864*JpM)+(-6.9334*Tmsk)+(0.2554*Hw)+(0.2897*RbCy)+(0.3711*Logged)+(1.3149*Hw*TMsk)+(-0.2456*Hw*JpM))$$

Figure 4-1. Predictive habitat quality maps for the Prince Albert Greater Ecosystem in late winter.

The transition probability analysis showed areas where habitat quality changed (increased or decreased) between 1966 and 2006. Areas on the landscape that increased in habitat quality in the past 40 years included north and northeast of Montreal Lake, north of Weyakwin Lake, and northeast and west of Clark Lakes (Figure 4-2). Areas on the landscape that decreased in habitat quality include areas west of Bittern and Montreal Lakes, a large area west, east and southeast of Smoothstone Lake. The majority, 88% of low quality habitat, stayed low quality habitat between 1966 and 2006 (Table 4-4). Only 6% of high quality habitat in 1966 remained high quality habitat in 2006. Most of the high quality habitat in 1966 changed to medium or medium-low habitat levels, 29% and 27%, respectively. Eighteen percent of medium low habitat in 1966 changed to high quality habitat in 2006.

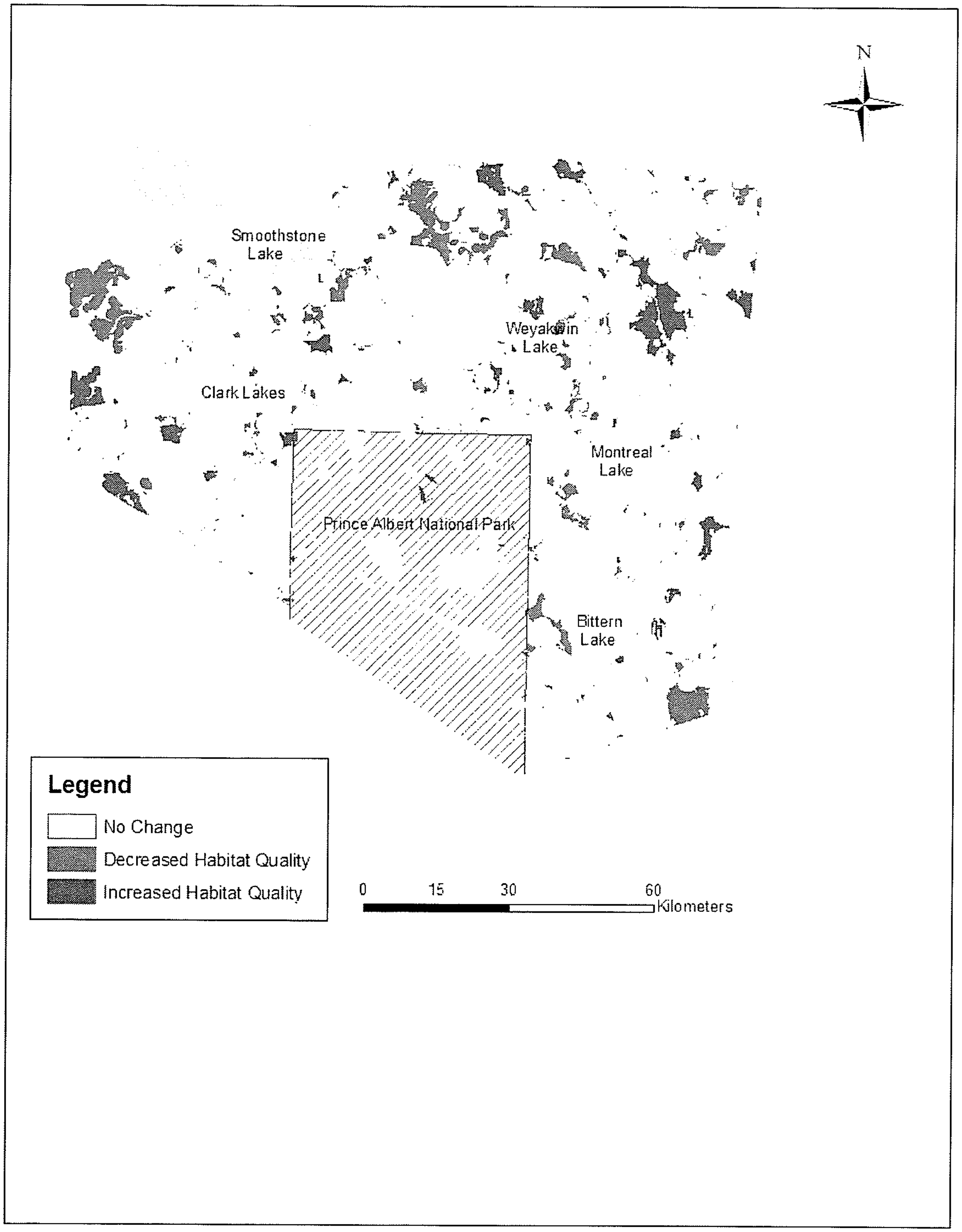


Figure 4-2. Transition probability map for changes in habitat quality between 1966 and 2006 on the Prince Albert Greater Ecosystem in the late winter season.

Table 4-4. Transition probabilities for caribou habitat quality between 1966 and 2006 on the Prince Albert Greater Ecosystem, Saskatchewan.

	1966				
2006	Low	Medium-Low	Medium	Medium-High	High
Low	0.8801	0.0253	0.0477	0.0946	0.1789
Medium-Low	0.0124	0.1082	0.3072	0.3614	0.2743
Medium	0.0220	0.2725	0.3069	0.3431	0.3000
Medium-High	0.0394	0.4079	0.2379	0.1496	0.1841
High	0.0461	0.1861	0.1002	0.0513	0.0627

4.3.2 Connectivity

Based on the results of the GEEs from the predictive model, there was significant selection for distance to treed muskeg and the interaction terms of jack pine*hardwood and treed muskeg*hardwood. Although, distance to jack pine was not significant the interaction term involving jack pine was significant. Expert opinion on boreal caribou habitat selection in this ecozone, the jack pine*hardwood interaction term, and comparative results of random and telemetry point distance to jack pine led us to believe jack pine is a driving factor in habitat selection. Therefore, we choose to use jack pine as a reference category when using GEEs to develop cost surface values.

The results of the GEE indicated caribou select treed muskeg over all other habitat types (Table 4-5). In comparison to jack pine mature stands caribou are one eighths as likely to use recent burn/coniferous young, one sixteenth as likely to use water, one third as likely to use coniferous intermediate, mixedwood, open muskeg or hardwood, two thirds as likely to use black spruce larch or spruce mature, and five sixths as likely to use jack pine/black spruce mature. Cost values for each habitat type range from 1.13 for jack pine/black spruce mature to 10.38 for recent burn/coniferous young. A cost value of

100 was assigned to water. There was no use of roads or recently logged areas by caribou so cost values could not be derived from the telemetry data. The maximum cost value of 100 was assigned to roads and logged because of documented caribou avoidance of logged areas (Chubbs et al., 1993; Cumming and Beange, 1987; Smith et al., 2000; Johnson and Gilligham, 2002; Lander, 2006) and roads (Cameron et al., 2005; Nellemen and Cameron, 1996) and based on the results of the predictive habitat model presented earlier in this document.

Table 4-5. Results of generalized estimating equations used to evaluate habitat selection within the winter home range and with jack pine mature as the reference category

Variable	β	SE	P	OR	95% CI for OR		Cost OR(-1)
					Lower	Upper	
JPM	-	-	-	-	-	-	1
JpBsM	-0.12747	0.4056	0.0017	0.88032	-2.0696	-0.4798	1.13595079
SprM	-0.3244	0.3843	0.3985	0.722961	-1.0076	0.4287	1.38320048
Ci	-1.447	0.7206	0.0446	0.235275	-2.8594	-0.0396	4.25034434
BSL	-0.4327	0.4168	0.2992	0.648755	-1.2497	0.3842	1.54141373
TMsk	0	0.4057	0.0549	1	-0.0164	1.5738	1
Mw	-1.0494	0.6759	0.1205	0.350148	-2.3742	0.2754	2.85593704
OpMsk	-1.5243	0.3273	<0.0001	0.217773	-2.1659	-0.8827	4.59192809
Hw	-1.6015	0.7658	0.0365	0.201594	-3.1025	-0.1004	4.96046755
RbCy	-2.3408	0.6926	0.0007	0.096251	-3.6982	-0.9833	10.3895449
Water	-5.2946	0.674	<0.0001	0.005019	-6.6086	-3.9806	100
Roads	-	-	-	-	-	-	100
Logged	-	-	-	-	-	-	100
Constant	0.0851	0.3224	0.094	-	-	-	-

The ECS values and MPGs differ for the 2006 and 1966 landscapes. On the 2006 landscape, high quality patches connected gradually over low cost distances from 0 to 4000. There were large increases in ECS at cost distances of 5200, 10600 and 13600 (Figure 4-3). In comparison, the ECS increased faster on the 1966 landscape up to 5000 with large increases in ECS at cost distances of 7000, 8800 and 106000 (Figure 4-3). At a cost distance of 21200, all clusters on the 2006 landscape are connected. In comparison, the 1966 landscape is fully connected at much lower cost distance threshold of 13600.

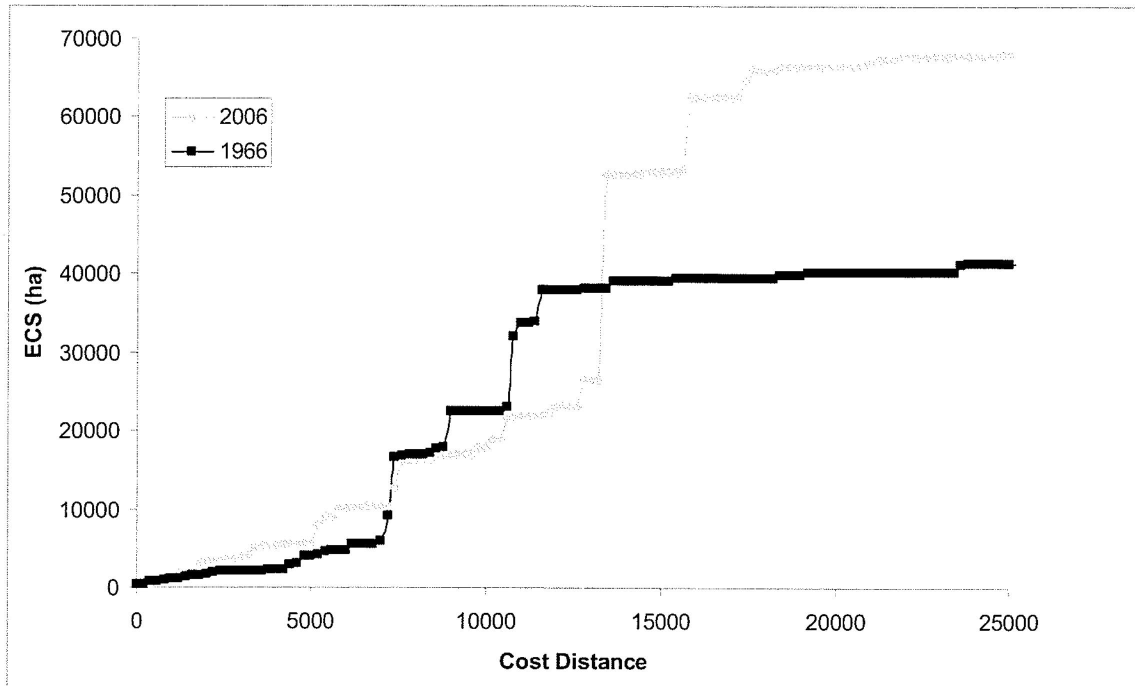


Figure 4-3. Expected cluster size computed for increasing cost distances for 1966 and 2006 on the Prince Albert Greater Ecosystem

The threshold of MPGs for the 2006 landscape show clusters of well-connected habitat northeast of Montreal Lake, east of Bittern Lake, north of the Park and the northwest corner of the Park (Figure 4-4, 4-5). At a cost value of 5200 the east side of Montreal Lake is connected as is the northwest corner of the Park with an area north of the Park. Increasing the cost distance to 10600 connects the north and east side of Montreal Lake and a large cluster forms on the far west side of the study area west of Clark Lakes. At a cost distance of 13600 east of Smoothstone Lake and north of the Park both connect with the area near Montreal Lake. In comparison on the 1966 landscape clusters of connected habitat occur directly north and west of Montreal Lake, northwest of Weyakwin Lake (Figure 4-4, 4-5). Although there are fewer high quality patches on the 1966 landscape these patches are spread more evenly throughout the landscape resulting in a fully connected landscape at a lower cost distance.

Figure 4-4. Minimum planar graph links for the Prince Albert Greater Ecosystem landscape for a) 1966 threshold of 5200, b) 2006 threshold of 5200, c) 1966 threshold of 8800 and d) 2006 threshold of 8800.

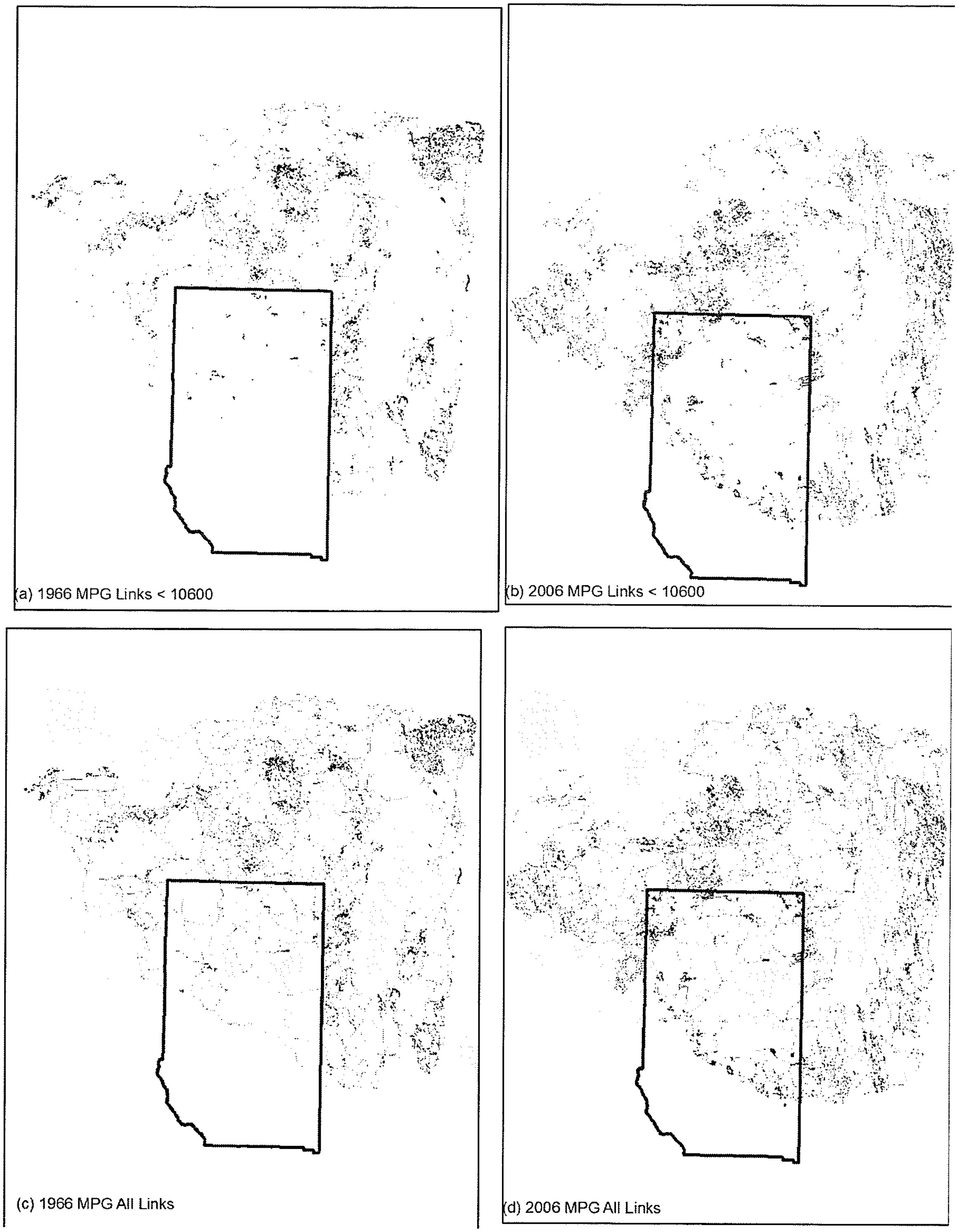


Figure 4-5. Minimum planar graph links for the Prince Albert Greater Ecosystem landscape for a) 1966 threshold of 10600, b) 2006 threshold of 10600, c) 1966 all links and d) 2006 all links.

4.4 Discussion

Between 1966 and 2006 many changes occurred in the PAGE. Recent research in this area showed an ageing landscape and increased anthropogenic activities in the past 40 years (Arlt and Manseau, 2009). Maturing landscapes and anthropogenic activities are common in the boreal forest, particularly in recent decades (Johnson et al., 1998; Walker, 1999; Harvey et al., 2002.). The PAGE landscape in 1966 was young due to extensive fires. Generally, these fires burned unsuppressed in both the Park and surrounding area (Weir, 1996) as fire prevention and fire suppression were not practiced (R. Davies, pers. comm.). In 1966 a very small anthropogenic footprint was observed. There were few roads and, at this point, commercial logging was not yet established. In comparison, in 2006, fire suppression was common throughout the study area and there was a substantial increase in roads and logging. Commercial forestry began in the early 1970's and was well established by 1980 leading to significant landscape changes. This increase in commercial forestry, the development of roads, cottaging areas and settlements led to extensive fire suppression efforts (Arsenault and Manseau, 2009). The change in fire suppression efforts following settlement and industrial development are not unique to this area and have been observed in other regions including Ontario and Quebec (Bergeron, 1991), Minnesota (Clark, 1988), British Columbia (Johnson and Larsen, 1991) and Alberta (Larsen, 1997). The change in fire interval and industrial development over the past 40 years has produced a landscape dominated by mature forests, cutblocks and roads (previous chapter; Arlt and Manseau, 2009).

Caribou in the PAGE showed similar selection and avoidance patterns to other boreal caribou herds. The PAGE animals selected mature jack pine and treed muskeg similar to caribou in British Columbia (Johnson et al., 2000; Johnson et al., 2003), Newfoundland (Mahoney and Virgil, 2003), Alberta (Brown et al., 2000b; Schneider, 2000) and Manitoba (Hirai, 1998; Brown et al, 2000a; O'Brien et al., 2006). Mature conifers such as jack pine are selected to minimize predation risk (Bergerud, 1988; Seip, 1992) and provide forage (Brown and Theberge, 1990; Schaefer and Pruitt, 1991; Schaefer, 1996). The PAGE individuals showed avoidance for disturbance (burned and logged areas) and hardwood stands. This trend in habitat avoidance also occurs in the previously cited studies (Cumming and Beange, 1987; Schaefer and Pruitt, 1991; Chubbs et al, 1993; Smith et al., 2000; Dyer et al, 2002; Johnson and Gilligham, 2002; Lander, 2006).

Other studies have used predictive modeling to evaluate habitat of caribou (Dyke, 2008) and other species (Pereira and Itami, 1991; Clark et al., 1993; Carey et al., 1992; Lehmkuhl and Raphael, 1993; Mladenoff et al., 1995). The model produced in this study was validated and when overlaying the telemetry data onto the predictive map it is evident that caribou are utilizing the predicted high quality areas. Applying this model to a historical landscape may help to see ways in which the habitat quality of an area has changed. The maps and transition probabilities showed less area covered by high quality habitat in 1966 than in 2006. Although there are fewer high quality patches on the 1966 landscape, these patches occur throughout the entire landscape whereas, on the 2006 landscape, high quality patches occur in a few isolated areas. This suggests a lesser amount of high quality habitat in 1966 but an overall better connected landscape. In 2006, the animals are confined to certain areas of the landscape as roads and cutblocks transect their range.

Transition probabilities provide information on areas that changed between 1966 and 2006. From the transition probabilities, it can be noted that between 1966 and 2006 ninety-four percent of the high quality habitat decreased in quality while only twelve percent of low quality habitat increased in quality. If this trend continues, high quality habitat will continue to decrease in area providing less high quality areas for caribou to range. In fact, Arsenault and Manseau, (2009) showed that individual range sizes have decreased significantly over the past 10 years.

Predictive maps are useful in determining areas of high quality habitat but one can only speculate about movement corridors and movement throughout the landscape. The use of minimum planar graphs helps us assess the likelihood an animal will travel a given path between high quality patches or groups of high quality patches (O'Brien et al., 2006). O'Brien et al (2006) states that the matrix surrounding these high quality patches may not all be equally unsuitable so the use of MPG's help provide a more functional representation of landscape connectivity for caribou (With et al., 1997). This technique allows us to identify various thresholds at which parts of the landscape become functionally isolated from one another.

The ECS of the 1966 landscape increased faster, resulting in a connected landscape at a much lower cost distance thresholds, when compared to the 2006 landscape. This means that from the animal's perspective, it would be easier to move throughout the 1966 landscape than the 2006 landscape. Although there are fewer high quality patches on the 1966 landscape, similar to what we see in the predictive maps, these patches are spread more evenly throughout the landscape resulting in a fully connected landscape at a lower cost distance. O'Brien (2006) found that woodland caribou selected large clusters of high quality habitat patches over the high quality habitat patches themselves. It is expected that on the 1966 landscape, a connected landscape with less overall high quality habitat, there would have been a higher survival rate than on a less connected landscape with more overall high quality patches, as in the 2006 landscape. Large clusters of well-connected habitat patches and the resulting habitat mosaic are important in providing both food and cover.

Results from both the predictive map and connectivity analysis point to a less connected landscape with isolated habitat patches in 2006. The reason these clusters occur on the landscape is related to changes in fire frequency and anthropogenic disturbance in the past 40 years. The change in fire frequency since the 1960s is common in the boreal forest (Johnson et al., 1998; Walker, 1999; Harvey et al., 2002.) and fire suppression efforts have created a mature landscape that generally is preferred by woodland caribou. However, in addition to a mature landscape, a fourteen fold increase in road coverage coupled with drastic increases in cutblocks (Arlt and Manseau, 2009) act to transect these clusters of high quality habitat. Boreal caribou tend to avoid logged areas (Chubbs et al., 1993; Cumming and Beange, 1987; Smith et al., 2000; Johnson and Gilligham, 2002; Lander, 2006) and areas near roads and trails (Cameron et al., 2005; Nellemen and Cameron, 1996). Often, logging and road development displace caribou (Chubbs et al., 1993; Dyer et al., 2001) and since these activities lead to more permanent landscape changes, they can result in range retraction (Bradshaw et al., 1997; Thomas and Gray, 2002) as habitat clusters become isolated. Range retraction is already occurring on the PAGE landscape (Arsenault and Manseau, 2009) and will continue unless changes occur in the management of this landscape.

The boreal ecotype of woodland caribou occur at low densities on the landscape making landscape connectivity crucial for the survival of the species. The thresholds identified represent distances animals are willing to travel. The least-cost paths or links may be used to represent movement corridors throughout their range. The landscape in 2006 is highly fragmented mostly due to roads and cutblocks, the use of the MPG's may suggest areas for restoration of the landscape in order to ensure the animals can move throughout the area. These MPG's could suggest specific roads that could be decommissioned and specific areas of the landscape where forestry should be restricted. There are many differences between the 1966 and 2006 landscape. Much of these differences can be attributed to forestry related disturbance. Connectivity measures may be used in the future to work with industry to choose areas where harvesting will have the least affect on animals and areas where restoration may be necessary to ensure the animals can move throughout their range with ease.

Boreal woodland caribou are sensitive to landscape change and the long-term persistence of local populations is essential for the conservation of the species (Thomas and Gray, 2002; Environment Canada 2007). The ageing landscape and associated increase in mature forests throughout the PAGE should be favorable for woodland caribou as these are habitat types selected by the animals (Hirai, 1998; Brown et al., 2000a; Brown et al., 2000b; Rettie and Messier, 2000; Schneider et al., 2000; Thomas and Gray, 2002; Mahoney and Virgl, 2003; Lander, 2006; O'Brien et al., 2006). This is a working landscape with significant levels of fragmentation. Restoring the natural fire process and limiting future anthropogenic activities will ensure critical habitat is available for boreal woodland caribou in the future. Overall, it is critical to ensure that sufficient habitat and adequate connectivity within and between clusters of habitat exist, thereby allowing caribou to move freely throughout their range.

5. Summary, Implications and Recommendations

5.1 Summary of Results

Caribou, being habitat specialists, are sensitive to landscape change. As woodland caribou are listed as a threatened species by the Species at Risk Act, identifying changes to habitat within their home ranges is critical to guide management and conservation efforts. Understanding historical changes to the landscape may help to identify areas with potential for restoration to ensure the persistence of the species.

5.1.1 Changes in caribou distribution

The extent of caribou use of the National Park portion of the PAGE has changed over the last 50 years, with very limited use detected since the 1980s.There have been multiple surveys conducted throughout the Park in recent years and only one observation was made over the last 14 years, in 2007. Caribou are still present over most of the provincial crown land portion of the PAGE although their distribution is clustered and their movements limited resulting in smaller home ranges when compared to 10 years ago (Arsenault and Manseau 2009).

5.1.2 Landscape changes

Transition probabilities showed similar trends in the National Park and provincial crown land portion of the PAGE. Less than 27% of the coniferous young/burn class remained in that class. A large portion of these stands aged to coniferous mature or to hardwood/mixedwood. Fifty four percent of National Park land and 68% of provincial crown land remained in the coniferous mature class. A substantial portion of land within the PAGE as a whole also transitioned to coniferous young/recent burn class. These results indicated an ageing landscape.

The predominant change to older aged stands suggests an ageing forest over the PAGE landscape as a whole. However, the transitioning of large tracts of crown land in the PAGE to coniferous young/burn stands is coincident with a noticeable increase in the number of cut blocks and the development of road and trails network. The amount of area logged increased from 0 ha logged in 1966 to 58211 ha logged in 2006. The road network increased 14-fold on the provincial crown land portion of the PAGE, from 342 km to 4730 km over the same 40-year period.

Changes in landscape metrics indicated an increase in mature coniferous stands over the study timeframe, both in the National Park and on the provincial crown land portions of the PAGE. There was an increased number of mature coniferous patches, from 436 to 544 (Park) and 4874 to 5398 (Province). A significant decrease in the mean shape index only occurred outside the Park, indicating a drop in shape complexity, which often occurs following logging activities.

Change in coniferous young/burn stands between 1966 and 2006 also followed a similar trend in both portions of the PAGE. There were a decreased number of patches between 1966 and 2006. The change in area-weighted mean patch size of coniferous young/burn differs between the Park and provincial crown land; the observed increase in the National Park and decrease on crown land is likely due to natural disturbance in the Park and a combination of natural and anthropogenic disturbance on crown land. An increased mean nearest neighbor distance was also detected for both areas indicating patches of the same cover type occurred farther from one another.

Treed muskeg and hardwood/mixedwood both exhibited the least overall change throughout the study area. The only noticeable change in treed muskeg stands was a significant decrease in area-weighted mean patch size, both in the Park and on the provincial crown land. These changes were coincident with a history of logging that accelerated over the study period along with fragmentation of the landscape.

Finally, changes in landscape configuration measured through distance metrics were only significant on the provincial crown land portion of the PAGE. Distances between habitat classes selected by boreal caribou (mature coniferous and treed muskeg) and those avoided (hardwood/mixedwood) were significantly less on provincial crown land in 2006 when compared to 1966. The juxtaposition of selected and avoided habitat types and the recently constructed roads and trails network clearly point to a reduction in the functional attributes of selected habitat types for caribou.

5.1.3 Predictive mapping

A total of 8991 telemetry points were used in developing the model and an equal number of random points were generated. The greatest difference between mean distances of telemetry and random points occurred in the logged, road and treed muskeg habitat classes. Telemetry points occurred closer to treed muskeg and farther from recent logged and roads. Due to correlations between the distance variables roads and recent logged and between mixedwood and hardwood, mixedwood and roads were not included in the global model.

Within the Prince Albert Greater Ecosystem, during the late winter season, caribou showed a significant response to treed muskeg, recent burn/coniferous young, cutblocks and the interaction term of jack pine*hardwood and treed muskeg*hardwood. The interaction terms were included in the model due to the documented influence of hardwood stands on habitat selection (Thomas and Gray, 2002).

The predictive habitat map produced for 2006 shows large but isolated clusters of high quality habitat throughout the study area. In 2006, the largest cluster of high quality habitat was north, northeast, and east of Montreal Lake. The predictive map for 1966 showed high quality habitat distributed more evenly across the landscape.

The transition probability analysis showed areas where habitat quality changed (increased or decreased) between 1966 and 2006. Areas on the landscape that increased in habitat quality in the past 40 years included north and northeast of Montreal Lake, north of Weyakwin Lake, and northeast and west of Clark Lakes. Areas on the landscape that decreased in habitat quality include areas west of Bittern and Montreal Lakes, a large area west, east and southeast of Smoothstone Lake. The majority, 88% of low quality habitat, stayed low quality habitat between 1966 and 2006. Only 6% of high quality habitat in 1966 remained high quality habitat in 2006.

5.1.4 Connectivity changes

Based on the results of the generalized estimating equations (GEEs), distance to treed muskeg and the interaction terms of jack pine*hardwood and treed muskeg*hardwood were significant. Although, distance to jack pine was not significant the interaction term involving jack pine was significant. Expert opinion on boreal caribou habitat selection in this ecozone, the jack pine*hardwood interaction term, and comparative results of random and telemetry point distance to jack pine led us to believe jack pine is a driving factor in habitat selection. Therefore, we choose to use mature jack pine as nodes and as a reference category when using GEEs to develop cost surface values.

The ECS values and MPGs differ for the 2006 and 1966 landscapes. On the 2006 landscape, high quality patches connected gradually over low cost distances from 0 to 4000. At a cost distance of 21200, all clusters on the 2006 landscape were connected. In comparison, the 1966 landscape is fully connected at much lower cost distance threshold of 13600.

The MPGs for the 2006 landscape showed clusters of well-connected habitat northeast of Montreal Lake, east of Bittern Lake, north of the Park and the northwest corner of the Park. At a cost value of 5200 the east side of Montreal Lake is connected as is the northwest corner of the Park with an area north of the Park. Increasing the cost distance to 10600 connects the north and east side of Montreal Lake and a large cluster forms on the far west side of the study area west of Clark Lakes. At a cost distance of 13600 east of Smoothstone Lake and north of the Park both connect with the area near Montreal Lake. In comparison on the 1966 landscape clusters of connected habitat occur directly north and west of Montreal Lake, northwest of Weyakwin Lake. Although there are fewer high quality patches on the 1966 landscape, these patches are spread more evenly throughout the landscape resulting in a fully connected landscape at a lower cost distance.

5.2 Management Implications

The results of this study have the following implications for the Smoothstone-Wapaweka management unit:

1. Understanding historical changes to the landscape can help us determine changes in caribou range. Due to the irregular survey effort, particularly on provincial crown land, it was difficult to ascertain whether changes in caribou distribution were due to changes in survey effort over time or if they were actual changes in areas of use. It would be easier to determine changes in distribution if there was a consistent survey effort using the same methods.
2. Creating a seamless vegetation layer was challenging. The provincial crown land is surveyed regularly by forestry companies as part of their harvesting agreement. Due to these regular forest inventories the vegetation maps created were very accurate. The national park is not surveyed regularly and the surveys conducted as not as detailed. This made it difficult to compare the areas due to the different scales and frequency of the surveys and also made creating a comparable vegetation classification difficult.
3. The Prince Albert Greater Ecosystem landscape currently has fourteen times the amount of linear features than in 1966 and the area covered by cutblocks increased five fold in the past 20 years. For caribou to persist on this landscape and to ensure this long-ranging species can move throughout their home range the reduction or restoration of linear features should be prioritized.
4. Based on the connectivity analyses the areas north of Montreal Lake, northeast of the park and the area south of Clark Lake need to be restored. Connectivity should

be restored between the Bittern Lake area and Montreal Lake area. Connectivity should also be restored between the national park and the rest of the range.

5. It is recommended that the telemetry data, predictive maps and connectivity analysis be used in future decision making with respect to forestry operations and human development. Continued monitoring of these animals is essential and the development of a monitoring plan should be developed.

5.3 Future Research

1. Regular aerial surveys need to continue with an effort towards the same methodology throughout the entire range. Using the same methodology for each survey would enable identification of changes in distribution.
2. Levels of fragmentation should be monitored to ensure fragmentation levels in this range do not increase. Opening the peatlands for industrial access should not be permitted. This would affect the core caribou habitat and would lead to further fragmentation.
3. The study area should be expanded to the east and north. There are caribou herds in these areas and connectivity analysis could help to determine the ability of these herds to interact, which would ensure persistence of the species in central Saskatchewan.
4. The predictive modeling method used in this study should be used and validated on other herds in the province. In addition, this method should be used on this

herd in different seasons because areas of use differ depending on the season and important areas might be missed if only the winter predictive map is used.

The main purpose of this research was to gain a better understanding of how the Prince Albert Greater Ecosystem has changed in the past 40 years and how this has impacted woodland caribou distribution and habitat. This study met the objectives by assessing changes in caribou distribution, quantifying landscape changes and assessing changes in habitat quality using predictive mapping and connectivity analyses. The knowledge gained from this research can hopefully be used to guide future management decisions by provincial and federal governments with the ultimate goal of the persistence of this herd into the future.

6. Literature Cited

Acton, D.F., Padbury, G.A. and Stushnoff, C.T. 1998. *The ecoregions of Saskatchewan.* Canadian Plains Research Center, University of Regina. 205 pp.

Andrén, H. 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. Oikos 71: 355-366.

Arlt, M.L. and Manseau, M. 2009. Historical changes in caribou distribution and land cover in and around Prince Albert National Park: land management implications. - Rangifer (submitted).

Armstrong, T., Racey, G., and Booky, T. 2000. Landscape level considerations in the management of forest-dwelling woodland caribou (*Rangifer tarandus caribou*) in northwestern Ontario. Rangifer Special Issue No. 12: 187-189.

Arsenault, A.A. 2003. Status and conservation management framework for woodland caribou *(Rangifer tarandus caribou)* in Saskatchewan. Saskatchewan Environment. Fish and Wildlife Tech. Rep. 2003-03. 40 pp.

Arsenault, A.A. 2005. Status and Management of wildlife in Saskatchewan, 2002 and 2003. Saskatchewan Environment, Resource Technical Report 2005-2. 89pp.

Arsenault, A.A. 2009. Sampling guidelines and disturbance impact thresholds for vertebrate species at risk and sensitive species in Saskatchewan. Saskatchewan Environment. Lands Branch Technical Report – 2009-draft.

Arsenault, A. A. and Manseau, M. 2009. Land management strategies for the recovery of boreal woodland caribou in central Saskatchewan. – Rangifer. (submitted).

Banfield, A.F. 1961. A revision of the reindeer and caribou, genus *Rangifer. Natural Museum of Canada*: Bulletin 177.

Bélisle, M., and Desrochers, A. 2002. Gap crossing decisions by forest birds: an empirical basis for parameterizing spatially explicit, individual-based models. Landscape Ecology. 17: 219-231.

Bergerud, A.T. 1988. Caribou, Wolves and Man. Trends in Ecology and Evolution. 3:68-72.

Bergerud, A.T., and Elliot, J.P. 1986. Dynamics of caribou and wolves in northern British Columbia. Canadian Journal of Zoology. 64: 1515-1529.

Bergerud A.T., and Page, R.E. 1987. Displacement and dispersion of parturient caribou at calving as antipredator tactics. Canadian Journal Zoology. 65: 1597-1606.

Bergeron, Y. 1991. The influence of island and mainland lakeshore landscapes on the boreal forest fire regimes. Ecology 72: 1980-1992.

Bergman, C.M., Schaefer, J.A., and Luttich, S.N. (2000). Caribou movement as a correlated random walk. Oecologia. 123: 364-374.

Beyer, H.L. 2004. Hawth's Analysis Tools for ArcGIS. Available online at http://www.spatialecology.com/htools.

Boyce, M.A., Vernier, P.R., Nielsen, S.E., and Schmiegelow, F.K.A. 2002. Evaluating resource selection functions. Ecological Modelling. 157:281-300.

Bradshaw, C.J., Hebert, D.M., Rippin, A.B, and Boutin, S. 1995. Winter peatland habitat selection by woodland caribou in northeastern Alberta. Canadian Journal of Zoology. 73: 1567-1574.

Bradshaw, C.J., Boutin, S., and Hebert, D.M. 1997. Effects of petroleum exploration on woodland caribou in northeastern Alberta. Journal of Wildlife Management. 61 (4): 1127-1133.

Brooks, C.P. 2003. A scalar analysis of landscape connectivity. Oikos. 102:433-439.

Brown, K.G., Elliott, C., and Messier, F. 2000a. Seasonal distribution and population parameters of woodland caribou in central Manitoba: implications for forestry practices. Rangifer Special Issue No. 12:85-94.

Brown, G.S., Mallory, F.F., and Rettie, W.J. 2003. Range size and seasonal movement for female woodland caribou in the boreal forest of northeastern Ontario. Rangifer Special Issue No. 14:227-233.

Brown, G.S., Rettie, W.J., Brooks, R.J., and Mallory, F.F. 2007. Predicting the impacts of forest management on woodland caribou habitat suitability in black spruce boreal forest. Forest Ecology and Management. 245: 137-147.

Brown, W.K., Rettie, W.J., Wynes, B., and Morton, K. 2000b. Wetland habitat selection by woodland caribou as characterized using the Alberta Wetland Inventory. Rangifer Special Issue No. 12:153-157.

Brown, W.K., and Theberge, J.B. 1990. The effect of extreme snow cover on feeding site selection by woodland caribou. Journal of Wildlife Management. 54(1): 161-168.

Burnside, N.G., Smith, R.F., and Waite, S. 2003. Recent historical land use change on the South Downs, United Kingdom. Environmental Conservation. 30(1): 52-60.

Burt, W.H. 1943. Territoriality and home range concepts as applied to mammals. Journal of Mammalogy. 24: 346-352.

Burt, W.H., and Grossenheider, R.P. 1998. Peterson Field Guide: Mammals. Houghton Mifflin, New York.

Cameron, R.D., Smith, W.T., White, R.G., and Griffith, B. 2005. Central Arctic caribou and petroleum development: Distributional, nutritional and reproductive implications. Arctic 58:1-9

Carey, A.B., Horton, S.P. and Biswell, B.L. 1992. Northern spotted owls: influence of prey base and landscape character. Ecological Monographs. 6:223-250.

Chubbs, T.E., Keith, L.B., Mahoney, S.P., and McGrath, M.J. 1993. Responses of woodland caribou (*Rangifer tarandus caribou*) to clear-cutting in east-central Newfoundland. Canadian Journal of Zoology. 71: 487-493.

Clark, J.S. 1988. Effect of climate change on fire regimes in northwestern Minnesota. Nature. 334:233-235.

Clark, J.D., Dunn, J.E. and Smith, K.G. 1993. A multivariate model of female black bear habitat use for a geographic information system. Journal of Wildlife Management. 57: 519-526.

Cook, T. 1990. *Methods of Dendrochronology: Applications in the Environmental Sciences.* Klumer Press, Boston.

Cook, W.M., Anderson, R.M., and Schweiger, E.W. 2003. Is the matrix really inhospitable? Vole runway distribution in experimentally fragmentated landscape. Oikos. 104: 5-14.

COSEWIC 2002. Assessment and update status report on the woodland caribou *Rangifer tarandus caribou* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa. xi + 98 pp.

Coxson, D.S., and Marsh, J. 2001. Lichen chronosequences (postfire and postharvest) in lodgepole pine (*Pinus contorta*) forests of northern interior British Columbia. Canadian Journal of Botany. 79: 1449-1464.

Cumming, H.G., and Beange, D.B. 1987. Dispersion and movements of woodland caribou near Lake Nipigon, Ontario. Journal of Wildlife Management. 51:69-79.

Cumming, H.G., and Beange, D.B. 1993. Survival of woodland caribou in commercial forests of northern Ontario. Biological Conservation. 72 (3): 411.

Darby, W.R., and Pruitt Jr., W.O. 1984. Habitat use, movements, and grouping behaviour of woodland caribou, *Rangifer tarandus caribou*, in southeastern Manitoba. Canadian Field-Naturalist. 98 (2): 184-190.

Downes, C.M., Theberge, J.B., and Smith, S.M. 1986. The influence of insects on the distribution, microhabitat choice and behaviour of the Burwash caribou herd. Canadian Journal of Zoology. 64:622-629.

Dyer, S.J., O'Neill, J.P., Wasel, S.M., and Boutin, S. 2002. Quantifying barrier effects of roads and seismic lines on movements of female woodland caribou in northeastern Alberta. Canadian Journal of Zoology. 80:839-845.

Dyer, S.J., O'Neill, J.P., Wasel, S.M., and Boutin, S. 2001. Avoidance of industrial development by woodland caribou. Journal of Wildlife Management. 65 (3): 531-542.

Dyke, C. 2008. Characterization of woodland caribou (*Rangifer tarandus caribou*) calving habitat in the boreal plains and boreal shield ecozones of Manitoba and Saskatchewan. M.N.R.M. Thesis. University of Manitoba. Winnipeg, Manitoba.

Ecological Stratification Working Group. 1998. A national ecological framework for Canada. Research Branch, Centre for Land and Biological Resources Research, Agriculture and Agri-Food Canada and Environment Canada, State of the Environment Directorate, Ecozone Analysis Branch, Ottawa Ont.

Edge, W.D., Marcum, C.L., and Olsen, S.L. 1985. Effects of logging activities on the home-range fidelity of elk. Journal Wildlife Management. 49: 741-444.

Environment Canada. 2007. Recovery strategy for the Woodland Caribou *(Rangifer tarandus caribou)*, Boreal Population, in Canada (Draft), June 2007. Species at Risk Act Recovery Strategy Series. Ottawa: Environment Canada. v + 48 pp.

Environmental Systems Research Institute Inc. 2006. ArcMap 9.2 GIS and Mapping Software. ESRI, Redlands, California, USA.

Fahrig, L. 1997. Relative effects of habitat loss and fragmentation on population extinction. Journal Wildlife Management. 61: 603-610.

Fall, A., and Fall, J. 2001. A domain-specific language for models of landscape dynamics. Ecological Modelling. 141:1-18.

Fall, A., Fortin, M-J., Manseau, M. and O'Brien, D. 2007. Spatial graphs: Principles and applications for habitat connectivity. Ecosystems. 43: 123-137.

Ferreras, P. 2001. Landscape structure and asymmetrical inter-patch connectivity in a metapopulation of the endangered Iberian lynx. Biological Conservation. 100: 125-136.

Ferguson, S.H., Bergerud, A.T., and Ferguson, R. 1988. Predation risk and habitat selection in the persistence of reminant caribou population. Oecologia 76:236-245.

Ferguson, S.H., and Elkie, P.C. 2004. Seasonal movement patterns of woodland caribou (*Rangifer tarandus caribou*). Journal of Zoology London. 262: 125-134.

Fitzmaurice, G.M., Laird, N.M., and Ware, J.H. 2004. *Applied Longitudinal Analysis*. John Wiley and Sons, Hoboken, New Jersey.

Forman, R.T., and Godron, M. 1986. *Landscape Ecology*. New York, John Wiley and Sons.

Goodwin, B.J., and Fahrig, L. 2002. How does landscape structure influence landscape connectivity? Oikos. 99: 552-570.

Government of Saskatchewan. 2002. Caring for natural environments: A biodiversity action plan for Saskatchewan's future 2004-2009.

Gustafson, E.J., and Gardner, R.H. 1996. The effect of landscape heterogeneity on the probability of patch colonization. Ecology. 77: 94-107.

Harary, F. 1969. *Graph Theory*. Addison-Wesley Series in Mathematics. Reading, Massachusetts.

Hardin, J.W. and Hilbe, J.M. 2003. *Generalized Estimating Equations*. Chapman and Hall, New York.

Hargis, C.D., Bissonette, J.A. and David, J.L. 1998. The behaviour of landscape metrics commonly used in the study of habitat fragmentation. Landscape Ecology. 13: 167-186.

Harris, L.D. 1984. *The fragmented forest: Island biogeographic theory and the preservation of biotic diversity*. Chicago: University of Chicago Press. 211 p.

Harvey, B.D., Leduc, A., Gautier, S. and Bergeron, Y. 2002. Stand-landscape integration in natural disturbance-based management of the southern boreal forest. Forest Ecology and Management. 155:369-385.

Haynes, K.J., and Cronin, J.T. 2004. Confounding of patch quality and matrix affects in herbivore movement studies. Landscape Ecology. 19: 119-124.

Heggem, D.T, Edmonds, C., Neal, A.C., Bice, L. and Jones, K.B. 2000. A landscape ecology assessment of Tensas River Basin. Environmental Monitoring and Assessment. 64: 41-54.

Hirai, T. 1998. An evaluation of woodland caribou calving habitat in the Wabowden Area, Manitoba. M.N.R.M. Thesis. University of Manitoba, Winnipeg, Manitoba.

Hosmer, D.W. and Lemeshow, S. 2000. *Applied Logistic Regression, Second Ed.* Wiley, New York.

Jackson, V.L., Laack, L.L. and Zimmerman, E.G. 2005. Landscape metrics associated with habitat use by ocelots in South Texas. Journal of Wildlife Management. 69(2):733-738.

James, A.R., Boutin, S., Hebert, D.M., and Rippin, A.B. 2004. Spatial separation of caribou from moose and its relation to predation by wolves. Journal of Wildlife Management. 68(4): 799-809.

Johnson, C.J., Alexander, N.D, Wheate, R.D., and Parker, K.L. 2003. Characterizing woodland caribou habitat in sub-boreal and boreal forests. Forest Ecology and Management. 180: 241-248.

Johnson, C.J., and Gillingham, M.P. 2002. A multiscale behavioural approach to understanding the movements of woodland caribou. Ecological Applications. 12 (6): 1840-1860.

Johnson, C.J. and Gilligham, M.P. 2005. An evaluation of mapped species distribution models used for conservation planning. Environmental Conservation. 32:117-128.

Johnson, C. J., Parker, K.L., and Heard, D.C. 2000. Feeding site selection by woodland caribou in north-central British Columbia. Rangifer Special Issue No. 12:159-172.

Johnson, C. J., Parker, K.L., and Heard, D.C. 2001a. Foraging across a variable landscape: behavioral decisions made by woodland caribou at multiple spatial scales. Oecologia 127:590-602.

Johnson, E.A. 1992. *Fire and vegetation dynamics: Studies from the North American boreal forest.* Cambridge University Press, Cambridge, England.

Johnson, E.A. and Larsen, C.P. 1991. Climatically induced change in fire frequency in the Southern Canadian Rockies. Ecology. 72:194-201.

Johnson, E.A., Miyanishi, K., and Bridge, S.R. 2001b. Wildfire regime in the boreal forest and the idea of suppression and fuel buildup. Conservation. Biology. 15(6): 1554-1557

Johnson, E.A., Miyanishi, K. and Weir, J.M. 1998. Wildfires in the western Canada boreal forest: landscape patterns and ecosystem management. Journal of Vegetation Science. 9:603-610.

Keitt, T.H., Urban, D.L., and Milne, B.T. 1997. Detecting critical scales in fragmented landscapes. Conservation Ecology. [online] 1, 4.
Available from: http://www.consecol.org/vol1/iss1/art4/

Kilpatrick, H.J., Spohr, S.M., and Lima, K.K. 2001. Effects of population reduction on home ranges of female white-tailed deer at high densities. Canadian Journal Zoology. 79: 949-954.

Kilpatrick, H.J. and Lima, K.K. 1999. Effects of archery hunting on movement and activity of female white-tailed deer in an urban landscape. Wildlife Society Bulletin. 27: 433-440.

Kinley, T.A. 2003. Characteristics of early-winter caribou, *Rangifer tarandus caribou*, feeding sites in the southern Purcell Mountains, British Columbia. Canadian Field-Naturalist. 117(3): 352-359.

Klein, D.R. 1982. Fire, lichens and caribou. Journal of Range Management. 35:390-395.

Koper, N and Manseau, M. 2009. Generalized estimating equations and generalized linear mixed-effects models for modeling resource selection. Journal of Animal Ecology. 46: 590-599.

Lander, C. 2006. Distribution and movements of woodland caribou on disturbed landscapes in West-central Manitoba: Implications for forestry. M.N.R.M. Thesis. University of Manitoba, Winnipeg, Manitoba.

Larsen, C.P.S. 1997. Spatial and temporal variations in boreal forest fire frequency in northern Alberta. Journal of Biogeography. 24:663-673.

Lemkuhl, J.F. and Raphael, M.G. 1993. Habitat pattern under northern spotted owl locations on the Olympic Peninsula, Washington. Journal of Wildlife Management. 57:302-315.

Mahoney, S.P., and Virgl, J.A. 2003. Habitat selection and demography of a nonmigratory woodland caribou population in Newfoundland. Canadian Journal of Zoology. 81:321-334.

Manley, B.F., McDonald, L.L. Thomas, D.L., McDonald, T.L. and Erickson, W.P. 2002. *Resource Selection by Animals: Statistical Design and Analysis for Field Studies 2nd Edition.* Klumer, New York.

Manseau, M., Fall, A., O'Brien, D., and Fortin, M-J. 2002. National Parks and the protection of woodland caribou: a multi-scale landscape analysis method. Research Links. 10:24-8.

Marcot, B.G., and Chin, P.Z. 1982. Use of graph theory measures for assessing diversity of wildlife habitat. In: Mathematical models of renewable resources, Proceedings of the First Pacific Coast Conference on Mathematical Models of Renewable Resources. Humboldt State University, Arcata, CA.

McGarigal, K. and Marks, B.J. 1995. FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. General technical report US Department of Agriculture, Forest Service.

McGarigal, D., Romme, W.H., Crist, M., and Roworth, E. 2001. Cumulative effects of roads and logging on landscape structure in the San Juan Mountains, Colorado (USA). Landscape Ecology. 16: 327-349.

Mladenoff, D.J., Sickley, T.A., Haight, R.G. and Wydeven, A.P. 1995. A regional landscape analysis and prediction of favorable gray wolf habitat in the northern Great Lakes Region. Conservation Biology. 9:279-294.

Mosnier, A., Ouellet, J., Sirois, L., and Fornier, N. 2003. Habitat selection and home range dynamics of the Gaspé caribou: a hierarchical analysis. Canadian. Journal of Zoology. 81:1174-1184.

Nellemann, C., and Cameron, R.D. 1996. Effects of petroleum development on terrain preferences of calving caribou. Arctic 49 (1): 23-28.

O'Brien, D., Manseau, M., Fall, A., Fortin, M-J. 2006. Testing the importance of spatial configuration of winter habitat for woodland caribou: An application of graph theory. Biological Conservation. 130 (1): 70-83

Okabe, A., Boots, B., Sugihara, K. and Nok Chiu, S. 2000. *Spatial Tessellation: Concepts and Applications of Voronoi Diagrams.* Wiley, New York.

Overall, J.E. and Tonidandel, S. 2004. Robustness of generalized estimating equation (GEE) tests of significance against misspecification of the error structure model. Biometrical Journal. 46:203-213.

Parks Canada. 1986. Prince Albert National Park Resource Description and Analysis. Parks Canada Western Service Center, Winnipeg, Manitoba.

Pattie, D.L., and Hoffman, R.S. 1999. *Mammals of the North American Parks and Prairies*. Edmonton Alberta.

Pereira, J.M. and Itami, R.M. 1991. GIS based habitat modeling using logistic multiple regression: A study of the Mt.Graham red squirrel. Photogrammetric engineering and remote sensing. 57: 1475-1486.

Pitt, W.C., and Jordan, P.A. 1994. A survey of the nematode parasite, *Parelaphostrongylus tenuis*, in the white-tailed deer, *Odocoileus virginianus*, in a region proposed for caribou, *Rangifer tarandus caribou*, re-introduction in Minnesota. Canadian Field-Naturalist. 108(3):341-346.

Poole, K.G., Heard, D.C., and Mowat, G. 2000. Habitat use by woodland caribou near Takla Lake in central British Columbia. Canadian Journal of Zoology. 78:1552-1561.

Post, E., Boving, P.S., Pedersen, C., and MacArthur, M.A. 2003. Synchrony between caribou calving and plant phenology in depredated and non-depredated populations. Canadian Journal of Zoology. 81:1709-1714.

Prince Albert Model Forest. 2008. Prince Albert Model Forest Annual Report 2007-08. [online] URL: http://mfiqlx.sasktelwebhosting.com/pubs/Annual %20report%20Final%20July%2031.08.doc

Prince Albert National Park. 2008. Prince Albert National Park Management Plan. Parks Canada

Reed, W.J. and Errico, D. 1986. Optional harvest scheduling at the forest level in the presence of the risk of fire. Canadian Journal of Forest Research. 16: 266-278.

Rettie, W.J, and Messier, F. 1998. Dynamics of woodland caribou populations at the southern limit of their range in Saskatchewan. Canadian Journal of Zoology. 76: 251-259

Rettie, W.J., and Messier, F. 2000. Hierarchical habitat selection by woodland caribou: its relationship to limiting factors. Ecography 23:466-478.

Rettie, W.J. and Messier, F. 2001. Range use and movement rates of woodland caribou in Saskatchewan. Canadian Journal of Zoology. 79:1933-1940.

Rettie, W. J., Sheard, J.W., and Messier, F. 1997. Identification and description of forested vegetation communities available to woodland caribou: relating wildlife habitat to forest cover data. Forest Ecology and Management. 93:245-260.

Ricketts, T.H. 2001. The matrix matters: effective isolation in fragmented landscapes. American Naturalist. 158: 87-99.

Rock, T.W. 1988. An assessment of survey techniques and population characteristics of woodland caribou (*Rangifer tarandus caribou*) on three study areas in Saskatchewan. Saskatchewan Wildlife Branch, Wildlife Population Management Information. 16pp.

Rock, T.W. 1992. A proposal for the management of woodland caribou in Saskatchewan. Saskatchewan Natural Resources. Wildlife Tech. Rep. 92-3. 28 pp.

Rominger E.M., and Oldemeyer J.L. 1990. Early winter diet of woodland caribou in relation to snow accumulation, Selkirk Mountains, British Columbia, Canada. Canadian Journal of Zoology. 68:2691-2694.

SAS Institute, Inc. 2003. The SAS system for Windows Version 9.1. Carey.

Saskatchewan Environment. 2007. Recovery strategy for boreal woodland caribou (*Rangifer tarandus caribou*) in Saskatchewan. Saskatchewan Environment. Fish and Wildlife Tech. Rep. 2007-draft. 46 pp.

Saskatchewan Environment. 2009. Natural forest pattern standards and guidelines for the Saskatchewan provincial forest. Draft 12, Feb 2009, Saskatchewan Environment, Forest Service Branch. 23 pp.

Saunders, D., Hobbs, R.J. and Margules, C.R. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology. 5: 18-32.

Schaefer, J.A. 1988. Fire and woodland caribou (*Rangifer tarandus caribou*): An evaluation of range in southeastern Manitoba. M.Sc thesis. University of Manitoba, Winnipeg, Manitoba.

Schaefer JA. 1996. Canopy, snow and lichens on woodland caribou range in southeastern Manitoba. Rangifer Special Issue 9:239-244.

Schaefer, J.A., and Pruitt W.O. 1991. Fire and woodland caribou in Southeastern Manitoba. Wildlife.Monographs. 116: 1-39.

Schneider, R. R., Wynes, B., Wasle, S., Dzus, E., and Hiltz, M. 2000. Habitat use by caribou in northern Alberta, Canada. Rangifer 20:43-50.

Seip, D.R. 1991. Predation and caribou populations. Rangifer, Special Issue No. 7: 46-52.

Shriver, W.G., Hodgman, T.P., Gibbs, J.P., and Vickery, P.D. 2004. Landscape context influences salt marsh bird diversity and area requirements in New England. Biological Conservation. 119: 545-553.

Smith, K.G., Ficht, J.E., Hobson, D., Sorenson, T.C, and Hervieux, D. 2000. Winter distribution of woodland caribou in relations to clear-cut logging in west-central Alberta. Canadian Journal of Zoology. 78: 1433-1440.

Southworth, J., Nagendra, H. and Tucker, C. 2002. Fragmentation of a landscape: Incorporating landscape metrics into satellite analysis of land-cover change. Landscape Research. 27:253-269.

Stuart-Smith, A.K., Bradshaw, C.J., Boutin, S., Hebert, D.M., and Rippin, A.B. 1997. Population parameters of woodland caribou relative to landscape patterns in northeastern Alberta. Journal of Wildlife Management. 61:622-633.

Sweaner, P.Y., and Sandegren, F. 1989. Winter-range philopatry of seasonally migratory moose. Journal of Applied Ecology. 26:25-33

Taylor, P.D., Fahrig L., Henein K., and Merriam, G. 1993. Connectivity is a vital element of landscape structure. Oikos 68: 571-572.

Terry, E.L., McLellan, B.N., and Watts, G.S. 2000. Winter habitat ecology of mountain caribou in relation to forest management. Journal of Applied Ecology 37: 589-602.

Thomas, D. C. and Gray, D. R.. 2002. Update COSEWIC status report on the woodland caribou *Rangifer tarandus caribou* in Canada. Committee on the Status of Endangered Wildlife in Canada. Page xi + 98, Ottawa.

Tischendorf, L., and Fahrig, L. 2000. On the usage and measurement of landscape connectivity. Oikos 90: 7-19.

Trottier, T.W. 1988. The natural history of woodland caribou. (*Rangifer tarandus caribou*). Saskatchewan Parks, Recreation and Culture. Wildlife Branch.

Turner, M.G., Gardner, R.H. & O'Neill, R.V. 2001. *Landscape Ecology in Theory and Practice: Pattern and Process.* Springer, New York.

Valkenburg, P., McNay, M.E. and Dale, B.W. 2004. Calf mortality and population growth in the Delta caribou herd after wolf control. Wildlife Society Bulletin. 32:746-456.

Van der Ree, R., Bennett, A.F., and Gilmore, D.C. 2003. Gap-crossing by gliding marsupials: thresholds for use of isolated woodland patches in an isolated agricultural landscape. Biological Conservation. 115: 241-249.

Walker, L.C. 1999. *The North American forests: Geography, ecology and silviculture.* CRC Press, Florida.

Weir, J.M.H., Johnson, E.A. & Miyanishi, K. 2000. Fire frequency and the spatial age mosaic of the mixed-wood boreal forest in western Canada. Ecological Applications. 10(4):1162-1177.

Weir, J.M.H. 1996. Fire frequency in the boreal mixedwood. M.Sc. Thesis. University of Calgary, Calgary Alberta.

Whitney, G.G. 1994. *From Coastal Wilderness to Fruited Plain: A History of Environmental Change in Temperate North America, 1500 to present.* Cambridge University Press, Cambridge, Great Britain.

Whittington, J., St. Clair, C.C., and Mercer, G. 2005. Spatial response of wolves to roads and trails in mountain valleys. Ecological Applications. 15:543-553.

Whittington, J., St. Clair, C.C., and Mercer, G. 2004. Path tortuosity and the permeability of roads and trails to wolf movement. Ecology and Society 9. Available from: <http//www.ecologyandsociety.org/vol9/iss1/art4/>

Williams, M. 1989. *Americans and their forest: A Historical Geography.* Cambridge University Press, Cambridge, Great Britain.

With, K.A., Gardner, R.H., and Turner, M.G. 1997. Landscape connectivity and population distributions in heterogeneous environments. Oikos. 78: 151-169.

With, K.A., Cadaret, S.J., and Davis, C. 1999. Movement responses to patch structure in experimental fractal landscapes. Ecology 80: 1340-1353.

Wittmer, H.U., McLellan, B.N., Seip, D.R., Young, J.A., Kinley, T.A., Watts G.S., and Dennis, H. 2005. Population dynamics of the endangered mountain ecotype of woodland caribou (*Rangifer tarandus caribou*) in British Columbia, Canada. Canadian Journal of Zoology. 83:407-418.